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Reconstructing the development of eutrophication in Louisiana salt marshes

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Abstract

We collected sediment cores from three salt marsh ponds in coastal Louisiana to test the usefulness of proxies of eutrophication. One-centimeter increments of ²¹⁰Pb- and ¹³⁷Cs-dated sediment were analyzed for diatoms, pigments (phaeophytin and chlorophyll *a*), biogenic silica, percentage organic matter, percentage carbon, and percentage nitrogen. Both sediment chlorophyll *a* and a diatom-based trophic index (TI) were significantly and positively correlated with riverine or local nutrient indices. Two diatom species, *Amphora copulata* Giffen and *Navicula yarrensii* Grunow, were significantly and negatively correlated with riverine and local nutrient indices. These results suggest that these variables can be used as potential indicators of trophic status. Results from a complete-linkage cluster analysis on the diatom assemblage data demonstrated that the sediment cores could be split into three time periods: early 1900s (pre-1930s/1940s), mid-1900s (1930s/40s to 1960s/1970s), and late 1900s (1960s/70s to 1990s). Examination of the sediment chlorophyll *a* and TI data over these time periods, coupled with an analysis of variance of nutrient inputs between the time periods, suggests that nutrient loading increased dramatically from the mid-1960s to the mid-1970s. This study demonstrates that (1) a retrospective analysis of sediment cores can be conducted in highly variable salt marsh ponds and (2) these salt marsh environments are already affected by the higher nutrient loads from both riverine and local processes occurring over the last 50 years. Additional nutrient loading, e.g., from river diversion projects for the lower Mississippi River, may exacerbate eutrophication already evident in the marsh environment.

Eutrophication, defined by Nixon (1995) as the increased rate of primary production, often due to increased nutrient inputs, represents the greatest single threat to coastal ecosystem health (Cloern 2001). Researchers have debated how much phosphorus- or nitrogen-based compounds contribute to eutrophication at any specific time and/or locale (Rabalais 2002). In either case, however, it is clear that both phosphorus and nitrogen loadings to aquatic systems have increased severalfold since preindustrial times because of increased inputs of phosphate and nitrate-based fertilizers, atmospheric

N deposition, and domestic/agricultural waste water runoff (Vitousek et al. 1997; Bennett et al. 2001).

Nutrient enrichment can lead to a (temporary) removal of nutrient limitation, resulting in increased plant/algal biomass in an aquatic ecosystem. The increase in algal biomass can lead to (1) decreased light levels that hinder benthic photosynthetic processes and (2) higher biological oxygen demand (BOD), due to respiration of the large algal biomass and/or consumer biomass (e.g., bacteria and grazers). A higher BOD can lead to incidences of hypoxia, an ongoing problem for many coastal areas globally (146 known areas were documented by Diaz et al. 2004).

Eutrophication also results in altered nutrient ratios (e.g., N×:×P, Si×:×N, Si×:×P), which can result in a change in the phytoplankton community composition (Officer and Ryther 1980; Anderson et al. 2002). N and P enrichment may lead to secondary nutrient limitation, such as silica limitation, which can result in hindered growth of diatoms and a subsequent shift toward flagellates (Officer and Ryther 1980). Some harmful algal bloom species may also take advantage of such altered nutrient ratios (Anderson et al. 2002). For example, increased P inputs may cause toxic cyanobacterial blooms (Anderson et al. 2002 and references therein), and increased N inputs coupled with a decreasing Si×:×N ratio may be responsible for the large presence of

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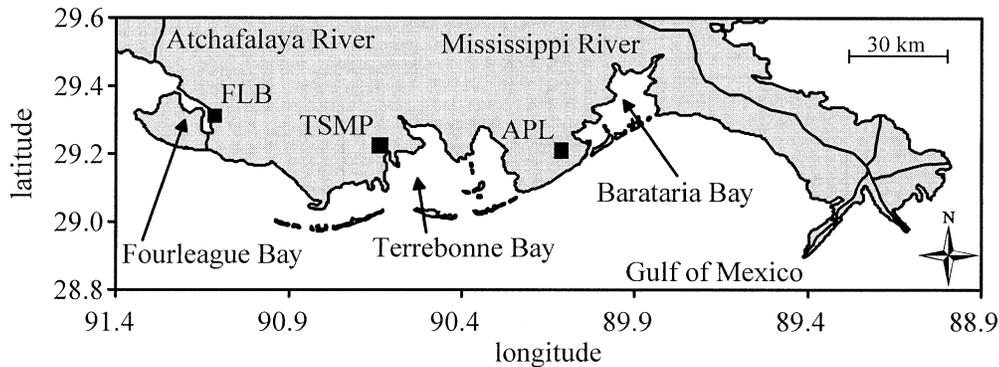


Fig. 1. A map of central coastal Louisiana illustrating the locations of the three study sites: Fourleague Bay marsh site (FLB), the Terrebonne Bay salt marsh pond (TSMP), and Airplane Lake (APL).

Pseudo-nitzschia spp. in coastal Louisiana (Parsons et al. 2002).

One of the best studied areas and clearest examples of the detrimental effects of coastal eutrophication are the shelf waters of Louisiana. The primary source of nutrients responsible for the eutrophication of coastal Louisiana is the Mississippi–Atchafalaya River complex, which serves as a major conduit transporting nonpoint source sediment and nutrients from a watershed encompassing 41% of the conterminous United States. Nitrogen and phosphorus concentrations in the Mississippi River have doubled since the 1950s because of the increased use of fertilizer and subsequent runoff in the watershed (Turner and Rabalais 1991; McIsaac et al. 2001). Additionally, the concentration of silicate in the Mississippi River decreased by 50% since the 1950s, probably in response to upstream sequestering of silica by freshwater diatoms impounded behind reservoir dams (Turner and Rabalais 1991).

Sediment core studies of Louisiana shelf waters have also documented historical changes in phytoplankton productivity. Biogenic silica (the major constituent of the diatom frustule) increased proportionally with the increased nutrient loading of the Mississippi River at a continental shelf site (Turner and Rabalais 1994). The diatom assemblage appears to have shifted toward smaller, more lightly silicified species (e.g., *Skeletonema* spp., *Cyclotella choctawhatcheeana*, Rabalais et al. 1996; Parsons unpubl. data).

While the changing water quality of the Mississippi River and the subsequent eutrophication of Louisiana shelf waters have been well studied, effects in the salt marshes and estuaries of Louisiana are less understood. The available data, summarized in Rabalais et al. (1995), suggest high nutrient loading in some estuaries. Data prior to the 1970s, however, are scant, making a detailed evaluation of estuarine eutrophication difficult. The conclusions drawn in Rabalais et al. (1995) indicate that water column chlorophyll biomass appears to have increased two to tenfold since the 1950s in some areas where data are available, indicating an increase in productivity that could be due to increased nutrient loads.

