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Projected risks to groundwater-dependent terrestrial vegetation caused by changing climate and groundwater abstraction in the Central Perth Basin, Western Australia

Olga Barron*, Ray Froend, Geoff Hodgson, Riasat Ali, Warrick Dawes, Phil Davies and Don McFarlane

Abstract The effect of potential climate change on groundwater-dependent vegetation largely depends on the nature of the climate change (drying or wetting) and the level of current ecosystem dependency on groundwater resources. In south-western Australia, climate projections suggest a high likelihood of a warmer and drier climate. The paper examines the potential environmental impacts by 2030 at the regional scale on groundwater-dependent terrestrial vegetation (GDTV) adapted to various watertable depths, based on the combined consideration of groundwater modelling results and the framework for GDVT risk assessment. The methodology was tested for the historical period from 1984 to 2007, allowing validation of the groundwater model results' applicability to such an assessment. Climate change effects on GDTV were evaluated using nine global climate models under three greenhouse gas emission scenarios by applying the climate projections to groundwater models. It was estimated that under dry climate scenarios, GDTV is likely to be under high and severe risk over more than 20% of its current habitat area. The risk is also likely to be higher under an increase in groundwater abstraction above current volumes. The significance of climate change risk varied across the region, depending on both the intensity of the change in water regime and the sensitivity of the GDTV to such change. Greater effects were projected for terrestrial vegetation dependent on deeper groundwater (6 to 10 m).

KEYWORDS Climate change; groundwater-dependent ecosystem; global climate model; risk assessment

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1 Introduction

Much of Australia's internationally recognised rich biodiversity is associated with groundwater discharge zones, which can be localised (i.e. wetlands or rivers) or diffuse (i.e. areas of terrestrial vegetation). The ecosystems associated with these zones are known as groundwater-dependent ecosystems; they require the presence or input of groundwater to maintain some or all of their ecological function, composition or structure (Eamus and Froend, 2006; Murray et al., 2006). Hatton and Evans (1998) identified four broad ecosystem classes within which groundwater plays an important role: terrestrial vegetation, river baseflow systems, aquifer and cave systems, and wetlands. These ecosystems were identified as being potentially vulnerable to groundwater abstraction and climate-induced hydrological change.

Terrestrial vegetation dependency on groundwater is commonly validated at the local scale and extrapolated via modelling to the regional scale. The level of dependency can be assessed based on the volume of groundwater transpired by individual species as part of the water balance estimation (Thorburn et al., 1993; Mensforth et al., 1994; Doody et al., 2009). It was shown that for groundwater-dependent terrestrial vegetation (GDTV), total evapotranspiration can be greater than annual rainfall (O'Grady et al., 2011, Batelaan et al., 2003, Batelaan and De Smedt, 2007). In addition, isotope methods can be used to separate between water sources in unsaturated and saturated profiles (Dawson and Ehleringer, 1991; Jackson et al., 1999; Adams and Grierson, 2001). When this information is linked to the position of the groundwater table and their seasonal and inter-annual variability under the studied vegetation, it allows identification of the possible range of groundwater depths which can satisfy vegetation water demands (Klijn and Witte, 1999, Zencich et al. 2002; Froend and Sommer, 2010; Leany et al., 2011). This localised information can be extended to identify the potential distribution of GDTV at a regional scale using the observed or modelled watertable depth.

In contrast, the sensitivity of GDTV to long-term and short-term change in groundwater regime has been studied to a lesser extent. Documented impacts vary from progressive shifts in species composition to profound impacts on ecosystem function leading to biotic extinction (e.g. Jump et al., 2010; Chmielewski et al., 2005; Froend and Sommer, 2010; Sommer and Froend, 2011). The nature and severity of impacts are related to both the magnitude and the rate of hydrological regime change, with rapid drawdown leading to

1 significant impacts on dependent ecosystems (Froend and Sommer, 2010). Examples of
2 drawdown-induced modification to groundwater-dependent vegetation are few and
3 generally local-scale assessments, single species responses over short timeframes, or are
4 modelled regional assessments (e.g., Cooper, 2006, Doody et al., 2009, Muñoz-Reinoso,
5 2001). Resulting from either groundwater abstraction or climate change, the impacts of
6 regional groundwater drawdown require assessment at a larger scale, particularly when
7 employed in the context of water resource management. In such cases, detailed knowledge
8 of the potential response of all affected ecosystems is generally not known and therefore
9 requires up-scaling of known, generalised, smaller-scale responses to a regional scale
10 (Leaney et al., 2011; Froend and Loomes, 2004).

11 In many regions with arid or semi-arid climates (including the larger part of Australia),
12 rainfall is highly seasonal and, where present, shallow groundwater promotes the
13 establishment of groundwater-dependent ecosystems. Such regions also often rely on
14 groundwater as the major water resource for agriculture and other human needs; however,
15 this use may lead to a reduction in groundwater availability to GDTV, causing its
16 deterioration (Muñoz-Reinoso, 2001; Elmore et al., 2003; Mata-González et al., 2011). In
17 such cases, the effect of groundwater abstraction was frequently found to be more
18 significant than changes in climate conditions.

