



Dynamics of particles and phosphorus in canals of the Lower Everglades, Florida, USA

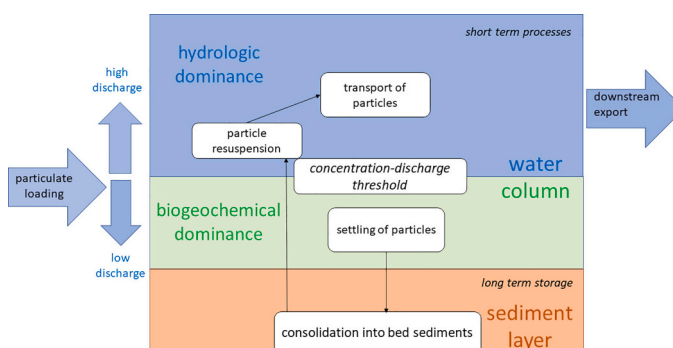
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HIGHLIGHTS

- Concentration–Discharge (C–Q) relationships are a useful tool to describe particle and phosphorus export behaviors in managed canals.
- High temporal resolution data improves the ability to determine discharge-driven concentrations and transport.
- Discharge affects the mass accumulation and settling distances of particles.
- Canal sediments are long-term integrators of short-term discharge-driven particle dynamics.

GRAPHICAL ABSTRACT



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ABSTRACT

Water flow (discharge) can affect water quality by influencing the concentration and transport of waterborne contaminants. The effects of discharge on phosphorus (P) and particle concentrations in managed canals, were described using concentration-discharge (C–Q) relationships, accumulation of suspended and settling particles, and the physicochemical characteristics of these particles and bed sediments. Piecewise regression analysis on C–Q relationships revealed slope inflections that denoted thresholds, where P-behavior changed from low to high discharge. The C–Q relationships generally showed higher concentrations at higher discharges. In three of the four Lower Everglades canals studied, long-term (1995–2019) lower temporal resolution data (daily to weekly) was adequate to describe the influence of discharge on P concentrations. However, in one site, the L–29 Canal, higher temporal resolution data (minutes to hours over weeks), derived from acoustic sensors, was necessary to produce C–Q relationships. In the L–29 Canal, discharge affected the transport, settling, and sediment accrual at distances from the S333 inflow structure. Sediment traps showed higher discharge led to a greater accumulation of suspended particles that were transported and settled farther downstream. Generally, downstream surface sediments in the L–29 Canal had greater organic matter, lower bulk density and higher TP than those of the upstream site, reflecting long-term effects of discharge. Understanding the effects of discharge on particles and associated nutrients, especially at discharge thresholds that lead to concentration increases, can inform the operation of managed canals to reduce contaminant loading to downstream sensitive ecosystems.

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1. Introduction

Waterborne transport of nutrients from areas of high generation (e.g., agricultural fertilizers) can cause the detrimental eutrophication of susceptible downstream environments. Phosphorus (P) is an often-limiting nutrient that exists in particulate and dissolved forms. Particulate P can be in mineral forms, often associated with calcium or iron, or in organic matter, both living and detrital (Trefry and Fox, 2021; Labry et al., 2013; Yoshimura et al., 2007). Particulate P transport can be a significant source of total P loading to rivers and canals. For example, in South Florida, particulate P can range from 14 % of TP in canals draining wetlands to 54 % in those draining agricultural areas (Daroub et al., 2002a; Onwuka, unpublished data).

Transport of particulate P in flowing water depends on several factors including water discharge, channel morphology, and the physicochemical characteristics of the particles (e.g., organic matter content, mineral content, bulk density) (Reddy et al., 1999; Daroub et al., 2002b; Rose et al., 2018; Fugate et al., 2021; Onwuka et al., 2023). During low discharge events, P-bearing particles can settle from the water column to the sediment surface where they can undergo biogeochemical processes including diagenesis and decomposition, or can accrue into long-term, relatively stable sediments (Svendsen and Kronvang, 1993; Reddy et al., 1999; Diaz et al., 2006). Greater discharge increases particle loading and when discharge exceeds a critical threshold, accrued sediments can be resuspended by shearing forces (Reddy et al., 1999; Daroub et al., 2002b; Fugate et al., 2021). Heavy mineral particles are not transported as easily as lighter organic particles (Daroub et al., 2002b). Thus, understanding the dynamics of particles, P, and discharge are crucial for P management, and hydrologic and ecosystem restoration of ecosystems.

Concentration-Discharge (C-Q) relationship is a widely-used approach to predict the effects of discharge on suspended particle concentrations and particulate-bound nutrients, including P (Godsey et al., 2009; Moatar et al., 2017; Onwuka et al., 2021). Particle export behaviors are determined by a power-law function of the relationship between particle concentration and discharge. Decreasing concentration with increased discharge (flow) indicates dilution, while increasing concentration with increased discharge denotes mobilization and enrichment (Zhang, 2018). An inflection point in the slope (b) of the C-Q plot can exist where the behavior of a constituent changes. This inflection is termed the discharge threshold and denotes the discharge where the constituent behavior changes between relatively low (below threshold) and high (above threshold) discharges (Moatar et al., 2017; Underwood et al., 2017). Particle export behaviors that depend on discharge are chemodynamic (hydrologically-dominated). Concentrations that are insensitive to changes in discharge (slopes not significantly different than 0) are chemostatic (biogeochemically-dominated). A common C-Q relationship export behavior for particles is chemostasis at low discharge and chemodynamic enrichment or mobilization at high discharge. The mobilization of sediment and dislodging of sorbed nutrients at higher discharge (e.g., P) increases their water column concentrations (Diamond and Cohen, 2018).

Historically, the Everglades was a continuous system originating in the Kissimmee River-Lake Okeechobee basin that flowed southward into Florida Bay and the southeast coast. Starting in the early 20th century, an extensive water management system - a network of canals, levees, roads, and water control structures - was created to drain the Everglades wetlands for agriculture and urban development (Davis and Ogden, 1994; Reddy and DeLaune, 2008). These changes in hydrology and the resultant change in land cover led to the loss of approximately 50 % of the original Everglades (Sklar et al., 1999; Sklar et al., 2001; Reddy and DeLaune, 2008; McVoy et al., 2011). Hydrologic alterations for agricultural and urban development compartmentalized the Lower Everglades (defined here as that portion of the system south of Lake Okeechobee) into the Everglades Agricultural Area (EAA), the Lower East Coast (LEC), and the Everglades Protection Area (EPA; that includes

three water conservation areas (WCAs) and Everglades National Park, ENP) (Fig. 1). These three areas generally correlate to agricultural, urban, and “natural” wetland land use respectively.

The historically P-limited, oligotrophic Everglades have been negatively impacted by alterations that have affected the hydrology of the system and has led to P-induced ecosystem changes, often associated with proximity to canals (Noe et al., 2001; Childers et al., 2003). Continuous loading of P-rich water from agricultural exports to canals, largely the result of the Central and South Florida Project of 1948 (Light and Dineen, 1994), led to the accumulation of “legacy P”. In the WCAs, inventories estimated P concentrations in canal sediments to be twice that in surrounding wetland soils (Wang et al., 2011). Canals are ubiquitous features in the South Florida landscape and are different from natural water bodies such as rivers, lakes, and streams. Canal discharges can fluctuate widely, by operation of water control structures such as pumps and gated spillways (e.g., G136, S333, etc.), ranging from no-discharge periods (where the canal acts like a reservoir) to high-discharge periods. These canals are steep-sided and have depths that range from a few meters to 11 m, and unlike rivers, have reduced or non-existent ability for bankfull discharge that can reduce flow velocities (Carter et al., 2010; UFIFAS, 2023).

