Predicted risks of groundwater decline in seasonal wetland plant communities depend on basin morphology

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Abstract

In regions of the world where the climate is expected to become drier, meeting environmental water needs for wetlands and other dependent ecosystems will become increasingly challenging. Ecological models can play an important role, by quantifying system responses to reduced water availability and
predicting likely ecological impacts. Anticipating these changes can inform both conservation and monitoring effort. We used water-plant functional group models to predict the effects of a declining water table for two wetland types reliant on the surface expression of groundwater but of contrasting basin morphology. Our interest was in quantifying the relative sensitivity of these wetland types to different amounts of groundwater decline. For the shallower, grass-sedge wetland, terrestrial plant probabilities increased markedly for declines between 0.25 and 0.5 m, but amphibious and submerged functional groups changed predictably, or not at all. However, mean inundated area reduced by over 70% for a 0.5 m groundwater decline, suggesting loss of area posed the greatest risk in this wetland type. In the deeper, steep-sided interdunal wetland, inundated area changed little, but models suggest clear transitions in plant functional group composition. Sedge-group probabilities increased sharply for declines between 0.25 and 0.5 m, while declines between 0.5 and 1.0 m predicted the loss of submerged species. As might be anticipated, the risks associated with groundwater level decline depend on basin morphology. However, by quantifying probable ways in which this will manifest in different wetland types, model predictions improve our ability to recognise and manage change.

Keywords

Groundwater-dependent ecosystem
Plant functional group
Predictive model
Wetland bathymetry
Wetland monitoring
Wetland typology

Electronic supplementary material

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Introduction

As climatic uncertainty increases, so does the need for ecological predictions to anticipate environmental impacts to inform planning and decision-making (Clark et al. 2001; Sutherland 2006). This is particularly poignant for aquatic ecosystems, where escalating human pressure on water resources will compound any increase in climatic stress (Dudgeon et al. 2006; Vörösmarty et al. 2000).
Wetlands are highly susceptible to impacts from water resource development (Brinson and Malvarez 2002), and the estimated two-thirds reduction in global wetland area occurring since 1900 is mainly attributed to the direct actions of humans (Davidson 2014). Wetland plant communities are effective indicators of overall wetland condition Spencer et al. (1998), so understanding how wetland vegetation will change with water availability could help to plan effective management options. Correlative ecological models can predict these changes in aquatic plant species (Auble et al. 2005; Booth and Loheide 2012), but water resources planning usually applies over spatial scales that encompass large numbers of species and wetland types. To make predictions that can inform management at these scales requires a means to generalise responses in vegetation across diverse wetlands that differ in their species composition; the use of plant functional groups is one means to achieve this (Casanova 2011; Deane et al. 2017b; Merritt et al. 2010).

Functional groups classify species based on the life-history, morphological or reproductive traits that pre-dispose them to respond reasonably consistently to changes in an environmental condition of interest (Merritt et al. 2010). For example, the water plant functional groups of Brock and Casanova (1997) were originally developed to relate plants of seasonal wetlands in Australia to different depths and durations of surface inundation. They have been widely used in a range of studies relating wetlands to water regime (Casanova and Brock 2000; Johns et al. 2015; Raulings et al. 2011), in quantifying wetland biodiversity (Deane et al. 2016) and for water allocation (Casanova 2011). Statistical models of plant functional groups can be used to project the likely distribution of each group in response to changes in hydrology (Deane et al. 2017b; Merritt et al. 2010). Once the functional group response is understood for a given hydrological scenario, this can then be related back to the species comprising each classification for regional water resource planning applications (Merritt et al. 2010).

A further source of variability for environmental planners is the hydrogeomorphic differences among wetlands, which along with vegetation are commonly used to classify wetlands into typologies (e.g., Brinson 1993; Cowardin et al. 1979). Grouping wetlands into typologies is often a priority task for regional wetland managers, because the behaviour of wetlands within a given type should be more similar than between them. For example, plant functional groups have a similar average response to water regime among wetland types, but can differ in their prevalence along a salinity gradient (Deane et al. 2017b). Wetland typologies help account for such variations. In addition to the influence of water quality, typologies will also reflect changes in water regime due to different combinations of basin morphologies, substrates, and hydrological
conditions. Quantifying differences in the susceptibility of different wetland types to changes in water availability is critical for understanding and ameliorating any potential ecological impacts.

