Received: 14 February 2024 | Revised: 20 March 2024 | Accepted: 11 June 2024

DOI: 10.1002/hyp.15213

RESEARCH ARTICLE

WILEY

Effects of climate variability and change on groundwater impacts of forestry plantations

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Funding information
Green Triangle Forest Industries Hub; CSIRO; the Natural Heritage Trust, Australian Government; the South Australian Department of Water, Land and Biodiversity Conservation; South East Natural Resources Management Board; The South East Catchment Water Management Board; Forestry SA; the Glenelg Hopkins Catchment Management Authority; Forest and Wood Products Australia; the Green Triangle Regional Plantation Committee; Auspine; Green Triangle Forest Products; Timbercorp

Abstract

Quantifying water use of various water consumers is an essential part of sustainable water management. Annual evapotranspiration (ET) of plantation forests often exceeds that of dryland agriculture, which in South Africa and South Australia has resulted in restrictions on plantation development. In the latter case, water licences are issued to commercial forestry plantations to account for higher ET compared to dryland pasture. Unlike irrigated crops, it is not practicable to measure water use of plantations directly and so in South Australia a set of 'deemed' average water use rates has been applied since 2013, based on species and depth to groundwater. Since South Australia's 'deemed' rates were calculated, additional plot-scale measurements of annual ET from plantations <2 years old and post-canopy closure have been used to quantify various components of ET. This has enabled development of two empirical ET models for plantations in South Australia's Lower Limestone Coast, and facilitated an advanced understanding of the effect of plantations on hydrological processes, particularly in relation to groundwater use. In this study, we applied these models to estimate rotation-averaged annual ET and net groundwater impacts (net groundwater extraction plus recharge reduction compared to pasture) of plantations, driven by climate and groundwater depth, for comparison with the deemed rates. The modelling suggests that the groundwater impacts of plantations vary in space and time and that the deemed rates over-estimate these impacts, on average. Accounting for variation in the effects of climate on the various components of ET, both spatially and temporally, may allow for more flexible rules for water resource allocation than using any simple, rule-of-thumb approach.

KEYWORDS
evapotranspiration, groundwater management, groundwater recharge, plantation water use

1 | INTRODUCTION

Land use change can have important effects on hydrological processes and the benefits derived from use of natural resources, including water, which may require a public policy response (Greenwood, 2013; van der Zel, 1995). Such policy responses must balance policy complexity with ease of application to ensure sufficient accounting for natural variability identified through research.

Establishing tree plantations on natural grassland or previously cleared agricultural land may have the greatest impact on water resources of any land use change by increasing the proportion of precipitation that becomes evapotranspiration (ET) and reducing runoff to streams or groundwater recharge (Benyon et al., 2006; Bosch & Hewlett, 1982; Brown et al., 2005; Brown et al., 2013; Zhang et al., 2001). However, the impacts are not always the same, being influenced by soil depth and water holding capacity, climate, and physiological properties such as genotype and forest age (Andréassian, 2004; Rubilar et al., 2022). Further, if tree roots can enter the capillary fringe above the water table to access groundwater, trees can transpire additional water and plantations can become a net groundwater user (Benyon et al., 2006; Doody & Benyon, 2010). Tree plantations can, therefore, have important impacts on local and sometimes regional water resources, which in some circumstances may require regulation (Brookes et al., 2017; Greenwood, 2013; van der Zel, 1995). The first such regulation was a permit system introduced in South Africa in 1972, which restricted development of new plantations in catchments identified as being stressed (van der Zel, 1995). Subsequently, under South Africa's National Water Act (1998), plantation forestry was identified as a "Stream Flow Reduction Activity", potentially requiring forestry plantations to hold a water licence (Dye & Versfeld, 2007). Concerns about the water use impacts of forestry plantations have been raised in other countries including Ethiopia (Gebrehiwot, 2015), Australia, Brazil, Chile, China, India and the USA (Garcia-Chevesich et al., 2017), but few have developed specific policy responses based on a detailed understanding of how key hydrological processes such as interception, evaporation and transpiration vary with plantation species and age, climate, and accessibility of groundwater.

Planting trees often provides hydrological benefits, for example to restore local groundwater balances to prevent dryland salinity after previous clearing of deep-rooted natural vegetation (Schofield, 1992), for improving the physical and chemical properties of degraded soils (Garg, 1998; Mishra et al., 2004) or for reducing erosion risk (Spiekermann et al., 2022) and can provide direct economic and social benefits. In other cases, higher water use of tree plantations is seen as a threat to sustainable use of water resources that may require management or regulation (Anonymous, 2004; Brookes et al., 2017; Greenwood, 2013; SENRMB, 2019; van der Zel, 1995; van Dijk et al., 2006; Vertessy et al., 2003). It can also present a threat to nearby groundwater dependent ecosystems such as wetlands, cave aquifer systems and terrestrial and riparian vegetation (Deane et al., 2018).

In Australia, concerns over the water resource impacts of large-scale forestry plantations since the late 1990s has resulted in a need for accurate measurements of water use by the main commercial tree species (Anonymous, 2004; Greenwood, 2013). As research results become available, policy makers are faced with decisions on how to balance the practicability of policy implementation with accuracy and precision in achieving desired policy outcomes. An overly simplistic interpretation of the science may result in adverse policy outcomes, whereas high policy complexity may make application unworkable. The research presented here demonstrates the difference that would be made to policy outcomes by adopting a scientifically slightly more rigorous, but still implementable, policy response than a simple rule-of-thumb approach, to the issue of plantation forest water use licensing, using the Lower Limestone Coast (LLC) of South Australia as a case study.

The LLC region of South Australia provides an environment that sustains a vital commercial forestry industry. Softwood plantations (*Pinus radiata* D. Don) established over the past 130 years cover approximately 1050 km^2, providing a primary source of income for the region (SENRMB, 2019). These plantations were mostly established over deeper karstic groundwater (>10 m) and likely rely only on rainfall in a region devoid of surface water. *Eucalyptus globulus* Labill. ($\sim$400 km^2) were rapidly introduced the late 1990s, targeting shallow groundwater zones. Concerns were raised around potential lowering of the local groundwater levels with unknown levels of groundwater extraction which would lead to impacts to surrounding groundwater dependent ecosystems, especially wetlands (Brookes et al., 2017).

To fully understand and manage plantation impacts on water resources, accurate measurements and predictions of evapotranspiration (ET) across the entire rotation are needed, taking account of soil and groundwater recharge in the fallow period between rotations, lower ET in the early part of each rotation, as well as peak ET once the plantation has fully occupied the site. The consequences of precipitation and evaporative demand variability, both spatially and temporally, also need to be understood and accounted for.

To address these needs, the aims of this research were to: (1) based on previous research, develop a climate-driven model of plantation annual ET that includes accounting for lower water-use in the early part of each rotation and for spatial and temporal variability in annual precipitation and evaporative demand and (2) apply this model to a case study region, namely the LLC Prescribed Wells Area of southeast South Australia, where plantation water use is currently licensed, to compare modelled groundwater impacts of plantations across the region and over time with current deemed impacts and licensed water allocations from the 2013 Water Allocation Plan. Throughout this study, the term 'groundwater impacts' is used to refer to net groundwater extraction plus recharge reduction compared to pasture.

