

## Chapter 8

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# Climate Change Impacts on Low Oxygen Zones and its Effects on Fisheries

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### Summary

A large-scale hypoxic zone ( $< 2 \text{ mg O}_2 \text{ l}^{-1}$ ) in the coastal waters of the northern Gulf of Mexico, recently exceeding 20,000 km<sup>2</sup>, overlaps with habitat and fishing grounds of commercial fish and shrimp species. We have developed a simple eutrophication model that accurately describes changes in surface and bottom oxygen concentrations for a station within the core of the Gulf of Mexico hypoxic zone. A sensitivity analysis revealed that the model is highly sensitive to external forcing, yet sufficiently robust to withstand order-of-magnitude changes in the nitrate flux of the Mississippi River. Model simulations indicated that bottom water hypoxia in the northern Gulf of Mexico has intensified in recent historical time, as a probable consequence of increased net productivity and an increase in the vertical flux of the organic carbon. Apparently, the long-term increase in riverine nutrient fluxes has been the primary factor controlling this historical decline in oxygen concentrations.

Nevertheless, the influence of climatic factors on nitrate flux has been significant and could further increase as a result of global climate change.

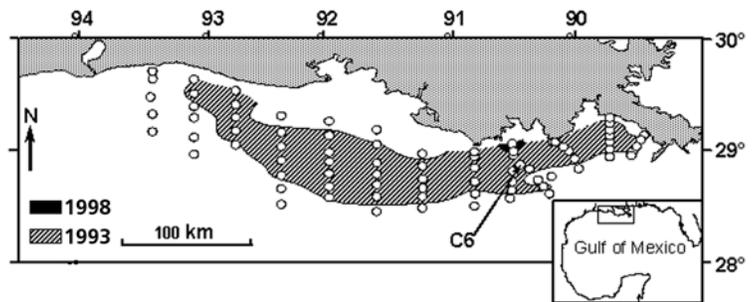
In contrast to a relatively high degree of confidence associated with the projected temperature increases, the effects of global climate change on the hydrological cycle are less certain, particularly on regional scales. The annual Mississippi River runoff, for example, was projected to decrease by 30% for the Canadian model, but to increase by 40% for the Hadley model by the year 2099. Model simulations further suggest that altered freshwater and nutrient fluxes would have important implications for water column stability, net productivity and global oxygen cycling in the northern Gulf of Mexico. Direct and indirect fisheries losses would likely be exacerbated if hypoxia expands in space or time as a result of global climate change.

## 8.1 Introduction

General circulation models (GCMs) forced by enhanced greenhouse gas concentrations have projected a global temperature increase of 2 to 6 °C over the next 100 years (IPCC, 1996). In contrast to a relatively high degree of confidence associated with the projected temperature increases, the effects on the global hydrological cycle are less certain, particularly on regional scales. Miller and Russell (1992) examined the impact of global warming on the annual runoff of the world's 33 largest rivers using a limited area model nested in a GCM. For a CO<sub>2</sub> climate, the runoff increases were detected in all studied rivers in high northern latitudes, with a maximum of +47%. At low latitudes there were both increases and decreases, ranging from +96% to -43%. Importantly, the model results projected an increase in the annual runoff for 25 of the 33 studied rivers. The northern Gulf of Mexico (Figure 1), which receives inflows of the Mississippi River - the sixth largest river in the world (Milliman and Meade, 1983), is one of the coastal areas that would experience increased freshwater input under the simulated 2x/CO<sub>2</sub> scenario. The annual Mississippi River runoff would increase 20%, and a higher runoff would occur during the May-August period (Figure 2). Wolock and McCabe (1999) estimated the potential effects of climate change on mean annual runoff for U.S. rivers based on Canadian and Hadley model projections. The estimates for the Mississippi River differed greatly between the two models. The annual Mississippi River runoff was projected to decrease by 30% for the Canadian model, but increase by 40% for the Hadley model by the year 2099.

The northern Gulf of Mexico would likely be highly sensitive to changes in freshwater inflow, because the combined discharges of the Mississippi and Atchafalaya Rivers account for 98% of the total freshwater inflow into the northern Gulf of Mexico (Dinnel and Wiseman, 1986). The nutrient-rich plumes of these two rivers rapidly form the Louisiana Coastal Current that flows predominantly westward along the Louisiana coast, and then southward along the Texas coast. Riverine nutrients are confined within the upper 10 m by a strong seasonal pycnocline ( $\Delta\sigma_t = 4 - 10 \text{ kg m}^{-3}$ ), which persists from April through October (Rabalais et al., 1991). Given this physical setting, it is not surprising that biological processes in the northern Gulf of Mexico are strongly

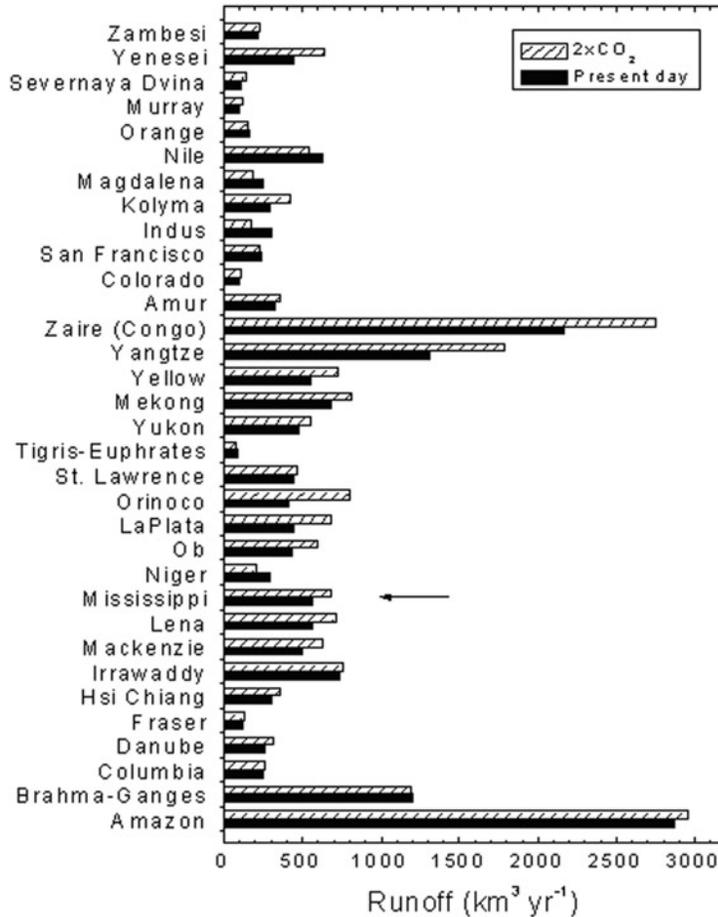
influenced by the pattern and relative magnitude of riverine freshwater runoff (Justić et al., 1993). Changes in the areal extent of hypoxic (< 2 mg O<sub>2</sub> l<sup>-1</sup>) bottom waters provide a representative example of the riverine influence on coastal productivity processes (Figure 1). The northern Gulf of Mexico is presently the site of the largest (up to 22,000 km<sup>2</sup>) and most severe coastal hypoxic zone in the western Atlantic Ocean (Rabalais et al., 1999). Hypoxia normally occurs from March through October in waters below the pycnocline, and extends between 5 and 60



