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ARTICLE

Impervious Surface, Summer Dissolved Oxygen, and Fish Distribution in Chesapeake Bay Subestuaries: Linking Watershed Development, Habitat Conditions, and Fisheries Management

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Abstract

We estimated target and limit impervious surface reference points (ISRPs) based on Chesapeake Bay dissolved oxygen (DO) criteria, and we examined associations and relationships among the percentage of watershed in impervious surface (IS), summer DO, and the presence of indicator species (blue crab *Callinectes sapidus*, white perch *Morone americana*, striped bass *M. saxatilis*, and spot *Leiostomus xanthurus*) in bottom waters of nine brackish subestuaries of Chesapeake Bay. Ideally, a target ISRP represented a level of development that maintained mean bottom DO at 5 mg/L or greater, while an ISRP threshold represented development that degraded mean bottom DO to less than 3 mg/L. The proportion of bottom trawls containing each indicator species rapidly declined from about 0.40 to 0.10 when DO fell below 3 mg/L, whereas the proportion remained at about 0.50 when DO was above 5 mg/L. The IS percentage had a significant negative influence on mean bottom DO and the odds that indicator species were present in midchannel bottom waters (0.8–7.0 m deep). Watersheds at or below a target IS of 5.5% (rural watershed) maintained mean bottom DO above 3 mg/L, but mean DO was only occasionally at or above 5 mg/L. Mean DO seldom exceeded 3 mg/L in watersheds with an IS value above 10% (suburban threshold). Comprehensive watershed management will be needed to offset significant degradation of bottom-water fish habitat in brackish subestuaries if rural lands are converted to suburban areas.

Assessments of human-induced perturbations of fish populations have typically focused on fishing (Boreman 2000), and biological reference points have been developed to guide the level at which fish can be safely harvested from a stock (Sissenwine and Shepherd 1987; Caddy and McGarvey 1996). Managers also take action to avoid negative impacts from habitat loss and pollution (Boreman 2000). A habitat-based corollary to biological reference points would be habitat degradation reference points that guide habitat conservation or activities attempting to compensate for habitat loss, such as harvest reductions, hatchery enhancement, and habitat restoration.

Increased residential development associated with human population growth conflicts with demands for fish production and fishing opportunities in coastal areas (Pearce 1991) and has been identified as a threat to Chesapeake Bay (Chesapeake Bay Program 1999). Stock declines due to the habitat effects of development were cited when moratoria on the harvest of yellow perch *Perca flavescens* were imposed for 20 years (starting in 1989) in Chesapeake Bay subestuaries located between Washington, D.C., and Baltimore, Maryland (Jensen 1993; Yellow Perch Workgroup 2002; Uphoff et al. 2005; Maryland Fisheries Service 2010a).

The extent of impervious surface (IS; paved surfaces, buildings, and compacted soils) has been used as an indicator of watershed development because of its effect on habitat and aquatic life in freshwater systems and because it is a variable

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in many water quality and water quantity models (Arnold and Gibbons 1996; Cappiella and Brown 2001; Wheeler et al. 2005; NRC 2009). Impervious surface increases runoff volume and intensity, erosion, sedimentation, temperature, contaminant loads, and nutrient loads in streams (Wheeler et al. 2005; NRC 2009). Altered and impaired fish communities are characteristic of urban streams, and fish communities become less abundant, less diverse, and more dominated by pollution-tolerant and nonnative species (Wheeler et al. 2005). Threshold effects of urban cover and IS are evident for stream fish (Wheeler et al. 2005; Stranko et al. 2008) and anadromous fish spawning (Limburg and Schmidt 1990). Adverse physical and chemical changes in South Carolina tidal creek ecosystems occurred when the IS level in a watershed exceeded 10–20%, and sensitive macrobenthos, penaeid shrimp, and spot *Leiostomus xanthurus* were negatively associated with IS (Holland et al. 2004). Fecal coliform loadings in North Carolina and South Carolina coastal watersheds increased linearly with IS (Mallin et al. 2000; Holland et al. 2004).

We applied the target and limit concept (Caddy and McGarvey 1996; Rice 2003) to develop IS reference points (ISRPs) for brackish subestuaries of Chesapeake Bay in Maryland based on associations and relationships among IS, habitat quality, and fish responses. A target ISRP would represent a "safe" percentage of watershed development associated with maintenance of nursery and adult habitat requirements, whereas an ISRP threshold would represent degradation to the point where a significant portion of habitat cannot meet the requirements of fish.

The occupation of bottom and shore zone habitat by a suite of indicator species was used as a measure of responses to IS-related degradation. Use of indicator species is widespread in studies of pollution and environmental conditions (Rice 2003). We selected five indicator species × life stage combinations (hereafter, "indicator species"), including two anadromous species (age-0 and age-1 and older [age-1+] white perch *Morone americana*; age-0 striped bass *M. saxatilis*) and two marine-origin species (age-0 spot; all ages of blue crab *Callinectes sapidus*). The indicator species are widespread and support important fisheries in Chesapeake Bay, and they are sampled well by commonly applied seine and trawl techniques (Bonzek et al. 2007). Furthermore, the bay serves as an important nursery area for these species (Lippson 1973; Funderburk et al. 1991).

We chose summer (July–September) dissolved oxygen (DO) as an indicator of IS-related habitat degradation because fish require well-oxygenated water and because DO provides insight into the pollution status of a water body (Limburg and Schmidt 1990; Breitburg 2002). In Chesapeake Bay, low DO is identified as a problem in mesohaline bottom waters of the main-stem and lower reaches of large subestuaries that have a stratified water column during late spring through early fall (Hagy et al. 2004; Kemp et al. 2005; Batiuk et al. 2009). Nutrient enrichment from agricultural fertilizers is the primary reason for these low-DO conditions, although urban areas can also exhibit high nutrient loading (Kemp et al. 2005; Brush 2009; NRC 2009).

We used a DO concentration of 5 mg/L as a target and 3 mg/L as a threshold for development of the ISRPs. Dissolved oxygen concentrations of 5 mg/L or greater are considered desirable for many Chesapeake Bay living resources (Batiuk et al. 2009). Chesapeake Bay DO criteria for deepwater fishes and shellfish call for maintaining a 30-d mean of 3 mg/L during June 1–September 30 in bottom waters (Batiuk et al. 2009). We hypothesized that an increase in IS would lead to degraded DO in bottom waters (i.e., meeting the target less and falling below the threshold more) and thus to a decline in occupation of bottom waters by indicator species. Occupation of shallow water would have a much wider range of potential responses (from declining to increasing) since fish and blue crabs become restricted to oxygenated shallows when hypoxia is extensive (Eby and Crowder 2002).

