Independent review of science underpinning reductions to licensed water allocation volumes in the Lower Limestone Coast water allocation plan

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Executive summary

Background and structure of the review

In 2013, a Water Allocation Plan (WAP) was adopted for the Lower Limestone Coast Prescribed Wells Area (LLC PWA). The WAP set out the rules for protecting the resource for all water users, now and into the future. The WAP included proposed reductions to water allocations of irrigators that were identified as being required to achieve the goals of the WAP based on a risk assessment undertaken in 2012. The management areas subject to the largest reductions are Coles, Short, Frances, Hynam East, and Zones 3A and 5A. All six of these management areas had cuts to water allocations in 2016, with additional cuts planned for 2018 and subsequent years.

Due to concerns raised about the science that underpinned the 2012 risk assessment, the Minister for Environment and Water, The Honourable David Speirs MP, placed the 2018 reductions on hold to allow a review of the science to be conducted. The South East Natural Resources Management Board engaged the Goyder Institute for Water Research to undertake this review. An independent review panel was therefore established to consider the science that underpinned the policies that led to proposed reductions to allocations in these six management areas. Specifically, the Panel's tasks were to:

1) Review the science and data used to carry out the risk assessment in 2012 that led to water allocation reductions, focusing on groundwater, groundwater modelling, recharge rate assessments, groundwater-dependent ecosystems (GDEs), surface water – groundwater interactions, and ecology.
2) Review the science and data that have become available since 2012 that affects the assessment of risks to the groundwater resources (and dependent users, including the environment) in the LLC PWA; and
3) Make recommendations to the South East Natural Resources Management Board about the information to be used in the re-run of the risk assessment in 2019, and gaps in data that should be addressed to improve the science for the risk assessment or for the 2022 review.

Importantly, several topics were explicitly excluded from the Panel's brief, most notably a review of the volumetric conversion rates and deemed rates of water use and extraction depths of plantation forestry.

The Panel concentrated its review in four key areas, each presented as a chapter of this report and summarised in the subsections below. For the most part, the review relied on existing documentation although it proved necessary to undertake additional analysis of water level trends.

Total available recharge

Total potential water demand (total allocations plus forestry deemed rates and stock and domestic use) has previously been estimated to exceed 100 percent of total available recharge (TAR) for the six management areas: Coles (200%), Frances (140%), Hynam East (210%), Short (170%), Zone 3A (120%) and Zone 5A (140%). Despite multiple technical investigations, the uncertainty of recharge estimates across the region remains high, and the large fraction of recharge assigned to be available for allocation (90%) lacks a strong scientific basis. Nevertheless, despite uncertainty associated with their precise values, volumetric allocation limits are a useful management tool. Although water level and salinity trends are more reliable indicators of unsustainable use, they can only be measured at discrete locations. Therefore, the use of allocation limits helps prevent unsustainable patterns of groundwater use developing in areas that are remote from observation bores. The use of allocation limits also reduces the possible need for future reductions in allocation when unsustainable use rates are detected in observation bores. However, TAR estimates are too imprecise to be used as indicators of sustainable groundwater use.

Water level and salinity trends

In 2007–2012, water levels were declining in 28 out of 80 observation bores within the six focal management areas. This included declines in three of four observation bores in Coles; in five of 11 observation bores in Short; in all four observation bores in Frances; in two of five observation bores in Hynam East; in five of 45 observation bores in Zone 3A; and in nine of 11 bores in Zone 5A. Only 29 observation bores had data for the longer 2002–2012 period, but water levels were declining in all of these bores. From 2007–2012, salinity increased in 10 of 33 observation bores. These data were used in the subsequent risk assessment.

However, a different pattern is evident for some management areas for 2014–2018. Across the six management areas, of the 28 bores whose water levels were declining in 2007–2012, ten are no longer monitored; water levels in two are declining more rapidly; one is declining at the same rate; five are declining more slowly; and ten are no longer declining (nine are now rising and one is stable). Of the six bores whose water levels were declining at rates exceeding the trigger level of 0.1 m/y, one is no longer monitored and none of the others have water levels that are still declining at this rate. In Coles, Short, Zone 3A and Zone 5A, all bores that showed declining trends in water levels in 2007–2012 (and that are still monitored) either have increasing trends in 2014–2018, or reductions in the rate of decline. The reasons for the changes in trends over time have not been systematically investigated across the LLC PWA, or within the six key management areas, and so it is unclear whether or not the rising trends and/or reductions in rates of decline will be continued in the future.

Groundwater-dependent ecosystems

The science and data about wetland GDEs in the LLC PWA that were used to inform the risk assessment in 2012 exceeded equivalent knowledge about most of the GDEs elsewhere in Australia at the time. Regional inventories had shown that less than six percent of the original wetland extent in the South East remained, mostly in degraded and fragmented condition, with 77 percent of the remnant wetlands highly likely to be groundwater-dependent. Biodiversity in many of these remnant wetlands had declined and species composition of aquatic plants had shifted to those more tolerant of saline conditions. Satellite images dating back to 1987 show that in the elevated areas of the southern Naracoorte Ranges (Hydrogeological Zones HZ3 and HZ7), many wetland GDEs have changed from permanent or seasonal wetlands to now being consistently dry whereas those of adjacent flats (HZ5 and HZ6) partially recovered in recent years with above-average rainfall. There are now reduced flows down riverine GDEs such as Mosquito Creek which enters the Ramsar-listed Bool Lagoon. Frances and Hynam East appear to lack wetland GDEs but they may support subterranean and terrestrial GDEs which may be at risk from changes to the groundwater regime. Zone 5A, Short, Coles and Zone 3A support respectively 27 (70 ha), 328 (580 ha), 366 (1352 ha) and 1122 (4680 ha) wetlands. One wetland in Coles (32 ha) and three in Zone 3A (totalling 131 ha) are ranked as high-value GDEs.

Declines in groundwater levels and/or increases in salinity should be sufficient justification for ameliorative action, particularly when high-value GDEs are present. The Panel believes that the rate of water level decline that is considered unsustainable over a 5-year window should depend on the presence, value and resilience of GDEs within the area. Larger water level declines might be permitted for short periods of time if GDEs are not present within the immediate vicinity. Where important and sensitive GDEs are present, lower rates of decline may be tolerable as long as lasting damage to the GDEs will not occur. Implementation of such an approach requires that all GDEs have been identified and sufficiently assessed, and their likely ecological responses are fully understood. This is currently not the case. Furthermore, subterranean and terrestrial GDEs in the LLC PWA have not been fully surveyed. In management areas with greater depths to groundwater (>5 m), remnant fragments of groundwater-dependent terrestrial native vegetation probably represent important refuges for native plants and animals in the largely cleared landscape. Studies that link ecological condition of GDEs to water regime in the LLC PWA are rare, yet essential to guide reliable risk assessments for GDEs, especially those deemed of high value.

Risk assessment

The 2012 Risk Assessment uses a series of look-up tables of groundwater and ecosystem condition and community and ecological dependency. Cut-off values separate levels within each category, and simple equations and weights allow the individual category scores to be combined to provide an overall risk rating for each management area. These overall risk ratings underpinned the decision to reduce water allocations in some management areas.

Although the general risk assessment approach is commendable, the Panel has identified several anomalies in the assessment. In particular, the reasons for the cut-off values between the different risk levels are unclear and, in some cases, this materially affects the classification of risk. The relative weights ascribed to the different factors are also not clearly justified and sometimes appear arbitrary, potentially creating further anomalies in the resulting risk assignments. For example, the groundwater consequence score determined for GDEs in each management area is always less than (or occasionally equal to) the community consequence score. As only the higher of these two scores is ultimately used in the assessment of risk, GDEs effectively have no influence on this aspect of the final risk rating. It is also inappropriate to use aquifer thickness as part of the assessment of ecological consequence (as is currently the case) because surface-expression GDEs (e.g., groundwater-dependent wetlands and terrestrial vegetation) may lose essential access to the groundwater if the water table falls, irrespective of the thickness of the aquifer system. Data confidence categories for aquifer thickness also did not consider all available information, and this has a material effect on some of the risk assignments. Perhaps most importantly, it is unclear how the water level and salinity trends observed in individual bores were combined to determine overall trends for the different management areas; in Coles and Short the overall rate of water level decline assigned to the management area is greater than the trend in any observation bore within that management area. These multiple anomalies lead to specific concerns with the high-risk ratings determined for many of the management areas, and generally undermine confidence in the risk assessment process. Notwithstanding this, there is sufficient information in water level and salinity trends and GDEs to inform water allocation decisions.

Key recommendations

To improve understanding of the groundwater resource and GDEs, and to address some of the issues raised in this review, the Panel has made multiple recommendations at the end of each chapter. For brevity in this Executive Summary, we present five high-level recommendations:

1. That coverage of the existing monitoring network is expanded (while maintaining existing bores), and reasons for changes in water level trends over time are more systematically examined than has been the case to-date. The salinity monitoring network should also be expanded;
2. That groundwater modelling of the LLC PWA is updated to include a suite of subregional models that can answer specific questions at an appropriate range of spatial and temporal scales;
3. That 1-D spatial unsaturated zone modelling is conducted to examine expected salinity increases due to historic clearing of vegetation and/or recycling of groundwater associated with irrigation activities and forestry plantations;
4. That hydrological and ecological monitoring of wetland GDEs is expanded (including expanded use of satellite data), and that reconnaissance assessments of subterranean and vegetation GDEs be carried out as a matter of urgency; and,
5. That future risk assessments use a greater number of levels of the different factors; better justify the cut-off values between the levels; and examine the influence on the final risk assignment of the weightings applied to the different factors. These changes would improve confidence in results of the risk assessment process.

Management area summaries

There is a large uncertainty associated with estimation of total available recharge, and the Panel identified several anomalies in the Risk Assessment that underpinned the decision to reduce water allocations.

Nevertheless, there is a large amount of information on GDEs and water level and salinity trends that together can underpin water allocation decision-making in the six management areas of particular interest for this review:

Coles:

Widespread declines in water levels were observed in 2002–2012 (10-year trend; 3/3 bores) and in 2007–2012 (5-year trend; 3/4 bores) although there has been significant recovery since that time (0/3 bores have declining levels in 2014–2018). This management area contains a single salinity observation bore, and this shows rising salinity. Coles supports 366 identified wetland GDEs, covering 1352 ha, with one wetland that is ranked as a high-value GDE. The reasons for the apparent recovery in water levels within the past few years should be investigated to determine whether it is likely to be sustained. If water level recovery is not sustained, then the declining water levels will negatively impact GDEs and stock and domestic users. The number of water level and salinity observation bores within this management area should be increased to improve confidence in future water level and water quality assessments.

Short:

Widespread declines in water levels were observed in 2002–2012 (4/4 bores) and in 2007–2012 (5/11 bores), but with significant recovery since then (0/6 bores have declining levels in 2014–2018). One of four salinity observation bores shows rising salinity. Short supports 328 wetland GDEs, covering 580 ha. Reasons underlying the apparent recovery in water levels within the past few years should be investigated to assess whether it is likely to be sustained. This is especially important in view of the large number of GDEs in this management area.

Frances:

This management area displays widespread and continuing declines in water levels (2/2 bores in 2002–2012, 4/4 bores in 2007–2012 and 3/3 bores in 2014–2018). Although no wetland GDEs have been identified within this management area, other GDEs are likely. Declining water levels may impact irrigators and stock and domestic groundwater users as well as unidentified GDEs.

Hynam East:

Significant declines in water levels were observed in 2002–2012 (4/4 bores) and in 2007–2012 (2/5 bores), with some recovery since then (1/3 bores has a declining level in 2014–2018). One of two salinity observation bores has rising salinity. In view of the contrasting water level and salinity trends (3/5 bores display rising water level trends in 2007–2012 as do 2/3 bores in 2014–2018), the Panel believes that the number of observation bores is insufficient. It is recommended that reasons for the divergent water level trends are examined, and additional water level and salinity monitoring bores are installed.

Zone 3A:

While all available observation bores showed declining water level trends in 2002–2012 (13/13 bores), only 3 of 45 bores displayed declines in water levels in 2007–2012, with some showing more recent recovery (1/31 bores showed a declining trend in 2014–2018). Six of 15 salinity observation bores show rising salinity. Of the six focal management areas, Zone 3A supports the most wetland GDEs (1122 covering 4680 ha), with three (totalling 131 ha) rated as high-value GDEs. The Panel recommends that the reasons for recovery in water levels and the rising salinity be investigated. Threats posed by the declining water levels and rising salinity to the high-value GDEs should be specifically examined.

Zone 5A:

Nine of 11 bores displayed declining water level trends in 2007–2012, and water levels in half of these continue to decline (5/10 bores have declining trends in 2014–2018). Water levels in all three available observation bores also declined in 2002–2012. One of eight salinity monitoring bores has rising salinity. Zone 5A supports 27 identified wetland GDEs, covering 70 ha. The widespread and sustained water level declines within this management area pose risks for GDEs, irrigators and stock and domestic groundwater users.

1 Background and scope

In 2013, a Water Allocation Plan (WAP) was adopted for the Lower Limestone Coast Prescribed Wells Area (LLC PWA). The WAP set out the rules for protecting the resource for all water users, now and into the future. The WAP included proposed reductions to water allocations of irrigators that were identified as being required to achieve the goals of the WAP based on a risk assessment undertaken in 2012. The management areas subject to the largest reductions are Coles, Short, Frances, Hynam East, and Zones 3A and 5A. All six of these management areas had cuts to water allocations in 2016, with additional cuts planned for 2018 and subsequent years.

Due to concerns raised about the science that underpinned the 2012 risk assessment, the Minister for Environment and Water, The Honourable David Speirs MP, placed the 2018 reductions on hold to allow a review of the science to be conducted. The South East Natural Resources Management Board engaged the Goyder Institute for Water Research to undertake this review. An independent Expert Panel of scientists was established to undertake this review. The Panel comprised of Professor Craig Simmons (Chair), Professor Peter Cook, Dr Lu Zhang and Professor Andrew Boulton.

This independent review considers the science that underpins the policies that led to proposed reductions to allocations, specifically in management areas Zone 5A, Zone 3A and Hynam East and its implications for three other management areas, the Hundreds of Coles and Short and the Frances Management Area in Zone 6A (Figure 1). Specifically, the Expert Panel's tasks were to:

1. Review the science and data used to carry out the risk assessment in 2012 that led to water allocation reductions, focusing on groundwater, groundwater modelling, recharge rate assessments, groundwater-dependent ecosystems (GDEs), surface water – groundwater interactions, and ecology.
2. Review the science and data that has become available since 2012 that affects the assessment of risks to the groundwater resources (and dependent users, including the environment) in the LLC PWA.
3. Make recommendations to the South East Natural Resources Management Board about the information to be used in the re-run of the risk assessment in 2019, and gaps in data that should be addressed to improve the science for the risk assessment or for the 2022 review.

The development of a revised risk assessment method did not form part of the scope of the study. In addition, the following items were specifically omitted from the study scope, as they would require a full review of the LLC Water Allocation Plan (WAP):

- Volumetric conversion rates
- Forestry deemed rates and extraction depths
- The ability to trade and the hydrological assessment technique
- Hydrogeological management boundaries
- Baseline period for GDEs and forestry
- Policy interpretation of science around trigger levels

The documents that formed the basis of the 2012 Risk Assessment were most critically assessed as part of this review. These were:

1. Brown K, Harrington G and Lawson J (2006) Review of groundwater resource condition and management principles for the Tertiary Limestone Aquifer in the South East of South Australia. South Australia. Department of Water, Land and Biodiversity Conservation. DWLBC Report 2006/2.
2. Latcham B, Carruthers R and Harrington G (2007) A new understanding on the level of development of the unconfined Tertiary Limestone Aquifer in the South East of South Australia. South Australia. Department of Water, Land and Biodiversity Conservation. DWLBC Report 2007/11.

3. SKM (2009) Classification of groundwater – surface water interactions for water dependent ecosystems in the South East, South Australia. Final Report, April 2009.
4. Conesa D, Harding C and Mustafa S (2015) Risk Assessment. Risk to the Tertiary Limestone Aquifer and its users from current allocation and extraction. Prepared for the 2013 Lower Limestone Coast Water Allocation Plan. Department of Environment, Water and Natural Resources. Version 14, January 2015.
5. Cranswick RH (2018) Groundwater resource condition assessment of the eastern Lower Limestone Coast PWA. Draft of DEW Technical Report, April 2018.

Document 4, above, is referred to as the 2012 Risk Assessment (or simply the Risk Assessment) throughout the current report, as it had been substantially completed at that time, and was used to inform the 2013 Water Allocation Plan. Other documents and computer files that were used in preparation of this review are listed in Appendix A. In additional to reviewing key documents, the Panel also met with stakeholder groups and government representatives. Individuals who were consulted are listed in Appendix B.

There has been a large number of groundwater studies within the LLC PWA, and the task of the Review Panel is not to summarise this work. Therefore, some familiarity with the previous work is assumed by the reader of this document, although a very short summary focusing on the concerns about changing groundwater levels and salinities is provided here.