Previous studies in South Florida canals quantified P release from sediments by processes including desorption and diffusion under conditions of zero or low discharges (Diaz et al., 2006; Wang and Li, 2010; Das et al., 2012a). These chemostatic processes do not account for the total phosphorus (TP) flux as they have not considered hydrologically-driven P release through sediment transport and resuspension. A more complete estimation of P release through canals requires quantifying the effects of hydrologic drivers, such as discharge, and their contributions to net P export. The South Florida Water Management District (SFWMD) has a well-developed, robust, and extensive series of monitoring protocols and stations distributed across most of South Florida, with data readily available in an online repository (DBHYDRO-<https://www.sfwmd.gov/science-data/dbhydro>). However, data collection and reporting in this database are too temporally coarse (low resolution –weekly to monthly measurements) to capture the high frequency data needed to understand the impact of short-term (minutes to hours) changes in discharge. Short-term changes in discharge commonly occur in manipulated (pumped flows through control structures) canals. Characterization of discharge-influenced transport is best obtained with temporally and spatially high-resolution measurements or surrogate estimates (Onwuka et al., 2023). The objectives of this study were to quantify the effects of managed canal discharges on suspended particles and P export in South Florida's Everglades, using both long term (course resolution over 25 years) and short term (hourly over several days) data. A guiding principle in Everglades Restoration is “getting the water right” (SFERTF, 2010) with an objective of sending increased water deliveries into the southern Everglades and ultimately Florida Bay. Concurrently, the extensive eutrophication that has occurred due to P enrichment of the oligotrophic system must be avoided. Therefore, it becomes critical to determine how to maximize the delivery of clean water while simultaneously reducing the negative effects of increased P. Not ensuring waters delivered to the Everglades are clean may partially defeat the purpose of restoration and may incur additional costs to remediate. Specific objectives were to determine how canal discharge affects particle and P transport within the Lower Everglades canals. Predictions include: 1) increased P concentrations would be observed during relatively high discharge, 2) at some critical discharge, there would be a change from chemostatic to chemodynamic behavior of P, and 3) physicochemical characteristics of sediments (suspended and bed) will vary according to the magnitude of discharge and proximity to canal inflow structures.

2. Materials and methods

2.1. Study site

Canals that drain and interact with the EAA, LEC, and EPA, including the Tamiami Canal (L-29) were selected for this study. The L-29 Canal is a major EPA canal that delivers water to ENP. The L-29 Canal was created by limestone excavation subsequently used to build the Tamiami roadway that connected Miami to Tampa. As part of the Modified Waters Deliveries project, about 5.8 km of bridges have been created along the roadway to enable more freshwater input to ENP (McLean, 2015; USACE, 2020). Additional details on the L-29 Canal are found in Onwuka et al., 2023.

2.2. Discharge and water column phosphorus data

Discharge and P data for twenty-five years (1995–2019) were used in this study. The sites selected on the canals were the L-1 Canal (G136), located in Hendry County at the G136 structure (26°40'03.5"N 80°56'57.3" W), West Palm Beach Canal (S5a) at the S5a structure (26°41'02.7"N 80°22'03.9" W), South New River Canal at the S9 structure (26°03'41.5"N 80°26'36.8" W), and the L-29 Canal (S333) at the S333 structure (25°45'43.2"N 80°40'26.2"W) (Fig. 1a). These sites' mean daily discharge and weekly grab total phosphorus (TP, which is a measure of dissolved and particulate P forms) and where available, total dissolved phosphorus (TDP) concentration data were downloaded from DBHYDRO (<https://www.sfwmd.gov/science-data/dbhydro>, accessed on 12 October 2022). In cases where TDP was reported, calculated total particulate phosphorus (TPP_{cal}) was determined as TP minus TDP.

In the L-29 Canal, which receives majority of its discharge from the S333 structure, velocity and echo intensity (a byproduct of velocity measurements) from an upward-looking acoustic Doppler current

profiler (ADCP) (SonTEK-a Xylem Brand, San Diego, CA, USA), was used to determine higher temporal resolution (every 20 to 50 min) discharge and total suspended solids (TSS; subsequently converted to TPP) respectively at two sites. The sites were Upstream (25°45'40.8"N, 80°40'18.8"W) located near the S333 structure (data collected in November 2021) and Downstream (25°45'40.6"N 80°39'06.3"W), approximately 2.4 km downstream of S333 and located before the start of the 4.2 km bridge (data collected in June 2021) (Fig. 1b). To convert the velocity to discharge, the velocity from the ADCP was multiplied by the cross-sectional area (CSA) of the canal. The CSA was estimated using the average water depth (approx. 4.5 m) and the width of the canal (approx. 34 m). Mean daily discharges in the L-29 Canal at Downstream (25°45'41"N, 80°39'17"W) and Interior (25°45'41.0"N 80°32'12.0"W) were also downloaded from the United States Geological Survey (USGS) -National Water Information System (<https://waterdata.usgs.gov/nwis>, accessed on 26 October 2022). The discharges at Interior had both positive and negative values. The negative discharges were taken as flows in the reverse direction (east to west) caused by factors including gate operations at a pump at the eastern end of the canal, and wind (Lopez Roque, 2022). Such negative values were converted to positive values for analysis.

A detailed description of the methods used to convert echo intensity (EI) from the ADCP to TPP can be found in Onwuka et al., 2023. In brief, calibrations were done between EI and measured TSS at each site, and the calibration coefficients were used to convert the EI to TSS estimates (TSS_{est}) for the ADCP deployment period at each site. The measured TSS (used in the calibration) was filtered from 1-L water samples collected using a peristaltic pump and were kept on ice and transported to the lab. In the lab, pre-dried (105 °C for 1 h) and pre-weighed 47 mm GFF Whatman filters were used to filter a known volume of the collected water samples. After filtration, the filter and particles (residue) were dried again, cooled in a desiccator, and weighed. The TSS was calculated