Correlative models can be used to quantify wetland responses to changes in water availability, provided predictions are within the range of variability in the hydrological variables used to build the models Sutherland (2006). Their use in modelling novel conditions, such as those predicted under long-term climate projections, is not reliable and instead requires mechanistic models (Purves and Pacala 2008; Sutherland 2006). However, human impacts on water dependent ecosystems are predicted to exceed any impacts related to climate change up until at least 2025 (Vörösmarty et al. 2000) and potentially until 2050 (Grouillet et al. 2015). Correlative models are still of use in contemporary resource management for predicting short to medium-term changes that will occur for incremental changes in hydrology within the current range of variability.

Our aim in this study was to estimate the likely ecological change in wetland vegetation that would occur at different levels of groundwater drawdown. We used water plant functional group models (Deane et al. 2017b) to quantify changes in vegetation and accounted for differences in sensitivity among wetland types by building models based on two contrasting basin morphologies. We selected groundwater lowering scenarios that were within the range of current variability, yet still relevant to current water allocation decision-making in the region. Models highlight clear differences in response between the two wetland types that warrant different monitoring and planning considerations.

Methods

Study region

This study was based in the South East of South Australia, an area transformed by water resource development over the last 150 years. This began in the 1800s, with the creation of an extensive network of surface water drains intended to lower water tables and thereby increase the area available to agriculture (Harding 2012). It is estimated today that less than 6% of the pre-European wetland area persists; yet wetlands still number over 16,000 and include nationally and internationally significant sites (Environment Australia 2001; Harding 2012). Water for irrigation and domestic supply in the South East is drawn from a regional unconfined aquifer (Tertiary Limestone Aquifer-TLA) that varies in thickness from 10 to 300 m (Brown et al. 2001). Depths to groundwater are generally shallow and most of the region’s wetlands (77% by number; 96% by area) are dependent upon surface expression from this groundwater system to maintain seasonal wetland habitat (SKM 2009). Many
wetlands are considered sensitive to even small declines in groundwater level (Cook et al. 2008; DFW 2010; Harding et al. 2015). Climate projections for the region differ in the magnitude and pace of change, but agree on the direction, with reduced annual and spring rainfall totals expected (Charles and Fu 2014). Long-term climate data bear out these projections, with statistically significant declining trends in annual and winter rainfall already evident (Chowdhury et al. 2015). Environmental concerns around the drying climate are further complicated by the likelihood of increased groundwater extraction (McFarlane et al. 2012).

Simulated wetlands

Using an existing wetland typology for the region (Butcher et al. 2011), we selected two wetland types of contrasting geometry and basin morphology (where morphology refers to such characteristics as maximum depth and slope of the wetland surface): Grass-sedge wetlands tend to be of a shallow, gently sloping profile, while inland interdunal wetlands are deeper with steeper sides and a flat base. Both types are prevalent across the region and between them represent ~ 75% of wetlands with a high likelihood of groundwater dependence (SKM 2009).

To select basins that were characteristic of each of the wetland typologies for modelling, we first analysed the natural variation in wetland basin characteristics. We used a LiDAR (light detection and ranging) derived 2 m digital elevation model (DEM; vertical accuracy ± 0.15 m) for the South East region (Wood and Way 2011) and the regional wetland polygon layer within a geographical information system to calculate zonal summary statistics (maximum depth, mean slope and total area; Table 1). For each wetland type we then selected around 20 wetlands with values for all statistics that were close to the mean values in all basin morphology measures for that typology. We then manually filtered these representative wetlands to remove basins where physical disturbances such as excavation or drains had altered the natural basin morphology, or where the digital elevation model was affected by dense overhanging vegetation or the presence of standing water at the time of LiDAR capture. While LiDAR is not able to detect the elevation of the wetland soil surface in the presence of water, this affected less than 2% of wetlands and is unlikely to have influenced the calculation of summary statistics. The final two wetland basins were chosen as generally representative of unaltered basin morphology distinct to the two types. The regional DEM was then clipped to the chosen basin geometry and the elevations were used to estimate inundation for modelling. Grass-sedge wetlands were represented by a small (~ 5 ha), irregular shaped, shallow basin with mean and maximum depths of approximately 0.5 and
0.8 m respectively. Inland interdunal wetlands were represented by a larger (~ 15 ha), roughly circular deflation basin with mean and maximum depth values of approximately 0.8 and 1.2 m (Fig. 1).

