

JOURNAL OF ENVIRONMENTAL HYDROLOGY

The Electronic Journal of the International Association for Environmental Hydrology

On the World Wide Web at <http://www.hydroweb.com>

VOLUME 19

2011

IDENTIFYING THE MAJOR VARIABLES CONTROLLING TRANSPORT OF WATER AND ANALYTES FROM AN ALLUVIAL AQUIFER TO STREAMS

V. Rasiah¹
J.D. Armour¹
P.N. Nelson²

¹Department of Environment and Resource Management,
Queensland, Australia

²James Cook University, Queensland, Australia

Reliable information on groundwater (GW) lateral flow characteristics is required for estimation of GW extraction, environmental flow requirements, contaminant loading from GW to surface water bodies (SWB), and aquifer remediation. Lateral flow from a shallow alluvial aquifer was investigated by applying parametric and non-parametric statistics to flux-theory based outputs obtained using time series hydraulic head (HH) and analyte concentration data. The emphasis of the investigation was to identify the major variables that control the export of contaminants from GW to SWB. Point measurements from 4 shallow wells (10-12 m deep) installed along a 1.1 km transect perpendicularly crossing a creek were taken at 7 - 12 day intervals from January through June (wet season) over 3 years in a wet tropical catchment in north-eastern Australia. The HH during two wet seasons at north upslope varied from 4.84 m to 12.37 m with mean, median, and coefficient of variation (CV) of 8.86 m, 8.73 m, and 17% respectively. At the downslope the corresponding values were 3.59-6.21 m, 4.81 m, 4.58 m, and 13%, respectively. Similar temporal trends were observed at the south upslope and downslopes. Nitrate-N concentrations at the north upslope varied from 23 to 1340 $\mu\text{g L}^{-1}$ with mean, median, and CV of 691 $\mu\text{g L}^{-1}$, 609 $\mu\text{g L}^{-1}$ and 23%, respectively. Similar trends were observed at north downslope and at up- and down-slopes of southern transect. The lateral hydraulic gradient (LHG) from north upslope to downslope varied from 4.12×10^{-3} to $9.92 \times 10^{-3} \text{ m m}^{-1}$ and the corresponding flow velocity (V_x) from 3.63×10^{-3} to $3.48 \times 10^{-2} \text{ m d}^{-1}$. Nitrate-N flux from north upslope to downslope varied from 1.0×10^{-4} to $4.4 \times 10^{-3} \text{ g m}^{-2} \text{ d}^{-1}$, similar trends were observed for EC and Cl and also along southern transect. These suggest that analyte fluxes followed the LHG indicating conservative transport of the former from upslopes to downslopes. The conservative transport was reconfirmed by significant associations between HH and analyte fluxes; R^2 18-70% for EC, 24-52% for Cl, and 52-76% for nitrate. Travel time for 650 m, computed using mean V_x varied from 5.8 to 69 yrs and the variations depended on the values of saturated soil hydraulic conductivity (K_s) used. The results indicate contaminant export extrapolations from point measurements to landscape scales depend on our ability to incorporate spatial and temporal variabilities in V_x and analyte fluxes, reliable information in K_s , and macropore bypass flow. We believe this is one of the few studies that have coupled flux-theory and statistics to identify and assess the major variables that control contaminant export from GW to SWB.

INTRODUCTION

Groundwater (GW) flow directions and fluxes are important for GW extraction, estimation of environmental flow requirements, contaminant loading and export, and aquifer remediation purposes. When published information is not available, it is generally assumed that GW discharge and solute transport occur towards streams, which may not always be the case (Praamsma et al. 2009; Larkin and Sharp, 1992). The GW recharge/discharge could vary spatially and temporally with the variations dependent on a number of factors/variables (e.g. vegetation, precipitation, climate, topography, geology and soil types) making it difficult, complex, and uncertain to quantify (O'Driscoll et al. 2010; Gu and Riley 2009; Praamsma et al. 2009; Rein et al. 2009; Rasiah et al. 2007). Despite these difficulties, the delineation of flow directions and rates is important, particularly in regions where recharge and discharge occur simultaneously. The importance is largely linked to contaminant export from GW to surface water bodies (SWB), as even small discharges have been shown to deliver large quantities of environmentally sensitive contaminants to off-site water bodies (LaSage et al. 2008; Kalbus et al. 2007). Further, contaminant export from GW discharge is often disregarded when estimating the total contaminant export load to streams, largely because reliable site-specific GW hydraulic data are often unavailable.

To partially overcome the aforementioned difficulties researchers have resorted to modelling to simulate flow directions and contaminant fluxes (Crosbie et al. 2009; Jolly and Rassam, 2009; Cook and Robinson, 2002). The model output and its reliability depend partially on the quality of the experimental data used for model calibration and validation and the model outputs have rarely been assessed using statistics. Further, model calibration and validation are crucial to improve the understanding of the impact of changes in climate patterns and/or land-use management practices on GW hydrology and the interaction with SWB. For calibration and validation purposes researchers have usually used GW and stream water chemistry data along with limited hydrology information (Crosbie et al. 2009; Jolly and Rassam, 2009; Cook and Robinson, 2002). Thus, in regions where very limited data are available in GW hydrology and its chemistry, the first step towards improving the understanding of lateral-flow transport processes is to undertake GW hydrological measurements before undertaking modelling studies.

In the north-east wet tropics of Australia export of N and P from intensively cultivated agricultural catchments has been partially linked to the health and sustainability of the UN listed Great Barrier Reef, GBR (Baker, 2003; Brodie et al. 2003). The current export load estimates are based primarily on surface runoff. However, studies from this region provide evidence for the presence of large quantities of nitrate in the leachates collected below crop root-zones (Moody et al. 1996) and in GW (Rasiah et al. 2010; 2005; 2003). These results show the nitrate concentrations and the loads in the GW are much higher than those in surface run-off and the authors have suggested the potential of it to SWB via base-flow discharge. The potential export issue was partially addressed by Rasiah et al. (2010) using statistical evidence of 3-way linkage between the nitrate in leachate, GW, and drain-water. However, the linkages were not supported by flux-theory applied to porous media. Furthermore, GW base-flow discharge in the wet tropics occurs throughout the year and it accounts for more than 60% of the total annual stream flow (Cook et al. 2001), highlighting the need to refine the total export estimates. Thus, in this study lateral flow from a shallow alluvial aquifer was investigated applying parametric and non-parametric statistics to flux-theory based outputs obtained using time series hydraulic head (HH) and analyte concentration data. The emphasis of the investigation was to identify the major variables that controlled the export of contaminants from GW to SWB.

MATERIALS AND METHODS

Study catchment

The study was conducted in the Mulgrave River Catchment (MRC) located between 17° 01' and 17° 24' S and between 145° 37' and 145° 58' E, covering an area of 1983 km² in north-east Queensland, Australia (Figure 1a). Approximately 1137 km² of the catchment is in the Wet Tropics World Heritage Area, 346 km² under state forest and timber reserve, 232 km² sugarcane, 55 km² grazing, and 8 km² horticulture.

Geology, soils, aquifer, and rainfall pattern

The aquifer under the present Mulgrave River is the result of thousands of years of sedimentary deposits that have accumulated from river movements and floods across the valley over time (Russell and Isbell, 1986). The alluvium consists of coarser-grained channel deposits within finer grained fan deposits. The aquifer is predominantly recharged by rainfall infiltration. Although some information is available on surface soil hydraulic properties those on sub-surface are generally very scarce (Hair, 1990; Bell et al. 2005; Bonell et al. 1983).

The monthly rainfall distribution shown in Table 1 indicates that it varied within and between years and the variations were high, particularly during the rainy season, January through June. The cumulative percolation during rainy season could be greater than 700 mm yr⁻¹ (Bonnell, 1983) and claimed to be approximately equal to total annual discharge, which accounts for approximately 60% of the total annual flow in streams (Cook et al. 2001). The soils of the catchment area are typically of low fertility (particularly deficient in phosphorus and nitrogen) and exhibit poor soil structure (Russell and Isbell, 1986).

