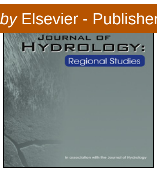
Journal of Hydrology: Regional Studies 4 (2015) 686–698

Journal of Hydrology: Regional Studies

journal homepage: www.elsevier.com/locate/ejrh



A national inventory of seawater intrusion vulnerability for Australia

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ARTICLE INFO

Article history:
Received 20 May 2015
Received in revised form 6 August 2015
Accepted 8 October 2015
Available online 1 December 2015

Keywords:
Coastal aquifers
Seawater intrusion
Vulnerability indicators
Sharp-interface solution

ABSTRACT

Study region: Twenty-eight coastal aquifer case study areas across Australia.
Study focus: Seawater intrusion causes degradation of groundwater resources in coastal areas. The characterization of seawater intrusion is difficult and expensive, and there is therefore a need to develop methods for rapid assessment of seawater intrusion as part of large-scale screening studies in order to guide future investment. We use a steady-state analytic approach to quantify seawater extent and propensity for change in seawater extent under different stresses, in combination with findings from a previous qualitative investigation, which relies on a data-based assessment of regional trends.
New hydrological insights for the region: The combination of methods identified areas of highest risk to SWI including unconfined aquifers at Derby (WA) and Esperance (WA), and confined aquifers at Esperance (WA) and Adelaide (SA). The combination of analytic and qualitative approaches offers a more comprehensive and less subjective seawater intrusion characterization than arises from applying the methods in isolation, thereby imparting enhanced confidence in the outcomes. Importantly, active seawater intrusion conditions occur in many of Australia's confined coastal aquifers, obviating the use of the analytical solution, and suggesting that offshore groundwater resources provide significant contributions to these systems.

1. Introduction

Coastal aquifers are important sources of freshwater supply in Australia (Werner, 2010). Seawater intrusion (SWI), which is the inland encroachment of seawater, has been highlighted as a risk to Australia's coastal aquifers in all states and the Northern Territory due to pressures associated with, for example, increased water demand and climate change (Voice et al., 2006; Werner, 2010). Developing a national-scale evaluation of SWI has been identified as a necessary step towards prioritizing efforts to manage these resources sustainably (Ivkovic et al., 2012).

Around the vast Australian coastline, there is extensive variability in geology, climate, land use, surface water effects, tidal ranges and groundwater use that produces a wide range of coastal aquifer situations. This poses a significant hindrance to the development of a national overview of the state of coastal aquifers with respect to SWI, particularly given the complex nature of the density-dependent flow and transport processes accompanying SWI. In addition, the extent of monitoring

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http://dx.doi.org/10.1016/j.ejrh.2015.10.005

and investigations specific to SWI are highly variable, with detailed SWI investigations (e.g., nested-piezometer monitoring of the freshwater–seawater interface, hydrochemical analyses to elucidate salinity sources, three-dimensional models of dispersive, density-dependent flow and transport, etc.) having occurred for only a few areas (Werner, 2010). Hence, methods that rely on relatively limited information are needed to produce a national overview of SWI in Australia. To achieve this, the method should identify current and emerging risk areas, while taking into account coastal aquifer responses to changes in the key drivers of SWI. This information can then be used to prioritize areas requiring more detailed SWI investigations in the future.

SWI is a complex process and this makes SWI assessment relatively difficult and expensive (Werner et al., 2013). As a result, large-scale reviews of SWI for North America (Barlow and Reichard, 2010), South America (Bocanegra et al., 2010), Europe (Custodio, 2010) and Africa (Steyl and Dennis, 2010) have involved particularly simple methodologies, leading to largely subjective descriptions of hydrogeological settings, and the scales and modes of SWI. There is a lack of quantitative and systematic characterization of individual aquifers, for the purposes of ranking and comparison, within large-scale reviews of SWI. This precludes repeatability of the assessment, and prevents comparison between different field sites. Efforts to standardize the investigation of large-scale SWI vulnerability (defined here as the propensity for SWI to occur) have used methods such as the GALDIT (Chachadi and Lobo-Ferreira, 2007; Lobo-Ferreira et al., 2007; Santha Sophiya and Syed, 2013; Recinos et al., 2015) and CVI (SLR) (Ozyurt, 2007) approaches. Werner et al. (2012) highlight that these methods lack theoretical underpinnings, require subjective rankings, and are based on only a subset of the key factors that influence SWI. For example, SWI vulnerability arising from changes in sea-level, recharge and/or extraction is not captured directly, if at all, and aquifer fluxes are not directly considered.

Recently, an alternative large-scale method has been developed by Werner et al. (2012), who proposed a set of SWI vulnerability indicators for continental unconfined and confined aquifer systems. The method is based on the steady-state, sharp-interface equations of Strack (1976, 1989), and consequently incorporates the main physical mechanisms of SWI, albeit under idealized conditions. The basic premise is that partial derivative equations quantify the propensity for SWI as rates of change in SWI extent for a range of different stresses, e.g., increased extraction, reduced recharge and sea-level rise (SLR). Using this approach, SWI vulnerability can be easily and rapidly quantified. A relatively small number of hydrogeological parameters are required for the method and this makes it suitable for application within data-poor areas. Further, SWI vulnerability to different stresses can be easily compared due to the simple nature of the underlying equations. The method was applied by Werner et al. (2012) to four coastal aquifer systems, where detailed SWI assessments have been carried out, and there was general agreement between their approach and the vulnerability determinations obtained from more detailed investigations. Morgan et al. (2013) applied the Werner et al. (2012) method as part of a first-order assessment of SWI vulnerability for the multi-layered Willunga Basin aquifer system in South Australia, and found that the approach offered useful insights into the relative vulnerability of aquifers at that site. Recently, Morgan and Werner (2014) extended the Werner et al. (2012) vulnerability indicators method to freshwater lens systems in strip islands.

Werner et al. (2012) recommended that additional case studies should be evaluated to produce an extensive database of SWI vulnerability indicators. This would allow for the conversion of vulnerability indicators to descriptive vulnerability definitions (i.e., high, moderate, low) and allow rankings of other SWI cases, thereby offering guidance to future large-scale studies of SWI vulnerability. The aim of this investigation is to address this knowledge gap by applying the methods of Werner et al. (2012) and Morgan and Werner (2014) to aquifers in 28 case study areas across Australia, where seawater intrusion was considered a threat by national groundwater leaders (Ivkovic et al., 2012). The degree to which Australian aquifers are currently vulnerable to SWI, and potentially vulnerable in the future as a consequence of over-extraction and anticipated climate change impacts, will be considered. Conceptualization and parameterization of each case study site were carried out by Ivkovic et al. (2013), and the resulting parameter values are adopted in the current analysis.

