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Watershed Land Use Is Strongly Linked to PCBs in White Perch in Chesapeake Bay Subestuaries

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We related total PCBs (t-PCBs) in white perch (*Morone americana*), an abundant estuarine resident that supports a valuable recreational and commercial fishery in the mid-Atlantic region, to the amount and spatial arrangement of developed land in watersheds that discharge into 14 subestuaries of Chesapeake Bay. We considered the intensity of development in watersheds using four developed land-use measures (% impervious surface, % total developed land, % high-intensity residential + commercial [%high-res/comm], and % commercial) to represent potential source areas of PCBs to the subestuaries. We further evaluated the importance of source proximity by calculating three inverse-distance weighted (IDW) metrics of development, an approach that weighted developed land near the shoreline more heavily than developed land farther away. Unweighted percentages of each of the four measures of developed land explained 51−69% of the variance in t-PCBs. However, IDWs markedly improved the relationships between % developed land measures and t-PCBs. Percent commercial land, weighted by its simple inverse distance, explained 99% of the variance in t-PCBs, whereas the other three measures explained as much as 93−97%. PCBs historically produced or used in commercial and residential areas are apparently persisting in the environment at the scale of the watersheds and subestuaries examined in this study, and developed land close to the subestuary has the greatest unit effect on t-PCBs in fish. These findings provide compelling evidence for a strikingly strong linkage between watershed land use and t-PCBs in white perch, and this relationship may prove useful for identifying unsampled subestuaries with a high risk of PCB contamination.

Introduction

Polychlorinated biphenyls (PCBs) are a group of organochlorine compounds that resist degradation in the environment and are widely distributed in aquatic ecosystems. PCBs are highly lipophilic and bioaccumulate in lipid-rich tissues of biota. Because of their toxicity, PCBs present a health risk to both humans and a variety of other organisms (*1*, *2*). Although banned from production in the United States in 1979, PCB levels in many aquatic ecosystems remain sufficiently high to contaminate food webs and cause consumption advisories for a wide range of valuable fish and shellfish species (*3*).

Major sources of PCBs in estuaries are thought to be legacy pools of past point-source releases by manufacturing and from nonpoint sources associated with the general use, storage, and disposal of these persistent compounds (*4*, *5*). The sources, spatial extent, and magnitude of PCB contamination is well-known for some estuaries (e.g., Hudson River Estuary), and recent efforts to link PCB levels in resident and migratory fish to specific areas of contamination have been quite successful (*6*, *7*). However, estuaries are hydrologically open systems affected by long-distance transport of contaminants from upstream and downstream areas, so the distribution of PCBs in many estuaries is spatially heterogeneous and difficult to predict. For example, in Chesapeake Bay, highly industrial areas such as Baltimore Harbor are known hotspots for PCBs due to decades of manufacturing-related discharges into the environment (*8*). However, less-industrial subestuaries of Chesapeake Bay currently have consumption advisories posted for several fish species due to PCB contamination, suggesting that more diffuse, nonpoint sources of PCBs may be important (*9*, *10*) or that other factors may be involved in contaminating biota in these small tributaries.

Watershed characteristics are being increasingly used as indicators of physical, chemical, and biological endpoints in aquatic systems (*11*, *12*). Many investigators have successfully linked watershed land use to ecosystem health of freshwater streams (*13*, *14*), demonstrating watershed analysis to be a practical, inexpensive alternative to ground-based monitoring, particularly at broad geographic scales. Although estuarine examples are fewer in number, linkages between watershed land use and downstream estuarine conditions have also been demonstrated (*15*−*17*). In a particularly relevant study, Comeleo et al. (*18*) sampled sediments in 26 subestuaries of Chesapeake Bay and found that concentrations of heavy metals and organic contaminants were strongly correlated to the amount of developed land within watersheds discharging into subestuaries. Predictions of contaminants were improved using a distance-weighting approach that accounted for distances between developed land in watersheds and sampling stations in the subestuaries. These results suggest that quantification of land-use patterns in watersheds may be useful for predicting PCB contamination in estuarine ecosystems.

