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Relationships between watershed stressors and sediment contamination in Chesapeake Bay estuaries

Randy L. Comeleo[1], John F. Paul[2], Peter V. August[3], Jane Copeland[1], Carol Baker[3], Stephen S. Hale[1,2] and Richard W. Latimer[2]
[1]R.O.W. Sciences, Inc., c/o U.S. Environmental Protection Agency, Atlantic Ecology Division, Narragansett, RI 02882, USA; [2]U.S. Environmental Protection Agency, Atlantic Ecology Division, Narragansett, RI 02882, USA; [3]Department of Natural Resources Science, University of Rhode Island, Kingston, RI 02881, USA

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Abstract

Three methods for assessing the relationships between estuarine sediment contaminant levels and watershed stressors for 25 Chesapeake Bay sub-estuaries were compared. A geographic information system (GIS) was used to delineate watersheds for each sub-estuary and analyze land use pattern (area and location of developed, herbaceous and forested land) and point source pollution (annual outflow and contaminant loading) using three landscape analysis methods: (1) a watershed approach using the watershed of the estuary containing the sampling station, (2) a 'partial watershed' approach using the area of the watershed within a 10 km radius of the sampling station and (3) a 'weighted partial watershed' approach where stressors within the partial watershed were weighted by the inverse of their linear distance from the sampling station. Nine sediment metals, 16 sediment organics and seven metals loading variables were each reduced to one principal component for statistical analyses. Relationships between the first principal components for sediment metals and organics concentrations and watershed stressor variables were analyzed using rank correlation and stepwise multiple regression techniques. For both metals and organics, the watershed method yielded R^2 values considerably lower than the partial and weighted partial watershed analysis methods. Regression models using stressor data generated by the weighted partial watershed landscape analysis method explained 76% and 47% of the variation in the first principal component for sediment metals and organics concentrations, respectively. Results suggest that the area of developed land located in the watershed within 10 km of the sediment sampling station is a major contributing factor in the sediment concentrations of both metals and organics.

1. Introduction

The U.S. Environmental Protection Agency (EPA), in cooperation with other Federal agencies that have resource management, monitoring, and research responsibilities, is implementing a long-term monitoring program, the Environmental Monitoring and Assessment Program (EMAP), that will provide the public, scientists, and interested parties with information that can be used to evaluate the overall health of the nation's ecological resources (U.S. EPA 1988; NRC 1990). EMAP has been conducting regional surveys to measure indicators of the health of plants and animals, the quality of their surroundings, and the presence of pollutants.

During the summers of 1990–93, data were collected from approximately 450 sampling stations located in estuarine waters of the mid-Atlantic region (mouth of Chesapeake Bay to Cape Cod). At each of the sampling locations, measurements (or indicators) were made of kinds and abundance of fish, incidence of conspicuous abnormalities in fish, kinds and abundance of organisms living in sediments, measures of water quality (such as concentration of dissolved oxygen), concentrations of contaminants in the sediments and toxicity of sediments to sensitive organisms (Holland 1990). Analyses have been conducted with individual indicators and combinations of indicators, and these results have been expressed as various aspects of estuarine condition (Weisberg *et al.* 1993; Schimmel *et al.* 1994; Strobel *et al.* 1994).

One of the objectives of EMAP is to provide information that helps explain, when unacceptable ecological conditions are observed, why these unacceptable conditions may be occurring. This is being accomplished through the examination of associations, or empirical relationships, between indicators of ecological condition and stressors potentially impacting the ecological systems. Important stressors include those that occur within the watersheds of affected estuarine waters (*i.e.*, watershed stressors). These stressors can be natural, such as climatic fluctuations and resultant storm patterns, or anthropogenic, such as population demographics, land use patterns and point source loadings of pollutants.

The concept that human activities within a watershed may affect the ecological condition of adjacent waters appears obvious, but it has been difficult to show direct relationships between estuarine condition indicators and watershed stressors. This has been due, in part, to the scarcity of estuarine monitoring data collected with consistent methods and procedures at a sufficient number of sites to make statistical analysis meaningful. Though several investigators have demonstrated relationships between watershed-based stressors and water quality in freshwater systems (Detenbeck *et al.* 1993; Osborne and Wiley 1988), few studies have addressed this issue in estuarine systems (Valiela *et al.* 1992; Correll *et al.* 1992), especially in terms of sediment quality.

In this paper, we compare three methods for assessing the relationships between estuarine sediment contaminant concentrations and watershed stressors for sub-estuaries within Chesapeake Bay. The stressors evaluated were coastal land use pattern and point source pollution. Chesapeake Bay and its associated watershed were selected for study because of the availability of EMAP sediment contaminant data, contemporary point source pollutant data and satellite imagery-derived land cover data.

2. Methods

2.1. Study area

Twenty-five small sub-estuaries (< 260 km^2 of estuarine surface area) within Chesapeake Bay and their associated coastal watersheds were selected as study sites. Small estuaries were chosen because their physical characteristics (small in areal extent, semi-enclosed, relatively shallow, in close proximity to land-based stressors) enhance the possible coupling of estuarine sediment quality to surrounding stressors. Additionally, contaminant dilution effects due to interactions with the main body of the bay are reduced. It has been suggested that these smaller systems are the initial estuarine water bodies to be affected by contaminants from adjacent land areas (Valiela *et al.* 1992). The number of estuary/watershed systems selected for analysis was limited to 25 out of a possible 47 for which sediment contaminant data were available due to incomplete land use data and the inability to accurately delineate several watersheds on the relatively flat mid-Atlantic Coastal Plain. The watersheds are located throughout the Chesapeake Bay area (Fig. 1), with 16 on the inner coastal plain and western shore of the bay, and with nine on the outer coastal plain and eastern shore.

