Coastal hypoxia responses to remediation

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Abstract

The incidence and intensity of hypoxic waters in coastal aquatic ecosystems has been expanding in recent decades coincident with eutrophication of the coastal zone. Because of the negative effects hypoxia has on many organisms, extensive efforts have been made to reduce the size and duration of hypoxia in many coastal waters. Although it has been broadly assumed that reductions in nutrient loading rates would reverse eutrophication and consequently, hypoxia, recent analyses of historical data from European and North American coastal systems suggest little evidence for simple linear response trajectories. We review existing data, analyses, and models that relate variations in the extent and intensity of hypoxia to changes in loading rates for inorganic nutrients and labile organic matter. We also assess existing knowledge of physical and ecological factors regulating oxygen in coastal marine waters and examine a broad range of examples where hypoxia responses to reductions in nutrient (or organic matter) inputs have been documented. Of the 22 systems identified where concurrent time series of loading and $O_2$ were available, half displayed relatively clear and direct recoveries following remediation. We explored in detail 5 well-studied systems that have exhibited complex, non-linear responses to loading, including apparent “regime shifts.” A summary of these analyses suggests that $O_2$ conditions improved rapidly and linearly in systems where remediation focused on organic inputs from sewage plants, which were the primary drivers of hypoxia. In larger more open systems where diffuse nutrient loads are more important in fueling $O_2$ depletion and where climatic influences are pronounced, responses to remediation tend to follow non-linear trends that may include hysteresis and time-lags. Improved understanding of hypoxia remediation requires that future studies use comparative approaches and consider multiple regulating factors including: (1) the dominant temporal scales of the hypoxia, (2) the relative contributions of inorganic and organic nutrients, (3) the influence of shifts in climatic and oceanographic processes, and (4) the roles of feedback interactions whereby $O_2$-sensitive biogeochemistry, food-webs, and habitats influence the nutrient and algal
1 Introduction

Depletion of dissolved oxygen from coastal waters is a widespread phenomenon that appears to be growing globally (Rabalais and Gilbert, 2009). There is considerable interest in this phenomenon because low oxygen causes physiological stress for most marine metazoans. Oxygen concentrations below approximately 20% saturation (“hypoxia”=O₂<62.5 µM) interrupt normal metabolism and behavior of fish and invertebrates causing reduced growth and increased mortality (e.g., Diaz, 2001). Extended periods of hypoxia and relatively brief exposure to anoxia (zero O₂) tend to cause mortality for many marine animals (e.g., Vaquer-Sunyer and Duarte 2008). Hypoxia may also affect predator-prey interactions and food web structures, with low O₂ zones providing more tolerant organisms extended habitat for foraging and refuge from predation (e.g., Nestlerode and Diaz, 1998; Decker et al., 2004). Low oxygen levels also alter the oxidation-reduction balance in marine sediments and associated biogeochemistry. For example, hypoxia and anoxia tend to reduce or eliminate coupled nitrification-denitrification (e.g., Kemp et al., 1990) and inorganic phosphorus sorption to metal oxide-hydroxide complexes (Slomp and van Cappellen, 2007; Middelburg and Levin, 2009).

Dissolved oxygen depletion results at various time and space scales from an imbalance between biological and physical sources and sinks for O₂. In shallow tidal rivers and lagoons with well-mixed water columns, the balance between ecosystem photosynthesis, respiration, and air-sea exchange generally controls O₂ concentrations (D’Avanzo and Kremer, 1994). In deeper systems with stratified water columns, concentrations in bottom waters are usually set by the balance between O₂ respiration and physical O₂ replenishment from both vertical mixing across the pycnocline and horizontal transport from oxygenated seaward waters (e.g., Kemp et al., 1992). For stratified systems, O₂ concentrations are generally lower in the bottom layer, par-
particularly in warmer months. In protected shallow coastal waters, \( \text{O}_2 \) generally varies markedly on hourly scales, with hypoxia often occurring at night when photosynthesis is zero (e.g., Tyler et al., 2008). In slightly deeper microtidal systems, periodic stratification may allow episodic hypoxia to occur on daily-to-weekly scales with wind-driven changes in vertical mixing (Park et al., 2007). In much deeper (>100 m) coastal seas and fjords, strongly stratified water columns result in virtually permanent hypoxia (or anoxia) that tends to change only in its size and position with decadal-scale variations in circulation (e.g., Zillén et al., 2008). Low \( \text{O}_2 \) waters in bottom layers of partially stratified estuaries and river-plume-shelf systems generally appear only in summer when stratification and respiration are strongest (e.g., Rabalais and Gilbert, 2009).

There is mounting evidence that eutrophication (i.e., anthropogenic nutrient and organic enrichment of tidal waters) is contributing to the expansion of occurrence, intensity and duration of hypoxic conditions in coastal waters worldwide (e.g., Diaz and Rosenberg, 2008; Rabalais and Gilbert, 2009). Additional nutrients tend to fertilize growth, sinking and decomposition of phytoplankton in bottom waters of estuaries, bays, lagoons and inland seas. For many coastal systems in the industrialized regions of the world, there have been major socio-economic commitments to remediate hypoxic zones by reducing nutrient loading from the adjacent catchment and overlying atmosphere (Boesch, 2002; Carstensen et al., 2006). Although reduction in anthropogenic nutrient loading to coastal systems is the primary means that has been employed for remediation of hypoxia associated with eutrophication, biomanipulation approaches have also been suggested. For example, there has been much discussion and analysis of potential impacts of re-establishing diminished populations of benthic filter-feeding bivalve populations as a means for reversing eutrophication and hypoxia by reducing phytoplankton biomass (e.g., Cerco and Noel, 2007; Petersen et al., 2008). In addition, engineering solutions (including enhanced vertical mixing, increased horizontal exchange and mechanical air-bubbling) have been discussed as options for mitigating human-induced coastal hypoxia (Stigebrandt and Gustafsson, 2007; Conley et al., 2009c). Although it has been broadly assumed that reductions in nutrient loading
rates would reverse associated eutrophication, recent analyses of historical data from European and North American coastal systems suggest little evidence for simple linear response trajectories (Duarte et al., 2009; Conley et al., 2009b).

The purpose of this paper is to review existing data, analyses and models that relate variations in the extent and intensity of hypoxia to changes in nutrient and labile organic loading. We review existing knowledge on physical and ecological factors regulating oxygen conditions in coastal marine waters and examine a broad range of examples where hypoxia responses to nutrient (or organic matter) reduction have been documented. We also focus on five large, well-studied coastal marine ecosystems that have exhibited non-linear responses to changes in nutrient loading, and we discuss key mechanisms that have been suggested to explain the hypoxia response trajectories. We conclude with summary statements of implications for remediating and managing low O\textsubscript{2} waters.

2 External factors controlling hypoxia

Although recent decades have seen widespread observations of hypoxic conditions in coastal marine waters worldwide (e.g., Diaz and Rosenberg, 2008), the relative importance of specific physical and ecological conditions in regulating oxygen differs substantially among these diverse systems. Therefore, it is expected that responses of hypoxic coastal waters to remediation will also differ. We anticipate that those systems that are influenced most strongly by human activities will be most likely to respond to reductions in anthropogenic influence.

2.1 Typology of coastal hypoxia

Drawing from previous hypoxia classification schemes based on duration and dominant time-scales of low oxygen (e.g., Diaz and Rosenberg, 2008), we define four broad categories of hypoxia: (1) permanent, (2) persistent seasonal, both stratified...
and vertically mixed, (3) episodic, and (4) diel. Permanent hypoxia occurs primarily in shelf regions, large fjords, and inland seas in which strong stratification isolates the bottom layer of deep water columns (>100 m), leading to persistent bottom-water hypoxia/anoxia (e.g., Helly and Levin, 2004; Gilbert et al., 2005) that tends to change only in its size and position with annual-to-decadal scale variations in circulation (e.g., Helly and Levin, 2004; Zillén et al., 2008; Chan et al., 2008). Persistent seasonal hypoxia occurs in many stratified temperate estuarine and shelf regions where the combination of spring flow and summer heat strengthen stratification, promote phytoplankton growth and stimulate respiration of sinking organic matter (e.g., Rabalais and Gilbert, 2009). Seasonal hypoxia may also occur in shallow well-mixed estuaries and tidal rivers that are heavily loaded with large inputs of labile organic material that is respired in warmer months (e.g., Soerter et al., 2006). Episodic hypoxia tends to occur at irregular intervals (weeks to decades) in productive, shallow (5–15 m), weakly-stratified microtidal coastal systems that are generally subjected to wind mixing. These systems are susceptible to occasional hypoxic conditions that are terminated by frontal wind events (e.g., Stanley and Nixon, 1991); however, they may be prolonged by extended warm calm weather (Møhlenberg, 1999) or exacerbated following major storm events that deliver large pulsed organic loading (Peierls et al., 2003). Diel hypoxia tends to occur in shallow productive lagoons and bays, when night-time respiration of organic matter produced during the day exceeds O\textsubscript{2} replenishment via air-sea exchange. Typically, daytime O\textsubscript{2} levels in these shallow systems are high (often supersaturated) because of strong photosynthetic O\textsubscript{2} production. Although this paper considers all types of hypoxia, it generally focuses on coastal systems with seasonal hypoxia (persistent or variable) because these systems tend to be well-studied and are often heavily influenced by human activities. In contrast, systems with diel hypoxia are less studied, while systems with permanent hypoxia tend to be dominated by natural processes that are difficult or impossible to remediate.
2.2 Factors driving physical and ecological processes

In many coastal systems, density stratification is sufficient to create a bottom layer isolated from surface waters and impede downward mixing of O$_2$ from surface waters, thereby reducing physical replenishment and allowing depletion of bottom water O$_2$ (e.g., Kemp et al., 1992, 2005). Buoyancy of the upper layer is increased and stratification is strengthened by seasonal inputs of freshwater (Boicourt, 1992) and warming of surface waters (e.g., Welsh and Eller, 1991). Relatively weak stratification in systems such as the Neuse River estuary, Long Island Sound, and Mobile Bay can be disrupted by typical summer wind events (e.g., Turner et al., 1987; Stanley and Nixon, 1992; O’Donnell et al., 2008). In any given year, stronger stratification, created by larger freshwater input or warmer surface water, is more resistant to disruption by wind events (Lin et al., 2008). Ventilation of bottom-water hypoxia may involve relatively complex mechanisms, where for example wind stress induces straining of density fields (e.g., Scully et al., 2005), lateral tilting of the pycnocline (Malone et al., 1986), alteration of far-field coastal circulation (e.g., Wiseman et al., 1997), or interaction with spring-neap tidal cycles (Sharples et al., 1994). In stratified systems with estuarine circulation, bottom-water O$_2$ pools are also replenished by landward transport of O$_2$-rich water from downstream sources (e.g., Kuo et al., 1991; Kemp et al., 1992). Because hypoxia in stratified coastal systems is confined to the bottom layer, respiration must be fueled by labile organic matter, typically organic particles sinking from the upper water column (e.g., Hagy, 2005; Chen et al., 2006; 2009).

Although water-column stratification is a key control on persistent seasonal hypoxia for many systems, other well-mixed coastal waters experience intermittent or persistent hypoxic conditions that are confined to the warm season. For example, vertically mixed shallow brackish tidal rivers and saline lagoons may experience relatively continuous summertime low O$_2$ concentrations if they are receiving heavy loads of labile organic wastes. In industrialized regions of the world prior to 1990, and even today in densely populated developing countries, large discharges of organic wastes can cre-
ate high rates of \( \text{O}_2 \) demand that often lead to hypoxic conditions throughout the water column (e.g., Andrews and Rickard, 1980; Soetaert et al., 2006; Diaz and Rosenberg, 2008; Yin et al., 2008). If these systems are relatively turbid due to suspended sediment inputs and resuspension, photosynthesis and associated \( \text{O}_2 \) production would be severely light-limited. In this case, vertical mixing of the water column is typically induced by winds and/or tidal turbulence, and hypoxia results from a sink-source imbalance where community respiration exceeds the rate of \( \text{O}_2 \) replenishment via air-water exchange. In contrast, when shallow, clear-water coastal systems (e.g., lagoons) receive substantial inputs of inorganic nutrients, photosynthetic production (often dominated by benthic plants) represents an important \( \text{O}_2 \) source, leading to diel-scale cycling between supersaturated \( \text{O}_2 \) concentrations during the day and hypoxic conditions at night (e.g., MacPherson et al., 2007; Tyler et al., 2008). Although *diel hypoxia* is generally confined to the warmer summer months, its occurrence and intensity tends to vary on daily-to-weekly time-scales associated with periodic fluctuations in sunlight and tides, as well as rain and wind events (e.g., Shen et al., 2008). There are surprisingly few reports of diel-scale hypoxia in the scientific literature; however, recent evidence suggests that this phenomenon is widespread in shallow eutrophic waters (e.g., Wenner et al., 2004).

Key ecological controls on seasonal hypoxia in coastal waters involve the production and delivery of labile organic matter to the region of \( \text{O}_2 \) depletion. The origin of the organic matter that fuels respiratory \( \text{O}_2 \) sinks can either be from sources within the aquatic system or from external sources, including the adjacent watershed or ocean (Bianchi, 2007). Major external sources of organic material to coastal waters can be derived from runoff of terrestrial plant debris, adjacent phytoplankton biomass from river-borne or oceanic-upwelling sources, and anthropogenic inputs of particulate and dissolved organics, (e.g., sewage effluents in non-industrialized nations). For non-stratified coastal systems, respiration and hypoxia may be driven by inputs of dissolved organic matter (e.g., Andrews and Rickard, 1980; MacPherson et al., 2007). To fuel bottom respiration in stratified waters, however, organic matter must be in the form of
particles capable of sinking to the bottom layer. Most bottom water hypoxia is fueled by sinking of living and detrital phytoplankton cells, whether they are transported from external sources or produced internally in overlying waters. The high rates of particulate organic input generally needed to support bottom-layer hypoxia, however, tend to be from algal production in overlying waters driven by inputs of inorganic nutrients from adjacent sources. Recent reviews of anthropogenic hypoxia suggest that O₂ depletion in stratified coastal waters is most often driven by nutrient-stimulated production of organic matter (e.g., Diaz and Rosenberg, 2008).

