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Injection of fresh river water into a saline floodplain aquifer as a salt interception measure in a semi-arid environment

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ABSTRACT

Floodplains in arid and semi-arid environments are hydrologically and ecologically essential components of the landscape. However, floodplain salinization has been highlighted as a significant risk for riparian tree health and river water quality. While various salt management measures have been developed, some of these have limited application in arid and semi-arid regions because of insufficient infrastructure and limited availability of supplemental environmental flows. Fresh river water injection into a saline floodplain aquifer can lead to environmental improvement using a relatively small amount of water and without the need for water disposal infrastructure. To explore the impacts of fresh river water injection on floodplain salinity, a physically-based, fully integrated numerical model was developed and calibrated against the observed data from a trial conducted in September 2006 at Clark's Floodplain in the Lower Murray in South Australia. It is shown that injection of an increased volume of river water leads to a larger extent of the subsequent freshwater lens. In addition, it is shown that for a given injection volume, it is more efficient to inject at a lower injection rate and for a longer duration. Also, the interface of the saturated/unsaturated zone appears to be the most effective injection screen depth. Moreover, in this case, a linear configuration of the injection wells was more effective compared to a rectangular configuration. Overall, the fresh river water injection is only able to maintain a temporary and spatially limited local freshwater lens. For long-term salt management, the river water injection needs to be periodically repeated in the absence of overbank flooding in the meantime.

1. Introduction

Floodplains are an integral part of a river basin's landscape, supporting agriculture, recreation and industry (Berens et al., 2009a; Holland et al., 2009b; Vugteveen et al., 2006), particularly, in arid and semi-arid environments where water resources are limited. Thus, arid and semi-arid floodplains and their adjacent wetlands often have higher biodiversity value than surrounding areas, providing habitat for many aquatic and riparian species, including vegetation, birds, mammals and fish species (Alaghmand et al., 2014a; Doble et al., 2006). But riverine floodplains have been gradually isolated from their parent rivers and are now amongst the world's most vulnerable and impacted ecosystems (Berens et al., 2009b; Mussared, 1997; Tockner and Stanford, 2002).

In the Murray River in South Australia, the installation of a series of weirs, constructed to regulate the river flow, along with irrigation practices on the highlands adjacent to the floodplains, has increased recharge by a three orders of magnitude. This has caused saline groundwater in the floodplain to rise (Alaghmand et al., 2013a; Jolly et al., 2002). River regulation and diversions for consumptive use have altered the extent, duration and timing of overbank flooding (Holland et al., 2013). These factors have increased the rates of soil salinization and this has degraded environmentally significant riparian vegetation health such as river red gum (*Eucalyptus camaldulensis*) and black box (*Eucalyptus largiflorens*) (Berens et al., 2009a; Cunningham et al., 2007; Holland et al., 2009a; Jolly et al., 1996).

Delivering environmental flows has been recommended as a floodplain management strategy to enhance the affected floodplains

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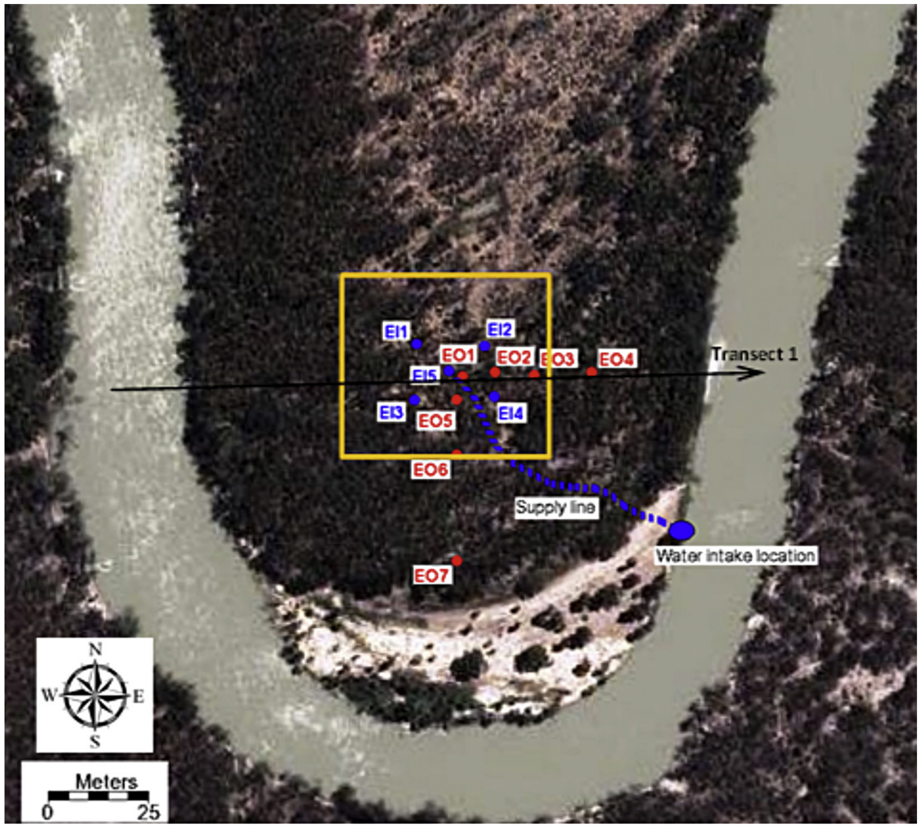
(Alaghmand et al., 2014b; Arthington and Pusey, 2003; Hughes and Rood, 2003). These environmental flows are provided via releases of water from large storages (Richter and Thomas 2007). However, this is often of limited feasibility in arid and semi-arid environments due to the large volume of water required. In such regions, recharge from rainfall is unlikely to occur, therefore the river and groundwater are the most likely available sources for water supply. If the groundwater aquifer is naturally saline, the river often has to supply the water demands including irrigation, potable water and the generation of hydroelectric power (Tockner and Stanford, 2002). Hence, the amount of water allocated for the environment is often limited. For instance, in the Murray River in South Australia only 1% of the total flow was explicitly available for environmental flows in 2006–2007 (Berens et al., 2009a). Even if enough water is available, delivery of environmental flows is usually associated with some technical difficulties (Berens et al., 2009a).

Managed aquifer recharge (MAR) is a method to balance water supply with demands. The goal is to generate environmental improvement using small amounts of water relative to surface flooding and water extraction techniques, and without the need for water disposal infrastructure. It consists of injecting water into an aquifer during a period of excess, in order to keep it stored until required (Antoniou et al., 2013; Pyne, 1995). In MAR, excess surface water is injected into a subsurface aquifer for subsequent recovery. This can be undertaken in order to create a reservoir of fresh water by displacing the natural saline groundwater in the capillary fringe (Eastwood and Stanfield, 2001). In such situations, injection of fresh river water into the floodplain shallow saline aquifer can be implemented as a novel engineering technique for environmental flow delivery (Berens et al., 2009a). This is particularly beneficial for trees that are able to access fresh water from both surface infiltration and the groundwater such as *E. camaldulensis*

Fig. 1. Location of Site E on Clark's Floodplain on the Lower Murray River in South Australia (Purple dotted line shows the floodplain perimeter). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)



(a)

(b)

Fig. 2. (a) Configuration of injection (blue dots) and observation (red dots) wells. Rectangle in yellow represents the perimeter of the injection zone, (b) Photo of the aquifer breach on 8 November 2006 next to injection well EI4 at Site E (Berens et al., 2009a). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

Table 1
Specifications details of the injection and observation wells.

