



Physicochemical characteristics of a southern Lake Michigan river plume

Yusuf Jameel ^{a,*}, Sarah Stein ^b, Emily Grimm ^c, Charles Roswell ^b, Alan E. Wilson ^d, Cary Troy ^c, Tomas O. Höök ^{b,e}, Gabriel J. Bowen ^a

^a Geology and Geophysics, University of Utah, Salt Lake City, UT, USA

^b Department of Forestry and Natural Resources, Purdue University, West Lafayette, IN, USA

^c Lyles School of Civil Engineering, Purdue University, West Lafayette, IN, USA

^d School of Fisheries, Aquaculture, and Aquatic Sciences, Auburn University, Auburn, AL, USA

^e Indiana Sea Grant, Purdue University, West Lafayette, IN, USA

ARTICLE INFO

Article history:

Received 27 July 2017

Accepted 9 January 2018

Available online 17 January 2018

Communicated by Jay Austin

Keywords:

Freshwater river plumes

Stable isotopes

Plume biogeochemistry

Nearshore processes

ABSTRACT

Riverine inputs are a major source of nutrients to the Laurentian Great Lakes and have important effects on nearshore biological processes, where mixing between river and lake water leads to formation of heterogeneous river plumes. We examined the physical and chemical characteristics of the St. Joseph River plume in southern Lake Michigan between May and October 2011, and in October 2012, June 2013 and April 2014. Specific electric conductivity and stable isotopes of water were used to quantify the fraction of river water (FRW) at sampling sites in Lake Michigan. Both tracers predicted similar patterns of FRW among sites; however, there was a systematic offset between the two methods, and specific electric conductivity method under-predicted the FRW by ~5%. We observed a distinct, seasonally varying river plume, with plume size correlated with flow rate of St. Joseph River. Within the plume, sediments and nutrients were non-conservative and exhibited significant and seasonally varying losses that we attribute to settling of particle-bound nutrients and/or nutrients in particulate phase below the plume. The characteristics and the spatiotemporal heterogeneity of the river plume documented here may have important implications for the nearshore biogeochemistry of the Great Lakes and for understanding the roles of these features in ecological processes in nearshore areas.

© 2018 International Association for Great Lakes Research. Published by Elsevier B.V. All rights reserved.

Introduction

Rivers and streams are a major source of nutrients and sediment to the Laurentian Great Lakes and many other large lacustrine systems. The mixing of river water with lake water results in the formation of river plumes, which are heterogeneous environments relatively enriched in nutrients and suspended sediments compared to the lake. Similar to oceanic river plumes, which are highly productive, act as a sink for sediments and pollutants, and have important ecological effects (Grimes, 2001; Grimes and Finucane, 1991; Peterson, 2003), freshwater river plumes can also be zones of high productivity and planktonic concentration (Johengen et al., 2008; Lohrenz et al., 2004; Vanderploeg et al., 2007) offering survival and growth advantages to many different fish species (Grimes and Kingsford, 1996; Jude and Pappas, 1992). Freshwater plumes have also been shown to play an important role in biogeochemical cycling of nutrients (Hecky et al., 2004; Howell et al., 2012; Morrice et al., 2004, 2009; Steinman et al., 2009).

Lacking the salinity gradients of their oceanic counterparts, freshwater plumes are generally characterized by distinct chemical and thermal properties, with potential implications for the physicochemical and biochemical processes occurring within the plumes (Stephens and Minor, 2010). Unlike oceanic plumes, which have been examined from ecological, chemical and physical perspectives in great detail, relatively few studies have examined freshwater plumes and river mouth zones within the Great Lakes (Larson et al., 2013, 2016; Marko et al., 2013; Tan et al., 2016). In their recent review, Larson et al. (2013) highlighted several critical knowledge gaps related to the understanding of these systems, including a lack of understanding of the impact of these zones on lake ecosystems.

Accurate delineation of freshwater plumes are challenging due to lack of salinity gradient between the mixing waters. Due to their conservative behavior, stable isotopes of water (SIW) and specific electric conductivity (SEC) have previously been used to quantify mixing between different freshwater sources (Klaus and McDonnell, 2013; Laudon and Slaymaker, 1997; Matsubayashi et al., 1993; McDonnell et al., 1991) and show potential promise for characterizing freshwater river plumes. In general, they are thought to provide robust and accurate estimates of the relative contribution of sources. However, few studies have used

* Corresponding author.

E-mail address: yusuf.jameel@utah.edu (Y. Jameel).

both tracers in combination or directly compared the results they produce. In many systems, river and lake water have distinct isotope ratios ($\delta^{18}\text{O}$ and $\delta^2\text{H}$) and SEC due to differences in the residence time of water and dissolved ion concentrations in these systems. Here, we use such differences to estimate fractional contributions of lake and river water at one river mouth in Lake Michigan and compare results obtained with the two tracer types to inform their future application in freshwater plume studies.

River plumes may form an active interaction zone between nutrient and sediment rich (river) and poor (lake) environments. Within plumes, riverine inputs are transformed by biogeochemical processes (Krieger et al., 1992; Morrice et al., 2004), buried by sedimentation, or remobilized (e.g., through storm-induced resuspension). These active interactions in turn affect lake biogeochemical cycles and ecosystems and, in general, are not well understood for the Great Lakes (Larson et al., 2013). To improve our understanding for one such system within Lake Michigan we determined the fate of nutrients and sediments in the St. Joseph River plume by comparing the observed concentrations with those expected assuming conservative mixing between lake and river water. We characterized nutrient transport to and through the plume environment, to identify processes that may govern the fate of nutrients delivered to the plumes and their ultimate influence on near-shore lake ecological processes.

Methods

Watershed characteristics

The St. Joseph River has a catchment area of $>12,000\text{ km}^2$ and an annual average discharge of $97\text{ m}^3/\text{s}$ (USGS streamflow-gaging station 014101500 at Niles, Michigan). The watershed is home to >1.5 million people living in 15 counties that span the watershed. St. Joseph River is the third largest river flowing into Lake Michigan. Excluding the water category, land use in the watershed, is dominated by agriculture (69%), followed by developed (20%) and forest (11%, source: 2006 National land cover database), while bedrock is composed mainly of siliciclastic rocks (shale and sandstone).

Site description and sampling protocol

The St. Joseph River – Lake Michigan system (SJS) was sampled 1–3 times per month from May 2011 to October 2011 during non-windy and calm days between 10 AM to 6 PM (see Electronic Supplementary Material (ESM) Table S1 for wind velocity and direction during sampling events). The sampling scheme for each survey included six sites, one site near the mouth of the river and five sites in the lake near the river mouth. The lake sites were symmetrically distributed; with 1 site situated directly offshore of the river mouth ($\sim 1\text{ km}$ from the river site) and 2 pairs of sites to the right and left of the river mouth (positioned $\sim 500\text{ m}$ and 1000 m alongshore from the river mouth along the $\sim 10\text{ m}$ depth contour, Fig. 1). Subsequently, the river mouth site will be referred as the “river” site and the sites in the lake will be referred as “open water” sites.

The sampling protocol at each site included measuring SEC, temperature (T) and dissolved oxygen (DO) for the entire water column. We also measured transparency with a Secchi disk at each site. In addition, depth integrated water samples were collected from the top 1 m of the water column using a rigid tube sampler, then transferred to a 5 L Nalgene bottles and immediately stored on ice. Water samples were later analyzed in the lab for their stable isotopic composition ($\delta^{18}\text{O}$ and $\delta^2\text{H}$), total phosphorus (TP), soluble reactive phosphorus (SRP), total nitrogen (TN), total suspended solids (TSS) and chlorophyll a using protocols specified below. All parameters were measured on most samples, but on a few occasions, some sites were not sampled at all or were sampled incompletely due to logistical constraints.

