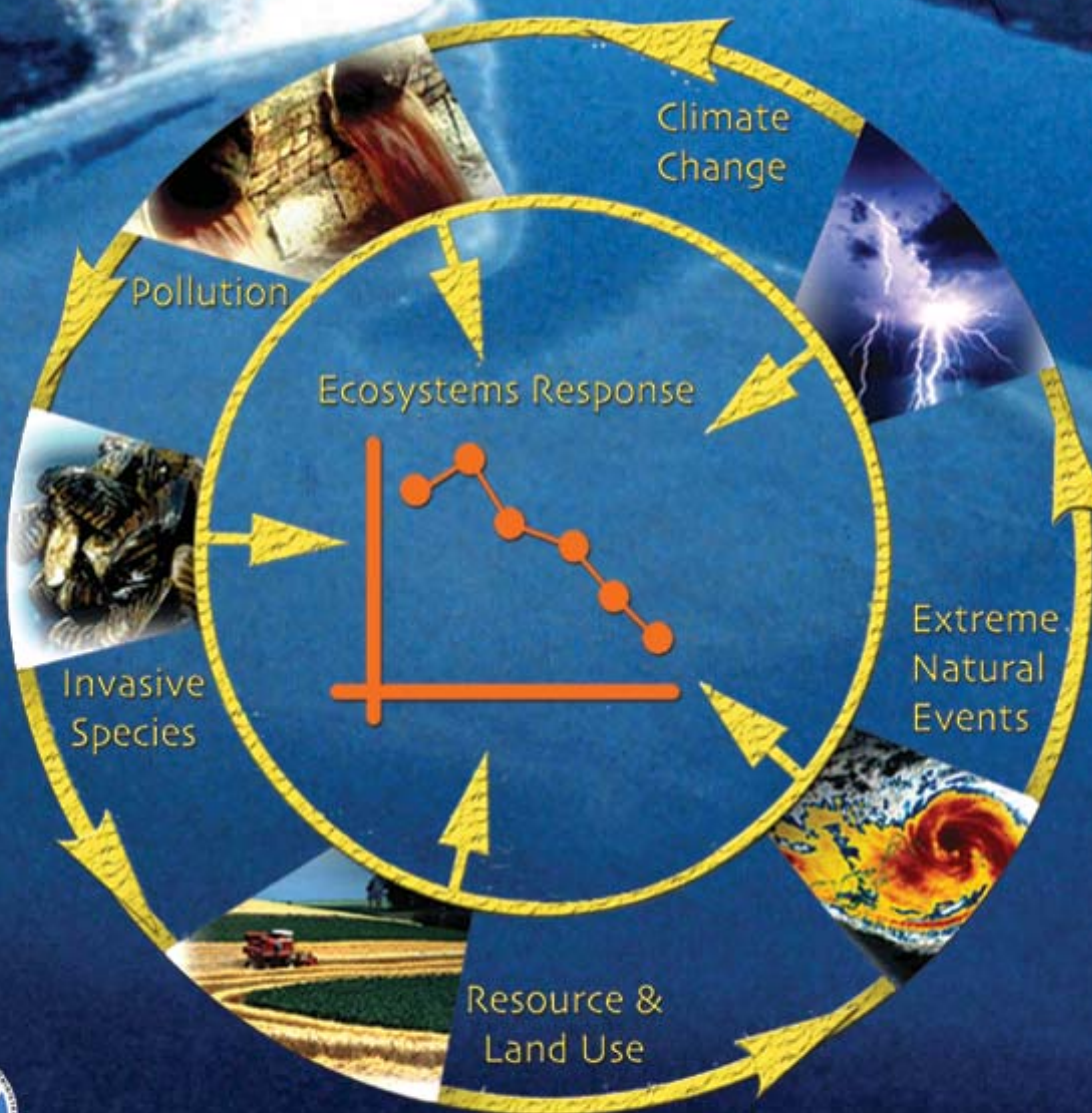


ECOLOGICAL FORECASTING:

New Tools for Coastal and Marine Ecosystem Management



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Nathalie Valette-Silver and Donald Scavia, Editors

**NOAA Technical Memorandum NOS NCCOS 1
July 2003**

**US Department of Commerce
Donald L. Evans
Secretary**

**National Oceanic and Atmospheric Administration
Conrad C. Lautenbacher, Jr.
Vice Admiral US Navy (Ret.)
Under Secretary of Commerce for Oceans and Atmosphere**

**National Ocean Service
Richard W. Spinrad, PhD
Assistant Administrator for NOAA National Ocean Service**

Citation for this Report

N.J. Valette-Silver and D. Scavia. 2003. Ecological Forecasting: New Tools for Coastal and Ecosystem Management. NOAA Technical Memorandum NOS NCCOS 1. 116 pp.

Introduction to Ecological Forecasting: New Tools for Coastal and Marine Ecosystem Management

Nathalie J. Valette-Silver and Donald Scavia¹

Abstract

Policy makers, natural resource managers, regulators, and the public often call on scientists to estimate the potential ecological changes caused by both natural and human-induced stresses, and to determine how those changes will impact people and the environment.

To develop accurate forecasts of ecological changes we need to: 1) increase understanding of ecosystem composition, structure, and functioning, 2) expand ecosystem monitoring and apply advanced scientific information to make these complex data widely available, and 3) develop and improve forecast and interpretative tools that use a scientific basis to assess the results of management and science policy actions.

Introduction

Ecological forecasts predict the impacts of physical, chemical, biological, and human-induced change on ecosystems and their components (CENR 2001).

Sustained ecosystem health and delivery of good and services are essential to the US economy. For example, each year U.S. ecosystems provide over \$227 billion in added value to the US economy (USDA 1999, CENR 2001). Ecosystems also provide other, harder to quantify, but critical benefits: air and water purification; waste detoxification and decomposition; pollination of crops and natural vegetation; maintenance of biological diversity; and recreational and spiritual renewal (Daily *et al.* 1997). Finally, ecosystem viability is important for national security because ecological damage, environmental degradation, and depletion of natural resources can easily give rise to political unrest and conflict among nations (NSTC 1995).

To help sustain ecosystem health, decision-makers must take into account effects of both natural and human-induced stressors on ecosystem structure, functioning, and productivity. A central role for science in the decision-making process is to provide forecasts of the magnitude, direction, and nature of those effects (Graedel 2001).

Ecological processes operate on a wide array of time and space scales. During the last decade, using techno-

logical and scientific innovations, scientists were able to develop and test ecological forecasts in ways that were not feasible only a few years ago (Clark *et al.* 2001). Within the next few decades, technological advances will bring even greater capabilities that are needed for ecosystem analysis at unprecedented scales: local, regional, and global.

Ecological forecasts also require a better understanding of how complex ecological communities form and evolve, including interactions among species, the physical environment, evolutionary history, and the ‘assembly rules’ by which ecosystems are formed. Many important ecological processes depend on the little-studied ‘hidden players’ such as microbes, fungi, and marine and soil invertebrates. Understanding interactions at very small scales will be made possible through recent and future developments of new micro- and nano-sensing technologies.

Ecological systems are strongly influenced by physical and biogeochemical processes and by historical events that introduce time lags and non-linearity, shape complex changes in community composition and functioning, and affect the capacity of ecosystems and communities to be restored once damaged. Unprecedented developments in our ability to sequence genomes and compute protein structures allow researchers to understand better ecosystem functioning. At the same time, understanding and observations of coupled physical-chemical-biological

¹ National Ocean Service, National Oceanic and Atmospheric Administration, Silver Spring, MD 20910

systems are producing larger scales, new theories and models of interactions within these complex systems. The integration of these genomic insights with ecological theory and larger-scale ecosystem dynamics, along with new computational and modeling technologies, holds the promise of revolutionizing our knowledge about how ecosystems respond to natural and human influences (CENR 2001).

The Benefits of Ecological Forecasting

Forecasting future conditions based on a series of potential scenarios is a new tool that allows for the evaluation of management options and associated tradeoffs. In addition to the forecasts, the data and information collected to develop and test them enrich scientific knowledge and improve the decision making process.

Ecological forecasts strengthen the link between research and management. When managers understand and can clearly articulate forecast needs, scientists can in turn better understand those needs. In particular, when scientists and managers work together to define, develop, interpret, and evaluate ecological forecasts, the researchers stay better informed about manager's needs and the managers help identify gaps in scientific knowledge.

Developing and evaluating forecasts can also help set science priorities. Among potential research, modeling, and monitoring efforts, it is possible to set priorities based on the need to create and increase the reliability of desired forecasts. For example, monitoring priorities should support and validate forecasts, modeling priorities should lead to quantitative forecasts, and research priorities should increase the information content or decrease forecast uncertainties.

Forecasting the Effects of Ecosystem Stresses

Extreme natural events, climate change, land and resource use, pollution, and invasive species are five key drivers of ecosystem change (CENR 2001) that interact across wide time and space scales (hours to decades and local to global) (Figure 1).

- **Extreme natural events** can cause significant ecosystem change. It is important for resource managers to forecast their effects on ecosystems as well as their interactions with other stressors. Forecasting the ecological disturbance from natural

Principal Scales of Stress

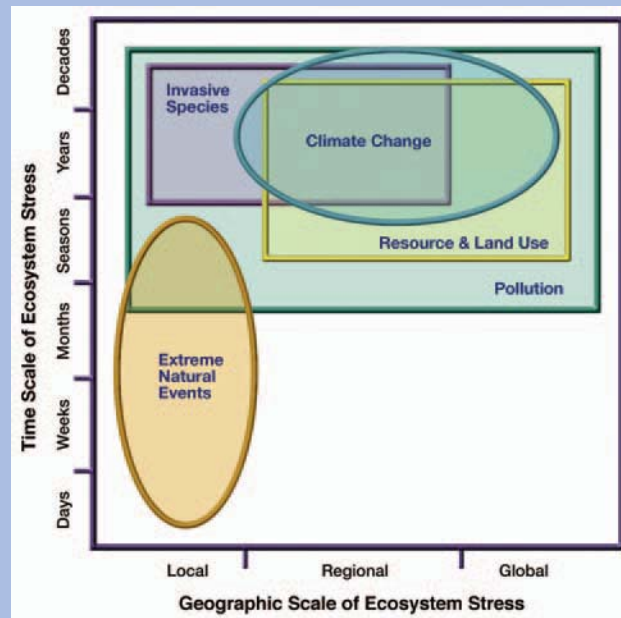


Figure 1. Time/Space Scale of Ecosystem Response. The five key ecosystem stressors - pollution, land and resource use, invasive species, extreme natural events, and climate change - can challenge the integrity of ecosystems and impede the delivery of their goods and services. These stressors can act alone or together, and their cumulative effects are poorly understood. Ecosystem responses are as varied as the inputs that strain them, playing out in scales from hours to decades and from local to global.

hazards such as wildfire, flood, drought, hurricanes, windstorms, and other similar events will allow resource managers and the public to anticipate, minimize and mitigate their impact.

- **Climate change and variability** are likely to have widespread and significant impacts on ecosystems. Resource managers and policy makers must account for these potential changes by developing coping and adaptation strategies to minimize impacts on ecosystems, species, and valued ecological resources.
- **Changes in land and resource use** (e.g., changes in marine resource harvests, conversion of land from forest to agriculture or from agriculture to urban) affect ecosystems ability to function properly. Forecasts of the ecosystem implications of these shifts, and their consequent impacts on environment and society, will help guide management and policy actions.

- **Pollution** and in particular harmful chemicals and excess nutrients remain one of the American public's top concerns. Demands for forecasts of pollution impacts present significant challenges. While the known and measurable point sources of pollution have been fairly well handled, the diffuse non-point sources are more difficult to measure and control. State and local governments as well as natural resource management agencies are seeing increased costs of meeting water and air quality standards and goals in addition to increased uncertainties in achieving desired ecological outcomes.
- The introduction and spread of **non-native species** impacts the abundance and the very existence of native species as well as the proper functioning of healthy ecosystems. The prevalence of this problem in most US ecosystems, and its integral link to domestic and international activities, make forecasts of alien species introduction and spread a critical and badly-needed management tool.

Most ecosystems are subjected to mixtures of these stresses. For example, a hurricane may open the door for new species invasions, and the success of that invader may be enhanced by altered climate, the extent to which the marine resources are used, and the extent of pollution in the environment being invaded. Building the capacity to forecast the cumulative effects of multiple stresses is one of ecology's most significant research challenges.

Some of the most severe and longest-lasting ecosystem effects result from chronic but subtle influences. Shorter-term, extreme events are also important drivers of both short-term and longer-term ecosystem change.

Next Steps

To make progress, an integrated science-management effort is needed to 1) understand the kind of forecasts desired, 2) adequately communicate the value and limits of forecasts, and 3) understand the iterative nature of forecasts in adaptive management. Already some scientists and modelers are providing rudimentary forecasts, but the efforts are neither well-organized nor coordinated. Presently, forecasts are still limited by gaps in understanding, data, and models. Building an ecological forecasting capability will require 1) integration across disciplines, 2) inclusion of the social and economic sciences; 3) development of ways to minimize forecast uncertainty; and 4) use of technological innovations.

Understanding complex ecosystems is one of the most significant forecasting challenges. Improvements in understanding ecosystem composition, structure, and functioning have occurred recently. However, there is still a clear research need in order to provide decision makers with reliable ecological forecasts at the larger time and space scales (National Academy of Sciences 2001. President's Committee of Advisors on Science and Technology 2000). Reliable ecological forecasts cannot be made or tested without accurate and timely information about historical and current ecosystem conditions and without models integrating ecological, hydrological, chemical and physical parameters.

Likewise, the efficacy of management actions, based on those forecasts, cannot be evaluated without monitoring ecosystem change. Rapid advances in remote and *in situ* sensing provide new opportunities to gather these data. Effective forecasts rely on synthesizing, analyzing, and interpreting complex information, ranging in scale from molecular to regional and from both the social and natural sciences.

This need for synthesis of large, widely distributed, diverse data sets, and the ability to analyze, model, and interpret them at varying scales, expands the boundaries of current computation and information science. New applications of geo-spatial and temporal data management, simulation, and visualization will require increased collaboration between computer and ecological scientists.

Both deterministic and statistical models are needed, and to be most effective, scientists and decision-makers must have confidence in them. Efforts are needed to define and evaluate the information content of models, which is controlled by uncertainty in model structure, parameters, and inputs. Advancements will require developing new modeling approaches, and formalized assessment of error propagation in both models and associated field data.

Finally, it must be understood that a forecast is an iterative process that must be reevaluate and improved along the way.

Conclusion

At present, rudimentary ecological forecasts are being made and used. Some examples are explored within this volume (Figure 2). Developing and improving forecasts provide a focus for the science-policy interface. They also help set science priorities, in particular what

to monitor, characterize and map, and what models are needed. Which natural processes need to be understood and what social and economic science is needed. Advancing ecological forecasting requires collaboration among scientists and managers at the Federal and state level, academia, non-governmental organizations, and the private sector.

Acknowledgements

We would like to thank all the scientists and managers who participated in the sessions organized in 2001 around the theme of Ecological Forecast at the meetings of the Coastal Zone 2001 (Cleveland, Ohio), Estuarine Research Federation ERF'01 (St. Petersburg, FL) and the Ecological Society of America ESA 2001 (Madison, WI). We also want to acknowledge the efforts of the members of the Committee on Environment and Natural Resources, Subcommittee on Ecological Systems who participated in the formulation of a 2001 CENR report on ecological forecasting (CENR 2001).

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Principal Scales of Stress

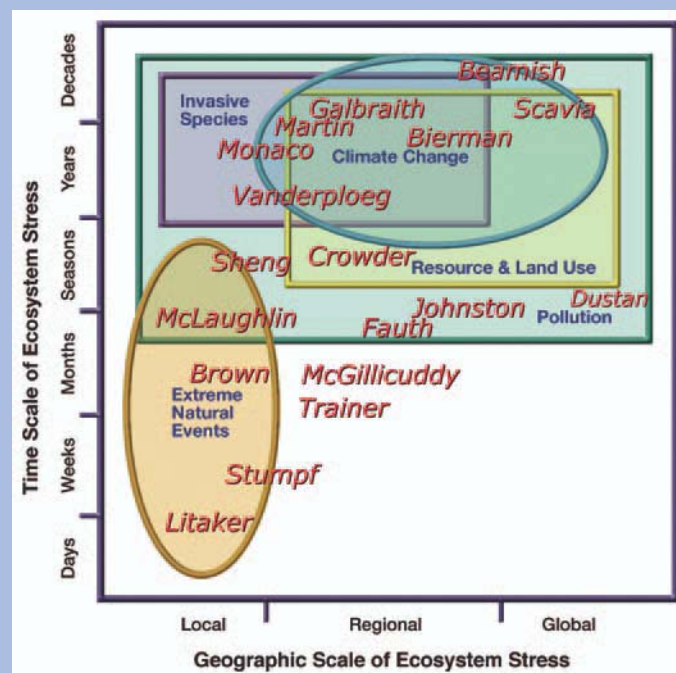


Figure 2. Time/Space Scale of Ecosystem Response. The name of the first author in each paper presented in this document is displayed on the time/space scale of ecosystem response diagram showing where their contribution lays. Their position on the diagram also identifies their contribution to a specific stressor.

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Mid-range Prediction of Coral Bleaching: A Molecular Diagnostic System Approach

John E. Fauth¹, Craig A. Downs², John C. Halas³, Phillip Dustan⁴, and Cheryl M. Woodley⁵

Abstract

One challenge facing ecological forecasters is making accurate, mid-range predictions of changes to ecosystem health. Accurately forecasting disease and degradation months in advance could allow resource managers time to minimize or ameliorate stressors that threaten ecosystem structure, function, and integrity.

*This research has resulted in a molecular biomarker system to assess the physiological status of corals by assaying specific parameters of cellular physiology indicative of individual colony health. Field samples of boulder corals (*Montastraea annularis*) from the Florida Keys, USA, showed a strong correlation between molecular biomarker levels and bleaching of individual coral colonies. Logistic regression identified molecular biomarkers capable of predicting the onset of coral bleaching 3-6 months before symptoms became apparent.*

Biomarkers useful in forecasting included glutathione (an antioxidant), lipid peroxides (which indicate functional lipid levels and membrane integrity) and chloroplast small heat shock proteins (chlpsHsp, which protect photosystem II activity during oxidative stress). Relationships observed between water depth, biomarker levels, and bleaching suggest deep-water corals cannot accumulate enough chlpsHsp to prevent Photosystem II from dissociating and generating the oxidative stress that causes bleaching.

Molecular biomarkers provide useful tools for ecological forecasting because they assay vital cellular and physiological parameters. Understanding the mechanistic basis of degradative processes should help scientists and resource managers identify and alleviate the environmental stresses that trigger them.

Introduction

Accurate ecological forecasts made months in advance can give resource managers time to minimize or ameliorate stresses that threaten ecosystems (Clark *et al.* 2001, National Biological Information Infrastructure 2001). Forecasting stress responses to intervene and prevent damage requires linking changes at lower levels of biological organization with the fitness of individuals, and accurately assessing the long-term, ecosystem-level consequences.

Stress often indirectly affects ecosystems by overwhelming defenses at lower levels of the biological hierarchy: molecular, cellular, and organism-level homeostatic processes. Stress then reduces individual fitness, alters demographic parameters, and diminishes the structure, function, and resilience of communities which comprise the biotic component of ecosystems. Changes in molecular and cellular parameters may precede ecosystem-level responses to chronic stress by days, months, or years.

¹ Department of Biology, College of Charleston, 66 George Street, Charleston, SC 29424; e-mail fauthj@cofc.edu. Current address: Department of Biology, University of Central Florida, Orlando, FL 32816

² EnVirtue Biotechnologies, Inc., 35 W. Piccadilly St., Winchester, VA 226015

³ Florida Keys National Marine Sanctuary, U.S. National Oceanic and Atmospheric Administration, P.O. Box 1083, Key Largo, FL 33037

⁴ Department of Biology, College of Charleston, 66 George Street, Charleston, SC 29424

⁵ US National Oceanic and Atmospheric Administration, Hollings Marine Laboratory, 331 Ft. Johnson Rd., Charleston, SC 29412

A molecular diagnostic system (MDS) has been developed to monitor cellular stress responses and to develop forecasting capabilities (Downs *et al.* 2000, Downs, Fauth & Woodley 2001, Downs, Dillon & coauthors 2001, Woodley *et al.* 2001, Downs, Shigenaka & coauthors 2002, Downs, Fauth & coauthors 2002). The MDS assays specific cellular and molecular parameters indicative of cellular physiological status, including whether 1) cell structural integrity was compromised, 2) cells experienced an oxidative stress, and 3) defenses were mounted to a particular stress (i.e., heat, pesticide, heavy metal).

Using molecular biomarkers to assess organism or ecosystem health is a popular idea with considerable evidence of its effectiveness (reviewed by Feder & Hoffman 1999). However, most studies examine only a single or several physiological parameters to gauge overall response to acute and chronic stress. Surprisingly few studies evaluate overall physiological status with multiple, specific molecular biomarkers (Schaeffer *et al.* 1988, Stegeman *et al.* 1992, Joannis & Storey 1996a,b, Downs *et al.* 2000, Downs, Fauth & Woodley 2001, Downs, Dillon & coauthors 2001, Woodley *et al.* 2001, Downs, Shigenaka & coauthors 2002, Downs, Fauth & coauthors 2002). The goal is to develop a comprehensive molecular diagnostic system to assess ecosystem health, providing resource managers timely information to better understand, monitor, and manage at-risk ecosystems.

While in theory the MDS can be applied to any living organism, the first forecast to be developed was for bleaching of reef-building corals in the Florida Keys National Marine Sanctuary (FKNMS), USA. The forecast arose from a research partnership between the Center for Coastal Environmental Health and Biomolecular Research (US National Oceanic and Atmospheric Administration/National Ocean Service), the FKNMS, EnVirtue Biotechnologies, Inc., and academic scientists. There were three objectives: 1) to identify probable mechanisms of coral bleaching, 2) to identify factors that influenced susceptibility of individual coral colonies to bleaching, and 3) to develop a method of forecasting bleaching of individual coral colonies. Once tested and verified, forecasts could be used by National Marine Sanctuary managers to identify vulnerable coral reefs, to assess the relative importance of different stresses on coral bleaching, and to set quantitative recovery endpoints for restoring reefs degraded by chronic stress.

Case Study

During bleaching, corals lose their symbiotic algae and/or their photosynthetic pigments. Bleaching robs corals of nutrients, slows growth (Hughes and Tanner 2000) and recovery from injuries (Mascarelli and Bunkley-Williams 1999), increases susceptibility to diseases (Kushmaro *et al.* 1996), delays reproduction (Hughes and Tanner 2000, Omori *et al.* 2001), and can kill entire colonies (Carriquiry *et al.* 2001).

Bleaching is caused by diverse stresses (reviewed in Williams and Bunkley-Williams 1990, Brown 1997a,b). However, heat stress is the primary trigger of mass bleaching events that have degraded coral reefs worldwide over the past 20 years (Williams and Bunkley-Williams 1990, Goreau *et al.* 1992, Goreau and Hayes 1993, Brown *et al.* 1996, Brown 1997a,b). Recently, Stone and co-workers (Huppert and Stone 1998, Stone *et al.* 1999) showed episodes of mass coral bleaching were associated with increased severity and frequency of El Niño events, which are symptomatic of global warming. Increased ocean temperatures compounded with other agents of coral degradation (e.g., pollution, disease, predation) pose an increasing and significant threat to the health and vitality of coral reef ecosystems worldwide (Dustan 1999).

The US National Oceanic and Atmospheric Administration currently uses remote sensing of unusually warm surface waters to identify areas where bleaching may occur (Carriquiry *et al.* 2001, NOAA National Environmental Satellite, Data, and Information Service 2002). However, even during extreme sea surface temperature anomalies, phylogenetic, spatial, temporal, bathymetric, and size-related variation in coral bleaching and mortality (Williams and Bunkley-Williams 1990, Hughes and Tanner 2000, Hueerkamp *et al.* 2001, Loya *et al.* 2001) can obscure both the intensity of bleaching events and their underlying causes. This is the situation for corals along the Florida Keys, which are subjected to different stressors from Key Largo to the Dry Tortugas (Dustan 1999, Porter *et al.* 1999).

A series of recent papers (Downs *et al.* 2000, Woodley *et al.* 2001, Downs, Fauth & coauthors 2002) demonstrated how heat stress affects coral MDS parameters under both laboratory and field conditions. In the laboratory, corals (*Montastraea faveolata*) heat-stressed under normal photosynthetically active radiation (PAR) produced more chloroplast small heat

shock protein (chlpHsp) than either the controls or corals heat-stressed in the dark (Downs *et al.* 2000, Woodley *et al.* 2001). Chloroplast small heat shock protein protects Photosystem II activity during oxidative stress (Downs *et al.* 2000, Downs, Fauth & coauthors 2002); when this defense is overwhelmed, oxyradicals are produced. In cnidarians, these reactive oxygen species cause lipid peroxidation, protein degradation, and DNA damage, and ultimately tissue and cell death (Hawkrige *et al.* 2000).

As expected, corals heat-stressed under PAR had significantly higher levels of lipid peroxide than those heat-stressed in the dark, indicating greater damage to lipid membranes. A similar pattern was seen in ubiquitin levels, which signals protein degradation (Downs *et al.* 2000). Superoxide dismutases (SOD) such as Cu/Zn and Mn SOD defend against oxidative damage by scavenging oxyradicals. Levels of Mn SOD and especially Cu/Zn SOD were significantly higher in corals heat stressed under PAR than in the dark, further evidence that elevated water temperatures caused oxidative stress (Downs *et al.* 2000, Woodley *et al.* 2001, Downs, Fauth & coauthors 2002). Independent researchers also concluded that heat stress causes oxidative stress, and leads to coral bleaching (Jones *et al.* 1998).

In 1999, coral colonies (*Montastraea annularis* species complex) sampled off Key Largo, FL, were exposed to elevated ocean temperatures for several months (Figure 1). There were strong positive correlations between oxidative damage and subsequent bleaching during the March-October sampling period (Woodley *et al.* 2001, Downs, Fauth & coauthors 2002). Corals with high levels of chlpHsp and SODs – typically colonies in shallower water – had less oxidative damage and did not bleach.

In contrast, corals with low chlpHsp and SODs – mostly those colonies in deeper water – experienced oxidative stress, had higher chaperonin levels (Hsp 60 and Hsp 70) and protein turnover activity, and bleached in late summer. Thus, bleaching was tightly coupled to the anti-oxidant and cellular stress capacity of coral colonies, suggesting bleaching was a final defense against oxidative stress (Downs *et al.* 2000, Woodley *et al.* 2001, Downs, Fauth & coauthors 2002).

This forecast exploits the time lag between temperature-induced cellular responses of corals (including upregulation of chlpHsp and antioxidant enzymes) and accumulation of damage products and subsequent

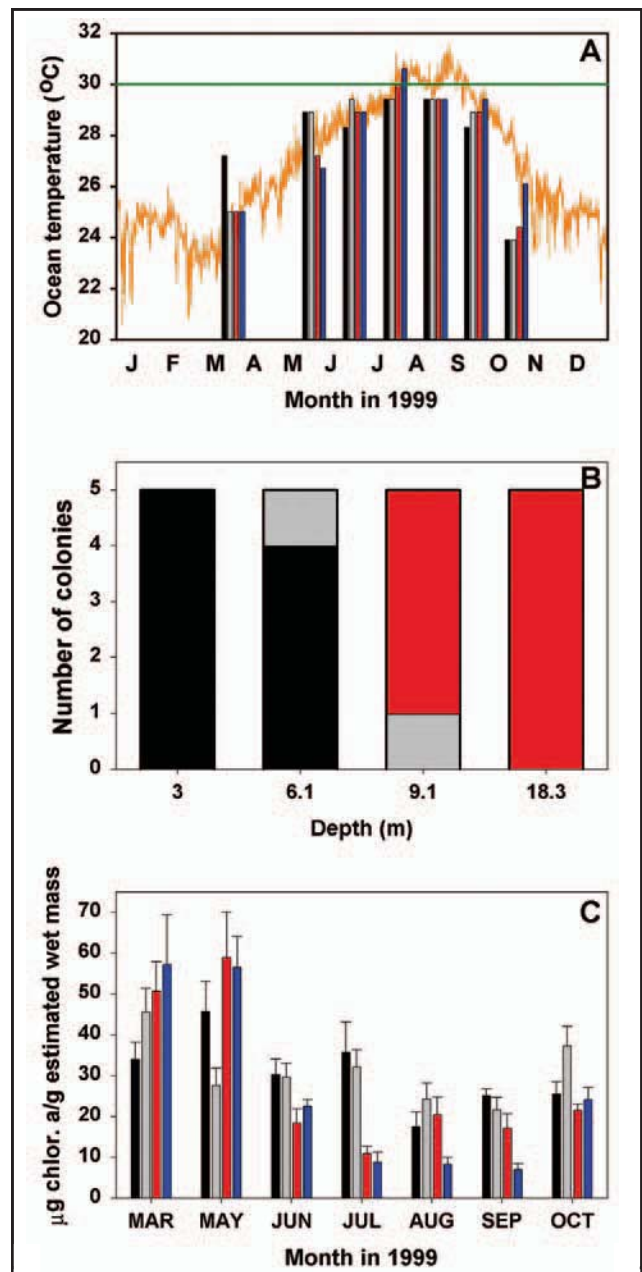


Figure 1. (A) Seasonal variation in ocean temperatures recorded beside coral colonies (*Montastraea annularis* species complex) sampled near Molasses Reef, off Key Largo, FL, USA. Bars show depth of colonies: black = 3.0 m, grey = 6.1 m, red = 9.1 m, blue = 18.3 m. Temperature beside the corals in summer differed significantly from all other months except May. Yellow line is mean sea surface temperature data recorded by a C-Man Buoy at Molasses Reef, 1-3 miles from our sites. Green line indicates the 30°C temperature threshold at which bleaching often occurs. (B) Colony condition in September, 1999 varied significantly among depths. Healthy colonies, black bars; paling colonies, gray bars; bleached colonies, red bars. (C) Chlorophyll a levels reflect photosynthetic activity and density of symbiotic zooxanthellae inhabiting coral tissue, providing a quantitative measure of bleaching. Bars show mean (± 1 SE) concentration of chlorophyll a in five coral colonies repeatedly sampled at each depth. Differences among months and the month \times depth interaction were statistically significant (Woodley *et al.* 2001).

bleaching. Using logistic regression, the relationship between MDS parameters in spring and early summer and coral condition in September was examined. One biomarker sampled in March (chlpsHsp) explained significant variation in bleaching six months later, when bleaching became visible to SCUBA divers (Figure 2). The variation in bleaching explained by chlpsHsp levels (35 percent) was greater than that explained in an identical analysis using March ocean temperature (27 percent) as the independent variable. In both analyses, the proportion of variation explained by logistic regression was determined by the coefficient of determination (R^2).

Ocean temperatures continued rising through late spring. Corals in deeper water did not produce enough chlpsHsp to prevent oxyradicals from damaging cell components. Therefore, colonies in deeper water accumulated significantly more lipid peroxide (LPO) than colonies in shallower water (Woodley *et al.* 2001, Downs, Fauth & coauthors 2002).

May LPO levels also were significant predictors of bleaching (Logistic regression: $X^2 = 28.9$, $P < 0.0001$, 2 df, $R^2 = 0.76$). Colonies with <30 micromoles LPO/mg total protein in May were almost certain to be healthy in September, while those with greater damage were likely to pale or bleach.

As water temperatures continued increasing, the superoxide dismutases assayed – those produced only by the coral and not by their algal symbionts (Downs *et al.* 2000, Woodley *et al.* 2001, Downs, Fauth & coauthors 2002) – also could be used to forecast coral condition. June levels of Cu/Zn SOD, which localizes to the cytosol, explained variation in coral condition more than Mn SOD, which was restricted to the mitochondria (34 percent versus 19 percent, respectively). Colonies with high levels of these antioxidant enzymes were significantly less likely to pale or bleach than those with low antioxidant levels (Logistic regressions: both $X^2 = 7.33$, $P < 0.03$, 2 df). Thus, three separate biomarkers could be used to forecast coral bleaching, anywhere from three to six months in advance of sea surface temperature anomalies.

This forecasting method is in the early stages of development and testing. Eventually, forecasts could be used like hurricane forecasts – to alert resource managers and reef-based industries to the likelihood of a mass bleaching event in the upcoming year.

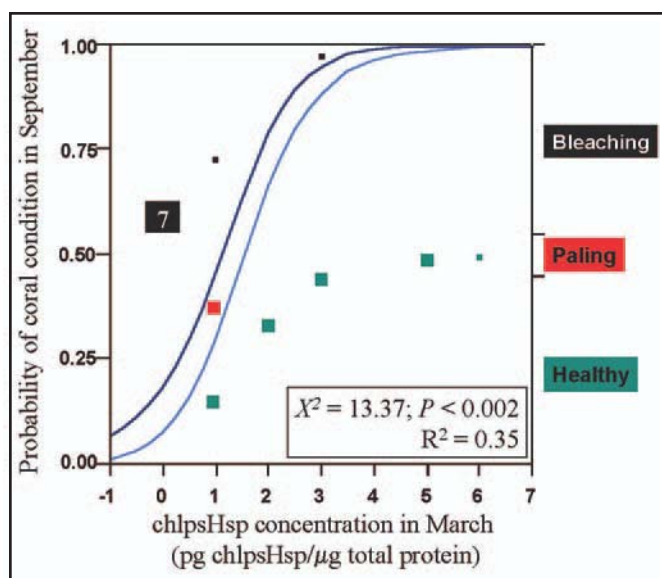


Figure 2. Cumulative logistic plot of the probability a coral colony was healthy (green), paling (red), or bleaching (black) in September, as a function of chloroplast small heat shock protein levels in March. Symbol size is proportional to the number of colonies with the same values. Smooth blue curves partition the probability axis among the three conditions, with healthy at lower right, bleaching at upper left, and paling in between. The probability of a colony having a specified condition when sea surface temperatures exceed 30°C for a prolonged period is given by the vertical distance between its probability curves. For example, the probability of a coral with 1 pg chlpsHsp in March being healthy six months later is 30 percent; of paling, 15 percent, and of bleaching, 55 percent.

Because the MDS allows identification of potential stressors, it also can be used to differentiate between global/regional and local forcing functions (i.e., climate warming and El Niño events versus point source pollution). Once sufficient data is obtained, it should be possible to forecast the effectiveness of different management options on coral bleaching.

Lessons Learned and Next Steps

The National Action Plan to Conserve Coral Reefs calls for a science-based ecosystem approach to coral reef conservation, and adaptive management to track and respond to environmental change and other anthropogenic threats (US Coral Reef Task Force 2000). This project includes close cooperation between a US NOAA/NOS biotechnology laboratory, US NOAA National Marine Sanctuaries managers, a private biotechnology company, and faculty from several colleges and universities.

The project has developed from an initial test of a promising new technology to a broader, more management-based initiative. Recently a wider geographic region was sampled, including two more sites off Key

Largo and within Biscayne National Park. The expanded project includes additional sentinel species in the coral reef community: an alga (*Halimeda opuntia*), corallivorous snail (*Coralliophila abbreviata*), bicolor damselfish (*Pomacentrus partitus*), and white grunts (*Haemulon plumieri*).

Infrastructure is in place to transfer forecasts of coral bleaching from the research and development stage to the operational stage. EnVirtue Biotechnologies, Inc. created monospecific polyclonal antibodies against the coral cellular parameters discussed above, which substantially enhanced MDS resolution and precision. The new antibodies also require less coral tissue, minimizing damage to stressed colonies. EnVirtue Biotechnologies, Inc., also developed a robotic, high-throughput system which greatly reduces the time required for multiple assays on a large number of samples. Currently, acquiring samples using SCUBA divers and grinding the samples under liquid nitrogen in preparation for MDS assays are the most time-consuming steps.

While the forecasting model used here was simplistic, it improves on efforts based solely on sea surface temperature, which cannot predict small-scale or bathymetric variation in bleaching. Verifying the accuracy of MDS-based forecasts requires obtaining more samples from diverse areas, and from additional species of corals, an effort already underway in the Florida Keys. The forecast also can be improved using multivariate statistical techniques, such as discriminant function and neural network analyses.

The most difficult steps remaining are 1) linking MDS parameters to other important life history traits of coral, such as growth, reproduction, survivorship, and susceptibility to disease, and 2) identifying the stressors impacting corals at different sites. Overcoming these hurdles will require vertically integrated sampling and experimentation, similar to what was successfully used to investigate declining amphibian populations (Collins *et al.* 2001).

Conclusions

There is an inverse relationship between expression of protective antioxidant/stress proteins and the incidence of oxidative damage and bleaching in boulder corals sampled within the Florida Keys National Marine Sanctuary, USA. Corals with high levels of antioxidants and heat-shock proteins, which protect against the damaging effects of unusually high ocean temperatures,

did not bleach. Those with low levels of protectant molecules suffered more oxidative damage, and were more likely to bleach (lose/expel their symbiotic algae). This correlation is similar to that observed in several human diseases (i.e., Alzheimer's disease, Parkinson's disease), where elevated levels of antioxidant and stress proteins protect cells from the degenerative influences of oxidative stress.

A simple regression procedure shows bleaching of individual coral colonies could be forecast months in advance from knowledge of specific biomarker levels. More sophisticated forecasts can be made using multiple markers and multivariate statistical techniques, but these will require larger data sets to test the accuracy of predictions than our logistic regression method.

Acknowledgements

Coral samples were collected under Florida Keys National Marine Sanctuaries and National Marine Fisheries Service permits. Statistical analyses were supported by NSF Grant DEB9727039 and EPA Grant R825795 to J. E. Fauth.

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The Importance of Incorporating Information on Climate-Related Marine Ecosystem Changes into Fisheries Management

R.J. Beamish, C.M. Neville and R.M Sweeting¹

Abstract

*The Strait of Georgia is a semi-enclosed marine ecosystem on the West Coast of Canada. The majority of the population of British Columbia is concentrated around the Strait, making it an important area for recreation and commerce. It is also the early marine rearing area for Pacific salmon (*Oncorhynchus* sp.) that ultimately account for up to 40 percent of Canada's total Pacific salmon catch.*

There were climate-related shifts in the dynamics of the Strait of Georgia after 1977, 1989, and 1998. The change after 1989 resulted in a reduced capacity to produce coho. The change after 1998 doubled the productivity of the ecosystem. In 2000, following the 1998 regime shift, there was a dramatic increase in primary production as indicated by the doubling of euphausiids and the modelled increase of phytoplankton. A comparison of the modelled biomasses of key species before and after the change in 2000 indicated the trophic structure of the Strait of Georgia will continue to adjust to the new, increased productivity. Future regime shifts will occur and the natural fluctuations may be affected in frequency and amplitude by greenhouse gas induced climate change.

Introduction

Many stock assessments require assumptions that the biological and physical relationships between a particular stock and their ecosystems have less of an impact on the population dynamics of the stock than fishing. Consequently studies to understand how marine ecosystems function frequently are viewed as being less important than the requirement to determine acceptable catch levels of the commercially-important species within the ecosystems. Recent evidence that climate has profound impacts on fish populations (Beamish *et al.* 1998, Finney *et al.* 2000) is a clear indication that ecosystem dynamics are an essential component of stock assessment. Forecasting the consequences of climate related changes in marine ecosystems on fish populations will become more important because this is a period of significant environmental change (IPCC 2001a,b).

Case Study

This study shows the Strait of Georgia ecosystem (Figure 1) is profoundly affected by decadal-scale changes in climate. This discovery is important because the Strait of Georgia may be the most important marine ecosystem on Canada's west coast. It is the early marine rearing area for Pacific salmon that traditionally accounted for about 40 percent of the total Canadian catch. In the 1970s, an assumption that marine ecosystems were not limiting Canadian Pacific salmon abundance led to a belief that the wild salmon production at the time could be doubled to historic levels by artificially rearing more juveniles in fresh water (Fisheries and Environment Canada 1978). Catches did reach historic high levels in the mid-1980s (Figure 2), although the enhanced portion of the catch was only about 13 percent (Beamish and Noakes 2003). However, by the late 1990s, catches were at the lowest levels in history even though the

¹ Fisheries and Oceans Canada, Pacific Biological Station, Nanaimo, B.C. V9T 6N7

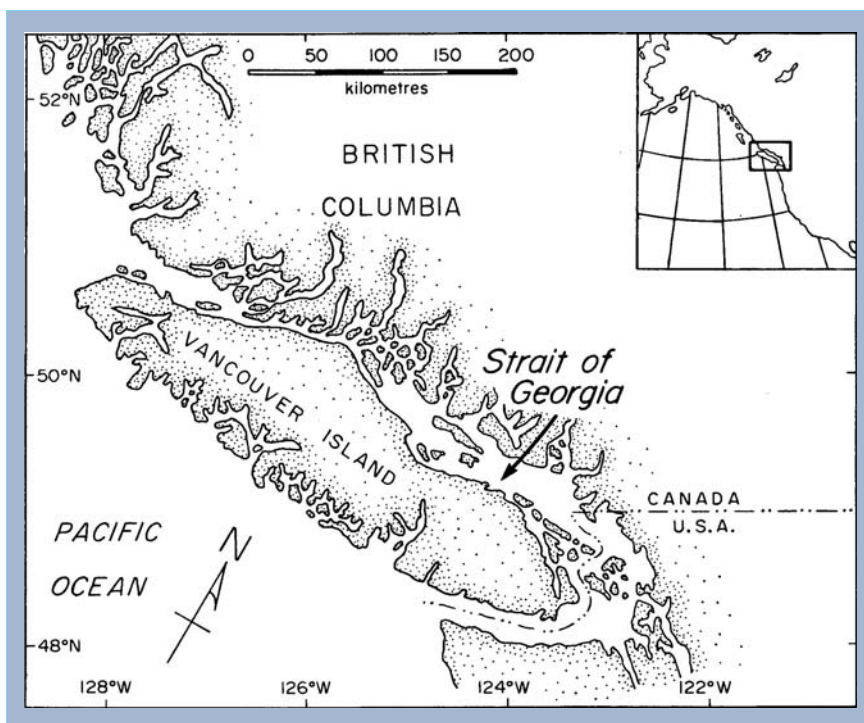


Figure 1. The Strait of Georgia is located between Vancouver Island and the mainland of British Columbia on the west coast of Canada.

releases of artificially reared salmon juveniles remained about constant.

We now know that it was incorrect for the enhancement program to assume that most of the natural losses regulating salmon abundance occurred in fresh water. Not only is ocean mortality important, but also, mortality trends shift quickly in response to climate. This paper summarizes the evidence that ecosystem variability in the Strait of Georgia is an essential component of fisheries management.

Regimes and the Strait of Georgia – The oceanography of the Strait of Georgia is strongly influenced by the flows of the Fraser River (Thomson 1981). As freshwater flows out from the Fraser River into the surface of the Strait and eventually into the open Pacific Ocean, denser, nutrient rich bottom water flows into the Strait of Georgia. In the spring, the combination of temperature, winds, sunlight, and nutrients initiate the production cycle that regulates the relative abundances of plants and animals in the ecosystem.

Interannual changes in climate can affect ecosystem productivity (Parsons *et al.* 1981, Gargett 1997). However, recent studies indicate that the dynamics of the Strait of Georgia are influenced by persistent states of climate and ocean regimes that shift abruptly (Beamish and Neville 1999). It

is not known how regime shifts alter the long-term dynamics of the Strait of Georgia naturally. This is particularly worry-some because we also do not know how global warming will affect the natural occurrence of these regimes. Forecasting the changes to ecosystems, therefore, requires an understanding of the consequences of regime shifts, as well as the trophic relationships within a regime.

In recent years, large-scale regime shifts have occurred in 1977, 1989, and probably in 1998 (Hare and Mantua 2000, McFarlane *et al.* 2000). In the Strait of Georgia, the impact of these shifts is evident in the temperatures of the surface, at 10 m, in bottom waters, and in the annual trend in Fraser River flow (Figure 3).

Regime changes are not cycles in which parameters oscillate back and forth. Regimes represent new ecosystem organizations both physically and biologically. Consequently, the regime after 1977 was characterized by reduced discharge from the Fraser River and increasing water temperatures. After 1989 there was not a major change in the annual pattern of Fraser River flows, but spring freshets were earlier, ocean temperatures continued to increase, and the dominant winter wind direction changed from winds predominately from the SW and SE to winds predominately from the SE. The changes in wind direction were associated with increases in sea levels and lower surface salinities. The

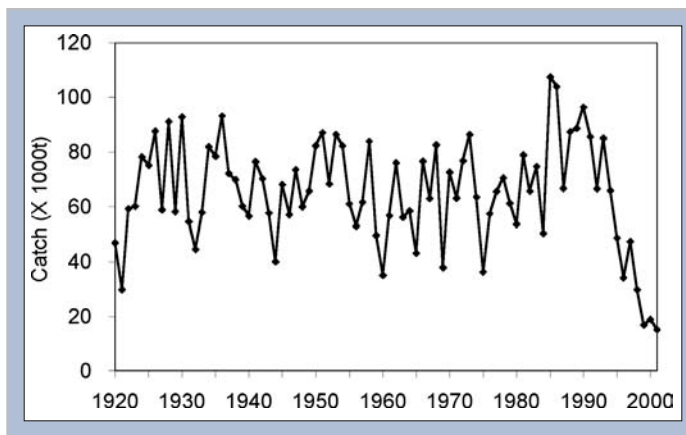


Figure 2. The total Canadian catch of all species of Pacific salmon. The estimate for 2001 is preliminary.