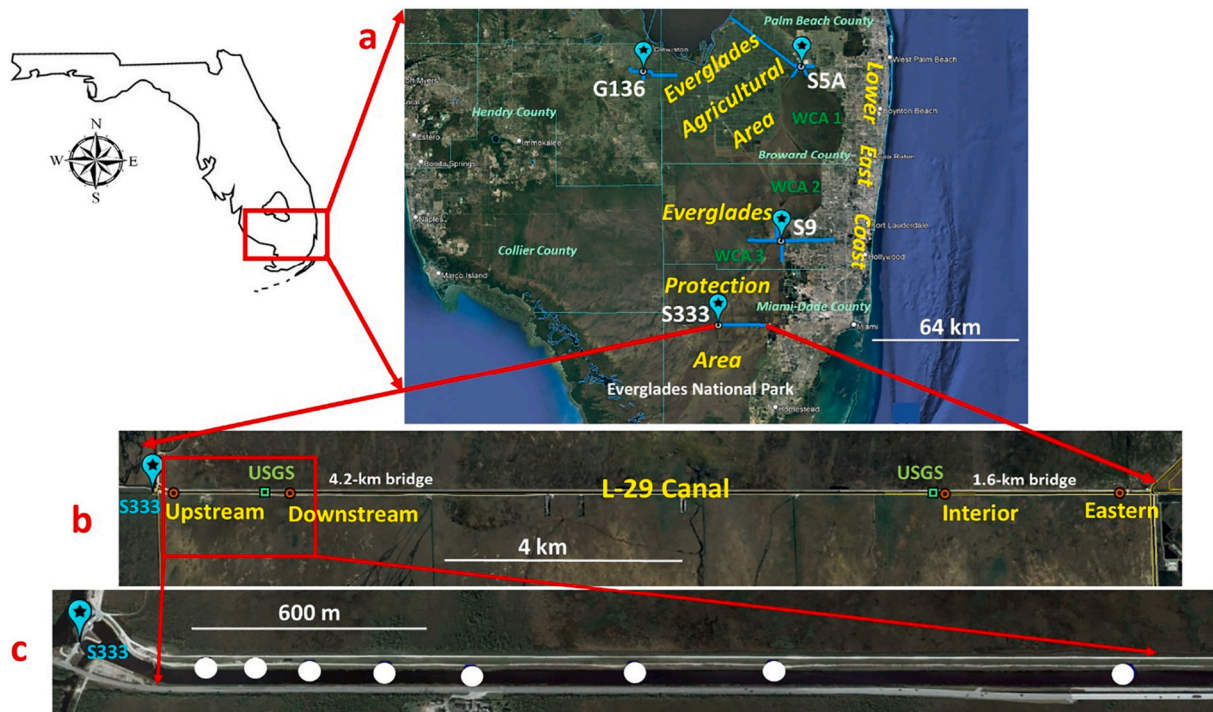


Fig. 1. a) Satellite image of Lower Everglades in South Florida showing canal sites in the Everglades Agricultural Area (G136, S5a), Lower East Coast (S9), and Everglades Protection Area (S333). The South Florida Water Management District (SFWMD) monitors these sites with the data stored in their online repository - DBHYDRO. b) L-29 portion of the Tamiami Canal showing four sites where samples were collected: The sites are: Upstream, Downstream, Interior, and Eastern. Water, velocity, suspended particles, and bed sediments were collected at all sites, except at EA (only bed sediments). c) Sites 1–8 are the sites of single bottom traps (settling particles collection) between Upstream and Downstream. Downstream is at the western bridge opening that allows flow into Everglades National Park (ENP). Maps were created from Google Earth (2022). The United States Geological Survey (USGS) discharge sites in the L-29 Canal are shown.

as the difference between pre-filtered and post-filtered weights divided by the volume of water filtered. Next, the measured TSS were analyzed for TPP ($\mu\text{g mg}^{-1}$) using the ascorbic acid method after digestion, as described below. This TPP ($\mu\text{g mg}^{-1}$) was then multiplied to the corresponding TSS_{est} (mg L^{-1}) to get estimated TPP (TPP_{est} , $\mu\text{g L}^{-1}$) in the water column.

2.3. Concentration–Discharge (C–Q) relationships

Concentration–Discharge (C–Q) relationship, a power law function, can be linearly expressed as:

$$\log_{10} C = \log_{10} a + b \log_{10} Q \quad (1)$$

where a and b are the coefficients that represent the intercept (same unit as concentration) and slope (unitless), respectively (Moatar et al., 2017). For each canal, discharge and P concentrations (both measured and estimated) were paired by date or time. Since discharges are regulated, the data used were limited to periods when canals were flowing. The data pairs were log-transformed and regressed to yield slopes (b) (Moatar et al., 2017) that were used to quantify the effect of discharge on P concentrations. Methodology details and statistical analysis for C–Q relationships are available in Onwuka et al., 2021.

2.4. Sediment collection and analysis

High-aspect ratio traps (HARTs), with a 10:1 (length to diameter) ratio, were deployed in the L-29 Canal in triplicates (trap tree) and singly (bottom traps) to collect suspended particles in the water column and near the bed, respectively, in 2021 (Fig. 1b and S1a). The trap tree (Fig. S1a) was constructed of PVC with a center mast of about 4 m in length to which three arms, one meter apart, were attached to hold HARTs at varying depth in the water column. HARTs were positioned so the openings were nominally 1 m apart (vertically) and were offset from each other by 120° to collect suspended particles near the surface, middle and deep layers of the canal water column. The trap tree was deployed once at Upstream (in November, Upstream_Nov) and at Interior (in August, Interior_Aug) and twice at Downstream (Downstream_July and Downstream_Dec) for two to three-week periods. The bottom traps (Fig. 1c and S1b) were deployed on the bottom of the canal to collect settling suspended particles. These were deployed twice between Upstream (BT1) and Downstream (BT8) for two-week periods between October and November 2021.

Triplicate intact sediment cores were collected from four sites in the L-29 Canal using a piston corer with clear plastic inserts. The corer was extended into the water column (using PVC pipe add-ons) and through the sediments until refusal, denoting the canal bottom was reached. Cores at Upstream, Downstream, and Interior ($25^\circ 45' 39.1''\text{N}$ $80^\circ 32' 03.5''\text{W}$) were collected in April 2021, and at Eastern ($25^\circ 45' 40.6''\text{N}$ $80^\circ 30' 13.4''\text{W}$) in September 2021. The cores were sectioned with depth according to physically observable differences.

Trapped sediment and sectioned core sediment samples were either immediately processed or stored in a refrigerator until analysis. Bulk density (BD) was determined by drying subsamples at 80°C until a constant weight. All subsample weights were done on an analytical balance with a readability to 0.0001 g. Bulk density was estimated by dividing the dry weight by the sediment core volume. Subsamples were then ground and used to determine organic matter (OM) and total phosphorus (TP). Ashing subsamples at 550°C following the loss-on-ignition method provided OM content (Nelson and Sommers, 1996). The TP was determined through dry combustion with MgSO_4 in a muffle furnace for 2 h at 550°C , followed by dissolution of ash in 0.24 M HCl (Solórzano and Sharp, 1980) and finally by colorimetric analysis for orthophosphate P via a Technicon Autoanalyzer II System (Pulse Instruments, Ltd.; USEPA 365.1; USEPA, 1983). All phosphorus analyses were done in the NELAC Certified CREST-CACHE Laboratory at Florida

International University where the analytical precision is 2 %, accuracy is 98 %, and the MDL is $0.44 \mu\text{g g}^{-1}$ for TP in sediment/soil.

2.5. Statistical analysis

Statistical analysis of sediment physicochemical variables were performed in Microsoft Excel and R (R Institute for Statistical Computing, Vienna, Austria) using RStudio (RStudio Inc., Boston, MA, USA). Assumptions of normality and homogeneity of variance were verified, and ANOVAs followed by the Post Hoc Tukey tests were conducted to determine statistically significantly different means. Where ANOVA assumptions were not met, the data were log-transformed. Because discharge data is not usually normally distributed (Onwuka et al., 2021), the non-parametric equivalent of ANOVA, the Kruskal-Wallis test was conducted and the Dunne's post-hoc test applied to determine the statistical difference between the discharges at the sites. Linear regressions were conducted to measure the relationship, and the strength and direction of association between discharge (predictor variable) and physicochemical measurements (response variables).

