Lower Limestone Coast forest water accounting groundwater model

DEWNR Technical report 2017/14

Government of South Australia
Department of Environment, Water and Natural Resources

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Foreword

The Department of Environment, Water and Natural Resources (DEWNR) is responsible for the management of the State's natural resources, ranging from policy leadership to on-ground delivery in consultation with government, industry and communities.

High-quality science and effective monitoring provides the foundation for the successful management of our environment and natural resources. This is achieved through undertaking appropriate research, investigations, assessments, monitoring and evaluation.

DEWNR's strong partnerships with educational and research institutions, industries, government agencies, Natural Resources Management Boards and the community ensures that there is continual capacity building across the sector, and that the best skills and expertise are used to inform decision making.

Sandy Pitcher
CHIEF EXECUTIVE
DEPARTMENT OF ENVIRONMENT, WATER AND NATURAL RESOURCES

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Summary

Management of groundwater in the Lower Limestone Coast Prescribed Wells Area is administered through the Lower Limestone Coast Water Allocation Plan (LLCWAP). The LLCWAP accounts for and sets limits on groundwater development to promote sustainability of groundwater resources and the environmental and economic services they support. Allocation limits are typically based on estimates of annual groundwater recharge.

Both irrigators and plantation forests are licensed water users in the LLCWAP, with rates of plantation recharge interception and groundwater use based on CSIRO studies (Dillon et al., 2001; Benyon and Doody, 2004). In developing the LLCWAP, a groundwater model was used to test scenarios related to future impact of the plantation estate on groundwater, particularly in the Coles and Short management areas (Aquaterra, 2010a). The water accounting models used in the LLCWAP and tested in the model assume that hardwood plantations are managed on a 10 year growth cycle. However much of the hardwood plantation estate in this area has been in place for more than 10 years. Furthermore, the Coles and Short management areas have been identified as over-allocated in the LLCWAP, with groundwater declines of up to 5 m observed over the past 15 years. Proposed reductions in allocation in Coles and Short may involve changes in land use including conversion of plantation forest back to pasture.

In this study, the groundwater flow model developed by Aquaterra (2010a) was revised and recalibrated to assess the forest water accounting models, and run scenarios related to potential changes in land use. **The two main conclusions from this study are:**

- The forest water accounting models and recharge rates used in the Lower Limestone Coast Water Allocation Plan can be considered appropriate for quantifying forest impacts on groundwater, with some exceptions.
- Reduction in the area of hardwood forest in Coles and Short is likely to lead to recovery in groundwater levels and less drawdown impacts on Bool Lagoon. However the level of recovery will be spatially variable, and dependent upon the extent of land use change and potential changes in groundwater recharge. Any potential reduction in recharge (e.g. as a result of climate change) would impact the amount of groundwater recovery. The potential impact of increased recharge in fallow years (Benyon and Doody, 2009) was also investigated, however no significant impact on modelled groundwater levels was observed.

This report also identifies a number of areas in which further work could be conducted to improve confidence in the model results, and help refine groundwater management in the area. Recommendations are detailed in Section 7.2. In summary they relate to the following areas:

- Define resource condition limits (RCLs) for groundwater resources in the area, particularly in relation to high value water dependent ecosystems such as Bool Lagoon.
- Extend the eastern boundary of the model into western Victoria.
- Conduct further parameter uncertainty analysis on the model as groundwater management is reviewed, particularly in relation to recharge, hydraulic conductivity and evapotranspiration (ET).
- Improved understanding and simulation of groundwater processes around drains.
- Constraining groundwater model ET, both from shallow watertables in areas of pasture, and ET from plantation forests using groundwater, possibly through the use of satellite data.
- Revisit model scenarios as more information on land use change in the study area becomes available.
- Assess the influence of potential changes in rainfall seasonality under future climates, and possible impacts of vegetation feedbacks from increased emissions.

1 Introduction

1.1 Background

In the Lower Limestone Coast region of South Australia, groundwater resources are managed in accordance with the Lower Limestone Coast Water Allocation Plan (LLCWAP, SENRMB 2013). This plan accounts for and sets limits on groundwater development to promote sustainability of groundwater resources and the environmental and economic services they support. Allocation limits are typically based on estimates of annual groundwater recharge.

Plantation forests are included in the LLCWAP as licensed groundwater users, based on many studies in the region which have demonstrated that groundwater recharge is reduced under forests (Holmes and Colville, 1970; Allison and Hughes, 1972; Dillon et al., 2001; Mustafa et al., 2006), and that plantation trees may directly extract groundwater where the watertable is shallow (less than 6 m, Benyon and Doody, 2004). Incorporating plantation forests into groundwater management has involved the development of forest water accounting models (Harvey, 2009) and assessing the impact of different forest management scenarios on groundwater resources in the region using a numerical groundwater model (Aquaterra 2010a,b). The water accounting models and scenario testing included assumptions regarding the growth and harvest cycling of the main plantation forest species, *Pinus radiata* (softwood) and *Eucalyptus globulus* (hardwood).

While softwood plantation forestry is considered a 'mature' industry in the region, with a long history and harvest cycles that closely match those used in the forest water accounting models, the same cannot be said for hardwood plantation forestry. Hardwood plantations expanded rapidly in the late 1990s and early 2000s, and have remained in place longer than originally anticipated or accounted for. In a review of the forest water accounting models, Harvey (2017) found that while the forest water accounting models were broadly accurate, there has been a doubling of the period of direct groundwater extraction under many areas of hardwood plantation. Harvey (2017) also found that in some cases, plantations are likely to be extracting groundwater from greater depths than those used in the accounting models (more than 6 m below ground level).

The LLCWAP also identifies the management areas of Coles and Short (Figure 2.1) as over-allocated based on the amount of plantation forestry present. Reductions in groundwater allocations are proposed in the LLCWAP, and this may result in changes in land use from plantation forest back to pasture. The impact of this land use change on groundwater resources is currently unknown.

Given the discrepancies between hardwood forest water accounting models and observed land use practices, there is a need to revisit the numerical groundwater model to assess the impact of the current forest management practices on groundwater resources. Furthermore there is a need to assess how these plantations will be managed in the future in respect to rotation length and reafforestation method, and what impact different management regimes may have on groundwater resources.

1.2 Aims and objectives

The aim of this study is to use a numerical groundwater flow model to assess the impact of different plantation forest management scenarios on groundwater resources in the LLC. Specifically the project objectives are to:

- Update and re-calibrate the WR2010 groundwater model developed by Aquaterra (2010a), to help assess the forest water accounting models.
- Use the updated model to run different prediction scenarios for plantation forest and groundwater management.

2 Conceptual model

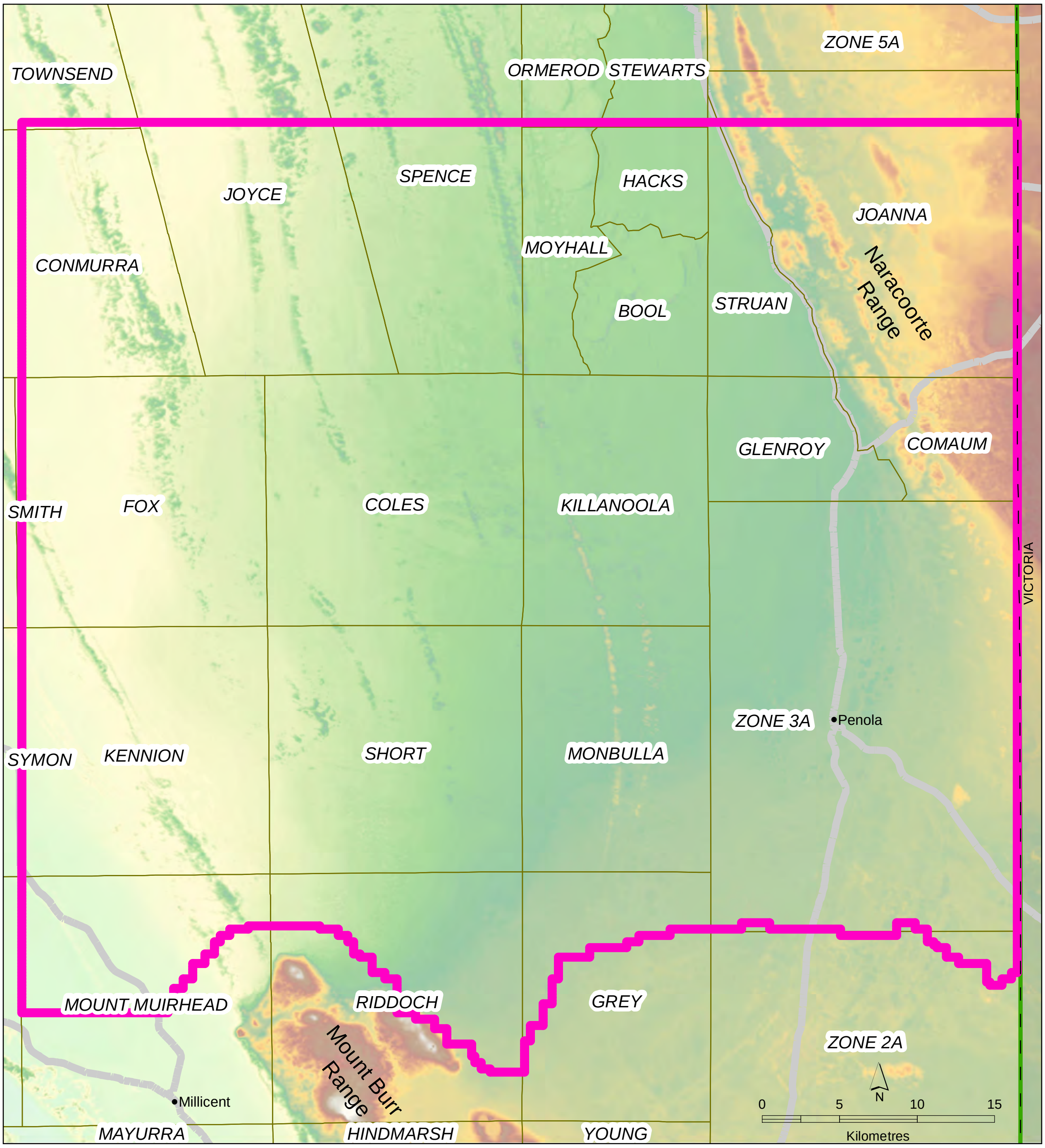
2.1 Climate and topography and administrative boundaries

For the purposes of this project, the study area is based on the model domain boundary used by Aquaterra (2010a, b, see Figure 2.1). Surface elevation varies from the Naracoorte Range in the east (~115 m AHD) to the low lying flats in the west (~12 m AHD). With the exception of the break in slope at the Naracoorte Range, the topography is relatively flat, with gentle undulations created by remnant dune ridges. The south-eastern boundary of the study area is met by the Mount Burr Range, a former volcanic range dating from at least 20,000 years ago, which is overlain by Quaternary sediments (Sheard, 1983).

The study area is situated in the Lower Limestone Coast Prescribed Wells Area (LLCPWA), in which groundwater is managed under the LLCWAP. The LLCPWA is further subdivided into 61 management areas. Groundwater is allocated on a management area scale, and allocation volumes are based on estimates of groundwater recharge within each management area (Brown et al., 2006).

The climate, like much of the lower South East of South Australia, is typified by hot, dry summers and cool, wet winters. Mean annual rainfall measured at Penola Post Office (station 26025) is 657 mm/y (BoM, 2017). Rainfall trends have been declining in the region since the mid-1990s, and this is reflected in the cumulative deviation in mean annual rainfall measured at Penola (Figure 2.2). Estimates of potential evapotranspiration (ET) for the region range from ~ 950 mm/y south of the study area to 1400 mm/y in the north (Natural Resources South East, 2017).

Figure 2.1 Surface topography and groundwater management areas in the study area



Study area

Lower Limestone Coast Prescribed Wells Area

Groundwater management areas

SA - Victoria Border

Primary Road

Digital elevation model

Elevation (m AHD)

High : 200

Low : 0

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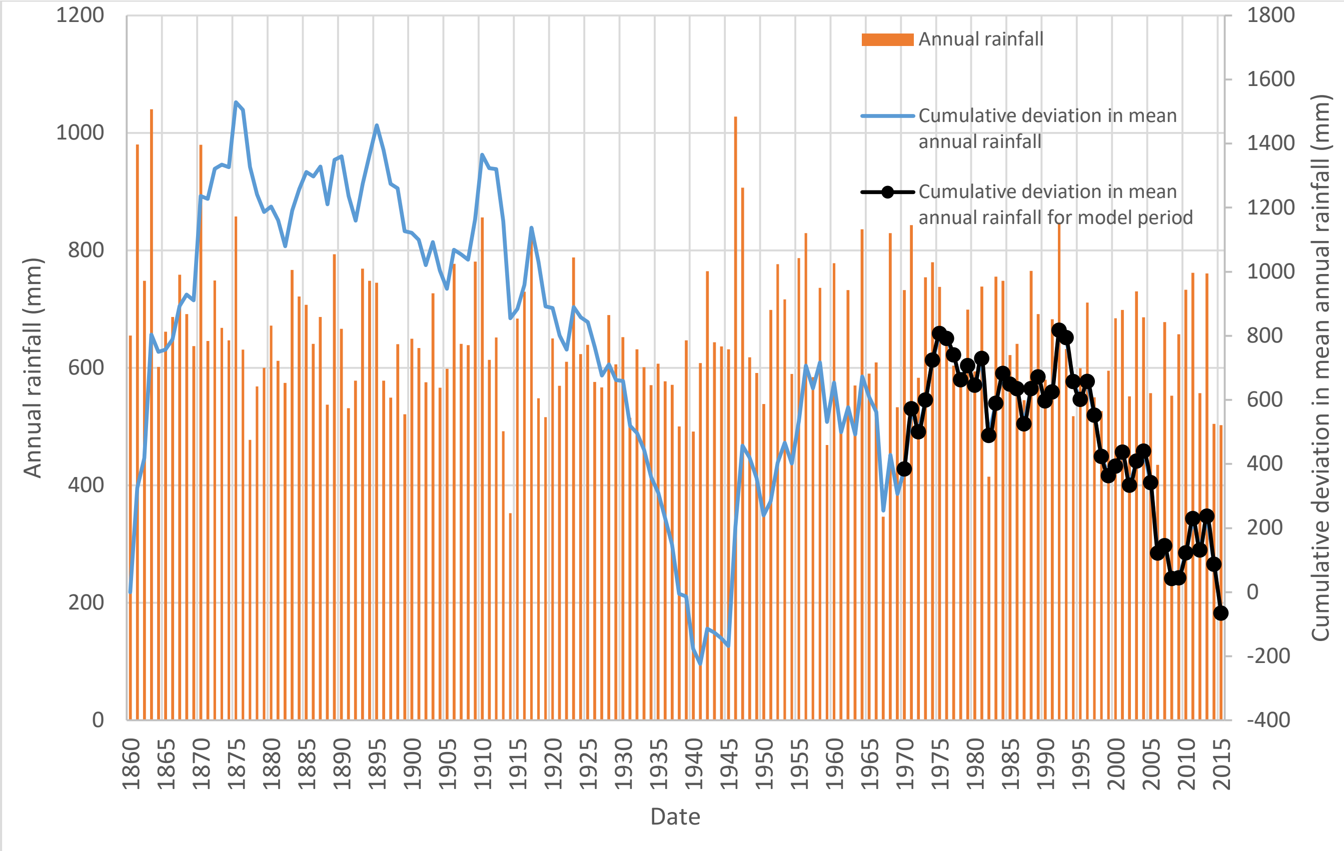


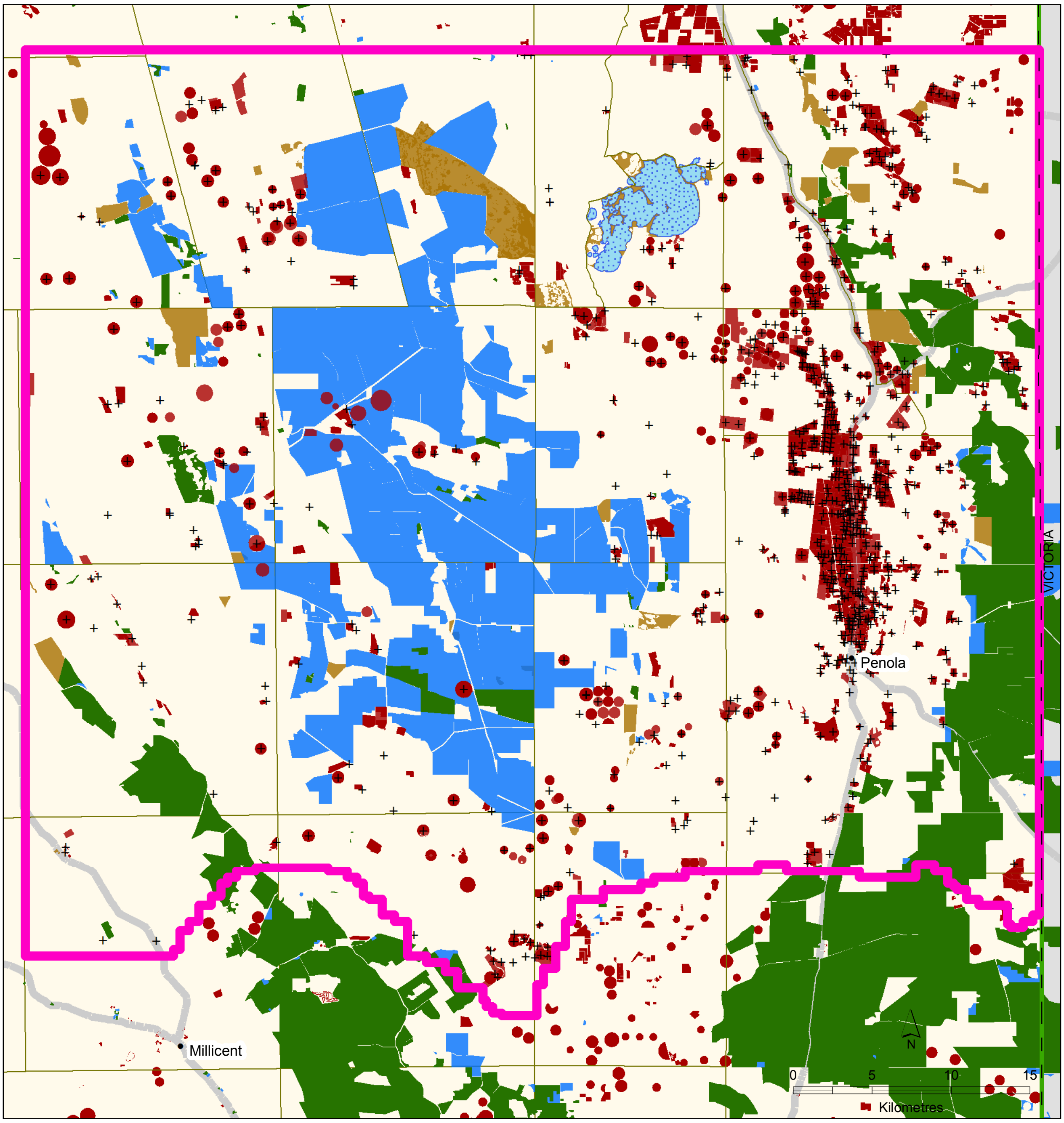
Figure 2.2. Cumulative deviation in mean annual rainfall measured at Penola (station 26025)

2.2 Land use

The majority of the land in the study area is used for dryland (non-irrigated) agriculture, predominantly livestock grazing on improved pastures. There is an area of concentrated irrigation north of Penola, where drip irrigated vineyards are the main land use. Irrigation of grain and pasture is dispersed throughout the rest of the study area (Figure 2.3). In all parts of the study area, water for irrigation is drawn exclusively from groundwater.

Plantation forestry is a significant land use in the region, occupying ~ 585 km^2, or 17% of the study area. The forest estate consists of softwood (*Pinus radiata*) and hardwood (*Eucalyptus globulus*) plantations. Most of the softwood plantations are located along the dune range extending north-west from the Mount Burr Range, and east of Penola. Softwood plantation forestry is considered a 'mature' industry in the region, with a long history and well established management techniques (Harvey, 2017). Hardwood plantations are centred in the management areas of Coles and Short (Figure 2.3). Hardwood plantations were first established the late 1990s, and expanded rapidly in the early–mid 2000s (Harvey 2017). For example, ~160 km^2 of hardwood forests were established in the study area in 2000 alone. While the hardwood plantations were originally intended to be managed on 10 year growth cycles (Harvey, 2009), the majority of the hardwood plantations have been in place for more than 10 years (up to 19 years, (Harvey, 2017)).

Figure 2.3 Land use in the study area (areas not highlighted are dryland agriculture)



Study area
Hardwood plantation
Irrigation
Softwood plantation
Native vegetation
Groundwater management areas
Bool Lagoon
Groundwater extraction wells
Lower Limestone Coast Prescribed Wells Area
Primary Road
SA - Victoria Border

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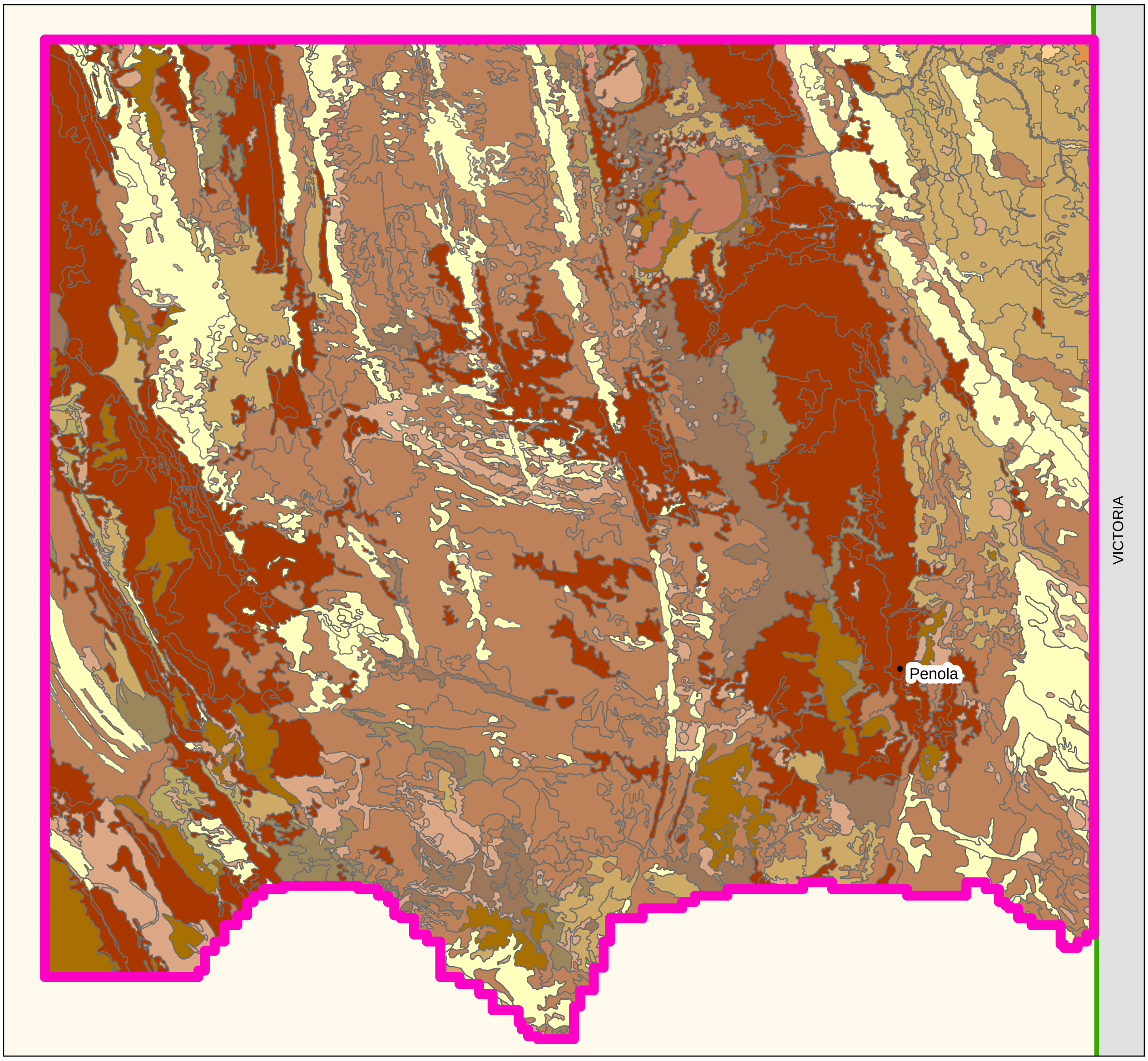
2.3 Soil and geology

Soil type varies across the study area, but a few dominant categories can be described based on the classifications of Hall et al. (2009). The eastern part of the study area and Naracoorte Range is dominated by deep loam and deep sand profiles. Deep sand profiles are also associated with the remnant dune ridges that run through the study area. On the flats, shallow soils overlying calcrete and limestone are more common, a famous example being the shallow, red-brown clay *Terra Rossa* soils in the region of irrigated vineyards north of Penola. In the central part of the study area where hardwood plantations are concentrated, shallow, sandy soils overlying clay dominate the landscape (Figure 2.4).

Surface geology is dominated by the Quaternary Padthaway and Bridgewater Formations (Figure 2.5). The Bridgewater Formation is a calcareous sand and sandstone, which is usually elevated above the surrounding landscape, owing to its origins as dune deposits formed during Pleistocene sea level transgressions and regressions (Belperio, 1995). The Padthaway Formation, a limestone with silt and marl interbeds, lies between the dune ridges of the Bridgewater Formation (Figure 2.5). Both of these formations are underlain by the Tertiary Gambier Limestone (Figure 2.6), which outcrops in the Naracoorte Ranges. The Gambier Limestone increases in thickness towards the west (up to 320 m thick in the study area, Figure 2.6), and consists of interbedded layers of fossiliferous limestone with marls (Alley and Lindsay, 1995).

The Gambier Limestone is underlain by the Tertiary Narrawaturk Marl, a calcareous mudstone. This transitions into the Mepunga Formation, a thin (9–12 m thick) layer of sparsely fossiliferous quartz grit. These formations overlay the Tertiary Dilwyn Formation, a sequence of gravelly sands and sandstone interbedded with muds and siltstones. Carbonaceous clays at the top of the Dilwyn Formation typically mark its upper boundary, and act as a confining layer for groundwater in the formation. The cross-section in Figure 2.6 does not map the full depth of the Dilwyn Formation, as drillholes typically only extend beyond these depths for mineral prospecting. Furthermore, the focus of this study is the groundwater resources in the upper Quaternary and Tertiary Limestone Formations.

Figure 2.4 Soil type descriptions in the study area (adapted from the descriptions of Hall et al., 2009)



Soil type

- Gradational/calcereous soils
- Shallow soils on calcrete or limestone
- Sand overlying clay soils
- Deep sand soils
- Deep loamy soils
- Cracking clay soils
- Wet soils
- Deep uniform to gradational soils

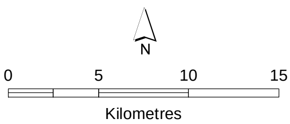


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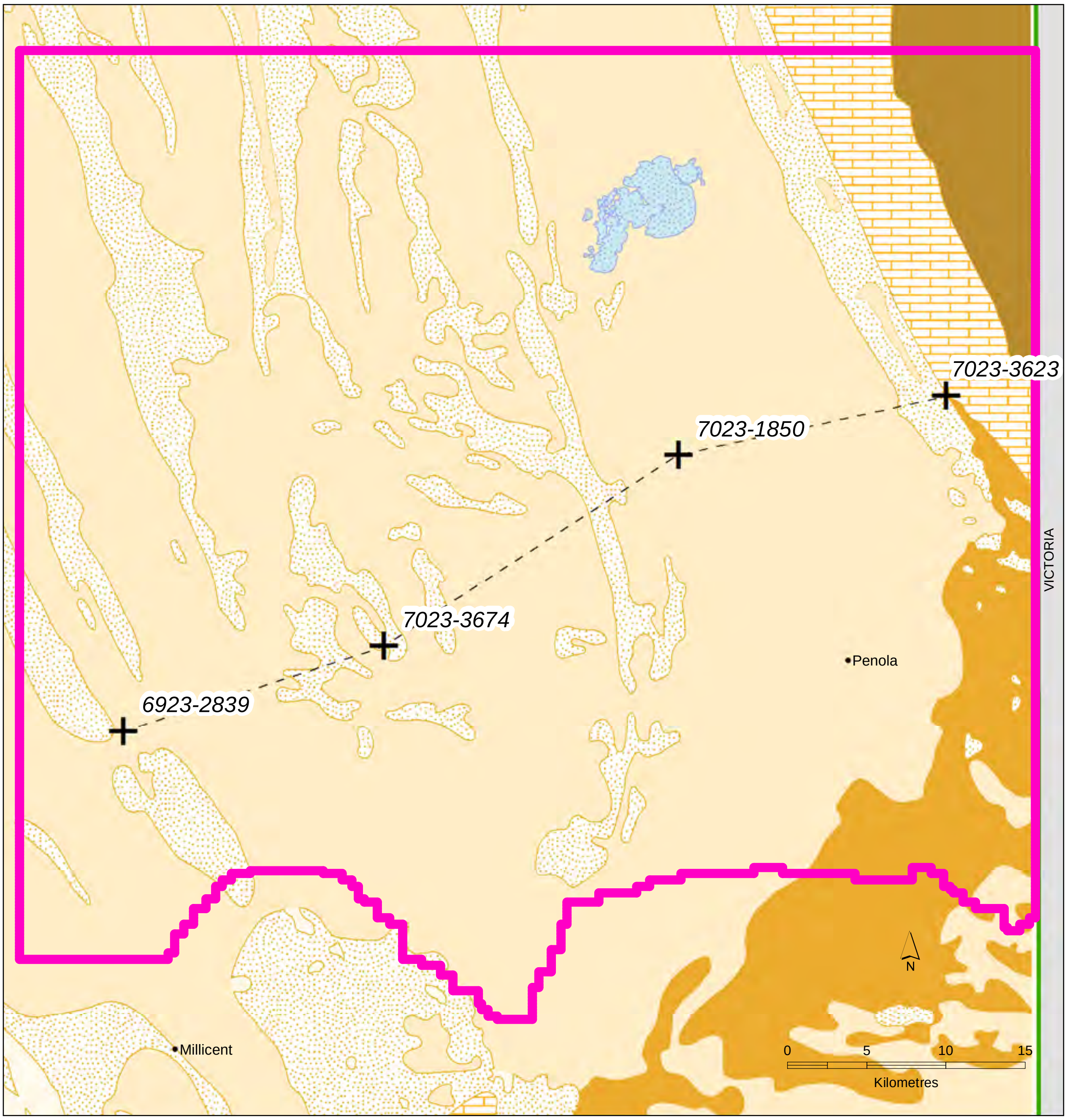
Figure 2.5 Geology in the area (see Figure 2.6 for cross-section)



Study area
Lacustrine/playa sediments
Stratigraphy wells
Geology
Padthaway Formation
Bridgewater Formation
Gambier Limestone
Parilla Sand
Molineaux Sand

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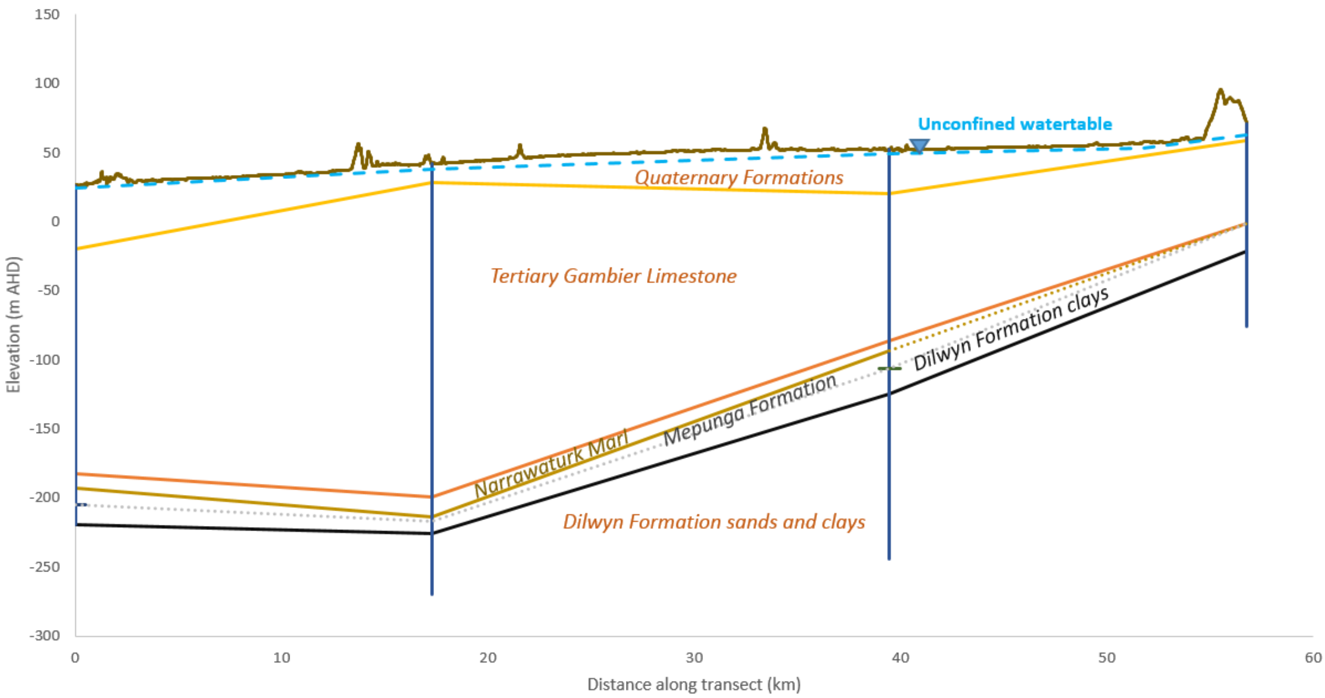


Figure 2.6. Cross-section through wells shown in Figure 2.5

2.4 Hydrogeology

2.4.1 Aquifers

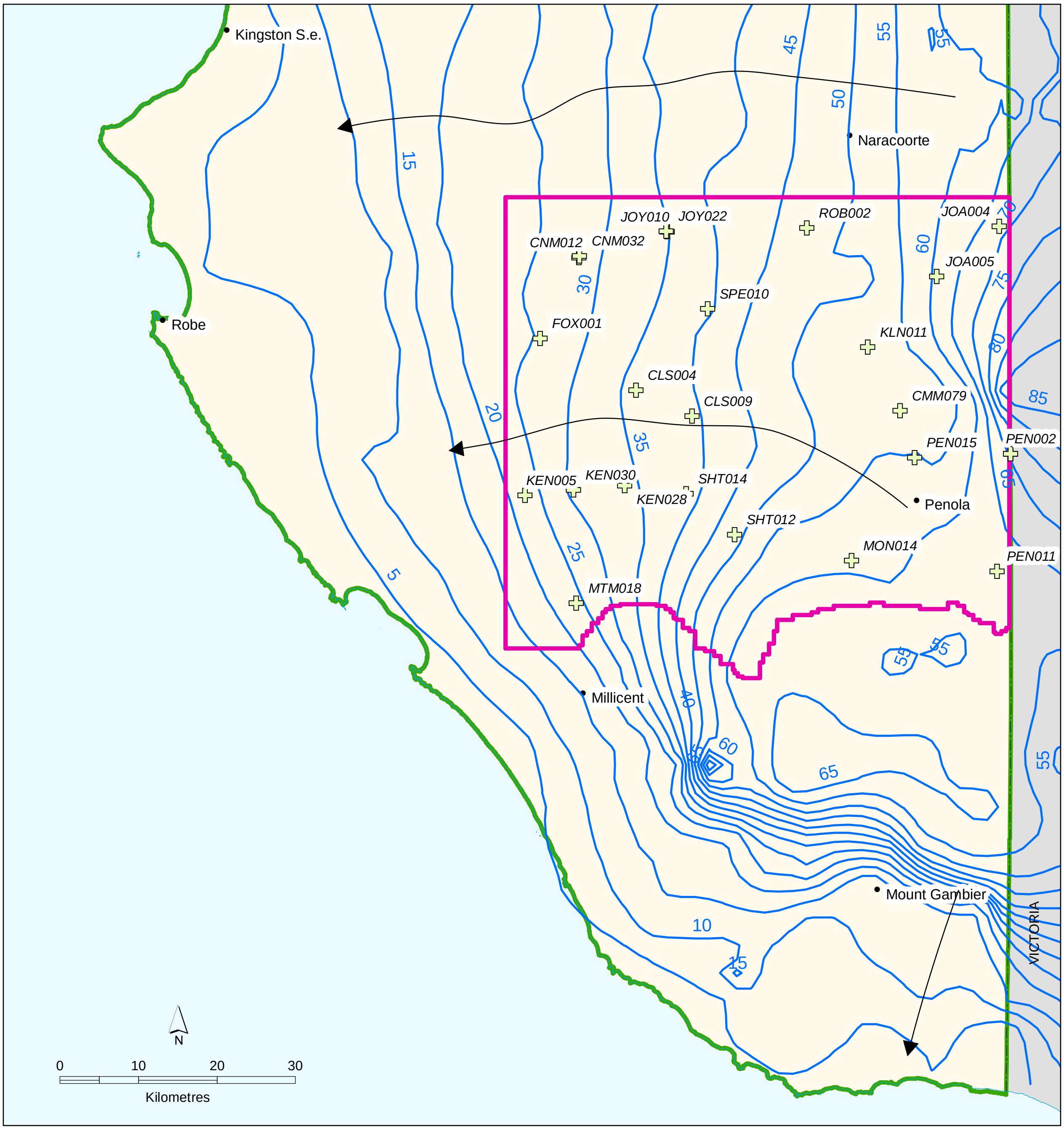
The Quaternary Padthaway and Bridgewater Formations and underlying Gambier Limestone form one continuous, regional unconfined aquifer, which for management purposes is referred to as the Tertiary Limestone Aquifer (TLA, SENRMB 2013). Underlying the TLA, the carbonaceous clays at the top of the Dilwyn Formation form a regional aquitard, which confines the underlying aquifer in the Dilwyn Formation, typically referred to as the Tertiary Confined Sand Aquifer (TCSA). Groundwater pressure is higher in the TLA than the TCSA throughout most of the study area, however west of the Joyce, Fox, Kennion and Mount Muirhead management areas (Figure 2.1), pressure in the TCSA is higher, and becomes artesian further west of the study area near Kingston.

South of the study area there is evidence of downward leakage from the TLA recharging the TCSA through fractures and faults (Brown et al., 2001). However the level of interaction between the two aquifers in the study area is unknown. Furthermore, the TCSA is not thought to have any influence on groundwater level trends in the TLA in the study area, particularly under plantation forest. Therefore the TCSA has not been included in previous models of the TLA in this area, and will not be considered further in this study.

2.4.2 Groundwater flow and trends

Groundwater in the TLA flows from the east/south-east towards the west (Figure 2.7). Figure 2.7 is based on South Australian groundwater level data, however the aquifers do extend into western Victoria, with groundwater flowing from across the border into South Australia. Depth to water ranges from 18 m below ground level (bgl) in the east to less than 0.5 m bgl in the west. The regional groundwater mound that extends from the south east of the study area to the north of Mount Gambier has previously been attributed to thinning of TLA units in the uplifted areas south of the model domain (Holmes and Waterhouse, 1983).

Figure 2.7 Potentiometric surface and flow direction in the unconfined Tertiary Limestone aquifer



Observation wells

Study area

Potentiometric surface (spring 2016)

Groundwater flow direction

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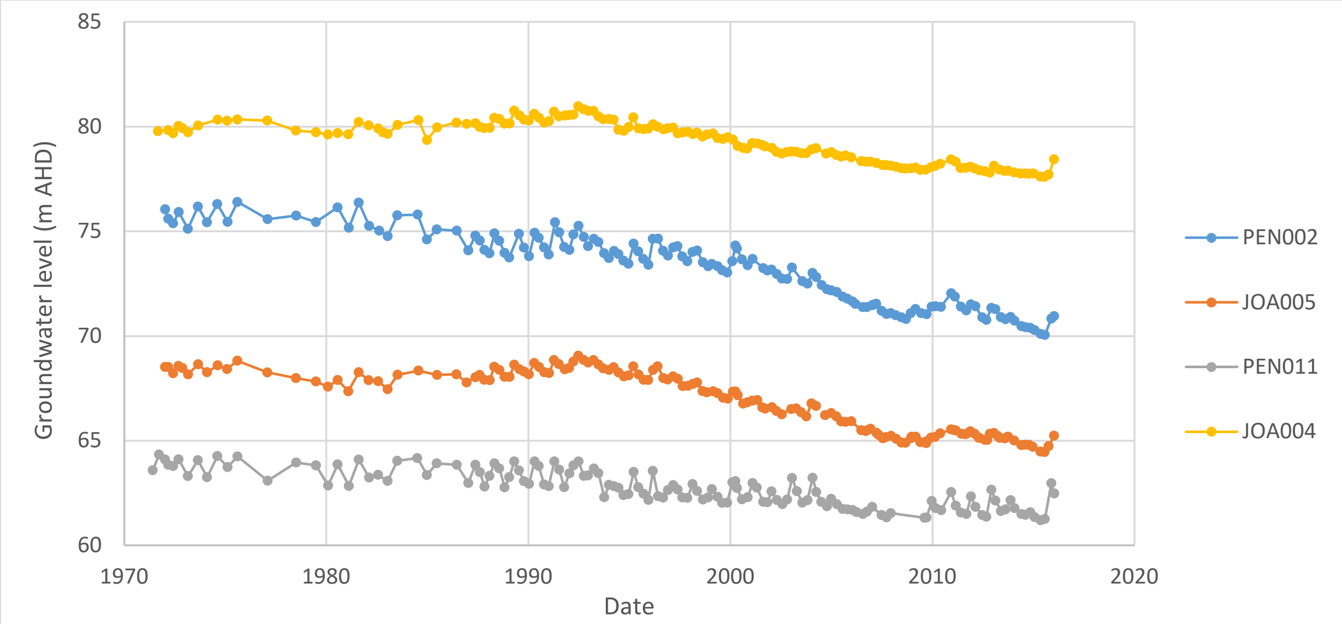


Figure 2.8. Groundwater levels along the eastern boundary of the study area

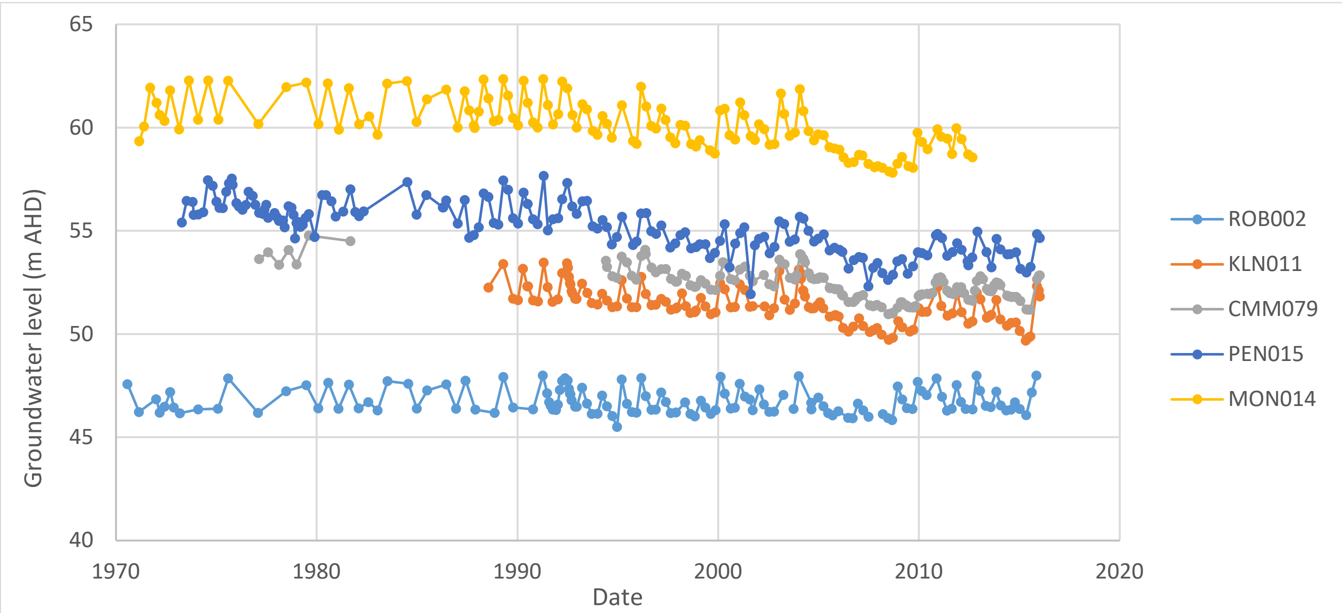


Figure 2.9. Groundwater levels in the eastern part of the study area (dryland farming and vineyards)

Groundwater level trends vary across the region. Along the eastern boundary of the study area in the Naracoorte Range, trends were stable until the 1990s, and have been declining since then (Figure 2.8). Within this trend, seasonal fluctuations can be observed, most likely driven by rainfall recharge in winter–spring (groundwater levels peak), and discharge via groundwater extraction in summer–autumn. The overall declining trend since the mid-1990s matches rainfall trends in the area (Figure 2.2).

Groundwater level trends show strong seasonal fluctuations along the flats west of the Naracoorte Range (Figure 2.9). While groundwater levels have remained stable in some areas (ROB002), in other areas declines were observed in the mid-2000s, most likely due to low rainfall (Figure 2.2) and increases in irrigation extraction north of Penola (discussed later).

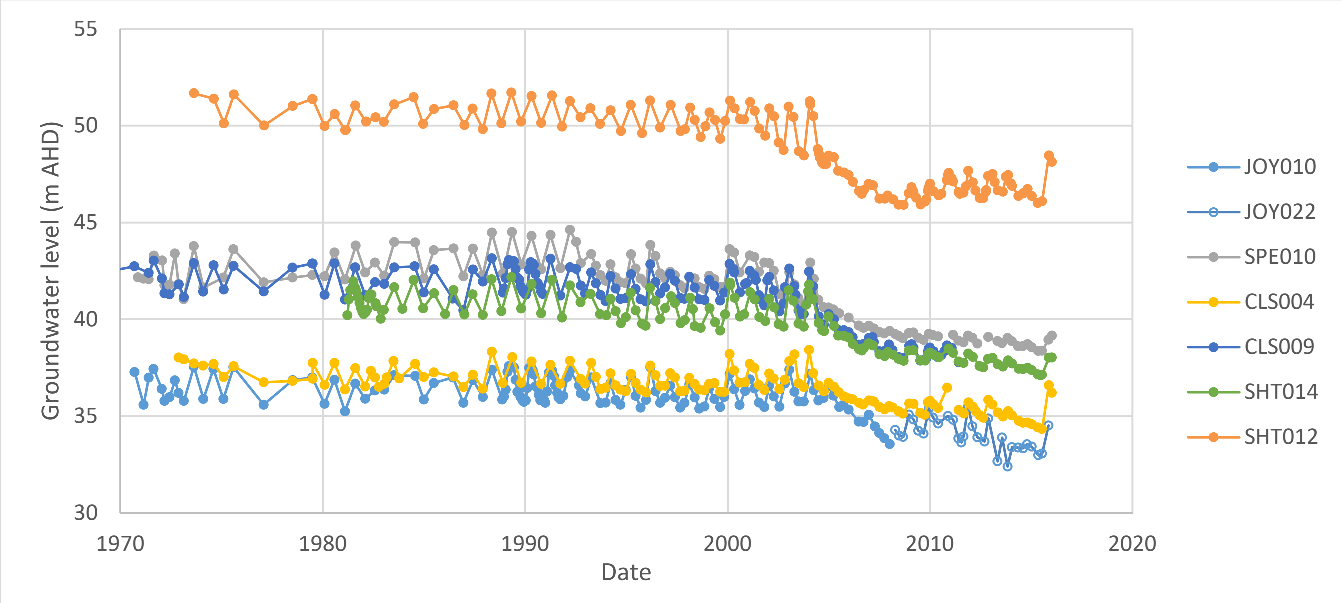


Figure 2.10. Groundwater levels in the central part of the study area (hardwood plantation from 2000 onwards)

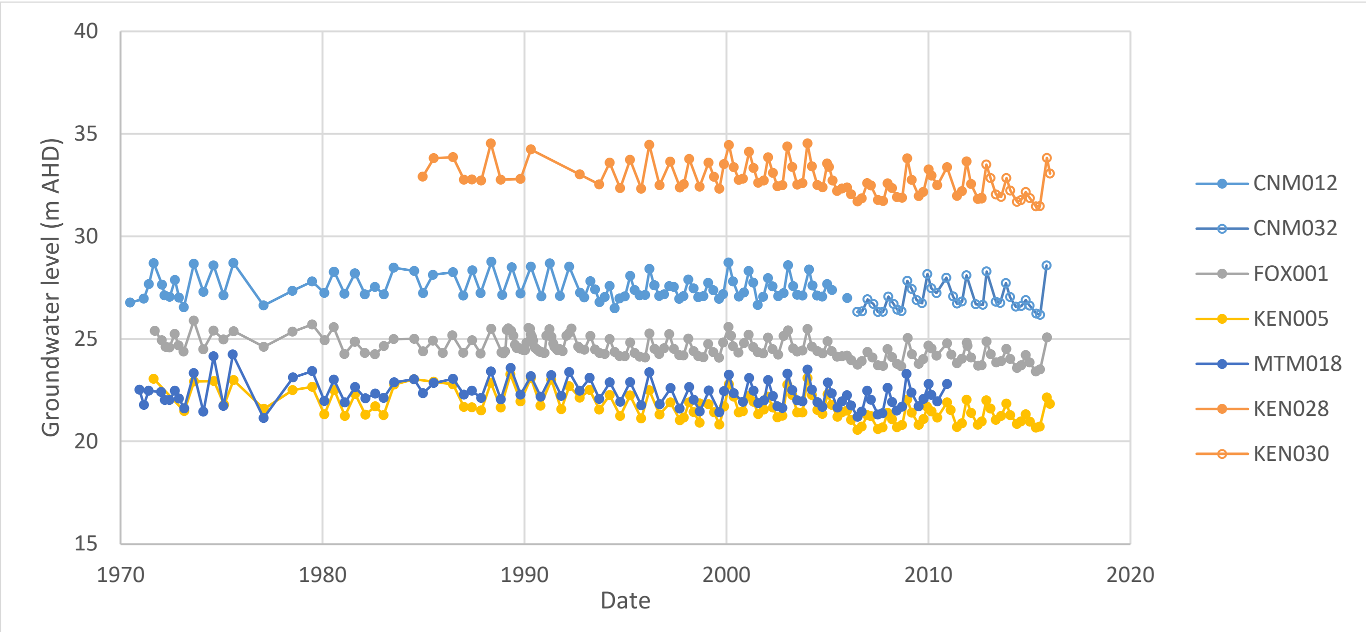


Figure 2.11. Groundwater levels in the western part of the study area (dryland farming and grazing, shallow watertable)

In the central part of the study area, groundwater levels were stable, with seasonal fluctuations up to 2 m, until the mid-2000s. From the mid-2000s onwards, fluctuations became muted and groundwater levels started to decline (Figure 2.10). While this corresponds with declines in rainfall in the mid-2000s, the trends are largely driven by the widespread expansion of hardwood plantation forest since the early 2000s (rainfall has been declining since the mid-1990s).

In the western part of the study area, groundwater level trends have been stable, with seasonal fluctuations of ~2 m each year (Figure 2.11). These fluctuations are driven by rainfall recharge in winter–spring (groundwater levels peak) and evapotranspiration from the shallow watertable (often less than 2 m below ground level) in summer–autumn. Worth noting is that these wells, in areas where plantation forest is absent, show only slightly reduced seasonal fluctuations during the mid-2000s drought, compared to the wells in adjacent management areas which are influenced by plantation forests (Figure 2.10).

2.4.3 Aquifer properties

For parameterizing numerical groundwater models in the area, hydraulic conductivity values (transmissivity divided by aquifer thickness) of 25 to 78 m/d have been used by Aquaterra (2010a). A larger regional model which encompasses the study area used values ranging from 0.2 to 1300 m/d (Morgan et al., 2015), while a model for the Padthaway area north of the study area used 5 to 200 m/d (Wallis, 2008). Reported hydraulic conductivity values based on aquifer test data in the study area range from 1.1 to 242 m/d.

The high hydraulic conductivity (K) values are based on transmissivity (T) values from aquifer tests in the region, where T = Kb (b is aquifer thickness). In a review of aquifer test data south of the study area, Mustafa and Lawson (2002) report transmissivity values ranging from 20 to 25,000 m^2/d, while Stadter and Love (1987) report values north of the study area ranging from 190 to 14,000 m^2/d. The large values are attributed to the development of karst features in the limestone aquifers. Aquifer test values within the study area report transmissivities of 230 to 3140 m^2/d.

2.4.4 Recharge and evapotranspiration

Groundwater recharge occurs throughout the study area via diffuse rainfall infiltration, and is influenced by soil type and land use. However little to no recharge occurs beneath native vegetation and closed canopy plantation forests (discussed further below). Many methods have been used to estimate recharge for multiple purposes across the LLC (Wood, 2011). Brown et al., (2006) gave estimates of recharge based on the watertable fluctuation method for each management area, ranging from 50 mm/y in the Joanna management area where watertables are deep, to 180 mm/y in the Monbulla management area. The recharge rates reported by Brown et al. (2006) are the basis for groundwater allocation in the LLCWAP. In scaling up these estimates to give recharge volumes, it is assumed that no recharge occurs under areas of native vegetation or lakes (Brown et al., 2006). Crosbie and Davies (2013) revisited the watertable fluctuation method for these management areas and reported rates ranging from 13 to 154 mm/y. The lower rates are likely caused by the authors including watertable fluctuation data from low rainfall periods from the mid-2000s.

Crosbie et al (2015) examined spatial patterns in evapotranspiration (ET) across the region using satellite based data for water balances (expressed as recharge = precipitation – satellite based ET). The authors found large areas where the water balance was in deficit in the study area by more than 250 mm/y, attributed to ET from plantation forest areas (discussed further below). However the water balance was also found to be in deficit in shallow watertable areas. Comparisons between recharge estimates from satellite based water balances and field based estimates (e.g. chloride mass balance estimates in Allison and Hughes, 1978; Wohling, 2008; Wood, 2011) showed a systemic bias in the satellite based estimates, in that they needed to be corrected by + 45 mm/y across the study area. It should be noted that the algorithm used to calculate ET from satellite data was calibrated to ET measurements from seven eddy covariance towers around Australia, none of which were located in the South East (Guerschman et al., 2009). Furthermore, the satellite based estimates of ET represent ET from plant reflectance, the unsaturated zone and the soil surface, not just groundwater.

Doble et al., (2015) extended this work to develop estimates of net recharge (recharge minus ET) across the area for input to groundwater flow models. The net recharge estimates are based on unsaturated zone modelling for different land uses, soil types, and climate zones. However the results of this work have not yet been ground truthed and the model did not perform as expected in plantation forest areas, hence further work is required. For groundwater models in the area, depth-dependent ET from shallow watertables outside forest areas, has been simulated to occur at a maximum rate of 500 mm/y with an extinction depth 2–3 m below ground level (Wallis 2008; Aquaterra 2010a). Wallis (2008) report this rate is consistent with CSIRO studies and the Bureau of Meteorology Climate Atlas, and the extinction depth is consistent with the approach presented by Shah et al., (2007).

2.4.5 Plantation forest impacts on groundwater

Many studies in the LLC have found that groundwater recharge is reduced under plantation forest compared to dryland agriculture (Holmes and Colville, 1970; Allison and Hughes, 1972; Dillon et al., 2001; Mustafa et al., 2006) through both canopy interception and tree water use. Additionally, based on tree water use studies in the region, Benyon and Doody (2004) found that plantation forests may extract groundwater where watertables are shallow. The authors found that for eight out of nine study sites where the watertable was within 6 m of the ground surface, plantation forests were using groundwater at a mean annual rate of 435 mm/y.

In translating studies on forest impacts into groundwater management policy, four forest water accounting models have been developed (Harvey, 2009). These four models summarise the influence of plantation forests on groundwater, over the lifecycle of a plantation, in terms of:

1. Reduction in recharge under hardwood plantations
2. Reduction in recharge under softwood plantations
3. Groundwater extraction by hardwood plantations
4. Groundwater extraction by softwood plantation.

Reduction in recharge applies to the deemed recharge rate for a given management area (as described by Brown et al., 2006). The groundwater extraction models apply to any areas where the depth to groundwater was less than 6 m below the ground surface in June 2004, and extraction rates reach a maximum of 364 mm/y when the plantation reaches maturity. The models are based on biophysical principles and the information in Benyon and Doody (2004) and were *"refined by a technical working group convened by the South East Natural Resources Management Board during the stakeholder consultation process of September–November 2006"* (Harvey, 2009). Full details of the water accounting models can be found in Harvey (2009), but for illustrative purposes, for a hardwood plantation in management area where the groundwater recharge rate is 100 mm/y (e.g. the Fox management area), water accounting models one and three would apply over the 10 year life of the plantation, and the vertical groundwater flux (recharge to and ET from the watertable) would follow Figure 2.12. For groundwater management purposes, these forest water accounting models are further simplified by averaging extraction over the 11 year plantation cycle (10 years of growth plus one year of post-harvest clean up), such that the 'annualised' extraction is 182 mm/y.

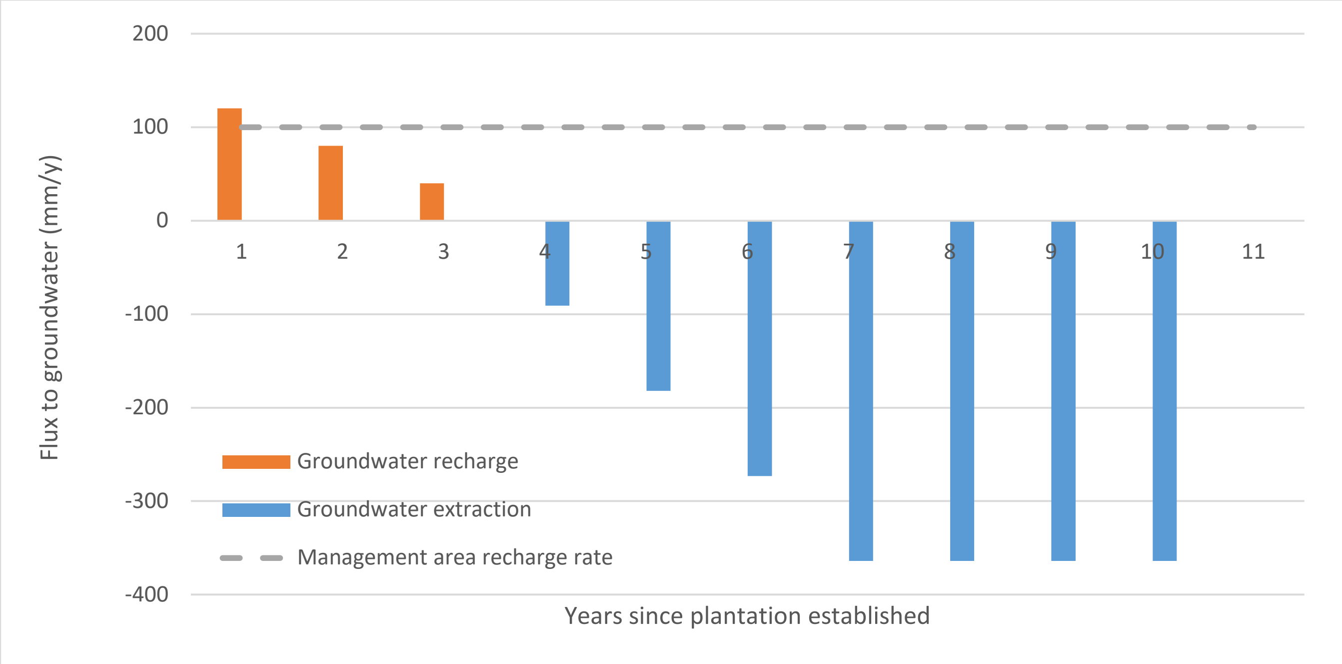


Figure 2.12. Conceptual forest water accounting model for a hardwood plantation with an 11 year cycle (10 years of growth and one year of post-harvest clean up), in a management area where recharge is 100 mm/y

While the forest water accounting models have been found to be broadly accurate (Harvey, 2017), most hardwood plantations have been in place longer than originally anticipated or accounted for, and hence are extracting more groundwater than accounted for. Furthermore, Harvey (2017) and Crosbie (2015) both found that the maximum depth at which plantations can extract groundwater is likely to be more than 6 m. Crosbie et al., (2015) discussed the influence of soil type on extinction depth of groundwater use by plantation forest in the South East. Based on analysis of satellite data, Crosbie et al., (2015) found that plantations use groundwater up to a depth of 6 m below ground level where soil texture is light (less than 5% clay), whereas they could be using groundwater at greater depths where soil textures are heavier (up to 20 m where clay content is greater than 25%).

2.4.6 Groundwater extraction for irrigation, stock and domestic use

In addition to groundwater extraction by plantation forests, groundwater is also extracted for irrigation purposes. Based on metered groundwater extraction data from Harrington and Li (2015) and from DEWNR's internal database, average groundwater extraction in the study area is 36,117 ML/y, however metered data only exists from 2009. The majority of this extraction occurs in the eastern parts of the study area, particularly in region north of Penola (Figure 2.3).

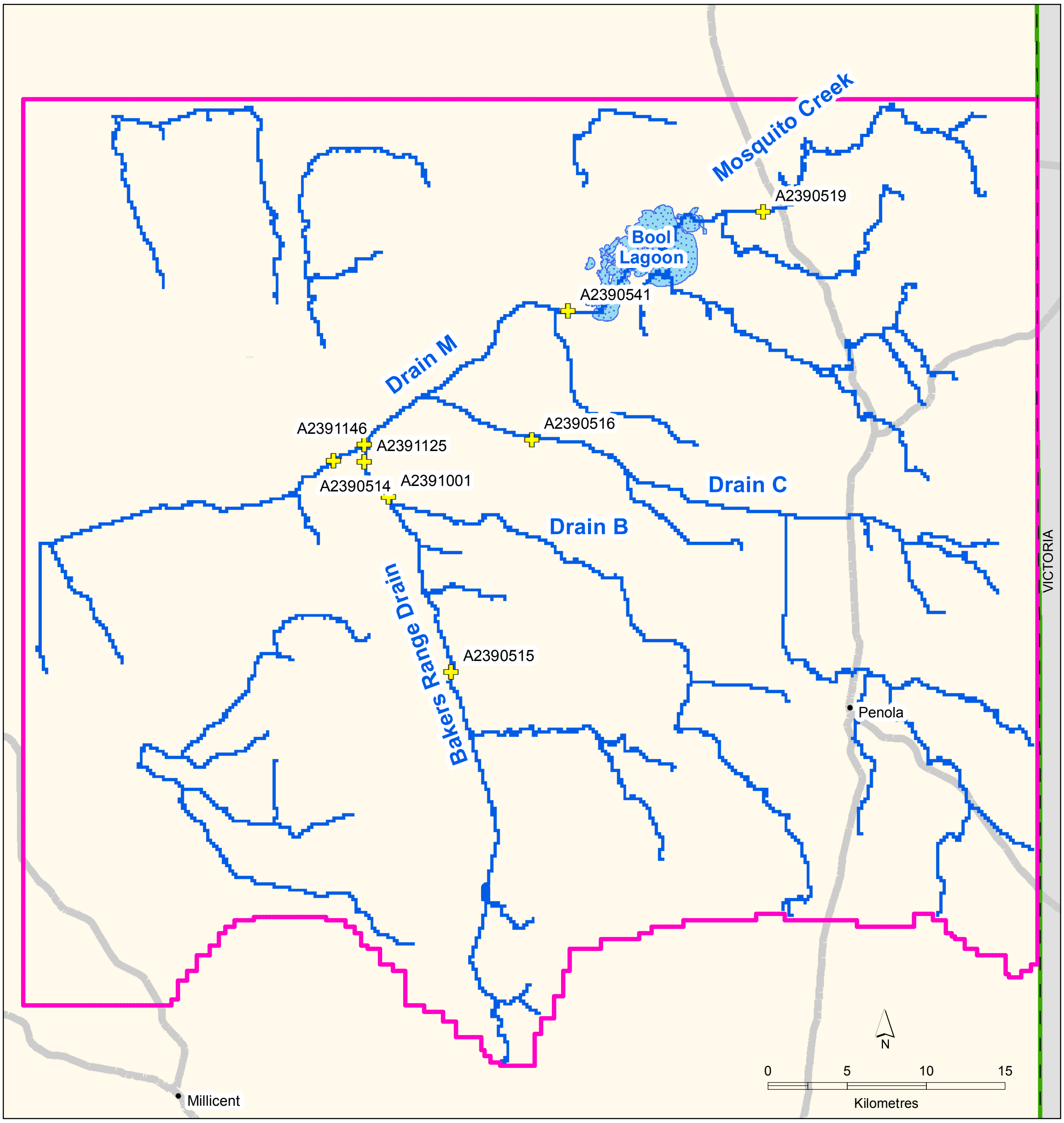
No metered data on stock and domestic groundwater use exists in the region. The LLCWAP provides estimates of use based on potential stocking rates within each management area, and estimates of the population in each management area (SENRMB, 2013). These estimates give a total volume of 12,094 ML/y in the study area, however it is unclear if stock use estimates take into account variability in land use (areas of pasture versus areas of plantation forest). Harvey (2017) provides an estimate of stock water use of 0.03 ML/ha for pasture land use in the study area. Subtracting land uses such as plantation forest, crop irrigation, lakes/wetlands, residential and native vegetation from the study area leaves ~2300 km^2 suitable for grazing. Thus stock water use estimated from this number is 6927 ML/y, assuming all available pasture is being stocked at capacity (Harvey, 2017). Domestic water use is estimated by Harvey to be ~3.35 ML/y per dwelling. Land use mapping shows 36 rural residential properties in the study area, thus domestic water use is estimated to be 120.6 ML/y. A total estimated stock and domestic value of 7047 ML/y is significantly less than the volume estimated in the LLCWAP, and neither figure can be validated due to lack of data.