We tested the hypothesis that the amount and spatial arrangement of developed land in watersheds would be significantly linked to concentrations of total PCBs (t-PCBs) in biota from subestuaries of Chesapeake Bay. We examined (a) the strength of correlations between different measures of developed land in the watershed and t-PCBs and (b) the relative improvement in our predictions of t-PCBs afforded by inverse distance-weighting developed land to account for proximity to the subestuaries. We focused on t-PCBs in white perch (*Morone americana* Gmelin), a widely distributed estuarine resident that supports a valuable commercial and recreational fishery throughout Chesapeake Bay. White perch are an ideal indicator species for detecting watershed linkages to PCBs because they spend most of their lives within or near

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TABLE 1. Watershed Area and Percent of the Watershed in Developed Land Uses for the 14 Subestuaries of Chesapeake Bay Sampled for t-PCBs in White Perch in 2002

subestuary	watershed area (km^2)	subestuary area (km^2)	% impervious	% total developed	% high-res/ comm	% commercial	t-PCBs (ng/g wet weight)[a]
Back	144	18	35.6	66.5	15.2	13.9	310.2
Battle	46	2	0.7	4.5	0.2	0.2	8.2
Bird	66	3	18.0	26.8	7.8	7.2	101.3
Langford	97	13	1.3	0.8	0.5	0.5	12.2
Patapsco, Upper	662	14	24.6	41.1	14.5	9.7	623.9
Piankatank	529	37	0.4	1.0	0.3	0.3	3.4
Severn	172	36	13.5	25.5	6.5	5.9	95.6
Southeast	141	2	0.9	0.8	0.2	0.2	8.3
St. Clements	119	14	1.2	4.7	0.5	0.5	19.6
St. Mary's	182	37	2.8	6.8	1.6	1.6	18.1
Totuskey	171	2	0.7	0.7	0.3	0.3	7.1
Tred Avon	96	26	5.6	11.4	2.4	2.3	38.2
Warwick	106	9	13.3	26.9	8.8	8.5	31.2
Wye	205	27	1.1	2.4	0.7	0.7	23.6

[a] White perch fillets; mean concentration of two composites for some subestuaries; others represent only 1 composite of 4 or 5 fish.

specific subestuaries (*19*). White perch also prey upon small fish and epibenthic invertebrates, consumers of allochthonous detritus running off the land and accumulating in sediments. Moreover, white perch are semianadromous, moving into freshwater tributaries to spawn and back down into the subestuaries as nursery and feeding habitat, so their life cycle spans a zone that continuously exposes them to runoff from the watershed. Finally, because PCB-related consumption advisories have recently been posted for several subestuaries and many other locations have yet to be assessed, there is great interest in developing geographical indicators of PCBs in this region. This is one of the first attempts to quantitatively link land use across multiple watersheds to PCBs in aquatic biota (*20*) and the first in an estuarine ecosystem.

Methods

Sample Collection. White perch were collected from 14 subestuaries of Chesapeake from 1 July−3 September 2002 (Figure 1). Subestuaries were selected to span a range of watershed land use from primarily forested or agricultural to highly developed (Table 1). We selected subestuaries to span a gradient of development because we hypothesized that developed land would be a more significant source of PCBs than other types of land use. Watersheds were also selected to be relatively similar in size, ranging from 46 to 662 km^2, although most were approximately 100−200 km^2 (Table 1). Greater details are provided in ref *21*.

White perch were collected from six sampling stations distributed throughout the middle-to-upper reaches of each subestuary. Individual subestuaries were the observational units for analysis, so white perch were collected at multiple locations across the subestuary to integrate t-PCB levels at the subestuary scale. Stations were located within 10−50 m of the shoreline, and white perch were collected using a pair of fyke nets employed for 24 h at each station. Two composites of 4 or 5 legal-sized fish were collected from 11 of the 14 subestuaries (≥200 mm was the legal minimum size at the time of collection, but current sportfishing regulations in Maryland impose no size restrictions on white perch). One composite was collected from the remaining three subestuaries (Bird, Piankatank, and Warwick). Fish <200 mm were included in one of the two composites from Southeast Creek (mean ± 1 SD: length = 178 ± 17 mm; weight = 159 ± 40 g; n=5). For all other composites, mean length (mm) and weight (g) ranged from 205 to 268 and from 126 to 255, respectively.

PCB Congener Analysis. Whole white perch fillets (skin-on) from all individuals in each composite were homogenized for analysis of 85 PCB congeners. Fish were analyzed skin-on in accordance with MDE and U.S. EPA protocols for fish tissue analysis for evaluating risk to human consumption, and fish handling and preparation for analysis followed U.S. EPA (*22*). Congener analyses were conducted by the University of Maryland Center for Environmental Science-Chesapeake Biological Laboratory (UMCES-CBL) using methods previously described by Kucklick et al. (*23*). t-PCBs (ng/g wet weight) were calculated by summing concentrations of all congeners detected above the minimum detection limit.

