Examining temporal and spatial changes in surface water hydrology of groundwater dependent wetlands using WOfS (Water Observations from Space)

Southern Border Groundwaters Agreement area, South East South Australia

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Disclaimer

Foreword

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Summary

The absence of long-term surface water level monitoring of wetlands has often been cited as a key limitation for understanding relationships between groundwater, rainfall and surface water variations (Harding et al., 2015; Taylor et al., 2015). Methods for detecting surface water from optical satellite imagery are however considered well established and operational (Mueller et al., 2016). Mapping of inundation from remotely sensed imagery (Landsat satellites) via the Australian continental scale Water Observations from Space (WOfS) (Geoscience Australia, 2014; Mueller et al., 2016) dataset, used in conjunction with a LiDAR (Light Detection and Ranging) derived Digital Elevation Model (DEM) offered the potential for re-creating approximations of historical surface water hydrographs for larger, open water wetland basins, and therefore had the potential ability to determine temporal and spatial surface water hydrological change in relation to groundwater levels. The objective of this study was to examine recent historical (past ~30 year) variations in surface water expression in wetland groundwater dependent ecosystems (GDEs) in relation to groundwater level change in a region where recent extraction estimates have been found close to or exceeding permissible allocation volumes in the Border Groundwaters Agreement. By establishing relationships between surface water and groundwater levels in multiple wetland GDEs over a relatively long-term period, we were able to characterise trends in surface water response to groundwater level change over time and estimate the relative impacts of contemporary rainfall variability and groundwater extraction/interception on observed changes in surface water inundation metrics of selected wetland GDEs.

The approach utilised the outputs from Geoscience Australia's WOfS products combined with a LiDAR DEM to approximate historic surface water hydrographs for 12 open water wetland GDE basins, with analysis indicating that it was possible to obtain both maximum and minimum annual surface water levels for the majority (>80%) of years over the WOfS data capture time-span. The use of WOfS derived data in conjunction with a LiDAR DEM to produce hydrographs was also shown to improve the usefulness of remotely sensed products for monitoring spatial changes in wetland surface water inundation – where wetland basin bathymetry influences the effectiveness of simply using spatial extent remotely sensed data (Deane et al., 2017a). The method developed in this report has the potential to be an efficient and cost-effective method of interpreting temporal surface water change at both wetland and landscape scales.

The relationship between groundwater level declines and the surface water hydrology of wetland GDEs was shown to vary based on distinct hydrogeological zones (HZs) where the aquifer performs similarly (after Harrington & Currie, 2008): higher elevation zones (HZs 3 and 7) and lower elevation zones (HZs 5 and 6). A strong linear relationship was found between surface water levels and nearby groundwater levels (R^2>0.7 for all except one site), with wetlands becoming dry when groundwater levels were on average 0.3–0.5 m below the base of the wetland. Groundwater levels in the unconfined aquifer were generally shown to be higher than wetland bed level during periods of surface water expression, indicating that the groundwater discharge from the unconfined aquifer had a potentially strong influence on surface water hydrology.

WOfS hydrograph analysis of wetlands within HZs 5 and 6 indicated seasonally inundated wetland GDEs strongly related to rainfall responsive groundwater levels. Periods where wetlands failed to inundate coincided with the time period referred to as the Millennium drought (late 1996 to mid-2010) (Australian Government Bureau of Meteorology, 2015). Surface water expression at least partially returned with rainfall and groundwater level recovery after approximately 2010. Whilst it is likely that groundwater extraction from the unconfined aquifer has a cumulative impact on groundwater levels (Cranswick, 2018), rainfall variability was shown to be the more likely driver of surface water change in the selected wetland GDEs in HZs 5 and 6.

In comparison, analysis of WOfS derived hydrographs of wetland GDEs in HZs 3 and 7 indicated that GDEs that were permanently inundated in the 1980s and 1990s were completely dry by 2006, and this was potentially attributable to significantly declining groundwater levels. With use of groundwater level scenarios generated by groundwater hydrograph regression analysis of nearby observation wells (Cranswick, 2018), we found that the observed absence of water from 2005–15 was unable to be entirely accounted for by rainfall variability alone. We showed that whilst rainfall variability was a contributing factor to the decline and loss of wetlands with high dependency on the unconfined aquifer, the impacts from surrounding landuses such as plantation forestry and groundwater extraction for irrigation, which are both extensive in HZs 3 and 7, were likely to be significant contributors to the observed losses.

The demonstrated contemporary impacts to GDEs (as also identified by Harding et al., (2015) and Cranswick (2018)) indicate a clear need for establishing ecologically relevant groundwater condition triggers and resource condition limits. The methods presented in this report, utilising the nationally available WOfS datasets, has the potential to contribute the monitoring of GDE responses to groundwater level decline and ultimately contribute to the establishment of environmental water requirements and the determination of limits of acceptable change for selected GDEs. This report was developed in conjunction with the groundwater resource condition assessment of the eastern Lower Limestone Coast PWA (Cranswick, 2018).

1 Introduction

Groundwater as a resource assists in meeting domestic, agricultural and industrial needs, and the human use of groundwater resources is increasing globally (Vörösmarty et al., 2000). The significance of groundwater in maintaining the health of aquatic ecosystems is often underestimated or unknown, resulting in a lack of scrutiny of groundwater policy and management (Nevill et al., 2010). Nevill et al. (2010) identify that groundwater overdraft (defined as abstracting groundwater at a rate which prejudices ecosystem or anthropocentric values) can substantially impact on groundwater dependent ecosystems (GDEs), and that impacts may occur over time scales at variance to those used in water planning and regulation.

Groundwater is the predominant water resource in the South East NRM region (the Region) of South Australia, with the majority of water for economic and domestic activity allocated from the regional unconfined aquifer. Depth to groundwater in the unconfined aquifer is generally shallow and responsive to variations in rainfall (Brown et al., 2001), with the Region's wetland ecosystems largely supported by the interaction of groundwater and surface water. The majority of the >16 000 wetlands in the region (77% by number and 96% by area) have been identified as having a high likelihood of interaction with unconfined aquifer groundwater (SKM, 2009). Increased groundwater extraction and changes in landuse, combined with declines in rainfall since the 1970s has resulted in widespread drawdown of groundwater across the Region (DFW, 2010). Increases in the reliance on groundwater extraction for human uses as a result of reduced rainfall, is a trend not unique to the Region, and has been witnessed in other parts of Australia where the reduced availability of surface water, and relative reliability of groundwater, have led to increased extraction (McFarlane et al., 2012).

There is clear and growing evidence that GDEs are at risk in the Region as a result of declining groundwater levels (Cook et al., 2008; DFW, 2010; Harding et al., 2015; Brookes et al., 2017; Deane et al., 2017a), despite provisions in water allocation planning for the needs of dependent ecosystems (SENRMB, 2013). Deane et al. (2017a) and Brookes et al. (2017) surmised that even relatively small declines (< 0.3–0.6 m) in groundwater level would result in the loss or degradation (gradual terrestrialisation) of wetland GDEs. However, without systematic long-term surface water monitoring of wetlands at a regional scale, it has not been possible to quantify the contemporary losses of GDE wetland ecosystems from previously observed groundwater levels.

The absence of long-term surface water level monitoring of wetlands has also often been cited as a key limitation for understanding relationships between groundwater, rainfall and surface water variations (Harding et al., 2015; Taylor et al., 2015), and therefore for demonstrating the impacts and potential risks to GDEs from groundwater use and climate change (Harding et al., 2015; Deane et al., 2017a). Methods for detecting surface water from optical satellite imagery are however considered well established and operational (Mueller et al., 2016), within limitations inherent to the imagery, including vegetation obstruction of the water target, cloud or cloud shadow, and pixel size of the imagery (Jones, 2015; Mueller et al., 2016). Mapping of inundation from remotely sensed imagery (Landsat satellites) via the Australian continental-scale Water Observations from Space (WOfS) (Geoscience Australia, 2014; Mueller et al., 2016) dataset, used in conjunction with a LiDAR (Light Detection and Ranging) derived Digital Elevation Model (DEM) offered the potential for re-creating approximations of historical surface water hydrographs for larger, open water wetland basins, and hence the potential ability to determine temporal and spatial surface water hydrological change in relation to groundwater levels.

The objective of this study was to examine temporal surface water hydrological trends in wetland GDEs, in relation to groundwater level change in the Border Groundwaters Agreement (BGA) area of the Lower Limestone Coast (LLC) prescribed wells area (South East, South Australia): a region where some allocation limits exceed Permissible Annual Volumes (PAV) set by the South Australian–Victorian Border Groundwaters Agreement (BGA) (Cranswick, 2018). By establishing relationships between surface water and groundwater levels in multiple wetland GDEs in the study area, we aimed to determine the relative impacts of contemporary rainfall variability and groundwater extraction/interception on observed changes in surface water inundation metrics for a selection of wetland GDEs.

This report was developed in conjunction with the groundwater resource condition assessment of the eastern Lower Limestone Coast Prescribed Wells Area (PWA) (Cranswick, 2018), from which the results of groundwater hydrograph regression analysis of relevant observation wells are sourced for analysis in this report.

1.1   Study area

The BGA area covers an area that extends 20 km either side of the South Australian–Victorian border, from the coast to the River Murray (Figure 1.1). There were 5183 wetlands mapped within the South Australian side of the BGA area, 57% of which were identified as likely to be dependent on the unconfined aquifer to varying degrees (SKM, 2009). The likelihood of groundwater dependency of wetlands in the BGA-area north of Zone 5A is low because of the greater depth to the unconfined aquifer (SKM, 2009) (Figure 1.1). Most wetland GDEs in the BGA area occur within the areas bounded by Zone 1A to Zone 4A.

Figure 1.1 shows that there are six distinct hydrogeological zones (after Harrington & Currie, 2008) which cover the study area. Hydrogeological zones (HZ) were defined using spatial analysis of groundwater behaviour and biophysical factors that are somewhat independent of current management boundaries (Harrington & Currie, 2008). Deep geological fault zones form natural boundaries between hydrogeologically distinct land units, with differing saturated aquifer thickness and groundwater hydrograph behaviour (depth to water, seasonal fluctuation, and trends in groundwater level). It follows that similarities in groundwater–surface water interactions also occur within the HZs, hence their boundaries are also used here to group and discuss general trends in GDE surface water behaviour.
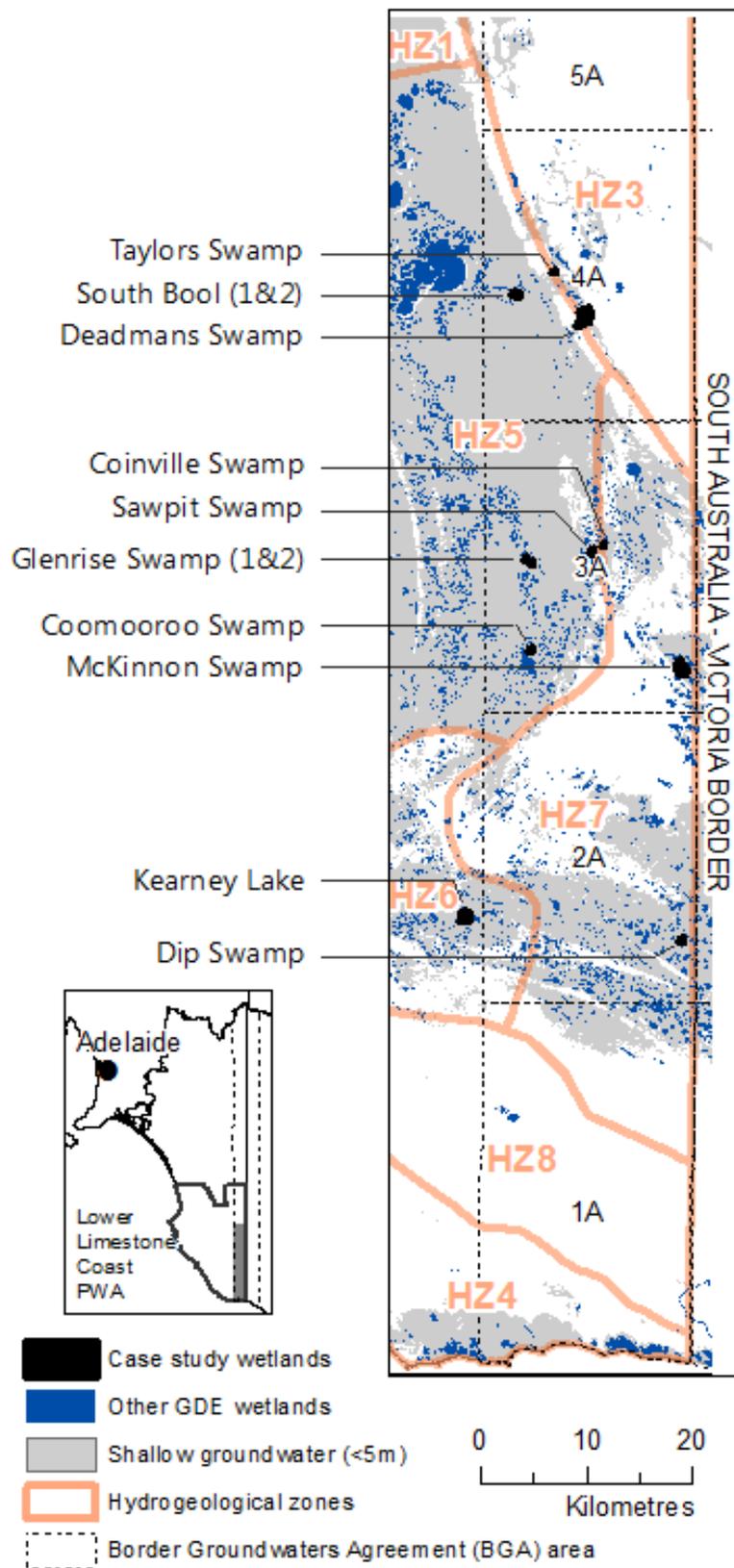
Figure 1.1. Location of the southern BGA area and of case study wetland GDEs