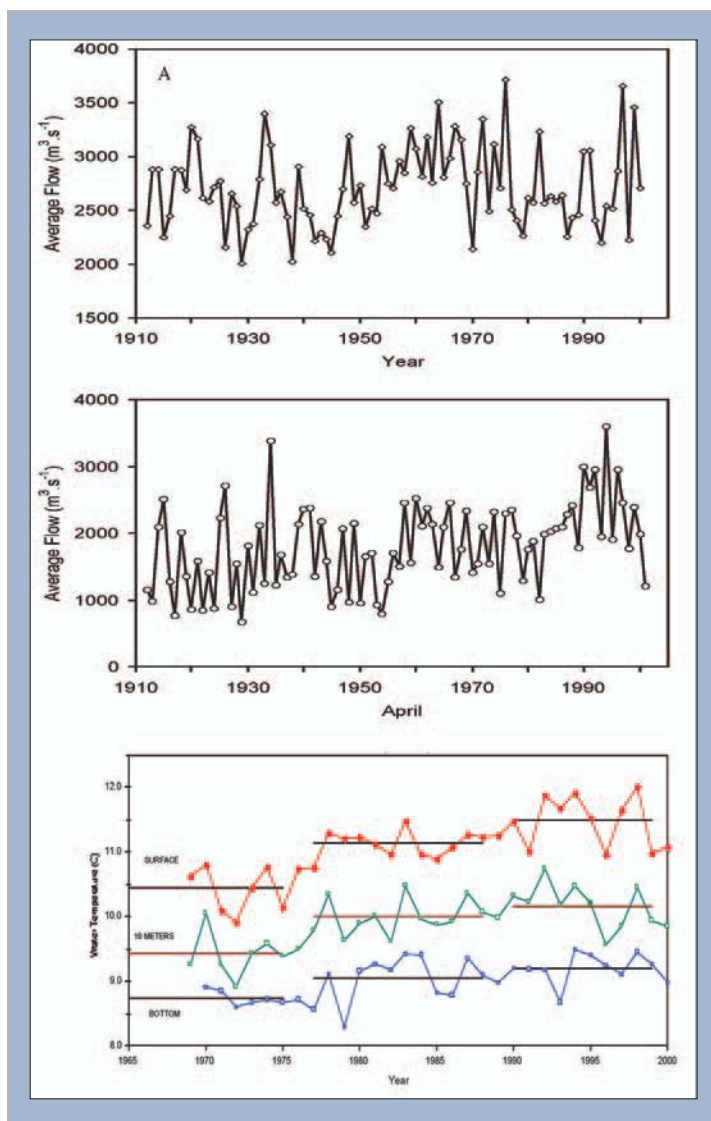


Figure 3. (A) Average annual flow (April to March) of the Fraser River from 1912 to 2000. The extreme variation in flow from 1996 to 2000 is unique to the time series. (B) Average daily flow of the Fraser River in April from 1912 to 2001. Increased average April flows in the 1990s indicate an earlier spring freshet. April flows decreased in 2001. (C) The average annual water temperature at surface, 10m and bottom (>200m) at Nanoose Bay in the Strait of Georgia.

changes in the ecosystem after 1998 remain to be determined, but appear to be characterized by extreme variation in total annual Fraser River discharge, lower temperature, and reduced storminess in the winter with a more even distribution of winds from the SE and SW.

The biological changes associated with regimes and regime shifts profoundly affected the dynamics of salmon species in the Strait, as well as the economics of local communities. After the 1977 change, local stocks of chinook salmon had reduced marine survivals (Beamish *et al.* 1998), whereas Pacific herring and hake stocks increased in abundance (McFarlane *et al.*

2000, Beamish *et al.* 2001a). The decline in the marine survival of chinook was one of the early indications that there was an ocean carrying capacity that limited the productivity of Pacific salmon, and that the bottleneck to salmon production was not only the number of juvenile salmon that entered the ocean.

The 1989 regime shift was associated with a collapse of the recreational fishery in the Strait of Georgia. Coho marine survival declined from approximately 15 percent in the 1970s and 1980s to less than 2 percent in the 1990s (Beamish *et al.* 2001a). The decline in marine survival occurred synchronously throughout the southern coho stocks and was associated with a change in migration behavior. In 1991, and from 1994 through to 2000, virtually all coho left the Strait of Georgia and did not return during the sports fishing season, collapsing one of the most important sports fisheries in Canada.

In 2000, following the 1998 shift, there was a dramatic increase in production in the Strait of Georgia as indicated by a large increase in euphausiid biomass (Beamish *et al.* 2002). Significantly increased early marine survival was evident for juvenile coho, chum, and pink salmon. In addition, individual size of juvenile coho, chum, and pink was larger than previous years. The individual size of euphausiids was also significantly larger in 2000 compared to 1999.

Ecopath was used as a trophic accounting model (Christensen and Pauly 1992). The model was balanced for 32 functional groups for the Strait of Georgia ecosystem in 1998 (Figure 4). Euphausiids are a major prey species that in 1998 had an ecotrophic efficiency (the biomass consumed by predators or transported out of the ecosystem) of 0.76. After the regime change in 2000 a new model that increased the consumption on the euphausiids by increasing the biomass of the consumers of euphausiids was required. The biomass increases included shellfish (50 percent), small pelagics (270 percent), miscellaneous demersal fish (250 percent), salmon (100 percent), hake (40 percent) and herring (40 percent) (Beamish *et al.* 2001b). After increasing the consumption of euphausiids, the ecotrophic efficiency was 0.58. The low ecotrophic efficiency of euphausi-

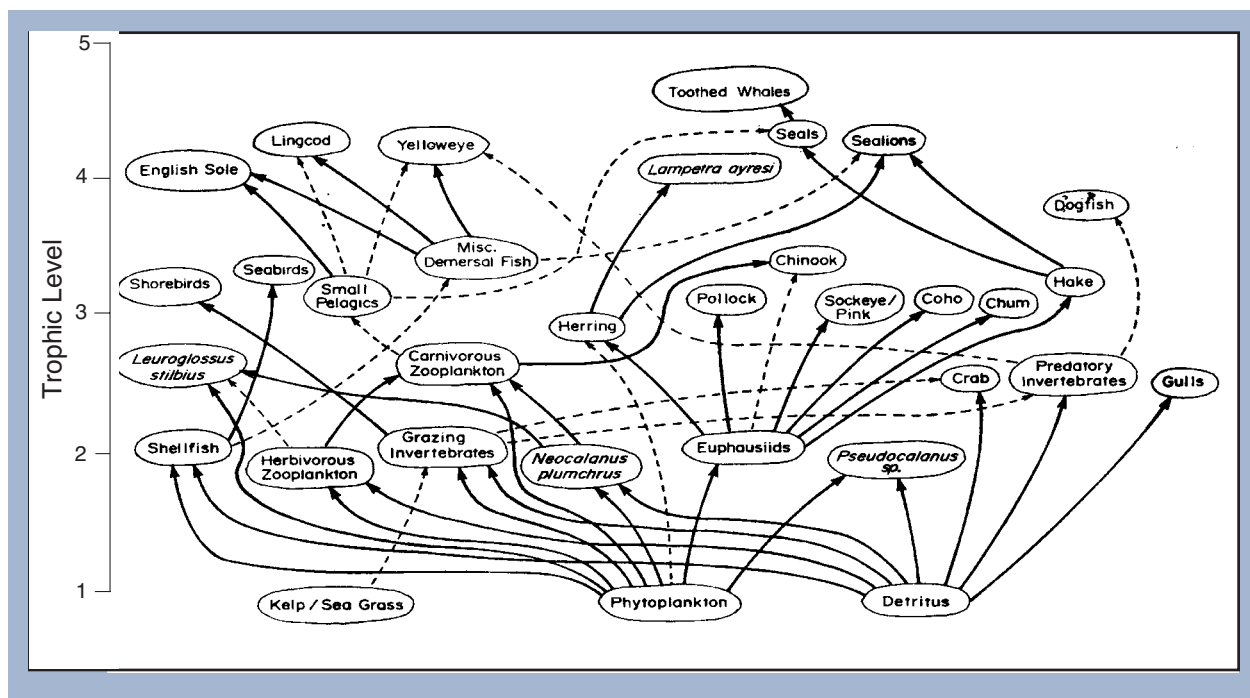


Figure 4. A schematic representation of the relationship in 1998 between the 32 function groups in the Strait of Georgia Ecopath model. Prey items representing greater than 25 percent (solid lines) and 20-25 percent (dashed lines) of a functional groups diet are represented in the diagram. Prey items representing less than 20 percent of a functional groups diet are not included.

ids indicates the ecosystem in the Strait of Georgia will continue to adjust to the new increased productivity.

A consistent indicator of changing regimes has been the intensity of the Aleutian Low, the area in the North Pacific bounded by 100.5 kPa in the winter. From 1977 to 1989, the area of low pressure increased in the winter in the subarctic Pacific and the strength of the low intensified. After 1989 there were weak lows. Other indicators of regime shifts are the temperature-based Pacific Decadal Oscillation (Hare and Mantua 2000) and the atmospheric forcing index (McFarlane *et al.* 2000) that combines an index of the Aleutian Low, the Pacific Decadal Oscillation and a measure of atmospheric circulation.

For the Strait of Georgia, an aggregate of physical and biological indices may best define when regimes shift. However, the research needed to develop this index remains to be funded. Management of individual species may be more effective using ecosystem species management charts (Figure 5, Beamish and McFarlane 1999). These charts are still experimental, but highlight key physical and biological relationships that both alter and stabilize recruitment.

Lessons Learned

Environmental factors have not been a major consider-

ation in fisheries management because there was little evidence that such factors were non-random. The discovery of persistent climate and ocean states that shift suddenly from one state to another indicates the need to re-think how marine fisheries are managed. The re-thinking also applies to the science that supports resource management.

Climate is an important consideration in determining fishing quotas. Fishing impacts remain important, but they can no longer be considered the sole regulator of population size. Once climate is recognized as important in the natural regulation of fish abundance, what humans add to the air becomes as important to fish as what gets added to the water. Also, recognizing the role of climate in the regulation of fish abundance and the discovery of regimes may make it possible to forecast the average productive state of an ecosystem.

Conclusion

Forecasting marine ecosystem change has not occurred possibly because stock assessment scientists tend to view the ocean as rather homogeneous over time and space. However, the scientists studying the impacts of global warming may provide the needed focus by developing models with an ability to capture decadal-scale processes such as the Pacific Decadal Oscillation and North Atlantic Oscillation.

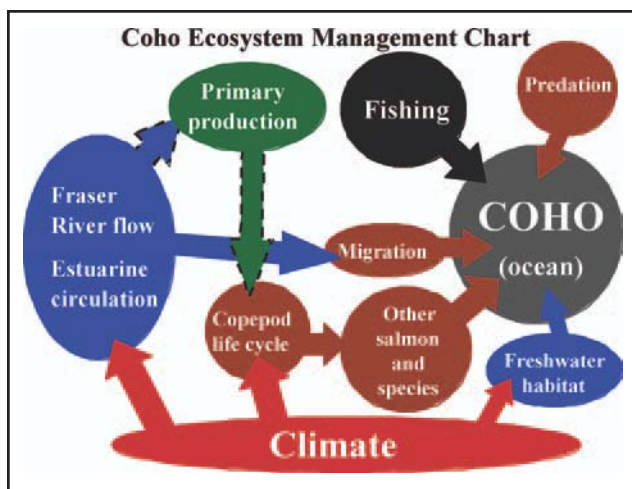


Figure 5. Coho ecosystem management chart. The chart identifies the relationships that may be most influential in the regulation of abundance. The three key areas are fishing, climate-ocean environment, and associated species. Climate, as a regime, affects Fraser River flows, which may affect primary productivity in the Strait. Climate also affects the freshwater habitat, but the principal impact of climate is on the timing of copepod migration into the surface waters. The timing of this migration affects the survival of key competitors for food of coho during their summer feeding period. Growth during the summer affects the survival in the winter and fall and eventually determines brood year strength. Predation is important but adult abundance is determined through growth. A manager needs to know if the regime is productive or less productive for coho and this would be determined using climate indices and timing of copepod migration. In favourable regimes, more smolts would be required to achieve the maximum productivity. In less favourable regimes, the manager needs to be sensitive to the impacts of associated species and other coho. In less favourable regimes, adding more coho smolts or other potential competitors may also increase competition for food resources, which are less available.

The importance of regimes is evident in the increased number of publications examining decadal scale oscillations. Since 2000, the Strait of Georgia appears to be entering a more productive phase. The well-documented, abrupt increase in productivity was related to a large-scale regime shift in 1998 and should provide the evidence needed to begin to change fisheries management science. Hopefully the surprising changes in 1977, 1989, and now in 1998 will be recognized as a signal of how little is known and the learning needed before advising policy makers of the future impacts of climate and climate change.

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Global Climate Change and Sea Level Rise: Potential Losses of Intertidal Habitat for Shorebirds

H. Galbraith¹, R. Jones², R. Park and J. Clough³, S. Herrod-Julius⁴, B. Harrington⁵, G. Page⁶

Abstract

Global warming is expected to result in acceleration in current rates of sea level rise, inundating many low-lying coastal and intertidal areas. This could have important implications for organisms that depend on these sites, including shorebirds that rely on them for feeding habitat during their migrations and in winter.

This case study modeled the potential changes in the extent of intertidal foraging habitat for shorebirds at five sites in the United States that currently support internationally important numbers of migrating and wintering birds: Willapa Bay (WA), Humboldt Bay (CA), San Francisco Bay (CA), Bolivar Flats (TX) and Delaware Bay (NJ). Parts of these sites are currently managed as nature reserves because they are crucial stopover sites for migrating shorebirds.

It is likely many shorebird populations may be limited by availability of feeding habitat at migration and wintering sites (Goss-Custard 1980) and loss of these habitats could have important conservation implications. Foraging habitat loss (together with other climate change-induced changes in their tundra breeding habitats) could result in major reductions in bird populations. Using the SLAMM 4 model, projections were made on the impact of rising sea levels on intertidal shorebird feeding habitat at the study sites.

The results suggest major changes in the ability of coastal sites to continue providing habitat for important shorebird populations and communities. In areas where sea walls are constructed to protect human interests, the ecological impacts may be intensified. Finally, at some sites, lessening existing stressors such as aquifer depletion might mitigate the greatest impacts.

Introduction

Sea level rise and the resulting coastal habitat loss are potentially major new threats to ecological resources, including migratory shorebirds. These birds typically breed in the arctic or subarctic and migrate south to spend the winter in Central and South America. These long-distance travelers require a succession of food-rich stopover sites where they can replace fat reserves. During their migrations, most of the birds feed mainly on sandflats or mudflats with dense invertebrate populations.

Sea level rise may result in the loss or changes to such habitats. Also, at most coastal sites, sea level rise is not the only factor threatening shorebird habitats. Land reclamation, urbanization, and industrialization, together with land subsidence due to aquifer depletion add to the potential rate of intertidal habitat loss.

As sea levels rise, state and local authorities will be challenged to find ways to protect currently dry land. Simply constructing protective structures may meet this requirement. However, what are the potential implications for coastal natural habitats and how should

¹ Galbraith Environmental Sciences, 289 Wiswall Hill Rd., Newfane, VT 05345, USA; hg1@sover.net, and to whom correspondence should be sent.

¹ Stratus Consulting Inc.

² EcoModeling

³ US Environmental Protection Agency

⁴ Manomet Center for Conservation Sciences

⁵ Point Reyes Bird Observatory

planners balance the needs of society with those of nature? This study is an attempt to begin addressing these questions.

Case Study

This predictive study has four main components.

- 1) Selecting study sites on the east, west, and Gulf of Mexico coasts of the United States that have at least hemispheric importance for their shorebird populations. Each of the five sites (Willapa Bay (WA), Humboldt Bay (CA), San Francisco Bay (CA), Bolivar Flats (TX) and Delaware Bay (NJ); Figure 1) may support up to a million or more birds annually.

- 2) Quantifying the current intertidal feeding habitats for shorebirds at each study site using maps from the U.S. Fish and Wildlife National Wetlands Inventory.

- 3) Identifying probabilistic sea level change scenarios at each site. The scenarios used in the analyses were obtained from US EPA (1995) supplemented in southern San Francisco Bay with data from local tide gauges. The tables in US EPA (1995) allow the estimation of probabilistic sea level changes for specific sites and are partly based on historical rates of sea level change superimposed on projected 50 percent and 5 percent probability global sea level change by 2100 (34 cm and 77 cm, respectively). These were obtained from tide gauges at or close to each site. The projections for 50 percent and 5 percent probability are based on assumed global temperature increases of 2°C (50 percent probability) and 4.7°C (5 percent probability), respectively. These temperature change estimates conform closely to the estimates in the most recent Intergovernmental Panel on Climate Change (IPCC) 'best estimate' for 2100 (IPCC 2001).

- 4) Forecasting changes in the intertidal feeding habitats in response to sea level change. Habitat change in response to sea level rise was modeled using the fourth version of the Sea Level Affecting Marshes Model (SLAMM 4). SLAMM uses a decision tree to convert the habitat type occurring in a 30-meter

cell to another for a given change in the inundation regime. The variables included in this process are elevation, habitat type, slope, sedimentation, accretion and erosion rates, substrate type (rock, sand, mud, etc.), over wash, the degree of exposure to the open ocean, salinity, the water table in the affected area, and the extent to which the affected area is protected by sea walls. Further details regarding SLAMM in general have been presented in Park *et al.* (1993). SLAMM 4 is the most recent version of this model, originally developed in the early 1980s.

The changes in the extent of intertidal habitats projected by SLAMM 4 for each study site are shown Table 1. These projections indicate that with the exception of Bolivar

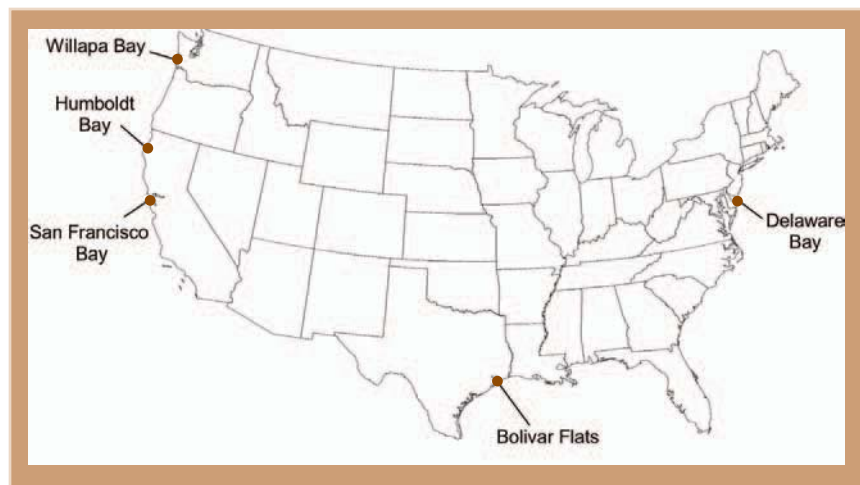


Figure 1. Location of study sites.

Flats, all of the study sites will lose tidal flat habitat, but the extents of loss will be site-specific. The habitat loss will be most marked at southern San Francisco Bay, Humboldt Bay, and Delaware Bay. Substantial areas of tidal flats are projected to be lost at these sites as soon as 2050, even using the 50 percent probability scenario. Without considering any future acceleration due to climate change (keeping the current rate of sea-level rise), southern San Francisco Bay may lose more than half of the current tidal flats by 2100. Using the historical rate of sea level rise, Bolivar Flats may lose all the tidal flat habitat by 2100.

This level of habitat loss is likely to result in major reductions in shorebird populations at these sites. Indeed, if the 50 percent probability predictions for south San Francisco Bay and Delaware Bay are accurate, it is difficult to imagine how these sites could continue to support anything close to the current numbers of shorebirds.

| | | <i>Historical Rate of Sea Level Change*</i> | | <i>50% Probability*</i> | | <i>5% Probability*</i> | |
|--|----------------|---|-------------|-------------------------|-------------|------------------------|-------------|
| <i>Habitat</i> | <i>Current</i> | <i>2050</i> | <i>2100</i> | <i>2050</i> | <i>2100</i> | <i>2050</i> | <i>2100</i> |
| <i>Willapa Bay</i> | | | | | | | |
| Tidal Flats | 21,157 | -0.7 | -0.7 | -7.5 | -18.1 | -25.8 | -61.5 |
| Salt Marsh | 3,455 | 12.8 | 12.8 | 9.5 | 10.5 | 13.6 | 12.8 |
| Upland and Other | 62,389 | -0.5 | -0.5 | -0.5 | -0.7 | -0.8 | -1.3 |
| <i>Humboldt Bay</i> | | | | | | | |
| Tidal Flats | 1,078 | -0.1 | -0.1 | -13 | -28.6 | -42.4 | -91.3 |
| Salt Marsh | 40 | 72.6 | 72.6 | 88.9 | 175.6 | 229.2 | 1,886 |
| Upland and Other | 12,750 | -0.2 | -0.2 | -0.3 | -0.6 | -0.7 | -6 |
| <i>San Francisco Bay – Northern</i> | | | | | | | |
| Tidal Flats | 4,117 | 0 | -4 | -11.9 | -39.4 | -35.9 | -80.7 |
| Salt Marsh | 613 | 0 | 0 | 0 | 0 | 0 | -18.1 |
| Upland and Other | 1,294 | 0 | 0 | 0 | 0 | 0 | 0 |
| <i>San Francisco Bay – Southern</i> | | | | | | | |
| Tidal Flats | 12,039 | -12.9 | -53.9 | -24 | -69.9 | -42.9 | -83.1 |
| Salt Marsh | 3,534 | 0.8 | -50.7 | -2.2 | -63.2 | -11.6 | -82.9 |
| Upland and Other | 75,694 | 0 | -0.2 | -0.1 | -0.5 | -0.2 | -0.6 |
| <i>Bolivar Flats</i> | | | | | | | |
| Tidal Flats | 398 | -14.6 | -93.8 | -37.6 | 1.8 | -80.6 | 1073 |
| Salt Marsh | 5,774 | 4.3 | 40.9 | 4.7 | 48.5 | 14.2 | 53.5 |
| Upland and Other | 18,275 | -1.3 | -12.9 | -1.4 | -17.6 | -4.4 | -53.1 |
| <i>Delaware Bay</i> | | | | | | | |
| Tidal Flats | 2,665 | -6.1 | -23 | -19.8 | -57.4 | -43.1 | 19.8 |
| Salt Marsh | 13,776 | 6.4 | 9.3 | 9 | 12.2 | 11.3 | -4.2 |
| Upland and Other | 20,538 | -3.5 | -5.5 | -5.3 | -7.5 | -6.9 | -11.3 |
| * The historical rate of sea level change projections are based on actual past sea level changes measured at the site. The 50 percent probability projections represent future sea level change with an assumed 2°C warming (US EPA's 'best estimate' temperature scenario); 5 percent probability projections represent future sea level change with an assumed 4.7°C warming (US EPA's 5 percent probability temperature scenario) | | | | | | | |

Table 1. Current area (ha) and projected future percent changes in intertidal and upland habitat at the study sites under three sea level rise scenarios.

This study also illustrates another important point – climate change does not happen in a vacuum. Climate change impacts will interact with other stressors. For example, the modeling predicts the extent of habitat loss at any one site will vary depending on local geomorphologic and anthropogenic factors. These include local land surface movements (subsidence, isostatic rebound, etc.), human exploitation patterns in the area (e.g., of aquifers), and human responses to the inland movement of the coastline.

At sites where crustal movements exacerbate the rate of sea level rise, the loss of feeding habitat is likely to be accelerated. For example, under the 50 percent probability scenario, southern San Francisco Bay will lose about 70 percent of its intertidal feeding habitat by 2100. Comparing that with the corresponding

prediction for northern San Francisco Bay (a loss of about 39 percent) shows much of the habitat loss in the southern bay is likely from factors un-related to, but exacerbating, the rise in sea level.

In parts of southern San Francisco Bay, the land surface has historically been subsiding partly because of aquifer depletion and crustal compaction. This subsidence, superimposed on global sea level rise, creates a much larger affected area. If current trends continue, by 2100 crustal subsidence alone is likely to have been responsible for about 1.5 m of the 1.8 m change in sea level predicted by the 50 percent probability scenario. In this case, human exploitation of local natural resources greatly exacerbates the likely effects of global climate change. In contrast, Willapa Bay has only modest rates of habitat loss projected because

the global sea level rise is being mitigated by crustal rise.

All the above model predictions assume that no new coastal protection structures will be installed. However, it is likely that the local human populations at these sites will protect valuable areas from sea level rise. To evaluate the likely influence of human responses, one simple protection scenario is considered for Bolivar Flats – all current dry land areas will be protected with new sea walls. This model reduced the amount of upland habitat to be lost, but had a 20percent increase in intertidal habitat loss by 2100 under the 50 percent probability scenario. This is because building sea walls stops the coastal system from simply moving inland and thereby exacerbates the effects on intertidal areas. Therefore, the protection measures work because up-

land habitats are protected, however, this protection is at the expense of the intertidal habitats.

Lessons Learned

Three are major lessons this in study. All have important conservation policy and management implications. First, there will be major changes in the ability of coastal sites to continue providing foraging and migration habitat for shorebirds. If the projections are correct, sites like San Francisco Bay may suffer serious adverse impacts. Second, the greatest impacts might be mitigated by lessening existing stressors such as aquifer depletion. Third, constructing sea walls to protect human interests may intensify the ecological impacts.

Conclusions/Next Steps

This study did not attempt to incorporate the potential effects of climate change-induced habitat loss for shorebirds in their wintering or breeding areas. The late winter months may comprise energy ‘bottlenecks’ for many shorebirds and less than adequate food reserves can result in increased mortality rates (Davidson and Evans 1982). Loss of feeding habitat due to sea level change could reduce the carrying capacity of wintering areas in Central and South America. Also, the high-latitude areas where shorebirds breed are expected to have the greatest habitat impacts of climate change.

Recent studies show climate-induced habitat changes may already be occurring in the arctic and subarctic areas of North America (Chapin *et al.* 1995). The combined effects of habitat change on their breeding

areas (Zockler and Lysenko 2000), and intertidal habitat loss at their wintering and migratory staging sites could, potentially, have even more severe effects than could be brought about by any one factor.

Future research will adopt a ‘life-cycle’ approach by incorporating climate change-induced effects on the wintering and breeding grounds. This should provide a more comprehensive appraisal of the likely effects of climate change on this group of migratory birds.

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Forecasting Climate Impacts on Coastal Ecosystems

Donald Scavia¹, John C. Field², Donald F. Boesch³

Abstract

Increases in greenhouse gases are expected to lead to increased air and ocean temperatures, increases in oceanic CO₂, and changes in precipitation patterns. An assessment of the potential impacts of these and other changes on coastal and marine systems (Boesch et al. 2000, Scavia et al. 2002) was done as part of the National Assessment of Potential Consequences of Climate Variability and Change (NAST 2001).

These analyses used a range of forecasting capabilities, from prediction to scenario analysis. In most cases, ecological scenarios were built upon predictions and simulations of changes in the physical

Introduction

Over half of the United States population lives on the 17 percent of land considered coastal (Culliton 1998), and these areas become more crowded every year. Within the next 25 years, the coastal population is likely to increase by approximately 25 percent (about 18 million people) with most of the growth occurring in the already crowded states of California, Florida, Texas and Washington (Boesch *et al.* 2000). This growth, along with rising affluence, is increasing the demand for coastal and marine resources, which increases both the stress on those resources and their vulnerability to climate change and variability. Conversely, climate change will interact with these existing and increased stresses, potentially accentuating their negative impacts.

Forecasts of Climate Change – The climate change scenarios used in this analysis were based on general circulation model (GCM) simulations conducted by the United Kingdom's Hadley Centre for Climate Prediction (HadCM2) and the Canadian Climate Centre (CGCM1), referred to here as the Hadley and the Canadian models, respectively (NAST 2001). Output from these models – scenarios – were used as two plausible climate futures rather than accurate predictions of what will happen.

Over the next 100 years, global warming is expected to accelerate sea level rise from 9 to 88 cm. While

climate change effects on hurricane frequency are hard to discern, hurricane wind strength could increase as a result of elevated sea surface temperatures. Regardless of potential changes in frequency and intensity, coastal storms and surges will be riding in on a higher sea level.

The hydrologic cycle is likely to change with a changing climate. However, regional projections of precipitation vary considerably among the models. That has a significant impact on the projected effects of river runoff. Water-balance models (Wolock and McCabe 1999), based on the precipitation and temperature projections from the climate models, provide forecasts of river runoff that vary considerably (Figure 1). For example, Hadley projections result in a 34 percent increase in total runoff along US Atlantic and Gulf coasts, but the Canadian model projections result in a 32 percent decrease. These analyses suggest changes in the freshwater delivery to coastal environments are highly uncertain. Also, both models predict an increase in flashiness of extreme rainfall-runoff events, which can significantly increase the chemical and sediment load delivered to the coast.

Mean temperature of the upper 300 meters of the oceans has increased by 0.31°C (Levitus *et al.* 2000), primarily due to human-caused climate change (Levitus *et al.* 2001). The decline in aerial extent and thickness of Arctic ice over the last 20 years (Johannessen *et al.* 1999, Rothrock *et al.* 1999) has also been attributed to

¹ National Ocean Service, National Oceanic and Atmospheric Administration, Silver Spring, Maryland 20910

² College of Ocean and Fisheries Science, University of Washington

³ Center for Environmental Studies, University of Maryland

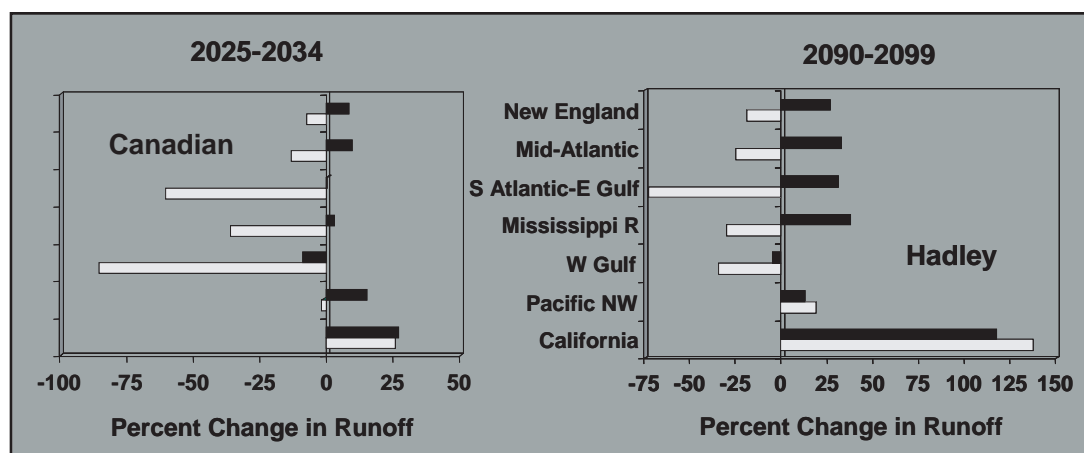


Figure 1: Projected changes in average annual runoff for basins draining to coastal regions from the Canadian and Hadley Centre General Circulation Models. Data from Wolock and McCabe (1999).

human-induced global changes (Vinnikov *et al.* 1999). The ocean currents, fronts, and upwelling and downwelling zones that influence the distribution and production of marine ecosystems are likely to change in response to the above alterations in temperature, precipitation, runoff, salinity, and wind. While these changes will occur over a wide range, some projections show significant alterations in the deep-water ‘conveyor belt’ circulation, with potentially dramatic feedback on large-scale climate patterns (Driscoll and Haug 1998; Broecker *et al.* 1999).

Case Study

The forecasts just described discuss changes in the climate drivers most relevant to coastal and marine systems. These were used to assess potential impacts on shorelines and developed areas, wetlands, estuaries, and coral reefs. Because the ability to accurately forecast quantitative impacts is low at this time, these climate forecasts were extended into coastal and marine physical and ecological impacts through ecosystem scenarios. These results are described by Boesch *et al.* (2000) and Scavia *et al.* (2002) and summarized here.

Shorelines and Developed Areas – Rising sea level can inundate lowlands, erode beaches, cause barrier islands to migrate inland, increase coastal flooding, and increase the salinity of rivers, bays, and aquifers. Changes in the frequency of severe storms and increased rainfall intensity could further aggravate flooding and storm damage. Land most vulnerable to inundation is in Florida, Louisiana, North Carolina, Texas, and Maryland (Titus and Richman 2001). Even those states with relatively little low land, however,

have shorelines that are vulnerable to erosion and storm damage.

Storm-surge floods, waves, and coastal erosion are natural processes that pose hazards. Increasing economic damages over time have been shown to be largely a consequence of increased development and appreciation in value of coastal property (Pielke and Landsea 1999).

While there is not yet any clear evidence for climate-induced changes in the frequency or intensity of hurricanes and coastal storms, there is little doubt that future waves and storms will be superimposed on a rising sea. In addition to increased erosion and flooding, a rising sea may also increase salinity of freshwater aquifers, alter tidal ranges in rivers and bays, change sediment and nutrient transport, and alter patterns of coastal chemical and microbiological contamination. Secondary impacts, including inundation of waste disposal sites and landfills, may introduce toxic materials into the environment, posing new threats to the health of coastal populations and ecosystems. Steep, rocky shoreline areas might experience relatively little change, but low-lying, gently sloping areas could experience severe inundation.

Coastal Wetlands – Coastal marshes, mangroves, forested wetlands, mudflats, and estuarine beaches provide critical refuge and forage opportunities for wildlife, fishes, and invertebrates supporting the commercial and recreational livelihoods of many communities. Survival of these ecosystems under a changing climate depends primarily on their ability to adapt to change, but also on whether human activities impair their natural adaptive capacity. Natural

biological and geological processes allow responses to gradual changes. However, accelerated sea level rise threatens these habitats with inundation, erosion, and saltwater intrusion. If landward margins are armored, effectively preventing inland migration, then wetlands could be lost if they are unable to accumulate substrate at a rate adequate to keep pace with future increased rates of sea level rise.

Changes in the timing and volume of freshwater delivery to coastal wetlands will also be critical, yet perhaps the most difficult to assess. In contrast to uncertainties associated with regional impacts of climate change on hydrology, it is clear increased human population and coastal development will create higher demands for freshwater resources. Changes in delivery of fresh water will affect sediment supply, which is needed to allow wetlands and mangroves to cope with rising sea level (Reed 1990, 1995). This is particularly critical for areas where wetlands are not able to migrate because they are blocked by bluffs, coastal development, or shoreline protection structures. In steeply sloped or developed areas where inland migration is not possible, wetlands and mangroves will have to rely on vertical growth to adjust.

Estuaries – Estuarine impacts from climate change will be manifest through exacerbation of current stresses, including those imposed by a significantly altered nitrogen cycle (Howarth *et al.* 1996; Vitousek *et al.* 1997). Eutrophication can lead to greater phytoplankton biomass, decreased water clarity, anoxic and hypoxic bottom waters, more frequent and longer lasting harmful algal blooms, degraded sea grasses and corals, altered plankton and the benthic community structures, and loss of biotic diversity and fisheries (Jørgensen and Richardson 1996; NRC 2000).

Estuaries vary significantly in their vulnerability to eutrophication. Climate change will likely influence that vulnerability in several ways, including changes in mixing characteristics caused by alterations in freshwater runoff, and changes in temperature, sea level, and exchange with the coastal ocean (Kennedy 1990; Peterson *et al.* 1995, Moore *et al.* 1997, Najjar *et al.* 2000). Changes in freshwater inflow, air temperatures, and precipitation patterns can also influence water residence time, nutrient delivery, dilution, vertical stratification, and control of phytoplankton growth rates (Malone 1977, Cloern 1991, 1996, Howarth *et al.* 2000, Buckley *et al.* 1998). Increased air temperature may also lead to earlier

snowmelt and the resulting peak in freshwater inflow. In those cases, summer flows may be reduced as a result of greater evapotranspiration. This would increase estuarine salinity and modify stratification and mixing, thus influencing biotic distributions, life histories, and biogeochemistry.

Decreased freshwater runoff will increase estuarine water residence time, whereas increased runoff will decrease residence time (Moore *et al.* 1997). Impacts in regions experiencing decreased precipitation may be exacerbated because water managers will likely store more freshwater within the watershed in response to drought. The effects of altered residence times can be considerable because, even at their fastest growth rates, phytoplankton populations are only able to double once or twice per day.

Increased and earlier freshwater inflow can also modify salinity regimes. For example, shifting freshwater flows from spring to winter will contribute to an overall rise in estuarine salinity. If this effect strengthens significantly over the next century, increased winter flows and decreased spring flows will decrease salinities in the wet season and dramatically increase salinity levels in the dry season, further stressing an ecosystem already heavily impacted by a wide range of other problems.

Changing precipitation patterns can also influence nutrient delivery. The effects of precipitation variability on delivery of nitrogen may also be magnified by land use practices. Nitrate tends to build up in soils during dry years, largely from reduced uptake of soil nutrients by crops, and is flushed into streams at much larger rates during subsequent wet years (Goolsby *et al.* 1999). Thus, a wet year that follows dry years tends to produce the largest fluxes of nitrate from the catchment to the Gulf of Mexico. If future precipitation regimes are more variable, a trend suggested by many climate models, this could increase the net impacts to coastal areas affected by nutrient over-enrichment and eutrophication.

Coral Reef Ecosystems – Coral reef communities, damaged by acute events, will recover if other environmental factors remain generally favorable. Conversely, coral communities may persist under sub-optimal conditions in the absence of acute stresses. However, once a chronically stressed reef is severely damaged, recovery is unlikely (Hughes 1994).

This is important because chronic stresses, such as nutrient loading and overfishing are likely to increase the vulnerability of coral ecosystems to climate-related

acute stresses such as high-temperature episodes. Reef ecosystems are susceptible to changes in the frequency and/or magnitude of temperature extremes because many coral species live near their upper limits of thermal tolerance. Bleaching can occur when temperatures exceed the maximum needed for survival. If the high-frequency SST variation of the past 20 years continues and is superimposed on a general warming trend, accelerated bleaching and mass mortality events could increase dramatically in many tropical reefs.

When dissolved in seawater, increased carbon dioxide reduces alkalinity and calcium carbonate concentrations. This, in turn, decreases calcification rates of reef-building corals and coralline algae. These calcification rates are likely to decline between 17 and 35 percent by the year 2100 (Figure 2, Gattuso *et al.* 1999, Kleypas *et al.* 1999a, 1999b); as much as one-third of that decrease may already have occurred. The resulting reduced skeletal density and/or growth rate will increase reef vulnerability to physical damage, bioerosion, some forms of predation, and the corals' ability to compete for space.

The effect of increased atmospheric CO₂ will be greatest at the northern and southern margins of coral distributions because CO₂ is more soluble in cooler waters. Therefore, these effects will be most severe at higher latitudes, reducing the ability of reefs to expand their ranges poleward as might otherwise be expected in response to ocean warming.

Changes in sediment, fresh water, and nutrient loads, as discussed in the Estuaries and Coastal Wetlands, will also impact coral ecosystems adjacent to coastal watersheds (Jickells 1998, Moffat 1998, Lapointe 1999). Negative impacts can be expected in areas where changes in land use and precipitation patterns lead to increased sedimentation, eutrophication, or algal growth.

Lessons Learned

Adaptation and Coping Strategies – There are significant uncertainties in forecasts of some important climate forces on coastal systems, particularly at regional scales. While forecasts of changes in atmospheric CO₂, air temperature, and sea level are becoming more reliable as model and data resolution increase, potential changes in the timing and strength of regional river runoff, coastal ocean and estuarine temperatures, and

coastal circulation remain uncertain. However, enough is known to begin assessing their potential impacts.

The assessment (Boesch *et al.* 2000, Scavia *et al.* 2002) was able to draw on expert opinion and relatively simple, and in many cases, qualitative analysis to extend forecasts of climate change into potential scenarios of impacts on coastal systems. The assessment identified important areas that should be supported to improve forecasts of these impacts. For

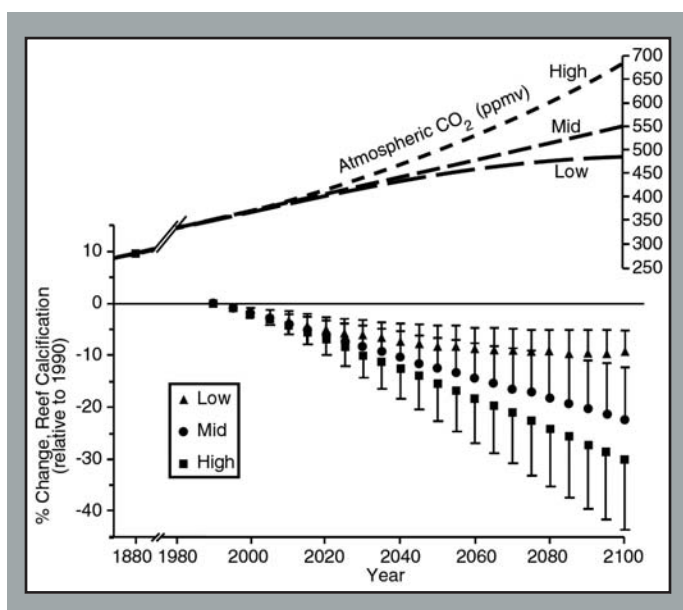


Figure 2: Reductions in coral calcification rates suggested by model results of increasing atmospheric carbon dioxide (Gattuso *et al.* 1999)

coastal systems, these center on developing quantitative predictions of:

- How sea level rise, erosion, coastal storms, and upland development impact the transformation of coastlines and wetlands;
- How changes in precipitation, freshwater loads, and land-use will influence the susceptibility of estuaries to nutrients; and
- How changes in ocean temperature and CO₂ will impact the growth and biogeography of coral ecosystems.

Although these potential impacts of climate change and variability will vary from system to system, it is important to recognize they will be superimposed upon, and in many cases intensify, other ecosystem stresses (pollution, harvesting, habitat destruction, invasive species, land and resource use, extreme natural events), which may lead to more significant consequences. Thus, there is a growing need for models and analysis tools capable of assessing multiple stresses.

Climate change likely over the next century will have many consequences for most U.S. coastal ecosystems, and some of these may substantially alter human dependencies and interactions with these complex and linked systems. The climatic effects will be superimposed upon, and interact with, a wide array of current stresses, including excess nutrient loads, invasive species, habitat destruction, and toxic chemical contamination. While the ability of these ecosystems to cope with or adapt to climate change or variability is compromised by extant stresses, the inverse is also likely to be true – ecosystems will be better suited to deal with climate variability and change if other stresses are significantly reduced.

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Simulating Coastal Environments with Landscape Models: Applications in the Mississippi Delta and Lake Erie

Jay F. Martin¹

Abstract

The survival and creation of Mississippi Delta marshes and the impacts of altered river inputs, accelerated sea-level rise, and management proposals were examined with a series of landscape models. The spatially-explicit models integrated routines and simulated interactions between ecological and hydrodynamic processes. In the future, models of this type will be used to predict impacts of land use policies on water quality and habitat distribution along the Lake Erie coastline.

Introduction

Spatially explicit landscape models can be used to test hypotheses about landscape functioning and evolution and how human intervention may alter the landscapes. These models analyzed how natural forces affect ecosystem function and habitat change in an area of the Mississippi delta extending west from the Mississippi River to Freshwater Bayou (Martin *et al.* 2002). This allowed management proposals to be based on a more accurate understanding of coastal and delta systems.

Large-scale management plans are often based on untested hypotheses about landscape functioning and how management changes alter this functioning. Spatial models provide an opportunity to test hypotheses by simulating the dynamics of the natural system and predicting the effects of proposed management plans through time and across the landscape (Costanza *et al.* 1990). In the future, linking such models with land-use change modules has the potential to quantitatively predict impacts of governmental land use policies upon coastal water quality and habitats.

Case Study

Spatial Modeling of the Mississippi Delta – Presently, many of the world's deltas, including the Mississippi River Delta, are rapidly deteriorating, primarily due to

human activities (Day *et al.* 1997, Day *et al.* 1995, Milliman *et al.* 1989). Anthropogenic actions isolate delta wetlands from river inputs and threaten the vitality of these systems and their ecological and economic resources. Isolated marshes succumb to subsidence and convert to open water as rapidly as 102 km²/yr (Reyes *et al.* 2000). As a result of losing 787 km² of land, the Barataria Basin within the Mississippi Delta suffered a 26 percent decline in net primary productivity from 1956 to 1988 (Day *et al.* 1997). Over the next 50 years, the loss of public resources from continued land loss across the Mississippi Delta is estimated at more than \$37 billion (Bourne 2000). Sea level rise has the potential to exacerbate these processes in deltas across the globe.

Spatially-explicit landscape models were used to evaluate the effects of accelerated sea level rise and altered river inputs on the survival and creation of Mississippi Delta marshes. Scenarios reducing river inputs resulted in losses of marsh coverage; increased river inputs created new marsh and demonstrated the potential to sustain delta marshes. The models used in this study simulate dynamic and spatial interactions between ecological, edaphic (e.g., soil and sediments), and hydrodynamic processes. They are unique in their ability to predict the growth and regression of deltas for multiple decades.

