A groundwater recharge perspective on locating tree plantations within low rainfall catchments to limit water resource losses

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Abstract

Despite the many studies that consider the impacts of plantation forestry on groundwater recharge, and others that explore the spatial heterogeneity of recharge in low rainfall regions, there is little marriage of the two subjects in forestry management guidelines and legislation. Here we carry out an in-depth analysis of the impact of reforestation on groundwater recharge in a low-rainfall (>700 mm annually), high-evapotranspiration paired catchment characterised by ephemeral streams. Water table fluctuation (WTF) estimates of modern recharge indicate that little groundwater recharge occurs along the topographic highs of the catchments (average 18 mm/yr); instead the steeper slopes in these areas direct runoff downslope to the lowland areas, where most recharge occurs (average 78 mm/yr). Recharge estimates using the chloride mass balance (CMB) method were corrected by replacing the rainfall input Cl^- value with that for streamflow, because most recharge occurs from infiltration of runoff through the stream bed and adjacent low gradient slopes. The calculated CMB recharge values (average 10 mm/yr) are lower than the WTF recharge values (average 47 mm/yr), because they are representative of groundwater that was mostly recharged prior to European land clearance (>200 years BP). The tree plantation has caused a progressive drawdown in groundwater levels due to tree water use; the decline is less in the upland areas.

The results of this study show that spatial variations in recharge are important considerations for locating tree plantations. To conserve water resources for downstream users in low rainfall, high evapotranspiration regions, tree planting should be avoided in the dominant zone of recharge, i.e. the topographically low areas and along the drainage lines, and should be concentrated on the upper slopes, although this may negatively impact the economic viability of the plantation.

1 Introduction

Tree plantations are known to have the potential to reduce groundwater recharge and surface water flows (e.g. Bell et al., 1990; Benyon, 2002; Bosch and Hewlett, 1982; Jobbagy and Jackson, 2004; Scanlon et al., 2007; van Dijk et al., 2007), particularly in low rainfall, high evapotranspiration regions where the high transpiration demands of the trees make them a significant user in the water balance (e.g. Benyon et al., 2006; Fekeima et al., 2010; Jackson et al., 2005; Schofield, 1992). This is often regarded as a negative aspect of tree plantations, but may be a positive outcome if the aim of a particular forestry project is to reduce groundwater levels, e.g. to decrease groundwater salinization (discussed further below). Groundwater recharge in low rainfall regions is also affected by a variety of other factors that cause substantial spatial variability, in particular topography, soil characteristics and geology (e.g. Delin et al., 2000; Scanlon et al., 2002; Schilling, 2009; Webb et al., 2008; Winter, 2001). However, the important conclusions made in the recharge studies have not been brought together with the results of tree plantation studies and directly applied to water resource management problems accompanying the establishment of tree plantations (Farley et al., 2005).

Since the earliest work on defining groundwater systems, recharge has been shown to be controlled predominantly by topography: the majority of groundwater recharge occurs at topographic highs, and discharge is mostly in topographic lows where the upward hydraulic gradient prevents recharge from occurring (Domenico and Schwartz, 1998; Schilling 2009). However, in arid and semi-arid regions, recharge following rainfall events often occurs predominantly in local depressions and along ephemeral streams (diverging from early conceptual models), due to the focussing of overland flow in these areas. Water tables under ephemeral streams are generally below the streambed (except during extended rainfall events), and therefore upwards groundwater gradients do not occur most of the time.

Infiltration beneath these areas may also be encouraged by the presence of preferential pathways, along which infiltrating water may more readily reach the water table (Delin et al., 2000; Scanlon et al, 2002; Schilling, 2009; Winter, 2001). In south-eastern Australia in particular, it has been observed that recharge can vary significantly within catchments due to multiple modes of recharge (Cartwright et al., 2007).

Vegetation can significantly impact groundwater recharge due to transpiration and by intercepting rainfall and overland flow (Scanlon et al., 2002; Winter, 2001); changing land-use can therefore affect recharge patterns. For example, land salinisation has occurred in large parts of south-eastern Australia due to the replacement of native forest by pasture and crops that use less water; this has led to increased recharge which raised water tables, causing saline groundwater to come to the land surface and discharge into surface water features (Allison et al., 1990; Bennetts et al., 2006, 2007). In contrast, afforestation of cleared farmland is likely to decrease recharge, due to the high rate of transpiration by the actively growing, closely planted trees, as well as the interception of overland flow and evaporation from the canopy (Benyon et al., 2006). In particular, the evergreen Eucalyptus tree plantations commonly planted in south-eastern Australia take up and transpire significantly more water than pasture, their canopy intercepts more rainfall and allows it to evaporate, and their roots reach greater depths than grasses, meaning they can extract water over a larger volume of the soil column (Bosch and Hewlett, 1982; Feikema et al., 2010; Hibbert, 1967). This recharge reduction is the reason why some studies have suggested using targeted tree plantations to reduce recharge in areas where there are high rates of saline groundwater discharge (e.g. Bennetts et al., 2007). Tree plantations also sequester carbon dioxide, prompting ongoing debate over the trade-off between increased water use by trees versus their increased carbon sequestration potential (Farley et al., 2005). As such, efforts over the past few decades in south-eastern Australia to reforest land that was cleared in the late 1800s by European settlers (Schofield, 1992) are now causing difficulties for land managers trying to define sustainable action plans for surface water and groundwater (Dalhaus et al., 2008; Jackson et al., 2005; Nicholson et al., 2006).

A whole catchment approach is key to managing groundwater recharge in the context of land use change (Cartwright et al., 2007). However, despite the evidence that recharge is often concentrated in topographic lows, groundwater management strategies in south-eastern Australia typically operate on the assumption that recharge occurs primarily in the upper parts of catchments, particularly along the ridgelines. Current regulations for tree plantations in

Australia focus on the percentage of a given catchment that can be forested, rather than what areas should be planted to maintain or intercept groundwater recharge, depending on the management application.

