



Original research article

# Groundwater decline and tree change in floodplain landscapes: Identifying non-linear threshold responses in canopy condition



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## ABSTRACT

Groundwater decline is widespread, yet its implications for natural systems are poorly understood. Previous research has revealed links between groundwater depth and tree condition; however, critical thresholds which might indicate ecological 'tipping points' associated with rapid and potentially irreversible change have been difficult to quantify. This study collated data for two dominant floodplain species, *Eucalyptus camaldulensis* (river red gum) and *E. populnea* (poplar box) from 118 sites in eastern Australia where significant groundwater decline has occurred. Boosted regression trees, quantile regression and Threshold Indicator Taxa Analysis were used to investigate the relationship between tree condition and groundwater depth. Distinct non-linear responses were found, with groundwater depth thresholds identified in the range from 12.1 m to 22.6 m for *E. camaldulensis* and 12.6 m to 26.6 m for *E. populnea* beyond which canopy condition declined abruptly. Non-linear threshold responses in canopy condition in these species may be linked to rooting depth, with chronic groundwater decline decoupling trees from deep soil moisture resources. The quantification of groundwater depth thresholds is likely to be critical for management aimed at conserving groundwater dependent biodiversity. Identifying thresholds will be important in regions where water extraction and drying climates may contribute to further groundwater decline.

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## 1. Introduction

Riparian and floodplain woodland vegetation provide habitat, corridors and refugia for numerous species and play a major role in channel hydraulics, sediment transport and nutrient filtration (Naiman and Décamps, 1997). However, these

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functions may be compromised where riparian and floodplain vegetation are degraded or lost. Degradation of riparian and floodplain vegetation have been linked to a number of biophysical factors, including climate, fire, grazing and disease (Jurksis, 2005), all of which can interact in complex ways across scales. Several recent studies have shown that groundwater decline is also a key driver of change for some vegetation communities. To date, changes in groundwater have been linked to differences in canopy condition (e.g. Cunningham et al., 2011), population characteristics (e.g. recruitment and survivorship (e.g. Horner et al., 2009)) and vegetation community composition (e.g. Stromberg et al., 1996; Elmore et al., 2006), as well as decreased shoot water potential, leaf mortality and branch dieback (Cooper et al., 2003). More generally, other studies have also shown links between groundwater decline and the degradation of riparian habitats (Stromberg et al., 1992; Busch and Smith, 1995; Stromberg et al., 1996).

While the importance of groundwater and the consequences of groundwater decline for biodiversity are recognised in temperate (e.g. Harding, 1993; Fojt, 1994; Smith, 2000; Danielopol et al., 2003; Wamelink et al., 2013), tropical (e.g. Lamontagne et al., 2005) and arid climates (e.g. Fensham et al., 2011), few studies have investigated the nature of the relationship between groundwater decline and ecological response and, in particular, whether critical threshold responses (*sensu* Groffman et al., 2006) can be identified. Groundwater depth thresholds might represent 'tipping points' (*sensu* Laurance et al., 2011) beyond which the vulnerability of groundwater dependent species and communities to other water related stressors, such as drought, could substantially increase and from which recovery is uncertain (Groffman et al., 2006; Laurance et al., 2011).

The literature reflects two models of ecological response to groundwater decline in riparian and floodplain tree species: (i) a linear response, where tree condition declines across a gradient of groundwater depths (e.g. Cunningham et al., 2011; González et al., 2012); or (ii) a threshold response, where tree condition remains relatively stable until groundwater depth declines below a particular level, after which tree condition declines rapidly (e.g. Horton et al., 2001; Cooper et al., 2003). Some studies have hypothesised non-linear responses but have fitted their data using linear regression and not specifically tested for thresholds (e.g. Shafroth et al., 2000). Others have selected surrogate thresholds based on maintaining mortality rates at a defined level (e.g. < 50% in González et al., 2012). As a consequence, the question of ecological threshold responses to groundwater decline remains unresolved.

Despite the limited empirical evidence in the literature, a threshold response of riparian/floodplain tree condition to groundwater decline, whether due to climatic influences or over extraction, may be reasonably conjectured on the basis of genetic, physical and physiological constraints to tree rooting depth (Schenk and Jackson, 2002; Naumburg et al., 2005). Riparian and floodplain tree species tend to be mesic and highly dependent on access to reliable water sources including in-stream surface water, soil moisture and shallow groundwater (e.g., Thorburn and Walker, 1994; Holland et al., 2006; Stromberg and Patten, 1996). Such species may be poorly adapted to cope with moisture deficit; hence, their condition tends to be closely related to their hydrological status (Bacon et al., 1993; Cooper et al., 2003; McDowell et al., 2008). Drought and anthropogenically modified hydrological conditions may result in a range of stress induced responses in trees (e.g. Rice et al., 2004), including adaptive canopy thinning and branch sacrifice (dieback) to reduce transpiration demand and conserve hydraulic status (Tyree and Sperry, 1988; Rood et al., 2000). Extreme moisture deficit may also result in embolism, cavitation and failure in transporting xylem tissue, ultimately causing death (Tyree and Ewers, 1991; McDowell et al., 2008). Consequently, when groundwaters decline below riparian and floodplain tree rooting depths, access to a secure water resource, required for these species to persist in dry surface water environments, becomes less likely.