It is important to recognize that the approaches of Werner et al. (2012) and Morgan and Werner (2014) have a number of limitations arising from the simplification of the conceptual system and the assumptions inherent in the analytical model. For example, key elements of SWI vulnerability are not captured, including temporal factors (e.g., seasonality and interannual climate events such as droughts), spatial variations (e.g., in recharge, pumping, aquifer properties and geometry), physical processes (e.g., land-surface overtopping, outflow face at the shoreline to accommodate submarine groundwater discharge, tidal impacts) and other important elements (e.g., the salinity of individual pumping wells, previous incidences of SWI, management practices, and the degree of knowledge and understanding of coastal aquifer processes). In order to overcome a number of these limitations, this study will use the results from a previous qualitative investigation of SWI vulnerability by Ivkovic et al. (2012), which relies on a data-based assessment of regional and temporal trends. Results from the two approaches will be used to provide a complementary evaluation of SWI vulnerability. To be clear, our goal is not to compare results of the two approaches, rather it is to carry out a national-scale assessment of SWI vulnerability for Australia using two separate methodologies that provide information on differing elements of SWI vulnerability. While the analytic approach of Werner et al. (2012) and Morgan and Werner (2014) offers insight into the theoretical extent of seawater within an aquifer, the qualitative approach of Ivkovic et al. (2012) evaluates regional and temporal trends in factors that are thought to increase SWI vulnerability.

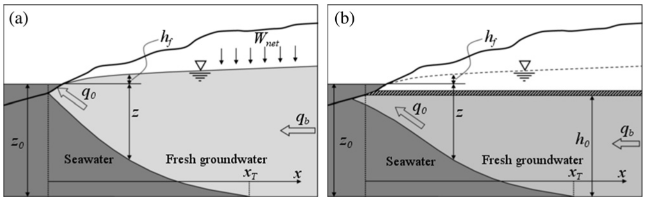

Fig. 1. Description of hydrogeological parameters for: (a) unconfined aquifer and (b) confined aquifer settings (adapted from Werner et al., 2012).

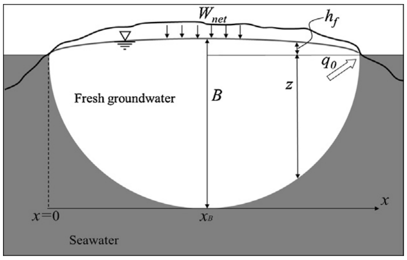

Fig. 2. Description of hydrogeological parameters for a freshwater lens (adapted from Morgan and Werner, 2014).

2. Methods

2.1. Analytic approach

The Strack (1976, 1989) solution for the steady-state position of the sharp freshwater-saltwater interface is the basis for the methods described by Werner et al. (2012) and Morgan and Werner (2014). The conceptual model used by Werner et al. (2012) for unconfined and confined aquifers is provided in Fig. 1 and notation is summarized in Table A1 of the Appendix. SWI vulnerability for current conditions was determined using the extent of SWI within the aquifer. For unconfined aquifers, this is determined using a mixed convection ratio M, and for confined aquifers it is determined using the wedge toe x_T, which is the maximum inland extent of the interface under steady-state conditions, as described further in Section 3.1.1. SWI vulnerabilities to future stresses, including SLR, recharge change and change in the seaward groundwater flux (e.g., associated with large-scale pumping effects) were determined using partial derivatives. The resulting equations are provided in Table A2 (unconfined aquifers) and Table A3 (confined aquifers) of the Appendix.

It is not possible to directly compare the vulnerability indicators (i.e., the partial derivatives) because they each have different dimensions. However, comparison can be carried out use normalized sensitivities (Kabala, 2001), which require prediction of future stress changes. Normalized sensitivities (also referred to as scaled partial derivatives) are calculated by multiplying the predicted stress change by the partial derivative, e.g., $\Delta z_0 \left(\partial x_T/\partial z_0\right)$ for SLR, to arrive at an approximation for the linearized change in toe location for the applied stress. It is important to note that the value of $\Delta z_0 \left(\partial x_T/\partial z_0\right)$ does not equate to Δx_T, because the relevant equations are non-linear.

Morgan and Werner's (2014) conceptual model of a freshwater lens is illustrated in Fig. 2, and notation is summarized in the Appendix. The interface does not intersect the aquifer basement and there is no wedge toe. Instead, SWI is quantified using changes in the freshwater thickness at the centre of the lens B, as well as the volume of freshwater in the lens. SWI vulnerability to future stresses of SLR and recharge change was quantified using equations developed for both flux-controlled and head-controlled boundary conditions, which is consistent with the approach taken for continental aquifers by Werner et al. (2012). The resulting equations are given in Table A4 of the Appendix. For brevity, we refer the reader to Werner et al. (2012) and Morgan and Werner (2014) for details of the underlying theory.

The methodology described above was applied to the 28 case study areas shown in Fig. 3, involving a total of 44 aquifers (the confined and unconfined aquifers of multiple-aquifer systems were considered separately). Using publicly available information, simplified cross-sectional conceptualizations of the case study areas were developed by Ivkovic et al. (2013), who also assigned representative aquifer parameters to each setting. Water levels, recharge and extraction were assessed for the period 2000–2010, unless data were limited and then long-term average values were reported. We use the hydrogeological parameters determined by Ivkovic et al. (2013) to firstly approximate the steady-state extent of the seawater wedge, and then to estimate the propensity for change in seawater extent under SLR, recharge change and (for non-island cases) change in the seaward groundwater flux.

