PERFORMANCE REPORT FOR FEDERAL AID GRANT F-63-R, SEGMENT 8

2017

MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

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Maryland Department of Natural Resources
Fishing and Boating Services
Tawes State Office Building B-2
Annapolis, Maryland 21401
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State of Maryland Department of Natural Resources

Larry Hogan
Governor

Boyd Rutherford
Lt. Governor

Mark J. Belton
Secretary

Department of Natural Resources Mission

For today and tomorrow,
the Department of Natural Resources
inspires people to enjoy and live in
harmony with their environment,
and to protect what makes Maryland
unique – our treasured Chesapeake
Bay, our diverse landscapes and our
living and natural resources.

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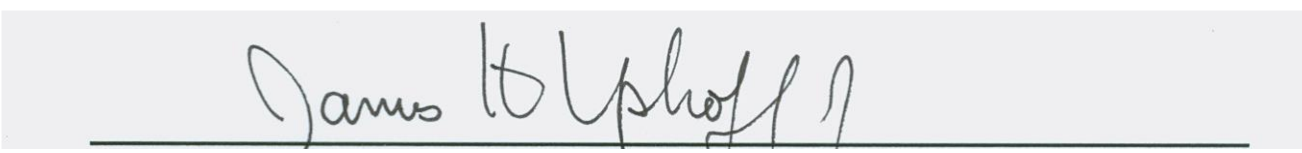

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Mattawoman Creek	Mattawoman Watershed Society
Bush River	Anita Leight Estuary Center
Choptank River	Jim Thompson MD DNR
Mapping	Marek Topolski MD DNR

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Report Organization

This report was completed during December, 2018. It consists of summaries of activities for Jobs 1–4 under this grant cycle. All pages are numbered sequentially; there are no separate page numbering systems for each Job. Job 1 activities are reported in separate numbered sections. For example, Job 1, section 1 would cover development reference points (Job 1) for stream spawning habitat of anadromous fish (Section 1). Tables in Job 1 are numbered as section number – table number (1-1, 1-2, etc). Figures are numbered in the same fashion. Throughout the report, multiple references to past annual report analyses are referred to. The complete PDF versions of many past annual reports can be found under the Publications and Report link on the Fisheries Habitat and Ecosystem (FHEP) website page on the Maryland DNR website. The website address is http://dnr.maryland.gov/fisheries/Pages/FHEP/pubs.aspx . Table 1 provides the page number for each job and section.

Table 1. Job and section number, topic covered, and page number.

SURVEY TITLE: MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS
PROJECT 1: FINFISH HABITAT AND MANAGEMENT

Job 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern: development targets and thresholds

Executive Summary

Spatial Analyses - We used property tax map based counts of structures in a watershed (C), standardized to hectares (C/ha), as our indicator of development. We developed an equation to convert annual estimates of C/ha to estimates of impervious surface (IS) calculated by Towson University from 1999-2000 satellite imagery. Estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were 0.27, 0.83, and 1.59 C/ha, respectively (Uphoff et al. 2012). Percent of watershed in agriculture, forest, and wetlands were estimated from Maryland Department of Planning spatial data.

Correlation analysis suggested negative associations of C/ha with agriculture, forest, and wetlands. Examination of scatter plots for these comparisons suggested a negative hyperbolic curve (power function) would provide a stronger description for the comparison of percent agriculture with C/ha. Remaining land use combinations were not significantly correlated with one another.

Section 1, Stream Ichthyoplankton - Proportion of samples with Herring eggs and-or larvae (P_{herr}; Blueback Herring, Alewife, and Hickory Shad) provided a reasonably precise estimate of habitat occupation based on encounter rate. Regression analyses indicated significant and logical relationships among P_{herr}, C/ha, and conductivity consistent with the hypothesis that urbanization was detrimental to stream spawning. Estimates of P_{herr} were more strongly related to C/ha than conductivity. Estimates of P_{herr} were consistently high in the three watersheds dominated by agriculture. Importance of forest cover could not be assessed with confidence since it was possible that forest cover estimates included residential tree cover. Conductivity was positively related with C/ha in our analysis and with urbanization in other studies. Herring spawning became more variable in streams as watersheds developed. The surveys from watersheds with C/ha of 0.46 (~7% IS) or less had high P_{herr}. General development targets (C/ha or impervious surface) worked reasonably well in characterizing habitat conditions for stream spawning of Herring.

Ranges of P_{herr} in study streams may have indicated variability in suitable habitat rather than abundance of spawners. In developed watersheds, a combination of urban and natural stream processes may create varying amounts of ephemeral spawning habitat annually and dampen spawning migrations through increased conductivity. Observed variation in P_{herr} would indicate wide annual and regional fluctuations in population size. However, stock assessments of Alewife and Blueback Herring indicate they are in decline or are at depressed, stable levels rather than fluctuating.

Section 2, Yellow Perch Larval Presence-Absence Sampling - Annual L_p, the proportion of tows with Yellow Perch larvae during a standard time period and where larvae would be expected, provides a cost-effective measure of the product of egg production and survival through the early postlarval stage. General patterns of land use and L_p emerged from the expanded analyses conducted for this report: L_p was negatively related to development, positively associated with forest and agriculture, and not associated with wetlands.

At least five habitat related factors can be identified that potentially contribute to variations in L_p: salinity, summer hypoxia, maternal influence, winter temperature, and watershed development. These factors may not be independent and there is considerable potential for interactions among them.

Years of high spring discharge favor anadromous fish recruitment in Chesapeake Bay and may represent episodes of hydrologic transport of accumulated organic matter from riparian marshes and forests of watersheds that fuel zooplankton production and feeding success. Amount of organic matter present in L_p samples was negatively influenced by development in Chesapeake Bay subestuaries. Wetlands appeared to be an important source of organic matter for Yellow Perch larvae in subestuaries we studied. Higher DO and pH values in urbanized large subestuaries (Patuxent and Wicomico rivers) during L_p surveys indicate their water quality dynamics were different from the rural, agricultural Choptank River watershed.

Section 3: Estuarine Community Sampling in Summer - Dissolved Oxygen Dynamics - Correlation analyses of DO with temperature and C/ha in subestuaries sampled since 2003 indicated that DO responded to temperature and C/ha differently depending on salinity classification. Mean bottom DO in summer surveys declined with development in mesohaline subestuaries, reaching average levels below 3.0 mg/L when development was beyond its threshold, but it did not decline in oligohaline or tidal-fresh subestuaries. The extent of bottom channel habitat that can be occupied does not appear to diminish with development in tidal-fresh and oligohaline subestuaries due to low DO.

Inspection of the scatter plot of percent of watershed in agriculture versus median bottom DO in mesohaline subestuaries indicated an ascending limb of median DO when agricultural coverage went from 6.0 to 40.9% that was comprised entirely of western shore subestuaries. Median DO measurements beyond this level of agricultural coverage (42.6-71.6% agriculture) were from eastern shore subestuaries and the DO trend appeared to be stable or slightly declining. Agricultural coverage and C/ha were strongly and inversely correlated, so the positive trend of DO with agriculture when agricultural coverage was low was likely to reflect development's negative impact. A dome-shaped quadratic model of median bottom DO and agricultural coverage that did not account for regional differences fit the data well. Modest declines in bottom DO would occur with increases in agriculture in subestuaries with 45%-71% of their watershed covered in agriculture. Predicted median bottom DO at the highest level of agriculture observed would equal 4.2 mg/L, which is between the DO target and threshold.

Section 3: Choptank River Subestuaries - We have explored DO trends in mesohaline Broad Creek, Harris Creek (not sampled in 2017), and Tred Avon River since 2007. These watersheds are similar in agricultural and forest cover, but these adjacent watersheds have undergone development at different levels. Broad and Harris creeks have just passed the target level of development, while Tred Avon River is approaching the development threshold. Tred Avon River provides an opportunity to evaluate modern stormwater management's ability to offset water quality deterioration. During 2017, bottom DO readings below the threshold (DO < 3.0 mg / L) were more frequent in the more developed Tred Avon River than Broad Creek. Seven percent of bottom DO measurements during 2006-2017 in Tred Avon River were below the DO threshold and 32% were below the DO target; in Broad Creek (samples since 2012), 1% were below the threshold and 14% of all DO values were below the target.

Section 3: Middle, Northeast, Severn, and Wicomico (western shore) Rivers – Four additional subestuaries were sampled during 2017: oligohaline Middle River (2009-2017; above threshold development), tidal-fresh Northeast River (2007-2017; above target development), and mesohaline Severn (2003-2005, 2017; above threshold development) and Wicomico (2003, 2012, and 2017; at target development) Rivers. Median Secchi measurements in Middle River ranged from 0.5m to 1.1m during 2009-2017; 2015, the year Zebra Mussels appeared, had the greatest Secchi depth. Zebra Mussels were not observed in 2016-2017 and Secchi depths declined and appeared to return to pre-Zebra Mussel levels. Northeast River median Secchi depth measurements ranged from 0.3m to 0.5m; Severn River ranged from 1.0 m to 1.2 m; and Wicomico River remained steady at 0.5 m. Bottom DO (mg/L) did not appear to fluctuate dramatically from year to year in Middle, Northeast, Severn, and Wicomico Rivers. Median bottom DO estimates were typically at or near the target level in all rivers except Severn River (0.1 mg / L to 2.2 mg / L) over their time-series. Measurements of pH for Middle, Northeast, Severn, and Wicomico Rivers were typically between 7 and 8, but Northeast River pH measurements appeared higher than the others. Since 2015, both Middle and Northeast Rivers exhibited lower total finfish geometric mean (GM) trawl catches. Severn River exhibited a slight increase in total finfish GM trawl catches between 2003-2005 and 2017; Wicomico River's total finfish trawl GM in 2017 was in the middle of the available estimates.

We separated all subestuaries sampled from 1989-2017 by salinity class, then ranked annual all species trawl GMs to find where the Middle, Northeast, Severn, and Wicomico Rivers ranked when compared to other subestuaries in their respective salinity classes. Middle River had one GM within the top ten oligohaline subestuary GMs; five GMs in the middle; and three GMs in the bottom ten. Northeast River had three GMs within the top ten tidal-fresh subestuary GMs; eight in the middle; and one in the bottom ten GMs. Severn River had the last four ranked GMs for mesohaline subestuaries and Wicomico River had all five GMs ranked in the middle.

Overall, the relative conditions at Northeast and Wicomico Rivers have been fairly stable over the available time-series. Middle River conditions have been declining. Severn River conditions remained poor and were the worst among the four subestuaries analyzed.

STATE: MARYLAND

SURVEY TITLE: MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

PROJECT 1: HABITAT AND ECOLOGICAL ASSESSMENT FOR RECREATIONALLY IMPORTANT FINFISH

JOB 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern

COMMON BACKGROUND for Job 1, Sections 1-3.

"*It is the whole drainage basin, not just the body of water, that must be considered as the minimum ecosystem unit when it comes to man's interests."* (Odum 1971).

Fishing has been the focus of assessments of human-induced perturbations of fish populations (Boreman 2000) and biological reference points (BRPs) have been developed to guide how many fish can be safely harvested from a stock (Sissenwine and Shepherd 1987). Managers also take action to avoid negative impacts from habitat loss and pollution that might drive a fish population to extinction (Boreman 2000) and typically control fishing to compensate for these other factors. A habitat-based corollary to the BRP approach would be to determine to what extent habitat can be degraded before adverse conditions cause habitat suitability to decline significantly or cease.

Forests and wetlands in the Chesapeake Bay watershed have been converted to agriculture and residential areas to accommodate increased human populations since colonial times (Brush 2009). These watershed alterations have affected major ecological processes and have been most visibly manifested in Chesapeake Bay eutrophication, hypoxia, and anoxia (Hagy et al. 2004; Kemp et al. 2005; Fisher et al. 2006; Brush 2009). Human population growth since the 1950s added a suburban landscape layer to the Chesapeake Bay watershed (Brush 2009) that has been identified as a threat (Chesapeake Bay Program or CBP 1999). Land in agriculture has been relatively stable, but fertilizer and pesticide use became much more intensive (use had increased) in order to support population growth (Fisher et al. 2006; Brush 2009). Management of farming practices has become more intense in recent decades in response to eutrophication (Kemp et al. 2005; Fisher et al. 2006; Brush 2009). Through previous research under F-63, we have identified many negative consequences of watershed development on Bay habitat of sportfish and have used this information to influence planning and zoning (Interagency Mattawoman Ecosystem Management Task Force 2012) and fisheries management (Uphoff et al. 2011). We have less understanding of the consequences of agriculture on sportfish habitat and have redirected some effort towards understanding impacts of agricultural land use on sportfish habitat.

Job 1 investigates two general alternative hypotheses relating recreationally important species to development and/or agriculture. The first hypothesis is that there is a level of a particular land-use that does not significantly alter habitat suitability and the second is that there is a threshold level of land-use that significantly reduces habitat suitability (production from this habitat diminishes). The null hypothesis would be an absence of differences. In general, we expect habitat deterioration to manifest itself as reduced survival of sensitive live stages (usually eggs or larvae) or limitations on use of habitat for spawning or growth (eggs-adults). In either case, we would expect that stress from habitat would be reflected by dynamics of critical life stages (abundance, survival, growth, condition, etc.).

Development associated with increased population growth converts land use typical of rural areas (farms, wetlands, and forests) to residential and industrial uses (Wheeler et al. 2005; National Research Council or NRC 2009; Brush 2009) that have ecological, economic, and societal consequences (Szaro et al. 1999). Ecological stress from development of the Bay watershed conflicts with demand for fish production and recreational fishing opportunities from its estuary (Uphoff et al. 2011; Uphoff et al 2015). Extended exposure to biological and environmental stressors affect fish condition and survival (Rice 2002; Barton et al. 2002; Benejam et al. 2008; Benejam et al. 2010; Branco et al. 2016).

Impervious surface is used as an indicator of development because of compelling scientific evidence of its effect in freshwater systems (Wheeler et al. 2005; NRC 2009) and because it is a critical input variable in many water quality and quantity models (Arnold and Gibbons 1996; Cappiella and Brown 2001). Impervious surface itself increases runoff volume and intensity in streams, leading to increased physical instability, erosion, sedimentation, thermal pollution, contaminant loads, and nutrients (Beach 2002; Wheeler et al. 2005; NRC 2009). Urbanization may introduce additional industrial wastes, contaminants, stormwater runoff and road salt (Brown 2000; NRC 2009; Benejam et al. 2010; McBryan et al. 2013; Branco et al. 2016) that act as ecological stressors and are indexed by impervious surface. The NRC (2009) estimated that urban stormwater is the primary source of impairment in 13% of assessed rivers, 18% of lakes, and 32% of estuaries in the U.S., while urban land cover only accounts for 3% of the U.S. land mass.

Impact of development on estuarine systems has not been well documented, but measurable adverse changes in physical and chemical characteristics and living resources have occurred at IS of 10-30% (Mallin et al. 2000; Holland et al. 2004; Uphoff et al. 2011). Habitat reference points based on IS have been developed (ISRPs) for Chesapeake Bay estuarine watersheds (Uphoff et al. 2011). They provide a quantitative basis for managing fisheries in increasingly urbanizing Chesapeake Bay watersheds and enhance communication of limits of fisheries resources to withstand development-related habitat changes to fishers, land-use planners, watershed-based advocacy groups, developers, and elected officials (Uphoff et al. 2011; Interagency Mattawoman Ecosystem Management Task Force 2012). These guidelines have held for Herring stream spawning, Yellow Perch larval habitat (they are incorporated into the current draft of Maryland's tidal Yellow Perch management plan), and summer habitat in tidal-fresh subestuaries (Uphoff et al. 2015). Preserving watersheds at or below 5% IS would be a viable fisheries management strategy. Increasingly stringent fishery regulation might compensate for habitat stress as IS increases from 5 to 10%. Above a 10% IS threshold, habitat stress mounts and successful management by harvest adjustments alone becomes unlikely (Uphoff et al. 2011; Interagency Mattawoman Ecosystem Management Task Force 2012; Uphoff et al. 2015). We have estimated that impervious surface in Maryland's portion of the Chesapeake Bay watershed will exceed 10% by 2020. We expect adverse habitat conditions for important forage and gamefish to worsen with future growth. Managing this growth with an eye towards conserving fish habitat is important to the future of sportfishing in Maryland.

We now consider tax map derived development indices as the best source for standardized, readily updated, and accessible watershed development indicators in Maryland and have development targets and thresholds based on it that are the same as ISRPs (Uphoff et al.

2015; Topolski 2015). Counts of structures per hectare (C/ha) had strong relationships with IS in years when all were estimated (1999-2000; Uphoff et al. 2015). Tax map data can be used as the basis for estimating target and threshold levels of development in Maryland and these estimates can be converted to IS. Estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.27, 0.83, and 1.59 C/ha, respectively. Tax map data provide a development time-series that goes back to 1950, making retrospective analyses possible (Uphoff et al. 2015).