Here we present the findings from a paired catchment study in south-western Victoria, Australia, where one catchment is planted with a tree plantation, and the adjacent catchment is covered with pasture. This approach largely removes the variables of climate, topography, soil and geology, with the only major difference between the two catchments being vegetation cover. Previous paired catchment studies on the impact of tree plantations tended to focus on surface water responses to afforestation, while groundwater has been somewhat neglected (Brown et al., 2005). In this study conceptual models of groundwater flow (based on ^{14}C and tritium groundwater dating) and groundwater recharge estimates (based on the water table fluctuation and chloride mass balance methods) are used to assess the impact of a *Eucalyptus globulus* plantation on the hydrologic and hydrogeologic regime. This contextualisation is then used to discuss the best areas to site tree plantations within low rainfall catchments.

2 Background

This study is part of a multi-site, paired-catchment investigation into the impacts of land use and climate change on the quality and quantity of groundwater and surface water resources in western Victoria, Australia (Adelana et al., 2014; Camporese et al., 2013, 2014; Dean et al., 2014; Dresel et al., 2012).

2.1 Site description

The study area consists of a pair of small, adjacent catchments at Mirranatwa in south-western Victoria, one (referred to as the eucalypt catchment) covered predominantly in a recently planted (July 2008) *Eucalyptus globulus* (Blue Gum) plantation (0.8 km^2), the other (referred to as the pasture catchment) is mostly pasture for grazing sheep (0.4 km^2; Fig. 1).

2.1.1 Geology

Both catchments are underlain by the same weathered/fractured aquifer, the Devonian Dwyer Granite (390-395 Ma; Hergt et al., 2007; VandenBerg 2009). The upper ~20 m of the granite is well-weathered, porous and permeable saprolite; below this is relatively fresh, fractured bedrock. The fractured granite aquifer extends no deeper than 150 m, as below this depth the fracture conductivity is negligible due to the high lithostatic pressure (Boutt et al., 2010; Cook, 2003; Dept. Sustainability and Environment, 2012). The granite saprolite is generally thicker beneath the lower parts of the catchment than along the ridges, and is overlain by up to 7 m of alluvial/colluvial material along and adjacent to drainage lines. This alluvium/colluvium is clay-rich and impermeable in places, causing temporally variable artesian behaviour in some of the bores along the drainage lines in both catchments. The topography of the site (hills in the middle of a broad valley, Fig. 1) means both catchments are local ground water systems, and there are no regional groundwater inputs. There is 50 m of relief in the eucalypt catchment, and 30 m in the pasture catchment; both catchments comprise reasonably steep hills separated by a marked break in slope from the more or less flat topography along the drainage lines (Fig. 1).

2.1.2 Climate and Land use

The climate is Mediterranean, maritime/temperate (Cfb in the Köppen classification); the average annual rainfall since records began in 1901 for the area is 672 mm (± 125 σ), while pan evaporation is around 1350 mm annually, exceeding rainfall for the majority of the year, excepting the winter months of May to September (Dean et al., 2014). Runoff ratios for the pasture and eucalypt catchments are 3.0% and 3.3% respectively (based on the stream hydrograph records from February 2011 to February 2014), and both streams are ephemeral.

Vegetation of the area prior to European settlement was mostly open eucalypt woodland (Department of Sustainability and Environment, Victoria). Following European settlement there was extensive land clearance, and the catchments were entirely converted to pasture by 1869 (White et al., 2003). 76% of the northern catchment was subsequently converted to an *E. globulus* plantation in July 2008 (Fig. 1). Prior to the planting of the eucalypts, the eucalypt plantation catchment (Euc – Table 1) was used for grazing, and was virtually identical to the pasture grazing catchment (Pas – Table 1) immediately to the south. During the planting of the trees the eucalypt catchment was ripped to an average depth of 800 mm and mounded to an average height of 300 mm. The tree density is 1010 stems per ha (2.2 m between trees along a row, and 4.5 m between rows), and fertilizer was applied following ripping and mounding at 60 kg per ha (McEwens Contracting, pers. comm.). The tree rows run east-west across the slope in the main north-eastern part of the catchment, and north-south (~down the slope) to the west of H Addinsalls Road (Fig. 2).

2.1.3 Catchment instrumentation

The pasture catchment has 13 bores drilled to different depths, and the eucalypt catchment has 10 bores (the bores may be considered to be piezometers – they are screened towards the bottom of the casing over a discrete two metre interval; Table 1). Seven bores in the eucalypt catchment and two bores in the pasture catchment were drilled for this project in late 2009; the other bores were installed in the late 1980s in the pasture catchment, and the mid-1990s in the eucalypt catchment. A groundwater logger was installed in every bore in the eucalypt catchment in August 2009, measuring at a minimum four hour time interval, and eight bores in the pasture catchment have loggers measuring at the same frequency. There is a v-notch weir at the end of each catchment on both streams, with one bore immediately adjacent to the eucalypt catchment weir and two next to the pasture catchment weir (Fig. 1). The bores adjacent to the weirs have Campbell CS450-L pressure transducers (accuracy ±0.01 m) measuring water level and electrical conductivity (EC) at 30-minute intervals, while the other bores have Schlumberger Mini Diver loggers (accuracy ±0.025 m) measuring only water level. At the weirs the surface water level was measured using a standard V-notch construction, and electrical conductivity (EC) was recorded using a logger in the weir pool (Dresel et al., 2012). Prior to installation of groundwater loggers in the older bores, groundwater levels were generally measured manually bi-monthly.

There are two small dams in each catchment, ranging in size from 10 m^2 to 50 m^2; they are not large enough to significantly impact the hydrology of the site (Fig. 1). The roads at the site are single lane and unsealed, and although they are less permeable than the normal ground surface and therefore promote runoff, their very small area means that they have negligible impact on the site hydrology.

3 Methods

Groundwater levels, surface water flow and rainfall data were collected from August 2009 to February 2013 for this study, with some older long-term groundwater level data from manual measurements going back as far as 1986 available from the Victorian Department of Environment and Primary Industries archives. Groundwater and surface chemistry is available from sampling campaigns between August 2010 to August 2011 (Dean et al., 2014).