2 | METHODS

2.1 | Overview of the methods used to develop and apply the model

1. As described in detail in Section 2.3 below, two existing, semi-empirical models of evapotranspiration (Benyon et al., 2009; Benyon & Doody, 2009) were combined to enable estimation of whole-rotation actual annual evapotranspiration (ET_a), net groundwater extraction and recharge reduction compared to pasture, using rotation length, annual precipitation (P), annual potential evapotranspiration (ET_p), and depth to groundwater (less than or greater than 6 m) as model inputs.
2. As described in detail below in Sections 2.4 and 2.5.1, gridded daily P and ET_p estimates on a 0.1° grid covering the LLC (98 grid

points) for the period 1900 to 2020 were aggregated to annual totals. These were used as inputs to the model developed in (1) to derive rotation-average ET_a estimates at each grid point for rotations beginning in each year for a *E. globulus* rotation of 11 years, with and without groundwater accessible, and for a *P. radiata* rotation of 30 years, also with and without access to groundwater. For each grid point and each combination of rotation length (11 and 30 years) and groundwater accessibility (accessible, not accessible), the mean annual rotation-average ET_a and groundwater impacts were calculated for the entire 120 years and for the decades 1991–2000, 2001–2010 and 2011–2020. The periodic, rotation-average groundwater impacts for the four combinations of rotation length and groundwater accessibility were mapped across the study area to examine variability caused solely by spatial and temporal variability in climate (Results, Section 3.2).

3. As described in detail below in Section 2.5.2, data on actual plantation areas in 2022 by location and species were obtained from the plantation forestry companies. Maps of actual depth to groundwater were available for 1995, 2004, and 2021. The 2022 plantation area was overlaid onto each of the three groundwater depth maps to produce three maps of plantation area by species and depth to groundwater. These three maps were intersected with the gridded whole-rotation estimates of groundwater impacts for each of the past three decades, producing nine maps of groundwater impacts for the 2022 plantation estate. For each of the nine maps, the total groundwater impact summed for the whole plantation area present in 2022 was compared with the total 'deemed' groundwater impact used as the basis for water allocations in the 2013 LLC Water Allocation Plan (Results, Section 3.3).

2.2 | Description of the study region and current water licensing for commercial tree plantations

The study was conducted in the LLC Prescribed Wells Area in the lower southeast of South Australia, an area of $\sim$14 500 km^2 (SENRMB, 2019). Plantations currently cover about 10% of the region. The topography, climate, land use, geology, hydrogeology, and soils of the region are described in detail by Harrington et al. (2011), Harrington et al. (2013), SENRMB (2019) and summarized in Crosbie et al. (2015). In summary, the study area is a gently undulating coastal plain of marine origin located in the southeast corner of South Australia, bounded by the Southern Ocean on the west and south sides, the South Australia/Victoria border on the east side and the Murray-Darling Basin to the north. The region slopes gently towards the sea, with elevation increasing from sea-level in the west and south to 70 m in the north and northeast. In between, topographic highs are generally around 30–50 m. The Koppen climate classification is Temperate, with cool, wet winters and warm, dry summers. Monthly mean minimum and maximum temperatures are $\sim$5 and $\sim$13°C in Austral mid-winter (July) and $\sim$12 and $\sim$26°C in mid-summer (January). About 75% of the mean annual precipitation of $\sim$600–800 mm falls between April and October, with monthly means of around 100 mm in mid-winter and 25 mm in mid-summer. Soils are generally duplex, with sandy A horizons and clay B horizons. High infiltration rates of surface soils mean there is little surface runoff. Most of the region's water supplies for towns and agriculture are derived from two regional aquifer systems: the Gambier Tertiary limestone unconfined aquifer and the Dilwyn confined aquifer. Land use of the region is a mixture of dryland and irrigated agriculture, plantation forestry and native vegetation, including numerous, small wetlands, most of which are potentially dependent on groundwater (Brookes et al., 2017; Castellazzi et al., 2019; Doody et al., 2017). Extensive clearing of the original native vegetation and construction of drains in the past has resulted in a highly modified landscape, with <6% of the original vegetation left. Prior to the arrival of Europeans, 44% of the area was wetlands. It is now <6% (SENRMB, 2019).

A carefully designed water allocation plan (WAP), based on a rigorous, science-based understanding of the region's water resources, is essential to ensure long-term sustainable use of the groundwater resource for towns, industries, and the environment (SENRMB, 2019). Irrigation in the LLC requires a volumetric water licence and irrigation water use is metered via the pipe system (usually centre-pivot) used to move water from underground to crops. After a decade of research, consultation and planning, the LLC became the second jurisdiction in the world to adopt regulation of plantation forestry based on water use and the first to incorporate dry land plantation forestry into a regional groundwater licensing system (Brookes et al., 2017; Greenwood, 2013; SENRMB, 2019).

Of $\sim$1445 km^2 of plantation forestry in the LLC in 2013, $\sim$72% was *P. radiata* and 28% *E. globulus* (Downham & Gavran, 2018; SENRMB, 2019). Since 2013, these forests have required a licence for their groundwater impacts (Greenwood, 2013). Because it is not possible to meter plantation water use, licensing is based on simple rule-of-thumb, or 'deemed' rotation average water use (Harvey, 2018; SENRMB, 2019). It has been established that where these plantations are located with median depth to groundwater of 6 m or less, the trees can usually transpire groundwater (Benyon et al., 2006). Based on some of this work and using additional assumptions about lower ET at the beginning of each rotation, deemed rotation-average rates of groundwater extraction are currently 182 mm year^{-1} for *E. globulus* and 166 mm year^{-1} for *P. radiata*. In addition, at any location with commercial plantations, it is deemed that there is substantially lower groundwater recharge, irrespective of depth to the water table. This reduction is fixed as a proportion of the mean annual recharge expected under pasture in each location. The 2013 water allocations assumed that all forestry plantations present at that time reduced groundwater recharge to zero.

Plantations of *P. radiata* have been grown commercially in the region since the late 19th century. Currently this species is grown in rotations of 30–35 years, with several commercial thinnings and a final clear-fell harvest producing a mix of products including sawn timber, veneer and pulpwood. At the start of each new rotation, residual vegetation is cleared, and weeds are controlled with herbicides. The site may be deep ripped and mounded. Bare-rooted seedlings are planted along the rip-lines. Spacing is typically 3 m between rows and

2 m within rows, giving initial planting density of 1667 trees ha^{-1}. At ~10 years of age every fifth row is removed, with selective thinning of the remaining rows. There are usually two or three additional selective thinnings so that final stocking at harvest is ~200 trees ha^{-1}.

Commercial *E. globulus* plantations were first established in the LLC in 1998. These are grown on a 10–12-year rotation for pulpwood using similar establishment methods to *P. radiata* but usually at a lower initial stocking density (~1200 trees ha^{-1}) and there is no thinning.

2.3 | Evapotranspiration modelling and climate data

Our study used a comprehensive data set of ET_a and root zone soil water measurements collected over 10 years (1999–2008; 2–6 years of data from each site) from 23 plantation sites in the southeast of South Australia and southwest Victoria, Australia (Benyon et al., 2006; Benyon et al., 2009; Benyon & Doody, 2009; Benyon & Doody, 2015; Doody, Benyon, & Gao, 2023) to derive two empirical models that predict annual ET_a based on annual precipitation, annual potential ET (ET_p) and median depth to groundwater. One model applies to the early part of the rotation, up to two years after planting, while the other applies for most of the rotation, post-canopy closure. Details of the measurement methods and data used to derive the two models, as well as uncertainties in the measurements of individual ET components, have been published previously (Benyon et al., 2006; Benyon et al., 2009; Benyon & Doody, 2009; Benyon & Doody, 2015; Doody, Benyon, & Gao, 2023).

In this study we used interpolated annual precipitation and ET_p estimates (Jeffrey et al., 2001; Queensland Government, 2024; Stone et al., 2019) on a 0.1-degree grid across the study region in the two models to derive rotation-averaged ET_a at each grid point (98 points in total) for each of the past three decades (1991–2020) as well as 120-year (1901–2020), long-term averages.