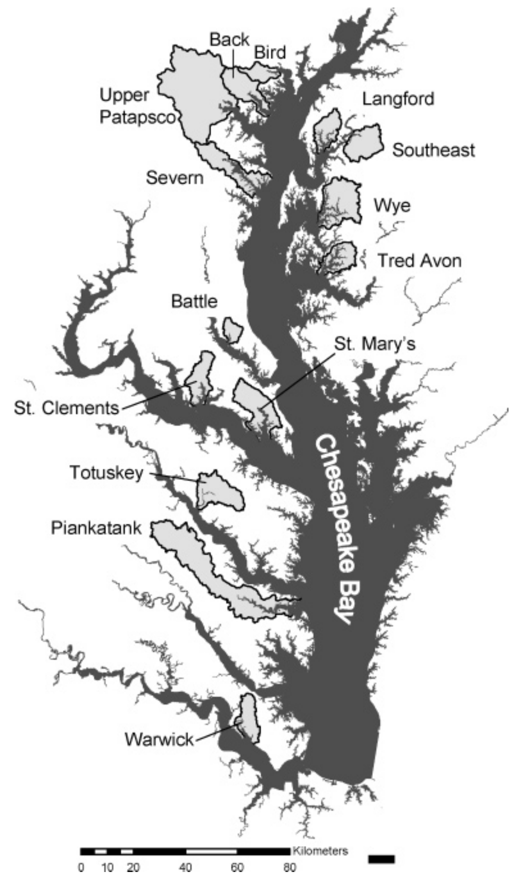

FIGURE 1. Names and locations of the 14 subestuaries and their watersheds.

Watershed Delineation and Land-Use Analysis. Watershed boundaries around each subestuary were delineated manually using a 1:24 000 digital elevation model (DEM) expressed as a 30-m raster (USGS National Elevation Data set, www.usgs.gov). Percentages of each watershed covered by developed land were calculated using the RESAC impervious surface map (*24*) and National Land Cover Data (NLCD) land cover map (*25*). These two maps are raster data sets developed from 30-m Landsat thematic mapper images taken during 1999−2000 and 1992, respectively. Watershed delineation and land-use analysis were accomplished in ArcGIS 8.2 (ESRI, Inc., Redlands, CA). Four different representations of developed land were considered: (1) impervious-surface cover (impervious), which was the sum across all pixels of the fraction of impervious surface in each pixel, (2) total cover of developed land (total developed), defined as the sum of NLCD low- and high-intensity residential and commercial cells, (3) the sum of NLCD high-intensity residential and commercial cells (high-res/comm), and (4) the sum of NLCD commercial cells (commercial) (Table 1). Percent impervious land is a metric of urban runoff potential (*26*) and a weighted index of both the amount and intensity of development in a watershed. The three NLCD development metrics were considered separately to test the hypothesis that commercial and/or high-res/comm land would be the strongest correlates of t-PCBs when compared to total developed land, which included low-intensity residential land, which is a less likely source area for t-PCBs.

Inverse-distance weights (IDWs) were applied to each of the four developed land-use measures to test the hypothesis that development closer to the subestuary would be a stronger predictor of t-PCBs than development farther away. Linear distance (m) was calculated between each cell and the shoreline of the subestuary for all land-cover cells in a watershed (Figure 2). Counts of distances were aggregated into unequal interval distance classes: 0−250 m, 251−500, 501−1000, 1001−2000, 2001−5000, 5001−10 000, and >10 000. The ranges were wider for greater distances because the inverse distance functions are less sensitive to distance at large distances than at small ones (Figure 2). The highest distance in each range was used to represent all cells within the range except for the >10 000 range, which was represented by a distance of 25 000 m. Three IDWs were applied: (1) $d^{-0.5}$, the square root of the inverse distance (2) d^{-1}, inverse distance, and (3) d^{-2}, the inverse distance squared. Of the IDWs, d^{-2} most strongly emphasized developed land close to subestuary shorelines (Figure 2). The equation for calculating inverse-distance-weighted percent developed land is as follows

$$\text{IDW \% developed land} = 100\cdot\sum_{i=1}^{C} n_D W_C / \sum_{i=1}^{C} n_T W_C \quad (1)$$

where C is the number of distance classes, n_D is the number of developed cells in distance class i, W_C is the inverse-distance weight for distance class i where d = the maximum distance between a cell in distance class i and the subestuary (e.g., 0−250 m distance class was assigned a distance of 250 m), and n_T is the total number of land-cover cells in distance class i. See ref *27* for greater details.

Data Analysis. Simple linear regression was used to test for significant (p<0.05) relationships between unweighted or distance-weighted developed land-use measures and t-PCBs in white perch among the 14 subestuaries. Mean t-PCBs were used as observations for subestuaries containing two composites because we were interested in spatially integrated estimates of t-PCB levels at the subestuary scale (Table 1).

