

2024 Forecast: Summer Hypoxic Zone Size in the Northern Gulf of Mexico

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Abstract

A hypoxic water mass with oxygen concentrations $\leq 2 \text{ mg l}^{-1}$ forms in bottom waters of the northern Gulf of Mexico continental shelf each spring/summer and lasts into the fall but can be disrupted by strong storms. The low oxygen conditions in the Gulf's most productive waters stresses organisms and may even cause their death so that living resources are threatened, including humans who depend on the fish, shrimp and crabs caught there. The main driver of the low oxygen zone size is nitrogen from the Mississippi River watershed that fertilizes the Gulf's surface waters creating excessive amounts of algal biomass. This biomass sinks to the bottom layer and sediments where decomposition leads to oxygen depletion. Various models use the May nitrogen load of the Mississippi River as the main driving force to predict the size of this hypoxic zone in late July. The warming of the continental shelf, however, has altered the phytoplankton-zooplankton-fish food web and this has shrunk the size of the zone created per nitrogen loading. A new model was developed to account for this warming effect, and it was used for the first time this year.

The June 2024 forecast of the hypoxic zone size for late July 2024 is that it will cover 5107 km^2 (1972 mi^2) of the bottom of the continental shelf off Louisiana and Texas. The 95% confidence interval is that it will be between 4687 and 5569 km^2 (1810 and 2150 mi^2). This estimate is based on the assumption that there are no hurricanes or unusual wind events in the two weeks before the monitoring cruise, or during the cruise.

The predicted hypoxic area is about 125% of the size of Rhode Island and equal to 38% of the 1985-2023 average of $13,546 \text{ km}^2$ ($n = 34$ including years with storms; two years had no cruise). If the area of hypoxia becomes as large as predicted, then it will be about the size of the Hypoxia Action Plan goal to reduce the zone to less than $5,000 \text{ km}^2$ (1931 mi^2) by 2035. Only minor reductions in the nitrate loading from the Mississippi River to the Gulf of Mexico have occurred since the formulation of the Hypoxia Action Plan environmental goal in 2001. *The reduction in the predicted size of the hypoxia area this year is solely due to ocean warming, not to a decline in nitrate loading from the Mississippi River.*

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Introduction

Hypoxic water masses in bottom waters occur when the oxygen concentration falls below 2 mg l^{-1} . The hypoxic water of the northern Gulf of Mexico is distributed on the continental shelf west of the Mississippi River and onto the northern Texas continental shelf, from near shore to as far as 125 km offshore, and in water depths up to 60 m (Rabalais et al., et al., 2007a; Jarvis et al., et al., 2021; Figure 1). It has been found in all months but is most persistent and severe in spring and summer (Turner et al., et al., 2005; Rabalais et al., 2007a,b). The July distribution of hypoxic waters most often is a single continuous zone along the Louisiana and adjacent Texas shelf. Hypoxia also occurs east of the Mississippi River delta but covers less area and is ephemeral. These areas are sometimes called ‘dead zones’ in the popular press because of the absence of commercial quantities of shrimp and fish in the bottom layer – something that is of economic consequence to the fishery (Purcell et al., 2017; Smith et al., 2017). The number of dead zones throughout the world has been increasing in the last several decades and currently totals over 500 (Díaz and Rosenberg 2008; Rabalais et al., 2010; Conley et al., 2011; Breitburg et al., 2018). The hypoxic zone off the Louisiana coast is the second largest human-caused coastal hypoxic area in the global ocean. A recent description of hypoxia development in the northern Gulf of Mexico is in Rabalais and Turner (2019). A well-written and recent overview of low oxygen zones in rivers, lakes, estuaries and oceans is in Kirchman (2021).

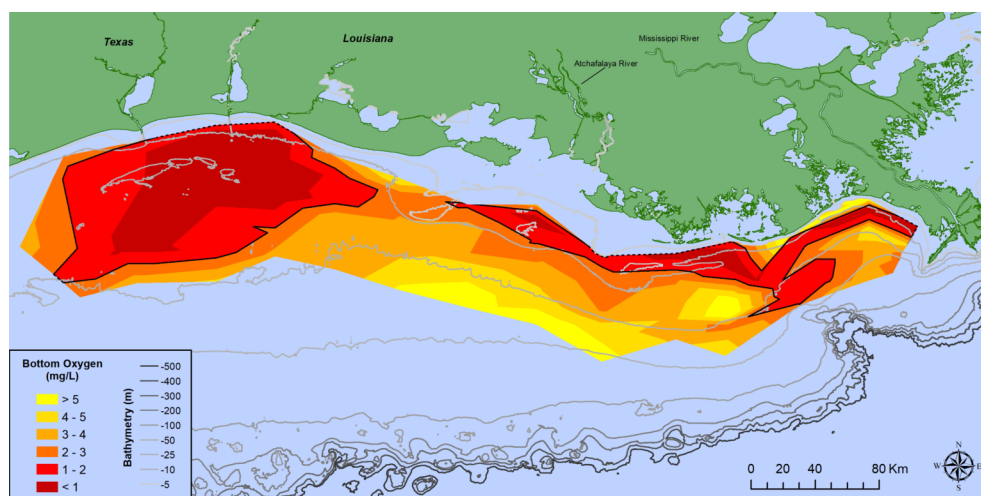


Figure 1. Oxygen concentrations in bottom water across the Louisiana-Texas shelf from July 25 – 31, 2021. Data source: N.N. Rabalais, C.N. Glaspie, and R.E. Turner, Louisiana State University. Funding: NOAA, NCCOS.

