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Science and management in four U.S. coastal ecosystems dominated by land-ocean interactions

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Abstract. The influence of science in the recognition of the effects of landscape changes on coastal ecosystems and in the development of effective policy for managing and restoring these ecosystems is examined through four case studies: Chesapeake Bay, San Francisco Bay, the Mississippi Delta, and Florida Bay. These ecosystems have undergone major alterations as a result of changes in the delivery of water, sediments and nutrients from their watersheds. Both science and management have been challenged by the spatial, functional and temporal scale mismatches inherent in the watershed-coastal ecosystem relationship. Key factors affecting the influence of science on management include (1) sustained scientific investigation, responsive to but not totally defined by managers; (2) clear evidence of change, the scale of the change and the causes of the change; (3) consensus among the scientific communities associated with various interests; (4) the development of models to guide management actions; (5) identification of effective and feasible solutions to the problems.

Keywords: Chesapeake Bay; Environmental policy; Estuaries; Florida Bay; Mississippi Delta; Nutrients; River discharge; Salinity; San Francisco Bay.

Introduction

The emerging, widespread environmental threats confronting coastal ecosystems around the world, such as eutrophication, hydrologic disruption, introduction of non-indigenous species, and global climate change, pose new challenges to environmental policy, management and science. Meeting these challenges requires different approaches from those used to manage traditional problems such as point-source discharges of industrial and municipal effluents, coastal land use, direct habitat destruction, and oil spills (Anon. 1994). In particular, there is a growing appreciation that coastal ecosystems are heavily influenced by human activities on the land–often hundreds of kilometers from the coast–as well as by activities in the coastal zone itself. This is well exemplified in the United States, with its large continental land mass and rivers which drain into coastal ecosystems important in terms of their economic value and natural heritage.

In this paper I will relate the role science has played in understanding these important connections between activities on the land and the coastal zone and guiding effective management solutions based on experiences in four ecosystems: Chesapeake Bay, San Francisco Bay, the Mississippi Delta, and Florida Bay (an ecosystem which has recently undergone major changes which have just begun to attract concerted scientific appraisal) (see Fig. 1). My analysis of the history of the influence of science is based in part on the published perspectives of others and in part on my own experiences as a research scientist, scientific administrator, or scientific advisor within each of the four coastal ecosystems (see also Boesch 1995). After reviewing these four case studies, I will then attempt to draw some generalities and to develop recommendations for improving the effectiveness of science in guiding the formulation of effective policies and the implementation of these policies.

Chesapeake Bay

The largest estuary in the United States, the Chesapeake Bay has been the site of extensive scientific research and is the subject of what is probably the world's most ambitious effort to manage and restore a coastal ecosystem. Despite the great attention the Chesapeake has received by scientists and environmental management, the dimensions of environmental change that has taken place in the Chesapeake Bay and its tributary sub-estuaries and their relationship to changes on the land have not been appreciated until relatively recently. The most pervasive and consequential environmental change has been an increase in nutrients reaching the bay. Nutrient enrichment has caused changes in plankton communities, productivity, the extent of bottom-water hypoxia, and increased turbidity. Increased hypoxia is responsible for reductions in some living resources and increased turbidity has caused great reductions in submersed vascular vegetation.

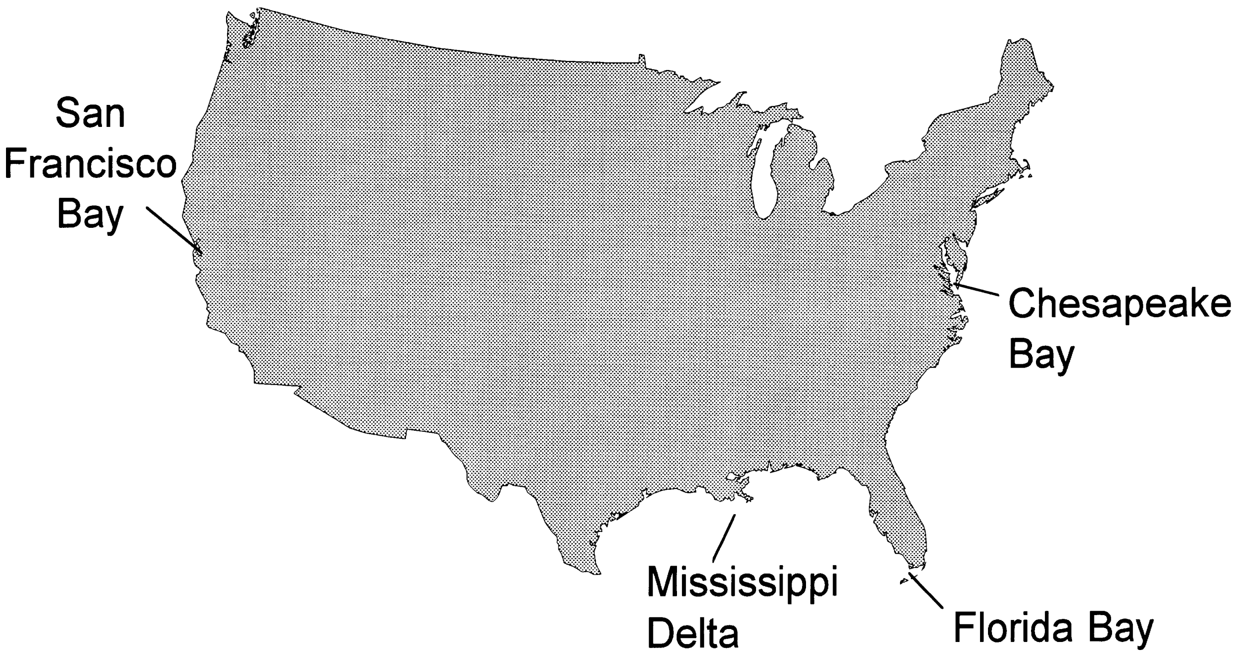

Fig. 1. Conterminous United States showing the location of the four coastal ecosystems discussed.