2.2. Data sources

The sediment contaminant data were collected in the Chesapeake Bay sub-estuaries during the July–August period over the years 1990–93. The sub-estuaries were selected for sampling in a given year based upon a probability-based sampling design employed across the entire mid-Atlantic region (Holland 1990). The actual sampling site within each sub-estuary was randomly selected within the constraint that the sampling vessel could collect bottom sediments (*i.e.*, depth at least two m and non-rocky substrate). The top two cm of multiple sediment samples were collected with a stainless steel grab sampler, homogenized and analyzed for a suite of organic and inorganic contaminants. The concentrations (μg/g dry weight) of 15 metals were measured using inductively coupled plasma-atomic emission spectrometry or atomic absorption tech-

Table 1. Sediment contaminants selected for analysis.

Metals	Organics	
Silver	Total PAHs	Phenanthrene
Arsenic	Low Mol. Wt. PAHs	Benz(a)anthracene
Cadmium	High Mol. Wt. PAHs	Benz(a)pyrene
Chromium	Acenapthene	Chrysene
Copper	Acenaphthylene	Dibenz(a,h)anthracene
Mercury	Anthracene	Flouranthene
Nickel	Fluorene	Pyrene
Lead	2-Methylnaphthalene	
Zinc	Naphthalene	

niques. Likewise, the concentrations (ng/g dry weight) of 63 organic contaminants were determined using gas chromatography or mass spectrometry (Weisberg *et al.* 1993). Of these, 9 metals and 16 organics known to cause measurable effects in sediment organisms (Long *et al.* 1995) were selected for analysis in this study (Table 1.) All are EPA Priority Pollutants designated under Section 307(a) of the Clean Water Act and 10 are pollutants on the Chesapeake Bay Toxics of Concern List (Chesapeake Bay Program 1994).

Two watershed stressors were used in this study: land use pattern and point source pollution. Land use pattern, in this study, refers to the composition and location of land use types within a watershed. We assumed that land use pattern would be a suitable surrogate for nonpoint source pollution information because the severity of nonpoint source pollution is closely related to land use (Kim and Ventura 1993). Land use data were obtained from the EMAP Landscape Characterization Group Chesapeake Bay Watershed Pilot Project (U.S. EPA 1994a). The data were derived from Thematic Mapper (TM) satellite imagery of the Chesapeake Bay watershed (1989–1991) which were classified into five land cover categories at a classification accuracy of ≥ 85%. The five land cover classes were developed land, herbaceous land, forested land, barren land and water. The developed land category is composed of areas of anthropogenic use including urban commercial areas, suburban residential areas and major highway systems. The herbaceous class includes lawns, grasslands, agricultural fields, pastures and herbaceous wetlands. The forested category includes woody shrubs and trees. Barren land refers to exposed soil, sand or rock areas such as quarries, landfills, gravel pits and mines. The water category includes areas of natural and impounded standing water. The original data set had a spatial resolution of 25 m but was subsequently resampled to a 250 m grid cell size to reduce the processing time for spatial analyses.

Point source pollution data were obtained from the NOAA National Coastal Pollutant Discharge Inventory for the Virginian Province (Pacheco 1993). The location of active major point source discharge sites (Fig. 1) as well as 1991 estimates of annual pollutant loading (kg/yr) and annual outflow ($m^3/yr \cdot 10^6$) at each discharge site were used in this study. Loading estimates were available for seven (arsenic, cadmium, chromium, copper, lead, mercury and zinc) of the nine metals measured in sediments. These seven metals are present in the environment from natural and anthropogenic sources and are also among the most frequently measured pollutants at monitored discharge sites (Pacheco 1993). Loading data from discharge sites were unavailable for organic contaminants.

2.3. *Watershed delineation*

We delineated watersheds for each sub-estuary within the study area for which EMAP sampling data were available (n = 47) using one-degree (1:250,000 scale) United States Geological Survey (USGS) Digital Elevation Model (DEM) data processed with the GRID module of the Environmental Systems Research Institute (ESRI) ARC/ INFO GIS software package. The accuracy of computer delineations for many of the Coastal Plain watersheds was questionable because the horizontal and vertical resolution of the elevation data (93 m and 1 m, respectively) were inadequate to discern low topographic relief. It was possible to accurately delineate more watersheds on the west shore of Chesapeake Bay because the inner coastal plain ('Tidewater' area) has relatively high topographic relief compared to the outer coastal plain (Delaware-Maryland-Virginia peninsula or 'Eastern Shore'). The computer-derived watersheds were reviewed and enhanced through on-screen interpretation using DEM-derived shaded relief maps and hydrography (1:100,000 scale USGS Digital Line Graph) as backdrops. In areas of low topographic relief where the accuracy of computer-interpreted water-