Well name	Purpose	Drilled date	Reference elevation	Ground elevation (m AHD)	Diameter (m)	Total depth	Screen depth (m)
EI1	Injection	12 April 06	13.1	12.4	0.2	7.0	3.0–6.0
EI2	Injection	11 April 06	13.3	12.9	0.2	7.0	3.0–6.0
EI3	Injection	11 April 06	13.6	13.0	0.2	7.0	3.0–6.0
EI4	Injection	12 April 06	13.0	12.5	0.2	6.0	2.0–5.0
EI5	Injection	12 April 06	13.5	13.0	0.2	7.0	3.0–6.0
EO1	Observation	7 April 06	13.8	13.0	0.08	12.5	2.5–11.5
EO2	Observation	8 April 06	13.4	12.4	0.08	7.5	2.0–7.0
EO3	Observation	8 April 06	13.4	12.4	0.08	7.5	2.0–7.0
EO4	Observation	8 April 06	13.5	12.3	0.08	7.5	2.0–7.0
EO5	Observation	7 April 06	13.2	12.4	0.08	7.5	2.0–7.0
EO6	Observation	9 April 06	12.9	12.0	0.08	7.5	2.0–7.0
EO7	Observation	10 April 06	13.6	12.5	0.08	7.3	1.8–6.8

(Mensforth et al., 1994; Thorburn et al., 1994). It would also reduce the water losses due to evaporation (Berens et al., 2009a). Berens et al. (2009a,b); Berens et al. (2009a,b) reported the application of river water injection into a floodplain saline aquifer as an alternative methodology for managing the health decline of riparian vegetation. However, due to technical issues including problems of aquifer clogging and breaching, and the installation and operation of the injection bores was a costly and intrusive operation. Therefore, they ceased the operation after 57 days. Thus, the small volume of injected water (4.9 ML of the allocated 10 ML) only slightly improved soil water availability in the capillary fringe.

The aim of this paper is to explore the hypothesis: is fresh river water injection to a saline floodplain aquifer a suitable floodplain management option? We further explore how this can improve water availability for riparian trees. In this context we study the following questions:

- Is fresh river water injection a plausible mechanism to create a thin freshwater lens at the saturated/unsaturated interface zone where tree roots uptake available soil water?
- Can fresh river water injection mobilize some of the solute mass stored in the unsaturated zone?

The impacts of factors such as injection rate, allocated water volume, pump screen depth and injection well configuration are also examined to determine how these variables influence the state of saline groundwater displacement. This displacement is studied in terms of extent, depth and duration. For this, a fully integrated, physically-based numerical model is developed and calibrated based on observed data collected during a field trial conducted at a study site.

2. Material and methods

2.1. Field trial setup

The study site, known as Site E, is in the southern part of Clark's Floodplain (140°340E, 34°220S), which is located downstream of Lock 4 on the Lower Murray River in South Australia and covers an area of approximately 5 km^2 (Fig. 1). It has a semi-arid environment with a mean annual rainfall of 264 mm and potential annual evaporation of approximately 1800–2000 mm (BOM, 2013). The 1500 ha Bookpurnong irrigation district, developed since 1964, is located adjacent to Clark's Floodplain (Telfer and Overton, 1999). Excess irrigation drainage beneath this district induces groundwater mounding, which hydraulically displaces regional saline groundwater into the floodplain alluvial aquifer and has caused seepage of saline groundwater at the edge of the river valley (Holland et al., 2013). For further details on the study site the reader is referred to Doble et al. (2006),Alaghmand et al. (2013b) and Holland et al. (2013).

A total of five injection wells, each 200 mm in diameter, and seven observation wells, each 80 mm in diameter, were installed at Site E in April 2006 using a rotary mud method, as suggested for unconsolidated aquifers by Segalen et al. (2005). The environmental flow volume allocated for this trial was 10 ML of Murray River water. Data-loggers were fitted in all seven observation and five injection wells to record groundwater levels. Observation wells EO1, EO2, EO3 and EO5 were also monitored for conductivity and temperature. The injection trial began on 19 September 2006 and ceased on 8 November 2006. The injection rate varied between 0.8 and 1.25 l s^{-1}. Over the injection period a total of 4.9 ML of water was injected into the aquifer. The injection ceased on 8 November 2006 due to injection well and

Table 2
Specifications of the defined scenarios.

Scenario name	Injection rate (l.s^{-1})	Volume (ML)	Screen depth (m)	Pump configuration	Time steps (days)
A	1.25	10	3–6	Five-point rectangular	19–111
B	2	10	3–6	Five-point rectangular	19–76
C	5	10	3–6	Five-point rectangular	19–42
D	2	20	3–6	Five-point rectangular	19–134
E	2	10	3–10	Five-point rectangular	19–76
F	5	10	3–6	Single-point	19–134
G	5	10	3–6	Five-point linear	19–42

aquifer clogging with biological and particulate matter, resulting in the breaching of the confining clay layer approximately 1.5–2.5 m away from the injection wells. For further details on the trial conducted in September 2006 at Clark's Floodplain the reader is referred to Berens et al. (2009a). Fig. 2 shows the configuration of the injection and observation wells at Site E, while Table 1 summarises the specification details.

2.2. Base case model

A numerical model was developed which simulated the observed behaviour of a saline floodplain aquifer induced by fresh river water injection at the study site. The calibration model (hereafter referred to as the base case model) included the design specifications for the injection trial. The base case model was run from 1 September 2006 to 1 December 2006. This included injection of 10 ML water via five-point rectangular configuration scenarios (see Fig. 2). The base case model and all the simulated scenario were based on the assumption that an aquifer breach would not occur.

2.3. Scenarios

To explore the impacts of fresh river water injection on the saline floodplain aquifer, the base case model was modified for various different scenarios (Table 2). These included seven scenarios with different injection rates, volumes and screen depths. Also, two alternative injection wells configurations were tested, namely single-point and five-point linear. The study time frame was from 1 September 2006 to 1 September 2007 in 365 daily time-steps. The saline groundwater displacement induced by the defined injection scenarios were analysed in terms of extent and duration.

2.4. Numerical model development

2.4.1. HydroGeoSphere

Surface water–groundwater interactions were simulated using the groundwater flow model HydroGeoSphere (HGS) (Therrien et al., 2006). HGS is a physically-based numerical model describing fully-integrated surface and unsaturated and saturated flows in the subsurface. HGS models flow in the unsaturated zone using the Richards equation. This is a significant advantage over models that do not explicitly consider the unsaturated zone (Brunner and Simmons, 2012). Also, HGS is capable of simulating density-dependent flow. Density-dependent flow can play an important role particularly in fractured geological units which was not the case here. The geology of the River Murray valley at the study site consists of an upper layer of heavy Coonambidgal clay, underlain by Monoman Sands (Holland et al., 2009a). Hence, density-dependence was not considered in the scope of this research. Also, density-dependent solute transport problems are very sensitive to discretization (Diersch and Kolditz, 2002). Therefore, considering the required fine discretization and the extent of the study area (92 ha), running the 3D fully-integrated model for the study site under density-dependent conditions would not have been computationally plausible. For further details on the code and a recent software review, the reader is referred to Therrien et al. (2006) and Brunner and Simmons (2012). HGS requires pre- and post-processor tools in order to handle input preparation (complex topography and grid) and visualization of the outputs. In this study, Grid Builder (McLaren, 2005) and Groundwater Modelling System (GMS) (AquaVeo, 2011) were used to generate the model grid. GMS was also used to visualize and interpret the model outputs. In this study, HGS used the control volume finite element approach to solve surface and subsurface flow and transport. Here an initial time step of 0.1 days, a maximum time step of 1 day and a

