

Reassessing Hypoxia Forecasts for the Gulf of Mexico

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Gulf of Mexico hypoxia has received considerable scientific and policy attention because of its potential ecological and economic impacts and implications for agriculture within its massive watershed. A 2000 assessment concluded that increased nitrate load to the Gulf since the 1950s was the primary cause of large-scale hypoxia areas. More recently, models have suggested that large-scale hypoxia did not start until the mid-1970s, and that a 40–45% nitrogen load reduction may be needed to reach the hypoxia area goal of the Hypoxia Action Plan. Recently, USGS revised nutrient load estimates to the Gulf, and the Action Plan reassessment has questioned the role of phosphorus versus nitrogen in controlling hypoxia. In this paper, we re-evaluate model simulations, hindcasts, and forecasts using revised nitrogen loads, and test the ability of a phosphorus-driven version of the model to reproduce hypoxia trends. Our analysis suggests that, if phosphorus is limiting now, it became so because of relative increases in nitrogen loads during the 1970s and 1980s. While our model suggests nitrogen load reductions of 37–45% or phosphorus load reductions of 40–50% below the 1980–1996 average are needed, we caution that a phosphorus-only strategy is potentially dangerous, and suggest it would be prudent to reduce both.

Introduction

Oxygen depletion can have deleterious ecological effects in coastal waters and has been receiving considerable scientific and policy attention (1–3). In the northern Gulf of Mexico, regions where oxygen concentrations are below 2 mg/L (hypoxia) have averaged over 15 600 km² since 1993, making it one of the largest hypoxic areas in the world (4, 5).

In 2001, an intergovernmental task force delivered to the President and the Congress an Action Plan (6) designed to reduce the 5-year running average hypoxic zone to less than 5000 km². The Action Plan was based primarily on a 2000 integrated assessment that evaluated the causes, consequences, and potential remedies for Gulf hypoxia (2, 5), and called for a reassessment of progress every five years. This paper addresses two important considerations that arose during the reassessment process that began in 2005 (7): (1) USGS revised estimates for nutrient loads to the Gulf (8) and (2) evidence suggesting that primary production in the Gulf may now be limited by phosphorus instead of nitrogen, especially in spring (9–12).

Limited monitoring, a highly variable ocean climate, complicated trophic dynamics, and complex biogeochemical

cycles can make empirical determination of the causes of hypoxia difficult. Detailed models have played an important role in interpolating across missing data and interpreting observations (13–16), and statistical models have closely linked nitrate flux and other constituents to the size of Gulf hypoxia (17). A simple, biophysically based model has been used to hindcast and forecast Gulf hypoxic areas in response to changes in nutrient loads (18, 19), continental precipitation (20), and nitrogen from offshore produced-water discharges by oil rigs (21). This simpler model has also been used to hindcast and forecast the effects of nutrient loads on Chesapeake Bay hypoxia (22).

Scavia et al. (18, 19) and Turner et al. (23) suggest that historical nitrogen loads were not sufficient to produce significant areas of oxygen-depleted bottom waters before the mid 1970s. Scavia et al. predicted areas of hypoxia between 5000 and 10 000 km² in 1973–1975, with significant large-scale regions occurring between 1978 and 1985. Although there were no comprehensive shelf-wide cruises during that period, these model estimates are consistent with the existing, albeit limited, empirical records (47). Scavia et al. (18) also examined the response of the model to the 30% nitrogen load reduction recommended in the Action Plan. They concluded that a 40–45% reduction might be necessary to achieve the goal of reducing the extent of hypoxia to 5000 km² or less. Donner and Scavia (20) also used this model to assess what load reductions might be necessary under anticipated future climate fluctuations.

The purpose of this paper is to use that previously developed and calibrated model to evaluate the effects of new USGS load estimates and to assess the potential for phosphorus control of hypoxia dynamics for both current interannual variability and long-term trends.

