Interaction between shallow groundwater, saline surface water and contaminant discharge at a seasonally and tidally forced estuarine boundary

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Abstract

This paper presents findings from a 2-year field investigation of a dissolved hydrocarbon groundwater plume flowing towards a tidally and seasonally forced estuarine river system in Perth, Western Australia. Samples collected from transects of multiport wells along the riverbank and into the river, enabled mapping of the fine scale (0.5 m) vertical definition of the hydrocarbon plume and its longitudinal extent. Spear probing beneath the river sediments and water table, and transient monitoring of multiport wells (electrical conductivity) was also carried out to define the zone of mixing between river water and groundwater (the hyporheic zone) and its variability. The results showed that groundwater seepage into the estuarine surface sediments occurred in a zone less than 10 m from the high tide mark, and that this distance and the hyporheic transition zone were influenced by tidal fluctuations and infiltration of river water into the sediments. The dissolved BTEXN (benzene, toluene, ethylbenzene, the xylene isomers and naphthalene) distributions indicated the behaviour of the hydrocarbon plume at the groundwater/surface water transition zone to be strongly influenced by edge-focussed discharge. Monitoring programs and risk assessment studies at similar contaminated sites should therefore focus efforts within the intertidal zone where contaminants are likely to impact the surface water and shallow sediment environments.

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1. Introduction

As urban and industrial development continues to expand around the world’s rivers and coastlines, so does the rate of unintentional release of contaminants to subsurface and surface waters and the need for...
effective assessment of such environments (Winter, 2000). Hydrologists have long known that surface waters and groundwater are intrinsically linked systems (e.g. Glover, 1959; Cooper, 1959; Clement et al., 1996; Simpson et al., 2003). Areas around streams, rivers, lakes and coastal environments represent zones of interaction and transition between the two systems where dissolved constituents such as pollutants can be diluted, exchanged, transformed or destroyed. Identifying predominant processes affecting solute exchange across transition zones is therefore, critical in assessing contaminant fluxes to the sediment/water interface, and ultimately in estimating contaminant exposures for the receiving ecosystems.

Groundwater/surface water interactions in estuarine environments are influenced by a number of processes forming complex spatially and temporally variable systems. Density contrasts between the typically fresh groundwater and saline to brackish marine and estuarine surface waters leads to mixing and convective circulation at the groundwater discharge boundary so that the system is characterised by the intrusion of saltwater into the adjacent coastal aquifer (Glover, 1959; Cooper, 1959; Reilly and Goodman, 1985; Ataie-Ashtiani et al., 1999; Simpson and Clement, 2004). Tidal activity can often induce a fluctuating water table as well as infiltration of surface water into sediments, forming a surficial mixing zone with groundwater discharging from the adjacent aquifer (Robinson et al., 1998; Ataie-Ashtiani et al., 1999; Boudreau and Jorgensen, 2001; Acworth and Dasey, 2003). Although there is still no single conceptual definition for such a surficial mixing zone, the terms ‘hyporheic zone’, ‘subsurface estuary’ and ‘groundwater/surface water interface’ or ‘GSI’ are gaining common usage in the scientific literature. White (1993) conceptually defined the hyporheic zone as ‘the saturated interstitial area beneath the stream bed and into the stream banks that contain some proportion of channel water or that have been altered by channel water infiltration’. This definition may be broadened to include rivers, lakes, estuaries and coastal environments where surface water infiltrates into the underlying sediments and interacts with groundwater.

