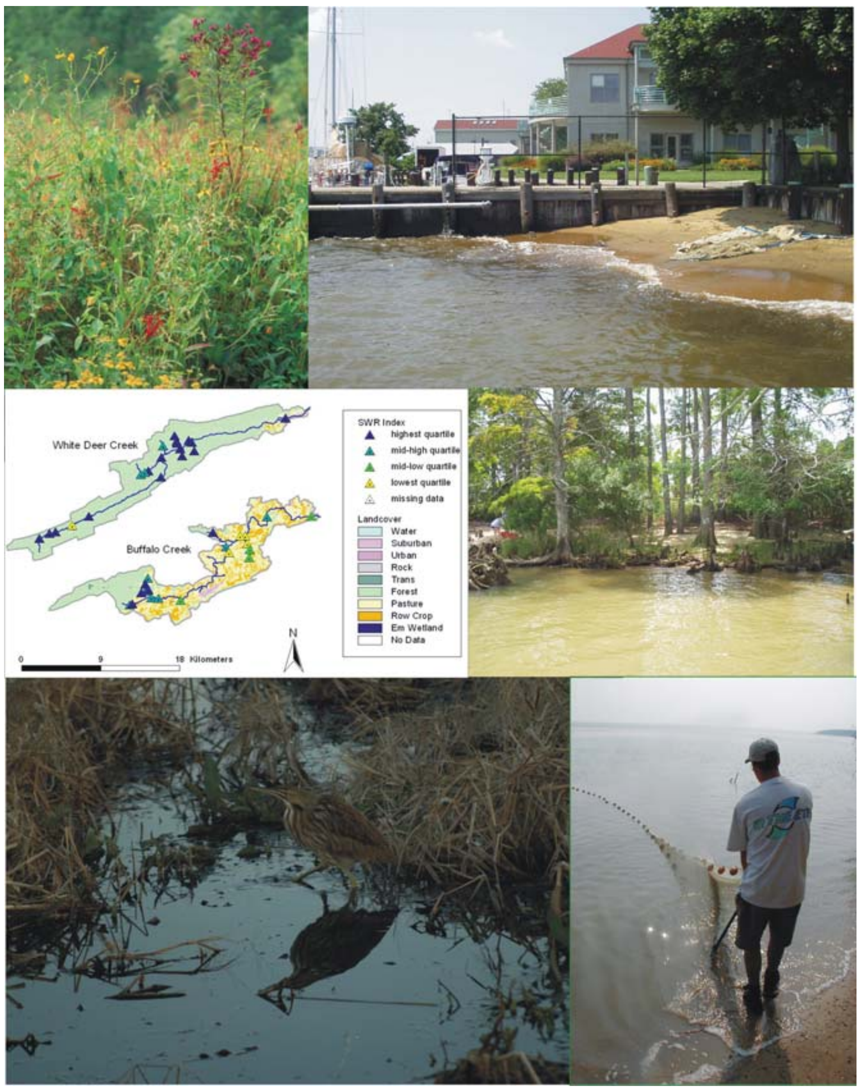
# INTEGRATION OF ECOLOGICAL AND SOCIOECONOMIC INDICATORS FOR ESTUARIES AND WATERSHEDS OF THE ATLANTIC SLOPE




18 February 2006

Atlantic Slope Consortium

# FOREWORD

The Atlantic Slope Consortium (ASC) was conceived to bring together a multidisciplinary team of natural scientists, social scientists, and managers to explore innovative and practical ways to assess and improve the condition of aquatic resources along the Atlantic Slope. Toward that end, we brought together nearly 40 investigators from six institutions:

Pennsylvania State University
Virginia Institute of Marine Science
Smithsonian Environmental Research Center
East Carolina University
Environmental Law Institute
FTN Associates, Ltd.

To accomplish our goals, we convened a dozen intensive “all-hands” meetings in different ecoregions across the Atlantic Slope. Many other meetings involving subsets of our team were held as were numerous conference calls and email communications. Members of the ASC participated in a wide array of conferences, workshops, and outreach activities reporting on the progress of our collective work over the 5-year project period. We joined similar groups funded through U.S. EPA’s STAR Program – the EaGLes – Estuarine and Great Lakes Environmental Indicators Program, to collaborate on complementary projects and to compare notes on administering large, multi-institutional research projects.

Throughout this venture, the levels of creativity, diligence, and camaraderie displayed were truly astounding. The many participants of the ASC’s first project, from research scientists to academic faculty and graduate students, from agency scientists and managers to technicians and clerical staff, should acknowledge to themselves that they have created a body of work that will influence the way environmental resources are assessed, managed, and conserved for decades to come. We give special thought to a colleague who we regret is no longer with us, Charles Taillie. As the ASC’s Director, it has been both a privilege and a pleasure to guide this project to its conclusion. My sincere appreciation goes out to each and every one who contributed to our collective success.

Although there are many individuals from the member institutions and from amongst our collaborating organizations to thank, two colleagues stand out and should be named to acknowledge their essential contributions. First, this project would not have been initiated without the foresight and persuasion of Tom DeMoss of U.S. EPA Region 3 and the MAIA team. He had the vision and provided the encouragement to propel us forward and keep us relevant which has been, and is, a continuing MAIA theme. He, and MAIA Team members, provided us with mid-course corrections when we might have veered from the logical path to completion. Lastly, our Project Officer from USEPA, Barbara Levinson, deserves our deepest gratitude for adeptly keeping us within the administrative and fiscal boundaries while always encouraging us to do our most capable work. I know the ASC Team and the other EaGLe

Directors will join me in thanking Barbara for her guidance, insight, and humor. It was a wonderful journey made much better by her enthusiastic participation in all phases of the project.

This research has been supported by a grant from the U.S. Environmental Protection Agency's Science to Achieve Results (STAR) Estuarine and Great Lakes (EaGLe) program through funding to the Atlantic Slope Consortium, U.S. EPA agreement R-82868401. Although the research described in this report has been funded wholly or in part by the United States Environmental Protection Agency, it has not been subjected to the Agency's required peer and policy review and, therefore, does not reflect the view of the Agency and no official endorsement should be inferred.

Robert P. Brooks, Director, Atlantic Slope Consortium
February 2006

The recommended citation for this report is:

Brooks, R.P., D.H. Wardrop, K.W. Thornton, D. Whigham, C. Hershner, M.M. Brinson, and J.S. Shortle, eds. 2006. Integration of ecological and socioeconomic indicators for estuaries and watersheds of the Atlantic Slope. Final Report to U.S. Environmental Protection Agency STAR Program, Agreement R-82868401, Washington, DC. Prepared by the Atlantic Slope Consortium, University Park, PA. 96 pp. + attachments (CD).

# EDITORS* AND AUTHORS

## Atlantic Slope Consortium (ASC)

Pennsylvania State University (PSU)
Robert P. Brooks*
Denice H. Wardrop*
Brian Armstrong
Amy Balog
Joseph A. Bishop
Tatiana Borisova
Mary M. Easterling
Barry M. Evans
James C. Finley
Ann N. Fisher
Jeremy Hite
Kristen Hychka
Maurie Caitlin Kelly
Jennifer Kipp
Suzy Laubscher
Elizabeth P. Marshall
Wayne L. Myers
Egide Nizeyimana
Timothy J. O'Connell
Robert E. O'Connor
G. P. Patil
Christopher Pfeiffer
Richard Ready
Jennifer Rubbo
Kristen Saacke-Blunk
James S. Shortle*
Richard Stedman
Charles Taillie
Liem Tran

Smithsonian Environmental Research Center (SERC)
Dennis F. Whigham*
Matthew Baker
William DeLuca
Charles L.Gallegos
Anson H. Hines
Thomas E. Jordan
Ryan King
Peter P. Marra
Donald E. Weller

Virginia Institute of Marine Science (VIMS)
Carl Hershner*
Marcia Berman
Donna Marie Bilkovic
Kirk J. Havens
David O'Brien
Lyle M. Varnell

East Carolina University (ECU)
Mark M. Brinson*
Chris Bason
Richard Rheinhardt

Environmental Law Institute (ELI)
James M. McElfish

FTN Associates, Ltd. (FTN)
Kent W. Thornton*
Christina Laurin
Bernadette Schane

## Synopsis Authors

Robert P. Brooks of Penn State Cooperative Wetlands Center, 302 Walker Building, Pennsylvania State University, University Park, PA 16801

Kent W. Thornton of FTN Associates, Ltd., 3 Innwood Circle, Suite 220, Little Rock, AR 72211

Denice H. Wardrop of Penn State Cooperative Wetlands Center, 302 Walker Building, University Park, PA 16801

## Introduction Author

Robert P. Brooks, Mary M. Easterling, Joseph A. Bishop, Jennifer Rubbo, Brian Armstrong, and Jeremy Hite of Penn State Cooperative Wetlands Center, 302 Walker Building, Pennsylvania State University, University Park, PA 16801

## Message 1 Authors

Denice H. Wardrop of Penn State Cooperative Wetlands Center, 302 Walker Building, University Park, PA 16801

Carl Hershner and Kirk J. Havens of Virginia Institute of Marine Science, PO Box 1346, Gloucester Point, VA 23062

Kent W. Thornton of FTN Associates, Ltd., 3 Innwood Circle, Suite 220, Little Rock, AR 72211

## Message 2 Authors

Dennis F. Whigham, Ryan King, William DeLuca, Peter P. Marra, Anson H. Hines, and Charles L. Gallegos of Smithsonian Environmental Research Center (SERC), Box 28, Edgewater, MD 21037

Donna Marie Bilkovic and Carl Hershner of Virginia Institute of Marine Science (VIMS), Box 1346, Gloucester Point, VA 23062

## Message 3 Authors

Robert P. Brooks, Mary M. Easterling, Joseph A. Bishop, Jennifer Rubbo, Brian Armstrong, and Jeremy Hite of Penn State Cooperative Wetlands Center, 302 Walker Building, Pennsylvania State University, University Park, PA 16801

Mark M. Brinson and Richard Rheinhardt of East Carolina State University, Howell Science Complex, N-108, Greenville, NC 27858

Matthew Baker, Ryan King, and Donald E. Weller of Smithsonian Environmental Research Center, Box 28, Edgewater, MD 21037,

David O'Brien and Kirk J. Havens of Virginia Institute of Marine Science, PO Box 1346, Gloucester Point, VA 23062

## Message 4 Authors

James S. Shortle of the Penn State College of Agricultural Science, 112-C Armsby, Pennsylvania State University, University Park, PA 16802

Kent W. Thornton of FTN Associates, Ltd., 3 Innwood Circle, Suite 220, Little Rock, AR 72211

## Synopsis
## Integration of Ecological and Socioeconomic Indicators for Estuaries and Watersheds of the Atlantic Slope

## Introduction

Coastal ecosystems, and their watersheds, are at risk from human activities, past and present. With over half of the world's human population residing within 100 km of coastlines, and increasing densities likely in the coming decades, it is essential that humans be considered part of, not apart from, these valuable aquatic ecosystems. Sixty-five percent of the monitored coastal estuaries exhibit signs of moderate to high levels of eutrophication (Bricker et al. 1999). "Dead zones" in coastal waters are increasing, not only in the Gulf of Mexico, but also in Chesapeake Bay, Delaware Bay, Long Island Sound, and other estuaries around the country (EPA 2004). Recognizing the critical influences that human activities have on the ecological condition of the interconnected aquatic ecosystems of coastal areas – wetlands, streams, rivers, and estuaries– the Atlantic Slope Consortium (ASC) team spent five years developing a suite of ecological and socioeconomic indicators for assessing and managing the condition of these vital resources in the Mid-Atlantic region (Figure 1).

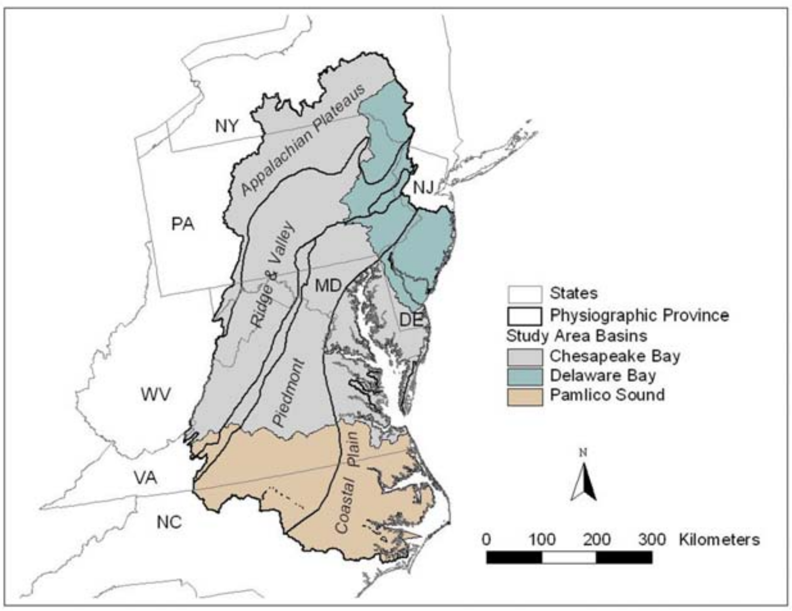



**Figure 1.** ASC study area - Mid-Atlantic Slope.

## Atlantic Slope Consortium

The ASC is a collaboration among scientists with the Pennsylvania State University, the Smithsonian Environmental Research Center, the Virginia Institute of Marine Science, East Carolina University, the Environmental Law Institute, and FTN Associates, Ltd. The ASC project, formally entitled "Development, Testing, and Application of Ecological and Socioeconomic Indicators for Integrated Assessment of Aquatic Ecosystems of the Atlantic Slope in the Mid-Atlantic States," is one of five projects funded nationally by the U.S. Environmental

Protection Agency Office of Research and Development through its Estuarine and Great Lakes (EaGLe) Indicator Research Program, part of EPA's STAR Grants Program. With the numbers of people living and working in the coastal zone increasing in the U.S., pressures on these critical resources have correspondingly increased. The EaGLe Program was designed to develop a new generation of ecological indicators that could aid managers in determining the condition and diagnosing the cause of degradation in estuarine ecosystems (Niemi et al. 2005). The ASC chose to extend investigations upstream of estuaries to include contributing watersheds.

## The Atlantic Slope

The project study area is the Mid-Atlantic Slope, encompassing three major drainage basins that extend from the Appalachian Mountains to the Atlantic Ocean: the Delaware, the Susquehanna-Chesapeake, and the Albemarle-Pamlico (Figure 1). This area includes portions of eight states and the District of Columbia: Delaware, Maryland, New Jersey, New York, North Carolina, Pennsylvania, Virginia, and West Virginia. Aquatic resources in this area have been heavily impacted by urbanization, agricultural production, mining and other human activities. Although much has been done to restore and protect freshwater and estuarine resources in this area, threats to life and health for both humans and other biota continue to be major issues of concern.

## Goal

The goal of this project was to develop a set of indicators for coastal systems that are ecologically appropriate, economically reasonable, and relevant to society to further inject science into natural resources management decisions. This suite of indicators can contribute to integrated assessments of the health and sustainability of aquatic ecosystems in the region. The indicators were developed based on ecological and socioeconomic information compiled at the scale of estuarine segments and small watersheds, with clear linkages to larger scales (Figure 2).

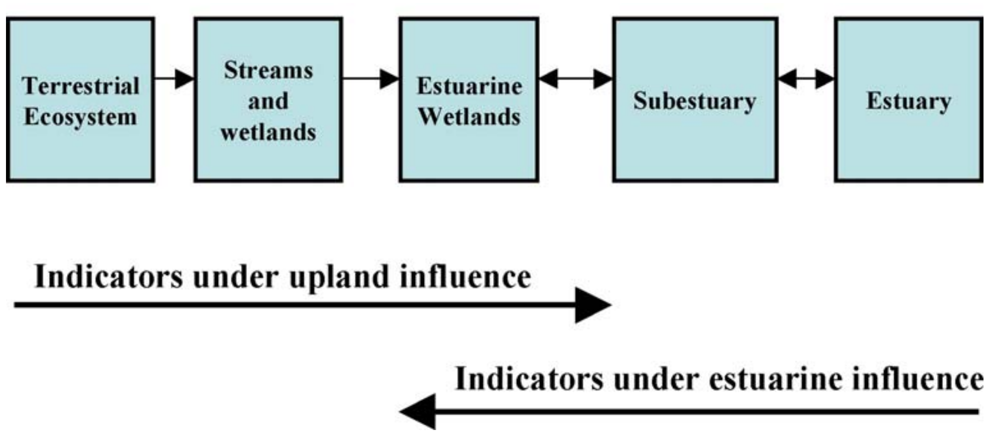



**Figure 2.** Conceptual model for purposes of identifying indicators.

# Conceptual Framework and Premises

This project was guided by a number of premises. Our first premise is that humans are part of, not apart from, coastal ecosystems and their watersheds. Individuals make choices concerning their use of private property based on their needs, desires, and perceptions. In a given community, watershed, or region, these collective decisions result in characteristic patterns of land use which we call *social choices*. These *social choices* can affect aquatic resources which are common public resources available to all. Since society has designated the uses they want for aquatic or public resources, government has been charged by society to attain and sustain these designated uses. We term these types of public decisions *societal choices*. When private *social choices* about land use affect the public's designated uses (*societal choices*) by altering the condition of aquatic resources, conflicts can arise.

We addressed these issues based on our second premise, which was that it is not possible to describe a single reference condition for the varied landscapes contained within the watersheds and estuaries of the Mid-Atlantic Region. Land use patterns are determined by both ecological and cultural factors, and these relationships vary across space and time. This means that there is no optimal management solution with universal applicability throughout the Mid-Atlantic region. The options available to managers, therefore, are dependent upon multiple benchmarks reflecting these varied landscapes that have evolved from social choices made over time by landowners (Hershner et al. In press).

To decipher this variability, a classification system of landscape patterns emerged for watersheds and estuaries of the Mid-Atlantic Region based on six social choice categories: two forest types, two mixed land use types, agriculture, and urban (Wardrop et al. 2005). New methods, analytical techniques, and indicators, developed during this project, demonstrated that landscape patterns can be linked to the condition of aquatic resources, from headwaters to estuaries. While there is no "best" landscape pattern that aligns with social or societal choices within watersheds, there are landscape patterns associated with non-attainment of designated uses for aquatic ecosystems.

A third premise guiding our search for indicators was that if they are to be useful, indicators must be practical. Part of the practicality of using indicators is helping managers choose among the many measurement techniques available. Through our surveys, we learned that environmental managers are not looking for a "silver bullet," but rather a suite of indicators to assess resources. We also learned that the perceptions of scientists, managers, and citizens about condition of aquatic ecosystems can vary, which has implications about how indicator information should be communicated. The ASC developed a taxonomy that informs the user about the type of indicator, relevant spatial and temporal scales, and relevant questions being asked (Wardrop et al. In press, Figure 3). Thus, regardless of whether an indicator is being used to assess a biological, chemical, physical, geographic, or cultural attribute, or whether the time frame of measurement is hours or years, the user can select the appropriate tool for the question being asked or the decision to be made. Being able to classify each indicator provides an improved level of certainty for the user.

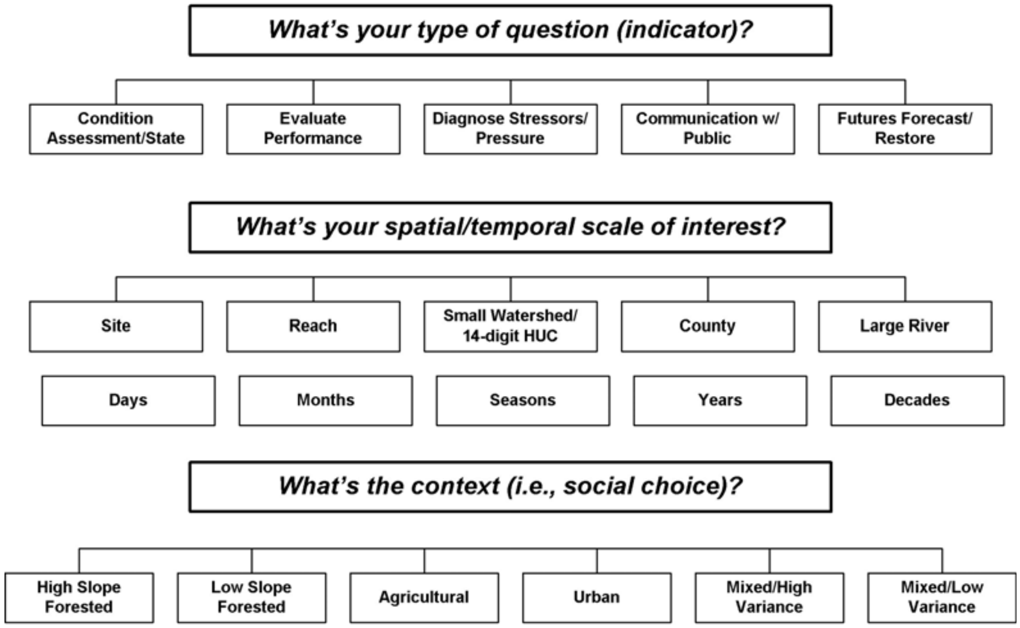



**Figure 3.** Taxonomy of Ecological Indicators

Rather than let these indicators stand alone, we have produced a coherent conceptual framework that shows how the indicators can be useful to environmental managers and understandable to citizens. We have described this framework in Message 1 of the ASC Synthesis Report. This message is followed by Messages 2, 3, and 4, which present our findings about estuarine, watershed, and socioeconomic indicators, respectively. Environmental managers in the region now have a variety of indicators in their "tool box" to assess the condition of aquatic ecosystems.

## Management Messages

Four important messages have come out of this project:

1. A taxonomy for classifying indicators based on the type of questions they can answer, what spatial and temporal scale they reflect, and the *social choices* they address, helps resource managers choose indicators that are most appropriate for their use (Figure 3).

2. Estuarine fish and wetland bird community indicators conclusively demonstrated that both

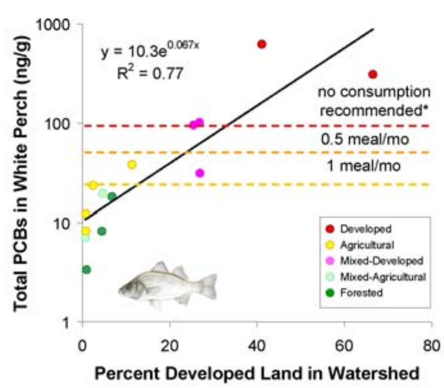



**Figure 4.** Total PCBs in white perch in relation to percent developed land in the watershed. USEPA (1999) guidelines for cancer health endpoints.

the amount of development in the watershed and its proximity to the estuary or wetland contribute to the condition of these aquatic resources. In general, the greater the amount of development and its proximity, the greater the degradation of aquatic resource condition. (Figure 4).

3. Strong linkages were found not only between the amount of development and proximity on stream and wetland condition in small watersheds, but also the patterns of land use (Figure 5).

4. Socioeconomic indicators can be combined with environmental indicators to show most communities in the Mid-Atlantic region do not have the quality of life possible, even when accounting for urban and rural differences (Figure 6).

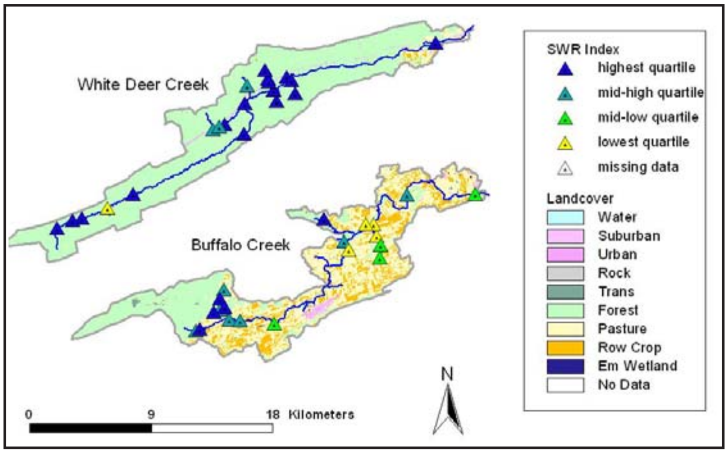



**Figure 5.** Nested SWR Index and Lanscape index.


**Figure 6.** Frontier concept showing the distinction between rural and urban regions and where Community A currently is compared to where it could be (regional curve).

Each of these messages is explained in greater detail in the ASC Synthesis Report and each indicator is described fully in technical papers encoded on a CD attached to the Synthesis Report.

## Completing the Vision

We have made significant progress toward our goal of developing a suite of ecological and socioeconomic indicators for the aquatic ecosystems of the Mid-Atlantic Region. Yet, more work is needed. In particular, additional field studies and modeling efforts are needed to complete the linkage of regional headwaters to estuaries. The contributions of large rivers to material transport and processing, and how receiving estuaries are affected by those inputs, remains poorly understood and difficult to predict. In addition, sets of indicators that span the variations in salinity and depth across estuaries have not been developed sufficiently. As further development, testing, and implementation of indicators occurs, these indicators can be classified using the taxonomy to ensure that the full range of indicator types are available to managers.

An ASC Synthesis Report and accompanying CD describe in detail the findings of the Atlantic Slope Consortium's project to develop ecological and socioeconomic indicators that describe the condition of aquatic resources in the Mid-Atlantic Region.

## Literature Cited


Bricker, S.B., C.G. Clement, D.E. Pirhalla, S.P. Orlando, and D.R.G. Farrow. 1999. National Estuarine Euthorphication Assessment: Effects of Nutrient Enrichment in the Nationa's Estuaries. NOAA, National Ocean Service, Special Projects Office and the National Centers for Coastal Ocean Science. Silver Spring, MD.

Hershner, C., K. Havens, D.H. Wardrop and D.M. Bilkovic. A practical concept for developing indicators of aquatic ecosystem health. EcoHealth (in press).

Niemi, G., D. Wardrop, R. Brooks, S. Anderson, V. Brady, H. Paerl, G. Rakocinski, M. Brouwer, B. Levinson, and M. McDonald. 2004. Rationale for a New Generation of Indicators for Coastal Waters. Environmental Health Perspectives: Annual Review Issue. 112(9):979-986.

US EPA. 2004. National Coastal Condition Report II. EPA-620/R 03/002. U.S. Environmental Protection Agency. Office of Research an Development/Office of Water. Washington, DC.

Wardrop, D.H., J.A. Bishop, M. Easterling, K. Hychka, W.L. Myers, G.P. Patil, and C. Taille. 2005. Use of landscape and land use parameters for classification and characterization of watersheds in the Mid-Atlantic across five physiographic provinces. Environmental and Ecological Statistics 12:209-223.

Wardrop, D.H., C. Hershner, K. Havens, K. Thornton and D.M. Bilkovic. Developing and communicating a taxonomy of ecological indicators: a case study from the Mid-Atlantic. EcoHealth (in press).

# TABLE OF CONTENTS

## LIST OF APPENDICES

(included on accompanying CD)

# INTRODUCTION

We start with a profile of our study area to provide geographic context for the project, and then present our goals, approach, and results.

## Study Area

> "…rivers and principal creeks rise in the high country…or in the ridges continuous therewith. Flowing between the groups of hills, and forking at frequent intervals, they run swiftly in a general southeast direction until the wider valleys are reached, and then they stretch more broadly onward to empty into the estuaries…" (Scharf 1881, p.14).

Scharf's description of Baltimore County (1881) (see Sidebar) is applicable to the entire study area of the Atlantic Slope Consortium project. The project focused on the Atlantic Slope of the Mid-Atlantic States from the central Appalachians to the Atlantic Ocean (Figure 7). This area includes the major drainage basins of the Susquehanna River, Delaware River, and the Albemarle-Pamlico, Delmarva and North Carolina coastal bays. A diversity of ecoregions that are representative of a significant portion of the East Coast occur in the study area, including the unglaciated Appalachian Plateau, glaciated Pocono Plateau, Ridge and Valley Province, Piedmont, Mid-Atlantic Coastal Plain, and Southeastern Coastal Plain ecoregions. The study area includes portions of New York, Pennsylvania, Maryland, Delaware, New Jersey, Virginia, West Virginia, North Carolina, and the District of Columbia. The study area contains a mosaic of urban (e.g., Philadelphia, Baltimore, Washington DC, Richmond, Norfolk) and rural (e.g., forests and farmlands of the Delmarva Peninsula, recreational and tourism areas of the Pocono's, forested Northern Tier counties of Pennsylvania) landscapes (see Table 1). Overall, the size of the study area is 108,000 $mi^2$.