METHODS

During 2003, we sampled nine subestuaries within two regions (midbay and Potomac River) of Chesapeake Bay; seven of the nine subestuaries were also sampled during 2004–2005 (Figure 1). All subestuaries had watershed areas of less than 60,000 ha, and eight had watershed areas less than 18,000 ha (Table 1). The West and Rhode rivers (Figure 1), two joined embayments, were considered a single subestuary. Salinities in the nine subestuaries averaged 4–11‰ annually when all depths and stations within a subestuary were combined.

In general, four evenly spaced sample sites were located in the upper two-thirds of each subestuary (i.e., linear distance along the center from head to mouth). Sites were not located near the subestuary mouth to reduce influence of main-stem Chesapeake Bay or Potomac River waters on measurements of watershed water quality. All of the sites on a given river were sampled on the same day during daylight, and there were 2 visits/month during July–September (i.e., 6 visits/year).

A 4.9-m semiballoon otter trawl sampled midchannel bottom habitat. The trawl was constructed of treated nylon-mesh netting with a mesh size of 38 mm (all measurements are stretch mesh) in the body and 33 mm in the cod end. An untreated, 12-mm knotless-mesh liner covered the cod end. A single tow (6 min at 3.2 km/h) was made in the same direction as the tide during each site visit. Trawl sites were located in the deepest portion of the channel at a station, and upstream trawl sites were generally shallowest; the approximate median depths were 2.0 m (range = 0.8–3.5 m) at station 1 (furthest upstream), 3.0 m (range = 1.8–6.2 m) at station 2, 4.0 m (range = 2.0–6.8 m) at station 3, and 4.2 m (range = 3.2–7.0 m) at station 4 (furthest downstream).

A 30.5- × 1.2-m bagless beach seine made of knotted, 6.4-mm stretch mesh was used to sample shore zone (shallow) habitat adjacent to a trawl site. One end of the seine was held on shore, while the other end was stretched perpendicular to shore as far as depth permitted. The end furthest from shore was pulled with the tide to the beach in a quarter arc and was then

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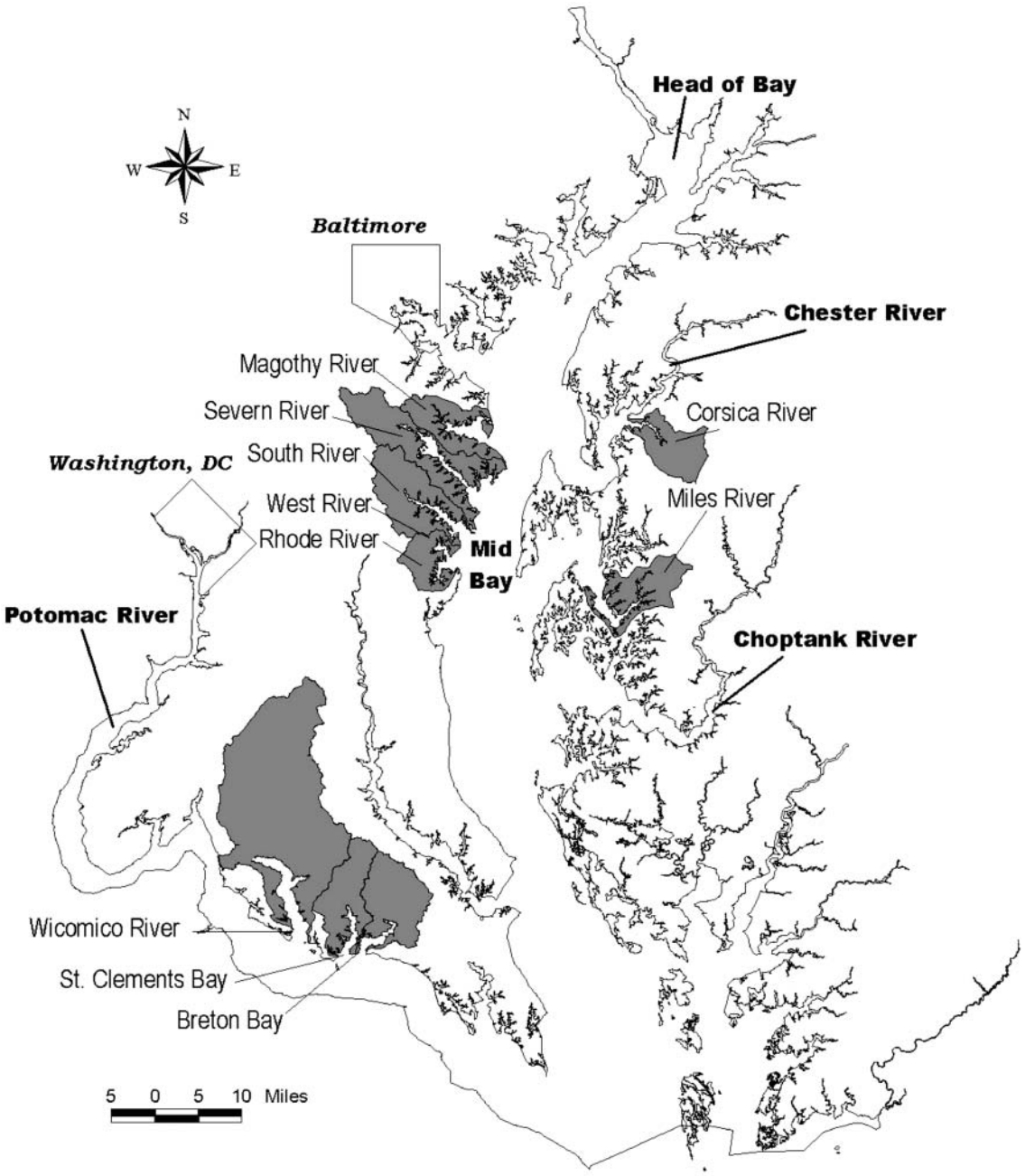

FIGURE 1. Locations of subestuaries in the Maryland portion of Chesapeake Bay (sampled during 2003–2005), their watershed boundaries (gray shading), and general geographic regions (bold letters).

pursed. A single seine haul was made at a site. Obstructions or lack of beaches prevented seining at some sites.

Maximum depth (m) of each trawl sample was recorded, and water temperature (°C), DO (mg/L), and salinity (‰) were measured (YSI Model 85 meter) at the surface, middle, and bottom of the water column at the trawl site and at the surface of the seine site. Middepth measurements were not made at sites where the difference between surface and bottom was less than 1.0 m.

For each watershed, IS area and watershed area estimated by Towson University from Landsat satellite imagery with 30-m/pixel resolution (eastern shore of Chesapeake Bay in 1999; western shore in 2001; Barnes et al. 2002; D. Sides, Towson University, personal communication) were used to calculate the IS percentage as

$$\text{IS}\,(\%) = (\text{IA}/\text{TA}) \times 100,$$

TABLE 1. Region, subestuary watershed, years sampled, watershed area, percent of watershed in impervious surface (IS), and land cover estimates (percent urban, agriculture, forest, and wetland; Maryland Department of Planning) for Chesapeake Bay subestuaries (Figure 1).