Methods

Nutrient Flux Estimates. We use the new USGS estimates for two locations at the bottom of the drainage basin: the Atchafalaya River at Melville, LA (MELV), and the Mississippi River main stem at St. Francisville, LA (STFR). These load estimates take into account regulation of water through the Old River Control Structure, initially completed in 1963. Since 1977, the Structure has prevented the Atchafalaya River from capturing the flow of the Mississippi by maintaining roughly 30% of the total Mississippi River flow down the Atchafalaya (24).

The revised USGS flux data (8) include nitrate plus nitrite (referred to herein as NO₃), total Kjeldahl nitrogen (TKN), and total phosphorus (TP). We calculated total nitrogen (TN) as the sum of NO₃ and TKN. The USGS Regression Method uses a seven-parameter regression to estimate daily nutrient flux, summed across the time period of interest (monthly or yearly). A second method, the Composite Method, combines the regression method with a period-weighted approach that takes into account residual concentrations, defined as regression-model predicted concentration minus observed concentration. It does this by applying the residual concentrations to the fluxes for days when samples are not collected. The Composite Method is considered more accurate than the Regression Method; however, Composite Method values were not reported for water years that did not have at least ten samples per year. Therefore, we used the Composite Method data whenever possible and Regression Method estimates for years in which the composite data were not available. Strong linear correlations between the two estimates support this choice.

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As in the original application (18) and consistent with a number of regression-based studies (17, 23), May–June spring loads are effective drivers of modeled summer hypoxia. For STFR, May–June NO₃ loads were available for water years 1968–2006, TKN loads were available after 1975, and TP loads were available for 1975–2006. May–June MELV NO₃ loads were available for 1979–2006, TKN loads were available after 1980, and TP loads were available for 1979–2006.

For flux estimates prior to 1968, we relied on the data set used in the original model application, as reported by Goolsby et al. (25, 26). However, because their STFR flux estimates included Old River Diversion flux estimates, we used streamflow data from the Army Corps of Engineers for the Mississippi River and the Old River Diversion to correct flux and streamflow estimates for the Mississippi (27).

Reconstructing TN and TP Spring Load Histories. For hindcasts, we reconstructed TN and TP spring load histories from 1955 to 2006. As in the original model application, we took advantage of strong correlations between TKN and NO₃ to estimate TKN (Figure S1a, Supporting Information) for the period 1955–1974 at STFR. We used the corrected NO₃ flux estimates from Goolsby et al. (25) for 1955–1967 and from Aulenbach et al. (8) thereafter:

$$\text{TKN} = 11.971\text{NO}_3^{0.7148} \quad (R^2 = 0.75; n = 700)$$

We used two methods to complete the MELV TN record. In the first case, we used a regression relating MELV and STFR loads (Figure S1b, Supporting Information):

$$\text{TN}_M = 0.8039\text{TN}_S^{0.9258} \quad (R^2 = 0.93; n = 320)$$

This method assumes that the relationship between the Mississippi and Atchafalaya flows remained consistent between the periods 1955–1967 and 1968–2006; however, while that stabilized after 1977, it varied between 15 and 40% between 1963 and 1977. For comparison, we calculated MELV TN flux from STFR TN flux by assuming TN concentration was the same at both locations, and then estimated nutrient flux down the Atchafalaya (MELV) by multiplying the Mississippi flux (STFR) by the ratio of the discharges at the two locations. The results of the two methods are shown in Figure 1a, along with comparison to measured TN flux at MELV for 1979–1996. We used the mean of the two estimates for the MELV TN reconstruction. The resulting trends for TN load (Figure 1b) show a significant increase between 1968 and the late 1980s, followed by significant interannual variability with no trend thereafter.