Sediment and total phosphorus export from the catchment are classified as medium risk whilst total nitrogen export as high (http://www.gbrmpa.gov.au/__data/assets/pdf_file/0004/7456/Russell_Mulgrave.pdf). Nutrient and pesticide exports from agricultural catchments, including MRC, are partially linked to the sustainable health of the GBR (Baker 2003; Brodie et al. 2003). The total N and P export in 1996 from MRC were 1440 t yr⁻¹ 153 t yr⁻¹, respectively, compared with 490 t N yr⁻¹ and 24 t P yr⁻¹ estimated for pre-settlement (http://www.gbrmpa.gov.au/__data/assets/pdf_file/0004/7456/Russell_Mulgrave.pdf).

Groundwater monitoring

Point measurements were conducted from wells installed on an undulating landscape along a 1.1 km long transect perpendicularly crossing Behana Creek (Figure 1b). The locations of the wells along the transect and the associated soil profile characteristics are provided in Table 2. Textural description of regolith profiles 1 m depth increments indicate the profiles ranged from clayey to gravelly. In this paper the data from wells 1b, 3a, 4, 6, and 7 are discussed. Boreholes (96 mm diameter), ranging from 10 m to 12 m deep, were drilled using a hydraulic rig. After drilling, PVC pipes (43 mm internal diameter) with tightly sealed bases were inserted into boreholes to serve as monitoring wells. Prior to insertion a segment of each pipe was slotted and wrapped with 250 mm seamless polyester filter socks to facilitate water inflow but prevent coarse sand particles entering the wells. Coarse sand was back-filled around the slotted section and a bentonite collar was placed just above the slotted portion of the pipe to prevent water entry from

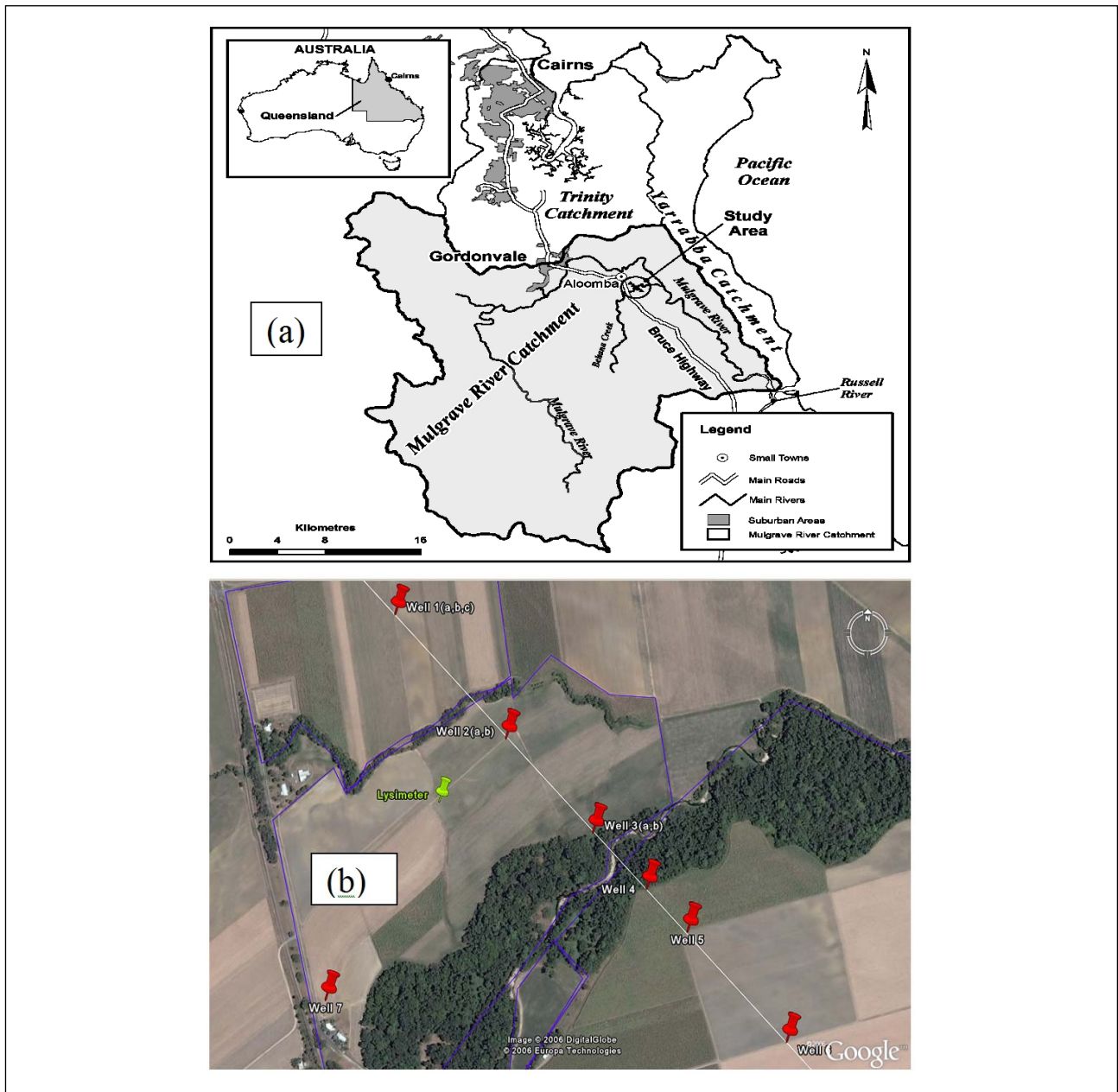


Figure 1. The wet tropical Mulgrave River Catchment in north-east Queensland, Australia, and the experimental site (Figure 1a). Location of the wells along the transect crossing Behana Creek (Figure 1b).

Table 1. Monthly rainfall (mm) distributions in 2007, 2008, and 2009 compared with the long-term average from 1941 to 2009.

Monthly rainfall distribution (mm month ⁻¹) at the experimental site													
	Jan	Feb	Mar	Apr	May	June	July	Aug	Sept	Oct	Nov	Dec	Total
Experimental site													
2007	279	1084	306	180	181	60	47	29	5	83	130	209	2590
2008	486	669	1076	38	74	37	61	7	59	104	88	218	2915
2009	933	883	218	187	107	14	7	14	10	57	450	99	2978
Average	566	879	533	135	121	37	38	17	25	81	223	175	2828
Coefficient of variation	59	24	89	62	45	62	73	67	121	29	89	38	7
January-June													2270
Long-term distribution, average of the data collected at Cairns Airport and Post Office.													
Long term	404	445	428	229	101	62	36	37	42	44	102	182	2117
January-June													1700

Table 2. Textural characterization of soil profiles, well elevation, and the distance between wells.