3. Results

3.1. Concentration–Discharge (C–Q) relationship for phosphorus in the Lower Everglades canals based on lower versus higher temporal resolution data

There was a range in TP concentrations from the northern canals to the southern canals (Table 1). At G316 and S5a, the mean TP concentrations were over an order of magnitude higher than at S9 and S333 (see also axes in Fig. 2). On average, the TPP_{cal} accounted for between 30 and 40 % of TP in the northern canals. The range in TP concentrations (min to max) varied by as much as two orders of magnitude. Especially in the northern canals, median values were all below the mean, suggesting that many of the measured concentrations were generally lower than the mean, but average concentrations were subjected to skewing by

Table 1

Summary statistics of phosphorus (from C–Q relationship pairs) and suspended particles in the canals. Lower temporal resolution data (weekly to monthly from DBHYDRO) includes: total phosphorus (TP), total dissolved phosphorus (TDP) and calculated total particulate phosphorus ($\text{TPP}_{\text{cal}} = \text{TP} - \text{TDP}$). Higher temporal resolution data (mins to hours, this study) includes: estimated total suspended solids, TSS_{est} (from echo intensity), and estimated total particulate phosphorus (TPP_{est} from TSS_{est}).

Canal/site	Constituent	Mean $\pm$ SD	median	min	max	n
Lower temporal resolution data						
G136	TP	105 $\pm$ 104	65	17	674	458
	TPP_{cal}	30 $\pm$ 15	27	8	107	255
	TDP	59 $\pm$ 64	37	7	425	255
S5a	TP	144 $\pm$ 70	128	32	822	725
	TPP_{cal}	57 $\pm$ 64	39	4	729	219
	TDP	105 $\pm$ 62	88	12	413	219
S9	TP	15 $\pm$ 7	14	9	86	425
S333	TP	13 $\pm$ 9	10	4	140	739
Higher temporal resolution data						
L-29 Upstream	$\text{TSS}_{\text{est}}^a$	3.91 $\pm$ 0.82	3.62	2.61	6.33	148
	$\text{TPP}_{\text{est}}^a$	3.83 $\pm$ 0.80	3.55	2.55	6.20	148
L-29 Downstream	$\text{TSS}_{\text{est}}^b$	6.84 $\pm$ 0.37	6.86	5.97	7.68	225
	$\text{TPP}_{\text{est}}^b$	15 $\pm$ 0.80	14.7	12.8	16.5	225

TDP data not reported (DBHYDRO) for S9 and S333. a and b are statistically different according to ANOVA.

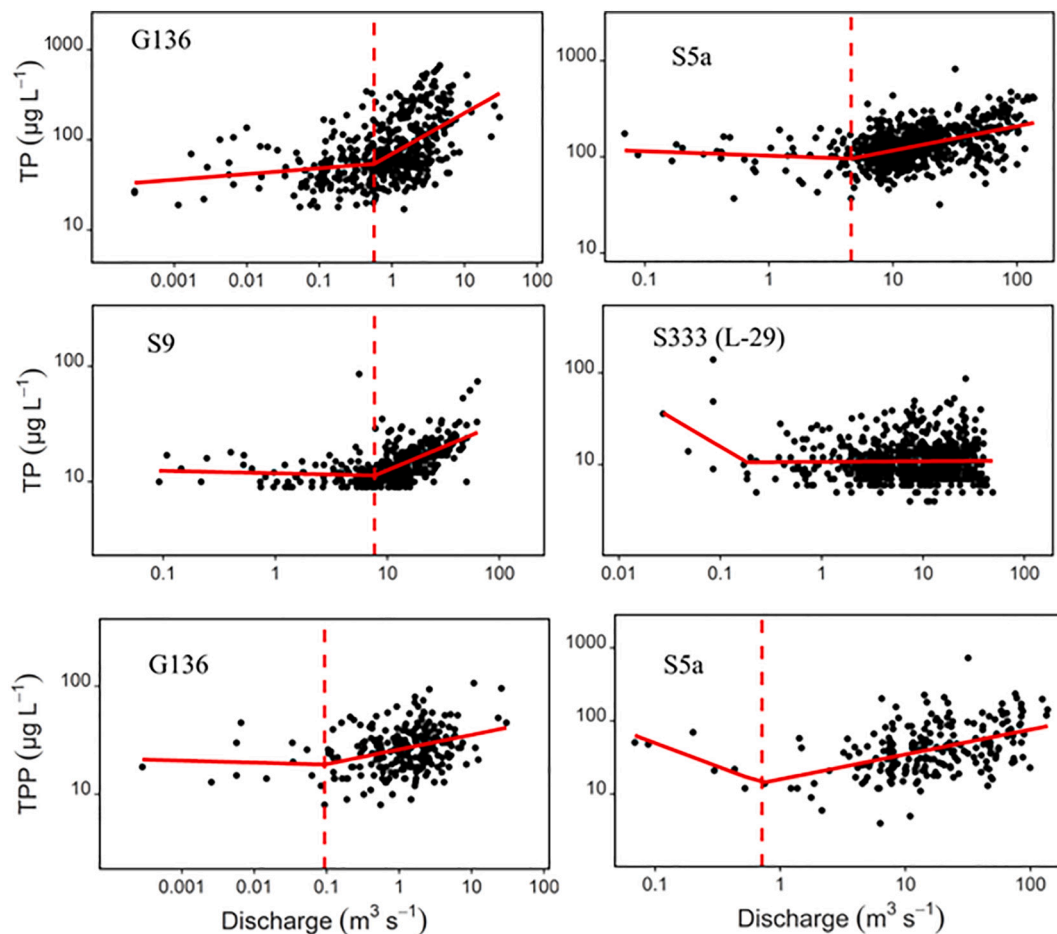


Fig. 2. C–Q relationship models with piecewise (solid red line) regressions for total phosphorus (TP) in canals G136 and S5a, that border the Everglades Agricultural Area, S9 that borders the urbanized East Coast and S333, in the Everglades Protection Area. There is a change in TP from north to south within the Lower Everglades as shown by differentially scaled axes. Calculated total particulate phosphorus (TPP_{cal}) derived from TP–TDP for G136 and S5a, where data was available are also shown.

high values. As expected, the higher temporal resolution data, with many measurements collected over one to three weeks, showed less variation and a narrower range in concentrations (Table 1).