Understanding salt marsh eutrophication is important to wetland management and conservation. Salt marshes are an important nursery habitat for many organisms including brown shrimp (*Farfantepenaeus aztecus*), white shrimp (*Li-*

topenaeus setiferus), spotted seatrout (*Cynoscion nebulosus*), and red drum (*Sciaenops ocellatus*). Additionally, the salt marshes of Louisiana contain important oyster harvesting grounds, and a degraded water quality may affect this industry as well as introduce human health concerns (e.g., *Vibrio vulnificus*, pollutants, harmful algal blooms (HABs); Anderson et al. 2002).

The fate of Louisiana salt marshes is of concern because the loss rates are dramatically high, e.g., the annual wetland losses were 12,700 ha from 1955 to 1978 ($0.86\% \text{ yr}^{-1}$; Baumann and Turner 1990) and represent the dominant coastal land loss in the lower 48 states. The planned and existing diversions of Mississippi River water into the coastal marshes of Louisiana are meant to counter land loss under the assumption that the emergent vegetation growth is limited by nutrient availability and that there is a net “accretion deficit” that the suspended sediments in the diverted water will compensate for, once deposited on the marsh surface. There is an ongoing debate over whether river diversions will have positive, neutral, or negative effect on the marsh ecosystem.

The sparse environmental water quality data before the 1970s and the unknown effects of river diversions led to this study, the purposes of which were to (1) examine indicators of past productivity to establish whether eutrophication has already occurred and (2) set a time frame for the occurrence of major water quality changes. The results of this study may be useful to gain insight into the potential effects of river diversions, as well as demonstrating the usefulness of sediment core studies in reconstructing salt marsh environmental conditions when historical data do not exist.

Materials and methods

Site descriptions—We sampled three salt marsh ponds in south-central coastal Louisiana, representing distinct watersheds east of Fourleague Bay (FLB), northwest of Terrebonne Bay (Terrebonne salt marsh pond [TSMP]), and west of Barataria Bay at Airplane Lake (APL) (Fig. 1). We hypothesized that the FLB site would be most influenced by riverine processes due to its proximity to the Atchafalaya River delta, that the APL site would be most influenced by

Table 1. ^{137}Cs and ^{210}Pb core dating results for the three study sites: Airplane Lake (APL), Fourleague Bay (FLB), and the Terrebonne salt marsh pond (TSMP).

Core data	APL	FLB	TSMP
Year core was collected	1993	1994	1992
Postextrusion core length (cm)	105.7	74.3	68.7
Depth of ^{137}Cs peak (cm)	9.5	36.5	35.5
^{137}Cs sedimentation rate (cm yr $^{-1}$)	0.33	1.19	1.24
^{210}Pb sedimentation rate (cm yr $^{-1}$)	0.31	0.8	0.8
Average sedimentation rate (cm yr $^{-1}$)	0.32	1.0	1.0
Time frame of each core	1664*–1993	1924–1992	1920–1994

* Year was calculated assuming the whole APL core had an average sedimentation rate of 0.32 cm yr $^{-1}$. Adapted from Parsons et al. (1999).

the discharge of the nearby Mississippi River, and that the TSMP site would be most influenced by local processes due to its intermediate distance from both the Mississippi and Atchafalaya Rivers.

Formulation of riverine nitrogen-loading indices—Riverine and local nitrogen-loading indices (RNI and LNI, respectively) were formulated to provide a proxy measurement for the annual input of nitrogen from these sources into the three watersheds. Data on nitrogen concentration in the Mississippi River are available since 1955 and indicate that nutrient concentrations began to rise in the 1950s due to increased fertilizer production (Turner and Rabalais 1991). It was crucial, however, to have data that extended back before the 1950s to provide a baseline of nutrient inputs before the economic expansion of the post-World War II era. A proxy riverine nitrogen-loading indicator (RNI) was therefore formulated with fertilizer use and river discharge data to provide the baseline and subsequent conditions.

Fertilizer use within the Mississippi River watershed (MRW) was determined by gathering data on annual fertilizer use (U.S. Department of Agriculture [USDA] 1929–1990) for each of the 22 states having significant land area within the watershed. Seven states (New York, Maryland, Virginia, North Carolina, Georgia, Alabama, and Texas) were excluded because of the small area of each within the watershed. The proportion of the agricultural land of each state located within the MRW was estimated from land use maps (United States Department of Interior 1970), and fertilizer use was then calculated from these proportions. MRW fertilizer use was the only nutrient source used to represent the external nitrogen inputs, since earlier work has demonstrated that increased fertilizer use since the 1950s is primarily responsible for the increased nutrient loading of the Mississippi River (Turner and Rabalais 1991; McIsaac et al. 2001). Annual Mississippi River discharges from Tarbert Landing, Mississippi, and annual Atchafalaya River discharges from Simmesport, Louisiana, were used to estimate the loading of MRW nitrogen into the Gulf of Mexico and as a relative measure of potential loading to the three study sites via mixing at tidal passes.

The RNI was calculated by multiplying annual nitrogen fertilizer use by annual river discharge for each year between 1930 and 1990 and then dividing this value by the average product of these variables. This calculation resulted in a unitless measure of riverine nitrogen inputs (the RNI) that was

>1 during years with nitrogen inputs above the overall average and <1 during below-average years. The RNI for APL used Mississippi River discharge data, whereas the RNI for TSMP and FLB used Atchafalaya River discharge data (due to the closer proximity of the Atchafalaya River to these two sites). A correlation analysis of the RNI versus available nitrate flux estimates from the Mississippi River (post-1955, Parsons et al. 2002) demonstrated that the RNI was a good proxy indicator ($r^2 = 0.61$; $p < 0.0001$).

Formulation of local nitrogen input indices—Local nitrogen inputs to the three watersheds were determined from historical population, fertilizer use, and livestock data collected from Louisiana Department of Agriculture documents (Annual Report of the Agricultural and Industrial Statistics and Crop Report, Louisiana Department of Agriculture 1872–1930; Annual Fertilizer Tonnage and Analysis Report, Louisiana Department of Agriculture 1948–1994). The data were collected from 1900 to 1994 for the parishes that contained a significant amount of land area within any of the three watersheds (data were scant before 1900). The parish data were then converted to watershed data according to the proportion of cropland, pastureland, and residential areas of each parish present within each watershed (Louisiana Coastal Resources Atlas, Anon. 1977). Population, fertilizer use, and livestock data, which are the primary sources of anthropogenic nutrients, were converted to weight nitrogen and added together to calculate an estimate of annual local nitrogen loading for each watershed. Population numbers were converted to weight nitrogen using the USDA conversion factor (8681.85 g N person $^{-1}$ yr $^{-1}$; Goolsby and Battaglin pers. comm.). Livestock data were converted to g N animal $^{-1}$ yr $^{-1}$ using the conversions in the Livestock Waste Facilities Handbook (Anon. 1985).