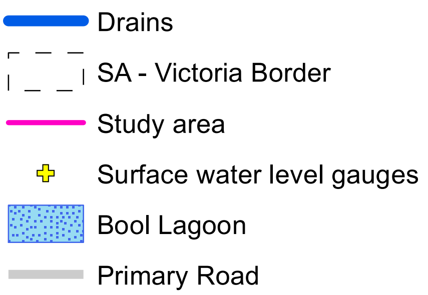
2.5 Surface water and drains

2.5.1 Bool Lagoon

The Bool Lagoon complex is the main surface water feature in the study area (Figure 2.13). Bool Lagoon is an ephemeral wetland which receives both surface water and groundwater inflow. Surface water inflow comes from ephemeral flows in Mosquito Creek. Smith et al., (2015) characterized Bool Lagoon as a regional groundwater discharge site, but note that both groundwater recharge and discharge occur intermittently. Groundwater discharge into Bool Lagoon is likely to occur in up gradient parts of the wetland, with additional discharge through ET from the shallow watertable. However when inflow from Mosquito Creek raises the surface water level in Bool Lagoon, there is likely to be some recharge of surface water back into groundwater.

Figure 2.13 Location of surface water features in the study area





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Bool Lagoon is a Ramsar listed wetland and subject to management intervention (Department for Environment and Heritage, 2006). When the volume of water in Bool Lagoon exceeds 20 GL, excess water is released into Drain M, which flows through the remainder of the study area, before eventually discharging at the coast. Taylor et al., (2015) presented a water balance for Bool Lagoon based on a review of available data. The water balance (Figure 2.14) shows that while both inflow from Mosquito Creek (D_{in}) and outflow to Drain M (D_{out}) were high from 1987 to 1996, reduced rainfall (P) from 1996 onwards resulted in decreased flow volumes. From 2006 onwards, evaporation from surface water (E) was no longer a component of the water balance as the wetland was often dry, and ET from underlying sediments (ET_{sed}) increased. G_{net} in Figure 2.14 represents the estimated net groundwater balance for Bool Lagoon, where Gn_{et} values less than 0 equates to net groundwater discharge into Bool Lagoon, which is the dominant trend in G_{net}.

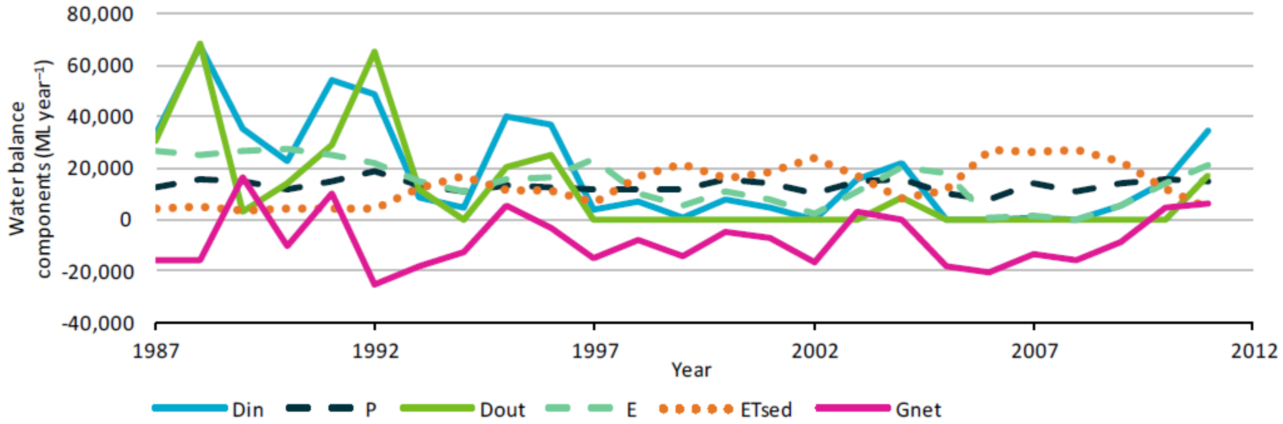


Figure 2.14. Water balance for Bool Lagoon (from Taylor et al., 2015)

2.5.2 Drainage network and surface water-groundwater interactions

Drain M also receives discharge from the Bakers Range drain, which flows from the south of the study area. The drainage network in the LLC is designed to regulate flows in wetlands like Bool Lagoon, and prevent surface water inundation in low lying areas (SENRMB, 2013). Drain flows are gauged at a number of locations in the study area (Figure 2.13), however most have periods of missing data. For example, A2390515 is missing all data from 1993-2010. Nevertheless a trend of higher flows in the 1970s and 1980s, and lower flows from the mid-2000s onwards can be observed (Figure 2.15), consistent with rainfall trends in the area (Figure 2.2).

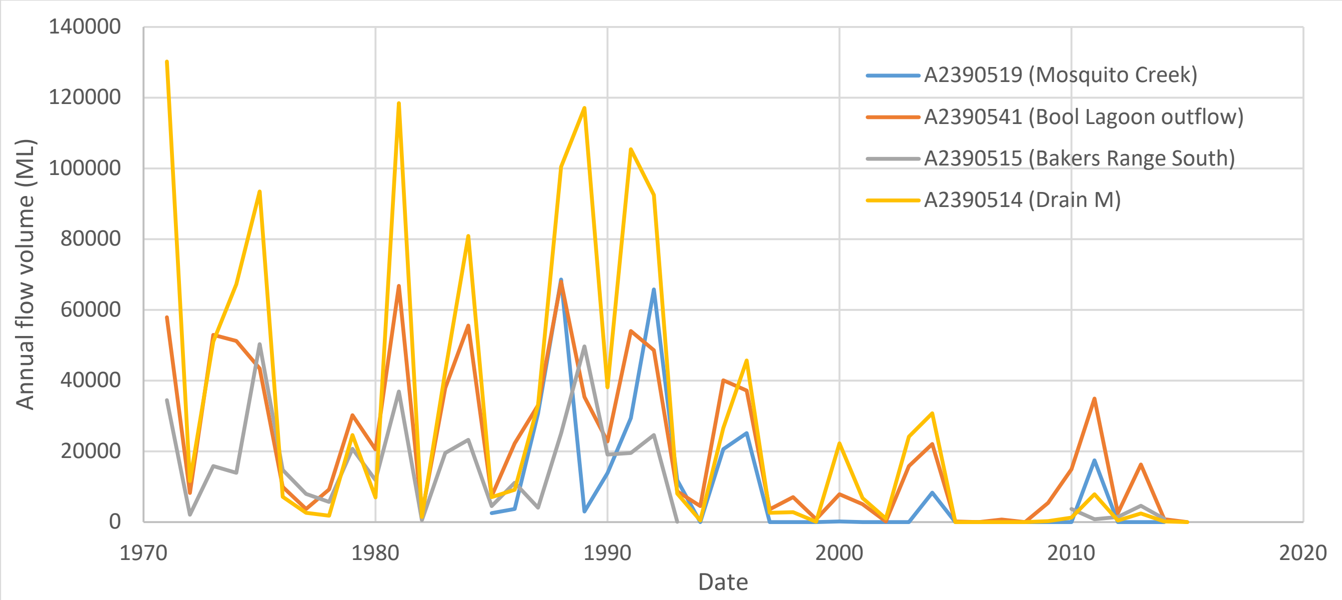


Figure 2.15. Measured drain flows in the study area (Bakers Range South data missing from 1993–2010)

In parts of the study area where the watertable is shallow, drains are also likely to intercept groundwater (Mustafa et al., 2006). The spatial and temporal variability of groundwater fluxes to drains are not well understood (Harrington et al., 2013). Wood and Way (2011) estimated groundwater fluxes to the Bakers Range Drain and Drain M at two points in the study area to be 800–1500 L/d/m, based on 2004 water levels. They attempted the same calculation for 2008 water levels, however groundwater levels were too far below the bottom of the drain for any discharge to occur.

Harvey (2017) identified groundwater level trends adjacent to the Bakers Range drain in the Short management area (observation well SHT012) which may be influenced by surface water in drains recharging groundwater in 2009–11. Groundwater level declines due to plantation forests have in some cases lowered groundwater levels below drains. Peaks in drain flow measured 7 km downstream of SHT012 correlate well with peaks in groundwater (Figure 2.16). An analytical solution from Walton (1997) can be used to estimate the potential flux from the drain to groundwater:

$$q_h = h_o\sqrt{ST/\pi t}$$

Where q_h is the discharge from the drain to the aquifer (L^2/T) per unit drain length on one side, h_o is the rise in drain level, S is the aquifer storage coefficient, T is aquifer transmissivity and t is the time since the rise in water level. For a T of 1000 m^2/d, S of 0.1, and a rise in drain level h_o of 1.46 m, a discharge of 23 m^2/d per metre of the drain length on one side is estimated. However the rise in groundwater levels in SHT012 in 2009 also matches a rise in rainfall trends (Figure 2.16). Furthermore a rise in groundwater level can be observed in observation wells throughout the LLC following above average rainfall from 2009-11 (DEWNR, 2013).

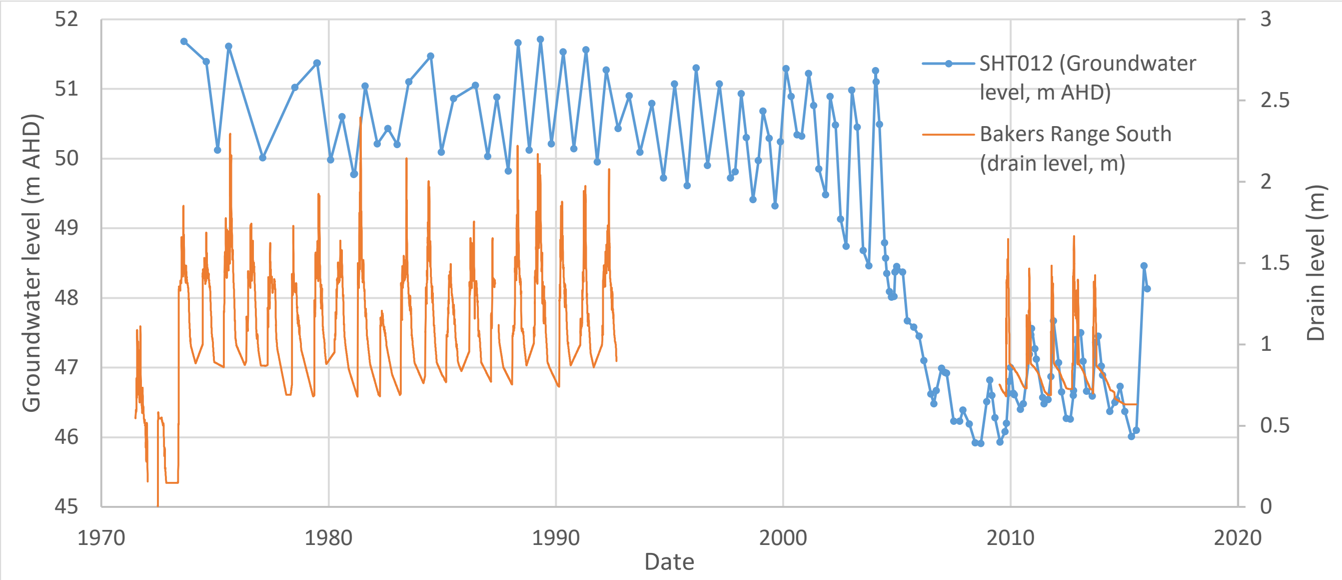


Figure 2.16. Groundwater level in SHT012 and in Bakers Range Drain (7 km downstream of SHT012)

2.6 Water balance

Based on the data available, an approximate water balance for the study area can be presented. Many of the fluxes in the water balance such as rainfall recharge, ET of shallow groundwater and groundwater extraction for irrigation are likely to vary from year to year, hence only a broad estimate of the water balance can be made. Nevertheless, it provides a good conceptual understanding of the system for development of the numerical model.

2.6.1 Groundwater inflows

2.6.1.1 Regional inflow

Lateral groundwater inflow from the east of the study area can be estimated with Darcy's Law:

$$Q = TiL \qquad \text{(Equation 1)}$$

Where *Q* is the volumetric flux across the boundary, a function of aquifer transmissivity (*T*), the hydraulic gradient of groundwater flowing in (*i*), and the length of the boundary (*L*). Along the eastern inflow boundary, transmissivity values of 50–4400 m^2/d have been recorded. The hydraulic gradient varies slightly along this boundary, being steeper in the north, and less steep in the south. For the purposes of calculating inflow, an average gradient of 3.33 x 10^{-3} (5 m/1500 m), and a transmissivity value of 3000 m^2/d is used. Thus the inflow volume along this 55 km long boundary is 200,750 ML/y.

Inflow from the south-eastern boundary of the study area, where the gradient is very flat (2.5 x 10^{-4}) is likely to be 6844 ML/y, again using a transmissivity of 3000 m^2/d.

2.6.1.2 Diffuse recharge

The LLCWAP (SENRMB, 2013) gives volumetric estimates of recharge in each groundwater management area in the LLC, based on the vertical recharge estimates given by Brown et al. (2006). The total recharge volume given by these estimates is 333,324 ML/y. An assumption here is that recharge is evenly distributed in the management areas. In other words, only 50% of the Joyce management area is included in the study area, therefore only 50% of the recharge volume for Joyce is included in this estimate.

The LLCWAP also estimates the volume of recharge intercepted by plantation forest in the study area, based on areas of plantation extent at June 2012. The total recharge interception volume is 85,228 ML/y. Thus the net recharge volume (rainfall recharge minus plantation interception) is 248,096 ML/y.

2.6.1.3 Recharge from drains and surface water

Bool Lagoon has been identified as an area where groundwater may be recharged intermittently from surface water, however the site is considered to be predominantly a groundwater discharge area (Smith et al., 2015), hence no recharge at Bool Lagoon is included in the water balance. The potential for groundwater recharge to occur from drain flows during high flow events has been identified in the Short management area (Harvey, 2017). However the rise in groundwater levels noted by Harvey (2017) also matches a rise in rainfall trends, and is observed in groundwater levels throughout the LLC (DEWNR, 2012). Given these factors, and the difficulty in up-scaling estimates of drain recharge to groundwater in the South East, where seepage fluxes may vary by three orders of magnitude over relatively short drain reaches (Noorduijn et al., 2014), an estimate of drain recharge to groundwater is not included in the water balance in this study.

2.6.2 Groundwater outflows

2.6.2.1 Regional outflow

There is scant transmissivity data from aquifer tests in the western part of the model domain, with only one value of 496 m^2/d found. Using this value along the 57 km eastern boundary, and a gradient of 1×10^{-3} (5 m/5000 m), yields an estimate of 10,320 ML/y groundwater outflow from the study area.

2.6.2.2 Groundwater extraction for irrigation

Based on metered groundwater extraction data, the average groundwater extraction in the study area from 2009-16 is 36,117 ML/y. For the purposes of the water balance 7047 ML/y is the adopted rate of stock and domestic groundwater use (Section 2.4.6). Lack of data on stock and domestic use is an acknowledged limitation, however it is only likely to be a relatively small component of the total water balance.

2.6.2.3 Groundwater extraction by plantation forests

The LLCWAP gives estimates of groundwater extraction by plantation forests, based on forests present at June 2012. This gives a total estimated groundwater extraction of 87,795 ML/y in the study area (SENRMB, 2013).

2.6.2.4 Evapotranspiration

There are currently no regional scale measurements of evapotranspiration (ET) of shallow groundwater from pasture (un-forested) land use in the South East, only estimates based on modelling. In a regional groundwater flow model of the LLC, Morgan et al., (2015) estimated shallow groundwater ET to be 986,000 ML/y, or about 40% of total groundwater flows. For the purposes of this water balance, we assume groundwater ET to be 40% of total outflows, in this case 95,000 ML/y. Averaged over the area where the watertable is within 2 m of the surface in the study area (~ 783 km^2) based on current groundwater levels, this yields an ET flux rate of ~ 121 mm/y. This may be an over-estimate compared with satellite based water balances reported by Crosbie et al., (2015), but it provides a reasonable estimate for the purposes of this water balance.

2.6.2.5 Groundwater discharge to drains and surface water

No measurements of the total groundwater discharge to drains exist for the study area. Using a regional groundwater flow model of the entire LLC, Morgan et al. (2015) estimated the groundwater flux to drains to be 222,000 ML/y for the period 2004–13, compared to the mean measured discharge of drains to the sea of 129,000 ML/y. This means that groundwater discharge to drains is ~1.7 times greater than the drain discharge at the coast, noting that evaporation further removes water from the drains. Applying the same proportion to flows measured along Drain M at Callendale, the most 'downstream' gauge in the study area, an estimate of groundwater discharge to drains in the study area of up to 222,000 ML/y is obtained, with an average volume of 6720 ML/y for the past 10 years. It should be noted that this is only a very broad estimate, as quantifying surface water-groundwater fluxes on a regional scale in the South East remains a challenge.

2.6.3 Water balance summary

Table 2.1 presents a preliminary water balance for the study area based on contemporary estimates of groundwater inflows and outflows (volumes are rounded to the nearest gigalitre). Overall inflows are greater than outflows, however the surplus in the water balance is not reflected groundwater levels in the area. As discussed in Sections 2.6.1 and 2.6.2, there is some uncertainty in the water balance volumes presented, due to spatial variability in aquifer properties and hydraulic gradient, and lack of data for fluxes such as stock and domestic use, ET and groundwater discharge to drains. The value of regional groundwater outflow in Table 2.1 is almost certainly an under estimate.

Spatial variability is also an important consideration when examining the net water balance. While this simple water balance is positive, there is variability in inflows and outflows across the study area. For example, in the management areas of Coles and Short, the cumulative impact of plantation forest extraction and recharge interception is greater than the estimated recharge volume. For Coles and Short, the water balance from these fluxes (recharge minus forest impacts) is in deficit in both areas, by 16,028 and 8750 ML/y respectively. This is reflected in the declining groundwater trends observed in these areas since the mid-2000s (Figure 2.10). However elsewhere in the study area, hydrographs have been relatively stable, for example, the pasture parts of the Fox and Kennion management areas (Figure 2.11). Hence the water balance is likely to be close to zero or positive in these areas.

Table 2.1. Preliminary water balance for the study area

Inflows	Volume (GL/y)
Regional inflow	207
Recharge	333
Total inflows	*540*
Outflows	
Irrigation extraction	36
Stock and domestic use	7
Plantation extraction	88
Plantation recharge interception	85
Evapotranspiration	95
Regional outflow	10
Groundwater discharge to drains	6
Total outflows	327
Inflows - outflows	***213***

2.7 Conceptual model summary

Regional groundwater flow in the study area occurs through the Quaternary and Tertiary aquifers, which for management purposes are considered one continuous, unconfined aquifer, referred to as the Tertiary Limestone Aquifer (SENRM, 2013). Groundwater flow is from the elevated Naracoorte Range in the east, to low lying plains in the west. Inflows from diffuse recharge slightly outweigh inflows from regional groundwater flow. ET from shallow groundwater is likely to make a large contribution to outflow, based on estimates from previous modelling exercises. Groundwater extraction by plantation forests outweighs extraction for irrigation in the study area, however the two processes are spatially variable. Likewise the water balance in the study area varies spatially. Overall there is a net positive water balance, and stable groundwater levels are observed. However in areas of plantation forests, the water balance is in deficit, and declining groundwater levels are observed.

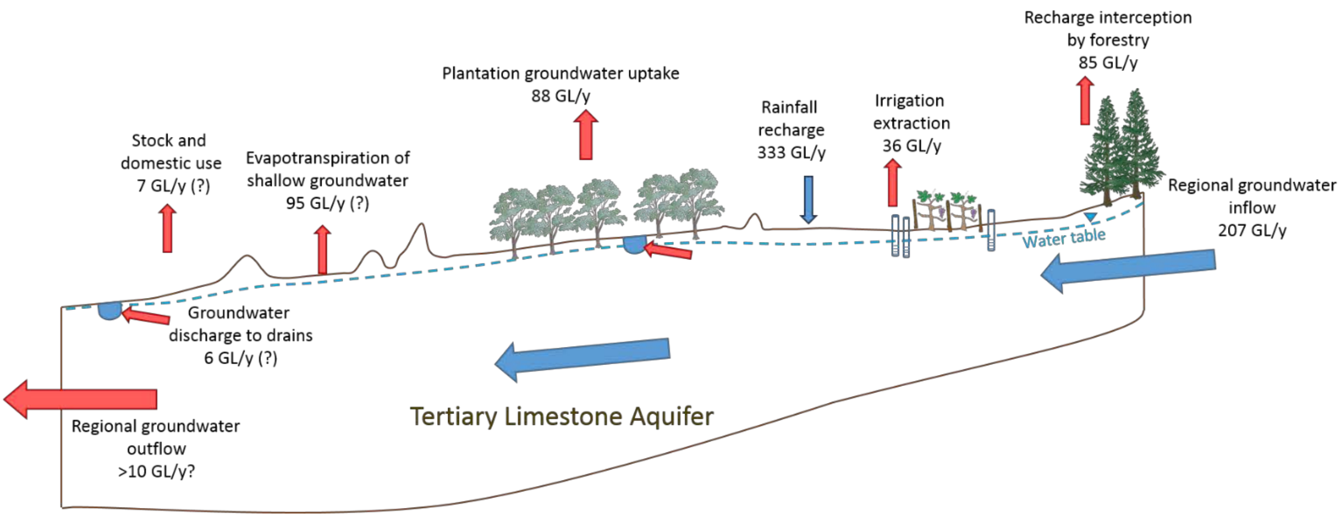


Figure 2.17. Conceptual water balance model of the study area (forest and vineyard symbols taken from Integration and Application Network, 2017)

3 Model description

3.1 Previous numerical models

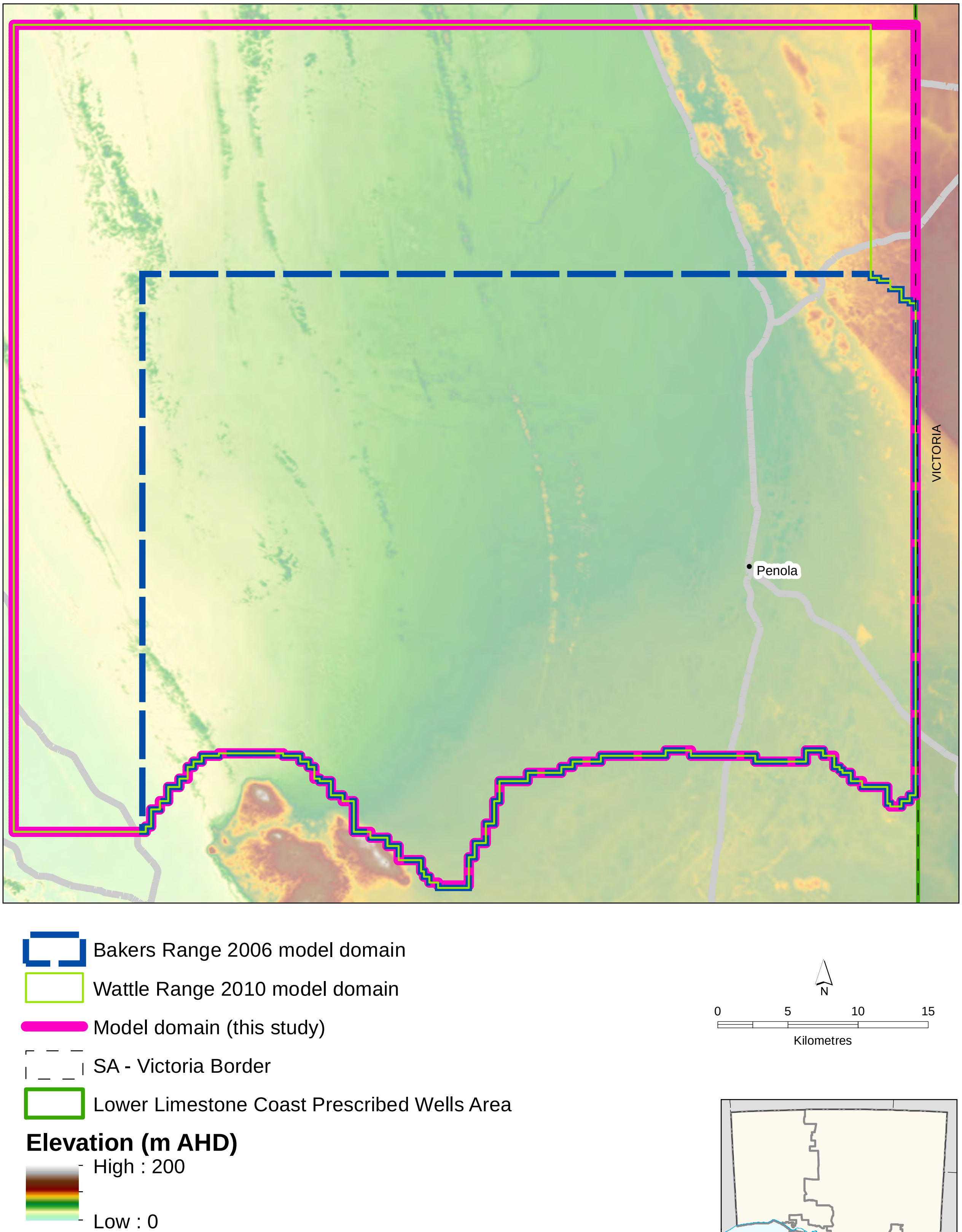
The model described in the report is a revised version of the Wattle Range 2010 model developed by Aquaterra (2010a), itself an expansion and revision of an earlier model developed by the Government of South Australia (Mustafa et al., 2006). While a considerable amount of work went into both models, the following sections presents only a brief summary of each model. More focus is given to the revisions made as part of this study, which includes description of the previous models (Section 3.2). The reader is referred to Mustafa et al. (2006) and Aquaterra (2010a, b) for full details of the previous models.

3.1.1 Bakers Range model

The Bakers Range groundwater model was constructed as part of a project investigating land use impacts on groundwater level and drain flow in the Bakers Range and Wattle Range region (Figure 3.1) in the early 2000s (Mustafa et al., 2006). The main land use change in the region of interest was the rapid expansion of hardwood (Tasmanian blue gum, *Eucalyptus globulus*) plantation forests from 1999 onwards.

The model was constructed in the Groundwater Modelling System (GMS) Version 6 program (Brigham Young University, 2005), using the MODFLOW code (Harbaugh, 2005). The two-layer model domain covered 1995 km^2 (Figure 3.1). The upper layer represented a combination of Bridgewater Formation and the ‘upper’ Tertiary Gambier Limestone, while the lower layer represented the ‘lower’ Tertiary Gambier Limestone. The major components of the water balance in the model were recharge, evapotranspiration (ET) under plantation forests, groundwater pumping for irrigation, groundwater outflow via drains, regional inflow from a constant head boundary and outflow from a head-dependent flow boundary. Recharge varied according to land use, averaging 200 mm/y in pasture, and reducing to 0 mm/y under plantation forests. ET was simulated for plantation forest areas with the MODFLOW ET package. ET rates (107–671 mm/y) and the extinction depth (6 m) were based on rates given by Benyon and Doody (2004). Groundwater pumping for irrigation was based on estimated crop water requirements for 2003. Drains were simulated with the MODFLOW drain package, with an estimated conductance term, and measured drain dimensions. A constant head boundary was used along the east and south-eastern boundary of the model to represent regional groundwater inflow, and a general head boundary was used along the western boundary to simulate regional groundwater outflow, with remaining boundaries being no-flow boundaries.

Figure 3.1 Domains for the previous groundwater models developed in the area



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A steady state model was calibrated to 1970–71 conditions, and a transient model was calibrated from 1970–95, with hydraulic conductivity varied in four zones in Layer 1 and six zones in Layer 2 to achieve calibration. Hydraulic conductivity varied from 15 to 55 m/d, with values based on available aquifer test data in the region, and specific yield ranged from 0.07 to 0.15 for these zones. The calibrated model was then used to run two alternative validation scenarios from 1995–2005. The two scenarios differed in terms of the presence and absence of plantation forests, commencing in 1999. The results showed a very good model fit to data in the Short management area for the forest validation scenario in which recharge rates were reduced while ET increased in forested areas in accordance with the findings of Benyon and Doody (2004). The results showed a poor fit for the alternative forest absence scenario, demonstrating that plantation forests have a clear interception and extraction effect.

Limitations identified in the Bakers Range model related to the lack of groundwater extraction data, as no metered extraction data was available, and the locations of extraction wells were not consistently known (Mustafa et al., 2006). Nevertheless the model provided an important first step in simulating the impacts of plantation forests on groundwater in the LLC.

3.1.2 Wattle Range model (WR2010)

Following the investigations into plantation forest impacts on groundwater in the LLC (Benyon and Doody, 2004; Mustafa et al., 2006; Benyon et al., 2006), water accounting models were developed to incorporate plantation forests in groundwater management through revision of the LLCWAP (Harvey, 2009). The water accounting models for forest, represented by rates of recharge interception and direct extraction, were tested in a revised version of the Bakers Range model – the Wattle Range model (WR2010, Aquaterra 2010a). The model was revised within the Groundwater Vistas program (Rumbaugh and Rumbaugh, 2011a) which also uses the MODFLOW code (Harbaugh, 2005). The domain was enlarged to cover the full extent of the hardwood plantation estate, which had expanded since 2006, to give a new domain area of 3440 km^2 (Figure 3.1). Additional changes to the model included:

- The two model layers were combined into one, as the lower layer was relatively thin and deep, and hence assumed not to have any impact on model results.
- The model thickness was revised based on updated aquifer thickness data.
- The drains were reassigned based on updated data and an extension of the drainage network since the Bakers Range model was constructed.
- Recharge was reduced from the uniform rate of 200 mm/y under pasture, to rates given by Brown et al., (2006), which vary from 50 to 180 mm/y according to groundwater management areas in the domain.
- Incorporation of ET across the entire domain for all pasture and native vegetation land use areas at a rate of 500 mm/y with an extinction depth of 2 m.
- Revised groundwater extraction data which was based on metered extraction, but which did not cover the Penola–Coonawarra area.
- Revised implementation of plantation forest recharge interception and groundwater extraction, where rates were based on water accounting models described by Harvey (2009). Furthermore, groundwater extraction by plantation forests was simulated as negative recharge (Aquaterra 2010a). The negative recharge approach was applied to robustly represent the management settings implicit in the WAP, without incurring substantial data processing overheads, although it suffers the limitation of not being a depth-dependent process. Two test scenarios which simulated the process with the MODFLOW ET package (Aquaterra 2010b).

A manual model calibration was performed for the period prior to hardwood forest development (1970–99), during which hydraulic conductivity values were slightly increased to range from 25 m/d to 78 m/d (Figure 3.2), and a uniform specific yield value of 0.08 was applied to the whole domain.

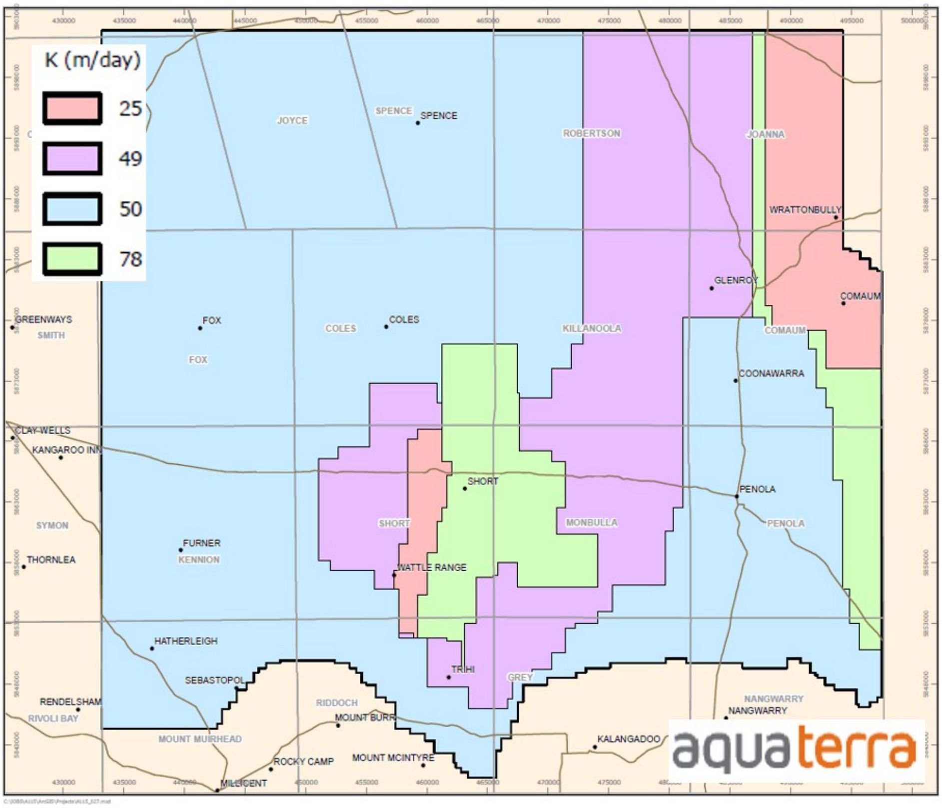


Figure 3.2. Hydraulic conductivity (K) zones used in the WR2010 model (taken from Aquaterra, 2010a)

A "blind verification" was then performed in which the model was run from 1999–2009 with calibrated parameter values, but with the addition of plantation forests intercepting recharge and extracting groundwater from 1999 onwards, at rates outlined in Harvey (2009). This simulation provided a good fit to observations of groundwater decline under plantation forests, validating the forest water accounting models. An additional simulation from 1970–2009 was run utilising parameter estimation software PEST (Doherty, 2010) to see if hydraulic conductivity values could be further optimised to improve model performance. However the PEST calibrated hydraulic conductivity values and model fit to data were similar to those obtained from the manual calibration, hence the manually calibrated model was used to run additional scenarios.

Eighteen scenarios were simulated exploring different plantation forest management options and different projected climate scenarios on groundwater in the area (Aquaterra, 2010b). Additionally two history matching scenarios were run in which plantation forest extraction of groundwater was simulated with the ET package, with extinction depths of 6 m and 9 m. Both simulations provided good fits to observed data, with the 6 m scenario providing a better fit in some areas, and the 9 m providing a better fit in other areas.

As with the Bakers Range model, the implementation of groundwater pumping data in the model was a recognised limitation. The WR2010 did not include any groundwater extraction data in the Penola–Coonawarra area (groundwater management area Zone 3A), where there is a significant amount of extraction for irrigation. Furthermore, the spatial implementation of forest impacts was based on collated information which reflected the best understanding of the forest coverage at the time. More accurate representation of groundwater extraction and the spatial extent of forest impacts were recognised areas for model improvement. Aquaterra (2010b) also recommended that future iterations of the model should simulate forest groundwater extraction with the ET package, rather than as negative recharge, especially if solute transport modelling is to be considered. While this was tested with two scenarios, Aquaterra (2010b) suggested that the model should be formally re-calibrated if the ET package is used, which was beyond the scope of their study.

3.2 Model revision

As part of this study, the WR2010 model was further revised based on currently available data, as well as the recommendations provided by Mustafa et al. (2006) and Aquaterra (2010a,b). Cumulatively, changes to boundary conditions and implementation of forests can be considered revisions of the underlying conceptualisation of the system, and thus a formal model re-calibration is required as recommended by Aquaterra (2010b). The following sections describe the changes made to the WR2010 model as part of this study.

3.2.1 Model domain, grid and surface refinement

The eastern boundary of the WR2010 model was extended northwards from the Bakers Range model boundary, such that the top north-eastern corner of the WR2010 model domain lies ~3 km west of the South Australian–Victorian border (Figure 3.1). As part of this project, the eastern boundary has been extended to the border in the north-eastern corner (Figure 3.1). This is because the entire eastern boundary is conceptualised as an inflow boundary (regional groundwater inflow from Victoria), and initial model runs as part of this project proved to be difficult to match potentiometric contours along the border with the segmented boundary. The revised model domain area is 3500 km^2.

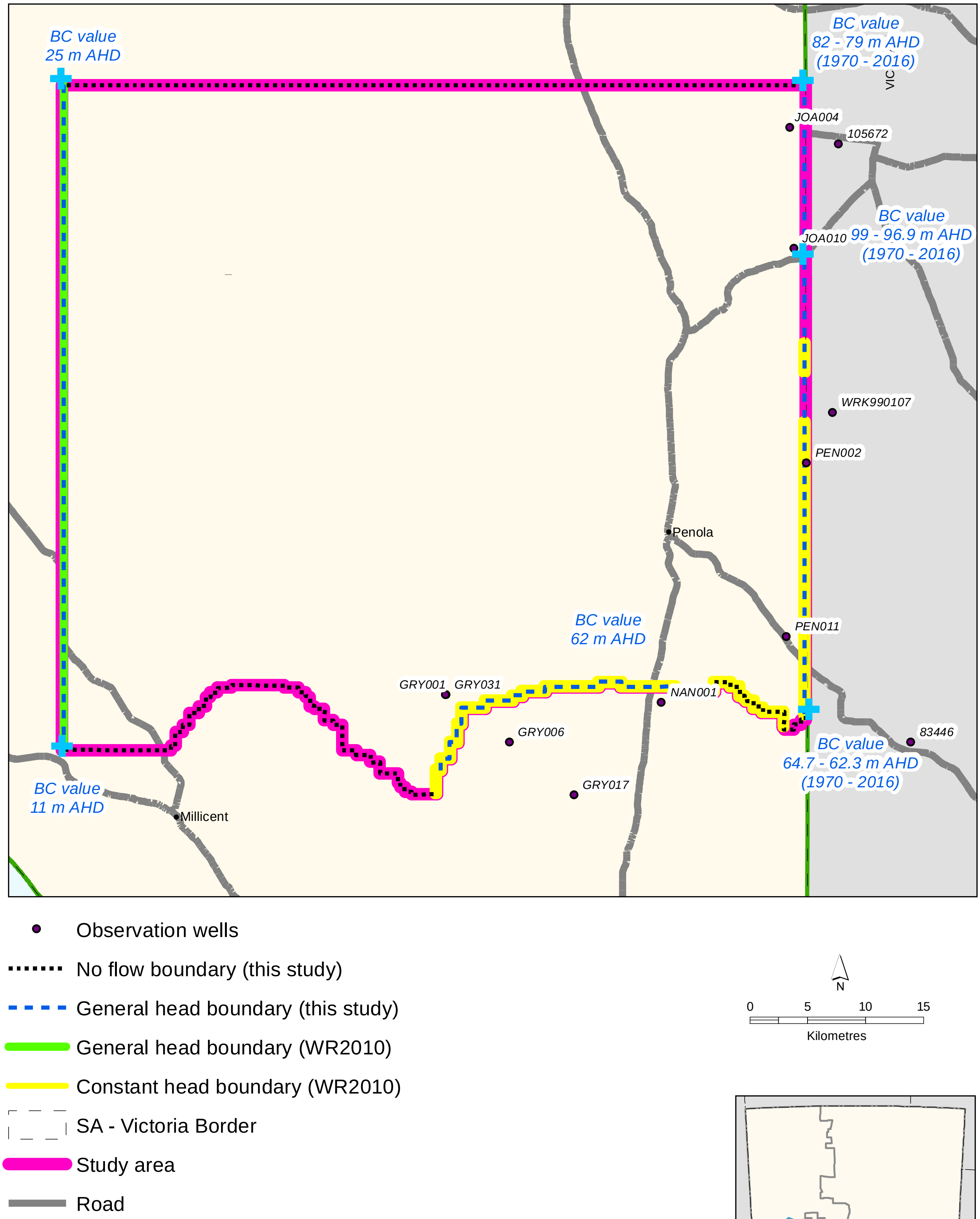
The WR2010 used uniform cell sizes of 200 m x 200 m. As Aquaterra (2010b) noted, the use of the ET package with a specified extinction depth requires accurate specification of the surface elevations in the model. Thus in order to refine surface elevations in the forested areas, the grid was refined in the Coles and Short management areas to give cell sizes of 100 m x 100 m. Refinement of Coles and Short areas invariably refined the surrounding forested areas to give cell sizes of 100 m x 200 m. This refinement gave a total of 175,865 cells.

Following the refinement of the grid, surface elevations were re-assigned based on a digital elevation model (DEM) of the region. The digital elevation model is based on airborne laser scanning data collected between October 2007 and May 2008, with a reported vertical accuracy root mean squared error of 0.5 m. Gridded ground elevation points are given at a scale of 10 m x 10 m (Location SA, 2017). This meant average elevation values from the DEM were assigned to the model cells, hence the accurate representation of ET may be limited by the model grid discretisation.

3.2.2 Domain boundary conditions

In the Bakers Range and WR2010 models, the eastern boundary is treated as a constant head boundary, with sections of no-flow boundary (Figure 3.3). This provides regional groundwater inflow, and matches patterns in the potentiometric surface in South Australia. However in reviewing the model, it was decided that the entire eastern boundary is likely to be an inflow boundary, as the unconfined aquifer extends into western Victoria (Barnett et al., 2015), and regional groundwater flow occurs from Victoria into South Australia (Morgan et al., 2015). Furthermore, groundwater levels along this boundary in both South Australia and Victoria have shown declining trends since 1970 east of Penola, and since the 1990s elsewhere (Figures 3.4–3.9). Therefore the eastern boundary was changed to a general head boundary to simulate groundwater inflow along the regional gradient, with time varying head values based on trends in Victoria and South Australia (Figures 3.4–3.9), and a high conductance value of 300,000 m^2/d so as not to restrict inflow.

Figure 3.3 Model boundary condition locations and values



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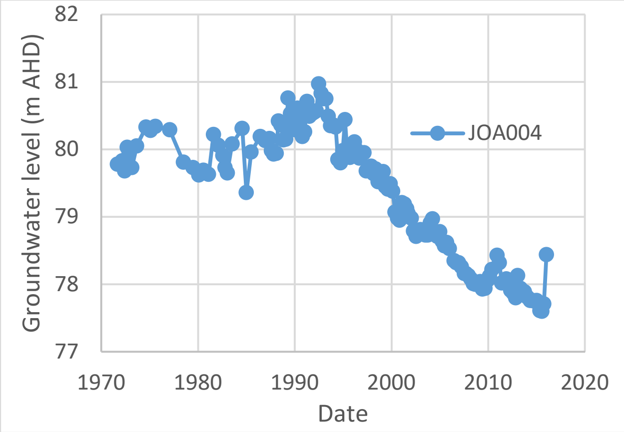


Figure 3.4. Groundwater levels in JOA004 (see Figure 3.4 for location)

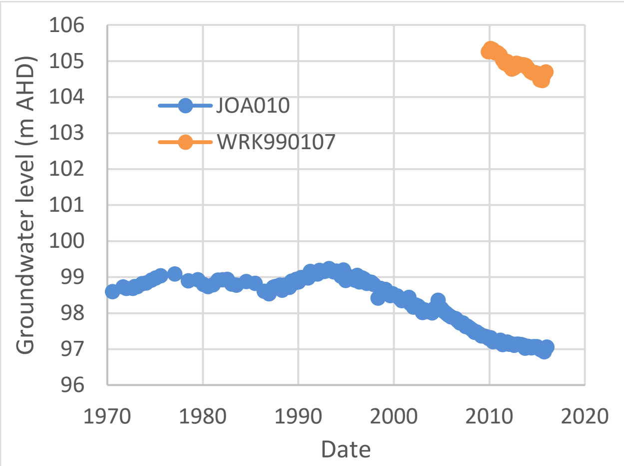


Figure 3.5. Groundwater levels in JOA010 and WRK990107 (see Figure 3.4 for location)

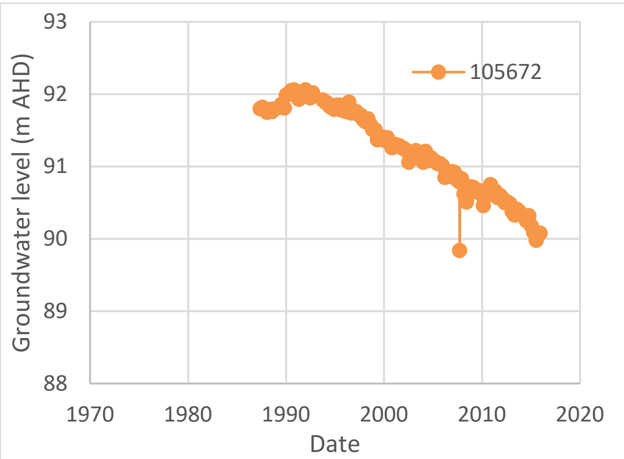


Figure 3.6. Groundwater levels in 105672 (see Figure 3.4 for location)

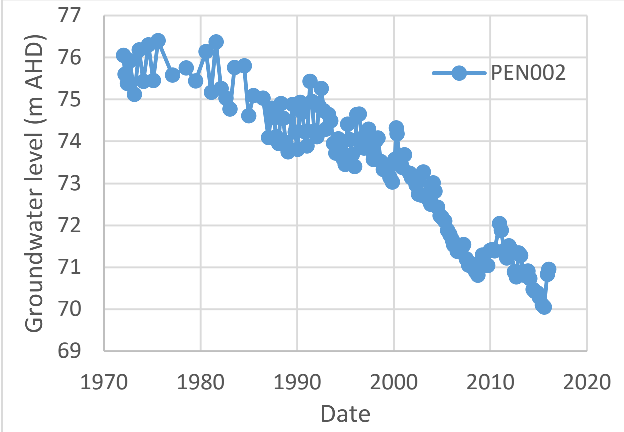


Figure 3.7. Groundwater levels in PEN002 (see Figure 3.4 for location)

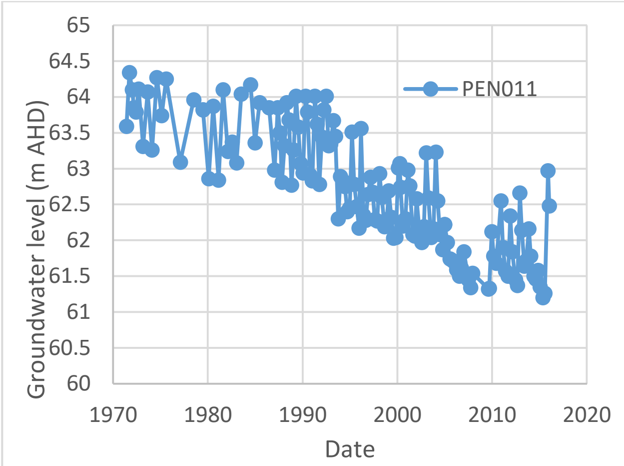


Figure 3.8. Groundwater levels in PEN011 (see Figure 3.4 for location)

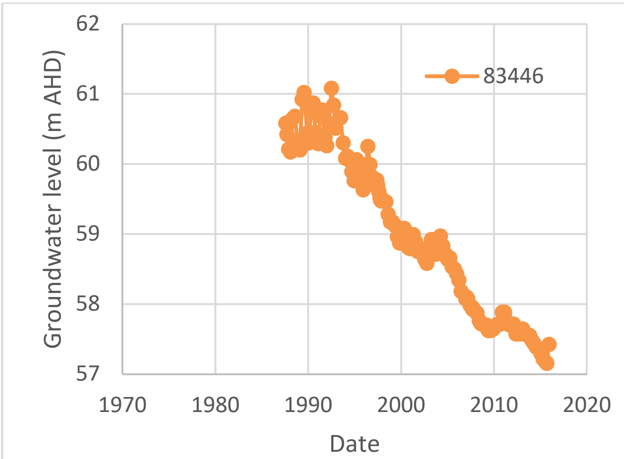


Figure 3.9. Groundwater levels in 83446 (see Figure 3.4 for location)

The south-eastern portion of the model domain is contoured to replicate a regional groundwater divide. The regional groundwater mound that can be observed in this area (Figure 2.7) has previously been attributed to thinning of TLA units in the uplifted areas south of the model domain (Holmes and Waterhouse, 1983). This section was previously set as a constant head boundary from the south east of the domain to the western edge of the Grey management area. However potentiometric contours for some years appear perpendicular to this boundary at the eastern edge of the domain (Brown et al., 2006). Hence it has been spatially refined, and made a general head boundary to replicate groundwater inflow from the south. Groundwater levels south of the divide vary spatially, with declines in the mid-1990s and mid-2000s. However inside the model domain near this boundary, groundwater levels have been relatively stable, despite declines from reduced rainfall with the millennium drought (note GRY031 replaces GRY001 in 2011; Figure 3.10). As these trends can be simulated within the domain with time-varying recharge, the southern boundary condition is set as a general head boundary set to 62 m AHD based on groundwater levels in this area to simulate regional groundwater inflow, but with levels that do not change with time (Figure 3.3).

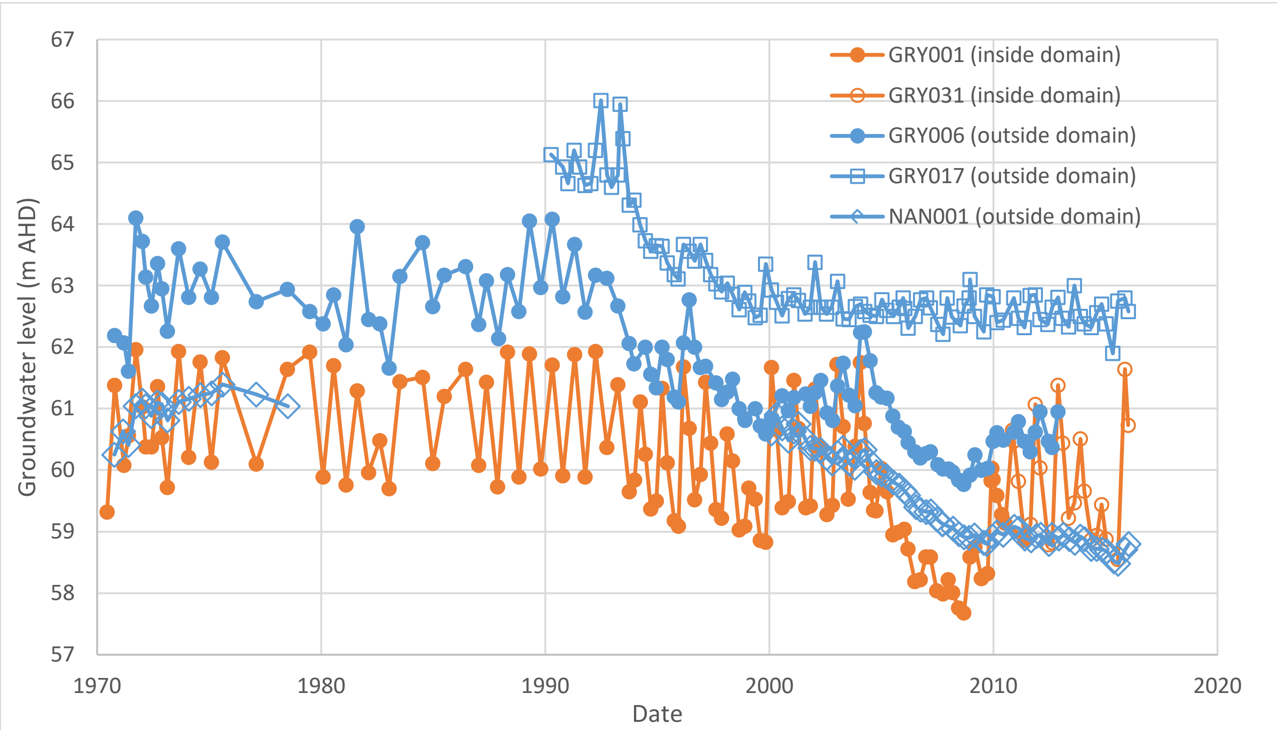


Figure 3.10. Groundwater levels inside and outside of the model domain near the south-eastern boundary (see Figure 3.3 for locations)

The southern boundary through the Riddoch and Mount Muirhead areas remains a no flow boundary, contoured to follow the extent of the Mount Burr range. The northern boundary of the model also remains a no-flow boundary. The western boundary of the model remains a general head boundary to simulate regional groundwater outflow. The head levels along this boundary do not change with time, given groundwater levels have been relatively stable since the 1970s in this part of the study area, much the same as groundwater levels in the western part of the model domain (Figure 2.11).

3.2.3 Groundwater extraction by pumping

In this study, a groundwater extraction data set for the South East developed by Harrington and Li (2015) is used to give extraction volumes and locations for the calibration period. These data are based on metered groundwater extraction data collected between 2009 and 2013, and assigns a pumping well to each metered extraction point. Harrington and Li (2015) used pumping well drill dates to create a groundwater extraction data set for 1970–2014, where historical (unmetered) groundwater extraction is estimated based on average extraction rates from the metered data. For this model, which runs through till the 2015–16 irrigation season, the data set was extended based on reported groundwater usage from DEWNR's internal database (Figure 3.11).

The average groundwater extraction volume from metered data reported by Harrington and Li (2015) for the model domain is 36,117 ML/y. This is significantly higher than the 15,506 ML/y extraction previously used in the model (Aquaterra, 2010a). However most of the groundwater pumping for irrigation is located in the eastern part of the model domain, particularly in the region north of Penola (Figure 2.3). Less than 5% of the groundwater extraction for irrigation occurs in the Coles and Short management areas, where hardwood plantation forest is most prevalent, and groundwater model performance is of most interest.

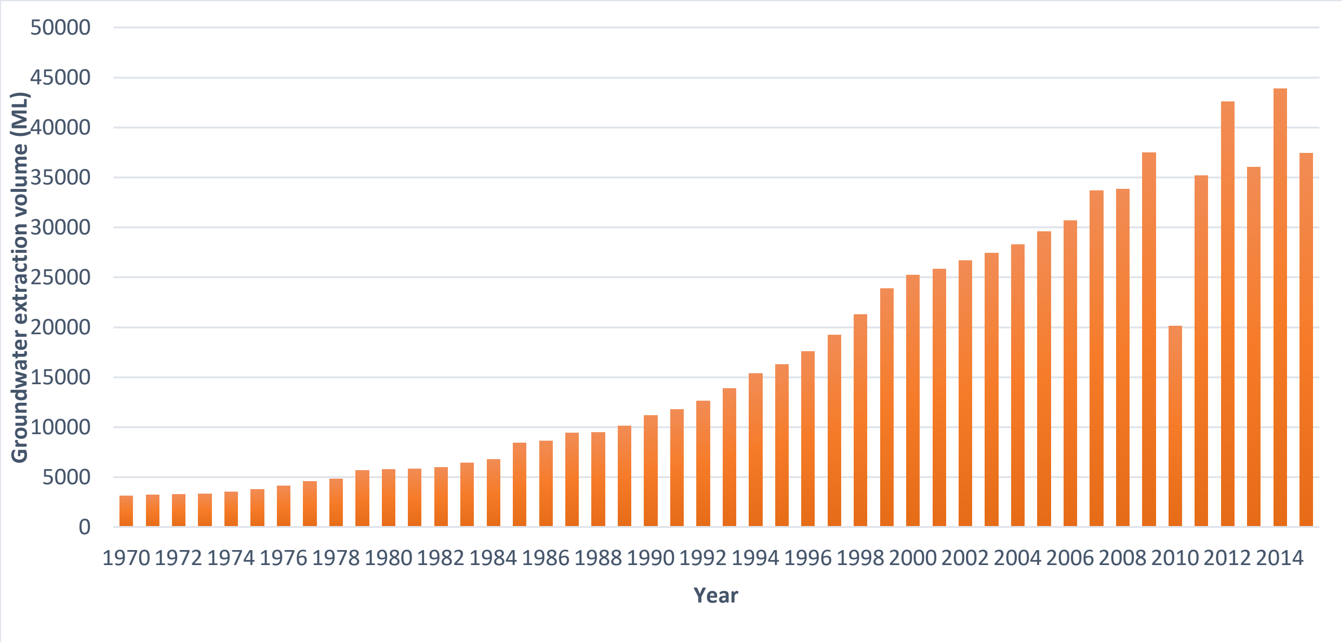


Figure 3.11. Measured (2009–16) and estimated groundwater extraction in the study area (taken from Harrington and Li, 2015, and DEWNR data)

3.2.4 Groundwater extraction by plantation forests

The previous models of Mustafa et al., (2006) and Aquaterra (2010a,b) used land use maps and estimates of the plantation forest footprint to simulate forested areas. Since the incorporation of plantation forests in the LLCWAP (SENRM, 2013), new data has been made available on the size and spatial extent of plantation in the study area. It should be noted that the data on plantation forest extent made available for this project at December 2016, is likely to be different to that used to develop water accounts in the LLCWAP (i.e. Table 1 in the LLCWAP). In addition to this, the date at which plantations were established is available for most plantation areas (Figure 3.12). The hardwood plantation estate, most of which is in the study area, has a bi-modal age distribution, with the majority of plantation being between 7–10 or 13–15 years old at December 2014 (Figure 3.12, Harvey, 2017).

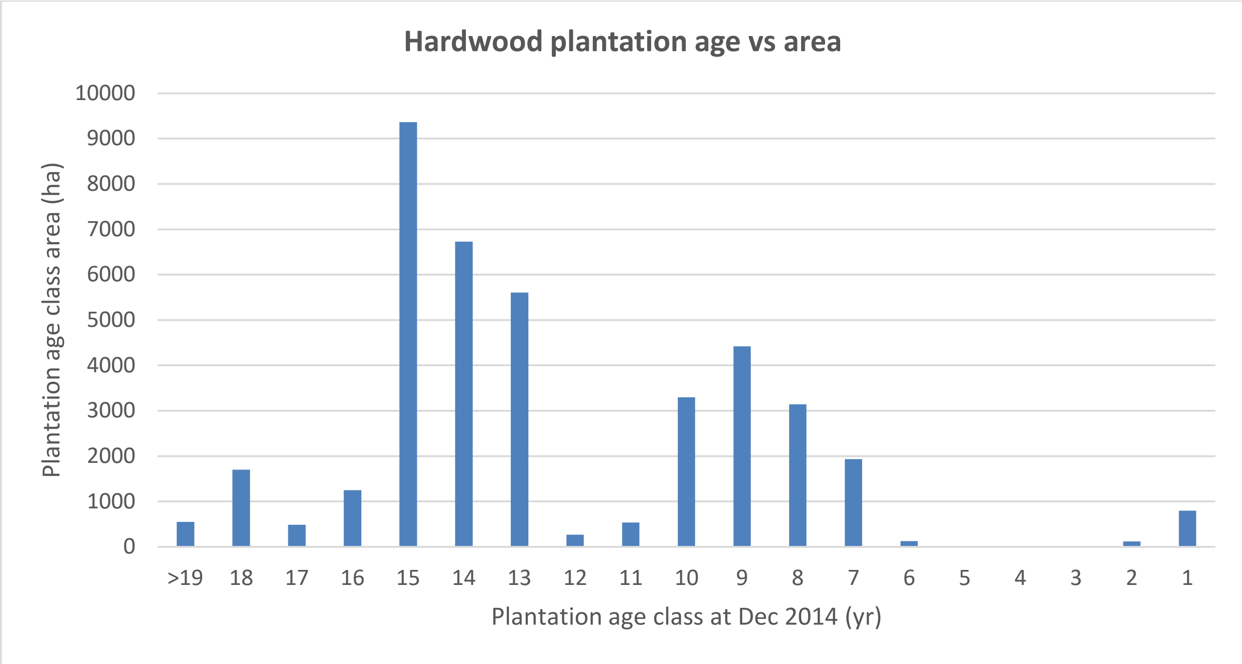


Figure 3.12. Hardwood plantation age class data (taken from Harvey, 2017)

This revised model uses this new data to simulate time varying groundwater extraction by plantation forests. Forested areas are introduced in the model based on their plantation date (Figure 3.12), and simulated groundwater extraction follows the rates given by the adopted forest water accounting models (Harvey, 2009). Thus most plantations begin extracting groundwater in the fourth year following planting, and extraction increases to a maximum rate of 3.64 ML/ha. Extraction is simulated using the ETS package in MODFLOW (Banta, 2000). Both the 'proportion of extinction depth' (PXDP) and 'proportion of maximum extinction depth' (PETM) parameters were set to 0.99, so that ET rates are constant with depth. The extinction depth for ET is set at 9 m. While the current policy adopts a 6 m extinction depth for plantation groundwater extraction (SENRM, 2013), both Crosbie et al., (2015) and Harvey (2017) have noted that extinction depth is likely to be greater. Aquaterra (2010b) found a 9 m extinction depth to be suitable for simulating plantation forest ET, hence that value is used here. Furthermore, there are observations wells in plantation forests in the study area which have shown groundwater levels decline deeper than 6 m below ground level (e.g. CLS002, CLS006, MON016, SPE010).

3.2.5 Evapotranspiration outside of forested areas

The previous model (Aquaterra, 2010a) used a maximum ET rate of 500 mm/y outside of forested areas, consistent with the model of Wallis (2008) for the Padthaway area to the north of the study area. Wallis (2008) cited CSIRO measurements and Bureau of Meteorology climate statistics in applying a rate of 500 mm/y. Padthaway has a lower mean annual rainfall of 516 mm/y, compared to 656 mm/y at Penola. Padthaway also shows higher pan evaporation measurements from 2009–16 than Coonawarra, 14 km north of Penola (BoM, 2017). Furthermore, Benyon and Doody (2004) reported ET rates for pasture in the study area ranging from 447–484 mm/y. Based on this, a maximum ET rate of 450 mm/y with an extinction depth of 2 m has been applied in the revised model.

3.2.6 Recharge

Several data sets for recharge in the South East are now available. Morgan et al., (2015) developed an unsaturated zone model for the entire LLC using LEACHM, to simulate monthly recharge from 1970–2013 for input to a regional groundwater model. However the model only accounts for four changes in land use from 1970–2013. In this study, the annual changes in plantation forest extent and consequently recharge over the past 20 years are of specific interest. Revising the LEACHM model to account for these changes is considered beyond the scope of this study, hence the LEACHM recharge model will not be used.

Doble et al., (2015) used unsaturated zone modelling with the WAVES program to develop a recharge data set for the South East. The modelling was used to develop a new net recharge (NETR) package for MODFLOW, which accounts for both recharge and ET. Different land uses (e.g. plantation forests) are specified with 26 different vegetation parameters in WAVES. However it is not known how these differ from the forest water accounting models in terms of recharge interception, and Doble et al. (2015) reported the WAVES modelled recharge be too high in plantation areas. As with the LEACHM model, revising the WAVES modelling to simulate the temporal variation in recharge under plantation forests is considered beyond the scope of this study.

As the purpose of the model is to assess the current groundwater management arrangements and forest water accounting models, and run scenarios for future groundwater management, recharge in the model will be based on management area recharge rates given by Brown et al., (2006). These rates were estimated using the watertable fluctuation method and a specific yield of 0.1 for the TLA. The average recharge rate for each management area is set as a percentage of average May–September rainfall from 1970–2015. The period from May–September is used as this is when more than 60% of annual rainfall occurs, and groundwater levels typically peak in September (Figures 2.8–2.11). It also corresponds with the bi-annual stress period used in the model. Recharge then varies each year, based on annual deviations in May–September rainfall. Thus for a management area where the adopted recharge rate is 50 mm/y (e.g. the Joanna management area), recharge is set as 13% of May-September rainfall from 1970–2015, to give a long term average recharge of 50 mm/y. Likewise where the adopted recharge rate is 180 mm/y (e.g. Monbulla), recharge is 40% of May–September rainfall, and varies annually to give a long term average of 180 mm/y (Figure 3.13). Following the approach in Brown et al., (2006), areas classified as lakes or native vegetation are assigned no recharge.

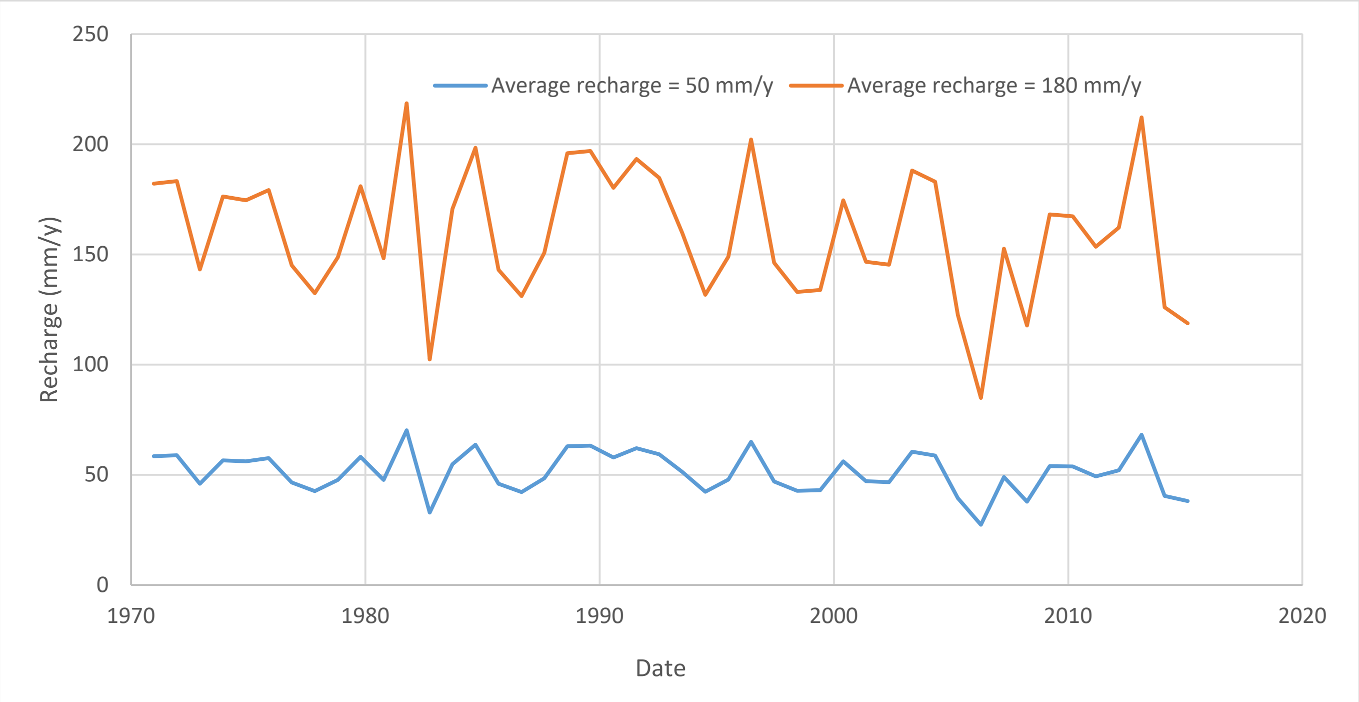


Figure 3.13. Upper and lower estimates of recharge in the model, where recharge varies in each management area

As with the plantation direct extraction, recharge interception by plantation forests is incorporated using the new information on the age classes of the plantation estate (Harvey, 2017). Recharge interception is applied to the recharge in each management area, based on the age of forest plantation (Figure 3.12), and following the forest water accounting models described by Harvey (2009), see Figure 2.12.