¹ Ecological Engineering Group, Department of Food, Agricultural, and Biological Engineering, The Ohio State University, 590 Woody Hayes Drive, Columbus, OH 43210-1057.

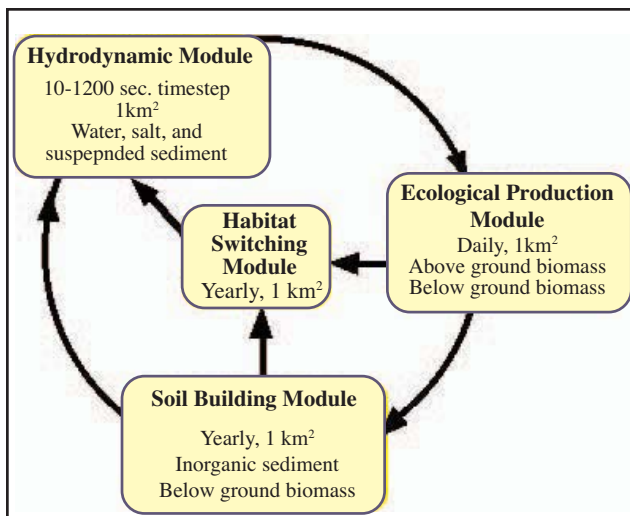


Figure 1. The landscape models passed information between modules operating at different time steps. The ecological module responded to daily salinity and flooding averages from the hydrodynamic module. The soil module accumulated inputs from the ecological module to calculate annual changes in elevation. Possible habitat change took place each year as salinity and vegetative biomass crossed thresholds. The hydrodynamic routine responded to changes in elevation and habitat dependent on Mannings coefficients. In the Mississippi Delta sModel, unlike the BTELSS, all modules had a 1-km² spatial resolution.

The spatial models integrate hydrodynamic routines with ecological and soil modules across 1 km² spatial grids (Figure 1, Martin *et al.* 2000, Reyes *et al.* 2000). Significant advances to the application of spatial models in aquatic-terrestrial interfaces accompanied the development of the Barataria-Terrebonne Ecological Landscape Spatial Simulation model (BTELSS) and the Mississippi Deltas Model. The explicit hydrodynamic module, much enlarged study area, and improved habitat switching algorithm are substantial advances of the BTELSS that built on a previous modeling study of coastal Louisiana (Costanza *et al.* 1990).

The Mississippi Deltas Model is characterized by a higher resolution hydrodynamic module, operating at 1 km² and variable time steps, as well as the ability to simulate the conversion of open water areas to marsh habitats. An improved sediment algorithm and marsh colonization routine are pertinent processes to create new land habitats in delta environments of the Mississippi Deltas Model.

Forcing functions consist of historical data from 1956 through 1995 (Reyes *et al.* 2000) that include climatic variables, river flow and sediment concentration, relative sea level rise, tides, and salinity. For calibration, the Mississippi Deltas Model was initialized with a 1978 habitat map from the United States Fish & Wildlife Service (USFWS), and using data series from 1978 to 1988, simulated 10 years of habitat change. The resulting 1988 habitat map was compared to a 1988 USFWS habitat map with a spatial statistics routine (Costanza 1989) that returned a value of 94.9 out of a possible 100. A validation value of 95.4 was produced when the Mississippi Deltas Model was initialized in 1988, ran for 10 years, and compared to the 1998 USFWS habitat map. Further calibration and validation of the models included comparing predicted habitat trends with historic rates of change, and comparing recorded and predicted salinity and suspended sediment concentrations at specific locations. The historical data series were repeated to simulate conditions for scenarios into the future (Martin 2000, Reyes *et al.* 2000).

Barataria Basin (NAP) scenarios simulated continuing current trends and management for each study area beginning in 1988, and offered a baseline of comparison for other simulations. These scenarios had distinctly different outcomes for each study area in the BTELSS and Mississippi Deltas Model (Table 1).

In the Barataria Basin, total marsh coverage declined by more than 56 percent during the 30 year No Action Plan (Figure 2, Martin *et al.* 2000). This marsh loss is due to the interaction of natural processes such as high

| | | Marsh Coverage | |
|--|------|--------------------|----------------------|
| Scenario | Year | (km ²) | (% change from 1988) |
| Barataria Basin (BTELSS simulations) | | | |
| Initial condidtions | 1988 | 1949 | |
| NAP (RSLR=1.2 cm-yr ⁻¹) | 2018 | 849 | -56.4 |
| RSLR=2.2 cm-yr ⁻¹ | 2018 | 196 | -89.9 |
| Mississippi Deltas Model (MDM) study area | | | |
| Initial condidtions | 1988 | 1696 | |
| NAP (RSLR=0.5 cm-yr ⁻¹) | 2058 | 1925 | 13.5 |
| Double river input | 2058 | 2117 | 24.8 |
| Half river input | 2058 | 1713 | 1.0 |
| RSLR=1.2 cm-yr ⁻¹ | 2058 | 1596 | -5.9 |
| RSLR=2.2 cm-yr ⁻¹ | 2058 | 1041 | -38.6 |
| NAP-No Action Plan, RSLR-Relative Sea-Level Rise | | | |

Table 1. Changes in marsh coverage due to scenarios simulated with the BTELSS and Mississippi Deltas Model.

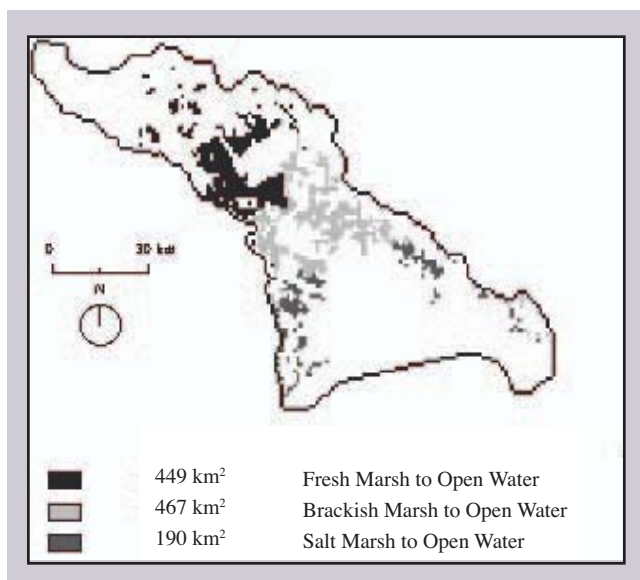


Figure 2. Location of cells that were converted from marsh to open water during the 30 year no action plan in Barataria from 1988 to

rates of relative sea level rise which characterized the Barataria Basin (Penland and Ramsey 1990), and human modification such as levees and dikes that eliminate the input of river sediments to marshes (Martin *et al.* 2000).

The Mississippi Deltas Model study area experienced a 13.5 percent gain in marsh during the 70 year No Action Plan, primarily due to the expansion of fresh marsh coverage associated with the progradation of the Atchafalaya Delta (Figure 3). The contrast in habitat change in these two areas is largely due to regional differences in river inputs. Because levees and closed river outlets prevent inputs from the Mississippi River, the marshes of Barataria Basin cannot keep pace with Relative Sea Level Rise (comprised of both subsidence and eustatic sea level rise) and will continue to convert to open water.

The Atchafalaya River deposits river sediments in its delta, raising the elevation of the substrate in open water areas until floods decrease, allowing vegetative colonization. Subsequent organic soil formation leads to additional elevation and eventual habitat change from open water to marsh. While the Mississippi Deltas Model study area as a whole experienced land gain, marshes more distant from the river mouth, such as those in Pointe au Fer and Marsh Island, suffered losses of marsh in the No Action Plan (Figure 3). These areas received less river input and could not keep

pace with the relative sea level rise of 0.5 cm yr^{-1} , characteristic of the Mississippi Deltas Model study area (Penland and Ramsey 1990).

Other Mississippi Deltas Model simulations in which Atchafalaya River inputs were doubled and halved demonstrated the importance of river flow in sustaining marsh habitats across the model study area (Table 1). Doubling flows increased both the water and sediment delivered to the estuary and increased the growth of the Atchafalaya Delta by 125 km^2 compared to the No Action Plan (Figure 4).

Additionally, marshes farther removed from the deltas experienced greater accretion, which reversed the trend in these areas from marsh loss in the No Action Plan to marsh gain. Pointe au Fer and Marsh Island gained 17 and 31 km^2 of marsh, respectively, during this scenario. Across the study area, there were total gains of 421 km^2 of marsh when the flows were doubled. Reducing

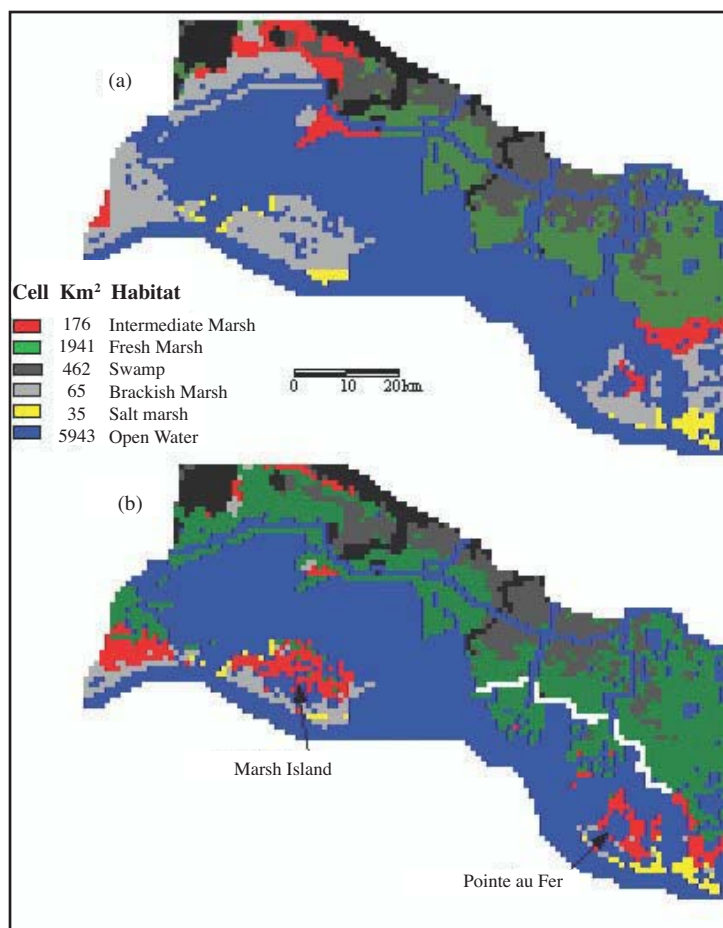


Figure 3. The 1988 United States Fish and Wildlife Service habitat map (A) was aggregated into 6 habitat classifications (swamp, fresh marsh, intermediate marsh, brackish marsh, salt marsh, and open water) and used to initialize the 70-year Current Conditions scenario that terminated in 2058 (B). The white line in the Current Conditions scenario map denotes the shoreline in 1988. Please refer to Table 1 for habitat coverages.

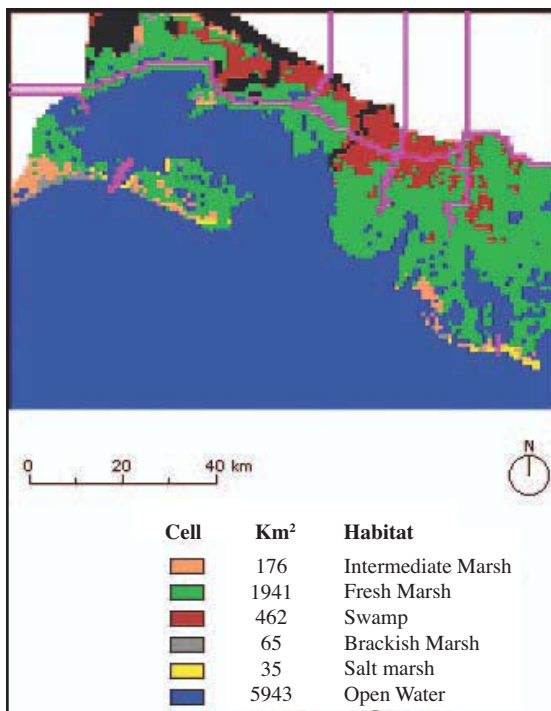


Figure 4. Resulting habitat distribution for the year 2058 for the Mississippi Deltas Model simulation in which the flow and sediment discharge of the Atchafalaya River and

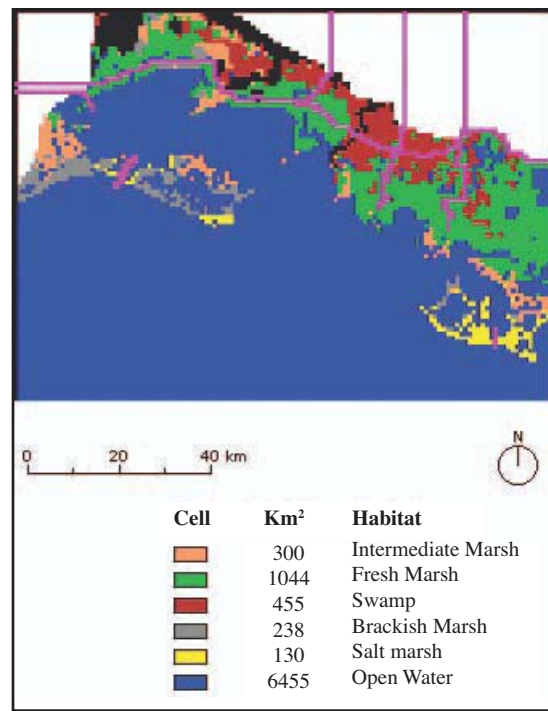


Figure 5. Resulting habitat distribution for the year 2058 for the Mississippi Deltas Model simulation in which the flow and sediment discharge of the Atchafalaya River and Wax Lake Outlet were halved.

Atchafalaya River flow by 0.5 decreased the growth of the Atchafalaya Delta and led to greater land loss across the study area (Figure 5).

Alterations in river inputs in the Mississippi Delta and other deltas throughout the world demonstrate the relevance of this scenario. Natural and anthropogenic changes have reduced the amount of sediments reaching the Nile, Indus, Ebro, Po, and Mississippi deltas by over 75 percent (Day *et al.* 1997, Kesel 1988). Current projects, such as the Three Gorges Dam along the Yangtze River, insure continued alterations of river inputs and impacts upon delta evolution.

Because of the ability to simulate the links and feedbacks between hydrodynamics and ecological processes across large areas over long time periods, models such as the Mississippi Deltas Model are critical tools to quantitatively assess environmental consequences of historical and future river and marine alterations upon deltas.

Further scenarios demonstrate that subsidence combined with accelerated sea level rise will amplify relative sea level rise rates across low-lying deltas over the next 100 years, jeopardizing the ecological and economic services provided by these systems (Day *et al.* 1995, Milliman *et al.* 1989). The BTELSS predicted marsh loss to increase by 89.9 percent in the isolated Barataria

Basin as a result of increasing the relative sea level rise from the No Action Plan rate of 1.2 cm-yr⁻¹ to 2.2 cm-yr⁻¹ (Table 1, Reyes *et al.* 2000). In the Mississippi Deltas Model study area, the trend of marsh gain was reversed to marsh loss when relative sea level rise was increased from the No Action Plan value of 0.5 cm-yr⁻¹ to 1.2 and 2.2 cm-yr⁻¹ (Table 1).

While it is impossible to revert to the natural delta cycle due to human settlement and use of these environments, these models provide quantitative predictions of various alternatives to balance human needs and long-term vitality of delta systems. Results demonstrate maintaining and restoring natural river inputs can create and preserve delta marshes and mitigate present and future deterioration of delta ecosystems.

Applications to Lake Erie – A new effort is underway to develop a spatial watershed model that can quantitatively predict impacts of governmental land use policies upon coastal water quality and habitats. In addition to allowing more complete evaluations of land use policies, models of this type will illustrate links between public actions and environmental changes. Because governmental policies are often evaluated in relative isolation, unforeseen environmental consequences often affect adjacent ecosystems. Cumulative effects of non-point source pollution on coastal environments and

aquatic life is one example. Unforeseen consequences are a concern, both nationally and in the Lake Erie watershed (Ohio Lake Erie Commission 1998).

The overall goal is to develop a spatial watershed model to quantify land use, habitat, and water quality dynamics in coastal areas of Lake Erie that have resulted from changes in land use policies including zoning and public water provision regulations. The initial application will focus on how such policies influence the development of agricultural, forest and natural lands within the Sandusky watershed (Lake Erie, Ohio), and in turn, how these changes impact coastal water quality and habitat change.

This is an ideal setting to develop this model because of the availability of land use change data, a 22 percent gain in developed land from 1982 to 1992, and past research of coastal habitats in Sandusky Bay. Results from this research will aid designing environmental and growth management strategies aimed at protecting and restoring the water quality and natural habitats of the Great Lakes.

By integrating ecological and physical processes with government policies and human behavior, these models will represent a significant advance in disciplines of landscape ecology and environmental management. During the past 15 years, landscape models have been developed to predict and analyze the interaction of physical and ecological processes in coastal zones (Reyes *et al.* 2000, Costanza *et al.* 1990, Sklar *et al.* 1985).

Another type of landscape modeling has sought to predict land use change based on governmental policies (Irwin and Bockstael 2001, Bockstael and Bell 1997, Landis 1995). The new models will unite these fields by developing a watershed model to predict environmental patterns arising from government actions and individual choices regarding land use. By incorporating ecological, physical and economic system processes and relationships, landscape models can identify both short- and long-term impacts of land use and other policies across watersheds and coastal zones (Martin *et al.* 2000). The integration of climatological, hydrological and ecological processes with human decisions creates the need for a tool to effectively manage coastal watersheds and to evaluate a variety of government policies that influence land use change.

The completed watershed model will be composed of four modules: 1) a land use module, 2) a hydrodynamic module, 3) an ecological unit module, and 4) a habitat

change module (Figure 6). Integrating these modules will quantitatively predict the causes and feedbacks between humans and the natural environment.

The land use module will predict watershed land use patterns resulting from policy decisions such as protecting sensitive environmental areas from development, and regulating public water and sewer provision. Inputs to rivers within the watershed will be calculated based on land use types and climatic conditions.

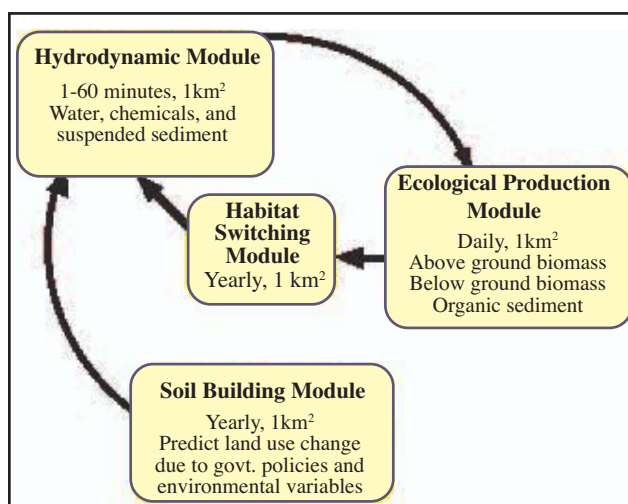


Figure 6. Similar to previous landscape models (Martin *et al.* 2000, Reyes *et al.* 2000) the Sandusky watershed model will pass information between modules operating at different time steps. Changes in land use predicted by the land use change module will alter water quantity and quality inputs to the hydrodynamic module. The hydrodynamic module will also respond to coastal habitat changes to predict water elevations and water quality throughout the watershed. Responding to these variables, the ecological unit module will predict changes in ecological productivity and substrate elevation. Habitat change may take place each year as water level and vegetative biomass crossed thresholds. All of the modules will operate with a 1 km² spatial resolution.

The hydrodynamic module will simulate the transport of water and materials from the watershed through rivers to the coastal zone. Biological and physical processes regulating water quality will be included within the hydrodynamic module.

The ecological unit module will respond to hydrodynamic variables and water quality to simulate biological productivity and habitat change (Fitz *et al.* 1996) within the coastal zone.

When completed, the models will be able to analyze how a variety of changes in the natural and manmade environment affect coastal zones. Potential scenarios to be evaluated include the effects of reductions or increases in erosion and fertilizer use and government

policies regarding land use zoning and the provision of public services such as roads and public sewers.

Lessons Learned

The general patterns predicted by these models are not surprising. For instance, the facts that more river flow and sediment will lead to delta growth and less river flow and sediment will lead to delta decline have been part of geological literature for many years (Roberts 1997). The benefit of these models lies in their ability to quantify the spatial and temporal effects of management plans and changes in forcing functions, which allows for more precise evaluation system manipulations.

For instance, the coverage of marsh was predicted to decrease 12.5 percent (compared to the No Action Plan) if river flow was reduced by one half. Further scenarios could predict the reduction in river flow needed to allow for desired decrease in marsh growth to offset another public works project such as flood management. These models are best applied to quantify the location and timing of landscape changes resulting from management plans and environmental changes.

Conclusion

Scenarios of the Mississippi Delta exemplify the potential of landscape models to quantitatively evaluate management plans in conjunction with natural alterations and lead to a more precise understanding of coastal response to perturbations. During the 15-year development of these models (Costanza *et al.* 1990, Sklar *et al.* 1985), and in current applications in the Everglades (Fitz *et al.* 1996), the goal has been to predict large-scale temporal and spatial patterns resulting from regional impacts. Such models can predict effects of complex interactions, over lengthy time periods and across entire landscapes.

Spatially-explicit models linking policies to land use changes and land use changes to environmental changes can be used to evaluate the impacts of existing and proposed policies. By establishing and quantifying these links, these models will elucidate the environmental and economic trade-offs of governmental policies that impact coastal water quality and habitats.

For example, this approach can account for the cumulative effects of small-scale land use changes on eroded soils and the eutrophication of downstream water bodies. Consequently, the watershed models will be capable of analyzing the effects of government

policies that may require ten or twenty years to fully impact the landscape and may have far-removed consequences.

These models can also evaluate potentially beneficial effects of restoration initiatives. With limited resources, it is imperative to identify restoration plans with the greatest potential to improve coastal water quality and maintain natural habitats. These models will be capable of predicting which land use policies will have the greatest impacts on coastal environments.

Acknowledgements

The United States Army Corps of Engineers, Louisiana Sea Grant Program, and Ohio Sea Grant Program funded this work. Thank you to Drs. Enrique Reyes, Paul Kemp, John Day and Hassan Mashriqui for their help and support throughout these projects.

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Ecological Perspective: The Decline of Carysfort Reef, Key Largo, Florida 1975-2000

Phillip Dustan¹

Abstract

The coral reefs of the Florida Keys are in catastrophic decline from nested multiple stressors. This includes direct and indirect anthropogenic impacts across local and global scales. Between 1974 and 1999, living coral cover on Carysfort Reef, Key Largo, declined 92 percent. Similar ecological degradation has occurred on many reefs throughout the Florida Keys, including Molasses Reef, Looe Key, and Sand Key, but there was no other formal documentation. Starting in 1995, the USEPA Coral Monitoring Project documented a 38 percent loss of living coral cover over four calendar years, documenting a decline from 10.3 percent to 6.4 percent.

Introduction

Coral populations in the Florida Keys (Figure 1) have declined precipitously since 1974, when quantitative monitoring began (Dustan 1977a, Dustan, 1985, Dustan and Halas 1987, Porter and Meier 1992, Porter *et al.* 2001a). Between 1974 and 1982, living coral cover and diversity on Carysfort Reef increased in the shallow areas of the reef, while there was significant decline on the deeper fore reef terrace.

Changes in shallow water reefs seemed driven by physical destruction of dominant stands of *Acropora palmata* (elkhorn coral). Cover increased because the lush, three-dimensional habitat was reduced to planar rubble by boat groundings and storms. Then smaller colonizing species settled on the open substrate. On deeper portions of the reef, colonies were dying from disease and sediment damage, and were not replaced by recruitment (Dustan and Halas 1987).

In 1984, observations on the phenotypic condition of over 9800 corals on 19 different reefs in the Key Largo region revealed 60 percent exhibited signs of physical or biological stress, 5-10 percent were infected with disease (Black Band and White Plague) and about 33 percent appeared healthy. Surprisingly, virtually all the reefs had approximately the same level of stressed corals. This argued for widespread stress as causal

agents (such as reduced water quality), rather than stressors localized to specific reefs (Dustan 1993). During the same period, high rates of coral tissue loss and mortality were documented throughout the Keys (Porter and Meier 1992).

Carysfort Reef continued to decline, and by June 1996 coral cover in the shallows decreased to approximately 10 percent and to 14 percent in the deeper habitat zones. In July 1998, virtually every colony of *Montastrea annularis* species complex was infected with White Plague; large colonies (>1 m diameter) were rapidly dying. Since the skeletal growth rate of *M. annularis* has been measured at 5 to 10 mm/yr., these colonies were over 100 years old. In 2000, coral cover on Carysfort had dropped below 5 percent, and corals ceased providing significant contribution to reef framework construction (Table 1).

Carysfort Reef has entered ecological collapse. Similar ecological degradation occurred on many reefs throughout the Florida Keys. The US Environmental Protection Agency (USEPA) Coral Reef Monitoring Project measured a Keys-wide coral cover loss of 38 percent between 1996 and 2000 (Porter *et al.* 2001a). Carysfort Reef, however, is the only reef with quantitative monitoring that began in the mid-1970s (Dustan and Halas 1987, Porter and Meier 1992).

¹ Department of Biology, University of Charleston, SC 29424

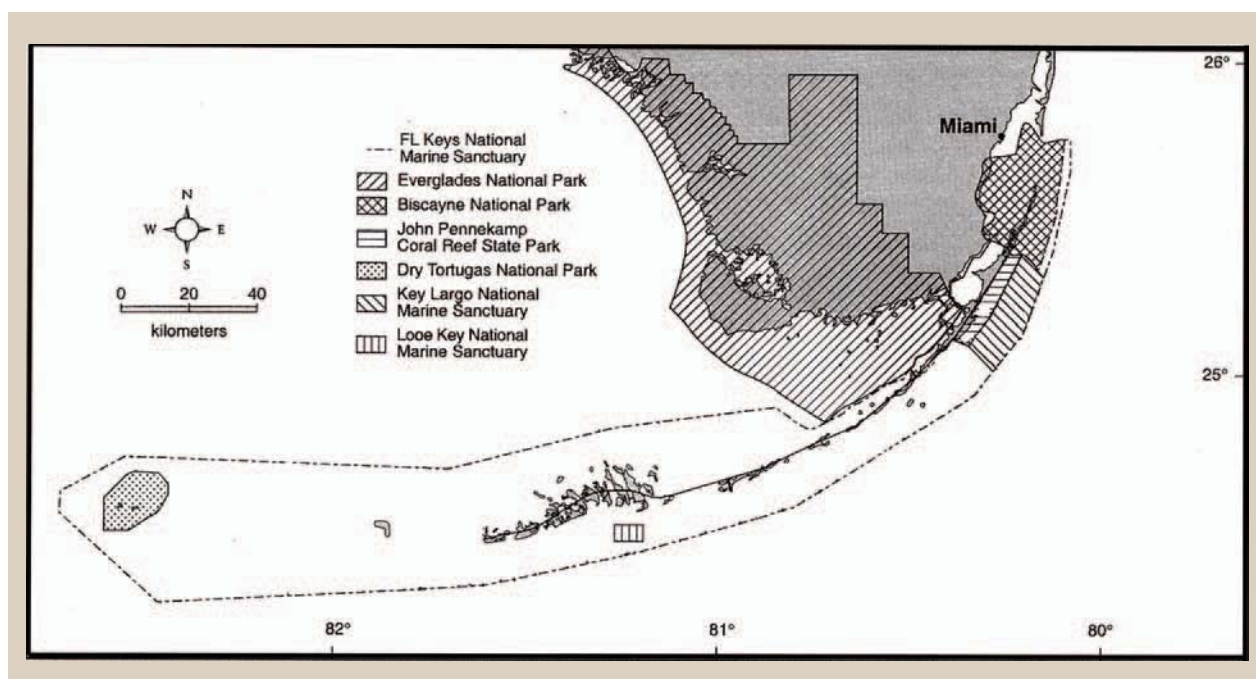


Figure 1. Location map showing south Florida and the Florida Keys including the Dry Tortugas.

Quantitative long-term monitoring in the Dry Tortugas at the ‘other end’ of the Keys revealed coral cover on Bird Key decreased an estimated 20-25 percent between 1975 and 2000. It appears corals in the Dry Tortugas are not stressed to the same degree as in the ‘mainland’ Florida Keys. Dry Tortugas reefs are buffered from Key West by 65 miles of ocean, which may explain why these reefs are somewhat healthier than the Keys reefs.

| Reef Zone | 1974-5 | 1982-3 | 1996 | 2000 | Change |
|-----------|--------|--------|--------|-------|--------|
| Shallow | 42% | 53% | 10.60% | 4.10% | -90% |
| Deep | 62% | 50% | 13.50% | 4.70% | -92% |

Table 1. Percent cover of living coral on Carysfort Reef between 1974 and 2000. (Data from Dustan and Halas 1987, Porter et al. 2001a)

In 1990, Congress created the Florida Keys National Marine Sanctuary and mandated the USEPA institute a water quality assurance and protection plan for the Florida Keys. From 1996 to 2000, the USEPA Coral Monitoring Project documented significant increases in the geographical distribution of coral diseases and the number of species with diseases (Tables 1 and 2, Porter et al. 2001b, Richardson 1998, Patterson et al. 2002).

Coral bleaching also became more common throughout the Keys. Similar disease trends appear in other marine organisms, most notably in the Florida Keys where gorgonians (soft corals), have been infected with *Aspergillus sydowii*, an opportunistic terrestrial soil fungus (Smith et al. 1996).

Case Study

Detection of Ecological Change using Satellite Imagery – A time series of 20 Landsat Thematic Mapper images for Key Largo, taken from 1982 to 1996, revealed changes on Carysfort Reef could be detected from space (Dustan et al. 2001). Changes in image brightness and spectral band ratios suggested a shift from a coral to algal-dominated community structure,

although the trends were not statistically significant. Texture analysis (spatial heterogeneity) generated by habitat patchiness or annual phenology (seasonal change) decreased over time, which was consistent with the

observed loss of corals and subsequent algal overgrowth.

Texture analysis was applied to examine change over time by looking at individual pixels. Each represented the same geographic point on the earth. High values indicate variability, while areas with low temporal texture signify little or no change. In effect, a change in temporal texture is an analysis that can identify hot spots without specific target classification.

When applied to Carysfort Reef, the analysis revealed areas of the reef known to have suffered the greatest ecological change yielded the highest temporal texture. Patterns detected by remote sensing were a clear

| Number of Stations with Disease | | | | | |
|--|-------|------|------|------------------|------------------|
| | WH | BB | OD | Disease Stations | Percent Diseased |
| 1996 | 7 | 7 | 16 | 26 | 16% |
| 1997 | 61 | 11 | 66 | 95 | 59% |
| 1998 | 97 | 28 | 92 | 131 | 82% |
| Increase n (96-98) | 90 | 21 | 76 | 105 | |
| Increase % (96-98) | 1285% | 300% | 475% | 404% | |
| Disease Types: WH=White Plague, BB=Black Band Disease, OD=Other Diseases | | | | | |

Table 2. Distribution of coral diseases by station 1996-1998. Data from USEPA Coral Reef Monitoring Project (Porter et al. 2001b)

diagnosis of known ecological change without having to classify the imagery (Figure 1, Dustan *et al.* 2001). This raises the possibility that measures of variability alone may be diagnostic without needing to fully interpret the spectral information. Furthermore, comparison of patterns over larger geographic areas may be employed in a GIS hunt for degradative forcing functions.

Detecting levels of habitat change without generating a precise classification opens the possibility of using currently orbiting technology to examine large areas of reef tract (or other ecosystems) for change. If hot spots are detected, more directed field studies and higher precision remote sensing from aircraft or 'next generation' high resolution orbiting sensors could be focused on these critical areas to decipher potential causal factors.

Stress at Nested Scales – Stresses to Florida Keys coral reefs reside within a series of nested scales, making it is extremely difficult to determine the individual contribution of each stressor. Growing at the northern limits of reef development implies high levels of naturally occurring stress. Many corals live long, so reef populations are now exposed to amplifications of naturally occurring stress, as well as newer stresses.

The impact of stress accumulates and is probably synergistic. In the Florida Keys, disease is a significant source of colony mortality. Already stressed corals may have weakened immune systems that make them even more susceptible to disease. Coral recruitment and regeneration rates are low.

Global and Mesoscale Stress – The Keys are downstream from almost every sediment or nutrient source in the Caribbean basin and Gulf of Mexico. Materials

wash into the sea from both the west and east coasts of the Florida peninsula as well as through the Everglades. The area of sediment influx extends from the Mississippi River watershed to throughout the Caribbean Sea. Sediments from rivers as distant as the Orinoco and Amazon have been identified on Carysfort Reef (Dustan, unpublished).

In 1993, floodwaters from the US midwest combined with the Loop

Current (which flows into the Florida Current) and reduced both salinity and oxygen concentrations along Keys reefs. On September 14, 1993, decreases in salinity (from 36 to below 32 ppt) and oxygen (from 7.1 to 5.2 ppm) were recorded on Molasses Reef (Porter, personal communication). While the magnitude of the signal is not usually as large, the event clearly showed the hydrologic connectivity between the interior watersheds of North America and the Florida Keys.

Coral populations of the Florida Keys, like reefs everywhere, are also subject to global-scale stresses such as global warming and increased ultraviolet light exposure due to ozone thinning. Coral bleaching has been correlated to increased water temperatures in the late summer and early fall. Particularly serious mass bleaching has occurred in the last 20 years. Most of

| Number of Coral Species with Disease | | | | | |
|--------------------------------------|------|------|------|------------------|------------------|
| | WH | BB | OD | Diseased Species | Percent Diseased |
| 1996 | 3 | 2 | 8 | 11 | 27% |
| 1997 | 22 | 4 | 22 | 28 | 68% |
| 1998 | 28 | 7 | 28 | 35 | 85% |
| Increase n (96-98) | 25 | 5 | 20 | 24 | |
| Increase % (96-98) | 833% | 250% | 188% | 218% | |

Disease Types: WH=White Plague, BB=Black Band Disease, OD=Other Diseases

Table 3. Distribution of coral diseases by species 1996-1998. Data from USEPA Coral Reef Monitoring Project (Porter et al. 2001b)

these events were recorded in many areas of the wider Caribbean and thus have been linked to warming seas.

However, recent work points to another, previously unsuspected ecological stressor: transatlantic African dustfall (Shinn *et al.* 2000). African dust is composed of the major crustal elements including iron, phosphorus, sulfate, aluminum, and silica, which may enhance growth of tropical marine algae, and there is a correlation between increased amounts of dust and various disease outbreaks. Isolated African dust collected in the Carib-

bean contains spores from *Aspergillus sydowii*, a fungus that causes disease in Caribbean gorgonians and is also a potential cause of other synchronous Caribbean-wide coral diseases, including those that killed staghorn coral (*Acropora cervicornis*) and the sea urchin (*Diadema antillarum*). Near extinction of these organisms in 1983 correlates with the period of highest annual dust transport to the Caribbean since measurements began in 1965.

Regional Stress – Humans began manipulating the surface hydrology of South Florida shortly after Florida became a state in 1855. Large-scale drainage and flood control programs drained thousands of acres of swampland for agriculture and constructed hundreds of miles of canals to control the surface flow of water from Lake Okeechobee southward to Biscayne Bay. Massive efforts to channel water flow resulted in excess freshwater, nutrients, and sediments flowing into coastal waters, including Florida Bay (cf. Davis and Ogden 1994). Fertilizers, organic carbon, and urban and commercial dumping further enriched the watershed's effluent as it flowed into the sea. Although concentrations may be diluted, these agents may still affect reef health, as reef ecosystems have evolved to be very efficient in trapping and retaining nutrients even in concentrations that are technically lower than the limit of detection.

Upstream of the Keys, effluent from cities, towns, and farms slowly bleeds into the sea through canals, rivers, and coastal bays. While the distribution of point sources (i.e., sewage outfalls, wastewater disposal by septic systems or deep well injection, or agricultural irrigation canals) is controlled through permitting processes, effluents are not easily traceable once they enter the tropical shallow-water ecosystems. For example, in Key Largo, effluent from a Class 5 shallow-well injection package plants migrates through the Key Largo limestone at rates of 3 to 30 m/d, depending on tides, weather condition, etc. Since this is the preferred

method for waste disposal by developments and hotels, it is a significant pollution threat for the nearshore coastal environment (Reich *et al.* 1999, Cable *et al.* 2002).

Urbanization in the Florida Keys has been intense, with most development centered on Key Largo in the Northern Keys, Marathon in the Middle Keys, and Key West in the South. The human population of the Florida Keys grew tenfold from 1870 to 1990, with about 30 percent of residents living in Key West. Tourism doubles the population during peak times, with two-thirds of the visitors participating in water-related activities. Over 24,000 boats are registered in Monroe County and approximately 14 percent of visitors (roughly 365,000 people) bring boats with them when they visit the Keys.

Localized Stress – Boat groundings and anchor damage have been widespread and have caused severe damage to shallow reefs throughout the Florida Keys (Dustan 1977b, Davis 1977). In 1999, there were 540 small boat groundings documented in the Florida Keys National Marine Sanctuary. There have been at least 5 major ship groundings on reefs since 1985, (Hudson and Diaz, 1988). Installation of hundreds of mooring buoys throughout the Keys has greatly reduced anchor

| Year | Protection Event or Scientific Finding |
|------|---|
| 1935 | Dry Tortugas National Monument by President Roosevelt |
| 1960 | John Pennekamp Coral Reef State Park established December 10, 1960 |
| 1968 | Biscayne National Park established |
| 1969 | Skin Diver Magazine sounds alarm on reef degradation (Barada 1975) |
| 1972 | Marine Protection, Research and Sanctuaries Act |
| 1973 | Coral Diseases discovered in Key Largo (Antonius 1974) |
| 1974 | Beginning of long term reef monitoring at Carysfort Reef (Dustan 1977b) |
| 1975 | Key Largo National Marine Sanctuary established |
| 1981 | Looe Key National Marine Sanctuary established |
| 1983 | Significant reduction of corals at Carysfort Reef since 1974 |
| 1984 | Key Largo Coral Vitality Study ~60% corals stressed (Dustan 1993) |
| 1987 | Severe coral bleaching throughout Caribbean |
| 1991 | Florida Keys National Marine Sanctuary and Protection Act 1990 |
| 1992 | Reef coral degradation estimated at ~5% loss/yr. (Porter and Meier 1992) |
| 1995 | USEPA Water Quality Protection Plan Coral Reef Monitoring project, |
| 1997 | Outbreak of White Band Disease 2. pathogen identified (Richardson 1998) |
| 1998 | President Clinton issued Executive Order 13089 on Coral Reef Protection. First US Coral Reef Task Force Meeting at Biscayne National Park. |
| 1999 | Carysfort Reef living coral coverage below 5% |
| 2000 | Proposed Tortugas 2000 National Marine Sanctuary |
| 2001 | USEPA Coral Monitoring reports 38% loss of living coral 1996-2000 |
| 2002 | Florida Keys' Particularly Sensitive Sea Area shipping restrictions declared |

Table 4. Florida Keys Coral Reef Ecosystem Timeline outlining significant political protection events and ecological findings.

damage on reefs and may serve as a means to regulate access to reef areas.

Starting in 1982, there was a mass mortality of *Diadema antillarum* throughout the Caribbean Sea. Populations have still not recovered to anywhere near original levels (Lessios, 1988). Following the mass mortality, macro algal populations throughout the Caribbean soared as they were released from herbivore pressure. So while nutrient levels were apparently rising from increased urbanization, levels of herbivory on the reef plummeted. At present, these two scenarios cannot be uncoupled and there remains a hot debate on which stressor is ecologically important (see Lapointe 1999, Hughes and Connell 1999).

The Combination – In summary, increased sedimentation smothers coral tissue; increased nutrients and a dramatic reduction in herbivory has resulted in algal overgrowth; and elevated temperatures promote bleaching and disease. Reefs near population centers have the increased pressures from sewage, runoff, garbage dumping, and greatly increased levels of harvesting. Coral diseases, first discovered in the Keys in the 1970s, have increased over 400 percent since 1996 (Porter *et al.* 2001b).

Protective Measures – The Florida Keys were placed under state and Federal protection beginning with the creation of the Dry Tortugas National Monument in 1935 by President Franklin Roosevelt. The rest of the Keys were protected through the Florida Keys National Marine Sanctuary Protection Act of 1990 (Table 4).

Presently, the Florida Keys National Sanctuary (FKNMS) uses the concept of multi-use zoning for varying levels of resource protection. Mooring buoys have been installed throughout the Keys to protect reefs from anchor damage and sanctuary officers patrol the waters. To reduce large vessel groundings, a series of radar beacons have been placed on outer reef lighthouses and the area has been declared a Particularly Sensitive Area to help warn commercial shipping traffic of the reefs. Less than one percent of the FKNMS has been designated as no take zones and appropriate sewage treatment and issues surrounding non-point pollution have yet to be adequately addressed.

Forecast – Concerns about reef degradation have been published in the popular literature since 1969 and clearly expressed in the scientific literature since 1977 (Dustan, 1977b, 2000, Dustan and Halas 1987, LaPointe, 1997, 1999, Porter and Meier 1992, Porter

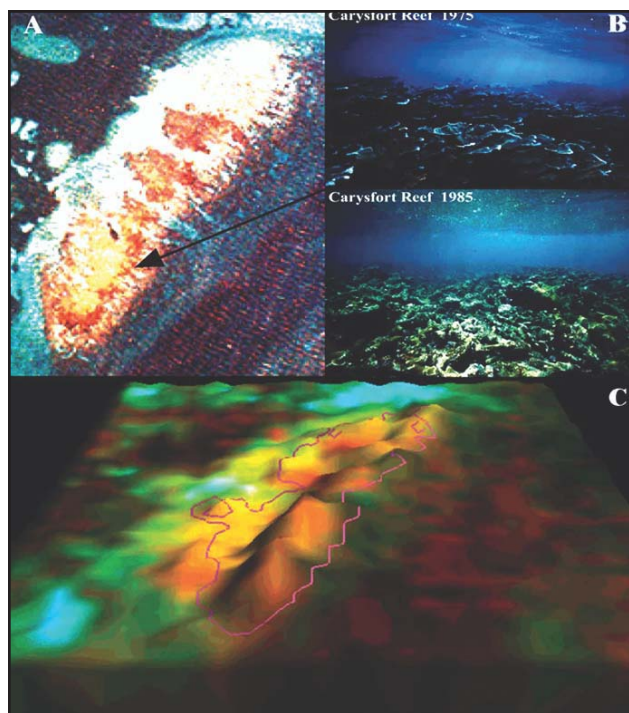


Figure 2, Carysfort Reef, Key Largo, FL. This composite of remote sensing and underwater photography reveals changes that occurred between 1975 and 2000, when the reef lost over 90 percent of its living coral cover. A) NAPP high altitude aerial photograph showing general reef features. Shallow areas (bright yellow and brown) are reef flat, and the seaward edge of the shallow reef. The lighthouse and its shadow can be seen above and to the left of the arrow tip. The arrow points to the *A. palmata* zone which can be seen underwater in B, a pair of photographs taken from the same vantage point ten years apart. C) Three-dimensional visualization of reef change with temporal texture as the vertical axis. Colors are true colors from TM image. Current living coral cover is below 5 percent (Dustan *et al.* 2001).

et al. 2001a). Furthermore, these studies suggest that the rate of loss of living coral tissue may have accelerated in the last 25 years (Figure 2).

Reef corals are slow growing, colonial organisms and clearly, at the present rates of loss, Florida Keys reefs will not be able to sustain their ecological integrity. However, because there are multiple stressors and they are nested, managers have no clear solutions and are reluctant to apply conservation measures beyond fisheries regulations, enforcement, and permitting.

The Florida Keys do not have tertiary sewage treatment, and much of the region is without sewage treatment beyond septic tanks. These contaminate canals and coastal waters due to the porosity of the limestone rock of the Keys. The USEPA Water Quality Assurance Program is an attempt to collect long-term ecosystem data on water quality, seagrass beds, and coral reefs, which will be used to drive ecosystem management.

Since the inception of the Program in 1995, the reefs have lost at least 38 percent of their living coral cover and little has been done to ameliorate the situation. The US Coral Reef Task Force, originated by Presidential Order 13089, has produced an Action Plan for Coral Reef Conservation, but the plan has not been implemented. Ecological forecasts of catastrophic reef decline have been accurate, but policy-makers and governments have not responded with appropriate actions.

Next Steps

Reefs of the Florida Keys will take hundreds of years to regrow, once their environment has been restored. Reefs have interconnected symbioses and trophic interactions because they evolved in energy-rich, nutrient-poor waters. These very same adaptations make reefs vulnerable to degradation. Ironically, the many values of coral reefs – as a fisheries resource, for coastal protection and building materials, and as tourist attractions – now are contributing to their steady and rapid decline.

As clearly as observational science has demonstrated this, we still lack knowledge concerning the specific factors involved and the magnitude of their relative contributions. This will require detailed long-term ecological monitoring of marine and terrestrial communities and physical processes. These tasks are greatly complicated by the geography of the Keys which places them at the convergence of the South Florida hydro-scape and waters exiting the Caribbean Sea and Gulf of Mexico through the Straits of Florida.

