CSIRO PUBLISHING

Soil Research, 2021, **59**, 44–59
https://doi.org/10.1071/SR19375

Salinity risk assessment of an irrigation development using treated coal seam gas water in the Condamine River catchment

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Abstract. All irrigation developments inherently carry a salinity risk, due to an unavoidable change in the water and salt balance. The time frame in which either land or water salinity will develop is driven by the ability of the landscape to absorb the change of water and salt supplied. Factors that influence this are landscape attributes, such as the size of the unsaturated zone and its properties (permeability and drainage), management considerations (land-use changes, water application rate and crop water use) and climate variability (temperature and rainfall). This study assessed the risk of secondary salinity expression occurring in an irrigation area in the Condamine-Balonne catchment in southern inland Queensland, Australia. The objectives were to (1) define the depth, size and properties of the unsaturated zone and regolith, (2) define deep drainage rates for past, present and future land uses and (3) assess this information to calculate the risk that groundwater table rise may result in surface salinity expression. Data collected during field investigations was used to conceptualise the regolith architecture, undertake hydrogeological modelling, estimate the available moisture storage capacity of the unsaturated zone and model paddock deep drainage characteristics. The work identified that irrigation-induced deep drainage had started to mobilise salt stores in the unsaturated zone. It also identified connectivity between land management and salt discharges into the Condamine River. As the water supply for the scheme is scheduled to continue until 2030, there is a clear risk of the unsaturated zone moisture storage capacity being exceeded, leading to both land and surface water salt expressions.

Additional keywords: electrical resistivity tomography (ERT), hydrogeology, Hydrus, water balance.

Received 10 December 2019, accepted 22 July 2020, published online 24 August 2020

Introduction

The potential for irrigation developments to cause secondary salinity is not a new phenomenon, as about one-third of the world's irrigated lands have reduced productivity due to waterlogging and salinisation (Fernández Cirelli *et al*. 2009). Previous studies have clearly defined the underlying principles relevant to all irrigation enterprises, regardless of the irrigation method, crop type, geographic location, soil type, source of irrigation water or irrigation water quality. These include that salt moves with water, irrigation water is more saline than rainfall, deep drainage is inevitable and necessary, and irrigation causes a change in the soil water balance (Shaw and Yule 1978; Ayers *et al*. 1985; SalCon 1997; Wichelns and Oster 2006; Charman and Murphy 2007; Duncan *et al*. 2008; Tolmie *et al*. 2011; Biggs *et al*. 2013; Silburn *et al*. 2013). Based on these principles, any irrigation salinity risk assessment can be focused to capture changes that fuel potential water movement in the landscape. Ultimately it is how the potential change impacts the environment that drives the risk assessment, as this change can be expressed in many forms.

Holistically, there is a simple and underlying conceptual model that applies to salinity risk; that of a simple bucket representing the unsaturated zone. When the rate of water input exceeds the storage capacity of the unsaturated zone, increased deep drainage or recharge will result in groundwater mounding (i.e. causing the depth of groundwater to rise) (Stevens *et al*. 2003). The unsaturated zone extends from the soil surface down to the saturated zone and importantly can include a zone(s) of reduced permeability. This may not necessarily lead to the development of a perched aquifer or water table, although it has the potential to if the overlying unsaturated zone is filled at a rate faster than drainage of the regolith below. Although the use of a bucket-type conceptual model is simplistic, it provides a method by which worst case scenarios can easily be calculated while providing the capacity to include further complexity, e.g. lateral losses. It also directs the data collection process to address both spatial and temporal changes within the unsaturated zone for the study area.

Salt will move through the landscape with water; therefore, the movement and distribution of salt is determined by the hydrology of the landscape (Shaw 1994). Changing land use from perennial, deep-rooted native vegetation to shallower-rooted annual crops or pastures alters the soil water balance (Walker *et al*. 1999). This change causes a decrease in transpiration and an increase in water draining below the root zone (i.e. deep drainage). Irrigated farming systems further alter the pattern of soil water use, storage and throughflow compared with areas under native vegetation. This leads to greater soil water balance perturbations, with irrigated systems experiencing increased frequency and magnitude of deep drainage episodes. Deep drainage losses under irrigated agriculture are inevitable and, in fact, essential to prevent salts from building up in the cropping zone of the soil profile. Even if very low salinity water (<0.95 dS m^{-1}) (ANZG 2018) is used for irrigation, there is often still a landscape salinity risk in all irrigation schemes. This is because in many arid and semiarid landscapes, salts naturally accumulate in the soil profile (unsaturated and saturated zones) over long time periods (Duncan *et al*. 2008). Vervoort *et al*. (2003) stated that the leaching fraction defines the amount of water needed to cause salts to move lower in the soil profile and, depending on the water quality, this generally represents 10–20% of the applied water. Thus, while the application of additional water to a landscape will add salt, it will also mobilise existing salt when deep drainage occurs.

Since commercial coal seam gas (CSG) production commenced in the Surat Basin in Queensland in early 2006 (Helmuth 2008), the industry has undergone a rapid expansion, with numerous gas-producing wells coming online. The CSG is a natural gas, consisting primarily of methane, which is adsorbed by coal. The CSG is released by constructing wells to dewater coal seams and reduce the hydraulic pressure that keeps the gas *in situ* (Averina *et al*. 2008). It is a process that results in vast amounts of saline water being brought to the surface. The Queensland Government's *Coal Seam Gas Water Management Policy 2012* (EHP 2012) encourages CSG producers to use their CSG water in a way that protects the environment and maximises its productive use as a valuable resource (referred to as beneficial use). For example, following treatment, CSG water can be used for irrigation. Within the Queensland Murray–Darling Basin, the Queensland Government is obligated to assess the salinity risk associated with new irrigation schemes developed post-1999. These obligations are outlined by the Basin Salinity Management Strategy 2030 (MDBMC 2015) and its predecessors. The quality of this recycled CSG water is regulated by guidelines and legislation; however, the regulation of any impact on the landscape to which it is applied is not as straightforward.

Salinity risk is an estimate of the likelihood of secondary salinity expression (in any form) in a spatially defined area within a defined time-span. Risk is a function of human activities, interacting with the inherent biophysical hazard (Grundy *et al*. 2007; Searle *et al*. 2007) and indicates the probability that certain actions, including land-use management, will lead to salinity being expressed (Moss *et al*. 2001). Therefore, a salinity risk assessment should aim to capture the influences of both historic and current land-use management practices upon the inherent landscape properties. The objectives of this study were to (1) define the depth, size and properties of the unsaturated zone and regolith, (2) define deep drainage rates for past, present and future land uses and (3) assess the information to calculate the risk of secondary salinity expression occurring in an irrigation area in the Condamine-Balonne catchment in southern inland Queensland, Australia. This paper conveys the key findings from a detailed three-year project on salinity risk assessment. Further details can be found in Crawford *et al*. (2019).

Materials and methods

Study area

The study area covers 4500 ha, and is located in the Condamine-Balonne catchment of the Murray–Darling Basin in southern inland Queensland (Fig. 1). The study area is roughly bounded by the towns of Miles, Chinchilla and Condamine (26.9283°S, 150.1319°E) and has an average annual rainfall of 585 mm (Welsh *et al*. 2014). Average monthly rainfall data for Miles show that the highest rainfall occurs in January, December and February (96.1, 91.2 and 73.6 mm respectively), highlighting a summer-dominant rainfall pattern (Bureau of Meteorology 2011). Mean maximum temperature in the hottest month of the year (January) is 33.2°C which coincides with a mean daily evaporation rate of 7.3 mm. The lowest average annual temperature (11.7°C) occurs in July. The main land uses for the area are dryland and irrigated cropping, cattle grazing and native vegetation.

Soil and geological features

The study area is part of a relict floodplain that is incised by the active drainage network of the Condamine River and its tributaries. It is primarily covered by Quaternary sediments with limited small outcrops of the underlying late Jurassic to early Cretaceous Kumbarilla Beds. The latter is further divided into several subunits with the uppermost unit of the Gubberamunda Sandstone Formation (GSF) overlaying the Westbourne Formation. The GSF typically comprises permeable medium and coarse-grained, poorly cemented quartzose sandstone with lesser amounts of conglomerate and fine-grained clastic sediments. It is thought to have been deposited in a high-energy, shallow water environment. The Westbourne Formation typically comprises less permeable interbedded shales and siltstones and very fine-grained quartzose sandstone.