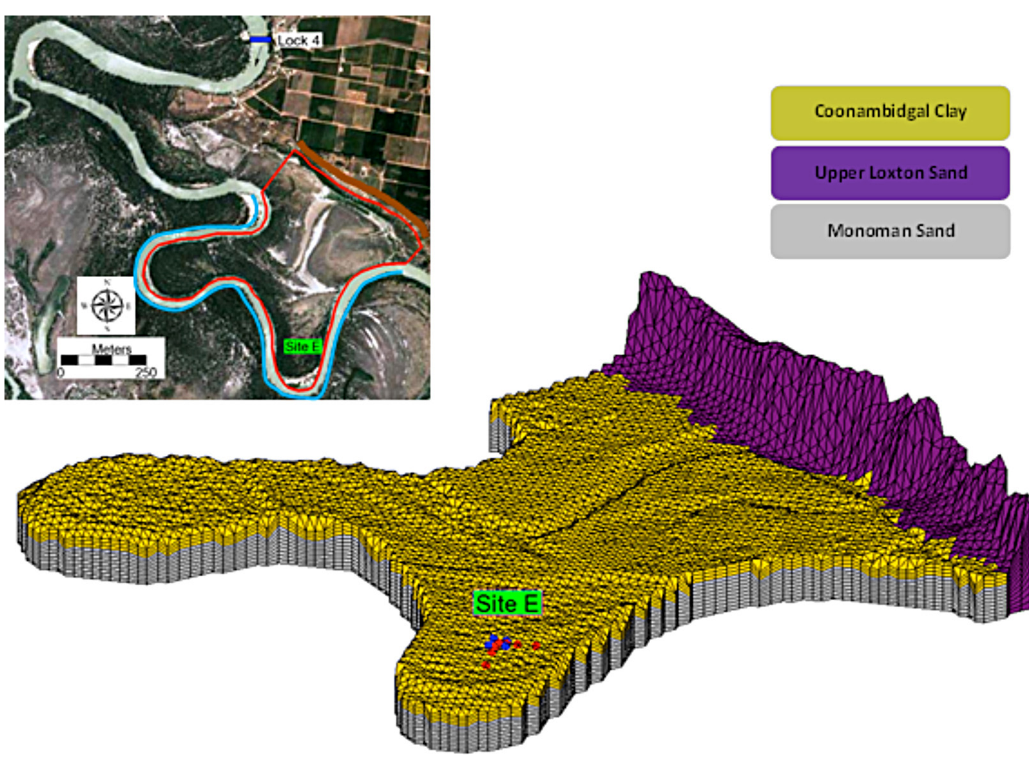


Fig. 3. 3D visualization of the geometric grid of the study site including soil types (Z magnification = 10). Red line in the inset map shows the perimeter of the model geometry grid. Blue and brown lines in the inset map show the location of constant first type (Dirichlet) and time-varying first-type (Dirichlet) boundary conditions.

Table 3
Soil parameter values of the model for the study site.

Model parameter	Soil type			Units
	Monoman Sand	Coonambidgal Clay	Upper Loxton Sand	
Porosity	35	60	40	%
Hydraulic conductivity	20	0.1	10	$m\ d^{-1}$
Specific storage	1.6×10^{-4}	2.0×10^{-3}	1.0×10^{-4}	m^{-1}
Evaporation limiting saturation (min)	0.05	0.25	0.15	
Evaporation limiting saturation (max)	0.9	0.9	0.9	
Longitudinal dispersivity	5	5	5	m
Transverse dispersivity	0.5	0.5	0.5	m
Residual water content	0.04	0.04	0.04	
a	1.69	0.28	0.8	m^{-1}
n	8.25	2.52	3.6	

maximum time step multiplier of 1.25 were used. The model solves non-linear equations for variably-saturated subsurface flow, surface flow and solute transport. To solve the non-linear equations, HGS uses the Newton–Raphson linearization method. Newton iteration parameters include Newton maximum iterations (25), Jacobian epsilon ($10.0\,d^{-5}$), Newton absolute convergence criteria ($1.0\,d^{-5}$), Newton residual convergence criteria ($1.0\,d^{-3}$) and flow solver maximum iterations ($1.0\,d^{-5}$).

2.4.2. Geometry grid

The geometric grid was based on a LiDAR Digital Elevation Model of the study site with a 10 m grid resolution and 15 non-uniform sub-layers. Fig. 3 shows the resulting grid of Clark's Floodplain covering 238 ha from the floodplain break of slope to the Lower Murray River main channel. In this case, the length of the river bank was 5950 m and the distance from the river bank to the floodplain break of slope was between 170 m and 1850 m. The final grid consisted of 226,629 nodes and 444,840 elements. The ground elevation ranges between 9.8 m Australian Height Datum (AHD) and 38.4 m AHD.

2.4.3. Model parameters

The floodplain aquifer consisted of three soil types, namely a continuous 10 m thick layer of Monoman Formation sand, overlaid by a spatially variable, 2–5 m thick layer of semi-confining heavy Coonambidgal Clay, and Upper Loxton Sand in the adjacent highland (Fig. 3). Soil properties and van Genuchten function (a and n) (van Genuchten 1980) parameters were adopted from Doble et al. (2006) and Jolly et al. (1993) (Table 3). Longitudinal and transverse solute dispersivity values were estimated through model calibration. The hydraulic properties of the surface domain (river bed and floodplain corridor) were divided in the model into two categories, namely main channel (river) and floodplain. Note that, riverbed hydraulic conductivity can vary spatially and temporally over several orders of magnitude (Calver, 2001; Wang et al., 2014). This can be due to erosion/deposition events (Hatch et al., 2010), biological activities (Treese et al., 2009) and temperature dependent material properties (Engeler et al., 2011). However, Irvine et al. (2012) showed that understanding the state of connection is the key to quantifying potential errors produced by numerical models when homogeneous equivalents are obtained by means of model calibration. They concluded that in connected and disconnected flow regimes, as defined by Brunner et al. (2009a,b); Brunner et al. (2009a,b), homogeneous equivalence is an efficient assumption. However, in a transitional regime a homogeneous equivalent will result in errors for simulations of both a rising and falling water table (Irvine et al., 2012). In this case, the river-floodplain is a connected flow regime during the study period. Furthermore, the vegetation coverage of the floodplain was classified as *Eucalyptus* trees and grass. The evapotranspiration parameter values for both categories were adopted from Hingston et al. (1997), Banks et al. (2011), Doody et al. (2009) and Verstrepen (2011). For further details on the model parameters the reader is referred to Doble et al. (2006), Doody et al. (2009) and Alaghmand et al. (2013b). Moreover, sensitivity analysis was conducted in this research prior to the model calibration. Among the tested parameters, Monoman Sand hydraulic conductivity and porosity, van Genuchten alpha for Monoman Sand and Transpiration extinction depth (*Eucalyptus*) show relatively significant impact on the groundwater head and salinity (Alaghmand 2014).

2.4.4. Boundary and initial conditions

Two types of boundary conditions were used in the model including first-type (Dirichlet) boundaries of prescribed head/concentration and second-type (Neumann) boundaries of prescribed flow/solute flux (Therrien et al., 2006). In the subsurface (porous media) domain, a constant first type (Dirichlet) boundary

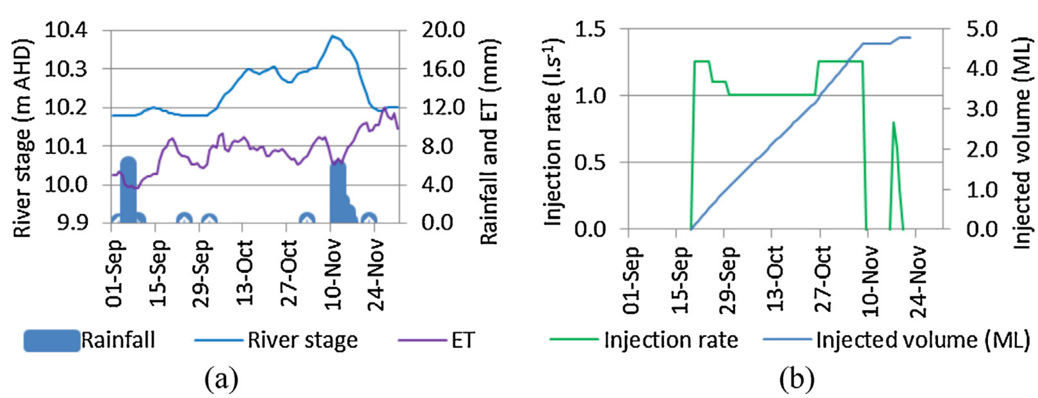


Fig. 4. (a) Recorded river stage, daily rainfall and daily ET during the trial; (b) Total injection rate and injected volume of river water.