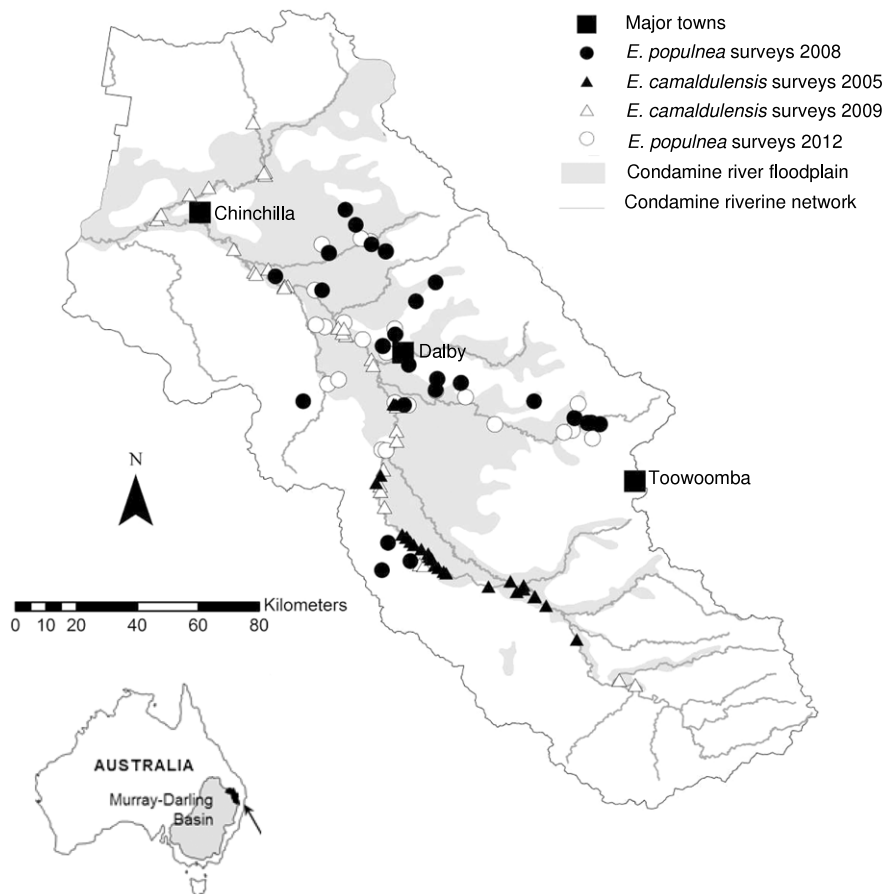
In this paper, we investigate the nature of the relationship between dominant tree condition and groundwater depth in remnant floodplain communities in an Australian agricultural landscape experiencing significant groundwater decline. Groundwater levels have declined by up to 25 m in the study region in recent decades; these declines are comparable in magnitude and extent to those documented in other parts of the world where levels of groundwater use are unsustainable (Gleeson et al., 2012). Empirical evidence for the existence and quantification of thresholds is vital to support robust decision-making aimed at maintaining ecological resilience.

Two commonly encountered riparian and floodplain tree species (*Eucalyptus camaldulensis* Denh. and *E. populnea* F. Muell.) in this landscape show evidence of poor canopy condition and significant dieback across parts of their distribution (Batterham, 2008; Reardon-Smith, 2011; Kath, 2012; Fritz, 2013). We investigate the nature of tree condition responses to groundwater decline in this landscape and use regression tree (Elith et al., 2008) and threshold identification (Cade et al., 1999; Cade and Noon, 2003; Baker and King, 2010; Kail et al., 2012) approaches to identify and quantify any non-linear (threshold) responses to groundwater depth. We also discuss the need for an evidence-based approach to defining ecological thresholds to support sustainable groundwater resource management aimed at ensuring the persistence of riparian and floodplain species and communities in areas where groundwater resources support a range of uses.

## 2. Material and methods

### 2.1. Study area

The Condamine catchment, covering approximately 24,434 km<sup>2</sup>, is located in the northern region of the Murray–Darling Basin in southern Queensland (Fig. 1). The catchment is characterised by a highly variable sub-tropical to semi-arid climate predominantly influenced by the intra-decadal dynamics of the El Niño–Southern Oscillation (ENSO) (Stone et al., 1996). Mean minimum and maximum temperatures range from  $-1.3^{\circ}\text{C}$  in July to  $33.2^{\circ}\text{C}$  in December (BoM, 2010). Long term



**Fig. 1.** Map of the Condamine catchment study area showing the extent of the Condamine riverine network, floodplain, major towns and tree condition survey sites for *Eucalyptus populnea* (floodplain sites) and *E. camaldulensis* (riparian and wetland sites).

(1890–2010) mean annual rainfall across the catchment ranges from 673 to 691 mm (BoM, 2010), but is subject to significant inter-seasonal and inter-annual variation with extended drought and intense rainfall periods common throughout the historical record (Murphy and Ribbe, 2004). Annual rainfall in the ‘wet’ La Niña phase can total 1200 mm compared to 250 mm per annum in the ‘dry’ El Niño phase of the ENSO (BoM, 2010). Stream flow variability in the Condamine River is also high and increases with distance downstream (QDEH, 2009).

The alluvial floodplain sections of the Condamine catchment and its groundwater resources are extensively developed for agriculture (~ 60% of the land area is dedicated to cropping) (QDERM, 1999; CSIRO, 2008). The development of irrigated cropping in the catchment since the 1960s has been accompanied and supported by significant surface and groundwater water resource development (Thoms and Parsons, 2003) to mitigate risk associated with the highly variable rainfall and stream flow of the region (McKay et al., 1999; Porter, 2002; Dafny and Silburn, 2014). Current groundwater extraction rates exceed estimated recharge by 38% in over 90% of years in the historical record (CSIRO, 2008). Groundwater levels in the region have been declining steadily for many years with the exception of localised areas where direct recharge occurs from the river and its larger tributaries (Porter, 2002). In some areas, declines of 15–25 m over a period of 4 decades (1967–2007) are evident (Kelly and Merrick, 2007).

