National Land and Water Resources Audit

Implementation Project 2

Theme 2 – Dryland Salinity

Using Natural Resource Inventory Data to Improve the Management of Dryland Salinity in the Great Southern, Western Australia

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Summary

The synoptic assessment of salinity risk and the likely costs and benefits associated with various management options is crucial to natural resource management decision-making in southern Australia. A variety of methods have been proposed and tested for assessing various aspects of salinity risks and costs, but no large region of Australia has ever had a comprehensive risk assessment across the range of biophysical and economic issues with forecasts of the effectiveness of different levels of intervention. This National Land and Water Resources Audit Implementation Project (referred to locally as *Salt Scenarios 2020*, or *SS2020* for short) attempted to provide such an assessment (at a scale of around 1:100,000).

The existing methods of monitoring and predicting salinity (based on variables derived from widely-available Landsat TM data and existing contour data; albeit with improved variable extraction from the DEMs) are being applied to the rest of the agricultural area of WA as part of the Land Monitor Project, funded in part by NHT. Collecting accurate contour data (2-metre) is a major part of the NHT project. This Audit project was proposed to allow other fundamental data sets, and especially groundwater levels from bore-hole data, to be used to significantly improve predictions in lower-rainfall areas as well as refine the predictions in the high rainfall areas.

The Great Southern is an area of considerable economic and environmental value populated by 60,000 people. In 1996, it was estimated that about 30% of the cleared land and associated vegetation and water resources are at risk from becoming salt-affected over the next 30 years unless high-water use farming systems and farm forestry are adopted over large parts of the region (Ferdowsian et al., 1996).

Four key questions arise with respect to the future of this region as affected by dryland salinity:

- How large will the problem eventually be under current land practices? How large might it be in the year 2020?
- What is at risk if the area under threat grows that large?
- To what degree can we change the eventual extent of salinity with land use alternatives that are both feasible and available?
- What are the costs and benefits of intervening with these alternative land uses?

Ultimately, the SS2020 Project aimed to provide some guidance to state, regional and local planners and managers regarding salinity risk in the Great Southern. The analyses underpinning this guidance were based on similar data employed by NLWRA projects under Theme 2 – Dryland Salinity.

Key findings

- In what was initially believed to be a relatively hydrologically data-rich region, screening the data for quality (accuracy in positions) and reliability (source and quality control of data capture) reduced the useful data to a fraction of the original database. It is, however, still data-rich compared with many areas in southern Australia.

- Two new methods for analysing trends in bore hydrographs were developed. The first provides different linear trends for different segments of the data, and at the same time estimates the amplitude(s) of the seasonal response(s) in the data. The second, called HARTT (hydrograph analysis of rainfall and time trends), separates the effect of atypical rainfall events from the underlying time trend and the lag between rainfall and its impact on groundwater is explicitly represented. It can cope with irregularly spaced data and missing observations, which are common problems in hydrograph analysis.

- Statistically repeatable methods for developing groundwater level maps for large regions with sparse data failed to produce a robust and reliable result. It remains to be determined whether this is due to an inherent lack of relationship between depth to water table and land form variable in local aquifer systems which are not well-connected, or whether including other explanatory variable (such as rock-platform relationships, time since clearing, land use and depth to regolith – if they were widely available, would lead to improved relationships. The relationships are likely to be better when the groundwater levels approach equilibrium, especially in intermediate and regional aquifer systems (e.g. south Stirlings, west Midlands)

- Previous methods for producing groundwater surfaces worked well in an area in which the surface and groundwater data were related with the local area, but such relationships could not be established in other regions.

- The application of the Land Monitor salinity risk maps based on predictions of shallow watertables for equilibrium conditions were useful for extrapolating expert knowledge on the ultimate extent of salinity risk and for identifying the areas in which assets may be at risk at the regional scale.

- For the 2.3 m ha of area mapped for salinity risk, 1157 buildings, 3333 km of roads, 15,682 farm dams, and 101,877 ha of perennial (remnant) vegetation is at risk to salinity over the next 20 years. The methodology developed cannot determine whether the risk will be realised and is based on regional-scaled interpolation.

- Groundwater modelling suggests that either: (a) most of this landscape requires large-scale intervention to change salinity risk, apart from the steepest, wettest sites in the western portion of the region; (b) risk may be modified by engineering options; or (c) the do-nothing scenario will result in larger saline areas developing.

- For three case study areas that were examined in detail, the discounted value of production losses under the "business as usual" scenario ranged between $800 and $1090 for every hectare of land that is predicted to have a shallow watertable and eventually become saline. In aggregate terms, potential losses for each of the two 30,000 hectare case studies ranged between $4.5 and $6.1 million. The larger of the three case study areas (comprising 107,250 ha) was estimated to potentially suffer losses of approximately $8 million. In addition to these production losses, a significant amount of new damage is expected to be inflicted on infrastructure and perennial vegetation. Across all three case study areas, within the zone predicted to have a shallow watertable in the future within 20 years, together with up to 54 farm buildings, and 540 dams, and 10,050 hectares of perennial vegetation.

- Hydrological modelling conducted by SS2020 suggests that most remediation strategies do not offer a great deal of off-site control. Hence, if a strategy is to produce a net benefit, most of the benefits will need to come from land prevented from becoming saline because of a treatment planted on that land. This means that treatments will need to be commercially profitable in their own right if they are to be attractive to farmers and even desirable from a community-wide perspective. A priority for future work will be to determine the extent to which this general conclusion applies to the SS2020 region, and to identify areas where there is potential to realise off-site benefits from treatments.

Implications

The implications of this project for the management of dryland salinity in Australia are profound and of immediate application, but must be tempered by the lack of degree of detail in salinity risk assessments possible with existing data and methods. The original vision of a regional, synoptic model of salinity risk under various scenarios, based on a regional hydraulic head surface, could not be realised. Instead, inferences on the economics and impacts of alternative land uses were largely based on case study modelling in regions with reasonable data.

Nevertheless, the following results are likely to hold for most parts of the region.

For the 2.13 m ha mapped by Evans et al. (2000), 0.65 m ha (31%) is expected to be at risk of salinity at equilibrium; this is the same value as estimated by Ferdowsian et al. (1996) for the entire Wheatbelt, and underscores the scale of the challenge in WA.

There are large numbers of farm and community assets at risk of developing shallow watertable and salinity, including dams, roads and remnant vegetation.

For all but the most dissected landscapes, intervention in the form of land-use change would need to be substantial and widespread. When alternative enterprises (e.g., trees) are limited to the most appropriate soils only, the ultimate extent of salinity does not significantly change, although time to impact is delayed.

Salinity risk abatement treatments associated with the low-recharge land-use systems modelled only benefits the land they are planted on. Thus, such systems must largely be profitable in their own right. The non-farm assets at risk require large-scale, largely non-economically-driven intervention.

The most immediate and obvious implication is for the analysis of where public funds should flow in aid of salinity control. Given the scale of revegetation required to substantially change the ultimate extent of salinity, and limited public funds, it does not appear feasible with the available low-recharge farming systems and tools which are at our disposal to make up the gap in farm profitability associated with alternative, low-recharge farming systems. On the other hand, recognising the local nature of the benefits, public assets such as key remnant vegetation sites, roads and townsites will require large, direct investment in remedial works. Given the nature of the hydrology in WA (dominance of local flowsystems), it is arguable that the highest-priority uses of those limited funds will mostly be in engineering works in or around the assets to be protected, rather than for revegetation on farms.

The implications of buying time as opposed to reducing the area ultimately salinised are also profound. While “low-recharge” farming will ultimately leave almost as much land at risk to salinity as current practice, the possibility that decades of time might elapse before the full impact of salinity is realised has considerable social and economic value (while also providing protection to the 70% that won't be saline). If nothing else, it may give families and government more time to adapt to a salinised landscape.

Decision-makers need confidence that the technical estimates and data are sound and robust. The SS2020 project has highlighted the difficulty in making robust inferences even for regions that are relatively data-rich. In fact, the density of reliable groundwater data in the WA wheatbelt and elsewhere in Australia is disturbingly low, much lower than expected on the face of existing databases. When these databases are interrogated and screened for quality control, a minority of the data proved to be useful. Many bores had very infrequent time series data and short records. Only 711 bores in the study area had adequate time series data to interpret trends. Of these, only 154 had surveyed or DGPS location data. This serious gap in our knowledge about a landscape undergoing profound and disturbing change greatly compromises our technical ability to provide advice.

These results challenge our perceptions regarding what are the most important and useful data for assessing salinity risk, and the most effective ways of using them. While in theory the traditional hydrogeological approach which relies on groundwater levels provides a more definitive assessment, the sparseness and current apparent lack of reliability of this data across southern Australia does not recommend this approach for a national methodology. In contrast, synoptic, widely-available data (high-quality elevation data, high-resolution Landsat satellite reflectance data) were shown to be of substantial use, while still requiring a degree of expert input into the analysis. Of the geographical coverages examined, the results from this project would lead to a questioning of the direct use of soil mapping units in deriving salinity risk; it appears that there is little correspondence, at any scale, between the occurrence of high-saline watertables and the soil units available to us.

Nevertheless, the need for bore hydrographic monitoring as input into defining the effectiveness of treatments is indisputable. There is a serious deficiency in southern Australia in a sustained and coordinated approach to monitoring the effectiveness of remedial works, including those supported under public funding such as the Natural Heritage Trust.

The scenario development and subsequent modelling presented here challenges the accepted belief that relatively limited recharge control is the basic tenant of salinity management. The evidence suggests that in areas that are close to equilibrium, the benefits accrued from engineering options and productivity from saline land are likely to outway the benefits from recharge control (unless the latter are economic in their own right).

Recommendations

Borehole data play an essential role in understanding the hydrogeology and the salinity of groundwater, and are valuable for reconnaissance coverage of an area. This study showed that in the dissected landscapes with existing bore data, reliable hydraulic head surfaces could not be created.

An important outcome from this study is the need for accurate bore location data. Very few of the 3000 bores have known accurate locations and therefore could not be related to their position in the landscape with confidence. Only 154 bores in the study region had surveyed or DGPS location data at the time of this study. Moreover, these bores are often clustered in space, as they generally form part of catchment-based studies and thus do not provide adequate spatial representation to derive groundwater surfaces. It is recommended that GPS-based georeferencing standards be adopted for locating new and existing groundwater bores where these are to be used in spatially-explicit modelling.

When accurately located bore data are available, it needs to be established if and where other datasets, such as soil landscape maps, new-generation soil maps derived from radiometric data and time since clearing can be used to produce accurate hydraulic head surfaces.

There is a need for continued monitoring of selected bores to improve trend analyses and for installation of new bores in key areas. The HARTT methodology developed during this study will allow incomplete bore records to be analysed better than was previously possible. The majority of bores have been drilled to monitor treatment effects or are already located in discharge zones, which makes modelling using the methods developed in SS2020 problematic. More bores are required in the midslope and upper slopes to identify trends in groundwater systems. The recently developed AgBores database managed by Agriculture WA should provide the vehicle for the storage and manipulation of the time series data required by projects such as SS2020.

A comprehensive monitoring strategy needs to be developed, based on a hierarchical approach, in which broad-scale, satellite-based monitoring (using high-resolution Landsat and high quality elevations complemented by more detailed ground-based measurements for selected key areas. Support is needed to undertake further research to produce comprehensive risk assessments for southern Australia, using a combination of DEM-derived variables, variables derived from Landsat TM and borehole data. Research methods aimed at the development of maps of groundwater levels based on sparse data need to be evaluated. Methods that use sparse bore data and soil landform data to infer areas at risk face even greater problems of accuracy than encountered in this study.

There is a need for a general method to set priorities for the investment of public funds based on the kind of knowledge generated by SS2020. In particular, more hydrogeological understanding is required for the Perth and south-coastal systems, to aid in determining the optimal location and impact of treatments.

The methodology for the spatio-economic assessment of salinity adopted in this report provides a systematic basis for categorising the types of economic impacts associated with a treatment strategy. Hydrological modelling conducted by SS2020 suggests that, in many instances, a very small proportion of the benefits from a treatment strategy are likely to be generated by the protection of land and assets away from the treatment site. Hence, if a strategy is to produce a net benefit, most of the benefits will need to come from the direct protection of land on a treatment site. This implies that treatments will need to be commercially profitable in their own right. Indeed, to be attractive for adoption before salinity has become an imminent threat, the treatments will need to be almost as profitable as existing agricultural land uses. A priority for future work will be to determine the extent to which this general conclusion applies to the SS2020 region, and to identify geographical areas where there is potential to realise off-site benefits from treatments. The economic analyses are based on the promise that everything classified to be at risk suffers an economic loss. If (as is generally the case) not all land mapped as at risk goes saline (eg roads), then these results will overestimate the risk.

While the treatments modelled in the case studies show that without major intervention, almost as much land will be left at risk to salinity as current practice, the possibility that decades of time might elapse before the full impact of salinity is realised has considerable social and economic value. A much greater emphasis is required to protect the 70% of the landscape that will not have a future shallow watertable. Analyses are required to spatially locate those catchments which are most responsive to various treatments.

1. Introduction

The broad-scale assessment of salinity risk and the likely costs and benefits associated with various management options is a crucial input to decision-making in natural resource management in southern Australia. While a variety of methods have been proposed and tested for assessing various aspects of salinity risks and costs, there has been no comprehensive risk assessment across a range of biophysical and economic issues for a large region of Australia, nor has there been any major attempt to forecast the effectiveness of different levels of intervention. This Implementation Project for the National Land and Water Resources Audit (the project is referred to locally as *Salt Scenarios 2020*, or *SS2020* for short) attempted to provide such an assessment.

The Great Southern region of Western Australia[1] provides the study area for SS2020. The area is of considerable economic and environmental value, and is populated by 60,000 people. It was estimated by Ferdowsian et al. (1996) that about 30% of the cleared land and associated vegetation and water resources were at risk from becoming salt-affected over the next 30 years unless high-water use farming systems and farm forestry were adopted over large parts of the region.

Farmers in the Great Southern have already lost 5 to 10 per cent of their productive land to salinity. The rate of increase has accelerated recently, as aquifers have filled since clearing took place in the past 30 to 100 years. The potential to lose up to 30 per cent of the most productive land in the state has caused a response at both the state level (Salinity Action Plan, clearing controls, and revegetation schemes) and the local level (landcare and catchment groups, experimenting with high-water-use farming systems, drainage schemes). The areas thought to be most at risk in the next 15 years have the highest rainfalls, yield potentials and possibilities for diversification.

In the south and southeast of the region lie two areas of mega-diversity associated with the Stirling Range and Fitzgerald River National Parks. Agencies and community groups are developing a macro-corridor concept region that provides the framework for a coordinated approach to integrated land and nature conservation management in the Fitzgerald Biosphere region. A system of macro- and micro-corridors will provide a stable environment to protect the biodiversity values of the areas around the parks, and provide a lifeline between the coastal-reserves and parks and inland reserves within the South Coast and Southern Wheatbelt. Lake Toolibin, in the northeast, is the last remaining freshwater lake in the Wheatbelt, while the Dongolocking Reserves in the east are outliers of invaluable remnants of native vegetation within a heavily cleared environment.

[1] For the purposes of the SS2020 project, the region comprises the Western South Coast (Mt Barker Landsat TM scene) and the Upper Blackwood Catchment (Dumbleyung and the Bunbury scene as far west as the Towerinning Catchment). The total area is about 30,000 km^2 (3 M ha).

Salinity monitoring and risk prediction methods, developed for the high-rainfall Kent Catchment as part of the National Dryland Salinity Program (NDSP), have had a marked impact on how landholders and government agencies viewed the salinity threat in this 100,000 ha area. Predictive methods were developed which had an accuracy of 80% (when compared with qualitative predictions made by local experts) in selected test areas. It was apparent, however, that improved risk predictions in the Kent and in the lower-rainfall wheatbelt would require more accurate contour data and a better understanding of groundwater levels and trends where salinity was still not expressed at the surface.

The basic methods of monitoring and predicting salinity developed are being extended to the rest of the agricultural area of WA as part of the Land Monitor Project, funded in part by the NHT. Collecting accurate contour data (2-metre) is a major part of the NHT project. This Audit project was proposed to allow other fundamental data sets, and especially groundwater levels from borehole data, to be used to significantly improve predictions in lower-rainfall areas.

The region has relatively dense trend data, such as changes in groundwater levels, and information on salt-affected land and vegetation condition over the past ten years. It also has good inventory data (such as soil-landform maps), accurate digital elevation models (2-metre contours), socio-economic data and 250,000 ha of airborne geophysics. Almost all of these data are likely to become available to other rural areas within WA over the next 2 to 3 years. They are also available to a greater or lesser extent in other parts of Australia which face salinity problems. The challenge for the SS2020 project was to see if these data could be used more effectively, so that government agencies and the farming community could make better decisions about resource allocations. The Project involved the pooling of data and expertise of six state agencies and three CSIRO divisions.