**Figure 1.** Map of the northern Gulf of Mexico showing station grid and location of station C6. Shaded areas represent the distribution of hypoxic (< 2 mg O<sub>2</sub> l<sup>-1</sup>) bottom waters during August 1988 and July 1993.

km offshore (Rabalais et al., 1996). During the drought of 1988 (a 52-year low discharge record of the Mississippi River), however, bottom oxygen concentrations were significantly higher than normal, and formation of a continuous hypoxic zone along the coast did not occur in midsummer (Figure 1). The opposite occurred during the Great Flood of 1993 (a 62-year maximum discharge for August and September), when the areal extent of summertime hypoxia doubled with respect to the average hydrologic year (Rabalais et al., 1998). Hypoxia in the coastal bottom waters of the northern Gulf of Mexico develops as a synergistic product of high surface primary productivity, which is also manifested in a high carbon flux to the sediments, and high stability of the water column. Likewise, the 1993 event was associated with both an increased stability of the water column and nutrient-enhanced primary productivity, as indicated by the greatly increased nutrient concentrations and phytoplankton biomass in the coastal waters influenced by the Mississippi River (Dortch, 1994; Rabalais et al., 1998).

Climate change, if manifested by increasing riverine freshwater inflow, could affect coastal and estuarine ecosystems in several ways. First, changes



**Figure 2.** Average annual discharges and projected 2xCO<sub>2</sub> discharges for 33 world's major rivers (data obtained from Miller and Russell 1992). Mississippi River is indicated by an arrow.

in freshwater inflow will affect the stability of the water column, and this effect could be enhanced due to changes in sea surface temperatures. Vertical density gradients are likely to increase, which could decrease vertical oxygen transport and create conditions in the bottom water favorable for the development of severe hypoxia or anoxia (Justić et al., 1996). Second, the concentrations of nitrogen (N), phosphorus (P), and silicon (Si) in riverine freshwater inflows are typically an order of magnitude higher than those in coastal waters (Justić et al., 1995a; 1995b). The mass fluxes of riverine nutrients are generally well correlated with integrated runoff values (Turner and Rabalais, 1991; Goolsby et al., 1999). Consequently, the nutrient inputs to the coastal ocean are expected to increase as a result of the increasing riverine runoff, which could have an immediate effect on the productivity of coastal phytoplankton. Third, the stoichiometric ratios of riverine nutrients, Si:N, N:P and Si:P, may differ from those in the coastal ocean (Justić et al., 1995a). Increased

freshwater inflow, therefore, may also affect coastal phytoplankton communities by increasing or decreasing a potential for single nutrient limitation and overall nutrient balance (Smayda, 1990; Dortch and Whitedge, 1992; Justić et al., 1995a; 1995b; Turner et al., 1998). A decrease in freshwater inflow would have the opposite effects to those described above.

Here we review probable implications of climate change for the Gulf of Mexico hypoxic zone and its effects on fisheries. In this report we are focusing on two areas: (1) coupling between climate variability, freshwater runoff of the Mississippi River, and hypoxia in the coastal northern Gulf of Mexico, and (2) potential implications of global climate change for coastal fisheries in the hypoxic zone. In this analysis we use our previously published physical-biological model (Justić et al., 1996; 1997) and extensive long-term data sets collected at a station within the core of the Gulf of Mexico hypoxic zone (C6; Figure 1).

## 8.2 Methods

### 8.2.1 The Study Area

The study area encompasses the Louisiana coastal waters (Figure 1). Station C6, located in the inner section of the hypoxic zone, was used as a reference site for studies of impacts of climate change. This site was chosen because of the longest and most consistent oceanographic data records (1985 – present) that are available for the northern Gulf of Mexico. Three distinct oceanographic features of this region facilitated the application of a two-box modeling scheme. First, between the beginning of April and the end of October, a strong pycnocline ( $\Delta\sigma_t = 4 - 10 \text{ kg m}^{-3}$ ) is typically found at the average depth of 10 m (Rabalais et al., 1991). Because the depth is only about 20 m, the pycnocline virtually divides the upper and the lower water column into two distinct water bodies of approximately equal volumes. Second, the horizontal oxygen transport in the inner section of the hypoxic zone appears to be of lesser importance than the vertical oxygen transport. This is suggested by a high coherence between changes in vertical temperature gradients and changes in bottom oxygen concentration. In contrast, a strong tidal signal, which would indicate horizontal transport, is not present in the periodograms of oxygen data series from station C6 (Rabalais et al., 1994). Also, maximum lateral displacement of water parcels that can be expected due to diurnal and semidiurnal currents is only about 3 km (Rabalais et al., 1994), which is not likely to affect the inner section of a 60 km wide hypoxic zone. Third, because of the high turbidity of the continental shelf waters near the Mississippi River, primary productivity below the depth of 10 m is low (Lohrenz et al., 1990), and may be considered insignificant when compared to vertical oxygen transport.

### 8.2.2 Data Sources

The data on temperature, salinity, and dissolved oxygen concentration were obtained from a series of monitoring cruises conducted during the period June 1985 - October 1993. Our sampling station (C6; Figure 1) was occupied on a biweekly to monthly basis. Standard water column profile data were obtained from a Hydrolab Surveyor or a SeaBird CTD system with SBE 13-01 (S/N 106) dissolved oxygen meter. The dissolved oxygen measurements were calibrated with Winkler titrations (Parsons et al., 1984) that were periodically carried out during deployment of

the instruments. Continuous (15 - min intervals) temperature and oxygen measurements were also obtained at station C6 from July 1990 onward, using an Endeco 1184 pulsed dissolved oxygen sensor. The instrument was deployed at the depth of 19 m, approximately 1m above the seabed. Predeployment and postdeployment calibrations of the pulsed dissolved oxygen sensors were performed in accordance with factory specifications. Continuous oxygen measurements were controlled during hydrographic surveys of the study area, by comparison with Winkler titrations, Hydrolab Surveyor, or a SeaBird CTD data.

The daily-averaged discharge values for the lower Mississippi River at Tarbert Landing (August 1954 - May 2000) were provided by the U.S. Army Corps of Engineers. Those daily-averaged discharges are inferred from data-adaptive models of discharge versus water level, whose accuracy is normally higher than 90% (Bratkovich et al., 1994). Monthly discharge averages, used in the nitrate flux calculations, were computed from the daily-averaged discharge values. Monitoring station Tarbert Landing is located in Mississippi, 13 km downstream from the inlet channel to the Old River control structure, where one-third of the Mississippi River is diverted to the Atchafalaya River. The discharge at Tarbert Landing, therefore, accounts for about 70% of the total Mississippi and the Atchafalaya River discharge.