The capability to partition the stress will require a strong commitment to basic ecological research on understanding underlying processes and ecosystem response to change, and information modeling. The research and management communities must increase their levels of collaboration and cooperation to identify knowledge gaps and support decision-making processes.

Conclusion

The decline of the Florida Keys coral reefs has been driven by a wide variety of diffuse stressors applied over a wide range of multiple scales. There is no one major cause, and so there is no magic ‘silver bullet’ to correct the situation. Federal and state efforts have been aimed at regulating harvesting and physical destruction while water quality issues encompassing sewage

treatment and watershed runoff have been largely ignored owing to the magnitude of the problems.

Future research needs to be targeted at nested scale process studies. At the same time, application of conservation methods that do work cannot be delayed.

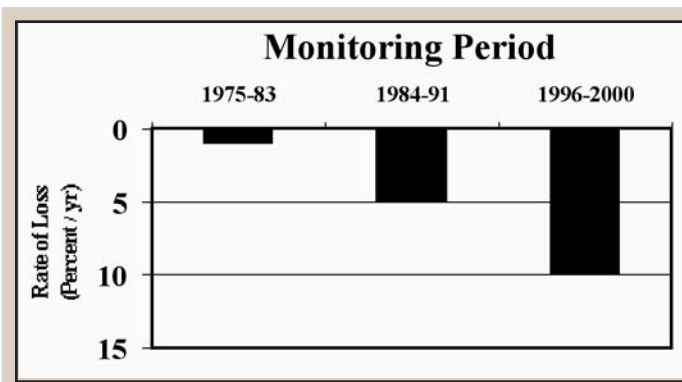


Figure 3. Comparison of measured rates of the loss of living coral cover in the Florida Keys 1974-2000. Data from Dustan and Halas, 1987, Porter and Meier, 1992, and Porter et. al, 2001a).

Measures such as marine protected no-take zones, tertiary sewage, and other aspects of watershed conservation can and should be implemented immediately, or it will be said that the coral reefs of the Florida Keys disappeared on our watch, while we were watching.

Acknowledgements

Discussions with numerous colleagues helped me formulate this communication including Gene Shinn, Rick Schwabacher, Ouida Meier, John Fauth, Jim Porter, Dennis Slice, John and Judy Halas, and Vladimir Kosmynin. Portions of this research were funded by USEPA, NOAA, and the Jeanette and Lafayette Montgomery Foundation.

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Nowcasting Sea Nettle Distributions in the Chesapeake Bay: An Overview

C. W. Brown¹, R. R. Hood², T. F. Gross³, Z. Li^{1,3}, M. B. Decker⁴, J. E. Purcell⁵, and H. V. Wang⁶

Abstract

Sea nettles (Chrysaora quinquecirrha), a stinging jellyfish, become abundant every summer in Chesapeake Bay and interfere with recreational and commercial activities. The adverse impact of their occurrence may be mitigated if their presence could be monitored and predicted. We have developed a procedure that produces weekly maps or 'nowcasts' of the likelihood of encountering sea nettles in the bay by flagging the geographic locations where ambient conditions coincide with their preferred salinity and temperature ranges. Estimates of near-real time temperature and salinity are derived from a hydrodynamic model of the Chesapeake Bay and satellite imagery, and the empirical habitat model relating these two environmental variables to sea nettle presence was developed using in-situ data. These maps, as well as general information about sea nettles, are staged on a web site (<http://coastwatch.noaa.gov/seanettles>) for dissemination. This project represents an initial step to develop an approach that combines data derived from disparate sources, such as numerical circulation models, operational satellites, and moored sensor systems, in order to generate nowcasts and forecasts of the distribution pattern of marine biota.

Introduction

It has long been recognized that a certain combination of physical, chemical, and biotic factors may lead to the development and persistence of biological events. For example, a specific algal bloom may form if a 'seed' population exists in a water column when environmental conditions, such as light level and nutrient concentrations, become favorable for that species. Accordingly, if the abiotic and biotic conditions necessary for the growth and maintenance of an organism, i.e., its habitat, are adequately defined, one should be able to predict the potential occurrence of an organism if one could predict the relevant conditions. This is the underlying basis for 'gap analysis', a technique employed by the terrestrial and aquatic research communities to estimate the potential distribution of a species by locating the geographic regions where the environmental conditions match the

habitat requirements of the species (e.g., Bushing, 1997; Jennings, 2000; Zacharias and Howes, 1998).

Recent advances in technology and telecommunications permit us to monitor numerous environmental factors in real- or near-real time and some can even be forecasted; data can be retrieved from geographic databases, derived from satellites, accessed from *in-situ* deployed instruments, and simulated by numerical models. Although we still have a considerable way to go to predict biotic events in the future, the current capability of retrieving and simulating environmental information with little or no time delay offers the potential to locate and predict the presence of organisms in near-real time, i.e., generate a 'nowcast' of the species abundance in time and space, provided that their habitat can be defined. This approach is similar to that of gap analysis, except near-real time data are employed. To

¹ NOAA / NESDIS Office of Research & Applications, Camp Springs, MD

² Horn Point Laboratory, University of Maryland Center of Environmental Science, Cambridge, MD

³ NOAA / NOS Coast Survey Development Laboratory, Silver Spring, MD

⁴ Department of Ecology and Evolutionary Biology, Yale University, New Haven, CT

⁵ Shannon Point Marine Center, Western Washington University, Anacortes, WA

⁶ Virginia Institute of Marine Science, College of William and Mary, Gloucester Point, VA

evaluate the viability of this ecological approach, we have developed a prototype system to nowcast the distribution of sea nettles (*C. quinquecirrha*), a species of stinging jellyfish, in the Chesapeake Bay.

Case Study

Sea nettles (*C. quinquecirrha*), a species of stinging scyphomedusan jellyfish, become abundant every summer in the Chesapeake Bay. From April through early summer, when temperature and salinity are favorable, the bottom-dwelling polyps bud off or strobilate into immature medusae, called ephyrae, which measure approximately 0.1 cm in diameter (Figure 1). The ephyrae grow rapidly into medusae, the adult form typically called jellyfish. To complete their life cycle, the adult nettles shed their gametes into the water and their fertilized eggs form larvae that attach to hard surfaces such as oyster shells, where they grow into tiny polyps that develop tentacles. The polyps over winter in a dormant state attached to the bottom (Bryant and Pennock 1998; Cargo and Schultz 1966; Cargo and Schultz 1967).

(Baird and Ulanowicz 1989) and may impact finfish production in the Bay (Purcell and Arai 2000). The adverse impact of sea nettles may be mitigated if their presence can be monitored and predicted in near real time.

We have developed a procedure to nowcast the distribution pattern of *C. quinquecirrha* in the Chesapeake Bay by exploiting our knowledge of their temperature and salinity preferences and our ability to acquire these two environmental variables in near-real time. Specifically, a habitat model relating sea nettle presence to sea-surface temperature and salinity is applied to estimates of ambient temperature and salinity derived from a hydrodynamic model of the Chesapeake Bay and satellite imagery. Weekly maps illustrating the likelihood of encountering sea nettles are generated by flagging the geographic locations where ambient conditions coincide with their preferred salinity and temperature ranges. The most current nowcast, as well as information describing sea nettle life history and the procedure used in creating the maps, are staged on a web site (NOAA 2003) for dissemination to the public and interested agencies.

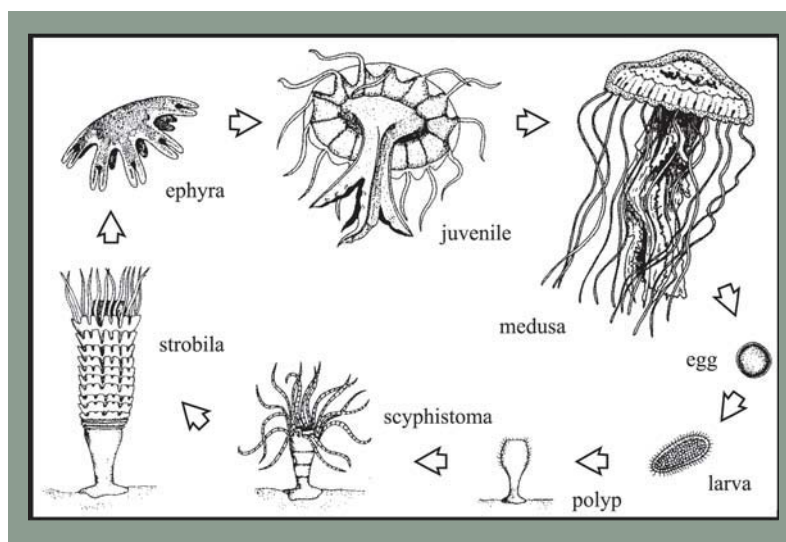


Figure 1. Life cycle of sea nettles, *Chrysaora quinquecirrha*. From T.L Bryant and J.R. Pennock 1998.

Sea nettles are biologically important in Chesapeake Bay and they interfere with recreational activities. Their presence deters swimming and other water activities because contact with their stinging tentacles is painful and may induce allergic reactions. Sea nettles are also voracious predators, devouring copepods, fish eggs and larvae, and comb jellies. This predation affects food web dynamics and energy flow at several trophic levels

Salinity – The environmental variables used in the nettle nowcast system are derived from existing data sources and a modified version of a well-validated hydrodynamic model. The distribution of surface salinity in the Chesapeake Bay is simulated by using a near-real time version of the Curvilinear Hydrodynamics in 3 Dimensions (CH3D) model. CH3D was originally developed by Sheng (1987) and later extensively modified at the US Army Corps of Engineers Waterways Experiment Station for the Chesapeake Bay (Johnson *et al.* 1991, Wang and Johnson 2002). It is a curvilinear, finite difference Z-coordinate model and is currently used, in combination with a water quality model, by the EPA Chesapeake Bay Program to

study the potential benefits of proposed nutrient loading reductions. Our version of CH3D was modified to run using near-real time forcing and is unique in that it allows us to simulate ambient salinity in an estuary. The error in the simulated salinity is estimated from coincident *in-situ* salinity measurements to be 2-3 in the Practical Salinity Scale (PSS) and it tends to be greater in the tributaries than the mainstem of the Bay.

Sea-Surface Temperature – The sea-surface temperature (SST) can be estimated by using either our near-real time version of the CH3D hydrodynamic model or derived from data acquired from the NOAA Advanced Very High Resolution Radiometer (AVHRR) satellites. We currently generate nowcasts of sea nettle distributions using model generated SST with an estimated error of 2-3 °C (Li *et al.* 2002).

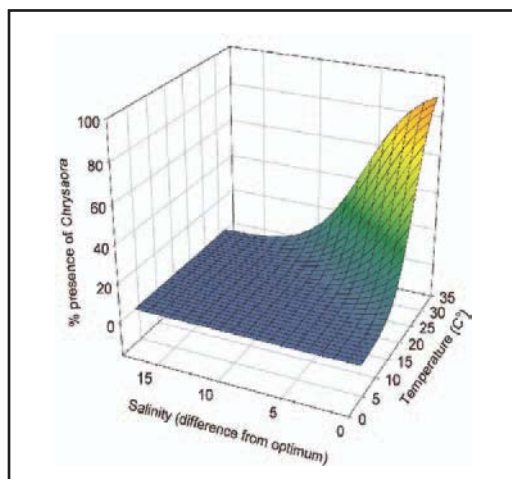


Figure 2. Graphical representation of the logistical equation describing the probability of encountering sea nettles, *Chrysaora quinquecirrha*, in the surface waters of the Chesapeake Bay as a function of salinity and sea-surface temperature.

Although not yet fully implemented, we are developing the ability to utilize more accurate SST of Chesapeake Bay derived from AVHRR data processed and distributed by the NOAA CoastWatch Program. AVHRR data are acquired with a wide-field scanning system that enables one to two images per day with a ground resolution of 1.1 km at nadir. A weekly composite of SST is formed and georeferenced to the CH3D derived salinities using a Geographic Information System. The composite is created by selecting for maximum temperature values in daily images estimated using the NOAA N(on) L(inear) SST daytime and nighttime split-window algorithm to eliminate clouds. Coast Watch NLSST from this region possesses a positive bias of up to 0.4 °C and standard deviation of 1.0 °C (Li *et al.* 2001).

Habitat Model – Though salinity and temperature were known to be important physical factors influencing the presence of *Chrysaora* in the Chesapeake Bay, a quantitative relationship between them on a Bay-side scale did not exist. We developed a bay-wide model to predict the probability of encountering sea nettles and their density in surface waters of the Chesapeake Bay

by analyzing coincident data of *Chrysaora* density, salinity and sea-surface temperature collected during the spring, summer and fall of 1987-2000 in surface waters (0-10 m) of the bay and selected tributaries (n = 1064). *In-situ* nettle density was measured using a 1-m², 280 mm mesh Tucker trawl with an attached flow-through meter.

Concentrations of *Chrysaora* were found within a relatively narrow, well-defined range of temperature and salinity (26-30 °C and 10-16 PSU). The *in-situ* data were used to derive an empirical (“logit”) model that estimates the likelihood of sea nettle encounter (p_{nettle}) as a function of salinity and temperature (Figure 2):

$$p_{nettle} = e^{\text{logit}} / [e^{\text{logit}} + 1],$$

where $\text{logit} = -6.995 + (0.30 * \text{temperature}) - (0.469 * \text{salinity}_{opt})$, temperature is in degrees celsius, and $\text{salinity}_{opt} = 13.5$ PSS.

A Type 1 Long-likelihood Ratio Test with 2 degrees of freedom comparing the full model (including temperature and salinity) to a reduced model with neither of the factors indicated temperature and salinity have a significant effect on the observed frequency of encountering sea nettles (LR Test statistic = 475.8, $p < 0.001$).

Validation Activities – Validation of nettle nowcasts has been initiated but remains rudimentary at this stage; our preliminary nowcasts began in May 2001. Comparison of retrospective *Chrysaora* ‘nowcasts’ (Figure 3A, 3B) with coincident *in-situ* measurements (Figure 3D, 3E) indicate our model predictions performed well on a bay wide scale, replicating the relative downstream-upstream position of nettles in the Chesapeake Bay during the summer of 1996 and 1999. On a finer scale, however, the nowcasts fail to predict lateral variations in sea nettle distributions in the mainstem of the Bay, which suggests that other factors, such as wind and surface convergences, strongly influence distributions of nettles in surface waters.

Lessons Learned

Validation is essential in establishing the accuracy of the generated nowcasts. Paramount in this verification activity is the availability of accurate *in-situ* observations for comparison against the model predictions. Accurate observations from throughout the Chesapeake Bay, however, have proven difficult to acquire. More effort should have been devoted to this aspect of the project from the beginning.

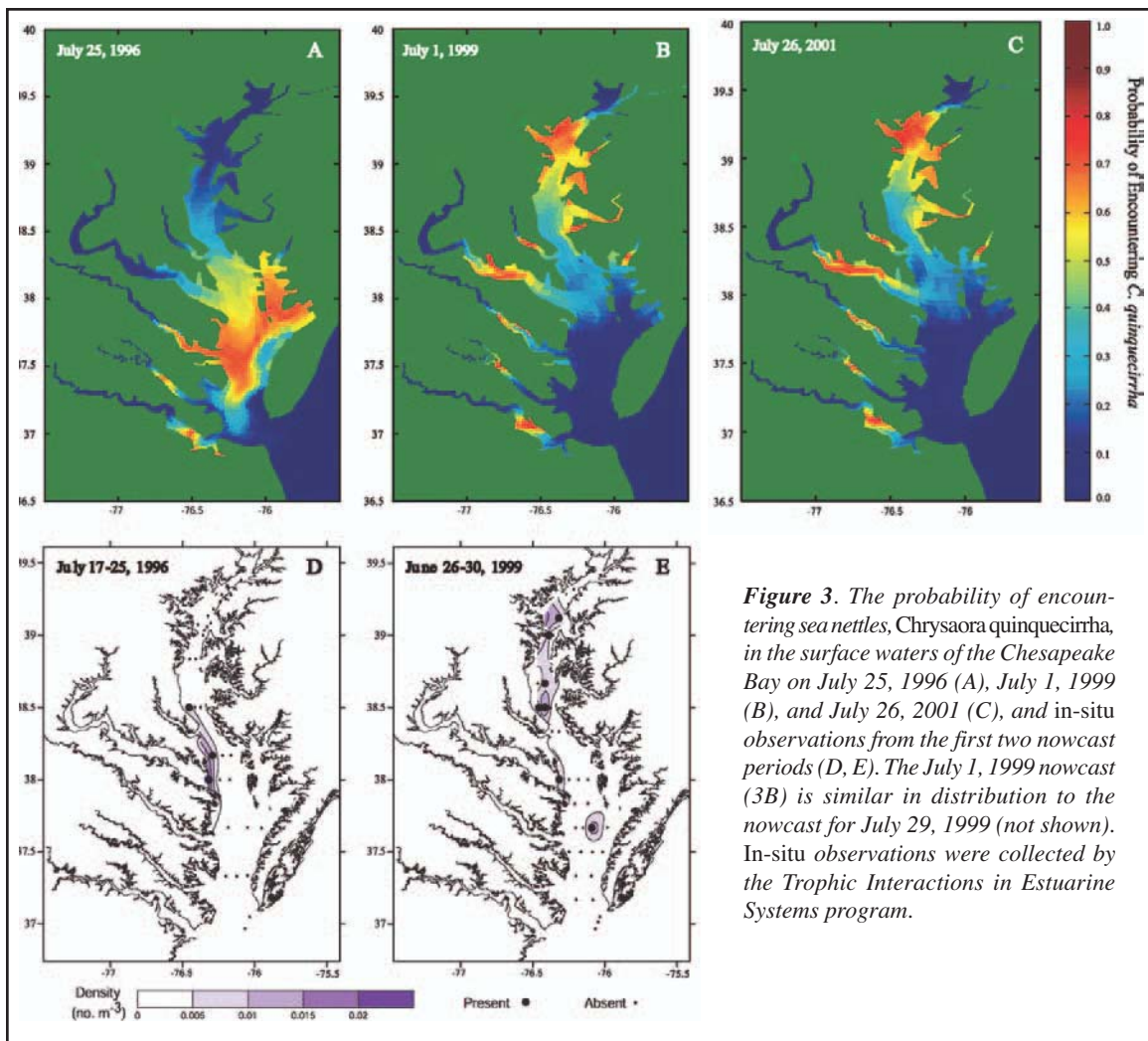


Figure 3. The probability of encountering sea nettles, *Chrysaora quinquecirrha*, in the surface waters of the Chesapeake Bay on July 25, 1996 (A), July 1, 1999 (B), and July 26, 2001 (C), and in-situ observations from the first two nowcast periods (D, E). The July 1, 1999 nowcast (3B) is similar in distribution to the nowcast for July 29, 1999 (not shown). In-situ observations were collected by the Trophic Interactions in Estuarine Systems program.

Next Steps

We will continue to evaluate the accuracy of nowcasts by comparing them against additional *in-situ* observations of *C. quinquecirrha* collected at locations throughout the Bay and obtained from numerous sources, such as the Alliance for Chesapeake Bay. We also plan to analytically estimate the total error of the nowcasting procedure and the relative influence of its individual components by performing a Monte Carlo simulation and other appropriate analyses. Future advances in modeling salinity (and temperature) in the Bay, particularly in the tributaries, should considerably improve our nowcasting ability.

Once the sources of error in the model are identified and quantified, the model can also be used to better understand the spatial and temporal variability of this currently under-sampled species. For example, the inter-annual variability of sea nettle distributions throughout the Chesapeake Bay can be examined by hindcasting the salinity and temperature fields with CH3D and

applying our sea nettle habitat model as illustrated for similar dates during July 1996, 1999, and 2001 in Figure 3 (3A, 3B, 3C). In 1996, an extremely wet year, our model predicts that nettles should have been located principally in the lower reaches of the Bay's mainstem. While in 1999 and 2001, more typical years for rainfall, our model predicts that nettles were positioned further upstream in both the mainstem and several tributaries.

Conclusion

This project represents an initial step towards developing an approach that combines near-real time data derived from disparate sources, such as numerical circulation models, operational satellites, and moored sensor systems, to generate nowcasts and forecasts of the distribution patterns of marine biota. Sea nettles provide an ideal prototype because their distribution is tightly constrained by two physical factors (temperature and salinity) that are relatively easy to measure and predict in near-real time.

In theory, this approach can be applied to any organism, but many obstacles and challenges remain. For example, the habitat of other target species, such as harmful algae, will likely be much more difficult to define than that of sea nettles. Moreover, we cannot routinely measure and estimate key chemical and biological factors, such as some nutrient concentrations and food availability, which will be needed to define the habitat of many species. Nonetheless, we believe the approach described here, in conjunction with new technological capabilities, is a powerful tool that holds great promise for predicting the distribution pattern of organisms in the ocean.

Acknowledgements

Funding has been provided by a NOAA Ocean Remote Sensing grant to C.W.B. and a Maryland Sea Grant Project Development grant to R.R.H.

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Forecasting Harmful Algal Blooms in the Gulf of Mexico

R.P. Stumpf¹ and M.A. Culver²

Abstract

To respond to harmful algal blooms (HABs) in the Gulf of Mexico, managers require four types of forecasts: 1) monitoring the extent of identified HABs, 2) distinguishing new blooms as HAB or non-HAB, 3) forecasting the location of a HAB since the last known position; and 4) predicting conditions favorable for initiation of a new HAB. Beginning in 1999, we have provided the Gulf states with bulletins with the objective of providing these types of forecasts. The bulletins started with satellite ocean color imagery from SeaWiFS/OrbView-II showing chlorophyll in response to HAB reports from the states (type 1). In 2000, wind data from coastal meteorology and forecasts from the National Weather Service were added to indicate direction of bloom movement (type 3). HAB detection algorithms (type 2) have been incorporated into the analysis in 2001, and added to the bulletins in 2002. The next addition will use models to predict events (type 4). The bulletins have provided early warning of some events and provided guidance to sampling of blooms when they have spread. The results help the understanding of the distribution of these blooms, particularly along the Florida coast.

Introduction

Blooms of the toxic dinoflagellate *Karenia brevis* have several significant impacts on public health, wildlife, and tourism in the Gulf of Mexico (Tester and Steidinger, 1997). The organism is the cause of neurotoxic shellfish poisoning (NSP) in shellfish, leading to closure of shellfish beds at 5 cells ml⁻¹. At medium to high concentrations (> 100 cells ml⁻¹), the organism can cause fish kills and lead to death of marine mammals, including the endangered manatee. Airborne toxins can cause human respiratory distress as well. Because the blooms reappear nearly every year in the Gulf, routine monitoring and prediction are essential for public health and for anticipating wildlife response.

Case Study

Management response to harmful algal blooms (HABs) of *Karenia brevis* in the Gulf of Mexico requires four types of forecasts: 1) monitoring movement of previously identified HABs; 2) distinguishing new blooms as HAB or non-HAB; 3) forecasting the location of a HAB since the last known position; and 4) predicting conditions favorable for initiation of a new HAB.

Beginning in 1999, bulletins have provided these types of forecast information to the states (Stumpf *et al.* 2003).

The bulletins are based on the integration of several data sources: ocean color imagery from the SeaWiFS-OrbView-II satellite purchased and processed for coastal waters by NOAA CoastWatch; wind data from coastal meteorological stations; field observations of bloom location and intensity provided by the states of Florida and Texas; and forecasts from the National Weather Service. The bulletins began with bloom monitoring (type 1) and limited advisories on transport (type 3).

In autumn 2000, the bulletins had improved forecasts and included the first attempts to predict conditions favorable for bloom development in Florida (type 4). In 2002, features considered to be potential HABs were flagged.

Figure 1 shows an example bulletin from September 2001. The bulletin includes several parts: 1) an image of chlorophyll concentration for the Gulf of Mexico with chlorophyll processed with a Gulf of Mexico algorithm (Stumpf *et al.* 2000) (in the example bulletin, the bloom can be found where chlorophyll is >2 µg L⁻¹ in the box), 2) a plot of wind from an appropriate loca-

¹ NOAA National Ocean Service, Center for Coastal Monitoring and Assessment, Silver Spring, MD, USA

² NOAA Coastal Services Center, Charleston, SC, USA

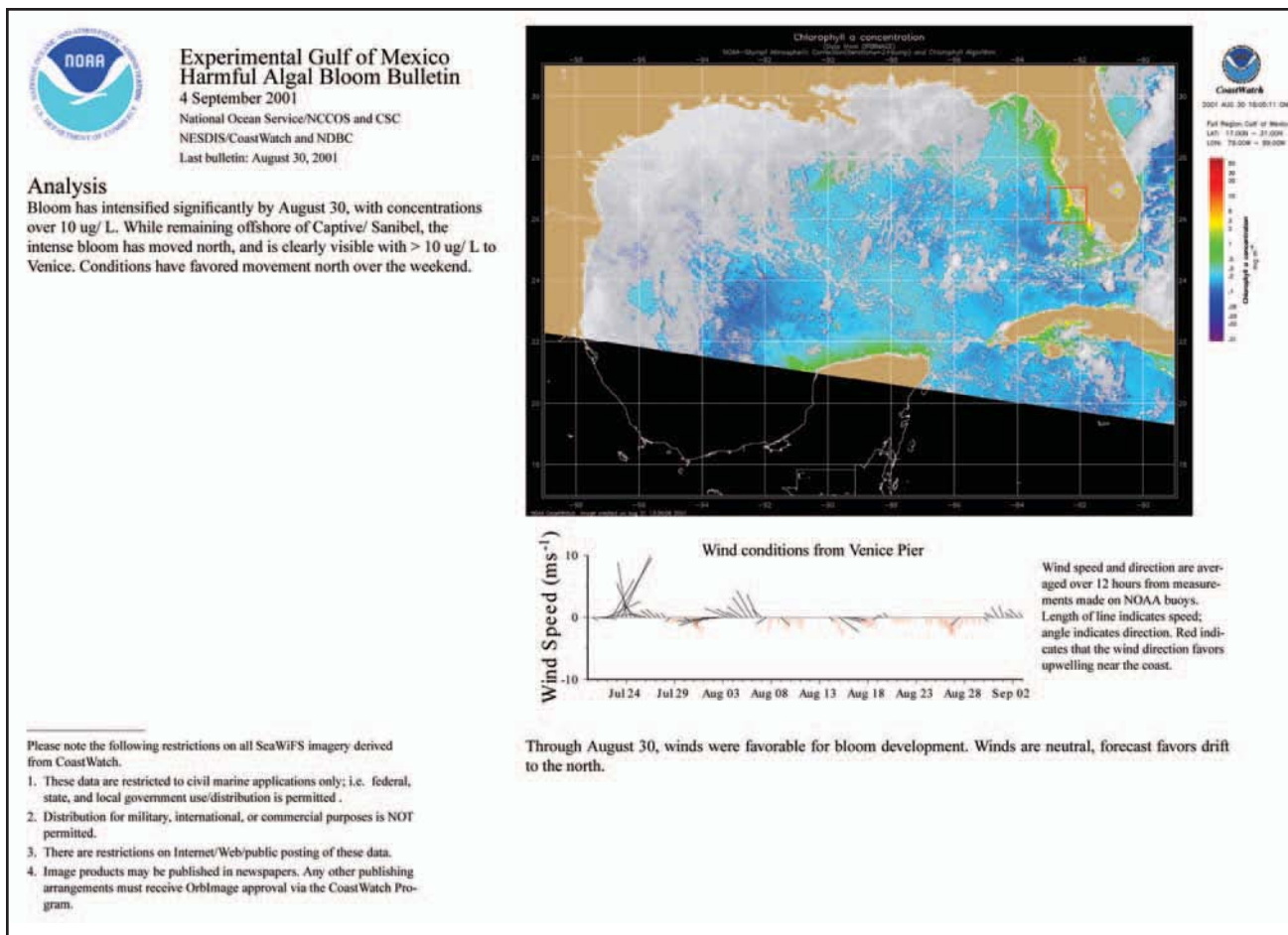


Figure 1. Example HAB forecasting report for the Gulf of Mexico. The image shows chlorophyll concentrations in the Gulf, and the wind speed and the discussion below it indicate whether conditions are favorable for developing a bloom. This is summarized in the discussion on the left.

tion (in the example, Venice Pier is at the coast where the top of the box intersects the coast), 3) discussion of relevant wind conditions (below the wind plots), and 4) interpretation of conditions (at the upper left).

Type 1 Forecasts: Monitoring Known Blooms – are conducted primarily through observations of chlorophyll concentration each 100 cell ml^{-1} of *K. brevis* has approximately $1 \text{ }^{\circ}\text{g L}^{-1}$ of chlorophyll (Tester and Steidinger 1997). With background chlorophyll along the shelf generally $<1 \text{ mg}$, some estimates of concentration and bloom edge can be made through the chlorophyll imagery (see chlorophyll image in Figure 1).

Type 2: Identifying Where High Chlorophyll Concentrations May Be HABs – involves a new method proposed by Stumpf (2001) from work done by Thomas (2000). The bloom is identified as an anomaly in the chlorophyll field (Figure 2). This capability also has potential for monitoring the blooms, and has been implementing in bulletins starting in late 2002.

Type 3: Forecasting Transport – draws on imagery and field observations together with observed and predicted winds to forecast along-coast movement. Tester *et al.* (1991) showed that wind plays a significant role in redistribution of *K. brevis* blooms. Assessing the along-coast wind transport can provide the forecast of the current position from the last known position.

Type 4: Predicting Future Blooms – draws on climatologic information and current winds. Stumpf *et al.* (1998) found HABs were correlated with upwelling-favorable winds. The late summer and autumn winds are monitored for these conditions and are noted in the bulletins.

The bulletins have provided information on the extent of blooms, particularly offshore areas where sampling is problematic. The state of Florida was correctly advised of the potential for a bloom to occur at the end of September, 2001 (forecasting type 4) and the state was alerted to the position of previously unknown blooms

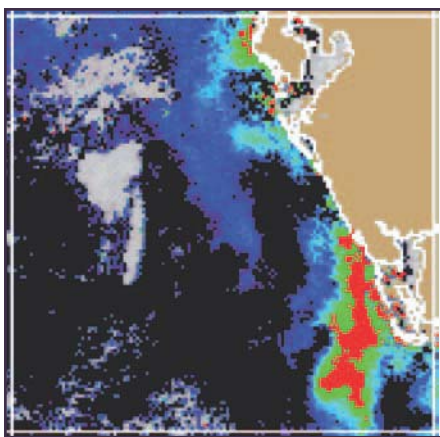


Figure 2. Identifying a HAB from an anomaly in the chlorophyll field. The red area is a probable bloom.

in January 2000 and October 2000 (forecasting type 2). Monitoring has continued, including estimates of direction and distance of transport. The combination of warning and rapid detection is a significant aid to the states in responding to these blooms.

Lessons Learned

The research aspects have developed in response to the need for information. Results and observations from monitoring blooms have been interpreted to develop the new techniques. From the beginning, it was known this program would be successful only with clear and direct communication with those locally responsible for field monitoring programs. Direct communication assured the necessary information was available, and the field data essential to analysis and interpretation of imagery was obtained promptly. Analyzing data during blooms produces insights that lead to improved forecasts. For example, the HAB detection algorithm was developed from interpretations of the imagery made during blooms in 1999.

Forecasting HABs is analogous to hurricane forecasting. For hurricane forecasts, the models and synthesized data give reasonable results for movement or strength. But ultimately a forecaster must provide guidance to the users on what results should be considered. Models used in HAB forecasting will be improved and expanded, reducing the amount of interpretation required. But some expert interpretation will always be needed because of the complexity of HABs and the data sets used. Training analysts to interpret HAB data should not be difficult, and bulletins do not need to be issued more than 2-3 times per week. This reduces the number of analysts needed.

Conclusion/Next Steps

The HAB bulletins provide additional information to the Gulf states, particularly Florida, for monitoring HABs. Evaluation of events along the Texas coast is needed, as well as evaluation of existing models and new analytical capabilities for incorporation into the bulletins. In addition, a strategy for training analysts must be developed, so the bulletins can move from experimental (although near-real time) to a demonstration phase, and ultimately to an operational program.

Acknowledgments

Imagery is processed by the NOAA CoastWatch program, headed by Kent Hughes. Personnel implementing various aspects of the processing are: Michael Soracco, Ramesh Sinha, Heng Gu, and Varis Ransibrahmanakul. The HAB-Bulletins are generated through semi-automated procedures by efforts of Kirk Waters and Andrew Meredith at NOAA Coastal Services Center. Communication with Earnest Truby of the Florida Marine Research Institute is essential to the value of the bulletins in Florida.

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The Challenges of Forecasting and Managing Toxigenic *Pseudo-nitzschia* Blooms on the US West Coast

Vera L. Trainer¹ and Barbara M. Hickey²

Abstract

The transfer of the toxin, domoic acid, to shellfish, crustaceans, seabirds, finfish and marine mammals has been recently documented on the US west coast. Data collected during west coast cruises in 1997-2001 indicate that often the highest toxin levels and greatest numbers of toxic cells are positioned in water masses associated with offshore eddies or in upwelling zones near coastal promontories. Such cruise data are essential in the characterization of offshore initiation sites that will lead to the effective placement of automated sensors such as moored arrays.

*In addition, beach monitoring is a necessary component of regional species characterization, resulting in the development of specific molecular and biochemical tools needed to assist managers in each coastal area. Indeed, beach samples collected in 1998 indicated that a *P. pseudodelicatissima* bloom was responsible for razor clam toxicity on the Washington coast, whereas toxin produced by *P. australis* resulted in sea lion mortalities in central California. Complete characterization of physical, biological and chemical conditions that favor harmful *Pseudo-nitzschia* blooms, only possible through large-scale, synergistic collaboration, is a prerequisite for forecasting these events. A forecasting capability will substantially improve the management of valuable coastal resources and the protection of human health, both of which are affected by these toxins.*

Introduction

While domoic acid (DA) poisoning was first recognized in an outbreak on Prince Edward Island, Canada (Wright *et al.* 1989), most of the known toxic events since then have occurred on the US west coast. Domoic acid, a neurotoxin produced by species of the genus *Pseudo-nitzschia* (PN), was first implicated in the illness and death of brown pelicans and Brandt's cormorants in Monterey Bay, California in 1991 (Work *et al.* 1993). About one month following the toxic bloom in California, levels of DA above the regulatory limit of 20 µg/g in shellfish tissue were found in edible parts of razor clams (*Siliqua patula*) and Dungeness crabs (*Cancer magister*) on the Washington coast (Wekell *et al.* 1994). In 1998, impacts of DA to the health of marine life and to the fisheries economy were documented in several regions along the west coast. In particular, California sea lions (*Zalophus californianus*) in central California were severely affected by DA poisoning (Scholin *et al.* 2000)

and high levels of toxin in razor clams in Oregon and Washington resulted in beach closures for more than a year and a half (Adams *et al.* 2000, Trainer *et al.* 2001).

Beach and harvest closures resulting from the toxigenic PN blooms have a severe impact on both coastal economies and on tribal communities. In 1991, the closure of Washington State beaches to recreational and commercial shellfish harvesting resulted in a \$15-20 million revenue loss to local fishing communities (Horner and Postel 1993, Anderson 1995). The commercial Dungeness crab industry on which Washington's Quileute tribe depends for employment lost 50% of their income in 1998 due to harvest closures (Trainer and Wekell 2000). The entire razor clam harvest of the Quinault tribe, on which they depend for both subsistence and commercial revenue, was also lost in the fall of 1998 (Trainer and Wekell 2000). With sufficient warning of an incoming bloom, tribal fishers could seek alternative buyers for eviscerated crab, and shellfish managers might have longer lead times to schedule closures.

¹ National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Northwest Fisheries Science Center, Marine Biotxin Program, 2725 Montlake Blvd. E., Seattle, Washington 98112

² School of Oceanography, University of Washington, Box 357940, Seattle, Washington 98195

Case Study

Physical Oceanography and Transport Processes –

In contrast to most US east coast environments, the US west coast shelf is relatively narrow, so nutrient-rich deeper water can be effectively brought to the surface by wind-driven upwelling that occurs in the growing season along the entire coastal boundary (Hickey 1989, Landry *et al.*, 1989). In contrast to most east coast coastal areas, nutrient input from coastal rivers is negligible except in the associated estuary and right at the river mouth. Wind-driven upwelling of nutrients from deeper water layers fuels coastal productivity, resulting in both a strong seasonal cycle and several day fluctuations that mimic changes in the wind direction and, hence, upwelling. During an upwelling, phytoplankton respond to the infusion of nutrients near the coast, and this ‘bloom’ is moved offshore, continuing to grow while depleting the nutrient supply. When winds reverse (usually during storms), the bloom moves back toward shore where it can contact the coast or enter coastal estuaries.

In contrast to the east coast, alongshore topography of the coastline is relatively straight and wind systems are large scale. Thus currents and water properties (e.g., temperature, stratification, etc.) are similar over relatively large distances (>500 km) along the coast and are reasonably predictable, given information about the alongshore structure of the coastal wind field (Battisti and Hickey 1984).

Although the west coast topography and forcing are generally large scale, several coastal promontories and offshore banks and other topographic features exist. Such features may be particularly important to toxic PN growth and/or retention (Figure 1). For example, in regions where large coastal promontories occur, phytoplankton are swept offshore and southward by the meandering jets and/or eddies that form where the coastal jets detach from the shelf. These blooms rarely return to the coast. On the other hand, in regions where banks and more complex mesoscale topography occur, retention is more likely. Under certain wind conditions, phytoplankton in these retention areas can return to the coast, and if toxic, affect organisms on beaches and in estuaries.

The Juan de Fuca Eddy, a Retentive Feature off the Northern Washington Coast – The counter-clockwise cold eddy off the Strait of Juan de Fuca (also called the “Tully” eddy, Tully 1942) is located southwest of Vancouver Island and offshore of northern Washington. The eddy, with a diameter of about

50 km, forms in spring and declines in fall (Freeland and Denman, 1982), a result of the interaction between effluent from the Strait, southward wind-driven currents along the continental slope, and the underlying topography, a spur of the Juan de Fuca submarine canyon. A dominant feature of circulation patterns off the northern Washington coast, it is visible in summertime satellite imagery as a relative minimum in sea surface temperature and generally a relative maximum in chlorophyll *a*.

A connection between the eddy and the Washington coast was demonstrated in July 1991, when oil spilled in the eddy was found on the Washington coast 6 days later (Venkatesh and Crawford, 1993). This suggests that under certain ocean conditions, phytoplankton residing in surface waters of the Juan de Fuca eddy can impact the Washington coast.

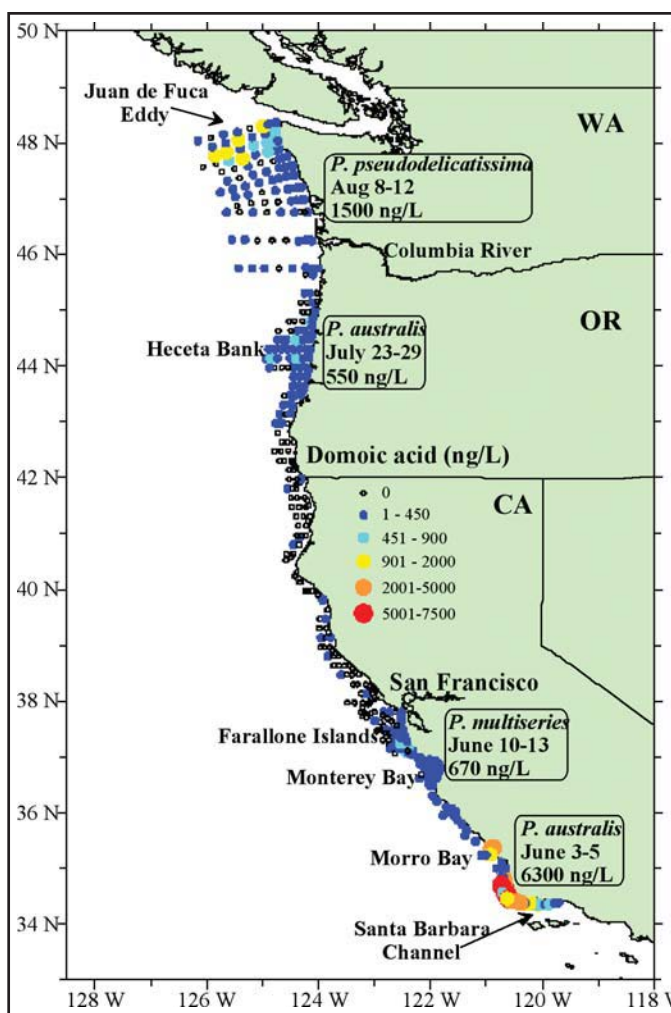


Figure 1. Particulate DA in PN species on the US West Coast in 1998. Maximum concentrations of DA and toxic species are indicated to the right of each area of high toxin. Each of these areas is associated with relatively retentive circulation patterns. (adapted from Trainer *et al.* 2001)

Sources of Toxic PN to the Washington Coast – Data collected on ships of opportunity in recent years show that the Juan de Fuca eddy usually contains much higher concentrations of particulate DA than elsewhere on the Washington coast (e.g., Figure 1). High levels of particulate DA are usually associated with high cell counts of PN (generally *P. pseudodelicatissima*). In extensive surveys in 1997 and 1998 (Horner *et al.* 2000), the highest concentrations of PN cells associated with the Juan de Fuca eddy were measured in the upper 5 m (Figure 2).

All measurements made during cruises and beach sampling of seawater and shellfish in 1997 and 1998 are consistent with the possibility that during some years, DA from this eddy appears to move southward in prolonged upwelling events and then onshore

during the first major storm of the fall season, resulting in high levels of DA in razor clams on coastal beaches. Such events result in immediate closure of coastal clamming beaches, often for the entire season (or longer, due to the slow depuration of DA from razor clam tissue).

Collaborations and Tools Instrumental in Developing Reliable Harmful Algal Bloom Forecasts – The impacts of DA on Washington coastal communities are currently being studied in a 5-year, multi-agency, multi-disciplinary program by the Olympic Region Harmful Algal Bloom (ORHAB) project, supported by NOAA's Coastal Ocean Program. The goals of this monitoring project are to 1) investigate the origins of open-coast toxic blooms, 2) monitor where and when the toxic species are present on the coast, 3) assess the environmental conditions under which blooms occur and are transported to intertidal shellfish populations, and 4) explore methods that can be used to forecast harmful algal blooms (HABs). The monitoring sites (Figure 3) were chosen because they have harvestable razor clam or mussel populations.

The ORHAB group is a forum for collaboration and cooperation among Federal, state, and local government agencies, academic institutions, coastal Native tribes, marine resource-based businesses, and public interest

groups. Its mission is to support applied and basic research on HABs and to build a greater local capacity to monitor and mitigate the effects of such events. The ultimate goal is to sustain a long-term monitoring project into the future without reliance on Federal support.

The US west coast HAB forecasting capability is in its

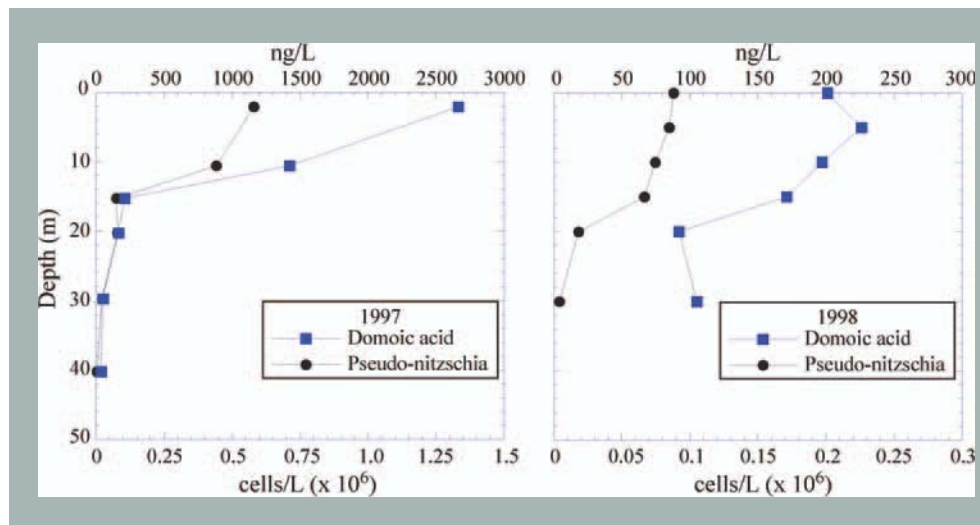


Figure 2. Typical vertical profiles of PN densities and DA concentrations in the Juan de Fuca eddy region.

early stages. Information to be used in developing an effective forecast includes the following.

- Dominant processes are large scale and can be 'sensed' over long distances.
- Upwelling is the dominant nutrient source to the US West Coast.
- Isolated topography-related retention areas may promote HABs. Possible HAB initiation sites, such as the Juan de Fuca eddy, have been identified.
- DA and PN are often located in the upper 20 m of the water column, so sensors ('Environmental Sensing Platform,' see the discussion in Next Steps) can be placed near the surface to detect cells and toxins.
- Blooms move to the coast when winds relax or reverse.
- Strong collaborative West Coast partnership, namely ORHAB, will be instrumental in providing monitoring results that will form the cornerstone of future forecasts. The collaborators in ORHAB will be the primary end users of the HAB forecasts.