Soils and land resources of the study area are described and mapped at a scale of 1 : 250 000 by Maher (1996). Detailed 1 : 50 000 scale soils mapping is available for part of the study area associated with the original 'Fairymeadow Road Irrigation Pipeline Scheme' Beneficial Use Approval application made to the Queensland Government in 2013. The survey carried out by Thompson and Shields (2013) covered 13 342 ha, identifying eight soil types. Most of the soils in the study area are described as Black and Grey Vertosols (Isbell and National Committee on Soil and Terrain 2016) (Vertisols by the World Reference Base

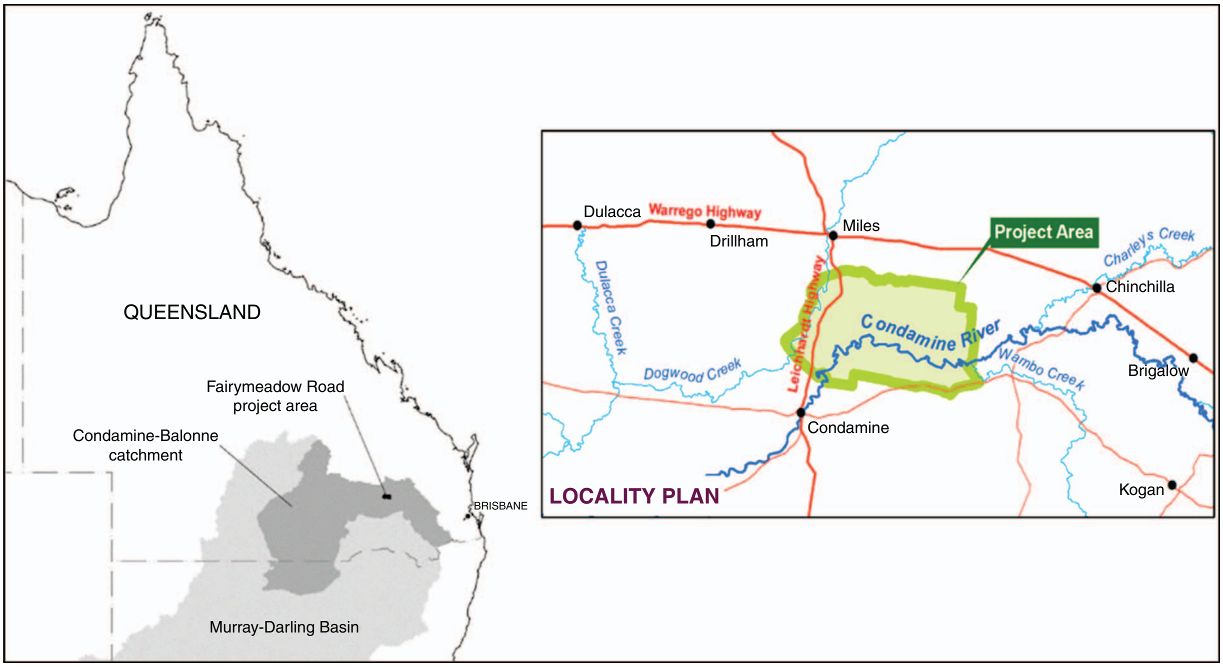

Fig. 1. Location of the Fairymeadow Road Irrigation Pipeline Scheme within the Condamine-Balonne catchment in the Queensland Murray–Darling Basin.

classification (IUSS 2014)), with some texture contrast soils (Grey Sodosols and Grey Chromosols), Solonetz and Luvisols (WRB).

Land-use change analysis

In order to understand the contribution of historical land-use changes to deep drainage, an historical land-use record was compiled in decadal intervals from 1956 to 2018 using aerial photography and satellite imagery. Land use was divided into categories of grazing, dryland cropping, irrigated cropping, native vegetation (thinned and natural) and mining. Current and historical land use was used to estimate the spatial pattern of groundwater recharge and the proportional role of historical vs modern deep drainage.

Assessing the unsaturated zone

Electrical resistivity tomography (ERT) is a geophysical technique for imaging subsurface structures from measurements made at the surface. Within the study area, eight transects were surveyed using ERT (Fig. 2, Table 1) to determine depth of the unsaturated zone and the presence of hydrological impediment zones which can influence the fate of deep drainage. The ERT surveys were undertaken using an ABEM Terrameter LS resistivity meter in various configurations. All transects except #3 used a 2 × 32 (set of two electrode cables with 32 take-outs each) increasing spread with the Gradient8 protocol and a pin spacing of 2.5 m. Transect #3 used the above spread and protocol with a pin spacing of 5 m. The overall length of each transect was achieved by using the extension (roll-along) data acquisition technique (Loke 2000). All data were processed to produce the associated 2D inversion models and transect pseudosections using the RES2DINV© Geotomo Software (2000–2010), based on a smoothness-constrained Gauss–Newton least-squares inversion technique (Sasaki 1989). For the sake of brevity, only four representative transects will be discussed in depth in the Results and Discussion.

Soil sampling

Data from deep coring (six cores ranging within 4.6–14.6 m depth) and historical groundwater bore logs was used to validate the ERT imagery and determine the size and nature of the unsaturated zone. A total of 92 new soil profiles were described in the study area using the guidelines and terminology of National Committee on Soil and Terrain (2009), all located north of the Condamine River. These profiles aided in the interpretation of the ERT transects (47), chloride mass balance determination (27) and soil classification and soil map checking (18). Chemical analysis was undertaken on 85 of these profiles. The analysis included pH (1 : 5, soil : water), electrical conductivity $(EC)_{1:5}$ (1 : 5, soil : water), Cl (automated ferricyanide method), water soluble nitrate (automated colourimetry), aqueous 1 M ammonium chloride (pH 7) extractable cations, alcoholic 1 M ammonium chloride (pH 8.5) extractable cation exchange capacity – the methods used are described in Rayment and Lyons (2011) (method codes: 4A1, 3A1, 5A2, 7B1, 15A1 and 15C1 respectively) – and particle size analysis (sieve and hydrometer method) (Standards Association of Australia 1977; Thorburn and Shaw 1987).

Surface and groundwater sampling

In February 2016, an investigation was undertaken by boat along the length of the Condamine River contained within the

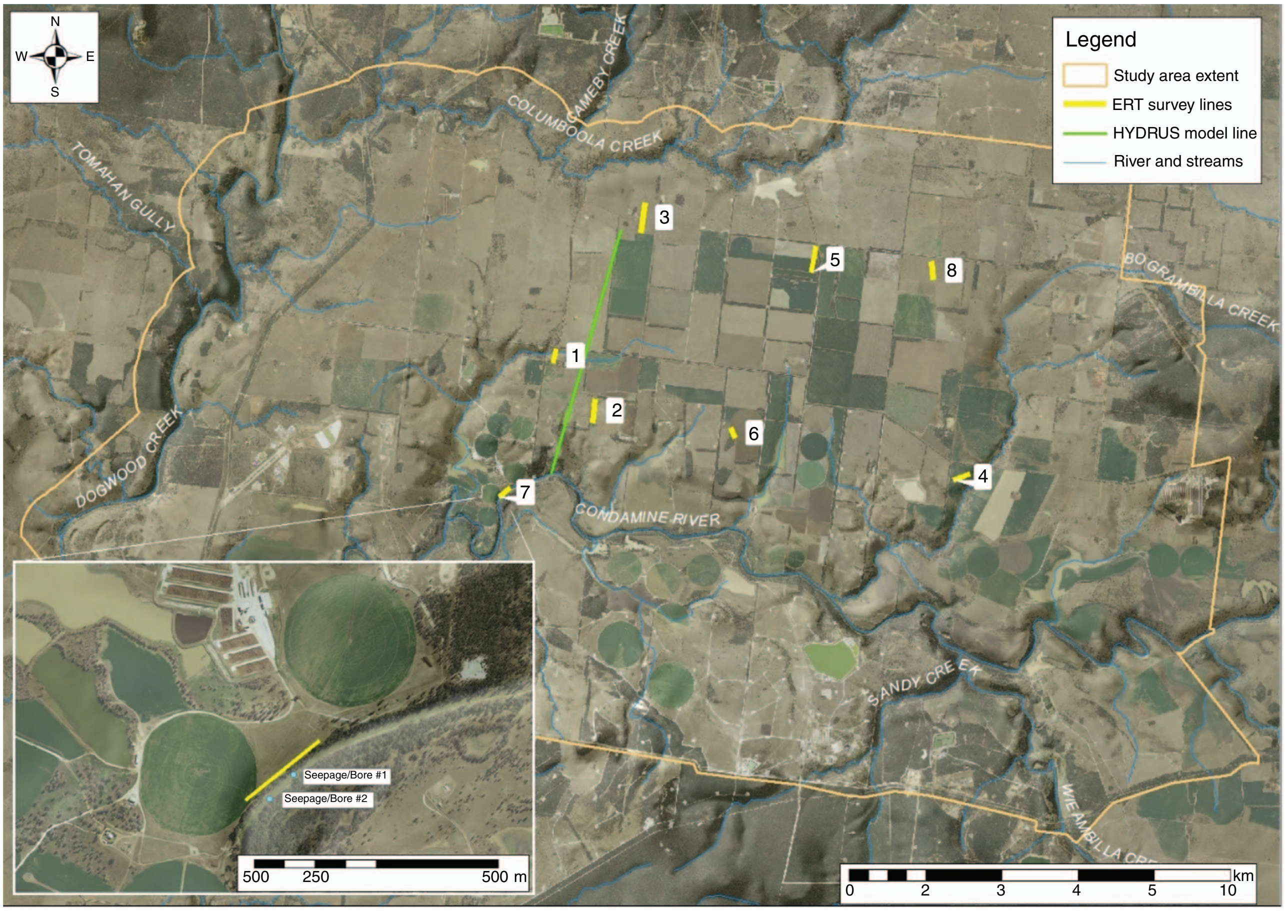

Fig. 2. Location of ERT transects, two of the identified seepage sites that were sampled and the cross-section used for HYDRUS-2D modelling. The underlining image (Queensland Government, digital ortho-rectified aerial photograph, 40 cm mosaic, dated 2013) is a reflection of the land use at the beginning of the irrigation scheme.