2 Method

A generalised conceptual model of GDE function adapted from Harding et al. (2015) (Figure 2.1), identifies relationships between rainfall, groundwater and surface water levels influencing surface water hydrometric parameters (depth, duration, frequency and area inundated). By establishing quantitative relationships between components of the conceptual model, it is possible to interpret and predict temporal and spatial changes in wetland hydrology to selected groundwater level scenarios (e.g. Harding et al., 2015, Deane et al., 2017a). This approach assumes a strong and direct relationship between groundwater and surface water level, but does not quantitatively assess the magnitude of groundwater–surface water exchange fluxes, which are influenced by the wetland geometry, properties of shallow sediments, and any preferential pathways that may exist within the aquifer in contact with the wetland. The approach also assumes other significant surface water inputs or runoff are insignificant in the cases to which it is applied. Hence the approach used here is a simplification of the likely complex groundwater–surface water relationships and therefore there is associated uncertainty in the relationships developed. However, the conceptual model is considered to be valid when applied to wetlands that are in known groundwater discharge landscape positions with an absence of other major surface water inputs or runoff (e.g. Harding et al. 2015; Deane et al., 2017a).
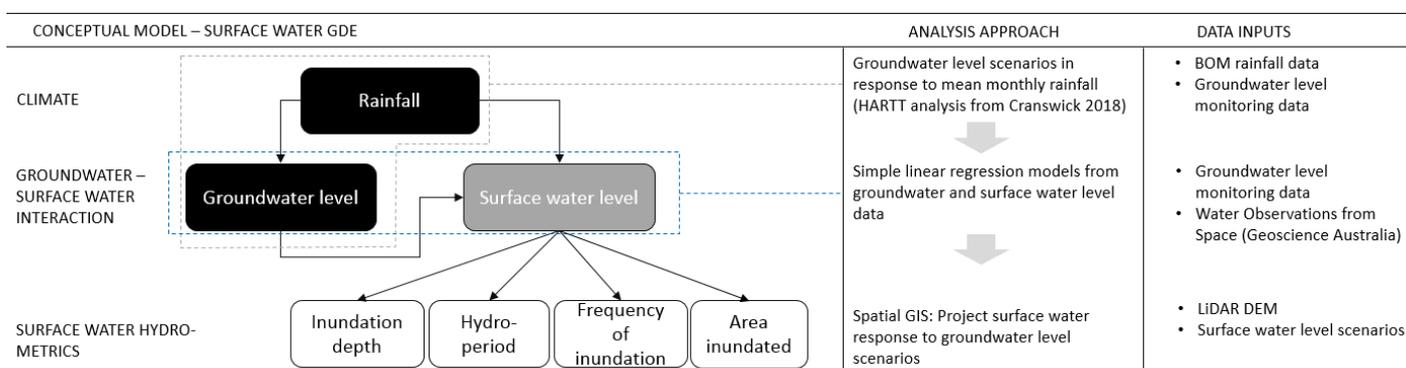
Figure 2.1.   Generalised GDE conceptual model, analysis approach and key data inputs

The application of the conceptual model also requires both wetland surface water level and local groundwater level data, normally limiting its application to well instrumented and regularly monitored sites (Harding et al., 2015). Taylor et al. (2015) identified sparse spatial and temporal monitoring for surface water in wetlands as one of the main challenges in quantifying groundwater–surface water interactions in the LLC PWA. Wetland GDE sites with surface water level monitoring equipment are few, and often have only short contemporary data capture periods (SKM, 2010), whereas groundwater monitoring infrastructure networks have been established across the region, often with data records from 1970s. To overcome these data limitations, our approach was to utilise remotely sensed water observations from the nationally available Water Observations from Space (WOfS) dataset (Geoscience Australia, 2017) to approximate historical surface water hydrographs for a selection of wetland GDEs in the study area where long-term groundwater level data was available. The resulting surface water hydrographs were then used to establish relationships between groundwater and surface water levels, using these relationships to apply groundwater level scenarios (sourced from Cranswick (2018)) to analyse and interpret spatial and temporal changes and trajectories of GDEs in the study area (Figure 2.2).
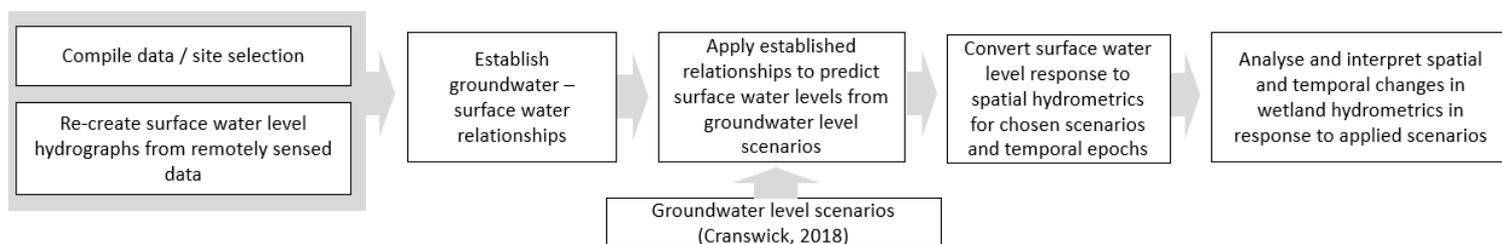
Figure 2.2.   Workflow diagram – analysis approach

2.1 Remotely sensed temporal surface water analysis

The WOfS dataset from Geoscience Australia provides temporal (date specific) surface water observations derived from the Australia-wide archive of Landsat 5 imagery (1987–2011), and Landsat 7 imagery (2000 to present). A water detection algorithm was developed by Mueller et al. (2016) to derive the WOfS dataset based on a decision tree classifier, and a comparison methodology using a logistic regression, implemented on every pixel to produce classifiers of water or non-water. The analysis procedure generated standardised and structured surface water classifications for each 25 x 25 m pixel, for each observation over the 27 years of data capture with an overall classification accuracy assessment of 97% (Mueller et al. 2016).

By overlaying each date specific classified water observation (25 x 25 m pixels) with a LiDAR DEM (at 2 x 2 m resolution), we derived maximum seasonal inundation height data (in m AHD), along with 'wetland dry' indicators in order to approximate historical surface water hydrographs for a selection of GDEs in the study area.

An initial review of yearly summarised (Turner et al., 2014) WOfS data of a selection of high rainfall years where water was expected to be present at its maximum extent (1992) was conducted in order to identify the suitability of individual wetland GDEs for remote detection of inundation. Selected sites were relatively large, open water basins, with limited obscuring vegetation (often in an agricultural landscape where grazing allowed clearer observations of surface water presence). Sites were also required to be in close proximity of suitable long-term groundwater level monitoring infrastructure and be plausibly consistent with assumptions of the generalised GDE conceptual model (Figure 2.1). As such, sites were further filtered to exclude those with surface water inputs from creeks or drains, and those with potentially large local surface water catchments (as determined by the LiDAR DEM and soil drainage properties (Maschmedt, 2002)). A total of 12 wetland basins made up the final selection, with a mean individual area of 17.12 ha (minimum: 5.4 ha; maximum 82.77 ha) (Figure 1.1).

Data processing and analysis was completed in ArcGIS® (ESRI). The classified WOfS raster data from 1987–2013 (totalling 362 individual raster scenes) were clipped to the maximum obtainable water level of each wetland basin (determined using the LiDAR DEM) and the clipped rasters renamed to reflect the date of capture and exported to a file geodatabase using ArcGIS models and Python script (Python Software Foundation 2017) (Appendix A, Workflow 1).

The 2 x 2 m LiDAR DEM for each wetland was converted from a grid to a point file, retaining elevation data for each point, and attributed with x and y projected coordinates. A second Python script was then developed to extract multi-values from each of the clipped WOFL (Water Observation Feature Layer) rasters to each wetland DEM point (Appendix A, Workflow 2). The resulting point feature class was exported as a .txt file and modified in MS Excel® to format the date fields. From the Excel files, IF logical function was used to retrieve elevation data for classified 'water in pixel' points, and then maximum inundated elevation (AHD) values calculated for each raster scene date. The wetland was recorded as dry where a 'no water in pixel' classifier was retrieved for the lowest elevation level of the wetland site. 'Cloud' and 'cloud shadow' errors at highest elevations of the wetlands were also identified, indicating that high water elevation data may be obscured by cloud. The resulting date and inundation elevation data were then manually filtered for potential cloud errors, anomalies were visually checked with the original raster date scenes, and the data then used to reconstruct hydrographs for each of the selected wetland sites.

2.1.1 Validation of WOfS derived hydrographs

Surface water monitoring data was available for three of the selected wetland sites as part of a broader regional GDE monitoring network (SKM, 2010) within the study area, however WOfS derived hydrographs from only one of the monitored sites was able to be produced due to vegetation obstruction. For this site, the variation between the WOfS derived hydrographs and observed data were examined.

2.2 Establishing simple linear relationships between groundwater and surface water

Where it is known that there is a strong connection between a wetland and the groundwater system and assumptions of the generalised conceptual model (Figure 2.1) generally apply, predictions of groundwater levels (and scenarios) can be used to assess the likely change in surface water parameters in a wetland GDE based on establishing empirical relationships between surface water and groundwater levels (Chambers et al., 2013; Harding et al., 2015; Deane et al.,

2017a). Simple linear regression models have previously been successfully fitted to groundwater and surface water level monitoring data in highly groundwater dependent wetlands in the South East NRM region, and used to predict changes in wetland hydrology for a given decline in groundwater level (Harding et al., 2015; Deane et al., 2017a). This approach however has relied on the availability of surface and groundwater monitoring data at suitable spatial and temporal scales.

To overcome this data shortfall, correlations between WOfS derived surface water hydrographs for each of the 12 selected wetlands and data from nearby (within ~3 km), shallow (<15 m drill depth) groundwater observation wells completed within the unconfined aquifer were established. Observation well quarterly (approximately four times per annum) monitoring data over the same time period as the derived surface water data (1987–2013) was sourced. Both derived surface water data and groundwater observations were converted to mean monthly data. Linear regression equations (Equation 1) were fitted to time periods where surface water was (at least seasonally) expressed.

$$Surface\ water\ level\ (y) = groundwater\ level\ (x) \times a + b \qquad\qquad\qquad\qquad \text{Eq1}$$

Where, a is the slope of the regression line and b is the intercept point of the regression line on the y axis.

Given significant declining groundwater levels observed near to some of the wetlands, and in the absence of any measures of soil moisture, strong linear relationships ceased to be maintained beyond the point where groundwater levels declined well below ground level and no longer interacted with surface water (and where the wetlands were recorded as dry).

2.3   Simulating surface water scenarios

Spatial representation of summarised temporal hydrometric (level and frequency) data from the WOfS derived hydrographs in 10-year epochs for each wetland site was produced by calculating average water levels (in m AHD) and subtracting the corresponding value of the DEM. Results were then displayed as spatial wetland hydrometric scenarios for each epoch by projecting the 2 x 2m DEM point file x and y coordinates in ArcGIS. The individual linear regression relationships between surface water expression and groundwater level were used to estimate seasonal maximum surface water inundation where no data from the WOfS analysis was available to fill existing data gaps. From the spatial representations, variation in wetland inundated area, and spatial changes in surface water depths and frequency of inundation could be illustrated.

2.3.1   Simulating surface water response to groundwater level change

A simple regression based analysis was performed by Cranswick (2018) using an adaptation of the HARTT (Hydrograph and Rainfall Time Trend) approach originally developed by Ferdowsian et al. (2001) to approximate linear responses in groundwater levels to any deviation from mean monthly rainfall, and then statistically estimate any additional trend in groundwater levels for a selection of observation wells in the study area. The approach has high explanatory power in cases where rainfall is the primary recharge mechanism to the unconfined aquifer, and there is a good understanding of the historical and current landuse and groundwater practices in the area of interest. The critical aspect of the regression analysis was the time trend component which may be attributed to processes such as groundwater extraction and/or landuse change (such as plantation forestry which has the capacity to both access groundwater directly and intercept recharge). It should be noted that the analysis does not explicitly identify these components, or the proportion of their combined influence on groundwater levels. In Cranswick (2018), the approach was applied in monthly time-steps using the cumulative deviation from mean 1900–85 rainfall (CDMR) from the nearest or most appropriate weather station as input (see further details in Cranswick, 2018)

The mean value used for the rainfall dataset (determined by the rainfall period chosen, i.e. 1900–85) adds some uncertainty and non-uniqueness to this analysis because it was possible for the influence of other processes to be captured within this deviation. A calibration period where the processes influencing the hydrograph were well known is also critical for the interpretation of any time trends that may exist. A calibration period from 1985–2016 was used in all analyses unless there was a clear change in the responses seen in the hydrograph. The approach assumes that the influence of the time-trend factor (e.g. groundwater extraction or landuse change) was uniform over the selected period – this approximates the average influence of any such other non-rainfall factor on the hydrograph.

We utilised the outputs from the hydrograph regression analysis (Appendix B) from Cranswick (2018) of groundwater levels from observation wells available nearby two of our selected GDE sites: Deadmans Swamp and Dip Swamp (Figure 1.1). Both sites are located within forestry plantation matrix and in areas where significant declines in groundwater levels have been observed.

Projections of groundwater levels with and without the time trend component provided two groundwater level scenarios from which to estimate the likely influence of rainfall variability alone on surface water level:

Scenario 1)   Estimated rain + trend: statistical estimate of trends in groundwater levels from mean rainfall and time trends

Scenario 2)   Estimated rainfall only: statistical estimate of groundwater level response to deviations from mean rainfall (time trend removed)

Scenario responses were interpreted over three 10-year epochs (1985–95; 1995–2005; 2005–15). Ten-year epochs were chosen to reflect approximately the 10-year water planning and policy cycle for the study region (SENRMB, 2013). Furthermore, as there are accumulating degrees of uncertainty between derived WOfS hydrographs, the hydrograph regression analysis (Cranswick, 2018), and assumptions inherent in the general GDE conceptualisation, it was considered more useful to view the outputs in terms of general trends over longer time-periods.

We utilised the surface water – groundwater relationships established for the two corresponding wetland sites, both with strong (R^2>0.7) linear relationships between surface water and groundwater, to determine estimated surface water levels for each groundwater level scenario, and calculated mean maximum seasonal surface water levels for each of the 10-year epochs. Mean surface water hydrometrics (depth and frequency) were displayed spatially in ArcGIS by converting the surface water levels (m AHD) to water depths by subtraction of the LiDAR DEM.

Finally, the difference in the mean maximum seasonal surface water expression between the estimated surface water level from Scenario 1 and Scenario 2 (rain influence alone) for the most recent 10-year epoch (2005–15) was calculated, providing an indication of the likely impact of groundwater usage on surface water expression within two individual wetlands in the study area.