19 However, the effect of climate change on GDTV is more complex. Projected global
20 warming is likely to lead to an increase in water demand. At the same time, elevated CO₂
21 (a main greenhouse gas) allows plants to increase their water-use efficiency (Medlyn et al.,
22 2001; Leuzinger and Korner, 2007; Loehle, 2007; Kohler et al., 2010). This can lead to an
23 increase in leaf area, a reduction in transpiration, or both (Eckhardt and Ulbrich, 2003).
24 Conversely, a projected reduction in rainfall in many regions is likely to reduce overall
25 water availability. The combination of these factors is likely to have an impact on
26 renewable groundwater resources and their sensitivity to climate change (Barron et al.,
27 2011), as groundwater recharge is largely dependent on land cover. An assessment of the
28 effect of climate change on GDTV is needed to account for these major governing
29 processes and their changes under future climatic conditions.

30 When climate change impact on mean annual rainfall is considered, the most consistent
31 projections in its reduction are in regions with Mediterranean-type climates (Bates et al.,
32 2008), including south-west Australia. Here drying climatic conditions will lead to an
33 increasing dependency on groundwater resources for urban and irrigation water supply,

causing an additional stress to water-limited ecosystems. This poses a particular challenge to sustainable use of groundwater resources in the region, due to the uncertainty of potential ecological impacts of groundwater use on groundwater-dependent vegetation under a changing climate. The region has experienced a 10–15% rainfall reduction since the early to mid-1970s (Bates et al., 2008) and the further rainfall reduction is projected (Charles et al., 2010). The consequences of reduced rainfall and enhanced evaporation, as well as increased groundwater abstraction, are seen to have profound impacts on vulnerable vegetation (Horwitz et al., 2008).

The objectives of this study were to provide a regional analysis of potential risks to GDTV in south-west Australia from both projected changes in future climate and increases in groundwater resource exploitation. Our analyses aim to identify areas where impacts are more likely to occur, as a first approximation to the risk posed to GDTV from future climate and development, as well as to identify the applicability of the currently available tools for such analysis.

2 Study area

The main topographic features of the south-west region of Australia are the coastal plains with elevations below 150 m AHD (often less than 100 m AHD), escarpments which separate the plains from the inland Darling Plateau with an average elevation of about 350 m AHD, and the Leeuwin-Naturaliste Ridge in the south (Figure 1). Almost all rivers arise on the plateau before flowing over the flatter coastal plains into the Indian Ocean, often through estuaries.

Up to 80% of mean annual rainfall for the region occurs during winter and spring (from May to October). Annual rainfall has a pronounced latitudinal gradient with the highest in the south of the area (1290 mm) and the lowest in the north (305 mm).

The Swan Coastal Plain marks the deep sedimentary Perth Basin which (onshore) is a narrow strip, 30–100 km wide by 1000 km long, from Geraldton along the western coastline down to the south-west of Western Australia. The basin contains one of Australia's most important groundwater resources (AWR, 2005) and at the surface is characterised by a dunal landscape with low relief and a shallow watertable.

The specific hydrogeological and hydrological conditions of south-west Australia, in association with its profound geographic isolation from other temperate areas, have

1 resulted in unique ecological systems. There is an unusually high degree of endemism
2 in plant and animal species, and many water-limited ecosystems are recognised at the
3 international, national and state level for their conservation value. High biodiversity is
4 also a hallmark of these ecosystems; it's estimated they contain 1% of the world's
5 total number of plant species and approximately 1000 species yet to be named
6 (Hopper and Gioia, 2004). Many endemic species are associated with ecosystems
7 sensitive to hydrological change, such as karstic systems (Jasinska and Knott, 1994;
8 Knott and Jasinska, 1998) and freshwater wetlands (Horwitz, 1997; Pinder et al.,
9 2004) and are therefore threatened by increased groundwater use, reduced rainfall and
10 increased evaporation.

11 Although a larger portion of this area was studied previously (CSIRO, 2009a and
12 2009b; Barron et al., 2012), only the region central to the area shown in Figure 1 is
13 considered here. Within this area there are a number of significant water-dependent
14 ecosystems and many of them partly or fully reliant on groundwater. This is also the
15 most densely populated area in the state of Western Australia and contains the state
16 capital, Perth. Groundwater is becoming a dominant source of domestic and industrial
17 water supply with Perth being largely dependent on groundwater from the Gnangara
18 Mound, from which more than 170 GL is extracted annually. Other uses, including
19 agriculture, private water supply schemes and mining, account for an additional 394
20 GL of licensed groundwater extraction per annum.

21 **3 Method**

22 The methodology used for the analysis of climate change impact on GDTV was
23 developed from the approach commonly adopted for investigation of such impact on
24 water resources. It involved the use of ensemble global climate model (GCM) outputs,
25 their down-scaling (spatially and temporally) (Mpelasoka and Chiew, 2009), and
26 application of the time series generated for rainfall and other climate parameters to
27 groundwater models to estimate changes in aquifer recharge and groundwater
28 resources under future climate conditions (e.g. Scibek et al., 2007; Mileham et al.,
29 2009; Ng et al., 2010; Crosbie et al., 2011; Dams, et al., 2012; Ali et al., 2012a).

30 The results of groundwater modelling were analysed deploying a risk assessment
31 framework developed for the groundwater-dependent vegetation of the Perth region

(Froend and Loomes, 2004). The risk rating was identified based on the antecedent groundwater level prior to the study period and both the magnitude and the rate of change in groundwater levels over the investigated period.

Details of the adopted methodology are outlined below.