In 2006, Brown et al. (2006) reviewed levels of groundwater use and water level and salinity trends in the LLC PWA and identified declining groundwater level trends in management areas Frances, Joanna, Kongorong and Zone 5A, which they attributed (at least in part) to groundwater use for irrigation. Coles and Hynam East were not identified as hot spots based on water level and salinity trends, but had rates of water use exceeding estimated available recharge. The number of management areas that were considered to have unsustainable levels of use (due to declining groundwater level trends or rising groundwater salinity) or allocation rates exceeding estimated rates of available recharge changed over time as recharge rates were revised (e.g., Latcham et al., 2007) and additional groundwater data became available. The 2012 Risk Assessment identified eight management areas with High- or Very High-risk ratings: Coles, Frances, Hynam East, Myora, Short, and Zones 2A, 3A and 5A. Myora and Zone 2A had reductions to their allocations in 2016. The other six management areas had partial reductions in 2016, with further reductions scheduled to occur in 2018, 2020 and 2022. The 2018 reductions have not yet been implemented.

This review is divided into seven sections. Section 1 describes the background and scope of the review. Section 2 describes the general philosophy of groundwater management, particularly as it relates to the use of volumetric allocations. This is provided as a background to some of the discussion that follows. The subsequent sections contain the Panel's review of the data and science that underpin the 2012 Risk Assessment, recent data and science that have become available since the Risk Assessment, and additional data and science that might improve future iterations of the Risk Assessment. Section 3 deals with aquifer recharge; Section 4 with water level and salinity trends; Section 5 with groundwater-dependent ecosystems; and Section 6 with the Risk Assessment itself.

This focus of this review is the condition of the unconfined aquifer (trends in depth to water and water quality) and its associated GDEs in the management areas of Coles, Short, Frances, Hynam East, Zones 3A and 5A because these areas are where the main reductions in allocations are targeted. Specific questions, compiled by the South East Natural Resources Management Board in consultation with stakeholder groups, were also asked of the Panel, and answers to these are provided in break-out boxes in the relevant sections of the report. One question about data gaps in the assessment is not specifically addressed in a break-out box because this topic is discussed throughout the report.

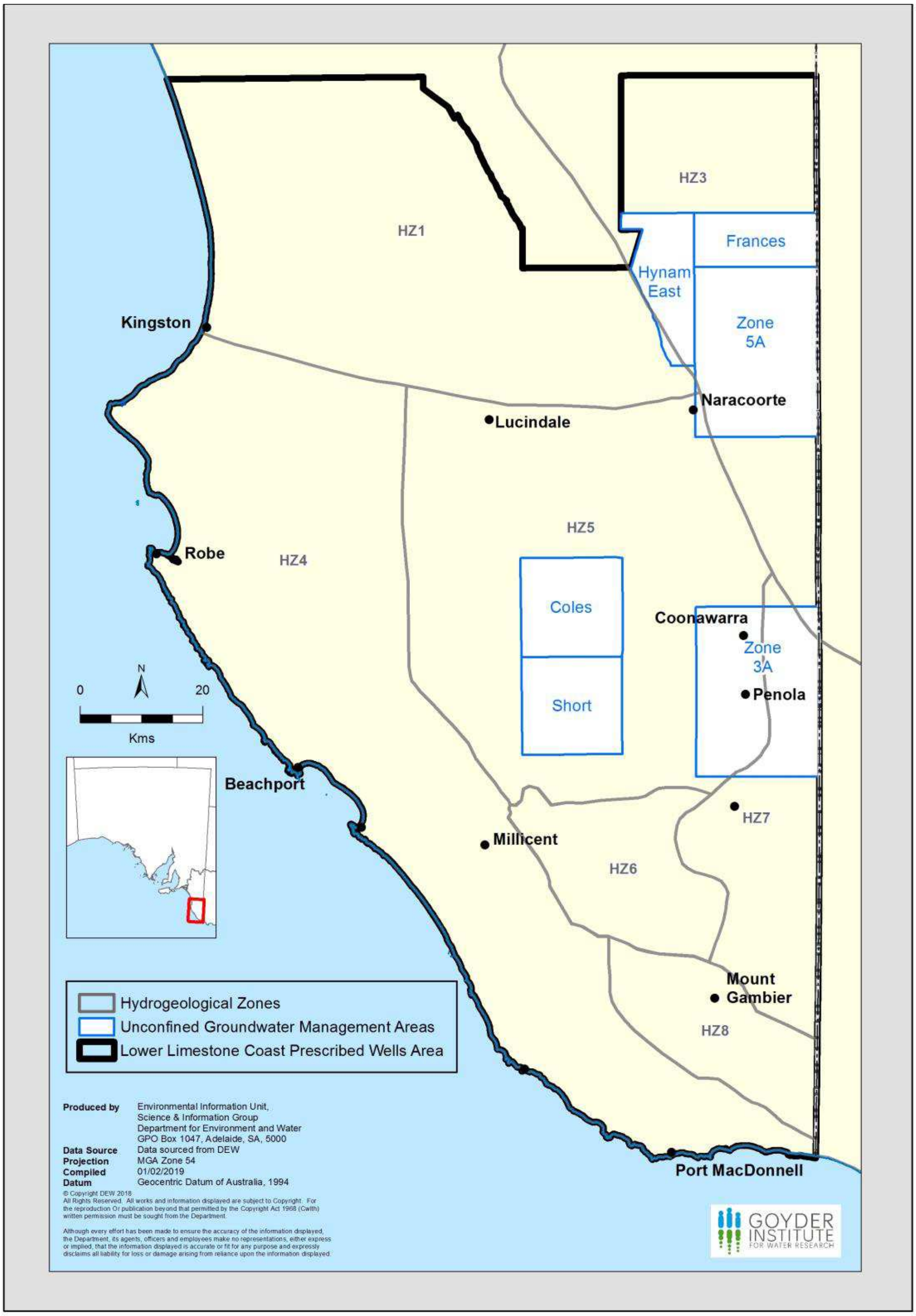

Figure 1. Location of the Lower Limestone Coast Prescribed Wells Area, showing hydrogeological zones (Cranswick, 2018) and the six key groundwater management areas that are the focus of the current review.

2 Groundwater management approaches

Traditionally, groundwater was managed by ensuring that groundwater extraction was below the long-term recharge rate - a concept referred to as ‘safe yield’. However, this approach has been widely criticised in the scientific literature, principally because it ignores the dependence of ecosystems on groundwater and so can have serious environmental consequences (Bredehoeft, 1997; Sophocleous, 2000; Bredehoeft 2002). Any groundwater extraction upsets the hydrological equilibrium of a groundwater system, and sustained extraction will cause long-term and potentially irreversible ecological impacts. Initially, the effect of extraction is to produce a loss in storage and a corresponding decline in water levels. However, in time a new configuration of the water balance is reached (assuming constant rates of extraction and stable climatic conditions), with the natural discharge now reduced by an amount equal to the volume extracted. Ecosystems that depend on natural groundwater discharge will necessarily be impacted.

Since all groundwater pumping will have environmental impacts, there is a need to assess what the impacts of different levels of pumping will be in the short-term, and once the system reaches a new hydrological equilibrium. The ‘sustainable yield’ of the system can be thought of as the amount of discharge that can be acceptably reduced. The question of what is acceptable should consider both community values and the needs of future generations. It also should acknowledge that ‘novel’ or ‘hybrid’ ecosystems (Hobbs et al., 2014) will likely emerge once the system reaches a new equilibrium after sustained groundwater extraction. For example, some GDEs may be irretrievably lost whereas others will be substantially altered (e.g., from permanently inundated groundwater-fed wetlands to seasonally inundated wetlands with very different plants and animals).

Having recognised that the environment is a legitimate user of groundwater, it is tempting to determine an ‘environmental allocation’ and hence to define the ‘sustainable yield’ to be some fraction of the long-term recharge rate, with the remaining amount available for the environment. However, there are several problems with this approach. The first, of course, is in defining the volume of water that is required by the environment (or in this case, various GDEs of the LLC PWA) to maintain function and persistence. The second, is that if groundwater extraction lowers the water table, some types of GDEs (Section 5) may no longer have access to the water resource (e.g., if the water table drops below the root zone of groundwater-dependent vegetation). Similarly, groundwater extraction that alters the groundwater regime and water quality will have different effects on different GDEs in a given landscape (Kath et al., 2018). Therefore, the environmental consequences of groundwater extraction will depend on the characteristics of individual GDEs and their distributions in the landscape (Section 5), the underlying groundwater system and the particular extraction regime, not simply on the volume of water that is taken.

Box 2.1. In 2012, was the use of Total Available Recharge (TAR) as the basis for allocations a sound approach for achieving the objectives of the WAP? Was the principle of assigning 90% of estimated recharge to determine the TAR technically robust in view of the objectives of the WAP? Was the principle of assigning 10% of estimated recharge for the environment appropriate?

The use of volumetric limits on groundwater use (i.e., volumetric allocations) is an effective means for managing groundwater at regional scales but will not necessarily prevent local areas of water table decline and associated impacts. A combination of volumetric limits, water level triggers and buffer zones are necessary for local-scale management and to protect dependent ecosystems. Use of Total Available Recharge (TAR) is a reasonable basis for determining regional allocations, and it is common practice to set allocations as a fraction of recharge. The 90% figure for allocation for human use (10% for the environment) within the Lower Limestone Coast is arbitrary, and towards the high end of values typically applied within Australia. Lower allocation fractions for human use are more usual in areas where there are groundwater-dependent ecosystems deemed to be valuable or of high priority. Nevertheless, allocation of a fraction of recharge for the environment will not necessarily achieve environmental goals, as any groundwater extraction can cause a reduction in groundwater levels, which can reduce the access by GDEs to the aquifer.

Box 2.2. What would be the likely hydrogeological response if reductions were based on usage and not allocation?

Reducing groundwater allocations does not necessarily reduce groundwater use, particularly if the level of use is less than the allocation. Almost certainly, the reduction in use (in percentage and absolute terms) will be less than the reduction in allocation.

The hydrogeological response will depend on the level of usage reduction, not the level of reduction in allocation. Other factors being equal, a reduction in use should lead to a reduction in the rate of water level decline. It may lead to a water level rise in some areas. However, the magnitude of the response is difficult to predict without a reliable groundwater model. This does not currently exist for the LLC PWA.

In the LLC PWA, allocations have been reduced in areas where they were greater than 90% of Total Available Recharge (TAR), and they have been reduced to the level of 90% of TAR. This will not ensure that the groundwater level decline will cease, and that groundwater-dependent ecosystems will be protected. However, while it is difficult to predict the hydrogeological response, greater reductions in use will lead to a greater likelihood of a reduction in water level decline. If usage rather than allocation was restricted to 90% of TAR, the likelihood of a reduction in the rate of groundwater decline would be less.

However, restricting allocations to 90% of TAR will not necessarily be sufficient to arrest the decline in water levels and protect groundwater-dependent ecosystems. Greater reductions may be required to achieve this objective.

Managing groundwater using volumetric limits alone is seldom sufficient to protect ecosystems that depend on groundwater (Box 2.1, Box 2.2), and other management tools and approaches will be required. These typically include (i) monitoring water levels near high-priority GDEs and reducing groundwater extraction when water levels drop below specified limits (trigger levels); (ii) excluding groundwater extraction from within defined distances of these GDEs; or (iii) a combination of the two strategies (Noorduijn et al., 2018).

The volumetric allocation approach embodied in the sustainable yield concept is useful because managing to water levels alone is too crude, due to the time delay between pumping and impact at a remote location. Without appropriately chosen volumetric limits, a situation might develop in which large allocations are initially awarded which then have to be reduced once bore levels inevitably decline. Although it is clear that the annual allocation volume should be less than the recharge rate, the exact figure represents a balance between the dual goals of groundwater development and environmental protection. It cannot be determined from scientific analysis, without pre-defining the locations and extraction volumes of individual pumping bores and acceptable levels of groundwater level decline at specific locations. It is important also to acknowledge that the use of volumetric allocations does not mean that there is a tipping point where a resource moves from being used sustainably to being overused. All groundwater extraction will have impacts on GDEs; the questions are about the spatial and temporal timescales over which these impacts occur, whether we are able to accurately measure them and whether the impacts are deemed ‘acceptable’.

More sophisticated approaches to groundwater management rely on groundwater models. Numerical groundwater models can predict the spatial distribution of groundwater drawdown over time due to groundwater extraction, and can hence predict changes in groundwater availability to ecosystems. If individual licence applications are assessed against the predictions of a groundwater model, then the need for volumetric allocation limits, buffer zones and trigger levels is lessened. Where it is not feasible to rerun model simulations for each new license application (or trade), groundwater models can be used to develop more robust management approaches, and to determine separation distances of extraction bores from other users and from important ecosystems that will prevent unacceptable impacts. Of course, development of groundwater models has its own challenges, and the development of reliable models relies on appropriate conceptualisation of aquifer systems, and availability of field data for calibration.

3 Estimates of aquifer recharge

Recharge is the amount of water that infiltrates into an aquifer. It is measured as a volume of water per unit area per unit time, usually as mm/year. Recharge is controlled by climatic factors (e.g., rainfall and potential evaporation) and catchment characteristics (e.g., land use and aquifer properties). It is arguably the most difficult water balance component to determine and large uncertainty is generally expected for recharge estimation. Although few studies have attempted to rigorously quantify uncertainty, estimates of recharge uncertainty due to uncertainty of parameters used in calculations range between 10% and more than 100% (Cook et al., 1995; Timlin et al., 2001; Xie et al., 2018). Even greater uncertainties are likely due to assumptions of some of the methods (termed 'structural uncertainty'), and difficulties in extrapolating from point measurements to regionally-averaged values.

Several methods are available for estimating recharge and the choice of a method depends on the spatial scale of interest (e.g., plot or catchment) and data availability (Walker and Zhang, 2002). When additional water is added to an aquifer by recharge or removed by discharge over a relatively short period of time, then one would expect the water table to fluctuate. This is the basic idea behind the water table fluctuation (WTF) method for estimating recharge, which has been used in the South East (Brown et al., 2006). The WTF method requires accurate estimation of aquifer specific yield. Specific yield is defined as the volume of water released from storage by an unconfined aquifer per unit surface area of aquifer per unit decline of the water table, and varies from 0.01 to 0.30 depending on the texture, bulk density and pore structure of the aquifer material. The time for rainfall to induce a response at the water table may vary from hours for a shallow groundwater system (water table depth less than a few metres) to days, months or even years for a deep groundwater system (water table depth several metres or tens of metres). For this reason, areas with deep water tables do not show rapid responses to rainfall, and so the WTF method is problematic in these areas.

It is important to differentiate the recharge caused by a single rainfall event from average recharge over a longer period (e.g., seasons); the latter is more important from a management point of view. The long-term trend in the water table can indicate whether the system is in equilibrium (i.e., average recharge equals average discharge). A declining water table generally indicates decreased recharge or increased discharge, or both (see Section 4). Water table fluctuations must be interpreted together with other information about nearby activities, such as pumping and land use change, to obtain appropriate recharge estimates. Recharge can vary greatly over a catchment and this makes it difficult to obtain reliable areal average estimates of recharge.

In the Water Allocation Plan (WAP) for the LLC PWA, the total available recharge (TAR), which is the volume of water that is considered to be available for use each year, is calculated as 90% of the estimated annual recharge to the unconfined aquifer. Clearly, accurate estimation of recharge is important for the implementation of the WAP and significant effort has been made towards recharge estimation. However, the 90% figure for the fraction of recharge that is allocated is arbitrary, and not justified in any of the reports examined by the Panel (see Box 2.1). As explained in Chapter 2, while setting allocation limits as a fraction of recharge is a useful management approach, the chosen fraction cannot be easily determined using scientific methods. To some extent, improved estimates of recharge will result in more sustainable allocation limits, however, since recharge is multiplied by an arbitrary fraction to determine TAR, increasingly precise estimates of recharge do not necessarily result in more robust TAR values. However, accurate estimates of aquifer recharge are important for our understanding of the groundwater system, and are required for groundwater models (see Chapter 4).

3.1 Information available to 2012

There has been a long history of recharge studies within the LLC WAP region (e.g., Allison and Hughes, 1978; Walker et al., 1990; Wood, 2011; and references therein), although most of these have been site-specific and difficult to extrapolate across the region. Given the extensive number of observation bores available in most of the LLC PWA, the water table fluctuation (WTF) method has generally been used for regional assessments, and was used to estimate recharge for each management area, assuming a specific yield of 0.1 (Brown et al.,

Box 3.1. Were all estimates of recharge that were available at the time considered? Was the method by which the recharge volume for each management area was determined appropriate, in view of the science used to support it?

Yes, all estimates of recharge that were available at the time were considered and these included recharge estimates using the groundwater table fluctuation method (Brown et al. 2006), environmental chloride and tritium method of Allison and Hughes (1978) and Bradley et al. (1995). These recharge estimates were made using the best methods and data available at the time. Studies carried out since 2012 evaluated and compared the recharge estimates for the South East region and concluded that the groundwater table fluctuation method produced consistently higher recharge estimated compared with the chloride mass balance and water balance methods. Recharge is the most difficult water balance component to estimate and it can vary considerably both in space and time. This suggests that large uncertainty can be expected with any recharge estimates and it is important to quantify their uncertainty.

2006). In areas where the WTF method proved to be inappropriate, the recharge rates were determined from the geochemical approach used by Allison and Hughes (1978). Recharge estimates for individual management areas were made by area-weighting the different soil/land-use types (Bradley et al., 1995; Brown et al., 2006). The resulting recharge rates showed considerable variation ranging from 15 to 200 mm/year with a median of 83 mm/y and a geometric mean of 73 mm/year. In a few cases, however, the method used to determine the recharge rate is not clearly described and difficult to determine from the documentation.

The WTF method was also used to estimate the change in recharge with time using monitoring data collected over the period of 1970 to 2012 with an average record length of 21 years (Brown et al., 2006). The recharge showed a long-term decreasing trend over this period. However, for some bores the decreasing trend is not statistically significant.

