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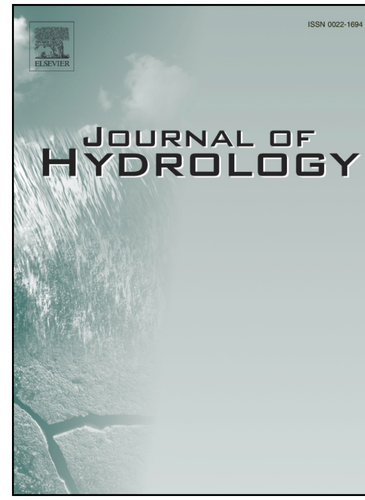

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PII: S0022-1694(14)00649-0
DOI: http://dx.doi.org/10.1016/j.jhydrol.2014.08.039
Reference: HYDROL 19836

To appear in: *Journal of Hydrology*

Received Date: 14 June 2014
Revised Date: 15 August 2014
Accepted Date: 19 August 2014

Please cite this article as: Fan, J., Oestergaard, K.T., Guyot, A., Lockington, D.A., Estimating groundwater recharge and evapotranspiration from water table fluctuations under three vegetation covers in a coastal sandy aquifer of subtropical Australia, *Journal of Hydrology* (2014), doi: http://dx.doi.org/10.1016/j.jhydrol.2014.08.039

Estimating groundwater recharge and evapotranspiration from water table fluctuations under three vegetation covers in a coastal sandy aquifer of subtropical Australia

Junliang Fan [1,2,]* , Kasper T. Oestergaard [1], Adrien Guyot [1,2], David A. Lockington [1,2]

[1] *School of Civil Engineering, The University of Queensland, St Lucia 4072, Australia*

[2] *National Center for Groundwater Research and Training, Flinders University, Bedford Park 5042, Australia*

* **Corresponding author**.

Junliang Fan

School of Civil Engineering,

The University of Queensland,

St Lucia, QLD 4072, Australia

E-mail: j.fan@uq.edu.au

Abstract

To evaluate potential hydrological impacts of changes in vegetation over a shallow sandy aquifer in subtropical Australia, we estimated groundwater recharge and discharge by evapotranspiration (ET_g) under three vegetation covers. Estimates were obtained over two years (November 2011–October 2013) using the water table fluctuation method and the White method, respectively. Depth-dependent specific yields were determined for estimation of recharge and ET_g. Our results show that the average annual gross recharge was largest at the sparse grassland (~52% of net rainfall), followed by the exotic pine plantation (~39% of net rainfall) and then the native banksia woodland (~27% of net rainfall). Lower recharge values at forested sites resulted from higher rainfall interception and reduced storage capacity of the vadose zone due to lower elevations when the water table approaches the soil surface. During 169 rain-free days when the White method was applied, pine trees extracted nearly twice as much groundwater through ET_g as the banksia, whereas no groundwater use by grasses was detected. Groundwater use is largely controlled by meteorological drivers but further mediated by depth to water table. The resulting annual net recharge (gross recharge minus ET_g) at the pine plantation was comparable to that of the banksia woodland but only half of the corresponding value at the grassland. Vegetation cover impacts potential groundwater recharge and discharge, but in these subtropical shallow water table environments estimates of potential recharge based on rainfall data need to take into account the often limited recharge capacity in the wet season.

Keywords

Pine plantation; banksia woodland; water table fluctuation method; White method; depth-dependent specific yield.

Introduction

Vegetation plays a significant role in the groundwater hydrological cycle due to its impact on groundwater recharge and transpirative discharge; conversely, groundwater hydrology impacts sensitive vegetation in shallow water table environments (e.g., wetlands or riparian areas). Vegetation affects groundwater recharge, and thus sustainable yields, indirectly by rainfall interception losses as well as extraction of infiltrating rainwater before it reaches the water table (Le Maitre et al., 1999).

The impact of changes in vegetation cover on groundwater hydrology has been investigated for a range of environments, mostly in (semi)arid or temperate areas with deep aquifer systems (e.g., Scanlon et al., 2005; Mao and Cherkauer, 2009; Brauman et al., 2012; Nosetto et al., 2012). Deep-rooted woody vegetation was generally found to reduce streamflow and groundwater recharge (Matheussen et al., 2000; Crosbie et al., 2010), compared to shallower-rooted grasses and crops, and they tend to tap groundwater with deeper rooting systems (Benyon et al., 2006; Pinto et al., 2013). For example, Scanlon et al. (2005) found that the conversion of natural shrublands with agricultural ecosystems in southwest US altered the water flow from discharge through *ET* (i.e., no recharge) to recharge (9–640 mm yr^{-1}). Benyon et al. (2006) reported that plantations of *Pinus radiata* D.Don and *Eucalyptus globulus* Labill. used groundwater at an average rate of 435 mm yr^{-1} (40% of total water use) in the Green Triangle of southeast Australia. However, while coastal systems are under pressure from human development as well as potential stresses due to climate change, there are few studies quantifying the hydrological effects of vegetation cover changes in coastal areas characterized by shallow aquifer systems with highly permeable sediments.

Like other coastal and island sand mass aquifers around the world, significant resources of high quality groundwater are located on Bribie Island for water supply to coastal communities and local wetland vegetation. Over the past three decades, exotic pine tree plantations have been developed on the island largely for timber production, particularly in the natural distribution areas of native vegetation (e.g., banksia woodland and grassland). The changes in vegetation cover can potentially affect the local hydrological processes, e.g., groundwater recharge and evapotranspiration (*ET*).

In shallow water table environments, groundwater recharge and groundwater use by vegetation via evapotranspiration (ET_g) can be estimated from analyses of water table fluctuations (e.g., Scanlon et al., 2005; Crosbie et al., 2005; Mould et al., 2010; Zhu et al., 2011; Yin et al., 2013; Fahle and Dietrich, 2014). For such analyses, quantification of the aquifer's specific yield (S_y) is considered the main source of uncertainty as its error is translated directly to final estimates (Scanlon et al., 2002; Loheide et al., 2005). Various methods (e.g., laboratory experiment, field study and numerical modelling) are available for determining specific yield, but they usually produce inconsistent values (Neuman, 1987; Crosbie et al., 2005). Specific yield is often considered constant in hydrological studies. However, researchers have recognized that it is dependent on water table depth and drainage time (Duke, 1972; Nachabe, 2002; Shah and Ross, 2009), particularly in a shallow water table environment due to a the truncation of the equilibrium soil moisture profile at the soil surface (Childs, 1960). Use of a constant specific yield can lead to the recharge and ET_g being significantly overestimated (Sophocleous, 1985; Loheide, 2005). Loheide et al. (2005) suggested the readily available specific yield can be used to obtain reasonable estimates of ET_g when the water table depths > 1 m, but the dependence of S_y on the water table depth needs to be considered for water table depths < 1 m. In spite of this, the depth-dependant specific yield has seldom been adopted for the estimation of recharge and ET_g in published studies (e.g., Crosbie et al., 2005; Carlson Mazur et al., 2013).

Here, we investigate shallow water table fluctuations in response to rainfall and ET_g under three vegetation covers to gain a better understanding of the hydraulic relationship between vegetation and groundwater in shallow sandy aquifers. Specific objectives of this study are to: (1) examine how water table depth varies daily and seasonally under a pine plantation, a banksia woodland and a sparse grassland; (2) determine depth-dependent specific yields under both rising and falling water table conditions; (3) estimate daily and seasonal groundwater recharge and ET_g under three contrasting vegetation covers; and (4) identify the controlling factors on groundwater yields in shallow sandy aquifer systems.

Materials and methods

Site description

The study was undertaken on an unconfined surficial aquifer on Bribie Island (26°59´04´´S, 153°08´18´´E), southeast Queensland, Australia (Fig. 1). The island stretches approximately 30 km from north to south and has an average width of 5 km with a total area of 144 km^2 (Isaacs and Walker, 1983). This area experiences a subtropical climate with a hot humid summer and a mild dry winter. Mean annual rainfall over the past 30 years is 1405 (± 338) mm with 70% of annual rainfall typically occurring during the wet season (November to April). The average monthly temperature ranges from 16.2 ℃ in July to 26.6 ℃ in January. Bribie Island has an average elevation of ~5 m Australian Height Datum (AHD) with the maximum value of 13 m AHD. The topography consists largely of the elevated areas which correspond to two parallel sand dune ridges and a separating swale. However, the island is generally considered to be one of low relief. The extensive unconfined upper aquifer consists of fine to medium sands lying over cemented low permeability layers, with an average water table depth of ~1.2 m below land surface. Using a constant head permeameter (Eijkelkamp-

Agrisearch Equipment, Giesbeek, the Netherlands), an average saturated hydraulic conductivity of 8.5 m d^{-1} was determined for the unconsolidated sands.

Fig.1

The exotic pine trees have replaced large areas of native banksia vegetation along the two major sand dune ridges on the island. To minimize the effect of tides and groundwater pumping on water table fluctuations, three field sites with different vegetation cover were carefully selected in the interior of the island (Fig.1). These were along a belt transect which was normal to the coastline and crossing a relatively elevated section (dune). The transect is aligned with expected groundwater flow to adjacent wetlands. Two study sites were established in the pine plantation and banksia woodland, ~400 m from each other. The site areas are 50 m × 50 m (~8.4 m AHD) and 25 m × 25 m (~7.8 m AHD) for pine plantation and banksia woodland, respectively (Fig.1). The pine hybrid (*Pinus elliottii* Engelm. × *Pinus caribaea* Morelet var. *hondurensis*) was planted in 2001 with roughly 5.0 m × 2.5 m spacing. The pine trees reached an average height of 13.3 m, with a stem density of 840 trees ha^{-1} and a stand basal area of 23.6 m^2 ha^{-1}. The native banksia woodland was largely dominated by wallum banksia (banksia *aemula* R.Br.) with an average tree height of 6.8 m. The woodland had a stem density of 371 tree ha^{-1} and a basal area of 21.3 m^2 ha^{-1}. A third grassland site (30 m × 30 m) between the other two sites (but closest to the pine plantation at around 50 m distance) was covered with sparse grasses (*Leptocarpus tenax* R.Br.) and with a higher surface elevation of ~9.3 m AHD.