It was expected that SS2020 would show how maps of changes in saline land and vegetation condition could be combined with groundwater levels and digital elevation data at 2-metre contours to identify areas of environmental, economic and social value at risk from salinity. To do this, it would integrate the two methods recommended by the NLWRA -- groundwater levels combined with soils and landform data, and products derived from Landsat TM (Nulsen and Evans 1998) -- together with other data sets (including DEM-derived variables and airborne geophysics where available) to improve the mapping and prediction of saline areas. It would also show the limitations of available data for this purpose.

Four key questions arise with respect to the future of this region as affected by dryland salinity:

- How large will the problem eventually be under current land practices? How large might it be in the year 2020?
- What is at risk if the area under threat grows that large?
- To what degree can we change the eventual extent of salinity with land use alternatives that are both feasible and available?
- What are the costs and benefits of intervening with these alternative land uses?

The SS2020 Project aimed to provide some answers to these questions, to provide some guidance to state, regional and local planners and managers regarding salinity risk in the Great Southern. The analyses underpinning this guidance were to be based on similar data to those employed by other NLWRA projects under Theme 2 – Dryland Salinity.

2. Methods

The SS2020 Project had the following four main components:

1. A comprehensive analysis of groundwater levels from borehole and other data (such as rainfall data), including trends at individual locations and a description of the regional borehole groundwater-level surface in 1996/97.

2. Integration of this estimated surface and rates of rise with current data layers (surface DEM, satellite-derived surface salinity, DEM-derived water flows, upslope vegetation, landuse) using an amalgamation of approaches from CMIS[2] and CLW[3] to improve the prediction of areas at risk of salinity in 2020.

3. What-if scenarios based on the likely effects of sensible management interventions on groundwater levels, and from using the improved predictions from component 2.

4. Economic and other analyses of "do-nothing", and of the results of the interventions in component 3.

Specific objectives included:

- a comprehensive analysis and description of groundwater data, resulting in regional maps of groundwater levels, current trends and forecasts;
- improved salt risk prediction maps for a 30,000 km^2 part of WA's wheat- and wool-producing areas – these maps will be produced by integrating the groundwater data with other spatial and point data sets (including Landsat TM and DEMs) in an innovative manner;
- improved maps showing valuable infrastructure, land, and vegetation resources at risk from salinity;
- more accurate statistics on the areas at risk from salinity;
- maps showing the spatial distribution of economic losses to farmers and the community if current practices continue, and simple what-if scenarios based on changes in the groundwater-level trends if different land-use practices are adopted; and
- economic benefits and costs (at the whole-farm level) of incorporating management practices available to farmers, plus improvements in other values from management (e.g. ecological benefits).

SS2020 specifically set out to develop improved methods for salinity risk prediction, by integrating a comprehensive description of the borehole data with existing approaches based on Landsat TM data and DEM-derived variables. The following sections outline the methods that were developed and used.

2.1 Comprehensive Description of the Groundwater Data

There were large datasets available relating to groundwater levels and groundwater salinity across the region, although the data were not evenly distributed in space and time.

One of the aims of SS2020 was to relate water levels to DEM-derived variables (eg position in landscape); the relationship could then be extrapolated to form a water-level surface for any region where high-quality DEMs are available (eg 2m-contour DEMs from Land Monitor). The resulting trends could then be adjusted according to intervention scenarios.

Borehole data were obtained for all known and identifiable bores within the SS2020 study area. Most of the data were extracted from the "AgBores" database maintained by Agriculture Western Australia, with some additional data being provided by the Water and Rivers Commission. AgBores contains data from thousands of boreholes which have been monitored over the past 20 years or

[2] CSIRO Mathematical and Information Sciences
[3] CSIRO Land and Water

so. Hydrologists from AGWEST provided subsets of bores where time series (trends) were available and the bores were likely to accurately represent the water levels of the landscapes in which they were located.

2.2 Trends in Groundwater Levels

Appropriate and effective synthesis of the borehole data into a regional description of salinity and its trend is essential to any salinity risk analysis. This required developing methods to estimate long-term groundwater trend from periods of record that were both discontinuous and of different duration.

Analyses of trends in bore hydrographs were undertaken by three methods.

The first method was the traditional approach (fitting a line by eye or by linear least squares regression to trend data) and was applied in all cases.

For the second method, inspection of a number of sets of borehole data indicated that there were two basic patterns to the groundwater responses. A linear trend (or trends) described the medium- to long-term changes in the groundwater level, while a periodic response described the seasonal variations. The observed trends in the groundwater level could sometimes be described by either one of these patterns and sometimes by a combination of both. For many boreholes, different periods of time exhibited different (roughly) linear trends - these periods are referred to here as segments.

A new approach was developed which allows for different linear trends in different segments over the duration of the records. A 12-month seasonal cycle can be superimposed on the segments; the amplitude of the cycle can also vary from segment to segment. Three distinct types of change are allowed from segment to segment. Both the linear trend and periodic response can change and a discontinuity may occur. The linear trend can be continuous and the periodic responses can be the same for both segments. The periodic response can be continuous and the linear trends can be the same for both segments. Or the linear trend and periodic response can change at the same time, but in a continuous manner. These four types of threshold changes may happen for a single bore during the sampling period.

A challenging issue is the selection of an optimal model. In regression analysis, a model fit can be measured by its sum of squared errors; the latter decreases as the number of parameters increases. A criterion is needed so that not too many parameters are used in selecting competing models. A modified Akaike Information Criterion is adopted here, which penalises the sum of squared errors by a function of the number of free parameters.

The suitability of the threshold models has been examined by applying them to a series of bores from the south-west of Western Australia (see Key Findings below).

A third approach, called HARTT (hydrograph analysis of rainfall and time trends), separates the effect of atypical rainfall events from the underlying time trend and the lag between rainfall and its impact on groundwater is explicitly represented. Rainfall is represented as an accumulation of deviations from average rainfall. It can cope with irregularly spaced data and missing observations, which are common problems in hydrograph data sets.

The regression model is $Depth_t = k_0 + k_1 \times AMRR_{t-L} + k_2 \times time$, where Depth is the depth of groundwater below the ground surface, AMRR is Accumulative Monthly Residual Rainfall (mm), time is months since observations commenced, L is the length of time lag (in months) between rainfall and its impact on groundwater, and k_0, k_1 and k_2 are parameters to be estimated.

Accumulative Monthly Residual Rainfall is given by $AMRR_t = \sum_{i=1}^{t}(M_{i,j} - \overline{M_j})$ where $M_{i,j}$ is rainfall in month i (a sequential index of time since the start of the data set) which corresponds to the jth month of the year, $\overline{M_j}$ is mean monthly rainfall for the jth month of the year, and t is also months since the start of the data set. Figure RF.1 illustrates the calculation of AMRR for an example bore. The AMRR variable is calculated as the difference between the other two variables shown. The AMRR variable tends to have relatively low within-year fluctuations because, in calculating AMRR, the fluctuations in actual rainfall tend to be offset by seasonal variation in average monthly rainfall. An alternative measure to AMRR is Accumulative Annual Residual Rainfall (AARR; mm), which is given by $AARR_t = \sum_{i=1}^{t}(M_i - \overline{A}/12)$, where $\overline{A}$ is mean annual rainfall. Because $\overline{A}$ is a constant, the fluctuations in M_i are not moderated as they are for AMRR, so AARR has higher within-year fluctuations. For this reason, it is expected to be well correlated with data from bores with shallow groundwater levels (less than three metres deep) which typically have seasonally-fluctuating water tables.

For both AMRR and AARR, construction of the variables was based on a data set commencing in 1957. This pre-dates the earliest recording of depth to groundwater by 33 years, allowing long lag effects of rainfall on groundwater to be detected, if they occur. In cases where it produced models with a higher R^2 (most of which were shallow bores), AARR was substituted for AMRR in the regression model. The value of L was estimated separately for each bore by selecting the value that resulted in the highest R^2 for the regression. Thus L does not necessarily represent the lag until either the first impact or the largest impact of rainfall on watertable depth, but the lag that produces the highest statistical correlation.

2.3 Relating Water Levels to DEM-derived Variables

2.3.1 Within a sub-catchment

The approach developed by Salama et al. (1996) for the hydrogeomorphic analysis of regional spatial data (HARSD) was implemented for the Ucarro subcatchment (west of Katanning). The technique involves three main components: (i) stratifying the landscape into units with similar hydrological properties; (ii) constructing an hydraulic head surface (HHS) under the assumption that the groundwater surface is a smoothed version of the surface topography, and that a linear relationship exists between measured groundwater levels (bore readings) and surface elevation (as measured by a DEM) and (iii) a flow net analysis. The first stage attempts to classify the landscape into units with similar hydrological properties, by using a DEM to derive variables such as elevation, slope, curvature and break of slope; these are then grouped according to their statistical distributions. The classification may be arbitrary initially, then adjusted to reflect existing classifications such as landform maps, soil maps, etc. For the Ucarro study area, the correlation between groundwater levels and surface elevation suggested that no classification was necessary (see below).

2.3.2 Fitting Surfaces for the Entire Study Region

Two main data sets were compiled to fit surfaces for the entire study region.

The first was provided by AGWEST hydrologists, who extracted bores which had adequate time series data and were located where treatment or landuse changes would not bias the trends in the water levels (see Appendix I of Hodgson 2000). This data set only provided 568 bores (see Figure 3 of Hodgson 2000), many of which were clustered together spatially. Water levels were extracted by applying the trend as identified by AGWEST hydrologists (Method 1 above) to the most recent readings for each bore and calculating a water level for 16 January 1999 (see Hodgson 2000 for further details).

A second data set was derived to provide better spatial coverage of the entire region. Bore locations were extracted based on their apparent accuracy, by only including locations (eastings and northings in metres) which did not end in "00" or "50", the assumption being that the locations were then entered in the database with apparent accuracy better than 50m (see Hodgson 2000 for further discussion). Water levels were extracted for all bores which satisfied this criterion and which had at least one reading in 1990s. The most recent reading was extracted. These restrictions provided a data set of 1257 bores (see Appendix II of Hodgson 2000) with a reasonable spatial distribution throughout the study region (Figure 2) which is hereafter referred to as the "augmented data set". It was accepted that there could be biases from treatments and that bores with readings early in the 1990's may have changed significantly.

The following variables – referred to below as explanatory variables - were derived from the DEM and Landsat data to examine the variation in the depth to groundwater (depth), as follows:

- average height of drainage area
- average slope of drainage area
- flow slope (or downhill slope)
- height above nearest salt
- height above nearest stream
- elevation
- water accumulation (or upslope area) - and smoothed versions
- flow path length
- percentage upslope cleared area
- percent cleared (derived from water accumulation and upland cleared)
- total upslope cleared area

Plots between the explanatory variables and depth to groundwater were examined to see if there were any strong and obvious relationships.

A variety of statistical approaches were used to model the relationship between depth to groundwater and the explanatory variables, including standard multiple linear regression, generalized additive models and robust regression. Discriminant procedures were used in an attempt to stratify the catchments into more homogeneous units.

2.4 Land Monitor Salinity Risk Prediction

It was not possible to establish relationships between depth to the watertable and the explanatory variables to derive a regional hydraulic head surface (see section 3.3.3). The approach adopted by the Land Monitor project relating to salinity risk prediction is summarised.

The Land Monitor project aims to provide information about land condition, specifically salinity and the status of remnant vegetation, for the whole of the south-west agricultural region of Western Australia. It is a collaborative project involving Agriculture WA, CSIRO, CALM, DOLA, Water and Rivers Commission, the Department of Environmental Protection and Main Roads WA.

The Land Monitor DEMs and current salinity maps are used to create the derived explanatory variables listed above.

The methodology used for predicting salinity risk in Land Monitor is as follows:

- create the above derived variables – see, eg, Caccetta (1999);

- create a decision tree that relates the derived variables to salinity risk for the ground truth areas provided (Evans et al., 1996); and
- use the tree to extrapolate the decision tree model and predict risk areas.

It is important to note a significant change in the terminology used to describe the Land Monitor prediction maps. The term *predicted risk areas* is assumed to mean areas that will have high watertables in the future (ie discharge areas), parts of which are likely to become saline. While waterlogging may be common, predicted risk areas will not necessarily be salt-affected throughout.

Feature selection procedures were used to determine the optimal subset of DEM-derived variables for predicting salinity risk areas (see Section 4 of Evans, 2000a). The results were derived using 50% of the available ground data for training and the remaining 50% for testing. By reducing the number of variables, the aim is to reduce the size of the decision tree (hence rendering it simpler to interpret) and improve the accuracy and visual outlook of the risk maps that are produced. Three heuristic approaches were investigated: an information theory approach; a forward selection approach; and a simple forward selection approach (see Evans, 2000a, Section 4.1).

2.5 Flow Tube Modelling of the Effect of Recharge Management Strategies

As part of the need to estimate the impact of salinity on assets under different management scenarios, members of the WA Salinity Council R&D Technical Committee (RDTC) were given the task of preparing estimates of the effect of various recharge management strategies on the extent of dryland salinity development in the wheatbelt of Western Australia. This enabled the team involved in the SS2020 project to use its methods to affect salinity policy within WA.

The principal deliverables required of this modelling exercise were:

1. a graph of salt-affected land through time with and without the Salinity Action Plan - the approximate area of land for both scenarios under the following land uses was to be estimated: commercial saline, non-commercial saline, commercial woody perennials, remnants and nature conservation plantings, commercial perennial pastures and fodder shrubs, commercial annuals;
2. regional graphs of salt-affected land through time under different recharge reduction scenarios and possible methods whereby the reductions could be achieved; and
3. details of likely changes in salt-affected land severity categories through time and the consequences of these for R, D and E.

Fifteen sites were modelled; their locations are shown in Figure 1 of Clarke et al. (1999). The 15 sites can be divided into seven complete (or almost complete) catchments and eight hillslope sections. Land surface topography, basement topography, current water level and rate of rise data for these sites were provided by AGWEST staff.

Decisions on the parameters to be modelled, such as thickness and saturated hydraulic conductivity (K_{sat}) of the aquifer, and recharge to the aquifer, were made using published data where possible.

CSIRO Land and Water's (CLW) Flow Tube model was adopted for the modelling, based on a formulation provided by Warwick Dawes (CLW) and developed for the one-dimensional analysis of aquifers in the Liverpool Plains, Northern NSW (see Dawes, et al.(2000). This model is based on the numerical solution of the Boussinesq equation for a one-dimensional domain.

The Boussinesq equation is a second order differential equation stated in terms of the piezometric head of the aquifer, and involves the specific yield of the soil along the flow tube, the cross-sectional area of the flow tube, the saturated hydraulic conductivity and the net recharge function to the aquifer, dependent on space and time. The initial condition is given by water levels measured in the field through a series of piezometers positioned along the stream line of the particular flow tube. At the downstream end of the flow tube, where the watertable reaches the soil surface or where a creek is present, a constant piezometric head is imposed. At the uphill end of the flow tube, a no-flux boundary is imposed. Additional specifications are required for a consistent representation of the seepage face when the calculated piezometric heads reach the soil surface. In this case, the head in the model is set equal to the land surface elevation and the respective excess volume of water included between the calculated piezometric head surface and the land surface elevation is removed as saturation excess flow from the flow tube. A maximum threshold discharge across the soil surface is imposed during the simulations. If the calculated discharge exceeds the maximum allowed discharge, the pressure heads are once again set equal to the land surface elevation and the surplus volume is redistributed along the flow tube.

The soil of the study catchments is generally layered, consisting of a saprolite aquifer overlain by the low-conductivity pallid zone aquitard. The layered structure of the hydraulic conductivities has been accounted for by assuming a single effective conductivity over the whole cross section of the flow tube cells. The effective conductivity is estimated as the harmonic mean of the conductivities associated with the individual layers.

The governing differential equation is solved through an explicit finite-difference scheme for a number of model cells. The mass conservation of the model is checked at every time step. The simulations were carried out starting from the imposed initial condition over a time-period of 100 years with a 1-day time-step. The isochrons of the head surface (lines approximating the watertable at a particular time) were represented at five-year intervals.

Further details are given in Clarke et al. (1999).

The principal assumption made is that the Flow Tube model is one-dimensional. It was run as a 1m wide tube, and no allowance was made for convergence. Convergence may exacerbate the currently modelled extent of dryland salinity. While disregarding convergence was recognised to be unrealistic of actual catchment conditions, it was postulated that it would not significantly alter the relative impact of recharge reductions on extent of ‘salinity’. By contrast, running the model as a single flow tube, often along the drainage line (particularly for the sites classified as catchments), will tend to underestimate the effectiveness of treatments since catchments have the greatest inertia to change in salinity extent in this direction.

Estimates of the impact of treatment scenarios on the extent of dryland salinity were made by assuming that all land lying along a flow tube was potentially salt-affected when the watertable was within 1 m of the soil surface. All results are therefore represented as the percentage of the flow tube which has a watertable within 1 m of the surface.