Table 1. Details of ERT transects carried out in the study area

Transect number	Land use	Transect length (m) and direction	Transect depth (m)	No. of deep calibration cores	No. of shallow calibration cores	Start coords[A]	Finish coords[A]
1	Pasture	397.5, N–S	25	3	1	226451, 7035159	226365, 7034768
2	Pivot	650, N–S	35	4	2	227477, 7033836	227408, 7033207
3	Pasture	800, N–S	70	3	4	228786, 7038981	228682, 7038186
4	Dryland cropping	480, SW–NE	30	2	1	236864, 7031710	237313, 7031864
5	Pivot	720, N–S	30	2	4	233266, 7037850	233152, 7037144
6	Pivot	320, NW–SE	30	–	5	231029, 7033085	231165, 7032796
7	Pivot	397.5, SW–NW	30	3	–	224942, 7031262	225247, 7031507
8	Pasture	477.5, S–N	35	3	–	236345, 7036958	236301, 7037437

[A]Coordinates are listed as Eastings, Northings (all zone 56).

study area (14.5 km) to identify potential seepage zones. Seepage zones were identifiable by the presence of water percolating from the river bank, above the standing river water level. Over time, this has led to an accumulation of iron by-products due to oxidation reactions. In particular, rust-red coloured zones (if ferric iron precipitates), pale green coloured zones (if ferrous iron precipitates), black sludge (similar to monosulfidic black ooze (Sullivan *et al.* 2018)) and the presence of iron bacteria (oily or shiny film coating the iron precipitates). These by-products will form a convex delta structure and have an increased bank vegetation presence. Five seepage zones were located within a 4.4-km stretch in the western section of the Condamine River. Photos, locations and site descriptions were recorded at each of the five seepage zones. Subsequent sampling and field visits focused only on two of these sites (seepage #1 and #2 in Fig. 2) due to ease of access and the volume of water available for sampling. Seepage water samples were collected and submitted for laboratory analysis in March 2016, March 2017, May 2017, March 2018 and March 2019, along with surface water samples from a nearby centre pivot irrigator and CSG water storage dam. Laboratory analysis included pH and EC (automated titration with temperature compensation), alkalinity (pH end point automated titration), major cations (inductively coupled plasma optical emission spectroscopy) and anions (ion chromatography), metals (inductively coupled plasma mass spectrometry), nutrients (segmented flow analysis) and stable isotopes. Methods used are as described in APHA and AWWA (2012) (method codes 4500-H B, 2510, 2320 B, 3120 B, 4110 B and 3125B; nutrients: 4500-NH_3, 4500-NO_3, 4500-P G, 4500-N_{org} D and 4500-P B) respectively. The method for stable isotope analyses was performed as per Thermo-Fisher Scientific (2010), following protocols and calculations detailed in Carter and Barwick (2011).

In May 2017, an ERT transect was carried out (transect #7) parallel to seepage #1 and #2 to identify possible flow paths of the seepage water. Two likely paths were identified and subsequently, and two groundwater bores were installed in late May 2017. The two bores were constructed on top of the northern bank of the Condamine River in close proximity to the two seepage sites. On 21 July 2017, CTD-Diver loggers were installed in the bores to monitor groundwater level and EC, taking daily readings at 0900 hours. A Diver-Baro logger (records atmospheric pressure) was also installed so that the water level data from the loggers could be adjusted for atmospheric pressure fluctuations. Both bores were triple purged before sample collection in accordance with Sundaram *et al*. (2009) and groundwater samples were collected on 21 July 2017 (before loggers were installed), 21 March 2018 and 14 March 2019.

Hydrogeological modelling

A single north–south cross-section, supported by ERT surveys and sufficient soil and land-use information was selected for the hydrogeological model analysis (Fig. 2). The cross-section aligned with cross-section III of Dafny (2015). Water flow across the unsaturated zone under the selected cross-section was modelled using HYDRUS-2D software (Šimůnek *et al*. 2016). For the modelling exercise, the following simplifications and estimations were made. Only areas where the semi-impermeable surface dipped south towards the Condamine River were modelled. Deep drainage rates varied over time, in accordance with land uses. Average deep drainage rates were allocated to each land use, based on Yee Yet and Silburn (2003). Deep drainage was assumed to occur under a 1.5-m Vertosol, precluding the need to model the topmost Vertosol layer or crack-flow mechanisms.

The subsurface layer was assumed to comprise a clay loam[1] with a typical hydraulic conductivity of 0.0624 m day^{-1} (or 2.6 mm h^{-1}). The Qa–JKk[2] contact zone was set with a hydraulic conductivity of 1 m day^{-1} (365 m $year^{-1}$ or 41 mm h^{-1}) and Kx : Kz anisotropy of 10 : 1. The Jurassic sediments were set with a very low hydraulic conductivity (0.001 m $year^{-1}$ or 3×10^{-11} m s^{-1}) that typifies impermeable sandstones and an anisotropy of 30 : 1. Initial conditions were a dry unsaturated zone, following decades of nearly no recharge under natural woodlands. The model was run for 100 years, beginning on 1 January 1930. It included a 30-year period of reset and a ‘prediction’ of 13 years. The model was informed that the area was cleared for grazing in the 1960s, allowing minimum recharge. Thereafter, varying areas were set to dryland or irrigated cropping.

Results

Land-use change analysis

Historical land-use maps of the study area were produced for 1956, 1969, 1974, 1982, 1990, 2001, 2006, 2013 and 2018[3] using aerial photography and satellite images. As land use changed, the water balance and deep drainage rates also changed. These historical land-use maps help understand the potential salinity risk of increased water application associated with intensifying land-use (i.e. changing from natural vegetation to dryland cropping to irrigated cropping results in increased deep drainage rates). Land use in 1956 was predominately grazing (70%) with most of the remaining area under wooded vegetation (28%) (Table 2). In 1969, the area of grazing had peaked (79%), wooded vegetation area decreased to 19% and dryland cropping area had increased to just over 1%. In 1982, the area of dryland cropping had increased to 9%, then increased again to 14% in 1990. Wooded vegetation had reduced to 16% in 1990, while small

Table 2. Summary of land-use change from 1956 to 2015 in the study area

Land-use category	Change in land use, 1956–2015 (%)	1956 (% total area)	2013 (% total area)	2015 (% total area)
Grazing	–14	70	57	56
Dryland cropping	+17	1	21	18
Irrigated cropping	+9	0	6	9
Total woody vegetation	–15	28	13	13
Mining	+1	0	1	1
Water	+2	1	3	3

[1]Clay loam was used to encompass the sand/clay sequence typically seen in these soils
[2]Qa–JKk = Quaternary alluvium/Kumbarilla Beds
[3]Dates were restricted to those years when aerial photography was performed across the study area

areas (0.5%) of irrigated cropping were evident. In 2013, dryland cropping areas had increased to 21%, irrigated cropping to 6%, while grazing area had reduced to 57% and wooded vegetation to 13%. The introduction of the Fairymeadow Road Irrigation Pipeline scheme in 2014 resulted in a 67% increase (6% to 9%) in irrigated cropping area from 2013 to 2015. Overall changes in land use during 1956–2015 are summarised in Table 2. During this 59-year period, grazing reduced by 14%, wooded vegetation cover reduced by 15%, dryland cropping area increased by 17% and irrigated cropping area increased by 9%.

Assessing the unsaturated zone

Subsurface resistivity data

The ERT transect values were interpreted by dividing the images into subgroups, each of which was calibrated using soil cores and drill logs. In Fig. 3*a*, the dark blue areas represent high EC/low resistivity; generally associated with clay, high moisture or saline material. The green areas are a mid-resistivity zone, indicating a weathered substrate, sandy clay or a lower moisture status. The red areas (low EC/high resistivity) are typically associated with sandy or dry material. Transect #3 (Table 1, Fig. 2), located in the central-northern part of the study area, can be clearly delineated into three zones (Fig. 3*b*). First, the 0–10 m depth zone represents the clay soil profile with low resistivity (high EC). The second zone (depth 10–40 m) is a highly resistive area that represents the GSF. The third zone (depth 40–70 m) situated underneath the GSF was more conductive and therefore thought to be the Westbourne Formation. Transect #3 also identified an apparent fault at 450–500 m (dark blue zone) starting approximately at 50-m depth, which was later confirmed by secondary assessment (Cranfield 2017). Although lithological zones can still be identified in ERT transects #5 and #8 (located in the north-eastern part of the study area), the underlying boundary was not as easily discriminated because the regolith material was all highly conductive (Figs 4*a*, *b* and 5*a*, *b*). Historical drainage features were identifiable in both transects, i.e. a wide relict drainage depression 320–420 m along transect #8 (Fig. 4*a*, *b*); and a narrow relict drainage depression 420–520 m along transect #5 (Fig. 5*a*, *b*).