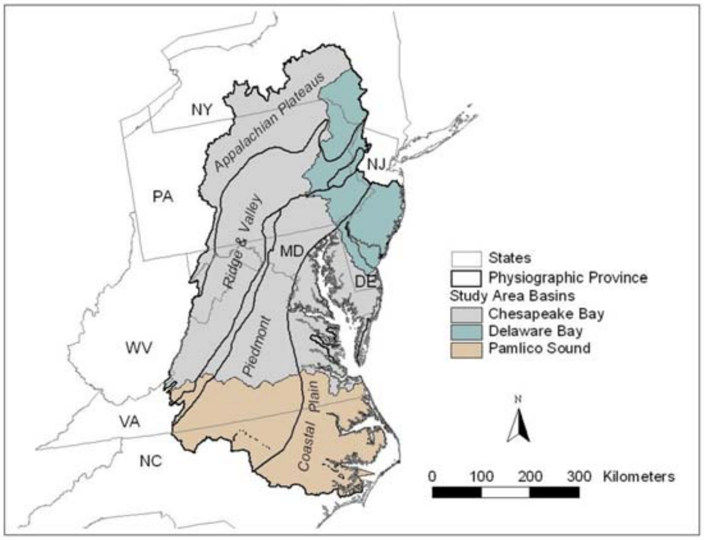



**Figure 7.** Atlantic slope region.

**Table 1. Examples of urban and rural landscapes in the ASC study area.**

| Urban Landscapes | Rural Landscapes |
|---|---|
| Philadelphia | Forests of the Delmarva Peninsula |
| Baltimore | Farmlands of the Delmarva Peninsula |
| Washington, DC | Recreational and tourism areas of the Poconos |
| Richmond | |
| Norfolk | |

## Project Background

As early as 1881, Scharf provided us with an assessment of the condition of the aquatic ecosystems of the region (see Sidebar), albeit in narrative form, and already changes were taking place – the conversion of forests to fields, the founding of towns and cities, and the depletion of natural resources. Not long afterward, the U.S. Congress passed the Rivers and Harbors Act of 1899 to begin to protect the nation's waterways from dredge and fill activities in navigable waters. The impacts to estuarine resources and associated aquatic ecosystems resulting from human development have been devastating and well documented (e.g., Paul et al. 1998). The degradation of these waters spawned regulatory and non-regulatory responses aimed at remediating both point and nonpoint pollution (e.g., Clean Water Act of 1972, Chesapeake Bay Agreement). Through implementation of the Clean Water Act and associated state laws, the goal of returning to "fishable and swimmable" conditions has been achieved to some degree and in some areas. Threats to life, human and other biota, however, continue to be major issues of concern in the Atlantic Slope region.

## Project Team

The Atlantic Slope Consortium (ASC) was made up of 43 individuals from six institutions; the Pennsylvania State University, the Smithsonian Environmental Research Center, the Virginia Institute of Marine Science,

### Baltimore County from Scharf 1881

Baltimore County forms an important part of the great continental belt of country known as the ***Atlantic Slope*** of North America. Supplied by nature with an abundance of water and wood, with soils easily cultivated, and capable of yielding ample harvest of all the cereals, vegetables, and all the best fruit of temperate climates, it rest only with the inhabitants to advance their own interests by adjusting themselves to the surrounding physical conditions.

Structurally, it possesses the most important elements which give strength, variety, and character to the Atlantic region. The contours of surface are chiefly brought into prominence by the underlying reliefs of hard rocks and of the solid materials derived from them. For convenience, the surface of the county may be divided into an upland region, a midland basin, and a lowland border.

**Uplands.** In the very midst of these lower hills an abrupt ridge of dark fissured rocks occasionally rises, where a rapid stream has cut a deep ravine in its downward flow. These waters are still clear, and do good service in furnishing power to flour mills which stand hid away here and there in unsuspected dells or hollows.

**Midlands.** It presents a wide area of open country, depressed below the general level, occupied by large farms, and wooded only on the hills and ridges which project into it. General affluents of the Gunpowder cross it, and an abundant supply of good drinking-water is obtained from wells.

**Lowlands.** The lowland section is an alluvial belt of country which bounds the hills of archaean rocks on their tide-water sides… Only a few years ago this section was much wilder than now, the waters were abundantly stocked with fish and reptiles, and wading birds – such as the great blue heron, the egret, lesser heron, and belted kingfisher – held complete sway over the humble inhabitants of every cove, pool, and swamp.

East Carolina University, the Environmental Law Institute, and FTN Associates, Ltd. These individuals have expertise in a number of disciplines related to the study and management of watersheds and estuaries, socioeconomics, and Geographic Information Systems (GIS).

The ASC was formed to conduct one of five projects funded nationally by the U.S. Environmental Protection Agency through its Estuarine and Great Lakes (EaGLe) Indicator Research Program, part of EPA's STAR Grants Program. The ASC project was formally titled "Development, Testing, and Application of Ecological and Socioeconomic Indicators for Integrated Assessment of Aquatic Ecosystems of the Atlantic Slope in the Mid-Atlantic States."

## Project Goal

The goal of the ASC project was to develop and test a set of indicators in coastal systems that were ecologically appropriate, economically reasonable, and relevant to society. Our suite of indicators can produce integrated assessments of the condition, health and sustainability of aquatic ecosystems, based on ecological and socioeconomic information compiled at the scale of estuarine segments and small watersheds, and with clear connections to smaller and larger scales.

Given limited resources to direct toward the assessment and protection of ecosystem health, a suite of ecological and socioeconomic indicators, if properly selected, evaluated, and synthesized, can help scientists, managers, and policy makers document trends, prioritize issues, and target management activities. By providing a reliable expression of environmental stress or change, ecological indicators can integrate impacts that are spatially and temporally disparate. The concept of using ecological indicators to assess ecological integrity was summarized by McKenzie et al. (1992), while Messer (1992) discussed concerns regarding the development of regional indicators. For a regional ecological monitoring network to be useful, it must relate closely to societal concerns and be defensible for decision-makers (Brooks 1991, Messer 1992, Angermeier and Karr 1994). Noss (1990) and Poiani et al. (2000) discussed the importance of directing conservation and monitoring efforts at four scales that should be addressed by ecological indicators: regional landscape, community/ecosystem, population/species, and genetic. In this project, we demonstrated how suites of indicators can encompass the first three levels of organization and be used for making decisions or assessing risk at a variety of scales. In addition, we provided information on the uncertainty associated with the indicators and thresholds at which significant and noticeable changes begin occurring in aquatic ecosystems because of human activity.

## Project Objectives

The specific project objectives were:

1) To develop and test ecological and socioeconomic indicators of aquatic resource condition, construct models that use environmental, geographic, and stressor

data to predict indicator responses, and use models to link upstream watersheds and downstream estuaries.

2) To develop large scale measures for characterizing landscape attributes and land-use patterns to serve as predictors of a range of environmental conditions.

3) To deliver a nested suite of indicators to managers, where the implications of aggregating models at various scales are considered, and for which reliability is known.

## Project Approach

Our research was guided by a question based on Scharf's statement about Baltimore County (see sidebar); ***Since his time, have the inhabitants of the Atlantic Slope advanced their own interests by adjusting themselves to the surrounding physical conditions?*** The ASC team sought to address the question of how humans interact with the aquatic environments in the 21st century by developing ecological indicators of the condition of the Atlantic Slope's aquatic resources and understanding the socioeconomic perceptions and values that lead to the adoption and acceptance of indicators. In attempting to answer the question above, our development and selection of indicators was guided by several premises.

Our first premise was that humans are part of, not apart from, all ecosystems and human activities, needs, and desires need to be considered for natural resources management to be practical and sustainable. The most common benchmark used for grading the health of ecosystems is the condition of the ecosystem prior to large scale changes by human society. However, because humans do live in and affect these systems, it may not be possible to achieve the level of ecosystem health that existed before large-scale settlement. Therefore, indicators need to help managers and communities understand the impacts they have on coastal ecosystems, and what, if anything, they can do to improve it. Also, methods are needed that can help managers and communities use indicators to make informed decisions about what they want, and can get, from coastal ecosystems.

Indicators also need to help managers and communities understand how the health of coastal ecosystems affects their society, economy, culture, and quality of life. Therefore, indicators are needed that monitor elements of coastal systems that are directly relevant to human activities, needs, and desires.

Our second premise was that the collective choices people make in their use of private property in their community, and in the region results in characteristic patterns of land use we call *social choices*. Individuals make choices concerning their use of private property based on their needs, desires, and perceptions. These *social choices* affect water resources, which are recognized as one of many common public resources available to all.

Our third premise was is that society as a whole designates the uses desired for public water resources. We call these *societal choices*. Society has designated the uses for public water resources through federal and state legislation (e.g., Clean Water Act, the Safe Drinking

Water Act), and water quality standards. Government has been charged by society to make sure these designated uses are attained and supported. Conflicts can arise when private *social choices* affect the ability of public water resources to sustain or attain the desired uses designated through *societal choices*.

Our fourth premise was that if they are to be useful, indicators must be practical. One aspect of this practicality is that managers need help identifying the indicators that are appropriate and most useful for their situation. The portfolio of assessment tools for aquatic ecosystems has been expanded substantially in the past several decades with improved levels of detection for chemical pollutants, expanded use of biological indicators (e.g., Barbour et al. 1999), and implementation of citizen monitoring programs (e.g., Ely 1998). Still, gaps remain in the availability of appropriate indicators for some ecosystems (e.g., tidal and freshwater wetlands, headwater streams, riparian corridors). The available set of ecological and socioeconomic indicators for assessing and predicting environmental integrity, health, and sustainability is not complete, nor is it formulated for consistent assessments across aquatic ecosystem types or scales. As a consequence, management activities are not effectively or efficiently applied across the wide array of political divisions. Thus, application of the existing array of indicators for coastal ecosystems could be improved. To do so, a common language is needed for managers of terrestrial and aquatic resources, so that their assessments and subsequent management activities can be linked from headwater streams to estuaries, including wetlands, floodplains and rivers, lakes and reservoirs, and tidal wetlands and coastal bays. Only in this way can these critical ecosystems be protected. Environmental managers are not looking for a single indicator, or "silver bullet", but rather a suite of indicators to assess resources. Therefore, an indicator classification tool is needed to help managers compare and select indicators. Being able to classify each indicator provides greater certainty about the relevance of the results for a particular application.

To be useful, indicator results need to be communicated to the public in a meaningful and timely way. In this project, we learned that scientists, managers, and citizens often have different perceptions about the condition of water resources. This has implications for how information about indicators should be communicated.

Another aspect of an indicator's usefulness is how easy it is to measure. Indicators that use methods that are already in common use, are reasonably inexpensive, and provide results in a relatively short time are preferred.

Our fifth premise was that coastal estuarine ecosystems are affected by human activities on lands that drain directly to the ecosystem, and by human activities in the watersheds of streams and rivers that drain these watersheds to the system, which can be far away from the coast. During this project, we sought to connect science with management, research with practice, and decision-making with public attitudes. To do so, coasts, estuaries, rivers, streams, lakes, and wetlands must be viewed as one integrated system, and, when combined with the contributing terrestrial areas (i.e., watersheds), they comprise a watershed or estuarine system. Only a fully integrated approach of inventory, assessment, and restoration can effectively and efficiently protect the nation's waters and the biota dependent upon them

(Brooks et al. in press). The availability of a defensible and useful suite of environmental indicators is essential for this to happen. With this project, we wanted to close the loop between integrated assessment and integrated management. That has been the strength of the approach used by the ASC.

Our sixth premise was that most natural resource management decisions are made at the scale of estuarine segments and small watersheds. Our approach was to define an appropriate and relevant assessment and management unit that was applicable to wetland, lake, stream, and estuarine systems alike. A unit within an estuarine system was denoted as an estuarine segment. Estuarine segments were composed of deepwater areas, vegetated and unvegetated shallows, tidal wetlands and creeks, and the adjacent terrestrial habitats. An equivalent unit upstream of estuaries was denoted as a small watershed. These areas were typically sized as tens to hundreds of $km^2$ (U.S. Geological Survey, 14-digit Hydrologic Unit Code, HUC), and encompassed several stream or river reaches, adjacent riparian corridors, associated wetlands and waterbodies, and the contributing terrestrial drainage basin.

There are logical reasons why estuarine segments and small watersheds served as the focal units to organize data collection and analysis for our indicator development and application:

- Units of this size are central to the entire dimensional range of information collected, spanning several orders of magnitude. The majority of environmental data are collected at smaller scales, such as points, plots, reaches, and sites. These data can be efficiently aggregated up to the scale of an estuarine segment or small watershed without significant loss of resolution. Similarly, socioeconomic data are usually collected at the person or family level, and can be aggregated on census blocks to represent communities and municipalities.
- Based on our discussions with managers and decision-makers, management activities for aquatic ecosystems can be effectively targeted and reported for units of this scale, such as towns and counties. Thus, there would be a convergence of scales for both assessment and management, for both ecological and socioeconomic data.
- A nested or hierarchical approach to data collection, assessment, and management provides opportunities to address larger emergent properties and regional issues at landscape and ecoregion scales. Yet, it still allows investigators and managers to trace the origins of those data, and transmit the risks of relying on that information to make decisions.

The linkages among aquatic ecosystems, terrestrial surroundings, and societal activities are postulated, sometimes confirmed, but rarely integrated. To achieve better ecosystem management, it is essential that a higher level of synthesis take place, one where indicators clearly link to stressors, to sources, and to solutions.

Our seventh premise was that there is no single reference condition or benchmark for the varied physical, social, and biological landscapes within the watersheds and estuarine systems of the Mid-Atlantic Slope. Landscape configurations are determined by both ecological and cultural factors, and these relationships vary across space and time. This means that there is no optimal management solution with universal applicability. Managers, therefore, need multiple reference conditions or benchmarks to reflect the variable landscapes they manage, which are controlled by physiographic settings and by land use patterns that have evolved from social choices made over time by landowners (Hershner et al. in press).

## Report

In this project, the team developed a number of new indicators (over 30 to date) applicable to estuarine ecosystems and their upstream watersheds in the Mid-Atlantic Slope. In addition, the team also developed a classification system for indicators. Details about the project's findings can be found in the remainder of this document and in the attached CD, which contains the appendices and many of the published papers and unpublished reports produced during this project.

This report is organized based on four messages that highlight the results of the project. Message 1 describes the classification taxonomy for indicators developed in the project. This taxonomy shows how the indicators can be useful to environmental managers and understandable to citizens through classification, recognition of the appropriate reference state, and practical applications. Message 2 presents our findings about estuarine indicators. Message 3 presents our findings about watershed indicators. Message 4 adds social relevance to the ecological indicators by discussing what managers and citizens consider useful in selecting and interpreting indicators, and how communities can make informed decisions about maintaining and restoring the health of water resources in their watersheds and estuarine systems. In Table 2, the indicators we developed are listed and briefly described. In Appendix A (found on the accompanying CD), the indicators developed by the ASC are presented as brief 1 to 2 page synopses formatted so that managers can select indicators based on the type of question they wish to answer, the appropriate spatial and temporal scales for implementing the indicator, and the context in terms of "social choice" land use (Wardrop et al. in press).

Based on the findings compiled in this report, environmental managers along the Mid-Atlantic Slope and elsewhere now have at their disposal a variety of measures to indicate the health of aquatic ecosystems. The methods presented here will allow communities of similar population densities, economic status, and cultural backgrounds to be compared. This offers managers and citizens a chance to make decisions within the context of similar communities. Once analyzed, these monitoring results should be communicated in a timely way to the public.

[THIS PAGE INTENTIONALLY LEFT BLANK]

# MESSAGE 1

## Introduction

A healthy aquatic ecosystem is one that can sustain its intended uses. This simple statement defines the approach our research team has taken in the search for useful indicators of aquatic system conditions in the Mid-Atlantic region. This has been a useful strategy based on our desire that indicators meet tests of both technical merit and practicality.

Developing indicators of aquatic ecosystem health that are practical tools for resource management means the concept of health must be defined so that it is possible to measure, and that the indicators link health directly with management needs and practices. This objective helped us focus on ecosystem services rather than the fuzzy concept of sustainability. This objective also incorporates the selection of reference conditions and provides a direct connection to the water quality management goal of attaining designated aquatic uses. Focusing on ecosystem services reduced the number of options for indicator development.

Indicators intended to inform management must be practical. The underlying metric must be something that has a reasonable cost: information ratio. Metrics that require unique analytical capabilities are likely to find only limited application, whereas metrics using commonly available analytical capabilities and technologies will have comparatively greater utility.

A final challenge in the development of indicators of aquatic ecosystem health is identifying those metrics that show a response over the entire gradient of stress from no or minimal stress to severely stressed. An underlying construct in many indicator development efforts has been the presumption that health of aquatic ecosystems can be described along a gradient of stress. Increasing levels of stress are assumed to result in a corresponding reduction in health. This dose-response model suggests that indicators are inadequate if they cannot detect changes over the gradient in stressors.

## A Taxonomy of Indicators

We recognized that if the indicators developed during the ASC project are to be integrated into environmental decision-making, it is imperative to provide a framework for indicator selection and use. Management efforts are generally directed at answering the following basic questions:

- How big is the problem (e.g., where is the resource, and what is its condition)?
- Is it getting better or worse?
- What's causing it?

- What can be done to fix the problem (e.g., how can we improve the health of the impaired system, and what level of health can be maintained)?
- Once action is taken, is management making a difference?
- How can any of the above be communicated to the public?

How do we know which indicator to use to answer any or all of the above questions? There are many existing frameworks for indicator selection (e.g., Dale and Beyeler 2001; SAB 2002; US EPA 2003; Noss 1999; Parker et al. 1999; Kelly and Harwell 1990). However, many of these frameworks are concerned with the use of ecological indicators only for describing system condition, status, and trends (Noss 1999), degree of stressor impact (Parker et al. 1999), or system sustainability (Azar et al. 1996). Therefore, each represents only a narrow range of questions posed to managers. A more general framework is required, one that is broad enough to address the range of decisions that an environmental manager must make (e.g., assessment through restoration), as well as other issues affecting the general public. However, the framework must also be detailed enough to cover the technical concerns that developers of indicators consider essential to their proper use (e.g., the spatial/temporal extent over which the indicator is valid). Finally, because the framework is meant to support decision-making, it must also address the existing social and environmental constraints of the management unit (i.e., what is the predominant land use). We propose a taxonomy based upon three elements: the type of question being asked, the spatial and temporal scale of interest, and the context or land use (*social choices*). The proposed taxonomy is depicted in Figure 8; each element is described in detail below.

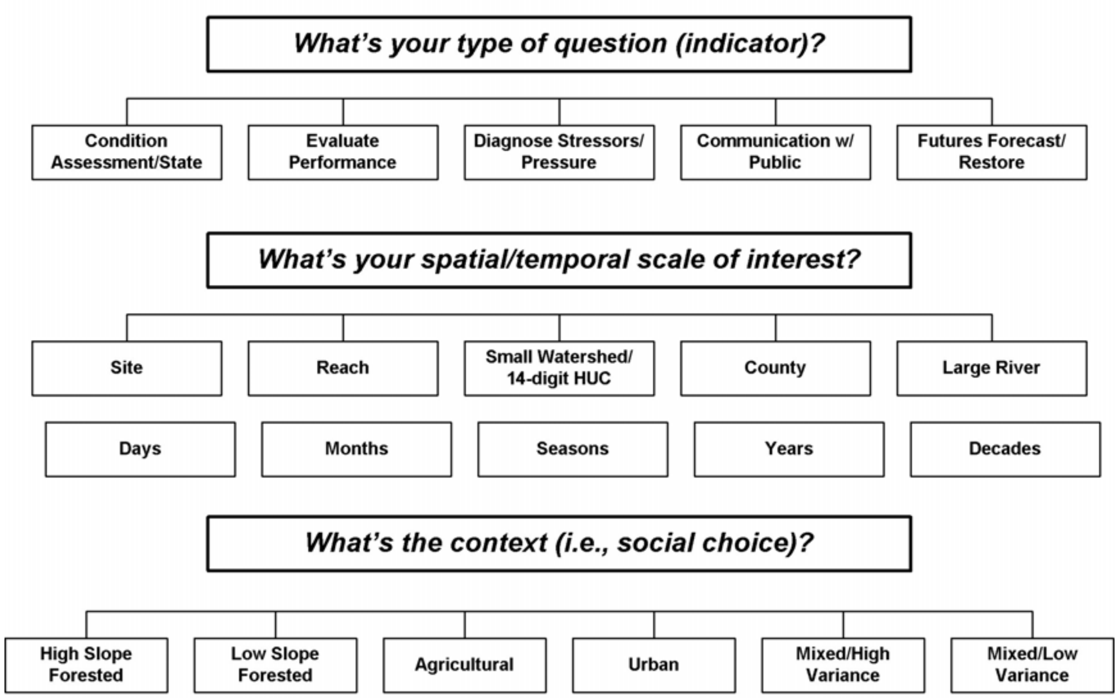



**Figure 8.** Taxonomy of Ecological Indicators

## Type of Question

Given the basic questions listed previously, we propose the following categories in our taxonomy:

1) Condition assessment/state: snapshot of the current state of the ecosystem. With condition indicators, measurements are compared to a threshold or value(s) to indicate whether the system is in good or poor condition. Examples are Indices of Biotic Integrity (IBIs) that have been developed for a number of organisms including fish and birds. Trends in ecological health can be assessed by monitoring condition indicators over time.

2) Performance evaluation: evaluating the effectiveness of management actions. Evaluation indicators must embody two criteria: (1) responsiveness to management actions, and (2) relevance at the management spatial and temporal scale. An example might be increased fish IBI scores because best management practices to reduce stream bank erosion were implemented in a watershed.

3) Stressor diagnosis: identification of factors causing a change in condition and demonstration of clear relationship between cause and condition. Examples include stream bank erosion (stressor) and decreased diversity in fishes (condition) or increased nutrient loading (stressor) and estuarine harmful algae blooms (condition). Identification of factors at a multitude of spatial and temporal scales is desirable. For many management decisions, particularly at larger spatial scales, associations or correlations among condition and stressor indicators, rather than cause-effect relationships, can be sufficient.

4) Communication to the public: encouraging comprehension of condition in a clear and understandable form. These indicators must be both useful and relevant (Jackson et al. 2000). Examples include both the Sneaker Index (i.e., being able to see your sneakers in waist-deep water) and blue crab abundance in Chesapeake Bay (http://www.chesapeakebay.net/indicators.htm).

5) Futures assessment: estimating the probable trend in condition, or assessing the vulnerability of a system to a particular event or activity. These indicators are most often utilized at large spatial and temporal scales. Examples include regional responses to climate change, such as impacts to agricultural and forestry production, fresh water quality and quantity, and biodiversity.

## Spatial and Temporal Scale

Ecological indicators document the state of ecological structure such as biotic diversity or rate of ecological function, or production. Indicators may measure processes directly (such as primary productivity of seagrass beds), or infer structure from pattern (such as utilizing Indices of Biotic Integrity as descriptors of community structure). Ecological patterns emerge, and processes operate, at a range of spatial and temporal scales. This leads us to the inevitable conclusion that the relevant scales must be specified when selecting indicators.

Most resource management decisions occur at local levels (i.e., county, community, land zones). Therefore, we felt a relevant scale for monitoring coastal indicators would be a small watershed (i.e., USGS 14-digit hydrologic category, which are typically tens to hundreds of $km^2$ and encompass several stream or river reaches, with adjacent riparian corridors, associated wetlands and waterbodies, and the contributing drainage basin), or an estuarine segment (composed of deepwater areas, vegetated and unvegetated shallows, tidal wetlands and creeks, and the adjacent terrestrial habitats). Thus, indicators developed during the ASC project can be validated at the scale at which most management decisions are made and implemented. The categories of spatial and temporal scale designated in the taxonomy are a first attempt to recognize applicable scales for coastal indicators, and include scales both larger/longer and smaller/shorter than the small watershed or estuarine segment. It is expected that users of the taxonomy will decide what categories of scale are relevant to them.

## Context of the Question

Identification of context requires us to ask ourselves the following: to whom do we want to be compared? What is a useful comparison? In many cases, the health of the system is compared to how far that system has departed from an ideal condition. In environmental management, the ideal has traditionally been a system devoid of human impact. While this comparison has utility in a general assessment of condition, it has little relevance in environmental decision-making for a variety of reasons. First, there are few, if any, systems or landscapes devoid of human impact. Second, a pristine condition is often unattainable, so a more realistic benchmark needs to be identified. For example, in an urbanized watershed, restoration to a pristine condition is neither possible nor sustainable. What is needed is a relevant benchmark for an urban watershed: what are realistic expectations for the best urban watershed? The problem is one of identifying system condition benchmarks for watersheds having different human use contexts. The conditions one would expect to find in a forested wilderness are vastly different than those expected in an urban watershed.

The research mandate of the ASC project was to identify relevant benchmarks for small watersheds and estuarine segments within the context of various “social choices.” The term “social choice” is used to mean the predominant land use in a watershed because these land use patterns are the “cumulative” result of individual social choices. When land use patterns in small watersheds across the mid-Atlantic are examined, four major categories can be identified: forested, agriculture, urban, and mixed (i.e., no one land use type is predominant) (Figure 9). For each context or social choices category we can ask three questions: (1) How “good” can the environment be, given those social choices; (2) What are the causes of its current condition; and (3) what can be done to improve condition?

The framework indicates there are both multiple ecological states and multiple reference conditions that satisfy various social choice and spatial/temporal scale categories. All the

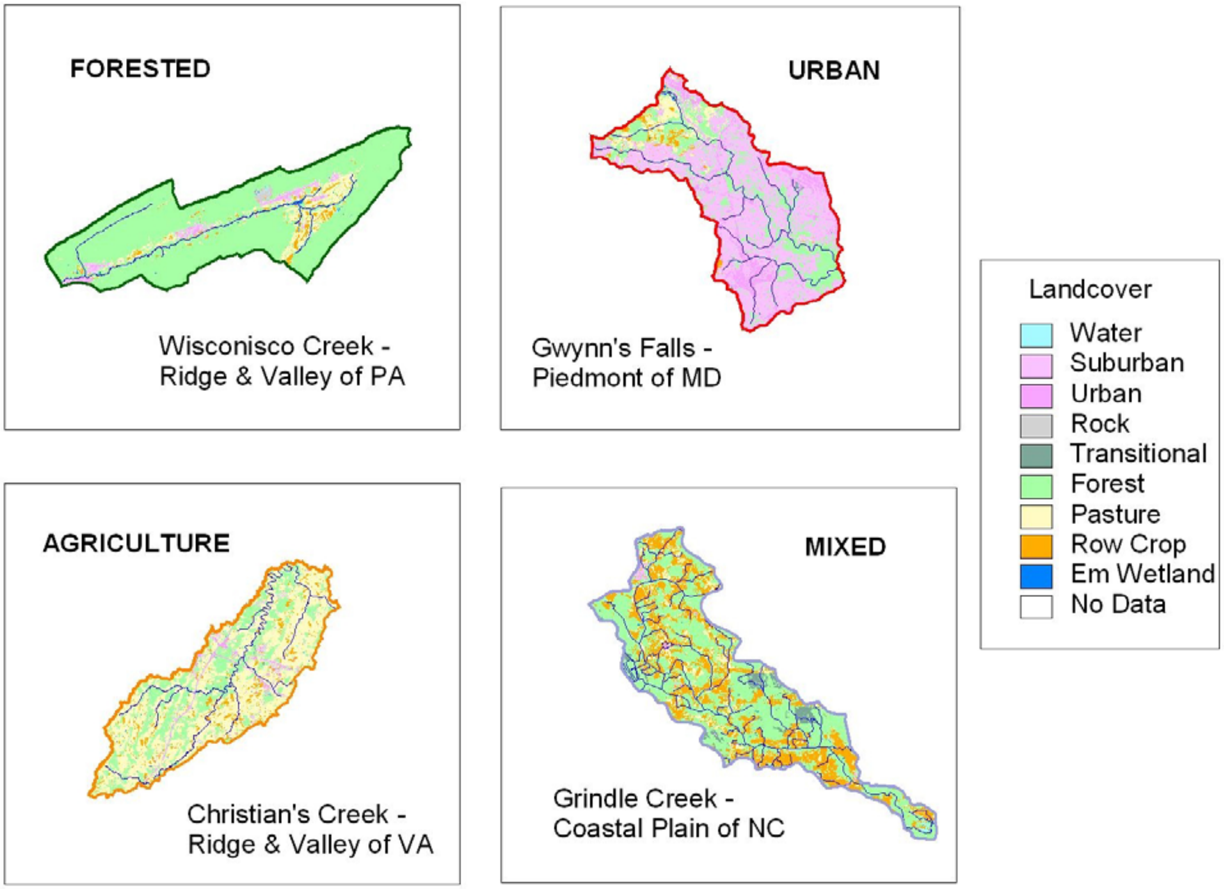



**Figure 9.** Predominant land use categories or "social choices" evaluated in ASC watersheds.

ASC indicators were characterized within this taxonomy (Table 2). Table 2 includes the indicator name, type of question, spatial and temporal scale, context, method of measurement, and a brief description of the indicator for each of the ASC indicators. A Fish Community Index (FCI) developed for the ASC will provide an example of how the framework can be used to select an indicator, as well as determining the usefulness of the indicator.