3 Results and discussion

3.1 Remotely sensed surface water hydrographs for wetlands

A total of 362 individual date specific water observation raster scenes from the WOfS dataset were analysed for each of the 12 selected wetlands sites. Error returns (primarily cloud and cloud shadow interference) were significant, with between approximately 40–70% of all WOFLs returning errors (Table 3.1). In the study area, seasonal distribution of cloudiness varies predominantly in line with seasonal variations in rainfall, with winter months generally cloudier than the summer months (Australian Government Bureau of Meteorology, 2012). The study area experiences an average of 100–150 days of rainfall greater than 1 mm annually (Australian Government Bureau of Meteorology, 2007), equating to approximately 30–40% of days annually. As such, a high rate of cloud errors were expected from Landsat derived remotely sensed data sources given local climate conditions. Cloud errors were very high throughout winter months, however at least one clear image was obtainable for detecting the seasonal high water expressions in spring for 72-88% of all years, and seasonal low water expressions in autumn for 84–100% (Table 3.1) for all sites. The relative decrease in seasonal cloud cover presence throughout summer and autumn months account for the overall number of 'dry' observations exceeding 'wet' observations for all sites, where many of the selected wetlands displayed seasonal wetting and drying cycles. The higher proportion of 'dry' observations may also be influenced by increasingly dry conditions in the wetlands observed over the time period of the data capture.

Table 3.1.   **Summary of WOfS generated surface water observations**

Sites	South Bool 1	South Bool 2	Glenrise Swamp 1	Glenrise Swamp 2	Coomooroo Swamp	Kearney Lake	Taylors Swamp	Deadmans Swamp	Sawpit Swamp	Coinville Swamp	McKinnons Swamp	Dip Swamp
Number of wet observations	35, 10%	48, 13%	26, 7%	41, 11%	32, 9%	32, 9%	35, 10%	71, 20%	28, 8%	41,11%	15, 4.1%	21, 6%
Number of dry observations	157, 43%	170, 47%	133, 37%	135, 37%	103, 28%	157, 43%	120, 33%	51, 14%	84, 23%	81, 22%	107, 30%	81, 22%
Number of errors	170, 47%	144, 40%	203, 56%	186, 51%	227, 63%	173, 48%	207, 57%	240, 66%	250, 69%	240, 66%	240, 66%	260, 72%
Number of years with at least one clear spring* observation	22, 88%	20, 80%	22, 88%	22, 88%	22, 88%	22, 88%	22, 88%	19, 76%	18, 72%	20, 80%	18, 72%	19, 76%
Number of years with at least one clear autumn* observation	23, 92%	25, 100%	24, 96%	25, 100%	22, 88%	23, 92%	24, 96%	24, 96%	221, 84%	20, 80%	21, 84%	21, 84%

*For this analysis spring: September–November; autumn: January–April

Regardless of the high error rate due to cloud cover, the available data provided between 102 (Dip Swamp) and 218 (South Bool 2) clear observations from which to recreate surface water hydrographs (Figure 3.1). The number of remotely sensed surface water observations able to be obtained roughly compare to the overall number of manually (4 times annually) monitored groundwater observation wells over the same time span of data capture. Vegetation obstruction was considered a likely influence for the Taylors Swamp site, where surface water levels to full supply level (FSL) were unable to be detected from the WOfS data during the same period as other wetlands were all inundated to FSL (Figure 3.1).
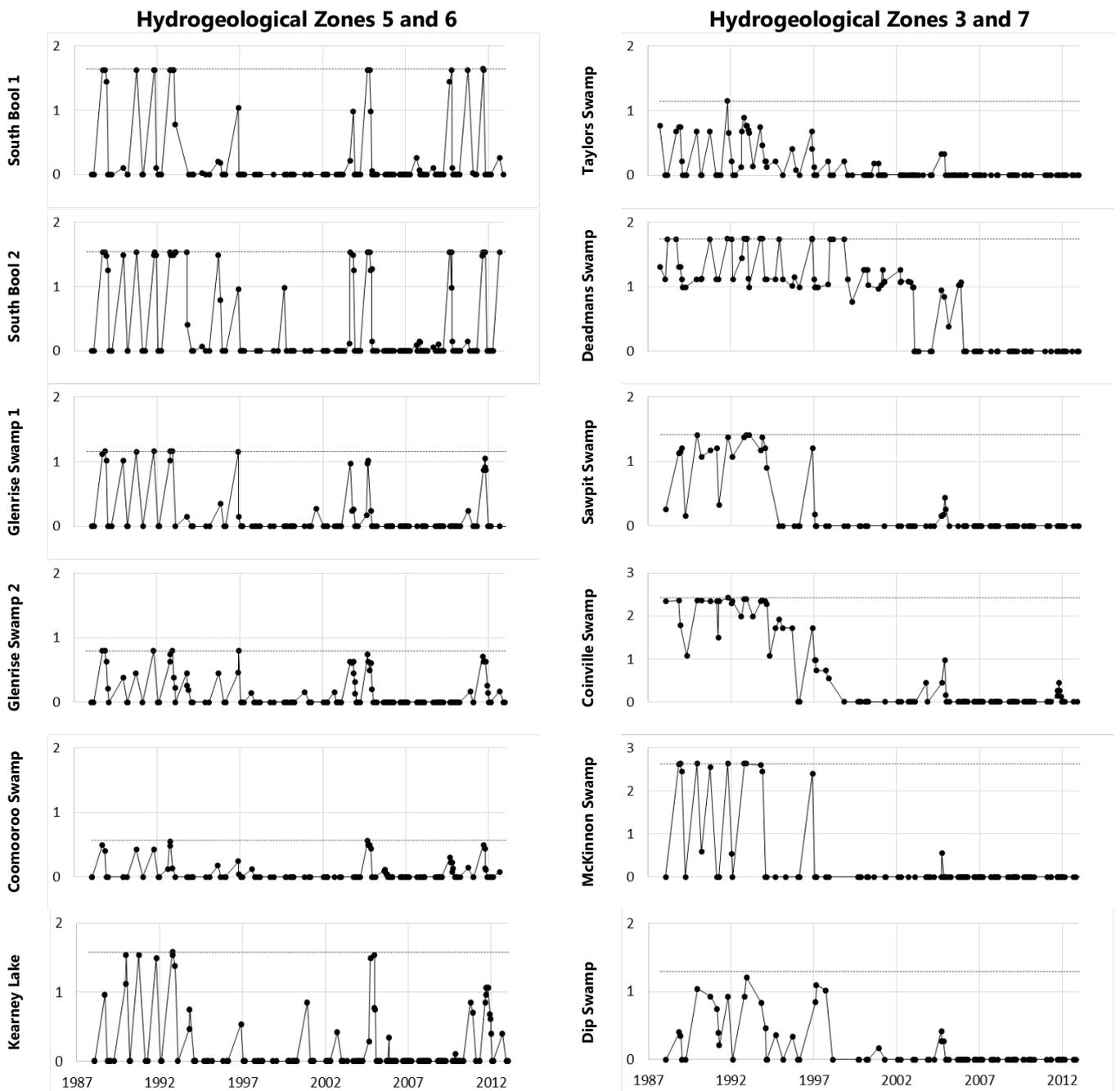
Figure 3.1.   Surface water hydrographs produced from WOfS and LiDAR DEM analysis for a selection of wetland basins in the study area between 1987 and 2013. Y-axis is surface water level (m). Dashed line shows wetland basin full supply level (FSL)

Validation of the resulting derived hydrographs with existing surface water monitoring data was impeded by lack of on-ground surface water observations for wetlands where surface water was also detectable via remote sensing. The Deadmans Swamp site has had surface water monitoring infrastructure installed since 2009 (SMK, 2010), however this has largely coincided with a period where the wetland has been dry (Figure 3.2). The comparison of observed and WOfS derived hydrographs for this site is therefore unable to validate the accuracy of WOfS derived surface water levels (when present), with only very shallow inundation event (< 0.2 m) recorded in 2011 which was not detectable via the WOfS data analysis. Underestimations of the extent of water in locations that contain mixed water and vegetation in pixels was identified as a limitation of the WOfS product by Mueller et al. (2016). Additionally, small areas of inundation (relative to a WOfS pixel size of 25 x 25 m) would also account for non-detection of small areas of shallow inundation. The comparison of observed and WOfS derived surface water data for Deadmans Swamp does however

indicate that WOfS was reliable for the detection of absence of water, with 4 of the 5 'dry' gauge board observations correctly identified as dry by the WOfS outputs (Figure 3.2).
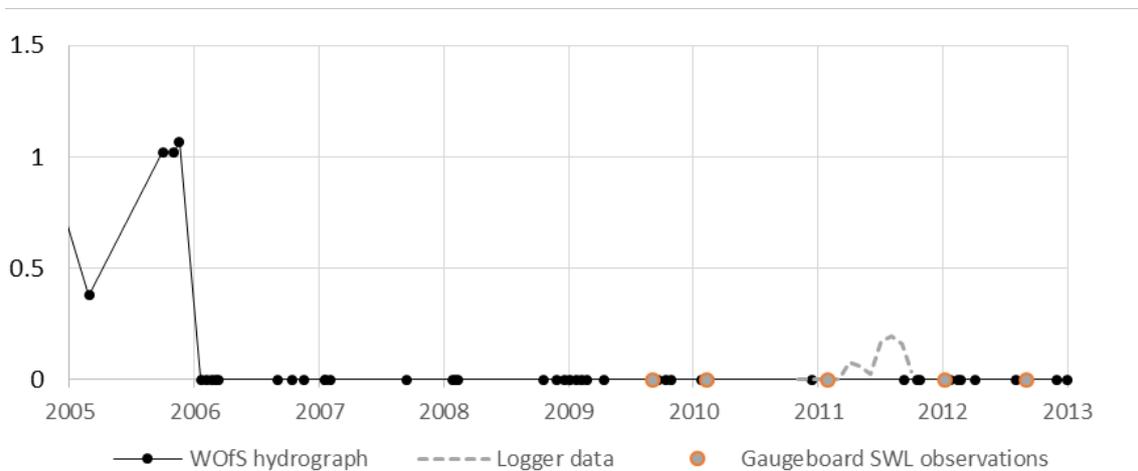
Figure 3.2.   Comparison of WOfS derived hydrograph data with surface water observation data available between 2009 and 2013 (Deadmans Swamp)

Surface water hydrographs for wetlands have been separated into two distinct groups based on similar hydrograph behaviour and location relative to hydrogeological zones (Figures 3.1 and 3.3) Those located in shallow groundwater areas of the lower elevation flats are within HZs 5 and 6, and those on the relatively higher elevations of the Naracoorte Ranges on eastern side of the Tartwaup fault are within HZs 3 and 7 (Figure 1.1). There are six selected wetland sites in each of the HZ groups (refer to Figure 1.1 for locations).

Figure 3.3 presents summarised annual hydro-period groups determined from the WOfS hydrographs for each of the analysed wetlands within both HZs in conjunction with rainfall (cumulative deviation from mean rainfall (CDMR)) trends from rainfall stations located within the zones and nearby the selected wetlands. Hydro-period groups were divided between permanent (water present in the wetland for entire 12-month period), seasonal (wetland wets and dries within a 12 month period), and dry (no water detected). Distinction is made as to whether or not the surface water level reached the FSL (as indicated by the DEM).

General trends observed in HZs 5 and 6 (Figure 3.3) indicate wetlands with seasonal wetting and drying phases, generally reaching FSL from 1987 through to approximately 1992. The majority of wetlands remained dry throughout the period 1997–2002, coinciding with below average rainfall. Between 2003 and 2004 water levels recovered briefly, with the majority of wetlands reaching FSL again in 2004 in response to a higher rainfall year. The following years of 2005–10 indicate another period where the majority of wetlands were dry, again coinciding with below average rainfall, however surface water inundation returns (although not all wetlands are reaching FSL) with rainfall recovery from 2009. The period where wetlands failed to inundate generally coincides with the time period referred to as the Millennium drought which affected much of southern Australia, influenced by strong El Nino weather patterns (Australian Government Bureau of Meteorology, 2015). As such, wetland hydro-periods within HZs 5 and 6 appear to closely align with rainfall variation as do groundwater level variations (Cranswick, 2018).

The majority of selected wetlands in HZs 3 and 7 were permanently inundated through approximately 1987 to 1993 (Figure 3.3). Trends in wetland hydro-periods observed in HZs 3 and 7 were similar to HZs 5 and 6 until approximately 2009 where surface water inundation failed to return after the end of the Millennium drought. The high rainfall year in 2004 also indicates that wetlands in Zones 3 and 7 failed to fully recover hydro-periods to prior FSLs, unlike those in HZs 5 and 6. Of these selected sites, the last evidence of any areas of permanent water was recorded in 2004. Rainfall declines in HZs 3 and 7 vary more widely, with the most northerly rainfall station (Wrattonbully) showing greater declining trends. Despite the variation in rainfall trends, the difference in hydro-period recovery from approximately 2009 onwards between the two zone groups indicates that other factors in HZs 3 and 7, may have had a substantial impact on wetland hydro-periods, in addition to the impact due to rainfall variation.
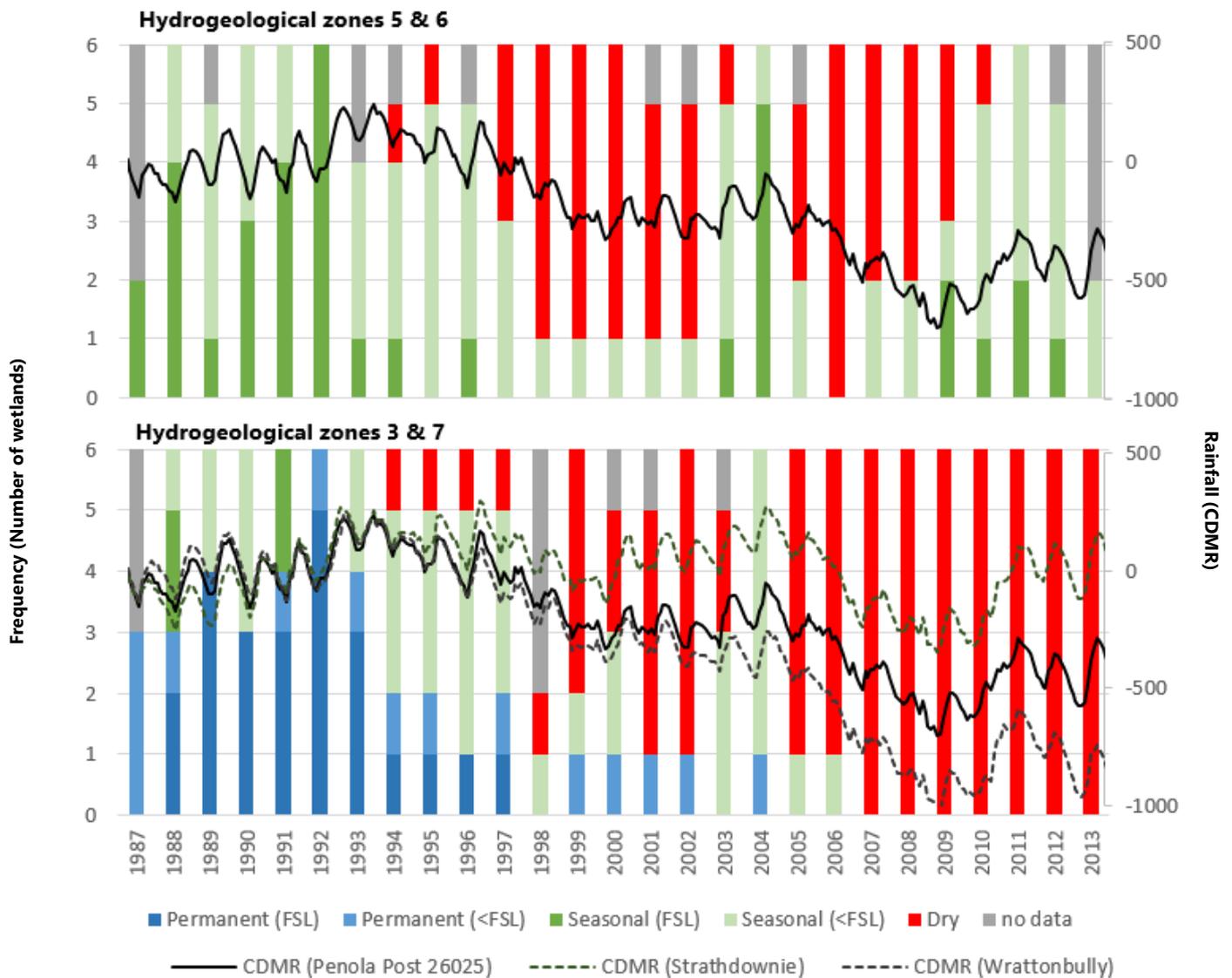
Figure 3.3. **Comparison of annual changes in hydro-periods for a selection of 12 wetland basins in Hydrogeological Zones 5 & 6, and 3 & 7 in relation to rainfall.** *Permanent (FSL):* **Wetland reached FSL and at least some area of the wetland remained inundated throughout the entire 12 month period;** *Permanent (<FSL):* **At least some area of the wetland remained inundated throughout the entire 12 month period, but did not reach FSL;** *Seasonal (FSL)*: **Wetland both wets and dries, and reached FSL at some time during the 12 month period;** *Seasonal (<FSL)*: **Wetland both wets and dries, but does not reach FSL during the 12 month period;** *Dry*: **Wetland did not record any surface water inundation during the 12 month period;** *no data*: **no WOfS observations available to determine seasonal high water level.** *CDMR*: **Cumulative deviation from mean rainfall (mean calculated from 1980 to 1995).**