Targeted plume surveys

In addition to the temporal sampling described above, we also conducted extensive spatial sampling in October 2012, June 2013 and April 2014 to map the extent of the plume in detail under contrasting seasonal conditions. We sampled 22 sites in October 2012 (fall), 30 sites in June 2013 (summer) and 35 sites in April 2014 (spring). The number of sites sampled and spatial pattern of the surveys was determined by visual observation at the time of sampling to provide

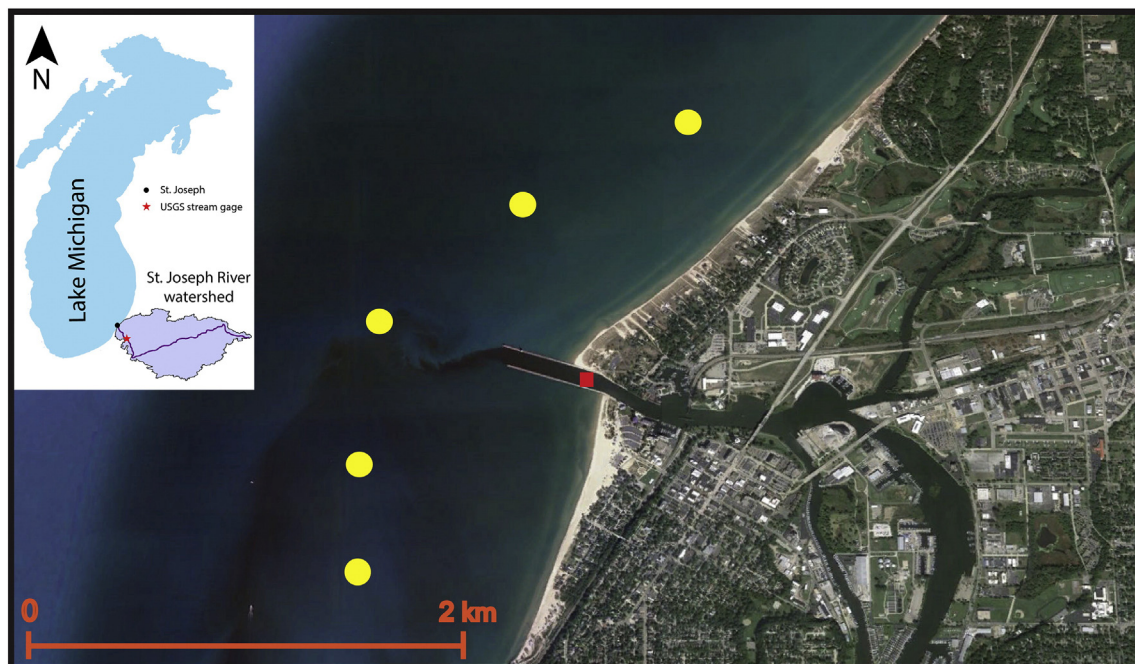


Fig. 1. St. Joseph river mouth zone, showing the location of open water (yellow circle) and river (red square) sites sampled in the lake and the river during the 2011 surveys (Map data: Google Earth). Inset: Map showing approximate location of St. Joseph River within Lake Michigan.

comprehensive sampling of the plume throughout its extent. As our aim in these spatial surveys was primarily to map the plume we only measured T, SEC, DO, and SIW and did not collect additional samples for water quality analyses. Additionally, our measurements (for T, SEC and DO) were depth specific, sampled at an interval of 0.5 m from the surface to the bottom. Water samples for isotopic analysis (i.e. SIW) were collected from two depths, surface and bottom (1 m above the lake bed), at each site.

Sample analysis

Physical parameters of water (T, SEC and DO) were measured with a YSI sonde (model 6600E V2, YSI Incorporated). Manufacturer specifications list the accuracy of ± 0.15 °C for temperature and analytical uncertainty of $\pm 2\%$ of the observed value for SEC. Water samples for isotopic analysis collected in 2011 were analyzed at the Purdue Stable Isotope (PSI) Facility, Purdue University. Aliquots of 1 μL of water were pyrolyzed using a Thermal Combustion Elemental Analyzer (TCEA); isotopic composition was determined using a Delta V Plus isotope ratio mass spectrometer (IRMS; Thermo Fisher Scientific). The samples collected during the targeted plume surveys (in and after 2012) were analyzed at the SIRFER facility, University of Utah. 1.2 μL of water was injected into a heated vaporizer and the isotopic composition of the resulting water vapor was determined using a Picarro L2130-i cavity ringdown spectrometer (CRDS). Details of the instrument set-up and data reduction are available in Good et al. (2014). Accuracy and precision for both methods were checked using secondary internal laboratory reference waters and have been described in Good et al. (2014) and Nielson and Bowen (2010). For SIW, uncertainty (1σ) was 0.30‰ and 0.7‰ ($\delta^{18}\text{O}$ and $\delta^2\text{H}$, respectively) for the samples collected in 2011, and $\pm 0.04\%$ and $\pm 0.20\%$ ($\delta^{18}\text{O}$ and $\delta^2\text{H}$, respectively) for samples collected in and after 2012.

Water samples collected in Nalgene bottles were filtered (using Whatman GF/F filters using a nominal pore size of 0.4–0.7 μM) within 24 h of sample collection. Total phosphorus (TP) was measured by persulfate digestion of unfiltered water, followed by colorimetric analysis of soluble reactive phosphorus (SRP) using long-path length spectrophotometry (Jarvie et al., 2002). Total nitrogen (TN) was measured using ultra violet spectrophotometry following persulfate digestion (Gross and Boyd, 1998), and chlorophyll a was measured by fluorometry (model 10 AU, Turner Designs, Inc., Sunnyvale, Calif.) after a 24 h dark extraction at 4 °C in 90% ethanol.

Data analysis

Calculation of fraction river water

A linear mixing model was used to calculate the fractional contribution of river water at open water sites, using the observations of SIW and SEC. The fraction river water (FRW) at each “open water site” (s) for tracer I , either SIW (using either $\delta^{18}\text{O}$ or $\delta^2\text{H}$) or SEC, was:

$$FRW_I(s) = \frac{X_I(s) - X_I(l)}{X_I(r) - X_I(l)} \quad (1)$$

where X_I is the tracer value observed for the open water site (s) or river (r) or lake (l) endmember, respectively. For each survey, the lake endmember ($FRW_I = 0$) value was defined as the lowest (for SEC) or highest (for SIW) value observed among all open water sites (ESM Table S2). The use of endmember values specific to each survey was considered necessary to isolate effects of local river input targeted here from larger spatial-scale and longer time-scale variability in near-shore water properties unrelated to the local river (Larson et al., 2013). This approach requires that our sampling design was adequate to obtain lake water unaffected by river input at one or more sites during each survey, which is supported by visual observations suggesting

that the plume rarely affected >3 open water sites for a given survey and the relatively low variance of lake water (as compared with river water) endmember tracer values. It also assumes that we were able to obtain an undiluted river sample ($FRW_I = 1$). To compare FRW estimates obtained using two tracers (I and J), we calculated pairwise differences between tracer estimates for each sample (R_{I-J}) as well as the average difference for each tracer pair across all samples (\bar{R}_{I-J}).

The uncertainty in the fraction river water estimate for each site (s) and tracer (I) was:

$$\varepsilon_I(s) = \sqrt{\left[\frac{X_I(s) - X_I(r)}{X_I(r) - X_I(l)} \varepsilon_I(l) \right]^2 + \left[\frac{X_I(s) - X_I(l)}{X_I(r) - X_I(l)} \varepsilon_I(r) \right]^2} \quad (2)$$

where $\varepsilon_I(l)$ and $\varepsilon_I(r)$ were the analytical uncertainties in the tracer values for the lake and river endmembers.

To highlight the spatial extent of the plume, we defined samples with $FRW_I > 0.15$ to be inside the plume and those with the FRW_I lower than 0.15 to be outside the plume. This threshold value, though subjective, was chosen because it corresponded well with the visual extent of suspended sediment carried by the plume during field surveys. All plume/non-plume classification and in-plume nutrient modeling used FRW calculated using $\delta^{18}\text{O}$ values (FRW_O) only. The rationale behind using $\delta^{18}\text{O}$ was the nominally non-conservative nature of SEC (discussed subsequently) within the plume and higher uncertainties associated with $\delta^2\text{H}$ measurements compared to $\delta^{18}\text{O}$. We also calculated the average FRW of all open water sites (\bar{FRW}_I) except the lake end member site for each survey date. For the temporal surveys, where a standardized sampling design was used, this value provides a measure of the strength of river water influence on the nearshore lake environment.

In-plume nutrient dynamics

For each open water site (s), we calculated the concentrations ($\hat{C}_T(s)$) of the measured water quality and nutrient parameters (TSS, TP, TN and chlorophyll a) that would be expected assuming conservative mixing of the respective parameter between lake and river water. This was done by weighting the observed end member values for the parameter according to the (conservative) mixing ratio determined by the relative $\delta^{18}\text{O}$ concentrations for the site ($FRW_O(s)$; Eq. (1)):

$$\hat{C}_T(s) = C_T(l)(1 - FRW_O(s)) + C_T(r)(FRW_O(s)), \quad (3)$$

where $C_T(l)$ and $C_T(r)$ are the nutrient or water quality concentrations for the lake and river endmembers, respectively. For each survey, the river endmember value was obtained from the measurement at the river site. Lake endmember values were estimated as the daily average of values for all open-water samples with $FRW_O \leq 0.05$, allowing us to average nutrient variability sometimes observed at multiple sites with estimated FRW_O not significantly different from zero.

We also calculated seasonal loads of TP, TN and TSS delivered to the Lake Michigan from May 2011 to October 2011 defined as the sum of the product of monthly average flow rate and monthly average concentration of each metric (TP, TN and TSS) for the river site.