Region	Subestuary watershed	Years sampled	Watershed area (ha)	IS (%)	Urban (%)	Agriculture (%)	Forest (%)	Wetland (%)
Midbay, east	Corsica River	2003–2005	9,699	4.1	5.7	65.4	28.1	0.6
	Miles River	2003–2005	11,078	3.4	12.1	56.1	30.4	1.4
Midbay, west	Magothy River	2003	9,131	20.2	61.1	6.0	32.8	0.0
	Severn River	2003–2005	17,907	19.5	47.3	11.1	41.2	0.2
	South River	2003–2005	14,745	10.9	29.0	19.9	50.5	0.4
	West and Rhode Rivers	2003–2005	6,586	5.0	17.7	36.4	44.9	1.0
Potomac River	Breton Bay	2003–2005	14,205	5.3	11.3	26.3	61.5	0.5
	St. Clements Bay	2003–2005	11,990	4.4	7.0	40.9	51.3	0.8
	Wicomico River	2003	59,363	4.3	7.4	30.1	56.7	1.6

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where IA is the estimated IS area in the watershed and TA is the estimated total area of the watershed (Table 1). Water area was excluded from this calculation. Watersheds with IS values of 5.5% or less were categorized as rural landscapes (low IS), and those with IS values of 10.0% or more were designated as suburban landscapes (high IS).

We computed the correlation of IS estimates for the nine watersheds with each of four land cover (%) estimates (urban, agriculture, forest, and wetland) from 1994 as determined by the Maryland Department of Planning (MDDNR 1999; Table 1). Correlation analysis also evaluated associations between the IS, urban, agricultural, forest, or wetland percentage and annual estimates of mean surface DO or bottom DO for all stations within a subestuary (hereafter, "mean surface DO" or "mean bottom DO"; $N = 23$ for each comparison). These analyses explored (1) whether IS estimates were correlated with another indicator of development (percent urban land cover); (2) general associations among major landscape features in our study watersheds; and (3) associations between land cover types and DO in the two major fish habitat categories (shore zone and bottom channel). Inspection of scatter plots indicated that hyperbolic associations were possible among some land use variables, and an inverse transformation was used (Sokal and Rohlf 1981). Urban land cover consisted of high- and low-density residential, commercial, and institutional acreages (MDDNR 1999) and was not a direct measure of IS.

Our primary interest was in the relationships between land use (IS in particular) and DO in shore zone and bottom channel waters. Historical changes in forest, agriculture, wetland, and developed lands in the Chesapeake Bay watershed have been associated with changes in nutrient loading, assimilation, and buffering that influence DO in main-stem Chesapeake Bay (Kemp et al. 2005; Brush 2009). However, other variables were potential influences on DO, and correlation analysis determined the direction and strength of associations between annual mean surface or bottom DO and annual means of temperature, salinity, and depth (depth was analyzed in relation to bottom DO only; $N = 23$ for each comparison). Significant associations between temperature or salinity and DO were possible because temperature and salinity influence DO saturation and stratification (Kemp et al. 2005), whereas chronically low DO is associated with deeper waters (below the pycnocline) of Chesapeake Bay (Hagy et al. 2004). The annual means of surface or bottom DO in summer at all sites within a subestuary were selected for the analyses in order to match the geographic scale of IS estimates (whole watershed) and to characterize chronic conditions.

The significance of correlations between land use, water temperature, salinity, or bottom depth and the mean surface or bottom DO was adjusted for multiple comparisons by dividing the desired α (0.05) by the number of comparisons (i.e., the standard Bonferroni correction; Nakagawa 2004). This procedure and a sequential Bonferroni correction are commonly applied in the field of ecology and evolution; however, there is no formal consensus as to when these procedures should be applied, and both exacerbate problems of low statistical power (Nakagawa 2004).

Average depth of sites within a subestuary corresponded closely with IS estimates, and thus there was a potential for confounding of the depth and IS effects. We used separate linear regressions of bottom DO (dependent variable) on depth (independent variable) for low- and high-IS subestuaries to explore the influence of depth. This analysis used concurrent bottom depth and bottom DO measurements obtained from each site visit during 2003–2005. Analysis was confined to depths that were common to both of the IS categories.

We used separate analysis of covariance (ANCOVA) models to describe the response of annual mean bottom or surface DO to IS. Mean bottom DO or surface DO was modeled as the response variable, IS was the covariate, and year was a factor encompassing annual differences in characteristics that influence DO (e.g., stratification, organic and nutrient loading, sedimentation, temperature, and salinity; Baird et al. 2004; Kemp et al. 2005). We first tested for heterogeneity of slopes by including a year $\times$ IS interaction term in each model (Littell et al. 2002). If the interaction was not significant ($P > 0.05$), it was dropped from the model. Each ANCOVA model was then used

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to estimate the common slope for either mean bottom DO or mean surface DO with IS and year. If the common slope was significant, least-squares means (means adjusted for the covariate, IS) were calculated for each year to compare DO values among years. Analysis of covariance was conducted with the GLM procedure in the Statistical Analysis System (SAS; Littell et al. 2002).

The proportions of all bottom DO measurements that met or exceeded the 5-mg/L target level ($N_{\text{target}}/N_{\text{total}}$) or that were at or below the 3-mg/L threshold ($N_{\text{threshold}}/N_{\text{total}}$) in low- or high-IS subestuaries during 2003–2005 were estimated (where N_{target} = number of measurements that met or exceeded 5 mg/L, $N_{\text{threshold}}$ = number of measurements that were at or below 3 mg/L, and N_{total} = total sample size). The SD of each proportion was estimated as

$$\text{SD} = (\{[N_x/N_{\text{total}}] \times [1 - (N_x/N_{\text{total}})]\}/N_{\text{total}})^{0.5},$$

where N_x equals either N_{target} or $N_{\text{threshold}}$ for the respective low-IS or high-IS subestuary calculations (Ott 1977).

Catch data were treated as presence–absence to (1) estimate relative abundance of each indicator species within a bottom DO category as the proportion of trawl samples that contained the given target species and (2) analyze occupation of shore zone and bottom channel habitat by each indicator species at the sample level. Presence–absence was ecologically meaningful, minimized errors and biases in sampling, and reduced statistical concerns about the lack of normality and the high frequency of zero catches, which was expected given the hypothesis that increases in IS lead to depleted DO, thereby causing reduced occupation of bottom waters by indicator species (Green 1979; Bannerot and Austin 1983; Mangel and Smith 1990). Proportions of positive- or zero-catch indices were previously found to be robust as indicators of abundance for yellowtail snapper *Ocyurus chrysurus* (Bannerot and Austin 1983), age-0 white sturgeon *Acipenser transmontanus* (Counihan et al. 1999), eggs of Pacific sardine *Sardinops sagax* (Mangel and Smith 1990), and eggs of Chesapeake Bay striped bass (Uphoff 1997) and as indicators of performance for the longfin inshore squid *Loligo pealeii* fishery (Lange 1991).