3.2 Linear relationships between groundwater level and surface water expression

Linear regression equations were developed for each of the 12 selected wetland sites, representing relationships between WOfS derived surface water hydrographs and nearest long-term groundwater level monitoring in the regional unconfined aquifer, and are presented in Appendix C (regression lines for each shown in Figure 3.4). In the majority of cases, surface water expression at sites indicated relatively strong linear relationships ($R^2 > 0.7$) with the observed unconfined aquifer level. Groundwater levels are generally shown to be higher than wetland bed levels during surface water expression (Appendix C and Figure 3.4). There is the potential to introduce some uncertainty at the site scale due to the use of remotely derived surface water hydrographs, monitored groundwater level data from

observation wells that were not specifically designed for the purpose of establishing empirical relationships with surface water features, and the assumption of direct groundwater–surface water interaction in the absence of significant surface water runoff. Nevertheless, the analysis suggests that the 12 wetlands are likely to be largely seasonal groundwater discharge GDEs with strong relationships between groundwater levels and surface water expression in the study area. This is also evidenced by and in agreement with other individual site analyses in the study region (Taylor et al., 2015; Harding et al., 2015; Deane et al., 2017a) and the regional-scaled GDE likelihood classification (SKM, 2009).
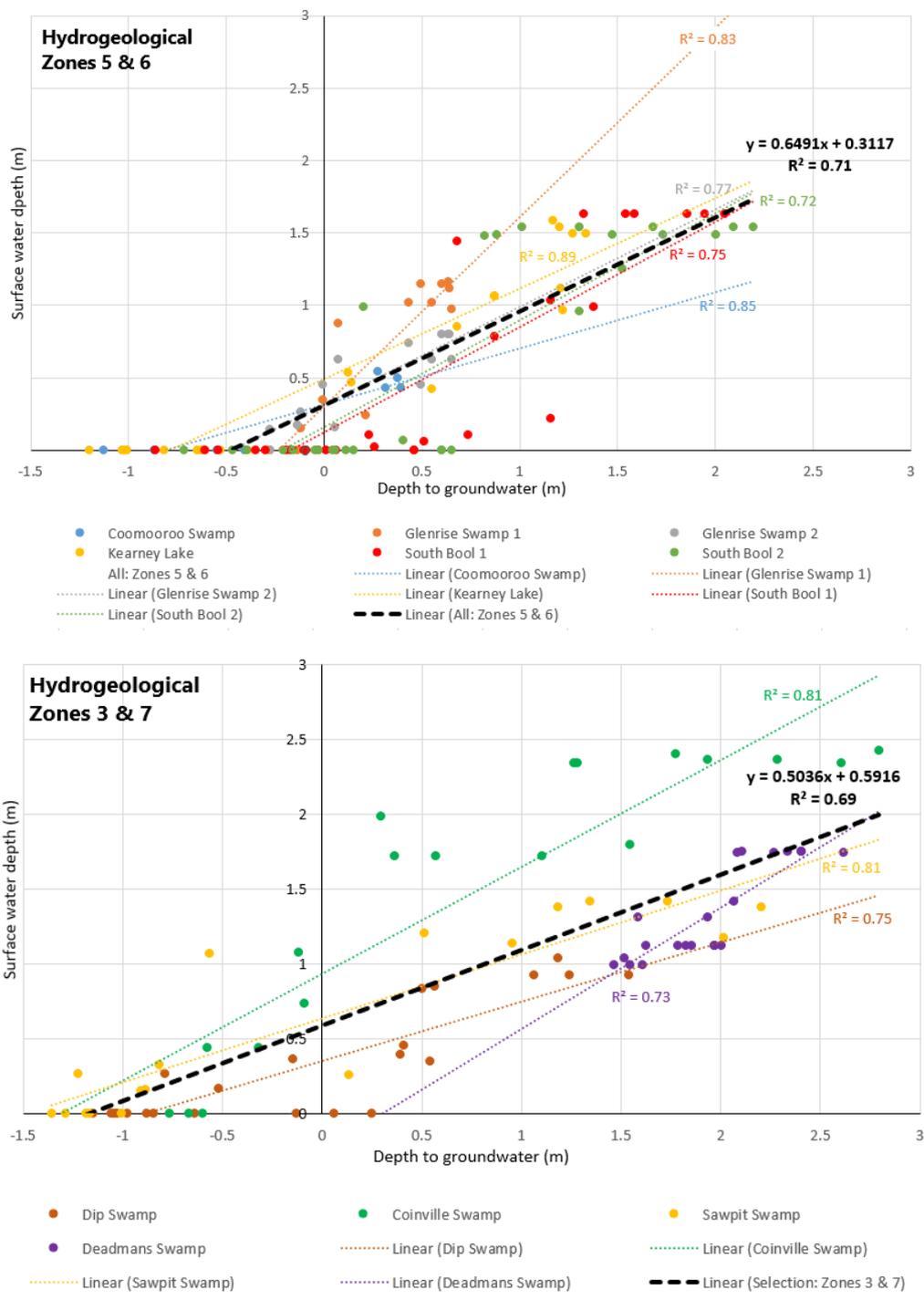
Figure 3.4. Correlation of surface water depth in wetlands and depth to groundwater within the two hydrogeological zone groups

Given the limitations of the data used, general empirical relationships between groundwater and surface water level at the hydrogeological zone group scale could be used as an estimate of general surface water response in GDEs to

groundwater change (Figure 3.4). Wetland GDEs in HZs 5 and 6 exhibit strong relationships between surface water presence and groundwater level above ground surface (i.e. groundwater discharging conditions, also shown by Taylor et al. (2015)). The majority of wetlands investigated in this zone group were shown to be dry (zero surface water depth) at times when groundwater level is on average more than 0.3 m below ground level (Figure 3.4). Combined linear regression of surface and groundwater level data for all analysed wetlands in HZs 5 and 6 gives the equation:

$$\text{Surface water level} = \text{Groundwater level} \times 0.65 + 0.31 \qquad (R^2 = 0.71)$$

This relationship explains over 70% of the variation in surface water level (Figure 3.4). Groundwater and surface water level linear regressions for analysed wetlands in HZs 3 and 7 displayed wider variation across the zone group, likely influenced by variations in local landuse, topographic and groundwater conditions. Generally, wetlands recorded 0 surface water inundation (dry) as groundwater declined on average more than 0.5 m below ground level. Combined linear regression of surface water and groundwater level data for all analysed wetland GDEs in HZs 3 and 7 resulted in the equation (Figure 3.4):

$$\text{Surface water level} = \text{Groundwater level} \times 0.5 + 0.59 \qquad (R^2 = 0.69)$$

Similarly, strong discharging conditions (where groundwater elevations exceeded wetland bed level during surface water expression) were also observed for the selected wetlands in HZs 3 and 7.

These relationships should however be interpreted with reference to the assumptions of the analysis, including the assumptions regarding the absence of clogging layers and insignificant inputs from surface water runoff. Consequently, and without detailed conceptualisation of each of the wetland sites, the causal relationship between groundwater level and surface water inundation may in reality be more complex.

3.3   Surface water scenario simulations

A spatial interpretation of WOfS derived changes in hydrometrics for each of the selected wetland GDE sites at the DEM pixel scale (2 x 2 m pixel) are provided in Appendix D, presented in averaged 10-year epochs.

In HZs 5 and 6, seasonally inundated extent (area) of wetlands averaged between 38 to 95% of maximum possible inundated area at FSL in the most recent epoch (2005–15) (Table 3.2). Shallow wetlands (e.g. Glenrise Swamps and Coomooroo Swamp) indicate relatively large reductions in area inundated (with only an average of 38 to 63% of original wetland extents inundated in 2005–15 epoch) due to declining maximum surface water levels. In contrast, the deeper, flat-bottomed topography of South Bool deflation basins remain inundated over more than 90% of their original area at FSL, even when only reaching 38–43% of the maximum obtainable surface water depth (Table 3.2). The influence of basin bathymetry on total wetland area loss, as a result of surface water level decline was also observed by Deane et al. (2017a) indicating that for some wetland basins, areal extent inundated can be a poor indicator of hydrological change.

Both mean maximum water depths and mean area inundated declined at a relatively steady rate over the three 10-year epochs in HZs 5 and 6 (Figure 3.5). Mean hydrometrics show wetlands at near FSL, inundating every year (10 in 10 years) in the 1985–95 epoch, declining in frequency to between 3–8 in 10 years and between 17 and 32% of maximum obtainable surface water depths in the 2005–15 epoch (Figure 3.5). Frequency of inundation was also most significantly altered for shallower wetland basins (Glenrise Swamps and Coomooroo Swamp) (Table 3.2 and Appendix D).

Surface water trends within selected wetland GDEs in HZs 3 and 7 are significantly different to the trajectories of those in HZs 5 and 6, with the most significant changes to area inundated, depth and frequency occurring in the 2005 to 2015 epoch (Figure 3.5). The majority of wetland GDEs analysed in HZs 3 and 7 averaged zero, or near-zero surface water depths (dry) during this period (Appendix D and Table 3.2). Frequency of inundation has declined from 10 in 10 years for all analysed wetlands in the 1985–95 period, to 0–3 in 10 years in the 2005–15 period. Mean area inundated and mean surface water depths have also declined to a small fraction of the original extent (Table 3.2).
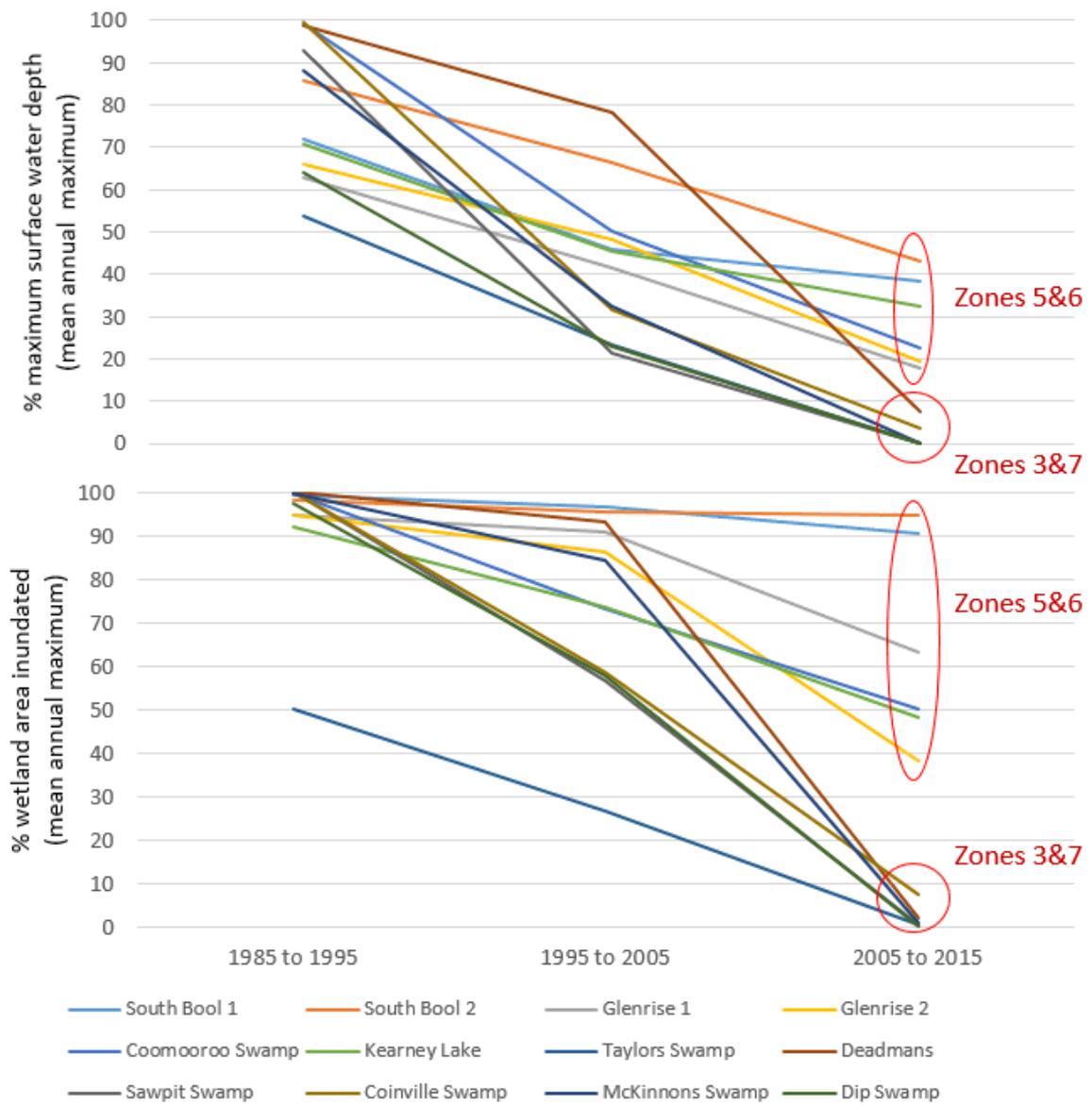
Figure 3.5. Percentage change in mean annual surface water depth and wetland area (from FSL) over three 10-year epochs