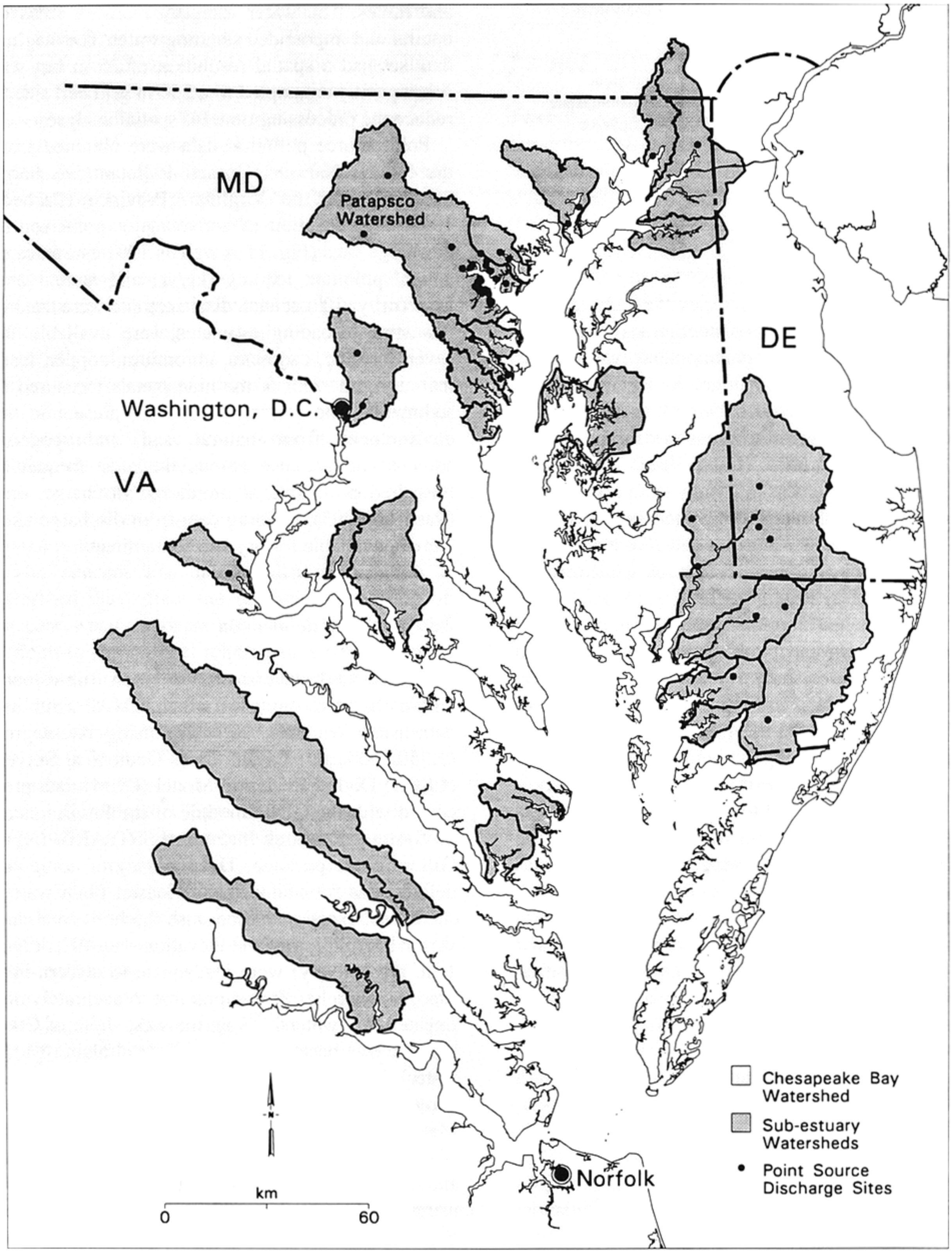

Fig. 1. Location of point source discharge sites in the 25 selected estuary/watershed systems in the Chesapeake Bay area.

shed boundaries was questionable, drainage divides were manually located by interpreting a line between stream drainage systems. The 25 most accurate watershed delineations were selected and used to define the estuary/watershed systems for this study.

2.4. Landscape analyses

Three methods for analyzing the landscape surrounding each sampling station were used: (1) a watershed approach using the watershed of the estuary containing the sampling station, (2) a 'partial watershed' approach using the area of the watershed within a 10 km radius of the sampling station and (3) a 'weighted partial watershed' approach where stressor values within each partial watershed were weighted by the inverse of the linear distance from the stressor location to the sampling station (Fig. 2). In order to judge the validity of selecting a station radius of 10 km for delineating partial watersheds, rank correlation coefficients for the area of developed land in watersheds within 2, 5, 10, 15 and 20 km of each sampling station with sediment metals levels were compared.

The ARC/INFO GRID module was used to determine the area of each land use class found within each watershed and partial watershed. Calculations for barren land were not performed because this land use type was poorly represented (< 0.5%) within the study area. The vector module of ARC/INFO was used to determine the total annual outflow and contaminant loading from all point discharge sites located within each watershed and partial watershed. For the weighted partial watershed approach, annual outflow and metals loading values at each discharge site within the partial watersheds were first weighted by the inverse of the linear distance from the discharge site to the sampling station. Likewise, each land use pixel within the partial watersheds was assigned an initial value of '1' and weighted similarly. A total of 11 watershed stressor variables were calculated for each of the three landscape analysis methods: the area of developed land (DEV), the area of herbaceous land (HERB), the area of forested land (FOR) and the annual outflow (FLOW) and metals loading rates (AS, CD, CR, CU, PB, HG, ZN) from all point discharge sites. Stressor variables were tested for normality using the Shapiro-Wilk test and examining normal probability plots (SAS Institute 1990). All variables, except the area of herbaceous land in partial watersheds, were non-normally distributed ($p \leq .01$, Shapiro-Wilk test). Consequently, a rank transformation was applied to all watershed stressor variables prior to use in subsequent statistical analyses.

