Use of GIS to Study the Evolution of Environmental Change in a Large Estuarine System

B.L. Preston[1]

[1]**CSIRO Marine and Atmospheric Research, Aspendale, VIC, benjamin.preston@csiro.au**

Keywords: GIS, estuary, toxicity, water quality, biodiversity, climate change.

EXTENDED ABSTRACT

Geographic information systems (GIS) have been widely used for examining land/water interactions, such as changes in land cover in watersheds that may influence water quantity and quality in neighbouring surface waters. However, GIS has seen limited application in the analysis of stressor/response relationships within individual basins. The Chesapeake Bay estuary represents one of the largest, most productive, and most monitored estuarine systems in the United States and has undergone marked change over the past three centuries in response to anthropogenic activities in the watershed.

Since 1984, data collected via the Chesapeake Bay Program's (CBP) basinwide monitoring efforts have provided information on spatial and temporal changes in benthic biodiversity, water quality and temperature, and (to a lesser extent) concentrations of organic and inorganic toxicants in the sediment. The lack of a standardised sampling scheme across these variables, however, results in spatial and temporal inhomogeneities in the data which are a challenge for data analysis. GIS tools were used to circumvent these problems by aggregating and spatially relating different data sets.

The Shannon-Weaver (SW) index of biodiversity was used as an indicator of the health of the benthic community. Point observations of June/July/August (JJA) SW values were transformed to a percentile scale, and a spatial interpolation was performed to model benthic health throughout the basin (Figure 1), revealing sizeable areas of benthic impairment (classified as the lower 20th percentile for SW). When this same analysis was conducted over three different time periods (1987-91, 1992-96 and 1997-01), the extent of high impact areas decreased substantially, while low impact areas increased. Although significant differences were found in JJA water quality indicators between high and low impact areas (upper 20th percentile for SW), water quality could not explain these temporal changes in biodiversity. However, concentrations of sediment toxicants were both spatially and temporally associated with high impact areas. Multiple regression analysis among specific locations in the basin also suggested a significant influence of toxicants on SW values, although water quality variables also explain a significant fraction of observed variance.

Finally, a signal of climate change was readily identifiable in Chesapeake Bay temperature data. Although this does not appear to be a major driver at present, projected changes over the 21st century suggest climate change as an important emerging driver of environmental change in the estuary.

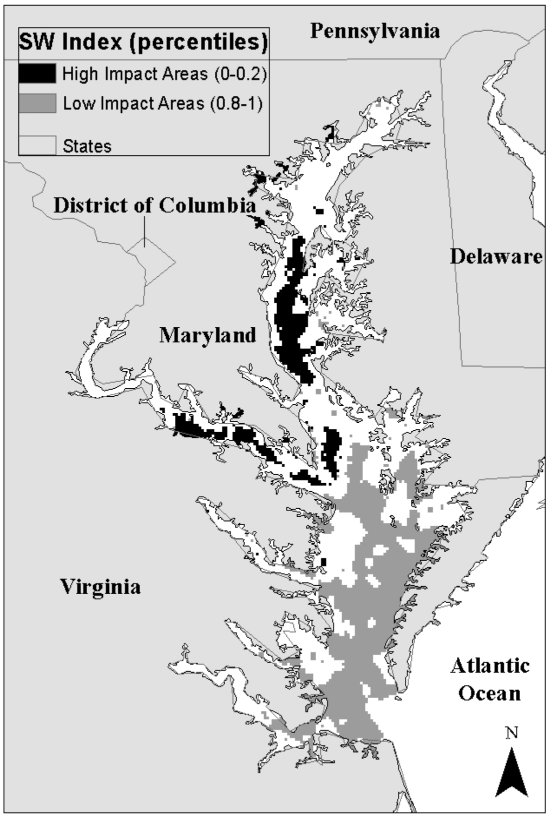

Figure 1. Spatial model of Shannon-Weaver (SW) biodiversity in the Chesapeake Bay. Values for SW (1987-2001; n=2687) were converted to a percentile scale and a spatial interpolation was performed using a second-order inverse distance-weighted algorithm.