2.2. Qualitative approach

The qualitative approach of Ivkovic et al. (2012) considered a range of factors thought to influence SWI, including: ratio of groundwater extraction to recharge; groundwater levels (i.e., minimum groundwater level that at least 20% of monitoring

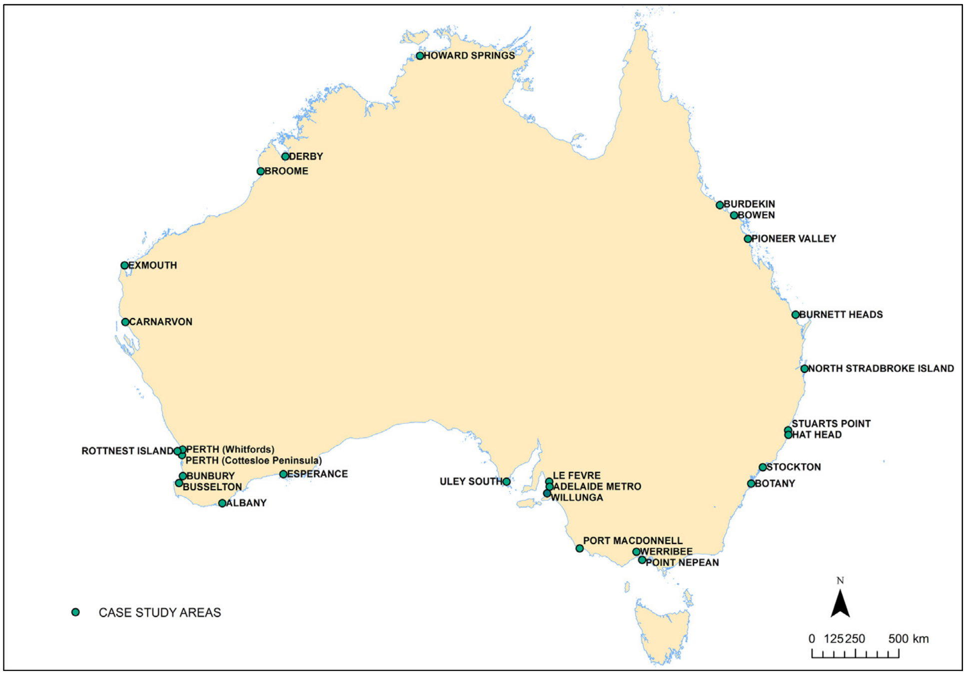

Fig. 3. Case study areas.

bores fell below during 2000–2010); the current level of SWI-specific knowledge, monitoring and management; rainfall trends; salinity trends (i.e., change in maximum salinity value from 1990–1999 to 2000–2010 that was exceeded by at least 20% of bores); and the tidal setting. These factors address a number of the limitations associated with the analytic approach noted within Section 1. The qualitative approach used a matrix-style indexing method to rank each site according to the vulnerability to SWI. Each factor was assigned a weighting (of between 1 and 4), which represents the (subjectively determined) relative importance of the factor in terms of SWI vulnerability. Each factor was also subdivided into classes, with a rating (of between 1 and 10, with 10 representing highest vulnerability to SWI) assigned to each class. For example, the tidal setting factor was divided into three classes: macro-tidal (rating of 10), meso-tidal (rating of 5), micro-tidal (rating of 1). High tides associated with macro-tidal settings cause elevated time-averaged coastal head conditions, and involve more extensive inland propagation of seawater through tidal creeks and rivers (e.g., Werner and Gallagher, 2006), and hence a rating of 10 was used. Weightings and ratings were multiplied together for each factor and then summed to obtain a final indexing score for each site. Indexing scores were categorized into three groups for assigning SWI vulnerability: Low (17–51), Moderate (52–102), High (103–170). For brevity, we refer the reader to Ivkovic et al. (2012) for further details of the method.

The qualitative approach was applied by Ivkovic et al. (2012) to the 28 case study sites shown in Fig. 3 (i.e., the same sites that were assessed using the analytical approach). When multiple (i.e., stacked) aquifers existed at a single location, an overall characterization was given for all aquifers at that location because the data did not allow for assessment of individual aquifers.

3. Results and discussion

3.1. Analytical approach

3.1.1. Theoretical SWI extent

The estimated steady-state extent of seawater in Australian unconfined aquifers is listed in Table 1. Given that the equations assume steady-state conditions, the following values of seawater extent represent the condition of the respective aquifers under average stresses for the period 2000–2010, assuming that enough time has passed for equilibrium conditions to have established. As such, the results based on the Werner et al. (2012) equations may differ to field observations of seawater extent in each case, but these differences in themselves are nonetheless informative. Unconfined aquifers are ranked using magnitude of the mixed convection ratio M (Eq. (A.7)). $M = 1$ is a significant value and threshold of stable/unstable conditions. Unstable interface conditions occur for values of $M \geq 1$, where the freshwater discharge to the coast is insufficient for the wedge toe to reach a steady-state location, and subsequently SWI is driven by a hydraulic gradient sloping downwards in the inland direction. As pointed out by Werner et al. (2012) and Mazi et al. (2014), a theoretical tipping point of complete seawater intrusion has been exceeded under these conditions, leading to complete aquifer salinization. This is expected to result in more aggressive SWI, and most likely involves situations where the wedge toe is actively moving inland. The

Table 1
Seawater extent in unconfined aquifers.

Case study	Aquifer	x_T (m)	$x_T{}'$ (−)	M (−)	V_{sw} (m^3/m)
Derby (WA)	Wallal/Erskine sandstone	Unstable	1.0	11	Unstable
Exmouth (WA)	Cape range group	Unstable	1.0	4.5	Unstable
Burdekin (QLD)	Unconfined	Unstable	1.0	3.5	Unstable
Broome, Cable Beach (WA)	Broome sandstone	7700	0.42	0.66	46,000
Port MacDonnell (SA)	Tertiary limestone	15,000	0.30	0.51	400,000
Broome, coconut wells (WA)	Broome sandstone	5500	0.24	0.42	35,000
Burnett Heads, Moore Park (QLD)	Elliott formation	390	0.23	0.41	560
Botany sands (NSW)	Botany sand beds	210	0.19	0.35	500
Esperance (WA)	Superficial/Pallinup	630	0.15	0.28	410
Perth, Whitfords (WA)	Superficial	1100	0.10	0.18	5600
Busselton (WA)	Superficial	44	0.06	0.12	29
Bowen (QLD)	Unconfined	440	0.04	0.08	290
Le Fevre (SA)	Semaphore sands	42	0.04	0.08	41
Willunga (SA)	Quarternary	200	0.04	0.08	390
Stockton (NSW)	Stockton sand beds	55	0.04	0.08	27
Hat head (NSW)	Coastal sands	120	0.03	0.07	140
Uley South (SA)	Bridgewater formation	220	0.03	0.06	220
Stuarts point (NSW)	Coastal sands	110	0.03	0.06	130
Nth Stradbroke, East (QLD)	Unconfined	37	0.02	0.04	150
Albany, ocean side (WA)	Werrillup formation sand	34	0.02	0.04	22
Werribee (VIC)	Alluvium/fractured rock	42	0.02	0.03	56
Burnett heads, Bargara (QLD)	Elliott formation	90	0.01	0.03	130
Carnarvon (WA)	Riverbed sand	89	0.01	0.02	30
Bunbury (WA)	Superficial	59	0.01	0.02	59
Albany, harbour side (WA)	Superficial	6	0.01	0.02	2
Pioneer valley (QLD)	Unconfined	200	0.01	0.01	240
Nth Stradbroke, West (QLD)	Unconfined	9	0.00	0.00	37

Table 2
Seawater extent in confined aquifers.