2.3.1 | Plot-scale measurements of ET_a

Between 1999 and 2008, the Commonwealth Scientific and Industrial Research Organization (CSIRO) collected measurements of precipitation in the open, net precipitation under the tree canopies, transpiration (T), using sap flow sensors, evaporation from the forest floor (E), using mini lysimeters, and changes in root zone soil water to between 3 and 6 m depth, using a neutron moisture meter, in representative sample plots at 23 locations within the region, including 13 *E. globulus* and 10 *P. radiata* plantations. Measurements at approximately monthly intervals included totals of precipitation, throughfall and, in some cases, stem flow (four sites), evaporation from the forest floor, soil water, daily transpiration and at 18 sites, depth to groundwater. All measurement methods, results and measurement uncertainties have been published previously in Benyon et al. (2006, 2009); Benyon and Doody (2009); Benyon and Doody (2015); and Doody, Benyon, and Gao (2023). Although most of the measurements were confined to the period post-canopy closure, at three sites the measurements included the period of fallow after harvest and at two sites for 2 years after replanting with seedlings (Benyon & Doody, 2009, 2015).

2.3.2 | Modelling annual ET_a and groundwater recharge for fallow to age 2 years

Benyon and Doody (2009) observed a strong correlation (R^2 0.97, p < 0.01) between annual recharge and annual precipitation in the fallow period and up to 2 years after replanting with seedlings. They fitted a simple linear regression to this relationship which we have used to estimate recharge in the first 2 years of each rotation based on mapped annual precipitation:

$$R = 0.801 \times P - 207.8, \tag{1}$$

where R = recharge (mm year^{-1}) in any given year for a plantation <2 years old and P = precipitation in that year (mm year^{-1}).

2.3.3 | Comparison of early rotation recharge with estimates derived from water table fluctuations

Groundwater recharge estimated using the watertable fluctuation method (Brown et al., 2006; Crosbie et al., 2015; Latcham et al., 2007) in two groundwater observation wells located close to each other in *E. globulus* plantations in the 'Coles' groundwater management area (well numbers CLS002 and CLS049), provided additional validation of modelled groundwater recharge occurring before planting and after forest harvesting. Groundwater levels in CLS002 were measured intermittently between 1970 and 2000 and then two to four times per year between 2001 and 2011 and between 2015 and 2017. A new well was installed as a replacement in late 2015 with an overlap in monitoring of both wells in 2016 and 2017 in the 2 years post harvesting of blue gums in that area. Watertable rises measured in both wells in 2016 in the year following forest harvesting were used to calculate recharge using aquifer specific yield values of 0.15 and 0.20 (Brown et al., 2006; Earle, 2006). The interpolated rainfall for that year was used to predict recharge using Equation (1). This was compared with recharge estimated by the watertable fluctuation method (Section 3.1).

2.3.4 | Modelling annual ET_a for closed-canopy plantations

Benyon et al. (2009) used multiple linear regression analysis to fit an empirical relationship between closed-canopy annual ET_a and annual P, ET_p and groundwater accessibility. Previous research (Benyon et al., 2006, 2009), indicated some plantations in the region access groundwater in addition to rainfall: usually when median depth to groundwater (dtw) is <6 m. In the regression analysis, a dummy variable was used to represent those sites using groundwater. In subsequent application of the model, it was assumed that any closed-canopy plantation with dtw <6 m would use groundwater at the rate predicted by the empirical equation.

The single regression equation relating ET_a to P, ET_p and groundwater accessibility derived by Benyon et al. (2009) is given as Equation (2):

$$ET_a = 0.716 \times P + 542 \times D1 + 0.282 \times D1 \times P + 0.930 \times D2 \times ET_p - 552, \tag{2}$$

where: ET_a is actual annual evapotranspiration (mm year^{-1}); P is annual precipitation (mm year^{-1}); ET_p is potential evapotranspiration (Morton's wet aerial potential ET, Morton, 1983); D1 is a dummy variable having a value of 1 if the plantation does not access groundwater or 0 if it does; D2 is a dummy variable having a value of 1 if the plantation accesses groundwater and 0 if it does not.

For plantations without access to groundwater (assumed to be when dtw >6 m), Equation (2) simplifies to Equation (3):

$$ET_a = 0.998 \times P - 10. \tag{3}$$

However, use of this equation in the LLC will always give ET_a each year of around 11 mm lower than P, meaning modelled recharge of ∼11 mm year^{-1}. This is an artefact of fitting the regression line rather than being real recharge and therefore for the purposes of modelling whole-rotation annual ET_a and recharge we have assumed that after canopy closure, at locations with >6 m dtw, annual ET_a is always equal to annual P and therefore net recharge is zero.

For plantations with access to groundwater (dtw <6 m), Equation (2) simplifies to Equation (4):

$$ET_a = 0.716 \times P + 0.930 \times ET_p - 552. \tag{4}$$

2.4 | Calculation of rotation-average ET_a, recharge and groundwater extraction

To apply the models to develop estimates of rotation-average ET_a and groundwater impacts across the LLC, various assumptions about rotation length, age at which canopy closure occurs in each species, and ET_a between age 2 years and canopy closure and after thinning were necessary (summarized in Table 1). Also detailed for comparison are the assumptions used in calculating the 'deemed' average water use rates described by Harvey (2018) to derive rotation-average

TABLE 1 Assumptions used for modelling whole-rotation average annual ET_a for this study and by Harvey (2018).

	This study		Harvey, 2018	
	P. radiata	***E. globulus***	***P. radiata***	***E. globulus***
Rotation length (years)	30	11	36	11
Years to canopy closure	5	2	5	2
Times thinned	3	0	4	0
Max dtw for extraction (m)	6	6	6	6
Age at maximum groundwater extraction (years)	6–30	3–10	10	6–10
ET_a post canopy closure, dtw >6 m	$ET_a = P$	$ET_a = P$	$ET_a = P$	$ET_a = P$
Max ET_a, dtw <6 m (mm year^{-1})	$0.716 \times P + 0.93 \times ET_p - 552$	$0.716 \times P + 0.93 \times ET_p - 552$	P + 364	P + 364
Recharge during 1-year fallow (mm year^{-1})	$0.801 \times P - 207.8$	$0.801 \times P - 207.8$	1.2 × local pasture recharge rate	1.2 × local pasture recharge rate
Recharge 1st year after planting	$0.801 \times P - 207.8$	$P - ET_a$	1.0 × local pasture recharge rate	0.8 × local pasture recharge rate
Recharge 2nd year after planting	$P - ET_a$	$P - ET_a$	0.8 × local pasture recharge rate	0.4 × local pasture recharge rate
Recharge 3rd year after planting	$P - ET_a$	$P - ET_a$	0.6 × local pasture recharge rate	0
Recharge 4th year after planting	$P - ET_a$	$P - ET_a$	0.4 × local pasture recharge rate	0
Recharge 5th year after planting	$P - ET_a$	$P - ET_a$	0.2 × local pasture recharge rate	0
Max recharge 1st year after thinning	0	Not applicable	0.5 × local pasture recharge rate if dtw >6 m	Not applicable
Rate of ET_a decline after peak ET_a, dtw <6 m	No decline	No decline	11 mm year^{-1} and each thinning reduces ET_a to P + 100 mm for 1 year	No decline

Abbreviations: dtw, median depth to watertable; ET_a, annual actual evapotranspiration; ET_p, annual potential evapotranspiration; P, annual precipitation.

groundwater extraction for *P. radiata* and *E. globulus* plantations at locations with median depth-to-watertable <6 m and rotation-average recharge reduction (compared to pasture) for each species at all locations within the LLC (Table 1). Of note, for groundwater allocations to plantations present in 2013, the groundwater extraction estimates of Harvey (2018) were used, but recharge under current plantations was assumed to be zero. That is, for calculating recharge reduction compared to pasture, no allowance was made for recharge in the early part of each rotation or after thinning.