## Case Study

As part of the ASC project, Bilkovic (2004) developed and tested a Fish Community Index (FCI) as an indicator of ecosystem health in the unique environment of nearshore, shallow water estuarine systems. The resulting FCI is presented as a case study to demonstrate the utility of the taxonomy.

Biotic and habitat variables are often developed together, because they generally represent the response and stressor axes, respectively, of the cause - effect (stressor-condition) curve (Karr and Chu 1999). Bilkovic (2004) linked response to probable stressors by evaluating the FCI in relation to habitat condition metrics that were assessed at multiple spatial scales (subtidal habitat, shoreline condition and watershed land use). Within the study area (Chesapeake Bay), habitat conditions were characterized in estuarine segments that represented the variability in dominant land use types of surrounding watersheds. The FCI fits into the taxonomy as follows.

## Type of Question

The Index of Biological Integrity was first proposed by Karr (1981) as a comprehensive and integrated indicator of biological condition. The effectiveness of multi-metric biotic indices extends to estuarine systems, and use of fish-based IBI has expanded to estuarine ecosystems (Deegan et al. 1997; Jordan and Vaas 2000; Hughes et al. 2002; Bilkovic 2004). The FCI is a straightforward *indicator of condition.*

At each estuarine segment in the study, the following were measured: shoreline land use, shoreline structures (piers, riprap, etc.), subtidal habitat, and macrobenthic and fish communities. The segments were part of an experimental design that was stratified according to dominant watershed land use. Biotic responses were correlated with habitat condition in the nearshore area and along the shoreline (Figure 10). Since correlations between habitat and biota were noted, if clear stressor-condition relationships can be determined and thresholds of response established, then shoreline surveys can become an essential *diagnostic* management tool.

Links among habitat metrics were evidenced between subtidal habitat and shoreline condition, as well as riparian and watershed land use. For example, as shoreline condition improved, the amount of subtidal habitat increased (e.g., woody debris, amount of submerged aquatic vegetation). These relationships provide opportunities for the development of *restoration* measures. If thresholds of shoreline alteration can be established that impact fish commu-

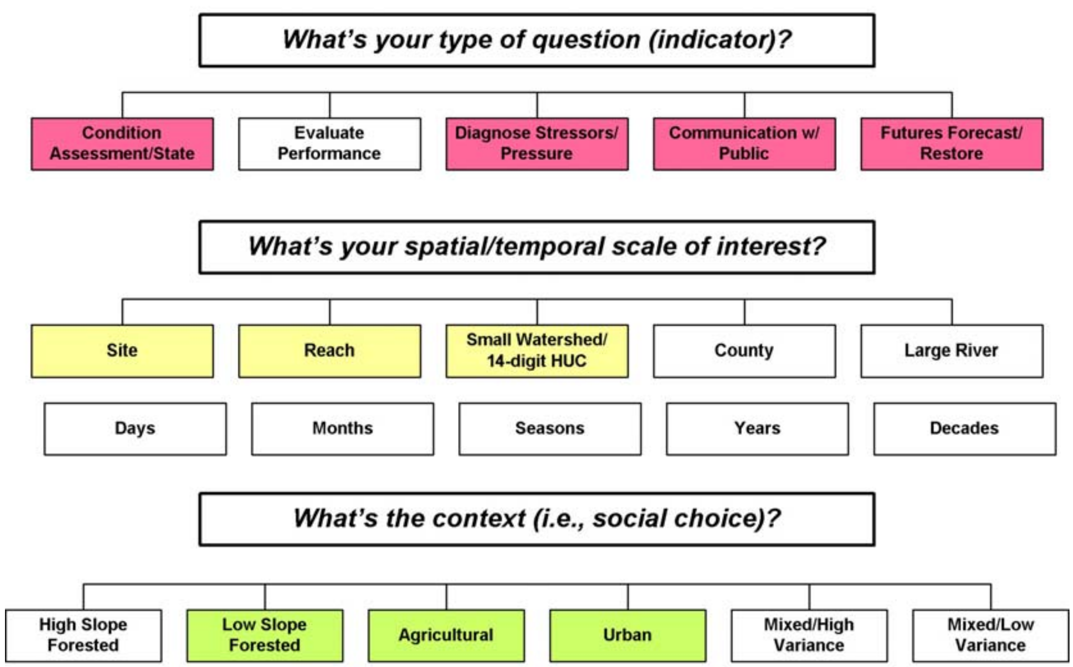



**Figure 10.** Fish Community Index Indicator Chart.

nities, then the shoreline condition assessment (currently underway) will provide a spatially flexible tool to predict and test for expected biotic responses. These indicators could then be utilized to *predict future biotic responses* to a given shoreline condition scenario.

### Spatial and Temporal Scale

The spatial scale of the FCI ranges from site to watershed level depending on the associated habitat feature. For instance, when assessing subtidal habitat, the indicator addresses site-level impacts, and when assessing watershed land use, the FCI addresses watershed level impacts. Temporally, the FCI currently operates over a short-time scale. However, the use of long-term monitoring data allows for the expansion of the time scale to multiple years.

### Context of the Question

In order to develop and test the FCI and habitat measures, 25 watersheds (14-digit HUC) were selected throughout low to moderate salinity regions of the Chesapeake Bay. Each watershed was placed into one of three broad land use categories based on principal land use percentages: forested, agricultural, or developed (see Table 3) and the FCI was mapped onto these broad land use categories. Developed and agricultural watersheds had significantly lower FCI scores than did forested ones.

## Conclusions

Primary elements of the taxonomy are intended to explicitly address three major obstacles to effective identification of impaired areas and their restoration: the type of question being asked, the spatial and temporal scale of interest, and identification of appropriate benchmark reference domains. Indicators are categorized as to which fundamental question they are useful in answering. They are also categorized so managers can select those that are useful for their spatial (stream reach, watershed, ecoregion, state) and temporal (day, season, year) management frame of reference and interest. Finally, the taxonomy identifies major classes of landscape patterns that emerge from individual social choices, in order to provide the necessary context for the questions being asked. The taxonomy indicates there are both multiple ecological states and multiple reference conditions that satisfy various social choice and spatial/temporal scale categories.

We believe the three primary elements of question, scale, and context are the most compelling basis of the taxonomy. We envision the taxonomy providing the following assistance:

- Guiding indicator selection
- Evaluating existing ecosystem indicator programs in order to identify conditions or stressors that are currently lacking indicators
- Designing and developing new ecosystem indicator programs.

**Table 2. Ecological indicators developed for the Mid-Atlantic Region. The applicable uses are given according to Atlantic Slope Consortium categories. A full list of indicators and further information is available at www.asc.psu.edu.**

| Indicator | Uses by ASC Indicator-Selection Framework | Measurement Method | Description |
|---|---|---|---|
| Abundance of Common Reed (*Phragmites australis*) in brackish wetlands of Chesapeake Bay | *Type of Question:* Condition Assessment<br>*Spatial Scale:* Wetland and Estuarine segment<br>*Temporal Scale:* Seasonal to Annual<br>*Context:* Piedmont & Coastal Plain Watersheds | Field Observation | Common reed more abundance in wetlands downstream from developed watershed. |
| Bio-optical Model for Determining Habitat Suitability for Submerged Aquatic Vegetation (SAV) in Estuarine Segments of Chesapeake Bay | *Type of Question:* Condition Assessment<br>*Spatial Scale:* Estuarine Segment<br>*Temporal Scale:* Seasonal to Annual<br>*Context:* Land-Use Decisions in Coastal Zone | Field Observation | Determines the level of suspended solids that allows SAV survival; gives level in relationship to land use. |
| Blue crab (*Callinectes sapidus*) abundance | *Type of Question:* Condition Assessment<br>*Spatial Scale:* Shoreline segment, estuarine segment, and regional level<br>*Temporal Scale:* Seasonal<br>*Context:* All Land Covers | Field observation | Correlates juvenile crabs abundance with shoreline wetlands, forested watersheds, and estuarine segments with average salinity. |
| Fish Community Index (FCI) in Estuaries | *Type of Question:* Condition Assessment. Communication, Forecast<br>*Spatial Scale:* Site to Small Watershed<br>*Temporal Scale:* Days to Years<br>*Context:* Low Slope Forested, Agricultural, Urban | Field Observation | Biotic integrity index for fish communities for application in the near-shore estuarine environment. |
| Index of Marsh Bird Community Integrity (IMBCI) | *Type of Question:* Community Integrity Assessment, Stressor Diagnosis<br>*Spatial Scale:* Marsh to Estuarine segment<br>*Temporal Scale:* Years to Decades<br>*Context:* Emergent Marshes within any land-cover context | Field Observation | Provides sensitive indicators of stress to estuarine wetland systems. |
| Index of Waterbird Community Integrity (IWCI) | *Type of Question:* Community Integrity Assessment, Stressor Diagnosis<br>*Spatial Scale:* Estuarine segment<br>*Temporal Scale:* Years to Decades<br>*Context:* Estuarine segments and associated watersheds with any land cover | Field Observation | Examines the relationship between land cover and estuarine ecosystem integrity. |
| Inverse-Distance Weighted Cropland | *Type of Question:* Condition Assessment, Stressor Diagnosis<br>*Spatial Scale:* Reach to Watershed<br>*Temporal Scale:* Seasons to Decades<br>*Context:* Agricultural, Urban, and Mixed Watersheds | GIS | Gives greater weight to croplands closer to water bodies while including effect of more distant croplands. |

| Indicator | Uses by ASC Indicator-Selection Framework | Measurement Method | Description |
|---|---|---|---|
| Inverse-Distance Weighted Developed Land | *Type of Question:* Condition Assessment, Stressor Diagnosis<br>*Spatial Scale:* Reach and Watershed<br>*Temporal Scale:* Season to Decades<br>*Context:* Urban and Mixed Watersheds | GIS | Emphasize developed land near a feature more than distant developed land. |
| Inverse-Distance Weighted Impervious Cover | *Type of Question:* Condition Assessment, Stressor Diagnosis<br>*Spatial Scale:* Reach and Watershed<br>*Temporal Scale:* Seasons to Decades<br>*Context:* Urban and Mixed Watersheds | GIS | Emphasizes impervious land near a resource more than impervious land further away. |
| Macrobenthic Community Indices in Estuaries (B-IBI, W-Value) | *Type of Question:* Condition Assessment; Communication; Forecast/Restore<br>*Spatial Scale:* Site to small watershed<br>*Temporal Scale:* Days to Years<br>*Context:* Low-Slope Forested, Agricultural, Urban | Field Observation | Measures biotic integrity of the near-shore estuarine environment. |
| Macroinvertebrate Assemblage Composition (Bray-Curtis Dissimilarity) | *Type of Question:* Condition Assessment, Evaluate Performance; Stressor Diagnosis<br>*Spatial Scale:* Reach and Watershed<br>*Temporal Scale:* Seasons to Decades<br>*Context:* Effects most evident in urban and mixed watersheds | Field Observation, GIS | Examines relationships between watershed land cover, in-stream habitat, and water chemistry, and macroinvertebrate assemblages. |
| Nitrate, Total N and Total P Concentrations in Estuarine Segments of Chesapeake Bay | *Type of Question:* Condition Assessment<br>*Spatial Scale:* Estuarine segment<br>*Temporal Scale:* Short-term to Seasonal<br>*Context:* All land cover types | Laboratory Analysis, GIS | Examines relationships between watershed characteristics and nitrate nitrogen, and phosphate. |
| Polychlorinated biphenyls (PCBs) in White Perch | *Type of Question:* Condition Assessment<br>*Spatial Scale:* Watershed Level<br>*Temporal Scale:* Seasonal to Annual<br>*Context:* Urban land cover | Laboratory Analysis | Showed high probability for estuarine segments near commercial land to have White Perch with high PCB levels. |
| Shoreline Condition | *Type of Question:* Condition Assessment, Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years, resample every 5 years to assess change<br>*Context:* Applicable in estuarine tidal areas | GIS, Field Observation | Reports riparian land use, bank characteristics, and structural modifications intended to reduce shoreline erosion. Data can be used with other GIS products for spatial assessment and analysis. |
| Source Land Proportion Weighted by Inverse Riparian Buffer Width | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach to Watershed Level<br>*Temporal Scale:* Months to Decades<br>*Context:* Agricultural, Urban, and Mixed Watersheds | GIS | Estimates effective proportion of land-cover type in the watershed draining to a specific stream point. |

| Indicator | Uses by ASC Indicator-Selection Framework | Measurement Method | Description |
|---|---|---|---|
| Source-Specific Mean Riparian Buffer Width | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach to Watershed<br>*Temporal Scale:* Seasons to Decades<br>*Context:* Agricultural, Urban, and Mixed Watersheds | GIS | Quantifies the potential of riparian buffers to reduce the impact of a specific land cover on aquatic systems. |
| Spot Sampled Average Stream Nitrate Concentration | *Type of Question:* Condition Assessment; Performance Evaluation; Stressor Diagnosis<br>*Spatial Scale:* Reach to Large River<br>*Temporal Scale:* Seasons to Decades<br>*Context:* All Watersheds | Laboratory Analysis | Measures nitrogen pollution in streams and is a potential predictor of aquatic biological responses. |
| Stream-Wetland Riparian (SWR) Index | *Type of Question:* Condition Assessment, Stressor Diagnosis<br>*Spatial Scale:* Reach to Small Watershed<br>*Temporal Scale:* Seasons to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation | Simultaneously assesses condition of all aquatic components and tallies on-site stressors; compiles site and reach data across a watershed. |
| Stream-Wetland Riparian Landscape Index | *Type of Question:* Condition Assessment<br>*Spatial Scale:* Proximal Landscape, Contributing Area, Small Watershed, County<br>*Temporal Scale:* Seasons to Decades<br>*Context:* Agricultural, Urban, Mixed Watersheds | GIS | Examines landscape extent and patterns that may affect aquatic ecosystem condition. |
| Beaver Impoundment Presence | *Type of Question:* Condition Assessment<br>*Spatial Scale:* Reach to Small Watershed<br>*Temporal Scale:* Seasons to Years<br>*Context:* Agricultural, Urban, and Mixed Watersheds | Field Observation and Aerial Photographs | Used to assess significant alterations of habitats by beaver activities. Can significantly alter existing conditions. |

| **Stream Channel Condition** | | | |
|---|---|---|---|
| **Subindicator** | **Uses by ASC Indicator-Selection Framework** | **Measurement Method** | **Description** |
| Channel-Riparian Zone Connection (CRZC) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation | Indicates degree to which free-flowing stream has been channeled or incised. |
| Factors Affecting Riparian Zone (FARZ) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation; Laboratory Analysis | Indicates degree to which pollution and other factors affect the riparian zone of a reach. |

| Stream Channel Condition | | | |
|---|---|---|---|
| **Subindicator** | **Uses by ASC Indicator-Selection Framework** | **Measurement Method** | **Description** |
| Habitat Quality of Riparian Zone (HQRZ) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation | Indicates condition of riparian zone habitat. Used with other riparian zone indicators for riparian zone function. |
| Instream Woody Structure (IWS) | *Type of Question:* Condition Assessment and Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watershed | Field Observation | Indicates the amount and quality (decay) of large downed wood in stream channels. |
| Near Stream Cover (NSC) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation | Composite of total biomass, based on the proportion of cover classes within the near-stream buffer zone. |
| Pollution Affecting Stream (PAS) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation; Laboratory Analysis | Determines degree to which a stream reach is polluted by the types and concentrations of pollutants and their distances upstream. |
| Riparian Zone Cover (RZC) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation | Multivariate descriptor that incorporates variations in biomass and distance from stream edge. |
| Sediment Regime (SR) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation | Indicates extent to which excess sediment is carried by a stream. |
| Stream Bank Stability (SBS) | *Type of Question:* Condition Assessment; Stressor Diagnosis<br>*Spatial Scale:* Reach<br>*Temporal Scale:* Days to Years<br>*Context:* Agricultural, Urban, Mixed Watersheds | Field Observation | Indicates degree of excessive bank erosion in higher order streams. |

[This Page Intentionally Left Blank]

# MESSAGE 2

## The Estuarine Segment Approach

Numerous studies have demonstrated that human activities on land can have negative impacts on estuarine ecosystems (Nixon 1995; National Research Council 2000; Bosch et al. 2001, 2003). Few studies, however, have quantified the direct linkages between particular land-use patterns and estuarine responses. One reason that it has been difficult to quantify linkages between specific land use patterns and estuarine responses is that most monitoring studies have focused on large open water systems (e.g., the mainstream of Chesapeake Bay or the large rivers that flow into it). Large-scale monitoring studies of this type are useful in tracking temporal changes for indicators of estuarine health. The Chesapeake Bay Program, for example, has developed a diverse array of indicators for monitoring the Bay (http://www.chesapeakebay.net/indicat.htm). The Chesapeake Bay Foundation uses a wide range of indicators to produce an annual scorecard of the health of Chesapeake Bay (http://www.cbf.org/site/PageServer?pagename =sotb_2004_index).

Monitoring programs and large-scale models such as those developed by the Chesapeake Bay Program (http://www.chesapeakebay.net/pubs/iannewsletter11.pdf) have been used to develop management plans, but they have limited use in guiding small scale land-use decisions because they do not have the sensitivity to quantify the relationships between specific land-use patterns and estuarine indicators at a scale that is appropriate for making management decisions.

The objective of this part of the ASC project was to identify linkages between patterns of land-use and environmental indicators in shallow estuarine habitats. To accomplish this objective we used existing data and also sampled estuarine segments of Chesapeake Bay that were linked to a watershed that was large enough to support at least one perennial stream but small enough for field teams to sample the several habitats within the subestuary in a reasonable period of time (i.e., one or two days).

## Estuarine Segment Selection and Characterization

Smithsonian Environmental Research Center (SERC) and Virginia Institute of Marine Science (VIMS) scientists selected estuarine segments based primarily on land-use patterns. But they differed in the selection criteria.

*SERC Estuarine Segments* - SERC scientists selected estuarine segments independent of watershed size, except for the criteria (described in more detail below) that the watershed was large enough to support at least one perennial stream that flowed into the estuarine portion of the segment. They initially screened more than 75 potential estuarine segments based on salinity regime. An initial goal was to minimize the complicating effects of salinity on estuarine biota by selecting estuarine segments that were in the mesohaline portion (i.e., intermediate salinity - between freshwater and seawater) of Chesapeake Bay. To be included as an estuarine segment, the watershed portion had to: (1) be dominated by one of the land-use types described in Table 3; (2) discharge directly into Chesapeake Bay or into the mesohaline portion of one of the large river systems; and (3) be large enough to have at least one perennial stream that flowed into the subestuary. In addition to shallow subtidal habitats, each estuarine segment included a small (< 2 ha or 5 ac), medium (2–7 ha or 5-18 ac) and large (> 7 ha or 18 ac) brackish tidal wetland. The 32 estuarine segments that were chosen for study were distributed along a north-south axis of Chesapeake Bay (Figure 11) and, with the exceptions of portions of the Back River, Bird River, Gwynns Falls, Jones Falls, Bird River watersheds, were within the Coastal Plain province. Topography of the Coastal Plain varies from rolling hills on the western shore to flat terrain on the central and southern portions on the eastern shore. Land-use patterns on the watersheds of each segment were used as surrogates for human disturbance levels.

**Table 3. Land-use categories used to characterize the watershed portions of estuarine segments.**

| Land Use Category | Criteria |
|---|---|
| Forested | Greater than 65% total forest covers (forest, mixed, forest wetland) and <10% urban |
| Agricultural | Greater than 50% total agricultural covers (pasture, crop) |
| Developed | Greater than 50% total urban covers (low and high residential and industrial areas) |
| Mixed Developed | 20-50% total urban covers |
| Mixed Agricultural | 20-50% total agricultural covers |

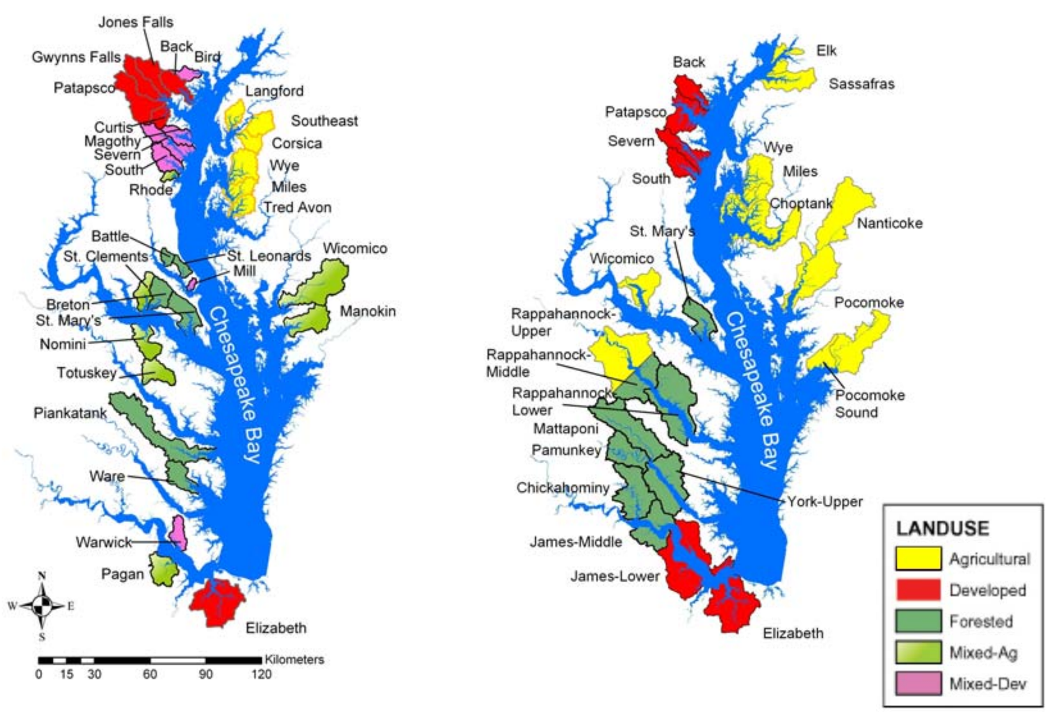



**Figure 11.** SERC (left) and VIMS (right) estuarine segments.

*VIMS Estuarine Segments* – VIMS scientists selected 23 estuarine segments in the oligo-to-mesohaline (i.e., low to intermediate salinity) portions of Chesapeake Bay based on: watershed land use classification, salinity regime, and accessibility. United States Geological Survey (USGS) designated 14-digit hydrologic unit codes (HUC) were used to select watershed sampling units and watershed land-use classification was based on principal land use percentages derived from the National Land Cover Database (NLCD, 30 m raster coverage). Because of the sizes of 14-digit HUCs, the number of available watersheds in each land use category was limited, the VIMS estuarine segments, therefore, only included three land-use categories: forested, agricultural (including the mixed-agriculture category in the SERC classification), and developed (including the mixed-developed category in the SERC classification) (See Table 3). Similar to SERC estuarine segments, the VIMS watersheds were distributed along a north-south axis of Chesapeake Bay (Figure 11) and, with the exceptions of portions of the Back, Patapsco, Severn, and Elk river watersheds, were within the Coastal Plain province.

## General Description of Sampling Methods

Data collection in each subestuary was tailored to each project (projects described in more detail below) and the reader is referred to the published articles for details.

In general, water quality parameters (e.g., temperature, salinity, dissolved oxygen, pH) were measured in the field at several sites in each subestuary. Field collected water samples were returned to the laboratory for further analyses (e.g., total suspended solids, nitrate-nitrogen ($NO_3$-N), total nitrogen (Total N) and total phosphorus (Total P)).

Subtidal habitats were sampled in several projects. Benthic samples were collected using coring devices (e.g., Ekman Grab) and habitat assessments (e.g., amount of woody debris, presence of submersed aquatic vegetation, characteristics of adjacent shoreline) were conducted. Fish and crabs were sampled using Fyke nets and nearshore seining and samples of White Perch (*Morone americana*) were retained and analyzed for PCB concentration.

Foraging waterbirds and birds that nested in brackish wetlands were sampled in the field as was wetland vegetation. Samples of Common Reed (*Phragmites australis*), an invasive wetland plant species, were collected in the field and analyzed in the laboratory for nitrogen content.

## Results

Table 4 provides a summary of the estuarine indicators that could be related to land-use patterns. In some instances, the indicators responded to land-use patterns only at the watershed scale. In other instances, the indicators responded to land-use patterns at the scale of the entire watershed and at the local scale, especially conditions close to the subestuary. One indicator (wetland breeding birds) only responded to local land-use patterns. In two instances (PCBs in White Perch and abundance of Common Reed), we were able to determine that indicators responded to conditions at the watershed scale, but more strongly to the relationship between land-use conditions and proximity to the estuary.

In the next section we report results for selected indicators. Detailed information on these indicators can be found in journal publications on these indicators. The publications are cited in the Reference chapter.

**Table 4. Estuarine indicators identified for estuarine segments of Chesapeake Bay.**

| *Indicator* | *Watershed* | *Local Land Use* |
|---|---|---|
| Macrobenthos Indices | X | X |
| Fish Community Index | X | X |
| Abundance and leaf nitrogen content of Common Reed (*Phragmites australis*) | X | X |
| Blue crab and bivalve abundance | X | X |
| PCBs in White Perch (*Morone Americana*) | X | |
| Waterbird Community Integrity | X | X |
| Marsh Bird Community Integrity | | X |
| SAV Abundance | X | |

# BLUE CRAB AND BIVALVE ABUNDANCE

## Background

The goal of this project was to explore relationships between regional (e.g., salinity), watershed (e.g., land use), and local (e.g., land use, water quality, habitat) factors on the abundance of blue crabs and species of *Macoma*, common clams that are blue crab prey.

A number of socioeconomic and ecological attributes make blue crabs (*Callinectes sapidus*) potentially ideal indicators of environmental conditions in estuarine ecosystems. Blue crabs are distributed throughout Chesapeake Bay and other estuaries of the East and Gulf coasts of North America and disperse across a wide range of salinities following settlement in the relatively high salinity zone. Blue crabs are also highly prized by humans for food and are the most important commercial fishery in the Mid-Atlantic region. As the dominant benthic predator and as prey for some larger predators, they also play a critical role in energy transfer in estuaries. Blue crabs feed intensively on bottom organisms living in the sediment, particularly clams, suggesting that the spatial distribution of blue crabs might be tied to natural and anthropogenic factors that affect the distribution and abundance of bivalve prey. In addition, blue crabs may be sensitive to anthropogenic shoreline modifications because natural nearshore habitats such as woody debris and marsh creeks are important for both juveniles and molting crabs as refugia from predation. Finally, blue crabs are sensitive to hypoxia (low dissolved oxygen concentrations), thus, their distribution may be directly influenced by cultural eutrophication commonly associated with developed and agricultural land use in watersheds.

## Findings

Classification And Regression Tree (CART) analysis, a type of statistical analysis, indicated that 46% of the variance in blue crab abundance was explained by salinity (9%), watershed land use (17%) and shoreline marsh habitat (19%) (Figure 12). Crab abundance was greatest at intermediate and higher salinities (>16 ppt), but in lower salinities crabs were most abundant along wetland shorelines in forested and mixed land use watersheds. Juvenile crabs <85 mm (~3 in) were more strongly associated with wetland shorelines, particularly in estuarine segments with forested and mixed land use watersheds.