In order to make the models representative of as wide an area as possible and to avoid a level of unsubstantiated and unnecessary detail, the simplest description of thickness and saturated hydraulic conductivity of the aquifer, and recharge to the aquifer, were selected. The model was calibrated by comparison of early (but not initial) modelled rates of rise of the watertable with the present rate of rise of the piezometric surface ‘averaged’ for the catchment.

2.6 Spatio-economic Assessment of Salinity Scenarios and Control Treatments

2.6.1 Overview

Two types of scenarios were examined: a "business as usual" scenario, which assumed minimal changes in farm management from current practices; and a number of "treatment" scenarios. The treatments constituted a variety of different agroforestry strategies for reducing recharge to the groundwater system. The analysis did not examine the benefits and costs of rehabilitating discharge areas with saltland agronomy. Nor were engineering works, such as pumping and draining, considered by the analysis. A planning horizon of 20 years was used for both scenarios.

Three case study areas were selected for the purposes of making an *ex ante* assessment of the economic impacts associated with each scenario. The treatments were developed and designated to each study area in consultation with hydrologists, agronomists, foresters and farmers. This consultative process also produced a set of criteria that were used to map the location of treatments within each study area.

The agroforestry treatments selected for analysis were:

- plantations of bluegums
- belts of bluegums with conventional agriculture in the alleys
- belts of maritime pines with conventional agriculture in the alleys
- belts of oil mallee with conventional agriculture in the alleys.

The modelling framework consisted of a spreadsheet simulation model for estimating economic impacts and a geographical information system (GIS) which served as a physical accounting tool. It supplied the area statistics required by the economic analysis. The GIS was also used as a means of mapping the distribution of economic impacts across a study area once the impacts had been calculated.

A spreadsheet was used to analyse the stream of annual costs and/or benefits associated with each scenario over a 20-year period. These calculations were based on physical information from the GIS together with economic data on the profitability of agriculture and treatments on an array of land management units (or soil types). These land units were fundamental to the analysis because their underlying productivity determined the size of costs imposed by encroaching salinity, the opportunity costs incurred by planting treatments on agricultural land, and the benefits from preserving land in a productive state.

The overlaying capabilities of the GIS were used to produce a two-way matrix containing estimates of the area of each land category that fall within each land management unit. The overlay maps included digitised maps of salinity (current and predicted), land management units, treatment locations, and various other layers of resource information such as perennial vegetation, main roads, and farm dams.

2.6.2 Business as usual scenario

The costs incurred under the "business as usual" scenario were equated to *future* losses predicted to be suffered as a result of expanding salinity, but did not include ongoing losses from land that is already saline in 2000. Both the economic costs to production and the extent of physical damage to farm buildings, farm dams, roads, and perennial vegetation were examined. Two important assumptions were made:

1. Costs imposed by the current level of salinity were considered to be "sunk costs" and netted out of the calculation. This assumption was made because the treatments examined by this analysis are likely to be incapable of reclaiming land that is currently saline.

2. In calculating production losses, the analysis allowed for the possibility of some adaptive management, by assuming that salt-affected land has some grazing value. This was considered important because it is reasonable to expect that ways will be developed to use salt-land more efficiently as the scale of affected land increases.

Further details of the methods and data used to calculate losses under the business as usual scenario are contained in Van Bueren et al. (2000).

2.6.3 Treatment scenarios

The net benefits of controlling salinity under each of the treatment scenarios were quantified by estimating production benefits and the extent to which infrastructure and off-farm assets were protected. The gains and losses from implementing a treatment were measured *relative to* the outcomes predicted to eventuate under "business as usual". The direct costs of treatments were included in the analysis, but other costs such as administration and extension of the programs were excluded. Furthermore, it was assumed that all treatments were implemented in the first year of the planning horizon.

The impacts of each treatment were estimated by quantifying the changes in profitability associated with each of the following categories of land:

A. Land planted to a treatment that is prevented from becoming saline because of the treatment.

B. Land planted to a treatment that is predicted to never be at risk from salinity.

C. Agricultural land adjacent to a treated area that is prevented from becoming saline.

D. Land adjacent to a treated area that is prevented from becoming saline and supports infrastructure or natural assets.

Profitability is expected to increase for land categories C and D, and possibly A, when a treatment is implemented. In most cases, profits for category B are expected to fall because forestry returns are generally less than those of agriculture. The aggregate impact of a particular treatment scenario was calculated by estimating the per hectare change in profitability associated with each category of land, together with the area of each category within a case study.

Further details of the methods and data used to estimate the economic impacts of each treatment scenario are documented in van Bueren et al. (2000).

2.6.4 Case study applications

Three study areas were selected. The southern-most study area was the Upper Kent catchment, comprising an area of 107,000 hectares. The other areas, each comprising 30,000 hectares, were located in the Woodanilling and Boscabel districts (see Figure S, Appendix C of van Bueren et al. 2000). Each study area has different hydrological characteristics.

Boscabel has two distinct systems: the western sediments and eastern rocky hills. The Beaufort River flats run north-west through the middle of the region. Much of the study area is underlain by a regional aquifer, meaning that the ground water table is unlikely to be responsive to control treatments. Approximately 12 percent of the study area is already salt-affected. The region has a mean annual rainfall of 500mm.

The Woodanilling case study area lies just 10km east of the Boscabel site but the landscapes are quite different. While Boscabel has a high potential for an increase in salinised area, the equilibrium level of salinity in Woodanilling is not expected to exceed 20 percent of the landscape (approximately 7 percent is currently salt-affected). Furthermore, the aquifers in this area tend to be localised, implying that groundwater levels could be more responsive to recharge control. The region receives 450mm of rainfall annually.

The Upper Kent catchment was chosen as a study site principally due to the large amount of salinity mapping and hydrological investigation that has already been undertaken previously. That work indicated that approximately 18 percent of the catchment is affected by salinity. The aquifer underlying this area is predominantly regional. Rainfall in this region ranges from 750mm in the south to 500mm in the northern part of the catchment.

The treatments corresponding to each study area are summarised in Table 1. Originally it was intended to analyse the impact of these treatments at two scales of implementation. However, in the absence of a hydrological model, this could not be achieved. Instead, only the "high-scale" of treatment was adopted for each area, as shown in Table 1.

The economic returns for each treatment were estimated using partial budgets for the "forestry enterprise" and gross margins for the agricultural component of the treatment. With the exception of the bluegum plantations (where there is no agricultural component), a joint treatment return was calculated by combining returns from the "forestry enterprise" with the agricultural gross margins according to the ratio of trees to alley land. The assumptions used to calculate forestry returns and agricultural gross margins are contained in van Bueren et al. (2000), together with a summary of the joint treatment returns (see Appendix A).

		Treatments details	
Case study	Treatment name	High scale	Very high scale
Upper Kent	Bluegums	Bluegum belts and plantations on gravel slopes	Bluegum belts and blocks on gravel slopes plus non-saline waterlogged soils
Woodanilling	Oil mallee	Oil mallee belts on suitable soils with annual crop:pasture rotation in alleys	Oil mallee on suitable soils with lucerne phase-farming in alleys
Boscabel	Maritime Pines	Pine belts on suitable soils with annual crop:pasture rotation in alleys	Pine belts on suitable soils with lucerne phase-farming in alleys

Table 1: The treatments examined in each case study area

2.7 Component Reports - Further Information

Clarke, C., George, R., Reggiani, P. and Hatton, T. 1999. The effect of recharge management on the extent and timing of dryland salinity in the wheatbelt of Western Australia. *Preliminary computer modelling.* Report to WA Salinity Council, June 1999.

Evans, F.H. 2000. Land Monitor salinity risk prediction. Dumbleyung and Mt Barker regions. *CSIRO CMIS Task Report* No. 2000/45.

George, R., Clarke, C., Hatton, T., Reggiani, P., Herbert, A., Ruprecht, J., Bowman, S. and Keighery, G. 1999. The effect of recharge management on the extent of dryland salinity, flood risk and biodiversity in Western Australia. *Preliminary computer modelling, assessment and financial analysis.* Report to WA Salinity Council, 19 July 1999.

Hodgson, G.A. 1999. Application of HARSD landscape classification and groundwater surface mapping techniques to study catchment at Ucarro. *Task Report* GAH 99-2.

Hodgson, G.A. 2000. A comprehensive analysis of borehole data from the SS2020 project area. *CSIRO CLW Technical Report* (in press).

Shao, Q., Campbell, N.A., Ferdowsian, R. and O'Connell, D. 1999. Analysing trends in groundwater levels. *CSIRO CMIS Technical Report* 99/37.

Shao, Q. and Hodgson, G.A. 1999. Statistical modelling for trends in groundwater level in Towerrinning. *CSIRO CMIS Technical Report* 99/110.

Van Bueren, M., Pannell, D. and Hodgson, G.A. (2000). Spatio-economic Assessment of Salinity Scenarios and Control Treatments. *CSIRO CLW Technical Memorandum (in press).*

3. Key Findings and Applications

Description of the Groundwater Data

Bore data were examined for location quality as well as date of last reading and time series information. Several data sets of differing quality were derived for use in modelling regional water level surfaces. It was found that very few bores had high-quality (+ or - 5m) location data. Out of approximately 3000 bores in the study area, only 154 had surveyed or DGPS location data. The vast majority of bores have positional data which are estimated by a range of different methods which have varying errors. The most common method of estimating locations is by reference to points taken from topographic maps or aerial photographs. In this case, the scale of map used will largely control the accuracy of estimation. For example, on a 1:100,000 topographic map, a 1cm error will equate to a 1000 m error, while on a 1:25,000 map, the same error will result in a 250 m error.

The SS2020 project established an integrated database for the study region containing water levels and trends where available (Hodgson, 2000). This included a GIS database which linked bore information to digital elevation data (and derived variables), soil information, land management units, and other spatial data sets. Maps were created of bore locations for bores with reliable location information (Figure 1) and for bores with trends identified (Figure 2). These clearly identify areas where there is little hydrological information and demonstrate the biases in the data that have been collected to date.

Two main data sets were compiled to fit surfaces for the entire study region.

The first was provided by AGWEST hydrologists, who extracted bores which had adequate time series data and were located where treatment or landuse changes would not bias the trends in the water levels (see Appendix I of Hodgson 2000). This data set only provided 568 bores (see Figure 3 of Hodgson 2000), many of which were clustered together spatially. Water levels were extracted by applying the trend as identified by AGWEST hydrologists to the most recent readings for each bore and calculating a water level for 16 January 1999 (see Hodgson 2000 for further details).

A second data set was derived to provide better partial coverage of the entire region. Bore locations were extracted based on their apparent accuracy, by only including locations (eastings and northings in metres) which did not end in "00" or "50", the assumption being that the locations were then entered in the database with apparent accuracy better than 50m (see Hodgson 2000 for further discussion). Water levels were extracted for all bores which satisfied this criterion and which had at least one reading in 1990s. The most recent reading was extracted. These restrictions provided a data set of 1257 bores (see Appendix II of Hodgson 2000) with a reasonable spatial distribution throughout the study region (see Figure 4 of Hodgson 2000) which is hereafter referred to as the "augmented data set". It was accepted that there could be biases from treatments and that bores with readings early in the 1990's may have changed significantly.

A full description of the groundwater databases can be found in Hodgson (2000).

3.2 Trends in Groundwater Levels

3.2.1 Segmented regression approach

The method outlined in Section 2.2 identifies the linear trend in groundwater levels as well as the seasonal (periodic) response for different time segments. Shao et al. (1999) give a number of examples.

The approach was applied and examined in detail for the Towerrinning region of the SS2020 study area (Shao and Hodgson,1999). A software module has been developed which runs under S-Plus; the method has been tested by a number of AGWEST hydrologists.

The examples presented in Figure 3(a) and 3(b) illustrate the power and flexibility of the segmented regression approach for describing borehole data. The approach provides different linear trends for different segments of the data, and at the same time estimates the amplitude(s) of the seasonal response(s) in the data. However, the method does not provide an estimate of long-term trend, nor does it account for rainfall during the period of observation.

There are several limitations to the segmented regression approach, including:

- it requires that the number and position of join points or break points be specified separately from the actual statistical estimation;
- the attributes of each segment have to be related to management practices or to changes in rainfall in subsequent analyses; and
- the method does not estimate explicitly the underlying long-term trend in groundwater levels, nor does it take into account rainfall during the period of observation, and so makes a limited contribution to the forecasting of future levels.

The ability to separate the linear and cyclic trends in hydrographs, and to separate segments during which both trends are constant, has the potential to permit a better understanding of landscape response to hydrological changes such as differences in rainfall pattern and land use. However, because this analysis is tied to the detail of the historic pattern of these hydrological changes, it is of less use to predict what the hydrograph trend will be in the future. If the thresholds are caused by changes in rainfall rather than land use, linear trends fitted by eye and experience may be more useful for prediction than the trend analysis if the hydrograph covers a long period, because of the likelihood (or possibility) that the pattern of rainfall changes will continue into the immediate future. The HARTT method of determining hydrograph trends and of separating the effect of treatments from rainfall is potentially useful since the rainfall record is much longer and complete than are bore records, though the impact of assigning rainfall to a bore by interpolating from the nearest stations needs to be assessed.

3.2.2 Hydrograph analysis of rainfall and time trends

Figures 2 - 4 illustrate results for three individual bores: deep, moderate and shallow respectively. They show trends typical of other bores of similar depth. The deep bore (Figure 2) has a rapidly rising water table. The pattern of rise is well captured by the estimated regression model (R^2 = 0.97). The rise does not follow a simple linear trend, but the deviations from linearity are explained almost completely by the *AMRR* variable. The moderately deep bore (Figure 3) has a lower rate of rise (less than half that of Figure 2). *AMRR* and time again explain groundwater level well, although this time there is some unexplained variation (R^2 = 0.80). The data deviates substantially from a smooth linear trend, but the main variations are captured well by the model. The shallow bore (Figure 4) has a slightly falling underlying trend. R^2 is lower again than for the deeper bores (0.71), though again the broad deviations from linearity are captured by the rainfall variable (*AARR* in this case). As well as illustrating the quality of fit from the model, the analyses of these three bores also demonstrate the potential biases in estimation of groundwater trends that occur in a simple regression model based only on time. Using the HARTT model, the estimated underlying time trends of groundwater change are 0.022, 0.0079 and –0.0033 m/month for the three bores, whereas the simpler linear model results in estimates of 0.022, 0.0071 and –0.0046 m/month.

For the bores analysed to date, there is considerable variation in the estimated parameters for the *AMRR* and *AARR* variables. In some cases, 1 mm of above-average rainfall results in a 1 mm rise in groundwater levels, while in others, the rise is considerably more than 1 mm. As would be expected, the lower parameters tend to be for deeper bores and very shallow bores, although the variation at all depths is high. The rates of groundwater rise tend to be greater for deep bores than for shallow bores. This partly reflects the greater role of discharge in landscapes with shallow bores; this discharge is effective in removing water from the system and lowering the rate of groundwater rise. Indeed, for bores with initial depths below 5 m, the average rate of groundwater rise is approximately zero.

The model tends to fit groundwater data more precisely for deeper bores. Conveniently, it is more important to be able to predict groundwater rise for deeper bores. Indeed, for farmers with shallow bores, where the approach has greatest difficulty in explaining data series and predicting trends, there is little use for such information for these bores. Where the information is most needed, the quality of model estimates appears high.

The method has a number of clear strengths:

- simplicity
- separation of underlying time trend from unusual or atypical rainfall effects
- consistency of results with intuitive hydrological explanations - high degree of fit to data for bores with depths greater than 5 m
- capacity to make predictions of depth to groundwater for a period ahead equal to the length of the lag estimated in the model
- capacity to make less precise predictions even further ahead based on forecasts of rainfall for the period

Some limitations can also be identified:

- without the addition of further variables to the model, the quality of explanation of data for shallow bores is not high
- trend results depend on calculated mean levels of rainfall. If means have changed during or after the period of rainfall data, the power of the model to forecast future groundwater levels is reduced. Changes in the within-year distribution of rainfall may also reduce predictive power.
- depth changes over time, so *L* and the coefficients of *AMRR* and *time* would actually change during the period of estimation. This requires a more sophisticated approach to estimation than has been undertaken to date.
- for time series data like these, serially correlated error terms are common, and this has not been allowed for in the estimation process.

The HARTT approach is simple to apply with standard regression methods. It provides high-quality fits to observed data in all but shallow bores (for which trend estimation is of less interest in any case). It allows the separation of atypical rainfall events from the underlying time trend. Results are highly consistent with hydrological expectations. The method seems likely to have wide applicability in situations with rising groundwater trends. An additional use is to predict what groundwater levels would have been in the absence of actual changes in land use affecting additions to groundwaters.

3.3 Hydraulic Head Surface

This study has shown that a good relationship can be established between average depth to groundwater and surface elevation for the deep bores at Ucarro for a stratified data set. The success of the approach depends heavily on establishing a suitable stratification (or hydrogeomorphic classification) in which the relationships are linear.