1. INTRODUCTION

The Chesapeake Bay estuary is a large ecological system comprised of a primary basin with an areal extent of over 11,000 km^2, with a surrounding catchment area of approximately 175,000 km^2. A fingerprint of human influence on the estuary can be identified from the early 18th century, initiated by population growth and land clearance for agriculture (Willard et al. 2003). By the mid-20th century, rapid urbanisation and industrialisation within the watershed amplified land-use changes, but also introduced new stressors to the estuary in the form of anthropogenic toxicants, many of which accumulated in sediment. The current management challenge for the estuary is addressing nutrient loading and eutrophication, which contributes to significant hypoxia within the basin during warm months. Recent assessments of regional climate change in the Chesapeake Bay region point to an emerging driver that is likely to pose new challenges for the ecosystem.

Determining how these various drivers contribute to environmental outcomes and trends within the estuary is a significant challenge, particularly when one considers the latter 20th-century, when multiple anthropogenic drivers, specifically water quality, toxicants, and climate change, were acting simultaneously. A comprehensive environmental monitoring program, under the management of the Chesapeake Bay Program, has been in place in the Chesapeake Bay since the early 1980s, providing invaluable data regarding drivers and system response over an interesting period in the history of the estuary. However, due to spatial and temporal inhomogeneities in data collection, integrated analysis of different drivers and ecosystem responses have proven difficult. A number of studies have utilised geographic information system-based (GIS) approaches to resolve some of these inhomogenieties and elucidate how anthropogenic drivers may have and continue to interact to affect the Chesapeake Bay estuary. This paper reviews those studies in the larger context of evaluating how human/ecosystem interactions evolve over time.

2. BIOLOGICAL INDICATORS

Monitoring data from the Chesapeake Bay provide a continuous, but spatially heterogenous, view of the health of benthic biodiversity in the Chesapeake Bay since 1984. Several studies by Preston et al. have utilised the Shannon–Weaver index (SW) of benthic biodiversity to characterise Chesapeake Bay ecological health (Newman 1995; Preston 2002a, 2002b, 2004a; Preston and Shackelford 2002), and use of this indicator has been validated against a range of indicators (Preston 2002a). Data from 1984-2001 indicate that the overall trend in the Chesapeake Bay basin is toward increasing values of SW, indicative of an improvement in environmental health (MDNR, 2004; Preston 2004a), yet significant impairment of benthic habitat persists. Interpolation among 2,687 individual observations from fixed and single-use monitoring stations provides some indication of the spatial patterns associated with SW over this time period (Figure 1). The area corresponding with the lower 20th percentile for SW indicates the northern basin and lower extent of the Potomac River (downstream from the cities of Baltimore and Washington, respectively) as key areas of significant ecological impairment (see also MDNR, 2004). Meanwhile, the southern half of the basin is largely associated with high values for SW (upper 20th percentile), suggesting limited impairment. Preston (2004a) examined how this spatial pattern has changed over time, finding that when one compared the period 1987-91 with 1997-01, the extent of high-impact areas decreased by 44%, while low-impact areas increased by over 300%. These spatial patterns and trends in biological diversity are indicative of a system that has been differentially stressed, but one where one or more drivers have changed within the last 20-30 years, resulting in some improvement in biological indicators. This finding is novel, and one that has been overlooked by regulatory agencies relying solely upon a declining number of diffuse point observations to assess biodiversity (MDNR, 2004). Accounting for this pattern of environmental change requires evaluation of the various drivers that have affected the system.

3. EVALUATION OF ENVIRONMENTAL DRIVERS

Monitoring data from the Chesapeake Bay are available for a broad range of physical and chemical variables as well as sediment toxicant concentrations. These data enable the exploration of how different drivers have contributed to observed patterns and trends in benthic biodiversity. The three drivers considered here are water quality, sediment toxicants, and climate change. Ten different water quality parameters, water temperature, and 60 different toxicants were selected as indicators of these drivers. Water quality and temperature data were derived from samples collected during summer months (June/July/August) between 1987 and 2001 from 152 fixed and random monitoring stations located throughout the basin, resulting in 205 to 2,295 observations per parameter. All water quality data were collected in compliance with the Chesapeake

Table 1. Comparison of mean benthic water quality parameters between high-impact and low-impact areas of Chesapeake Bay, USA during summer months (May-September; 1987-2001). n=number of observations per parameter per area. SE represents the standard error of the mean. High/Low indicates the ratio of mean concentration in the high impact areas to the low impact areas. * indicates significant difference ($p<0.05$) by two-sided t-test. † indicates significant difference ($p<0.0001$) by two-sided t-test.