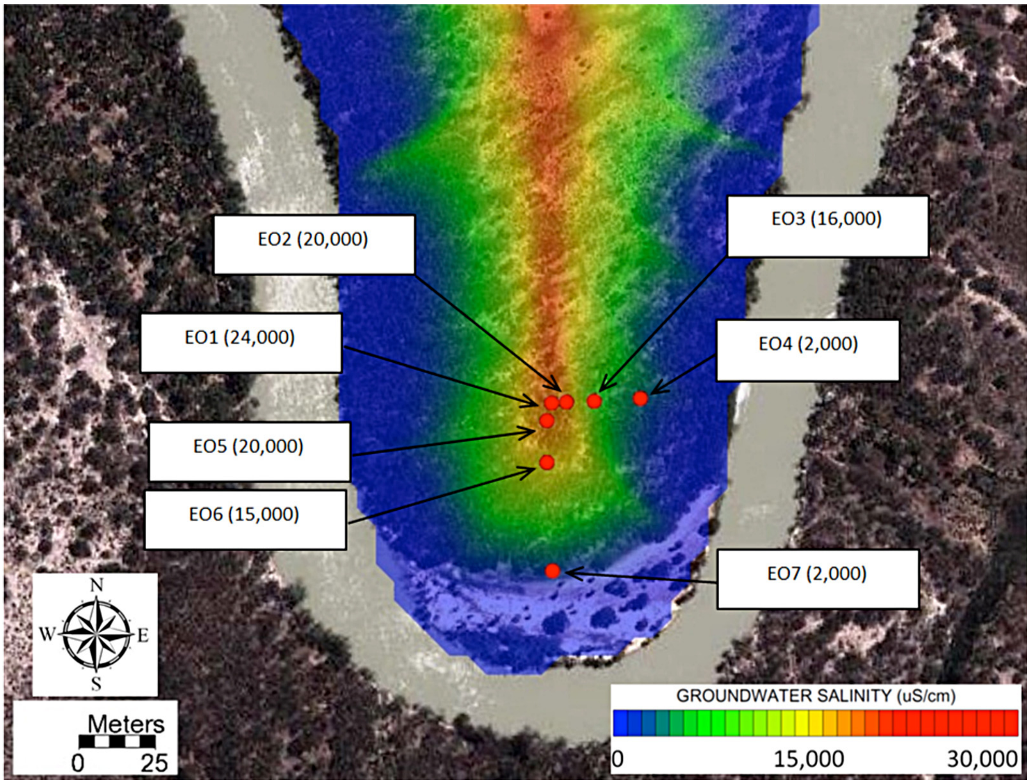


Fig. 5. Simulated groundwater salinity at the beginning of the study period (pre-trial) on 1 September 2006. Values in brackets represent the observed groundwater salinity.

condition of 12 m AHD constant head was specified at the north-eastern part of the domain to represent the potentiometric head in the regional Upper Loxton Sand aquifer at the edge of the floodplain (AWE, 2013). The observed river levels for the surface domain were assigned at the river side of the model using a time-varying first-type (Dirichlet) boundary condition. In this regard, the observed water levels downstream of Lock 4 were applied to the river nodes of the model (WaterConnect, 2013). The locations of the boundary conditions are displayed in the inset map in Fig. 3. Moreover, the solute boundary conditions were represented using

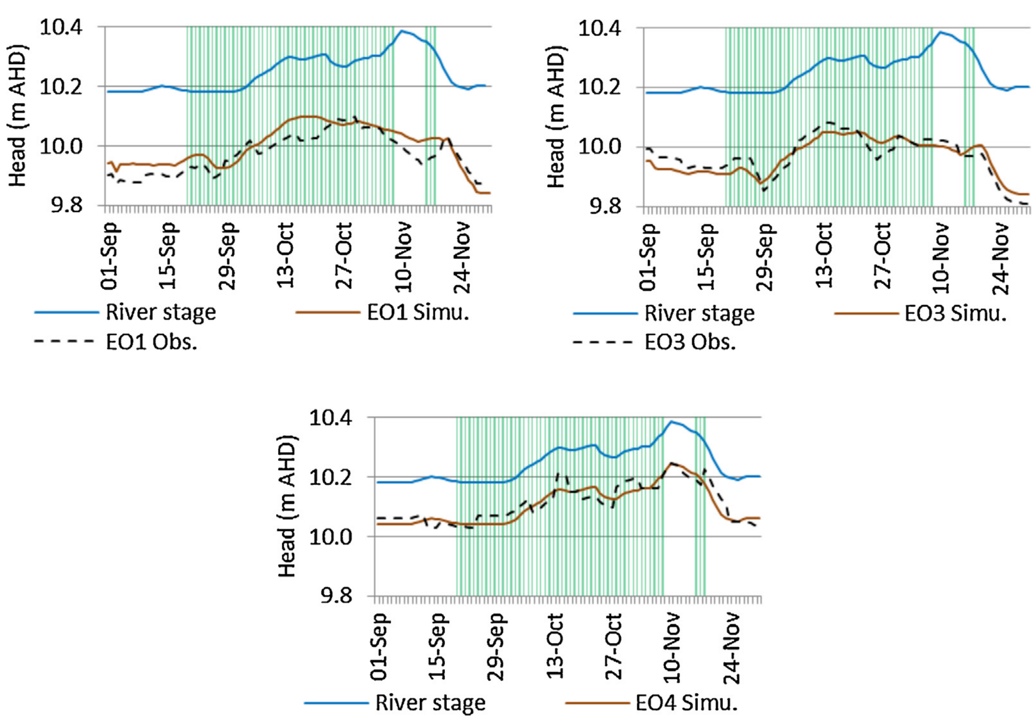
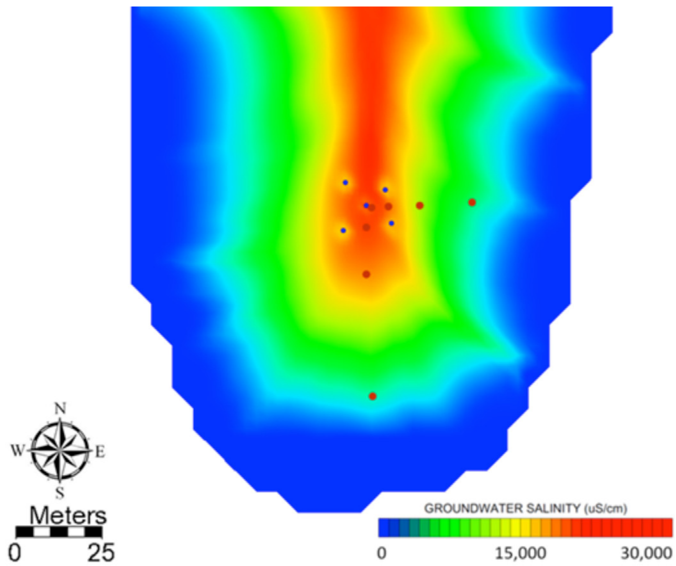
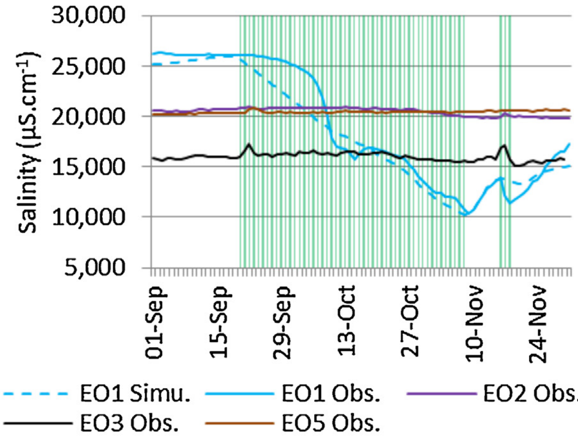


Fig. 6. Simulated and observed groundwater heads at observation wells EO1, EO3 and EO4. Light green pattern represents the injection trial period. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)



(a)



(b)

Fig. 7. (a) Plan view of simulated groundwater salinity distribution at Site E on 1/12/2006 (time step = 92 days); (b) Simulated and observed groundwater salinity at observation well EO1. Light green pattern represents the injection trial period. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

a first-type (Dirichlet) constant concentration boundary condition. Hence, constant values were applied at the subsurface outer boundary nodes (representing regional groundwater flow from the Loxton aquifer with salinity of 30,000 $\mu S\,cm^{-1}$) and the river nodes (200 $\mu S\,cm^{-1}$) (Holland et al., 2013). Potential evapotranspiration and rainfall were simulated using a time-varying second-type (Neumann) boundary condition according to recorded data (BOM, 2013). Also, the five injection wells were represented in the model using recorded injection rates and durations obtained from Berens et al. (2009a). Fig. 4 shows the recorded river stage, rainfall and ET during the trial at Site E as well as total injection rate and injected volume of river water.