It was the intention of the SS2020 project to produce an hydraulic head surface (groundwater level) for the entire study area, based on the relationship between the bore water levels as described above and variables derived from the Land Monitor DEM. With this surface, groundwater trends could be projected forward in time (to 2020, for instance) to estimate the areas at risk of salinity under current climate and land management. Modified trends reflecting recharge reductions associated with alternative land uses could also be projected in this manner.

The study found that there are currently insufficient reliable hydrological data to construct meaningful water levels surfaces for the entire study region. A local surface could be produced with some degree of confidence only where there is reliable information and an effective stratification. There is clearly a need for more accurate location data for bores. The bore data that are available are strongly biased towards areas of very shallow groundwater and some are even in discharge zones. More bores are required higher in the landscape for the hydraulic gradient to be captured and regional trends to be fully understood. There are few bores where trends can be interpreted, and monitoring of existing bores should be continued to improve the long-term trend approximations. The methodologies developed, including the HARRT approach, should be explored further where good bore data are (or become) available.

3.3 Relating Water Levels to DEM-derived Variables

3.3.1 Within a sub-catchment

Groundwater levels were obtained from all 29 deep piezometers in the Ucarro subcatchment for the month of May 1998. The average trend (absolute value) in the 29 bores was 0.04m/y.

Linear regression was performed for surface elevation in metres AHD (Australian Height Datum) against monthly average depth to groundwater (Figure 4), with and without the removal of two spurious points. The exclusion of the two points changes the r^2 from 0.25 to 0.84 (n=27, dotted line); the latter more clearly shows the general dependence of groundwater depth on topography. This plot contrasts with those in Figures 2.1 and 2.2 of Hodgson (2000); the two spurious points are not evident when monthly average water level is plotted against surface elevation, as recommended by Salama et al. (1996). While the resulting mathematical relationships can be shown to be equivalent, there is a spurious correlation between groundwater elevation and surface elevation which can obscure the true closeness of the relationship with depth to groundwater.

In addition to bore data, salinity maps were used to define areas of discharge (groundwater close to surface) to provide more control on the HHS. It is reasonable to assume that where there is saline discharge then the groundwater is effectively in contact with the surface. A salinity map for the Ucarro catchment was obtained and sixteen points within well-defined areas of salinity were chosen to supplement the bore data. In reality, discharge can occur when groundwater levels are within 2 metres of the surface, because evaporation and capillary rise can facilitate discharge in such situations. For this reason, it was decided that the groundwater levels for all the discharge points would be set to one metre below the surface elevation at the discharge points.

Once a HHS has been constructed, it can be used to map the discharge area for the region of interest. Hodgson (2000) shows the mapped discharge zones for Ucarro for May 1998 and the change in discharge area given a universal rise of 1m in groundwater. The most significant point to note is that the discharge area almost doubles with only a 1 metre rise in groundwater.

For a single sub-catchment like Ucarro, a good relationship can be established between average depth to groundwater and surface elevation. However, at larger scales, where multiple catchments and diverse landscapes are involved, such as the SS2020 study area, pure elevation-driven predictions are unlikely to be this accurate unless a more repeatable method for partitioning the landscape can be established.

This simple case study has shown that the vetting of anomalous data is critical for accurate prediction. The inclusion of data from known discharge areas has the potential to improve the fit. Since these extra points represent grouped data in a statistical sense, they cannot simply be included as extra points in the regression. Moreover, a high percentage of discharge points will clearly influence the slope of the line. However, it would seem sensible to use the salinity data to improve the spatial representation of the data.

There is a need for validation and testing of the predictions. This could be achieved through "leave some out" techniques, in which a portion of the input points are reserved for validating the predictions. There will also be a significant number of bores within the SS2020 study area which do not have sufficient trend data to be included in the prediction of trends, but which could be used to test predictions for particular dates. It is anticipated that if the trends in groundwater levels can be accurately identified, then a prediction of a water level for a bore can be made for any given date. Thus predictions into the future can be made with a reasonable estimate of error. The

HARTT method of Ferdowsian et al. has the potential to provide us with a method that can do this with good precision.

3.3.2 Fitting surfaces for the entire study area

The next part of this report investigates the use of DEM-derived variables in establishing a HHS. The concept of "relative elevation" is examined to address the problem of large-scale variation and the reliance on pure elevation within the regression. If similar relationships between relative elevation and groundwater depth can be identified, then this should prove a more robust and statistically acceptable method. This method is applicable to series of local aquifers, each of which has a separate discharge area.

Using the DEM-derived variables listed in Section 2.3.2, a model was derived for the Kent area which provided a fit to the bore data (R^2=0.56) such that the resulting hydraulic head was plausible. However, for the Boscabel region, the fit to the bore data yielded R^2 values as low as 0.003; the resulting maps of hydraulic head surface made no real sense. For the other areas, the results lie between these two extremes -- in each of these cases (Woodanilling, Gnowangerup and the South Stirlings), the resulting hydraulic head surface has a number of plausible features but also has a number of problems such as excessively steep gradients.

Two points should be noted:

- In some areas, up to one third of the bores have been effectively removed from the modeling process by the robust fitting procedure -- if the depths of these bores are accurate, then clearly the hydraulic head surface will be erroneous at these points.
- In some areas (Boscabel is the worst instance), the fitted robust regression model is driven by a very small number of points -- if these points are inaccurate, then the model will likewise be inaccurate.

Reasons why modelling the hydraulic head may not be possible include:

- the explanatory variables may not reflect the physical processes that determine groundwater depth -- this is certainly at least partly true, since groundwater depth is determined by underground local features and time since clearing that are not reflected in the data;
- the poor quality of the bore data base may preclude effective modelling over large areas.

There is no obvious explanation as to why the approach gave some sensible results in the Kent but not in Boscabel. Considerably more is now known about the state of the borehole data than at the beginning of the project. However, none of the bores in the Kent region has a surveyed location and there is no compelling reason to suppose that the quality of the Kent data is superior to that of the other regions. All of the bores in the Woodanilling area are quite shallow and there is a sampling bias in that the bores are located where there is an imminent danger of salinity (or to monitor a treatment in such a case), so there are no bores in the higher parts of the landscape. The result of this is that the fitted hydraulic head was shallow everywhere.

In the Toolibin area, an attempt was made to use the DEM-derived variables to partition the landscape into units such that the depth to groundwater could be more effectively predicted within each unit. This was unsuccessful. An area with deep groundwater (the eastern area) could not be separated from an area of shallow groundwater (the southern area) on the basis of the DEM-derived variables. However, the areas were clearly better fitted by separate regression models. It has been suggested that elapsed time since clearing is a variable that could help distinguish between these areas; unfortunately it is not widely available.

In summary, the application of a regression-based approach at the regional scale failed to produce a hydraulic head surface of sufficient reliability at the regional scale. This is an important finding as it casts doubt on the suitability of using a few bore records to extrapolate to larger areas, a common technique used to estimate risk from salinity.

3.4 Land Monitor Salinity Risk Prediction

Table 2 shows the accuracies for areas in the Dumbleyung and Mt Barker Landsat TM scenes where ground data were available.

ground data area	**accuracy**	
	salt	**not salt**
Toolibin	91.5	87.5
Broomehill	70.2	92.9
Cranbrook	81.6	73.6
Tambellup	77.0	82.7
Kent	68.1	83.6
South Stirlings	85.5	39.9

Table 2: Accuracies for areas in the Dumbleyung and Mt Barker Landsat TM scenes where ground data were available

The following results should be noted:

- better results are achieved in the Dumbleyung region when only the Broomehill ground data are used;
- the best results for the Cranbrook and Tambellup areas are achieved when ground data from the Cranbrook, Tambellup and the Kent catchments are combined;
- the best results for the Kent catchment area are achieved when only local ground data from the Kent are used; and
- results for the South Stirlings area are poor - the use of regional training data produces large errors of commission but the use of local training data produces large errors of omission.

The South Stirlings should be modelled separately to the remainder of the Mt Barker scene; additional training data will be required to accurately map the South Stirlings area. Examination of the extrapolated risk maps south of this point suggests that risk areas are also being over-estimated in the higher rainfall zones; additional training data are also required for these areas.

The prediction map for the Dumbleyung region has been produced using the decision tree classifier trained on only the Broomehill data. The accuracies achieved in the two areas with ground data were acceptable and further ground-truthing has been performed in the Boscabel and Towerinning areas. Detailed assessments of the salinity risk maps at specific geographically located areas were noted for both the Boscabel and Woodanilling areas. In general:

1. The overall impression of the risk maps is that they are acceptable.

2. Errors in local valleys are of two types: some should show less risk extending up the valley and some should show more.

3. Broad valley floors are exclusively mapped as risk. However, there are many areas where the groundwater is near-surface (ie. 1-2m) within the broad valleys that may be slightly higher in elevation than the true valley floor (less than 1m height differences can be significant) that will

not become salt-affected. These areas may currently be supporting a healthy crop or pasture cover.

While the results are acceptable in the Kent, Cranbrook and Tambellup catchments, risk areas are severely over-estimated in the South Stirlings and southern higher rainfall areas. More training data are required to correct these problems. The current version of the map for these areas will not be released until it is significantly improved.

It should also be noted that in the North Stirlings area where accurate DEMs were not available at the time of processing and 10m contour data were used in their place, the majority of the broad valley floors have been mapped as risk areas. This problem may be fixed when the more accurate Land Monitor DEMs are incorporated into the analysis.

In conclusion, the method implemented by Evans (2000) was able to reproduce and extrapolate locally developed maps based on hydrogeological expertise in a wholly repeatable and reliable manner. However, given the lack of data and hydrogeological understanding for the sedimentary geology south of the Stirling Ranges, the method could not be applied there. For the remaining 2.13 m ha mapped, 0.65 m ha (31%) is expected to be at risk of salinity at equilibrium. The maps of salinity risk form the basis for the economic analyses and evaluation of assets at risk.

While maps of salinity risk at equilibrium do not provide the predictive functionality of the method originally intended in SS2020, and do not lend themselves to scenario modelling, they are useful for risk assessment and identifying assets at risk if we continue with current land uses in the region. They have proven to be the only reliable approach for determining risk at a broad scale.

3.5 Flow Tube Modelling of the Effect of Recharge Management Strategies

A number of different recharge scenarios were modelled at each site in order to examine the impact of each individual management strategy. In addition, the trend of the change in salinity extent with increasing recharge management was examined to see if there were any differences in the way individual sites responded to increasing recharge management. The five recharge reduction levels and likely landscape treatment scenarios are listed in Table 3.

% of base case	Scenario
200	Episodic recharge, higher rainfall, together with do nothing different (worst case)
100	Do nothing different from today (base case)
50	Assumes SAP2 totally implemented; only likely in high rainfall zone (perennials?)
25	SAP2 + widespread installation of groundwater pumps and drains
10	Deep-rooted, perennial plants “effectively planted everywhere”

Table 3: Recharge scenarios modelled

The generalised results are shown in Figures 5 and 6.

Detailed cross sections of each site modelled for each recharge scenario, showing groundwater levels (isochrons) at five-year intervals for 100 years, are included as Appendix 2 of Clarke et al. (1999). Graphs of the amount of recharge (mm year^{-1}) plotted against the proportion of the flow tube where water levels are within 1m of the land surface are shown for each catchment in Figure 2 of Clarke et al. (1999), and for each hillslope in Figure 3 of Clarke et al. (1999). Graphs of the proportion of recharge compared to base case recharge for each scenario, plotted against the ratio of the length of the flow tube where levels are within 1m of the land surface for each scenario to the length for the base case, are shown for each catchment in Figure 4 of Clarke et al. (1999), and for each hillslope in Figure 5 of Clarke et al. (1999).

There appear to be three patterns of response by whole catchments (see Figure 4 of Clarke et al., 1999). Skeleton Rocks shows a very rapid response to changes in recharge; North Baandee, Toolibin and Welbungin show an intermediate response; and Merredin, Mills Lake and Towerrinning have a very sluggish response. This difference in performance in relation to recharge management is thought to be driven, in part, by geomorphological differences. Skeleton Rocks valley has a dominantly convex landform (although it is also has the lowest recharge), while the intermediate group are essentially flat, and the “sluggish” group have a more complex land form with a number of concave areas within the profile. It may be that there is no ‘significant difference’ between the latter groups.

Figure 5 shows lines drawn to represent the behaviour of the flat and the concave groups of catchments. It is expected that most catchments will behave within the envelope formed by the shaded area between these two lines. Convex landforms may follow the pattern of Skeleton Rocks.

While Figure 5 summarises the likely result of change in area of salinised land induced by management of recharge, it does not address the change in onset time caused by management of recharge. It would be possible to model the latter effect using the Flow Tube model, but as a constant time has been modelled (100 years) here, a constant proportion of salinised flow tube would need to be modelled, to establish any meaningful relationship. This would require more iterations of the model. However, examination of the isochrons (the five- year interval water level lines) in the graphs in Appendix 2 of Clarke et al. (1999) give some indication of this effect.

There is an essentially continuous range of behaviours for the hillslope sites (see Figure 5 of Clarke et al., 1999), although most respond as expected when the recharge is reduced below the base case, by changing slowly at first and then increasing in rate of response as recharge is reduced. The exceptions are Arrowsmith, which has a high saturated hydraulic conductivity and is very flat, and Narrogin, which is relatively steep with only a small proportion of the slope discharging. The reasons for this scatter of performance are as yet unresolved.

Because the parameters which are controlling the variable response at the hillslopes modelled are yet to be determined, it is suggested that attention should be focused on the results from the modelling of the seven complete catchments (Figure 5).

3.5.1 Computer modelling, assessment and financial analysis of the effect of recharge management on the extent of dryland salinity

Following the presentation of the results in the previous Section to the State Salinity Council (SSC), a follow-up study was commissioned to review a range of treatment scenarios which would allow the SSC to determine which impacts may reduce the extent and impact of dryland salinity in Western Australia. Intervention at high, medium and low rainfall sites within the agricultural area Were studied. The intervention scenarios were based on current treatments and were to be optimistic. The study also addressed the implications of the results of the modelling on biodiversity, water resources and infrastructure; and to undertake a financial analysis of the cost of the scenarios if applied across the agricultural area, and the gap between the costs to farmers and the costs to the wider community of the intervention.

Three sites were chosen to be representative of the eastern, central and western agricultural area. The catchments selected were North Baandee (rainfall 330 mm year^{-1}) located ~50 km northwest of Merredin; Toolibin (rainfall 400 mm year^{-1}) located ~20 km southeast of Wickepin; and Date Creek (rainfall 600 mm year^{-1}), located ~20 km southwest of Darkan (Figure 1). It must be noted that these catchment do not explicitly represent other agricultural areas nearby the chosen catchment, nor do they represent different hydrogeological areas such as the Perth and Bremer Basins. Appendix 2 of George et al. (1999) lists the hydrological parameters used in the Flow Tube model for each of these catchments, and the detailed methodology is described in Appendix 3 of George et al. (1999).

At each of the selected catchments, three levels of recharge management were modelled in addition to the current "do-nothing differently" situation (base case). The three levels of recharge management were based on currently available systems already being established by farmers in the regions. The options were chosen to be both optimistic and realistic. Each of the three levels of intervention were chosen to develop, as practically as possible, on "intermediate" recharge reductions reported by Clarke et al. (1999).

The results of the groundwater modelling compared to the base case are presented graphically in Figure 6 and an interpretation is presented in Figure 7 Additionally, for each of the 11 management changes modelled within the three catchments (Table 4), one cross-section showing the depth to groundwater at five-year increments over a period of 100 years was produced, and model output is also available for several (medium) cases modelled for 300 years and for sensitivity to changes in permeability. The results are summarised below, and are presented in Appendix 4 of George et al. (1999).

	Landscape changes to the water balance		
Location	**Low**	**Medium**	**High**
North Baandee Eastern Wheatbelt 330 mm year^{-1}	Two rows of oil mallee alleys (2 x 1.5 m) over 100% of landscape at a nominal belt spacing of 100 m. The alleys comprise high water use annual crops & pastures.	Sandplain soils (10% catchment) planted to tagasaste; Two rows of oil mallee alleys (2 x 1.5 m) over the rest of the landscape at a nominal belt spacing of 50 m.	Medium intervention plus groundwater pumping at two locations at the break of slope and in the valley floor
Toolibin Central Wheatbelt 400 mm year^{-1}	Sandplain soils planted to tagasaste (10 %) plus phase farming system over the remainder (90%), consisting of 5 years lucerne, 5 years of cropping and pasture	1. Sandplain soils planted to Pines (10 %) plus a phase farming system over the remainder (90%, 5 years lucerne, 5 years cropping and pasture), except for two blocks of oil mallees at the break of slope and near the saline area (each 10% of the area). 2. Sandplain soils planted to Pines (10 %) plus a phase farming system over the remainder within the matrix of an oil mallee alley system (50 m spacing)	Medium intervention plus groundwater pumping at two locations at the break of slope and in the valley floor
Date Creek Western Wheatbelt 600 mm year^{-1}	1. Commercial farming of eucalypts or pines in blocks over 30% of the upper slopes. The remainder comprise optimum water use annual crops and pastures. 2. Commercial farming of eucalypts in plantations immediately adjacent to the saline land. The remainder comprise the current system of annual crops and pastures.	Commercial farming of eucalypts in blocks over 30% of the upper slopes plus a phase farming system over the remainder (70%, 5 years lucerne, 5 years cropping and pasture)	Medium intervention plus groundwater pumping in the valley floor (one well).