Water Quality Variable	High Impact Areas		Low Impact Areas		Ratio
	n	Mean ± SE	n	Mean ± SE	High/Low
Dissolved oxygen (mg/L)	1028	4.2 ± 0.1	786	5.8 ± 0.07	0.72†
Salinity (g/L)	1027	12.4± 0.2	799	21.9 ± 0.2	0.57†
pH	1289	7.6 ± 0.01	968	7.9 ± 7.9	0.96†
Total Suspended Solids (mg/L)	1296	14.7 ± 0.4	1002	22.1 ± 22.1	0.67†
Conductivity (μmhos/cm)	1291	21022.2 ± 294.1	1004	34689.2 ± 185.5	0.61†
Turbidity (NTU)	39	7.2 ± 1.2	166	16.0 ± 0.9	0.45†
Chlorophyll-A (mg/L)	863	14.4 ± 1.0	441	6.3 ± 0.3	2.27†
Dissolved organic carbon (mg/L)	764	3.4 ± 0.07	337	2.6 ± 0.04	1.30†
Total nitrogen (mg/L)	1118	1.0 ± 0.02	853	0.5 ± 0.01	1.85†
Total phosphorus (mg/L)	1144	0.07 ± 0.002	853	0.06 ± 0.001	1.20†
Water Temperature (^{o}C)	1292	22.2± 0.1	1005	22.6 ± 0.1	0.98*

Bays Program's standard methods and quality assurance/quality control management plan (Preston 2002a). Sediment toxicant data were collected from 485 monitoring sites, resulting in 192 to 756 observations among 60 different toxicants: 10 metals, 12 pesticides, 20 polycyclic aromatic hydrocarbons (PAHs), and 18 polychlorinated biphenyl (PCB) congeners. Due to spatial inhomogenities in monitoring stations, water quality, toxicant, and temperature data were related to biological indicators through two different GIS-based data manipulations. First, those areas corresponding to the lower and upper 20th percentiles for SW were converted to polygon features in GIS, which were subsequently used to select water quality, toxicant, and temperature data corresponding with these respective regions. These data were subsequently analysed and aggregate results from these broad regions were compared. The second approach integrated water quality, toxicant, and temperature data on a site-specific basis. GIS tools were used to link water quality, sediment toxicant, temperature, and biological indicators at 485 sites in the basin, which corresponded with toxicant monitoring sites. Values for SW were assigned using the aforementioned spatial model. Values for water quality and temperature data were assigned by assuming values at the 485 sites corresponded with values at the nearest neighbouring water quality monitoring site (see Preston and Shackelford 2002; Preston 2004a).

3.1. Water Quality Variables

Comparison of 10 different water quality parameters between low and high impact areas revealed significant differences in all cases (Table 1). High impact areas had elevated phosphorus and nitrogen concentrations coincident with higher chlorophyll-a concentrations and reduced dissolved oxygen. Collectively, these indicators suggest eutrophication as a potential factor driving spatial patterns of biodiversity. In addition, high impact areas were also associated with lower salinity, suspended solids, and turbidity, the causes and implications of which were less clear, and are likely related to a combination of water inflows, anthropogenic activity, and inherent estuary physical/chemical gradients.

Site-specific evaluation of water quality variables also reflected this high degree of spatial association with benthic biodiversity. Among the 485 sites, chlorophyll-a, nitrogen concentrations and dissolved oxygen showed the highest correlation with site-specific biodiversity values. However, salinity, pH, and conductivity were also correlated, suggesting the influence of natural variability on spatial patterns of biodiversity. A multiple regression among nine water quality variables (turbidity excluded due to low sample size) resulted in an explained variance in benthic biodiversity of approximately 52% ($n=205$; $r^2=0.52$; $p<0.0001$).

Comparison of temporal trends among these variables within the high impact areas suggested little that could explain observed temporal changes in spatial patterns of biodiversity. Mean values for each indicator were calculated for three time periods, 1987-1991; 1992-1996; and 1997-2001. When compared to the 1987-2001 mean, chlorophyll-a and dissolved organic carbon concentrations were observed to increase over time. Remaining variables demonstrated high temporal variability, with values during the middle time period, 1992-1996, often higher or lower than those for the other two.

