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A hydrologic tracer study in a small, natural wetland in the humid tropics of Costa Rica

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Abstract Growing populations and food demand in the tropics are leading to increased environmental pressures on wetland ecosystems, including a greater reliance on natural wetlands for water quality improvement. Effective assessment of wetland treatment potential requires an improved understanding of the hydraulic and biogeochemical factors that govern contaminant behavior, however detailed studies of flow through natural, tropical wetlands are scarce. We performed a tracer study using a conservative salt (potassium bromide) to examine the hydraulic behavior of a small, natural wetland in the Costa Rican humid tropics and modeled observed breakthrough curves using the 1-D advection–dispersion equation. Velocities in the wetland were extremely slow, from less than 4 m day^{-1} to a maximum of $\sim$30 m day^{-1}, and were distributed across several flowpaths, illustrating a spatial heterogeneity of flow and velocities. Modeled dispersion coefficients were also low (33 $\pm$ 33 m^2 day^{-1}). Estimated residence times suggested high potential pollutant removal capacity over a range of influent concentrations, reinforcing the environmental services provided by this and other small tropical wetlands. The study also highlighted how small variations in wetland topography and vegetation yield strong differences in transport patterns that affect transport and mixing in densely vegetated, heterogeneous wetland systems. Empirical data on the hydraulics, and resulting ecosystem functions, of small, distributed wetlands may provide support for improved conservation and management of these important ecosystems.

Keywords Tracer study · Wetland hydrology · Bromide · Residence time · Water quality · Humid tropics · Ecosystem services

Abbreviations

Br^-	Bromide
BTC	Breakthrough curve
τ	Residence time

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Introduction

Hydraulic characterization of wetlands is fundamental to quantifying the diverse set of services provided by these complex ecosystems. Hydrologic functions of wetlands include flood control and flow regulation (Brouwer et al. 1999), erosion control and sediment retention (Hey and Philippi 2006), aquifer discharge and recharge (Ewel 1990; Choi and Harvey 2000; McLaughlin et al. 2014), and maintenance of water levels and flows in receiving waterbodies during dry seasons (Keddy 2010). Ecological functions are inextricably tied to these hydrologic processes and include habitat and biodiversity conservation (Gibbs 2001), biomass and nutrient transformation (e.g., Junk et al. 1989), and water quality improvement (Kadlec and Wallace 2008). Both natural and constructed wetlands are increasingly being used to remove nutrients (Bachand and Horne 2000; Reilly et al. 1999), metals (Debusk et al. 1996; Kadlec and Wallace 2008), pesticides (Schulz and Peall 2001), and industrial solvents (Doble and Kumar 2005) from municipal, agricultural, and stormwater runoff. The focus of this study is the potential for small natural wetlands to provide a variety of ecosystem services within developing agricultural watersheds in the humid tropics.

A greater reliance on natural wetlands for water quality improvement in developing watersheds requires a better understanding of the hydraulic and biogeochemical factors that govern contaminant behavior. Field tracer studies are an effective way to study wetland hydraulics, velocities, and pathways through hydrological systems (Harden et al. 2003) and to assess residence time distribution and vertical and horizontal water mixing conditions (Martinez and Wise 2003). A precondition for studying wetland hydraulics is the availability of robust tracer methods adapted to the conditions of the study area. A tracer is a non-reactive, non-sorbing solute released into a water system to determine its hydraulic characteristics. Tracer release is generally made at the wetland inlet or an upstream point to study the resulting time of arrival, concentration, and dispersion of the tracer through the system. The most popular surface and groundwater tracers are chemical salts containing chloride, lithium, or bromide (Br^-). Bromide is the most widely used tracer in natural wetland systems (Martinez 2001) since it is found at very low background concentrations in the environment compared to chloride. Bromide is also known to be conservative compared with lithium, which can adsorb by ion exchange to sediments. However, in wetland systems with tranquil water flow it is often difficult to track Br^- because of high dilution rates.

While several authors have explored the hydraulic function of constructed wetlands using chemical tracers (e.g., Grismer et al. 2001; Kadlec 1994; King et al. 1997; Martinez and Wise 2003), hydraulic studies of natural (i.e., not constructed) wetlands are exceedingly scarce (Stern et al. 2001). With notable exceptions in some large, charismatic systems such as the Everglades in south Florida (USA) (Harvey et al. 2005; Ho et al. 2009; Variano et al. 2009), description of flow through natural wetlands is generally qualitative (Stern et al. 2001). Furthermore, while contaminant reduction in wetlands has been relatively well studied in temperate climates (Kadlec and Wallace 2008), less is known about the treatment capabilities of tropical wetlands, which have received less attention from the scientific and management communities (Bullock 1993; Ellison 2004; Junk 2002; Nahlik and Mitsch 2006; Roggeri 1995). We could find no relevant hydraulic studies in natural, tropical wetlands. An increased focus on the role and function of these imperiled ecosystems is required, particularly as they face widespread degradation due to the increasing appropriation of land and water associated with rapidly increasing populations and food demand in the tropics (Daniels and Cumming 2008; Junk 2002).

Due to their abundance and ubiquitous distribution throughout the landscape, small wetlands in the tropical landscape of Central America likely play a critical and multifaceted role in the environmental quality of the area (water storage, flood control and water quality improvement). The case study of a small Central American wetland presented here and in a related investigation (Kaplan et al. 2011) aims to quantify the function of these natural wetland systems and generate hydrological information in support of public decision-making regarding wetland use, conservation, and preservation. The specific objective of this study was to conduct a field tracer study to explore the hydraulic characteristics and resulting ecosystem services of a small natural wetland in the humid tropics of Costa Rica.