Annual precipitation data collected by the Louisiana State Climatological Office at their Houma station was used to provide an estimate for runoff processes. This station was the closest station to the three study sites and had the longest continuous precipitation records (from 1930). We assumed that runoff due to precipitation would provide the primary mobilizing agent to transport locally derived anthropogenic nutrients to the coastal marshes. It was hypothesized that in high-precipitation years, runoff will be higher, and, therefore, nutrient loadings will be higher. A possible argument against this hypothesis is that high runoff could dilute the nitrogen source, decreasing the loadings, or keeping them relatively

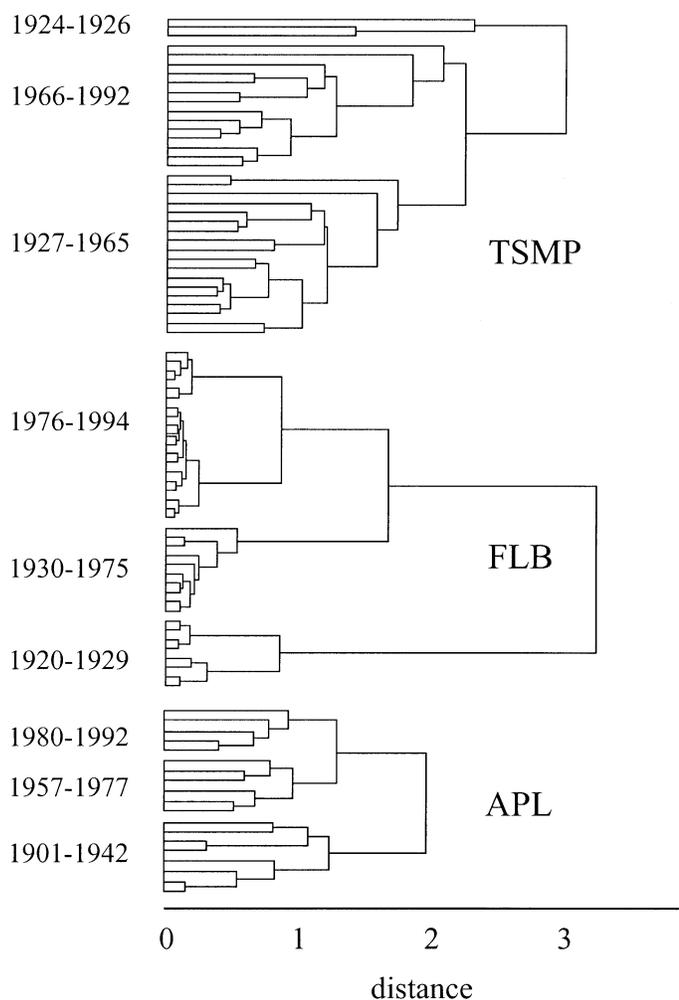


Fig. 2. Complete-linkage cluster analysis dendrograms indicating the different time periods determined from the changing diatom assemblage for each estuary. The distance scale is in Euclidian distance.

constant. However, this argument can be countered by Goolsby and Battaglin (pers. comm.), who demonstrated that the river flood of 1993 (caused by excessive rainfall) did not dilute river nitrate concentrations, but rather increased the flux of nitrate to the Gulf of Mexico. Additionally, Goolsby and Pereira (1996) demonstrated that pesticide and fertilizer concentrations increased during high-precipitation/runoff events, indicating that increased runoff mobilizes more nitrogen, rather than diluting nitrogen levels in the river water. Finally, in low runoff years, other mechanisms of removal (i.e., photolysis, bacterial degradation, etc.) would have more time to operate before the nutrients were washed into the rivers and bayous to be transported to the coastal zone.

The local nitrogen-loading index (LNI) was formulated by multiplying the annual local nitrogen-loading value by the annual precipitation for each year between 1930 and 1994 and then dividing this value by the average product of these variables. This calculation resulted in a unitless measure of local nitrogen inputs (the LNI) that was >1 during years

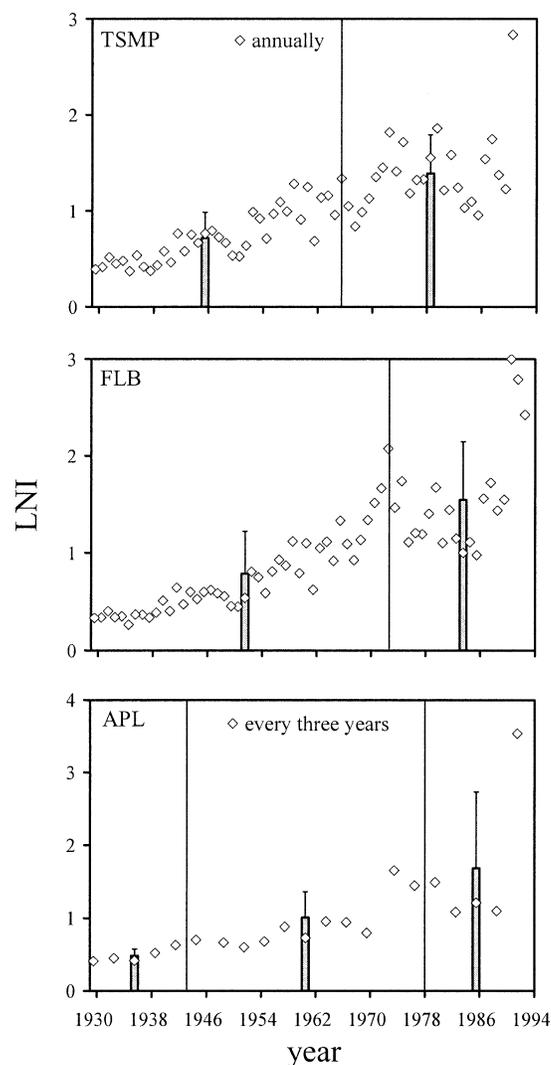


Fig. 3. The local nitrogen-loading index (LNI) for each of the study sites from 1930 to 1994. The diamonds indicate the LNI values determined annually (TSM and FLB) and every 3 yr (APL). The vertical lines in each graph separate the time periods according to the cluster analysis results shown in Fig. 2. The bars with error bars within each period represent the average LNI value (and standard deviation) within each time period.

with nitrogen inputs above the overall average and <1 during below-average years.