In this analysis, we used the monthly records (August 1954 - May 2000) of nitrate concentration at St. Francisville. St. Francisville is located in Louisiana, approximately 430 km upstream from the Mississippi River Delta. The average monthly nitrate fluxes were computed by multiplying the average monthly nitrate concentrations with the respective monthly discharge averages. Data sources and analytical methods used to determine nitrate concentrations are discussed in Turner and Rabalais (1991) and Goolsby et al. (1999). Nitrogen is often considered to be the limiting nutrient for the growth of the estuarine and coastal phytoplankton (e.g. D'Elia et al., 1986). The data from the northern Gulf of Mexico indicate that the frequency of stoichiometric nitrogen limitation is on the order of 30% (Justić et al., 1995a).

### 8.2.3 Model Formulation

We adopted our previously published two-box modeling scheme (Justić et al., 1996), which assumes uniform properties for the layers above and below

the average depth of the pycnocline. The model includes mathematical descriptions of relevant physical and biological processes that affect oxygen cycling in shallow, river-dominated, coastal waters (Justić et al., 1996, 1997). The oxygen concentration in the upper water column changes as a result of biological oxygen production and consumption, oxygen transport in the horizontal and vertical direction, and atmospheric exchanges. By neglecting horizontal oxygen transport due to advection and diffusion, the oxygen balance in the upper water column ( $O_{ts}$ ,  $\text{g O}_2 \text{ m}^{-2}$ , 0 – 10 m) may be described by the expression

$$\partial O_{ts} / \partial t = -F_{Ot} - D_o + NP \quad (1)$$

where  $t$  is time (d),  $F_{Ot}$  is the total air-sea oxygen flux ( $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ ),  $D_o$  is the diffusive oxygen flux through the pycnocline ( $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ ), and  $NP$  is the net primary productivity expressed in terms of oxygen equivalents ( $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ ). Because of the high turbidity of the continental shelf waters near the Mississippi River, primary productivity below the depth of 10 m is low (Lohrenz et al., 1990), and may be considered an insignificant term when compared to vertical oxygen transport. Thus, the balance equation for oxygen in the lower water column ( $O_{lb}$ ,  $\text{g O}_2 \text{ m}^{-2}$ , 10 – 20 m) includes only two terms: oxygen uptake due the benthic and water column respiration ( $R$ ), and oxygen resupply from the upper water column via turbulent diffusion ( $D_o$ ):

$$\partial O_{lb} / \partial t = -R + D_o \quad (2)$$

A description of the individual model terms from Eqs. 1 and 2 is discussed below.

Oxygen transport through the sea surface is dependent on the difference in the partial pressure of the gas in the surface layer and in the atmosphere. The transfer velocity across the air-sea boundary is thought to be a function of the temperature and the wind speed. In our calculations we used the formulation proposed by Stigebrandt (1991), which takes into account the effect of gas transfer due to bubbles:

$$F_{Ot} = V(O_s - 1.025 O_2') \quad (3)$$

Here  $F_{Ot}$  is the total air-sea oxygen flux ( $\text{g O}_2 \text{ m}^{-2} \text{ d}^{-1}$ ),  $V$  is transfer velocity ( $\text{m d}^{-1}$ ),  $O_s$  is the average surface oxygen concentration ( $\text{g O}_2 \text{ m}^{-3}$ , 0-10 m), and  $O_2'$  is the oxygen saturation value ( $\text{g O}_2 \text{ m}^{-3}$ ). Nega-

tive  $F_{Ot}$  values obtained from Eq. 3 indicate that the oxygen flux is directed towards the water column. The  $O_s$  value was computed by dividing the  $O_{ts}$  value (Eq. 1) with the thickness of the upper water column (10 m). The oxygen saturation value was computed from the observed temperature data and estimated salinity values using the equation of Weiss (1970). Surface salinity values for station C6 were calculated from the Mississippi River runoff data, using a time-delayed linear model ( $\tau = 2$  months;  $r^2 = 0.8$ ;  $p < 0.001$ ) developed by Justić et al. (1996). The transfer velocity was computed from a formula given by Liss and Merlivat (1986),

$$V = 5.9 Sc^{-0.5} (aW + b) \quad (4)$$

where  $Sc$  is Schmidt number, and  $W$  is wind speed ( $\text{m s}^{-1}$ ). The values of constants  $a$  and  $b$  depend on the wind speed. In the interval where  $3.6 < W < 13 \text{ m s}^{-1}$  these constants are equal to 2.85 and  $-9.65$ , respectively (Liss and Merlivat, 1986). At oxygen saturation levels above 125%, a modification of Eq. 4. was used that takes into account the oxygen surplus,

$$V = 5.9 Sc^{-0.5} (aW + b) (O_s / O_2')^2 \quad (5)$$

where  $O_s$  is the ambient surface oxygen concentration ( $\text{g O}_2 \text{ m}^{-3}$ , 0-10 m), and  $O_2'$  is the oxygen saturation value ( $\text{g O}_2 \text{ m}^{-3}$ ). Schmidt numbers for oxygen were computed from surface temperature data ( $T_s$ ), using a simple analytical expression derived by Stigebrandt (1991):

$$Sc = 1,450 - 71 T_s + 1.1 T_s^2 \quad (6)$$

The vertical diffusive flux of oxygen (DO) was estimated from the equation:

$$D_o = -K_z (\partial O_2 / \partial z) \quad (7)$$

where  $K_z$  is the vertical eddy diffusivity ( $\text{m}^2 \text{ s}^{-1}$ ),  $O_2$  is ambient oxygen concentration ( $\text{g O}_2 \text{ m}^{-3}$ ), and  $z$  is depth (m). The model assumes that the only properties of the stratified water column controlling  $K_z$  are the turbulent kinetic energy dissipation rate ( $\epsilon$ ), and the buoyancy frequency (= Brunt-Väisälä frequency) ( $N$ ):

$$K_z = a \epsilon N^{-2} \quad (8)$$

Various values for parameter  $a$  have been sug-

gested (e.g., Denman and Gargett, 1983). Because of the high stability of the water column in the northern Gulf of Mexico (Rabalais et al., 1999), we adopted the value of 0.8 (Weinstock, 1978). This value is thought to be valid for strong and intermediate stratification, where the Cox number is less than 2500 (Caldwell et al., 1980). We assumed that the turbulent energy dissipation rate ( $\epsilon$ ) at the depth of 10 m is in the range of  $10^{-7} \text{ m}^2 \text{ s}^{-3}$ , which is likely to be an upper estimate. Corresponding values were obtained from microstructure measurements in the upper ocean during high winds (Dillon and Caldwell, 1980). Buoyancy frequency  $N$  ( $\text{s}^{-1}$ ) was calculated from the expression

$$N^2 = (g/\rho_w) (\partial\rho/\partial z) \quad (9)$$

where  $g$  is the acceleration due to gravity ( $9.81 \text{ m s}^{-2}$ ),  $\rho_w$  is the average density of the water column ( $\text{kg m}^{-3}$ ), and  $\partial\rho/\partial z$  is the vertical density gradient ( $\text{kg m}^{-4}$ ) between the upper (0-10 m) and the lower (10-20 m) water column. Vertical density gradients were computed from a multiple regression of  $\partial\rho$  on salinity and temperature ( $r^2 = 0.85$ ;  $p < 0.001$ ), as explained in Justić et al. (1996).