Next Steps

This understanding of west coast physical processes in combination with remote sensing tools are a key to success of this program. The 'Environmental Sensing Platform' (Scholin *et al.* 1999) is one instrument, which when placed in likely initiation sites for HABs, will enhance

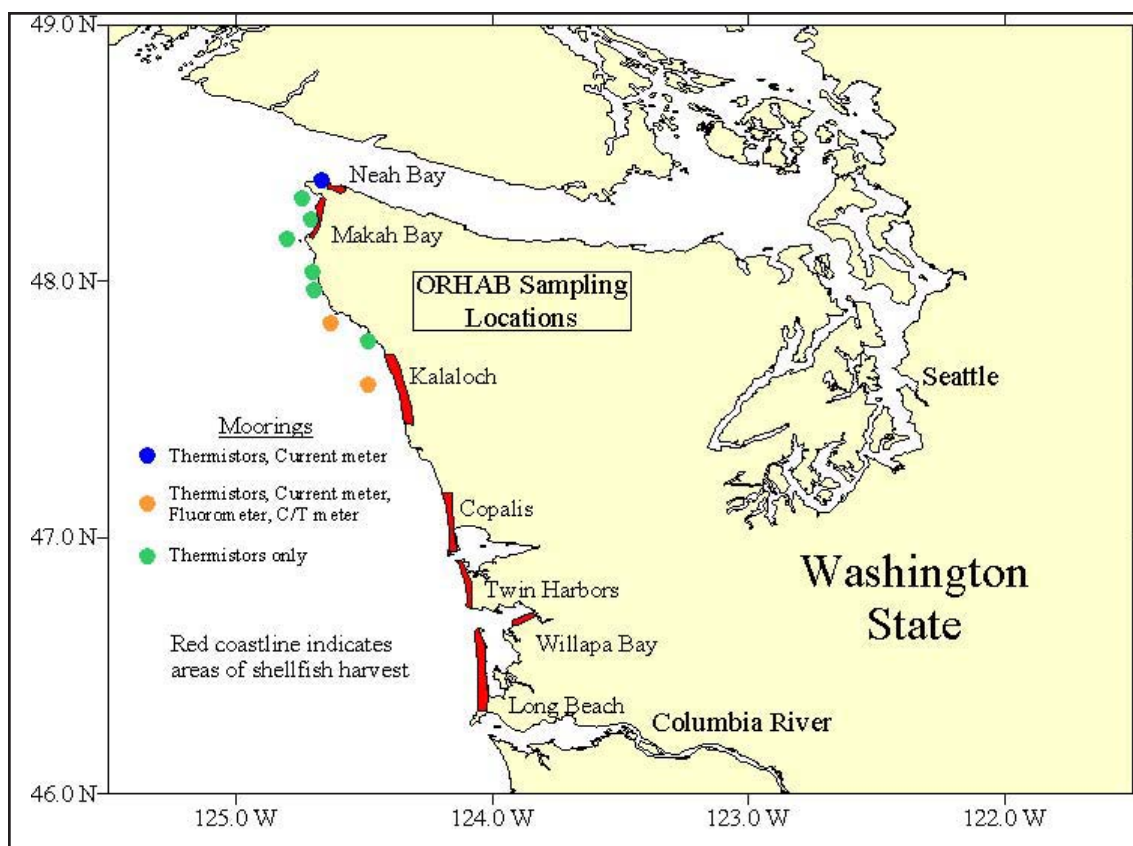


Figure 3. ORHAB monitoring sites, including areas of beach (seawater) sampling (Neah Bay and Makah Bay by Makah Tribe, Kalaloch and Copalis by Quinault Tribe, Twin Harbors and Long Beach by Washington Dept. of Fish and Wildlife, Willapa Bay by Washington Dept. of Ecology and Pacific Shellfish Institute). Razor clam domoic acid analyses are done by Washington Dept. of Health and Northwest Fisheries Science Center (NWFSC). Moorings are maintained by the Olympic Coast National Marine Sanctuary and data are analyzed by University of Washington and NWFSC.

the US west coast forecasting of toxigenic PN blooms. This instrument has a stand-alone function that uses genetic techniques to record the presence and abundance of micro-organisms. Seawater is collected in discrete samples. Installed at several locations along the coast, it serves as a sentinel for toxic algal blooms. A project to make the instrument more user-friendly has been proposed as a collaboration between ORHAB and the Monterey Bay Aquarium Research Institute.

In conjunction with data streams from moored arrays containing sensors such as fluorometers, current meters, nutrient sensors, thermistors (see the locations of ORHAB moorings and sensors, Figure 3), as well as ocean satellite images, this autonomous instrument will be a vital tool in providing reliable early warnings for HABs (most likely within days). This will allow the early opening of clamming season or will give commercial fishers time to seek an alternative buyer (in the case of Dungeness crab).

In the future, the US west coast will need to focus research and monitoring in the following areas, allowing

refinement of the HAB forecasting capability. Greater accuracy and longer lead time for prediction of coastal HABs will be achieved when the following questions have been answered.

- In the field, what are the conditions (chemical, biological, and physical) that make PN cells toxic? A number of lab studies have focused on aspects of this question, but limited field studies have been done.
- What makes one location a more viable initiation site for HABs than another? What are the environmental factors that result in bloom initiation and propagation?
- Can a biological/physical model be configured to effectively link the biological growth (DA production and cell survival) and physical transport of PN cells to coastal sites where fisheries are impacted?

Conclusion

The challenges faced on the west coast due to HAB-related mammal mortalities, widespread closures of beaches to shellfish harvest, and human illness can only be met by sustained beach monitoring programs such

as ORHAB and dedicated research cruises. These cruises will provide the means to answer the above questions through the observation of natural populations of PN *in situ*. Complete characterization of physical, biological and chemical conditions that favor harmful PN blooms, only possible through large-scale synergistic collaboration, is a prerequisite for forecasting these events. A forecasting capability will substantially improve the management of valuable coastal resources and the protection of human health, both of which are affected by toxigenic PN.

Acknowledgments

This project was partially funded by the Olympic Region Harmful Algal Bloom (ORHAB) program that was supported by the Monitoring and Event Response of HAB (MERHAB) under the National Centers for Coastal Ocean Science. The authors extend their sincere thanks to all ORHAB collaborators.

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Hindcasting Blooms of the Toxic Dinoflagellate *Alexandrium* spp. in the Western Gulf of Maine

D.J. McGillicuddy, Jr.¹, C.A. Stock¹, D.M. Anderson¹, R.P. Signell²

Abstract

Three-dimensional coupled physical-biological models have been developed to facilitate forecasting regional Alexandrium distributions in the Gulf of Maine. The hydrodynamic basis is a primitive equation model with advanced turbulence closure; it is forced by tides, winds, heat fluxes, and river discharge. Explicitly modeled aspects of Alexandrium population dynamics include germination from resting cysts, vegetative growth, and mortality.

Quantitative evaluation of model skill in a hindcast mode indicates the model is capable of simulating the observed large-scale patterns in Alexandrium abundance to within a factor of two to three. Sensitivity analysis suggests uncertainties in loss terms (such as respiration, predation and encystment) may be the most critical limitation of the coupled model in its present form.

Future developments may yield a predictive model with skill sufficient to produce meaningful ecological forecasts. However, the need for both physical and biological data streams to drive the models must not be underestimated. An integrated observational and modeling strategy will be essential in the context of ecological forecasting.

Introduction

Toxic or harmful algal blooms (HABs) are a serious economic and public health problem throughout the US and the world. In New England, the most serious HAB issue is paralytic shellfish poisoning (PSP), a potentially fatal illness that occurs when humans eat shellfish that have accumulated toxins as they feed on dinoflagellates in the genus *Alexandrium* (see review by Anderson 1997). These organisms are responsible for human illnesses and occasional death due to PSP, repeated closures of shellfish beds in both nearshore and offshore waters, the mortality of larval and juvenile stages of fish and other marine animals (White *et al.* 1989), and even the death of marine mammals such as humpback whales (Geraci *et al.* 1989).

Accurate forecasts of *Alexandrium* blooms in this region would therefore have important application, including the management of shellfish and finfish resources, aquaculture siting, and effluent discharge. Furthermore, there are a multitude of coastal zone

management issues that could use the hydrodynamic forecasts on which ecological predictions of *Alexandrium* would be based.

Case Study

Within the western Gulf of Maine (GOM) region, which extends from Penobscot Bay to Massachusetts Bay, a coupling has been inferred between the abundance and distribution of *Alexandrium* and a buoyant coastal current that travels from northeast to southwest. A conceptual model of this link has been proposed, known as the River Plume Advection Hypothesis (Franks and Anderson 1992a,b).

Critical features of this system include: 1) a source population of cells located to the north of Massachusetts Bay near the Androscoggin-Kennebec estuary; 2) freshwater outflow from the Androscoggin-Kennebec estuary resulting in a coastally-trapped, buoyant plume that supports the growth of *Alexandrium* cells and transports them to the south and west; and 3) plume

¹ Woods Hole Oceanographic Institution

² US Geological Survey

behavior, as influenced by the volume of freshwater outflow, the local wind stress, and underlying GOM circulation. All of these combine to regulate the alongshore and cross-shore location of the plume, its associated cells, and PSP toxicity.

As part of Ecology and Oceanography of Harmful Algal Blooms-Gulf of Maine (ECOHAB-GOM) and its

| | |
|-----------------------|---|
| Modeled inputs (T, S) | Bi-monthly climatological model with tides (Lynch <i>et al.</i> 1996) |
| Cruise data (T, S) | Upstream sections and offshore stations |
| River discharge | USGS gauges at 4 major rivers |
| Wind stress | Uniform distribution derived from Portland Buoy |
| Surface heat flux | Derived from temperature at Portland Buoy |
| Insolation | From land-based sensors at Woods Hole |

Table 1. Data sets used to force the circulation model.

predecessor programs, a set of numerical models has been developed to study these dynamics in a three-dimensional context. Thus far, these coupled physical/biological models have been used only for research purposes. Quantitative demonstration of predictive skill in a hindcast mode is a necessary precursor to an operational model. The following example describes recent efforts to assess the hindcast skill of one model.

The hydrodynamic basis of the following coupled physical/biological simulations is the Princeton Ocean Model (POM; Blumberg and Mellor 1987) with variable horizontal resolution of 2-4 km and 12 vertical levels. Vertical mixing is internally generated from the Mellor-Yamada 2.5 turbulence closure model. This physical model is forced by various modeled inputs as well as land-based and shipboard observations (Table 1). A population dynamics model for *Alexandrium* has been incorporated into this physical framework.

Alexandrium has a very complex life cycle (Figure 1). It can lie dormant as a resting cyst in the sediments for very long time periods, perhaps a decade or more (Anderson 1984; Keafer *et al.* 1992). Germination is regulated by a complex set of processes, including an endogenous clock (Anderson and Keafer 1987) and physiological responses to environmental factors such as temperature (Anderson 1980), light and oxygen availability (Anderson *et al.* 1987). Once emerged from the sediments, the cells swim toward the surface to begin a phase of vegetative growth.

Photosynthetic production is fundamentally limited by light and the availability of nutrients, however, maximum growth occurs only within a specific range of temperature and salinity. When faced with environmental stress such as nutrient limitation, the vegetative cells form gametes that subsequently fuse into a zygote (Anderson *et al.* 1984). The zygote then encysts, and the cycle is complete.

The model for *Alexandrium* is summarized in Figure 2. The ecosystem in which *Alexandrium* resides is not explicitly modeled; ecosystem effects are parameterized through their influence on the vital rates of *Alexandrium*'s population dynamics processes. The top panels show the various factors that contribute to the input of cells due to germination. On the far left is the observed distribution of cysts, showing the peak offshore of Casco Bay. In the middle is the functional fit to laboratory data on the endogenous clock, shown as open circles. On the right is the germination rate as a function of light and temperature, based on laboratory

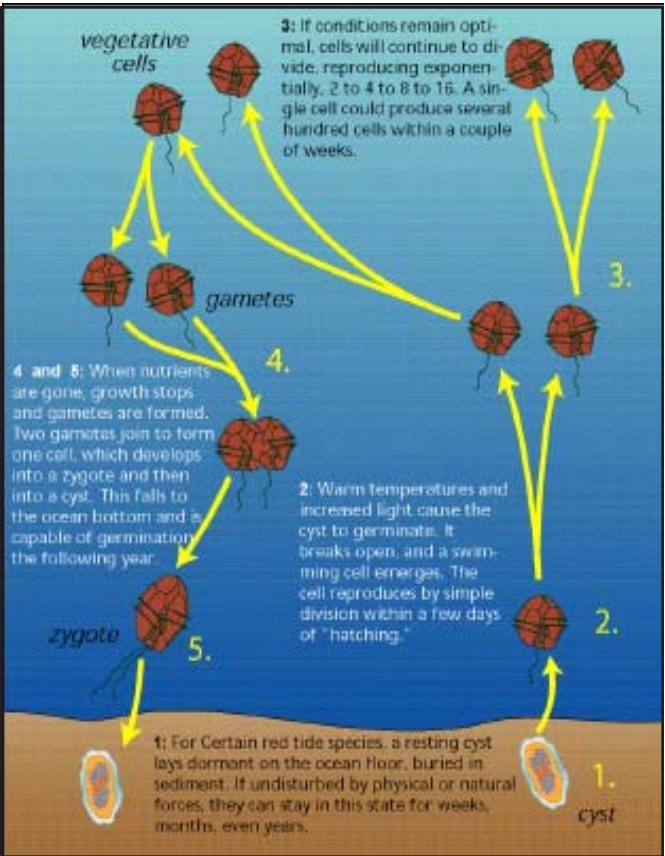


Figure 1: Life cycle of Alexandrium.

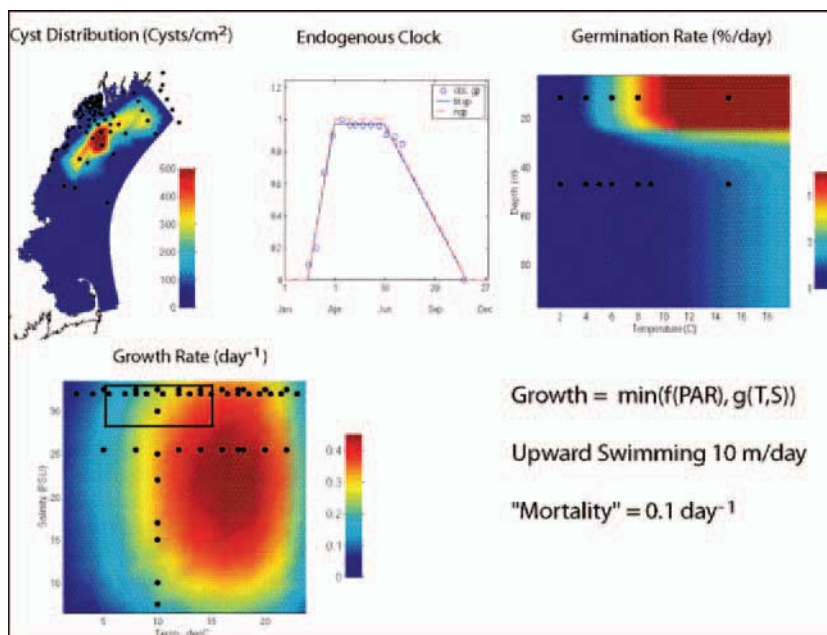


Figure 2: Population dynamics model for *Alexandrium*.

measurements (indicated by black dots). The vertical axis actually represents light intensity but has been scaled to depth.

The function describing vegetative growth is based on data from Watras *et al.* (1982). It suggests that *Alexandrium* has a preferred temperature and salinity range for optimal growth. The range of temperature and salinity conditions that *Alexandrium* cells encounter in the simulations described here is indicated by the black rectangle. The overall growth rate is based on this function and is modulated by the ambient light field as per Liebig's law of the minimum (Liebig 1845). In other words, at each time step, the lesser of the two growth rates predicted by the temperature/salinity and light-limitation functions is specified. Initial experiments with simple representations of nutrient limitation have been started, but these effects are not included here. An upward swimming rate of 10m/day and a net 'mortality' of 0.1 per day are specified. Mortality is in quotes here because it represents the net effect of all loss processes, which include predation and encystment.

This coupled model has been used to hindcast a data set collected in the Western Gulf of Maine in 1993 as part of the Regional Marine Research Program (RMRP). An example snapshot from a hindcast simulation is shown in Figure 3; the entire simulation is available

in animated form on the web site (McGillicuddy *et al.* 2002). The salinity field shows the impact of fresh water plumes originating at the Kennebec/Androscoggin and Merrimack rivers. These tend to propagate down the coast, and also undergo significant cross-shore displacements due to the effects of upwelling and downwelling winds. This particular simulation suggests offshore initiation of the bloom because the impact of the germination input from the offshore maximum in the cyst distribution is clearly evident. Features in the cell distribution are advected downstream in the coastal current, and are also clearly impacted by wind-driven upwelling and downwelling.

Quantitative evaluation of hindcast accuracy requires careful consideration of how the model results should be compared with observations. One might think this issue is as simple as subsampling the model solutions at the space/time points at which data are available. However, such an approach is problematic because the observations contain small-scale patchiness not resolved in the model. Plots of autocorrelation functions for both simulated and observed fields bear this out. Thus, it is necessary to average both the model results and the observations on scales resolved by both.

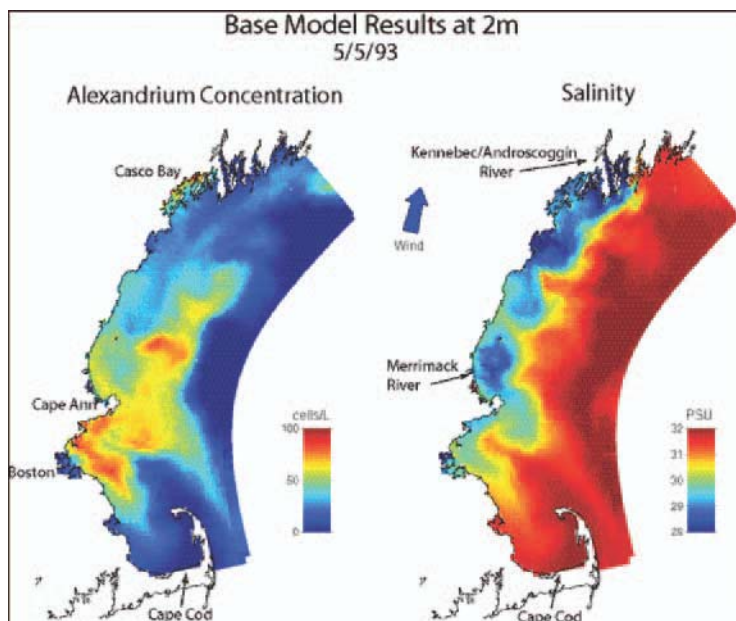


Figure 3: Example snapshot of model output.

Therefore, nine zones in which the mean values of the simulated and observed concentrations are defined and compared for each cruise (Figure 4). An error metric J quantifies the degree to which the simulated means (sim) match the observed means (obs) for each zone during each cruise: $J = \max(\text{sim}, \text{obs}) / \min(\text{sim}, \text{obs})$. A perfect fit would result in a value of 1.0. This penalty function grows larger as the simulation diverges from the observations. A variety of error metrics is possible,

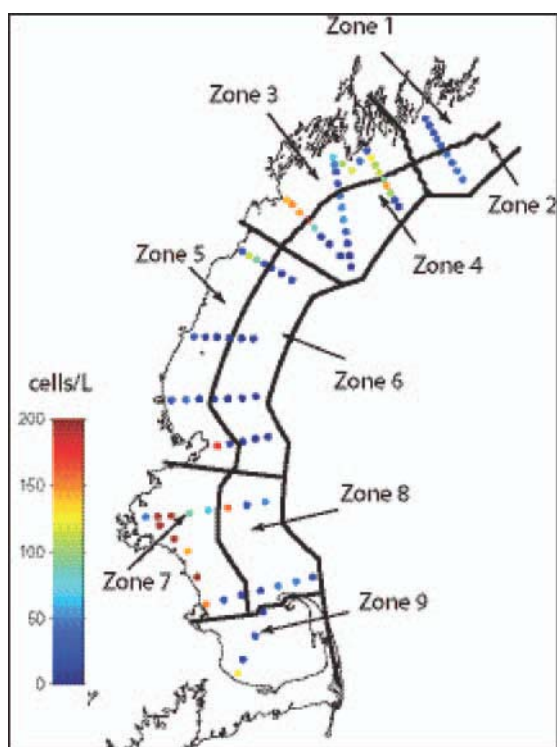


Figure 4: Typical observational coverage during one cruise. Colored dots represent observations of *Alexandrium fundyense* concentration (cells per liter). The nine zones chosen for model evaluation are indicated.

and all have various advantages and disadvantages. One appealing aspect of this particular metric is that errors at low concentrations are weighted just as heavily as those at high concentrations. In summary, this approach yields a quantitative method of evaluating the large-scale spatial distribution, timing, and magnitude of the bloom.

There is yet another issue confounding evaluation of the skill of the model predictions: the simulations are sensitive to model parameters that are not perfectly known. Therefore, the adjustable parameters are tuned within their envelope of reasonable values in a manual procedure for parameter estimation. More sophisticated estimation procedures have been used in the context of zero- and one-dimensional models (e.g., Vezina and Platt 1988, Fasham and Evans 1995, Hurtt and Armstrong

1996, Friedrichs and Hofmann 2001), and will be pursued in the future. At present, the computational demands of those approaches make them impractical for the three-dimensional system of interest here.

Quantitative evaluation of hindcast skill facilitates testing hypotheses concerning bloom controls by ascertaining whether one model structure fits the data significantly better than another. As an example, we compare the results of two different simulations: one without grazing of *Alexandrium*, and one that includes a spatially uniform loss term of 0.1 per day (Figure 5). Each of the four panels shows the space/time distribution of the error metric. Time runs along the ‘cruise’ axis with cruises 1-5 in April to June. Space runs along the ‘zone’ axis, with zones 1-9 corresponding to those indicated in Figure 4. The zones have been separated into inshore and offshore groups. The height of the vertical bars indicates the magnitude of the error metric, with low model values in blue and high model values in red.

In the simulation that does not include mortality, cell concentration is systematically over-predicted late in the run in the southern portion of the domain. The average value of the error metric is 3, so on average the predicted values are within a factor of three of the observations. Explicit treatment of loss processes via a uniform loss rate of 0.1 per day improves the simulations dramatically, reducing the large errors late in the run in the southern portion of the domain. The error metric drops to an average of 2.33, so the predictions are now within a factor of two of the observations.

The conclusion is that a loss term (which may represent some combination of predation and encystment) plays a key role in controlling the development and termination of the bloom. One important caveat is the circulation fields are assumed to be correct; the possibility that mismatches between the simulated and observed concentrations of *Alexandrium* could be a result of unknown deficiencies in the modeled circulation cannot be excluded. Future research will formulate measures of misfit for the physical fields as well as the biological fields.

Lessons Learned

The hindcasts of the 1993 RMRP data do not rely on data assimilation to keep the circulation fields on track. The rationale for adopting a purely ‘forward’ modeling approach hinged on the fact the circulation in this relatively small region is largely controlled by riverine buoyancy inputs and wind-forced dynamics. Such is

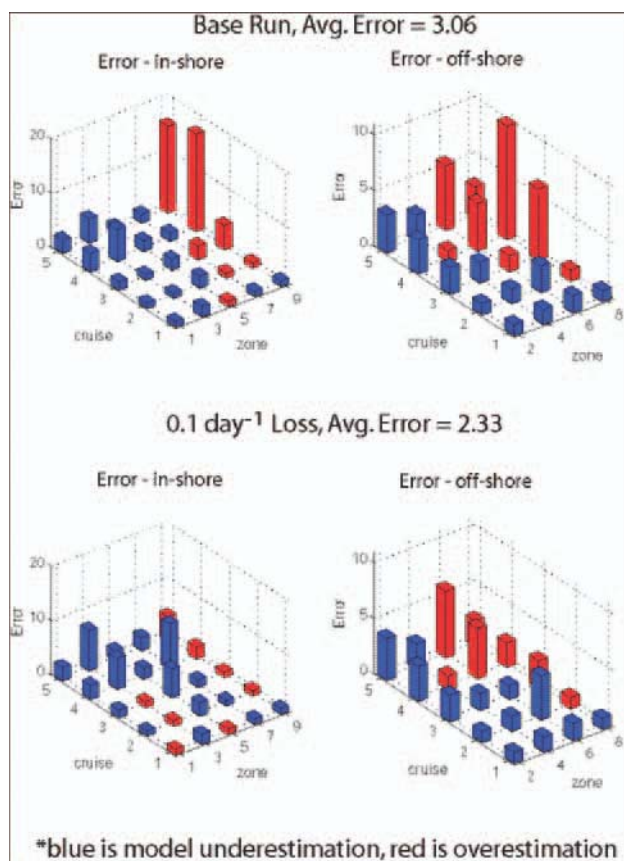


Figure 5. Quantitative evaluation of hindcasts based on two different biological formulations (see text).

clearly not the case in larger domains where a wide variety of other processes influence the circulation.

Results from the ECOHAB-GOM program have shown the dynamics of *Alexandrium* blooms are regional in scope, spanning the waters from the Bay of Fundy down into Massachusetts and Cape Cod Bays (Anderson *et al.* 2000; Townsend *et al.* 2001). Thus, a gulf-wide modeling approach will be necessary in an operational context. Given the complex hydrodynamics characteristic of this large regional domain, data assimilation will be an essential element of skillful prediction.

An operational system based on a data-assimilative model implies an observational network to drive it. Design of such a network would be facilitated by Observational System Simulation Experiments (OSSEs; see Robinson *et al.* 1998; McGillicuddy *et al.* 2001). The idea is model simulations can serve as a space/time continuous representation of reality, which is then subsampled in a specified fashion to produce a simulated data set. The simulated data are then fed into an analysis scheme in which they are synthesized into a reconstruction of 'reality.' Direct comparison of the reconstructed field with the 'truth' as defined by the

original simulation provides a quantitative evaluation of that particular sampling strategy.

In summary, *meaningful ecological forecasts of this system will require accurate real-time data streams together with advanced data assimilation techniques.*

Conclusion/Next Steps

Coupled physical-biological models of *Alexandrium* in the Gulf of Maine have matured to the point that it is now feasible to assess their operational suitability and potential value. A strategy for this could consist of three main elements: 1) thorough evaluation of the predictive skill in a hindcast model using data from ECOHAB-GOM and its predecessor programs; 2) improvement of the models from what is learned in that evaluation; and 3) formulation of a plan for transition of the models to operational use. The second item implies the dynamics of *Alexandrium* blooms are not fully understood, thus additional research is needed on the underlying physical/biological interactions that control these blooms.

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Forecasting Oyster Mortality in Relation to Variations in Freshwater Inflow into the Apalachicola-Chattahoochee-Flint (ACF) Basin

Mark E. Monaco¹ and Robert J. Livingston²

Abstract

This paper was part of a symposium on Ecological Forecasting sponsored by NOAA's National Centers for Coastal Ocean Science (NCCOS) at the 16th Biennial Conference of the Estuarine Research Federation in November 2001. The summary below complements and updates previously published articles on the topic of forecasting and assessing changes in the Apalachicola Bay, Florida ecosystem due to variations in freshwater inflow (Christensen et al. 1998; Livingston et al. 2000).

*The eastern oyster, (*Crassostrea virginica*), was selected to study the impacts of changes in freshwater inflow on oyster mortality, growth, and ecology. Optimum oyster habitat in Apalachicola Bay is a function of estuarine circulation, nutrient and salinity concentrations, habitats, basin morphometry, and freshwater inflow. Several scenarios of the amount of freshwater inflow entering into the Bay were evaluated to provide an ecological forecast on potential impacts on oyster populations. At the most productive oyster bars in the eastern Bay, predation, a biological determinant of oyster growth, was predicted to increase due to high salinity concentrations and stable salinity regimes associated with reduced freshwater inflow. In addition, at the western bars where greatest young individual oyster growth rates were observed in relatively high salinity waters, potential oyster mortality increased with decreased freshwater inflow.*

Introduction

This paper summarizes an oral presentation given at the 16th Biennial Conference of the Estuarine Research Federation in November 2001. The paper entitled *Forecasting Oyster Mortality in Relation to Variations in Freshwater Inflow into the Apalachicola-Chattahoochee-Flint (ACF) Basin*, is part of an ongoing cooperative study between NOAA's National Centers for Coastal Ocean Science's (NCCOS) Biogeography Program and the Florida State University (FSU) Center for Aquatic Research and Resource Management. The paper was part of a symposium on ecological forecasting sponsored by NOAA's NCCOS (NOAA/NOS 2001). This summary complements and updates previously published articles on the topic of forecasting and assessing changes in the Apalachicola Bay, Florida ecosystem due to variations in freshwater inflow (Christensen *et al.* 1998; Livingston *et al.* 2000).

The US Army Corps of Engineers requested NOAA/NCCOS to participate as a cooperating agency in the preparation of the draft environmental impact statement (EIS) for the Apalachicola-Chattahoochee-Flint (ACF) basin. NOAA is participating on the inter-agency team because of several legislative mandates: develop and review of EISs (National Environmental Policy Act); assess potential impacts on essential fish habitats (Magnuson-Stevens Fishery Conservation & Management Act); determine if any endangered species will be affected (Endangered Species Act); and assess potential impacts to the Apalachicola Bay National Estuarine Research Reserve (Marine Sanctuaries Act). This study was conducted jointly by the Apalachicola Bay EIS Assessment Team that was led by NCCOS and FSU consulting with the Northwest Florida Water Management District, University of South Florida, and the Florida Fish and Game Commission.

¹ NOAA/NOS National Centers for Coastal Ocean Science's Biogeography Program, Silver Spring, MD

² The Florida State University, Center for Aquatic Research and Resource Management, Tallahassee, Florida

Because of limited resources and data, the NCCOS/FSU team focused on the eastern oyster (*Crassostrea virginica*), a key ecological species, to assess potential impacts from reduced freshwater into the ACF basin on oyster mortality, growth, and ecology. It was selected for two primary reasons: 1) its importance to the ecology and economics of the Apalachicola Bay region and, 2) the availability of a unique database on oyster biological attributes (e.g., percent mortality at oyster bars) housed at FSU.

Although oysters can survive in relatively high and dynamic salinities, high and stable salinity regimes enable predators and diseases to attack oyster populations. The dominant marine predator of oysters in Apalachicola Bay is the oyster drill (*Thais haemastoma*, Livingston *et al.* 2000). Thus the study focused on the indirect impacts of reduced freshwater inflow into Apalachicola Bay on the eastern oyster population due to increased mortality from predation and disease.

The results of the ecological forecast have been published in Christensen *et al.* 1998, Livingston *et al.* 2000, and the US Army Corp of Engineers Draft EIS. Plans are to conduct a new assessment if a freshwater inflow allocation formula can be agreed to between the States of Georgia, Florida, and Alabama. To date, no final freshwater allocation formula for the ACF basin has been approved by the States to enable final assessment studies. If, and when, a final allocation formula is approved, the oyster ecological forecast will contribute information and assessment to the Federal Inter-agency commissioner and the states on the potential biological impacts of reduced freshwater inflow on oyster populations within the Bay. The ecological forecast will assist the Federal commissioner in evaluating the allocation formula and contribute to the commissioner's decision to concur, or not concur, with the proposed freshwater inflow allocation formula. This decision will be partially based on the collective assessment of Federal agencies that comprise the ACF Inter-agency Team.

Case Study

The ACF ecological forecast case study was developed to address the problem of competing uses of water between the three states for water allocation. Relative to the ACF issues, the primary issues are between Georgia, who requires additional water to support domestic and agricultural needs, compared to Florida

who has generally focused on maintaining sufficient flow for ecological and economic concerns in the Apalachicola Bay region. NCCOS's role in this management issue was to develop a set of forecasts using various scenarios of the amount of freshwater inflow into the Bay to predict potential ecological impacts to the estuary.

Therefore, the objectives of the ecological forecast were to:

- 1) Model the variation in estuarine salinity and temperature regimes under various freshwater inflow scenarios.
- 2) Develop models of potential oyster growth and mortality under various freshwater inflow scenarios.
- 3) Integrate the physical and biological models to provide an assessment of the potential impacts of changes in freshwater inflow on the Apalachicola Bay oyster community.
- 4) Develop digital maps of oyster mortality and growth estimates under various freshwater inflow scenarios.

Methods—Study results were developed by integrating several biological and physical models using Geographical Information System (GIS) technology to demonstrate how oyster biological responses (mortality and growth) could potentially change with reduced freshwater inflow. A series of digital maps were produced that showed potential oyster mortality due to predation throughout the Bay under various demands for freshwater in drought, low-moderate, and wetter than normal summer flow conditions. Based on available data, a range of potential oyster mortalities under various alternative flow conditions were modeled and mapped in Christensen *et al.* 1998.

The integrated biological (mortality) and physical (hydrodynamic) models enabled NCCOS to predict potential oyster mortality using three historical flow years, eight simulated alternative operation and demand scenarios, and two existing operations with different demand scenarios (Christensen *et al.* 1998). The scenarios bracketed Bay-wide simulated salinity regimes (i.e., oyster habitat) under different demands for freshwater. Three years with different water inflows were selected for the assessments based on available oyster mortality and growth data: 1985 with low to moderate flow conditions, 1986 with drought flow conditions, and 1991 with wet flow conditions.

Based on this information, the potential of oyster mortality was forecasted under the various flow scenarios and varying levels of human demands on

those flow scenarios. Potential oyster mortality was grouped into 10 percent mortality categories (0-1; 1-4; 4-9; 9-16; 16-25; 25-36; 36-49; 49-64; 64-81; 81-100 percent mortality) (Christensen *et al.* 1998). These categories were color-coded into map polygons and the maps represent the spatial distribution of potential oyster mortality (Christensen *et al.* 1998).

Results – Low to Moderate Hydrologic Year (1985) – Digital maps showing potential oyster mortality under 1985 high flow operations with 1995 demands for water were modeled and mapped (Christensen *et al.* 1998). Under the high flow operations, the peak in potential oyster mortality occurred in the 4-9 percent (not biologically significant) category, indicating that over 50 percent of the Bay would not experience significant potential oyster mortality. Most important, at the largest and most productive eastern oyster beds of Cat Point and East Hole, potential oyster mortality was less than 10 percent (Figure 1). However, in the far western portions of the Bay near Scorpion Bar, the model predicted

tions and demands. This was evidenced by 71 percent (1985 flows with 2050 low flow operations and demands) of the potential oyster mortality cells throughout the Bay fell in mortality categories ranging from about 10 to 50 percent.

Under the ‘worst case’ scenario for 1985 flow conditions with 2050x demands, a peak in potential oyster mortality throughout the Bay occurred at 9-16 percent mortality category (2050x corresponds to a 2.5 times increase in agricultural consumptive demands). However, the largest and most productive bars at Cat Point and East Hole could experience an increase in potential oyster mortality from <10 percent under 1995 demands to 10-25 percent under 2050x demands. In addition, the commercially important bars of St. Vincent Sound could experience potential oyster mortality ranging from 16-81 percent; with most of the bars predicted to experience about 25-50 percent mortality. The mean potential oyster mortality would increase from 11 percent in 1985 with 1995 consump-

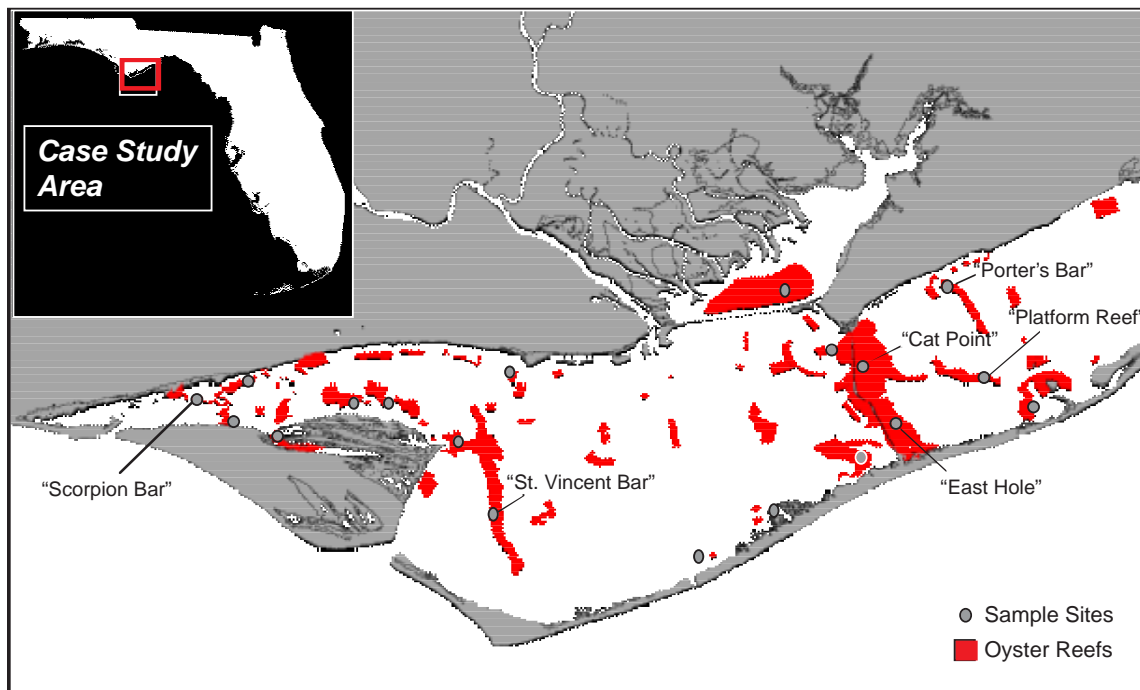


Figure 1. Oyster reef sampling sites in Apalachicola Bay, FL

this area could experience high potential oyster mortality (>50 percent) under the same flow conditions.

Under the scenario for 1985 flow conditions with 2050 human consumptive demands, significant oyster mortality could occur over a greater geographic range when compared to the 1985 with 1995 high flow opera-

tive demands to 16 percent under 2050x consumptive demands.

To better understand and interpret the difference in predicted mortality across the various alternative demand scenarios for 1985, a spatial calculation of percent change in mortality was run. This went from the

base, or best case flow condition – the 1985 high freshwater inflow operations with 1995 demands, to an extreme condition – the 1985 with low freshwater inflow with year 2050 demands. The analysis reflects differential mortality as a percentage of the original (base year) condition for the alternative flow analysis of 1985. The analysis indicated that a large portion of the oyster reefs in the eastern portion of the Bay, including East Hole and Cat Point, were characterized by the greatest increases in mortality (>25 percent of base) from the high flow condition to the low flow condition. The western reefs of St. Vincent Sound would also exhibit large increases in potential mortality (>5-25 percent of base). Thus, the most productive oyster beds throughout Apalachicola Bay are predicted to experience the greatest increases in mortality as freshwater inflow is reduced due to predation and disease.

No Action Alternative – Under the ‘no action’ alternative (current reservoir operations), few noticeable changes in the patterns of potential mortality were detected between the 1985 flow conditions with 1995 consumptive demands and the 1985 flows with projected 2050 demands. However, comparing the cumulative distribution of calculated mortality over the entire Bay, the 1985 flow with 1995 demands consistently showed a slightly lower mortality rate. There was an overall greater geographic range of low potential mortality throughout the Bay (0-9 percent) under the 1995 demand scenario, while the 2050 scenario indicated a 3 percent greater spatial coverage for the 9-16 percent mortality range relative to the 1995 demand scenario.

Although the differences appear to be slight, placing the 2050 demands on existing operations would likely result in elevated oyster mortalities baywide. This analysis, when compared to the alternative flow scenarios, indicates that the greatest effects on Apalachicola Bay oysters will result from reservoir operations resulting in low river flows.

Lessons Learned

The development of this ecological forecast has been critical to providing peer-reviewed science to the draft EIS and for State and Federal managers to understand potential biological impacts to the Apalachicola Bay ecosystem. The forecast only focused on the potential impacts of increased oyster mortality, decreased

growth, and implications to the overall ecology of Apalachicola Bay (Livingston *et al.* 2000). The study was designed in cooperation with Federal agencies and the State of Florida. This approach was critical so the analyses were able to be input into the State of Florida’s decision process as they developed draft freshwater inflow allocation formulas. Most important, the assessment enabled NOAA to meet the request of the US Army Corp of Engineers to be a member of the Inter-Agency Team to provide a technical review of a range of proposed freshwater inflows into the ACF basin and ultimately into Apalachicola Bay.

Improving the ecological forecast requires additional data to validate the potential oyster mortality models, since resource constraints enabled only limited model validation. Additional field studies to further refine and validate the model are urgently needed because there is currently a major drought in freshwater that may have significant impacts on the capability of the models to predict ecosystem impacts. This will limit the information content and quality available to managers in deciding the potential for oyster mortality. However, until additional studies are implemented, the joint NCCOS and FSU study to predict oyster mortality in Apalachicola Bay due to changes in freshwater inflow stands as the most robust study to support management actions relative to flow regimes into the Bay.

Conclusions/Next Steps

Optimum oyster habitat in Apalachicola Bay is a function of estuarine circulation, nutrient and salinity concentrations, habitats, basin morphometry, and freshwater inflow (Livingston *et al.* 2000). At the most productive oyster bars in the eastern Bay, predation, a biological determinant of oyster growth, was due to the indirect effects of salinity limitation of marine predators and disease as well as the actual proximity of these bars to Gulf entry points into the Bay. In addition, at the western bars where greatest young individual oyster growth rates were observed in relatively high salinity waters, potential oyster mortality increased with decreased riverine freshwater inflow.

The intricate associations of the physical habitat (salinity) and the biological responses of oysters is part of the complex interplay of the various life stages of the oyster with both physical and biological aspects of the Bay ecosystem. Many of these attributes are either directly (increased salinities) or indirectly (predation) associated with freshwater from the Apalachicola River.

The Apalachicola Bay accounts for about 90 percent of Florida's and 10 percent of the national commercial oyster fishery (Livingston *et al.* 2000). Without the nutrient loading from the river, coupled with reduced salinities and variations in salinity concentrations (i.e., adequate freshwater inflow) that enable continued oyster growth and protection from marine predators, the commercial value of Apalachicola Bay's oyster production may be damaged.

At this time, no final allocation formula has been developed or agreed upon by the States. If, and when, a final ACF freshwater inflow allocation formula is published, NOAA will continue to participate in the technical review process by the Inter-agency Review Technical Review Team. Plans are to have the NCCOS and FSU team re-run the potential oyster mortality models using the final allocation formula and provide an assessment report to the Federal Commissioner.

Additional model runs to predict potential oyster mortality under the current reservoir operations will be conducted to assess the biological impacts from the current southeast drought conditions that the ACF basin is experiencing. In addition, in consultation with the National Marine Fisheries Service, NCCOS will formulate an assessment on the potential impacts of reduced freshwater inflow into Apalachicola Bay on Federally managed commercial species (e.g., shrimp).

Acknowledgements

We would like to thank all members of the NOAA/NCCOS Biogeography Program who contributed to the

development of this assessment with special thanks to Mr. John Christensen and Mr. Tim Battista. In addition, we thank Mr. John Klein and Mr. Russell Ives in supporting the development of the hydrological model component of the study. We thank members of the FSU assessment team lead by Mr. Glen Woodsum and hydrological model runs conducted by Boris Galperin of the University of South Florida. Finally, we thank Ms. Jamie Higgins for assisting in the production of this manuscript.

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A Potential Ecoforecast for Protozoal Infections of the Eastern Oyster (*Crassostrea virginica*) in the Upper Chesapeake Bay

Shawn M. McLaughlin¹ and Stephen J. Jordan²

Abstract

Perkinsus marinus and Haplosporidian *nelsoni* cause devastating infections in populations of the eastern oyster, *Crassostrea virginica*, along the US Atlantic coast and Gulf of Mexico. Salinity and temperature are considered major controlling factors in the prevalence and infection intensity of these two parasites. Fishery managers and oyster growers use this relationship to make short-term predictions of potential outbreaks of disease in oyster populations and to determine the best timing, or sites, for planting and harvesting of oysters. Predicting medium or long term fluctuations in oyster disease in the Chesapeake Bay is limited; however, by the inability to accurately predict medium- and long-term fluctuations in weather patterns such as occurrence and duration of drought conditions.

Several mathematic models have been developed to assess the health of oyster populations in the Gulf of Mexico and Chesapeake and Delaware bays. An impediment to the accuracy of mathematical models is the current severe gap in knowledge regarding the parasites, the host, and their interactions. Continuous advances in shellfish pathology research can be expected to improve the precision of predictive models. A population dynamics model recently developed by Jordan and Vanisko (submitted) is discussed as a potential tool for forecasting oyster disease in the upper Chesapeake Bay.

Introduction

Over the last 50 years, two pathogenic parasites, *Perkinsus marinus* and Haplosporidian *nelsoni*, have played a major role in the decline of the oyster fisheries in the United States. The parasites cause serious disease in oysters and lead to severe epizootics (epidemics among animals). *Perkinsus marinus* was first discovered in the Gulf of Mexico during losses of oysters in the 1940s and causes a disease commonly referred to as Dermo (Mackin *et al.* 1950). The range of the parasite now extends from Mexico (Burreson *et al.* 1994) to Maine (Barber *et al.* 1997). Oyster epizootics occur in several regions including the Chesapeake Bay (Andrews and Hewatt 1957, Andrews 1988, Burreson and Ragone-Calvo 1996). *Haplosporidium nelsoni* is known by the acronym MSX (Multinucleated Sphere Unknown). MSX was first associated with epizootic mortalities in eastern oysters in the Chesapeake and Delaware Bays in the late 1950s and early 1960s (Andrews 1966, Haskin *et al.* 1966). The range of MSX now extends along the Atlantic coast

from Florida to Maine with major epizootics continuing to occur in the mid-Atlantic and northeastern states (Ford and Tripp 1996, Barber *et al.* 1997).