3.2. Toxicant Variables

Comparison of sediment toxicant concentrations in the low and high impact areas revealed that all toxicants were elevated in the high impact areas, often by an order of magnitude. These elevated levels were statistically significant for all 10 of the metals, 2 of the 12 pesticides, and 7 of the 20 PAHs. These results suggest that like water quality variables, toxicant concentrations exhibit a high degree of spatial association with benthic biodiversity. As a further test of this association, Preston (2002b) assessed how gradients in sediment toxicant concentrations vary around SW high impact areas. Concentrations for the majority of the toxicants for which data were available decreased significantly with increasing distance from the high-impact areas. Furthermore, of the 18 toxicants listed by the CPB as “toxics of concern,” 15 (83%) were also identified as priority contaminants based simply upon the strength of spatial gradients associated with impaired benthic biodiversity (Preston 2002b).

A further test of the role of toxicants in influencing benthic biodiversity was performed via a screening-level ecotoxiocological risk assessment. Field et al. (2003) published logistic models of the relationship between 37 organic and inorganic contaminants and the response of invertebrate toxicity tests, based upon robust data sets drawn from coastal and estuarine ecosystems. Toxicant data from the Chesapeake Bay were selected for the 37 toxicants included in Field et al. (2003), and classified into one of two categories, depending upon whether the observation was made within low- or high-impact areas of Chesapeake Bay (Figure 1). Concentrations of samples below detection limits were treated as zero values. For PCB congeners, irregular sampling and analysis prevented meaningful quantification of PCB concentrations useful for estimating toxicant effects. For the remaining toxicants, within each category, the 50th and 90th percentile sediment concentration (SC50 and SC90, respectively) was calculated for each contaminant based upon cumulative probability distributions. SC50s and SC90s were then used as input parameters into the logistic models utilised by Field et al. (2003). Model outputs represented the estimated probability of sediment toxicity for each contaminant within the low- and high-impact areas.

Results for metals are presented in Table 2, and indicate that sediment concentrations of anthropogenic contaminants in Chesapeake Bay observed between 1987 and 1999 were sufficiently high to present a significant risk to benthic invertebrates. However, risk varied among different classes of contaminants. For example, the risk associated with pesticides was generally low, with negligible risk of toxicity at median (SC50) concentrations and low risk (~20%) at SC90s. Although the risk of toxicity at SC50s was also low for metals and PAHs, risks increased sufficiently at SC90s for toxic effects to be a reasonably likely (20-60%) occurrence in 10% of samples. Metals were observed to have the highest risk, averaging a 60% chance of toxicity at the SC90.

As with water quality indicators, site-specific evaluation of toxicant concentrations reflected a high degree of spatial association with benthic biodiversity. Among the 485 sites, PCBs, cadmium and copper showed the highest correlation with SW (-0.47, -0.37, and -0.35, respectively). A range of other organic and inorganic toxicants were also

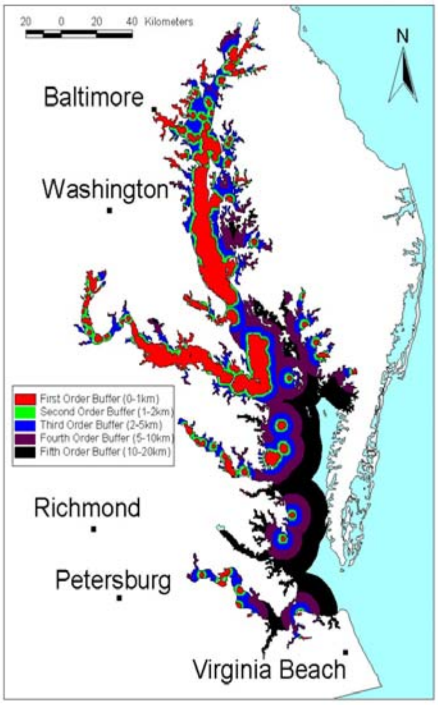

Figure 2. Illustration of spatial buffers around high impact areas (Preston, 2002b).