A few regressions using simple percentages exhibited mild departures from normality in the residuals (Shapiro-Wilk test, 0.01<p<0.05). However, we chose to conduct and report all regression models using untransformed data because (a) we hypothesized that increases in watershed developed land would have a unit effect (linear) on t-PCBs, (b) departures from normality have a trivial effect on significance levels in highly significant regressions that explain very large amounts of variance (*28*), and (c) transformed data would have confounded the effect of distance weighting in the regressions. We expected that distance weighting would improve the linear relationship (and concomitantly the assumption of normality of residuals) between developed land and t-PCBs because we hypothesized that the spatial distribution of subestuary development was a potential cause of variation in the analyses.

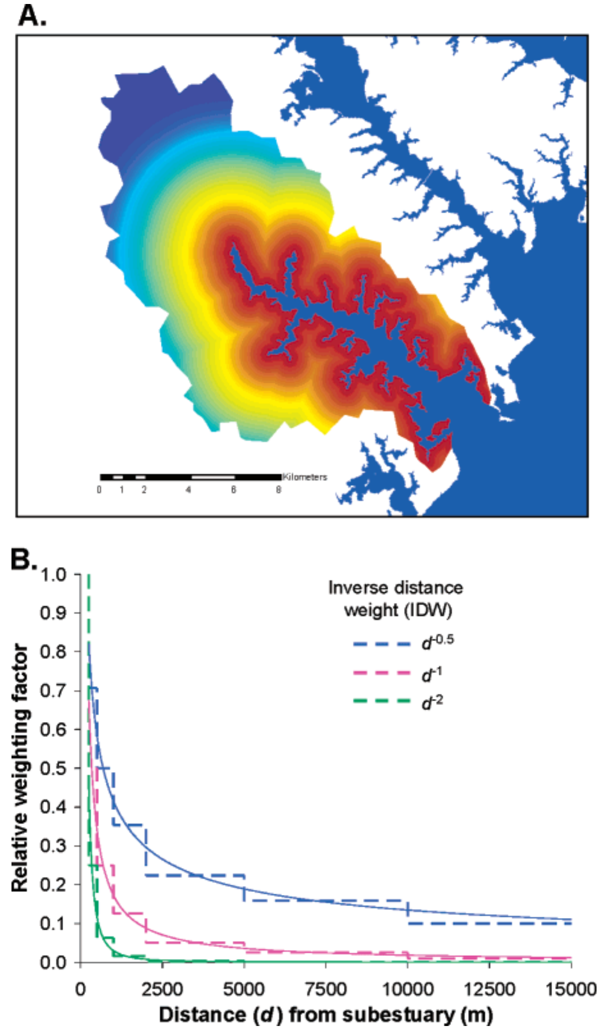

FIGURE 2. (A) Linear distance from the subestuary represented by a gradient of colors spanning red (near) to blue (far). Distances (m) were used to assign raster cells into distance classes used in the calculation of inverse-distance weighted (IDW) % developed land in each watershed (B). The dashed curves show the relative weight across the distance classes, and the smooth curves illustrate the fit of the inverse-distance functions (d^{-x}) to linear distance from the subestuary.

Results

All unweighted developed land-use measures were significant predictors of t-PCBs in white perch, explaining 51−69% of the variance among the 14 subestuaries (Table 2). Percent high-res/comm land was the best predictor of t-PCBs among the unweighted developed-land-use classes.

Distance-weighting markedly improved the linear fit of each land-use predictor and t-PCBs in white perch among the 14 subestuaries (Table 2). Variance explained (r^2)

TABLE 2. Parameters for Regressions of t-PCBs in White Perch (ng/g Wet Weight) Against Unweighted and Distance-Weighted Percent of Watershed in Developed Land Measures for Chesapeake Bay Subestuaries