Temperature and salinity have been identified as important environmental factors affecting interactions between eastern oysters and the parasites *P. marinus* (Andrews and Hewatt 1957, Andrews 1988, Burreson and Ragone-Calvo 1996, Ford 1996) and *H. nelsoni* (Farley 1975, Ford and Haskin 1982, Ford 1985, Matthiessen *et al.* 1990). In general, elevated temperatures (20-25° C) and salinities (>12 ppt) favor proliferation of the parasites and significant increases in parasite activity. Prevalences may be observed during extended periods of low rainfall (Andrews 1988). In regions where oyster populations harbor significant numbers of parasites, a mild winter and dry spring may result in increased water salinities and high parasite prevalences (Burreson and Ragone-Calvo 1996). Significant rainfall in the early summer; however, may create less favorable conditions for parasite proliferation, thus prevalences and mortalities

¹ NOAA National Ocean Service, Cooperative Oxford Laboratory Oxford, MD 21654

² Environmental Protection Agency, Gulf Ecology Division, Gulf Breeze, FL 32561

| Publication | Location | Type | Topic | Variables |
|----------------------------|--------------------------------|--|--|--|
| Hoffman et al. 1992 | Gulf of Mexico, Chesapeake Bay | Population | Reproduction of eastern oyster | Temperature; food supply |
| Powell et al. 1992 | Gulf of Mexico | Population | Growth and reproduction of eastern oyster | Filtration rate |
| Hofmann et al. 1994 | Gulf of Mexico | Population | Adult size and reproductive effort in eastern oyster | Temperature; salinity, food supply; turbidity |
| Powell et al. 1994 | Galveston Bay, Chesapeake Bay | Population | Mortality rate of eastern oyster | Food supply; recruitment; fishing; disease; predators |
| Hoffman et al. 1995 | Galveston Bay | Coupled oyster population-parasite model | Prevalence and infection intensity of <i>P. marinus</i> in the eastern oyster | Temperature; salinity, food supply; turbidity; parasite density; oyster mortality |
| Powell et al. 1995 | Galveston Bay | Population | Population abundance of eastern oyster | Reduced phytoplankton stock |
| Powell et al. 1996 | Gulf of Mexico | Coupled oyster population-parasite model | Triggering mechanisms for <i>P. marinus</i> epizootics | Temperature; salinity; food supply; turbidity; recruitment and juvenile mortality; parasite virulence and disease resistance |
| Ford et al. 1999 | Delaware Bay | Coupled oyster population-parasite model | <i>H. nelsoni</i> infection process in oysters | Parasite proliferation; sporulation; transmission; spread in tissues; salinity; temperature; oyster and parasite mortality |
| Powell et al. 1999 | Delaware Bay, Chesapeake Bay | Coupled oyster population-parasite model | Transmission of <i>H. nelsoni</i> in oysters | Salinity; temperature; season |
| Paraso et al. 1999 | Delaware Bay | Coupled oyster population-parasite Model | <i>H. nelsoni</i> infection intensity and prevalence in oysters | Salinity |
| Brousseau and Baglivo 2000 | Long Island Sound | Linear | Progression of <i>P. marinus</i> disease in eastern oysters | Temperature |
| Ragone-Calvo et al. 2000 | Lower Chesapeake Bay | Individual-based oyster model | Transmission, growth, and death of <i>P. marinus</i> within the eastern oyster | Temperature; salinity |
| Klinck et al. 2001 | Delaware Bay | Fisheries | Time-dependent mortality rates in the eastern oyster | Disease; recruitment into market stock |

Table 1. Examples of mathematic models available for oyster health assessment.

may not be as high as expected. Prolonged drought conditions may lead to elevated parasite prevalences in areas where the parasites are endemic in oyster populations. Indeed, the mild winters and drought conditions of the late 1980s and early 1990s have been associated with increased parasite prevalences and geographic range extensions (Burrenson and Ragone-Calvo 1996, Ford 1996).

In the last decade, a number of mathematical models have been developed to assess the relative outcomes of disease virulence and pressure under varying conditions for eastern oyster populations (Table 1). There is a consensus among the authors that these models can help decision-makers determine how best to manipulate oyster stocks around *P. marinus* and *H. nelsoni* in a particular geographic region. Moreover, a dual disease model presently under development will improve the ability to examine the relationship between *P. marinus* and *H. nelsoni* and better predict how each produces mortality in oysters under a suite of environmental conditions.

Case Study

State agencies are the regulators of commercial oyster fisheries and those with significant oyster resources conduct routine surveys to monitor the health of oyster populations. The Maryland Department of Natural Resources (MDNR) Cooperative Oxford Laboratory (COL) conducts annual disease surveys of oysters in the upper Chesapeake Bay.

Sixteen years of Chesapeake Bay oyster disease and population data collected by MDNR provide the basis of a prototype population dynamics model developed by Jordan and Vanisko (submitted). The primary purpose for its development was to predict outcomes of management options for resource restoration and fisheries enhancement. The potential for the model to predict the effects of oyster disease on the population is examined in this study.

Jordan and Vanisko's population dynamics model simu-

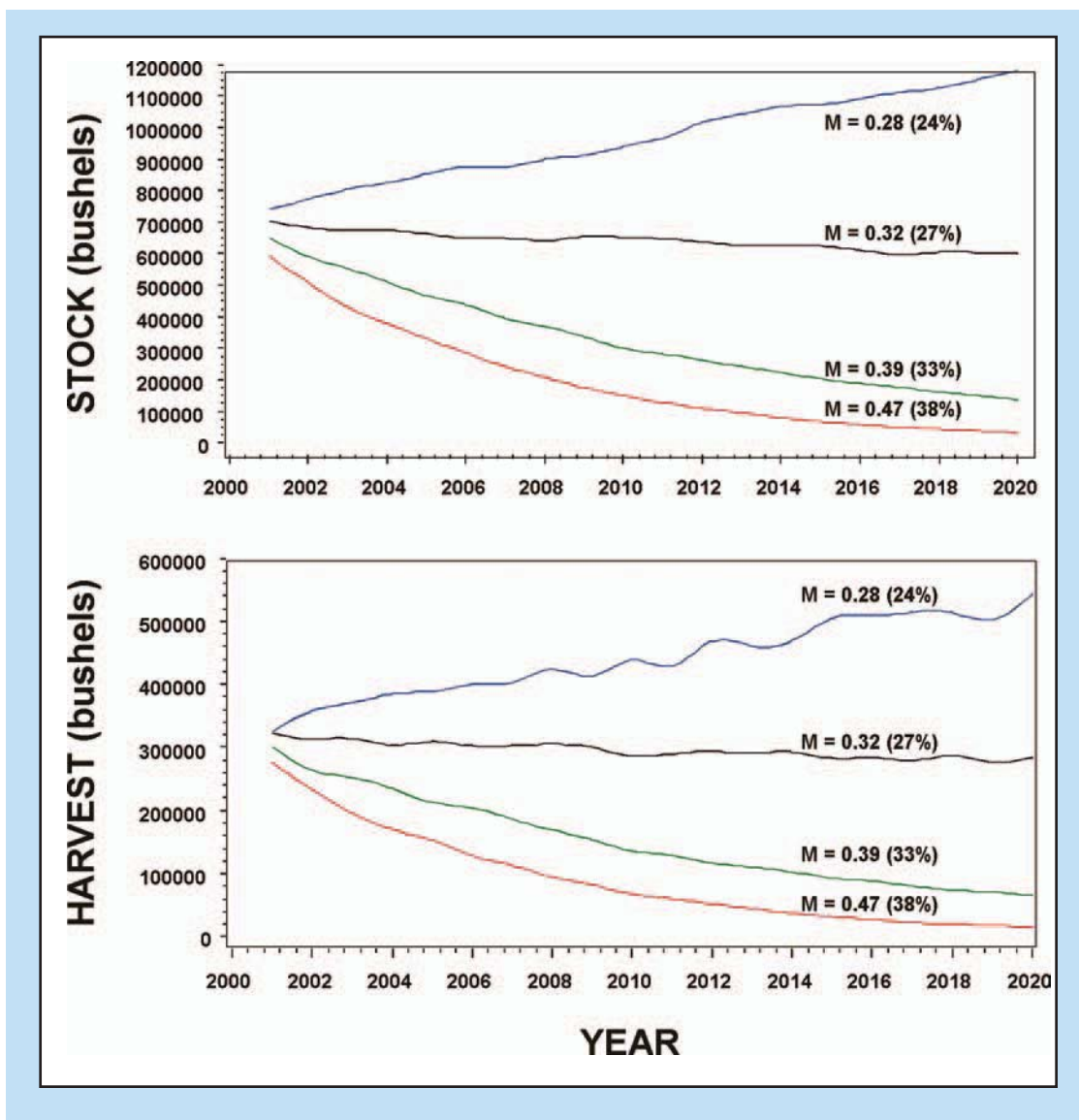


Figure 1. Forecasts of Maryland Chesapeake Bay market oyster stock (top plot) and harvest (bottom plot) based on four scenarios with different natural mortality rates. M = instantaneous rate of natural mortality; finite annual rates of natural mortality are shown in parentheses. $M = 0.39$ is the mean observed rate from 1985-2000, $M = 0.47$ is a 20 percent increase, $M = 0.32$ is a 20 percent decrease, and $M = 0.28$ is a 30 percent decrease in the instantaneous natural mortality rate.

lates harvestable stock and harvest based on 16 years of monitoring data from which variable parameters for natural mortality, fishing mortality and recruitment to the harvestable stock have been calculated. Fishing mortality and recruitment are random variates with the baseline means and standard deviations from the time series. The natural mortality parameter in the model is based on 16 years of box counts (empty articulated oyster shells) at 43 oysters bars in Maryland. Predation, disease, senility and other causes of natural mortality are implicit in the mean and standard deviation of the mortality term.

In the following simulations, the means of natural mortality were varied to illustrate long-term reductions or

increases in salinity (Figure 1). The graphs show the means of 1000 simulations for each scenario for each year, smoothed by a trend line. The parameters fishing mortality, natural mortality, and recruitment (gross recruitment to the market oyster stock) varied randomly about the observed or simulated mean, constrained by the observed standard deviations, which were not varied. A long-term reduction in salinity would be expected to result in fewer oyster mortalities due to *P. marinus* or *H. nelsoni* infections. Higher mortalities would be expected with long-term increases in salinity.

The mean rates of market oyster mortality modeled are 24 percent, 27 percent, 33 percent and 38 percent. The

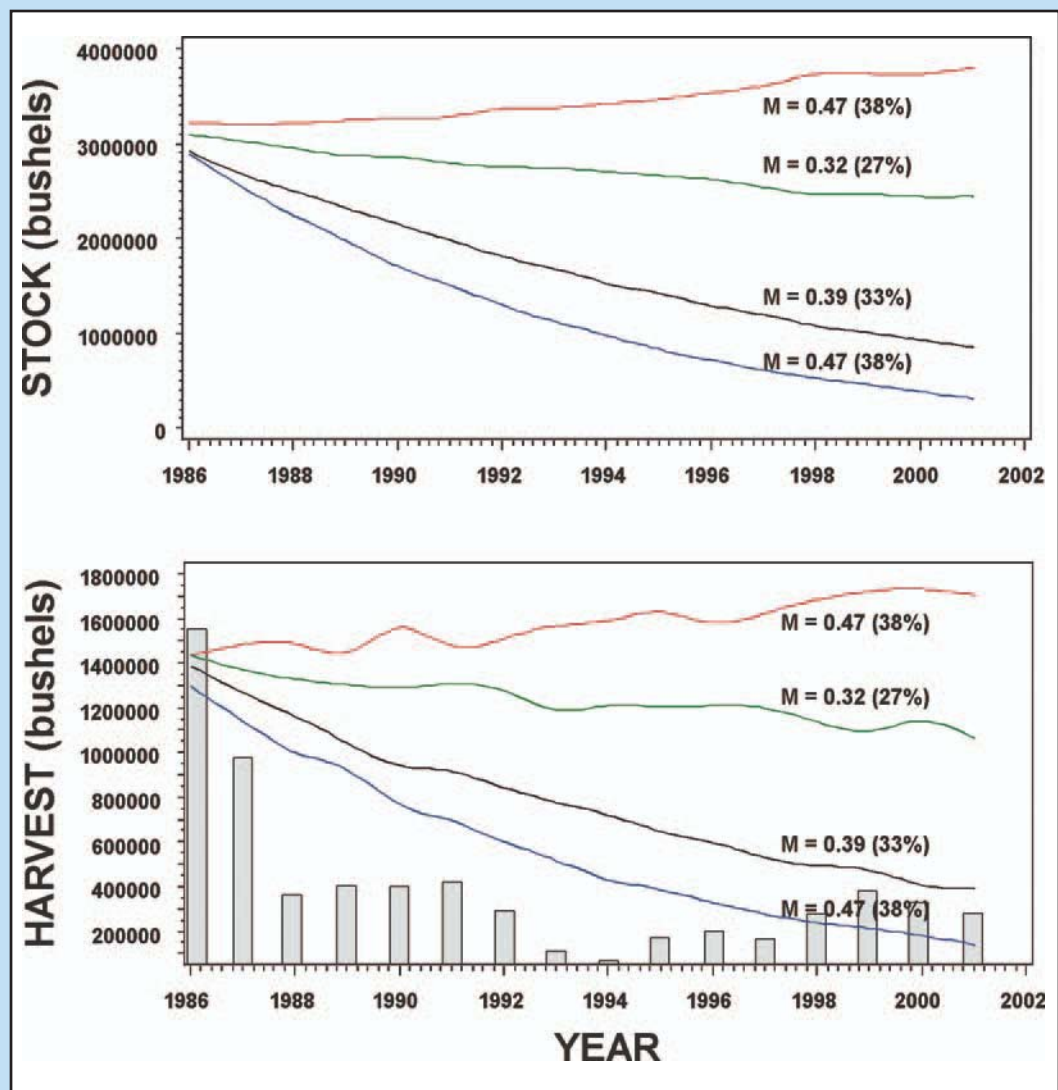


Figure 2. Hindcasts of Maryland Chesapeake Bay market oyster stock (top plot) and harvest (bottom plot) based on four scenarios with different natural mortality rates. Bars represent actual reported harvest. M = instantaneous rate of natural mortality; finite annual rates of natural mortality are shown in parentheses. $M = 0.39$ is the mean observed rate from 1985-2000, $M = 0.47$ is a 20 percent increase, $M = 0.32$ is a 20 percent decrease, and $M = 0.28$ is a 30 percent decrease in the instantaneous natural mortality rate.

baseline mean from 16 years of monitoring data was approximately 33 percent ($M = 0.39 \pm 0.25$ standard deviation). That is, 33 percent of the market oysters died in an average year. A simulated 20 percent reduction in instantaneous natural mortality rate equated to 27 percent, a 30 percent reduction to 24 percent of oysters dying per year; 38 percent annual mortality was a simulated 20 percent increase in natural mortality rate.

Note the vertical scales are different on the graphs. The simulated salinity increase results in the stock crashing over a 20-year period, more rapidly than the predicted decline at the observed natural mortality rate. Only a

modest increase in stock, or no increase, occurs with simulated decreases in salinity over the same time period. Based on the time series of data used to parameterize the model, the mortality rates simulated can be used to predict the corresponding changes in salinity. A linear regression of arcsine-transformed annual oyster mortality against salinity (measured in the fall at each of 43-64 oyster monitoring sites in Maryland) was employed to give indications of mean salinities associated with the simulated mortality rates. The relationship was significant ($p < 0.0001$) but imprecise ($r^2=0.08$, $N=444$). We note that the salinity measurements were made only once per

year at each site, and that factors other than salinity influence diseases and mortality. Nevertheless, salinity was significantly correlated (Spearman's r ; $p < 0.0001$) with *P. marinus* prevalence and infection intensity, and with *H. nelsoni* prevalence (*H. nelsoni* intensity was not measured quantitatively). Predicted salinity at the observed mean annual mortality rate (33 percent) was 16.5 ppt, at 24 percent mortality 11.9 ppt, at 27 percent mortality 13.6 ppt, and at 38 percent mortality 18.3 ppt. This analysis suggests that long-term mean salinity would have to decrease by about 3 ppt to prevent further decline of the market oyster stock, and by nearly 5 ppt to achieve a substantial increase in stock in the absence of other interventions.

Jordan and Vanisko's population dynamics model, when parameterized separately for 3 salinity zones in northern Chesapeake Bay, indicated that an oyster fishery could not be sustained when long-term mean salinity exceeded 14 ppt because of the high mortality rates associated with Dermo and MSX diseases. The model further shows that oyster fisheries could be sustained or restored at lower salinities with moderate reductions in fishing mortality to compensate for disease-related mortality.

Hindcasts simulate the past and are used to verify models by comparing results with actual data. Figure 2 shows the same variations in natural mortality depicted in Figure 1, simulated for the time span of data used to parameterize the model. The baseline simulation ($M = 0.39$) reproduces the trend and end members of harvest over this period reasonably well, but does not capture the intervening variations in harvest. These variations might be explained by interannual environmental effects on recruitment and natural mortality, in combination with non-random variations in fishing mortality. No model-independent estimates of absolute stock size exist for comparison with the simulated stock.

Lessons Learned

Development of population dynamic models to predict oyster disease in the Chesapeake Bay requires a baseline of information acquired through regular, and costly, monitoring of oyster populations.

Jordan and Vanisko's population dynamics model may support forecasting oyster disease in the upper Chesapeake Bay. The primary purpose for its development; however, is to predict outcomes of management options for oyster fisheries enhancement and restoration. Additional verification of the model as a forecasting tool is needed.

There are gaps in knowledge related to the biology and life cycles of the oyster parasites, reservoirs of infection, oyster defense mechanisms, susceptibility and resistance to disease, and host-parasite interactions that are not adequately addressed in stock assessment models. Further, the identification of two *Perkinsus* species, *P. marinus* and *P. chesapeaki*, in softshell clams *Mya arenaria* of the upper the Chesapeake Bay (McLaughlin and Faisal 2000) may compound the interpretation of prevalence data used in present oyster-disease model development. Should these gaps be filled, prediction of oyster disease outbreaks using computer-based simulation and mathematical models programs will become more precise.

Forecasts of pending oyster disease may be useful to state agencies and shellfish farmers; however, announcing such forecasts to the general public may negatively affect consumer confidence and expectations.

Next Steps

Research to better understand shellfish biology and pathology, the biology and life cycles of parasitic pathogens, and host-pathogen interactions needs to be supported. For example, the invertebrate immune system and defensive mechanisms have yet to be completely understood. In addition, the impacts of other stressors on the health of estuarine and marine organisms such as disturbances in food availability, reproductive failures, increased sedimentation, presence of contaminants or toxins in the water, and the effects of multiple stressors and increased susceptibility to disease remain to be understood. All of these variables are critical to the development of predictive models.

The relationship between elevated salinity and increased prevalence of oyster disease has been well documented. Thus, an alternative to the stock assessment or population dynamics modeling approach to predicting oyster disease may be to directly predict changes in salinity. For example, stream flow data recorded by United States Geological Survey or climate models could be used to predict variations in salinity in the Chesapeake Bay. A predicted change in salinity could then provide the basis for predicating increases or decreases in oyster disease. The ability to predict salinity changes could be further applied to forecasts of ecological events of importance to other users of the Chesapeake Bay. Reduced salinity due to increases in stream flow may provide an opportunity to predict a higher potential for fish kills resulting from higher nutrient levels, stimulation of algal blooms, and decreased oxygen in the bay. Conversely, predicted

increases in salinity may be used to forecast a wider distribution and higher number of sea nettles in Chesapeake Bay waters.

Emerging diseases in marine species have been associated with ongoing degradation of coastal habitats and global climate change (Harvell *et al.* 1999). The limited amount of data available on the epidemiology of marine diseases is a serious obstacle to linking climate changes with apparent increases in outbreaks of disease (National Research Council 2001). Collection of long term spatially resolved disease surveillance data and centralization of data into electronic databases would provide baselines against which changes could be detected and models developed and validated (National Research Council 2001). Interdisciplinary collaboration among epidemiologists, ecologists, meteorologists, climate modelers, information technologists, and other scientists should be fostered at national and international levels. Expansion of recent efforts to apply remote sensing and other satellite systems to epidemiological investigations should be supported.

Conclusions

Scientists and fishery managers are able to make short-term predictions regarding potential outbreaks of disease in oyster populations due to well established associations between salinity fluctuations and disease. Competency in predicting medium- or long-term fluctuations in oyster disease and other ecological events is severely limited by the inability to accurately predict medium- and long-term fluctuations in weather patterns such as occurrence and duration of drought conditions.

A multi-disciplinary approach to understanding disease pathogenesis and mechanisms in estuarine and marine organisms and the ecological processes influencing disease dynamics is critical for effectively anticipating and predicting disease outbreaks.

Acknowledgements

The authors gratefully acknowledge Fred Kern (NOAA Cooperative Oxford Laboratory, Oxford, Maryland) and Chris Judy (Maryland Department of Natural Resources, Annapolis, MD) for insights into oyster disease forecast development. We thank Don Scavia, Nathalie Valette-Silver, Fred Kern (NOAA) for their constructive reviews. Jane Keller (NOAA) provided editorial assistance. Maryland Department of Natural Resources staff conducted annual Chesapeake Bay oyster disease surveys.

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Ecological Forecasting of Impacts of Ponto-Caspian Species in the Great Lakes: Describing, Understanding, and Predicting a System in Transition

Henry A. Vanderploeg¹

Abstract

*This paper summarizes results relevant to ecological forecasting from a study that described, explained, and predicted Great Lakes ecosystem impacts of six Ponto-Caspian (the region including the Caspian, Black, and Azov Seas) endemic species of mussels, crustaceans, and fishes that recently invaded the Great Lakes via ballast water (Vanderploeg et al. 2002). These are the zebra mussel (*Dreissena polymorpha*), the quagga mussel (*D. bugensis*), the predatory cladoceran (*Cercopagis pengoi*), the benthic amphipod (*Echinogammarus ischnus*), the round goby (*Neogobius melanostomus*), and the tubenose goby (*Proterorhinus marmoratus*).*

The ecology and possible mechanisms of ecosystem impact were reviewed for each species using information from a variety of studies and sites, including the Great Lakes. This information was combined with experiments and monitoring on the Great Lakes to describe ecosystem change, the underlying invader species mechanisms of impact, and predict future changes.

Introduction

Several Ponto-Caspian endemics have recently invaded the Great Lakes: the zebra mussel (*Dreissena polymorpha*), the quagga mussel (*D. bugensis*); the predatory cladoceran (*Cercopagis pengoi*), the benthic amphipod (*Echinogammarus ischnus*); the round goby (*Neogobius melanostomus*), and the tubenose goby (*Proterorhinus marmoratus*). All these euryhaline species arrived in ballast water.

The Great Lakes have undergone and continue to undergo profound ecological changes in response to these invaders. There are also changes from oligotrophication, piscivorous fish stocking, and other exotic species, some of which are in concert with effects of the Ponto-Caspian species. So it is difficult to separate impacts from these invaders from other anthropogenic changes and makes it challenging to forecast the impacts of these species on water quality and fisheries of the Great Lakes.

It is important to document changes in the system since invasion and attribute them to the invaders. But it is also necessary to relate – ideally quantitatively – the

changes through a set of plausible underlying mechanisms to these recent invaders to define relative contributions of all factors. This is an overview of results from a large synthesis relevant to ecological forecasting of spread and ecosystem impacts of Ponto-Caspian species in the Great Lakes and other water bodies (Vanderploeg *et al.* 2002).

Case Study

The study began with a synthesis of the ecology of each Ponto-Caspian invader and the mechanisms they use to spread and affect ecosystems. This was derived from studies in the laboratory, in the Great Lakes, and in other systems.

To generate as holistic as possible case studies of invader-induced change, the underlying causes, and predictions for the future, information from the ecological synthesis was applied to areas in the Great Lakes where the ecosystem has been monitored. This exercise was extremely important not only for impact description but also for revealing mechanisms and interactions not suspected from prior knowledge of each

¹ GLERL/NOAA, 2205 Commonwealth Blvd., Ann Arbor, Michigan 48105

invader alone or from current understanding of aquatic ecosystem processes.

The areas of study included shallow bays and basins – Saginaw bay (inner bay mean depth = 5.1 m), western Lake Erie (mean depth = 5.7 m), Lake St. Clair (mean depth = 3.0 m) Long Point Bay on Lake Erie (inner bay mean depth = 2 m) – as well as deep basins of Lakes Erie, Ontario and Michigan (Figure 1). Long Point Bay was of particular interest in that it is one of the most important staging areas for waterfowl in North America.

Lessons Learned

The zebra and quagga mussels are changing ecosystem function through ecosystem engineering (increased water clarity and reef building), fouling native mussels, high particle filtration rate with selective rejection of colonial cyanobacteria in pseudofeces, and alteration of nutrient ratios. They also facilitate the rapid spread of their Ponto-Caspian associates, the benthic amphipod and the round goby, which feed on zebra mussels.

High concentrations of zebra mussels and quagga mussels in shallow regions of the Great Lakes (<30 m) continue to dominate energy flow and nutrient cycling by their high filtration rate and nutrient excretion. Dreissenids are able to clear the water in spring in bays and shallow basins of the Great Lakes, but not in summer because of reduced filtering and selective rejection of unpalatable algae in pseudofeces. This same selective rejection mechanism may be increasing the probability of toxic blooms of the colonial cyanobacterium *Microcystis* on Saginaw Bay and western Lake Erie.

Dreissena, round gobies, and *Echinogammarus* form an ‘invasional meltdown’ assemblage in the Great Lakes that is becoming an important part of the food web in many areas of the Great Lakes. That is, rather than restricting invasion by other species, zebra mussels facilitated invasion of the Great Lakes by their Ponto-Caspian associates. *Dreissena* provides substrate (shell) and food (pseudofeces and feces) to *Echinogammarus*. Round gobies and other fishes feed on *Echinogammarus*. Round gobies are efficient predators of

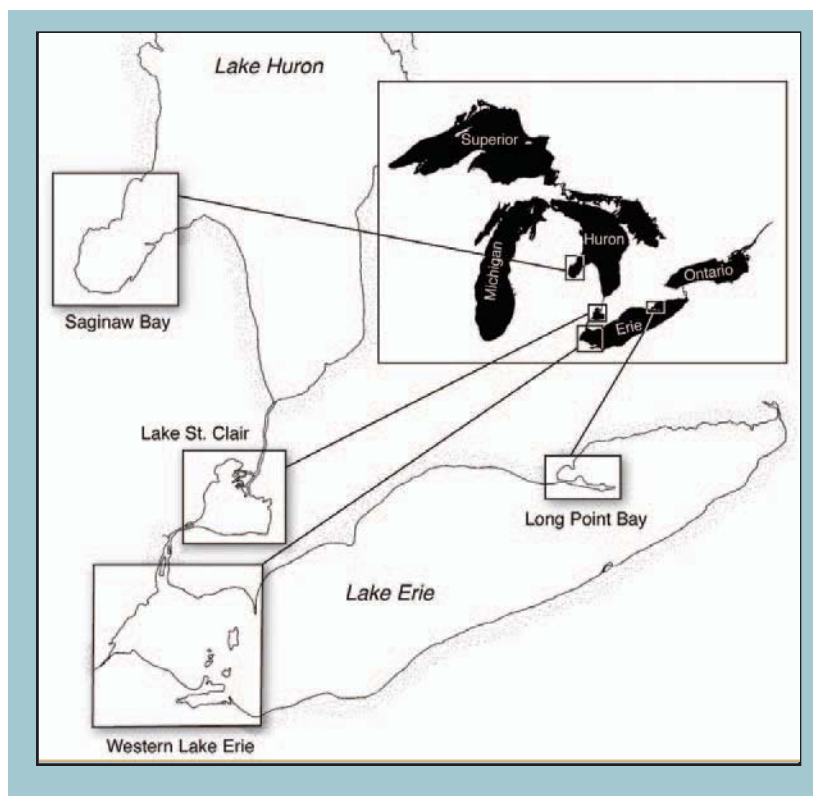


Figure 1. Case study sites, where monitoring and experimental work permitted description and mechanisms of invader impact.

Dreissena, and are eaten by a variety of game fishes. Tubenose gobies, which are too small to feed on *Dreissena*, are not expected to cause significant impacts.

In some shallow systems, certain sport fishes and waterfowl have benefited from habitat and food web changes caused by the zebra mussel (Figure 2). Zebra mussels are a food source for diving ducks. After the zebra mussel invasion of Long Point Bay, waterfowl use days increased from 0.04 to 3.5 million. Game fishes such as smallmouth bass and muskellunge have additional feeding and spawning habitat created by the macrophytes that grew because of the increased light resulting from suspended particle removal by zebra mussel filtration. Improved habitat and predation on round gobies have resulted in increased populations of these game fishes. In Lake St. Clair, a particularly important area for sport fishing, abundance of smallmouth bass and muskellunge increased 3-fold.

On the other hand, there are negative consequences to the new food web. Populations of walleyes, which prefer low light, have decreased by 50 to 75 percent in Lake St. Clair and the reproductive success of diving ducks may be affected by contaminants associated with zebra mussels. Thus, there is concern that Long Point

Bay may be a magnet for contamination of continental populations of certain diving ducks. And the zebra mussel → round goby → game fish food chain may increase contaminant loads in game fishes.

Zebra mussels, through their fouling native unionid mussel shells, have removed them from most areas except for refuge areas in marshes and wave-exposed shallow areas. Surprisingly, despite dominance of shallow benthic communities by dreissenids, non-dreissenid benthic invertebrates other than mussels remained at pre-invasion levels. The provision of substrate by mussel shells and increased primary production of benthic plants enhanced by increased water clarity (Figure 2) may have compensated for the expected loss of the phytoplankton food base removed by mussel filtering. Microzooplankton have strongly declined in shallow areas as a result of dreissenid filtering, but crustacean zooplankton have not been greatly affected in areas examined, and growth of fish dependent on them has not decreased.

The quagga mussel, a close relative of the zebra mussel that can colonize soft substrates, has invaded many deep areas of the Great Lakes (>30 m) and is extending the impacts of *Dreissena* to this region since it too is a voracious filter feeder. *Diporeia*, an amphipod that has historically dominated the benthos in deep water (>30 m), is now in decline in many of the Great Lakes (Michigan, Huron, Ontario, Erie); this decline may harm many fishes that prey on it. It is hypothesized that dreissenids are intercepting algae before they can reach *Diporeia* at the sediment surface.

Cercopagis pengoi, a predatory cercopagid cladoceran with large tail spine, is protected from predation by small fishes because of its tail spine, and will compete with small fishes for cladocerans and other zooplankton (Figure 3). It is now found in abundance in summer in both nearshore and offshore waters of Lake Ontario and inshore waters of Lake Michigan. Distribution and impact of *Cercopagis* in Lake Michigan may be controlled somewhat by predation by *Bythotrephes*,

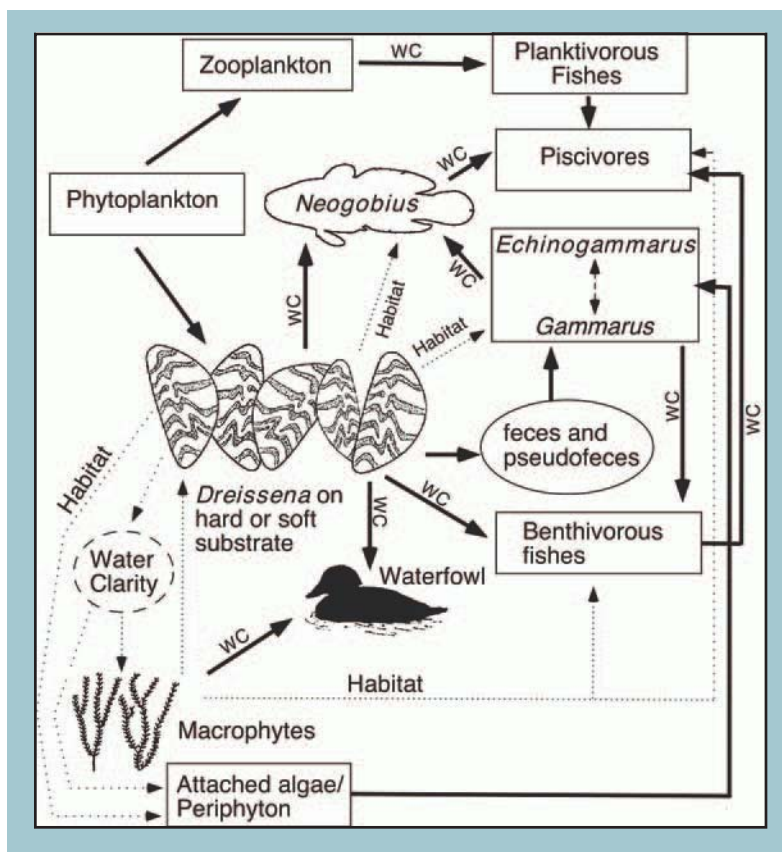


Figure 2. Food-web and ecosystem-engineering mechanisms of impact of the Ponto-Caspian benthic assemblage (*Dreissena*, *Neogobius*, and *Echinogammarus*) in bays (e.g. Saginaw Bay, Long Point Bay), interconnecting waterways (Lake St. Clair) and nearshore or shallow (Lake Erie) areas of the Great Lakes. Solid arrows indicate food-web connections. The dashed double-headed arrow indicates potential competitive interactions between the native amphipod, *Gammarus*, and the non-indigenous amphipod *Echinogammarus*, and the dotted arrows indicate the influence of ecosystem engineering through the mechanisms of habitat provision of mussel shells (autogenic engineering) and water clarity (allogenic ecosystem engineering). WC (for water clarity) is placed next to solid arrows for predation by vertebrate (visual) predators (fishes and waterfowl) to indicate possible increases in predation rate associated with increased water clarity that results from zebra mussel filtering. Because their filtering activities increase the water clarity, zebra mussels promote growth of benthic plants, providing habitat for fishes. The shells of the mussels provide habitat for a variety of invertebrates. The zebra mussel → round goby → piscivore food chain and increased macrophytes (cover and spawning habitat of fishes) have resulted in increased production of certain fish species in shallow areas.

another larger cercopagid that invaded the Great Lakes from northern Eurasia in 1985.

Next Steps

The Great Lakes ecosystems have been radically modified, are in a state of transition, and impacts will continue to evolve. For example, over 90 percent of the secondary production, grazing, and nutrient excretion in many areas of the Great Lakes is associated with zebra and quagga mussels. Thus, old rules of ecological forecasting and management no longer apply.

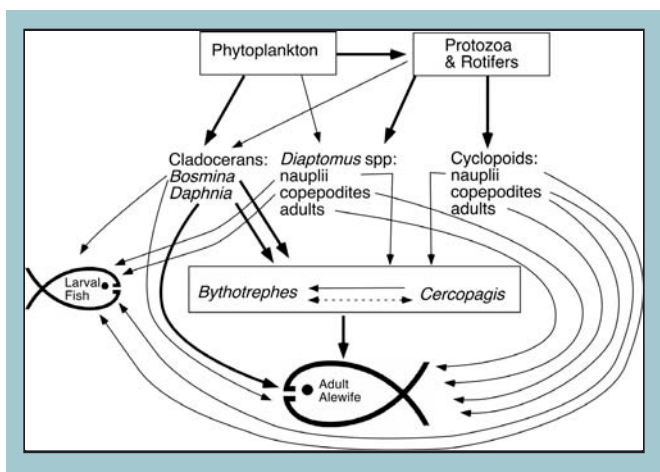


Figure 3. The role of *Cercopagis* and *Bythotrephes* in the food web of larval fish. *Cercopagis* and *Bythotrephes* prey on many of the same zooplankton that larval fish feed on. Therefore they are in competition with them. *Bythotrephes* feeds on and competes with (dotted double headed arrow indicates competition) *Cercopagis*, and adult alewives feed on *Bythotrephes*. By preying on *Bythotrephes*, adult alewives may be allowing *Cercopagis* to find a niche.

Therefore, careful monitoring of the Great Lakes will be necessary to continue to observe new impacts as they emerge, and experimental work must be done to develop new rules for making ecological forecasts and managing these systems

Conclusion

Predictions of impact require intimate knowledge of food web structure and function in the area of concern. Extrapolations of impacts from other sites, such as those in Eastern Europe, often have limited value for the Great Lakes because of differences in lake morphometry, nutrient loading, and species composition, particularly fishes. The ability to understand present impacts and predict future impacts is limited by insufficient monitoring and understanding of Great Lakes food webs. The Great Lakes need to be carefully monitored to observe changes and to evaluate impacts of these changes. Experimental work in conjunction monitoring will be necessary to develop ecological forecasting tools and models.

Acknowledgements

Thanks to the Canadian Journal of Fisheries and Aquatic Sciences for permission to reprint Figure 2.

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Extreme Events and Ecological Forecasting

R. Wayne Litaker and Patricia A. Tester¹

Abstract

Almost all extreme events lasting less than several weeks that significantly impact ecosystems are weather related. This review examines the response of estuarine systems to intense short-term perturbations caused by major weather events such as hurricanes. Current knowledge concerning these effects is limited to relatively few studies where hurricanes and storms impacted estuaries with established environmental monitoring programs.

Freshwater inputs associated with these storms were found to initially result in increased primary productivity. When hydrographic conditions are favorable, bacterial consumption of organic matter produced by the phytoplankton blooms and deposited during the initial runoff event can contribute to significant oxygen deficits during subsequent warmer periods. Salinity stress and habitat destruction associated with freshwater inputs, as well as anoxia, adversely affect benthic populations and fish. In contrast, mobile invertebrate species such as shrimp, which have a short life cycle and the ability to migrate during the runoff event, initially benefit from the increased primary productivity and decreased abundance of fish predators.

Events studied so far indicate that estuaries rebound in one to three years following major short-term perturbations. However, repeated storm events without sufficient recovery time may cause a fundamental shift in ecosystem structure (Scavia et al. 2002). This is a scenario consistent with the predicted increase in hurricanes for the east coast of the United States.

More work on the response of individual species to these stresses is needed so management of commercial resources can be adjusted to allow sufficient recovery time for affected populations.

Introduction

Short-term natural events (days to weeks) causing significant stress to estuarine environments are primarily weather related: hurricanes, tropical storms or northeasters. These powerful weather systems damage habitats through direct wind or wave action and alter habitats by loading freshwater, nutrients, and pollutants into the estuary. On a global time scale, periodic disturbances by these weather systems are highly predictable along the Gulf of Mexico and the eastern coast of the United States. Precisely predicting when and where they will occur on shorter time scales (year-to-year), however, is highly unpredictable.

This high degree of variation makes detailed studies of the short and long-term effects of storms on estuarine environments and populations difficult. Most of the objective knowledge concerning the effects of these

perturbations comes from ongoing long-term studies that coincided with a hurricane passing through the study area. The notable studies are the effect of Hurricane Agnes on the Chesapeake Bay in 1972, and Hurricanes Fran (1996) and Floyd (1999) on North Carolina estuaries (Figure 1).

As a result of the limited number of long-term studies, separating the effects of these storms from normal environmental variation can only be estimated from current studies of less extreme runoff and wind events. These smaller events generally occur much more frequently and hence are easier to study because the effects of a larger than normal spring runoff or a major wind event often mimic, to a lesser degree, the stress and excess nutrient loading that accompanies a tropical storm or hurricane.

The remainder of this paper briefly reviews what can

¹ Center for Coastal Fisheries and Habitat Research, National Ocean Service, NOAA, 101 Pivers Island Road, Beaufort, NC 28516

accurately be predicted about the effects of major storm events on estuarine systems, what information managers will need to adequately respond to these storms, and what research is needed before there can be accurate forecasting of these effects.

To understand the effects of short-term, intense perturbations such as hurricanes on estuarine systems, it is important to consider three specific aspects of each storm: the amount of storm runoff, the time of year the storm occurs, and the amount of direct physical damage caused by the storm. The amount of rainfall and subsequent runoff varies tremendously between storms and directly determines the salinity stress. In addition, it directly affects the overall amounts of nutrients, sediments, and toxicants introduced into the estuary. Secondly, the time of year the storm occurs is critical because the total oxygen consumption by an estuary is directly related to ambient temperatures. The warmer the water, the more rapidly nutrients are taken up by phytoplankton and converted to carbon sources that promote bacterial growth and respiration. Given that warm water holds relatively little oxygen, this increased bacterial respiration often leads to hypoxia or anoxia. Lastly, storms vary greatly in the amount of direct physical damage they inflict through wind and wave action. This damage is of particular concern when it destroys environmentally sensitive areas such as breeding habitats of endangered species.

Case Study

There are no current models that adequately address how these intense short-term events will impact estuaries. Most of the available data on the response of estuaries to short-term, large scale perturbations comes from Hurricane Agnes's strike on Chesapeake Bay in 1972, and Hurricanes Fran and Floyd, which made landfall in North Carolina in 1996 and 1999 (Tester *et al.* 2003), respectively. Therefore, the following case study represents a conceptual model based on current knowledge of how various components of estuarine ecosystems are thought to respond to short-term, large-scale weather-related events.

Salinity Stress – The freshwater input during and after major storms, particularly hurricanes, determines the mortality among sessile organisms as well as displacement and mortality of mobile species (Jury *et al.* 1995). The mesohaline region of the estuary is generally the most impacted by runoff during these events. When runoff reduces the salinity in this region to less than 10

psu for more than a week, extensive mortality occurs among benthic organisms such as sponges, tunicates, bryozoans, coelenterates and mollusks (Andrews 1973, Smock *et al.* 1994). Some species may be eliminated.

Although species with pelagic larvae and fast growth rates generally repopulate the region within three months, repopulation may take years for species with limited dispersal capacity. Furthermore, if the salinity stress coincides with the breeding season of a particular organism, fecundity and survival will be affected. This must be taken into account when anticipating the effects of hurricanes and tropical storms on an estuary (Livingston *et al.* 1999). The closer the storm occurs to the peak breeding season, the greater the chances of severe population impacts.

The consequences of acute or prolonged salinity stress on fish stocks are less understood. Preliminary studies indicate that runoff causes extensive mortalities of estuarine fish populations (Dorf and Powell 1997, Paerl *et al.* 1998). The long-term effects on each population are now beginning to be rigorously quantified (Crowder, personal communication). Considerable work remains to adequately quantify losses and determine how mortalities affect future reproduction and recruitment. These studies should also include a rigorous examination of how food web disruption caused by runoff affects the recovery of various fish species, particularly mesohaline bottom feeders.

Little is known about how changes in the lower trophic levels caused by increased runoff impact fish populations. This information is critical to accurately understand how fish stocks will be affected by runoff events. Stressed fish are more susceptible to disease and parasites (Noga 2000, Paerl *et al.* 2001). Therefore, fish mortality due to these causes should also be investigated immediately following the initial stress event and in subsequent years when lowered food abundance and anoxic events may again stress these populations. Ideally, once the general effects of runoff-induced mortality and stress are known, local fisheries managers could adjust catch limits to enable the recovery of fish populations.

Since prolonged salinity stress will severely affect an estuarine ecosystem, one critical need is to accurately predict runoff in a watershed under different conditions. While it provides information for implementing more accurate evacuation plans, having a precise estimate of the potential runoff also will help sewage treatment facilities and farm operations reduce unintended discharges. Large releases of waste material adversely af-

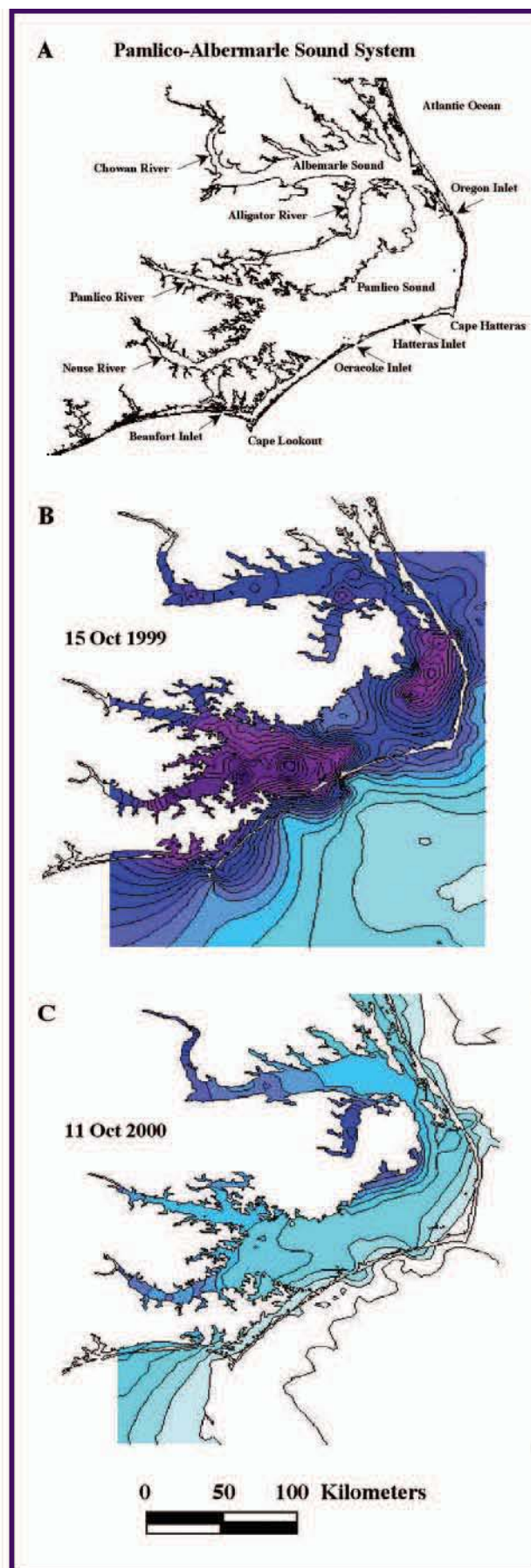
fect estuarine water quality for weeks. They also pose a public health threat (Mallin *et al.* 1999). The NOAA weather service, academic scientists and the USGS have made great strides in developing runoff prediction models (Pietrafesa *et al.* 1997). However, much work remains to improve these models, as was shown from the unexpectedly severe flooding in North Carolina during Hurricane Floyd.