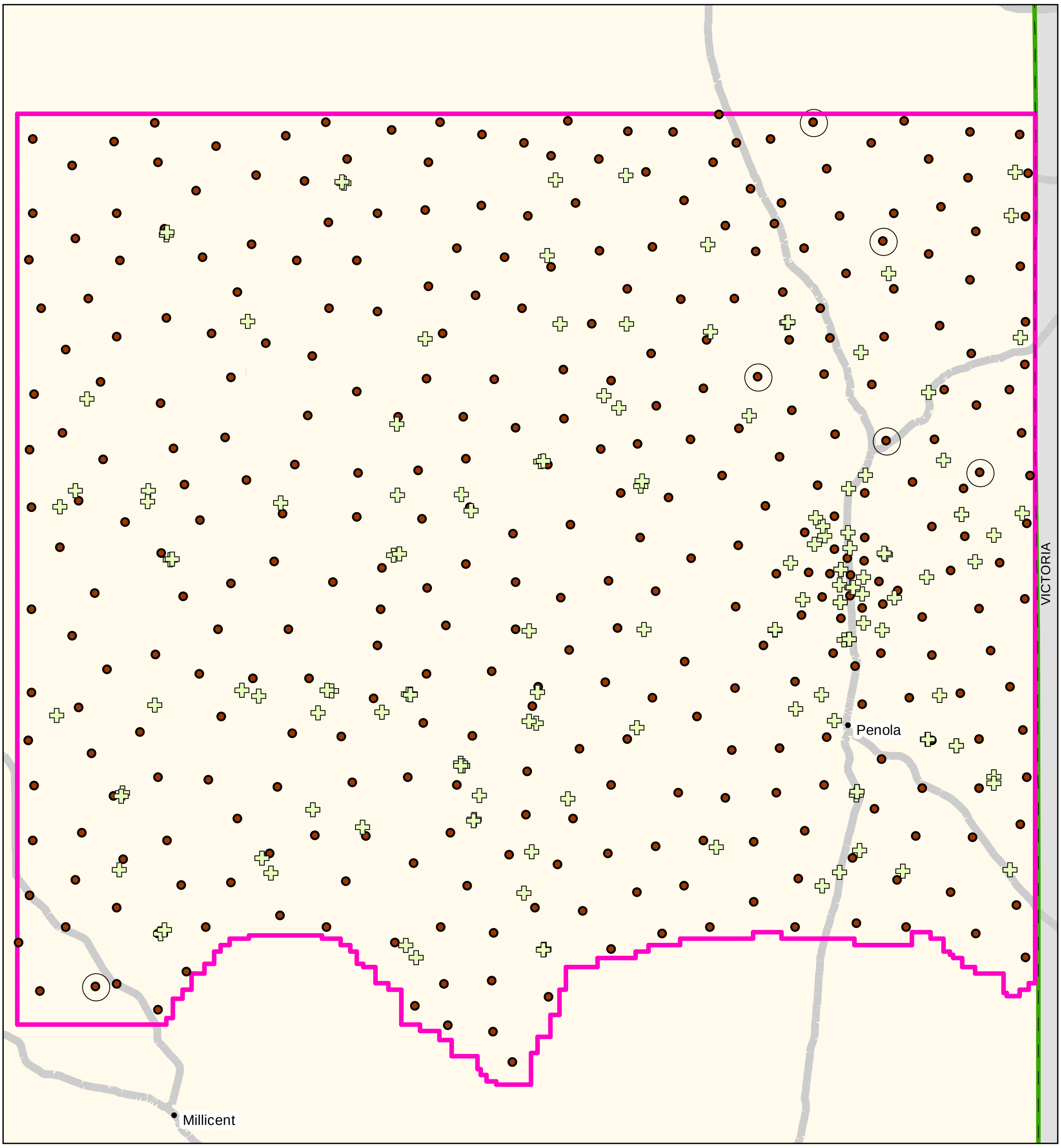
3.2.7 Aquifer properties and calibration

Previous iterations of the model have used a zoned hydraulic conductivity (K) approach, with values varying from 25–78 m/d (Aquaterra, 2010a, see Figure 3.2). A kriged (interpolated) K distribution based on pilot point values of

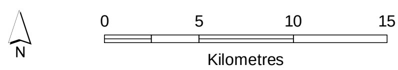
K, determined through the use of parameter estimation software (PEST), was also derived by Aquaterra (2010a). However there was minimal difference in the model calibration statistics, hence the zoned K model was used. Although both calibration approaches yielded the same result, an advantage of calibration with pilot points and PEST is that it provides a useful first step for later performing detailed uncertainty analysis on the model. Assessing uncertainty in groundwater model results is recommended by industry guidelines (Middlemis, 2001; Barnett et al., 2012) however it is often not performed. While a formal model uncertainty analysis was not conducted as part of this project, it is considered a key recommendation for future work. Hence it was decided to use PEST (Doherty, 2010) to calibrate aquifer properties in this model, using pilot points.

Pilot points were initially placed in the model domain using the Target Triangulation option in Groundwater Vistas (Harbaugh and Harbaugh, 2011b), where observation/calibration targets were connected to form a network of triangles in the model domain, and a pilot point was placed in the centre of each triangle. This resulted in 95 pilot points for K, and a higher density of points where more observations are present, consistent with recommendations given by Doherty et al., (2010). Doherty et al., (2010) also recommend using as many pilot points as numerically practical in model calibration, while Rumbaugh and Rumbaugh (2011b) recommend including pilot points between model boundaries and calibration targets. Hence an additional 254 pilot points were placed in the domain using the Fill Gaps option in Groundwater Vistas, where pilot points were created within every 10 cells of existing points. An additional 6 pilot points were introduced at locations where aquifer test data (and measured K values) were available, to give a total of 355 pilot points (Figure 3.14).

Figure 3.14 Pilot point and aquifer test locations



Observation wells
Pilot points
Aquifer test location
Study area
Primary Road
SA - Victoria border



Government of South Australia
Department of Environment, Water and Natural Resources

Produced by: Science, Monitoring and Knowledge Branch
Map Projection: Lambert Conformal Conic
Map Datum: Geocentric Datum of Australia 1994
Date: February 2017

Only six archived aquifer test data points could be found in the study area on the Department of Environment, Water and Natural Resources database, SA Geodata. Hydraulic conductivity values from these data points range from 1–127 m/d, however values up to 242 m/d are found within 2 km of the study area boundary. Thus pilot points at locations of aquifer test data were given an initial value based on their reported K value, and narrow upper and lower bounds within which they could vary of ± 20 m/d. The remaining pilot points were given initial values based on their zoned K value, and lower and upper bounds of 1–250 m/d respectively, based on the reported range of 1–242 m/d in and around the study area.

The distribution of hydraulic conductivity values following model calibration (see Section 4) is shown in Figure 3.15. While there are only six measurements of K in the study area, there are 4855 reported values of aquifer yield. Yield measurements are reported in litres per second, based on measurements or estimates of yield made while wells are being developed with equipment available to the well driller. This data does not correlate exclusively with aquifer transmissivity, as it depends on well construction and the method used to derive a yield value, be it from pumping or airlift. The resulting value is at best an estimate (National Uniform Drillers Licensing Committee, 2012). It does nevertheless provide a basic spatial map of relative aquifer properties.

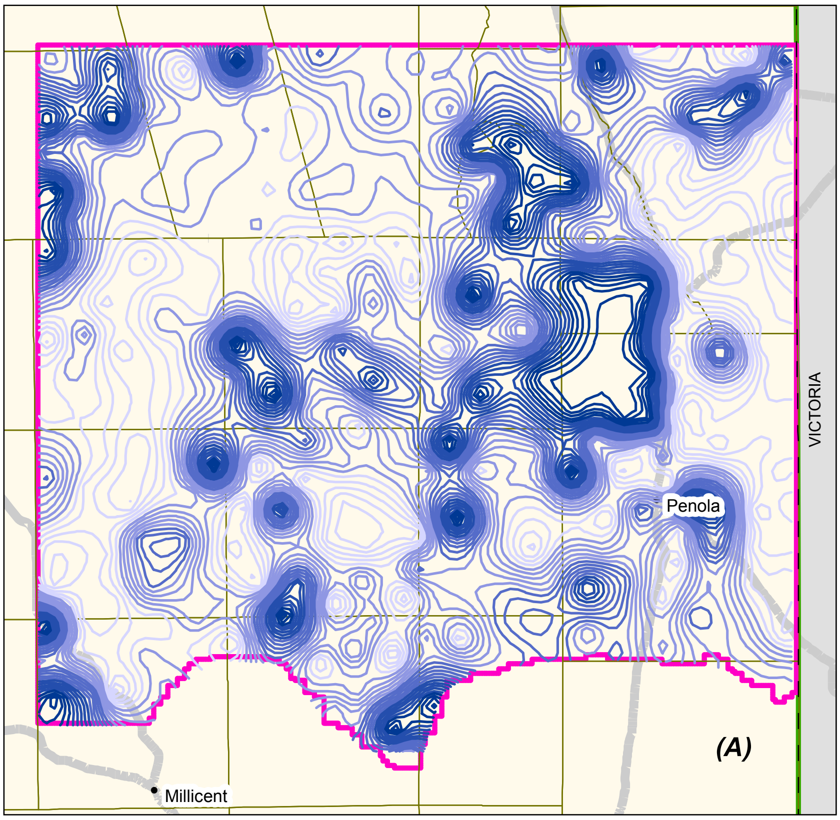
The hydraulic conductivity values derived by the model calibration match some of the regional spatial trends in aquifer yield in the study area. For example, higher aquifer conductivity is given in the south-western portion of the Coles management area, consistent with higher yield observations. Areas of higher hydraulic conductivity are also given in the top half of the Zone 3A management area, extending north-west into Glenroy, Killanoola and Struan. These patterns are broadly reflected in the aquifer yield data.

The calibrated hydraulic conductivity field does not match that derived by Aquaterra (2010a) using pilot points. However the previous model of Aquaterra (2010a) used observation data from 36 wells in the central part of the domain, and was calibrated using 38 pilot points, with lower and upper parameter bounds of 5–150 m/d. Hence the two calibrations cannot be expected to match. Nevertheless the pilot point calibration by Aquaterra (2010a) did produce high conductivity values in the north-west of Zone 3A, with five pilot points yielding conductivity values at the upper bound of 150 m/d. This is also an area of high hydraulic conductivity in the current calibration, and the aquifer yield data (Figure 3.15).

Locally the calibration has yielded some "bullseye" values around pilot points. As Doherty and Hunt (2010) point out, such points in a calibration field may be either a true reflection of heterogeneity, or an outcome of employing too few pilot point parameters in the calibration process. Thus employing more pilot points, or employing additional forms of parameter regularization may provide a 'smoother' hydraulic conductivity field. However, both aquifer test (Mustafa & Lawson, 2002) and aquifer yield data (Figure 3.15) for the area suggest large local variations and "bullseyes" in aquifer parameters are likely to exist. This is expected in the Lower Limestone Coast due to karst development in the Tertiary Limestone (Harrington et al., 2011). Thus no further regularization or addition of pilot points in model calibration is attempted here. However future work on this model or groundwater modelling in the South East in general could explore these options, perhaps with the use of well yield data.

The approach to simulating recharge in this study is based on watertable fluctuation estimates of recharge, which assume a specific yield value of 0.1. Hence a uniform specific yield value of 0.1 is applied throughout the model domain. Reported specific yield values for the TLA in the South East range from 0.075 to 0.3 (Mustafa and Lawson, 2002).

Figure 3.15 Hydraulic conductivity distribution in the model compared with reported aquifer yield



(A) Hydraulic conductivity distribution in the model

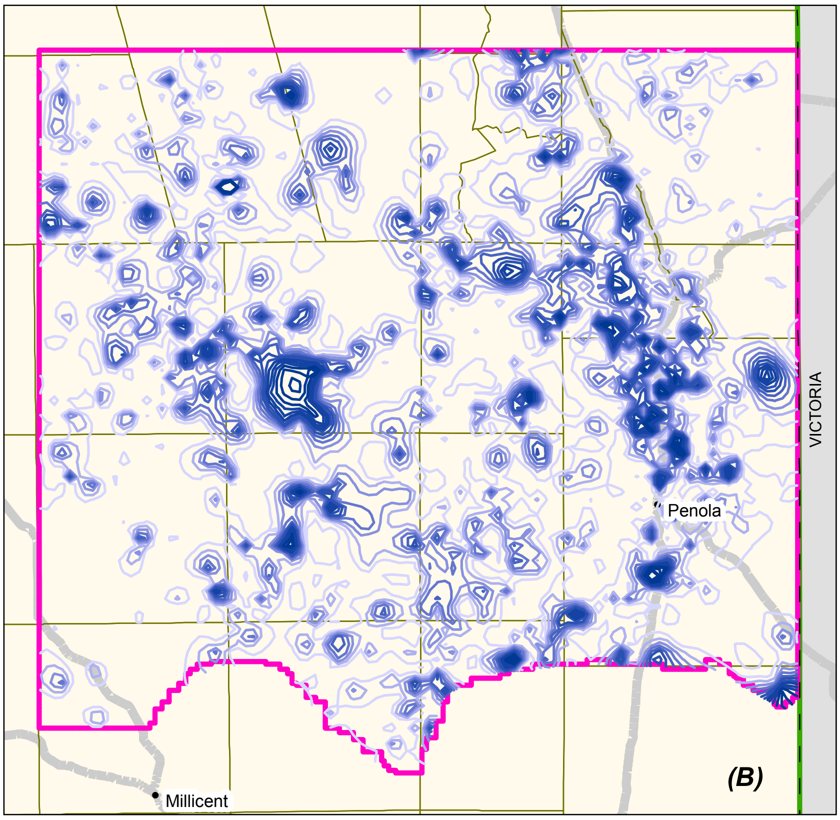
Hydraulic conductivity (m/d)

0 - 50
50 - 100
100 - 150
150 - 200
200 - 250
Groundwater management areas
Study area
SA - Victoria border
Primary Road
Lower Limestone Coast PWA

N

0 5 10 15
Kilometres

Government of South Australia
Department of Environment, Water and Natural Resources



(B) Reported aquifer yield in the study area

Aquifer yield (L/sec)

0 - 10
10 - 20
20 - 30
30 - 40
40 - 50
50 - 75
75 - 100
> 100

Produced by: Science, Monitoring and Knowledge Branch
Map Projection: Lambert Conformal Conic
Map Datum: Geocentric Datum of Australia 1994
Date: February 2017

3.2.8 Observation/calibration data

The previous model of Aquaterra (2010a) included 37 observation wells in calibration/validation. This has been expanded to 142 observation wells in the current model (Figure 3.14). However it should be noted that 17 of these 142 wells are replacement wells, installed to replace degraded or dry observation wells. In general, quarterly groundwater measurements are available at most locations, however the data is not consistent (see calibration plots in Appendix for time series observation data), and the 142 wells used gives a total of 10,872 head observations in the transient model.

3.2.9 Initial conditions

The original Bakers Range model of Mustafa et al., (2006) used a steady state model calibrated to average 1970-71 groundwater levels to develop initial heads for the transient model, which commences in 1970. The steady state model was calibrated to 1970–71 groundwater levels because there is scant groundwater observation data prior to 1970 (Mustafa et al., 2006). The same approach has been taken here. The steady state model was revised along with the transient model as described above (changes to model extent, boundary conditions, hydraulic conductivity). Here the steady state model is calibrated to March 1971 groundwater levels, as this is the first time when there is a reasonable coverage of observation wells across the study area.

3.2.10 Time discretisation

The WR2010 model was divided into two stress periods per year to simulate a winter recharge period occurring from mid-May to mid-September (123 days), and a summer groundwater pumping/ET period occurring from mid-September to mid-May (242 days). Each stress period was divided into five time steps. In this study, the winter recharge period has been extended from the start of May till the end of September (153 days), as recharge is based on monthly rainfall, and September is often a high rainfall month. Thus the summer groundwater pumping/ET stress period lasts 212 days. Winter recharge stress periods are divided into five time steps as per the WR2010 model and summer ET periods are divided into eight time steps. Increasing the number of time steps in the ET time periods was to keep mass balance errors below 1% when the ETS package is active, and did not have any impact on model calibration.

3.2.11 Assumptions and components of the model that were not revised

The scope of this project was to update and re-calibrate the WR2010 model. While several components of the model have been revised or updated, and the model re-calibrated accordingly, a number of components of the model have not been changed. This is because either there is no scientific justification or new data which warrants changing these components, or changing these components was considered beyond the scope of this project. Furthermore components of the model that remain unchanged are assumed not to have any significant impact on the results of interest in the model (groundwater levels under plantation forests in Coles and Short and surrounding management areas). However this has not been rigorously demonstrated as doing so would warrant constructing a new model, which is considered beyond the scope of this study. The components of the model that have not been changed include:

- Model platform (Groundwater Vistas) and code (MODFLOW-2005).
- Model layering – one-layer model representing flow in the unconfined aquifer, with no simulated interaction with the underlying confined aquifer.
- Model layer thickness – a stratigraphic model for the study area has been developed since the WR2010 model (Barnett et al., 2015), however it does not present new data on aquifer thickness within the study area, hence aquifer thickness in the model has not changed.
- Drain conductance – while drainage cells were refined following the grid refinement, no changes were made to drain elevations or conductance values.

- Bool Lagoon – As discussed in Section 2.5.1, Bool Lagoon is predominantly an area of groundwater discharge, with perhaps some groundwater recharge when surface water levels are high. However it also receives surface water inflow from Mosquito Creek, hence accurately simulating the dynamics of Bool Lagoon is considered beyond the scope of this study. Consistent with the previous models, the elevation of Bool Lagoon is accurately depicted in the model, and discharge to Drain M through Bool Lagoon is simulated. However the Lagoon is not simulated as lake or drain, and groundwater discharge to the Lagoon is approximated from a mass balance analysis of the volume of groundwater ET in the Lagoon area.

4 Model calibration

4.1 Steady state model

The purpose of calibrating a model to steady state conditions is to provide a broad match to head measurements for long term "average" conditions as well as to allow the simulated head to come into equilibrium with the specified boundary conditions. This also provides initial conditions for the transient model to be run and further calibrated (Middlemis, 2001). Following the original Bakers Range model, the steady state model was calibrated to 1971 groundwater levels, with the assumption that these represent average 1960–70 groundwater levels (Mustafa et al., 2006). While the steady state model provided initial conditions for the transient model, it was through calibration of the transient model that final hydraulic conductivity values were obtained. These values were then fed back into the steady state model and the fit to heads assessed. Overall the fit to data for the steady state model is good, with a root mean squared (RMS) error of 1.82 m, and a scaled root mean square error (SRMS) of 3.7% (Figure 4.1).

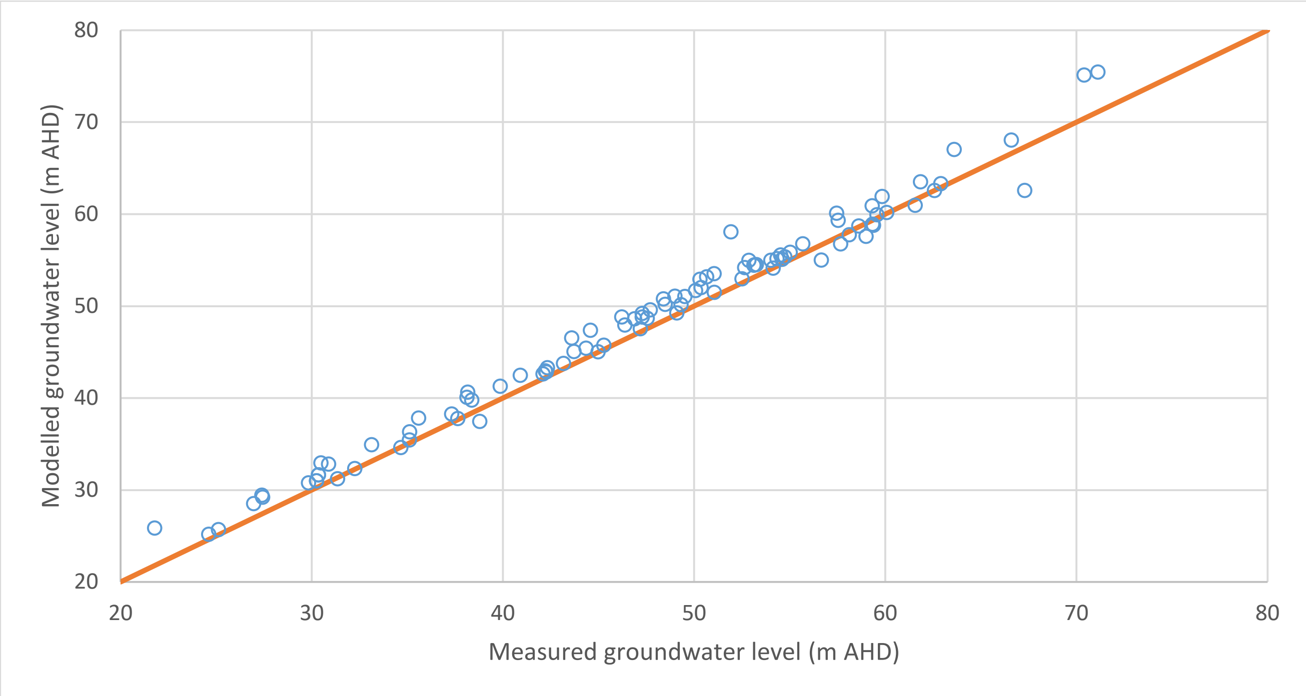


Figure 4.1. Measured vs. modelled groundwater levels for the steady state model (RMS error = 1.82 m)

4.2 Transient model

Transient calibration was performed using parameter estimation software PEST (Doherty, 2010) to estimate hydraulic conductivity using the pilot point technique, by comparing modelled groundwater level to measured groundwater level at 142 observation wells across the study area for the period from 1970 to 2015. In order to reduce the number of PEST iterations required to calibrate the model, the SVD-assist method was used to generate 35 super-parameters from the 355 pilot point parameters. The overall calibration is good, with a sum of squared measured versus modelled residuals of 7587 (the objective function in PEST), an RMS error of 0.83 m, and scaled RMS of 1.06%. However the scaled RMS is scaled over the range of observation values, in this case 20.56–99.24 m AHD (range of 78.68 m). Only 25% of calibration targets have values greater than 60 m AHD (Figure 4.2), located in the east and north-eastern part of the model domain, very close to the eastern boundary. The main area of interest in the model is in the central part of the domain, where plantation forests are concentrated. In these areas of interest, groundwater levels span the range of 20–60 m AHD. The RMS calculated over this range is 0.79 m, and the scaled RMS is 1.98% (Figure 4.3).

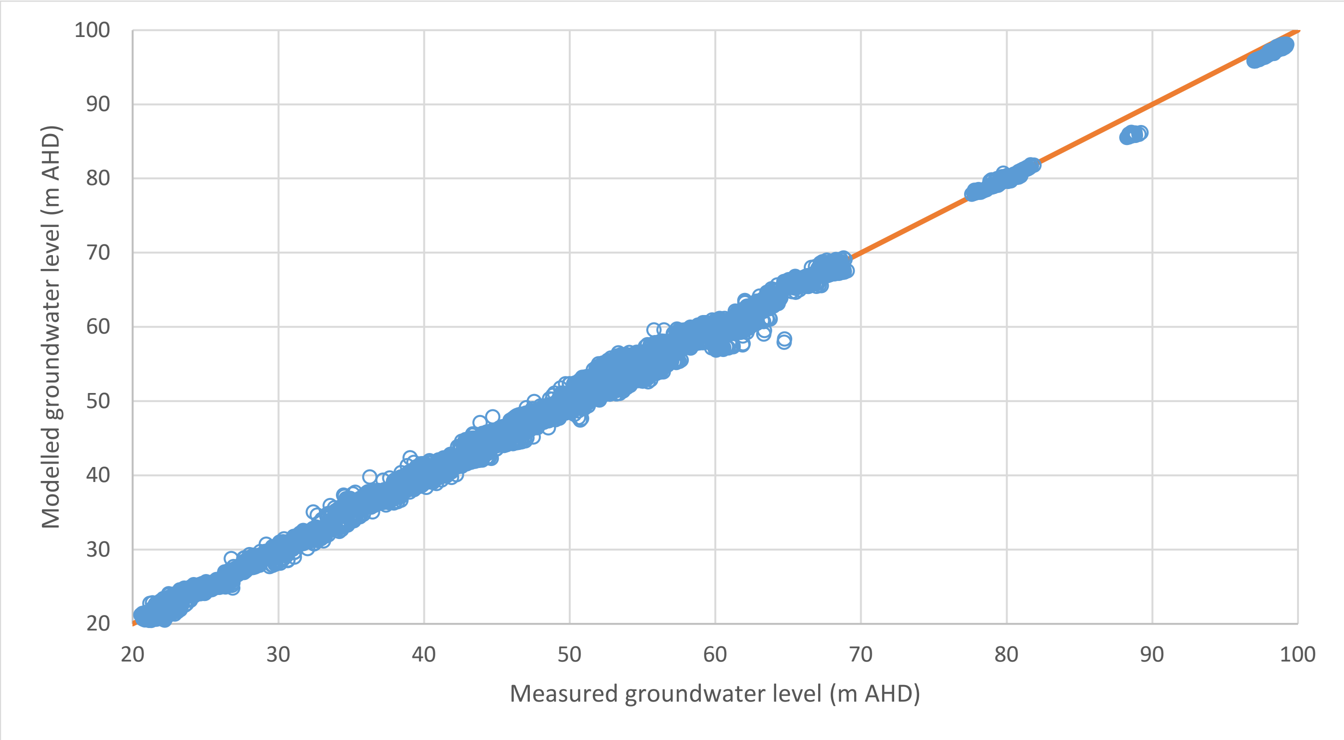


Figure 4.2. Measured versus modelled groundwater levels for the transient model (RMS = 0.83 m)

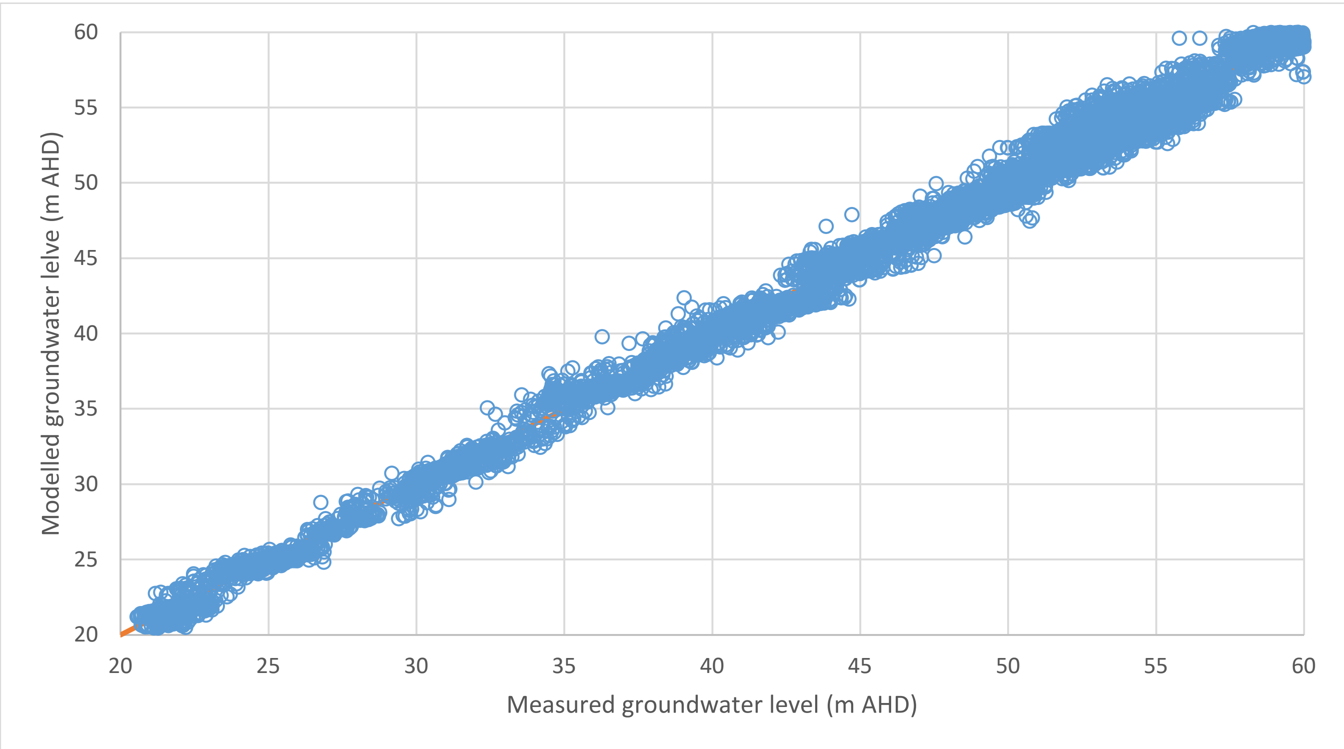


Figure 4.3. Measured versus modelled groundwater levels for the transient model for the range for levels between 20–60 m AHD (RMS = 0.79 m)

Individual fits to hydrographs vary in different parts of the model (see Appendix). In general the model fit is good in central parts of the model domain, which is the main area of interest for model scenarios. Key hydrographs (measured and modelled) in the Coles and Short management area generally demonstrate a good fit (Figures 4.3-4.8), with the declines in groundwater level from 2005 onwards matching well using the annual forest water use values adopted in the LLCWAP (SENRMB, 2013). The 9 m extinction depth for plantation extraction does not adversely impact on simulated groundwater levels. This is demonstrated at wells in which water levels have not declined below 6 m (e.g. SHT014, Figure 4.9), which show a good fit to measured data.

Worth noting is the match between measured and simulated groundwater levels in SHT012. While the match is good overall, the model slightly under-predicts groundwater level declines from 2005 to 2010. In the model SHT012 is surrounded by plantation forests established between 1999 and 2000, hence recharge is zero in this part of the domain from 2005 to 2010. Thus the modelled rate of plantation forest extraction may be an underestimate at this location for this period. Also in 2010–12, a rise in groundwater level is not well matched, meaning the extraction may be overestimated for this period. This may be due to the influence of the Bakers Range Drain, which is within 50 m of SHT012, as discussed in Section 2.5.2, with drain flow potentially recharging the aquifer locally. A better fit may be obtained by increasing forestry extraction and simulating enhanced recharge in this area, however it is not clear to what extent these changes should be up-scaled across the rest of the domain. As the purpose of the model is to assess the adopted models of forest water use on groundwater levels, the current fit at SHT012 is considered adequate, and no further calibration at this location is attempted. A more detailed analysis of spatial and temporal variability in recharge and plantation groundwater use would be useful though, particularly for assessing the impact of model assumptions on predictive scenarios.

Poorer fits are observed closer the model boundaries, particularly the eastern boundary and the management areas of Joanna, Comaum and Zone 3A (Appendix). Initial heads in this part of the domain appear too high and a period of model re-adjustment is observed from 1970–71 (Figures 4.14, 4.15). However this re-adjustment does not seem to influence overall model behaviour. This was tested by re-running the model with constant recharge and ET, and no pumping or forest extraction from 1970–2015. After re-adjustment in the first stress period, all observation showed static trends, with levels only fluctuating seasonally with recharge and ET. Thus this initial model adjustment is considered acceptable as it does not influence model performance in the hardwood forests areas of most interest. Given the level of irrigation development, there may be interest in further developing this model for scenario analysis specific to Zone 3A and other parts of the Border Designated Area. However based on the calibration here, it is recommended that the eastern boundary be extended into Victoria and the model re-calibrated should this be desired.

In parts of Zone 3A, modelled groundwater levels are too high (see CMM052, CMM074, CMM075, CMM076 and CMM078 in Appendix). Worth noting is that recharge rates for Zone 2A and Zone 3A given by Brown et al., (2006) of 95 and 100 mm/y were increased to 140 and 120 mm/y respectively as part of a review conducted by Latcham et al., (2007). However more recent modelling by Somaratne et al., (2016) simulated groundwater flow in the region using a recharge rate of 87 mm/y for Zone 3A. Given that groundwater levels are too high in parts of Zone 3A using the Brown et al. (2006) value of 100 mm/y in this study, recharge rates in Zone 3A should be investigated further as groundwater policy in the region is reviewed.

The model reproduces potentiometric surfaces for the aquifer reasonably well (Figure 4.16). The difference in modelled potentiometric surfaces from September 2000 to September 2015 shows the regional impact of groundwater level declines in relation to forests (Figure 4.17). Two drawdown cones of 5.5 m and 5 m are centred on around the Coles and Short management areas (respectively). The drawdown radiates outward from the main areas of plantation forest, with a 1 m drawdown contour extending as far east as Bool Lagoon. To investigate the influence of forests on this drawdown, the model was re-run with no forests present. Under this scenario, a rise in groundwater level of 0.3 m was observed between 2000 and 2015 at the eastern edge of Bool Lagoon. Elsewhere in the domain, small rises were likewise observed. Thus the drawdown depicted in Figure 4.17 can largely be attributed to plantation forests.

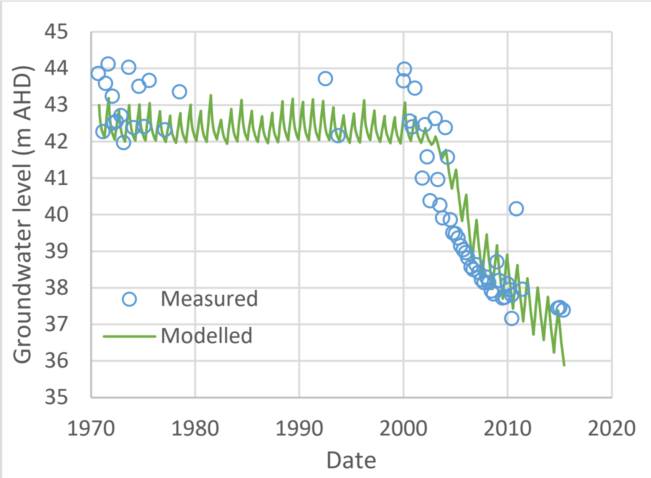


Figure 4.4. Measured and modelled groundwater levels in CLS002

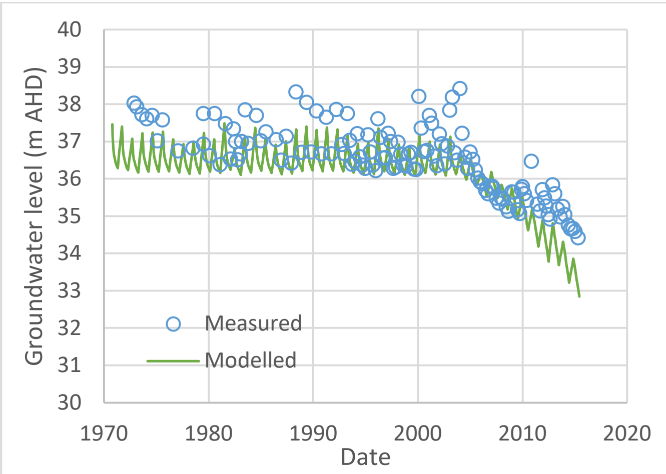


Figure 4.5. Measured and modelled groundwater levels in CLS004

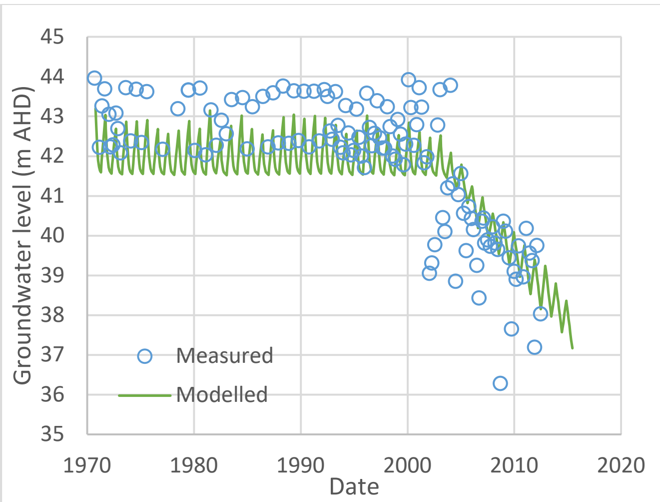


Figure 4.6. Measured and modelled groundwater levels in CLS006

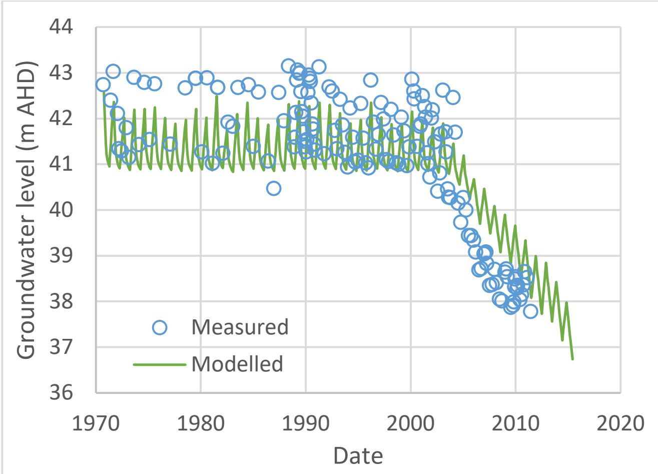


Figure 4.7. Measured and modelled groundwater levels in CLS009

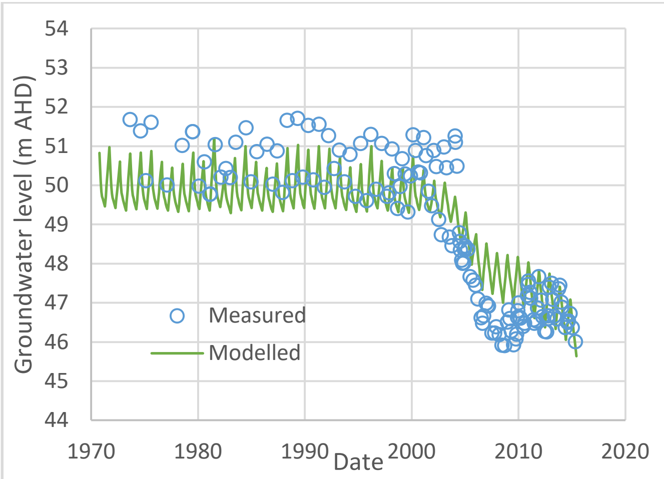


Figure 4.8. Measured and modelled groundwater levels in SHT012

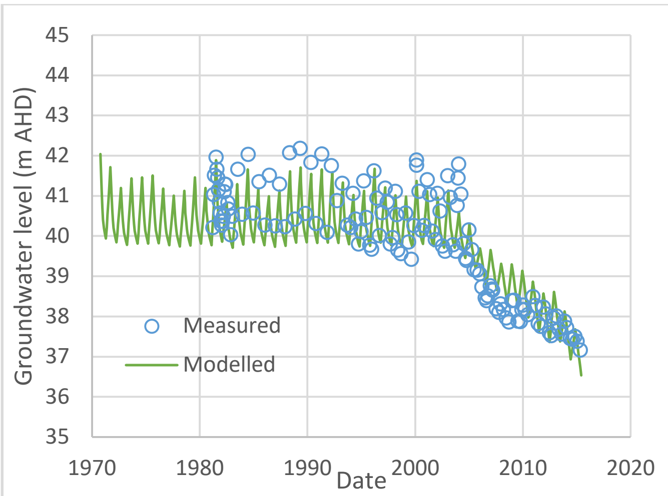


Figure 4.9. Measured and modelled groundwater levels in SHT014

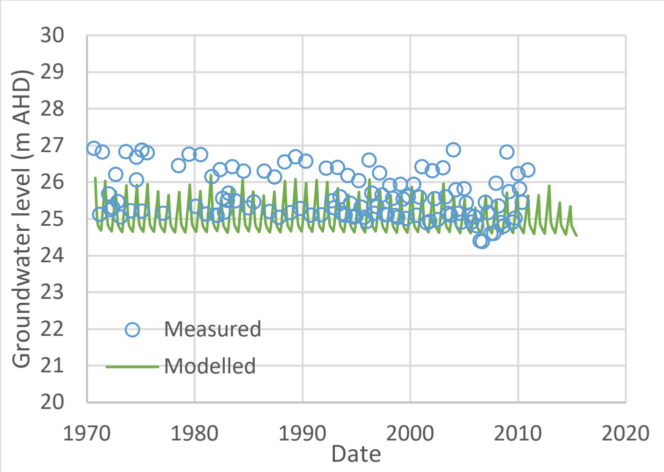


Figure 4.10. Measured and modelled groundwater level in FOX004

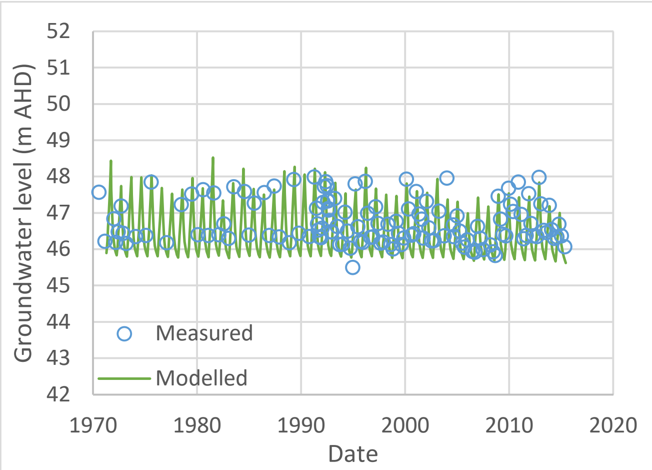


Figure 4.11. Measured and modelled groundwater level in ROB002

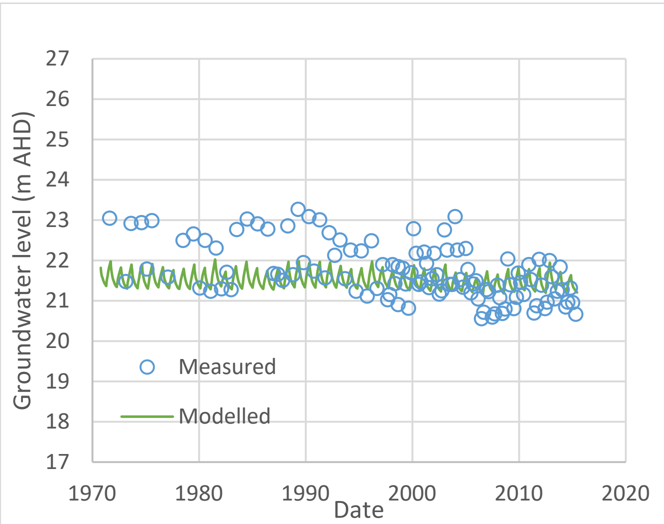


Figure 4.12. Measured and modelled groundwater level in KEN005

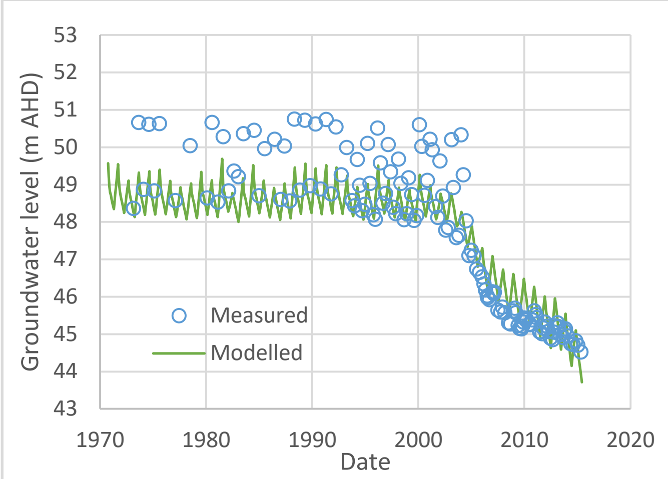


Figure 4.13. Measured and modelled groundwater level in MON016

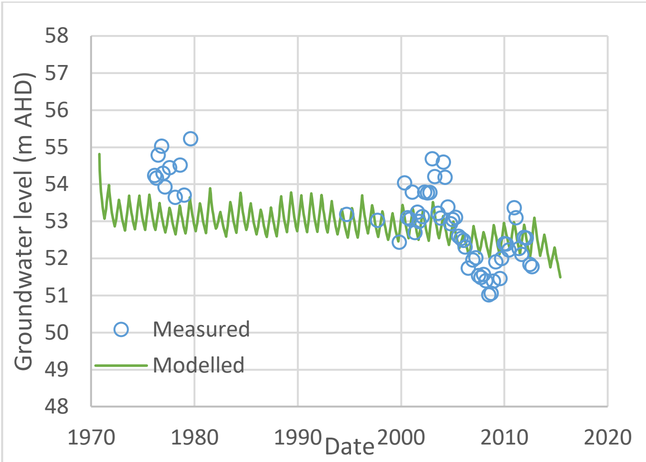


Figure 4.14. Measured and modelled groundwater level in CMM056

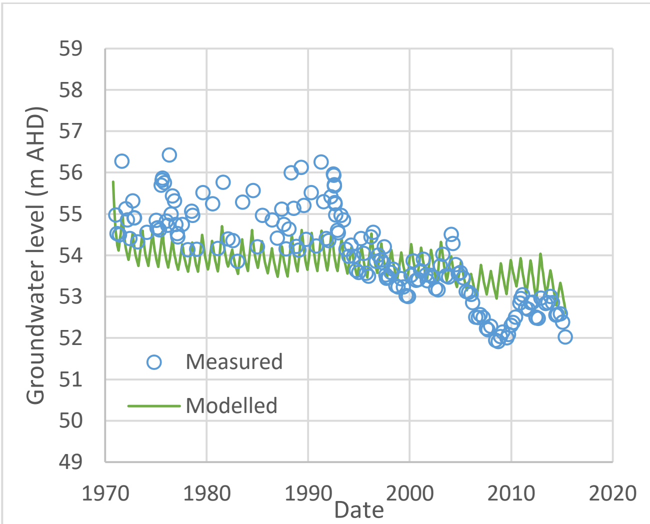
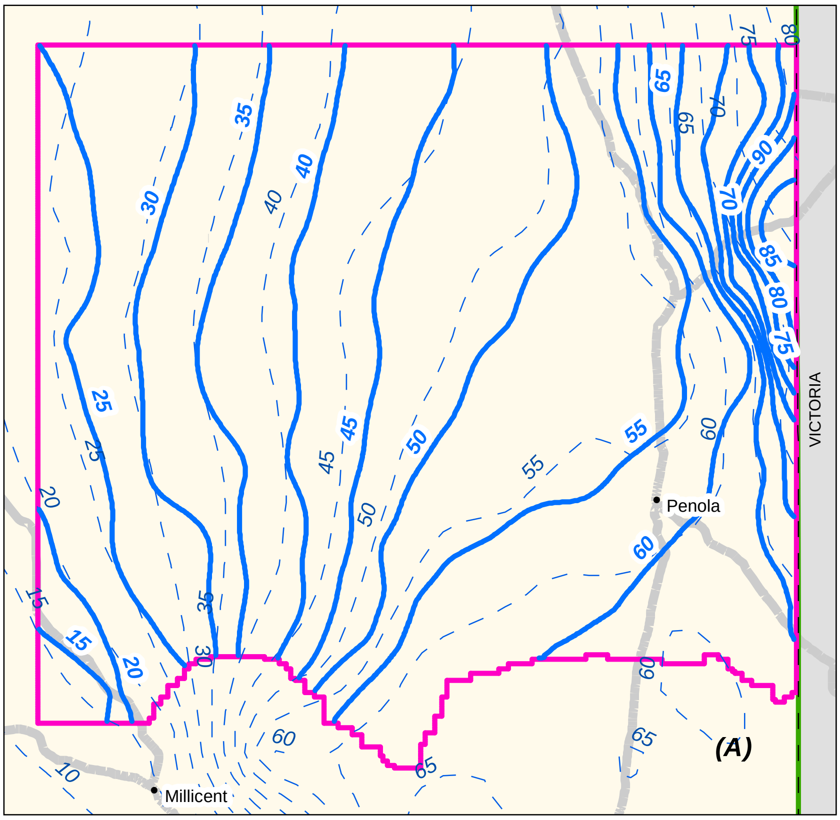


Figure 4.15. Measured and modelled groundwater level in CMM022

Figure 4.16 Measured and simulated potentiometric surfaces in 2000 and 2015



(A) Measured and modelled potentiometric surface at September 2000

Modelled September 2000

Measured September 2000

Study area

SA - Victoria border

Primary Road

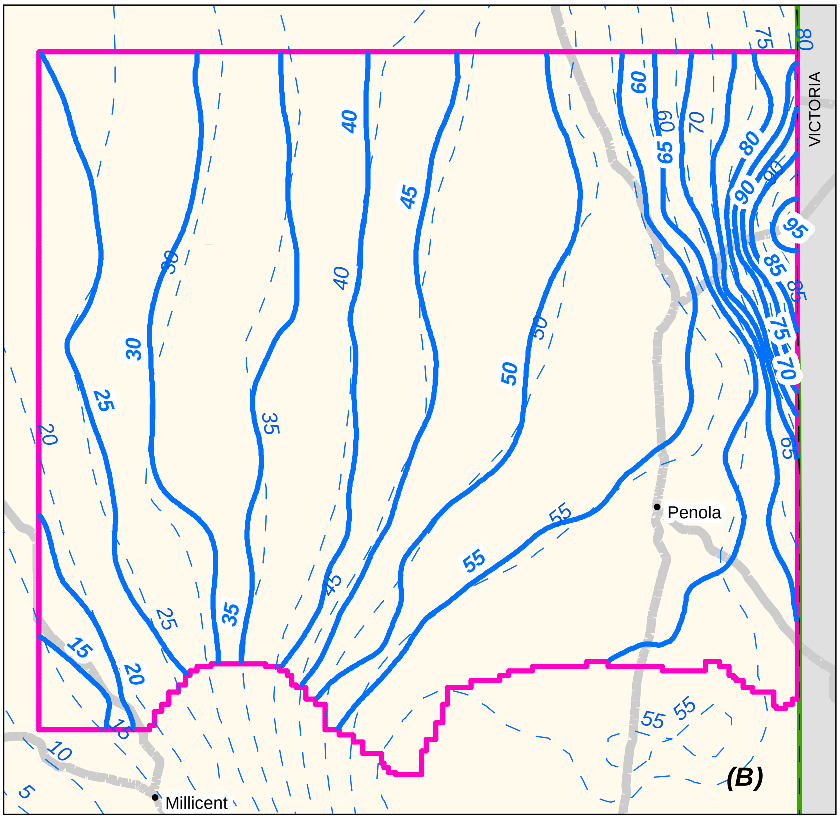
Lower Limestone Coast PWA

0 5 10 15

Kilometres

N

Government of South Australia
Department of Environment, Water and Natural Resources



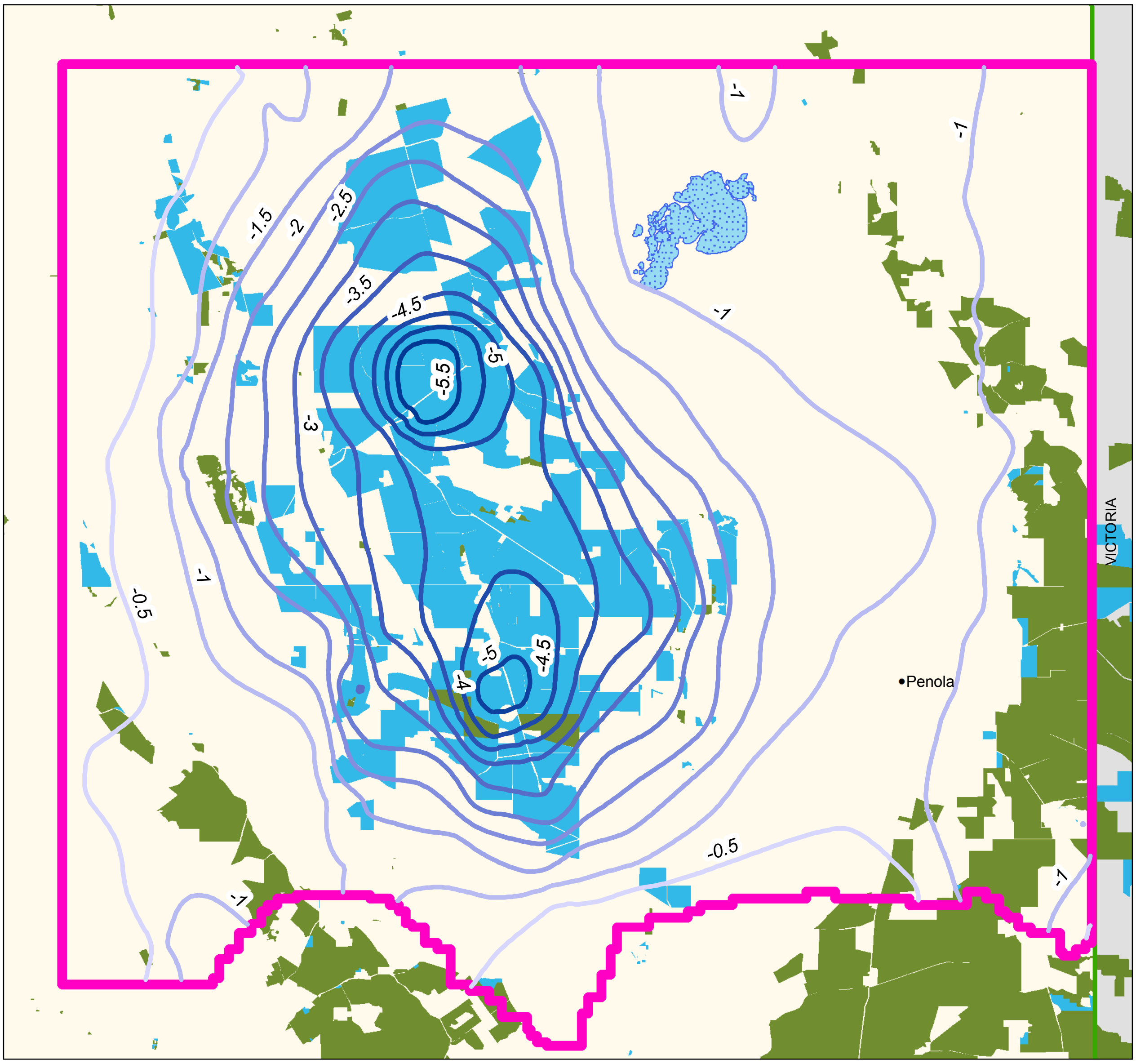
(B) Measured and modelled potentiometric surface at September 2015

Modelled September 2015

Measured September 2015

Produced by: Science, Monitoring and Knowledge Branch
Map Projection: Lambert Conformal Conic
Map Datum: Geocentric Datum of Australia 1994
Date: February 2017

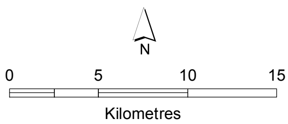
Figure 4.17 Simulated drawdown in the Tertiary Limestone aquifer from Sept 2000 to Sept 2015



Drawdown 2000-2015 (m)

-0.5
-1
-1.5
-2
-2.5
-3
-3.5
-4
-4.5
-5
-5.5

Study area
Bool Lagoon
Hardwood plantation
Softwood plantation



Produced by: Science, Monitoring and Knowledge Branch
Map Projection: Lambert Conformal Conic
Map Datum: Geocentric Datum of Australia 1994
Date: February 2017

Government of South Australia
Department of Environment, Water and Natural Resources

4.3 Modelled water balance

The simulated water balance for the entire domain shows that the model is dominated by inputs from seasonal recharge and outputs from evapotranspiration (ET) (Figure 4.18). Annual fluxes in GL/y are reported in Table 4.1 for selected years. There are some notable differences between simulated fluxes (Table 4.1) and estimated fluxes given in the preliminary water balance (Table 2.1). The preliminary water balance in Section 2 over-estimated the volume of regional inflow along the eastern boundary, and significantly underestimated the volume of ET and regional groundwater outflow. However the preliminary estimates of these volumes in Section 2 were based on a simple calculation (boundary inflow and outflow) and an estimate of ET being 40% of outflows based on a large regional groundwater model of the entire LLC (Morgan et al., 2015). In this model, the watertable is generally quite shallow (less than 2 m below ground level) hence outflows from ET are not surprisingly proportionally higher than other models for larger parts of the South East.

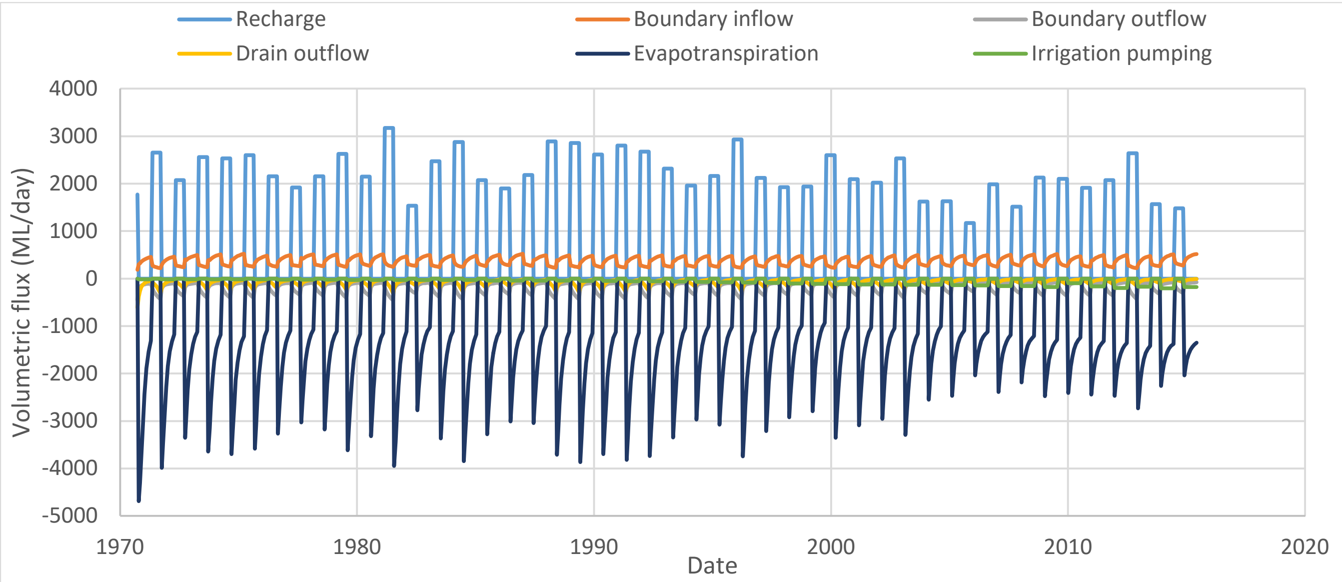


Figure 4.18. Simulated flux rates for the entire model domain

Table 4.1. Mass balances for the entire model domain

	Year					
	1980	**1990**	**2000**	**2005**	**2010**	**2015**
INFLOWS (GL)						
Recharge	329	400	398	250	321	226
Boundary inflow	141	131	125	135	134	145
OUTFLOWS (GL)						
ET	387	415	374	319	339	333
Drains	30	37	32	20	20	12
Pumping	6	11	25	30	20	37
Boundary outflow	67	75	72	61	67	58
Gain in storage (GL)	312	369	369	242	312	228
Loss from storage (GL)	332	376	349	287	302	297
Change in storage (GL)	-20	-7	20	-45	10	-69
Error (%)	-1.88E-02	-3.18E-02	-1.21E-02	-1.39E-02	-1.07E-02	-3.76E-03

For the entire model domain, peak ET rates appear to decrease with time (Figure 4.18). This is because declines in groundwater levels in parts of the model domain drive the watertable below the extinction depth for ET in pasture areas (2 m below ground level). However ET tends to persist at higher rates for longer after 2000, as forest ET is occurring from deeper watertables.

In forested areas however, ET generally increases in time in response to plantation water use. This is most evident in the Coles and Short management areas, which show large increases in ET from plantation groundwater use, and reductions in recharge from plantation interception after plantation establishment in the early 2000s (Table 4.2). These increases in ET and decreases in recharge are responsible for the modelled declines in groundwater level in these areas (Figures 4.4–4.9 and 4.17). Regional groundwater inflow also increases into the Coles and Short area over time, as declining groundwater levels increase the gradient, drawing in more inflow. Regional groundwater outflow decreases likewise as a result of the drawdown (Table 4.2).

Table 4.2. Mass balance in the Coles and Short management areas

	Year					
	1980	**1990**	**2000**	**2005**	**2010**	**2015**
INFLOWS (GL)						
Recharge	65	79	78	37	39	27
Regional inflow	68	68	68	74	89	96
OUTFLOWS (GL)						
ET	64	75	63	57	78	92
Drains	3	4	3	1	0	0
Pumping	0	1	1	1	2	1
Regional outflow	70	71	71	66	58	50
Gain in storage (GL)	60	73	73	38	51	44
Loss from storage (GL)	64	75	63	53	60	65
Change in storage (GL)	-4	-2	10	-15	-8	-21
Error (%)	-2.7E-03	-5.1E-03	-9.5E-01	-2.8E-05	-4.0E-01	2.9E-01

4.4 Groundwater discharge to drains

Figure 4.19 plots modelled groundwater discharge to drains 'upstream' of the Callendale regulator against measured flows at the Callendale regulator. Simulated groundwater discharge to drains is relatively small compared to measured historical drain flows. However gauged drain flows, particularly along Drain M, include intercepted surface water runoff and outflows from Bool Lagoon. In the model however the drains purely intercept shallow groundwater, hence the two data sets cannot be expected to correlate. It may be necessary to couple a surface water flow model with the groundwater flow model here in order to better constrain groundwater fluxes to drains in the study area, however this was considered beyond the scope of this study.

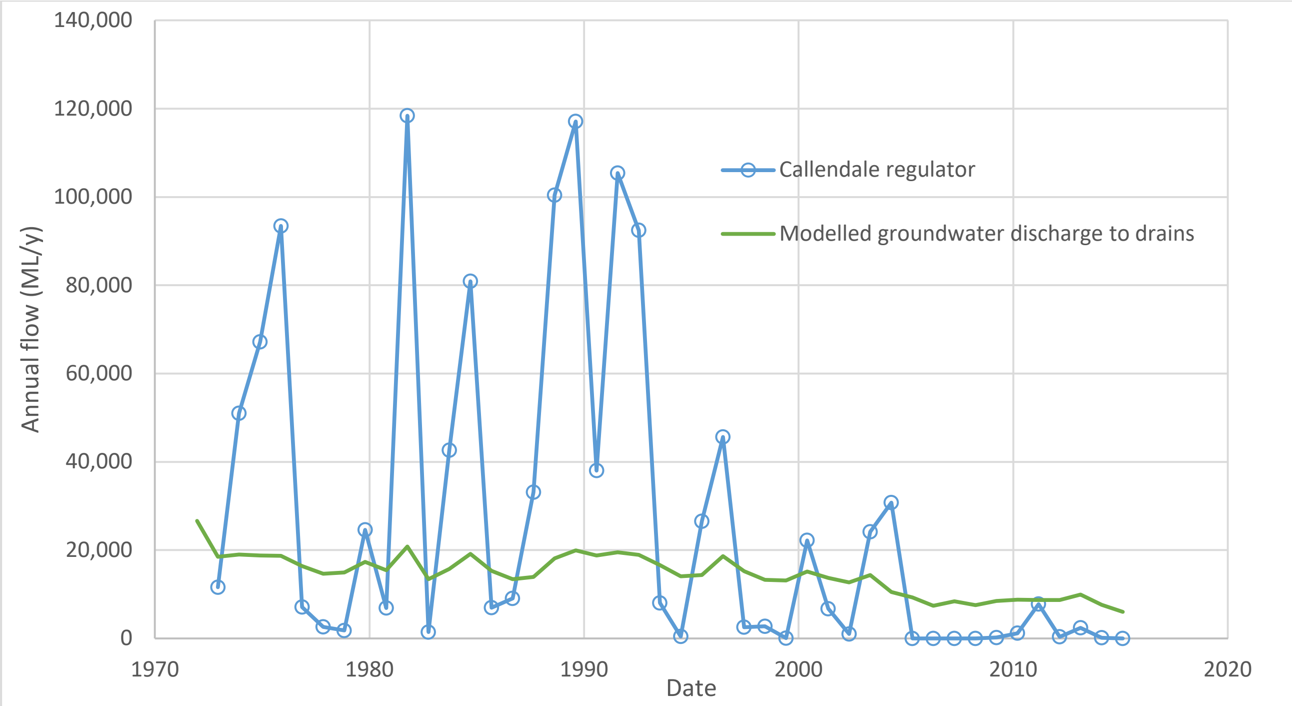


Figure 4.19. Modelled groundwater discharge to drains and measured drain flows at the Callendale regulator on Drain M

4.5 Bool Lagoon

The LLCWAP considers impacts on groundwater resource condition in terms of rates of groundwater level decline, with a trigger level for unacceptable declines set at a rate of 0.1 m/y over five years. However recent groundwater modelling studies in the South East have explored the use of resource condition limits (RCLs) as indicators of groundwater resource condition and management targets. For example, maintaining artesian pressure for productive use in the confined aquifer near Kingston (Wood and Pierce, 2015), and maintaining hydraulic gradient and saturation in more productive aquifer zones in Tatiara (Cranswick and Barnett, 2017; Li and Cranswick, 2017). The Bool Lagoon complex is Ramsar-listed wetland, and considered a wetland of high conservation value in the LLCWAP (SENRMB, 2013). Hence a potential future RCL for the study area could relate to maintaining groundwater discharge in Bool Lagoon.