The initial conditions for the base case model were obtained from a steady-state flow and transport model (constant river level and ET) that represented the condition of the river-floodplain system the prior to the study period. Therefore, hydraulic head and solute concentration outputs from the initial model compared favourably with recorded data from the observation wells and the EM31 survey results undertaken at the beginning of the study period. Fig. 5 shows the simulated groundwater salinity distribution on the 1st day of the trial (1 September 2006). The observed groundwater salinity levels at the observation wells show a good agreement with the simulated results.

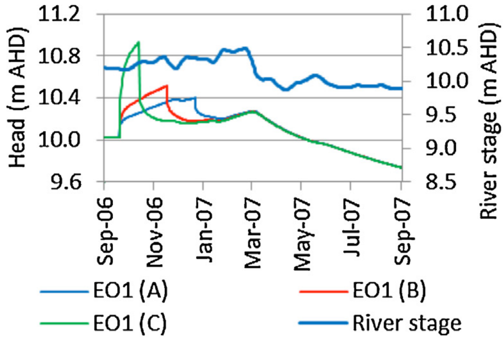
2.5. Model calibration

Calibration was undertaken using an iterative trial-and-error method. The hydraulic conductivity, porosity, dispersivity (longitudinal and transverse) and leaf area index were adjusted within known ranges and reasonable limits in order to achieve an acceptable match to observed data. The water table and salinity were compared to the observed data at observation wells EO1, EO3 and EO4. Moreover, the EM31 data were available to check the patterns of groundwater salinity.

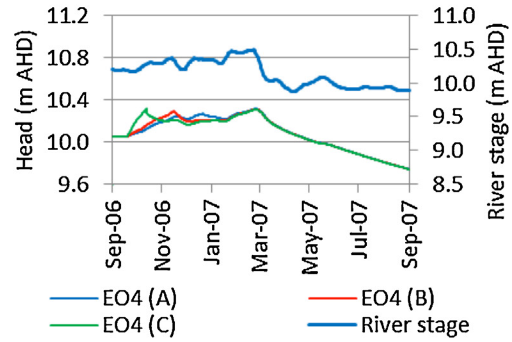
3. Results and discussion

3.1. Base case model

The calibrated model was developed with the aim to reproduce the observed behaviour of the flow and solute dynamics of the floodplain aquifer over the period 1 September 2006 to 1 December 2006. Thus, the model performance in terms of groundwater dynamic was tested by visual comparison between the observed and simulated series of groundwater levels at observation wells along transect 1 (EO1, EO3 and EO4). Fig. 6 shows a good agreement between the simulated and observed data. During the trial, the injection rates were between $0.8\,l\,s^{-1}$ and $1.25\,l\,s^{-1}$ between 19 September 2006 and 8 November 2006. This produced a small

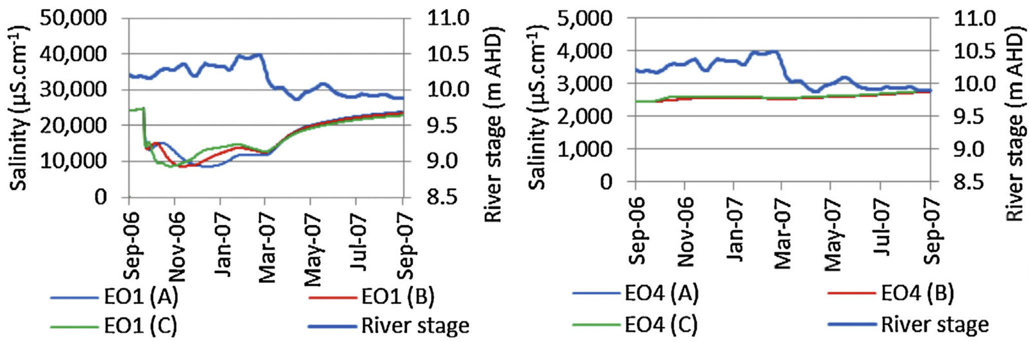


(a)



(b)

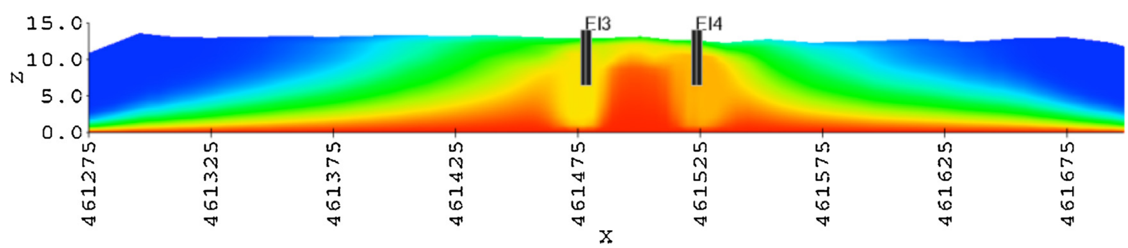
Fig. 8. Simulated groundwater head for the injections with $1.25\,l\,s^{-1}$, $2\,l\,s^{-1}$ and $5\,l\,s^{-1}$ injection rates at Site E.



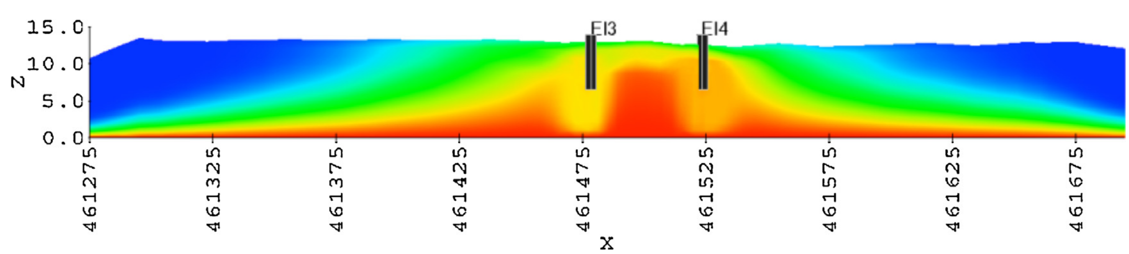
(a) (b)

Fig. 9. Simulated groundwater salinity for the injection with $1.25\,\mathrm{l\,s^{-1}}$, $2\,\mathrm{l\,s^{-1}}$ and $5\,\mathrm{l\,s^{-1}}$ injection rates at Site E.

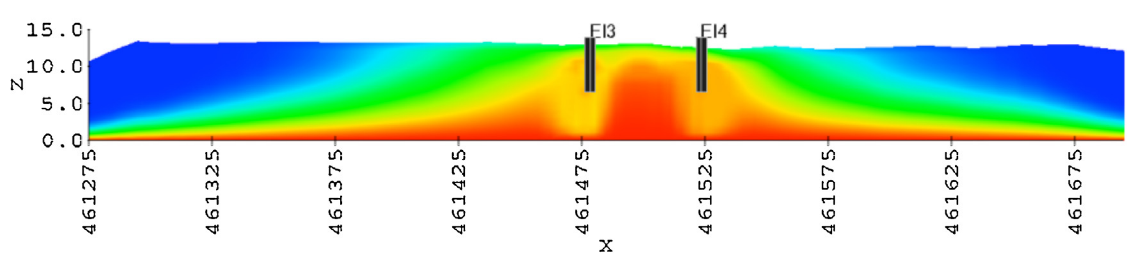
but rapid increase in groundwater level at the observation wells closest to the injection wells. For instance, along transect 1, a 0.05 m head increase was recorded at EO1 and EO3, which were located less than 25 m from the injection wells. Also, a small but rapid reduction in the groundwater level was detected when the trial was stopped following the aquifer breach. This was not observed in the other observation wells further away. It appears recorded groundwater level at EO4 was due to river stage