Sediment core collection—Three sediment cores were collected at each site with 7.62-cm diameter, 1.3-m long acrylic tubes. The longest sediment core from each site was split the next day into 1-cm increments on a precision core extruder. Visual inspections and length measurements of the core were recorded before and during core extrusion. Core compaction was measured during core extrusion. Each 1-cm increment was homogenized and subdivided for core dating and constituent analyses. Core samples were labeled according to midpoint depth below surface; i.e., the surface sample (0–1 cm) was labeled as “0.5 cm,” the next sample (1–2 cm) as “1.5 cm,” etc. The majority of the subsamples (i.e.,

Table 2. ANOVA results for each indicator tested against the diatom-based time periods. A plus denotes a significantly greater time period mean, a minus denotes a significantly lower mean, a zero denotes an insignificant result, and nd represents no data. Significance was tested against a Bonferroni corrected $\alpha = 0.003$. The variables in bold had consistent ANOVA results over all locations.

Variable	Late 1900s			Mid-1900s			Early 1900s		
	APL	FLB	TSMP	API	FLB	TSMP	API	FLB	TSMP
RNI	+	+	+	0	-	-	-	nd	nd
LNI	0	+	+	0	-	-	0	nd	nd
Si:N (by mole)	0	-	-	0	+	+	0	nd	nd
Fertilizer (% of LNI)	+	+	+	0	0	0	-	0	0
Population (% of LNI)	0	0	0	0	0	0	-	0	-
Cattle (% of LNI)	-	-	-	0	0	0	+	0	0
TP ($\mu\text{g g}^{-1}$ dsw)	+	-	+	0	-	0	0	+	-
Chl a ($\mu\text{g g}^{-1}$ dsw)	+	+	+	0	0	0	0	0	-
Carbon (% dsw)	0	-	0	0	-	0	0	+	0
Nitrogen (% dsw)	0	0	0	0	0	0	0	+	0
BSi (% dsw)	+	+	0	0	0	0	-	-	0
Valves (No. g^{-1} dsw)	+	0	+	-	0	0	-	0	-
BSi (pg valve ⁻¹)	0	0	0	0	0	0	0	0	+
S-W diversity index	+	-	0	0	+	0	-	0	0
TI	+	+	+	0	0	0	0	0	0

APL, Airplane Lake; FLB, Fourleague Bay; TSMP, Terrebonne salt marsh pond; RNI, riverine nitrogen-loading index; LNI, local nitrogen-loading index; TP, total pigments; dsw, dry sediment weight; Chl a, chlorophyll a; BSi, biogenic silica; S-W, Shannon-Weiner; TI, trophic index.

core dating, biogenic silica, CHN, and diatoms) were dried at 60°C and stored prior to analysis, while subsamples collected for pigment analysis were frozen (-80°C) until analyzed. Some of the dried samples (i.e., all but diatoms) were ground in a Wiley mill and then pulverized with a mortar and pestle in preparation for the specific analyses outlined below.

Sediment core dating—Selected core subsamples were dated by ¹³⁷Cs and ²¹⁰Pb following the techniques outlined in Milan et al. (1995) for ¹³⁷Cs and Cutshall et al. (1983) for ²¹⁰Pb using a Princeton Gamma-Tech 60-mm diameter intrinsic germanium “N” type coaxial detector (40% efficiency) interfaced to an EG&G Ortec 92X spectrum master integrated gamma-spectroscopy system. Samples for ²¹⁰Pb analysis were held for 2 weeks before analysis to allow for equilibrium between atmospheric ²²²Rn and ²¹⁰Pb. Samples for ¹³⁷Cs were counted for at least 4 h to yield a counting error of 10% in the vicinity of the 1963/1964 peak, corresponding to the peak ¹³⁷Cs fallout for the southeastern United States (Pennington et al. 1973). Samples for ²¹⁰Pb were counted briefly with a source of ²¹⁰Pb to measure the sample self-absorption potential. The samples were then recounted for 24 h to obtain a significant amount of net counts above background. Additional counts were conducted for ²¹⁴Pb activity so that supported ²¹⁰Pb could be determined. Supported ²¹⁰Pb was subtracted from the total to obtain excess ²¹⁰Pb, from which sedimentation rates were determined. The ¹³⁷Cs-based sedimentation rates were determined by dividing the difference between the date each core was collected and 1963/1964 by the depth where the peak in ¹³⁷Cs activity occurred. Sedimentation rates were determined from the excess ²¹⁰Pb using the constant rate of supply (CRS) model, assuming that there is negligible migration of ²¹⁰Pb in the sediment

and that the supply of excess ²¹⁰Pb is constant over time. Sedimentation rates, therefore, were determined through the inverse of the slope of the least-squares regression line that was fitted to each ²¹⁰Pb profile, resulting in an average sedimentation rate over the length of the sediment core analyzed. We chose this method because we were interested in overall trends, rather than specific, short-term (i.e., [semi]annual) variations in sedimentation.

The ¹³⁷Cs and ²¹⁰Pb estimated sedimentation rates were averaged to give an overall sedimentation rate that would apply for the total length of each sediment core because there are often different estimates for accretion rates using the ¹³⁷Cs or ²¹⁰Pb methods. This difference may be due to the length of time that the ¹³⁷Cs or ²¹⁰Pb methods estimate accretion rates. Sediment volume decreases from organic decomposition, compaction of solids, and dewatering, and each of these processes may change with age. The results of the ¹³⁷Cs method will, thus, tend to yield higher accretion rates than the results of the ²¹⁰Pb method (Kearny et al. 1994; Cochran et al. 1998).

Sediment analyses—Subsamples were analyzed for carbon, hydrogen, and nitrogen sediment content as a measure of sediment organic content. These analyses were conducted on a Control Equipment elemental analyzer model 240-XA (TSMP and APL), and a Perkin-Elmer 2400 CHNS/O series II analyzer (FLB). Subsamples were analyzed for biogenic silica content by using a time-digestion method adapted from DeMaster et al. (1983). Sediment pigment analyses were conducted using 90% acetone as an extracting agent over a 24-h period at 4°C. The concentrations of chlorophyll a (Chl a) and phaeopigments were measured on a Turner Type-10 fluorometer.

