Predicted stream flow and salinity changes after afforestation in the Southwest Goulburn

A regional BC2C model application to develop commercial environmental forestry

Albert I. J. M. van Dijk, Xiang Cheng, Jenet Austin, Mat Gilfedder, Peter B. Hairsine

Prepared for the Commercial Environmental Forestry project, funded by the Natural Heritage Trust

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Cover: View on the property of Mr. David Freeman in Whiteheads Creek, near Seymour. In the foreground a gully and saline discharge area can be recognised by the occurrence of spiny rush (it has been fenced to promote regeneration). On the hillside in the background a recently planted block of forest can be seen. Whiteheads Creek is an area with considerable dryland salinity problems and an area where modelling suggests environmental benefits can be achieved by afforestation. (Photograph Albert van Dijk, 19 March 2004)
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Executive Summary

Returning tree cover to reduce groundwater recharge has the potential of reversing some of Australia’s salinity problems. Plantation forests in low-to-medium rainfall areas (400-800 mm year⁻¹) are currently not widespread and, at the lower end of this range, typically commercially marginal or unprofitable. However, their salinity benefits may provide an incentive for expansion. The Commercial Environmental Forestry (CEF) project was jointly initiated by DAFF/NHT and CSIRO to develop forestry systems for profitability, environmental benefits and regional community development in those low-to-medium rainfall zones. The project integrates knowledge and models of the catchment water and salt balance, forest productivity, and economics. The Southwest Goulburn catchment has been selected as the CEF project focus area.

Earlier hydrological modelling suggested that ~450 km² or 12% of the Southwest Goulburn catchment had potential for realising salinity benefits through afforestation (Van Dijk et al., 2004a, b). However, this result was obtained with a water yield and salinity model (BC2C) that was intended to show relative regional differences in water and salt yield across the entire Murray-Darling Basin, taking into account that data availability is low in most of the basin. By contrast, the hydrogeology of the Southwest Goulburn is relatively well understood and local hydrological and hydrogeological data are available. This provides a scope for further model calibration; that is, providing it with more and better input and changing model parameters to reproduce the observed distribution of water and salt. This was the purpose of the present report.

We first reviewed the existing hydrological and hydrogeological knowledge and data. Secondly, we analysed available records of stream flow and salinity for five subcatchments within the Southwest Goulburn to assist BC2C model calibration. These included the catchments of the Sunday, Hughes, Sugarloaf and Major Creeks (282–609 km²), which had continuous daily flow and discontinuous EC records from 1971–1973 to 2001–2003. Constant fractions of tree cover were presumed. By contrast, the small (3.2 km²) Pine Creek catchment was fully converted to pine plantation between 1986-1988, with stream flow and EC being recorded on a daily basis ever since. This dataset therefore represents a rare field experiment of how water and salt discharge change after afforestation. All data were interpreted with a custom-built stream salinity analysis tool that largely uses the same concepts as BC2C but operates on a daily time-step. Although some ambiguity in data interpretation could not be avoided, this gave useful insights in the hydrological functioning of the different catchments and provided guidance for BC2C calibration.

The results of hydrogeological review and stream flow analysis were combined with higher detail maps of climate and land cover to provide a realistic parameterisation of the BC2C model for the Southwest Goulburn. The resulting spatial distribution of expected water and salt generation changes differs substantially from earlier estimates obtained with the basin-wide BC2C version, emphasising the importance of local calibration. The improved model application suggests that as much as 49 to 72% of the Southwest Goulburn area produces a stream salinity benefit after 5 to 30 years, respectively. The general observation, made in the earlier preliminary model application, that environmental benefits can be achieved much more effectively by carefully targeted afforestation is maintained.

The model provides information on relative differences in the effectiveness of achieving environmental benefits by CEF between subcatchments, and a quantitative estimate of its magnitude and timing. The uncertainty in model predictions increases from water yield via salt yield to stream EC, the latter accumulating all uncertainty in the former two quantities. The capacity for higher detail, hillslope-level predictions is being developed through the FLUSH modelling framework, as a specific application of the methodology developed by partners in the CRC for Catchment Hydrology’s Salt Prediction project (project 2C) (Gallant et al., 2004). This will improve the spatial detail and realism of predictions, but not necessarily the uncertainty, which requires additional data collection.
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Acronyms used

AHD        Australian Height Datum
ANU        Australia National University
BC2C       Biophysical Capacity to Change model (Dawes et al., in press)
BOM        Australian Government Bureau of Meteorology
BRS        Bureau of Rural Sciences, DAFF
CEF        Commercial Environmental Forestry
CRC        Cooperative Research Centre
CSIRO      Commonwealth Scientific and Industrial Research Organisation
DAFF       Commonwealth Department of Agriculture, Forestry and Fisheries
DEM        Digital Elevation Model
DPIE       Commonwealth Department of Primary Industry and Energy
DPI        Department of Primary Industries, Victoria
EC         Electrical Conductivity (a measure of salt concentration)
FLUSH      Framework for Land Use and Spatial Hydrology (Gallant et al., in press)
GIS        Geographical Information System
GFS        Groundwater Flow System (Coram et al., 2000)
IHACRES    Identification of unit Hydrographs and Component flows from Rainfall, Evaporation and Stream flow data (Jakeman and Hornberger, 1993)
MDBC       Murray-Darling Basin Commission
NDSP       National Dryland Salinity Program
RWC        Rural Water Corporation
SALMOD     SALimity MODel (custom built for this report)
SILO       Australian climate database available from BOM
SKM        Sinclair Knight Merz
SPIF       Scenario Planning Investment Framework
LAI        Leaf Area Index
Acknowledgements

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1. Introduction

1.1. Background

Australia is facing severe salinity problems. Clearing of native vegetation has led to increased recharge of groundwater systems in many areas, mobilising salt that is now degrading land and water resources. Returning deeper rooted and more permanent vegetative cover aims to reduce groundwater recharge in an effort to mitigate land and stream salinity, although the groundwater response time can be long.

To combat stream salinisation, the Murray-Darling Basin Commission aims to establish 1.5 million hectares of targeted woody vegetation by the year 2050 for a projected cost of $17 billion. The Murray-Darling Basin Salinity Management Strategy proposes farm forestry for revegetation in low-to-medium rainfall areas (400-800 mm y\(^{-1}\)). Parallel to this, the ‘Plantations 2020 Vision’ was initiated jointly by governments and the forestry industry in 1997 (DPIE, 1997). It aims to enhance regional wealth creation and international competitiveness by trebling the area of commercial tree crops by 2020 and is on track in achieving this expansion.

Despite these seemingly complimentary goals, traditional forestry investments have made little progress in extending forestry into lower rainfall areas, due to low productivity, marginal profitability, lack of established transport and processing infrastructure, and a reluctance to diversify from traditional farming enterprises (A. Gerrand, BRS, Canberra, pers. comm., 2004). The off-farm private investors (industry and financial sectors) needed to underpin profitable forestry require sound information to indicate that such investment would be commercially viable.

Forestry may become commercially viable in some of these areas if environmental benefits can be taken into account. In addition to traditional wood products, potential benefits of ‘commercial environmental forestry’ (CEF) include decreased land salinity, carbon sequestration, soil conservation and rehabilitation, reduced stream nutrient and sediment loads, and enhanced landscape diversity, ecological connectivity and biodiversity. There will also be economic and social benefits for rural communities (Alexandra, 2002).

The Commercial Environmental Forestry project was initiated by the Commonwealth Department of Agriculture, Fisheries and Forestry (DAFF) and CSIRO through the Natural Heritage Trust program. The project aims to develop commercially viable and environmentally beneficial farm forestry systems in low-to-medium rainfall zones typical of much of Australia. Specific aims are: to quantify the environmental benefits of plantations; improve predictions of plantation productivity; and increase financial returns for growers, processors and investors in farm forestry.

The CEF concept is being developed and tested in a focus area where it is most likely to be viable. For six potentially suitable focus catchments in New South Wales and Victoria, Van Dijk \textit{et al.} (2004a) used the Biophysical Capacity to Change (BC2C) model (Dawes \textit{et al.}, in press) to estimate the likely changes after 15 years in catchment stream flow and salinity for an afforestation scenario. This study contributed to the eventual selection of the Southwest Goulburn as research focus area.
The BC2C model was developed as a synoptic prioritisation tool to identify broad regions within the Murray-Darling Basin and regions within where land use change was most likely to have an effect on Murray River salinity (Dawes et al., in press; Dowling et al., 2004). To enable this for the many data-poor areas within the basin, this required a parsimonious and broad-brush approach to the representation of hydrogeology. This approach was suited for comparison of the six prospective focus areas (Van Dijk et al., 2004a) and a first, qualitative assessment of the likely effectiveness in achieving environmental outcomes by targeted as opposed to random afforestation (Van Dijk et al., 2004b).

The CEF project aims to identify locations within the catchment where the greatest overall CEF benefits can be expected. This requires the ability to predict the effect of locally targeted afforestation on subcatchment stream outcomes (changes in water and salt yield and stream salinity). This goes beyond the original application of the BC2C model. Fortunately, the data availability in the Southwest Goulburn is greater than in many other areas. These data can be used for better model parameterisation by: (i) providing more accurate and detailed input to the model; and (ii) calibration, that is optimising empirical parameters in the model structure to reproduce observed patterns in water and salt distribution within the catchment. This allows realistic prediction of stream outcomes at subcatchment outlets and so can help to identify those subcatchments where desired stream outcomes are most likely to be achieved.

It should be stressed that the structure of the BC2C model is not suited to assess particular patterns of planting within a catchment, because it does not describe local interactions within hillslopes. For this purpose, models of forest growth, hillslope hydrology and groundwater behaviour are being linked in the FLUSH framework (Framework for Land Use and Spatial Hydrology, Gallant et al., 2004) that is to be tested in the focus area. Because the two modelling approaches essentially use the same terrain and stream flow data for calibration they will produce consistent results.

1.2. Aim and structure of this report

The aim of the present report is to parameterise the BC2C model for the Southwest Goulburn so that it provides subcatchment water and salt yield predictions that are in accordance with the available hydrological knowledge and data.

This is done through the following steps:

- A review of the hydrological and hydrogeological knowledge and data available for the Southwest Goulburn (Section 2);

- Analysing the available stream flow quantity and quality data and determining the implications for BC2C model application (Section 3).

- Deriving a BC2C model parameterisation based on the above information (Section 4).

In Section 4, the results obtained with the improved BC2C model application are also compared with those from the earlier synoptic BC2C version (Van Dijk et al., 2004a, b). The main conclusions and scope for improvement are discussed in Section 5.
2. Regional hydrogeology

2.1. Introduction

This section provides a brief summary of the available knowledge and data relevant to BC2C application in the Southwest Goulburn. This summary draws rather heavily on a recent and comprehensive literature review of local hydrogeology by Cheng et al. (2003).

2.2. Regional hydrogeological classification

The National Classification of Catchments (Coram, 1990) can be seen as the first consistent framework for classifying parts of the landscape on the basis of their hydrogeological behaviour. It was used to produce the national scale groundwater flow system (GFS) map (Coram et al., 2000). This map was further refined for the Murray-Darling Basin, resulting in the mapping used in BC2C (Dowling et al. 2004). Following a series of workshops facilitated by the MDBC, regional salinity province maps have been developed across the upland parts of the basin, showing the distribution of GFSs within each region. For the Goulburn-Broken catchment, a map with 13 different GFSs was produced, eight of which occur in the Southwest Goulburn (NDSP & MDBC, 2001; see Figure 1 and Table 1).

Below, the classification and the underlying regional hydrogeology are briefly described, with an emphasis on the implications for the present BC2C application. Where available, for each GFS estimates are given of groundwater salinity (as electrical conductivity, EC in µS cm⁻¹), specific yield (S, the water yield for a unit decrease in groundwater level), the hydraulic conductivity (K, m d⁻¹, a measure of how rapidly water can move through a unit volume of aquifer), and T (m² d⁻¹, the product of K and aquifer thickness D, and therefore a measure of how much water can be transported through the GFS at a given pressure difference). Most values given are quoted from Cheng et al. (2003).

Most of the Southwest Goulburn area is dominated by old (Cambrian to Devonian) marine sediments and granites. These rocks are partially overlain by younger deposits, notably Quaternary volcanic basalt outflows and alluvium. Four important hydrogeological landscape units may be recognised in the area (Figure 1):

1. **Granite** hills occur in the western and eastern extremities of the area (Mollisons Creek and Hughes Creek, respectively – see Figure 5 for the location of these subcatchments) and are typically found above 300 m AHD (above mean sea level). The granites are sparsely fractured but weathered and well-drained. Stream valleys cut deep into the rocks and little alluvium is found in them. The upper, up to 30 m thick zone, is weathered into coarse quartz sand or gravel in a silty clay matrix. Groundwater flows through this weathered zone rather than through fractures and typically forms small-sized, local unconfined aquifers. Values of K are normally <1 m d⁻¹ and S is 0.05–0.2. Groundwater in these GFSs is relatively fresh with EC typically 300 µS cm⁻¹, occasionally with values up to 3000 µS cm⁻¹. Salinity does not appear to be a major issue in this landscape, with little or no apparent saline surface discharge and relatively low stream salinity values (for example, <250 µS cm⁻¹ on a flow-weighted average annual basis for Hughes Creek; Section 3; Table 3).
Old sedimentary rocks occupy a large part (~60%) of the Southwest Goulburn and are found in the middle and northern part of the catchment, typically between 150–400 m AHD. These uplifted, gently folded and dissected rocks consist mainly of a mix of Palaeozoic marine sandstones, siltstones and mudstones. Streams typically follow a strongly branching flow system.

Table 1. Main hydrogeological units, groundwater flow system classes and their properties as estimated through expert consultation: groundwater salinity (in mg l⁻¹), specific yield (S), hydraulic conductivity (K) and transmissivity (T). See Figure 1 for GFS distribution. Source: NDSP/MDBC (2001) (Note: these salinity values were not used in the present BC2C application, see Section 4).

<table>
<thead>
<tr>
<th>Geology</th>
<th>Groundwater flow system</th>
<th>salinity mg l⁻¹</th>
<th>S  %</th>
<th>K  m d⁻¹</th>
<th>T  m² d⁻¹</th>
</tr>
</thead>
<tbody>
<tr>
<td>Granite</td>
<td>local Weathered granites</td>
<td>500</td>
<td>15</td>
<td>0.5</td>
<td>10</td>
</tr>
<tr>
<td></td>
<td>local Fractured sedimentary and metamorphic rocks</td>
<td>750</td>
<td>2.5</td>
<td>1.5</td>
<td>2.5</td>
</tr>
<tr>
<td>Old sedimentary rocks</td>
<td>local Weathered fractured sedimentary rocks</td>
<td>2,500</td>
<td>2.5</td>
<td>1.0</td>
<td>2.5</td>
</tr>
<tr>
<td></td>
<td>local Fractured Cambrian/Ordovician rocks</td>
<td>1,200</td>
<td>2.5</td>
<td>2.5</td>
<td>2.5</td>
</tr>
<tr>
<td>Basalt</td>
<td>local Fractured basalts</td>
<td>1,200</td>
<td>2.5</td>
<td>3.5</td>
<td>25</td>
</tr>
<tr>
<td></td>
<td>regional Riverine plains</td>
<td>1,000</td>
<td>20</td>
<td>25</td>
<td>20</td>
</tr>
<tr>
<td>Alluvial deposits</td>
<td>local Upland alluvium</td>
<td>600</td>
<td>15</td>
<td>20</td>
<td>20</td>
</tr>
<tr>
<td></td>
<td>local Tertiary gravel caps</td>
<td>2,400</td>
<td>7.5</td>
<td>0.55</td>
<td>5.0</td>
</tr>
</tbody>
</table>

2. *Old sedimentary rocks* occupy a large part (~60%) of the Southwest Goulburn and are found in the middle and northern part of the catchment, typically between 150–400 m AHD. These uplifted, gently folded and dissected rocks consist mainly of a mix of Palaeozoic marine sandstones, siltstones and mudstones. Streams typically follow a strongly branching flow system.
pattern and in some areas have incised deeply into the bedrock, before being partially filled up again with alluvium. Areas of saline discharge at the surface are mainly found in this landscape unit, and stream salinity values can be moderate (>500 µS cm⁻¹; see Section 3).

The GFSs in this unit are of local to intermediate scale and are subdivided into three classes based principally on their degree of weathering and age.