The ERT transect #2 is an example of the highly resistive GSF overlying the conductive Westbourne Formation (Fig. 6*a*, *b*). It also clearly shows the separation of the surface soil profile (highly conductive blue zone) and the two underlying lithologies (highly resistive red zone and mildly conductive green zone). Subsequent deep cores confirmed the presence of a coarse-grained lithology (longitudinal distance north to south 0–400 m) and fine-grained lithology (longitudinal distance north to south 400–650 m) (Fig. 6*b*). The coarse-grained lithology is representative of the materials found within the GSF. The fine-grained lithology was either a highly weathered pallid (white) zone of the GSF or Westbourne Formation exposed by geological erosion processes.

The study area can also be divided into two broad sections. The central-northern section (where ERT surveys #1 to #3

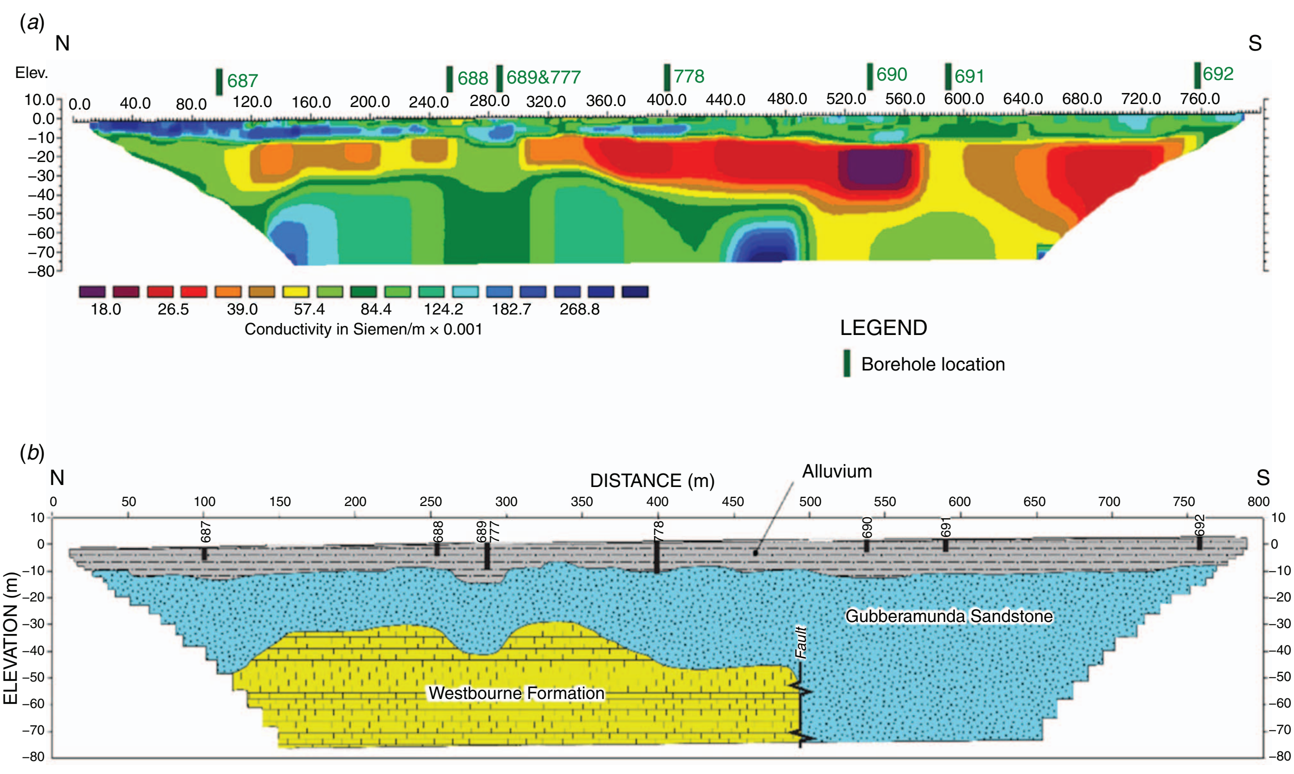

Fig. 3. (*a*, *b*) Pasture ERT survey (transect #3) measuring to a depth of 70 m. Note the inferred fault at 450–500 m (dark blue zone). Interpretation showing the soil profile depth, geological classifications and identified fault.

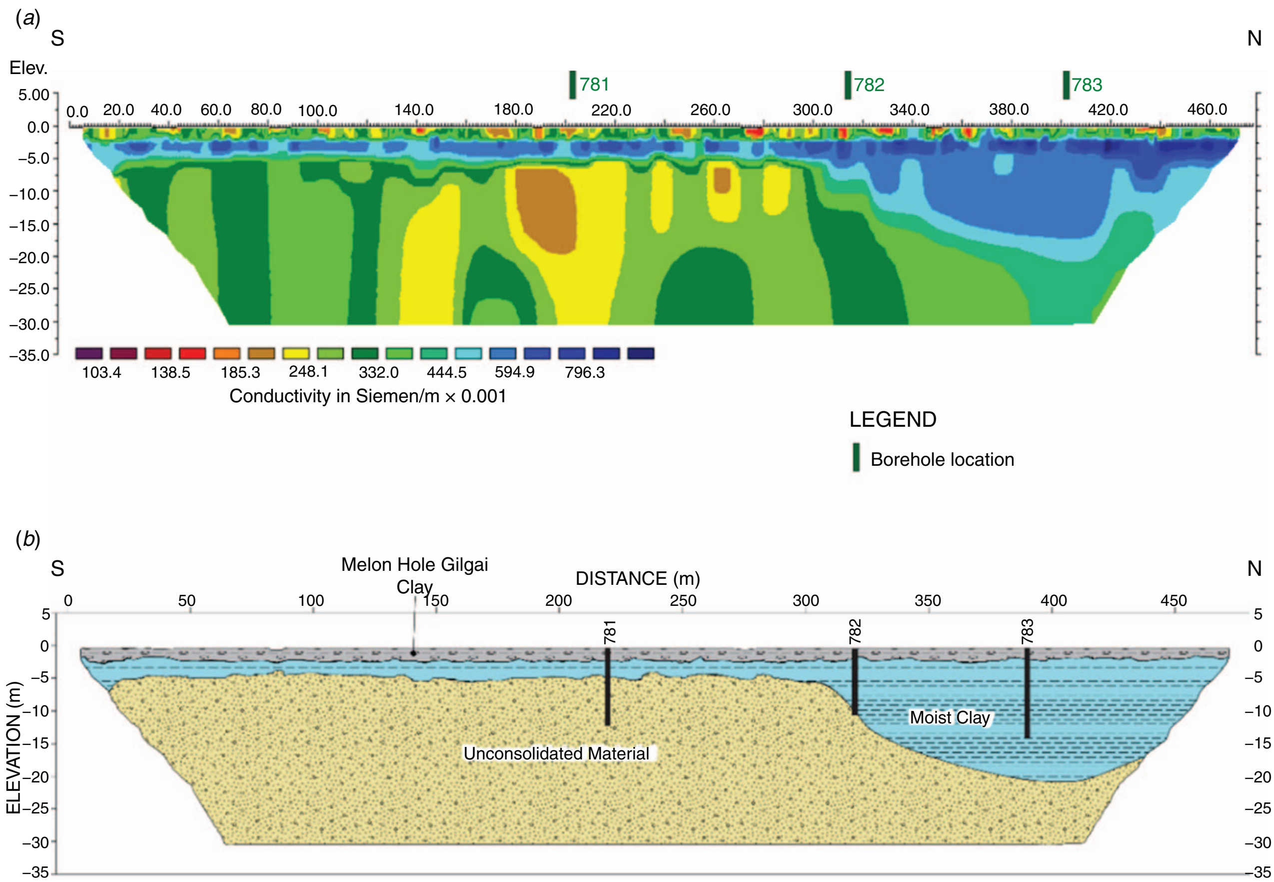

Fig. 4. (*a*, *b*) Pasture ERT survey (transect #8) measuring to a depth of 35 m. Note the highly conductive values (green and blue zones) associated with moist clay soils underneath pasture and a large relict drainage depression (320–420 m). Interpretation showing the soil profile depth and lithological characteristics.

were situated) depicts the highly resistive GSF overlying the conductive Westbourne Formation. In the north-eastern section of the study area (ERT surveys #5 and #8), conductive material (interpreted as unconsolidated material) overlying a very conductive parent material of mudstone or siltstone was evident. A potential relict flow path of the Condamine River (or other large drainage system) was identified in both of these transects. Unconsolidated material containing pockets of identified sandy alluvium in both #5 and #8 ERT transects indicates that the previous depositional system was of higher energy than the current system which consists of finer material.

Six deep cores were extracted within the elevated old alluvial plain, which is where the more recent irrigation developments have occurred. The results from these cores identified the depth to hydrological impediment to be 7–13 m below ground level. The presence of high levels of calcium carbonate concretions or nodules and an alkaline pH at 7–10 m suggests evidence of an old or extinct watertable. Further evidence includes a pH inversion–reversion sequence occurring at similar depths across all cores, i.e. the pH was alkaline at the surface, became acid (~4) in the subsoil (1 m), before returning to alkaline pH below 7 m.