Long-term trends and decadal-scale cycles in climatic forcing can also exert control over O₂ concentrations in bottom waters via changes in temperature, salinity, freshwater inputs, and wind stress. For example, recent increases in water temperature (e.g., Nixon et al., 2004), which are expected to continue with increases in atmospheric CO₂ concentrations, will have direct and indirect consequences for hypoxia. The direct effects include decreased solubility of O₂ in water and enhanced respiration rates, while indirect effects include changes in food webs resulting from spatial and temporal shifts in species distribution and abundance (e.g., Najjar et al., 2000; Pyke et al., 2008). In addition, long-term increases in relative sea level occurring in many coastal regions worldwide (Holgate and Woodworth, 2004) may result in elevated bottom water salinities (Hilton et al., 2008), thus potentially enhancing stratification and reducing ventilation of deep waters. Long-term increases or decreases in freshwater input caused by global climate change will influence coastal hypoxia in many coastal systems by increasing or decreasing (respectively) the stratification strength and nutrient delivery rate (e.g. Justic et al., 2003; Arnell, 1999). Lastly, long-term trends and decadal-scale shifts in atmospheric pressure fields and circulation (e.g., Ogi et al., 2003) may alter the magnitude and direction of wind stress, causing changes in vertical mixing and oxygenation of O₂-depleted bottom waters in coastal systems.
3 Internal processes controlling hypoxia

Although external forcing of physical and biological processes has a strong influence on coastal ecosystem dynamics, including the development of hypoxia, internal ecosystem structure and associated processes are also important. For example, internal processes regulate key biogeochemical fluxes, including production and consumption of organic carbon and cycling of inorganic nutrients. These processes, which create positive and negative feedbacks within the ecological system, strongly influence dynamics of dissolved oxygen in coastal water columns (e.g., Kemp et al., 2005). In this section we review important internal ecological processes that, on one hand, may be inhibited by eutrophication and hypoxia and, on the other, are capable themselves of reducing development and expansion of hypoxia.

Oxygen depletion in most stratified coastal systems is ultimately supported by surface layer phytoplankton production and particulate sinking to the bottom layer. Herbivorous grazing in the upper water column tends to impede sinking of algal cells and detritus to the lower layer. However, most marine zooplankton are relatively less effective grazers compared to large-bodied cladocerans in lakes, which can strongly control phytoplankton biomass (e.g., Jeppesen et al., 2007). Marine zooplankton (primarily copepods) are less effective because of lower filtering efficiency and strong top-down control by planktivores (e.g., Roman and Gauzens, 1997; Stock and Dunne, 2009). On the other hand, marine suspension-feeding benthic bivalves can effectively control phytoplankton growth, especially in shallow coastal systems (e.g., Prins et al., 1998; Dame and Olenin, 2005), leading to the suggestion that mussels, oysters and other reef-forming benthic bivalves could potentially regulate phytoplankton sufficiently to reduce hypoxia in eutrophic coastal systems (e.g., Officer et al., 1982; Newell and Ott, 1999). A requirement for this to be effective is that benthic grazers must have access to upper mixed layer water where they can graze rapidly growing cells and retain organic matter in the shallow aerobic waters (e.g., Pomeroy et al., 2006; Newell et al., 2007). Although field-scale documentation of benthic grazing impacts mitigating coastal hy-
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Bottom water oxygen concentrations can influence the balance between decomposition and preservation of organic matter deposited on the seafloor through a variety of complex interactions (e.g., Middelburg and Levin, 2009). The fraction of organic matter deposited on the sediments tends to increase with organic matter deposition rate, possibly because high rates of organic input fuel oxygen depletion, which retards decomposition. This makes it challenging to resolve the relative importance of hypoxia, per se, as a control on decomposition versus physical effects of rapid of burial (e.g., Hedges and Keil, 1995). Numerous experiments where natural organic matter is allowed to decompose under controlled conditions with and without O$_2$ have been generally inconclusive (e.g., Westrich and Berner, 1984); however, recent laboratory and field investigations tend to support the idea that decomposition rates are retarded by absence of O$_2$ due to a range of mechanisms including loss of macrofauna activity and sulfide inhibition of microbial activities (e.g., Middelburg and Levin, 2009). Recent papers have speculated that relatively labile organic matter produced in one year could be buried and preserved under seasonally hypoxic conditions, until it is exposed by subsequent physical disturbance in the following year, when decomposition (and O$_2$ demand) would increase under aerobic conditions (e.g., Turner et al., 2008; Bianchi et al., 2008).
Many of the sediment biogeochemical processes that strongly influence porewater chemistry and associated nutrient recycling and retention in coastal sediments are clearly influenced by low water column O\(_2\) and associated sediment oxidation-reduction (redox) profiles. For both nitrogen (N) and phosphorus (P), benthic recycling efficiency (the fraction of inputs of organic N and P to sediments that efflux back to overlying water) tends to increase with decreasing bottom water O\(_2\) concentrations (e.g., Kemp et al., 2005). Particulate organic nitrogen delivered to bottom water and the sediment surface is decomposed via hydrolysis reactions using one of several available terminal electron acceptors (primarily O\(_2\), NO\(_3\), Mn (III, IV), Fe (III), and SO\(_4^{2-}\)), generating inorganic ions of nitrogen (NH\(_4^+\)) and phosphorus (PO\(_4^{3-}\)) as end-products (Middelburg and Levin, 2009). In the presence of O\(_2\), NH\(_4^+\) tends to be oxidized completely to NO\(_3^-\) (or to NO\(_2^-\) and N\(_2\)O) by chemoautotrophic nitrifying bacteria. Although nitrification may be limited by NH\(_4^+\) availability in sediments with low organic content, rates in eutrophic coastal systems are more often controlled by depth of O\(_2\) penetration into NH\(_4^+\)-rich fine-grain organic sediments (e.g., Henriksen and Kemp, 1988). A substantial fraction of the NO\(_3^-\) generated in nitrification is generally reduced in surrounding anaerobic zones via denitrification to gaseous N\(_2\) (or N\(_2\)O) – forms that are virtually unavailable for assimilation by plants (e.g., Seitzinger, 1988). Lower redox conditions and high sulfide concentrations favor dissimilatory reduction of NO\(_3^-\) back to NH\(_4^+\) over denitrification (e.g., Tiedje, 1987) and strongly inhibit nitrification (e.g., Joye and Hollibaugh, 1995). Thus, hypoxia and anoxia greatly reduce nitrification and denitrification rates (e.g., Kemp et al., 1990). Although anammox (anaerobic oxidation of NH\(_4^+\) to N\(_2\) with NO\(_2^-\)) may occur in hypoxic environments, it is limited by availability of NO\(_2^-\), and rates tend to be substantially lower than denitrification in coastal sediments (Revsbech et al., 2006). Similar dynamics involving hypoxia and PO\(_4^{3-}\) recycling are attributable to completely different mechanisms. Under normoxic conditions, dissolved PO\(_4^{3-}\) binds to oxides and hydroxides of Fe and Mn, forming amorphous solid-phase substances that are retained in sediments (Froelich, 1988). In contrast, hypoxic conditions promote reduction of Fe and Mn to soluble states, thereby releasing bound PO\(_4^{3-}\) (Froelich et al., 6900
The presence of free sulfide, which has a very high affinity for binding sites on Fe and Mn, further promotes rapid release \( \text{PO}_3^{4-} \) and efflux to overlying waters (e.g., Caraco et al., 1989).

Many benthic invertebrate macrofauna (e.g., polychaetes, bivalves, amphipods) are highly susceptible to physiological stresses or mortality from bottom-water hypoxia and anoxia (e.g., Diaz and Rosenberg, 1995; Levin, 2003). Healthy benthic faunal communities can, however, exert strong influence on N and P cycling in coastal marine sediments (e.g., Aller, 1982). Although direct excretion by these organisms tends to increase nutrient recycling, activities of many species also retard recycling of \( \text{NH}_4^+ \) and \( \text{PO}_4^{3-} \) by enhanced \( \text{O}_2 \) advection into sediment porewaters. Macrofauna burrows, tunnels and tubes that penetrate (0.2–10 cm) into sediments are ventilated by natural circulation and by active animal pumping of overlying water (e.g., Aller, 1988). Macrofaunal ventilation tends to stimulate sediment nitrification and strengthen its coupling to denitrification by increasing the effective area of oxic-anoxic interfaces (e.g., Pelegri and Blackburn, 1995). Enhanced \( \text{O}_2 \) penetration into coastal sediments also retards dissolution of Fe-Mn-oxide-hydroxide complexes, promoting burial of \( \text{PO}_4^{3-} \) rather than release to overlying waters (e.g., Welsh, 2003; Middelburg and Levin, 2009). Feeding activities of other benthic fauna can dramatically alter sediment biogeochemistry by homogenizing or vertically transporting particles within the upper (0–30 cm) sediment column (e.g., Francois et al., 2001). Field observations and modeling studies suggest that vertical mixing of P-bound particles can reduce \( \text{PO}_4^{3-} \) release from sediments to overlying water in summer (e.g., DiToro, 2001). In summary, hypoxia and anoxia can further stimulate \( \text{NH}_4^+ \) and \( \text{PO}_4^{3-} \) recycling to overlying waters by reducing benthic macrofauna bioturbation.

Meadows of tidal marsh and seagrass plants effectively mitigate eutrophication and hypoxia along the coastal margins through dissolved nutrient uptake and particulate nutrient trapping (e.g., Kemp et al., 2005). Plant biomass accumulation in marshes and seagrass beds can store \( 10^3 \) more dry weight (dw) than phytoplankton, with plant stands sometimes exceeding 1000 g dw m\(^{-2} \) (e.g., Valiela, 1995). Integrated nutrient
pools contained in these macrophytes plant tissues and associated sediments can dominate coastal biotic nutrient budgets (e.g., Bricker and Stevenson, 1996; Kemp et al., 2005). These plants can respond to N and P enrichment by incorporating higher nutrient concentrations into their leaves (e.g., Duarte, 1990). In addition, denitrification rates in marsh and seagrass sediments are often much higher than those in nearby unvegetated sediments, because of enhanced nitrification associated with O₂ transported by roots into sediments and interception of nitrate-rich groundwater flux from watersheds (Bricker and Stevenson, 1996). The largest impact that these plants have on coastal N and P budgets is derived from their intense trapping of suspended nutrient-rich particles (e.g., Kemp et al., 2005; Boynton et al., 2008). As with benthic macrofauna, however, marsh and especially seagrass plants are also highly vulnerable to negative effects of coastal eutrophication, including reduced water clarity (e.g., Orth et al., 2006; Darby and Turner, 2008).

4 Theoretical trajectories for hypoxia response to remediation

Although low-oxygen zones are a natural part of many stratified coastal systems, anthropogenic increases in loading of inorganic nutrients and labile organic wastes have caused hypoxia to occur more widely, more frequently, and with greater severity in coastal regions throughout the world (Diaz and Rosenberg, 2008). Coastal eutrophication results in a spectrum of ecological changes in addition to hypoxia, including algal blooms, reduced water clarity, loss of seagrasses, and changes in food-webs (e.g., Kemp et al., 2005). Many responses to eutrophication are undesirable because of associated degradation of habitat for demersal fish and benthic invertebrates (Breitburg, 2002). Growing interest in these problems has motivated major socioeconomic commitments by regional, national and international authorities to reduce hypoxia and other environmental problems associated with coastal eutrophication. Consequently, expensive strategies are being devised to reduce watershed and atmospheric inputs of inorganic nutrients and organic wastes to coastal waters (Boesch et al., 2001; Rabalais
et al., 2007). Numerical models are often used to provide quantitative guidance to this mitigation process. Although these models generally predict simple linear reductions in hypoxia in response to reduced nutrient loading (Arhonditsis and Brett, 2004), recent data suggest that coastal ecosystem responses to nutrient reduction are often more complex (Duarte et al., 2009).

Scheffer et al. (2001) have illustrated a broader range of possible aquatic ecosystem response to changes in nutrient loading (Fig. 1). In the simplest case, responses of hypoxia and other eutrophication effects might be relatively smooth, continuous and linear, where effects increase and decrease along the same pathway in lock-step with changes in nutrient inputs (Fig. 1a). Alternatively, hypoxia might exhibit little response to an initial increase or decrease in nutrients until the system approaches a “threshold” where relatively small changes in nutrient input cause an abrupt system change (Fig. 1b). In this case, hypoxia again follows the same basic pathway in response to nutrient increase (eutrophication) and nutrient decrease (oligotrophication). If, however, nutrient increases change the fundamental ecosystem character – including trophic structure, habitat conditions, and biogeochemical cycles – the system may follow a distinctly different trajectory in response to nutrient input declines (Fig. 1c, d). These altered ecosystems become resistant to a change in state: relatively larger nutrient reductions and longer recovery times are required to induce a reversal of eutrophication effects (e.g., hypoxia) than the nutrient increases and degradation period that originally led to these effects. Because many coastal ecosystems are also experiencing perturbations from other factors (e.g., climate change, fishing harvest, species invasion) that can alter hypoxia responses, the “baseline conditions” may have changed during multi-decade time intervals between periods of nutrient increase and decrease. Such baseline shifts can lead to situations where complete recovery to pre-eutrophication conditions cannot be readily achieved simply with reduced nutrient loading (Fig. 1e, f) due to changes in other factors that affect hypoxia (e.g., Duarte et al., 2009).

To our knowledge, there are very few mechanistic models that have effectively predicted responses of eutrophication-induced coastal hypoxia to remediation, particularly
reductions in nutrient loading. Although relatively simple models have been used to hind-cast responses of shallow lakes to such remediation efforts (e.g., Scheffer and Jeppesen, 2007), few numerical forecasts have been documented for coastal systems (Soetaert and Middelburg, 2009). Detailed retrospective observations showing how hypoxia has changed with remediation is limited to a few coastal systems (e.g., Diaz and Rosenberg, 2008).