Systematic mapping of the area of hypoxia in bottom waters of the northern Gulf began in 1985 at geographically fixed stations (Appendix Figure 1). Its size from 1985 to 2023 ranged between 40 to 22,720 km² during late July to early August and averaged 13,546 km² (5,230 mi²). Monthly, and bi-monthly monitoring on two transects off Terrebonne Bay, LA, and the Atchafalaya delta, LA, ended in 2012. The annual number of hypoxia-focused cruises peaked 20 years ago and is now at the bare minimum (Appendix Figure 2).

Hypoxic Water Formation

Hypoxic water masses form because the consumption of oxygen in bottom water layers exceeds the re-supply of oxygen from the atmosphere and photosynthesis. The re-aeration rate is negatively influenced by stratification of the water column, which is primarily dependent on the river's freshwater discharge and accentuated by summer warming. The overwhelming supply of organic matter respired in the bottom layer is from the downward flux of organic matter produced in the surface layer. The transport to the bottom layer is the result of sinking of individual cells, as the excretory products of the grazing predators (zooplankton) that 'package' them as fecal pellets, or as aggregates of cells, detritus and mucus. The decomposition of this organic matter declines as it falls through the water column (Turner et al., 1998), but the descent rate is sufficiently rapid so that most respiration occurs in the bottom layer and sediments, not within the water column.

The amount of organic matter produced in the surface waters is primarily limited by the supply of nitrogen, not phosphorus (Scavia and Donnelly 2007; Turner and Rabalais 2013), and previous indicators of phosphorus deficiency are not as reliable as they were once thought to be (Fuentes et al., 2014). The evidence for this conclusion is that the supply (loading) of nitrogen (primarily in the form of nitrate-N) from the Mississippi River watershed to the continental shelf within the last few decades is positively related to chlorophyll *a* concentration (Walker and Rabalais 2006; $R^2 = 0.30 - 0.42$), the rate of primary production (Lohrenz et al., 1997, $R^2 > 0.77$; Lohrenz et al., 2008), and the spatial extent of the hypoxic area in summer (Turner et al., 2012; $R^2 > 0.9$). The size of the shelfwide hypoxic zone has increased since it began occurring in the 1970s simultaneously with 1) the rise in carbon sequestration in sediments, 2) indicators of increased diatom production, and 3) shifts in benthic foraminiferal communities (Turner and Rabalais 1994; Sen Gupta et al., 1996; Rabalais et al., 2007b; Turner et al., 2008). There is, therefore, a series of cause-and-effect arguments linking nitrogen loading in the river to phytoplankton production, bottom water oxygen demand, and the formation and maintenance of the largest human-caused coastal hypoxic zone in the western Atlantic Ocean.

The oxygen consumption creates a zone of hypoxia that is constrained by the geomorphology of the shelf, horizontal water movement, stratification, and vertical mixing (Obenour et al., 2012; Justić and Wang 2014). The significance of reducing nutrient loads to these coastal waters is based on the coupling between the organic matter produced in response to these nutrients and its respiration in the bottom layer (MRNGoM HTF 2001, 2008; Rabalais et al., 2002, 2007a, 2010; SAB 2007).

Mississippi River Discharge and Nitrogen Loading

Hypoxic conditions are dependent on river discharge because of the influence that water volume and salinity have on the physical structure of the water column and on the nutrients delivered to the coastal zone. The US Geological Survey (USGS) provides monthly estimates of river discharge and nitrogen concentration (<http://toxics.usgs.gov/hypoxia/mississippi/>), which are used to calculate the nitrogen loading for the Mississippi River watershed into the Gulf of Mexico. The nutrient load is calculated by multiplication of the discharge volume and the concentrations of nutrients, particularly nitrogen (Figure 2).

The discharge from the Mississippi River and Atchafalaya watershed in May 2024 was $33,000 \text{ m}^3 \text{ s}^{-1}$ (cms), which is the 27th largest in 40 years from 1985 to 2024, and equal to about 62% of the average May discharge. The concentration of nitrate has been declining slightly over the last 20 years, but the increase in river discharge means that the total loading has remained the same in recent decades or is increasing (Figure 2; Sprague et al., 2011; Crawford et al., 2019). Nitrogen loading is predominately from the Midwest, the upper Mississippi River subbasin and the Ohio River subbasin (Figure 3) and comes from agricultural land (Figure 4).

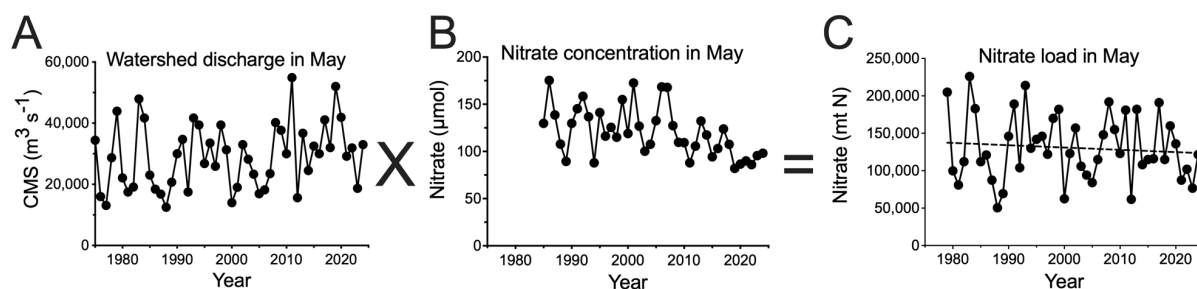


Figure 2. The discharge of the Mississippi River (A), the concentration of nitrate (B), and the resultant nitrate load (C) in May. The discharge and nitrate concentrations are from the United States Geological Survey. N.S. = the simple regression line is not significant.