During most of the surveys, observed sample concentrations, $C_T(s)$, were lower than $\hat{C}_T(s)$, suggesting loss of dissolved or suspended material within the plume. To model the pattern and quantify the fraction of loss across the river to open-lake gradient, we fit a regression model (RM) to the relationship between C_T values for all open-water samples in a given survey and FRW_O (Fig. 2A). We evaluated several model forms and chose to use an exponential model, which consistently had the lowest Akaike Information Criterion (AIC). Models comparisons

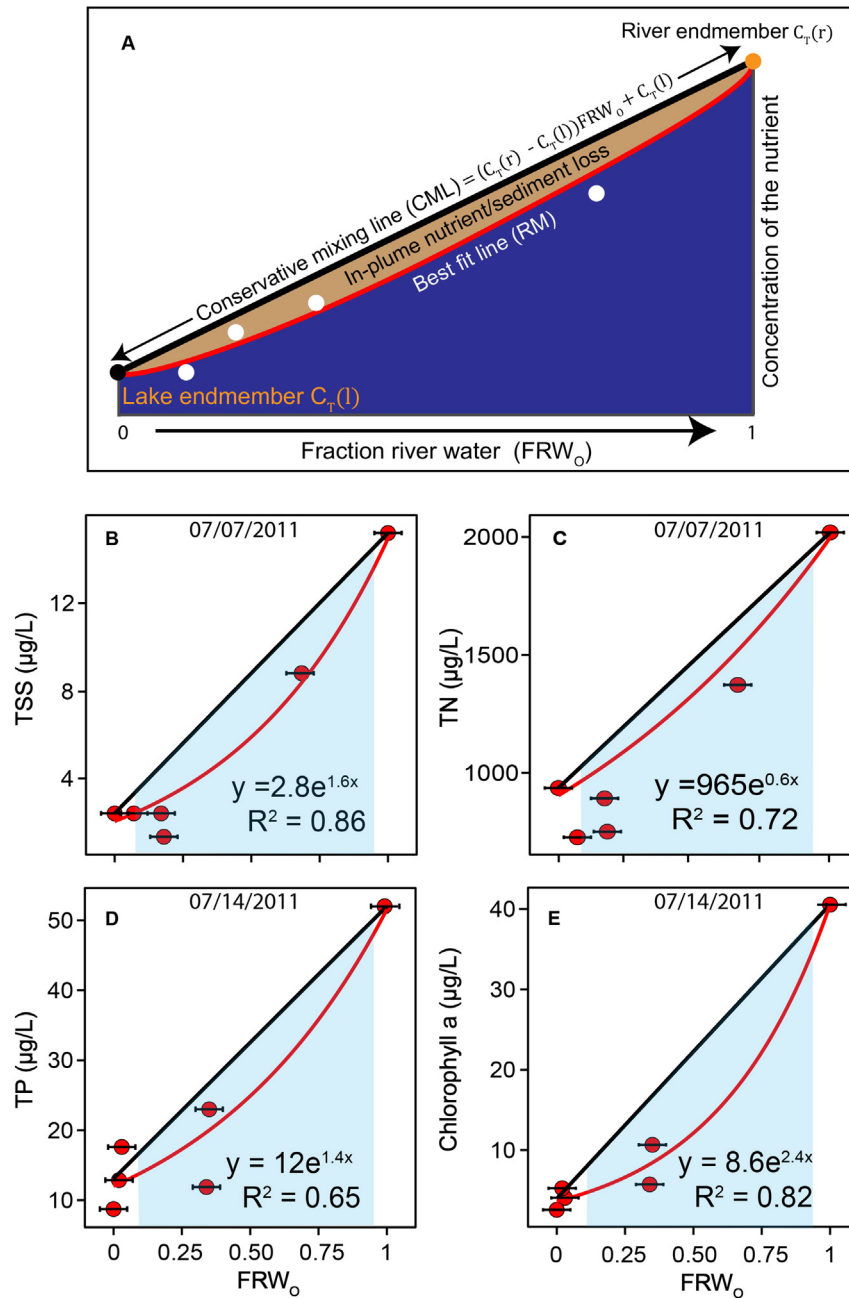


Fig. 2. (A) Schematic illustrating our approach to diagnosing and quantifying non-conservative losses of nutrients/sediments within the river plume. The black line joining lake and river endmember (shown by black and orange circle respectively) represents the conservative mixing line (CML) between lake and river endmember. White circles are hypothetical open lake observations and the red curve represents the best fit line (RM in Eq. (4)) to the observations. The area between RM and CML, highlighted in brown gives the in-plume sediment and nutrient loss. (B-E) Example FRW_0 – nutrients/sediment relationships observed at SJS for selected surveys; the black line and red curve represents the CML and the best fit line (RM) to the observations. The best fit line is constrained to pass through the lake and river endmembers. Shaded light blue region represents area within the plume ($0.15 < FRW_0 < 1$). The horizontal error bars in 2B–D are the uncertainties associated with FRW_0 estimates.

(ESM Table S3) and the rationale behind choosing the exponential model are discussed before ESM Table S3 in supplementary material. We constrained the regression model through lake and river endmember values. This approach, provided a realistic and conservative relationship between non-plume, in-plume and river sites, which was necessary considering our limited open water sampling (3–5 per survey) and the occasional large variability observed, in open water concentration of sediment and nutrients (for example see Fig. 2C). We then estimated the total mass of observed constituent lost in the plume (N_T) by integrating the difference between the conservative

mixing line for each constituent (CML, see Eq. (4) and Fig. 2A) and their respective RM curve over the river to open-lake gradient:

$$N_T = Q * \int_{FRW_0=0}^{FRW_0=1} (CML - RM) dFRW_0, \quad (4)$$

Where Q is river discharge measured at the USGS gage at Niles, MI. CML is defined as

$$CML = (C_T(r) - C_T(l))FRW_0 + C_T(l) \quad (5)$$

All statistical calculations were performed using R statistical software (R Core Team, 2015).

Results

River endmember SIW and SEC values were significantly lower and higher, respectively, than lake endmember values (paired t-test, $p < .05$; Fig. 3, ESM Table S2). For all surveys, the difference between endmembers was $>1.3\text{‰}$, 3.9‰ and $247 \mu\text{S/cm}$ for $\delta^{18}\text{O}$, $\delta^2\text{H}$ and SEC respectively.

Mixing calculations excluding lake endmember sites showed that approximately 65% of open water sites had $FRW_O < 0.15$, 25% had FRW_O between 0.15 and 0.50 and 10% had $FRW_O > 0.50$. Most of the locations with $FRW_O > 0.50$ were sampled in spring 2014. In 2011, only 30% of the open water sites had $FRW_O > 0.15$. The highest FRW_O values were observed in June, with the maximum value occurring during our second sampling event on 16 June 2011, and values declined progressively through the end of sampling in October (Fig. 4A). We observed a strong positive correlation between St. Joseph River flow rate and FRW_O across all 2011 surveys ($R^2 = 0.31$, $p < .05$, $n = 9$, Fig. 4B). A similar trend was observed for the spatial surveys (fall (October 2012), summer (June 2013) and spring (April 2014)), with the highest FRW_O in spring 2014 and lowest in fall 2012. We observed a large, northward trending, buoyant plume during summer 2013 and spring 2014 surveys but did not observe any significant surficial river plume in fall 2012 (Fig. 5). In fall 2012, most bottom water samples were also classified as non-plume; however, at the site closest to the river mouth the bottom water sample FRW was much higher than the surface sample ($FRW_{O(bottom)} = 0.38$, $FRW_{O(surface)} = 0.14$).

Each of the conservative tracers gave similar estimates of FRW for each site and survey (Fig. 6). Estimates based on $\delta^{18}\text{O}$ and $\delta^2\text{H}$ were most strongly correlated ($r^2 = 0.94$, $p < .05$). SEC and $\delta^2\text{H}$ gave the most dissimilar results but were still strongly correlated ($r^2 = 0.90$, $p < .05$). For the majority of the samples, FRW_O and FRW_H values were within 4% of each other and the mean pairwise difference for FRW estimated using $\delta^{18}\text{O}$ and $\delta^2\text{H}$ (\bar{R}_{O-H}), was -0.001 (ESM

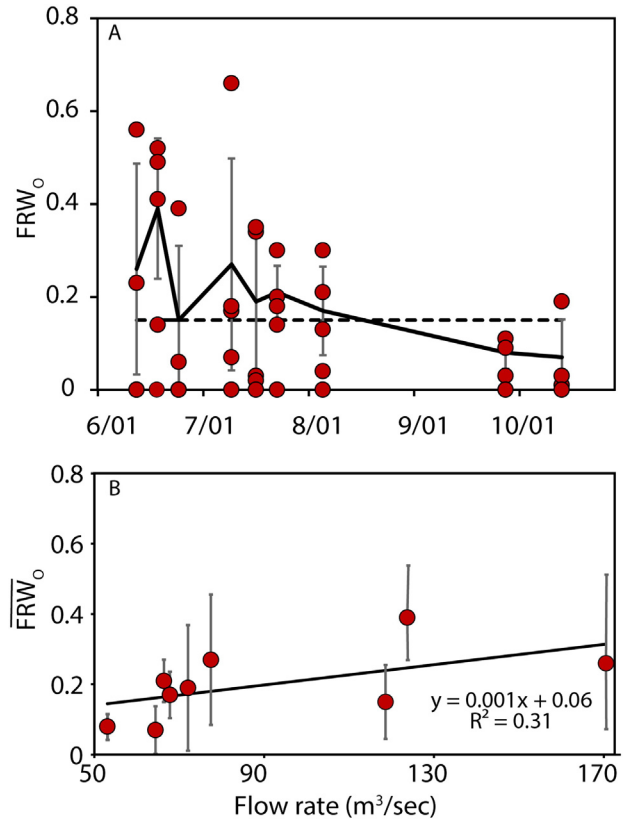


Fig. 4. (A) Temporal trend of fraction river water (FRW) observed at the lake sites. The red circle represents the FRW calculated for each open water site during the different surveys. The black line and the error bar represent FRW and the associated standard deviation (1σ) for each survey. The black dashed line presents the in-plume cutoff value of $FRW_O = 0.15$. The x-axis represents the month/date of the year 2011. (B) Correlation between FRW and flow rate for the 2011 surveys. The error bars represent the associated standard deviation (1σ).