3.1 Future climate

The future climate scenarios are derived from the historical climate by modifying individual climate parameters, such as temperature and rainfall, according to the statistical properties of climates generated by 15 GCMs under 40 specific ‘storylines’ (Charles et al., 2010). Each storyline varies the projected future global emissions of carbon dioxide, methane, nitrous oxide and sulphate aerosols; resulting in different levels of global warming. Seasonal adjustments per degree of global warming, and for each climatic variable, are determined from the simulated values for each GCM within individual computational cells. The climatic sequences and statistics derived from GCM outputs are discussed by Charles et al. (2010), and were produced on a 0.05 degree × 0.05 degree grid over continental Australia and Tasmania.

The outputs from nine selected GCMs (those which most closely reproduced southern hemisphere weather phenomena and historic rainfall patterns over south-west Western Australia; Smith and Chiew, 2009) were used for further analyses. These GCMs, with three estimates of global warming (by 0.7°C, 1.0°C and 1.3°C compared to 1990), were applied for a selection of dry, median and wet future climate sequences and reported for 2030 (CSIRO, 2009a; Ali et al. 2012a). The scaling factors, defined from GCM outputs, were applied to the historical climate data to generate the projected sequences of climate parameters. From these results, three scenarios were selected for further analysis and denoted as *median*, *wet* and *dry* future climate scenarios, representing the median as well as the extreme ends of the nine selected GCMs’ spectra.

The climate record from 1975–2007 was adopted as a base for climate data used in this study, considering that during this period both meteorological and groundwater data across the region were the most reliable. Additionally, this period was found to be representative of recent climate in the south-west following a discrete drying shift in the late 1960s and early 1970s (IOCI, 2006; Baines and Folland, 2007; Charles et

al, 2007). Data from a computational cell closest to an existing climate station was used to generate climate sequence for future scenarios.

The *median*, *wet* and *dry* scenarios assumed that groundwater abstraction would continue unchanged (i.e. at 2007 levels) until 2030. Furthermore, a future climate and *development* scenario was deployed, based on the *median* climate scenario, which incorporated full development of water resources (extraction up to the maximum level defined in current water allocation plans in the region).

3.2 Groundwater modelling

A groundwater model known as PRAMS (Perth Regional Aquifer Modelling System; Cymod Systems, 2004), largely based on MODFLOW-96 (McDonald and Harbaugh, 1996), was used to estimate groundwater levels in year 2030 under the scenarios described above. The model, covering more than 10,000 km², is currently used by Western Australia's water authorities to facilitate water management decisions in the region. Model calibration and validation were based on monitoring data obtained from more than 1000 bores.

PRAMS incorporates a dynamically linked vertical flux model (VFM), replacing the functions of the recharge and evaporation modules in MODFLOW. The VFM dynamically calculates daily, and aggregates to monthly, the flux of water to or from groundwater as a result of land surface and root zone unsaturated water balance. This net flux can be positive or negative depending on combinations of soil type, vegetation characteristics, local climate and groundwater depth as estimated by MODFLOW (Silberstein et al., 2002; Barr et al., 2003).

VFM for vegetated surfaces is based on the WAVES model (Zhang and Dawes, 1998), which is a physically based model utilising Richards' equation for vertical one-dimensional moisture flow and redistribution (Short et al., 1995), the Penman-Monteith combination equation for determining evaporative demand (Monteith, 1981), and the integrated rate method for modelling vegetation dynamics in response to light, temperature, water availability and concentration of atmospheric CO₂ (Hatton et al, 1992; Wu et al, 1994). Therefore, it allows for rain interception by canopies and water use by plants from the unsaturated zone as well as from groundwater. The WAVES model has been well tested for climate change simulations (Crosbie et al., 2010; Green et al., 2007; McCallum et al., 2010).

1 The VFM outputs are monthly net recharge values at each computational cell, and
2 they do not include separate estimates of gross recharge and total evapotranspiration.
3 Nevertheless, the area with negative net recharge was considered a groundwater
4 diffuse discharge zone.

5 The model includes the main Perth Basin aquifers: the Superficial Aquifer with a free
6 (unconfined) watertable; and the confined Leederville and Yarragadee Aquifers. The
7 vertical profile is represented by 13 layers, on a uniform horizontal grid 500 m × 500
8 m, aligned north–south. A no-flow boundary condition was assigned for northern,
9 southern and eastern boundaries; the latter coincided with the Darling Scarp. The
10 western boundary is a constant head boundary, associated with groundwater discharge
11 to the ocean. The VFM operates in each MODFLOW cell, but uses unique
12 combinations of soil, vegetation and climate to minimise run time overheads.

13 The model was calibrated for the historical period 1975–2007 and used to predict
14 outcomes from future climate scenarios from 2007–2030 (CSIRO, 2009a).
15 Groundwater abstractions were kept constant at 2007 levels in all scenarios except the
16 *development* scenario, as described above.

17 PRAMS is considered to be adequately calibrated and validated for groundwater
18 resource assessment (Cymod, 2004; DoW, 2009). The model was calibrated over the
19 20 years from 1980 to 1999, and validation statistics derived for the 5 year period
20 2000 to 2004. In the unconsolidated sandy surface aquifer there were 362 bores in the
21 top 10 m, and 302 bores in the lower 10 to 40 m. Validation runs yielded an average
22 absolute error of 1.6 m between observed and predicted groundwater level, and a root
23 mean square error of 2.4 m. While the model was developed to facilitate the
24 management of groundwater resources in the region, its applicability to environmental
25 implications of groundwater use and climate variability has not been previously
26 tested.