## 2.2. Assessing tree condition on the Condamine floodplain

Tree condition data used in this study were derived from four recent surveys investigating the canopy condition of *E. camaldulensis* (Reardon-Smith, 2011; Kath, 2012) and *E. populnea* (Batterham, 2008; Fritz, 2013) in floodplain woodlands in the Condamine catchment. These surveys were conducted independently and in different years (from 2005 to 2012). However, a common field method was used to assess canopy tree condition, allowing the data from each to be combined for analysis. Each study used the Foliage Index (FI) method (Banks, 2006); also referred to as ‘crown vigour’ by Cunningham et al. (2007) to estimate condition for each mature tree sampled. We define mature trees as > 10 cm diameter at breast height. The FI was based on a visual estimate of the proportion of crown that contained foliage relative to a healthy tree with a full crown (after Banks, 2006). Cunningham et al. (2007) report this measure to be a simple, consistent and reliable

**Table 1**  
Description of predictor variables and their quantification methods and source.

Variables	Description and justification	Analysis procedure and notes	Data source
Lateral river connectivity index (no units)	Connectivity between the riverine system and non-riverine wetlands. This represents spatial variation in elevation and flood frequency.	Derived using a DEM and the path distance function in Arc Map version 9.2 (ESRI, 2006) (Details Appendix A)	QDNRM (2005). Digital elevation model
Upstream area of nearest stream segment (m <sup>2</sup> )	Upstream area of stream segment nearest to each site. This may correlate with longitudinal differences (e.g. river regulation) along the stream that may influence tree condition.	Calculated in Arc Map version 9.2 (ESRI, 2006). (Details Appendix A)	BoM (2012a).
Groundwater depth (m) (1987, 2000, 2005, 2009 and 2012)	Past groundwater conditions (1987) and groundwater conditions for each year coinciding with tree condition surveys.	Interpolated groundwater depths using inverse distance weighting in Arc Map version 9.2 from measurements made at bores throughout catchment.	QDERM. Groundwater data (2009).
Groundwater depth decline (m) (1987–2009).	Change in groundwater depth from 1987 to 2009. This provides a proxy for differences in rates of groundwater decline which may influence tree condition.	As above	As above
Groundwater flow system	Groundwater flow systems are associated with recharge and discharge, as well as groundwater management, both of which influence relationships between groundwater and tree condition.	The groundwater flow system underlying each site was assessed in Arc Map version 9.2. (Details Appendix A)	BoM (2012b).
Mean annual Max. Temperature (°C)	Long term mean maximum temperature (1985–2005) preceding surveys that may influence groundwater conditions.	Calculated using Australian Bureau of Meteorology mean annual rainfall data from 1987 to 2005 and the Data Visualisation and Analysis software FERRET version 6.02 (Hankin et al., 1996) (Details Appendix A). Other time periods (1980–2008, 2005–2012), were also assessed, but all were strongly correlated (Pearson $R > 0.7$ ).	BoM (2010).
Mean annual Rainfall (mm).	Long term mean rainfall (1985–2005) preceding surveys that may influence groundwater conditions.		BoM (2010).
Survey year	The year of each survey.	-	Field surveys
Tree density at site (tree/ha).	Tree density may correlate with tree condition (e.g. negatively, if competition for resources, or positively, by buffering from external disturbances).	Direct count of mature trees.	Field surveys

indicator of stand condition. This study uses the site level Average Foliage Index (AFI) as the primary response variable in the analyses conducted. Survey year and the density of trees assessed at each site were included as factors in analyses to account for the effect of observer bias and seasonal conditions and differences in the number of trees surveyed.

### 2.3. Mapping groundwater depths across the Condamine catchment

Available groundwater depth records for years coinciding with the years of tree condition surveys (i.e. 2005 (644 monitoring bores), 2008 (451 bores), 2009 (434 bores) and 2012 (467 bores)) were interpolated using inverse distance weighting in Arc Map version 9.2 (ESRI, 2006) to provide a spatially continuous estimate of groundwater depth across the catchment. A spatially continuous surface for 1987 (the earliest year for which groundwater depth data is widely available) was also derived and used to provide a proxy for groundwater depth change (calculated as the change in groundwater depth

from 1987 to 2009). Cross validations showed that interpolated rasters predicted recorded groundwater depths reasonably well across the catchment (Pearson  $R = 0.65$ ,  $R^2 = 0.43$ ).

A range of additional hydro-climatic and hydrogeological variables were also assessed and included in models to account for any influence these may have on relationships between groundwater depth and tree condition (details given in [Appendix A](#)). A summary of all predictor variables used in analysis is given in [Table 1](#).

We initially ran models with interpolated groundwater depths for all years, but due to significant correlations ( $p < 0.01$ ) between groundwater depths between years, final models included only the 2005 interpolated groundwater depths as: (i) the 2005 dataset was the most highly correlated with other years; (ii) the 2005 dataset provided groundwater depths that all tree condition surveys were exposed to (i.e. all surveys were carried out in 2005 and after); and (iii) in initial models, 2005 groundwater depth had the greatest relative importance of all groundwater variables ([Section 3.1](#)).

#### 2.4. Data analysis – Identifying and quantifying thresholds

Recent developments in statistical techniques have seen an increase in the number of methods that can be used to identify and quantify thresholds ([Kail et al., 2012](#)). However, differences between these methods may lead to uncertainty in interpretation ([Kail et al., 2012](#)). To account for any discrepancies in threshold identification and quantification that may occur with the method used, we used a range of techniques to check for consistency in any threshold values identified. Boosted regression trees (BRTs) were used to qualitatively assess relationships, while quantile regression and Threshold Indicator Taxa Analysis (TITAN) were used to quantify threshold values. Within BRTs, 'tree species' was included as a factor within the analysis, while quantile regression and TITAN threshold values were quantified individually for each species. Each of the techniques used is outlined below ([Sections 2.4.1–2.4.3](#)).