Well location along the transect and identification numbers					
Profile depth (m)	North upslope well 1b	North downslope well 3a	South down-slope well 4	South upslope well 6	East upslope well 7
0- 1	Clay loam	Loam	Silty clay	Clay loam	Loam
1-2	Clay loam	Silt	Silty clay	Loam+ gravel	Silty clay
2- 3	Silty	Silt	Fine sandy silt	Loam+ gravel	Silt
3-4	Fine sand +clay	Loam+ gravel	Fine sand + loam	Mottled clay	Sandy loam
4-5	Fine sand + clay	Gravel	Clay	Gravel+ mottled clay	Silt+ mottled clay
5-6	Mottled clay	Gravel	Sandy clay	Silt + mottled clay	Gravel + mottled clay
6-7	Mottled clay	Sandy gravel	Clay	Mottled clay	Clay
7-8	Mottled clay	Gravel	Clay	Gravel + mottled clay	Clay
8-9	Mottled clay	Gravel	Clay	Sand + mottled clay	Mottled clay
9-10	Mottled clay	Gravel	Clay	Sand + mottled clay	Mottled clay
10-11	Gravel + clay	-	-	-	Mottled clay
11-12	Silty clay	-	-	-	Mottled clay
Distance between wells and elevation.					
Distance (m)	1b_3a = 648	3a_7 = 713	6_4 = 461	-	1b_7 = 830
Elevation (m)	12.51	6.50	11.63	10.55	13.31
†The elevation is Australian height datum (AHD). 1b_3a refers to the distance from north upslope well 1b to downslope 3a and similarly for the other wells.					

above the collar. Above this collar, the space between the pipe and bore-wall was tightly back-filled with grout and soil material to the soil surface. The water-table levels reported in the text are the depth to groundwater (DGW) from soil surface. Daily rainfall during the investigation period was recorded in an automated weather station located at the study site.

The DGW measurement and water sampling were conducted at 7- 12 day intervals from January through to May (wet season) commencing in 2007 and ending in 2009. Monitoring and sampling were scheduled to occur 12 to 24 hrs after major rain events, and after dry-spells that lasted for at least 2 - 3 days. The former provided information in rising GW and the latter on receding water. The DGW was measured using a special tape and water samples were collected following the procedures described in Alexander (2000) for analyte analysis. The samples were kept at approximately 4°C until arrival in the laboratory, where they were frozen until being analysed for nitrate-N, electrical conductivity (EC) and Cl, using the procedure described by Rayment and Higginson (1992) in a National Association of Testing Authorities (NATA) accredited laboratory.

Cropping and fertilizer history at the site

The experimental site was under native rainforest before being deforested for cultivation in early 1940's and since deforestation it has been under sugarcane crop production until now. Before mid-1980s a trash-burn sugarcane production system was practiced; a green-blanket system was adopted beginning in 1990. The N-fertiliser input at the site during the study period ranged from 100 to 120 kg N ha⁻¹ yr as urea and/or diammonium phosphate. The N-fertiliser was split applied, once at planting in June-August and for ratoon crops and another in December before the onset wet season.

Theory of water and solute flux

The lateral flow gradient (LHG) between 2 wells is defined as:

$$\text{LHG} = (H_1 - H_2) / L \quad (1)$$

where H_1 and H_2 are hydraulic heads in 2 wells, where $H_1 > H_2$, and L is the shortest distance between the wells (Fetter, 1999).

Solute mass-flux (F_x) in one-dimensional flow is defined as:

$$F_x = V_x \zeta C \quad (2)$$

where V_x is average linear velocity, C is concentration of a given soluble analyte in GW, and ζ is effective porosity through which water flows (Fetter, 1999).

$$V_x = (K_s / \zeta) * \text{LHG} \quad (3)$$

where K_s is saturated hydraulic conductivity. The K_s in Equation (3) was taken as 25.37 mm hr⁻¹ (Ninghu et al. 2010) and ζ as 0.15 (Rasiah et al. 2003).

Statistical analysis

The non-parametric statistical parameters median and coefficient of variation (CV) were computed for the time series hydraulic head (HH) and analyte concentrations data in GW. The parametric statistics mean and correlation coefficient were computed for the water and analyte fluxes computed applying flux-theory to the time series data and for the above. The SAS (1991) software package was used for the statistical analysis.

RESULTS AND DISCUSSION

Behaviour of hydraulic heads

During the 2007 wet season the hydraulic head (HH) at the northern upslope varied from 4.84 to 10.75 m compared with 3.59 to 6.17 m at downslope (Figure 2). The HH in 2009 varied from 5.56 to 12.35 m at the upslope and from 4.14 to 6.21 m at downslope. At the southern upslope the HH varied from 3.76 to 10.44 m and from 3.57 to 6.73 m at downslope in 2007. It varied from 4.49 to 10.15 m at the southern upslope and from 3.54 to 6.79 m at downslope during the 2009 season. It is apparent the HH varied within and between wet seasons at a given landscape position. It also varied along landscape positions at a given point in time and between landscapes on either side of the creek. Regardless of all these variations, the HH began to increase early in January with the onset of rain, fluctuated (rose and receded) during mid January-April, and rapidly decreased during April-May to dry-season levels before it began to increase again during the following January. The HHs usually increased after rainfall events and decreased between events.

The dependence of HHs on rainfall (Table 1) was explored by regressing (parametric statistics) HH against the cumulative rainfall received (CRF) between two consecutive monitoring (Table 3). The slopes of the equations indicate significantly different HH responses among the wells to the impact of CRF. The R^2 values suggest the downslope wells, nearer to the creek, were more responsive to CRF than upslope wells, far away from the creek. Higher responses of downslope wells could be attributed to bank storage influence (US Geological Survey Circular 1139). The CVs and median (non-parametric statistics) for the northern upslope wells are higher than downslope well (Table 4). A similar trend was observed for the southern upslope and downslope

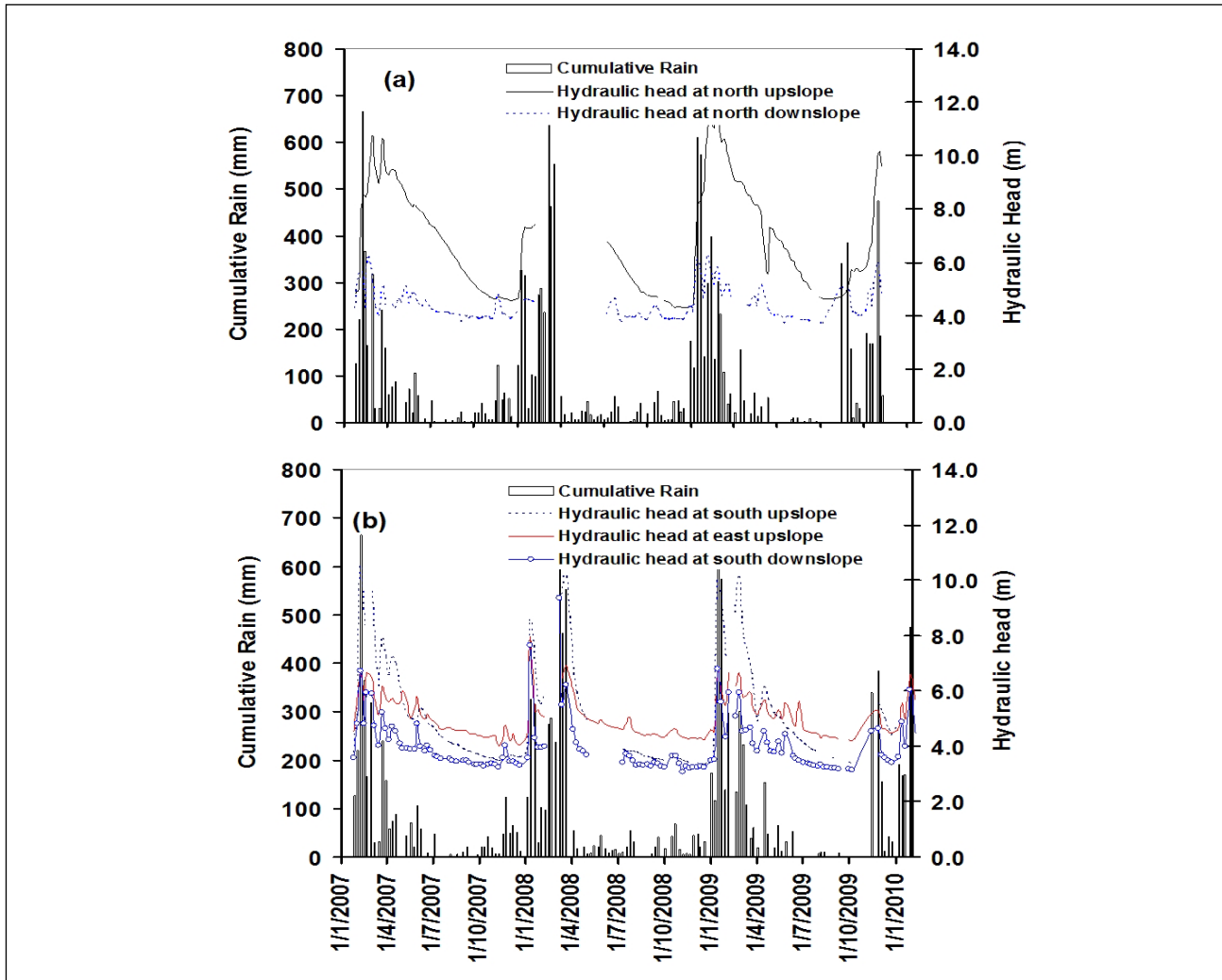


Figure 2. Time series plot showing the influence of cumulative rainfall between two consecutive monitoring on hydraulic head.