2.5. Principal components analysis of sediment contaminant and loading variables

Principal components analysis (PCA) was used to assess the relationships among the individual sediment contaminants and to reduce the number of variables in subsequent statistical analyses. PCA was performed using a correlation matrix to standardize the deviation from the mean for each contaminant (Neff and Marcus 1980). Because metals and organics are somewhat distinct chemical groups, PCA computations were performed separately on the group of 9 metals and the group of 16 organic chemicals. Similarly, PCA computations were performed on the seven metals loading rates. All principal components were non-normally distributed ($p \leq .01$, Shapiro-Wilk test) and were rank transformed prior to use in subsequent statistical analyses. Rank correlation coefficients (r) were calculated between the principal components and the original variables to help interpret PCA results.

2.6. Correlation and regression

Pairwise rank correlations were performed as a nonparametric measure of the association between each of the watershed stressors and selected principal components for metals and organics. This procedure reduces the weights assigned to outliers in the raw data (Conover and Iman 1981; Potvin and Roff 1993). A rank correlation matrix of the stressor indicators was also constructed to examine the relationship between watershed stressor variables.

Nonparametric stepwise multiple linear regression models using ranked data (Conover and Iman 1981; Potvin and Roff 1993) were developed to examine the interrelationship among selected prin-

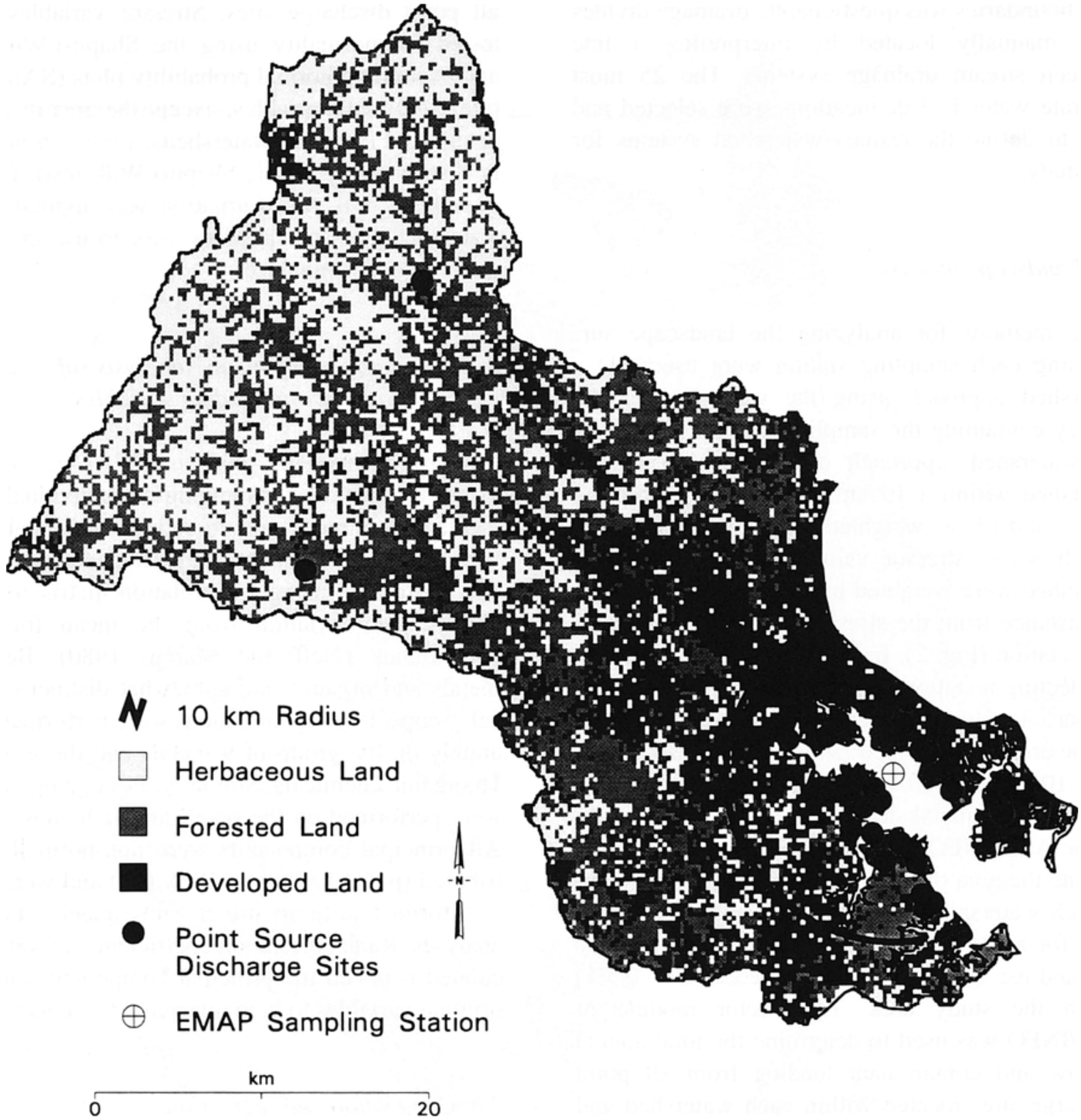

Fig. 2. Land use and point source discharge sites within the Patapsco River watershed and 10 km partial watershed in relation to the sediment sampling station.