27 **3.3 GDTV risk assessment**

28 The ecological implications of groundwater modelling results were inferred by using a
29 desktop assessment of ecological risks to GDTV based on an approach developed by
30 Froend and Loomes (2004): the framework originally developed for water resource
31 allocation planning in the Perth region. Froend and Loomes (2004) suggest that in the

1 Perth region, groundwater shallower than 10 m below ground is highly likely to
2 support vegetation. The degree of groundwater use by phreatophytes (groundwater-
3 dependent plants) increases with reduction in the depth to groundwater (Froend and
4 Zencich, 2001). These conclusions were based on long-term, local-scale
5 investigations of terrestrial vegetation responses to decline in watertable along with
6 the analysis of seasonal water-source partitioning of species growing along a depth to
7 groundwater gradient that ranged in depth from 2.5 to 30 m.

8 Zencich (2003) demonstrated that phreatophytes are more tolerant to groundwater
9 decline with increasing depth to watertable and lower proportion of groundwater use;
10 primarily due to the larger volume of unsaturated zone exploitable by plant root
11 systems (as a result of the associated increase in unsaturated zone thickness that
12 comes with groundwater decline) (Zencich et al., 2002). Froend and Sommer (2010),
13 in their analysis of Perth-region phreatophytic plant community dynamics and
14 response to rainfall reduction and groundwater drawdown over 30 years, identified
15 depth to groundwater and rate of drawdown as the dominant biophysical drivers of
16 floristic spatial and temporal patterns. Progressive vegetation change occurred under
17 lower rates of watertable drawdown (9 cm per year), whereas threshold response
18 (marked by extensive and rapid mortality) was observed under higher rates (50 cm per
19 year) of drawdown.

20 In adopting the Froend and Loomes (2004) framework in the current study, firstly the
21 risk assessment identified the watertable levels thresholds which are specific for the
22 hydrological regime required to maintain terrestrial vegetation (Figure 2). Depth to
23 groundwater (H) was grouped into three zones: 0 to 3 m, 3 to 6 m and 6 to 10 m,
24 representing decreasing vegetation dependency on groundwater with increasing depth
25 to watertable. Secondary, the risk categories (extreme, high, medium or low) are
26 estimated using the magnitude and rate of groundwater level drawdown over a
27 considered period of time, as illustrated in Figure 2.

28 The predictive groundwater modelling was then used to define changes in
29 groundwater levels between 2007 and 2030 within the area of shallow groundwater
30 (less than 10 m) by comparing the maximum depth to groundwater in the Superficial
31 Aquifer between these years for future *wet*, *median*, *dry* and *development* scenarios.
32 The absolute change in the maximum depth to groundwater over the simulation period

(ΔH) and the rate of groundwater level drawdown (or $\Delta H/t$, where t is the simulation period of 23 years) were calculated for each computational cell. These allowed the estimation and mapping of the risk rank for each computational cell within the model as low, medium, high or extreme as shown in Figure 2.

In order to further explore the applicability of PRAMS results to the ecological risk assessment, the method was also applied to a historical period of 23 years between 1984 and 2007, allowing comparison of observed changes in vegetation with the risk assessment resulting from the deployed methodology. Over this period a 15% reduction in annual rainfall was observed which coincided with a three-fold increase in groundwater abstraction in the region.

Ecological risk projections based on the PRAMS results over the historical period, was validated using monitoring data. The analysis included a comparison between observed and modelled data at monitoring locations for both the depth to groundwater (H), groundwater level change (ΔH) and the rate of the level changes between two dates. Following the recommended approach (Barnett et al., 2012) agreement between observed and modelled groundwater levels was assessed using Scaled Mean Sum of Residuals (SMSR), Scaled Root Mean Fraction Squares (SRMFS) and Coefficient of Determination (CD). Using the water level monitoring data in 400 bores in the Superficial Aquifer (where data were available for both dates) and simulated watertable levels within the corresponding cells, the locations were classified in terms of the potential GDTV occurrence (zone 1: depth to groundwater 0 to 3 m, zone 2: depth to groundwater between 3 to 6 m, and zone 3: depth to groundwater between 6 to 10 m). Finally, based on observed and simulated data the risk to GDTV was estimated at the location of individual bores and the results compared. The spatial distribution of risk zones, identified from the modelled results and over the historical period, was compared with the areas where changes in vegetation were recorded in response to past changes in watertable levels.

The historical analysis also served to establish the limits of the deployed methodology prior to its application to future climate scenarios. It is recognised that the GDTV ecological risk framework, as developed for the Gnamangara area (Figure 1), may have limitations when applied to regions with different lithology (for example, heavier soils with greater water retention than sandy soils, such as on the Dandaragan Plateau;

Figure 1). Furthermore, the variation in depth to watertable, whilst significant in terms of ecological risk assessment, may be challenged by the groundwater model accuracy in some areas of the Perth region. Considering these limitations, the approach was used only as a means of approximating risk posed to GDTV by future climate change and current abstraction on a regional scale, and highlighting areas where the potential risk is likely to occur. We recognised that future work should encompass both refinement of the groundwater modelling and localised ecological studies under more variable site conditions.