To construct TP flux estimates, we used two scenarios for years without measurements. The first scenario assumed that TP load is only a function of streamflow because, unlike TN, there is no significant trend in measured TP loads between 1975 and 2006 (Figure 1c), indicating that most of the interannual variability was due to changes in flow. Under this scenario, we estimated TP load from 1955 to 1968 from the following regression based on flow and TP loads from Aulenbach et al. (8) (Figure S1c, Supporting Information):

$$\text{TP} = 0.2167\text{Flow}^{1.0912} \quad (R^2 = 0.82; n = 700)$$

Because flow data were not available between 1955 and 1968 for MELV, we used the following regression to estimate TP at MELV from STFR (Figure S1d, Supporting Information):

$$\text{TP}_M = 0.5239\text{TP}_S^{0.9678} \quad (R^2 = 0.76; n = 332)$$

While it is most likely that TP load did not increase significantly between 1955 and 1968 (see discussion), we included a second scenario for comparison. Here we assume that TP load would still vary with streamflow as above, but that the loads increased by a factor of 2 since 1955 to reach

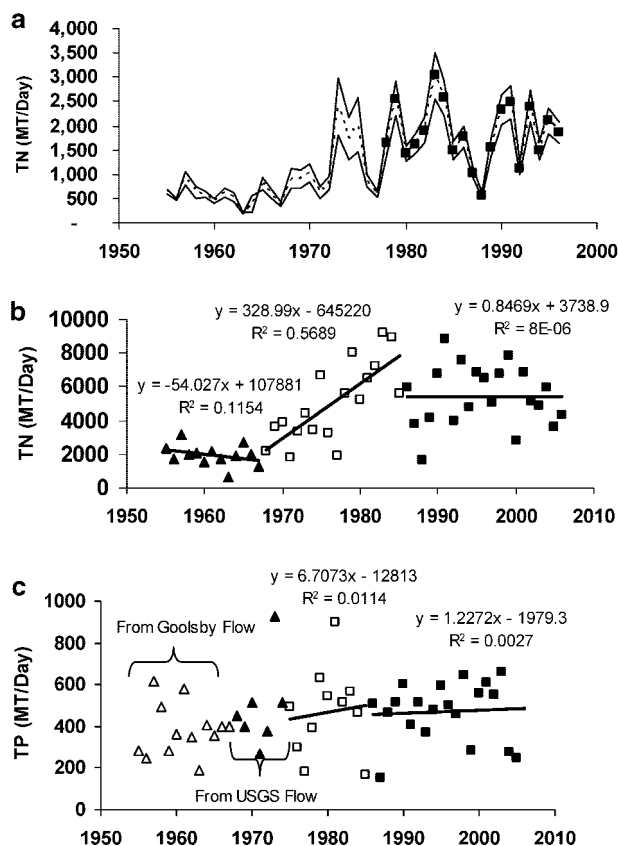


FIGURE 1. Estimates of Atchafalaya TN flux using two different methods (solid lines) compared to measurements between 1979 and 1996 (a), and reconstructed TN (b) and TP loads (c). Dashed line represents the mean of the two methods.

the average load from the first 10 years of record (i.e., 1975–1984 for STFR and 1979–1988 for MELV).

The resulting reconstructed 1955–2006 TP and TN loads are shown in Figure 2. We recognize the uncertainty associated with these reconstructed loads, but because their primary purpose is to explore long-term trends, not to provide precise annual forecasts, we believe they are appropriate for this analysis.

Hypoxia Data. Hypoxia spatial extent estimates (Figure 3) are from northern Gulf of Mexico cruises conducted between mid-July and mid-August and cross-shelf transects annually since 1985 (excluding 1989) (4, 5). Interpolation schemes are described in the Supporting Information.