Interpretation of a given species' absence from a site can be ambiguous (Green 1979) since absence could reflect the fact that (1) the site was never occupied because it was outside of the species' range or (2) the habitat had deteriorated to the point that the species could no longer occupy it. To minimize ambiguity in interpreting absence, we compiled seine and trawl catches to calculate the percentage of sites where each indicator species was encountered at least once. A high percentage of occurrence among all sites (~90%) indicated that a species was likely to occur at all sites and that sustained absence in bottom waters was therefore related to habitat conditions at the site.

We examined the relationships between the relative abundance of each indicator species and the DO target or DO threshold by using a descriptive model as a standard of comparison (Pielou 1981). Relative abundance was estimated for each indicator species as the proportion of bottom trawls that contained the given indicator species i (PT_i). Bottom DO was categorized into 1-mg/L bins, and the PT_i within DO bins between 1 and 8 mg/L was estimated. After inspecting scatter plots, we chose a Weibull function to describe the increase in PT_i as an asymmetric, ascending, asymptotic function of DO bin midpoint:

$$\text{PT}_i = \text{PT}_k\{1 - \exp[-(\text{DO}/S)^b]\},$$

where PT_k is the asymptotic PT_i of indicator species in bottom trawls as DO approaches infinity, S is a scale factor equal to the value of DO at which $\text{PT}_i = 0.63 \times \text{PT}_k$, and b is a shape factor (Pielou 1981; Prager et al. 1989). The Weibull function is a sigmoidal curve that depicts asymmetric ecological relationships (Pielou 1981). The Weibull model was fitted by use of the NLIN procedure in SAS (Gauss–Newton algorithm; Freund and Littell 2000), and 95% confidence intervals of the model parameters for each indicator species were compared to determine whether significant differences were indicated. If none of the three parameter estimates differed, the data were pooled to develop relationships among species exhibiting similar responses. A common relationship was possible since the Chesapeake Bay DO criteria that were the basis for the DO target and threshold were developed to protect a diverse array of aquatic living resources (Batiuk et al. 2009).

Logistic models tested the influence of IS on the odds that a given indicator species was present in a sample from shore zone (seine) or bottom channel (trawl) habitat (SAS 1995; Wright 1998). Five analyses (one per species) were conducted for each habitat. To isolate the influence of IS, DO was deliberately omitted from these logistic regressions.

Distance and regional abundance were added to the logistic models as indicators of migration and abundance, which could influence the presence of indicator species within the shore zone and bottom channel habitats of the subestuaries. Distances from the mouth of each subestuary to the center of major striped bass spawning areas or white perch nursery areas (Lippson 1973) were measured (Table 2). Potomac River subestuaries were assigned a distance from Potomac River spawning or nursery areas, and the remaining bay tributaries were assigned a distance from the head-of-bay spawning or nursery areas. Distance from the mouth of Chesapeake Bay was used to test whether the occupation of a site by spot and blue crabs was influenced by distance from marine waters (Table 2). Regional (Potomac River or head of bay) relative abundances of age-0 and age-1+ white perch, age-0 striped bass, and age-0 spot in the Maryland Juvenile Striped Bass Survey were estimated as geometric mean catches per seine haul (Table 3; Bonzek et al. 2007; Durell and Weedon 2010; E. Durell, MDDNR, personal communication). Regional indices for blue crabs were not available. Annual densities of all blue crab life stages in a Chesapeake Bay winter dredge survey were used as an index of baywide relative abundance (Table 3; Maryland Fisheries Service 2010b); this survey

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TABLE 2. Distance (km) from the mouth of each subestuary to (1) the mouth of Chesapeake Bay (marine), (2) the center of major regional (head of bay or Potomac River) striped bass spawning areas, or (3) the center of major regional white perch nursery areas.

Subestuary	Region	Marine	Striped bass	White perch
Magothy River	Head of bay	240.3	57.1	47.6
Severn River	Head of bay	229.8	67.6	58.1
South River	Head of bay	221.4	76.0	66.5
Rhode River	Head of bay	217.6	81.1	70.3
West River	Head of bay	216.9	80.5	71.0
Corsica River	Head of bay	261.0	82.1	70.3
Miles River	Head of bay	232.1	101.1	55.8
Breton Bay	Potomac River	165.6	99.5	49.6
St. Clements Bay	Potomac River	169.0	96.1	46.2
Wicomico River	Potomac River	178.3	86.7	36.9

is considered the primary indicator of Chesapeake Bay blue crab status by the Chesapeake Bay Stock Assessment Committee (Chesapeake Bay Program 2010).

Only main effects were considered in the logistic regression models of shore zone or bottom channel habitat occupation (i.e., we only considered the odds that an indicator species' presence in a shore zone or bottom channel sample was influenced by IS, distance, or regional abundance; we did not consider any potential interactions among these three factors). This analysis was conducted with the LOGISTIC procedure in SAS (SAS 1995).

RESULTS

Percent IS was highly correlated with the percentage of the watershed in urban land ($r = 0.97$, $P < 0.0001$), agriculture ($r = -0.81$, $P = 0.0085$), and wetland ($r = -0.78$, $P = 0.014$) but not with percent forest ($r = -0.23$, $P = 0.55$; Figure 2). Correlations between IS and the inverse of the proportion of land cover in agriculture ($r = -0.83$, $P = 0.0055$) or wetland ($r = -0.89$, $P < 0.0032$) were stronger, indicating that hyperbolic declines as IS increased were possible.

TABLE 3. Indices of relative abundance used in logistic regression analysis of indicator species and life stages in Chesapeake Bay regions. Relative abundance for fish is the geometric mean catch per standard seine haul (Durell and Weedon 2010); blue crab relative abundance is indicated by density (crabs/1,000 m^2) in Chesapeake Bay as estimated by a winter dredge survey (Maryland Fisheries Service 2010b).

Species and life stage	Region	Index 2003	2004	2005
White perch, age 0	Potomac River	20.1	5.6	6.4
	Head of bay	69.1	22.2	15.4
White perch, age 1+	Potomac River	3.2	4.7	2.0
	Head of bay	2.1	4.4	6.2
Striped bass, age 0	Potomac River	12.8	2.4	7.9
	Head of bay	11.9	4.2	8.5
Spot, age 0	Potomac River	0.5	0.7	1.9
	Head of bay	0.02	0.03	1.3
Blue crab, all ages	Chesapeake Bay	39.8	30.7	45.3

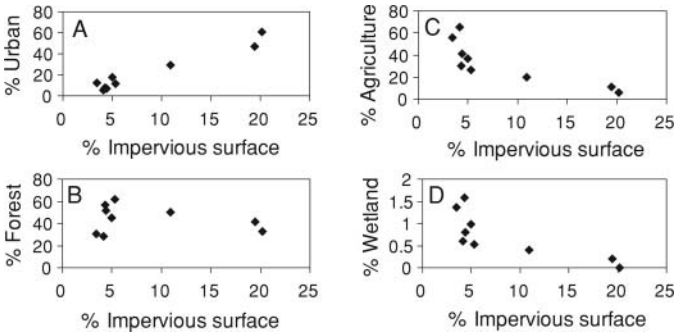

FIGURE 2. Land cover (%) as **(A)** urban land, **(B)** forest, **(C)** agriculture, and **(D)** wetland plotted in relation to the percentage of watershed in impervious surface for nine Chesapeake Bay subestuaries.