3.1 Rainfall and streamflow

Daily rainfall measurements were available from a Bureau of Meteorology station (089019) approximately two kilometres south of the study site; rainfall was also measured in the study catchments and showed an excellent correlation with the Bureau of Meteorology station. Due to significant gaps in the onsite data, the Bureau of Meteorology station data was used for consistency throughout the study period. To determine rainfall patterns, cumulative deviation from the monthly mean (CDM) values were calculated alongside daily values (section 4.1.1), whereby the difference between a given monthly rainfall total and the average for that month (calculated from the entire station's data record of 1901 to 2012), was cumulatively summed from one month to the next (modified from Craddock, 1979). The CDM values represent the longer term rainfall patterns, with a sustained negative trend for drought periods and positive values indicating wetter than usual periods, and match well with the longer term hydrographs (section 4.1).

Streamflow in both catchments is ephemeral, and was measured at 30-minute intervals at V-notch weirs at both catchment outlets and summed to annual totals, and a total for the complete study period, 2009-2013. To allow comparison between catchments, volumes were converted to depth equivalents (mm) by dividing by the respective catchment area. Streamflow is derived predominantly from direct runoff, as the proportion of groundwater input into the stream is small (discussed further below).

3.2 Grain size analysis

The grain size of the saprolite was used to estimate the average specific yield value for this aquifer over the whole study site, as the geology of the two catchments is very similar (see section 3.6.1). During drilling of five bores on the eucalypt catchment, samples of the regolith were taken at one metre intervals to a depth of 10 m, or until bedrock was encountered. Samples were sieved using a two-millimetre sieve and the material that passed through was then analysed using a Malvern Mastersizer 2000.

3.3 Groundwater composition

All 23 groundwater bores across the entire site were sampled once each over a period of a year, from August 2010 to August 2011. Seasonal variability in groundwater composition is considered negligible due to the age of the groundwater at the study site (mostly >200 years;

Table 1), and repeat sampling produced virtually identical field parameters (Dean et al., 2014). Subsamples for Cl^- were filtered with 0.45 µm filter paper and analysed using Ion Chromatography. Groundwater sampling, Cl^- analyses and calculations of volume-weighted, average rainfall Cl^- concentrations are described in more detail in Dean et al. (2014).

3.4 ^{14}C analysis and tritium analysis

Dating of the groundwater was carried out to determine the time period over which recharge has occurred. Groundwater samples from all the bores at the study site were ^{14}C dated and no corrections were applied, as there is no indication that the radiocarbon ages have been compromised by "dead" carbon in the regolith; standard error of groundwater ages is 25-100 years (Dean et al., 2014). In addition, seven bores in the eucalypt catchment and 11 bores in the pasture catchment (including the shallowest and deepest bores and a range in between), were analysed for tritium (standard error in these measurements was 0.04-0.13 tritium units (TU); Dean et al., 2014; Table 1). The methodologies for both are described in more detail in Dean et al. (2014).

3.5 Radon (^{222}Rn)

Radon surveys were carried out on groundwater and surface water samples in both catchments to ascertain whether there is a significant contribution of groundwater to surface water flow. The ^{222}Rn content of surface water and groundwater was measured using the gas-extraction for H_2O accessory of the Durridge RAD-7 radon detector. The RAD-7 is an alpha particle detector that measures the decay of the radon daughters, ^{214}Po and ^{218}Po. Samples from weirs, bores and dams (disconnected surface water bodies; Fig. 1) were collected in 250 ml vials and aerated for five minutes to degas the radon into the air circulation within the instrument, which takes four measurements (five minutes each), and then gives the mean ^{222}Rn concentration in Bq/L; the average standard error for measurements using this instrument is 10% (Durridge Co. Inc., 2010).

3.6 Groundwater recharge

To ensure robust estimates of groundwater recharge, two different, well established methods were used, namely the water table fluctuation method and chloride mass balance method.

While both methods are in widespread use, they have known deficiencies that are discussed below.

3.6.1 Water table fluctuations

The water table fluctuation (WTF) method for measuring groundwater recharge was first applied in the 1920s (Healy and Cook, 2002; Meinzer, 1923) and has since been refined (e.g. Jie et al., 2011; Scanlon et al., 2005; Sophocleus, 1991). The principle of this method is that rises in the groundwater hydrograph of an unconfined aquifer provide an estimate of recharge to the water table, calculated from:

$$R = S_y \frac{\Delta h}{\Delta t} \qquad (1)$$

where recharge (R) is the product of the specific yield of the aquifer (S_y) and the change in hydrograph height (Δh) over a given time interval (Δt). This method assumes that recharge occurs vertically from piston flow and that water discharges continuously from the aquifer, causing a drop in the water table when recharge is not occurring. Therefore the change in hydrograph height from which recharge is calculated is the sum of the rise in the hydrograph, together with the decline in the hydrograph that would have occurred in the absence of recharge over the same time period (Healy and Cook, 2002; Jie et al., 2011). Several techniques have been developed to estimate the hydrograph decline: the graphical approach – where the exponential decay curve of the hydrograph is manually extended to coincide with the peak of the next recharge event (Delin et al., 2007), the master recession curve approach – where regression functions are assigned to simulate the potential hydrograph decline for each data time-step (Heppner et al., 2007), and the RISE approach – where the assumption is made that in the absence of recharge, no decline in the water table occurs (Jie et al., 2011; Rutledge, 1998).

It proved difficult to apply the graphical and master recession curve methods in the present study because these methods focus on the section of the hydrograph recession limb which decays exponentially, whereas the recession limbs in the Mirranatwa hydrographs often had significant sections which were steep and straight (Fig. 3); this can lead to the underestimation of actual groundwater recharge, as has been highlighted elsewhere (Cuthbert, 2014). In addition, because the streams in both study catchments are ephemeral, groundwater discharge as baseflow occurs only occasionally; the majority of groundwater discharge occurs at the bottom of the catchments and downstream of the catchment boundaries. This intermittent baseflow means that the recession curve in the hydrographs following a recharge event may not be exponential (as observed in the hydrographs). Because the assumption of an exponential recession curve is implicit in the graphical and master recession curve WTF methods, the RISE approach was adopted, i.e. the decay curve of the hydrograph was ignored. Applying the RISE approach means that the values calculated in this study potentially underestimate actual recharge, but when compared with the graphical approach carried out for sections of the hydrographs where exponential recession curves were evident, gave very similar values.