Table 3. Simple linear relationship between the hydraulic head and cumulative rainfall (CRF) received between two consecutive water-table measurements

The regression equations for the different wells	R ²
HH 1b = 6.36 (0.20) + 5.56 x 10 ⁻³ (1.29 x 10 ⁻³) CRF	0.37
HH 3a = 4.09(0.04) + 3.27 x 10 ⁻³ (2.61 x 10 ⁻⁴) CRF	0.57
HH 3b = 4.14(0.04) + 3.73 x 10 ⁻³ (2.56 x 10 ⁻⁴) CRF	0.62
HH 6 = 4.21(0.21) + 1.07 x 10 ⁻² (8.00 x 10 ⁻⁴) CRF	0.61
HH 4 = 3.49(0.06) + 5.68 x 10 ⁻³ (3.26 x 10 ⁻⁴) CRF	0.70
HH 7 = 4.70(0.05) + 3.78 x 10 ⁻³ (3.32 x 10 ⁻⁴) CRF	0.48

[†]The equations are significant at P < 0.05. R² = coefficient of determination. HH 1b refers to the hydraulic head in the north upslope well 1b and similarly for the other wells. The numbers within parenthesis are standard errors of estimates.

wells. This suggests the temporal variations at upslopes were more rapid than downslope and this seems to contradict the claim made previously that downslope wells were more responsive to the impact of CRF than upslope. The contradiction is attributed to the differences in impact of individual storm (CRF) on HH vs. seasonal rainfall differences. The HH response to CRF reflects the impact of individual storms whereas CVs characterise the impact of the differences in seasonal rainfall distributions on HH. Higher responses of the downslope wells to CRF could also be attributed to creek water influence after storm events. The mean (parametric statistics) HHs are

Table 4. Selected descriptive statistics for hydraulic head and analytes.

Physico-chemical properties	Mean	Median	Min-Max	CV
North upslope well 1b				
Hydraulic head (m)	8.86(0.22)	8.73	4.84-12.37	17(45)
Electrical conductivity (dS m ⁻¹)	0.081(0.003)	0.075	0.032-0.134	25(45)
Nitrate-N (µg L ⁻¹)	691(96)	609	23-1340	66(23)
Chloride (µg L ⁻¹)	8798(489)	8740	3660-20000	30(31)
North downslope well 3a				
Hydraulic head (m)	4.81(0.09)	4.58	3.59-6.21	13(44)
Electrical conductivity (dS m ⁻¹)	0.062 (0.001)	0.063	0.048-0.090	13(44)
Nitrate-N (µg L ⁻¹)	892(290)	1100	70-1680	73(5)
South downslope well 4				
Hydraulic head (m)	4.62(0.14)	4.30	3.54-9.34	23(53)
Electrical conductivity (dS m ⁻¹)	0.148 (0.005)	0.153	0.056-0.242	27(52)
Nitrate-N (µg L ⁻¹)	96 (33)	80	10-213	78(5)
South upslope well 6				
Hydraulic head (m)	6.57(0.26)	5.99	3.79-10.44	27(51)
Electrical conductivity (dS m ⁻¹)	0.087 (0.003)	0.080	0.043-0.186	31(51)
Nitrate-N (µg L ⁻¹)	640(90)	460	4-1320	78(31)
Chloride (µg L ⁻¹)	8853(473)	10000	4020-11600	26(25)
East upslope well 7				
Hydraulic head (m)	5.62 (0.10)	5.15	4.55-9.52	14(60)
Electrical conductivity (dS m ⁻¹)	0.060 (0.003)	0.058	0.022-0.197	34(59)

of limited use to characterise temporal behaviour, however, the significant differences between wells along the northern or southern transect segment indicate the modifications imposed by spatial differences or systems variables impact on HH along the transect. The mean HH tended to decrease towards the creek, as expected topographically driven HH differences and GW flow.

The rapid fluctuations in HH (water-table) in our study suggest that matrix flow probably didn't play a major role in the recharge/discharge processes. Instead we suggest substantial proportion of the flow was via macropores (bypass flow) and flow through the buried undecomposed cane stools of green trash. The zero-till during ratoon phase (4-5 years) and the green trash might have provided conditions favourable to preserve macropores and/or the stools linked flow, leading to high infiltration and rapid bypass flow. Despite the existence of spatial and temporal variations in HH in this or other wet tropical environments, few workers have attempted to incorporate these behaviours in GW recharge/discharge modelling at catchment scale. The non-inclusion could produce questionable outputs that are often used for regional and local water budget estimations, water resource management, nutrient cycling, and contaminant load estimations and export.

Lateral flow gradients and directions

The LHGs during rainy seasons were almost all positive but there were negative gradients after June, particularly from well 6 to 4 (Figure 3). Because the major emphasis of our work is in base-flow discharge during wet seasons and immediately thereafter, we focus our attention for the period from mid-January to mid-June. The LHG from northern upslope to downslope varied from 4.12×10^{-3} to $9.92 \times 10^{-3} \text{ m m}^{-1}$ and the corresponding variation for the southern upslope to downslope was 4.48×10^{-3} to $8.05 \times 10^{-3} \text{ m m}^{-1}$ (Figure 3a). The LHGs (Table 5 and Figure 3a) indicate that GW from northern upslope flowed laterally towards downslope and east upslope (well 7), from the latter to southern downslope, and from the southern upslope to downslope (Figure 3b). The mean LHGs suggest that more GW flowed from the northern aquifer segment towards the creek than the southern segment. Because more water flowed from the northern segment than