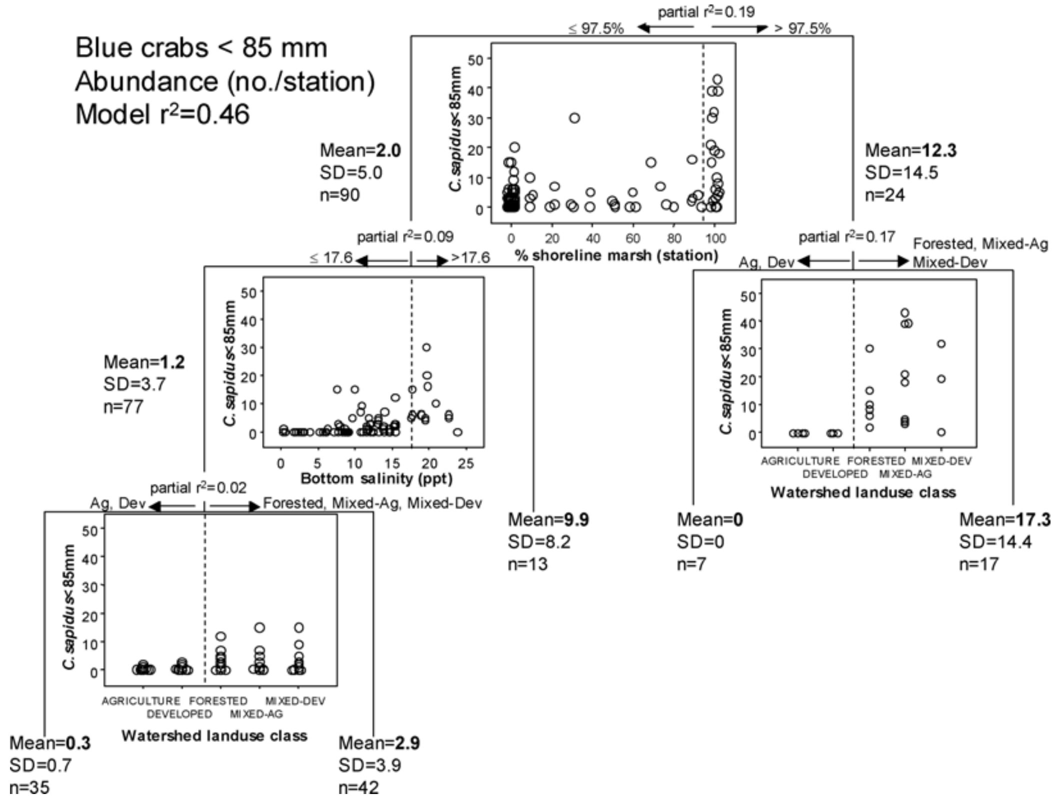



**Figure 12.** Results from CART analysis of juvenile blue crabs < 85 mm (no. station$^{-1}$). Scatter plots illustrate the response of juvenile blue crab abundance at each level of the tree. The vertical line in each plot identifies the value of the predictor (x) that best explained variation in juvenile blue crabs. Values of predictors are shown to the left and right of each split above each scatter plot. Variance explained ($r^2$) for each predictor is shown above each split. Means, standard deviations (SD), and number of stations (n) summarize properties of the data to the left and right of splits in each scatter plot.

Clams (*Macoma*) were similarly associated with wetland shorelines, but mainly in muddy bottoms at moderate-to-high salinities; however, the best CART model only explained 25% of variance in bivalve abundance. These results were consistent with predictions that shoreline wetlands and watershed land use may have important effects on these taxa along the estuarine salinity gradient, and are consistent with hypotheses based on previous descriptive and experimental research linking blue crabs and deposit-feeding clams to habitats rich in particles of plant leaf pods, broken stems, and other organic matter worked in from the watershed. These findings are described in detail in King et al. (2005).

## Implications

Within habitat characteristics (salinity, shoreline condition, substrate type, abundance of wetlands) are important factors influencing the abundance of blue crabs and clams. Land-use at the scale of the entire watershed is also important and the lowest abundances of both organisms occur in estuarine segments that are downstream of watersheds dominated by development and agriculture. Land-use, therefore, can be used as an indicator of estuarine conditions but the target organisms (blue crabs and clams) could also be monitored to track conditions within subestuarine habitats. The application of blue crabs and clams within the framework of ASC indicators can be found in Appendix A.

## ABUNDANCE AND LEAF NITROGEN CONTENT OF COMMON REED (*Phragmites australis*)

### Background

We hypothesized that the distribution and abundance of *Phragmites* may be linked to land use through pathways at both local scales (e.g., disturbance, nitrogen enrichment, and salinity reductions caused by adjacent land use) and watershed scales (e.g., enhanced nitrogen availability in surface water linked to agricultural and developed land uses in adjacent watersheds). To test this hypothesis, we examined the relationship between *Phragmites* distribution and abundance data collected from 90 tidal wetlands located within 30 estuarine segments spanning over 250 km of Chesapeake Bay to digital land-cover data summarized at both local and watershed scales. We also explored the potential linkage between land use and increased nitrogen availability at the watershed scale and *Phragmites* leaf-tissue nitrogen, an indicator of enrichment (Bertness et al. 2002).

*Phragmites australis* is an invasive species in North America, particularly in the Mid-Atlantic region (Chambers et al. 1999, Sillman and Bertness 2003) and an introduced species appears to be responsible for the recent spread (Saltonstall 2002). *Phragmites* impacts on wetlands ecosystems are considered to be negative so control or eradication management practices are often used (e.g., Philipp and Field 2005 and references therein). What factors are responsible for *Phragmites* invasion and spread? Development of nearshore areas and within-wetland disturbances and increased nitrogen are associated with an increased abundance, cover and spread of *Phragmites* in New England tidal wetlands (Minchinton and Bertness 2003, Sillman and Bertness 2003). Tidal wetlands of Chesapeake Bay have also seen marked increases in the occurrence and abundance of *Phragmites* (reviewed by Rice et al. 2000). However, less is known about the process of invasion and spread in Chesapeake Bay compared to the more comprehensively studied New England salt marshes.

The Chesapeake Bay watershed is rapidly urbanizing and is the fastest growing and culturally enriched coastal region in North America (e.g., Culliton et al. 1990, Boesch and Greer 2003). Cultural eutrophication has been related to point and non-point source nitrogen inputs from agricultural and urban (developed) lands (e.g., Jordan et al. 1997a, Boesch and Greer 2003, Jordan et al. 2003a). Thus, given the mechanistic relationships reported elsewhere, the increase in anthropogenic nitrogen and shoreline disturbances caused by agricultural and developed land uses may be at least partially responsible for the expansion of

*Phragmites* in Chesapeake Bay. However, no previous study has empirically examined such relationships in this estuarine ecosystem and no study in any region has examined linkages between land use and *Phragmites* among many wetlands spanning a geographical extent as great as that of Chesapeake Bay.

## Findings

For wetlands that had *Phragmites*, abundance was best explained by the following factors, in order of importance, by percent inverse distance weighted (% IDW) development (see Sidebar), % IDW forested land, and northing or longitude (Figure 13). If % IDW development was >15%, *Phragmites* abundance increased dramatically (Figure 13 – top diagram). When % IDW development was ≤15%, wetlands in estuarine segments with ≤34% IDW forested land tended to have higher *Phragmites* abundance (Figure 13 – bottom left diagram). Wetlands in the middle and northern regions of Chesapeake Bay also had more *Phragmites* (Figure 13 – bottom right diagram).

Examination of data for all 90 sites showed that *Phragmites* was almost always present when the watershed associated with the subestuary had <39% forested land cover. When watershed forest cover was >39%, abundance was higher in estuarine segments that had higher percentages of development near the subestuary.

Nitrogen concentration in leaves was also highest when % IDW developed land exceeded 14% (Figure 14). In 2002, a drought year with lower runoff into the estuaries, %N in estuarine segments with agricultural watersheds (Figure 14 - bold bubbles in left diagram) were not consistently higher compared to forested systems and were much lower compared to developed watersheds. In 2003, a wet year with higher runoff from agricultural fields, we found the same relationship between % IDW and %N but leaf nitrogen concentration tended to be higher at sites with agricultural watersheds (i.e., higher values for bold bubbles in right diagram in Figure 14 compared to same in left diagram). Additional information will be available in King et al. (in prep.).

### Inverse Distance Weighting

Activities on land closest to water bodies generally have the greatest effects on the quality or condition of a water body and its biological organisms. If the runoff from two parking lots is identical, and one of these parking lots is 1 yd from the receiving water body, while the other is 1,000 yds from the same water body, the pollutants from the parking lot only 1 yd from the water body would affect the water body more than pollutants from the parking lot 1,000 yds away. If distance from a stream, wetland, or estuary was used to weight the importance of the land use, the parking lot 1,000 yds away would be weighted higher than the lot 1 yd away, but this is the opposite of which parking lot's pollutant runoff is more important. Therefore, the inverse of this distance is used for weighting, so the land use closest to the water body is weighted as being more important to the quality or condition of the aquatic ecosystem.

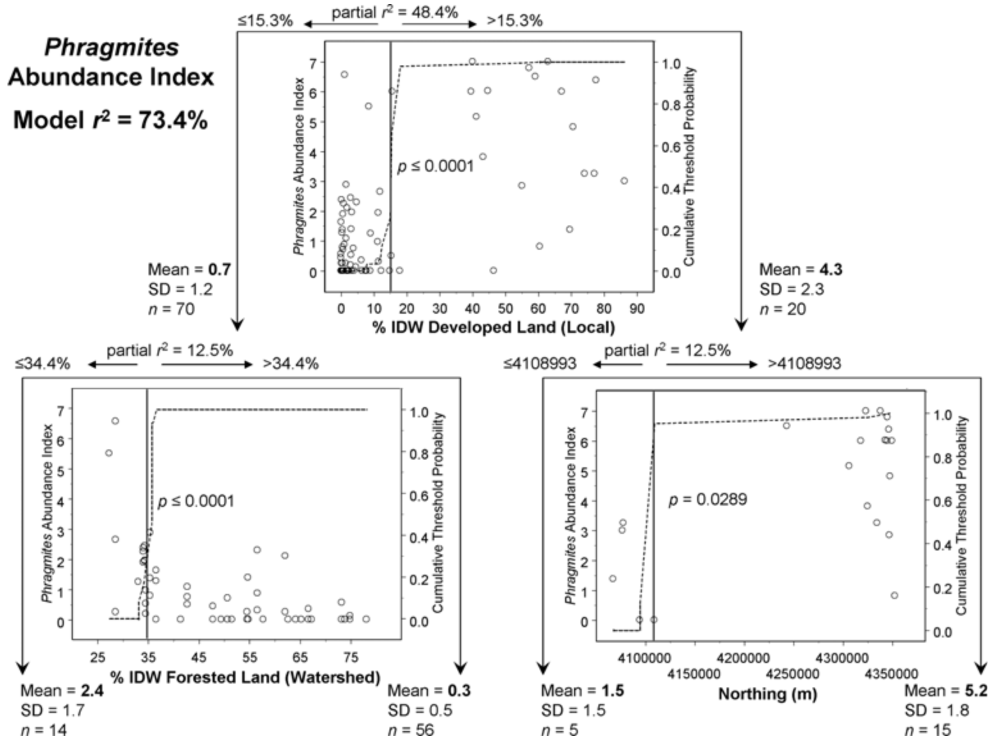



**Figure 13.** Results from CART analysis of *Phragmites* abundance. Scatter plots illustrate the abundance of *Phragmites* at each level of the tree. The vertical line in each plot identifies the value of the predictor (x) that best explained variation in *Phragmites* abundance. Values of predictors are shown to the left and right of each split above each scatter plot. Variance explained ($r^2$) for each predictor is shown above each split. Means, standard deviations (SD), and number of stations (n) summarize properties of the data to the left and right of splits in each scatter plot.

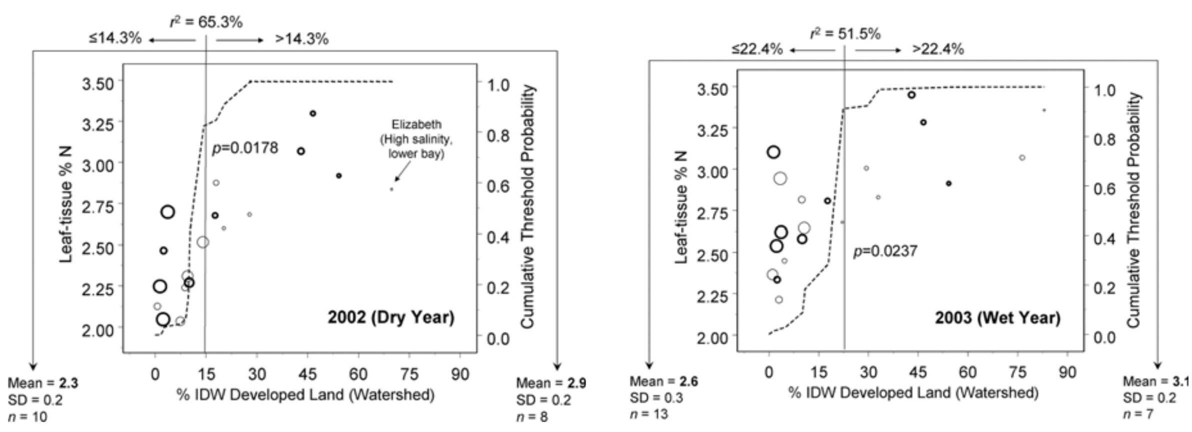



**Figure 14.** Scatter plots of the relationship between nitrogen in *Phragmites* leaves and the inverse-distance weighted (IDW) percentage of the watershed area that is developed. The left diagram is for 2002 the driest year on record in the region and the right diagram is for 2003, the wettest year in the region. Bold circles are estuarine segments sampled in both years. The size of the circles indicates the relative amount of IDW agircultural land in the watershed. The vertical line in each plot identifies the value of the predictor (x) that best explained N in leaves of *Phragmites*. Values of predictors are shown to the left and right of each split above each scatter plot. Variance explained ($r^2$) for each predictor is shown above each split. Means, standard deviations (SD), and number of stations (n) summarize properties of the data to the left and right of splits in each scatter plot.

## Implications

Land-use, especially the amount of development at the watershed and local scale, are important factors contributing to the abundance of *Phragmites* and the nitrogen content of leaves. Land-use, therefore, can be used as an indicator of estuarine conditions but the target species (*Phragmites*) could also be monitored to track conditions within subestuarine habitats.

# MACROBENTHOS INDICES

## Background

Our objective was to examine the influence of shoreline alteration and watershed land use on nearshore macrobenthic (organisms, visible without magnification, living on or in the sediment) communities using established indices for related estuarine environments.

Human modification within watersheds arguably has the strongest impact on aquatic condition at the land-water interface. Biotic multimetric indices have been used extensively as measures of condition in a variety of systems, most recently estuaries. The characterization of ecosystem condition using integrative indices was initially developed for, and applied in, freshwater systems. Multimetric biological indices such as benthic indices of integrity, however, have shown promise as methods for assessing condition in estuaries due to their predictable and integrative response to stressors.

Benthic macroinvertebrates have a long history as indicator organisms due to the ease of collection, their immediate and measurable response to impairment, and the fact that they are mostly sedentary, consequently reflecting local conditions. Macrobenthic community indices have been successfully applied in estuarine systems and may be useful as condition or diagnostic indicators in the critical nearshore ecosystem.

Shallow-water tidal habitats provide essential nursery and spawning areas, protection from predators, and foraging opportunities for numerous fish, shellfish and crustacean species. This critical resource area is under intense and increasing pressure from a variety of uses and users and the impact of shoreline and watershed land use on nearshore biotic communities is a fundamental ecosystem management question. Evaluation of the ability of macrobenthic community indices to characterize the influence of shoreline alteration and watershed land use in nearshore estuarine environments could lead to the development of viable management tools.

## Findings

Biotic responses were correlated with habitat condition along the shoreline and in the watershed, with the highest scores (i.e., best condition) associated with forested watersheds. Nonparametric changepoint (statistical) analyses indicated that ecological thresholds existed in response to developed land use at the site and watershed scale. There was a significant reduction in Benthic Biotic Index scores at the site and watershed levels when the amount of developed shoreline exceeded 10% and developed watershed exceeded 12%, respectively (Figure 15, left diagram).

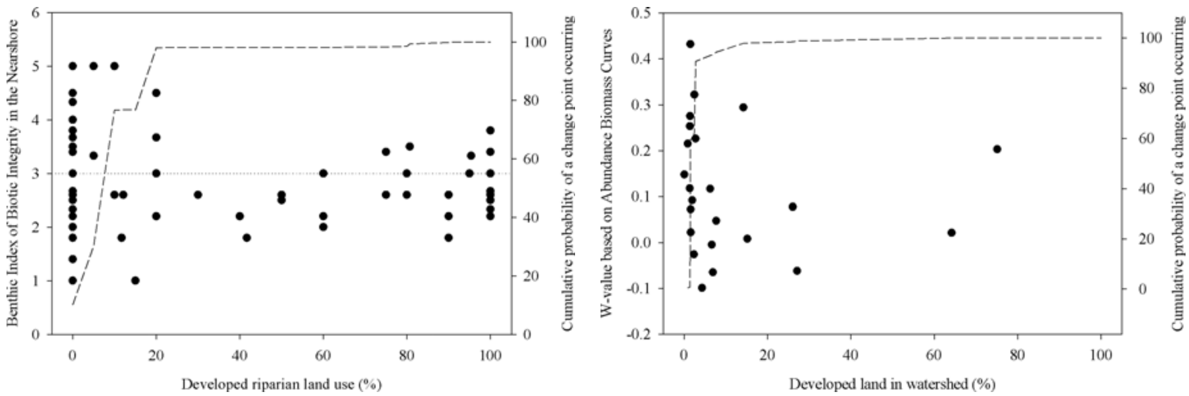



**Figure 15.** Results of non-parametric changepoint analyses for a) percent development of the shoreline (150 m of water's edge) of study sites and the benthic index of biotic integrity in the nearshore (B-$IBI_N$) (left diagram); and b) percent development within the watershed and W-value (right diagram). The W-value, a statistical measure of abundance biomass curves, interprets high values as indicative of a less-disturbed or reference system. The B-$IBI_N$ is scaled from one to five, with scores less than three indicative of stressed conditions (dashed horizontal line). The cumulative probability curve represents the cumulative probability that a changepoint occurred at various levels of development. Significant macrobenthic community responses (p = 0.05) were measured with the B-$IBI_N$ and W-value when developed lands were 10 and 12%, respectively. There was a 95% cumulative probability of an ecological threshold occurring at 20 and 14% developed lands for the $BIBI_N$ and W-value, respectively.

The addition of shoreline land use information enhanced the discriminatory ability of the indices in a given landscape. In particular, the site scale Benthic Biotic Index shows promise for elucidating gradients of condition within landscapes with varying degrees of shoreline alterations. Since shoreline forests and wetlands may diminish the effects of urban land use in localized areas, the inclusion of detailed site-specific information may be indispensable for defining condition. Additional details can be found in Bilkovic et al. (in review).

## Implications

Nearshore macrobenthic communities responded to land use conditions at local (site) and watershed scales. Index scores decreased with anthropogenic alterations to the landscape (e.g., developed watersheds), and thresholds were identified for shoreline and watershed developed land use (10% to 12%) beyond which a negative response in macrobenthic communities occurred. Watershed and shoreline land use may be effective integrative measures of stress that are able to infer the state of degradation in a system. The integration of shoreline and watershed land use measures with macrobenthos indices can lead to practical management tools with particular application on small watershed scales.

Ecosystem approaches to condition assessment should incorporate a variety of indicators that measure different scales or types of stressors. The measure of prey community (e.g. macrobenthic) responses to habitat condition adds a layer of information about the nearshore system that will aid managers in prioritizing and targeting sites or watersheds for restoration or protection.

# FISH COMMUNITY INDEX (FCI)

## Background

The goal of this subproject was to develop and test fish community metrics in the nearshore Chesapeake Bay and evaluate relationships among fish communities and habitat condition assessed at multiple spatial scales (subtidal habitat, shoreline condition and watershed land use).

Fish community characteristics have been used since the early 1900s to measure relative ecosystem health. Within the last 20 years, advances have stemmed from the development of integrative measures of ecological condition, such as the Index of Biotic Integrity (IBI), which relates fish communities to abiotic and biotic conditions of the ecosystem. Fish community IBIs were first developed for use in freshwater, Midwestern streams, and subsequently modified for application in Great Lakes bays, reservoirs, streams and large rivers throughout the United States and other countries. The common thread that connects the various IBIs is a multimetric approach, which describes biotic community structure and function and relates it to the ecosystem or habitat. The use of fish community-level response as an indicator affords many advantages: (1) high public interest; (2) multi-trophic response that integrates aquatic condition; (3) assessment of both habitat and biotic condition as well as cumulative effects; (4) assessment of large-scale regional effects due to their mobility; (5) ease of identification on-site; and (6) availability of long-term monitoring data.

Estuarine systems are arguably some of the most complex aquatic systems. Their natural variability compounds the problems of detecting anthropogenic impacts. Until now, use of fish community IBIs in estuarine systems has been limited, with varying degrees of success. With growing recognition that effective management of estuarine systems can only occur at ecosystem levels, the need for further development of these metrics is widely accepted.

Within estuaries, nearshore habitat provides essential nursery and spawning areas, protection from predators, and foraging opportunities for numerous fish species. This critical resource area is under intense and increasing pressure from a variety of uses and users and generally exists without an operative comprehensive management plan. For instance, the cumulative impact of shoreline armoring has been demonstrated to drastically reduce available shallow-water habitat structure and associated fish communities. Evaluation of nearshore habitat and shoreline condition in conjunction with descriptions of biological communities may establish links between landscape and the biota lending guidance to managers. This association may provide the basis for development of a diagnostic indicator of estuarine condition.

# Findings

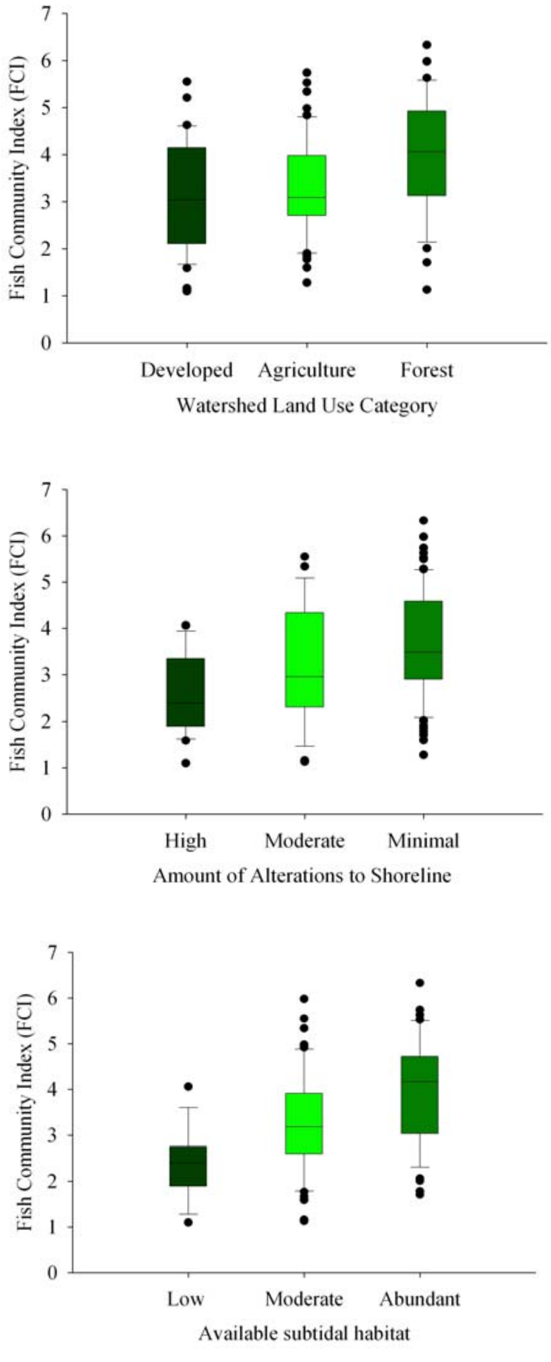



**Figure 16.** Fish Community Responses (FCI) in relation to habitat condition states assessed at various spatial scales: watershed land use (top), shoreline condition (middle), and subtidal habitat (bottom).

Biotic responses were correlated with habitat condition in the nearshore, shoreline and watershed. Fish Community Index (FCI) scores were significantly lower in developed and agriculture watersheds than in watersheds dominated by forests (Figure 16, top), and there were also negative impacts associated with local land use patterns and nearshore habitat conditions. The lowest average FCI scores were found in areas with highly altered shoreline conditions and minimal subtidal habitat (Figure 16, middle and bottom). This is intuitive, since the direct biotic response may be due to changes in nearshore habitat, with indirect impacts due to watershed land use. These results are supported by recent studies describing the relationship between shoreline alteration and nearshore/littoral habitat condition.

Links among habitat conditions were substantiated in the relationships between subtidal habitat and shoreline condition, as well as shoreline and adjacent watershed land use. Shoreline condition and subtidal habitat measures were significantly correlated indicating a negative association between shoreline alterations and

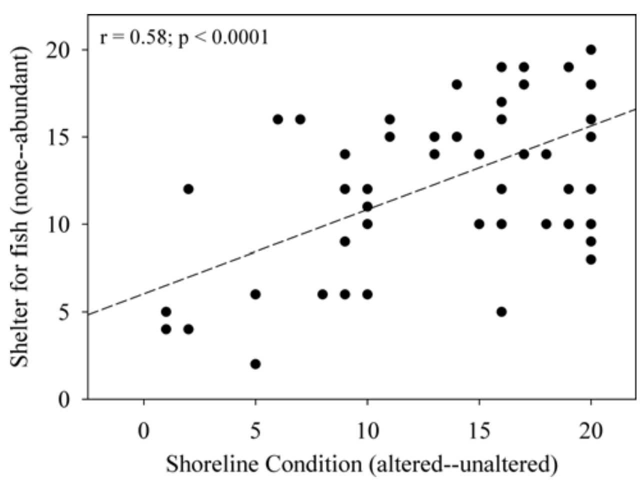



**Figure 17.** Comparison between available subtidal habitat (scaled from none to abundant habitat) and shoreline condition (scaled from highly altered to unaltered states) per site.

available subtidal structural habitat (Figure 17). Dominant watershed land use was reflected in shoreline land use conditions for all three of the categories (developed, agricultural, forested) (Figure 18). More detailed information can be found in Bilkovic et al. (2005).

## Implications

Habitat conditions at multiple spatial scales (subtidal habitat, shoreline condition and watershed land use) are correlated with the Fish Community Index scores. These measures may be used as indicators of estuarine condition in addition to the biological functional response as reflected in the FCI. For instance, since correlations between habitat and biota were noted, if mechanistic processes can be determined and thresholds of response established, then shoreline condition surveys become an essential diagnostic management tool.

# MARSH BIRD COMMUNITY INTEGRITY

## Background

Our objective was to construct a community index based on marsh birds designed to estimate the integrity of the marsh bird community as well as to provide insight into the integrity of the entire marsh ecosystem. We used basic ecological principles to develop the index of marsh bird community integrity (IMBCI) and then subsequently tested the sensitivity of marsh bird community integrity to independently quantified land-use disturbances.

Birds are considered ideal for use in a community index because they are easy to survey and their life histories are relatively well

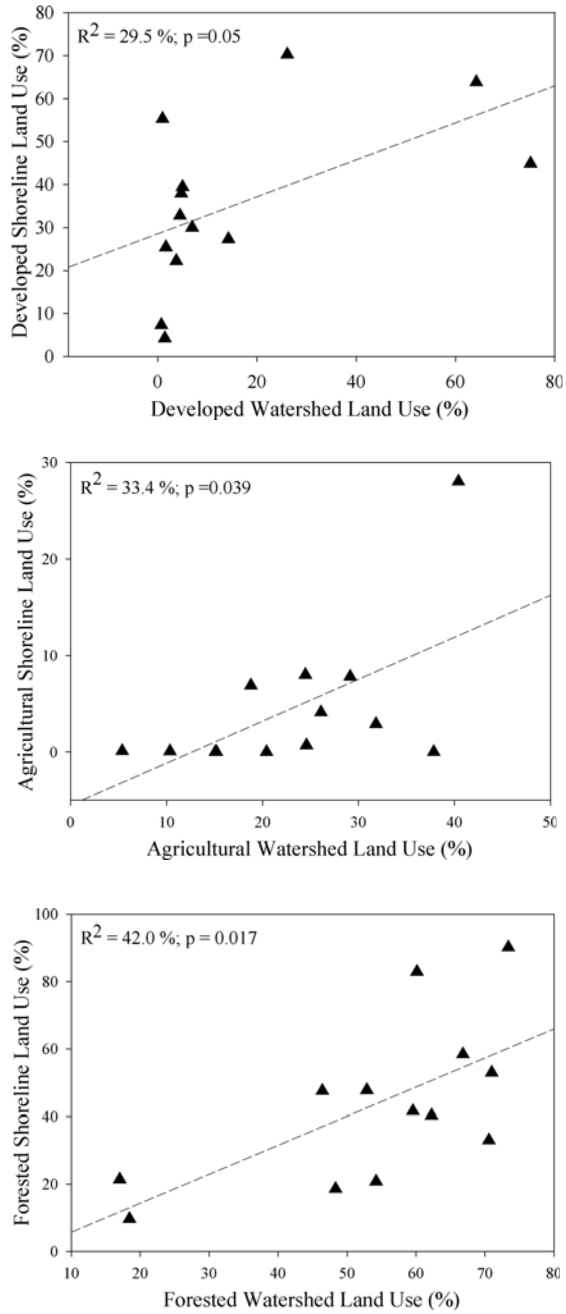



**Figure 18.** Comparison between percentages of each land use type in a watershed, and the corresponding riparian land use category: Developed (top), Agricultural (middle), or Forested (bottom). VIMS-Center for Coastal Resources Management (CCRM) Shoreline Condition Survey and NLCD land use data were extracted from a subset of thirteen watersheds in the Chesapeake Bay: Back, Battle, Breton Bay, Chickahominy, Elizabeth, Lower Rappahannock, Lower James, Pagan, Piankatank, Severn, St. Clements, St. Mary's, and Totuskey.