The area of major spawning tributaries used by Striped Bass, White Perch, Yellow Perch, Alewife, Blueback Herring, Hickory Shad, and American Shad are typically on the receiving end of large amounts of agricultural drainage because of their location at the junction of large fluvial systems and brackish estuaries. Trends in juvenile indices of these species are similar, indicating similar influences on year-class success (Uphoff 2008).

Agricultural pesticides and fertilizers were thought to be potential sources of toxic metals implicated in some episodic mortality of Striped Bass larvae in Bay spawning tributaries in the early 1980s (Uphoff 1989; 1992; Richards and Rago 1999; Uphoff 2008). A correlation analysis of Choptank River watershed agricultural best management practices (BMPs) and estimates of postlarval survival during 1980-1990 indicated that as many as four BMPs were positively associated with survival (Uphoff 2008). Two measures that accounted for the greatest acreage, conservation tillage and cover crops, were strongly associated with increased postlarval survival (r = 0.88 and r = 0.80, respectively). These correlations cannot explain whether toxicity was lowered by BMPs, but it is possible that reduced contaminant runoff was a positive byproduct of agricultural BMPs aimed at reducing nutrients (Uphoff 2008).

Agriculturally derived nutrients have been identified as the primary driver of hypoxia and anoxia in the mainstem Chesapeake Bay (or Bay; Hagy et al. 2004; Kemp et al. 2005; Fisher et al. 2006; Brush 2009). Hypoxia is also associated with transition from rural to suburban landscapes in brackish Chesapeake Bay subestuaries (Uphoff et al. 2011). Hypoxia's greatest impact on gamefish habitat occurs during summer when its extent is greatest, but hypoxic conditions are present at lesser levels during spring and fall (Hagy et al. 2004; Costantini et al. 2008). Episodic hypoxia may elevate catch rates in various types of fishing gears by concentrating fish at the edges of oxygenated waters, masking associations of landings and hypoxia (Kraus et al. 2015).

Habitat loss due to hypoxia in coastal waters is often associated with fish avoiding DO that reduces growth and requires greater energy expenditures, as well as lethal conditions (Breitburg 2002; Eby and Crowder 2002; Bell and Eggleston 2005). There is evidence of cascading effects of low DO on demersal fish production in marine coastal systems through loss of invertebrate populations on the seafloor (Breitburg et al. 2002; Baird et al. 2004). A long-term decline in an important Chesapeake Bay pelagic forage fish, Bay Anchovy, may be linked to declining abundance of the common calanoid copepod *Acartia tonsa* in Maryland's portion of Chesapeake Bay that, in turn, may be linked to rising long-term water temperatures and eutrophication that drive hypoxia (Kimmel et al. 2012). Crowding in nearshore habitat, if accompanied by decreased growth due to competition, could lead to later losses through size-based processes such as predation and starvation (Breitburg 2002; Eby and Crowder 2002; Bell and Eggleston 2005). Exposure to low DO appears to impede immune suppression in fish and Blue Crabs, leading to outbreaks of lesions, infections, and disease (Haeseker et al. 1996; Engel and Thayer 1998; Breitburg 2002; Evans et al. 2003). Exposure of adult Carp to hypoxia depressed reproductive processes such as gametogenesis, gonad maturation, gonad size, gamete quality, egg fertilization and hatching, and larval survival through endocrine disruption even though they were allowed to spawn under normoxic conditions (Wu et al. 2003). Endocrine disruption due to hypoxia that could reduce population spawning potential has been detected in laboratory and field studies of Atlantic Croaker in the Gulf of Mexico (Thomas and Rahman 2011) and Chesapeake Bay (Tuckey and Fabrizio 2016).

Impacts of hypoxia may not be entirely negative. Costantini et al. (2008) examined the impact of hypoxia on Striped Bass 2 years-old or older in Chesapeake Bay during 1996 and 2000 through bioenergetics modeling and concluded that a temperature-oxygen squeeze had not limited growth potential of Striped Bass in the past. In years when summer water temperatures exceed 28°C, hypoxia could reduce the quality and quantity of habitat through a temperature-oxygen squeeze. In cooler summers, hypoxia may benefit Striped Bass by concentrating prey and increasing encounter rates with prey in oxygenated waters (Costantini et al. 2008).

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General Spatial and Analytical Methods used in Job 1, Sections 1-3

Spatial Methods - We used property tax map based counts of structures in a watershed, standardized to hectares (C/ha), as our indicator of development (Uphoff et al. 2012; Topolski 2015). This indicator has been provided to us by M. Topolski (MD DNR). Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MD DOP 2013). All tax data were organized by county. Since watersheds straddle political boundaries, one statewide tax map was created for each year of available tax data, and then subdivided into watersheds. Maryland's tax maps are updated and maintained electronically as part of MD DOP's Geographic Information System (GIS) database. Files were managed and geoprocessed in ArcGIS 9.3.1 from Environmental Systems Research Institute (ESRI 2009). All feature datasets, feature classes, and shapefiles were spatially referenced using the NAD_1983_StatePlane_Maryland_FIPS_1900 projection to ensure accurate feature overlays and data extraction. ArcGIS geoprocessing models were developed using Model Builder to automate assembly of statewide tax maps, query tax map data, and assemble summary data. MdProperty View tax data are annually updated by each Maryland jurisdiction to monitor the type of parcel development for tax assessment purposes. Tax data through 2014 were available for this report. To create watershed land tax maps, each year's statewide tax map was clipped using the MD 8-digit watershed boundary file; estuarine waters were excluded. These watershed tax maps were queried for all parcels having a structure built from 1700 to the tax data year. A large portion of parcels did not have any record of year built for structures, but consistent undercounts should not have presented a problem since we were interested in the trend and not absolute magnitude (Uphoff et al. 2012). Mattawoman Creek C/ha declined between 2011 and 2012 and then returned to a higher level in 2013. We replaced the 2012 estimate of C/ha for Mattawoman Creek with the average of 2011 and 2013.

Uphoff et al. (2012) developed an equation to convert annual estimates of C/ha to estimates of impervious surface (IS) calculated by Towson University from 1999-2000 satellite imagery. Estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.27, 0.83, and 1.59 C/ha, respectively (Uphoff et al. 2012).

Percent of watershed in agriculture, forest, and wetlands were estimated from Maryland Department of Planning spatial data. The MD DOP forest cover estimates have a minimum mapping unit of 10 acres that mixes forest cover in residential areas (trees over lawns) with true forest cover, clouding interpretation of forest influence (R. Feldt, MD DNR Forest Service, personal communication). An urban category was available as well, but was not featured in many subsequent analyses since we have adopted C/ha as our preferred index of development. Land use and land cover (LULC) shapefiles for the years 2002 and 2010 were downloaded from http://planning.maryland.gov/OurProducts/downloadFiles.shtml. Maryland Department of Planning LULC shapefiles for the year 1994 were obtained from the Maryland Department of Natural Resources. The shapefiles are vector polygons projected in NAD_1983_StatePlane_Maryland_FIPS_1900. General categories of LULC queried were urban land uses, agriculture, forest, and wetlands. Metadata for the LULC categories is available for download from the Maryland Department of Planning. Shapefiles are provided for each Maryland jurisdiction and as an aggregated statewide file.

The statewide LULC shapefiles were clipped using boundary shapefiles for each watershed of interest. Once clipped, polygon geometry was recalculated. Polygons designated as water were omitted when calculating watershed area; that is only land was considered when calculating the ratio of LULC for each category. For each LULC category, polygons were queried and the total land area in hectares was calculated. The ratio of LULC was its total hectares divided by the total watershed hectares to the nearest tenth of a hectare.

Statistical Analyses – A combination of correlation analysis, plotting of data, and curve-fitting was used to explore trends among land use types (land that was developed or in agriculture, forest, or wetland) and among fish habitat responses. Fish habitat responses were the proportion of stream samples with Herring eggs and/or larvae (P_{herr}; Section 1), proportion of subestuary samples with Yellow Perch larvae (L_p; Section 2), or subestuary bottom dissolved oxygen in summer (Section 3).

Correlations among watershed estimates of C/ha and percent of watershed estimated in urban, agriculture, forest, and wetland based on Maryland's Department of Planning spatial data (Maryland Department of Planning 2013) were used to describe associations among land cover types. Urban land consisted of high and low density residential, commercial, and institutional acreages (Maryland Department of Natural Resources or MD DNR 1999) and was not a direct measure of IS. These analyses explored (1) whether C/ha estimates were correlated with another indicator of development, percent urban and (2) general associations among major landscape features in our study watersheds. Scatter plots were inspected to examine whether nonlinear associations were possible. Land use was assigned from Maryland Department of Planning estimates for 1973, 1994, 1997, 2002, or 2010 that fell closest to a sampling year. We were particularly interested in knowing whether these land uses might be closely correlated enough (r greater than 0.8; Ricker 1975) that only one should be considered in analyses of land use and L_p and P_{herr}. We further examined relationships using descriptive models as a standard of comparison (Pielou 1981). Once the initial associations and scatter plots were examined, linear or nonlinear regression analyses (power, logistic, or Weibull functions) were used to determine the general shape of trends among land use types. This same strategy was pursued for analyses of land use and L_p or P_{herr}. Level of significance was set at $P < 0.05$. Residuals were inspected for trends, non-normality, and need for additional terms. A general description of equations used follows, while more specific applications will be described in later sections.

Linear regressions described continuous change in variable Y as X changed:

$$Y = (m \cdot X) + b;$$

where m is the slope and b is the Y-intercept (Freund and Littel 2006). Multiple regression models accommodated an additional variable (Z):

$$Y = (m \cdot X) + (n \cdot Z) + b;$$

where n is the slope for variable Z and other parameters are as described previously (Freund and Littel 2006). We did not consider multiple regression models with more than two variables. Potential dome-shaped relationships were examined with quadratic models (Freund and Littell 2006):

$$Y = (m \cdot X) + (n \cdot X^2) + b.$$

The linear regression function in Excel or Proc REG in SAS (Freund and Littel 2006) was used for single variable linear regressions. Multiple linear and quadratic regressions were analyzed with Proc REG in SAS (Freund and Littell 2006).

Examination of scatter plots suggested that some relationships could be nonlinear, with the Y-axis variable increasing at a decreasing rate with the X-axis variable and we fit power, logistic growth, or Weibull functions to these data using Proc NLIN in SAS (Gauss-Newton algorithm). The power function described a relationship with a perceptible, but declining increase in Y with X by the equation:

$$Y = a \bullet (X)^b;$$

where a is a scaling coefficient and b is a shape parameter. The symmetric logistic growth function described growth to an asymptote through the equation:

$$Y = b / ((1 + ((b - c) / c) \bullet (\exp(-a \bullet X)));$$

where a is the growth rate of Y with X, b is maximum Y, and c is Y at X = 0 (Prager et al. 1989). The Weibull function is a sigmoid curve that provides a depiction of asymmetric ecological relationships (Pielou 1981). A Weibull curve described the increase in Y as an asymmetric, ascending, asymptotic function of X:

$$Y = K\{1 - \exp[-(Y / S)^b]\};$$

where K was the asymptotic value of Y as X approached infinity; S was a scale factor equal to the value of Y where Y = 0.63 • K; and b was a shape factor (Pielou 1981; Prager et al. 1989).

Confidence intervals (typically 95% CIs) of the model parameters for each indicator species were estimated to examine whether parameters were different from 0 (Freund and Littel 2006). If parameter estimates were not different from 0, the model was rejected.

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Section 1: Stream Ichthyoplankton Sampling

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Introduction

Urbanization associated with increased population growth became a factor in the decline of diadromous fishes in the late 20th century (Limburg and Waldman 2009). Increased impervious surfaces have altered hydrology and increased diadromous fish habitat loss (Limburg and Waldman 2009). Anadromous fish egg densities (Alewife and White Perch) in the Hudson River exhibited a strong negative threshold response to urbanization (Limburg and Schmidt 1990). We were interested in understanding how reference points for development (impervious surface reference points or ISRPs, or C/ha reference points) developed for Chesapeake Bay subestuaries (Uphoff et al. 2011) were related to anadromous fish spawning in streams in Maryland's portion of Chesapeake Bay.

Surveys to identify spawning habitat of White Perch, Yellow Perch and "Herring" (Blueback Herring, Alewife, American Shad, and Hickory Shad) were conducted in Maryland during 1970-1986. These data were used to develop statewide maps depicting anadromous fish spawning habitat (O'Dell et al. 1970; 1975; 1980; Mowrer and McGinty 2002). Many of these watersheds have undergone considerable development and recreating these surveys provided an opportunity to explore whether spawning habitat declined in response to urbanization. Surveys based on the sites and methods of O'Dell et al. (1975; 1980) were used to sample Mattawoman Creek (2008-2017), Piscataway Creek (2008-2009 and 2012-2014), Bush River (2005-2008 and 2014), Deer Creek (2012-2015), Tuckahoe Creek (2016-2017), Choptank River (2016-2017), and Patapsco River (2013-2017; Figure 1-1).

Mattawoman and Piscataway Creeks are adjacent Coastal Plain watersheds along an urban gradient emanating from Washington, DC (Table 1-1; Figure 1-1). Piscataway Creek's watershed is both smaller than Mattawoman Creek's and closer to Washington, DC. Bush River is located in the urban gradient originating from Baltimore, Maryland, and is located in both the Coastal Plain and Piedmont physiographic provinces. Deer Creek is within a conservation district, and is located entirely in the Piedmont north of Baltimore, near the Pennsylvania border (Clearwater et al. 2000). Bush River and Deer Creek drainages are adjacent to each other. The Choptank River has an agricultural watershed that is entirely within the eastern shore's Coastal Plain. Ichthyoplankton surveys were conducted in the upper reaches of the Choptank River and Tuckahoe Creek, a tributary of the Choptank River. Both systems are predominantly agricultural and the Choptank River is a major tributary of the Chesapeake Bay. The Patapsco River watershed is located within both physiographic provinces, with rolling hills over much of its area that are characteristic of the eastern division of the Piedmont province, while to the southeast the watershed lies in the Coastal Plain bordering the western side of the Chesapeake Bay (O'Dell et al. 1975; Table 1-1; Figure 1-1).

We developed two indicators of anadromous fish spawning in a watershed based on presence/absence of eggs and larvae: occurrence at a site (a spatial indicator) and proportion of samples with eggs and larvae (a spatial and temporal indicator). Occurrence of eggs or larvae of an anadromous fish group (White Perch, Yellow Perch, or Herring) at a site recreated the indicator developed by O'Dell et al. (1975; 1980). This spatial indicator was compared to the extent of development in the watershed (counts of structures per hectare or C/ha) between the

1970s and the present (Topolski 2015). An indicator of habitat occupation in space and time from collections that started in the 2000s was estimated as proportion of samples with eggs and-or larvae of anadromous fish groups. Proportion of samples with an anadromous fish group was compared to level of development (C/ha) and conductivity, an indicator of water quality strongly associated with development (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012).

Methods

Stream sites sampled for anadromous fish eggs and larvae during 2005-2017 were typically at road crossings that O'Dell et al. (1975; 1980) determined were anadromous fish spawning sites during the 1970s. O'Dell et al. (1975; 1980) summarized spawning activity as the presence of any species group (White Perch, Yellow Perch, or Herring) egg, larva, or adult at a site. O'Dell et al. (1975; 1980) sampled eggs and larvae with stream drift ichthyoplankton nets and adults were sampled by wire traps.

All collections during 2005-2017, with the exception of Deer Creek during 2012-2015, Choptank River and Tuckahoe Creek during 2016-2017, and Patapsco River during 2013-2017, were made by citizen volunteers who were trained and monitored by program biologists. During March to May, 2008-2015, ichthyoplankton samples were collected in Mattawoman Creek from three tributary sites (MUT3-MUT5) and four mainstem sites (MC1-MC4; Figure 1-2; Table 1-2). Tributary sites MUT4 and MUTX were selected based on volunteer interest and added in 2010 and 2014, respectively; MUTX was discontinued in 2015 due to restricted access and limited indication of spawning. All mainstem sites were sampled in 2016-2017, while the only tributary site sampled was MUT3; beaver dams blocked spawning access to MUT4 and MUT5. Piscataway Creek stations were sampled during 2008-2009 and 2012-2014 (Figure 1-3; Uphoff et al. 2010). Bush River stations were sampled during 2005-2008 and 2014 (Figure 1-4; McGinty et al. 2009; Uphoff et al. 2015). Deer Creek sites SU01-SU04 were sampled in 2012 and sampling continued in 2013-2015 with the addition of site SU05 (Figure 1-5). Choptank River (CH100-CH111; Figure 1-6) and Tuckahoe Creek (TUC101-TUC110; Figure 1-7) sites were sampled in 2016-2017. Patapsco River samples (four sites; Figure 1-8) were collected by US Fish and Wildlife Service from 2013-2017 and were added to this data set. Table 1-2 summarizes sites, dates, and sample sizes in Mattawoman, Piscataway, Deer, and Tuckahoe Creeks, and Bush, Choptank, and Patapsco Rivers during 2005-2017.