Although numerous studies have addressed groundwater and solute inputs to surface water bodies (e.g. Harvey et al., 1987, Gallagher et al., 1996, Portney et al., 1998, Krabbenhoft et al., 1990, Lorah and Olsen, 1999, Winter, 2000; Tobias et al., 2001), few studies to date have examined near-shore groundwater discharge in detail. Studies of note however, include those by Robinson et al. (1998); Robinson and Gallagher (1999); Smith and Turner (2001); Linderfelt and Turner (2001); Simpson et al. (2003) and the initial study by Westbrook et al. (2000) related to the current work. Robinson et al. (1998) presented results from a field investigation on unconfined groundwater discharge to estuarine waters at Chesapeake Bay, Virginia, showing strong tidal and seasonal controls on fresh groundwater discharge associated with infiltration of surface water into tidally exposed sediments. Robinson and Gallagher (1999) further developed a two-dimensional, field scale, finite-element model based on density dependent fluid flow with water table and dynamic tidal boundary conditions. The model was able to reproduce the Chesapeake Bay field data on the movement of the near-shore water table, groundwater salt concentrations and groundwater discharge rates and patterns but was unable to replicate short-term salt fluctuations in the hyporheic zone due to the wave action of tides within the intertidal zone (Robinson and Gallagher, 1999). Simpson et al. (2003) performed transport experiments in a sand tank to study the characteristics of the seepage-face zone that exists near a groundwater/surface water interface. Their study concluded that seepage-face zones, which are dominated by strong hydraulic gradients, play an important role in influencing the localized flow and solute transport processes in shallow unconfined aquifers.

Field observations and mathematical modelling of density dependent groundwater/surface water interaction in the Swan-Canning estuary by Smith and Turner (2001) showed that density contrasts between the estuary and adjacent fresh groundwater system are sufficient to drive mixed-convection cells that give rise to circulation of river water in the aquifer, providing a mechanism to transport nutrients between the nutrient-rich pore fluids in the riverbed sediments and groundwater. Further, results indicated unconfined groundwater preferentially discharges into the Swan River along the outside of river meanders, with very low discharges or at times saline river water recharge along the inside of meanders (Smith and Turner, 2001;
Linderfelt and Turner, 2001). Short-term fluctuations such as wave-induced displacement, hyporheic flux and tidal oscillation appeared to cause nutrient release from the sediment pore fluids, particularly in low flow summer periods (Linderfelt and Turner, 2001).

Although these findings highlight the complexities of groundwater discharge at an estuarine boundary, to the author’s knowledge no studies have investigated the importance of the near-shore processes in understanding the transport of point-source contaminant plumes across the estuarine transition zone. This paper presents findings from a 2-year field investigation of a dissolved phase gasoline groundwater plume, primarily comprised of BTEXN (benzene, toluene, ethylbenzene, the xylene isomers and naphthalene), that discharges to an estuarine beach on the foreshore of the Canning River in Perth, Western Australia. These compounds are known or suspected carcinogens and may represent a threat to the quality of fresh water resources (see, for example ANZECC, 1992). Interpretation of the data is used to develop an increased understanding and improved conceptual model of groundwater/surface water dynamics, especially for this study site. The model identifies spatial and temporal characteristics of groundwater/surface water interactions and examines the effects of these interactions on contaminant discharge to the estuarine environment. Ongoing research at the site uses the conceptual model as a basis for investigations into geochemical controls on groundwater quality and contaminant fate in the near-shore region.

2. Site location and background data

The study site (Figs. 1 and 2) is situated on the foreshore of the Canning River, approximately 6-km south-southwest of Perth’s central business district, Western Australia, and approximately 15 km upstream from the Indian Ocean. Groundwater at the site is impacted with dissolved-phase petroleum hydrocarbons, including the BTEX (benzene, toluene, ethylbenzene, and xylene isomers) compounds and naphthalene (N). These impacts originate from a historical, unintentional, light nonaqueous phase liquid (LNAPL) release from an underground storage tank (UST) located approximately 80 m from the river’s edge. The plume follows an easterly flow path from the UST source as governed by the local groundwater hydraulics.

The Canning River forms part of the Swan-Canning river and estuary system, which is characterised by cyclical density stratification related to changing surface flow dynamics. The estuary system is tidal, being open to exchange with the Indian Ocean and seasonally forced by dominant winter rainfall. During spring and summer a high-density wedge of saline ocean water intrudes into the estuary and it’s rivers, reaching as far as 45 km upstream from the ocean. High rainfall periods during autumn and winter give rise to fresh water flushing of the estuary causing regression of the saline wedge. Smith and Turner (2001) showed that the seasonal density contrasts between the estuary and the adjacent fresh water aquifers were sufficient to drive mixed-convection cells and saline water into the adjoining surficial aquifers.