Surface and groundwater data

Water quality results from the seepages, groundwater bores and surface water samples are listed in Table 3 and presented in Figs 7 and 8. Results were collected over a three-year period, from February 2016 to March 2019. Seepage water and groundwater clearly differed from the Condamine River water and other surface water. The seepages and their adjacent bores were saline and dominated by sodium chloride, with a sodium adsorption ratio above 7. The EC, nitrate, sodium and chloride results of the seepage water and groundwater were above applicable water quality objectives (McNeil *et al*. 2018; Newham *et al*. 2018). The Condamine River samples were non-saline and non-sodic, with low levels of cations, anions and nutrients. The samples collected from a nearby dam and centre pivot irrigator were also non-saline and non-sodic.

Results of surface water and groundwater analysis are shown in a Piper diagram (Fig. 7), with different water types grouped (indicated by red circles). The mix of ions identified in the groundwater bores was generally similar to those in their associated seepage, but their quality differed slightly, but consistently; suggesting that they derived from slightly different sources. It is likely that the difference in

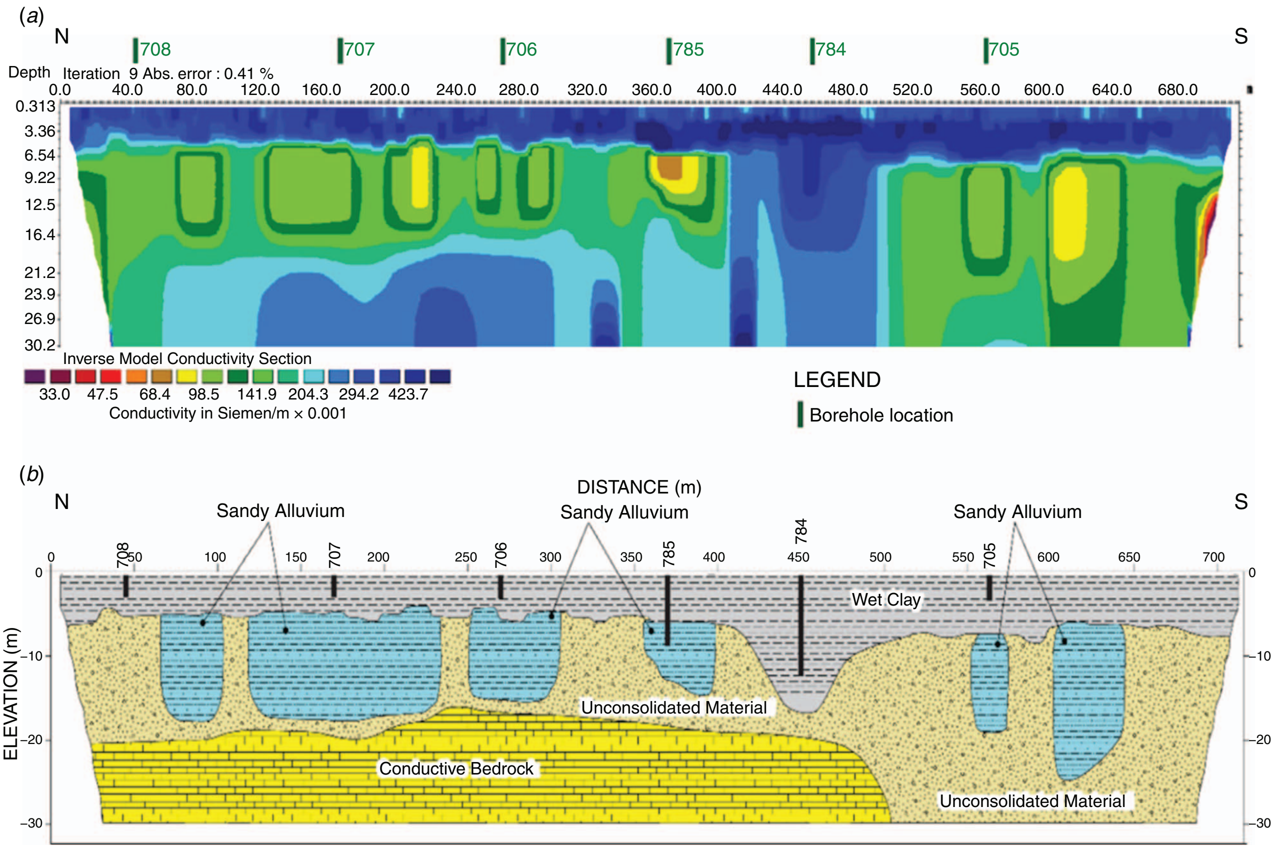

Fig. 5. (*a*, *b*) Pivot ERT survey (transect #5) measuring to a depth of 30 m. Note the highly conductive values (blue and green zones) associated with moist clay soils and a deep relict drainage depression (420–520 m). Interpretation of ERT transect #5 showing soil profile depth and lithological characteristics.

water quality between the two bores was due to the proximity of the nearby centre pivot; however, quantifying the degree of influence is difficult from the available dataset.

Stable isotopes (Fig. 8) and pesticides were also analysed from the seepages, groundwater bores and Condamine River water. The stable isotope data suggest relatively rapid rainfall recharge, and interaction between the seepages and groundwater in the bank area as a result of river flow events. This is demonstrated by the closeness of fit to the local meteoric water line (LMWL). If influenced by artesian water, the isotope results would reflect a greater separation below the LMWL. When sampled in March 2018, 22 pesticides (all below ANZECC guideline values (NHMRC and NRMMC 2011)) were detected in the Condamine River water (Crawford *et al*. 2019). Of these 22 pesticides, 15 were found in the seepages. Only two (metabolites of atrazine) were found in the bores. All 22 pesticides have previously been found in the river (Standley *et al*. 2017); the values detected in 2018 were lower than those found previously in the Condamine River (J Standley, pers. comm.).

Hydrogeological modelling

The base-case HYDRUS-2D model successfully simulated deep drainage moving through the unsaturated zone and a development of a saturated, perched water lens near the Qa–JKk contact (Fig. 9). At the end of the simulation (t = 2030), the saturated zone extended almost throughout the entire section and reached a thickness of 4 m above the Qa–JKk contact and into the clay loam subsoil. Although there was no seepage into the Condamine River, it was within the river seepage risk zone.

Following the base-case model assessment, it was decided to remove the JKk sediments from the model domain, thus treating the Qa–JKk contact as an impermeable boundary. The revised model successfully simulated deep drainage percolation through the unsaturated zone, seepage and the development of a saturated, perched water lens near the Qa–JKk contact (which thickened with time as a result of increased deep drainage). Under this scenario, seepage into the Condamine River occurred as early as 1966, and reached ~0.08 L min^{-1} in the mid-2010s. At the end of the simulation (t = 2030), the water mound in the saturated zone extended throughout the entire section, reached a maximum thickness of 5.4 m above the Qa–JKk contact and seepage reached >0.1 L s^{-1}.

Discussion

At the commencement of the irrigation scheme in 2014, the conceptual model suggested that irrigation of the proposed