4 Results

4.1 Validation of GDTV risk framework using historical groundwater level analysis

The adopted approach for GDTV risk analysis is based on relatively small changes in groundwater levels (less than 2.75 m, see Figure 2) and comparison between observed and modelled results for the historical period from 1984 to 2007 was undertaken to validate the methodology. The results of this comparison suggested that the currently available modelling results are adequate for GDTV risk analysis at a regional scale, as indicated by the following points:

- 1. The difference between observed and modelled water levels is within the accuracy defined by the model spatial discretisation.*

A difference between observed and modelled groundwater levels could be expected, arising from both the spatial averaging associated with the deployment of the 500 m by 500 m horizontal grid in the model and the topography in the vicinity of a particular observation bore. The regional groundwater gradient in the Superficial Aquifer is relatively low, varying between 0.002 and 0.03 (0.2–3.0 m per 100 m). The mean value of the groundwater gradient over the model domain is 0.0028 with a standard deviation (STD) of 0.0024. This indicates that the variation in ‘real’ groundwater levels within an area covered by an individual model cell of 500 m by 500 m can be on average 1.4 m or 2.6 m for ± 1 STD. The maximum difference within a single model cell was estimated as 15.2 m. Therefore the differences (Δh) between observed groundwater levels in the individual bores (H_o) and predicted by the model

1 (H_m), or $\Delta h = H_o - H_m$, should be expected to fall within a similar range if the model
2 results adequately represent the groundwater table in the region.

3 The statistical distribution of the Δh values at the available monitoring locations for
4 2007 in the Superficial Aquifer shows that its mean and standard deviation are equal
5 to 1.6 m and 1.6 m respectively, with a maximum $\Delta h = 9.5$ m. In more than 85% of
6 bores the difference in water levels fell within ± 1 STD from the Δh mean value. This
7 estimate is consistent with the values expected, based on deployed model
8 discretisation and regional groundwater gradients of the Superficial Aquifer.

9 The relationship between modelled and observed water levels is shown for the years
10 1984 and 2007 in Figure 3. SMSR and SRMFS for both years are less than 5 and the
11 CDs are 0.97 and 0.96 for the two years respectively. This suggests a good agreement
12 between the modelled and observed water levels, though a greater deviation from the
13 1:1 line is related to shallower groundwater levels.

14 2. *The changes in groundwater levels between the years 1984 and 2007 within*
15 *individual bores are comparable for observed and modelled data.*

16 The comparison between the changes in groundwater levels both observed (ΔH_o) and
17 modelled (ΔH_m) over a period between 1984 and 2007 in the Superficial Aquifer is
18 shown in Figure 4. SMSR and SRMFS for both years are less than 5 and CD is 0.91,
19 indicating a good agreement between observed and modelled data.

20 3. *The observed and modelled changes in water level produce similar results when*
21 *GDTV zones are estimated for 1984 and risk indices are calculated for*
22 *groundwater level changes between 1984 and 2007.*

23 The GDTV zones for 1984 were identified for the locations of the groundwater
24 observation, using water level monitoring data and the model results for the cells,
25 associated with those bores locations. In 72% of all locations the identical GDTV
26 zones were detected for both the observed and modelled results. In 27% of cases,
27 there was a disagreement between a zone type identified by the observed and
28 modelled results with only a one zone shift.

Using the risk assessment approach, described above, and the groundwater level changes between 1984 and 2007, the risk for the identified GDTV types was estimated using water level monitoring data and the model results for the cells, associated with those bores locations. In 56% of cases the risk category for all GDTV zones was in agreement between observed and modelled results (in **Error! Reference source not found.** shown as disagreement index 0) and a further 29% show a one risk category shift.

The results of the risk assessment for GDTV over the period from 1984 to 2007 are shown in Figure 5 combined for three GDTV zones. The risk results are shown only within the vegetated area mapped in 2007 (CSIRO, 2009a). The areas with recorded changes in vegetation (Chang et. al., 2012; DEC, 2009) were identified as high risk GDTV zones using the adopted methodology. Areas where modelled and observed water levels are not in agreement were also identified in Figure 5. The inspection of results indicates that the disagreement between modelled and observed results occurs in the areas where localised groundwater recharge is expected from ephemeral streams flowing from the Dandaragan Plateau in the North West area of the model domain. This finding leads to the conclusion that details of this interaction were not adequately incorporated in the current model; however, it is worth pointing out that a detailed investigation in this area was not within the scope of the original groundwater modelling.

4.2 Future climate projections and their effect on renewable groundwater resources and GDTV risk

The nine selected GCMs used in this study projected rainfall changes between +4.5% and –14.2% averaged across the study area, and between +10.3% and –17.5% at individual stations, based on warming of 0.7° to 1.3°C by the year 2030.

Projected changes in groundwater net recharge and associated changes in groundwater levels are illustrated in Figure 6 and discussed in detail in Ali et al. (2012a, 2012b); Dawes et al. (2012). The greatest changes in groundwater levels are associated with the *dry* future climate scenario, where a reduction in water level is projected for most of the study area, particularly in the area between Yanchep and Gingin, located to the north of areas of public groundwater abstraction on the Gnangara Mound.

1 The moderate reduction in groundwater level observed under the *median* future
2 climate is projected to become greater when anticipated additional abstraction takes
3 effect. The most noticeable changes in watertable between the *median* and
4 *development* scenarios are in the east from Lancelin (Gnangara Mound) and in the
5 south of the model domain. At the same time, rising watertables are projected in the
6 north-east of the study area under all future climate scenarios. However this was
7 attributed to the effect of the recent land clearance for dry-land agriculture occurred
8 prior to 2007, the year which marks the start of the assessment period reported in this
9 paper. The land clearance appears to lead to an increase in recharge in this region
10 regardless of drying climate.