For the management areas that are the focus of this study, total recharge was estimated to be 28,031, 4,881, 3,973, 33,997, 60,176 and 20,867 ML/y for Coles, Frances, Hynam East, Short, Zone 3A and Zone 5A, respectively. These represent mean recharge rates of 120, 30, 25, 150, 120 and 40 mm/y, respectively. Based on these values, the total potential water demand (total allocations plus forestry deemed rates and stock and domestic use estimates) is estimated at 200, 140, 210, 170, 120 and 140 percent of TAR for Coles, Frances, Hynam East, Short, Zone 3A and Zone 5A, respectively (SENRMB, 2015; Appendix D).

Recharge is the most difficult water balance component to estimate and it can vary considerably both in space and time (Scanlon et al., 2002). The methods used for estimating recharge in the WAP were appropriate (Box 3.1) and represented the best estimates available at the time. However, the WTF recharge estimates may not be accurate due to the lack of accurate estimates of the specific yield and possible unrepresentativeness of the groundwater bores used in some management areas. Other factors such as pumping may also affect the accuracy of the WTF recharge estimates. The various methods for estimating recharge integrate over different spatial and temporal scales, and so their reconciliation can be challenging. Although estimating uncertainty of recharge estimates can be difficult, it is noteworthy that the median difference between recharge rates estimated by Brown et al. (2006) for each management area and those available prior to this study was approximately 50%. This provides some indication of the uncertainty in the values.

3.2 Information available since 2012

Several studies have been conducted since 2012 to estimate recharge in the South East, including Crosbie and Davies (2013) and Crosbie et al. (2015). These studies updated the recharge estimates of Brown et al. (2006) with longer records of water table observations and better represented the effects of climate variability on recharge because the records included observations from a relatively dry period. These studies also compared the recharge estimates using the WTF method with those obtained from water balance modelling incorporating satellite-derived evapotranspiration estimates. They also identified some weaknesses of the previous recharge studies, including inaccurate estimates of the specific yield and the effects of pumping on water table fluctuations.

While the most recent studies used the water balance method for estimating recharge with satellite-derived estimates of evapotranspiration (Crosbie et al., 2015; Doble and Crosbie, 2017), due to uncertainties associated with this method, the authors chose to calibrate it using the WTF method. There is some uncertainty associated with this calibration. Nevertheless, the method can be used to produce recharge estimates with much higher spatial and temporal resolutions than either the WTF or geochemical methods. After calibration, the water balance method produced lower estimates of recharge than those currently used in the WAP, with the greatest discrepancies in Coles and Short. In these management areas, adopted recharge rates in the WAP are 120 and 150 mm/y, respectively, whereas the water balance method indicated negative net recharge for the 2001 – 2010 period. Note that net recharge is defined as gross recharge minus evapotranspiration from groundwater and hence net recharge can be either positive or negative. Positive net recharge is generally found in cropping and pasture areas, while negative recharge is often associated with plantation areas. Coles and Short have the most commercial forestry of the six management areas that are the focus of this review, and this is probably the cause of the discrepancy. The water balance method measures net recharge, whereas the WTF should reflect gross recharge. The negative recharge rates estimated by the water balance for Coles, Short and Zone 5A are consistent with declining water level trends in 2002 – 2012. However, further analysis is needed on the implications of the water balance estimates of net recharge for water allocation in the LLC PWA.

3.3 Key findings and recommendations

Despite a number of studies to assess groundwater recharge within the LLC PWA region, the uncertainty of the recharge estimates remains large. The large uncertainty is due to assumptions involved with most of the methods used to estimate recharge, and difficulties in extrapolating from point-based approaches to regional averages. Further, improved estimates of recharge do not necessarily lead to a more rigorous estimate of TAR, as the recharge rate is multiplied by a fraction (here 90%) that has been arbitrarily determined. Due to the large uncertainty in TAR, and issues discussed in Section 2, ratios of groundwater allocation or use to TAR are poor indicators of sustainability. The Panel has a number of recommendations to improve estimates of recharge within the LLC region, although due to the above difficulties, we do not consider this work to be the highest priority. Nevertheless:

3.1. The Panel recommends that studies are conducted to obtain accurate estimates of the specific yield and its spatial variation. This would help to produce better recharge estimates in the LLC PWA. The recharge estimates rely heavily on the WTF method, which for simplicity assumed a constant value of the specific yield of 0.1. In reality, the specific yield depends on the texture, bulk density and pore structure of the aquifer material.

3.2. The Panel recommends that an assessment of the possible role of processes other than recharge on water level changes (e.g., groundwater pumping) is undertaken to determine their potential effect on recharge estimates. It is important that bores used for recharge estimation are representative of the region and not affected by local-scale processes, an issue raised by the local community.

3.3. The Panel recommends that uncertainties associated with variables used to calculate recharge are assessed, and that corresponding uncertainties in recharge rates are quantified. Some of the uncertainty in recharge is associated with the assumptions inherent to the different methods, and some is due to the approaches used to extrapolate point scale estimates to regional averages. Uncertainty resulting from some of the assumptions can be difficult to quantify, and best assessed by comparing recharge rates obtained using different methods. The effect of recharge uncertainty on allocation/TAR values should be explicitly acknowledged when these values are presented, and inherent uncertainty should be incorporated into use of this data in the risk assessment.

3.4. The Panel recommends further work to compare recharge rates obtained using different methods within the LLC PWA, and particularly to examine differences between gross and net recharge rates, and to compare net recharge rates with water level trends.

4 Water level and salinity trends

Short-term changes in groundwater levels can result from short-term changes in pumping rates or recharge rates, both of which may be consequences of climatic variations. Over longer timescales, water table decline can result from increases in groundwater extraction or changes in the locations of extraction bores. In this case, the water table decline occurs as the groundwater system adjusts to the new extraction regime. It does not necessarily mean that groundwater use is unsustainable, but rather that rates of loss from the groundwater are currently greater than rates of replenishment in the immediate vicinity of the area of extraction. As the system adjusts, water levels should eventually stabilise, but it can be difficult to predict when (and at what levels) this will occur. It can also be difficult to distinguish between short-term groundwater level fluctuations, groundwater level adjustments to changes in patterns of groundwater extraction, and groundwater level declines due to unsustainable use. Salinity increases can result from inflow of more saline water in response to groundwater pumping, or from infiltration of surplus irrigation water. Salinity increases can also result from leaching of salt from the unsaturated zone, and this can be triggered by an increase in recharge associated with land use change.

Since water table position is generally determined from observation bores, the issue of *representativeness* of observation bores requires consideration. Of course, the rate of water level decline will vary across management areas, depending on proximity to points (or areas) of groundwater extraction (Box 4.3) and variation in recharge and aquifer properties. Other factors being equal, water levels in bores close to points of groundwater extraction will decline at faster rates than in bores further from such points. However, it does not make sense to disregard bores near points of groundwater extraction any more than it makes sense to disregard bores that are distant from extraction points. If numerous bores are available and distributed across the management area both near and far from groundwater extraction points, then this will give a more reliable picture of average water level trends. Observation bores should also be located in particular areas of interest, such as close to GDEs or other groundwater users. Where only a few observation bores are available, the most reliable approach is to use an appropriately calibrated groundwater model to estimate likely water level trends in areas where observation bores are not available. Such models also have value in predictive analysis (i.e., what will future declines be under different groundwater use scenarios?). A reliable and robust groundwater model could also be used for quantifying and understanding impacts of the relative contributions of groundwater abstractions and climatic variations or climate change on water level trends.

The LLC WAP defines 'resource condition triggers' for rates of groundwater level decline and salinity increase to protect the resource. These are: (i) a mean decrease in water level of no more than 0.1 m/y; and (ii) a mean increase in salinity not exceeding 2% per year, both measured over the preceding 5-year period. However, the Risk Assessment also considers lower rates of water table decline and salinity increase, as well as water level trends over a 10-year period. Although the above trigger values have not been justified, they are reasonable values to ensure protection of most GDEs and other groundwater users, as discussed below. A 5-year window is relatively short for revealing trends, as it is readily affected by climate variability. However, as some GDEs can be rapidly impacted by relatively small declines in water levels, the Panel considers this time period to be appropriate. Nonetheless, where longer time-series data exists, there is merit in also examining trends over much longer time periods to put the 5-year trends into a broader context and to illustrate the existence of sustained trends (e.g., steady prolonged decline in groundwater level) or otherwise. Cumulative water level declines and salinity increases over long periods of time are as important as rates of decline.

4.1 Information available to 2012

For the 2012 Risk Assessment (see Section 6), trends in water level were calculated based on: (i) a linear regression on data for the 5-year period 2007 – 2012; and (ii) the absolute difference in water level over the 10-year period between March 2002 and March 2012. Salinity trends were calculated over the 5-year period from 2007 to 2012. Over the 10-year period between March 2002 and March 2012, water table declines were recorded in all observation bores within the six key management areas (Frances, Short, Coles, Hynam East, Zone 3A and Zone 5A), although bore coverage was limited in Frances (two bores) and Coles and Zone 5A (three bores each) (Box 4.1). Water table declines averaged greater than 0.1 m/y throughout much of management areas Frances, Coles, Short and Zone 5A, and also in parts of Hynam East and Zone 3A. Rates of decline averaged more than 0.2 m/y throughout much of Coles and parts of Short.

It is unclear why different approaches were used to examine 5-year and 10-year water level trends. The use of absolute water level differences between two dates to determine trends can be problematic if there is significant variability between readings, as the calculated trend can be highly influenced by the precise start and end dates. Linear regression analysis can also be greatly affected by individual readings, although the Panel considers that this is not the case for the observation bores within the LLC PWA. Thus, because the WAP defines permissible rates of water level decline with reference to the 5-year time period, and also because the Panel believes that the method used for the 5-year time period is preferable to that used for the 10-year period, the following discussion focusses on the 2007 – 2012 linear regression analysis. Over this period, significant declines were observed throughout most of Coles, Frances and Zone 5A, and in parts of Short, Zone 3A and Hynam East, although the number of observation bores differs greatly between the management areas, and the rates of decline vary greatly between individual bores (Figure 2).

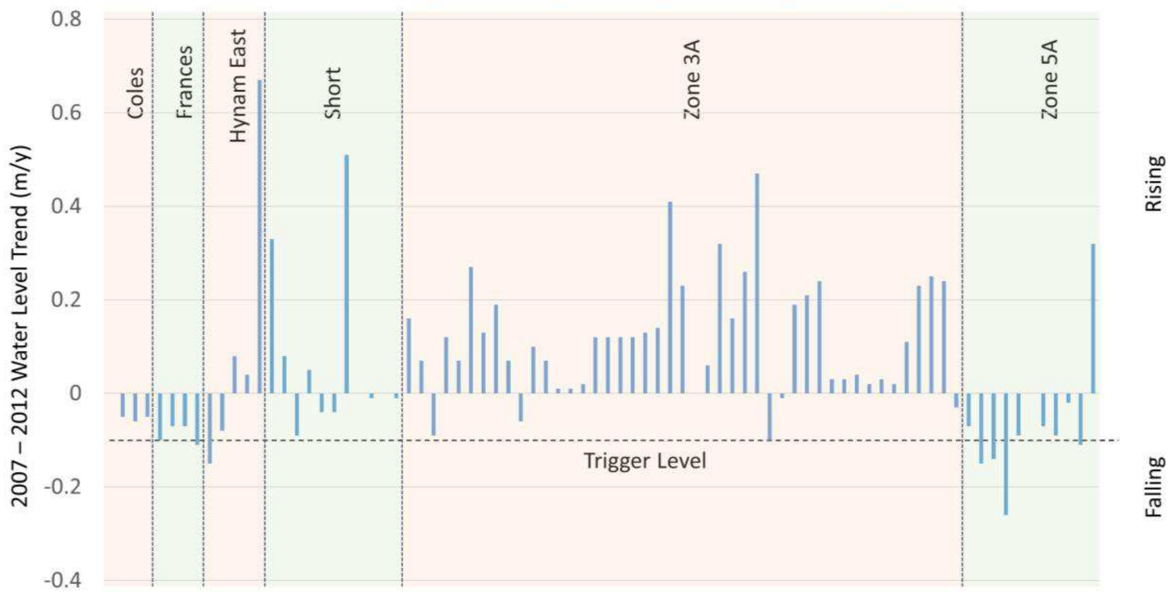

Figure 2. 5-year water level trends for the period 2007 – 2012. Data was used to inform the Risk Assessment, and is from spreadsheets supplied to the Panel. Exceptions are trends for bores CMM057, PEN097 and PEN098, which have been recalculated by the Panel (see Appendix C). Each vertical bar represents an individual bore, and bores are arranged in alphabetical and numerical order within each management area, as per Appendix C. Positive values reflect rising water level trends and negative values reflect declining trends.

The Risk Assessment (see Section 6) assigns water level trends for each management area based on observed trends in suitable bores within and adjacent to the management areas. However, the process for assigning the management area trend is not clearly documented, and is unclear to the Panel (Box 4.2). The Panel has examined the original bore data used to determine 5-year linear regression trends for the six management areas that are the focus of this review, and has recalculated these trends using the same method (see Appendix C). Trends were found to differ for only three of 80 observation wells for which 5-year trends were recalculated. However, when individual bore trends were compared with overall management area trends reported in the Risk Assessment, several discrepancies were identified:

Box 4.1. Was there an appropriate coverage of groundwater monitoring wells in the LLC Prescribed Wells Area, and were they of appropriate depth?

Although varying greatly among the different management areas, the number of observation bores that were available for the 2007 – 2012 water level and salinity trend assessment is considered acceptable for most management areas. However, the coverage is too sparse in some management areas, particularly with respect to salinity observations in Coles and Hynam East. The Panel is also concerned that a large number of water level observation bores (30% within the six key management areas) are no longer monitored. Long term records are essential for groundwater management, and changing the locations of observation bores or discontinuing observations impacts on our ability to understand groundwater trends and hence to manage groundwater systems. Where trends are significantly different between different bores (such as in Hynam East), then a larger number of observation bores may be justified. The Panel also believes that more observation bores in areas that are currently considered to be under stress and/or close to GDEs would be beneficial.

The median depth (relative to the position of the water table) of the bores used for the 2007 – 2012 water level trend analysis in the six key management areas is 6 m, which is considered appropriate for monitoring the position of the water table. The median depth of the bores used for the salinity trend assessment is 15 m, which may be too deep to monitor changes in salinity due to infiltration processes or soil water extraction by vegetation. The Panel notes that salinity increases exceeding 2% per year (the trigger level) were observed in six of 33 bores, and none of these bores were deeper than 10 m (no depth information is available for one of the bores). Increases in salinity exceeding the trigger level occur in almost 25% of bores less than 10 m deep (five of 21 bores).

- In Coles, of the four bores that were used, water levels in three were declining at rates of 0.05 – 0.06 m/y while the water level in the fourth was stable. Yet the management area was assigned an overall water level trend of declining at > 0.1 m/y (Appendix C; Category 2 in the Risk Assessment).
- Frances, Short and Zone 5A were assigned overall water level trends of declining at > 0.1 m/y. In Frances, water level in only one of four bores used for the analysis was declining at greater than 0.1 m/y. If the water level trend observed in all four bores is averaged, this results in a declining trend of 0.09 m/y, still less than the trigger value of 0.1 m/y. In Short, water levels in none of the 11 bores that were used were declining at this rate, and only one of them had water levels declining at a rate exceeding 0.05 m/y. In Zone 5A, water levels in nine of the 11 bores that were used show declining trends, but only four of them were declining at greater than 0.1 m/y. The mean trend of all 11 bores is -0.06 m/y, still less than the trigger value of -0.10 m/y.
- Hynam East and Zone 3A were assigned overall water level trends of declining at $\leq$ 0.1 m/y. In Hynam East, five bores were used for the analysis. Three of these had rising water level trends, one was declining at 0.08 m/y and one was declining at 0.15 m/y. In Zone 3A, of the 45 bores used for the analysis, water levels in only five were declining. Rates of water level decline for these five bores were between 0.01 and 0.10 m/y.

The assigned management area trends for Frances, Zone 3A and Zone 5A could be explained by the application of a precautionary principle, but the rating for Hynam East is not consistent with uniform application of such an approach. Ratings for Coles and Short appear to be in error. The assigned water level trends for these management areas are thus inconsistent with the trends observed in bores within or adjacent to these areas. The implications of this inconsistency for the conclusions from the Risk Assessment for the six management areas are discussed in Section 6. Over the ten-year time period, all management areas were assigned to Category 3 in the Risk Assessment (more than 1 m decline over the 10-year period), and this rate of change was observed in 3/3 observation bores in Coles, 3/4 in Short, 2/2 in Frances, 2/4 in Hynam East, 4/13 in Zone 3A and 3/13 observation bores in Zone 5A (Appendix D).

In terms of salinity, management areas Coles, Short, Hynam East and Zone 5A were all assigned to Salinity Category 1 in the Risk Assessment, which corresponds to salinity increasing at more than 0.5% per year and less than 1% per year (Table 6 in Conesa et al., 2015). These management areas have respectively one out of

Box 4.2. Was observation wells monitoring data that was available at the time used appropriately to understand depth to water table and the nature of groundwater level trends?