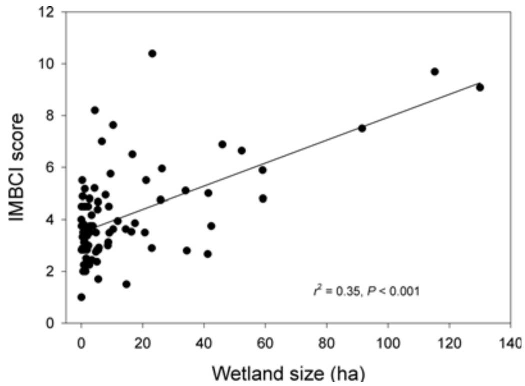



**Figure 19.** Results of a regression analysis showing the relationship between the index of marsh bird community integrity (IMBCI) and wetland size for 91 wetlands in the Chesapeake Bay, USA. IMBCI scores are calculated by scoring species attributes on a generalist to specialist gradient.

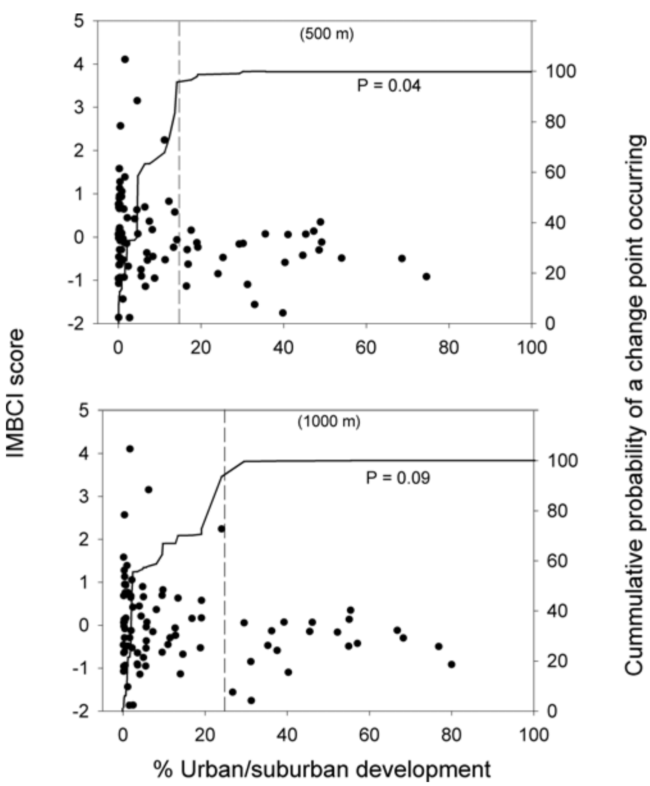



**Figure 20.** Results of non-parametric changepoint analyses for percent development within 500 m and 1000 m of wetland study sites and index of marsh bird community integrity (IMBCI) scores controlled for wetland size using standardized residuals from Figure 19. The cumulative probability curve represents the cumulative probability that a changepoint occurred at various levels of development. IMBCI scores are calculated by scoring species' attributes on a generalist to specialist gradient. Dashed lines indicate the percent of development within a wetland buffer required to produce a 95% cumulative probability of an ecological threshold occurring.

defined. Previous research has shown that birds are linked to the overall ecological integrity of their respective ecosystem. This is true primarily because birds are sensitive to habitat fragmentation, landscape composition, and changes in habitat structure. Birds may also be particularly good indicators because species at higher trophic levels can be sensitive to disturbances at lower levels. Therefore, it is unlikely that a marsh with low ecological integrity can support a high-integrity marsh bird community.

## Findings

Wetland size had a significant influence on IMBCI scores (Figure 19). Changepoint (statistical) analysis revealed a changepoint or threshold occurred when >14% of the area within 500 m of the marsh was developed. IMBCI scores decreased significantly as the percent of developed area increased beyond 14%. In fact, there was 95% probability that IMBCI scores would decline when >14% of the area was developed and a 60% probability of a change occurring when as little as 6% of the land within 500 m of a wetland was developed (Figure 20, top). However, changepoints were not significantly detected when agriculture or forest land use were tested against IMBCI scores at the 500 m scale.

Changepoint analysis also revealed a 95% probability of a changepoint occurring with ≥ 25% development within 1000 m with a 60% chance of a changepoint occurring when 8.5% of the 1000-m buffer was developed (Figure 20, bottom). Again, changepoints were not detected for agriculture or forest land use at the 1000-m scale. In addition, changepoints in IMBCI scores were not detected for percent development, agriculture, or forest at the watershed scale. More detailed information can be obtained from Deluca et al. (2004).

## Implications

Changepoints identified in this study represent ecological thresholds, beyond which the ecological integrity of the marsh bird community and potentially the entire marsh ecosystem becomes significantly compromised. These relationships were only identified at relatively local scales (500-m and 1000-m buffers), so it appears that local land cover is the best predictor of marsh ecosystem integrity. Furthermore, our results indicate that developed land use is the primary stressor to marsh bird communities of the Chesapeake Bay.

We demonstrated that the IMBCI is a reliable indicator of marsh bird community integrity that may assist in the assessment of the integrity of the entire marsh ecosystem. IMBCI scores, combined with the identification of a land-use threshold, are easily interpreted and provide rapid assessment approaches for communicating complex ecological data to natural resource managers and conservation planners. By helping to bridge the gap between scientists and regional conservation decision makers, the IMBCI could become a valuable tool to the ongoing efforts of restoring and maintaining the ecological integrity of coastal wetlands.

# WATERBIRD COMMUNITY INTEGRITY

## Background

We developed an index of waterbird community integrity (IWCI) to provide insight into estuarine ecosystem integrity and used it as a tool to: (1) determine land-cover types that influence waterbird community integrity; (2) identify relevant geographic scales at which land cover influences IWCI scores; and (3) test if ecological thresholds exist in the amount of land-cover disturbance that causes significant declines in IWCI scores.

We modified the IMBCI (DeLuca et al. 2004) to develop the index of waterbird community integrity (IWCI). We defined waterbirds as all species that forage exclusively or opportunistically on aquatic estuarine organisms (i.e., gulls, terns, waders, raptors, kingfishers, and waterfowl). Theoretically, the waterbird community is an ideal indicator because it is at the top of the estuarine food web. Therefore, this indicator is potentially sensitive to stressors influencing the system at multiple trophic levels. Furthermore, as a community that is closely tied to a functioning subestuarine ecosystem, it has high potential as an indicator to be sensitive to stressors at both the watershed and local scales (DeLuca et al. 2004, Hale et al. 2004).

## Findings

In 2002 and 2003, one single-predictor model, which included developed land cover, was a significant predictor of IWCI scores (Table 5). Depending on the year, this model was between 13 and 26 times more likely to describe variation in IWCI scores than any of the seven remaining candidate models (Table 5). Because development was the only predictor with strong support in both years, we focused subsequent analyses on this land use.

**Table 5. Relative ranking of models using land cover variables to describe variation in index of water bird community integrity (IWCI) scores. Columns give model notation, number of estimable parameters (*K*), second-order Akaike's information criterion values ($AIC_c$), $AIC_c$ differences (Ä$AIC_c$), and $AIC_c$ weights ($w_i$).**

| Model | *K* | $AIC_c$ | Ä$AIC_c$ | $w_i$ |
|---|---|---|---|---|
| 2002 | | | | |
| development | 3 | 68.5 | 0.0 | 0.752 |
| null | 2 | 73.7 | 5.2 | 0.056 |
| dev + forest | 4 | 74.1 | 5.6 | 0.046 |
| dev + agriculture | 4 | 74.1 | 5.6 | 0.046 |
| agriculture | 3 | 74.3 | 5.8 | 0.041 |
| ag + forest | 4 | 74.8 | 6.3 | 0.032 |
| dev + ag + forest | 5 | 76.5 | 8.0 | 0.014 |
| forest | 3 | 76.6 | 8.1 | 0.013 |
| 2003 | | | | |
| development | 3 | 52.1 | 0.0 | 0.903 |
| null | 2 | 58.6 | 6.5 | 0.035 |
| dev + forest | 4 | 59.7 | 7.6 | 0.020 |
| dev + agriculture | 4 | 59.7 | 7.6 | 0.020 |
| dev + ag + forest | 5 | 61.9 | 9.8 | 0.001 |
| agriculture | 3 | 62.3 | 10.2 | 0.001 |
| ag + forest | 4 | 62.6 | 10.5 | 0.001 |
| forest | 3 | 62.8 | 10.7 | 0.000 |

As total development increased, IWCI scores decreased significantly at the watershed, IDW, and 500 m scales in 2002 and 2003 (Table 6). Suburban development also had a significant negative impact on IWCI scores at the watershed, IDW, and 500 m scales for both years (Table 6). The relationship between total development and IWCI scores was consistently stronger than the relationship between suburban land cover and IWCI scores (Table 6). In addition, more variation was explained in IWCI scores when the two geographic scales emphasizing local land cover (IDW and 500 m) were used as predictors (Table 6). Increasing urban land cover also lead to lower IWCI scores in 2002 and 2003 at the watershed scale, however, the relationship between IWCI scores and the IDW and 500 m scales were not linear.

**Table 6. Results of linear regressions for IWCI scores and three land-cover types at three different geographic extents in a dry (2002) and wet (2003) year. Results are summarized as $r^2$ and *P* value.**

| | Geographic Extent | | | | | |
|---|---|---|---|---|---|---|
| | Watershed | | Watershed (IDW) | | 500-m buffer | |
| **Land Cover** | **2003** | **2003** | **2002** | **2003** | **2002** | **2003** |
| Development | 0.51, 0.001 | 0.54, <0.001 | 0.57, <0.001 | 0.60, <0.001 | 0.54, 0.001 | 0.57, <0.001 |
| Suburban/rural | 0.43, 0.004 | 0.47, <0.001 | 0.54, 0.001 | 0.57, <0.001 | 0.55, 0.001 | 0.54, <0.001 |
| Urban | 0.40, 0.007 | 0.51, <0.001 | NL* | NL* | NL* | NL* |

**Relationships between urban land cover and IWCI scores were not linear and were therefore analyzed with a changepoint analysis to test for the presence of an ecological threshold (see Figure 21).*

Changepoint analysis indicated that in 2002, when as little as 4% of the IDW land cover within a watershed was urbanized, there was a 94% probability of a threshold response in waterbird community integrity (Figure 21a). When testing the 500 m buffer scale in 2002, we found that when 4% of land cover was urbanized within 500 m of the subestuary there was an 85% probability of a threshold response in waterbird community integrity (Figure 21b). In 2003, when 5% of IDW land cover was urban it lead to a 99.9% probability of a threshold (Figure 21c). Finally, in 2003 we found that when there was as little as 5% urbanization within 500 m of the shoreline it resulted in a 99.9% probability of a threshold occurring (Figure 21d). Additional detailed findings can be found in Deluca et al. (in prep).

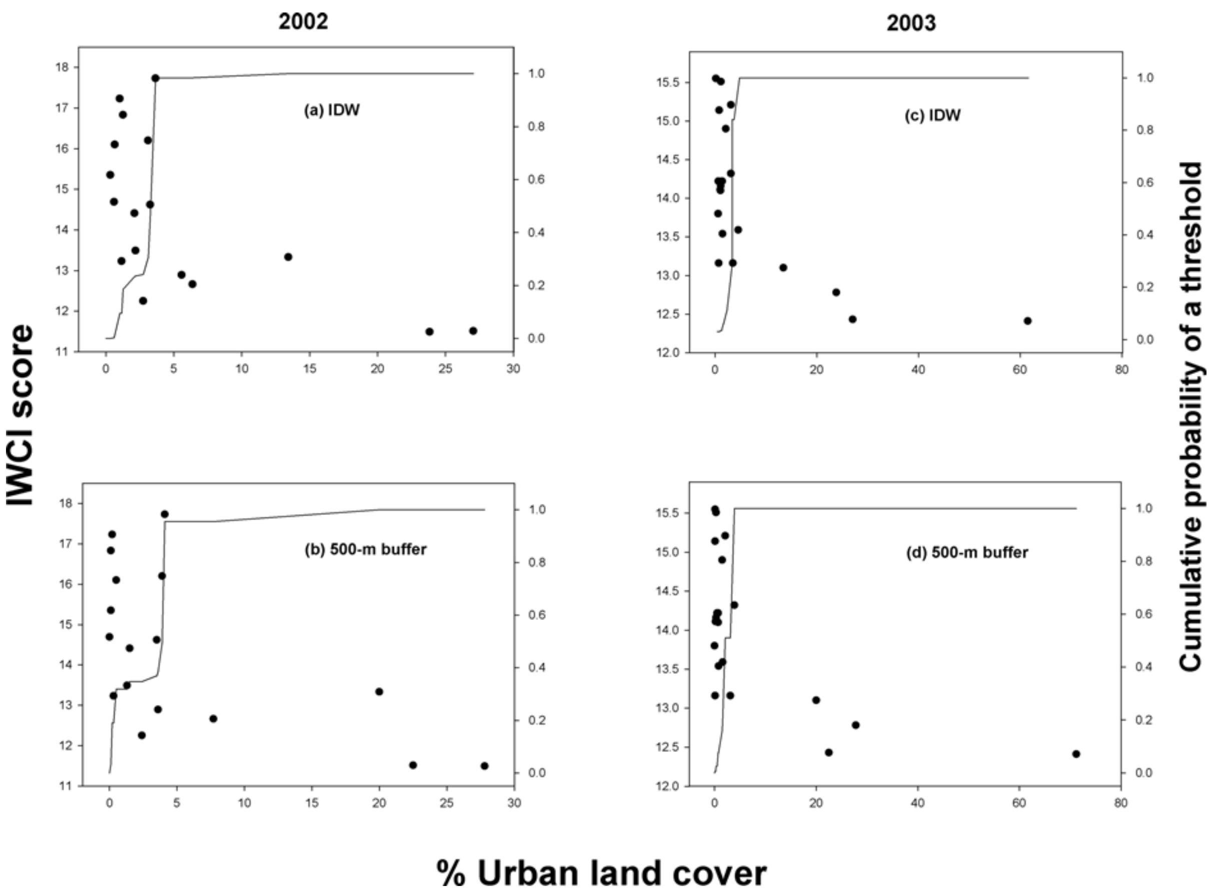



**Figure 21.** Results of changepoint analyses for percent urban development and index of waterbird community integrity scores (IWCI) in 2002 (a, b) and 2003 (c, d) for two different spatial scales; inverse distance weighted (IDW) land cover within the watershed (a, c) and a 500-m buffer around the subestuary (b, d). The solid lines depict the cumulative probability that an ecological threshold will occur with increasing urban development.

## Implications

The IWCI clearly identified developed land cover as the primary stressor influencing waterbird community integrity (Table 6). The waterbird community is particularly sensitive to urban development, as it exhibits a threshold response to alarmingly low levels of disturbance near the shoreline. From a management perspective, the threshold response to urban development at the IDW and 500 m buffer scales, offer clear management guidelines of how much coastal development estuarine ecosystems can tolerate before a collapse in ecological integrity can be expected. A compromised waterbird community, at the top of the estuarine food web, may have significant implications on the entire ecosystem through altered top-down food web relationships and controls (Baird et al. 2004).

# PCBS IN WHITE PERCH (*MORONE AMERICANA*)

## Background

The goal of this project was to develop statistical models that predict total PCBs (t-PCBs) in an economically and ecologically valuable fish species in Chesapeake Bay using different types of urban land use from estuarine watersheds.

Polychlorinated biphenyls (PCBs) are a group of organochlorine compounds that resist degradation in the environment and are widely distributed in aquatic ecosystems. PCBs accumulate in fat-rich tissues of biota. Because of their toxicity, PCBs present a health risk to both humans and a variety of other organisms. Although banned in the U. S. in 1979, PCB levels in many aquatic ecosystems remain sufficiently high to contaminate food webs and cause consumption advisories for a wide range of valuable fish and shellfish species.

Major sources of PCBs in estuaries are thought to be legacy pools of past point-source releases by manufacturing and from nonpoint sources associated with the general use, storage, and disposal of these persistent compounds. However, the sources, spatial extent, and magnitude of PCB contamination are not well characterized and have proven difficult to predict, presumably because estuaries are hydrologically open systems affected by long-distance transport of contaminants from upstream and downstream areas. However, some recent studies have successfully linked land use data from small estuarine watersheds to various sediment contaminants. Given that PCBs are known to be associated with industrial or other urban land uses, these previous findings suggested to us that quantification of land-use patterns in watersheds may be useful for predicting PCB contamination in downstream estuarine ecosystems.

We tested the hypothesis that the amount and spatial proximity of urban land in watersheds would be significantly linked to concentrations of total PCBs (t-PCBs) in biota from estuarine segments of Chesapeake Bay. We examined: (1) the strength of correlations between different measures of developed (urban) land in the watershed and t-PCBs; and (2) the relative improvement in our predictions of t-PCBs afforded by weighting urban land by its inverse distance from the shoreline to account for proximity to the estuarine segments. We focused on t-PCBs in White Perch (*Morone americana*), a widely distributed estuarine fish that supports a valuable commercial and recreational fishery throughout Chesapeake Bay. White perch are an ideal indicator species for detecting watershed linkages to PCBs because they spend most of their lives within or near specific estuarine segments. White perch also prey upon small fish and bottom-dwelling invertebrates, which are consumers of fine organic particles running off the land and accumulating in sediments. Moreover, White Perch are semi-anadromous, moving into freshwater tributaries to spawn with the young moving back down into the estuarine segments to find a nursery and feeding habitat, so their life cycle spans a zone that continuously exposes them to runoff from the watershed. Finally, because PCB-related consumption advisories have recently been posted for several estuarine segments and many other locations have yet to be assessed, there is great interest in developing geographical indicators of PCBs in this region.

## Findings

All unweighted developed land-use measures were significant predictors of t-PCBs in White Perch, explaining 51% to 69% of the variance among the 14 estuarine segments. Percent high residential/commercial land was the best predictor of t-PCBs among the unweighted developed-land-use classes (Figure 22, top).

Inverse-distance-weighting markedly improved the linear fit of each land-use predictor and t-PCBs in White Perch among the 14 estuarine segments (Figure 22, bottom). Inverse-distance weighted percent commercial land, was the best predictor of t-PCBs of any of models considered and accounted for nearly all the variance ($r^2$ = 99%).

Two estuarine segments had distinctly higher levels of t-PCBs than the other estuarine segments and may have had disproportionately strong effects on the regressions; so the effect of removing these two observations from the analysis was evaluated. All land-use classes remained significant predictors of t-PCBs using the reduced (n=12) set of observations. In particular, inverse-distance-weighted models for % high-residential/commercial and % commercial land exhibited large improvements in explain-

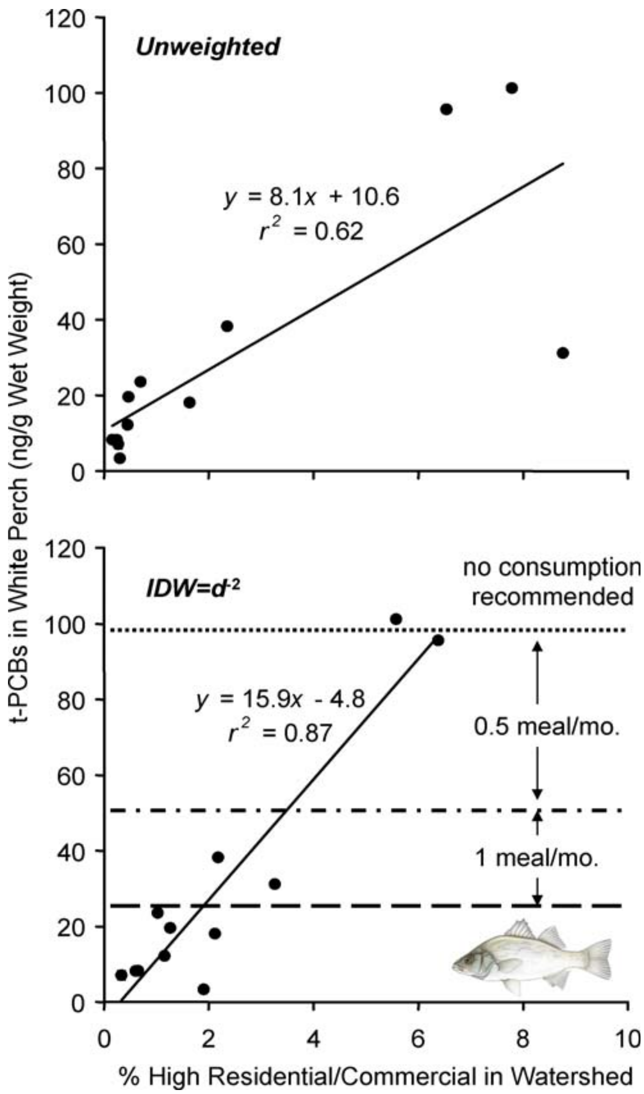



**Figure 22.** Regressions of unweighted (top) and inverse-distance weighted (IDW) % high-intensity residential/commercial (bottom) in watersheds on t-PCBs in white perch across 12 subestuaries, excluding the two locations with the hightest levels of developed land and t-PCBs. Dashed lines illustrate levels of t-PCBs that correspond to U.S. EPA (1999) consumption advisories for cancer health endpoints.

ing variance over unweighted models. Percent high-res/comm and % commercial land explained 87% and 86% of the variance in White Perch t-PCB concentrations, respectively among all predictors in the reduced data set. Additional information can be found in King et al. (2004).

## Implications

Our study is novel because we demonstrated a remarkably strong relationship between the amount of developed land in watersheds, weighted by its proximity to the water, and PCBs in White Perch across many tributaries of Chesapeake Bay. No previous study has demonstrated such a relationship between watershed land use and contaminants in fish, particularly among multiple watersheds. Perhaps more importantly, we also showed that very little watershed development, particularly near shorelines, corresponded to levels of PCBs that were unsafe for human consumption. Thus, these findings were not just limited to highly urban areas where we already know the water is badly polluted. Although PCBs have been banned since 1979, new consumption advisories for several fish species have been posted across many Chesapeake Bay tributaries because of PCBs, and these advisories have been big news for communities previously unaware of this problem. Our study suggests that PCBs historically produced and used in this region are persisting in the environment at the scale of these watersheds, and urban runoff may still be acting as a source of legacy PCBs to downstream aquatic habitats.

The relationships we discovered will be very important to managers because they may be used as tools for predicting areas that have a high probability of PCB contamination. Moreover, because many other contaminants are associated with development, these models will likely be very useful for identifying other types of contamination in estuaries. Many new contaminants are still in production and use, including flame retardants (PBDEs), metals, and emerging contaminants such as pharmaceuticals, and may well be related in a similar way to the amount and spatial proximity of development in watersheds.

The study also helped confirm that White Perch may be an ideal species for assessing bioaccumulation of estuarine contaminants associated with watershed runoff because of its small home range on an individual level but broad distribution across a wide range of salinities that span the length of Chesapeake Bay.

On a broader front, this study points to the importance of better understanding the impacts of development on estuaries. Our study highlights the implications of development on the health of aquatic ecosystems. It links environmental and ecological conditions in estuaries to land use in their associated watersheds. There may be other contaminants at unsafe levels in estuaries that we have yet to discover that are related to urbanization.

# BIO-OPTICAL INDICATORS

## Background:

Communities of submersed aquatic vegetation (SAV) are highly valued habitats because of the functions they perform in coastal systems. These functions include, among others, provision of refuge and nursery habitat for juvenile fish, shellfish and crabs, sediment stabilization, and food for certain waterfowl. Loss of valuable SAV habitat has been one of the most deleterious effects of pollution in numerous coastal systems along the Atlantic slope. Presence or absence of SAV is, therefore, a powerful indicator of estuarine water quality. Efforts to preserve and restore seagrasses have focused mainly on factors affecting water clarity, because of the inherently high light requirement of seagrasses. The attenuation of light in water is controlled by the concentrations of three parameters: suspended particulate matter (SPM), phytoplankton chlorophyll (Chl), and colored dissolved organic matter (CDOM). The goal was to develop an optically based indicator of habitat suitability for SAV, and explore its variation with land use in the local watershed.

## Findings

Concentrations of chlorophyll were higher in estuarine segments with developed watersheds, while CDOM was higher in segments with developed and mixed agricultural watersheds. Concentrations of TSS were remarkably independent of land use in the local watershed, including the reference site. Specific-absorption coefficients were significantly higher in segments with developed and mixed-developed watersheds. Specific-scattering coefficients were also elevated somewhat in these land uses. Using these specific-absorption and –scattering coefficients in bio-optical modeling routines, we determined water quality thresholds (diagonal lines in Figure 23) that delineate conditions that will support SAV (low concentrations, points near the origin) from those that will not (concentrations falling outside the thresholds. The green shaded area represents the approximate contribution of phytoplankton to TSS, and is an area that should have few or no points).

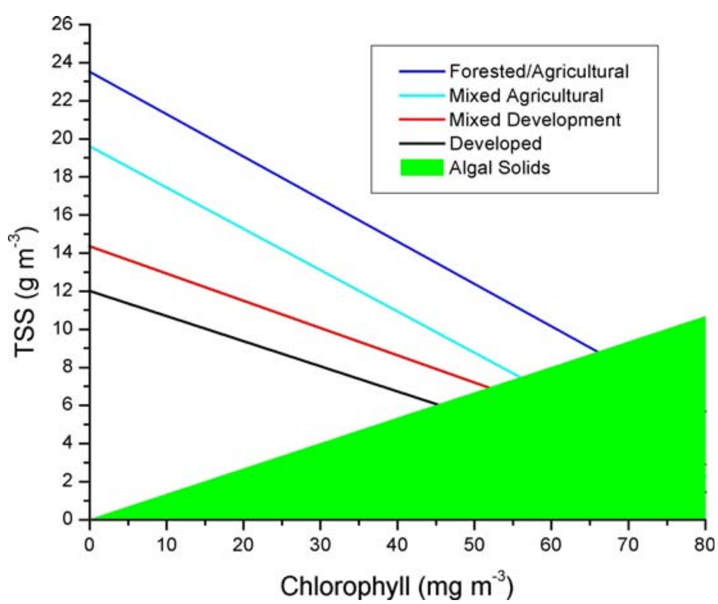



**Figure 23.** Water quality thresholds for SAV growth in estuarine segments of Chesapeake Bay with differing land use in their watersheds. Differences as development increased were due to higher concentrations of CDOM as well as higher specific-absorption and -scattering coefficients of suspended particulate matter.

## Implications

Not only did estuarine segments with developed watersheds have higher concentrations of optically significant water quality constituents (especially chlorophyll), but the water quality requirements for segments with developed watersheds were considerably more stringent than less developed watersheds. The results imply that greater management effort is expected to be required to restore SAV in developed watersheds. Optical properties of the particulate matter and bio-optical modeling offer improved insight into mechanisms responsible for loss of SAV.

# MESSAGE 3

## Small Watershed Approach

The ecological condition of streams, wetlands, and riparian areas depends upon land use and other activities upstream. Therefore, classifying regions first by social choice and physiographic setting allows better distinction of variation due to natural sources (slope, soils, extreme natural events, etc.) and those due to human activities (hydrologic alteration, pollution, etc.). Certain physical, chemical, and biotic indicators respond to the degree and type of human activities, and thus can predict these effects on the condition of aquatic ecosystems. Indicators are desired that respond to different types of questions, are useful at various spatial and temporal scales, and are relevant in a range of physiographic settings, watershed land use types, or social choices.

All of the indicators described here (see Table 2) address ecological condition. The spatial scale differs among indicators, but generally applies either to characterizing a small watershed (e.g., 14-digit HUC or smaller) or to a particular site or reach of stream. Indicators associated with Level 1 Landscape assessments typically use remote sensing or GIS scenes and generally apply to watershed scales; however, the aggregation of data at a watershed scale also can be used to interpreted to apply the condition of a downstream point whether it be a site or a reach.

Indicators that require field observations (Level 2 Rapid or Level 3 Intensive assessments) can provide information for interpreting ecological condition at the site level as well as upstream conditions. In fact, in-stream biotic indicators (e.g., IBIs) may reflect conditions upstream more than they do the surrounding habitat in the floodplain and riparian zone (Brooks et al. In press). Individual site conditions, if randomly selected within a watershed, can be aggregated to provide a statistically based estimate of a stream network at small watershed scales (Rheinhardt et al. in review).