Some consequences of water quality degradation with nitrate contamination include higher sewage treatment costs (Dearmont et al., 1998), seafood price decreases (e.g., Smith et al., 2017), compromises to fish reproduction (Tuckey and Fabrizio 2016) and increased frequency and duration of harmful algae events inshore and offshore (Lopez et al., 2008). There are links between nitrate in drinking water and birth defects [neural tube and spinal cord including spina bifida, oral cleft defects and limb deficiencies (Brender et al., 2013)], and bladder and thyroid cancer (Ward et al., 2018). The strictly nutrient-related issues are co-developing with ocean acidification and climate change and the cumulative and synergistic interactions may be even more socially and ecologically significant than just nitrate contamination (Moss et al., 2011). Plastics fill oceans (Lavers and Bond 2017), pharmaceuticals are distributed in sewage (Kasprzyk-Hordern 2009; Wilkinson et al., 2022), the COVID-19 virus and other viruses spread in partially treated sewerage wastes from: a) aging septic tanks (Farkas et al., 2020), b) unconstrained wetland treatment systems with insufficient hydrologic controls, and, c) overloaded treatment systems.

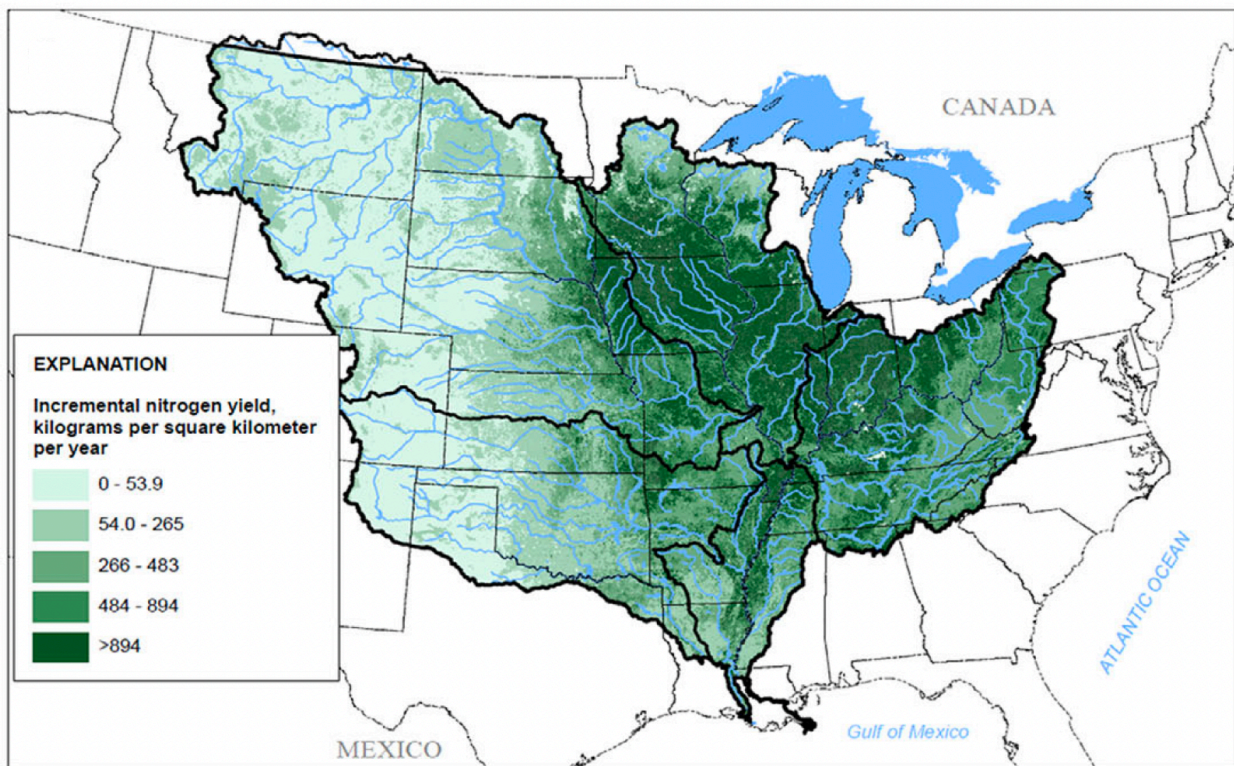


Figure 3. The yield of nitrogen (kg km^{-2}) from the Mississippi River watershed (from Robertson and Saad 2021).

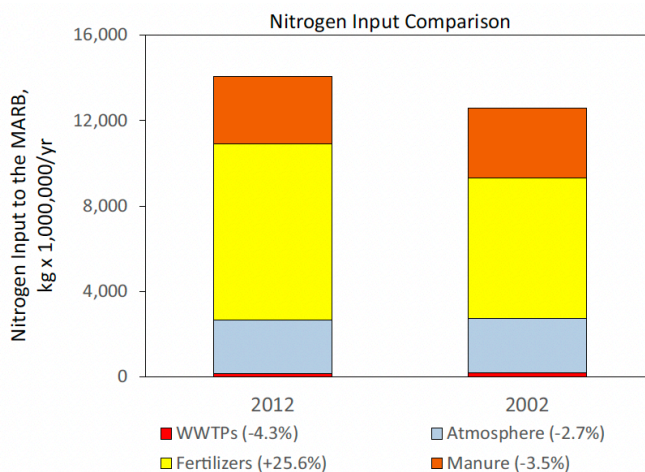


Figure 4. The percent of nitrogen loading in the watershed that is sourced to wastewater treatment plants (WWTP), fertilizers, atmospheric deposition and manure in 2012 and 2002 (from Robertson and Saad 2021).