Observation monitoring data available at that time was appropriately used to understand water table depth and groundwater level trends in available bores. Both a 10-year absolute change in water table depth and a 5-year linear regression were used as indicators of trends. However, the process by which trends in individual bores were used to determine an overall trend in each management area is not clearly documented and is unclear to the Panel. In Coles and Short, the overall trend assigned for the management area is greater than the trend recorded in any of the observation bores that were used.

one, one out of four, one out of two and one out of eight available salinity monitoring bores where salinity is rising at more than 0.5% per year. Zone 3A was also classified as Category 1. Salinity in six of its 15 bores is rising at more than 0.5%/y, but five of these have salinity rising at > 1%/year. Frances has three salinity monitoring bores and salinity in none of these is rising at more than 0.5% per year, and so the management area was assigned to Salinity Category 0 (corresponding to salinity declining or between 0.01-0.5% increase).

The justification for assigning an entire management area to Salinity Category 1 when salinity in only one bore is rising at the designated rate is not described in the reports, but discussions with DEW staff suggest that it is due to application of a precautionary principle. The Panel notes that salinity trends are difficult to determine from a small number of observation bores, as such data tends to reflect very local conditions. The location and representativeness of salinity monitoring bores are thus more critical than for water level observations. The salinity trends assigned to the management areas have important implications for the results of the risk assessment, and this is also discussed in Section 6.

Earlier, Brown et al. (2006) had carried out an analysis of water level trends within the unconfined aquifer system on 556 bores, and salinity trends on 229 bores. (Fewer bores are monitored for salinity than for water level.) At that time, trends were determined by linear regression over periods of 5 years, 10 years and the full period of record. Overall, water levels in 108 out of 556 bores were declining at > 0.1 m/y over the previous 5 years (the period of assessment appears to be 1999 – 2004 but is not clearly stated). Water level declines in 158 out of 484 bores exceeded this rate over the previous 10 years, with some water level declines up to 0.4 m/y. Salinities in 82 out of 229 bores exceeded a 10 mg/L salinity trigger over 5 years, and 62 out of 175 bores exceeded this salinity trigger over the preceding 10-year period. Based on analysis of this data (particularly the 10-year trend results), Brown et al. (2006) identified a number of 'hot-spot' areas, including the management areas of Frances, Myora and Zones 3A and 5A. Of these, Zone 3A was highlighted as also having total groundwater use significantly greater than total available recharge. Coles, Short and Hynam East were not identified as hot-spots based on groundwater trends at that time.

The observed declines in groundwater level are likely to have resulted principally from the combination of groundwater use for irrigation, plantation forestry and a decrease in rainfall over the previous 20 years. It is unclear whether the decrease in rainfall represents climate change or whether it is a temporary phenomenon. Accurately apportioning the reduction in groundwater levels to different causes is difficult. Although in some areas, there is a correlation between water level trends and changes in rainfall, such correlations can be affected by increased groundwater extraction in low rainfall years. As groundwater metering has only been widespread in the region over the past few years, it is not possible to compare water level trends with changes in groundwater pumping. Despite this uncertainty, the objective of protecting GDEs requires ameliorative action, irrespective of the principal cause of the decline in levels.

4.2 Information available since 2012

An analysis of water level trends from 2012 – 2016 was included in the 2016 Status Report (DEWNR, 2017), and in Cranswick (2018). Cranswick (2018) noted that for the period 2012 – 2016, water levels in 169 out of 226 monitoring bores in Hydrogeological Zones HZ3, HZ5 and HZ7 (which collectively contain the six management areas, Figure 1) were declining. Of those 169 bores, water levels in 69 were declining at rates exceeding 0.1 m/y. Considering only those bores that were used for the 2012 Risk Assessment, only one of the four bores used in the Coles management area had a declining trend in water level (at 0.04 m/y) for the period 2012 – 2016 (one was no longer monitored). Analysis by the Panel of more recent trends revealed that all three of the bores that continue to be monitored have had rising water level trends for the period 2014 – 2018. For Frances and Hynam East management areas, trends in water levels in 2012 – 2016 and 2014 – 2018 were broadly similar to those observed in 2007 – 2012, although in Hynam East the rate of decline in one of the five bores had significantly increased. For Short, all six bores that were used in the Risk Assessment and that are still monitored have rising water level trends for 2014 – 2018. For Zone 3A, water levels in only one of 31 bores that were used for the Risk Assessment and that are still monitored are currently declining. For Zone 5A, water levels in four of the bores that were declining are now rising or stable, and water levels in no bores are currently declining at rates in excess of 0.1 m/y (Figure 3).

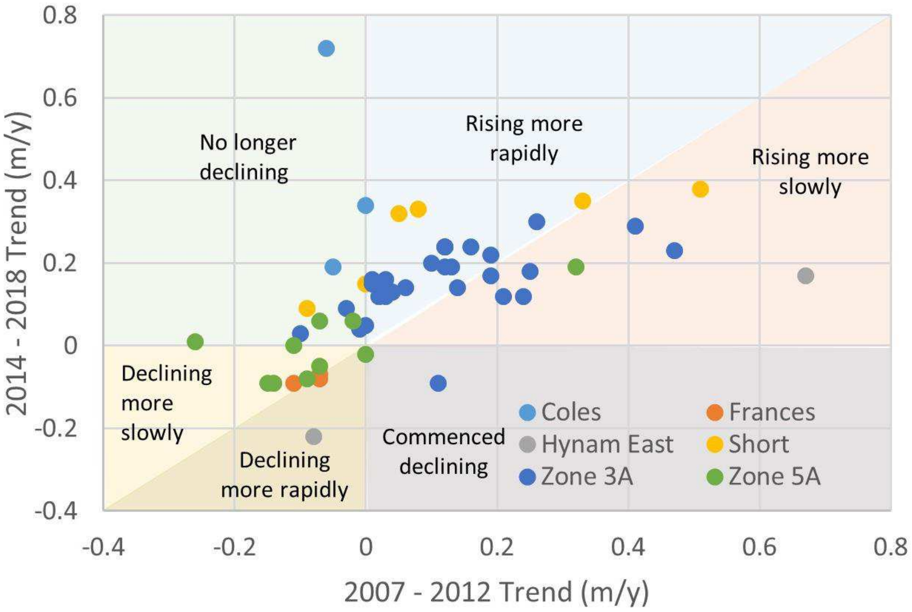

Figure 3. Comparison between 5-year water level trends for different management areas for the periods 2007 – 2012 and 2014 – 2018. Data from 2007 – 2012 was used to inform the Risk Assessment. Exceptions are trends for bores CMM057, PEN097 and PEN098 in 2007 – 2012, which have been recalculated by the Panel (see Appendix C). Positive values reflect rising water level trends and negative values reflect declining trends.

Box 4.3. Considering climate change, was the method of using trigger levels of a mean (arithmetic) decrease (note: the Panel has interpreted this to refer to an 'increase' rather than 'decrease') in depth to water tables of 10 centimetres per year over the preceding 5 and 10 years in the risk assessment, a suitable measure for assessing condition of the resource in view of the objectives of the WAP?

Water level triggers based on a mean decrease in water level (an increase in depth to water table) of 10 cm per year over the preceding 5 and 10 years have been used in the risk assessment. Of course, the rate of water level decline will vary across management zones, depending on proximity to points of groundwater extraction and aquifer properties. The process that is used in the risk assessment to assign water level trends to management areas, based on trends observed in individual bores is unclear.

In areas where groundwater-dependent ecosystems (GDEs) are present, the trigger level value of 0.1 m/y, although somewhat arbitrary, is considered to be reasonable for most but not all GDEs. Over periods of 5 – 10 years, this would constitute groundwater level declines of 0.5 and 1.0 m, respectively. For GDEs, such changes in water table are likely to be important (e.g., the NSW Aquifer Interference Policy (DPI, 2012) deems a cumulative decline in water table should not exceed 0.2 m near a high-value GDE). Although climate change may be partially responsible for the observed declines in groundwater levels, rates of decline of this magnitude are a cause of concern if GDEs are to be protected. Where GDEs are not present or not considered of sufficient value to be protected, then a greater rate of decline might not cause long-term risks to the resource, provided that this rate of decline was not sustained over much longer periods of time. However, water levels declining by more than 2 – 3 metres are likely to cause significant impacts to stock and domestic water users.

Box 4.4. How resilient is the groundwater resource to climate and land uses (such as forestry, irrigation, and drainage)? (This question to be targeted for the Management Areas where reductions are to be implemented.)

In the six management areas where reductions are to be implemented, groundwater levels in the period 2007 – 2012 were declining at 28 out of 80 available observation bores, with rates of decline up to 0.26 m/y. In some areas, increases in salinity were observed and there were also declines in the condition of groundwater-dependent ecosystems (Section 5). Groundwater levels have declined largely through a combination of a reduction in groundwater recharge due to reduced rainfall since the late 1990s, recharge interception and groundwater use by forestry plantations, and groundwater extraction for irrigation. Groundwater levels would be expected to recover if rainfall increases, and if forestry developments and extraction of water for irrigation are reduced. However, if only some of these changes occur, then only partial recovery of groundwater levels would be expected. The timescale for this recovery is difficult to accurately predict but is likely to be on the order of years to decades, depending on the area under consideration. Where salinity levels have increased, the timescale for recovery (reduction) of salinity could be hundreds of years or longer. It is important to recognise that many ecosystems dependent on groundwater might never recover to their former state or condition, even if groundwater levels and salinity return to pre-disturbance states.

Considering all six management areas (Frances, Coles, Short, Hynam East and Zones 3A and 5A), of the 28 bores whose water levels were declining in 2007 – 2012, ten are no longer monitored, water levels in two are declining more rapidly in 2014 – 2018, one is declining at the same rate, five are declining more slowly, and ten are no longer declining (nine are now rising and one is stable; Figure 3 and Appendix C). Of the six bores whose water levels were declining at rates exceeding the trigger level of 0.1 m/y, one is no longer monitored and none of the others have water levels that are still declining at this rate. In management areas Coles, Short, Zone 3A and Zone 5A, all bores that showed declining trends in 2007 – 2012 (and that are still monitored) either have increasing trends in 2014 – 2018, or reductions in the rate of decline. The reasons for the reductions in declining water levels have not yet been systematically investigated.

Recently, Cranswick (2018) undertook a statistical analysis of rates of water level decline in an attempt to separate the changes in rainfall from those caused by human activities (land use change and groundwater pumping). The analysis assumed that fluctuations in the water table could be decomposed into: (i) a direct relationship with rainfall, so that the water level decline would be proportional to the difference between the annual rainfall and the mean rainfall that occurred from 1900 – 1985; and (ii) a linear rate of decline over more recent times due to human activity. The analysis yielded successful results on 26 of 44 bores. Unsuccessful analyses occurred either in areas with very shallow or deep water tables (where responses of the water table to rainfall are muted) or may have resulted from more complex effects of human activity on the water table than the assumed linear trend. Where the analysis was successful, linear water table trends attributed to human activity ranged from -0.22 m/y (declining trend) to +0.02 m/y (rising trend), with a mean declining trend of 0.075 m/y. In all cases, the declining trend was apparent for more than 10 years, and sometimes more than 30 years. However, the study area for this analysis did not encompass the entire LLC PWA, and the analysis was not attempted on any bores from Coles, Short, Frances, Hynam East or Zone 5A. Although the analysis was attempted on five bores in Zone 3A or immediately adjacent to this zone (Box 4.5), it was not successful for any of these bores. Nevertheless, this work showed long-term declining groundwater levels that could not be attributed solely to variations in rainfall.

Since 2012 there have also been a number of developments related to construction of a groundwater model of the LLC PWA (Morgan et al., 2015). Despite these advances, existing models of the LLC region remain inadequate for making confident and reliable predictions about the groundwater resource and impacts of groundwater abstraction at the scale required for groundwater management. There are a number of reasons for this, and it is beyond the scope of the current report to provide a detailed analysis. Nevertheless, it is worth noting that there are idiosyncratic features of the LLC region that make the development of robust and reliable models challenging. For example, the region is very flat and the hydraulic gradients are therefore very small. This creates challenges for groundwater modelling. The large size of the region and the small scale at which water allocation decisions must be made means that a one-size-fits-all regional model will not be

Box 4.5. Was it scientifically robust for the risk assessment to use groundwater observations in neighbouring management areas if the hydrogeological setting is similar? Will groundwater use on the Victorian side of the border diminish the ability of reductions to achieve their objectives (such as improved groundwater level trends, salinity trends, and lateral groundwater flow)?

Groundwater flow does not respect management boundaries, and so it is appropriate to consider both the impact of groundwater extraction for irrigation in adjacent management areas on water level declines, and to examine water level trends in adjacent management areas sharing similar hydrogeological settings in the assessment of water use. Based on typical aquifer transmissivity values of 200 – 2000 m^2/day, and assuming a specific yield of 0.1, areas of groundwater level declines due to groundwater extraction for irrigation and forestry activities could extend over distances of 5 – 20 km within 10 years. It is therefore reasonable to examine impacts over these distances, even if the examination occurs in neighbouring management areas.

For the same reasons, groundwater extraction on the Victorian side of the border may influence groundwater level trends on the South Australian side. However, notwithstanding a possible contribution from groundwater pumping in Victoria, observation of significant and sustained declines in groundwater levels within the LLC PWA may impact GDEs and stock and domestic groundwater users unless ameliorative action is taken. Whether the reductions in allocations will achieve their objectives is currently uncertain, although a reduction in groundwater use will make it more likely that the groundwater level and salinity trends will improve. However, the magnitude of the improvement will depend upon other factors, including climate and groundwater use in neighbouring management areas and on the Victorian side of the border. Groundwater use on the Victorian side of the border would not be expected to impact salinity trends on the South Australian side in the short term, due to the relatively slow rate of groundwater movement (typically less than a few metres per year).

suitable for answering specific questions at a large range of spatial and temporal scales. It is thus envisaged that a regional scale model will ultimately provide a framework that can be used to develop smaller sub-regional models which will provide management tools for assessing likely groundwater level changes due to groundwater extraction, and for deconvolving impacts of irrigated agriculture, plantation forestry and climate change on observed groundwater level trends.

4.3 Key findings and recommendations

In 2007 – 2012, water levels were declining in 28 out of 80 observation bores within the six focal management areas. This included declines in three of four observation bores in Coles; in five of 11 observation bores in Short; in all four observation bores in Frances; in two of five observation bores in Hynam East; in five of 45 observation bores in Zone 3A; and in nine of 11 bores in Zone 5A. Only 29 observation bores had data for the longer 2002 – 2012 period, but water levels were declining in all of these bores over this period. Salinity was increasing in 10 of 33 observation bores. However, a different pattern is evident for some management areas for 2014 – 2018. Across the six management areas, of the 28 bores whose water levels were declining in 2007 – 2012; ten are no longer monitored; water levels in two are declining more rapidly in 2014 – 2018; one is declining at the same rate; five are declining more slowly; and ten are no longer declining (nine are now rising and one is stable). In management areas Coles, Short, Zone 3A and Zone 5A, all bores that showed declining trends in 2007 – 2012 (and that are still monitored) either have increasing trends in 2014 – 2018, or reductions in the rate of decline. The reasons for the changes in trends over time have not been systematically investigated across the LLC PWA, or within the six key management areas. The process for determining overall water level and salinity trends within management areas from individual bore trends also is not described in the Risk Assessment and is unclear to the Panel. There appear to be a number of anomalies, particularly in Coles and Short, which were both assigned overall water level trends of declining at > 0.1 m/y even though no bores within these areas are declining at rates greater than 0.1 m/y.

The Panel has a number of recommendations about addressing gaps in knowledge and data related to water level and salinity trends:

4.1. The Panel recommends that additional resources are made available to maintain and expand the monitoring network. Long-term bore-specific records are essential for effective groundwater management. Changing the locations of observation bores or discontinuing observations severely constrains the ability to understand groundwater trends and hence to manage groundwater systems. The Panel is particularly concerned about the large number of bores that were used to inform the 2012 Risk Assessment, but which are no longer monitored. Twenty-four (30%) of the bores that were used to inform the 2012 Risk Assessment in the six key management areas are no longer monitored. The Panel also believes that more observation bores are required to cover gaps in the network, particularly in areas that are currently considered to be under stress, and adjacent to GDEs.

4.2. The Panel recommends that a detailed analysis should be undertaken of the reasons for the recent trends in rising water levels that are observed in a number of bores. Consideration should also be given to extending the analysis by Cranswick (2018) to the entire observation bore network to identify the effect of variations in rainfall on groundwater levels. This will improve understanding of the reasons for water table declines and how these trends vary across the LLC PWA. Other approaches for deconvolving the effects of climate variability and/or climate change on groundwater levels should also be considered. Where there are clear regional differences in water level trends across a management area, then the causes for these differences should also be investigated.

4.3. The Panel recommends that consideration should be given to developing groundwater models to predict water level changes across groundwater management areas. Groundwater models represent the most reliable means for inferring water table trends where observation bores are not available. They also have value in predictive analysis (i.e., what will future declines be under different groundwater use scenarios?), and in deconvolving impacts of irrigated agriculture, plantation forestry and climatic variation on water levels. We expect that a suite of subregional models will need to be developed at smaller spatial scales than most existing models (e.g., for each of the smaller jurisdictions and management areas examined in this review). This may involve a "parent-daughter" modelling framework where smaller, subregional models are developed within a larger regional modelling context. Finally, and importantly, we encourage a rigorous and comprehensive sensitivity and uncertainty analysis of, amongst other things, model conceptualisation, parameterisation and boundary conditions. This will help to identify what things the model is and is not sensitive to as well as to quantify the uncertainty of model predictions. This will, in turn, provide a structured approach to focus future modelling and measurement studies on reducing uncertainty and risk as a basis for environmental decision making, in areas where – on the basis of risk – this is deemed warranted. This will also inform, and be informed by, the risk assessment approach.