Ichthyoplankton samples were collected in all systems and years using stream drift nets constructed of 360-micron mesh with a rectangular 300 • 460 mm opening. The stream drift net configuration and techniques were the same as those used by O'Dell et al. (1975). The net frame was connected to a handle so that the net could be held stationary in the stream. A threaded collar on the end of the net connected a mason jar to the net. Nets were placed in the stream for five minutes with the opening facing upstream. Collections in Choptank River and Tuckahoe Creek during 2016-2017 were made using stream drift nets at wadeable sites or using a conical plankton net towed from a boat (see Section 2 for a description of ichthyoplankton sampling by boat) at sites too deep to wade. This mimics collections made by O'Dell et al. (1980) within the Choptank River drainage, specifically Tuckahoe Creek. For both types of collection, nets were retrieved and rinsed in the stream by repeatedly dipping the lower part of the net and splashing water through the outside of the net to avoid sample contamination. The jar was removed from the net and an identification label describing site, date, time, and collectors was placed both in the jar and on top of the lid before it was sealed. Samples were fixed immediately after collection by DNR staff, or were placed in a cooler with ice for transport and preserved with

10% buffered formalin after a volunteer team was finished sampling for the day. Water temperature (°C), conductivity (µS/cm), and dissolved oxygen (DO, mg/L) were recorded at each site using either a hand-held YSI Model 85 meter or a YSI Pro2030 meter. Meters were calibrated for DO each day prior to use. All data were recorded on standard field data forms and double-verified at the site during volunteer collections. Approximately 2-ml of rose bengal dye was added to each sample in order to stain the organisms pink to aid sorting.

Ichthyoplankton samples were sorted in the laboratory by project personnel. All samples were rinsed with water to remove formalin and placed into a white sorting pan. Samples were sorted systematically (from one end of the pan to another) under a 10x bench magnifier. All eggs and-or larvae were removed and were retained in a small vial with a label (site, date, and time) and stored with 20% ethanol for later identification under a microscope. Each sample was sorted systematically a second time for quality assurance (QA). Any additional eggs and-or larvae found were removed and placed in a vial with a label (site, date, time, and QA) and stored with 20% ethanol for identification under a microscope. All eggs and larvae found during sorting (both in original and QA vials) were identified as either Herring (Blueback Herring, Alewife, and Hickory Shad), Yellow Perch, White Perch, unknown (eggs and-or larvae that were too damaged to identify) or other (indicating another fish species) and the presence or absence of each of the above species was recorded. The three Herring species' eggs and larvae are very similar (Lippson and Moran 1974) and identification to species can be problematic. American Shad eggs and larvae would be larger at the same stages of development than those identified as Herring (Lippson and Moran 1974) and none have been detected in our surveys.

Methods used to estimate development (C/ha) and land use indicators (percent of watershed in agriculture, forest, wetlands, and urban land use) are explained in **General Spatial and Analytical Methods used in Job 1, Sections 1-3.** Development targets and limits and general statistical methods (analytical strategy and equations) are described in this section as well. Specific spatial and analytical methods for this section of the report are described below.

Mattawoman Creek's watershed was 24,441 ha and estimated C/ha increased from 0.87 to 0.93 during 2008-2017; Piscataway Creek's watershed was 17,642 ha and estimated C/ha increased from 1.41 to 1.50 during 2008-2014; Bush River's watershed was 36,038 ha and estimated C/ha increased from 1.37 to 1.51 during 2005-2014; and Deer Creek, a spawning stream with low development, had a watershed of 37,697 ha and estimated C/ha was 0.24 during 2012-2015 (Table 1-1). The upper portion of the Choptank River (watershed area = 38,216 ha and developmental level = 0.18 C/ha) and a tributary of the Choptank River, Tuckahoe Creek (watershed area = 39,388 ha and developmental level = 0.07), were added in 2016-2017 as spawning streams with high agricultural influence and low watershed development (Table 1-1; Figure 1-1). Deer Creek, and Choptank River and Tuckahoe Creek, collections were made by DNR biologists from the Fishery Management Planning and Fish Passage Program at no charge to this grant. Patapsco River's watershed equaled 93,895 ha and estimated C/ha was 1.11-1.12 during 2013-2017. Collections in the Patapsco River were made at no charge to this grant.

Conductivity measurements collected for each date and stream site (mainstem and tributaries) during 2008-2017 from Mattawoman Creek were plotted and mainstem measurements were summarized for each year. Mainstem sites would be influenced by development in Waldorf, while the monitored tributaries would not. Unnamed tributaries were excluded from calculation of summary statistics to capture conditions in the largest portion of habitat. Comparisons were made with conductivity minimum and maximum reported for Mattawoman Creek during 1991 by Hall et al. (1992). Conductivity data were similarly summarized for Piscataway Creek mainstem stations during 2008-2009 and 2012-2014. A subset of Bush River stations that were sampled each year during 2005-2008 and 2014 (i.e., stations in common) were summarized; stations within largely undeveloped Aberdeen Proving Grounds were excluded because they were not sampled every year. Conductivity was measured with each sample in Deer Creek in 2012-2015, in the Choptank River and Tuckahoe Creek in 2016-2017, and in the Patapsco River in 2013-2017.

A water quality database maintained by DNR's Tidewater Ecosystem Assessment (TEA) Division provided conductivity measurements for Mattawoman Creek during 1970-1989. These historical measurements were compared with those collected in 2008-2017 to examine changes in conductivity over time. Monitoring was irregular for many of the historical stations. Table 1-3 summarizes site location, month sampled, total measurements at a site, and what years were sampled. Historical stations and those sampled in 2008-2016 were assigned river kilometers (RKM) using a GIS ruler tool that measured a transect approximating the center of the creek from the mouth of the subestuary to each station location. Stations were categorized as tidal or non-tidal. Conductivity measurements from eight non-tidal sites sampled during 1970-1989 were summarized as monthly medians. These sites bounded Mattawoman Creek from its junction with the estuary to the city of Waldorf (Route 301 crossing), the major urban influence on the watershed. Historical monthly median conductivities at each mainstem Mattawoman Creek non-tidal site were plotted with 2008-2017 spawning season median conductivities.

Presence of White Perch, Yellow Perch, and Herring eggs and-or larvae at each station in 2017 was compared to past surveys to determine which sites still supported spawning. We used the criterion of detection of eggs and-or larvae at a site (O'Dell et al. 1975; 1980) as evidence of spawning. Raw data from early 1970s collections were not available to formulate other metrics.

Sites where Herring spawning was detected (site occupation) during the current study and historical studies were compared to changes in C/ha. Historical site occupation was available for Mattawoman Creek mainstem stations sampled in 1971 by O'Dell et al. (1975) and Hall et al. (1992) during 1989-1991. Hall et al. (1992) collected ichthyoplankton with 0.5 m diameter plankton nets (3:1 length to opening ratio and 363μ mesh set for 2-5 minutes, depending on flow) suspended in the stream channel between two posts instead of stream drift nets. Historical site occupation was available for Piscataway Creek in 1971 (O'Dell et al. 1975), Bush River in 1973 (O'Dell et al. 1975), Deer Creek in 1972 (O'Dell et al. 1975), and Tuckahoe Creek, 1976-77 (O'Dell et al. 1980). The sites sampled by O'Dell et al. (1975) in the Patapsco River, were not the same as those sampled during 2013-2017, but were within a similar area.

The proportion of samples where Herring eggs and/or larvae were present (P_{herr}) was estimated for Mattawoman Creek mainstem stations (MC1-MC4; Figure 1-2) during 1991 and 2008-2017, Piscataway Creek (2008-2009 and 2012-2014), Bush River (2005-2008 and 2014), Deer Creek (2012-2015), Choptank River (2016-2017), Tuckahoe Creek (2016-2017), and Patapsco River (2013-2017). Counts of Herring eggs and larvae were available for 1991 (C/ha = 0.46) in a tabular summary in Hall et al. (1992) at the sample level and these data were converted to presence-absence. Herring was the only species group with adequate sample sizes for annual P_{herr} estimates with reasonable precision. Mainstem stations (PC1-PC3) and Tinkers Creek (PTC1) were used to estimate P_{herr} in Piscataway Creek (Figure 1-3). Only sites in streams that were sampled in all years (sites in common) in the Bush River drainage were analyzed (Figure 1-4; see Uphoff et al. 2014 for sites sampled in other years). Deer Creek stations SU01, SU04, and SU05 corresponded to O'Dell et al. (1975) sites 1, 2, and 3 respectively (Figure 1-5). Two additional sites, SU02 and SU03 were sampled and analyzed in this system as well. The mainstem of the Choptank River had not been sampled previously, so 12 stations (CH100-CH111; Figure 1-6) were added in that system for analysis. Tuckahoe Creek stations TUC101, TUC102, TUC103, and TUC108 correspond to O'Dell et al. (1980) sites 4, 5, 6, and 8 respectively (Figure 1-7). Eight additional sites were sampled in this system and analyzed as well. Sampling in the Patapsco River was within an area similar to that of O'Dell et al. (1975), but sites were different (Figure 1-8).

The proportion of samples with Herring eggs and/or larvae present was estimated as:

$$^{(1)}\ P_{herr} = N_{present} / N_{total};$$

where $N_{present}$ equaled the number of samples with Herring eggs andor larvae present and N_{total} equaled the total number of samples taken. The SD of each P_{herr} was estimated as:

$$^{(2)}\ SD = [(P_{herr} \bullet (1 - P_{herr})) / N_{total}]^{0.5} \text{ (Ott 1977).}$$

The 90% confidence intervals were constructed as:

$$^{(3)}\ P_{herr} \pm (1.44 \bullet SD).$$

White Perch and Yellow Perch have been present in samples at the downstream-most one or two stations in Mattawoman Creek during 1989-1991 (Hall et al. 1992) and 2008-2017. We pooled data into two-or-three year intervals (1989-1991, 2008-2009, 2010-2012, 2013-2015, and 2016-2017) to estimate the proportion of samples with White or Yellow Perch eggs and larvae in order to gain enough precision to separate these estimates from zero. Formulae for estimating proportions, SD's, and 90% CI's were the same as for estimating P_{herr} (see above). White Perch spawning occurred at MC1 and MC2. Yellow Perch spawning was only detected at Station MC1.

Regression analyses examined relationships of development (C/ha) with standardized conductivity measurements (median conductivity adjusted for Coastal Plain or Piedmont background level; see below), C/ha and Herring spawning intensity (P_{herr}), standardized conductivity with P_{herr}, and estimates of watershed percentage that was agriculture or forest with P_{herr}. Data were from Mattawoman, Piscataway, Deer and Tuckahoe Creeks, and Bush, Choptank, and Patapsco Rivers. Thirty-four sets of estimates of C/ha, percent agriculture, percent forest, and P_{herr} were available (1991 estimates for Mattawoman Creek could be included), while 33 estimates were available for standardized conductivity (Mattawoman Creek conductivity data were not available for 1991). Examination of scatter plots suggested that a linear relationship was the obvious choice for C/ha and P_{herr}, that either linear or curvilinear relationships might be applicable to C/ha with standardized conductivity and standardized conductivity with P_{herr}, and that quadratic relationships best described the relationships of percentage of a watershed that was either agriculture or forest and P_{herr}. Power functions were used to fit curvilinear models. Linear regressions were analyzed in Excel, while the non-linear regression analysis used Proc NLIN in SAS (Freund and Littell 2006). A linear or nonlinear model was considered the best description if it was significant at $\alpha < 0.05$ (both were two parameter models), it explained more variability than the other (r^2 for linear or approximate r^2 for nonlinear), and examination of residuals did not suggest a problem. We expected negative relationships of P_{herr} with C/ha and standardized conductivity, while standardized conductivity and C/ha were expected to be positively related.

Conductivity was summarized as the median for the same stations that were used to estimate P_{herr} and was standardized by dividing by an estimate of the background expected from a stream absent anthropogenic influence (Morgan et al. 2012). Piedmont and Coastal Plain streams in Maryland have different background levels of conductivity. Morgan et al. (2012) provided two sets of methods of estimating spring base flow background conductivity for two different sets of Maryland ecoregions, for a total set of four potential background estimates. We chose the option featuring Maryland Biological Stream Survey (MBSS) Coastal Plain and Piedmont regions and the 25th percentile background level for conductivity. These regions had larger sample sizes than the other options and background conductivity in the Coastal Plain fell much closer to the observed range estimated for Mattawoman Creek in 1991 (61-114 μS/cm) when development was relatively low (Hall et al. 1992). Background conductivity used to standardize median conductivities was 109 μS/cm in Coastal Plain streams and 150 μS/cm in Piedmont streams. For Bush and Patapsco Rivers, whose watersheds run through both physiographic provinces, conductivities were standardized using 150 μS/cm of Piedmont streams since sampling locations were solely within that region.

Results

Development level of Piscataway, Mattawoman, and Deer Creeks, Bush River, and the Choptank River drainage (which includes Tuckahoe Creek) watersheds started at approximately 0.05 C/ha in 1950, while Patapsco River was approximately 0.17 C/ha at that time (Figure 1-9). Surveys conducted by O'Dell et al. (1975, 1980) in the 1970s, sampled largely rural watersheds (C/ha < 0.27) except for Piscataway Creek (C/ha = 0.48) and Patapsco River (C/ha = 0.43). By 1991, C/ha in Mattawoman Creek was similar to that of Piscataway in 1970. By the mid-2000s, Bush River and Piscataway Creek were at higher suburban levels of development (~1.30 C/ha) than Mattawoman Creek (~0.80 C/ha) and Patapsco River (~1.02 C/ha). Deer Creek (zoned for agriculture and preservation) and the Choptank River drainage (predominantly agricultural) remained rural through 2017 (0.24 and 0.13 C/ha, respectively; Figure 1-9).

Conductivity measurements in mainstem Mattawoman Creek during 2008-2017 never fell within the range observed during 1991 (Figure 1-10). Conductivity in Mattawoman Creek tributaries sampled during 2008-2017 often fell within the range observed during 1991 (Figure 1-10).

In 2017, conductivity measurements in mainstem Mattawoman Creek were elevated in March and April (> 130 μS/cm) and declined in May, with only one date (5/14/17) falling below the 1991 maximum (114 μS/cm; Figure 1-10). Conductivity measurements in tributary MUT3 in 2017 were above the 1991 maximum during the month of March, and had values similar to those observed in the tributaries during 2010-2013 the rest of the time (Figure 1-10). Conductivities at Mattawoman Creek's mainstem stations in 2009 were highly elevated in early March following application of road salt in response to a significant snowfall that occurred just prior to the start of the survey (Uphoff et al. 2010). Measurements during 2009 steadily declined for nearly a month before leveling off slightly above the 1989-1991 maximum. Temperatures were higher and snowfall lower in 2017 than in 2014 and 2015, with a conductivity pattern similar to 2010-2013 and 2016 (Figure 1-10). During 2014 and 2015, temperatures were colder and snowfall was higher; conductivities were elevated and similar to 2009. In general, highest conductivity measurements were at the most upstream mainstem site (MC4) and declined downstream to the site on the tidal border. This, along with low conductivities typically seen at the unnamed tributaries, indicated that development at and above MC4 associated with Waldorf affected water quality (Figure 1-10).

Table 1-4 provides summary statistics for each stream and year where conductivity was measured during spawning season. Conductivities were usually elevated beyond background levels in all streams studied during 2008-2017 and median conductivities ranged from 1.14- to 2.8-times times expected background levels. In general, Deer Creek and Choptank River appeared to have consistently low conductivity and Patapsco River and Piscataway Creek had consistently high conductivity. Mattawoman Creek exhibited the highest inter-annual variation (1.14- to 1.94-times background). Bush River and Tuckahoe Creek were similarly elevated (1.39- to 1.69-times for the former and~1.40-times for the latter) even though Tuckahoe Creek was much more rural.

During 1970-1989, 73% of monthly median conductivity estimates in Mattawoman Creek were at or below the background level for Coastal Plain streams; C/ha in the watershed increased from 0.25 to 0.41. Higher monthly median conductivities in the non-tidal stream were more frequent nearest the confluence with Mattawoman Creek's estuary and in the vicinity of Waldorf (RKM 35; Figure 1-11). Conductivity medians were highly variable at the upstream station nearest Waldorf during 1970-1989. During 2008-2016 (C/ha = 0.87-0.93), median spawning survey conductivities at mainstem stations MC2 to MC4, above the confluence of Mattawoman Creek's stream and estuary (MC1), were elevated beyond nearly all 1979-1989 monthly medians and increased with upstream distance toward Waldorf. Most measurements at MC1 fell within the upper half of the range observed during 1970-1989 (Figure 1-11). None of the non-tidal conductivity medians estimated at any mainstem site during 2008-2017 were at or below the Coastal Plain stream background criterion (109 μS/cm).