cipal components for sediment metals and organics (the dependent variables) and watershed stressor (independent) variables (developed land, herbaceous land, forested land, annual outflow and PC1 for annual metals loading). A significance level of $p \leq 0.15$ for independent variables to enter and stay in the model was used. Variance inflation factors (VIFs) were calculated to identify intercorrelated independent variables. The VIF is a widely used means of detecting multicollinearity in a data set (Morrison *et al.* 1992; Philippi 1993). A VIF greater than 10 for a variable indicates a problem with multicollinearity (Neter *et al.* 1985). To test the validity of each regression model, plots of residuals against predicted values of the dependent variable were prepared and inspected for patterns in the residuals (Zar 1984). All analyses were performed separately for metals and organics and for each of the three landscape analysis methods: watershed, partial watershed and weighted partial

Table 2. Summary statistics (mean, standard deviation, range) for watershed and partial watershed variables for 25 estuaries.

			Watershed				Partial Watershed			
Variable	Abbrev	Units	Mean	SD	Min	Max	Mean	SD	Min	Max
Area	–	km^2	538	632	61	2333	114	57	40	263
Developed land area	DEV	%[1]	15	20	1	71	18	25	0	79
Herbaceous land area	HERB	%[1]	37	17	11	74	35	20	7	72
Forested land area	FOR	%[1]	45	16	11	70	42	19	8	71
Annual outflow	FLOW	$m^3/yr \cdot 10^6$	56	134	0	417	25	83	0	399
Arsenic loading	AS	kg/yr	296	875	0	4126	239	839	0	4126
Cadmium loading	CD	kg/yr	103	308	0	1443	84	294	0	1443
Chromium loading	CR	kg/yr	393	1164	0	5479	320	1116	0	5479
Copper loading	CU	kg/yr	348	1016	0	4752	283	972	0	4752
Lead loading	PB	kg/yr	393	1207	0	5709	326	1159	0	5709
Mercury loading	HG	kg/yr	3	8	0	38	2	8	0	38
Zinc loading	ZN	kg/yr	1465	4453	0	21088	1204	4281	0	21088

[1]Land use variables are given in % for comparing watersheds, actual variable values are areas.

watershed. SAS software was used for all statistical analyses (SAS Institute 1990).

3. Results

3.1. Landscape characteristics

The characteristics of the 25 watersheds selected for study were highly variable (Table 2). Watersheds ranged in size from 61 to 2,333 km^2. The percentage of developed land in watersheds ranged from 1 to 71% and was the most abundant land use in three of the 25 watersheds. Forested land was the most abundant land use in 13 and herbaceous land in nine of the watersheds. Point source outflow and metals loadings also varied widely among watersheds. Comparable variation in landscape characteristics was measured for partial watersheds.

The results of partial watershed analyses using varying distances from the sampling station indicated that a 10 km distance is a reasonable selection for delineating partial watersheds. Rank correlations were strongest ($r \geq 0.81$, $p < .001$) between sediment metals levels and the area of developed land in watersheds within 5, 10 and 15 km of the sampling station. Correlations were somewhat weaker ($r \leq 0.75$, $p < .001$) using distances of 2 and 20 km from the sampling station.

3.2. Principal components analyses

The first eigenvectors for each of metals and organics showed positive component correlations of similar magnitude for nearly all contaminants (Table 3). Arsenic, which was weakly correlated with PC1, was the only contaminant strongly correlated with PC2. We interpret PC1 to reflect the overall amount of metals or organic contaminants at a site; the larger PC1, the greater the amount of sediment contamination. The first principal components for metals (METPC1) and organics (ORGPC1) were used to represent the level of sediment contamination.

The results of principal components analyses with metals loading variables were nearly identical for all three landscape analysis methods. Significant ($r \geq 0.93$, $p < .0001$) positive rank correlations were observed between PC1 for metals loading (LOADPC1) and the loading rates for each of the seven individual contaminants. The first eigenvector showed uniform component correlations for all contaminants. LOADPC1 explained 99.9% of the standardized variance in metals loading rates in watersheds, partial watersheds and weighted partial watersheds and was used to represent metals loading rates.

3.3. Correlation and multiple regression analyses

In the three landscape analysis methods, pairwise rank correlations indicated that four of five stressor

Table 3. Eigenvectors and significant (p < .05) rank correlations (r) between the sediment concentration of individual contaminants and the first (PC1) and second (PC2) principal components. Principal components analyses were performed separately for metals and organics.