11 Net groundwater recharge is projected to reduce over most of the study area, shown in
12 warm colours on the plot “Change in net recharge” in Figure 6 (also discussed in Ali
13 at al. 2012b and Dawes et al. 2012). However, an increase in net recharge was
14 identified in the areas where the watertable is shallow, shown in cool colours on the
15 plot “Change in net recharge” in Figure 6. These areas coincide with the zones of
16 diffuse groundwater discharge, where evapotranspiration losses exceed gross recharge
17 (“negative” net recharge zones). In such areas under a drying climate, a reduction in
18 rainfall and therefore gross recharge led to groundwater level reduction (Figure 6, plot
19 “Changes in watertable”), and hence decrease in evaporative losses from watertable.
20 Though the resulting net recharge still remains negative, it is however greater than
21 under historical climate (“less negative”). Therefore in the areas with particularly
22 shallow groundwater an increase in annual net recharge can be expected even under
23 conditions where annual rainfall is projected to reduce.

24 According to the groundwater model results, the area with a watertable shallower than
25 10 m covers 3525 km² or 36% of the modelled domain, with areas of shallow
26 watertable (<3 m), watertable at 3 to 6 m, and watertable at 6 to 10 m accounting for
27 12%, 13% and 11% of the total model domain area respectively (Figure 7). Under the
28 future climate and development scenarios, the extent of these areas is projected to
29 alter due to changes in watertable, with reductions in the areas with shallower-
30 watertable-depth categories becoming gains in the areas with deeper-watertable-depth
31 categories.

1 The ecological risks associated with future *median* climate projections are illustrated
2 in Figure 8. Under this scenario, the areas with identified ecological risks are small for
3 most GDTV zones. Low risk is the dominant category with the exception of areas
4 near or within the Gngangara Mound (Figure 8 and Figure 9), where the changes in
5 groundwater level are significant.

6 Under the *median* climate scenario, high and severe risks are not likely to occur for
7 zone 1 terrestrial vegetation (depth to watertable <3 m). For zone 2 vegetation (depth
8 to watertable 3 to 6 m), only 2% (by area) is projected to be at a high or severe risk.
9 For zone 3 vegetation (depth to watertable 6 to 10 m), about 8% of the area is
10 projected to be at a high or severe risk.

11 As expected, the greatest risk to GDTV is projected to occur under the *dry* future
12 scenario. The area where GDTV is projected to be under severe and high risk
13 increases to 10%, 21% and 38% for terrestrial vegetation with a depth to watertable of
14 0 to 3 m, 3 to 6 m and 6 to 10 m respectively (Figure 8). Similarly to the *median*
15 future climate scenario the majority of high and severe risk areas are in the central and
16 northern area, where groundwater abstraction, assumed to be equal to current
17 groundwater use, is particularly high.

18 The *development* scenario, for which a *median* future climate scenario was used but
19 with the groundwater abstraction rates higher than current, leads to an increase of 54–
20 60% in the areas where moderate, high and severe GDTV risk categories are
21 projected, when compared to the *median* future climate without development (Figure
22 9). However, the low risk category still dominates within the study region.

23 **4.3 Association between GDTV zone, groundwater balance** 24 **and increased GDTV risk under future climate projections**

25 The results of the analysis indicate that the greater risks for GDTV are due to a significant
26 reduction in recharge, such as under the most dry climate scenario, and/or an increase in
27 groundwater discharge, such as higher groundwater abstraction volumes. The risks
28 appeared to be more significant for vegetation established over deeper groundwater, i.e. 6
29 to 10 m below ground level. Though the first two conclusions are expected, the higher risk
30 for GDTV in zone 3 is at a first glance counterintuitive, particularly when in the adopted

1 framework the threshold values for this class were the highest amongst all GDTV classes
2 considered.

3 To investigate the reasons for these projections, groundwater model results were further
4 analysed to more methodically explore the changes in groundwater recharge and watertable
5 under future climate and development scenarios within the identified GDTV categories.
6 VFM application to the MODFLOW model facilitated such analysis as it accounted for
7 land cover types and depth to groundwater table by adapting process-based models for
8 unsaturated flow simulation.

9 It is a well established fact that GDTV influence both groundwater diffuse recharge
10 and groundwater diffuse discharge. The combination of these fluxes influences
11 resulting annual net recharge, and, among other hydrodynamic factors, defines the
12 annual watertable fluctuation. Within the vegetation root zone in a soil profile, the
13 amount of rain water reaching the watertable (or gross recharge) is lower where the
14 depth to groundwater is greater (Renger et al., 1986). This is due to a higher
15 proportion of infiltrated water retained in the unsaturated zone of a greater thickness.
16 This water is used by vegetation and replenished annually during wet season. Under
17 the drying condition the reduction in rainfall leads to reduction in water flux that can
18 reach the watertable (gross recharge) after replenishing this relatively constant water
19 storage in the unsaturated zone. Considering that this is likely to lead to lowering the
20 watertable and increasing unsaturated zone thickness, the amount of water reaching
21 the watertable becomes progressively lower, leading to a more significant reduction
22 watertable and gross recharge.

23 Compared to zones 1 and 2, the unsaturated zone is larger in zone 3 that under the
24 same conditions leads to relatively lower gross recharge. Hence following from the
25 above discussion there is a greater sensitivity of watertable fluctuation to changes in
26 the rainfall regime than in other zones. This is illustrated in Figure 10: compared to
27 other zones, a progressive reduction in net recharge in zone 3 is greater, when
28 stepping from wet to dry climate scenarios.