distance weight	land-use class											
	% impervious			% total developed			% high-res/comm			% commercial		
	slope	intercept	r^2	slope	intercept	r^2	slope	intercept	r^2	slope	intercept	r^2
	All Subestuaries (n=14)											
unweighted	12.1 (2.9)	−10.7 (39.1)	0.60*	6.6 (1.7)	−9.9 (41.9)	0.55**	26.6 (5.1)	−19.9 (34.4)	0.69*	27.5 (7.7)	−8.4 (43.9)	0.51*
$d^{-0.5}$	13.8 (1.7)	−25.6 (23.7)	0.85*	7.7 (1.4)	−28.8 (33.3)	0.72**	22.3 (1.6)	−19.4 (14.1)	0.94**	27.8 (3.2)	−26.3 (22.6)	0.86**
d^{-1}	12.7 (0.6)	−18.7 (10.0)	0.97**	8.1 (0.9)	−38.3 (22.3)	0.87**	15.2 (0.7)	0.8 (8.9)	0.97**	19.8 (0.6)	−8.9 (6.2)	0.99**
d^{-2}	11.5 (0.6)	−7.6 (9.7)	0.97**	8.3 (0.7)	−40.3 (16.7)	0.93**	11.8 (0.9)	13.1 (13.2)	0.94**	14.3 (0.9)	8.7 (11.2)	0.96**
	High PCBs Excluded (n=12)[a]											
unweighted	4.7 (0.8)	7.1 (5.9)	0.79**	2.6 (0.5)	6.0 (7.3)	0.71*	8.1 (2.0)	10.6 (8.0)	0.62*	8.4 (2.2)	11.0 (8.3)	0.59*
$d^{-0.5}$	5.5 (0.9)	5.6 (6.2)	0.79**	2.6 (0.6)	5.8 (8.1)	0.66*	9.6 (2.0)	7.6 (7.4)	0.69*	10.4 (2.2)	7.1 (7.4)	0.70*
d^{-1}	6.6 (1.1)	3.4 (6.5)	0.79**	2.9 (0.7)	3.6 (8.4)	0.67*	12.5 (2.0)	2.1 (6.4)	0.80**	14.2 (2.0)	0.6 (6.0)	0.83**
d^{-2}	7.8 (1.4)	0.9 (7.1)	0.77*	3.3 (0.7)	0.2 (8.2)	0.71*	15.9 (2.0)	−4.8 (5.7)	0.87**	17.8 (2.3)	−5.3 (6.0)	0.86**

[a] Observations from the subestuaries with the two highest levels of PCBs (Upper Patapsco and Back) were excluded from this analysis. One standard error of the parameter is given in parentheses. $*P < 0.005$; $**P < 0.0001$. No intercept was significantly different from zero ($P > 0.05$).

increased by as much as 48% once proximity to the subestuary was taken into account. The two most extreme IDWs (d^{-1} and d^{-2}) resulted in the greatest improvements in model r^2 (Table 2, Figure 3). Percent commercial land, weighted by d^{-1}, was the best predictor of t-PCBs of any of models considered ($r^2 = 99\%$; Table 2, Figure 4). Distance-weighting of % total developed land improved its predictive ability from 55% to 93%; however, intercepts for distance-weighted total developed models became increasingly negative (although not significantly different from zero) as IDWs increased in magnitude (Table 2, Figure 4). In contrast, d^{-1} and d^{-2} weights applied to % impervious, % high-res/comm, and % commercial land moved intercepts closer to zero while also improving the fit to the data compared to unweighted and $d^{-0.5}$ models (Table 2, Figures 3 and 4).

Two subestuaries (Upper Patapsco and Back) had distinctly higher levels of t-PCBs than the other subestuaries and may have had disproportionately strong effects on the regressions; so the effect of removing these two observations from the analysis was evaluated. All land-use classes remained significant predictors of t-PCBs using the reduced (n=12) set of observations (Table 2). In particular, distance-weighted models for % high-res/comm and % commercial land exhibited large improvements in r^2 over unweighted models (Table 2, Figure 5). Percent high-res/comm and % commercial land yielded the highest r^2 values (87% and 86%, respectively; IDW=d^{-2}) among all predictors, intercepts that were closer to zero than unweighted models, and slopes that were very similar to models built using the full data set (Table 2).

Discussion

All of the measures of developed land-use evaluated in this study were strongly related to t-PCBs in white perch fillets. Very few studies have developed empirical relationships between watershed land use and t-PCBs in biota across multiple watersheds (*20*). Moreover, no published study has documented such a strong empirical relationship between watershed land use and levels of t-PCBs in fish in an estuarine ecosystem. Apparently, PCBs historically produced or used in commercial and residential areas are persisting in the environment at the scale of the watersheds and subestuaries examined in this study. This finding also provides compelling evidence that environmental and ecological conditions in subestuaries of Chesapeake Bay, and perhaps other estuarine ecosystems, can be strongly tied to land use in their associated watersheds (*17*).

The strength of the associations between watershed development and t-PCBs in white perch improved markedly when developed land was weighted by its inverse distance to the subestuary. This is consistent with the relationships between distance-weighted development in watersheds and sediment contaminants in subestuaries described by Comeleo et al. (*18*). In our study, the strongest relationships were observed with the two most extreme IDWs (d^{-1} and d^{-2}). These weights most heavily emphasized land use relatively close to the subestuary shorelines. The effect of distance weighting on our results was most evident in the Upper Patapsco and Back watersheds. In the Upper Patapsco, most land adjacent to the subestuary is heavily urbanized, particularly by industry, while upper reaches of its watershed are primarily forested or agricultural with relatively little development. The Back watershed, which has a much larger percentage of its watershed in developed land than the Upper Patapsco (Table 1), has less development, particularly less industry, near the subestuary. Accordingly, white perch in the Back had much lower t-PCBs than white perch from the Upper Patapsco. Distance-weighting effectively captured differences in the arrangement of developed land and resulted in much better predictions of t-PCBs than were achieved with whole-basin measures of watershed development that ignored distance.