Hypoxic Zone Size

Models for predicting the size of the hypoxic zone rely on July cruise data, primarily because there are no comparable shelfwide data for other months. Data on the size of the hypoxic zone in late July from 1985 to 2023 are based on annual field measurements (data available at <http://www.gulfhypoxia.net>). Data for 10 years were not included in the analysis because there were hurricane conditions just before or during the cruise, no funding, no vessel, or the 2010 BP

oil spill occurred (1988, 1989, 1997, 2003, 2005, 2008, 2010, 2016, 2019, and 2020). Data for 1978 to 1984 are estimated from contemporary field data. The estimates for before 1978 assume that there was no significant hypoxia then and are based on results from various models and sediment core analyses. The storms changed currents, disrupted the stratified water column, and re-aerated the water column. It may take a few days to several weeks, depending on water temperature and initial dissolved oxygen concentration, for respiration to reduce the dissolved oxygen concentration to $\leq 2 \text{ mg L}^{-1}$ after the water column stratification is re-established.

Prediction for 2024

We used a new model to forecast the hypoxic zone in the northern Gulf of Mexico in July 2024 that is described in a manuscript under peer review. We predict that it will cover 5107 km^2 (1972 mi^2) of the bottom of the continental shelf off Louisiana and Texas (Figure 5). The 95% confidence interval is that it will be between 4687 and 5569 km^2 (1810 and 2150 mi^2). This estimate is based on the assumption that there are no significant tropical storms occurring in the two weeks before the monitoring cruise, or during the cruise.

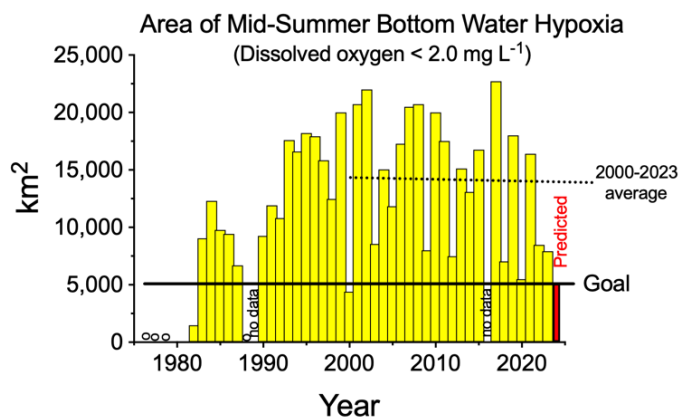


Figure 5. Area of the hypoxic zone from 1979 to 2023 and the predicted value for 2024 (red bar). The Hypoxia Action Plan restoration ‘goal’ is $5,000 \text{ km}^2$.

Hypoxia Models and Model Accuracy

The previous models predicting the size of the hypoxic zone in July are based on the May total nitrite+nitrate nitrogen load (note: concentration \times discharge equals the load) to the Gulf from the main stem of the Mississippi River and the Atchafalaya River. [The nitrite+nitrate loading will be referred to here as “nitrate” loading because the nitrite component is a minimal component of the two] The residence time of the surface waters along this coast is about 2 to 3 months in the summer, hence the 2 to 3 month lag between the loading rate calculated in May and the size of the hypoxic zone in late July. These models must change as the ecosystem evolves. For example, the size of the hypoxic zone for the same amount of nitrate loading is increasing at about three times over the last 40 years (Turner et al., 2012). Further, the models will eventually be adjusted to account for the limited space on the shelf for hypoxia to occur (a geographic and physiographic constraint). The process-based ecosystem models are a platform to greatly expand understanding how the physical and biological factors interact over all months are increasingly accurate and visually-appealing, but also require additional seasonal data to validate them as conditions change throughout the year and among years.

The unstated hypothesis implied by these models is that the system can be treated as a chemostat limited by N, in the same way that the chlorophyll *a* concentration or algal biomass in lakes might be modeled by P loading to the lake. The Streeter–Phelps type models initiated by Scavia and colleagues also incorporate this nutrient dose:response framework (Scavia et al., 2003, 2004; Scavia and Donnelly 2007) in their predictive schemes. These models assume that the size of the zone is driven mostly by what happens in the current year and that other influences cause variation around a relatively stable baseline suite of factors. An example of secondary influences might be seasonal or annual variations in wind speed and direction or freshwater volume. The reference point for calibrating model is the behavior of the system in recent history. We use the last 23 years of data on the relationship between hypoxic zone size and nutrient loading for this model. Others do something similar. For example, the USGS uses the last five years of data to calibrate the ‘LOADSET’ model, and Scavia and Donnelly (2007) update the coefficients in their model annually by using rolling 3- to 5-year averages for coefficients (Evans and Scavia 2010). Their recent numerical adaptation has the effect of adjusting model input with each year, but not explaining the biological/physical basis for these changes any better than one of our earlier models did with the ‘year’ term. The year term in our model is, in other words, descriptive, but not explanatory.

Nutrient load models are robust for long-term management purposes, but they are less robust when short-term weather patterns move water masses or mix up the water column (Rabalais et al., 2018). Some of the variations in the size of the Gulf hypoxic zone result from re-aeration of the water column during storms. The size of the summer hypoxic zone in 2008, for example, was less than predicted because of the influence of Hurricane Dolly. Tropical Storm Don was a similar complication in 2011. Climate changes may alter the spring initiation of hypoxia formation, duration and frequency. The timing of hypoxia in the Chesapeake Bay, for example, is earlier with climate warming (Testa et al., 2018). The needed detailed seasonal data necessary to make phenological comparisons for the Gulf of Mexico are not known. The prediction in 2018 was noteworthy for the great disparity between the much larger size of the hypoxic zone predicted by *all* models and the actual size. The predicted size of the forecast from four models ranged from 12,949 to 17,523 km², but the measured size was 7040 km².