- **Local GFSs in fractured sedimentary and metamorphic rocks** occur in the central and southern part of the catchment (Figure 1). They consist of Silurian-Devonian (sometimes metamorphosed) marine rocks and form moderate to steep hills with thin hillslope soils and minor sediment deposits on the lower slopes and valley floors. The rocks are usually only superficially weathered and the main aquifers are developed in fractured zones that can extend to more than 100 m below the surface. The local nature of the fractures makes these systems very heterogenous and localised. Fractures can make up as much as 2% of the rock volume, but because many fractures are not connected to the GFS, the actual S is usually much smaller. Values for K are typically 0.1–2 m d⁻¹ and 10–50 m² d⁻¹ for T, whereas EC is usually 800–3,000 µS cm⁻¹. The response to changed land use is expected to be fairly rapid in this GFS (within one or two decades) but equilibrium conditions may well take much longer to establish.

- **Local to intermediate GFSs in weathered fractured sedimentary rocks** are similar to those described above, but are highly weathered, often to depths of 30 m or more. They are found among the unweathered rock and are most frequent in the undulating hills near to the alluvial plain in the lower central part of the catchment. These systems appear to be the main contributors of stream salt and often have saline discharge areas (for example, Whiteheads Creek; see cover). They normally consist of less weathered, permeable rocks forming the main aquifer that are overlain by a weathered, kaolinised and less permeable layer. Values of K are typically <1 m d⁻¹ but up to 10 m d⁻¹ in fractured zones, with T values of <30 m² d⁻¹. The EC varies between 800 to >30,000 µS cm⁻¹. Groundwater is transmitted largely by the underlying fractured rock and discharges in valley floors and at breaks-of-slope. An important issue in managing salinity in this landscape is the low permeability of the aquifer, in particular in the lower parts of the catchment. The associated response times for these weathered sedimentary rock aquifers are expected to be longer than for those in unweathered rock.

- **Local to intermediate GFSs in fractured Cambrian/Ordovician rocks** are found along the Mount Camel Range at the north-western edge of the area (Figure 1). This long and narrow range consists of fractured ancient volcanic and marine sedimentary rocks with a weathered surface layer on the slopes. The fractures cause a highly heterogenous groundwater system with K of 1–20 m d⁻¹, S of 0.001–0.3 and T of 15–170 m d⁻¹. Preferential flow is along the north-south axis. Groundwater EC is normally 1,500–15,000 µS cm⁻¹. The groundwater system appears to be compartmentalised and numerous saline discharge areas occur at the foot slopes at either side of the range.

3. **Basalt plains, hills and cones** show up on the geological map as elongated units among sedimentary rocks in the Sunday and Sugarloaf Creek subcatchments. These rocks were formed by volcanic outflows and have a flat to gently undulating topography. They can be highly weathered, very porous and surface runoff from these areas is typically low. The groundwater behaviour depends primarily on the degree of jointing and open spaces in the rock that formed during the outflows, and reported K and T values are 1–20 m d⁻¹ and 10–600 m² d⁻¹, respectively. Groundwater salinity is also variable, between 800–5,000 µS cm⁻¹.
water table is generally >30 m below the surface, but the salinity processes in this landscape are not well understood. Saline discharge occurs mainly at valley heads and break-of-slope, or at the interface with the underlying rocks. The GFS response time may be relatively fast in the smaller, permeable local systems.

4. **Alluvial deposits** occur in a pattern of fingers with broad deposits extending from the Goulburn river plain into some of its tributaries. The alluvium of the Goulburn River and some tributaries have numerous terraces and paleo-channels, resulting in a network of oxbows and swamps along the Goulburn River. In the higher parts of the catchment, local alluvial deposits can also be found.

- **Regional GFSs in riverine plains** are found in the vicinity of the Goulburn River. Groundwater in some of the alluvial upland systems (see below) is directly connected to the groundwater system in the principal river deposits, which can be 80 m deep. Groundwater flow is dominated by sandy and gravelly stream bed deposits between finer floodplain deposits. Values of $T$ can be as high as 500–3000 m$^2$ d$^{-1}$ with $K$ up to 25 m d$^{-1}$. Groundwater EC is 800–8,000 µS cm$^{-1}$ and increases downstream (see Section 2.4). This groundwater system probably receives direct recharge from the river network during flood episodes. Water tables are relatively shallow and show a rising trend. Because of the large, slow-responding nature of this groundwater system, revegetation measures are probably not applied most effectively here.

- **Local to intermediate GFSs in upland alluvium** are most common in the northern half of the Southwest Goulburn area. The make up of these systems is similar to the regional system described above but smaller. Values for $K$ and $T$ of 10–30 and 300–700 m$^2$ d$^{-1}$, respectively, have been observed. These aquifers are also typically less saline (<1,500 µS cm$^{-1}$) and thought to be recharged primarily during flood events. Salinity problems have not yet manifested themselves in this landscape, but there is potential for this as the system may be filling up from the river plain upwards.

- **Local GFSs in Tertiary gravel caps** are found on top of small to moderate hills south of Mangalore airport that have remained after river erosion (Figure 1). They are a mixture of clay and gravel cemented by iron oxides. The resulting often low permeability and relatively small sizes caps produce many small groundwater cells. Not much is known about salinity processes in these deposits, but groundwater EC is usually low to moderate. The small size of these systems suggests that they may respond to revegetation relatively rapidly.

2.3. **Salinity process knowledge**

Cheng *et al.* (2003) also reviewed previous investigations in the area that provide some important clues about the local hydrogeological behaviour.

Earlier reports have suggested that ‘wash-off’ is an important mechanism driving stream salinity in the Southwest Goulburn (RWC, 1988; SKM, 1996). Wash-off refers to the process through which saline groundwater does not discharge into the stream directly, but comes to the surface and evaporates. The accumulated salt may subsequently be washed into the stream by surface runoff. This has implications for the modelling of stream salinity, but also has practical implications. For example, it suggests that salt delivery to the stream may be prevented not only by reducing groundwater recharge, but also by reducing the generation of runoff on or near the surface during storms. Further work to unravel the local importance of
different salt delivery mechanisms has been recommended in many previous studies, but is essentially still lacking (e.g. RWC, 1988; SKM, 1996; ANU, 2001; Cheng, 1999; Cheng et al., 2003).

Allan (1994) mapped saline discharge areas based on the occurrence of salt-tolerant plant species. The total area of saline discharge in the Southwest Goulburn was estimated to be at least 725 ha. More recently, SKM (2001) modelled the likely occurrence of discharge areas from groundwater, soil and topographic attributes and known incidence of discharge. This suggested the total area of salt discharge to be many times greater. The SKM (1996) report suggested that the groundwater system was reasonably close to equilibrium with present land use, and therefore stream salinity was not expected to increase substantially in future.

Cheng (1999) gave salinity scores to 69 subcatchments within the Goulburn-Broken based on observed (changes in) groundwater tables, surface salinity discharge and streamflow EC. Six of the 16 highest scoring subcatchments were located in the Southwest Goulburn: Whiteheads, Major, Kurkuruc, Cornella, Mollisons and Dry Creeks. Cheng et al. (2003) undertook a literature review (on which this chapter is largely based), collated data, and selected two case study areas for further study, each less than 50 km² in size and dominated by sedimentary rocks. One was the upper Gardiner Creek, the other the Hamilton/Dry Creek area (draining into Kurkuruc and Sunday Creeks, respectively). In both catchments, boreholes were drilled and tested, streamflow and EC were monitored, and soils were mapped and analysed. Although hydrological data collection up to the time of reporting was limited because of dry conditions, interpretation of the collected and collated data and hillslope hydrological modelling already provided some useful insights. The model suggested that about 50% of these areas needed to be planted to achieve a salt yield reduction of 30%. However, less drastic land use changes could still achieve some salt yield reductions. The modelling also pointed at salt accumulation in the root zone as a potential threat to long-term tree production in lower parts of the landscape. The collected hydrogeological data pointed at the importance of localised geological structures such as fracture zones and formation boundaries for salt mobilisation and delivery.

The Pine Creek catchment is a small (3.2 km²) catchment in the Southern part of the Southwest Goulburn near Broadford and was converted from grassland to *Pinus radiata* forest in 1986–1988. The changes in stream flow and quality during the first six years after afforestation (1989–1994) were reported by HydroTechnology (1995). Before planting, the property suffered severe soil erosion but land salinisation was not a problem although stream salt yields were substantial. Over the reported period (1–6 years after afforestation), both water and salt yields were reduced significantly, while groundwater levels also declined by ~2 m. This experimental site is revisited in Section 3.

### 2.4. Borehole salinity data

Salinity measurements for 2104 boreholes distributed across the Goulburn-Broken catchment were obtained from the Groundwater Database provided by DSE Victoria (www.dse.vic.gov.au/dhre/grndwtr/grdata.htm). Data provided include the location of the bore, its depth, intended bore use, the depth the bore is screened to intersect the aquifer, a typical salinity analysis, the type of bore owner and the date the bore was constructed. Data sources include the former Rural Water Corporation and Minerals and Energy Department, and other Government Departments, industry and the community.
The associated distribution of salinities is shown in Figure 2 and is approximately log-normal, with ~15% having an EC greater than twice the mean EC of 1500 µS cm\textsuperscript{-1}. This distribution includes a variety of different hydrogeological units, however. It also includes a variety of borehole types, depths, uses, and times and ways of measurement.

To study differences in groundwater salinity between GFS classes, the measured borehole EC values were assigned to GFS classes based on the regional salinity province map (Figure 1). This may have lead to some misclassifications (the GFS where mapped on the basis of surface geology and topography, whereas the GFS may be located in an underlying, different hydrogeological formation). Despite the classification and measurement uncertainties, some clear differences between GFS classes are revealed (Table 2). Furthermore, closer inspection of the data within each class in most cases also showed a relationship between groundwater EC and mean annual rainfall that can be described by an inverse power equation (Figure 3). This was not the case for boreholes in fractured basalts, which occurred over a limited rainfall range only (mean 1274 µS cm\textsuperscript{-1}, Figure 3), and tertiary gravel caps, which had only one borehole (628 µS cm\textsuperscript{-1}).

Although annual rainfall explains up to 64% of the variance in borehole salinity, the scatter around the best-fit relationship is still considerable (Figure 3). An indication of the spatial variability in local groundwater salinity can be obtained through normalising the remaining variation; that is, expressing it as a fraction of values predicted by the regression equations in Figure 3. The resulting distributions are shown in Figure 4. The skewness of the distribution reflects the degree to which salt is concentrated in local stores. Alluvial aquifers appear to have a high degree of variation in salinity. River and upland alluvium deposits are likely to receive groundwater from the adjoining aquifers, and texture and geomorphology are also determined by nearby geology. Therefore, groundwater salinity can be expected to vary significantly with the surrounding geological landscape, which may explain the greater spatial variation. Granites show relatively little variation in EC, although ~10% of the boreholes have a salinity of more than four times the estimated average (Figure 4). It should be noted that
planting trees in parts of the landscape with high groundwater salinity does not necessarily lead to a greater reduction in salt yield. A factor that potentially decreases land use change effectiveness in these areas is that the fraction of excess rainfall (rainfall minus water use) that reaches the groundwater system may well be smaller (see Section 3.6).

The average and range of salinities shown in Figure 3 agree reasonably well with earlier reported values (NDSP & MDBC, 2001; Cheng et al. 2003) (Table 2). The borehole data suggest less difference in salinity between weathered and unweathered rocks than reported by earlier studies. This may be related to the fact that both rock types occupy roughly the same area. This increases the possibility that boreholes were assigned to the one GFS, but in reality had groundwater from the other GFS. Alternatively, the two GFSs may indeed not have very different salinity in reality. The salinity for Cambrian and Ordovician sedimentary rocks and granites were at the lower end of the range of earlier reported values, whereas that for upland alluvium was at the higher end. The only groundwater EC value measured in tertiary gravel deposits was much lower than estimated by regional hydrogeologists. These deposits occur only in a few places in the Southwest Goulburn area and therefore their importance is limited (see Section 4).

Figure 3. Relationship between mean annual rainfall and borehole salinity for seven of the eight GFSs found in the Southwest Goulburn area. Data are for the whole Goulburn-Broken catchment. Best-fit power equations are also given. Note that vertical axes have a logarithmic scale.
Figure 4. Local variation in normalised borehole salinity. Measured borehole salinity was divided by the salinity value estimated from mean annual rainfall using the equations in Figure 3. Legend refers to GFS types: (1) fractured sedimentary rock, (3) weathered sedimentary rock, (4) fractured Cambrian rock; (6) riverine plains; (7) weathered granites; (8) fractured basalts; and (9) upland alluvium.

Table 2. Summary of borehole salinity data and estimates derived by regional expert consultation (NDSP/MDBC, 2001; Table 1) and by literature review (Cheng et al., 2003; Section 2.2). All values are in $\mu$S cm$^{-1}$ and rounded to the nearest two significant numbers or multiple of ten.

<table>
<thead>
<tr>
<th>Groundwater flow system</th>
<th>Borehole EC</th>
<th>regional consultation</th>
<th>Cheng et al. (2003)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fractured sedimentary rocks</td>
<td>830</td>
<td>30–10,000 (371)</td>
<td>800</td>
</tr>
<tr>
<td>Weathered fractured sedimentary rocks</td>
<td>900</td>
<td>100–4,000 (144)</td>
<td>1,200</td>
</tr>
<tr>
<td>Fractured Cambrian/Ordovician rocks</td>
<td>1,500</td>
<td>100–7,000 (37)</td>
<td>3,900</td>
</tr>
<tr>
<td>Weathered granites</td>
<td>330</td>
<td>30–7,000 (162)</td>
<td>1,900</td>
</tr>
<tr>
<td>Fractured basalts</td>
<td>1,300</td>
<td>300–5,000 (87)</td>
<td>1,900</td>
</tr>
<tr>
<td>Riverine plains</td>
<td>2,000</td>
<td>50–10,000 (1060)</td>
<td>1,600</td>
</tr>
<tr>
<td>Upland alluvium</td>
<td>1,200</td>
<td>30–10,000 (179)</td>
<td>950</td>
</tr>
<tr>
<td>Tertiary gravel</td>
<td>630</td>
<td>n/a (1)</td>
<td>3,800</td>
</tr>
</tbody>
</table>

$N$=number of boreholes; n/a=not available
3. Stream flow data analysis

3.1. Introduction

The availability of stream flow and quality data in the Southwest Goulburn is greater than most other areas in the Murray-Darling Basin. These data can be used in two ways:

- long-term average water and salt yields can be used directly for BC2C model calibration;
- daily data can be analysed to make inferences that assist in BC2C calibration.

This section addresses the latter of these two uses. The BC2C model cannot be used directly to analyse daily data, as the model structure only describes long-term average fluxes. Therefore, a simple, custom-built model was used that largely has the same conceptual structure as the BC2C model but with added temporal detail.

The model includes a surface water module and a subsurface module consisting of three parallel linear reservoirs with different response times. The surface water component only serves to provide estimates of drainage from the root zone and does not affect the eventual BC2C model calibration, which was based directly on long-term average stream flow measurements (Section 4). The main purpose of the modelling presented in this section was to obtain estimates of the partitioning of deep drainage into near-surface, sub-surface and groundwater pathways, respectively. This was done by fitting the model to the stream flow and salinity records.

The model analysis provides a better understanding of the dominating salt sources and pathways to the stream and its implications for salinity modelling; as well as estimates of the groundwater flow system (GFS) attributes that influence predictions of stream salinity and its response to land use change by BC2C model.

3.2. Data description and processing

The daily stream flow data and discontinuous instantaneous EC data were downloaded from the Victoria Water Resource Data Warehouse web site (//www.vicwaterdata.net/). Typically, these stations have been established at convenient locations, often some distance upstream from the confluence. The subcatchment is defined here as the area that contributes flow to the gauging station (cf. Figure 5). These data include daily stream flows for a number of subcatchments in the Southwest Goulburn.

Measurements of stream flow electrical conductivity (EC in $\mu$S cm$^{-1}$) are discontinuous and only available for four subcatchments: Sunday, Sugarloaf, Hughes and Major Creek. In addition, a valuable dataset containing daily measurements both stream flow quantity and quality has been collected for the small Pine Creek subcatchment (3.2 km$^2$, nested inside the Sunday Creek catchment) since its conversion from grassland to plantation forest in 1986–1988.