The Model. The model (18) is an adaptation of the Streeter–Phelps river model that predicts oxygen concentration downstream from point sources of organic matter loads using mass balance equations for oxygen-consuming organic matter (B), in oxygen equivalents (i.e., BOD), and dissolved oxygen deficit (D). Assuming no upstream oxygen deficit, and ignoring longitudinal dispersion, the model's steady state solution for D is

$$D = [a/(b - a)B_0(e^{-ax/v} - e^{-bx/v})]$$

where B_0 is the oxygen demand at the source, t is time (days), x is distance downstream (km), a is the first-order decay constant (day^{-1}), b is the first-order rate constant for vertical oxygen flux (day^{-1}), and v is a combination of net downstream advection and a calibration term (km day^{-1}). Organic matter load (B_0) was approximated by multiplying TN or TP loads by the Redfield ratio to convert nitrogen or phosphorus to algal carbon (5.67 gCgN^{-1} ; 41 gCgP^{-1}), and by assuming an oxygen equivalent (e.g., respiratory ratio) of

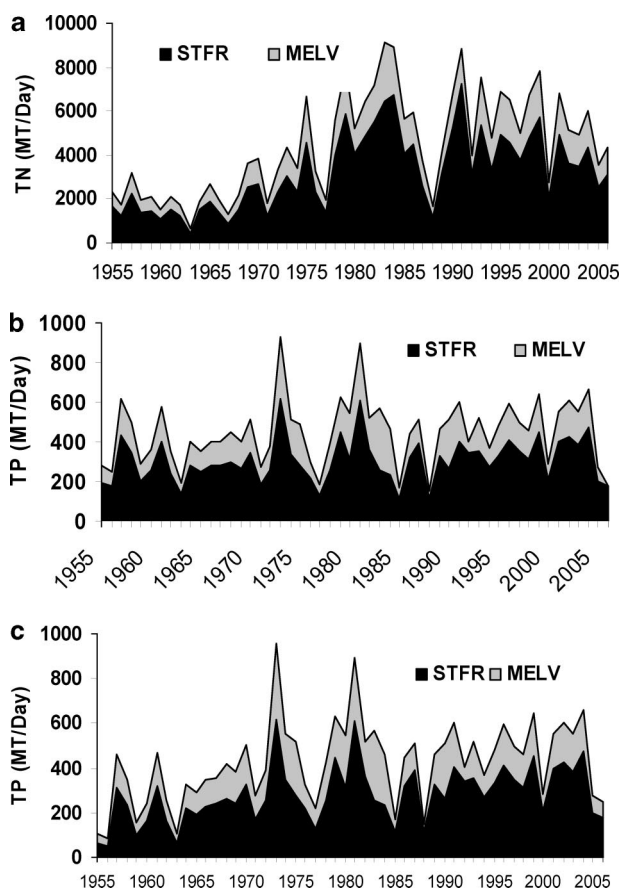


FIGURE 2. Reconstructed nutrient loads for the Mississippi and Atchafalaya Rivers for TN (a), TP assuming no long-term trend (b), and TP assuming loads doubled since 1955 (c).

3.47 gO₂gC⁻¹. As in the original application, we assumed 50% of the Mississippi River load moved east or offshore and did not contribute to hypoxia development (28, 29), and that 50% of the surface algal production settled below the pycnocline (30, 31). Scavia et al. (18) described the validity of the model's assumptions, including the use of May–June loads as a substitute for organic matter below the pycnocline, the one-dimensional flow of subpycnocline water movement, and the modeling of subpycnocline oxygen consumption and cross-pycnocline oxygen flux as first-order processes (see Supporting Information).

Calibration. Holding coefficient values for organic matter decay (*a*) and vertical oxygen flux (*b*) from the original application (0.003 and 0.01 day⁻¹) constant across all years and treating the mouths of the Mississippi and Atchafalaya rivers as point sources, we calculated summer steady state, subpycnocline oxygen concentrations at different distances from the point sources. We estimated hypoxic zone length by summing segments with concentrations below the definition of hypoxia and converted it to hypoxic area with a regression of area and length (slope = 38.84; *R*² = 0.82; Figure S2, Supporting Information). Using the nutrient load from each year between 1985 and 2006, the model was calibrated to the measured hypoxic zone areas by varying only one coefficient, *v*. In the original Streeter–Phelps formulation, *v* represents net downstream advection. However, because in this application it is the only calibration term, it also represents all unmodeled process and associated uncertainties. Collapsing all uncertainties into this one coefficient allows us to characterize it effectively in the Monte Carlo simulations described below.