As discussed in Section 3.2.11, Bool Lagoon is not modelled as a lake or drain, however changes in groundwater discharge to the Lagoon can be approximated based on an analysis of the mass balance, and volume of groundwater lost to ET in the lagoon area. Table 4.3 shows a decrease in both groundwater ET and discharge to Drain M (which runs through Bool Lagoon in the model domain) from 2000–15, which is likely related to groundwater level decline. While these trends are consistent with the whole domain in general (Table 4.1), Bool Lagoon shows increasing volumes of groundwater outflow from 2000–15, which is not consistent with the rest of the model (Table 4.1). The increase in groundwater outflow from the Bool Lagoon area is due to drawdown in the plantation forest areas (Figure 4.17) creating a steeper flow gradient and inducing greater volumes of groundwater flow to the west. Thus the drawdown from plantation forests is influencing the groundwater balance of Bool Lagoon. This was confirmed by re-running the model from 1970-2015 without any plantation forests present. Groundwater levels between 2000 and 2015 showed a slight rise at Bool Lagoon under these conditions, compared to the declines observed in Figure 4.17.

Table 4.3. Mass balance for the Bool Lagoon area for selected years

	Year					
	1980	1990	2000	2005	2010	2015
INFLOWS (GL)						
Groundwater inflow	48.9	49.1	49.0	48.4	48.9	48.3
OUTFLOWS (GL)						
ET	10.7	10.8	10.6	10.5	10.3	10.0
Drains	4.3	4.8	4.3	3.6	3.7	2.6
Groundwater outflow	34.0	33.6	33.9	34.4	34.8	36.0
Gain in storage (GL)	2.7	2.8	2.9	2.6	2.8	2.5
Loss from storage (GL)	2.8	2.9	2.6	2.6	2.8	2.8
Change in storage (GL)	-0.1	-0.1	0.2	-0.1	0.0	-0.3

5 Model sensitivity and limitations

5.1 Background

Uncertainty in model results can arise from multiple areas. For example conceptual uncertainty can be present in a groundwater model when there are gaps in our knowledge and/or data on the processes occurring and influencing groundwater flow. Structural uncertainty results from our inability to construct models that capture and simulate real-world processes exactly (Doherty and Hunt, 2010), which is inherent given that models are a simplification of reality. Parameter uncertainty results from our inability to characterise the heterogeneity of natural groundwater systems, and the difficulty in quantifying spatial and temporal variability in fluxes such as recharge or historical groundwater extraction. Consequently the groundwater model solution is non-unique and any number of realistic parameter combinations (e.g. hydraulic conductivity and recharge) could yield an equally well calibrated model. The following section discusses these areas of uncertainty in relation to the current study, and how some aspects of uncertainty may be reduced in the future with improved knowledge and data. In calibrating the model with PEST we have also assessed the sensitivity of calibration parameters on the model outputs. This sensitivity analysis is also presented below.

5.2 Key areas of uncertainty

5.2.1 Conceptual uncertainty

The conceptual groundwater flow model for the area is broadly understood. Rainfall recharge and regional inflow from Victoria are key drivers of groundwater flow through the study area, and discharge occurs through multiple processes (Figure 2.17). However there is some uncertainty in key processes. For example the magnitude of evapotranspiration (ET) from shallow watertables (less than 2 m below ground) is not well understood. One method to look at this in more detail would be to compare modelled ET outputs across the model domain with ET estimates derived from satellite based data. For example the 'CSIRO MODIS reflectance based scaling ET' (CMRSET, Guerschman et al., 2009) data could be used to further investigate the spatial and temporal variability of ET in the study area, and compared to modelled rates of groundwater ET (Purczel et al., 2016). However this is considered beyond the scope of the current project.

Surface water–groundwater interactions around drains is another area of conceptual uncertainty. Historically drains in the lower South East were constructed to reduce surface inundation, while drains in the upper South East were constructed to intercept saline groundwater (Harding, 2014). Given the shallow depth to groundwater in much of the western part of the study area, it is likely that drains in the study area intercepted groundwater historically. However groundwater level declines associated with plantation forests in parts of the study area, mean that groundwater levels are now below drain bottom elevations in some areas. Hence any drain flow may now be recharging groundwater locally at certain times (see Section 2.5.2). Capturing the spatial and temporal variability in surface water–groundwater interactions around drains is considered beyond the scope of this study. More work using environmental tracers and differential flow gauging would be needed to better inform the spatial and temporal variability of these fluxes (Harrington et al., 2012).

5.2.2 Structural uncertainty

Relevant to our model, the incorporation of ET in groundwater models has received recent attention (Doble and Crosbie, 2016). With regards to ET in plantation forest areas, this study has used policy-adopted transient ET rates to calibrate the model. These are implemented in the model using the ETS package, and simulating a depth-constant rate of ET. In reality the growth and transpiration rate of vegetation is likely to vary in time and space. This may be better incorporated in the model using a net recharge approach based on unsaturated zone modelling, as outlined by Doble et al. (2015). However the approach described by Doble et al. (2015) has not been calibrated, and further refinement would be required. This was considered beyond the scope of this study.

5.2.3 Parameter uncertainty – hydraulic conductivity

The calibration approach in this model involved matching simulated groundwater levels to measured groundwater levels by varying hydraulic conductivity values using the pilot point technique in PEST (Doherty, 2010). As discussed in Section 3.2.6, recharge is assumed to be equal to rates used in the LLCWAP (SENRMB, 2013; Brown et al., 2006), with some variability around yearly rainfall. As these recharge estimates are based on the watertable fluctuation method, and assume a specific yield of 0.1, specific yield has been left at 0.1. Furthermore simulated rates of plantation groundwater extraction are based on policy adopted rates, and groundwater extraction rates from 1970–2009 are based on average values of metered data from 2009–15.

The hydraulic conductivity values derived by the PEST calibration are within realistic bounds for the aquifer, and the spatial patterns in conductivity match spatial patterns in aquifer yield (Figure 3.16). However the results must be considered non-unique, and allowing other parameters such as recharge, ET and pumping to vary in PEST would no doubt yield an alternative hydraulic conductivity data set. To properly assess the impact of this uncertainty on model predictions, the model would need to be re-calibrated allowing all sensitive parameters to vary, and pursuing an approach such as the null-space Monte Carlo analysis to assess the impact of different parameter realisations on model predictions. Unfortunately this type of non-linear uncertainty analysis was beyond the scope of this project.

Nevertheless the model was subjected to a re-calibration in which both hydraulic conductivity at 355 pilot points and recharge rates were allowed to vary. Recharge was varied by applying a recharge multiplier to each of the winter stress periods. This resulted in a better model calibration (RMS = 0.34 m) and a different set of hydraulic conductivity values. Figure 5.1 plots the hydraulic conductivity values derived from the two calibrations at each pilot point against each other. Note that the upper bound in both calibrations was 250 m/d. While there is a lot of scatter, an overall correlation between K values in both calibrations is seen. In both cases more than 75% of the pilot point values are less than 125 m/d (Figure 5.2).

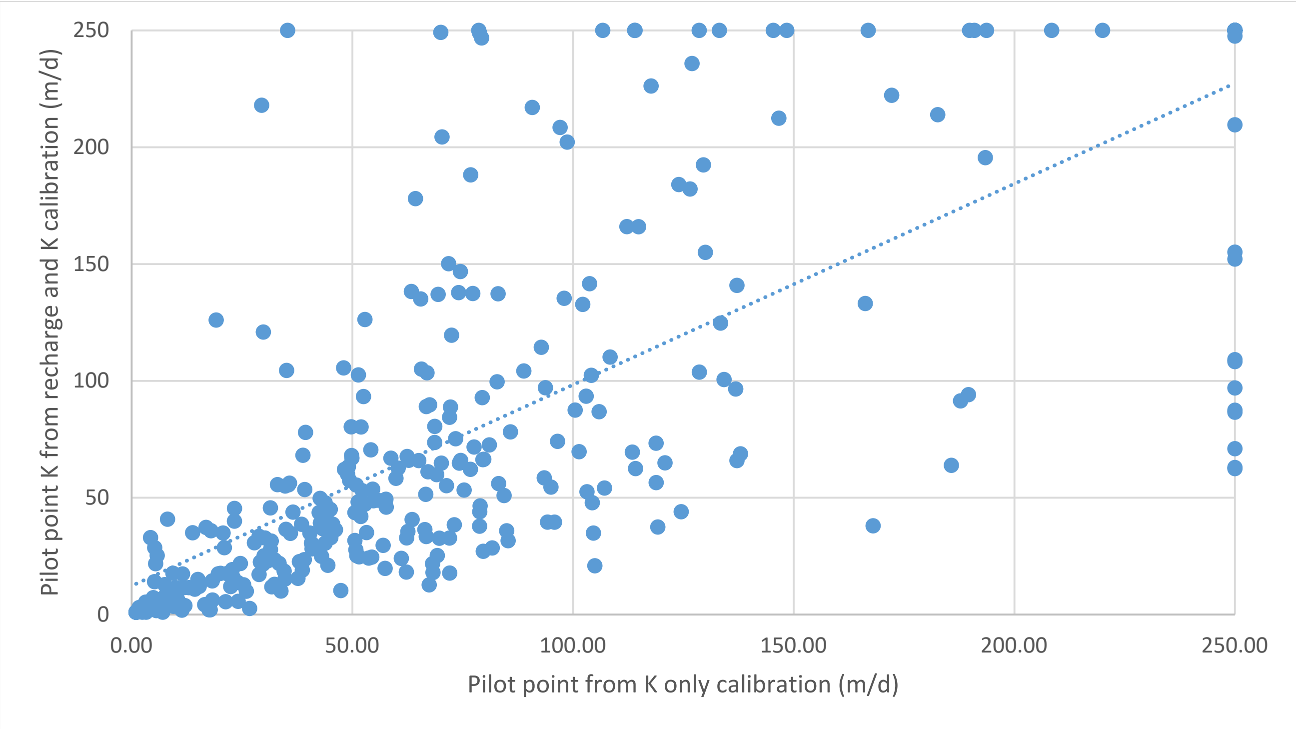


Figure 5.1. Pilot point hydraulic conductivity (K) values from two model calibrations – one calibration to head with only K varying, and one calibration to head allowing both K and recharge to vary

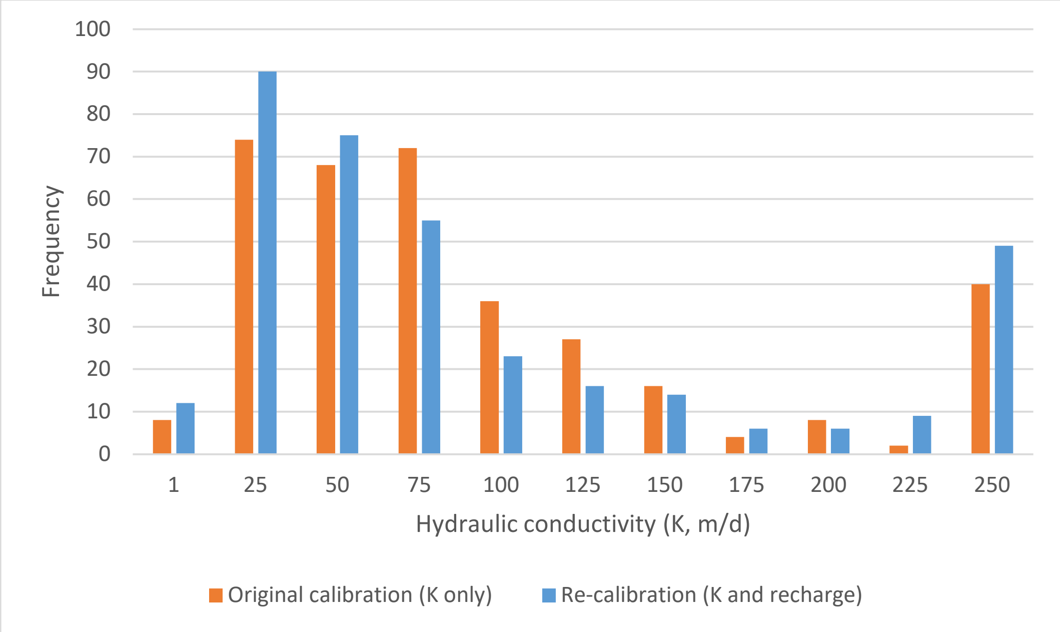
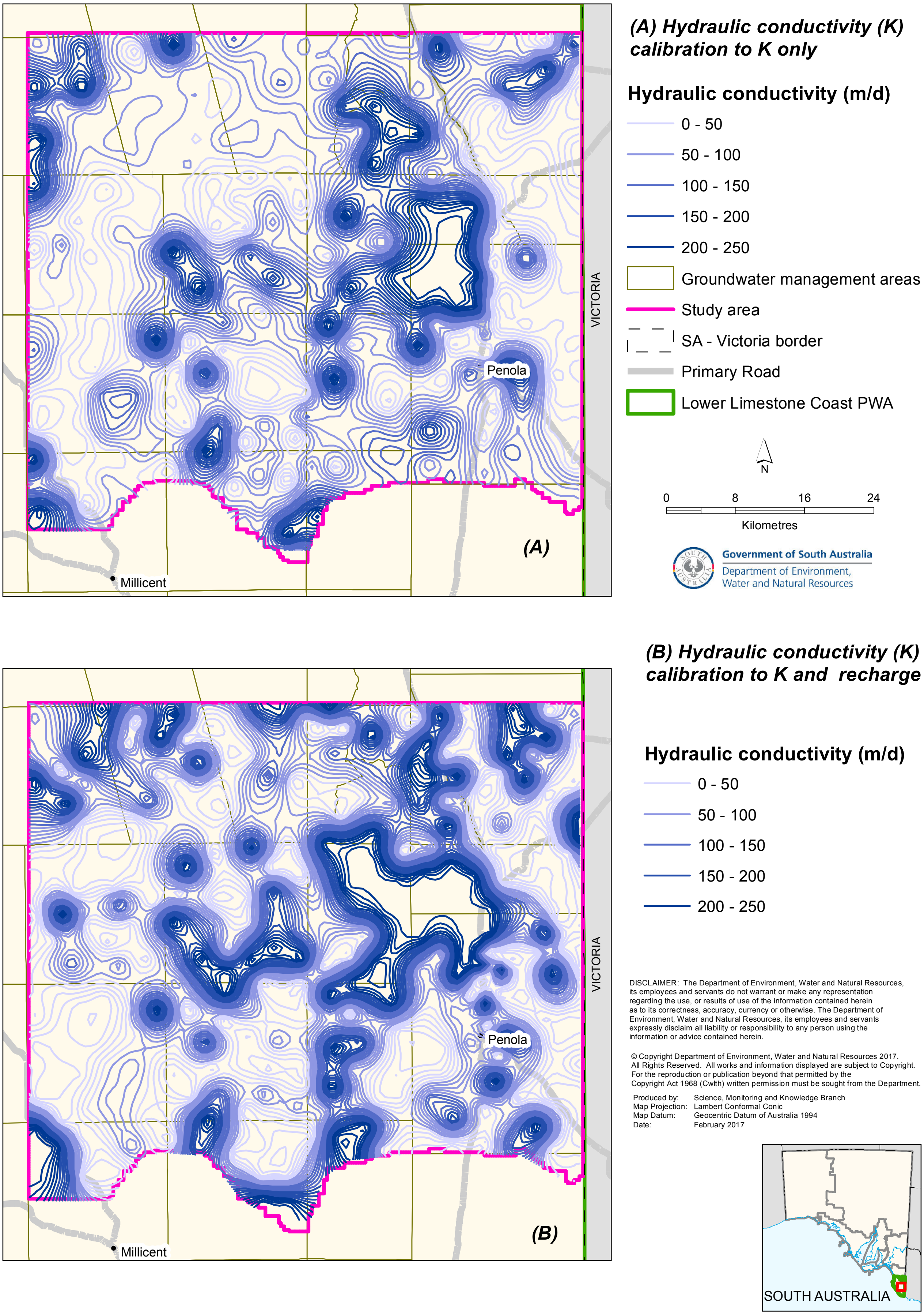


Figure 5.2. Distribution of hydraulic conductivity (K) values at pilot points from two model calibrations

The spatial distribution of hydraulic conductivity from the two calibrations show similar patterns. Both calibrations show zones of higher hydraulic conductivity in the north-west corner of Zone 3A, extending to the north and west (Figure 5.3). Both calibrations also show higher hydraulic conductivity in the southern half of the Coles management area. Thus the similar trend in aquifer properties given by both calibrations improves overall confidence in the hydraulic conductivity values used here.

Figure 5.3 Spatial distribution of hydraulic conductivity in the model from two calibrations



5.2.4 Parameter uncertainty - recharge

The second calibration does highlight uncertainty in the recharge rates used to calibrate the model. The recharge rates used to calibrate the model are based on values used in the LLCWAP (SENRMB, 2013; Brown et al., 2006), with annual variations based on rainfall. Recharge multipliers derived from the second calibration are generally greater than 1 up to the mid-1990s, suggesting that historical recharge may have been higher than estimated for this period. However from the mid-1990s onwards, recharge multipliers less than 1 are observed, suggesting that recharge may have been lower than estimated for this period. Care must be taken in interpreting these results, as the extent to which time-varying recharge can be constrained in a model is diminished when other parameters such as hydraulic conductivity are allowed to vary (Knowling and Werner, 2017). Furthermore, the low recharge multipliers from 2005 to 2008 may be influenced by groundwater levels in plantation forest areas which showed minimal fluctuations during this period, meaning that recharge interception in plantation areas may have been greater than estimated.

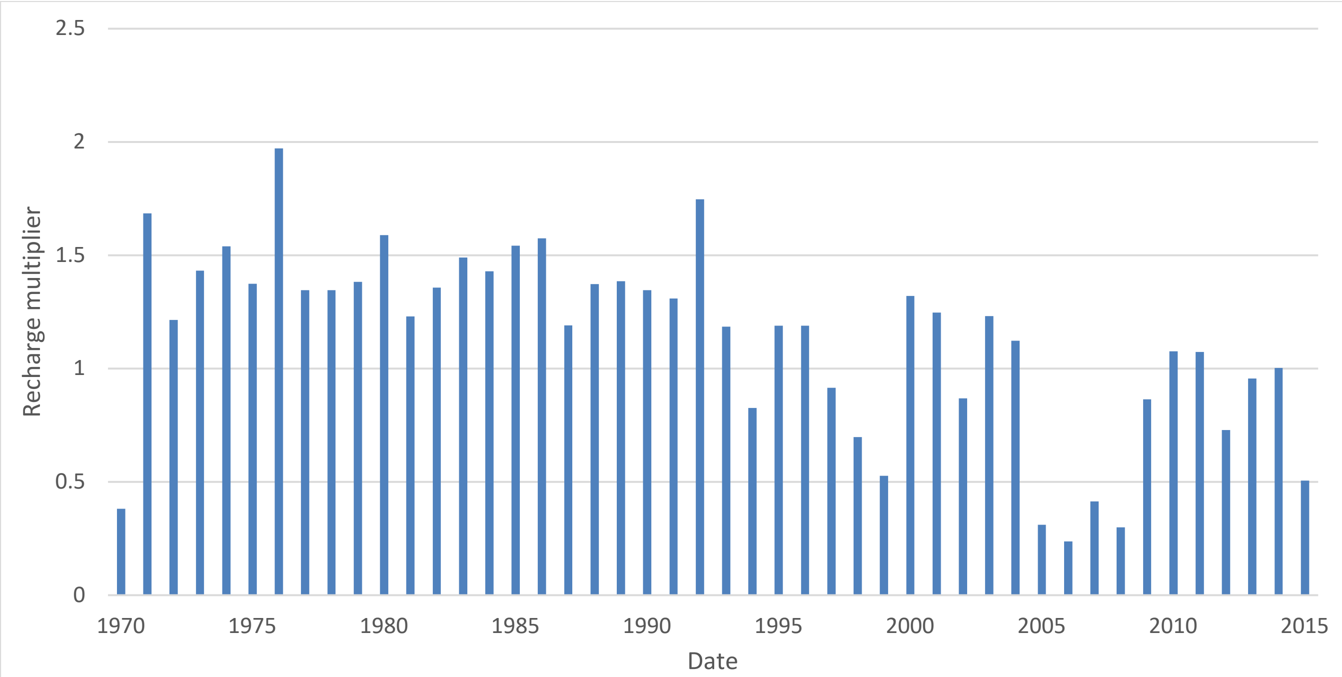


Figure 5.4. Recharge multipliers derived from an alternative model calibration

5.2.5 Parameter uncertainty - summary

A detailed non-linear parameter uncertainty analysis has not been performed here, as it was beyond the scope of the project. Thus the full impact of parameter uncertainty on model predictions cannot currently be quantified. However, the alternative model calibration in which both recharge and hydraulic conductivity were allowed to vary has yielded some insight into parameter uncertainty. Hydraulic conductivity values derived from the two calibrations show similar values and spatial distributions, both of which conform with available information on the spatial variability in aquifer properties. This provides additional confidence in the hydraulic conductivity values used in the model for future scenarios (Section 6). While uncertainty in recharge and plantation forest groundwater use has not been fully explored, the values used here are consistent with those used in groundwater management. Thus given the confidence in hydraulic conductivity values used in the model, the groundwater recharge and plantation water use parameters used in the model (and the LLCWAP) can be considered appropriate for simulating the impact of plantation forests on groundwater levels in the study area. Uncertainty in these parameters could be further explored in future as the LLCWAP is reviewed.

5.3 Parameter sensitivity

One of the key processes performed by PEST is calculation of the Jacobian matrix, which quantifies the sensitivity of all model outputs to each adjustable parameter (Doherty and Hunt, 2010). Figure 5.5 plots the composite sensitivities derived from the Jacobian matrix, for the calibration in which both recharge and K were adjusted. Note the vertical scale lacks resolution given the number of parameters (355 K pilot points and 46 recharge multipliers). With a few exceptions, Figure 5.5 shows that model results are generally more sensitive to recharge multipliers (denoted RM1970-RM2015) than to hydraulic conductivity (K1-355). The model used for scenario analysis here adopts recharge rates based on those used in the LLCWAP, hence it is recommended that future work involve an uncertainty analysis to quantify the effect of uncertainty in recharge rates on model predictions.

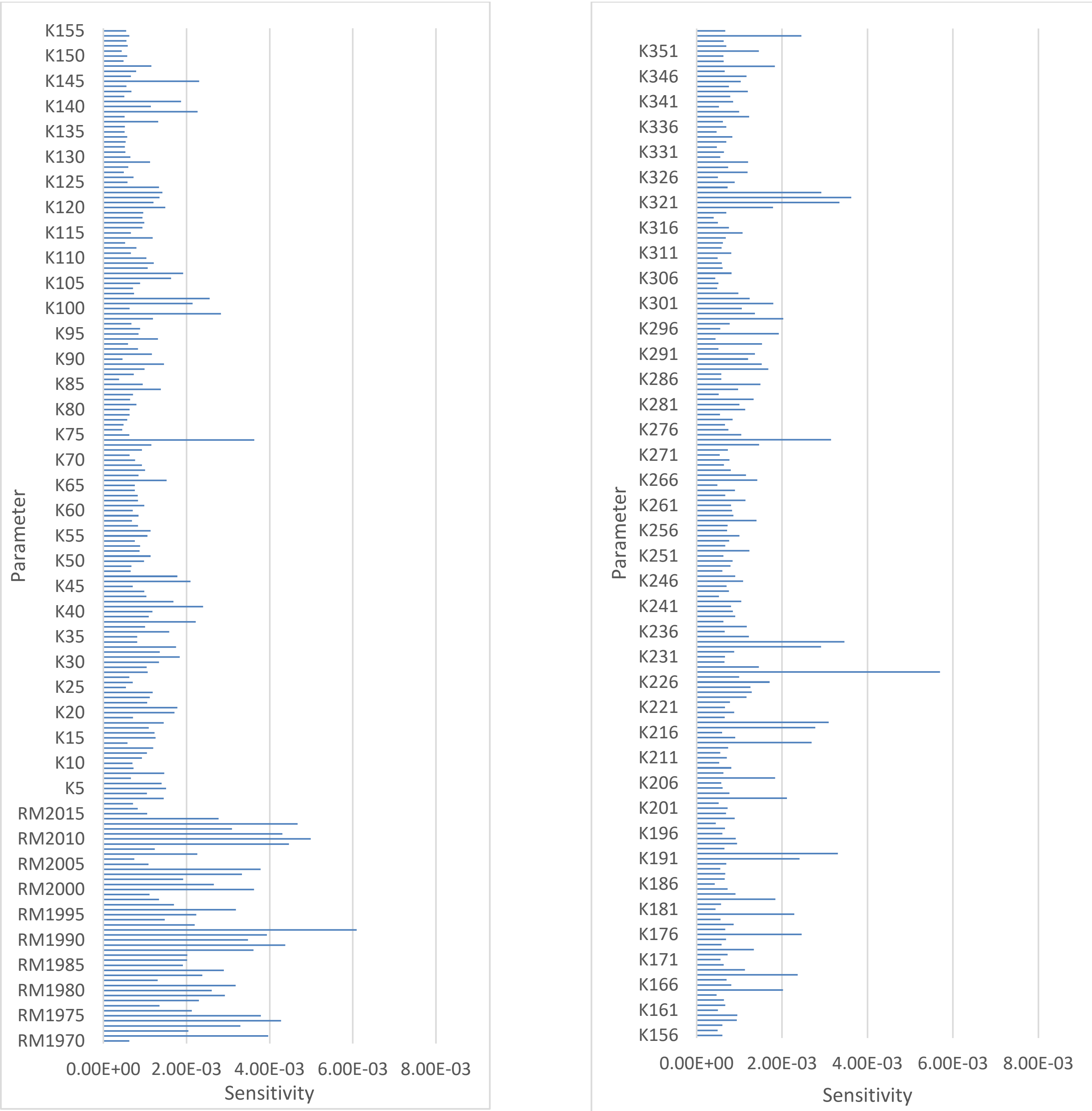


Figure 5.5. Parameter sensitivity derived from the Jacobian matrix in PEST

A separate, more traditional sensitivity analysis was performed for specific yield (Sy), by varying Sy between 0.05 and 0.2, and observing the effect on model calibration. The range was chosen based on the range of values used in groundwater models for other parts of the LLC (Wallis, 2008; Aquaterra, 2010a; Morgan et al., 2015). Figure 5.5 shows that variation in Sy has minimal impact on the overall model calibration. Slightly better calibration statistics maybe achieved with a lower Sy of 0.09, however the difference is very small, and thus the value used in the model (0.1) can be considered appropriate. It is likely that Sy would show greater sensitivity if a pilot point approach were used to calibrate this parameter, rather than having one value for the entire domain. However this approach was considered beyond the scope of this study.

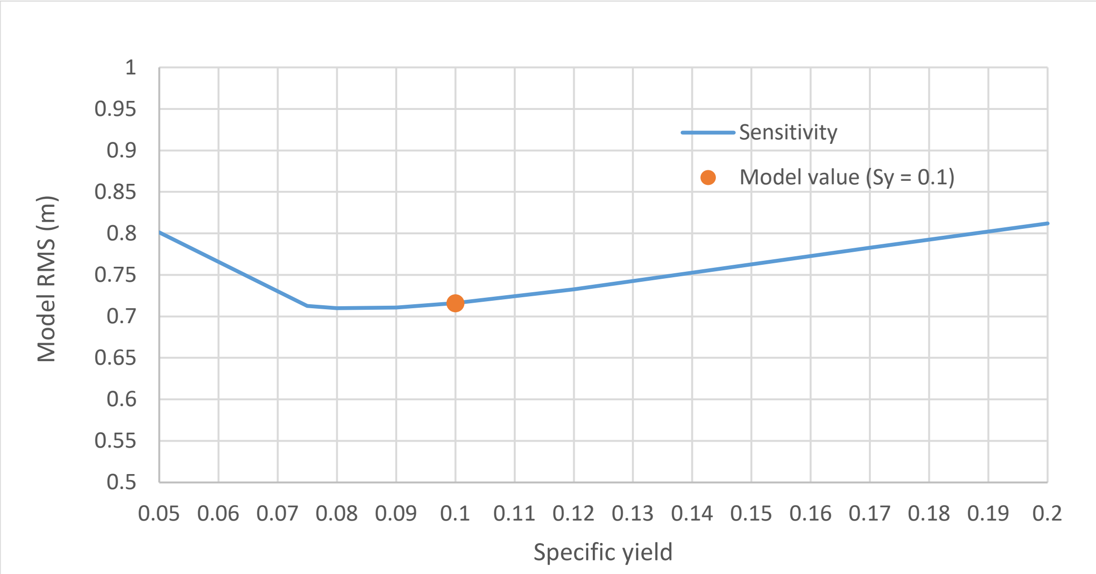


Figure 5.6. Parameter sensitivity derived from the Jacobian matrix in PEST

5.4 Model capability and limitations

In order to manage expectations on what the model can and cannot be used for, it is important to define its capabilities and limitations. The purpose of the model was to assess the forest water accounting models used in the LLCWAP, and run scenarios of future groundwater management. Based on the calibration and analysis of parameter uncertainty (Section 5.2.5) the models capabilities can be summarized as follows:

- The model is capable of simulating groundwater levels in areas of plantation forest, particularly the management areas of Coles, Short, Monbulla and Spence. Declines in groundwater level are accurately simulated using the rates of recharge, recharge interception and plantation groundwater use specified in the LLCWAP.
- The model is capable of assessing changes in the groundwater balance in areas of hardwood plantation over time, particularly increases in ET and decreases in recharge.
- The model is capable of simulating groundwater level trends reasonably well across the entire study area, and can thus be used to assess changes in the groundwater balance over time.
- The model is capable of simulating groundwater levels around Bool Lagoon, and providing estimates of temporal changes in the groundwater balance around Bool Lagoon in terms of shallow groundwater ET and lateral flow.
- The model is suitable for running scenarios to assess the impacts of future changes in land use on groundwater levels and water balances in the areas of hardwood forests and around Bool Lagoon.

- The model is suitable for undertaking an analysis to quantify the effects of uncertainty on resource management options and policies. Such and analysis could further be used to investigate alternative policy arrangements for the LLCWAP.

The models capabilities are tailored to its purpose, hence the models limitations also stem from its purpose, as well as some of the uncertainties discussed in Section 5.2. Specifically:

- The model is not well calibrated close to the SA–Victoria border in the Zone 3A management area, hence the model should not be used for detailed scenario analysis in areas close to the border. While the model broadly simulates groundwater levels in Zone 3A, the model domain should be extended further east if detailed scenario analysis in this area is required.
- The model is not capable of accurately simulating surface water–groundwater interactions around drains in the study area. There are currently no reliable estimates of groundwater discharge to drains with which to constrain this aspect of the model, hence more detailed field investigations would likely be required in order to improve this part of the model.
- The model considers only flow, therefore no conclusions regarding the impact of plantation forests and associated clearance on groundwater salinity can be made. To do this solute transport would need to be added to the model, and the transport model would need to be calibrated. Detailed monitoring and field investigations would be required to provide this data for calibration.
- The model considers only flow in the unconfined Tertiary Limestone Aquifer, hence it assumes no interactions with the underlying Tertiary Confined Sand Aquifer. There is no evidence of significant interactions with the TCSA in this area, hence this is not thought to have any significant impact on the model in its current use.

6 Model scenarios

6.1 Background

The Lower Limestone Coast Water Allocation Plan sets target management limits (TMLs) for groundwater allocation from the unconfined Tertiary Limestone Aquifer in the South East, based on estimates of recharge. In the Coles and Short management areas, the cumulative impact of recharge interception and direct extraction by plantation forest exceeds the TML volume by nearly 50% (SENRMB, 2013). Hence the LLCWAP proposes reductions to allocations attached to forest water licences to address over-allocation in these management areas. Specifically, the LLCWAP states that *"as part or all of compartments are clearfelled, the volume of allocation equivalent to the clearfelled area should be reduced from the forest water licence, until such time as the total volume of reductions to allocations attached to forest water licences for that management area is achieved."*

The LLCWAP sets out the principles for reductions in allocations so as not to exceed TMLs. However the exact details of how the plantation forest estate in the study area will be managed in the future, and how allocation reductions will be achieved, is currently unclear. Hence there is a need to explore different potential future land use change scenarios.

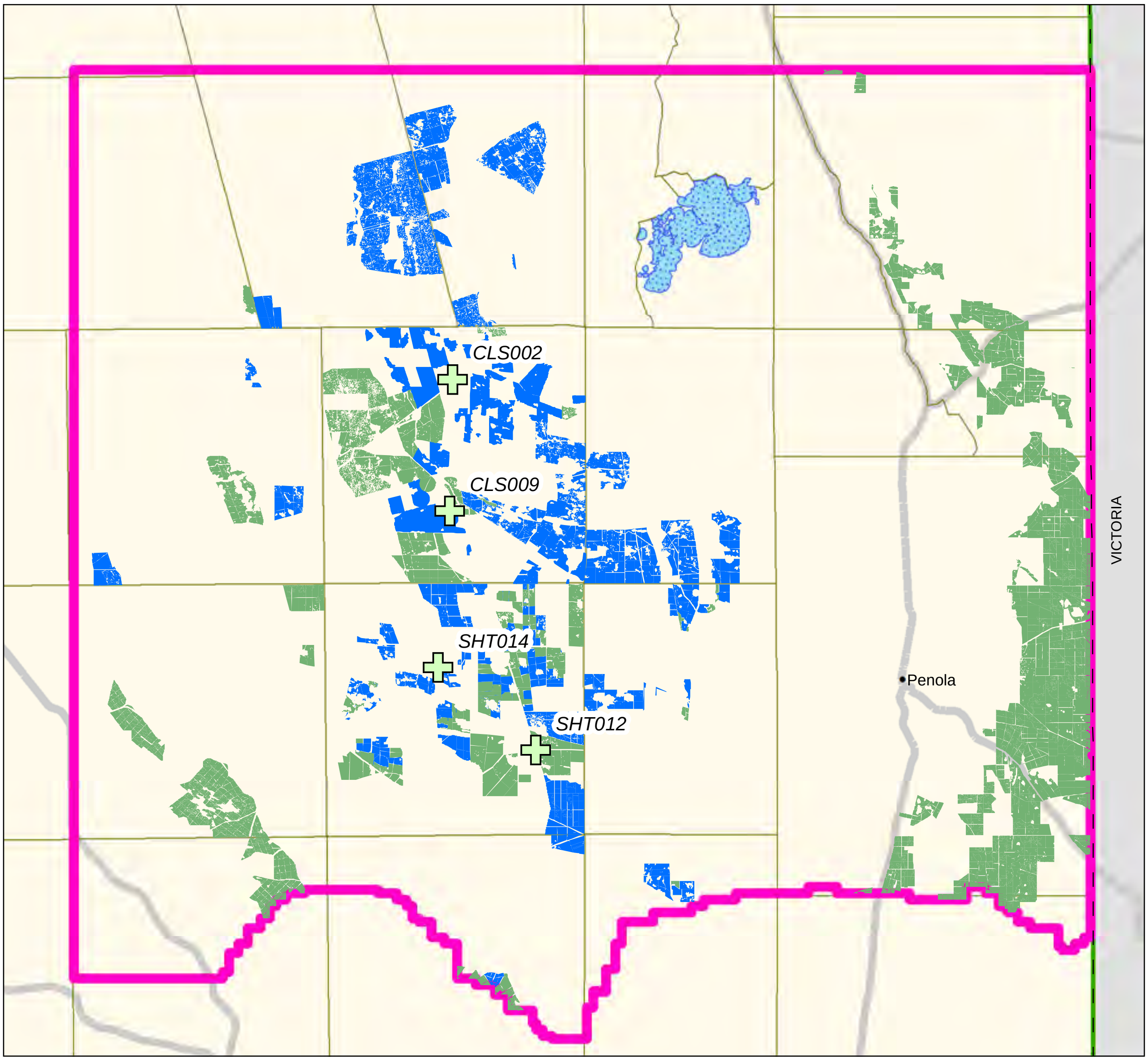
At the time of this study (and subsequent to the model being calibrated), only preliminary information on plantation forest clearance in Coles and Short was available. This preliminary information, taken from water licencing information collected by DEWNR, indicated that 3550 hectares (35.5 km^2) was cleared in 2015 in Coles and Short, while 5100 ha (51 km^2) was cleared in 2016. However it is thought that 92% of the forest cleared in 2015 was 'coppiced' – cleared to the tree stump and the forest allowed to regenerate from there – rather than trees being replanted. In 2016 it is thought that 51% of the clearance was coppiced, while 49% was cleared.

This preliminary clearance data is included in the model scenarios, such that scenarios start in 2015. The model scenarios then consider the following changes in land use:

1. A change of land use in Coles and Short such that by 2022, 40% of the hardwood plantation area becomes pasture, 30% becomes softwood plantation, and 30% remains as hardwood. Long term average recharge is assumed to occur, and the scenario runs from 2015–40 (Figure 6.1).
2. A change of land use in Coles and Short such that by 2022, 20% of the hardwood plantation area becomes pasture, 40% becomes softwood plantation, and 40% remains as hardwood. Long term average recharge is assumed to occur, and the scenario runs from 2015–40 (Figure 6.2).
3. Scenario 1 under a reduced rainfall climate.
4. Scenario 2 under a reduced rainfall climate.
5. Scenario 1 but with increased recharge in fallow years of plantation, based on information reported in Benyon and Doody (2009).

In all scenarios, irrigation pumping and areas of native vegetation remain constant from 2015–40. It should be noted that the model calibrated by varying hydraulic conductivity only (not recharge) is used here to run scenarios. This is because the recharge rates, specific yield and forest ET rates are consistent with values currently used in policy, and therefore suitable for future predictions. The model is suitable for further scenario investigations which may consider alternative policy settings, recharge rates and rates of plantation water use, however this has not been conducted as part of this study.

Figure 6.1 Land use distribution at 2040 used in Scenarios 1, 3 and 5



Study area

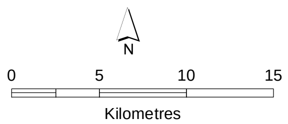
Observation wells

Softwood

Hardwood

Bool Lagoon

SA - Victoria Border



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Date: February 2017

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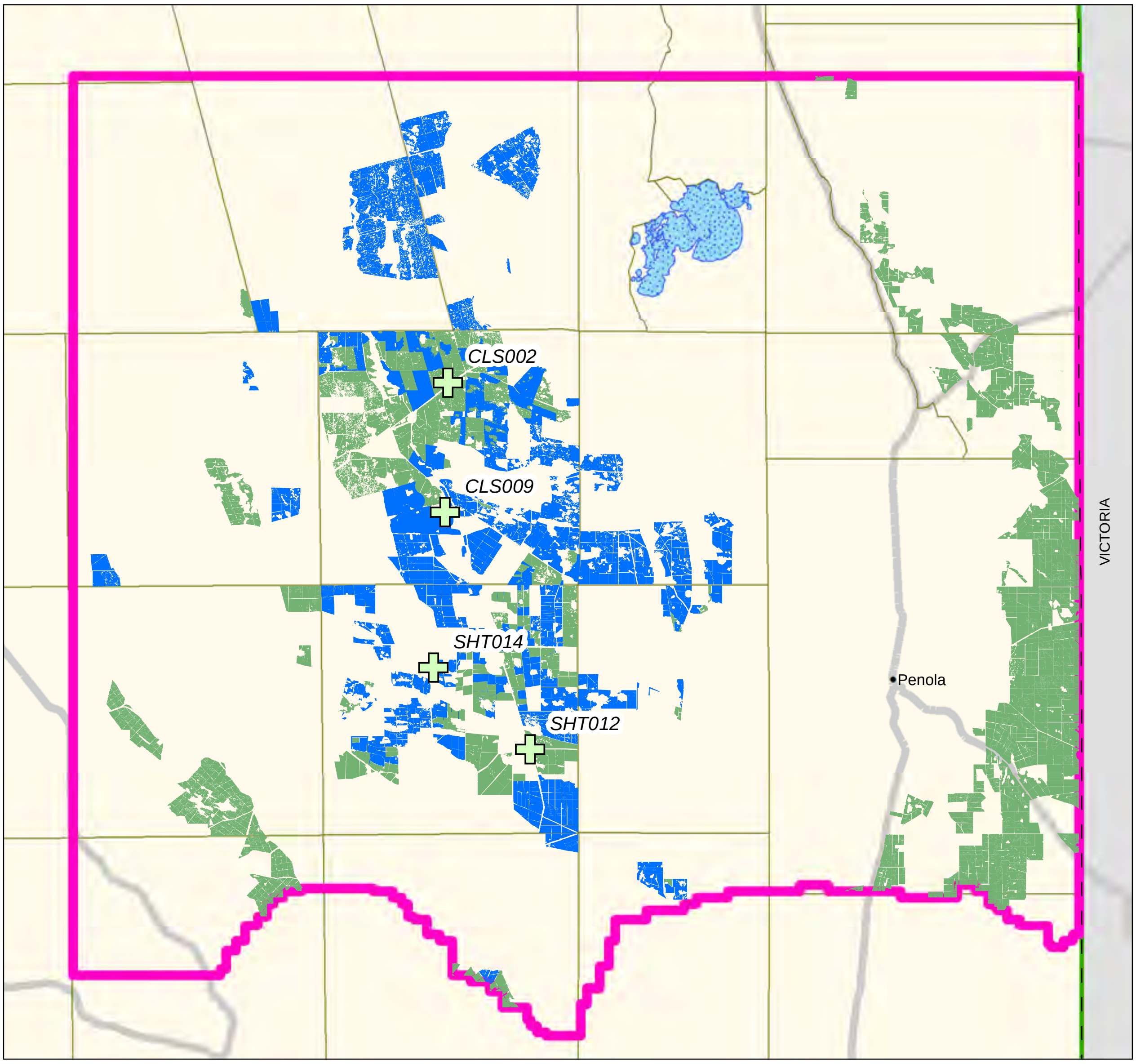
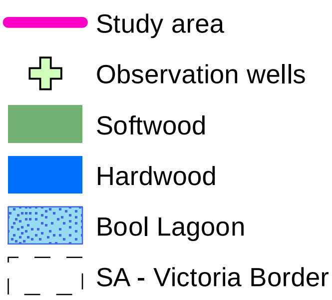
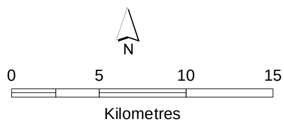
Figure 6.2 Land use distribution at 2040 used in Scenarios 2 and 4







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6.2 Future climate scenarios background

Previous work on groundwater resources in South Australia has investigated the impact of climate change on groundwater recharge through the use of unsaturated zone modelling (Green et al., 2011; 2012). For example, Green et al. (2011) reported that a 5% decrease in rainfall was likely to result in a 23% reduction in groundwater recharge to the fractured rock aquifers in the Clare Valley. However in the sedimentary basins on the Eyre Peninsula, a 5% reduction in rainfall was likely to result in a 15% reduction in groundwater recharge (i.e. a scaling factor of three, Green et al., 2012). The Australian Groundwater Modelling Guidelines state that using scaling factors derived from such modelling are suitable for simulating climate change impacts on recharge in groundwater models (Barnett et al., 2012). This approach was recently adopted by Li and Cranswick (2016) who used a scaling factor of three to scale the impacts of climate change on groundwater recharge in the Barossa Valley.

Charles and Fu (2015) have recently reported on the potential impacts of climate change on rainfall in the South East of South Australia. The authors summarised the output of six Global Climate Models (GCMs) and two representative concentration pathways (RCPs) which describe an intermediate greenhouse gas emission scenario (RCP 4.5) and a high emission scenario (RCP 8.5). The authors report decreases in annual rainfall in the South East of 3.5% and 4.4% by 2030 for Scenarios RCP 4.5 and 8.5 respectively. By 2050, there are larger decreases in rainfall of 5.4% to 6.6% for Scenarios RCP 4.5 and 8.5 respectively. However the authors report that changes in spring and summer rainfall are likely to be greater than changes in autumn and winter rainfall.

For Scenarios 2 and 4, it is assumed that mean annual rainfall decreases by 5%, falling in between the projections for 2030 and 2050. The 5% reduction is constant from 2015–40. It is assumed this results in a 15% impact on groundwater recharge (a scaling factor of three). Thus groundwater recharge in each management area is 15% lower than the long term averages used in Scenarios 1 and 3. The potential influence of changes in rainfall seasonality is not necessarily captured in this approach. Further work is warranted here, including using the results of climate modelling studies along with the application of unsaturated zone models for the study area. This would also allow for evaluation of the potential influence of increased emission scenarios on vegetation feedbacks, including increased ET and potentially increased groundwater use.

6.3 Scenario 1

Scenario 1 considered the clearance of plantation forest in Coles and Short from 2015 onwards, such that by 2022, 40% of the plantation estate had been cleared and returned to pasture, 32% is converted to softwood plantations, and 28% remains as hardwood. The slight variations in change in softwood and hardwood (32% and 28% as opposed to 30% and 30%) are due to the way in which land use data is dealt with in the model. That is for ease of implementation, all forests planted in a particular year are grouped as one land use area. The spatial and temporal distribution of clearance is related to plantation age such that older plantations are cleared first. Table 6.1 shows the areas cleared each year, while Figure 6.1 shows the spatial distribution of land use by 2022.

Table 6.1. Areas of clearance in Coles and Short from 2015 to 2022 in Scenario 1

	Areas (hectares)			
Year	**Total cleared**	**Converted to pasture**	**Remaining hardwood**	**Converted to softwood**
2015	3,609	530	3,079	0
2016	5,133	2,117	3,016	0
2017	6,947	3,372	0	3,575
2018	2,448	1,370	0	1,078
2019	3,018	0	548	2,470
2020	2,926	1,712	983	231
2021	760	0	760	0
2022	1,537	1,537	0	0
Total (hectares)	26,378	10,638	8,386	7,354
Percent	100	40	32	28

The areas remaining as hardwood are initially coppiced – allowed to regenerate from the stump and grown for a further eight years. After this remaining hardwood plantations are clear felled and managed on the 10 year growth cycle with a replant for the next rotation, as per the forest water accounting model characterised average plantation. Recharge is assumed to be constant each year at the management area rates given by Brown et al. (2006), and the scenario runs from 2015 to September 2040. Based on the areas of forest considered in the model, the 40% pasture represents 10,638 hectares in the Coles and Short management areas. In terms of direct extraction, assuming plantation extraction can be calculated using an 'annualised' rate of 1.82 ML/ha/y over an 11 year plantation cycle (10 years of growth and one year of post-harvest clean up, see Section 2.4.5), the conversion of 10,638 hectares to pasture means that 19,361 ML/y is no longer being extracted by hardwood plantations in Coles and Short. It is considered that this scenario alone will reduce plantation impacts on groundwater to within the TML by 2022 without any dependency on irrigation licence reductions.

For hardwood forests areas outside of Coles and Short, it is assumed that older parts of the plantation estate are also progressively cleared. It is assumed that 40% of this area is also converted to pasture after clearance, while 60% remains as hardwood plantations. Softwood plantation areas in Zone 3A, Mount Muirhead and Kennion are assumed to remain the same land use.

The results of Scenario 1 show that groundwater levels recover by 4.2 to 6.5 m in Coles and Short following the progressive change in land use from 2015 onwards (Figures 6.3 to 6.6). This recovery generally returns groundwater levels close to pre-forest (2000) groundwater levels, although groundwater levels still fluctuate with ongoing cycles of forest plantation and clearance, particularly where hardwood plantations are present and managed on 10 year growth cycles. This is demonstrated by the hydrograph for CLS002 (Figure 6.3), where groundwater levels show a declining trend from 2029 as hardwood is replanted, but then recover at the end of the 10 year growth cycle as forest is cleared for replanting. The degree to which groundwater levels recover depends on the spatial distribution of land use change, which may vary. While some information on the area of forest cleared in 2015 and 2016 was available at the time of this study, detail on the spatial distribution of this clearance was not available, and accurately reconstructing the distribution from aerial imagery proved difficult. It should also be noted that these scenario results assume long term average recharge occurs.

As groundwater levels recover in Scenario 1, an increase in groundwater ET from Bool Lagoon is observed. Table 6.2 gives the water balance for the Bool Lagoon area for 2000 and 2015 from the calibrated model, and 2020, 2030 and 2035 from the scenario. Table 6.2 also shows a reduction in the volume of groundwater outflow from the Bool Lagoon area as the watertable recovers in Coles and Short and the gradient for through flow becomes less steep. This suggests that the change in land use simulated in Scenario 1 would promote groundwater discharge in the Bool Lagoon area.

Table 6.2. Mass balance for the Bool Lagoon area for Scenario 1 (2020, 2030, 2035)

	Calibrated model		Scenario 1		
	2000 (pre-forest)	2015 (post forest)	2020	2030	2035
INFLOWS (GL)					
Groundwater inflow	48.8	48.5	49.6	48.9	48.1
OUTFLOWS (GL)					
ET	10.7	9.9	9.5	10.2	10.3
Drains	4.4	2.6	2.2	3.2	2.4
Groundwater outflow	33.7	36.2	37.4	35.1	35.8
Gain in storage (GL)	2.9	2.5	2.8	2.8	1.9
Loss from storage (GL)	2.8	2.6	2.2	2.4	2.4
Change in storage (GL)	0.1	-0.1	0.6	0.4	-0.4

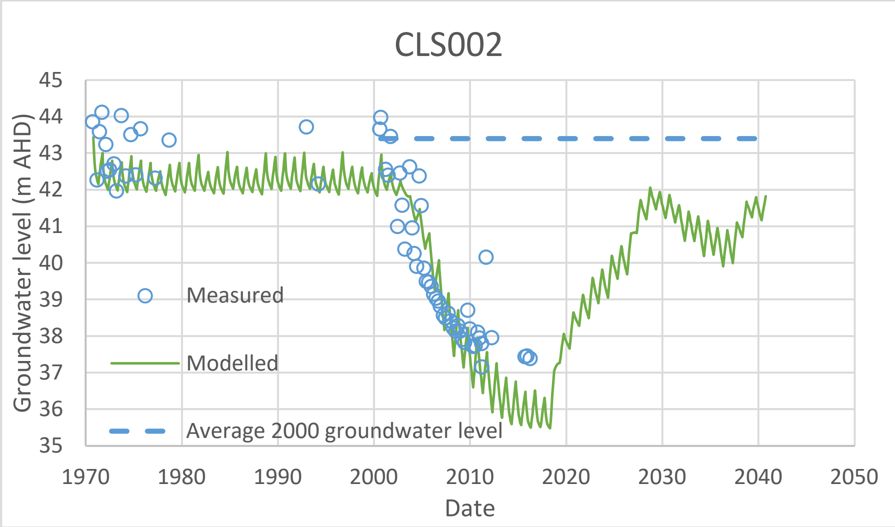


Figure 6.3. Groundwater levels in CLS002 for Scenario 1

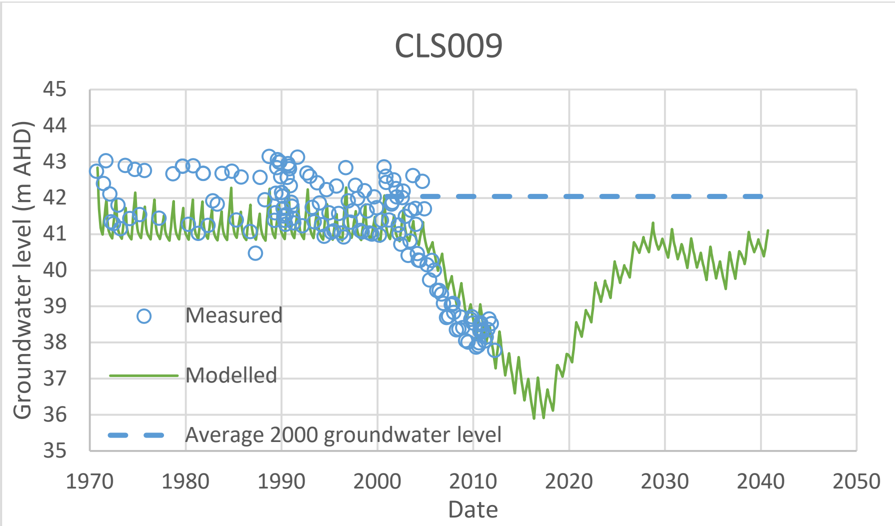


Figure 6.4. Groundwater levels in CLS009 for Scenario 1

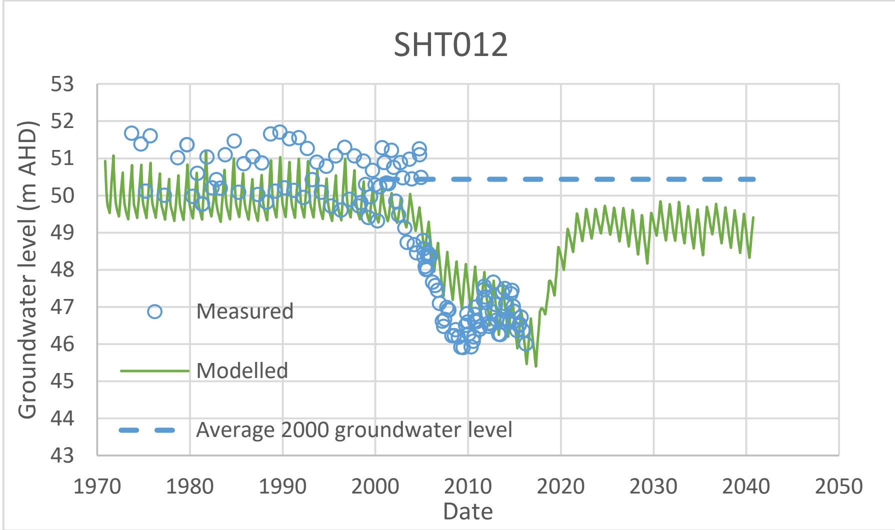


Figure 6.5. Groundwater levels in SHT012 for Scenario 1

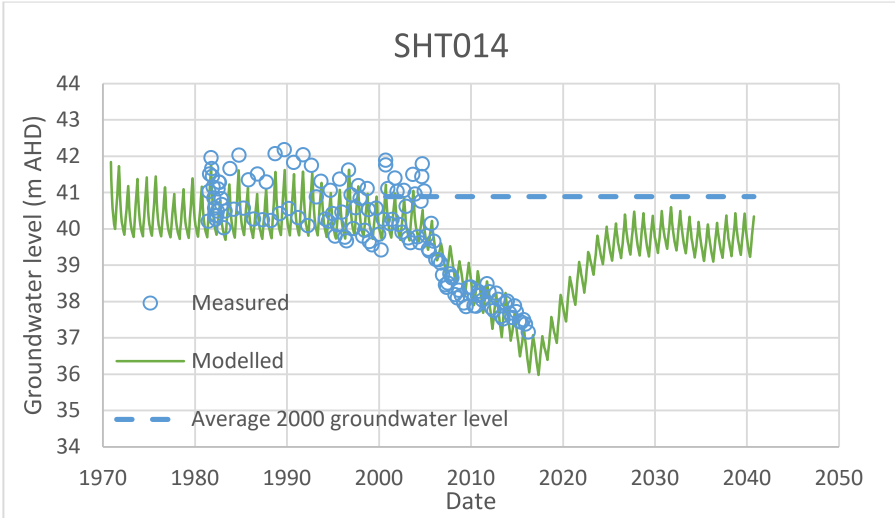


Figure 6.6. Groundwater levels in SHT014 for Scenario 1

6.4 Scenario 2

Scenario 2 considered the clearance of plantation forest in Coles and Short from 2015 onwards, such that by 2022, 20.3% of the plantation estate had been cleared and returned to pasture, 38.4% is converted to softwood plantations, and 41.3% remains as hardwood. The slight variations in change in softwood and hardwood (32% and 28% as opposed to 30% and 30%) are due to the way in which land use data is dealt with in the model. That is for ease of implementation, all forests planted in a particular year is grouped as one land use area. The spatial distribution of clearance is related to plantation age such that older plantations are cleared first. Table 6.3 shows the areas cleared each year, while Figure 6.2 shows the spatial distribution of land use by 2022. Compared to Scenario 1, conversion of 5354 ha to pasture means 9744 ML/y of direct extraction by hardwood plantations is removed from the water balance.

Table 6.3. Areas of clearance in Coles and Short from 2015 to 2022 in Scenario 2

	Areas (hectares)			
Year	**Total cleared**	**Converted to pasture**	**Remaining hardwood**	**Converted to softwood**
2015	3,609	530	3,079	0
2016	5,133	2,117	3,016	0
2017	6,947	0	0	6,947
2018	1,918	1,187	276	455
2019	2,530	529	2,001	0
2020	2,701	231	0	2,470
2021	2,003	760	1,243	0
2022	1,537	0	1,270	267
Total (hectares)	26,378	5,354	10,885	10,139
Percent	100	20.3	41.3	38.4

The areas remaining as hardwood are initially coppiced – allowed to regenerate from the stump and grown for a further eight years. After this remaining hardwood plantations are clear felled and managed on a 10 year growth cycle with a replant for the next rotation, as per the forest water accounting model characterised average plantation. Recharge is assumed to be constant each year at the management area rates given by Brown et al. (2006), and the scenario runs from 2015 to September 2040. It is considered that this scenario would require a reduction in potentially un-used irrigation licences in Coles and Short to achieve the allocation reductions specified in the LLCWAP.

For hardwood forest areas outside of Coles and Short, it is assumed that older parts of the plantation estate are also progressively cleared. It is assumed that 40% of this area is also converted to pasture after clearance, while 60% remains as hardwood plantations. Softwood plantation areas in Zone 3A, Mount Muirhead and Kennion are assumed to remain the same land use. This is consistent with Scenario 1.

The results of Scenario 2 show groundwater levels recovering by 3.9 to 4.5 m in Coles and Short following the land use changes from 2015 onwards (Figures 6.7–6.10). Groundwater levels do no quite recover to pre-forest levels, and ongoing fluctuations are observed as forest management cycles are carried out. As with Scenario 1, the spatial extent of groundwater level recovery depends on the spatial variation in land use change. However in general, it is observed that groundwater levels to do not recover as much as in Scenario 1. This is because less land area is converted to pasture.

As with Scenario 1, the recovery in groundwater levels results in an increase in modelled groundwater ET from the Bool Lagoon area (Table 6.4). A small reduction in groundwater outflow from the Bool Lagoon is also observed as the watertable recovers in Coles and Short and the gradient for through flow becomes less steep. However the increase in ET and decrease in outflow is not as large as in Scenario 1, given that groundwater levels in Coles and Short have not recovered as much as in Scenario 1.

Table 6.4. Mass balance for the Bool Lagoon area for Scenario 2 (2020, 2030, 2035)

	Calibrated model		Scenario 2		
	2000 (pre-forest)	**2015 (post forest)**	**2020**	**2030**	**2035**
INFLOWS (GL)					
Groundwater inflow	48.8	48.5	49.6	49.1	48.2
OUTFLOWS (GL)					
ET	10.7	9.9	9.4	10.1	10.2
Drains	4.4	2.6	2.2	3.1	2.3
Groundwater outflow	33.7	36.2	37.4	35.4	36.1
Gain in storage (GL)	2.9	2.5	2.8	2.8	1.9
Loss from storage (GL)	2.8	2.6	2.2	2.3	2.3
Change in storage (GL)	0.1	-0.1	0.6	0.4	-0.4

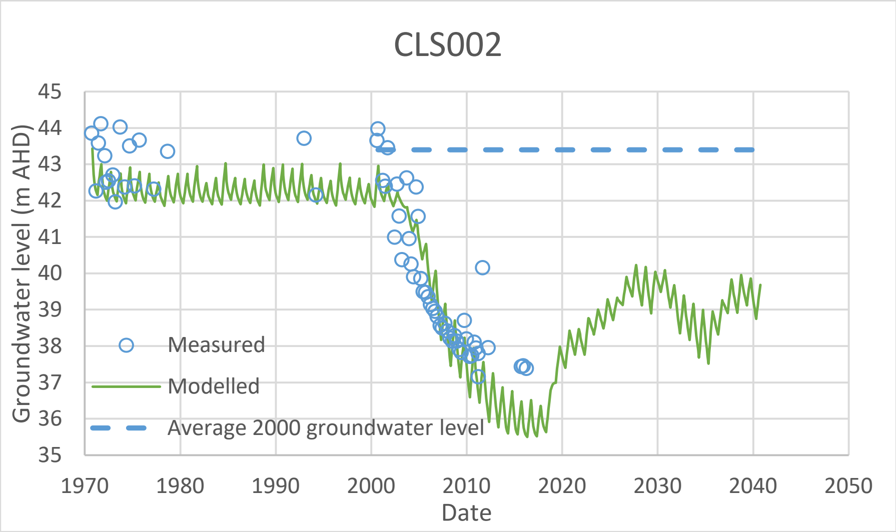


Figure 6.7. Groundwater levels in CLS002 for Scenario 2

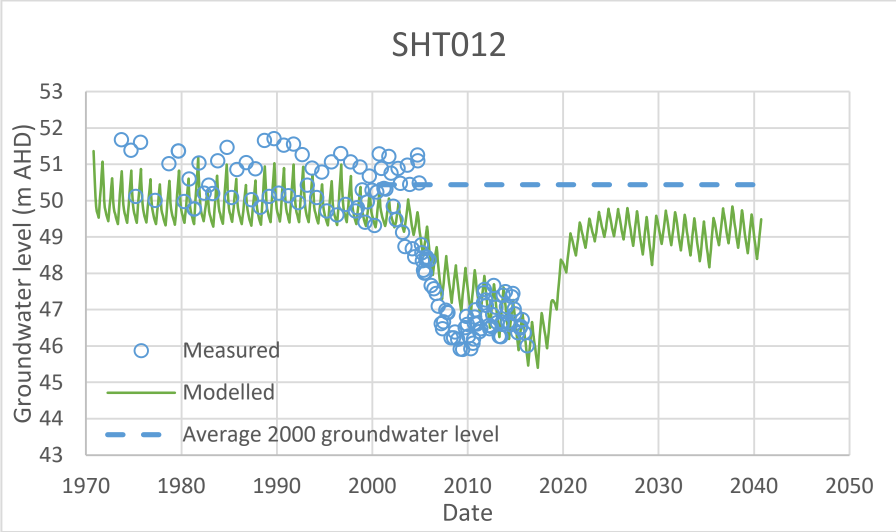


Figure 6.9. Groundwater levels in SHT012 for Scenario 2

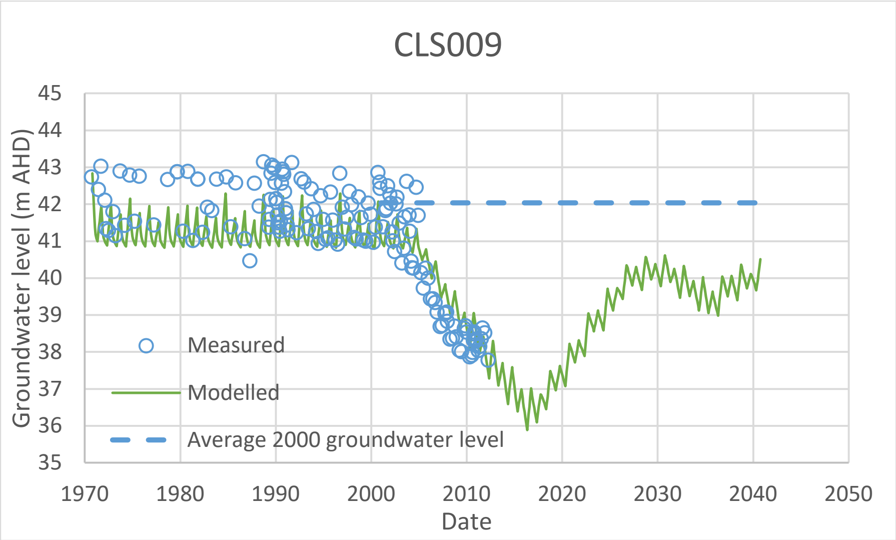


Figure 6.8. Groundwater levels in CLS009 for Scenario 2

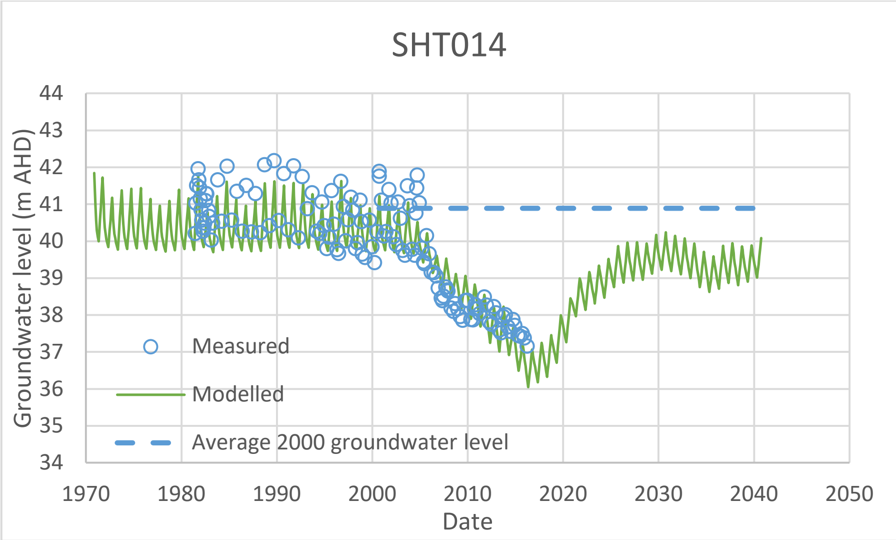


Figure 6.10. Groundwater levels in SHT014 for Scenario 2

6.5 Scenario 3

Scenario 3 simulates the same change in land use as Scenario 1, in which by 2022, 40% of the plantation estate had been cleared and returned to pasture, 30% is converted to softwood plantations, and 30% remains as hardwood. However in Scenario 3, recharge from 2015 onwards is reduced from the long term average by 15%. The results are thus similar to the results of Scenario 1, however groundwater level recoveries are generally 0.4-0.5 m less than in Scenario 1 (Figure 6.11–6.14). Groundwater ET in the Bool Lagoon area does still increase over time following the changes in land use (Table 6.5), however not to the same level as in Scenario 1.