Mean surface DO was not significantly correlated with landscape variables, mean salinity, or mean temperature (Table 4). Mean bottom DO was significantly ($P < 0.0001$) and negatively correlated with IS ($r = -0.82$), urban land cover percentage ($r = -0.78$), and bottom depth ($r = -0.81$); mean bottom DO was significantly and positively correlated with the percentage of

TABLE 4. Correlations between (1) percentage of watershed in impervious surface (IS), land cover variables (percent urban, agriculture, forest, and wetland), mean annual salinity (‰; all depths), mean annual temperature (°C; all depths), or mean annual bottom depth (m) and (2) mean annual surface dissolved oxygen (DO; mg/L), mean annual bottom DO, or mean annual bottom depth in Chesapeake Bay subestuaries. All water quality variables were measured during July–September. Asterisks indicate significance ($P < 0.05$) after Bonferroni adjustment for multiple comparisons ($N = 23$ for each comparison).

Variable	Statistic	Surface DO	Bottom DO	Bottom depth
IS	*r*	0.29	−0.82	0.88
	P	0.19	<0.0001*	<0.0001*
Urban	*r*	0.27	−0.78	0.84
	P	0.21	<0.0001*	<0.0001*
Agriculture	*r*	−0.53	0.63	−0.77
	P	0.01	0.001*	<0.0001*
Forest	*r*	0.33	0.16	0.01
	P	0.12	0.48	0.97
Wetland	*r*	−0.28	0.73	−0.59
	P	0.20	<0.0001*	0.003*
Salinity	*r*	−0.42	0.12	0.08
	P	0.05	0.59	0.73
Temperature	*r*	−0.39	−0.23	0.05
	P	0.07	0.28	0.84
Bottom depth	*r*	0.21	−0.81	
	P	0.33	<0.0001*	

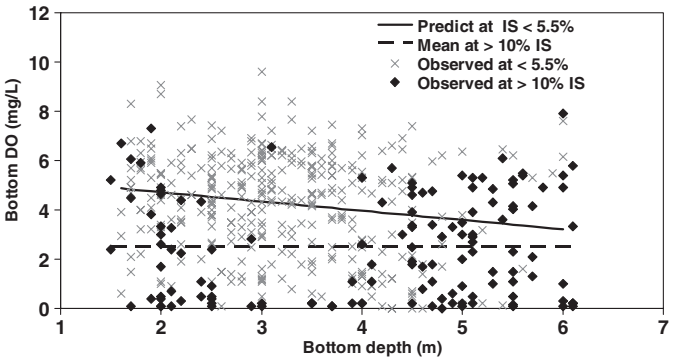

FIGURE 3. Observed bottom dissolved oxygen (DO; measured during July–September 2003–2005) versus bottom depth in Chesapeake Bay subestuaries with less than 5.5% of the watershed in impervious surface (IS; gray ×) and subestuaries with greater than 10.0% IS (black diamonds). Trends in DO are indicated by lines. Analysis was based only on depths that were common to both subestuary types (low and high IS).

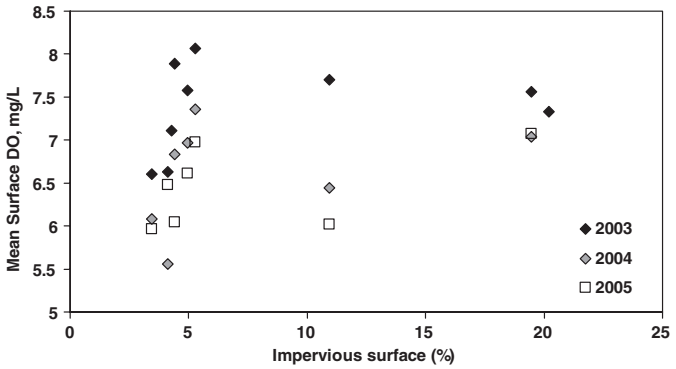

FIGURE 4. Plot of annual mean surface dissolved oxygen (DO; measured during July–September 2003–2005) versus percentage of watershed in impervious surface for Chesapeake Bay subestuaries.

the watershed in agriculture ($r = 0.63$) and wetland ($r = 0.73$). Mean bottom depth had significant correlations with the same land use variables as mean bottom DO (Table 4).

Bottom depth explained little variation in bottom DO measurements within rural or suburban subestuaries at depths that were common to both subestuary types (1.5–6.1 m). The relationship between bottom depth and bottom DO was significant, weak, and negative when IS was low ($r^2 = 0.028$, $P < 0.002$, $N = 343$; Figure 3). Average decline in bottom DO (DO_B) with bottom depth (B; m) for rural subestuaries was described by the equation

$$DO_B = (-0.37 \times B) + 5.46.$$

The SE was 0.12 for the slope and 0.40 for the intercept. Bottom DO was predicted to decline from 4.8 mg/L at a bottom depth of 1.5 m to 3.2 mg/L at a bottom depth of 6.1 m when IS was low. No relationship between bottom depth and DO was detected when IS was high ($r^2 = 0.002$, $P = 0.59$, $N = 133$); bottom DO averaged 2.5 mg/L at bottom depths between 1.5 and 6.1 m (Figure 3).

Among-year differences in the slope of the relationship between mean surface DO or mean bottom DO and IS were not detected (i.e., year × IS interaction term was not significant). Subsequent ANCOVAs of mean surface DO ($R^2 = 0.45$, $F = 5.60$, $P = 0.0063$) or mean bottom DO ($R^2 = 0.79$, $F = 24.46$, $P < 0.0001$) with IS and year were significant; each analysis had 9 model df and 13 error df. Year had a significant influence on surface DO ($F = 6.93$, $P = 0.005$), but IS did not ($F = 1.90$, $P = 0.18$; Figure 4). Significant effects of both IS ($F = 61.95$, $P < 0.0001$) and year ($F = 5.53$, $P = 0.013$) on mean bottom DO were detected (Figure 5). The relationship between IS and mean bottom DO in 2005 was described by the equation

$$\text{Mean DO}_B = (-0.17 \times \text{IS}) + 4.58.$$

The common slope and 2005 intercept were significant (pairwise t-tests: $P < 0.0001$; Littell et al. 2002); the SE of the slope was 0.02, and the SE of the intercept was 0.29. The additional effect of year on the 2005 intercept estimated for 2003 (intercept = 0.57; SE = 0.31) was marginally significant ($P = 0.08$). The additional effect estimated for 2004 was more pronounced (intercept = 1.09; SE = 0.33) and was significant ($P = 0.0036$; Figure 4).