	PC1		PC2	
Contaminant	Eigen-vector	r	Eigen-vector	r
Metals				
Silver	0.31	0.64	–0.32	–
Arsenic	0.10	0.41	0.84	0.93
Cadmium	0.37	0.93	–0.10	–
Chromium	0.35	0.93	0.31	–
Copper	0.36	0.96	0.14	–
Mercury	0.34	0.87	–0.24	–
Nickel	0.35	0.97	–0.10	–
Lead	0.37	0.95	–0.06	–
Zinc	0.37	0.96	0.08	–
% Variation Explained	77.6		13.8	
Organics				
Total PAHs	0.27	0.98	–0.02	–
Low Mol. Wt. PAHs	0.25	0.92	0.27	0.45
High Mol. Wt. PAHs	0.26	0.95	–0.18	–
Acenapthene	0.26	0.81	0.20	–
Acenaphthlylene	0.25	0.81	0.24	–
Anthracene	0.26	0.87	–0.04	–
Fluorene	0.24	0.88	0.28	–
2-Methylnaphthalene	0.23	0.67	0.39	0.57
Naphthalene	0.22	0.57	0.41	0.40
Phenanthrene	0.26	0.95	0.08	–
Benz(a)anthracene	0.25	0.93	–0.25	–
Benz(a)pyrene	0.25	0.88	–0.22	–
Chrysene	0.25	0.96	–0.28	–
Dibenz(a,h)anthracene	0.24	0.83	–0.34	–
Flouranthene	0.26	0.91	–0.24	–
Pyrene	0.26	0.95	–0.21	–
% Variation Explained	84.8		11.2	

variables were significantly correlated (p < .05) with METPC1 (Table 4). Correlation results for stressor variables with ORGPC1 were similar with the exception that the area of herbaceous land in watersheds was not correlated with ORGPC1. The area of forested land was not correlated with METPC1 or ORGPC1. All significant correlations of herbaceous land with METPC1 and ORGPC1 were negative. Correlations of stressors with PC1 were generally stronger for metals than for organics. Correlations of METPC1 and ORGPC1 with the stressor variables were generally lowest for watersheds and highest for weighted partial watersheds. Several watershed stressor variables were significantly correlated (p < .05, Table 5). The strongest correlations were observed between FLOW and LOADPC1 for each landscape analysis method.

A significant amount of variation in the level of sediment metals and organics was explained by the regression models (Table 6). Coefficients of multiple determination (R^2) ranged from 0.468 to 0.781. Among all three landscape analysis methods, stressors consistently accounted for a greater percentage of the variation in sediment metals levels than sediment organics levels. Regression models using stressor indicator data generated by the weighted partial watershed landscape analysis method explained the greatest percentage of the variation in both sediment metals and organics levels. For metals, regression models using stressor indicator data generated by the partial and weighted partial watershed landscape analysis methods yielded model R^2 values considerably higher than the R^2 value from the regression model using data generated by the watershed landscape analysis method. Plots of residuals versus predicted values showed a random distribution of residuals for each regression model with R^2 values of zero for regressions against predicted values.

4. Discussion

4.1. Relationships between watershed stressors and sediment contamination

Our results suggest that variation in sediment contamination levels among small sub-estuaries within Chesapeake Bay is related to the human activities associated with developed land use and point source discharges of pollutants. The influence of these watershed stressors on sediment contamination appears to be greatest when they are located within 10 km of the sediment sampling station. The strength of the associations between sediment contamination level and each of the watershed stressors improved noticeably when only those stressors within the partial watershed were considered. Weighting stressor values within the 10 km partial watershed by the inverse of the distance from the sampling station improved the strength

Table 4. Significant ($p < .05$) rank correlations for the first principal component (PC1) of nine sediment metal and twenty sediment organic concentrations with watershed stressor variables using the watershed (Wshed), partial watershed (Partial) and weighted partial watershed (Weighted) landscape analysis methods. Correlations which were not significant are indicated by 'ns'. Loading data were unavailable for organic contaminants.

		Sediment Metals – PC1			Sediment Organics – PC1		
Watershed Stressors	Code	Wshed	Partial	Weighted	Wshed	Partial	Weighted
Area of Developed Land	DEV	0.46	0.81	0.84	0.54	0.68	0.70
Area of Herbaceous Land	HERB	−0.42	−0.61	−0.62	ns	−0.57	−0.56
Area of Forested Land	FOR	ns	ns	ns	ns	ns	ns
Annual Outflow	FLOW	0.59	0.78	0.78	0.47	0.58	0.61
Metals Loading – PC1	LOADPC1	0.44	0.73	0.74	–	–	–

Table 5. Correlation matrix showing significant ($p < .05$) rank correlations for watershed stressor (independent) variables using three landscape analysis methods. Correlations which were not significant are indicated by 'ns'.

	Landscape Analysis Method											
	Watershed				Partial Watershed				Weighted Partial Watershed			
	DEV	HERB	FOR	FLOW	DEV	HERB	FOR	FLOW	DEV	HERB	FOR	FLOW
HERB	ns	–	–	–	−0.62	–	–	–	−0.66	–	–	–
FOR	ns	0.84	–	–	ns	0.55	–	–	ns	0.50	–	–
FLOW	0.62	ns	ns	–	0.69	−0.60	ns	–	0.70	−0.58	ns	–
LOADPC1	0.59	ns	ns	0.84	0.70	−0.57	ns	0.91	0.70	−0.56	ns	0.93

DEV = area of developed land; HERB = area of herbaceous land; FOR = area of forested land; FLOW = annual outflow; LOADPC1 = first principal component for metals loading.

of this relationship slightly. In a similar study, Osborne and Wiley (1988) found that land use within a buffer zone of surface waters has a greater influence on river water quality than does the land use within the entire watershed.