The net productivity of the surface water column ( $NP$ ;  $\text{gC m}^{-2} \text{d}^{-1}$ ) was computed from the time-delayed regression model developed by Justić et al. (1996):

$$NP_t = -0.34 + 3.93 \times 10^{-7} (N\text{-NO}_3)_{t-1} \quad (10)$$

where  $N\text{-NO}_3$  is the nitrate flux of the Mississippi River ( $10^6 \text{ kg d}^{-1}$ ), and subscripts  $t$  and  $t-1$  denote the current and the preceding month, respectively. The Eq. 10 was developed based on the Mississippi River nitrate flux data for the period 1985-1992 and net productivity estimates at station C6. The cross correlation coefficient (ccc) indicated that the two time series are highly coherent (ccc = 0.73;  $p < 0.01$ ), and that a time-delay of one month is justified (Justić et al., 1997). Conversion of carbon to oxygen equivalents, so that Eq. 1 is dimensionally correct, was carried out using a ratio of 3.47 by weight (mol. C: mol.  $\text{O}_2 = 106 : 138$ , Redfield et al., 1963).

The rate of respiration ( $R$ ) in the lower water column is proportional to the amount of detritus present, and may be described by the first order decay relation

$$R = -k C_s \quad (11)$$

where  $k$  is the decay constant, or respiration constant, ( $\text{d}^{-1}$ ), and  $C_s$  is the pool of sedimentary organic carbon that is available for decomposition ( $\text{g C m}^{-2}$ ). Vertical flux of organic detritus to the bottom waters is described as a function of surface net productivity at some earlier time. For the northern Gulf of Mexico, Justić et al. (1993) showed that there is a significant coherence ( $r = 0.85$ ;  $P < 0.01$ ) between the net productivity of the upper water column (0-10 m) and the oxygen deficit in the lower water column (10-20 m), implying a time-lag of 1 month. Thus, the respiration rate ( $R$ ) in the lower water column at any given time  $t$  may be expressed in terms of the net productivity rate  $NP(t)$  at some earlier time  $t_0$  (Officer et al. 1984, 1985), so that

$$R(t) = k(t) \int_{-\infty}^t a NP(t_0) \exp\left[-\int_{t_0}^t k(t_1) dt_1\right] dt_0 \quad (12)$$

where the proportionality constant  $\alpha$  describes the fraction of  $NP$  that reaches the lower water column. Carbon uptake during respiration was converted to oxygen equivalents using a ratio of 3.47 by weight (mol. C : mol.  $\text{O}_2 = 106 : 138$ , RQ = 0.77, Redfield et al. 1963). Rabalais et al. (1991) suggested that around 50% of surface primary production may be reaching the bottom ( $\sim 20 \text{ m}$  on average) in the northern Gulf of Mexico. Based on the data for the period 1985-1992, Justić et al. (1997) estimated that the average respiration rate ( $R$ ) of the lower water column (10-20 m) at station C6 accounted for 47% of the  $NP$  in the upper water column (0-10 m). Accordingly, a value of  $\alpha = 0.47$  was used in this study. The respiration constant  $k$  is often described by an empirical relationship of the form

$$k = k_0 (T_b/10)^a (O_b/6)^b \quad (13)$$

where  $k_0$  is the multiplying factor ( $\text{d}^{-1}$ ),  $T_b$  is the average temperature of the lower water column (10-20 m,  $^{\circ}\text{C}$ ),  $O_b$  is the average oxygen concentration ( $\text{g O}_2 \text{ m}^{-3} = \text{mg O}_2 \text{ l}^{-1}$ ) in the lower water column, and  $a$  and  $b$  are constants. The  $O_b$  value was computed by dividing the  $O_{ib}$  value (Eq. 2) with the thickness of the lower water column (10 m). In this study, we have adopted the values of  $k_0 = 0.008$ ,  $a = 1.1$ , and  $b = 0.4$ . The later two values were originally proposed for the Chesapeake Bay (Officer et al., 1985), while the  $k_0$  value of 0.008 was the upper limit of the range

of values for this constant proposed for Patuxent estuary (Boynton et al., 1980). Those estimates for  $k_o$ ,  $a$  and  $b$  provided a good fit to the benthic and epibenthic respiration rates observed in the coastal waters of the northern Gulf of Mexico.

Eq. 12. defines the net productivity as a surrogate for excess carbon in the upper water column (0-10 m) that is available for export to the lower water column (10-20 m). The instantaneous vertical carbon flux ( $S_f$ ,  $gC\ m^{-2}d^{-1}$ ) due to the sedimentation of organic material from the upper water column may be described as

$$S_f(t) = \alpha NP(t_o) \quad (14)$$

Accordingly, the balance equation for organic carbon in sediments ( $C_s$ ,  $gC\ m^{-2}$ ) may be written as

$$\partial C_s / \partial t = S_f(t) - R(t) - E_c \quad (15)$$

where  $R(t)$  is the respiration rate in the lower water column, expressed here in terms of carbon equivalents ( $gC\ m^{-2}\ d^{-1}$ ), and  $E_c$  ( $gC\ m^{-2}\ d^{-1}$ ) is the loss of sedimentary carbon due to resuspension and export. The continental shelf of the northern Gulf of Mexico is a highly dynamic system where wind-driven sediment resuspension may be a driving force in exporting sediments to the outer shelf and slope. Seasonal deposition rates can be locally high, but decadal sediment accumulation rates are significantly lower (Wiseman et al., 1999). In computing the organic carbon accumulation rates, we assumed that 50% of the sedimented organic carbon that is not subsequently decomposed is ultimately exported from the study area.

Equations 1, 2 and 15 represent a system of coupled, non-linear, ordinary differential equations. In simulations experiments, the equations were solved using the Runge-Kutta integration method of the fourth order, and an integration step of 0.01 month (0.3 days).