At an IS value of 5.5% or lower ($N_{total} = 391$), the probability that bottom DO would meet or exceed the 5-mg/L target criterion was 0.42 (SD = 0.02), and the probability that bottom DO would be at or below the 3-mg/L threshold was 0.25 (SD = 0.03). At an IS level of 10% or more ($N_{total} = 179$), the probability that bottom DO would meet or exceed the target was 0.14 (SD = 0.03), and the probability that bottom DO would be at or below the threshold was 0.63 (SD = 0.10).

Confidence intervals of Weibull function parameters describing relationships between the PT_i for each indicator species and bin midpoints for bottom DO often overlapped among species. In some instances, parameters were not well estimated (not different from zero), reflecting low df ($N = 8$ for each species in

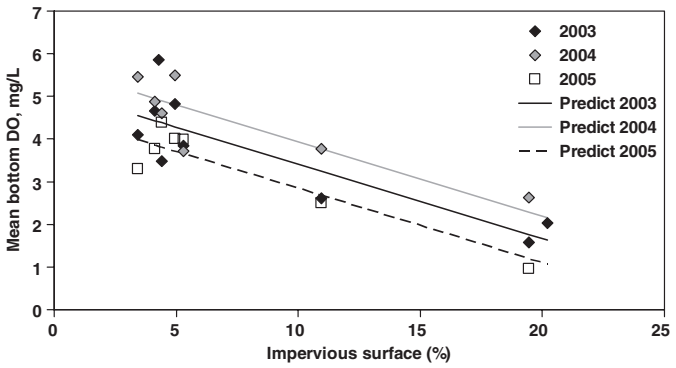

FIGURE 5. Relationships between annual mean bottom dissolved oxygen (DO; measured during July–September 2003–2005) and percentage of watershed in impervious surface for Chesapeake Bay subestuaries (ANCOVA: $P < 0.05$). Annual relationships are indicated by lines.

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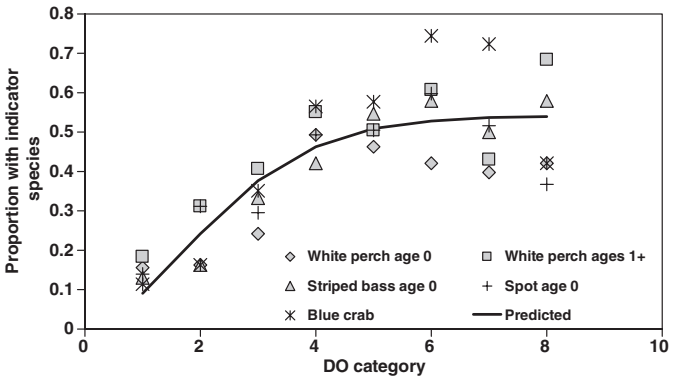

FIGURE 6. Observed (symbols) and predicted (solid line; Weibull function) proportions of tows containing various indicator species in relation to bottom dissolved oxygen (DO; mg/L) category midpoint for Chesapeake Bay subestuaries.

a three-parameter model). Therefore, the five sets of PT_i values at each DO bin (i.e., 1 set/indicator species) were pooled into a single relationship. The relationship between PT_i and bottom DO bin midpoint for all indicator species combined was described by

$$PT_i = 0.54 \times \{1 - \exp[-(DO/2.71)^{1.70}]\}$$

($R^2 = 0.85$, $P < 0.0001$, $N = 40$; Figure 6). The approximate SEs for PT_k, S, and b were 0.03, 0.27, and 0.52, respectively. Predicted PT_i declined steadily from 0.38 at the 3-mg/L DO threshold to 0.09 at 1 mg/L. Predicted PT_i increased little with increasing DO beyond the 5-mg/L target ($PT_i = 0.51$ at 5 mg/L; $PT_i = 0.54$ at 8 mg/L; Figure 6). Pooling of the data across indicator species assumed a common spatial distribution among subestuaries and a common response to DO. The high amount of variation in PT_i explained by bottom DO supported the assumption of a common response among the indicator species.

The assumption of a common spatial distribution, which was needed to pool PT_i across indicator species, was supported by high site occupation. The percentage of sites where an indicator species was encountered at least once was 89% for spot, 97% for age-1+ white perch, and 100% for blue crabs, age-0 white perch, and age-0 striped bass. Sustained absence in bottom channel waters would largely represent a loss of suitable habitat rather than habitat that was unsuitable to begin with.

Logistic regressions of indicator species presence in the shore zone versus IS, regional abundance, and distance did not detect a negative influence of IS (Table 5). A significant ($P < 0.0001$) positive influence of IS on the odds of age-0 white perch and all stages of blue crabs being present was detected (odds ratio = 1.07 for both). An influence of IS on shore zone occupation was not detected for the remaining indicator species. Regional relative abundance had a significant ($P < 0.0001$) positive influence on the indicator fish species but not on blue crabs ($P = 0.54$). Distance from a spawning area had a significant negative influence on the presence of age-0 and age-1+ white perch but not on the presence of age-0 striped bass ($P = 0.61$). Distance from the mouth of Chesapeake Bay exerted a significant ($P < 0.0001$) negative influence on blue crab presence in the shore zone but did not significantly influence age-0 spot presence ($P = 0.13$; Table 5).

Logistic regressions of indicator species presence in bottom channel samples versus IS, regional abundance, and distance detected a significant (P ranged from <0.0001 to 0.0004) negative influence of IS (i.e., these species were more likely to be present in bottom waters as IS decreased; Table 6). Odds ratios indicated that the IS effect on presence was greatest for age-0 white perch (odds ratio = 0.73) and lowest for age-1+ white perch (odds ratio = 0.94). Presence of age-0 spot, age-0 striped bass, and blue crabs in bottom channel samples was influenced similarly by IS (odds ratio = 0.83–0.84). Regional abundance had a significant positive influence (P ranged from <0.0001 to 0.028) in all five sets of logistic regressions for bottom channel habitat. Distance exerted a significant negative influence on the presence of age-0 and age-1+ white perch, a significant positive influence on spot presence (all $P < 0.0001$), and no influence on the presence of age-0 striped bass ($P = 0.12$) or blue crabs ($P = 0.25$; Table 6).

