

3-2022

Peak chlorophyll a concentrations in the lower Mississippi River from 1997 to 2018

R. Eugene Turner

Charles S. Milan

Erick M. Swenson

James M. Lee

Follow this and additional works at: https://digitalcommons.lsu.edu/oceanography_coastal_pubs



Part of the [Marine Biology Commons](#), and the [Oceanography Commons](#)

Peak chlorophyll *a* concentrations in the lower Mississippi River from 1997 to 2018

R. Eugene Turner ,* Charles S. Milan, Erick M. Swenson, James M. Lee

Department of Oceanography and Coastal Sciences, Louisiana State University, Baton Rouge, Louisiana

Abstract

Large and turbid rivers have varying temperatures, light conditions, nutrient availability, and nutrient ratios that may affect phytoplankton communities and occur within a changing world of point and nonpoint source nutrient loadings. We investigated how these physical and chemical factors affect Chlorophyll *a* (Chl *a*) concentrations in the Mississippi River, the largest river in North America, by sampling 878 times from February 1997 to December 2018 near its terminus at Baton Rouge, Louisiana. We hypothesized that nutrient concentrations and ratios were significant factors limiting phytoplankton biomass accumulations in this turbid river. The Chl *a* concentrations were in the “poor” water quality category when above $20 \mu\text{g Chl } a \text{ L}^{-1}$ 12% of the time. Two percent of the samples were $> 40 \mu\text{g Chl } a \text{ L}^{-1}$ and occurred on declining discharges. Results from graphical analysis and a principal component regression analysis showed that the highest Chl *a* values were constrained to when dissolved silicate: dissolved inorganic nitrogen (DIN) molar ratios $< 1 : 1$ and dissolved DIN : phosphate molar ratios $< 16 : 1$, which is when the phytoplankton community likely consists of non-siliceous phytoplankton. Increasing light conditions and reducing turbulence, which happens when river water is diverted into calmer and shallower waters, will create a phytoplankton bloom—perhaps becoming a harmful algal bloom—as has happened previously.

The surface waters of large and turbid rivers are thinly illuminated and constantly mixing with the thicker darkness below. River phytoplankton grow and accumulate there only if the brief and rapid exposure to light is enough to sustain them during their high frequency descents and ascents through strong gradients of light found within a thin portion of the total water column. These bursts of sunlight exposure (sunflecks) may account for 62–99% of the cumulative sunlight exposure (Gardner et al. 2020). Phytoplankton may be constrained by these physical realities, but not completely; they can survive if growth rates, buoyancy and light adaptations are sufficient to overcome the limitations of brief irradiances and predation. But they also need sufficient concentrations and ratios of essential nutrients to take advantage of brief exposures of light for photosynthesis to occur. The concentrations of three of these—nitrogen, phosphorous and silica—have been changing in the last 200 yr as humans

learned to use nitrogen derived from industrial processes, to mine, process and distribute phosphorus, and to influence various element cycling rates through hydrologic engineering, to name just three factors.

Plankton communities in rivers are indicators of water quality, and the presence of harmful algal blooms (HABs) can be a particularly nasty one (Hallegraeff et al. 2021). Some HABs are toxic to wildlife and livestock, particularly non-siliceous species of cyanobacteria, which may create taste and odor problems when the rivers are used for drinking water sources. Dodds et al. (2009), for example, estimated that water quality compromises cost at least \$2.2 billion yr^{-1} in the U.S. The resulting social pressures to restore water quality in large rivers has had more success in reducing P loading from point sources, principally through wastewater treatment, than from nonpoint sources; agriculture has been stubbornly resistant to nutrient reductions (Grizzetti et al. 2012; Thorsøe et al. 2021). Dams differentially trap dissolved silicate (DSi) over N and P (Humborg et al. 2006; Maavara et al. 2020) and increase water residence times and trap sediments. There is evidence that higher summer temperatures drive total abundance but decrease the toxicity of some blooms, and that cyanobacterial abundance is strongly influenced by the length of the summer (Ho and Michalak 2020). Reservoirs and lakes can be sites of nitrogen fixation (Scott et al. 2009; Moutinho et al. 2021) that feed nitrogen and cyanobacteria into the

*Correspondence: euturne@lsu.edu

This is an open access article under the terms of the Creative Commons Attribution-NonCommercial-NoDerivs License, which permits use and distribution in any medium, provided the original work is properly cited, the use is non-commercial and no modifications or adaptations are made.

Additional Supporting Information may be found in the online version of this article.

larger hydrologic network, especially as discharges decline (Knowlton and Jones 1997, 2000).

The consequences of changes in the nutrient concentrations and ratios that arise are often viewed through the lens of Redfield molar ratios (106 : 16 : 1 :: C : N : P; Redfield 1958; Redfield et al. 1963). Nutrient ratios in the water are seen as physiological thresholds differentiating nutrient limitations for phytoplankton growth. A dissolved inorganic silicate : nitrogen ratio < 1 : 1 (DSi : DIN) is widely used to prescribe the potentially favorable conditions for non-siliceous phytoplankton (Officer and Ryther 1980; Turner et al. 1998; Billen and Garnier 2007; Romero et al. 2013). But are nutrient ratios limiting in a turbid river where light availability is obviously limiting growth?

We studied the turbid Mississippi River, the largest river in North America, to examine how discharge, light, temperature, and nutrient ratios affect Chlorophyll *a* (Chl *a*) concentrations—an indicator of phytoplankton biomass. It drains 41% of the conterminous U.S. including 32 states and 2 Canadian Provinces and empties into the Gulf of Mexico. The Mississippi River naming comes from various Algonkian dialects meaning “Great River” and became comingled by colonialists with the “Muddy River”—a name given for the Missouri River by Native American Sioux (pronounced “Minisose”) and Osage (pronounced “Nishodse”) (Leland 1953). These namings are appropriate for its watershed area, discharge, and sediment yield, which are the third, ninth, and fifth, respectively, of the largest rivers in the world (Syvitski and Milliman 2007). The concentration of dissolved nitrate and phosphorus increased, and dissolved silicate decreased over the last 200 yr to change nutrient ratios; in particular, the average DSi : N molar ratio went from 4 : 1 to 1 : 1, or less (Turner and Rabalais 2003; Turner et al. 2007). The suspended sediment concentrations rose with colonization of the watershed and then decreased with soil conservation and dam construction, which surely affected light conditions in the river (Tweel and Turner 2012). Chl *a* concentrations at the end of the river are not well-known, beyond occasional monitoring data and a study by Duan and Bianchi (2006) who collected information on phytoplankton communities using a chromatograph program separating taxonomic groups in 23 samples collected from September 2001 to August 2003.

Our main objectives were to examine the physical and chemical factors influencing Chl *a* concentration in the Mississippi River which we did by using observations on river discharge, suspended material, light, and nutrient ratios collected over 22 yr. We used Chl *a* concentration as an indicator of biomass and hypothesized that nutrient concentrations and ratios were significant factors limiting phytoplankton biomass accumulations in this turbid river. In this context, patterns found in the variability of these relationships inform how future changes in these physical and chemical attributes have

consequences to phytoplankton within the river and when light conditions improve.