Table 1

Basin morphology summary statistics from calculated from a regional digital elevation model (Wood and Way 2011)

<table>
<thead>
<tr>
<th>Wetland type</th>
<th>Grass-sedge (n = 7581)</th>
<th>Inland interdunal (n = 413)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Average depth (m)</td>
<td>Max depth (m)</td>
</tr>
<tr>
<td>Mean</td>
<td>0.5</td>
<td>0.9</td>
</tr>
<tr>
<td>SD</td>
<td>0.7</td>
<td>1.2</td>
</tr>
</tbody>
</table>

‘Grass-sedge’ and ‘Inland interdunal’ refer to wetland types under the typology of (Butcher et al. 2011) for wetlands in the South East region of South Australia

Fig. 1

Basin morphology for the two modelled wetland types derived from the digital elevation model a grass-sedge; b inland interdunal basins. Contours show 0.1 m increments in elevation, note different axis scales

Plant functional groups

The plant functional group system on which models were based was developed in Australia to detect wetland plant responses to surface water inundation and classifies plants into three broad categories (Brock and Casanova 1997): terrestrial (intolerant of flooding), amphibious (tolerates or responds to flooding and drying) and submergent (intolerant of desiccation). Deane et al. (2017b) then
modelled eight different groups based on sub-divisions of the above categories: two terrestrial (Tdry—terrestrial dry; Tdamp—terrestrial damp), four amphibious (Af1l—amphibious fluctuation tolerator low-growing; Aftw—amphibious fluctuation tolerator woody; Afte—amphibious fluctuation tolerator emergent; Afrp—amphibious fluctuation responder plastic growth) and two submerged groups (Se—submerged emergent; Sr—submerged r-selected). The relative depth-duration tolerance for each plant functional group modelled along with a list of species in the study region found in Table 2. For further information refer to Brock and Casanova (1997), Casanova (2011) and Deane et al. (2017b).

**Table 2**

Plant functional group water regime preferences, model structures and coefficients

<table>
<thead>
<tr>
<th>Functional group name (code)</th>
<th>Water regime preference</th>
<th>Model structure and coefficient values (^{a})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Terrestrial dry (Tdry)</td>
<td>Will not tolerate inundation and tolerates low soil moisture for extended periods.</td>
<td>$-8.0$ to $5.28 \times \text{MIs}$</td>
</tr>
<tr>
<td>Terrestrial damp (Tdamp)</td>
<td>Will tolerate inundation for short periods (&lt; 2 weeks) but require high soil moisture</td>
<td>$-2.29$ to $3.26 \times \text{HP} + 1.99 \times \text{SI} - 4.39 \times \text{MI} - 3.71 \times \text{I}$</td>
</tr>
<tr>
<td></td>
<td>throughout their life cycle.</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Functional group name (code)</th>
<th>Water regime preference</th>
<th>Model structure and coefficient values (^{a})</th>
</tr>
</thead>
<tbody>
<tr>
<td>Amphibious fluctuation tolerators-low</td>
<td>Fluctuating water levels, plants do not respond morphologically to flooding and</td>
<td>$-2.74$ to $0.16 \times \text{HPs} - 1.29 \times \text{MIs}$</td>
</tr>
<tr>
<td>growing (Aftl)</td>
<td>drying and are generally small herbaceous species.</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Amphibious fluctuation tolerators-</td>
<td>Fluctuating water levels, plants do not respond morphologically</td>
<td>$-3.40 + 1.92 \times \text{HPs} - 5.02 \times \text{MIs}$</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Functional group name (code)</td>
<td>Water regime preference</td>
<td>Model structure and coefficient values</td>
</tr>
<tr>
<td>-----------------------------</td>
<td>-------------------------</td>
<td>----------------------------------------</td>
</tr>
<tr>
<td>woody (Aftw)</td>
<td>to flooding and drying and are large perennial woody species.</td>
<td>2.66 + 1.26 × SI − 2.09 × MI − 1.098 × SI:MI</td>
</tr>
<tr>
<td>Amphibious fluctuation tolerators-emergent (Afte)</td>
<td>Fluctuating water levels, plants do not respond morphologically to flooding and drying and will tolerate short-term complete submergence (&lt; 2 weeks).</td>
<td>0.41 + 0.09×MIs + 0.73×Siqr − 0.62 × HPs − 0.60 × M</td>
</tr>
<tr>
<td>Amphibious fluctuation responders-plastic (Afrp)</td>
<td>Fluctuating water levels, plants respond morphologically to flooding and drying (e.g. increasing above to below ground biomass ratios when flooded).</td>
<td>0.01−0.50 × HP − 0.53 × SI + 2.42 × MI</td>
</tr>
<tr>
<td>Submergent r-selected (Sr)</td>
<td>Common in temporary wetlands that hold water for longer than 4 months.</td>
<td>− 2.79 + 2.42 × HPs + 2.95 × Siqr</td>
</tr>
<tr>
<td>Submerged-Emergent (Se)</td>
<td>Static shallow water &lt; 1 m or permanently saturated soil.</td>
<td></td>
</tr>
</tbody>
</table>