DISCUSSION

Impervious surface was the landscape feature that was best associated with degraded bottom habitat in nine brackish subestuaries of Chesapeake Bay. Mean bottom DO in a subestuary was negatively influenced by IS, but mean surface DO was not. In bottom waters within low-IS (≤5.5%) watersheds, the 5-mg/L DO target was unlikely to be met, on average, but the 3-mg/L threshold was avoided. In bottom waters within high-IS (≥10%) watersheds, the DO threshold was unlikely to be avoided. Rapid declines in PT_i from about 0.4 to 0.1 occurred when DO declined from the 3-mg/L threshold to 1 mg/L, and PT_i remained at about 0.50 at DO concentrations beyond the 5-mg/L target. The odds that bottom channel habitat would be occupied by indicator species was reduced by IS, but IS did not negatively influence indicator species' presence in the shore zone.

Annual differences among mean bottom DO estimates in the same subestuary were substantial, but these differences did not affect our interpretation of the influence of IS. Mean bottom DO did not fall below the 3-mg/L DO threshold when IS was less than 5.5%; mean bottom DO did not rise above the threshold when IS was approximately 20%. At 11% IS, one mean DO value was above 3 mg/L, while two means were below 3 mg/L.

Development since 1950 has added a suburban landscape layer to the Chesapeake Bay watershed (Brush 2009). Land use typical of rural areas (farms, wetlands, and forests) has been converted to residential and industrial uses (Wheeler et al. 2005; NRC 2009). Fish habitat quality in brackish subestuaries, as indicated by mean bottom DO during summer, was positively

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TABLE 5. Parameters from logistic regressions ($N = 520$ for each regression) of indicator species presence in the shore zone (seine samples) of Chesapeake Bay subestuaries versus (1) the percentage of watershed in impervious surface (IS), (2) distance (km) from a major spawning area (striped bass) or nursery area (white perch) or from the mouth of Chesapeake Bay (spot or blue crabs), or (3) regional relative abundance indices (see Table 3). Lower and upper 95% confidence limits (CLs) for the odds ratio are presented.

Parameter	Estimate	SE	Wald χ^2	Wald χ^2 *P*-value	Odds ratio	Lower 95% CL	Upper 95% CL
				White perch, age 0			
Intercept	1.37	0.53	6.60	0.0102			
IS	0.07	0.02	9.84	0.0017	1.07	1.03	1.12
Distance	−0.10	0.02	34.40	<0.0001	0.90	0.87	0.93
Index	0.09	0.01	60.31	<0.0001	1.09	1.07	1.11
				White perch, age 1+			
Intercept	1.34	0.50	7.15	0.0075			
IS	−0.01	0.02	0.56	0.454	0.99	0.96	1.02
Distance	−0.07	0.01	22.24	<0.0001	0.93	0.91	0.96
Index	0.47	0.07	46.28	<0.0001	1.60	1.40	1.83
				Striped bass, age 0			
Intercept	−0.40	1.87	0.04	0.8323			
IS	−0.01	0.04	0.06	0.8075	0.99	0.92	1.07
Distance	0.01	0.03	0.26	0.6089	1.02	0.96	1.08
Index	0.17	0.03	23.97	<0.0001	1.19	1.11	1.27
				Spot, age 0			
Intercept	−2.88	0.76	14.51	0.0001			
IS	−0.02	0.02	0.70	0.4042	0.984	0.947	1.022
Distance	0.01	0.01	2.29	0.1299	1.009	0.997	1.02
Index	1.92	0.18	110.50	<0.0001	6.809	4.762	9.737
				Blue crab, all ages			
Intercept	5.05	0.97	26.88	<0.0001			
IS	0.07	0.02	17.96	<0.0001	1.074	1.039	1.109
Distance	−0.04	0.01	41.41	<0.0001	0.966	0.955	0.976
Index	−0.01	0.02	0.38	0.5367	0.989	0.956	1.024

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associated with two indicators of rural land use (percent land cover in agriculture and wetland) and was negatively associated with IS and urban land use estimates. Changes in bottom DO with IS in Chesapeake Bay subestuaries generally agreed with findings elsewhere that habitat quality in fluvial and tidal streams declines with IS and becomes degraded at IS greater than 10% (Arnold and Gibbons 1996; Cappiella and Brown 2001; Beach 2002; Holland et al. 2004; NRC 2009).

Smaller Hudson River watersheds (<40 km^2) appeared to be more susceptible to capture by urban sprawl than larger ones (Limburg and Schmidt 1990), so the reader is cautioned against applying the DO–IS results from these smaller tributaries to larger tributaries or to the entire Chesapeake Bay. Hypoxia already occurs over an extensive area of Chesapeake Bay, but its watershed is still largely in forest cover and agriculture represents the largest human land use (Hagy et al. 2004; Kemp et al. 2005). Historically, nutrient pollution has been attributed to agriculture, but urban land may produce greater loading on a per unit basis (Kemp et al. 2005; Wheeler et al. 2005; NRC 2009). Turf cover is an important component of suburban development that may now constitute a larger acreage than row crops, pasture, or freshwater wetlands in the Chesapeake Bay watershed (Schueler 2010). Accurate data on application of fertilizer, pesticides, irrigation, and turf management are lacking, but nitrogen input from lawns is potentially high (Schueler 2010). Nitrogen processing may become reduced in suburban watersheds as riparian zones and floodplains become disconnected from stream channels by stormwater management, and increased IS causes groundwater recharge to lessen and soils to dry out (Craig et al. 2008; Kaushal et al. 2008; Brush 2009; NRC 2009). Depletion of DO in response to increased IS may be reinforced by benthic release of nitrogen and phosphorus into overlying subestuary waters (Kemp et al. 2005).

Nutrient loading from agricultural fertilizers is considered a large influence on hypoxia in main-stem Chesapeake Bay (Kemp et al. 2005; Brush 2009), but mean bottom DO in the set of subestuaries we studied was positively correlated with the percentage of watershed in agriculture. This divergence of DO response between the main stem and subestuaries may reflect differences in watershed size, nutrient loading, and nutrient

TABLE 6. Parameters for logistic regressions ($N = 588$ for each regression) of indicator species presence in midchannel bottom habitat (trawl samples) of Chesapeake Bay subestuaries versus (1) the percentage of watershed in impervious surface (IS), (2) distance (km) from a major spawning area (striped bass) or nursery area (white perch) or from the mouth of Chesapeake Bay (spot or blue crabs), or (3) regional relative abundance indices (see Table 3). Lower and upper 95% confidence limits (CLs) for the odds ratio are presented.