Field data acquisition

To characterize water table fluctuations for the vegetation covers, each field site was instrumented with a cluster of three monitoring wells (in triangle arrangement at 20–40 m spacing) equipped with pressure transducers (Level Troll 300, In-Situ Inc., USA). The average water levels obtained from three wells were used for estimates of recharge and ET_g at each site. Monitoring wells were installed to a depth of 2.0 m using a 51 mm diameter, 1.5 m long PVC screen and 1.5 m PVC riser. Augered sand was backfilled around the wells to a depth of 0.25 m below land surface and granular bentonite was then packed around land surface to avoid preferential flow. Apart from water pressure measurements, atmospheric pressure was monitored using a barometric datalogger (Baro Troll 100, In-Situ Inc., USA) to obtain water levels. The monitoring wells were vented to connect with the atmosphere and prevent air compression inside the PVC tubing. The water level data were measured from 1 November 2011 to 31 October 2013 and automatically recorded at 15-min intervals. Data were collected quarterly from the pressure transducers and the water table depth was manually measured by a dip meter during each field trip to check the logged water level values.

An automatic weather station was installed on a 15-meter-high mast located above the canopy and in the center of the pine plot to measure meteorological variables, including temperature and relative humidity, wind speed and direction, solar radiation and soil heat flux. Potential evapotranspiration (PET) was estimated using the Penman-Monteith equation (Monteith, 1965) with parameters obtained from the pine plantation (Fan et al., 2014). Gross rainfall was measured using a tipping-bucket rain gauges (RG3-M, Onset Computer Corp., USA) located in a nearby well-exposed clearing next to the banksia woodland. To obtain the net rainfall (throughfall plus stemflow) reaching the forest floor, throughfall was measured using 15 tipping-bucket rain gauges in the pine plantation and 8 troughs connected to 8 rain gauges in the banksia woodland. Stemflow was also collected in the pine plantation and banksia woodland using 6 and 8 collars connected to rain gauges, respectively. A detailed description of rainfall and throughfall measurements was presented by Fan et al. (2014).

Groundwater recharge estimation using the water table fluctuation method

The water table fluctuation (WTF) method is widely used to estimate spatially-averaged gross recharge for unconfined shallow aquifers (Healy and Cook, 2002; Delin et al., 2007):

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

where R is the estimated gross recharge (m); $S_y(h)$ is the depth-dependent specific yield; Δh is peak rise in water level attributed to the recharge period (m); Δt is the time of the recharge period. The WTF method assumes rises of groundwater levels in unconfined aquifers are only due to recharge water arriving at the water table (Healy and Cook, 2002; Scanlon et al., 2002). The method is best applied in areas with shallow water tables that demonstrate sharp rises in water levels over short time periods, which is applicable in our coastal sandy environment.

The water level rise in Eq. (1) during a recharge event was calculated as the difference between the peak of the water level rise and the low point of the extrapolated antecedent recession curve at the time of the peak, which is the trace that the well hydrograph would have followed in periods of no rainfall. Similar to Crosbie et al. (2005), the effects of evapotranspiration from the water table, lateral flow in and outwere coupled into the rate of water table decline. The master recession curve (MRC) approach was used to obtain the projected groundwater decline in each of the monitoring wells (Heppner and Nimmo, 2005; Crosbie et al., 2005; Heppner et al., 2007), rather than using more subjective graphical extrapolation methods. The MRC approach assumes that higher groundwater levels lead to larger hydraulic gradients and thus to larger water table declines due to discharge. During rain-free days, the decline rate was calculated as the decline in the groundwater level per day. To describe the relationship between rate of water table decline and depth to water table depth , regression functions are fitted to available data. The potential groundwater level that would have occurred under rain-free conditions can be calculated for a given groundwater level during rainfall events using these regression functions.

Groundwater evapotranspiration estimation using the White method

White (1932) developed an empirical method to quantify daily groundwater use by vegetation via evapotranspiration from the analysis of shallow water table fluctuations. The White method assumes: (1) diurnal water table fluctuations are caused by plant water use; (2) night-time water use from vegetation is negligible; and (3) a net inflow rate during night (midnight and 4 a.m.) is representative as a daily average rate. The daily groundwater evapotranspiration (ET_g) is obtained using the following equation (White, 1932):

$$ET_g = S_y(h)(24r \pm \Delta s) \quad (2)$$

where r is the net inflow rate between midnight and 4 a.m. (mm h^{-1}) and Δs is the net change of water table during a 24-h period (mm d^{-1}). A slight modification to the original White method suggested by Loheide et al. (2005) was applied in this study, where r was estimated as the average value of the net inflow rates calculated between midnight and 6 a.m. on the day of interest and the following day.

Determination of specific yield

The specific yield is defined as the volume loss or gain of water per unit area of aquifer associated with a corresponding unit drawdown or rise in water table (Freeze and Cherry, 1979):

$$S_y = \frac{V_w}{A\Delta z} \quad (3)$$

where V_w is the volume of water released or stored, A is the aquifer area and Δz is the change (decline or rise) in water table elevation. This definition is misleading as the specific yield can vary with depth to water table and with the time scale of observation (Duke, 1972; Said et al., 2005). The variation in specific yield beyond the daily time frame in this study was neglected due to the fast response of the water table in our sandy environment. To obtain reasonable estimates of recharge and ET_g, depth-dependent specific yields under falling and rising conditions were determined using laboratory-based drainage experiments on extracted cores (Cheng et al., 2013) and the ratio of water table rise to rainfall amount for different water table elevations using the field observations (Carlson Mazur et al., 2013), respectively.

Two undisturbed soil columns were excavated from the study sites using 80 cm high steel pipes with an inner diameter of 15 cm. In the laboratory, each column was slowly saturated from the bottom to minimize the trapped air and drained layer by layer using 8 taps identically spaced on the side of columns (3 replicates). Each drainage was stopped when a steady hydraulic state in the soils was reached (i.e., no further drainage out of column). Based on Eq. (3), the specific yield was calculated for each layer using the weight of the drained water recorded by a balance (SP402 Scout-Pro, Ohaus, USA), cross-sectional area of the column (177 cm^2) and the drawdown in water table (10 cm). The calculated specific yield for each layer was considered as the value corresponding to the midpoint between two drainage levels.

Specific yield was also estimated from the response of the water table to each rainfall event as the ratio of water level rise to net rainfall depth at each site. Rainfall events were only included if the previous rainfall within one week had replenished the soil moisture over the entire unsaturated zone and thus caused water level rises. Large rainfall events when the water table was within 0.5 m of the surface with substantial runoff potential were not used for S_y estimation.

Results and discussion

Seasonal and diurnal water table fluctuations in response to rainfall and ET_g

Annual gross rainfall during the hydrological years 2012 and 2013 was 2093 mm and 1493 mm, respectively, which were higher than the long-term mean of 1405 mm. As we previously presented (Fan et al., 2014), the annual rainfall interception losses were estimated at 16.4% of gross rainfall for banksia woodland and 22.7% for pine plantation. Thus we take the resulting net rainfall under banksia woodland as 1737 mm in 2012 and 1239 mm in 2013, and the corresponding net rainfall under the pine plantation as 1633 mm and 1164 mm, respectively. The interception loss from sparse grassland was considered to be minimal since the grasses were small and sparsely distributed. Major rainfall that occurred during the wet season (November–April) accounted for ~76% of the annual rainfall (Fig.2). No rainfall occurred in both August 2012 and 2013, the driest months recorded since 1983.

Fig.2

Seasonal fluctuations in water table depth were clear under the three vegetation covers and the fluctuation patterns were similar (Fig.3). Over the 2-year period, the depth to water table under the sparse grassland varied from 0.21 m to 1.77 m and averaged 1.02 m. The depth to water table averaged 0.55 m at the woodland wells and 0.68 m at plantation wells, ranging from ponded conditions to 1.47 m and from 0.02 m to 1.53 m, respectively. Water table fluctuations were not evaluated if the water table was above the land surface. Water table rises of between 0.02 m to 0.97 m were recorded in response to various rainfall events. Depending on the amount of rainfall and the initial depth to water table, the water table rise peaked from 0.5 to 73 hours of its initiation (Fig.3). This is an appropriate time frame (hours or a few days) for application of the water table fluctuation method (Healy and Cook, 2002). However, these water table rises were not necessarily resulted from recharge. Infiltrating rainwater can trap air in the unsaturated zone and cause the Lisse effect (Heliotis and DeWitt, 1987). Trapped air potentially reduced the profile water storage capacity, with less water to raise the same water table relative to that without the entrapped air effects (Nachabe et al., 2004). The increase of air pressure in the unsaturated zone can partially cause the rises of water table. Although rapid water table rises were recorded by the pressure transducers (Fig.3), rises in water level comparable to expected values based on the depth of given rainfall and a gradual dissipation of the water table rise indicated that the Lisse effect could be considered minimal in our coastal sandy environment (Healy and Cook, 2002; Crosbie et al., 2005).

Fig.3

In the absence of rainfall events, diurnal fluctuations of groundwater levels were observed under the pine plantation and banksia woodland, whereas the sparse grassland hydrograph exhibited a continuous declining curve (Fig. 4). Over the 2-year period, diurnal water table fluctuations were detected at a depth of up to ~1.0 m below land surface (mbls) in the pine plantation and ~0.8 mbls in the banksia woodland, but the fluctuation magnitude was significantly reduced beyond 0.8 mbls for pine plantation and 0.6 mbls for banksia woodland. The water table declined during the daytime because of tree water use and rebounded to a level slightly lower than the maximum level of the previous day during the night when transpiration significantly diminished or ceased. The daily highest water level occurred between 6 a.m. and 8 a.m. and the daily lowest water level occurred at 4 p.m.–6 p.m. Diurnal fluctuation of the water table suggests that both pine and banksia trees are accessing the groundwater.

Fig.4

The amplitudes of groundwater fluctuations at our tree sites suggest that the root zone there developed a maximum root depth of 1.0 m, with the majority of active roots in the upper 0.8 m for the pine plantation and 0.6 m for the banksia woodland. We observed the fine root (diameter < 2 mm) distributions of pine and banksia trees by excavation adjacent to trees and found a high root length density in the upper 0.5 m. In general, the rooting depths of woody vegetation have been found to be highly variable with a mean maximum depth of 7.0±1.2 m for trees and 5.1±0.8 m for shrubs (Canadell et al., 1996). The shallow and spreading rooting systems for trees at our sites were likely to be associated with their growth adapting to the shallow water table conditions. No diurnal water table fluctuation occurred at the grassland site because the grasses here had relatively shallow root depths (0.1 m) compared to trees, but the depth to water table at this site was often larger (> 0.3 m) than those at the forested sites due to its higher elevation. This undetectable fluctuation can also be ascribed to the low water requirements by the sparse grasses.