## 8.3 Results

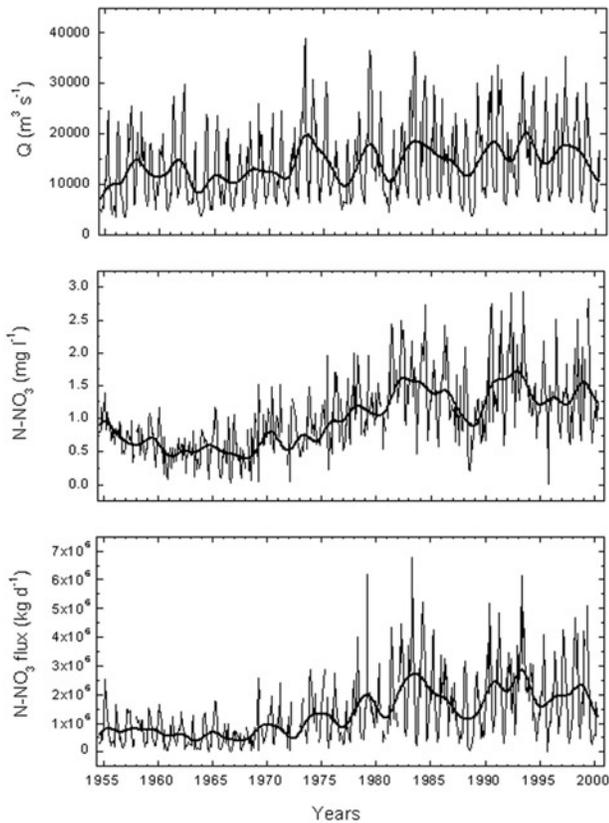
### 8.3.1 Trends in the Mississippi River Discharge and Nitrate Flux 1954-2000

The average nitrate flux of the lower Mississippi River increased 3.3-fold between 1954-1967 and 1983-2000 (Figure 3). During the same time period, the average nitrate concentration increased 2.3-fold

while the average discharge increased 40%. Partitioning of the observed trend in nitrate flux among the two flux components, nitrate concentration and discharge, revealed that about 80% of the observed increase in flux could be explained by only the increase in nitrate concentration (Figure 4). This indicates that historical increase in the anthropogenic nutrient inputs has had a far greater impact on the lower Mississippi River nitrate flux than a change in climate. Nevertheless, the influence of climatic factors on nitrate flux has been significant and may further increase as a result of global climate change. This argument is supported by two lines of evidence: First, the residual component of nitrate flux, obtained by removing a trend from the time-series, is controlled primarily by the variability in discharge, i.e. climatic factors. Also, there is a highly significant relationship between discharge and nitrate flux for the period 1983-2000 (Figure 5). Linearity in discharge-flux relationships was also observed in the 1954-1967 and 1968-1982 data subsets. The regression slopes for 1954-1967 ( $\alpha=62.35$ ;  $R=0.81$ ;  $p<0.01$ ) and 1968-1982 ( $\alpha=96.12$ ;  $R=0.69$ ;  $p<0.01$ ), however, were significantly lower ( $p<0.01$ ) in comparison with the slope for the 1983-2000 ( $\alpha=146.76$ ;  $R=0.86$ ;  $p<0.01$ ). Clearly, these differences between the three periods are attributable to historically lower nitrate concentrations during 1954-1967 and 1968-1982 (Figure 3).

Sensitivity of nitrate flux to altered discharge was estimated by computing the percent increase in flux that corresponds to a 1% increase in the average discharge for the 1983-2000 ( $15,874\ m^3\ s^{-1}$ ). Our estimated flux sensitivity value was 1.16%, which was somewhat higher than a value of 1.1% that was reported for the lower Mississippi River by Alexander et al. (1996). In general, flux sensitivity values above 1% are indicative of rivers where runoff, either from agricultural, urban, or forested lands, is the predominant source of nitrate (Alexander et al. 1996). Because of the amplified influence of discharge on nitrate flux in these rivers (Figure 5), nutrient management efforts for the Mississippi River (Brezonik et al., 1999; Goolsby et al., 1999) may be more challenging.

While a detailed discussion of complex watershed processes affecting nitrate flux remains beyond the scope of this paper, there are several ways in which these anticipated changes in precipitation and increases in discharge may influence nitrate flux. First, the higher precipitation would leach more

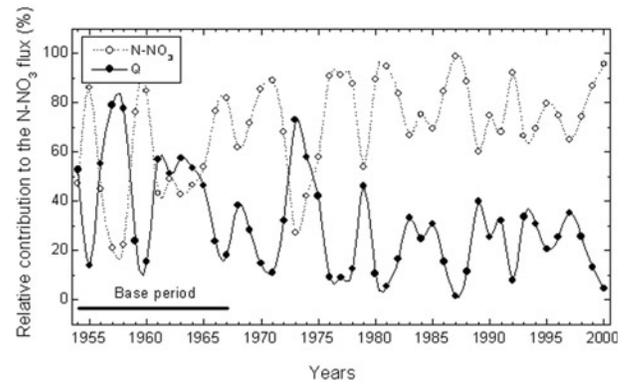


**Figure 3.** Monthly averages (1954-2000) of the lower Mississippi River discharge ( $Q$ ), nitrate concentration ( $N-NO_3$ ), and nitrate flux ( $N-NO_3$  flux). Smoothed curves are estimated third order polynomial fits on 12-month weighted averages.

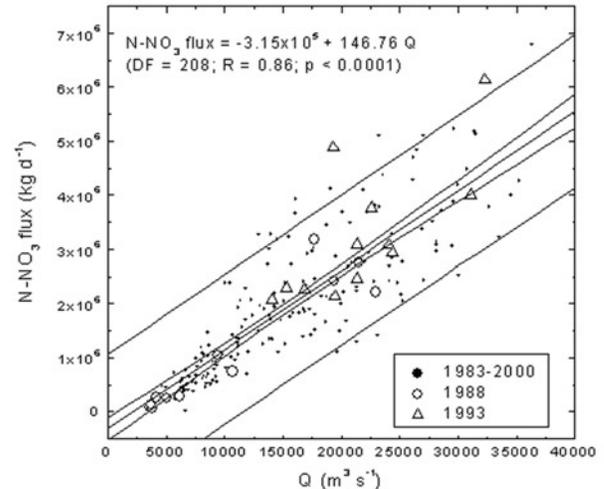
nitrate flux from soils into tributaries and the mainstem of the Mississippi River (Goolsby et al., 1999). Second, unless riverine nitrate concentrations are reduced, the higher discharge would necessarily lead to an increased nitrate flux (Figure 5). Lastly, the higher discharge would decrease the water residence times in canals, lakes and small streams in the upper parts of the watershed. This would substantially reduce nitrogen losses due to denitrification (Howarth et al., 1996; Alexander et al., 2000), and ultimately result in a higher nitrate concentration in the mainstem of the Mississippi River. Thus, it is impossible at this time to reliably predict future trends in nitrate concentration of the lower Mississippi River. Nevertheless, a hypothetical 20% increase in the Mississippi River discharge (Miller and Russell, 1992) would result in a nitrate flux that greatly surpasses the historical flux values for the period 1954-2000 (Figures 3 and 6).

### 8.3.2 Model Simulations

Calibration results for surface and bottom oxygen concentrations are illustrated in Figure 7. The model

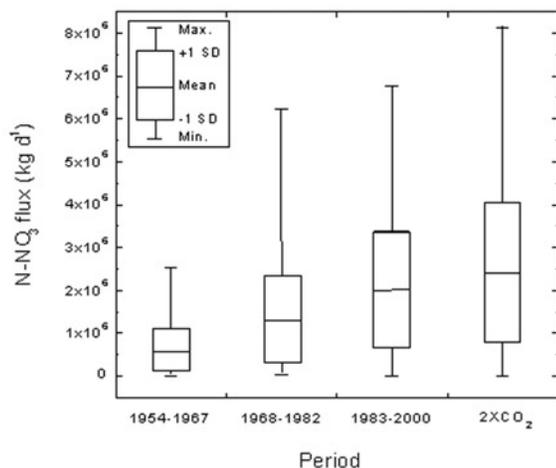


**Figure 4.** Partitioning of observed trends in the  $N-NO_3$  flux of the lower Mississippi River into anthropogenic ( $N-NO_3$  concentration) and climatic (discharge,  $Q$ ), components. Symbols indicate relative contributions (%) of the two flux components, based on deviations from averages of the base period 1954-1967.