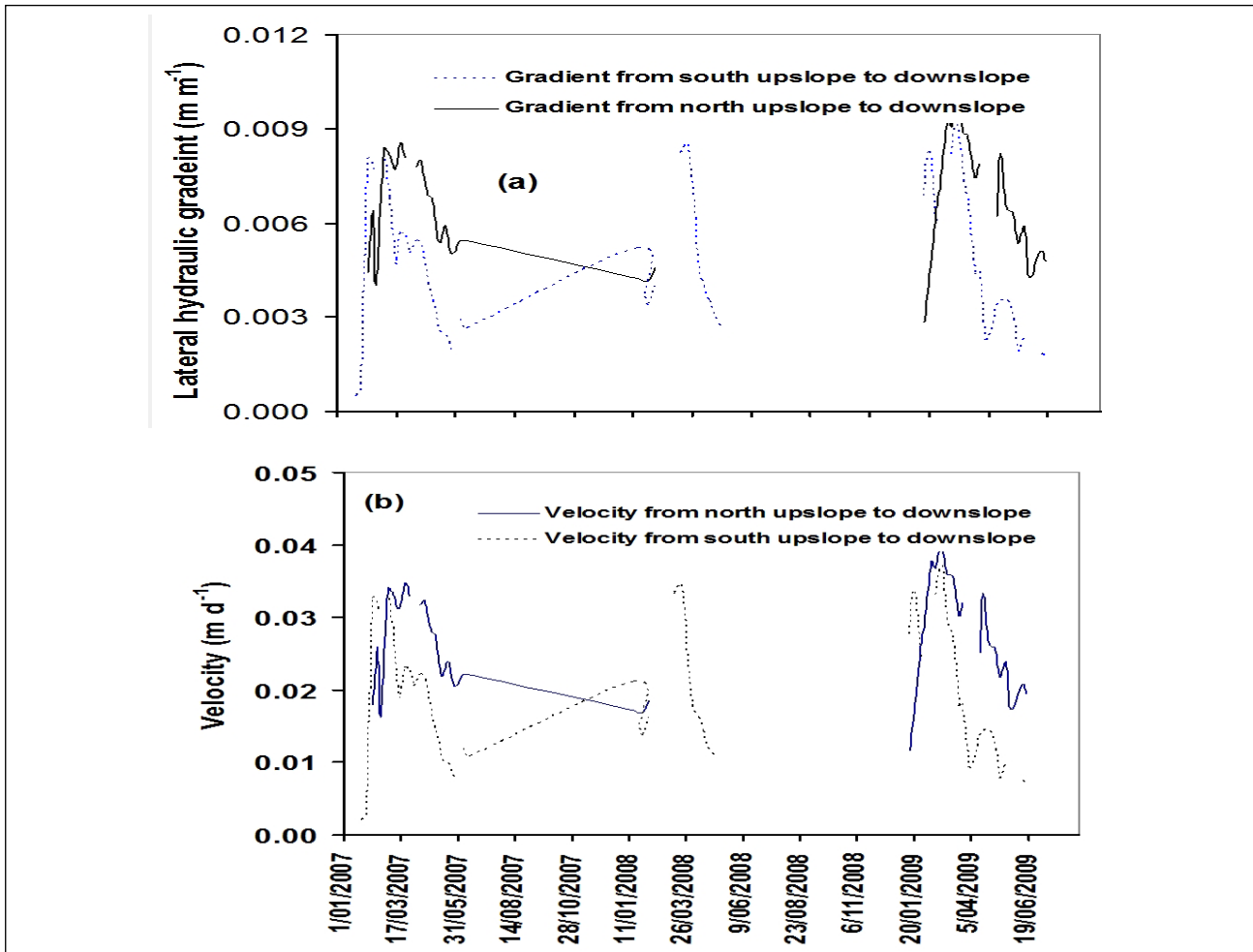


Figure 3. Time series plots showing the influence of topography on either side of the creek on lateral hydraulic gradient and flow velocity. Note the differences in units in Y-axis.

Table 5. A summary for flow gradients, velocities, and analyte fluxes from upslopes to downslopes.

Flux directions				
Statistical parameters	North upslope to downslope	South upslope to downslope	North upslope to east upslope	East upslope to south downslope
Lateral hydraulic gradient ($m\ m^{-1}$)				
Mean	6.29×10^{-3} (2.98×10^{-4})	4.18×10^{-3} (3.29×10^{-4})	3.96×10^{-3} (1.94×10^{-4})	2.57×10^{-3} (3.64×10^{-4})
CV	31	58	33	113
Velocity ($m\ d^{-1}$)				
Mean	2.55×10^{-2} (1.21×10^{-3})	1.70×10^{-2} (1.34×10^{-3})	1.61×10^{-3} (7.88×10^{-4})	1.57×10^{-3} (2.21×10^{-4})
Nitrate flux ($g\ m^{-2}\ d^{-1}$)				
Mean	2.79×10^{-3} (3.05×10^{-4})	1.35×10^{-3} (2.10×10^{-4})	1.50×10^{-3} (1.74×10^{-4})	Not available
CV	62	85	53	
Ion flux ($g\ m^{-2}\ d^{-1}$)				
Mean	1.92×10^{-1} (1.28×10^{-2})	1.49×10^{-1} (1.69×10^{-2})	1.22×10^{-1} (7.41×10^{-3})	2.54×10^{-2} (1.83×10^{-3})
CV	44	82	39	49
Chloride flux ($g\ m^{-2}\ d^{-1}$)				
Mean	3.19×10^{-2} (5.33×10^{-3})	2.47×10^{-2} (2.90×10^{-3})	2.16×10^{-2} (1.81×10^{-3})	Not available
CV	64	60	46	

south, we suggest the former could be a larger source for contaminant export to the creek than the latter, provided the contaminant concentration in the north is equal to or greater than south.

Linear velocity

The temporal and spatial behaviour of flow velocity, V_x , (Figure 3) is similar to LHG as the only variable used in the computation of V_x . The V_x from north upslope to downslope during the wet season varied from 3.63×10^{-3} to $3.48 \times 10^{-2} \text{ m d}^{-1}$ and that from the south upslope to downslope 1.9×10^{-3} to $3.27 \times 10^{-2} \text{ m d}^{-1}$. The mean V_x of the northern landscape was $2.60 \times 10^{-2} \text{ m d}^{-1}$ and that of the south $1.70 \times 10^{-2} \text{ m d}^{-1}$ (Table 5). Extrapolations reveal it could take approximately 69 yr for GW to travel 650 m from north upslope to downslope. This travel time seems to be unrealistically long given regolith characteristics (Table 2), rainfall patterns, and the rapid fluctuations in water-table. According to Equation (3) the static variables that controlled V_x were K_s and effective porosity. We used the site-specific K_s (25 mm hr^{-1}) obtained from slug test (Ninghu Su et al. 2010) for the computation of V_x , however, other reports indicate the K_s for the wet tropics could vary from 30 to 300 mm hr^{-1} (Australian Natural Resource Atlas - www.anra.au/topics/soils/pubs/national/agriculture_asris_she.html). K_s value of 150 mm hr^{-1} reduced the travel time to approximately 11 yrs and 5.8 yr for $K_s = 300 \text{ mm hr}^{-1}$. The order of magnitude difference in V_x estimates show the need for reliable data in K_s and effective porosity before making decisions in travel time, particularly when contaminant export from GW are modelled. Our results indicate that even experimentally determined site-specific K_s values are limited use at this site. The limitation of the experimentally determined site-specific K_s value is attributed to masking effect by bypass flow (see elsewhere also).

De Vries and Simmers (2002) reported that despite numerous studies, experimental determination of recharge/discharge remains an uncertainty and suggested that if bypass flow plays a role in recharge/discharge, then flux-theory based recharge/discharge results are limited use. Scanlon et al. (2002) reported that choosing appropriate methods for recharge/discharge is often difficult because of the complexities involved in incorporating space-time scale variations and bypass flow issues. Our results add another dimension to the aforementioned uncertainties with regard to V_x obtained using experimentally determined K_s and space-time scale variations in V_x .

In order to clarify the issue of appropriate value of K_s for travel time (V_x) estimation we modified and tested the approach proposed by Healy and Cook (2002). These workers showed that changes in GW levels over time and specific yield data can be beneficially utilized for the estimation of recharge/discharge and the advantage of this method is not only its simplicity and accuracy, its insensitivity to the mechanism by which water percolates through soil profile. In our modified approach we used water-table recession data to compute recession rates and used the rates as K_s to compute V_x . For example, over 8 days (11/02/2009 and 18/02/2009), water-receded by approximately 1.0 m and 0.85 m in the southern up- and down-slopes wells and by 0.60 m and 0.50 m at northern up- and down-slopes wells, respectively. These recessions translated to recession rates of 0.07 m d^{-1} to 0.14 m d^{-1} . These values were then used as K_s to compute V_x . The computed V_x are comparable to the V_x obtained using K_s values of 75 mm hr^{-1} and 150 mm hr^{-1} . The travel time estimated using the water-table recession rates as K_s are 22 and 11 yrs, respectively. These travel times seem to be a vast improvement over the 69 yr period that was computed using the site-specific experimentally determined K_s . Still, we believe that even an 11 yr travel time to cover 650 m is too long for this site. Although not shown here, we believe the most appropriate K_s for this site could be obtained by applying inverse minimum residual sum squares

procedure on the time series water-table recession data (Rasiah et al. 1992), and this exercise is beyond the scope of this study (may be in another paper).