2.5 | GIS analysis to apply the model to estimate impacts of plantations existing in the lower limestone coast in 2022

2.5.1 | Gridded groundwater impacts

Modelled rotation-average groundwater impacts were calculated for each of the 98, 0.1° grid points, for each of four combinations of rotation length (11 and 30 years) and dtw (<6, >6 m) using annual climate data at each point from 1991 to 2000, 2001 to 2010, 2011 to 2020 and 1900 to 2020. See Figure 1 for an example of gridded groundwater impact for the 30-year rotation, shallow watertable scenario for a dry decade and an average decade. Simple descriptive statistics, including mean and standard deviation, were used to examine the spatial variability between grid points in modelled groundwater impacts.

2.5.2 | GIS analysis of total groundwater impacts of plantations present in 2022

Using a geographic information system (GIS), he gridded rotation-average estimates of groundwater impacts were intersected with the 2022 LLC plantation estate, using data on plantation area by species provided by the forestry companies, and the LLC unconfined aquifer management areas (Figure 1). For examining the effects of decadal variations in climate and depth to groundwater on net groundwater impacts across the LLC, nine scenarios were modelled. All scenarios used maps of the 2022 plantation estate (Figure 1) but with climates from three different decades (1991–2000, 2001–2010, and 2011–2020) and dtw mapped in 1995, 2004 and 2021. In addition to calculating total groundwater impacts across the region, modelled groundwater impacts were calculated separately for three groundwater management areas out of 61 within the LLC, classified in 2013 as 'very high-risk management areas' due to high concentrations of plantations (SENRMB, 2019). These were 'Coles', 'Short' and 'Zone 2A' (Figure 1).

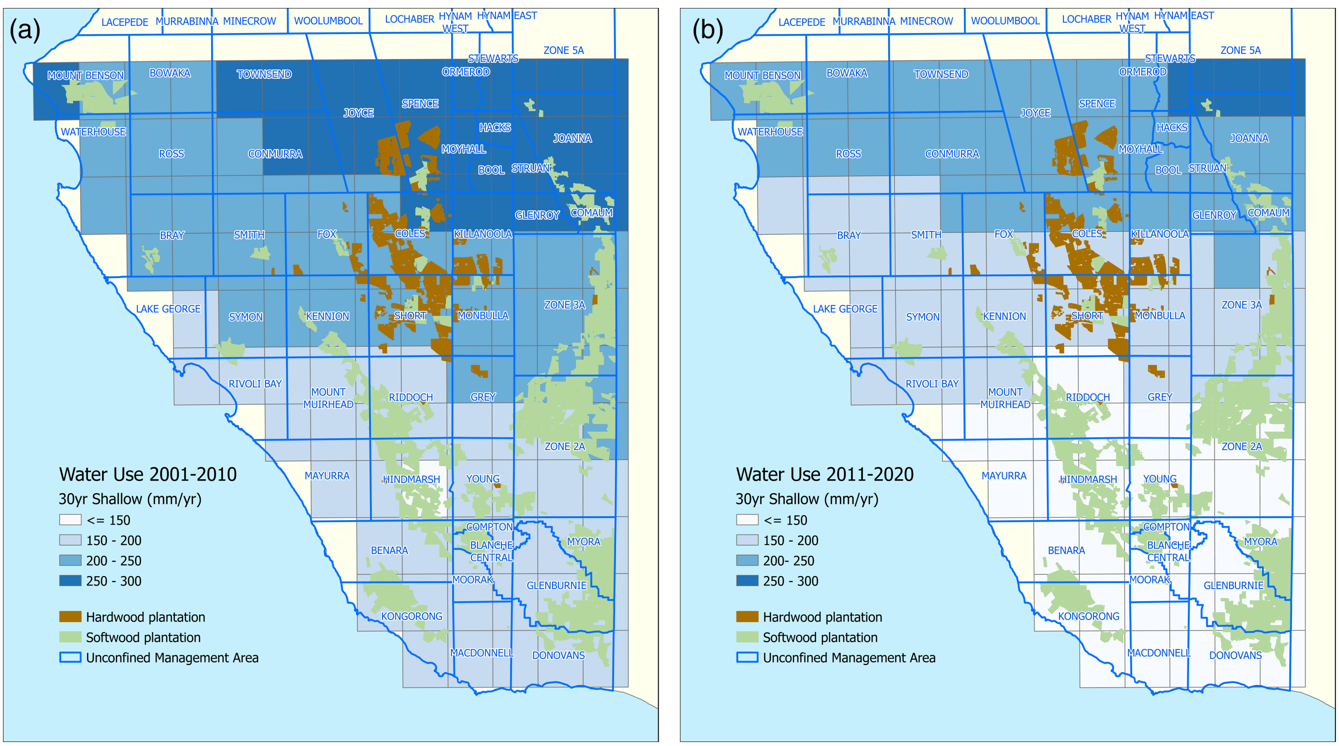

FIGURE 1 Maps showing names and locations of groundwater management areas within the Lower Limestone Coast and existing hardwood (*E. globulus*) and softwood (*P. radiata*) plantations in 2022. 'Unconfined Management Area' refers to the Gambier Tertiary limestone unconfined aquifer management areas. Black lines show the 0.1 degree grid. White and blue shading shows examples of the modelled hypothetical groundwater impacts of plantations (i.e. the gridded groundwater impacts estimates): in this case the 30-year rotation with <6 m dtw (not accounting for actual plantation locations and dtw) using (a) 2001–2010 (dry) and (b) 2011–2020 (average) rainfall climate scenarios.

3 | RESULTS

3.1 | Validation of early rotation and post-harvest recharge using the watertable fluctuation method

Watertable levels and fluctuations from pre-plantation establishment on ex-pasture in the early 2000s to harvesting of the plantations in 2015 and for the post-harvesting period to 2023 around observation wells CLS002 and CLS049 are shown in Figure 2. The most relevant period for model validation is the year or two of plantation establishment (2002–2003) and first year after harvest (2016) when the land would have been largely bare of vegetation, as this is when the model (Equation 1) predicts high recharge compared to pasture.

In 2001, prior to any plantation establishment in the area and with rainfall of 620 mm close to the long-term average of 640 mm, recharge estimated using specific yield (Sy) of 0.15 was 135 mm, which is similar to the rate of 120 mm assumed for pasture in the Coles management area in the 2013 WAP and much less than the 277 mm predicted by Equation (1). Despite below average rainfall in 2002 (586 mm), calculated recharge was higher (219 mm for Sy of 0.15 and 292 mm for Sy of 0.20, compared with 262 mm predicted by the model for fallow land). Commencement of plantation establishment in that year is likely to have resulted in removal of pasture, reducing ET_a and increasing recharge in line with model predictions. The years 2003 and 2004 were wet years and recharge estimated using Sy of 0.15 was 338 and 371 mm compared with 370 and 368 mm predicted by the model for the early rotation period. From 2005, when canopies of the plantations would have reached closure and net groundwater extraction would have started, there was a steady decline in watertable levels, with no fluctuations between 2005 and 2008 during a prolonged drought period. A wetter year in 2009 (686 mm rainfall) resulted in a rise in the water table that winter, although this is likely to have been exceeded by groundwater extraction over the following summer. Water level data for the period 2010 to 2015 were intermittent and therefore are not shown but generally the watertable depth remained at around 7 m.