Table 3.2. Summary of 10-year epoch hydrometrics for selected wetland sites with use of WOfS-derived hydrographs and empirical relationships between groundwater and surface water expression

	Sites	Hydrogeological Zones 5 and 6						Hydrogeological Zones 3 and 7					
		South Bool 1	South Bool 2	Glenrise Swamp 1	Glenrise Swamp 2	Coomooroo Swamp	Kearney Lake	Taylors Swamp	Deadmans Swamp	Sawpit Swamp	Coinville Swamp	McKinnons Swamp	Dip Swamp
Mean annual maximum ha inundated (% inundated from FSL)	1985-1995	11.96 ha (99.25%)	5.73 ha (98.1%)	5.15 ha (94.58%)	11.86 ha (94.9%)	6.91 ha (99.57%)	24.14 ha (92.08%)	2.71 ha (50.11%)	11.24 ha (100%)	15.0 ha (99.94%)	6.99 ha (100%)	82.40 ha (99.55%)	14.59 ha (97.51%)
	1995-2005	11.65 ha (96.67%)	5.58 ha (95.48%)	4.94 ha (90.86%)	10.8 ha (86.37%)	5.08 ha (73.18%)	19.3 ha (73.53%)	1.43 ha (26.45%)	10.47 ha (93.17%)	8.5 ha (56.61%)	4.10 ha (58.44%)	69.67 ha (84.17%)	8.62 ha (57.61%)
	2005-2015	10.9 ha (90.5%)	5.53 ha (94.59%)	3.43 ha (63.11%)	4.76 ha (38.11%)	3.47 ha (50.03%)	12.62 ha (48.13%)	0.03 ha (0.54%)	0.24 ha (2.16%)	0.05 ha (0.36%)	0.51 ha (7.33%)	0.57 ha (0.69%)	0 ha (0%)
Mean maximum depth (m) (% of depth at FSL)	1985-1995	1.65 m (72.04%)	1.32 m (85.6%)	0.69 m (62.63%)	0.52 m (65.81%)	0.56 m (99.27%)	1.12 m (70.6%)	0.62 m (53.85%)	1.76 m (98.58%)	1.31 m (92.77%)	2.49 m (99.53%)	2.4 m (87.97%)	0.82 m (63.83%)
	1995-2005	0.76 m (45.92%)	1.02 m (66.38%)	0.46 m (41.67%)	0.38 m (48.32%)	0.28 m (50.1%)	0.72 m (45.28%)	0.27 m (23.5%)	1.4 m (78.15%)	0.3 m (21.27%)	0.79 m (31.7%)	0.88 m (32.34%)	0.3 m (23.0%)
	2005-2015	0.63 m (38.32%)	0.66 m (43.03%)	0.19 m (17.65%)	0.15 m (19.48%)	0.13 m (22.36%)	0.51 m (32.5%)	0 m (0%)	0.13 m (7.55%)	0.002 m (0.17%)	0.09 m (3.7%)	0 m (0%)	0 m (0%)
Maximum frequency of inundation (% years inundated)	1985-1995	10 in 10 years (100%)	10 in 10 years (100%)	10 in 10 years (100%)	9 in 10 years (90%)	10 in 10 years (100%)	10 in 10 years (100%)	10 in 10 years (100%)	10 in 10 years (100%)	10 in 10 years (100%)	10 in 10 years (100%)	10 in 10 years (100%)	10 in 10 years (100%)
	1995-2005	10 in 10 years (100%)	10 in 10 years (100%)	6 in 10 years (60%)	8 in 10 years (80%)	7 in 10 years (70%)	9 in 10 years (90%)	8 in 10 years (80%)	9 in 10 years (90%)	8 in 10 years (80%)	9 in 10 years (90%)	8 in 10 years (80%)	8 in 10 years (80%)
	2005-2015	8 in 10 years (80%)	8 in 10 years (80%)	3 in 10 years (30%)	3 in 10 years (30%)	5 in 10 years (50%)	7 in 10 years (70%)	1 in 10 years (10%)	3 in 10 years (30%)	1 in 10 years (10%)	2 in 10 years (20%)	0 in 10 years (0%)	0 in 10 years (0%)

3.3.1 Estimated changes in surface water from groundwater level scenarios: Deadmans Swamp and Dip Swamp

The linear relationships between groundwater and surface water established for two GDEs in HZs 3 and 7 (Deadmans Swamp and Dip Swamp – refer to Appendix C) were used to estimate the likely contribution of rainfall variability alone on the observed significant decline in surface water presence.

Deadmans Swamp is located in the interdunal depression of the Naracoorte Ranges, within aeolian sand dunes (Taylor et al. 2015). The topography and sandy soils local to the site suggests limited local catchment runoff is likely. Limited data suggest that some clay sediments are found beneath the southern basin of Deadmans Swamp (SKM, 2010), however for this analysis, we assume it does not significantly limit groundwater–surface water exchange. Similarly, Dip Swamp has limited potential for surface water runoff, located in the topographic flats of Dismal Swamp complex, in an area dominated by sandy soils. There is no information on Dip Swamp regarding potential for presence of clogging clay sediments. As such, for both sites we assume insignificant surface water runoff potential and insignificant clogging layers between the wetland and unconfined aquifer. Neither assumption has been comprehensively tested, therefore the analysis based on linear relationships should be viewed within the constraints of the assumptions.

Estimated groundwater responses to rainfall trends from hydrograph regression analysis provided two groundwater level hydrograph scenarios for each site. These were sourced from Cranswick (2018): 1) Estimated rainfall and trend (with time trend); 2) Estimated rainfall only (without time trend) (Appendix B). Figure 3.6 illustrates the estimated surface water response for both groundwater scenarios for Deadmans Swamp (regression analysis for observation well JOA005) and Dip Swamps (regression analysis for observation well MIN015). Mean wetland surface water hydrometrics (maximum annual water depth and frequency) were calculated for both scenarios and are displayed spatially in 10-year epochs, compared to that observed from WOfS derived hydrograph analysis (Figure 3.7).

Scenario 1 (simulation of surface water expression from rainfall and a non-rainfall time trend) indicates that the applied linear relationships to the groundwater hydrograph regression analysis reasonably predict the WOfS hydrograph output trends, although often fail to predict the full magnitude of seasonal variation, instead approximating average or seasonal maximum surface water levels (Figure 3.6). As such, the results are potentially best interpreted as indicating average water levels over 10-year time intervals (Figures 3.7 and 3.8), rather than absolutes. Both sites show Scenario 2 (simulation of expected surface water expression from rainfall variation alone) plotting well above the observed surface water levels (Figure 3.6). Decreasing surface water level trends are evident in the 2005–15 period (corresponding with a period of below average rainfall), however the scenario indicates that surface water expression would have been expected to occur annually throughout the dry period (although no longer reaching FSL), and show at least partial recovery post-2010. The simulated hydrographs for Scenario 2 do not align with the WOfS surface water level data (Figure 3.6), implying that, as also indicated in initial analysis of hydrometric trends (Figures 3.3 and 3.5), impacts from regional groundwater resource usage (i.e. potentially plantation forestry and extraction for irrigation) is likely to be involved in the observed decline in surface water expression at these sites providing the broad assumptions of the analysis are true.

At the Deadmans Swamp GDE site, comparisons between Scenarios 1 and 2 at the 10-year epoch time-scale indicate that rainfall was likely the main driver of changes in inundation between the 1985–95 to 1995–2005 epochs, with <13% difference in maximum surface water extent and mean annual maximum depth (Figures 3.7 and 3.8) between the two. In the 2005–15 epoch, there is some deviation between the two scenarios, where inundated area was estimated to be ~30% greater, and maximum surface water depth ~50% greater in Scenario 2 (Figure 3.8). These estimations indicate that surface water inundation in Deadmans Swamp may have been substantially impacted by rainfall variability alone before 2005–15 (Figures 3.6 and 3.7), however would have likely maintained shallow seasonal surface water expression of groundwater. The observed absence (or minimal expression) of surface water expression of groundwater in the 2005–15 epoch was unable to be accounted for by rainfall variability alone in this analysis, suggesting that other groundwater resource impacts (i.e. both direct extraction and reduced recharge under forestry) are at least partially contributing to the current reduction in surface water availability to this site.
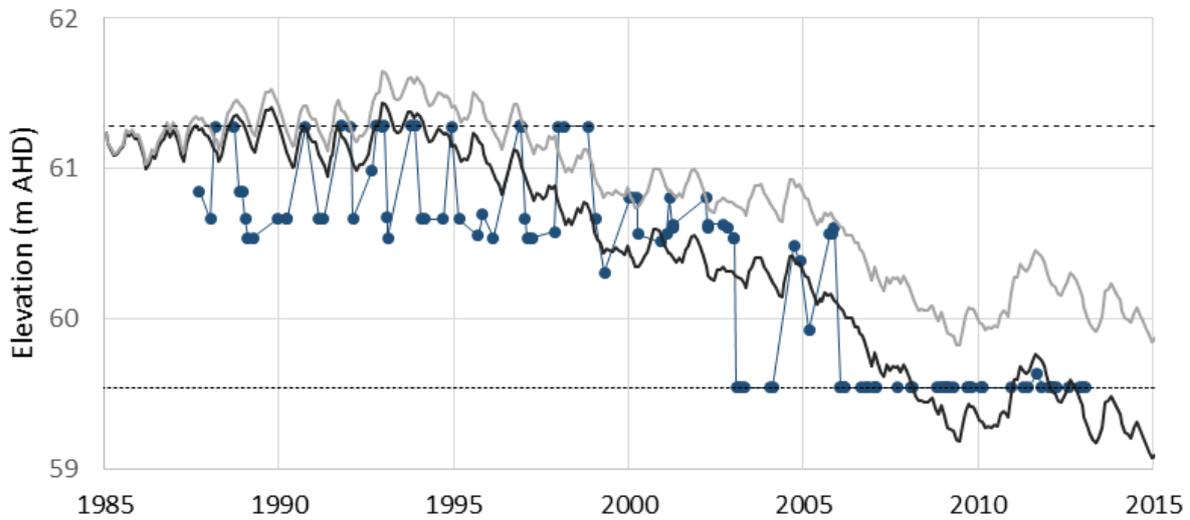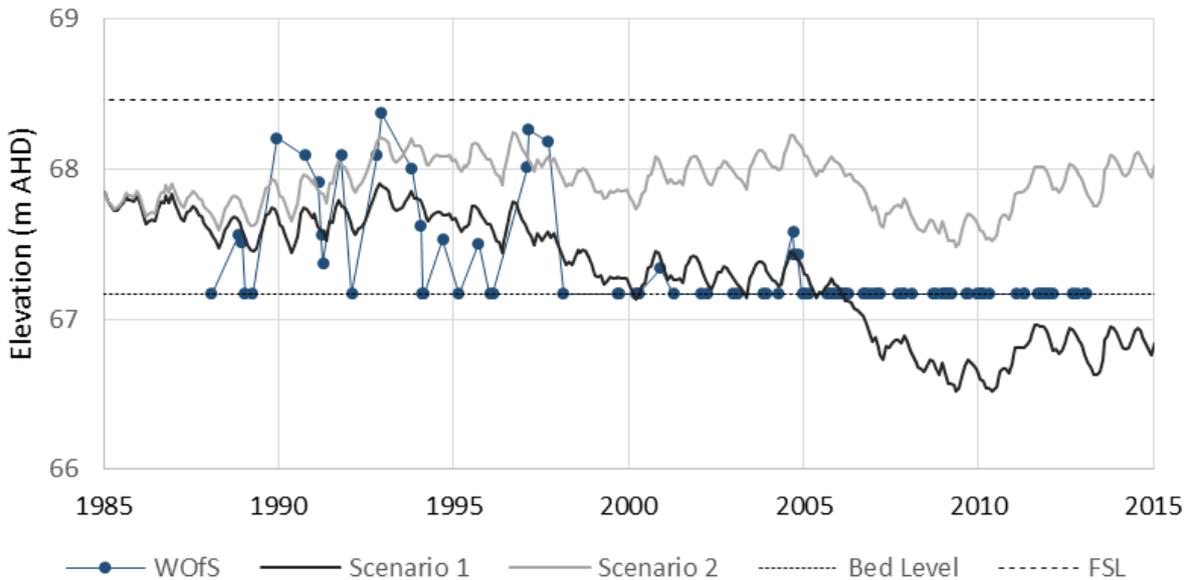
Figure 3.6. **Estimated surface water hydrographs for Deadmans Swamp and Dip Swamp from hydrograph regression derived groundwater level scenarios. Scenario 1: estimating observed surface water (rain + trend); Scenario 2: estimating surface water response to rainfall variation (rain only); WOfS: Surface water hydrographs produced from WOfS analysis; Bed Level: lowest elevation of the wetland basin; Full Supply Level (FSL).**

Comparison between Scenarios 1 and 2 for the Dip Swamp GDE site indicate more substantial differences between 1985–95 and 2005–15 epochs (Figures 3.7 and 3.8) than that of Deadmans Swamp. Scenario 2 (surface water expression as a result of rainfall variation alone) estimates that only minor changes in wetland hydrometrics between all three epochs were expected, indicating that above 90% of the observed reduction in wetland area inundated since 1985–95, and more than 50% of maximum surface water levels in the most current epoch were unexplained by rainfall variation (Figure 3.8). Therefore, the loss of surface water expression of groundwater at Dip Swamp observed throughout the 2005–15 epoch is likely a result of groundwater resource usage and landuse change in the region.