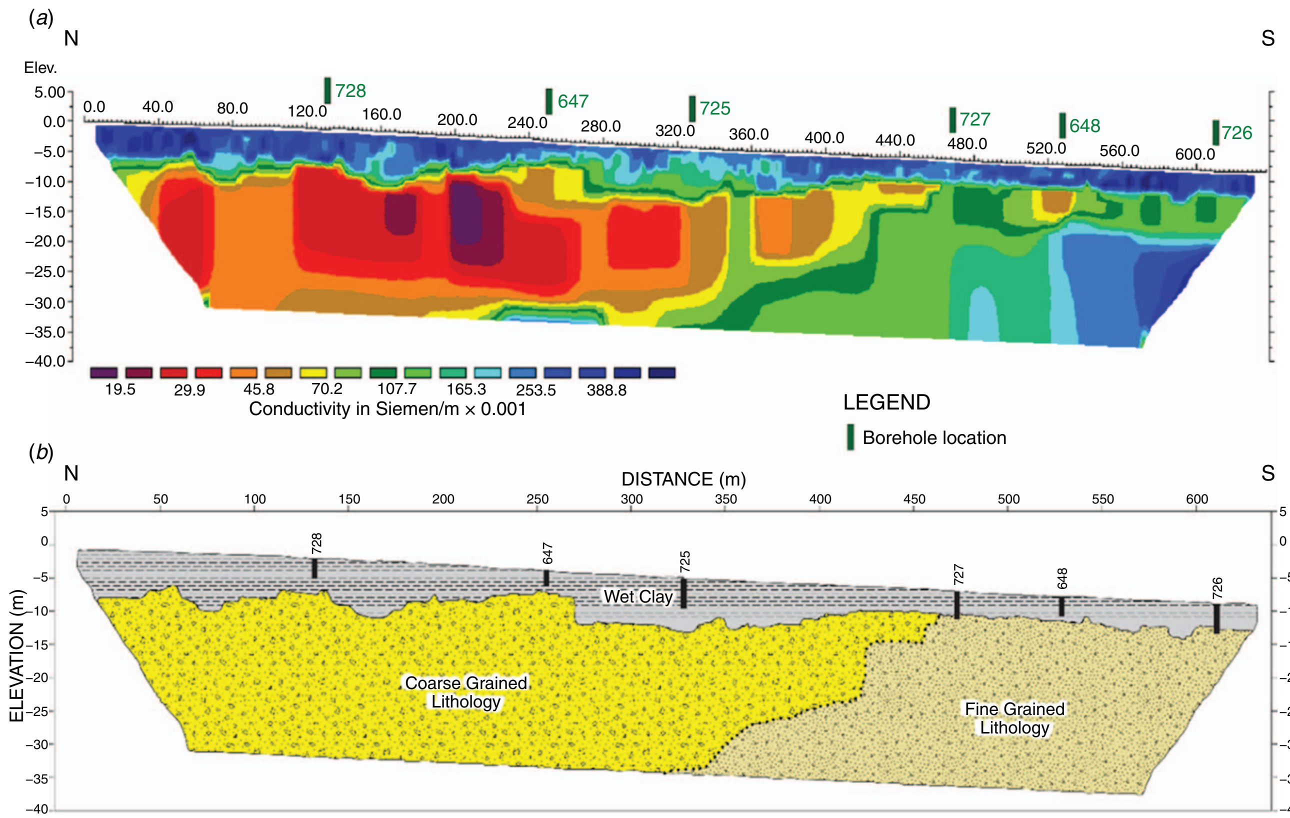

Fig. 6. (*a*, *b*) Dryland cropping ERT survey (transect #2) measuring to a depth of 35 m. Interpretation showing depth of soil profile and lithological characteristics.

Table 3. Laboratory results for the seepages, bores, Condamine River, pivot and dam, from Feb. 2016 to Mar. 2019

The first number is the average result and the numbers in brackets are the range. Applicable surface water and groundwater quality objectives are also shown Number of samples analysed for each water type were one (pivot and dam), three (bores #1 and #2) and five (river, seepage #1 and #2)

	#1		#2		Surface water			Surface water quality objectives	Groundwater quality objectives[C]
	Seepage	Bore	Seepage	Bore	River	Pivot	Dam		
pH	7.2 (6.9–7.5)	7.2 (6.9–7.7)	7.3 (7.4–7.9)	7.4 (7.1–7.7)	7.4 (7.1–7.6)	8.7	7.7	6.9–7.9[A]	7.8
EC (μS cm^{-1})	4174 (2830–6130)	5360 (5140–5530)	4498 (3340–5320)	5853 (5200–6720)	246 (136–431)	510	334	155–220[A]	2700
Chloride (mg L^{-1})	1246 (808–1760)	1563 (1450–1670)	1310 (939–1600)	1503 (1290–1690)	36 (19–73)	75	47	400[B]	608
Nitrate (mg L^{-1})	2 (<0.6–7.1)	15 (14–16)	27 (18–33.5)	44 (<0.6–81)	1 (<0.6–1.5)	<0.6	3	0.02[A]	0.50
Calcium (mg L^{-1})	201 (127–287)	244 (237–250)	135 (49–162)	189 (182–198)	12 (7–19)	15	15	–	40
Sodium (mg L^{-1})	451 (275–704)	613 (580–647)	725 (504–829)	835 (734–980)	27 (16–50)	73	39	300[B]	586
SAR	5.9 (4.5–7.9)	7.4 (7.2–7.7)	11.3 (10.3–12.4)	11.1 (10–12.7)	1.5 (1.2–2.2)	3.5	2.0	2–8[B]	18.1

[A]Draft water quality objectives for the Lower Condamine Floodplain for a moderately disturbed system, under low and high flow conditions (Newham *et al.* 2018).
[B]Values from the ANZECC Water Quality Guidelines (ANZG 2018).
[C]Water quality objectives for QMDB zone #8 Lower Condamine alluvium (McNeil *et al.* 2018). Values presented are 50th percentiles.

lands could present a high risk of regolith salt stores being mobilised towards the Condamine River. This was based on the close proximity of the development to the Condamine River, presence of a hydraulic gradient towards the river, high salt stores in the landscape and existing regolith data. The identification of a saline seepage zone indicated that hydrogeological connection was already occurring. Based on first principles, irrigation schemes are inherently associated with a higher risk of salinisation, because the volume of water added will always be greater than any natural system could provide. The overall salinity risk depends, in part, on the characteristics of the unsaturated zone and the underlying bedrock. The presence of and depth to any zones of reduced permeability will determine deep drainage and will ultimately determine how fast the unsaturated zone will fill. The overall aim of this study was

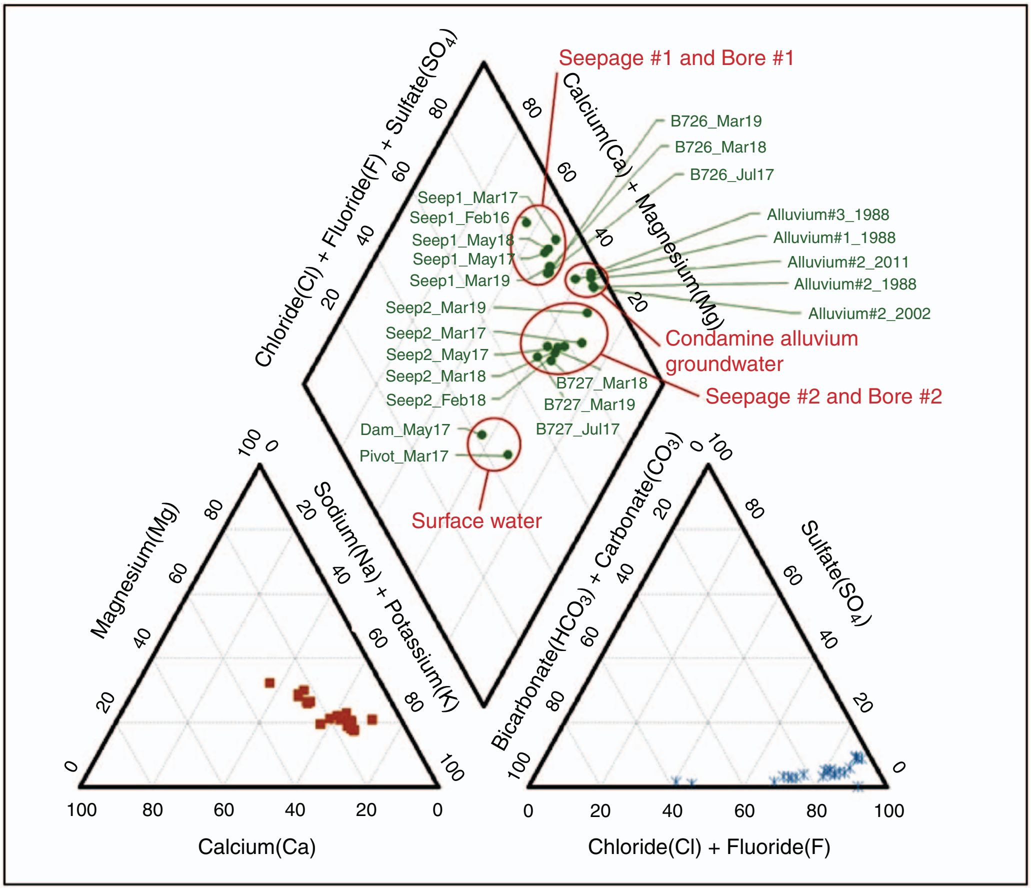

Fig. 7. Piper diagram showing the results for the two seepages, two groundwater bores and two surface water points. Results for nearby (within ~20 km) Condamine alluvium groundwater are also shown for comparison purposes.

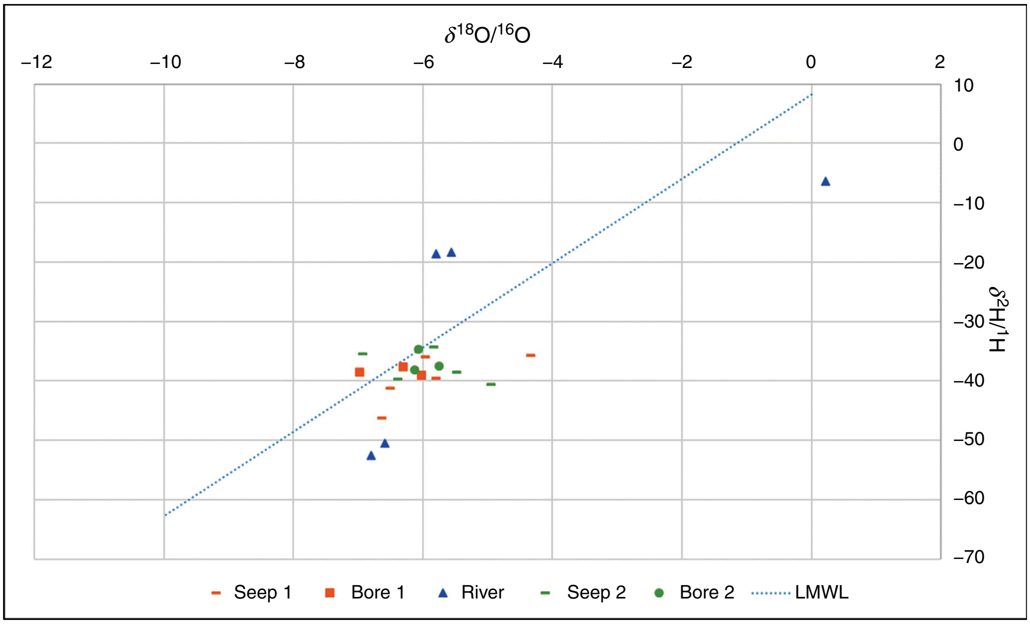

Fig. 8. Stable isotope results for the two seepage points, two groundwater bores and Condamine River plotted against the LMWL (local meteoric water line).