Results

Model Calibration. The recalibrated model, driven with the new TN loads, is similar to the original application (Figure S3, Supporting Information). We were able to explain virtually all of the interannual variability for the 1985–2006 calibration by adjusting only *v*, and the resulting set of calibration terms had mean and standard deviation similar to those used in the original application (0.56 ± 0.20 vs 0.53 ± 0.16).

We were also able to explain virtually all of the interannual variability in hypoxic area with the model driven by TP loads (Figure S3, Supporting Information), but only after assuming the P recycle rate was twice the N recycle rate by doubling the initial BOD estimate after making the conversions described above. Without that adjustment, the model was not able to produce hypoxia in several of the years (1985, 2000, 2005, 2006) for any values of *v*, and it required using significantly larger values (0.96 ± 0.39) to match observations in the other years. By assuming P cycles twice as fast as N, the calibration terms were indistinguishable from those used in the N calibration (0.51 ± 0.32). We discuss the justifications and implication of doubling the recycle rate below.

Monte Carlo Simulations. Because all uncertainty associated with the model structure, coefficients, and inputs is accumulated in the calibration term, and because it is not possible to know what that value was in the past or will be in the future, we captured that uncertainty through Monte Carlo simulations. For each hindcast and forecast, we ran the model 1000 times with values of *v* drawn from a normal distribution with mean and standard deviation of the combined set of 40 *v* values used in the 1985–2006 calibrations with TN and TP loads (0.52 ± 0.27).

Hindcasts. We ran Monte Carlo simulations for 1955–2006 using reconstructed TN and TP loads. For the model driven by TN (Figure 3a), most of the 1985–2006 observations fall within the error bounds of the model output and, as in Scavia et al. (18), the hindcasts indicate that large-scale hypoxia did not begin until the mid-1970s. After that time, the hypoxic zone increased until the 1980s, and has been rather steady with significant interannual variability ever since. For the TP scenarios (panels b and c in Figure 3), most of the 1985–2006 observations also fall within the error bounds of the model output. However, unlike Scavia et al. (18) and the TN-driven results herein, TP-driven hindcasts produce significant large-scale hypoxic zones in the 1950s and 1960s.

Required Load Reductions. On the basis of the calibration to the new TN loads, we recalculated load reductions needed to meet the Action Plan goal to reduce hypoxia to 5000 km². The results were similar to those presented in the original work (18, 19); to meet that goal in an average year requires a TN load reduction of approximately 37% from the 1980–1996 average. To meet that goal in most years requires approximately a 45% reduction. Similar projections based on TP loads suggest 40 and 50% TP load reductions to meet the 5000 km² goal.

Discussion

One purpose of this analysis was to explore phosphorus as a potential driver of interannual and long-term changes in hypoxic area. It was not unexpected that we would be able to calibrate the model equally well with TP and TN loads because those loads are correlated (*r* = 0.66) during the calibration period (i.e., 1985–2006). However, for the TP-driven model, we were not able to produce a satisfying calibration by only varying the calibration term, *v*. We had to use significantly larger values and, for four years, we were not able to produce hypoxia at all. There was simply not enough organic matter produced for the given TP load, based on the same assumptions used for TN. Therefore, for the final TP calibration, we assumed phosphorus cycled twice