The C–Q relationship models of the lower temporal resolution data showed that canal discharges had quantifiable effects on TP concentrations in three of the four canal sites and on TPP at G136 and S5a (where this data was available) (Fig. 2, Table 2). For TP at G136, S5a and S9, enrichment behaviors were observed with higher discharges

(positive linear regression slope, b) but at S333, the pattern ($b = -0.02$) observed was chemostatic because the slope was not statistically different from zero. Piecewise regression analysis further revealed the presence of statistically significant slope inflections (discharge thresholds) for TP at G136, S5a, and S9 (Table 2). At G136, there was a slight enrichment pattern at low discharge ($b = 0.06$) and a stronger enrichment pattern at higher discharge ($b = 0.45$). At S5a, and S9, there was a chemostatic pattern at low discharge and an enrichment pattern at high

Table 2

Concentration–Discharge (C–Q, log–log) relationship, linear and piecewise regression slopes using lower temporal resolution data (DBHYDRO): total phosphorus (TP) and calculated total particulate phosphorus (TPP_{cal}), and higher temporal resolution data (from estimated total suspended solids, TSS_{est}): estimated total particulate phosphorus (TPP_{est}) in Lower Everglades canals. Statistically significant inflections in b (discharge thresholds) according to Davies' test and P-score test⁺ are listed.

Data resolution	Phosphorus	Canal/Site	Data pairs (n)	Linear regression slopes (b)	Davies' Test or P-score Test	Discharge threshold (m^3s^{-1}) [CI]	Piecewise regression slopes (b)	
							Low	High
Lower	TP	G136	458	0.22*	$p < 0.0001$	0.57 [0.34, 0.95]	0.06*	0.45*
		S5a	725	0.17*	$p < 0.0001$	4.60 [2.68, 7.87]	−0.05	0.25*
		S9	425	0.16*	$p < 0.0001$	7.67 [6.23, 9.45]	−0.02	0.41*
		S333	739	−0.02	$p = 0.003$	0.18 [0.08, 0.43]	−0.65	0.01
	TPP _{cal}	G136	255	0.10*	$p = 0.03^+$	0.09 [0.01, 1.28]	−0.02	0.14*
		S5a	219	0.24*	$p < 0.001$	0.712 [0.23, 2.21]	−0.64*	0.34*
Higher	TPP _{est}	L-29 Upstream	148	0.29*	$p < 0.0001$	52.54 [50.30, 54.88]	−0.04	1.92*
		L-29 Downstream	225	−0.15*	$p < 0.001$	16.12 [14.76, 17.60]	0.03	−0.24*

Significant p values at 0.05 (*) according to Student's t -test (linear slopes) and ANOVA test of independence (piecewise slopes) indicating slopes are significantly different from zero. * indicates a significant difference. Confidence Interval [CI, 95 %].

discharge. At S333, although the p -value for the Davies' test was significant, ANOVA showed that these slopes were not statistically different from zero. Furthermore, visual inspection of the C-Q plot showed that there was not a definite pattern, and a few extreme data pairs skewed the low discharge slope (Fig. 2). Therefore, chemostasis was exhibited at low and high discharge. The discharge thresholds ranged from $0.57 \text{ m}^3 \text{ s}^{-1}$ at G136 in the northern part of the Lower Everglades to $7.67 \text{ m}^3 \text{ s}^{-1}$ at S9 farther south in the region. For TPP_{cal} , there was a chemostatic pattern (G136) and dilution pattern (S5a) at low discharge, and enrichment patterns at high discharge (Table 2, Fig. 2). Similar TPP analysis could not be conducted because TDP data was not available at S9 and S333.

The estimated total particulate phosphorus (TPP_{est}), from the higher temporal resolution data estimates conducted in the L-29 Canal, was statistically significantly higher at Downstream than at Upstream (Fig. 3, Table 1). In the L-29 Canal, discharge had a significant relationship with TPP_{est} at both Upstream and Downstream sites (Fig. 3, Table 2). At Upstream, TPP_{est} was enriched at higher discharge (positive linear b) while at Downstream there was a dilution at higher discharge (negative linear b). Piecewise regression analysis also revealed the presence of statistically significant discharge thresholds at both sites. At Upstream, there was a chemostatic pattern at low discharge ($b = -0.04$) and a strong enrichment pattern at high discharge ($b = 1.92$) where higher discharge increased the concentration of TPP. The discharge threshold occurred at $52.54 \text{ m}^3 \text{ s}^{-1}$ (Fig. 3, Table 2). At Downstream, there was a chemostatic pattern at low discharge ($b = 0.03$) and dilution at high discharge ($b = -0.24$), with the threshold at $16.12 \text{ m}^3 \text{ s}^{-1}$.

3.2. Physicochemical properties of suspended and settling particles

3.2.1. Trap tree

Across the sites, in the L-29 Canal, and although the deployments varied in time, the greatest mass accumulation rates (MAR) of suspended particles were observed at the upstream site closest to the S333 inflow structure (Table 3, Fig. 4a). During most deployments, regardless of site, MAR and volumes increased from the top trap to the low trap (Table 3, Fig. 4a). The interior site (Interior_Aug), farthest from S333, had the least MAR and the least vertical differences in suspended particle accumulation. Generally, there was little variation in BD, OM, and TP content vertically within sites and overall between sites, although, almost all variables were lowest at Interior_Aug compared to the other sites, and largely due to greatly reduced sediment volumes collected. Discharge varied between 0 and $40 \text{ m}^3 \text{ s}^{-1}$ during trap tree deployments (Fig. 4b). The highest discharge correlated with the highest MAR at the Upstream_Nov site.

3.2.2. Bottom traps

Bottom traps were deployed during significantly different discharges, one high ($20\text{--}40 \text{ m}^3 \text{ s}^{-1}$) and one low ($<10 \text{ m}^3 \text{ s}^{-1}$) (Fig. S3). The bottom traps (BT 1–8) during high discharge had the highest MAR at

Table 3

Trap tree suspended particle characteristics at upstream (Upstream_Nov), downstream (Downstream_July, Downstream_Dec), and interior (Interior_Aug) stations in the L-29 Canal and at three different depths within the canal water column.

Trap position	Volume	Mass Accumulation Rate	Bulk Density	Organic Matter	Total Phosphorus
	cm^3	$\text{g cm}^{-2} \text{ d}^{-1}$	g cm^{-3}	%	mg kg^{-1}
Upstream_Nov					
Top	97.75	0.0151		35.06	905
Mid	140.36	0.0426	0.108	38.54	937
Low	192.17	0.0516	0.096	36.61	940
Downstream_July					
Top	36.50	0.0080	0.079	38.44	954
Mid	75.00	0.0148	0.071	38.47	964
Low	85.00	0.0181	0.076	37.71	1011
Downstream_Dec					
Top	82.94	0.0167	0.103	38.14	988
mid	72.73	0.0133	0.093	40.00	1025
Low	140.36	0.0276	0.100	39.01	990
Interior_Aug					
Top	6.00	0.0005	0.027		1239
Mid	3.50	0.0003	0.032		1097
Low	5.50	0.0008	0.054	34.74	887

Blank cells are either lost samples or insufficient samples for analysis.

the farthest downstream site, BT8 (Table 4, Fig. S2). During low discharge (and with additional bottom traps) there was an increase in MAR from upstream (BT2) to downstream with a peak at BT5 (Table 4, Fig. S2). Under both discharge conditions, the BDs decreased with distance while the OM content generally increased. However downstream of BT5 (BT6 and BT8), during low discharge, there was an increase in BD and a decrease in OM. The TP concentration in settled particles did not greatly vary with one exception (BT4).