The stream flow data available for downloading are derived from stream water level records converted to stream flow using so-called rating curves and are quality controlled and gap-filled (see //www.vicwaterdata.net/). Two types of data are available, based on manual and
automated water level recordings, respectively. Because of the many gaps in the manual measurements and the lack of corresponding EC measurements, we only used the more recent automatically recorded flow time series, starting between 1971–1975 in the four larger subcatchments and 1988 (shortly after afforestation) in Pine Creek. Except for Major Creek, all gauging stations appear to be still active.

Daily EC measurements were only available for Pine Creek. Manual instantaneous EC measurements have been made infrequently in the other catchments since 1976. Streamflow EC was measured monthly, except for two intensive measuring campaigns during which EC at the Sunday Creek and Sugarloaf Creek gauging stations was recorded every 4 hours (1 July – 1 November 2003; Figure 10). Because only daily total flow volumes were available, we used the average daily EC values during these periods.

Without daily EC measurements for the four larger subcatchments, we could not estimate salt discharge directly for most days. Instead, the available daily flow and EC data were used to construct water-salt discharge rating curves for each of the four larger subcatchments. Using these rating curves, the missing data was estimated assuming that equal flow rates have equal salt loads. Because EC measurements were made at fairly regular intervals and throughout most of the period with flow data, the available data is expected to represent a reasonably representative and unbiased sample. In that case, the approach followed will produce realistic estimates of longer-term mean salt yield. The same approach cannot be used to investigate the inter-annual variation in salt yields, because on an annual basis the sample size is too small to derive reliable water-salt rating curves.

For the range of EC values dealt with in this report, it is assumed that 1 µS cm⁻¹ is equivalent to 0.64 mg l⁻¹ salt (conversely, 1 mg l⁻¹ ~ 1.6 µS cm⁻¹). The uncertainty or error in this conversion can be expected to be within 15%, depending on actual EC (e.g. McCutcheon et al., 1992).

### 3.3. Sub-catchment characteristics

The locations of the subcatchments and corresponding gauging stations are shown in Figure 5, details on data availability and subcatchment characteristics are given in Table 3. Below follows a brief description of each subcatchment.

**Sunday Creek** (337 km²) is located in the Southern part of the area. It is largely dominated by fractured sedimentary rocks with numerous isolated pockets of more weathered rocks (Figure 1). The topography is hilly and elevation between 157 m AHD at the gauging station to 780 m in the far south, where it borders onto a granite outcrop. Mean annual rainfall is 820 mm y⁻¹, of which about 14% leaves the catchment as runoff, and tree cover (which can include isolated trees as well as forests) is substantial at 42% (Table 3). The estimated mean annual salt yield is 37 t km⁻² y⁻¹ for this subcatchment, whereas the flow-weighted EC (the ratio of mean annual salt and water yields) is 534 µS cm⁻¹ (flow-weighted EC values will be lower than the simple mean of all individual measurements, as low flows occur more often than high flows, and typically will have higher EC values).
Figure 5. Location of the five subcatchments within the Southwest Goulburn that have stream flow and salinity measurements (in the darker shade; solid dots indicate the gauging stations). The other gauging stations (open dots) have flow records of varying length, but no or insufficient EC data, or receive flows from upstream.

Table 3. Water and salt discharge from the four subcatchments in the Southwest Goulburn that have flow and EC measurements. Flow was measured continuously, EC and salt loads are estimated based on more than a hundred discontinuous EC measurements at each station.

<table>
<thead>
<tr>
<th>Stream</th>
<th>Sunday Creek</th>
<th>Hughes Creek</th>
<th>Sugarloaf Creek</th>
<th>Major Creek</th>
<th>Pine Creek</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Station details</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Location</td>
<td>Tallarook</td>
<td>Tarcombe Road</td>
<td>Ash Bridge</td>
<td>Graytown</td>
<td>Broadford</td>
</tr>
<tr>
<td>Station ID</td>
<td>405212</td>
<td>405228</td>
<td>405240</td>
<td>405248</td>
<td>405290</td>
</tr>
<tr>
<td>Flow records</td>
<td>from 13-Mar-75 to 31-Dec-03</td>
<td>from 6-May-75 to 31-Dec-03</td>
<td>from 6-Feb-73 to 31-Dec-03</td>
<td>from 20-Apr-71 to 18-Jul-01</td>
<td>from 20-Sep-88 to 23-Jun-88</td>
</tr>
<tr>
<td>EC records</td>
<td>from 3-Feb-78 to 18-Feb-04</td>
<td>from 12-Oct-76 to 8-Jun-88</td>
<td>from 7-Oct-76 to 18-Feb-04</td>
<td>from 6-Oct-76 to 18-Feb-04</td>
<td>from 20-Sep-88 to 9-Dec-03</td>
</tr>
<tr>
<td><strong>Subcatchment characteristics</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Area * km²</td>
<td>337</td>
<td>471</td>
<td>609</td>
<td>282</td>
<td>3.2</td>
</tr>
<tr>
<td>Mean annual rainfall mm y⁻¹</td>
<td>820</td>
<td>838</td>
<td>714</td>
<td>625</td>
<td>768</td>
</tr>
<tr>
<td>Mean annual PET mm y⁻¹</td>
<td>1138</td>
<td>1188</td>
<td>1164</td>
<td>1234</td>
<td>1149</td>
</tr>
<tr>
<td>Woody Cover %</td>
<td>42</td>
<td>24</td>
<td>16</td>
<td>52</td>
<td>**</td>
</tr>
<tr>
<td><strong>Flow statistics</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Mean daily runoff ML d⁻¹</td>
<td>102</td>
<td>211</td>
<td>204</td>
<td>46</td>
<td>0.45</td>
</tr>
<tr>
<td>Mean daily salt load t d⁻¹</td>
<td>85</td>
<td>78</td>
<td>197</td>
<td>55</td>
<td>0.29</td>
</tr>
<tr>
<td>Flow-weighted EC µS cm⁻¹</td>
<td>534</td>
<td>238</td>
<td>617</td>
<td>766</td>
<td>154</td>
</tr>
<tr>
<td>Mean annual runoff mm y⁻¹</td>
<td>111</td>
<td>164</td>
<td>122</td>
<td>60</td>
<td>51</td>
</tr>
<tr>
<td>Mean annual salt load t km⁻² y⁻¹</td>
<td>37</td>
<td>25</td>
<td>42</td>
<td>29</td>
<td>11</td>
</tr>
</tbody>
</table>

* as reported by data provider (http://www.vicwaterdata.net/)
** afforested between 1986 and 1988
Hughes Creek (471 km²) forms an eastern extension of the catchment area. It is mostly underlain by granites, but a small area on the western edge of the sub-catchment consists of sedimentary rocks and has some surface outbreaks of saline groundwater. The topography is hilly with relatively steep slopes in places, and elevation is 164–682 m. Mean annual rainfall is 838 mm y⁻¹, of which 20% runs off, and tree cover is rather low at 24%. Salt load and flow-weighted EC are both lower than for the other three larger subcatchments, at 25 t km⁻² y⁻¹ and 238 µS cm⁻¹, respectively.

Sugarloaf Creek (609 km²) receives flow from two distinctly different landscapes. The granite uplands are similar to those in the Hughes Creek subcatchment and largely drain into Mollisons Creek. The rest of the area is made up of the same sedimentary rocks found in Sunday Creek, but is partially covered by more recent basalt outflows (Figure 1). Some of the sedimentary rock hillsides drain into Kurkuruc Creek before it merges with Mollisons Creek to form Sugarloaf Creek. Saline discharge to the surface is found in the lower, less hilly part of the catchment. Elevation is 152-752 m, mean rainfall 714 mm y⁻¹ and runoff fraction 17%. Trees cover is lower than in the other subcatchments (16%). Salt yields from this catchment are high at 42 t km⁻² y⁻¹, while mean flow-weighted streamflow EC is 617 µS cm⁻¹.

Major Creek (282 km²) is a more undulating subcatchment north of Sugarloaf Creek. It differs from the above three catchments in that it has a lower elevation (142-547 m), less rainfall (625 mm y⁻¹), a lower runoff fraction (10%) and more substantial alluvial deposits in the valley bottoms. Otherwise the geology is of the same sedimentary nature as found in Sunday Creek and the lower Sugarloaf Creek subcatchments. It also has substantial (52%) tree cover. Estimated sub-catchment salt yield is 29 t km⁻² y⁻¹, which is not particularly high in comparison with the other subcatchments, but because runoff is about half that of the other areas, mean streamflow salinity is relatively high at 766 µS cm⁻¹.

The much smaller Pine Creek (3.2 km²) subcatchment is located along the Hume highway near Broadford and Kilmore at an altitude of 250-400 m, and is part of the Sunday Creek subcatchment. Virtually all of the catchment is contained within the 516 ha Glenburnie estate. Before 1986, the property contained open grasslands and experienced severe soil erosion, but land salinisation was not a significant problem. The creek itself was 3-5 m wide and had incised ~1.2 m into the valley bottom, exposing fractured shales. There was evidence of groundwater discharge into the creek near the catchment outlet (HydroTechnology, 1995). The entire property was planted to Pinus radiata in 1986 (203 ha), 1987 (251 ha) and 1988 (6 ha) by Smorgon Forests, with ~3 ha of trees having failed by the end of 1987.

Stream flow and EC were automatically recorded since 20 September 1988, that is, about 1-2 years after most trees were planted. These data provide key information on the magnitude and rate of change in water and salt yields after afforestation in this hydrogeological setting. Annual water and salt yields as reported in Table 3 represent mean values for the first 16 years after pine planting and contain a strong decrease in salt and water yields. These mean numbers should be interpreted with caution and it is more useful to look at the change in fluxes over time (see Section 3.5).

Infrequent manual readings of groundwater level and EC have been taken in two boreholes within the Pine Creek subcatchment. One borehole (D2, 307 m AHD) is located approximately in the middle of the catchment, about 15 m above the valley bottom at the lower end of a divergent hillslope, whereas the other (D1, 273 m AHD) is located in the valley bottom, close to the creek and catchment outlet. Both boreholes showed lowering
groundwater levels between 1988–1994, from 21 to 23 m below surface in the hillside borehole (D2) and from 5 to 8 m below surface in the valley bottom (D1). Most of the decline occurred during 1992 only (HydroTechnology, 1995). Groundwater salinity in the higher bore rapidly decreased from ~2,000 to ~1,000 μS cm⁻¹ during 1989, but no trend was obvious in the valley bottom (~3,500 μS cm⁻¹).

3.4. Method of analysis

A simple lumped conceptual process model was custom-built for this study, to analyse stream flow and EC data. The model, called SALMOD for convenience, is not intended as a predictive process model. The conceptual structure of the model is similar to other commonly used linear reservoir models (e.g. IHACRES; Jakeman and Hornberger, 1993) but with salt concentrations added to the stores (Figure 6). The model equations can be found in Appendix A.1. In summary, the model has two main components that were fitted in sequence to the available data:

1. a near-surface water balance component that calculates drainage from the root zone as a function of rainfall, soil water retention properties and two vegetation-specific properties: (i) the rainfall fraction that is intercepted and evaporated from the wet canopy, and (ii) the maximum rooting zone soil moisture storage $S_{\text{max}}$ (which can be interpreted as the product of rooting depth and available soil moisture at field capacity per metre depth);

2. a sub-surface component that directs different fractions of soil water drainage to the stream via three stores with varying response times and EC signatures.

The model application procedure is described in more detail in Appendix Section A.2. Daily rainfall amounts for each sub-catchment were synthesised by weighting daily rainfall from six climate stations and scaling the resulting averages to agree with estimated mean catchment rainfall (derived from the BOM SILO data base, Table 3). For Pine Creek, local rainfall measurements were available and no processing was needed. Daily pan evaporation data from Mangalore airport (near Seymour) were scaled to mean catchment potential evapotranspiration (PET) to derive estimates of daily PET (Table 3). Relative fractions of each subcatchment covered by woody and non-woody vegetation were derived by GIS analysis of land cover data (cf. Section 4.3), except for Pine Creek, where forest canopy development was estimated using a mathematical function. The difference between rainfall interception by non-woody (5%) and woody vegetation (20%) was defined a priori based on literature data (e.g. Bruijnzeel, 2000), whereas tree rooting depth was assumed to be ten times that of non-woody vegetation. It should be stressed that this model component does not affect the eventual BC2C model calibration, for which the measured long-term average stream flow data were used directly (Section 4). The purpose of the surface water balance model component is to provide estimates of rainfall in excess of soil water storage capacity. Three of the parameters in this component were fitted to match observed cumulative flow and its temporal pattern (see Section 3.5).

Subsequently, five parameters describing the sub-surface pathways of drainage were fitted to daily flow data and observed daily salt yields. Groundwater salinity was not fitted but estimated from EC measurements at low stream flow conditions. An additional parameter needed to be introduced to describe the groundwater behaviour for Pine Creek. In analysing these data, it became obvious that salt yields suddenly plummeted in 1992, which appeared to
coincide with stabilisation of the groundwater table. We simulated the associated sudden stop in groundwater discharge by introducing a minimum baseflow rate ($Q_{G,\text{min}}$ in mm d$^{-1}$), below which no baseflow occurs.

**Main model assumptions**

As in any model, the complexity of the system has been simplified through a variety of underlying assumptions, which may not all be valid and in that case can compromise the interpretation. A few important simplifying assumptions and their effect on calculations are outlined below and discussed fully in Section 3.6.

Using cumulative runoff to fit the water balance model, as was done here, will only give theoretically accurate results if soil water drainage is very rapid and the groundwater store is in equilibrium. If this is not the case, runoff can be boosted by water from slow releasing stores during dry periods and fitting the model to this will lead to an underestimation of the effect of soil moisture availability on $E_t$. Indeed, there was some dependency between the surface and sub-surface model components, which was addressed by iterative model fitting (Appendix A.2).

All rainfall is assumed to infiltrate, which any infiltration in excess of storage capacity draining out of the root zone. The model assumes return floe and sub-surface storm flow to be the dominant quick flow mechanisms, and does not explicitly account for infiltration excess.

Figure 6. Illustration of the SALMOD model structure. Shown fluxes are rainfall ($P$), rainfall interception ($E_i$), evapotranspiration ($E_t$), drainage from the root zone ($D$), groundwater recharge ($R$), quick flow ($Q_q$), slow flow ($Q_s$), groundwater discharge ($Q_g$) and stream flow and EC measured at the subcatchment gauging station ($Q_{\text{tot}}$ and $C_{\text{tot}}$, respectively). The thickness of the lines corresponds to the approximate relative size of the fluxes predicted for Pine Creek at 50% tree cover.
or saturation overland flow. If these latter processes are more important, the implication will be that the relative fraction of drainage that enters the fastest subsurface stores will be overestimated. The relative fractions going to the other two stores may be underestimated to make up for the overestimate of soil drainage and fit the measured stream flow pattern. Whether surface runoff is explicitly modelled or not probably does not affect the modelled actual size of the fluxes going into the two slower stores.

Any time needed for water draining from the root zone to travel downwards to the groundwater table is not modelled. Rather this travel time is implicitly combined with the lateral groundwater travel time in the overall fitted groundwater response.

Using a single reservoir to represent the GFS implicitly assumes that one, fully connected groundwater system underlies the modelled subcatchment. This is an important simplification, as some GFSs are likely to consist of separate parallel cells. These cells may not act as linear reservoirs, but instead show a stronger baseflow recession in the initial stages as has been commonly observed elsewhere (Chapman, 1999). Modelling groundwater discharge into the stream as a linear reservoir also assumes that there is no build-up of saline groundwater in temporary stores close to the stream. Furthermore, assigning different, fixed EC values to water that travels through the groundwater system and water that travels through the unsaturated zone, respectively, suggests that these flowpaths do not cross before reaching the stream. In reality, the transport of salt to the stream is probably more complex.

Any evaporation of stream flow is not accounted for. This can lead to overestimations of stream flow and underestimations of EC (but not salt load) during periods of low flows. This leads to convex-upward baseflow recession curves, that is, temporal baseflow changes that show an increasing downwards slope when plotted on a log-normalised scale. A similar curve will also result if some of the flow is lost to an underlying aquifer and/or bypasses the gauging station below the stream bed (Chapman, 1999). These processes can become important where low flows are low in an absolute sense. This is the case in most of the Southwest Goulburn, where creeks (almost) stop flowing altogether during parts of the year.

3.5. Results and discussion

A more comprehensive account of the fitting procedure, resulting parameters and graphs to illustrate the degree to which SALMOD explained the observations are provided in Appendix A.3. Below the main results are summarised.