Methods

Sampling

All samples were taken from a wharf overhanging the left descending riverbank of the Mississippi River in Baton Rouge, LA, U.S. (30°26'34" N, 91°11'29" W), located 233 km (365 km of river distance) from the debouching end (Fig. 1). The average water depth at Baton Rouge is 15 m, the navigation channel is 15 m deep, and it is 16 m deep on nautical charts at the sampling site. The average width of the river at Baton Rouge is about 700 m.

We used a “seasoned” plastic bucket to collect surface water starting in February 1997 and ending December 2018 for a total of 878 collection events occurring at least once monthly. Water temperature data collection started in August 2002 ($n = 694$). The daily discharge data is from the United States Geological Survey (USGS) station at Tarbert Landing, LA (https://nwis.waterdata.usgs.gov/usa/nwis/qwdata/?site_no=07295100), located 124 river km upstream from Baton Rouge, LA.

Water quality analysis

The water quality analyses are identical to those described in Turner et al. (2019). Light penetration (cm) was measured with a Secchi disk (SD) and a commercial laboratory thermometer was used to measure water temperature immediately after sample collection. Suspended matter samples were first filtered through pre-weighed grade C borosilicate glass fiber filters (1.1 μm) that were pre-combusted at 550°C for 1 h. The filters were then dried at 60°C for 24 h, and then reweighed to determine the total suspended matter (TSS), heated to 550°C for 1.0 h, and then reweighed again to determine the amount of inorganic and organic suspended matter.

Water samples were frozen in polystyrene vials until determination of dissolved forms of phosphorus (P), nitrogen (N), and silicate (Si) using either a Technicon Autoanalyzer II (USEPA Method 353.2 for ammonium and nitrate + nitrite (DIN)), USEPA Method 365.2 for phosphate (DIP), and Technicon Method 186-72W/B for silicate) or a Lachat Quick-Chem 8000 Flow Injection Analyzer using the Lachat Methods approved by USEPA: method 31-107-06-1-B for ammonium, method 31-107-04-1-C for nitrate + nitrite (herein “nitrate”), method 31-115-01-1-H for inorganic phosphate (DIP), and method 31-114-27-1-C for silicate. A 5-point standard curve was used and QC standards were analyzed before, during and after each set of samples analyzed. Total nitrogen (TN) and total phosphorus (TP) concentrations were measured using a Technicon Autoanalyzer II or LaChat Quick-Chem after persulfate wet oxidation digestion (Raimbault et al. 1999). The TP and TN analyses were done on unfiltered samples. The coefficient of



Fig. 1. The Mississippi River watershed and location of the sampling station at Baton Rouge, LA. The photograph is from the state capitol observation deck, 2 km upstream from the interstate 10 highway bridge in the foreground on the left (downstream) side. Water samples are from within 0.6 km upstream of the bridge shown in the enlargement. Photograph by RETurner.

determination for the standard curve was > 0.98 for all nutrient analyses.

The Chl *a* concentration ($\mu\text{g Chl } a \text{ L}^{-1}$) was estimated using a Turner Model 10 fluorometer calibrated with a chemical supply house Chlorophyll standard and measured with a spectrophotometer (Lohrenz et al. 1999). The average percent difference for replicate Chl *a* values was 4.68%. We divided the Chl *a* concentrations into three groups: below $20 \mu\text{g Chl } a \text{ L}^{-1}$, between 20 and $39.9 \mu\text{g Chl } a \text{ L}^{-1}$, and $> 40 \mu\text{g Chl } a \text{ L}^{-1}$. The lowest concentration was chosen because a review of water quality in the U.S. classified water quality as “poor” with when the Chl *a* concentrations are $> 20 \mu\text{g Chl } a \text{ L}^{-1}$ (National Coastal Condition Report IV 2012; table 1–5). Whether the river was rising or falling for when $> 40 \mu\text{g Chl } a \text{ L}^{-1}$ was determined by comparing the daily discharge at Tarbert Landing, LA, at the day of sampling to the discharge 30 d earlier.

Statistics

We used Prism 9.1.2 software © 2021 (GraphPad Software, La Jolla, CA) for statistical analyses. The data were averaged and the standard error of the mean ($\mu \pm 1 \text{ SEM}$) for each analyte was calculated for each year. Violin plots of the Chl *a* concentration data were constructed to visually identify the width of the distribution of points in proportional to the number of points at that Y value. A frequency plot of all Chl *a* concentrations separated concentrations into $2 \mu\text{g L}^{-1}$ bin sizes. We performed a linear regression of the log-transformed

DIN : DIP data and the DSi : DIN data at a significance level of $p < 0.05$. There were 186 missing temperature values over the 880 sampling dates, resulting in 694 data rows with: (1) temperature, Secchi disk depth, and river discharge, (2) concentrations of suspended sediment, TN, TP, DIN, DIP, and DSi, and, (3) molar ratios of DSi : DIN and DIN : DIP. A principal component regression (PCR) was run on the untransformed data with the Parallel Analysis option to use the fewest number of variables in order to describe the most variance in the dependent variable Chl *a*, while retaining variance in the original data. Data on the percent oxygen saturation in surface waters of the Mississippi River at Plaquemine, LA, was obtained from the Louisiana Department of Environmental Quality (<https://waterdata.deq.louisiana.gov/>). Plaquemine, LA, is 34 km downstream from Baton Rouge, LA.

Results

The Chl *a* concentrations in all 878 samples ranged from 3 to $60 \mu\text{g Chl } a \text{ L}^{-1}$, with a mean value of $13.8 \pm 0.2 \mu\text{g Chl } a \text{ L}^{-1}$ (Fig. 2A). There were 19 sampling events that had Chl *a* concentration greater than $40 \mu\text{g Chl } a \text{ L}^{-1}$, but none occurred between 2007 and 2016. The months with the highest averages occurred from August through November (Fig. 2B). The percentage of samples in the three concentration classes (< 20 , $20\text{--}39.9$, $> 40 \mu\text{g Chl } a \text{ L}^{-1}$) were 87.8%, 10.4%, and 1.8% of the total (Fig. 2C), with average values of 11.6 ± 0.1 , 26.5 ± 0.6 , and $45.3 \pm 1.3 \mu\text{g Chl } a \text{ L}^{-1}$,