Through paleontological and chemical analyses of cores of sediment from the depositional deep trough of the bay, Cooper (Cooper & Brush 1991; Cooper 1995) demonstrated that the sedimentation rate, primary production (reflected in the accumulation of carbon and biogenic silica), and extent and severity of anoxia (reflected in the formation of iron pyrite from reduced sulphur) began to change dramatically around 1760 following widespread land clearing for agriculture by European colonists (Fig. 2). In addition, the diatom community shifted from a diverse community with nearly equal representation by centric and pennate forms to a less diverse community heavily dominated by centric diatoms as the enriched and more turbid system became dominated by planktonic rather than benthic primary production. Although, according to Cooper & Brush (1991), intense seasonal anoxia is mainly a phenomenon of the late twentieth century, the eutrophication of the Chesapeake started with European settlement and, in particular, the proliferation of agriculture.

The combination of a large watershed (166000 km ^{2}), significant volume in relation to freshwater input and tidal exchange, and partial stratification disposes the Chesapeake Bay to nutrient retention and recycling. Therefore, this ecosystem is particularly sensitive to nutrient inputs from the watershed (Boynton et al. 1995). Yet, for many years both the scientific and management community focused on smaller scale human impacts associated with activities directly affecting portions of the bay while assuming that, with the exception of widespread overfishing of certain species, the bay was in good health overall.

Malone et al. (1993) examined in considerable detail how this view began to change in the early 1970s and how the importance of nutrient loadings to the Chesapeake Bay was embraced by scientists, managers and policy makers. By the 1960s the upper end of the Potomac River sub-estuary below Washington, DC had become obviously over-enriched as evidenced by massive algal blooms and depleted oxygen (Jaworski 1990). Following the recent successes in addressing over-enrichment problems in Lake Erie, one of the North American Great Lakes, major federal investments were made in 1972 to provide advanced treatment, in particular phosphorus removal, of municipal wastewaters of metropolitan Washington. This was after all the nation's capital and these were the days of the Great Society when government was thought to accomplish what it set out to do. The results were dramatic: water quality greatly improved, nuisance algal blooms retreated, and fish returned to the upper Potomac (Jaworski 1990). This experience had the effect of instilling confidence in regional environmental managers that commitment to waste treatment would yield positive results, but it also focused attention on point sources of pollutants, obscuring the effect of non-point sources on the bay.

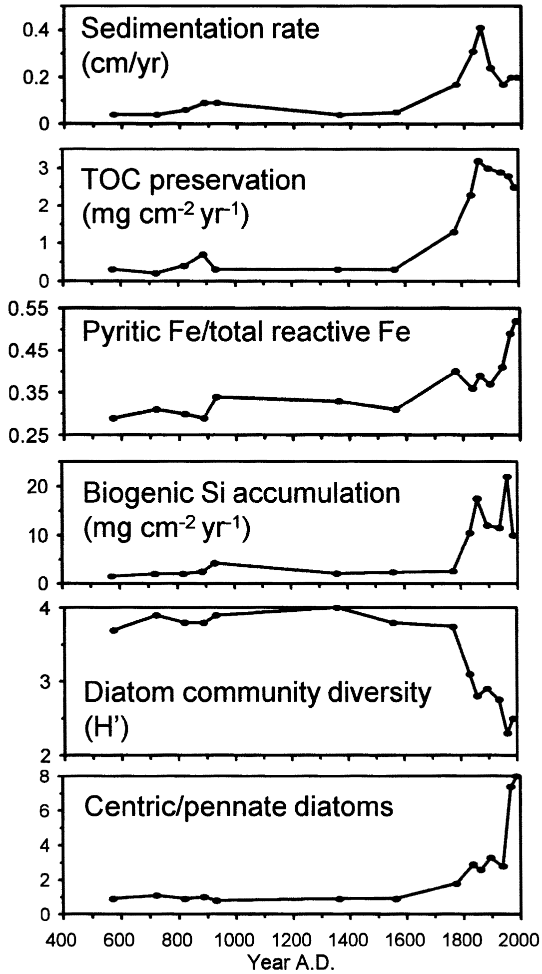

Fig. 2. The history of eutrophication of the Chesapeake Bay as revealed in a sediment core (R4-50) from the central channel (Cooper & Brush 1991; Cooper 1995). Sedimentation increased greatly following extensive land clearing for agriculture around 1760. The increased deposition of total organic carbon (TOC) and biogenic silica reflect the significant enrichment of the estuary by nutrients following this landscape change. A reduction of diatom community diversity and an increase in centric diatoms reflect a shift from a benthic-dominated to a plankton-dominated, light-limited system. Seasonal hypoxia, reflected by an increase of pyritic iron, has intensified in the latter half of this century.