Herring spawning was detected at all mainstem stations in Mattawoman Creek (MC1-MC4) during 1971 and 1991 (Table 1-5). Herring spawning in fluvial Mattawoman Creek was detected at two mainstem sites during 2008-2009 and all four mainstem stations during 2010-2017. Herring spawning was not detected at tributary site MUT3 during 2008-2010, but was consistently present from 2011-2016. In 2017 herring spawning was not detected at MUT3. Spawning was intermittently detected at MUT4 and MUT5 in sampling during the 2000s. During 1971 and 1989-1991, White Perch spawning occurred annually at MC1 and intermittently at MC2. Stream spawning of White Perch in Mattawoman Creek was not detected during 2009, 2011, and 2012, but spawning was detected at MC1 during 2008, 2010 and 2013-2017, and at MC2 during 2013-2014 and 2016-2017. Spawning was detected at MC3 during 1971 and 2016. Station MC1 was the only stream station in Mattawoman Creek where Yellow Perch spawning has been detected in surveys conducted since 1971. Yellow Perch spawning occurred at station MC1 every year except 2009 and 2012 (Table 1-5).

Herring spawning was detected at all mainstem sites in Piscataway Creek in 2012-2014 (Table 1-6). Stream spawning of anadromous fish had nearly ceased in Piscataway Creek between 1971 and 2008-2009. Herring spawning was not detected at any site in the Piscataway Creek drainage during 2008 and was only detected on one date and location (one Herring larvae on April 28 at PC2) in 2009. Stream spawning of White Perch was detected at PC1 and PC2 in 1971, was not detected during 2008-2009 and 2012-2013, but was detected at PC1 in 2014 (Table 1-6).

Changes in stream site spawning of Herring, White Perch, and Yellow Perch in the Bush River stations during 1973, 2005-2008, and 2014 were not obvious (Table 1-7). Herring eggs and larvae were present at three to five stations (not necessarily the same ones) in any given year sampled. Occurrences of White and Yellow Perch eggs and larvae were far less frequently detected during 2005-2008 than 1973 and 2014 (Table 1-7).

O'Dell et al. (1975) reported that Herring, White Perch, and Yellow Perch spawned in Deer Creek during 1972 (Table 1-8). Three sites were sampled during 1972 in Deer Creek and one of these sites was located upstream of an impassable dam near Darlington (a fish passage was installed there in 1999). During 1972, Herring spawning was detected at both sites below the dam (SU01 and SU03), while White and Yellow Perch spawning were detected at the mouth

(SU01). During 2012-2015, Herring spawning was detected at all sites sampled in each year. White Perch spawning was not detected in Deer Creek in 2012 but was detected at three sites each in 2013 and 2014, and two sites in 2015. Yellow Perch spawning detection has been intermittent; evidence of spawning was absent in 2013 and 2015, while spawning was detected at two and three sites in 2012 and 2014, respectively (Table 1-8).

While the Choptank River itself had not been sampled prior to 2016 (Table 1-9), O'Dell et al. (1980) reported Herring, White Perch, and Yellow Perch spawned in its drainage (Tuckahoe Creek) during 1976-1977 (Table 1-10). Twelve sites were sampled during 1976-77 after installation of a fish ladder at the dam for the lake at Tuckahoe State Park. Sampling sites were established above and below the dam to determine the effectiveness of the fish ladder in passing anadromous and estuarine species (O'Dell et al. 1980). During 1976-77, White Perch, Yellow Perch, and Herring were collected downstream of the dam/fishway, while White Perch were documented on the upstream side. O'Dell et al. (1980) noted that this species might have been trapped behind the dam when it was built and that its presence did not necessarily indicate successful migration through the fish ladder since no other species were documented on the upstream side. Sites in common between current sampling (2016-2017) and the O'Dell et al. (1980) study included TUC101-TUC103 and TUC108 (Table 1-10). Herring spawning was detected at all sites sampled in 2017 with the exception of TUC109. A new fish ladder was installed in 1993 to replace the one referenced in O'Dell et al. (1980) and has been shown to pass Herring (J. Thompson, MD DNR, personal communication). White Perch spawning was detected in all but the two most upstream sites, both of which were located above the dam. In 2017 Yellow Perch spawning was detected at all sites below the dam, with the exception of TUC105, but not above the dam (Table 1-10).

Herring, White Perch, and Yellow Perch spawning during 2013-2017 occurred within the same reach of Patapsco River as sampled by O'Dell et al. (1975; Figure 1-8, Table 1-11). Herring spawning was detected at all sites sampled in the Patapsco River in 2013-2017, with the exception of MBSS 593 in 2016. White Perch and Yellow Perch spawning was more variable, with spawning presence being detected in as few as one site, and as many as all sites, throughout the sampling period (Table 1-11).

The 90% confidence intervals of P_{herr} (Figure 1-12) provided sufficient precision for us to categorize four levels of stream spawning: very low levels at or indistinguishable from zero based on confidence interval overlap (level 0); a low level of spawning that could be distinguished from zero (level 1); a mid-level of spawning that could usually be separated from the low levels (level 2); and a high level (3) of spawning likely to be higher than the mid-level. Stream spawning of Herring in Mattawoman Creek was categorized at levels 1 (2008-2009), 2 (2010 and 2012), and 3 (1991, 2011, and 2013-2017). Spawning in Piscataway Creek was at level 0 during 2008-2009, at level 2 during 2012, and at level 1 during 2013-2014. Bush River Herring spawning was characterized by levels 0 (2006), 1 (2005 and 2007-2008), and 2 (2014). Deer Creek (2012-2015), Tuckahoe Creek (2016-2017), and Choptank River (2016-2017) are the least developed watersheds and were characterized by the highest level of Herring spawning (level 3) in all years sampled (Figure 1-12).

The 90% CI's of proportions of samples with White Perch eggs and larvae at Mattawoman Creek's stations MC1 and MC2, pooled in 2-to-3-year intervals, indicated less stream spawning occurred during 2008-2012 than during 1989-1991 and 2013-2017 (Figure 1-13). The 90% CI's for stream spawning of Yellow Perch (at MC1 only) overlapped for all years indicating significant change in stream spawning had not been detected up to that point. Stream spawning of Yellow Perch in 2013-2017, however, does appear to have increased somewhat (Figure 1-13). Anecdotally, fishermen targeting Yellow Perch just downstream of Mattawoman's MC1 site indicated that 2016 had the highest number of adults seen and caught in recent (10+ year) memory (C. Hoover, MD DNR, personal communication).

Uphoff et al. (2017) examined associations among three land cover parameters: C/ha, agricultural land cover, and forest cover. They reported that there were strong, negative correlations between agricultural watershed percentages with C/ha; that forest cover and agriculture were strongly and negatively correlated; and that forest cover was poorly correlated with C/ha (Uphoff et al. 2017). MD DOP forest cover estimates mix forest cover in residential areas (trees over lawns) with true forest cover, clouding interpretation of forest influence. Uphoff et al. (2017) determined that subsequent analyses with P_{herr} beyond comparisons with C/ha were likely to be confounded by the close negative correlations so statistical analyses with land uses other than C/ha were not pursued. The preference for using C/ha in analyses is two-fold: we have already done considerable work using C/ha, and C/ha provides a continuous rather than episodic time-series. We did note, however, when these other land uses were predominant for particular P_{herr} outcomes.

Standardized conductivity increased with development, while P_{herr} declined with both development and standardized conductivity. Regression analyses indicated significant and logical relationships among P_{herr}, C/ha, and standardized median conductivity (Table 1-12). The relationship of C/ha with standardized median conductivity was linear, significant, and positive ($r^2 = 0.39$, P = 0.0001, N = 33; Figure 1-14). Estimates of P_{herr} were linearly, significantly, and negatively related to C/ha ($r^2 = 0.55$, P = <.0001, N = 34). Negative linear and curvilinear (power function) regressions similarly described the relationship of P_{herr} and standardized median conductivity ($r^2 = 0.22$, P < 0.0066; or approximate $r^2 = 0.20$, P < 0.0001, respectively), with linear regression explaining only slightly more variability (N = 33; Figure 1-15). Low estimates of P_{herr} (≤ 0.4) were much more frequent beyond the C/ha threshold (0.83 C/ha) or when standardized conductivity was 1.5-times or more than the baseline level (Figure 1-15). Estimates of P_{herr} were consistently above 0.6 in the three watersheds dominated by agriculture (Deer Creek, Tuckahoe Creek, and Choptank River; Figure 1-15). The only watershed in this analysis dominated by forest cover was Mattawoman Creek and only one estimate (1991 at 62.6% forest cover and C/ha = 0.46) represented development below the C/ha threshold. The 1971 estimate of P_{herr} was above 0.6 and was consistent with watersheds dominated by agriculture. Remaining estimates for Mattawoman Creek were represented by 53.9% forest cover with C/ha increasing from 0.87 in 2008 to 0.93 in 2014. Estimates of P_{herr} exhibited a much greater range, 0.08-0.77 (half had P_{herr} above 0.6), at these higher levels of development and lower forest cover, than less developed agricultural systems (0.62-0.87; Figure 1-15).

Discussion

Proportion of samples with Herring eggs and/or larvae (P_{herr}) provided a reasonably precise estimate of habitat occupation based on encounter rate. Regression analyses indicated significant and logical relationships among P_{herr}, C/ha, and conductivity consistent with the hypothesis that urbanization was detrimental to stream spawning. Estimates of P_{herr} were consistently high in the three watersheds dominated by agriculture. Importance of forest cover could not be assessed with confidence since it was possible that forest cover estimates included residential tree cover. Conductivity was positively related with C/ha in our analysis and with urbanization in other studies (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003;

Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012), but the relationship was not particularly strong.

Herring spawning became more variable in streams as watersheds developed. The surveys from watersheds with C/ha of 0.46 or less had high P_{herr}. Estimates of P_{herr} from Mattawoman Creek during 2008-2017 (C/ha was 0.87-0.93) varied from barely different from zero to high. The Mattawoman Creek time-series suggested that P_{herr} increased and stabilized at a higher level after stabilization of C/ha, i.e., P_{herr} rose from lowest levels in 2008-2009 to a consistently higher level after a preceding decade of high growth leveled off. Intensity of watershed change may be an important additional consideration along with level of development. Eggs and larvae were nearly absent from fluvial Piscataway Creek during 2008-2009, but P_{herr} rebounded to 0.45 in 2012 and then dropped again to 0.2 in 2013-2014 (C/ha was 1.41-1.50). The rebound in Herring spawning in Piscataway Creek during 2012 was concurrent with the lowest mean and median conductivities encountered there in the four years sampled. Variability of Herring spawning in Bush River during 2005-2008 and 2014 involved "colonization" of new sites as well as absence from sites of historical spawning (Uphoff et al. 2014). Limburg and Schmidt (1990) found a highly nonlinear relationship of densities of anadromous fish (mostly Alewife) eggs and larvae to urbanization in Hudson River tributaries, reflecting a strong, negative threshold at low levels of development.

Ranges of P_{herr} in study streams may have indicated variability in suitable habitat rather than abundance of spawners. In developed watersheds, a combination of urban and natural stream processes may create varying amounts of ephemeral spawning habitat annually and dampen spawning migrations through increased conductivity. Observed variation in P_{herr} would indicate wide annual and regional fluctuations in population size. However, stock assessments of Alewife and Blueback Herring along the Atlantic coast indicate they continue to be depleted and near historic lows (ASMFC 2009a; 2009b; 2017; Limburg and Waldman 2009; Lipkey and Jarzynski 2015; McClair and Jarzynski 2018). In the most recent Atlantic States Marine Fisheries Commission assessment, stocks in Maryland were either listed as no trend (or highly variable), stable (essentially depressed, but no trend) or unknown (ASMFC 2017; McClair and Jarzynski 2018). Maryland stock assessments do not appear to have enough resolution to address whether P_{herr} varied due to changes in suitable habitat or watershed-specific spawning stock variation.

Processes such as flooding, riverbank erosion, and landslides vary by geographic province (Cleaves 2003) and influence physical characteristics of streams. Unconsolidated layers of sand, silt, and clay underlie the Coastal Plain province and broad plains of low relief and wetlands characterize the natural terrain (Cleaves 2003). Coastal Plain streams have slow flows and sand or gravel bottoms (Boward et al. 1999). The Piedmont is underlain by metamorphic rocks and characterized by narrow valleys and steep slopes, with regions of higher land between streams in the same drainage. Most Piedmont streams are of moderate slope with rock or bedrock bottoms (Boward et al. 1999). The Piedmont province is an area of higher gradient change and more diverse and larger substrates than the Coastal Plain (Harris and Hightower 2011) that may offer greater variety of Herring spawning habitats.

Urbanization and physiographic province both affect discharge and sediment supply of streams (Paul and Meyer 2001; Cleaves 2003) that, in turn, could affect location, substrate composition, and extent and success of spawning. Alewife spawn in sluggish flows, while Blueback Herring spawn in sluggish to swift flows (Pardue 1983). American Shad select spawning habitat based on macrohabitat features (Harris and Hightower 2011) and spawn in moderate to swift flows (Hightower and Sparks 2003). Spawning substrates for Herring include gravel, sand, and detritus (Pardue 1983); these can be impacted by development. Strong impacts of urbanization on lithophilic spawners are well documented and range from loss of suitable substrate, increased embeddedness, lack of bed stability, and siltation of interstitial spaces (Kemp 2014). Broadcasting species, such as Herring, could be severely affected since they neither clean substrate during spawning nor provide protection to eggs and larvae in nests (Kemp 2014). Detritus loads in subestuaries are strongly associated with development (see Section 2) and urbanization affects the quality and quantity of organic matter in streams (Paul and Meyer 2001) that feed into subestuaries. Organic matter may be positively impacted by nutrients and negatively impacted by fine sediment from agriculture (Piggot et al. 2015).

Elevated conductivity, related primarily to chloride from road salt (but including most inorganic acids and bases; APHA 1979), has emerged as an indicator of watershed development (Wenner et al. 2003; Kaushal et al. 2005; Morgan et al. 2007; Morgan et al. 2012). Use of salt as a deicer may lead to both "shock loads" of salt that may be acutely toxic to freshwater biota and elevated baselines (increased average concentrations) of chloride that have been associated with decreased fish and benthic diversity (Kaushal et al. 2005; Wheeler et al. 2005; Morgan et al. 2007; 2012). Commonly used anti-clumping agents for road salt (ferro- and ferricyanide) that are not thought to be directly toxic are of concern because they can break down into toxic cyanide under exposure to ultraviolet light. Although the degree of breakdown into cyanide in nature is unclear (Pablo et al. 1996; Transportation Research Board 2007), these compounds have been implicated in fish kills (Burdick and Lipschuetz 1950; Pablo et al. 1996; Transportation Research Board 2007). Heavy metals and phosphorous may also be associated with road salt (Transportation Research Board 2007).

At least two hypotheses can be formed to relate decreased anadromous fish spawning to conductivity and road salt use. First, eggs and larvae may die in response to sudden changes in salinity and potentially toxic amounts of associated contaminants and additives. Second, changing stream chemistry may cause disorientation of spawning adults and disrupted upstream migration. Levels of salinity associated with our conductivity measurements are very low (maximum 0.2 ppt) and anadromous fish spawn successfully in brackish water (Klauda et al. 1991; Piavis et al. 1991; Setzler-Hamilton 1991). A rapid increase might result in osmotic stress and lower survival since salinity represents osmotic cost for fish eggs and larvae (Research Council of Norway 2009).

Elevated stream conductivity may prevent anadromous fish from recognizing and ascending streams. Alewife and Blueback Herring are thought to home to natal rivers to spawn (ASMFC 2009a; ASMFC 2009b), while Yellow and White Perch populations are generally tributary-specific (Setzler-Hamilton 1991; Yellow Perch Workgroup 2002). Physiological details of spawning migration are not well described for our target species, but homing migrations in anadromous American Shad and Salmon have been connected with chemical composition, smell, and pH of spawning streams (Royce-Malmgren and Watson 1987; Dittman and Quinn 1996; Carruth et al. 2002; Leggett 2004). Conductivity is related to total dissolved solids in water (Cole 1975) which reflects chemical composition.

An unavoidable assumption of regression analyses of P_{herr}, C/ha, and summarized conductivity was that watersheds at different levels of development were a substitute for time-series. Extended time-series of watershed-specific P_{herr} were not available. Mixing physiographic provinces in this analysis had the potential to increase scatter of points, but standardizing median conductivity to background conductivity moderated the province effect in analyses with that variable. Differential changes in physical stream habitat and flow with urbanization due to differences in geographic provinces could also have influenced fits of regressions. Estimates of C/ha may have indexed these physical changes as well as water chemistry changes, while standardized conductivity would only have represented changes in water chemistry. Estimates of C/ha explained more variation in P_{herr} (55%) than standardized conductivity (22%). Liess et al. (2016) developed a stress addition model for meta-analysis of toxicants and additional stressors of aquatic vertebrates and invertebrates and found that the presence of multiple environmental stressors could amplify the effects of toxicants 100-fold. This general concept may offer an explanation for the difference in fit of P_{herr} with C/ha and median conductivity, with conductivity accounting for water quality and C/ha accounting for multiple stressors.