Raw bore hydrograph data collected using data loggers at the site contain small fluctuations due to the impact of barometric pressure on the water column in the bore (Fig. 3a; Rasmussen and Crawford, 1997). The fluctuations in the water level and the barometric pressure are normally inversely correlated (Butler et al., 2011), and can be readily corrected (Rasmussen and Crawford, 1997; Toll and Rasmussen, 2007). At the study site these fluctuations are clearly positively correlated with barometric fluctuations (Fig. 3a), and as a result normal barometric compensation techniques could not be applied. Two types of groundwater level sensors were used, Schlumberger Mini Diver loggers (accuracy ±0.025 m) and Campbell CS450-L pressure transducers (accuracy ±0.01 m); the Campbell sensors are vented and therefore technically do not need compensating for barometric pressure changes, while the Schlumberger sensors require barometric compensation and barometric loggers were installed in the middle of both catchments to collect barometric data for this purpose. The barometric effect shown in Fig. 3a is consistent across all the Schlumberger sensors in both catchments, regardless of landscape position. Figure 3a is based on Figure 1 from Butler et al. (2011), and the data from this study was prepared in the same manner, so the positive correlation is not an artefact of data processing error. Barometric forcing was evident in the Campbell sensor data also, despite their being vented, so this data was treated in the same way as the Schlumberger data (see below).

A 15-day moving average was used to remove the barometric fluctuations but retain the overall response to rainfall (Fig. 3b). The 15-day timestep is a narrow enough time period to incorporate recharge events and reflect the general trend of the hydrograph, but removes the small barometrically forced fluctuations that bear no relationship to rainfall (Fig. 3). Recharge was then calculated using equation [1], where Δh was taken as the sum of the increases in groundwater level over the timestep, and then summed for the entire length of the record.

When there was a drop in groundwater level from one timestep to the next, this was taken as zero recharge. The measurement uncertainty of the loggers (± 0.025 m) was used as the threshold for recognition of recharge for each 15-day timestep. The RISE method was also used to calculate recharge for the longer-term hydrographs (generally bi-monthly measurements taken prior to logger installation).

A specific yield value of 0.095 ± 0.014 was calculated for the unconfined saprolite aquifer from the average grain size (clay to coarse sand; Table 2) of all the bore samples analysed (see Section 3.2), using the general relationship between specific yield and grain size in Healy and Cook (2002, Tables 1 and 2). The estimation of specific yield is a potential source of considerable error in recharge calculations as it can vary spatially, although it can be assumed to be independent of time (Healy and Cook, 2002). The specific yield value calculated here is comparable to other values from weathered granites in the region (0.043 – Hekmeijer and Hocking, 2001; 0.075 – Edwards, 2006). When calculating recharge for the study site, this specific yield was applied to bores that are screened within the saprolite, and is assumed to be representative for the whole site because of the relatively uniform nature of the soils (Table 2).

3.6.2 Chloride mass balance

The chloride (Cl^-) mass balance (CMB) method for calculating recharge is based on the relationship between Cl^- in groundwater and in precipitation, assuming that all Cl^- in the groundwater is derived from rainfall and remains in solution within the groundwater system, that direct recharge (R, in mm) occurs via piston flow, and that runoff is negligible:

$$R = P\frac{C_p}{C_{gw}} \qquad (2)$$

where P is the amount of rainfall (mm), C_p is the concentration of Cl^- in P, and C_{gw} is the concentration of Cl^- in groundwater (Allison and Hughes, 1978; Scanlon et al., 2002). R was calculated at all bores using the groundwater Cl^- content (Table 1), and rainfall Cl^- content was the median value from three different sampling periods at nearby sites (Fig. 1): 1954-1955 at Cavendish (Hutton and Leslie, 1958), 2003-2004 at Hamilton (Bormann, 2004), and 2007-2010 at Horsham (Nation, 2009); all Cl^- values were volume weighted based on rainfall during the sampling periods in these studies. These three sampling periods include a wet period (1954-1955) and two dry periods (2003-2004 and 2007-2009). The median rainfall Cl^- from all of these studies is 4.3 ± 0.9 mg/L, and the annual rainfall is 672 ± 125 mm (1σ); the uncertainties associated with each value were used to estimate the overall uncertainty in the recharge values calculated. *R* is strongly governed by C_p in this equation, so it is important to take into account the variability in C_p.

4 Results and discussion

4.1 Groundwater recharge estimates

Recharge estimates calculated using both the WTF and CMB methods range from 0.8 ± 0.3 to 161 ± 24 mm/yr (Table 3), a very wide range that matches recharge calculations from similar climatic areas in Australia (5 – 250 mm/yr; Allison and Hughes, 1978; Cook et al., 1989), and elsewhere from low rainfall regions around the world (0.2 – 35 mm/yr; Scanlon et al., 2006).

4.1.1 Water table fluctuation method

The groundwater hydrographs vary significantly across the study site (Fig. 4), indicating substantial variation in groundwater recharge. Because hydrographs from the upper parts of the catchment show a limited response to rainfall patterns, both in the detailed groundwater logger data (Fig. 4) and the longer term monitoring data for the older bores (Fig. 5), recharge values calculated using the WTF method are relatively low for these areas in both catchments (average 18 mm/yr; 3% of rainfall).

In contrast, bores on or close to drainage lines show a much greater sensitivity to sustained rainfall and streamflow events (e.g. for bore Pas96, rises in the hydrograph directly correspond to flow in the ephemeral stream channel; Fig. 6). As a result, recharge values calculated from logger data and longer-term hydrographs using the WTF method are relatively high for low-lying areas in both catchments (average 78 mm/yr; 12% of rainfall; Fig. 4; Table 3). These recharge trends have been consistent over the past 20-30 years (Fig. 5).