Using the weighted partial watershed approach, regression models explained 78% of the variation in sediment metals levels and 49% of sediment organics levels. These results compare favorably with the results of similar watershed studies in lacustrine and riverine systems, especially considering the complexity of tidal circulation and sediment transport in estuarine systems. Detenbeck *et al.* (1993) found that regression equations using watershed variables explained from 14 to 76% of the variation in lake water quality variables. Osborne and Wiley (1988) found that watershed land use variables explained 49 to 95% of the variation in stream nutrient concentrations.

Partial R^2 results obtained from stepwise multiple regression analyses indicate that the area of developed land may be a dominant factor in determining the levels of both sediment metals and organics. Using the weighted partial watershed approach, approximately 70% of the variation in sediment metals levels and 49% of the variation in sediment organics levels could be explained as a function of the area of developed land alone. Although the area of developed land was the first independent variable to enter in five or six stepwise regressions, correlations between annual outflow and sediment contamination levels were nearly as strong as those obtained for the area of developed land and sediment contamination levels (Table 4). Annual outflow was more strongly correlated with sediment metals levels than sediment organics levels and appears to have an additional effect on the level of metals in sediments. This relationship is consistent with the fact that significant amounts of inorganic contaminants (*e.g.*, metals) are often released at point discharge sites (U.S. EPA 1994b).

Land use classes such as developed land may act as pollutant sources while others may function as pollutant 'sinks' within a watershed (Detenbeck *et*

Table 6. Results of stepwise multiple regressions with the rank of the first principal component for sediment metals (METPC1) or organics (ORGPC1) as the dependent variable and the rank of the area of developed land (DEV), herbaceous land (HERB), forested land (FOR), annual outflow from point discharge sites (FLOW) and the rank of the first principal component for metals loading[a] (LOADPC1) as independent variables. Results for the watershed (Wshed), partial watershed (Partial) and weighted partial watershed (Weighted) landscape analysis methods are given. Independent variables shown met the 0.15 significance level to enter and remain in the stepwise regression model.

Group	Analysis Method	Parameter	Coefficient Value	SE[b]	VIF[c]	Partial R^2	Model R^2	Adjusted[d] R^2	F-Value[e]
Metals	Wshed	Intercept	9.39	2.72	–	–	–	–	–
		FLOW	0.482	0.185	1.6	0.348	–	–	–
		FOR	–0.499	0.144	1.1	0.196	–	–	–
		DEV	0.295	0.182	1.7	0.051	0.595	0.537	10.3
	Partial	Intercept	–0.305	1.89	–	–	–	–	–
		DEV	0.517	0.150	1.9	0.651	–	–	–
		FLOW	0.507	0.181	1.9	0.092	0.743	0.720	31.8
	Weighted	Intercept	–0.476	1.75	–	–	–	–	–
		DEV	0.571	0.139	1.9	0.704	–	–	–
		FLOW	0.466	0.168	1.9	0.077	0.781	0.761	39.3
Organics	Wshed	Intercept	10.5	2.74	–	–	–	–	–
		DEV	0.653	0.157	1.1	0.291	–	–	–
		FOR	–0.459	0.157	1.1	0.198	0.489	0.443	10.5
	Partial	Intercept	4.11	2.26	–	–	–	–	–
		DEV	0.684	0.152	1.0	0.468	0.468	0.445	20.2
	Weighted	Intercept	3.86	2.20	–	–	–	–	–
		DEV	0.703	0.148	1.0	0.494	0.494	0.472	22.5

[a]The rank of the first principal component for metals loading (LoadPC1) was not used in regression analyses with sediments organics
[b]SE = Standard Error
[c]VIF = variance inflation factors
[d]The explanatory power of the model is adjusted for the number of parameters in the model
[e]$p < .001$.

al. 1993). Forested land located in riparian and other low-lying areas may trap sediments and cause a reduction in the amount of nonpoint pollutants which reach surface waters (Karr and Schlosser 1978; Peterjohn and Correll 1984; Whigham and Chitterling 1988). Detenbeck *et al.* (1993) found that increasing area of watershed forested land was associated with a reduction in lake trophic state. In our study, however, there were no significant correlations between the area of forested land and sediment quality for any of the landscape analysis methods. This may be to due the position of forested lands relative to nonpoint sources of contaminants within the watersheds. We used the GRID module of the ARC/INFO GIS to calculate the average elevation of developed versus forested land as an indication of the position of each land use type within each watershed. The average elevation of watershed forested land was higher than developed land in 19 of 25 watersheds. This suggests that forested lands are generally located above developed lands in most of the watersheds and are not 'in position' to function as contaminant sinks.