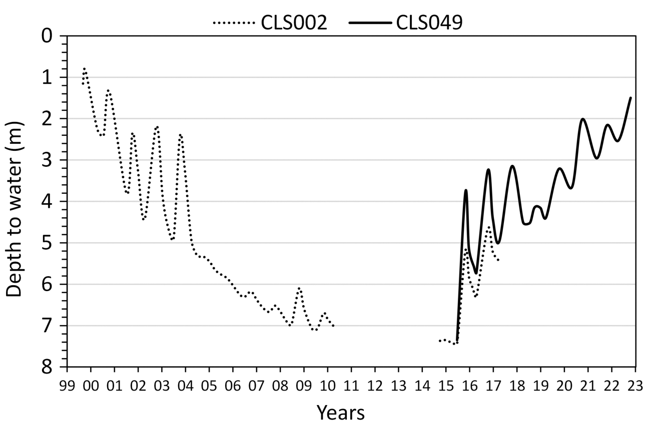

FIGURE 2 Depth to watertable in observation wells CLS002 and CLS049 from pre-planting (2001) to post-harvest (2016).

The plantations were not replanted after harvesting in 2015 and the type of vegetation cover that established after harvesting of the plantations is not known but there has been a notable rise in the watertable over the past 7 years. For the post-harvest year, 2016, calculated recharge in CLS002 was 332 and 442 mm using Sy of 0.15 and 0.20, compared with 370 mm predicted by Equation (1). For the new well, CLS049, recharge that year was 534 mm (Sy 0.15) and 732 mm (Sy 0.20), which is higher than predicted by Equation (1), and would correspond to Sy of 0.11.

These results indicate that recharge estimated independently using the watertable fluctuation method was similar to, or greater than, that predicted by Equation (1) for the year following harvest. This provides further evidence that Equation (1) gives reliable, and perhaps conservative, predictions of recharge in the period between rotations.

3.2 | Modelled groundwater impacts of plantations in the LLC and comparison with deemed rates

The modelled decadal average groundwater impacts of plantations vary across the study area and from decade to decade (Figures 3 and 4). Predicted actual groundwater impacts are not shown in Figures 3 and 4, as these do not take account of actual groundwater depths or plantation locations. Rather, these are the expected impacts that would occur if each combination of groundwater accessibility (<6, >6 m) and rotation length (11 and 30 years) was present at each 0.1 degree grid point.

Modelled groundwater extraction increases from south to north and is highest in the northeast (Figures 3a, c, 4a, c) and is lower for the decade 2011–2020, which had mean annual P closer to the 120-year average, than for 2001–2010, which was the driest decade of the past 120 years.

Modelled groundwater extraction for the 30-year rotation (*P. radiata*) averaged 176 mm year^{-1} using the 2011–2020 climate but 219 mm year^{-1} using the 2001–2010 climate (Table 2). For the 11-year rotation (*E. globulus*) the decadal means were respectively 163 and 211 mm year^{-1} (Table 2). Decadal climate had less effect on recharge reduction, the respective decadal means being 91 and 95 mm year^{-1} (long rotation) and 61 and 64 mm year^{-1} (short rotation) for the average and dry decades.

Spatial variation in modelled impacts was quite high, with standard deviations among grid points of ~17% to ~48% of regional mean values in each decade (Table 2). In the decade with average precipitation (2011–2020), mean groundwater extraction varied across the region from a minimum of 103 mm year^{-1} to a maximum of 254 mm year^{-1} for the long rotation and from 80 to 253 mm year^{-1} for the short rotation, compared with the deemed rates of 166 and 182 mm year^{-1} for long and short rotations that are currently applied uniformly across the LLC (Table 2). For the same decade, recharge reduction varied spatially from 30 to 164 mm year^{-1} for the long rotation and from 4 to 129 mm year^{-1} for the short rotation, compared to the deemed rates for existing plantations that vary from 50 mm to 200 mm across the LLC (Table 2).

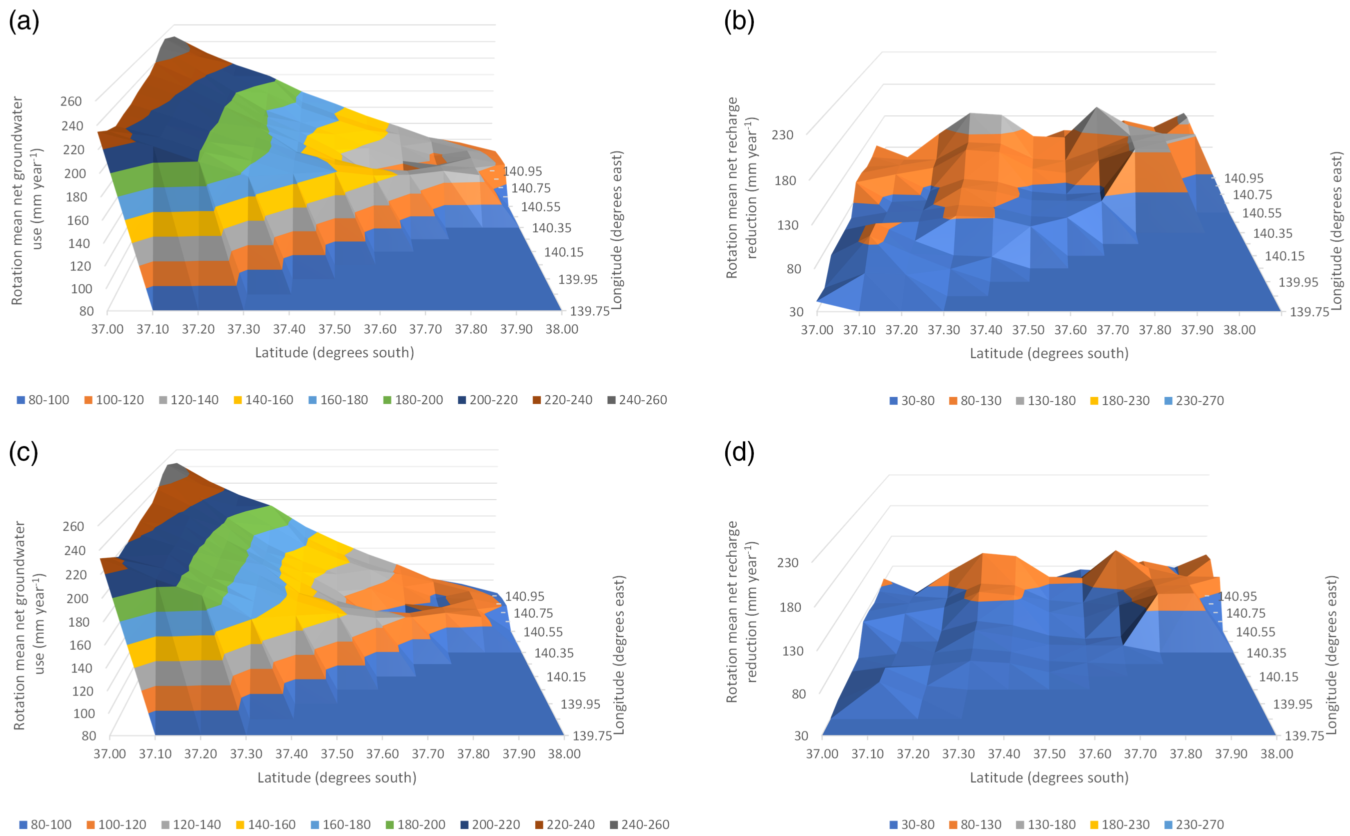

FIGURE 3 Three-dimensional representations of modelled rotation average groundwater impacts for the decade 2011–2020 across the Lower Limestone Coast for: (a) 30 year rotation, groundwater accessible; (c) 11 year rotation, groundwater accessible; (b) 30 year rotation, groundwater not accessible; and (d) 11 year rotation, groundwater not accessible. These represent the expected impacts at each grid point if each combination of rotation length and groundwater accessibility was present at that location. These do not represent actual impacts as they do not account for actual plantation locations, rotation lengths and depths to groundwater. The flat blue area in the southwest corner (bottom right) of each panel is ocean.