The greater recharge in the lower-lying areas is predominantly because the steeper slopes in the upland areas direct runoff downslope to the lowland areas, which are consequently saturated for longer with a greater volume of runoff. In addition, runoff velocities across the lower areas decrease due to the reduction in slope, allowing more infiltration into the soil. Runoff from the upland areas is aided by the low permeability, silty soils (Table 2), and infiltration in the lower lying areas, particularly through the stream bed, is increased by the greater depth of weathering (9 m depth to bedrock in the pasture catchment and 30 m in the eucalypt catchment, except at the very bottom of this catchment).

Two of the lowland bores (Euc84 and Euc85) show very similar recharge patterns to upland bores (e.g. Euc83), i.e. little recharge, due to the presence of a localised confining layer (both bores frequently go artesian; Fig. 4).

Two of the upper slope bores show high recharge (Pas74 and Pas 78), due to preferential recharge down fractures in the granite (Section 4.2; Fig. 5).

4.1.2 Chloride mass balance method

Recharge values calculated from the CMB method (equation 2) are much lower than the WTF method values, often by an order of magnitude or more (Table 3), e.g. Pas96 has recharge values of 1.1 ± 0.4 mm/yr (CMB) and 161 ± 24 mm/yr (WTF), and Pas82 has a CMB value of 8.8 ± 3.3 mm/yr and a WTF value of 26 ± 4 mm/yr. Furthermore, the bore hydrographs used to calculate the WTF recharge values indicate that there is much more recharge occurring in the lowland areas than is indicated by the CMB values.

The most likely explanation for the mismatch between the CMB and WTF results is that the input Cl^- value used in the CMB method should be for runoff/streamflow rather than rainfall, because most recharge occurs from infiltration of surface flow through the stream bed and across the low gradient slopes adjacent to the streams, as previously discussed.

To account for this difference, the CMB values were recalculated using the volume and Cl^- content of streamflow (assumed to be the same as runoff here) in place of rainfall in equation 2:

$$R = RO\frac{C_{ro}}{C_{gw}} \qquad (3)$$

where RO (mm) is the estimated amount of runoff that would reach a given bore, and C_{ro} is the estimated Cl^- concentration of the runoff (volume weighted).

The volume of runoff at a particular bore (RO) is calculated using streamflow as a proxy for runoff, by dividing the average streamflow per year by the amount of the catchment that could theoretically provide runoff to the bore location (i.e. a bore in the middle of the catchment is only going to receive approximately half the runoff that could potentially recharge a bore at the bottom of the catchment). The Cl^- concentration of the runoff (C_{ro}) is calculated from the average EC measured at each weir (May 2010 to February 2013), converted to Cl^- using the

EC:Cl^- ratio for the study site dataset (0.39 and 0.37 for the pasture and eucalypt catchments respectively). Equation 3 was only applied to bores in the lowland parts of the landscape where runoff is likely to recharge the groundwater. Because of the highly variable nature of the streamflow Cl^-, the potential variation in recharge values calculated from equation 3 is large, and this is seen in the error values (1σ – Table 3).

The recalculated recharge values generated from equation 3 are much closer to the WTF recharge values, but are still generally a factor of five to 15 lower. This may reflect the fact that the groundwater across the study site is mostly >200 years old, indicating that the CMB values are generally representative of recharge rates under native vegetation prior to land clearance during European settlement in the late 1800s, whereas the WTF values represent recent recharge (August 2009 to February 2013). The older, pre-European settlement vegetation caused lower recharge, as these trees transpire much more water from groundwater and the soil zone than modern pasture. This disparity between modern and pre-European recharge rates has been observed elsewhere in south-eastern Australia (e.g. Allison et al., 1990; Bennetts et al., 2006, 2007; Cartwright et al., 2007).

The CMB method estimates of recharge do not vary significantly between the two catchments, showing that both catchments behaved in a similar fashion before measurements began, prior to the establishment of the plantation. This corrects for the lack of a calibration period prior to the change in land use, a potential source of considerable error (Brown et al., 2005).

4.2 Topographic controls on recharge

Recharge estimates using the WTF method (Table 3) show that within the local groundwater systems of the study catchments, variations in recharge predominantly reflect differences in topography. Dominant areas of recharge are not along the topographic highs of the catchments, as in the traditional conceptual model of recharge, but are instead analogous with more arid regions, where most recharge occurs in topographic depressions (Scanlon et al., 2002).

Recharge rates increase as surface elevation decreases (Fig. 7). The steeper slopes of the upland areas promote runoff rather than infiltration, aided by low permeability, silty soils (Table 2). Overland flow is focused into topographic lows and along drainage lines. Here the granite is most weathered, as indicated by the greater depth to bedrock here (9 m in the pasture catchment, and 30 m in the eucalypt catchment except at the very bottom of this catchment), encouraging recharge to occur, particularly through the stream bed.

4.3 Influence of fractures on groundwater recharge

The ^{14}C data (Table 1) shows that most of the groundwater at the study site is older than the tree plantation, but the groundwater in some bores (Pas74, Pas 80, Pas81, Pas 82, Pas 96, Euc91, Euc93 and Euc97) also contains measurable tritium, indicating a component of younger groundwater (<50 years old). Recharge in fractured rock aquifers like granite is controlled to some extent by the fracture network (Cook, 2003), which forms multiple recharge pathways. In the study area this has allowed mixing of young groundwater (containing tritium) with much older groundwater (as shown by the ^{14}C dates; Table 1). The hydrograph for the upslope bore Pas74 (Fig. 4) shows high recharge following rainfall events (in contrast to most of the other upslope bores), most likely because it is located on a fracture in the granite that allows rapid recharge, as shown by the dilute groundwater with low Cl^- concentrations (Dean et al., 2014) and the presence of significant amounts of tritium (Table 1).

This dual porosity (matrix and fracture flow) influence on recharge has been observed elsewhere in south-eastern Australia where there was disparity between the residence times of groundwater samples (Cartwright et al., 2007). Nevertheless, the dominant recharge control across both catchments is topography rather than fracture heterogeneity, as shown by the relatively flat hydrographs for most of the upland bores, and strongly oscillating hydrographs in the lowland bores (Fig. 4).