Table 2. Estimated concentrations of metals in Chesapeake Bay (1987-1999), and associated probability of toxicity based upon Field et al. (2002). n represents the number of samples for each contaminant. SC50 and SC90 represent the 50th and 90th percentile concentrations (mg/kg dry weight) in the low and high-impact areas, and P represents the probability of toxicity expressed as a percent.

CONTAMINANT	Low-Impact Areas					High-Impact Areas				
	n	SC50	P(%)	SC90	P(%)	n	SC50	P(%)	SC90	P(%)
Antimony	36	0.09	3	7.00	76	64	0.56	18	2.47	51
Arsenic	51	3.80	9	9.00	25	89	11.10	30	31.10	64
Cadmium	53	0.06	3	0.24	13	89	0.62	30	2.30	64
Chromium	53	24.20	9	43.70	18	89	77.70	32	282.77	71
Copper	53	12.00	7	31.00	19	89	35.30	22	173.00	68
Lead	53	15.80	11	35.40	24	89	33.00	22	149.00	64
Mercury	55	0.00	0	0.08	12	89	0.08	12	0.32	39
Nickel	53	11.30	16	23.00	30	89	37.00	43	75.60	64
Silver	40	0.00	0	0.09	10	66	0.24	21	0.74	41
Zinc	53	57.60	11	200.00	43	89	169.00	37	497.00	74

significantly correlated with benthic biodiversity, although these correlations tended to be weaker than water quality indicators. A multiple regression among concentrations for the 10 metals, and total PCB, PAH, and pesticide concentrations resulted in an explained variance in benthic biodiversity of approximately 47% (n=126; r^2=0.47, p<0.0001).

The spatial pattern of toxicity must also be considered in conjunction with temporal trends in sediment toxicant concentrations. Preston (2004a) analysed changes in sediment concentrations over three time periods: 1987-1991; 1992-1996; and 1997-1999; relative to the 1987-1999 mean. Results indicated a substantial reduction in sediment concentrations over this time period, either due to burial via sedimentation (reducing bio and ecosystem availability) or degradation. The

largest reductions were seen for cadmium, mercury, silver, and the total pesticide, PAH, and PCB concentrations. These reductions averaged 50% for metals, 90% for PAHs, 75% for PCBs, and 80% for pesticides. Generally, these reductions in sediment concentrations were comparable to observed differences in SC90s between low- and high-impact areas, and thus reductions of 50-90% in SC90s over time may contribute to a significant reduction in the net impact of toxicity on benthic organisms.

3.3. Climate Change

An assessment of climate variability and change in the Chesapeake Bay region indicated surface air temperatures in the region increased over the 20th century (MARA Team, 2001). Preston (2004b) utilised monitoring data from the Chesapeake Bay to quantify the effects of climate change on water temperatures in the surface (<15 m) and subsurface (>15m) of the basin. Long-term (1949-2002) trends of 0.16°C/decade and 0.21°C/decade were calculated for the surface and subsurface, respectively, suggesting net late-20th century warming of approximately 0.8-1.0°C.

There does not appear to be any significant relationship between this warming trend and spatial or temporal patterns in benthic biodiversity. Although both temperatures and biodiversity suggest positive trends, correlation among annual mean temperatures in the basin and observed mean SW over the period 1984-2002 is poor. Furthermore, Preston (2004b) found a small difference in mean benthic JJA temperature between the low and high impact areas of Chesapeake Bay (22°C vs. 23°C, respectively), but this is unlikely to be ecologically significant. However, on a site-specific basis, log-transformed temperatures ranked higher than all other water quality variables except chlorophyll-a, with respect to their correlation with benthic biodiversity among 485 sites, although, this correlation remained poor (0.12) (Preston 2004b). Preston and Shackelford (2002) also found that including temperature with other water quality variables in a multiple regression increased model fit with respect to explaining spatial variance in benthic biodiversity, yet using more recent data (Preston, 2004), found temperature to be an insignificant variable.

Preston (2004b) also revealed that there has been spatial variability in Chesapeake Bay warming trends over the past half century. Two latitudinal transects were drawn at equal intervals across the basin (38.7 and 37.8 N), resulting in three different regions. The three regions all reflected the warming trends observed for the basin as a whole, but there was a north-south gradient in warming with warming trends increasing from north to south among both surface and subsurface observations (Figure 3). Warming trends in the southern-most region were 0.20-0.23^{o}C/decade compared with 0.11-0.13^{o}C/decade in the northern-most region.