**Figure 5.** Regression of nitrate flux ( $N-NO_3$  flux) on discharge ( $Q$ ) for the period 1983-2000. The values for the drought of 1988 and the flood of 1993 are shown in different symbols. The two pairs of lines parallel to the regression line denote the 95% confidence limits and the 95% prediction limits, respectively.

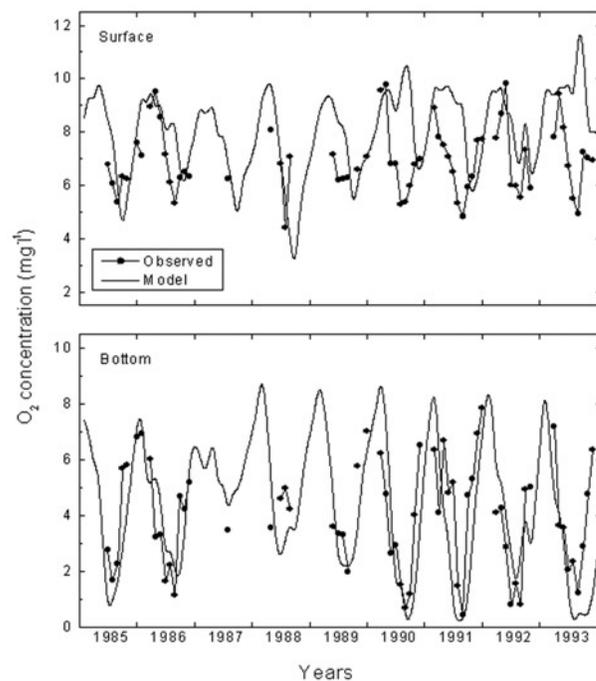
was calibrated on the basis of a 1985-1993 data set for the Mississippi River and the Northern Gulf of Mexico. The 1985 – 1993 period included three average hydrologic years (1985, 1986 and 1989), a record flood year (1993, a 62-yr record high discharge), two years with above average discharge (1990 and 1991), three years with below average discharge (1987, 1988 and 1992), and a record drought year (1988, a 52-yr record low discharge). Given the time-span of the data, we considered the 1985 – 1993 data subset to be suitable for model calibration. A sensitivity analysis revealed that the model is highly sensitive to



**Figure 6.** Box-plots showing nitrate flux ( $\text{N-NO}_3$  flux) statistics for 1954-1967, 1968-1982, and 1983-2000, as well as model projections for a  $2\times\text{CO}_2$  climate.

external forcing, yet sufficiently robust to withstand an order of magnitude change in the nitrate flux between successive months, such as those encountered during the flood of 1993. For the bottom layer (10 – 20 m), the model agrees exceptionally well with the observed values, both in terms of the annual and interannual variability. The agreement between the model and the data is also very good for the surface layer (0 – 10 m), with the exception of 1990, 1991, and 1993, for which the predicted summertime oxygen concentrations were somewhat higher than observed.

Model simulations for a station within the core of the present day hypoxic zone (C6; Figure 1) indicated a decadal trend of increase in the oxygen concentrations in the upper water column (0 – 10 m) and decrease in the lower water column (10 – 20 m) (Figure 8). The annual average oxygen concentration at 10-20 m depth decreased from  $6.6 \text{ mg l}^{-1}$  in 1955 – 1965 to  $4.2 \text{ mg l}^{-1}$  during 1990 – 2000. As expected, the differences in summertime oxygen concentrations between those periods are even greater. The average oxygen concentration in the lower water column during August decreased from  $5.8 \text{ mg l}^{-1}$  in 1955 – 1965 to  $0.9 \text{ mg l}^{-1}$  during 1990 – 2000. The model has identified the mid 1970s as a start of the recurring hypoxia ( $< 2 \text{ mg O}_2 \text{ l}^{-1}$ ) in the lower water column. This result should, however, be interpreted with caution, because the model only predicts the average oxygen concentration for the entire lower water column. It is probable that hypoxia in the near bottom waters was sporadically present during the late 1960s and early 1970s, and perhaps even earlier than that



**Figure 7.** Observed and predicted monthly averages of surface (0-10 m) and bottom (10-20 m) oxygen concentrations at station C6 for the period June 1985-November 1993.

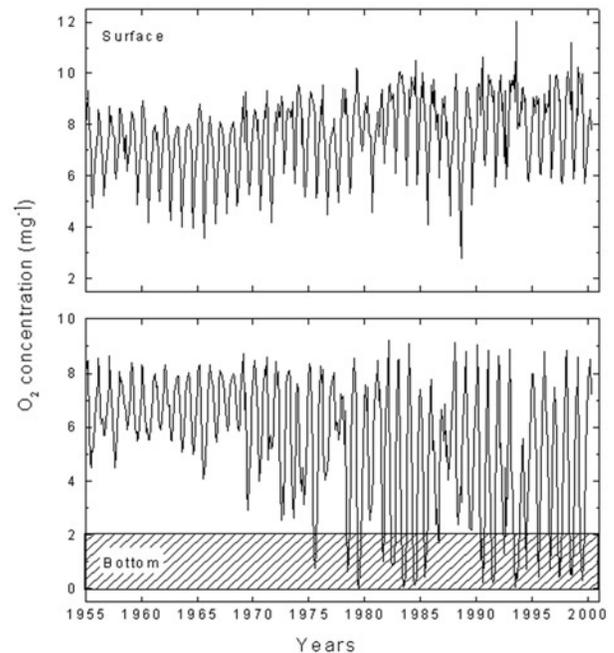
(Figure 8). The model results also suggest that the annual average oxygen concentration of the upper water column (0 – 10 m) increased from  $7.0 \text{ mg l}^{-1}$  in 1955 – 1965 to  $8.4 \text{ mg l}^{-1}$  during 1990 – 2000.