Deemed rates of groundwater extraction and recharge reduction assumed for the 2013 LLC (WAP) and calculated by Harvey (2018) are also presented in Table 2. The regional average modelled groundwater extraction is 10 mm year^{-1} higher than the deemed values for the long rotation but 19 mm year^{-1} lower than deemed for the short rotation. The deemed values do not account for any spatial or climate-induced temporal variation in groundwater extraction. The 2013 WAP did not allow for any recharge under existing plantations and therefore deemed recharge reduction rates assumed in the WAP are substantially higher than predicted by the model, with the greatest difference being for short rotations. For 2011–2020, the WAP regional mean recharge reduction is 32% greater than modelled for the long rotation (120 mm year^{-1} deemed compared with 91 mm year^{-1} modelled) and 97% greater than modelled for the short rotation (120 mm year^{-1} deemed compared with 61 mm year^{-1} modelled).

3.3 | Modelled regional impacts of existing plantations

The previous section illustrated how spatial and temporal variation in climate would affect hypothetical groundwater impacts of plantations but did not consider actual groundwater impacts of plantations that existed in 2022, accounting for locations of plantations (Figure 1) within the LLC. Here, we summarize GIS analysis results applying the model to existing plantations across the LLC in 2022 using nine different scenarios (Section 2.5.2; climate from three decades and dtw maps from 1995, 2004 and 2021).

Modelled total groundwater impacts for 1343 km^2 of plantations in the LLC (area in 2022) and deemed total impact from the 2013 WAP which were based on a 7.5% greater plantation area (1444 km^2) are presented in Table 3. Table 3 presents total volumes summed for the whole Lower Limestone Coast, in ML, also expressed as percentages of the 2013 WAP value in brackets. Climate and mapped median dtw both had substantial impacts on modelled groundwater impacts. Averaged across the three dtw scenarios, the total impact was 259 530 ML year^{-1} (84% of the 2013 WAP total) using the 2001–2010 climate but only 225 541 ML year^{-1} (73% of the WAP) using the 2011–2020 climate, and 242 691 ML year^{-1} (79% of the WAP) using the 1991–2000 climate. There were also differences between the three dtw scenarios, with the mean impact reducing from 253 963 ML year^{-1} (82%) using the 1995 dtw to 245 341 ML year^{-1} (80%) using the 2004 dtw and 228 455 ML year^{-1} (74%) using the 2021 dtw. This is due to increasing dtw in some areas, meaning that

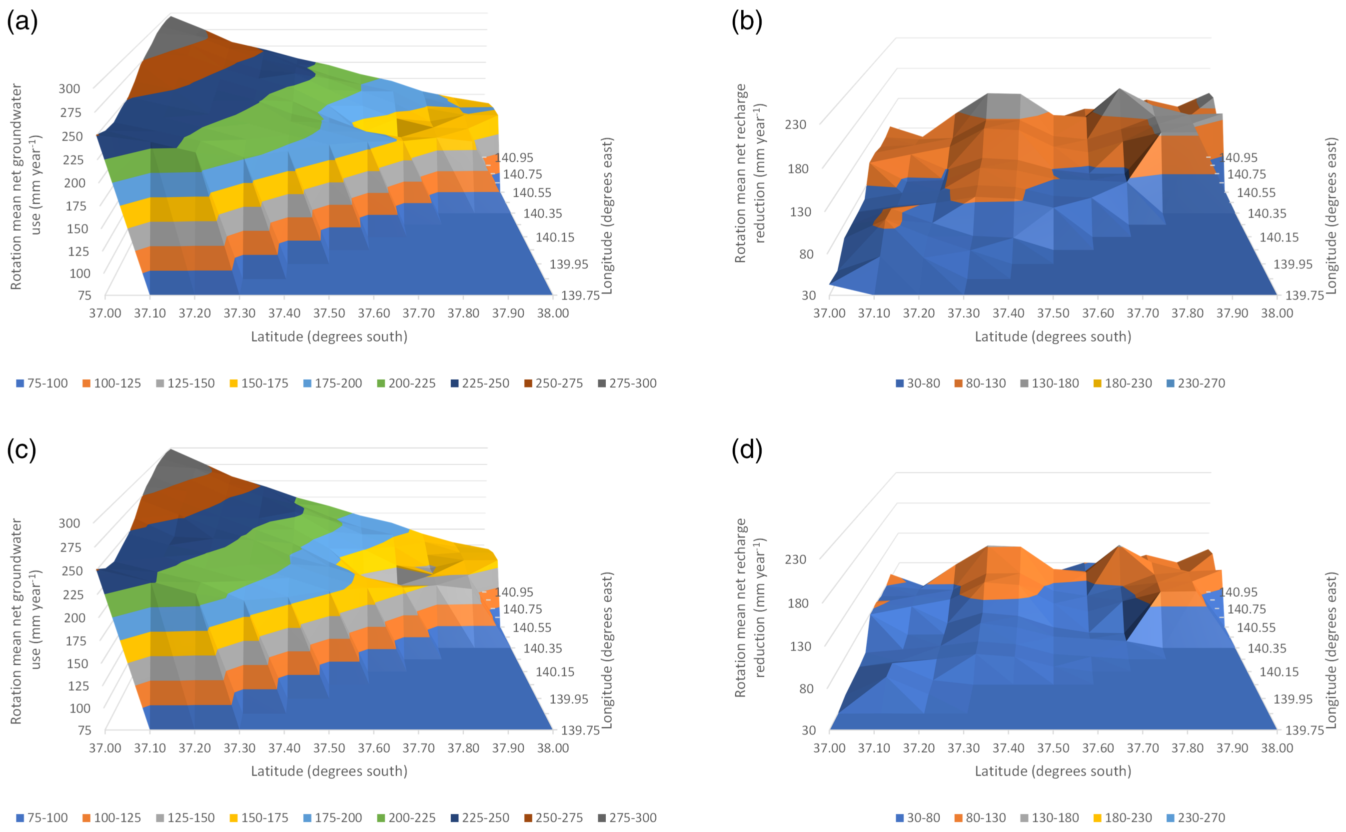

FIGURE 4 (a,b,c,d) The same as Figure 3, but for the decade 2001–2010.

over time, a smaller plantation area has mapped dtw less than the 6 m threshold assumed for groundwater extraction.

For all nine scenarios, analysis suggests the total groundwater impact is less than the ∼284 746 ML year^{-1} assumed in the 2013 WAP (after adjustment for the 7.5% reduction in plantation area). The smallest difference from the 2013 WAP data was 12 169 ML year^{-1} (4% of the 2013 WAP) less, and when assuming the driest climate (2001–2010) is used with the shallowest dtw (1995); the largest difference from the 2013 WAP was 71 206 ML year^{-1} (23%) when adopting the wettest climate (2011–2020) with the deepest dtw (2021). The mean difference, averaged across all nine scenarios, is 42 160 ML year^{-1}, or 15% of the deemed total impact determined for the 2013 WAP after adjustment for reduced plantation area.

3.4 | Modelled groundwater impacts for three 'very high risk' groundwater management areas

High concentrations of plantations in management areas Coles, Short and Zone 2A in 2013 (Figure 1) meant that these were considered very high-risk groundwater management areas and groundwater was deemed to be overallocated. To bring the groundwater back into balance, there has been a prohibition on replanting some of these plantations once the current rotation is completed. Consequently, between 2013 and 2022, there were reductions in plantation area of 42% in Coles, 14% in Short and 2% in Zone 2A (Table 4).