Parameter	Estimate	SE	Wald χ^2	Wald χ^2 P-value	Odds ratio	Lower 95% CL	Upper 95% CL
White perch, age 0							
Intercept	4.34	0.59	53.41	<0.0001			
IS	−0.32	0.03	84.80	<0.0001	0.73	0.68	0.78
Distance	−0.18	0.02	73.74	<0.0001	0.84	0.81	0.87
Index	0.09	0.01	108.75	<0.0001	1.09	1.07	1.11
White perch, age 1+							
Intercept	3.45	0.47	54.41	<0.0001			
IS	−0.06	0.02	12.73	0.0004	0.94	0.91	0.97
Distance	−0.13	0.01	77.24	<0.0001	0.88	0.85	0.90
Index	0.26	0.07	15.72	<0.0001	1.30	1.14	1.47
Striped bass, age 0							
Intercept	1.92	1.21	2.50	0.1137			
IS	−0.18	0.03	33.58	<0.0001	0.84	0.79	0.89
Distance	−0.03	0.02	2.43	0.1191	0.97	0.94	1.01
Index	0.06	0.03	4.83	0.028	1.06	1.01	1.11
Spot, age 0							
Intercept	−3.28	0.72	20.68	<0.0001			
IS	−0.17	0.02	51.56	<0.0001	0.84	0.80	0.88
Distance	0.02	0.01	19.28	<0.0001	1.02	1.01	1.03
Index	1.80	0.19	92.23	<0.0001	6.07	4.20	8.78
Blue crab, all ages							
Intercept	1.88	0.85	4.85	0.0277			
IS	−0.18	0.02	71.48	<0.0001	0.83	0.80	0.87
Distance	0.01	0.00	1.32	0.2513	1.01	1.00	1.01
Index	−0.04	0.02	4.74	0.0295	0.97	0.94	1.00

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processing by wetlands and streams. We did not measure nutrient loads, but all nine subestuaries were considered to have excessive nutrient loads (Maryland Department of Environment 2008). Wetlands buffer eutrophication by trapping and assimilating nutrients (Kemp et al. 2005), and agricultural watersheds had more area in wetlands. Wetlands constituted 0.0–0.4% of watershed area in the three suburban subestuaries, 0.6–1.4% in the two watersheds where agriculture was the dominant land use, and 0.5–1.6% in the remaining rural watersheds. Agricultural watersheds should lack stormwater structures that disconnect streams from their interface with riparian zones, which serve as "hot spots" for denitrification (Kaushal et al. 2008). Negative correlation between agricultural land use and IS indicated that watersheds with major amounts of agriculture will undergo a lesser extent of IS-related hydrological changes that increase downstream transport of nitrogen and reduce denitrification (Kaushal et al. 2008).

Absence of a significant positive association between IS and forest cover in this study may reflect the limited number of watersheds sampled, the remnant nature of forest cover adjacent to Chesapeake Bay, inability to account for forest fragmentation in our analyses, and difficulty in separating forest and turf cover hidden below tree canopies within suburban and exurban regions (Breitburg et al. 1998; Brush 2009; Schueler 2010).

Within and near DO-depleted waters, fish and mobile macroinvertebrates experience increased mortality, altered trophic interactions, and impaired reproduction, immune responses, and growth (Haeseker et al. 1996; Engel and Thayer 1998; Breitburg et al. 2002, 2009; Evans et al. 2003; Rudolph et al. 2003; Baird et al. 2004). Once severe hypoxia becomes established, fish yields and abundances plummet (Breitburg 2002). Persistent hypoxia in suburban Chesapeake Bay subestuaries was identified as a factor that precluded recovery of yellow perch after long-term harvest prohibitions (Uphoff et al. 2005).

Additional IS-related stressors exist that are not associated with hypoxia, and the negative effects of multiple stressors usually exceed those of the worst single stressor alone (Breitburg et al. 1998; Folt et al. 1999). Development leads to altered hydrologic features in streams that provide spawning habitat for anadromous fish (Limburg and Schmidt 1990; Konrad

and Booth 2005). Altered hydrology and groundwater recharge associated with development to 20% IS were implicated as causes of potentially lethal salinity levels for yellow perch eggs and larvae in the upper estuarine reach of the Severn River, Maryland (Uphoff et al. 2005). Significant polychlorinated biphenyl concentrations in white perch were closely related to IS in 14 Chesapeake Bay subestuaries (King et al. 2004). Anthropogenic chemicals like polychlorinated biphenyls disrupt endocrine function associated with reproduction in fishes; are associated with depressed survival, malformation, and abnormal chromosome division in eggs and larvae; and are associated with reduced growth and survival skills in larvae (Longwell et al. 1992, 1996; Colborn and Thayer 2000; McCarthy et al. 2003).

We propose a general ISRP framework that could be considered when managing common estuarine resident species and their habitat in Chesapeake Bay and its subestuaries. Sensitive species or specific habitats may require greater protection from development than is provided by these guidelines for estuarine waters (Stranko et al. 2008).

In rural subestuary watersheds (IS $\leq$ 5.5%), fish habitat would generally be considered unimpaired and management actions that deal with harvest or reintroduction would be most appropriate. Preserving watersheds at this level of IS would be a viable fisheries management strategy. Setting this level of IS as a target is not based on attainment of target DO conditions. Rather, the use of this target acknowledges that these low levels of IS are attainable along the Maryland portion of Chesapeake Bay given its development history. Thus, the target IS largely avoids threshold DO conditions.

Above an IS of 10% (suburban landscape), habitat stress mounts and comprehensive watershed management strategies (stormwater management, sewage treatment, riparian buffers, stream and wetland restoration, etc.) become vital. Unfortunately, we were not able to find examples of the successful restoration of estuarine habitat that had been degraded by watershed development. Opportunities to improve conditions may involve innovative stormwater management (NRC 2009), creation of wet, marshy conditions throughout watersheds (Brush 2009), and reconnection of streams to riparian areas (Kaushal et al. 2008). Lovell and Johnston (2009) identified opportunities to upgrade human-dominated landscapes through seminatural landscape elements. Palmer (2009) emphasized science-based prioritization schemes for restoration that focus on processes and limiting factors at whole-watershed scales.

Managers may be faced with choosing watersheds upon which to target restoration efforts, and watersheds that are closer to the IS target should be more likely respond positively to remediation of limiting factors. This remediation may result in a favorable regime of habitat conditions for fisheries but will not necessarily yield the exact mix of species that occurred prior to development. As IS increases well beyond 10%, the presence of strong multiple stressors makes it less likely that remediation will eliminate or significantly reduce habitat stress. Restoration efforts should also consider future plans for watershed development. The interim between target and threshold IS appears to be brief, and the lack of watersheds between 5.5% and 10.0% IS in our study was not a choice on our part.

Several methods of estimating IS are available (Cappiella and Brown 2001; NRC 2009), and ISRPs developed here may not be compatible with IS estimates made by different techniques. However, targets and thresholds should exist that are relative to the technique used to estimate IS.

As the proportion of watershed that is affected by development increases, the effectiveness of fisheries management shifts from harvest control to landscape management, habitat conservation, and restoration. In the Chesapeake Bay region, many of these responsibilities now lie with agencies that are not involved in fisheries management. Fisheries managers need to effectively and openly communicate the potential for quality of life, sustainability, and services (fish, fishing opportunities, and ecological services) to be lost because of habitat degradation. Such communication will allow stakeholders, responsible agencies, and governing bodies to make informed, overt decisions about trade-offs between development and conservation of the rural landscapes needed to support fisheries.

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