5 Observed hypoxia responses to remediation

Perhaps the most important assumption of the hypothetical response trajectories presented above is that nutrient loading is the primary driver of hypoxia. Much of the evidence suggesting nutrient loading reduction as an approach for hypoxia remediation is derived from retrospective observation of parallel increasing trends of nutrient loading and hypoxia (e.g., Turner et al., 2005; Hagy et al., 2004; Kauppila et al., 2005; Conley et al., 2009a). However, there is also strong theoretical and experimental evidence linking nutrient loading, algal productivity, organic particle sinking, and bottom water O\textsubscript{2} consumption, providing a mechanistic expectation for the responses that have been observed and evidence that nutrients are likely the primary driver. A possible exception is vertically mixed systems with significant inputs of sewage effluents rich in both dissolved inorganic nutrients and labile organics, where the organics may sustain respiration and hypoxia directly (Andrews and Rickard, 1980; Soetaert et al., 2006).

We compiled from the published literature a number of parallel time series of both hypoxia indices and nutrient (and organic matter) loading for several coastal systems to test theoretical expectations of system response to remediation. Our analysis includes systems with hypoxia of varying duration (seasonal, episodic, diel), with different anthropogenic inputs fueling hypoxia (nutrients, organic matter), and in different system types (well-mixed tidal rivers and shallow lagoons, as well as stratified estuaries, inland seas, and continental shelves). Comparisons of realized response trajectories to the hypothetical trajectories described above are made where relevant. In general,
this analysis suggests that $O_2$ conditions improved rapidly and linearly in systems with large reductions in discharges of labile organic matter from sewage treatment plants that had been sustaining $O_2$ consumption and hypoxia (Table 2). In larger stratified systems where diffuse input of inorganic nutrients was the primary driver of hypoxia through growth, sinking and decomposition of algal cells, the response to remediation tended to exhibit hysteresis and other non-linear behavior (Table 2).

Improved and more widely applied secondary sewage treatment in the 1960s, 1970s and 1980s led to major reductions in loads of dissolved and particulate labile organic material (or biochemical oxygen demand, BOD) to coastal waters. Where BOD from sewage treatment plants was the major source of oxygen demand, the situation usually involved large municipal plants discharging into the often shallow upper reaches of estuaries. One example is the inner Thames estuary, which received high loads of nutrients and organic matter from two major London sewage treatment plants through the 1960s and 1970s, causing summer dissolved $O_2$ to remain well below saturation levels (e.g., Andrews and Rickard, 1980) for a stretch of river (>20 km) seaward of London Bridge (Tinsley, 1998). Installation of secondary treatment at the major sewage treatment plants reduced BOD loads by 80% in the early 1970s and quickly returned summer $O_2$ levels to near-saturation (Fig. 3), followed by a recovery of fish, benthic macroinvertebrates, and benthic algal communities (Andrews and Rickard, 1980). The remediation trajectory of the BOD load versus % $O_2$ saturation indicates threshold behavior, where $O_2$ conditions improved slowly until ~70% of the load was removed, followed by rapid improvement during the final 30% of BOD removal (Fig. 3). Although no clear explanation exists for this response in the literature, the threshold may represent a synergy of biological and physical conditions. Up to the point where oxygen recovered, respiration, particularly in sediments, may have been saturated with respect to organic loading, such that reduced oxygen demand occurred only when availability of organic substrates became limiting. Interacting with this biological-physical threshold is the possibility that turbidity declined following wasteload reductions sufficiently to allow $O_2$ production by benthic algae (Andrews and Rickard, 1980). Improvements in other
sewage treatment plants in the area in the late 1970s may also have contributed to rapidly improving O₂ conditions (Tinsley, 1998). Most other shallow well-mixed estuaries receiving high sewage loads have exhibited positive linear responses (rather than threshold responses) to reduced BOD loads in recent decades. These include the Delaware River estuary (Patrick, 1988), the lower Hudson River and adjacent estuaries (Brosnan and O'Shea, 1996; O'Shea and Brosnan, 2000), and the Mersey estuary, (Jones, 2006).

The Scheldt estuary is another example of a shallow, turbid, and eutrophic upper estuarine system that responded strongly to changes in nutrient and organic matter loading (Soetaert et al., 2006). The tidal Scheldt, whose densely populated catchment includes parts of France, Belgium, and the Netherlands, is macrotidal with a mean depth of 10–12 m. Nutrient and organic loads to the Scheldt increased through the 1970s and were linearly related to an O₂ decline resulting from respiration of organic matter and oxidation of NH₄⁺ (Fig. 4). Because the upper Scheldt is turbid and is characterized by high nutrient levels, light limitation of phytoplankton generally prevailed and nutrient loads did not contribute to O₂ depletion via production of algal biomass. When improved sewage treatment reduced BOD loads in the mid-1970s, O₂ returned to pre-load levels (Fig. 4) over a 20-year period. In using DIN concentration as a proxy for N loading to the Scheldt, we observed that the slope of O₂ increase following DIN reduction was flatter than the slope of O₂ decline during increasing DIN (Fig. 4). This favorable shifting baseline scenario may be related to the fact that the fraction of DIN in a form that could be oxidized (NH₄⁺/DIN) declined steeply (Fig. 4) during this period (Soetaert et al., 2006). This success of remediation in the Scheldt despite complicating changes in nutrient ratios, nitrogen biogeochemistry, and climate (Soetaert et al., 2006) underscores the dominance of point sources in controlling O₂ in some shallow tidal estuaries.

Laajalahti Bay is another example of a simple linear response of low O₂ to loading. This shallow (mean depth=2.4 m), well-mixed, and semi-enclosed estuary is located west of Helsinki and is connected to the Gulf of Finland by a series of straits and
sounds (Kauppila et al., 2005). Although the well-mixed nature of the estuary generally prevented anoxia from occurring during the peak of eutrophication in this estuary, \( O_2 \) percent saturation was frequently below 30% during the period of highest nutrient and BOD loads in the mid 1960s (Fig. 5). In response to the realization that Laajalahti Bay was one of the most polluted coastal waters near Helsinki, improvements in sewage treatment in the late 1960s led to a steep decline in nutrient and BOD loading to the estuary. \( O_2 \) concentrations increased linearly following a decline in BOD loading, but subsequently stabilized in the early 1980s (Fig. 5). Further remediation occurred when Helsinki wastewaters were diverted to the outer archipelago in the mid-1980s, resulting in a second increase in \( O_2 \) to near saturation levels (Fig. 5). This second \( O_2 \) increase was significantly and linearly correlated with a decline in chlorophyll-\( a \), which was related to decreased TN loading; however, \( O_2 \) was likely also affected by removal of all wastewater BOD loads following the diversion (Kauppila et al., 2005). Thus, both phases of remediation in Laajalahti Bay caused a linear increase in \( O_2 \), one via reduced BOD input and a second linked to reduced inputs of inorganic nutrients (and BOD). As in all the examples presented above, shallow and well-mixed waters appear to respond positively and rapidly to nutrient and organic matter load reductions. Such systems are good targets for remediation because they do not have naturally occurring low \( O_2 \) conditions, and hypoxia can be readily controlled via improved sewage-treatment.

In coastal systems where only nutrient loads were reduced, few examples exist where data show that hypoxia decreased markedly with decreased nutrient loading. Where positive \( O_2 \) responses have been documented (e.g., Mallin et al., 2005), increases were relatively small despite significant declines in nutrient concentrations. To improve \( O_2 \) conditions, reductions in nutrient loading must first cause decreases in the phytoplankton biomass and production that fuels \( O_2 \) consumption. Although non-linear responses of phytoplankton biomass to nutrient loading reduction have been reported for many coastal systems (e.g., Duarte et al., 2008), there is a growing number of examples where reductions in algal biomass have been linearly correlated with decreasing nutrient loading (e.g., Henriksen, 2009; van Beusekom et al., 2009). In many large
stratified coastal systems, physical processes (e.g., wind stress, river flow, and tidal mixing) play key roles in $O_2$ depletion, where variations in ventilation of bottom waters may dominate the $O_2$ balance and control hypoxia formation. Under these conditions, climate-induced changes in circulation and mixing at decadal or longer scales might mask hypoxia responses to decreased nutrient loading, even if organic production and ecosystem respiration decline significantly. Recent studies have revealed complex dynamic relationships between hypoxia, nutrient loading, food webs, and climate for a number of well-studied coastal systems including Chesapeake Bay and its tributaries (e.g., Hagy et al., 2004; Testa et al., 2008), the northern Gulf of Mexico (Turner et al., 2009), the Black Sea (Oguz and Gilbert, 2007), and the Baltic Sea (Conley et al., 2009a). In the following section, we review and analyze these case studies toward improved understanding of hypoxia responses to remediation in large coastal ecosystems.

6 Complex relationships between nutrient loading and hypoxia

6.1 Patuxent River estuary

The Patuxent River estuary is a tributary system of Chesapeake Bay whose watershed straddles two major metropolitan areas. Two-layered circulation occurs for most of the year in the estuary, with a generally seaward-flowing surface layer and landward-flowing bottom layer. The mesohaline region of the Patuxent estuary has a deep channel (>10 m) that is flanked by broad shoals (<6 m) on each side. Bottom waters of the Patuxent River estuary develop hypoxia between May and September each year, with high interannual variation driven to a large extent by changes in freshwater inputs and associated nutrient loading and stratification (Testa et al., 2008). Although low-$O_2$ bottom water has been reported occasionally in the estuary since at least the 1940s (Newcombe et al., 1939), signs of eutrophication have been evident for the last 5 decades following increased urbanization in the upper watershed (D’Elia et al., 2003).
In an effort to decrease hypoxia and other eutrophication effects, upgraded sewage treatment reduced loads of TN (via Biological Nitrogen Removal, or BNR) and TP (via chemical precipitation) from point sources (which comprised 45% of the total N and P inputs to the estuary) by 40–60% in the mid 1980s for P and early 1990s for N (Fig. 7). Associated with nutrient loading reductions, there have been significant decreases in DIN and DIP concentrations throughout the estuary, as well as declines in phytoplankton chlorophyll-a (chl-a) and light-saturated carbon fixation for upper regions of the estuary (Testa et al., 2008). Bottom water O₂ concentrations in the upper estuary increased rapidly following the nutrient load reductions, where hypoxia now rarely occurs during summer (Fig. 7). Despite the improvements in the upper estuary and reduced transport of N and P to the lower estuary, the total estuarine hypoxic volume did not change (Fig. 7).

In fact, we found two separate positive correlations between NO₃⁻ load from the watershed and hypoxic volume when data were parsed into pre- and post-BNR years, with more hypoxic volume per unit NO₃⁻ load after BNR was initiated (after 1990) than before (Testa et al., 2008). Elevated hypoxia per unit N load (Fig. 6) corresponded to increases in summer chl-a, turbidity, and surface-layer net O₂ production in the lower estuary in years following BNR (Fig. 7). This degradation of water quality in the lower estuary and maintenance of high hypoxic volume appears to be linked to three unexpected changes in external and internal factors driving hypoxia. First, after 1990 annual input of DIN to the Patuxent River estuary (via estuarine circulation) from the nutrient-rich Chesapeake Bay bottom layer began to increase at a mean rate sufficient to offset the reduced loading from upstream (Fig. 7). Secondly, several consecutive years with above-average river flow and associated watershed nutrient loading in the mid-1990s contributed to stratification and elevated N-loading from non-point sources (Testa et al., 2008). Lastly, an abrupt increase in populations of the planktivorous ctenophore, Mnemiopsis leidyi (Purcell and Decker, 2005) significantly reduced summer zooplankton abundance (and potential phytoplankton grazing) for the calanoid copepod, Acartia tonsa, from 1990 to 2002 (Fig. 7). Thus, to relieve hypoxia in estuaries like the Patuxent, which are tidally
connected to nutrient-rich down-stream waters, nutrient inputs must be managed from both landward and seaward sources, while potential effects of trophic interactions must also be considered in the course of evaluating hypoxia responses to remediation.

6.2 Northwestern shelf of the Black Sea

The Black Sea, which is the largest enclosed sea in the world, receives freshwater, nutrient, and sediment loads from a large fraction of Europe and from some regions of Asia via the Danube and other rivers (Lancelot et al., 2002). Centuries of human impact have affected the Black Sea, but large ecosystem changes over the past 50 years caused more dramatic changes in nutrient and O₂ levels, as well as ecological changes at all trophic levels (Oguz and Gilbert, 2007). A deep and permanently anoxic basin exists in the central part of the Black Sea, but seasonal hypoxia (i.e., during warm months) has occurred on the northwestern shelf of the sea, which is directly influenced by the Danube River discharge. Areas of hypoxic bottom-water have been observed to varying extents since the early 1970s (Mee, 2006). The development, variability, and ultimate removal of the hypoxic zone in the northwestern shelf is an example of how the extent of hypoxia can respond to changes in nutrient loads, as mediated by climatic variability and food web shifts resulting from over-fishing.

Hypoxic waters, which were first observed on the northwest shelf in the summer of 1973, occurred annually until 1993, with substantial inter-annual variability (Fig. 8, Mee, 2006). The extent of hypoxia on the northwestern shelf has been linked to elevated phytoplankton biomass, which increased from the early 1970s to a peak in the 1980s with increased nitrogen fertilizer use and N loads from the Danube River (Mee, 2006; Oguz and Gilbert, 2007). Following the collapse of the former Soviet Union and Warsaw Pact governments in Eastern Europe in the late 1980s, agricultural subsidies were greatly reduced, causing abrupt and substantial declines in N fertilizer use and animal agriculture. The resulting large decrease in Danube River N loads was quickly followed by the disappearance of the associated hypoxic area by 1993; a small area of hypoxic water has returned in only one year since then (Fig. 8, Mee, 2006; Oguz and
Parallel time-series of hypoxia and N fertilizer application (a proxy for N loading) suggest a threshold response of hypoxia to increasing nutrient input, where N loads increased for more than a decade before hypoxia appeared (Fig. 8). An explanation for this threshold is not clear, but Danube River loading (and chl-a in the open Black Sea) did not begin to increase until 1970 (Oguz and Gilbert, 2007), suggesting that increased fertilizer use did not translate into increased loads until this time. As fertilizer use began to decline in the late 1980s, hypoxic areas were still observed for ~5 yr (1989–1993). This brief hysteretic time-lag may be linked to a period of increased phytoplankton production on the northwestern shelf associated with depressed zooplankton grazing in the open Black Sea, and with enhanced upward mixing of nutrient-rich deep water into the photic zone (Oguz and Gilbert, 2007). By the mid-1990s, sustained low nutrient loads, recovered zooplankton and benthic communities, and milder climatic conditions, all helped prevent hypoxic events (Mee, 2006; Oguz and Gilbert, 2007). Although the interaction of “top down” effects and other external conditions appears to have modulated the effect of N loading on hypoxia in the Black Sea, this system provides a relatively rare example of how a major reduction in nutrient loading can dramatically reduce or eliminate hypoxia.