4.4. The Panel recommends that 1D spatial unsaturated zone modelling is conducted to examine expected salinity increases due to historic clearing of vegetation (flushing of salt that had accumulated within the unsaturated zone prior to land clearance) and due to recycling of groundwater associated with irrigation activities and forestry plantations. Salinity trends are difficult to monitor, as bore data tends to reflect very local conditions, and so observation networks need to be accompanied by numerical modelling. This modelling is distinct from, and entirely independent from, the numerical groundwater modelling described above. An increase in the number of salinity monitoring bores is also strongly recommended.

5 Understanding of groundwater-dependent ecosystems

Groundwater-dependent ecosystems (GDEs) are ecosystems that require access to groundwater to meet all or some of their water requirements to maintain the communities of plants and animals, the ecological processes they support, and the ecosystem services they provide (Richardson et al., 2011). These groundwater requirements apply over space and time, and are typically considered in terms of flow, level (or depth to water table), pressure and quality. It is important to consider these requirements *collectively* as components of the 'groundwater regime' (Kath et al., 2018) because there can be impacts on a GDE if, for example, the water quality declines even though the other components remain unchanged. This principle applies to all three broad types of GDEs:

(1) 'Subterranean GDEs', such as aquifers (e.g., karstic, fractured rock and alluvial) and caves occupied by microbes and sometimes animals,

(2) 'Aquatic GDEs', such as ecosystems dependent on the surface expression of groundwater (e.g., base-flow rivers and streams, wetlands, seeps, springs), and,

(3) 'Vegetation GDEs', such as ecosystems dependent on the subsurface presence of groundwater which is often accessed via the capillary fringe (the non-saturated zone above the saturated zone of the water table) when roots penetrate this zone.

Although these three types of GDEs all occur in the LLC PWA, risk assessments have focused on the potential effects of altered groundwater regime on the second type, especially the groundwater-dependent wetlands. This pragmatic decision reflects the abundance and knowledge about this type of GDE in the region. However, it means that risks to other GDEs such as remnant patches of native groundwater-dependent vegetation have potentially been underestimated so far.

5.1 Information available to 2012

The science and data about wetland GDEs in the LLC PWA that were used to inform the risk assessment in 2012 exceeded equivalent knowledge about most of the GDEs elsewhere in Australia (and indeed much of the world) at the time. Much of this information about GDEs in the South East came from work collated or done by a taskforce formed by the South Australian State Government to support its efforts to review the policy options guiding more sustainable water use in a region where groundwater levels were declining (summarised in Brookes et al., 2017).

Regional inventories (e.g., Harding, 2006; Taylor, 2006) had shown that <6% of the original wetland extent in the South East remained, mostly in degraded and fragmented condition. Seventy-seven percent of these remnant wetlands were deemed highly likely to be groundwater-dependent (Brookes et al., 2010), although groundwater-dependence and its spatial and temporal variability had been directly measured for only a handful of South East wetlands (e.g., Howe et al., 2007; Cook et al., 2008).

This paucity of empirical data necessitated application of a wetland vegetation components (WVC) model (Ecological Associates, 2009) to conceptualise how changes in groundwater levels might affect the ecological character of the wetlands. Validated by field surveys of the wetland vegetation of a subset of SE wetlands (Cooling et al., 2010), a refined WVC model was applied to 72 focus wetlands that were considered of high priority, had high biodiversity and had ample data describing their ecological characteristics. Of these 72 wetlands, 69 had been identified as having a high degree of groundwater dependence (SKM, 2009). Losses of WVCs with declining water levels were then assessed to provide estimates of risk to wetland GDEs at various levels of groundwater decline. This approach was the best option in light of the limited data and resources available at the time but inevitably many wetland GDEs could not be assessed, particularly those deemed to have only moderate groundwater dependence and/or relatively low biodiversity that may already have been declining due to altered groundwater regimes. Subsurface and terrestrial GDEs were not included in the assessment (see later).

The WVC model development and subsequent risk assessment also drew on the empirical evidence that many of the remaining wetlands in the LLC PWA had declined in biodiversity and there had been significant shifts in species composition of aquatic plants to those more tolerant of saline conditions (Goodman, 2010). These shifts corresponded with fresher wetter conditions before 2000 and more saline and drier conditions post-2000 (Goodman, 2012).

A final major advance in the science of risk assessment of GDEs in the South East pre-2012 was the application of the Water-dependent Ecosystem Risk Assessment Tool (Water-RAT). This is a GIS tool to provide baseline information to identify significant water-dependent ecosystem assets and processes and incorporate spatial distribution and connectivity issues and associated development threats and risks (Harding, 2012). The output was then used to specify zones of high risk of potential impacts to high-value groundwater-dependent ecosystems (GDEs) from water-affecting activities and developments (discussed in Section 6). One significant limitation of this analysis was the large number of wetlands in the South East in the 'not assessed' category, prompting a further inventory of unassessed wetlands judged from aerial photography to be in good condition.

Although application of Water-RAT generated a classification of groundwater dependency for the mapped wetlands and improved coverage of wetland inventory data to identify high-value wetlands, several data gaps (e.g., aquifer transmissivity, extent of groundwater catchment zones for significant GDEs) could not be filled and limitations of the project scale were identified (Harding, 2012). The implications of these constraints for the risk assessment of GDEs (Box 5.1) are discussed more fully in Section 6.

Box 5.1. Was the method used to determine the risks to the ecological value and vulnerability to changes in depth to water of GDEs robust?

Yes. The method is closely similar to others recommended and used in other Australian states (e.g,. Serov et al., 2012 for New South Wales; IESC, 2019 in eastern Australia). The criteria used to assess ecological value of wetland GDEs in the LLC PWA are logical and, with one minor difference, matched those used by Serov et al. (2012). Assessing vulnerability of different types of GDE to groundwater declines is more challenging because of our current limited understanding about the details of how specific ecological processes and biota in each GDE rely on different aspects of the groundwater regime. However, the modelling approaches used for the vulnerability assessment of wetland GDEs in the LLC PWA are currently the best available in the absence of detailed site-specific data. Where ground-truthed data are available, they have broadly confirmed the conclusions drawn from the modelling although there may be long time lags (multiple years) in detecting some ecological responses in GDEs (e.g., loss of vulnerable species, impaired or lost ecological function) to increasing groundwater depth.

5.2 Information available since 2012

Since 2012, further information has been collected to enhance the assessment of risks to the GDEs in the LLC PWA. There are now longer time-series of data on changes in depths to water table and other hydrogeological information (described in Section 4). Surface water and groundwater monitoring infrastructure was installed at 13 key GDEs identified in the LLC WAP in 2009 but apart from several sites used for individual studies, monitoring and maintenance of most of the network have not been resourced since 2010.

Despite data gaps (usually due to logger failure and battery expiry), report cards have been prepared for each of the 13 GDE sites presenting all data from the nine-year monitoring period (Harding, 2018). The network is suitable for monitoring hydrological conditions at key high-value GDEs but less effective for monitoring highly groundwater-dependent wetlands other than those at major discharge sites at the coast (e.g., karst rising springs). Consequently, the current monitoring network may not be suitable to provide early warning indicators of potential impacts to the most vulnerable GDEs in some hydrogeological zones (HZs) (Harding, 2018), including those in the six management areas of interest in this review (see below).

To partly address these problems of limited spatial and temporal data on wetland GDEs in the South East, nationally available remotely sensed Water Observations from Space (WOfS) dating back to 1987 have been used to cost-effectively re-create historic records of surface water regimes of many more GDEs than could be feasibly monitored manually (Harding et al., 2018). Hydrographs of these water regimes used in conjunction with DEMs of basin bathymetries derived from LiDAR (Deane et al., 2017) enable the assessment of 30-year variations in surface water expression in GDEs where groundwater levels have changed in response to rainfall variability and groundwater extraction and interception.

Results from these post-2012 analyses of GDEs corroborate many of the pre-2012 findings. Widespread declines in groundwater continue to occur across most of the eastern LLC due to a combination of climate variability and licensed groundwater use for irrigation and by plantation forestry (Cranswick, 2018). In the elevated areas of the southern Naracoorte Ranges (HZ3 and HZ7, Figure 1), many wetland GDEs have changed from permanent or seasonal wetlands to now being consistently dry whereas those of adjacent flats (HZ5 and HZ6) partially recovered in recent years with above-average rainfall (Cranswick, 2018; Harding et al., 2018). Thus, the assessment of risks to GDEs (Section 6) is informed by strong evidence that the projected continued decline in groundwater level will increase the risk of reduced seasonality of remaining wetland GDEs and reduced access to groundwater in shallow bores for stock and domestic uses. These impacts are also evident in the reduced flows down riverine GDEs such as Mosquito Creek which enters the Ramsar-listed Bool Lagoon. Permanent groundwater-fed pools in Mosquito Creek provide important habitat refuges for nationally threatened species including the Southern Bell Frog (*Litoria raniformis*) and Yarra Pygmy Perch (*Nannoperca obscura*).

Six management areas are of particular interest in this review. Two of them (Frances and Hynam East) lack wetland GDEs (Appendix D) but field and desktop assessment may indicate that they support subterranean and terrestrial GDEs which may potentially be at risk from changes to groundwater regime and water quality. The other four management areas support 27 to 1122 wetland GDEs (70 to 4680 ha) of which one in Coles (32 ha) and three in Zone 3A (totalling 131 ha) are ranked as high-value GDEs (Appendix D; Harding, 2012). Again, field and desktop assessments are likely to indicate these four other management areas support subterranean and terrestrial GDEs.

Of the six focal management areas in this review, the only one for which there are hydrographic data on changes in water regime in wetland GDEs is 3A (Figure 1). In this management area, hydrographs for six wetlands (Coomooroo Swamp, Glenrise Swamp 1 and 2 in HZ5 and Coinville Swamp, Sawpit Swamp and McKinnon Swamp in HZ7, Figure 1) were derived using WOfS integrated with LIDAR DEMs. All three of the wetlands in HZ5 showed decadal patterns of reductions in mean annual maximum inundation, mean maximum depth and maximum frequency of inundation, especially in 2005-2015 (Table 1). In HZ5, wetlands with seasonal wetting and drying phases typically reached full supply level from 1987-1992, and then most remained dry during 1997-2002, a period of below-average rainfall. Higher rainfall in 2003-2004 helped water levels recover briefly and most wetlands filled to full supply levels in 2004. Over 2005-2010 (the latter half of the Millennium Drought), most wetlands dried again. The resumption of rain in 2009-2010 and the rise in groundwater levels restored patterns of seasonal surface water to 2015 patterns although not all wetlands reached full supply level (Harding et al., 2015). GDE wetland hydro-periods in HZ5 appear to align closely with rainfall variation, along with variations in depth to groundwater (Cranswick, 2018).

Trends in the three GDE wetlands in HZ7 resembled those in the HZ5 wetlands for the first two decadal periods but there was no evidence of recovery of the seasonal hydro-period in the last decade, with one swamp (McKinnon) remaining dry (Table 1). From 1992-2007, permanent and seasonal wetlands in HZ7 became dry, and even during years with above-average rainfall (2004) there was no significant recovery (Harding et al., 2018). This suggests that the groundwater system is no longer sustaining these wetland GDEs with direct discharge (Cranswick, 2018), severely impacting the aquatic biota and processes in these GDEs.

The marked differences in wetland GDE hydro-period responses to changes in groundwater levels in the two hydrogeological zones within 3A make it impossible to generalise about GDE responses in this management area as well as other areas where more than one hydrogeological zone occurs. This wetland-specific behaviour also reiterates the importance of gathering adequate spatial and time-series data on hydrological and ecological responses to altered groundwater regimes in individual wetlands because there appear to be serious constraints to extrapolating responses to this area's wetland GDEs where data are sparse or lacking.

Table 1: Ten-year summaries of mean annual maximum inundation, mean maximum depth and maximum frequency of inundation for six swamps over 30 years (1985-2015) in Zone 3A (Fig 1). Hydrometric data derived from WOfS and empirical relationships between groundwater and surface water expression presented in Table 3.2 in Harding et al. (2018). FSL = full supply level.

		HYDROGEOLOGICAL ZONE HZ5			HYDROGEOLOGICAL ZONE HZ7		
		GLENRISE 1	GLENRISE 2	COOMOOROO	SAWPIT	COINVILLE	MCKINNON
Mean annual max ha inundated (% inundated from FSL)	1985-1995	5.15 (94.6)	11.86 (94.9)	6.91 (99.6)	15.0 (99.9)	6.99 (100)	82.4 (99.6)
	1995-2005	4.94 (90.9)	10.8 (86.4)	5.08 (73.2)	8.5 (56.6)	4.1 (58.4)	69.7 (84.2)
	2005-2015	3.4 (63.1)	4.76 (38.1)	3.47 (50.0)	0.05 (0.4)	0.51 (7.3)	0.57 (0.7)
Mean max depth (m) (% of depth at FSL)	1985-1995	0.69 (62.6)	0.52 (65.8)	0.56 (99.3)	1.31 (92.8)	2.49 (99.5)	2.4 (88.0)
	1995-2005	0.46 (41.7)	0.38 (48.3)	0.28 (50.1)	0.3 (21.3)	0.79 (31.7)	0.88 (32.3)
	2005-2015	0.19 (17.6)	0.15 (19.5)	0.13 (22.4)	0.002 (0.2)	0.09 (3.7)	0 (0)
Max frequency of inundation (% years inundated)	1985-1995	10 in 10 y (100)	9 in 10 y (90)	10 in 10 y (100)	10 in 10 y (100)	10 in 10 y (100)	10 in 10 y (100)
	1995-2005	6 in 10 y (60)	8 in 10 y (80)	9 in 10 y (90)	8 in 10 y (80)	9 in 10 y (90)	8 in 10 y (80)
	2005-2015	3 in 10 y (30)	3 in 10 y (30)	7 in 10 y (70)	1 in 10 y (10)	2 in 10 y (20)	0 in 10 y (0)

5.3 Key findings and recommendations

The science and data about wetland GDEs in the LLC PWA that were used to inform the 2012 Risk Assessment exceeded equivalent knowledge about most of the GDEs elsewhere in Australia at the time. Regional inventories had shown that less than six percent of the original wetland extent in the South East remained, mostly in degraded and fragmented condition, with 77 percent of the remnant wetlands highly likely to be groundwater-dependent. Biodiversity in many of these remnant wetlands had declined and species composition of aquatic plants had shifted to those more tolerant of saline conditions. In the elevated areas of the southern Naracoorte Ranges, many wetland GDEs have changed from permanent or seasonal wetlands to now being consistently dry whereas those of adjacent flats partially recovered in recent years with above-average rainfall.

Declines in groundwater levels and/or increases in salinity should be sufficient justification for ameliorative action, particularly when high-value GDEs are present. The Panel believes that the rate of water level decline that is considered unsustainable over a 5-year window should depend on the presence, value and resilience of GDEs within the area. Larger water level declines might be permitted for short periods of time if GDEs are not present within the immediate vicinity. Where important and sensitive GDEs are present, lower rates of decline may be tolerable as long as lasting damage to the GDEs will not occur. Implementation of such an approach requires that all GDEs have been identified and sufficiently assessed. This does not currently appear to be the case, and subterranean and terrestrial GDEs in the LLC PWA have not been fully surveyed. In management areas with greater depths to groundwater (>5 m), remnant fragments of groundwater-dependent terrestrial native vegetation may represent important refuges for native plants and animals in the largely cleared landscape. Studies that link ecological condition of GDEs to water regime are generally lacking.

The Panel has several recommendations about addressing gaps in knowledge and data to improve the science for the risk assessment or for the 2022 review:

5.1. The Panel strongly supports the recommendations by Harding (2018) to make additions to the current monitoring network in HZ1, HZ3 and HZ5 because reliable monitoring data are fundamental to identifying limits of acceptable changes for specific GDEs relative to local hydrogeological variability and then being able to detect when changes approach or exceed these limits. Sufficient resources should be made available to maintain infrastructure and continue this monitoring.

5.2. The Panel recommends further use of remotely sensed methods (e.g., Harding et al., 2018) to derive longer-term data sets of surface water expression and to continue monitoring GDEs at a coarser scale at more sites (surveillance monitoring) than currently feasible manually. This is necessary given the climatic variability in recent decades and resulting trends in groundwater behaviour (Sections 3 and 4), and the need to deconvolve the potential causes of these trends in groundwater behaviour and ecological responses. Data from this recommendation would supplement those from Recommendation 5.1 in identifying limits of acceptable change to inform policy decisions about future resource condition limits.

5.3. The Panel recommends further hydrogeological data (e.g., aquifer transmissivity, surface-groundwater interactions, recharge characteristics (Section 3), groundwater water quality) are needed in the vicinity of GDEs to better describe local features and characteristics of individual GDEs, especially in high-risk areas. Such data would improve predictions of likely lag times between groundwater changes and ecological responses of specific GDEs and inform risk management strategies (e.g., setting buffer zones, water allocations, Section 6). Groundwater-dependency of many GDEs is temporally variable due to declining groundwater levels caused by rainfall variability and groundwater extraction/interception over the last 30 years (Cranswick and Herpich, 2018). Individual GDEs vary in their susceptibility to threatening processes but this sensitivity is poorly known for most GDEs.