Herbaceous land may function as a contaminant source or sink depending upon the specific class of herbaceous land being assessed. Agricultural land can be a source of contaminants when agricultural pesticides are applied (U.S. EPA 1994b), whereas herbaceous wetlands such as wet meadows, freshwater marshes and salt marshes may function as pollutant sinks or transformers (Kadlec and Kadlec 1979). Because herbaceous wetlands were not differentiated from agricultural land within the herbaceous category of the land use data set used in this study, we could not effectively interpret the relationships between the area of herbaceous land and sediment contamination levels. Our results, however, did show a negative relationship between sediment contaminant levels and the amount of surrounding herbaceous land. These results may reflect the effects of herbaceous upland and wetland processes such as sedimentation, microbial transformation and uptake and storage of contaminants by plants (Bastian and Benforado 1988). Calculations based on 1973 USGS digital land use and land cover data (Fegas *et al.* 1983) show that herbaceous wetland comprised between 0 and 27% of the watersheds and 0 to 65% of the partial watersheds we studied. These percentages are comparable to wetland cover percentages in watersheds from other studies where an improvement in water quality was related to the area of wetland cover (Johnston *et al.* 1990; Detenbeck *et al.* 1993).

4.2. Data accuracy and uncertainty

Inaccuracy and uncertainty in parameter estimates can contribute to the difficulty in explaining variation in sediment contamination. The accuracy of a nonpoint source pollution model is dependent upon the resolution of the data as well as the specificity of land use categories (Kim and Ventura 1993). Data resolution, or cell size, influences computer system performance through its effects on data storage requirements and processing speed. Generally, data volume, disk space requirements and operation response times decrease as grid cell size increases. Our land use data set had an initial spatial resolution of 25 m but was resampled to a 250 m grid cell size to reduce the processing time for spatial analyses. This increase in cell size can reduce accuracy when assessing the extent of each land use category (Wehde 1982). The individual and collective characteristics of landscape features influence the rate of information loss when data are resampled to a coarser resolution. Several studies have shown that small, complex and isolated features in heterogeneous landscapes are more readily lost as data resolution is decreased (Meentemeyer and Box 1987; Turner *et al.* 1989).

Point source pollution data used in this study exhibited varying degrees of uncertainty. When possible, discharge estimates were derived from monitoring sources, however, when monitoring data were unavailable, estimates were then based on National Pollutant Discharge Elimination System (NPDES) permit limit requirements or 'typical pollutant concentration' data (Pacheco 1993). For example, 78% of the annual outflow estimates were based on monitoring sources. Conversely, much less monitoring data were available for metals loading estimates. The uncertainty in loading estimates may explain why metals loading rates were not as strongly correlated to sediment metals levels as were annual outflow rates.

While it is plausible that the top two cm of sediment contains only contemporary pollutants released to the watershed in recent years, it is also possible that the sediment contamination values reflect cumulative or high historical pollutant loading and would not necessarily be correlated with current, single-year, point source loading measurements within the watershed (Detenbeck *et al.* 1993). In this case, a time-weighted estimate of cumulative loading based on historical loading rates might be more strongly correlated with levels of contamination in surficial sediments. Additionally, our analyses did not address several other sources of pollutant loading including atmospheric deposition loadings, shipping and boating loadings and groundwater loadings (U.S. EPA 1994b) which may also contribute to the variation in sediment contamination levels.

4.3. Attributes and limitations of landscape analysis methods

The complicated pattern of land use and terrain within each watershed contributes to the variation in sediment contamination among estuaries as well as the difficulty in explaining this variation. The definition of land use pattern, in this study, was limited to the composition and location of land use types within a watershed. A broader definition would also include the collective spatial characteristics of features in a land use data set, including individual feature characteristics such as unit size and shape, and topological relationships such as unit isolation and elevation within the watershed

(Ritters *et al.* 1995). These characteristics of pattern may be related to the amount of nonpoint source pollutants which actually reach the surface waters of an estuary (Johnston *et al.* 1990), but were not considered in this study.

Our watershed landscape analysis method used a 'lumped parameter' approach (Maidment 1993), where watershed stressor values (the parameters) were spatially averaged over the entire watershed. The assessment of land use pattern was limited to simple calculations of the area of each land use class within each watershed. No other individual or topological characteristics were used to describe land use pattern. Similarly, estimates of point source pollution (outflow and loading) were summed over the entire watershed. Using this approach, the assumption is made that all pixels of developed land or point discharge sites contribute equally to sediment contamination, regardless of distance from the sampling station. This is a simplistic approach which ignores the effects of attenuation, dispersion and uptake of contaminants as they move through soil, water and wetlands. Not unexpectedly, correlations and R^2 values were lowest using this landscape analysis method.

In a more realistic distributed parameter approach (Maidment 1993), watershed stressor values are calculated as a function of location in the watershed. Our partial watershed method is actually a simplified distributed system where stressor values from locations in the watershed beyond 10 km from the sampling station are weighted by a value of zero while values from locations within 10 km from the sampling station are weighted by one. The assumption here is that stressors located further than 10 km from the sediment sampling station make no contribution to sediment contamination. This simple weighting scheme accounted for significant improvement in R^2 values from the watershed to partial watershed approaches used in this study.

In our weighted partial watershed approach, stressor values within each partial watershed were weighted based on a function of their distance from the sampling station. The linear distance weighting improved the fit of the data to the regression model slightly, compared to the partial watershed model. This suggests that stressors within 10 km of the sampling station contribute similarly to sediment contamination levels or that a non-linear weighting function may be more appropriate. Repetition of the weighted partial watershed analysis using an inverse distance squared weighting model yielded correlation and R^2 values slightly lower than the values obtained using the original inverse distance weighting function. The spatially distributed approach featured in our hybrid and weighted partial hybrid landscape analysis methods are relatively new to hydrologic modeling and is an area where GIS may contribute significantly (Maidment 1993).

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