29 Under *wet, median, dry* future climate and *development* scenarios the resulting change
30 in watertable (defined here as $\Delta H = H_{future} - H_{historical}$) relative to 2007 is also greater
31 for areas where the watertable is initially between 6 to 10 m deep. Figure 11 shows

1 the frequency distribution of ΔH , using a percentage of groundwater model cells with
2 a 2007 watertable depth of less than 3 m (zone 1), 3 to 6 m (zone 2) and 6 to 10 m
3 (zone 3). In all zones, more than 90% of cells show a reduction in the watertable level
4 under *dry* climate scenarios; however, in zone 3, more than 50% of the cells show a
5 reduction greater than the maximum threshold in ΔH for the extreme risk class (2.75
6 m; Figure 2). The smallest ΔH values are projected for zone 1 where only 9% cells
7 exceeded the extreme risk threshold. As these groundwater level changes are long-
8 term averages, it is unlikely they are either seasonal or short term in their effects.

9 On the other hand groundwater use by vegetation (or groundwater diffuse discharge)
10 is greater when plants are established over a shallower watertable; and if it is
11 shallower than 6 m, groundwater can contribute up to 50% of vegetation water use
12 (Zencich et al. 2002). A reduction in gross recharge followed by a lower watertable
13 leads to a reduction in evapotranspiration losses from the groundwater table, which, to
14 some extent, buffers the overall reduction in net recharge under future climate
15 scenarios in this region. However, such a buffering effect is more pronounced in areas
16 where the initial watertable was shallow (<3 m). The net recharge in those areas is
17 projected to increase under all future climate scenarios (Figure 10). As a result, the
18 overall impact of climate change on groundwater levels appears to be less significant
19 in areas of the most shallow groundwater occurrence (<3 m).

20 Such observations can explain why the overall risk for GDTV is lower in the southern
21 areas of the study region, which appears to be less vulnerable to climate variability.
22 The lower degree of projected risk for all categories is due to the shallow watertable,
23 leading to relatively higher than elsewhere gross recharge sufficient to maintain a high
24 watertable under a climate with less rainfall and overall an increase in net
25 groundwater recharge (Figure 6). There is also limited groundwater abstraction in this
26 area, which appears to have a profound effect on GDTV risk in the north of the
27 region, as discussed below.

28 As mentioned above, GVDV risk criteria were defined based on the 30 years of
29 historical observations. It appears that the future climate projections are likely to lead
30 to exceeding the identified risk thresholds particularly for vegetation established over
31 a deeper groundwater (zone 3). The recent tree mortality cases, reported for the region
32 of Gngangara Mound (DEC, 2011), followed a few years of particularly low rainfall.

Those cases were related to vegetations in the regions of deep groundwater occurrence and were not found in the regions where groundwater was shallow. To some extent this validates the above observations, which are also discussed in the following section.

4.4 Effect of groundwater development on GDTV risk

Increase in groundwater abstraction is projected to have further effect on water level and net recharge (Figure 10 and Figure 11). When *median* and *development* scenarios are compared, additional groundwater abstraction leads to an increase in net recharge in the zone 1 and zone 2, while having a limited impact on net recharge in zone 3. As groundwater abstraction causes watertable drawdown, as illustrated on Figure 11, when *median* and *development* scenarios are compared, it also leads to reduction in groundwater losses to evapotranspiration, and hence net recharge. This is most profound in the areas with a shallow watertable (<6 m, zone 1 and zone 2).

Higher GDTV risks under both the historical (Figure 4) and future climate scenarios (Figure 9) were identified in regions with significant groundwater abstraction, i.e. around Gnangara Mound in the central part of the study area (to the east between Perth and Lancelin) and Jandakot Mound (to the west from Armadale). Under the historical and future climate scenarios, abstraction was applied within all production aquifers, including confined systems. Although the impact of groundwater abstraction from the confined aquifers on the watertable (and therefore GDTV) was not assessed, it is likely that the impact of groundwater abstraction from the unconfined Superficial Aquifer is more immediate. As the abstraction volumes were reported for individual groundwater management areas (GMAs) and included abstraction from all modelled aquifers combined, it is not feasible to define the relationship between the abstraction volumes and GDTV risk for the future climate scenarios.

However, under the future *development* scenario, which was based on the *median* climate scenario and full development of water resources up to the maximum allocation level, the modification in groundwater abstraction was applied only to the Superficial Aquifer. This allowed for comparing the increase in groundwater abstraction in individual GMAs with the relevant changes in GDTV risks between the *median* and *development* scenarios, caused by groundwater abstraction only (Figure 1212). The results indicate that within GMAs, groundwater abstraction led to an

1 increase in risk to GDTV for all depth-to-watertable zones. However, there were
2 differences between GMAs in the degree of change in GDTV risk, mostly likely due
3 to variations in hydrogeological conditions between GMAs.

4 **5 Discussion and Conclusion**

5 It has been demonstrated globally, that the most consistent projections in mean annual
6 rainfall reduction are related to regions within a Mediterranean climate type (Bates et
7 al., 2008). In such regions as the Mediterranean countries, California in the USA and
8 south-western Australia, drying climatic conditions are expected to lead to an
9 increasing dependency on groundwater resources, causing additional stresses to
10 water-limited ecosystems.