Due to the generalisations made at the conceptual level for this analysis, and in the use of groundwater hydrograph regression analysis derived groundwater level scenarios, there is a possibility that the impact of groundwater resource impacts (other than rainfall) could be over or under estimated. However, the observed significant declines in

groundwater levels have undoubtedly had an influence on groundwater–surface water interactions in the region. The cause of the decline in groundwater levels, and hence surface water expression, may also be, in part, a function of changing soil water, and local runoff processes unable to be captured in this scale of analysis.

3.4 Estimation of loss of aquatic ecosystem values as a result of groundwater resource use

Figure 3.9 illustrates the difference in mean maximum surface water level between predictions for Scenarios 1 and 2 for Deadmans Swamp and Dip Swamp for the most current epoch (2005–15). The scenario analysis of Deadmans Swamp and Dip Swamp indicated a mean seasonal maximum water level difference of estimated -0.66 m and -0.76 m respectively was unexplained by rainfall variation alone in this epoch (Figure 3.9). Maximum surface water level is a significant hydrological predictor in the establishment of wetland plant communities (Casanova & Brock, 2000; Deane et al., 2017b). Brookes et al. (2017) found that a decline in surface water level of 0.3 m resulted in the loss of at least one distinct wetland vegetation community, with declines of between 0.6–0.9 m resulting in the loss of all aquatic vegetation in a majority of wetlands analysed, largely influenced by basin bathymetry (Deane et al., 2017a). Similar thresholds of groundwater level change (0.25–0.3 m, equivalent to ~0.2–0.38 m surface water reduction) were found by Deane et al. (2017a) to result in clear changes in predicted aquatic plant functional group spatial distribution. The magnitude of predicted changes were also broadly consistent with risk categories developed for wetland vegetation due to groundwater decline (low: <0.25 m, moderate 0.25–0.5 m, high 0.5–0.8 m) based on long-term monitoring data (1996–2010) from the Swan Coastal Plain, Western Australia (Loomes et al., 2013).

If we cautiously assume that thresholds between plant functional groups (species with similar hydro-niche requirements) are determined by ~0.25 m maximum surface water depth ranges, in the Scenario 2 prediction, Dip Swamp would potentially have up to four depth hydro-niches supporting distinct aquatic vegetation communities, and Deadmans Swamp up to three depth hydro-niches. These are broadly spatially matching the distributions of 0.25 m increment surface water level change shown in Figure 3.9. The current vegetation at Dip Swamp has almost entirely transitioned to terrestrial plant species (primarily pasture grasses), whilst Deadmans Swamp maintains small areas of shallow (< 0.25 m) ephemeral inundation during high rainfall periods, with the majority of the wetland basin gradually transitioning to a species composition less tolerant of inundation (Clarke et al., 2015). Application of hydrological-niche models for plant functional groups (Deane et al., 2017a; 2017b) to predict ecosystem response offers the potential to further model response. This would make use of multiple hydrometrics which could be derived from WOfS hydrographs which were not able to be directly investigated as part of this project. It can be concluded that significant loss of aquatic ecosystem values has already occurred due to declines in the potential maximum surface water levels obtainable (in the absence of impacts other than climate) of > 0.6 m, resulting in largely dry wetland basins.
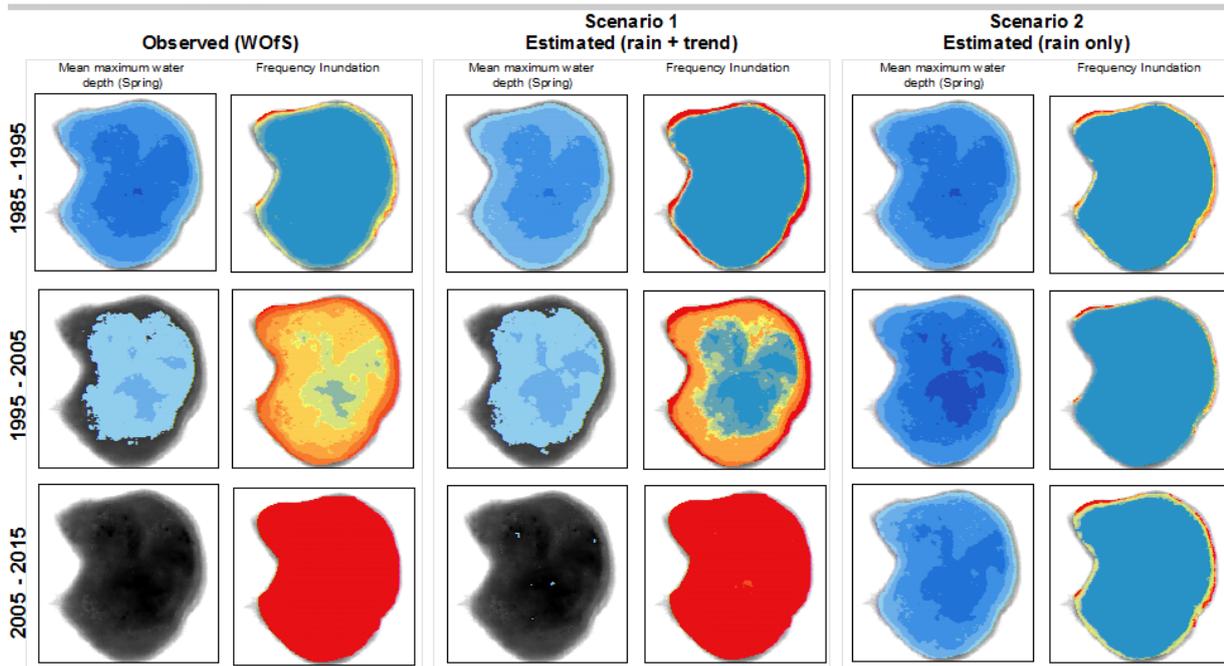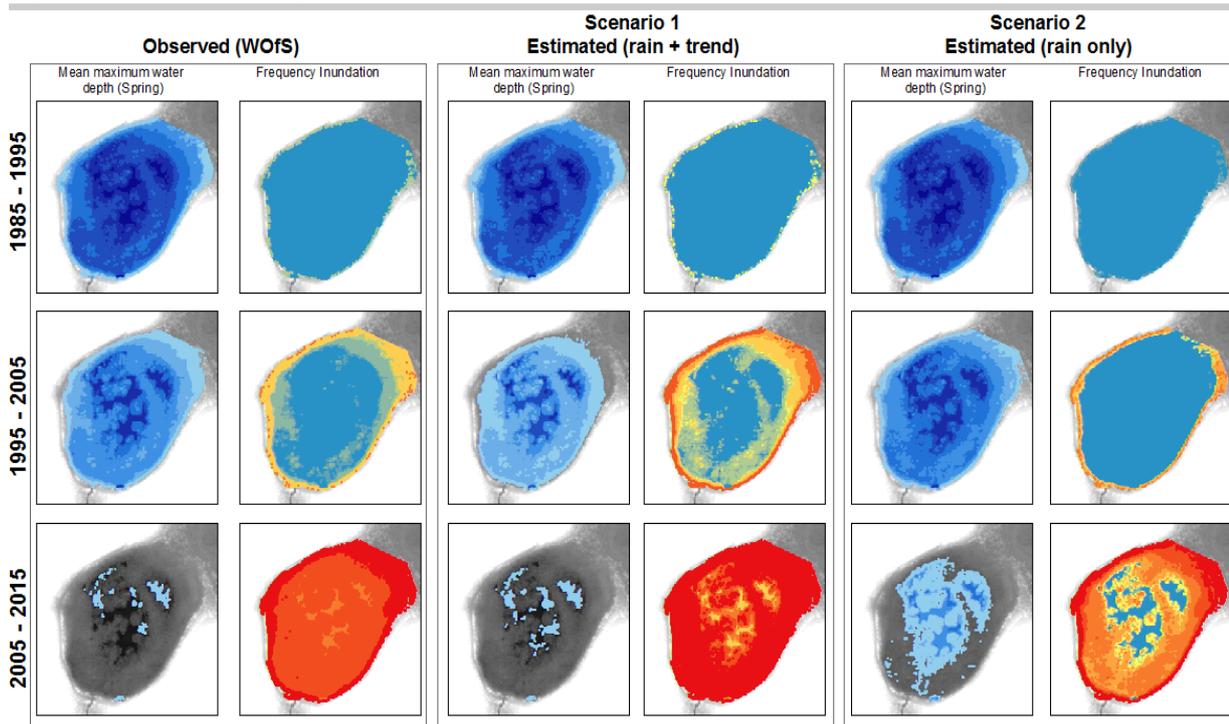
Figure 3.7. Estimated spatial surface water response to groundwater level scenarios at two wetland GDEs. Scenarios produced from hydrograph regression analysis: Deadmans Swamp: hydrograph regression analysis using observation well JOA005 and Wrattonbully rainfall station. Dip Swamp: hydrograph regression analysis using observation well MIN015 and Strathdownie rainfall station.
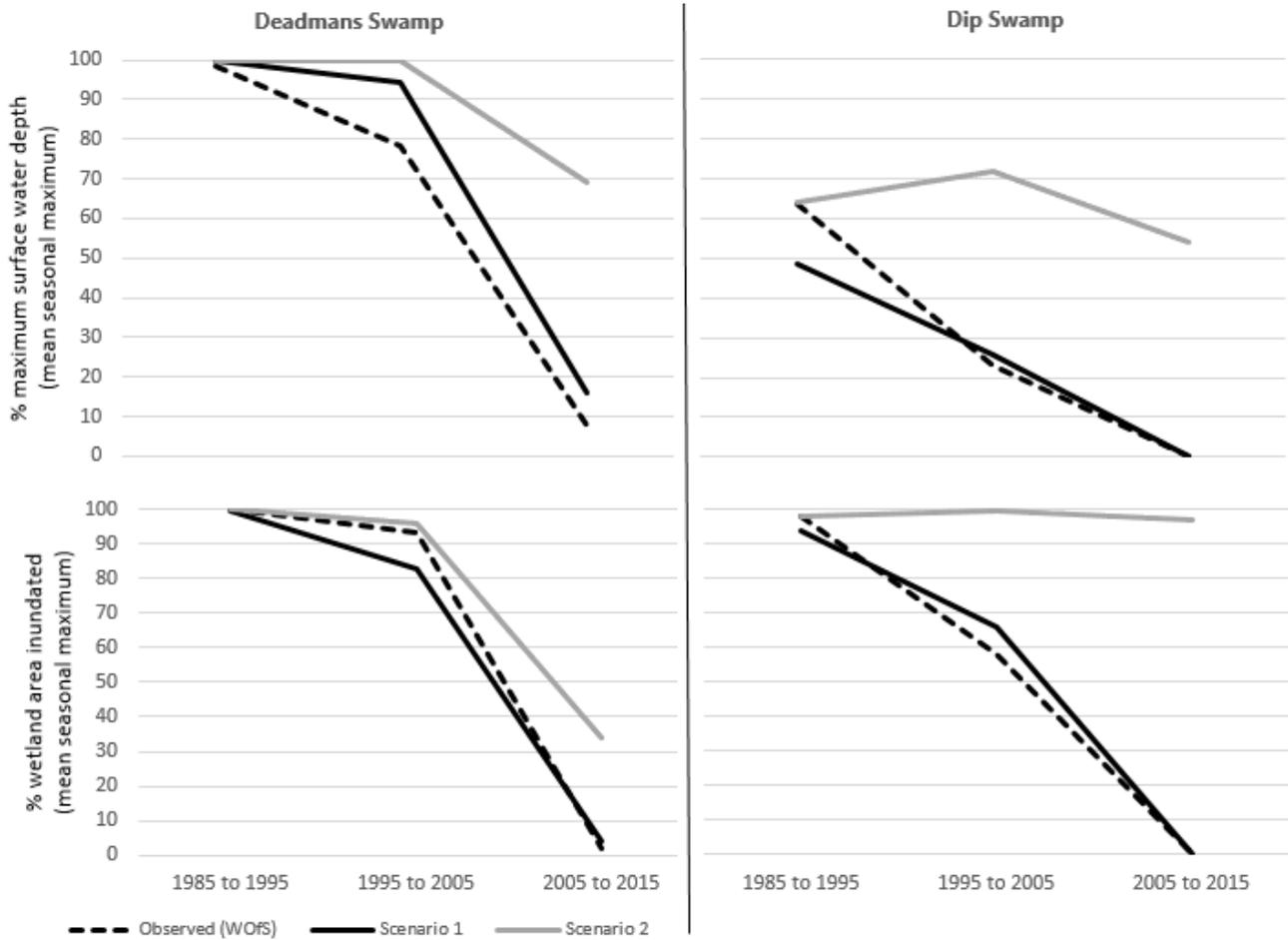
Figure 3.8. Comparison of percentage change in surface water depth and wetland area (from FSL) over three 10-year epochs for Scenarios 1 and 2 at the Deadmans Swamp and Dip Swamp wetland GDE sites
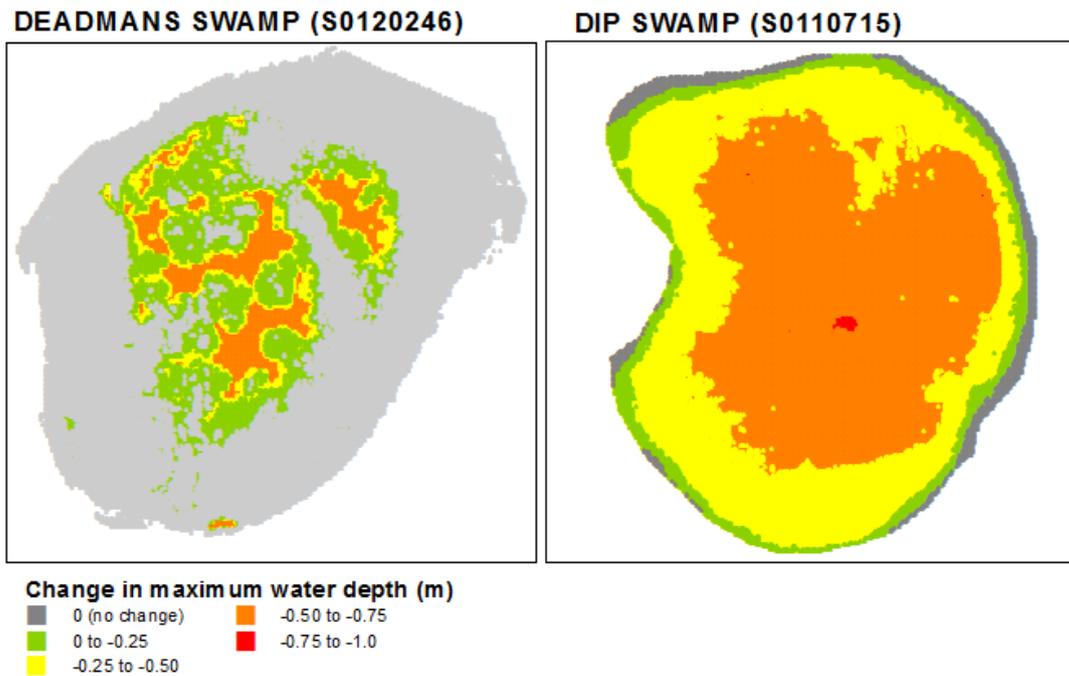
Figure 3.9. Difference in mean maximum water depth between Scenarios 1 and 2 for the 2005–15 epoch presented in 0.25 m thresholds

4 Conclusion

4.1   WOfS surface water hydrographs

The Water Observations from Space (WOfS) product provides a nationally consistent tool for understanding surface water across Australia both spatially and temporally (Mueller et al., 2016). We have shown that outputs from Geoscience Australia's WOfS products combined with a LiDAR DEM can be used to approximate historic surface water hydrographs for open water wetland basins. These data can be used to analyse trends in both spatial and temporal hydro-periods and hydrometrics (depth and frequency) of wetlands, at a scale suitable for determining broad surface water–groundwater relationships. Cloud and cloud shadow errors were significant, especially given the South East NRM region's cool temperate climate conditions. However analysis indicated that it was possible to obtain both maximum and minimum annual water levels for the majority (>80%) of years over the WOfS data capture time-span.