Other models predicting oxygen dynamics on this shelf are in Bierman et al. (1994), Justić et al. (2003), Scavia and Donnelly (2007), Forest et al. (2011), Scavia et al. (2003, 2004), Justić and Wang (2014), Fennel et al. (2016), Justić et al. (2017), Testa et al. (2017), Laurent et al. (2018), Fennel and Laurent (2018), and Ou and Xue (2022). Other forecasts for this year will be from the University of Michigan (<http://scavia.seas.umich.edu/hypoxia-forecasts/>), Dalhousie University (<http://memg.ocean.dal.ca/news/>), North Carolina State University (<https://obenour.wordpress.ncsu.edu/news/>), and the Virginia Institute of Marine Science (http://www.vims.edu/research/topics/dead_zones/forecasts/gom/index.php). The NOAA ensemble predictions are based on these models (<http://www.noaa.gov/media-releases>). These models do not always produce similar results, and model improvement is one focus of ongoing research efforts supported by the NOAA National Centers for Coastal Ocean Science.

Climate Warming

We developed a new model used in 2024 that incorporates rising oceanic temperatures which are ubiquitously documented on the global and local scales in various reports. Oceans are

warming this century because they store about 80-90% of the heat accumulating in the atmosphere which has been rising (Levitus et al., 2005). Surface water temperatures in the central Gulf of Mexico have risen at about the same rate as in the global ocean (Figure 6). The bottom water temperatures on the continental shelf of the northern Gulf of Mexico have been routinely measured on hypoxia cruises during late July to early August and rose about $0.41\text{ }^{\circ}\text{C decade}^{-1}$ from 1963 to 2023 (Figure 7). This water warming has been projected to increase the deoxygenation of the ocean because of reduced mixing in more highly stratified layers (Levin 2018) and lower oxygen solubilities. Some models predict that the hypoxic zone in the northern Gulf of Mexico will increase by 1 to 10% (Eldridge and Roelke 2010; Altieri and Gedan 2015; Lehrter 2017). The authors frequently acknowledge that models are insufficient to capture the many biogeochemical processes, omitting temperature effects on biological components or using only one phytoplankton type (diatoms) to simplify modeling. But the biology is not constant and changes with temperature. Warming will have an effect on food web components involved in the creation of the organic matter in surface waters and its downward flux where it is decomposed and deoxygenates the bottom layer.

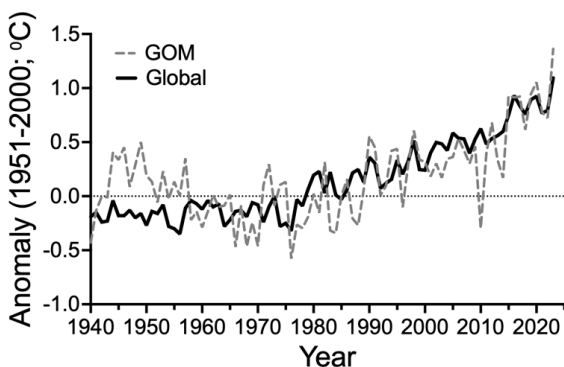
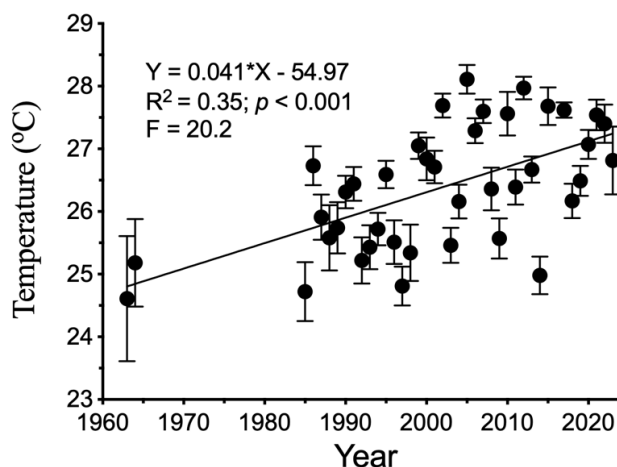


Figure 6. Average annual surface water temperatures in the global ocean between 90°S and -90°N , and in the central Gulf of Mexico between 20°N and 30°N , and 98°W and 92°W . The data file was generated using Climate Reanalyzer (<https://ClimateReanalyzer.org>).

Figure 7. The July bottom water temperature on the continental shelf of the northern Gulf of Mexico increased at $0.41\text{ }^{\circ}\text{C decade}^{-1}$ from 1963 to 2023. Updated from Turner et al. (2017).



These temperature increases affect everything from bacteria to fish (Millien 2006; Daufresne et al., 2009). Temperature, for example, is inversely related to size of individual organisms of the same species (the ‘temperature-size rule’; Atkinson et al., 2003). The growth rates in ectotherms have a skewed optimum at higher temperatures, beyond which performance drops off rapidly (Thomas et al., 2017; Figure 8), although the shape of the curve varies among species. These relationships can be subtle. Between 20 and $30\text{ }^{\circ}\text{C}$, for example, the natural

mortality of zooplankton and handling time of prey increases, but the intrinsic growth rate of phytoplankton prey and capture rate decreases (Zhao et al., 2002).

The quality and quantity of carbon flows within and among water layers will be affected by these temperature induced consequences to have a cumulative effect on ecosystems. For example, Laws et al.'s (2000) seminal analysis of 11 steady state systems, including polar, Red Sea, Greenland, and open ocean environments, showed that the percent export from surface to deeper water of new primary production of the total production was associated with both temperature and primary production. This transfer from surface to deeper layers is quantified as an export function (ef), which is the % of the surface production that sinks to deeper water, and some is buried in sediments; it is commonly known as the 'biological pump' that sequesters carbon. They found that temperature was negatively correlated with ef , a relationship that explained 86% of the variability in ef . Dunne et al. (2005) and Laws et al. (2011) used much larger data sets to show the same outcome. Maiti et al. (2013) demonstrated regional variability in ef for the Antarctic, and several groups have demonstrated that there is a stronger effect of temperature in nutrient replete coastal systems like the northern Gulf of Mexico than in nutrient deficient systems such as the open ocean (Cael and Follows 2016; Mara  n et al., 2018). Reducing the percent of the carbon produced in aquatic surface layers that is exported into the bottom layers (ef) will favor a lower consumption of oxygen and *shrink* the size and duration of oxygen-depleted waters ($<2 \text{ mgO}_2 \text{ L}^{-1}$; 2 ppm) in stratified bottom layers.