In the absence of vertical flow at depths > 12 m (data not shown) and based on the aforementioned water-table recession rates we suggest there was substantial lateral flow discharge between rainfall events. Others (e.g. Cook et al. 2001) have reported the source for more than 60% of the total perennial flow in creeks and rivers in the wet tropics was base-flow discharge. Our LHG data indicates the major proportion of the discharge occurred between rainfall events during wet seasons.

Coupling flux-theory and statistics to analyse the behaviour of analytes

Nitrate-N mass-flux from north upslope to downslope during the 2007 wet season varied from 1.0×10^{-4} to $4.4 \times 10^{-3} \text{ g m}^{-2} \text{ d}^{-1}$, Cl from 1.77×10^{-2} to $1.01 \times 10^{-1} \text{ g m}^{-2} \text{ d}^{-1}$, and total solutes (EC) from 3.0×10^{-2} to $3.8 \times 10^{-1} \text{ g m}^{-2} \text{ d}^{-1}$ (Figure 4). The variations along the southern transect segment were 1.0×10^{-2} to $2.24 \times 10^{-1} \text{ g m}^{-2} \text{ d}^{-1}$ for nitrate, 7.14×10^{-3} to $4.95 \times 10^{-2} \text{ g m}^{-2} \text{ d}^{-1}$ for Cl, and 2.2×10^{-2} to $5.83 \times 10^{-1} \text{ g m}^{-2} \text{ d}^{-1}$ for EC (Figure 5). The CVs indicate analyte concentrations varied during and between wet seasons and in some cases the variations were larger than HH (Table 4). It is apparent from the values of CVs for analyte concentrations the temporal variation was highest for nitrate, followed by Cl and EC. The time-series plots indicate there were close associations between LHG and analyte fluxes (Figures 4 and 5). Because LHG was used in the computation of analyte fluxes we regressed the latter against HH and found that approximately 18 to 70% of ion (EC) fluxes were controlled by HH, compared with 24 to 52% for Cl, and 52 to 76% for nitrate (Table 6). It should be noted the linear fit of nitrate vs. HH was obtained for a zero intercept. There seems to be contradiction between the characterisation of temporal variations by CV and R^2 . We have discussed this aspect previously and suggested the CV's reflect overall seasonal variation and R^2 the influence of individual storm (CRF). Regardless of the differences in responses, the primary driver responsible for the temporal variations in analytes was rainfall induced variations in HH.

Chloride distribution/redistribution in unsaturated soil profiles has extensively been used as signature indicator to delineate subsurface flow pathways (Radford et al. 2009; Rasiah et al. 2005; O'Geen et al. 2002). The relatively high R^2 for EC and Cl suggest that EC along with Cl can be used as signature indicator or conservative tracer to track lateral flow of water and solutes in non-saline saturated regolith in this wet tropical environment. Though the R^2 for zero intercept for nitrate vs. HH linear fit was high there might have been nitrate attenuation by denitrification or dissimilatory reduction to ammonium (Kellogg et al. 2005; Hanson and Hoffman, 1994). This implies that nitrate export from GW to streams may be low, perhaps because the travel time required or the residence time GW is relatively long compared with the rate of nitrate attenuation reductions. The aforementioned long travel time is for export from upslope to the creek, but in reality the export to creek was from downslope. The travel time from downslope to creek is approximately 1 month for $K_s = 300 \text{ mm h}^{-1}$ or approximately 1 yr for $K_s = 25 \text{ mm hr}^{-1}$, implying export risk is significant. The anion adsorption capacity of these alluvial soils is not known to suggest nitrate adsorption in soil matrix played any role in the fate of nitrate in GW. The negative intercepts for EC and Cl suggest potential reverse fluxes of analytes from the creek to GW and negative LHG obtained after mid-June support this claim.

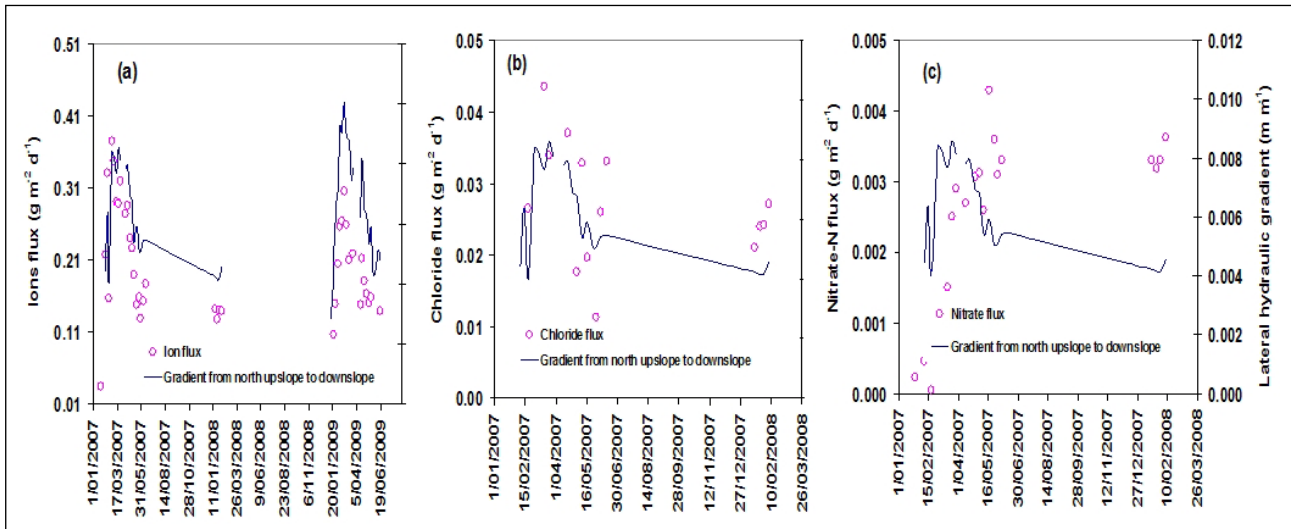


Figure 4. Time series plots showing the influence of lateral hydraulic gradient on conservative transport of ions (Figure 4a), chloride (Figure 4b), and nitrate (Figure 4c) from northern upslope to downslope. Note the differences in units in both X and Y axes.