The development of indicators and their calibration requires substantial effort. However, the actual practice of assessing the condition of a site or a watershed can take advantage of these efforts, and apply them to resource management or regulatory programs. Consequently, managers do not incur the cost of development and calibration when indicators have been developed for the region of interest or for particular programmatic purposes. Indicators developed for the Atlantic Slope generally use the same metrics used in indices for other regions, but they are calibrated within regional climatic, soil, biotic, and cultural conditions. If indicators are not calibrated within a physiographic region, they may be ineffective at separating variation due to natural sources from those due to human alterations.

## General Description of Activities

There were two main thrusts in the ASC analyses of small watershed segments: (1) developing an integrated assessment protocol for simultaneous rapid assessment of the conditions of streams, adjacent wetlands, and adjacent riparian zones at an assessment point; and (2) developing improved geographic models for predicting the chemical and biological conditions of streams from watershed characteristics, particularly land cover.

Scientists from PSU, ECU, and SERC cooperated in the development of the integrated rapid method for simultaneously assessing the condition of streams and the adjacent wetlands and riparian zone. To develop and test the assessment protocols, field observations were collected from small watersheds throughout the Atlantic Slope. Twenty-four study watersheds (14-digit HUCS) were selected to represent the range of ecoregion and land use types in the Atlantic Slope (Figure 7). Field teams sampled about 20 randomly selected stream locations within each selected watershed and measured conditions in the stream and near stream zones. The resulting measurements were synthesized to produce an SWR (Stream, Wetland, and Riparian) index for the entire stream/near-stream complex, and to develop other indicators that applied to separate components of that complex. ECU scientists supplemented this effort with an evaluation of the influence of beaver impoundments as a potential indicator.

SERC scientists led the geographic modeling effort, which focused on improving the methods and models for using watershed characteristics, particularly land cover, as landscape indicators of stream water quality and biotic condition. This effort exploited available water quality data from previous SERC studies of streams draining small watersheds (e.g., Jordan et. al. 1997a,b,c, Liu et al. 2000. Weller et al. 2003) and available data on physical, chemical, and biotic condition of streams from the Maryland Biological Stream Survey (cite them). We used statistical models to relate the chemical and biological data (dependent variables) to independent variables derived from analyzing digital watershed maps with a geographic information system (GIS). We focused on the independent variables including physiographic province and land cover, especially cropland and developed land. We were especially interested in how the spatial arrangement of land cover moderated its influence on stream responses, so we explored new methods and metrics for accounting for two important aspects of spatial arrangement: The distance of disturbed areas to assessment points and the presence and the distribution of riparian buffers along hydrologic flow paths connecting disturbed areas to streams. We used correlation, regression, multiple regression, and threshold analysis to relate responses to land cover, distance-weighted land cover, and new metrics describing riparian buffer distribution.

## Results

The development of the SWR index demonstrated that rapidly assessed field indicators can be tailored into effective tools for quantifying stream, wetland, and riparian condition within physiographic regions and land-use categories. The analysis also revealed patterns in stressor distributions among physiographic regions and social choices categories. Hydrologic modifications and measures of sediment and erosion were by far the most dominant stressors in all physiographic regions and land uses. Vegetation modification was also very prevalent, and invasive species were particularly common in Coastal Plain and mountain settings than in the Piedmont. Coastal plain streams had fewer nearby stressors than streams in other physiographic provinces, but the numbers of stressors for urban lands were similar across provinces.

The geographic modeling effort developed enhanced methods for identifying and calibrating landscape indicators of stream responses, and those methods were applied to yield some specific recommended indicators (next section). We identified a number of spatial challenges that arise in relating land cover to stream responses, and we presented statistical methods to surmount those challenges (King et al. 2005). Our work also produced methodological improvements in integrating community data into response indices (King et al. 2005) and in automatically delineating watershed boundaries (Baker et al. 2006). Although land cover alone is a useful indicator of stream condition, we showed that indicator models can be improved by incorporating information on the spatial patterns of land cover through distance-weighting of source areas (King et al. 2005, manuscripts in preparation) or through new functional riparian metrics that consider distribution of riparian buffers along hydrologic flow paths connecting source areas to streams (Baker et al. in press, Baker et al. submitted). Our analyses show that both the amount and spatial arrangement of cropland and development in a watershed can have a significant impact on nutrient discharges (King et al. 2005, manuscripts in preparation). Similarly, the amount and spatial arrangement of developed land (or impervious surface) significantly affect the response thresholds of stream macroinvertebrate communities (King et al. 2005, manuscripts in preparation). Compared to traditional ways of quantifying riparian patterns, our functionally-based riparian metrics were more interpretable and more independent of watershed land cover (Baker et al. in press, Baker et al. submitted, Baker et al. in preparation). Analyses incorporating those metrics show that riparian buffer configuration is correlated with reduced nutrient discharges in some but not all provinces within the Chesapeake Bay watershed (Baker et al. in preparation).

The general conclusions summarized above are embodied in the example results for selected indicators presented in the next section. Additional ASC indicators of watershed stream, wetland, and riparian condition and their taxonomic characteristics (e.g., type of question, scale, context) are listed in Table 2.

## INVERSE-DISTANCE WEIGHTED CROPLAND

### Background

Croplands closer to water bodies can be stronger sources of sediment and nutrients to aquatic systems, while discharges from more distant croplands may be attenuated by a variety of processes along transport pathways before reaching a water body (Soranno et al. 1996).

This index is based on the proportion of cropland in a watershed, modified by giving greater weight to cropland areas closer to water bodies or sampling stations while still including some effect of more distant croplands. The metric is calculated from digital land cover, elevation, and stream maps using a GIS. For every pixel in a watershed, we calculate the horizontal distance to a water body or sampling station along the steepest descent flow line determined by landscape topography. It the distance is measured to streams, flow lines derived from digital elevation surfaces must first to be modified to match the stream maps. All pixels are weighted by the inverse of this distance (1/distance) and weighted cropland properties are estimated by dividing the sum of all weighted cropland pixels by the sum of all weighted pixels in the watershed (King et al. 2005).

## Findings

In one application, distance weighting of cropland proportion improved predictions (higher $r^2$) of stream nitrate concentrations measured in the Maryland Biological Stream Survey for small coastal plain watersheds (<600 ha or 1,500 ac), but not for larger watersheds (King et al. 2005). In another test using stream chemistry data from 429 Chesapeake Bay subwatersheds in 4 physiographic provinces, distance weighting cropland (distance to streams) improved predictions of stream nitrate concentration the Coastal Plain, but not in other 3 physiographic provinces. Distance weighted cropland proportion can range from 0 to 100% and will differ increasingly from simple cropland proportion as cropland distribution becomes less uniform, perhaps by preferential location of croplands on uplands or on floodplains.

## Implications

The percentage of cropland in a watershed is a commonly-used indicator of sediment and nutrient concentrations in streams. Distance weighting can improve predictions for small watersheds, but the benefits of distance weighting vary among physiographic settings.

## INVERSE-DISTANCE WEIGHTED DEVELOPED LAND

## Background

Urban development adjacent to a stream reach and throughout the watershed can degrade stream ecosystems, but near-stream development may have a stronger effect. By weighting land cover near the reach more heavily, the metric preferentially emphasizes local, acute effects of development (e.g., riparian forest removal, dumping hazardous materials) while still incorporating the effects of more distant development (watershed-scale hydrological modification, nonpoint source runoff).

Inverse-distance-weighted (IDW) index of developed land gives greater emphasis to developed land near a feature of interest (e.g., a stream reach) than developed land located farther away. The metric is calculated from digital land cover and stream maps using a GIS. Within an individual watershed, every pixel is assigned a distance (meters) to an assessment point using simple Euclidean distance. Pixels are then weighted by the inverse of their distance (1/distance) to the assessment point. The sum of distance-weighted developed land (residential and commercial) is divided by the sum of distance-weighted total land in the watershed to yield a distance-weighted developed land percentage.

## Findings

Relationships between developed land and macroinvertebrate assemblages were examined among 295 Coastal Plain streams in Maryland (King et al. 2005). Assemblages exhibited an ecological threshold between 21 and 32% when unweighted developed land in watersheds was used (Figure 24). Beyond 32%, the probability was almost 100% that all streams were biologically impaired. However, the apparent developed-land threshold dropped to as low as 18% when weighted by its inverse distance to the sampled reach of stream, with 100% certainty of a threshold above 23% IDW developed land. IDW % developed land can range from 0 to 100%, and deviates most significantly from unweighted % developed land among watersheds with distinctly different spatial patterns of urbanization. Higher values indicate greater probabilities of stream impairment, particularly above 18-23% in Coastal Plain streams.

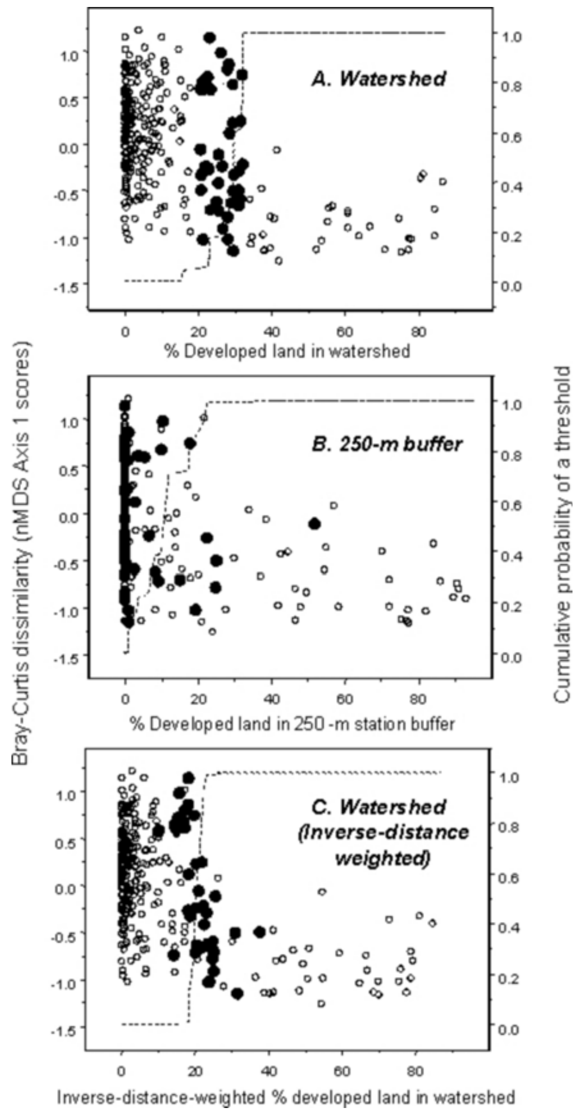



**Figure 24.** Scatterplots of the threshold effect of developed land on macroinvertebrate assemblage composition (Bray-Curtis dissimilarity expressed as nonmetric multidimensional scaling [nMDS] Axis 1 scores). (A) Percentage developed land in the watershed. (B) Percentage developed land within a 250-m radius buffer of the sampling station. (C) Percentage developed land in the watershed weighted by its inverse distance (IDW; in meters) to the sampling station. The dotted lines indicate the cumulative probability of an ecological threshold in response to increasing percentage developed land. Samples within the watershed-scale threshold zone of 21-32% developed land in panel (A) are highlighted in black in panels (A)-(C).

## Implications

Developed land contributes to stream impairment. Distance-weighted developed land provides a more discriminating indicator of the threshold effects on stream macroinvertebrate communities than does the simple proportion of developed land because distance-weighting accounts for the stronger effects of development near an assessment point without ignoring more distant land. The index can also be useful in targeting best management practices and ecological restoration by identifying not only what land use practices should be changed to improve conditions, but also by informing where in a watershed those changes would be most effective.

# SOURCE-SPECIFIC MEAN RIPARIAN BUFFER WIDTH

## Background

Riparian buffer width has long been considered a key measure for estimating buffer effects on water chemistry and other stream responses (Lowrance et al. 1997, Weller et al. 1998). However, watershed scale analyses have relied on the proportion of buffer within a fixed distance of streams as the measure of buffering potential (Jones et al. 2001). Our calculation of the source-specific mean riparian buffer width metric provides a more functionally based measure that focuses on that portion of the riparian system that is actually connected to a source area, and measures width along paths of likely hydrologic transport.

Source-specific mean riparian buffer width examines the potential of riparian buffers to reduce the effects of a specific land cover on aquatic systems. The source area can be any land cover type (such as cropland or developed land) that can affect stream responses. The metric is calculated from digital land cover, elevation, and stream maps using a GIS. Prior to analysis, digital elevation surfaces must first be modified to align with stream maps (Baker et al. 2006). Within a watershed, all surface flow paths leading downhill from source areas to a stream are identified, and then the width of riparian buffer (forest or wetland contiguous with streams) is calculated for each flow path. Mean width is averaged across all flow paths weighted by the source land cover area contributing to a flow path (Baker et al. in press, Baker et al. submitted).

## Findings

Mean riparian buffer width for croplands was quantified for 503 small (6-48,000 ha) watersheds within 4 major physiographic provinces of the Chesapeake Bay drainage (Lui et al. 2000). Source-specific mean widths were compared with a more commonly-used measure—the percentage of forest within 100 m of a stream. Figure 25 shows that source-specific mean buffer width for cropland captured large differences among watersheds that had similar values of forest with 100 m of streams (Baker et al. in press).

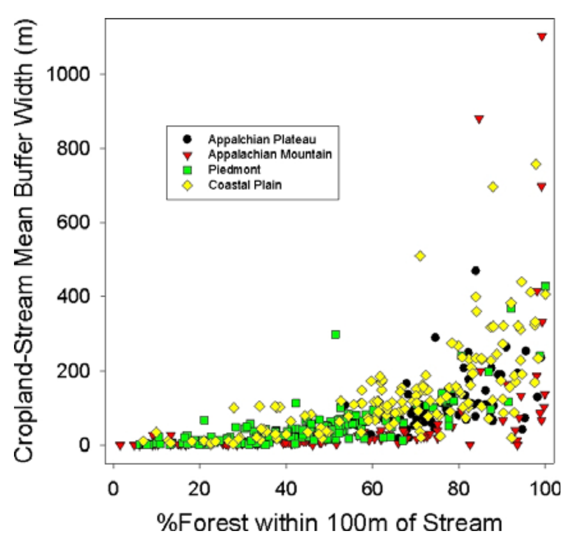



**Figure 25.** The average buffer width along flow paths connecting cropland to streams (vertical axis) varied widely among watersheds which had similar forest percentages in the area within 100 m of streams (horizontal axis), particularly in more forested watersheds (right side of plot). Mean buffer width can represent the different buffer potentials of watersheds that seem the same according to the commonly used but less discriminating metric on horizontal axis.

## Implications

Source-specific mean riparian buffer width considers only that part of the riparian system that is likely connected to a source area and then integrates the buffering potential along the likely lines of flow from source areas to streams. This indicator is derived from combining measurements about the characteristics of the riparian buffer along the source- to-stream transects (Lowrance et al. 1997). This functional connectivity provides a more discriminating indicator of buffering potential across whole watersheds than the common method of calculating the percentage of forest within a fixed distance of streams. It could be useful as a planning tool for identifying where the restoration of forested buffers in watersheds might be most effective, thus providing insight into more economically efficient approaches to stream and riparian restoration.

# STREAM, WETLAND, RIPARIAN (SWR) INDEX

## Background

Components of the sampling protocol are based on rapid assessment methods developed and tested by the EPA (e.g., Stream Habitat Assessment – Barbour et al. 1999) and by the Cooperative Wetlands Center at the Pennsylvania State University (e.g., stressor checklist, riparian buffer score). Metrics used to compute the SWR Index were selected based on a conceptual model of their relationship to aquatic system condition.

A Stream, Wetland, Riparian (SWR) Index was developed to produce simultaneous assessments of condition for these interrelated components of aquatic ecosystems. A GIS was used to select about 20 stream-centered points for 24 small watersheds stratified by Mid-Atlantic ecoregions and land use type. In 2003, aspects of hydrology, soils, vegetation, and topography were measured in one 100 m x 100 m plot per site using a rapidly implemented sampling protocol (<2 hr). Observations of on-site stressors were recorded. Landscape metrics are computed from 1-km radius circles centered on each point. We combined the floodplain-wetland and stream measurements into an indicator of overall condition for small watersheds (SWR Index) and examined the relationship with the Landscape Index (get reference). Comparisons were made to assessments derived from existing biological and chemical data from intensive studies. This indicator is composed of the following floodplain-wetland metrics: buffer width, basal area, number of tree species, abundance of invasives, number of stressors, and these stream condition metrics: Stream Habitat Assessment score, incision ratio, and number of stressors.

## Findings

The sampling protocol was applied at approximately 20 sites in each of 24 watersheds in the Atlantic Slope study region, representing a range of land cover types and physiographic regions. Values of the index were compared with Index of Biotic Integrity (IBI) values for fish and benthic macroinvertebrates in selected watersheds of the study region. For the most part, the SWR Index agreed well with these biotic indices. Agreement was better for macroinvertebrates at the site level, and better for fish at the watershed scale. The SWR index was also compared with a landscape-level (GIS-based) index (Figure 26).

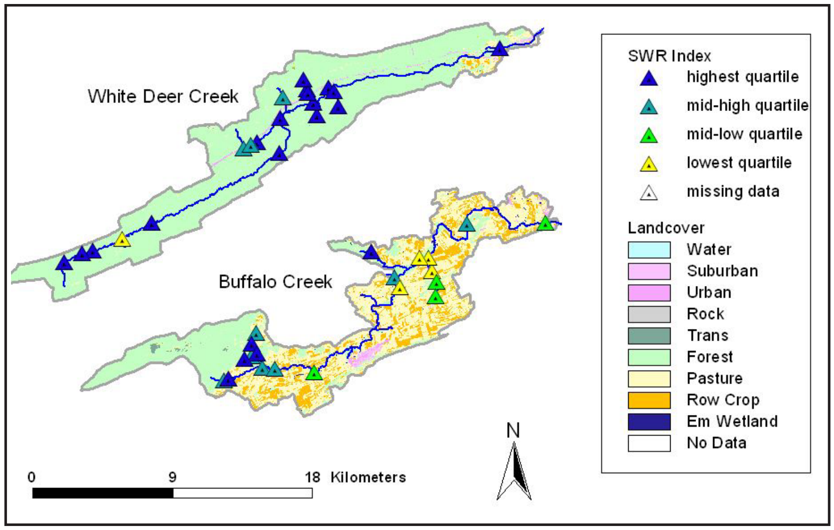



**Figure 26.** Nested SWR Index and Lanscape index.

For sites where we had both Level 2 and Level 3 measurements, we found a highly significant correlation between the SWR Index and the benthic IBI, but the correlation with the fish IBI was weaker, and the link with $NO_3$ was very weak. One would expect benthic invertebrates to be more influenced by site-level conditions than fish (which are more mobile) or $NO_3$ (which integrates over a larger upstream area).

When Level 3 measurements were compared with the average SWR Index in their upstream contributing area, all three Level 3 indices were correlated with the SWR Index. The relationship was strongest for the benthic IBI, followed by the fish IBI, and $NO_3$. This suggests that looking at multiple sites in the upstream area may give us a broader representation of condition at a point. Although the correlation with the benthic IBI was somewhat weaker than the site-to-site comparison, the relationships of the SWR Index with fish and nitrate were strengthened.

We also compared the average of all SWR points in a HUC-14 watershed with the average of all Level 3 points for a HUC-14 watershed. The correlation was statistically significant for the fish IBI, nearly so for the benthic IBI, but not for the nitrate. The very small sample size made relationships difficult to discern and statistical significance difficult to achieve. However, the indication is that our sample of 20 SWR points provides a reasonable estimate of biological condition, but not of chemical condition, at the watershed level.

## Implications

Rapid assessment protocols for riparian and stream condition are important for identifying water and habitat quality problems at watershed scales. More intensive methods that require considerably more time are still necessary prior to deciding on restoration activities for specific projects. The rapid assessment methods, when applied to randomly chosen sites within a watershed, provide an unbiased evaluation at watershed scales. The SWR Index strives to simultaneously assess all components of these aquatic ecosystems, rather than to treat each separately. The SWR Index should complement existing stream, wetland, and floodplain monitoring programs. It is designed to be used with the Landscape Index. Since their development, both the SWR Index and Landscape Index have been incorporated into monitoring programs of specific units of the National Park Service and by the state of North Carolina.

# SPOT SAMPLED AVERAGE STREAM NITRATE CONCENTRATION

## Background

Nitrate is often the dominant chemical form of nitrogen lost from watersheds through storm runoff and stream flow, and nitrate is even more predominant over other nitrogen forms in the discharges from croplands, developed lands, and other areas disturbed by human activities. Nitrate is highly soluble, so it easily enters the soil and follows subsurface transport pathways to streams. Therefore, much of the nitrate loss from watersheds occurs in baseflow between storms, and nitrate concentrations are less temporally variable than concentrations of other nitrogen forms transported episodically on particles during storms.

Stream nitrate concentrations can be a direct measure of nitrogen pollution in streams and a potential predictor of aquatic biological responses. Nitrate concentration rises with increasing proportions of agricultural or developed land in a watershed. To measure spot sampled average stream nitrate concentration, multiple water samples are collected from a stream reach during non-storm conditions, with at least one sample in each season. The samples are analyzed chemically for the concentration of nitrate, and the results are averaged to estimate the mean annual nitrate-nitrogen concentration in baseflow.

## Findings

For 66 Chesapeake Basin subwatersheds of differing land cover proportions in 4 major physiographic provinces, we compared average spot sampled nitrate concentration to flow-weighted average nitrate and total nitrogen concentrations measured with 1-3 years of weekly composite flow-proportional sampling at an automated stream sampling station. Average spot sampled nitrate concentration was a very strong predictor of nitrate ($r^2$=.98) and total nitrogen ($r^2$=.98) concentrations from the much more labor intensive and expensive automated sampling. Spot sampling is a cost effective approach to accurately gauge nitrate pollution across many sites in broad, regional studies or assessments.

Average spot sampled nitrate concentration in stream water increased with increasing amounts of developed land or cropland in a watershed, ranging from 5 mg N/L in completely forested watersheds to 20 mg N/L or more in heavily agricultural watersheds. The increase per unit area is steeper for cropland, and the rate of increase for a land use type can vary with physiographic setting (Jordan et al. 1997a, b, and c; Lui et al. 2000; Jordan et al. 2003b; Weller et al. 2003).

## Implications

The high degree of agreement between intensive automated sampling of streams and spot samples for nitrate in streams gives confidence that this indicator is reliable. Because of the cost savings, the approach can be applied to many more streams and reaches, and thus improve the effectiveness of water quality monitoring programs.

# BEAVER IMPOUNDMENT PRESENCE

## Background

Beaver (*Castor canadensis* Kuhl) populations have experienced a resurgence in recent years in the ASC study area (Arner and Hepp 1989, Butler 1991). Beaver impoundments increase the surface area of wetlands, depth of flooding, and residence time of water relative to streams. These impoundments are mostly found in low order streams, which constitute the majority of stream networks and are generally closest to sediment and nitrate sources. Nearly everywhere in North American where beaver impoundments have been studied, they reduce the downstream transport of nutrients and sediments, and alter the floodplain habitat from one of emergent marsh and forest to those dominated by submerged aquatic vegetation and shallow ponds.

## Findings

In a study of 13 beaver impoundments in the inner coastal plain of North Carolina, impoundments significantly decreased nitrate and TSS concentrations relative to control reaches (Bason 2004). The presence of beaver impoundments should be considered a factor that improves water quality. However, this indicator is very tentative at this point because the metrics have not been worked out to quantitatively predict the effects of individual impoundments on water quality at watershed scales.

## Implications

The increase in water depth and wetland area from beaver activities convert forested habitats to other wetland types. This can be perceived as a negative effect by landowners, particularly if timber revenues are lost, agricultural land is flooded, or property is devalued in other ways. The open habitat created by beaver impoundments can lead to the proliferation of invasive and exotic species (Barden 1987). These potential negatives must be weighed against the advantages of water quality improvement through reduction in nitrate concentrations and suspended sediments. It is currently not possible to evaluate the effect of an individual beaver impoundment on water quality within a watershed. However, additional analyses are on-going.

# MESSAGE 4

## Human Dimensions

Indicators can provide scientific information for answering questions related to the condition, trends, and causes of problems in aquatic ecosystems. As we move toward answering the latter three questions stated at the beginning of this report,

- What can we do about it?" (management)
- "Are we making a difference?" (performance measures)
- "How do we tell the story?" (communication)

we bring in another dimension besides space and time—the human dimension. Effective management, performance measures, and communication requires consideration of socioeconomic as well as ecological indicators, information, and insight.

The ASC included the human dimension because we think it is critical in understanding and resolving conflicts between *social choices* (land uses) and *societal choices* (designated uses). Ecological, cultural, social, economic, and political factors influence both social and societal choices. Gaining a better understanding of how indicators from each of these sectors interact and affect decisions can contribute to better communication among various stakeholders and better, more informed policy and management decisions.

Useful environmental indicators must have ecological validity and reliability. But they must also be meaningful to and relevant for intended audiences. These audiences include decision makers in environmental and resource management and planning agencies, as well as stakeholders to whom decision makers must be responsive. Moreover, given limited resources for assessing and protecting ecosystem health, the indicators to which society devotes resources should add significant value to environmental management. Choices about types of indicators, the scales at which they are gathered, and the precision with which they are measured should be guided by the value of the information for management relative to the costs of developing and maintaining the indicators.

The goal of the human-dimensions research was to provide scientific results that support the choice and communication of suites of environmental indicators that environmental managers and other audiences will find useful for:

- Characterizing the condition of resources and ecosystems at multiple scales
- Diagnosing likely causes of degraded conditions
- Evaluating (when linked with hydrological, ecological process, socioeconomic, and other models) the probable consequences of changes in measurable landscape attributes.

- Setting management priorities and selecting management strategies.

The human dimensions research emphasized:

1) How managers use indicators and their desired indicator characteristics.
2) When factors affect indicator use in public decisions.
3) The effects of commingling socioeconomic and environmental indicators.
4) Differences in how scientists and informed citizens perceive ecosystem condition.
5) How additional information improves decision making.

## DESIRED CHARACTERISTICS AND USE OF INDICATORS

### Background

Managers use indicators to monitor and assess environmental condition and trends, set agency priorities, enforce regulations, measure human and economic consequences of changes in ecological condition, and communicate with stakeholders. In addition, greater emphasis is being placed on government agencies to define desired environmental outcomes and assess the effectiveness of management practices and policies in achieving these desired outcomes.

Personal interviews were conducted with 46 government officials from state and federal agencies and interstate commissions to determine:

1) How environmental indicators were used by managers in assessment and management decisions, and

2) What characteristics were desired in environmental indicators used in decision-making.

### Findings

Managers preferred suites of indicators with issue-dependent elements rather than a single index or indicator because they were able to construct a more complete picture of environmental condition and the factors contributing to this condition with suites of indicators. Individual indicators were used in assessing attainment of individual water quality standards (WQS) (e.g., dissolved oxygen concentration or fecal coliform bacteria counts). However, indicators were considered most useful when they also provided insight into sources and factors responsible for existing conditions, including non-attainment of WQS. Environmental indices that provided a single number (e.g., fish index of biotic integrity), but that did not provide diagnostic information about environmental condition were not considered as useful as suites of indicators.

The attributes that made indicators useful depended on the specific purpose for the indicator. For example:

- For *monitoring and assessment*, indicators must be sensitive to the relevant spatial and temporal scale, and must be adaptable to improving technology.
- For *setting priorities*, managers considered the ability to measure impairment as the most useful indicator attribute. Indicators that allowed agencies to identify impairments influence the distribution of agency resources.
- For *regulatory enforcement*, managers considered scientific accuracy and consistency in measuring standards as the most important attributes. Indicators must hold up in court. Ambiguity in indicator interpretation was not acceptable.
- For *communication*, indicators must be adaptable to different audiences and concerns. Officials communicate with a wide variety of stakeholders, ranging from other regulatory agencies to elementary school children.

Each official and agency involved in indicator development had specific goals for the application of indicators. These goals often dictated what kinds of indicator data were collected, where data were collected, and how often data collection occurred. Differing perspectives on indicator development were also apparent between managers and scientists. Managers used indicators as information to contribute to decisions, while scientists used indicator information to understand relationships (e.g., cause-effect) in ecosystems. A significant challenge identified by respondents was achieving consistency between the metrics that scientists obtain and the data that managers need.

While indicator development was important, many managers stated that having tools and approaches for transforming existing raw metrics into useful formats was equally important. There is a wealth of indicator data available for some systems, but these data are in difficult formats, or are not readily available, so this information can not be readily used. If greater access to information were available through indicator clearinghouses or similar vehicles, the data might be applied to a much broader set of problems and in a broader variety of ways than it has been in the past.