3.3. Physicochemical characteristics of L-29 Canal sediments

The surface sediment layer at the Upstream was consistently, statistically different from the surface sediment layers at other sites farther downstream (Table 5). For TP, Upstream (72 mg kg^{-1}) and Downstream (1042 mg kg^{-1}) were statistically different from each other and from the other two sites, while Interior (703 mg kg^{-1}) and Eastern (559 mg kg^{-1}), farthest from S333, were not statistically different. The OM and BD were dramatically and statistically lower and higher respectively in the surface sediments at Upstream than in the surface sediments of the other three sites. Within sediment profiles, TP and OM generally decreased with depth across all sites except at Upstream, where TP and OM were

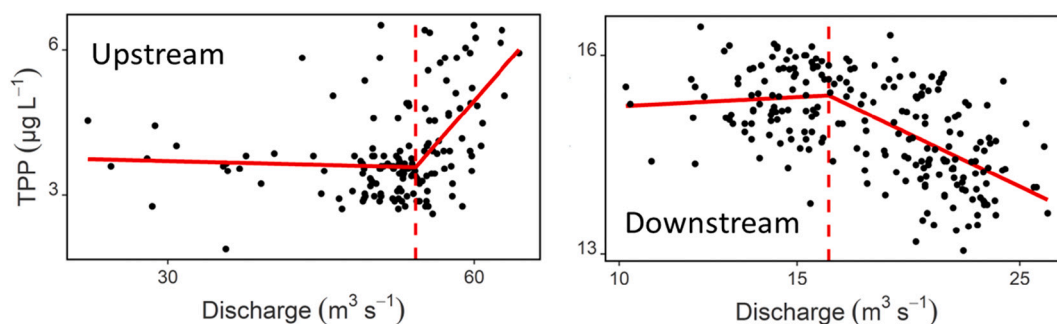


Fig. 3. C-Q relationship with slope inflections (discharge thresholds) for estimated total particulate phosphorus (TPP_{est}) at Upstream and Downstream sites in the L-29 Canal.

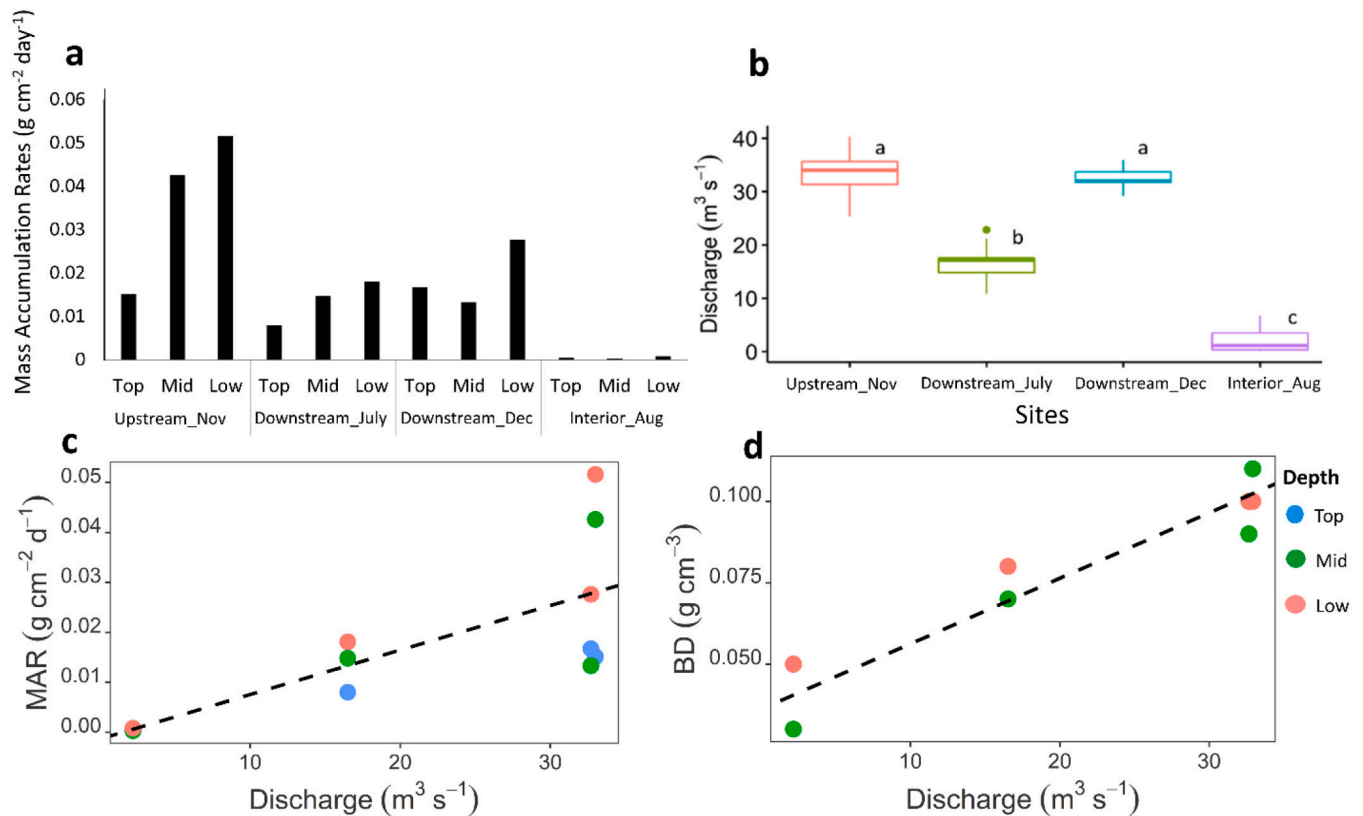


Fig. 4. a) Mass accumulation rates (MAR) of suspended particles during trap tree deployments. b) Box and whisker plot of discharge during the deployment of trap trees. Trap trees were deployed at the Downstream site during low (Downstream_July) and high (Downstream_Dec) discharges. Plots of c) mass accumulation rate (MAR) vs discharge ($r^2 = 0.50$, $p < 0.01$, $n = 12$, and d) Bulk density (BD) vs discharge ($r^2 = 0.90$, $p < 0.0001$, $n = 11$) of the trap trees. Included in these plots are the trap positions (depths): top, mid, and low. (For BD, the top values are masked by the mid).

Table 4

Settling particle characteristics and accumulation in bottom traps (BT) during high discharge (28 Oct to 11 Nov 2021) and low discharge (11 Nov to 24 Nov 2021) deployments.

Bottom traps	Approx. distance from S333 structure m	Volume cm^3	Mass accum. rates $\text{g cm}^{-2} \text{ day}^{-1}$	Bulk density g cm^{-3}	Organic matter %	Total phosphorus mg kg^{-1}
High discharge deployment ^a						
BT1	312	100.81	0.0484	0.170	19.72	723
BT6	1279	54.87	0.0098	0.060	48.20	965
BT8	2392	441.51	0.0659	0.053	65.02	655
Low discharge deployment ^b						
BT2	425	108.46	0.0371	0.113	36.26	877
BT3	547	198.30	0.0572	0.096	44.69	8789
BT4	717	354.74	0.0928	0.087	51.07	2909
BT5	916	586.98	0.1201	0.066	56.10	858
BT6	1279	172.78	0.0366	0.071	46.09	906
BT8	2392	118.67	0.0329	0.095	38.14	898

^a Only three traps were available to be deployed at that time.