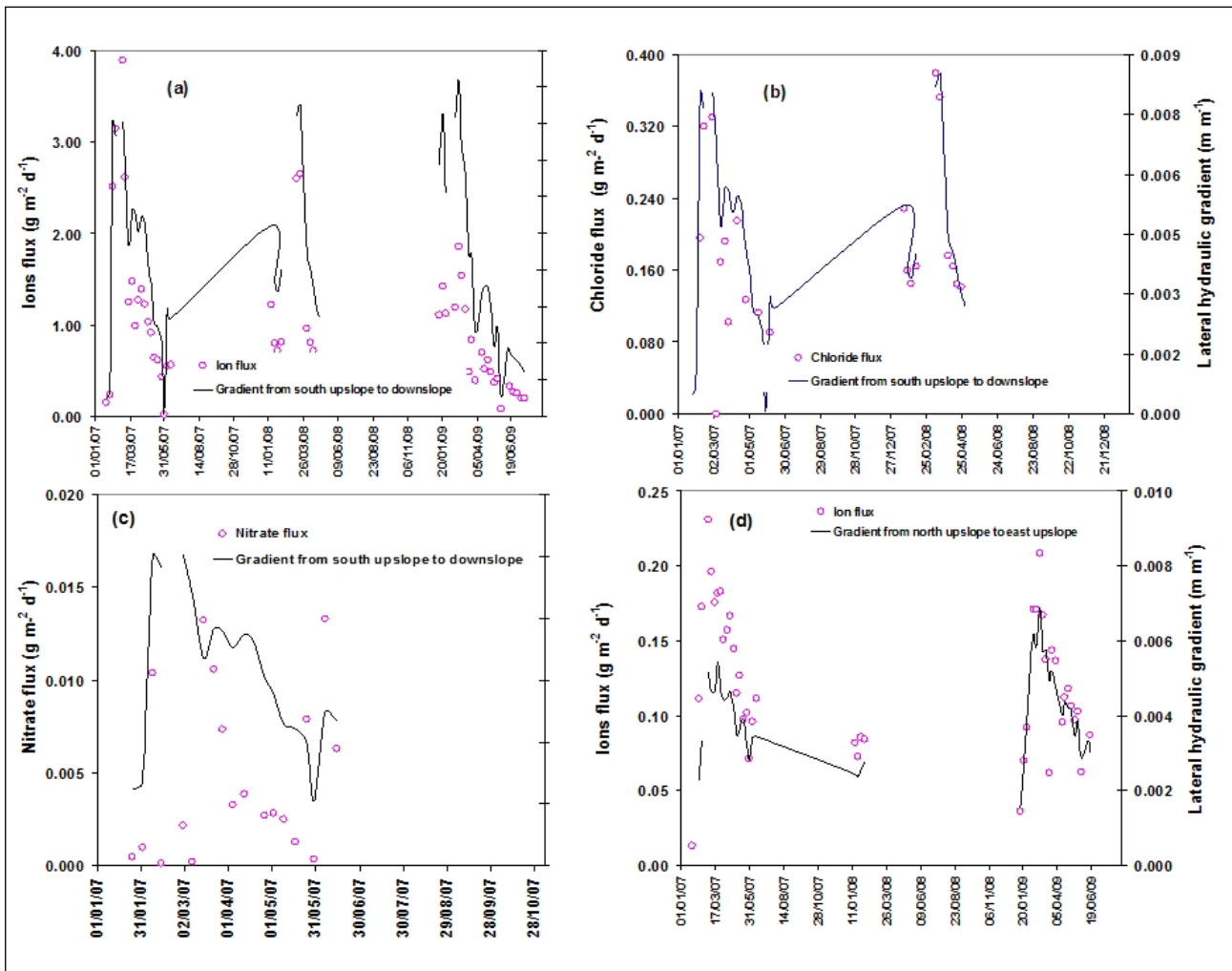


Figure 5. Time series plots showing the influence of lateral hydraulic gradient on conservative transport of ions (Figure 5a), chloride (Figure 5b), and nitrate (Figure 5c) from southern upslope to downslope and ion transport from northern upslope to east upslope (Figure 5d). Note the differences in units in both X and Y axes.

Table 6. Simple linear relationship between mass-fluxes of analytes and hydraulic head (HH).

Flux directions				
Regression parameters	North upslope to downslope	South upslope to downslope	North upslope to east upslope	East upslope to south downslope
Ion flux				
Intercept	-1.71×10^{-1}	-2.11×10^{-1}	-1.23×10^{-1}	-2.21×10^{-2}
Slope	4.12×10^{-2}	5.66×10^{-2}	2.75×10^{-2}	8.64×10^{-3}
R ²	0.53	0.70	0.62	0.18
Chloride flux				
Intercept	-5.30×10^{-2}	-1.67×10^{-2}	-2.48×10^{-2}	Nota available
Slope	1.01×10^{-2}	6.18×10^{-3}	5.14×10^{-3}	
R ²	0.24	0.52	0.43	
Nitrate flux				
Intercept	0	0	0	Nota available
Slope	2.60×10^{-4}	1.92×10^{-4}	1.71×10^{-4}	
R ²	0.69	0.52	0.76	
†The relationship for nitrate flux vs. HH was significant only with zero intercept fit. All the equations are significant at $P < 0.05$ and R ² is coefficient of determination				

O’Driscoll and Dewalle (2010) showed that GW discharge had a strong influence on stream N concentration and that on en-route to streams the N concentration decreased by 30%. Using model results, Rein et al. (2009) reported that contaminant export estimation based on point measurements may be misleading because point measurements may vary spatially and complicated by temporally varying GW flow dynamics. Our results indicate mass-flux of analytes depended largely on the Ks values used, thereby creating an accuracy uncertainty if point measurements are used for extrapolations without reliable data in Ks. In this regard the range in Ks values provided in the Australian Natural Resource website (see elsewhere) seems to be promising and the selection of appropriate site-specific Ks value should be based on site-specific water-table rise/recession data as we did.

CONCLUSION

Point measurements from shallow groundwater (GW) in an alluvial aquifer showed that hydraulic head (HH), lateral hydraulic gradient (LHG), flow velocity (Vx) and travel time (TT) varied temporally and spatially within and between wet seasons. GW flowed laterally towards a creek during wet seasons. The TT computed using mean velocity (Vx) varied from 5.8 to 69 years over a distance of 650 m and this large variation depended primarily on the value of saturated hydraulic conductivity (Ks) values used in the computation of Vx. However, a TT of even 5.8 yrs seems to be too long, given the visual observations in rapid base-flow discharge between rainfall events. This suggests that even the experimentally determined site-specific Ks value was limited use for the estimation of Vx. For such situations, we suggest the use of times series water-table rise/recession rates for the estimation of Ks. The water-table rise/recession method is not laborious and is insensitive to the mechanisms (matrix and/or macropore flow) by which water percolates through soil profiles. The temporal and spatial variations in electrical conductivity (EC), nitrate and chloride (Cl) concentrations and mass-fluxes of these analytes followed LHG, suggesting conservative lateral transport of these towards the creek. We reconfirmed the conservative transport process by regressing analyte mass-fluxes against HH. We conclude that contaminant export extrapolations from point measurements to field scale depend on the ability of end-users to incorporate spatial and temporal variations in flow velocity, gathering reliable information in Ks, and bypass flow characteristics. We believe this is one of the few studies that

have coupled flux-theory and statistics to identify the major variables that control contaminant export from GW to surface-water.

ACKNOWLEDGMENTS

Dr. Chris Carroll (Principal Scientist) and Ms. Glynis Orr (Senior Hydrologist) at the Department of Environment and Resource Management, Queensland, Australia, reviewed this manuscript. We extend our sincere thanks and gratefully acknowledge their contribution via comments, suggestions, and editorial input. The financial support provided by the Australian Research Council, the field and laboratory support provided by Mr. D. H. Heiner, Ms. T. Whiteing, Mr. M. Dwyer, and Ms. D. E. Rowan are gratefully acknowledged.