Paul et al. (*15*) suggested that much of the variation explained by distance weighting in Comeleo et al.'s (*18*) sediment contaminant study may have been explained by the amount of nonforested wetland in a watershed, which may have acted as a sink for contaminants (note: analysis of PCB data was not reported in refs *15* or *18*). We explored this hypothesis using our data set and found that % nonforested wetland was not a significant term in any multiple regression model using any of the four measures of unweighted, whole-basin developed land. Thus, the combination of % nonforested wetlands with unweighted % developed land could not explain variation in t-PCBs that simple regressions using distance weighted developed land could.

We further scrutinized the association between distance-weighted watershed developed land and t-PCBs by excluding observations from the Upper Patapsco and Back and reanalyzing the data using the remaining 12 subestuaries. We were concerned that these two subestuaries, which had the highest concentrations of t-PCBs, had such a strong influence on the regressions that they would mask potentially weak relationships within less-contaminated subestuary watersheds. However, all developed land-use measures remained highly significant predictors of t-PCBs, and distance-weighting of high-res/comm and commercial land improved predictions of t-PCBs by up to 25% and 27%, respectively. This reanalysis is a strong indication that our overall results are not spurious. Moreover, our results, reinforced by the findings of Comeleo et al. (*18*), indicate that strong linkages

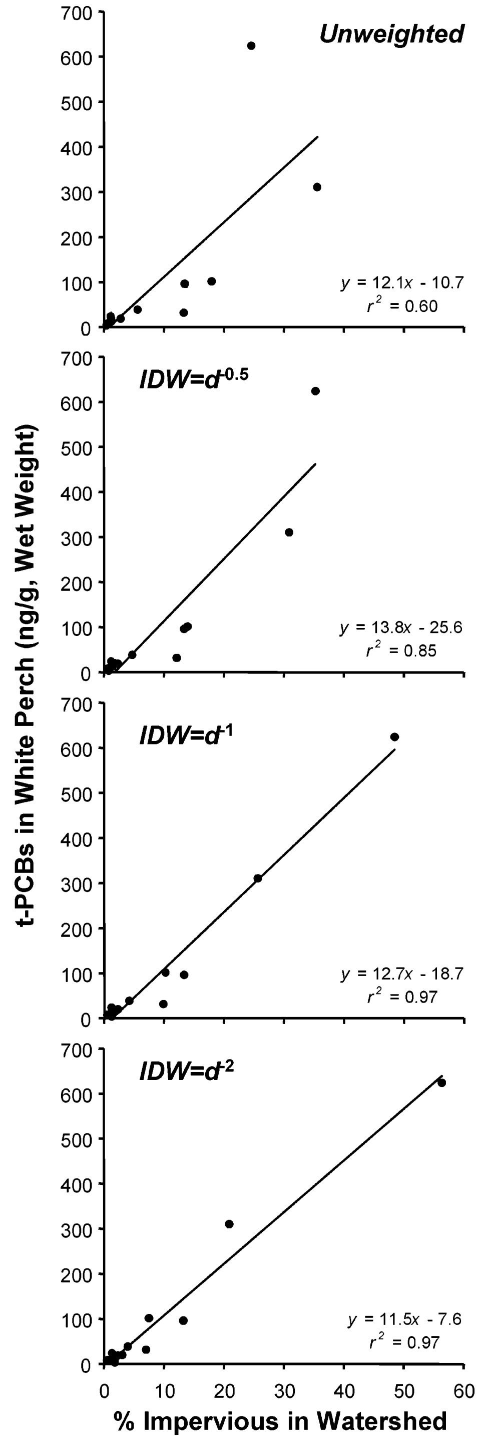

FIGURE 3. Regressions of unweighted and inverse-distance weighted (IDW) % impervious in watersheds on t-PCBs in white perch across the 14 subestuaries.

between subestuaries and their proximal watersheds are not unique to a particular endpoint or location.