^b More traps were created and deployed, but BT1 and BT7 were not recovered during retrieval.

lowest in the upper 15 cm (Table 5). The BD generally increased with depth at all sites except at Upstream, which had the highest BD at the surface. There was a positive relationship between TP and OM across all sites combined ($r^2 = 0.52$, $p < 0.0001$, Fig. S4). The greatest BD were observed in the top layers at Upstream which were characterized by lower OM and were sandier in texture (Fig. S5).

4. Discussion

4.1. Dynamics of particles and phosphorus in the water column of the Lower Everglades canals

The fate and transport of suspended particles and particulate P in the canals of the Lower Everglades is a function of discharge (Stuck et al., 2002).

During high discharge, canal water becomes enriched with TP

Table 5

Mean ($\pm$ SD) sediment characteristics with depth in collected cores. Total phosphorus (TP), organic matter (OM) and bulk density (BD) values of Upstream, Downstream, Interior, and Eastern sediments were observed in the sediment profiles ($n = 3$).

Site/Depth Increment	TP	OM	BD
cm	mg kg ⁻¹	%	g cm ⁻³
Upstream			
0–15	72 $\pm$ 46a	0.71 $\pm$ 0.20a	1.30 $\pm$ 0.06a
15–24	214 $\pm$ 208	4.89 $\pm$ 5.87	1.01 $\pm$ 0.41
24–35	320 $\pm$ 67	10.96 $\pm$ 5.67	0.72 $\pm$ 0.27
35–44	336 $\pm$ 14	12.85 $\pm$ 2.92	0.55 $\pm$ 0.16
44–56	345	10.88	0.67
Downstream			
0–11	1042 $\pm$ 134b	24.32 $\pm$ 1.19b	0.18 $\pm$ 0.01b
11–21	632 $\pm$ 144	18.77 $\pm$ 6.31	0.36 $\pm$ 0.15
21–33	372 $\pm$ 44	18.76 $\pm$ 2.92	0.41 $\pm$ 0.06
33–40	316 $\pm$ 6	19.09 $\pm$ 0.13	0.45 $\pm$ 0.09
Interior			
0–11	703 $\pm$ 87c	31.30 $\pm$ 1.37b	0.15 $\pm$ 0.01b
11–19	261 $\pm$ 83	13.87 $\pm$ 2.53	0.66 $\pm$ 0.2
19–30	138 $\pm$ 103	15.26 $\pm$ 1.22	0.52 $\pm$ 0.04
30–38	169 $\pm$ 20	16.47 $\pm$ 2.3	0.55 $\pm$ 0.06
Eastern			
0–8	559 $\pm$ 165c	19.24 $\pm$ 10.7b	0.35 $\pm$ 0.19b
8–16	539 $\pm$ 27	20.91 $\pm$ 1.58	0.3 $\pm$ 0.04
16–23	468 $\pm$ 180	15.56 $\pm$ 5.78	0.66 $\pm$ 0.29
23–32	208 $\pm$ 8	9.98 $\pm$ 2.58	0.88 $\pm$ 0.18
32–40	207 $\pm$ 35	17.14 $\pm$ 1.02	0.53 $\pm$ 0.08

* $n = 1$, ** $n = 2$ ANOVA and Tukey's test were performed on the surface cores to determine significant differences between the sites.

because of an increase in inflowing upstream particles and resuspended sediments (Onwuka et al., 2021), as was observed in three of the four canals in this study. High discharge leads to advection-driven transport of P including the entrainment of particulate P from bed sediments, and the desorption of P from suspended particles (Diamond and Cohen, 2018; Reddy et al., 1999; Withers and Jarvie, 2008) as was confirmed by our C–Q plots when TPP data was available (G136 and S5a). During low discharge, particles settle to the canal bed where they can be biogeochemically processed (Fig. 5). Chemostatic and/or the slight enrichment behavior at lower than threshold discharge suggests

biogeochemical processes (e.g., adsorption-desorption, biologic uptake, and decomposition of OM), contribute to regulating the availability of TP in the water column (Withers and Jarvie, 2008).

The effects of discharge on particle behavior varies with distance from an inflow (Daroub et al., 2007). In this study there was a strong enrichment behavior of particulate phosphorus at higher discharges near the inflow structure and an apparent dilution of TPP downstream (where the “high” discharges were below 30 m³ s⁻¹). The dilution behavior of particulate constituents, like P, in riverine systems at high discharge has been ascribed to the exhaustion of particles from processes such as their deposition in floodplains (Rose et al., 2018; Zhang, 2018; Onwuka et al., 2021). Although overbank discharge doesn't occur in the L–29 Canal, particle exhaustion may have occurred because most particles settled in upstream regions. Additionally, the L–29 Canal has a low particle load with average TSS in downstream regions around 10 mg L⁻¹ (Onwuka et al., 2023). Low TSS values like these are found in <5 % of lotic systems worldwide (Trefry and Fox, 2021). However, the TSS in the L–29 has higher P content than global averages for rivers, suggesting particle transport in this area can still have a significant environmental affect (Onwuka et al., 2023).

The lower temporal resolution monitoring data at S333 (located at the start of the L–29 Canal) did not produce a discernable C–Q pattern because the weekly sampling could not capture the variability in TP in relation to the discharge releases. The apparent lack of discharge effect on TP exports at S333 as shown in the C–Q relationship is misleading, because downstream of S333 in the L–29 Canal, the effect was evident. Since canal discharge releases in south Florida can be rapid and instantaneous, water quality estimates that match this frequency are necessary for adequate monitoring, evaluation, and environmental restoration purposes. Higher temporal resolution data (e.g., ~hourly measurements as made in the L–29 Canal) discerned the effects of discharge on P where lower resolution data did not.

The C–Q discharge thresholds separate low discharge chemostasis (biogeochemical driven) and high discharge mobilization (hydrologic driven) behavior and are at least partially dependent on physicochemical characteristics of the particles. Across the Lower Everglades, the more northern sites, proximal to the EAA (G136 and S5a), had lower TP discharge thresholds, reflecting the easily transportable, high OM, low BD, and P-rich particles characteristic of the EAA. Previous analysis in the EAA canals showed sediments with OM as high as 54 % (Das et al., 2012b), suspended particles with up to 77 % OM and BD as low as 0.08 g cm⁻³, and TP concentrations >2500 mg kg⁻¹ (Bhadha et al., 2017).