Also in 1972, the entire Chesapeake watershed was affected by record floods associated with the passage of the tropical storm Agnes. The resulting freshet had profound effects on the Chesapeake Bay and its river sub-estuaries in terms of circulation, sedimentation, chemical inputs, biotic changes, and declines in important fisheries. It forced the scientific community and some of the management community to begin to think of the bay not as a vast arm of the sea, but as an estuarine ecosystem heavily influenced by its watershed (Malone et al. 1993). Following Agnes, concern about large scale changes in the bay, such as the declines of submersed vascular plants in both the upper bay and lower bay, stimulated Congressional pressures on the Federal government to study and fix the problems afflicting this ecosystem. The resulting multi-year Chesapeake Bay study began in 1978 with heavy financial support by the U.S. Environmental Protection Agency. It focused on aquatic vegetation, toxic materials and nutrient enrichment. Interestingly, the studies of nutrient enrichment never advanced to the stage of field research, involving instead a series of workshops and conferences. Although this did help coalesce opinions about the importance of non-point sources, for example, it was left to the studies of submersed vegetation to identify widespread nutrient over-enrichment as the primary culprit in the disappearance of these bay grasses.

Meanwhile, controversies developed over the effects of population growth and expanding wastewater discharges into the Patuxent River sub-estuary, just to the north of the Potomac. On one side of the controversy were estuarine scientists, who had been studying the Patuxent and had become alarmed at signs of over-enrichment, and their allies, officials of the rural local governments at the lower end of the estuary. On the other were state and federal environmental officials who were planning the wastewater treatment plants to handle the population growth in suburban upstream areas. A particular bone of contention was the need to remove nitrogen as well as phosphorus in wastewater treatment. State and federal managers and their engineering consultants, borrowing from the experience in the upper Potomac sub-estuary, held that phosphorus removal was all that was required. But the upper Potomac is freshwater, and the estuarine scientists pointed to literature which indicated that N rather than P tended to be the limiting nutrient for marine phytoplankton. The phytoplankton of the mesohaline lower Patuxent, they argued, was likely to be N-limited, therefore costly nitrogen removal would be required in the new treatment plants to avoid further degradation in the estuary. A lawsuit ensued, with estuarine scientists appearing for the plaintiff in opposition to the very agencies which supported their research. The matter was ultimately settled in 1981 by a 'charette' in which the parties committed to hammer out a consensus during a time-constrained meeting. The agreement to remove N was a milestone in the scientific influence on nutrient management policies.

With the conclusion of the five-year Chesapeake Bay study, the three states (Virginia, Maryland and Pennsylvania) which occupy most of the bay's watershed, the District of Columbia which includes the nation's capital, and the federal government represented by the Environmental Protection Agency endorsed the first Chesapeake Bay Agreement in 1983, thus launching the intergovernmental Chesapeake Bay Program for restoration of the bay. A Scientific and Technical Advisory Committee was formed and in 1986 it released a report which presented clear and compelling evidence that both P and N removal would be required to improve water quality in the bay and its tributaries and, very importantly, that cost efficient technologies were available for the combined removal of these nutrients. This scientific consensus provided the rationale and credibility for the bold action of the Second Chesapeake Bay Agreement in 1987 which committed the signatories to achieving a 40 % reduction in controllable inputs of N and P by the year 2000.

To the scientists involved in these debates this adaptation to scientific understanding may seem to have been painfully slow, but, because I was away from the Chesapeake scientific community during the 1980s, I was impressed by the speed of the effect of science on the management paradigm. Hennessey (1994) reviewed the Chesapeake Bay Program and observed that the evolution and refinement of its objectives and its use of monitoring and scientific information evidenced an effective application of adaptive management. But, in the sense of the originators of this concept, adaptive management involves a more structured approach to environmental management in the face of high uncertainty which emphasizes learning and pursuing multiple options (Walters 1986). While this may be true of the Chesapeake Bay Program viewed from a distance or over several decades, adaptive management as a concerted process requires even tighter linkages and shorter time steps.

Two technical tools have been of central importance in the Chesapeake Bay Program since its inception: modeling and monitoring. Over ten years of monitoring of water quality and living resources has made continuous and otherwise unattainable environmental data available to regional scientists for use in extending their research. Furthermore, many researchers actually perform some of the monitoring. There has likewise been a mutualistic relationship between researchers and modelers. An array of linked and coupled models has been developed to predict water quality and ecological conditions in the bay in response to inputs of energy, water and nutrients from the atmosphere, watershed, and coastal ocean. The water quality model of the main stem of the bay started as a hydrodynamic model of the type used in sanitary engineering analyses of the effects of biological oxygen demand of wastes on oxygen conditions. Through the creative tension between scientific critics of the model's assumptions and the practically minded engineers, many new discoveries about biological and chemical processes in the bay have been incorporated into the water quality model (e.g. regarding factors affecting nutrient limitation of primary production, effects of animals on nutrient flux at the seabed, and grazing and settling rates of different forms of phytoplankton) such that today it is one of the most realistic and effective coastal ecosystem models that exist. So much have managers come to rely, and perhaps over-rely, on this model that many will only believe something when it is 'confirmed' by the model. A case in point is the announcement in 1994 that the model has demonstrated that phytoplankton production in the bay is N-limited –this eight years after results of mesocosm experiments were published which clearly showed this to be the case!