Model parameters

All parameter values estimated beforehand or optimised are listed in Table A.1 in Appendix A. The fitted water balance model parameters were in reasonable agreement. Fitted maximum soil water storages for grassland were in the range of 63–198 mm for the four large catchments, but lower for Pine Creek (25 mm). Fitted values for the soil water availability response function coefficient $\sigma$ were 1.2–2.3, but higher for Major Creek (4.6) and Pine Creek (5.4), indicative of a later but more sudden drop in ET at low soil water availability. The ‘crop factor’ $k_c$ (the ratio of actual over potential ET under conditions with a dry canopy and sufficient soil moisture) was 0.7-1.2, but there was a strong negative correlation ($r^2=0.95$) between $k_c$ and the area covered by trees (Table A.1). Possibly this is because contrary to model assumptions woody and non-woody vegetation do not have equal transpiration rates under well-watered conditions. However, the variation in $k_c$ may also compensate for the
prescribed and possibly over-estimated differences in rainfall interception and rooting depth. Because of this uncertainty, we did not attempt to correct or use separate $k_c$ values for tree and non-tree cover, respectively. Another reason not to do this was that the BC2C model uses a different method to estimate water use (Section 4.3).

Baseflow salinity was estimated as the mean stream flow salinity for low flow conditions. This produced values of about 1,000 µS cm$^{-1}$ for Sunday and Sugarloaf Creeks, and even higher values for Major Creek (2,555 µS cm$^{-1}$) and Pine Creek (3,868 µS cm$^{-1}$). A much lower value of 346 µS cm$^{-1}$ was found for Hughes Creek, which agrees with the predominance of granites in this subcatchment, compared with the mainly sedimentary substrate of the other subcatchments. Baseflow salinity agreed well with groundwater salinity estimates from regional expert consultation (NDSP & MDBC, 2001) and borehole data (cf. Section 2.2). Moreover, baseflow salinities agreed extremely well with catchment average values of groundwater salinity used in the improved BC2C model application, which are estimated from mean rainfall and GFS class (Section 4.3; Figure 7a, b) and fully independent of streamflow data. This result increases our confidence in these groundwater salinity estimates. The agreement is much less for Pine Creek and highlights the increased spatial variability at this smaller spatial scale.

An estimated 31–66% of excess rainfall ran off within a few days as ‘quick flow’. This fraction appeared to be inversely related to annual runoff, and as a result estimated annual quick flow was similar for all catchments (Table A.3). The fraction of this quick flow that reached the catchment outlet within one day (identical to reservoir coefficient $k_Q$) was 0.28-0.35 for the four larger subcatchments and 0.58 for Pine Creek. This difference can be attributed to the time lag associated with water travelling through the stream network.

Fitted recharge fractions were small for Sugarloaf Creek, Major Creek (both ~1%), Sunday Creek and Pine Creek (both ~6%), but somewhat larger for Hughes Creek (15%). The storage
The coefficient of the groundwater system \( k_G \) was between \( 7.8 \times 10^{-4} \) (Pine Creek) and \( 25.9 \times 10^{-4} \) (Major Creek) \( \text{d}^{-1} \) (Table A.2). This suggests that after a (hypothetical) sudden stop in recharge, groundwater discharge would be reduced by 50% within a year for Major Creek and after ~2.5 years for Pine Creek. These periods (calculated \( t_{50} = -\ln(0.5)/k_G \) ), can be considered equivalent to the groundwater response half-times \( (t_{50}) \) in the BC2C model.

**Stream flow**

Cumulative runoff and its seasonal pattern were modelled with good accuracy for all subcatchments (Figure A.1 in Appendix A). Major Creek flows were overestimated systematically during the later part of the measurement period (after ~1985), which may be related to land use change or a change in the configuration of the gauging station.

Daily flows were also simulated reasonably well in most cases (Figure A.2). The best and worst fits are shown in Figure 8 as flow-duration curves, showing the fraction of time that a particular flow is exceeded. While the fit is good for Hughes Creek, it shows a slight overestimation of low flows. This overestimation was much more for Major Creek and also occurred for the other subcatchments. Accurate modelling of these low flows is important because they reflect the behaviour of the groundwater system. Under-estimation of high flows and overestimation of low flows is a common feature of catchment hydrological models. We aimed to decrease this last effect by including the simulation of log-normalised flows as a fitting constraint, which puts greater weighting on low flows. This apparently did not remove all of the bias, however.

Average annual groundwater discharge was estimated at 26 mm y\(^{-1}\) (16% of total flow) for Hughes creek, but much less for the other four subcatchments (3-7 mm y\(^{-1}\), or 3-6% of total flow). Total flow was also highest in Hughes Creek 167 mm y\(^{-1}\) versus <124 mm y\(^{-1}\) for the others. Two processes potentially explain the overestimation of low flows. Firstly, baseflow
may decrease faster than is modelled by a linear reservoir. This phenomenon is well known and can in fact be expected for unconfined aquifers (Chapman, 1999). Secondly, the actually measured low flows are very low indeed when expressed in mm per unit catchment area (10 ML d\(^{-1}\) corresponds with ~ 0.1 mm d\(^{-1}\) for the four larger subcatchments; Figure 8). It means relatively small losses (such as evapotranspiration from open water and the riparian zone, leakage to an underlying aquifer, and even bypass flow through the streambed) can significantly decrease measured flows.

High flows could be fitted reasonably well considering the simplicity of the model and the fact that estimates of catchment average daily rainfall were used. Annual average values of quick runoff, that is, excess rainfall running off within a few days, typically via (near-) surface pathways, were comparable between subcatchments and was 43 mm y\(^{-1}\) (coefficient of variation CV=0.18), or an average 6% (CV=0.16) of annual rainfall (Table A.1). Slower flowpaths, acting at a time scale of a few weeks, were modelled to contribute another 16-90 mm y\(^{-1}\) or 31-65% of total annual flows.

Salt yield and stream salinity

Daily salt yields could not be modelled very accurately. Nash-Sutcliffe model efficiency, which in this case can be interpreted as the part of variance in the observations that is explained by the model (similar to \(r^2\)), was still reasonable for Hughes, Sunday and Pine Creeks (0.43-0.51), but low for Sugarloaf Creek (0.36) and Major Creek (0.14) (Table A.3). Daily observed and modelled salt loads for Hughes Creek and Major Creek are shown in Figure 9. Low salt yields, occurring at low flows, tend to be overestimated by the model, whereas the highest salt yields, occurring during peak flows, were sometimes substantially under-estimated.

The lack of model performance points to an important weakness in the conceptual model. The model assumes that saline groundwater is discharged directly into the stream and therefore reaches the gauging station as baseflow. However, salt contributions from baseflow were modelled to be only 3–8 t km\(^{-2}\) y\(^{-1}\) (Table A.3). In fact, a comparison of modelled and

Figure 9. Observed versus modelled daily salt loads for Hughes Creek and Major Creek, for which model application was most and least successful, respectively (Appendix A).
observed streamflow rates suggests that the model even overestimated baseflow, and therefore direct salt inputs from baseflow are probably even less.

The model also assumes that near-surface (quick and slow) flowpaths are not in contact with salt stores and therefore have a much lower salt concentration. This is not supported by the data (cf. Figure A.4). Without coming into contact with salt, rainwater near the surface will be concentrated by evapotranspiration. Assuming a rainfall salinity of 12 $\mu$S cm$^{-1}$ (7.5 mg l$^{-1}$), the resulting EC of near-surface water can be expected to be larger than rainwater EC by approximately the ratio of rainfall and runoff. This produces values of 60-120 $\mu$S cm$^{-1}$ for the four larger subcatchments (rainfall-runoff ratio 5-10; Table A.3) and values increasing from 35 to 650 $\mu$S cm$^{-1}$ for Pine Creek as runoff decreases. The fitted near-surface flow EC was only in this range for Pine Creek (68 $\mu$S cm$^{-1}$). For the other catchments, best-fit EC values were significantly higher than ‘theoretical’ values by as much as three to five times (228–531 $\mu$S cm$^{-1}$; Table A.2). This suggests water following these relatively rapid flowpaths mobilises additional salt.

These findings are strengthened by Figure 10, which shows the available direct estimates of salt yield for Sunday and Sugarloaf Creeks together with modelled values for a four month period. Salt yields associated with baseflow are overestimated, but so is salt discharge during periods with intermediate flow conditions. Large pulses of salt appear to occur on single days with significant rainfall and runoff, suggesting that salt is ‘flushed’ out of temporary stores during these events. Different processes can be invoked to explain this behaviour (see Section 3.6), but none of these is described in the model.

Temporal changes in water and salt yield from Pine Creek

We consider the results for Pine Creek separately, because the pattern in annual water and salt yields provides some important clues about the behaviour of this groundwater system. Groundwater discharge suddenly came to a stop in 1992, about 6 years after afforestation. We simulated this by introducing a minimum baseflow threshold (0.02 mm d$^{-1}$), below which discharge stops altogether. With this provision, the model reproduced the annual variation in water and salt yield from the Pine Creek subcatchment very well (Figure 11). Without using a minimum baseflow threshold, salt yields decrease somewhat less rapidly (the dashed lines in Figure 11b). More importantly, in that case modelled EC values would be very high indeed, because total runoff (having a modelled EC of 68 $\mu$S cm$^{-1}$) decreases much faster than groundwater discharge (3,868 $\mu$S cm$^{-1}$).

Two explanations can be formulated for the sudden drop in salt yield. Firstly, it is possible that the groundwater system continued to discharge to the surface after 1992, but that this salt did not reach the catchment outlet anymore, instead perhaps being evaporated in local salt scalds. If this is the case, field evidence of salt scalds and patches of poor growth would be expected. Alternatively, saline groundwater may still be leaving the catchment through alluvium in the valley bottom, but pass underneath the flume and therefore go unmeasured. Both mechanisms are feasible given the small volumes involved: the minimum baseflow threshold of 0.02 mm d$^{-1}$ could be evaporated from a ~100 m$^2$ area (assuming wet soil evapotranspiration rates), or alternatively equates to a modest ~45 l min$^{-1}$ passing under the flume. However, local borehole data also shows a rather sudden decline of groundwater level in 1992, after which it appears to stabilise again (HydroTechnology, 1995). Comparison of the maximum drop in stored groundwater that was modelled (21 mm) to the drop in groundwater level (~2–3 m for the two boreholes) suggest a specific yield of about 1%. This
agrees well with the low fracture space in these sedimentary rocks (~2%; cf. Section 2.2). If groundwater discharge was to continue, another 2–3 m drop in groundwater level since 1992 would be expected. The borehole data available for this report extended until 1994 only, but more recent measurements may be available (see Section 5.2).

3.6. Conclusions and implications for modelling

A simple model was used to analyse stream flow and salinity data for one small (3.2 km$^2$) and four larger (282-609 km$^2$) subcatchments in the Southwest Goulburn area. The purpose of this exercise was to gain a better understanding of the dominating pathways of salt to the stream;
and, secondly, obtain estimates of groundwater flow system (GFS) attributes that determine stream salinity and its response to land use change.

In the Pine Creek subcatchment (3.2 km²), the conceptual model appeared to agree rather well with observations. The decrease in water yield upon afforestation was simulated well by the model, and through the inclusion of a minimum groundwater discharge threshold, the temporal pattern in water and salt discharge as well as groundwater levels could be reproduced well. The groundwater salinity value of 3,868 µS cm⁻¹ that was estimated from low flows agrees well with the ~3,500 µS cm⁻¹ measured in a borehole in the valley bottom, but is higher than observations in a borehole further upslope (HydroTechnology, 1995; cf. Section 3.3). These observations, together with the sudden drop in groundwater discharge after about six years following afforestation, demonstrate the highly dynamic nature and spatial variability of groundwater salinity that cannot be reflected in a simple lumped model.

For the four larger subcatchments, the water balance could also be modelled well. However, this was not the purpose of model application, and a different approach to water balance modelling is used in the BC2C model. The model over-estimated low flows (which can be interpreted as groundwater baseflow), but despite this the salt yields associated with baseflow and those from rainfall together amounted to less than 20-45% of the observed salt yields. The
rest appears to be mobilised and transported out of the catchment during peak flow events. The limited importance of baseflow discharge agrees with the results of Cheng et al. (2003), who studied salinity processes in the Southwest Goulburn and two smaller areas within.

A variety of mechanisms can explain the importance of salt transport during and shortly after rainfall events, three of which are illustrated in Figure 12. Saline groundwater discharge upslope from the stream may (partially) evaporate before the salt is washed off by overland flow. Flushing of saline water or solution of salt in the unsaturated zone by sub-surface flow may also occur where groundwater comes near to the surface (not illustrated). Saline groundwater may also build up in permeable substrate in the rock aquifer (for example in fracture zones) and be purged into the stream by infiltrated water during rainfall events. Alternatively, it may build up in alluvium near the stream (for example in coarse-textured paleo-channels) and be flushed by overbank flow or water from upslope (Summerell, in prep.; Figure 12). Field methods are available to distinguish between these flowpaths using chemical or isotope signatures in combination with (airborne or on-ground) geophysical exploration methods.

The apparent importance of these two-step salt delivery processes, that is, first mobilisation of saline groundwater and then flushing during peak flows, complicate the interpretation and modelling of stream data. For a start, the modelled recharge fractions will be too low. An alternative estimate of sub-catchment average recharge fraction $F_G$ (the fraction of excess rainfall that becomes groundwater recharge) can be obtained if the following assumptions are made:

- salt concentrations in recharge and water following shallow flow paths are equal and depend on rainfall salinity $C_{\text{rain}}$ (estimated to be 7.5 mg l$^{-1}$);
- with the exception of atmospheric salt contributed by shallow flow paths, catchment salt yield is ultimately derived from groundwater mobilised by recharge;
- discharged groundwater has a salinity that equals the value $C_G$ (µS cm$^{-1}$) estimated for baseflow (cf. Section 4.3);

- over the period of measurement (~30 years), mean annual groundwater recharge $R$ equals total groundwater discharge (whether directly as baseflow into the stream or indirectly, contributing to two-step processes), and catchment water yield $Q_{tot}$ equals excess rainfall (rainfall $P$ minus ET, all mean annual values in mm or ML km$^{-2}$).

In that case, a recharge fraction $F_G$ can be estimated from the salt balance. Of all salt precipitating in rainfall ($P_{C_{rain}}$), a fraction $1-F_G$ reaches the stream directly and so contributes to catchment salt yield. The remaining part of total catchment salt yield $Y_S$ (t km$^{-2}$) is derived from the groundwater system; dividing this remaining salt yield by groundwater salinity $C_G$ yields the corresponding amount of groundwater discharge. Under equilibrium conditions ($R$ equals groundwater discharge, and excess rainfall equals $Q_{tot}$) this give recharge fraction $F_G$ as:

$$F_G = R/Q_{tot} = \frac{1000Y_S - (1-F_G)P_{C_{rain}}}{C_G} \quad (1)$$

The factor 1000 converts to kg km$^{-2}$ y$^{-1}$. Under the stated assumptions, the definitions of parameters and variables in equation (1) are consistent with the BC2C model. The required values of $P$, $Q_{tot}$, $Y_S$ can be found in Table 3, and $C_G$ values are listed in Table A.2 (values were converted to mg l$^{-1}$).

Resulting estimates of recharge fraction are 0.26 (Major Creek), 0.53 (Sugarloaf Creek), 0.38 (Sunday Creek) and 0.57 (Hughes Creek). These values are clearly much larger than those fitted by the model. For Pine Creek ($F_G=0.04$), observed annual salt yields and baseflow were simulated much better than for the larger subcatchments, which increases confidence in the fitted value of 0.06. A strong correlation exists between baseflow salinity and the apparent recharge fraction, mainly because the variation in catchment salt yields is smaller than the variation in baseflow salinity [cf. equation (1)] (Figure 13). It is conceptually reasonable to expect that for two systems with identical aquifer characteristics and excess rainfall (rainfall minus water use), the one with a higher recharge fraction leads to higher groundwater recharge, and therefore shorter groundwater residence times and lower salinity. In Section 2.4 it was shown that groundwater salinity increases as mean annual rainfall decreases, and combined with Figure 13, this would suggest that for the same GFS, recharge fraction decreases with rainfall. While this is not impossible, there is no strong physical rationale for such a relationship, and the five catchments represented in Figure 13 have more or less similar mean annual rainfall (628-838 mm y$^{-1}$). We therefore did not use the regression equation in Figure 13 directly to estimate recharge fraction but took a slightly different approach (see Section 4). Expanding the present analysis to other catchments in future may provide more information on the relationship between rainfall, water use, hydrogeology, recharge fraction and groundwater salinity.