We compared four measures of developed land to evaluate whether t-PCBs were more closely associated to particular types of development. Percent commercial land, when weighted by d^{-1}, explained 99% of the variance in t-PCBs across the 14 subestuaries. Distance-weighted (d^{-1}) % impervious and % high-res/comm land each explained 97% of the variance, but these metrics were highly correlated (r=0.97 and 0.92, respectively) to % commercial among the 14 subestuaries. When % total developed land, which included low-intensity residential areas, was used as predictor, the r^2 dropped to 87% for the same IDW. Total developed land remained the weakest predictor when Upper Patapsco and Back were excluded from the analysis. Low-intensity residential areas may be less important as past or present source areas of PCBs, and much of their association with PCBs may be attributed to autocorrelation with PCB sources in more intensive areas of development in the same watershed. However, the high correlation among these variables, coupled with our relatively small sample size, makes it difficult to ascribe greater importance to any one class of developed land.

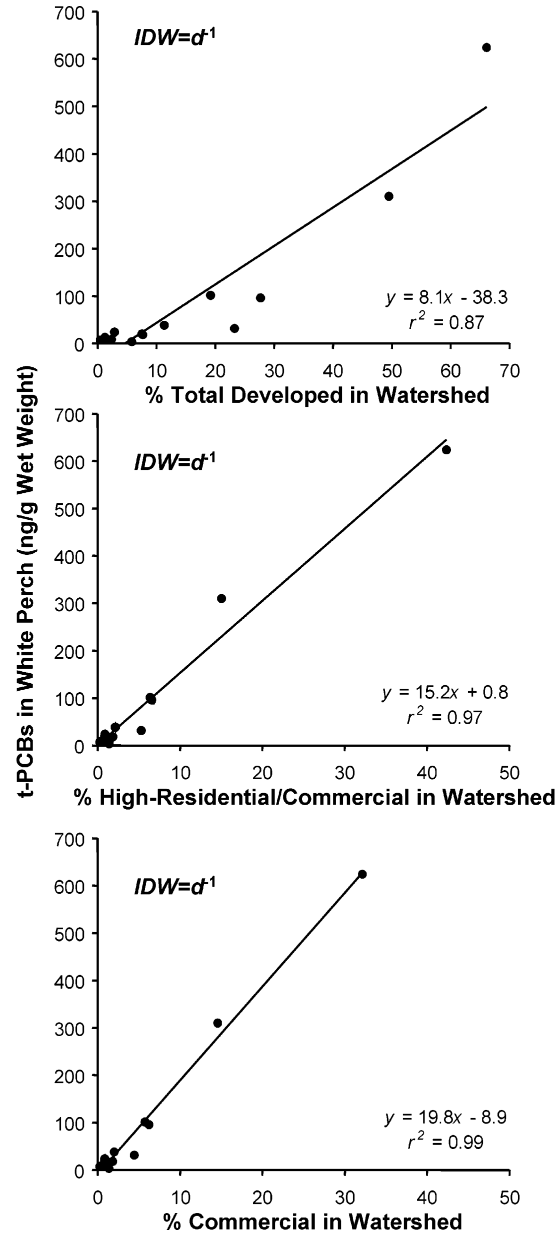

FIGURE 4. Regressions of inverse-distance weighted (IDW=d-1) % total developed, % high-intensity residential/commercial, and % commercial in watersheds on t-PCBs in white perch across the 14 subestuaries.

There is relatively little evidence of active sources of PCBs in the Chesapeake Bay watershed (*29*), although it is not clear that such sources have been well characterized (*30*). One recent (1995−1996) study from the Anacostia River Estuary, a highly urbanized tributary of Chesapeake Bay, showed that detectable amounts (up to 28.9 ng/L) of dissolved- and particulate-phase t-PCBs were present in streams discharging into the tidal Anacostia, particularly