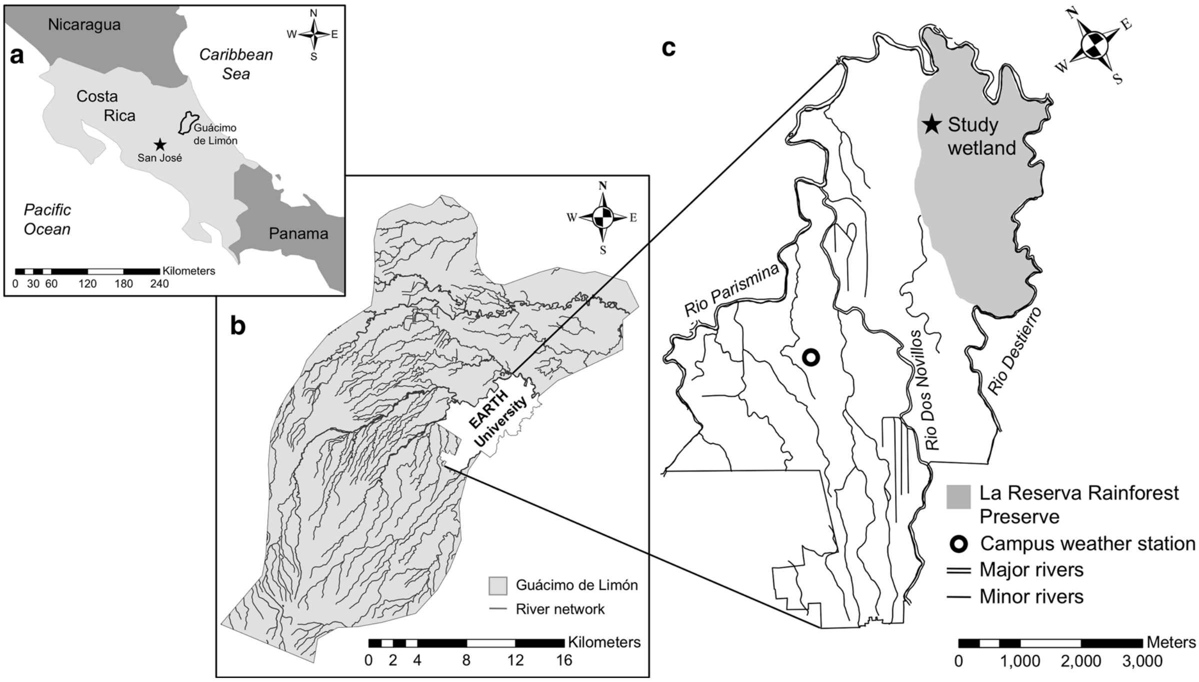

Fig. 1 Location of the study wetland within "La Reserva" rainforest preserve on the campus of EARTH University in Guácimo de Limón, Costa Rica (*source* Kaplan et al. 2011)

Materials and methods

Geographic setting

The study was carried out in the humid tropics of Costa Rica on the campus of EARTH University (Escuela de Agricultura de la Región Tropical Húmeda). The campus is located 60 km from the Caribbean coast in the canton (i.e., county) of Guácimo de Limón (Fig. 1a, b) and has elevations ranging between 20 and 30 m above sea level. Average annual rainfall and temperature recorded at EARTH between 1996 and 2008 were 3 227 mm and 24.5 °C, respectively (W. Rodríguez Chacón, unpublished data). The watershed has undergone little urban development but is home to intensive agricultural activities, dominated by banana production.

EARTH University contains several small, natural wetlands on clayey, hydromorphic soils (Aquepts) (Mitsch et al. 2008). The ~1.5-ha wetland investigated in this study was located in a ~10-ha subwatershed of the ~400-ha "La Reserva" rainforest preserve (Fig. 1c). Elevations in the subwatershed ranged from 20 to 30 meters above sea level, with topographic variation on the order of 1–2 m within the wetland itself. Irregular topography forms several waterlogged basins with three main forested branches (dominated by swamp palm [Raphia taedigera Mart.]) that join together in a central herbaceous marsh dominated by a variety of graminoids and forbs (Fig. 2). A small fourth branch joins the herbaceous marsh from the west. Organic wetland soils are composed primarily of poorly decomposed plant material, while upland soils are primarily oxisols with low organic content. The wetland has a single outlet downstream (Q in Fig. 2) with variable, but apparently perennial, outflow and no specific surface water inlet (Kaplan et al. 2011). Direct precipitation is the main source of water to the wetland, with some contribution from runoff, particularly during the largest storms (Kaplan et al. 2011), and little groundwater flow due to clayey alluvial soils underlying the basin (Nahlik and Mitsch 2006), although groundwater flows have not been directly measured. Additional information on wetland topography, soils, hydrology, and vegetation can be found in Gallardo and César (2006), Nahlik and Mitsch (2006), Kolln (2008), Mitsch et al. (2008), and Kaplan et al. (2011).

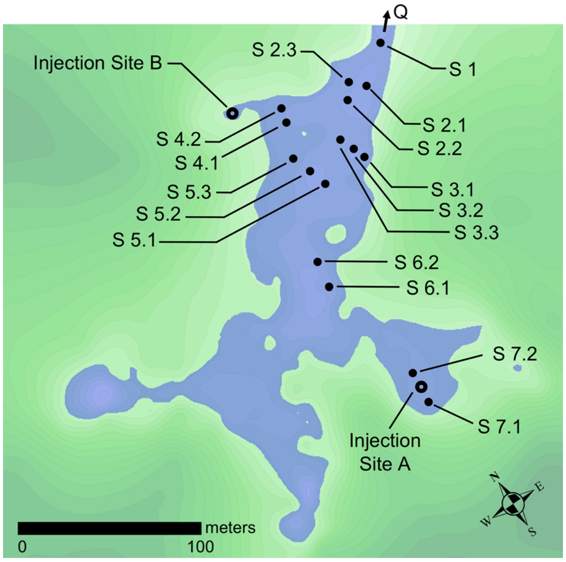

Fig. 2 Map of the study wetland showing tracer injection and monitoring sites and wetland outflow (*Q*)

Tracer preparation, injection, and sampling

The Br^- tracer was injected into the wetland on May 13th, 2008. The tracer was applied to the wetland at two different sites as point sources, each as a single injection. Injection points were identified in initial field reconnaissance, and were located at two accessible, upstream points located in opposite branches (Injection Sites A and B in Fig. 2). Site A was located in a forested branch with sections of shallow channelized flow, and Site B was located in a primarily herbaceous branch with slower, shallow sheet flow. Sampling points were distributed downgradient from these points and across assumed flow paths (Fig. 2). Approximately 1 h prior to injection, two ~100 L plastic barrels were filled with water from the wetland at each injection site. Next, 13.6 kg (nominally 30 lb) of photograde potassium bromide (KBr; 99.92 % purity; DigitalTruth Photo, Houston, TX, USA) was dissolved into each injection barrel by carefully pouring and stirring until completely dissolved (~10 min). Given a KBr solubility of 535 g L^{-1} at 25 °C, the injection solution was at ~25 % KBr saturation. The initial bromide concentrations [Br^-] measured in the injection barrels were 89.8 g L^{-1} at injection site A and 67.4 g L^{-1} at injection site B (concentration differences are due to the difference in volumes of the injection barrels). Before tracer injection, samples were secured from each injection barrel. Finally, the entire contents of the two injection barrels were poured into the wetland at each injection site over 15–30 s. The two injections occurred within 9 min of each other.

Samples were collected from the eighteen injection and monitoring sites for 3 weeks after injection. Samples were collected twice daily early in the tracer test to better capture potential rapid peaks of tracer passing through the sampling site, after which samples were collected once daily to capture breakthrough curve tails. Samples were collected manually in 40 mL septum vials (ThermoFisher, Suwanee, GA, USA) that were completely filled and capped under water. A total of 424 samples (24 per site) were collected. Samples were stored at 4 °C after collection and kept closed at all times until chemical analysis. For transport from Costa Rica to Gainesville, FL, samples were packed cold in styrofoam containers, transported by plane, and replaced in cold storage at 4 °C upon their arrival (total travel time ~16 h). Associated hydrologic data were collected in a companion study (Kaplan et al. 2011); rainfall, potential evapotranspiration (PET), and wetland outflow (Q) measured during the period of the tracer study are summarized in Fig. 3.