Variability of specific yield with depth in shallow water table environments

In our laboratory experiments, the water drainage from each layer of the soil columns was fast due to the high conductivity of our well-sorted aeolian sands and normally ceased within 24 hours of initiation. Considering the daily timeframe used in the White method, the time-dependency of specific yield at our sites is ignored. Similar to other studies (Schilling, 2007; Shah and Ross, 2009; Carlson Mazur et al., 2013), specific yields obtained from both laboratory and field methods were found to vary with water table depth, with low S_y values close to the soil surface (Fig.5). Specific yields then increase with increasing depth to water table as more groundwater is drained from the soil profile and finally approaches a quasi-constant of 0.25 when the water table is more than ~1.0 m from the surface. This is consistent with the finding by Loheide et al. (2005), who argued depth-dependency of specific yield has to be considered when water table depths < 1 m. We derived sigmoid functions (Venegas et al., 1998) to describe the dependence of specific yield on depth to water table (p<0.05).

Fig.5

The S_y values calculated from the field water table response to rainfall were smaller than those obtained from drainage experiments, especially at the middle range of depths to water table. The difference in S_y obtained under falling and rising water table conditions may be due to hysteresis (entrapped air) in the soil water characteristic and air encapsulation below water table, where there is a difference in the volume of water able to be held at saturation and the volume able to be fully drained (Fayer and Hillel, 1986; Nachabe, 2002). Encapsulated air is likely to reduce the value of specific yield achieved from a rising water table compared with that determined by drainage from near full saturation situation in the laboratory. Similar discrepancies in S_y under wetting and drying conditions have been found by others (Said et al., 2005; Shah and Ross, 2009; Logsdon et al., 2010). It can also be partially due to the difference in spatial scales used in the S_y determination (ten-meter vs sub-meter). The rainfall-water table response method is able to provide information about the variation of specific yield with depth, but it is expected to give an overestimate of S_y due to the inclusion of infiltrating rainwater retained by the soil (Logsdon et al., 2010; Carlson Mazur et al., 2013). Logsdon et al. (2010) investigated the effect of soil wetting on S_y estimation in a crop field. They indicated the rainfall-rise method produced much higher S_y values if the amount of vadose zone water was not subtracted from rainfall depth, suggesting caution is required when applying the method when soils are dry prior to a rainfall event. We thus ignored all the rainfall events with long previous dry periods in this study. Only the rainfall events with recent rainfall where soil moisture was likely to be replenished and resulted in water level rises were considered.

Crosbie et al. (2005) tested different methods to estimate S_y and found that the rainfall-water table response provided the most reasonable estimates for recharge, which they attributed to S_y being calculated using the same temporal and spatial scale in which it was applied. Therefore, S_y from the water table response to rainfall was adopted to obtain estimates of recharge, whereas S_y determined by the drainage experiments was used to estimate ET_g since it corresponded to draining conditions.

Groundwater recharge under the three vegetation covers.

The relationship between the rate of water table decline and depth to water table (Fig.6) shows that the higher the water table is, the greater the decline rate. On average, the rate of water table decline decreased from ~5 cm d^{-1} to less than 1 cm d^{-1} as the water table elevation decreased from near the ground surface to 1.0 m below the surface. The decline rate incorporated factors affecting water level decreases, e.g., groundwater evapotranspiration, lateral flow in and out. A negative power function was fitted between the bin median of discharge rate and depth to water table. We presume that the water table recession behavior is unique to each site, which largely depends on the rates of discharge from the recharge site to the central swale or the ocean. The grassland site shows a higher rate of water table decline than forested sites most likely as a result of its associated higher hydraulic head gradient.

Fig.6

Recharge for each monitoring well was estimated by multiplying the groundwater level rise by the specific yield corresponding to the average level during each rainfall event using the equations in Fig. 5. There is an obvious seasonal trend in the estimated recharge with the major recharge occurring during the wet summers and autumns (Fig.7). Generally, the recharge pattern is similar to that of gross rainfall, with largest amounts of rainfall and recharge in December and January 2012, January and February 2013, during which the heavy rainfall replenished soil moisture and generated groundwater recharge. Although the annual recharge averaged from three vegetation sites amounted to 620 mm (40% of annual average net rainfall), the monthly average recharge distributions showed significant variations. The monthly average recharge estimated for the year 2011–2012 ranged from 11 mm in October 2012 to 208 mm in December 2011, representing 9%–73% of the monthly average net rainfall (Fig.7). For the year 2012–2013, the monthly average recharge varied from 8 mm in September 2013 to 221 mm in January 2013 (11%–67% of the monthly average net rainfall).

Fig.7

The reduction in recharge as a percentage of monthly net rainfall in late summer and autumn is due to the greater influence of the shallow water table, whereas the low percentage of recharge in winter and early spring is mainly ascribed to drier soils with higher moisture holding capacity and smaller rainfall events. There were several rainfall events that did not cause an increase in the water table elevation during the dry season, particularly at the grassland site with its thicker unsaturated zone. During these periods, most infiltrating rainwater was stored in the unsaturated zone and did not apparently recharge the shallow aquifer. Hence, major groundwater recharge primarily occurred in the early summer following the dry season. In this case, the lower water table and drier vadose zone had the largest capacity to accept more recharge after the significant replenishment of soil moisture by the frequent heavy summer storms in our subtropical coastal environment (Fig.7).

Temporal recharge patterns for the three field sites are similar due to the similar rainfall patterns between sites; however the magnitude of recharge is different. In the pine plantation, annual recharge amounted to 521 mm (31% of net rainfall) for year 2012 and 589 mm (49% of net rainfall) for year 2013, whereas annual recharge in the banksia woodland was less, with 357 mm (21% of net rainfall) for year 2012 and 449 mm (36% of net rainfall) for year

2013. Much greater annual recharge occurred in the grassland, where total recharge amounted to 1037 mm (49% of net rainfall) and 830 mm (56% of net rainfall) for years 2012 and 2013, respectively. Lower recharge values in the pine plantation and banksia woodland can be expected since ~20% of gross rainfall was intercepted by forest canopies. This was also due to the generally shallower water table at the forest sites limiting recharge, whereas the grassland had a relatively larger capacity to capture more infiltrating rainfall as groundwater recharge. The forested sites were lower in elevation and had several continuous weeks of near-saturated soil conditions in the wet season, and therefore, recharge was restricted during these periods. The difference in the annual recharge values at each site between the years 2012 and 2013 was attributed to differences in the rainfall in each year.

Overall, these recharge values are higher than results obtained in other studies on Bribie Island. For example, soil water balance modelling by Ishaq (1980) resulted in a recharge value of 13% of rainfall while Isaacs and Walker (1983) calibrated a numerical model for the southern part of the island using a recharge value of approximately 20%. Harbison (1998) estimated recharge values of 15% and 30% based on sodium and chloride mass balance respectively. The Department of Natural Resources reported a recharge value of 22% of total rainfall for the whole island using calibrated models (DNR, 1988). However, much lower recharge estimates have also been reported for the whole island, e.g., 8% by the Department of Natural Resources (DNR, 1996) and 7% by Harbison (1998). Since the estimated recharge values obtained by the water table fluctuation method were event-based gross recharge compared with the above steady state recharge rates, higher recharge percentages were expected in this study. Using the WTF method, Crosbie et al. (2005) estimated recharge for 6 field sites in a similar coastal sand-bed aquifer of Newcastle, Australia. The reported recharge percentage values ranged from 58% to 65% of gross rainfall. A deeper average water table (~2 m) than ours was recorded, which probably accepted more infiltrating rainwater and resulted in higher available recharge than that in our shallower water table environment. The water table fluctuation method appeared to produce reasonable recharge values in our study, but the uncertainties in recharge estimates directly resulted from the uncertainty in specific yield under rising water table condition has to be acknowledged. Despite this, the water table fluctuation method was useful to compare the relative influence of various vegetation on groundwater recharge in this environment.

Applying a water balance method, Brauman et al. (2012) found recharge for both cattle pastures and native forests in the highly permeable basalt catchments of tropical leeward Hawaii island were close to 100% of gross rainfall (range = 87% to 106 %), where difference in recharge under different vegetation covers was attributed to the direct fog interception by native forests. They concluded vegetation has small effects on water quantity in areas with highly permeable substrates and intense storms due to fast percolation of water beyond the rooting zone. In our study, the estimated annual recharge was lower (25%–35%) in native and planted forests than that of grassland (50%). In contrast to fewer interception losses in their study as a result of supplement by fog and clouds, ~20% of gross rainfall was intercepted by tree canopies and evaporated back into the atmosphere at our sites, which greatly reduced the potential recharge. The highly permeable sandy aquifer can potentially accept large amount of net rainfall as they suggested, but our shallow water table led to significant rejection of recharge in the wet season. The excess rainwater acted as overland flow to feed the central swale or the wetland through drainage channels along the tracks.

Groundwater discharge via evapotranspiration under the three vegetation covers

Groundwater evapotranspiration was estimated by multiplying the daily net inflow rate and the net fall of water table by the specific yield corresponding to the daily average level using the equations in Fig. 5. The White method was not applied during recharge events or when the water table was below the maximum rooting depth. Over the 2-year study period, this method was applied to 82 days in 2012 and 87 days in 2013 for banksia woodland and pine plantation. The results show that the daily ET_g generally decreases from summer to winter (Fig.8). The decline in ET_g during the winters reflects the decrease in the transpiration rate as the atmospheric evaporative demand is three times lower than that in the summer. The highest daily ET_g rates were observed in January 2012 and February 2013. The annual cumulative ET_g estimated by the White method amounted to 208 mm in 2012 and 217 mm in 2013 for pine plantation, while the corresponding values for banksia woodland were 111 mm and 131 mm, respectively. The daily ET_g over the year 2012 averaged 2.8 mm d^{-1} (range=1.0 to 5.1 mm d^{-1}) in plantation and 1.5 mm d^{-1} (range=0.4 to 4.1 mm d^{-1}) in woodland, while the corresponding values in 2013 were 2.9 mm d^{-1} (range=0.5 to 5.8 mm d^{-1}) and 1.7 mm d^{-1} (range=0.3 to 3.7 mm d^{-1}), respectively. The estimated ET_g for pine plantation was closer to the PET (0.8–6.8 mm d^{-1}) compared to banksia woodland. This was partially caused by the difference in PET between the pine plantation and the banksia wood since the PET was only calculated using parameters from the pine plantation. Although the seasonal patterns of ET_g between banksia and plantation were similar, the estimated ET_g for banksia woodland is approximately half of the corresponding values for pines. The higher ET_g at the pine plantation is largely explained by much higher tree density. The ET_g for banksia can also be restricted by weaker sensible heat transfer at the canopy surface caused by greater boundary layer resistance from its relatively broad leaves (Oke, 1978) and the higher aerodynamic resistance due to its lower canopy height (Valente et al., 1997).