11 The hydrological consequences of future climate change and resource development on
12 groundwater-dependent ecosystems can only be estimated using predictive models,
13 including GCMs, recharge and groundwater models as well as vegetation response
14 models. However, these models are necessarily conceptual simplifications of reality,
15 based on limited observation and our current understanding of the system. Some
16 short-comings of the climate change projections on water resources, including
17 groundwater, have been discussed in recent publications. These issues commonly
18 relate to the wide range of GCM results (Allen et al., 2010; Crosbie et al. 2011) and
19 down-scaling methods for the same region (Crosbie et al., 2011; Holman et al., 2009;
20 Mileham et al., 2009), the limitation in projection of rainfall intensity and the
21 uncertainties in the vegetation response to changing conditions. The latter is likely to
22 have significant implications for water resource evaluation under a future climate as
23 terrestrial vegetation can influence both diffuse groundwater recharge and diffuse
24 groundwater discharge. Due to the paucity of long-term (decadal) analysis of
25 groundwater-dependent vegetation response to hydrological change, it is also unclear
26 if existing observations of vegetation response to variability in climatic conditions and
27 changes in groundwater levels are adequate for future projections (Naumburg et al.,
28 2005).

29 The present study is not an exception to such limitations; however, the adapted
30 methodology accounted for some of these short comings. Using multiple GCMs
31 provided a possible range of future climate scenarios rather than a single outcome. A

1 daily down-scaling approach posed certain limitations as historical climate data was
2 proportionally adjusted based on GCM projections (Mpelasoka and Chiew, 2009). As
3 reduction of rainfall was projected for the region, this led to an overall reduction in
4 rainfall intensity, which is also suggested by independent analysis (Charles et al.,
5 2010). It was also shown that daily down-scaling projections provide a wider range in
6 rainfall changes than the alternative methods, i.e. statistical (Fowler et al., 2007) or
7 dynamic down-scaling (Maraun et al., 2010), or both (Crosbie, et al., 2011).

8 The VFM adopted for recharge modelling in PRAMS (which takes into account the
9 effect of vegetation on water balance) is likely to reduce uncertainty in groundwater
10 resource sensitivity to climate change. More commonly, recharge is not dynamically
11 incorporated in groundwater modelling, and changes under future climate scenarios
12 have to be estimated independently. Although, it is not likely that VFM can reproduce
13 the changes in water-use efficiency by vegetation under the future climate and CO₂
14 emission scenarios, the model outputs compared well with groundwater levels in the
15 Superficial Aquifer in the historical analysis over the period 1984 to 2007.

16 The GDTV risk assessment framework used in this study is based on historical
17 observations for a selected vegetation type common to the Swan Coastal Plain
18 (Froend and Loomes, 2004) and as such may not adequately represent vegetation
19 response to changing hydrological conditions at the regional scale of this assessment.
20 Although application of the framework to historical records of hydrological change
21 and comparison with observed vegetation decline showed a degree of confirmation,
22 the variety of vegetation and soil types in the study region are likely to account for a
23 higher degree of variability in response and risk than predicted. In addition, there is
24 little information on the long-term response dynamics of phreatophytic vegetation in
25 the study region, outside of the Swan Coastal Plain (Froend and Sommer, 2010). Even
26 if GDTV identified as being at high risk is actually impacted (death of plants) by
27 groundwater drawdown in the future, the vegetation may demonstrate resilience and
28 persist under an altered ecohydrological state. However, the potential for true
29 resilience (recovery) may be limited for some vulnerable vegetation types if a
30 progressive reduction in water availability persists (Sommer and Froend, 2011).

31 In conclusion, groundwater-dependent ecosystems are diverse, uniquely adapted
32 features of the Western Australian landscape and their retention and the continuation

of the ecological services which they provide, requires considered water resource management. The results of our research, within the limitations described above, provide a quantitative and spatial representation of the potential pressures on vegetation from climate and groundwater consumption at a regional scale. The projected impact of future climate scenarios on those habitats varies spatially and for different GDTV types:

- There is a very limited impact on GDTV under future *wet* scenarios, while 30–40% of potential GDTV may be affected to some degree under a *median* future climate and 70–80% under a *dry* future climate. It is acknowledge though that climate change can lead to more complex impact on vegetation, such as duration of the dry seasons (which becomes increasingly longer), high temperature during some month and heatwaves and others, which may increase the overall ecological risks.
- The impact is expected to be most noticeable for terrestrial vegetation over deeper groundwater (6 to 10 m). This is due to a greater reduction in net groundwater recharge, leading to the reduction in groundwater levels, in those areas compared to the areas with shallower groundwater.
- Additional groundwater abstraction exacerbates the risk to GDTV.

Identification of ‘high risk’ represents a higher potential for unacceptable impacts (e.g. the death of all or part of current populations) to phreatophytic vegetation. Depending on the magnitude and (particularly) the rate of drawdown, groundwater-dependent vegetation can adapt at the individual and population level to new hydrological conditions (Sommer and Froend, 2011). Change in groundwater level is likely to be gradual in the absence of groundwater abstraction, and this may potentially lead to gradual changes in abundance and composition of GDTV.

Reduction in uncertainties in future climate projection and adequate representation of vegetation response to changing climate conditions should be a focus of future investigations of climate change impact on both water resources and vegetation resilience or adaptation to new conditions. The latter is related to both the biophysical assessment of vegetation water use in response to high atmospheric CO₂ concentration

1 along with a warmer climate and the adequate representation of such responses in
2 predictive models.

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GDTV zones in the Central Perth Basin

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