Obstruction of remote water observations from dense vegetation is commonly reported as an issue with the use of remote sensing for wetlands (Jones, 2015; Turner et al., 2015; Mueller et al., 2016). The use of WOfS data products for recreating hydrographs is likely to only perform well for wetland basins that are not significantly vegetated (either naturally or artificially), as were the majority of sites selected for the analysis presented within this report. Vegetation obstruction in some of our case study sites was considered likely to cause some minor misclassification of shallow inundation as dry (e.g. Deadmans Swamp) and also result in potential masking of true maximum inundation extents (e.g. Taylors Swamp). We attempted to avoid significant issues from vegetation obstruction by purposefully selecting wetland sites for analysis that were located within the agricultural matrix, where stock grazing and other vegetation clearing activities had removed dense surrounding and emergent plants.

The analysis of several wetlands within a land system (with similar hydrogeology, hydrology, climate, topography and landuse setting) enables any general trends in both temporal and spatial surface water behaviour to become apparent. These trends could then be used as surrogates for hydrological responses in similar wetland types within a land system, regardless of vegetation. The use of WOfS derived data in conjunction with a LiDAR DEM to produce hydrographs also improves the usefulness of remotely sensed products for monitoring spatial changes in wetland surface water inundation. We found that wetland basin bathymetry significantly influenced the potential for monitoring hydrological change remotely. Deeper flat-bottom deflation basins typical of inland interdunal wetlands in the region (e.g. South Bool Lagoons) showed little change in inundation extent over the three 10-year epochs, despite an approximate 40% decrease in maximum surface water level. Similar findings regarding the influence of wetland bathymetry on the magnitude of change of inundation extent and hydro-niche distributions was found by Deane et al., (2017b). The approach also has the ability to overcome some cloud errors and vegetation obstruction (particularly for large wetland bodies), by recording the highest elevation of inundation and utilising the DEM to indicate spatial area inundated. By utilising the LiDAR DEM to convert spatial extent data to elevations, far greater capacity for remotely monitoring of changes in hydrometrics can be achieved and has the potential to be an efficient and cost-effective method of interpreting temporal surface water change at landscape scales. Improvements to the remotely sensed hydrological outputs could also be achieved with the addition of strategic site based surface water monitoring equipment to validate and calibrate remote methods and any subsequent upscaling of results.

The WOfS derived hydrographs for our selected wetland basins were grouped based on similar patterns in surface water expression over time and similar hydrogeological characteristics. The two groups were based on Harrington and Currie (2008) hydrogeological zones, amalgamating HZs 5 and 6 (lower elevation area on the Mosquito Creek flats), and HZs 3 and 7 (higher elevations from southern Naracoorte Ranges to the Tartwaup fault zone). Strong relationships between groundwater level and surface water expression (with R^2>0.7 for all except one site) were able to be established with use of the WOfS derived hydrographs and nearest groundwater monitoring observation well data in both zone groups. Groundwater levels in the unconfined aquifer were generally shown to be higher than wetland bed level during periods of surface water expression, indicating that the wetlands analysed were likely GDEs with strong connection to groundwater discharge and were shown to become dry when groundwater levels were on average > 0.3–0.5 m below the lowest elevation of the wetland bed.

4.2   Conclusions for Hydrogeological Zones 5 and 6

WOfS hydrograph analysis of wetlands within Hydrogeological Zones 5 and 6 indicated wetland GDEs with seasonal wetting and drying phases strongly related to rainfall responsive groundwater levels, where inundation periods appeared to closely align with rainfall variations. Periods where wetlands failed to inundate coincided with the time period referred to as the Millennium drought (Australian Government Bureau of Meteorology, 2015) where groundwater was shown to decline below the wetland bed levels, resulting in an absence of surface water expression. Surface water expression at least partially returned with rainfall and groundwater level recovery after approximately 2010. Significant changes in both mean maximum water depths and mean frequency of inundation was shown over the 30-year timeframe of analysis with frequency of inundation reducing from annual inundation in the 1985–95 epoch to between 3 and 8 in 10 years in the most recent 2005–15 epoch.

Whilst it is likely that groundwater extraction (for irrigation) from the unconfined aquifer is contributing to the observed decline in groundwater levels (Cranswick, 2018), we show that climate (rainfall variation) and its impact on groundwater recharge (and therefore groundwater level) is the primary driver of surface water change in GDEs for HZs 5 and 6. Projected future groundwater level declines in response to climate change predictions (Costar et al., 2017; Cranswick, 2018) will pose an increased risk of GDEs becoming more frequently dry due to loss of discharge from the unconfined aquifer. In view of this, the management of groundwater in these zones will likely require adjustment if groundwater levels are to be maintained at high enough elevations to provide groundwater discharge that meets the environmental water needs of aquatic ecosystems within wetland GDEs into the future. It is however plausible that, due to the simplification of the conceptual model and subsequent analysis, that the recorded response over time of the wetlands in HZs 5 and 6 could also be partially explained by changing rainfall runoff relationships.

4.3   Conclusions for Hydrogeological Zones 3 and 7

Wetland GDEs in Hydrogeological Zones 3 and 7 were shown by the analysis of WOfS derived hydrographs to have experienced significant declines in surface water expression, where wetlands that were permanently inundated during the 1985–95 epoch were completely dry in the 2005–15 epoch. All analysed wetland GDEs in these zones were dry by 2006–07, and failed to recover surface water inundation post the Millennium drought. The timing of observed absence of surface water expression coincided with significantly declining groundwater levels (Appendix C) in these zones, with monitored groundwater levels recording an almost 3 m change in seasonal maximum levels over the analysis timeframe, which we conclude resulted in loss of surface water expression of groundwater for many GDEs from the unconfined aquifer. With the use of groundwater level scenarios generated by groundwater hydrograph regression analysis of nearby observation wells (Cranswick, 2018), we found that the observed absence of water in the wetlands in the 2005–15 epoch was unable to be entirely accounted for by rainfall variability alone. As much as 90% of the observed change in surface water extent was unable to be attributed to rainfall variation at the Dip Swamp GDE site, and as much as 50% at Deadmans Swamp.

It was shown that whilst rainfall variability was a contributing factor to the decline and loss of wetlands with high dependency on the unconfined aquifer (especially Deadmans Swamp), that other impacts from surrounding landuse, such as plantation forestry and groundwater extraction for irrigation (both landuses which dominate in HZs 3 and 7) were likely to be significant contributors to the observed losses. These findings are likely to apply only to GDEs with very strong dependency on the unconfined aquifer, and under the assumptions of the conceptual model applied.

Other wetland ecosystems within HZs 3 and 7 that are less strongly dependent on the unconfined aquifer (i.e. either receive significant surface water inputs, or have less permeable substrate types) or are reliant on potentially perched localised sand aquifers (SKM, 2010) are likely not to be as heavily impacted by observed groundwater level declines in the regional unconfined aquifer. Projected future groundwater level declines (Costar et al., 2017; Cranswick, 2018) are however likely to increase the degree of disconnection between groundwater and GDEs in HZs 3 and 7 resulting in further loss and continued terrestrialisation of aquatic ecosystems. Significant increases in groundwater levels of at least 0.8–1.5 m (back to the seasonal high levels recorded in 2004) would be required to re-establish seasonal surface water inundation in highly groundwater dependent wetlands in this zone.

4.4 Water management implications

The demonstrated contemporary impacts to GDEs under current and past groundwater allocation policy in the Region (as also identified by Harding et al. (2015) and Cranswick (2018)) indicate a clear need for establishing ecologically relevant groundwater condition triggers and resource condition limits in water allocation plans. Examples of managing declining groundwater levels as a result of climate change and increased extraction has been well documented and demonstrated in Western Australia (GSS Taskforce, 2009; McFarlane et al., 2012). Many of the recommendations of the GSS Taskforce (2009) for managing the environment under a drying climate are potentially applicable in the Region, and the BGA area. Some of the key proposed management options identified by Harding et al. (2015) and with reference to the GSS Taskforce (2009) included:

- An adaptive management approach to the monitoring of the environmental impacts of groundwater decline be developed, that can separate the role of climate, and anthropogenic extraction/land use. This would include long-term monitoring of indicator species and/or vegetation community change and site hydrology (both groundwater and surface water) to detect ecosystem change, confirm conceptual understandings of the impacts of groundwater decline, and inform frequent review of management actions.

- Maintaining hydrogeologically and ecologically relevant set-back distances for new extraction and plantation forestry to high value GDEs, and monitoring the effectiveness of current dependent ecosystem protection strategies in water allocation plans.

- Improved knowledge of specific groundwater and surface water interactions and dynamics for GDEs in the South East NRM region could provide ecologically significant groundwater management levels (environmental water requirements) for high priority/value ecosystems.

The methods presented in this report, utilising the nationally available WOfS datasets, has the potential to contribute to the monitoring of GDE responses to groundwater level decline and ultimately contribute to the establishment of environmental water requirements and the determination of limits of acceptable change for selected GDEs. It is also important to note that a quantitative assessment of both the magnitude and timing of the impact of groundwater extraction and use by plantation forestry on groundwater discharge to wetland GDEs has not been conducted to date. This could be achieved by developing a local to sub-regional scale numerical groundwater model and testing future climate and extraction scenarios against a series of potential resource condition limits.

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6 Appendices

Appendix A. GIS models and scripts for WOfS hydrograph data retrieval

Workflow 1: Clip WOFLs and export to geodatabase

(1)   WOFL raster were clipped to wetlands of interest using an ArcGIS model:
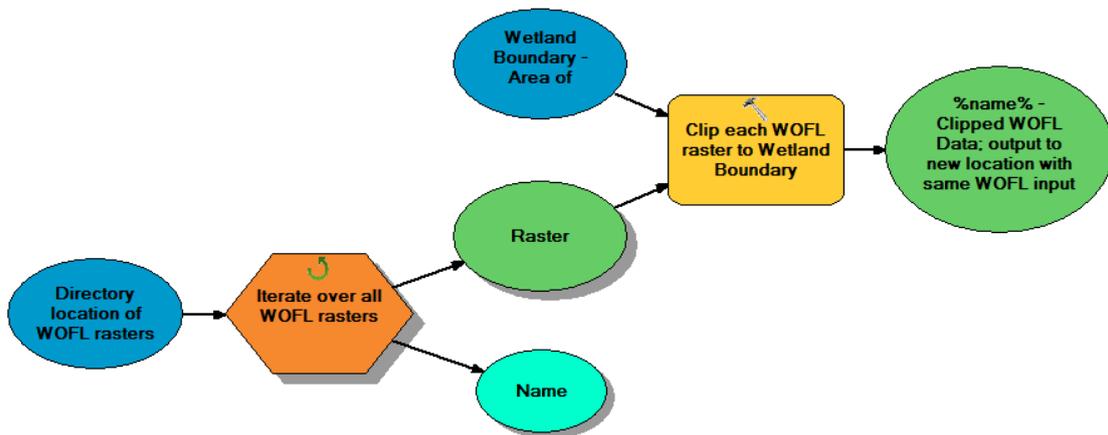
(2)   The clipped rasters were renamed using the following python script:

```
# ------------------------------------------------

# Import standard modules...
import time, sys, os, string # , glob #, math
# Import system modules...
import arcpy
from arcpy import env
# Set the ArcGIS for Desktop Advanced product by importing the arcinfo module...
import arcinfo
# Check out any necessary licenses
arcpy.CheckOutExtension("spatial")
# Overwrite any existing files...
arcpy.env.overwriteOutput = True
# Start time for process...
startTime = time.asctime()
startClock = time.clock()
#local variables
##localDataWS = "D:/Border_Groundwater/Data/TRIAL/Grids/rename"
try:
    print("")
    print("================================================")
    print("GIS Tool: rename wofs tiffs.py")
    print("================================================")
    print("")
    print("Commence Processing...")
    print("")
    print("------------------------------------------------")
```

```
    # Initialize...
    count = 0
    print("i")

    # Define working directory...
    inLocalDataWS = "d:/Border_Groundwater/Data/TRIAL/Grids"
    outLocalDataWS = "d:/Border_Groundwater/Data/TRIAL/Grids/rename"
    print inLocalDataWS
    print outLocalDataWS
    # Set the current workspace
    arcpy.env.workspace = inLocalDataWS
    print("ii")
    print("---------------------------------------------------")
    # Get and print a list of GRIDs from the workspace
    rdList = arcpy.ListRasters("LS5*", "TIF")
    print rdList
    for rd in rdList:

        # Increment counter...
        count = count + 1
        print ("Loop: ") + str(count)
        inRaster = str(rd)
        inRasterPath = inLocalDataWS + "/" + inRaster
        inRasterRF = inRaster.replace("-","_")
        outRasterName = inRaster[22:32] + ".tif"
        outRasterPath = outLocalDataWS + "/" + outRasterName

        print ("Input raster is : ") + inRasterPath
        print ("Output raster is: ") + outRasterPath
        print ("---------------------------------------------------")
        # Process: Copy Raster...
        arcpy.CopyRaster_management(inRasterPath, outRasterPath)
        print ("1.") + str(count)
        print("---------------------------------------------------")

# Get and print a list of GRIDs from the workspace
    rdList = arcpy.ListRasters("LS7*", "TIF")
    print rdList
    for rd in rdList:

        # Increment counter...
        count = count + 1
        print ("Loop: ") + str(count)
        inRaster = str(rd)
        inRasterPath = inLocalDataWS + "/" + inRaster
        inRasterRF = inRaster.replace("-","_")
        outRasterName = inRaster[23:33] + ".tif"
        outRasterPath = outLocalDataWS + "/" + outRasterName

        print ("Input raster is : ") + inRasterPath
        print ("Output raster is: ") + outRasterPath
        print ("---------------------------------------------------")
        # Process: Copy Raster...
        arcpy.CopyRaster_management(inRasterPath, outRasterPath)
```

```
            print ("1.") + str(count)
            print("------------------------------------------------")
except:
    # Check in any necessary licenses
    arcpy.CheckInExtension("spatial")

    # Bail-out and print error messages...
    print ("------------------------------------------------")
    print arcpy.AddMessage(arcpy.GetMessages(1))
    print ("")
    print ("--- ERROR!! ---")
    print ("")
    print("================================================")

# Check in any necessary licenses
arcpy.CheckInExtension("spatial")
print "Start  : " + startTime
print "End    : " + time.asctime()
endClock = time.clock()
elhours =int((endClock - startClock)/3600)
elmins = int((endClock - startClock)/60) - (elhours * 60)
elsecs = int((endClock - startClock) - (elmins * 60) - (elhours * 3600))
print "Elapsed: " + str(elhours) + " hrs " + str(elmins) + " mins " + str(elsecs) + " secs"
print("================================================")

# End time for process...
```