REFERENCES

- Alexander, D.G. 2000. Hydrographic procedure for water quality sampling. Water Monitoring Group, Water Resource Information & Systems Management. Department of Natural Resources, Brisbane, Australia.
- Baker, J. 2003. A Report on the Study of Land-Sourced Pollutants and their Impacts on Water Quality in and Adjacent to the Great Barrier Reef. Department of Primary Industries, Brisbane. QLD, Australia.
- Bell, M.J., B.J. Bridge, G.R. Harch, and D.N. Orange. 2005. Rapid internal drainage rates in Ferrosols. *Aust. J. Soil Res.*, Vol. 43, pp. 443-455.
- Bonell, M.D., A. Gilmour, and D.S. Cassell. 1983. A preliminary survey of the hydraulic properties of the rainforest soils in the tropical North-East Queensland and their implications for the runoff process. *Catena*, Vol. 4, pp. 57-78.
- Brodie, J., L. McKergow, I.P. Prosser, M. Furnas, A.O. Hughes, and H. Hunter. 2003. Sources of Sediment and Nutrient Exports to the Great Barrier Reef World Heritage Area. ACTFR Report 03/11. Townsville, Australia.
- Cook, P.G., and N.I. Robinson. 2002. Estimating groundwater recharge in fractured rock from environmental ^3H and ^{36}Cl , Clare Valley, South Australia. *Water Res. Res.*, Vol. 35(2), pp.1136-1142.
- Cook, P.G., A.L. Herczeg, and K.L. McEwan. 2001. Groundwater recharge and stream baseflow: Atherton Tablelands, Queensland. CSIRO Land and Water. Technical Report 08/01, April 2001. Canberra, Australia.
- Crosbie, R.S., J.L. McCallum, and G.A. Harrington. 2009. Estimation of groundwater recharge and discharge across northern Australia. 18th World IMACS/MODSIM Congress, Cairns, Australia 13-17th July 2009. (<http://mssanz.org.au/modsim09>)
- De Vries, J.J., and I. Simmers. 2002 Groundwater recharge: an overview of processes and challenges. *Hydrogeology Journal*, Vol. 10(1), pp.5-17
- Fetter, C.W. 1999. *Contaminant Hydrogeology*, 2nd ed., Prentice Hall, ISBN 0137512155.
- Gu, C., and R.J. Riley. 2009. Combined effects of short term rainfall patterns and soil texture on soil nitrogen cycling – A modelling analysis. *J. Contaminant Hydrology*, Vol. 112, pp. 141-154.
- Hair, I.D. 1990. *Hydrogeology of the Russell and Johnstone Rivers Alluvial valleys, North Queensland*. ISBN 1034 7399, Department of Resource Industries, Brisbane, Australia.
- Hanson, G.C., and P.M. Hoffman. 1994. Denitrification rates in relation to groundwater level in a peat soil under grassland. *J. Environmental Quality*, Vol. 23, pp. 917-922.
- Healy, R. W., and P.G. Cook. 2002. Using groundwater levels to estimate recharge. *Hydrology Journal*, Vol.10, pp. 99-109
- Jolly, I.D. and D.W. Rassam. 2009. A review of modelling of groundwater-surface water interactions in arid/semi-arid floodplains. 18th World IMACS / MODSIM Congress, Cairns, Australia 13-17 July 2009. <http://mssanz.org.au/modsim09>.
- Kalbus, E., C. Schmidt, M. Bayer-Raich, S. Leschik, F. Reinstorf, G.U. Blacke, and M. Schirmer. 2007. New methodology to investigate potential contaminant mass fluxes at the stream-aquifer interface by combining integral pumping tests and streambed temperatures. *Environmental Pollution*, Vol. 148, pp. 808-816.
- Kellogg, D.Q., A.J. Gold, P.M. Groffman, K. Addy, M.H. Stolt, and G. Blazewski. 2005. In-situ ground water denitrification in stratified, permeable soils underlying riparian wetlands. *J. Environmental Quality*, Vol. 34,

pp. 524-533.

- Larkin, R.G., and J.M. Sharp Jr. 1992. On the relationship between river-basin geomorphology, aquifer hydraulics, and groundwater flow direction in alluvial aquifers. *GSA Bulletin*, Vol. 104, pp. 1608-1620.
- LaSage, D.M., A.E. Fryar, M.A. Mukherjee, N.C. Sturchio, and L.J. Heraty. 2008. Groundwater-derived contaminant fluxes along a channelized Coastal Plain stream. *J. Hydrology*, Vol. 360, pp. 265-280.
- Moody, P.W., J.R. Reghenzani, J.D. Armour, B.G. Prove, and T.J. McShane. 1996. Nutrient balances and transport at farm scale- Johnstone River Catchment. In: Hunter HM, Eyles GE, Rayment G. (Eds.), *Downstream Effects of Land Use*. Department of Natural Resources, Brisbane, Australia.
- Ninghu Su., P.N. Nelson, and S. Connor. 2010. Determining aquifer hydraulic parameters using slug tests and the solution of a fractional diffusion-wave equation. *J. Hydrology* (in review).
- O'Geen, A.T., P.A. McDaniel, and J. Boll. 2002. Chloride distributions as indicators of vadose zone stratigraphy in Palouse loess deposits. *Vadose Zone Journal*, Vol. 1, pp. 150-157.
- O'Driscoll, M.A., and D.R. Dewalle. 2010. Seep regulates stream nitrate concentration in a forested Appalachian catchment. *J. Environmental Quality*, Vol. 39, pp. 420-431.
- Praamsma, T., K. Novakowski, K. Kyser, and K. Hall. 2009. Using stable isotopes and hydraulic head data to investigate groundwater recharge and discharge in a fractured rock aquifer. *J. Hydrology*, Vol. 366, pp. 35-45.
- Radford, B.J., D.M. Silburn, and M. Forster. 2009. Soil chloride and deep drainage responses to land clearing for cropping at seven sites in central Queensland, northern Australia. *J. Hydrology*, Vol. 379, pp. 20-29.
- Rasiah, V., J.D. Armour, A.L. Cogle, and S.K. Florentine. 2010. Nitrate import-export dynamics in groundwater interacting with surface-water in a wet-tropical environment. *Aust. J. Soil Res.*, Vol. 48, pp. 361-370.
- Rasiah, V., J.D. Armour, and A.L. Cogle. 2007. Statistical characterisation of impact of system variables on temporal dynamics of groundwater in highly weathered regoliths. *Hydrological Processes*, Vol. 21, pp. 2435-2446.
- Rasiah, V., J.D. Armour, and A.L. Cogle. 2005. Assessment of variables controlling nitrate dynamics in groundwater: Is it a threat to surface aquatic ecosystems? *Marine Pollution Bulletin*, Vol. 51, pp. 60-69.
- Rasiah, V., J.D. Armour, T. Yamamoto, S. Mahendrarajah, and D.H. Heiner. 2003. Nitrate dynamics in shallow groundwater and the potential for transport to off-site water bodies. *Water Air and Soil Pollution*, Vol. 147, pp. 183-202.
- Rasiah, V., G.C. Carlson, and R. A. Kohl. 1992. Assessment of functions and parameter estimation methods in root water uptake simulation. *Soil Sci. Soc. Am. J.*, Vol. 56, pp. 1267-1272.
- Rayment, G.R., and F.R. Higginson. 1992. *Australian Laboratory Handbook of Soil and Water Chemical Methods*, Inkata Press, Sydney, Australia.
- Rein, A., S. Bauer, P. Dietrich, and C. Beyer. 2009. Influence of temporally variable groundwater flow conditions on point measurements and contaminant mass flux estimations. *J. Contaminant Hydrology*, Vol. 108, pp. 118-133.
- Russell, J.S., and R.F. Isbell. 1986. *The Australian Soils: the Human Impact*. St. Lucia, University of Queensland Press. ISBN 0702219681.
- Scanlon, B.R., R.P. Langford, and R.S. Goldsmith. 1999. Relationship between geomorphic settings and unsaturated flow in an arid setting. *Water Res. Res.*, Vol. 35(4), pp. 983-999.
- Statistical Analysis Systems. 1991. *SAS/STAT Procedure Guide for Personal Computers, Version 5*. Statistical Analysis Systems Institute Inc. Cary, NC.
- von Asmuth, J.R., and M. Knotters. 2004. Characterising groundwater dynamics based on a system identification approach. *J. Hydrology*, Vol. 296, pp. 118-134.

ADDRESS FOR CORRESPONDENCE

V. Rasiah
Department of Environment and Resource Management
5B Sheridan Street
PO Box 937
Cairns QLD 4870
Australia

rasiah_v@derm.qld.gov.au