Now, the principal weak links in the models concern the watershed rather than the estuary. The watershed model is an adaptation of generic streamflow modeling and is not built on as rich an understanding of how this particular ecosystem works as the bay water quality model. There have not been comparable investments in advancing hydrology, geochemistry and ecology in the watershed and the terrestrial and freshwater scientists are not well linked with their estuarine counterparts. This is unfortunate because, as the Chesapeake Bay Program emphasizes non-point source control within the tens of hydrologic units which comprise the Chesapeake watershed through what is known as Tributary Strategies, models well grounded in scientific understanding will be essential in guiding local communities to the most cost-effective targets for reducing the nutrients which actually reach the bay.

San Francisco Bay

The San Francisco Bay estuary, including the large tidal delta at the confluence of the Sacramento and San Joaquin rivers, is perhaps the major U.S. estuary most modified by human activity (Nichols et al. 1986). Although the Spanish settled in the area in 1769, the bay remained little affected until the discovery of gold in the Sierra Nevada foothills in 1848. Hydraulic mining of ore resulted in massive downstream sedimentation in the upper bay. Virtually all of the freshwater marshes of the delta were reclaimed for agriculture and salt marshes were filled or diked. Of the original 2200 km^2 of tidal marsh, only 125 km^2 of undiked marshes remained in 1986. Once abundant populations of commercial fishery species have been over-harvested and otherwise affected by habitat degradation to the point that only herring and anchovies are harvested today. Many nonindigenous species of invertebrates and fishes were introduced either purposefully or inadvertently with transplanted oysters or via ship ballast. Many of these exotics have established populations and, in fact, now constitute the dominant biota in the bay (Nichols et al. 1986; Carlton & Geller 1993).

Presently, the most significant management issue for the San Francisco Bay estuary is the consumption and diversion of fresh water for agriculture and for urban uses in central and southern California. To serve these needs the world's largest human-made water system removes about 40% of the historic flow of the Sacramento-San Joaquin river system for local consumption upstream and in the delta, and exports another 24% in aqueducts for agricultural and municipal consumption elsewhere. By the mid-1980s the flow actually reaching the estuary had decreased to less than 40 % of historic levels, and was projected to decline below 30 % by the year 2000 (Nichols et al. 1986).

The consequences of reduced river inflow to the estuarine ecosystem have been profound and include interference with migrations of anadromous fish species (i.e. fishes that migrate into fresh water to spawn); changes in estuarine circulation and increased residence time; upstream movement of isohalines and the null zone where sediments and phytoplankton accumulate; and suppression of the pelagic food web (Nichols et al. 1986). However, the lack of clear consensus among scientific and technical experts representing different interests on these effects and relationships led to the continued low priority given to the estuary in water allocation decisions, particularly in the face of strong and tangible interests of the other water users. The lack of definable and achievable objectives for the estuary led to what Kimmerer & Schubel (1994) referred to as 'regulatory gridlock'. The lack of technical consensus may be attributed to the difficulties in bridging the advocacy coalitions (Sabatier 1994) of agency representatives, resource users and scientists which form around specific resources or concerns, e.g. agricultural and municipal water supplies, fisheries, wildlife, or water quality.

A major breakthrough recently occurred as a result of a concerted effort to forge consensus on what management criteria should be used to guide water allocation to the estuary. This was effected through a series of workshops (Kimmerer & Schubel 1994) and an innovative statistical analysis relating the position of isohalines in the estuary to key ecosystem variables, including several directly related to important living resources (Jassby et al. 1995). It was shown that the longitudinal position in the estuary of the two practical salinity units (psu) isohaline measured 1 m off the bottom was directly related to such variables as total input of organic carbon including *in situ* production, biomass of molluscs, survival from egg to young-of-the year and year class strength of striped bass, survival of salmon smolts passing through the delta, and the abundance of several important prey species. Freshwater inflow can thus be regulated by managing releases from upstream dams and withdrawals from the delta to maintain the desired position of the 2 psu isohaline. After years of impasse and protracted negotiation between the Federal and State governments, this management guideline has now been included in the 1994 agreement for water allocation.

Mississippi Delta

Even in comparison to such expansive catchment areas as those of the Chesapeake and San Francisco bays, the catchment of the Mississippi River is vast, over 3.3 million km^2, including 41% of the conterminous United States. In contrast to the other coastal ecosystems considered here, the coastal area receiving water of the Mississippi is not a semi-enclosed embayment, but a distributary delta and the open continental shelf. Like the Chesapeake Bay and San Francisco Bay watersheds and river tributaries, the Mississippi watershed has also been greatly changed by agricultural conversion and damming (Meade 1995). But the flow of the Mississippi has also been greatly affected by channel deepening and straightening for navigation and by an extensive flood-control system of earthwork levees, revetments, weirs and dredged channels that has isolated most riverine wetlands in the flood plain (Turner & Rabalais 1991).

At the mouth of the Mississippi a vast distributary deltaic plain has been constructed by fluvial and marine processes during the past 7000 years, following Holocene transgression of sea level (Boesch et al. 1994). This deltaic plain includes extensive tidal (estuarine) wetlands and lagoons between the active or abandoned distributaries of the Mississippi Delta. Normally, the river and its distributaries –several of which were active at the same time– would overtop their banks during spring floods, bringing fresh water and sediments to the extensive wetlands of the interdistributary lagoon basins. However, soon after colonization of New Orleans by the French in 1719, construction of flood protection levees and closure of minor distributary channels began, culminating in an unbroken barrier of levees extending to the hub of the distributaries of the present active delta, a rather modest delta precariously perched at the edge of the continental shelf. Following catastrophic floods in 1927 a controlled diversion of 30% of the flow of the Mississippi and Red rivers was made into the Atchafalaya River in order to relieve the hydrologic inefficiency that resulted from so constraining a long channel to the sea. Thus, today the mighty Mississippi has two effective mouths, the deep-water birdsfoot delta and the entry of the Atchafalaya into a large shallow embayment.