Sample analysis and quality control

Samples were analyzed for Br^- using high-pressure liquid chromatography (HPLC) with electrochemical detection (DIONEX ICS-90 ion chromatography equipped with an AS40 autosampler; Thermo Fisher Scientific, Sunnyvale, CA, USA) within 3 months of sample collection. Most samples had a small amount of fine sediment and were filtered (Nylon Pores Size 0.2 µm, Fisher Scientific, Pittsburgh, PA, USA) prior to HPLC injection. Blank samples of distilled–deionized water were run between each sample set to purge the HPLC, followed by a Br^- dilution standard to compute the calibration curve. The calibration curve was a linear relation between the signals detected at the retention time for Br^- (13.5 min). The software Peaknet 6 (Thermo Fisher Scientific, Sunnyvale, CA, USA) allowed a direct calculation of [Br^-] for each sample. The detection limit of Br^- was 0.01 mg L^{-1}. Samples with expected high concentration (e.g., from injection barrels and initial samples close to the injection location) were diluted to keep the data in the

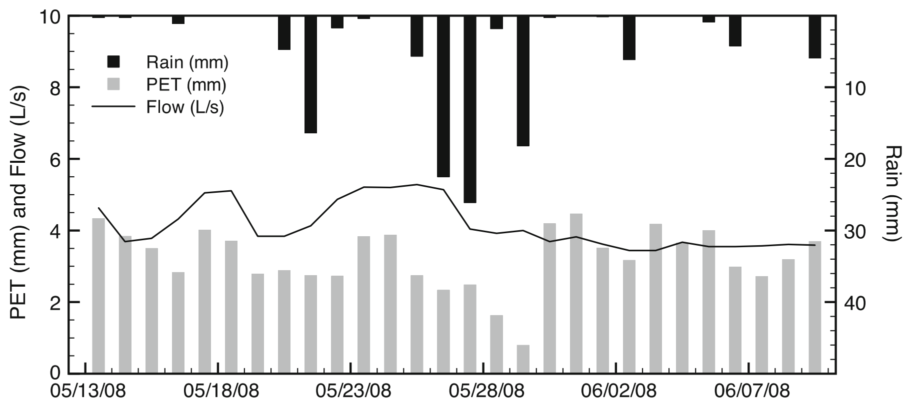

Fig. 3 Rain, PET (potential evapotranspiration), and wetland outflow measured during the study period

linear scale of the calibration curve and analyzed at the end to avoid biasing peak detection of samples with lower concentration.

Estimating transport parameters

Observed Br^- concentrations were modeled using the CXTFIT code (Toride et al. 1995) implemented in STANMOD (Studio of Analytical Models; Šimůnek et al. 1999) (available at www.ars.usda.gov/Services/docs.htm?docid=8960), which applies the 1-dimensional advection–dispersion equation (ADE). CXFIT uses a nonlinear least-squares method to optimize parameter estimation based on observed data by minimizing the sum of squared errors (Šimunek et al. 1999). Tracer injection was modeled as a pulse input, and initial $[Br^-]$ was set to zero. Moment analysis was applied to all breakthrough curves (BTCs) to estimate initial values for velocities and dispersion coefficients following Pang et al. (2003). While we knew the injection concentration and total tracer mass injected, the tracer concentration observed at each measurement location was a function of spatially heterogeneous local flows, which included not only flows emanating from the tracer injection sites, but also flow from other wetland areas which had $[Br^-] = 0$. To account for the diluting effect of these additional flows on the observed BTCs, we allowed the CXFIT code to optimize model performance by treating pulse duration as an additional fitting parameter.

The model was used to fit observed data along different flow paths through the wetland from injection to measurement sites. At locations where tracer peaks from both injection sites were observed, BTCs were first separated by assuming a rapid exponential decay after passage of the first tracer peak (based on observed behavior of the first BTC peaks). Separated peaks were then fit individually to estimate transport parameters for each pathway and the two models were combined (i.e., added) for comparison with observed BTCs. For all locations the upstream boundary was assumed to be one of the injection sites, so the fitted transport parameters (v and D) represent average transport conditions along the entire flowpath being modeled (i.e., from the injection site to the measurement location), not between specific measurement locations.

Results

Tracer breakthrough curves

At both injection sites, $[Br^-]$ decreased rapidly to less than 1 % of the injected concentration as tracer flowed downgradient (Fig. 4a). Initial dilution was $\sim 10\times$ greater at Site A, indicating higher flow in the upstream portion of the eastern branch compared to the northwestern branch. The tracer injected at Site A showed minor evidence of upgradient flow (to Site 7.1) concurrent with a large rainfall event on May 21 (Fig. 3), however the concentration of this "peak" (1.25 mg L^{-1}) was small compared to the initial $[Br^-]$ of 286 mg L^{-1} at the injection site immediately following tracer release (Table 1). Injection Site B was at the wetland-upland boundary, and it was assumed that no upgradient transport occurred at this site.

Breakthrough curves observed at all other sites showed relatively low $[Br^-]$ due to dilution, with