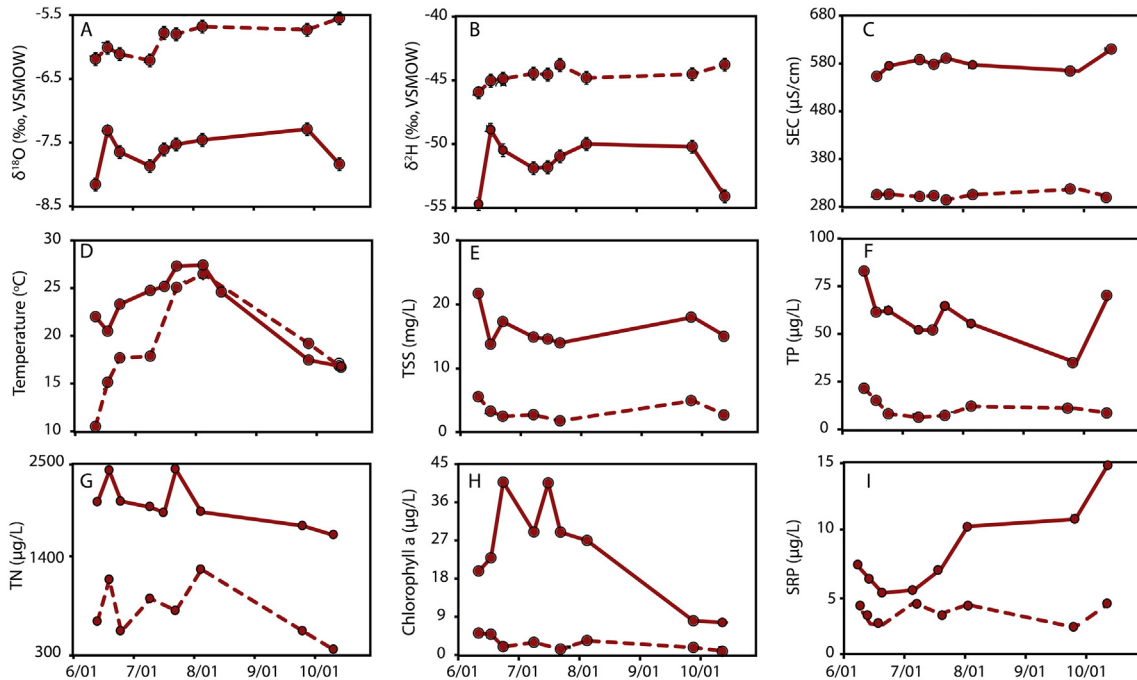


Fig. 3. Differences in physicochemical metrics between the lake (dashed lines) and river (solid lines) endmembers. The x-axis represents the month and day of the year 2011.

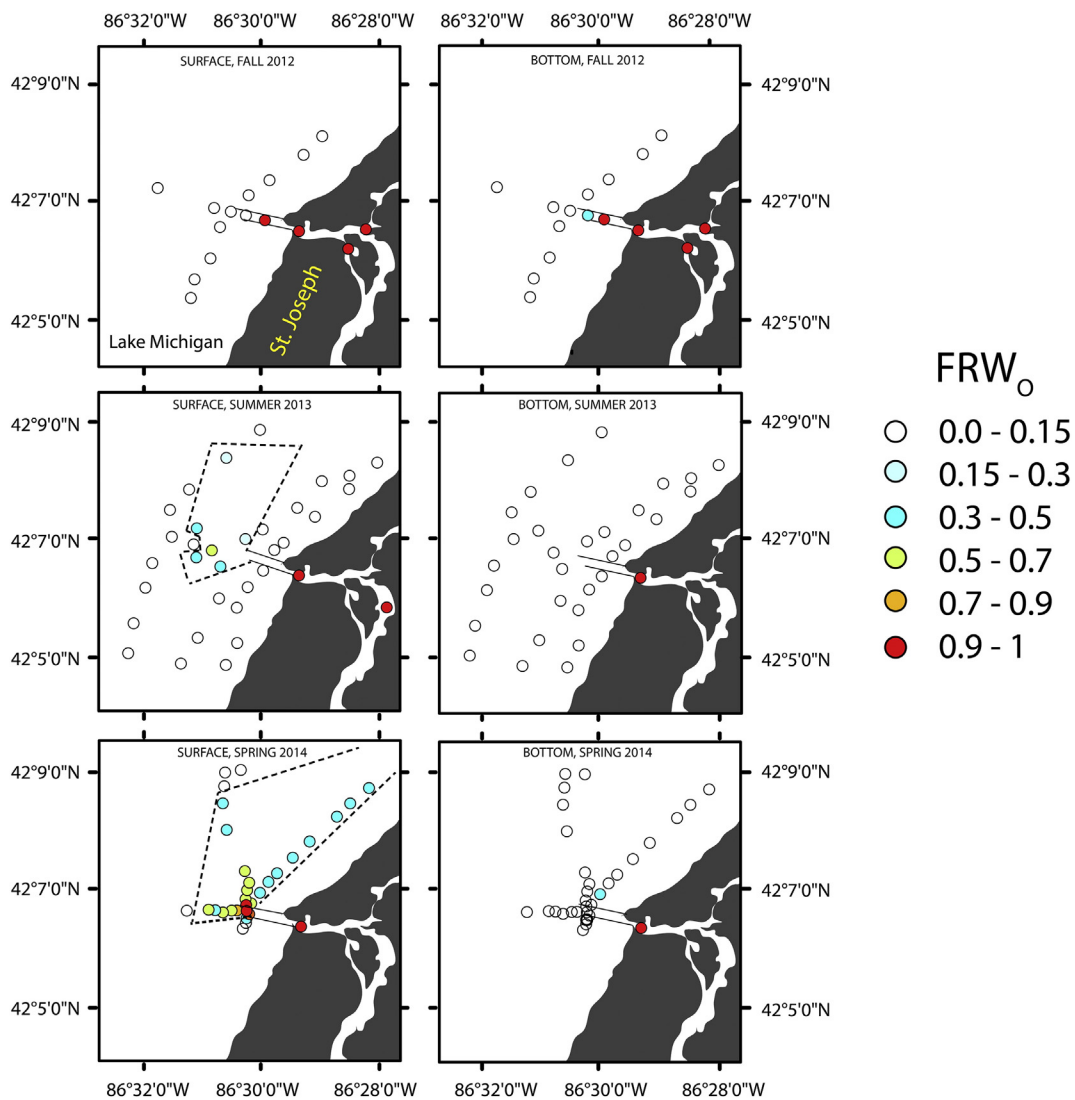


Fig. 5. FRW_O observed at the lake sites for October 2012 (fall), June 2013 (summer) and April 2014 (spring) spatial surveys. The black dashed line encloses sites with $FRW_O > 0.15$ and delineates the approximate plume extent.

Fig. S1). Data from some of the 2011 surveys show differences $>5\%$, but even in these cases the data are arrayed along the 1:1 line (Fig. 6B) and R_{O-H} values show little evidence for systematic bias. However, despite an overall similarity in pattern, systematic offsets between values estimated using SIW and SEC do exist. FRW_{SEC} values were significantly lower than FRW_O and FRW_H (paired t-test, $p < .05$) and significantly different from 1:1 line (ANOVA, $p < .05$). Pairwise (mean \pm SD) difference between $FRW_O - FRW_{SEC}$ was $0.04 (\pm 0.07)$, ESM Fig. S1).

For physicochemical and nutrient measurements, the river site was significantly different than non-plume ($FRW_O < 0.15$) and plume ($0.15 < FRW_O < 1$) open water sites (paired t-test, $p < .05$, Table 2). Significant differences were also evident between non-plume and plume sites except for T, SRP and DO (paired t-test, $p < .05$, Table 2). We did not observe any temporal pattern in the river and open water physicochemical and nutrient parameters except for average water temperature (\bar{T}) differences between the river and open water sites. During spring and early summer, open water \bar{T} was much lower than river \bar{T} ; this contrast decreased progressively throughout the summer sampling campaigns, and eventually open water \bar{T} was comparable to (October 2011) or higher (October 2012) than river \bar{T} in fall (Table 1).

In-plume samples, in general, exhibited higher concentrations of nutrients and TSS than the lake endmember; however, for many surveys these concentrations were lower than the calculated mixing values assuming conservative mixing (Fig. 2B-E) suggesting in-plume nutrient and sediment loss.