In Coles and Short, the choice of which dtw map to use in the analysis had little effect (<1%) on the modelled groundwater impacts of plantations (Table 4), meaning that the variation in dtw over the past 30 years has little effect on plantation impacts on groundwater in those management areas. In these two areas, using the most recent climate (2011–2020) in the model resulted in lower predicted groundwater impacts than the deemed impacts after adjustment for reduced plantation area. The difference was ∼3000 ML year^{-1} (12%) in Coles and ∼5400 ML year^{-1} (18%) in Short. In contrast, dtw had a strong influence on predicted groundwater impacts of plantations in Zone 2A, with 26 755 ML year^{-1} impact using the 2021 dtw compared with 30 598 ML year^{-1} using 2004 dtw, 37 209 ML year^{-1} using the 1995 dtw and 32 375 ML year^{-1} deemed impact (using the deemed rates in the WAP) following adjustment for the reduced plantation area. Based on the most recent climate and dtw mapping, modelled groundwater impact of plantations in Zone 2A was ∼1280 ML year^{-1} (5%) less than the adjusted deemed impact used by the WAP and using the 1995 dtw the modelled impact is 7% lower than the deemed impact.

4 | DISCUSSION

While concerns about the water resource impacts of forestry plantations have been raised in at least half a dozen countries (Garcia-Chevesich et al., 2017; Gebrehiwot, 2015), only two

TABLE 2 Summary statistics for modelled decadal mean groundwater impacts across the LLC for an average decade (2011–2020) and the driest decade (2001–2010).

	30 year, <6 m dtw	30 year, >6 m dtw	11 year, <6 m dtw	11 year, >6 m dtw
2011–2020 climate (average)				
Mean	176	91	163	61
Std. deviation	42	30	48	29
Min.	103	30	80	4
Max	254	164	253	129
2001–2010 climate (dry)				
Mean	219	95	211	64
Std. deviation	38	31	44	29
Min.	141	34	120	9
Max	296	168	299	133
Deemed values, 2013 LLCWAP				
Mean	166	120	182	120
Std. deviation	0	34	0	34
Min.	166	50	182	50
Max	166	200	182	200
Deemed values using Harvey (2018) assumptions				
Mean	166	100	182	94
Std. deviation	0	29	0	27
Min.	166	42	182	39
Max	166	166	182	156

Note: All values are rotation-average annual impacts on groundwater in mm year^{-1}.

TABLE 3 Total groundwater impacts (net extraction plus recharge reduction compared to pasture) using the model applied to the 1343 km^2 of plantations that existed in the Lower Limestone Coast (LLC) in 2022 for the nine combinations of decadal climate and mapped median depth to watertable.

	Climate decade		
Median dtw year	1991–2000	2001–2010	2011–2020
1995	253 975 (82.5)	272 577 (88.5)	235 337 (76.4)
2004	245 452 (79.7)	262 826 (85.4)	227 746 (74.0)
2021	228 637 (74.3)	243 187 (79.0)	213 540 (69.4)
2013 LLCWAP	307 834 (100)	307 834 (100)	307 834 (100)
2013 LLCWAP using 2022 plantation area	284 746 (92.5)	284 746 (92.5)	284 746 (92.5)

Note: Deemed total impact from the 2013 Lower Limestone Coast Water Allocation Plan (LLCWAP) is provided for comparison. All values are in ML year^{-1}. Note plantation area for the 2013 water allocation plan was 1444 km^2. Figures in brackets give the values expressed as percentages of the 2013 LLCWAP total.

TABLE 4 Modelled total net groundwater impacts of plantations in three very high-risk groundwater management areas using the 2022 plantation area, the 2011–2020 climate and the 1995, 2004 and 2021 dtw maps.

	Coles	Short	Zone 2A
Plantation area in 2013 (km^2)	139.7	119.5	209.6
Plantation area in 2022 (km^2)	81.5	102.6	204.8
1995 dtw (ML year^{-1})	21 094 (51.1)	25 231 (64.1)	37 209 (98.0)
2004 dtw (ML year^{-1})	20 921 (50.7)	25 246 (64.2)	30 598 (80.6)
2021 dtw (ML year^{-1})	21 044 (51.0)	25 194 (64.0)	26 755 (70.5)
LLCWAP 2013 (ML year^{-1})	41 256 (100)	39 347 (100)	37 950 (100)
LLCWAP using 2022 plantation area, 2004 dtw (ML year^{-1})	23 950 (58.1)	30 665 (77.9)	32 375 (85.3)

Note: Lower Limestone Coast Water Allocation Plan (LLCWAP) refers to the LLCWAP. Figures in brackets give the values expressed as percentages of the 2013 LLCWAP total.

jurisdictions have enacted policies to directly manage these impacts and therefore the literature on water management policy responses to large scale plantation developments is scant. In South Australia's LLC, the regional-scale plantation ET and groundwater impacts predicted using two simple empirical models derived from a decade of plot-scale measurements which elucidate the key hydrological processes, are less than those currently applied by resource managers, using a simple rule-of-thumb approach. Averaged across nine scenarios of climate and dtw, the modelled impact is ~15% less than would be deemed based on the 2022 plantation area and 21% less than in the 2013 WAP. Modelled groundwater impacts vary in space and in time, being least in the southern part of the region and during wetter decades. Modifying water licences to account for spatial variation may encourage plantation developments in areas with lower groundwater impacts and discourage plantation development in areas of greater impact.

4.1 | Using model outputs for groundwater management

For all nine scenarios modelled, the total groundwater impact of plantations is less than deemed in the 2013 WAP. Under the current plan, the area of plantations in three 'very high-risk' groundwater management areas must be reduced by ~20 000 ha (~43%) compared to the 2013 area, of which ~8000 ha has already been removed. The 2013 plan deemed that 46 878 ha of plantation then present in Coles, Short and Zone 2A had a groundwater impact totalling 118 553 ML year^{-1}, resulting in overallocation of 49 634 ML year^{-1}. Our modelling suggests that the current groundwater impact of plantations in these three areas is 72 993 ML year^{-1} (38% less than the deemed impact in the 2013 WAP), meaning overallocation is now 4074 ML year^{-1}, an impact equivalent to ~1630 ha of *E. globulus* plantation or ~1543 ha of *P. radiata* plantation in areas with <6 m dtw.

Across the entire LLC, using the 2011–2020 climate and 2021 dtw mapping, the modelled groundwater impact of the current plantation estate is ~71 000 ML year^{-1} (23%) lower than deemed, equivalent to ~28 000 ha of plantations. This lower impact is due mainly to assumptions about groundwater recharge in the early part of each rotation. Measured recharge in the first 2 years of a rotation is 2–3 times greater than that assumed for pasture (Benyon & Doody, 2009), not zero as assumed in the 2013 WAP, meaning an average of ~24% less recharge reduction for a 30-year rotation and ~49% less for an 11-year rotation.

More generally, the results demonstrate that maximizing the use of existing data on vegetation water use for water licensing of land uses for which actual water use is not easily monitored can be achieved without the need for complex models. The climate-driven empirical models used here do not require any modelling expertise but would provide more accuracy and flexibility in policy outcomes than a rule-of-thumb approach.

4.2 | Model uncertainties

The two empirical models used here are based on a decade of previously published plot-scale measurements of the water balance at 23 plantation sites within the study region. Despite this, there are still gaps in our understanding of how plantation water use varies in space and time. These are discussed below.

4.2.1 | Rate of increase in water use between replanting and canopy closure

Evapotranspiration from fallow land and very young plantations is much less than that of plantations post-canopy closure. A linear increase in annual ET after age 1 year to a maximum at canopy closure was assumed. However, little data exists for *P. radiata* plantations aged ~2 to 6 years and for *E. globulus* aged 1–3 years and therefore the rate of increase in ET up to canopy closure is uncertain.