Table 6.5. Mass balance for the Bool Lagoon area for Scenario 3 (2020, 2030, 2035)

	Calibrated model		Scenario 3		
	2000 (pre-forest)	**2015 (post forest)**	**2020**	**2030**	**2035**
INFLOWS (GL)					
Groundwater inflow	48.8	48.5	48.5	48.6	47.8
OUTFLOWS (GL)					
ET	10.7	9.9	8.9	10.0	10.1
Drains	4.4	2.6	1.7	2.8	2.2
Groundwater outflow	33.7	36.2	37.3	35.4	35.8
Gain in storage (GL)	2.9	2.5	2.6	2.7	1.8
Loss from storage (GL)	2.8	2.6	2.1	2.3	2.2
Change in storage (GL)	0.1	-0.1	0.6	0.4	-0.4

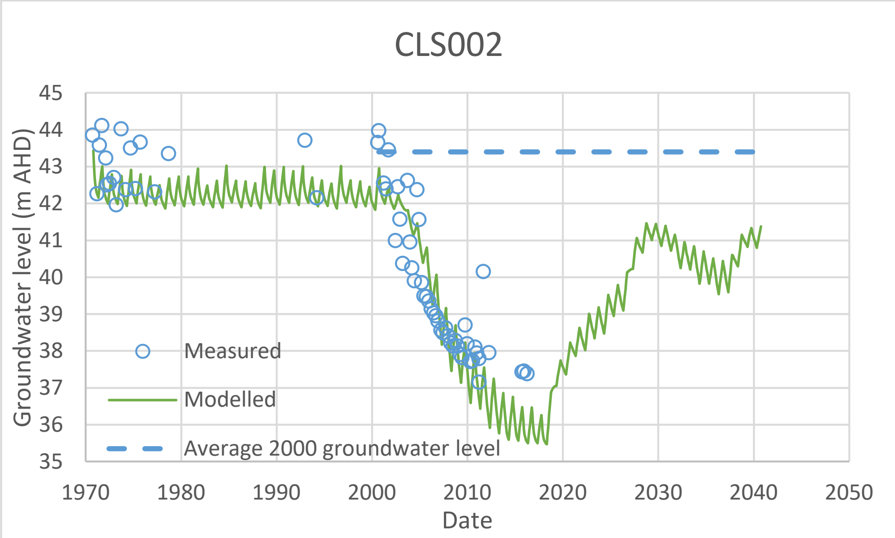


Figure 6.11. Groundwater levels in CLS002 for Scenario 3

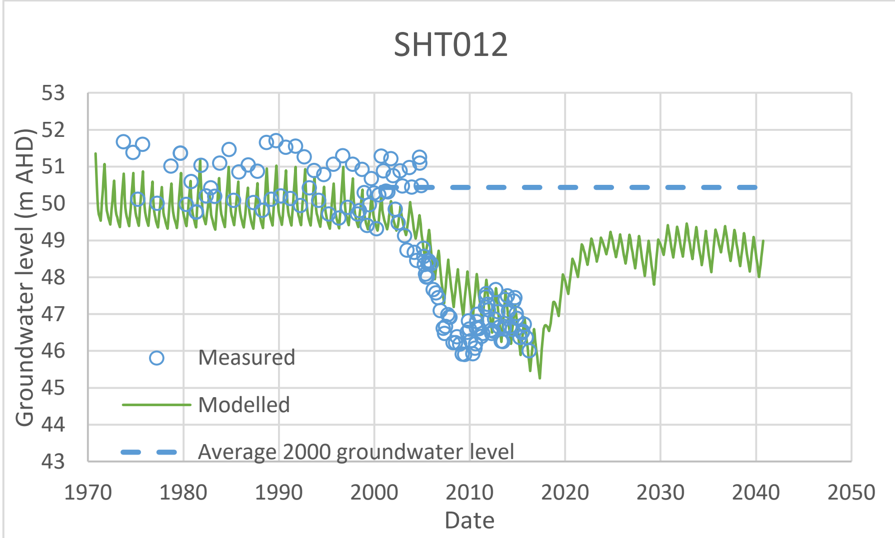


Figure 6.13. Groundwater levels in SHT012 for Scenario 3

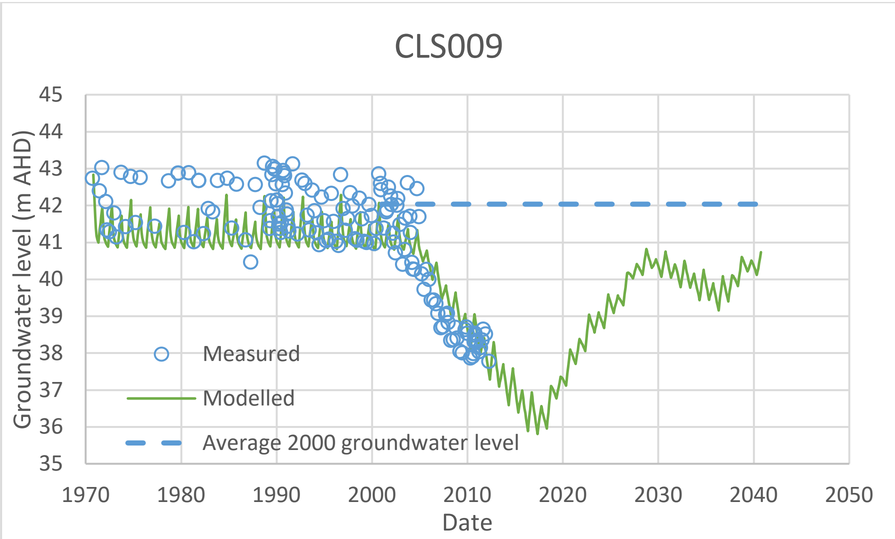


Figure 6.12. Groundwater levels in CLS009 for Scenario 3

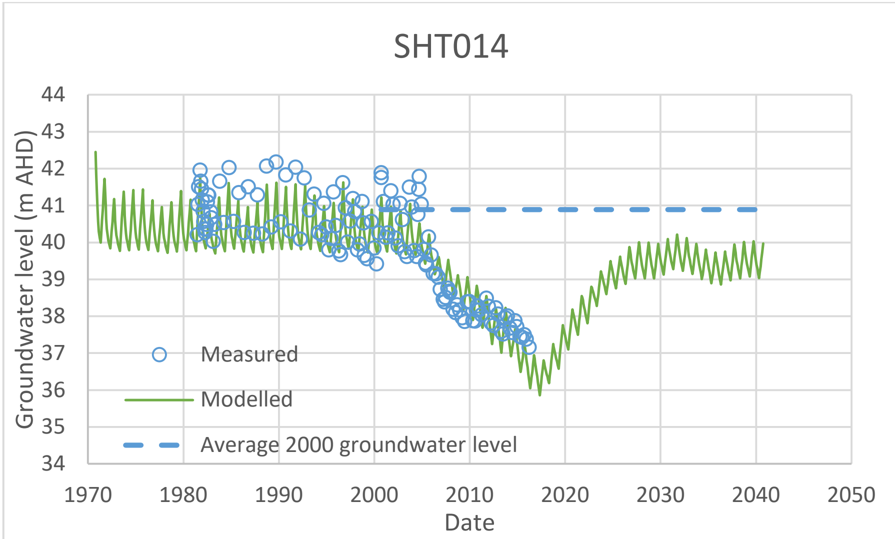


Figure 6.14. Groundwater levels in SHT014 for Scenario 3

6.6 Scenario 4

Scenario 4 simulates the same change in land use as Scenario 2, in which by 2022, 20% of the plantation estate had been cleared and returned to pasture, 40% is converted to softwood plantations, and 40% remains as hardwood. However recharge from 2015 onwards is reduced from the long term average by 15%. As with the comparison between Scenarios 1 and 3, the reduction in recharge results in groundwater levels being up to 0.57 m lower in Scenario 4 compared to Scenario 2 (Figure 6.15–6.18). Groundwater ET in the Bool Lagoon area does increase as watertables still recover following the change in land use (Table 6.6), however the recovery is less than in Scenario 2. There is little change in the volume of groundwater outflow from the Bool Lagoon area compared to Scenario 2 though, as lower groundwater levels associated with reduced recharge are similar across the domain meaning there is little change in the gradient between Bool Lagoon and Coles and Short.

Table 6.6. Mass balance for the Bool Lagoon area for Scenario 4 (2020, 2030, 2035)

	Calibrated model		Scenario 4		
	2000 (pre-forest)	2015 (post forest)	2020	2030	2035
INFLOWS (GL)					
Groundwater inflow	48.8	48.5	48.5	48.7	47.8
OUTFLOWS (GL)					
ET	10.7	9.9	8.9	9.9	10.0
Drains	4.4	2.6	1.7	2.6	2.1
Groundwater outflow	33.7	36.2	37.3	35.7	36.1
Gain in storage (GL)	2.9	2.5	2.6	2.6	1.8
Loss from storage (GL)	2.8	2.6	2.1	2.3	2.2
Change in storage (GL)	0.1	-0.1	0.5	0.4	-0.4

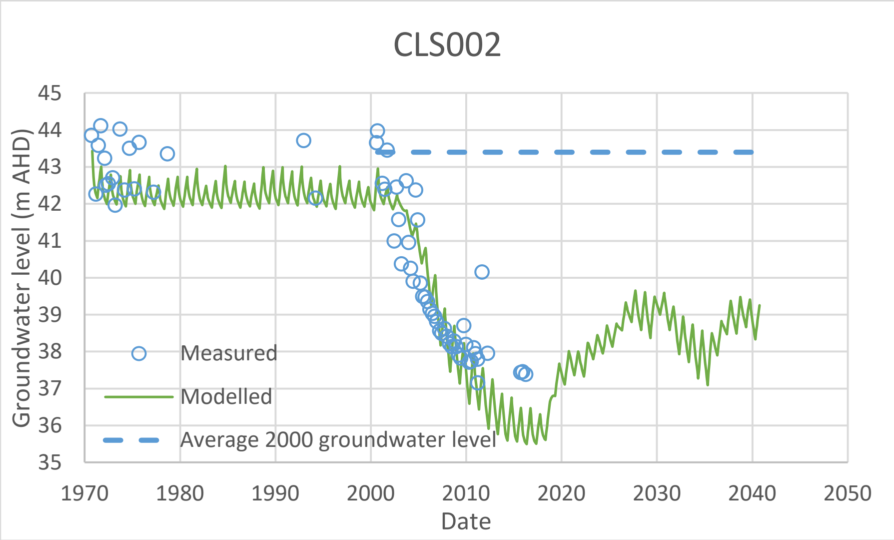


Figure 6.15. Groundwater levels in CLS002 for Scenario 4

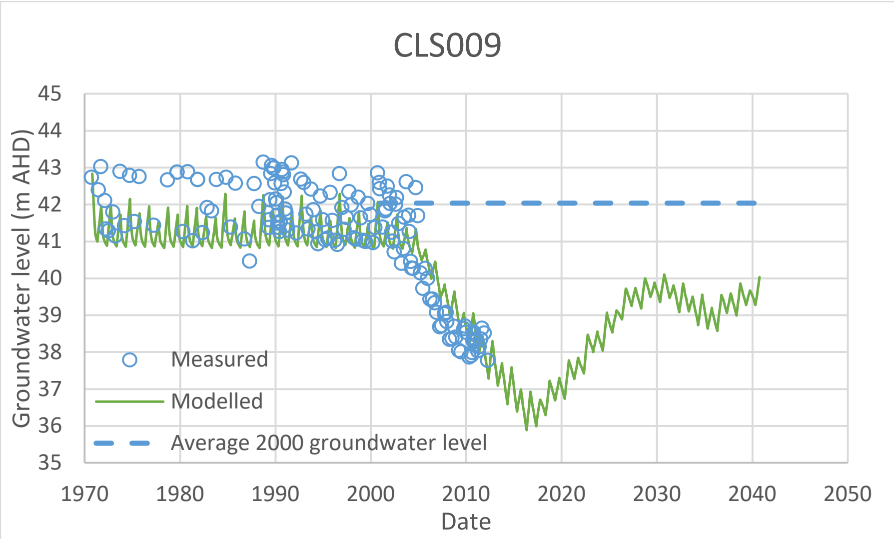


Figure 6.16. Groundwater levels in CLS009 for Scenario 4

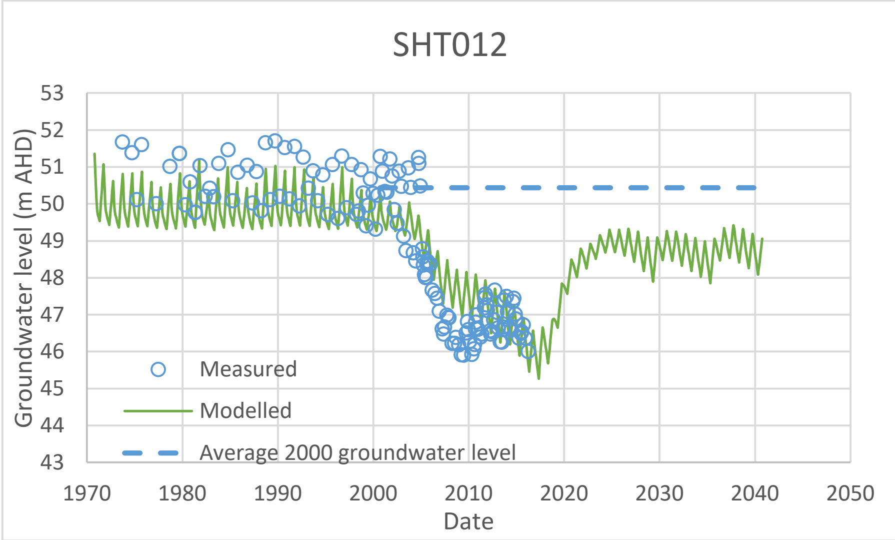


Figure 6.17. Groundwater levels in SHT012 for Scenario 4

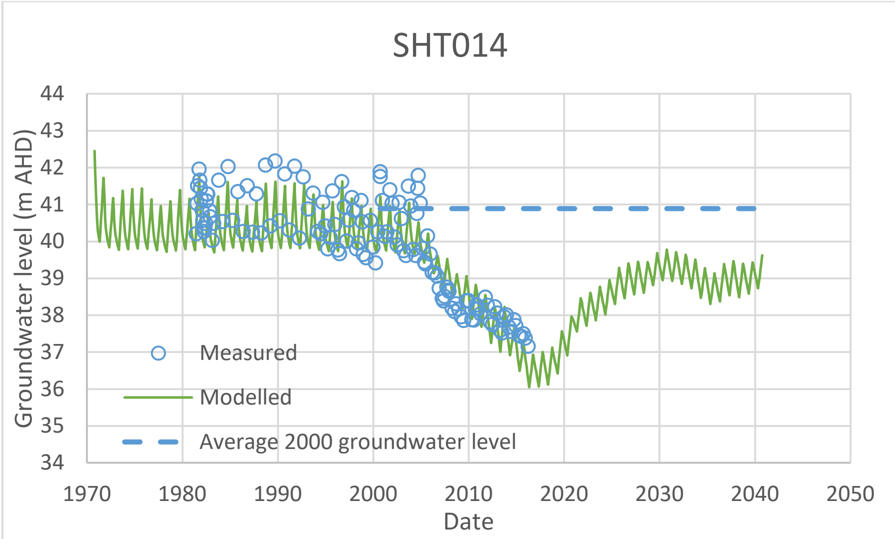


Figure 6.18. Groundwater levels in SHT014 for Scenario 4

6.7 Scenario 5

Scenario 5 was developed after consultation with the Green Triangle Regional Plantation Committee (GTRPC) and Natural Resources South East in July 2017. The scenario considers the potential for increased recharge at the commencement of a plantation cycle, following the work of Benyon and Doody (2009). Benyon and Doody (2009) reported that land clearance and weed control practices in between forest plantations may result in enhanced groundwater recharge during this fallow period. Based on water balance data collected at three study sites (two softwood and one hardwood plantation), Benyon and Doody estimated recharge to be 1.4 to 1.6 times greater than the management area recharge rate in the year after plantation clearance. This is higher than the factor of 1.2 used in the LLCWAP forest water accounting models (Harvey, 2009). Recharge in the second year was estimated to be 1.9 times greater than the management area recharge rate at the hardwood site, although the authors note that the results may not be accurate as the site was not managed as a typical hardwood plantation after harvesting.

While research questions may remain regarding the age at which recharge ceases during forest development and the actual recharge occurring during this period, to simulate increased recharge following clearance, Scenario 5 considers increases in recharge up to 2 times the management area recharge rate in the first year of plantations. This is followed by a linear decrease in recharge to canopy closure. Canopy closure occurs after three years for hardwood plantations, and after six years for softwood, consistent with the LLCWAP forest water accounting models. Furthermore, softwood is managed on a 32 year plantation cycle with three thinnings, based on feedback provided by the GTRPC. The recharge models for hardwood and softwood are provided in Table 6.7 and 6.8 respectively.

In terms of land use change, Scenario 5 follows the assumptions in Scenario 1. That is reductions in allocation are achieved by the conversion of 40% of the plantation estate in Coles and Short from forest to pasture. The remaining 60% of the area is split between replanted hardwood plantations (32%) and conversion of hardwood to softwood plantations (28%).

The results of Scenario 5 are only slightly different to the results of Scenario 1, with groundwater levels recovering up to 0.22 m more in Scenario 5 as a result of enhanced recharge after clearance (Figures 6.19 to 6.22). However following the increased recharge, groundwater levels decline to the same level in Scenario 5 as Scenario 1. There is little difference between Scenario 1 and 5 for the water balance at Bool Lagoon (Table 6.9), thus the enhanced recharge in fallow years does not reduce impacts on Bool Lagoon.

Table 6.7. Recharge model for hardwood plantations in Scenario 5 compared to LLC values (used elsewhere in this report)

	Recharge as percent of management area rate	
Year of forest rotation	Scenario 5	LLCWAP
1	200	120
2	133	80
3	67	40
4	0	0
5	0	0
6	0	0
7	0	0
8	0	0
9	0	0
10 (harvest)	0	0
11 (clean up)	0	0

Table 6.8. Recharge model for softwood plantations in Scenario 5 compared to LLC values (used elsewhere in this report)

	Recharge as percent of management area rate	
Year of forest rotation	**Scenario 5**	**LLCWAP**
1	200	120
2	167	100
3	135	80
4	100	60
5	70	40
6	35	20
7	0	0
8	0	0
9	0	0
10	0	0
11	0	50
12	0	0
13	50	0
14	0	0
15	0	0
16	0	0
17	0	50
18	0	0
19	0	0
20	50	0
21	0	0
22	0	0
23	0	50
24	0	0
25	0	0
26	0	0
27	50	0
28	0	0
29	0	0
30	0	50
31	0	0
32	0 (harvest)	0
33	0 (clean up)	0
34	200	0
35	167	0 (harvest)
36	135	0 (clean up)

Table 6.9. Mass balance for the Bool Lagoon area for Scenario 5 (2020, 2030, 2035)

	Calibrated model		Scenario 5		
	2000 (pre-forest)	2015 (post forest)	2020	2030	2035
INFLOWS (GL)					
Groundwater inflow	48.8	48.5	49.6	48.9	48.1
OUTFLOWS (GL)					
ET	10.7	9.9	9.5	10.2	10.3
Drains	4.4	2.6	2.2	3.3	2.5
Groundwater outflow	33.7	36.2	37.4	35.1	35.8
Gain in storage (GL)	2.9	2.5	2.8	2.8	1.9
Loss from storage (GL)	2.8	2.6	2.2	2.4	2.4
Change in storage (GL)	0.1	-0.1	0.6	0.4	2.4

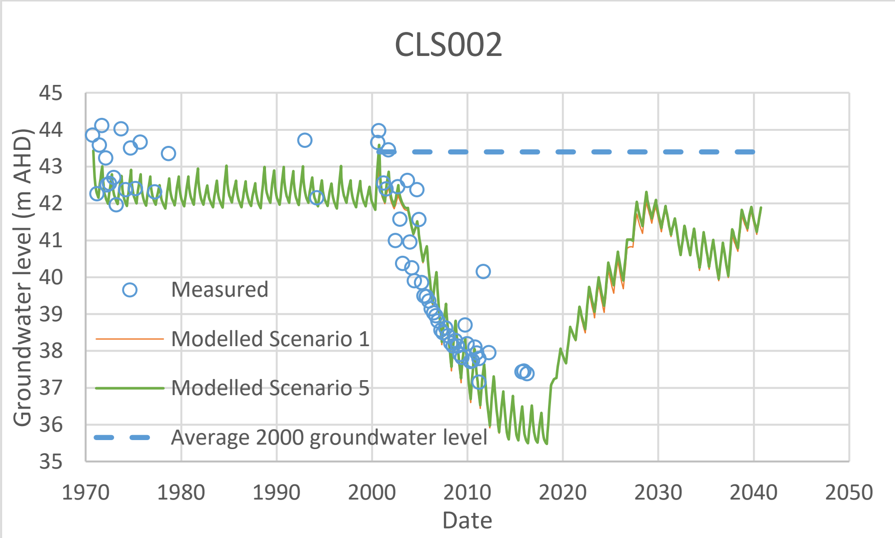


Figure 6.19. Groundwater levels in CLS002 for Scenario 5

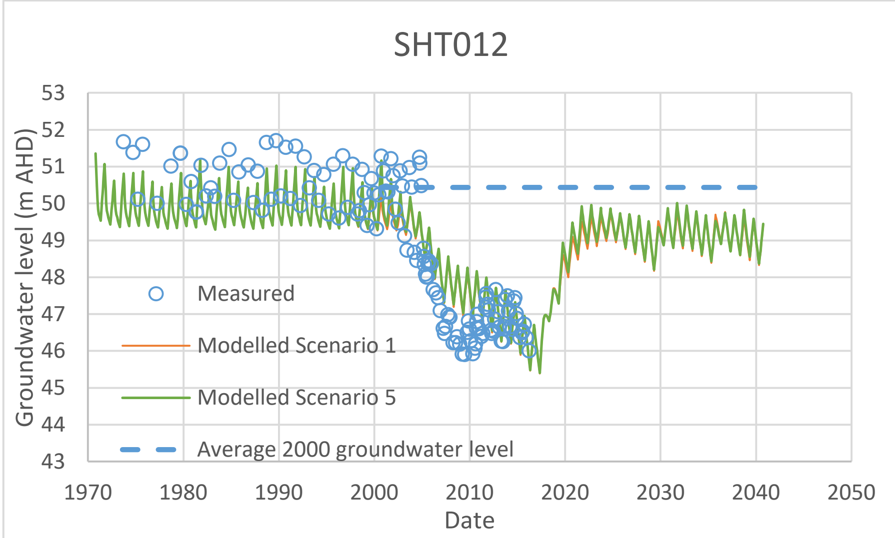


Figure 6.21. Groundwater levels in SHT012 for Scenario 5

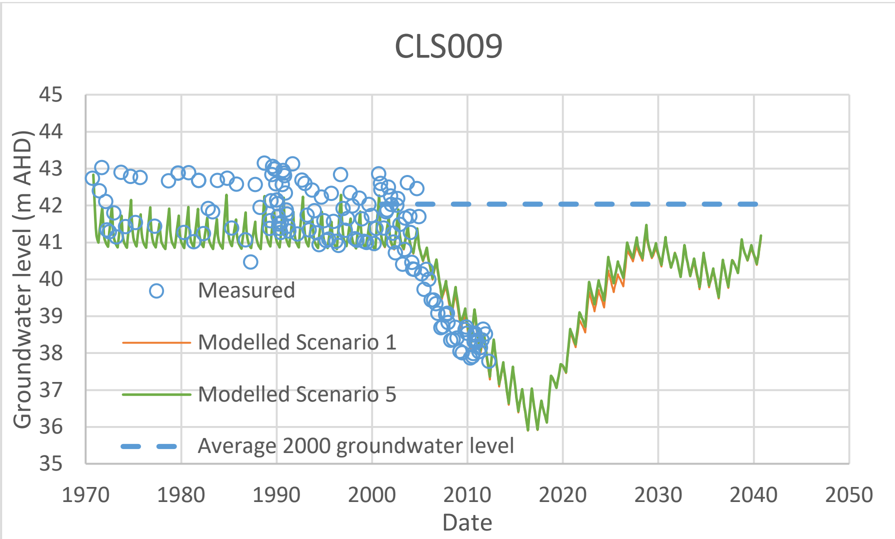


Figure 6.20. Groundwater levels in CLS009 for Scenario 5

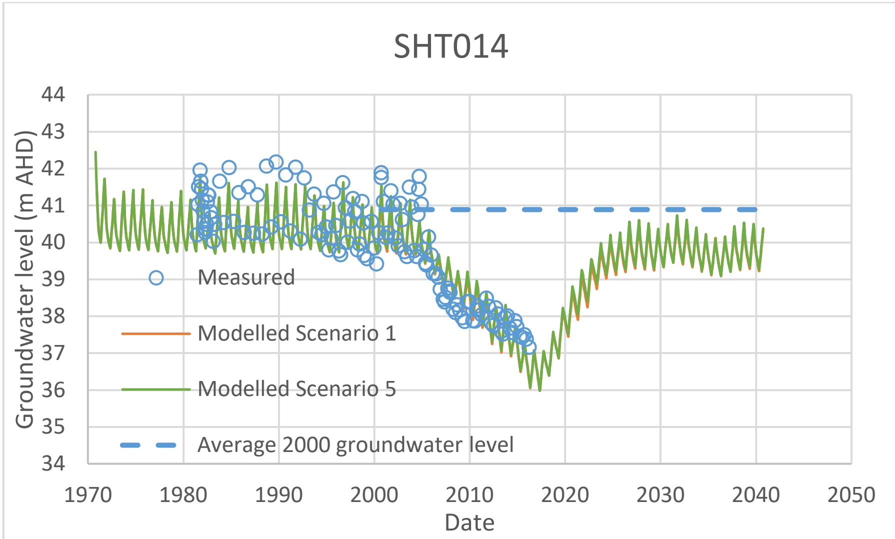


Figure 6.22. Groundwater levels in SHT014 for Scenario 5

6.8 Summary of scenarios

The scenarios modelled here demonstrate that a change in land use in the Coles and Short management areas, which involves the conversion of a percentage of the existing hardwood plantation estate to pasture, will result in some groundwater level recovery. However the amount of groundwater level recovery and its spatial variation will depend upon how much land use change takes place. This is currently not known, and it is recommended that model scenarios be reviewed as more information about land use change in Coles and Short becomes available.

Based on the scenarios modelled here, a reduction in hardwood plantations in Coles and Short will lead to an increase in groundwater levels and thus an increase in groundwater ET and reduction in groundwater outflow from the Bool Lagoon area compared to 2015, which is likely to promote groundwater discharge at the site. However the water balance at Bool Lagoon is also likely to be influenced by surface water inflows from Mosquito Creek, which are not considered here. Therefore these results are only indicative of groundwater changes around Bool Lagoon, rather than persistence of surface water in the lagoon.

A potential reduction in groundwater recharge resulting from reduced rainfall associated with climate change will also impact on groundwater levels. Comparison between land use change scenarios with (Scenario 3) and without (Scenario 1) climate change show groundwater levels are up to 0.5 m lower when recharge is reduced. However the difference in groundwater level is greater between the two different land use scenarios (Scenarios 1 and 2), than scenarios with and without climate change impacts on recharge. The climate change scenarios assume a static 15% reduction in long term average recharge, and further work could be done to consider more complex climate change scenarios. It is recommended that more information on potential future land use change should be considered before additional climate change scenarios are considered. This is because the combined impacts of recharge interception and groundwater extraction by plantations are likely to have a greater impact on groundwater levels in the area than changes in groundwater recharge alone.

Scenario 5 shows that increased recharge during fallow years of plantations, as discussed in Benyon and Doody (2009), leads to small (~0.5 m) increases in groundwater level following clearance of plantations. However groundwater levels decline to the same level as Scenario 1 as plantations reach maturity, and there is no difference in the impact on Bool Lagoon. This scenario nevertheless provides a useful initial sensitivity analysis of recharge under plantation forests. It would be useful to conduct a further sensitivity analysis of recharge as well as groundwater extraction from plantations, including testing groundwater extraction rates covering the range detailed in Benyon and Doody (2004).

7 Conclusions and recommendations

7.1 Conclusions

In this study, a groundwater model for the Wattle Range area in the LLC was updated and re-calibrated using the ETS package to simulate groundwater extraction by plantation forests. Calibration was achieved by varying hydraulic conductivity at 355 pilot points using PEST. The model shows a good fit to measured groundwater levels and trends in the region, and the hydraulic conductivity values derived through calibration conform with existing knowledge of aquifer properties. The model has been used to run a series of predictive scenarios to 2040, to assess the impact of potential land use changes and climate changes on groundwater level and flow. Based on the calibration and scenarios, the following can be concluded:

- **The forest water accounting models and recharge rates used in the Lower Limestone Coast Water Allocation Plan can be considered broadly appropriate for quantifying plantation forest impacts on groundwater.**

  The model used fixed annual rates of groundwater recharge and plantation forest impacts (recharge interception and direct extraction) consistent with values used in the LLCWAP and recharge rates given by Brown et al., (2006). A good calibration was achieved by varying hydraulic conductivity alone, and the conductivity values obtained are consistent with available data on aquifer parameters. It should be noted that the model uses groundwater extraction rates based on the forest water accounting models described by Harvey (2009), with hardwood plantations reaching a maximum extraction rate of 364 mm/y, which continues until the plantation is cleared. The model has not considered the 'annualised' rates of extraction of 182 mm/y described in the LLCWAP.

  The model was re-calibrated with alternative recharge rates, and the calibration was slightly better. However the hydraulic conductivity values derived from the alternative calibration were broadly similar to those from the first calibration. Hence in the absence of a full uncertainty analysis, the model parameters can be considered adequate for the purposes of supporting the LLCWAP. The influence of uncertainty in hydraulic conductivity, recharge and groundwater extraction by plantation forests on model predictions could be revisited using this model as the LLCWAP is reviewed in future. In particular, quantifying uncertainty in recharge and its impact upon model simulations should be a priority, given that groundwater allocation is based upon estimates of recharge.

- **Reduction in the area of hardwood forest in Coles and Short is likely to lead to recovery in groundwater levels and less drawdown impacts on Bool Lagoon. However the level of recovery will be spatially variable, and dependent upon the extent of land use change and potential changes in groundwater recharge.**

  The LLCWAP (SENRMB, 2013) identifies the Coles and Short management areas as over-allocated based on forest impacts on groundwater, hence reductions in groundwater use are likely to be achieved through changes in land use, including clearance of forests. The scenarios modelled here show that varying levels of forest clearance and change in land use are likely to lead to recovery in groundwater levels in Coles and Short. However the spatial distribution and extent of land use change cannot be predicted at this stage, hence recoveries will vary spatially. The scenarios have been projected to 2040 assuming both long term average recharge occurs, and a reduction in recharge due to reduced rainfall. The model should be revisited in the next 5 to 10 years as more information on land use change becomes available.

7.2 Recommendations

The model can be considered suitable for simulating groundwater processes and the impact of plantation forests on groundwater levels in the study area. Thus the model and scenarios run are suitable for supporting the LLCWAP. However further work could be done to improve both the model performance from a technical perspective, as well as how the model can be used to assist in future reviews of groundwater policy in the LLC.

Recommendations for use of the model to assist in future groundwater policy review include:

- <u>Definition of resource condition limits</u>

  As part of this study, the groundwater balance around Bool Lagoon was reported on, as Bool Lagoon is an important wetland identified as groundwater dependent in the LLCWAP. While the LLCWAP stipulates set back distances for groundwater development around sites such as Bool Lagoon, there are currently no resource condition indicators (RCIs) or resource condition limits (RCLs) for maintaining specific conditions around Bool Lagoon. Recent work by Cranswick and Barnett (2017) has developed measurable RCLs for groundwater resources in the upper South East, to help guide groundwater management settings. Development of measurable RCIs and RCLs for Bool Lagoon and other groundwater users in the study area would help in interpreting model results in terms of the potential impact of land use change, and also help future reviews of groundwater management in the region. This would also align groundwater management review in the study area with work conducted in the upper South East (Cranswick and Barnett, 2017).

- <u>Extension of the eastern boundary</u>

  The main purpose of this model was to simulate groundwater processes in response to forest plantations in and around the Coles and Short management areas. As part of the re-calibration, additional data from surrounding management areas was incorporated and the domain boundary extended slightly in the north-east. However the model results close to the eastern boundary show an initial adjustment in the first few stress periods (see plots of CMM wells in Appendix). The simulation of groundwater processes close to the SA–Victoria border may be of interest in future, and this model could be drawn upon for such work. However it is recommended that the eastern boundary of the model be extended further east, possibly to the same extent as the model of Morgan et al. (2015), to remove the influence of boundary conditions on initial groundwater levels close to the border. This would increase the usefulness of this model as a tool to understand and test management scenarios both in the hardwood plantation forest areas as well as in the SA-Victoria border zone.

- <u>Further uncertainty analysis</u>

  A full parameter uncertainty analysis was not performed as part of this project. However a recalibration with alternative recharge rates (Section 5.2.4) demonstrated that while the model used in this study is suitable for the purposes of this project, further parameter uncertainty work would be useful. If the model were to be used as a tool to investigate alternative policy settings based on a RCL approach, then understanding any impacts of parameter uncertainty on model predictions will be critical for providing confidence in model results. Additional data on aquifer properties would also be useful in this context.

Recommendations for areas of technical improvement to the model include:

- <u>Constraining modelled ET</u>

  Rates of groundwater ET from plantation forests in the model were based on rates adopted in the LLCWAP, which were based on the rates described by Benyon and Doody (2004) and refined during stakeholder consultation (Harvey, 2009). Rates of groundwater ET from pasture areas in the model were likewise based on values reported by Benyon and Doody (2004), as well as values used in existing groundwater models for the area. This approach was considered valid for testing the forest water use policy with the model in this study. However ET is likely to vary spatially and temporally under all land use types. Understanding this variability in ET, especially in relation to plantation groundwater use, may help guide the review of water plans in the region in future. One way to improve this understanding and constrain modelled groundwater ET, would be to compare model outputs to estimates of ET from remote sensing data, which provides detailed spatial and temporal coverage. Doble and Crosbie (2017) cite such comparisons as a growing field of research. The CMRSET data developed by Guerschman et al., (2009) could be used for this purpose. However the CMRSET data would need to be calibrated against field estimates of ET for the study area (T Doody (CSIRO) 2017, pers comm., 10 May), which was considered beyond the scope of this project. Calibration of the CMRSET data to water balances ET estimates from Benyon and Doody (2004), or installation of an eddy covariance tower for measuring ET in a plantation area in the South East, would therefore be useful future projects in order to constrain satellite based ET estimates in the area. If reliable ET estimates could be obtained from satellite data, then further calibration and parameter uncertainty work could be done with the model, in order to understand spatial and temporal variability in groundwater ET, and assist future groundwater management reviews.

- Simulation of groundwater processes around drains

  The model currently simulates the drainage network with the MODFLOW Drain package, consistent with previous models of the area (Aquaterra, 2010a; Morgan et al., 2015). However the spatial and temporal variability of groundwater discharge to or recharge from (see Section 2.5.2) drains is not accurately represented using this approach. Nevertheless, the main limitation in simulating these processes is a lack of data with which to constrain model fluxes. Following the recommendations in Harrington et al. (2012) additional data on groundwater processes around drains could be collected using differential gauging and environmental tracers to better constrain these fluxes. With this data it may be possible to simulate groundwater processes around drains more accurately, and if necessary, test an alternative to the Drain package (e.g. the MODFLOW Stream package). This would allow for a more integrated approach in considering management options and their implications for both groundwater and surface water in the region.

- Finer temporal and spatial resolution

  In relation to recommendations on constraining model ET, the CMRSET data referred to above is available at 8-day intervals. Thus to accurately simulate the temporal variation in groundwater ET, stress period lengths in the model should be decreased from bi-annual to monthly. Furthermore, the model grid could be spatially refined further so that more accurate surface elevations can be assigned. More refined surface elevations mean groundwater ET can be simulated in a more refined way, as modelled ET is dependent upon the elevation of the watertable beneath the model surface.

- Future land use change

  The scenarios modelled in this study assess potential changes in land use over the next two decades. The changes simulated are based on the currently available data on hardwood forestry clearance in 2015 and 2016, and speculations on future clearance and management practices (e.g. coppiced forest management versus clear felling). However detailed information on the spatial extent of clearing is not currently available, and difficult to reconstruct from aerial photography. If future management of the plantation estates differs significantly from the scenarios modelled here, then it is recommended that the model scenarios be revisited with more accurate information, to assess potential longer term groundwater responses.

- Changes in groundwater quality

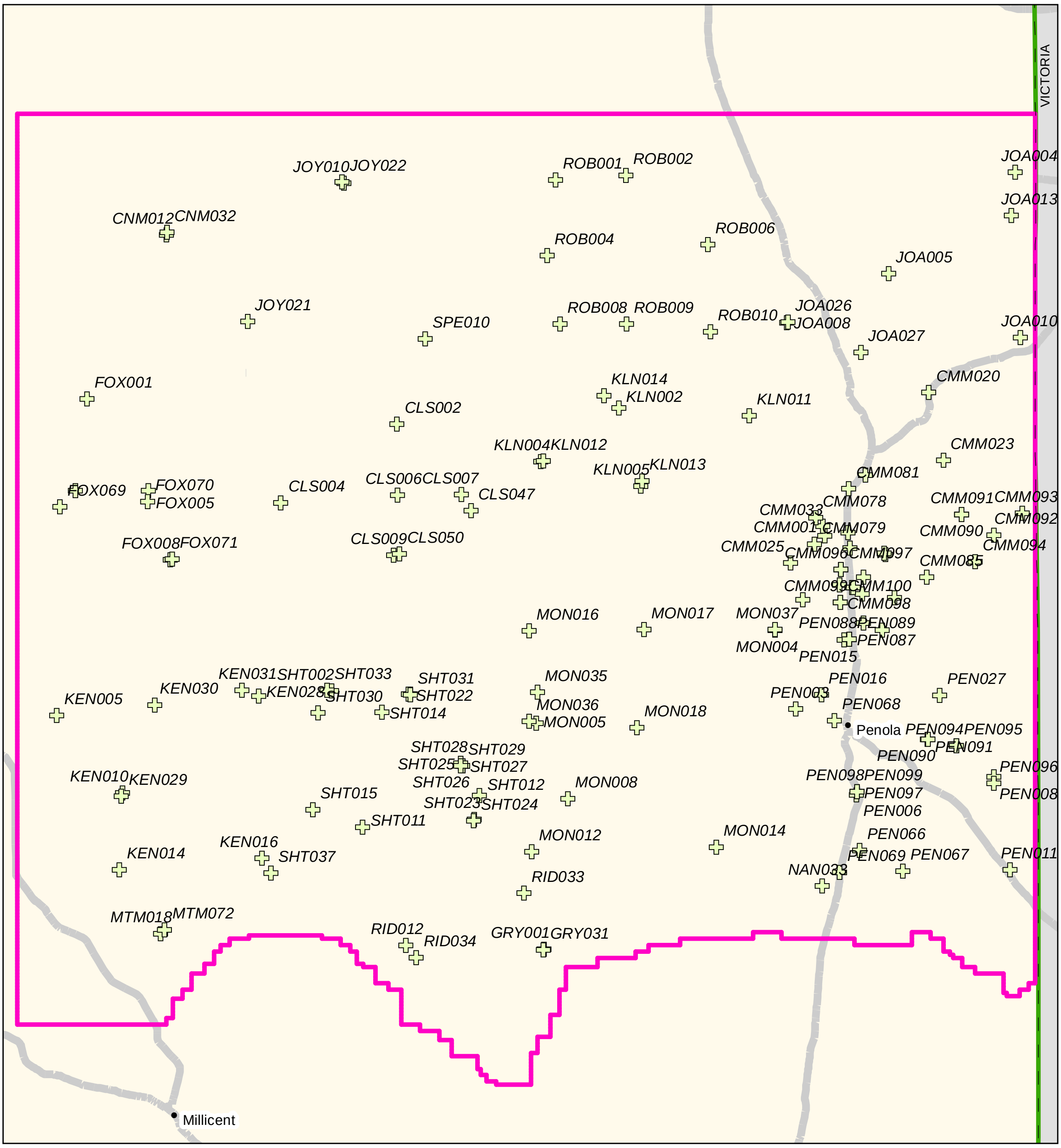
  Mustafa et al. (2006) identified the potential for salt to accumulate in the unsaturated zone under plantation forests, as plants use groundwater and soil water, and leave salts behind. However the total salt accumulation over the time that hardwood plantations have been in place is not known. If ongoing groundwater monitoring reveals changes in groundwater salinity as large areas of plantation are cleared, then the model should be updated to simulate solute transport processes and assess long term impacts.

- Future climate scenarios

  The reduced rainfall scenarios simulated in this study are relatively simplistic in applying a rainfall/recharge scaling factor derived from separate unsaturated zone modelling studies. This approach does not explicitly take into account the potential impact of changes in seasonality of rainfall, or possible vegetation feedback in response to climate change. Increases in CO_2 may result in increases in plant water use (Trancoso et al., 2017), which in this study area could lead to increased groundwater ET. More detailed scenario analysis, perhaps through the use of unsaturated zone modelling, would be required to look at these processes in more detail.

8 Appendix

Figure 8.1 Location of observation wells used in model calibration



Observation wells

Study area

Primary Road

SA - Victoria border

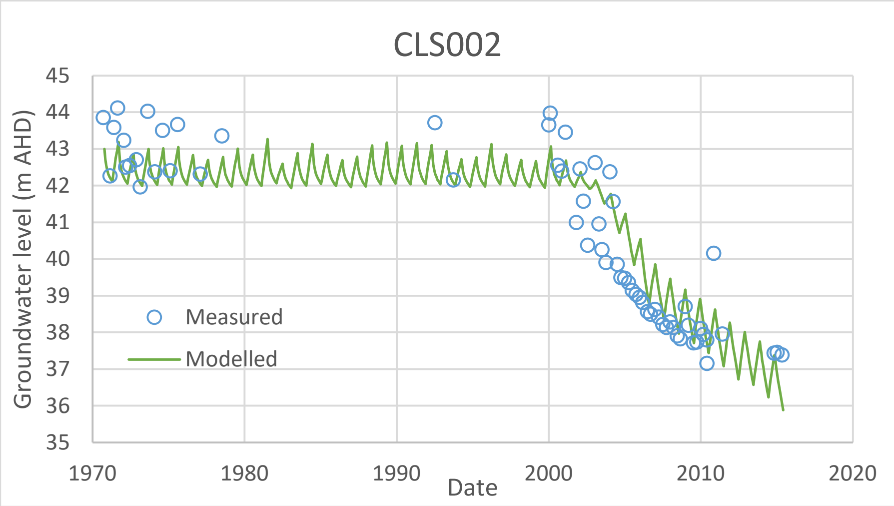
N

0 5 10 15

Kilometres

Produced by: Science, Monitoring and Knowledge Branch
Map Projection: Lambert Conformal Conic
Map Datum: Geocentric Datum of Australia 1994
Date: February 2017

Government of South Australia
Department of Environment, Water and Natural Resources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45
44
43
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35
1970
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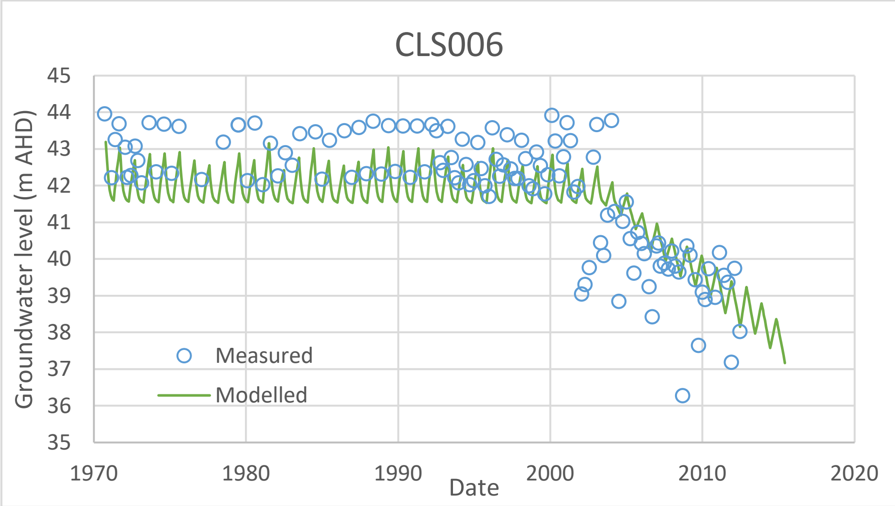
Date
Measured
Modelled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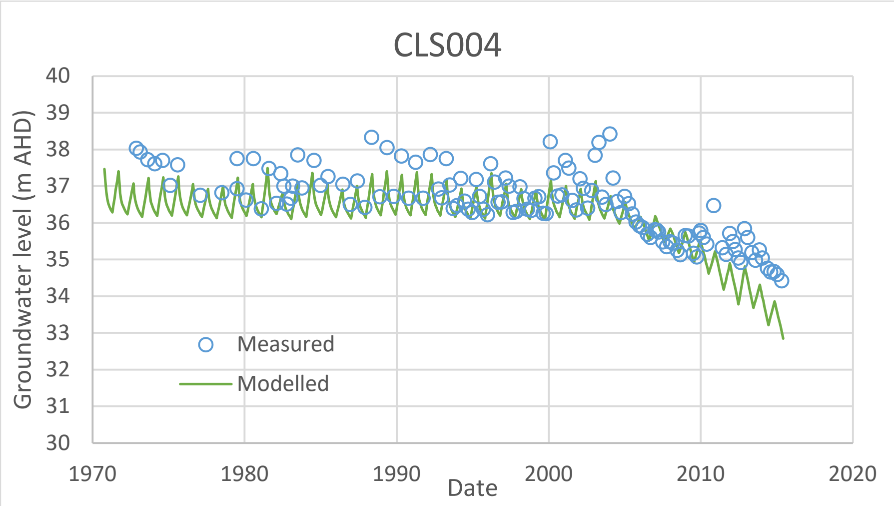
Measured
Modelled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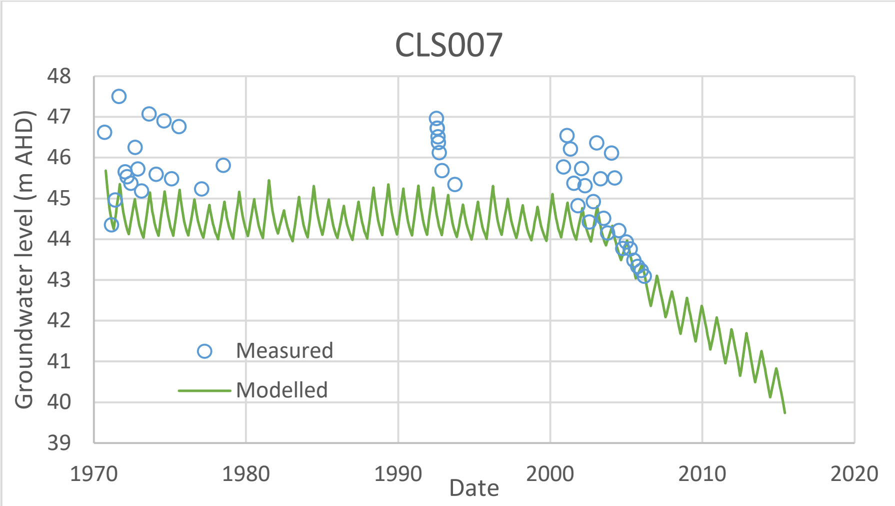
Measured
Modelled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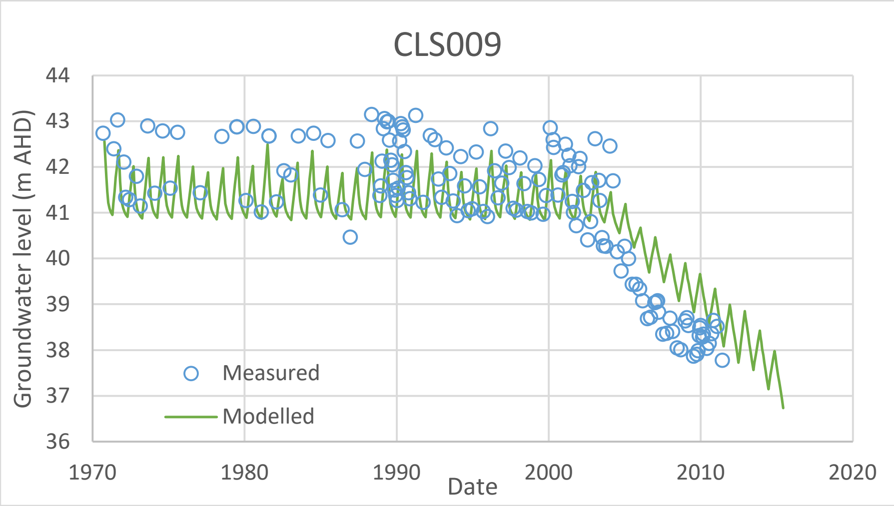
Date
Measured
Modelled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Measured
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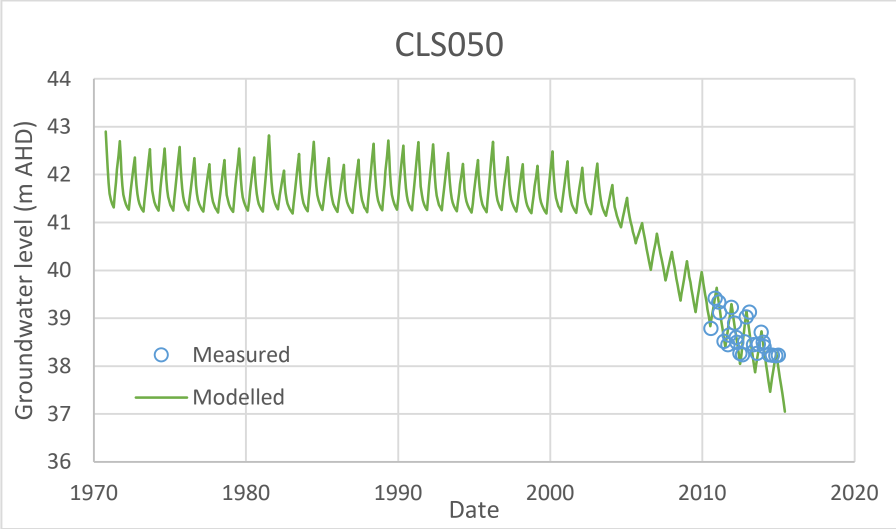
Date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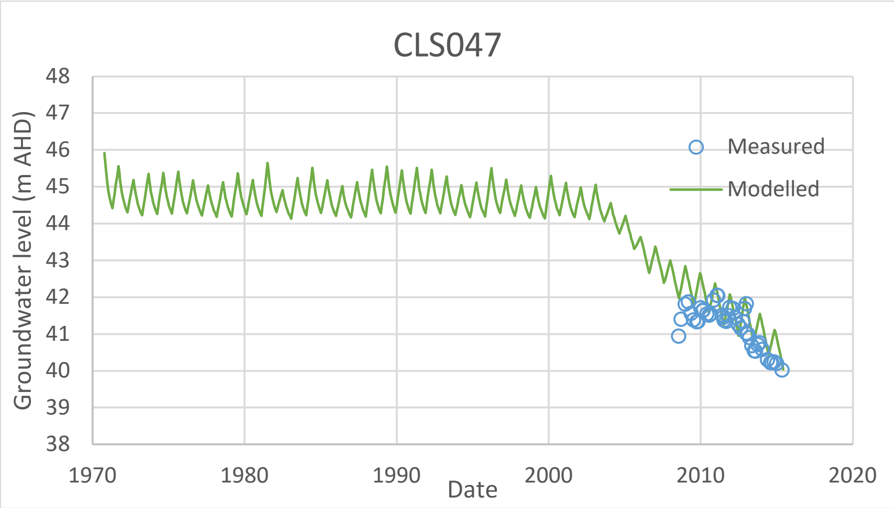
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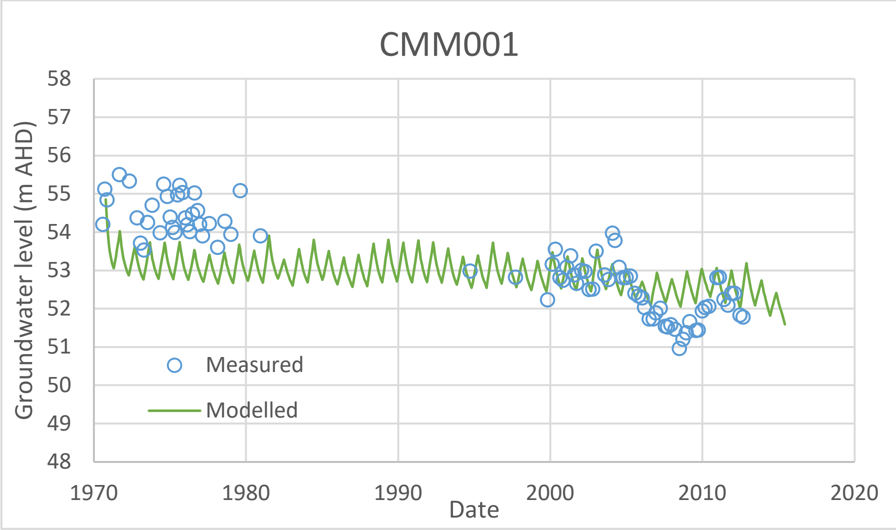
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Modelled
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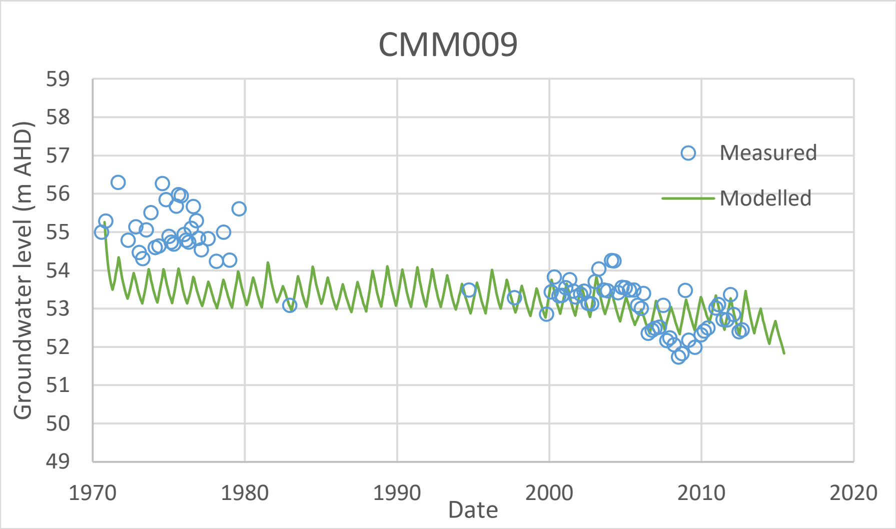
Date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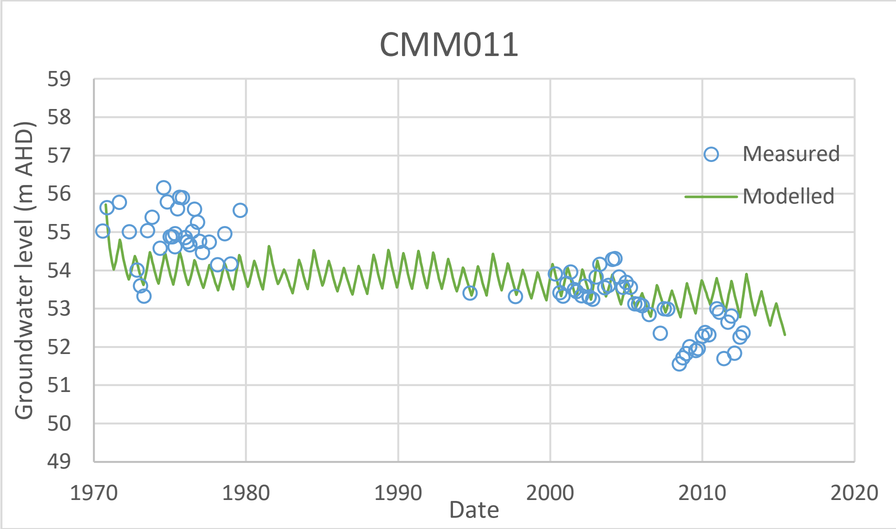
Measured
Modelled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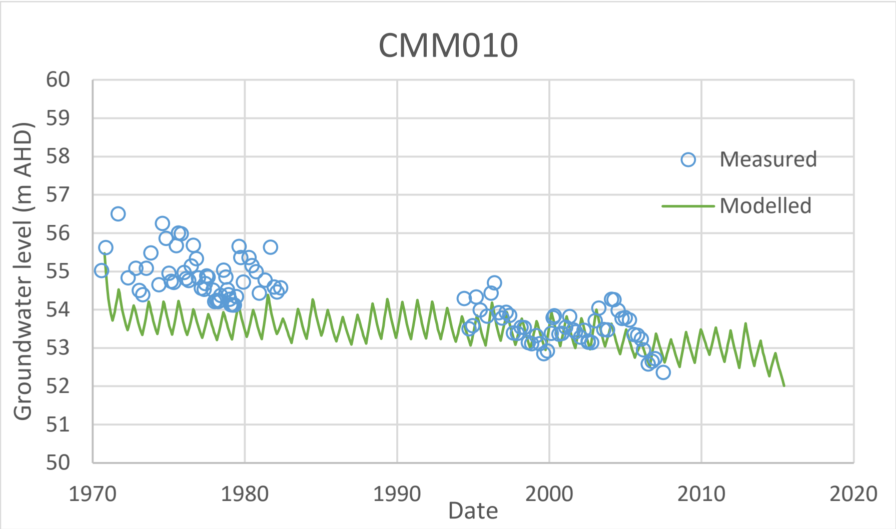
Modelled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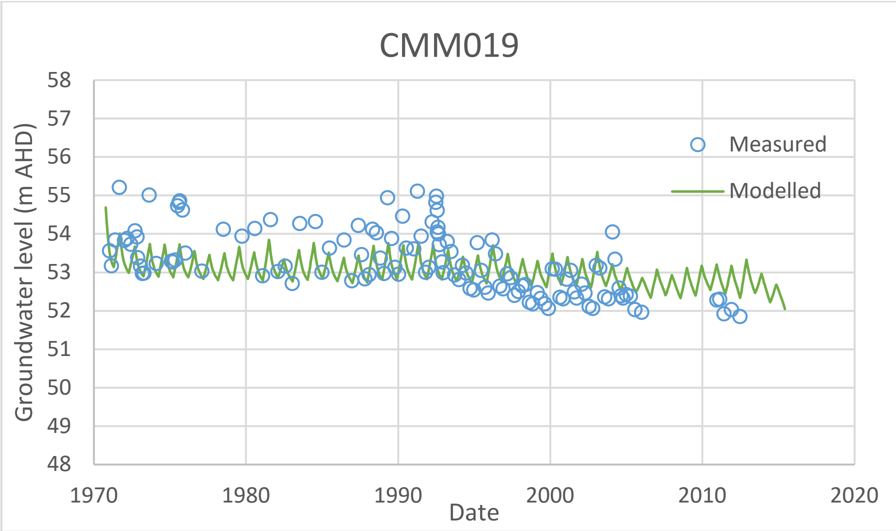
Measured
Modelled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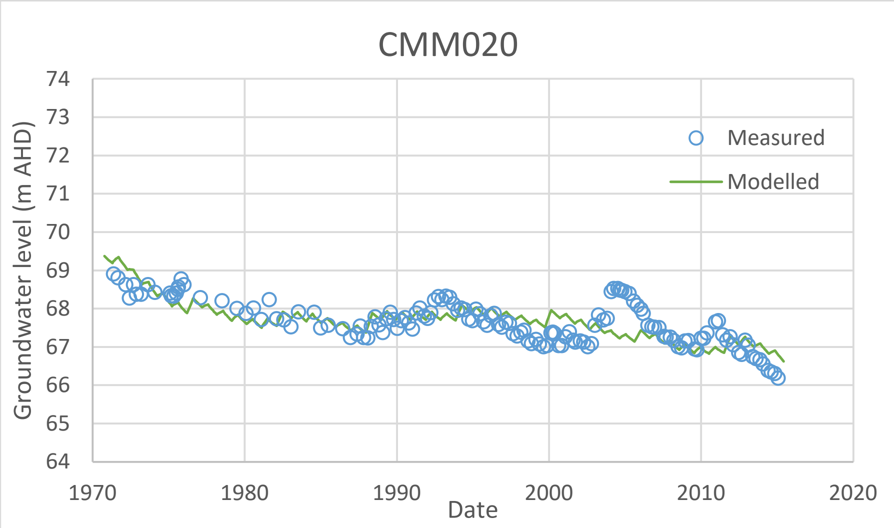
Measured
Modelled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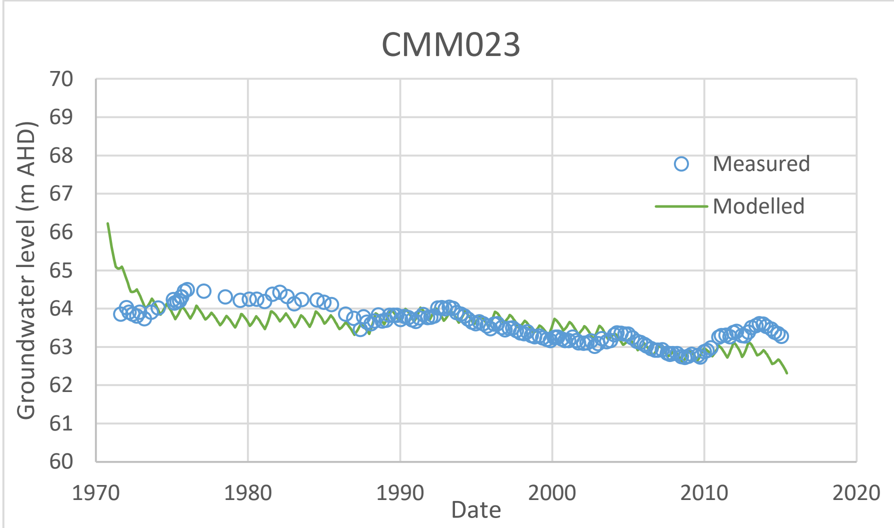
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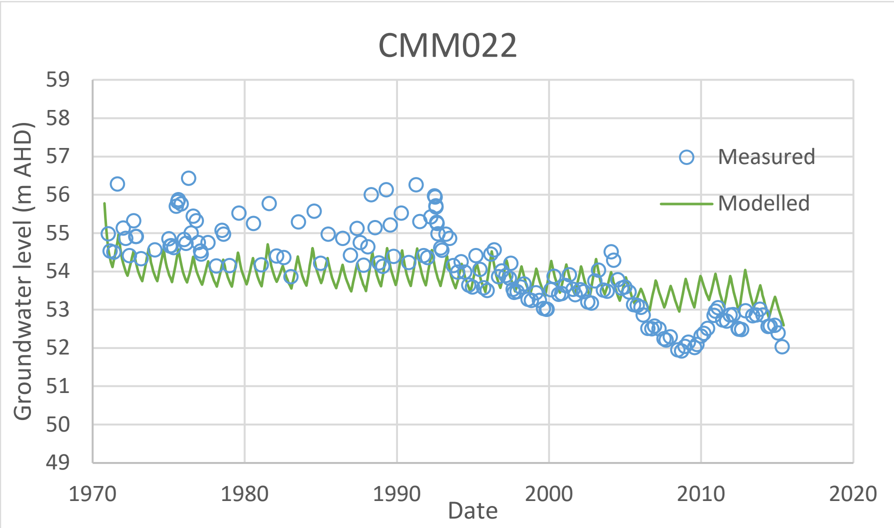
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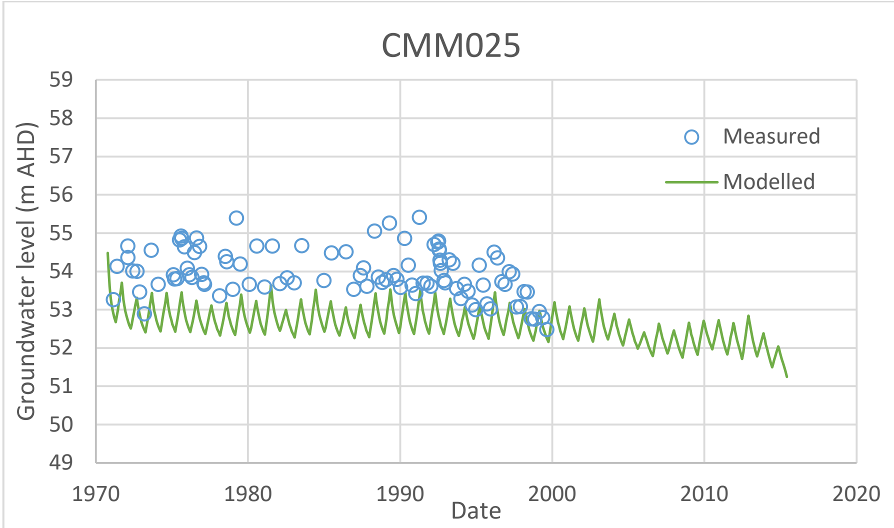
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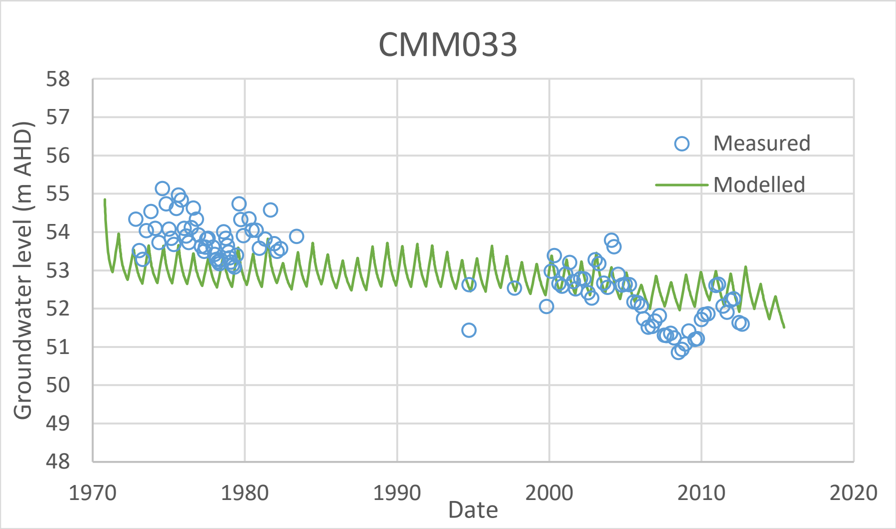
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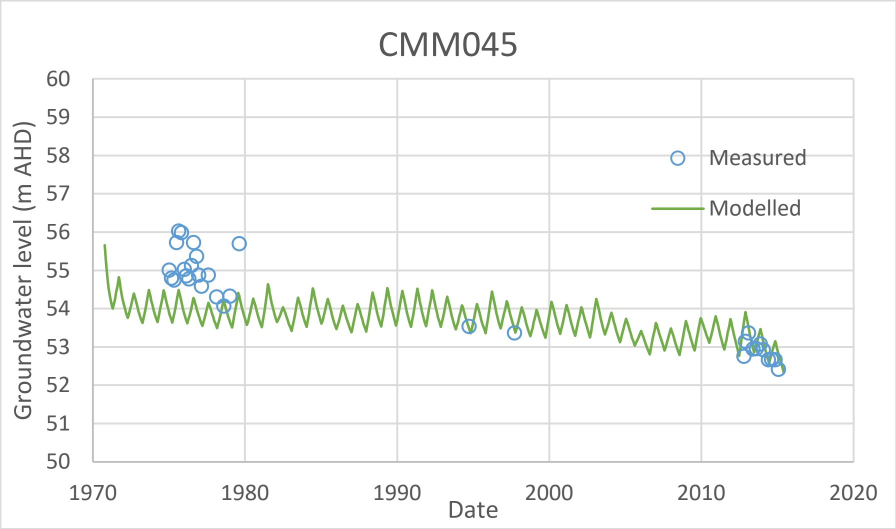
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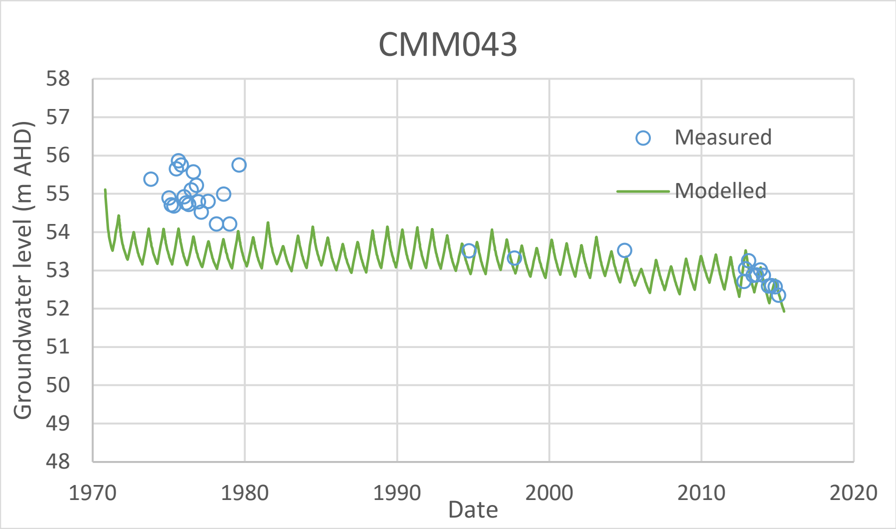
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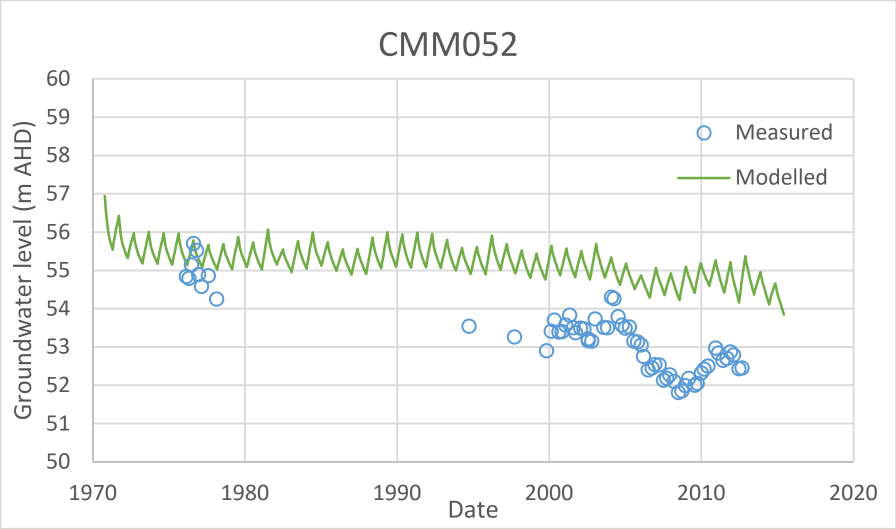
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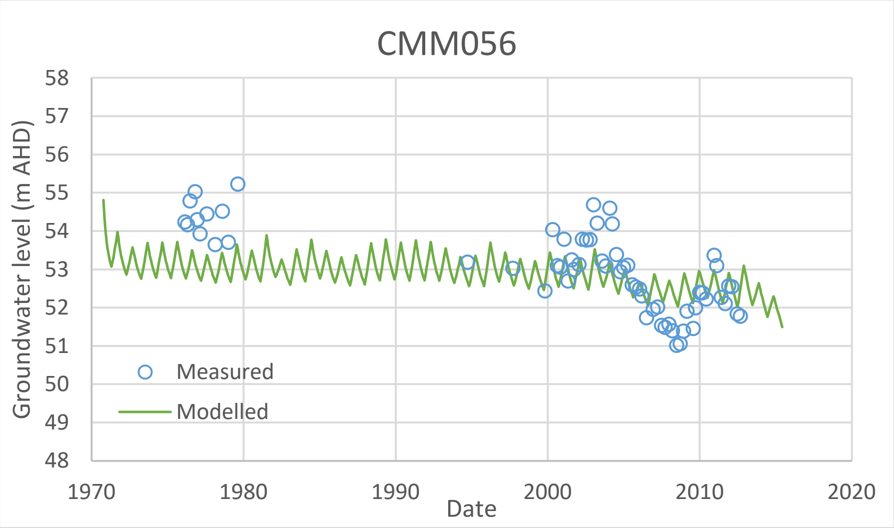
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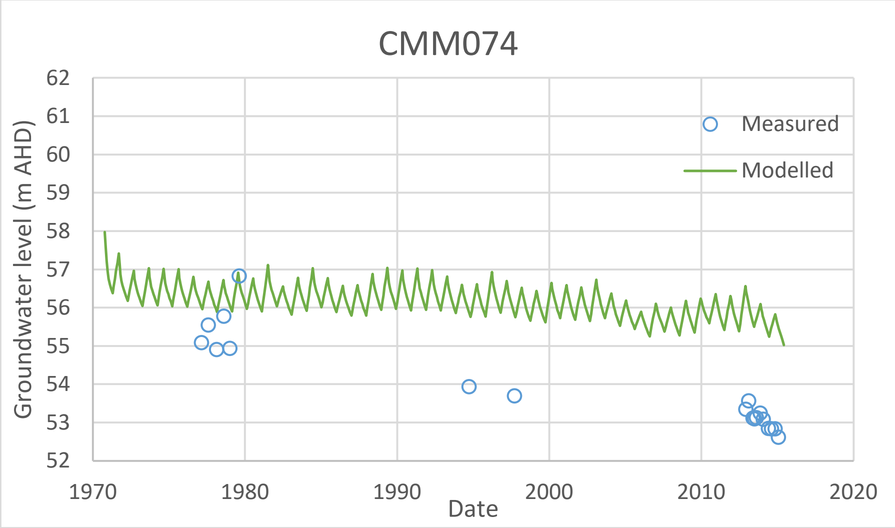
50
49
48
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