(a) Scenario A, time step 111 days



(b) Scenario B, time step 76 days



(c) Scenario C, time step 42 days

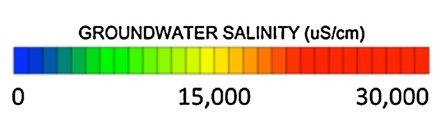
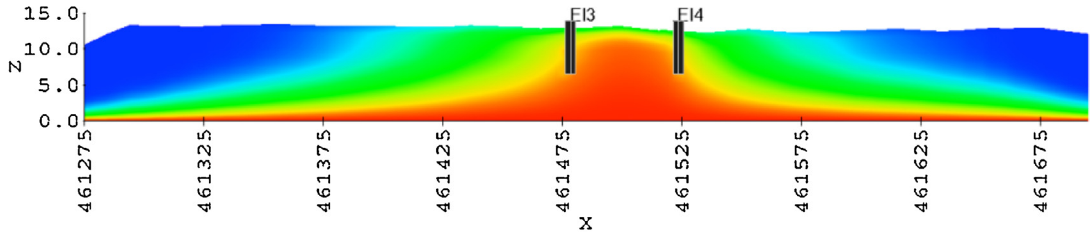
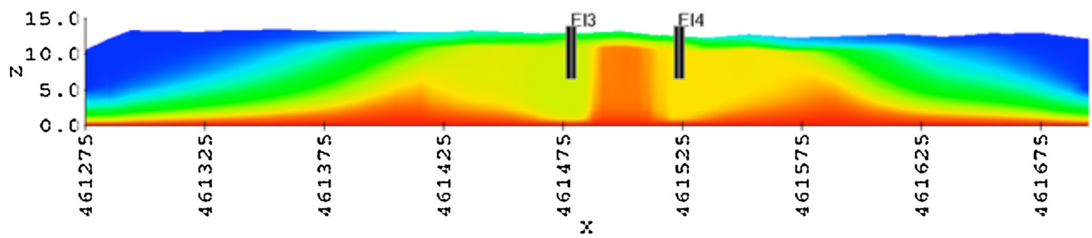


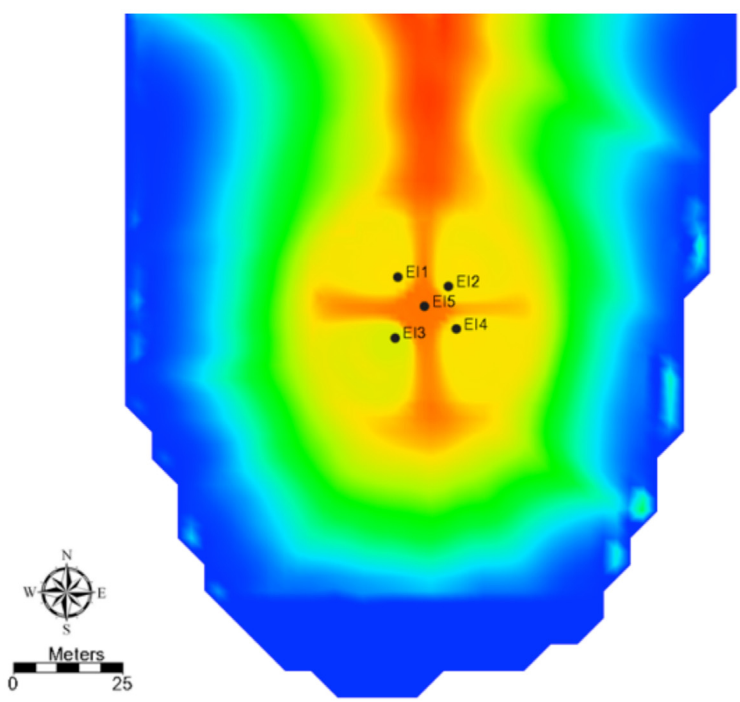
Fig. 10. Simulated groundwater salinity distribution for scenarios A, B and C at time steps 111, 76 and 42 days, respectively (Z magnification = 3).



(a) Scenario D, time step 19 days



(b) Scenario D, time step 134 days



(c) Plan view of Site E for scenario D, the time step 134 days

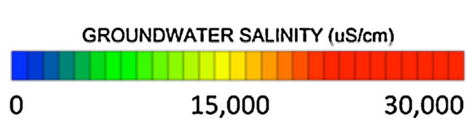


Fig. 11. Simulated groundwater salinity distribution for scenario D at pre and post injection time steps (Z magnification = 3).

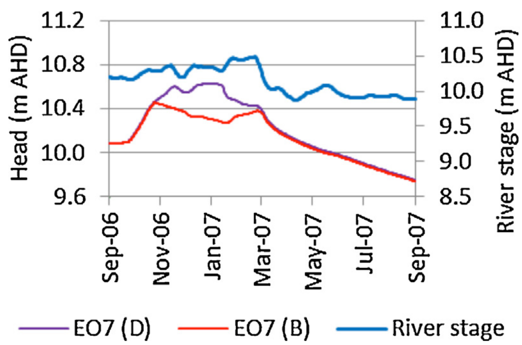


Fig. 12. Simulated groundwater heads at observation well EO7 for scenarios B and D during the injections.

fluctuations rather than the injection trial. In fact, observation wells EO4 and EO7 located closer to the river banks are influenced with the river fluctuations rather than the injection trial.

The groundwater salinity results show that the calibrated model is able to reproduce the observed dynamic in an acceptable manner (RMSE (m); EO1: 0.069, EO3: 0.081 and EO4: 0.074). The simulated results are consistent with the EM31 survey which was conducted on 2 November 2006 before the aquifer breach. As a general pattern, two distinct zones were identified, namely a near-river fresh zone (includes EO4 and EO7, with salinity less than 5000 μS cm^{-1}) and a saline zone over the rest of the floodplain (Fig. 7a). However, relatively fresh zones were formed in the immediate vicinity, approximately 10–15 m from each injection well but this did not extend to the adjacent observation wells, with the exception of EO1. A reduction in the groundwater salinity (from 26,000 μS cm^{-1} to 10,000 μS cm^{-1}) was only exhibited at EO1 which is situated less than 5 m from EI5 (Fig. 7b). Soil water availability at the capillary fringe (depths greater than 2 m) displayed a similar behaviour (Berens et al., 2009a).

3.2. Scenarios

Seven scenarios are defined to explore different river water injection strategies (Table 2). All the scenarios run for 1 year from 1 September 2006 to 1 September 2007 (366 time steps). To be able to compare the numerical model results to the conducted field trial at the study site, the injection simulation commences on 19 September 2006 (time step 19) in each scenario.

3.2.1. Injection rate

Scenarios A, B and C inject a total of 10 ML of fresh river water to the saline floodplain aquifer starting from day 19. They differ by the rate at which the water is injected. Scenario A injects over 92 days at 1.25 l s^{-1}, Scenario B injects over 57 days at 2 l s^{-1} and Scenario C injects over 23 days at 5 l s^{-1}. Fig. 8 displays the simulated groundwater head dynamics at observation wells EO1 and EO4 during scenarios A, B and C. The water table response appears to be proportional to the distance from the injection well. For instance, observation well EO1 which is located 5 m from injection well EI5 shows a rapid 0.9 m increase in scenario A, while in the same scenario the groundwater head at observation well EO4 (30 m further) increases by less than 0.3 m. Due to the high hydraulic conductivity of the floodplain aquifer (Monoman Sand), the groundwater head fluctuation is almost instant. Fig. 8a shows that higher injection rates lead to higher groundwater heads. Furthermore, it can be seen that when the injection ceases, the