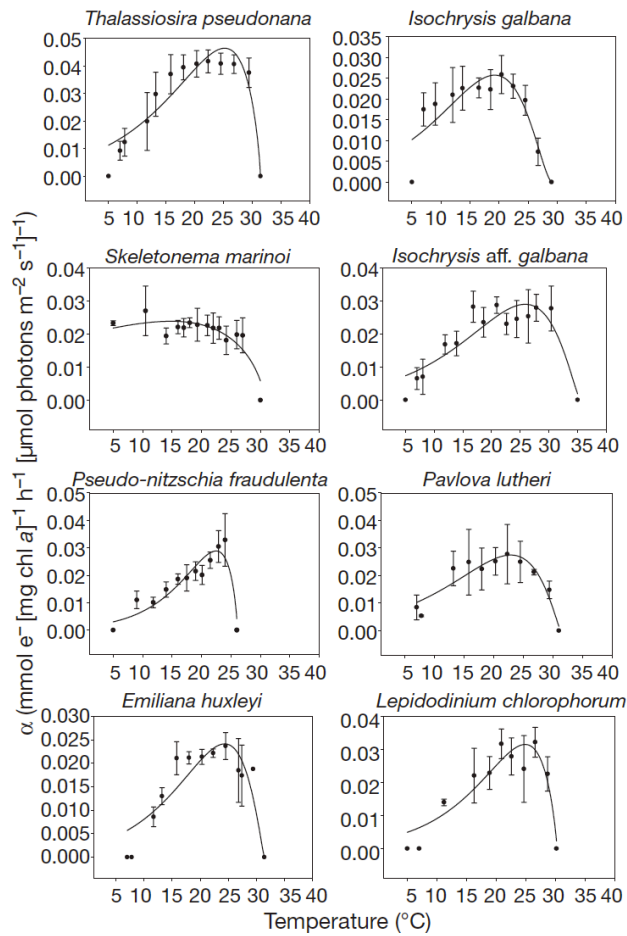


Figure 8. Examples of how the photosynthetic efficiency of 8 phytoplankton species changes with temperature. Means \pm SE of triplicate cultures are shown. From Claquin et al., 2008.

Reducing Nitrogen Loading

The nitrogen loading is predominantly driven by land use choices in the river watershed. *There has been no reduction in the quantity of nitrogen in the river after the Hypoxia Action Plan (MRNGoM HTF 2001) was forwarded to Congress in 2001.* The rise in nitrate concentration and decline in silicate concentration occurring in the two decades after the 1960s has become relatively stable over the last 40 years. The average annual nitrate concentration at St. Francisville, LA, for example, has been about the same from 1992 to 2015. The variability of nitrate and silicate concentrations is now moving coincidentally with variations in alkalinity. The rising nitrate concentrations are often ascribed to changing temporal and spatial applications of fertilizers. This is because fertilization increases loading rates to the land and converting forests to agricultural land and improving drainage, especially by installing subsurface tiles, increases the percent of the applied nitrogen that goes into drainage channels (Randall and Gross, 2001; Figure 9). The N fertilizer application rate increased almost three orders of magnitude from 1950 to 2015: from less than $0.01 \text{ g N m}^{-2} \text{ yr}^{-1}$ in 1850 to $9.04 \text{ g N m}^{-2} \text{ yr}^{-1}$ in 2015 (Cao et al., 2018) as fertilizer use climbed from 1.5 to 8.1 Tg N y^{-1} from 1960 to 2014 (Tian et al., 2020). Nitrogen fertilizer use in the Mississippi River basin now accounts for ~65% of the total fertilizer application in the continental United States (Tian et al., 2020).

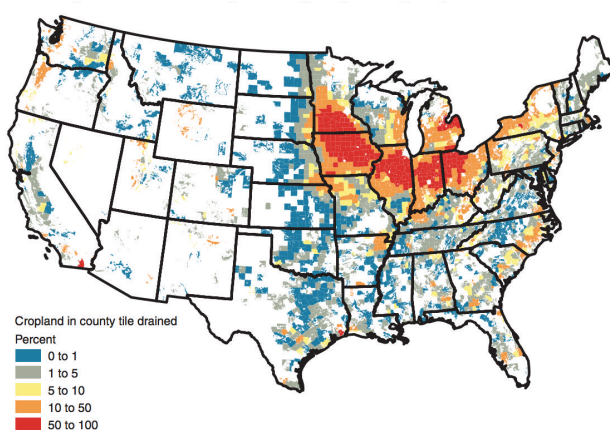


Figure 9.
Land under
drainage in
the US.
Source:
USDA,
Economic
Research
Service
calculations
based on

data from USDA, National Agricultural Statistics Service,
2013 Census of Agriculture.