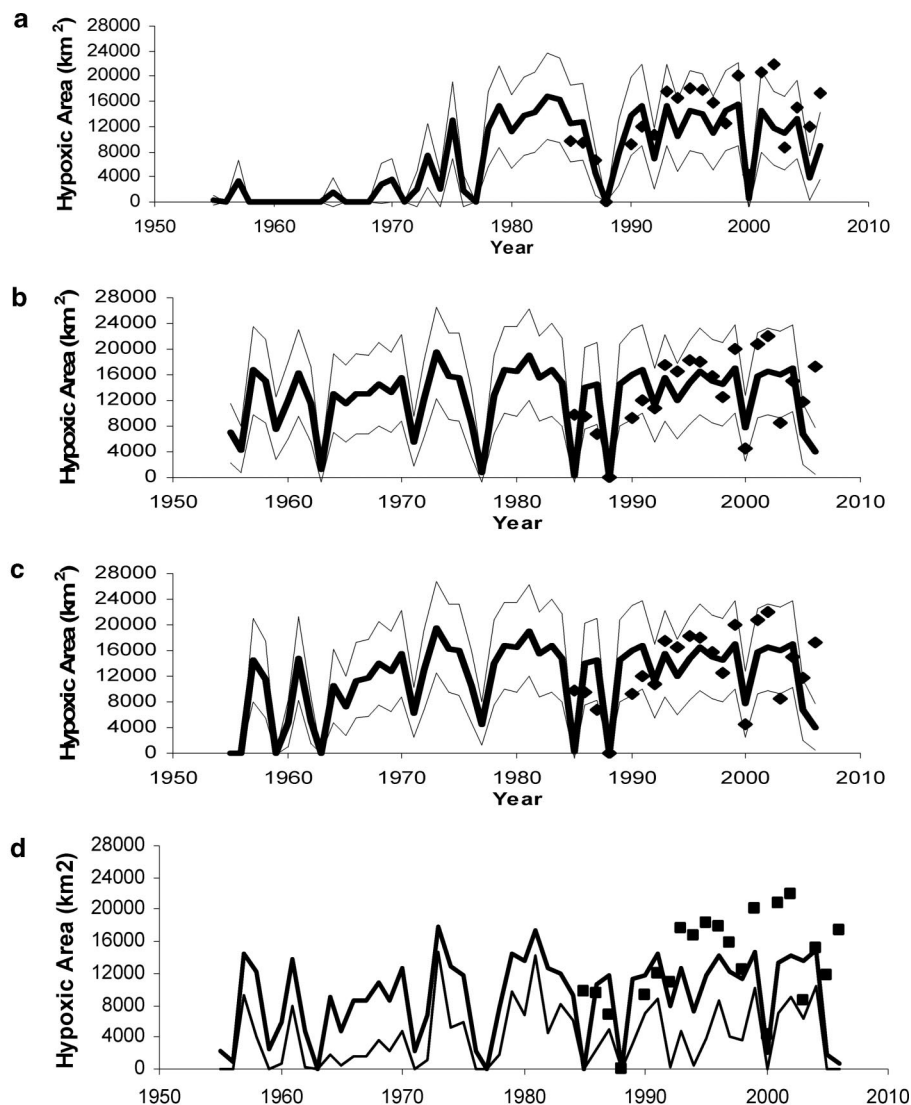


FIGURE 3. Simulated hypoxia for models driven by (a) TN, (b) TP assuming no long-term trend, and (c) TP assuming loads doubled since 1955. Model output is mean \pm standard deviation of 1000 Monte Carlo runs. Bottom panel (d) represents TP load assuming no long-term trend, but for assumptions of P recycle rates 50% higher than N (top line) and equal to N (bottom line).

as fast as nitrogen. With that assumption, we were not only able to produce a calibration that explained virtually all of the observed variability, but were able to do that with values of the calibration term, ν , comparable to those used for TN.

There are several lines of evidence supporting the assumption of P recycling as being more efficient than N on the time scale of the model computations for the northern Gulf of Mexico. While bioavailability of particle-bound P is low within freshwater portions of the Mississippi River, high ion abundances of Gulf seawater cause much of it to desorb and become available (32–36) down plume. Also, while organic P is largely ester-bound phosphates that are cleaved readily by easily produced alkaline phosphatase, organic N is largely covalently bound to a wide range of carbon compounds that require many different enzymes to synthesize (37). In addition, sediment diagenic processes increase biological availability of particle-bound P delivered to the Gulf (35) and low oxygen concentrations in bottom waters significantly increase P flux into overlying waters (38, 39). In contrast, benthic processes under these conditions provide a denitrification N sink that is relatively large compared to the loss of P through permanent sediment burial (3, 40). Measured fluxes of N and P from sediments provide a direct means to assess the role of sediment processes on the relative supply of N and P in the overlying water column. A

compilation of N and P flux studies shows a ratio of dissolved inorganic nitrogen to dissolved inorganic phosphorus flux ranging from 1 to 25, with a mean of 10:1 (41), which is enriched in P relative to the Redfield ratio.