Major changes have taken place in the Mississippi deltaic plain and in the offshore waters of the continental shelf during the latter half of the twentieth century. Best documented is the accelerated rate of coastal wetland loss from marshes and swamps to open water (Fig. 3), and conversion to more salt tolerant species of wetland vegetation. By the late 1960s, approximately 73 km^2 of vegetated wetlands were being lost per year (Boesch et al. 1994). The factors responsible for this massive wetland change are multiple, complex and difficult to apportion but are related to widespread channelization of the wetlands for navigation and oil and gas extraction, increasing salinity, and a deficit in the aggradation of soil (mineral sediments and peat) in the rapidly subsiding marshes. The high rate of relative sea level rise resulting from the rapid regional subsidence offers a model for forecasting the effects of accelerated eustatic sea level rise on coastal environments (Day & Templet 1989). Prevention by the levees of the introduction of fresh water and sediments into the wetlands and lagoons was certainly a factor in wetland loss, but it appears that channelization was largely responsible for the great increase in wetland loss between 1950 and 1980 (Boesch et al. 1994). Nonetheless, it is widely held that the long-

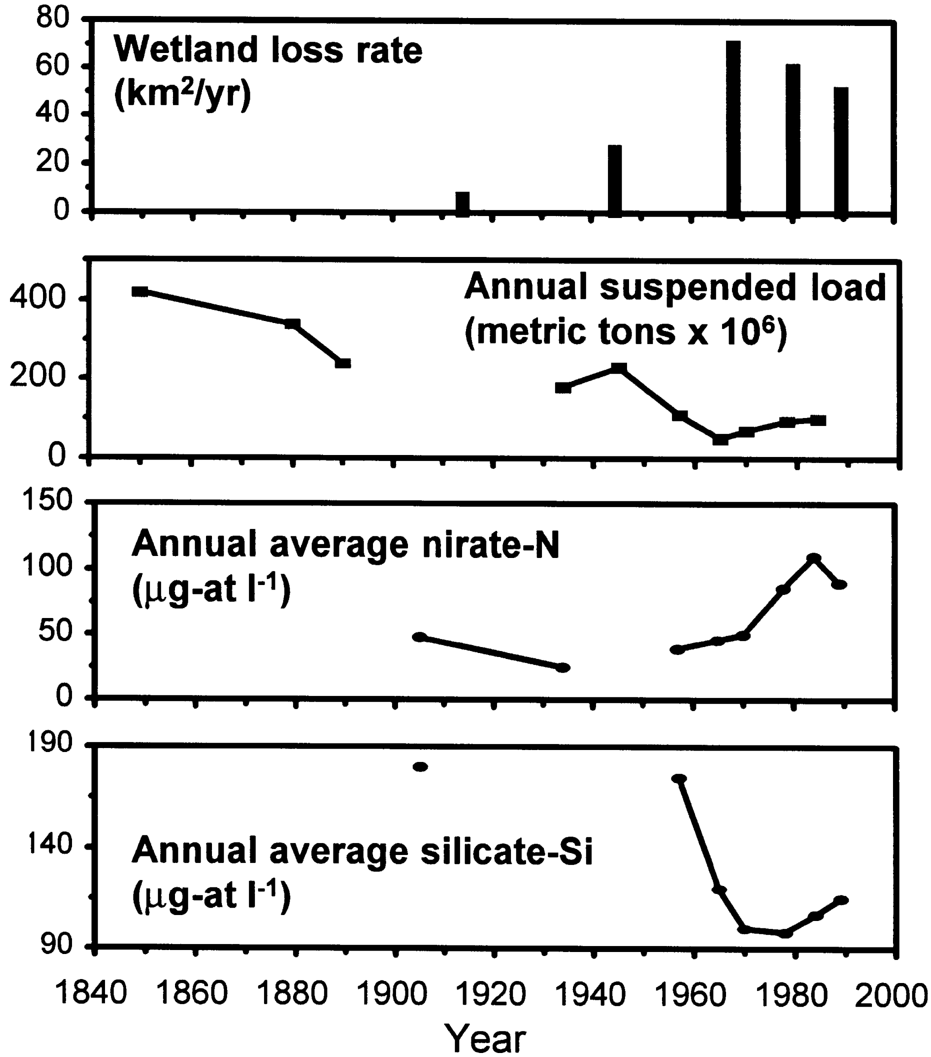

Fig. 3. Rapid changes in the Mississippi River Delta and its effluent have taken place in the latter half of the twentieth century. Coastal wetland loss rates are from data summarized by Boesch et al. (1994), suspended load estimates are from Kesel (1989), and average nitrate and silicate concentrations are taken from Turner & Rabalais (1994b).

term survival of wetlands in the deltaic plain must depend on the re-introduction of river flow into the interdistributary basins in order to stem salinity intrusion and supply sediments for wetland aggradation and progradation.