### 8.3.3 Future Scenarios: An Inverse Approach to Scientific Controversy

Assessments of future climate change scenarios for the northern Gulf of Mexico are complicated by the fact that model projections for the Mississippi River runoff are highly variable. The Canadian and the Hadley models projected a 30% decrease and a 40% increase, respectively, by the year 2099. In a series of modeling studies we investigated the impacts of various climate change scenarios and different nutrient control strategies for the Mississippi River (Table 1) on the Gulf's hypoxic zone. Model simulations were conducted in a hindcast mode, by increasing or decreasing the Mississippi River discharge, nitrate concentration, and nitrate flux relative to the observed monthly averages for the period 1954-2000 (Figure 3). Model results are summarized in Table 2. Nominal model simulation for the period 1954-2000 predicted 19 years with moderate hypoxia ( $< 2 \text{ mg O}_2 \text{ l}^{-1}$ ) and 16 years with severe hypoxia ( $< 1 \text{ mg O}_2 \text{ l}^{-1}$ ). A 30% decrease in the Mississippi River discharge for the same period would have significantly reduced

the number of years with moderate and severe hypoxia to 8 and 4, respectively (Table 2). For a scenario with 4°C increase in the average annual temperature and a 20% increase in the average Mississippi River discharge, the model predicts 31 year with moderate and 26 years with severe hypoxia. Importantly, model simulations suggest that pronounced hypoxia would not develop if the nitrate concentrations would had remained unchanged with respect to the period 1954 – 1967 ( $0.61 \text{ mg N l}^{-1}$ ). Thus, depending on future climate change scenarios and nutrient control strategies, hypoxia in the northern Gulf of Mexico may become more or less severe.

Nutrient reductions in the Mississippi River and their effects on the Gulf of Mexico hypoxic zone are currently high-profile concerns in scientific and public forums (e.g., Brezonik et al., 1999; Goolsby et al., 1999; Rabalais et al., 1999). There are examples of successful nutrient control programs, whose implementation in coastal and estuarine ecosystems under stress have resulted in a reversal of the eutrophication trend (Rosenberg, 1976; Cherfas, 1990; Johansson and Lewis, 1992). Those nutrient control programs, however, have been implemented in circumstances where external nutrient inputs were much lower than that of the Mississippi River, and their results may not be applicable to the northern Gulf of Mexico. Nevertheless, model results (Table 2) suggest that a large-scale reduction (~30%) in nitrogen concentration of the Mississippi River would eventually diminish the severity of hypoxia in the northern Gulf of Mexico. Because of the sensitivity of riverine nitrate flux to climatic influences, global climate change would likely have important implications for the northern Gulf of Mexico. Consequently, nutrient control efforts for the Mississippi River watershed that would be based solely on achieving a specific reduction in the non-point source loading, may have a limited success in controlling the eutrophication and hypoxia in the northern Gulf of Mexico. If, for example, the Mississippi River discharge would increase 20%, as predicted in some model scenarios (Miller and Russell, 1992), a reduction in nitrate flux in excess of 20% would be required only to prevent the eutrophication from worsening.



**Figure 8.** Simulated changes in the average surface (0-10 m) and bottom (10-20 m) oxygen concentration at station C6 for the period January 1955-May 2000. Shaded area in the lower chart denotes hypoxic conditions ( $< 2 \text{ mg O}_2 \text{ l}^{-1}$ ) in bottom waters.

### 8.3.4 Implications for Fisheries Food Webs

The effects of hypoxia on demersal and benthic communities will likely intensify as hypoxia stress worsens, due to either increase in areal extent, severity, or duration. Catches in trawls are negligible when the bottom dissolved oxygen concentration falls below  $2 \text{ mg l}^{-1}$  (Pavela et al., 1983; Renaud, 1986). Motile fishes and invertebrates migrate from the area or into the upper water column. Mass mortalities are likely, however, if they are trapped against the shore by a large anoxic water mass. This could become a serious problem in the northern Gulf of Mexico, if the areal extent of hypoxia increases. Heavy mortalities occur in the benthic infauna and species diversity is drastically reduced when ambient oxygen concentrations decrease below  $0.5 \text{ mg l}^{-1}$  (Gaston, 1985; Boesch and Rabalais, 1991; Rabalais et al., 1993; Rabalais et al., 2001). Presently, there is some recovery of the benthic community in the fall, and further recruitment in the subsequent spring. However, the overall structure of the benthic community is shifted in species composition and age structure, to a smaller-sized, lower biomass, polychaete dominated fauna. An increase in areal extent and severity of hypoxia will decrease recovery rates

and also reduce food resources (infauna) for recolonizing demersal groups, such as the commercially important penaeid shrimps. Further, alterations in benthic community structure will have implications for sedimentary processes, benthic pelagic coupling, and energy flow. Major alterations in benthic communities due to hypoxia stress, especially a reduction in diversity and biomass, will certainly alter the productivity base that leads to fishery stocks.

Pelagic species could also be impacted if hypoxic conditions extend high into the water column affecting their distribution and movement patterns. From August 1995 to February 1997, for example, dual beam hydroacoustics were employed on quarterly research trips to measure the density and in situ target strengths of fishes associated with a petroleum platform, South Timbalier 54 (Stanley and Wilson, 1998). During the survey in July 1996, observed dissolved oxygen levels were below 2 mg O<sub>2</sub> l<sup>-1</sup> in depths of 15 – 22 m. Within these depths fish density was essentially zero while from the surface to 15 m elevated fish densities were observed. Other potential fisheries impacts include: concentrated fishing effort with increased bycatch, local nearshore mortality of finfish and shellfish, and decreased recruitment due to impacts on zooplankton assemblages (Hanifen and Romaire, 1998; Craig et al., 2001).

Freshwater runoff, via its negative effect on salinity, is a critical parameter governing biological processes in the northern Gulf of Mexico estuaries and coastal waters. The annual yield of penaeid shrimp in the Gulf of Mexico is inversely related to the annual discharge of the Mississippi River, perhaps because of reduced estuarine salinities at high river flows (Turner, 1992; Mulholland et al., 1997). Penaeid shrimp postlarvae are mostly limited to estuarine habitats with salinities greater than 10 psu. In the case of Louisiana, salinities are primarily influenced by river flow and precipitation. Mississippi River discharge affects the lower estuaries, while rainfall affects the upper bays and estuaries. With heavy rains or high river flow, salinities in the marshes are reduced. If salinities are reduced beyond acceptable conditions, postlarvae do not move as high into the marshes, ultimately influencing adult stock. This is important because if the freshwater runoff increases as a result of global warming, estuarine salinities could decrease, possibly leading to reduced yields of shrimp and other species favoring higher estuarine salinities. Temperature also influences growth. Growth is inhibited in waters with a temperature below 20°C. Global warming may expand this region of high shrimp yield northwards, increasing shrimp harvest throughout the region, assuming that salt

**Table 1. Investigated model scenarios. Changes denote deviations from the observed monthly averages for the period 1954-2000.**

Nominal model	Observed monthly averages (1954-2000) of the Mississippi River discharge, nitrate concentration and nitrate flux.
Scenario 1.	30% decrease in the average Mississippi River discharge (Wolock and McCabe, 1999).
Scenario 2.	20% increase in the average Mississippi River discharge (Miller and Russel, 1992; Wolock and McCabe, 1999).
Scenario 3.	4°C increase in the average annual temperature of the northern Gulf of Mexico.
Scenario 4.	4°C increase in the average annual temperature of the northern Gulf of Mexico and 20% increase in the average Mississippi River discharge.
Scenario 5.	Mississippi River nitrate concentration remains unchanged with respect to the average for the period 1954-1967 (0.61 mg N L <sup>-1</sup> ).
Scenario 6.	30% reduction in the Mississippi River nitrate concentration (proposed management scenario).