5.4. The Panel recommends collecting relevant ecological data on wetland and other GDEs (especially those deemed of high value and/or at high risk) at appropriate spatial scales over a long time period to provide GDE-specific knowledge on how individual GDEs respond to changes in groundwater regime. This would indicate where 'tipping points' may occur in ecological responses to changes in salinity and/or depths to groundwater, and would improve the capacity to predict responses and assess potential risks of various management strategies. Such ecological sampling should be matched with the collection of hydrogeological data advocated in Recommendation 5.3, and should be timed to coincide with seasonal patterns in vegetation condition, wetland water regime and water temperature. It is also recommended that sampling sites include GDEs in a variety of Hydrological Zones and in management areas deemed at different levels of risk so that a suitably broad spectrum of GDEs is sampled to provide 'reference' and 'impact' sites if needed for specific analysis.

5.5. The Panel recommends development of a monitoring program to collect relevant data (e.g., field measurements, remotely sensed Normalized Difference Vegetation Index; Emelyanova et al., 2017) to assess changes in distribution and groundwater dependency of remnant patches of vegetation GDEs in the higher elevation hydrological zones, especially where surface-expression GDEs are rare or absent. Although the Panel acknowledges the pragmatic focus on monitoring aquatic GDEs, subterranean and vegetation GDEs in the LLC PWA are also likely to be imperilled by changes in groundwater regime. For example, declines in depth to groundwater in areas where surface expression of groundwater is rare or absent may exceed levels of access for groundwater-dependent vegetation, leading to loss of important remnant fragments of native vegetation.

6 Risk assessment

Risk assessment is a systematic process for identifying hazards that have the potential to cause harm (risk identification), analysing and evaluating risks associated with each hazard (risk analysis and evaluation), and determining appropriate ways to either eliminate the hazard or, if the hazard cannot be eliminated, controlling the risk (risk control). In water resource management, risk assessment is crucial to identify, analyse, evaluate and control risks to natural resources, community values and management objectives. Since 2012, risk assessments in water planning and management in South Australia have been structured according to the 'Risk Management Framework for Water Planning and Management (DEWNR, 2012a) and 'Risk Management Policy and Guidelines for Water Allocation Plans' (DEWNR, 2012b).

6.1 Information available to 2012

In 2001, initial WAPs were adopted for the Naracoorte Ranges, Comaum-Caroline and Lacepede-Kongorong PWAs (now amalgamated as the LLC PWA). These initial WAPs included a monitoring regime for depth to the water table and salinity in bores across the management areas. Based on these data, Brown et al. (2006) determined available recharge for each management area and identified a number of "hotspot" areas where depth to water level and/or salinity triggers specified in the 2001 WAPs had been exceeded. Subsequently, the South East Water Science Review (Brookes et al., 2010) proposed that groundwater resources were threatened in three areas: Zone 3A, the Coles and Short management areas, and the area south of Mt Gambier. This prompted the risk assessment in late 2012 (Conesa et al., 2015) to identify the level of risk posed by the potential demand for groundwater and current level of extraction (including recharge interception) to the community and GDEs in each management area. The term 'community' refers to users of groundwater for irrigation, commercial forestry, farm forestry, stock and domestic purposes, public water supply, industry, recreation, aquaculture and Aboriginal cultural purposes, as well as holders of water allocations.

When the 2012 Risk Assessment was being done, the concurrent LLC WAP had already had considerable input from existing licensees, the commercial forestry industry and local government through community and agency consultation, and so this was used in the Risk Assessment. Additional key stakeholders were identified as DEWNR, the SE NRM Board and the South East Aboriginal Focus Group, and there was consultation with the South East Aboriginal Focus Group during the valuation of wetlands for the risk assessment to identify wetlands of cultural significance within the LLC PWA.

The Risk Assessment used a series of look-up tables of groundwater and ecosystem condition and community and ecological value and dependency. Cut-off values separate levels within each category, and simple equations and weights allow the individual category scores to be combined to provide an overall risk rating for each management area. These overall risk ratings underpinned the decision to reduce water allocations in some management areas. The value of the groundwater resources to the community was measured as

Box 6.1. Is a risk assessment a scientifically robust method of water management?

Appropriate risk assessment is a crucial component of the suite of methods used in scientifically robust water management because it allows resource managers, decision-makers and developers to identify whether current or proposed activities are likely to affect assets such as existing groundwater bores and groundwater-dependent ecosystems. Failure to assess risks properly is likely to lead to disastrous economic, social and environmental consequences because livelihoods may be affected by loss of access to, for example, suitable quality of groundwater and there may be irreversible damage or loss of unique ecosystems. Therefore, appropriate risk assessment is a scientifically robust component of water management, along with other components such as setting acceptable levels of change, monitoring changes in indicators such as groundwater level and water quality, and implementing appropriate actions (e.g., modifying water allocations) if there is evidence of impending or current threats to the sustainable use of water resources.

their level of dependency on this resource. Dependency was determined by the current and potential extent of usage (relative to the whole management area) and level of activity of a variety of water-dependent activities (e.g., irrigation, commercial forestry), public water supply and stock and domestic requirements, and cultural value (where known) in each management area. The highest groundwater dependency determined for any category of water use (including cultural purposes) was used as the overall score for groundwater dependency of the community in each management area. Although selecting the highest groundwater dependency for any category of water use in a given management area is conservative (aimed at ensuring that the most vulnerable users are being considered in the risk assessment), there may be situations where this highly dependent use is restricted to only a small part of the management area and the resulting area-wide rating could be too stringent. It would be useful to assess how sensitive is the risk assessment to this potential bias by comparing outputs with, for example, those derived from area-weighted dependencies.

A risk assessment of GDEs in the LLC PWA by Harding (2012) focused on wetland GDEs and relied on the identification of assets, likelihood of impacts and existing threats to assets. Its goal was to identify zones of high risk of potential impacts from water-affecting activities and developments on high-value GDEs and ecological assets. These risk zones were not intended to be used as regulatory buffers or a policy tool (Harding 2012) but to provide an 'early warning' mechanism to alert resource managers, decision-makers and developers about potential risks to a high value GDE as a result of a proposed activity likely to affect groundwater within a certain distance of an asset. This approach to GDE risk assessment matched standard methods recommended for GDEs elsewhere (e.g., Serov et al., 2012 for New South Wales) and was scientifically robust (Box 6.1). However, as described in Section 5, there were constraints on the available field data, necessitating further wetland inventories and the use of conceptual models such as the WVC approach (Cooling et al., 2010). In particular, it was deemed important to assess GDEs in areas with shallow groundwater (< 5m to groundwater) because these systems are likely to be at greatest risk from water-affecting activities in the South East such as irrigation and forestry (Harding 2012).

The ecological value of the wetland GDEs was assessed based on four criteria: (1) landscape naturalness and connectivity; (2) diversity and richness; (3) threatened species and ecosystems; and (4) special features (details in Harding, 2007; 2012). These criteria match three of those used in the GDE risk assessment method proposed by Serov et al. (2012). The exception is the first one which Serov et al. (2012, p. 18) replaces with the condition of the aquifer and surrounding landscape. This is not a major difference and it is highly likely that use of either approach would provide similar assessments of the ecological value of a given GDE (Box 5.1). The assessment of assets, likelihood of impacts and existing threats informed the boundaries of groundwater development risk zones applied to high value water-dependent assets that also had a high likelihood of groundwater dependence. Derivation of these zones in the LLC PWA had some constraints (Harding, 2012) but, at the time, yielded zones considered to be sufficiently conservative and realistic to be used as an alert to a potential significant impact within the scope of the risk assessment tool. However, acceptable limits of change were not determined for GDEs in the South East, and probably vary depending on GDE type and degree of groundwater-dependency.

The 2012 Risk Assessment (Conesa et al., 2015) followed the release of methods and definitions established by the Department of Environment, Water and Natural Resources' Risk Management Framework for Water Planning and Management (DEWNR, 2012a) and the accompanying Risk Management Policy and Guidelines for Water Allocation Plans (DEWNR, 2012b). It was based on the condition of the groundwater resource between 2002 and 2012, the condition of GDEs at the time of the assessment, and the last two years of available groundwater extraction data (from 2010 and 2011). Following standard practice, risk level was based on likelihood and consequence. Likelihood criteria were a function of the extent to which potential demand and, in some cases, current extraction exceeded Total Available Recharge (TAR). Ecological value and dependency and community dependency were each multiplied by an aquifer vulnerability score to derive ecological and community consequence in each management area (Figure 4). For GDEs, consequence was determined by multiplying the highest measure of groundwater dependency by GDEs in the management area (on a 5-point scale) by the vulnerability of the aquifer in that management area.

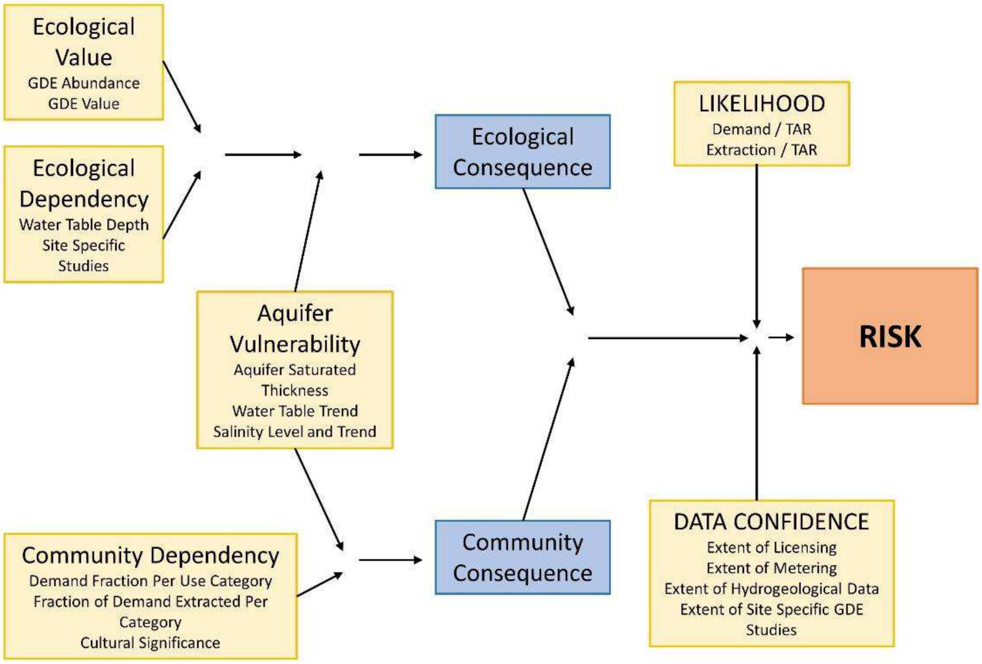

Figure 4. Schematic illustration of the risk assessment methodology used in the LLC WAP.

Groundwater resource vulnerability considered aquifer saturated thickness (the greater the saturated thickness, the less vulnerable the aquifer is considered) as well as evidence that an impact is imminent or already occurring (as determined from historical 5- or 10-year trends in depth to water table and salinity) and the groundwater salinity (which affects the aquifer productive use). Four risk categories (Very high, High, Moderate or Low) were generated separately for the community and GDEs in each management area, based on the ratings of likelihood, consequence and confidence level (Figure 4), and the higher risk was selected as the overall risk for each management area (Conesa et al., 2015). Areas designated as at Very High or High risk were considered to require ameliorative action. This includes the six management areas Coles, Frances, Short, Hynam East, and Zones 3A and 5A that are the focus of the current assessment.

The Panel has several concerns about the GDE risk assessment. Understandably, the focus of the Risk Assessment is very much on wetland GDEs rather than groundwater-dependent terrestrial vegetation or the completely subsurface ecosystems in caves, karsts and aquifers. Although this focus on wetlands is pragmatic, a full risk assessment would also consider whether groundwater drawdown and/or increasing salinities might adversely affect other types of GDEs in the LLC PWA. It is also questionable whether aquifer thickness should form part of the assessment of ecological consequence, because surface-expression GDEs (e.g., groundwater-dependent wetlands and terrestrial vegetation) may lose essential access to the groundwater if the water table falls, irrespective of the thickness of the aquifer system (Box 6.5). The Panel also is concerned that the groundwater consequence score determined for GDEs in each management area is always less than (or occasionally equal to) the community consequence score. Since the higher score is used in the assessment of risk, this results in GDEs having no influence on the final risk rating. Thus, although the LLC PWA contains GDEs of international significance, and one of the objectives of the Water Allocation Plan is to protect these ecosystems, the presence of GDEs does not influence the risk assessment ratings.

There is also a lack of clarity surrounding the reasons for the cut-off values which determine the boundaries between the different levels. In assessing the method, the Panel identified that, in some cases, the ultimate classification is sensitive to these boundary values. Although this is often the case in risk assessments, as there is not a lot of empirical data to establish evidence-based cut-off values, this issue becomes pronounced where only a small number of levels is defined (Box 6.4). For example, the risk assessment does not discriminate between areas where declines in water table only marginally exceed the trigger level of 0.1 m/y from areas where declines greatly exceed this value. A greater number of levels that include categories that

Box 6.2. What methodology should be used to determine whether there is sufficient data to assess the risk to the resource? In the situation where there is insufficient data to characterise the risk, what additional data is needed?

Determining whether there is sufficient data to assess the risk to a resource depends on whether the risk is expected to be high or low (in turn, a function of likelihood and consequence of one or more impacts) and the degree of uncertainty about the assessment. In situations where risk and uncertainty are both low, fewer data will be needed than when risk, uncertainty or both are high. The sufficiency of data is also determined by the degree to which risks need to be specified, and this varies on a case-by-case basis. For example, where risks are expected to be low, the risk assessment may only require a qualitative analysis of risks to the resource, and conceptual models or extrapolation to similar situations may be sufficient. On the other hand, if specific risks require quantification (as might occur in the case of an asset being considered of high value and at high risk), then field-collected data may be needed on temporal and spatial trends in the amount or quality of the resource and the drivers (e.g., declining water levels, increasing salinity) that threaten it. In the current situation, there are ample data to indicate material risks to the groundwater resource and GDEs. However, more work is needed to agree on which risks are deemed unacceptable and how to set resource condition limits, for example, to avoid or reduce unacceptable risks. The answer to the second part of this question is problem-specific, and depends on how severe the risks are expected to be and how feasible it is to collect relevant data to inform the risk assessment in a given situation. Where there is insufficient data to characterise the risk, the amount and type of additional data required depend on whether the risk assessment is qualitative or quantitative, and if the latter, what specific data gaps remain that prevent adequate assessment of risk to guide effective avoidance or mitigation strategies.

span a narrow range just below and just above the trigger level might result in some management areas being moved to higher or lower risk levels, and provide more reliable input for risk assessment and resource management.

The relative weights ascribed to the different factors are also not clearly justified and sometimes appear unreasonably arbitrary. This can create anomalies in the resulting risk assignments. For example, the ecological value of GDEs within a given management area appears to have little impact on the final risk rating, even though protection of GDEs is one of the main criteria for maintaining groundwater levels. GDE risk rankings in the LLC WAP (Table 21 in Conesa et al., 2015) always come out as Low or Moderate, and are never higher than the Community risk ranking. This may indicate that the weights in the risk assessment are inappropriate. Further, the risk assessment assigns a low likelihood of impact whenever extraction is less than 50% of the estimated TAR (Conesa et al., 2015, p. 33), and this means that the overall risk can never be rated as High or Very High, irrespective of the levels of the other factors. Given the uncertainty associated with estimates of TAR (Section 3), the Panel believes that the weighting applied to the Extraction/TAR ratio is too high.

Scenario testing of the assessment by the Panel reveals other anomalies. For example, suppose that: (i) a management area contains large numbers of high-value GDEs but with unknown cultural significance; (ii) groundwater levels have been declining at greater than 0.2 m/y for more than 10 years; (iii) groundwater allocation and use are assessed to both be 90 – 95% of TAR; and (iv) this information is known with a high level of confidence. Even though there appears to be material risk, application of the current risk assessment process could still identify the risk as Low, particularly if groundwater demand is split between several different categories of users.

Although GDE abundance and value are included in the risk assessment, the permissible rates of water table decline that are defined in the WAP do not change in regions that do not have any GDEs. In the absence of GDEs, it is more difficult to justify restrictions on water table drawdown, although it may be important for protection of stock and domestic bores. However, the existing GDE assessment is not exhaustive, and has not considered groundwater-dependent vegetation (see Section 5). Thus it is possible that management areas without identified GDEs may actually support some as yet unidentified GDEs.

Box 6.3. Does the risk assessment adequately consider the natural groundwater dynamics beneath areas of water use activities?

The risk assessment does not attempt to differentiate between the different possible causes of water level decline (e.g., forest activities, irrigation, climate change). It also does not consider the spatial distribution of these activities in relation to observation bores, other than through the use of groundwater management areas. Of course, the rate of water level decline and salinity change will vary across management zones, depending on proximity to areas of groundwater extraction, and variation in aquifer properties. Where large regional differences in water level or salinity trends are apparent across management areas, then average trends are not especially informative, and possible reasons for differences in trends should be considered. However, it should not necessarily be assumed that water use activities close to an observation bore are most responsible for observed trends in that bore and other possible causes, including local natural groundwater dynamics, should be considered.

The most reliable approach for unravelling some of these causes and their interactions would be through an appropriate groundwater model, although the available groundwater models are currently not considered to be sufficiently reliable for such an analysis. In some areas, increasing the density of observation bores would help assess and understand the link between water use activities and trends in water levels and salinity.