Significant changes in the composition of the Mississippi's effluent have also been well documented (Fig. 3). Average annual suspended sediment load in the lower river has declined by at least one-half since the late 19th century, ostensibly as a result of improved soil conservation practices and, particularly, the construction in the 1950s of dams which trap sediments (Kesel 1989). The concentration of dissolved nutrients has also changed from the 1950s; nitrate concentrations more than doubled and silicate concentrations have declined by 40% or more. Turner & Rabalais (1991) presented evidence suggesting that the increase in nitrate was coincident with the increase in use of chemical fertilizers in the U.S. In studies of nutrient concentrations throughout the length of the river, Antweiler et al. (1995) demonstrated that the source of most of this nitrate is the heavily agricultural, upper Mississippi basin over 1500 km upstream. Smith et al. (1987) suggested that increased atmospheric deposition of N in the industrialized Ohio River basin may also be a contributing factor; but Howarth et al. (1996) estimated that atmospheric deposition would account for no more than 25% of the anthropogenic N-loading of the Mississippi River system. The decline in concentrations of silicate are presumably related to reductions in suspended sediment loadings and to the biodeposition of removal of silicate by diatoms, the production of which has been enhanced by phosphorus enrichment and the construction of reservoirs and navigation pools which reduce turbidity and light limitation.

The consequences of the changes in nutrient delivery to continental shelf waters are incompletely known. Although hypoxic bottom waters reflective of eutrophication were occasionally reported in the literature, it was not until 1985 that the spatial and temporal extent of shelf hypoxia was surveyed. A region of up to 18200km ^{2} of the inner continental shelf east of the mouth of the Mississippi River and extending to the west from the Atchafalaya River occasionally to Texas, has been found to have bottom water oxygen concentrations too low (<2 mg/l) to sustain fishes and decapod crustaceans during the long summer season (Rabalais et al. 1991; N. Rabalais pers. comm.).

The key question of whether shelf hypoxia has spread or become more intense as a result of increased nitrate loading is difficult to answer because there were very few observations prior to 1985. In contrast to the wetlands of the Mississippi deltaic plain which have been studied intensively beginning in the 1960s, the continental shelf off Louisiana and Texas –an area that produces virtually all of the offshore oil and gas and a large portion of the fishery landings in the U.S.– remained a *mare incognito*. Nonetheless, analyses of sediment cores from the region of chronic summer hypoxia have shown changes in benthic foraminifera microfossils consistent with worsening hypoxia (Rabalais et al. 1996); an increase in the accumulation of biogenic silica (Turner & Rabalais 1994a); increased concentration of organic carbon; and C and N isotopic signatures (Eadie et al. 1994) consistent with increased productivity since the 1950s. In addition, the changed ratios of N, P and Si in the river discharge may have shifted the nutritional conditions for phytoplankton growth, perhaps favoring flagellates over diatoms (Turner & Rabalais 1994b; Rabalais et al. 1996).

Science has played an increasing role in environmental management of the wetlands of the Mississippi deltaic plain. The publication in 1981 of the first comprehensive measurements of land change rates stimulated much concern among resource managers, the general public and, eventually, political leaders. During the 1980s extensive research was conducted which greatly increased understanding of the causes of wetland loss. By 1990, the U.S. Congress had enacted the Coastal Wetlands, Planning, Protection and Restoration Act (CWPPRA) which established a process of planning and active restoration focused on Louisiana's coastal wetlands (Boesch et al. 1994).

Controversies among scientists and managers still rage on the effectiveness of various restoration and management techniques, including river diversions, barrier island restoration, and structural control of water levels. Planning and implementation of restoration projects within the inter-distributary basins are progressing with varying levels of involvement of the scientific community. An independent assessment of the process by a group of scientists emphasized the need for greater involvement of the scientific community in planning and monitoring and for a more holistic approach within the entire Mississippi deltaic plain as well as within each of its constituent interdistributary basins (Boesch et al. 1994). One factor that must be taken into account is the decline in suspended sediment loads, and thus basic building material, in the river as a result of dams put into place upstream.

On the other hand, policy-makers and managers have not yet developed any mitigative responses to the eutrophication of shelf waters, such as reduction of point and non-point sources as in the Chesapeake Bay. There are several potential reasons for this: the evidence for worsening eutrophication, although very strong, is new and not yet widely understood and accepted; the consequences of eutrophication to important resources, although potentially major, have not been well documented; hypoxia occurs offshore, out of view and without obvious massive fish kills; and the idea of trying to control nutrient discharges throughout the huge Mississippi watershed has been just too daunting for many to contemplate. In January 1995 a coalition of environment organizations petitioned the U.S. Environmental Protection Agency (EPA) to take action to control nutrient pollution of the Mississippi River under federal law which allows federal intervention when pollutants discharged on one state affect another downstream. So far the response of the EPA has been to initiate an assessment of the evidence, hold a workshop, and begin to discuss the problem with agricultural and other upstream interests.

There is an under-appreciated interaction between wetland restoration initiatives, particularly river diversions, and the eutrophication of coastal waters. Diffusing the rivers effluent more broadly over coastal marshes rather injecting it onto the continental shelf may effect some N removal but probably a small part of the total loading, particularly during high spring flow. On the other hand, introducing nutrient-rich river waters into an interdistributary bay or to the east of the river's mouth may stimulate excess phytoplankton growth in areas not now experiencing hypoxia. Large scale diversions, such as that proposed to divert flow from the birdsfoot delta to the east, could result in eutrophication of portions of the continental shelf not now experiencing hypoxia and affect stratification and buoyancy-driven circulation over a large scale.