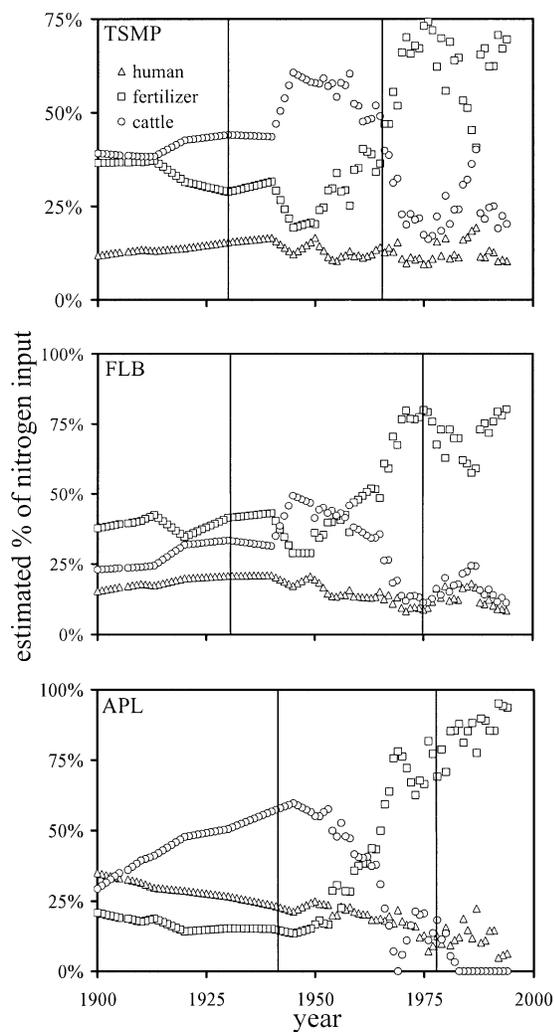


Fig. 4. The percentage breakdown of the source of nitrogen in each localized watershed: TSMP, FLB, and APL.

Diatom analysis—Subsamples were prepared for diatom analysis according to Parsons et al. (1999). At least 250 valves were counted per sample at $\times 750$ on a Zeiss Universal microscope using Nomarski illumination. Relative abundance data were determined for each diatom species by dividing the number of valves counted for a particular species by the total number of diatom valves counted for a particular sample. Diatom species were identified to their lowest possible taxon and assigned a trophic classification according to a previous literature review (summarized in Parsons 1996). Three broadly defined trophic classifications were defined (adapted from McCormick and Cairns 1994): eutrophic (nutrients are not limiting), mesotrophic (nutrients are sometimes limiting), and oligotrophic (nutrients are often limiting). The trophic groupings were then used to compute a diatom-based trophic index (TI), which was calculated as a ratio of the relative abundance of the eutrophic diatoms to oligotrophic diatoms. Diatom counts were initially conducted at 10-cm core-depth intervals to determine whether the diatom assemblage was changing within these periods.

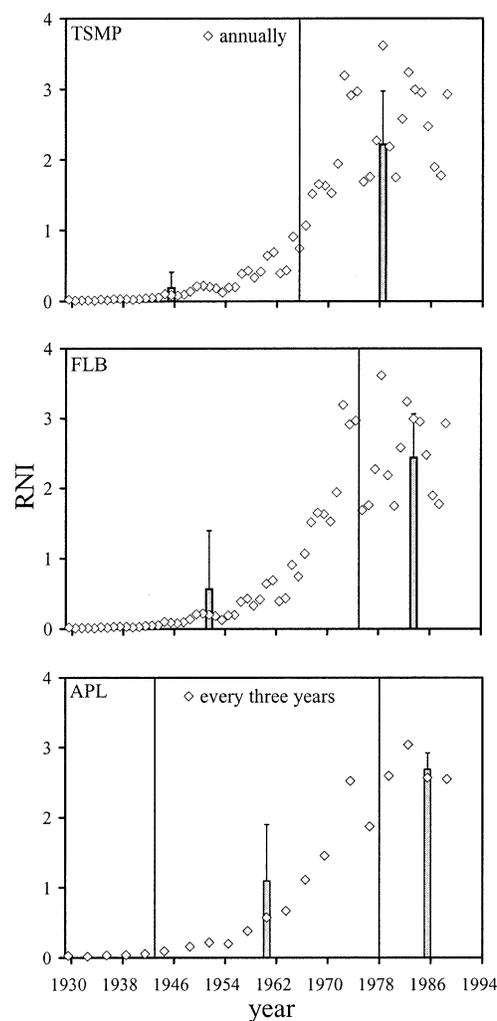


Fig. 5. The riverine nitrogen-loading index (RNI) for each of the three study sites from 1930 to 1994. The vertical lines in each graph separate the time periods according to the cluster analysis results shown in Fig. 2. The bars with error bars within each period represent the average RNI value (and standard deviation) within each time period.

Counts were then completed at 5-cm sections to improve the overall resolution of the analysis. Intensive 1-cm counts were completed where the assemblage changed within 5-cm intervals.

Statistical analyses—A complete-linkage cluster analysis was applied to the diatom assemblages (i.e., the relative abundance data) of each depth for each site to determine whether the diatom assemblage was significantly changing and when. Depth was not used as a variable in the cluster analysis, which rather relied solely on the diatom relative abundance data. Therefore, the clusters that were joined and the subsequent dendrograms that were built were based solely on the similarities/differences between the diatom assemblages from the different samples. The analysis was limited to those species that had a relative abundance of at least 1.5% in any sample to simplify the analysis. Complete-link-

Table 3. Summary of the Pearson correlation analysis results between the LNI and RNI versus the sediment-derived data. All variables except the trophic index (TI) are based on dry sediment weight (dsw). Pearson correlation coefficients (r) are the top number in each cell, and p values are the bottom number. Significance was tested against a Bonferroni corrected $\alpha = 0.003$.

Variable	TSMP		FLB		APL	
	RNI	LNI	RNI	LNI	RNI	LNI
Biogenic silica (% dsw)	0.719 0.000	0.691 0.000	0.066 0.615	0.810 0.000	0.589 0.006	0.426 0.054
Carbon (% dsw)	-0.552 0.000	-0.620 0.000	-0.468 0.000	-0.476 0.000	0.610 0.004	0.597 0.004
Nitrogen (% dsw)	0.120 0.360	-0.065 0.621	-0.374 0.003	-0.431 0.000	0.896 0.000	0.752 0.000
Chlorophyll a ($\mu\text{g g}^{-1}$ dsw)	0.683 0.000	0.680 0.000	0.597 0.000	0.650 0.000	0.850 0.000	0.786 0.000
Total pigments ($\mu\text{g g}^{-1}$ dsw)	0.565 0.000	0.522 0.000	-0.155 0.238	0.037 0.774	0.888 0.000	0.840 0.000
TI	0.573 0.001	0.246 0.181	0.611 0.001	0.642 0.000	0.732 0.004	0.562 0.036

TSMP, Terrebonne salt marsh pond; FLB, Fourleague Bay; APL, Airplane Lake; RNI, riverine nitrogen-loading index; LNI, local nitrogen-loading index.

age builds the tree by combining clusters (i.e., each depth sample) according to the maximum differences between their variables (i.e., the diatom assemblages), thereby providing greater separation and a more open, spread-out dendrogram. The cluster analysis was conducted using the SAS statistical program (SAS 8.2), and the dendrograms were constructed using the gtree.sas macro program (Friendly pers. comm.).