Hindcasts. With the TP- and TN-calibrated models, we explored long-term controls on hypoxia development. As in the original application and other reports (14–17, 23), hindcasts with TN loads indicate that large-scale hypoxia did not form consistently until the mid- to-late 1970s (Figure 3a). This is consistent with empirical water-column evidence collected between 1970 and 1985, before the shelf-wide surveys began, summarized by Turner and Allen (42) and Rabalais et al. (43, 44), as well as by retrospective analyses of sedimentary records. Chen et al. (45) analyzed sediment pigments of oxygen-sensitive bacteria and suggest that increases in the 1960s reflect enhanced preservation with hypoxia as well as nutrient-driven increases in production. Using paleo-records of chlorophyll and its degradation products, accumulation of biogenic silica and organic carbon of marine origin, oxygen-sensitive mineral composition, and microfossils from sediment cores from within and outside the hypoxic region, Rabalais et al. (46, 47) demonstrated that eutrophication and hypoxia changed gradually from 1955 to 1970, followed by a steady increase to the late 1990s. There has also been increased total organic carbon and biogenic

silica accumulation in recent sediments near the mouth of the Mississippi River (48, 49), although the spatial and temporal variations observed between dated sediment cores are large. Analyses of ^{210}Pb -dated sediment cores showed substantial increases in organic carbon accumulation rates between the 1950s and 1990s (50), with the rate significantly higher within the area of chronic hypoxia compared to a site outside that area, and composed primarily of marine carbon produced *in situ*. Several studies compared abundances of benthic foraminifera with low and high tolerance for low oxygen conditions in dated sediment cores (51–54). Comparisons of the foraminifera *Ammonia* and *Elphidium* abundance (51) indicate an overall increase in oxygen stress during the 1960s and 1970s. Several other indicators (e.g., the relative abundance of three low-oxygen tolerant species: *Pseudonoin atlanticum*, *Epistominella vitrea*, and *Buliminella morgani*; 53) and the A/P ratio (agglutinated to porcelaneous orders) (54) also indicate that increases in the intensity of hypoxic events have occurred over the past 50 years. All of these studies support the interpretation that hypoxia is a relatively recent phenomenon that has increased over the past few decades.

In contrast to the TN-driven model and the empirical evidence outlined above, hindcasts with TP (Figure 3b and c) produce large hypoxic areas throughout the 1950–1970 period, with some as large as 16,000 km². These results depend on our assumption that, since the 1950s, variability in TP load has been driven primarily by variability in water delivery. There are several lines of justification for this. First, while measured nitrogen loads increased dramatically in the 1970s and 1980s, there was no significant trend in measured phosphorus loads (8; Figure 1c), and those loads are strongly correlated with streamflow throughout the period of record (Figure S1c, Supporting Information). Second, simple linear regressions of orthophosphate and total phosphorus concentration data, available for the Ohio River at Cairo and the Mississippi River at East St. Louis for 1960 to 1971 (55–57), from STORET, showed no temporal trends. In addition, because phosphorus is far more particle bound than nitrate (the primary TN load) it is unlikely TP loads would have tripled as nitrate did. In fact, Goolsby et al. (25) concluded that TP loads may have actually declined during that period because of the dramatic reductions in suspended sediment in response to construction of dams and locks. Kesel (58, 59) showed that suspended sediment loads in the lower river declined from annual averages of 227 million metric tons in 1930–1952, before the completion of Missouri river dams, to 112 million metric tons in 1953–1962, and 82 million metric tons in 1963–1982, following completion of a number of Arkansas River dams. Finally, in David and Gentry's (60) estimates of N and P balances for Illinois, they indicate that the excess net P inputs between 1960 and 1970 most likely ended up in soils with a much smaller percentage leaving in rivers, compared to N. However, it is important to note that TP-driven hindcasts were unrealistic even under our second scenario that assumed TP loads doubled between 1955 and the 1970s (Figure 3c).