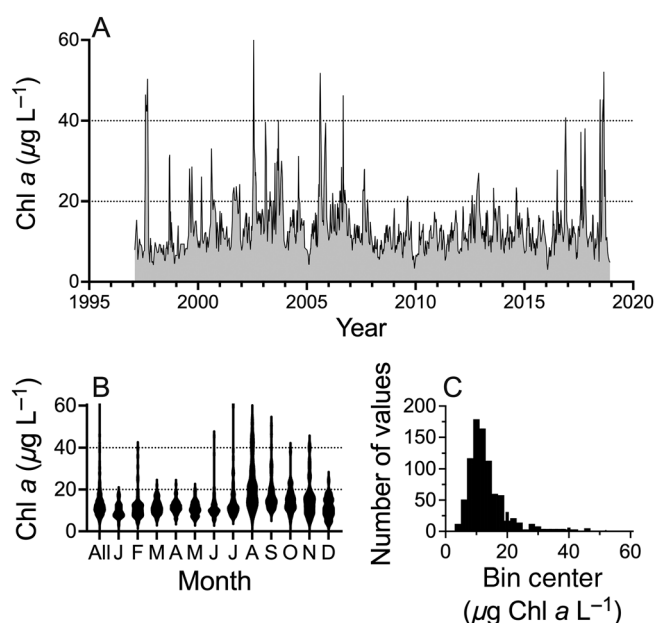


Fig. 2. The concentration of Chl *a* ($\mu\text{g L}^{-1}$) at Baton Rouge in 878 samples for 1997–2018. **(A)** Concentrations for each sampling date. A dotted line is at 20 and 40 $\mu\text{g Chl } a \text{ L}^{-1}$; **(B)** violin plots of monthly averages; **(C)** the frequency of $\mu\text{g Chl } a \text{ L}^{-1}$ values for all samples at different bin sizes.

respectively. There was no visual sign of a “step function” or “threshold” in the frequency distribution of Chl *a* values (Fig. 2C). All of the Chl *a* values $> 40 \mu\text{g L}^{-1}$ occurred when discharges in the previous 30 d were lower than when samples were collected (range = -3% to -68% ; average = -37%).

Physical factors and Chl *a*

Chl *a* concentrations were highest when river temperatures were above 28°C , and when discharge was lowest (Fig. 3A,B). One sample had a Chl *a* concentration that was high at intermediate temperatures (Fig. 3B), which is indicated by the dotted line. The Chl *a* concentration was not narrowly restricted to a range of Secchi disk depths (Fig. 3C). The highest Chl *a* values were at the intersection of high temperatures and low discharge (Fig. 3D), but not narrowly clustered at relatively high Secchi Disk depths and low discharge (Fig. 3E).

Suspended sediments and light

There is a curvilinear relationship between suspended sediments and discharge in which suspended sediment concentrations peak around $15\text{--}20 \text{ thousand m}^3 \text{ s}^{-1}$ and are lowest at both higher and lower river discharges (Fig. 4A). But light penetration, as measured by the Secchi disk depths (average $0.32 \pm 0.01 \text{ m}$; $n = 803$), increases as inorganic sediment concentration declines rapidly, but then decreases quite slowly below 100 mg L^{-1} (Fig. 4B).

Phytoplankton blooms, nutrients, and ratios

The concentration of DIN, DIP and DSi varied enormously throughout the 22 yr and the average concentrations were 100.2 ± 19.5 , 102.6 ± 2.23 , and $2.54 \pm 0.40 \mu\text{M}$, respectively ($n = 875$), for an average N : P ratio = 39.5, and DSi : DIN ratio of 1.0 (Figs. S1 and S2).

There was no peak in Chl *a* as the TN : TP, DIN : P or DSi : DIN molar ratios changed (Fig. 5A–C), except for higher Chl *a* values at the lowest DSi : DIN molar ratio (Fig. 5C). The concentration of nitrate was 74% of the TN concentration and well-constrained; the DIP concentration was about 55% of the TP concentration and more variable (Fig. S3). The Chl *a* concentrations $> 40 \mu\text{g L}^{-1}$ were confined to the quadrat with ratios above 16 : 1 :: DIN : DIP and $> 1 : 1 :: \text{DSi} : \text{DIN}$ (Fig. 5D). The two ratios were inversely related to each other in the river for all values ($R^2 = 0.33$; $p < 0.001$; $F_{1,873} = 422$, $n = 875$). In other words, the DSi : DIN ratio was lower as the DIN : DIP ratio increased (Fig. 5D). The annual average percent oxygen saturation in surface waters at Plaquemine, LA, was always 95%, or less (Fig. S5).

Principal component regression

The PCR analyses yielded many cumulative variances that depended on which variables were included. The highest cumulative variance in the Chl *a* concentration, 65 %, was described by 5 independent variables: Secchi disk depth, temperature, discharge, and the molar ratios of DSi : DIN, and DIN : DIP. The overall Goodness of Fit had an $R^2 = 0.34$ ($n = 694$; $F_{2,691} = 179$, $p < 0.0001$) with all parameter estimates were significant at $p < 0.001$. The PC1 axis is dominated by the three physical factors with temperature and Secchi disk depth in the same quadrat and discharge in the diagonally opposite quadrat, indicating contrasting influences (Fig. 6). The PC2 axis has the DSi : DIP molar ratio in the diagonally opposite quadrats with the DIN : DIP molar ratio, also indicating contrasting influences. The proportion of variance for the 1st and 2nd axis was 42.7% and 22.0% of the total variance for a cumulative total of 64.7% of the total variance. These PCR results for the molar ratios are in agreement with the data distribution shown in Fig. 5D.

Discussion

Suspended sediments and light

The curvilinear pattern between suspended sediments and discharge is consistent with the results of Mossa (1988). She analyzed the suspended sediment concentrations of sand, silt, and clay in the lower Mississippi River from 1972 to 1985 and found that the variations in discharge affected the timing of the peak in suspended sediment concentrations and load which preceded the discharge peak by weeks or even months, with longer periods in high discharge years compared to in low discharge years. There were nonlinear relationships between silt-clay fractions and discharge, but linear