Discussion

Spatiotemporal patterns in the plume

The river plume was clearly larger and more distinct in spring and summer than fall as evidenced by declining FRW values later in the year. During summer 2013 and spring 2014, the plume was thin and surface trapped, with an average depth of 2.2 m and 4 m respectively (ESM Fig. S2). The river water temperature far exceeded that of the lake, supporting substantial plume buoyancy, and no river water was observed at the open water bottom sites (Fig. 5). For both surveys the plume spread northwards, following the dominant large-scale cyclonic circulation of the lake (Beletsky et al., 1999).

The presence of higher FRW at bottom site closest to the river mouth in fall 2012 (Fig. 5) suggests a downwelling plume. Indeed, river water

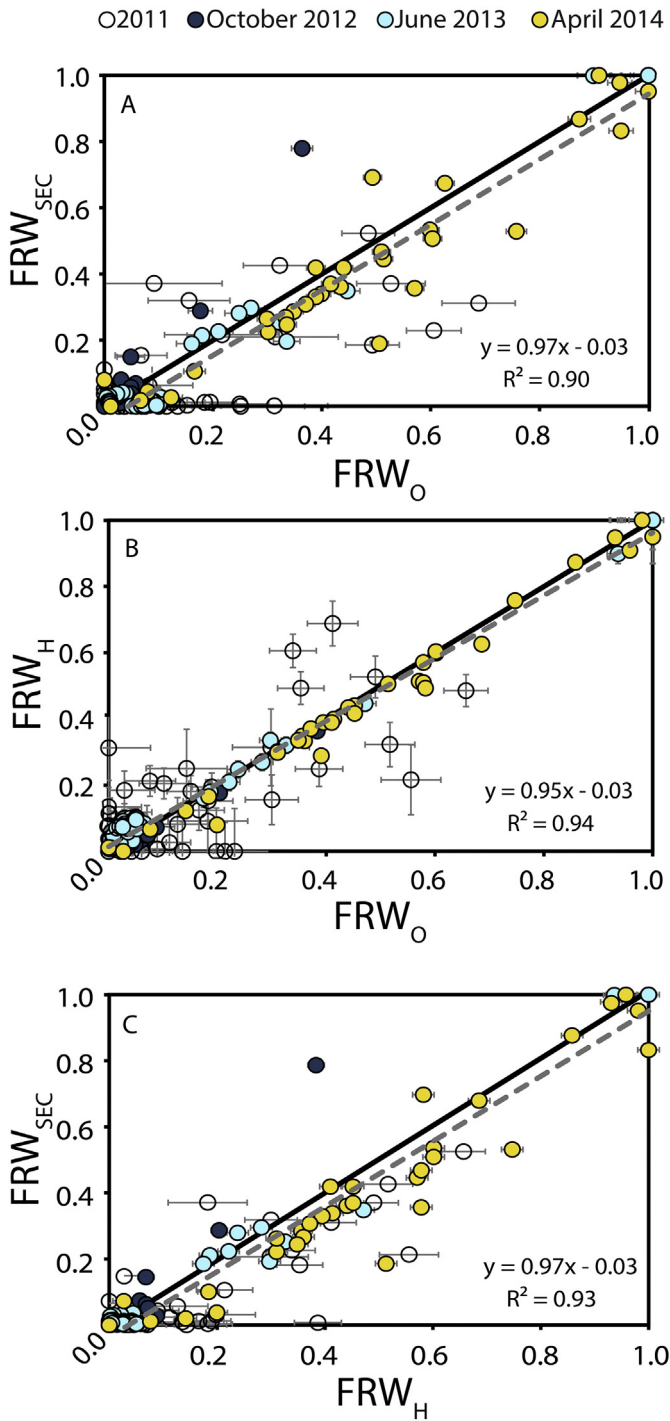


Fig. 6. Comparison of FRW values calculated using different tracers; A: SEC and $\delta^{18}\text{O}$, B: $\delta^2\text{H}$ and $\delta^{18}\text{O}$ and C: SEC and $\delta^2\text{H}$. The black and dashed gray lines represent the 1:1 and best fit relationships, respectively. The error bars represent the uncertainty associated with the calculation of FRW (Eq. (2)). Uncertainty values for FRW_{SEC} were <0.03 and are not shown. The regressions are statistically significant ($p < .005$) and significantly different from 1:1 line (ANOVA, $p < .001$ for EC and $\delta^{18}\text{O}$, and for SEC and $\delta^2\text{H}$ and $p = .02$ for $\delta^2\text{H}$ and $\delta^{18}\text{O}$).

during fall 2012 was 2 °C colder than the lake water, consistent with a negatively buoyant plume leading to sinking of river water as it impinges on the lake. The relatively low flow rate of the St. Joseph River, in conjunction with turbulent mixing within the sinking plume, likely contributed to the limited spatial extent of plume during this survey. Thus, both flow and density may affect the extent and nature of plume environment throughout the seasonal cycle.

Comparison of FRW calculated using different tracers

The systematic offset observed in FRW calculated using SIW and SEC as observed here (Fig. 6) is not unprecedented (Klaus and McDonnell, 2013; Ladouche et al., 2001; Laudon and Slaymaker, 1997; Pellerin et al., 2008). The observed pattern suggests that one or both the tracer groups are exhibiting non-conservative behavior. The largest differences are observed in 2011 (Fig. 6), when sparse spatial sampling may have limited the accuracy of our lake endmember estimates. However, since the same approach was used to estimate endmember values for both tracer families this cannot explain the systematic offset between FRW estimates. The observation of systematic FRW differences for the more spatial extensive surveys also suggests that this result is not an artifact of sampling design.

Evaporation of water within the plume could render water isotopes non-conservative. However, it is unlikely that evaporation has a significant effect on our SIW-based mixing estimates because the residence time of water in river plumes of southern Lake Michigan is on the order of a few hours (Grimm, 2013), which would be insufficient to allow measurable isotopic enrichment of plume water even at relatively high evaporation rates. In actuality, evaporation from the Great Lakes is minimal during the summer season, when most of our sampling occurred, due to relatively high over-lake humidity (Croley, 1989; Magnuson et al., 1997; Rasmussen, 1967, 1968; Spence et al., 2011). Further, even if evaporation did affect plume water SIW values, the impact on our mixing models would be an underestimation of FRW, an effect opposite of the observed discrepancy between SIW and SEC-based estimates.

Alternatively, a low bias in FRW_{SEC} values could indicate non-conservative behavior of SEC within the plumes. In this system we expect conductivity to be dominated by dissolved ions and charged suspended particles (Miller et al., 1988). SEC could be changed by electrochemical processes such as complexation, protonation, adsorption and flocculation (Miller et al., 1988; Stumm and Morgan, 1981), which are common when waters with different properties blend. In marine plumes and estuaries, non-conservative behavior and removal (or addition) of suspended sediments, nutrients, and metals is typical (Ayukai and Wolanski, 1997; Bainbridge et al., 2012; Fox et al., 2004; Milligan et al., 2007; Ndung'u et al., 2003; van den Berg et al., 1987). Interestingly, within the St. Joseph River plume, we observed non-conservative behavior of nutrients and suspended sediments (discussed subsequently), suggesting possible links between nutrients, suspended sediments, dissolved ions and SEC.

In the absence of salinity gradients, significant thermal and chemical gradients within Lake Michigan plumes could cause reactive interactions among nutrients, sediments, and dissolved ions in the river and the lake water (Larson et al., 2013). Several studies have shown that organic matter in rivers forms stable complexes with dissolved ions and can stabilize suspended sediments through the formation of organic particle coatings that prevent aggregation and coagulation (Beckett, 2013; Benjamin and Lawler, 2013; Gibbs, 1983). As productive river water (higher organic matter, high TP and TN) mixes with chemically distinct and less productive nearshore lake water (lower organic matter, low TP and TN) these complexes may be destabilized and broken down, ultimately leading to within-plume deposition of ions and sediment that contributed to SEC. Although this specific mechanism represents a hypothesis to be tested, previous work has documented particulate sedimentation and lowering of SEC in river mouth zones of the Great Lakes, supporting the pattern observed here (Baker, 2011; Bell, 1978).

Nutrient and sediment processing

Our 2011 water quality data suggest that the SJS delivered ~600 and 20,000 kg of phosphorus and nitrogen respectively per day in summer (June) and show little structured variation in nutrient concentrations over time. The seasonal load from June 2011 to October 2011 was 80

Table 1
Average river ($FRW_0 = 1$), plume ($0.15 < FRW_0 < 1$) and non-plume ($FRW_0 < 0.15$) values of the physiochemical and nutrient parameters for the different surveys. N: number of sites, T: Temperature, TSS: Total Suspended Sediments, TP: Total Phosphorus, TN: Total Nitrogen, CHL: chlorophyll a, SRP: Soluble Reactive Phosphorus and DO: Dissolved Oxygen. Temperature is calculated as average of top 1 m of the water column. Dashed lines represent no data in the respective date and/or site. The detection limits of the different parameters are as follows: TP – 3 $\mu\text{g/L}$, TN – 40 $\mu\text{g/L}$, TSS – 0.2 mg/L, SRP – 0.3 $\mu\text{g/L}$ and chlorophyll a – 0.1 $\mu\text{g/L}$.