Care should be taken when comparing the apparent recharge fractions calculated with equation (1) or fitted by the model with values derived from local water balance studies. Whereas the above values represent recharge as a fraction of stream flow, water balance studies usually express recharge fraction in relation to soil drainage. This drainage may re-enter the root zone downslope and be (partially) evaporated, therefore not contributing to stream flow. In the extreme case, all groundwater contributing salt via two-step processes may evaporate (for example from saline discharge areas or valley bottoms). When this volume of water is added to stream flow, the calculated apparent recharge fractions decrease to 0.22-
0.46. Because catchment water yield rather than soil drainage is estimated by the BC2C model, the earlier mentioned, higher recharge fractions are the appropriate ones to use.

The groundwater system in the Pine Creek subcatchment showed a rapid reaction to reduced recharge, with salt yields already more than halved within six years after afforestation. This response is much faster than typically assumed for this type of landscape (typically 15 years or more; e.g. Dowling et al., 2004; Vaze et al., 2004). For the larger subcatchments, the fitted groundwater response rate was modelled to be even more rapid. These results should be interpreted with much caution, however, as baseflow was substantially overestimated for three subcatchments, and direct baseflow appeared to deliver only a small part of total subcatchment salt yields. At the same time, the importance of two-step processes for salt delivery suggests that afforestation may also bring about a rapid decrease in groundwater discharge outside the Pine Creek subcatchment. The greater water use of forests leads not only to reduced recharge, but also reduced surface and near-surface storm flow. Therefore, it can be expected that less salt near the surface will be mobilised by these flow components. This issue is discussed further in Section 5.

Figure 13. Relationship between baseflow salinity and apparent recharge fraction, fitted by the stream flow model for Pine Creek (open dot) and estimated with equation (1) for the four larger subcatchments (closed dots).
4. BC2C model application

4.1. Model approach and background

This section briefly summarises the modelling approach that underlies the BC2C model and outlines the methods used to parameterise the model.

A number of different approaches have been taken in previous studies to model catchment hydrological and salinity processes. Broadly, these can be placed somewhere in a spectrum with the extremes of (1) fully mechanistic, spatially-distributed models; or (2) fully conceptual and lumped models. Both approaches have important applications.

Mechanistic, spatially distributed models embed the physical laws that govern hydrological processes and in principle can provide exact predictions with high spatial and temporal detail. These physical process descriptions typically involve many parameters, which can lead to problems and uncertainty if the model’s ‘hunger’ for data cannot be satisfied. This type of model is appropriate when input data with high spatial detail is available, for example for well researched scientific study catchments.

On the other hand, fully conceptual models are largely based on general understanding of, and broad assumptions about the processes involved. This provides the important advantage that a first-pass estimate of the relative importance of hydrological processes in different areas can be obtained with minimal data availability. However, such models cannot reproduce the local variability in processes. This makes them less useful as research tools, but better placed for regional planning in situations with relatively low knowledge and data availability.

In reality almost no model is fully mechanistic (as some simplifying assumptions always need to be made where knowledge and data are lacking) or fully conceptual (as concepts often reflect mechanistic thinking). Hybrids also exists, for example using a relatively highly mechanistic model of vegetation water use and recharge, and a more conceptual description of the groundwater flow system (GFS, Coram et al., 2000). As data on vegetation cover and climate (and to a regionally varying extent also soils) are now available at good spatial and temporal resolution, the greatest unknowns in salinity modelling are typically related to the groundwater system.

The Biophysical Capacity to Change (BC2C) model (Dawes and Gilfedder, in press) was developed to compare the expected spatial and temporal impacts of land use change on stream flow and salinity for different parts of the Murray-Darling Basin. Because of the limited empirical knowledge of the response of groundwater and stream salinity to land use changes; the large variation between different groundwater flow systems; the large area to be modelled; and the generally low data availability (on GFS properties in particular), the BC2C model was designed as a broad-scale low data-demanding model that can be classified towards the conceptual end of the model spectrum. A description of the assumptions, simplifications and algorithms underlying the model is given in Appendix B. The model uses spatial data on mean annual rainfall, rainfall salinity, tree cover and GFSs and creates output on an annual time step, assuming average climate conditions. In principle the model can be tested at any resolution depending on data availability, but the models conceptual structure implies that the basic scale of modelling is an idealised hillslope representative for a valley or subcatchment.
Earlier applications of the BC2C model in the CEF project (Van Dijk et al., 2004a, b) used the model parameterisation for the whole Murray-Darling Basin. While this has been useful in selecting a focus research area and further developing the CEF concept, the spatial results were suitable for demonstration purposes only, not for identifying areas within the Southwest Goulburn with the best stream outcomes after afforestation. This is particularly true for the effect of afforestation on salt generation; water yield change predictions have been shown to be rather well predictable with a broad-scale approach (but see Van Dijk et al., 2004b, and Section 4.4 for qualifications on this).

Because local targeting of CEF for maximum environmental benefits constitutes an important part of the project objective, more realistic predictions of the afforestation impacts on salt generation are required. Fortunately, the data availability in the Southwest Goulburn is greater than in many other areas, and it is therefore feasible to derive more certain predictions of subcatchment outcomes by using these data to parameterise the BC2C model. This can assist in prioritising subcatchments or valleys where desired stream outcomes are most likely to be achieved. Such a BC2C application is not suitable to identify more and less suitable CEF locations within these valleys and within hillslopes. The FLUSH model is being developed and tested in the focus area for this purpose (Gallant et al., 2004), and will be integrated with the methodology being developed in CRC for Catchment Hydrology’s Salt Prediction project (project 2C). As both the BC2C and FLUSH approaches use the same terrain and stream flow data for calibration, the results will be consistent.

4.2. Model structure

The BC2C model structure incorporates relationships between vegetation type and water use; the influence of vegetation type on the near-surface water balance; and the spatial and temporal relationships between changes in land use and changes in stream salinity for specific types of groundwater flow system. The model can operate in a GIS environment and requires spatial data on elevation, mean annual rainfall, rainfall salinity, tree cover and groundwater flow systems (GFSs; Coram et al., 2000). The output is generated on an annual time step and includes the water balance components and water and salt discharge for each delineated subcatchment.

The BC2C model has been tested using measured stream flow and salinity time series in a number of smaller catchments (Dawes et al., in press; Dowling et al., 2004). The model generally reproduced the observed trends well although the actual numbers did not always agree, which was attributed to the strong spatial heterogeneity in GFS characteristics in the reality. BC2C has also been used for the entire Murray-Darling Basin to provide a spatial analysis of the expected stream flow and salinity impacts of afforestation, with input data in grids with a resolution of 0.25 to ~10 km, depending on the type of data (Dowling et al., 2004).

The BC2C structure used here for the Southwest Goulburn shows similarities with that used in the Little River catchment (Evans et al., 2004). In summary, the typical model application proceeds through the following steps (more details and all algorithms are given in Appendix B):

1) For each grid cell, water use (the sum of all evapotranspiration components) is estimated based on land cover (forest or non-forest) from empirical relationships between long-term annual rainfall and water use. The BC2C model estimates
evapotranspiration using a simple but well-established empirical relationship that states that water use increases with rainfall (Zhang et al., 1999, 2001).

2) A map of groundwater flow systems (GFS) is used to estimate the fraction of excess rainfall that drains to the groundwater system. The rest of the excess rainfall reaches the stream via near-surface runoff.

3) A digital elevation model (DEM) is used to delineate hydrological landscape units (HLU) of a desired approximate size (1000 ha in this case, often representing individual valleys). For each HLU, the mean hydraulic gradient and flowpath length is estimated from the DEM.

4) Using the relative distribution of different GFSs for each HLU, mean values of the hydrological attributes hydraulic conductivity, transmissivity, and specific yield are calculated. This is combined with the topographic attributes and the estimated change in recharge to estimate how fast the groundwater system responds to land use change.

5) The results from all grid cells are aggregated to yield totals at the scale desired (subcatchments and catchment).

Outputs from the model are 100-m resolution grids with the changes in water yield, salt yield and salinity. The appropriate way of interpreting the results is by viewing them as estimates of what happens to water and salt outputs from an individual HLU when all the non-forested land in that HLU was to be afforested. There are two important reasons for this:

- The Zhang curves that underlie the water use estimates were based on catchment water yields, and are not necessarily accurate for part of a subcatchment or hillslope. Within larger catchments, water not used upslope may be used by vegetation downslope or further downstream.

- Some of the groundwater system attributes (recharge change, flowpath length and hydraulic head) are estimated by assuming full afforestation of a HLU, and using average HLU topographic and GFS characteristics.

For the above reasons, comparing results on a cell-by-cell basis is not appropriate, and a different modelling approach, such as FLUSH, is needed to derive more realistic predictions for different areas within the defined valleys (HLUs). The grids however still do convey useful information of terrain characteristics within HLUs, in particular on the local presence of GFSs with different characteristics and the areas with and without forest cover at present. Therefore, with these caveats in mind, there is still value in looking at the gridded data. It should also be stressed that, while hillslope level modelling will provide a better theoretical capacity to predict differential impacts at this level, this does not necessarily mean that the uncertainty is reduced. A real reduction in prediction uncertainty can only be achieved by additional data collection (see Section 5.2).

4.3. Materials and methods

The BC2C model was coded and run in the ArcInfo™ GIS environment. Water and salt yields were calculated for two scenarios, that is, present land use and full afforestation. This can still provide an estimate of the effects of partial afforestation (Section 4.1). To run the model, a number of spatial data sets and parameter values are required and are described below. All spatial input layers are shown in Appendix C.1.
A surface of mean annual rainfall was obtained from the Bureau of Meteorology (BOM, 1999). It was derived by interpolation of station data for the period 1961-1990 and had a resolution of 2.5 km. Interpolation was done using the ANU 3D spline fitting surface that estimates rainfall based on rainfall measured at nearby stations and topography, using a 1/40° resolution digital elevation grid.

A potential evapotranspiration (PET) surface was obtained from BOM’s SILO database (www.bom.gov.au/silo/) and has a 5-km resolution. Values represent mean annual Priestley-Taylor potential ET (mm y⁻¹) for the period 1980-99.

Tree cover was derived from the MDB C Structural Vegetation Mapping (M305) data set that was based on Landsat TM classification (Ritman, 1995). Classification was done at 25 m resolution, with trees being classified as woody vegetation >2m tall. The 25 m resolution classification was used to estimate fractional tree cover for 100 m blocks (16 cells).

A rainfall salt concentration surface was used to estimate salt inputs from rainfall (CSIRO Land and Water, unpublished). It was derived by interpolation of station measurements at 24 stations throughout the Murray-Darling Basin (Blackburn and McLeod, 1983). Interpolated values within the Southwest Goulburn values are 6.5–8.5 mg l⁻¹ (corresponding to annual atmospheric salt inputs of ~5-6 t km⁻² y⁻¹).

The regional salinity province map shown in Figure 1 was used to distinguish groundwater flow systems. It was mapped at 1:250,000. The GFS classes each had the following properties assigned to them:

- recharge fraction was estimated from average GFS salinity (described below).
- hydraulic conductivity (K), transmissivity (T), and specific yield (S) estimates were derived through regional expert consultation (NDSP & MDB, 2001; Table 1);
- groundwater salinity was estimated using the regression equations and average values presented in Section 2.4 using the mean annual rainfall layer;

A 25-m resolution digital elevation model (DEM) produced by AUSLIG was used for two main purposes:

- To delineate the subcatchments contributing flow to stream gauging stations. The required gauging station coordinates were retrieved from the Victoria Water Resource Data Warehouse web site (//www.vicwaterdata.net/).
- To disaggregate the landscape into hydrological landscape units (HLUs) of comparable size (~1000 ha) and estimate characteristic aquifer flow length L and gradient H needed to estimate catchment response time (see Appendix B, Section B.4).

Tools were available within ArcInfo™ to determine contributing area and the parameters H and L. Calculated water and salt yields for different scenarios were summed for each gauged subcatchment for comparison with measured values.

The vegetation parameter w describes how water use at low rainfall differs between forest and grassland and can reflect, among others, differences in rooting depth (Zhang et al., 2004). Instead of using the default values that describe a global set of catchment water yield data (cf. Van Dijk et al., 2004b), we used water yield measurements of subcatchments within the Southwest Goulburn to calibrate this parameters for woody and non-woody vegetation. This was done by optimising the value of w for woody and non-woody vegetation by the method of least squares to fit observed mean annual stream flow data for the four larger subcatchments presented in Section 3.
The data layer recharge fraction $F_G$ is an important determinant of salt yield but difficult to estimate. A relationship between groundwater salinity and recharge fraction was found for five subcatchments (Figure 13), but this cannot be expected to hold everywhere. Instead, using the average rainfall for the five subcatchments (753 mm $\text{y}^{-1}$), we estimated groundwater salinity for each GFS and subsequently estimated recharge fraction using the equation in Figure 13. This yielded recharge fractions of 0.43–0.58, with the lowest value for basalts and the highest for granites, respectively (Table 4).

4.4. Testing results

Below some of the main model results are presented and compared with subcatchment data. A comparison with findings of earlier reports (Van Dijk et al., 2004a, b) and implications for Commercial Environmental Forestry are provided in Section 4.5.

Annual average subcatchment water yields estimated by the improved BC2C model application agree well with measured catchment water yields (Figure 14a). The initial and later stage water yields from Pine Creek agree very well with the water yield lines representing no and full forest cover, respectively, which increases our confidence in these results (Figure 14b). The resulting change in water yield before and after afforestation modelled with the recalibrated Zhang curve parameters is greater than predicted using the original values. However, the recalibration was based on five subcatchments in the Southwest Goulburn only. While these represent rainfall regimes in the area reasonably well (mean subcatchment rainfall between 625 and 838 mm $\text{y}^{-1}$), we suspect that the recalibrated curves will underestimate water yields for wetter areas, and therefore the original Zhang curves should be used in other areas.

The Pine Creek catchment was fully forested. Present forestry guidelines exclude areas near the stream from planting, which means that reductions in catchment water yield (but potentially also in groundwater recharge and therefore salt generation) after any future afforestation may be slightly less than predicted. The differential impact of different afforestation scenarios on water yield are discussed in more detail in Van Dijk et al. (2004b).
In general, the BC2C model reproduced observed salt yield and flow-weighed salinity also reasonably well (Figure 14b and c). This was to be expected, as these data were used to estimate actual recharge fractions (Section 4.3). Stream salinity in the Pine Creek catchment 10–15 years after afforestation was strongly overestimated, however. The difference between measured and modelled salt yield was similar to the other cases, but because absolute values of salt and water yield was very small, this had a large effect on modelled salinity. This again emphasises the large uncertainty in subcatchment salinity predictions as both salt and water yield become increasingly small (cf. Section 3.6).
4.5. Changes in stream flow and salinity after afforestation

The improved BC2C model application was used to estimate potential impacts on water yield, salt yield and salinity. The effect of afforestation on stream salinity was calculated as the relative change in combined Goulburn-Broken River EC 5, 15 or 30 years after afforestation of one km$^2$ [equation (B.11) in Appendix B.5]. The spatial distribution of predicted relative EC change after 15 years is shown in Figure 15 and in Appendix C.3, expressed in millionths relative change in Goulburn-Broken salinity per km$^2$ afforested.

The results are markedly different from those obtained earlier in a preliminary study using the whole-of Murray-Darling Basin version of BC2C (Van Dijk et al., 2004a, b). Two important discrepancies in parameterisation explain most of this difference. Firstly, most areas that show up orange in Figure 15 represent weathered granites that were predicted to have high EC benefits in the earlier application. Across the Murray-Darling Basin both saline and relatively fresh granite aquifers occur, depending on local climate and topography (G. R. Walker, CSIRO Land and Water, Adelaide, pers. comm.). The basin-wide BC2C version therefore assumes average, moderate salinity, whereas in actual fact the granite aquifers in the Southwest Goulburn are fresher than other formations. The local sedimentary GFSs in the central part of the area also appear to have much faster response times than previously assumed for such systems across the Murray-Darling Basin (<10 vs. >100 years). This more rapid response can probably be attributed to local topography and agrees well with the observed response in the Pine Creek subcatchment. The differences outlined above demonstrate that the results of modelling should not be looked at in higher detail than warranted by the input data.