Coefficients represent predictions for moderate electrical conductivity (~ 1–6 m-Scm⁻¹), commonly found in the region (Deane et al. 2017b). Models predict the log-odds of obse the functional group for the water regime predictor values calculated over the antecedent 3-year period.

For a full description of model predictors, model building and validation and general res curves see Deane et al. (2017b). Water regime preferences based on Casanova (2011).

*MIs the mean maximum depth during the spring months (Sept–Nov), HP mean annual hydroperiod (proportion of days-per-year that water levels are at, or above, ground level mean annual sum exceedance value (the sum of depths of water greater than zero each yr). MI maximum depth observed for a continuous 14 days, HPs as for HP, but calculated over spring months only, Siqr the interquartile range of depths in spring months. All variables standardised using the distribution of hydrological predictor values used in model buildi
Groundwater scenarios

To protect groundwater dependent ecosystems in the South East region, water allocation plans utilise set-back distances for new groundwater use in the vicinity of wetlands identified as being of high ecological value SENRMB (2013). This is based on an assessment of perceived level of risk, which is evaluated using trends in groundwater level over a 5 year reporting period. The acceptable limit for all high-value groundwater dependent ecosystems is set at a mean (arithmetic) decrease in groundwater level of 0.05 m/y, with 0.1 m/y accepted as a region-wide trigger elsewhere SENRMB (2013). We based our modelled scenarios on the total change in groundwater level that would occur over a single 5-year period at the maximum limit of the acceptable groundwater decline cited in the water allocation plan, corresponding to 0.25 m and 0.5 m for high-value and other wetlands respectively. We also modelled a groundwater decline of 0.1 m in the shallow grass-sedge wetland and a 1 m decline in the deeper interdunal wetland to bracket these limits. Owing to the observed drying trend and consistent climate projections for lower annual and seasonal rainfall totals we did not consider any scenarios where groundwater levels increased.

Modelling wetland hydrological change

The plant functional group models require hydrological predictors based on surface water inundation statistics calculated over a 3--years antecedent period (Deane et al. 2017b). We therefore needed a representative hydrograph from wetlands in the region that could be used to relate changes in groundwater level under our lowering scenarios to changes in surface water inundation. To do this we used empirical relationships between surface and groundwater (Chambers et al. 2013; Harding et al. 2015), collating groundwater and surface-water logged water depth data from eight wetland basins with known groundwater dependence (SKM 2009).

We corrected these to observed manual readings and took 14--days running averages to remove the influence of isolated rainfall events, which increase short-term variability in surface water levels. All sites had strong linear relationships between surface and groundwater dynamics (mean ± SE for slope parameter = 0.89 ± 0.28; mean $R^2 = 0.87$; Online resource 1, Appendix S1). We selected the site where water level variation in the regional tertiary limestone aquifer explained the greatest amount of variation in wetland surface water levels ($R^2 = 0.96$) using simple linear regression to relate surface and groundwater levels where: Surface water level = groundwater level × 0.75, with the intercept of the regression adjusted to give the full supply level referenced to
the lowest elevation for each modelled basin under current water level variation (0.8 and 1.2 m for the shallow and deep wetlands respectively). To calculate a surface inundation hydrograph for each groundwater lowering scenario, we first subtracted the drawdown amount from the daily groundwater hydrograph and then used the regression equation to predict surface water level variations (Fig. 2).