The diatom-derived cluster groupings produced by the cluster analysis provided a means to divide the time frame represented by each sediment core into distinct time periods. An analysis of variance (ANOVA) was then used to determine how the productivity indicators (e.g., RNI, LNI, Si: \times N, biogenic silica, etc.) changed over the time periods. Significant differences in these variables versus the time periods were then used to determine (1) whether the diatom-based clusters reflected changing nutrient regimes and (2) how the nutrient regimes changed.

A Pearson's correlation analysis was conducted on the sediment-based data (biogenic silica, percentage carbon, percentage nitrogen, total pigments, Chl a , TI, and the relative abundance of individual diatom species) for each of the three sites to determine whether they were significantly and positively correlated with the RNIs and LNIs, thus indicating the potential role of these variables as historical indicators of nutrient inputs. Bonferroni α corrections were applied to the results to protect against Type I error. Together, the ANOVA and correlation results were then used to assess whether nutrient increases (and subsequent eutrophication) were evident in these salt marshes. All statistical analyses were computed using the Minitab 13.0 and SAS 8.2 statistical programs.

Results

Sediment core collection and dating—Detailed core dating results, presented in an earlier publication (Parsons et al. 1999), are summarized in Table 1. Compaction rates were

negligible (<10%). Sedimentation rates were determined to be 0.32 cm yr⁻¹ for APL and 1.0 cm yr⁻¹ for both TSMP and FLB. The cores contained a record of 68 yr (TSMP) to a potential period of 330 yr (if the whole APL core had an average sedimentation rate of 0.32 cm yr⁻¹). Since we only concentrated our analyses in the top 30 cm (92 years) of the APL core, however (*see below*), we will not make this assumption. The ¹³⁷Cs and ²¹⁰Pb sedimentation rates were in good agreement for APL but differed by 33% for the other two cores (*see earlier comments in methods section*).

Diatom analysis—A total of 33,726 diatom valves representing 205 species from 60 genera were encountered in 101 samples analyzed from the three sites. Twenty nine of these 205 species were >1% relative abundance in any sample and could also be assigned a trophic classification. The proportion of valves counted that could be classified ranged from a low value of 36% from TSMP-68 cm to a high of 87% from FLB-30 cm. Overall, an average of 67% of the valves identified and enumerated could be assigned a trophic classification.

A total of 37 samples were counted from FLB, 37 from TSMP, and 27 from APL. A paucity of diatoms in samples deeper than 30 cm in the APL core resulted in counts of fewer than 250 diatom valves. These deeper samples were therefore omitted from statistical analyses because of concern for inadequate representation of the diatom assemblages from these samples due to the low counts. Results from the complete-linkage cluster analysis (Fig. 2) demonstrate that the diatom assemblages shifted twice over the last century, resulting in three time periods in each of the three cores: early 1900s (pre-1930s/1940s), mid-1900s (1930s/40s to 1960s/1970s), and late 1900s (1960s/70s to 1990s).

Nutrient inputs—Local sources of nitrogen showed similar increasing trends in all three watersheds (Fig. 3). The ANOVA results indicate that the LNI is significantly higher

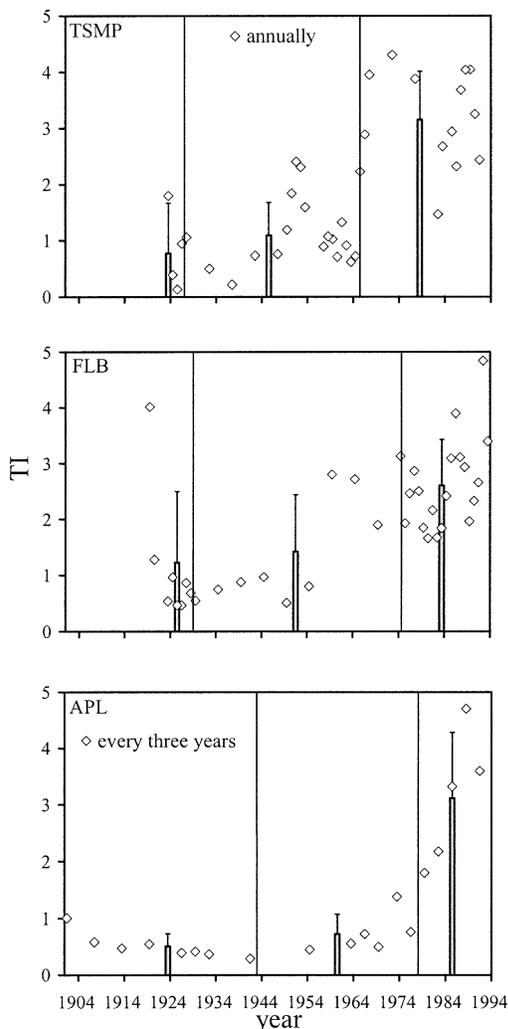


Fig. 6. The diatom-based trophic index (TI) for each of the three study sites. A higher TI value indicates that a greater proportion of diatom species are eutrophic species. The vertical lines in each graph separate the time periods according to the cluster analysis results shown in Fig. 2. The bars with error bars within each period represent the average TI value (and standard deviation) within each time period.

in the late 1900s versus the earlier time periods for all sites except APL (Fig. 3, Table 2). This is due in part to the Bonferroni α correction, which lowered the p value threshold to 0.003 to protect against type I errors (the LNI ANOVA p value for APL was 0.01). Fertilizer application became the most prominent source of anthropogenic nitrogen in the late 1950s and early 1960s (Fig. 4). The results from the ANOVA also indicate that fertilizer accounted for a significantly larger portion of anthropogenic nitrogen in the late 1900s versus the earlier time periods for all three sites (Table 2). Conversely, the human and cattle population constituted a decreasing proportion of the nitrogen inputs over the time periods (Table 2, Fig. 4). The RNI generally increased from the 1950s to the 1980s and remained relatively constant into the 1990s (Fig. 5). The results indicate that the RNI was