Agency officials also stressed the importance of communicating with stakeholders. Indicators must be presented to managers and decision-makers in a language they can understand and a format that they can use. Suites of indicators increased officials' ability to use a variety of different formats and approaches for communicating with stakeholders. Suggestions for improving the communication and presentation of indicator data included:

- Collect data for commonly used indicators *across* agencies,
- Use more visuals and graphics,

- Use color in reports and outreach,
- Use more maps and invest in GIS technologies,
- Establish an indicator clearinghouse accessible through the Internet,
- Build communication networks between scientists and managers, and
- Engage experts in both the natural sciences and the human dimensions of environmental behavior.

## Implications

Development and use of indicators has been most successful when an on-going dialog existed between scientists, managers, and stakeholders, and when all parties involved in water resource issues worked to communicate their data and knowledge in creative and audience-specific ways. Suites of indicators that describe not only condition, but also help diagnose the underlying factors or stressors contributing to that condition need to be provided to managers. Indicators are particularly useful for management when their information can be clearly and understandably communicated to the public.

# DESIGNING ENVIRONMENTAL INDICATOR SYSTEMS FOR PUBLIC DECISIONS

## Background

Information is needed not only for better management, but also for better public policy decisions. Societal choices have been made about the designated uses desired for aquatic resources. Indicators information can help inform the public on whether these uses are being attained and whether they can be attained.

In addition to interviews with managers on the use of environmental indicators, the Human Dimensions Team also examined factors (including laws, regulations, and policies) that affect the use of environmental indicators in public decisions. Many researchers assume if environmental indicators are scientifically valid, information from these indicators will be used in making public decisions.

## Findings

In general, there are three issues that affect indicator usefulness (McElfish and Varnell 2005). First, indicators must be relevant for the management purpose. In many instances, it is difficult to link indicators directly to management endpoints and purposes. These endpoints vary from general assessments of environmental condition to evaluating the effectiveness of individual permits. It is critical in designing effective environmental indicator systems to understand who the users are, what endpoints are being considered by management, which legal and jurisdictional constraints are applicable, and the technical sophistication of the user regarding application of indicators.

Second, the indicators must be appropriate for the geographic or spatial scale. For example, watershed-wide indicators might provide little guidance for management decisions related to development on an individual tract of land. Similarly, indicators for small, headwater streams might not be appropriate for assessing condition in the Susquehanna River.

Third, it is also important to consider the "delivery" system. How, when, where, and to whom will this information be provided? The delivery system must be capable of providing clearly understood and interpretable information when and where it is needed in the decision-making process.

## Implications

In designing suites of environmental indicators for public decisions, it is important to consider that:

- Indicators must provide information about specific endpoints used for management and policy decisions.
- Indicators must be appropriate for the geographic or spatial scale of the decision.
- Clear and interpretable indicator information must be able to be delivered to decision-makers when and where they need it.

# A NEW "FRONTIER" FOR ANALYZING ENVIRONMENTAL AND SOCIOECONOMIC INDICATORS

## Background

Managers indicated that information from both the natural sciences and social sciences (e.g., socioeconomic data and indicators) was important for environmental decision making, but how do you integrate these different types of data?

A key challenge of the ASC project was to develop methods for integrated assessment of the quality of life in alternative social choice contexts. Scientific assessments of quality of life face two fundamental challenges. One is to identify and measure the economic, social, and environmental factors that contribute to the determination of quality of life. A second is to aggregate across the alternative factors to produce a metric that can be used to assess conditions in different communities. *Frontier analysis* provides a method for aggregating across alternative factors.

Frontier analyses are approaches that have been used for economic analyses, but have not been previously used in the natural sciences. These techniques can be used to explore the efficiency with which quality of life is "produced" within assessment units such as counties, communities or watersheds (Marshall and Shortle 2005). Frontier analyses can be used for assessing not only the current quality of life in a community or county, but also the extent and direction of changes needed to achieve a feasible future quality of life defined by the performance of other communities or counties.

Quality of life indicators are broadly categorized into three dimensions – social, environmental, and economic. The concept underlying frontier analysis is that ecological, economic, and social constraints associated with a particular community define a maximum achievable quality of life for that community, and that this maximum achievable quality of life or frontier can be approximated by examining the performance of a community in achieving the maximum for various ecological, economic, and social factors. Each community (defined as counties here for a proof-of-concept) has a unique combination of attributes, and therefore a unique position along the continuum of possible values for these ecological and socioeconomic factors. A set of the counties will therefore form an outer boundary, or frontier, that defines the maximum achievable quality of life based on the combination of factors. The performance of those counties within the frontier can then be measured relative to the performance of those efficient counties that actually comprise the frontier (Figure 27) (Marshall and Shortle 2005).

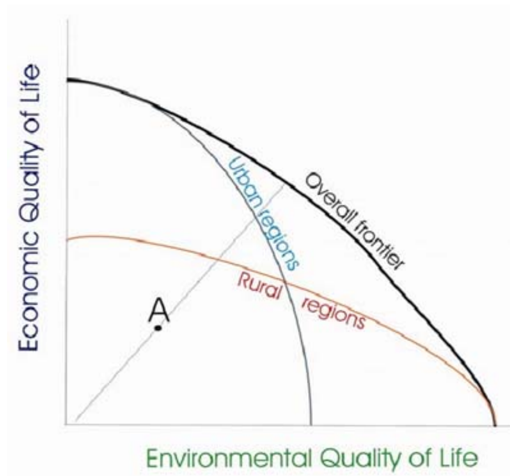



**Figure 27.** Frontier concept showing the distinction between rural and urban regions.

Data Envelopment Analysis (DEA) and Value Efficiency Analysis (VEA) are two statistical methods for integrating environmental, economic, and social indicators, such as those listed in Table 7 (Bowlin 1998, Charnes et al. 1994, Halme et al. 1999, Korhonen et al. 2001). DEA makes a weak, but reasonable, assumption that communities prefer to maximize "good" development outcomes (e.g., natural amenities, literacy, affordable living) and to minimize "bad" development outcomes (e.g., poverty, illiteracy, pollution). DEA evaluates

**Table 7. Variables included in quality of life model.**

| | |
|---|---|
| Environmental Dimension | EPA's cancer risk index (input)<br>% of land area developed (input) |
| Social Dimension | Teacher/pupil ratio (input)<br>% of population 25 and older who are high school graduates (input)<br># of arts, recreation, and entertainment establishments per developed square mile (output) |
| Economic Dimension | Median household income (output)<br>% of population below poverty level (input) |
| Non-Discretionary Amenity Variables | Amenity index (output) |

the relative efficiency of a community or county in maximizing the good outcomes and minimizing the bad outcomes. DEA provides a measure of that community or counties distance from the efficiency frontier (Figure 28). VEA is similar to DEA, but it permits the decision-maker to designate one community or county (i.e., production unit) as the "most preferred solution" (Figure 29). This most preferred solution, rather than the entire frontier, then becomes the standard against which all other communities or counties are compared. Because the quality of life in rural versus urban communities or counties reflects different social values, urban and rural counties were treated as separate subpopulations in the ASC DEA and VEA analyses.

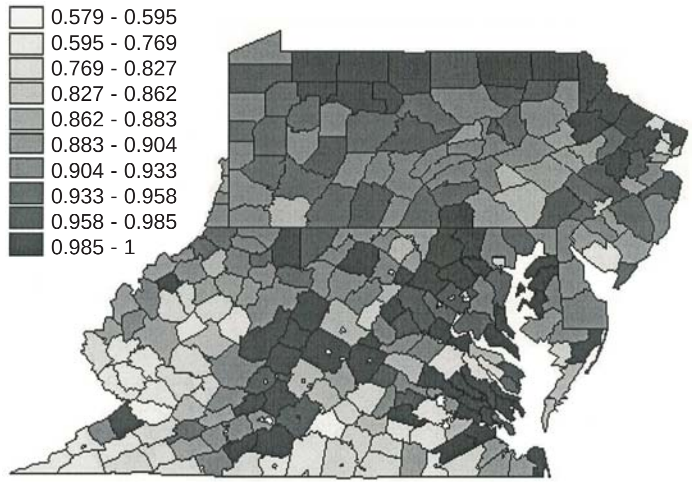



**Figure 28.** Range of DEA efficiencies for Mid-Atlantic counties, where the maximum efficiency is 1.0.

In this analysis we examined the relationships among undesirable outcomes (e.g., high cancer risk, percent of developed land, miles of impaired streams, and percent of population below the poverty level) and desirable outcomes (e.g., high teacher/pupil ratio, percent of population over 25 that are high school graduates, natural amenity index, miles/acres of wetlands, and miles of streams in good condition) in determining quality of life for the counties in the Mid-Atlantic region (Bloomquist et al. 1988, Deller et al. 2001, Marshall and Shortle 2005). We analyzed a subset of such indicators; information for each of these indicators in the ASC analyses were obtained from a number of different sources ranging from 1996 emissions data in EPA's National-Scale Air Toxics Assessment, to 2000 Census data, to the 2000 USDA Amenity Index.

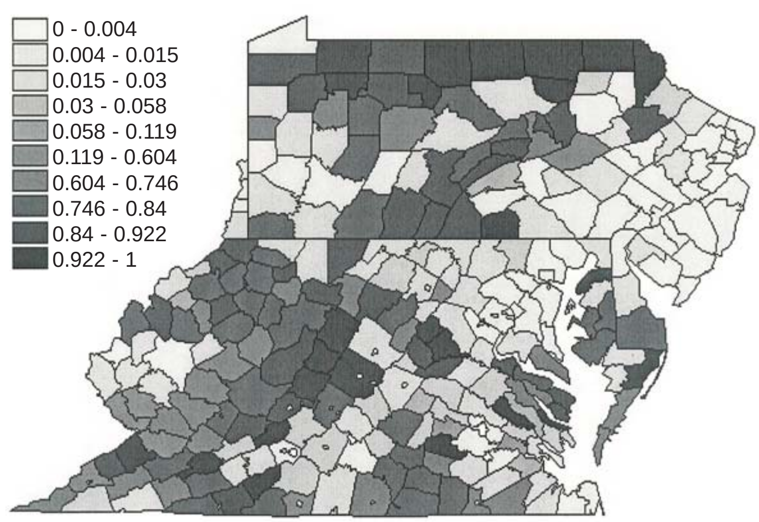



**Figure 29.** VEA efficiency scores when Amelia County, VA, is used as the reference county for urban counties and Floyd County, VA, is used as the reference county among the rural counties.

## Findings

DEA provided estimates of how a county performed with respect to a theoretical maximum frontier in producing quality of life in the Mid-Atlantic Region (Figure 28). The analyses provided a bench mark for "how far" a county was from the best that could be attained. Counties with high efficiency ratings, suggesting high quality of life, were scattered throughout the Mid-Atlantic Region, with the areas of lowest efficiency concentrated in West Virginia and Virginia. These counties tended to have low values for a number of indicators such as poverty level, percent high school graduates, and affordability.

VEA provided an estimate of where a county was with respect to what was considered the most preferred urban or rural county in the region. Based on population migration data, and the conjecture that people indicate preference by locational choices, Amelia County, VA was chosen as the reference county for urban counties and Floyd County, VA was chosen as the reference for rural counties. With VEA, the counties with low values were still scattered throughout WV, but a number of counties with low values were also found in eastern PA, MD, and central VA. VEA yielded a broad range of value efficiencies in the quality of life for Mid-Atlantic counties (Figure 27). VEA results, however, are highly sensitive to the reference county selected for comparison. If the VEA county selected as reference was highly unique, the scores of the remaining counties were far below their corresponding DEA scores. Despite this sensitivity, DEA and VEA provide a great deal of information about relative performance of counties in the production of quality of life, and where improvements could be obtained.

There were significant differences between rural and urban counties. In general, rural counties, when compared to their efficient frontiers, outperformed similar urban counties on the environmental dimension. Urban counties, when compared to their efficient frontiers, outperformed similar counties on the socioeconomic dimension. While it might appear that improved socioeconomic dimensions come at the expense of environmental dimensions (i.e., you can have good environmental or good economic condition, but not both), there is evidence that this is only true for areas with very high quality environmental conditions. Outside of very high quality natural environments, urban counties appear to make fewer environmental sacrifices in the achievement of economic development than do rural counties.

## Implications

Most counties in the Mid-Atlantic were below their potential for maximum achievable quality of life, but improved quality of life could be achieved in most counties. In general, rural counties out performed urban counties in the environmental dimension while urban counties outperformed rural counties in the socioeconomic dimensions. While some loss of environmental quality does occur in very high quality natural areas with economic development, it is possible to have both environmental and socioeconomic factors contributing to the quality of life in most Mid-Atlantic counties.

# HUMAN PERCEPTIONS VERSUS SCIENTIFIC ASSESSMENTS; AQUATIC ECOSYSTEMS AND QUALITY OF LIFE

## Background

The frontier analysis utilized data from secondary sources for integrating social, economic, and environmental factors to examine quality of life for county residents. A more fundamental approach is to ask residents questions about the relationship among environmental, economic, and social factors and human quality of life. A set of questions were formulated and asked through focus groups and sample surveys of residents living within the watersheds studied by ASC scientists. The focus groups and surveys addressed the following issues in the "human dimensions" of aquatic ecosystem indicators:

- What is the relationship between water quality and socio-economic indicators of quality of life?
- What is the relationship between *perceived* water quality and *perceived* quality of life—does water matter?
- How is water quality perceived by the public?
- What is the relationship between perceived and "actual" water quality (as provided by ecological scientists)
- What kinds of tradeoffs are people willing to make to obtain higher water quality (what elements, how good, and at what cost?)
- How are these relationships affected by baseline water quality conditions?

Focus groups were conducted in six different watersheds (Spring and Conodoguinet in PA; Gunpowder Falls and Southeast Creek, MD; and James and Ware Rivers, VA), with 53 participants in all. These focus groups provided insight into the background knowledge of watershed residents: their use of local water quality resources, the importance they place on water, their perceptions of local water quality, as well as factors that threaten water quality. These focus group results helped ensure that the mail surveys reflected local water-related concerns and issues.

A mail survey of residents of 9 watersheds that comprise a portion of watersheds studied in the larger Atlantic Slope project was implemented, and a total of 1,170 useable surveys were received.

While the focus group and resident surveys were being conducted, ASC scientists were assessing the quality of the watersheds they studied. The idea was to compare residents "perceived" water quality with scientific assessments of "actual" water quality. Because we were interested in examining both biological quality and recreational quality, scientists were asked to make judgments about both biological and recreational quality.

Because socioeconomic and demographic factors can affect the perception of quality of life, a comparison of these factors was made among watersheds. The mean age of respondents was similar across watersheds ranging from 48 to 54 years. Respondent sociodemographic characteristics were found to vary significantly across watersheds. Respondents differed strongly across watersheds in mean 2004 Household income (from a high of $93,535 in Conodoguinet Creek to a low of $42,772 in Clearfield Creek). Each of these figures is higher than that reported by Census data, suggesting that respondents were atypical of local residents. Clearfield Creek residents had the longest average residence time (24 years), more than twice that of Ware River respondents (10.8 years). Clearfield Creek residents had the lowest formal education of the respondents, with 55% having a high school diploma or less. In contrast, over 90% of Spring Creek, Conodoguinet Creek, Pamunkey River, and Chickahominy River respondents had at least some college education (Table 8).

**Table 8. Sociodemographic characteristics of respondents, by watershed.**

| | Mean Age | Mean Income | Mean Residence Tenure (in years) | % male | % with less than a High School Diploma | % with only High School Diploma | % with at least Some College | % with at least 2 year degree | % with at least 4 year degree | % with Advanced Degree (MS, PhD) |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | | | | | 7.5 | | |
| Spring Creek | 52 | $62,500 | 14.6 | 65.5 | 2.3 | 17.8 | 10.9 | 12.5 | 26.4 | 35.1 |
| Clearfield Creek | 54 | $42,772 | 24 | 67.8 | 7.1 | 48.2 | 15.2 | 18.2 | 13.4 | 3.6 |
| Ware River | 51 | $70,000 | 10.8 | 62.8 | 1 | 21.2 | 18.2 | 12.8 | 19.2 | 19.2 |
| Chickahominy River | 48 | $71,190 | 11.1 | 62.1 | 0 | 8.5 | 12.8 | 12.4 | 36.8 | 25.6 |
| Paminkey River | 53 | $92,200 | 13.7 | 76.1 | 2.1 | 11 | 17.2 | 15.3 | 30.3 | 24.1 |
| South East Creek | 54 | $70,741 | 16.4 | 58.8 | 7.3 | 22.6 | 16.1 | 7 | 19.4 | 18.5 |
| Conodoguinet Creek | 51 | $93,535 | 11.8 | 75.1 | 0.5 | 16.8 | 15.7 | 20.4 | 37.3 | 21.6 |
| Grindle Creek | 52 | $56,475 | 20.1 | 62.6 | 10.9 | 27.7 | 19.7 | 17.3 | 13.9 | 5.8 |
| Little Contentnea | 50 | $66,286 | 17.36 | 64.3 | 9.3 | 26.7 | 10.7 | | 26.7 | 8 |

A “*stated choice experiment*” was conducted to examine the value of improvements in water quality for recreational uses and biological integrity to watershed residents. For each watershed, average stated willingness to pay (WTP) per month per household was estimated for improvements in water quality that would affected 10% of the streams in the watershed.

## Findings

In general, most respondents rated water quality relatively low, with scores ranging from a high of 3.3 (Spring Creek watershed) to 2.3 (South East Creek watershed), with 1 being poor water quality to 6 being perfect water quality. Out of 9 watersheds, respondents in 6 watersheds indicated runoff from development was the greatest threat to water quality. Pollution from mining was the greatest perceived threat in Clearfield Creek watershed while pollution from agricultural chemicals was considered the greatest threat in Grindle Creek watershed. Perceived water quality was positively associated with respondents satisfaction with water recreation activities across all watersheds. It was the only factor in common with water recreation in all watersheds. A slightly above average quality of life was perceived in all watersheds, with the scores ranging from 3.6 to 4.0, on a six point scale from 1 = poor to 6 = perfect quality of life.

In the "stated choice experiment," willingness to pay to improve biological quality was higher than willingness to pay to improve recreation quality in all other watersheds except the Little Contentnea. The willingness to pay values for the Little Contentnea were not significantly different from zero. In general, watersheds with higher WTP values were associated with better quality watersheds.

Preliminary analyses suggest that the opinion of scientists about water quality differed from citizen perceptions. We are exploring these differences.

## Implications

Desired use of aquatic resources, such as recreation, affects public perception of the condition of the resource. Indicators of desired uses, therefore, can be as important as indicators of regulatory designated uses.

There is an apparent relationship between citizens' willingness to pay to maintain or improve aquatic resources in better quality watersheds. This has implications for watershed, stream, and wetland restoration. The public might be more willing to pay to maintain aquatic resources following restoration or improvement in the quality of these resources.

# VALUATION OF INFORMATION INVESTMENTS

## Background

Comprehensive approaches to aquatic ecosystem management require extensive information about existing conditions, threats to these conditions (e.g., development), and how conditions will respond to changes in these threats. Informed choices among alternative management strategies also require information on costs, societal goals, and tradeoffs. Given that information acquisition is costly, decisions are required about the types and amounts of information that should be sought.

A tool for guiding information investments examined in this project was the Expected Value of Information (EVOI). EVOI is a measure of the contribution that additional information makes to the outcome of decisions by reducing uncertainty.

The ASC project developed a procedure for estimating the expected value of information used for water quality management. The tool was demonstrated in case studies for reducing nitrogen loadings from crop land in the Pennsylvania portion of the Susquehanna River Basin (Borisova, 2004; Borisova et al., 2005; Sung, 2005; Ghosh, expected 2006).

## Findings

The case studies revealed that better information on the response of nitrogen loads to changes in farming practices, on the impacts of nitrogen loads on aquatic ecosystem conditions, on the economic benefits of increased ecological services, and on the costs of nutrient management practices all lead to improvements in the design of nutrient management policies as measured by a money metric of expected benefits of increased ecosystem services less the expected costs of nutrient reductions. The greatest gains from additional information come from investments in better understanding of ecosystem responses to changes in nutrient loads and the economic value of increased ecosystem services. The value of information used in aquatic ecosystem management is contingent on policy objectives, and the policy instruments used to achieve those policy objectives. For example, the value of information of all types was greater for a policy using quantity controls (e.g., nutrient credit trading) than for a policy using payments to farmers (or their inverse, charges) to induce adoption of nutrient management practices.

## Implications

Expected value of information tools can contribute to the selection of indicators used in the decision-making process. Value of information approaches may be an approach both for better quantifying the benefits of ecosystem services and communicating the importance of ecosystem services to the public.

# SUMMARY

The Atlantic Slope Consortium (ASC) was initiated with a goal of developing a set of indicators for coastal ecosystems that were ecologically appropriate, economically reasonable, and relevant to society to further inject science into policy and management decisions.

The indicators presented in this report were developed with the understanding that there can be conflicts among the cumulative individual *social choices* made within the watershed and *societal choices* made about the designated uses for our common aquatic resources.

Four primary messages emerged from this project as it conducted its research over the past four years:

1) A taxonomy for classifying indicators based on the type of questions they can answer, what spatial and temporal scale they reflect, and the *social choices* they address, helps resource managers choose indicators that are most appropriate for their use.

2) Estuarine fish and wetland bird community indicators, among others, conclusively demonstrate that both the amount of development and its proximity to the estuary or wetland contributes to the condition of these aquatic resources. In general, the greater the amount of development and its proximity, the greater the likelihood the resource would be degraded.

3) Strong linkages were found not only between the amount of development in small watersheds and its proximity to steams and wetlands on resource condition, but also the patterns of land use in these watersheds. For example, forest buffers interspersed along stream corridors or around a wetland had a significant effect in reducing sediment and nutrient loads to the aquatic resource.

4) Socioeconomic indicators can be combined with environmental indicators to show most communities in the Mid-Atlantic region do not have the quality of life that is possible, even when difference between rural and urban areas are considered.

# EPILOGUE

We have made significant progress toward our goal of developing suites of ecological and socioeconomic indicators for coastal ecosystems in the Mid-Atlantic region. Clearly, more work is needed, but we are encouraged that we have been able to demonstrate why we need to consider humans as part of, not apart, from ecosystems. Our research focused on small watersheds and estuarine bays. Additional field studies and modeling efforts are needed to link regional headwaters to estuaries. The contributions of large rivers to pollutant transport and processing, and how the estuaries are affected by these inputs, remains poorly understood and difficult to predict. In addition, we studied only a narrow range of salinities and depths across estuaries. We need to expand the range of salinities and depths. With the indicator taxonomy, every new indicator, as well as existing indicators, can be classified to ensure not only the full range of indicator types are available to managers, but also that each indicator is associated with the type of question being asked, its appropriate spatial and temporal scale, and that the indicator context is understood.

We think this is the future approach for the development of ecological indicators and their integration with social, economic, and political indicators. In applying this approach, informed policy and management decisions can be made, implemented, and explained to the public.

# LITERATURE CITED


Angermeir, P.L. and J.R. Karr. 1994. Biological integrity versus biological diversity as policy directives. *Bioscience* 44:690-697.

Arcement, G.J., Jr. and V.R. Schneider. 1989. Guide for selecting Manning's roughness coefficients for natural channels and flood plains. U.S. Geological Survey Water-Supply Paper 2339, U.S. Government Printing Office, Washington, DC.

Arner, D.H. and G.R. Hepp. 1989. Beaver pond wetlands: a southern perspective. in L.M. Smith, R.L. Pederson, and R.M. Kaminski, eds. Habitat Management for Migrating and Wintering Waterfowl in North America. Texas Tech University Press, Lubbock, Texas. p. 117-128

Azar, C., J. Holmberg, and K. Lindgren. 1996. Socio-ecological indicators for sustainability. *Ecological Economics* 18:89-112.

Baird D., R.R. Christian, C.H. Peterson, and G.A. Johnson. (2004) Consequences of hypoxia on estuarine ecosystem function: energy diversion from consumers to microbes. *Ecological Applications* 14:805-822.

Baker, M.E., D.E. Weller, and T.E. Jordan. 2006. Mapping watershed boundaries using digital elevation data: implications for landcover analysis of nutrient discharge. *Photogram. Engin. Rem. Sens.* 72.

Baker, M.E., D.E. Weller, and T.E. Jordan. In press. Improved methods for quantifying potential nutrient interception by riparian buffers. *Landscape Ecology.*

Baker, M.E., D.E. Weller, and T.E. Jordan. Submitted. Effects of stream map resolution on observed patterns of riparian buffer distribution and nutrient retention potential. *Landscape Ecology.*

Baker, M.E., D. E. Weller, and T. E. Jordan. In prep. a. Evaluating measures of riparian buffer configuration in geographic predictions of nutrient discharge. (in preparation)

Barbour, M.T., J. Gerritsen, B.D. Snyder, and J.B. Stribling. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrate and fish. 2nd ed. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Barden, L.S. 1987. Invasion of *Microstegium vimineum* (Poaceae), an exotic, annual, shade-tolerant, C-4 grass, into a North Carolina floodplain. *American Midland Naturalist* 118:40-45.

Bason, C. 2004. Effects of beaver impoundments on stream water quality in the coastal plain. M.S. Thesis, Department of Biology, East Carolina University, Greenville, NC.

Benke, A.C., I. Chaubey, G.M. Ward, and E.L. Dunn. 2000. Flood pulse dynamics of an unregulated river floodplain in the southeastern U.S. coastal plain. *Ecology* 81:2730-2741.

Bertness M.D., P.J. Ewanchuk, and B.R. Silliman. (2002) Anthropogenic modification of New England salt marsh landscapes. *Proceedings of the National Academy of Sciences (USA)* 99:1395-1398.

Bilkovic, D.M., C.H. Hershner, M.R. Berman, K.J. Havens, and D.M. Stanhope. 2004. Evaluating Nearshore Communities as Indicators of Ecosystem Health. in S. Bortone, ed. Estuarine Indicators. CRC Press, Inc. p. 365-379.

Blomquist, G.C., M.C. Berger, and J.P. Hoehn. 1988. New Estimates of the Quality of Life in Urban Areas. *American Economic Review* 78(1):89-107.

Boesch D.F., E. Burreson, W. Dennison, E. House, M.K. Kemp, R. Newell, K. Paynter, R. Orth, and R. Ulanowicz. 2001. Factors in the decline of coastal ecosystems. *Science* 293:1589-1591.

Boesch D.R. and J. Greer. Chesapeake Futures. Choices for the 21st Century. 2003. Chesapeake Bay Program Scientific and Technical Advisory Committee. Edgewater, MD.

Borisova, T. 2003. Coping with uncertainty in water quality management through choices of policy instruments and information investments. Ph.D. Dissertation, Agricultural, Environmental, and Regional Economics, The Pennsylvania State University, University Park, PA.

Borisova, T., J. Shortle, R. Horan, and D. Abler. 2005. The Value of Information for Water Quality Management. Water Resources Research. 41, W06004, doi:10.1029/2004WR003576.

Bowlin, W.F. 1998. Measuring Performance: An Introduction to Data Envelopment Analysis (DEA). *Journal of Cost Analysis* (Fall):3-27.

Breiman L., J.H. Friedman, R.A. Olshen, and C.J. Stone. Classification and Regression Trees. 1984. Monterey, CA, Wadsworth and Brooks/Cole.

Brinson, M.M., H.D. Bradshaw, and E.S. Kane. 1984. Nutrient assimilative capacity of an alluvial floodplain swamp. *Journal of Applied Ecology* 21:1041-1057.

Brinson, M.M., K. Miller, R.D. Rheinhardt, R.R. Christian, G. Meyer, and J.O'Neal. In prep. Development of Assessment Procedures for Restoration of Streams in Coastal North Carolina. A Report to the Ecosystem Enhancement Program, North Carolina Department of Environment and Natural Resources. Raleigh, NC.

Brooks, R.P. 1991. Restoration Ecology: Repairing the damage. in D.J. Decker, M.E. Krasny, G.R. Goff, C.R. Smith, and D.W. Gross, eds. Conserving Biological Resources: A practitioner's Guide. Westview Press, Boulder, CO. p. 67-81.

Brooks, R.P., D.H. Wardrop, and C.A. Cole. Inventorying and monitoring wetland condition and restoration potential on a watershed basis with examples from the Spring Creek watershed, Pennsylvania, USA. *Environmental Management* (in press).

Brooks, R.P., M.M. Brinson, R. Rheinhardt, K. Havens, D.O'Brien, J. Bishop, J. Rubbo, B. Armstrong, and J. Hite. A Stream, Wetland, Riparian (SWR) Index for assessing condition of aquatic ecosystems in small watersheds along the Atlantic Slope of the eastern U.S. (in preparation).