##### 2.4.1. Boosted regression tree analysis

BRTs were used to qualitatively investigate the relationship (i.e. linear or threshold) between groundwater depth and tree condition. BRTs are well suited for modelling complex functions and interactions between variables without making assumptions about the shape of the fitted functions ([De'ath, 2007](#)). Tree condition data from all studies combined were modelled as a function of groundwater depth (after [Elith et al., 2008](#)), with tree condition data arc-sine transformed to correct for boundedness ([Ahrens et al., 1990](#)). BRT analyses were performed in R version 2.15.0 ([R Development Core Team, 2011](#)) using the 'gbm' library ([Ridgeway, 2012](#)) supplemented with functions from [Elith et al. \(2008\)](#). Partial dependence plots were used to visualise the fitted functions from the BRT model ([Elith et al., 2008](#)). These plots show the influence of each variable on the response, while controlling for the average effect of other variables in the model ([Elith et al., 2008](#)). This allowed for the relationship between tree condition and groundwater depth to be assessed, while taking account of other factors (e.g. survey year, rainfall, temperature etc.). To account for residual spatial autocorrelation, which can bias parameter estimates, we followed the residual autocovariate method outlined by [Crane et al. \(2012\)](#).

##### 2.4.2. Quantile regression

Quantile regression ([Koenker and Bassett, 1978](#)) was used to assess the relationship between groundwater depth and tree condition across different parts of the probability distributions for both species. In contrast to classical linear regression techniques, which are restricted to estimating rates of change in the mean of the response variable ([Cade and Noon, 2003](#)), quantile regression allows the estimation of functional relationships between variables across the entire probability distribution ([Cade and Noon, 2003](#)). Quantile regression thus allows for differences in the relationship between a predictor and a response to be accounted for and provides estimates for multiple rates of change from the minimum to maximum response ([Cade and Noon, 2003](#)).

Quantile regression was performed on all quantiles from 0.05 to 0.95 for both *E. camaldulensis* and *E. populnea* tree condition data using the quantreg package ([Koenker and Ng, 2012](#)) in R version 2.15.0 ([R Development Core Team, 2011](#)). Bootstrapping ( $\times 1000$  replicates) was performed for each quantile, at 0.05 intervals, to identify and validate significant relationships. Quantiles showing significant relationships between groundwater depth and tree condition ( $p < 0.05$  and bootstrapped confidence intervals not including zero) were then used for threshold identification using the GUIDE software (after [Kail et al., 2012](#)). In contrast to [Kail et al. \(2012\)](#), who determined threshold values for the upper most quantile with the narrowest confidence interval only (indicative of a limiting factor), we examined thresholds for all significant quantiles to assess the full range of possible responses.

##### 2.4.3. Threshold identification

Generalised Unbiased Interaction Detection and Estimate software (GUIDE v12.6) ([Loh, 2002](#)) was used to quantify groundwater depth thresholds for each species. GUIDE identifies thresholds for quantiles of interest using a regression tree method ([Loh, 2010](#)) and allows for both quantile piecewise linear (QPL) and quantile piecewise constant (QPC) approaches. We calculated thresholds using both approaches, with QPL and QPC being robust methods for quantifying thresholds when data show wedge-shaped and stair-step relationships, respectively (after [Brenden et al., 2008](#); [Kail et al., 2012](#)). To visualise potential sharp changes in response, locally weighted quantile regression (loess-QR) was also used (after [Baker and King, 2010](#); [Koenker and Ng, 2012](#); [Kail et al., 2012](#)). For further details on quantile regression and GUIDE, please see [Loh \(2002\)](#) and [Kail et al. \(2012\)](#) for application in ecological contexts.

**Table 2**

Boosted regression model results groundwater depth in 2005 and all groundwater years together. (a) Model performance. (b) Relative importance of predictor variables. Residuals of the BRT model showed no spatial auto-correlation and were randomly distributed (Moran's Index =  $-0.05$ ,  $p > 0.05$ ).

(a) Model performance	All groundwater depth years model	Only groundwater depth (2005) in model
% Explained	28.90	26.92
Estimated CV deviance (se)	0.047 (0.004)	0.048 (0.004)
Training data correlation	0.693	0.665
CV correlation (se)	0.32 (0.055)	0.325 (0.097)
(b) Variable relative importance (%)		
Survey year	24.54	26.96
Groundwater depth (2005)	17.20	23.64
Tree density	14.03	20.00
Autocovariate	11.99	3.10
Groundwater decline (1987–2009)	6.68	9.35
Groundwater depth (2009)	5.20	n/a
Groundwater depth (2008)	4.91	n/a
Lateral connectivity index	3.74	5.90
Groundwater depth (1987)	3.24	n/a
Mean max. temperature	2.85	5.04
Upstream area	2.39	3.42
Groundwater depth matching survey year	1.22	n/a
Mean annual rainfall	1.04	2.42
Groundwater depth (2012)	0.84	n/a
Groundwater flow system	0.12	0.15
Species	0.00	0.00

n/a = not in model.

Threshold Indicator Taxa Analysis (TITAN) (Baker and King, 2010) was used to provide another threshold estimate for *E. camaldulensis* and *E. populnea* tree condition groundwater depth thresholds. TITAN utilises change point analysis, a non-parametric technique that orders and partitions observations along a gradient. TITAN works by splitting sample units into 2 groups at the value of the predictor variable (e.g. groundwater depth) that maximises association of the response of interest (e.g. tree condition) with one side of the partition.