4.4 The interplay between ephemeral stream flow and groundwater recharge and discharge

The streams at the study site are ephemeral, flowing on average only 40% of the time at the catchment outlets. When they are dry, recharge can occur readily along and near the streambeds as upwards groundwater gradients are not present, because the water table is below the base of the stream. As a result, bores in the lower parts of the catchments (e.g. Pas96 near the outlet of the pasture catchment; Fig. 6) show a clear, sometimes instantaneous link between recharge and runoff.

Following extended periods of wet weather, the ephemeral stream at the bottom of the eucalypt catchment is fed by groundwater discharge, as shown by the significant levels of ^{222}Rn measured at the weir (11 Bq/L; Fig. 8); however, the elevated ^{222}Rn measured in the Eucalypt stream could just be due to the close proximity of the granite bedrock to the surface at the bottom of this catchment. This is suggested by the high ^{222}Rn values in Pas95 and Euc92, both screened in granite bedrock, compared to the lower ^{222}Rn values in Euc90 and Pas96, which are screened in the weathered granite saprolite (Fig. 8). Regardless, the shallow granite bedrock at the outlet of the Eucalypt catchment (less than two metres below the surface; Fig. 9), forces groundwater towards the surface here. In contrast, the bedrock at the bottom of the pasture catchment is nine metres deep, so the water table lies more consistently below the base of the stream and there is less groundwater discharge; as a result, the pasture catchment has fewer low flows than the eucalypt catchment (Fig. 9) and lower ^{222}Rn levels (1 Bq/L at the weir; Fig. 8).

In both catchments, during periods of little or no rainfall, the water table lies below the surface, so recharge can occur along the length of the channel. When it begins to rain and the system wets up, the water table rises at the downstream end of the catchment and groundwater begins to discharge here (this occurs more frequently and to a greater extent in the eucalypt catchment). Continued rain raises the water table so it connects to the stream further upstream, increasing the length of the stream that receives groundwater discharge (Fig. 9; Adelana et al., 2014). When rainfall ceases, the water table drops and progressively disconnects from the stream, starting upstream, until it is completely disconnected throughout the catchment. This means that during smaller rainfall events, when the water table remains below the land surface and does not connect to the stream, recharge occurs along the length of the stream. During larger rainfall events, as the water table comes to the surface along the stream channel, the area of potential recharge decreases.

The groundwater hydrographs indicate that during the study period, recharge occurred readily in the lowland areas of both catchments, particularly when there was enough rainfall to generate consistent flow in the streams, while much less recharge is evident on the upper slopes. There is relatively little groundwater discharge along the streams, as shown by the ^{222}Rn data (Fig. 8), and groundwater within the catchments is lost predominantly through evapotranspiration, particularly when the water table is within two metres of the ground surface (as commonly occurs in south-eastern Australia, e.g. Bennetts et al., 2006, 2007); a small amount flows out at the bottom of the catchment.

4.5 Vegetation controls on recharge

The bore hydrographs in the eucalypt catchment show a clear overall declining trend of up to 3 m during the study period, evident even in artesian bores (Euc84 and Euc85), and regardless of landscape position (Fig. 4). This decline is not evident in hydrographs from the pasture catchment (Fig. 4), where the water table has increased by 0.5-1 m during the whole study period as a result of consecutive wet summers of 2010/11 and 2011/12 (Fig. 7). The tree plantation was a little over one year old when the main measurements of this study began, and as the age of the plantation increased, a steeper decline in groundwater depth was observed (Fig. 4).

The water level decrease in the eucalypt catchment, with no corresponding drop in the pasture catchment, is attributed to greater water use by the trees, as has been demonstrated elsewhere (e.g. Adelana et al., 2014; Bosch and Hewlett, 1982). The water table decline is less in the upland areas (Fig. 9), probably because recharge rates here are lower, so that the decrease in recharge due to tree water use has had relatively little impact. Furthermore, in the upland areas the water table is too deep for the vegetation to access the groundwater directly; Benyon et al. (2006), in a study in the same region of south-eastern Australia, found that deep-rooted eucalypts can only access groundwater up to a depth of six to eight metres. In the lowland areas the trees are able to reach the groundwater (Fig. 2), and this, combined with the interception of potential recharge in the soil zone by the growing plantation, causes the observed decline in groundwater level (Fig. 4). Although tree roots can provide preferential pathways for infiltration of rainfall to the water table (Burgess et al., 2001), any effect of this is masked by the overall impact of eucalypt water use. The increasing rate of decline in groundwater depth over time can be attributed to the greater water usage by the trees as they grow (Fig. 4).

The narrow areas immediately adjacent to the drainage lines in the eucalypt catchment are covered in grass and therefore there is less direct interception of potential recharge, but in fact these areas show the biggest decline in groundwater level (Fig. 7). The highest rates of recharge occur along the drainage lines and the adjacent trees will therefore have a substantial impact there, in particular because they are directly accessing the groundwater.

With groundwater levels in the eucalypt catchment still in decline at the end of the study period, five years since the establishment of the eucalypt plantation, there is no sign that the system is reaching equilibrium under the new land use. Brown et al. (2005) indicate that equilibrium would not be expected until more than five years after the land-use change occurred.

4.6 Management of tree plantations and recharge

Afforestation of farmland was widespread in south-eastern South Australia and south-western Victoria (known as the Green Triangle) from the 1980s through to the 2000s, with the plantation area expanding by 5-14% to 30,000 ha in Victoria alone (Adelana et al., in 2014; Benyon et al., 2006; Ierodiaconou et al., 2005). However, the subsequent development of tree plantations in the region has been hindered by a poor timber market (HVP Plantations, pers. comm.) and concerns that plantations use more groundwater and surface water than other land uses like farming. As a result, tree plantations in the state of South Australia must now be licenced as groundwater users (Govt. of South Australia, 2009), while it is hoped that the potential reduction in water availability resulting from reforestation will be offset by the beneficial gains of the carbon sequestration within the new trees (Schrobback et al., 2011).