Florida Bay

Florida Bay is a large (about 2200 km^2 in surface area), very shallow (average depth less than 1m) lagoon bordered on the north by the Florida mainland and on the south by the Florida Keys (McIvor et al. 1994). In contrast to the three other coastal ecosystems considered here it is tropical, frequently hypersaline, contains carbonate sediments and outcroppings, and has extensive seagrass meadows and mangroves.

Florida Bay has received relatively little attention and was thought to be little affected by human activities. Beginning in 1987, seagrass meadows, which had covered as much as 80 % of the bay's bottom began to die with the area of die-off as large as 18 % of the total area of the bay (Roblee et al. 1991; Boesch et al. 1993; McIvor et al. 1994). Blooms of cyanobacteria and other phytoplankton, which were first noticed as early as 1979, began to occur with increasing frequency and intensity, turning the once clear waters a turbid green. Populations of water birds, forage fish and juveniles of game fish seem to be reduced in the upper end of the bay, coincident with increased salinities. Many large sponges along the Florida Keys margin of the bay died, potentially threatening a significant decline in the catch of spiny lobsters, the juveniles of which use the sponges as critical habitats.

A number of scientists who were investigating these changes argued that most were related –one causing another– and have as a root cause changes in freshwater flow through the Everglades into Florida Bay (McIvor et al. 1994). A conceptual model of these cascading effects is presented in Fig. 4. Florida Bay lies at the distal end of the 28000 km^2 Kissimmee-Okeechobee-Everglades drainage basin. In order to reclaim wetlands for agriculture, furnish irrigation water, and provide flood protection for the sprawling population of the Miami region which encroaches on this watershed, an extensive series of canals has been dug which have the effect of moving water outside of this drainage basin to discharge sites along the Atlantic coast. As little as one-fifth of the historic surface water flow into the northeastern corner of the bay may now escape such diversion under the present water management regime. However, other scientists suggested that the changes may have been the manifestations of natural cycles, including the frequency of hurricanes, related to the filling in of the Florida Keys and occlusion of water exchange, or were caused by greater infusion of plant nutrients from the mainland watershed (Boesch et al. 1993).

In order to assess the evidence associated with the intense differences of opinion among scientists, which had become publicized by the news media, the Author was asked by the Assistant Secretary of the U.S. Department of the Interior to chair a panel of outside scientists. Although, our report was unable to resolve these scientific controversies, it did indicate that there was enough evidence that the deteriorating conditions in the northeastern part of the bay were related to increased salinities and that rediversion of freshwater flows was needed to restore this part of the bay. Further, the panel crafted a set of hypothesis-driven research, monitoring and modeling needs to address the questions critical to effective management. The structure of questions and scientific priorities developed by the panel are now being used as guidance of a greatly expanded ($7 million per year) research program coordinated by a Federal-state inter-agency committee. Although this scientific evaluation started with a more limited base of knowledge about this coastal ecosystem than in the case of the Chesapeake Bay, San Francisco Bay or Mississippi Delta, it now has the opportunity to resolve the unknowns through a more strategic and coordinated approach. It will be interesting to observe whether this

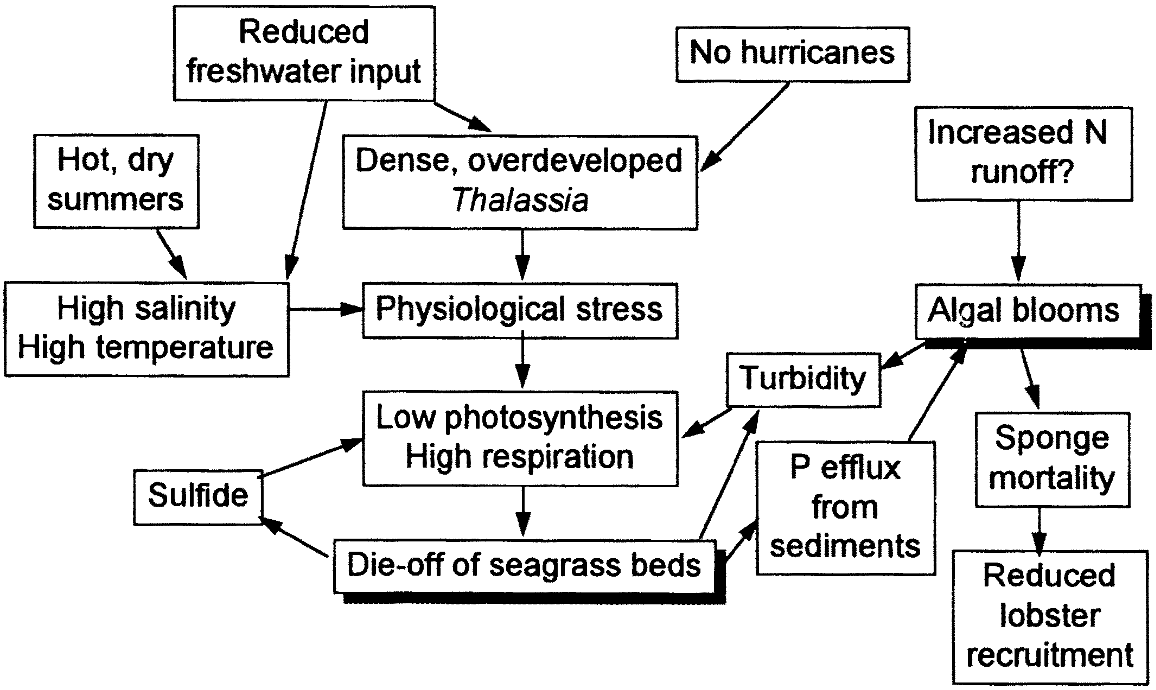

Fig. 4. Hypothetical model relating the potential causes of massive die-off of seagrasses and algal blooms in Florida Bay, based on McIvor et al. (1994) and Boesch et al. (1995).

speeds up the process by which effective management solutions are identified and implemented.