The actual change in Goulburn-Broken EC can be estimated by multiplying the relative changes in salinity shown in Figure 15 by flow-weighted average EC, which was 239 µS cm$^{-1}$ for 1985–1994. Similarly, an indication of the change in Murray River salinity can be obtained by multiplying the relative change by ~29.0 µS cm$^{-1}$, which is the number of EC units that the Goulburn-Broken catchment is estimated to contribute to salinity at Morgan (see Appendix B.11 for details). For example, the measured changes in water and salt after afforestation of the Pine Creek subcatchment correspond to a decrease of Goulburn-Broken EC of 47·10$^{-6}$ km$^2$ or 0.015% for the whole 3.2 km$^2$ plantation. This corresponds to a change of ~0.036 µS cm$^{-1}$, and reduced total catchment water yield by ~0.02%. Conversely, it can be estimated that to decrease Murray River EC by 0.1 µS cm$^{-1}$, a change of Goulburn-Broken salinity of ~0.34% or 0.82 µS cm$^{-1}$ will be needed. This is corresponds to ~25 ‘Pine Creek plantations’ or about 8,000 ha of new forest, and would reduce total Goulburn-Broken water yield by about 0.4%.

The distribution of areas with beneficial and rapid salinity changes in Figure 15 coincides well with the distribution of saline groundwater discharge areas as mapped by Allan (1994). While this finding makes intuitive sense, it is not trivial. It suggests the occurrence of such discharge areas may be a very useful indicator for determining good locations for afforestation for environmental benefits in the Southwest Goulburn. These saline discharge areas are typically also associated with erosion features and general land degradation, which may help to achieve multiple benefits by afforestation (upslope) of these areas. It remains to be investigated if this finding is specific for the Southwest Goulburn or has wider applicability.

The spatial distribution and variability in the environmental benefits of afforestation has implications for the importance of targeted afforestation. Following Van Dijk et al. (2004b),
we re-assessed correlations between site afforestation and salinity reduction and water yield for predictions by the improved BC2C model application. All 100-m blocks with expected salinity benefits after 15 years were ranked starting from (1) the greatest stream salinity benefit, (2) the greatest salt yield reduction, or (3) the smallest water yield loss. Subsequently, the respective targeted planting scenarios to achieve these outcomes were evaluated by assuming afforestation would proceed in order from the highest-ranked to the lowest-ranked block (for example, for the scenario targeted at maximising salinity benefits, it was assumed the block with the highest salinity benefit would be planted first). This also means that the overall effectiveness decreases with every additional block being planted. The cumulative effect on the ranking variable and the other two variables was calculated. For the ranking variable, this shows the magnitude of spatial variation and hence the effectiveness of a

Figure 15. Map of the relative change in Goulburn-Broken salinity (in millionths of present EC per km$^2$ of afforestation) after 5 years as predicted by the improved model application, together with mapped saline groundwater outbreaks (Allan, 1994; red lines). Blue colours indicate areas where afforestation is predicted to lead to a decrease of Goulburn River EC, whereas the orange areas indicate areas where this leads to an increase. Actual Goulburn-Broken salinity change can be estimated by multiplying the numbers with flow-weighted average EC (239 µS cm$^{-1}$ for 1985–1994) (For example, a 50·10$^{-8}$ reduction corresponds to $-50·10^{-6}·239 = -0.012$ µS cm$^{-1}$).
targeted planting strategy. For the remaining two variables, a straight line suggests that there is no correlation with the ranking variable and targeted planting does not affect this outcome any differently from random planting. A straight line should also result if expansion occurs in a completely random fashion, which might occur if incentives did not consider the degree to which benefits are realised.

Figure 16 shows the cumulative impacts on water yield, salt yield and stream salinity for the two targeting scenarios. The results in Figure 16a emphasise the effectiveness of targeted planting to reduce salinity: for example, 50% of the maximum achievable salinity reduction after 15 years can be realised by afforesting 19% of the Southwest Goulburn, equal to 28% of the total area with salinity benefits after 15 years (compared to 20% in Van Dijk et al., 2004b). This was predicted to lead to reductions of 29 kt y\(^{-1}\) in salt yield, 52 GL y\(^{-1}\) in water yield and 12% in Goulburn-Broken stream salinity. Caution must be exercised in attaching too much quantitative value to these findings, but they clearly illustrate the importance of targeting new plantation development in areas where salinity benefits can be expected to be greatest. The impact on salt yield is much the same as on stream salinity. Figure 16a further suggests that targeting plantations to maximise salinity benefits leads to somewhat higher-than-average water yield losses. For example, by planting the earlier mentioned 19% of the Southwest Goulburn to target salinity benefits, water yield loss is estimated to be 16% higher than if afforestation was random within the total area with expected salinity benefits. This difference becomes smaller as the afforested area increases, however (Figure 16a).

If plantation expansion is targeted to minimise water yield losses, a different picture appears (Figure 16b). While this approach can indeed save water, it also is much less effective in decreasing salinity. This is primarily because smaller water losses are predicted in the drier northern areas, underlain by slower responding groundwater systems. In these areas, groundwater discharge is still boosting water yields after 15 years, but as a consequence, salinity also remains relatively high.
When the same analysis was performed for modelled changes 5 or 30 years after afforestation, the overall findings remained much the same. The total area with expected salinity benefits increased from 49% (of the total 3,700 km$^2$ Southwest Goulburn area) after 5 years, via 67% after 15 years to 72% after 30 years (this represents 60%, 82% and 88% of the area that is not already fully forested, respectively). These numbers are significantly greater than those suggested in preliminary modelling (Van Dijk et al., 2004b). There are two main causes for this. Firstly: the new model parameterisation predicted greater salt yield reductions and shorter response times for many areas than before. Secondly, the higher resolution of tree cover mapping by looking at greater detail has ‘found’ more (partially) un-forested areas within forest patches.

In reality, the area with real potential for CEF will be further constrained by site productivity, block size, access and transport distance, land ownership, and other factors. Also, planting in different parts within subcatchments and on different sections of hillslopes will have a differential impact on water and salt yield. This will act to narrow down the area with CEF potential, but also to increase the effectiveness of targeted afforestation. The FLUSH modelling framework is being developed to assess differences in water and salt yield impacts within hillslopes (Gallant et al., 2004).
5. Conclusions

5.1. Summary of main findings

[1] The Southwest Goulburn area can be divided into three main hydrogeological and geomorphic units. Granites are found mainly in the higher part of the area, such as the Mollisons Creek and Hughes Creek catchments, and have relatively high and fresh stream flow. The northern plains are dominated by alluvial deposits along with sedimentary rocks. Although groundwater salinity is generally high in this area, the groundwater systems are expected to respond slower to changes in recharge. The southern-central and north-western hillsides are underlain by sedimentary rocks and basalts, which have saline groundwater and are expected to almost fully respond to reductions in recharge within three decades (Section 2.2).

[2] Measurements of borehole salinity for different groundwater flow systems (GFSs) in the Goulburn-Broken catchment generally agreed well with estimates from earlier studies. For most aquifer types, salinity increases non-linearly with decreasing mean annual rainfall. Groundwater salinity was typically >1000 µS cm\(^{-1}\) in areas with ~500 mm y\(^{-1}\) rainfall, and decreased to <200 µS cm\(^{-1}\) in areas with >1000 mm y\(^{-1}\) rainfall (Section 2.4).

[3] The increase of groundwater salinity with decreasing rainfall does not automatically imply that recharge reduction will have more effect in drier areas. Groundwater discharge also tends to be lower in drier areas, and this can potentially compensate for the higher groundwater salinity. For five subcatchments that experienced reasonably similar rainfall, there appeared to be a negative correlation between groundwater salinity and annual recharge, that is, the variation in annual salt yields was much smaller than expected based on groundwater salinity alone (Section 3.6).

[4] Stream flow analysis suggested that groundwater directly discharging into the stream as baseflow is not the most important mechanism of salt delivery. This confirms findings by earlier studies (Section 2.3). Instead, most salt appears to be transported through ‘two-step’ processes: first being brought close to the surface by groundwater and then being delivered to the stream during rainfall events. A variety of physical mechanisms can be conceptualised to explain such two-step processes (Section 3.6).

[5] Water and salt yield monitoring in Pine Creek after afforestation of its 3.2 km\(^2\) catchment area showed a gradual decrease in water yield during the first five years after afforestation and a rapid decrease in salt yield at the end of this period, which agrees with borehole observations. In this catchment, direct baseflow discharge fully explained the observed stream flow and quality patterns (Section 3.6).

[6] The rapid groundwater system response observed in the Pine Creek experiment agreed with BC2C model estimates based on dimensional analysis (Section B.4), in particular when the gradual onset of recharge reduction after afforestation is taken into account. Despite significant uncertainties, both stream data analysis and dimensional analysis suggested similarly rapid response times (<30 years) for the other four subcatchments and in fact for most of the sedimentary rock landscape (Section 3.6).

[7] Observed water yields for four subcatchments and its change over time in the Pine Creek subcatchment were consistent with the existing knowledge of vegetation water use and could be reproduced well. The improved model application confirmed that catchment afforestation typically leads to a decrease in water yield. For the 600-800 mm y\(^{-1}\) rainfall area in the
Southwest Goulburn, water yield reduction is estimated to be in the order of ~100-200 ML \(y^{-1}\) for each square kilometre of land converted to forest. However, a variety of local factors and land use characteristics can change this amount (cf. Van Dijk et al., 2004b) and needs to be taken into account when better estimates are required. In the absence of evidence to the contrary, it may be expected that smaller reductions in water yield will be associated with smaller reductions in recharge (Sections 3.6 and 4.4).

[8] The spatial and temporal patterns in salt yield change after afforestation predicted by the improved model application differ markedly from results obtained by the coarser, generalised model for the Murray-Darling Basin that was used in earlier assessments (Van Dijk et al., 2004a, b). This was expected and indeed was the purpose of the calibration described in this report. It reiterates the absolute necessity of regional knowledge and data to constrain the wide range of possible model outcomes (Section 4.4).

[9] This study reinforces the importance of careful targeting of CEF if environmental benefits are to be realised at all, and in the most efficient manner. Although the model predicted stream salinity reductions could be achieved through afforestation on most (72%) of the not yet forested land (~2,700 km\(^2\)), half of the predicted total achievable salinity reduction could already be realised on 700 km\(^2\) (19% of the area). Also, afforestation in some areas would decrease dilution flows and so in fact increase rather than decrease stream salinity. Overall, the best CEF opportunities were predicted for the southern-central part of the area, where the right combination of hilly topography, high groundwater and stream salinity, and moderate rainfall were found (Section 4.4).

[10] The areas for which the largest and most rapid onset of stream benefits was predicted generally corresponded well with areas where saline groundwater discharges to the surface. These saline discharge areas are typically also associated with erosion features and general land degradation, which may mean that there is good potential to achieve multiple benefits by afforestation (upslope) of these areas. It remains to be investigated if this finding is specific for the Southwest Goulburn or has wider applicability (Section 4.4).

5.2. Scope for improvement

Some of the key uncertainties in the predicted changes in water and salt yields and the scope for improvement are discussed below.

[11] The dominant salt delivery mechanisms in the Southwest Goulburn appear to be two-step processes. This also has important implications for model predictions, as two-step processes may well be affected more rapidly by afforestation, as both groundwater recharge and near-surface runoff are reduced. There is scope to distinguish between these mechanisms using the chemical and isotope signature of stream flow samples, in combination with on-ground or airborne geophysical exploration methods.

[12] Analysis of borehole salinity data demonstrated considerable spatial variation, with a small number of boreholes showing salinities of several times the average value. As the actual afforested area increases, the uncertainty associated with this spatial variability will decrease. Geophysical on-ground and airborne methods can be used to identify areas with high salt concentrations. However, the present study provided some circumstantial evidence that groundwater discharge rates may be inversely related to salinity. In other words, more saline stores may well receive less recharge, and so these salt stores are not necessarily more important than the surrounding, less saline groundwater. This makes intuitive sense, but more
work is needed to confirm this notion. A combination of geophysical and hydrogeological measurements can help to clarify this.

[13] Only well-designed field experiments similar to those conducted in the Pine Creek subcatchment can be expected to really improve our understanding of transient groundwater response to a changed water balance. Any future plantings in the Southwest Goulburn would provide an opportunity to monitor changes in stream flow and salinity that would be very valuable.

[14] Many of the areas for which beneficial stream outcomes were modelled contain saline groundwater outbreaks. Identification and prediction of these areas and areas where groundwater is close the surface has the potential to strongly improve the overall environmental benefit of CEF. Remote sensing techniques may provide cost-effective solutions to identify these areas.

[15] The present model primarily provides a comparison of expected water and salt yield changes between small subcatchments (so-called hydrological landscape units). Both the water and salt budget will vary significantly within these units. Some reasons for this are: (i) water not used in higher parts of the hillslope and catchment may still be evaporated further downslope (also potentially affecting forest productivity); and (ii) local topography typically exerts an important influence on groundwater behaviour. It is feasible to model these processes in a theoretically more consistent manner and for this purpose the FLUSH model is developed as a specific application of the methodology developed by partners in the CRC for Catchment Hydrology Salt Prediction project (project 2C) (Gallant et al., 2004).
References


NDSP and MDBC (2001) A part of Catchment Characterisation and Hydrogeological Modelling to Assess Salinisation Risk and Effectiveness of Management Options - A project to produce a framework and suitable outputs to ensure that funding and resources for salinity management is targeted towards appropriate management activities. Notes from the workshop “Classification of Goulburn-Broken Groundwater Flow Systems”. Benalla, Victoria.


Appendix A. Stream flow analysis

A.1. SALMOD model structure

A simple catchment model was fitted to observed stream flow and salinity data to obtain better understanding of the dominating pathways of salt to the stream, as well as estimates of the groundwater flow system (GFS) attributes that determine stream salinity and its response to land use change. The conceptual model structure is described in Section 3.4. Below the model equations and input data are described, whereas the model results are listed in Section A.2. The model was coded and run in the mathematical software package MatLab\textsuperscript{TM}.

Surface water balance component

Water loss by evapotranspiration (ET) was estimated as the weighted average of the water use of the respective fractions of non-woody and woody vegetation. For each vegetation type $x$ (being woody or non-woody), water use $E_i$ was calculated at a daily time step as the sum of interception losses and potential evapotranspiration ($E_{PET}$, mm d$^{-1}$) multiplied by a crop factor and modified by a water availability function:

$$E_x = F_x k_P P + f(S) k_c E_{PET}$$

(A.1)

and

$$f(S) = \left[1 - \left(\frac{S}{S^*}\right)\right]^{\sigma/|\sigma|}$$

(A.2)

where $E$ is total water use(mm d$^{-1}$), $P$ rainfall (mm d$^{-1}$), $k_P$ the fraction of rainfall lost to interception evaporation, $f(S)$ a water availability function, $k_c$ a crop factor, $S$ (mm) available soil moisture, $S^*$ the water available at field capacity – that is, water content in the root zone after a few days of free drainage minus the remaining water content after ET ceases – and $\sigma$ a coefficient that determines the shape of the water availability function (e.g. a linear response if $\sigma$=1, and a slow initial ET decrease if $\sigma$>1). Interception fractions were estimated a priori at 0.05 for non-woody and 0.20 for woody vegetation, based on literature (Table A.1).

Root zone moisture water storage $S_x$ was modelled as a running water balance with inputs from rainfall and losses from $E_x$:

$$S_x(t) = \min\left\{S_x^*, \max\left[0, S_x(t-1)+P(t)-E_x(t)\right]\right\}$$

(A.3)

where $t$ (d) denotes the time step. Any water in excess of field capacity $S_x^*$ was routed to the subsurface component as drainage ($D_x$, mm d$^{-1}$):

$$D_x(t) = S_x(t-1) - S_x(t) + P(t) - E(t)$$

(A.4)

No drainage was assumed to occur when water content was below field capacity. Although this is probably not realistic and some drainage is still likely to occur at lower water contents, this was not thought to create problems for the present application.
Sub-surface flow component

The area-weighted drainage $D$ was calculated using the fraction $F_x$ of area covered by vegetation type $x$ (woody or ‘tree’, and non-woody or ‘grass’ respectively):

$$D = (FD)_{tree} + (FD)_{grass}$$

(A.5)

For the four larger subcatchments, the areas covered by woody and non-woody vegetation were estimated from the 100 m resolution land cover data used in BC2C modelling (Section 4.3). For the Pine Creek subcatchment, the gradual increase in tree cover associated with growth of the young pine stand was modelled with an exponential saturation function as a function of time after planting ($t$):

$$F_{tree} = 1 - 10^{-\left(\frac{t}{T}\right)}$$

(A.6)

Where $T$ is the number of days (since afforestation, assumed to have been on 1 January 1987) at which 90% canopy closure is achieved, and was estimated to be 2300 days (6.3 years). The remaining area was assumed to have grass cover.