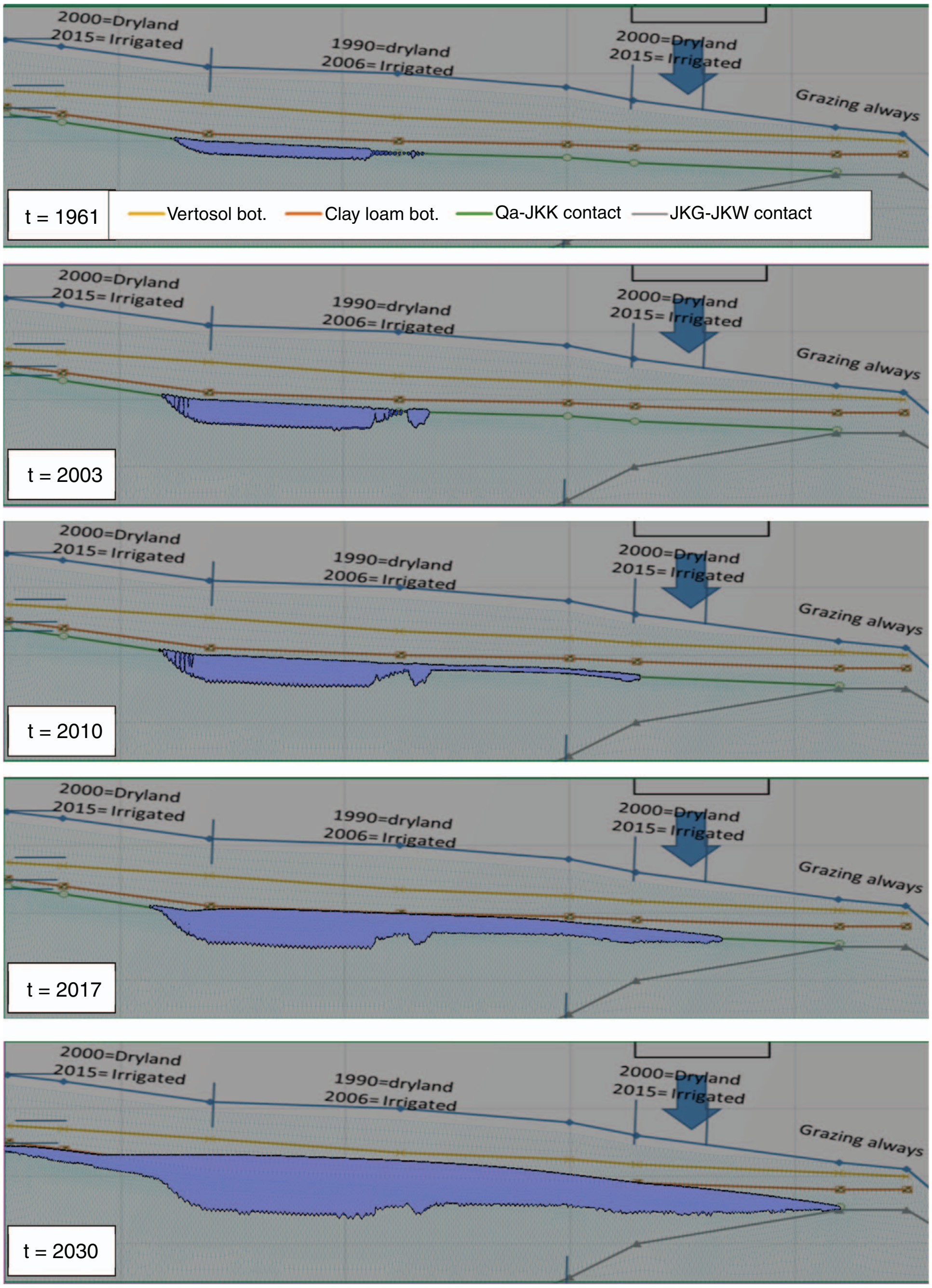

Fig. 9. Saturated zone development (shaded blue area) for the base-case model run (years 1961–2030).

to determine if an increased amount of water being applied to the landscape in the form of irrigation would cause detrimental impacts to the environment. These impacts could take the form of salinity expressions within the study area, rising watertables or saline discharge into the Condamine River.

Land-use change

Understanding land-use change is one of the key factors in the calculation of how much deep drainage has occurred over time. As deep drainage rates vary under different land uses, so too does the time to fill the unsaturated zone. If a hydrological impediment exists (i.e. a layer which impedes the vertical movement of water), the unsaturated zone will fill to different levels, depending on land management. Under native vegetation, it is expected that minimal deep drainage occurs, whereas under grazing, dryland cropping and irrigated cropping the amount of drainage occurring below the root zone increases. Yee Yet and Silburn (2003) reported that deep drainage significantly increased with irrigation in the

Queensland Murray–Darling Basin (QMDB). In their example, the average modelled deep drainage for native woodland on a grey Vertosol was <1 mm year^{-1}, whereas the average modelled deep drainage rate due to rainfall alone for irrigation cropping areas was 43 mm year^{-1}. They also reported that deep drainage under irrigation could range within 8–165 mm year^{-1}, meaning that the potential time to fill the unsaturated zone would vary substantially. A more recent review, however, found deep drainage on similar soil types to our study area (Vertosols) was typically in the range of 100–200 mm year^{-1} (Silburn *et al*. 2013). This provided a more consistent timeframe to work from, when calculating the time to fill the unsaturated zone under irrigation. Changes of land use and their associated deep drainage rates have been occurring within the study area for over 60 years, since 1956. Hence, understanding the time frame in which different land uses, in particular irrigation, have been occurring is a vital piece of information for salinity risk assessment.

The size of the irrigation footprint will inevitability influence the development of any potential hydrological mounding or lateral flow rates within the landscape (Yihdego 2017). In this study, the area under irrigation increased from ~2760 ha in 2013 to almost 4570 ha in 2015, corresponding to the commencement of the Fairymeadow Road Irrigation Scheme in March 2014. In 2017, the area under irrigation increased further to 4710 ha as landholders expanded their irrigation operations. Although additional water application can lead to increased deep drainage, this is not always the case. Efficient irrigation practices coupled with much shorter intercrop fallows can result in reduced frequency and magnitude of deep drainage events (Irmak *et al*. 2011). However, as stated earlier, deep drainage and the filling of the unsaturated zone will inevitability occur in some capacity, even using modern efficient irrigation systems. As mentioned above, since 2013 the area under irrigation in the study area has increased by nearly 60% (~1950 ha). This provides a clear separation between the areas of pre- and post-Fairymeadow Road Irrigation Scheme time periods, enabling a better basis to assess any potential water balance change. How long that change will take will be determined by the depth of the unsaturated zone and the presence of any impediment layers or aquitards.

Assessing the unsaturated zone

The initial conceptual model of the regolith architecture developed for the study area was limited by the coarse scale of available geological data and inconsistencies within the available groundwater bore logs. The presence of many CSG-related wells in the area did not provide further assistance, as their drill logs contained little surficial lithology detail. The interpretation of ERT transects aided in the determination of not only the depth of the soil profile, but also the underlying lithology of the geological unit (fine, medium or coarse sized). This information helped define the depth to the first layer with reduced permeability (i.e. a hydrological impediment). For example, initial investigation of groundwater bore logs suggested that sand seams up to 20-m thick were present under much of the area, potentially providing connectivity to the Condamine River via lateral flow. However, the ERT surveys and deep calibration cores showed that, in some areas, the sand seams were actually sandstone, not sand. This means that the likelihood of lateral flow to the Condamine River would be lower, due to the different hydraulic properties of sand vs sandstone. Within the study area, the Kumbarilla Beds are divided into several subunits, with the uppermost unit being the GSF, overlying the Westbourne Formation. Bore logs from within the study area identified the Westbourne Formation to be located deeper than inferred by the ERT transect. The presence of a fault in transect #3 (Fig. 3*a*, *b*), situated 450–500 m along the transect and at ~60 m below the surface, could provide insight as to why. Although the fault was subsequently confirmed by consulting geological mapping (Cranfield 2017), it does raise a question about the consistency of any depth to impermeable layers within the study area. These materials have varying permeabilities which influences deep drainage rates, likelihood of lateral flow and groundwater recharge. The shallower occurrence of this unit can be explained by the presence of the fault, which appears to have resulted in uplift of the Westbourne Formation. It is possible that the extent of faulting in the study area is under-mapped or not identified, which can lead to a misinterpretation of the underlying substrate, with implications regarding water movement into and through the substrate. Although early geological mapping covering the study area (e.g. BMR and CSQ (1971)) is useful, advances in technology such as radiometrics and geophysical surveys mean the potential to accurately interpret the underlying lithology has greatly increased. Improved geological and geophysical information also created new questions about the depositional history of the study area, with a split pattern emerging in the substrate material. This pattern consisted of a coarse–fine lithology roughly splitting the study area in half, potentially leading to vastly different calculated timeframes for secondary salinity risk in each section. Knowing the depth of the soil profile, and how it conforms to the substrate vastly improves the understanding of what will happen to deep drainage which is essential for groundwater and therefore salinity modelling.