Application of presence/absence data in management needs to consider whether absence reflects a disappearance from suitable habitat or whether habitat sampled is not really habitat for the species in question (MacKenzie 2005). Our site occupation comparisons were based on the assumption that spawning sites detected in the 1970s were indicative of the extent of habitat. O'Dell et al. (1975; 1980) summarized spawning activity as the presence of any species group's egg, larva, or adult (latter from wire fish trap sampling) for all samples at a site and we used this criterion (spawning detected at a site or not) for a set of comparisons. Raw data for the 1970s were not available to formulate other metrics. This site-specific presence/absence approach did not detect permanent site occupation changes or an absence of change since only a small number of sites could be sampled (limited by road crossings) and the positive statistical effect of repeated visits (Strayer 1999) was lost by summarizing all samples into a single record of occurrence in a sampling season. A single year's record was available for each of the watersheds in the 1970s and we were left assuming this distribution applied over multiple years of low development.

Proportion of positive samples (P_{herr}) incorporated spatial and temporal presence/absence and provided an economical, precise alternative estimate of habitat occupation based on encounter rate. Encounter rate is readily related to the probability of detecting a population (Strayer 1999). Proportions of positive or zero catch indices were found to be robust indicators of abundance of Yellowtail Snapper *Ocyurus chrysurus* (Bannerot and Austin 1983), age-0 White Sturgeon *Acipenser transmontanus* (Counihan et al. 1999; Ward et al. 2017), Pacific Sardine *Sardinops sagax* eggs (Mangel and Smith 1990), Chesapeake Bay Striped Bass eggs (Uphoff 1997), and Longfin Inshore Squid *Loligo pealeii* fishery performance (Lange 1991).

Unfortunately, estimating reasonably precise proportions of stream samples with White or Yellow Perch eggs annually would not be logistically feasible without major changes in sampling priorities. Estimates for Yellow or White Perch stream spawning would require more frequent sampling to obtain precision similar to that attained by P_{herr} since spawning occurred at fewer sites. Given staff and volunteer time limitations, this would not be possible within our current scope of operations. In Mattawoman Creek, it was possible to pool data across years to increase precision of estimates of proportions of samples with White Perch eggs and larvae (sites MC1 and MC2) or Yellow Perch larvae (MC1) for 1989-1991 collections to compare with 2008-2017 collections at the same combinations of sites. These estimates did not indicate a loss in stream spawning in downstream sites furthest from development (Waldorf).

Volunteer-based sampling of stream spawning during 2005-2017 used only stream drift nets, while O'Dell et al. (1975; 1980) and Hall et al. (1992) determined spawning activity with ichthyoplankton nets and wire traps for adults. Tabular summaries of egg, larval, and adult catches in Hall et al. (1992) allowed for a comparison of how site use in Mattawoman Creek might have varied in 1991 with and without adult wire trap sampling. Sites estimated when eggs and/or larvae were present in one or more samples were identical to those when adults present in wire traps were included with the ichthyoplankton data (Hall et al. 1992). Similar results were obtained from the Bush River during 2006 at sites where ichthyoplankton drift nets and wire traps were used; adults were captured by traps at one site and eggs and/or larvae at nine sites with ichthyoplankton nets (Uphoff et al. 2007). Wire traps set in the Bush River during 2007 did not indicate different results than ichthyoplankton sampling for Herring and Yellow Perch, but White Perch adults were observed in two trap samples and not in plankton drift nets (Uphoff et al. 2008). These comparisons of trap and ichthyoplankton sampling indicated it was unlikely that an absence of adult wire trap sampling would impact interpretation of spawning sites when multiple years of data were available.

The different method used to collect ichthyoplankton in Mattawoman Creek during 1991 could bias that estimate of P_{herr}, although presence-absence data tend to be robust to errors and biases in sampling (Green 1979; Uphoff 1997). Removal of 1991 data lowered the fit between C/ha and P_{herr} (from $r^2 = 0.55$, P = <.0001 to $r^2 = 0.54$, P = <.0001), but did not alter the negative relationship (95% CI's of slopes and intercepts of both models overlapped).

Absence of detectable stream spawning does not necessarily indicate an absence of spawning in the estuarine portion of these systems. Estuarine Yellow Perch presence-absence surveys in Mattawoman and Piscataway Creeks, and Bush River did not indicate that lack of detectable stream spawning corresponded to their elimination from these subestuaries. Yellow Perch larvae were present in upper reaches of both subestuaries, (see Section 2). Yellow Perch do not appear to be dependent on non-tidal stream spawning, but their use may confer benefit to the population through expanded spawning habitat diversity. Stream spawning is very important to Yellow Perch anglers since it provides access for shore fisherman and most recreational harvest probably occurs during spawning season (Yellow Perch Workgroup 2002).

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Table 1-1. Summary of subestuaries and their watershed size, Maryland Department of Planning (MD DOP) land use designation and estimates of land use types, and level of development (C/ha) during years sampled. DOP Year = the year DOP estimated land use that best matches sample year. Bush (w/o APG) refers to the portion of the Bush River watershed not including Aberdeen Proving Grounds.

River	Sample Year	DOP Year	C/ha	% Ag	% Forest	Watershed Size (ha)	Primary Land Use
Bush (w/o APG)	2005	2002	1.37	25.4	35	36,038	Urban
Bush (w/o APG)	2006	2002	1.41	25.4	35		
Bush (w/o APG)	2007	2010	1.43	18	29.9		
Bush (w/o APG)	2008	2010	1.45	18	29.9		
Bush (w/o APG)	2014	2010	1.51	18	29.9		
Choptank	2016	2010	0.18	55	27.8	38,216	Agriculture
Choptank	2017	2010	0.18	55	27.8		
Deer	2012	2010	0.24	44.6	28.4	37,697	Agriculture
Deer	2013	2010	0.24	44.6	28.4		
Deer	2014	2010	0.24	44.6	28.4		
Deer	2015	2010	0.24	44.6	28.4		
Mattawoman	1991	1994	0.46	13.8	62.6	24,441	Forest
Mattawoman	2008	2010	0.87	9.3	53.9		
Mattawoman	2009	2010	0.88	9.3	53.9		
Mattawoman	2010	2010	0.90	9.3	53.9		
Mattawoman	2011	2010	0.91	9.3	53.9		
Mattawoman	2012	2010	0.90	9.3	53.9		
Mattawoman	2013	2010	0.91	9.3	53.9		
Mattawoman	2014	2010	0.93	9.3	53.9		
Mattawoman	2015	2010	0.93	9.3	53.9		
Mattawoman	2016	2010	0.93	9.3	53.9		
Mattawoman	2017	2010	0.93	9.3	53.9		
Patapsco	2013	2010	1.11	24.4	30.4	93,895	Urban
Patapsco	2014	2010	1.12	24.4	30.4		
Patapsco	2015	2010	1.12	24.4	30.4		
Patapsco	2016	2010	1.12	24.4	30.4		
Patapsco	2017	2010	1.12	24.4	30.4		
Piscataway	2008	2010	1.41	10	40.4	17,642	Urban
Piscataway	2009	2010	1.43	10	40.4		
Piscataway	2012	2010	1.47	10	40.4		
Piscataway	2013	2010	1.49	10	40.4		
Piscataway	2014	2010	1.50	10	40.4		
Tuckahoe	2016	2010	0.07	66.6	25.4	39,388	Agriculture
Tuckahoe	2017	2010	0.07	66.6	25.4		

Table 1-2. Summary of subestuary watersheds sampled, years sampled, number of sites sampled, first and last dates of sampling, and stream ichthyoplankton sample sizes (N).

Subestuary	Year	Number of Sites	1st Sampling Date	Last Sampling Date	Number of Dates	N
Bush	2005	13	18-Mar	15-May	16	99
Bush	2006	13	18-Mar	15-May	20	114
Bush	2007	14	21-Mar	13-May	17	83
Bush	2008	12	22-Mar	26-Apr	17	77
Bush	2014	6	22-Mar	1-Jun	10	60
Choptank	2016	12	17-Mar	18-May	10	101
Choptank	2017	11	9-Mar	24-May	14	109
Deer	2012	4	20-Mar	7-May	11	44
Deer	2013	5	19-Mar	23-May	19	87
Deer	2014	5	2-Apr	28-May	12	60
Deer	2015	5	23-Mar	26-May	15	75
Mattawoman	2008	9	8-Mar	9-May	10	90
Mattawoman	2009	9	8-Mar	11-May	10	70
Mattawoman	2010	7	7-Mar	15-May	11	75
Mattawoman	2011	7	5-Mar	15-May	14	73
Mattawoman	2012	7	4-Mar	13-May	11	75
Mattawoman	2013	7	10-Mar	25-May	12	80
Mattawoman	2014	8	9-Mar	25-May	12	87
Mattawoman	2015	7	15-Mar	24-May	11	60
Mattawoman	2016	5	13-Mar	22-May	11	55
Mattawoman	2017	5	5-Mar	28-May	13	65
Patapsco	2013	4	19-Mar	30-May	22	40
Patapsco	2014	4	4-Apr	29-May	19	28
Patapsco	2015	4	25-Mar	28-May	18	32
Patapsco	2016	4	7-Mar	2-Jun	26	40
Patapsco	2017	4	9-Mar	6-Jun	21	40
Piscataway	2008	5	17-Mar	4-May	8	39
Piscataway	2009	6	9-Mar	14-May	11	60
Piscataway	2012	5	5-Mar	16-May	11	55
Piscataway	2013	5	11-Mar	28-May	11	55
Piscataway	2014	5	10-Mar	1-Jun	9	45
Tuckahoe	2016	10	16-Mar	16-May	12	97
Tuckahoe	207	10	8-Mar	23-May	11	102

Table 1-3. Summary of historical conductivity sampling in non-tidal Mattawoman Creek. RKM = site location in river kilometers from the mouth; Months = months when samples were drawn; Sum = sum of samples for all years.

RKM	Months	Sum	Years Sampled
12.4	1 to 12	218	1971, 1974-1989
18.1	4 to 9	8	1974
27	4 to 9	9	1970, 1974
30	8 and 9	2	1970
34.9	4 to 9	9	1970, 1974
38.8	8 and 9	2	1970

Table 1-4. Summary statistics of conductivity (µS/cm) for mainstem stations in Mattawoman, Piscataway, Deer, and Tuckahoe Creeks, and Bush and Choptank Rivers during 2005-2017. Unnamed tributaries were excluded from analysis. Tinkers Creek was included with mainstem stations in Piscataway Creek.

	Year												
Conductivity	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Bush													
Mean	269	206	263	237						276.7			
Standard Error	25	5	16	6						15			
Median	230	208	219	234						253.4			
Kurtosis	38	2	22	7						3.16			
Skewness	6	-1	4	0						1.56			
Range	1861	321	1083	425						606			
Minimum	79	0	105	10						107			
Maximum	1940	321	1187	435						713			
Count	81	106	79	77						60			
Choptank													
Mean												130.7	129.7
Standard Error												1.4	1.0
Median												133.2	129.8
Kurtosis												2.41	-0.05
Skewness												-1.07	-0.07
Range												89	49
Minimum												74	107
Maximum												163	156

Count	101	109

Table 1-4 cont.

	Year												
Conductivity	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
Deer													
Mean								174.9	175.6	170.3	191.8		
Standard Error								1.02	1.5	1.4	0.9		
Median								176.8	177.7	171.7	193.5		
Kurtosis								17.22	13.88	9.21	7.43		
Skewness								-3.78	-2.25	-2.42	-1.97		
Range								39.3	122	66	51		
Minimum								140.2	93	116	156		
Maximum								179.5	215	183	207		
Count								44	87	60	75		
Mattawoman													
Mean				120.1	244.5	153.7	147.5	128.9	126.1	179.4	181.8	180.3	151.2
Standard Error				3.8	19.2	38	2.8	1.9	2.4	9.1	6.5	4.1	3.7
Median				124.6	211	152.3	147.3	130.9	126.5	165.8	172.5	188.8	150.2
Kurtosis				2.1	1.41	1.3	8.29	-0.26	5.01	0.33	1.49	-0.80	-0.55
Skewness				-1.41	1.37	0.03	1.72	-0.67	-1.70	1.00	1.33	-0.68	-0.36
Range				102	495	111	117	49	96	261	185	93	102
Minimum				47	115	99	109	102	63	88	130	121	91
Maximum				148	610	210	225	151	158	350	315	214	193
Count				39	40	43	44	44	48	48	44	44	52
Patapsco													
Mean									406.2	282.5	346.8	310.4	340.3
Standard Error									48.7	8.0	18.2	30.6	15.1
Median									304.9	279.5	324.0	262.7	310.0
Kurtosis									12.13	-0.24	5.04	17.97	2.22
Skewness									3.33	0.42	1.97	3.99	1.36
Range									1554	166	487	1055	432
Minimum									245	219	216	188	175
Maximum									1799	385	703	1243	607
Count									40	28	32	40	40

Table 1-4 cont.

Conductivity	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
							Year						
Piscataway													
Mean				218.4	305.4			211.4	245	249.4			
Standard Error				7.4	19.4			5.9	6.9	11.1			
Median				210.4	260.6			195.1	238.4	230			
Kurtosis				-0.38	1.85			0.11	-0.29	2.56			
Skewness				0.75	1.32			0.92	0.73	1.50			
Range				138	641			163	173	274			
Minimum				163	97			145	181	174			
Maximum				301	737			308	354	449			
Count				29	50			44	44	36			
Tuckahoe													
Mean												152.2	155.9
Standard Error												2.4	1.7
Median												159.6	160.5
Kurtosis												-0.29	-0.18
Skewness												-0.68	-0.61
Range												103	82
Minimum												85	103
Maximum												188	185
Count												97	102

Table 1-5. Site-specific presence/absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Mattawoman Creek during 1971, 1989-1991, and 2008-2017. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-2.

	Year													
Station	1971	1989	1990	1991	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017
	Herring													
MC1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
MC2	1	1	1	1	0	0	1	1	1	1	1	1	1	1
MC3	1			1	1	1	1	1	1	1	1	1	1	1
MC4	1			1	0	0	1	1	1	1	1	1	1	1
MUT3	1				0	0	0	1	1	1	1	1	1	0
MUT4							0	0	1	0	0	0		
MUT5	1				1	0	0	0	0	0	1	0		
	White Perch													
MC1	1	1	1	1	1	0	1	0	0	1	1	1	1	1
MC2	0	0	1	0	0	0	0	0	0	1	1	0	1	1
MC3	1			0	0	0	0	0	0	0	0	0	1	0
	Yellow Perch													
MC1	1	1	1	1	1	0	1	1	0	1	1	1	1	1

Table 1-6. Site-specific presence/absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife) and White Perch spawning in Piscataway Creek during 1971, 2008-2009, and 2012-2014. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-3.

	Year					
Station	1971	2008	2009	2012	2013	2014
	Herring					
PC1	1	0	0	1	1	1
PC2	1	0	1	1	1	1
PC3	1	0	0	1	1	1
PTC1	1	0	0	1	1	0
PUT4	1		0	0	0	0
	White Perch					
PC1	1	0	0	0	0	1
PC2	1	0	0	0	0	0

Table 1-7. Site-specific presence/absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch spawning in Bush River streams during 1973, 2005-2008, and 2014. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-4.

	Year					
Station	1973	2005	2006	2007	2008	2014
			Herring			
BBR1	0	1	1	1	1	1
BCR1	1	0	0	1	0	1
BHH1	0	0	1	1	1	1
BJR1	0	1	1	1	0	1
BOP1	1	1	1	1	1	1
BWR1	1	0	0	1	0	1
			White Perch			
BBR1	1	0	0	0	0	1
BCR1	1	0	0	0	0	1
BHH1	0	0	0	0	0	0
BJR1	0	0	0	0	0	0
BOP1	1	0	0	1	0	1
BWR1	1	0	0	0	0	0
			Yellow Perch			
BBR1	1	0	0	0	0	0
BCR1	0	0	0	0	0	1
BHH1	0	0	0	0	0	1
BJR1	1	0	0	0	0	1
BOP1	0	0	0	0	0	0
BWR1	1	0	1	0	0	0

Table 1-8. Site-specific presence/absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Deer Creek during 1972 and 2012-2015. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-5.

	Year				
Station	1972	2012	2013	2014	2015
			Herring		
SU01	1	1	1	1	1
SU02		1	1	1	1
SU03		1	1	1	1
SU04	1	1	1	1	1
SU05	0		1	1	1
			White Perch		
SU01	1	0	1	1	1
SU02		0	1	0	1
SU03		0	0	1	0
SU04	0	0	1	1	0
SU05	0		0	0	0
			Yellow Perch		
SU01	1	1	0	1	0
SU02		1	0	1	0
SU03		0	0	1	0
SU04	0	0	0	0	0
SU05	0		0	0	0

Table 1-9. Site-specific presence/absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Choptank River during 2016-2017. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-6.