Fig.8

Our daily ET_g estimates (0.3–5.8 mm d^{-1}) are generally lower than other ET_g estimates for pines and woodland species using White methods. For example, Vincke and Thiry (2008) found the estimated ET_g for a Scots pine (*Pinus sylvestris* L.) stand growing on a sandy soil in

Belgium ranged from 0.7 to 7.5 mm d^{-1} (PET=1.0–8.5 mm d^{-1}). Gribovszki et al. (2008) obtained ET_g rates of 3.2–10.5 mm d^{-1} (PET=5.0–16.5 mm d^{-1}) for a phreatophyte ecosystem dominated by alder (*Alnus glutinosa* L. Gaertn.) in Hungary. Butler et al. (2007) obtained ET_g rates of 2.9–9.3 mm d^{-1} for a cottonwood forest (*Populus* spp.) with less amounts of mulberry (*Morus* spp.) and willow (*Salix* spp.) in USA. However, our ET_g estimates are comparable with estimated ET_g rates (1.7–6.3 mm d^{-1}) for oak (*Quercus* spp.) and maple (*Acer* spp.) trees by Nachabe et al. (2005) using diurnal fluctuations in the total moisture of sandy soil above a shallow water table (PET=2.0–7.5 mm d^{-1}). In the above studies, the PET was generally higher than our PET estimates of 0.8–6.8 mm d^{-1}, which probably resulted in the higher ET_g estimates accordingly as the ET_g are largely driven by the meteorological variables that control PET such as net radiation, temperature and humidity (Butler et al., 2007; Gribovszki et al., 2008). This difference can also result from differences in depth to water table and forest characteristics. However, the White method seems to be applicable for comparison of ET_g between exotic and native tree species in subtropical coastal environments.

The relationship between ET_g and PET was further analyzed (Fig.9), which confirmed that the groundwater discharge via root water uptake by both forests correlates strongly with PET, with higher daily ET_g rates corresponding to higher daily PET. Generally, there is a positive linear correlation between ET_g and PET for both forests, with substantial scatter around the trendline. This scatter is likely the result of changes in other environmental factors such as soil moisture availability, which can impact evapotranspiration and are not included in the PM equation.

Fig.9

Depth to water table is another important factor determining the groundwater contribution to transpiration. The ratio of ET_g to PET can reflect the fraction of *ET* that is derived from the groundwater. The ET_g/PET ratio increases for depths to water table which range from near-surface to ~30 cm for banksia woodland and to ~45 cm for pine plantation, and then decreases as the water table falls below these thresholds (Fig.10). Using numerical simulations, Shah et al. (2007) identified similar thresholds (31–36 cm) for forests with extinction depths of ~2.5 m in sandy soils. Root water uptake by trees was most likely stressed under anaerobiosis conditions when the water table was close to the land surface (Feddes et al., 1978). The roots gradually became active due to increased oxygen in the unsaturated soil as the water table fell towards the threshold depths. The ET_g fraction reached maximum value at threshold depths, when estimated ET_g rates were closer to PET. However, the ET_g fraction started to decrease when water table exceeded the thresholds as a result of a decreasing root density with depth and tended to a value of zero when water table approached maximum rooting depth. Although difference in PET between the pine plantation and the banksia wood was expected, the tendency of ET_g/PET ratio with depth to water table was considered to be similar.

Fig.10

A representative estimation of the net inflow rate throughout the day is required in the White method, because the used assumption of constant inflow rate is questionable in most cases due to changing hydraulic gradients between the recovery source and the monitoring site (Troxell, 1936; Loheide, 2008; Fahle and Dietrich, 2014). Various modifications have been developed for the original White method to improve ET_g estimates by deriving a time-dependent inflow rate (Gribovszki et al., 2008; Loheide, 2008) or an average rate across the day (Miller et al., 2010). Fahle and Dietrich (2014) evaluated different inflow estimation methods using hourly flow data measured from the lysimeter experiments. Compared to the original White method, better estimates of the inflow rate were obtained when using a two-night average value suggested by Loheide (2005) over longer time spans (6 p.m.–6 a.m.).

Thus, the average inflow rate estimated from the two-night values between midnight and 6 a.m. in this work was considered to be representative of the net inflow rate throughout the day of interest.

Conclusions and Implications

In this study, water table measurements in a sandy aquifer under three adjacent vegetation covers were collected over a 2-year period on Bribie Island in subtropical coastal Australia. Water table fluctuations were analyzed to estimate groundwater recharge and discharge through ET_g. The results show substantial seasonal variations in water table depth. The water table at the forested sites displayed a diurnal fluctuation at a depth of up to ~1.0 m, whereas the grassland site exhibited no diurnal fluctuations. For the two years studied, the estimated annual recharge at the sparse grassland site (49–56% of net rainfall) was larger than that at the pine plantation (31–49% of net rainfall), which in turn was larger than that at the banksia woodland (21–36% of net rainfall). The annual cumulative ET_g rates estimated by the White method were higher in the pine plantation than in the banksia woodland, with an average daily ET_g of 2.9 mm d^{-1} in pine plantation and 1.6 mm d^{-1} in banksia woodland for a total of 169 days during hydrological years 2012 and 2013.

The results from this study suggest recharge in the shallow sandy aquifer is dominated by early wet season rainfall but restricted by wet antecedent soil moisture conditions when the water table approaches the soil surface. Groundwater evapotranspiration was largely driven by meteorological variables, but also moderated by depth to water table. Considering the similar net annual recharge (gross rainfall minus ET_g) between the pine plantation and banksia woodland but much lower net annual recharge at the grassland, the conversions from native vegetation to exotic pine plantations may reduce the local water yields and lower the groundwater level in these areas, especially during the dry seasons and years. Future work will expand upon this study by examining total tree water use to better understand the hydrological effects of vegetation cover changes in shallow sandy aquifer systems.

Acknowledgments

This study was financially sponsored by the National Centre for Groundwater Research and Training (NCGRT), co-funded by the Australian Research Council and the National Water Commission. The authors thank Jeremy Canard for his assistance in the field installation and observations. We acknowledge the insightful suggestions from Marcus Fahle and one anonymous reviewer. Special thanks are due to HQPlantation Pty Ltd for providing the field sites.

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Figure Captions

Fig.1 Location map of the pine plantation, banksia woodland and sparse grassland.

Fig.2 Relationship between time to peak and peak rise of water table for different groups of initial water table depths.

Fig.3 Seasonal rainfall distribution and average water table fluctuations observed from three monitoring wells at pine plantation, banksia woodland and sparse grassland, for the period from 1 November 2011 to 31 October 2013.

Fig.4 Example of average diurnal water table fluctuations observed from three monitoring wells at pine plantation, banksia woodland and sparse grassland from 20 July to 20 August 2012.

Fig.5 Specific yield as a function of depth to water table from drainage experiments (●) and water table response to rainfall (○).The error bars represent one standard deviation from the mean. Well data form all three sites were used to derive specific yield from water table response to rainfall.

Fig.6 Box plots of water table decline rate binned into 0.2 m intervals by depth to water table.

Fig.7 Distribution of estimated monthly gross recharge and observed monthly net rainfall at each site over the 2-year period.

Fig.8 Daily potential evapotranspiration (PET) and estimated groundwater evapotranspiration (ET_g) by pine plantation and banksia woodland (a) from 26/12/2011 to 31/08/2012 and (b) 26/12/2012 to 31/08/2013. Due to recharge events or depths to water table larger than maximum root depths, no ET_g was detected between days with ET_g and beyond the above periods over the two years.

Fig.9 Relationship between daily groundwater evapotranspiration (ET_g) and potential evapotranspiration (PET) at the pine plantation and banksia woodland.

Fig.10 The ratio of groundwater evapotranspiration (ET_g) to potential evapotranspiration (PET) as a function of depth to water table at the pine plantation and banksia woodland.