CMM074
Groundwater level (m AHD)
62
61
60
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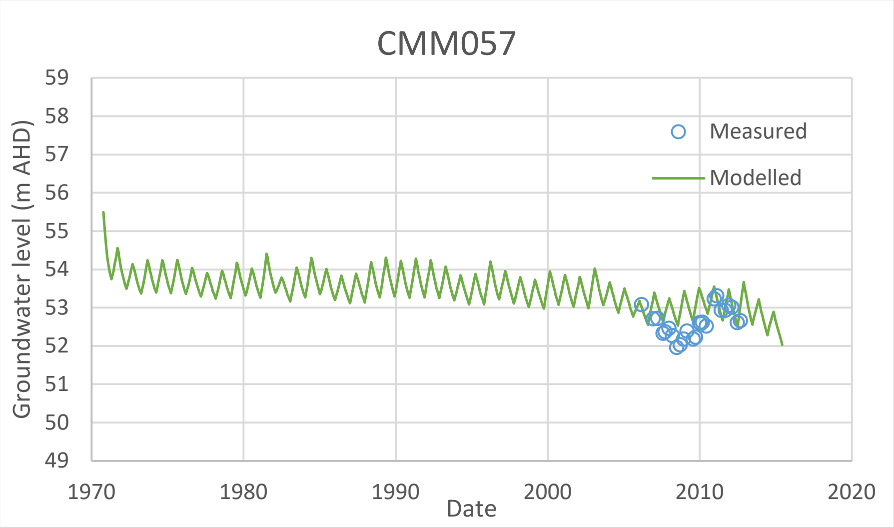
58
57
56
55
54
53
52
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


CMM057
Groundwater level (m AHD)
59
58
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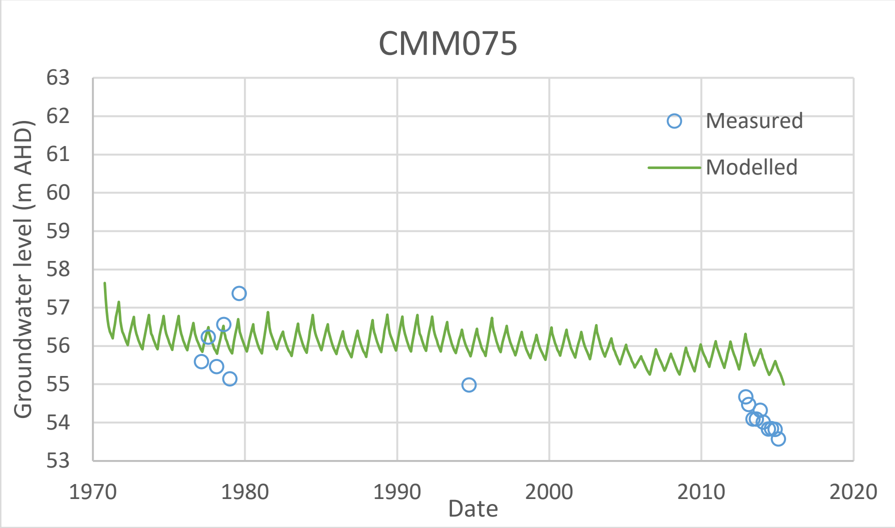
55
54
53
52
51
50
49
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


CMM075
Groundwater level (m AHD)
63
62
61
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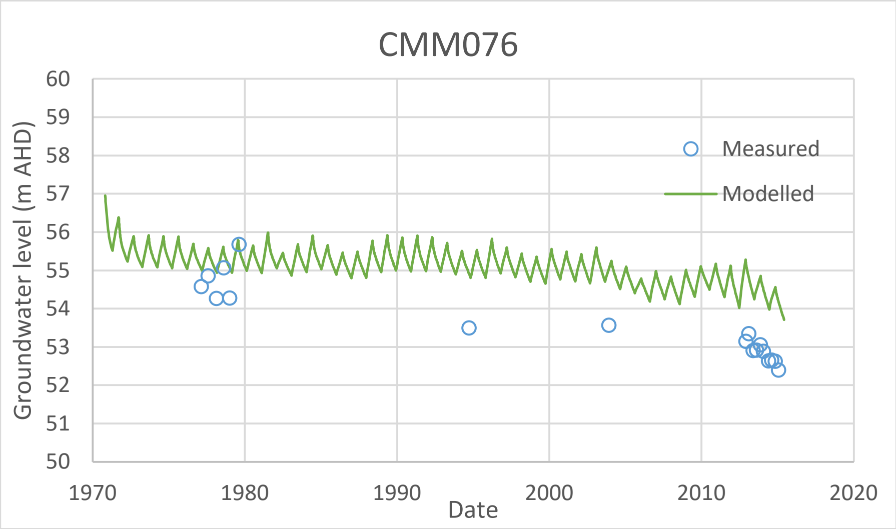
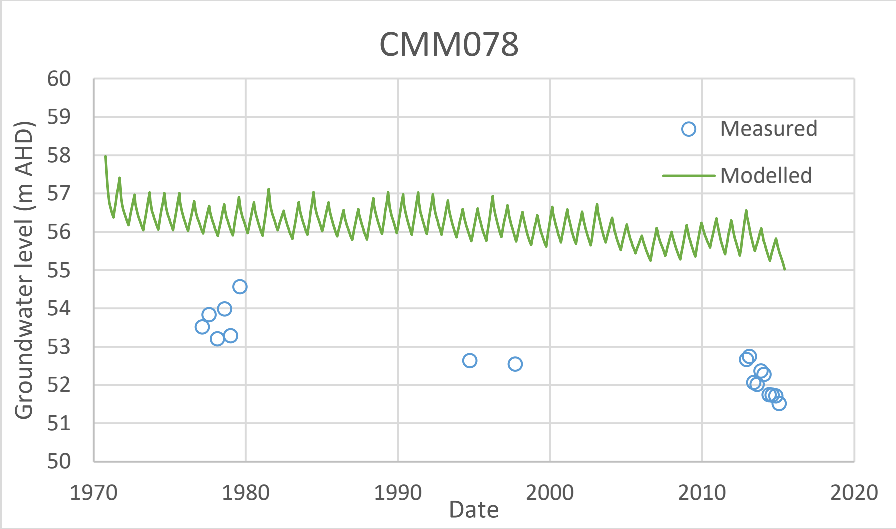
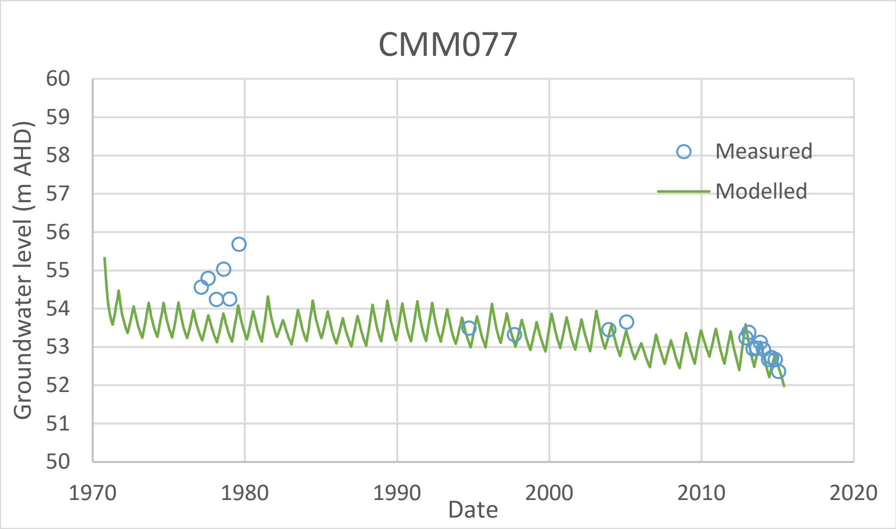
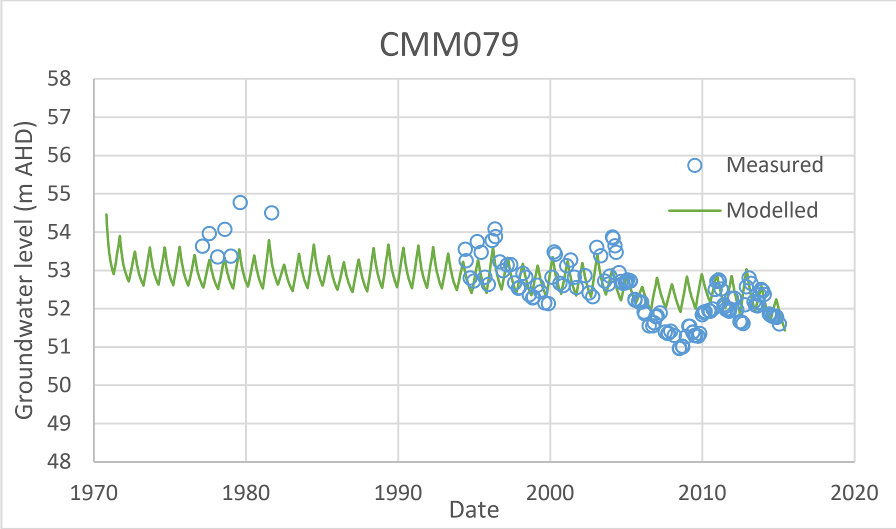
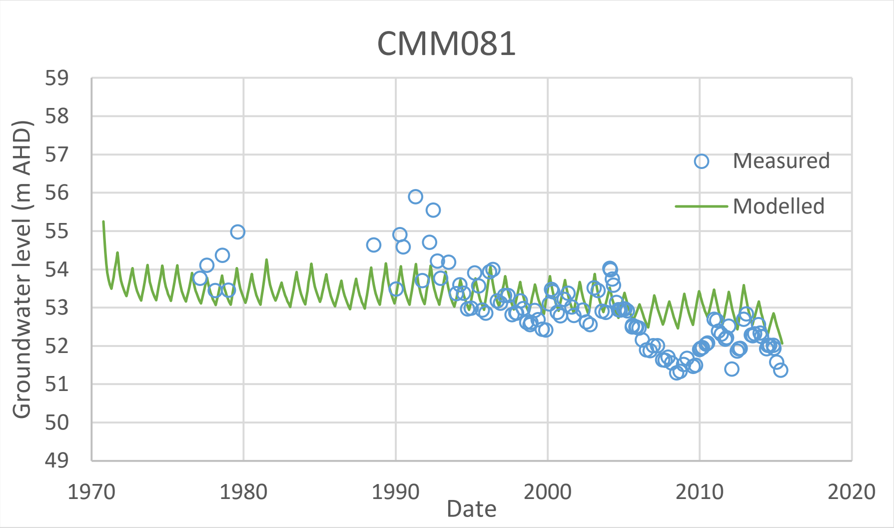
59
58
57
56
55
54
53
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


CMM076
Groundwater level (m AHD)
Date
Measured
Modelled


CMM078
Groundwater level (m AHD)
Date
Measured
Modelled


CMM077
Groundwater level (m AHD)
Date
Measured
Modelled


CMM079
Groundwater level (m AHD)
Date
Measured
Modelled


CMM081
Groundwater level (m AHD)
59
58
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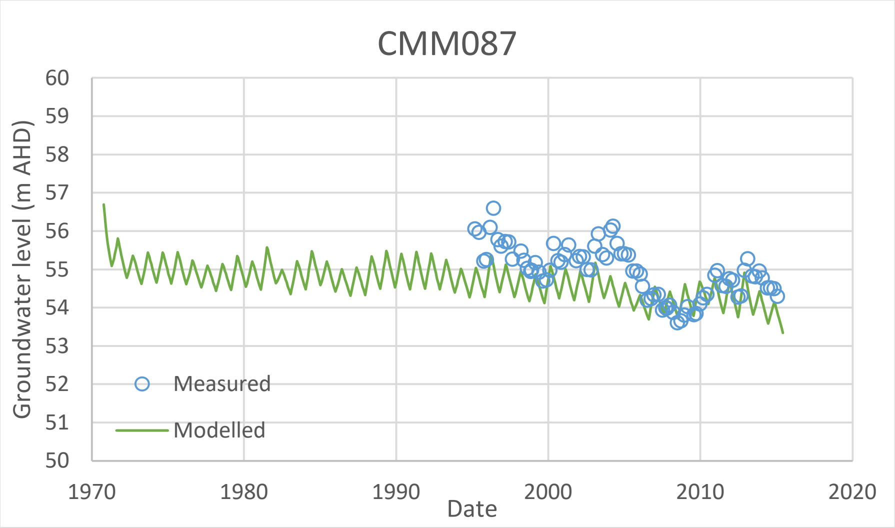
54
53
52
51
50
49
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


CMM087
Groundwater level (m AHD)
60
59
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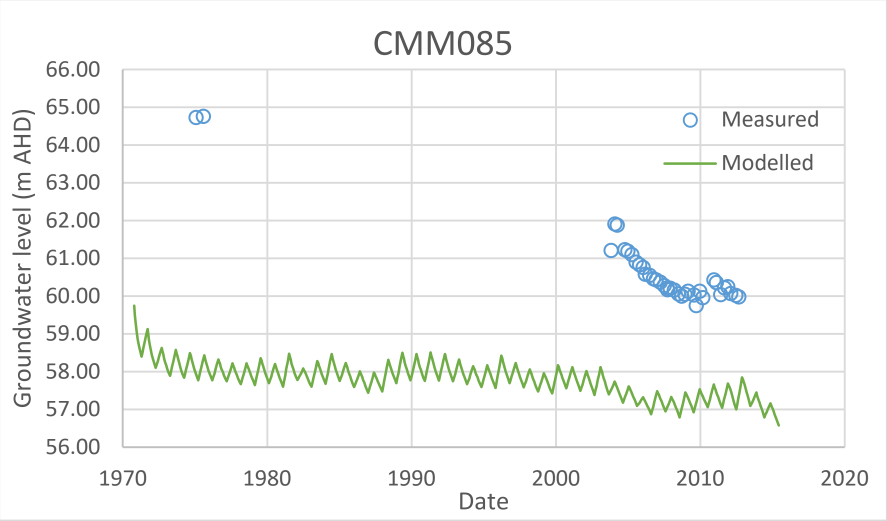
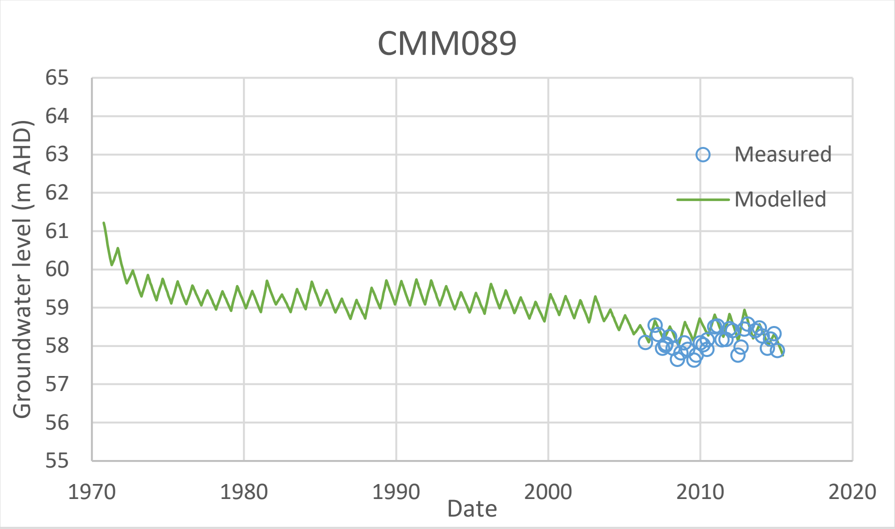
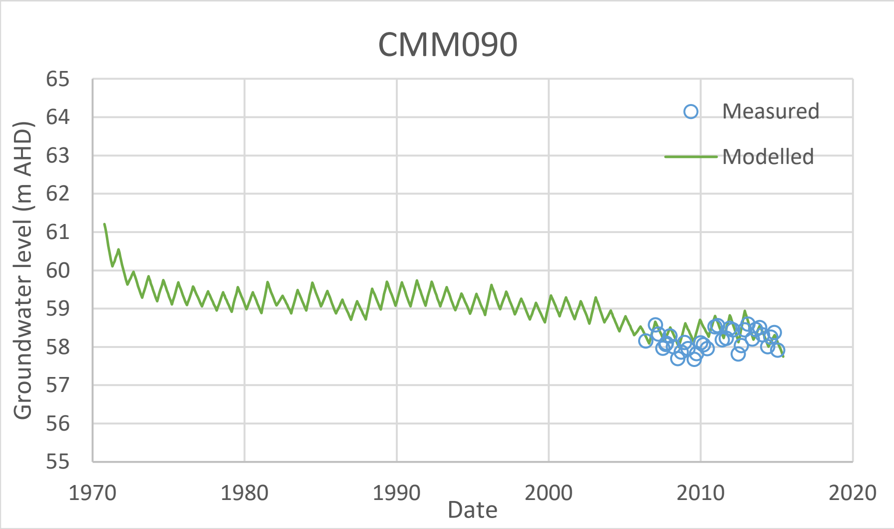
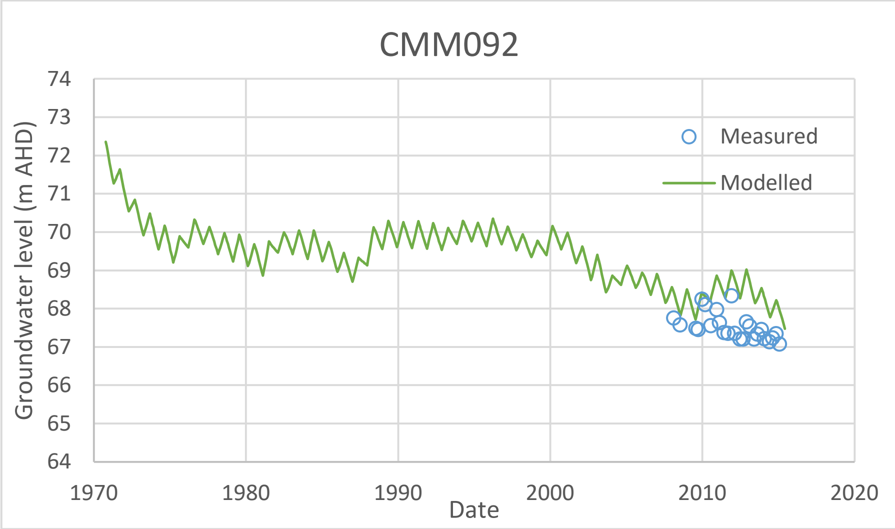
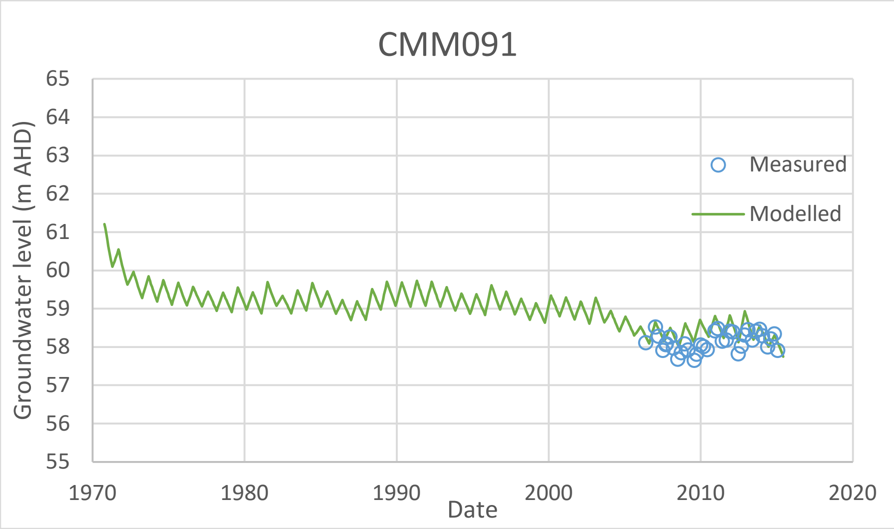
56
55
54
53
52
51
50
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

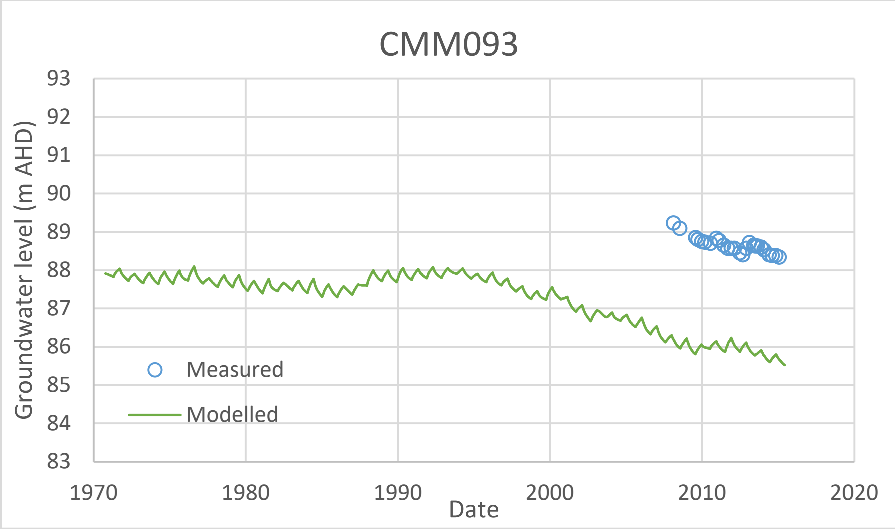

CMM085
Groundwater level (m AHD)
66.00
65.00
64.00
63.00
62.00
61.00
60.00
59.00
58.00
57.00
56.00
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

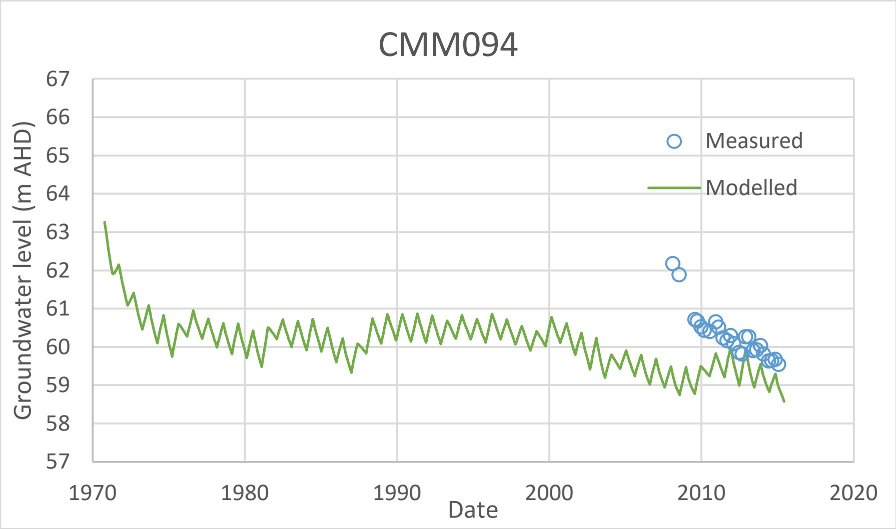

CMM089
Groundwater level (m AHD)
65
64
63
62
61
60
59
58
57
56
55
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

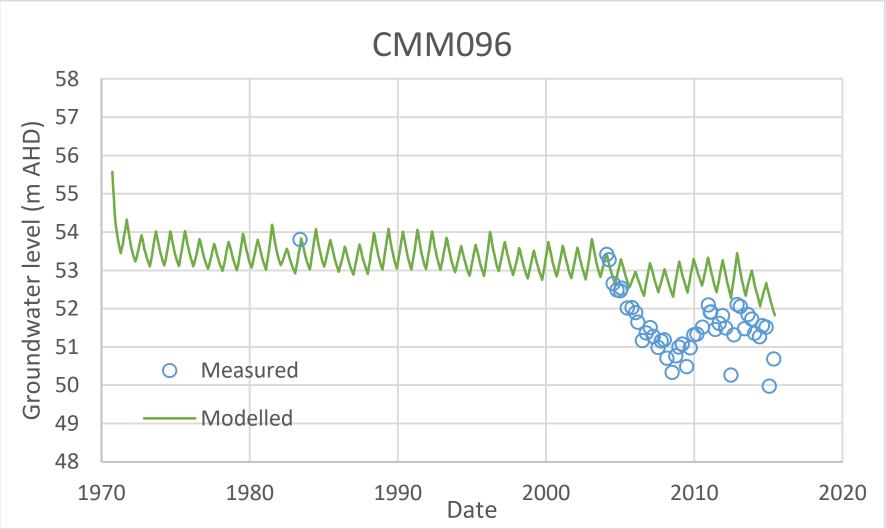

CMM090
Groundwater level (m AHD)
65
64
63
62
61
60
59
58
57
56
55
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

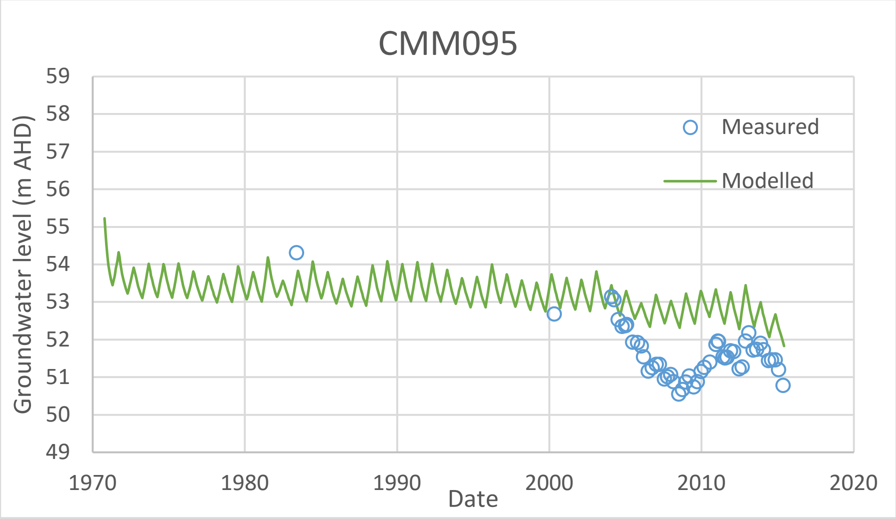

CMM092
Groundwater level (m AHD)
74
73
72
71
70
69
68
67
66
65
64
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

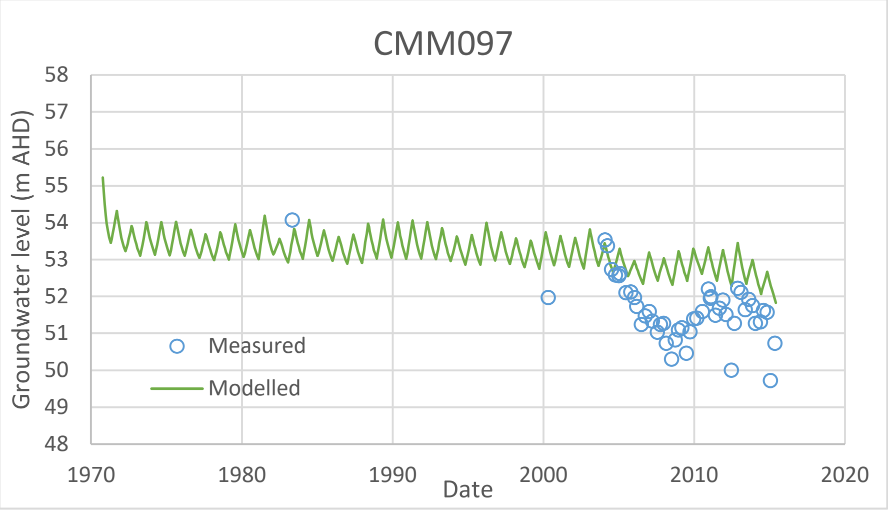

CMM091
Groundwater level (m AHD)
65
64
63
62
61
60
59
58
57
56
55
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


CMM093
Groundwater level (m AHD)
93
92
91
90
89
88
87
86
85
84
83
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


CMM094
Groundwater level (m AHD)
Date
Measured
Modelled
67
66
65
64
63
62
61
60
59
58
57
1970
1980
1990
2000
2010
2020


CMM096
Groundwater level (m AHD)
Date
Measured
Modelled
58
57
56
55
54
53
52
51
50
49
48
1970
1980
1990
2000
2010
2020


CMM095
Groundwater level (m AHD)
Date
Measured
Modelled
59
58
57
56
55
54
53
52
51
50
49
1970
1980
1990
2000
2010
2020


CMM097
Groundwater level (m AHD)
Date
Measured
Modelled
58
57
56
55
54
53
52
51
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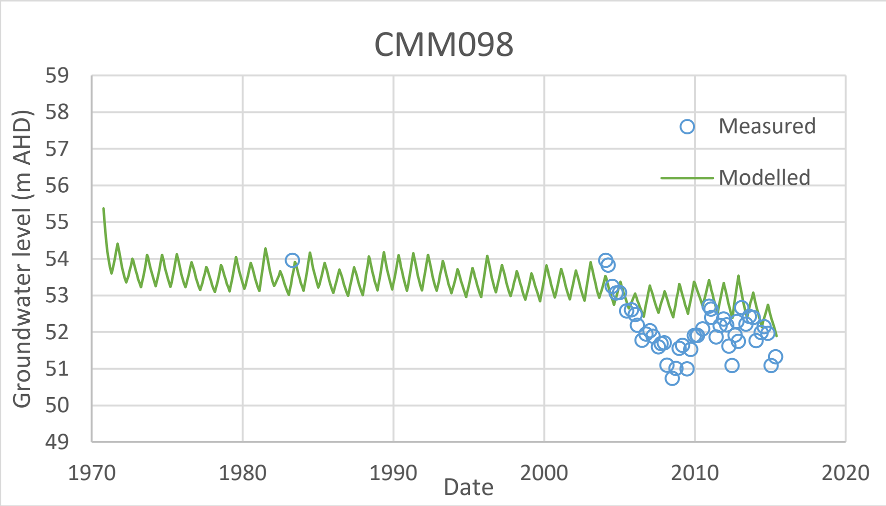
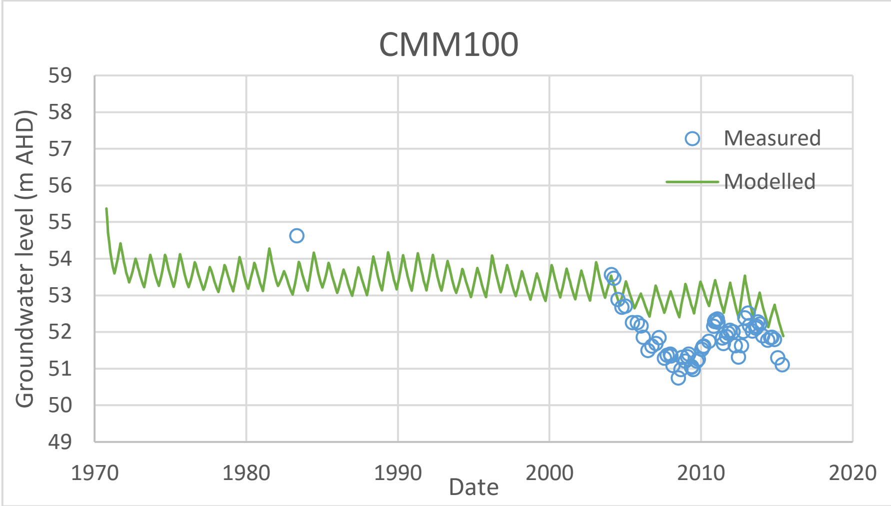
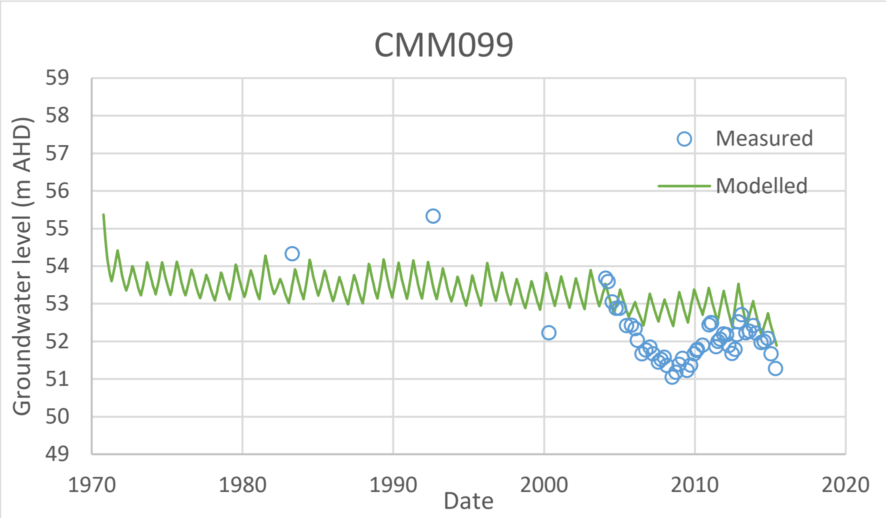
49
48
1970
1980
1990
2000
2010
2020



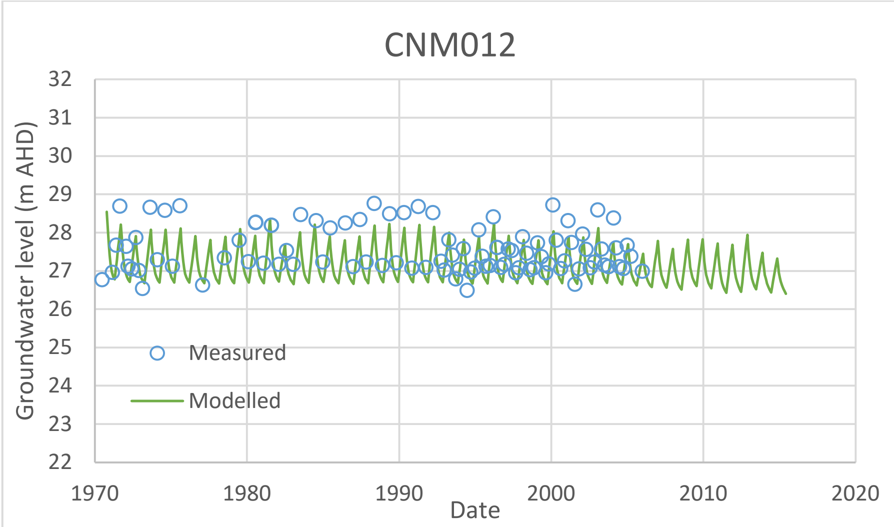
CMM098
Groundwater level (m AHD)
Date
Measured
Modelled



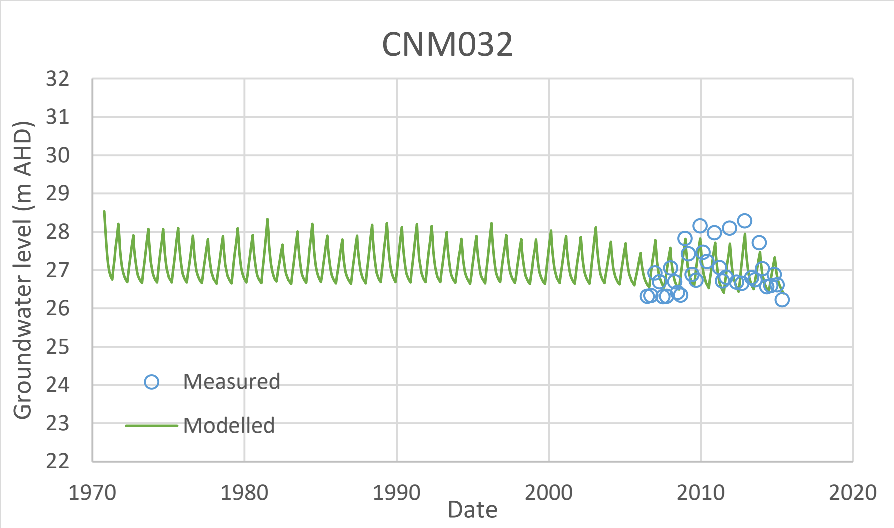
CMM100
Groundwater level (m AHD)
Date
Measured
Modelled

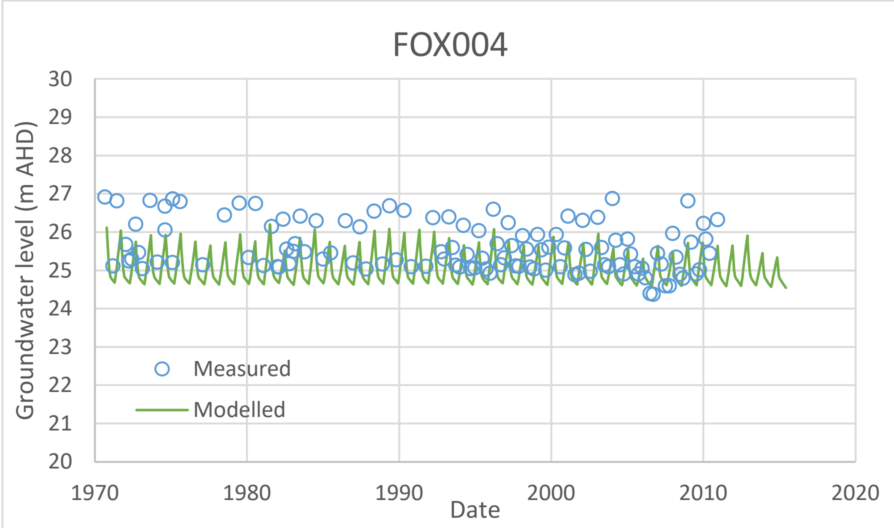


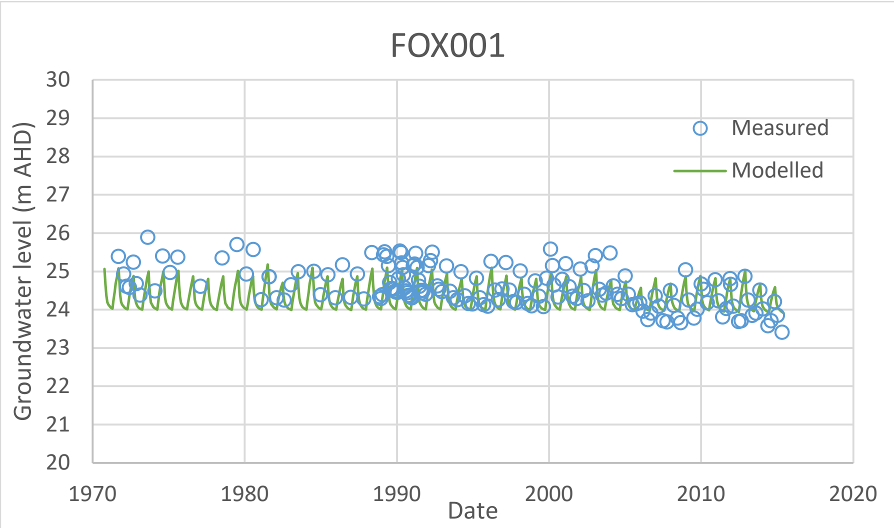
CMM099
Groundwater level (m AHD)
Date
Measured
Modelled

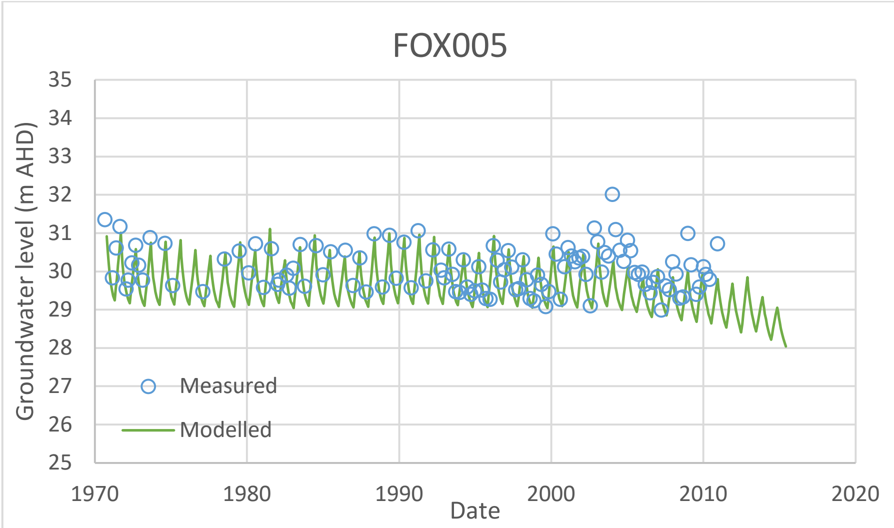


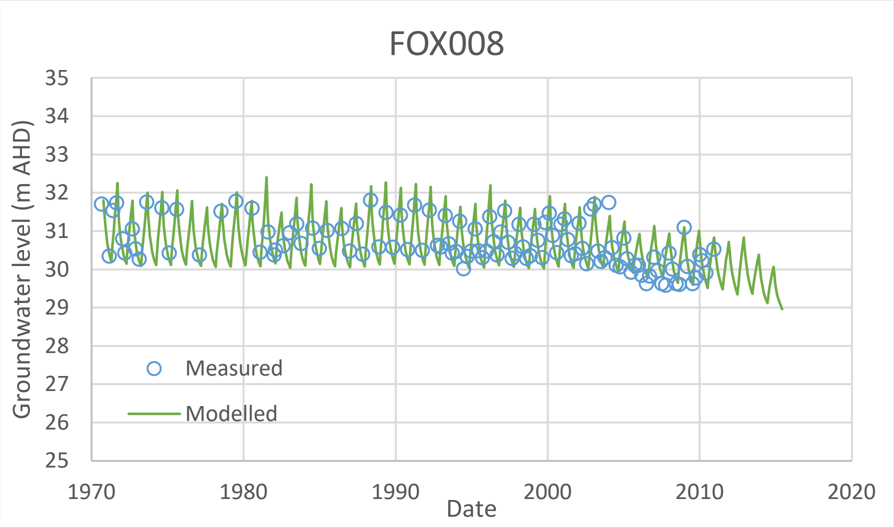
CNM012
Groundwater level (m AHD)
Date
Measured
Modelled

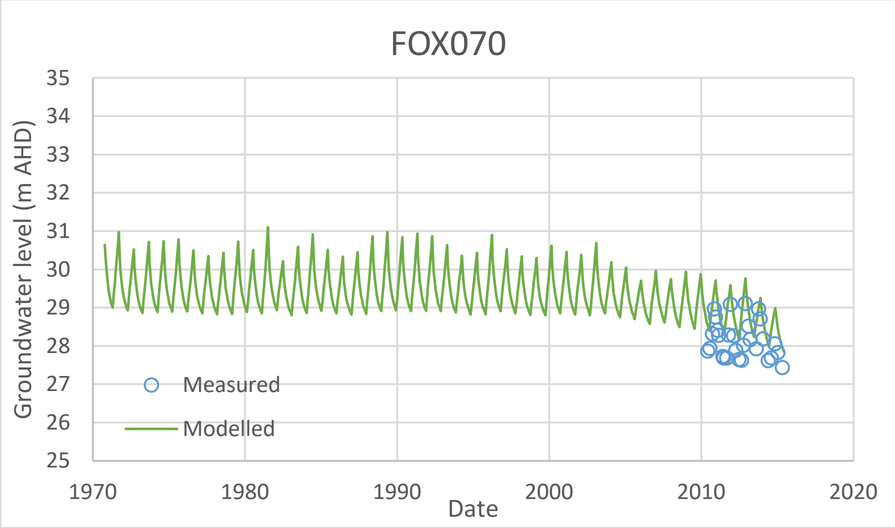

CNM032
Groundwater level (m AHD)
32
31
30
29
28
27
26
25
24
23
22
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

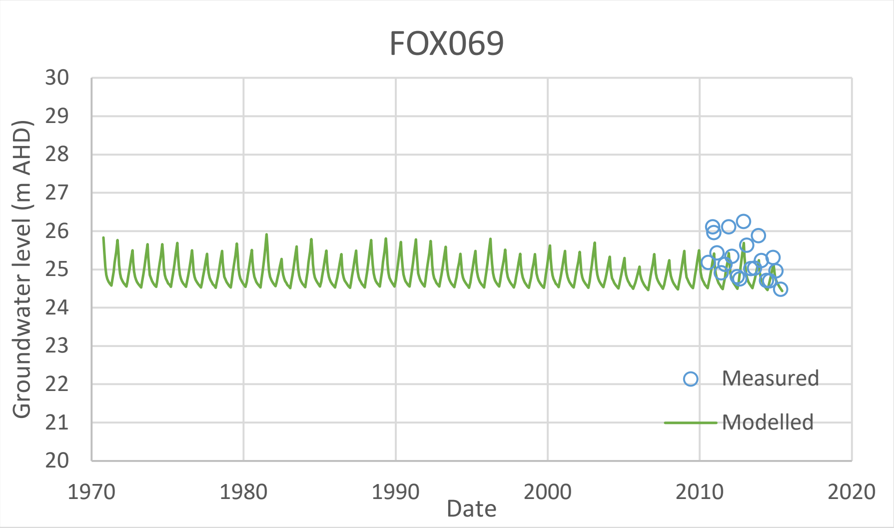

FOX004
Groundwater level (m AHD)
30
29
28
27
26
25
24
23
22
21
20
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

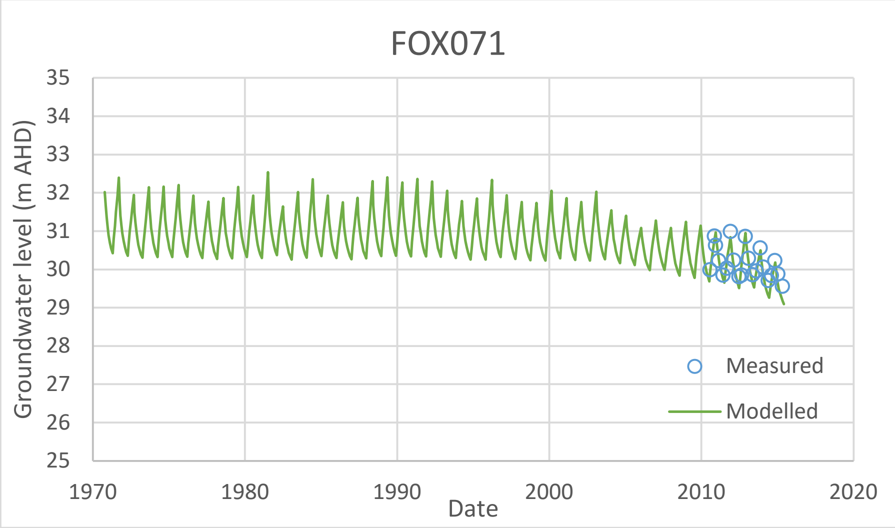

FOX001
Groundwater level (m AHD)
30
29
28
27
26
25
24
23
22
21
20
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

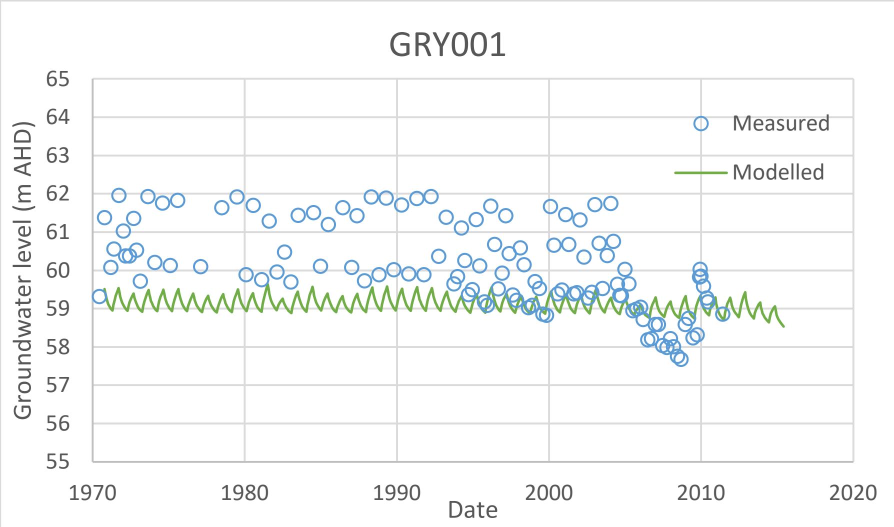

FOX005
Groundwater level (m AHD)
35
34
33
32
31
30
29
28
27
26
25
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

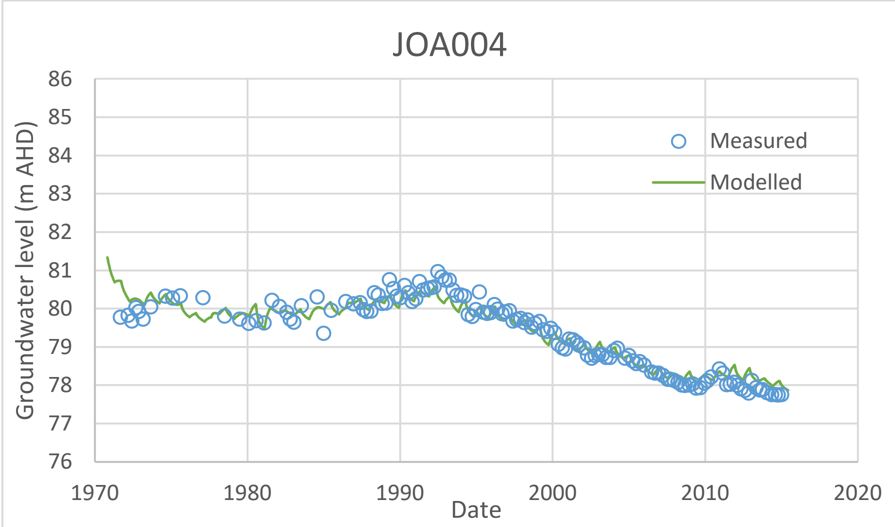

FOX008
Groundwater level (m AHD)
Date
Measured
Modelled

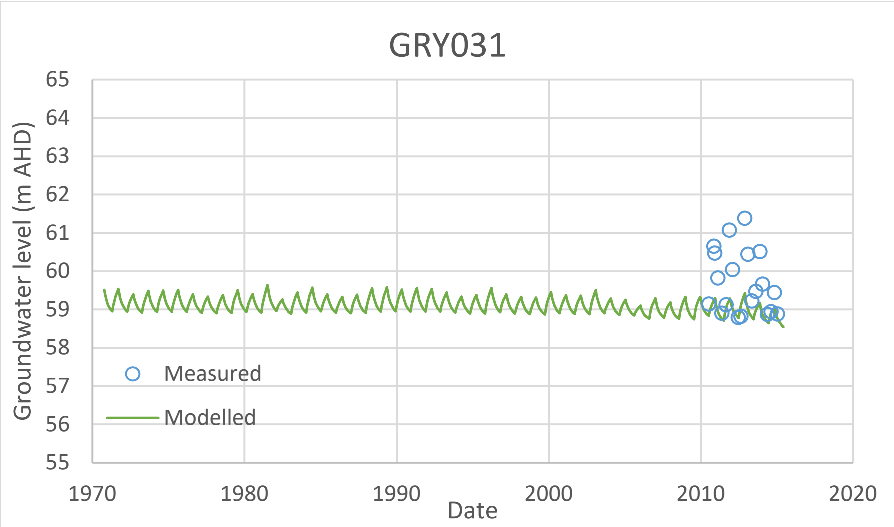

FOX070
Groundwater level (m AHD)
Date
Measured
Modelled

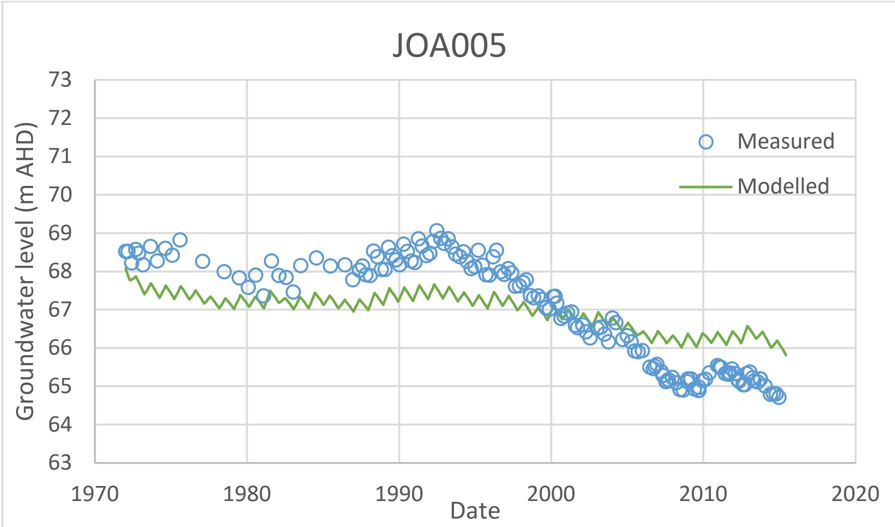

FOX069
Groundwater level (m AHD)
Date
Measured
Modelled

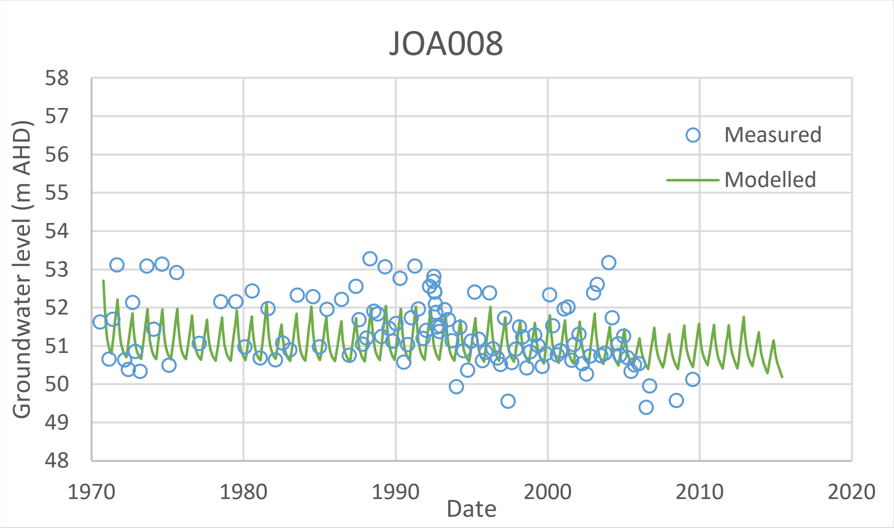

FOX071
Groundwater level (m AHD)
Date
Measured
Modelled

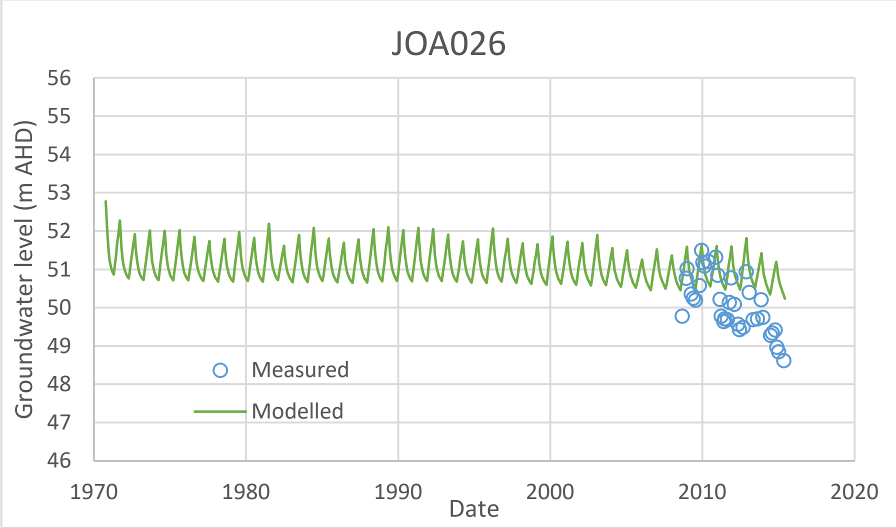

GRY001
Groundwater level (m AHD)
65
64
63
62
61
60
59
58
57
56
55
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

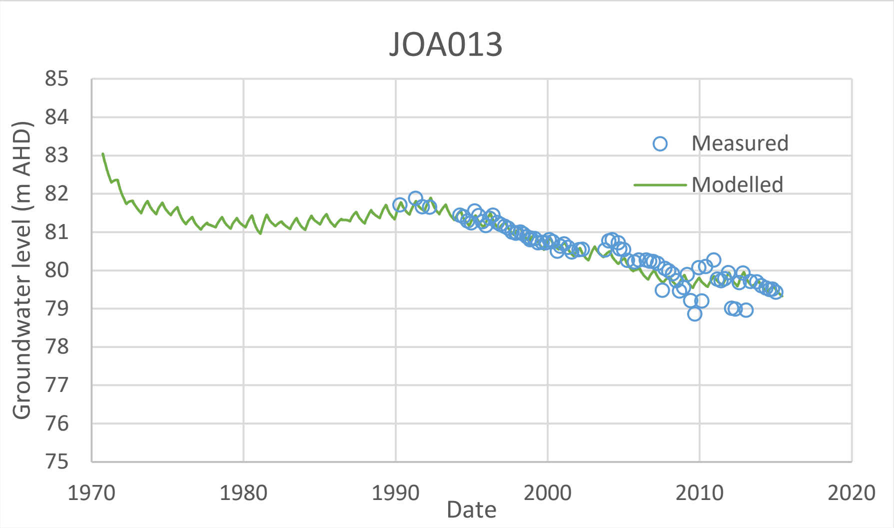

JOA004
Groundwater level (m AHD)
86
85
84
83
82
81
80
79
78
77
76
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