	Year					
Station	2016			2017		
	Herring	White Perch	Yellow Perch	Herring	White Perch	Yellow Perch
CH100	1	1	1	1	1	1
CH101	1	1	1	1	1	1
CH102	1	1	1	1	1	1
CH103	1	1	1	1	1	1
CH104	1	1	1	1	1	1
CH105	1	1	1	1	1	1
CH106	1	1	1	1	1	1
CH107	1	1	0	1	1	0
CH108	1	1	0	1	1	0
CH109	1	1	1	1	1	0
CH110	1	0	0	1	0	0
CH111	0	0	0			

Table 1-10. Site-specific presence/absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Tuckahoe Creek during 1976-77 and 2016-2017. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-7.

	Year		
Station	1976-77	2016	2017
	Herring		
TUC101	1	1	1
TUC102	1	1	1
TUC103	1	1	1
TUC104		1	1
TUC105		1	1
TUC106		1	1
TUC107		1	1
TUC108	0	1	1
TUC109		1	0
TUC110		0	1
	White Perch		
TUC101	1	1	1
TUC102	1	1	1
TUC103	1	1	1
TUC104		1	1
TUC105		1	1
TUC106		1	1
TUC107		1	1
TUC108	1	1	1
TUC109		0	0
TUC110		0	0
	Yellow Perch		
TUC101	1	1	1
TUC102	1	1	1
TUC103	1	1	1
TUC104		1	1
TUC105		1	0
TUC106		1	1
TUC107		1	1
TUC108	0	0	0
TUC109		0	0
TUC110		0	0

Table 1-11. Site-specific presence/absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Patapsco River during 1973 and 2013-2017. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-8.

O'Dell Sampling (1973)	
Station	Herring
Inland 1	0
Inland 2	1
Inland 3	1
Inland 4	1
Inland 5	0
	White Perch
Inland 1	1
Inland 2	1
Inland 3	0
Inland 4	1
Inland 5	0
	Yellow Perch
Inland 1	1
Inland 2	0
Inland 3	0
Inland 4	0
Inland 5	1

	Year				
Station	2013	2014	2015	2016	2017
			Herring		
USFWS Down River	1	1	1	1	1
USFWS Up River	1	1	1	1	1
MBSS 591	1	1	1	1	1
MBSS 593	1	1	1	0	1
			White Perch		
USFWS Down River	0	1	1	1	1
USFWS Up River	1	1	1	1	1
MBSS 591	0	1	0	1	1
MBSS 593	0	0	0	0	0
			Yellow Perch		
USFWS Down River	1	1	1	1	1
USFWS Up River	1	0	1	1	0
MBSS 591	0	0	0	1	0
MBSS 593	0	0	0	1	0

Table 1-12. Summary of best regression models for standardized conductivity (annual median/province background) versus development level (C/ha), proportion of samples with Herring eggs or larvae (P_{herr}) versus C/ha, and P_{herr} versus standardized conductivity.

Linear Model	Standardized conductivity = Structure density (C/ha)					
ANOVA	df	SS	MS	F	P	
Regression	1	1.58457	1.58457	19.45	0.0001	
Residual	31	2.52614	0.08149			
Total	32	4.11072				
r^2 = 0.3855						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.16009	0.10938	10.61	<.0001	0.93702	1.38317
C / ha	0.46321	0.10504	4.41	0.0001	0.24897	0.67744

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Structure density (C/ha)					
ANOVA	df	SS	MS	F	P	
Regression	1	1.31033	1.31033	39.15	<.0001	
Residual	32	1.07093	0.03347			
Total	33	2.38126				
r^2 = 0.5503						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.85201	0.06828	12.48	<.0001	0.71292	0.9911
C / ha	-0.41530	0.06637	-6.26	<.0001	-0.55049	-0.2801

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Standardized conductivity					
ANOVA	df	SS	MS	F	P	
Regression	1	0.50141	0.50141	8.49	0.0066	
Residual	31	1.83111	0.05907			
Total	32	2.33252				
r^2 = 0.2150						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.02113	0.19521	5.23	<.0001	0.62301	1.41926
Standardized Conductivity	-0.34925	0.11987	-2.91	0.0066	-0.59373	-0.10477

Figure 1-1. Watersheds sampled for stream spawning anadromous fish eggs and larvae during 2005-2017. Coastal Plain and Piedmont Regions are indicated.

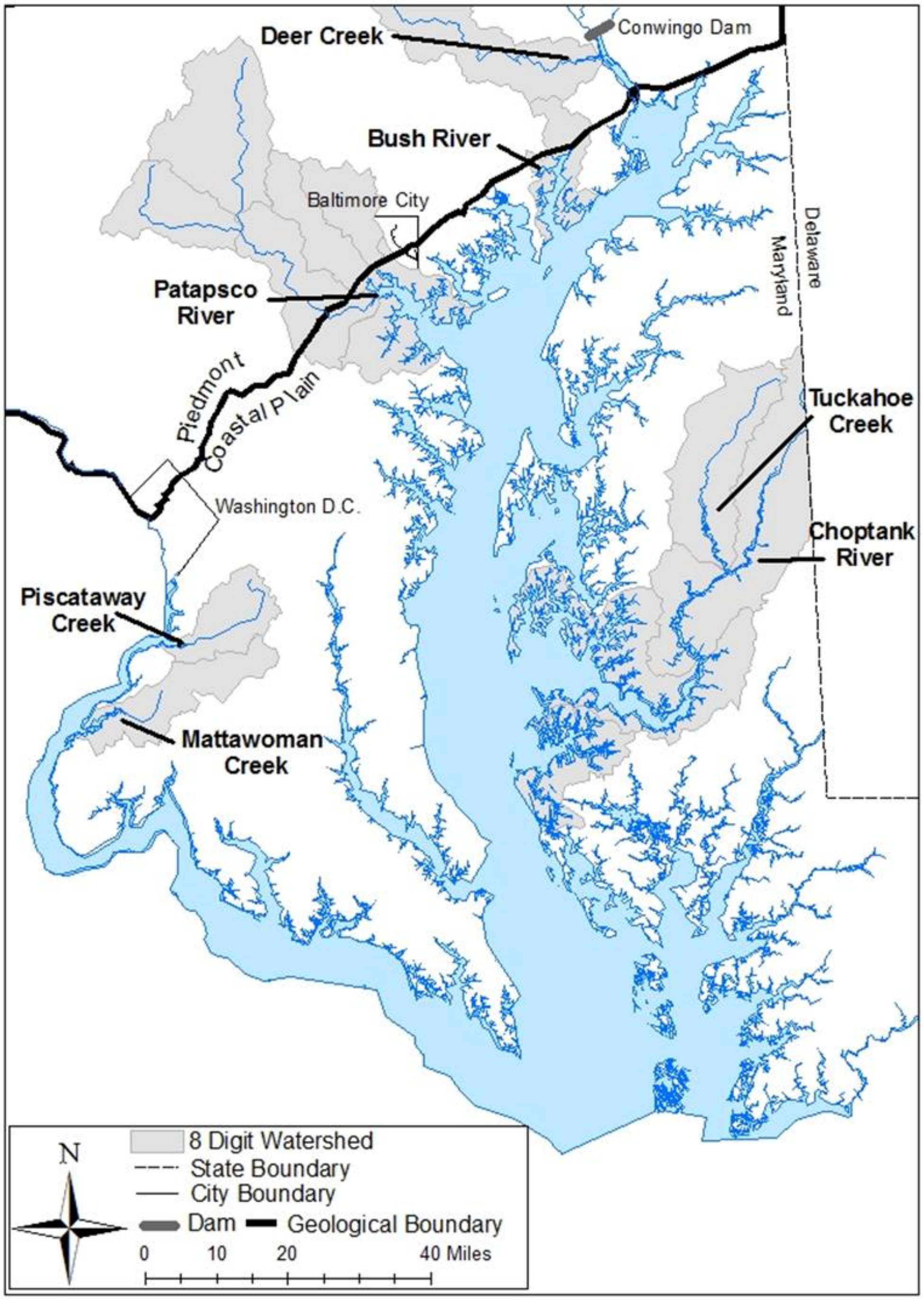


Figure 1-2. Mattawoman Creek's 1971 and 2008-2017 sampling stations. Bar approximates lower limit of development associated with the town of Waldorf.

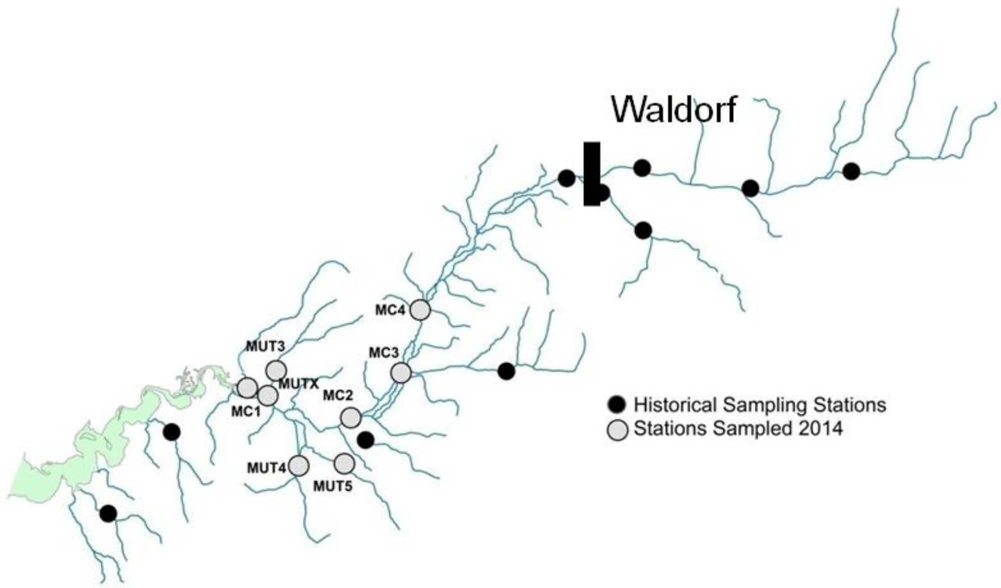


Figure 1-3. Piscataway Creek's 1971, 2008-2009, and 2012-2014 sampling stations.

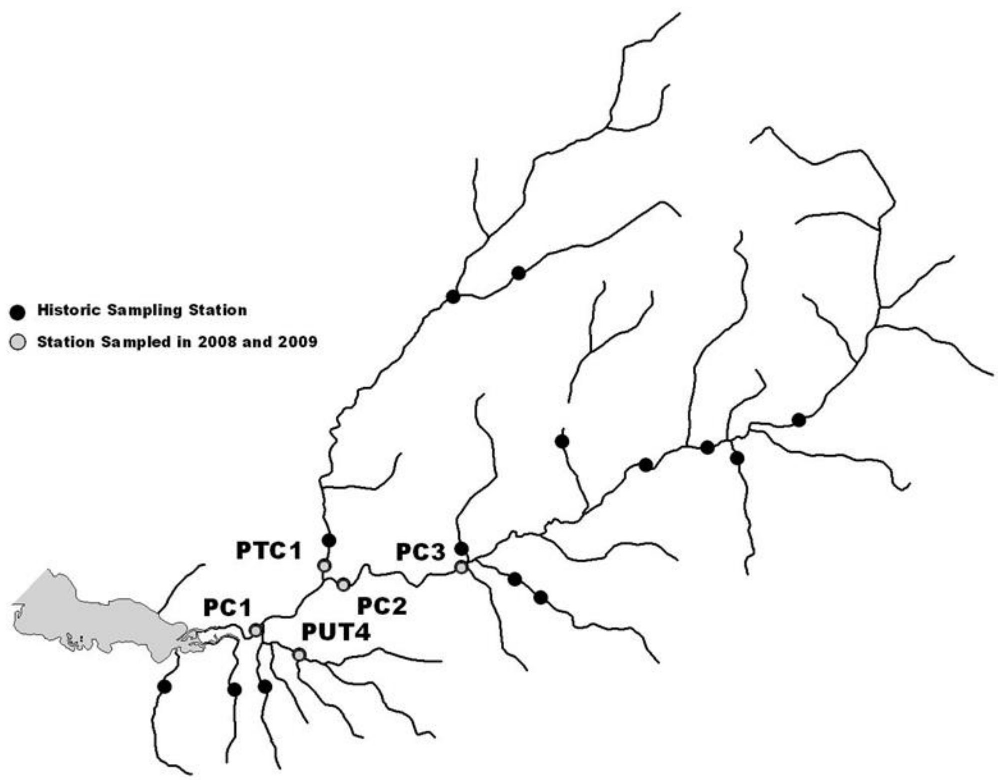


Figure 1-4. Bush River's 1973, 2005-2008, and 2014 sampling stations. Stations in Aberdeen Proving Grounds (APG) have been separated from other Bush River stations. Line delineates APG streams that were excluded.

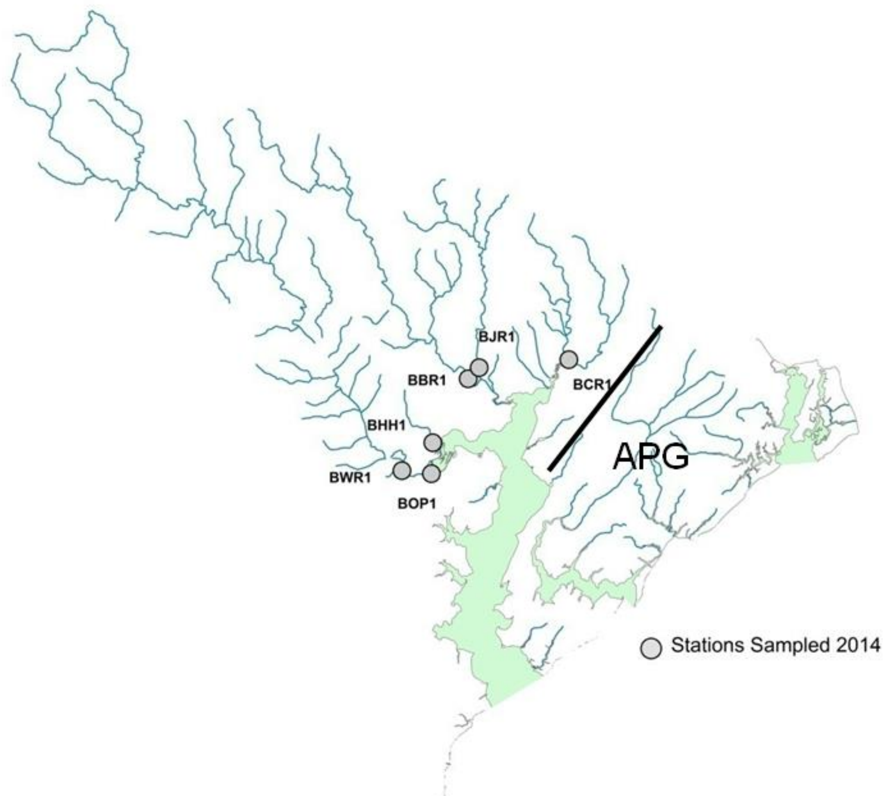


Figure 1-5. Deer Creek's 1972 and 2012-2015 sampling stations.

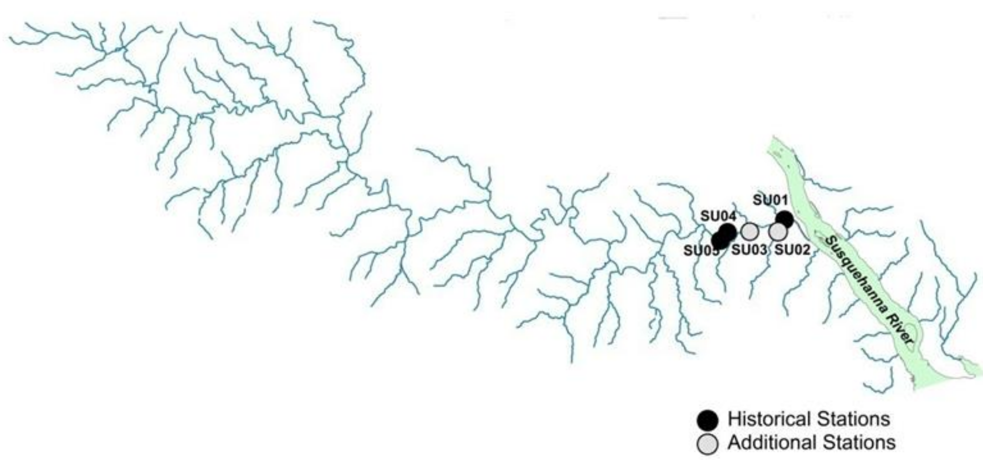


Figure 1-6. Choptank River's 2016-2017 sampling stations.