Date	Site	N	T (Celsius)	Secchi disk (meters)	TSS (mg/L)	TP ($\mu\text{g/L}$)	TN ($\mu\text{g/L}$)	CHL ($\mu\text{g/L}$)	SRP ($\mu\text{g/L}$)	DO (mg/L)
6/10/11	River	1	–	0.6	21.7	82.9	2077	19.8	6.6	–
	Plume	2	–	1.0	11.8	43.0	1002	11.3	2.4	–
	Non-plume	2	–	6.4	1.5	25.0	664	5.6	3.2	–
6/16/11	River	1	20.5	0.6	13.8	61.4	2448	22.9	5.5	9.1
	Plume	3	16.0	1.5	6.4	28.5	1279	10.1	4.7	9.8
	Non-plume	2	14.7	4.6	2.5	17.4	959	4.8	2.1	10.5
6/22/11	River	1	23.3	0.8	17.3	62.2	2085	40.7	4.5	8.9
	Plume	1	18.3	1.5	6.6	25.6	1159	12.4	2.7	9.9
	Non-plume	3	18.0	2.9	2.6	11.2	698	2.9	2.3	9.9
7/7/11	River	1	24.7	0.8	14.9	52.0	2019	29.0	4.7	9.2
	Plume	3	19.2	0.8	5.6	19.6	1062	12.5	1.6	9.4
	Non-plume	2	18.0	3.0	3.3	6.1	831	3.3	5.3	9.8
7/14/11	River	1	25.2	–	14.6	52.0	1952	40.5	–	9.3
	Plume	2	13.4	–	7.0	19.3	1345	9.0	3.9	10.1
	Non-plume	3	11.4	5.9	5.9	12.8	1335	4.4	3.8	10.4
7/20/11	River	1	27.3	0.9	14.0	64.6	2464	28.9	6.4	–
	Plume	3	24.8	2.9	3.2	18.0	1500	5.2	4.7	–
	Non-plume	2	25.6	8.5	2.0	6.5	840	1.3	2.7	–
8/2/11	River	1	27.4	1.0	–	55.4	1960	27.0	9.9	8.4
	Plume	2	25.0	2.5	–	18.8	1839	6.6	4.5	8.7
	Non-plume	3	26.2	5.8	–	17.6	1141	2.6	5.8	8.4
9/23/11	River	1	17.5	–	17.5	34.9	1798	8.0	10.5	9.4
	Plume	0	–	–	–	–	–	–	–	–
	Non-plume	4	19.2	–	3.4	9.1	565	2.0	1.7	9.2
10/9/11	River	1	16.8	–	15.0	70.1	1685	7.6	14.9	9.3
	Plume	1	16.4	–	2.9	11.7	1096	2.0	3.6	9.8
	Non-plume	3	16.7	–	3.1	9.0	478	1.5	3.1	9.6

metric tons (MT) of phosphorous, 3600 MT of nitrogen, and 26,000 MT of suspended sediment. This phosphorus load, for example, equals approximately 10% of the total phosphorus delivered to Lake Michigan in 2005 (Mida et al., 2010), suggesting that the SJS supports significant fluxes in the biogeochemical budget of Lake Michigan.

In-plume water quality and nutrient concentrations were lower than theoretically calculated conservative mixing values (Fig. 2), suggesting non-conservative losses. We observed similar patterns for TP, TN, TSS and chlorophyll a within the plume. Strong correlations between TSS and nutrients (Fig. 7) in our data suggest a likely link to particle-phase losses. Physical settling offers a parsimonious explanation for the observed pattern and is expected due to a reduction of turbulence and particle carrying capacity in the plume (Dagg et al., 2004; Kineke et al., 2000) and coagulation, aggregation, and settling as discussed above (Dagg et al., 2004; Lohrenz et al., 1999; Miller et al., 1988; Stumm and Morgan, 1981). This explanation is also consistent with in-plume non-conservative behavior of SEC. Although in-plume biological activity particularly photosynthesis, might play a role in the observed patterns, we did not observe any pattern in SRP and chlorophyll a suggesting increased in-plume photosynthesis (Table 1 and Fig. S3). As calculated by integration of difference between CML and RM line, average in-plume fractional loss was 25% for TSS, 27% for TP and 25% for TN (Table 3). This is similar to the values reported in (Marko et al.,

2013) for Muskegon River system, where 33% of TP was intercepted in the river mouth terminus.

We observed a weak seasonal trend in TSS fractional loss, which was higher during late summer and fall (Table 3). We attribute this pattern to thermally buoyant, spring and early summer plumes, which dissipate energy over larger area and longer time period leading to gradual and slow initial in-plume deposition (Wright and Nittrouer, 1995). This has been observed in many large buoyant river plumes such as the Mississippi, Amazon and Columbia (Wright and Nittrouer, 1995). In contrast, thermally neutral or hyperpycnal (bottom dwelling) plumes, (observed in late summer and fall) mix rapidly with the lake, leading to abrupt dissipation of energy and higher sedimentation (Wright et al., 1990; Wright and Nittrouer, 1995). High sediment deposition on the continental slope has been reported for many marine hyperpycnal plumes such as the Huanghe River (Kineke et al., 2000; Warrick and Milliman, 2003; Wright et al., 1990; Wright and Nittrouer, 1995).

Given the limited duration and extent of our sampling, the degree to which our estimates are representative of the full range of variability in the SJS plume system in general is uncertain. However, the fact that in most surveys we observed in-plume concentrations of nutrients and suspended sediments that were lower than expected for conservative mixing suggests that these freshwater plumes, like their oceanic counterparts (Dagg et al., 2004; Lohrenz et al., 2004, 1999), are zones of substantial nutrient and sediment loss.

Table 2
Mean of the difference between river, plume and non-plume sites for the different parameters. Bold values are significant at $p < .05$.

Paired t-test (Mean of the difference)		T (Celsius)	Secchi disk (meters)	TSS (mg/L)	TP ($\mu\text{g/L}$)	TN ($\mu\text{g/L}$)	CHL ($\mu\text{g/L}$)	SRP ($\mu\text{g/L}$)	DO (mg/L)
River-Plume	4.6	–0.9	9.7	37.5	800	18.3	4.0	–0.6	
River-Non-plume	4.1	–4.3	13.1	46.8	1219	21.7	4.6	–0.6	
Plume-Non-plume	0.3	–3.5	3.2	9.8	416	5.3	0.0	–0.1	

Table 3

In-plume fractional loss (f) of TSS, TP, TN and chlorophyll a . Due to insufficient data, TSS and TN loss was not calculated for some of the surveys (shown by dashed lines for the corresponding dates). Statistics shown include mean and standard deviation (SD) for all the surveys. For parameters, where RSS associated with the best-fit model was greater than that associated with the conservative mixing line (see ESM Table S2), we did not calculate the fractional loss in the plume (shown as NC: Not calculated). All values are in per cent.

Survey date	TSS	TP	TN	Chlorophyll a
6/16/2011	21	28	38	28
6/22/2011	10	NC	NC	NC
7/7/2011	32	34	27	34
7/14/2011	23	28	–	38
7/20/2011	35	32	20	NC
8/2/2011	–	26	–	39
9/23/2011	31	25	20	24
10/9/2011	28	33	NC	31
Mean	25	27	25	32
SD	7	6	7	5

Lotic-lentic relationship and the SJS plume

River mouth zones play an important role in lake biogeochemistry and ecology, capturing riverine inputs, reprocessing and redistributing them within the lake (Larson et al., 2013, 2016). Our study suggests some important links between riverine inputs and lake biogeochemistry/ecology. Our observations for SJS cannot necessarily be generalized to other plumes within the Great Lakes, but they do highlight specific processes and effects that may structure the biogeochemical and ecological impacts of some plume systems.

First, our analysis suggests that 20–30% of TP loading to the Lake during summer and fall is deposited below the plume. Given the abundance of dreissenids in river mouth zones (Larson et al., 2013) it seems likely that some fraction of this flux could contribute to the growth of dreissenids. Our data suggest that particulate P is the dominant phase of P delivery in the SJS system, with SRP constituting <15% of the total P flux. Within-plume reprocessing of particulate phosphorus into bioavailable phosphorus by dreissenids, then, could be an important driver of the growth of benthic algae (e.g., *Cladophora*) (Dayton et al., 2014; Hecky et al., 2004), and link between lotic and lentic nutrient cycling (Fahnenstiel et al., 1995a, 1995b; Ozersky et al., 2009; Wilson et al., 2006).

Second, freshwater river plumes and storm induced resuspension plumes are intrinsically different in their origin and spatial coverage, however, similar to our river plume; strong coupling between TP and TSS and TP consisting dominantly of particulate P have been reported for southern Lake Michigan resuspension plumes (Vanderploeg et al., 2007). Further, TP concentration around St. Joseph River mouth during

the 1998–2000 episodic plume studies was lower but comparable to the TP concentration observed in our river plumes (Vanderploeg et al., 2007). These associations between sediments and nutrients for both plume types suggest that sediments and nutrients deposited by the river plumes might be an important source of the storm induced resuspension plumes.