Total drainage was distributed over three stores conceptualised as so-called leaking buckets or linear reservoirs (cf. Jakeman and Hornberger, 1993): a rapidly draining near-surface store releasing quick flow, a slower draining fresh water store, and a slow-draining (saline) groundwater store (the GFS). The resulting quick flow can be conceptualised as storm flow processes (for example, saturation and infiltration-excess overland flow and sub-surface storm flow), whereas the slower draining store may represent lateral flow through the unsaturated zone, or a ephemeral fresh groundwater system perched on the saline groundwater system.

Water storage $h$ (mm) and flow $Q$ (mm d$^{-1}$) from each store for day $t$ is described by:

Table A.1. SALMOD parameter values. Letters indicate fitting parameters and their relationship. Note: groundwater salinity was directly estimated from measurements, whereas three additional parameters were used for the Pine Creek data to describe vegetation development (see text).

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Surface water balance component</strong></td>
<td></td>
</tr>
<tr>
<td>grass interception fraction, $k_p$</td>
<td>0.05</td>
</tr>
<tr>
<td>forest interception fraction, $k_f$</td>
<td>0.20</td>
</tr>
<tr>
<td>grass crop factor, $k_{PET}$</td>
<td>$a$</td>
</tr>
<tr>
<td>forest crop factor, $k_{PET}$</td>
<td>$a$</td>
</tr>
<tr>
<td>maximum soil water store for grass, $S^*$</td>
<td>$b$</td>
</tr>
<tr>
<td>maximum soil water store for forest, $S^*$</td>
<td>10$b$</td>
</tr>
<tr>
<td>soil water availability response coefficient, $\sigma$</td>
<td>$c$</td>
</tr>
<tr>
<td><strong>Sub-surface flow component</strong></td>
<td></td>
</tr>
<tr>
<td>fraction drainage to quick flow, $f_Q$</td>
<td>$d$</td>
</tr>
<tr>
<td>fraction drainage to slow flow, $f_S$</td>
<td>$1-d-e$</td>
</tr>
<tr>
<td>fraction slow flow going to groundwater, $f_G$</td>
<td>$e$</td>
</tr>
<tr>
<td>quick flow drainage coefficient, $k_Q$</td>
<td>$f$</td>
</tr>
<tr>
<td>slow flow drainage coefficient, $k_S$</td>
<td>$f/10$</td>
</tr>
<tr>
<td>groundwater discharge coefficient, $k_G$</td>
<td>$g$</td>
</tr>
<tr>
<td>groundwater salinity (µS cm$^{-1}$)</td>
<td>$h$</td>
</tr>
</tbody>
</table>

Sub-surface flow component
\[ h(t) = [h(t-1) + fD(t)][1-k] \]  
\[ Q(t) = [h(t-1) + fD(t)]k \]  

where \( f \) is the fraction of drainage going to the store and \( k \) (d\(^{-1}\)) is the fraction of storage that has drained after one time step. Because a range of \( f-k \) combinations can be expected to lead to similar results, it is useful to constrain the number of parameters (cf. Jakeman and Hornberger, 1993). To this end, \( k_S \) was fixed at one-tenth of \( k_Q \) (Table A.1). There is always a quick flow component that reacts to rainfall within a few days and therefore has a \( k \) in the order of \( 0.5<k<1.0 \). The slower component will have a \( k \) value that is ten times smaller and therefore respond at time scales in the order of a few weeks rather than days.

An additional parameter needed to be introduced to describe the groundwater behaviour for Pine Creek. In analysing these data, it became obvious that salt yields suddenly plummeted in 1992, which appeared to coincide with stabilisation of the groundwater table. We simulated the associated sudden stop in groundwater discharge by introducing a minimum baseflow rate \( (Q_{G,\text{min}} \text{ in mm d}^{-1}) \), below which no baseflow occurs.

Total stream flow \( (Q_{\text{tot}} \text{ ML d}^{-1}) \), EC \( (C_{\text{tot}}, \mu\text{S cm}^{-1}) \) and salt discharge \( (Y_S, \text{ t d}^{-1}) \) are calculated by:

\[ Q_{\text{tot}}(t) = A[Q_S(t) + Q_Q(t) + Q_G(t)] \]  
\[ C_{\text{tot}}(t) = C_Q \frac{[Q_S(t) + Q_Q(t)] + C_G Q_G(t)}{Q_{\text{tot}}(t)} \]  
\[ Y_S(t) = 0.64 \cdot 10^{-3} C_{\text{tot}}(t) Q_{\text{tot}}(t) \]

where \( A \) (in km\(^2\)) is the area of the subcatchment (km\(^2\) times mm equals ML), \( C_Q \) (\( \mu\text{S cm}^{-1} \)) is the fitted salinity of the ‘fresh’ quick and slow flow components, and \( C_G \) (\( \mu\text{S cm}^{-1} \)) is groundwater salinity and calculated from stream flow measurements. The factor \( 0.64 \cdot 10^{-3} \) converts from EC in \( \mu\text{S cm}^{-1} \) to equivalent salt concentration (g L\(^{-1}\)) (note that 1 Mg = 1 t).

### A.2. Modelling procedure

The near-surface and subsurface model components were fitted separately in sequence. The purpose of the near-surface model component was to provide daily estimates of rainfall in excess of soil storage capacity; it did not influence the eventual BC2C model calibration, which used stream flow data directly (Section 4). It used the following input data:

- **Daily rainfall** from six stations [Broadford (station code 88033), Pyalong (88050), Heathcote (88029), Ruffy (82037), Mangalore (88109), and Seymour (88053)] were weighted depending on their proximity to each catchment (double mass curves were inspected for quality control). The derived average daily rainfall amounts were scaled to agree with the estimated mean subcatchment rainfall (derived from the SILO data base). For Pine Creek, local rainfall measurements were available and therefore no processing was needed.

- **Daily pan evaporation** data from Mangalore airport (station 88109, ~10 km north of Seymour) were scaled to mean catchment potential evapotranspiration (PET) to synthesise estimates of daily PET for each subcatchment.

- The woody and non-woody vegetation cover fractions for each subcatchment covered by were derived by GIS analysis of land cover classification data (cf. Section 4.3), except for Pine Creek, where forest cover development was estimated using equation (A.6).

- The difference between rainfall interception fraction by non-woody (5%) and woody vegetation (20%) was defined a priori based on literature data.
The other parameters were fitted by minimising the difference between observed and modelled cumulative flows. This procedure ensures that both total flow and its temporal pattern are simulated. Fitted parameters were:

- the maximum soil moisture storage for grassland ($S_{grass}$ in mm). The value for woody vegetation ($S_{tree}$ in mm) was taken to be 10 times larger than that of the non-woody vegetation because of its greater rooting depth (Table A.1).
- the ‘crop factor’ ($k_c$, the ratio of actual evapotranspiration $E_t$ over PET);
- the soil moisture response function coefficient $\sigma$, determining how rapid $E_t$ decreases in response to declining soil moisture store.

Subsequently, the sub-surface model component was fitted using the available measurements of daily flow and EC. Drainage was divided over three stores: two fractions ($F_Q$ and $F_S$) going to stores that empty relatively quickly and can be taken to represent surface and near-surface flowpaths that are not in contact with saline groundwater, and one fraction ($F_G$) slower store that represents the groundwater flow system (GFS). Each of these stores has a reservoir coefficient ($k$ in d$^{-1}$), determining the fraction of the water in each storage that flows out of it in one day. The values of $k_Q$ and $k_G$ were fitted, whereas $k_S$ was set to be ten times smaller than $k_Q$. The fraction $F_S$ was not fitted but followed as the rest term needed to maintain conservation of mass ($F_S=1-F_Q-F_G$).

Groundwater salinity was estimated from the stream flow data. By visual assessment of flow-EC plots, it was estimated that flow rates equivalent to $1 \times 10^3$ mm d$^{-1}$ adequately represented conditions in which groundwater baseflow was the only source of water (see Figure A.2). Therefore, we estimated groundwater salinity as the mean EC value for these flow rates. Water passing through the two faster stores was assumed to have a lower EC than water passing through the GFS. A single salinity value ($C_Q$ in $\mu$S cm$^{-1}$) was fitted for the two quicker water flowpaths.

Overall, the model has eight fitting parameters: three for the surface water balance model component ($S^*, k_c$ and $\sigma$) and five for the sub-surface component ($F_Q$, $F_G$, $k_Q$, $k_G$ and $C_Q$). Given the available data, this was considered to be an appropriate number of parameters.

The full daily stream flow record (including occasional interpolated values) and all direct salt yield estimates based on observed flow and EC values in combination with equation (A.8c) were used for model fitting.

To initialise the model, the two near-surface stores were assigned a value of zero (empty), whereas the groundwater store was estimated by assuming it represented the long-term equilibrium between discharge [$k_G h_G(0)$] and recharge ($F_G$ times mean daily stream flow):

$$h_G(0) = \frac{F_G}{k_G} \bar{Q}_{tot,obs}$$  \hspace{1cm} (A.9)

The actual value of $h_G(0)$ is only likely to have had a noticeable effect for the first year(s).

Fitting was done using non-linear downhill search algorithms, where the objective function was to simultaneously maximise the sum of the fraction of explained variance in daily flows, log-transformed flows, and salt yields, respectively, and minimise the relative difference between total observed and modelled flows and salt yields. The evaluation criterion used for fitting was:

$$M(Q_{tot}) + M[\log(Q_{tot})] + M(Y_S) + abs \left( \frac{\sum Q_{tot,mod}}{\sum Q_{tot,obs}} - 1 \right) + abs \left( \frac{\sum Y_{S,mod}}{\sum Y_{S,obs}} - 1 \right)$$  \hspace{1cm} (A.10)

where Nash-Sutcliffe model efficiency $M$ is comparable to the fraction of variance that is explained. Log-transformed daily flows were included to mitigate the tendency of rainfall-runoff models to underestimate high flows and overestimate low flows. Accurate modelling of the latter was important as it was expected to represent mainly groundwater contributions to stream flow.
Because the parameterisation of the sub-surface component (in particular the baseflow component) had some effect on the modelled temporal flow pattern, the surface and sub-surface components were fitted in sequence until further iteration produced no significant improvements in model fit.

A.3. Model results

On the following pages, the results of model fitting are presented. The fitted parameter sets are listed below in Table A.2, and indicators of model performance are listed in Table A.3 and some results are shown in Figures A.1-A.4. The results are presented and discussed in Section 3.5.

Table A.2. Calculated and optimised SALMOD parameters values.

<table>
<thead>
<tr>
<th>Subcatchment</th>
<th>Sunday Creek</th>
<th>Hughes Creek</th>
<th>Sugarloaf Creek</th>
<th>Major Creek</th>
<th>Pine Creek</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Calculated parameters</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Catchment area</td>
<td>$A$ (km$^2$)</td>
<td>337</td>
<td>471</td>
<td>612</td>
<td>291</td>
</tr>
<tr>
<td>Mean annual rainfall</td>
<td>mm y$^{-1}$</td>
<td>820</td>
<td>838</td>
<td>714</td>
<td>625</td>
</tr>
<tr>
<td>Mean PET</td>
<td>$E_{\text{PET}}$ (mm y$^{-1}$)</td>
<td>1138</td>
<td>1188</td>
<td>1164</td>
<td>1234</td>
</tr>
<tr>
<td>Fraction woody cover</td>
<td>$F_{\text{tree}}$ (-)</td>
<td>0.42</td>
<td>0.24</td>
<td>0.16</td>
<td>0.52</td>
</tr>
<tr>
<td>Groundwater EC</td>
<td>$C_{G}$ (µS cm$^{-1}$)</td>
<td>1,193</td>
<td>346</td>
<td>917</td>
<td>2,555</td>
</tr>
<tr>
<td><strong>Optimised parameters</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Crop factor</td>
<td>$k_{c}$ (-)</td>
<td>0.80</td>
<td>1.07</td>
<td>1.15</td>
<td>0.66</td>
</tr>
<tr>
<td>Max. avail. water for non-woody vegetation</td>
<td>$S^{*}$ (mm)</td>
<td>198</td>
<td>76</td>
<td>63</td>
<td>93</td>
</tr>
<tr>
<td>Soil water response function coefficient</td>
<td>$\sigma$ (-)</td>
<td>1.23</td>
<td>1.86</td>
<td>2.26</td>
<td>4.58</td>
</tr>
<tr>
<td>Drainage fraction to quick flow</td>
<td>$F_{Q}$ (-)</td>
<td>0.43</td>
<td>0.31</td>
<td>0.33</td>
<td>0.66</td>
</tr>
<tr>
<td>Drainage fraction to GFS</td>
<td>$F_{G}$ (-)</td>
<td>0.056</td>
<td>0.154</td>
<td>0.012</td>
<td>0.009</td>
</tr>
<tr>
<td>Storage coefficient quick flow</td>
<td>$k_{Q}$ (-)</td>
<td>0.39</td>
<td>0.28</td>
<td>0.35</td>
<td>0.29</td>
</tr>
<tr>
<td>Storage coefficient GFS</td>
<td>$k_{G}$ (-)</td>
<td>$13.3 \cdot 10^{-4}$</td>
<td>$15.5 \cdot 10^{-4}$</td>
<td>$14.3 \cdot 10^{-4}$</td>
<td>$25.9 \cdot 10^{-4}$</td>
</tr>
<tr>
<td>Quick flow EC</td>
<td>$C_{Q}$ (µS cm$^{-1}$)</td>
<td>259</td>
<td>228</td>
<td>337</td>
<td>531</td>
</tr>
</tbody>
</table>

* Estimated with equation (A.6)
Table A.3. Indicators of model performance and modelled components of the water and salt budgets.

<table>
<thead>
<tr>
<th>Subcatchment</th>
<th>Sunday Creek</th>
<th>Hughes Creek</th>
<th>Sugarloaf Creek</th>
<th>Major Creek</th>
<th>Pine Creek</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Model efficiency for daily values of:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>cumulative flow</td>
<td>0.99</td>
<td>1.00</td>
<td>0.97</td>
<td>0.90</td>
<td>0.99</td>
</tr>
<tr>
<td>flow</td>
<td>0.50</td>
<td>0.60</td>
<td>0.42</td>
<td>0.38</td>
<td>0.63</td>
</tr>
<tr>
<td>log-transformed flow</td>
<td>0.47</td>
<td>0.72</td>
<td>0.35</td>
<td>0.11</td>
<td>0.38</td>
</tr>
<tr>
<td>flow-duration curve</td>
<td>0.95</td>
<td>1.00</td>
<td>0.84</td>
<td>&lt;0</td>
<td>0.64</td>
</tr>
<tr>
<td>salt yields</td>
<td>0.47</td>
<td>0.51</td>
<td>0.36</td>
<td>0.14</td>
<td>0.43</td>
</tr>
<tr>
<td><strong>Difference between mean modelled and observed:</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>stream flow (%)</td>
<td>5</td>
<td>2</td>
<td>2</td>
<td>13</td>
<td>-1</td>
</tr>
<tr>
<td>salt yield (%)</td>
<td>-39</td>
<td>3</td>
<td>-31</td>
<td>-6</td>
<td>3</td>
</tr>
<tr>
<td>flow-weighted EC (%)</td>
<td>-40</td>
<td>0</td>
<td>-32</td>
<td>-19</td>
<td>4</td>
</tr>
<tr>
<td><strong>Estimated stream flow components</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>quick flow (mm y(^{-1}))</td>
<td>50</td>
<td>51</td>
<td>40</td>
<td>42</td>
<td>32</td>
</tr>
<tr>
<td>slow flow (mm y(^{-1}))</td>
<td>60</td>
<td>90</td>
<td>80</td>
<td>22</td>
<td>16</td>
</tr>
<tr>
<td>baseflow (mm y(^{-1}))</td>
<td>7</td>
<td>26</td>
<td>4</td>
<td>4</td>
<td>3</td>
</tr>
<tr>
<td><strong>Estimated salt yield components</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>in quick and slow flow (t km(^{-2}) y(^{-1}))</td>
<td>18</td>
<td>21</td>
<td>26</td>
<td>22</td>
<td>3</td>
</tr>
<tr>
<td>in baseflow (t km(^{-2}) y(^{-1}))</td>
<td>4</td>
<td>5</td>
<td>3</td>
<td>6</td>
<td>8</td>
</tr>
</tbody>
</table>
Figure A.1. *Double-mass curves of stream flow*, in which cumulative observed and modelled flows are compared. The dotted diagonal line indicates 1:1 agreement. The horizontal segment in the curve for Sugarloaf Creek can be attributed to interpolation of missing stream flow data. The departure for Major Creek may be related to changes in land use or in the configuration of the gauging station. The serrated curve for Pine Creek can be attributed to overestimations of dry season flow.
Figure A.2. Flow duration curves. Flow-duration curves show the fraction of time a particular flow rate is exceeded and so provide a useful method of comparing flow characteristics, in this case for observed (solid line with closed dots) and SALMOD modelled (dotted line with open dots) daily flows. The model overestimates low flows in all cases (cf. figure A.2), although the agreement with modelled flows is still good for Hughes Creek.
Figure A.3. Daily salt yields. Shown for each subcatchment are salt yields determined from flow and EC measurement, and values simulated by the SALMOD model. Log-log scales are used because of the large variation in salt yields. In all cases there appears a tendency for the model to overestimate low salt yields.
Figure A.4. Flow-salinity relationships. Shown are, for each subcatchment, all available salinity (EC in $\mu$S cm$^{-1}$) data against corresponding flow rates (in ML d$^{-1}$, except for Pine Creek, shown in mm d$^{-1}$). The horizontal lines indicate groundwater EC as calculated over the flow range, and fitted near-surface flow EC (flow range chosen arbitrarily as the equivalent of $>1$ mm d$^{-1}$). Note that vertical scales vary.
Appendix B. BC2C model structure