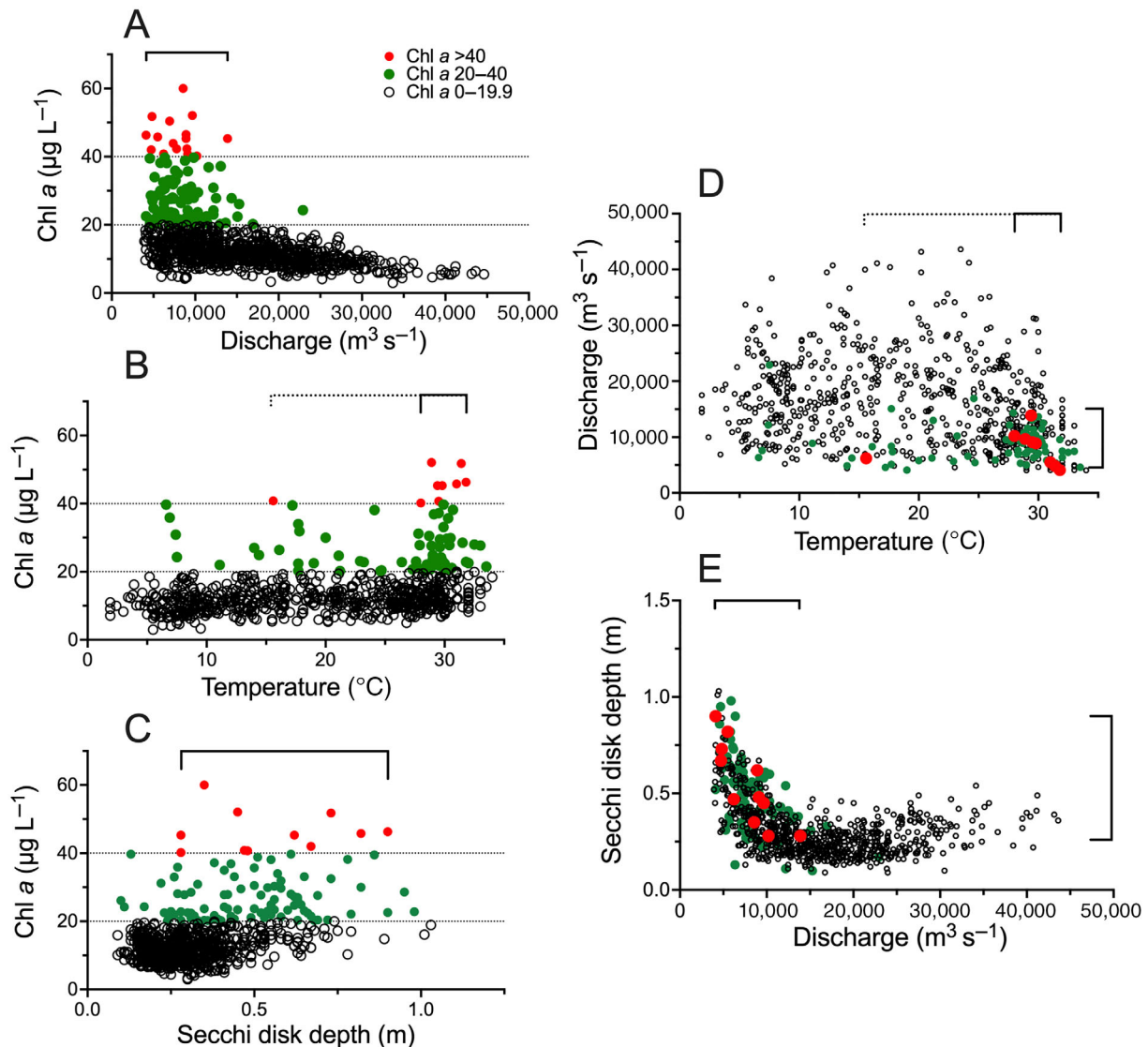


Fig. 3. The distribution of three categories of Chl *a* concentration ($\mu\text{g Chl } a \text{ L}^{-1}$) and three physical variables of the river. **(A)** Discharge ($\text{m}^3 \text{s}^{-1}$); **(B)** temperature ($^{\circ}\text{C}$); **(C)** Secchi disk depth (m); **(D)** discharge and temperature; **(E)** Secchi disk and discharge. The bars in each horizontal or vertical axis are the ranges of the Chl *a* concentration values > 40 $\mu\text{g L}^{-1}$.

relationships with sand concentrations and discharge; both contributed to the suspended sediment concentrations. She also described unexplained sediment peaks during both falling and rising stages that may have been caused by bank failures or turbulent fluctuations in stream velocity. Light penetration, as measured by Secchi disk depth measurements, is inversely related to the suspended sediment concentration and would be affected by these nonlinear relationships with discharge. Phosphorus is tightly bound to clay particles and so the concentration total phosphorus would be similarly affected in nonlinear and lagged ways with discharge. Both the light and nutrient regime, therefore, will be affected in nonlinear ways with discharge, which may influence Chl *a* concentrations.

Chl *a* concentrations

There is no simple relationship between Secchi disk depth and Chl *a* concentrations, but there were multiple influences from variations in discharge (presumably directly related to turbulence), temperature, light, and DSI : DIN and DIN : DIP ratios in the Mississippi River at Baton Rouge, LA. The conditions of low discharge, declining discharge, and higher temperatures favored higher Chl *a* concentrations. Secchi disk depths were only 2% of the average total river depth and so there was a strong effect of light limitation; but this effect did not completely overwhelm the biological responses, which were sensitive to nutrient ratios existing in a nutrient-rich medium. The Chl *a* concentrations were highest where the nutrients ratios were constrained to the Redfield boundaries of

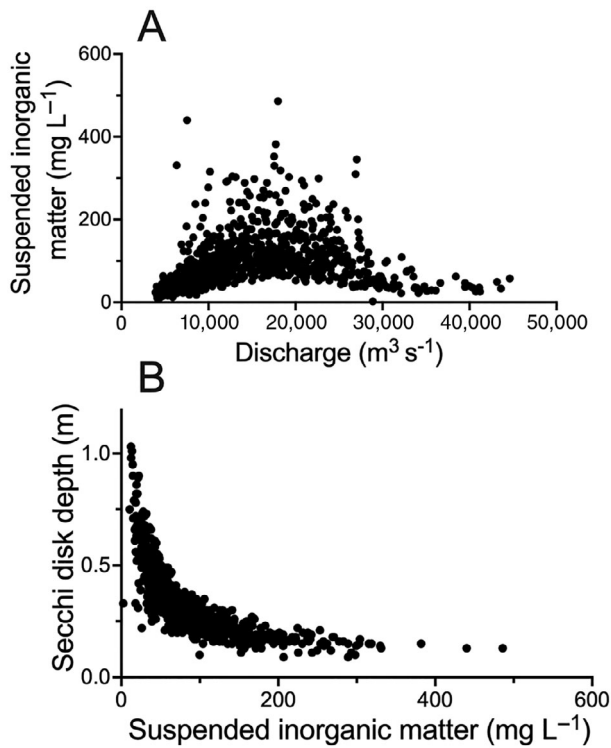


Fig. 4. The relationships between suspended inorganic matter (mg L⁻¹), discharge (m³ s⁻¹) and Secchi disk depth (m). **(A)** Suspended vs. discharge; **(B)** Secchi disk depth vs. suspended inorganic matter.

P and DSi limitation for non-siliceous phytoplankton, and perhaps being HABs. It would be interesting to expand coverage to simultaneously sample longitudinally, especially upstream, in order to improve our knowledge of how the variation in channel morphometrics influence velocity and affect species composition.

The average concentration of Chl *a* in all 878 samples ($13.8 \pm 0.2 \mu\text{g Chl } a \text{ L}^{-1}$) was populated 19 times over 21 yr by values more than three times that average, representing about 2% of the samplings. Twelve percent of the samples were in the “poor” water quality category. Duan and Bianchi (2006) found less Chl *a* in the river from 2001 to 2003 than we did for the same interval (average = $6.8 \pm 1.1 \mu\text{g Chl } a \text{ L}^{-1}$ ($n = 23$) vs. 18.2 ± 0.9 ($n = 89$)) and no values $> 30 \mu\text{g Chl } a \text{ L}^{-1}$. Ochs et al. (2013) also found about the same Chl *a* concentrations that we did over nine seasons (average = $9.8 \pm 1.0 \mu\text{g Chl } a \text{ L}^{-1}$, $n = 9$) and a plot of their Chl *a* concentration and river discharge shows an inverse relationship, as we found. The average annual Chl *a* values for the Mississippi River are lower than in the Missouri River (Knowlton and Jones 1997), the main channel of the Elbe and Rhine rivers (Lehmann and Rode 2001; Hardenbicker et al. 2014), Thames River (Bowes et al. 2012), and Seine River (Cugier et al. 2005), and less than in others at the river’s terminus (Loire River; Descy et al. 2011). It would be interesting to examine these rivers and others for similar relationships.