Groundwater modelling can be substantially improved with data from geophysical studies such as ERT surveys, as they provide detail about the depths of underlying layers, including the presence of any layers that may impede water movement. The associated calibration cores provide further information in this regard and should always be included in any geophysical study. In this study, the calibration cores identified a relict watertable, situated 7–10 m below the modern land surface. Hydrogeological impediments related to historical aquifer features may still influence water movement through the regolith. The relict watertable feature was identified by a pH inversion–reversion and high concentrations of carbonates. A pH inversion (going from alkaline surface soils to acid subsoils) is an interesting trait of cracking clays in this region that has been well documented, for example: Hubble and Isbell (1958), Maher (1996), Harris *et al*. (1999) and Biggs and Binns (2015). However, a pH inversion–reversion pattern has not been documented before.

This could be because soil profile descriptions were not deep enough, or because the specific circumstances have not been met elsewhere in the region. In order to change the pH values from acid back to alkaline at depth (reversion), a significant soil environmental change would be required – the presence of a watertable could provide such a catalyst. The inference of this in regard to salinity risk is that, once modern irrigation water reaches this depth (i.e. a zone of reduced permeability), saturation is more likely to occur.

The identification of several seepage points along a stretch of the Condamine River helped clarify the question of connectivity between the landscape and the river. Information gathered through deep soil coring, ERT and installation of groundwater bores suggests that in this particular part of the study area, water is moving laterally through the landscape, into groundwater and into the Condamine River. This is due to the underlying bedrock being less permeable, so water is moving laterally above this layer. Additionally, this area is located very close to several centre pivot irrigators, so the amount of water being added exceeds that able to move down through the profile and into the bedrock. Whether connectivity existed before agricultural development is unknown, as it is possible that water is simply following old flow paths. However, agricultural development has led to more water being added to the system and more water moving through the connecting strata, exacerbating the likelihood of discharge into the river.

The analysis of stable isotope data (Fig. 8) and the closeness of fit to the LMWL provides further evidence of the source of groundwater recharge. Several studies have utilised hydrochemistry to understand the sources of isotopic signatures (δ^2H and δ^{18}O) of groundwater and surface water in relation to rainfall events, recharge source and hydrogeochemical evolution of groundwater (Harrington *et al*. 2002; Vanderzalm *et al*. 2011; Dogramaci *et al*. 2012). On the basis of their isotopic content, elevations of groundwater recharge may be differentiated on the relative position of the grouping to LMWL. Although temporal and spatial variation exists within the water quality dataset, due to the closeness of fit to the LMWL the most likely source of groundwater within the bores and seepage sites is surface or shallow recharge. If the source was of deeper artesian groundwater the data cluster would display a greater separation from the LMWL. Additionally, the presence of high nitrate levels and atrazine breakdown products in the sampled groundwater points to a surface connected source.

An associated issue related to deep drainage under irrigation is that it tends to be concentrated in small areas, which can create locally higher watertables (Vervoort *et al*. 2003), i.e. groundwater mounding (Hantush 1967). Cook *et al*. (2007) showed that the maximum water table rise (in the centre of the irrigated area) is strongly dependent on the size of the irrigation area, and as the radius increases so too does the water table rise. They showed that the water table rise is linear with time initially, but the rate of rise reduces with time as the mound spreads laterally. The rate of water table rise will be reduced if water leaks downward or if there is a sink (e.g. river) that the water can flow to. Even with such sinks, the resulting water table rise may still be significant (Cook *et al*. 2007). Field investigations (flow rate within each bore, isotope data and an ERT survey) indicated that groundwater mounding could potentially be happening at one site in the study area. This was an area under centre pivot irrigation, adjacent to bore #2 and seepage #2 (Fig. 2). Whether that bore is on the shoulder of the mound or deeper within is not critical, but rather, the groundwater in this bore and in seepage #2 would be heavily influenced both volumetrically and chemically by water moving out of the mound. This helps explain the differences in water quality and water levels between the two bores. To better understand groundwater mounding in the study area, it would be necessary to (a) install and regularly monitor a grid of piezometers over the affected sub-areas and their environs and (b) conduct a series of repeat ERT surveys, along the same transects, looking for temporal changes. Ultimately the assessment of the unsaturated zone should clarify both local and catchment scale variability.

The depth of the unsaturated zone inherently varies across the QMDB, ranging from <5 to >100 m. This is dependent on several factors including geology, lithology, substrate permeability and porosity, landscape position, slope and landform (Biggs *et al*. 2013). Inherent inaccuracies in broad hydrological modelling will be magnified by inconsistency such as the fault line in transect #3 and the variable depths observed within the collected data (bore logs, deep cores and ERT surveys). This is why a localised approach to salinity risk assessment is critical to obtaining an accurate timeframe of connectivity to the river.

Hydrogeological modelling

The HYDRUS-2D simulations successfully described recharge and percolation at various rates, and the resulting redistribution of water in the subsurface over a period of 100 years (Fig. 9). The modelling suggests that most of the input water would have percolated vertically, and will not result in seepage into the Condamine River, i.e. the induced recharge due to the changes in land use is still 'filling' the unsaturated zone. This is contrary to field observations of seepage found in the vicinity of the centre pivots close to the Condamine River. It does, however, support the formation of mounds under the centre pivot irrigation sites. It is envisaged that with time, seepage and vertical recharge to the deeper aquifer will increase substantially, which should be reflected in the bore data and seepage flows. It will be the position in the landscape or the presence of an unmapped impediment that will largely determine if seepage occurs. The revised model case study showed that seepage into the Condamine River could be simulated successfully. However, this was only after amending several underlying assumptions, e.g. assuming the Qa–JKk contact was impermeable, changing Qa–JKk conductivity and anisotropy, using higher deep drainage rates and assuming that the seepage point lithology had changed. The revised modelling showed that the occurrence of seepage is linked to the lateral extent of the perched lens. That is, whenever there is a hydraulic continuity between the seepage point and the perched lens, there will be some seepage. The modelling suggested that the current understanding of the shallow hydrogeology in the study area and the deep drainage rates under various land uses is not sufficient.

Conclusions

Multiple lines of evidence were used to assess the risk of secondary salinity occurring within an irrigation scheme on the Condamine River, Queensland. The risk assessment encompassed the inherent landscape features that may contribute to salinity processes, the level of risk due to land management practices and the progression of salinity processes in the landscape. These factors were used to determine the amount of time before any potential secondary salinity might express. The main findings follow:

- Geophysical investigations identified that the Kumbarilla Beds were highly variable in the study area and that the thickness of the unconsolidated zone varied considerably. Under the majority of the irrigation scheme area, it was 2–10 m thick.
- The identification of several seepage points along a stretch of the Condamine River indicated hydraulic connectivity between the landscape and the Condamine River. Water quality analysis suggested a link to current localised irrigation practices.
- Hydrogeological modelling of recharge through the surficial sediments showed that the current understanding of deep drainage rates into the underlying regolith is insufficient as it did not replicate the known riverbank seepages.

An implication of these findings is that there is a clear knowledge gap in the hydrogeological interactions at the point of contact between the unconsolidated zone and the underlying bedrock. If underlying bedrock can constitute the arbitrary limit of the unsaturated zone then the calculated time to fill would be substantially reduced. Another implication is the measurable permeability of the Kumbarilla Beds. Further data collection would be required to determine if the seepage area was a localised phenomenon or if improvements can be made to both the overall risk assessment or modelling process. These could include the installation of piezometers around known seepage points to more accurately track groundwater or seepage movement. Another could be conducting slug tests of vertical permeability to a depth of ~50 m to better assess the permeability of shallow unconsolidated sediments and the underlying Kumbarilla Beds, including the upper weathered zone.

Conflicts of interest

The authors declare no conflicts of interest

Acknowledgements

Thanks and acknowledgement is made to all who contributed to the study, including Landholders: Simon Drury, Ash Geldard, Ian Geldard, Murray Geldard, Andrew Rushwood, Bruce Uebergang and David Uebergang; DNRME staff: Jenny Foley, Maria Harris, Sunny Jacobs, Tony King, Brad Oleksyn and Andrea Prior; and Department of Environmental Science staff: Jeremy Manders, Angus McElnea, Fred Oudyn, Sonya Mork, Dan Yousaf, Ashneel Sharma and Rob de Hayr. Thanks also go to Dan Smith and Jon Walton for reviewing the manuscript.

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Handling Editor: Nilantha Hulugalle