4.2.2 | Effect of thinning on water use and recharge

Pinus radiata plantations in the region are thinned three or four times each rotation, reducing stand density from 1667 trees ha^{-1} at planting to ~200 stems ha^{-1} at final harvest. The effects of thinning in these plantations on ET_a is currently not known. Thinning reduces stand density and therefore leaf area and sapwood area, but also increases penetration of sunlight and turbulence within the canopy, which may result in higher rates of transpiration from the remaining trees. Based on Harvey (2018), for estimation of groundwater extraction, the current WAP assumed some reduction of ET_a and an increase in recharge in the first year after each thinning. However, our analysis conservatively assumed no effect of thinning on ET_a or recharge. A recent global analysis of research on the effects of forest thinning on hydrological processes (del Campo et al., 2022) found that thinning does have significant effects on the various components of evapotranspiration, but overall, across all studies they analysed, there was no statistically significant effect on ET_a as a whole. del Campo et al. (2022) concluded that at least 50% of the stand basal area must be removed to see significant effects on hydrological processes. In the *P. radiata* plantations in the LLC, each thinning typically removes only 20%–35% of the basal area.

4.2.3 | Maximum depth from which plantations extract groundwater

Based on Benyon et al. (2006) and Benyon et al. (2009), plantation forests in the LLC are assumed to extract groundwater when median depth-to-watertable is <6 m. This assumption needs to be validated more broadly as it has a strong influence on the deemed groundwater use estimates for plantations in the region, especially in groundwater management areas such as Zone 2A where dtw is close to the 6 m threshold (Table 4). An additional complication is the effect of spatial variability on soil properties that may influence accessibility of groundwater and rates of extraction. Using bias-corrected satellite-based estimates of net water balances across the study region at a much higher density than previously available (250 m pixels), Crosbie et al. (2015) examined the relationship between net water balances and dtw in relation to vegetation type and soil texture. They agreed that 6 m is the maximum depth for groundwater use by plantations on light textured soils (clay content <5%) but their results suggested increasing depth for groundwater extraction with increasing clay content, being respectively 9 m, 13 m and 16 m for clay contents of 5%–10%, 10%–15% and 15%–25%. However, Benyon et al. (2006, 2009) found no evidence for extraction from >6 m at 10 plantation sites in the region, most with watertables ranging in depth from 8 to 16 m. Satellite-based estimates of ET_a require further validation against ground-based measurements for plantations located on soils with high clay content. Development, validation and application of satellite-based monitoring of plantation water use in the region is on-going (Doody, Gao, et al., 2023; Myers et al., 2023), and should ultimately provide more robust, ET_a estimates at finer temporal and spatial resolutions.

4.2.4 | Effect of falling water table and root impeding layers on groundwater extraction

In some parts of the region there has been a persistent fall in watertable levels (increasing dtw). Because the model assumes a step change from maximum groundwater use to no groundwater use below the 6 m median depth threshold, reductions or rises in the watertable can make substantial differences to modelled plantation impacts, especially in areas where the watertable is close to the 6 m threshold. For example, increasing depth to the watertable in Zone 2A has a substantial influence on the modelled impacts of plantations in different decades. Therefore, it is important to understand how the dynamic change in the watertable over time affects groundwater extraction by plantations: do plantations really stop using groundwater if the watertable falls from slightly less than to slightly greater than 6 m? On a similar note, do plantations extract groundwater at the same rate across the entire range in dtw over which extraction can occur and how is this affected by soil properties such as clay content?

4.2.5 | Effect of plantation age on groundwater use in *Pinus radiata*

Canopy closure in *P. radiata* plantations typically occurs at around 5–6 years of age. Ignoring possible effects of thinning, the canopy essentially remains closed for the next 25–30 years before the final harvest. Modelling here assumed peak ET_a continued for the entire rotation period after canopy closure, whereas Harvey (2018) assumed that at locations with shallow groundwater, ET_a in *P. radiata* peaked at age 11 years and from then on declined at 11 mm year^{-1}. Scott and Prinsloo (2008) observed that ET_a of *Eucalytus* and *Pinus* plantations in South Africa did begin to decline but not until 13 years of age in *Eucalyptus* and 26 years in *Pinus*, which for *E. globulus* is beyond the end of a typical rotation in the LLC. In the case of *Pinus*, the Scott and Prinsloo observations suggest there might be a diminished effect on groundwater in the final few years of each rotation, but certainly for much less of each rotation than assumed by Harvey (2018).

4.3 | Moving beyond rules-of-thumb and empirical models

While the water use of non-irrigated vegetation is difficult to measure directly, improved satellite-based methods for monitoring ET have been in development over the past 20 years. Recently, Doody, Benyon, and GAO (2023) applied machine learning methods in combination with region specific climate data, soil moisture and canopy greenness (after Doody, Gao, et al., 2023) to bias correct a national ET product (McVicar et al., 2022) based on field plantation ET_a data for both species collected in the LLC region (Benyon et al., 2006). This dataset provides dynamic plantation ET estimates at a temporal scale of monthly from 2000 until current using 30 m spatial resolution. Modelled 'AMLETT' ET outputs include calibration related to periods of plantation clear fall to enable long-term assessment of ET across the plantation estate for *P. radiata* and *E. globulus*. Another remote sensing-based ET model, FORETHIR, has been developed and calibrated to more recent monitoring in the LLC (Myers et al., 2023). A similar approach to the modelling presented within could be tested beyond the 23 sites reported, to potentially improve the estimates of net groundwater impacts regionally. As the national CMRSET/TERN ET dataset (McVicar et al., 2022; https://tern-landscapes.earthengine.app/view/cmrset-landsat-v22), is updated regularly, continual assessment of regional net groundwater impacts is possible, and the adoption of remote sensing tools may reduce the influence of the five areas of uncertainty noted above.

5 | CONCLUSION

Sustainable management of water resources requires accurate measurements or estimates of the impacts of all water users. When it is not practicable to directly meter or measure ET, the best available model of ET should be used that accounts for natural variability within the system, without necessarily requiring specialized expertise to apply. Accounting for variation in ET through the rotation and the impacts of climate on ET in both space and time may result in more flexible rules for water allocation and more accurate accounting of water use and water availability than using a uniformly applied, rule-of-thumb approach.

ACKNOWLEDGEMENTS

We gratefully acknowledge funding provided for this study by the Green Triangle Forest Industries Hub, especially Liz McKinnon and

Scott Whicker. We also gratefully acknowledge the substantial technical support provided for collection of the underlying data from 1999 to 2008 by Michele Michael, Colleen Bernie, David Gritton, Kevin Nickolls, Phill Keeley, Vijay Koul, Leroy Stewart and John Larmour. Collection and analysis of these data was funded by the CSIRO, the Australian Government's Natural Heritage Trust, the South Australian Department of Water, Land and Biodiversity Conservation, South East Natural Resources Management Board, The South East Catchment Water Management Board, Forestry SA, the Glenelg Hopkins Catchment Management Authority, Forest and Wood Products Australia, the Green Triangle Regional Plantation Committee, Auspine, Green Triangle Forest Products and Timbercorp. Access to field sites was provided by Woakwine, ForestrySA, Green Triangle Forest Products, Timbercorp, Victorian Department of Primary Industries and Balak Park Pty Ltd.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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How to cite this article: Benyon, R. G., Doody, T. M., Lawson, J., Hay, A., & Myers, B. (2024). Effects of climate variability and change on groundwater impacts of forestry plantations. *Hydrological Processes*, *38*(7), e15213. https://doi.org/10.1002/hyp.15213