As discussed in Section 4, it is unclear how the water level trends observed in individual bores were combined to determine overall trends for the different management areas, although it appears to have been based on a precautionary principle. In terms of the management areas of particular interest to this review, notwithstanding application of a precautionary principle, Coles and Short appear to have been incorrectly classified in terms of 5-year water level trends, although the Panel notes that this would not have affected their overall risk rating. The overall risk rating of management area Frances would be reduced from High to Moderate if the 5-year water level trend was changed from Category 2 (declining > 0.1 m/y) to Category 1 (declining ≤ 0.1 m/y) (Frances has only one of four monitoring bores declining at greater than 0.1 m/y.) In terms of salinity trends Zone 5A was assigned to Category 1 (salinity rising at more than 0.5% per year and less than 1% per year), but only one of the eight bores that were examined was rising at this rate, and none was rising more rapidly. The overall risk rating would decrease from High to Moderate if a lower salinity rating was assigned to this management area. Although the Panel supports the application of a precautionary principle, caution should be exercised in assessing overall management area trends from results from only a single bore. Additionally, the overall risk rating of management area Zone 5A would decrease from High to Moderate if the data confidence for aquifer thickness had been classified as High rather than Medium. The Panel believes that the classification of aquifer thickness confidence is in error, as the Risk Assessment considers only the number of observation bores that have aquifer thickness information. In reality, many

Box 6.4. Were the likelihood and consequence factors / criteria used in the risk assessment method suitable to determine the overall risks (to community and environmental values) of allocating volumes greater than TAR?

The likelihood and consequence factors used in the risk assessment method are suitable for determining the overall risks of groundwater extraction to the community and to the environment. Allocating volumes greater than TAR equates to mining the resource and will (and has been shown to) result in declining groundwater levels. For highly groundwater-dependent GDEs that have a high likelihood of being deprived of groundwater because of declining groundwater levels, the criteria used to measure the consequences and likelihood are suitable to demonstrate the potential for and outcomes of groundwater declines (e.g., conversion of permanent or seasonal wetlands to ones that are consistently dry as seen in the elevated areas of the southern Naracoorte Ranges (HZ3 and HZ7)). Stock and domestic users will also be impacted by water table declines. Allocation less than TAR can also lead to significant impacts on the environment and on some groundwater users. An increase in the number of levels of some of the factors, changes to the cut-off values between the levels, and changes to the relative weighting of the factors could be considered in future risk assessments.

Box 6.5. Was aquifer thickness data appropriately considered in the risk assessment?

The risk assessment used aquifer thickness data as part of an assessment of aquifer vulnerability. Aquifer vulnerability is used to inform both Ecological Consequence and Community Consequence of groundwater extraction in the risk assessment by Conesa et al. (2015). Five different values of aquifer thickness were used (between < 25 m and > 150 m), and greater values resulted a lower vulnerability rating, and hence lower overall risk. As with all classification systems, there is an element of arbitrariness concerning the cut-off values between the different aquifer thickness levels. However, the use of aquifer thickness to inform aquifer vulnerability for Community groundwater users (e.g., stock and domestic bores) is reasonable, as increased aquifer thickness will provide increased resilience of the resource to climate variations. However, aquifer thickness will not necessarily protect surface-expression GDEs, as these are most likely influenced by water level. Surface-expression GDEs such as wetland and groundwater-dependent terrestrial vegetation are unlikely to be able to access deeper groundwater resources if the water table declines rapidly or below either the beds of wetlands or the root zones of terrestrial vegetation.

other bores are available whose information can confirm aquifer thickness data obtained from observation bores (see Box 6.5). Other changes to the risk assessment to alter the weighting of different factors (e.g., to increase the weighting applied to declining water level trends) and vary cut-off values between factor levels could also affect the weightings.

6.2 Information available since 2012

Since 2012, there is now more temporal data on groundwater behaviour and GDEs to inform risk assessment of GDEs in the LLC PWA, including remotely sensed data for deriving surface-expressed water regimes in GDE wetlands (Section 5). This is especially true for the wetland GDEs in HZ7 where seasonal and permanent groundwater-dependent wetlands do not appear to have recovered after the Millennium Drought broke (Section 5). As GDE wetlands in HZ3 shared similar post-drought responses to those in HZ7, it would be expected that those in other management areas lying in HZ3 (e.g., Hynam East, Frances, 5A) would also have experienced similar substantial reductions in inundation frequency as well as loss or reduction of aquatic biota. Assessing ecological recovery in GDEs as a result of the reduced water allocations is challenged by the potentially long time (years to perhaps decades) that might be needed before recovery of impacted GDEs is evident. It is also possible that some species or ecological processes in some GDEs may be irreversibly lost or degraded.

Box 6.6. Was the application of data confidence ratings in the risk assessment robust?

Data confidence ratings were used to inform the level of risk associated with different Likelihood – Consequence combinations, so that a lower confidence led to a higher level of risk. This is a conservative approach, and is appropriate. The factors considered in determining the data confidence rating (extent of licensing and metering, extent of hydrogeologic data and extent of site-specific GDE studies) were also appropriate.

However, the Panel believes that the approach for determining the confidence rating assigned to aquifer thickness is incorrect, as it only considers the number of water level observation bores within each zone. In reality, aquifer thickness can be estimated from other bores, and so the confidence rating is too low as it fails to consider all of the data that can inform aquifer thickness.

The data confidence rating is important to the level of risk assigned to each of the management areas Hynam East, Coles, Short, Frances, and Zones 3A and 5A. Each of these areas currently has a confidence rating of MODERATE, and the overall level of risk would be reduced if the confidence rating was HIGH. In particular, Zone 5A has a MODERATE confidence rating for aquifer thickness, and if this were changed to HIGH, then the overall level of risk would decrease from HIGH to MODERATE.

There has also been an assessment of the vulnerability of stock and domestic (S&D) bores to drawdown, based on the depth of water in the bore, and this was mapped across region (Cranswick et al., 2018). The analysis used data from 11,800 S&D bores (approximately 37% of all S&D bores; the remainder did not have sufficient data available). In Hydrogeological Zones HZ3, HZ5 and HZ7 (which collectively contain the six management areas that are the focus of the current review), 23% of S&D bores have a water column depth of less than 3 m, and 45% have water column depths of less than 5 m. If this analysis is representative of the region, then it implies that a sustained water table decline of 0.1 m/y will cause 23% of all S&D bores within the LLC WAP region to be dry within 30 years, and 45% of all S&D bores to be dry within 50 years. The analysis is relevant to determining acceptable rates of water table decline.

There is also more groundwater level and salinity data available. Re-examination of the water level trends by the Panel for the period 2014 – 2018 shows that a number of bores that had declining water level trends in 2007 – 2012 now have rising trends (Section 4.2). Reasons for this have not been examined.

6.3 Key findings and recommendations

Quantitative risk assessment is always challenging, and involves the establishment of cut-off values within categories, and weighting schemes between different categories to determine overall risk levels. It is not unexpected that some issues have been identified. The Panel supports the use of a quantitative risk assessment approach, but has identified several anomalies in the 2012 Risk Assessment. In particular, the reasons for the cut-off values between the different risk levels are unclear and, in some cases, this materially affects the classification of risk. The relative weights ascribed to the different factors are also not clearly justified and sometimes appear arbitrary, potentially creating further anomalies in the resulting risk assignments. For example, the groundwater consequence score determined for GDEs in each management area is always less than (or occasionally equal to) the community consequence score. Since only the higher of these two scores is ultimately used in the assessment of risk, this results in GDEs having no influence on the final risk rating. It is also inappropriate to use aquifer thickness as part of the assessment of ecological

Box 6.7. Following a review of the spatial distribution of declining groundwater levels, rising groundwater salinity, locations of GDEs and other users versus the distribution of groundwater extraction and allocation, in how many Management Areas would a blanket adjustment to water allocations (i.e. % across Management Area) be expected to achieve the objectives of the Water Allocation Plan? Did the outcomes of the 2012 Risk Assessment and WAP reflect this?

As discussed in Box 4.5, groundwater extraction for irrigation or industry can affect water levels over distances of 5 – 20 km within 10 years. Groundwater level declines will be greatest close to extraction bores, but the area of impact can still be large. Thus, while regulating (or disallowing) extraction from close to important GDEs can be a valid management approach (see Chapter 2), it can only be successful when used in combination with regional scale approaches. As a general rule, regulating groundwater extraction close to GDEs and other users will reduce impacts in the short-term, but may not prevent longer term impacts. The size of many of the management areas in the LLC PWA is (in order-of-magnitude) similar to the area of influence of groundwater extraction over the 10-year time period. Thus, since the impacts of groundwater extraction may extend across the management area within this relatively short period of time, it makes sense to manage groundwater allocations on the management area scale, while also acknowledging the potential long-term effects at a regional scale.

Whether the proposed uniform reductions in allocations across the management areas will achieve the objectives of the Water Allocation Plan cannot be determined without a suitably calibrated and reliable groundwater model. Such a model is not currently available, and its development is not straightforward. A uniform reduction in groundwater use will reduce the rate of groundwater decline and reduce the impact on GDEs and other users, although the magnitude of the improvement will depend upon many factors. It is possible, perhaps even likely, that targeted reductions in allocation (and use) would cause a greater level of improvement (in water levels and GDE condition), but a numerical model would be required to determine the optimal pattern of use that would maximise economic returns yet best protect the environment and the region's water resources into the future.

consequence (as is currently the case), because surface-expression GDEs (e.g., groundwater-dependent wetlands and terrestrial vegetation) may lose essential access to the groundwater if the water table falls, irrespective of the thickness of the aquifer system. Data confidence categories for aquifer thickness also appear to have been incorrectly assigned. Perhaps most importantly, it is unclear how the water level and salinity trends observed in individual bores were combined to determine overall trends for the different management areas; in some cases (e.g., Coles and Short), the overall rate of water level decline assigned to the management area is greater than the trend in any observation bore within that management area. These multiple anomalies lead to specific concerns with the high-risk ratings determined for many of the management areas, and generally undermine confidence in the risk assessment process.

The Panel has several recommendations about addressing gaps in knowledge and data to improve the science for future GDE risk assessments:

6.1. The Panel recommends future risk assessments should consider using a greater number of levels of the different factors, better justify the cut-off values between the levels, and examine the influence on the final risk assignment of the weightings applied to the different factors. A sensitivity analysis should be conducted on the risk assessment process, to ensure that the outcomes are robust, and are not unduly influenced by small changes in the input data. Consideration should also be given to removing aquifer thickness as a factor for GDE risk, and including an additional factor related to depths of stock and domestic bores (Cranswick et al., 2018). All decisions taken in implementing the risk assessment (including how management area water level and salinity trends are obtained from individual observation bore trends) should be fully documented.

6.2. Although the risk assessment has successfully identified the key risks, the Panel recommends more assessment of the degree to which the risks are deemed unacceptable, ensuring full transparency and consultation with water users to establish specific and measurable resource condition limits (RCLs). A series of RCLs relevant to each hydrogeological zone and explicitly related to the conditions that pose significant risks to GDEs (and the groundwater resource) could include agreed limits to environmental water requirements, including water quality, for selected GDEs and agreed limits to the groundwater level decline in areas where access to groundwater is limited. Associated with each RCL would be one or more indicators that could be monitored so that appropriate management can be triggered (by resource condition triggers) before RCLs are exceeded. This approach is outlined in Cranswick et al. (2018) and endorsed by the Panel.

7 Conclusions and recommendations

A substantial body of scientific work underpins water allocation planning in the LLC PWA, supported by significant data and information regarding water level trends in the six management areas we reviewed. This water level trend data constitutes a primary data source for understanding the groundwater system response – it measures the primary and direct "effect" – and its importance cannot be understated. However, the number of available observation bores differs greatly between the different management areas, and there is a need for additional observation bores in some regions. Furthermore, the way in which water level trends obtained from observation bores have been interpreted in the Risk Assessment is problematic and unclear. The Risk Assessment, as the final part of the technical process, has limitations that are opportunities for future improvement.

Understanding both causes and effects and the relationships between them (causality) is critically important. Some effort has already been made to deconvolve effects of groundwater abstraction and climate change and early results suggest that declining water level trends are mostly related to groundwater abstraction. Further understanding the various and relative causes remains an important line of inquiry for future scientific investigations.

Current water allocation approaches are based on a percentage allocation of recharge. This approach is applied uniformly across subregions irrespective of critical factors such as the presence, or otherwise, of high-value groundwater-dependent ecosystems. The principle of managing to recharge has been widely criticised in the scientific literature. Firstly, the recharge rates are inherently uncertain. Secondly, the percentage of recharge allocated to abstraction is arbitrary. Thus, in essence, we have two numbers that are multiplied together – both either uncertain or arbitrary – that form the basis for the water allocation volume. The final result is therefore one which contains compounded uncertainty (or arbitrariness). Although volumetric allocation limits based on recharge are useful for management, due to the uncertainties, the ratio of use or allocation to TAR is not, in itself, an accurate indicator of unsustainable groundwater use.

In principle, groundwater models provide a useful tool for groundwater management that can offset some of the shortcomings of management based on recharge volumes and with sparse observation bores networks. However, the groundwater models of the LLC region developed to date remain inadequate for making confident and reliable predictions about the groundwater resource and impacts of groundwater abstraction. This is not intended to be a criticism of previous modelling efforts but is rather an acknowledgements of the difficulty of the task. An appropriate groundwater model could, however, address many of the shortcomings of the current approach. A one-size-fits-all approach – a groundwater model that is suitable for and can be used in all circumstances to answer all questions – is not appropriate and is unlikely to be fit for purpose. We expect that a suite of subregional models will need to be developed at smaller spatial scales than most existing models (e.g., for each of the smaller jurisdictions and management areas examined in this review or perhaps some combination of priority (high-risk) regions that are adjacent to each other and which might be grouped together in a single subregional model). Such models will need to be fit for purpose and constructed to answer specific questions at appropriate spatial and temporal scales. These will need to be appropriately parameterised to address model non-uniqueness, and should include rigorous and comprehensive sensitivity and uncertainty analyses that consider model conceptualisation, parameters and predictions.

A clear identification of assets (aquifers and groundwater-dependent ecosystems) remains profoundly important. This includes describing, mapping and understanding subterranean and vegetation GDEs in the LLC region. In order to predict important potential impacts on a GDE, understanding hydro-ecological responses (or causes and effects) remains key. This will also inform critical aspects of conceptualisation and parameterisation of requisite groundwater models that underpin impact assessments. All of these studies will, in turn, provide a structured approach to focus future modelling and measurement studies on reducing uncertainty and risk as a basis for environmental decision making, in areas where – on the basis of risk – this is deemed warranted. They will also inform, and be informed by, the risk assessment approach.

To date, most attention has focussed on groundwater level trends, but it should be recognised that, within the six management areas that are the focus of this review, groundwater salinity is rising at greater than

0.5%/y in 10 of the 33 observation bores. The increases in salinity could be due to historic clearing of vegetation (flushing of salt that had accumulated within the unsaturated zone prior to land clearance), or to recycling of groundwater associated with irrigation activities and forestry plantations. The potential for these processes to increase groundwater salinity and the magnitude and rate of this potential increase has not been examined, yet could become a significant limitation to groundwater use within parts of the LLC PWA. A groundwater model cannot directly predict the potential for salinity increases due to these processes, and additional, one-dimensional unsaturated zone models are required.

In summary, the Panel is impressed with the large body of scientific investigations conducted in the groundwater resources of the LLC PWA. However, to improve understanding of the groundwater resource and groundwater-dependent ecosystems, the Panel has made numerous recommendations that are highlighted in each chapter of this report. No attempt has been made to prioritise each recommendation and for brevity, they are not repeated here. However, the Panel draws attention to five main high-level recommendations:

1. That the coverage of the existing monitoring network is maintained and expanded, and reasons for changes in water level trends over time be more systematically examined than has been the case to-date. The salinity monitoring network should be expanded.
2. That groundwater modelling is updated to include a suite of subregional models that answer specific questions at an appropriate range of spatial and temporal scales.
3. That 1-D unsaturated zone modelling is conducted to examine expected salinity increases due to historic clearing of vegetation and due to recycling of groundwater associated with irrigation activities and forestry plantations.
4. That monitoring activities associated with wetland GDEs are expanded (including expanded use of satellite data), and that reconnaissance assessments of subterranean and vegetation GDEs be carried out as a matter of urgency. Studies that specifically link ecological condition of GDEs to water regime are also needed.
5. That future risk assessments use a greater number of levels of the different factors; better justify the cut-off values between the levels; and examine the influence on the final risk assignment of the weightings applied to the different factors. This would improve confidence in results of the risk assessment process.

The Panel's recommendations for the six management areas that are the focus of this review focus on observed water level and salinity trends and abundance and value of GDEs. These are summarised below.

Coles:

Widespread declines in water levels were observed in 2002–2012 (10-year trend; 3/3 bores) and in 2007–2012 (5-year trend; 3/4 bores) although there has been significant recovery since that time (0/3 bores have declining levels in 2014–2018). This management area contains a single salinity observation bore, and this shows rising salinity. Coles supports 366 identified wetland GDEs, covering 1352 ha, with one wetland that is ranked as a high-value GDE. The reasons for the apparent recovery in water levels within the past few years should be investigated to determine whether it is likely to be sustained. If water level recovery is not sustained, then the declining water levels will negatively impact GDEs and stock and domestic users. The number of water level and salinity observation bores within this management area should be increased to improve confidence in future water level and water quality assessments.