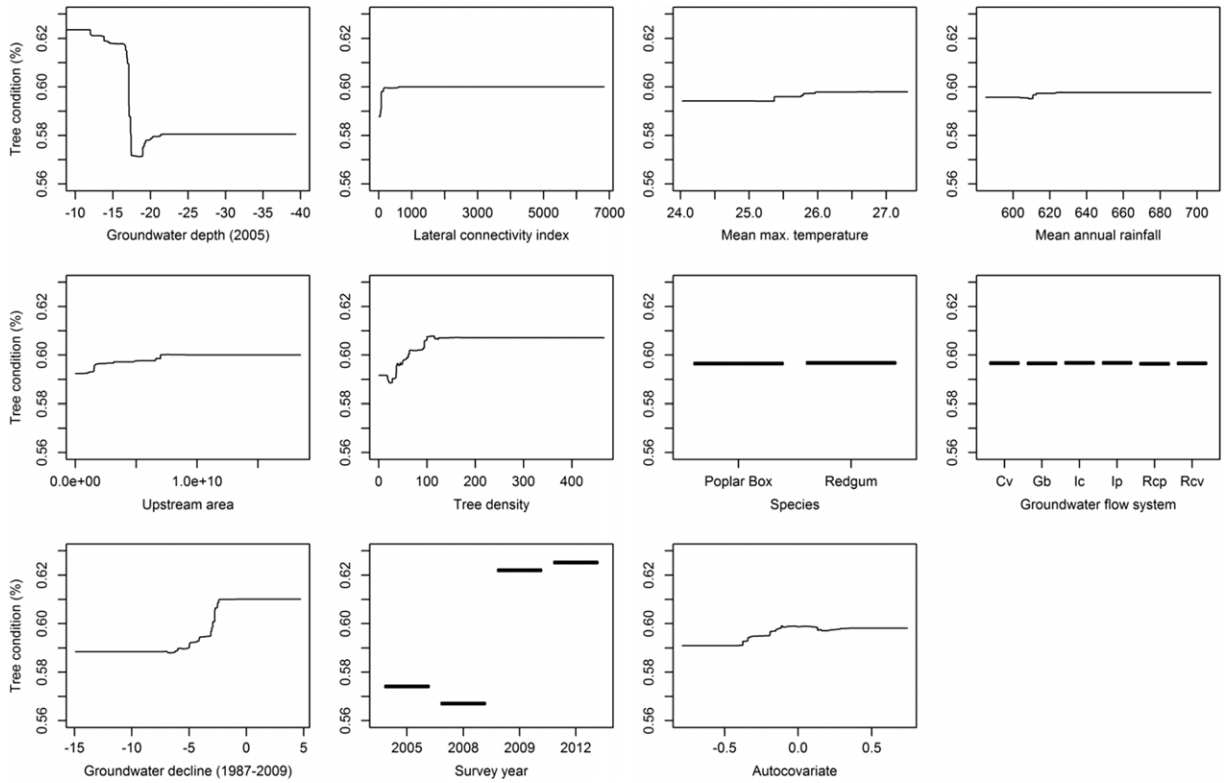
Within TITAN, bootstrapping ( $\times 1000$  replicates) was also used to assess the uncertainty around change points or threshold estimates. Uncertainty was assessed by taking the bootstrap 5th and 95th quantile change points, which provided a 90% confidence interval (CI) for the threshold value (Baker and King, 2010). Bootstrapping also provides two measures of the quality of responses, purity and reliability. *Purity* refers to the proportion of change-point directions ( $-$  or  $+$ ) within bootstrap replicates that agree with the observed response; a pure indicator is one with a consistent response direction (i.e.  $\text{purity} \geq 0.95$ ) (Baker and King, 2010). *Reliability* refers to the proportion of bootstrap replicate change points with indicator value scores (a measure of strength between a taxa and an external sample grouping) below a defined probability (e.g.  $p \leq 0.05$ ); *reliable* indicators have  $\geq 0.95$  of bootstrap replicates significant at  $p \leq 0.05$  (or some other defined level). Following Baker and King (2010), we used the cut-offs for significance criteria provided in the TITAN package (i.e. for  $p \leq 0.05$ ,  $\text{purity} \geq 0.95$  and  $\text{reliability} \geq 0.90$ ). TITAN was run with the TITAN package in R.2.9.2 (for further details see Baker and King, 2010).

### 3. Results

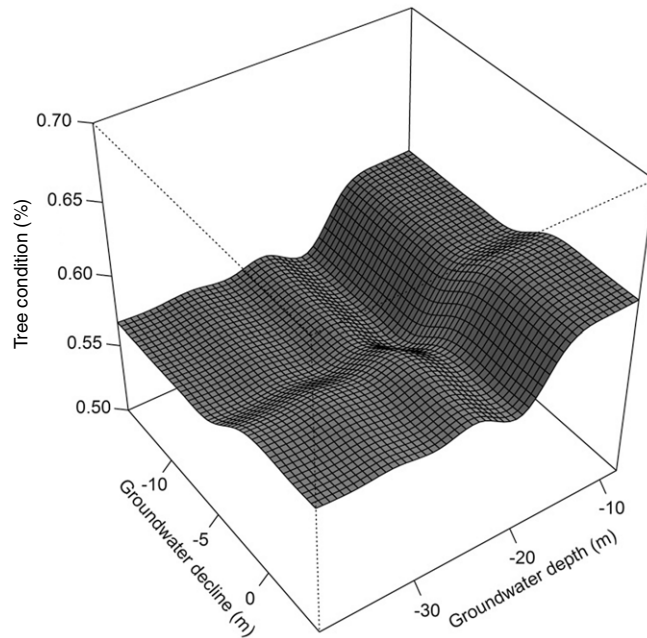
#### 3.1. Tree condition response to groundwater depth

The BRT model for 'tree condition' explained 26.9% of the variation in tree condition (CV correlation = 0.33) (Table 2). Within the model, the variable 'survey year' had the highest relative contribution to the model (27%); 'groundwater depth' explained 23.6%; 'tree density' 20.0%; and 'groundwater decline' 9.4% (Fig. 2). The remaining predictor variables all explained less than 6% of the variation within the model (Fig. 2). Models including all groundwater depth variables showed similar model performance and results (Table 2; Appendix B, Fig. B.1). Visual inspection of the partial plot of the relationship between 'groundwater depth' and 'tree condition' accounting for other predictors suggests a non-linear response of tree condition to groundwater depth, declining sharply between 12.5 and 20.0 m (Fig. 2). 'Groundwater decline' also showed an abrupt response in tree condition with groundwater declines of 2 and 5 m below 1987 levels (Fig. 2).

Interactions between groundwater depth and other predictors appeared weak, with relationships between tree condition and other variables unchanged regardless of groundwater depth (Appendix B, Fig. B.2). Some interaction between 'groundwater decline' and 'groundwater depth' was indicated, with sites in areas of equivalent groundwater decline having lower predicted tree condition when groundwater depths were lower (Fig. 3). For groundwater decline of between 0 and 5 m, tree condition was predicted to be around 10% lower in areas with groundwater depths below  $-20$  m relative to those with depths above  $-15$  m (Fig. 3).