Table 4: Treatments for three catchments selected for management modelling

Modelling conducted for **North Baandee** over a period of 100 years showed that:

- for the base case, current management systems caused 89% of the flow tube to develop a shallow watertable (seepage);
- establishing tagasaste on the sandplain and an 100 m spaced oil mallee alley system based on annuals (low case) reduced the wet length to 89% of the base case and watertable levels declined beneath the sandplain;
- establishing a 50 m oil mallee alley system based on annuals (medium case) reduced the wet length to 58% of the base case; and
- including a groundwater pump with the 50 m alley system (high case) reduced the seepage length to 21% of the base case -- it is likely that the impact of the pump was overestimated.

However, despite these encouraging results, modelling conducted over a 300-year period showed that while the systems bought time (25-60 years), most of the gains achieved, by all except the pumping system, were lost.

Additional modelling to test the sensitivity of the model to changes in currently measured rates of saturated hydraulic conductivity (K_{sat}) indicated that doubling K_{sat} made almost no difference to the final outcome. Increasing K_{sat} by five times reduced the saturated length of the flow tube to 37% from 58%, and increasing K_{sat} by an order of magnitude reduced the saturated length of the flow tube to almost zero.

Modelling conducted for **Toolibin** over a period of 100 years showed that:

- for the base case, current management systems caused 94% of the flow tube to develop a shallow watertable (seepage);
- establishing tagasaste on the sands and phase farming elsewhere (low case) reduced the wet length to 66% of the base case and watertable levels declined beneath the sandplain;
- establishing the previous system, while adding two blocks of trees (medium case 1) only reduced the wet length to 65% of the base case (the lower block of trees was swamped - killed - by rising groundwater) but replacing the blocks with 50 m oil mallee alleys (medium case 2) reduced the wet length to 35% of the base case; and
- including a groundwater pump with the 50 m alley system (high case) reduced the seepage area to 6% of the base case, while taking away the two blocks and replacing the lower one with a pump only reduced the wet length to 51% of the base case -- it is likely that the impact of the pump was overestimated.

Modelling conducted over 300 years on medium case 1 (pines and phase farming) showed that while the systems bought time (over 30 years), there was no long-term gain.

Modelling conducted for **Date Creek** over 100 years showed that:

- for the base case, current management systems caused 72% of the flow tube to develop a shallow watertable (seepage);
- establishing commercial trees on the upper 30% with annuals-based agriculture (low case 1) reduced the wet length to 64% of the base case, while only establishing one block of trees adjacent to the seepage area (no trees upslope - low case 2) reduced the length to 61% (in this case the groundwater rises through the trees when the trees do not access the groundwater, and remain near to the surface [< 3 m] when they are allowed to withdraw groundwater);
- establishing the same commercial upper-slope system described previously, while replacing the annuals with phase farming (medium case) reduced the wet length to 57% of the base case; and
- including a groundwater pump (high case) reduced the seepage area to 53% of the base case.

Modelling conducted over 300 years on the medium case (commercial trees and phase farming) showed that there was an additional gain of 16% (41% of base case) because groundwater levels were still declining after 100 years.

Figure 6 summarises the impact of all of the management changes on the length of the flow tube that has a watertable within 1 m of the surface. The Figure shows that low levels of intervention reduce recharge relative to the base case by a maximum of 50%; that medium levels of intervention reduce recharge to a maximum of about 30%; and that only medium and engineering systems (or effectively full reafforestation) reduces levels towards pre-clearing situations.

In most cases at North Baandee and Toolibin, recharge management buys between 25 years and greater than 60 years, although at equilibrium, these gains are eventually eroded by slowly rising groundwaters.

Economic Analysis

The methods used to analyse the economic costs and benefits of each of the recharge management strategies were independent of those described in Section 2.6. The main component of SS2020 used a GIS framework for analysing economic impacts, whilst the flow tube results were evaluated using a 50-year discounted cash flow analysis. The proposed strategies were assumed to be implemented over the first 10-year period. Other assumptions are listed in Appendix 5 of George et al. (1999). Table 3 of George et al. (1999) presents a summary of the results of the financial analysis. The results are considered to be indicative of the magnitude of costs and benefits but not sufficient as a basis for policy formation because:

- The analysis was limited to the on-farm benefits and costs.
- It was assumed that the length of saturated flow tube was salt-affected, whilst in reality the groundwater modelling is only capable of predicting where the water level is less than one metre from the surface.
- No allowance was made for production off salt-affected land.

3.5.2 Impacts of Salinity on Flood Risk

George et al. (1999) present details of a flood risk study of the Blackwood Catchment which investigated the impact of increasing areas of saline land on streamflow, and the impact of increasing areas of saline land on salinity levels in the Blackwood River (see also Bowman and Ruprecht, 2000).

A runoff routing model was developed for the Blackwood catchment and calibrated using actual flow events and data from five gauging stations within the catchment. Data from the flood event of January 1982 was incorporated into the model to investigate the potential impact of increased areas of saline land on streamflow. Analysis of data from experimental catchments indicates that runoff increases not only as a result of clearing, but also as a result of increasing areas of seepage (or "waterlogging") due to rising groundwater levels. Runoff stabilises once the expression of groundwater reaches equilibrium. Therefore, the following assumptions were implicit to the model:

- levels of clearing do not change (that is, present levels apply); and
- the change in area of saline land corresponds to the change in seepage area.

The effect of increasing areas of saline land was incorporated into the model by decreasing rainfall losses and increasing runoff parameters for the 1982 flood event (all other model parameters remained the same). The following three prediction scenarios were run using the model, assuming the full extent of groundwater expression:

- the area of saline land doubles throughout the catchment;

- the area of saline land doubles to Bridgetown, and trebles east of Bridgetown; and
- the area of saline land doubles to Bridgetown, trebles east of Bridgetown, and quadruples within the Arthur and Beaufort River systems.

As the percentage of cleared land affected by salinity is currently estimated to be 10% in the upper Blackwood, the model uses an upper limit of 40% for this area.

Results for the Beaufort River at Manywaters (GS 609015) suggest that there is a direct increase in flood flow with increasing area of saline land. Therefore it is possible that flood flows could quadruple in the upper Blackwood if the percentage of saline affected cleared land increases to 40%. Results for the Blackwood River at Darradup (GS 609025) in the lower reaches shows that flood flows could treble if the area of saline land trebles; however, the effect of the quadrupled saline land in the upper reaches is reduced.

A simple regional salt export model was used to estimate future stream salinities. The Blackwood Catchment was divided into eight rainfall zones, for which runoff and salt load parameters were assigned for "forested" and "fully cleared" conditions. Present parameters were based on data from Blackwood gauging stations and nearby catchments.

Future predictions were made by altering the "fully cleared" parameters. It was assumed that the area of saline land doubles in order to calculate the runoff parameters. The load parameters were calculated from groundwater recharge and groundwater salinity. Results suggest that salinity levels may increase by more than double, especially for extensively cleared catchments within the lower rainfall zones. Figure 4 of Paper 2 of George et al. (1999) shows the long-term salinity trend for the Blackwood River at Darradup (GS 609025), with the predicted salinity level at groundwater equilibrium. A range is provided for the predicted level, as an indication of the variation from year to year due to climate variability.

The general conclusions from this study are:

- If there is a three- to four-fold increase in the area of the wheatbelt with shallow watertables, there is a three- to four-fold chance of increased flood risk.
- Similarly, salinity levels may increase by more than double, especially for extensively cleared catchments within the lower rainfall zones

3.5.3 Impacts of Salinity on Biodiversity

Paper 3 of George et al. (1999) presents details of the impacts of salinity on biodiversity.

Following examination of the low, medium and high intervention simulations for the eastern, central and western catchments, the following generalities emerge.

The impacts of low intervention in the eastern and central wheatbelt are that most or all of the wetland, dampland and woodland communities on the heavier soils in the lower half of catchments will be lost. These include naturally saline systems that do not tolerate waterlogging (e.g. gypsophyllous ecosystems), as well as black morrell, salmon gum and gimlet woodlands. These constitute over half of the plant community types in the region; hundreds of plant species and a large but unknown number of invertebrates are at very high risk in this case.

In the wetter, western edge of the wheatbelt, there are generally fewer endemic species limited exclusively to the lower parts of the landscape. Nevertheless, given that there are so few examples of some of these communities left after clearing, concern is equally high. The fresh water lake, wetland and dampland systems are greatly at risk to salinity and waterlogging and these will be lost where they are in the lower half of catchments. Of equal concern are the wandoo, flooded gum and mallet reserves in valleys, given their rarity in this region. Generally, the impact will be lower extinction risk than the drier regions, but greater losses in terms of populations and genetic diversity within species.

On the basis of the hydrologic simulations of the eventual extent of groundwater discharge in these regions, the heathland and and shrubland communities characteristic of intermediate slopes and of particularly high biodiversity (e.g. duplex soil mallee and grey sand heaths) are also at risk.

The only remnant ecosystems not at risk to salinity in these regions under low intervention are upland communities on laterite or sand ridges.

Hydrologic simulations indicate that the impacts of medium intervention on biodiversity in the eastern and central wheatbelt are largely the same as for low intervention on valley floors, apart from perhaps buying some time. However, it is not clear what would be done with this additional time in the case of ecosystem protection. More is potentially gained in the western wheatbelt, if medium level intervention is implemented very quickly.

There is apparently a significant risk reduction to remnants on midslope positions (as described above) under medium intervention across the wheatbelt.

The incorporation of high-intervention groundwater pumping appears necessary to save the valley systems described above. This includes achieving biodiversity objectives in the Nature Conservation Recovery Catchments.

In saying this, it is recognised that trade-offs will have to be identified between those systems saved by pumping and those portions of the landscapes affected by disposal of that water. In this regard, it is worth recognising the eventual (or even current) health of the Avon and Blackwood Rivers under the do-nothing option. Salt loads, concentrations and flooding will increase into the future. Key inland ecosystems can only be saved through engineering intervention; it may be that using these rivers to dispose of drainage water to save valley remnants should be considered.

The fragmentation of ecosystems and its consequence for their persistence is well-understood, and much activity aimed at maintaining biodiversity is dedicated to linking up remnants. As the system salinises, options for re-integrating remnants decreases; what would have been potential corridors for revegetation across valleys will be cut off by salt-affected land.

Changes in flood regime anticipated in a salinising landscape will tend to decrease the biodiversity of aquatic and riparian systems along the whole length of the rivers. Many species are as sensitive to prolonged inundation as they are to salinity.

3.5.4 Scenario modelling - conclusions

The key linkage between treatments and salinity risk is the degree to which groundwater levels will respond, particularly over the next 20 years. Groundwater modelling was used to assess this linkage.

The principal conclusions on the impact of recharge reduction on the extent of salinity from the first phase of the modelling were:

- Catchments are resistant to reduction in recharge - a large reduction is needed to produce a relatively small reduction in the saturated proportion of the flow tube.
- There is some variation among catchments, with a 50% reduction in recharge producing between a ~10-40% reduction in saturated flow tube - some of this variation may be due to catchment shape.

- Increasing recharge from the current situation has little effect on the length of the saturated proportion of the flow tube (see below in relation to timing) and therefore salinisation.

The principal conclusions on the impact of recharge reduction on the timing of salinity were:

- Reduction in recharge extends the time for a particular proportion of the flow tube to become saturated - a 25% reduction can cause a delay of between 10 and20 years, and a 50% reduction between 20 and 60 years.
- Doubling recharge, which is assumed to take into account the effect of episodic recharge and wetter periods, may hasten the onset of salinity.

In the second phase, the three catchments on which Flow Tube modelling was conducted were chosen to be representative of the “wheatbelt” of Western Australia (~18 m ha). North Baandee is considered to be similar to much of the approximately 8 m ha of the low rainfall wheatbelt, while Toolibin is believed to be similar to the central wheatbelt (~4 m ha), and Date Creek, the western wheatbelt or woolbelt; (~2 M ha). However, this modelling has not explicitly modelled hydrogeological systems on the Bremer Basin (~1.5 m ha) or the Perth Basin (~3.5 m ha). In this analysis, it is considered that the conclusions from the Date Creek study are more likely to apply to these areas than the Eastern and Central case study sites, though they are not explicitly modelled.

Flow Tube modelling has confirmed that the typically flat Eastern and Central wheatbelt catchments are not very sensitive to considerable reductions in recharge. Discharge areas are very high as a result of the extremely low hydraulic gradient and permeability of the regolith. It suggests that most valley systems, covering an area of approximately 3.5m ha of land in the wheatbelt (Barrett-Lennard 1999, Salinity Council Report, June 25th, 1999), will develop a shallow watertable over the next 30-70 years.

In the long term (100 years), recharge management in these eastern and central regions significantly slows the rate of salinisation and restricts the area to the mid- and lower-valley systems. Recharge management during this period also prevents the loss of the lower hillsides. In addition, Clarke et al. (1999) postulated that some catchments within this region would develop significantly less areas with a shallow watertable as a result of their geomorphology (convex landforms). Despite the relatively high rates of recharge management, in the longer term, most of the valley systems and some of the lower slopes will develop the full extent of shallow watertables modelled in the base case. Once again, this is due to the fact that recharge rates remain greater than the capacity of the catchment to transmit groundwater without direct intervention with engineering systems. However, the results of the sensitivity analysis to changes in saturated hydraulic conductivity at North Baandee have some most important implications in this regard. There is some evidence, at least in areas of higher saturated hydraulic conductivity, that its value is twice the current measurements (Clarke et al. in preparation) and there is evidence in the western wheatbelt that major fault systems have saturated hydraulic conductivity five times higher than the surrounding granite (Clarke et al. 1999), but it is considered unlikely that it is ten times as high. However, the existence of a deep and well-connected fractured rock aquifer and more extensive permeable sediments are the only possible mechanisms that might allow such an increase in permeability.

This is a particularly important result, since in the Flow Tube model, increasing saturated hydraulic conductivity has the effect of increasing transmissivity (the product of saturated hydraulic conductivity and the aquifer thickness), so the same reduction of seepage length would have been achieved by keeping saturated hydraulic conductivity constant and increasing the aquifer thickness tenfold. The aquifer at North Baandee is 10 m thick so this would be an increase to 100 m if saturated hydraulic conductivity is at the level measured, or 50 m if saturated hydraulic conductivity is twice that measured. If there is a fractured rock aquifer in the basement underlying the saprolite aquifer at the base of the weathered zone, as is postulated, but about which little is known, then such an increase in thickness of the aquifer is possible and the gains achieved by the medium level of intervention would remain.

Clearly, investigation of the potential fractured rock system is a matter of some importance, since it would enhance the long-term effects of vegetation treatments in the eastern and central wheatbelts.

The impact of engineering systems may not be well modelled by Flow Tube. Modelling systems that take account of geological variability would better simulate the impact of pumps. However, engineering systems of management are capable of having a significant "local" impact, since the rates of groundwater discharge responsible for such high watertables are likely to be very low (typically < 50 mm year^{-1}). Management systems that locally increase discharge may allow productivity on areas of shallow watertables.

However, results from modelling at Date Creek (100- and 300-year scenarios) suggest that even at relatively low levels of intervention (upper 30% planted to commercial trees and the remainder farmed with better systems of crops and pastures), the area with shallow watertables will be maintained at present levels or may slightly reduce. Increasing the amount of perennial pastures improves the situation, although it may not reclaim significant areas of land.

In summary, for the Eastern and Central Wheatbelt:

- Modelling suggests that catchments in the Eastern and Central region, similar to North Baandee and Toolibin, do not significantly respond in the very long term (300 years) to recharge reductions modelled in this project.
- However, if there is a fractured rock aquifer of sufficient thickness and permeability beneath the weathered zone, as is postulated, the modelling suggests that the significant gains achieved after 100 years are retained.
- In any event, modelling suggests that medium levels of intervention buy a significant amount of time in catchments with deep watertables. Episodic recharge will reduce the amount of time "bought" by new farming systems.
- Only engineering systems significantly alter this outcome.
- Very preliminary economic analysis suggests that the interventions are not currently close to "break-even" for farmers. However, many of the assumptions involved in the analysis require testing and consideration, and there are a number of off-site issues -- such as loss of biodiversity, increased flood risk, and danger to rural towns -- that need to be taken into consideration. Therefore, all levels of intervention would need to be supported by the wider community.