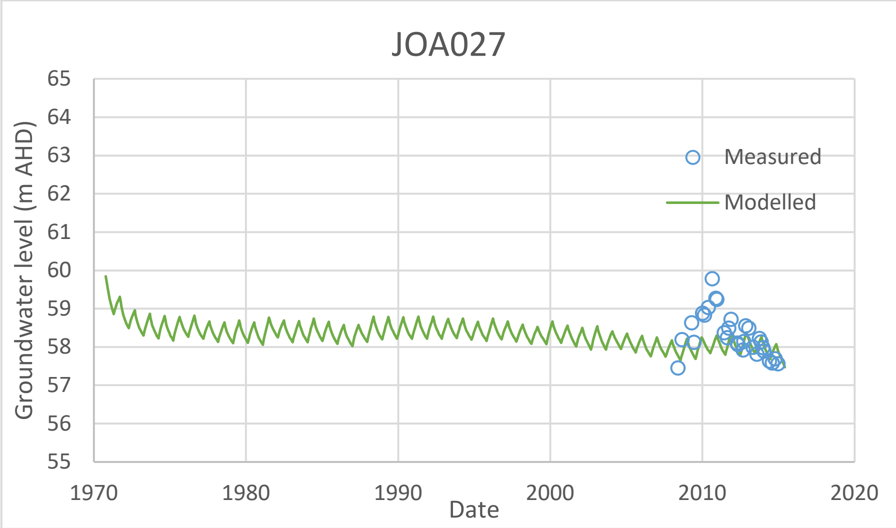

GRY031
Groundwater level (m AHD)
65
64
63
62
61
60
59
58
57
56
55
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

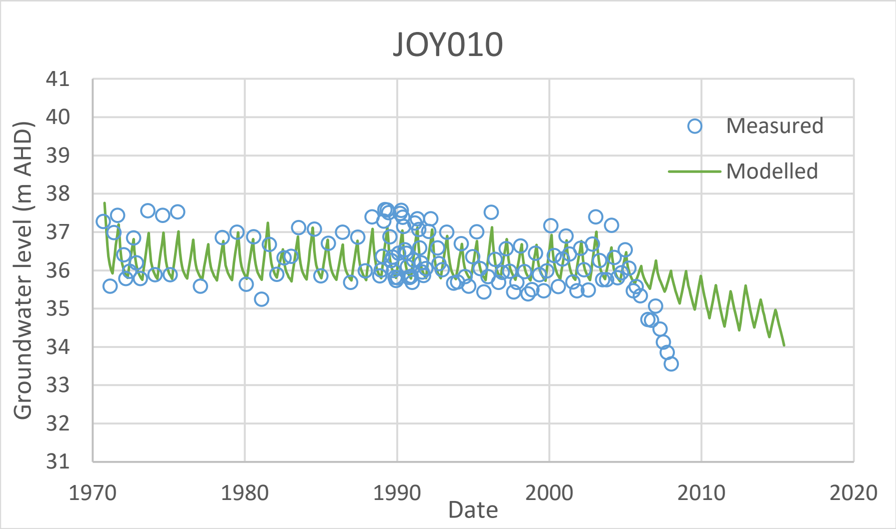

JOA005
Groundwater level (m AHD)
73
72
71
70
69
68
67
66
65
64
63
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

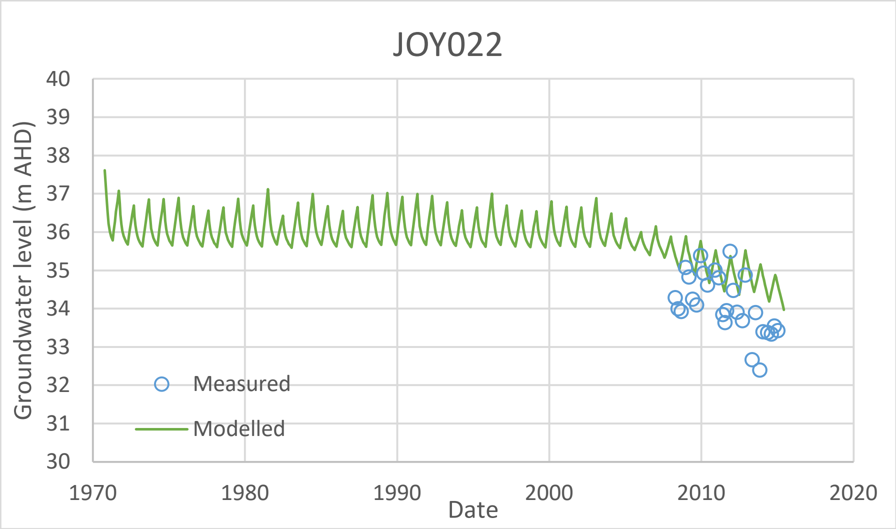

JOA008
Groundwater level (m AHD)
58
57
56
55
54
53
52
51
50
49
48
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

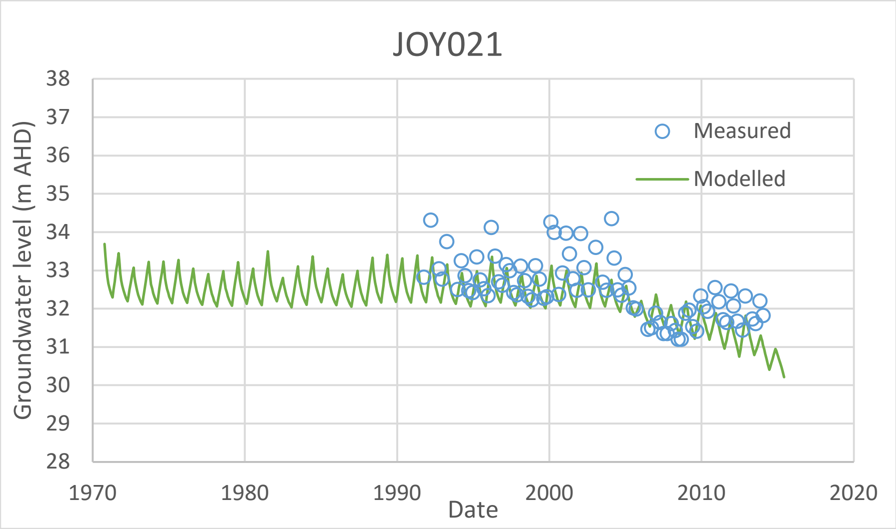

JOA026
Groundwater level (m AHD)
56
55
54
53
52
51
50
49
48
47
46
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

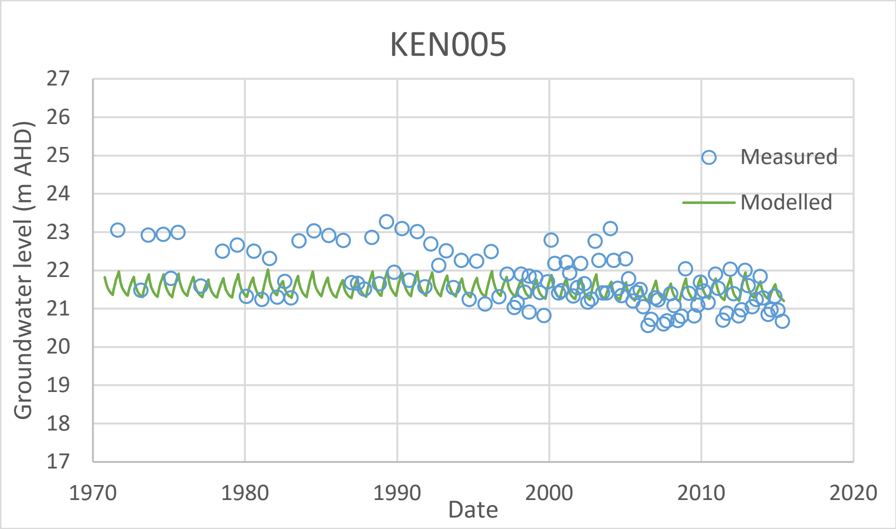

JOA013
Groundwater level (m AHD)
85
84
83
82
81
80
79
78
77
76
75
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

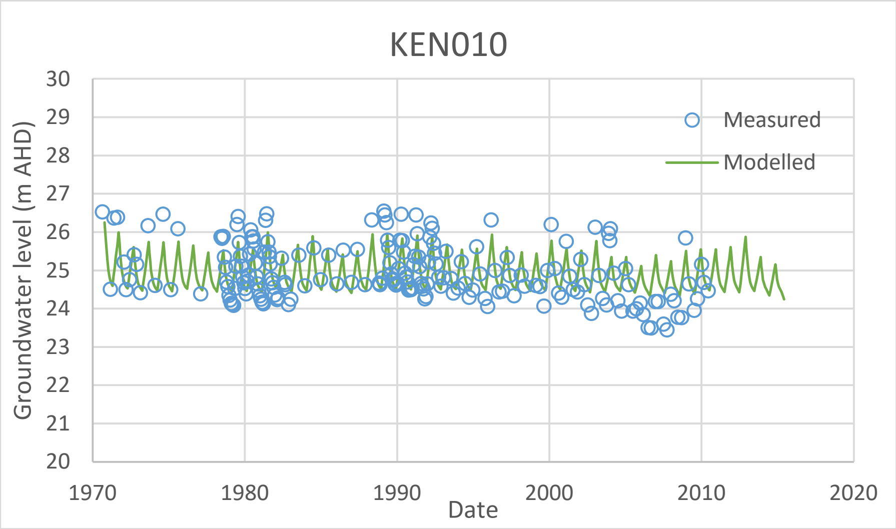

JOA027
Groundwater level (m AHD)
65
64
63
62
61
60
59
58
57
56
55
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled

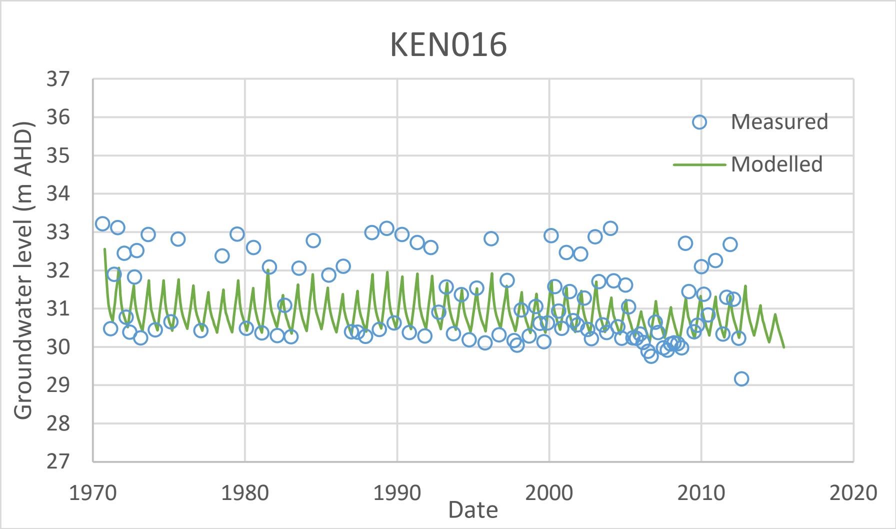

JOY010
Groundwater level (m AHD)
41
40
39
38
37
36
35
34
33
32
31
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date

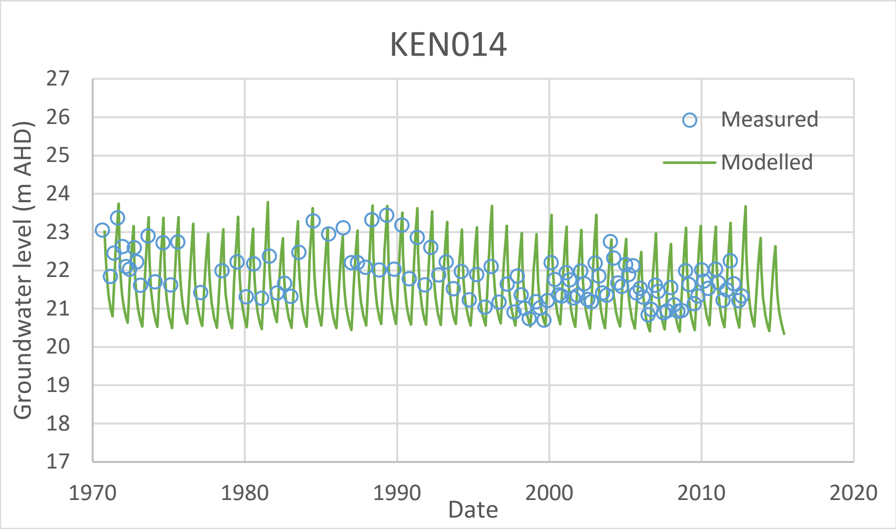

JOY022
Groundwater level (m AHD)
40
39
38
37
36
35
34
33
32
31
30
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


JOY021
Groundwater level (m AHD)
38
37
36
35
34
33
32
31
30
29
28
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


KEN005
Groundwater level (m AHD)
27
26
25
24
23
22
21
20
19
18
17
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


KEN010
Groundwater level (m AHD)
30
29
28
27
26
25
24
23
22
21
20
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


KEN016
Groundwater level (m AHD)
37
36
35
34
33
32
31
30
29
28
27
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


KEN014
Groundwater level (m AHD)
27
26
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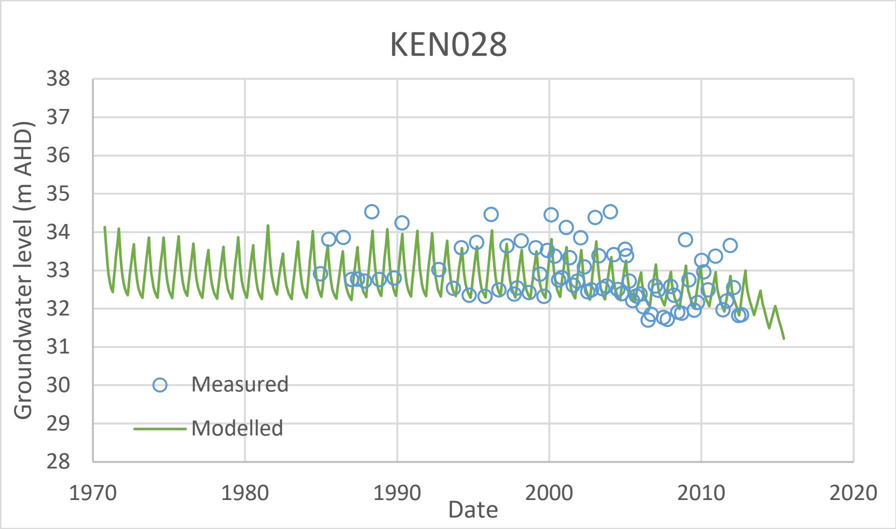
23
22
21
20
19
18
17
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


KEN028
Groundwater level (m AHD)
38
37
36
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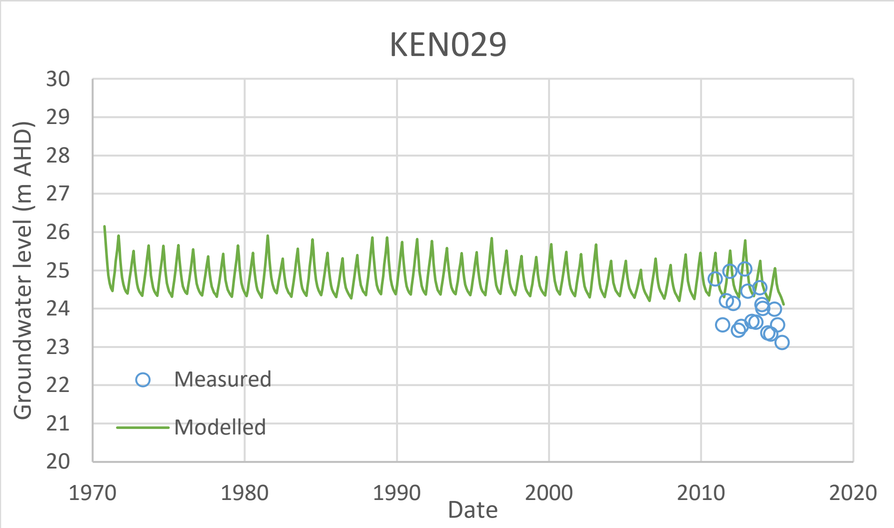
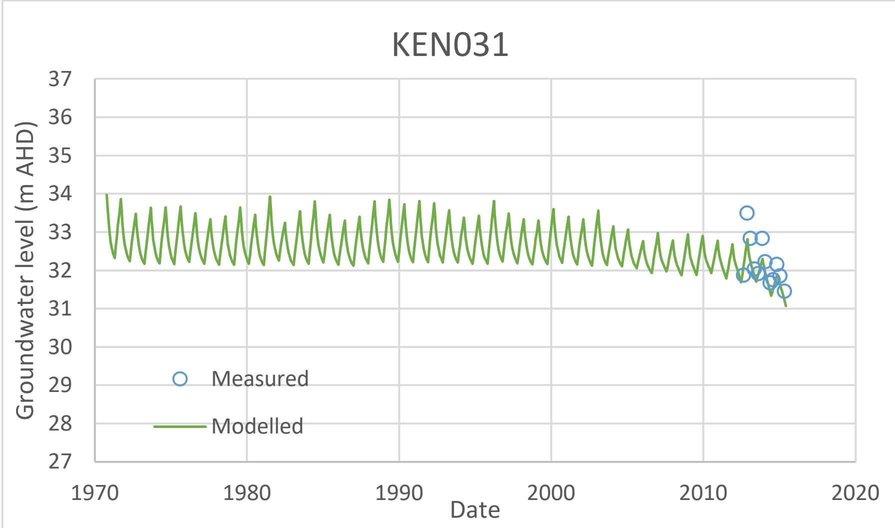
34
33
32
31
30
29
28
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


KEN029
Groundwater level (m AHD)
Date
Measured
Modelled
30
29
28
27
26
25
24
23
22
21
20
1970
1980
1990
2000
2010
2020


KEN031
Groundwater level (m AHD)
Date
Measured
Modelled
37
36
35
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33
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31
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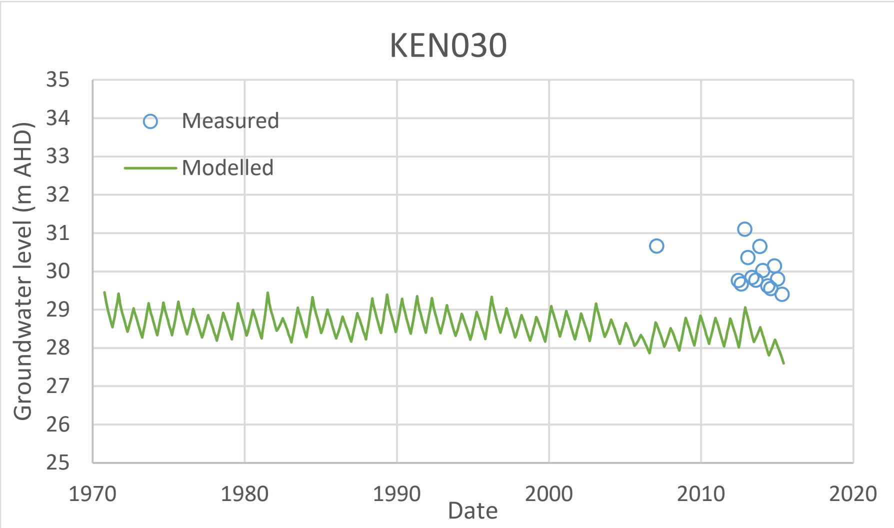
28
27
1970
1980
1990
2000
2010
2020


KEN030
Groundwater level (m AHD)
Date
Measured
Modelled
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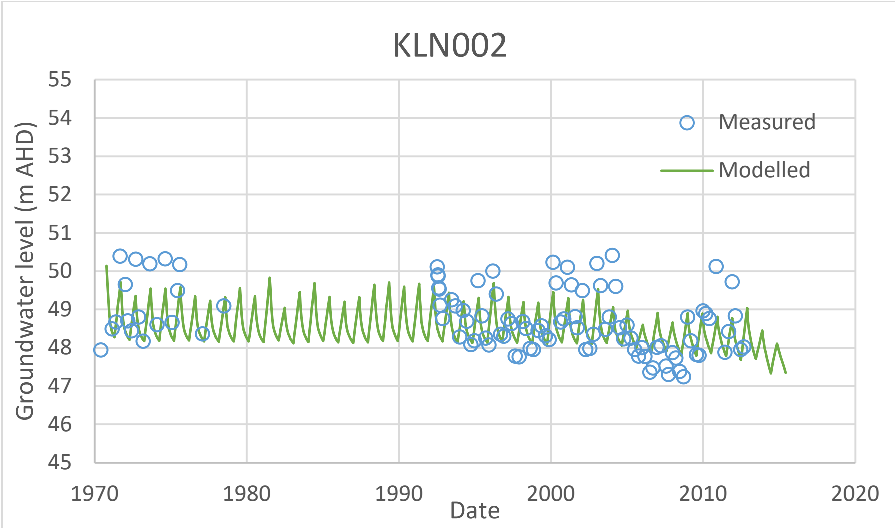
1970
1980
1990
2000
2010
2020


KLN002
Groundwater level (m AHD)
Date
Measured
Modelled
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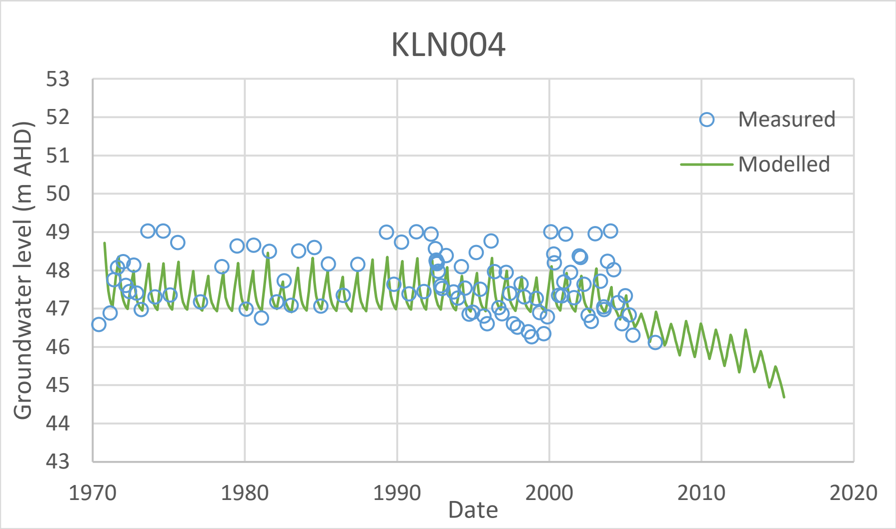
1970
1980
1990
2000
2010
2020


KLN004
Groundwater level (m AHD)
Date
Measured
Modelled
53
52
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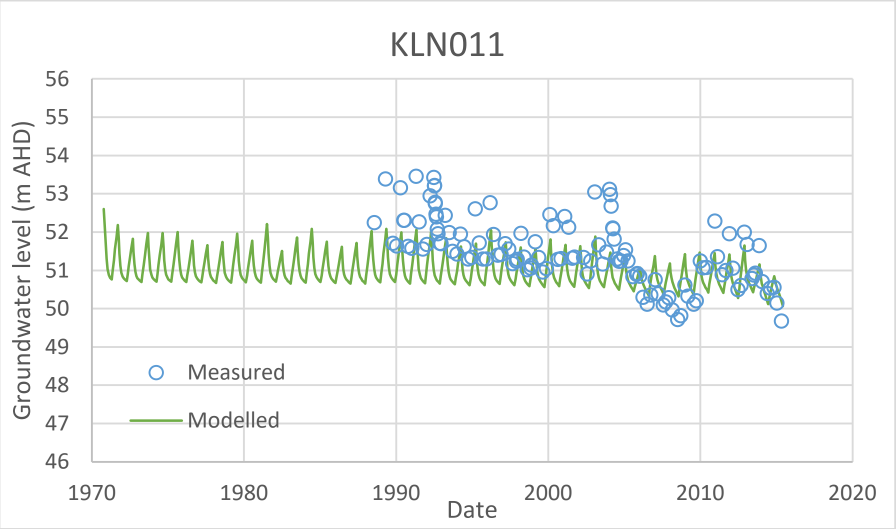
43
1970
1980
1990
2000
2010
2020


KLN011
Groundwater level (m AHD)
Date
Measured
Modelled
56
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54
53
52
51
50
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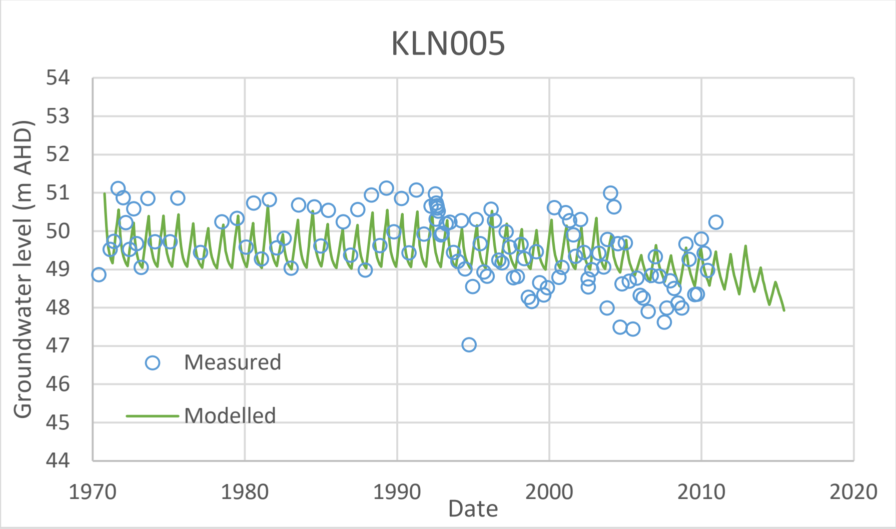
1970
1980
1990
2000
2010
2020


KLN005
Groundwater level (m AHD)
Date
Measured
Modelled
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1970
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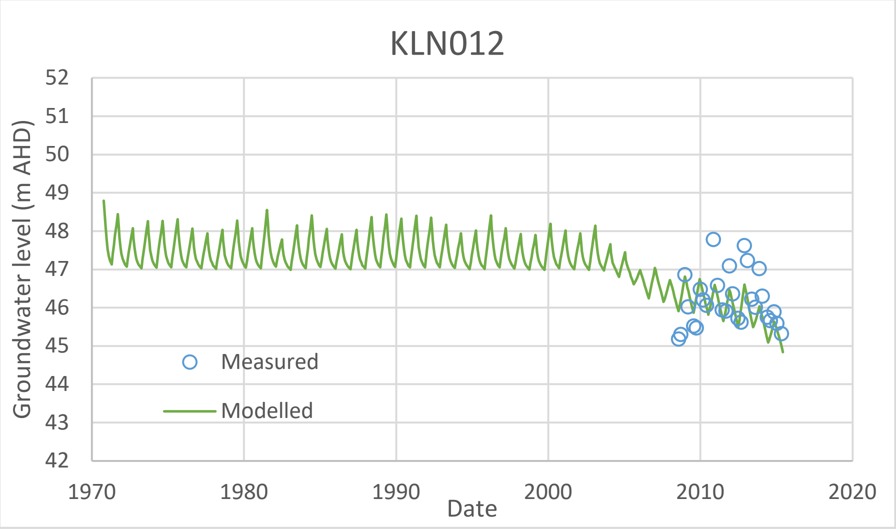
2000
2010
2020


KLN012
Groundwater level (m AHD)
Date
Measured
Modelled
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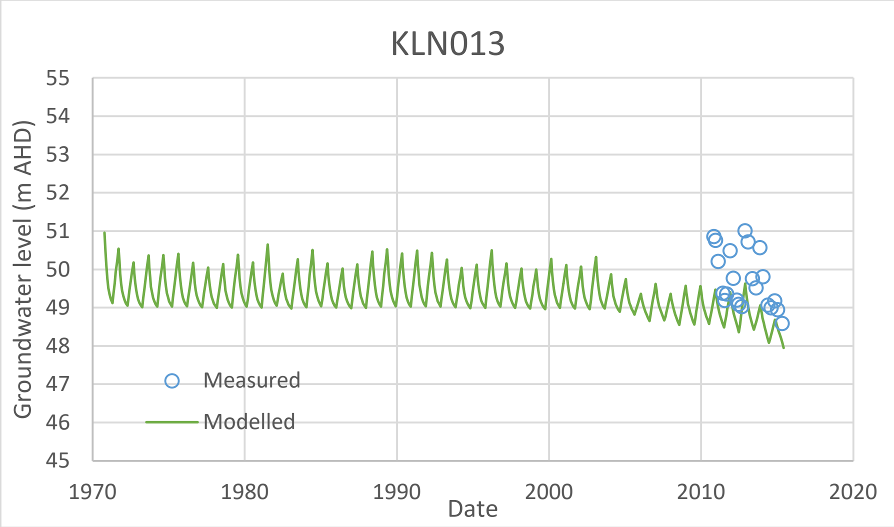
42
1970
1980
1990
2000
2010
2020


KLN013
Groundwater level (m AHD)
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49
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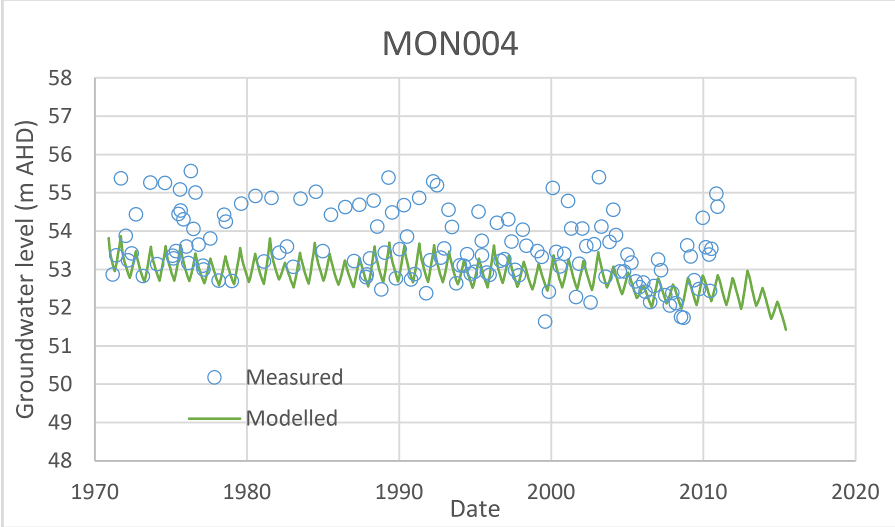
47
46
45
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


MON004
Groundwater level (m AHD)
58
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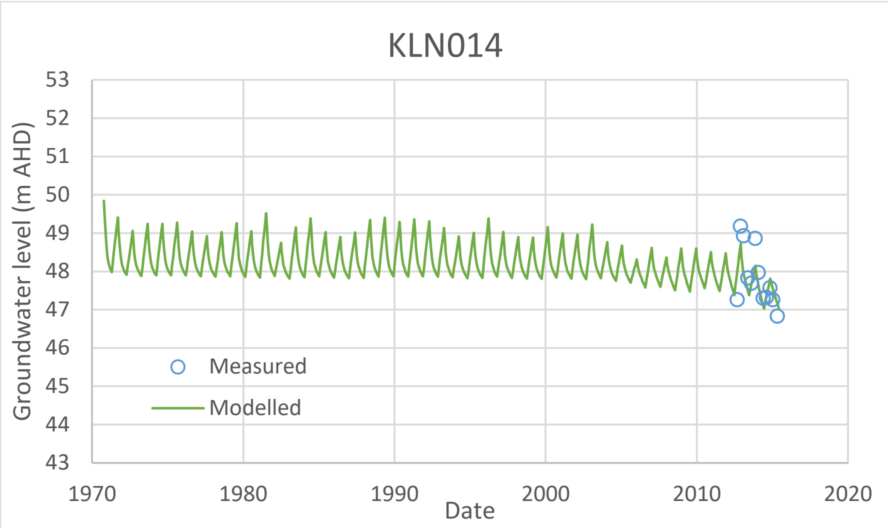
53
52
51
50
49
48
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


KLN014
Groundwater level (m AHD)
53
52
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48
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46
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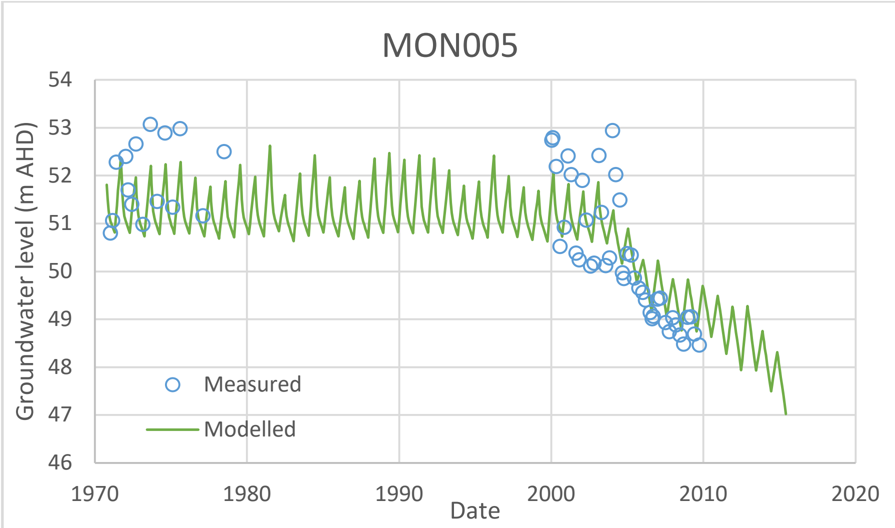
44
43
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


MON005
Groundwater level (m AHD)
54
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49
48
47
46
Measured
Modelled
1970
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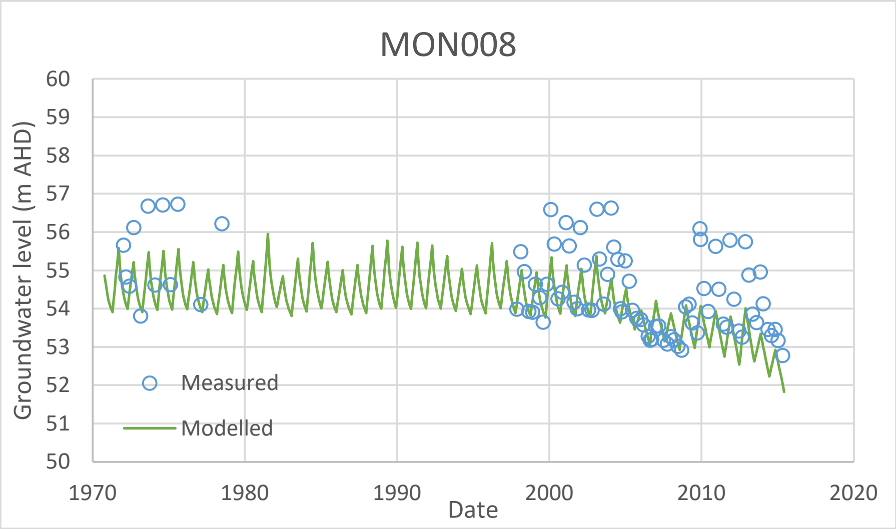
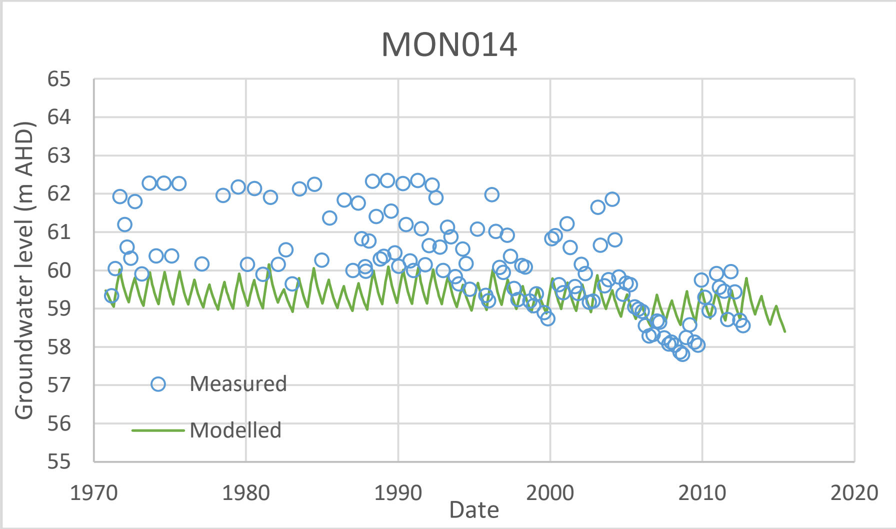
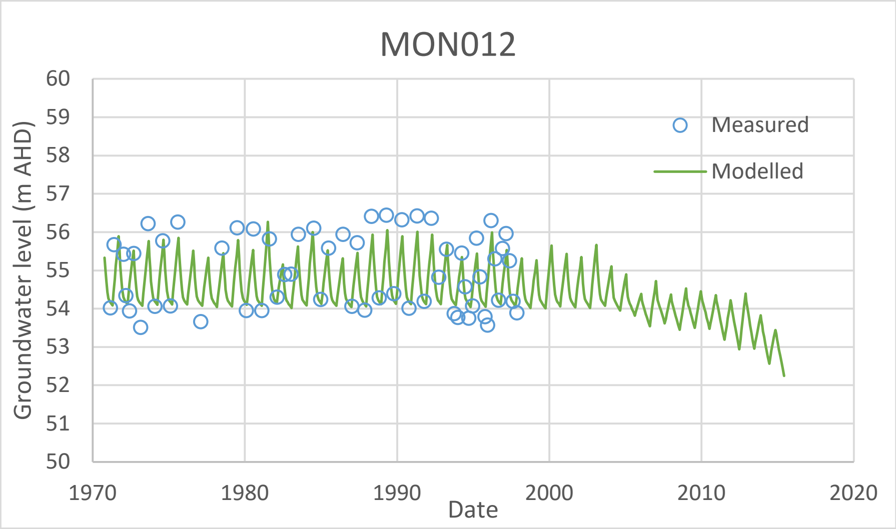
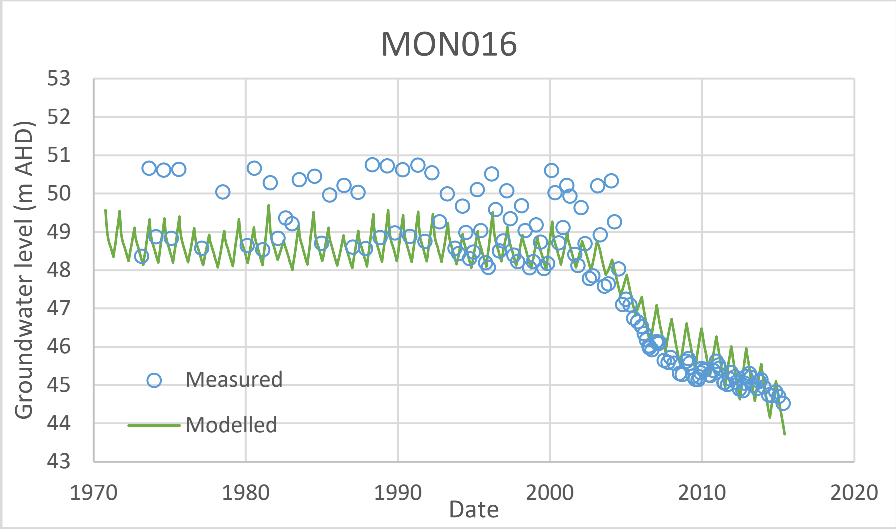
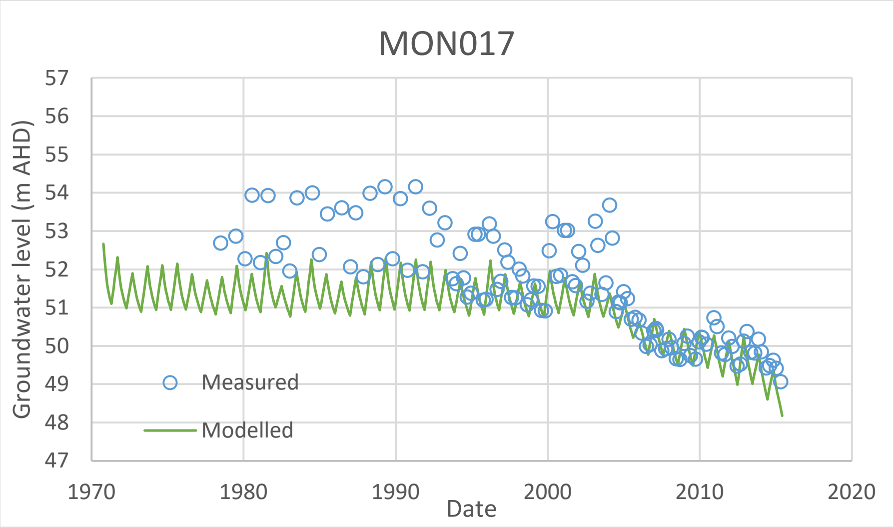
1990
2000
2010
2020
Date


MON008
Groundwater level (m AHD)
Measured
Modelled
Date


MON014
Groundwater level (m AHD)
Measured
Modelled
Date


MON012
Groundwater level (m AHD)
Measured
Modelled
Date


MON016
Groundwater level (m AHD)
Measured
Modelled
Date



MON017
Groundwater level (m AHD)
57
56
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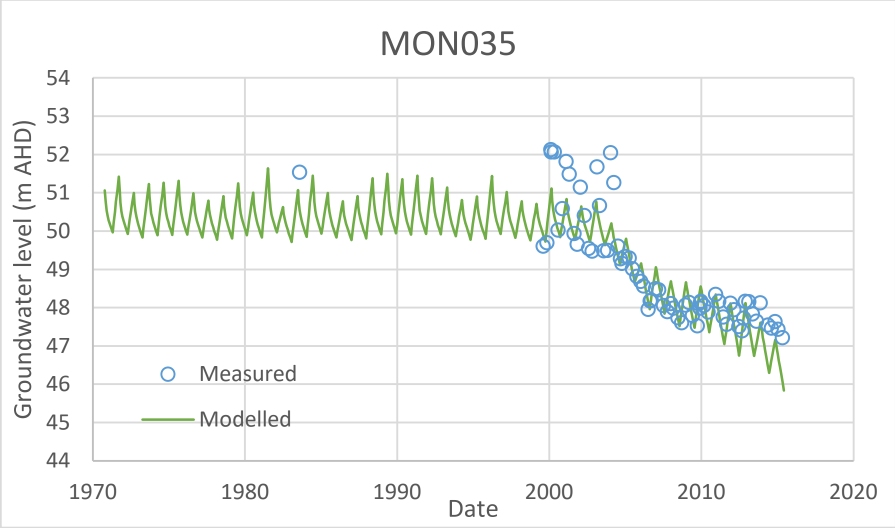
49
48
47
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled



MON035
Groundwater level (m AHD)
54
53
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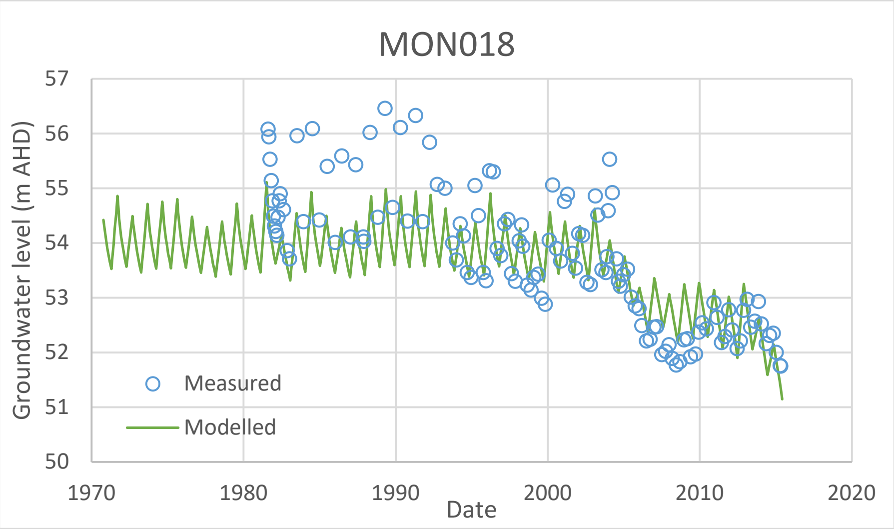
48
47
46
45
44
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled



MON018
Groundwater level (m AHD)
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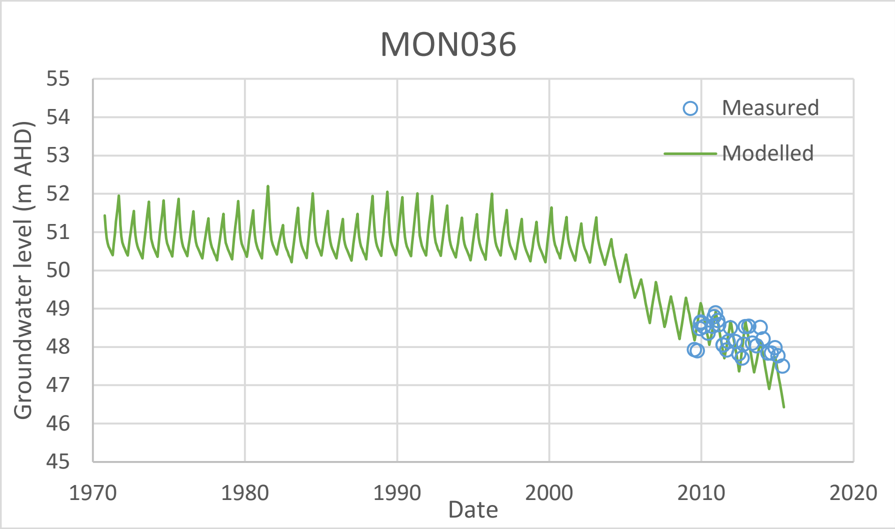
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled



MON036
Groundwater level (m AHD)
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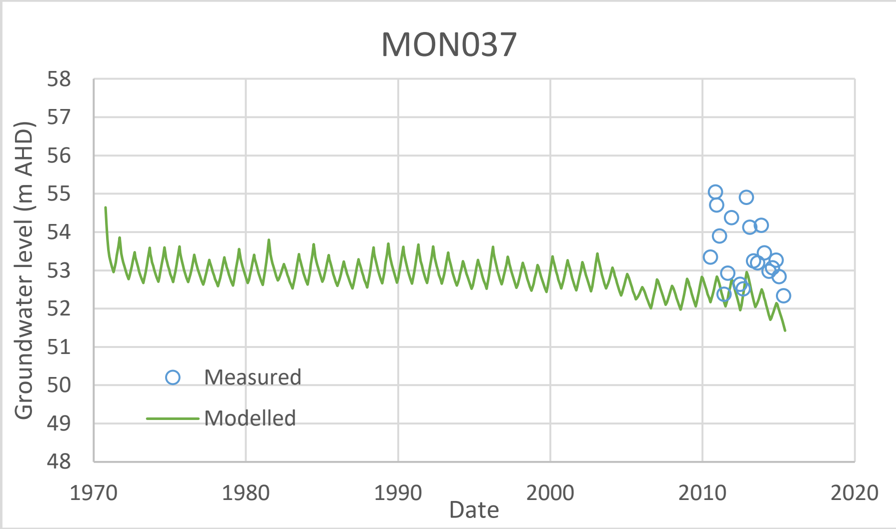
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


MON037
Groundwater level (m AHD)
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51
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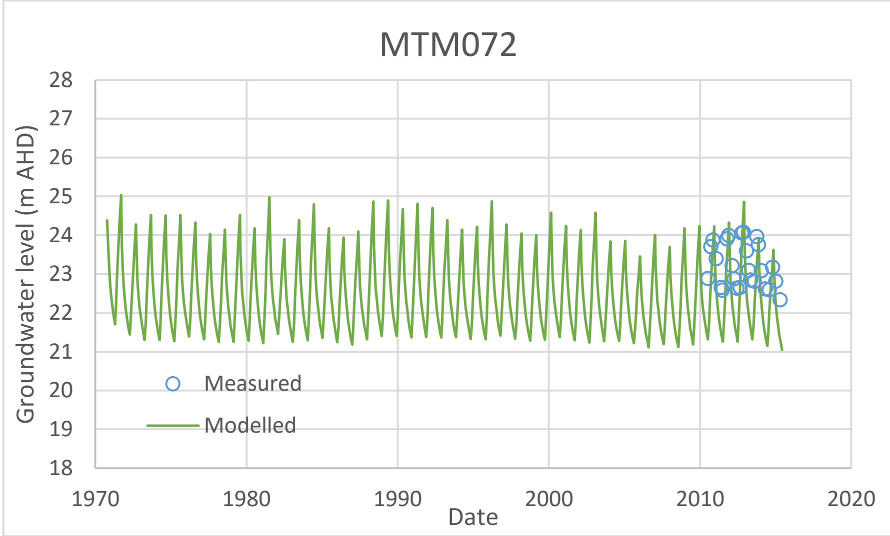
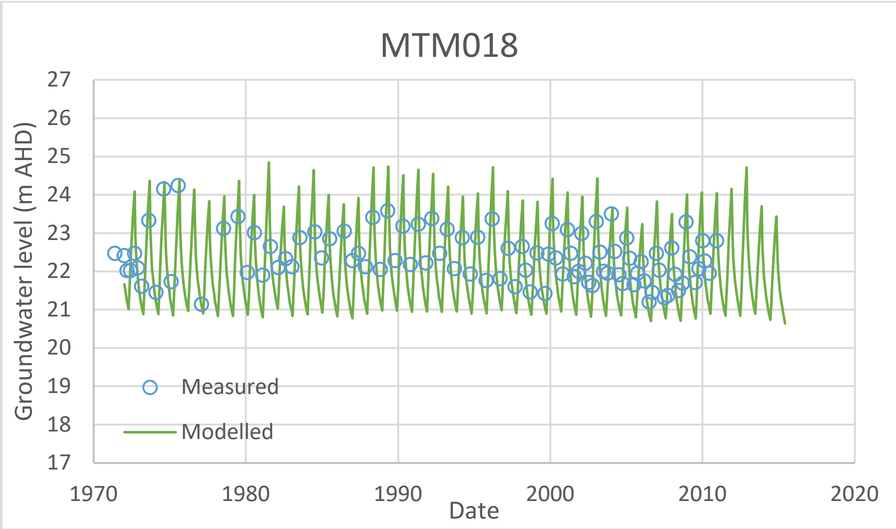
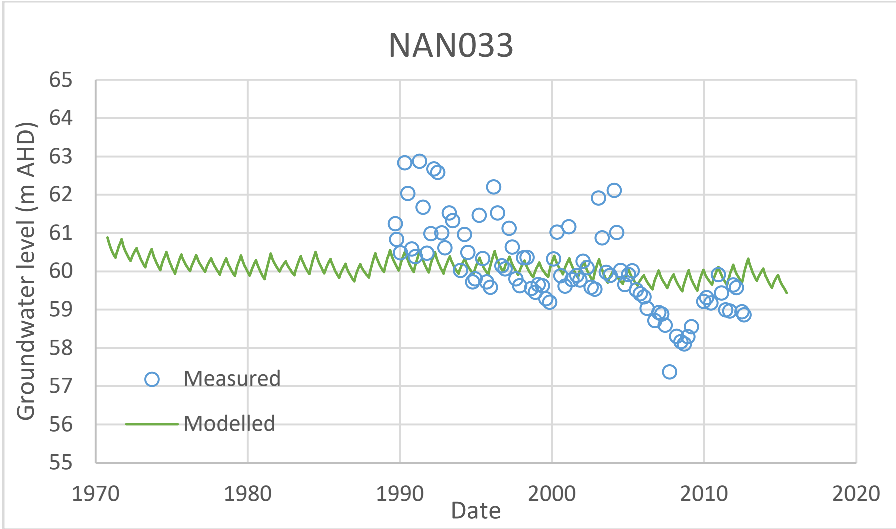
49
48
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


MTM072
Groundwater level (m AHD)
28
27
26
25
24
23
22
21
20
19
18
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


MTM018
Groundwater level (m AHD)
27
26
25
24
23
22
21
20
19
18
17
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


NAN033
Groundwater level (m AHD)
65
64
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61
60
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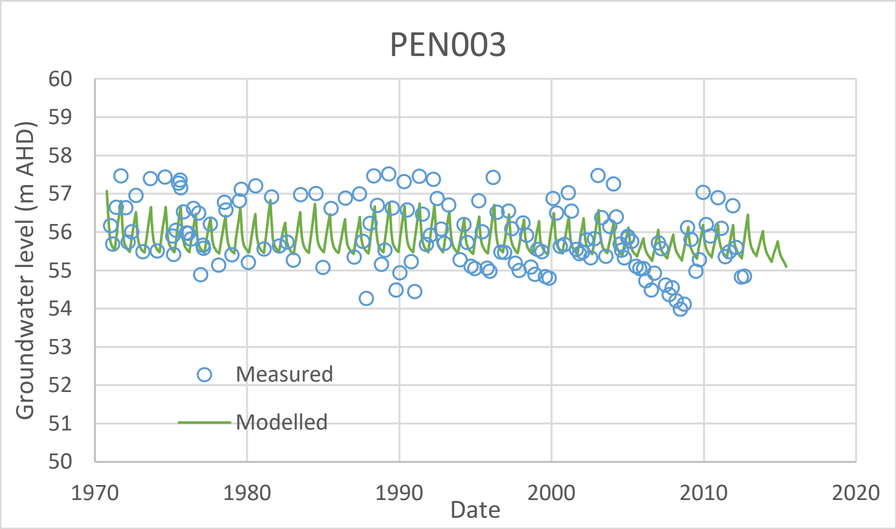
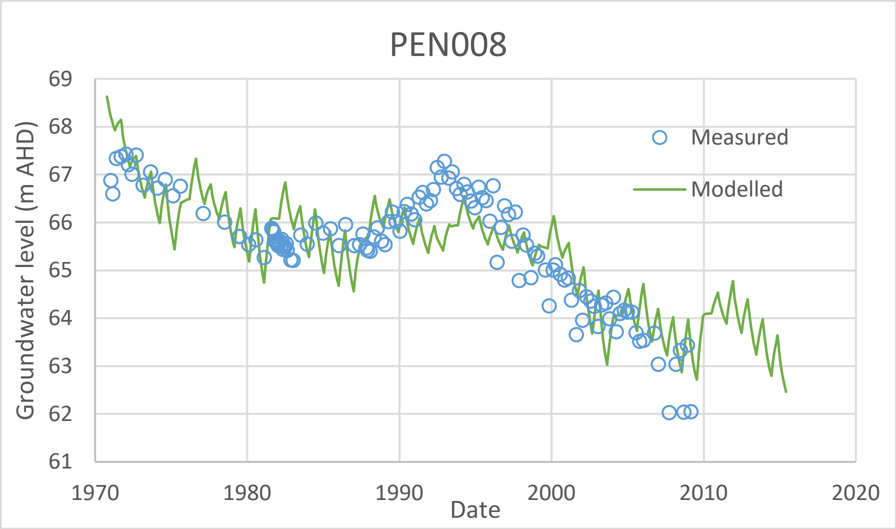
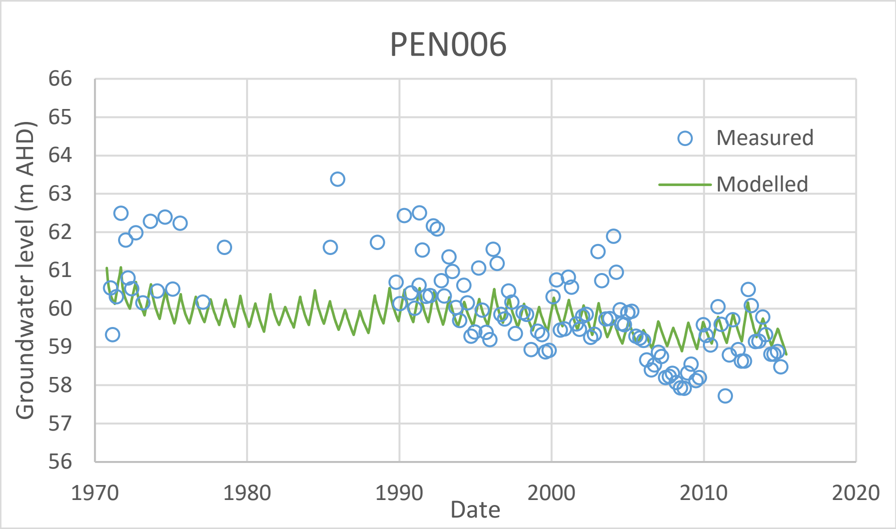
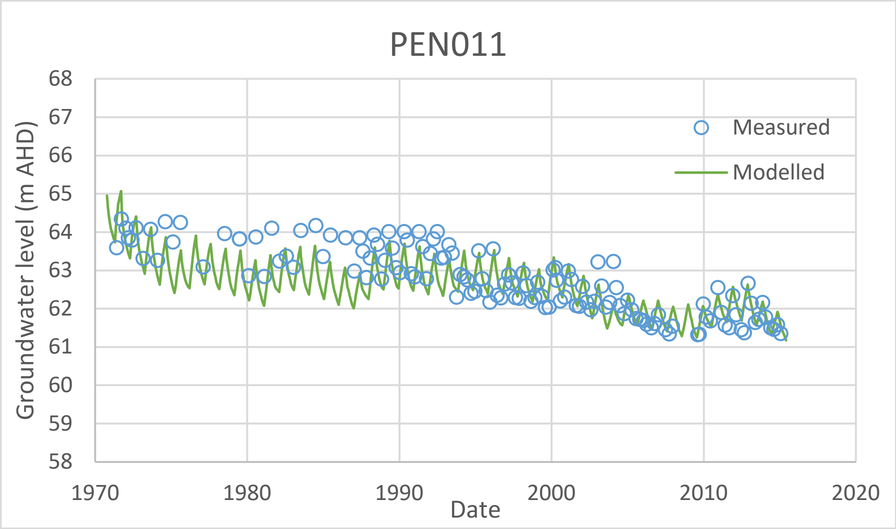
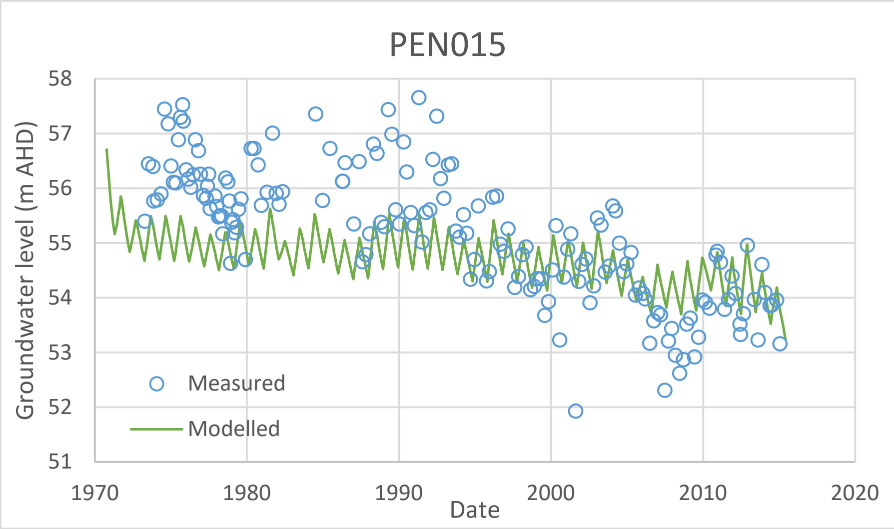
58
57
56
55
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


PEN003
Groundwater level (m AHD)
Date
Measured
Modelled


PEN008
Groundwater level (m AHD)
Date
Measured
Modelled


PEN006
Groundwater level (m AHD)
Date
Measured
Modelled


PEN011
Groundwater level (m AHD)
Date
Measured
Modelled


PEN015
Groundwater level (m AHD)
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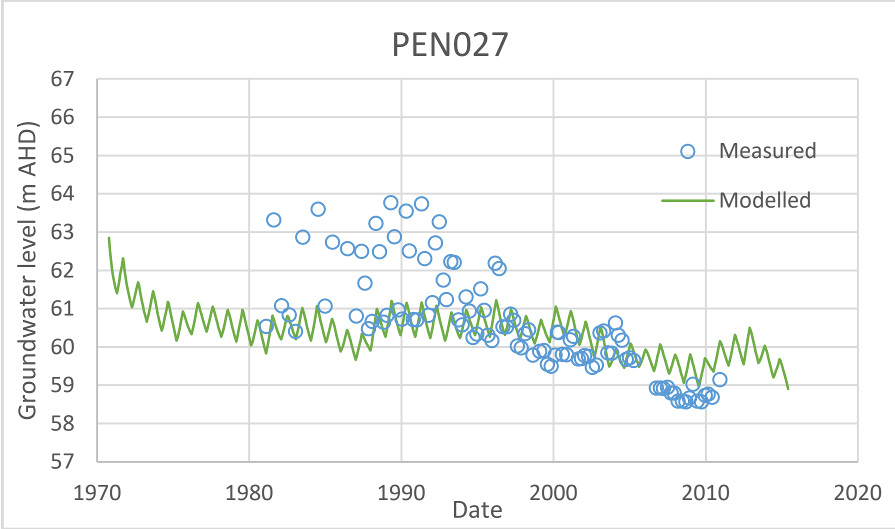
51
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


PEN027
Groundwater level (m AHD)
67
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59
58
57
1970
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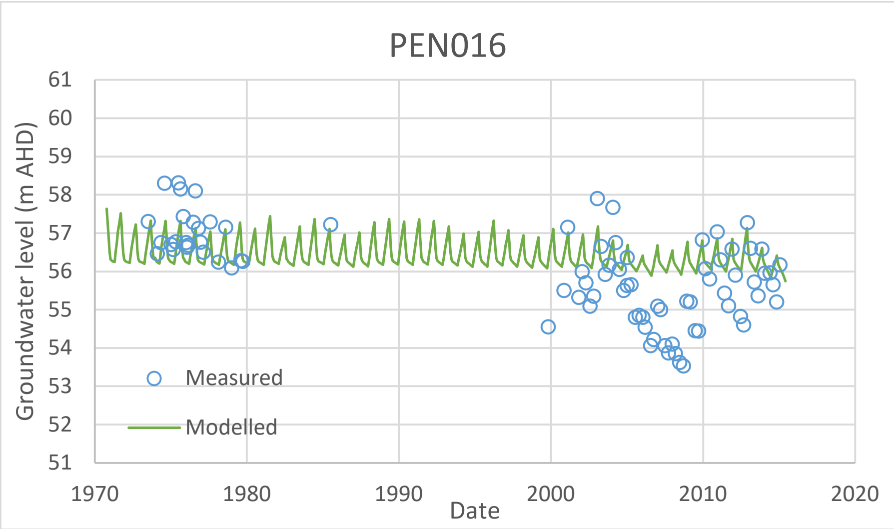
1990
2000
2010
2020
Date
Measured
Modelled


PEN016
Groundwater level (m AHD)
61
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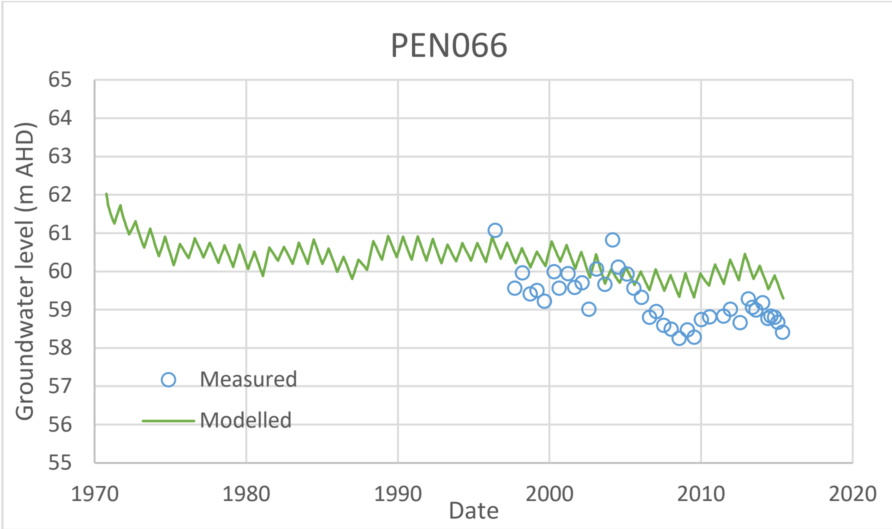
53
52
51
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