Case study	Aquifer	x_T (m)	V_{sw} (m^3/m)
Le Fevre (SA)	T1	Unstable	Unstable
Le Fevre (SA)	T2	Unstable	Unstable
Adelaide metro (SA)	T1	Unstable	Unstable
Adelaide metro (SA)	T2	Unstable	Unstable
Willunga (SA)	Maslin sands	Unstable	Unstable
Burnett heads, Bargara (QLD)	Fairymead beds	Unstable	Unstable
Esperance (WA)	Werillup	Unstable	Unstable
Bunbury (WA)	Yarragadee	670,000	6,700,000
Perth, Whitfords (WA)	Yarragadee	55,000	2,700,000
Willunga (SA)	Port Willunga formation	31,000	28,0000
Perth, Whitfords (WA)	Leederville	23,000	14,000
Carnarvon (WA)	Older alluvium	9400	21,000
Port MacDonnell (SA)	Tertiary sands	4200	170,000
Busselton (WA)	Leederville	3750	8125
Uley South (SA)	Wanilla sands	600	1800
Albany, harbour side (WA)	Pallinup/Werrilup	140	94
Howard springs (NT)	Koolpinyah/Coomalie dolomite	80	67

majority of M values for the unconfined aquifers have $M < 1$, with three cases i.e., Derby (WA), Exmouth (WA) and the Burdekin (QLD) having M values that ranged between 1 and 11.

Results for the 17 confined aquifers are shown in Table 2. Confined aquifers are ranked using x_T (i.e., the toe of the interface and furthest inland extent of seawater under steady-state conditions, Eq. (A.18)), which is more than 5 km from the coast in the majority of cases, and is unstable in seven cases. Unstable interface conditions occur because all of the heads within the aquifer are lower than the coastal density-corrected head at the base of the aquifer, and therefore the hydraulic gradient slopes downwards in the inland direction. In this case, freshwater flow to the coast has ceased and active SWI is occurring. Ranking of unstable aquifers was not possible because the method does not offer insight into the degree of unstableness; however, these were assumed to represent the greatest SWI extent. A major limitation of the confined aquifer results is that the aquifer is truncated at the shoreline boundary, whereas confined aquifers are expected to discharge offshore. Nonetheless, the current method is useful for the purposes of ranking based on SWI extent and stability conditions. There is very limited understanding of the offshore extension of confined aquifers in Australia, and hence it is currently not possible to include this element of confined aquifer SWI vulnerability within the present study.

Results for freshwater lens systems are listed in Table 3, with ranking based on the magnitude of maximum freshwater lens thickness B. Values of B ranged between 5 m for Perth, Cottesloe (WA) and 67 m for Point Nepean (Vic).

Table 3
Seawater extent in freshwater lens cases.

Case study	Aquifer	B (m)	V_{fw} (m^3/m)
Perth, Cottesloe (WA)	Tamala limestone	5	2400
Rottnest (WA)	Tamala limestone	17	12,000
Point Nepean (Vic)	Quaternary	67	140,000

Table 4
Unconfined aquifer vulnerability indicators for sea-level rise, recharge change and change in seaward groundwater flux.

Case study	Aquifer	Flux-controlled			Head-controlled	
		$\partial x_T/\partial z_0(-)$	$\partial x_T/\partial W_{\mathrm{net}}(\mathrm{d})$	$\partial x_T/\partial q_b(\mathrm{d/m})$	$\partial x_T/\partial z_0(-)$	$\partial x_T/\partial W_{\mathrm{net}}(\mathrm{d})$
Derby (WA)	Wallal/Erskine	Unstable	Unstable	Unstable	Unstable	Unstable
Burdekin (QLD)	Unconfined	Unstable	Unstable	Unstable	Unstable	Unstable
Exmouth (WA)	Cape range group	Unstable	Unstable	Unstable	Unstable	Unstable
Port MacDonnell (SA)	Tertiary limestone	120	−2.2.E+08	−10000	1600	−9.2.E+07
Broome, Cable Beach (WA)	Broome sandstone	100	−1.5.E+08	−5400	1800	−5.6.E+07
Esperance (WA)	Superficial/Pallinup	69	−9.2.E+07	−4500	480	−4.0.E+07
Broome, coconut wells (WA)	Broome sandstone	63	−1.7.E+07	−4300	660	−7.7.E+06
Burnett, Moore Park (QLD)	Elliott formation	60	−1.4.E+07	−1300	620	−6.7.E+06
Uley South (SA)	Bridgewater formation/Wanilla	48	−4.1.E+06	−1200	190	−2.0.E+06
Bowen (QLD)	Unconfined	45	−4.0.E+06	−770	120	−1.9.E+06
Carnarvon (WA)	Riverbed sand	36	−3.7.E+06	−390	52	−1.8.E+06
Perth, Whitfords (WA)	Superficial	32	−1.8.E+06	−290	160	−8.0.E+05
Uley South (SA)	Bridgewater formation	30	−1.3.E+06	9200	69	−6.5.E+05
Willunga (SA)	Quarternary	20	−8.3.E+05	−170	53	−4.1.E+05
Botany sands (NSW)	Botany sand beds	19	−7.3.E+05	−170	160	−3.6.E+05
Pioneer valley (QLD)	Unconfined	13	−6.7.E+05	−120	17	−3.3.E+05
Burnett, Bargara (QLD)	Elliott formation	12	−3.7.E+05	−120	19	−1.8.E+05
Busselton (WA)	Superficial	9	−5.6.E+05	−792	32	−2.7.E+05
Le Fevre (SA)	Semaphore sands	8	−2.0.E+05	−96	22	−8.8.E+04
Bunbury (WA)	Superficial	8	−1.9.E+05	−71	11	−9.6.E+04
Stockton (NSW)	Stockton sand beds	7	−1.8.E+05	−56	19	−9.1.E+04
Hat head (NSW)	Coastal sands	7	−1.7.E+05	−53	17	−8.4.E+04
Stuarts point (NSW)	Coastal sands	6	−1.7.E+05	−48	14	−8.3.E+04
Werribee (Vic)	Alluvium/FR	4	−1.5.E+05	−44	7	−7.6.E+04
Albany, ocean side (WA)	Werrillup formation	3	−7.7.E+04	−41	6	−3.8.E+04
Albany, harbor side (WA)	Superficial	2	−4.1.E+04	−22	3	−2.0.E+04
Nth Stradbroke, East (QLD)	Unconfined	2	−1.3.E+04	−20	3	−6.5.E+03
Nth Stradbroke, West (QLD)	Unconfined	0	−1.0.E+04	0	0	−5.0.E+03