2.1. Hydrogeology of the site

Groundwater of the regional Perth area is dominated by a shallow, unconfined Quaternary ‘superficial aquifer’ system, overlying confined tertiary and mesozoic sedimentary formations of the Perth basin.
The superficial aquifer sediments vary in composition from predominantly clayey in inland areas to sand and limestone successions in the coastal plain areas. The sediments have an average thickness of 20–45 m. Around the Swan-Canning estuary, the sediments are characterised by sequences of interbedded sand, silt and clay that directly overly the confining shale and siltstone of the early Tertiary Kings Park Formation (Davidson, 1995).

The shallow (top 10 m) hydrogeology at the site comprises a stratigraphy of well-sorted, medium-grained, unconsolidated quartz-rich sediments with thin shelly horizons underlain by a dense clay layer intersected at around 7.5 m below surface. The aquifer’s saturated porosity ranges from 0.25 to 0.3 m$^3$ m$^{-3}$, and the hydraulic conductivity ranges from 1 to 10 m day$^{-1}$. The average depth to water from the ground surface ranges from 2.02 m in the near source area at MW02 to 0.94 m at the foreshore at MW33 (see locations on Fig. 2). The hydraulic gradient from the source to the river foreshore varies seasonally between 0.005 and 0.01 m m$^{-1}$.

Dissolved oxygen concentrations of < 1 mg l$^{-1}$ indicated that the groundwater is typically anaerobic.

2.2. Historical contaminant characterisation data

Environmental assessments of the area since the early 1990s identified two areas of subsurface contamination: (1) a northern zone around the source at the site of the present study, and; (2) a southern zone situated at a site where underground storage tanks were previously located. Results from the initial site assessment showed that the subsurface was contaminated with hydrocarbon compounds. Soil contamination and dissolved phase total petroleum hydrocarbons (TPH) concentrations of > 5000 ppm and > 2 ppm, respectively, were reported from monitoring wells MW-01 and MW-02 situated at the UST location. Analysis of samples for hydrocarbon based on chain length indicated that both gasoline and diesel were possible source fuels.

Up to 37 monitoring wells had been installed at various localities around the contaminated site.
between 1993 and 1999. Most of these monitoring wells were installed to depths of 3 m below ground surface (approximately 1–2 m below the water table). The wells were screened over their whole depth intervals. Historic data from periodic sampling indicated a groundwater plume moving in an easterly direction away from the source towards the Canning River (see Fig. 2), with benzene concentrations in MW33, situated on the river bank, exceeding the ANZECC (1992) guideline value of 300 µg l⁻¹ in May 1996, October 1997, December 1998 and March 1999.

In December 1999, an additional three monitoring wells were installed (MWs 41–43) to depths of 6 m below ground level and screened over 2.5–6 m depth intervals. Subsequent sampling of MW43, situated 20 m west of the river bank showed dissolved phase hydrocarbon concentrations up to 1000 µg l⁻¹ benzene, 955 µg l⁻¹ ethylbenzene and 1600 µg l⁻¹ xylene. The presence of elevated BTEX concentrations at depths not previously sampled at the site suggested that the dissolved phase plume had migrated downward in the aquifer and highlighted the uncertainties involved when using shallow, fully screened wells to monitor groundwater plumes.

3. Methods

3.1. Water quality monitoring

To aid further definition of the plume, multiport wells (MPs) and spear probing (drive point sampling) were used to collect groundwater samples for analysis. As shown in Fig. 3, the multiport wells had sampling screens at 0.5 m depth intervals and comprised a bundle of individual 2.5 mm internal diameter nylon tubes, each extending to a different depth. Sampling screens (or ports) were made of 0.1 m long slotted stainless steel tubes covered by an inert nylon filter mesh. The MPs were installed by initially auguring to the water table using a half split stem auger followed by insertion of a temporary casing into the hole. Aquifer material was sludged (bailed) out of the casing while the casing was pushed deeper until the required depth was reached. The multiport bundle was then inserted into the temporary casing, which was removed to allow the hole to collapse around the installation. This method was used to minimise the zone of disturbance around each installation.