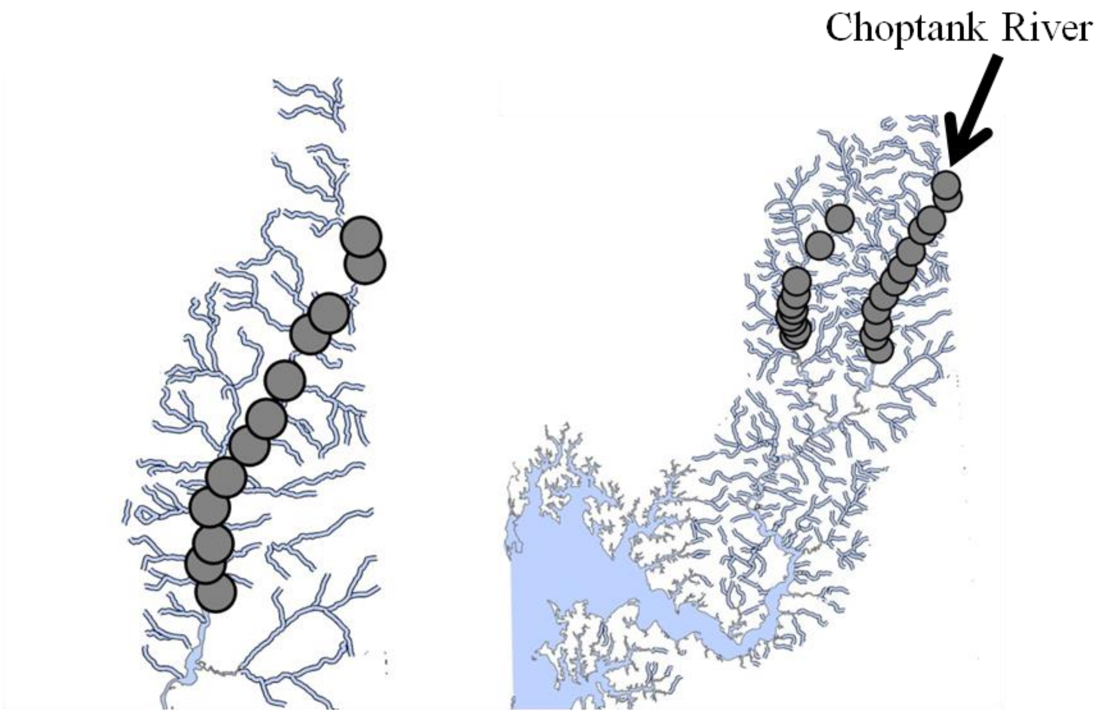


Figure 1-7. Tuckahoe Creek's 2016-2017 sampling stations.

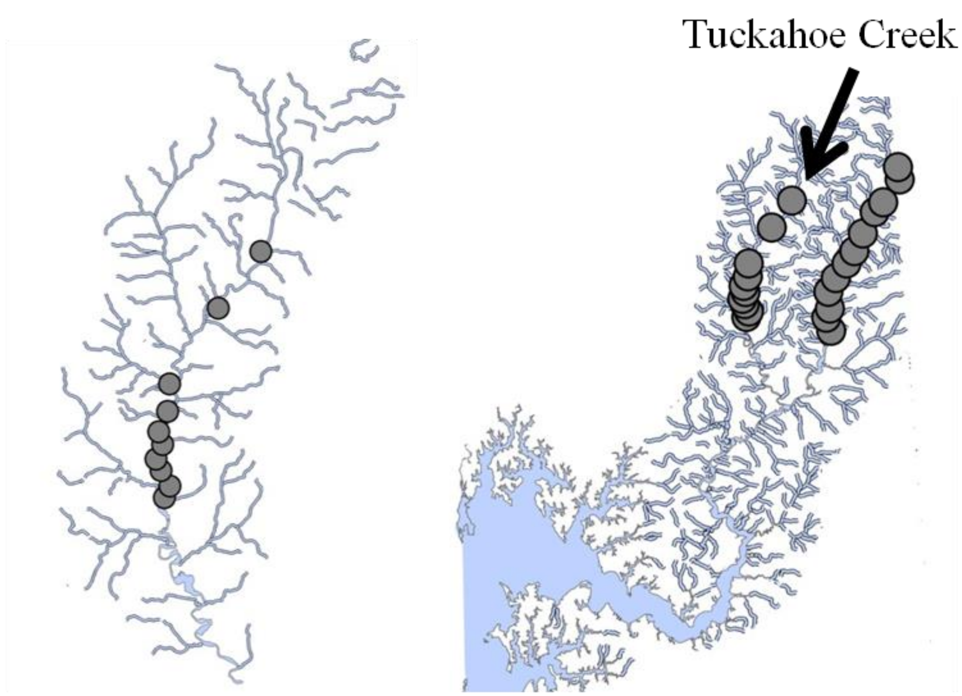


Figure 1-8. Patapsco River's 1973 and 2013-2017 sampling stations.

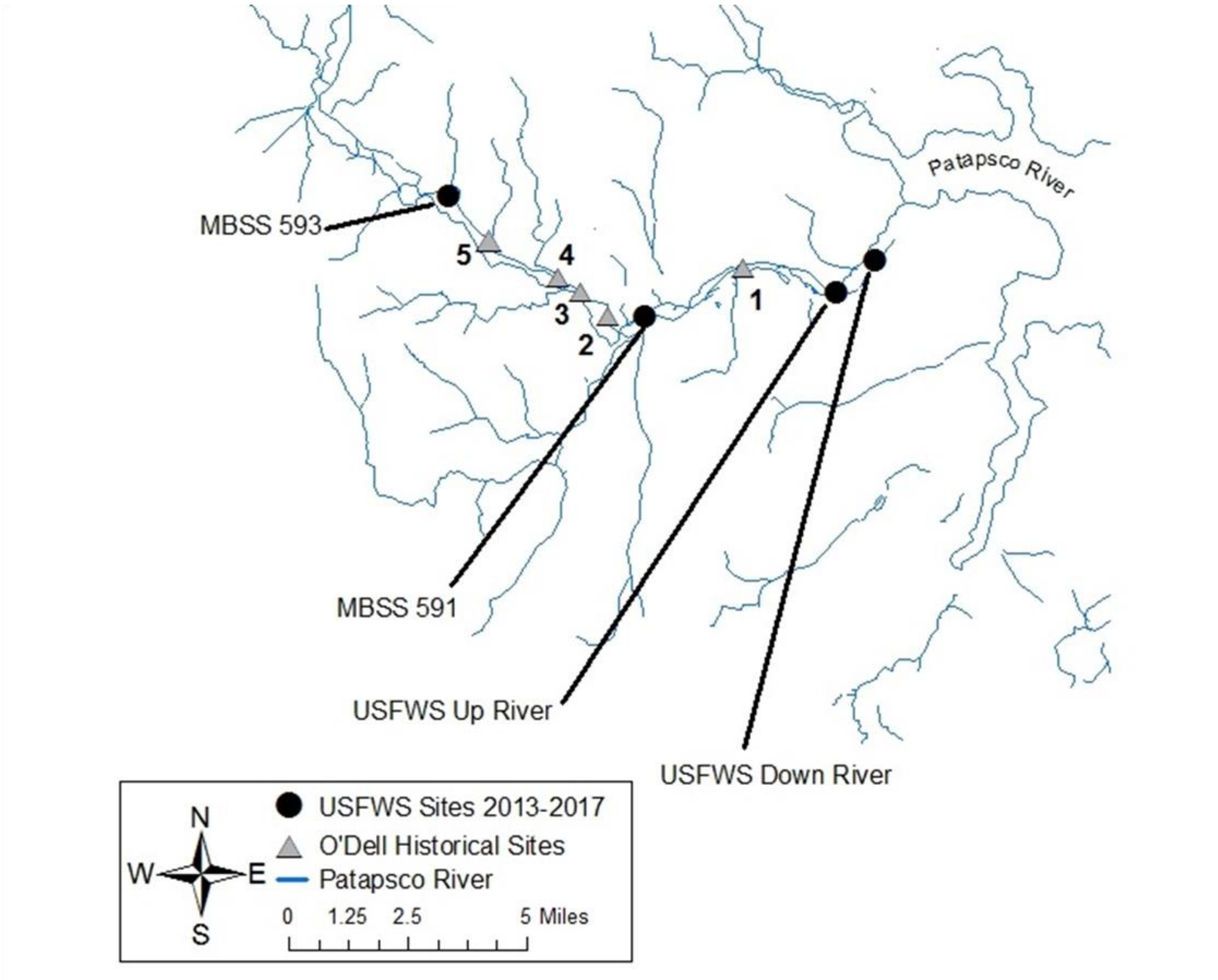


Figure 1-9. Trends in counts of structures per hectare (C/ha) during 1950-2014 in Mattawoman, Piscataway, and Deer Creeks, the Bush and Patapsco Rivers, and the Choptank River drainage watersheds. Updated estimates of C/ha were not available for 2015-2017. Large symbols indicate years when stream ichthyoplankton was sampled.

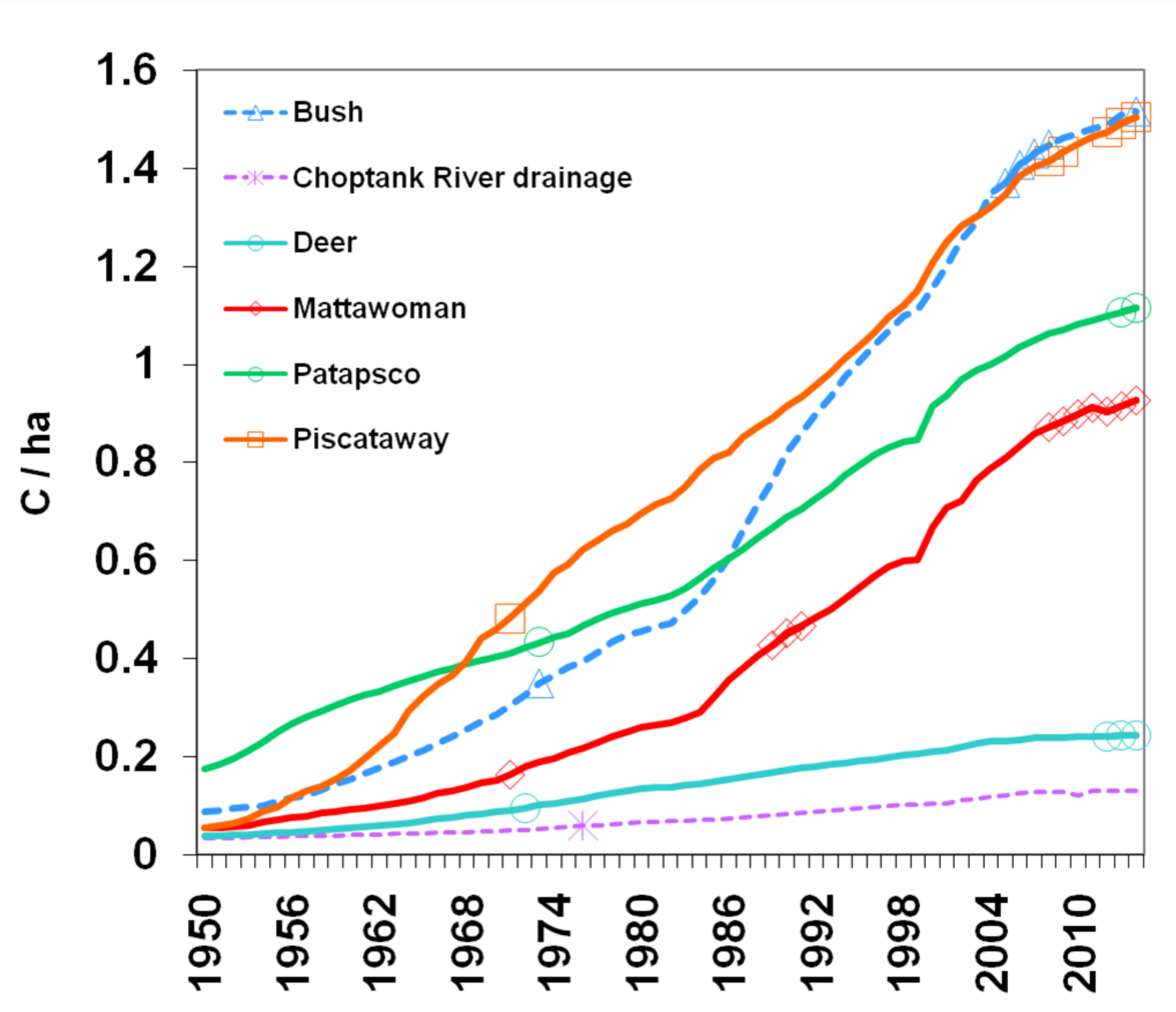


Figure 1-10. Stream conductivity measurements (µS/cm), by station and date, in Mattawoman Creek during (A) 2009, (B) 2010, (C) 2011, (D) 2012, (E) 2013, (F) 2014, (G) 2015, (H) 2016, and (I) 2017. Lines indicate conductivity range measured at mainstem sites (MC1 – MC4) during 1991 by Hall et al. (1992). Note changes in axis scale among years.

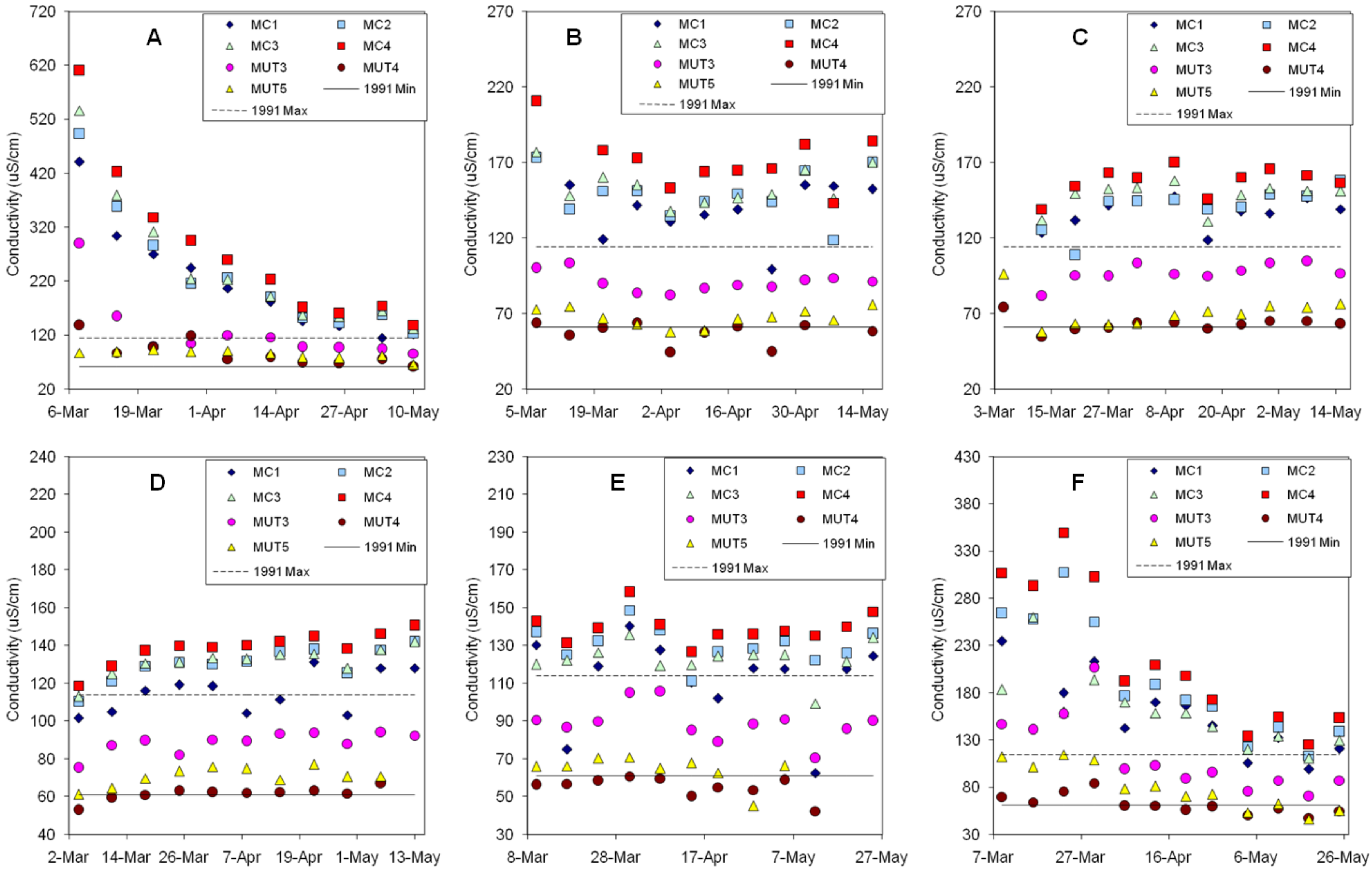
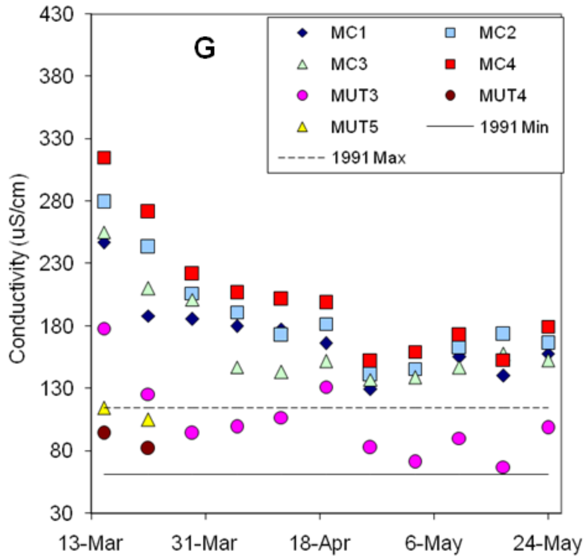
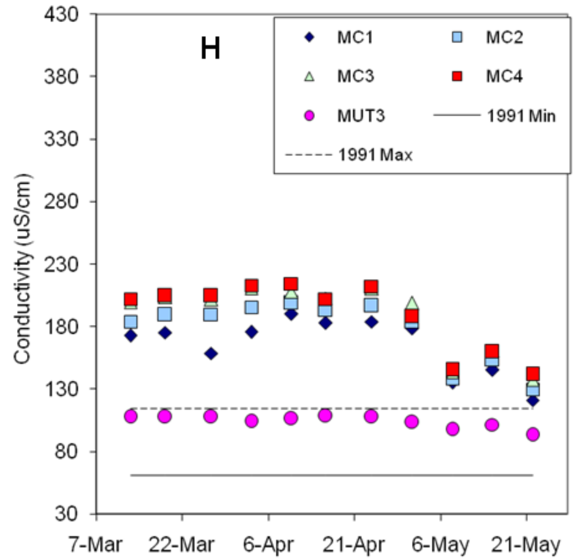


Figure 1-10 cont.