Fig. 2
Modelled surface water level scenarios. Each hydrograph is a linear transform of the 3 year baseline groundwater hydrograph with a slope value of 0.75.

This value was then subtracted from the elevation of each cell in each wetland-type model to calculate surface water predictors based on daily values of the time series above this elevation. This means that we assumed a flat and level water surface across each wetland basin and any daily surface water level that
fell below the elevation of the wetland grid cell was assigned a value of zero. Hydrological variables differ among the plant functional group model, but include hydroperiod and maximum depth calculated over annual and spring seasons. Table 2 shows the model structures and describes the water regime predictors, while detailed descriptions of model building and validation are described in Deane et al. (2017b).

As the functional group models use only surface inundation metrics, we were limited to drawdown scenarios that remained above the surface at least part of the three-year period used to calculate the predictors. As discussed above, for both wetlands we modelled 0.25 and 0.5 m declines in groundwater while for the deeper interdunal wetland, we added a 1.0 m decline scenario as it remained inundated across most of the wetland area. For the shallow grass-sedge wetland, we added an additional 0.1 m drawdown scenario to test fine-scale sensitivity to drawdown. The maximum year-to-year variation over the period of record at the site used in calculating the hydrographs was 0.58 m, so the 1 m scenario is technically an extrapolation beyond the observed variability in surface and groundwater data and less reliable. However, the constant linear relationship observed at all sites held for inter-annual differences in surface and groundwater levels of 0.87 m at a similar site, suggesting the 1.0 m scenario likely provides a valid prediction of the change in surface water level.

Modelling

We used the fixed effects from generalised linear mixed-models with binomial error structures, built and validated for eight plant functional groups in the region (Deane et al. 2017b) to predict their probability of presence. Deane et al. (2017b) modelled plant functional groups probabilities within three salinity classes. We used the models for ‘moderate’ salinity levels (varying between ~ 1000 and 6000 μS·cm⁻¹) as it supports all functional groups and is widely representative of wetlands in the region. Wetlands of low salinity should respond similarly to model predictions, but hyper-saline wetlands differ strongly in the prevalence of many functional groups and are therefore not represented.

We report the mean predicted probability for each functional group within each wetland type for each groundwater scenario. We quantified uncertainty for each of these predictions using 95% confidence intervals on the raw model probabilities from the diagonal of the variance–covariance matrix of model predictions (Deane et al. 2017b). Only model predictions made in cells that inundated are valid, and as water levels declined across different scenarios, the area of wetland that could be modelled decreased. We therefore report the change in probability for each functional group in the area remaining inundated.
at the maximum drawdown level modelled. That is, we used the minimum
inundated area as an analysis mask in probability calculations for all scenarios.

Directly comparing model probabilities (which range from 0 to 1) for different
water regime scenarios provides the most reliable means to assess the likelihood
of change in plant functional group cover. However, it is possible to convert
probabilities to presence-absence predictions by specifying a threshold
probability above which the functional group is considered present. While this
introduces an additional level of uncertainty, it arguably provides a more
intuitive illustration of potential changes in spatial distribution. We therefore
converted raw probabilities for the Aftt and Se groups in the deeper wetland to
presence–absence to show their potential change in cover graphically. To do this,
we used threshold values of between 0.8 and 0.3 respectively, which gave the
best model-validation statistics in model fitting (Deane et al. 2017b).

Results

Changes in plant functional groups

Changes in surface-water inundated area with declining groundwater levels
differed between the two sites due to the differences in basin depth and general
morphology (Table 2). The maximum drawdown scenarios reduced the surface-
inundated area by \( \sim 71\% \) for 0.5 m decline in the shallow wetland and \( \sim 20\% \) for
a 1.0 m drawdown in the deeper wetland (Fig. 3). Changes in plant functional
group probabilities for both wetlands were greater for groups at the extremes of
the hydrological gradient (Figs. 4, 5), particularly dry-adapted groups. In both
wetlands, terrestrial groups (Tdry, Tdamp) increased sharply at the maximum
modelled decline in groundwater level (Figs. 4, 5). Also consistent in both
wetlands were the responses of amphibious fluctuation-responders plastic
growth (Afrp), which remained near 0.5 regardless of the water level scenario;
and, submerged \( r \)-selected (Sr) which decreased linearly across scenarios,
although they are of lower probability in the shallow wetland (Figs. 4, 5).