Figure 1

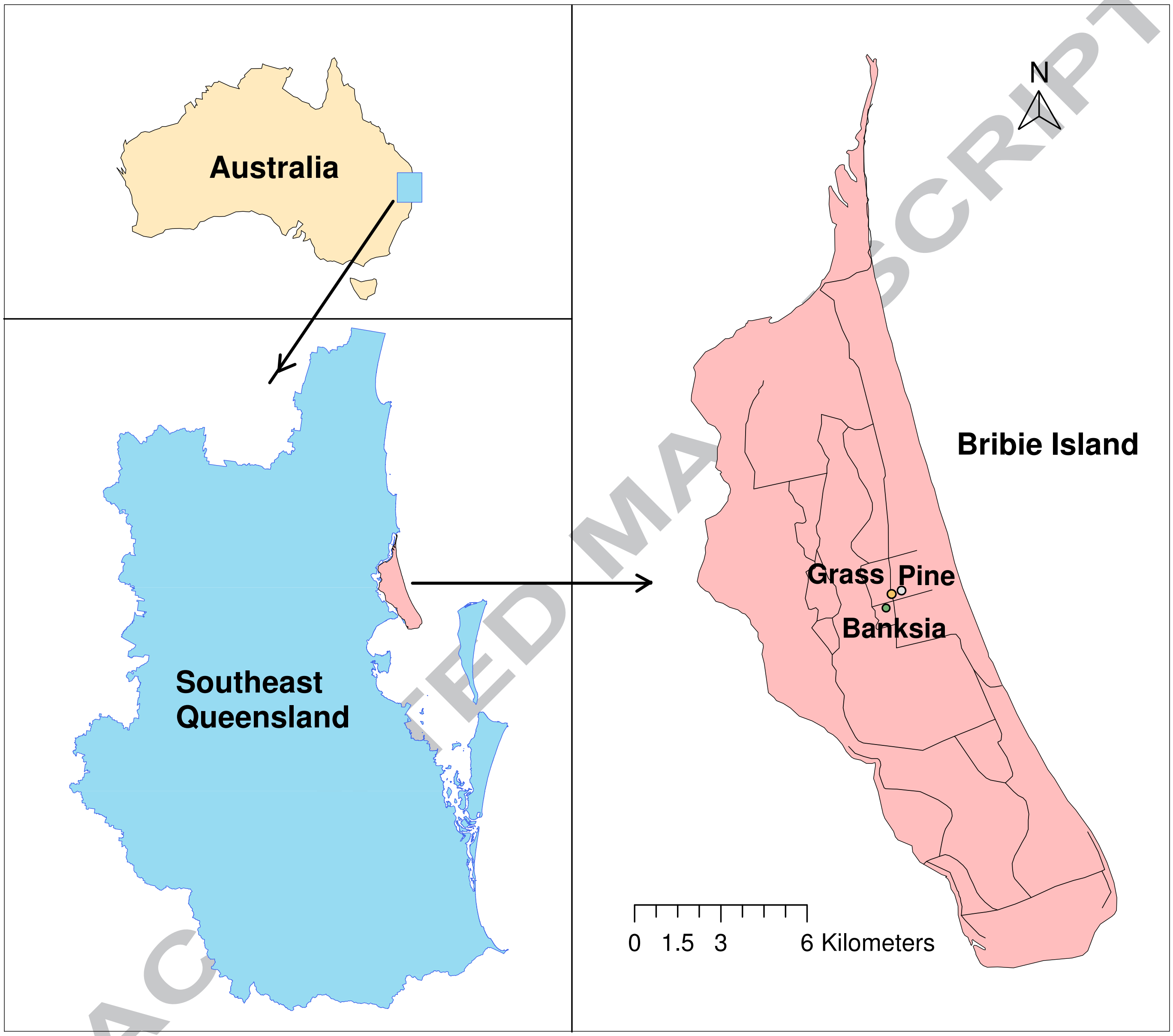
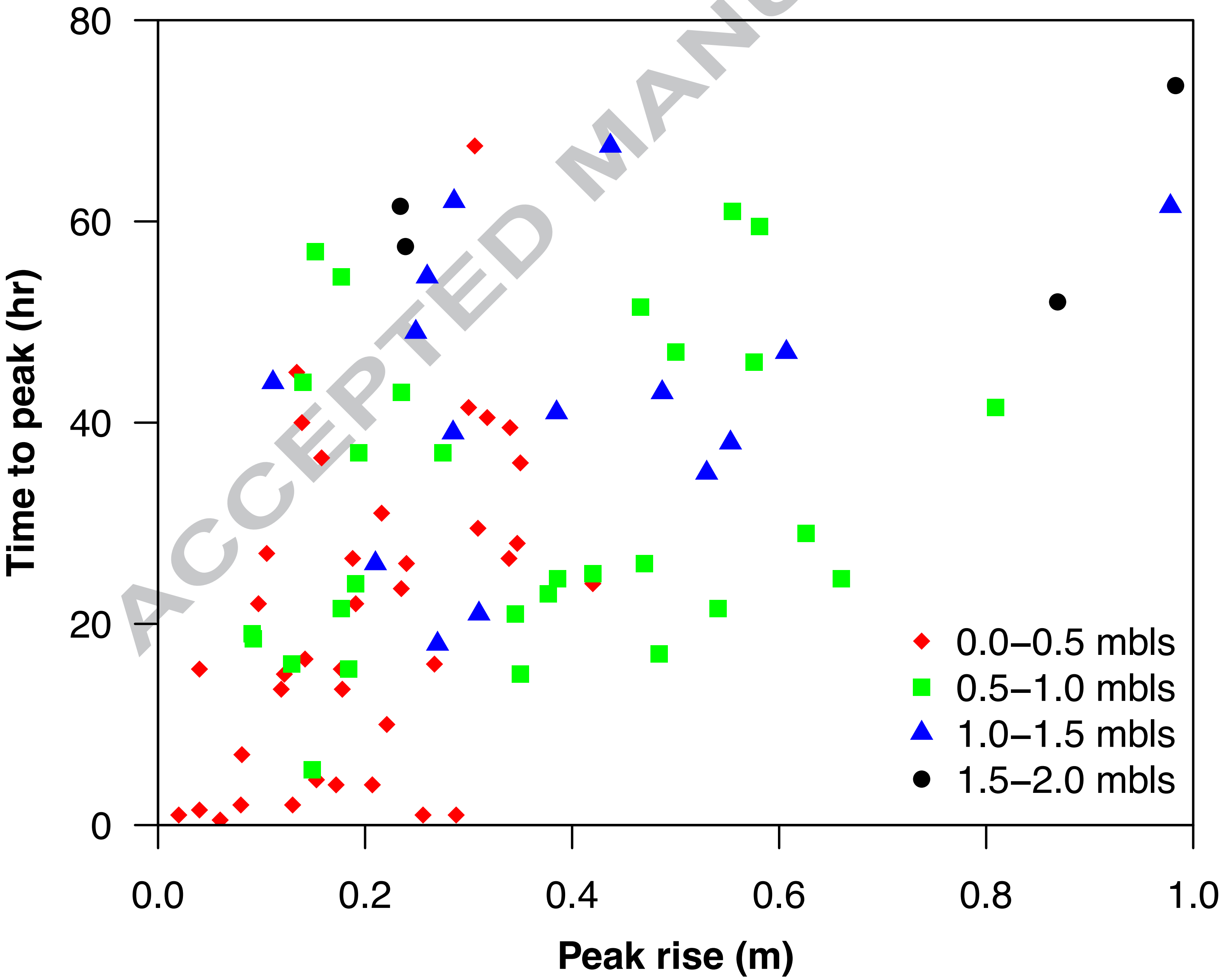
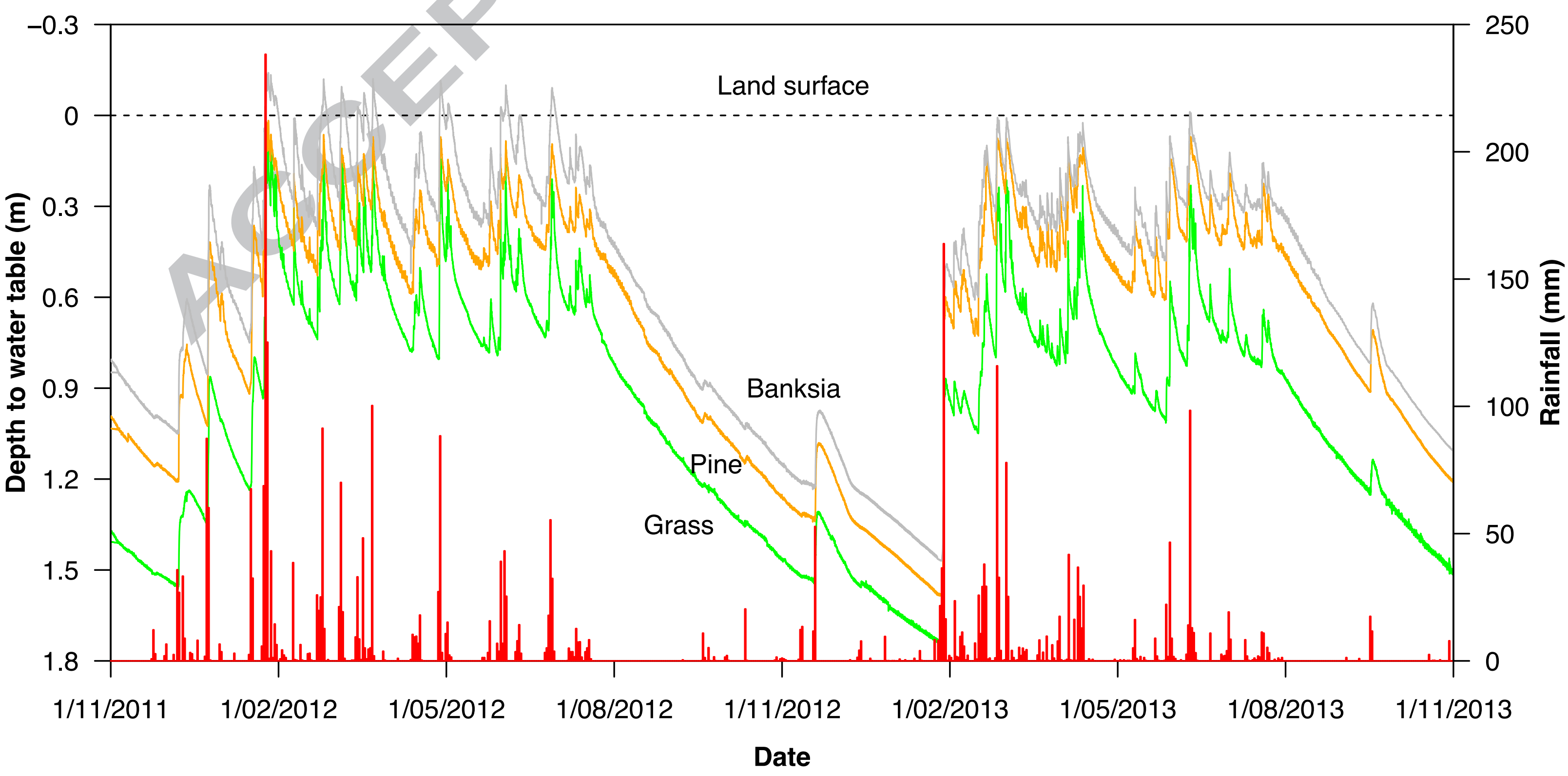

Australia
Southeast Queensland
N
Bribie Island
Grass
Pine
Banksia
0 1.5 3 6 Kilometers