The highest Chl *a* concentrations were after falling discharges. This is when light conditions improve, washout is lowered, and the incubating floodwaters in floodplains and oxbow lakes are feeding into the river (Knowlton and Jones 1997, 2000). Although the Chl *a* concentrations are nontrivial, the river’s ecosystem is driven by allochthonous carbon as Reynolds and Descy (1996) surmised for the largest rivers, and which is the case for this river given that the oxygen saturation is almost always less than 100%.

Water quality has improved in some subregions of the Mississippi watershed because of conservation (Kling et al. 2014; Rabotyagov et al. 2014; McIsaac et al. 2016), but not at the end of the river, as shown here. Stets et al. (2015) reviewed the trends in nitrate concentrations in tributaries across the U.S. from 1945 to 2008 and found that the nitrate concentration increased strongly during 1945–1980 at most stations and then stabilized from 1981 to 2008. The nitrate concentrations, however, increased in the Midwest U.S., and less so in the Eastern and Western U.S. During 1945–1980, nitrate concentrations in the agricultural areas of the Midwestern part of the Mississippi River watershed increased up to fivefold. They found that the greatest increases in river nitrate levels coincided with increased nitrogen inputs from livestock and agricultural fertilizer inputs.

Phytoplankton communities

Monitoring of plankton biomass at the lower end of the Mississippi River, especially when the river has low DSi : DIN molar ratios, might inform water quality management efforts, especially where river water is used for a drinking water source and might contain harmful algae. In 2015, the suspended sediment concentrations in the river were half of what it was in 1980 (Mize et al. 2018) which surely resulted in increased illumination and changed plankton populations quantity, if not quality. The eutrophication of the lakes, reservoirs and streams feeding into the Mississippi River are a potential source of cyanobacteria which may produce hepatotoxins (Paerl 2018). Tropical reservoirs, for example, have Chl *a* concentrations that are directly related to the TP concentration and P can be the main nutrient controlling nitrogen fixation rates when cyanobacteria are present (Moutinho et al. 2021). But there is no systematic monitoring of plankton communities in the lower Mississippi River that we know of, other than by Duan and Bianchi (2006) who measured the biomass of diatoms using a chromatographic analyses, which was a different method from our fluorometric analysis. Diatom biomass in their samples ranged between 30% and 80% of the total phytoplankton biomass. The proportional amount of Chl *a* as diatom biomass increased as the discharge volume became lower (Fig. S3), but only one of their 23 samples had a Chl *a* concentration $> 20 \mu\text{g Chl } a \text{ L}^{-1}$ (September 2002, $21.9 \mu\text{g L}^{-1}$; our 5 samples averaged $16.3 \pm 0.4 \mu\text{g L}^{-1}$ for that month). The molar ratios in Duan and Bianchi’s (2006) samples were on the edges of the Redfield nutrient quadrat that

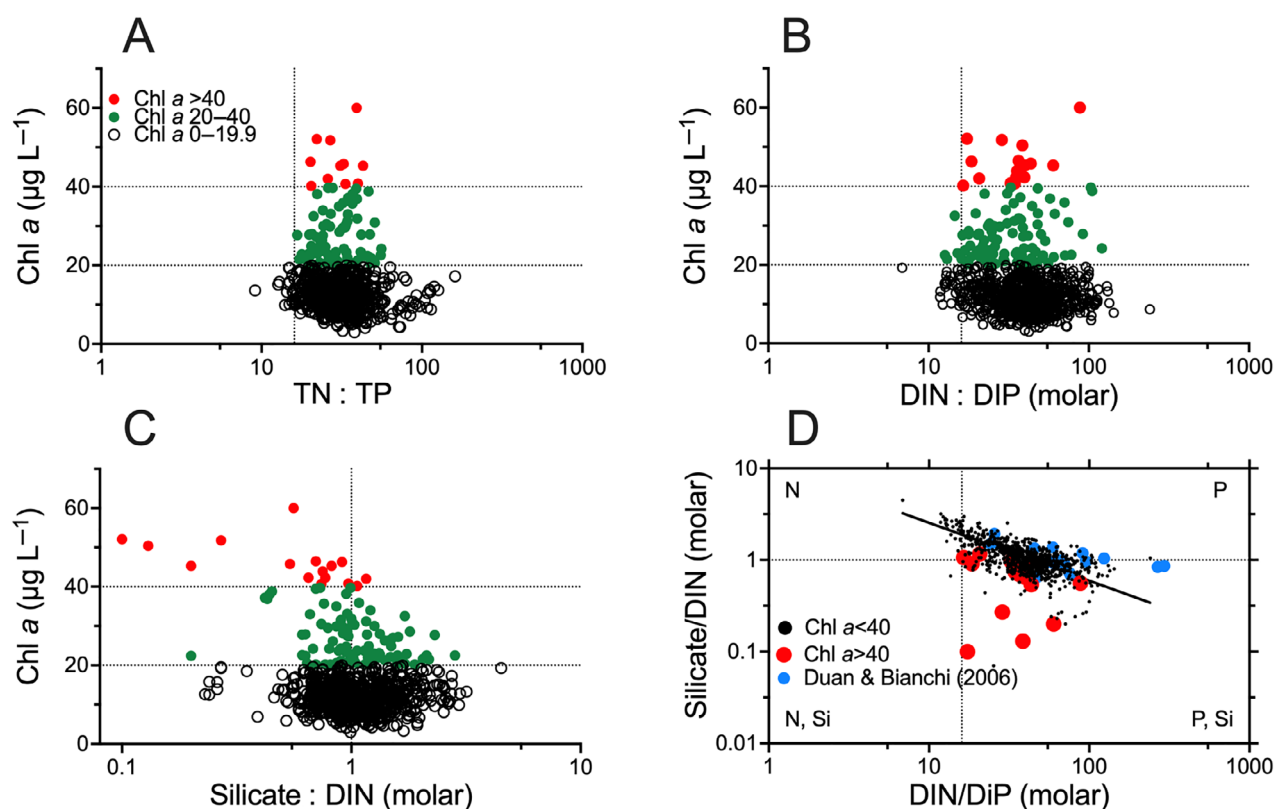


Fig. 5. The distribution of Chl *a* ($\mu\text{g L}^{-1}$) vs. the molar ratios. (A) TN : TP; (B) DIN : DIP, (C) DSi : DIN; and (D) DSi : DIN vs. DIN : DIP. The 23 blue dots are from Duan and Bianchi (2006) which are all $< 20 \mu\text{g Chl } a \text{ L}^{-1}$ except one. A linear regression line is shown for all of our data that is $< 40 \mu\text{g Chl } a \text{ L}^{-1}$.

our samples are in (Fig. 5). We speculate that higher Chl *a* values would have occurred if their samples had been in that lower right quadrat, and that diatoms would be largely absent

and replaced by non-siliceous phytoplankton, including cyanobacteria that might be toxic. Cugier et al.'s model of the past, present and future eutrophication in the Seine River (Cugier et al. 2005) suggested that diatoms were limited by P and DSi at most fluxes, which is when they found that dinoflagellates were dominant.