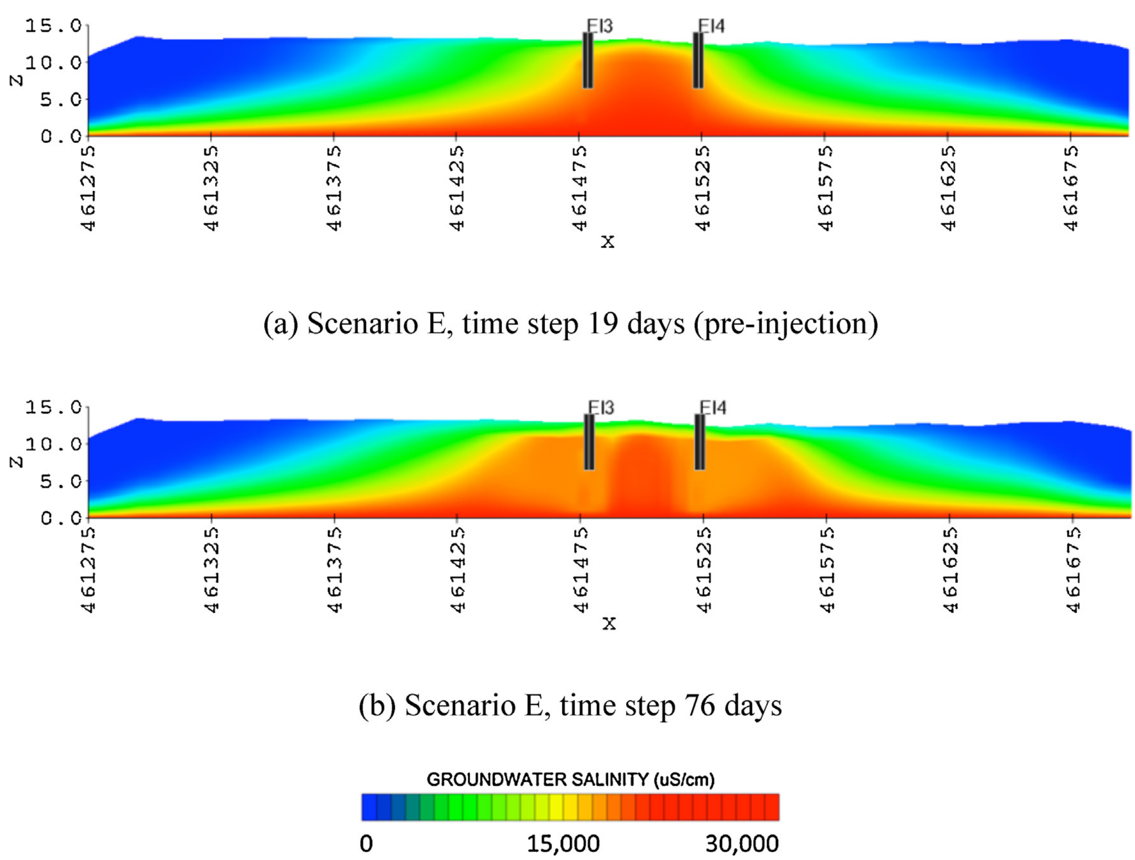


Fig. 13. Simulated groundwater salinity distribution for scenario E at pre and post injection time steps (Z magnification = 3).

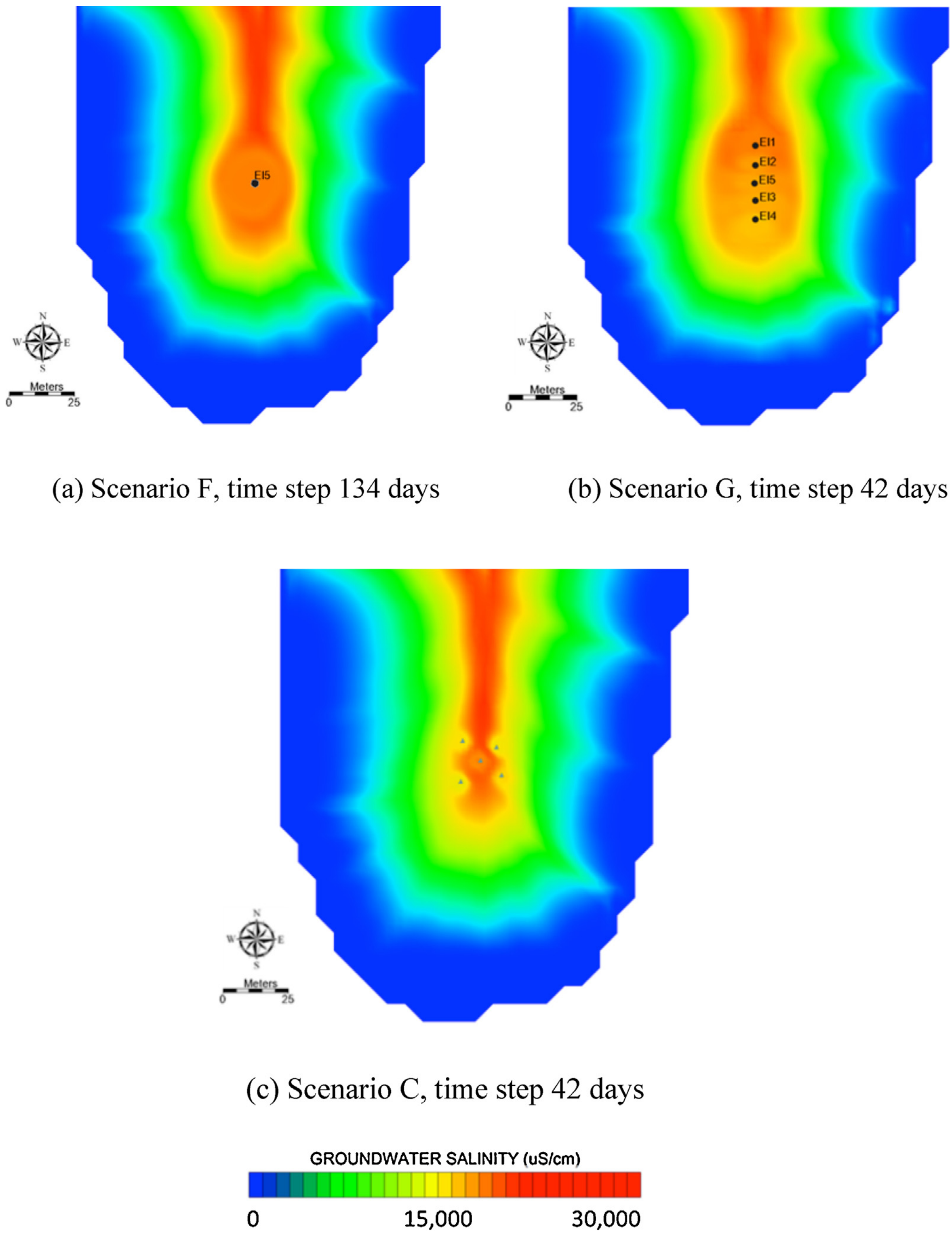
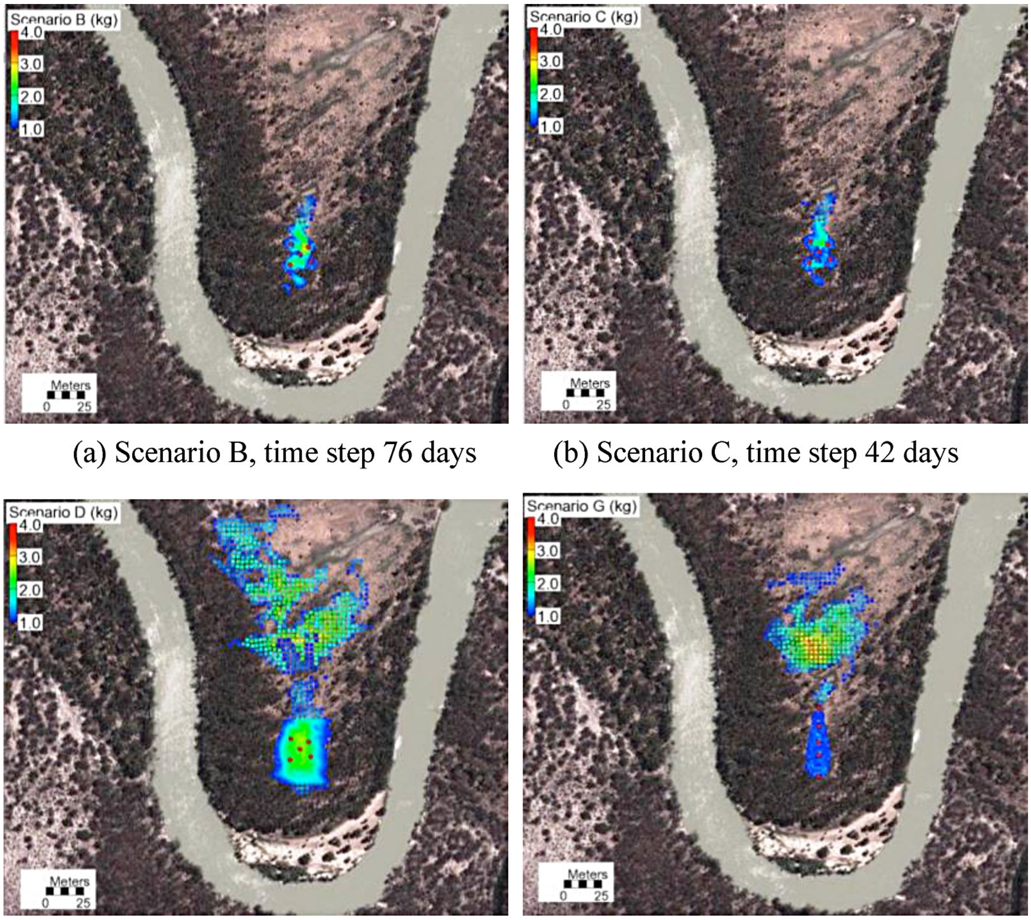


Fig. 14. Simulated groundwater salinity distributions for scenarios C, F and G.

groundwater dynamic is a function of the river fluctuation. Thus, during the river recession from March 2007 to September 2007, a decrease in groundwater head can be observed.