(3) Clipped rasters exported to a file geodatabase using ArcGIS model:
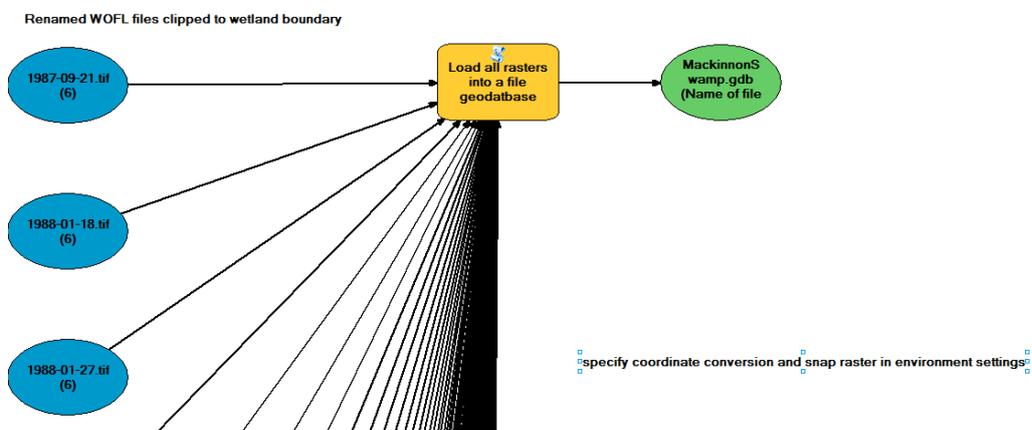
Workflow 2: Extract DEM point files and extract multi-values from WOFL rasters

    (1)   The DEM was clipped to the specified wetland, converted from a grid to a point file and attributed with x and y coordinates:
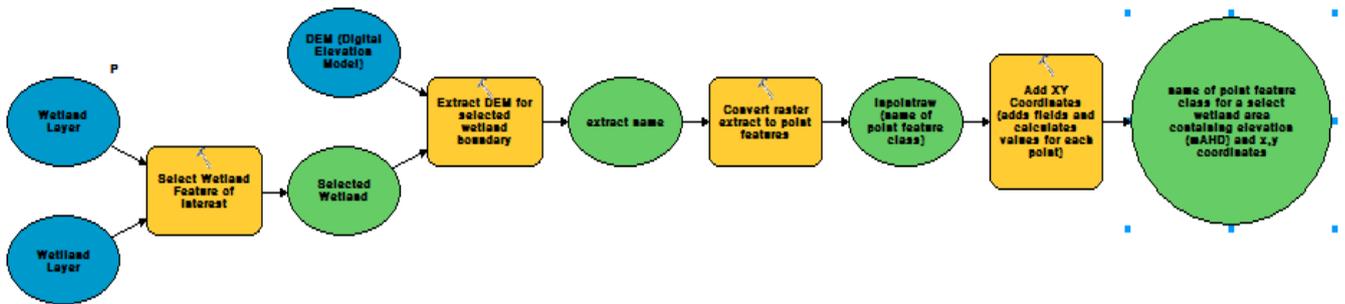
    (2)   The following python script was run to extract multi-values from each raster to each point:

```
# Import standard modules...

import time, sys, os, string , glob, math

# Import system modules...
import arcpy
from arcpy import env
from arcpy.sa import *

# set environment settings
env.workspace = r"e:\\Border_Groundwater\\Data\\Middlepoint.gdb" #location of rasters#

# set local variables
Wofs_rasters = "e:\\Border_Groundwater\\Data\\Middlepoint.gdb" #location of rasters
inpoints = "inpointraw" #this is location of point file

# check out the ArcGis Spatial Analyst Licence Extension
arcpy.CheckOutExtension("spatial")

# Get a list of rasters in the workspace
rasters = arcpy.ListRasters()

# Loop through the list of rasters
for inRaster in rasters:
    # set the output name for each output to be the same as the input
    outRaster = Wofs_rasters + "\\" + inRaster

    # Process: extract multi raster values to point feature class

    arcpy.gp.ExtractMultiValuesToPoints (inpoints, outRaster, "None")
```

Appendix B. Hydrograph regression analysis for selected observation wells from Cranswick (2018)

Nearest observation well to Deadmans Swamp:
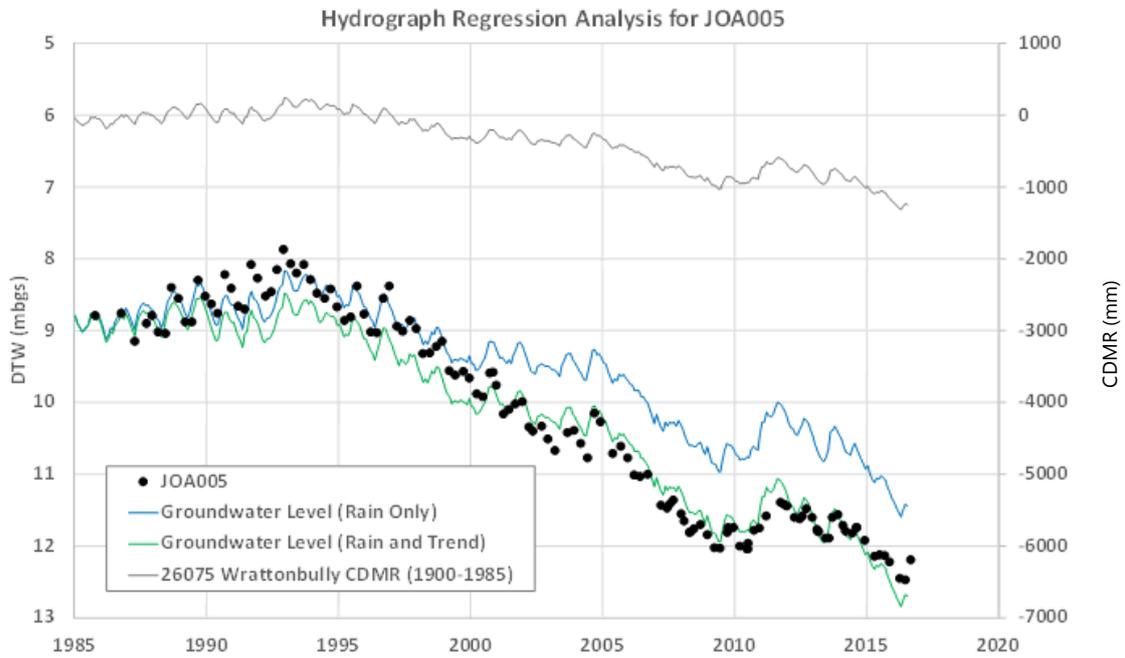
Nearest observation well to Dip Swamp:
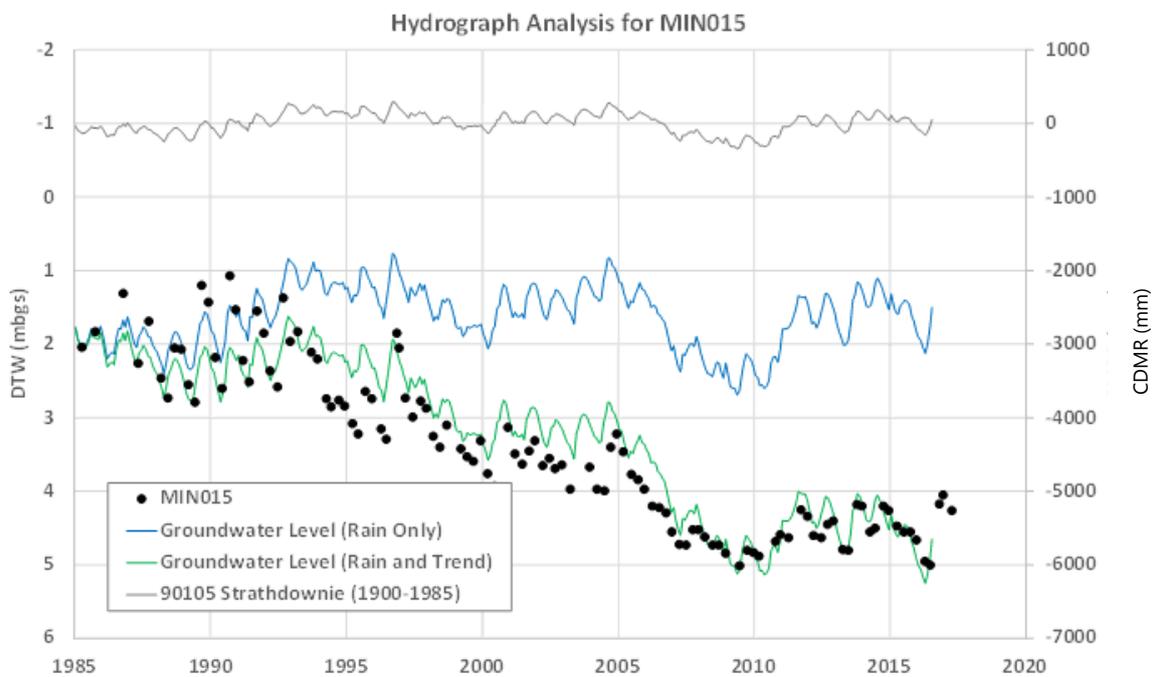
(sourced from Cranswick, 2018)

Appendix C. WOfS derived surface water hydrographs 1987 to 2013 with corresponding linear regression equations with nearby groundwater observation wells.

South Bool Lagoons 1

$SWL = 0.7242 \times JOA008 + 14.257$ $R^2=0.75$

Taylors Swamp

$SWL = 0.3479x \; JOA005 + 33.626$ $R^2=0.59$

South Bool Lagoons 2

$SWL = 0.7359 \times JOA008 + 13.655$ $R^2=0.72$

Deadmans Swamp

$SWL = 0.651 \times JOA005 + 16.879$ $R^2=0.73$

Glenrise Swamp 1

$SWL = 1.3038 \times PEN003 - 16.958$ $R^2=0.83$

Sawpit Swamp

$SWL = 0.427 \times PEN027 + 31.063$ $R^2=0.81$

Glenrise Swamp 2

$SWL = 0.6724 \times PEN003 + 17.764$ $R^2=0.77$

Coinville Swamp

$SWL = 0.7139 \times PEN027 + 14.371$ $R^2=0.81$

Coomooroo Swamp

$SWL = 0.3537 \times MON014 + 40.371$ $R^2=0.85$

McKinnon Swamp

$SWL = 2.0404 \times PEN011 - 64.709$ $R^2=0.76$

Kearney Lake

$SWL = 0.6303 \times YOU028 + 26.221$ $R^2=0.89$

Dip Swamp

$SWL = 0.3967 \times MIN015 + 40.87$ $R^2=0.75$

y-axis in m (AHD). SWL: Surface water level.

Appendix D. Spatial representation of surface water hydrometrics (depth, area and frequency) for analysed wetlands

SAWPIT SWAMP   S0109194

McKINNON SWAMP   S0106539

DIP SWAMP   S0110715

Observed (WOfS)

Mean maximum water depth (Spring) | Frequency Inundation

1985 - 1995
1995 - 2005
2005 - 2015

Average maximum water depth (m)
0 - 0.2
0.2 - 0.4
0.4 - 0.6
0.6 - 0.8
0.8 - 1
1 - 1.5
>1.5

Frequency of annual return inundation (years)
0 in 10 years
1 in 10 years
2 in 10 years
3 in 10 years
4 in 10 years
5 in 10 years
6 in 10 years
7 in 10 years
8 in 10 years
9 in 10 years
10 in 10 years

7 Units of measurement and abbreviations

7.1 Units of measurement used (SI and non-SI Australian legal)

Name of unit	Symbol	Definition in terms of other metric units	Quantity
hectare	ha	$10^4\,\text{m}^2$	area
kilometre	km	$10^3\,\text{m}$	length
metre	m	base unit	length
year	y	365 or 366 days	time interval

7.2 Abreviations

AHD   Australian Height Datum

BGA   Border Groundwaters Agreement (area)

CDMR   Cumulative Deviation from Mean Rainfall

DEM   Digital Elevation Model

DTW   Depth to water (groundwater)

DEW   Department for Environment and Water (Government of South Australia)

FSL   Full supply level (of a waterbody)

GDE   Groundwater dependent ecosystem

HARTT   Hydrograph and Rainfall Time Trend (analysis – Ferdowsian et al. 2001)

HZ   Hydrogeological Zone (Harrington & Currie 2008)

LiDAR   Light Detecting and Ranging (DEM)

LLC   Lower Limestone Coast

mbgs   meters below ground surface

NRM   Natural Resource Management

PWA   Prescribed Wells Area

SWL   Surface water level

WOFL   Water Observation Feature Layer (product of WOfS, Geoscience Australia)

WOfS   Water Observations from Space (Geoscience Australia)