Our results are also influenced by the assumption that phosphorus recycles twice as efficiently as nitrogen. While that assumption is justified above, we ran two additional hindcasts, assuming phosphorus recycles: (1) 50% faster than nitrogen, and (2) equal to nitrogen. It is clear that only by assuming a recycle rate twice that of nitrogen, does the model capture the observations between 1985 and 2006 (Figure 3b and c). When using phosphorus recycling that is equal to, or 50% higher than, that of nitrogen, the model underestimates hypoxia in the 1990s and 2000s, and still produces unrealistically large areas of hypoxia in the 1950s and 1960s (Figure 3d).

The inability of the TP-driven model to produce historical trends consistent with the empirical evidence suggests that, even if primary production and hypoxia are controlled currently by nitrogen or phosphorus, they were likely decoupled from TP loads in the past. If this were not true, we should have been able to provide more realistic hindcasts, given our ability to reproduce the more recent time period. The TN model's ability to provide hindcasts consistent with empirical evidence for major changes in eutrophication and hypoxia beginning 30 or 40 years ago supports the notion that it was the increase in TN loads during the 1970s and early 1980s that stimulated the dramatic increase in hypoxia. Although recent evidence points to phosphorus as important in controlling contemporary primary production (10–12), our analysis suggests that, if phosphorus is limiting, it became so because the larger relative increase in nitrogen loads during the 1970s and 1980s shifted the N:P load balance in favor of P limitation.

Management Implications. The focus of the integrated assessment (2) and subsequent Gulf Action Plan (6) was to reduce the size of the hypoxic zone by addressing nitrogen loads. However, with the system now apparently precariously perched between nitrogen and phosphorus limitation, one could theoretically decrease eutrophication and hypoxia by reducing the loads of either nutrient. If nitrogen loads alone were reduced, the system would likely revert to the stronger nitrogen limitation of the past and further reductions should reduce production and hypoxia. From this reanalysis with new USGS loads, we suggest TN load reductions of 37–45% below the 1980–1996 average are necessary to reach the Action Plan goal. It is also important to note that predicted long-term changes in midcontinental precipitation will also influence the timing and flux of water and nutrients, and thus hypoxia, even with no change in land use practices (16, 20, 61). In addressing this variability, Donner and Scavia (20) suggest that as much as a 50% reduction might be needed.

Although we have less confidence in our phosphorus model because of the inability to reproduce the longer-term trends, our analyses suggest that reducing TP loads between 40 and 50% could still satisfy the Action Plan goal. However, a phosphorus-only strategy is potentially dangerous because stronger phosphorus limitation in the vicinity of the river outflows would likely reduce productivity in the near-field, allowing more nitrogen to move down-plume to nitrogen-limited regions. This could potentially result in increased downfield production, and perhaps even more severe hypoxia west of where hypoxia currently forms (62), as was observed in North Carolina estuaries (63, 64) and the Paerl River in Hong Kong (65). Thus, management strategies should either reduce nitrogen loads, or reduce *both* nitrogen and phosphorus loads. The prudent approach would be to address both nitrogen and phosphorus because this should not only lead to reduced eutrophication and hypoxia in the Gulf of Mexico, but would also support Clean Water Act goals for the phosphorus-limited freshwaters of the basin.

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Supporting Information Available

Model description and assumptions and hypoxia data interpolation processes, graphs of load regressions (S1), hypoxia area-length regression (S2), and model calibration (S3). This material is available free of charge via the Internet at <http://pubs.acs.org>.

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