Fig. 9 shows the simulated groundwater salinity dynamics for scenarios A, B and C. Observation well EO1 responds strongly to the injection. However, a similar pattern is not apparent at observation well EO4 which is located further away and closer to the river. Moreover, the three scenarios can maintain a fresh water lens (at salinities less than 10,000 μS cm^{-1}) in the vicinity of the injection wells (Fig. 10). The formation of the fresh water lens takes 53, 34 and 18 days for scenarios A (lowest injection rate), B and C (highest injection rate), respectively. However, the three scenarios can displace the saline groundwater for the same duration. Both field and laboratory studies have found that groundwater salinity 15,000–20,000 μS cm^{-1} causes river red gum death (Eamus et al., 2006). In this case, the three scenarios can maintain the groundwater salinity at less than 15,000 μS.cm^{-1} for approximately 165 days (Fig. 9a). Moreover, the extent of the freshwater lens induced by the fresh river water injection can be interpreted according to Fig. 10. This shows the groundwater salinity along transect 1 at the last time step of the injection. It appears that the lateral magnitude of freshening is similar in all three scenarios, despite the different injection rates. Overall, it appears that for given fresh river water volumes, the three defined injection rates form the same freshwater lens extent. However, technically it should be considered that higher injection rates may increase the risk of aquifer breach due to increased pressure heads. Therefore, because of the same freshwater lens extents, lower injection rates seem to be more appropriate. Furthermore, scenarios A, B and C all show an increase in groundwater salinity between March 2006 and



(a) Scenario B, time step 76 days (b) Scenario C, time step 42 days

(c) Scenario D, time step 134 days (d) Scenario G, time step 42 days

Fig. 15. Simulated distribution of solute mass mobilization from the unsaturated zone for the defined scenarios.

September 2007. This is attributed to the river stage recession which leads to floodplain recharge by the saline regional groundwater aquifer.

3.2.2. Injected volume

The results of scenario D are useful for exploring the impact of fresh river water injection on the saline floodplain aquifer. Fig. 11 shows scenario D at time step 134 days which corresponds to the time by which a total of 20 ML of fresh river water has been injected into the floodplain aquifer. Comparing Figs. 11 and 10b show that, for a given injection rate ($2\,l\,s^{-1}$), an increased volume of injected fresh river water considerably increases the extent of the freshwater lens. Moreover, Fig. 12 shows an increase in groundwater head (up to 0.3 m) at observation well EO7 is induced by scenario D. This was not observed during scenarios A, B and C. Overall, it is shown that with a given injection rate, delivering more injected water leads to more effective floodplain salinity mitigation.

3.2.3. Injection screen depth

To explore the impact of injection screen depth on the freshwater lens, scenarios B and E can be compared. In both of these scenarios, 10 ML of river water is injected at a rate of $2\,l\,s^{-1}$, but with different injection screen depths (B: 3–6 m depth and E: 3–10 m depth). Fig. 13 shows the simulated floodplain aquifer salinity at the first (19 days) and last (74 days) time steps of the injection for scenario E. From Figs. 13 and 10b, it can be seen that locating the injection screen at the interface of the saturated/unsaturated zone (in this case at a depth of 3.5 m) is more effective for forming a freshwater lens. Furthermore, it appears that deeper injection depths cannot create a freshwater lens at the interface of the saturated/unsaturated zone, and even it may expands the saline groundwater plume (See Fig. 13a and b). In fact, injection of fresh water beneath the thick layer of saline groundwater fails to significantly mitigate the local aquifer salinity and create the freshwater lens. This configuration pushes the saline groundwater plume upward and increases the overall floodplain shallow aquifer salinity. It seems that selection of an appropriate injection screen depth is vital to obtain an efficient outcome. Otherwise, it may be detrimental for the health of the targeted vegetation. Consideration needs to be placed on the near surface (top of aquifer) groundwater displacement and the ability to create a freshwater lens to target the tree root zone, rather than the displacement of water at greater depths in the aquifer.

3.2.4. Injection pumps configuration

In the base case model and scenarios A–E the five injection wells are configured as a rectangular pattern. Scenarios F and G are defined to examine the impact of the configuration of the injection wells on the outcome of the floodplain aquifer freshening. Scenario F injects the 10 ML of fresh river water via one injection well (single-point). Fig. 14a–c shows a plan view of the simulated groundwater salinity distribution for scenarios C, F and G at the end of the injection period. It can be seen that at the end of the injection period different patterns can be observed. Scenario F leads to a circular dispersion of the fresh water and groundwater freshening, while scenario G forms a larger freshwater lens along the straight line where the injection wells are located. Even scenario G seems to be more effective at freshening the floodplain aquifer compared to scenario C. This shows that for given injection volumes, the extent of induced saline groundwater displacement can be different depending on the injection well configuration. Depending on the topography of the floodplain and distribution of the targeted stressed trees, different configurations can be considered to deliver the injected fresh water more efficiently.

3.2.5. Solute mass mobilization

Due to an altered overbank flooding regime, natural saline groundwater and high rate of evapotranspiration, there is often a significant amount of solute mass stored in the unsaturated zone in arid and semi-arid floodplains. Therefore, one of the benefits or aims of any salt management measure is to mobilise some of the solute mass stored in the unsaturated zone. Fig. 15 shows the distribution of the solute mass mobilization induced by scenarios B, C, D and G. The results of the model show that up to 4 kg m^3 of solute mass can be mobilised from the unsaturated zone.

It appears that scenarios B and C, despite different injection rates, mobilise almost the same amount of solute mass from the unsaturated zone, but that this is limited to the immediate vicinity of the injection wells. Moreover, injecting 20 ML of fresh river water in scenario D leads to the most effective outcome in term of solute mass mobilization from the unsaturated zone. After scenario D, scenario G shows the second most effective solute mass mobilization which is attributed to its injection well configuration.

4. Conclusion

Riparian trees in the Lower Murray River have been declining. An aquifer injection strategy was used to deliver fresh river water to the interface of the saturated/unsaturated zone to increase water availability for stressed trees. The results confirmed that the volume of injected water is the dominant factor to achieve an effective result. An increased injection volume can form a more extensive freshwater lens and can also mobilise more solute mass from the unsaturated zone. In addition, it was shown that for given injection volumes, injecting at lower injection rates and for longer durations produces a more sustainable and efficient strategy. Furthermore, the numerical model results showed that choosing a proper injection screen depth is vital for a successful outcome. It was shown that injecting the freshwater deep into the aquifer and beneath a thick layer of saline groundwater may lead to upward movement of saline groundwater which is unfavourable in terms of vegetation health. Moreover, it was demonstrated that the number and configuration of injection wells also plays an important role in the distribution of injected fresh water. Overall, it was demonstrated that fresh river water injection is a plausible mechanism to create a thin freshwater lens at the saturated/unsaturated interface and this may improve plant and tree health. This salt management measure is able to mobilize some of the solute mass stored in the unsaturated zone but this mainly depends on the injection well configuration and total volume of injected water. However, fresh river water injection can only maintain a temporary and spatially limited local freshwater lens which can be applied as short-term management strategy for protecting the health of declining vegetation. For long-term strategies, the injection needs to be periodically repeated unless there is overbank flooding in the meantime. This research is one of the first attempts to incorporate observed data into a physically-based, fully integrated model to explore such complex river-floodplain interactions.

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