**Fig. 2.** Boosted regression tree partial plots showing the relationship between floodplain tree condition (*E. camaldulensis* and *E. populnea* combined), and groundwater depth (m) and other predictor variables in the Condamine catchment. See Table 1 for variable units and details and Appendix B for description of variables and groundwater flow system abbreviation meaning.



**Fig. 3.** Boosted regression tree interaction plot between groundwater depth (m) and groundwater decline (m).

**Table 3**  
*E. camaldulensis* and *E. populnea* quantile regression results with threshold values for significant quantiles from GUIDE.

Quantile	Slope	95% CI <sup>a</sup>	Bootstrapped <sup>b</sup>			GUIDE Groundwater depth (m) threshold	
			Mean	se	p-value	QPL <sup>c</sup>	QPC <sup>d</sup>
<i>E. camaldulensis</i>							
0.200	0.035	0.007–0.039	0.035	0.013	0.009	–12.45	–17.16
0.250	0.036	0.011–0.044	0.036	0.012	0.004	–12.45	–17.16
0.300	0.035	0.009–0.047	0.035	0.012	0.003	–12.45	–17.16
0.350	0.032	0.017–0.048	0.032	0.011	0.005	–12.45	–17.16
0.400	0.029	0.021–0.042	0.029	0.011	0.011	–17.16	–17.16
0.450	0.029	0.015–0.046	0.029	0.012	0.021	–17.16	–17.16
<i>E. populnea</i>							
0.650	0.012	0.005–0.017	0.012	0.005	0.023	–18.95	–16.82
0.700	0.011	0.009–0.018	0.011	0.005	0.039	–15.63	–16.82
0.750	0.012	0.001–0.019	0.012	0.005	0.038	–15.63	–13.86
0.900	0.019	0.005–0.021	0.019	0.008	0.020	–21.95	–13.86

<sup>a</sup> CI = confidence interval.<sup>b</sup> Bootstrap resample = 1000, se = standard error.<sup>c</sup> QPL = quantile piecewise linear.<sup>d</sup> QPC = quantile piecewise constant.

### 3.2. Groundwater depth thresholds for tree condition

Groundwater depth was significantly associated ( $p < 0.05$ ) with quantiles 0.2 through to 0.45 for *E. camaldulensis* (Table 3) and significantly associated ( $p < 0.05$ ) with quantiles 0.65, 0.70, 0.75 and 0.9 for *E. populnea* (Table 3). All significant quantiles for *E. camaldulensis* and *E. populnea* showed declining tree condition as groundwater depth declined (Table 2). Results for all quantiles and quantile plots are given in Appendix B (Tables. B.1, B.2. and Fig. C.3). Quantiles 0.2, 0.25 and 0.3 showed the greatest rate of decline for *E. camaldulensis* condition in relation to groundwater depth with slopes of 0.035–0.036 (Table 2). *E. populnea* condition decline was greatest in relation to groundwater depth for quantiles 0.9 and 0.95, with both having a slope of 0.019 (Table 3). Residuals for all *E. camaldulensis* and *E. populnea* quantile regression models showed no spatial auto-correlation and were randomly distributed (Moran's Index = 0 – 0.01,  $p > 0.05$ ).

Groundwater depth thresholds derived from GUIDE software for significant quantiles ranged from 12.5–17.2 m for quantile piecewise linear (QPL) and 17.2 m for quantile piecewise constant regression (QPC) for *E. camaldulensis* (Table 3) and from 15.6–22.0 m (QPL) and 13.9–16.8 m (QPC) for *E. populnea* (Table 3). Visualised thresholds for these quantiles for locally weighted quantile regression were at approximately 12 m for quantile 0.3 and 18 m for quantile 0.4 for *E. camaldulensis* (Fig. 4a, Fig. 5b, respectively). *E. populnea* thresholds were approximately 19 m and 18 m for quantiles 0.65 and 0.75, respectively (Fig. 4c, d). Threshold responses for quantile 0.9 for *E. populnea* were less pronounced, but suggest a change at around 21 m (Fig. 4e).

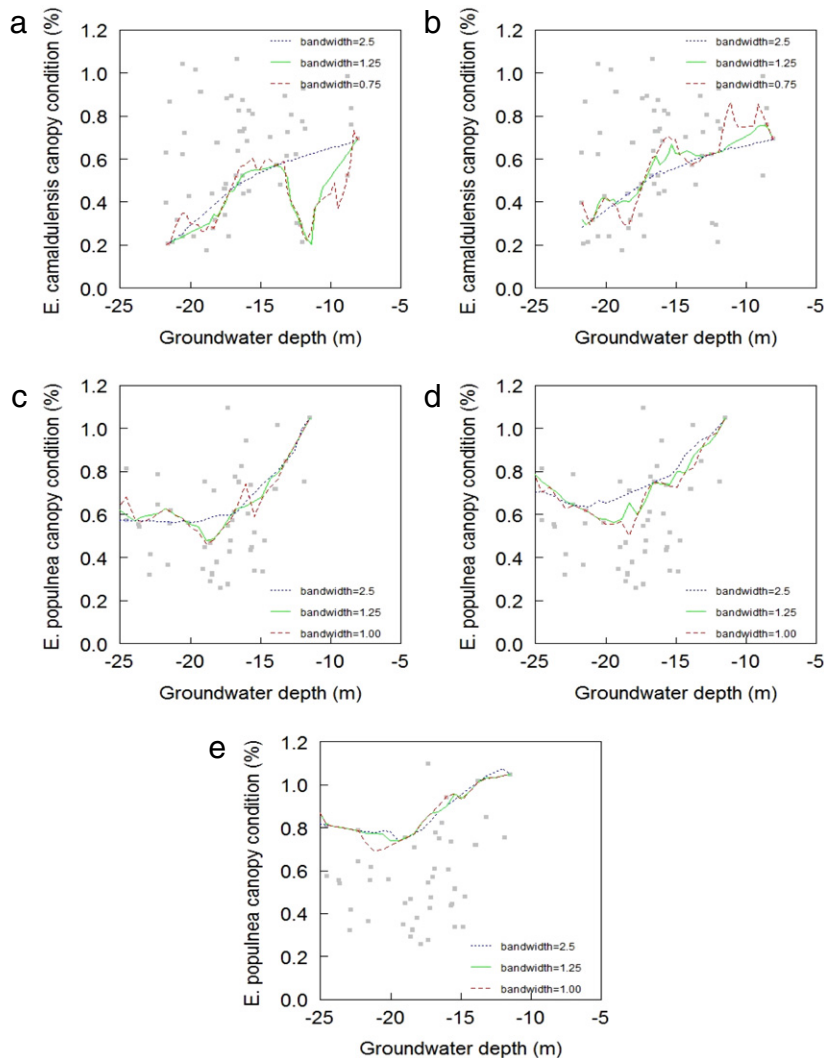
### 3.3. TITAN groundwater depth estimation