3.2. *SWI vulnerability indicators*

The propensity for change in seawater extent due to different stresses (SLR, recharge change and change in seaward groundwater flux) was calculated using derivative equations (i.e., SWI vulnerability indicators). SWI vulnerability indicators for unconfined aquifers are listed in Table 4. Ranking was based on the magnitude of SWI vulnerability indicators for SLR under flux-controlled conditions. Although ranking of aquifers would differ depending on the stress selected, changes in the order by ranking to different stresses were predominantly minor. Vulnerability indicators could not be calculated for aquifers with unstable interface conditions, but these were nonetheless ranked at the top of Table 4 given their state of instability.

SWI vulnerability indicators for confined aquifers are listed in Table 5. Steady-state interface location is insensitive to SLR under flux-controlled conditions (i.e., $\partial x_T/\partial z_0 = 0$) and is therefore not reported. As with unconfined aquifers, vulnerability indicators could not be calculated for confined aquifers with unstable interface conditions and a high ranking was assigned. Ivkovic et al. (2013) found that estimates of net recharge for confined aquifers were difficult to obtain. For this reason, a simplifying assumption of zero net distributed recharge was applied. This assumption is commonly made when carrying out simple first-order SWI assessments within confined aquifer systems (Custodio, 1987). Neglecting the SWI vulnerability of confined aquifers associated with recharge change was considered a reasonable assumption given the longer residence times of confined aquifers. Ranked vulnerabilities to recharge change and SLR for lens systems are shown in Table 6.

3.3. *Normalized sensitivities*

Normalized sensitivities were calculated for all aquifers using stress changes selected to reflect possible future scenarios under climate change and increased extraction. A SLR of 1 m was applied, which is within the range of values predicted by the IPCC (2013). Recharge change predictions are highly variable for different Australian locations. For example, Green et al. (2007) found a significant increase in recharge for North Stradbroke Island (Queensland), whereas both increase and decrease

Table 5
Confined aquifer vulnerability indicators for sea-level rise and change in seaward groundwater flux.

		Head-controlled	
Case study	Aquifer	$\partial x_T/\partial z_0(-)$	$\partial x_T/\partial q_b$(d/m)
Le Fevre (SA)	T1	Unstable	Unstable
Le Fevre (SA)	T2	Unstable	Unstable
Adelaide metro (SA)	T1	Unstable	Unstable
Adelaide metro (SA)	T2	Unstable	Unstable
Willunga (SA)	Maslin sands	Unstable	Unstable
Burnett heads, Bargara (QLD)	Fairymead beds	Unstable	Unstable
Esperance (WA)	Werillup	Unstable	Unstable
Bunbury (WA)	Yarragadee	4.2E + 07	−2.0E + 07
Willunga (SA)	Port Willunga formation	2.6E + 05	−9.8E + 05
Perth, Whitfords (WA)	Leederville	7.2E + 04	−1.4E + 06
Carnarvon (WA)	Older alluvium	3.9E + 04	93.5E + 06
Perth, Whitfords (WA)	Yarragadee	3.6E + 04	95.3E + 04
Busselton (WA)	Leederville	1.3E + 04	−2.7E + 05
Port MacDonnell (SA)	Tertiary sands	710	−870
Uley South (SA)	Wanilla sands	350	−360
Albany, harbor side (WA)	Pallinup/Werrilup	100	−790
Howard springs (NT)	Koolpinyah/Coomalie	10	−20

Table 6
Freshwater lens vulnerability indicators for sea-level rise and recharge change.

		Flux-controlled		Head-controlled	
Case study	Aquifer	$\partial B/\partial z_0(-)$	$\partial B/\partial W_{net}$(d)	$\partial B/\partial z_0(-)$	$\partial B/\partial W_{net}$(d)
Point Nepean (Vic)	Quarternary	0	3.1E + 05	−41	0
Rottnest (WA)	Tamala limestone	0	2.6E + 05	−41	0
Perth, Cottesloe (WA)	Tamala limestone	0	2.7E + 04	−41	0

in recharge was obtained for the Gnangara Mound (Western Australia). In contrast, Green et al. (2011) predicted a recharge reduction of up to 58% by 2070 in the Clare Valley, South Australia. For the sake of simplicity, a scenario involving a 25% reduction in recharge is applied within this analysis. Normalized sensitivities for recharge change in confined aquifers are not reported because SWI vulnerability to recharge change in confined aquifers is not considered. It is also difficult to predict future changes in the seaward groundwater flux arising from changes in extraction, but a 25% reduction has been employed (i.e., $\Delta q_b = -0.25\,q_b$) for consistency with the approach to future recharge changes. The results shown in Table 7 indicate that, for the stresses considered, unconfined aquifers were most sensitive to either SLR under head-controlled conditions (11 cases), recharge change under flux-controlled conditions (9 cases) and change in seaward groundwater flux (4 cases). Confined aquifers were most sensitive to SLR under head-controlled conditions in the majority of cases.

3.4. *SWI vulnerability classification*

The ranking of unconfined aquifers listed in Table 1 was used to assign a vulnerability classification, with the top one-third of aquifers classified as high (H), the middle third as moderate (M) and the bottom third as low (L) (Table 8). Similarly, a vulnerability classification of confined aquifers listed in Table 2 was carried out by assigning the top seven aquifers (i.e., those listed as having unstable interface conditions) as H, the next five aquifers as M and the remaining five aquifers as L (Table 9). There were too few lens cases for a vulnerability classification to be carried out.

The vulnerability classification is based on the theoretical SWI extent under current conditions (Tables 1 and 2), so as to be comparable to the results of the qualitative assessment, which subjectively combines several parameters relating to the present-day aquifer condition. Although not carried out in this study, vulnerability classifications can also be determined for future stresses using vulnerability indicators, such as for SLR, recharge change and increased extraction (Tables 4 and 5). This is not possible with the qualitative approach. By comparing results listed in Tables 1 and 4 for unconfined aquifers, and Tables 2 and 5 for confined aquifers, it can be seen that systems with large SWI extent under current conditions also tend to have high vulnerability indicator values. That is, when an aquifer has a large inland extent of SWI under current conditions, it also tends to have a high propensity for SWI to occur with future stress changes. This justifies the use of M, x_T and B for classifying vulnerability within the present study.