Fig. 3

Inundated area for baseline and maximum groundwater level decline scenario for
the two bathymetries: a grass-sedge (shallow) wetland b inland interdunal (deep)
wetland. Gridlines represent 100 m intervals
**Fig. 4**

Change in plant functional group probabilities for each hydrological scenario in the shallow (grass-sedge) wetland type. Functional group probabilities are calculated based on the area remaining inundated at the greatest decline in groundwater (0.5 m—see Fig. 3a†). Error bars indicate 95% confidence intervals in the modelled prediction.

**Fig. 5**

Change in plant functional group probabilities for each hydrological scenario in the deeper (inland interdunal) wetland type. Functional group probabilities are
calculated based on the area remaining inundated at the greatest decline in groundwater (1.0 m—see Fig. 43b). Error bars indicate 95% confidence intervals in the modelled prediction.

Other changes in plant functional group probability differed among the wetlands. The mean probability for the submerged-emergent (Se) group remains consistent in the deeper wetland up to and including a 0.5 m decline in groundwater, but decreases to almost zero between a 0.5 and 1.0 m decline (Figs. 5, 6). In contrast, probabilities for ‘Se’ in the shallow wetland decline consistently across scenarios, before dropping to near zero at the maximum (0.5 m) decline. While the probabilities for amphibious groups adapted to shallow inundation (Aftw, Aftl) increase linearly with decreasing groundwater level in the shallow wetland (Fig. 4), their probability remains extremely low in the deeper wetland until a rapid increase occurs for a 1 m decline (Fig. 5). The probability of the amphibious fluctuation tolerator emergent (Afte) group, (comprising largely sedge- and rush-type species; Tables 2, 3), remains consistently close to 100% for all scenarios in the shallow wetland (Fig. 4). However, in the deeper wetland it increases rapidly between 0.25 and 0.5 m declines and approaches 100% at the maximum (1.0 m) decline scenario.

Fig. 6
Change in predicted spatial distribution of two plant functional groups with declining groundwater level for the deeper (inland interdunal) wetland type. Top row *amphibious fluctuation tolerator emergent* (Afte) predominantly includes sedge genera such as *Baumea* or *Juncus*, while common species in the region from the submerged emergent (Se) functional group (bottom row) include *Triglochin procera* and *Typha domingensis*.

**Table 3**

Change in surface inundated area and major transitions in wetland plant functional group probabilities for the two wetland types

<table>
<thead>
<tr>
<th>Wetland type</th>
<th>GW scenario</th>
<th>% Inundated area at baseline</th>
<th>Change in PFG</th>
<th>Indicator species</th>
</tr>
</thead>
<tbody>
<tr>
<td>Shallow</td>
<td>0–0.25</td>
<td>71</td>
<td>Increase in Tdamp</td>
<td><em>Samolus repens</em>, <em>Lepidosperma longitudinale</em>, <em>Hypolaena Hypolemma</em>, <em>fastigiata</em></td>
</tr>
<tr>
<td></td>
<td>0.25–0.5</td>
<td>29</td>
<td>Increase in Tdry Loss of Se</td>
<td><em>Senecio pterophorus</em>, <em>Hypocharis glabra</em>, <em>Holcus lanatus</em>, <em>Triglochin procera</em>, <em>Eleocharis spachealata</em></td>
</tr>
<tr>
<td>Wetland type</td>
<td>GW scenario</td>
<td>% Inundated area at baseline</td>
<td>Change in PFG</td>
<td>Indicator species</td>
</tr>
<tr>
<td>--------------</td>
<td>-------------</td>
<td>-----------------------------</td>
<td>---------------</td>
<td>------------------</td>
</tr>
<tr>
<td>Deep</td>
<td>0.25–0.5</td>
<td>96–90</td>
<td>Increase in Afte</td>
<td><em>Baumea arthropylla, Baumea juncea, Polypogon monspeliensis</em></td>
</tr>
<tr>
<td></td>
<td>0.5–1.0</td>
<td>90–83</td>
<td>Loss of Se Increase in Tdamp, Increase in Aftw Increase in Aftl</td>
<td><em>Triglochin procera, Eleocharis spachealata, Samolus repens, Hypolaena fastigiata, Distichlis distichophylla, Leptospermum myrsinoides, Leptospermum lanigerum, Melaleuca halmaturorum, Lilaeopsis polyantha, Crassula helmsii, Isolepis platycarpa</em></td>
</tr>
</tbody>
</table>