Figure 2



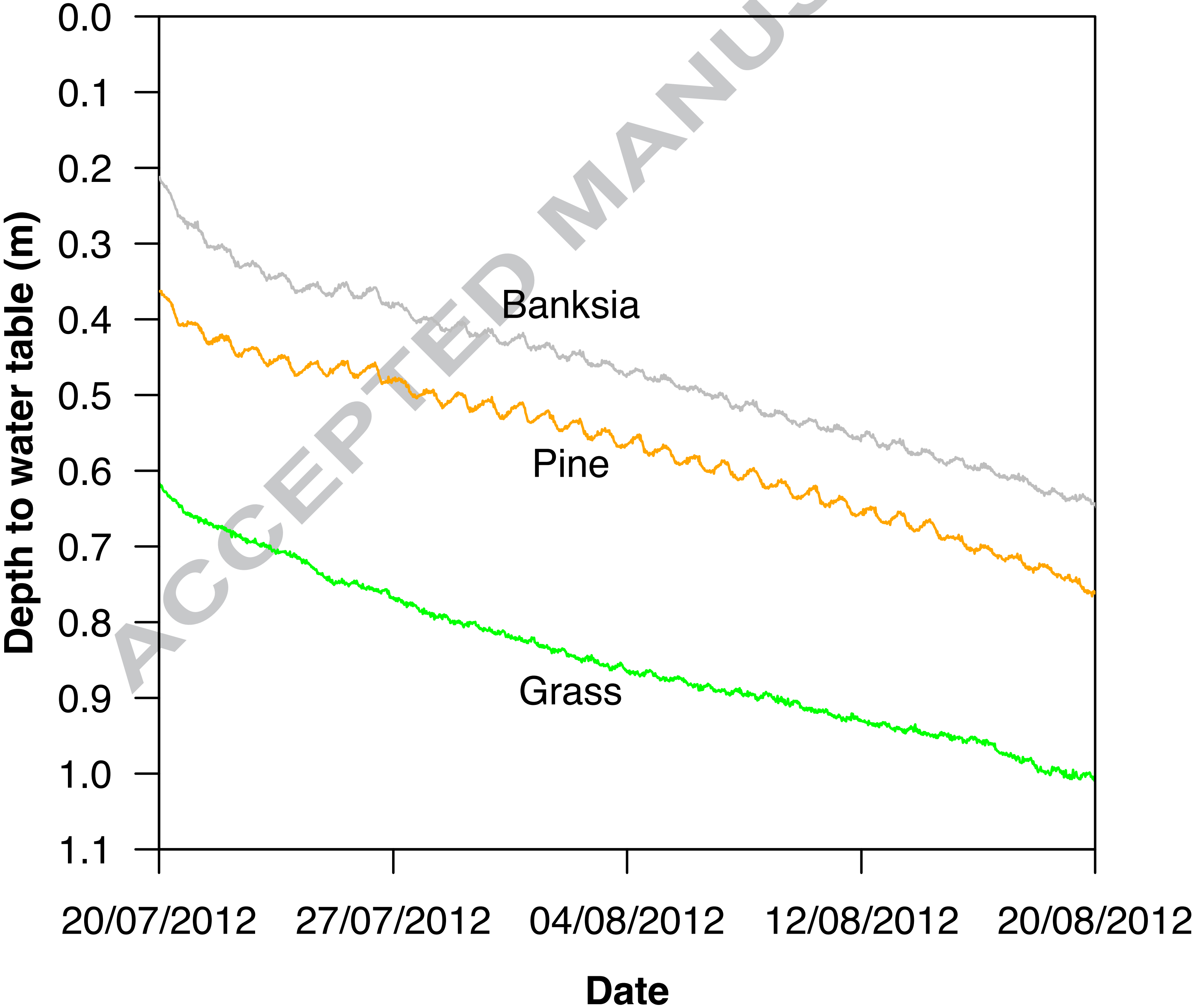
80
60
40
20
0
Time to peak (hr)
0.0
0.2
0.4
0.6
0.8
1.0
Peak rise (m)
0.0–0.5 mbls
0.5–1.0 mbls
1.0–1.5 mbls
1.5–2.0 mbls

Figure 3



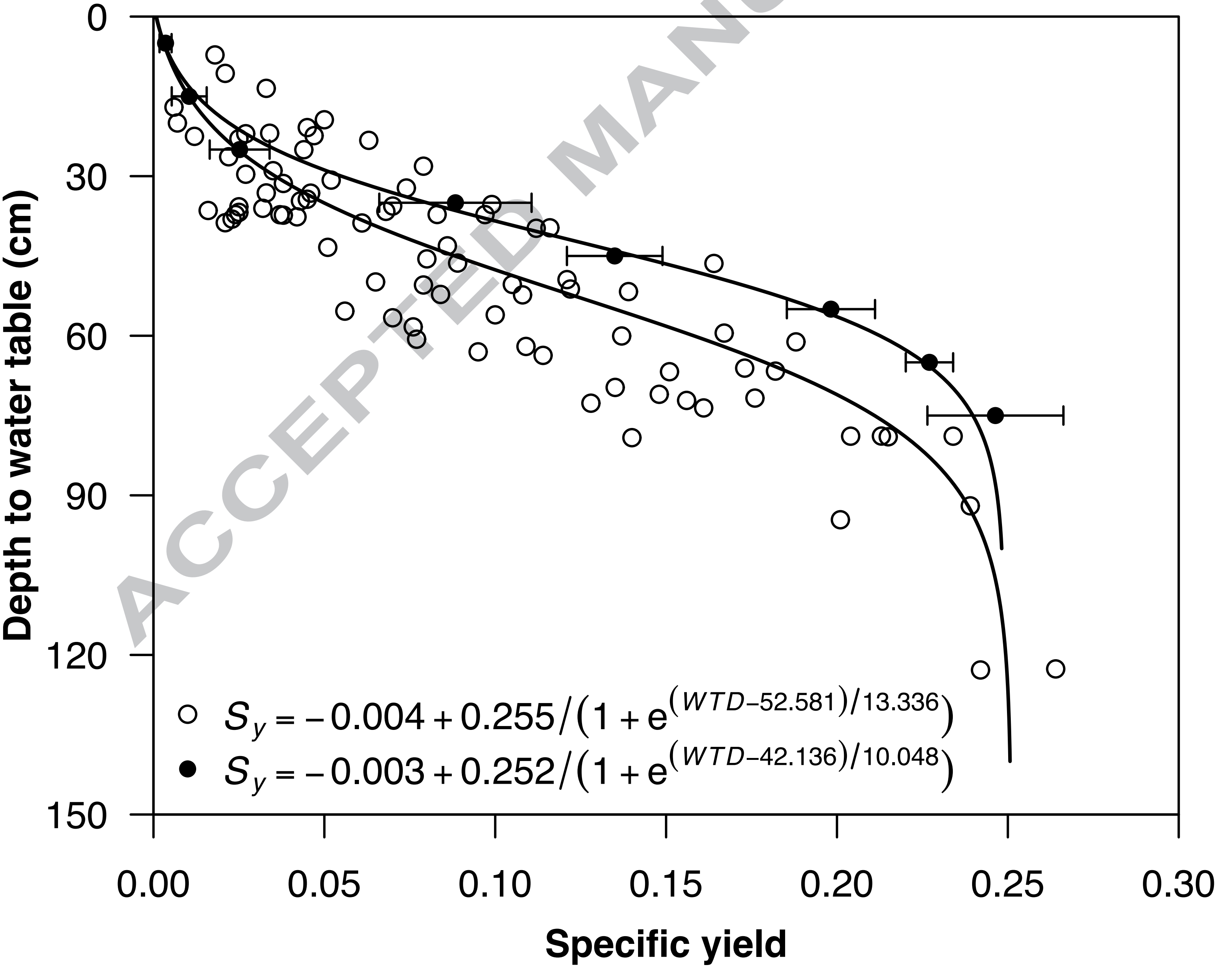
Land surface
Banksia
Pine
Grass
Depth to water table (m)
Rainfall (mm)
Date
−0.3
0
0.3
0.6
0.9
1.2
1.5
1.8
250
200
150
100
50
0
1/11/2011
1/02/2012
1/05/2012
1/08/2012
1/11/2012
1/02/2013
1/05/2013
1/08/2013
1/11/2013

Figure 4



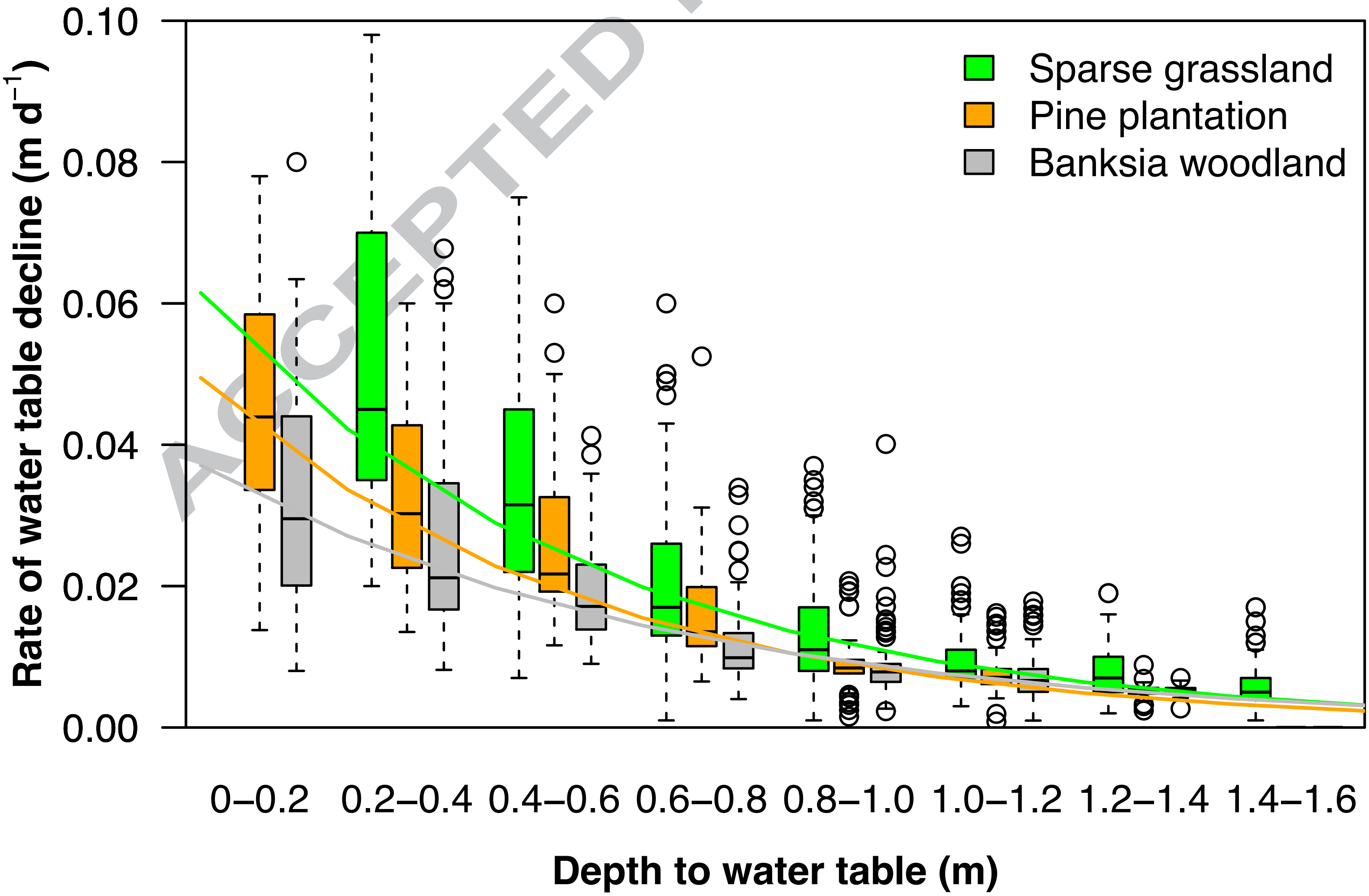
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
0.9
1.0
1.1
Depth to water table (m)
Banksia
Pine
Grass
20/07/2012
27/07/2012
04/08/2012
12/08/2012
20/08/2012
Date