Brown, M.T. and M.B. Vivas. 2005. Landscape development intensity index. *Environmental Monitoring and Assessment* 101:289–309.

Butler, D.R. 1991. The reintroduction of the beaver to the South. *Southeastern Geographer* 31:39-43.

Chambers R.M., L.A. Myerson, and K. Saltonstall. (1999) Expansion of *Phragmites australis* into tidal wetlands of North America. *Aquatic Botany* 64:261-273.

Charnes, A., W. Cooper, A.Y. Lewin, and L.M. Seiford. 1994. Data Envelopment Analysis: Theory, Methodology, and Applications. Boston: Kluwer Academic Press.

Clarke, K.R. 1993. Non-parametric multivariate analyses of changes in community structure. *Austral. J. Ecol.* 18:117-143.

Cooper, J.R. and J.W. Gilliam. 1987. Phosphorus redistribution from cultivated fields into riparian areas. *Soil Science Society of America Journal* 51:1600-1604.

Cuffney, T.F. 1988. Input, movement and exchange of organic matter within a subtropical coastal blackwater river-floodplain system. *Freshwater Biology* 19:305-320.

Culliton T.J., M.A. Warren, T.R. Goodspeed, D.G. Remer, D.G. Blackwell, and J. MacDonough. 1990. 50 Years of Population Change Along the Nation's Coast, 1960-2010. NOAA Strategic Assessments Branch. Rockville, MD.

Dale, V.H. and S.C. Beyeler. 2001. Challenges in the development and use of ecological indicators. *Ecological Indicators* 1:3-10.

De'Ath G. and K.E. Fabricius. (2000) Classification and regression trees: A powerful yet simple technique for ecological data analysis. *Ecology* 81:3178-3192.

Deegan, L.A., J.T. Finn, S.G. Ayvazian, C.A. Ryder-Kieffer, and J. Buonaccorsi. 1997. Development and validation of an Estuarine Biotic Integrity Index. *Estuaries* 20(3):601-617.

DeLuca W.V., C.E. Studds, L.L. Rockwood, and P.P. Marra. (2004) Influence of land use on the integrity of marsh bird communities of Chesapeake Bay, USA. *Wetlands* 24: 837-847.

Deller, S.C., S.T. Tsai, D. Marcooouiller, and D.B.K. English. 2001. The Role of Amenities and the Quality of Life in Rural Economic Growth. *American Journal of Agricultural Economics* 83(2):352-365.

EPA U.S. 2000. Mid-Atlantic Highlands Streams Assessment. EPA/903/R-00/015. U.S. Environmental Protection Agency, Washington, DC.

EPA U.S. 2001. How Will Climate Change Affect the Mid-Atlantic Region? EPA/903/F-00/002. U.S. Environmental Protection Agency, Region 3, Philadelphia, PA .

EPA U.S. 2003. A SAB Report: A Framework for Assessing and Reporting on Ecological Condition. EPA-SAB-EPEC-02-009A. U.S. Environmental Protection Agency, Washington, DC.

Ely, E. 1998. Monitoring Wetlands. *The Volunteer Monitor* 10(1):1-27.

Faith, D.P., P.R. Minchin, and L. Belbin. 1987. Compositional dissimilarity as a robust measure of ecological distance. *Vegetatio* 69:57-68.

Gilliam, J. W. 1994. Riparian wetlands and water quality. *Journal of Environmental Quality* 23:896-900.

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones. *BioScience* 41:540-551.

Hale S.S., J.F. Paul, and J.F. Heltshe. (2004) Watershed landscape indicators of estuarine benthic condition. *Estuaries* 27: 283-295.

Halme, M., T. Joro, P. Korhonen, S. Salo, and J. Wallenius. 1999. A Value Efficiency Approach to Incorporating Preference Information in Data Envelopment Analysis. *Management Science* 45:1, 103-115.

Harmon, M.E., J.F. Franklin, F.J. Swanson, P. Sollins, S.V. Gregory, J.D. Lattin, N.H. Anderson, S.P. Cline, N.G. Aumen, J.R. Sedell, G.W. Lienkaemper, K. Cromack Jr., and K.W. Cummins. 1986. Ecology of coarse woody debris in temperate ecosystems. *Advances in Ecological Research* 15:133-302.

Hershner, C., K. Havens, D.H. Wardrop and D.M. Bilkovic. A practical concept for developing indicators of aquatic ecosystem health. *EcoHealth* (in press).

Hughes, J.E., L.A. Deegan, M.J. Weaver, and J.E. Costa. 2002. Regional application of an index of estuarine biotic integrity based on fish communities. *Estuaries* 25(2):250-263.

Hupp, C.R., M.D. Woodside, and T.M. Yanosky. 1993. Sediment and trace element trapping in a forested wetland, Chickahominy River, Virginia. *Wetlands* 13:95-104.

Jackson, L.A., J.C. Kurtz, W.S. Fisher, and eds. 2000. Evaluation guidelines for ecological indicators. EPA/620/R-99/005. U.S. Environmental Protection Agency, Office of Research and Development, Research Triangle Park, NC.

Jacoby, J.M., M.R. Seeley, E.B. Welch, and R.R. Horner. 1990. Responses of periphyton to changes in current velocity, suspended sediment, and phosphorus concentration. *Freshwater Biology* 42:215-232.

Johnston, C.A. 1991. Sediment and nutrient retention by freshwater wetlands: Effects on surface water quality. *Critical Reviews in Environmental Control* 21:491-565.

Jones, K.B., A.C. Neale, M.S. Nash, R.D. Van Remortel, J.D. Wickham, K.H. Riitters, and R.V. O'Neill. 2001. Predicting nutrient and sediment loadings to streams from landscape metrics: A multiple watershed study from the United States Mid-Atlantic Region. *Landscape Ecology* 16(4): 301–312.

Jordan, T.E., D.L. Correll, and D.E. Weller. 1993. Nutrient interception by a riparian forest receiving inputs from adjacent cropland. *Journal of Environmental Quality* 22:467-473.

Jordan, T.J., D.L. Correll, and D.E. Weller. 1997a. Nonpoint source discharges of nutrients from Piedmont watersheds of Chesapeake Bay. *J. Am. Water Res. Assn.* 33:631-645.

Jordan T.E., D.L. Correll, and D.E. Weller. 1997b. Effects of agriculture on discharge of nutrients from Coastal Plain watersheds of Chesapeake Bay. *Journal of Environmental Quality* 26: 836-848.

Jordan, T.J., D.L. Correll, and D.E. Weller. 1997c. Relating nutrient discharges from watersheds to land use and streamflow variability. *Water Resour. Res.* 33:2579-2590.

Jordan, T.J., D.E. Weller, and D.L. Correll. 2003a. Sources of nutrient inputs to the Patuxent River estuary. *Estuaries* 26:226-243.

Jordan T.E., D.F. Whigham, K.H. Hofmockel, and M.A. Pittek. 2003b. Nutrient and sediment removal by a restored wetland receiving agricultural runoff. *Journal of Environmental Quality* 32: 1534-1547.

Jordan, S.J. and P.A. Vass. 2000. An index of ecosystem integrity for Northern Chesapeake Bay. *Environmental Science and Policy* 3:S59-88.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6):21-27

Karr, J.R. and E.W. Chu. 1999. Restoring life in running waters: better biological monitoring. Island Press, Washington, DC. 206 pp.

Kelly, J.R. and M.A. Harwell. 1990. Indicators of ecosystem recovery. *Environmental Management* 14:527-545.

King R.S., M.E. Baker, D.F. Whigham, D.E. Weller, T.E. Jordan, M.K. Hurd, and P.F. Kazyak. 2005b. Spatial considerations for linking watershed land cover to ecological indicators in streams. *Ecological Applications* 15: 137-153.

King R.S., J.R. Beaman, D.F. Whigham, A.H. Hines, M.E. Baker, and D.E. Weller. 2004. Watershed land use is strongly linked to PCBs in white perch in Chesapeake Bay subestuaries. *Environmental Science and Technology* 38: 6546-6552.

King R.S., A.H. Hines, F.D. Craige, and S. Grap. 2005a. Regional, watershed, and local correlates of blue crab and bivalve abundances in subestuaries of Chesapeake Bay, USA. *Journal of Experimental Marine Biology and Ecology* 319: 101-116.

King, R.S., M.E. Baker, D.F. Whigham, D.E. Weller, P.F. Kazyak, M.K. Hurd. Watershed urbanization and ecological thresholds in streams: effects of physiography, watershed size, and spatial arrangement of impervious cover. (in preparation)

King, R.S., M.E. Baker, D.F. Whigham, D.E. Weller, T.E. Jordan, P.F. Kazyak, M.K. Hurd. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. *Ecol. Applic.* 15: 137-153.

Korhonen, P., A. Siljamake, and M. Soismaa. 1998. Practical Aspects of Value Efficiency Analysis. IIASA Interim Report No. IR-98-042. International Institute for Applied Systems Analysis. Laxenburg, Austria.

Leps J. and V. Hadincova. 1992. How reliable are our vegetation analysis? *Journal of Vegetation Science* 3: 119-124.

Liu, Z.-J., D.E. Weller, D.L. Correll, and T.E. Jordan. 2000. Effects of land cover and geology on stream chemistry in watersheds of Chesapeake Bay. *J. Am. Water Res. Assn.* 36:1349-1366.

Lowrance, R., L.S. Altier, J.D. Newbold, R.R. Schnabel, P.M. Groffman, J.M. Denver, D.L. Correll, J.W. Gilliam, J.L. Robinson, R.B. Brinsfield, K.W. Staver, W. Lucas, and A.H. Todd. 1997. Water quality functions of riparian forest buffers in the Chesapeake Bay Watershed. *Environmental Management* 21:687-712.

Lowrance. R.R., R.L. Todd, J. Fail, O. Hendrickson, R. Leonard, and L.E. Asmussen. 1984. Riparian forests as nutrient filters in agricultural watersheds. *BioScience* 34:374-377.

Marshall, E., and J. Shortle. 2005. Using DEA and VEA to Evaluate Quality of Life in the Mid-Atlantic States. *Agricultural and Resource Economics Review* 34(2):185-203.

McElfish, J.M., Jr. and L.M. Varnell. 2006. Designing environmental indicators for use in public decisions. *Columbia Journal of Environmental Law* 31(1):101.

McKenzie, D.H., D.E. Hyatt, and V.J. McDonald, eds. 1992. Ecological indicators. Proceedings of the International Symposium on Ecological Indicators; 16-19 October 1990; Fort Lauderdale, FL. Elsevier Science Publishers, Ltd. Esses, England.

Meador, M.R. and R.M. Goldstein. 2003. Assessing water quality at large geographic scales: relations among land use, water physiochemistry, riparian condition, and fish community structure. *Environmental Management* 31: 504-517.

Messer, J.J. 1992. Indicators in regional ecological monitoring and risk assessment. Pages 135-146 in McKenzie, D.H., D.E. Hyatt, and V.J. McDonald, eds. Ecological indicators. Proceedings of the International Symposium on Ecological Indicators; 16-19 October 1990; Fort Lauderdale, FL. Elsevier Science Publishers, Ltd., Esses, England.

Minchinton T.E. and M.D. Bertness. 2003. Disturbance-mediated competition and the spread of *Phragmites australis* in a coastal marsh. *Ecological Applications* 13: 1400-1416.

National Research Council. Clean Coastal Waters: Understanding and Reducing the Effects of Nutrient Pollution. 2000. Washington, DC, National Academy Press.

Nixon S.W. 1995. Coastal marine eutrophication: A definition, social causes, and future concerns. *Ophelia* 41: 199-219.

Noss, R.F. 1990. Indicators for monitoring biodiversity: a hierarchical approach. *Conserv. Biol*. 4(4):355-364.

Noss, R.F. 1999. Assessing and monitoring forest biodiversity: A suggested framework and indicators. *Forest Ecology and Management* 115:135-146.

Parker, I.M., D. Simberloff, W.M. Lonsdale, K. Goodell, M. Wonham, P.M. Kareiva, M.H. Williamson, B. Von Holle, P.B. Moyle, J.E. Byers, and L. Goldwasser. 1999. Impact: toward a framework for understanding the ecological effects of invaders. *Biological Invasions* 1:3-19.

Paul, J.F., and T.B. DeMoss. 2001. Integration of environmental indicators for the U.S. Mid-Atlantic Region. *Human and Ecological Risk Assessment* 7(5):1555-1564.

Peterjohn, W.T. and D.L. Correll. 1984. Nutrient dynamics in an agricultural watershed: Observations on the role of a riparian forest. *Ecology* 65:1466-1475.

Philipp K.R. and R.T. Field. 2005. *Phragmites australis* expansion in Delaware Bay salt marshes. *Ecological Engineering* 25: 275-291.

Poiani, K.A., B.D. Richter, M.G. Anderson, and H.E. Richter. 2000. Biodiversity conservation at multiple scales: Functional sites, landscapes, and networks. *BioScience* 50(2):133-146.

Rheinhardt, R., M.M. Brinson, R.P. Brooks, M. McKenney-Easterling, J.M. Rubbo, J. Hite, and B. Armstrong. 2006. Development of a reference-based method for identifying and scoring indicators of condition for Coastal Plain riparian reaches. *Ecological Indicators* (in press).

Rheinhardt, R., M. Brinson, R. Christian, G. Meyer, C. Bason, E. Hardison. 2005. Development of Ecological Assessments for Planning Coastal Plain Stream Restoration in Coastal North Carolina. Report submitted to North Carolina Ecosystem Enhancement Program, North Carolina Department of Environment and Natural Resources, Raleigh, NC.

Rheinhardt, R., M. Brinson, R. Christian, K. Miller, G. Meyer. 2004. Developing and calibrating an indicator for biogeochemical condition of headwater riparian ecosystems. Poster presented at annual EPA EaGLe's meeting in Duluth, MN, October 1, 2004.

Rheinhardt, R.D., M.M. Brinson, R.R. Christian, K.H. Miller, and G.F. Meyer. In review. A reference-based framework for evaluating the ecological condition of stream networks in small watersheds. *Wetlands* (in review).

Rice D., J. Rooth, and J.C. Stevenson. 2000. Colonization and expansion of *Phragmites australis* in upper Chesapeake Bay tidal marshes. *Wetlands* 20:280-299.

Saltonstall K. 2002. Cryptic invasion by a non-native genotype of the common reed, *Phragmites australis*, into North America. *Proceedings of the National Academy of Sciences (USA)* 99:2445-2449.

Scharf, J.T. 1881. History of Baltimore City and County. Louis H. Everts, Philadelphia, PA. 947 pp.

Seastedt, T.R., M.V. Reddy and S.P. Cline. 1989. Microarthropods in decaying wood from temperate coniferous and deciduous forests. *Pedobiologia* 33:69-77.

Seitzinger, S.P. 1994. Linkages between organic matter mineralization and denitrification in eight riparian wetlands. *Biogeochemistry* 25:19-39.

Sillman B.R. and M.D. Bertness. 2004. Shoreline development drives invasion of *Phragmites australis* and the loss of plant diversity on New England salt marshes. *Conservation Biology* 18:1424-1434.

Simberloff, D., I.M. Parker, and P.N. Windle. 2005. Introduced species policy, management, and future research needs. *Front. Ecol. Environ.* 3(1):12-20.

Snyder, C.D., J.A. Young, and R. Villella. 2003. Influences of upland and riparian land use patterns on stream biotic integrity. *Landscape Ecology* 18:647-664.

Soranno, P.A., S.L. Hubler, S.R. Carpenter, and R.C. Lathrop. 1996. Phosphorus loads to surface waters: A simple model to account for the spatial pattern of land use. *Ecological Applications.* 6:865–878.

Sung, H. and J. Shortle. 2006. The Value of Sample Information In an Integrated Watershed Based Water Quality Protection Program with Application to the Conestoga Watershed. Working Paper. Department of Agricultural Economics and Rural Sociology, Penn State University, University Park PA.

Thorp, J.H., E.M. McEwan, M.F. Flynn, and F.R. Hauer. 1985. Invertebrate colonization of submerged wood in a cypress-tupelo swamp and blackwater stream. *American Midland Naturalist* 113:56-68.

Urban D.L. 1987. Landscape ecology: A hierarchical perspective can help scientists understand spatial patterns. *BioScience* 37:119-127.

Vargo, S.M., R.K. Neely, and S.M. Kirkwood. 1998. Emergent plant decomposition and sedimentation: response to sediments varying in texture, phosphorus content and frequency of deposition. *Environmental and Experimental Botany* 40:43-58.

Walker, L.R. and S.D. Smith. 1997. Community response to plant invasion. in J.O. Luken and J.W. Thieret, eds. Assessment and Management of Plant Invasions. Springer. New York, NY. p. 69-86.

Wardrop, D.H., and R.P. Brooks. 1998. The occurrence and impact of sedimentation in central Pennsylvania wetlands. *Environmental Monitoring and Assessment* 51:119-130.

Wardrop, D.H., J.A. Bishop, M. Easterling, K. Hychka, W.L. Myers, G.P. Patil, and C. Taille. 2005. Use of landscape and land use parameters for classification and characterization of watersheds in the Mid-Atlantic across five physiographic provinces. *Environmental and Ecological Statistics*. 12:209-223.

Wardrop, D.H., C. Hershner, K. Havens, K. Thornton and D.M. Bilkovic. Developing and communicating a taxonomy of ecological indicators: a case study from the Mid-Atlantic. *EcoHealth* (in press).

Weller, D.E., T.E. Jordan, and D.L. Correll. 1998. Heuristic models for material discharge from landscapes with riparian buffers. *Ecol. Applic.* 8:1156-1169.

Weller, D.E., T.J. Jordan, D.L. Correll, and Z.-J. Liu. 2003. Effects of land use change on nutrient discharges from the Patuxent River watershed. *Estuaries* 26:244-266.

Williams, G.P. 1978. Bank-full discharge of rivers. *Water Resources Research* 14:1141-1154.

# ATLANTIC SLOPE CONSORTIUM RESEARCH MEETINGS

**2001**

| | |
|---|---|
| February 19 | STAR Grant awarded from U.S. Environmental Protection Agency |
| April 24 – 27 | EaGLe Project Directors Meeting, Pensacola, FL |
| May 8 – 9 | First ASC All Hands Meeting, University Park, PA |
| June 11 | ASC Human Dimensions Group Meeting, University Park, PA |
| June 19 | ASC Watershed Group Meeting, Edgewater, MD |
| July 5 | ASC Human Dimensions Group Meeting, University Park, PA |
| September 13 | ASC Estuary Group Meeting, Edgewater, MD |
| December 3 – 4 | First Annual EaGLes Meeting, Morehead City, NC |
| December 12 – 13 | Second ASC All Hands Meeting, Berkeley Springs, WV |

**2002**

| | |
|---|---|
| March 13 – 14 | Third ASC All Hands Meeting, Berkeley Springs, WV |
| November 13 – 14 | Fourth ASC All Hands Meeting, Williamsburg, VA |
| December 4 – 6 | Second Annual EaGLes Meeting, Edgewater, MD |

**2003**

| | |
|---|---|
| April 23 – 24 | Fifth ASC All Hands Meeting, Milford, PA |
| November 6 – 7 | Sixth ASC All Hands Meeting, Greenville, NC |
| December 3 – 5 | Third Annual EaGLes Meeting, Bodega Bay, CA |

**2004**

| | |
|---|---|
| March 24 – 25 | Seventh ASC All Hands Meeting, Lewes, DE |
| September 16 – 17 | Eighth ASC All Hands Meeting, Harper’s Ferry, WV |
| Sept. 30 – Oct. 1 | Fourth Annual EaGLes Meeting, Duluth, MN |

**2005**

| | |
|---|---|
| January 25 – 26 | Ninth ASC All Hands Meeting, Edgewater, MD |
| May 25 – 26 | Tenth ASC All Hands Meeting, Edgewater, MD |

**2006**

| | |
|---|---|
| January 26 – 27 | Eleventh ASC All Hands Meeting, Blackwater Falls, WV |

# THESES AND DISSERTATIONS

## Fully or Partially Supported by the Atlantic Slope Consortium

Balog, Amy. 2003. User perspectives on environmental indicators for aquatic ecosystems: Results from water quality officials in the Mid-Atlantic states. M.S. Thesis, Environmental Pollution Control, The Pennsylvania State University, University Park, PA.

Bason, Christopher. 2004. Effects of beaver impoundments on stream water quality in the coastal plain. M.S. Thesis, Department of Biology, East Carolina University, Greenville, NC.

Bishop, Joseph. A multi-scale watershed approach to habitat integrity and diversity in Pennsylvania. Ph.D. Dissertation (in progress), Ecology Program, The Pennsylvania State University. Partially supported by ASC.

Borisova, Tatiana. 2003. Coping with uncertainty in water quality management through choices of policy instruments and information investments. Ph.D. Dissertation, Agricultural, Environmental, and Regional Economics, The Pennsylvania State University, University Park, PA.

Cai, Y. 2003. Integrating environmental and socioeconomic indicators of quality of life assessment using data envelopment analysis. M.S. Thesis, Agricultural, Environmental, and Regional Economics, The Pennsylvania State University, University Park, PA. Partially supported by ASC.

DeLuca, William V. 2003. Multi-scale factors influencing marsh bird communities of the Chesapeake Bay: Developing a marsh bird community index. M. S. Thesis, George Mason University, Fairfax, Virginia.

Hychka, Kristen. 2005. Landscape indicators of riparian health: A case study in the Upper Juniata River watershed. M.S. Thesis, The Pennsylvania State University, University Park, PA. Partially supported by ASC.

Laubscher, Susan. 2005. Aquatic macroinvertebrates in Pennsylvania riverine systems: their ecology and utility as bioindicators of condition. M.S. Thesis, Ecology Program, The Pennsylvania State University, University Park, PA.

Saake-Blunk, Kristen. 2005. Integrating ecological assessments to evaluate the condition of the Spring Creek watershed, Centre County, Pennsylvania. Master of Forest Resources Thesis, The Pennsylvania State University, University Park, PA.

# ATLANTIC SLOPE CONSORTIUM - PUBLICATIONS


Baker, M.E., D. E. Weller, and T. E. Jordan. Improved methods for quantifying regional patterns of riparian buffers. Landscape Ecology (in review).

Baker, M.E., D. E. Weller, and T. E. Jordan. 2005. Mapping watershed boundaries using digital elevation data: implications for land cover analysis of nutrient discharge. Photogrammetric Engineering & Remote Sensing (in press).

Biber, P.D., Paerl, H.W., Gallegos, C.L., Kenworthy, W.J., Fonseca, M.S., 2005. Evaluating indicators of seagrass stress to light limitation in North Carolina. In *Estuarine Indicators*, Stephen Bortone (ed.). CRC Press, Inc, Washington, DC, pp. 193-209.

Bilkovic, D.M., C.H. Hershner, M.R. Berman, K.J. Havens and D.M. Stanhope. 2004. Evaluating nearshore communities as indicators of ecosystem health. In *Estuarine Indicators*, Stephen Bortone (ed.). CRC Press, Inc., Washington, DC, pp. 365-379.

Bilkovic, D.M., M. Roggero, C.H. Hershner and K.H. Havens. Influence of land use on macrobenthic communities in nearshore estuarine habitats. Estuaries (in review).

Borisova, T., J. S. Shortle, R.D. Horan, and D.G. Abler. 2005. The value of information for water quality protection. Water Resources Research 41(6), W06004, doi:10.1029/2004WR003576.

DeLuca, W.V., C.E. Studds, L.L. Rockwood, and P.P. Marra. 2004. Influence of land use on the integrity of marsh bird communities of Chesapeake Bay, USA. Wetlands 24: 837-847.

Gallegos, C.L. and P.D. Biber. 2004. Diagnostic tool sets water quality targets for restoring submerged aquatic vegetation in Chesapeake Bay. Ecological Restoration 22: 296-297.

Gallegos, C.L., Jordan, T.E., Hines, A.H., Weller, D.E., 2005. Temporal variability of optical properties in a shallow, eutrophic estuary: Seasonal and interannual variability. Estuarine, Coastal and Shelf Science 64: 156-170.

Havens, K., C. Hershner, D.M. Bilkovic and D.H. Wardrop. November 2004. Assessment of Chesapeake Bay Program selection and use of indicators. EcoHealth (in review).

Hershner, C., K. Havens, D.H. Wardrop and D.M. Bilkovic. A practical concept for developing indicators of aquatic ecosystem health. EcoHealth (in review).

Horan, R., and J. Shortle. 2005. When two wrongs make a right: second best point-nonpoint trading. American Journal of Agricultural Economics 87(2):340-352.

Horan, R., J. Shortle and D Abler. 2004. Point-nonpoint trading programs and agri-environmental policies. Agricultural and Resource Economics Review 33 (1): 61-78.

King, R. S. and C. J. Richardson. 2003. Integrating bioassessment and ecological risk assessment: an approach to developing numerical water-quality criteria. Environmental Management 31:795-809.

King, R. S., A. H. Hines, F. D.Craige, and S. Grap. 2005. Regional, watershed, and local correlates of blue crab and bivalve abundances in subestuaries of Chesapeake Bay, USA. Journal of Experimental Marine Biology and Ecology 319:101-116.

King, R. S., J. R. Beaman, D. F. Whigham, A.H. Hines, M.E. Baker, and D.E. Weller. 2004. Watershed land use is strongly linked to PCBs in white perch in Chesapeake Bay subestuaries. Environmental Science and Technology 38: 6546-6552.

King, R. S., M. E. Baker, D. F. Whigham, D. E Weller, T. E. Jordan, M. K. Hurd, and P. F. Kazyak. 2005. Spatial considerations for linking watershed land cover to ecological indicators in streams. Ecological Applications 15: 137-153.

Marshall, E. and J. Shortle. 2005. Using DEA and VEA to evaluate quality of life in the mid-Atlantic states. Agriculture and Resource Economics Review 34(2):185-203.

Marshall, E., and J. Shortle. 2005. Urban development impacts on ecosystems. In S. Goetz, J. Shortle, and J. Bergstrom (eds.), Land Use Problems and Conflicts: Causes, Consequences and Solutions. Routledge Publishing, New York.

McElfish, J. M., Jr. and L. M. Varnell. 2005. Designing environmental indicators for use in public decisions. Columbia Journal of Environmental Law 31 (1) (in press).

Niemi, G., D. Wardrop, R. Brooks, S. Anderson, V. Brady, H. Paerl, C. Rakocinski, M. Brouwer, B. Levinson, and M. McDonald. 2004. Rationale for a new generation of indicators for coastal waters. Environmental Health Perspectives 112(9): 979-986.

Patil G.P., R.P. Brooks, W.L. Myers, D.J. Rapport, and C. Taillie. 2001. Ecosystem health and its measurement at the landscape scale: Towards the next generation of quantitative assessments. Ecosystem Health 7(4): 307-316.

Patil, G. P., J. Bishop, W.L. Myers, C. Taillie, R. Vraney, and D. H. Wardrop. 2004. Detection and delineation of critical areas using echelons and spatial scan statistics with synoptic cellular data. Environmental and Ecological Statistics 11 (2): 139 – 164.

Patil, G. P., R. P. Brooks, W. L. Myers, and C. Taillie. 2002. Multiscale advanced raster map analysis system for measuring ecosystem health at landscape scale–A novel synergistic consortium initiative. In Managing for Healthy Ecosystems, D. Rapport, W. Lasley, D. Rolston, O. Nielsen, C. Qualset, and A. Damania, (eds.). CRC Press/ Lewis Publishers, Boca Raton, FL, pp. 567-576.

Rheinhardt, R., M.M. Brinson, R.P. Brooks, M. McKenney-Easterling, J.M. Rubbo, J. Hite, and B. Armstrong. 2006. Development of a reference-based method for identifying and scoring indicators of condition for Coastal Plain riparian reaches. *Ecological Indicators* (in press).

Rheinhardt, R., M. Brinson, R. Christian, K. Miller, G. Meyer, C. Bason, and E. Hardison. 2005. Development of Ecological Assessments for Planning Coastal Plain Stream Restoration in Coastal North Carolina. Report presented to the North Carolina Ecosystem Enhancement Program.

Wardrop, D.H., C. Hershner, K. Havens, K. Thornton and D.M. Bilkovic. Developing and communicating a taxonomy of ecological indicators: a case study from the Mid-Atlantic. EcoHealth (in review).

Wardrop, D.H., J.A. Bishop, M. Easterling, K. Hychka, W.L. Myers, G.P. Patil, and C. Taille. 2005. Use of landscape and land use parameters for classification and characterization of watersheds in the Mid-Atlantic across five physiographic provinces. Environmental and Ecological Statistics 12 (2): 209-223.

THE ATLANTIC SLOPE CONSORTIUM