Indicator species are the most commonly observed species of each functional group from data used in Deane et al. (2017b)

Exotic species are indicated with an asterisk

Indicator species of transitions in plant functional groups

For the shallow wetland, increases in terrestrial species (particularly exotic forbs and grasses), and absence of submerged species such as *Triglochin procera* (Table 3) would be indicative of the transitions predicted for a decline between 0.25 and 0.5 m. For the deeper wetland, declines of between 0.25 and 0.5 are predicted to allow an expansion in sedge (Afte) species across the wetland, that increased more rapidly with greater declines in level (Figs. 5, 6; Table 3). This would likely relate to sedge species such as *Baumea arthropylla, B. juncea* and the exotic grass *Polypogon monspeliensis*, which are all common Afte species in the region (Table 3).

Discussion

Both wetland types are predicted to maintain distinct and consistent plant functional group compositions, were groundwater levels to decline by 0.25 m. If groundwater levels fell again by the same amount to 0.5 m below baseline, clear changes in the composition of plant functional groups are predicted. Perhaps counter-intuitively, the deeper wetland would be at greater risk of change in plant functional group composition over the remaining inundated area, with the expansion of sedge-cover across the basin a likely outcome. Changes in the functional group composition of the shallow wetland were less obvious at a 0.5 m groundwater decline, but its inundated area would be decreased by > 70%.
Cooling et al. (2010) mapped wetland vegetation at several sites in the study region and found a 0.3 m discretization of the elevation gradient described the observed vegetation zonation well. This is consistent with the change in sedge cover (functional group Afte) predicted for declines between 0.25 and 0.5 m (equivalent to ~0.2–0.38 m surface water reduction) in the deeper wetland. It also matches the change in terrestrial and submerged group probabilities for the same two scenarios in the shallow wetland. Our results and those of Cooling et al. (2010) therefore suggest a 0.25–0.3 m decline represents a threshold of surface water lowering where change in some functional group distributions is likely in these wetlands, irrespective of basin morphology. This level of decline is consistent with existing policy in the study region for protecting wetlands of high ecological value, but a single threshold provides little guidance on the likely change in wetland ecology. Our results illustrate the likely types of change expected for the most commonly occurring wetlands in the region, allowing more nuanced interpretation of these thresholds. Interestingly, the magnitude of predicted change in composition we found are broadly consistent with groundwater decline risk categories for wetland vegetation on the Swan Coastal Plain, Western Australia (low < 0.25 m, moderate 0.25–0.5, high 0.5–0.8 m), which were developed by analysis of long-term (1996–2010) monitoring data (Loomes et al. 2013).

Declining groundwater levels differed in the extent of change they produced in surface-inundated area for the two wetlands. The large decrease in inundated area for the grass-sedge wetlands for modest groundwater declines should be detectable through analysis of satellite imagery (e.g. Landsat 8), and this could provide a cost-effective means to monitor impacts in this wetland type over large areas. It would be more difficult to apply remote methods in the deeper inland interdunal wetland, which would remain inundated over more than 80% of its original area even at a 1.0 m decline in groundwater. Model predictions of the rapid expansion of emergent amphibious (Afte) species across the basin suggest this could be used as an indicator. If this proved difficult to quantify with remote methods, expansion of sedge or rush species from wetland fringes into core areas of the deeper wetland should be readily evident in field surveys, even using only rapid assessment techniques or photo points.