4. ASSEMBLING THE COMPONENTS

The availability of a broad range of spatially-referenced data on the physical, chemical, and biological condition of the Chesapeake Bay enables the testing of a range of hypotheses regarding the relationship between different drivers and the biological response of the ecosystem. The big picture is one of declining and emerging ecosystem drivers in conjunction with successful management of toxicants, unsuccessful (at least to date) management of nutrient loading, and the lack of management of climate change.

For the past decade, the primary issue in the management of the Chesapeake Bay has been nutrient loading and eutrophication, which has been linked with persistent hypoxia in the benthic zone and a deterioration of the benthic environment (MDNR, 2004). Various initiatives have been launched in the watershed, which have failed to check the flux of nutrients to the estuary. The analyses conducted here confirm the presence of reduced JJA dissolved oxygen in various regions of the basin, coincident with areas of impaired benthic habitat. These analyses also reveal a marked increase in measures of biodiversity over time, without a coincident change in dissolved oxygen. Thus, observed temporal patterns in benthic biodiversity cannot be satisfactorily explained by nutrient management efforts and oxygen dynamics.

In contrast, changes in sediment toxicant concentrations agree well with both spatial and temporal patterns of benthic biodiversity. They occur in concentrations sufficient to cause toxicity, and there are spatial and temporal trends in toxicant concentrations consistent with biodiversity. Large declines occur in a range of key toxicants post 1991, which coincides with the culmination of a range of regulatory measures to reduce toxicant releases to the environment during the 1980s. This suggests that reductions in toxicants within the basin have had a direct influence on benthic biodiversity. Subsequently, although toxicants appear to have been a significant driver of benthic biodiversity in the basin over the past 30 years, their influence has significantly declined, and eutrophication has become a major driver affecting further recovery and the future status of the basin.

Last, time-series data reveal that a signal of climate change is already apparent within the Chesapeake Bay. Several assessments have discussed the potential implications of observed and future climate change for the Chesapeake Bay ecosystem, and there is some evidence of community shifts in response to climate variability and change. Furthermore, the general consequences of temperature increases, such as reduced dissolved oxygen, would appear to interact with ongoing eutrophication challenges. Yet, the analyses here suggest that the long-term climate signal has yet to have a discernable influence on benthic biodiversity. It remains to be seen whether the influence of climate emerges as a major driver of future change in the Chesapeake Bay, whether the bay proves resilient to this forcing, or whether other drivers simply dominate. It also remains to be seen how human agency will affect the interaction between climate and the Chesapeake Bay ecosystem (Preston, 2004b).

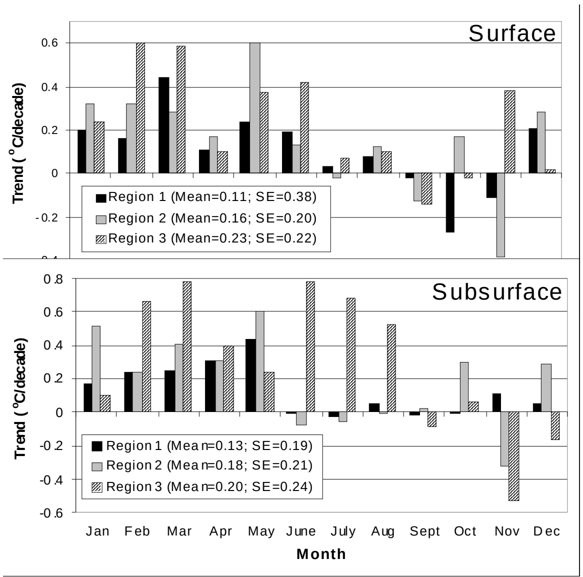

Figure 3. Monthly trends in surface (top) and subsurface (bottom) temperatures in the Chesapeake basin (1949-2002). Trend analysis was conducted on three different regions, revealing a north (Region 1)/south (Region 3) gradient.