After initial investigations using historical monitoring data, and pilot multiport and spear probe sampling, a transect approach was employed to monitor contaminant distributions. In this approach the principal axes of the contaminant plume were designated x, y and z, with x being the longitudinal direction of flow, y the lateral direction perpendicular to flow, and z the vertical direction. By orientating the sampling transects along the principal axes, lateral (y–z), longitudinal (x–z) and plan (x–y) control sections were mapped and a three-dimensional picture of the plume was constructed.

In total, five control sections were installed and sampled at the site, consisting of three lateral sections (A–A’, B–B’, C–C’), one longitudinal section (D–D’) and one plan section (see Figs. 2 and 4). The lateral sections, comprised of permanent multiport wells, were installed to capture the depth distribution of contaminants across the plume in both the on-shore and in-river sediments. The longitudinal section...
utilised both multiport wells and spear probe sampling to monitor groundwater quality and contaminant distribution along the estimated centre line of flow of the plume as it approached and discharged to the river. The plan section, which used only spear probe sampling over a grided area (2 m spaced north-south grid lines and 1 m east–west sample intervals), was used to investigate water quality and contaminant distribution in the area of groundwater discharge near the sediment/water interface. Samples from the plan section were taken from 0.20 m below the surface of the riverbed or 0.20 m below the groundwater table in the on-shore section of the sampling grid.

Two major monitoring events, designed to provide snap-shots of the plume distribution during summer and winter conditions, were carried out in January and August/September 2001, respectively. The MPs 9–13 were only sampled in August/September 2001. The monitoring events comprised sampling of all installed multiport wells for the lateral (y–z) control sections as well as spear probe sampling of the longitudinal (x–z) and plan (x–y) control sections. Sampling of selected multiport wells or control sections was also carried out intermittently over the investigation in order to obtain data for analysis of temporal trends.

Groundwater samples were recovered via suction directly from the access lines of the multiport boreholes using glass syringes after purging of the lines. This strategy was adopted to minimise disturbance of the groundwater and sorptive and/or volatile losses of the BTEX compounds (Davis et al., 1992, 1999). Groundwater samples were recovered from the spear probe in a similar manner, using a syringe to draw the sample through a nylon access tube from a small-screened interval at the head of the spear. Groundwater quality parameters measured at each sample location and depth were dissolved oxygen (DO), electrical conductivity (EC), pH and redox potential (or Eh). Groundwater samples taken from the multiport boreholes and spear probing were analysed for benzene, ethylbenzene, toluene, m- and p-xylene, o-xylene, 1,3,5-trimethylbenzene, naphthalene, 1-methylnaphthalene, 2-methylnaphthalene, m- and p-cresol, o-cresol and phenol. Duplicates were always taken for at least one or two locations per sampling period, and blanks were also analysed. The organic compounds were concentrated from groundwater samples using micro-solvent extraction techniques (Patterson et al., 1993), and analysed by Gas Chromatography.

3.2. Water level monitoring

River and groundwater levels were monitored via four online water level loggers. One logger monitored river water levels, while the remaining three monitored groundwater levels at increasing distances away from the river foreshore along the proximal path of the contaminant plume. The groundwater level loggers were located in existing monitoring wells located at 8 m (MW33), 28 m (MW43) and 55 m from the shoreline.

4. Results and discussion

4.1. On-shore distribution of contaminants

In the on-shore sediments between multiport transects A–A’ and B–B’, the distribution of the dissolved
BTEXN was as a thin, lenticular plume in cross-section, with approximate widths of 20–30 m (Fig. 5). However, at a finer concentration range, the width of the benzene plume near the source is much wider—possibly greater than 40 m wide. Table 1 compares the highest BTEXN (benzene, toluene, ethylbenzene and the xylene isomers).

Table 1

<table>
<thead>
<tr>
<th>Depth</th>
<th>Concentrations (µg L⁻¹)</th>
<th>Ratio of organics to benzene</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>B</td>
<td>T</td>
</tr>
<tr>
<td>A–A'</td>
<td></td>
<td></td>
</tr>
<tr>
<td>MP09</td>
<td>2.7</td>
<td>459</td>
</tr>
<tr>
<td>MP10</td>
<td>3.2</td>
<td>82</td>
</tr>
<tr>
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<td>195</td>
</tr>
<tr>
<td>MP12</td>
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<td>233</td>
</tr>
<tr>
<td>MP13</td>
<td>3.7</td>
<td>371</td>
</tr>
<tr>
<td>B–B'</td>
<td></td>
<td></td>
</tr>
<tr>
<td>MP03</td>
<td>1.1</td>
<td>7</td>
</tr>
<tr>
<td>MP02</td>
<td>2.5</td>
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<tr>
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<td>3.9</td>
<td>333</td>
</tr>
<tr>
<td>MP01</td>
<td>1.2</td>
<td>789</td>
</tr>
<tr>
<td>MP04</td>
<td>0.7</td>
<td>103</td>
</tr>
</tbody>
</table>