6.3 Baltic Sea

The Baltic Sea is a large and permanently stratified estuary in northern Europe that receives freshwater inflow from an expansive watershed and salt water inflow with North Sea intrusions via the Kattegat, Skaggerak, and Danish straits. Hypoxia has been present in the Baltic Sea to varying degrees since the Holocene, when the previously freshwater lake was connected to the adjacent ocean, and brackish conditions were established (Zillén et al., 2008). Evidence suggests that the extent of hypoxic water in the Baltic has, however, increased since 1950 (Jonsson et al., 1990). Expanding hypoxic areas appear to have accompanied increased nutrient loads derived from the large and populous Baltic watershed, which includes parts of Sweden, Finland, Den-
Concerted efforts by these countries to reduce eutrophication and hypoxia in the Baltic Sea have been unsuccessful thus far. Below we explore the factors responsible for controlling the extent of hypoxia in this ecosystem.

The time-series data describing Baltic Sea hypoxia span 1960 to the present (Fig. 9, Conley et al., 2009a). A steady decline in the extent of hypoxia from 1970 through 1993 is easily apparent against a small background of interannual variability (Fig. 9). The latter decade of the decline (1985–1995) is associated with a decrease in total phosphorus (TP) load and concentration (Fig. 9), which upon first inspection would suggest that nutrient load remediation efforts were successful. However, a steady increase in hypoxic area and dissolved inorganic phosphorus (DIP) occurred from 1993 to 2000, despite no apparent increase in TP load (Fig. 9). Increased DIP levels appear to be attributable both to sustained external P inputs and to increased internal P loading, where iron-bound P was presumably released from sediments in larger quantities during the expansion of hypoxic bottom area from the mid-1990s to early 2000s (Conley et al., 2002).

Shifts in climate and physical circulation appear also to have been involved in the decrease in hypoxic area prior to 1993, as well as the subsequent hypoxia increase (Fig. 9). The Baltic Sea proper is permanently stratified and has fairly limited exchange with adjacent oceanic waters. Thus, mechanisms replenishing O\textsubscript{2} to bottom waters via both vertical cross-pycnocline mixing and water renewal via the Danish straits are highly restricted. These processes are controlled by physical circulation, and associated stratification strength may set an upper limit on Baltic hypoxia (Conley et al., 2009a). The 1993 hypoxia minimum followed a 10-year period of low salt water inflows through the Danish straits (Fig. 9), which served to reduce stratification and enhance downward vertical mixing of O\textsubscript{2} (Conley et al., 2009a), as inferred by a steady decline in deep water (200 m) salinity in the Gotland Deep through the 1980s to a minimum in 1992 (Fonselius and Valderrama, 2003). A return of inflowing salt water, documented in one year in the early 1990s, probably continued and caused increasing salinity in the

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Gotland Deep from 1992–2000 (Fig. 9). This would be expected to increase stratification strength (Fonselius and Valderrama, 2003), contributing to expanding hypoxic area (Fig 9). It appears that changes in water renewal rate and stratification combined with increasing temperature (Fonselius and Valderrama, 2003; Omstedt et al., 2004), persistently high nutrient loads (Fig. 9), and enhanced P release from sediments (Conley et al., 2002, 2009a) have caused stable and increasing hypoxic areas in this system. Despite the importance of abiotic controls on current hypoxia in the Baltic, modeling studies indicate that effective nutrient management in the Baltic watershed will help to reduce eutrophication and hypoxia (Wulff et al., 2007).

6.4 Chesapeake Bay

Chesapeake Bay is a large estuary in the United States and receives freshwater, nutrient, and organic matter inputs from several rivers, the largest of which is the Susquehanna. Dramatic ecological changes have occurred in Chesapeake Bay during the past century as a result of nutrient enrichment associated with human population growth and watershed land-use changes (Kemp et al., 2005). Effects of Bay eutrophication have been compounded by extensive loss of key habitats, including meadow- and canopy-forming beds of submersed aquatic plants and reefs of the filter-feeding eastern oyster, both of which form habitats for diverse animal populations and sinks for suspended particles and dissolved nutrients. One of the most important ecologically damaging effects of nutrient enrichment in Chesapeake Bay has been a long-term increase in the duration and volume of bottom-water hypoxia during mid-summer (Hagy et al., 2004).

Because Chesapeake Bay has a large ratio of watershed–to-estuary area, a deep channel isolated from the atmosphere during summer stratification, and a long water-residence-time, it is particularly susceptible to hypoxia and related effects of eutrophication (Kemp et al., 2005). Although incidents of hypoxia were reported as early as the 1930s (Newcombe and Horne, 1938), direct measurements and geochemical indicators suggest that intense and recurrent hypoxia and anoxia were uncommon before the mid 1950s following the large post-WWII increase in nitrogen loading (Hagy et al., 2004).
Extensive efforts to curb nutrient (both N and P) enrichment of Chesapeake Bay have led to a stabilization and slight decline in spring nitrogen loading from the Susquehanna River since ∼1990 (Fig. 10). Here we focus on N-loading because it is viewed as the primary limiting nutrient for phytoplankton production (e.g., Fisher et al., 1992, 1999). During this same time period, however, the volume of hypoxia has continued to rise, resulting in an abrupt doubling of the hypoxia volume per unit spring N load (Hagy et al., 2004). This “regime shift” with respect to N effects on hypoxia (Kemp et al., 2005) appears to have been initiated in 1986 (Conley et al., 2009b).

The cause for this unexpected shift in hypoxia volume per N-loading is uncertain. However, it generally coincided with notable changes in several key factors that could have contributed to the hypoxia increase, including a sharp increase in water temperature, a potential decrease in nutrient retention in the upper Bay, and a rapid decrease in oyster abundance (indexed to harvest) and associated filtration capacity (Fig. 10). Monitoring data indicate that surface water temperatures have been generally increasing in the middle region of Chesapeake Bay since 1945, but a relatively rapid rise (∼1.0°C) occurred in the period 1980–1990 (Kaushal et al., 2009). This temperature rise would have reduced O2 saturation level by ∼0.25 mg/l and increased water column respiration by 0.06 mg l−1 d−1 (Sampou and Kemp, 1994), probably insufficient changes to have caused the observed regime shift. Although reductions in phytoplankton biomass in the upper region of the Bay since the late 1970s may have allowed a larger fraction of Susquehanna River N loads to be transported seaward to fuel increased phytoplankton biomass in seaward regions (Harding and Perry, 1997; Hagy et al., 2004; Paerl et al., 2004), nutrient transport estimates to support this hypothesis are lacking. The relative abundance and filtering capacity of the eastern oyster (Crasostera virginica) in Chesapeake Bay have declined dramatically over the past 150 yr due to over-fishing and two disease outbreaks, with the largest reduction (∼40×10^6 to ∼15×10^6 kg yr⁻¹) in upper Bay harvest occurring between 1890 and 1910 (Newell, 1988). Despite these problems, a relatively stable, but much reduced, harvest (varying between 5 and 10×10^6 kg yr⁻¹) persisted from ca. 1930 to the early 1980s, when a fi-
nal, drought-induced decline essentially eliminated the oyster fishery (Fig. 10, Kimmel and Newell, 2007). Assuming that harvest provides a relative index of oyster abundance, the loss in phytoplankton filtering capacity associated with this final decline in harvest might have contributed to the 1980s jump in hypoxia per N loading (Newell and Ott, 1999; Newell et al., 2007), especially if the drought of 1985–1986 caused high mortality of isolated, non-harvestable oysters (Kimmel and Newell, 2007). The fact that historical records indicate no clear signal in phytoplankton abundance corresponding to the earlier oyster decline, however, raises questions about the potential for this smaller more recent oyster reduction to have caused the abrupt shift in hypoxia per N loading. The timing of other large changes in the Bay ecosystem, including reductions in nutrient-trapping capacity resulting from losses in abundance of marsh and seagrass communities (Kemp et al., 2005), while important, appear to be out of phase with this hypoxia regime shift.

While many of these changes in the ecosystem may have contributed to increased hypoxia in Chesapeake Bay, none appear to be sufficiently large and synchronous with hypoxia trends to have caused this regime shift. The sub-pycnocline recycling of nitrogen is a key biogeochemical process that has, however, exhibited a time-course of change that parallels the hypoxia regime shift trajectory. Previous studies have suggested that as O_2 decreases, a larger fraction of the organic nitrogen deposited to sediments is recycled to overlying water as NH_4^+ and mixed vertically into the euphotic zone where it can stimulate further algal growth (e.g., Kemp et al., 1990, 2005). This relationship arises because the coupled nitrification-denitrification process that transforms fixed nitrogen (NH_4^+) into N_2 gas (which is unavailable for algal assimilation) is restricted under hypoxic conditions (Kemp et al., 1990). This switch to increased N recycling to NH_4^+ is further enhanced by the loss of benthic macrofauna species from the anoxic region of the Bay bottom (Kemp et al., 2005), who have the ability to increase O_2 penetration into sediments though ventilation of their tube and burrow habitats (e.g., Mayer et al., 1995). Preliminary analyses of time-series data from mid-Chesapeake Bay suggest an abrupt increase in mean summer bottom water NH_4^+ concentration per
unit spring TN loading that closely follows the trend in hypoxia per N loading (Testa et al., 2009). While this analysis suggests that hypoxia enhancement of benthic N recycling has played an important role reinforcing further O$_2$ depletion, it is difficult to imagine how this mechanism could have driven the observed regime shift in hypoxia per unit N loading.

On the other hand, abrupt changes in atmospheric forcing or continental shelf circulation might be strong enough to alter vertical or horizontal replenishment of bottom water O$_2$ in the Bay. For example, recent modeling and retrospective data analysis suggest that sea level rise tends to cause increases in salt flux and bottom-layer salinity in Chesapeake Bay (Hilton et al., 2008), which could have increased stratification. Other evidence suggests an increase in the latitude of the north wall of the Gulf Stream since the 1980s (Taylor and Stephens, 1998) that may have reinforced the trend associated with sea level rise by causing an increase in salinity at the Bay mouth (Lee and Lwiza, 2008). The dramatic shift from negative to positive values for the winter index of the North Atlantic Oscillation (NAO) in the late 1970s (Fig. 10, http://www.cgd.ucar.edu/cas/jhurrell/indices.html) may be related to the change in the Gulf Stream position (Taylor and Stephens, 1998). Such a shift in NAO might also lead to changes in the prevailing wind direction and intensity (Ogi et al., 2003) which could affect the strength of stratification and associated ventilation of hypoxic bottom waters in summer (e.g., Malone et al., 1986, Scully et al., 2005). Although many of the ecological and biogeochemical factors discussed here may have contributed to the initiation and resilience of this hypoxia regime shift, we conclude that physical factors are likely involved in the initiation of this change in hypoxia per N loading.

### 6.5 Northern Gulf of Mexico

A large region of the Gulf of Mexico’s northwest continental shelf is highly influenced by outflow of the Mississippi and Atchafalaya Rivers, with associated buoyancy causing stratification and river-borne nutrient loads supporting rich phytoplankton productivity (e.g., Bianchi et al., 2008). Under present conditions, bottom-water hypoxia occurs...
during summer months in this northern Gulf of Mexico (NGOM) region. The size of this low-O$_2$ zone has been mapped annually since the mid-1980s, when hypoxia was first recorded by direct observation (e.g. Rabalais et al., 2007). Fossil records of low-oxygen-tolerant foraminifera and modeling studies, however, suggest that infrequent hypoxia probably occurred back into the 1950s (Osterman et al., 2005; Greene et al., 2009) but that large-scale hypoxic zones likely did not occur prior to the 1970s (Justic et al., 2005; Greene et al., 2009). A significant trend of increasing areal extent of NGOM hypoxia corresponds strongly with a parallel trend of increasing nitrogen (N) loads from the Mississippi River (Turner et al., 2005), driven largely by the growth of intensive agricultural production in the Midwest region of the USA (Alexander et al., 2008). Since the mid-1980s, the July hypoxic zone has ranged in size from 40 to 22,000 km$^2$ and has been correlated with May TN and NO$_3^-$ loading from the Mississippi River (Scavia et al., 2004; Turner et al., 2008). A small (17%) decrease in TN loading between 1997 and 2005 (Turner et al., 2007), produced no significant decrease in the extent of hypoxia in July (Rabalais et al., 2007). Recent analyses have shown that the extent of hypoxia increased substantially in 1993 and thereafter, while N loading has changed very little, resulting in an increase in the area of hypoxia observed per unit of N loading (Turner et al., 2008; Greene et al., 2009).