**Table 2. Model results for selected scenarios described in Table 1. The simulation interval was 45 years (1954 – 2000).**

	Number of years with moderate hypoxia (< 2 mg O <sub>2</sub> l <sup>-1</sup> )	Number of years with severe hypoxia (< 1 mg O <sub>2</sub> l <sup>-1</sup> )
Nominal model	19	16
Scenario 1.	8	4
Scenario 2.	26	20
Scenario 3.	25	19
Scenario 4.	31	26
Scenario 5.	0	0
Scenario 6.	12	7

marsh nursery areas are not negatively affected by other factors, such as water level changes.

Louisiana commercial and recreational fisheries depend on life cycles of species located within shallow continental shelf waters that overlap with the hypoxic zone. Fishery-independent surveys reveal reduction or absence of shrimp in hypoxic waters (Craig et al., 2001). Both abundance and biomass of fishes and shrimp are significantly reduced where oxygen concentrations in bottom water fall below 2 mg l<sup>-1</sup> (Renaud, 1986). Under experimental conditions, white shrimp avoid water with dissolved oxygen concentrations below 1.5 mg l<sup>-1</sup> and brown shrimp are even more sensitive, avoiding water with oxygen concentrations below 2.0 mg l<sup>-1</sup>. The ability to detect and avoid hypoxic water leads to an observed blocking effect on juvenile shrimp emigrating from inshore nurseries to offshore feeding and spawning grounds. The life cycle of shrimp involves offshore (Gulf shelf) and inshore (estuarine) phases. Adults spawn on the Louisiana and Texas shelf. Resulting larvae immigrate as plankton via currents into coastal estuaries. Within the estuaries, postlarvae metamorphose into small juvenile shrimp that are benthic in habit. After about two months, intermediate size juveniles emigrate from the nursery and return to the outer shelf to complete their growth

into adults. The life cycle from egg to adult takes about 6 months. Larval, postlarval, sub-adult, and adult shrimp utilize habitats overlapping with the hypoxic zone, and, depending on the stage within the life cycle, their spawning grounds, feeding grounds, or migratory pathways could be impacted. A negative correlation between shrimp catch and the presence of hypoxia corroborates interference with shrimp migration (Zimmerman and Nance, 2001). In areas where hypoxia is widespread and persistent, shrimp catch is always low. In Louisiana, the nearshore concentration of shrimp is always higher than offshore, possibly because hypoxia impedes offshore movement. Since nearshore catches are comprised of young shrimp, productivity in growth to a larger size is lost. Production models conservatively estimate that several million pounds of shrimp are lost annually due to early harvest. Since hypoxia blocks access of migrating juvenile shrimp to offshore feeding grounds, losses in production due to lost feeding are also predictably large. These direct and indirect fisheries losses would be exacerbated if hypoxia expands in space and time as a result of global climate change.

## 8.4 Conclusions

The average nitrate flux of the lower Mississippi River increased from about  $0.6 \times 10^6$  kg d<sup>-1</sup> in 1954-1967 to about  $2 \times 10^6$  kg d<sup>-1</sup> in 1983-2000, which is a 3.3-fold increase. During the same time period, the average nitrate concentration increased 2.3-fold (from 0.61 mg N-NO<sub>3</sub> l<sup>-1</sup> to 1.37 mg N-NO<sub>3</sub> l<sup>-1</sup>), while the average discharge increased 40% (from 11,381 m<sup>3</sup> s<sup>-1</sup> to 15,874 m<sup>3</sup> s<sup>-1</sup>). Partitioning of the total increase in nitrate flux among the two flux components revealed that about 80% of the observed increase in flux may be explained by the increase in nitrate concentration. Nevertheless, the residual component of nitrate flux is controlled primarily by the variability in runoff, i.e., climatic factors. Nitrate concentration is also highly correlated with discharge at the low end of the discharge spectrum, up to about 13,000 m<sup>3</sup> s<sup>-1</sup>. This peculiar relationship is clearly affecting nitrate flux, primarily by amplifying variations in flux between flood and drought years. Consequently, future changes in the frequency of droughts or floods, or an overall change in freshwater discharge, may substantially alter the flux of nitrate to the northern Gulf of Mexico. Model simulations for the northern Gulf of Mexico indicated that bottom water hypoxia intensified about 30 years ago, as a probable consequence of increased net productivity and increase in the vertical flux of the organic carbon. Apparently, the long-term increase in riverine nutrient fluxes has been the primary factor controlling this historical decline in oxygen concentrations. Thus, riverine nutrient fluxes, via their influence on net productivity of the upper water column, play a major role in controlling the development of bottom water hypoxia and accumulation of organic carbon in coastal sediments.

Assessment of future climate change scenarios for the northern Gulf of Mexico is complicated by the fact that model projections for the Mississippi River runoff are highly variable. The Canadian and the Hadley models projected a 30% decrease and a 40% increase, respectively, by the year 2099. Nevertheless, model simulations suggest that altered freshwater and nutrient fluxes would have important implications for water column stability, net productivity, and global oxygen cycling in the northern Gulf of Mexico. Nominal model simulation for the period 1954-2000, for example, predicted 19 years with moderate hypoxia (< 2 mg O<sub>2</sub> l<sup>-1</sup>) and 16 years with severe hypoxia (< 1 mg O<sub>2</sub> l<sup>-1</sup>). A 30% decrease in the Mississippi River discharge for the same period

would have significantly reduced the number of years with moderate and severe hypoxia to 8 and 4, respectively. For a scenario with 4°C increase in the average annual temperature and a 20% increase in the average Mississippi River discharge, the model predicts 31 years with moderate and 26 years with severe hypoxia. Importantly, model simulations suggest that pronounced hypoxia would not develop if the nitrate concentrations would have remained unchanged with respect to the period 1954-1967 (0.61 mg N l<sup>-1</sup>). Thus, depending on future climate change scenarios and nutrient control strategies, hypoxia in the northern Gulf of Mexico may become more or less severe.

Nutrient reductions in the Mississippi River and their effects on the Gulf of Mexico hypoxic zone are currently high-profile concerns in scientific and public forums, largely because of the potentially negative consequences that hypoxia may have for habitat functionality and sustainability of the Gulf's coastal fisheries. Our model results suggest that a large-scale reduction (~30%) in nitrogen concentration of the Mississippi River would eventually diminish the severity of hypoxia in the northern Gulf of Mexico. Nevertheless, the areal extent and the severity of hypoxia are very sensitive to climate-induced changes in freshwater and nutrient fluxes. If, for example, the Mississippi River discharge would increase 20%, as predicted in some model scenarios, a reduction in nitrate flux in excess of 20% would be required only to prevent the eutrophication from worsening. Consequently, nutrient control efforts for the Mississippi River watershed that are based solely on achieving a specific reduction in the non-point source loading, may have a limited success in controlling the eutrophication and hypoxia in the northern Gulf of Mexico.

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