Fig. 5. Lateral depth-sections through the dissolved plume showing benzene, TEX and naphthalene concentrations in the near-source (A–A') multiport well control sections. AMG, Australian map grid, AHD, Australian height datum, MP, Multiport well. TEX = toluene, ethylbenzene and the xylene isomers.
(MPs 1–4 and 8) multiport wells and shows ratios of TEXN organics to benzene concentrations. Fig. 6 also shows detailed BTEXN distributions with depth from the centreline multiport wells.

Contaminants in the near-source section (A–A', Fig. 5) were observed to impact the groundwater over a thin vertical zone of approximately 1 m, at depths between 5.5 and 6.5 m below ground level (also see MP-11 data in Fig. 6). The contoured groundwater hydrocarbon concentration data in section A–A' (Fig. 5) was mostly dominated by the xylene, ethylbenzene and naphthalene compounds, with concentrations in MP11 (Fig. 6) reaching up to 13,850, 4401 and 806 \( \mu \text{g l}^{-1} \), respectively, in September 2001, while benzene was found at lower concentrations of 195 \( \mu \text{g l}^{-1} \). Benzene, however, was also observed as a more laterally persistent plume, extending further to the north where it becomes the dominant hydrocarbon in MP09 (see Table 1).

Contaminant levels and distributions in the near-shore section (B–B') indicate a down gradient vertical thickening and lateral narrowing of the plume combined with shallowing and southward migration of the plume centreline. Increased dispersion due to fluctuating near-shore groundwater pressure gradients (see Section 4.3) and possibly increasing vertical groundwater flow directions may explain the observed vertical thickening of the plume along transect B–B'. Benzene and naphthalene are the dominant hydrocarbons impacting groundwater, while the TEX compounds were almost at negligible concentrations (see Table 1).

Table 2 presents dissolved mass calculations for the individual and total BTEXN compounds in sections A–A' and B–B' for September 2001. The contaminant mass figures stated are based on an average section length of 27 m, a saturated aquifer thickness of 6 m, a control section width of 1 m and an aquifer porosity of 0.25. Using average contaminant concentrations calculated from multiport sampling, the dissolved mass was estimated using the equation (Wiedemeier et al., 1999)

\[
DM_t = C_{\text{avg}, t} \cdot bnLW
\]

where \( DM_t [M] \) is dissolved mass at time \( t [T] \), \( C_{\text{avg}, t} [\text{ML}^{-3}] \) is the average concentration at time \( t \), \( n \) is...
the aquifer porosity $[L^3L^{-3}]$, $b$ is the aquifer thickness $[L]$, $L$ $[L]$ is the lateral plume length and $W$ $[L]$ is plume width.

Clearly the near-source plume section is significantly higher in total contaminant mass with ethylbenzene and xylene comprising the dominant portion. Ratios to benzene along the plume centreline (i.e. through MP11/MP12 towards MP08/MP01) showed that the significant mass loss and probably biodegradation of the non-benzene compounds may have occurred. Benzene concentrations showed no enhanced depletion trend along the down gradient flow path, which is in accord with observations from a BTEX plume in a similar aquifer formation in Perth (Davis et al., 1999). Although this data suggests loss of contaminant mass along the plume flow path, it is recognised that it is difficult to draw such conclusions based on ‘snap shot’ sampling alone.