Consequences of light improvements and longer residence times

This analysis supports the conclusion that when phytoplankton in the Mississippi River are exposed to more light and lower water turnover that they will develop higher Chl *a* concentrations if other factors remain the same. A dramatic example of that potentiality happened when light conditions improved after river water was diverted through the Bonnet Carré Spillway and into the 3.7 m deep Lake Pontchartrain (Fig. 1) in 1997. The spillway was used to reduce the flooding potential for the downstream city of New Orleans and was at a time of relatively calm summer wind. The peak flow was $6793 \text{ m}^3 \text{ s}^{-1}$ and lasted 32 d during which it filled Lake Pontchartrain two times; the lake slowly mixed with oceanic water entering at the southeasterly end and was back to its salinity baseline value 4 weeks after the diversion closed, but not before there was an explosion of Chl *a* (average = $200 \mu\text{g Chl } a \text{ L}^{-1}$) and development of a toxic bloom consisting of

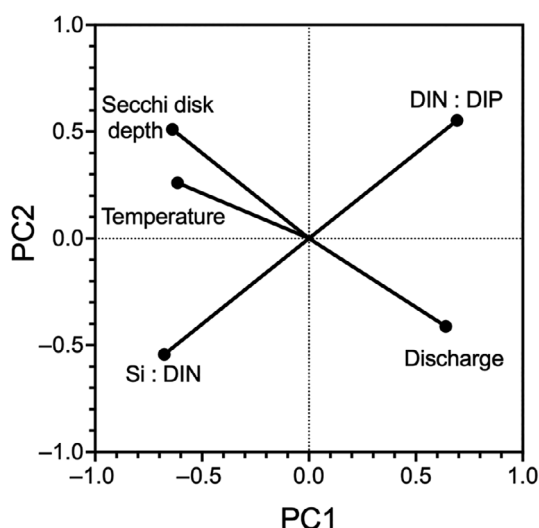


Fig. 6. Results from a principal component regression analysis that included Chl *a* concentration ($\mu\text{g L}^{-1}$) as the dependent variable. Axis PC1 includes three physical factors: Temperature, Secchi disk depth and river discharge volume. Axis PC2 includes DSi : DIN and DIN : DIP ratios located in diagonally opposing quadrats.

Anabaena spp. and *Microcystis* sp. (Dortch et al. 1997; Turner et al. 2004).

A second example is a diversion of river water into the Davis Pond diversion receiving area (marsh and swamp) and then Barataria Bay. There the discharge of the diverted water averaged $50 \text{ m}^3 \text{ s}^{-1}$ from 2003 to 2016. Less than half of the dissolved DIN and DIP were removed by the receiving wetlands. The Chl *a* concentration went from below $20 \mu\text{g Chl } a \text{ L}^{-1}$ to between 20 and $30 \mu\text{g Chl } a \text{ L}^{-1}$, and in proportion to the P loading added from the diversion and making its way to a lake 30 km away (Turner et al. In press).

These two examples inform the consequences of diverting river water at 10 times the discharge rate going into the Davis Pond diversion and then into estuaries via the planned Mid-Breton Sound Diversion, Lower Breton Sound Diversion, and Mid-Barataria Bay Diversion that are proposed to operate at 991, 1416, and $2124 \text{ m}^3 \text{ s}^{-1}$, respectively (Coastal Master Plan 2017). The estuaries receiving this water are about half of the depth of Lake Pontchartrain, some sediments in the diverted water will settle out to increase water clarity, and there will be longer water residence times. An estuary-wide phytoplankton bloom is a reasonable anticipated consequence, perhaps including HABs, particularly during the summer when winds are relatively low.

Conclusions

The Mississippi River phytoplankton are light limited, and their biomass was $> 40 \mu\text{g Chl } a \text{ L}^{-1}$ 2 % of the time and $> 20 \mu\text{g Chl } a \text{ L}^{-1}$ 12 % of the time. River discharge, temperature and light availability were the dominant factors controlling the variability of Chl *a* concentrations, but not exclusively. The highest Chl *a* values were constrained by both relatively high DIN : P and low Dsi : N ratios that are clustered around Redfield ratios. Changes in both ratios will affect species composition and the ratios where Chl *a* $> 40 \mu\text{g Chl } a \text{ L}^{-1}$ are indicative of non-siliceous plankton, such as cyanobacteria. Both ratios are influenced by anthropogenic factors, principally agricultural land use choices in the watershed. When river phytoplankton are relieved of most of the light-limitations, such as when rivers are diverted into calmer and shallower waters, then there will likely again create conditions for phytoplankton blooms—perhaps becoming a HAB bloom—as it has before.

Data Availability Statement

Data are available at the Dryad repository at <https://doi.org/10.5061/dryad.x95x69pkm>.

References

Billen, G., and J. Garnier. 2007. River basin nutrient delivery to the coastal sea: Assessing its potential to sustain new production of non-siliceous algae. *Mar. Chem.* **106**: 148–160. doi:[10.1016/j.marchem.2006.12.017](https://doi.org/10.1016/j.marchem.2006.12.017)

Bowes, M. J., E. Gozzard, A. C. Johnson, P. M. Scarlett, C. Roberts, D. S. Read, L. K. Armstrong, S. A. Harman, and H. D. Wickham. 2012. Spatial and temporal changes in chlorophyll-*a* concentrations in the River Thames basin, UK: Are phosphorus concentrations beginning to limit phytoplankton biomass? *Sci. Total Environ.* **426**: 45–55. doi:[10.1016/j.scitotenv.2012.02.056](https://doi.org/10.1016/j.scitotenv.2012.02.056)

Coastal Master Plan 2017. Coastal Restoration and Protection Authority, Baton Rouge, LA [accessed 2017 January 03]. Available from <http://coastal.la.gov/2017>

Cugier, P., G. Billen, J. F. Guillaud, J. Garnier, and A. Ménesguen. 2005. Modelling the eutrophication of the Seine Bight (France) under historical, present and future riverine nutrient loading. *J. Hydrol.* **304**: 381–396. doi:[10.1016/j.jhydrol.2004.07.049](https://doi.org/10.1016/j.jhydrol.2004.07.049)

Descy, J.-P., M. Leita, E. Everbecq, J. S. Smits, and J.-F. Deliége. 2011. Phytoplankton of the River Loire, France: A biodiversity and modelling study. *J. Plankton Res.* **34**(2): 120–135. doi:[10.1093/plankt/fbr085](https://doi.org/10.1093/plankt/fbr085)

Dodds, W. K., W. W. Bouska, J. L. Eitzmann, T. J. Pilger, K. L. Pitts, A. J. Riley, J. T. Schloesser, and D. J. Thornbrugh. 2009. Eutrophication of U.S. freshwaters: Analysis of potential economic damages. *Environ. Sci. Technol.* **43**: 12–19. doi:[10.1021/es801217q](https://doi.org/10.1021/es801217q)

Dortch, Q., T. D. Peterson, and S. Acheé. 1997. The 1997 algal bloom in Lake Pontchartrain. Chapter 6. In R. E. Turner, Q. Dortch, and N. N. Rabalais [eds.], *Effects of the 1997 Bonnet Carré opening on nutrients and phytoplankton in Lake Pontchartrain*. Rept. To the Lake Pontchartrain Basin Foundation.