3.5. *Qualitative approach*

Ivkovic et al. (2012) also classified aquifers as H, M or L vulnerability to SWI based on the results of the qualitative investigation (Tables 8 and 9). The qualitative approach of Ivkovic et al. (2012) resulted in 14 aquifers being classified as H, 28 as M and 2 as L. In contrast, the analytic method adopts a relatively even spread. The vulnerability classifications from

Table 7
Normalized sensitivities.

Case study	Aquifer	Flux-controlled			Head-controlled	
		$\frac{\partial x_T}{\partial z_0}\Delta z_0(\mathrm{m})$	$\frac{\partial x_T}{\partial W_{\mathrm{net}}}\Delta W_{\mathrm{net}}(\mathrm{m})$	$\frac{\partial x_T}{\partial q_b}\Delta q_b(\mathrm{m})$	$\frac{\partial x_T}{\partial z_0}\Delta z_0(\mathrm{m})$	$\frac{\partial x_T}{\partial W_{\mathrm{net}}}\Delta W_{\mathrm{net}}(\mathrm{m})$
Uley South (SA)	Bridgewater formation	30	57	41	69	28
	Wanilla sands	0	–	150	350	–
Port MacDonnell (SA)	Tertiary limestone	120	4600	4700	1600	1900
	Tertiary sands	0	–	1000	710	–
Le Fevre (SA)	Semaphore sands	8	11	0	22	5
Willunga (SA)	Quarternary	20	50	14	53	25
	Port Willunga formation	0	–	7300	260,000	–
Werribee (VIC)	Alluvium/fractured rock	4	11	1	7	5
Pioneer valley (QLD)	Unconfined	13	50	48	17	25
Burnett, Moore Park (QLD)	Elliott formation	60	110	360	620	49
Burnett, Bargara (QLD)	Elliott formation	12	23	78	19	11
Bowen (QLD)	Unconfined	45	110	100	120	55
*N*th Stradbroke, East (QLD)	Unconfined	2	9	7	3	5
*N*th Stradbroke, West (QLD)	Unconfined	0	2	2	0	1
Perth, Whitfords (WA)	Superficial	32	290	220	160	140
	Leederville	0	–	6000	72,000	–
	Yarragadee	0	–	14,000	3600	–
Esperance (WA)	Superficial/Pallinup	69	170	180	480	79
Albany, ocean side (WA)	Werrillup formation sand	3	8	1	6	4
Albany, harbor side (WA)	Superficial	2	1	1	3	1
	Pallinup/Werrilup	0	–	35	101	–
Busselton (WA)	Superficial	3	4	1	4	2
	Leederville	0	–	205	106	–
Bunbury (WA)	Superficial	8	15	7	11	7
	Yarragadee	0	–	150,000	42,000,000	–
Carnarvon (WA)	Riverbed Sand	36	22	8	52	11
	Older alluvium	0	–	2600	39,000	–
Broome, coconut wells (WA)	Broome sandstone	63	1600	1600	660	680
Broome, Cable Beach (WA)	Broome sandstone	100	2600	3100	1800	960
Hat head (NSW)	Coastal sSands	7	31	16	17	15
Stuarts point (NSW)	Coastal sands	6	28	16	14	14
Stockton (NSW)	Stockton sand beds	7	14	0	19	7
Botany sands (NSW)	Botany sand beds	19	58	7	160	26
Howard springs (NT)	Koolpinyah/Coomalie dolomite	0	–	20	10	–

Table 8
Unconfined aquifer vulnerability classifications from the analytic and qualitative approaches.

Case study	Aquifer	Analytic	Qualitative
Derby (WA)	Wallal/Erskine sandstone	H	H
Exmouth (WA)	Cape range group	H	M
Burdekin (QLD)	Unconfined	H	M
Broome, Cable Beach (WA)	Broome sandstone	H	M
Port MacDonnell (SA)	Tertiary limestone	H	M
Broome, Coconut Wells (WA)	Broome sandstone	H	M
Burnett heads, Moore Park (QLD)	Elliott formation	H	M
Botany sands (NSW)	Botany sand beds	H	M
Esperance (WA)	Superficial/Pallinup	H	H
Perth, Whitfords (WA)	Superficial	M	H
Busselton (WA)	Superficial	M	H
Bowen (QLD)	Unconfined	M	M
Le Fevre (SA)	Semaphore sands	M	M
Willunga (SA)	Quarternary	M	M
Stockton (NSW)	Stockton sand beds	M	M
Hat head (NSW)	Coastal sands	M	M
Uley South (SA)	Bridgewater formation	M	M
Stuarts Point (NSW)	Coastal sands	M	H
*N*th Stradbroke, East (QLD)	Unconfined	L	L
Albany, ocean side (WA)	Werrillup formation sand	L	M
Werribee (VIC)	Alluvium/fractured rock	L	M
Burnett Heads, Bargara (QLD)	Elliott formation	L	M
Carnarvon (WA)	Riverbed sand	L	H
Bunbury (WA)	Superficial	L	M
Albany, harbor side (WA)	Superficial	L	M
Pioneer valley (QLD)	Unconfined	L	M
*N*th Stradbroke, West (QLD)	Unconfined	L	L

Table 9
Confined aquifer vulnerability classifications from the analytic and qualitative approaches.

Case study	Aquifer	Analytic	Qualitative
Le Fevre (SA)	T1	H	M
Le Fevre (SA)	T2	H	M
Adelaide metro (SA)	T1	H	H
Adelaide metro (SA)	T2	H	H
Willunga (SA)	Maslin sands	H	M
Burnett heads (QLD)	Fairymead beds	H	M
Esperance (WA)	Werillup	H	H
Bunbury (WA)	Yarragadee	M	M
Perth, Whitfords (WA)	Yarragadee	M	H
Willunga (SA)	Port Willunga formation	M	M
Perth, Whitfords (WA)	Leederville	M	H
Carnarvon (WA)	Older alluvium	M	H
Port MacDonnell (SA)	Tertiary sands	L	M
Busselton (WA)	Leederville	L	H
Uley South (SA)	Wanilla sands	L	M
Albany, harbor side (WA)	Pallinup/Werrilup	L	M
Howard springs (NT)	Koolpinyah/Coomalie dolomite	L	M

Table 10
Comparison of vulnerability classifications for the analytic and qualitative approaches.