Nutrient Loading – Besides reducing the overall salinity regime, runoff adds inorganic nutrients, dissolved organic nutrients, and sediments to the estuary (Paerl *et al.* 1998). More than 50 percent of the annual nutrient loading can occur during a single, large runoff event (Hahl 1981).

Of all nutrients in the runoff, nitrogen and to a lesser extent, phosphorous, play a direct role in stimulating phytoplankton growth (Howarth 1988). In addition, most hurricanes and tropical storms occur in the warmer months when the phytoplankton are actively growing. Under these conditions, phytoplankton take up the inorganic N and P and grow rapidly. These same results occur during smaller storm events that also contribute nutrients into the estuary.

The biomass increase generally occurs within a week following nutrient input and lasts from a few weeks to three months, depending on the extent of loading (Loftus *et al.* 1972, Flint 1985, Bennett *et al.* 1986). In estuaries where flushing rates are high, excess nutrients and cells may be transported out of the system, resulting in a smaller bloom than found in the lagoonal systems that efficiently trap nutrients.

Figure 1. A) Record rainfall and extensive flooding in the Pamlico-Albermarle Sound system (PAS) accompanied hurricanes Dennis, Floyd, and Irene, which made landfall in coastal North Carolina in September-October 1999. The physical and chemical changes following this massive freshwater influx served as a large-scale natural experiment. In a matter of weeks, half of the annual supply of nutrients was delivered to the PAS as freshwater runoff. The relative abundance and distribution of colored dissolved organic matter (CDOM) in the PAS system are shown during the flooding events (B, 15 October 1999) and under a normal flow regime 1 year after the hurricanes (C, 11 Oct 2000). Light blue indicates relatively low CDOM, whereas dark purple indicates very high levels of CDOM in the system. The CDOM inputs into the PAS originate from the surrounding pine forests and swamp. Hence, CDOM serves as a visual proxy for both the amount of runoff from the surrounding coastal regions and overall nutrient inputs. The CDOM data were gathered during a series of over flights of the PAS using a NOAA observation plane equipped with NASA's airborne oceanographic light detection and ranging instrument (AOL3). Laser induced fluorescence (LIF) at 355 nm served as a measure of CDOM (Tester *et al.* 2003).



Phytoplankton blooms contain a large quantity of carbon, the base of the food chain. Zooplankton grazing communities would generally be major consumers of this increased productivity. However, zooplankton biomass is often significantly decreased following major runoff events (Shaheen and Steimle 1995). This further favors the development of phytoplankton blooms and subsequent bacterial use of newly-produced carbon that would otherwise have been consumed by the zooplankton community.

As phytoplankton cells in the bloom die and sink out of the water column, bacteria remineralize the phytoplankton. During warmer months, the complete remineralization of N and P is often complete within a few hours to several days (Nixon 1981). This efficient recycling mechanism amplifies the productivity stimulated by the initial nutrient input. Frequently, macrophytic algae also respond to the increased nutrient load and become so abundant they also contribute to severe anoxic events (Sfriso *et al.* 1987, Barranguet and Alliot 1995).

If runoff-induced phytoplankton blooms occur in stratified salt wedge estuaries, the cells will settle into a high-salinity wedge. This wedge does not mix effectively with the upper layer. Studies of salt wedge estuaries that have experienced a runoff event and bloom during warmer months indicate that if both the carbon supply from sinking phytoplankton and temperatures are sufficiently high, extensive bacterial respiration will cause the lower salt wedge to become hypoxic or anoxic (Taft *et al.* 1980, Bennett *et al.* 1986).

Transient hypoxic or anoxic events similarly occur in shallow estuaries when the primary production (phytoplankton or macrophytes) is sufficiently high so the biological oxygen demand (BOD) consumes all or most of the oxygen in the water column (Barranguet and Alliot 1995, Conley *et al.* 2000). Sometimes these hypoxic or anoxic zones can be quite large and lead to additional fish kills or the decimation of the benthic microfauna (Bianchi *et al.* 2000). Though anoxia tends to slow down remineralization of organic compounds in the benthos, significant amounts of ammonium and phosphate can be released (De Casabianca *et al.* 1997). If nutrients released below the pycnocline are transferred to the upper layers of the estuary via wind, tidal, or shear mixing forces, additional phytoplankton/macrophyte growth will occur, which will subsequently increase the BOD (Malone *et al.* 1986). This cycle accounts for the increasing number of anoxic events with increased nutrient loading in estuaries.

Accurate models are needed to address the link between hydrography, nutrients, phytoplankton biomass, and the potential for anoxia events. The latest generation of Doppler radar current meters, buoyed meter arrays, and real-time satellite uplinks make gathering these data much more practical and should lead to ever improving circulation models. Improved models and real-time monitoring will give environmental managers better information relating to potential anoxic events and their potential to affect environmental and public health. In addition to determining the effects of hurricanes, these models would also help predict the potential effects of increased nutrient loading from aquaculture, waste treatment plants, and agricultural runoff.

Sediment Loading – Runoff from tropical storms and hurricanes can load more than a year's worth of sediment into an estuary within several weeks (Gross *et al.* 1978). Sedimentation affects the benthic community directly by burying many benthic organisms so they cannot feed effectively. Indirectly, the sediments represent a large source of organic carbon, exacerbating the BOD problem during the warmer months (Paerl *et al.* 1998). The initial anoxic or hypoxic conditions produced by storm runoff are often not the result of bacterial action on increased algal production, but rather the immediate metabolism of organic material in the sediment. Over the next three to five years, additional nutrients will be released as this organic matter continues to be remineralized by benthic bacteria. The particulate organic matter therefore functions as a 'time-release fertilizer,' increasing productivity and resulting in more frequent and severe anoxia events in subsequent years (Paerl *et al.* 2001).

The increased productivity following runoff events can have both positive and negative effects. On the positive side, the increased productivity can result in a rapid increase in the abundance of certain invertebrates. For example, a general phenomenon is that shrimp harvests increase substantially the year after hurricanes or tropical storms cause large runoff events (Flint 1985). Presumably the shrimp, either directly or indirectly, take advantage of the increased primary productivity and the disruption of predatory fish populations. On the negative side, certain parts of an estuary, with fairly low flushing rates and high recycling rates, may experience nuisance algal blooms when there is little wind. These blooms greatly reduce the recreational value of the estuary and can supply enough carbon for the bacterial community to create anoxia and related fish kills.

Toxic Chemicals – The extent to which hurricanes load toxicants into estuaries depends largely on upstream sources. Large amounts of water coming into the system often dilute dissolved toxic compounds below levels of detection. Given sufficient outflow from the estuary during the event, they can be largely eliminated from the estuary. In that respect, the runoff can cleanse the estuary.

When the toxic substances are bound to sediment particles, the impact is less clear. If the upstream sediment is contaminated with heavy metals, pesticides, or other harmful chemicals, these substances can be retained in the system. After reaching the benthos, they either remain inertly attached to the particle or are mobilized to enter the food chain. If the sedimentation rate is sufficiently high, some of the labile material will be buried and effectively sequestered unless released by bioturbation. The amount of bioturbation following the sedimentation event is, therefore, an important component determining how rapidly toxic compounds enter the food chain.

In cases where bioturbation is important, the remineralization process may be delayed for months after the runoff event until the benthic community is reestablished. Similarly, physical resuspension of already contaminated sediments by wind and wave action can contribute to the renewed cycling of once sequestered toxic compounds (Tissue *et al.* 1992). To adequately understand how runoff will affect the distribution and cycling of toxic compounds in estuarine systems, more research is needed to estimate potential toxic loads in upstream watersheds and understand how certain toxins are transformed and transferred at different salinity regimes within the estuary.

Much is known regarding the migration and transformation of heavy metals, pesticides, and other organic compounds; however, accurate loading estimates are often lacking. To obtain a realistic picture of the potential threat to an estuary, toxic compounds will require frequent monitoring, particularly during runoff events. Estuaries receiving significant inputs of industrial waste or pesticides should be given priority monitoring to identify any significant deterioration in environmental quality. Given the limited resources available to environmental managers, the toxicants measured should also be carefully selected so the focus is on those most likely to cause adverse effects on human or animal health. Reviews should be made periodically to determine if other critical toxins should be measured or if more ap-

propriate and economical detection methods have been developed.

Wind and Wave Damage – Hurricanes, tropical storms, and particularly northeasters are also notable for the physical damage they cause to estuarine environments. Much of this damage may persist for years. In the broadest ecological terms, these physical disturbances can all be viewed as a means to open niche space. No matter how severe the disruption, the same or different species will eventually exploit the newly-created habitat.

Vascular plant communities surrounding estuaries are most visibly affected by major storms. Mortality and damage occurs from uprooting, stripping leaves and limbs, and salt spray damage. The periodic damage inflicted by hurricanes, therefore, plays a crucial role in the structure of these plant communities.

A good example is the mangrove community (Wanless *et al.* 1996). Hurricanes open space in mangroves by destroying trees, removing peat deposits, and by bringing poisonous anoxic sulfide rich sediments to the surface. Once the peat is gone, subsidence occurs; the sulfide released from the mud kills weakened trees and saplings. Despite this damage, the mangrove and other plant communities gradually respond with increased productivity after the storm and generally recover until the next storm produces similar damage.

The immediate effect of these storms, however, is to degrade the habitat for many vertebrate and invertebrate species. Storm surges also frequently disrupt littoral sand communities and deposit large amounts of sand onto existing marshes further inland (Courtemanche *et al.* 1999), causing significant habitat destruction. Many species are forced to disperse to find food and shelter, accounting for the increase in reported hornet and wasp stings and snakebites following hurricanes (Valiela *et al.* 1998).

The physical disturbances caused by major storms cannot be altered and is primarily of concern when there are endangered species within the affected habitat. In this case, strategies should be sought to encourage establishing as many spatially-dispersed habitats as possible to ensure the best chances for survival.

The timing of the physical disturbance must also be considered when determining its specific impact on various estuarine species. Juvenile oysters, for example, sometimes benefit from the reduction in predators and open surfaces created by the physical disturbance dur-

ing storms. However, if the storm event occurs during maximal recruitment or significantly affects the ability of the adult population to produce larvae, the effects on the population will be negative (Livingston *et al.* 1999). Similar effects on different life cycle stages would apply to hundreds of different organisms. Given the limited knowledge about many estuarine organisms, it will be difficult to incorporate timing issues into models that will accurately predict the effect of hurricanes, tropical storms and northeasters on estuarine environments and populations.

Conclusions/Next Steps

There is a broad understanding of how major storms affect estuarine environments, both positively and negatively. Better models of rainfall, runoff and estuarine circulation are needed to improve predictive capabilities. Without robust basic hydrographic models, it will not be possible to understand and predict the effects of a storm on an estuary. Once these models are available, then realistic model inputs for nutrients, toxicants and other factors can be incorporated and first order predictions made. Once these predictions are available, monitoring real time storm events is needed to test the models' predictions.

A particularly important model parameter to evaluate will be nutrient loading. Having accurate loading information is crucial in determining whether nutrient loading is increasing through time, either through increased hurricane activity or anthropogenic inputs. If eutrophication is allowed to proceed, there will be adverse long-term biogeochemical and trophic changes in estuarine and coastal habitats (Paerl *et al.* 2001). Only by carefully monitoring nutrient inputs and using sophisticated modeling will it be possible to predict those changes and justify the political support for modifying land use, agricultural, and waste treatment policies sufficiently to avoid an unalterable degradation in estuarine environments.

The next steps will be to integrate long-term monitoring projects with intensive acute response monitoring to determine how large, short-term perturbations such as hurricanes affect estuarine ecosystems. This is particularly important given the greater eutrophication stress being placed on estuaries and predictions for increased hurricane activity (Goldenberg *et al.* 2001).

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Reversing the Decline of Sea Turtles: Insights from Life-History Population Models

Larry B. Crowder¹

Abstract

All seven sea turtle species on earth are threatened or in danger of extinction. The life history of turtles, including late age at first reproduction, was successful for over 100 million years. But in the last century, sea turtles have flirted with extinction.

It is partly related to reproductive success. Some sea turtles begin reproducing in their teens - others wait three decades or more. Nesting females lay several clutches of eggs every few years over a reproductive span of decades. In some areas nearly all the eggs were lost to human and other predators.

Fishing gear has added to the problem – it isn't restricted to small prey. So the protection afforded to sea turtles by their size and hard carapace is inadequate.

Beginning nearly 20 years ago, more and more precise life-history population models have been developed for sea turtles. But even simple population forecasting techniques provide useful guidance to researchers and managers who deal with long-lived species and limited data. Analyses of these models focus researchers and managers on the life stages and processes that would have the greatest impact on population recovery.

A loggerhead model showed protection of large juveniles and adults was critical to recovery and paved the way for requiring Turtle Excluder Devices (TEDs) in the US in 1990. TEDs, along with habitat protection, has brought Kemp's ridley back from as few as 800 nests per year in the mid-1980s to nearly 6,000 nests in 2002.

Recent analyses suggest that TEDs may need to be enlarged to protect adult loggerheads and leatherbacks. And of course, TED regulations need to be enforced for sea turtles to benefit.

Introduction

One of the key roadblocks to integration across scales in population biology has been the failure to include all factors that might influence the status of a population in a common framework. Generally, researchers study what is known (e.g., behavior, physiology, toxicology, fisheries interactions) and hope the insights gained can be brought to bear on understanding the dynamics of these populations.

Understanding these dynamics is particularly critical for research on threatened and endangered species like sea turtles. If factor A reduces fecundity by 10 percent - is that important? Or if factor B reduces juvenile survival by 5 percent – is that important? Is factor A more

important than factor B? It is necessary to put these factors into a common 'currency' to address the 'so what?' questions at the population level. Namely, age- or stage-specific effects on population vital rates must be estimated (e.g. fecundity, growth, and survival). This type of integration is long overdue. And it is critical to forecasting the effects of environmental change and human activity on the future of sea turtles and other protected species.

All five species of sea turtles that nest in the US are protected under the Endangered Species Act (1973). Their population demographics reflect the effects of multiple stressors. These include a history of terrestrial habitat loss, terrestrial and aquatic habitat degradation, as well as direct hunting and non-target bycatch in several

¹ Duke University Marine Laboratory, Nicholas School of the Environment and Earth Sciences, Beaufort, NC 28516-9721, lcrowder@duke.edu

fisheries spreading over ocean basins (NRC 1990, Figure 1). Specifically, beach development, photo-pollution, beach renourishment, and beach armoring have reduced nesting habitat, compromised incubation environments, and reduced hatchling production.

These threats continue, but new threats include increased egg incubation temperatures due to global warming (further skewing sex ratios which are defined by incubation temperature), and loss of nesting habitat associated to rising sea levels on developed and armored beaches. In marine systems, accumulation of pollutants such as plastics, heavy metals, environmental estrogens, and oil products in pelagic nursery and demersal coastal habitats as well as rapid general degradation of marine and estuarine environments by nutrient runoff have the potential to further reduce turtle populations. In addition, sea turtles swim into a gauntlet of fishing gear including trawls, gill nets, pound nets and longlines as they migrate across ocean basins. The National Academy of Sciences panel (NRC 1990) concluded that interactions with fisheries were the most serious threat to sea turtle populations.

Case Studies

Sea turtle populations are monitored by counting nests (and less often nesting females) on nesting beaches. For many sea turtle populations, numbers have declined dramatically from historical peak values (Bjorndal and Jackson 2002), but others are stable or showing some signs of recovery (Figure 2). This data for population trends can be misleading, however, because nesting females often constitute a very small portion of the overall population. Loggerheads first nest in their early 30s, so adult females not only account for much less than one percent of the population, but they reflect nesting success and oceanographic conditions decades ago.

The following is a brief summary of the work in analysis and modeling for loggerhead and Kemp's ridley sea turtles.

Loggerhead Sea Turtles – Early deterministic matrix models of North Atlantic loggerhead sea turtles (Crouse *et al.* 1987, Crowder *et al.* 1994) outlined an approach

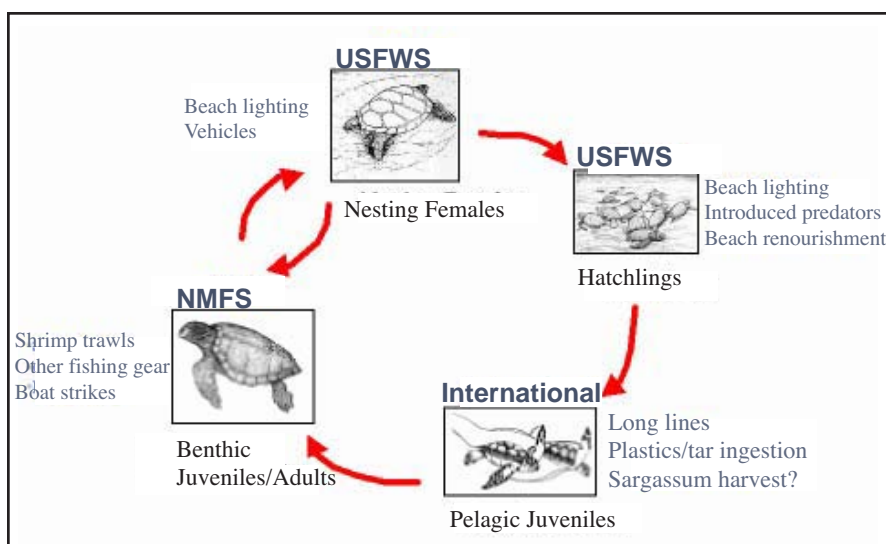


Figure 1. Life history of sea turtles, threats and management responsibilities in the US.

to compare multiple management scenarios and determined which of several proposed scenarios was most (or least) likely to enhance the probability of population recovery.

Using a technique called elasticity analysis, we can examine the effects on population growth rates of small proportional changes in survival, growth, and fecundity for a particular sea turtle population. Elasticity analysis of early deterministic models pointed to critical and sensitive aspects of the life history and emphasized where uncertainty in parameter values limited their predictive ability (Crouse *et al.* 1987, Crowder *et al.* 1994, Heppell *et al.* 1996, Heppell *et al.* 2000a, Heppell *et al.* 2002b). The analysis provided direction for researchers who wanted to reduce this uncertainty and so enhance predictions. It also pointed to sensitive stages (e.g., benthic juveniles) and processes (e.g., survival) which gave direction to managers.

In Australia, more detailed data allowed development of more elaborate stochastic difference equation models for green and loggerhead sea turtles (Chaloupka and Limpus 1996, Chaloupka 2001). Of course, a variety of model structures, from simple deterministic models to stochastic, individual-based, spatially-explicit models (Letcher *et al.* 1998, Walters *et al.* 2002) could be used. This approach uses the most appropriate modeling form as determined by the questions and the available data.

Although data on population vital rates of North Atlantic loggerhead turtles were too limited in the early 1980s to develop more complex models (e.g., stochastic or time-varying), these early analyses proved extremely useful. Elasticity analysis of the loggerhead

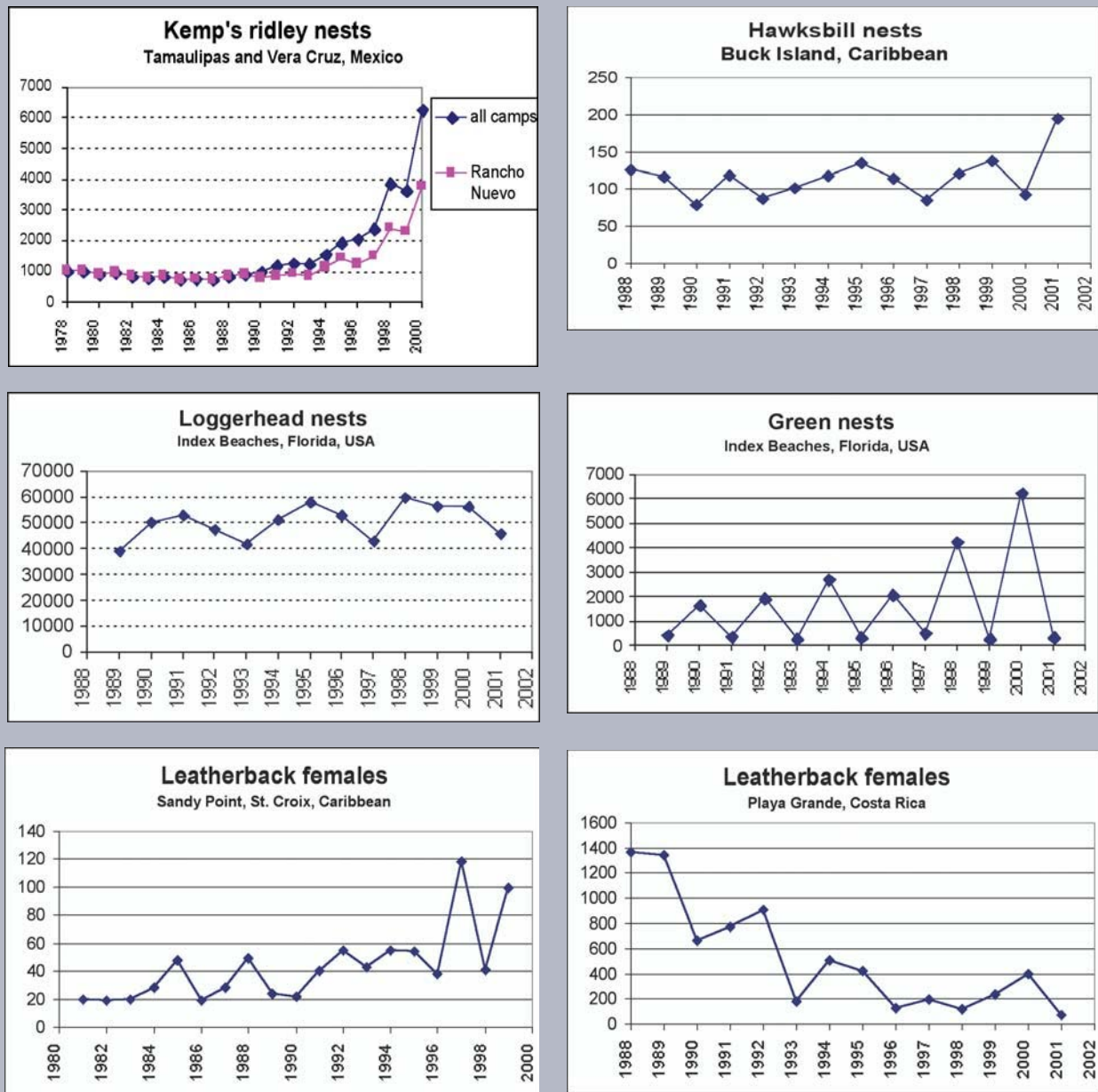


Figure 2. Examples of long time series of sea turtle abundance. Data generously provided by B. Witherington, Florida Fish and Wildlife Conservation Commission, Florida Marine Research Institute, Index Nesting Beach Survey Program (greens and loggerheads), the Gladys Porter Zoo and SEMARNAT/INE, Mexico (Kemp's ridleys), Z. Hillis-Starr and B. Phillips, Buck Island Reef NM Sea Turtle Research Program (hawksbills), D. Dutton and P. Dutton (leatherbacks, St. Croix) and R. Reina and J. Spotila (leatherbacks, Costa Rica).

model showed the greatest impact on the population could be made by increasing survival of large juveniles and sub-adults – precisely the sizes taken most often in trawl fisheries.

This evidence was used to support the implementation of Turtle Excluder Devices (TEDs) in trawl fisheries in the US just three years after the initial paper was published (Crouse *et al.* 1987). Subsequent analyses showed that TEDs were reducing the number of dead

loggerheads washed up on Atlantic beaches (Crowder *et al.* 1995) to levels that might allow population recovery (Crowder *et al.* 1994). These forecasts were followed by population increases in the South Florida sub-population (Heppell *et al.* 2002a), but these increases appear to have recently leveled off (Blair Witherington, pers. comm.). The northern population continues to decline at about 2 percent per year despite all management efforts to date.

The TED technology is now spreading globally (Epperly 2002). Recent analyses (NOAA 2001) support increases in the openings of TEDs to protect adult loggerheads and leatherbacks. It also suggests minimizing takes in longline fisheries are potentially important to recovery.

Kemp's Ridley – Kemp's ridley sea turtles nest primarily on one beach, Rancho Nuevo, Mexico. They were once considered the most endangered sea turtle, having only about 300 nesting females per year the mid 1980s.

Modeling Kemp's ridley population dynamics (Heppell *et al.* 1996, 2002c) drew attention to the need for protecting the in-water stages and lent support to efforts to require TEDs. It also showed a program of captive rearing of hatchlings, referred to as 'headstarting,' was unlikely to increase population numbers. Still, the headstarting program contributed to an important goal of establishing a second nesting population at Padre Island National Seashore.

Based on both protection of nests on the beach and TED regulations, Kemp's ridleys have been increasing steadily since the late 1980s at a rate of 11-13 percent per year (Heppell *et al.* 2002c.).

New analyses on the spatial distribution of turtles and shrimpers in the Gulf of Mexico pointed to important problems. One of these is the effect of the hypoxic 'dead zone' on demersal habitat for fishes, turtles, and marine mammals (Craig *et al.* 2001). Also, several areas within the Gulf that seemed to retain relatively high densities of turtles and so might be appropriate for additional protection as marine reserves (McDaniel *et al.* 2000). Two of these areas have since been included in protected areas as part of the expansion of the Dry Tortugas Marine Sanctuary off the southwestern tip of Florida and as a seasonal, nearshore shrimp closure (which also protects turtles) near Padre Island National seashore off the coast of Texas.

In a recent analysis of strandings data from Texas (the number of dead turtles washed up on the beach, Lewison *et al.* 2003), both sea turtle population growth and shrimping activity have contributed to the observed increase in strandings. TED compliance was a significant factor in accounting for annual stranding variability – low compliance was correlated with high strandings. Forecasts suggest that improved compliance with TED regulations will reduce strandings to levels that with other protective measures, should promote population recovery for both loggerhead and Kemp's ridley turtles. Local, seasonal fisheries closures, concur-

rent with TED enforcement, could reduce strandings to even lower levels.

The model forecasts the Padre Island shrimp closed area is also likely to reduce mortality of adult Kemp's ridley, potentially as much as 39 percent. In 2001 and 2002, this reduction was 46 percent and 38 percent, respectively. This seasonal closure, adjacent to a recently established Kemp's ridley nesting beach at Padre Island National Seashore, may promote long-term population persistence by establishing a robust, secondary nesting site.

Evolving Model Approaches – Subsequent model analyses have generalized the approach to understanding population dynamics of long-lived species (Heppell *et al.* 2000b) and to species with very limited demographic data (Heppell 1998, Heppell *et al.* 2000c). Chaloupka (2001) outlines many advantages of developing fully stochastic models, if the data are available. One basis for his analyses is that variation is probably greater in processes that influence fecundity and survival in the egg and early juvenile stages than for survival later. The qualitative effect on previous conclusions (based on the deterministic model) is reproduction and early survival are likely to become more important. This is similar to results from comparing deterministic and stochastic models for fish populations (Quinlan and Crowder 1999).

Recently, there is new information on the subpopulation structures of North Atlantic loggerheads (NOAA 2001). There are no fewer than five genetically distinct rookery populations. Most are increasing, but the northern nesting population, which produces most of the males for the regional population is not.

Historical evidence suggests that lost rookeries are seldom recolonized; losing the northern population presents two problems – loss of a distinct segment of the northwest Atlantic loggerhead population and loss of the major source of males for the remaining regional population.

And finally, managers are increasingly considering area-based management alternatives like marine protected areas. For these reasons, a spatially-explicit approach to loggerhead population dynamics is being formulated (Wyneken and Crowder 2001).

Lessons Learned

The key lesson from attempts to forecast population-level implications of management alternatives for long-lived species like sea turtles is significant progress can

be made with even limited data. Conducting even simple analyses requires critical aggregation and examination of data. These tools quickly and efficiently focus research and allow comparison of management alternatives, at least qualitatively. As data quality improves, new analytical tools can be applied.

For sea turtles and other long-lived species, management alternatives need to be considered even before all the data are in. In addition to forecasting, TEDs can work to reduce sea turtle mortality and alter population trends. The analysis also highlighted the importance of mortality at sea, requiring entirely new research efforts on the in-water phase of sea turtle life histories.

Conclusions

Population models provide a useful framework to integrate what is known and to clearly identify what is not. Analyses of these models also allows comparison of management alternatives based on the mode of action and impact to determine which are likely to contribute most to population recovery. Although the early models of sea turtle population dynamics were simplistic, the qualitative forecasts are robust for turtles (Heppell 1998) and for most long-lived species (Heppell *et al.* 2000b).

Model analyses not only identify poorly-known parameters, but point to factors expected to have the greatest impact on population growth. In this way, research priorities can be more efficiently focused.

Finally, these models are constrained by the available data and now can only make reliable qualitative predictions. Precise quantitative prediction requires additional data collection. Still, for ecological forecasting, qualitative predictions have provided useful guidance to managers responsible for the recovery of these endangered species.

Acknowledgements

I thank Debby Crouse for introducing me to the world of sea turtles. Selina Heppell and Melissa Snover have borne the brunt of model development for the last few years. I also thank Nancy Thompson, Sherry Epperly and other members of the Turtle Expert Working Group for all their assistance and encouragement. NMFS has supported much of this research, but recently EPA, Environmental Defense and Pew Charitable Trusts also have provided support. To all I am most grateful.

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Contaminant Forecasting at Hazardous Waste Sites in California Marine Waters

Deborah J. Johnston and Michael Martin¹

Abstract

Predicting or forecasting pollutant responses, after their removal, has been accomplished with DDTs for piscivorous birds and raptors at several marine sites in North America. Conversely, there are few examples of predictions of contaminant effects on other marine receptors or communities. In California, three major sites have been contaminated with organochlorine compounds (DDTs and PCBs): Moss Landing Harbor, Richmond Harbor, and the Palos Verdes Shelf.

The discharge of bioaccumulative contaminants has caused injuries and toxic effects to marine biota. Sediment, water column, and biota contaminant concentrations, which will protect marine receptors at these sites, were developed, based upon forecasting estimates. This data and information was evaluated to determine which cleanup and disposal (= remediation) options best protect State natural resources. At one site (Richmond Harbor), field measurements (chemistry and biology) were used to evaluate the effectiveness of the predicted environmental conditions after cleanup. The other two sites are in earlier stages of this process, however natural recovery predictions (i.e., without remediation) are presented.

Introduction

As a natural resource trustee, under United States and California laws and regulations, the remediation and restoration of hazardous waste contaminated sediments in estuarine and ocean waters is a responsibility of the Federal and State fish and wildlife agencies. This is under authorities of the Comprehensive Environmental Response, Compensation and Liability Act (CERCLA) of 1980, as amended, 42 U.S.C. 9601 *et seq.*, and the Clean Water Act (CWA), 33 U.S.C. 1251-1376, and the California Fish and Game Code (§ 2014). CERCLA, Section 107. They require the governor of each state to designate state officials who may act on behalf of the public as trustees for natural resources, which are defined to include “land, fish, wildlife, biota, air, water, groundwater, [and] drinking water supplies.” This Section 107 trustee designation is important because it validates the status of an agency as a relevant state authority in the CERCLA process (i.e., hazardous waste cleanup, which may affect natural resources) and clarifies its jurisdiction. The California Department of

Fish and Game has been delegated as one of the CERCLA Section 107 natural resource trustees.

Remedial actions (i.e., contaminant cleanup, removal, or response), at hazardous waste sites, are managed by Federal and State water quality regulatory agencies (i.e., US EPA and CalEPA). There are several regulatory actions that may be used by the agencies. However, the principal approach is to perform a Remedial Investigation/Feasibility Study (RI/FS) to determine the relative threat(s) or risk(s) of the contaminants to human health and the environment, and to develop cleanup alternatives. The selected alternative for remediation is included in the Record of Decision (ROD) for the site, which provides the analyses and justification for the selection of the remedy.

If remedial actions are judged insufficient to protect and restore natural resources injured by releases from a Superfund site, or if the use of the natural resource is lost or curtailed because of the releases, the natural resource trustees may seek to collect damages from responsible parties. Natural resource damages are monetary payments

¹ California Department of Fish and Game, 20 Lower Ragsdale Drive, Suite 100, Monterey, CA 93940

“for injury to, destruction of, or loss of natural resources, including the reasonable cost of assessing such injury, destruction, or loss resulting from such a release.”

These payments are considered compensation, and are intended to cover the past injury and residual costs, or losses, beyond whatever restoration can be achieved through remediation. Any payments to natural resource trustee agencies from responsible parties under these statutes are limited to: 1) reimbursement to the agency for its past costs of determining injuries and damages to natural resources and 2) expenditures for planning, monitoring, and implementation of restoration activities.

Case Study

The issues to be examined in this case study of Californian marine and estuarine sediment cleanups are threefold: 1) prediction of toxic chemical concentrations with remediation, 2) prediction of ecosystem response (i.e., selected biomarker response) following remediation, and 3) estimates of uncertainty in forecasting ecological responses for remedial actions.

This forecasting is at three sites, which are contaminated with organochlorine pesticides. The focus is to evaluate the feasibility of using ecological forecasting in restoration planning, to establish a framework for ecological prediction and assessment (confirmation), and to identify problems and recommendations for future forecasting needs. The forecasts are intended to serve as information and planning evaluations to portray estimates of ecological conditions resulting from the remedial actions (and designs), based upon experiences at the subject sites (Figure 1).

At these three sites (Figure 1), hazardous waste remediation has occurred or is in stages of planning. The sites are described in relation to the chemical and ecological response forecasts, a description of the forecasting methods, and the likely outcomes, based upon planned remedial action(s). Forecasts are generally qualitative predictors of chemical and ecological responses, however, portions of the predictions are based upon application of a quantitative equilibrium model of contaminant biomagnification in coastal food webs.

One of the forecast efforts (Montrose NRDA, Palos Verdes Peninsula) was used to evaluate the fate of contaminants in coastal sediments, as well as predicting the bioaccumulation potential for various food web compartments (Hydroqual, 1994). The second site (United Heckathorn, Richmond Harbor) involved a completed removal and remediation project (which anticipated risk reduction to no significant future effects), and subsequent field verification. The third site (Moss Landing Harbor) included a description of a planned project, using environmental monitoring data, with estimates of responses under several cleanup options.

Palos Verdes Peninsula DDT Contamination – The Palos Verdes Peninsula DDT contamination is the result of over four decades of industrial-produced DDT waste releases from a major municipal wastewater outfall. Over 1,800 metric tons were discharged. Approximately 100 metric tons of DDT remains sequestered in surficial sediments on the ocean bottom in the general vicinity of the outfall. An ongoing cleanup investigation, along with pilot scale remedial activities, is being conducted by the response agency to determine a recommended alternative for cleanup action.

The food chain mathematical model, based on organochlorine biomagnification, was used to estimate concentrations in several receptor species [fish (Dover sole, white croaker, and kelp bass), sea lions, peregrine falcons, eagles, and double-crested cormorants] that would result from exposure to sediments and water column contaminants at the site, located in approximately 60 meters of water, off of the Palos Verdes Peninsula, Los Angeles, California. The model is based upon two principal variables: 1) the bioaccumulation potential of the organochlorine contaminant [DDT (proportional to its octanol water partition coefficient or K_{ow})] and 2) the bioenergetic status of the receptor species (= trophic level).



Figure 1. Case Study Sites

The model also was used to project (estimate) future concentrations of organochlorine contaminants in sediments and water columns, as well as future contaminant levels in target species, under conditions of natural attrition. Several response actions were evaluated, using risk to human health and the environment as the receptors of concern. The initial engineering evaluation determined capping with a one meter cap, constructed of clean dredged materials was a potential feasible alternative, and is being evaluated further with pilot project investigations (EPA 2001). The modeling predictions for recovery responses in bird species are based upon ‘natural recovery,’ that is, with no remedial actions at the outfall site. Current pilot scale testing is being evaluated to determine the effects (benefits) of remedial actions for 1) risk reduction to human health and the environment, and 2) accelerated recovery of affected receptor species.



Figure 2. United Heckathorn Site.

Results of the modeling indicated levels of DDT in the sediments would increase from the current levels of 1 µg/g to 1.9 µg/g in 2030 due to erosional factors releasing DDT. The levels would then decrease to a final level of 0.15 µg/g in 2100. Based on the model results, it is expected that the peregrine falcon population counts will decrease, corresponding to the increase release of DDT predicted to occur by 2030. Then as DDT levels decrease, falcon populations should recover (in 2100) to what was predicted for the Channel Islands based on no exposure to DDT.

United Heckathorn DDT and Dieldrin Contamination – Lauritzen Channel is an industrial waterway adjacent to the former United Heckathorn facility in the inner Richmond Harbor area of San Francisco Bay, California (Anderson *et al.* 2000, Figure 2).

Contaminated marine sediments at this Superfund site were dredged and disposed in a land facility from late 1996 through early 1997 to remove DDT, and dieldrin, both of which pose high risks to wildlife. A study assessed the Lauritzen Channel marine environment immediately before and approximately one year after the dredging. The study included chemical analysis of sediments, tissue concentrations of transplanted mussels, toxicity testing of sediment samples, and characterization of benthic community structure. The receptors evaluated at this site were associated with benthic invertebrate’s chronic toxicity and benthic invertebrate community responses.

The pre-remediation forecast was that removal and upland disposal of DDT-contaminated sediments would completely eliminate bioaccumulation and deleterious effects to aquatic species from organochlorine compound exposure (Table 1). Results indicated that sediment toxicity to bivalve larvae (*Mytilus galloprovincialis*) decreased in post-remediation samples, but that toxicity to the amphipod (*Eohaustorius estuarius*) increased significantly, when exposed to post-remediation samples.

After remediation, assessment of benthos at this site suggested a transitional benthic community structure. In addition, post-remediation sediments remained contaminated by a variety of organic chemical compounds, including DDT, dieldrin, chlordane, polycyclic aromatic hydrocarbons, and polychlorinated biphenyls. Tissue concentrations of DDT and dieldrin in mussels (*M. galloprovincialis*) were lower than those in preremediation samples, indicating that although sediment concentrations of organochlorine pesticides remained high, concentrations of these chemicals in the water column were reduced after dredging. This suggests that their

| | Station 1 | | Station 2 | | Station 3 | | Station 4 | |
|--|-----------|-----------|-----------|-----------|-----------|-----------|-----------|-------|
| Chemical | Before | After | Before | After | Before | After | Before | After |
| SDDT1 | 77,700.00 | 21,361.80 | 47,800.00 | 27,833.00 | 26,000.00 | 15,555.00 | 2,740.00 | 840.2 |
| SDDT2 | 3,500.00 | 2,637.26 | 2,710.00 | 1,366.81 | 1,520.00 | 691.33 | 189 | 53.18 |
| Dieldrin3 | 748 | 371 | 528 | 619 | 442 | 196 | 35.7 | 25.8 |
| Dieldrin4 | 35.2 | 45.8 | 28.7 | 30.34 | 25.8 | 8.71 | 2.46 | 1.63 |
| % Fines | 92.3 | 23.5 | 85.2 | 91.59 | 85.9 | 94.45 | 89.5 | 92.58 |
| % Organic Carbon | 2.38 | 0.81 | 1.78 | 2.09 | 1.73 | 2.25 | 1.46 | 1.58 |
| 1. ug/kg dry weight 2. ug/g OC 3. ug/kg dry weight 4. ug/g OC | | | | | | | | |

Table 1. Sediment SDDT and Dieldrin Concentrations Before and After Site Remediation of the Lauritzen Channel.

bioavailability may have diminished after the remedial actions (Table 1).

Moss Landing Harbor Sediments DDT Contamination – Moss Landing Harbor lies at the head of the Monterey Bay Canyon. It receives non-point runoff from land based activities primarily agricultural nutrients and pesticides. Regular dredging of the harbor sediments has not occurred due to the high DDT, TBT, PCBs, and copper levels that exceed EPA standards for unconfined aquatic disposal of dredged sediment. The study proposes to collect data and design a model to project contaminated sediment movement (both geologically and biologically) in the Monterey Bay Canyon. This will enable resource managers to determine appropriate disposal locations for DDT-contaminated dredged sediments. This project is in the initial stages of data collection.

The model proposes to use Cs/K ratios with field and laboratory validation. It would provide a more site-specific analysis of DDT movement, both within the food chain (bioaccumulation) and within the canyon (locations dependent on how available the DDT is to the biota for uptake). Several of the important model components are unknown including: sediment transport within the Harbor and Canyon, benthic structure in the Canyon (one potential disposal site), and pollutant source contribution loads. However, based on model predictions controlling 10 percent of the agricultural runoff, DDT levels in sediment should decrease from current levels of approximately 3,000 ppb to less than 1,000 ppb. Based on uncontrolled agricultural runoff, modeling results predict DDT levels will decrease due to the initial dredging removal of the source load, but then return to current levels.

Lessons Learned

Ecological risk assessment and management guidance includes scientific/policy/management interactions with the development and implementation of a hazardous waste cleanup. This consisted generally of consultations and scientific reviews of risk assessment and engineering feasibility documents, as the projects progressed. Technical/scientific personnel interact with policy and decision-makers, to formulate plans and alternatives that minimize risk(s) to human health and the environment. Natural resource trust agencies are now being consulted

and involved with the development of cleanup plans, with efforts being coordinated and incorporated into the assessment process. The research and development, along with verification and monitoring, are key elements in the on-going assessment process. Most large California cleanup investigations to date include a monitoring and surveillance component.

Conclusion/Next Steps

The major impediments to improving the forecasts are the limited data on trophic transfers of toxic chemicals, as well as the abilities of current biomarkers to accurately predict or identify chronic toxic effects. Additional data on site-specific characteristics of contaminant exposure to sensitive receptors and understanding and interpreting the biomarker or community response to chemical insult(s) will increase the accuracy to predict the pathways and improve model prediction.

Few databases are widely available as comparative or interpretive tools. There are really only two major predictive models that have been developed (i.e., Equilibrium Partitioning and Food Web Models) to identify the pathways and exposure characteristics of bioaccumulative toxic pollutants. Other approaches might provide additional assistance in forecasts of chemical and ecological conditions in the marine and estuarine environment.

Acknowledgements

Partial travel expenses were funded by a grant from the California Association of Professional Scientists.

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An Integrated Modeling System for Forecasting the Response of Estuarine and Coastal Ecosystems to Anthropogenic and Climatic Changes

Y. Peter Sheng¹

Abstract

This paper describes the application of an integrated modeling system (CH3D-IMS) which consists of a hydrodynamic model, a wave model, a sediment transport model, a water quality model, a light attenuation model, and a seagrass model. The integrated system has been used to forecast the response of two estuaries – the Indian River Lagoon and Tampa Bay in Florida – to anthropogenic and climate changes.

Model simulations of the Indian River Lagoon in 1998 and Tampa Bay in the summer of 1991 are considered the baseline conditions. Simulations with reduced nutrient loads and climate change are then conducted to forecast the response of these estuaries. The forecast results are presented in terms of predicted changes in salinity, water quality, light, and seagrass biomass.

Introduction

During the past few decades, agricultural activities and increased population has led to increased loading of pollutants (total suspended solids, phosphorus, nitrogen, dissolved organic material, etc.) into numerous estuarine and coastal ecosystems. Excessive pollutant loads have resulted in deteriorated water quality and high concentrations of chlorophyll *a*, total suspended solids, and color (dissolved organic matter). Consequently, light (Photosynthetically Active Radiance or PAR), essential for seagrass growth, has been significantly attenuated, leading to loss of seagrass beds and fishery habitat. For example, there has been a significant loss of seagrass beds in many Florida estuaries, including Indian River Lagoon (Figure 1), Tampa Bay (Figure 2), and Florida Bay.

Resource management agencies for these Florida estuaries have been developing strategies to reverse the trend of seagrass loss by establishing pollutant load reduction goals (PLRGs), i.e., external loading rates which will lead to the desired water quality, light, and seagrass biomass targets. Sheng (1998) reviewed the various types of pollutant load reduction models – a regression model, a box model, and a process-based model.