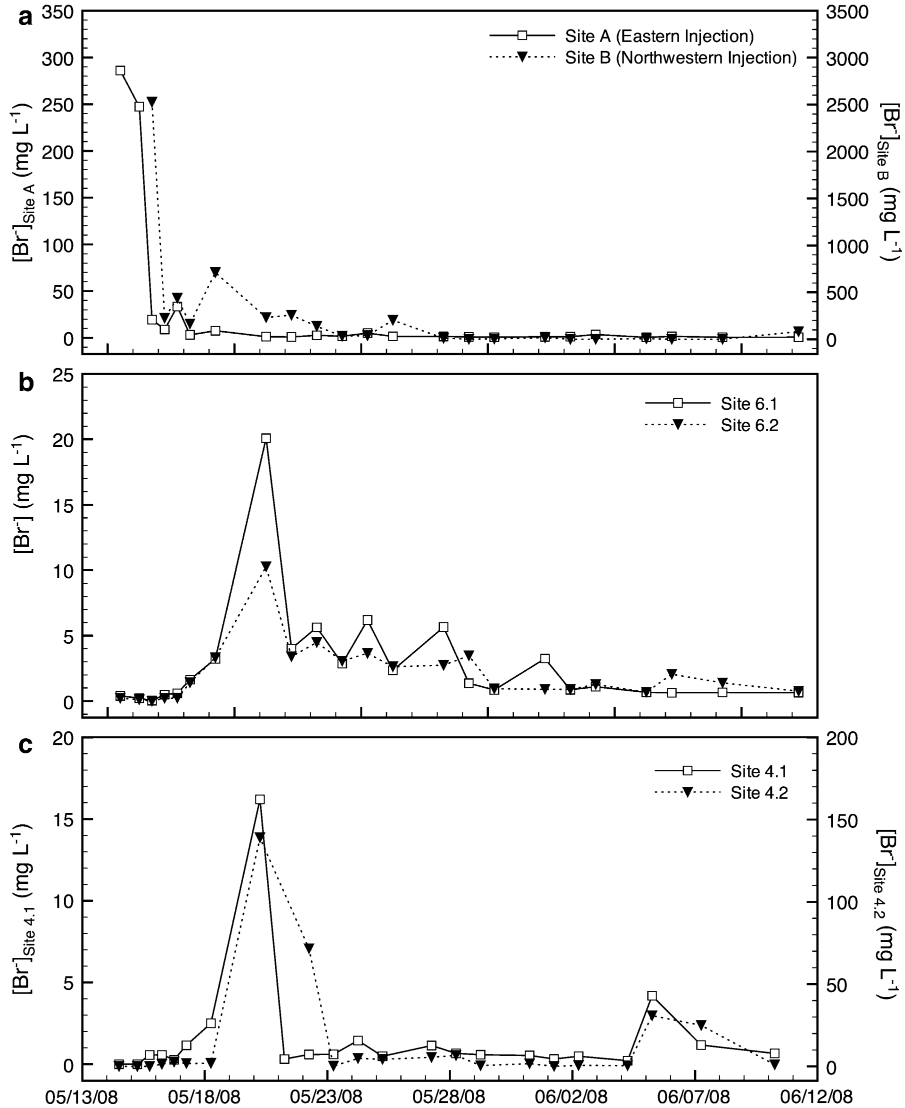

Fig. 4 **a** Bromide concentration [Br⁻] at Injection Sites A and B show rapid downstream transport away from injection sites (note different scales for Sites A and B). **b**, **c** [Br⁻] measured at Sites 4 and 6 showed one distinct peak (note different scales for Sites 4.1 and 4.2 in **c**)

concentrations decreasing with distance from injection sites. Consistent, single [Br⁻] peaks were observed at Sites 6.1 and 6.2 (Fig. 4b), which were located downstream of Injection Site A. While the shape of the two curves is similar, the peak at Site 6.2 is smaller, implying greater dilution at this location due to flow emanating from the western portion of the wetland, which did not have tracer injection (Fig. 2). Single primary peaks were also observed at Sites 4.1 and 4.2, (Fig. 4c) downstream of Injection Site B, with smaller secondary pulses apparent in early June, concurrent with a series of rainy days (Fig. 3) that raised the water level several cm in the northwestern branch. Br⁻ concentrations measured at Site 4.2 were ~10× than those at Site 4.1, again suggesting large differences in mixing and dilution over short spatial scales; lower concentrations at Site 4.1 were likely due to dilution of flow coming from the main body of the wetland that did not reach Site 4.2 due to variations in microtopography or vegetative resistance. Single BTC peaks at both Sites 4 and 6 indicate that only the tracer plume from the injection point upstream of these stations was captured at these locations. Although the tracer peaks arrived at Sites 4 and 6 on the same day, these results indicate faster flow in the eastern branch (from Injection Site A to Site 6) than in the northwestern branch (from Injection Site B to Site 4), given the greater distance between stations in the

Table 1 Breakthrough curve characteristics and at each sampling site and initial velocity estimates calculated using time between tracer peaks and distance from injection

Location	Time to peak 1 (days)	$[Br^-]$ peak 1 (mg L^{-1})	Time to peak 2 (days)	$[Br^-]$ peak 2 (mg L^{-1})	Distance from Injection Site A (m)	Distance from Injection Site B (m)	Eastern velocity (m day^{-1})	Western velocity (m day^{-1})
Eastern								
Site 7.1	0	1.25	–	–	~0	–	–	–
Site 7.2	0	285.95	–	–	~0	–	–	–
Site 6.1	5	20.08	–	–	77.9	–	15.6	–
Site 6.2	5	10.27	–	–	89.2	–	17.8	–
Site 3.1	5	0.87	–	–	141.7	–	28.3	–
Site 3.2	5	1.27	–	–	143.6	–	28.7	–
Site 3.3	9	0.91	–	–	151.0	–	16.8	–
Western								
Site 4.1	5	138.96	–	–	–	41.6	–	8.3
Site 4.2	5	16.20	–	–	–	36.3	–	7.3
Confluence								
Site 5.1	5	2.30	14	1.39	127.4	64.7	25.5	4.6
Site 5.2	5	1.92	13	1.06	130.3	59.7	26.1	4.6
Site 5.3	5	1.26	14	1.12	135.1	51.1	27.0	3.7
Site 2.1	5	1.07	13	1.40	164.4	52.4	32.9	4.0
Site 2.2	5	0.22	9	0.80	174.0	69.2	34.8	7.7
Site 2.3	5	0.42	9	0.73	176.4	58.5	35.3	6.5
Site 1	9	3.22	13	3.55	207.0	93.0	23.0	7.2
					Average velocity		26.0	6.0

eastern branch. This general inference was used to interpret double tracer peaks observed at other monitoring locations.

Two primary tracer peaks were observed at Sites 1, 2, and 5 (Fig. 5a–c), suggesting that these sites were located downgradient of the confluence of flow paths from the two injection sites. Double peaks at Sites 2 and 5 suggest the passage of water from two distinct flowpaths at these sites: an early sharp pulse from the tracer injected at Site A and a broader secondary pulse emanating from injection Site B towards the central herbaceous marsh and downstream towards the wetland outlet. Two primary peaks were also apparent at Site 1, however they were less clearly separated perhaps due to greater mixing near the wetland outlet as a result of backwater conditions behind the wetland outflow culvert. Interpretation of the BTCs at Site 3 (Fig. 5d) is less clear, although a general trend of multiple peaks was observed. These sites were variably flooded during sample collection and may have had different (and transient) connection with the main flowpaths, which may explain why $[Br^-]$ does not decrease to background concentrations during the sampling period; Br^- was delivered in initial inundating flow and remained in perched "puddles" and infiltration was likely very slow due to clayey alluvial deposits that underlie the wetland basin (Nahlik and Mitsch 2006). All downstream sites (i.e., 1, 2, 3, and 5) exhibited long tailing behavior, which was not completely captured by the sampling program due to time and travel constraints. Moment analysis of the BTC at Site 1 indicated mass recovery of ~40 % (based on outflow volume calculated in Kaplan et al. 2011), which would increase if sampling had continued until $[Br^-]$ decreased to background concentrations.