Short:

Widespread declines in water levels were observed in 2002–2012 (4/4 bores) and in 2007–2012 (5/11 bores), but with significant recovery since then (0/6 bores have declining levels in 2014–2018). One of four salinity observation bores shows rising salinity. Short supports 328 wetland GDEs, covering 580 ha. Reasons underlying the apparent recovery in water levels within the past few years should be investigated to assess whether it is likely to be sustained. This is especially important in view of the large number of GDEs in this management area.

Frances:

This management area displays widespread and continuing declines in water levels (2/2 bores in 2002–2012, 4/4 bores in 2007–2012 and 3/3 bores in 2014–2018). Although no wetland GDEs have been identified within this management area, other GDEs are likely. Declining water levels may impact irrigators and stock and domestic groundwater users as well as unidentified GDEs.

Hynam East:

Significant declines in water levels were observed in 2002–2012 (4/4 bores) and in 2007–2012 (2/5 bores), with some recovery since then (1/3 bores has a declining level in 2014–2018). One of two salinity observation bores has rising salinity. In view of the contrasting water level and salinity trends (3/5 bores display rising water level trends in 2007–2012 as do 2/3 bores in 2014–2018), the Panel believes that the number of observation bores is insufficient. It is recommended that reasons for the divergent water level trends are examined, and additional water level and salinity monitoring bores are installed.

Zone 3A:

While all available observation bores showed declining water level trends in 2002–2012 (13/13 bores), only 3 of 45 bores displayed declines in water levels in 2007–2012, with some showing more recent recovery (1/31 bores showed a declining trend in 2014–2018). Six of 15 salinity observation bores show rising salinity. Of the six focal management areas, Zone 3A supports the most wetland GDEs (1122 covering 4680 ha), with three (totalling 131 ha) rated as high-value GDEs. The Panel recommends that the reasons for recovery in water levels and the rising salinity be investigated. Threats posed by the declining water levels and rising salinity to the high-value GDEs should be specifically examined.

Zone 5A:

Nine of 11 bores displayed declining water level trends in 2007–2012, and water levels in half of these continue to decline (5/10 bores have declining trends in 2014–2018). Water levels in all three available observation bores also declined in 2002–2012. One of eight salinity monitoring bores has rising salinity. Zone 5A supports 27 identified wetland GDEs, covering 70 ha. The widespread and sustained water level declines within this management area pose risks for GDEs, irrigators and stock and domestic groundwater users.

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Appendix A – Additional material consulted

Reports

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Computer files

The following spreadsheets provided information on the 2012 Risk Assessment:

Use of DS for Risk Asst.xlsx (Provided to Review Panel on 10/10/2018)

VALUE TABLE community resource 16 Aug 2012.xlsx (Provided to Review Panel on 10/10/2018)

LLC Risk Assessment 24 Aug 2012 v12 reviewed Mar 2017.xlsx (Provided to Review Panel on 10/10/2018)

The following presentation provided information on analysis of climate trends:

Selected_rejected_linearregression_analysis_inprep_for PGC_20181019.pptx (Provided to Review Panel on 19/10/2018)

The following presentation provided information on aquifer properties:

Tvalues_Cranswick_unpublished_forPGC_20181019.pptx (Provided to Review Panel on 19/10/2018)

The following files provided information on groundwater level trend analysis:

5YrTrend_ WL_Trends_Map.xlsx (Provided to Review Panel on 23/10/2018)

Unconfined_Salinity_Trends_Map.xlsx (Provided to Review Panel on 23/10/2018)

LLC_WL_Status_2016.xlsx (Provided to Review Panel on 25/10/2018)

Hydrographs_forPGC_ColesShort.xlsx (Provided to Review Panel on 24/10/2018)

Hydrographs_forPGC_HynamEast_West.xlsx (Provided to Review Panel on 24/10/2018)

Hydrographs_forPGC_Location.xlsx (Provided to Review Panel on 24/10/2018)

Hydrographs_forPGC_Zone2A.xlsx (Provided to Review Panel on 24/10/2018)

Hydrographs_forPGC_Myora.xlsx (Provided to Review Panel on 24/10/2018)

Hydrographs_forPGC_Frances.xlsx (Provided to Review Panel on 24/10/2018)

Hydrographs_forPGC_Zone3A.xlsx (Provided to Review Panel on 24/10/2018)

Hydrographs_forPGC_Zone5A.xlsx (Provided to Review Panel on 24/10/2018)

Myora.JPG (Provided to Review Panel on 24/10/2018)

Zone2A.JPG (Provided to Review Panel on 24/10/2018)

Zone3A.JPG (Provided to Review Panel on 24/10/2018)

Coles_Short.JPG (Provided to Review Panel on 24/10/2018)

Zone5A_Coonawarra_Zoom.JPG (Provided to Review Panel on 24/10/2018)

Zone5A_Hynam_Frances.JPG (Provided to Review Panel on 24/10/2018)

Hydrograph_Locations.dbf (Provided to Review Panel on 24/10/2018)

Hydrograph_Locations.sbn (Provided to Review Panel on 24/10/2018)

Hydrograph_Locations.sbx (Provided to Review Panel on 24/10/2018)

Hydrograph_Locations.shp (Provided to Review Panel on 24/10/2018)

Hydrograph_Locations.shx (Provided to Review Panel on 24/10/2018)

Hydrograph_Locations.prj (Provided to Review Panel on 24/10/2018)

Appendix B – Individuals consulted

Stakeholders

Zone 5A, Hynam East, Clover Growers:

Nick Hillier

Andrew Shepherd

Tim Koch

Brett Dolling

John Young

Coral Young

Scott Longbottom

Tim Schultz

Mark Chester

Vignerons, SADA, Potatoes, Grape Wine Council:

James Freckleton

Stuart Sharman

Andrew Widdison

Allen Jenkins

Pete Balnaves

Graeme Hamilton

Terry Buckley

Australian Forest Products, Green Triangle Regional Plantation Committee:

Glenn Rivers

Darren Shelden

Rob Cains

Leon Rademeyer

Baden Meyers

Jim O'Hehir

Laurie Hein

Conservation Council SA:

Lachlan Farrington

Consultants

Glenn Harrington – Innovative Groundwater Solutions

Nikki Harrington – Innovative Groundwater Solutions

SE NRM Board

Fiona Rasheed – South East Natural Resources Management Board – Presiding Member

Kerry de Garis - South East Natural Resources Management Board - Member

DEW/Natural Resources South East

Abigail Goodman – Bush Management Advisor

George McKenzie – Technical Leader, Water Science and Monitoring

Claire Harding – Aquatic Ecologist

Jeff Lawson - Hydrogeologist

Roger Cranswick – Senior Hydrogeologist

Saad Mustafa – Senior Hydrogeologist

Cameron Wood – Senior Hydrogeologist

Steve Barnett – Principle Hydrogeologist

Neil Power – Director, Water Science and Monitoring

Daniela Conesa – Team Leader, Water Planning

CSIRO

Russell Crosbie – Research Scientist

Appendix C – Water level trends

The Panel has re-calculated 5-year linear regression water level trends for all of the bores from the six key management areas that were used in the 2012 Risk Assessment. Trends calculated for the 2007 – 2012 period (used in the Risk Assessment) were checked, as well as trends for the 2012 – 2016 period (that informed the 2016 Status Report). Trends on individual bores were provided to the Panel in spreadsheet form. The Panel also determined trends for the 2014 – 2018 period.

For the 2007 - 2012 period, the Panel calculated significantly different trends for three bores (CMM057, PEN097 and PEN098) to those contained in the spreadsheet provided to the Panel. For these three bores, the trends calculated by the Panel are provided in Table 2 alongside those provided by the Department. Differences might in part be due to correction of erroneous data on the database between initial analysis in 2012 and the recent analysis by the Panel (DEWNR, pers. comm).

For the 2012 – 2016 period, significant differences also were identified for three bores: CLS049, PEN002 and BIN051. Also, trends for PEN006 and BIN024 were reported for 2007-2012, but not for 2012-2016, even though data is available. These trends are also provided in parentheses below.

Table 2: Five-year (2007 – 2012) trends in watertable decline in six key water management areas. Data from 2007 – 2012 was used to inform the Risk Assessment, and was provided to the Review Panel in spreadsheet form. Trends for 2012 – 2016 are from DEWNR (2017), and 2014 – 2018 trendshave been calculated by the Review Panel from data in the Waterconnect website. Some of the trends for the earlier period are composites of data from more than one bore. Exact periods for trend analysis are 1/3/07-31/3/12, 1/1/12-31/12/16, and 1/1/14-31/10/18. NA means that water level readings have been discontinued. Numbers in parentheses for 2007 – 2012 and 2012-2016 were calculated by the Review Panel.

MANAGEMENT AREA	OBS NO.	2007 – 2012 TREND (M/Y)	RISK ASSESSMENT CATEGORY	2012 – 2016 TREND (M/Y)	2014 – 2018 TREND (M/Y)
Coles	CLS004	0.00		-0.04	+0.34
	CLS006	-0.05		NA	NA
	CLS049	-0.06		+0.11(+0.39)	+0.72
	CLS050	-0.05	>0.1 m/y	+0.02	+0.19
Frances	BIN005	-0.10		NA	NA
	BIN007	-0.07		-0.10	-0.07
	BIN032	-0.07		-0.09	-0.08
	HYN025	-0.11	>0.1 m/y	-0.12	-0.09
Hynam East	HYN001	-0.15		NA	NA
	HYN007	-0.08		-0.23	-0.22
	HYN009	+0.08		NA	NA
	HYN028	+0.04		+0.01	+0.13
	HYN035	+0.67	≤0.1 m/y	+0.03	+0.17
Short	SHT011	+0.33		-0.19	+0.35
	SHT012	+0.08		+0.07	+0.33
	SHT014	-0.09		-0.10	+0.09
	SHT015	+0.05		0.00	+0.32
	SHT026	-0.04		NA	NA
	SHT027	-0.04		NA	NA

	SHT032	+0.51		+0.13	+0.38
	SHT033	0.00		-0.17	+0.15
	SHT034	-0.01		NA	NA
	SHT035	0.00		NA	NA
	SHT036	-0.01	>0.1 m/y	NA	NA
Zone 3A	CMM001	+0.16		NA	NA
	CMM009	+0.07		NA	NA
	CMM011	-0.09		NA	NA
	CMM033	+0.12		NA	NA
	CMM052	+0.07		NA	NA
	CMM056	+0.27		NA	NA
	CMM057	+0.07(+0.13)		NA	NA
	CMM079	+0.19		-0.03	+0.22
	CMM084	+0.07		NA	NA
	CMM085	-0.06		NA	NA
	CMM087	+0.10		-0.06	+0.20
	CMM088	+0.07		NA	NA
	CMM089	+0.01		-0.05	+0.16
	CMM090	+0.01		-0.04	+0.15
	CMM091	+0.02		-0.04	+0.14
	CMM095	+0.12		-0.04	+0.19
	CMM096	+0.12		-0.06	+0.24
	CMM097	+0.12		-0.06	+0.24
	CMM098	+0.12		0.00	+0.24
	CMM099	+0.13		+0.01	+0.19
	CMM100	+0.14		-0.03	+0.14
	MON037	+0.41		+0.08	+0.29
	NAN033	+0.23		NA	NA
	NAN063	0.00		-0.09	+0.05
	PEN002	+0.06		-0.12(-0.23)	+0.14
	PEN003	+0.32		NA	NA
	PEN006	+0.16		(+0.12)	+0.24
	PEN015	+0.26		0.00	+0.30
	PEN016	+0.47		+0.10	+0.23
	PEN030	-0.10		-0.12	+0.03
	PEN067	-0.01		-0.09	+0.04
	PEN087	+0.19		-0.01	+0.17
	PEN088	+0.21		-0.08	+0.12
	PEN089	+0.24		-0.09	+0.12
	PEN090	+0.03		-0.04	+0.16
	PEN091	+0.03		-0.04	+0.14
	PEN092	+0.04		-0.05	+0.13

	PEN093	+0.02		-0.05	+0.12
	PEN094	+0.03		-0.05	+0.12
	PEN095	+0.02		-0.05	+0.12
	PEN096	+0.11		-0.15	-0.09
	PEN097	0.00 (+0.23)		-0.09	NA
	PEN098	0.00 (+0.25)		-0.05	+0.18
	PEN099	+0.24		-0.09	NA
	PEN105	-0.03	≤0.1 m/y	0.00	+0.09
Zone 5A	BIN024	-0.07		(-0.09)	-0.05
	BIN050	-0.15		-0.12	-0.09
	BIN051	-0.14		-0.09 (-0.21)	-0.09
	BIN052	-0.26		-0.10	+0.01
	BIN053	-0.09		-0.11	-0.08
	HYN036	0.00		-0.09	-0.02
	JES004	-0.07		-0.13	+0.06
	JES005	-0.09		NA	NA
	JES007	-0.02		-0.07	+0.06
	JES050	-0.11		-0.13	0.00
	NAR046	+0.32	>0.1 m/y	-0.05	+0.19

Appendix D – Management area summary

Table 3: Summary of key statistics by management area.

	COLES	SHORT	FRANCES	HYNAM EAST	ZONE 3A	ZONE 5A
Water Level Change 2002 – 2012	3/3 bores declining 3/3 declining > 0.1 m/y	4/4 bores declining 3/4 declining > 0.1 m/y	2/2 bores declining 2/2 declining > 0.1 m/y	4/4 bores declining 2/4 declining > 0.1 m/y	13/13 bores declining 4/13 declining >0.1 m/y	3/3 bores declining 3/3 declining > 0.1 m/y
Water Level Trend 2007 – 2012	3/4 bores declining 0/4 declining > 0.1 m/y	5/11 bores declining 0/11 declining > 0.1 m/y	4/4 bores declining 1/4 declining > 0.1 m/y	2/5 bores declining 1/5 declining > 0.1 m/y	3/45 bores declining 0/45 declining > 0.1 m/y	9/11 bores declining 4/11 declining > 0.1 m/y
Water Level Trend 2014 – 2018	3/3 bores rising	0/6 bores declining	3/3 bores declining 0/3 declining > 0.1 m/y	1/3 bores declining 1/3 declining > 0.1 m/y	1/31 bores declining 0/31 declining > 0.1 m/y	5/10 bores declining 0/10 declining > 0.1 m/y
Salinity Trend 2007 – 2012	1/1 bores rising > 0.5%/y	1/4 bores rising > 0.5%/y	0/3 bores rising > 0.5%/y	1/2 bores rising > 0.5%/y	6/15 bores rising > 0.5%/y	1/8 bores rising > 0.5%/y
Demand / TAR (%)	200	170	140	210	120	140
Risk Assessment Rating	VERY HIGH	VERY HIGH	HIGH	HIGH	HIGH	HIGH
GDE area (ha) (Harding 2012)	1352	580	0	0	4680	70
GDE number (Harding 2012)	366	328	0	0	1122	27
High-value GDE area (ha) (Harding 2012)	32	0	0	0	131	0
High-value GDE number (Harding 2012)	1	0	0	0	3	0
Percent GDEs not assessed ('confidence') (Harding 2012)	98.6	99.4	0	0	96.9	44.4
GDEs 'ecological value score[1]' (Table 17 in Conesa et al. 2015)	3	2	0 (No GDEs)	0 (No GDEs)	4	1

GDE 'susceptibility to groundwater change[2]' (Table 17 in Conesa et al. 2015)	4	4	1	1	3	3
Groundwater dependency (Table 17 in Conesa et al. 2015)	Minor (2.4)	Minor (1.6)	No GDEs	No GDEs	Minor (2.4)	Insignificant (0.6)
Hydrological conditions and predicted impacts on GDEs (Cranswick 2018, Fig 4.1)	Declining groundwater level trends have influenced the hydroperiods of GDE wetlands which have shown some recent recovery from previous drier conditions (HZ5)	Declining groundwater level trends have influenced the hydroperiods of GDE wetlands which have shown some recent recovery from previous drier conditions (HZ5)	No GDEs (MA in HZ3)	No GDEs (MA in HZ3)	Eastern half (E of Penola) in HZ5, western half (W of Penola) in HZ7 – In both HZs, declining gw trends have influenced the hydroperiods of GDE wetlands which have shown some recent recovery from previous drier conditions	Declining groundwater level trends have influenced the hydroperiods of GDE wetlands, showing a change from permanent to consistently dry conditions at a number of sites (HZ3)
Significant values (Harding 2012)		Yarra pygmy perch, Dwarf galaxias			Growling grassfrog	

NB The GDEs listed in this table are wetland GDEs; it is strongly recommended that this analysis be repeated for groundwater-dependent vegetation (phreatophytes) as these GDEs are likely to be present in the management areas for which no wetland GDEs have been recorded (e.g., Frances, Hynam East

[1]Ecological value ranging from 1 (Insignificant) to 5 (Very High) based on criteria in Table 5 in Conesa et al. (2015): abundance (numbers/area) of GDEs in the management area, condition (degraded to near-natural), value (low to high), priority (low to high), regional rarity and representativeness, and conservation significance (e.g., supporting endangered or vulnerable species, species sensitive to hydrological change and with limited powers of dispersal, groundwater supporting wetlands of international importance)

[2]Susceptibility to groundwater change ranging from 1 (Insignificant) to 5 (Very High) based on criteria in Table 5 in Conesa et al. (2015): likelihood of connection to Tertiary Limestone Aquifer, depth to water table, and categorisation by Fass and Cook (2005).

The Goyder Institute for Water Research is a partnership between the South Australian Government through the Department for Environment and Water, CSIRO, Flinders University, the University of Adelaide, the University of South Australia, and the International Centre of Excellence in Water Resource Management.