Although there is no clear explanation for this change in hypoxia per unit N loading, a range of physical and biological factors may have contributed to this hypoxia “regime shift” (Rabalais et al., 2007). Turner et al. (2008) suggested that increased sediment oxygen demand (SOD) could have caused the observed rapid increase in hypoxia. They hypothesized that intensified SOD in recent years was derived from organic matter produced in previous years but preserved and stored. They further suggested that this stored organic matter would be exposed (e.g., through resuspension events) in subsequent years to fuel enhanced respiration, thereby causing larger than expected hypoxic regions per unit nitrogen loading in a given year. Unfortunately, empirical support for this hypothesis (e.g., long-term records of SOD) is limited, and previous observations from warm marine systems suggest that the vast majority of labile organic
A related hypothesis is that there has been a recent increase in organic matter inputs from sources other than the Mississippi-Atchafalaya River System (Dagg et al., 2007; Bianchi et al., 2008). The primary postulated source for this increased organic matter input is cross-shelf transport of material from Louisiana’s eroding coastal wetlands (e.g., Barras et al., 2008). Although the general magnitude and biogeochemical nature of eroding wetland organic matter fluxes have been estimated, these organic fluxes have not been directly linked to increased Gulf hypoxia (Dagg et al., 2007). In addition, recent analyses indicate that, apart from increases associated with hurricanes in 2005, the rate of coastal land loss has been relatively constant for the last several decades (Barras et al., 2008). Others have emphasized the role of phosphorus (P) loading in regulating NGOM hypoxia (e.g., Sylvan et al., 2006; Dagg et al., 2007). Although statistical analysis reveals that N loading was by far the best predictor of hypoxia, inclusion of winter P concentrations in a multiple regression model did improve summer hypoxia predictions (e.g., Greene et al., 2009). However, the absence of any long-term trend in P loading (Sylvan et al., 2006) indicates that it is probably not responsible for the observed hypoxia regime shift.

The general open nature of the NGOM system, with unrestricted boundaries to the south along the Gulf and to the west across the broad continental shelf, makes hypoxia development and control highly sensitive to physical circulation processes. Key physical controls on NGOM hypoxia include freshwater inflow, local meteorological forcing, and interactions with the open Gulf circulation. Of these, only the impact of freshwater flow has been considered in analyses of changes and long-term trends in the extent of hypoxia (e.g., Justic et al., 2003; Turner et al., 2005; Greene et al., 2009). One way that a change in other physical factors could impact the extent of hypoxia (and its ratio to N loading), is by modulating the flux of freshwater and nutrients between the shelf and open Gulf (e.g., Bianchi et al., 2006). Current estimates indicate that only 43% of the Mississippi River flow and associated nutrients (not including the Atchafalaya River flow which is 30% of the total) enters the western shelf, whereas the remainder...
is directed eastward or otherwise lost to the open Gulf (Etter et al., 2004). Any change either in this fraction or in the rate of removal of water, nutrients, and sediments at the shelf margin (e.g., Bianchi et al., 2006) would likely impact the extent of hypoxia independently from the measured nitrogen loading rate. Summer winds are also important in regulating NGOM hypoxia, where up-coast (southwesterly) winds favor retention of freshwater (+nutrients and plankton) on the Louisiana-Texas shelf and enhance stratification and hypoxia, until September when down-coast winds (northeasterly) promote vertical mixing and Gulf-shelf exchange (Cochrane and Kelly, 1986). Although the extent of hypoxia can be reduced quickly when wind events (e.g., associated with frontal passage) reduce stratification (Rabalais and Turner, 2006), such events are relatively rare in summer. Clearly, changes in these physical forces are important in controlling NGOM hypoxia. Thus far, however, no evidence has been presented to suggest decadal scale changes in these processes that would be needed to explain the observed shift in hypoxia per unit of N loading in this system.

In summary, a long-term increase in the areal extent of hypoxia in the northern Gulf of Mexico since the early 1980s is linked principally to increases in N loading from the Mississippi-Atchafalaya River System following the intensification of agriculture in the watershed. Since 1993, observed hypoxic areas have been larger than expected from spring N loading alone, suggesting that one or more changes may have occurred in other potential ecological or physical drivers of hypoxia. Hypothesized causes include accumulation and preservation of labile sediment organic matter, relief of P-limitation on phytoplankton production, alternative organic carbon sources, and a sustained change in physical processes. Another possibility is that the massive discharge from the Mississippi Basin in 1993 caused an ecological state change in the shelf ecosystem. To date, however, supporting empirical evidence and modeling analyses are insufficient to provide rigorous tests of these hypotheses.
7 Concluding comments and implications for management

Our review of case studies around the world reveals the complexity of the hypoxia problem, from both the standpoint of understanding the controls on O\textsubscript{2} dynamics and also deciding what management actions are required to relieve depressed O\textsubscript{2}. Persistent and growing hypoxia in many other regions of the world – e.g., East China Sea (Chen et al., 2007) and Long Island Sound (O'Donnell et al., 2008 – remind the scientific community that efforts to understand and control hypoxia must continue during the next several decades, as human impacts on coastal systems continue to increase.

The varied trajectories of hypoxia responses to organic and nutrient loading in different systems, both hypothesized and realized, underscore the fact that managing and remediating hypoxia in coastal ecosystems must be system-specific. Physical features of the ecosystem (i.e., bathymetry, circulation, and stratification) appear to set limits on hypoxia extent by controlling O\textsubscript{2} replenishment via advective and diffusive O\textsubscript{2} inputs. These physical factors, thus, contribute to threshold responses of hypoxia to nutrient loading. Thresholds can also be ecological in nature when disruption of key feedback processes (e.g., grazing on algae, trapping of organic matter, and cycling of nutrients) alters the nature of the system's responses to loading. Where such thresholds exist, time-lags in ecosystem responses to reduced nutrient loading declines must be expected. Further time-lags may exist, for example, where accumulated nutrients take years to flush out of the system, or where benthic communities and associated food-webs require time to fully recover. In the case that benthic communities or other components of the food web are irreversibly altered, expectation of a return to historical conditions with remediation may be inappropriate. This is because climatic changes (i.e., increased temperature, altered freshwater input, changes in atmospheric circulation) during the degradation and remediation time periods may also induce a “shifting baseline”, where even if nutrient loading is reduced, climatic forcing and ecosystem dynamics will differ from previous periods, possibly in ways that alter the ecosystem's susceptibility to hypoxia.
As with strategies for management and remediation, expected responses to remediation should be adjusted to the specific system. In Long Island Sound, for example, despite reductions in municipal wastewater discharges, summer values for bottom O\textsubscript{2} have continued to decline because of long-term changes in wind velocity and associated increases in stratification (Wilson et al., 2008). In contrast, for shallow well-mixed coastal systems where point sources were the dominant contributor to hypoxia (e.g., the Scheldt, Forth and East River estuaries, Table 2), abrupt and large load reductions have induced successful remediation that followed relatively linear recovery trajectories. For other shallow coastal systems (e.g., upper estuaries of the Potomac and Thames Rivers) responses to point-sources nutrient remediation were substantial but delayed for a decade as these latter cases illustrate how adequate recovery time may need to be allowed prior to evaluating success of management actions.

Deep stratified systems have generally exhibited more complicated responses to remediation compared to shallower coastal systems. One of the few documented rapid and direct recoveries from hypoxia in a deep stratified system occurred in the northwestern shelf of the Black Sea, where severe political and economic shifts led to an unusually large and abrupt reduction in external nutrient loading. Similar systems, for which nutrient loads are primarily derived from diffuse sources (e.g., Chesapeake Bay, northern Gulf of Mexico, and Baltic Sea), have shown little recovery and in most cases, indicate continued expansion of hypoxia. Hysteresis in recovery trajectories and long delays between management and ecological responses appear in these systems due to alterations in biological and physical processes. It appears that, in many cases, the loss of key ecosystem feedback effects (e.g., biogeochemical cycles or food-web interactions) and/or major alterations in climatic or oceanographic conditions may have muted or counteracted expected hypoxia decreases in response to reduced nutrient loading. In such situations, hypoxia remediation may be enhanced by restoration of habitats for key filter-feeding bivalve reefs, tidal marshes and seagrass beds – all of which tend to filter and retain nutrients and organic material. Coupled management of point-sources and landscape modification may be necessary to remediate the nutrient
loading that controls hypoxia. Full recovery in these systems may require additional
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time for restoration of nutrient-impaired biological functions to occur.

The temporal scales of hypoxia remediation may also be linked to inherent phys- 
ical properties and processes that constrain hypoxia development in coastal systems. 
For instance, in shallow well-mixed estuaries that experience seasonal or diel hypoxia, 
limited stratification and adequate vertical mixing naturally favor oxygenation, thus ac-
celerating remediation time and favoring rapid linear recovery. Improvements in water 
clarity and restoration of benthic photosynthesis in these shallow systems may provide 
positive feedbacks that further accelerate recovery. In this case, decreased nutrients 
reduce phytoplankton biomass, improve water clarity, and enhance benthic algal and 
seagrass photosynthesis, which serves to oxygenate the water column and sequester 
nutrients within sediments. Deep, stratified, and enclosed estuaries are naturally more 
susceptible to hypoxic conditions and resistant to remediation. Stratified continen-
tal shelf regions have similar susceptibility to hypoxia; however, their comparatively 
open nature may make them more responsive to remediation because of character-
istic stronger water exchange with deeper offshore regions. In systems with episodic 
hypoxia, the climatic conditions that favor brief hypoxic events may be unresponsive to 
remediation, but reduced nutrient loads could decrease the frequency and intensity of 
hypoxic events by limiting phytoplankton biomass.

Lessons learned from this comparative analysis of hypoxia responses to remediation 
will be useful for coastal researchers and managers alike. Improved understanding of 
the dominant physical and biological processes is needed to characterize the natural 
susceptibility of a particular coastal system to hypoxia. Further information is needed 
concerning interactions among climatic trends and cycles, hydrodynamic circulation 
and mixing, biogeochemical cycles, and food-web relationships. Analyses should in-
volve application of sophisticated statistical methods and diagnostic studies using bio-
physical numerical models. In addition to understanding oceanographic and ecological 
processes, managers will need to educate stakeholders to achieve realistic results re-
garding the speed, extent, and nature of hypoxia responses to remediation. What
appears to be unsuccessful remediation may simply result from time-lags in ecological response to remediation actions. Successful remediation may require combinations of nutrient loading reductions, restoration of key ecological habitats, and hydrologic engineering solutions (e.g., river flow controls). Assuming that the scientific community is able to adequately understand factors regulating hypoxia, sustained remediation may require open communication among researchers, managers and stakeholders.

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References

Alexander, R. B., Smith, R. A., Schwarz, G. E., Boyer, E. W., Nolan, J. V., and Brakebill, J. W.: Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River basin, Environ. Sci. Technol., 42, 822–830, 2008.

Aller, R. C.: The effects of macrobenthos on chemical properties of marine sediments and overlying water, In: Animal-Sediment Relations, edited by: McCall, O. P. L. and Tevesz, M. J. S., Plenum Press, New York, 3–52, 1982.

Andrews, M. J. and Rickard, D. G.: Rehabilitation of the inner Thames Estuary, Mar. Pollut. Bull., 11, 327–332, 1980.

Arhonditsis, G. B. and Brett, M. T.: Evaluation of the current state of mechanistic aquatic biogeochemical modeling, Mar. Ecol.-Prog. Ser., 271, 13–26, 2004.

Arnell, N. W.: The effect of climate change on hydrological regimes in Europe: A continental perspective, Global Environ. Chang., 9, 5–23, 1999.
Balls, P. W., Brockie, N., Dobson, J., and Johnston, W.: Dissolved oxygen and nitrification in the upper Forth Estuary during summer (1982–1992): Patterns and trends, Estuar. Coast. Shelf S., 42, 117–134, 1996.

Banas, N. S., Hickey, B., Newton, J., and Ruesink, J.: Tidal exchange, bivalve grazing, and patterns of primary production in Willapa Bay, Washington, USA, Mar. Ecol.-Prog. Ser., 34, 123–139, 2007.

Barras, J. A., Bernier, J. C., and Morton, R. A.: Land area change in coastal Louisiana-A multidecadal perspective (from 1956 to 2006): US Geological Survey Scientific Investigations Map 3019, scale 1:250 000, 14 p. pamphlet, 2008.

Bianchi, T. S., Allison, M. A., Canuel, E. A., Corbett, D. R., McKee, B. A., Sampere, T. P., Wakeham, S. G., and Waterson, E.: Rapid export of organic matter to the Mississippi Canyon, EOS T. Am. Geophys. Un., 87, 565, 572–573, 2006.

Bianchi, T. S.: Biogeochemistry of Estuaries, Oxford University Press, Oxford, 2007.

Bianchi, T. S., DiMarco, S., Allison, M., Chapman, P., Cowan, J., Hetland, R., and Morse, J. W.: Controlling hypoxia on the U.S. Louisiana shelf: beyond the nutrient-centric view, EOS T. Am. Geophys. Un., 89, 236–237, 2008.

Boesch D. F., Brinsfield, R. B., and Magnien, R. E.: Chesapeake Bay eutrophication: Scientific understanding, ecosystem restoration, and challenges for agriculture, J. Environ. Qual., 30, 303–320, 2001.

Borsuk, M. E., Stow, C. A., Luetich, Jr., R. A. Pael, H. W., and Pinckney, J. L.: Modelling oxygen dynamics in an intermittently stratified estuary: Estimation of process rates using field data, Estuar. Coast. Shelf S., 52, 33–49, 2001.

Boicourt, W. C.: Influences of circulation processes on dissolved oxygen in the Chesapeake Bay, In: Oxygen Dynamics in the Chesapeake Bay, A Synthesis of Recent Research, edited by: Smith, D. E., Leffler, M., and Mackiernan, G., Maryland Sea Grant, College Park, M. D., 7–59, 1992.

Francois, F., Poggiale, J., Durbec, J., and Stora, G.: A new model of bioturbation for a functional approach to sediment reworking resulting from macrobenthic communities, In: Organism-sediment Interactions, edited by: Aller, J. Y., Woodin, S. A., and Aller, R. C., University of South Carolina Press, Columbia, S.C., 73–86, 2001.