Model scenarios predict plant functional group zonation if the current seasonal variations in groundwater were maintained, but shifted to a lower level and remained stable at this new level for a 3-year period. However, the length of time required for plant distributions to equilibrate to the new level, were it to arise, is unknown. Plant succession will likely have some inherent resistance as individuals can only re-establish up or down the elevation gradient as space
becomes available. Improved understanding of inter-annual variability is important to ensure responses indicate irreversible changes rather than natural community dynamics.

Reductions in groundwater level have the potential to completely transform dependent plant communities (e.g., Stromberg et al. 1996). If current climatic trends continue, managing successional changes in vegetation as wetlands transition to more terrestrial habitat will become increasingly important. At a 1.0 m decline, groundwater would no longer reach the surface of the shallow, grass-sedge wetland and such complete loss of wetland patches—however small in area—increases the risk of regional species loss (Deane et al. 2017a). Data from a formerly permanent wetland in the region affected by groundwater decline from 2006 showed exotic terrestrial species (Tdry and Tdamp) were present in more than 50% of quadrats by 2013, and these have the potential of completely dominating the wetland plant community. Interestingly, emergent amphibious species, presumably sustained by seasonal rainfall, remained widespread (98% of quadrats)—though not abundant—at the site (unpublished data). In deeper interdunal wetlands, declines of 0.5 m could create conditions suitable for the expansion of the exotic grass *Polypogon monspeliensis*, which is a high-threat weed species of nationally listed herbaceous wetlands to the east of the study region (TSSC 2012).

While invasion by agricultural weeds appears a likely initial outcome of wetland loss in the absence of management intervention, temperate wetlands in Australia often support species-rich ecotones suitable for terrestrial or facultative wetland plants (Brock and Casanova 1997; Deane et al. 2016). Plantings of native species suited to wetland ecotones could allow them to establish and migrate naturally down gradient as wetlands dried, reducing the likelihood of widespread invasions by exotic species. To support planning for optimal succession of the native plant community to a drier environment it is advisable to develop vegetation models that can predict vegetation based on soil moisture or depth to groundwater.

Our hydrological scenarios assume a linear dependence of wetland surface inundation on groundwater level, but surface water–groundwater connectivity in wetlands in the region is highly complex Taylor et al. (2015). Future work could couple plant functional group models to spatially explicit surface water–groundwater models in different connectivity settings to better understand the range of ecological responses. It is also important to consider how factors other than hydrology might affect the realised wetland communities, in particular salinity, which exerts strong controls on the prevalence of different functional groups (Deane et al. 2017b). Although sea water intrusion could become an
increasing concern in some coastal aquifers and wetlands (Mustafa et al. 2012), Goodman (2012) suggests the major salinization risks are associated with evapoconcentration of salt from shallow groundwater. Depending on wetland connectivity to local and regional groundwater flow systems, this process of salinization could occur on very different timescales (Taylor et al. 2015). Although wetland plants in the region can cope with short-term pulses of high salinity inflows (Goodman et al. 2010), sustained increases in salinity will alter the composition of wetlands (Goodman 2010, 2012) and would also result in very different model predictions for functional groups (Deane et al. 2017b). Other demonstrated influences on wetland plant composition that are not incorporated into model predictions include disturbance (Harding et al. 2015; Nicol et al. 2007), the state of wetland propagule banks (Goodman et al. 2011; Nicol and Ganf 2000; Nicol et al. 2003) and the rate of groundwater decline (Touchette et al. 2008).

Conclusions

We found wetland basin morphology not only affects the functional group composition of wetlands under baseline conditions, but also the ways in which they are sensitive to groundwater decline. Perhaps counter-intuitively, deeper wetlands were more likely to experience major shifts in functional group distribution than shallow wetlands across the area of inundation. More predictably, impacts on shallow wetlands are likely to be associated with loss of surface area and invasion by terrestrial weeds. By quantifying likely ecological transitions during the initial stages of global change, correlative ecological models can provide support for planning, policy evaluation and monitoring design.

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Compliance with ethical standards

Conflict of interest The authors declare that they have no conflict of interest.
Electronic supplementary material

Below is the link to the electronic supplementary material.

Supplementary material 1 (PDF 261 kb) Online Resource 1 - we provide a single PDF document with the results of linear regression analysis of surface water and groundwater levels for eight wetlands in the region

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