Duan, S., and T. S. Bianchi. 2006. Seasonal changes in the abundance and composition of plant pigments in particulate organic carbon in the lower Mississippi and Pearl Rivers. *Estuaries Coasts* **29**(3): 427–442. doi:[10.1007/BF02784991](https://doi.org/10.1007/BF02784991)

Gardner, J. R., S. H. Ensign, J. N. Houser, and M. W. Doyle. 2020. Light exposure along particle flowpaths in large rivers. *Limnol. Oceanogr.* **65**: 128–142. doi:[10.1002/lno.11256](https://doi.org/10.1002/lno.11256)

Grizzetti, B., F. Bouraoui, and A. Aloe. 2012. Changes of nitrogen and phosphorus loads to European seas success with P but not N which have been stubbornly resistant to change. *Glob. Chang. Biol.* **18**: 769–782. doi:[10.1111/j.1365-2486.2011.02576.x](https://doi.org/10.1111/j.1365-2486.2011.02576.x)

Hallegraeff, G. M., and others. 2021. Perceived global increase in algal blooms is attributable to intensified monitoring and emerging bloom impacts. *Commun. Earth Environ.* **2**: 1–10. doi:[10.1038/s43247-021-00178-8](https://doi.org/10.1038/s43247-021-00178-8)

Hardenbicker, P., S. Rolinski, M. Weitere, and H. Fischer. 2014. Contrasting shifts and long-term trends in phytoplankton dynamics in two large rivers. *Int. Rev. Hydrobiol.* **99**: 287–299. doi:[10.1002/iroh.201301680](https://doi.org/10.1002/iroh.201301680)

Ho, J. C., and A. M. Michalak. 2020. Exploring temperature and precipitation impacts on harmful algal blooms across

- continental U.S. lakes. *Limnol. Oceanogr.* **65**: 992–1009. doi:[10.1002/lno.11365](https://doi.org/10.1002/lno.11365)
- Humborg, C., M. Pastuszak, J. Aigars, H. Siegmund, C. M. Morth, and V. Ittekkot. 2006. Decreased silica land–sea fluxes through damming in the Baltic Sea catchment. Significance of particle trapping and hydrological alterations. *Biogeochemistry* **77**(2): 265–281. doi:[10.1007/s10533-005-1533-3](https://doi.org/10.1007/s10533-005-1533-3)
- Kling, C. L., and others. 2014. LUMINATE: Linking agricultural land use, local water quality and Gulf of Mexico hypoxia. *Eur. Rev. Agric. Econ.* **41**: 431–459. doi:[10.1093/erae/jbu009](https://doi.org/10.1093/erae/jbu009)
- Knowlton, M. E., and J. R. Jones. 1997. Trophic status of Missouri River Floodplain lakes in relation to basin type and connectivity. *Wetlands* **17**(4): 468–475. doi:[10.1007/BF03161512](https://doi.org/10.1007/BF03161512)
- Knowlton, M. E., and J. R. Jones. 2000. Seston, light, nutrients and chlorophyll in the lower Missouri River, 1994–1998. *J. Freshw. Ecol.* **15**: 283–297. doi:[10.1080/02705060.2000.9663747](https://doi.org/10.1080/02705060.2000.9663747)
- Lehmann, A., and M. Rode. 2001. Long-term behaviour and cross-correlation water quality analysis of the River Elbe, Germany. *Water Res.* **35**: 2153–2160. doi:[10.1016/S0043-1354\(00\)00488-7](https://doi.org/10.1016/S0043-1354(00)00488-7)
- Leland, J. A. C. 1953. Indian names in Missouri. *Names* **1**: 266–273. doi:[10.1179/nam.1953.1.4.266](https://doi.org/10.1179/nam.1953.1.4.266)
- Lohrenz, S. E., G. L. Fahnenstiel, D. G. Redalje, G. A. Lang, M. J. Dagg, T. E. Whitledge, and Q. Dortch. 1999. Nutrient, irradiance, and mixing as factors regulating primary production in coastal waters impacted by the Mississippi River plume. *Cont. Shelf Res.* **19**(9): 1113–1141. doi:[10.1016/S0278-4343\(99\)00012-6](https://doi.org/10.1016/S0278-4343(99)00012-6)
- Maavara, T., Z. Akbarzadeh, and P. Van Cappellen. 2020. Global dam-driven changes to riverine N:P:Si ratios delivered to the coastal ocean. *Geophys. Res. Lett.* **47**: e2020GL088288. doi:[10.1029/2020GL088288](https://doi.org/10.1029/2020GL088288)
- Mclsaac, G. F., M. B. David, and G. Z. Gertner. 2016. Illinois River nitrate-nitrogen concentrations and loads: Long-term variation and association with watershed nitrogen inputs. *J. Environ. Qual.* **45**: 1268–1275. doi:[10.2134/jeq2015.10.0531](https://doi.org/10.2134/jeq2015.10.0531)
- Mize, S. V., J. C. Murphy, T. H. Diehl, and D. K. Demcheck. 2018. Suspended-sediment concentrations and loads in the lower Mississippi and Atchafalaya rivers decreased by half between 1980 and 2015. *J. Hydrol.* **564**: 1–11. doi:[10.1016/j.jhydrol.2018.05.068](https://doi.org/10.1016/j.jhydrol.2018.05.068)
- Mossa, J. 1988. Discharge-sediment dynamics of the lower Mississippi River. *Trans. Gulf Coast Assoc. Geol. Soc.* **38**: 303–314.
- Moutinho, F. H. M., G. A. Marafão, M. do Carmo Calijuri, M. Z. Moreira, A. M. Marcarelli, and D. G. F. Cunha. 2021. Environmental factors and thresholds for nitrogen fixation by phytoplankton in tropical reservoirs. *Int. Rev. Hydrobiol.* **106**: 5–17. doi:[10.1002/iroh.202002057](https://doi.org/10.1002/iroh.202002057)
- National Coastal Condition Report IV 2012. United States Environmental Protection Agency Office of Research and Development/Office of Water, Washington, DC, EPA-842-R-10-003 [accessed 2012 October 15]. Available from <http://www.epa.gov/nccr>
- Ochs, C. A., O. Pongruktham, and P. V. Zimba. 2013. Darkness at the break of noon: Phytoplankton production in the lower Mississippi River. *Limnol. Oceanogr.* **58**: 555–568. doi:[10.4319/lo.2013.58.2.0555](https://doi.org/10.4319/lo.2013.58.2.0555)
- Officer, C. B., and J. H. Ryther. 1980. The possible importance of silicon in marine eutrophication. *Mar. Ecol. Progr. Ser.* **3**: 383–391. doi:[10.3354/MEPS003083](https://doi.org/10.3354/MEPS003083)
- Paerl, H. W. 2018. Mitigating toxic planktonic cyanobacterial blooms in aquatic ecosystems facing increasing anthropogenic and climatic pressures. *Toxins* **10**: 76. doi:[10.3390/toxins10020076](https://doi.org/10.3390/toxins10020076)
- Rabotyagov, S., and others. 2014. Cost-effective targeting of conservation investments to reduce the northern Gulf of Mexico hypoxic zone. *Proc. Natl. Acad. Sci. USA* **111**: 18530–18535. doi:[10.1073/pnas.1405837111](https://doi.org/10.1073/pnas.1405837111)
- Raimbault, P., W. Pouvesta, F. Diaz, N. Garcia, and R. Sempéré. 1999. Wet-oxidation and automated colorimetry for simultaneous determination of organic carbon, nitrogen & phosphorus dissolved in seawater. *Mar. Chem.* **66**(3–4): 166–169. doi:[10.1016/S0304-4203\(99\)00038-9](https://doi.org/10.1016/S0304-4203(99)00038-9)
- Redfield, A. C. 1958. The biological control of chemical factors in the environment. *Am. Sci.* **46**: 205–221.
- Redfield, A. C., B. H. Ketchum, and F. A. Richards. 1963. The influence of organisms on the composition of sea-water, p. 12–37. *In* M. N. Hill [ed.], *The sea*. **2**: John Wiley & Sons. Hoboken, New Jersey
- Reynolds, C. S., and J.-P. Descy. 1996. The production, biomass and structure of phytoplankton in large rivers. *Large Rivers* **10**(1–4): 161–187. doi:[10.1127/lr/10/1996/161](https://doi.org/10.1127/lr/10/1996/161)
- Romero, E., J. Garnier, L. Lassaletta, G. Billen, R. Le Gendre, P. Riou, and P. Cugier. 2013. Large-scale patterns of river inputs in southwestern Europe: Seasonal and interannual variations and potential eutrophication effects at the coastal zone. *Biogeochemistry* **113**: 481–505. doi:[10.1007/s10533-012-9778-0](https://doi.org/10.1007/s10533-012-9778-0)
- Scott, J. T., J. K. Stanley, R. D. Doyle, M. G. Forbes, and B. W. Brooks. 2013. River–reservoir transition zones are nitrogen fixation hot spots regardless of ecosystem trophic state. *Hydrobiologia* **625**: 61–68. doi:[10.1007/s10750-008-9696-2](https://doi.org/10.1007/s10750-008-9696-2)
- Stets, E. G., V. J. Kelly, and C. G. Crawford. 2015. Regional and temporal differences in nitrate trends discerned from long-term water quality monitoring data. *J. Am. Water Res. Assoc.* **51**: 1394–1407. doi:[10.1111/1752-1688.12321](https://doi.org/10.1111/1752-1688.12321)
- Syvitski, J. P. M., and J. D. Milliman. 2007. Geology, geography, and humans battle for dominance over the delivery of fluvial sediment to the coastal ocean. *J. Geol.* **115**: 1–15. doi:[10.1086/509246](https://doi.org/10.1086/509246)
- Thorsøe, M. H., M. S. Andersen, M. V. Brady, M. Graversgaard, E. Kilis, A. Branth Pedersen, S. Pitzén, and H. Valve. 2021. Promise and performance of agricultural nutrient