Third, our work provides baseline information on the character of an open water river plume that is relevant to understanding its potential effects on nearshore ecological processes. In oceanic river plumes and estuaries, warm nutrient and sediment rich river discharge may lead to high densities of larval fish and favor relatively high early life growth and survival, thereby providing recruitment advantage for young fish (Grimes, 2001; Grimes and Finucane, 1991; Grimes and Kingsford, 1996; Reichert et al., 2010). Within the Great Lakes, the Maumee River plume has been shown to improve recruitment by reducing larval predation mortality (Reichert et al., 2010). In Lake Michigan, storm induced resuspension plumes have shown positive effects on bacterial respiration and planktonic production (Johengen et al., 2008; Lohrenz et al., 2004); however, the effects of river plumes on fish in southern Lake Michigan is understudied. In this study, we show that the St. Joseph River plume provides a physically and chemically distinct zone that could be beneficial for juvenile and young fish, for example through higher early-season water temperatures or suspended sediments relative to the broader nearshore habitat. However, due to the relatively small size and short residence time of this plume its impact on young fish may be limited. Our data suggests that any positive impacts are likely to be locally significant primarily during spring and early summer.

Conclusions

Our spatiotemporal surveys characterize a freshwater river plume within Lake Michigan, highlighting the heterogeneity within this plume. The plume is influenced by riverine and nearshore lake properties, is highly seasonal, and encompasses a region of active deposition of nutrients and sediment emanating from the river. We observed numerous similarities between this freshwater plume and its oceanic analog, such as the strong influence of flowrates on plume properties and in-plume nutrients and sediments loss. Our research shows that St. Joseph River plume is distinct and an active ‘connector’ between lotic and lentic systems, hosting processes which may modulate the impact of river inputs on nearshore biogeochemistry and ecology. Our work also compared and evaluated two groups of tracers for quantifying mixing within freshwater plumes (SEC and SIW), and demonstrated inconsistencies between them. We suggest that in our study system the same processes that led to loss of nutrients and sediment within the plume led to non-conservative behavior of SEC. More generally our results highlight the importance of critically evaluating potential in-plume effects on tracers used in freshwater plume mixing studies.

Acknowledgements

We thank the volunteers who assisted with the sampling and Michael Chislock for water quality analyses. We would also like to thank an anonymous reviewer and Anett Trebitz for their feedback and review. This work was supported by Great Lakes Fishery Trust grant 1152.

Appendix A. Supplementary material

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.jglr.2018.01.003>.

References

Yasukai, T., Wolanski, E., 1997. Importance of biologically mediated removal of fine sediments from the Fly River plume, Papua New Guinea. *Estuar. Coast. Shelf Sci.* 44, 629–639.

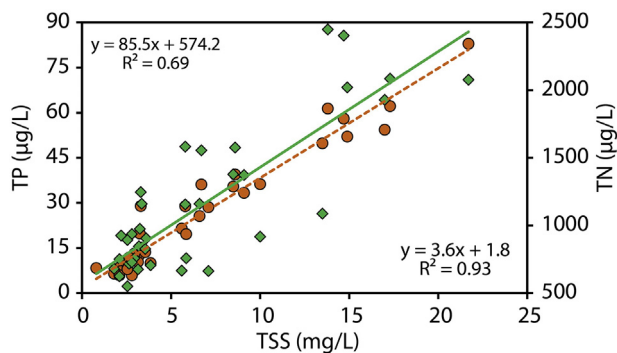


Fig. 7. Correlation between TSS and TP and between TSS and TN. The orange circle and the green diamond represent the relation between TSS and TP (dashed line, $R^2 = 0.93$, $p < .05$) and between TSS and TN (solid line, $R^2 = 0.69$, $p < .05$) respectively.