For example, the Tampa Bay Estuary Program established nitrogen load reduction goals for Tampa Bay by using simple regression models to relate nitrogen loads to chlorophyll *a*, light loss, and seagrass biomass in Tampa Bay. Sheng *et al.* (1996a) developed a process-based integrated modeling system to study the effect of reduced nitrogen loads in Roberts Bay and upper Little Sarasota Bay in Florida. (These bays are connected to the south part of Sarasota Bay.) Sheng *et al.* (1996b) and Yassuda and Sheng (1998) developed a process-based integrated modeling system for Tampa Bay. Sheng (2000) and Sheng *et al.* (2002) developed a process-based integrated modeling system for determining the pollutant load reduction goals for the Indian River Lagoon. After model calibration and validation, these process-based integrated modeling systems can be used to determine the response of the estuaries to different loading conditions and to determine the loading conditions corresponding to desired targets of water quality, PAR, and seagrass biomass. Resource management agencies can use these models to develop pollutant the load reduction goal (PLRG), total maximum daily load (TMDL), and minimum flow and levels (MFL) for various water bodies.

¹ Civil & Coastal Engineering Department, University of Florida

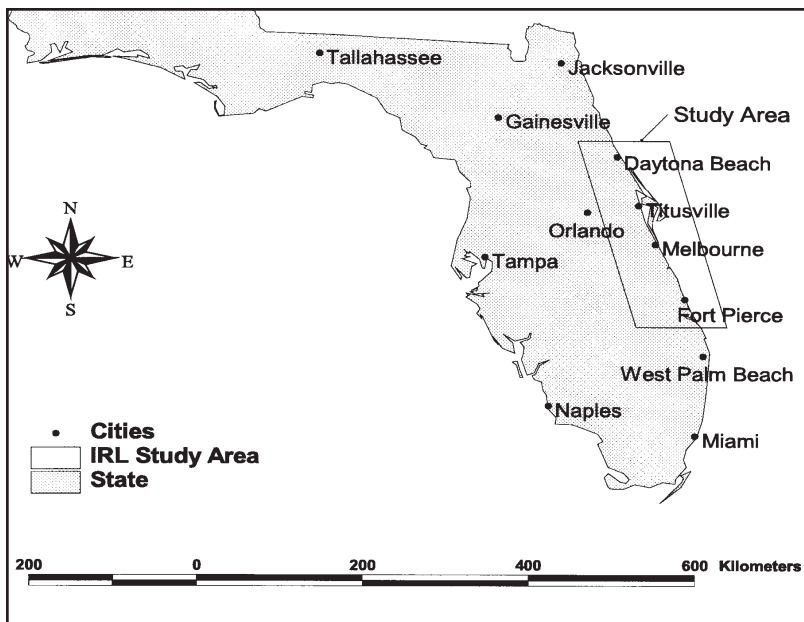


Figure 1A. The Indian River Lagoon study area.

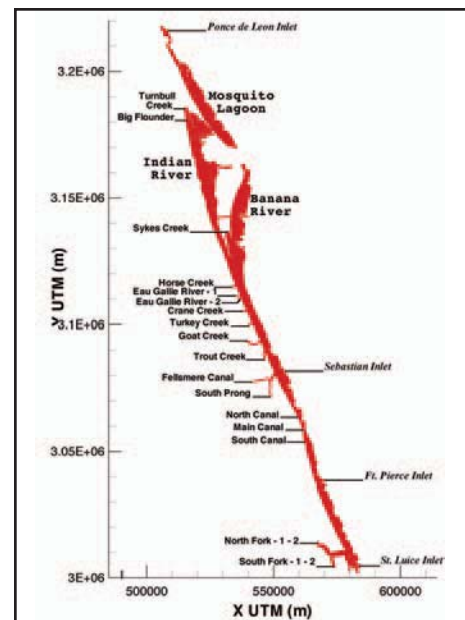


Figure 1B. Indian River Lagoon and four tidal inlets.

Case Studies

There are two case studies. Both studies use the CH3D-based integrated modeling system, CH3D-IMS (Sheng 2000, Sheng *et al.* 2002, University of Florida). CH3D is a 3-D curvilinear-grid hydrodynamic model originally developed by Sheng (1986, 1987) and has been significantly enhanced since then (e.g., Sheng 1989, Sheng *et al.* 1996a, 1996b). The CH3D model uses orthogonal as well as non-orthogonal curvilinear grids, hence it can be used to fit the irregular estuarine boundaries rather accurately. As shown in Figure 3, the CH3D-IMS consists of the 3-D hydrodynamic model CH3D, a wind wave model (SMB), a 3-D sediment transport model (SED3D), a 3-D water quality model (WQ3D), a light attenuation model (LA3D), and a seagrass model (SAV).

These component models are dynamically coupled. For example, light penetration in the water column is affected by water, chlorophyll *a*, total suspended solids, and dissolved organic matter. Light penetration in turn affects the growth of chlorophyll *a*. Seagrass growth is limited by the water temperature and salinity, as well as by light and nutrients.

The first case study uses the CH3D-IMS to forecast the response of the Indian River Lagoon to reduced nutrient input and total suspended solid loads from tributaries. The second case study applies the CH3D-IMS to forecast the response of Tampa Bay to reduced nitrogen and freshwater loading from a major tributary – the Hillsborough River.

Indian River Lagoon, Florida – The Indian River Lagoon (IRL) extends about 255 kilometers from Ponce Inlet to the North to St. Lucie Inlet to the South, with a width of 2-10 km and an average depth of 2 m. This lagoon is one of the most biologically diverse ecosystems in the world, with fishery resources valued at almost one billion dollars per year. In recent decades, however, both seagrass and fishery resources have declined.

To develop scientifically sound strategy to restore the ecosystem, efforts began at the University of Florida in 1996 to develop an Indian River Lagoon Pollutant Load Reduction model (IRLPLR). It is an integrated modeling system which can predict the response of the ecosystem to changes in pollutant loads. This model is based on the CH3D-IMS and validated with extensive physical, chemical, and biological data collected from the IRL. Critical model coefficients have been determined, based on laboratory and/or field experiments.

Validation of the CH3D-IMS with IRL data was presented in Sheng *et al.* (2002). Two types of model simulation have been conducted and validated. The first simulates hydrodynamics (water levels, currents and waves), sediment dynamics, and water quality dynamics during short-term events (storms). The second is the one-year continuous simulation during 1998. Model results have been compared with measured time and space varying data of water level, currents, salinity, suspended sediment concentration, nutrient concentrations (nitrogen and phosphorus), chlorophyll *a* concentration, dissolved oxygen concentration, and PAR.

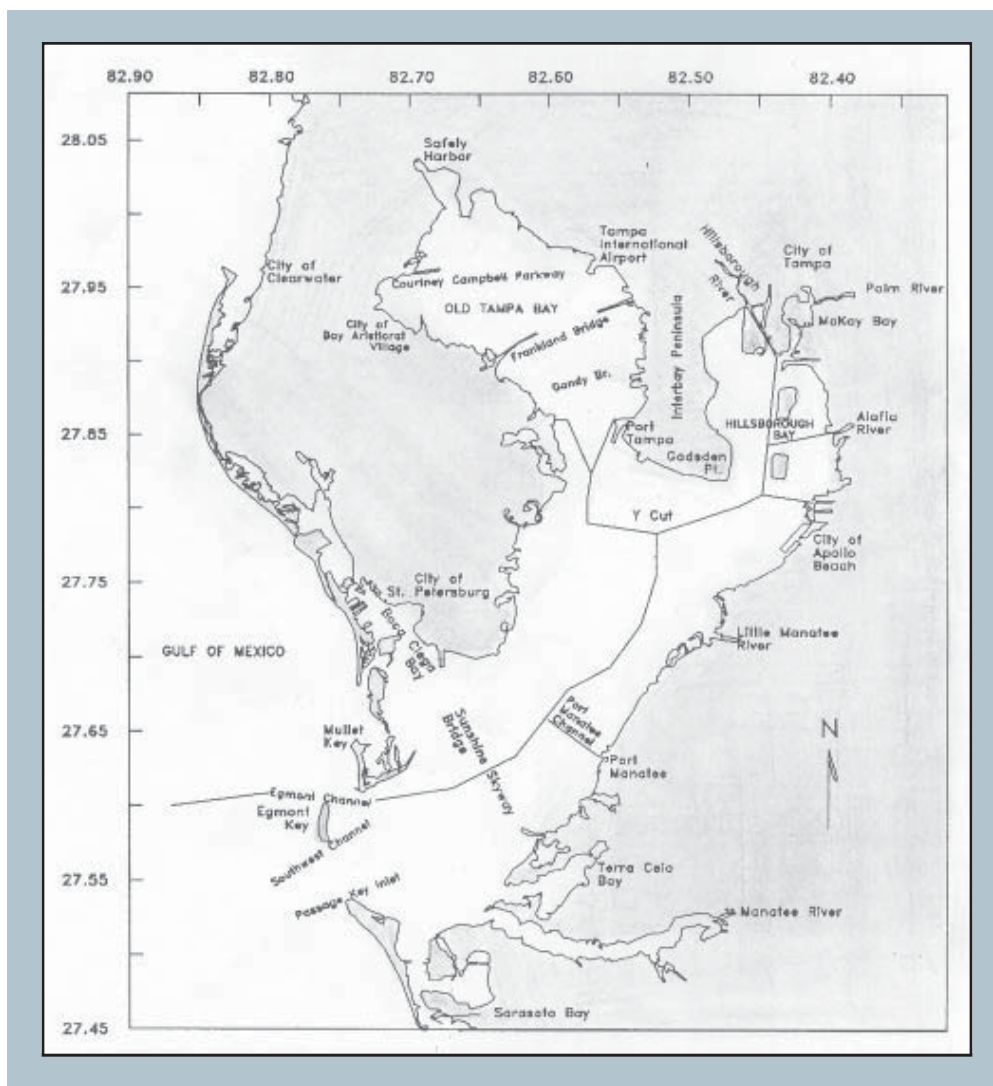


Figure 2. Tampa Bay, Florida.

Model and data showed more phosphorus loading in the South and more nitrogen loading in the North. Light attenuation is mainly influenced by total suspended solids, while chlorophyll *a* and color have secondary importance, particularly at high attenuation (low penetration) situation with the diffuse attenuation coefficient exceeding one.

The validated IRLPLR model has been applied to forecast the response of the IRL to reduced loads. Specifically, a quantitative forecast the response of the IRL to 1) 100 percent reduction of nutrient loads, 2) 100 percent reduction of nutrient and total suspended solids loads, 3) increased wind speed, and 4) less evaporation. For each of the forecasts, the validated IRLPLR model simulates the 1998 IRL condition, with somewhat different boundary conditions.

oped a seagrass restoration plan based on the regression models. The restoration plan, which only considers the light attenuation due to chlorophyll *a*, appears to have worked in much of Tampa Bay. Seagrasses have grown in response to nutrient load reduction. However, seagrass in Old Tampa Bay has failed to respond to lower nutrient loads and lower chlorophyll *a* concentrations. Hence, it is useful to use an integrated modeling system to examine the response of Tampa Bay to nutrient load reduction.

An earlier version of CH3D-IMS was validated with field data obtained in the summer of 1991 (Sheng *et al.* 1996b, Yassuda and Sheng 1998). The validated model forecasts the response of Tampa Bay to 100 percent reduction of freshwater and nitrogen loads, both dissolved and particulate (Sheng *et al.* 2001) from the tributaries.

For example, the first forecast is identical to the 1998 validation run except the nutrient loads (both dissolved and particulate) from the tributaries are set to zero. For the second forecast, all nutrient and total suspended solids loads from tributaries are set to zero. To examine the effects of increased wind and reduced evaporation, the third and fourth forecasts are conducted, respectively. These forecasts are preliminary applications of the IRLPLR model to determine the allowable pollutant load rate and the pollutant load reduction goal (PLRG). Results of the first and second forecasts are discussed in this paper.

Tampa Bay, Florida – Tampa Bay resource managers have devel-

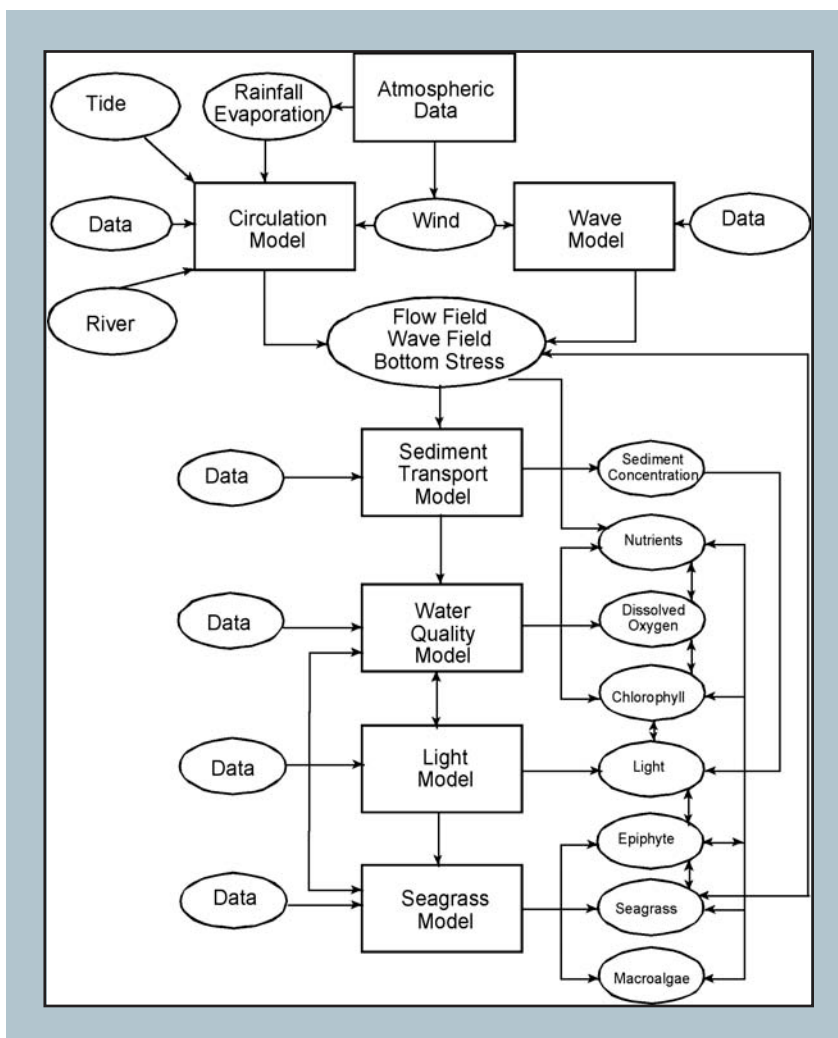


Figure 3. Block Diagram of an integrated modeling system: CH3D-IMS (From Sheng 2000).

These preliminary forecasts for Indian River Lagoon and Tampa Bay have not been validated with field data, since 100 percent reduction in loads of nutrients, freshwater, and total suspended solids cannot be easily achieved without complicated regulations. However, since both the IRL model and the Tampa Bay model have been validated with field data, the model forecasts are assumed to be fairly reliable.

Forecast uncertainty has not been addressed, but will be by performing ensemble simulations/forecasts. The IRL forecasts are expected to be used by the St. Johns River Water Management District, while the Tampa Bay forecasts may be used by the Tampa Bay Estuary Program and the Southwest Florida Water Management District.

Lessons Learned

The IRLPLR model has been validated with 1998 field data. It is in the process of being used to determine

pollutant load reduction goals (PLRGs) for various segments of the IRL (Figure 4). As an example, the 1998 IRL simulation is repeated with the nutrient and total suspended solid loads completely removed from the tributaries, and the results are compared with the 'with loading' results. Figure 5 shows the effect of complete nutrient and total suspended solid load reduction on the increased amount of PAR (Photosynthetically Available Radiance) at the bottom of Segments 5, 6, 7, and 8.

The increase in PAR is not very dramatic since total suspended solids are the dominant light attenuator, accounting for 78 percent of light attenuation, vs. the 5 percent and 16 percent loss to color (dissolved organic carbon and other matters) and suspended chlorophyll *a*, respectively (Christian and Sheng 2003). Although total suspended solids are not supplied through the tributaries, significant sediment re-suspension in the shallow IRL causes significant total suspended solids concentration to maintain significant light attenuation except during the summer months. During the summer months, the forecasted

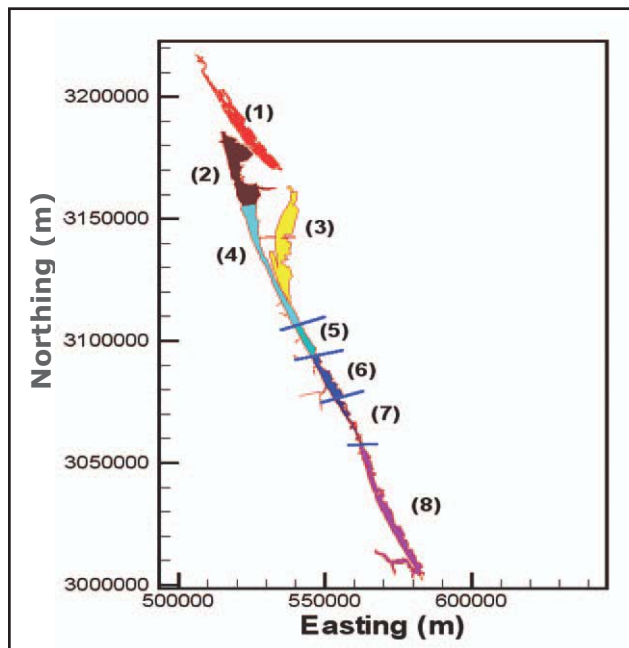


Figure 4. Segments of the Indian River Lagoon.

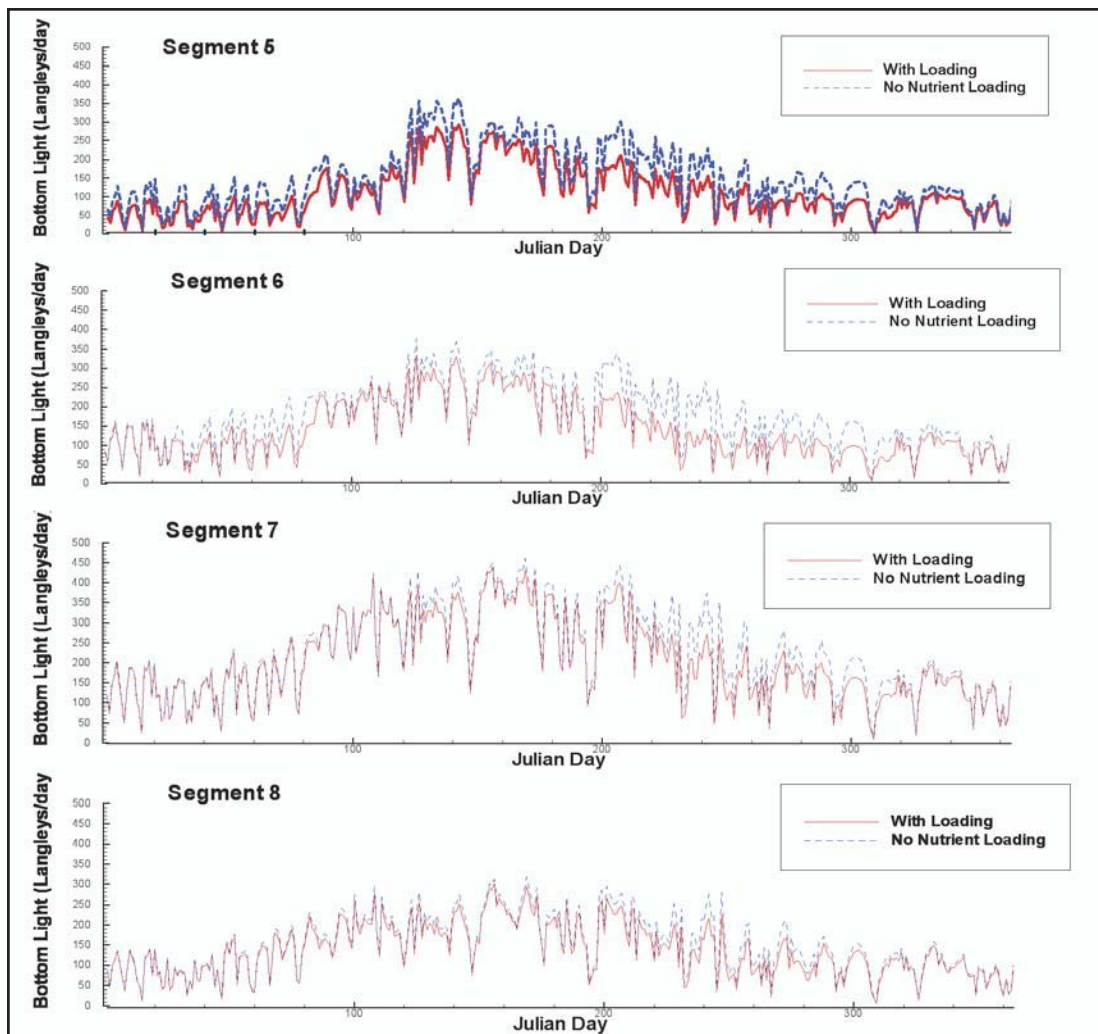


Figure 5. Bottom light in Segment 5,6,7, and 8 during 1998 with and without nutrient loading.

PAR show more noticeable increases, due to removal of significant color and total suspended solid loading and reduced sediment resuspension during these months.

The model forecasted the biomass of *Halodule*, the dominant seagrass species in some parts of the Indian River Lagoon, show noticeable increase at the end of June in 1998, due primarily to the increase in available light. Reduced nutrient loads did not limit the growth of seagrass, since sufficient nutrients already exist in the IRL. The SAV model, which was modified from

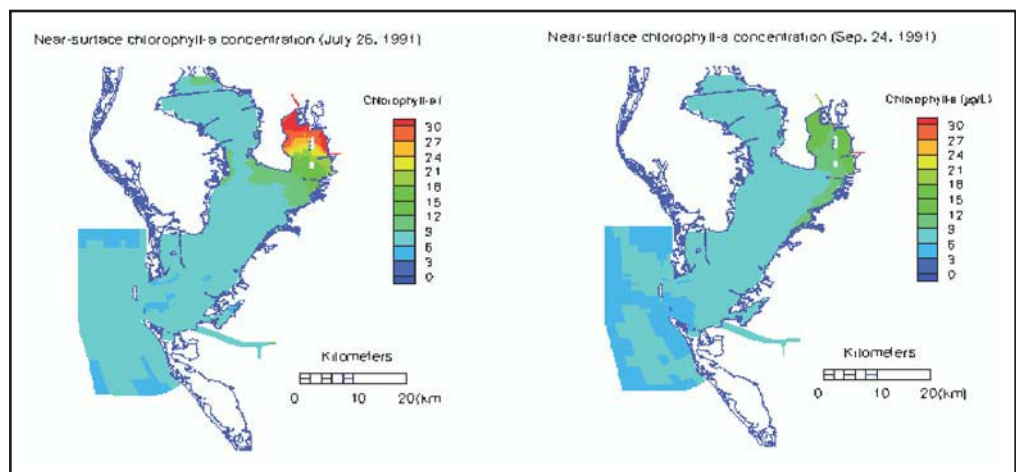


Figure 6. Near-surface Chlorophyll a in Tampa Bay during July and September 1991.

the model of Fong *et al.* (1997), is being further developed and validated with historical seagrass data.

The Tampa Bay simulations show the hypoxia and high Chlorophyll *a* concentrations in the Hillsborough Bay

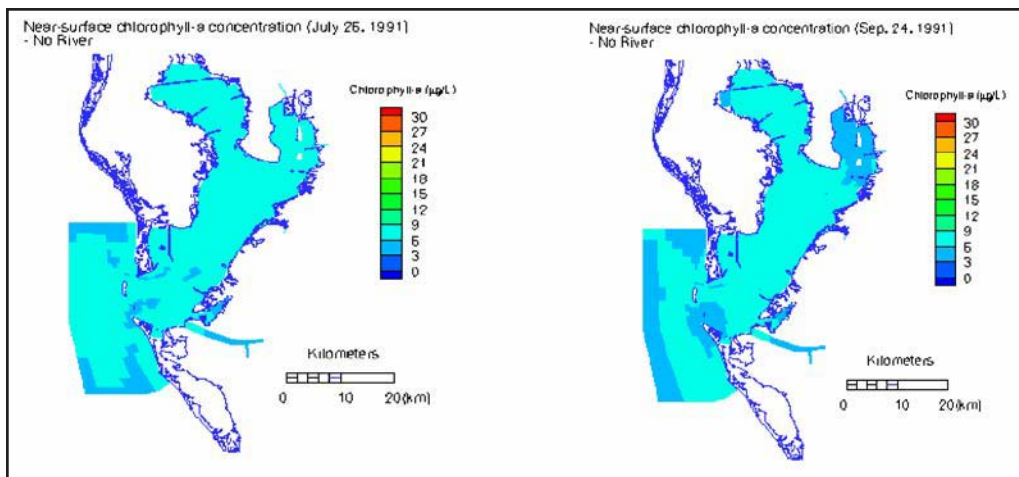


Figure 7. Near-surface chlorophyll *a* in Tampa Bay during July and September 1991 without river loading.

during the summer of 1991 (Figure 6) are significantly reduced if all the nutrient loads into Hillsborough Bay from the rivers are removed (Figure 7). The reduced Chlorophyll *a* concentration leads to increased light and some slight increase in *Thalassia*, which is the dominant seagrass species in Tampa Bay.

Both the Indian River Lagoon model and the Tampa Bay model are undergoing additional validation using long-term historical data and current data. The forecasts show the expected response of the estuarine systems under extreme conditions (i.e., complete reduction of nutrient loads). Since these are not realistic, there is no data to validate the forecasts. They are presented in this paper to represent preliminary applications of the CH3D-IMS to determine pollutant load reduction goals (PLRGs) and total maximum daily loads (TMDLs) for estuarine and coastal ecosystems. Additional forecasts should be obtained by considering a wide range of management scenarios (with 10-100 percent reduction of individual and combined parameters) and natural conditions (e.g., a typical year vs. an El Niño year, a relatively calm year vs. a year with many storms or hurricanes), and by focusing on each of the numerous watersheds.

To transfer the forecast from research and development to operational mode, it will be necessary to couple the integrated modeling system with additional component models (e.g., meteorological model, ocean model, etc.) and/or data network to allow dynamically-coupled real-time model forecasting. With forecasted natural conditions (wind, air temperature, river inflows, and ocean boundary conditions), it is possible to obtain real-time forecasts of the water quality and ecological conditions in these estuaries. For example, one can forecast the location, time, and duration of algal

blooms, hypoxia and seagrass dieoff. To verify the accuracy of the forecast and to incorporate modifications to improve the precision or accuracy, more model validation runs need to be done. In addition, inverse modeling techniques and data assimilation can be incorporated into the model simulations.

Significant uncertainties are associated with ecological forecasting using the integrated modeling system. Each component model in the basic integrated system contains inherent uncertainties associated with the model formulation, model coefficients, and model input (boundary and initial conditions). Monte Carlo simulations can be conducted for some of the component models, e.g., the light attenuation model and the seagrass model, which are basically 'box models'. For the three-dimensional hydrodynamic model, sediment transport model, and water quality model, ensemble simulations in combination with model sensitivity tests maybe used in place of the Monte Carlo simulations. Additional 'forecasting uncertainties' are associated with the forecasting of the forcing functions (i.e., forecasted model boundary conditions of wind, air temperature, river inflows, evaporation, and precipitation, etc.).

Next Steps

These two examples show ecological forecasting for two Florida estuarine ecosystems – Indian River Lagoon and Tampa Bay. However, much remains to be done to improve the forecasting. The well-being of ecological systems are threatened by both man-made and natural changes, hence improvement in ecological forecasting should lead to better management of ecological systems.

Improvement in ecological forecasting requires additional data, improvement in data gathering techniques, model advancements, as well as research into the fundamental ecological processes. The integrated modeling system presented here can be further expanded to include additional component models,

including atmospheric model, ground water model, and fishery model, etc.

Conclusions

Examples of ecological forecasting have been presented here using an integrated modeling system, CH3D-IMS. Specifically, the validated integrated modeling systems for these estuaries are used to forecast the response of these estuaries to reduced loads of nutrients, total suspended solids, and freshwater from tributaries. These forecasts of load reduction scenarios correspond to extreme conditions and hence have not been validated with data. However, since both the IRL model and the Tampa Bay model have been validated with field data, the forecasts are expected to be fairly reliable.

With forecasted natural conditions (wind, atmospheric temperature, river inflows, and ocean boundary conditions), it is possible to obtain real-time forecasts of the water quality and ecological conditions in these estuaries under more realistic load reduction conditions. For example, by coupling the CH3D-IMS with real-time weather forecast and assimilation of real-time ocean data, one can forecast the location, time, and duration of algal blooms, hypoxia, and seagrass die-off in these estuaries. Forecast uncertainty has not been quantified, but may be evaluated by a combination of model sensitivity study, Monte Carlo simulations or ensemble simulations.

Acknowledgement

The Indian River Lagoon model has been developed with support from the St. Johns River Water management District, while development of the Tampa Bay model has been supported by the Tampa Bay Estuary Program and the University of Florida.

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Forecasting Hypoxia in the Gulf of Mexico: Responses to Nutrient Loadings from the Mississippi River Basin

Victor J. Bierman, Jr.¹, Scott C. Hinz¹, William J. Wiseman, Jr.², Nancy N. Rabalais³, R. Eugene Turner²

Abstract

A zone of hypoxia (≤ 2 mg O_2/l) forms each spring and summer on the Louisiana-Texas continental shelf and stresses aquatic life. The principal causes are increased nitrogen loads from the Mississippi River Basin (MRB) combining with the physical stratification of Gulf waters.

A mass balance model was developed to address broad, macro-scale questions related to the impacts of nutrient loadings from the MRB on water quality in the northern Gulf of Mexico. Simulations were conducted for a range of nutrient load reductions to determine how much the bottom water dissolved oxygen concentrations improved. Reducing the nutrient load 20-30 percent led to forecasts of a 15-50 percent increase of bottom water dissolved oxygen concentrations. These results were used to develop a 30 percent nitrogen loading reduction target to meet the long-term Coastal Goal for reducing the areal extent of hypoxia in the northern Gulf of Mexico.

A major accomplishment in this study established watershed-scale links between nutrient loads from the MRB and water quality responses in the northern Gulf of Mexico. The study also called attention to some challenging questions facing policy-makers when implementing forecast results developed from research. In view of the complexities of the system, uncertainties in the level of knowledge, and the potentially long time period required to observe changes resulting from management action, an adaptive management approach was recommended.

This approach includes a comprehensive program of monitoring, modeling, and research to facilitate continual improvement in scientific knowledge and gradual adaptation of management approaches. Specific recommendations are made for reducing uncertainties and improving forecasting ability for water quality responses.

Modeling needs cannot be met independently of monitoring and research needs because models are only tools for synthesizing environmental data, not substitutes for these data. In the end, there must be compatibility among management questions, model capabilities and available data.

Introduction

The largest zone of oxygen-depleted coastal waters in the United States and the western Atlantic Ocean exists in the bottom waters of the Gulf of Mexico along the Louisiana-Texas continental shelf. As required by the Harmful Algal Bloom and Hypoxia Research and Control Act of 1998, the Mississippi River/Gulf of Mexico Watershed Nutrient Task Force (the Task Force) developed the Action Plan for Reducing, Mitigating and Controlling Hypoxia in the Northern Gulf of Mexico

(US EPA 2001). The Action Plan was formed by findings of the Committee on Environment and Natural Resources (CENR) Integrated Assessment of Hypoxia in the Northern Gulf of Mexico (CENR 2000) along with comments received and the six hypoxia assessment reports on which it was based.

The Action Plan describes a national strategy to reduce the frequency, duration, size and amount of oxygen depletion of the hypoxic zone of the northern Gulf of Mexico. In view of the complex nature of nutrient cy-

¹ Limno-Tech, Inc.

² Louisiana State University

³ Louisiana Universities Marine Consortium

cling and transport within the Basin and Gulf of Mexico, and the potentially long period of time required to observe changes resulting from management action, the Task Force recommended an adaptive management framework. Such an approach includes a comprehensive program of monitoring, modeling, and research to facilitate continual improvement in scientific knowledge and gradual adaptation of management approaches.

Case Study

The Gulf of Mexico is an ecosystem in distress because of hypoxia. Hypoxia occurs when dissolved oxygen concentrations are below the level necessary to sustain most animal life (≤ 2 mg O_2/l). A zone of hypoxia forms each spring and summer on the Louisiana-Texas shelf and stresses aquatic life (Figure 1). Since 1993, mid-summer bottom water hypoxia on the Louisiana-Texas Shelf has been larger than 10,000 km². The hypoxia zone has been expanding over time. The zone has more than doubled since 1985 (Rabalais *et al.* 1999) and extended to 20,000 km² in 1999, about the size of the state of New Jersey (Rabalais 1999).

The principal causes of hypoxia are increased nitrogen loads from the Mississippi River Basin (MRB) combining with the physical stratification of Gulf waters. Nutrient enrichment (eutrophication) resulting in dissolved oxygen depletion, has been widely observed elsewhere, including the Chesapeake Bay, Long Island Sound, and other coastal waters, as well as fresh water lakes and reservoirs. Nutrients stimulate algae growth which produces organic carbon. When this organic carbon sinks to the bottom and decays, dissolved oxygen is consumed. This happens mostly in spring and summer when the water column stratifies from fresh water runoff and the warming of surface waters, thus isolating bottom waters from oxygen replenishment.

Sediment cores from the hypoxic zone show algal production and deposition, as well as oxygen stress were much lower earlier in the 1900s, but significant increases occurred in the latter half of the twentieth cen-

tury. During this century, there also have been significant increases in nitrogen and phosphorus loads and decreases in silica loads in the Mississippi River. All of these have accelerated since 1950 (Turner and Rabalais 1991), and appear to have caused shifts in the offshore phytoplankton composition and an increase in primary production (Rabalais *et al.* 1996).

As part of the NOAA Nutrient Enhanced Coastal Ocean Productivity (NECOP) program, a mass balance model was developed for the Louisiana Inner Shelf portion of the northern Gulf of Mexico (Bierman *et al.* 1994). Available historical data and new data generated within the NECOP program permitted developing a preliminary, screening-level modeling analysis. The model addresses broad, macro-scale questions related to the impacts of potential reductions in nutrient loads from the Mississippi-Atchafalaya River system (Limno-Tech,

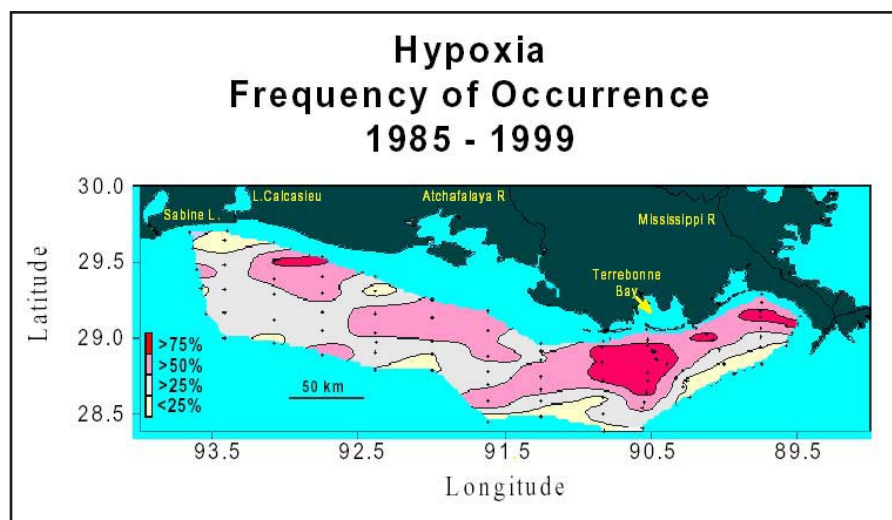


Figure 1. Frequency of occurrence of mid-summer hypoxia in the Gulf of Mexico – based on data from Rabalais, Turner and Wiseman (CENR 2000).

Inc. 1995). This was to determine whether dissolved oxygen concentrations in the Gulf are sensitive to changes in nutrient loads and to forecast the nutrient loads necessary to reduce seasonal hypoxia.

The forecast consisted of model simulations with reduced nutrient loads to determine the degree of improvement in bottom water dissolved oxygen concentrations on the Louisiana-Texas shelf. The rationale was that reductions in nutrient loadings from the MRB would reduce production and settling of organic carbon and hence, the consumption of dissolved oxygen in bottom waters. These simulations involved reductions from 10 to 70 percent in nitrogen and phosphorus loads from the MRB. This range was not intended to

represent reductions feasible in terms of technology, economics or what is socially acceptable; rather, it was to investigate whether reducing loads 20 to 30 percent were sufficient to produce a response, or whether reductions of up to 70 percent would be required.

Results indicate the dissolved oxygen concentrations were sensitive to reductions in nutrient loads; however, there were large uncertainties in the magnitude of the responses (Figure 2). The uncertainty is due to insufficient information on controlling physical, chemical and biological processes, and to the natural variability in hydrometeorological conditions in the northern Gulf of Mexico.

Reducing nutrient loads 20-30 percent showed bottom-water dissolved oxygen concentrations might increase 15-50 percent. Any increase in oxygen above the 2 mg/l threshold has a significant effect on marine life survival, so even small reductions in nitrogen loads are desirable. The forecast indicated somewhat greater responses when nitrogen loads were reduced as compared to phosphorus loads. However, the differences are not large, and considering the uncertainties in the modeling analysis, further investigation is needed to verify this.

These forecast results were used to develop a 30 percent nitrogen load reduction target to meet the long-term Coastal Goal for reducing hypoxia in the Gulf (US EPA 2001). Verification of forecast results will require reductions in nutrient loads from the MRB and continued efforts to monitor the zone of oxygen depletion on the Louisiana-Texas shelf. For such a large system, it may require years to a decade before significant changes can be observed in frequency, duration, size or amount of oxygen depletion.

Lessons Learned

This case study started in a research program (NECOP); however, a national science policy dialogue began with the initiation of the CENR process, with both successes and frustrations. The science in this overall case study helped drive the issues, alternatives and potential consequences. A major accomplishment was establishing watershed-scale links between nutrient loads from the MRB and water quality responses in the northern Gulf of Mexico. Successes were achieved by maintaining visibility with government agencies, legislators, and stakeholders through communication and public outreach, and by an independent peer review process for the scientific results.

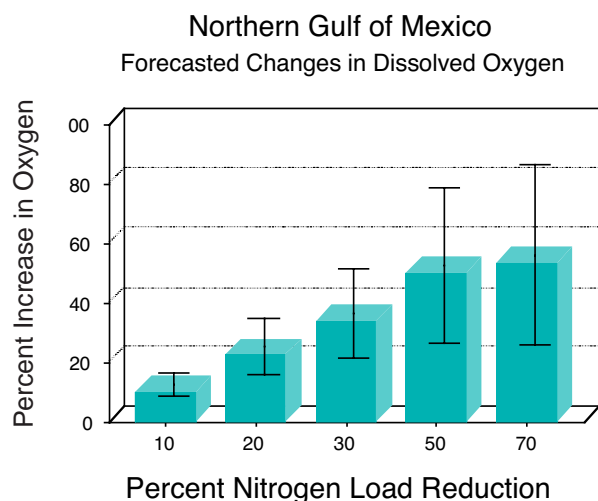


Figure 2. Forecasts from the NECOP water quality model for changes in bottom water dissolved oxygen concentration in response to nitrogen load reductions from the Mississippi River Basin.

Frustrations with this science policy dialogue included preconceptions and misconceptions, and polarization among the scientists involved. Earlier collaboration among government agencies, academics and stakeholders could have made the science more useful. Earlier dissemination of research results might have mitigated the sharp political battle lines that were drawn.

This case study called attention to some challenging questions policy-makers and researchers face in transferring forecast results from research and development to operational decisions. These are just some of the issues.

- When are research results ‘good enough’ to be used in decision-making?
- How should the need for scientific understanding be balanced with the management needs to forecast future behavior?
- How should the policy developed handle uncertainties in research results?
- How should costs and benefits in ecological systems be evaluated?

It became clear that research should be viewed as part of an overall management framework. A comprehensive, carefully targeted program of monitoring, modeling, and research is needed to facilitate continual improvement in scientific knowledge. Adjustments in management practices should be coupled with whatever initial nutrient management strategies are chosen. This adaptive management scheme should involve continual communication among all stakeholders for interpretation of new information and improved management actions.

Next Steps

The scientific impediments to improving forecast ability for water quality responses in the northern Gulf of Mexico can be categorized in terms of monitoring, research and quantitative modeling. The principal impediment is lack of a comprehensive database for the physical, chemical and biological processes that influence hypoxia. Although the existing database is comprehensive in many respects, the data were acquired primarily to characterize water quality responses, not to provide data for quantitative models or forecasting.

For these purposes, a quantitative conceptual model of ecosystem structure and function in the northern Gulf of Mexico should be conceived and used as a foundation for developing future monitoring plans. More funding by government agencies for long-term monitoring is essential to reduce the uncertainties and to verify the accuracy of existing forecast results.

Field data generated by a comprehensive monitoring program are necessary but not sufficient for developing, calibrating and validating quantitative water quality models. There is not yet a complete understanding of the physical, chemical and biological processes that influence water quality responses in the northern Gulf of Mexico. Research is needed to better understand these processes and to provide information for representing them in quantitative models. The most important unresolved research questions include:

- The influence of large-scale Gulf circulation on the Louisiana-Texas shelf;
- The influence of sediment diagenesis;
- The relationship between underwater light attenuation and primary productivity;
- The importance of silica limitation and shifts in phytoplankton species in influencing fate pathways for organic carbon; and,
- The processes controlling carbon, nutrient and oxygen dynamics and cycling.

Modeling advancements are also needed to improve forecasting ability, in particular, the ability to link nutrient loads from the MRB to frequency, duration and areal extent of hypoxia in bottom waters. As part of the CENR Gulf of Mexico Hypoxia Assessment, Brezonik *et al.* (1999) recommended the following refinements and enhancements to the existing NECOP water quality models.

- Advective flows and dispersive mixing coefficients should be determined using the output of a hydrodynamic model of Gulf of Mexico circulation.

- The temporal domain should be extended to include a continuous, time-variable representation of water quality conditions over the complete annual cycle.
- The spatial domain should be extended so that its seaward boundaries are beyond the influence of freshwater and nutrient inputs from the Mississippi-Atchafalaya River system.
- The vertical resolution should be refined to better represent near-bottom hypoxia on the Louisiana-Texas Shelf.
- The horizontal spatial resolution should be sufficiently refined to assess changes in the area and volume of hypoxia under different management strategies.
- Finer spatial-temporal resolution should be employed to represent the dynamics of nearshore waters (shore to 60 meter depth) and linkages with estuaries and offshore waters.
- Model calibration and validation should be conducted over several years (three or more) with varying Mississippi-Atchafalaya River hydrology (average, wet and dry years).
- The conceptual framework should be expanded to include a sediment diagenesis submodel and explicit representation of nutrient and dissolved oxygen mass balances between water column and sediment segments.
- The conceptual framework should be expanded to include diatom and non-diatom phytoplankton functional groups, and silica as a potential limiting nutrient.
- The water quality model should include a separate submodel for underwater light attenuation as a function of background color, biotic solids (phytoplankton) and abiotic solids.

These modeling needs cannot be met independently of the above monitoring and research needs. Models are only tools for synthesizing environmental data and can not be used as substitutes for these data. For a model to be useful, it must have the capabilities for addressing the principal management questions. For a model to be scientifically credible, there must be adequate field data for development, calibration and validation. In the end, there must be compatibility among the management questions, model capabilities, and the available field data.

More information on hypoxia, the Mississippi River Basin, and the Gulf of Mexico can be found on the following websites.

US Environmental Protection Agency

<http://www.epa.gov/msbasin/>

<http://www.epa.gov/gmpo/>

National Oceanic and Atmospheric Administration

http://www.nos.noaa.gov/products/pubs_hypox.html

<http://www.csc.noaa.gov/products/gulfmex/startup.htm>

<http://www.aoml.noaa.gov/ocd/necop/>

US Geological Survey

<http://www.wrcolka.cr.usgs.gov/midconherb/hypoxia.html>

Conclusions

Model forecast results indicate dissolved oxygen concentrations in the northern Gulf of Mexico were sensitive to reductions in nutrient loads. However, there were large uncertainties in the magnitudes of the responses due to lack of sufficient information on controlling physical, chemical and biological processes, and to natural variability in hydrometeorological conditions in the northern Gulf of Mexico. A major accomplishment was establishment of watershed-scale links between nutrient loads from the MRB and water quality responses in the northern Gulf of Mexico.

The study called attention to some challenging questions facing policy-makers and researchers in transferring forecast results from research and development to operational decisions. In view of the complexities of the system and uncertainties in the level of knowledge, research should be viewed as part of an overall management framework. A comprehensive, carefully targeted program of monitoring, modeling, and research to facilitate continual improvement in scientific knowledge and adjustments in management practices should be coupled to whatever initial nutrient management strategies are chosen. This adaptive management scheme should involve continual communication among all stakeholders for interpretations of new information and improved management actions. In particular, modeling needs cannot be met independently of monitoring and research needs.

Models are only tools for synthesizing environmental data and cannot be used as substitutes for data. In the end, there must be compatibility among the management questions, model capabilities, and the available data.

Acknowledgements

The NOAA Coastal Ocean Program Office and the U.S. EPA Gulf of Mexico Program Office provided support

to Limno-Tech, Inc., for this work through Louisiana State University and the University of Minnesota.

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