Initial velocity estimates and principle flow pathways

The time to first (and second, if applicable) $[Br^-]$ peak at each location is summarized in Table 1, along with the corresponding distances from the injection points. These data were used to elucidate primary flowpaths through the wetland and to provide an initial estimate

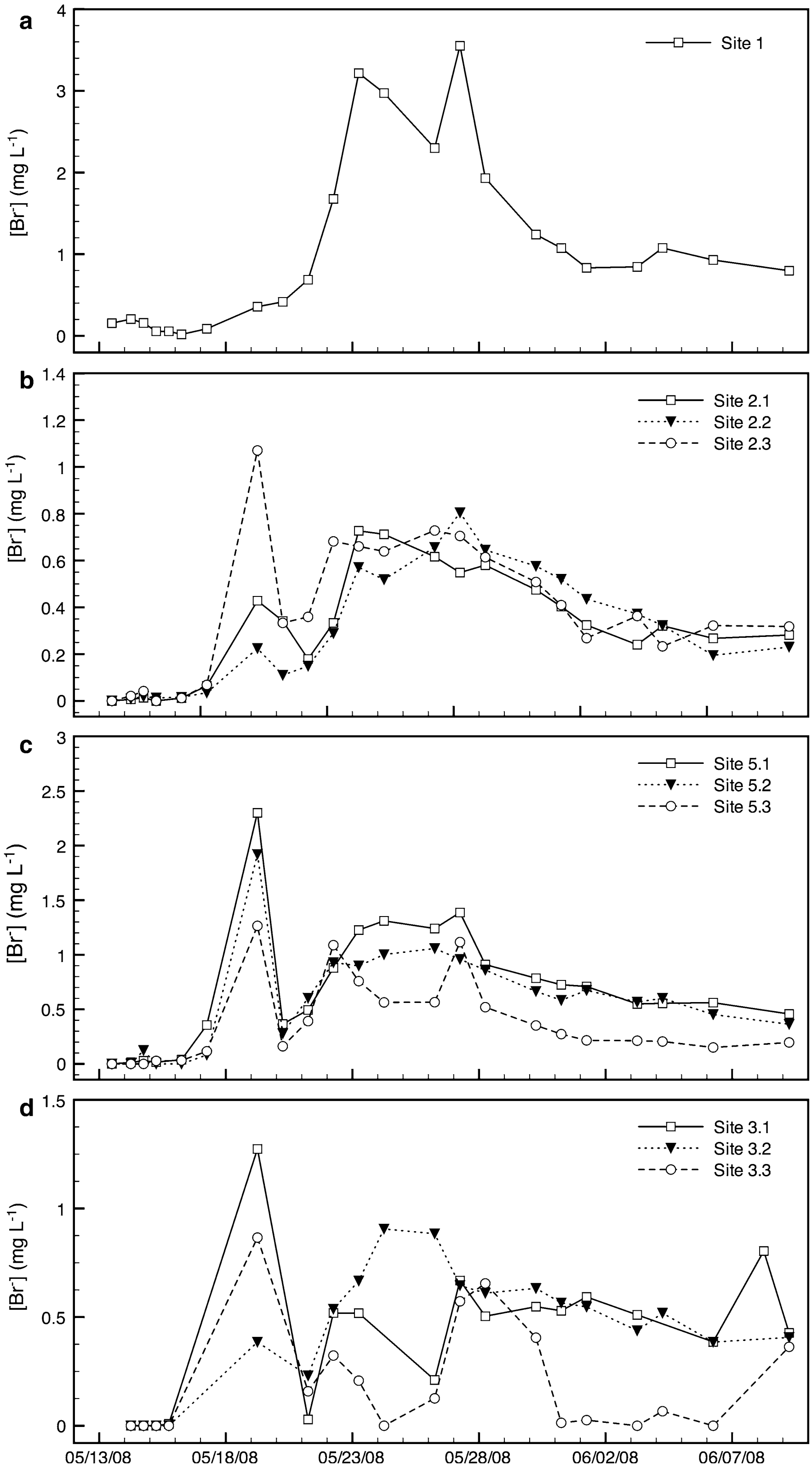

Fig. 5 Bromide concentration [Br$^-$] measured at Sites 1, 2, 3, and 5 showed evidence of double peaks, indicating that tracer from both injection sites flowed through the area

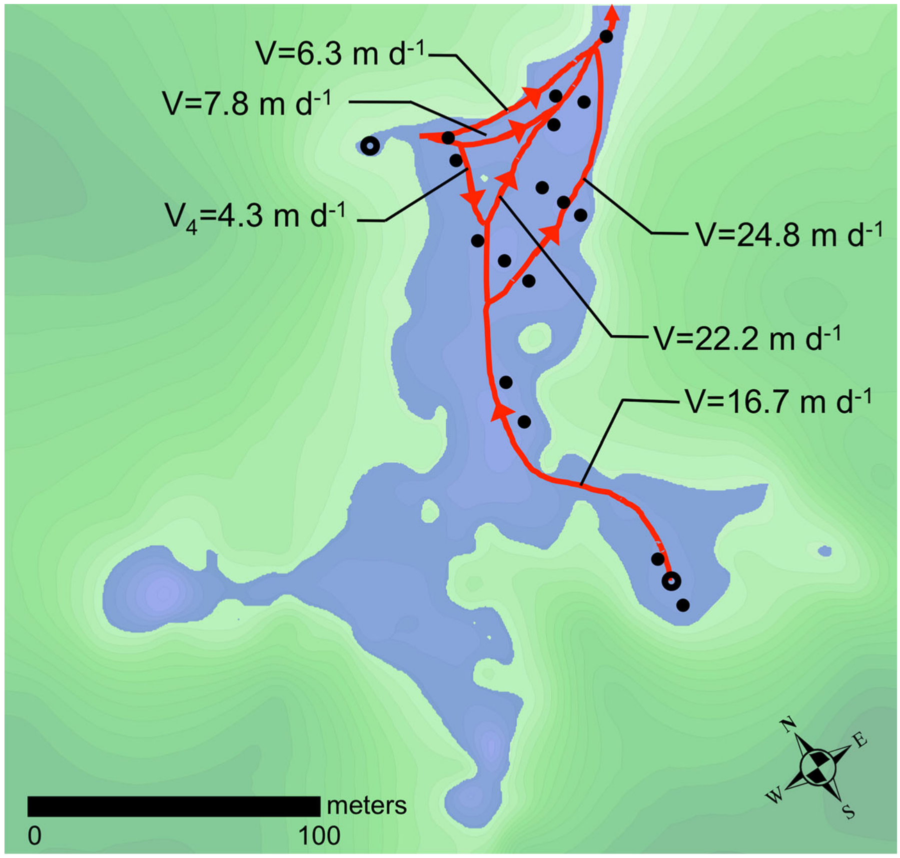

Fig. 6 Proposed flowpath distributions and average velocities along each path based on the simplified peak-to-peak analysis

of water velocities along different flowpaths. Calculated velocities using this simplified "peak-to-peak" method were markedly different in the eastern and northwestern flow channels, with substantially faster flow observed in the eastern wetland branch (Table 1). This was likely due to short sections of shallow channelized flow in the eastern branch, which was observed through the upstream, forested reach, as well as short stretches of channelized flow through the herbaceous marsh near the wetland outlet. Along the eastern branch, upstream velocities ranged from 15.6 to 17.8 m day^{-1}, while velocities over the entire eastern flowpath ranged from 16.8 to 35.3 m day^{-1}. In the western branch, velocities ranged from 4.0 to 8.3 m day^{-1} (Table 1).