		Qualitative		
		High	Moderate	Low
Analytic	High	5	11	0
	Moderate	6	8	0
	Low	2	10	2

the analytic method and the qualitative approach were equivalent in 15 of the 44 aquifers. 27 cases differed by one level of vulnerability (i.e., M–H or L–M), and two cases differed significantly (i.e., L–H) (Table 10).

The Derby (unconfined), Esperance (unconfined and confined) and Adelaide Metro (confined) aquifers were classified as H using both approaches. Derby was classified as H by the qualitative approach primarily because of a high extraction-to-recharge ratio, low groundwater levels, low knowledge, monitoring and management, and a macrotidal setting. Esperance had a high extraction-to-recharge ratio and low groundwater levels. Adelaide Metro was classified as H mainly because of the high extraction-to-recharge ratio and declining rainfall. Using the analytic method, Derby was classified as H because of unstable interface conditions, which are partly attributable to the large thickness of the aquifer (around 350 m at the coast). The unconfined Esperance aquifer has a moderately high mixed convection ratio due to low recharge and low groundwater levels. Unstable interface conditions were calculated for the confined Esperance and Adelaide Metro aquifers, noting that the groundwater heads in these aquifers are lower than the density-corrected heads at the coast, resulting in a landward-sloping hydraulic gradient.

There were two aquifers with a mixed classification combination of H and L in Tables 8–10. The unconfined Carnarvon and the confined Busselton aquifers were classified as H by the qualitative approach because of high extraction-to-recharge ratios and declining trends in rainfall. A small mixed convection ratio was calculated for Carnarvon and is primarily attributable to the aquifer being very thin (5 m at the coast), which restricts significantly the seawater extent. The seawater extent of the confined Busselton aquifer was limited by the relatively low hydraulic conductivity (i.e., 2 m/d) and high inland head (i.e., 1.5 m AHD at 1.5 km from the coast, where AHD refers to the Australian Height Datum, with 0 m AHD being approximately mean sea level).

4. Conclusions

In this study, we have undertaken an evaluation of the vulnerability of Australia's coastal aquifers using the analytic methods described by Werner et al. (2012) and Morgan and Werner (2014). The physically based, analytic method gives information on the extent of seawater in aquifers and the change from passive to active SWI, where active SWI infers that the interface is moving inland under a landward-sloping hydraulic gradient. The method also offers insights into the propensity for future stresses to change seawater extent. The analytic approach was used to complement the results of a qualitative assessment, and provides a quantitative and less subjective addition to the regional and temporal trends in factors obtained from the qualitative assessment that are thought to increase SWI vulnerability. The combination of the two methods identified a number of areas as being at high risk, including unconfined aquifers at Derby (WA) and Esperance (WA), and confined aquifers at Esperance (WA) and Adelaide metropolitan area (SA). Active SWI was found to occur in seven out of the seventeen confined aquifer systems assessed. This suggests that offshore groundwater resources provide significant contributions to these systems.

The benefits of the analytic approach are the ease of application and the requirement to consider the hydrogeological parameters that control SWI. However, without complementary methods that account for important, but otherwise unaccounted, features of the system, the approach may not be reliable. This supports the use of multiple lines of evidence, even for first-order assessments of SWI, given the strengths and weaknesses of different methods. We propose that the combined method of Werner et al. (2012) and Morgan and Werner (2014) is best applied as a complement to other sources of information regarding SWI vulnerability, such as the qualitative indexing approach of Ivkovic et al. (2012).

Acknowledgements

The authors gratefully acknowledge Baskaran Sundaram and the Geosciences Australia team, as well as Karen Ivkovic (Naiades Geohydrology), for helpful discussions. This work was funded by the National Centre for Groundwater Research and Training, a collaborative initiative of the Australian Research Council and the National Water Commission.

Appendix

Table A1
Notation.

The following symbols are used in this paper	
h_f	Freshwater head above mean sea level [L]
z	Depth to the interface from mean sea level [L]
z_0	Depth to aquifer base from mean sea level [L]
W_{net}	Net recharge (accounting for infiltration, evapotranspiration and distributed pumping) [L/T]
q_0	Freshwater discharge at the coast [L^2/T]
q_b	Lateral inflow from aquifers inland of the inland boundary [L^2/T]
x_b	Distance of the inland boundary from the coastal boundary [L]
h_b	Freshwater head at the inland boundary [L]
h_0	Saturated confined aquifer thickness [L]
x_B	Half width of island [L]
ρ_s	Seawater density [M/L^3]
ρ_f	Freshwater density [M/L^3]
δ	Density ratio, i.e., $(\rho_s - \rho_f)/\rho_f$ [–]
K	Hydraulic conductivity [L/T]
n	Porosity [–]
x_T	Wedge toe location [L]
x_n	Location of inland no flow boundary in an unconfined aquifer or freshwater lens [L]
$x_T{}'$	Scaled wedge toe position (i.e., x_T/x_n) [–]
V_{sw}	Volume of seawater per unit length of aquifer [L^2]
M	Mixed convection ratio [–]
B	Freshwater thickness at the centre of the lens [L]
V_{fw}	Freshwater volume per unit length of aquifer [L^2]

Table A2
SWI vulnerability equations for unconfined aquifers (after Werner et al., 2012).