### 4.2. Groundwater/surface water transition zone

Mapping of electrical conductivity and contaminant concentration data from the in-river and near-shore transects indicated the groundwater/surface water transition zone at the site to be characterised by: (1) saline water intrusion (salt wedge) landwards at the base of the aquifer; (2) shore focussed groundwater and plume discharge and (3) formation of a shallow hyporheic zone due to brackish/saline river water infiltration and mixing with groundwater below the sediment/water interface. These features are illustrated in the longitudinal (D–D’) and the plan sections from January and September 2001, as shown in Figs. 7 and 9.

Salinity differences between the groundwater and river water allow electrical conductivity measurements to be used to map the distribution of water types.

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**Fig. 7.** Pore water electrical conductivity ($\mu S \text{ cm}^{-1}$, shaded contours) and total BTEXN concentration ($\mu g \text{l}^{-1}$, line contours) from the longitudinal transect D–D’ for (A) January and (B) September 2001.
and transition zones in the subsurface, and the discharge of fresh groundwater towards the sediment/water interface. At the Canning River field site, fresh groundwater is characterised by EC values of $<1000 \mu S \text{ cm}^{-1}$, while river water EC is variable, generally ranging from 4000 $\mu S \text{ cm}^{-1}$ during winter months to 47,000 $\mu S \text{ cm}^{-1}$ during summer.

4.2.1. Longitudinal trends (transect D–D$'$)

Fig. 7 shows closely contoured electrical conductivity values along the longitudinal transect D–D$'$ for January and September 2001. Total BTEXN concentrations are indicated as line contours overlying the conductivity data. Saline water intrusion into the aquifer is clearly indicated by elevated pore water electrical conductivity underlying the fresh groundwater. At the salt wedge/groundwater interface, electrical conductivity values were transitional over a depth interval of 1.0–1.5 m, steadily increasing up to 45,000 $\mu S \text{ cm}^{-1}$ in the main body of the saltwater wedge. With increasing distance along the flow path towards and into the river sediments, the fresh water lens was observed to thin above a thickening salt wedge. Consequent groundwater discharge towards the sediment surface was observed, resulting in either the formation of a seepage face at the sediment surface or mixing with river water in a hyporheic zone. In this case, we define the hyporheic zone as the region beneath the sediment/surface water interface (i.e. the river bed) with electrical conductivity values between that of the onshore groundwater (i.e. $>1000 \mu S \text{ cm}^{-1}$) and river water (seasonally variable). Both the seepage face and hyporheic zones were variable in space and time—as shown by the changes in seepage zone length and EC when comparing January and September 2001 data (Fig. 7A and B). Changes in the distribution of the hyporheic zone are primarily a function of the local surface water (tidal oscillation) and groundwater hydraulics (this is discussed later). BTEXN concentrations along the transect showed the leading front of the plume ($500 \mu g \text{ L}^{-1}$) extending a maximum of only some 2.5–5 m into the river sediments from the shoreline or 10–12 m east of MP08, with abrupt lateral decreases in contaminant concentration occurring in the area of hyporheic mixing.

Depth profiles of electrical conductivity in shallow pore water along transect D–D$'$ (as depicted in Fig. 7) clearly show the higher salinity water of the hyporheic zone situated above fresh groundwater. The general trend was for the depth and salinity of the hyporheic zone to increase with increasing distance away from the shoreline. Conversely, the thickness of the fresh groundwater lens steadily decreases and is eventually pinched out where the salt wedge becomes continuous with infiltrating river water. The maximum hyporheic zone thickness recorded was at approximately a depth of 2 m into the river sediments.

4.2.2. Lateral distributions (transect C–C$'$)

Contaminant distributions from the in-river lateral section C–C$'$ (Fig. 8), are consistent with vertical flow dominated, shore-focussed behaviour of groundwater discharge as indicated by results from section D–D$'$. Elevated BTEXN levels were only intermittently recorded from sampling points in the upper 2 m of sediments indicating the rapid shallowing of the plume in the intertidal zone.

![Fig. 8. Depth distribution of BTEXN concentrations in MP05, MP06 and MP07 along transect C–C$'$ in river sediments during September 2001.](image-url)
4.2.3. Plan view distributions (rectangular grid)

Grid-based spear-probe sampling of shallow pore water (0.2 m below the sediment surface, Fig. 10) at the river discharge boundary showed BTEXN compounds strongly coincident with the detection of fresh to slightly brackish pore water. Fig. 4 shows the location of the rectangular grid used for this sampling. The distribution of measured electrical conductivity values and total BTEXN concentrations for summer (January) and winter (September) periods are shown in Fig. 9.