In summary, for the Western Regions:

- Modelling suggests that catchments in the Western region, similar to Date Creek, respond to recharge reductions modelled in this project relatively quickly. At higher levels of intervention, some land may be prevented from becoming saline.
- Engineering systems decrease the extent of land with a shallow water table.
- Economic analysis suggests that the low levels of intervention are most economic; however, none allows a "break-even" level to be achieved by farmers.
- All levels of intervention would need to be supported by the wider community.
- Groundwater pumping in this area will be more expensive, as the pumping has a smaller impact.

For all but the steepest of sections simulated (e.g., Nuniup, Boscabel), the planned interventions generally buy only a marginal decrease in the eventual extent of salinity but up to several decades of time before full salinisation.

If the recharge of the aquifers has been underestimated, or the saturated conductivity overestimated, the outlook for salinity risk is far more serious. Some cross-sections (e.g., the majority in the Upper Kent) indicate large fractions of the landscape at risk to salinity under the most likely scenarios of land management.

Most of this landscape in the SS2020 area is nearing equilibrium within the next 30 years, sooner in the western half than the eastern half. Thus, for most of the region, distinctions between 2020 and final equilibrium under current land use are marginal. These local results are wholly consistent with the general sensitivity outcomes for the wheatbelt reported by Clarke et al. (1999) and George et al. (1999).

Biodiversity

The conclusions from the modelling for the impacts of salinity on biodiversity are:

- Low levels of intervention anywhere in the wheatbelt will not offset in any significant way massive losses in biodiversity in key systems in valley bottoms and even midslope positions. These systems include high-value systems such as saline and freshwater wetlands and damplands, woodlands, and heaths. Medium levels of intervention offer little relief to biodiversity impacts in the eastern and central wheatbelt, but potentially large benefits in the western wheatbelt. Hundreds of plant species are at high risk of extinction; an unknown number of invertebrates are also at risk.
- Only groundwater pumping offers any reliable potential to save ecosystem elements in the lower third of catchments, and this will come as a trade-off against the systems into which the groundwater is discharged.

3.6 Spatio-Economic Assessment of Salinity Scenarios and Control Treatments

3.6.1 Overview

For reasons explained in other sections of this report, the hydrological modelling techniques employed by the SS2020 Project were unable to produce a hydraulic head surface and hence it was not possible to model the current level of salinity or predict future levels of salinity under each of the scenarios. As a compromise, a range of other data were drawn upon to facilitate the economic analysis.

Current areas of salinity for the Woodanilling and Boscabel regions were estimated using data from the Land Monitor Project (see Appendix C of van Bueren et al. 2000). For the Upper Kent study area, a salinity map from a previous study by Evans et al. (1996) was used. Equilibrium levels of salinity for all three case study regions under the Business as Usual scenario were determined using the method described in Evans (2000) as part of the Land Monitor Project, and reported in Section 3.4. It was assumed that equilibrium would be achieved by 2020 and that shallow groundwater was saline.

In the absence of a hydrological model for predicting responsiveness of groundwater to treatments, the following simplifying assumptions were made:

- Treatments are 100% effective at controlling salinity at the site where they are planted.
- The treatments do not have any off-site control. That is, land adjacent to treatments is not protected.

These assumptions allowed the economic analysis to proceed, but future work is required to determine their validity.

3.6.2 Business-as-usual scenario

A summary of the impacts of the business-as-usual scenario is given in Table 5. For the three case study areas that were examined, the discounted present value of total production losses over the 21 years ranged between $70/ha and $200/ha. These losses are averaged over every hectare within a case study area. Consequently, some localised areas suffer much greater losses while other parts of the study area remain unaffected. When these losses are expressed per hectare of newly salinised land, they amount to between $800 and $1090 per hectare. When expressed in terms of equivalent annual values, the losses are between $25 and $33 per hectare of newly salinised land or between $4.50 and $6.15 per hectare averaged over the whole area. The approximate distribution of costs are shown by maps included in Appendix C of van Bueren et al. (2000).

In aggregate terms, losses for each of the two 30,000 hectare case studies ranged between $4.5 and $6.1 million. The larger of the three case study areas was estimated to suffer losses of approximately $8 million. The analysis also produced some estimates of the extent to which infrastructure and remnant vegetation is predicted to become salt-affected. Across the three study areas, approximately 28 km of sealed roads are at risk, together with 54 buildings, 540 farm dams and 10,050 hectares of perennial vegetation.

	Woodanilling	**Boscabel**	**Upper Kent**
Percent of study area salt affected			
Current (2000)	7%	12%	18%
Predicted (2020)			
	17%	35%	33%
Size of study area	30,000 ha	30,000 ha	107,250 ha
On-farm impacts			
Cost of lost production (present			
values)			$7.86 mill
Aggregate for study area	$4.45 mill	$6.10 mill	$73/ha
Per hectare of study area	$150/ha	$200/ha	$816
Per hectare of newly salinised land	$1090	$897	
Number of farm buildings at risk			37
Number of farm dams at risk	16	1	330
	150	60	
Off-farm impacts			
Length of sealed roads at risk	13 km	8 km	7 km
Area of perennial vegetation at risk	970 ha	3256 ha	5820 ha

Table 5: Estimated future impacts of salinity for each of the case study areas. Impacts are additional to the damage already caused by the current level of salinity. A 5% discount rate was used.

3.6.3 Treatment scenarios

The results from the analysis of the treatment scenarios indicate the scale of economic and physical impacts that could be attributable to implementing a large-scale agroforestry program. However, the results are entirely conditional upon the simplifying assumptions that we were forced to make in the absence of a satisfactory hydrological model. As such, the results may be best regarded as a demonstration of how the spatio-economic modelling developed by this project could be applied in future, when better data are available.

Table 6 contains a summary of the key results that were calculated using a standard outlook for treatment returns. For Woodanilling, planting 65 percent of the study area to oil mallee belts would cause a net loss of $4.0 million in production (present value). This loss is due to the poor economic returns for oil mallee (relative to agriculture) and the planting of oil mallee on land with zero risk of becoming saline. In the case of Boscabel, belts of Maritime Pine were estimated to be sufficiently profitable so as to produce a net gain of $1.6 million relative to "business as usual". While losses are incurred by treating land that is not at risk from salinity, these losses are outweighed by the gains from planting pines on land at risk.

The net benefits from treating salinity were found to be greatest in Upper Kent. The strategy of planting bluegums belts and plantations was calculated to yield a net gain of $15 million (present value) for the region. Bluegums were estimated to be profitable even on land that has zero risk of becoming saline in the absence of a treatment. This is due to the competitive returns that were assumed for bluegum belts and plantations (relative to conventional agriculture).

The lower half of Table 6 contains estimates of the extent of protection offered by the treatments to infrastructure. While no effort was made to convert these physical measures into economic benefits, a technique of "threshold value analysis" could be used. For instance, in the case of Woodanilling, if the collective value of these protected assets are perceived to exceed $4.0 million in present value terms, then the control program would more than break-even.

The hydrological modelling work documented in other sections of this report indicates that, in many instances, treatments are unlikely to protect adjacent land from salinity. That is, most strategies do not offer a great deal of off-site control. Hence, if a strategy is to produce a net benefit, most of the benefits will need to come from increased returns off treated land, so that treatments will need to be commercially profitable in their own right if they are to be attractive to farmers and desirable even from a community-wide perspective. A priority for future work should be to determine the extent to which this general conclusion applies to the SS2020 region, and to identify geographical areas where there is potential to realise off-site benefits from treatments.

	Woodanilling	Boscabel	Upper Kent
On-farm impacts			
Production benefits or costs			
-Treatments on land at risk	$2.3 mill gain	$2.6 mill gain	$5.7 mill gain
-Treatments on land not at risk	$6.3 mill loss	$1.0 mill loss	$9.3 mill gain
-Agric. land saved adjacent to treated land	$0	$0	$0
Net impact on production	$4.0 mill loss	$1.6 mill gain	$15.0 mill gain
Protection of farm infrastructure *			
Number of farm buildings protected	20	0	14
Number of farm dams protected	144	33	134
Off-farm impacts			
Length of roads protected (km)	2km	2km	2km
Area of perennial vegetation protected (ha)	0	0	0

Table 6: Economic and physical impacts of treating salinity in three case study areas. The impacts are measured relative to "business as usual" outcomes. All economic impacts are expressed as net present values, using a 5 percent discount rate. Standard outlooks were used for forestry returns.

*Infrastructure located within treated areas was assumed to be protected. The assumption that treatments have no off-site control means that no protection is offered to perennial vegetation and infrastructure that is located outside the treatment areas.

3.6.4 Improved maps showing valuable infrastructure, land, and vegetation resources at risk from salinity, and more accurate statistics on the areas at risk from salinity

For the region covered by the Land Monitor risk map (the vast bulk of the SS2020 region), the following assets were identified at risk to salinisation at equilibrium under current land use (Table 7):

Infrastructure	Total	Salt-Affected at Equilibrium	Percentage of Asset
Buildings	7673	1157	15%
Roads (total)	11,864 km	3,333km	28%
Roads (sealed over 6m)	1767km	550km	31%
Roads (sealed under 6m)	575 km	168 km	31%
Dams	35599	15682	44%
Perennial (remnant) vegetation	544717 (ha)	101877 (ha)	19%

Table 7: Assets at risk in area mapped for salinity risk

More detailed statistics were produced for the scenario areas (Tables 7 – 9). Because the Land Monitor risk maps mask the roads and remnant vegetation as part of their production, it was not possible in all cases to calculate the increase in assets at risk between now and equilibrium.

LMU	Year 2000 A All area (ha)	B Current salt (ha)	B1 Treatment area (ha)	C Net unaffected area (ha)	Year 2020 D Final salt (ha)	E Net increase in salt (ha)
1 Waterlogged soils	6938	2099	0	4839	6603	4504
2 Deep sands	1633	269	0	1364	791	522
4 Sandy gravels	8084	396	?	7688	1440	1044
5. Sandy loams	6684	312	?	6372	1049	737
6. Saline soils, non arable and wetlands	1038	413	0	625	912	499
7.Perennial vegetation	5623	2 (masked)	0 (masked)	5621	3258	3256
Totals	30000	3489		26511	14055	10564

Note: 58% of perennial vegetation is affected at equilibrium.

Boscabel Infrastructure

Infrastructure	Year 2000 A All road (m)	B Current salt (m)	C Net unaffected area (m)	Year 2020 D Final salt (m)	E Net increase in salt (m)
Roads under 6m (sealed)	18868	438 (masked)	18430	3913	3475
Roads over 6m (sealed)	15661	1343 (masked)	14318	6162	4819

Infrastructure	Year 2000 A All	B Current salt	C Net unaffected	Year 2020 D Final salt	E Net increase
Buildings	29	1	28	2	1
Dams	286	96	190	155	59

Table 8: Boscabel area salinity risk statistics

	Year 2000			Year 2020	
LMU	A All area (ha)	B Current salt (ha)	C Net unaffected area (ha)	D Final salt (ha)	E Net increase in salt (ha)
1 Waterlogged soils	20741	4106	16635	10375	6269
2 Gravel slopes	43419	2806	40613	6171	3365
3 Wetlands and rock outcrops	3893	3425	468	3808	383
4 perennial vegetation	39201	8978	30223	14798	5820
Totals	107254	19315	87939	35152	15837

Note: 38% of perennial vegetation is affected at equilibrium.

Upper Kent Infrastructure

	Year 2000			Year 2020	
Infrastructure	A All road (m)	B Current salt (m)	C Net unaffected area (m)	D Final salt (m)	E Net increase in salt (m)
Roads under 6m (sealed)	21816	3562	18254	6469	2907
Roads over 6m (sealed)	51646	7921	43725	12273	4352

	Year 2000			Year 2020	
Infrastructure	A All	B Current salt	C Net unaffected	D Final salt	E Net increase
Buildings	574	34	540	71	37
Dams	1956	623	1333	953	330

Table 9: Upper Kent Area salinity risk statistics

Do Nothing	Year 2000				Year 2020	
LMU	A All area (ha)	B Current salt (ha)	B1 Treatment area (ha)	C Net unaffected area (ha)	D Final salt (ha)	E Net increase in salt (ha)
2 Waterlogged soils	3250	935	0	2315	2871	1958
3 Deep sands	96	13	0	83	23	20
4 Sandy gravels	1651	11	1640	1640	27	18
5 Sandy loams	21906	1025	20881	20881	3146	2138
6 Wetlands and rock outcrops	81	48	0	33	79	31
7. Perennial vegetation	3016	97 (masked)	0	2919	1067	970
Totals	30000	2129	22521	27871	7213	5231

Note: 35% of perennial vegetation is affected at equilibrium.

Woodanilling Infrastructure

Do Nothing	Year 2000			Year 2020	
LMU	A All road (m)	B Current salt (m)	C Net unaffected area (m)	D Final salt (m)	E Net increase in salt (m)
Roads under 6m (sealed)	29180	918 (masked)	28262	8721	7803
Roads over 6m (sealed)	12010	751 (masked)	11259	5772	5021
	Year 2000			Year 2020	
Infrastructure	A All	B Current salt	C Net unaffected	D Final salt	E Net increase
Buildings	340	37	303	53	16
Dams	858	48	810	298	150

Table 10: Woodanilling area salinity risk statistics

3.6.5 Spatio-economic Assessment of Salinity Scenarios and Control Treatments

The economic component of SS2020 has increased our knowledge of the strengths and limitations of using spatial data sets to facilitate cost-benefit analysis. In addition, it has provided some foundations for future economic studies of salinity by developing a systematic way of categorising the economic impacts of salinity across different land types.

The study has demonstrated how a GIS framework, coupled with cost-benefit analyses, can be used to investigate the economic consequences of regional strategies for managing salinity.

The analytical technique has a number of strengths, including:

- The economic impact maps produced by the analysis serve as a valuable communication tool for economists. The maps translate physical changes into economic outcomes and provide decision makers with an overview of how impacts are unevenly distributed across a region.
- The analysis identifies the ratio of on-farm to off-farm impacts, and the distribution of impacts across stakeholders. This information is valuable for devising "cost-sharing" policies once a preferred strategy has been selected.
- The analysis helps with priority setting by identifying which geographic regions are likely to yield the greatest economic returns to salinity management.
- Once the initial data are collected for a given region, the impacts of different scenarios and assumptions can be simulated quite rapidly. It allows judgements to be made about the optimal scale at which treatments should be implemented.
- The technique is readily transferable to other regions in Australia which have similar databases.
- The results force hydrologists to confront the economic implications of their recommendations for controlling salinity.

Perhaps the greatest criticism of the method is its high demand for data. Application of the technique is limited to those regions which have satisfactory sets of digitised resource information. A second weakness is that maps tend to hide the uncertainty that underlies modelled results. While a GIS often produces an impressive visual product, this can sometimes be misleading if the viewer does not understand the probabilistic nature of the mapped values. Quantifying and presenting risk and uncertainty is an important avenue for future research in economic-GIS modelling (and in fact for all GIS-based modelling).

3.6.6 General Implications

The implications of the SS2020 project for the management of dryland salinity in Australia are profound and of immediate application, but must be tempered by the degree of detail in salinity risk assessments possible with existing data and methods. The original vision of a regional, synoptic model of salinity risk under various scenarios, based on a regional hydraulic head surface, could not be realised. Instead, inferences on the economics and impacts of alternative land uses were largely based on local modelling in case study areas with adequate data, and an at-equilibrium decision-tree mapping at the regional scale.

Nevertheless, the results are sensible and likely to hold for most parts of the region.

For the 2.13 Mha mapped, 0.65 Mha (31%) is expected to be at risk of salinity at equilibrium; this is the same value as estimated by Ferdowsian et al. (1996) for the entire Wheatbelt, and underscores the scale of the challenge in WA. There are large numbers of farm and community assets at risk, including dams, roads and remnant vegetation.

For all but the steepest landscapes, intervention in the form of land use change must be substantial and widespread. When alternative enterprises (e.g., trees) are limited to the most appropriate soils only, the ultimate extent of salinity does not radically change, although time to impact is slowed. Salinity risk abatement associated with low-recharge land-use systems largely only benefit the land they are planted on. Thus alternative farming systems must largely be profitable in their own right. The non-farm assets at risk require large, local, largely non-commercially-driven intervention.

The most immediate and obvious implication is to the analysis of where public funds should flow in aid of salinity control. Given the scale of the revegetation required to substantially change the ultimate extent of salinity, and the limited amount of public funds, it does not appear possible to make up the gap in farm profitability associated with alternative, low-recharge farming systems. On the other hand, recognising the local nature of the benefits, public assets such as key remnant vegetation sites, roads and townsites will require large, direct investment in remedial works. Given the nature of the hydrology in WA, it is arguable that the highest-priority uses of those limited funds will mostly be in engineering works in or around the assets to be protected, rather than for revegetation on farms.

The implications of buying time as opposed to reducing the area ultimately salinised are also profound. While "high water use" farming will ultimately leave almost as much land at risk to salinity as current practice, the possibility that decades of time might be bought before the full impact of salinity is realised has considerable social and economic value. If nothing else, it may give families and government more time to adapt to a salinised landscape.