- Bainbridge, Z.T., Wolanski, E., Álvarez-Romero, J.G., Lewis, S.E., Brodie, J.E., 2012. Fine sediment and nutrient dynamics related to particle size and floc formation in a Burdekin River flood plume, Australia. *Mar. Pollut. Bull.* 65, 236–248.
- Baker, D., 2011. The sources and transport of bioavailable phosphorus to Lake Erie. Final Report: Part 2, US EPA/GLNPO, ID: GL 00E75401–1 (Chicago, IL).
- Beckett, R., 2013. *Surface and Colloid Chemistry in Natural Waters and Water Treatment*. Springer Science & Business Media, New York.
- Beletsky, D., Saylor, J.H., Schwab, D.J., 1999. Mean circulation in the Great Lakes. *J. Great Lakes Res.* 25, 78–93.
- Bell, G.L., 1978. Characteristics of the Oswego River Plume and its Influence on the Near-shore Environment.
- Benjamin, M.M., Lawler, D.F., 2013. *Water Quality Engineering: Physical/Chemical Treatment Processes*. John Wiley & Sons, New Jersey.
- van den Berg, C.M., Merks, A.G., Duursma, E.K., 1987. Organic complexation and its control of the dissolved concentrations of copper and zinc in the Scheldt estuary. *Estuar. Coast. Shelf Sci.* 24, 785–797.
- Croley, T.E., 1989. Verifiable evaporation modeling on the Laurentian Great Lakes. *Water Resour. Res.* 25, 781–792.
- Dagg, M., Benner, R., Lohrenz, S., Lawrence, D., 2004. Transformation of dissolved and particulate materials on continental shelves influenced by large rivers: plume processes. *Cont. Shelf Res.* 24, 833–858.
- Dayton, A.L., Auer, M.T., Atkinson, J.F., 2014. Cladophora, mass transport, and the near-shore phosphorus shunt. *J. Great Lakes Res.* 40, 790–799.
- Fahnenstiel, G.L., Bridgeman, T.B., Lang, G.A., McCormick, M.J., Nalepa, T.F., 1995a. Phytoplankton productivity in Saginaw Bay, Lake Huron: effects of zebra mussel (*Dreissena polymorpha*) colonization. *J. Great Lakes Res.* 21, 464–475.
- Fahnenstiel, G.L., Lang, G.A., Nalepa, T.F., Johengen, T.H., 1995b. Effects of zebra mussel (*Dreissena polymorpha*) colonization on water quality parameters in Saginaw Bay, Lake Huron. *J. Great Lakes Res.* 21, 435–448.
- Fox, J., Hill, P., Milligan, T., Boldrin, A., 2004. Flocculation and sedimentation on the Po River Delta. *Mar. Geol.* 203, 95–107.
- Gibbs, R.J., 1983. Effect of natural organic coatings on the coagulation of particles. *Environ. Sci. Technol.* 17, 237–240.
- Good, S.P., Mallia, D.V., Lin, J.C., Bowen, G.J., 2014. Stable isotope analysis of precipitation samples obtained via crowdsourcing reveals the spatiotemporal evolution of superstorm sandy. *PLoS One* 9, e91117.
- Grimes, C.B., 2001. Fishery production and the Mississippi River discharge. *Fisheries* 26, 17–26.
- Grimes, C.B., Finucane, J.H., 1991. Spatial distribution and abundance of larval and juvenile fish, chlorophyll and macrozooplankton around the Mississippi River discharge plume, and the role of the plume in fish recruitment. *Mar. Ecol. Prog. Ser.* 75, 109–119 (Oldendorf).
- Grimes, C.B., Kingsford, M.J., 1996. How do riverine plumes of different sizes influence fish larvae: do they enhance recruitment? *Mar. Freshw. Res.* 47, 191–208.
- Grimm, E., 2013. *Characterization and Mapping of Buoyant River Plumes in Southern Lake Michigan*. Purdue University.
- Gross, A., Boyd, C.E., 1998. A digestion procedure for the simultaneous determination of total nitrogen and total phosphorus in pond water. *J. World Aquacult. Soc.* 29, 300–303.
- Hecky, R., Smith, R.E., Barton, D., Guildford, S., Taylor, W., Charlton, M., Howell, T., 2004. The nearshore phosphorus shunt: a consequence of ecosystem engineering by dreissenids in the Laurentian Great Lakes. *Can. J. Fish. Aquat. Sci.* 61, 1285–1293.
- Howell, E., Chomicki, K., Kaltenecker, G., 2012. Tributary discharge, lake circulation and lake biology as drivers of water quality in the Canadian Nearshore of Lake Ontario. *J. Great Lakes Res.* 38, 47–61.
- Jarvie, H.P., Withers, J., Neal, C., 2002. Review of robust measurement of phosphorus in river water: sampling, storage, fractionation and sensitivity. *Hydrol. Earth Syst. Sci. Discuss.* 6, 113–131.
- Johengen, T.H., Biddanda, B.A., Cotner, J.B., 2008. Stimulation of Lake Michigan plankton metabolism by sediment resuspension and river runoff. *J. Great Lakes Res.* 34, 213–227.
- Jude, D.J., Pappas, J., 1992. Fish utilization of Great Lakes coastal wetlands. *J. Great Lakes Res.* 18, 651–672.
- Kineke, G., Woolfe, K., Kuehl, S., Milliman, J., Dellapenna, T., Purdon, R., 2000. Sediment export from the Sepik River, Papua New Guinea: evidence for a divergent sediment plume. *Cont. Shelf Res.* 20, 2239–2266.
- Klaus, J., McDonnell, J., 2013. Hydrograph separation using stable isotopes: review and evaluation. *J. Hydrol.* 505, 47–64.
- Krieger, K.A., Klarer, D., Heath, R., Herdendorf, C., 1992. Coastal wetlands of the Laurentian Great Lakes: current knowledge and research needs. Preface: a call for research on Great Lakes coastal wetlands. *J. Great Lakes Res.* 18, 525–528.
- Ladouche, B., Probst, A., Viville, D., Idir, S., Baqué, D., Loubet, M., Probst, J.-L., Bariac, T., 2001. Hydrograph separation using isotopic, chemical and hydrological approaches (Strengbach catchment, France). *J. Hydrol.* 242, 255–274.
- Larson, J.H., Trebitz, A.S., Steinman, A.D., Wiley, M.J., Mazur, M.C., Pebbles, V., Braun, H.A., Seelbach, P.W., 2013. Great Lakes river mouth ecosystems: scientific synthesis and management implications. *J. Great Lakes Res.* 39, 513–524.
- Larson, J.H., Frost, P.C., Vallazza, J.M., Nelson, J.C., Richardson, W.B., 2016. Do river mouths alter nutrient and seston delivery to the nearshore? *Freshw. Biol.* 61 (11), 1935–1949.
- Laudon, H., Slaymaker, O., 1997. Hydrograph separation using stable isotopes, silica and electrical conductivity: an alpine example. *J. Hydrol.* 201, 82–101.
- Lohrenz, S.E., Fahnenstiel, G.L., Redalje, D.G., Lang, G.A., Dagg, M.J., Whitedge, T.E., Dortch, Q., 1999. Nutrients, irradiance, and mixing as factors regulating primary production in coastal waters impacted by the Mississippi River plume. *Cont. Shelf Res.* 19, 1113–1141.
- Lohrenz, S.E., Fahnenstiel, G.L., Millie, D.F., Schofield, O.M., Johengen, T., Bergmann, T., 2004. Spring phytoplankton photosynthesis, growth, and primary production and relationships to a recurrent coastal sediment plume and river inputs in southeastern Lake Michigan. *J. Geophys. Res.: Oceans* 109.
- Magnuson, J., Webster, K., Assel, R., Bowser, C., Dillon, P., Eaton, J., Evans, H., Fee, E., Hall, R., Mortsch, L., 1997. Potential effects of climate changes on aquatic systems: Laurentian Great Lakes and Precambrian shield region. *Hydrol. Process.* 11, 825–871.
- Marko, K.M., Rutherford, E.S., Eadie, B.J., Johengen, T.H., Lansing, M.B., 2013. Delivery of nutrients and seston from the Muskegon River Watershed to near shore Lake Michigan. *J. Great Lakes Res.* 39 (4), 672–681.
- Matsubayashi, U., Velasquez, G.T., Takagi, F., 1993. Hydrograph separation and flow analysis by specific electrical conductance of water. *J. Hydrol.* 152, 179–199.
- McDonnell, J., Stewart, M., Owens, I., 1991. Effect of catchment-scale subsurface mixing on stream isotopic response. *Water Resour. Res.* 27, 3065–3073.
- Mida, J.L., Scavia, D., Fahnenstiel, G.L., Pothoven, S.A., Vanderploeg, H.A., Dolan, D.M., 2010. Long-term and recent changes in southern Lake Michigan water quality with implications for present trophic status. *J. Great Lakes Res.* 36, 42–49.
- Miller, R.L., Bradford, W.L., Peters, N.E., 1988. Specific conductance: theoretical considerations and application to analytical quality control. U. S. Geol. Survey., Water-Supply Pap. 2311.
- Milligan, T., Hill, P., Law, B., 2007. Flocculation and the loss of sediment from the Po River plume. *Cont. Shelf Res.* 27, 309–321.
- Morrice, J., Kelly, J.R., Trebitz, A.S., Cotter, A.M., Knuth, M.L., 2004. Temporal dynamics of nutrients (N and P) and hydrology in a Lake Superior coastal wetland. *J. Great Lakes Res.* 30, 82–96.
- Morrice, J., Trebitz, A., Kelly, J., Cotter, A., Knuth, M., 2009. Nutrient variability in Lake Superior coastal wetlands: the role of land use and hydrology. *The State of Lake Superior*, pp. 217–238.
- Ndung'u, K., Franks, R.P., Bruland, K.W., Flegal, A.R., 2003. Organic complexation and total dissolved trace metal analysis in estuarine waters: comparison of solvent-extraction graphite furnace atomic absorption spectrometric and chelating resin flow injection inductively coupled plasma-mass spectrometric analysis. *Anal. Chim. Acta* 481, 127–138.
- Nielson, K.E., Bowen, G.J., 2010. Hydrogen and oxygen in brine shrimp chitin reflect environmental water and dietary isotopic composition. *Geochim. Cosmochim. Acta* 74, 1812–1822.
- Ozersky, T., Malkin, S.Y., Barton, D.R., Hecky, R.E., 2009. Dreissenid phosphorus excretion can sustain *C. glomerata* growth along a portion of Lake Ontario shoreline. *J. Great Lakes Res.* 35, 321–328.
- Pellerin, B.A., Wollheim, W.M., Feng, X., Vörösmarty, C.J., 2008. The application of electrical conductivity as a tracer for hydrograph separation in urban catchments. *Hydrol. Process.* 22, 1810–1818.
- Peterson, M.S., 2003. A conceptual view of environment-habitat-production linkages in tidal river estuaries. *Rev. Fish. Sci.* 11, 291–313.
- R Core Team, 2015. *A Language and Environment for Statistical Computing*. R Found. for Stat. Comput., Vienna, Austria.
- Rasmusson, E.M., 1967. Atmospheric water vapor transport and the water balance of North America: part I, paper presented at characteristics of the water vapor flux field. *Mon. Weather Rev.* 95, 403–426.
- Rasmusson, E.M., 1968. Atmospheric water vapor transport and the water balance of North America: II. Large-scale water balance investigations. *Mon. Weather Rev.* 96, 720–734.
- Reichert, J.M., Fryer, B.J., Pangle, K.L., Johnson, T.B., Tyson, J.T., Drelich, A.B., Ludsin, S.A., 2010. River-plume use during the pelagic larval stage benefits recruitment of a lentic fish. *Can. J. Fish. Aquat. Sci.* 67, 987–1004.
- Spence, C., Blanken, P., Hedstrom, N., Fortin, V., Wilson, H., 2011. Evaporation from Lake Superior: 2: spatial distribution and variability. *J. Great Lakes Res.* 37, 717–724.
- Steinman, A., Chu, X., Ogdahl, M., 2009. Spatial and temporal variability of internal and external phosphorus loads in Mona Lake, Michigan. *Aquat. Ecol.* 43, 1–18.
- Stephens, B.M., Minor, E.C., 2010. DOM characteristics along the continuum from river to receiving basin: a comparison of freshwater and saline transects. *Aquat. Sci.* 72, 403–417.
- Stumm, W., Morgan, J.J., 1981. *Aquatic Chemistry: An Introduction Emphasizing Chemical Equilibria in Natural Waters*. John Wiley, New Jersey.
- Tan, J., Cherkauer, K.A., Chaubey, I., Troy, C.D., Essig, R., 2016. Water quality estimation of river plumes in southern Lake Michigan using Hyperion. *J. Great Lakes Res.* 42 (3), 524–535.
- Vanderploeg, H., Johengen, T., Lavrentyev, P.J., Chen, C., Lang, G., Agy, M., Bundy, M., Cavaletto, J., Eadie, B., Liebig, J., 2007. Anatomy of the recurrent coastal sediment plume in Lake Michigan and its impacts on light climate, nutrients, and plankton. *J. Geophys. Res. Oceans* 112.
- Warrick, J.A., Milliman, J.D., 2003. Hyperpynal sediment discharge from semiarid southern California rivers: implications for coastal sediment budgets. *Geology* 31, 781–784.
- Wilson, K.A., Howell, E.T., Jackson, D.A., 2006. Replacement of zebra mussels by quagga mussels in the Canadian nearshore of Lake Ontario: the importance of substrate, round goby abundance, and upwelling frequency. *J. Great Lakes Res.* 32, 11–28.
- Wright, L.D., Nitttrouer, C.A., 1995. Dispersal of river sediments in coastal seas: six contrasting cases. *Estuaries* 18, 494–508.
- Wright, L., Wiseman, W., Yang, Z.-S., Bornhold, B., Keller, G., Prior, D., Suhayda, J., 1990. Processes of marine dispersal and deposition of suspended silts off the modern mouth of the Huanghe (Yellow River). *Cont. Shelf Res.* 10, 1–40.