PEN066
Groundwater level (m AHD)
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58
57
56
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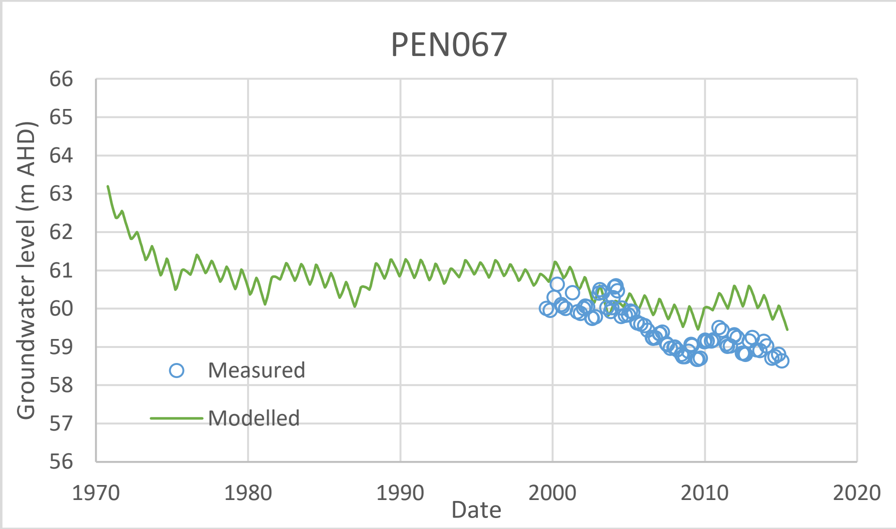
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


PEN067
Groundwater level (m AHD)
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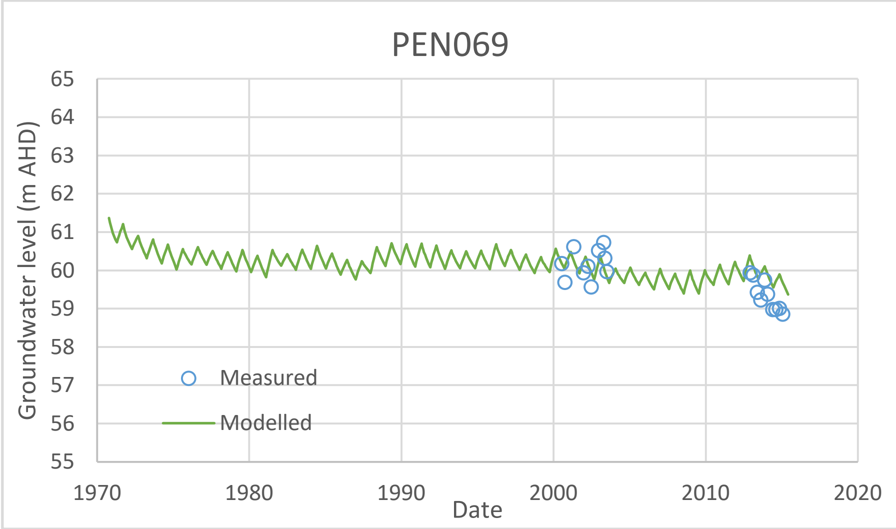
58
57
56
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


PEN069
Groundwater level (m AHD)
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1970
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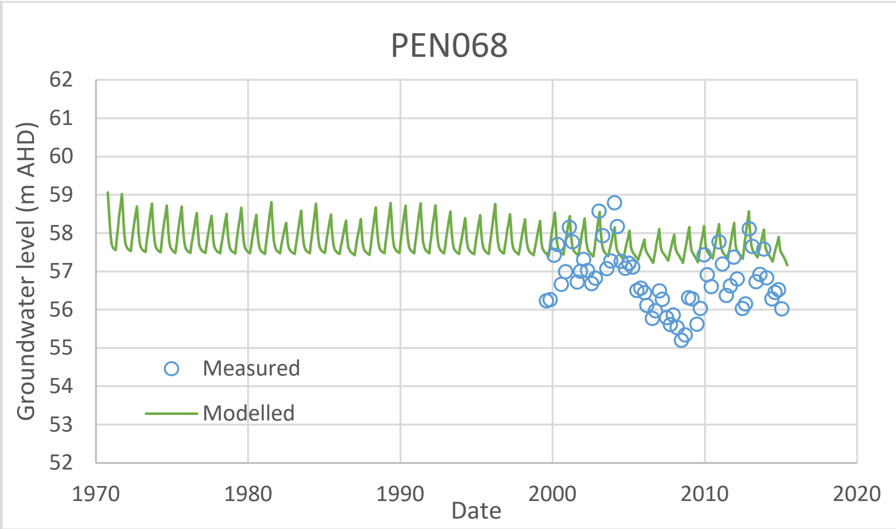
1990
2000
2010
2020
Date
Measured
Modelled


PEN068
Groundwater level (m AHD)
62
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53
52
1970
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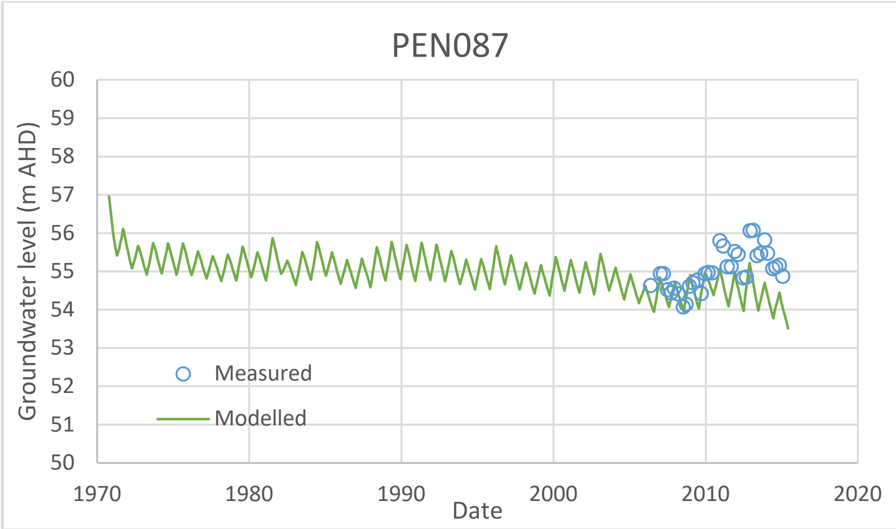
2000
2010
2020
Date
Measured
Modelled


PEN087
Groundwater level (m AHD)
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1970
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1990
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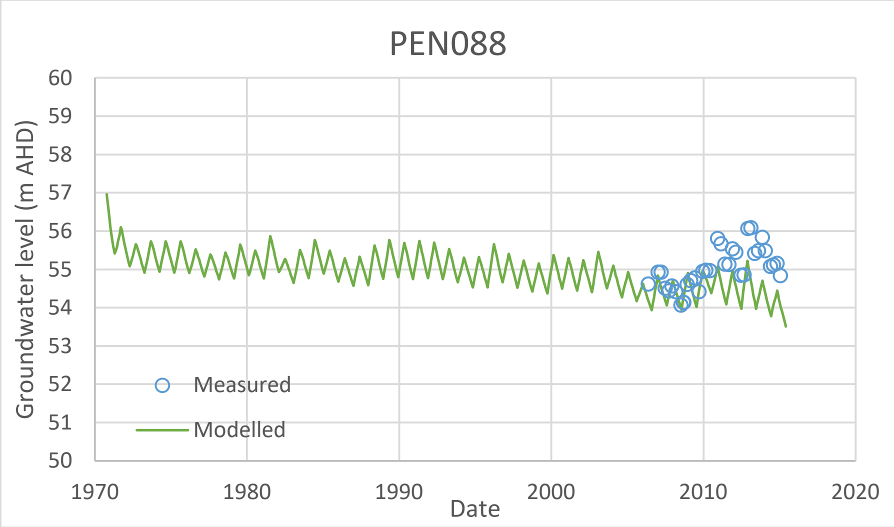
2010
2020
Date
Measured
Modelled


PEN088
Groundwater level (m AHD)
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51
50
Measured
Modelled
1970
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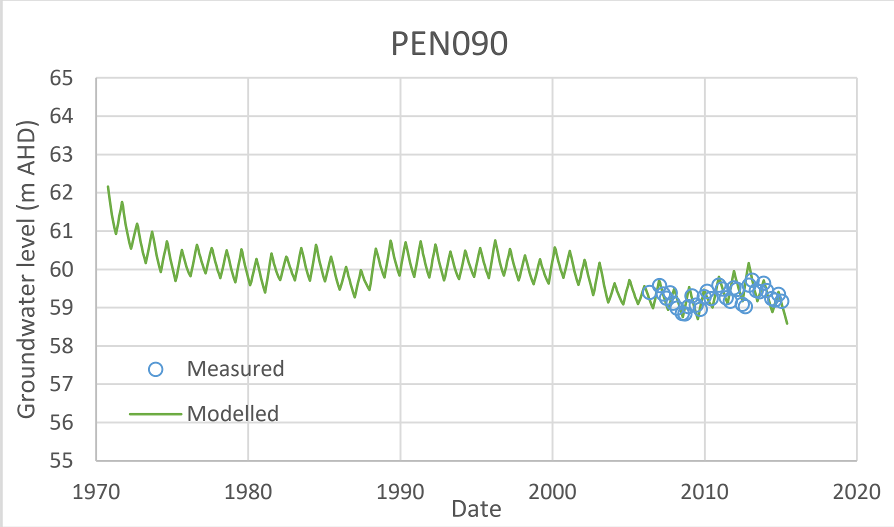
2000
2010
2020
Date


PEN090
Groundwater level (m AHD)
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56
55
Measured
Modelled
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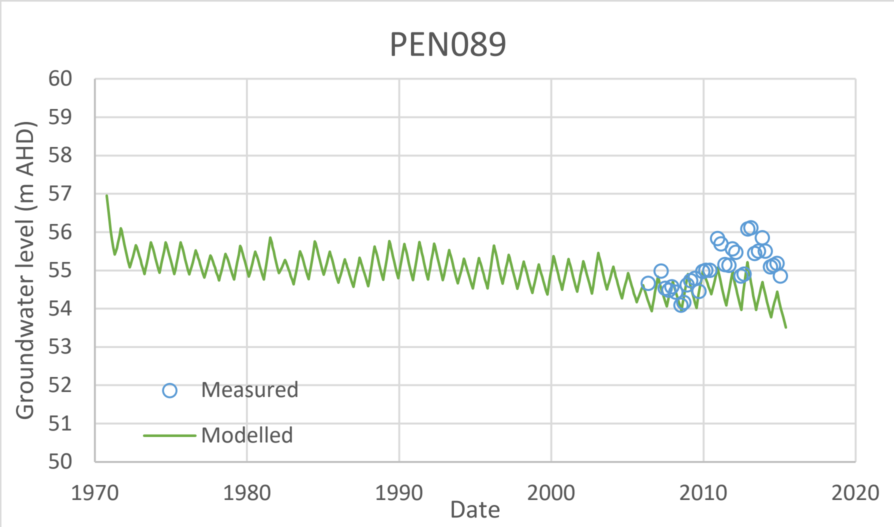
2010
2020
Date


PEN089
Groundwater level (m AHD)
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50
Measured
Modelled
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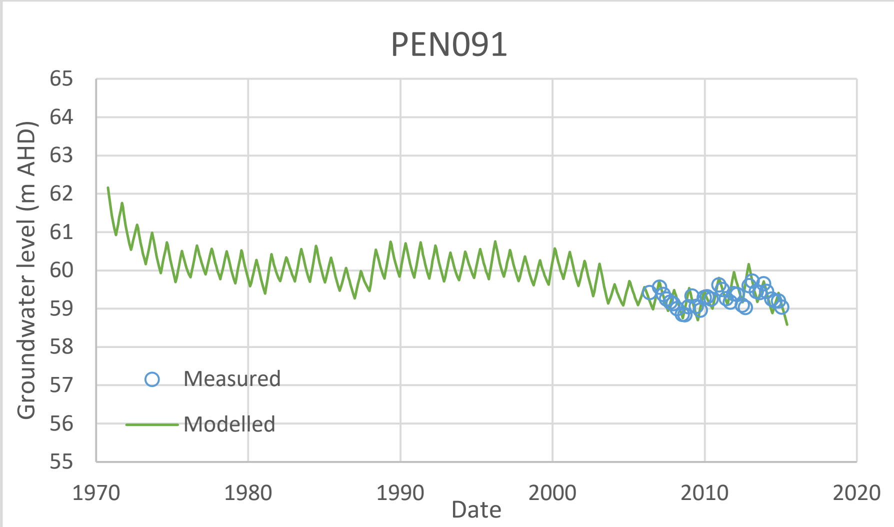
2020
Date


PEN091
Groundwater level (m AHD)
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56
55
Measured
Modelled
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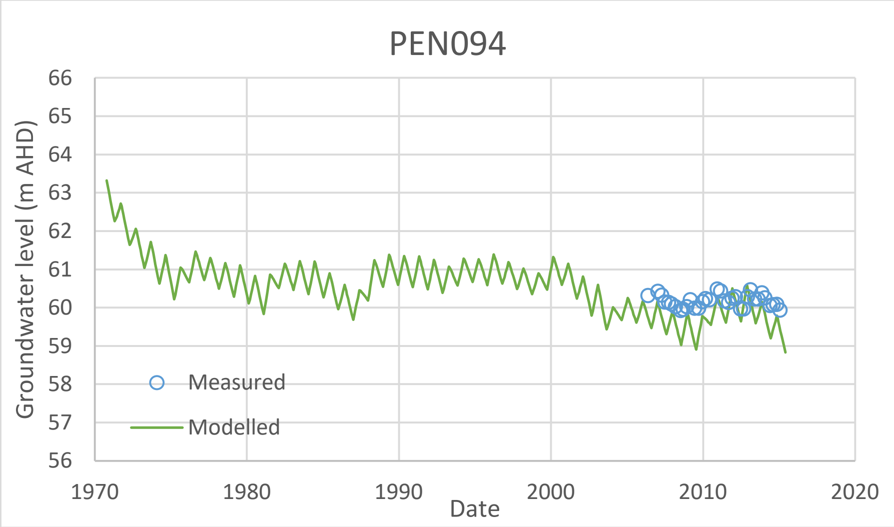
2010
2020
Date


PEN094
Groundwater level (m AHD)
66
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60
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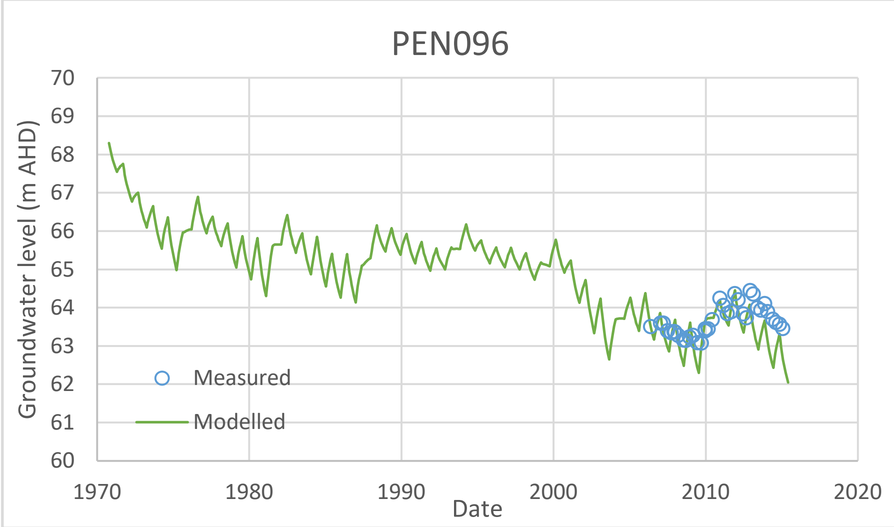
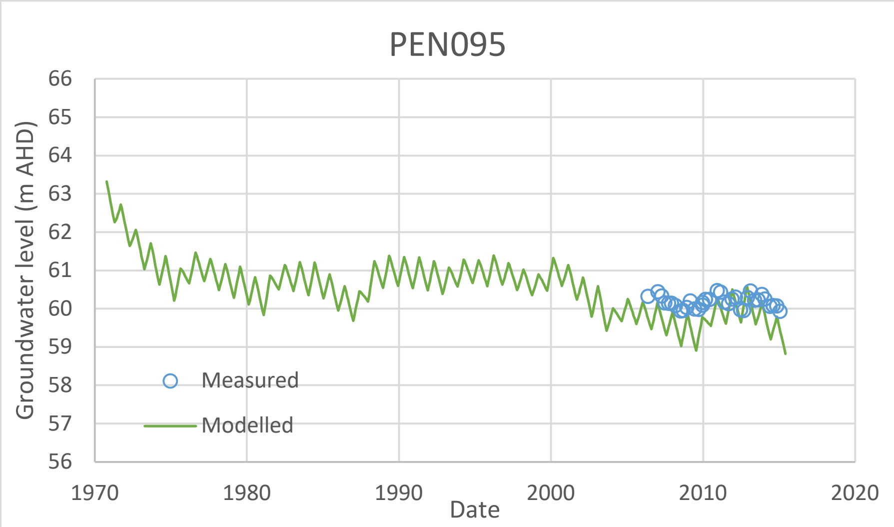
58
57
56
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


PEN096
Groundwater level (m AHD)
70
69
68
67
66
65
64
63
62
61
60
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


PEN095
Groundwater level (m AHD)
66
65
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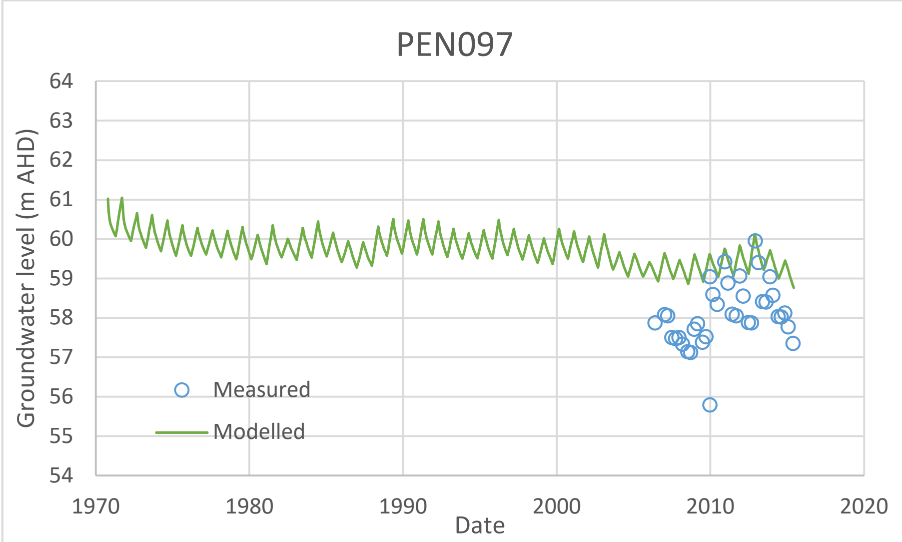
60
59
58
57
56
Measured
Modelled
1970
1980
1990
2000
2010
2020
Date


PEN097
Groundwater level (m AHD)
64
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54
Measured
Modelled
1970
1980
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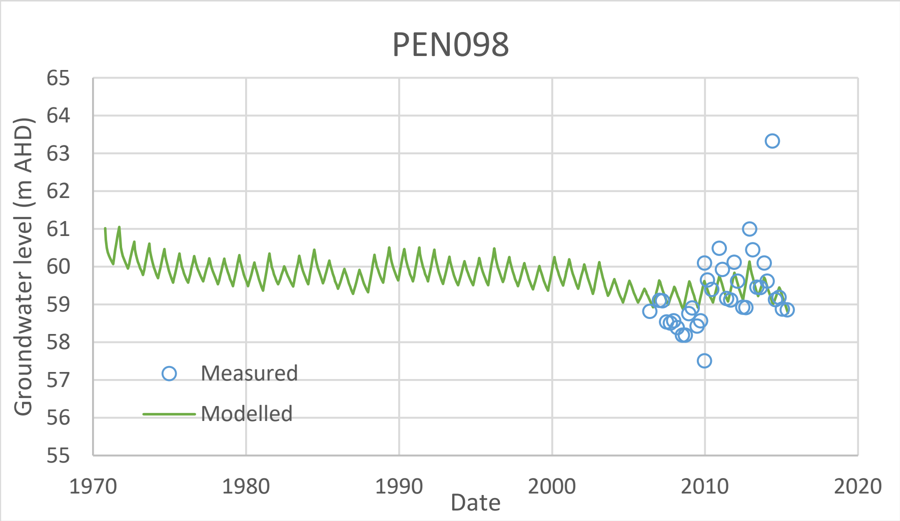
2010
2020
Date


PEN098
Groundwater level (m AHD)
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1970
1980
1990
2000
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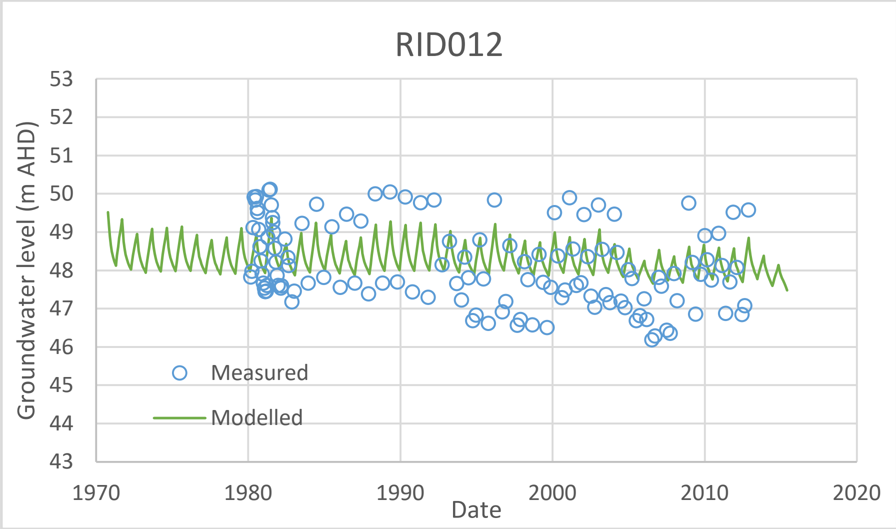
2020
Date
Measured
Modelled


RID012
Groundwater level (m AHD)
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52
51
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48
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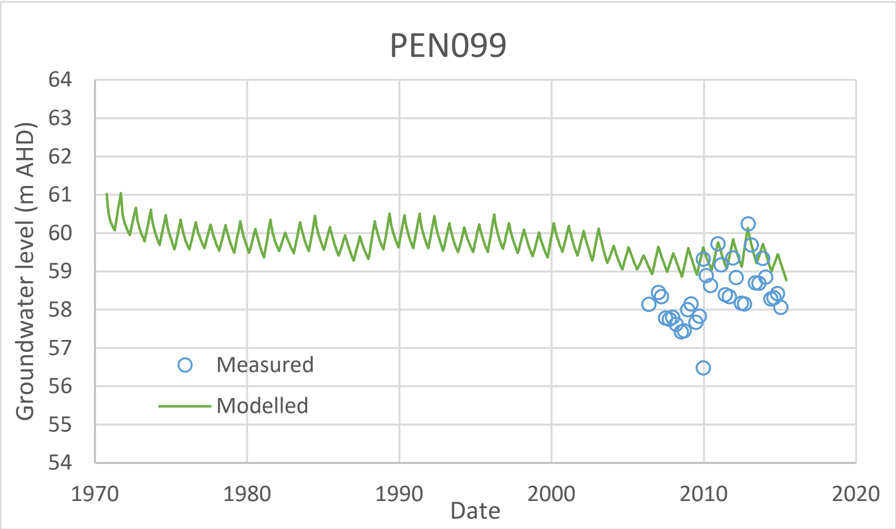
46
45
44
43
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled


PEN099
Groundwater level (m AHD)
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1970
1980
1990
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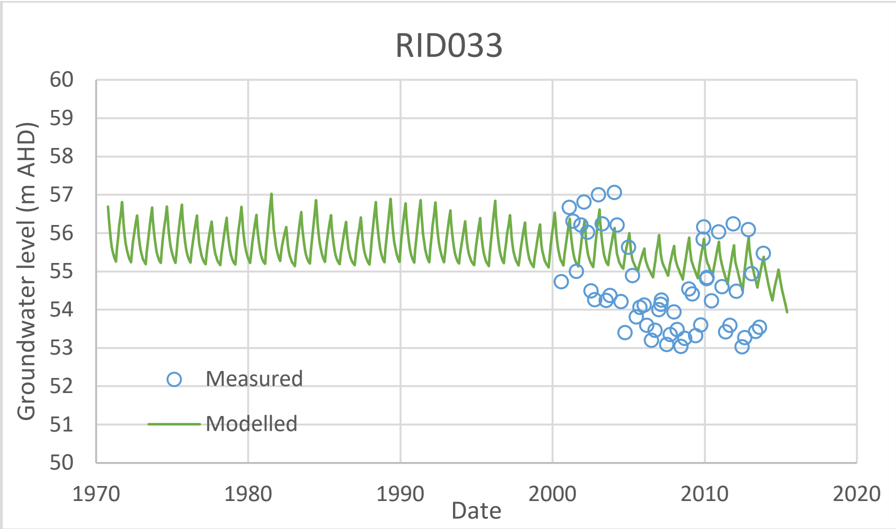
2010
2020
Date
Measured
Modelled


RID033
Groundwater level (m AHD)
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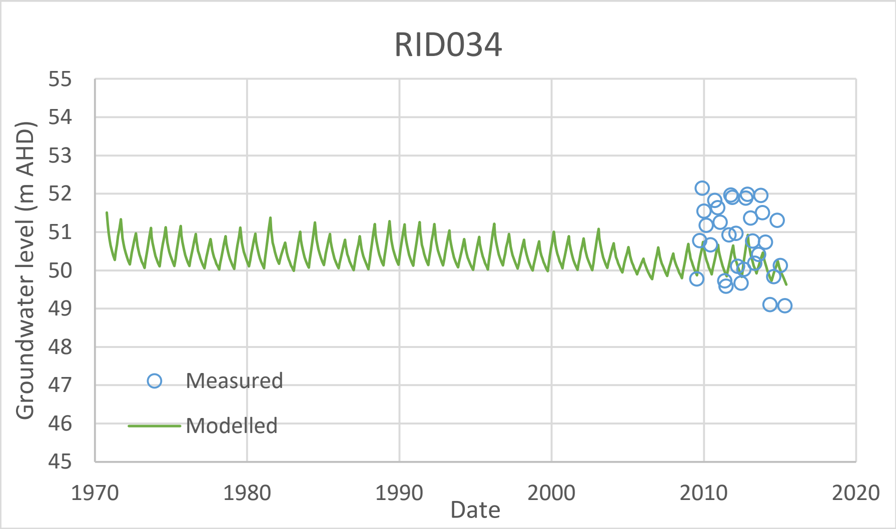
1970
1980
1990
2000
2010
2020
Date
Measured
Modelled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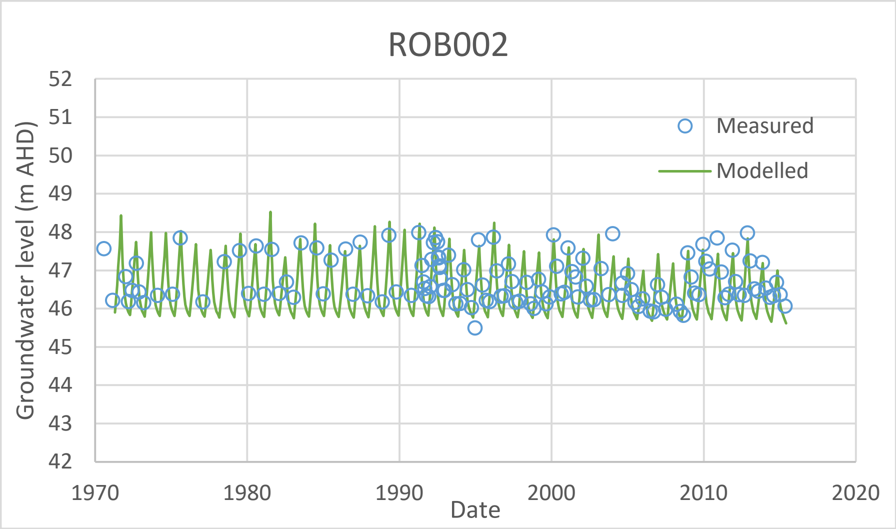
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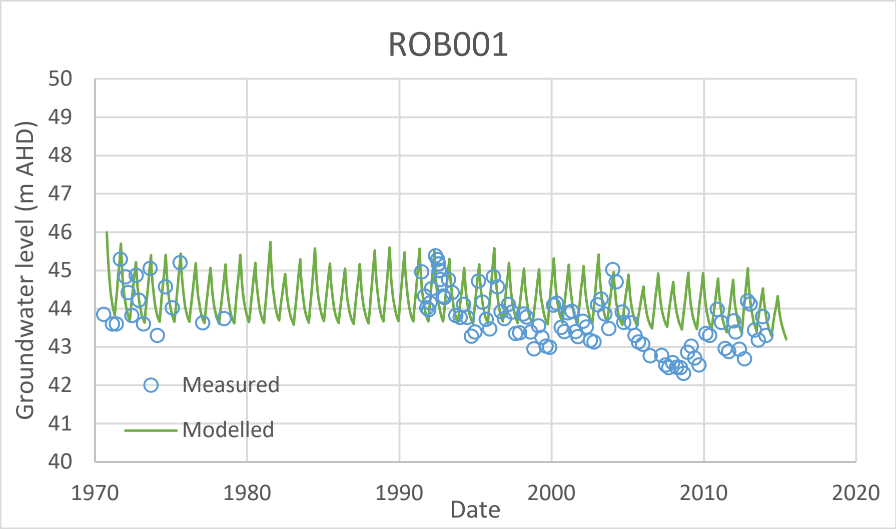
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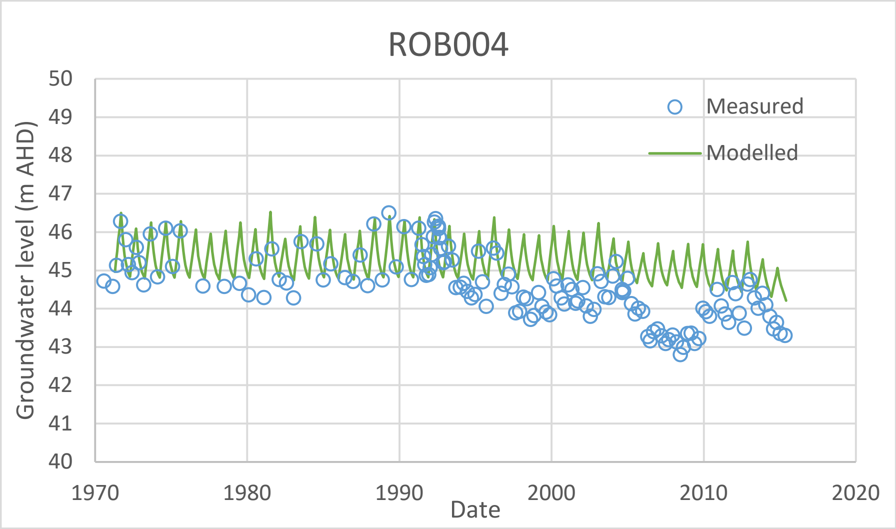
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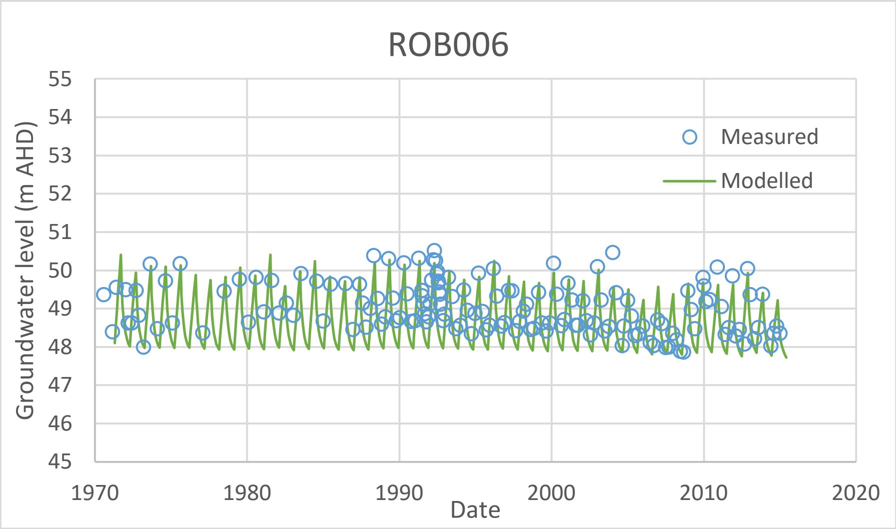
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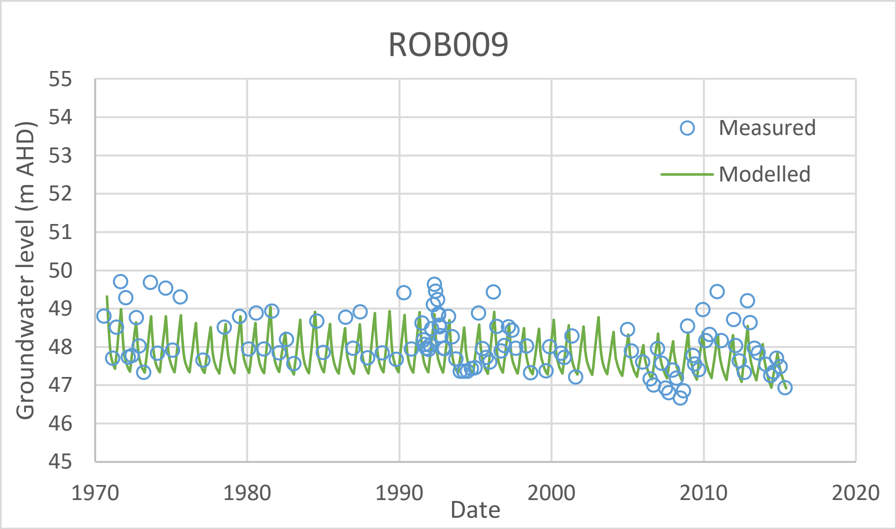
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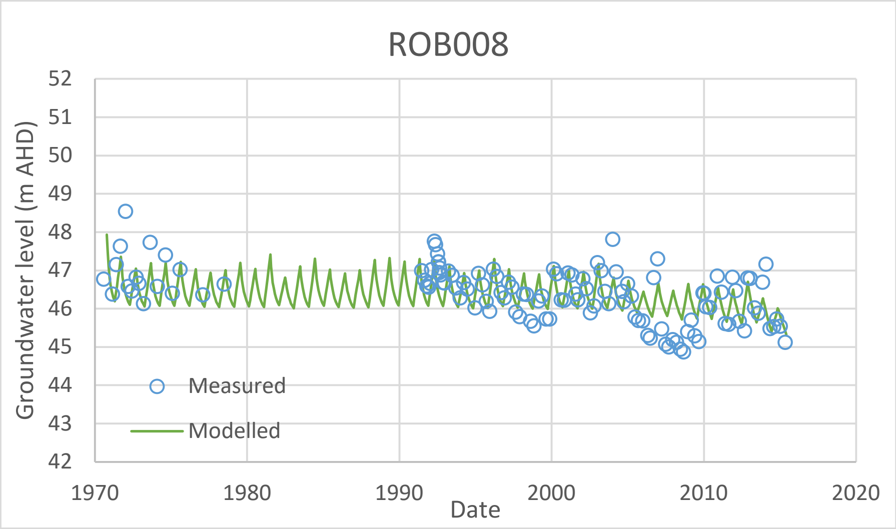
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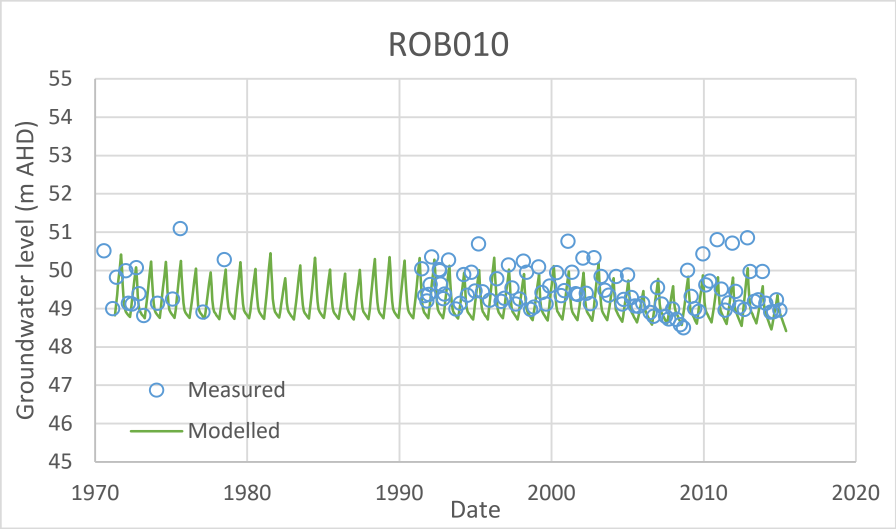
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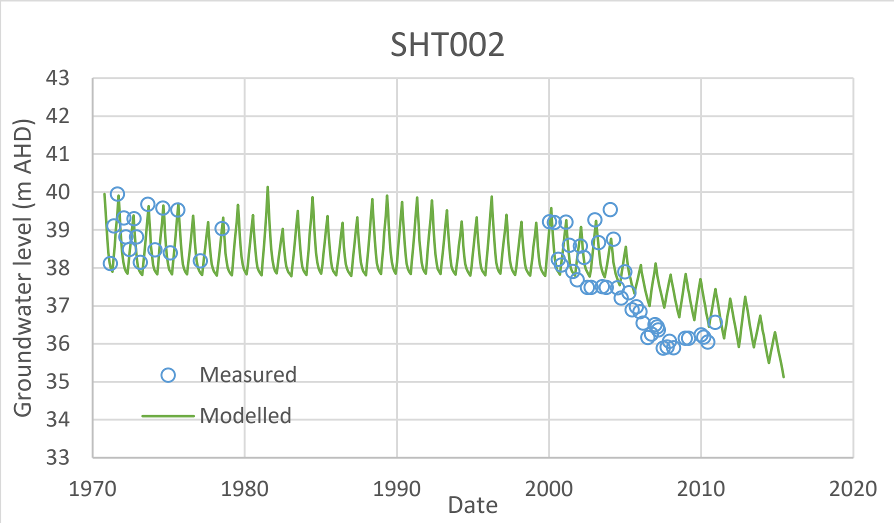
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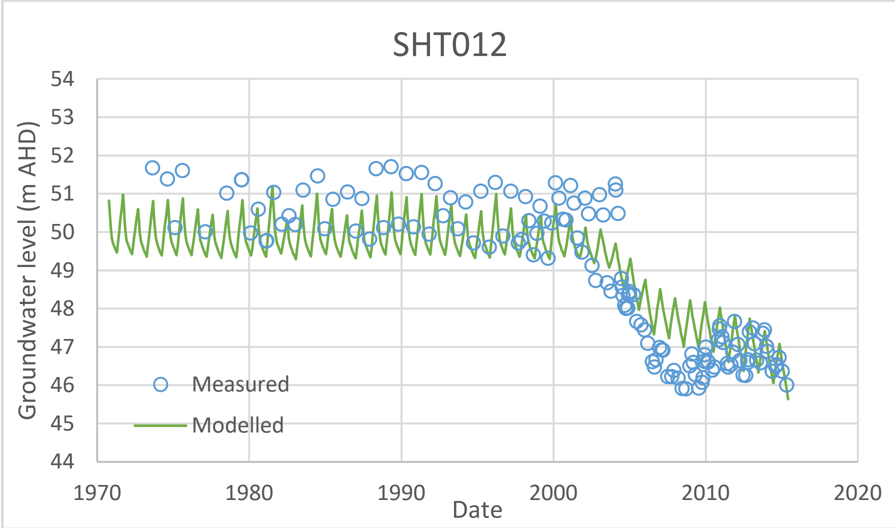
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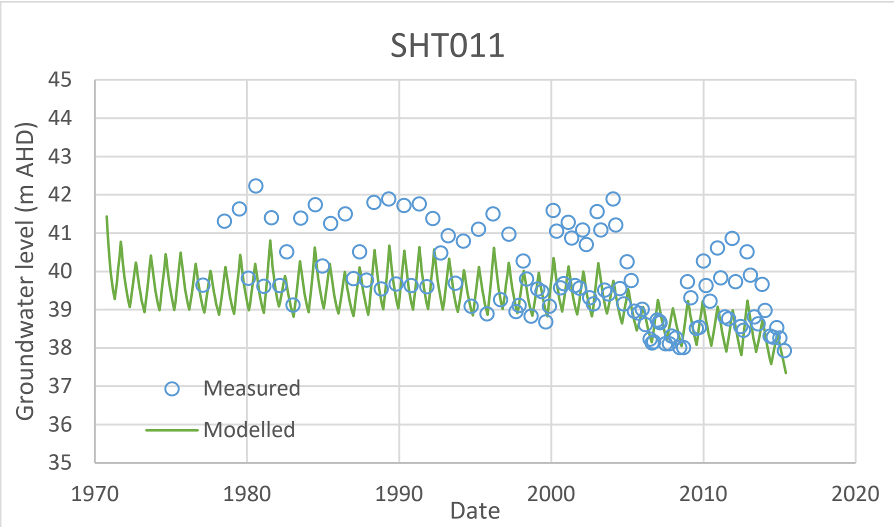
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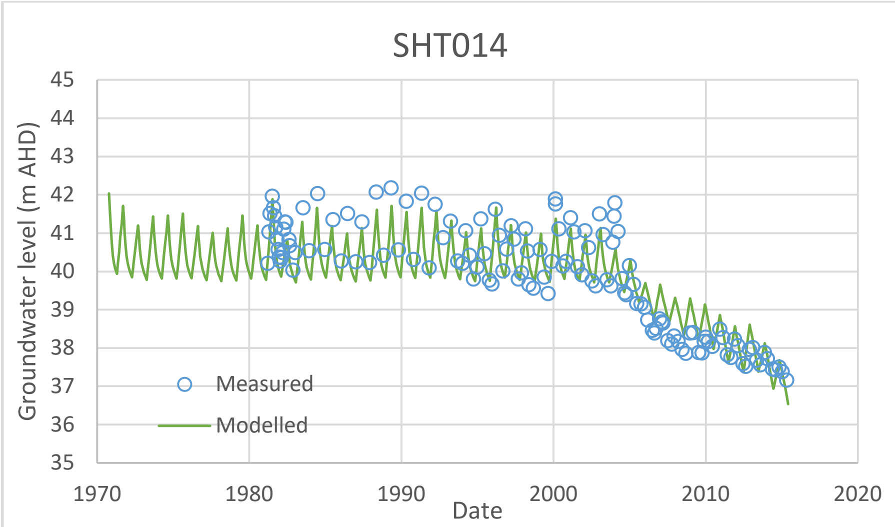
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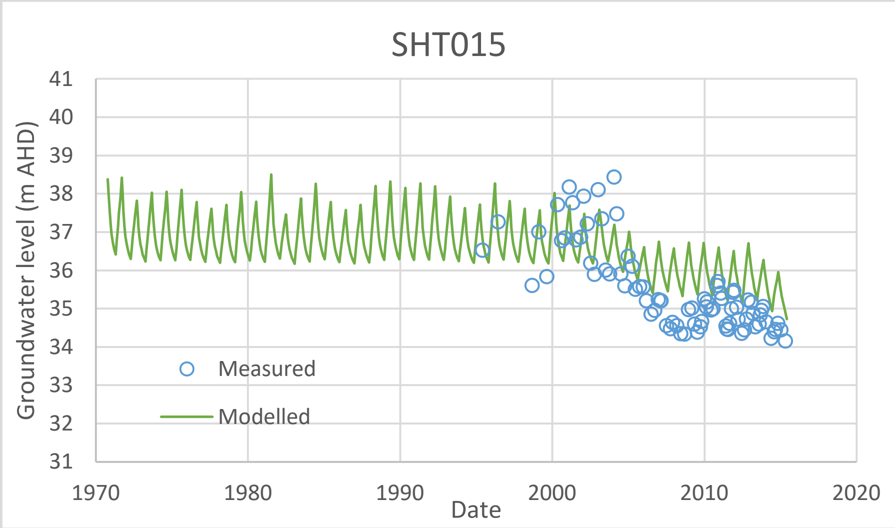
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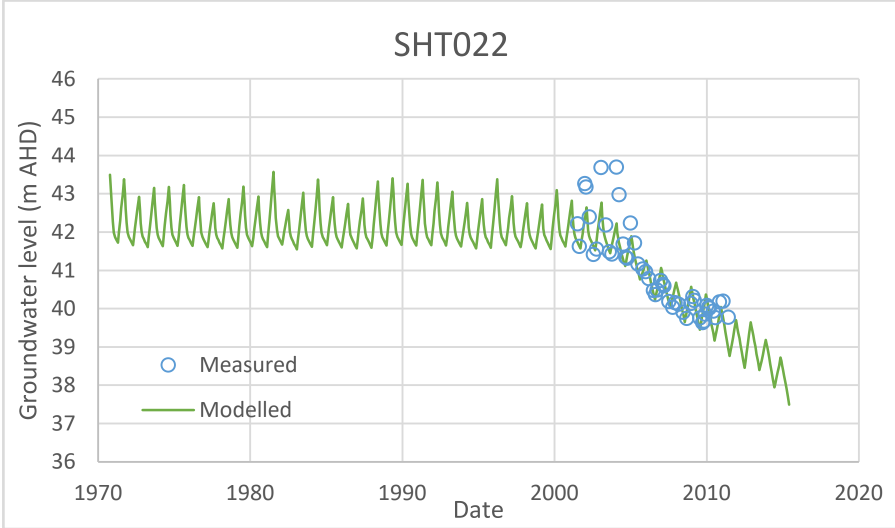
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Groundwater level (m AHD)
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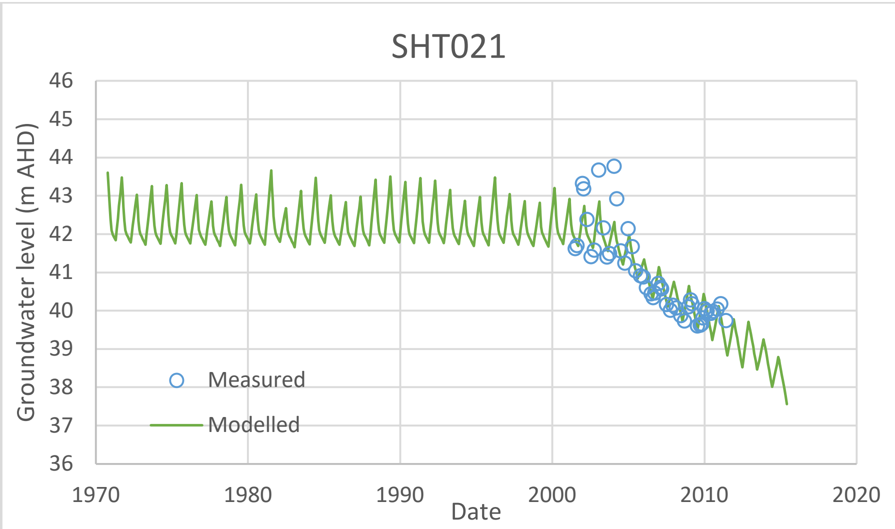
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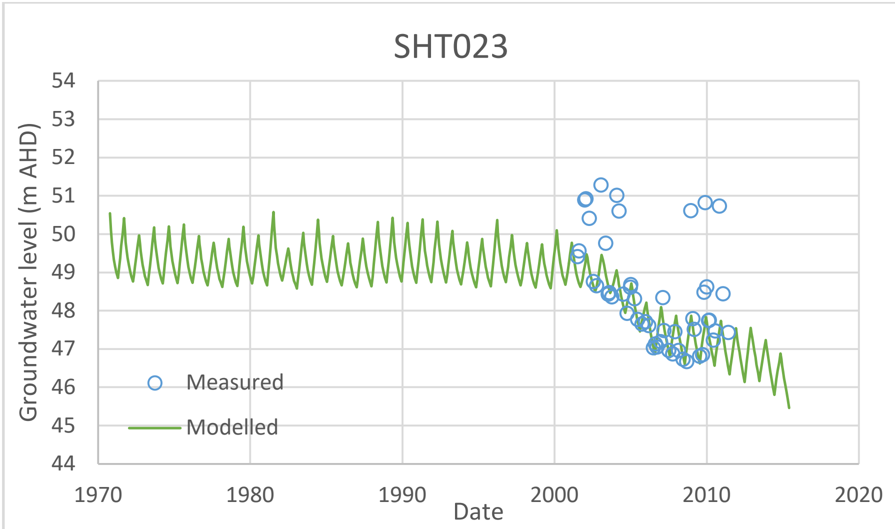
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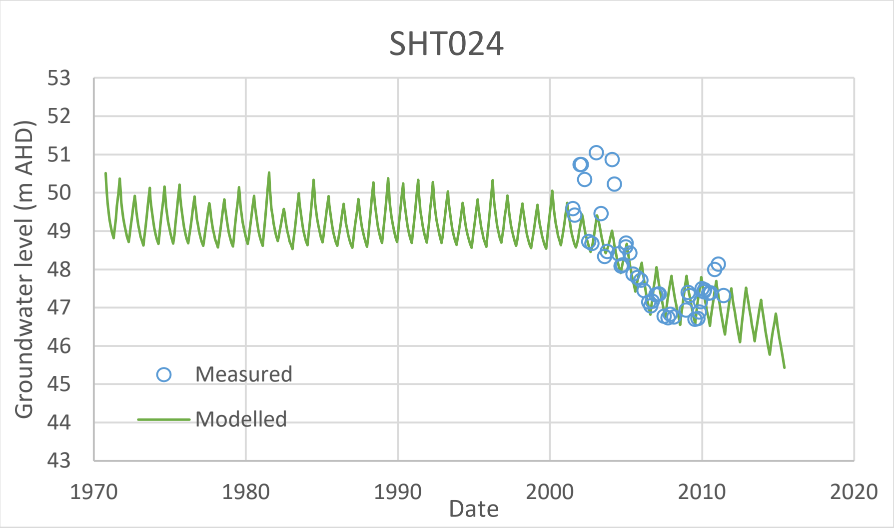
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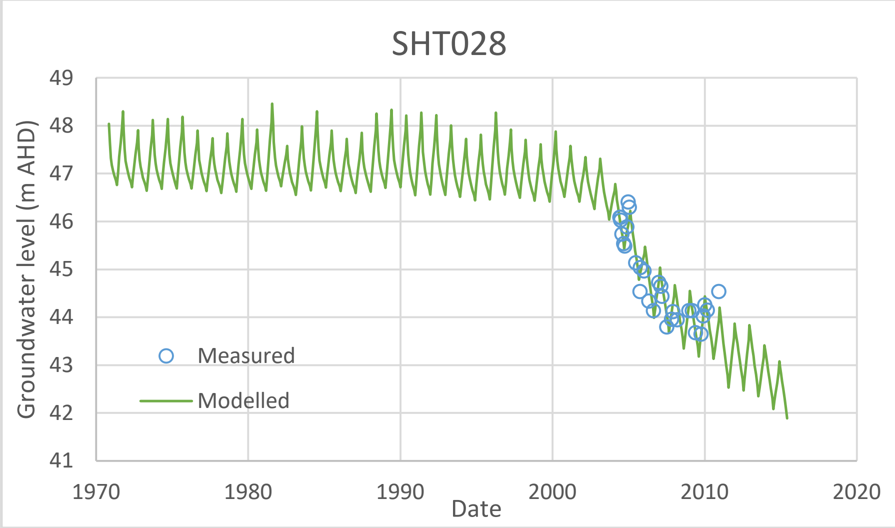
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Groundwater level (m AHD)
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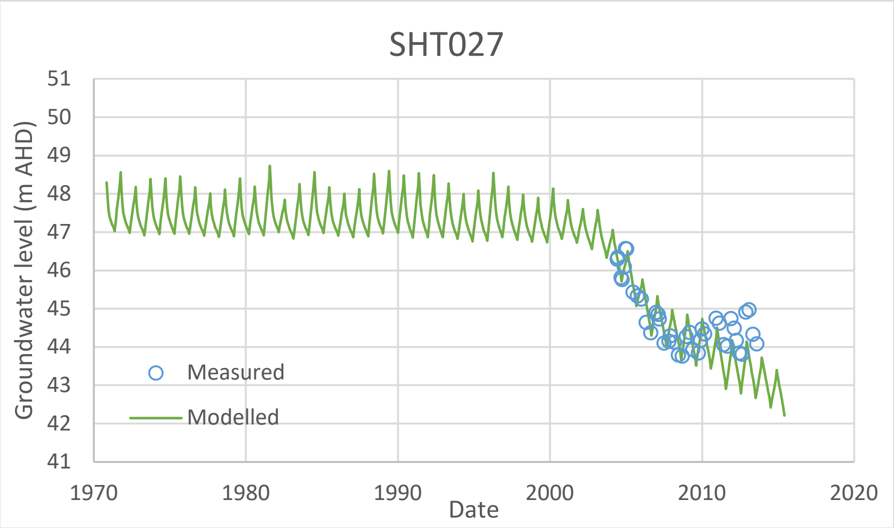
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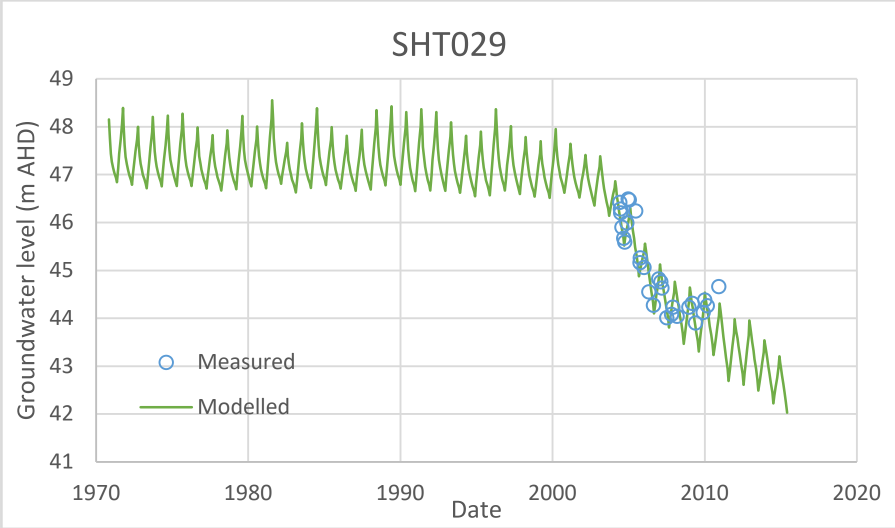
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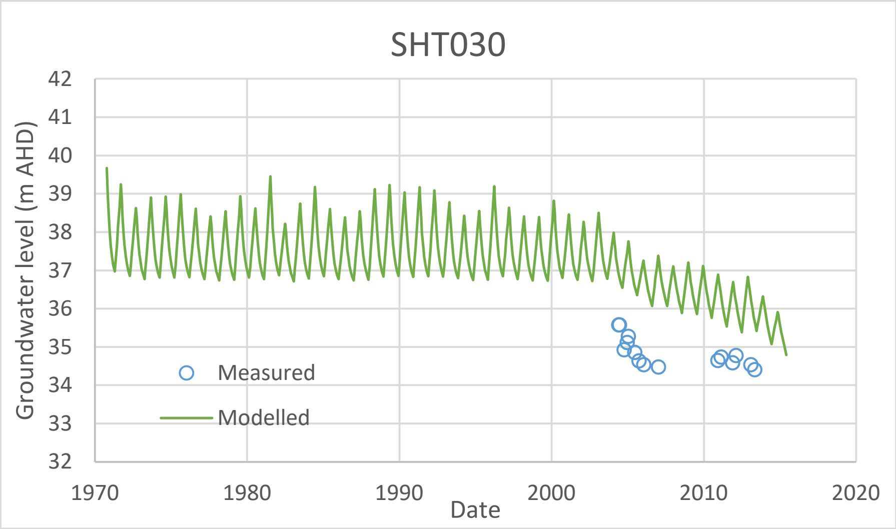
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Groundwater level (m AHD)
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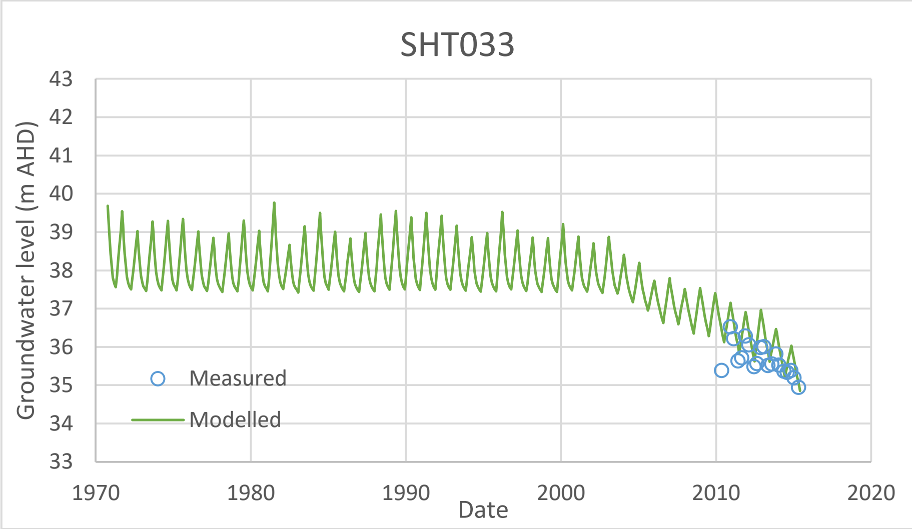
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Groundwater level (m AHD)
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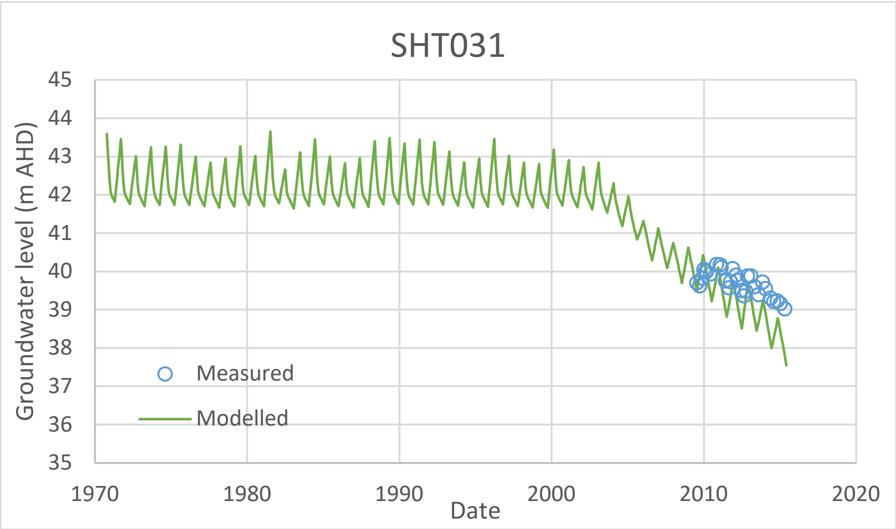
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Groundwater level (m AHD)
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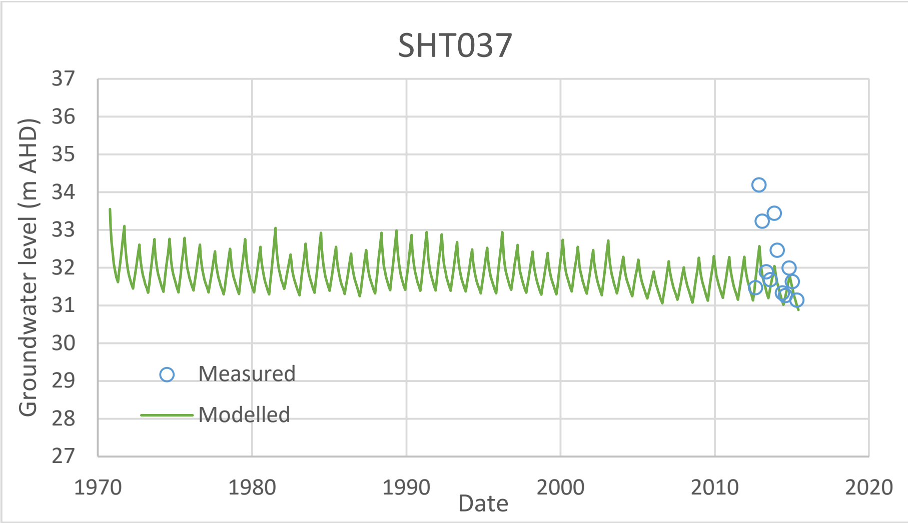
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Groundwater level (m AHD)
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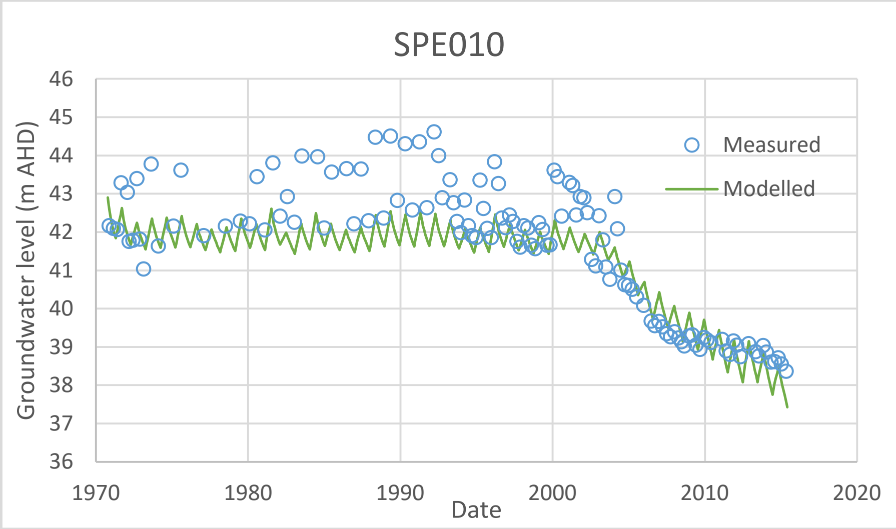
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Groundwater level (m AHD)
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9 Units of measurement

9.1 Units of measurement commonly used (SI and non-SI Australian legal)

Name of unit	Symbol	Definition in terms of other metric units	Quantity
day	d	24 h	time interval
gigalitre	GL	10^6 m^3	volume
gram	g	10^{-3} kg	mass
hectare	ha	10^4 m^2	area
hour	h	60 min	time interval
kilogram	kg	base unit	mass
kilolitre	kL	1 m^3	volume
kilometre	km	10^3 m	length
litre	L	10^{-3} m^3	volume
megalitre	ML	10^3 m^3	volume
metre	m	base unit	length
microgram	µg	10^{-6} g	mass
microlitrer	µL	10^{-9} m^3	volume
milligram	mg	10^{-3} g	mass
millilitre	mL	10^{-6} m^3	volume
millimetre	mm	10^{-3} m	length
minute	min	60 s	time interval
second	s	base unit	time interval
tonne	t	1000 kg	mass
year	y	365 or 366 days	time interval

10 Glossary

Aquifer — An underground layer of rock or sediment that holds water and allows water to percolate through

Aquifer, confined — Aquifer in which the upper surface is impervious (see 'confining layer') and the water is held at greater than atmospheric pressure; water in a penetrating well will rise above the surface of the aquifer

Aquifer test — A hydrological test performed on a well, aimed to increase the understanding of the aquifer properties, including any interference between wells, and to more accurately estimate the sustainable use of the water resources available for development from the well

Aquifer, unconfined — Aquifer in which the upper surface has free connection to the ground surface and the water surface is at atmospheric pressure

Aquitard — A layer in the geological profile that separates two aquifers and restricts the flow between them

Evapotranspiration (ET) — The discharge of water (soil water, groundwater and/or surface water) to the atmosphere a result of transpiration from plants and solar evaporation.

Hydraulic conductivity (K) — A measure of the ease of flow through aquifer material: high K indicates low resistance, or high flow conditions; measured in metres per day

Hydrogeology — The study of groundwater, which includes its occurrence, recharge and discharge processes, and the properties of aquifers; see also 'hydrology'

m AHD — Defines elevation in metres (m) according to the Australian Height Datum (AHD)

Model — A conceptual or mathematical means of understanding elements of the real world that allows for predictions of outcomes given certain conditions. Examples include estimating storm runoff, assessing the impacts of dams or predicting ecological response to environmental change

MODFLOW — A three-dimensional, finite difference code developed by the USGS to simulate groundwater flow

Pasture — Grassland used for the production of grazing animals such as sheep and cattle

Potentiometric head — The potentiometric head or surface is the level to which water rises in a well due to water pressure in the aquifer, measured in metres (m); also known as piezometric surface

Prescribed water resource — A water resource declared by the Governor to be prescribed under the Act, and includes underground water to which access is obtained by prescribed wells. Prescription of a water resource requires that future management of the resource be regulated via a licensing system.

PWA — Prescribed Wells Area

Ramsar Convention — This is an international treaty on wetlands titled *The Convention on Wetlands of International Importance Especially as Waterfowl Habitat*. It is administered by the International Union for Conservation of Nature and Natural Resources. It was signed in the town of Ramsar, Iran in 1971, hence its common name. The convention includes a list of wetlands of international importance and protocols regarding the management of these wetlands. Australia became a signatory in 1974.

Recharge — The infiltration of water into an aquifer from the surface (rainfall, streamflow, irrigation etc).

Specific yield (S_y) — The volume ratio of water that drains by gravity, to that of total volume of the porous medium. It is dimensionless

Surface water — (a) water flowing over land (except in a watercourse), (i) after having fallen as rain or hail or having precipitated in any another manner, (ii) or after rising to the surface naturally from underground; (b) water of the kind referred to in paragraph (a) that has been collected in a dam or reservoir

Tertiary aquifer — A term used to describe a water-bearing rock formation deposited in the Tertiary geological period (1–70 million years ago)

Transmissivity (T) — A parameter indicating the ease of groundwater flow through a metre width of aquifer section (taken perpendicular to the direction of flow), measured in m^2/d

Well — (1) An opening in the ground excavated for the purpose of obtaining access to underground water. (2) An opening in the ground excavated for some other purpose but that gives access to underground water. (3) A natural opening in the ground that gives access to underground water

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