Decision-makers need confidence that the technical estimates and data are sound. The SS2020 project has highlighted the difficulty in making robust inferences even for regions with relatively rich data. In fact, the density of reliable data in the WA wheatbelt is disturbingly low, much lower than our expectations on the face of existing databases. When these databases were interrogated and screened for quality control, a minority of the data proved to be useful. Many bores had very infrequent time series data and short records. Only 711 bores in the study area had adequate time series data to interpret trends. Of these, only 129 had surveyed or DGPS location data. This serious gap in our knowledge about a landscape undergoing profound and disturbing change greatly compromises our ability to provide sound, technically-based advice.

These results challenge our perceptions regarding what are the most important and useful data for assessing salinity risk, and the most effective ways of using them. While in theory the traditional hydrogeological approach which relies on groundwater levels provides a more definitive assessment, the sparseness and apparent lack of reliability of this data across southern Australia does not recommend this approach for a national methodology. In contrast, synoptic, widely-available data (high-quality elevation data, high-resolution Landsat satellite reflectance data) were shown to be of substantial use, while still requiring a degree of expert input into the analysis. Of the geographical coverages examined, the results from this project would lead to a questioning of the direct use of soil mapping units in deriving salinity risk; it appears that there is no direct correspondence, at any scale, between the occurrence of high saline watertables and the soil units available to us.

Nevertheless, the need for bore hydrographic monitoring in defining the effectiveness of treatments is indisputable. There is a serious deficiency in southern Australia in monitoring the effectiveness of remedial works, including those supported under public funding such as the Natural Heritage Trust.

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Rod Short	Agriculture Western Australia

Figure 1: Groundwater depths across the SS2020 project area, as assessed by bores. Sparseness of data precluded the development of a regional water level map even in this relatively data-rich region of Australia.

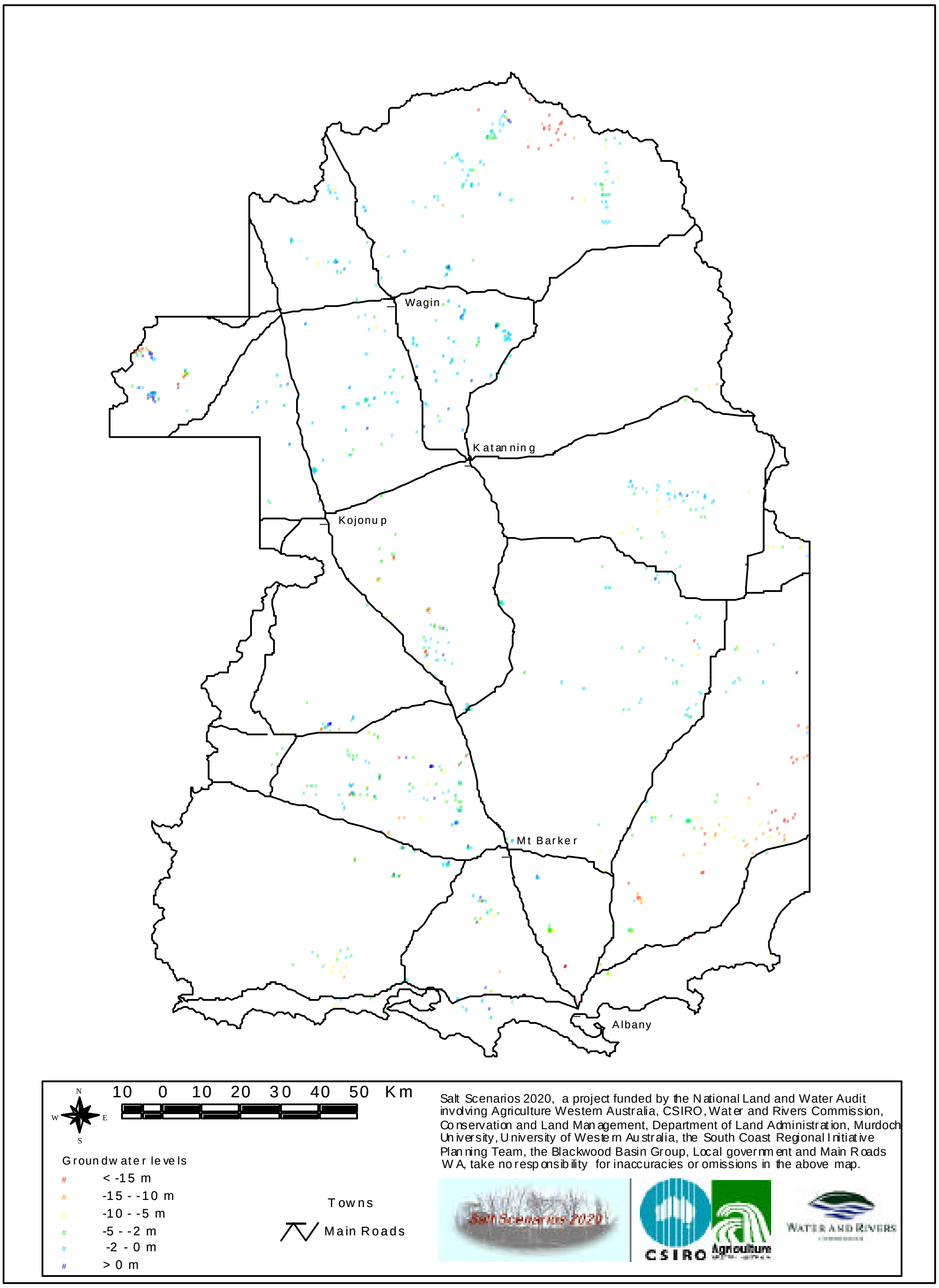


Figure 2: Groundwater trend maps for bores in the SS2020 project area. Falling bores may be a result of short-term records during relatively dry years; lack of reliable, long-term monitoring left gaps in our ability to establish salinity risk

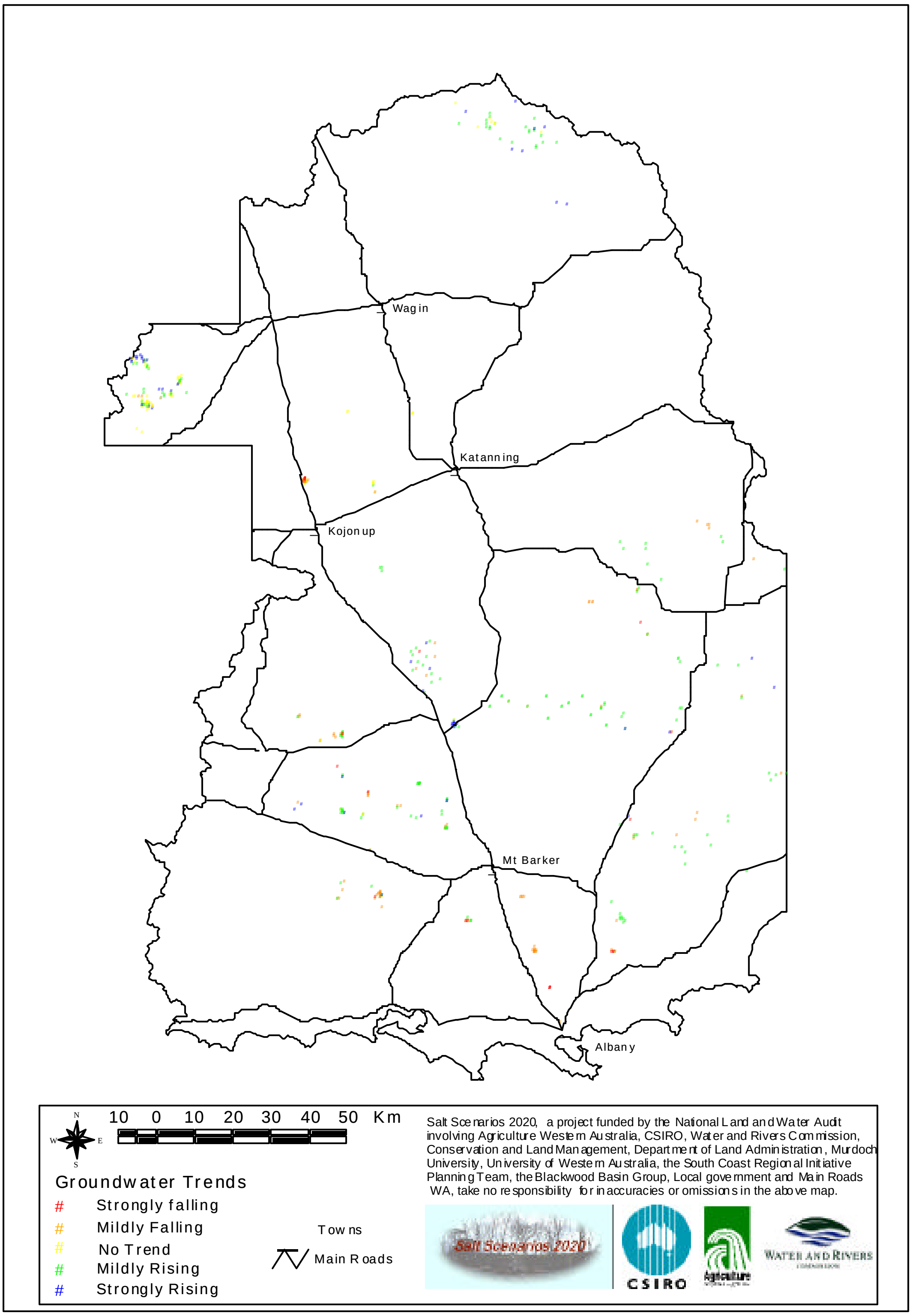


Figure 3(a): Observed and fitted depths for G. English Bore 5 showing two alternative models. From Shao et al. (1999).

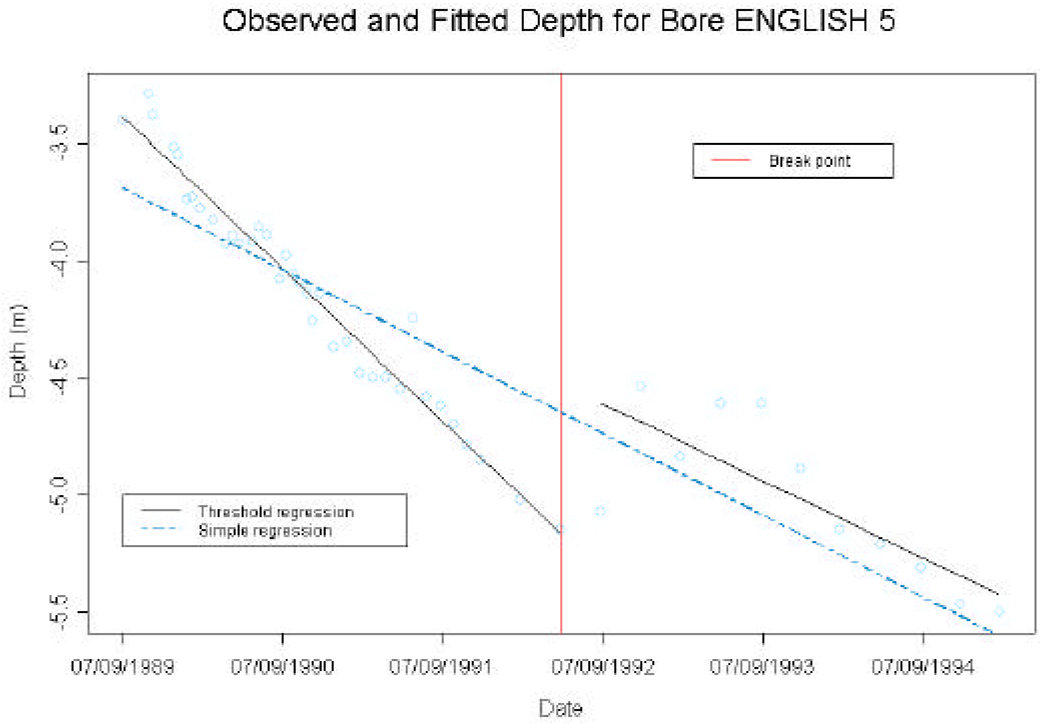


Figure 3(b): Observed and fitted depths for Bore DJ07I. From Shao et al. (1999). This analysis shows the full complexity accommodated by the method: periodicity, thresholds, and regression type.

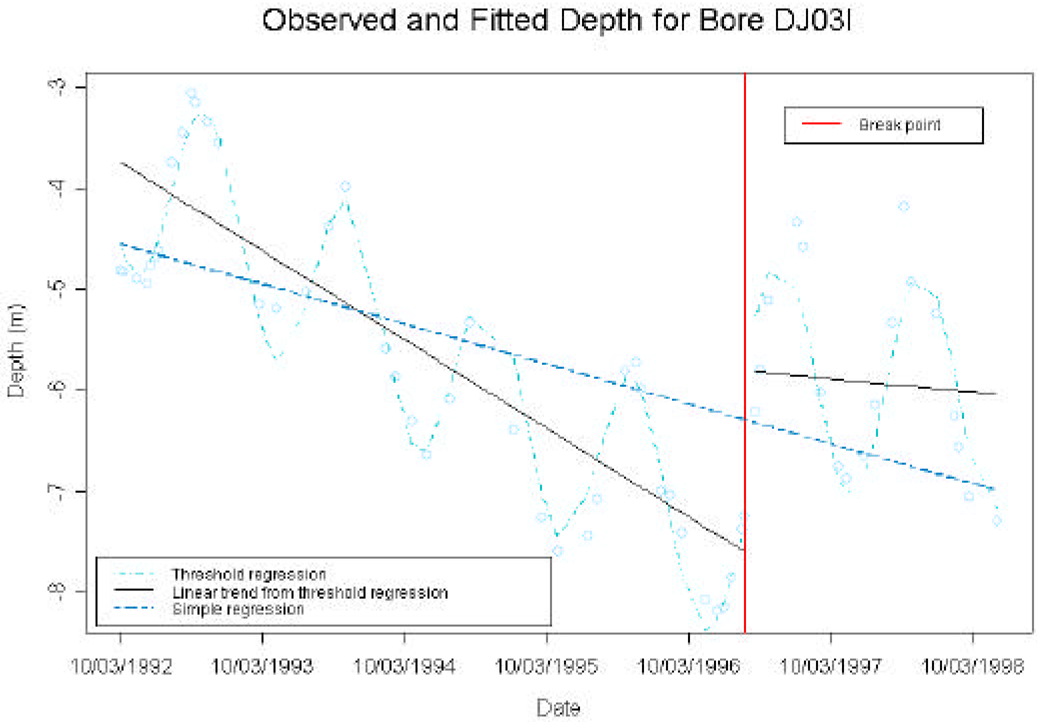


Figure 4: **Average Depth to Groundwater (May 1998) versus Surface Elevation for Ucarro Deep Bores. Alternative regression excluding two points from the hydrogeomorphic unit with shallow bedrock dramatically improved prediction. These points are not evident using the original HARSD formulation (without detailed analysis of residuals).**