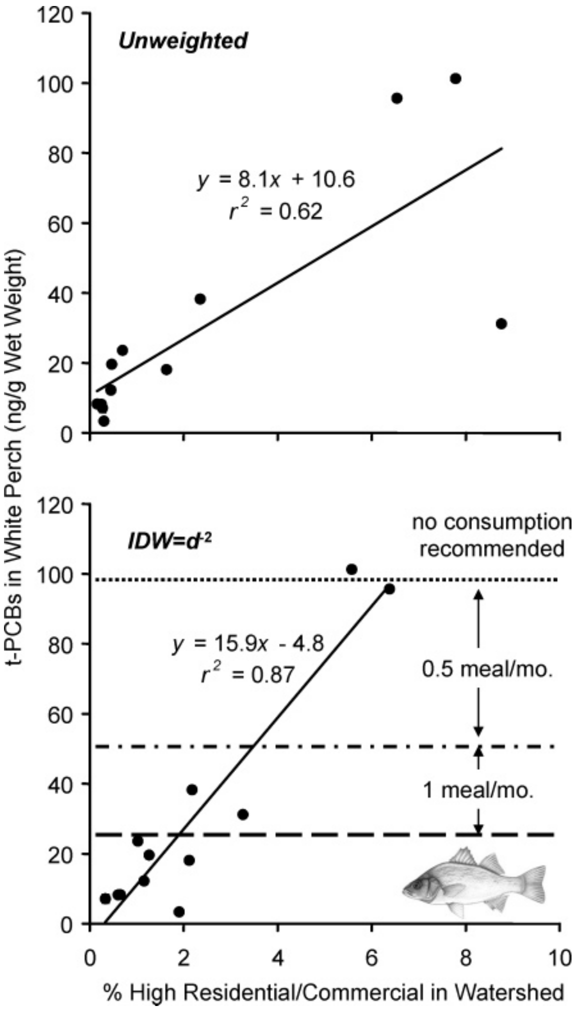

FIGURE 5. Regressions of unweighted and inverse-distance weighted (IDW) % high-intensity residential/commercial in watersheds on t-PCBs in white perch across 12 subestuaries, excluding the two locations with the highest levels of developed land and t-PCBs. Dashed lines illustrate levels of t-PCBs that correspond to consumption advisories for cancer health endpoints (U.S. EPA 1999).

during storm events (*10*). The finding that the highest sediment t-PCBs in Baltimore Harbor were measured near storm drain outlets (*9*) also highlights the importance of stormwater transport. It remains unclear whether PCBs in urban runoff represent legacy deposits of PCB-contaminated sediment already in streams and storm drains or result from new loadings originating from active sources on the land (e.g., landfills, contaminated soils). A recent finding of a previously unrecognized source of PCBs, deteriorating caulking material from old buildings, provides yet another example of possible mechanisms linking urbanized areas to PCBs (*31*). Urban runoff likely represents a current source of delivery of PCBs to downstream estuarine habitats, and our results may provide further indication of active loadings from adjacent portions of upstream watersheds.

The strong relationship between current (ca. 1990–2000) developed land uses and t-PCBs in white perch is somewhat surprising when one considers that PCBs have been banned since the 1970s and a significant amount of urbanization has occurred in this region since then. Current patterns of developed land are almost certainly highly correlated with historical urbanization, and this is particularly true for highly commercial areas such as Baltimore Harbor, which has changed little in the past 50 years. This correlation, coupled with the persistence of PCBs in the environment, provides a simple explanation for why current developed land is such a strong predictor or t-PCBs. Our work highlights the need for further exploration of the historical and current mechanisms of storage and delivery of PCBs from developed watersheds to downstream aquatic habitats as well as examination of current rates of decline, if any (*32*), in existing PCB levels in estuarine fishes in this region. Recent knowledge of these processes will be needed to forecast future PCB levels, particularly in response to land-use change (*33*).

The strength of relationships reported here suggest white perch to be an ideal species for assessing bioaccumulation of estuarine contaminants associated with watershed runoff. However, there are a number of other estuarine species (e.g., American eel (*34*), yellow perch, common carp, channel catfish, largemouth bass, several sunfish species) that exhibit combinations of life history, mobility/fidelity, salinity range, and feeding biology that also span the watershed-estuary interface to varying degrees. Our results suggest that t-PCB levels in these species may also be linked to watershed land use and could represent a risk both to human and fish population health. Further study is warranted to explore such linkages.

A recent increase in the average fish-meal size recommended by the U.S. EPA (*3*) to estimate consumption limits for PCBs has resulted in new consumption advisories for white perch in several Chesapeake Bay subestuaries (*35*). One of the key objectives of this study was to evaluate the utility of watershed analysis for identifying areas of high consumption risk of t-PCBs in fishes. Our results suggest that any of the developed land-use classes evaluated in our study may be useful indicators of t-PCBs in subestuaries of Chesapeake Bay. For example, following U.S. EPA (*3*) guidelines for cancer health endpoints, all subestuaries with >4% distance-weighted high-res/commercial land in their watersheds are highly likely (95% CL) to exceed t-PCB levels that would result in a consumption advisory of no more than 1 meal/mo. (Figure 5). Subestuaries with patterns of development similar to the Upper Patapsco and Back would be very likely to exceed PCB levels recommended for any consumption of fish based on these same guidelines. The models we describe were based on relatively few subestuaries and thus should be interpreted with caution. We recommend that additional white perch be collected and analyzed for t-PCBs from more subestuaries so that the generality of these relationships can be more completely assessed. However, we believe that these current models do provide convincing support for a strikingly strong linkage between watershed land use and PCBs, and this relationship may prove useful for identifying unsampled subestuaries with a high risk of PCB contamination.

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