The EC data show there were strong transient variations in discharge patterns that may be attributed to a combination of seasonal and tidal forcing. Data from the January sampling (Fig. 9, A and B) are consistent with characteristic summer hydraulic conditions where river salinity approaches a maximum

Fig. 9. Distribution of electrical conductivity (A and C) and total BTEXN concentrations (B and D) from spear probe sampling of the plan-section, during January (A and B) and August/September (C and D) 2001. Samples were recovered from 0.2 m below the river bed or 0.2 m below the water table (if over land).
and groundwater discharge is low. In this case, groundwater discharge was restricted to a small zone of limited lateral extent. Sustained high tidal stages during the January sampling period are likely to have compounded this effect, disturbing groundwater pressure gradients and stimulating hyporheic mixing below the sediment/water interface. Conversely, the August/September sampling period was designed to be representative of winter conditions where river salinity was low and groundwater discharge was relatively high. Overall, the field data (Fig. 9, C and D) showed a significant spatial increase of groundwater discharge to the river sediments, with contaminants travelling further into the river before being expressed near the surface. Low tidal stages during the September sampling are likely to have contributed to the enhanced discharge of groundwater across the sediment/water interface.

Of the hydrocarbon compounds, benzene and naphthalene concentrations were highest in the spear probed samples, as was the case for the MP samples nearer the river. The concentration of benzene ranged up to 550 µg l⁻¹ and naphthalene ranged up to 304 µg l⁻¹. Typically, however, maximum benzene and naphthalene concentrations were 100–300 µg l⁻¹, where detectable.

4.3. Tidal controls at the discharge boundary

Monitoring of river stage and on-shore water table heights showed that near-shore groundwater levels were strongly responsive to tidal oscillations. Water table heights measured approximately 8 m landward of the shoreline in MW33 are shown in Fig. 10 with simultaneous river stage heights over a 10 day period in early September 2001. Periods of groundwater flow reversal, where there is a net movement of river water into the groundwater system, were initially inferred during some high tide events when river water levels exceeded those of the near shore groundwater levels. However, monitoring at MP04 (also 8 m inland from the shoreline) over tidal periods showed no change in groundwater quality, with the salt wedge interface retaining a stable position. Pore water in river sediments showed infiltration of river water across the sediment/water interface to form a hyporheic zone but also indicated a stable salt wedge interface below the fresh groundwater lens. This suggests that over the time scale of a tidal period, the observed changes in near-shore groundwater levels occurred in response to changing pressure heads in the intertidal region of groundwater discharge. Tidal oscillations appeared to have had little effect on salt wedge intrusion distances into the aquifer. These observations are consistent with numerical modelling results published by Ataie-Ashtiani et al. (1999) for scenarios with aquifer depths much larger than tidal amplitudes. Flatter beach slopes are likely to intensify the surface water infiltration phenomenon associated with tidal fluctuations (Ataie-Ashtiani et al., 1999). Monitoring probes at 18 and 50 m inland from the shoreline showed little

![Fig. 10. River stage (Tide) and near-shore (MW33) water table elevations (Groundwater) over a 10-day period in September 2001.](image)

![Fig. 11. River stage and water table elevations (in MW33) over a 24-h period in September 2001. The inset table shows decreasing electrical conductivity of hyporheic pore water at three depths (0.05, 0.25 and 1.0 m below the sediment interface) on the outgoing tide.](image)
response (<0.05 m variation) to the changing river conditions (data not shown).

Transient sampling of water quality in the river sediments showed a relationship between tidal stage and groundwater seepage to the river sediments with freshening of hyporheic pore water on an outgoing tide and high groundwater head relative to the river stage. In Fig. 11, river stage tidal oscillations are compared with groundwater pressures over a 24 h period in September 2001. Electrical conductivity profiles taken over part of the tidal period are shown in the inset table. Changes in pore water pressure relative to the river stage taken in the upper 0.5 m of the sediment column using mini-piezometers indicated vertical pressure gradients of 0.04 m m⁻¹ in the hyporheic zone.