Boynton, W. R., Hagy, J. D., Cornwell, J., Kemp, W. M., Greene, S., Owens, M., Baker, J., and Larsen, R.: Nutrient budgets and management actions in the Patuxent River estuary, Maryland, Estuaries Coasts, 31, 623–651, 2008.
Breitburg, D.: Effects of hypoxia, and the balance between hypoxia and enrichment, on coastal fishes and fisheries, Estuaries, 25, 67–781, 2002.
Bricker, S. B. and Stevenson, J. C.: Nutrients in coastal waters: A chronology and synopsis of research, Estuaries, 19, 337–341, 1996.
Brosnan, T. M. and O’Shea, M. L.: Long-term improvements in water quality due to sewage abatement in the lower Hudson River, Estuaries, 19, 890–900, 1996.
Burdige, D. J.: The kinetics of organic matter mineralization in anoxic marine sediments, J. Mar. Res., 49, 727–761, 1991.
Caraco, N. F., Cole, J. J., and Likens, G. E.: Evidence for sulfate-controlled phosphorus release from sediments of aquatic systems, Nature, 341, 316–318, 1989.
Carstensen, J., Conley, D. J., Andersen, J., and Ærtebjerg, G.: Coastal eutrophication and trend reversal: A Danish case study, Limnol. Oceanogr., 51, 398–408, 2006.
Cerco, C. F. and Noel, M. R.: Can oyster restoration reverse cultural eutrophication in Chesapeake Bay? Estuaries Coasts, 30, 331–343, 2007.
Chen, C.-C., Gong, G.-C., and Shiah, F.-K.: Hypoxia in the East China Sea: One of the largest coastal low-oxygen areas in the world, Mar. Environ. Res., 64, 399–408, 2007.
Chen, C.-C., Shiah, F.-K., Chaing, K.-P., Gong, G.-C., and Kemp, W. M.: Effects of the Changjiang (Yangtze) River discharge on planktonic community respiration in the East China Sea, J. Geophys. Res., 114, 1–15, doi:10.1029/2008JC004891.2009, 2009.
Cochrane, J. D. and Kelly, F. J.: Low-frequency circulation on the Texas-Louisiana continental shelf, J. Geophys. Res., 91, 10645–10659, 1986.
Conley, D. J., Humborg, C., Rahm, L., Savchuk, O. P., and Wulff, F.: Hypoxia in the Baltic Sea and basin-scale changes in phosphorus biogeochemistry, Environ. Sci. Technol., 36, 5315–5320, 2002.
Conley, D. J., Carstensen, J., Ærtebjerg, G., Christensen, P. B., Dalsgaard, T., Hansen, J., and Josefson, A.: Long-term changes and impacts of hypoxia in Danish coastal waters, Ecol. Appl. 17, S165–S184, 2007.
Conley, D. J., Björck, S., Bonsdorff, E., Carstensen, J., Destouni, G., Gustafsson, B. G., Hietanen, S., Kortekaas, M., Kuosa, H., Meier, H. E. M., Müller-Karulis, B., Nordberg, K., Norkko, A., Nürnberg, G., Pitkänen, H., Rabalais, N. N., Rosenberg, R., Savchuk, O. P., Slomp, C. P., Voss, M., Wulff, F., and Zillén, L.: Hypoxia-related processes in the Baltic Sea, Environ. Sci. Technol., 43, 3412–3420, 2009a.
Conley, D. J., Carstensen, J., Vaquer-Sunyer, R., and Duarte, C. M.: Ecosystem thresholds
with hypoxia, Hydrobiologia, 629, 21–29, 2009b.
Conley, D. J., Bonsdorff, E., Carstensen, J., Destouni, G., Gustafsson, B. G., Hansson, L.-A., Rabalais, N. N., Voss, M., and Zillén, L.: Tackling hypoxia in the Baltic Sea: Is engineering a solution?, Environ. Sci. Tech., 43, 3407–3411, 2009c.
Dame, R. F. and Prins, T. C.: Bivalve carrying capacity in coastal ecosystems, Aquat. Ecol., 31, 409–421, 1998.
Dame, R. F. and Olenin, S.: The comparative roles of suspension feeders in ecosystems, NATO ASI Sci. Ser. 4 Earth Environmental Science, Springer-Verlag, Berlin, 2005.
Dagg, M. J., Ammerman, J. W., Amon, R. M. W., Gardner, W. S., Green R. E., and Lohrenz, S. E.: A review of water column processes influencing hypoxia in the northern Gulf of Mexico, Estuaries Coasts, 30, 735–752, 2007.
D’Avanzo, C. and Kremer, J. N.: Diel oxygen dynamics and anoxic events in an eutrophic estuary of Waquoit Bay, Massachusetts, Estuaries, 17, 131–139, 1994.
Darby, F. A. and Turner, R. E.: Effects of eutrophication on salt marsh root and rhizome biomass accumulation, Mar. Ecol.-Prog. Ser., 363, 63–70, 2008.
Decker, M. B., Breitburg, D. L., and Purcell, J. E.: Effects of low dissolved oxygen on zooplankton predation by the ctenophore Mnemiopsis leidyi, Mar. Ecol.-Prog. Ser., 280, 163–172, 2004.
D’Elia, C. F., Boynton, W. R., and Sanders, J. G.: A watershed perspective on nutrient enrichment, science, and policy in the Patuxent River, Maryland: 1960–2000, Estuaries, 26, 171–185, 2003.
Diaz, R. J. and Rosenberg, R.: Marine benthic hypoxia: A review of its ecological effects and the behavioral responses of benthic macrofauna, Oceanogr. Mar. Biol., 33, 245–303, 1995.
Diaz, R. J.: Overview of hypoxia around the world, J. Environ. Qual., 30, 275–281, 2001.
Diaz, R. J., Rhoads, D. C., Blake, J. A., Kropp, R. K., and Keay, K. E.: Long-term trends of benthic habitats related to reduction in wastewater discharge to Boston Harbor, Estuaries Coasts, 31, 1184–1197, 2008.
Diaz, R. J. and Rosenberg, R.: Spreading dead zones and consequences for marine ecosystems, Science, 321, 926–929, 2008.
Di Toro, D. M.: Sediment flux modeling, John Wiley, New York, 2001.
Duarte, C. M.: Seagrass nutrient content, Mar. Ecol.-Prog. Ser. 67, 201–207, 1990.
Duarte, C. M., Conley, D. J., Carstensen, J., and Sanchez-Camacho, J.: Return to “Neverland”: Shifting baselines affect eutrophication restoration targets, Estuaries Coasts, 32, 29–
Etter, P. C., Howard, M. K., and Cochrane, J. D.: Heat and freshwater budgets of the Texas-Louisiana shelf, J. Geophys. Res., 109, C02024, doi:10.1029/2003JC001820, 2004.

Fallesen, G., Andersen, F., and Larsen, B.: Life, death and revival of the hypertrophic Mariager Fjord, Denmark, J. Marine Syst., 25, 313–321, 2000.

Fisher, T. R., Peele, E. R., Ammerman, J. W., and Harding Jr., L. W.: Nutrient limitation of phytoplankton in Chesapeake Bay, Mar. Ecol.-Prog. Ser., 82, 51–63, 1992.

Fisher, T. R., Gustafson, A. B., Sellner, K., Lacouture, R., Haas, L. W., Wetzel, R. L., Magnien, R., Everitt, D., Michaels, B., and Karrh, R.: Spatial and temporal variation of resource limitation in Chesapeake Bay, Mar. Biol., 133, 763–778, 1999.

Fonselius, S. and Valderrama, J.: One hundred years of hydrographic measurements in the Baltic Sea, J. Sea Res., 49, 229–241, 2003.

Francois, F., Poggiale, J., Durbec, J., and Stora, G.: A new model of bioturbation for a functional approach to sediment reworking resulting from macrobenthic communities, In: Organism-Sediment Interactions, edited by: Aller, J.Y., Woodin, S.A., and Aller, R.C., University of South Carolina Press, Columbia, S.C., 73–86, 2001.

Froelich, P. N., Bender, M. L., and Luedtke, N. A.: The marine phosphorus cycle, Am. J. Sci., 282, 474–511, 1982.

Froelich, P. N.: Kinetic control of dissolved phosphate in natural rivers and estuaries: A primer on the phosphate buffer mechanisms, Limnol. Oceanogr., 33, 649–668, 1988.

Gilbert, D., Sundby, B., Gobeil, C., Mucci, A. and Tremblay, G.-H.: A seventy-two-year record of diminishing deep-water oxygen in the St. Lawrence estuary: The northwest Atlantic connection. Limnol. Oceanogr., 50, 1654–1666, 2005.

Greene, R. M., Legrter, J. C., and Hagy, J. D.: Multiple regression models for hindcasting and forecasting midsummer hypoxia in the Gulf of Mexico, Ecol. Appl., 19, 1161–1175, 2009.

Hagy, J. D., Boynton, W. R., Keefe, C. W., and Wood, K. V.: Hypoxia in Chesapeake Bay, 1950–2001: Long-term change in relation to nutrient loading and river flow, Estuaries, 27, 634–658, 2004.

Hagy, J. D., Boynton, W. R., and Jasinski, D.: Modeling phytoplankton deposition to Chesapeake Bay sediments during winter–spring: Interannual variability in relation to river flow, Estuar. Coast. Shelf S., 62, 25–40, 2005.

Hagy, J. D. and Murrell, M. C.: Susceptibility of a northern Gulf of Mexico estuary to hypoxia: An analysis using box models, Estuar. Coast. Shelf S., 74, 239–253, 2007.
Harding Jr., L. W. and Perry, E. S.: Long-term increase of phytoplankton biomass in Chesapeake Bay, 1950–1994, Mar. Ecol.-Prog. Ser., 157, 39–52, 1997.

Hedges, J. I. and Keil, R. G.: Sedimentary organic matter preservation: An assessment and speculative synthesis, Mar. Chem., 49, 81–115, 1995.

Helly, J. J. and Levin, L. A.: Global distribution of naturally occurring marine hypoxia on continental margins, Deep-Sea Res. Pt. I., 51, 1159–1168, 2004.

Henriksen, K. and Kemp, W. M.: Nitrification in estuarine and coastal marine sediments: Methods, patterns and regulating factors, In: Nitrogen Cycling in Coastal Marine Environments, edited by: Blackburn, T. H. and Sørensen, J., John Wiley, New York, 207–209, 1988.

Henriksen, P.: Long-term changes in phytoplankton in the Kattegat, the Belt Sea, the Sound, and the western Baltic Sea, J. Sea Res., 61, 114–123, 2009.

Holgate, S. J. and Woodworth, P. L.: Evidence for enhanced coastal sea level rise during the 1990s, Geophys. Res. Lett., 31, L07305, doi:10.1029/2004GL019626, 2004.

Jeppesen, E., Sondergaard, M., Pedersen, A., Jurgens, K., Strzelczak, A., Lauridsen, T., and Johansson, L.: Salinity induced regime shift in shallow brackish lagoons, Ecosystems, 10, 47–57, 2007.

Jones, P. D.: Water quality and fisheries in the Mersey estuary, England: A historical perspective, Mar. Pollut. Bull., 53, 144–154, 2006.

Jonsson, P., Carman, R., and Wulff, F.: Laminated sediments in the Baltic: A tool for evaluating nutrient mass balances, Ambio, 19, 152–158, 1990.

Joye, S. B. and Hollibaugh, J. T.: Influence of sulfide inhibition of nitrification on nitrogen regeneration in sediments, Science, 270, 623–625, 1995.

Justic, D., Turner, R. E., and Rabalais, N. N.: Climatic influences on riverine nitrate flux: Implications for coastal marine eutrophication and hypoxia, Estuaries, 26, 1–11, 2003.

Justic, D., Rabalais, N. N., and Turner, R. E.: Coupling between climate variability and coastal eutrophication: Evidence and outlook for the northern Gulf of Mexico, J. Sea. Res., 54, 25–35, 2005.

Kauppila, P., Weckström, K., Vaalgamaa, S., Korhola, A., Pitkänen, H., Reuss, N., and Drew, S.: Tracing pollution and recovery using sediments in an urban estuary, northern Baltic Sea: Are we far from ecological reference conditions? Mar. Ecol.-Prog. Ser., 290, 35–53, 2005.

Kaushal, S. S, Likens, G. E., Jaworski, N. A., Pace, M. L., Sides, A. M., Seekell, D., Belt, K. T., Secor, D. H., and Wingate, R.: Rising stream and river temperatures in the United States, Front. Ecol. Environ., in press, 2009.
Kemp, W. M., Sampou, P. A., Caffrey, J. M., Mayer, M., Henriksen, K., and Boynton, W. R.: Ammonium recycling versus denitrification in Chesapeake Bay sediments, Limnol. Oceanogr., 35, 1545–1563, 1990.

Kemp, W. M., Sampou, P. A., Garber, J., Tuttle, J., and Boynton, W. R.: Seasonal depletion of oxygen from bottom waters of Chesapeake Bay: Roles of benthic and planktonic respiration and physical exchange processes, Mar. Ecol.-Prog. Ser., 85, 137–152, 1992.

Kemp, W. M., Boynton, W. R., Adolf, J., Boesch, D., Boicourt, W., Brush, G., Cornwell, J., Fisher, T., Glibert, P., Hagy, J., Harding, L., Houde, E., Kimmel, D., Miller, W. D., Newell, R. I. E., Roman, M., Smith, E., and Stevenson, J. C.: Eutrophication of Chesapeake Bay: Historical trends and ecological interactions, Mar. Ecol.-Prog. Ser., 303, 1–29, 2005.

Kimmel, D. K. and Newell, R. I. E.: The influence of climate variation on eastern oyster (Crasostrea virginica) juvenile abundance in Chesapeake Bay, Limnol. Oceanogr., 52, 959–965, 2007.

Kuo, A. Y., Park, K., and Mustafa, M. Z.: Spatial and temporal variabilities of hypoxia in Rappahannock River, Virginia, Estuaries, 14, 113–121, 1991.

Lancelot, C., Staneva, J., Van Eeckhout, D., Beckers, J.-M., and Stanev, E.: Modelling the Danube-influenced north-western continental shelf of the Black Sea. II: Ecosystem response to changes in nutrient delivery by the Danube River after its damming in 1972, Estuar. Coast. Shelf S., 54, 473–499, 2002.

Lee, Y. J. and Lwiza, K. M. M.: Factors driving bottom salinity variability in the Chesapeake Bay, Cont. Shelf. Res., 28, 1352–1362, 2008.

Levin, L. A.: Oxygen minimum zone benthos: Adaptation and community response to hypoxia, Oceanogr. Mar. Biol., 41, 1–45, 2003.

Lin, J., Xu, H., Cudaback, C., and Wang, D.: Inter-annual variability of hypoxic conditions in a shallow estuary, J. Marine Syst., 73, 169–184, 2008.