Much of the total nitrogen yield in the basin is a consequence of structural changes in the landscape from building swales and tiling, not from solely (or simply) the nitrogen fertilizer application rates (e.g., Kaspar et al., 2003; McIsaac and Hu, 2004; Nangia et al., 2008; Tomer et al., 2003; Randall and Gross, 2001). Land drainage dries soils, as intended, but reduces denitrification, and moves the leached nitrate to waterways that reduce the potential sediment trapping and denitrification in riparian zones (Burt and Pinay, 2005). Randall and Gross (2001) showed that there was a close correspondence between tile drain water yield and nitrate yield, which is visually apparent when maps of the county-level nitrogen yield and drainage are compared. This is why McIsaac and Hu (2004) found that the 1945–1961 riverine nitrate flux in an extensively tile drained region in Illinois averaged $6.6 \text{ kg N ha}^{-1} \text{ year}^{-1}$, compared to 1.3 to 3.1

kg N ha⁻¹ year⁻¹ for the non-tile drained region, even though the nitrogen application was greater in the non-tile drained region. Arenas Amado et al. (2017) found that tiles “delivered up to 80% of the stream N load while providing only 15–43% of the water” in the 122 km² Otter Creek watershed in Iowa. Ikenberry et al. (2014) estimated that 97% of the nitrate flow occurred during 50% of the highest flows in a 5-year study of the 5132 ha Walnut Creek watershed in Iowa. They reported that two-thirds of row crop land in Central Iowa and the Minnesota Till Prairies Major Land Resources Area had subsurface tile drainage. Measurements of inorganic C concentrations in the intensively farmed Raccoon River, IA, by Jones and Schilling (2013) are another example of the consequence of drainage on nitrate concentrations. The alkalinity there doubled from 2000 to 2011 compared to in 1931 to 1944 (Jones and Schilling, 2013), during which there was an increase in drainage and nitrate concentrations, but not fertilizer applications (Hatfield et al., 2009). In summary, nitrogen yields from field to streams is generally dominated by fertilizer application which increased in the 1960s and stabilized in the last decades. More nitrate escapes farm fields when tiled, and, thus drainage improvements are a significant multiplier of these loadings.

Shifting to more diverse plant rotations will be needed to have a major effect on carbon losses and to sustain soil fertility and farming enterprises. Achieving this restoration does mean that it will result in negative consequences for farmers or consumers. Soil carbon restoration has many benefits important to sustaining agriculture because organic material holds water, minerals, nutrients and organisms giving it soil structure, resistance to erosion and increased soil fertility (Jackson et al., 2017; Chambers et al., 2016). Improving soil health will provide public benefits of improved water quality, flood reduction, enhanced wildlife habitat, reduced air pollutants and reduced global warming potential (Jordan et al., 2007; Boody et al., 2005; Ryan et al., 2018). Plant rotations will include cover crops, especially deep-rooted perennials (Poeplau and Don, 2015; Jungers et al., 2019; Paustian et al., 2019). Land under tile drainage can be a part of these efforts (Randall and Mulla, 2001; Dinnes et al., 2002). Tile drainage can enter buffer strips before reaching streams, drain into wetlands, or even not be used if row cropped fields are converted to perennials. Planting only shallow-rooted annuals leaves the ground without plant cover for more than half the year and results in more soil erosion (Heathcote et al., 2013). Replacement of annuals with deep-rooted perennials provides continuous living cover and reduces soil erosion (Song et al., 2014). Water quality has improved in some sub-watershed streams of the Mississippi watershed because of soil conservation (Rabotyagov et al., 2014, McIsaac et al., 2016; Garcia et al., 2016).

A signature example of alternative management of perennial grains is provided by Liebmann et al. (2013), Davis et al. (2012) and Tomer and Liebmann (2014) who conducted a 7-year field trial of different cropping systems for corn-soybean rotations. Some key findings were that, by using cover crops for 4 years, there was a 50% or more reduction in fossil fuel use, a doubling of employment, and no loss of profits. The diversification of crop coverage with small grains and legumes had a 91% reduction in fertilizer use, 97% reduction in herbicide use, and increased carbon storage. There is much work remaining – half of the US stream and river miles violate water pollution standards (Keiser and Shapiro, 2019). Whole system analyses of land use alternatives for small and large farms are needed to include not only greenhouse gas emissions, but also energy expenditures, wildlife, water quality, and social factors. Boody et al. (2005) found that alternative land management schemes using the same resources could create “improved water quality, healthier fish, increased carbon sequestration, and decreased

greenhouse gas emissions, while economic benefits include social capital formation, greater farm profitability, and avoided costs.” On-the-ground experiments at a watershed scale that include social governance (Meyfroidt et al., 2022) are recommended.

Fertilizer applications in the watershed are not always directly proportional to nutrient loading in the Mississippi River; their influence is modified by tile drainage, climate variation, plant choices, and farm management. Conversion of natural habitat to agricultural fields with improved drainage reduces plant cover (and increase in evapotranspiration) and more water is shunted into streams. This, in turn, changes the rate of soil accumulation and denitrification as well as alkalinity and the concentrations of alkalinity, nitrate, phosphate, and silicate. The integrated culmination of these land use changes across the watershed over the last 100 years reduced water residence time and nutrient processing in soil but increased concentrations in water bodies; the percent of nitrate exported from soils to water becomes higher as a result. Today agricultural lands are hemorrhaging the carbon needed to build and sustain healthy soil and a significant carbon sink is not being realized. Examples of accommodations for simultaneous water quality improvements while building soil health exist, but more on-the-ground examples are needed to integrate both the biogeochemical factors and practical socio-political-economic aspects of farming.

The replacement of deep-rooted perennials giving continuous living cover with short-rooted annuals leaving the ground bare for more than half the year results in more soil erosion (Heathcote et al., 2013). Putting perennials back into the landscape will reduce soil erosion and nutrient losses from farm fields. An excellent example of the use of perennial grains is provided by Liebmann et al. (2013) and Davis et al. (2012) who conducted a 7-year field trial of alternative cropping systems for corn-soybean rotations at Iowa State University. Some key findings were that by using cover crops for 4 years there was a 50% or more reduction in fossil fuel use, a doubling of employment, and the profits remained unchanged. The diversification of crop coverage with small grains and legumes had a 91% reduction in fertilizer use, 97% reduction in herbicide use, and increased carbon storage. Implementing these strategies requires changing significant embedded social, agronomic and political conditions that will benefit the agronomic community, the soil upon which it depends and may be a positive component of climate change adaptation. Maintaining the status quo will not be so helpful.