Rather than focus on a particular driver or management challenge (e.g., eutrophication and hypoxia), the studies described here attempted to build a broader understanding of how different drivers have influenced the bay over time, using a biological indicator of a key ecological community as a measurement endpoint. As a result, it is possible to identify the well-documented effect of water quality on the health of the estuary, which appears to persist despite management efforts. Yet, these studies also reveal that attempts to manage toxicants have been far more successful, with discernable benefits for the health of the estuary's benthic community.

5. GIS IN ENVIRONMENTAL ASSESSMENT

Geographic information, modelling, and simulation systems are increasingly common tools in environmental assessment. The work described here utilised geographic analysis tools as a means of organising historical monitoring data for analysis and diagnosis, rather than for predicting current or future status of the estuary. In particular, GIS was used to address two challenges in the analysis of monitoring data: a) the exploration of spatial relationships that may provide information on stressor/response relationships, hazard identification, and attribution and b) the resolution of spatial inhomogeneities in historical monitoring efforts that leave the original data poorly suited for comprehensive analysis and communication. The latter of these challenges, in particular, may explain why the copious amounts of monitoring data that exist within public and private institutions are underutilised. Data gaps and discontinuation of programs often make rigorous analysis quite difficult. This is alleviated in the Chesapeake Bay, due simply to the volume of data that is available. Support for a small, but continuously monitored network of water quality stations in the basin will ensure more meticulous record keeping with higher temporal resolution, although the spatial resolution needed for the investigation of certain questions will continue to be a challenge. Meanwhile, monitoring of toxicant concentrations in the basin largely terminated in 1999, preventing the analyses described here from being carried forward in time.

Although simulations are invaluable for exploring the sensitivity of complex environmental systems to different drivers and management decisions, these efforts must be integrated with more traditional analysis of observational data to confirm successful management efforts, identify failures, and maintain an accurate understanding of the net status of the system. In this regard, comprehensive monitoring remains invaluable.

6. REFERENCES

Field L.J., D.D. MacDonald, S.B. Norton, C.G. Ingersoll, C.G. Severn, D. Smorong, and R. Linkskoog (2003), Predicting amphipod toxicity from sediment chemistry using logistic regression models, *Environmental Toxicology and Chemistry* 21, 1993-2005.

MARA Team (Mid-Atlantic Regional Assessment Team) (2001), *Preparing for a Changing Climate: The Potential Consequences of Climate Variability and Change. Mid-Atlantic Foundations*, U.S. Global Change Research Program, Washington, DC, USA.

MDNR (Maryland Department of Natural Resources) (2004), *Chesapeake Bay Water Quality Monitoring Program: Long-Term Benthic Monitoring and Assessment Component – Level I Comprehensive Report, July 1984-December 2003 (Volume 1)*, R.J. Llansó et al. (eds), Versar, Inc., Colombia, Maryland, USA.

Newman, M.C. (1995), *Quantitative Methods in Aquatic Ecotoxicology*, CRC, Boca Raton, FL, USA.

Preston, B.L. (2002a), Spatial patterns in benthic biodiversity of Chesapeake Bay (1984-1999): Association with water quality and sediment toxicity, *Environmental Toxicology and Chemistry* 21, 151-162.

Preston, B.L. (2002b), Hazard prioritization in ecological risk assessment through spatial analysis of toxicant gradients, *Environmental Pollution* 117, 431-445

Preston, B.L. (2004a), GIS-based localization of impaired benthic communities in Chesapeake Bay: Associations with indicators of anthropogenic stress, In L.A. Kapustka et al. (eds), *Landscape Ecology and Wildlife Habitat Evaluation: Critical Information for Ecological Risk Assessment, Land-Use Management Activities, and Biodiversity Enhancement Practices*, *ASTM STP 1458*, ASTM International, West Conshohocken, PA, pp. 221-241.

Preston, B.L. (2004b), Observed winter warming of the Chesapeake Bay estuary (1949-2002): Implications for ecosystem management, *Environmental Management* 34, 125-139.

Preston, B.L., and J. Shackelford (2002), Multiple stressor effects on benthic biodiversity of Chesapeake Bay: Implications for ecological risk assessment, *Ecotoxicology* 11, 85-99.

Willard, D.A., T.M. Cronin, and S. Verardo (2003), Late-Holocene climate and Ecosystem History from Chesapeake Bay sediment cores, USA, *The Holocene* 13, 201-214.