TITAN estimated overlapping groundwater depth thresholds of 17.5 m (90% confidence interval, 12.5–20.8 m) for *E. camaldulensis* and 14.3 m (90% confidence interval, 14.33–26.5 m) for *E. populnea*. Estimates for both species had similar purity and reliability scores ( $>0.95$  and  $\geq 0.92$ , respectively). TITAN threshold estimation 90% confidence intervals also overlapped regardless of which groundwater depth year data was assessed, with all estimates falling between 12.1 and 22.6 m for *E. camaldulensis* and 12.6 and 26.7 m for *E. populnea* (Appendix B, Table. B.3).

## 4. Discussion

Previous studies have suggested groundwater depth thresholds may be present at which critical declines in riparian/floodplain tree condition and survivorship occur (e.g. Shafroth et al., 2000; Horton et al., 2001). However, few studies have tested or attempted to identify groundwater depth thresholds for such species. Furthermore, few studies have investigated tree condition responses in a region subject to chronic groundwater decline (defined here as a persistent fall in groundwater levels over time) and exhibiting a range of groundwater depths. Horton et al. (2001) found threshold responses in the canopy condition of three dominant riparian tree species to seasonal groundwater decline in drier than normal years in the Hassayampa River Reserve, Arizona, but groundwater recovery in normal–wetter years suggests only short-term impact on population processes and ecosystem function. Similarly, Stromberg et al. (1996) and Shafroth et al. (2000) report critical responses in riparian vegetation to groundwater depth fluctuations in response to seasonal rainfall variation. In western China, restoring groundwater to more favourable levels following a period of unsustainable extraction also resulted in recovery of riparian tree canopy condition (Chen et al., 2008). In contrast, groundwater levels on the Condamine floodplain show limited response to major rainfall periods or to reduced entitlements through groundwater licencing arrangements (Reardon-Smith, 2011); hence, the potential for groundwater depth recovery in this system is limited or likely slow. Chronic





**Fig. 4.** Locally weighted quantile regression plots for *E. camaldulensis* quantiles (a) 0.3 and (b) 0.4 and *E. populnea* quantiles (c) 0.65, (d) 0.75 and (e) 0.9. Different bandwidths represent different sensitivities to change. An intermediate bandwidth resulting in the sharpest change was selected visually and two additional bandwidths were included to show that lower bandwidths result in a curve strongly influenced by single data points, while higher bandwidths are less sensitive and mirror the linear quantile regression line (after Kail et al., 2012).

groundwater declines such as those occurring in the Condamine are not an isolated case, with vegetation in many other parts of the world exposed to similarly persistent groundwater declines (Gleeson et al., 2012). As such, vegetation responses to groundwater decline in the Condamine have significant global relevance.

Results from this study suggest non-linear threshold responses for both *E. camaldulensis* and *E. populnea* tree condition to groundwater depth, identifying thresholds of between 12.1 and 22.6 m for *E. camaldulensis* and between 12.6 and 26.7 m for *E. populnea*. The identified thresholds were consistent for all methods used, with TITAN estimates overlapping those derived through GUIDE using both quantile piecewise linear and quantile piecewise constant approaches. These values also concurred with visually estimated thresholds from boosted regression trees (~12–20 m) and locally weighted quantile regression (~12–22 m). These groundwater depths corroborate proposed thresholds of 12–15 m for *E. camaldulensis* mortality in floodplain plantation forests on the Murray River floodplain, south-eastern Australia (Horner et al., 2009) and 13–16 m for *E. camaldulensis* site-level dieback severity in riparian woodlands on the Upper Condamine floodplain (Reardon-Smith, 2011). They are also consistent with the estimates of maximum lateral root extension in this and similar species reported by Mensforth et al. (1994) and Canadell et al. (1996). Modelled groundwater depth thresholds for *E. populnea* condition are the first such evidence for this species, although groundwater depth was reported by Fritz (2013) to be a strong correlate of tree condition in this species.

Evidence of significantly poorer canopy condition in trees occurring where groundwater depths exceed the identified thresholds supports the contention that access to groundwater provides a critical resource for these species, particularly in the absence of alternative moisture sources (e.g. during drought). The depth to which tree roots can access and effectively

draw water in the absence of alternate sources may therefore indicate a threshold at which such stress responses become critical and provides a mechanism for the hypothesis that there is a threshold response of tree condition to groundwater decline. While maximum rooting depths in *E. camaldulensis* and *E. populnea* are unknown, critical groundwater depths identified in this study may represent a functional physiological limit to root growth. As suggested by Naumburg et al. (2005), groundwater decline may decouple tree roots from accessible moisture resources contributing to moisture deficit and impacting tree condition, particularly during extended drought conditions when surface soil moisture levels are depleted. Although, several studies have noted strong links with groundwater, the quality of this water is likely to be critical; in areas of high groundwater salinity, trees may rely more on deep soil moisture infiltrating after rainfall which may lie above the more saline groundwater (e.g. Holland et al., 2006). Generalisations about vegetation's dependence on groundwater and related thresholds should therefore be made cautiously, as there may be important exceptions (e.g. Holland et al., 2006). Isotope studies would confirm the source(s) of water used by trees and are likely to be an important area of future research. Nonetheless, if the tree condition-groundwater depth thresholds identified for *E. camaldulensis* and *E. populnea* in this study are associated with physiological limits to rooting depth for reaching groundwater and such limits apply to other species, then declining groundwater depths may have wide-ranging implications for the future condition, function and persistence of trees in environments where conditions such as extreme drought may increase vegetation's dependence on groundwater.