5. Conceptual model

A generalised conceptual model for the interaction between shallow groundwater, saline surface water and contaminant discharge at the Canning River site is shown in Fig. 12. The model, developed from the field observations presented above, shows three dominant hydrodynamic features that affect plume discharge patterns to river sediments at the site: (1) saline water intrusion landwards into the aquifer; (2) edge-focussed groundwater discharge and (3) formation of a shallow hyporheic zone due to saline river water infiltration mixing with groundwater below the sediment/water interface.

Hydrodynamic observations at the Canning River site were consistent with classical seawater intrusion models (Glover, 1959; Cooper, 1959; Reilly and Goodman, 1985; Ataie-Ashtiani et al., 1999)—so, as such, the conceptual model in its simple form as depicted in Fig. 12 is not new. However, the work here has help constrain the magnitude of the extent of the groundwater discharge zone into the river (the order of 5–10 m), its transient behaviour, and the variability induced in concentrations due to tidal and seasonal influences.

![Fig. 12. Conceptual model for groundwater/surface water interaction at the estuarine discharge boundary showing the hyporheic zone with respect to: (A) the predominant hydrodynamic processes and; (B) the general behaviour of the contaminant plume.](image-url)
With increasing distance towards the shoreline, groundwater movement above the saltwedge becomes dominated by vertical flow paths and increased velocities, with groundwater discharge to surface water occurring close to the shoreline (Bear, 1979). Even neglecting density effects, studies have shown that unconfined groundwater discharges preferentially at the shoreline of a surface water body (Duffy and Al-Hassan, 1988; Diersch, 1988). This phenomenon is commonly referred to as edge-focussed discharge (Smith and Turner, 2001). Mapping of the dissolved BTEXN distributions indicated the behaviour of the hydrocarbon plume at the groundwater/surface water transition zone to be strongly influenced by the edge-focussed discharge, with the leading front of the plume extending only some 2.5–5 m into the river sediments from the shoreline. Monitoring programs and risk assessments studies at similar contaminated sites should therefore, focus efforts within the intertidal zone where contaminants are likely to impact the surface environment.

The field observations from this study also show that the hyporheic zone, although often ignored in classical seawater intrusion models, can significantly influence the spatial and temporal discharge patterns of groundwater at the sediment/water interface. Neglect of this zone and its dynamics may introduce significant uncertainties when attempting to estimate groundwater and solute fluxes across the interface. Further, results indicated that the hyporheic zone plays an important role in controlling near-shore groundwater levels. During increasing tide levels, estuarine surface water infiltrates into the sediments and mixes with groundwater to form the hyporheic zone that effectively acts to retard groundwater discharge across the sediment/water interface. During the ebb period, the surface water level recedes past the groundwater discharge zone. This allows the formation of a seepage face after a lag period during which the infiltrated saline surface water is either diluted with groundwater discharge and/or drained down slope. As a result, the intertidal hyporheic zone acts as a fluctuating head discharge boundary.

6. Conclusions

Three-dimensional mapping of water quality and a dissolved petroleum groundwater plume has improved our understanding and conceptual model of groundwater and plume behaviour at an estuarine river discharge boundary. Results indicated that shore-focused discharge strongly influences the area of contaminated groundwater seepage to the estuarine environment. Electrical conductivity measurements show that groundwater seepage to the estuarine surface sediments occurs in a zone less than 10 m from the high tide mark, and mostly less than 5 m from the shoreline. Tidal activity plays an important role, inducing a fluctuating head boundary and the formation of a hyporheic zone due to infiltration of saline surface water in the intertidal sediments. Given that the presence of the hyporheic zone can influence flow, models not properly accounting for its presence will give a higher degree of uncertainty when estimating groundwater and solute fluxes across the sediment/water interface. Seasonal and tidal fluctuations of the interface location plays a role in redistributing hydrocarbon concentrations, governing the lateral extent of discharge into the river, and the proximity of high hydrocarbon concentrations to the sediment/surface water interface.

These observations and conceptual model, along with other results obtained in the study, will be used to more fully model the exchange processes and provide greater confidence in the bio-attenuation and transport processes that may occur where groundwater discharges to such surface water environments.

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References