Current conditions	
$h_f = \sqrt{\frac{2q_0x - W_{net}x^2}{K} + \left(1+\delta\right) z_0^2} - z_0$	where $x \geq x_T$(A.1)
$h_f = \sqrt{\left(\frac{\delta}{1+\delta}\right) \frac{2q_0x - W_{net}x^2}{K}}$	where $x \geq x_T$(A.2)
$q_0 = \frac{K\left(\left(h_b+z_0\right)^2 - \left(1+\delta\right)z_0^2\right) + W_{net}{x_b}^2}{2x_b}$	where $x_b \geq x_T$(A.3)
$q_0 = \left(\frac{1+\delta}{\delta}\right) \frac{K}{2x_b} h_b^2 + \frac{W_{net}x_b}{2}$	where $x_b \geq x_T$(A.4)
$x_T = \frac{q_0}{W_{net}} - \sqrt{\left(\frac{q_0}{W_{net}}\right)^2 - \frac{K\delta\left(1+\delta\right)z_0^2}{W_{net}}}$	where $W_{net} > 0$(A.5)
$x_T = \frac{K\delta\left(1+\delta\right)z_0^2}{2q_0}$	where $W_{net} > 0$(A.6)
$M = \frac{K\delta\left(1+\delta\right)z_0^2}{W_{net}x_n^2}$	where $W_{net} > 0$(A.7)
$V_{sw} = nz_0\left(x_T - \frac{x_n}{2}\left(\sqrt{\frac{1}{M}}\arcsin\left(\sqrt{M}\right) - \sqrt{1-M}\right)\right)$	$W_{net} > 0$(A.8)
Future stresses	
Flux-controlled setting	Head-controlled setting
Sea level rise $\frac{\partial x_T}{\partial z_0} = \frac{x_n M}{z_0\sqrt{1-M}}$ (A.9)	Sea level rise $\frac{\partial x_T}{\partial z_0} = \frac{x_n M}{z_0\sqrt{1-M}} + \frac{x_n M}{\delta z_0}\left(\frac{1-\sqrt{1-M}}{\sqrt{1-M}}\right)$(A.10)
Change in net recharge $\frac{\partial x_T}{\partial W_{net}} = -\frac{x_n M}{2W_{net}\sqrt{1-M}}$ (A.11)	Change in net recharge $\frac{\partial x_T}{\partial W_{net}} = -\frac{x_n M}{2W_{net}\sqrt{1-M}} + \frac{x_n\left(1-\sqrt{1-M}\right)}{2W_{net}\sqrt{1-M}}$(A.12)
Change in seaward groundwater flux $\frac{\partial x_T}{\partial q_b} = \frac{1}{W_{net}}\left(1 - \frac{1}{\sqrt{1-M}}\right)$(A.13)	

Table A3
SWI vulnerability equations for confined aquifers (after Werner et al., 2012).

Current conditions	
$h_f = \frac{q_0x - W_{net}x^2/2}{Kh_0} + \delta z_0 - \frac{\delta h_0}{2}$	where $x \geq x_T$(A.14)
$h_f = \sqrt{\left(2q_0x - W_{net}x^2\right)\frac{\delta}{K} + \delta z_0 - \delta h_0}$	where $x \geq x_T$(A.15)
$q_0 = \frac{K}{2\delta x_b}\left(2\delta h_b h_0 + \left(\delta h_0\right)^2 - 2\delta^2 z_0 h_0\right) + \frac{W_{net}x_b}{2}$	where $x_b \geq x_T$ and $h_b > \delta(z_0 - h_0)$(A.16)
$q_0 = \frac{K}{2\delta x_b}\left(h_b + \delta h_0 - \delta z_0\right)^2 + \frac{W_{net}x_b}{2}$	where $x_b \geq x_T$ and $h_b > \delta(z_0 - h_0)$(A.17)
$x_T = \frac{\delta K h_0^2}{2q_0}$	where $W_{net} = 0$(A.18)
$V_{sw} = \frac{n\delta K{h_0}^3}{6q_0}$	where $W_{net} = 0$(A.19)
Future stresses	
Flux-controlled setting	Head-controlled setting
Sea level rise, where $W_{net} = 0$ $\frac{\partial x_T}{\partial z_0} = 0$ (A.20)	Sea level rise, where $W_{net} = 0$ $\frac{\partial x_T}{\partial z_0} = \frac{\delta\left(1+\delta\right)K^2h_0^3}{2q_0^2x_f}$(A.21)
Changes in seaward groundwater flux, where $W_{net} = 0$ $\frac{\partial x_T}{\partial q_b} = \frac{\partial x_T}{\partial q_0} = -\frac{\delta K h_0^2}{2q_0^2}$(A.22)	

Table A4
SWI vulnerability equations for freshwater lenses (after Morgan and Werner, 2014).

Current conditions	
$B = x_\mathrm{B}\sqrt{\frac{W_\mathrm{net}}{K}(1+\frac{1}{\delta})}$(A.23)	
$V_\mathrm{fw} = n\int_{0}^{2x_\mathrm{B}}(h_\mathrm{f}-z)dx = n\frac{\pi}{2}x_\mathrm{B}B$(A.24)	
Future stresses	
Flux-controlled setting	Head-controlled setting
Sea level rise $\frac{\partial B}{\partial z_0} = 0$(A.25)	Sea level rise with land surface inundation $\frac{\partial x_\mathrm{T}}{\partial z_0} = \frac{\delta(1+\delta)K^2h_0^3}{2q_0^2x_\mathrm{f}}$(A.27)
$\frac{\partial V_\mathrm{fw}}{\partial z_0} = 0$(A.26)	$\frac{\partial V_\mathrm{fw}}{\partial z_0} = -n\frac{h_\mathrm{B}}{x_\mathrm{B}}\frac{1+\delta}{\delta}\sqrt{\frac{1+\delta}{\delta}\frac{K}{W}}\left(-\sqrt{2x_\mathrm{B}x_n-x_\mathrm{B}^2}+\frac{(x_\mathrm{B}-x_n)x_\mathrm{B}}{\sqrt{2x_\mathrm{B}x_n-x_\mathrm{B}^2}}+2x_n\arccos(1-\frac{x_\mathrm{B}}{x_n})-\frac{x_\mathrm{B}}{\sqrt{1-(1-x_\mathrm{B}/x_n)^2}}\right)$ (A.28)
Recharge change $\frac{\partial B}{\partial W} = \frac{B}{2W}$(A.29)	Recharge change $\frac{\partial B}{\partial W} = 0$(A.31)
$\frac{\partial V_\mathrm{fw}}{\partial W} = \frac{V_\mathrm{fw}}{2W}$(A.30)	$\frac{dV_\mathrm{fw}}{dW} = n\sqrt{\frac{1+\delta}{\delta WK}}\left\{\begin{matrix}\frac{1}{2}\left((x_\mathrm{B}-x_n)\sqrt{2x_\mathrm{B}x_n-{x_\mathrm{B}}^2}+{x_n}^2(\arccos(1-\frac{x_\mathrm{B}}{x_n}))\right)\\+\left((\frac{-x_\mathrm{B}}{2}+x_n)\sqrt{2x_\mathrm{B}x_n-x_\mathrm{B}^2}+\frac{1}{2}\frac{(x_\mathrm{B}-x_n)(x_\mathrm{B}^2-2x_\mathrm{B}x_n}{\sqrt{2x_\mathrm{B}x_n-x_\mathrm{B}^2}}\right)\\+\left(\left(x_\mathrm{B}x_n-2x_n^2\right)(\arccos(1-\frac{x_\mathrm{B}}{x_n})+\frac{x_\mathrm{B}x_n-x_\mathrm{B}^2/2}{\sqrt{1-(-1+x_\mathrm{B}/x_n)^2}}\right)\end{matrix}\right\}$(A.32)

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.ejrh.2015.10.005.

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