A reduction of groundwater recharge by plantations, as documented in this study, lowers the water table and can reduce stream flow. If this is the object of the reforestation, for example to reduce saline groundwater discharge, then this land-use change may well serve its purpose (Bennetts et al., 2007). However, the recent drought in south-eastern Australia (1997-2010) has exacerbated concerns that trees may be a significant user of local and regional water resources, reducing groundwater recharge, discharge and surface water availability (Jackson et al., 2005).

In order to reduce the impact on water availability, current regulation of tree plantations in south-eastern Australia focuses on the percentage of a catchment that may be planted. However, the present study shows that the location of the plantation within the catchment is significant also, with a smaller water table decline seen in the upland areas of the eucalypt catchment. Therefore to reduce the impact of plantations on groundwater recharge, tree planting should be avoided in the dominant zone of recharge, i.e. the topographically low areas and along the drainage lines, and should be concentrated on the upper slopes, where the water tables are deeper and the trees are less likely to access the groundwater and transpire it directly. At present, tree plantations in Victoria cannot be planted within 20 m of drainage lines, to avoid erosion of creek banks when the trees are removed (Dept. of Environment and Primary Industries, Victoria); we suggest that this currently restricted area along the drainage lines be expanded to include as much of the low topography parts of the site as practicable.

The expansion of the drainage line exclusion zone in tree plantations will have an added benefit in many regions of south-eastern Australia where the groundwater is saline. This is because the parts of the catchments where the saline groundwater is within a few metres of the land surface (generally the lowland areas) can have a negative effect on tree health; at the study site, the trees closer to the drainage lines are shorter and thinner than those upslope.

However, excluding tree planting from low elevation areas reduces the number of trees that can be planted within a catchment, and also means that trees are not planted in areas where (good quality) groundwater is shallowest and can be most readily accessed for tree growth. As the primary purpose of many tree plantations is the production of wood and pulp products for economic gain, this restriction will slow economic returns. To overcome this, consideration could be given to planting lower water use trees that can better cope with the upslope areas where water supplies for tree growth may be limited.

This management strategy of balancing economic and hydrologic perspectives when locating tree plantations within catchments will be applicable to other low-rainfall, high-evaporation regions, and should be considered for tree plantations in similar climatic areas worldwide.

5 Conclusions

While the importance of topography and ephemeral streams to focused recharge in low rainfall regions around the world has been known for some time, the implications of this aspect of the groundwater resource literature have not been incorporated into plantation management guidelines and legislation. In this study, it is shown that the majority of modern recharge at the study site, calculated from the water table fluctuation method, occurs in the lower parts of both study catchments (12% of rainfall versus 3% in the upland areas). Overland flow is focused into topographic lows and along drainage lines where greater infiltration can occur. Recharge calculations using a corrected chloride mass balance method gave lower values than modern recharge estimates because the groundwater across the study site is mostly >200 years old, representing recharge under native eucalypt forest prior to

European land clearance. Relatively little groundwater discharges into the streams or flows out at the bottom of the catchment; groundwater within the catchments is lost predominantly through evapotranspiration. Overall the tree plantation in this study caused a drawdown in groundwater levels, increasing over time as the trees aged, compared to a slight rise in groundwater levels in the pasture catchment.

The results of this study lead to the conclusion that both the hydrogeological and economic frameworks for commercial forestry need to be considered. If conserving groundwater recharge is a primary objective, tree planting should be avoided in the dominant zone of recharge, and concentrated on the upper slopes, where recharge is low enough that any further reduction will have minimal impact. We suggest expanding present regulations for tree plantations which specify that trees cannot be planted within a certain distance of drainage lines, including as much of the low topography parts of the site as practicable. Consideration should be given to the potential negative impact on the financial viability of a tree plantation. This management strategy is applicable to low-rainfall, high-evaporation regions worldwide.

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Figure captions:

Figure 1: Left - location of the study site in south-western Victoria, Australia. Right - location of the streams, weirs and bores and their reference numbers. “L” denotes the presence of a water level logger in a bore.

Figure 2: Orientation of the tree rows in the Eucalypt plantation and the area where tree roots may be able to reach groundwater up to depths of six and eight metres below the surface.

Figure 3: (a) barometric pressure (in equivalent cm of H_2O), groundwater logger data, rainfall and the 15 day moving average used for the water table fluctuation method estimates of groundwater recharge. The black dots represent the average groundwater level for the preceding 15 days. (b) Full record for the bore used in (a) – Euc90 – showing the complete removal of the large amount of barometric noise, but keeping the overall trend of the 15 day period.

Figure 4: bore hydrographs, rainfall and recharge estimates (in mm/yr from Table 3), for the water table fluctuation and chloride mass balance methods. Hydrographs are sorted by landscape position – lowland or upland.

Figure 5: Long-term hydrographs for bores with available data and cumulative deviation from mean monthly rainfall to show the relationship between groundwater levels and long term rainfall patterns.

Figure 6: pasture stream hydrographs (Dwyer’s Creek) and bores hydrographs from the bottom of the catchment (Pas96) and midway up the catchment (Pas75).

Figure 7: Cross section from bore Euc91 across both catchments to bore Pas74 showing recharge rates based on both methods used in this study, and the water table change over the course of the study period (see Fig. 1 for bore locations).

Figure 8: ^{222}Rn concentrations in the streams, measured at the weirs of both sites, and nearby bores. Surface water from further up the catchments is represented by water from dams located upslope in both catchments. The relatively high levels in the groundwater are a result of the decay of uranium present in the allanite and zircon of the granite.

Figure 9: Long section from bores Euc97 to Euc92 showing the effect of the shallow granite on the water table under different flow conditions shown in the flow duration curve below; (1) where low flows in the eucalypt catchment stream are sustained for longer due to some

groundwater discharge compared to virtually no groundwater discharge in the pasture catchment, (2) where the water table is at the surface and runoff is transported more quickly out of the eucalypt than in the pasture catchment, and (3) where there are some rare, very high flows, much higher than observed in the pasture catchment.

Table 1: Groundwater characteristics and bore construction.