In the eastern branch, flowpaths along the wetland edge were slower than central paths, while the opposite was true in the western branch. In the eastern branch, faster flow closer to the wetland center was likely due to the presence of several shallow channels away from the wetland edge. In the western branch, the opposite was true: the central flow path passed through flat areas of dense vegetation as shallow sheet flow, and a slightly more defined channel existed close to the wetland edge. Thus, flow may be distributed between two (or more) flow paths in the eastern branch: a central flow path (i.e., from Injection Site A → Site 6 → Site 5.3 → Site 3.3 → Site 2 → Site 1) or along the wetland edge (i.e., from Injection Site A → Site 6 → Site 5.1 → Site 3.1 → Site 1) (Fig. 5). Flow along the western branch appears to be distributed between at least two pathways: from Injection Site B → Site 4.1 → Site 2 → Site 1; from Injection Site B → and from Injection Site B → Site 5 → Site 3 → Site 2 → Site 1 (Fig. 5). Finally, [Br^-] observed at Site 1 was ~3× higher than at Site 2, suggesting a third possible flowpath from Site 4.2 to the outlet that bypasses Sites 2.1, 2.2, and 2.3 (Fig. 6). We note that these flowpaths were observed under relatively low-flow conditions and may be different during periods of higher flow (see "Discussion" section).

Table 2 Velocity (v), dispersion coefficients (D), and model goodness of fit (R^2) for 1-D advection–dispersion models of observed tracer data

	Eastern path			Northwestern path			Combined
	V (m day^{-1})	D (m^2 day^{-1})	R^2	V (m day^{-1})	D (m^2 day^{-1})	R^2	R^2
Site 6.1	14.6	14.6	0.67	–	–	–	–
Site 6.2	14.4	36.6	0.49	–	–	–	–
Site 4.1	–	–	–	7.6	1.9	0.86	–
Site 4.2	–	–	–	5.5	0.02	0.92	–
Site 5.1	24.5	31.0	0.93	4.0	18.8	0.88	0.88
Site 5.2	24.0	23.6	0.99	3.53	20.6	0.91	0.90
Site 5.3	26.0	28.5	0.99	3.8	11	0.73	0.74
Site 2.1	26.5	103.0	0.99	3.3	11.6	0.83	0.81
Site 2.2	30.8	91.7	0.98	4.18	18.4	0.96	0.95
Site 2.3	32.0	63.6	0.99	3.86	15.7	0.91	0.88
Site 1	15.8	98.7	–	3.7	4.3	–	0.91
Mean	23.2	54.6		4.4	11.4		
SD	6.3	33.1		1.3	7.3		

Modeled transport parameters

Velocities and dispersion coefficients (D) estimated from fitting Eq. (1) to the observed data are summarized in Table 2. The model was not fit to BTCs from Site 3 due to intermittent flowpath connection at that site. In general, the velocities fitted using Eq. 1 were in the same range as those calculated using the simpler peak-to-peak calculation, and the contrast between faster flows in the eastern channel and slower flows in the northwestern channel persisted. Average velocities were 23.2 m day^{-1} along eastern flowpaths and 4.4 m day^{-1} along northwestern flowpaths (compare to 26 and 6 m day^{-1} calculated using the peak-to-peak calculation). Dispersion coefficients were slightly higher along the eastern flowpath (average $\pm$ SD $D = 54 \pm 33$ m^2 day^{-1}) than the northwestern flowpath ($D = 11 \pm 7$ m^2 day^{-1}), but were generally within the lower range observed in densely vegetated wetlands (see "Discussion" section).

Model fits for single peak and separated BTCs were fair to excellent ($0.49 \le R^2 \le 0.99$), with the worst performance for sites 6.1 and 6.2, downstream of Injection Site A (e.g., Fig. 7a), where the simple ADE was unable to capture both the quick peak of tracer passing through the site and the observed tailing behavior (Table 2). For sites with two clear peaks, the combined BTCs yielded good model fits ($0.74 \le R^2 \le 0.95$) (e.g., Fig. 7b–c; Table 2). For Site 1, we were unable to separate and fit two distinct tracer peaks due to greater mixing near the wetland outlet. For this site, we instead optimized model fits by varying v and D parameters for both flow pathways to best match the observed BTC data (Fig. 7d), with good results ($R^2 = 0.91$).

Discussion

Environmental pressures from growing populations and agricultural development in the tropics have driven the loss and ecological degradation of wetland and other aquatic ecosystems due to both direct and indirect impacts (e.g., Daniels and Cumming 2008; Junk 2002). Quantification of the ecosystem services provided by natural wetlands may provide motivation to maintain, conserve, and restore wetlands in the landscape, however assessment of these services requires an improved understanding of wetland hydraulics, velocities, and flow pathways. This case study aimed to characterize the hydraulic properties of a small tropical wetland in an agricultural watershed to improve our understanding of tropical wetland function and generate information in support of public decision-making regarding wetland use, conservation, and preservation. While a previous study (Kaplan et al. 2011) presented a coarse estimate of residence times within the wetland, this detailed tracer study allowed us to map likely wetland flowpath distributions and estimate solute transport parameters, which varied greatly across wetland flowpaths.