MacPherson, T. A., Cahoon, L., and Mallin, M.: Water column oxygen demand and sediment oxygen flux: patterns of oxygen depletion in tidal creeks, Hydrobiologia, 586, 235–248, 2007.

Mallin, M. A., McIver, M., Wells, H., Parsons, D., and Johnson, V.: Reversal of eutrophication following sewage treatment upgrades in the New River estuary, North Carolina, Estuaries, 28, 750–760, 2005.

Malone, T. C., Kemp, W. M., Ducklow, H. W., Boynton, W. R., Tuttle, J. H., and Jonas, R. B.: Lateral variation in the production and fate of phytoplankton in a partially stratified estuary,
Coastal hypoxia responses to remediation

W. M. Kemp et al.

Mayer, M. S., Shaffner, L., and Kemp, W. M.: Nitrification potentials of benthic macrofaunal tubes and burrow walls: Effects of sediment NH$_4^+$ and animal irrigation behavior, Mar. Ecol.-Prog. Ser., 121, 157–169. 1995.

Mee, L.: Reviving dead zones, Sci. Am., 295, 78–85, 2006.

Middelburg, J. J. and Levin, L. A.: Coastal hypoxia and sediment biogeochemistry, Biogeoosciences Discuss., 6, 3655–3706, 2009, http://www.biogeosciences-discuss.net/6/3655/2009/.

Møhlenberg, F.: Effect of meteorology and nutrient load on oxygen depletion in a Danish microtidal estuary, Aquat. Ecol., 33, 55–64, 1999.

Najjar, R. G., Walker, H. A., Anderson, P. J., Barron, E. J., Bord, R. J., Gibson, J. R., Kennedy, V. S., Knight, C. G., Megenigal, J. P., O’Connor, R. E., Polsky, C. D., Psuty, N. P., Richards, B. A., Sorenson, L. G., Steele, E. M., and Swanson, R. S.: The potential impacts of climate change on the mid-Atlantic coastal region, Clim. Res., 14, 219–233, 2000.

Nestlerode, J. A. and Diaz, R. J.: Effects of periodic environmental hypoxia on predation of a tethered polychaete, Glycera americana: Implication for trophic dynamics. Mar. Ecol.-Prog. Ser., 172, 185–195, 1998.

Newcombe, C. L. and Horne, W. A.: Oxygen-poor waters in the Chesapeake Bay, Science, 88, 80–81, 1938.

Newcombe, C. L., Horne, W. A., and Shepherd, B. B.: Studies on the physics and chemistry of estuarine waters in Chesapeake Bay, J. Mar. Res., 2, 87–116, 1939.

Newell, R. I. E.: Ecological Changes in Chesapeake Bay: Are they the result of overharvesting the Eastern oyster (Crassostrea virginica)? In: Understanding the Estuary: Advances in Chesapeake Bay Research, edited by: Lynch, M. P. and Krome, E. C., Chesapeake Research Consortium Publication 129 (CBP/TRS 24/88), Gloucester Point, Virginia, 536–546, 1988.

Newell, R., and Ott, J.: Macrobenthic Communities and Eutrophication, In: Ecosystems at the Land-Sea Margin: Drainage Basin to Coastal Sea, edited by: Malone, T. C., Malej, A., Harding Jr., L. W., Smololka, N., and Turner, R. E., Coastal and Estuarine Studies Vol. 55. American Geophysical Union, Washington, D.C., 265–293, 1999.

Newell, R. I. E., Fisher, T. R., Holyoke, R. R., and Cornwell, J. C.: Influence of eastern oysters on nitrogen and phosphorus regeneration in Chesapeake Bay, USA. In: The Comparative Roles of Suspension Feeders in Ecosystems, edited by: Dame, R. and Olenin, S., NATO
Coastal hypoxia responses to remediation

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Science Series: IV – Earth and Environmental Sciences V. 47. Springer, Netherlands, 93–120, 2005.

Newell, R. I. E., Kemp, W. M., Hagy, J. D., Cerco, C. F., Testa, J. M., and Boynton, W. R.: Top-down control of phytoplankton by oysters in Chesapeake Bay, USA: Comment on Pomeroy et al. (2006), Mar. Ecol.-Prog. Ser., 341, 293–298, 2007.

Nixon, S. W., Granger, S., Buckley, B. A., Lamont, M., and Rowell, B.: A one hundred and seventeen year coastal water temperature record from Woods Hole, Massachusetts, Estuaries, 27, 397–404, 2004.

O'Donnell, J. H., Dam, H. G., Bohlen, W. F., Fitzgerald, W., Gray, P. S., Houk, A. E., Cohen, D. C., and Howard-Strobel, M. M.: Intermittent ventilation in the hypoxic zone of western Long Island Sound during the summer of 2004, J. Geophys. Res., 113, 1–13, 2008.

Officer, C. B., Biggs, R., Taft, J. L., Cronin, L. E., Tyler, M., and Boynton, W. R.: Chesapeake Bay anoxia: Origin, development, and significance, Science, 223, 22–27, 1984.

Ogi, M., Tachibana, Y., and Yamazaki, K.: Impact of the wintertime North Atlantic Oscillation (NAO) on the summertime atmospheric circulation, Geophys. Res. Lett., 30, 1704, doi:10.1029/2003GL017280, 2003.

Oguz, T. and Gilbert, D.: Abrupt transitions of the top-down controlled Black Sea pelagic ecosystem during 1960–200: Evidence for regime-shifts under strong fishery exploitation and nutrient enrichment modulated by climate-induced variations, Deep-Sea Res. Pt. I, 54, 220–242, 2007.

Omstedt, A., Pettersen, C., Rodhe, J., and Winsor, P.: Baltic Sea climate: 200 yr of data on air temperature, sea level variation, ice cover, and atmospheric circulation, Climate Res., 25, 205–216, 2004.

Orth, R., Carruthers, T., Dennison, W., Duarte, C., Fourquarean, J., Heck, K., Hughes, R., Kendrick, G., Kenworthy, J., Olyarnik, S., Short, F., Waycott, M., and Williams, S.: A global crisis for seagrass ecosystems, BioScience, 56, 987–996, 2006.

O'Shea, M. L. and Brosnan, T.: Trends in indicators of eutrophication in western Long Island Sound and the Hudson-Raritan estuary, Estuaries, 23, 877–901, 2000.

Osterman, L. E., Poore, R., Swarzenski, P., and Turner, R. E.: Reconstructing a 180-yr record of natural and anthropogenic induced low-oxygen conditions from Louisiana continental shelf sediments, Geology, 33, 329–333, 2005.

Pael, H. W., Valdes, L., Joyner, A., and Piehler, M.: Solving problems resulting from solutions: Evolution of a dual nutrient management strategy for the eutrophying Neuse River estuary,
North Carolina, Environ. Sci. Technol., 38, 3068–3073, 2004.
Park, K., Kim, C.-K., and Schroeder, W. W.: Temporal variability in summertime bottom hypoxia in shallow areas of Mobile Bay, Alabama, Estuaries Coasts 31, 54–65, 2007.
Parker, C. A. and O’Reilly, J. E.: Oxygen depletion in Long Island Sound: A historical perspective, Estuaries, 14, 248–264, 1991.
Patrick, R.: Changes in the chemical and biological characteristics of the upper Delaware River estuary in response to environmental laws. In Ecology and Restoration of the Delaware River Basin, edited by: Majumdar, S.K., Miller, E.W. and Sage L.E., The Philadelphia Academy of Science, Philadelphia, Pennsylvania, 332–359. 1988.
Peierls, B. L., Christian, R. R., and Paerl, H. W.: Water quality and phytoplankton as indicators of hurricane impacts on a large estuarine ecosystem, Estuaries, 26, 1329–1343, 2003.
Petersen, J. K., Hansen, J. W., Laursen, M. B., Clasuen, P., Carstensen, J., and Conley, D. J.: Regime shift in a coastal marine ecosystem, Ecol. Appl., 18, 497–510, 2008.
Pyke, C. R., Najjar, R. G., Adams, M. B., Breitburg, D., Kemp, W. M., Hershner, C., Howarth, R., Mulholland, M., Paolisso, M., Secor, D., Sellner, K., Wardrop, D., and Wood, R.: Climate change and the Chesapeake Bay: State-of-the-Science Review and Recommendations. A Report from the Chesapeake Bay Program Science and Technical Advisory Committee (STAC), Annapolis, MD, 2008.
Pomeroy, L. R., D’Elia, C. F., and Schaffner, L. C.: Limits to top-down control of phytoplankton by oysters in Chesapeake Bay, Mar. Ecol.-Prog. Ser., 325, 301–309, 2006.
Prins, T.C., Smaal, A., and Dame, R.: A review of the feedbacks between bivalve grazing and ecosystem processes, Aquat. Ecol., 31, 349–359, 1998.
Prueckl, J. E. and Decker, M. B.: Effects of climate on relative predation by scyphomedusae and ctenophores on copepods in Chesapeake Bay during 1987–2000, Limnol. Oceanogr., 50, 376–387, 2005.
Rabalais, N. N. and Turner, R. E.: Oxygen depletion in the Gulf of Mexico adjacent to the Mississippi River, in: Past and Present Marine Water Column Anoxia, edited by: Neretin, L. N., NATO Science Series: IV-Earth and Environmental Sciences, Kluwer Academic Publishers, Dordrecht, The Netherlands, 222–245, 2006.
Rabalais, N. N., Turner, R. E., Sen Gupta, B. K., Boesch, D. F., Chapman, P., and Murrell, M. C.: Hypoxia in the northern Gulf of Mexico: Does the science support the plan to reduce, mitigate, and control hypoxia? Estuaries Coasts, 30, 753–772, 2007.
Rabalais, N. N. and Gilbert, D.: Distribution and consequences of hypoxia, In: Watersheds,
Coastal hypoxia responses to remediation

W. M. Kemp et al.

Bays and Bounded Seas, edited by: Urban, E., Sundby, B., Malanotte-Rizzoli, P., and Melillo, J. M., Island Press, Washington, D.C., 209–226, 2009.

Reish, D. J.: The seasonal settlement of polychaete larvae before and after pollution abatement in Los Angeles-Long Beach harbors, California, B. Mar. Sci., 67, p. 672, 2000.

Revsbech, N. P., Risgaard-Petersen, N., Schramm, A., and Nielsen, L. P.: Nitrogen transformations in stratified aquatic microbial ecosystems, A. Van Leeu. J. Microb., 90, 361–375, 2006.

Roman, M. R. and Gauzens, A.: Copepod grazing in the equatorial Pacific, Limnol. Oceanogr., 42, 623–634, 1997.

Sampou, P. A. and Kemp, W. M.: Factors regulating plankton community respiration in Chesapeake Bay, Mar. Ecol.-Prog. Ser., 110, 249–258, 1994.

Scavia, D., Justic, D., and Bierman, Jr., V. J.: Reducing hypoxia in the Gulf of Mexico: Advice from three models, Estuaries, 27, 419–425, 2004.

Scheffer, M., Carpenter, S., Foley, J. A., Folke, C., and Walker, B.: Catastrophic shifts in ecosystems, Nature, 413, 591–596, 2001.

Scheffer, M. and Jeppesen, E.: Regime shifts in shallow lakes, Ecosystems, 10, 1–3, 2007.

Scully, M. E., Friedrichs, C., and Brubaker, J.: Control of estuarine stratification and mixing by wind-induced straining of the estuarine density field, Estuaries, 28, 321–326, 2005.

Seitzinger, S.: Denitrification in freshwater and coastal marine systems: Ecological and geochemical significance, Limnol. Oceanogr., 33, 702–724, 1988.

Sharples, J., Simpson, J. H., and Brubaker, J. M.: Observations and modelling of periodic stratification in the Upper York River Estuary, Virginia, Estuar. Coast. Shelf S., 38, 301–312, 1994.

Shen, J., Wang, T., Herman, J., Mason, P., and Arnold, G.: Hypoxia in a coastal embayment of the Chesapeake Bay: A model diagnostic study of oxygen dynamics, Estuaries Coasts, 31, 652–663, 2008.

Slomp, C. P. and Van Cappellen, P.: The global marine phosphorus cycle: sensitivity to oceanic circulation, Biogeosciences, 4, 155–171, 2007, http://www.biogeosciences.net/4/155/2007/.

Soetaert, K., Middelburg, J. J., Heip, C., Meire, P., Van Damme, S., and Maris, T.: Long-term change in dissolved inorganic nutrients in the heterotrophic Scheldt estuary (Belgium, The Netherlands), Limnol. Oceanogr., 51, 409–423, 2006.