- management policy: Lessons from the Baltic Sea. *Ambio* **51**: 36–50. doi:[10.1007/s13280-021-01549-3](https://doi.org/10.1007/s13280-021-01549-3)
- Turner, R. E., and N. N. Rabalais. 2003. Linking landscape and water quality in the Mississippi River Basin for 200years. *BioScience* **53**(6): 563–572. doi:[10.1641/0006-3568\(2003\)053\[0563:LLAWQI\]2.0.CO;2](https://doi.org/10.1641/0006-3568(2003)053[0563:LLAWQI]2.0.CO;2)
- Turner, R. E., N. Qureshi, N. N. Rabalais, Q. Dortch, D. Justić, R. Shaw, and J. Cope. 1998. Fluctuating silicate:nitrate ratios and coastal plankton food webs. *Proc. Natl. Acad. Sci. USA* **95**: 13048–13051. doi:[10.1073/pnas.95.22.13048](https://doi.org/10.1073/pnas.95.22.13048)
- Turner, R. E., Q. Dortch, and N. N. Rabalais. 2004. Inorganic nitrogen transformations at high loading rates in an oligohaline estuary. *Biogeochemistry* **68**(3): 411–423. doi:[10.1023/B:BIOG.0000031039.56794.29](https://doi.org/10.1023/B:BIOG.0000031039.56794.29)
- Turner, R. E., N. N. Rabalais, R. B. Alexander, G. McIsaac, and R. W. Howarth. 2007. Characterization of nutrient and organic carbon and sediment loads and concentrations from the Mississippi River into the northern Gulf of Mexico. *Estuaries Coasts* **30**: 773–790. doi:[10.1007/BF02841333](https://doi.org/10.1007/BF02841333)
- Turner, R. E., E. M. Swenson, C. S. Milan, and J. M. Lee. 2019. Spatial variations in chlorophyll *a*, C, N, and P in a Louisiana estuary from 1994 to 2016. *Hydrobiology* **834**(1): 131–144. doi:[10.1007/s10750-019-3918-7](https://doi.org/10.1007/s10750-019-3918-7)
- Turner, R. E., E. M. Swenson, C. S. Milan, and J. M. Lee. In press. Second thoughts: Mass removal efficiencies in water and consequences after a river diversion into a coastal wetland. *Hydrobiology*.
- Tweel, A. W., and R. E. Turner. 2012. Watershed land use and river engineering drive wetland formation and loss in the Mississippi River birdfoot delta. *Limnol. Oceanogr.* **57**: 18–28. doi:[10.4319/lo.2012.57.1.0018](https://doi.org/10.4319/lo.2012.57.1.0018)

Acknowledgments

Dr. Lauris Hollis and two anonymous reviewers are thanked for reviewing the manuscript. Dr. Jack Jones helpfully suggested to look at flow conditions prior to sample collection. This data collection and analysis was supported by the NOAA Coastal Ocean Program MULTISTRESS Award No. NA16OP2670 to Louisiana State University and NSF Rapid Grant DEB-1044599.

Conflict of Interest

None declared.

Submitted 23 July 2021

Revised 29 November 2021

Accepted 07 January 2022

Associate editor: John A. Downing