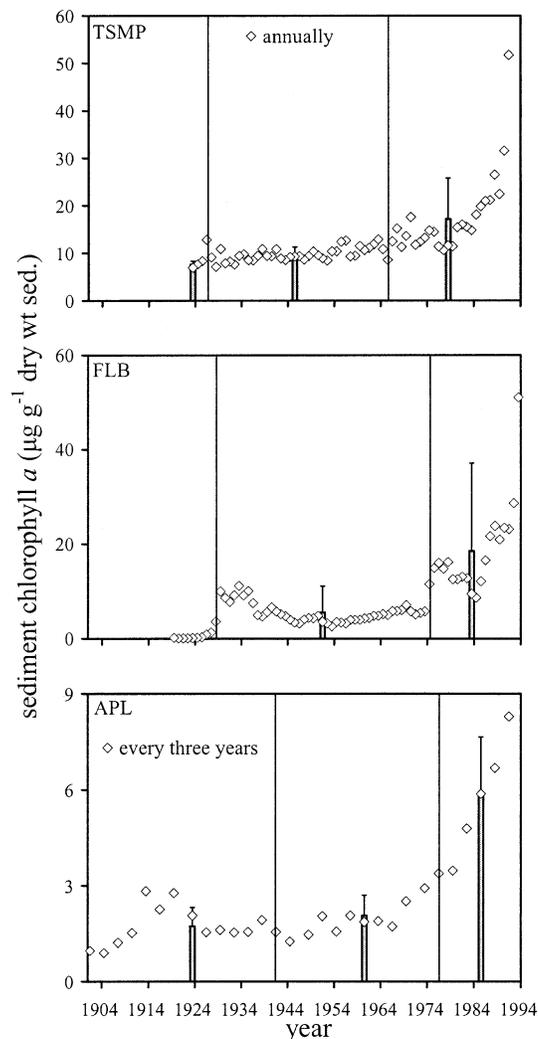


Fig. 7. Sediment Chl a concentrations ($\mu\text{g g}^{-1}$ dry weight sediment) for each of the three study sites. The vertical lines in each graph separate the time periods according to the cluster analysis results shown in Fig. 2. The bars with error bars within each period represent the average Chl a value (and standard deviation) within each time period.

significantly higher in the late 1900s versus the earlier time periods for all three sites (Table 2).

Sediment core trophic indicators—Two of the sediment-based parameters displayed significant differences between time periods according to the ANOVA results (Table 2) and were significantly and consistently correlated with the RNI and/or LNI over all three study sites (Table 3): the diatom-based trophic index and Chl a . Both variables increased significantly upcore at all three sites (Figs. 6, 7) and were significantly higher in the late 1900s compared with the earlier time periods (Table 2).

Eleven diatom species were significantly correlated ($p < 0.0007$) with the LNI and/or RNI (Table 4). The most significant results were for *Amphora copulata* Giffen and *Navicula yarrensii* Grunow, both of which were negatively cor-

Table 4. Diatom species displaying a statistically significant correlation (using a Bonferroni corrected $\alpha = 0.0007$) with either the LNI (L) or RNI (R). The correlated index is indicated (LNI or RNI), Pearson correlation coefficients are given (r), and p values are noted.

Species	LNI or RNI?	r	p value
<i>Achnanthes brevipes</i> var. <i>intermedia</i> (Kützing) Cleve	RNI	-0.41	0.0006
<i>Amphora acutiuscula</i> Kützing	RNI	0.50	<0.0001
<i>Amphora copulata</i> Giffen	both	-0.45 (L), -0.58 (R)	<0.0001
<i>Bacillaria paradoxa</i> Gmelin	LNI	0.40	0.0004
<i>Caloneis westii</i> (W. Smith) Hendey	RNI	-0.51	<0.0001
<i>Diploneis didyma</i> (Ehrenberg) Ehr.	RNI	-0.50	<0.0001
<i>Navicula consentanea</i> Hustedt	RNI	0.42	0.0003
<i>Navicula yarrensensis</i> Grunow	both	-0.51 (L), -0.62 (R)	<0.0001
<i>Nitzschia proxima</i> Hustedt	LNI	0.42	0.0002
<i>Nitzschia pubens</i> Cholnoky	LNI	0.45	<0.0001
<i>Tryblionella hungarica</i> (Grunow) Mann	LNI	0.42	0.0003

LNI, local nitrogen-loading index; RNI, riverine nitrogen-loading index.

related with both the LNI and RNI ($p < 0.0001$). Both species decreased in abundance over time (Fig. 8).

Discussion

This study demonstrates that sediment-based reconstructive studies are possible in a dynamic, shallow-water marsh environment, and that the results of such a study can provide useful data for addressing current water quality issues. The evidence presented here suggests that eutrophication is a recent phenomenon in Louisiana salt marshes, occurring primarily during the decade of the mid-1960s and mid-1970s. The diatom assemblages of all three sites exhibited two shifts (Fig. 2), which, when coupled with the RNI, LNI, and trophic indicator results (Figs. 3–8), represent changes from oligotrophic to mesotrophic conditions (between the late 1920s to early 1940s, site dependent), and from mesotrophic to eutrophic conditions between the mid-1960s and mid-1970s. The timing of the shift to eutrophic conditions is consistent with a global increase in the symptoms of eutrophication from the 1960s to the 1980s (Boesch 2002). At least two diatom species (*Amphora copulata* and *Navicula yarrensensis*) appear to be candidate indicator species of oligotrophic conditions (Fig. 8), although results suggest that nine others may also be useful indicator species (six for eutrophic conditions, three others for oligotrophic conditions, Table 4).

The increased nutrient load during the shift from mesotrophic to eutrophic conditions appears to be due to the increased use of fertilizer, as indicated by the predominance of fertilizer in the LNI in the 1960s (Figs. 3, 4) and the higher RNI since the 1960s (Fig. 5). Other studies have demonstrated that higher amounts of fertilizer consumption were most likely the cause of the changes in nitrogen concentrations in the mainstem of the Mississippi River (Turner and Rabalais 1991; McIsaac et al. 2001) and of recent (advanced) eutrophication in other estuaries (Boesch 2002).

Similar changes in the coastal ecosystem adjacent to the Mississippi River are directly linked to the changes within the watershed and nutrient loading to the continental shelf,

especially from nitrate, which tripled in the last half of the 20th century (Turner and Rabalais 1994; Rabalais et al. 1996, 2002). Evidence from long-term data sets and the sedimentary record demonstrates that indicators of increased marine productivity and subsequent worsening of oxygen stress are highly correlated with historic increases in riverine dissolved inorganic nitrogen concentrations and loads over the last 50 yr (Rabalais et al. 2002, 2004; Turner et al. 2004). Evidence comes in long-term changes in Secchi disk depth and diatom productivity, increased accumulation of diatom remains and marine-origin carbon in sediments, changes in phytoplankton pigment concentrations in sediments, and surrogates for worsening oxygen conditions in the sediments (e.g., glauconite abundance, benthic foraminiferan diversity and community composition, and ostracod diversity).