Soetaert, K. and Middelburg, J. J.: Modeling eutrophication and oligotrophication of shallow-
water marine systems: The importance of sediments under stratified and well-mixed conditions, Hydrobiologia, 629, 239–254, 2009.
Stanley, D. W. and Nixon, S. W.: Stratification and bottom-water hypoxia in Pamlico River estuary, Estuaries, 15, 270–281, 1992.
Stigebrandt, A. and Gustafsson, B. G.: Improvement of Baltic Proper water quality using large-scale ecological engineering, Ambio, 36, 281–286, 2007.
Stock, C. and Dunne, J.: Controls on the ratio of mesozooplankton production to primary production in marine ecosystems, Deep-Sea Res. Pt. I, in press, 2009.
Sylvan, J. B., Dortch, Q., Maier Brown, A. F. Morrison, W., and Ammerman, J. W.: Phosphorus limits phytoplankton growth on the Louisiana shelf during the period of hypoxia formation, Environ. Sci. Technol., 40, 7548–7553, 2006.
Taylor, A. H. and Stephens, J. A.: The North Atlantic Oscillation and the latitude of the Gulf Stream, Tellus, 50A, 134–142, 1998.
Testa, J. M., Kemp, W. M., Boynton, W. R., and Hagy, J. D.: Long-term changes in water quality and productivity in the Patuxent River estuary: 1985 to 2003, Estuaries Coasts, 31, 1021–1037, 2008.
Testa, J. M., Murphy, R., and Kemp, W. M.: Enhancement of sediment nitrogen recycling in Chesapeake Bay corresponds to increases in hypoxic water volume, in preparation, 2009.
Tiedje, J. M.: Ecology of denitrification and dissimilatory nitrate reduction to ammonium, In: Biology of Anaerobic Microorganisms, edited by: Zehnder, A. J. B., John Wiley, New York, 179–244, 1987.
Tinsley, D.: The Thames estuary: a history of the impact of humans on the environment and a description of the current approach to environmental management, In: A Rehabilitated Estuarine Ecosystem: The Environment and Ecology of the Thames Estuary, edited by: Attrill, M. J., Kluwer Academic Publishers, London, 2–25. 1998.
Turner, R. E., Schroeder, W. W., and Wiseman, Jr., W.J.: The role of stratification in the deoxygenation of Mobile Bay and adjacent shelf bottom waters, Estuaries, 10, 13–19, 1987.
Turner, R. E., Rabalais, N. N., Swenson, E. M., Kasprzak, M., and Romaine, T.: Summer hypoxia in the northern Gulf of Mexico and its prediction from 1978 to 1995, Mar. Environ. Res., 59, 65–77, 2005.
Turner, R. E., Rabalais, N. N., Fry, B., Atilla, N., Milan, C. S., Lee, J. M., Normandeau, C., Oswald, T. A., Swenson, E. M., and Tomasko, D. A.: Paleo-indicators and water quality change in the Charlotte Harbor estuary (Florida), Limnol. Oceanogr., 51, 518–533, 2006.
Turner, R. E., Rabalais, N. N., Alexander, R. B., Mclissac, G., and Howarth, R. W. : Characterization of nutrient, organic carbon, and sediment loads and concentrations from the Mississippi River into the Northern Gulf of Mexico, Estuaries Coasts, 30, 773–790, 2007.

Turner, R. E., Rabalais, N. N., and Justic, D.: Gulf of Mexico hypoxia: Alternate states and a legacy, Environ. Sci. Technol., 42, 2323–2327, 2008.

Tyler, R. M., Brady, D. C., and Targett, T.: Temporal and spatial dynamics of diel-cycling hypoxia in estuarine tributaries, Estuaries Coasts, 32, 123–145, 2009.

Valiela, I.: Marine Ecological Processes, 2nd Edition, Springer-Verlag, New York, 1995.

Van Beusekom, J. E. E., Loebl, M., and Martens, P.: Distant riverine nutrient supply and local temperature drive the long-term phytoplankton development in a temperate coastal basin, J. Sea Res., 61, 26–33, 2009.

Vaquer-Sunyer, R. and Duarte, C.: Thresholds of hypoxia for marine biodiversity, P. Natl. Acad. Sci. USA, 105, 15452–15457, 2008.

Welsh, B. L. and Eller, F. C.: Mechanisms controlling summertime oxygen depletion in western Long Island Sound, Estuaries, 14, 265–278, 1991.

Welsh, D. T.: It’s a dirty job but someone has to do it: The role of marine benthic macrofauna in organic matter turnover and nutrient recycling to the water column, Chem. Ecol., 19, 321–342, 2003.

Wenner, E., Sanger, D., Arendt, M., Holland, F. A., and Chen, Y.: Variability in dissolved oxygen and other water-quality variables within the national estuarine research reserve system, J. Coastal Res., Special Issue 45, 17–38, 2004.

Westrich, J. T. and Berner, R. A.: The role of sedimentary organic matter in bacterial sulfate reduction: The G model tested, Limnol. Oceanogr., 29, 236–249, 1984.

Wiseman, W. J., Rabalais, N. N., Turner, R. E., Dinnel, S. P., and MacNaughton, A.: Seasonal and interannual variability within the Louisiana coastal current: stratification and hypoxia, J. Marine Syst., 12, 237–248, 1997.

Wilson, R. E., Swanson, R. L., and Crowley, H. A.: Perspectives on long-term variations in hypoxic conditions in western Long Island Sound, J. Geophys. Res., 113, C12011, doi:10.1029/2007JC004693, 2008.

Wulff, F., Savchuk, O. P., Sokolov, A., Humborg, C., and Mörth, C.-M.: Management options and effects on a marine ecosystem: Assessing the future of the Baltic, Ambio, 36, 243–249, 2007.

Yin, K., Lin, Z., and Ke, Z.: Temporal and spatial distribution of dissolved oxygen in the Pearl ...
Coastal hypoxia responses to remediation

W. M. Kemp et al.

River Estuary and adjacent coastal waters, Cont. Shelf Res., 24, 1935–1948, 2004.
Zillén, L., Conley, D. J., Andrén, T., Andrén, E., and Gjörck, S.: Past occurrences of hypoxia in the Baltic Sea and the role of climate variability, environmental change and human impact, Earth-Sci. Rev., 91, 77–92, 2008.
Table 1. Characteristic categories of coastal hypoxia defined by time-scales of variability and partially controlled by water column depth.

| Hypoxia Type   | Dominant Time-Scale | Depth Range (m) | System Types       | Key Controls                          | Example Systems* |
|----------------|---------------------|-----------------|--------------------|---------------------------------------|-----------------|
| (1) "Permanent"| Decades-            | >300 m          | Silled Fjord,      | Depth, mixing, Stratification         | Black Sea^a      |
|                | centuries           |                 | Inland Sea         | Organic input flushing                 | Baltic Sea^b     |
| (2) "Persistent Seasonal" | Months     | 10–100 m       | Estuary, Shelf plume | River flow, Temperature, Organic input | Chesapeake Bay^c |
|                | –Stratified         |                 |                    |                                        | Pensacola Bay^d  |
|                |                     |                 |                    |                                        | Changjiang plume^e|
|                | –Mixed              | 5–15 m          | Tidal river        | River flow, Tidal range, Organic input | Thames Estuary^f |
|                |                     |                 |                    |                                        | Scheldt Estuary^g|
| (3) "Episodic" (Intermittent) | Weeks-years | 5–20 m         | Lagoons, Bays      | Wind & Tides, Storms, Organic input   | Mobile Bay^h     |
|                |                     |                 |                    |                                        | Neuse Estuary^i  |
| (4) "Diel"    | Hours-              | 1–5 m           | Lagoons, Bays      | Wind, Light, Nutrient input, Organic input | DE Inland Bays^j |
|                | -days               |                 |                    |                                        | Waquoit Bay^k    |

*References: a Mee (2006), b Conley et al. (2007), c Kemp et al. (1992), d Hayg and Murrell (2007), e Chen et al. (2007), f Andrews and Rickard (1980), g Soertaert et al. (2006), h Turner et al. (1987), i Borsuk et al. (2001), j Tyler et al. (2009), k D'Avanzo and Kremer (1994).
Table 2. Summary of reported changes in oxygen for coastal ecosystems in response to remediation (Nut.=Nutrients, Reg. Shift=Regime Shift, No Red.=No Reduction in Nutrient Load).

| System                | Hypoxia Type          | Loading Source | Target | Response       | Trajectory Type | Reference               |
|-----------------------|-----------------------|----------------|--------|----------------|-----------------|-------------------------|
| Danish Coastal Waters | Seasonal Stratified   | Diffuse/Point  | Nut.   | None           | Reg. Shift      | Conley et al. 2007     |
| N. Gulf of Mexico     | Seasonal Stratified   | Diffuse        | Nut.   | No Red.        | Reg. Shift      | Turner et al. 2008     |
| Lower Potomac         | Seasonal Stratified   | Diffuse/Point  | Nut.   | None           | Hysteresis      | Jaworski unpubl.       |
| Chesapeake Bay        | Seasonal Stratified   | Diffuse/Point  | Nut.   | –              | Reg. Shift      | Hagy et al. 2004       |
| NW Shelf Black Sea    | Seasonal Stratified   | Diffuse/Point  | Nut.   | +              | Hysteresis      | Mee 2006               |
| Baltic Sea proper     | Permanently Stratified| Diffuse/Point  | Nut.   | No Red.        | None            | Conley et al. 2009a    |
| Lajaalahti Bay        | Seasonal Mixed        | Point          | Nut./BOD | + | Linear | Kauppila et al. 2005 |
| Boston Harbor         | Seasonal Stratified   | Point          | Nut./BOD | + | Unknown | Diaz et al. 2008     |
| Delaware estuary      | Seasonal Mixed        | Point          | Nut./BOD | + | Linear | Patrick 1988        |
| Scheldt estuary       | Seasonal Mixed        | Diffuse/Point  | Nut./BOD | + | Linear | Soetaert et al. 2006 |
| Upper Potomac         | Seasonal Stratified   | Point          | Nut./BOD | + | Linear | Jaworski unpubl.     |
| Western LIS           | Seasonal Stratified   | Point          | Nut./BOD | – | Hysteresis | Wilson et al. 2008   |
| Thames estuary        | Seasonal Mixed        | Point          | BOD    | + | Threshold | Andrews and Rickard 1980 |
| Mersey estuary        | Seasonal Mixed        | Point          | BOD    | + | Unknown | Jones 2006           |
| Forth estuary         | Seasonal Mixed        | Point          | Nut./BOD | + | Linear | Balls et al. 1996   |
| Lower Hudson          | Seasonal Stratified   | Point          | Nut./BOD | + | Linear | Brosnan and O’Shea 1996 |
| New York Harbor       | Seasonal Stratified   | Point          | Nut./BOD | + | Linear | Parker and O’Reily 1991 |
| East River            | Seasonal Mixed        | Point          | Nut./BOD | + | Linear | Parker and O’Reily 1991 |
| Charlotte Harbor      | Seasonal Stratified   | Point          | Nutrients | + | Unknown | Turner et al. 2006  |
| Los Angeles Harbor    | Seasonal Mixed        | Point          | Nut./BOD | + | Unknown | Reish 2000          |
| Raritan Bay           | Seasonal Mixed        | Point          | Nut./BOD | + | Linear | Parker and O’Reily 1991 |
| New River estuary     | Seasonal Stratified   | Point          | Nut.   | + | Linear | Mallin et al. 2006   |
Fig. 1. Six hypothetical response trajectories of oxygen conditions in relation to changes in nutrient load. Trajectories include: (a) linear relationship of hypoxia to load with immediate responses, (b) direct threshold response to nutrient load, (c) delayed (hysteretic) response to nutrient load, (d) threshold response with hysteresis, (e) linear response with shifted baseline due to changes in other forcing variables, and (f) threshold response with hysteresis with a shifted baseline.
Fig. 2. Conceptual diagram of processes influencing hypoxia. The left side represents the suite of processes which increase hypoxic conditions, while the right side details processes that relieve hypoxia. Hypoxia is enhanced in most all systems by higher freshwater inputs, higher nutrient and BOD loads from more human-impacted watersheds, reduced nutrient filtration with loss of marshes and submerged aquatic vegetation, reduced filtration of phytoplankton by depleted bivalve communities, elevated temperature, reduced ventilation of deeper waters with atmospheric O$_2$, and enhanced sediment nutrient recycling with hypoxia-induced changes in biogeochemistry and loss of bioturbating benthic fauna.
Fig. 3. Time series (1955–1980) observations in the upper Thames River estuary (England) for (a) BOD load from major sewage treatment plants and summer O₂ % saturation and (b) relationship of O₂ % saturation deficit (concentration units below mean saturation) to BOD load showing threshold response of O₂ to reduced BOD load. Data sources referenced within text.
Fig. 4. Time series (1965–2002) observations in the Scheldt estuary (Netherlands) for (a) BOD and O$_2$ % saturation deficit, and for (b) DIN concentrations and fraction as NH$_4^+$ and (c) relationship of O$_2$ deficit index to DIN concentration, showing a favorable linear shifting baseline response of O$_2$ to reduced DIN and BOD loads. Data sources referenced within text.
Fig. 5. Time series (1965–2000) observations in Laajalahti Bay (Finland) of (a) BOD and TN loads and of (b) O$_2$ % saturation and chlorophyll-a. Relationship of (c) O$_2$ % saturation to BOD Load and (d) O$_2$ % saturation to chlorophyll-a. Data sources referenced within text.
Fig. 6. Multi-decade time series data of hypoxia per unit nitrogen load in the (a) Patuxent River estuary, USA, (b) northern Gulf of Mexico, USA, and (c) Chesapeake Bay, USA. Inset figures are relationships between N load and hypoxia for each system during periods before and after statistically significant change points (vertical dashed lines) in time-series of hypoxia per unit N load. Data sources referenced within text.
Fig. 7. Time series and trend lines (1985–2003) of annual mean June-August concentrations (a) *Mnemiopsis* spp. biovolume, (b) *Acartia tonsa* abundance, (c) chlorophyll-*a*, and (d) DIN, and (e) mean annual DIN loads from upstream sewage plants, (f) upper Patuxent summer (June–August) mean concentrations of bottom water O$_2$, (g) hypoxic volume days in the entire Patuxent River estuary, and (h) mean annual DIN inputs from Chesapeake Bay to the Patuxent River estuary. Data sources referenced within text.
Fig. 8. Time series (1960–2001) observations for (a) Danube River watershed nitrogen fertilizer use and summer hypoxic area in the northwest shelf of the Black Sea and (b) response trajectory of hypoxic area on the northwest shelf of the Black Sea to interannual changes in nitrogen fertilizer use derived from time series data. Data sources referenced within text.
Fig. 9. Time-series (1970–2000) of (a) annual TP loads to the Baltic Sea, (b) annual concentration of DIP (solid circles) and DIN (open circles) in the Baltic Proper for depths <100 m, (c) index of water inflow to the Baltic (solid bars) and deep water (200 m) salinity (grey circles) in the Gotland Deep, and (d) hypoxic area in the Baltic Proper, Gulf of Riga, and Gulf of Finland from 1970 to 2000. Data sources referenced within text.
Fig. 10. Time-series (1945–2007) of (a) Susquehanna River Flow at Conowingo, MD, (b) January–May NO$_3^-$ loads to Chesapeake Bay, (c) July hypoxic volume in Chesapeake Bay, (d) mean annual water temperature at Solomons, MD, (e) January to March North Atlantic Oscillation, and (f) annual Maryland (MD) and Virginia (VA) oyster harvest. Data sources referenced within text.