Post-cruise Assessment

The 2024 mapping cruise is scheduled for July 25 to August 1. The data will be posted as close to daily as possible at <http://www.gulfhypoxia.net>. Delays will be related to the QA/QC of dissolved oxygen data against Winkler titrations. The data from this year’s cruise will be used to quantify the relative merits of the assumptions of the models used to predict hypoxia size, and to compare them with other models. This is an example of how long-term observations are one of the best ways to test and calibrate ecosystem models, to recognize the dynamic nature of our changing environment(s), and to improve the basis for sound management decisions. The post-cruise assessment will be provided at the end of the summer shelfwide hypoxia cruise and posted on the same website where this report appears.

Acknowledgments

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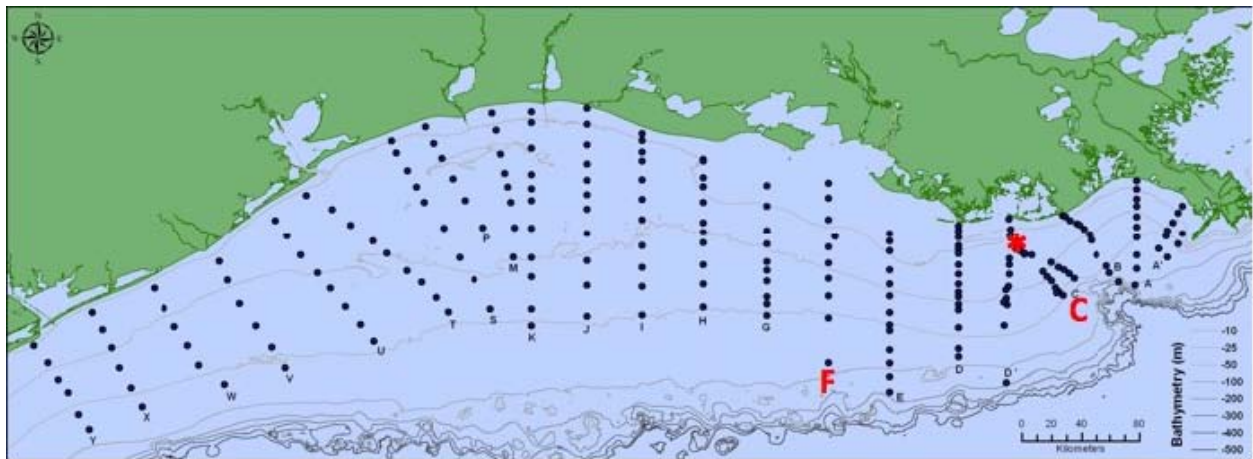
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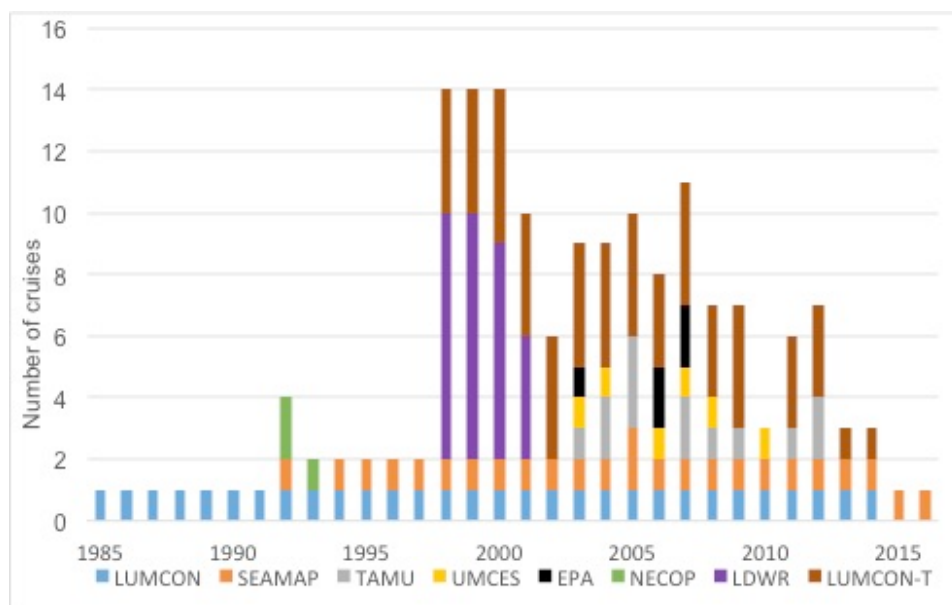
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Appendix



Appendix Figure 1. Location of hypoxia monitoring stations sampled in summer (but not all stations in every year, depending on location of hypoxic area), the transects off Terrebonne Bay (transect C) and Atchafalaya Bay (transect F), and the ocean observing system (asterisk) off Terrebonne Bay (no longer in operation).



Appendix Figure 2. The number of State, Federal and university cruises associated with hypoxia measurements in the northern Gulf of Mexico from 1985 to 2016. LUMCON = Louisiana Universities Marine Consortium; SEAMAP = Southeast Area Monitoring and Assessment Program; TAMU = Texas A&M University; UMCES = University of Maryland Center for Environmental Studies; EPA = U.S. Environmental Protection Agency; NECOP = Nutrient Enhanced Coastal Ocean Productivity; LDWR = Louisiana Department of Wildlife Research; LUMCON-T = transects sampled during the year by LUMCON. Source: Maiti et al., 2018; used with permission.