Bore ID	Earliest data from bore	Screen depth (m below surface)	Surface elevation (m AHD)	Radiocarbon age (yr BP)	1σ - error	Activity of ^3H (TU)	1σ - error	Logger	Groundwater Cl^- (mg/L)
Pasture Catchment									
Pas72 - Low	31/08/1986	9.4-11.6	259.55	1665	± 30	BD		N	3292
Pas73 - Low	31/08/1986	4-6.1	259.54	2055	± 30			N	3110
Pas75 - Low	31/08/1986	12-13.6	263.93	935	± 35			Y	2231
Pas76 - Low	31/08/1986	2.2-4.2	263.98	575	± 30	BD		Y	1595
Pas95 - Low	26/08/2009	22.8-24.8	254.13	3540	± 30	BD		Y (weir)	2732
Pas96 - Low	26/08/2009	5-7.55	254.18	345	± 25	1.12	±0.09	Y (weir)	2553
Pas74 - Up	31/08/1986	6.2-8.5	268.62	790	± 30	0.44	±0.04	Y	306
Pas77 - Up	31/08/1986	17.7-19.7*	271.11	Modern		2.84	±0.13	N	28
Pas78 - Up	31/08/1986	17.3-19.4	277.45	650	± 90	BD		Y	1185
Pas79 - Up	31/08/1986	23.65-25.65*	283.23	Modern		2.55	±0.12	N	38
Pas80 - Up	31/08/1986	23.3-24.4	288.23	115	± 30	1.24	±0.08	Y	2290
Pas81 - Up	31/08/1986	7.1-8.9	272.12	690	± 100	0.79	±0.08	N	668
Pas82 - Up	31/08/1986	23.2-24.8	283.54	430	± 30	0.60	±0.05	Y	329
Eucalypt catchment									
Euc84 - Low	12/11/1996	5.6-7.5	268.67	785	± 30			Y	3909
Euc85 - Low	12/11/1996	7.9-10	268.66	**		BD		Y	3537
Euc89 - Low	30/10/2009	26-28	261.80	7330	± 50			Y	2833
Euc90 - Low	30/10/2009	13-15	261.93	6980	± 45			Y	2788
Euc92 - Low	30/10/2009	26.2-29.2	255.43	20770	± 90	BD		Y (weir)	1490
Euc93 - Low	2/03/2010	11-14	263.31	725	± 30	0.73	±0.06	Y	1357
Euc83 - Up	12/11/1996	14.8-16.7	274.21	685	± 30	BD		Y	2064
Euc91 - Up	30/10/2009	33.9-35.9	280.02	415	± 30	0.39	±0.04	Y	1114
Euc94 - Up	30/10/2009	28-30	286.05	2060	± 30	BD		Y	2891
Euc97 - Up	30/10/2009	43.1-45.1; 57.6-59.6	291.74	5655	± 35	0.30	±0.04	Y	3494

BD denotes below detectable; * assumed screen depths; ** CO_2 concentration too low for analysis.

Table 2: Median grain size compositions for sampled profiles used to estimate a range of values for *Sy* in equation 1.

Bore ID	Clay (%)	Silt (%)	Fine sand (%)	Coarse sand (%)
Euc89 – Low	3	39	38	19
Euc91 – Low	3	39	40	18
Euc93 – Low	3	36	43	18
Euc94 – Up	3	35	44	18
Euc97 – Up	3	34	43	20

Table 3: Recharge (R) values using different methods for all the bores across both catchments.

Bore ID	R (mm/yr) – groundwater Cl^-	R (mm/yr) – groundwater Cl- with stream input correction	R (mm/yr) – water table fluctuation method	R (mm/yr) – long-term hydrograph water table fluctuation method
Pasture catchment – lowland landscape position				
Pas72 – Low*	0.9 ± 0.3	6.8 ± 4.6	[L]	[D]
Pas73 – Low*	0.9 ± 0.3	7.2 ± 4.8	[L]	[D]
Pas75 – Low	1.3 ± 0.5	3.9 ± 2.6	58 ± 9	38 ± 6
Pas76 – Low	1.8 ± 0.7	5.5 ± 3.7	77 ± 11	[D]
Pas95 – Low*	1.1 ± 0.4	24 ± 16	[C]	[D]
Pas96 – Low	1.1 ± 0.4	26 ± 17	161 ± 24	[D]
Pasture catchment – upland landscape position				
Pas78 – Up	2.5 ± 0.9	[C]	36 ± 5	[D]

Pas80 – Up	1.0 ± 0.4	[C]	12 ± 2	30 ± 5
Pas82 – Up	8.8 ± 3.3	[C]	26 ± 4	28 ± 4
Pasture catchment – possible fracture flow				
Pas74 – Up	9.4 ± 3.5	[C]	65 ± 10	56 ± 8
Pas77 – Up	102 ± 38	[C]	[L]	[D]
Pas79 – Up	76 ± 29	[C]	[L]	[D]
Pas81 – Up	4.3 ± 1.6	[C]	[L]	[D]
Eucalypt catchment – lowland landscape position				
Euc84 – Low*	0.7 ± 0.3	1.7 ± 1.3	[C]	[C]
Euc85 – Low*	0.8 ± 0.3	1.9 ± 1.4	[C]	[C]
Euc89 – Low	1.0 ± 0.4	5.7 ± 4.3	59 ± 9	[D]
Euc90 – Low	1.0 ± 0.4	5.8 ± 4.4	74 ± 11	[D]
Euc93 – Low	2.1 ± 0.8	8.0 ± 6.1	40 ± 6	[D]
Eucalypt catchment – upland landscape position				
Euc83 – Up	1.4 ± 0.5	[C]	10 ± 2	19 ± 3
Euc91 – Up	2.6 ± 1.0	[C]	17 ± 3	[D]
Euc94 – Up	1.0 ± 0.4	[C]	1.7 ± 0.2	[D]
Euc97 – Up	0.8 ± 0.3	[C]	26 ± 4	[D]
Eucalypt catchment – possible fracture flow				
Euc92 – Low*	1.9 ± 0.7	[C]	[C]	[D]

* denotes confined bores; [L] no logger present; [D] no data; [C] indicates that this calculation was not done for that bore as it did not meet the required conditions (see sections 3.6.1 and 3.6.2).