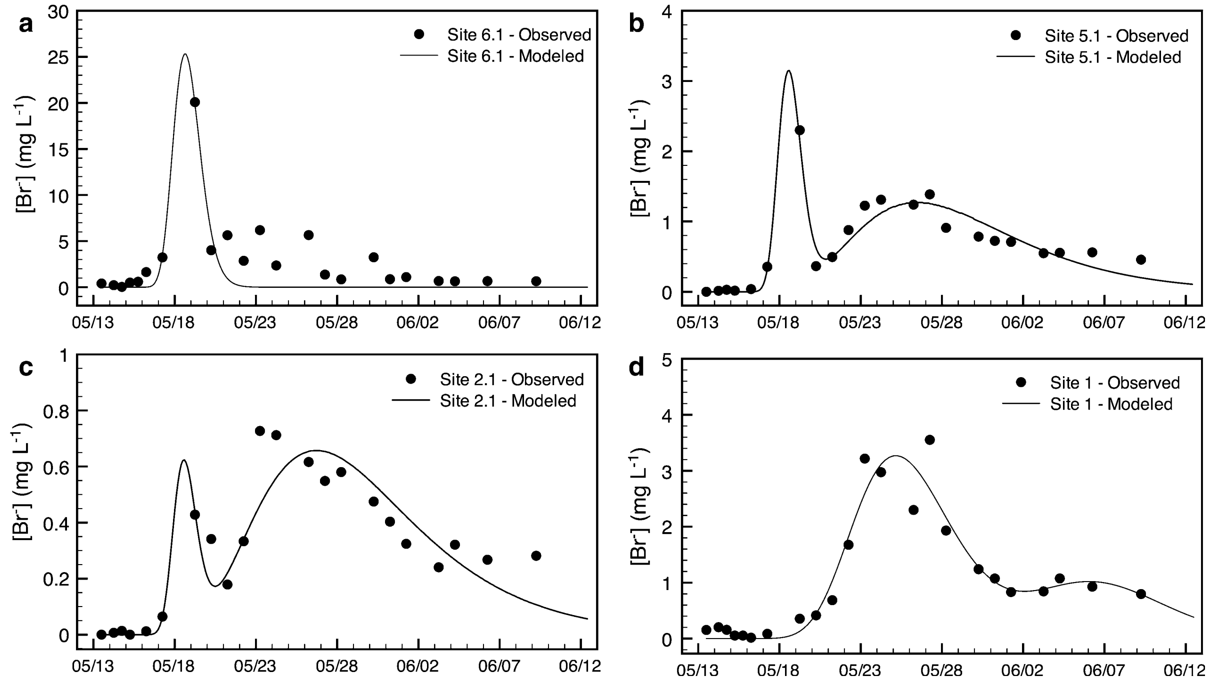

Fig. 7 Observed (*symbols*) and modeled (*lines*) breakthrough curves at **a** Site 6.1 downstream of Injection Site A; **b**, **c** Sites 5.1 and 2.1 downstream of the confluence of Injection Sites A and B; and **d** Site 1 at the wetland outlet

Modeling observed tracer BTCs using a steady state, 1-D ADE yielded good model fits in most cases (Table 2; Fig. 6), however this approach has clear limitations in dynamic and heterogeneous natural wetland systems. For example, while measured outflow was relatively constant over the study period (between 3.5 and 5 L s^{-1}; Fig. 3), precipitation events (Fig. 3) caused local water level changes that appeared to change flowpath connectivity over small spatial scales. In addition to the variable tracer signal observed at Site 3 (Fig. 5d), small secondary pulses of tracer at Sites 4.1 and 4.2 (downstream of Injection Site B) suggest that precipitation events in late May mobilized additional tracer mass that had been temporarily immobilized in "dead zones", but which contributed to solute transport once water levels increased. This is supported by inundation maps in Kaplan et al. (2011, Online Resource 1) that show variable connectivity as a function of inundation depth at this location. This variability is particularly important to consider in light of the timing of the study during a period of relatively low flows; the flowpaths and residence time distributions (see below) observed were specific to this flow regime, and additional tracer injections during the wet season would be required to characterize wetland hydraulics during that time.

The long tailing observed at most locations, is likely due to transient storage and release, which has been found to affect tracer transport in streams and wetlands (Runkel et al. 1998; Martinez and Wise 2003). Several authors have modeled this phenomenon using the one-dimensional transport with inflow and storage (OTIS) model (Runkel et al. 1998), however these applications have been restricted to modeling tracer BTCs at the outlet of treatment wetland systems (e.g., Martinez and Wise 2003; Keefe et al. 2004); OTIS application at multiple locations within a wetland is made difficult by the need to for accurate estimates of local flow and parameterization of main channel and storage zone cross-sections and storage zone exchange coefficients. In addition, noise in the [Br^-] signal across all sites can be partially attributed to the relatively coarse (temporal) sampling scale of sample collection. Despite these limitations, the results presented here provide a good, albeit simplified and time-specific, estimate of wetland transport properties and flowpath distribution.

Observed (and modeled) velocities in the study wetland were exceedingly slow: from less than 4 m day^{-1} to a maximum of ~30 m day^{-1} (Table 2). These velocities are an order of magnitude lower than those measured in the low-gradient Everglades (Florida, USA), where typical velocities have been found to range from a minimum of ~170 m day^{-1} (Leonard et al. 2006) to ~1 300 m day^{-1} (Schaffranek and Riscassi 2004). Indeed, observed velocities were closer in the range to those observed in pervious porous media (e.g., Bear 1988). In many parts of the wetland, the presence of free surface water was limited to shallow flooding of a few cm above a dense substrate of living and dead plant roots overlaying organic peat soils. The low velocities observed in this study suggest that a portion of the wetland flow was moving through this dense network, leading to extremely high friction and low velocities. Even in sections of the wetland where shallow sheet flow and shallow channelized flow were observed, a high density of emergent wetland vegetation (Kaplan et al. 2011) also created high friction and low velocity. These low velocities support slow release of water to downstream riverine systems during periods of low flow and correspond to long residence times, which enhance biogeochemical processing.

Modeled dispersion coefficients estimated with the 1-D ADE were correspondingly low, with higher average values of D (54.6 ± 33.1 m^2 day^{-1}) in the faster-flowing eastern flow path compared to the slower-moving northwestern path (11.4 ± 7.3 m^2 day^{-1}) (Table 2). These values are within the low range reported for other slow-flowing wetlands. For example, Min and Wise (2009) found dispersion coefficients to range between 8 and 6 700 m^2 day^{-1} for a treatment wetland in Florida. Similarly, Saiers et al. (2003) found dispersion coefficients to vary from ~2 to ~400 m^2 day^{-1} in a tracer study conducted in the Everglades. Conversely, larger-scale tracer studies in the Everglades by Ho et al. (2009) found longitudinal dispersion coefficients on the order of ~3,000 to ~22,500 m^2 day^{-1} (though these authors also note the effect of scale dependence). Assuming characteristic lengths of 207 and 93 m for the eastern and western flow paths respectively, Peclet numbers for the eastern and northwestern flow paths were 124 and 74, indicating advection-dominated flow (albeit at very low velocities).

The exceedingly low velocities observed in this study correspond to (relatively) long wetland residence times (τ), which support biogeochemical processing. Based on a 1-year wetland water balance, Kaplan et al. (2011) found the 95 % confidence interval of τ to be between 29 and 138 days. A moment analysis of the BTC observed at Site 1 near the wetland outlet (i.e., dividing the first moment by the zeroth moment) yielded a shorter mean τ of 18.9 days. Differences in τ values calculated in the two studies may be attributed to the fact that the water balance method accounted for the entire wetland volume over a 1-year monitoring period, while this study focused on specific flow paths over a shorter monitoring period. Moreover, calculating τ using the water balance method (i.e., dividing wetland outflow by wetland volume) assumes complete mixing (clearly not the case in this system) and tends to overestimate wetland residence time (Mitsch and Gosselink 2007). The lower value of τ measured in this tracer study may therefore be viewed as good (though conservative) estimate of average residence time during the study period. It should also be noted that the value of τ calculated in this study (i.e., using moment analysis) represents water traveling along the flow path from injection points to the outlet; water entering the system upstream or downstream of the injection points will have longer and shorter mean τ values, respectively. Moreover, this hydraulic characterization is specific to the flow regime observed during the study; during periods of higher rainfall, residence times will be shorter and, as noted above, flow paths will likely change.