If riparian and floodplain tree species commonly respond to groundwater depth thresholds, improved understanding of this relationship will have significant implications for biodiversity conservation and resource management elsewhere in the world where chronic groundwater decline is an issue (e.g. in much of Africa, the Americas, southern Europe, the Middle East, Southeast Asia and Australia; Dai, 2011). Riparian and floodplain vegetation support a wide range of ecosystem functions that would likely be impaired by changes in the condition and survivorship of key species sensitive to groundwater depletion (Naiman and Décamps, 1997). For example, population failure in dominant tree species could trigger cascading changes resulting in altered community composition and structure, which in turn could cause declines in other species (e.g. avifauna Betts et al., 2010). In the study area much of the mapped *E. camaldulensis* and *E. populnea* vegetation occurs in areas in which groundwater depths exceed the identified thresholds (Appendix C, Fig. C.1). Such spatial representations could provide important starting points for managers to identify ecosystems that may be at high risk from groundwater decline.

More specifically, if the tree condition-groundwater depth thresholds identified for *E. camaldulensis* and *E. populnea* in this study are associated with physiological limits to rooting depth and such limits also apply to other species, it is probable that declining groundwater depths across many regions of the world (as reported by Gleeson et al., 2012) will have wide-ranging ecological implications. In environments where dry climatic extremes exacerbate both societal and vegetation dependence on groundwater, knowledge of rooting depths could provide important indications of where thresholds may occur. However, whilst rooting depth is a plausible mechanism, rooting depth is difficult to ascertain directly; hence, threshold responses to groundwater decline remain difficult to verify. Although isotopic studies are able to identify the sources of water being utilised by vegetation (e.g. Dawson and Ehleringer, 1991), this is a resource intensive approach and difficult to execute over large spatial scales (e.g. entire forests or catchments). Hence, threshold identification, such as implemented in this study, provides a valuable approach to indirectly derive such information.

In combination with broader regional scale modelling approaches, fine scale physiological approaches will be needed to improve confidence about the mechanisms behind threshold responses to groundwater decline. Finer scale studies would also allow investigation of the relative importance of groundwater depth and the rate of groundwater change, as trees may be affected not only by declines in absolute depths but also the rate of decline (e.g. Shafroth et al., 2000). Variability in groundwater levels could potentially be important, especially in areas where seasonal irrigation activities may cause water tables to fluctuate above and below thresholds over time. Although, we focused on groundwater, in some instances, deep soil moisture may also be an important fine scale determinant of tree condition, particularly in areas of high groundwater salinity (e.g. Holland et al., 2006). In addition, measures of tree size and estimations of root length may also help better explain some of the patterns observed; tree size can correlate strongly with the water source being used (e.g. Dawson and Ehleringer, 1991) and younger trees with shallow roots may show threshold responses at different depths to those of larger trees. Incorporation of these factors would likely improve both fine and broad scale predictions and provide important information for targeting conservation activities at different scales. In combination, information about the sources of water being derived (e.g. Dawson and Ehleringer, 1991) and knowledge of rooting depths (e.g. Stromberg, 2013) in addition to statistically derived thresholds (as in this study) could provide robust grounds for conserving groundwater dependent biodiversity at both local and regional scales.

While threshold responses to groundwater decline were identified in this study, groundwater decline is not the only factor influencing tree condition in landscapes such as the Condamine catchment. Tree condition represents an integrated response to a range of environmental factors including defoliating insects, disease and pesticides (e.g. Wylie et al., 1992; Jurskis, 2005; Overton et al., 2006; Davidson et al., 2007; Reid et al., 2007) and the impact of groundwater depth thresholds should be considered alongside these. These additional factors may also help better explain the additional unexplained variation in tree condition in the current study. Further, an understanding of the interactions between finer scale drivers of tree condition, such as insect attack, and groundwater decline could also be important. In some instances, synergistic interactions across scales might be identified, providing important insight into how multiple stressors may drive declining tree condition and ecosystem change.

## 5. Conclusions

The ability to quantify critical groundwater thresholds for canopy condition in riparian/floodplain tree species provides valuable evidence for biodiversity conservation and resource use, improving our ability to predict the impact of groundwater depletion on riparian and floodplain ecosystems and to identify safe operating limits for sustainable water resource management in these regions. In this paper, we have identified clear threshold responses for *E. camaldulensis* and *E. populnea* canopy condition in a highly modified agricultural production landscape subject to chronic groundwater decline. We have presented an approach which allows for a relatively rapid assessment of groundwater–tree condition relationships over large spatial scales and contributes to addressing significant knowledge gaps in understanding groundwater decline impacts on ecosystems. Given widespread groundwater decline globally, and predictions that many semi-arid regions will experience increased incidence of drought under climate change (Dai, 2011), the susceptibility of riparian and floodplain tree species to drought, in combination with groundwater decline, will be an important issue for biodiversity conservation into the future. The findings and approaches outlined in this paper will inform policy and management to address these challenges and to better understand the broader ecological implications of groundwater decline.

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## Appendix A, B and C. Supplementary data

Supplementary material related to this article can be found online at <http://dx.doi.org/10.1016/j.gecco.2014.09.002>.

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