Although the southern-most L–29 Canal had higher discharge

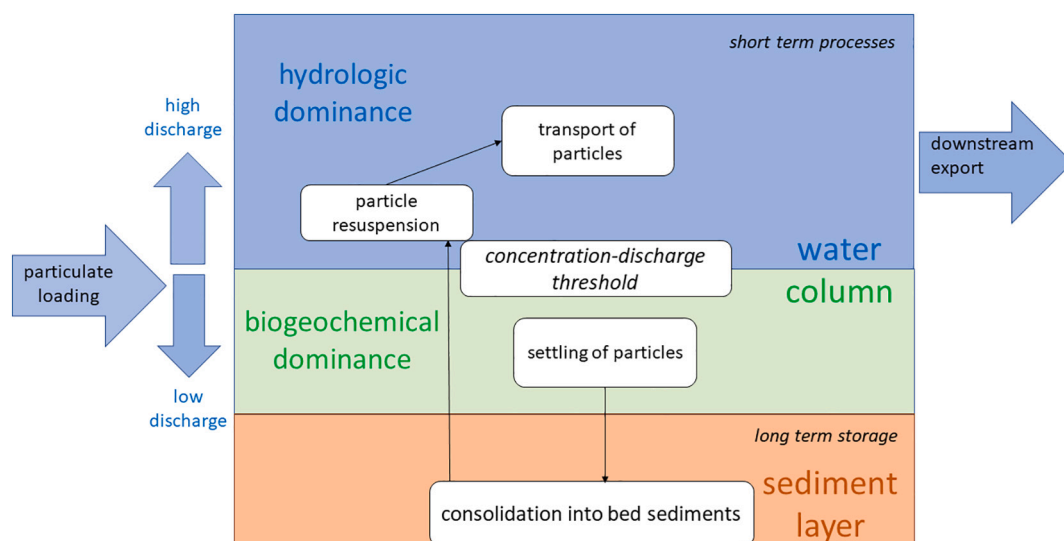


Fig. 5. An illustration of particle dynamics in the water column and sediment layer in a canal.

thresholds, suspended particles can still be a major source of P downstream in the EPA. Discharge and proximity to the inflow structure (S333) determined the quantity and characteristics of particles in the water column and that settled near the canal bed. During high canal discharges, materials originating from upstream were introduced to the L-29 Canal through the bottom-opening S333 spillway (thus creating velocities and particle discharge lower in the water column) and velocities were created that could erode and resuspend bed sediments (Stuck et al., 2002; Rose et al., 2018). During this high discharge, sediments accumulated at the lowest depths in the water column because higher density materials tend to settle out at greater depths, and because canal bed sediments can be resuspended (Wang et al., 2011; Dwino-vantyo et al., 2017). Comparing site specific differences in discharge (Downstream site) more particles accumulated in the bottom traps during high discharges than at lower discharges. The least accumulation of transported particles occurred at the farthest downstream (Interior) site that also had the lowest discharge. However, the volume of particles that settled to the canal bed often occurred farthest downstream of the inflow during high discharge compared to low discharge. These particles generally had low BD, and high OM, characteristic of light materials that do not readily settle but rather can be easily transported (like the EAA originating sediments above). This is further supported by the positive correlation between BD and discharge across all deployments and sites suggesting lighter suspended particles are easily transported at lower discharges while higher discharges are required to mobilize and transport heavier particles. Similarly, Trefry et al. (2009) found lower discharges transported less TSS with a higher OM in the Lake Worth Lagoon of South Florida. Furthermore, the particles at the top trap at Interior were green and had the highest TP content, indicating that they were perhaps rich in chlorophyll (floating algal cells; Onwuka, personal observation). These observations show that the decreasing effect of discharge in the interior parts of the canal can give rise to the domination of biogeochemical processes (O'Donnell and Hotchkiss, 2019) in determining the type of particles generated.

4.2. Bed sediment characteristics and their relationship to discharge in the L-29 Canal

Variations in discharge-driven transport and sedimentation of particles have led to distinct changes in the physicochemical characteristics of accreted sediments that vary with distance along the canal. Notable distinctions in BD, OM and TP were generally greatest within the upper approx. 24 cm of sediment profiles. Sediment profiles show a greater uniformity in physicochemical characteristics below approx. 24 cm and may identify differences in sediment accumulation that occurred with the construction and operation of the S333 structure in the mid-1970s (Light and Dineen, 1994). We suggest sediments deeper than about 24 cm accrued between the construction of the L-29-Tamiami Canal (1928) and the construction and operation of the S333 structure. At the S333 inflow structure, instantaneous releases from zero to $60 \text{ m}^3 \text{ s}^{-1}$ have been observed (Onwuka et al., 2023). Such sudden and large water releases potentially mobilized and transported light (organic) material in the surface sediments at the Upstream site further downstream and left behind the heavier (mineral) sediments. Similarly, Das et al. (2012b) found lower OM sediments in proximity to pumps in EAA farm canals and suggested this was due to the mobilization of light flocculent sediments.

Sediment management practices such as the implementation of agricultural best management practices (BMPs) and the creation of the treatment wetlands (e.g., stormwater treatment areas -STAs) have been largely successful in P export prevention and retention, leading to lower P concentrations (Wang et al., 2011). Some BMPs include sediment control initiatives that minimize the transport of sediments from farms, and sediment removal from canals. Examples include constructing canal sumps to trap sediments, reducing drainage velocities near pumps, and establishing canal cleanup endeavors (Daroub et al., 2011). Likewise,

the STAs are constructed wetlands that remove P through biological assimilation (emergent aquatic vegetation, submerged aquatic vegetation, periphyton), chemical adsorption, and settling and accretion of P in sediments (Chimney and Goforth, 2001; Das, 2010). However, studies have shown that particulate P is a dominant form of TP exported from STAs (Dierberg and Debusk, 2008), especially under wind-driven conditions (Fugate et al., 2021).

This study addresses the particulate P export in managed canals by quantifying the discharges that export P-rich particles into nutrient-sensitive downstream ecosystems including Everglades National Park. By relating canal discharges to the volume of particles that accumulate in the water column and settle in the bed to form sediments, the determination of the ranges of discharges that increase the export of particles can be identified and prevented. This is crucial for canal operational management to meet Everglades' restoration goals.

5. Conclusion

Higher discharges, above determined thresholds, increase the concentrations and potential downstream export of particles and P by importing more particulate mass and facilitating resuspension of canal sediments. Lighter organic, P-rich, particles are transported farther downstream at higher discharges leaving heavier mineral particles to settle near inflows. The effects of discharge are well described using C-Q relationship models that in many cases can be determined using long-term, low temporal resolution data, especially in systems that have large variations in both discharge and concentration. However, where variation in concentration is generally more constrained or where long-term lower temporal resolution monitoring data are not available, the use of higher temporal resolution (mins-hrs.) data over a relatively short duration (week) and over a range of discharges (characteristic of control structure operations) can help identify discharge thresholds and quantify the impacts of discharge on constituent export. Where possible, management operations should consider discharge thresholds to avoid increased downstream transport of high-concentration water.

Recognizing the effects of discharge can aid in realizing the goals of the Comprehensive Everglades Restoration Plan (CERP). Additionally, and in a larger context, excess phosphorus and nitrogen are the most extensive causes of water quality deterioration in the United States, and although nutrient load reduction efforts have mitigated some problems, their concentrations are still released in quantities that are environmentally harmful. Understanding discharge-driven concentration increases, via C-Q relationships, can aid in reducing and mitigating these effects.

CRediT authorship contribution statement

Ikechukwu S. Onwuka: Conceptualization, Methodology, Validation, Formal analysis, Investigation, Data curation, Writing – original draft, Visualization, Project administration, Funding acquisition. **Leonard J. Scinto:** Conceptualization, Methodology, Investigation, Resources, Writing – review & editing, Supervision, Project administration, Funding acquisition. **René M. Price:** Writing – review & editing, Funding acquisition. **Assefa M. Melesse:** Conceptualization, Writing – review & editing, Supervision.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data availability

Data will be made available on request.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.scitotenv.2023.166508>.

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