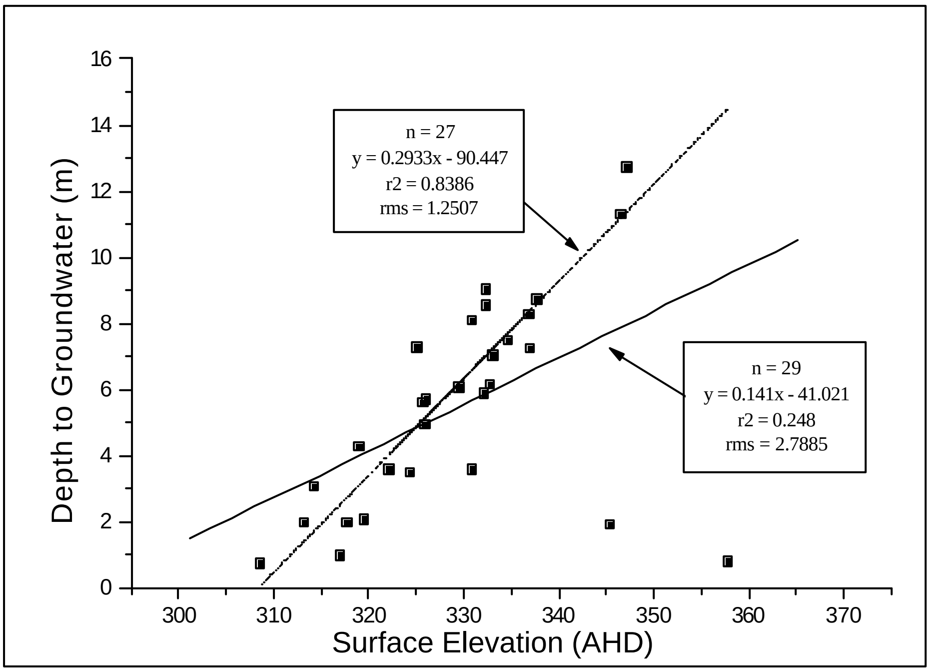


Figure 5: Overview of catchment behaviour

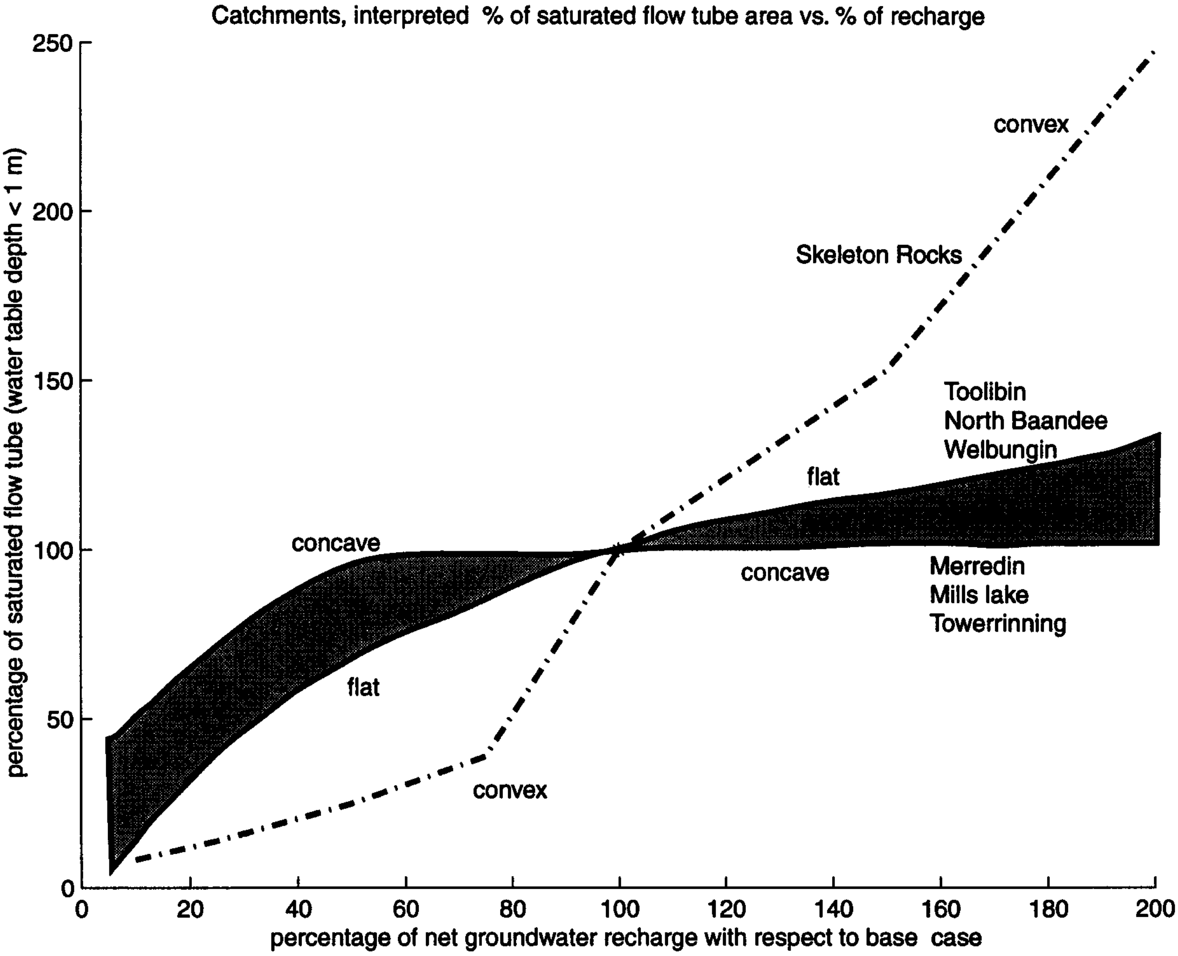
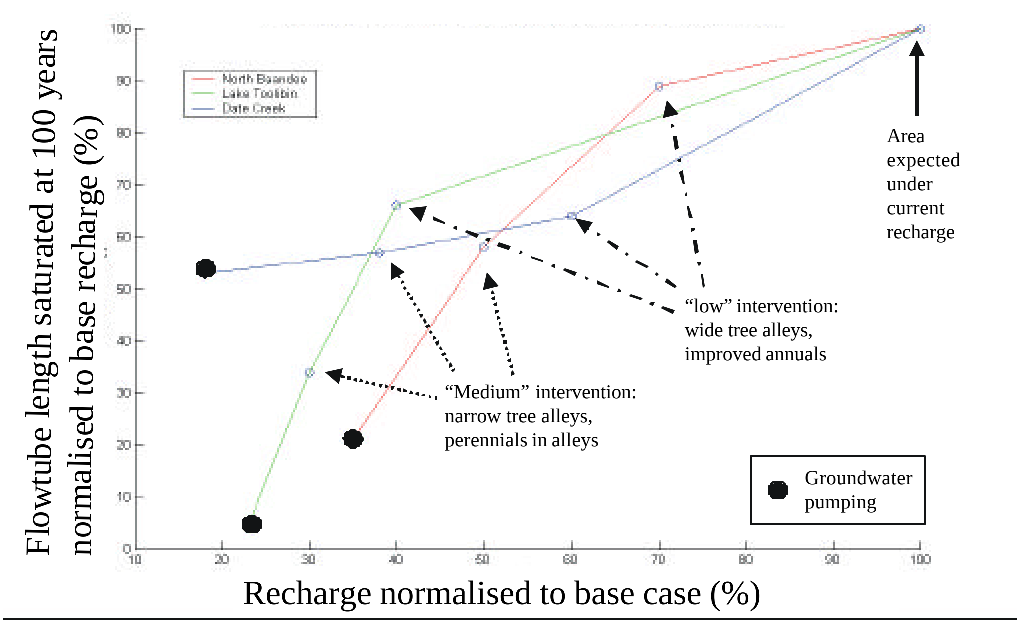


Figure 6: Impact of Recharge Reduction at each of the Three Sites



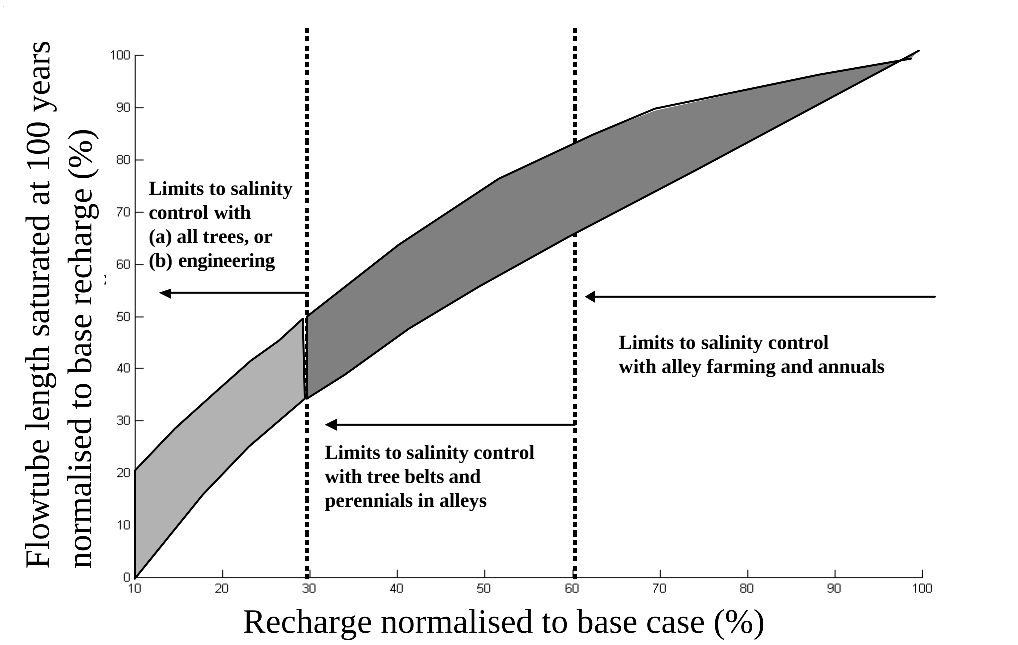


Figure 7: SS2020 area where equilibrium salinity risk was derived

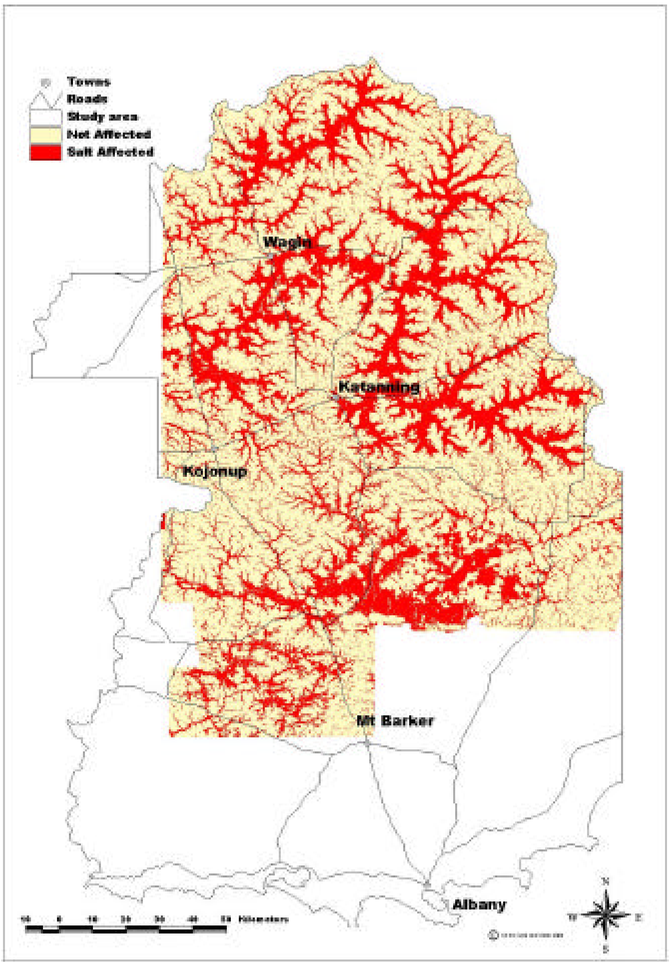


Figure 8: Calculation of *AMRR*

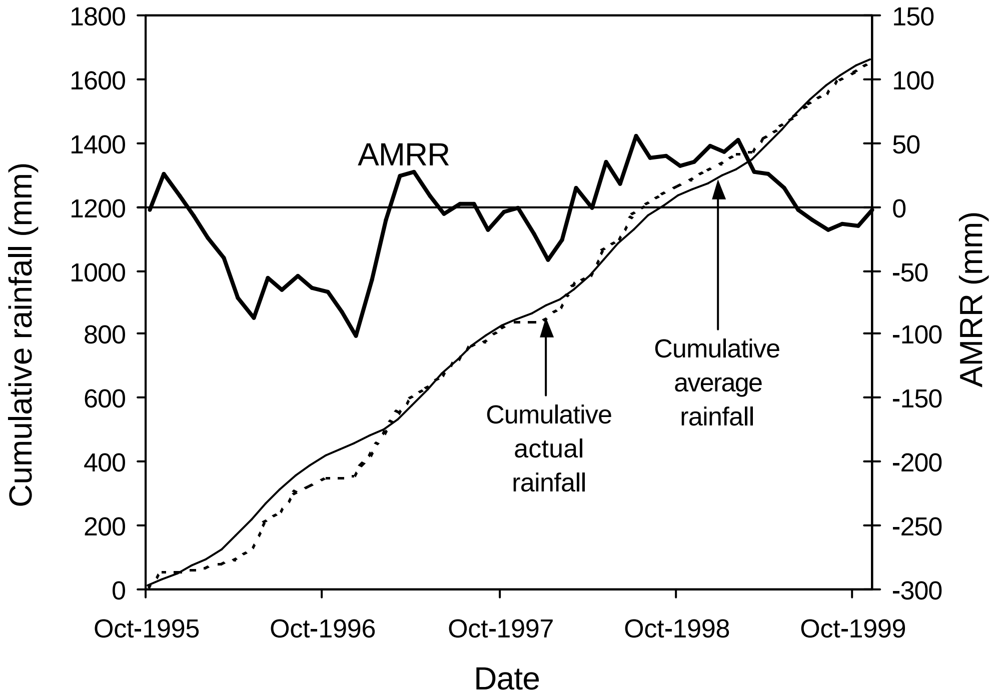


Figure 9: Hydrograph for a deep bore

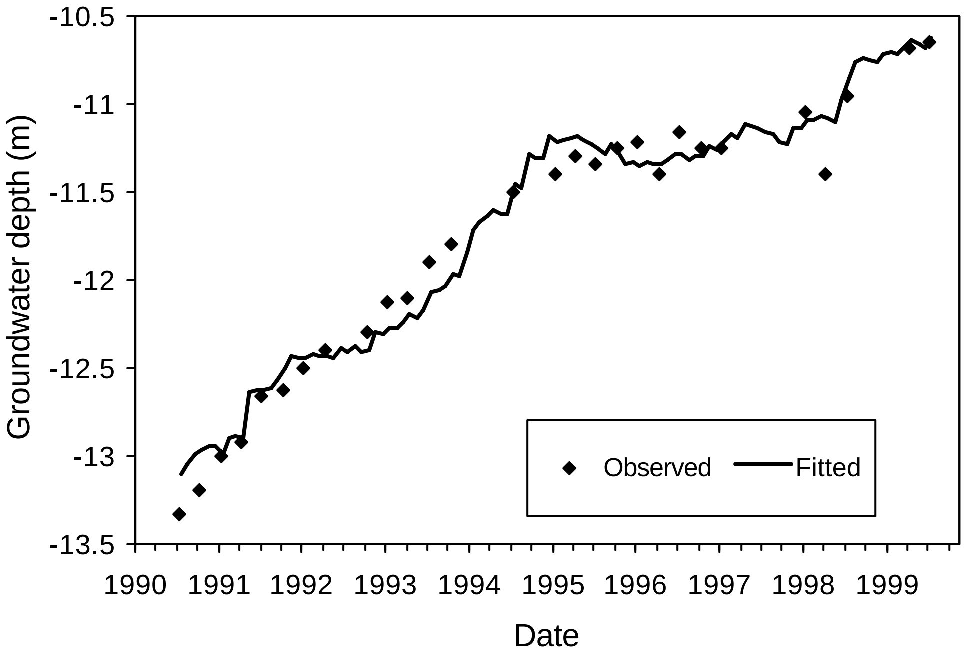


Figure 10: Hydrograph for a medium-depth bore

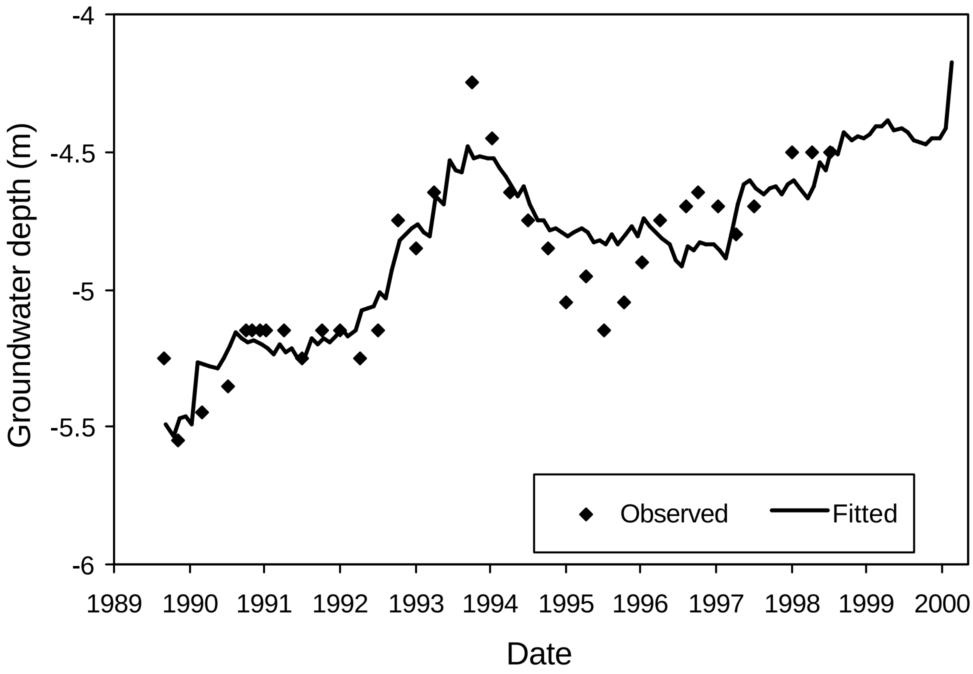


Figure 11: Hydrograph for a shallow bore