Figure 5



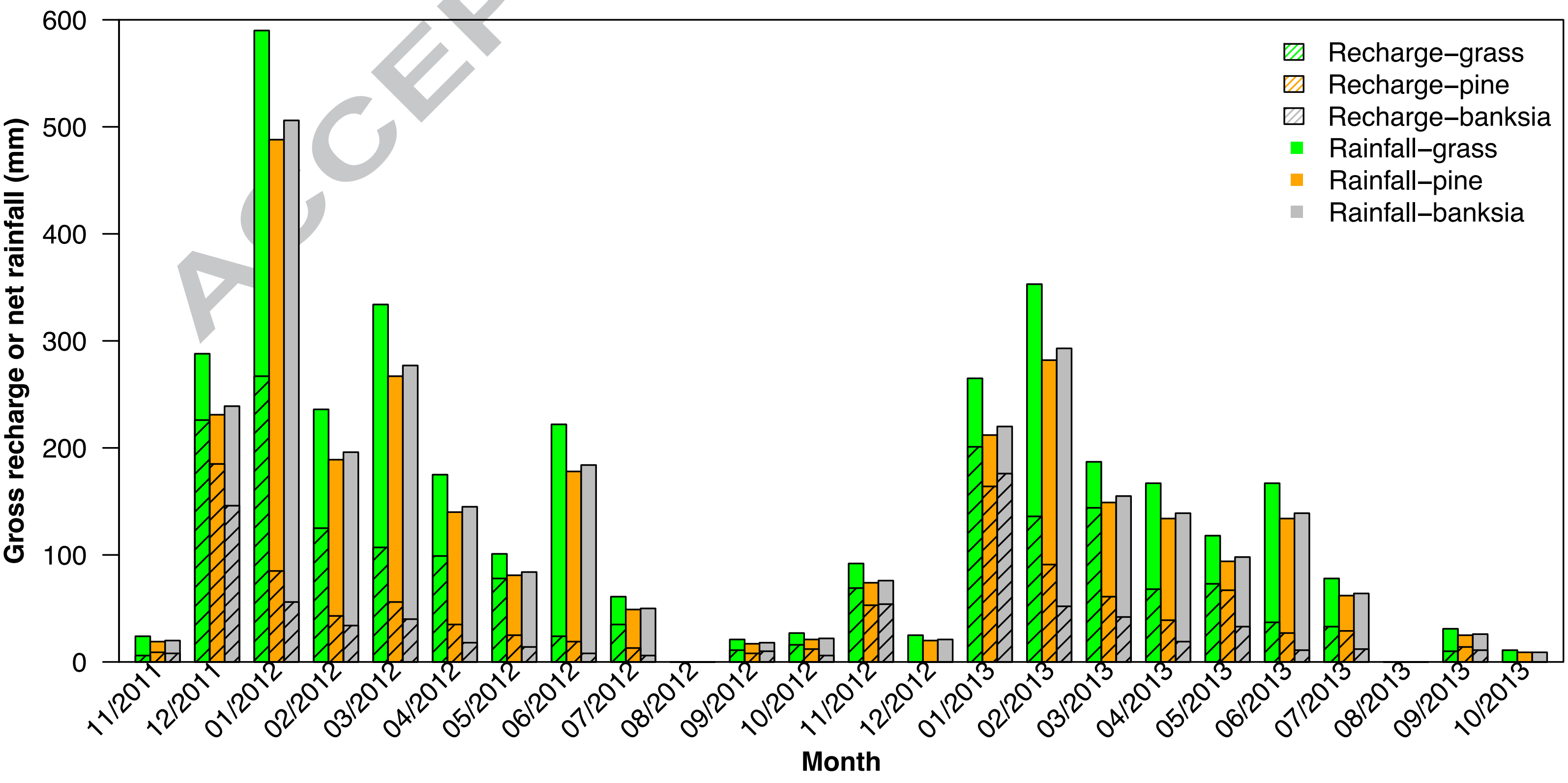
0
30
60
90
120
150
Depth to water table (cm)
0.00
0.05
0.10
0.15
0.20
0.25
0.30
Specific yield
$S_y = -0.004 + 0.255/(1 + e^{(WTD-52.581)/13.336})$
$S_y = -0.003 + 0.252/(1 + e^{(WTD-42.136)/10.048})$

Figure 6



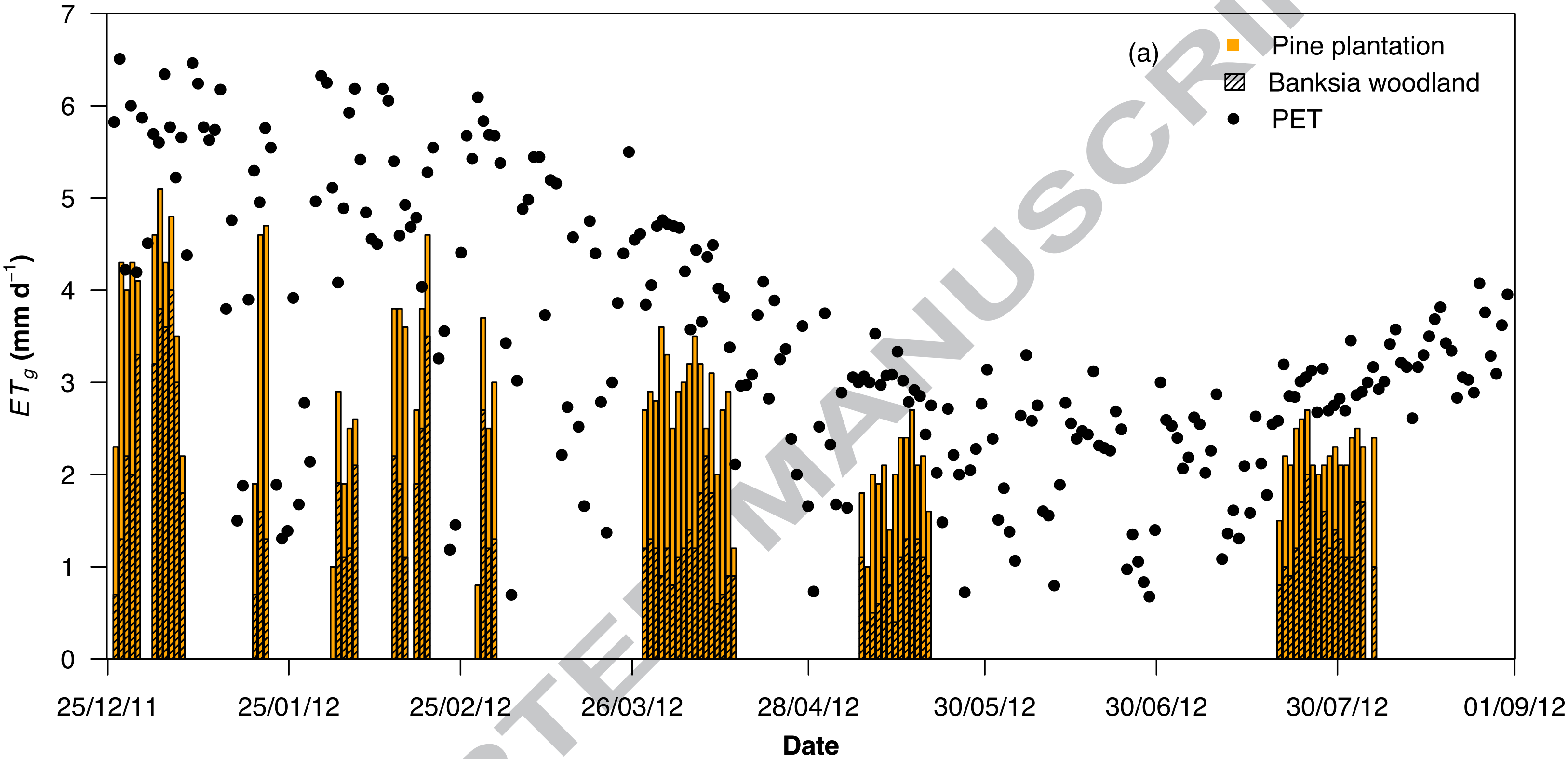
Sparse grassland
Pine plantation
Banksia woodland
Rate of water table decline (m d^{-1})
0.10
0.08
0.06
0.04
0.02
0.00
0–0.2
0.2–0.4
0.4–0.6
0.6–0.8
0.8–1.0
1.0–1.2
1.2–1.4
1.4–1.6
Depth to water table (m)

Figure 7



Gross recharge or net rainfall (mm)
600
500
400
300
200
100
0
Recharge−grass
Recharge−pine
Recharge−banksia
Rainfall−grass
Rainfall−pine
Rainfall−banksia
11/2011
12/2011
01/2012
02/2012
03/2012
04/2012
05/2012
06/2012
07/2012
08/2012
09/2012
10/2012
11/2012
12/2012
01/2013
02/2013
03/2013
04/2013
05/2013
06/2013
07/2013
08/2013
09/2013
10/2013
Month

Figure 8


(a)
Pine plantation
Banksia woodland
PET
ET_g (mm d^{-1})
0
1
2
3
4
5
6
7
25/12/11
25/01/12
25/02/12
26/03/12
28/04/12
30/05/12
30/06/12
30/07/12
01/09/12
Date