Discussion

Scale mismatches underlie the difficulties in recognizing, assigning causes to and effectively managing large-scale modifications of coastal ecosystems associated with changes in the delivery of fresh water, sediments and nutrients from the land. As Lee (1993) notes, "when human responsibility does not match the spatial, temporal or functional scale of natural phenomena, unsustainable use of resources is likely, and it will persist until the mismatch of scales is cured." In the case studies considered here, spatial scale mismatches occurred because of the lack of awareness of the consequences of the clearing of land, use of agricultural fertilizers, dam construction, or flood protection on the flux of materials into coastal ecosystems far removed from these activities. Functional scale mismatches occurred, for example, in the allocation of water among users when the interests of the estuary as a user was not considered or not considered important. Temporal scale mismatches occurred when the longer-term effects of actions were not understood or considered, for example the eventual unsustainability of Mississippi Delta wetlands deprived of periodic sediment subsidies from floods.

Both science and management have been challenged by the spatial, functional and temporal scale mismatches inherent in the watershed-coastal ecosystem relationship. Based on both the successes and problems identified in the four U.S. case studies considered here, five key factors seem to affect the degree and timeliness of influence of science on environmental policy and management:

1. *Sustained scientific investigation, responsive to but not totally defined by managers.* Scientific investigations and institutions were sustained over long periods in both the Chesapeake Bay and San Francisco Bay. In the Chesapeake this took place primarily in academic research institutions which were established and have been supported to provide scientific information to the states of Virginia (Virginia Institute of Marine Science) and Maryland (Center for Environmental and Estuarine Studies). In San Francisco Bay, despite the proximity to prestigious universities, this role was played largely by governmental agency scientists from the U.S. Geological Survey and the California Department of Fish and Game. In the Mississippi Delta, there had been sustained research on wetlands by university scientists, particularly through the Sea Grant Program, but little effort on the continental shelf until the late 1980s. In

contrast, Florida Bay has been the subject of little sustained research which could have detected and understood changes earlier. Now, there is new, intense scientific effort, but without the benefit of much corporate experience. It has also proved important that the research be responsive to the management issues, but not totally prescribed by environmental and resource managers who are more prone to support science based on present understanding than to invest in the potentially heretical investigation that have led to paradigm shifts.

2. *Clear evidence of change, the scale of the change and the causes of the change*. Because of the scale of changes and the confounding effects of other human activities and natural phenomena, it has frequently proved difficult to communicate in a clear and convincing way that the coastal ecosystem has indeed been affected by changes in the watershed. This is particularly so where, as is usually the case, historical data are lacking or sketchy. The Chesapeake Bay and Mississippi Delta examples provided here (Figs. 2 and 3) illustrate the power of historical analyses, but comparative ecosystem analyses must become increasingly applied to extend understanding from better studied coastal ecosystems to those potentially experiencing emerging problems (Anon. 1994). Also, more attention should be devoted to the process of making the connections necessary to promote the appropriate use of science in policy-making (Anon. 1995).

3. *Some level of consensus among the scientific communities associated with various interests.* The power of scientific consensus is illustrated by the Chesapeake Bay Scientific and Technical Committee report on nutrient controls and the recent San Francisco Bay workshop proposals for managing for optimal location of the 2 psu isohaline. High-level governmental agreements were concluded quickly after the articulation of each consensus. Surely management actions were being considered parallel with these processes, but broad consensus within the scientific community, including those scientists associated with different sectors such as fisheries, water supply, water quality, waste treatment and agriculture make difficult political decisions more acceptable or less risky. In a similar vein, Haas (1990) points out the critical importance of the international scientific community– an epistemic community in the terms of political scientists– in obtaining multinational agreement for the Mediterranean Action Plan. Tempering the reliance on consensus, however, is Walters' (1986) suggestion that adaptive management requires embracing alternatives rather than promoting consensus.

4. *The development of models to guide management actions*. Models are particularly important in making complex relationships understandable, defining management indicators, pointing to effective solutions, and assessing progress. These models may be process models or statistical models and range from descriptive to highly quantitative. They provide a means of articulating scientific information in a way that can be understood and used by managers. Of course, an oversimplified model or one based on false premises can be dangerous in the hands of these very same managers. For that reason, modeling must be a process which actively engages scientists as well as modelers and managers and must be closely coupled with monitoring as part of an adaptive management approach (Walters 1986).

5. *Identification of effective and feasible solutions to the problems*. In the cases studied here, management became engaged in responding to the problems only after effective and feasible solutions were identified (e.g. biological nutrient removal in the Chesapeake Bay, managed river diversions in the Mississippi Delta, and management for isohaline position in upper San Francisco Bay). Coastal environmental scientists are more oriented to uncovering problems than in identifying solutions, particularly when those solutions must be implemented far from the coast. Toward this end, much better communication and integration must take place among the scientific and engineering communities working on coastal ecosystems, watershed processes, agricultural practices, and waste treatment.

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