B.1. Introduction

This appendix provides some of the theory and algorithms of the BC2C model. More background on model philosophy, development and previous applications is given in Section 4.1. For a more comprehensive account the reader is referred to Dawes et al. (in press), Gilfedder et al. (2003), Smitt et al. (2003), Dawes et al. (in press), Dowling et al. (2004) and Evans et al. (2004). In the following sections, the method of calculation is presented for each of the following steps (cf. Section 4.1)

1) Water use, excess rainfall and catchment water yield
2) Recharge, atmospheric salt inputs and salt yield
3) Groundwater response time
4) Aggregation and processing

B.2. Water use, excess rainfall and catchment water yield

The present application of the BC2C model uses the ‘Zhang curves’ (Zhang et al., 1999, 2001, 2004) to estimate water use $E_x$ for a vegetation type $x$, as a function of mean annual potential evapotranspiration (PET), denoted by $E_{PET}$ (mm y$^{-1}$), and mean annual rainfall $P$ (mm y$^{-1}$). The corresponding equation is:

$$E_x = P \frac{1 + w_x \left(E_{PET} / P\right)}{1 + w_x \left(E_{PET} / P\right) + P / E_{PET}}$$  \hspace{1cm} (B.1)

Zhang et al. (1999) collated global catchment water yield data and derived best-fit values for $w_x$ of 0.5 for grassland catchments and 2.0 for forested catchments. However, where stream flow data area available, such as in the Southwest Goulburn, $w_x$ values can be locally calibrated (Zhang et al., 2004). This yielded values of 0.64 and 12 for grassland and trees, respectively (cf. Section 4.4).

Subsequently, excess rainfall is calculated as the difference between rainfall $P$ and water use $E_x$. Assuming that the groundwater system is in equilibrium over longer periods of time, at the subcatchment outlet excess rainfall will equal water yield:

$$Q = P - E_x$$  \hspace{1cm} (B.2)

The BC2C model uses equations (B.1) and (B.2) for grid cells to estimate local excess rainfall for equilibrium conditions under a given scenario. Scenarios used in this study were the (1991) vegetation distribution and full afforestation. Although calculations are made at grid level, the results should be interpreted as aggregated values at a greater subcatchment level, because equation (B.1) was developed and validated for this scale. Within the catchment, more complex processes may take place. For example, lower water use upslope may be compensated by evaporation of water that comes near the surface in lower parts of the catchments.

Also, catchment water yield only equals excess rainfall if there is no change in water storage (for example, in the soil, groundwater system and water bodies). While this is usually a good approximation for longer timescales (for example, >10 years), it may be invalid for shorter periods.
B.3. Recharge, atmospheric salt inputs and salt yield

The excess rainfall calculated with equation (B.1) is split into a fraction $F_G$ that recharges the groundwater system, and a remaining fraction that does not. Thus, recharge $R$ (mm y$^{-1}$) is calculated as:

\[ R = F_G (P - E_s) \quad \text{(B.3)} \]

Different $F_G$ values are assigned to different groundwater flow systems (GFSs). In the present study, they were estimated from specific yield and groundwater salinity (see Section 4.3).

Atmospheric salt inputs $Y_{rain}$ (t km$^{-2}$ y$^{-1}$) are calculated as a function of mean annual rainfall $P$ and rainfall salt concentration $C_{rain}^*$ (mg l$^{-1}$):

\[ Y_{rain} = \frac{1000 \times P \times C_{rain}^*}{1000} \quad \text{(B.4)} \]

The factor 1000 is required to keep the equation dimensionally correct. A grid of rainfall salinity is used, based on interpolation of rainfall station measurements. For the Southwest Goulburn, resulting $C_{rain}^*$ values are ~6.5–8.5 mg l$^{-1}$ (Figure C.1).

Once the groundwater system has reached a (new) dynamic equilibrium, mean annual recharge $R$ should equal groundwater discharge $Q_G$ (mm y$^{-1}$) over longer periods of time. It follows that mean annual salt discharge in groundwater $Y_G$ (t km$^{-2}$ y$^{-1}$) is:

\[ Y_G = \frac{1000 \times Q_G \times C_G^*}{1000} = \frac{0.64C_G}{1000} \quad \text{(B.5)} \]

where $C_G^*$ and $C_G$ are groundwater salinity expressed in mg l$^{-1}$ and µS cm$^{-1}$, respectively. The factor 0.64 converts between the two units. Subsequently, total equilibrium salt yield for each grid cell is calculated as:

\[ Y = Y_{rain} + Y_G \quad \text{(B.6)} \]

B.4. Groundwater response time

The rate of response of a groundwater flow system to changes in recharge depends, among others, on the geometry of the aquifer and its hydraulic properties. Because of the general lack of aquifer data and process knowledge to formulate and populate a fully distributed groundwater model, Gilfedder et al. (2003) developed a simpler method to derive a first estimate of response time and its difference between GFSs through dimensional analysis.

The rate of three different processes determines the overall groundwater response:

1. Vertical filling, proportional to:

\[ t_1 = \frac{1000Sd'}{(R - R_0)} \quad \text{(B.7a)} \]

2. Lateral flow, proportional to:

\[ t_2 = \frac{SL^2}{365KD} \quad \text{(B.7b)} \]

3. Gradient driven lateral flow, proportional to:

\[ t_3 = \frac{SL^2}{365KH} \quad \text{(B.7c)} \]

where:

- $t$ = representative response time (y)
- $S$ = specific yield or storativity (-)
- $d'$ = unsaturated zone thickness (m)
- $R$ = recharge rate under new land use (mm y$^{-1}$)
\( R_0 \) = recharge rate under previous land use (mm y\(^{-1}\))

\( L \) = flow length of the groundwater system (m)

\( K \) = hydraulic conductivity (m d\(^{-1}\))

\( D \) = aquifer thickness (m)

\( H \) = change in hydraulic head over the flow length (m)

The factors 1000 (mm to m) and 365 (day to year) are required to keep the equations dimensionally correct. The minimum of these three response times will dominate the overall response.

In the present application, estimates of specific yield \( S \), hydraulic conductivity \( K \) and aquifer depth \( D \) were available from expert consultation (Section 2.2; Table 1), whereas \( R \) and \( R_0 \) followed from equations (B.1) and (B.3) for the two respective scenarios. For each of these parameters, simple HLU mean values were calculated. This averaged any spatial variability within the HLUs that was predominantly associated with different GFSs and tree cover. The depth of the unsaturated zone \( d' \) was estimated at 15 m, based on hydrogeological cross-sections in three small subcatchments in the Southwest Goulburn (Gardiner, Hamilton and Dry Creeks) presented in Cheng et al. (2003, p. 141-142).

The geometric parameters flow length \( L \) and head difference \( H \) were estimated using a DEM analysis method described in Evans et al. (2004). This method first uses terrain analysis methods to distil small subcatchments from a digital elevation model (DEM). The average size of the derived subcatchments or hydrological landscape units (HLUs), can be changed to arrive at the most appropriate delineation. This choice is arbitrary; following Evans et al. (2004) we used 1000 ha as an average size. HLUs smaller than a threshold size (400 ha in this case) were merged with an adjoining HLU. For each HLU, the characteristic flow length \( L \) was estimated as half the length of the minor axis of an ellipse that best fitted the HLU shape. This gives a first estimate of project hillslope length, as the drainage line typically follows the major axis. A rough estimate of the hydraulic head difference \( H \) was derived from the minimum elevation \( h_{\text{min}} \) (m AHD) within the HLU (typically the outflow point) and the median elevation \( h_{50} \) (m AHD):

\[
H = 2(h_{50} - h_{\text{min}})
\]  
(B.8)

Clearly this is no more than a first pass estimate, but it serves to distinguish HLUs that include mainly flat morphology from those with steeper hillsides. The median elevation was used instead of maximum to suppress the effect that a few higher areas within the HLU would otherwise have.

For each HLU, the three response times were calculated with equation (B.7a-c). The time for the GFS to show half of the total response was subsequently estimated as the harmonic sum of these three values:

\[
t_{50} = \left( \frac{1}{t_1} + \frac{1}{t_2} + \frac{1}{t_3} \right)^{-1}
\]  
(B.9)

The approach outlined above, and equation (B.7a) in particular, assumes that recharge is changed instantly after land use change. This is of course not the case, as a newly established forest will take some time to grow and increase water use (in fact, water use shortly after afforestation may well be lower than before; Van Dijk et al., 2004b). This effect was also seen in the Pine Creek experiment, where it took about 4-5 years for water yields to decrease to values typical for forested catchments (Section 3.5, Figure 11a, Figure 14). The BC2C model did not simulate this transition. To account for it, we added 4 years to values of \( t_{50} \) calculated with equation (B.9).

A simple S-shaped logistic curve was used to model the transient change in salt yields (Gilfedder et al., 2003; Smitt et al., 2003; Evans et al., 2004):

\[
Y(t) = Y_0 + (Y - Y_0) \left(1 + \exp\left(4 \frac{t_{50} - t}{t_{50}}\right)\right)^{-1}
\]  
(B.10)
where \( Y(t) \), \( Y \) and \( Y_0 \) (all in t km\(^{-2}\) y\(^{-1}\)) are total salt yields for year \( t \) after land use change, and equilibrium conditions after and before land use change, respectively.

The procedure outlined above will only produce a very rough first-pass estimate of groundwater response, and has considerable uncertainty. This uncertainty is commensurate with the lack of knowledge of how groundwater systems really respond to land use change. The Pine Creek experiment is one of the few real-life experiments that give us some insight into these processes, and therefore provides a reality check on the predicted response (Section 4.5).

**B.5. Stream salinity impact**

The calculated change in salt and water yield can be used to assess the effect on mean flow-weighed stream salinity for the Goulburn-Broken catchment, as well as in the Murray River at Morgan (the usual point of reference for the Murray River).

In principle, modelled changes in water and salt yield could be compared to those measured at the Goulburn-Broken catchment outlets. However, this procedure would not account for the fact that a significant fraction of water is lost before reaching the catchment outlets. Jolly *et al.* (1997) estimated total mean salt and water yields over the period 1985-1994 within the Goulburn-Broken at 0.308·10\(^6\) t y\(^{-1}\) and 4.64·10\(^6\) ML y\(^{-1}\), respectively (total Goulburn-Broken catchment area is 24.530 km\(^2\)). Of this latter amount, only 2.01·10\(^6\) ML y\(^{-1}\) reached the catchment outlet (~10% of which via the Broken River); the remaining 57% was diverted. Using the net catchment salt and water yields, the flow-weighted average salinity for 1985-1994 can be calculated to have been 239 µS cm\(^{-1}\).

For each grid cell, the modelled change in grid cell water yield \( dQ \) (mm y\(^{-1}\)=ML km\(^{-2}\) y\(^{-1}\)) and salt yield \( dY \) (t km\(^{-2}\) y\(^{-1}\)), were compared with the catchment totals mentioned above. The relative change in Goulburn River salinity, \( \frac{dC}{C}_{GB} \) (as a fraction change per km\(^2\) of land with vegetation change) is subsequently given by:

\[
\left( \frac{dC}{C} \right)_{GB} = \frac{Y_{GB} + dY}{Y_{GB}} \frac{Q_{GB}}{Q_{GB} + dQ} - 1 = \left( 1 + \frac{dY}{0.308 \cdot 10^6} \right) / \left( 1 + \frac{dQ}{4.64 \cdot 10^6} \right) - 1 \tag{B.11}
\]

This procedure assumes that no salt is lost between the subcatchment and the Goulburn-Broken catchment outlets, and that diversions and internal losses are equal for all stream flow, regardless of where it is generated.

The change in Murray River salinity at Morgan can be estimated by taking into account that water from the Goulburn-Broken presently accounts for ~29.0 µS cm\(^{-1}\) of total Murray salinity (MDBC, unpublished; see also Dowling *et al.* 2004).

Results for the Pine Creek plantation (320 ha, ~232 mm y\(^{-1}\) water yield decrease, ~30 t km\(^{-2}\) y\(^{-1}\) salt yield decrease) can be used to illustrate the order of magnitude of the changes that can be expected. Equation (B.11) suggests that establishment of this forest has reduced Goulburn river salinity by 0.015% or ~0.036 µS cm\(^{-1}\) and reduced total Goulburn-Broken water yield by 0.02%.

It can be estimated that in order to decrease Murray River EC by 0.1 µS cm\(^{-1}\), a change of Goulburn-Broken salinity of ~0.34% or 0.82 µS cm\(^{-1}\) will be needed. This is corresponds to about 25 times the Pine Creek plantation, or about 8,000 ha. This would reduce total Goulburn-Broken water yield by about 0.4%.

The calculations above should be considered as approximate numbers only. Some simplifying assumptions need to be made to arrive at the above estimates, arguably the most important one being that none of the salt in diverted irrigation water is returned to the river. If, conversely, all salt is returned then afforestation would be more than two times as effective in reducing river EC. It is also noted that proper targeting of valleys and hillslope sections within valleys is likely to increase the effectiveness of afforestation.
Appendix C. Spatial input data and results

C.1. Spatial input data

Mean annual rainfall $P$ (mm y$^{-1}$)

Mean annual potential evapotranspiration $E_{PET}$ (mm y$^{-1}$)

Woody vegetation cover $F_{tree}$
Spatial input data (ctd.)

Rainfall salt concentration $C_P$ (mg l$^{-1}$)

Groundwater flow systems

Digital Elevation Model (artificially shaded)
C.2. Intermediate spatial output

Groundwater salinity $C_G$ ($\mu$S cm$^{-1}$)

Hydrological Landscape Units (HLUs) and estimated response half time $t_{50}$ (years)
Spatial output: intermediate results (ctd.)

Present water yield $Q_0$ (mm $y^{-1}$)

Present salt yield $Y_0$ (t $km^{-2} y^{-1}$)
C.3. Spatial output: changes in water and salt yields and stream salinity

Changes in water and salt yields and stream salinity 5 years after afforestation

Water yield change (mm y\(^{-1}\)) 5 years after afforestation

Salt yield change (t km\(^{-2}\) y\(^{-1}\)) 5 years after afforestation

Fraction change in Goulburn-Broken salinity (in millionths of present EC per km\(^{2}\) of afforestation) 5 years after afforestation
Changes in water and salt yields and stream salinity 15 years after afforestation

- Water yield change (mm $y^{-1}$) 15 years after afforestation
- Salt yield change (t km$^2$ $y^{-1}$) 15 years after afforestation
- Fraction change in Goulburn-Broken salinity (in millionths of present EC per km$^2$ of afforestation) 15 years after afforestation
Changes in water and salt yields and stream salinity 30 years after afforestation

Water yield change (mm y\(^{-1}\)) 30 years after afforestation

Salt yield change (t km\(^{-2}\) y\(^{-1}\)) 30 years after afforestation

Fraction change in Goulburn-Broken salinity (in millionths of present EC per km\(^2\) of afforestation) 30 years after afforestation