While increased applications of fertilizer appear to be causally related to eutrophication in these salt marshes, it is unclear whether the fertilizer causing these changes is mostly from local origins or from the Mississippi River. Several studies have demonstrated that enough freshwater from the Mississippi River enters through tidal passes to affect salinity patterns in these salt marshes (e.g., Parsons et al. 1999). It is possible, therefore, that riverine processes (i.e., the RNI) dominate over local processes (i.e., the LNIs) because a riverine signal is evident in the salt marsh sediments. Some of the sediment core trophic indicators used in this study, however, display a significant positive relationship with the LNI versus the RNI (e.g., biogenic silica at FLB, Table 3); indicating that local sources of nitrogen should not be discounted.

The state of Louisiana and the U.S. Army Corps of Engineers are studying the effectiveness of river diversions to increase riverine inputs of nutrients and suspended sediments to the coastal marshes. Discussion about the efficacy of diversions to conserve and restore coastal wetlands has centered on whether these diversions will increase sediment accretion and aboveground productivity enough (Martin 2002) to outweigh any detrimental effects (e.g., eutrophication, noxious algal blooms, HABs) of the diversions.

Our study indicates that the salt marshes of Louisiana are

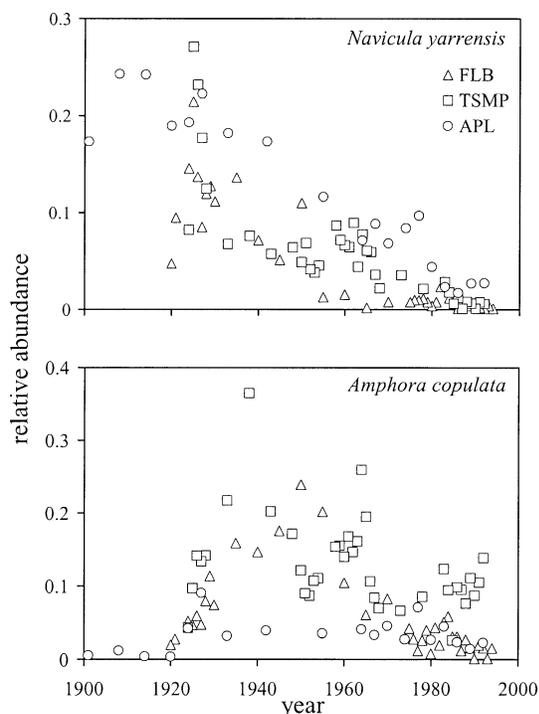


Fig. 8. The relative abundance of (a) *Navicula yarrensii* and (b) *Amphora copulata* at the three study sites.

already eutrophic and that the nutrient inputs from riverine inputs are a dominant factor in the progression of eutrophication. Additional nutrient inputs may exacerbate the present eutrophic state. While nitrate concentrations might decline as river water passes through the marsh environment, ammonium, total organic nitrogen (TON), and phosphate concentrations may increase, creating a different, yet still significant nutrient loading issue. For example, some HAB species are thought to be stimulated by ammonium and/or TON (Anderson et al. 2002). The coastal waters of Louisiana may be phosphate-limited during the spring (Rabalais 2002; Q. Dortch unpubl. data), and the outwelling of remineralized phosphate from the estuaries may remove this limitation resulting in additional primary production. Lastly, there is some question as to the threshold at which the marshes will become N saturated (Vitousek et al. 1997). Since the marshes are already showing evidence of eutrophication, it is possible that additional nutrient inputs may lead to rapid saturation, reducing the potential benefits of nitrate use. Further, nutrient additions to salt marshes have been shown to decrease soil organic matter accumulation (Morris and Bradley 1999, Table 3), which may create the opposite of the desired effect (i.e., the destruction rather than formation of marsh).

Six diversion structures are currently in operation on the lower Mississippi River. The largest of these is the Davis Pond structure, which is scheduled to release up to $25 \text{ m}^3 \text{ s}^{-1}$. The Davis Pond diversion became operational in January 2004, releasing $\sim 17 \text{ m}^3 \text{ s}^{-1}$ on a continual basis, but it is not yet up to its anticipated capacity. The nutrient loading effects of this diversion, however, were evident within the first half month of operation. The inflow of nutrient-rich Mississippi River water (with 40-fold higher nitrate concen-

trations vs. ambient shallow-water conditions) resulted in an eightfold increase in Chl *a* biomass in Lake Cataouatche, which received the diverted water after it passed through a 2,800-ha marsh (Rabalais unpubl. data). Such preliminary results indicate that water column productivity was enhanced by the diversion and that the marsh did not significantly remove the additional nutrient load of the diversion.

Evidence from earlier studies in the area suggests that diversions of high-nutrient waters will not only enhance primary productivity but also cause changes in phytoplankton species composition so that noxious, and even potentially toxic species, will become more abundant (Dortch et al. 1999, 2001; H. Rick pers. comm.). For example, a toxic cyanobacterial bloom occurred in Lake Pontchartrain in 1997 as a result of diversion of Mississippi River water through the Bonnet Carré diversion (Dortch et al. 2001). In addition, other harmful algal species have been documented from throughout the Barataria Bay system (Dortch et al. 1999, 2001), demonstrating that HAB species are present and may be stimulated by the introduction of the diverted riverine waters. *Pseudo-nitzschia* spp., for example, appear to be stimulated by both higher nitrate levels and low Si: \times N (Parsons et al. 2002), conditions typical of Mississippi River waters (Turner and Rabalais 1991). It is possible that toxigenic *Pseudo-nitzschia* spp. could be stimulated by the river diversions, especially if the estuaries become N saturated and if silica is removed by diatoms nearer the diversion.

Diverting Mississippi River water into the marshes will increase the nitrogen loading substantially. Eutrophication will increase further, and silica limitation could become prominent, possibly leading to a shift in the primary producers away from silica-dependent diatoms toward silica-independent flagellates and cyanobacteria (Officer and Ryther 1980). The resulting phytoplankton community shift would be detrimental to the health of the ecosystem, resulting in poorer food quality for the zooplankton and fish populations. Harmful algal blooms could increase in frequency, resulting in more frequent fish kills and possible toxic events affecting human health.

The initial hypotheses that the three study sites were not eutrophic in the past and that eutrophication is directly linked with anthropogenic activities that increase nutrient inputs to the marsh are supported by the results of these analyses. Increased fertilizer use, either from local or riverine sources, appears to be most responsible for the development of eutrophication. Further nutrient inputs are likely to exacerbate the effects of eutrophication in the marshes (i.e., noxious algal blooms, more HABs, poorer food quality for primary consumers), which may offset to an unknown degree the benefits of river diversions to the marshes. Therefore, while river diversions may, or may not, help reduce marsh loss or restore marshes, the accompanying nutrient loading could be detrimental and should be considered in restoration plans.

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