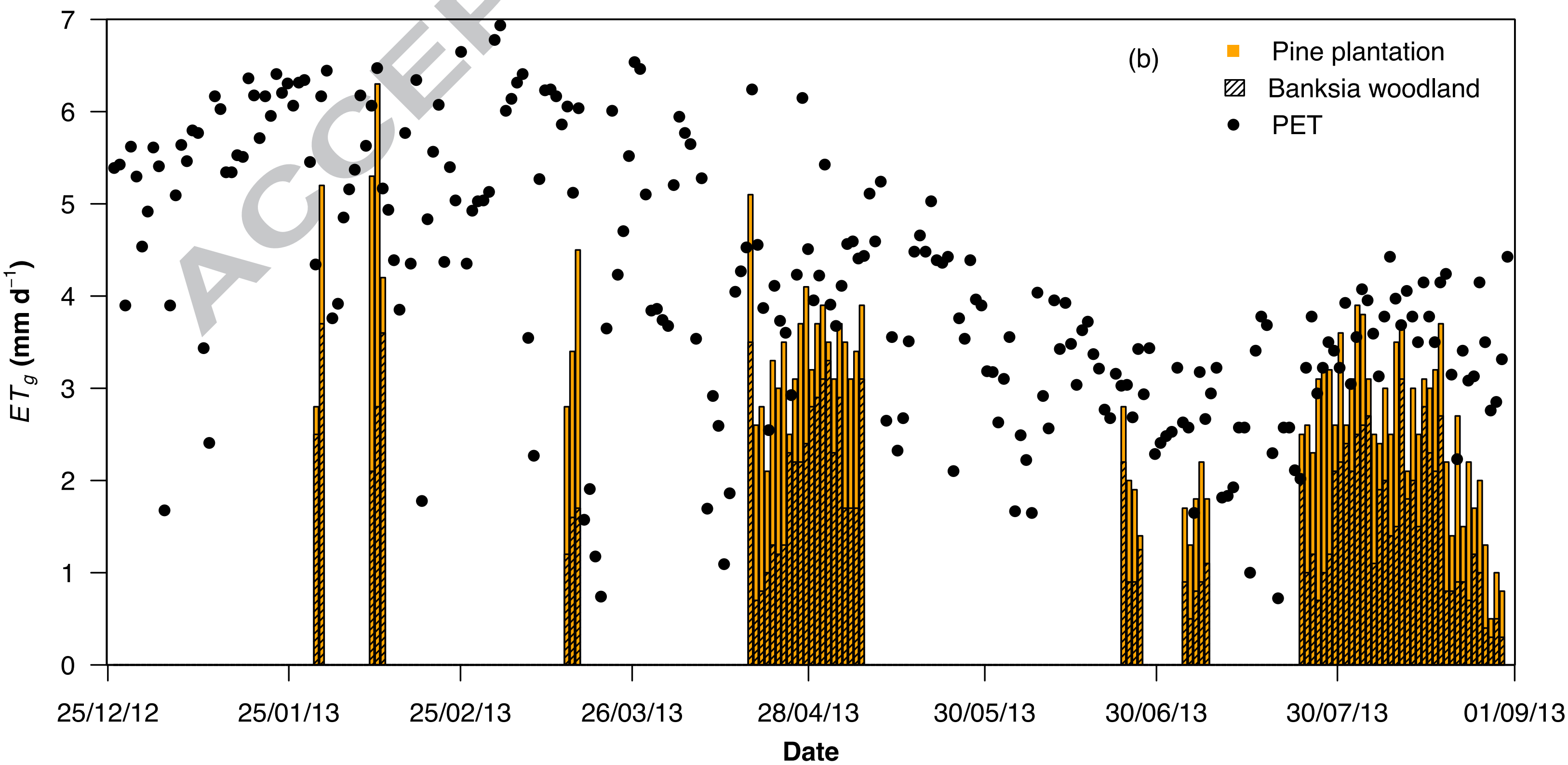
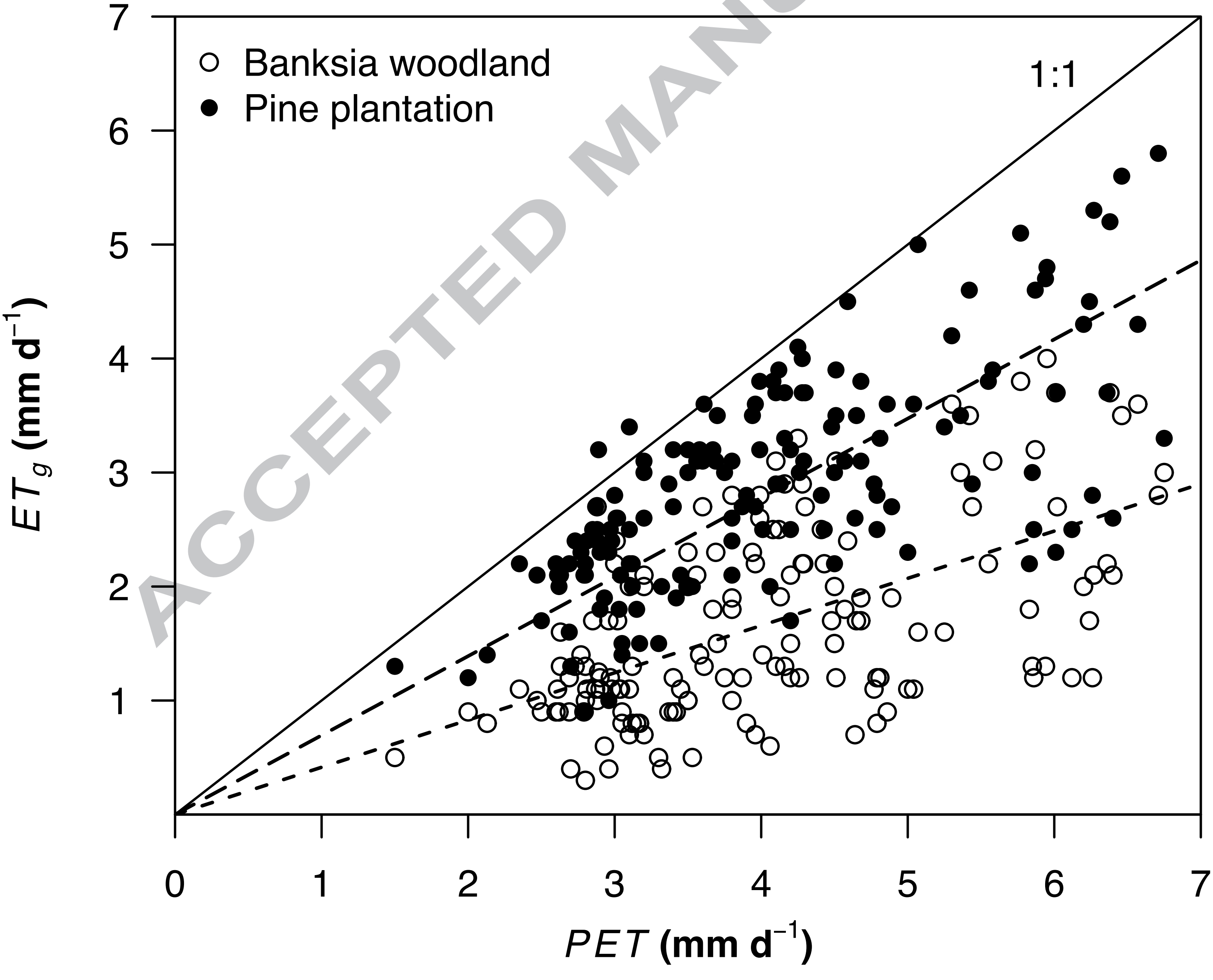
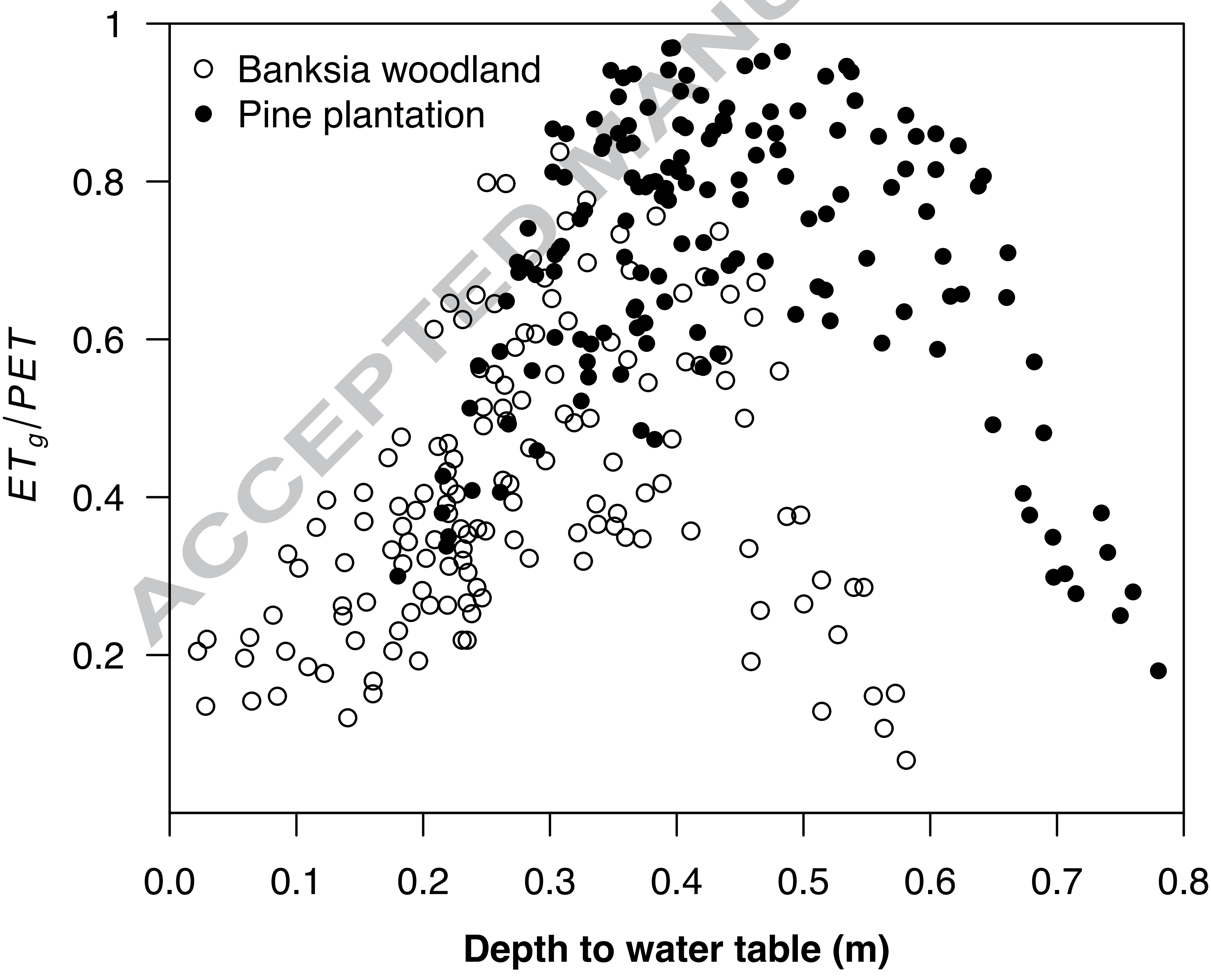

(b)
Pine plantation
Banksia woodland
PET
ET_g (mm d^{-1})
0
1
2
3
4
5
6
7
25/12/12
25/01/13
25/02/13
26/03/13
28/04/13
30/05/13
30/06/13
30/07/13
01/09/13
Date

Figure 9



Banksia woodland
Pine plantation
1:1
ET_g (mm d^{-1})
PET (mm d^{-1})
0
1
2
3
4
5
6
7

Figure 10



Banksia woodland
Pine plantation
ET_g/PET
1
0.8
0.6
0.4
0.2
0.0
0.1
0.2
0.3
0.4
0.5
0.6
0.7
0.8
Depth to water table (m)

- We estimated groundwater recharge and ET_g under three vegetation covers.
- Depth-dependent S_y were determined under rising and falling water table conditions.
- Lower forest recharge was due to higher interception and reduced recharge capacity.
- ET_g was controlled by meteorological drivers but mediated by depth to water table.