While this wetland has no specific surface water inlet, it is located in a watershed with intensive banana production and is likely a de facto treatment wetland for agricultural runoff that contains nutrients, fungicides, and pesticides. Using a simple model of pollutant removal (the k-C* model; Kadlec and Wallace 2008), Kaplan et al. (2011) estimated potential pollutant removal rates in this wetland over a range of influent concentrations and τ values, including the mean τ value (18.9 days) found in this study: 63.6–79.3 % for biological oxygen demand; 60.0–99.8 % for total suspended solids; 51.1–71.8 % for total nitrogen; and 34.2–42.7 % for total phosphorous. While these removal rates are based on simple models, they provide a baseline for understanding the important water quality improvement role of this and other small tropical wetlands in agricultural watersheds.

As noted above, these potential pollutant reduction rates are specific to the flow regime observed during this study. Critically, lower removal efficiencies are likely during periods of heavy rainfall, when large fluxes of water drive reductions in τ and pulses of agrochemicals may be expected. This effect is somewhat mitigated by the fact that wetland area and volume increase during periods higher flow periods, which serves to decrease hydraulic loading rate and increase residence time, respectively, given the same flow. However, the observed interannual variability in wetland area (1.35–1.60 ha) and wetland volume (7,000 to 10,500 m^3) is relatively low (Kaplan et al. 2011). On the other hand, we note that potential pollutant reduction rates listed above are for the shortest residence times calculated in Kaplan et al. (2011) (i.e., 2.5–21 days), which correspond to periods of high flow and high hydraulic loading rates. Thus, these moderate levels of treatment are likely achieved even during periods of higher flows.

The distribution of sampling locations across the wetland allowed us to delineate primary flow pathways and demonstrate mixing of eastern and northwestern flow paths (Fig. 4). This spatial heterogeneity in velocity, flowpath connection, and flow distribution is likely important for supporting a number of wetland functions by providing: (1) proximally located aerobic and anaerobic zones for nitrogen cycling (Kadlec and Wallace 2008); (2) differential sediment deposition zones (Johnston 1991) that support a variety of plant communities (Chambers et al. 1991) and a diversity of flow regimes and associated macroinvertebate habitats (Gallardo et al. 2008). Support for these three predictions are provided by the low nitrogen levels observed by Gallardo and César (2006) and the wide diversity of plant species noted by Kolln (2008) and Mitsch et al. (2008). It is also interesting to note the apparent "slow-down" upstream of the wetland outlet culvert that appears to delay and mix the tracer pulses from the two injection sites, indicating a backwater effect due to constricted outflow (Site 1; Fig. 3a). Notably, however, the effect is very local; it is not apparent at Site 2, less than 30 m upstream.

As noted by a multitude of authors (e.g., Zedler 2003; Hansson et al. 2005; Feld et al. 2009), wetlands in agricultural landscapes have the potential to provide myriad ecosystem services, but are limited in their ability to provide all potential services simultaneously or indefinitely. For example, Hansson et al. (2005) conclude that constructed wetlands may support high biodiversity and nitrogen retention rates via shallow depths and shoreline complexity, but that deeper wetlands are required to maximize phosphorus retention—at the expense of biodiversity. Similar tradeoffs are inherent in the delivery of ecosystem services by natural tropical wetlands, which can provide both resource (i.e., direct economic) values (e.g., agriculture, fisheries, forage, energy, etc.) and environmental values (biodiversity, nutrient processing and sediment retention, hydrologic buffering, climate stabilization, etc.) (Roggeri 1995).

For the case of small tropical wetlands in developing agricultural landscapes, the degree to which overall ecological integrity (e.g. water quality, soil structure, biodiversity, foodwebs, etc.) is degraded by the delivery of agricultural runoff is not well studied. In general, the addition of high-nutrient runoff to these systems has the potential to impact primary producer community structure (PPCS) via eutrophication (Sánchez-Carrillo et al. 2011), although this threat is reduced in forested wetlands and even herbaceous wetlands with dense canopies given reduced light availability in the water column. Moreover, consistently high air and water temperatures facilitate rapid biogeochemical processing (Kadlec and Wallace 2008) relative to similarly sized systems in temperate environments, potentially decreasing the relative impact of nutrient loading on PPCS. On the other hand, bioaccumulation or biomagnification of herbicides and pesticides is of concern in any system that may serve as a "sink" for these chemicals. Given the propensity of many organic chemicals to bind tightly to soils with high clay and organic matter contents (Grundl and Small 1993)—and the existence of multiple trophic levels for accumulation within wetland ecosystems (e.g., Snodgrass et al. 2000)—this of particular concern in wetlands; any benefits to downstream water quality must therefore be considered in light of these tradeoffs. Similar tradeoffs must be considered for the case of increased sediment retention, which negatively impacts wetland function (e.g., Gleason and Euliss 1998), but prevents sediment transport downstream to rivers and estuaries.

In addition to highlighting spatial heterogeneity in transport parameters, the tracer study also demonstrated the temporal variability of the flowpath connectivity and mixing driven by water level, topography, and vegetation over very short spatial scales (e.g., variable free surface water at Site 3 and order of magnitude differences in mixing at Site 4). A previous hydrologic study that used measured surface water elevation and wetland topography to determine inundation (Kaplan et al. 2011) suggested only small variations in the area flooded over wet and dry seasons. This and other wetland studies suggest that small changes in the nature of this inundation (i.e., flooding of tens of cm vs. minimal free surface water inundation above dense emergent plant stems and roots) can lead to vastly different transport patterns and pathways (e.g., Kaplan et al. 2012; Choi and Harvey 2014). Similarly, the hydraulic properties related to residence time and water quality enhancement potential related to residence time are specific to the relatively low-flow conditions observed during the study, although as noted above, moderate treatment efficiencies are likely even for shorter travel paths.

The results of this tracer study complement the hydrologic data presented in Kaplan et al. (2011). In particular, this study provides estimates of specific flow paths and allowed us to estimate transport parameters that may be used in future hydrodynamic modeling, while the previous study only provided a coarse estimate of bulk transport assuming complete mixing. On the other hand, the results presented here are limited to periods of relatively low flow, while the previous study presented a full year of hydrologic data. In short, this study benefits greatly from the longer-term hydrologic data collection and water balance presented in Kaplan et al. (2011) and provides finer-scale information about system hydraulics. A better understanding of these hydraulics is required to fully account for solute transport and mixing in densely vegetated, heterogeneous wetland systems like the wetland investigated in this case study. In addition to providing site-specific measures of wetland hydraulics, these results may be leveraged to develop hydrodynamic modeling efforts to analyze interactions between vegetation, topography, and water flow and to explore the relative importance that small-scale heterogeneity introduces on the effective hydraulic behavior.

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