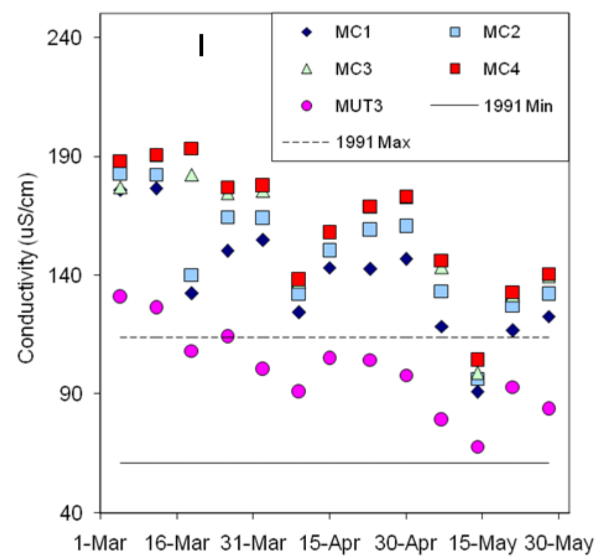


Figure 1-11. Historical (1970-1989) median conductivity measurements and current (2008-2017) anadromous spawning survey median conductivity in non-tidal Mattawoman Creek (between the junction with the subestuary and Waldorf) plotted against distance from the mouth. The two stations furthest upstream are nearest Waldorf. Median conductivity was measured during March-May, 2008-2017, and varying time periods (see Table 1-2) during 1970-1989.

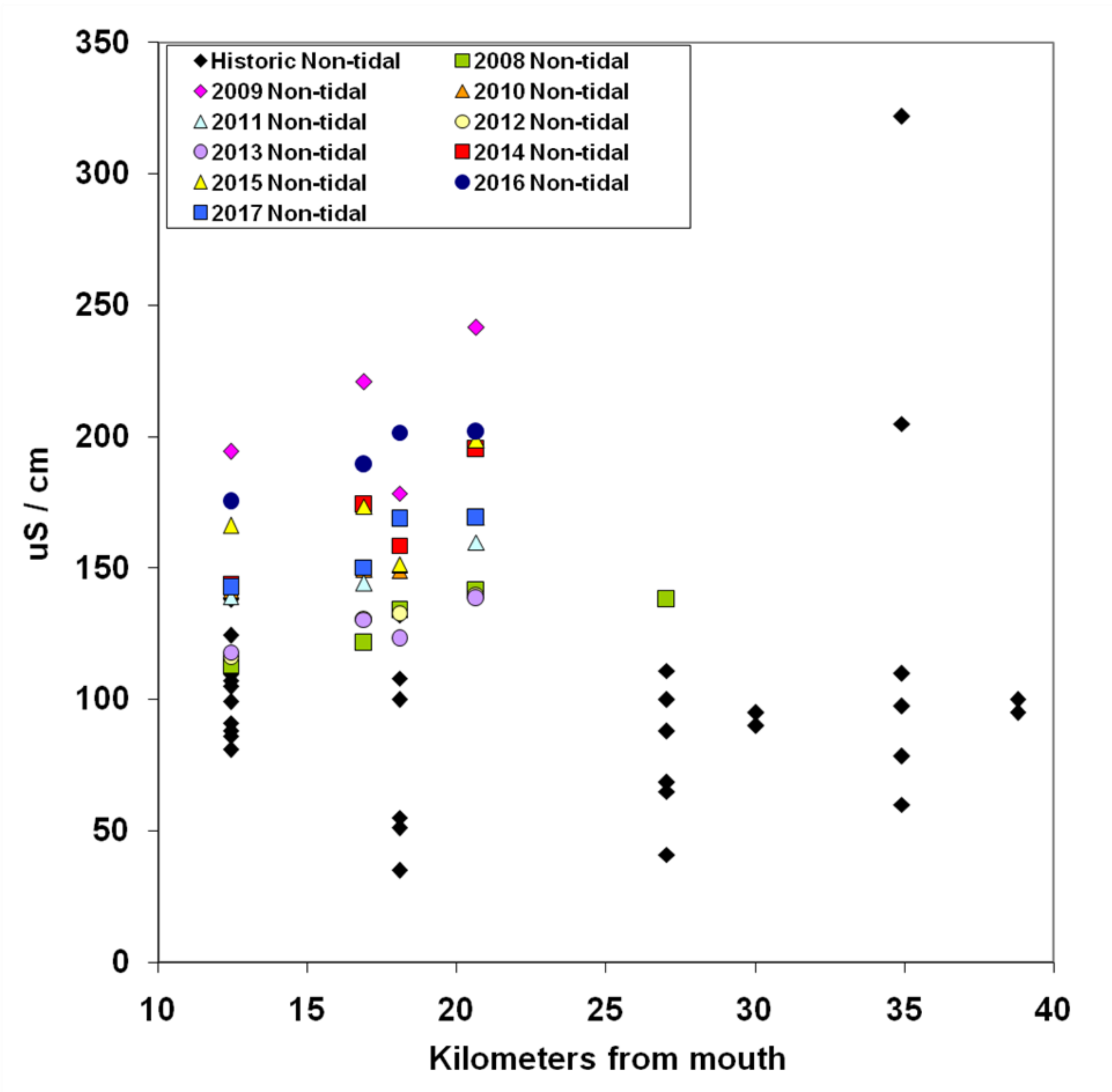


Figure 1-12. Proportion of samples (P_{herr}) with Herring and its 90% confidence interval for stream ichthyoplankton surveys in Mattawoman, Piscataway, Deer, and Tuckahoe Creeks, and Bush, Choptank, and Patapsco Rivers.

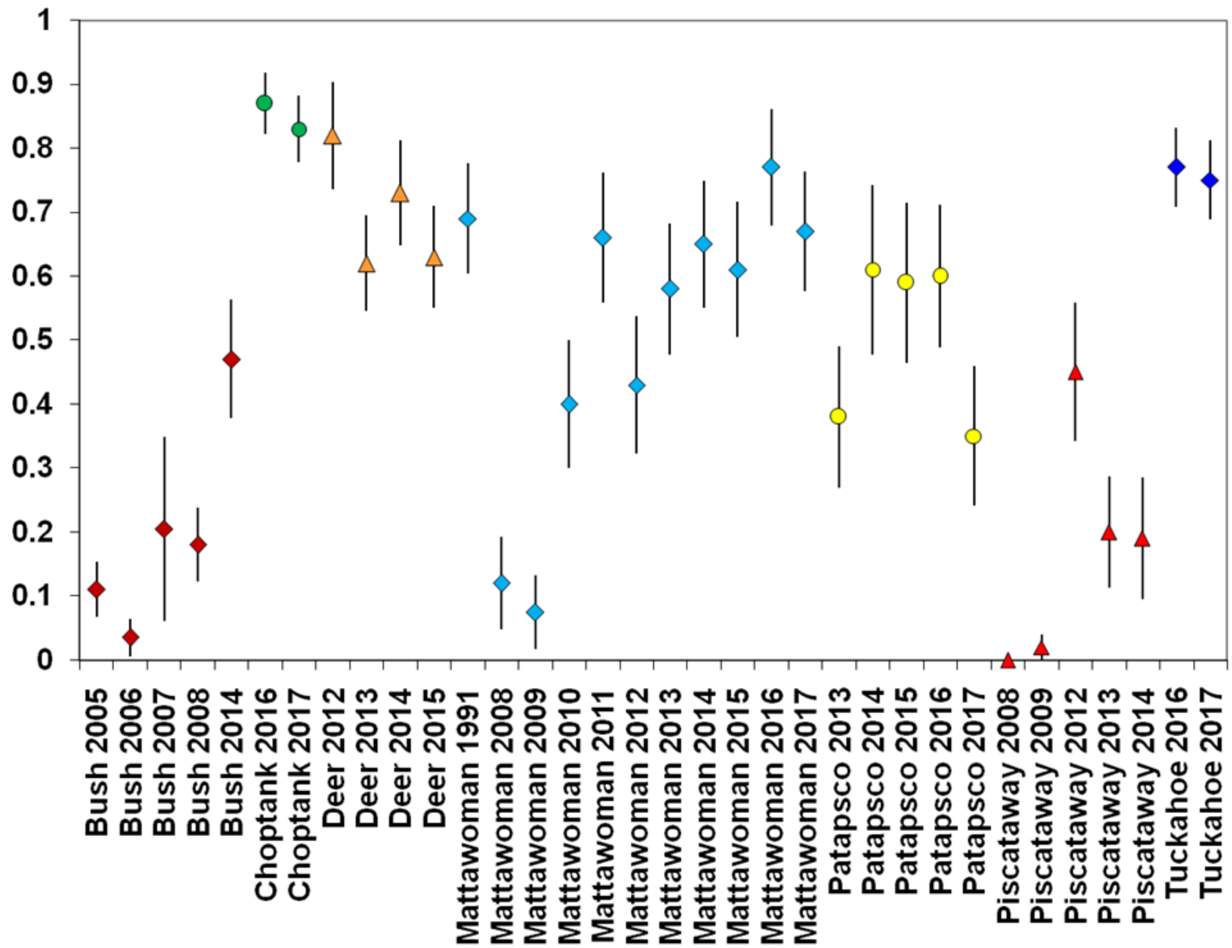


Figure 1-13. Mattawoman data pooled across years to form estimates of proportions of samples with White Perch (WP) eggs and-or larvae (sites MC1 and MC2) or Yellow Perch (YP) eggs and-or larvae (MC1) for 1989-1991 collections compared to 2008-2009, 2010-2012, 2013-2015, and 2016-2017 collections at the same combination of sites.

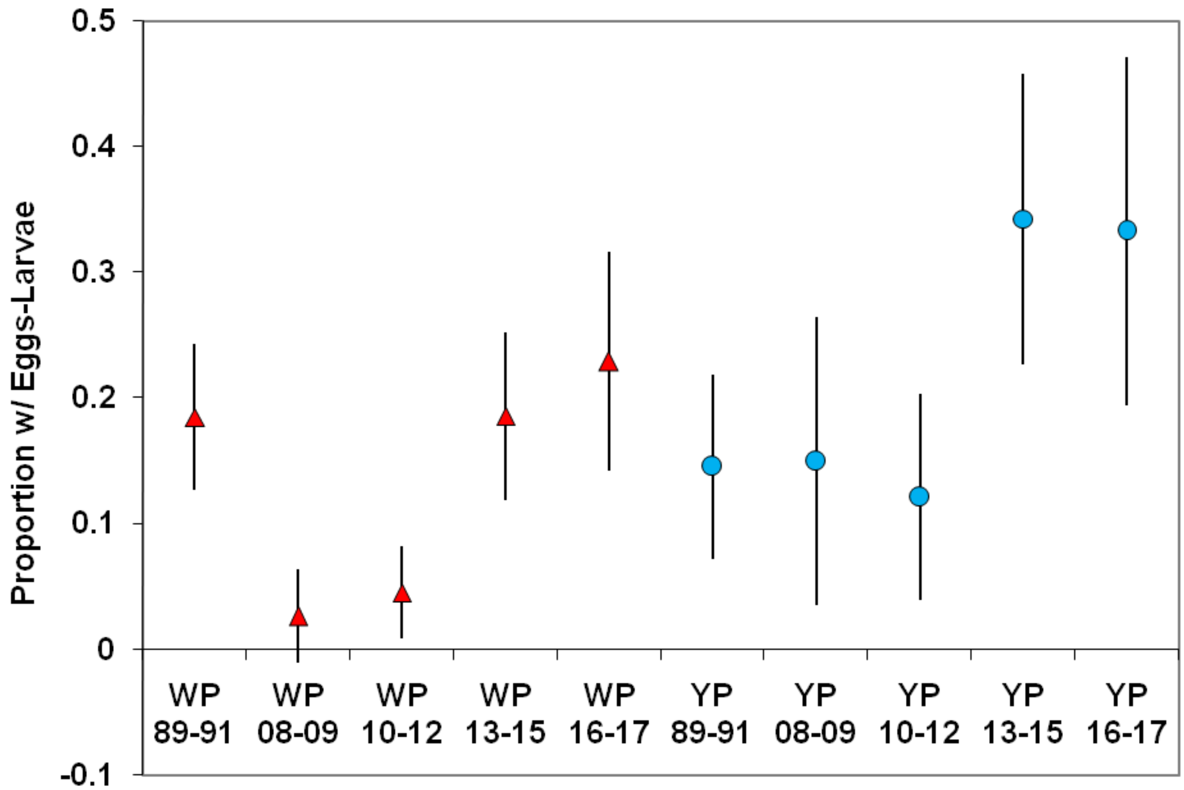


Figure 1-14. Standardized median conductivity during spring spawning surveys and level of development (C/ha). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

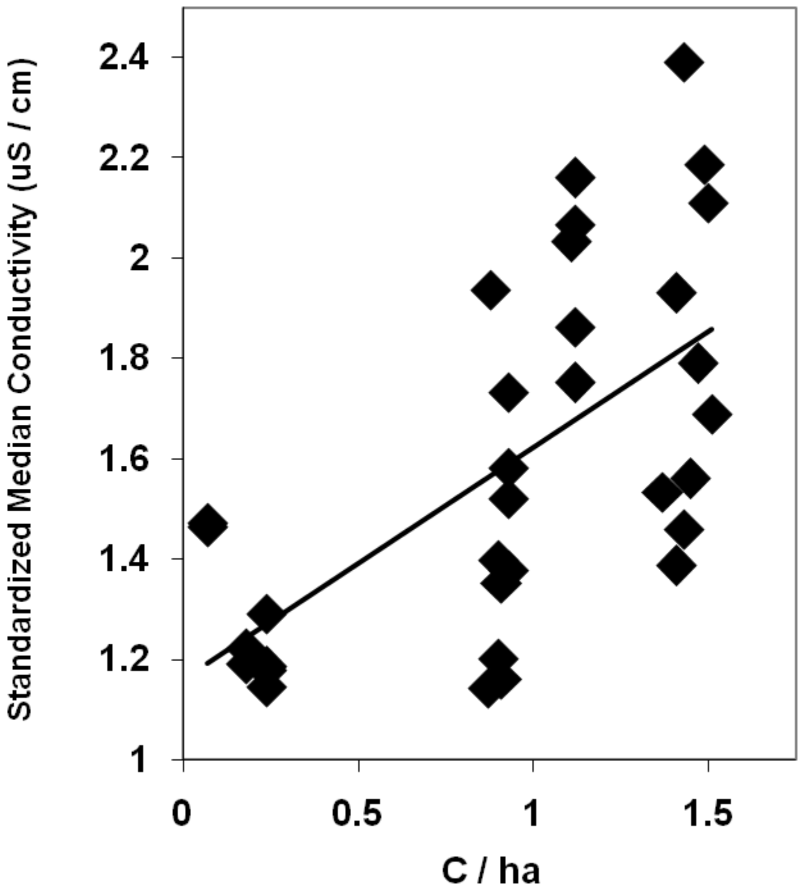
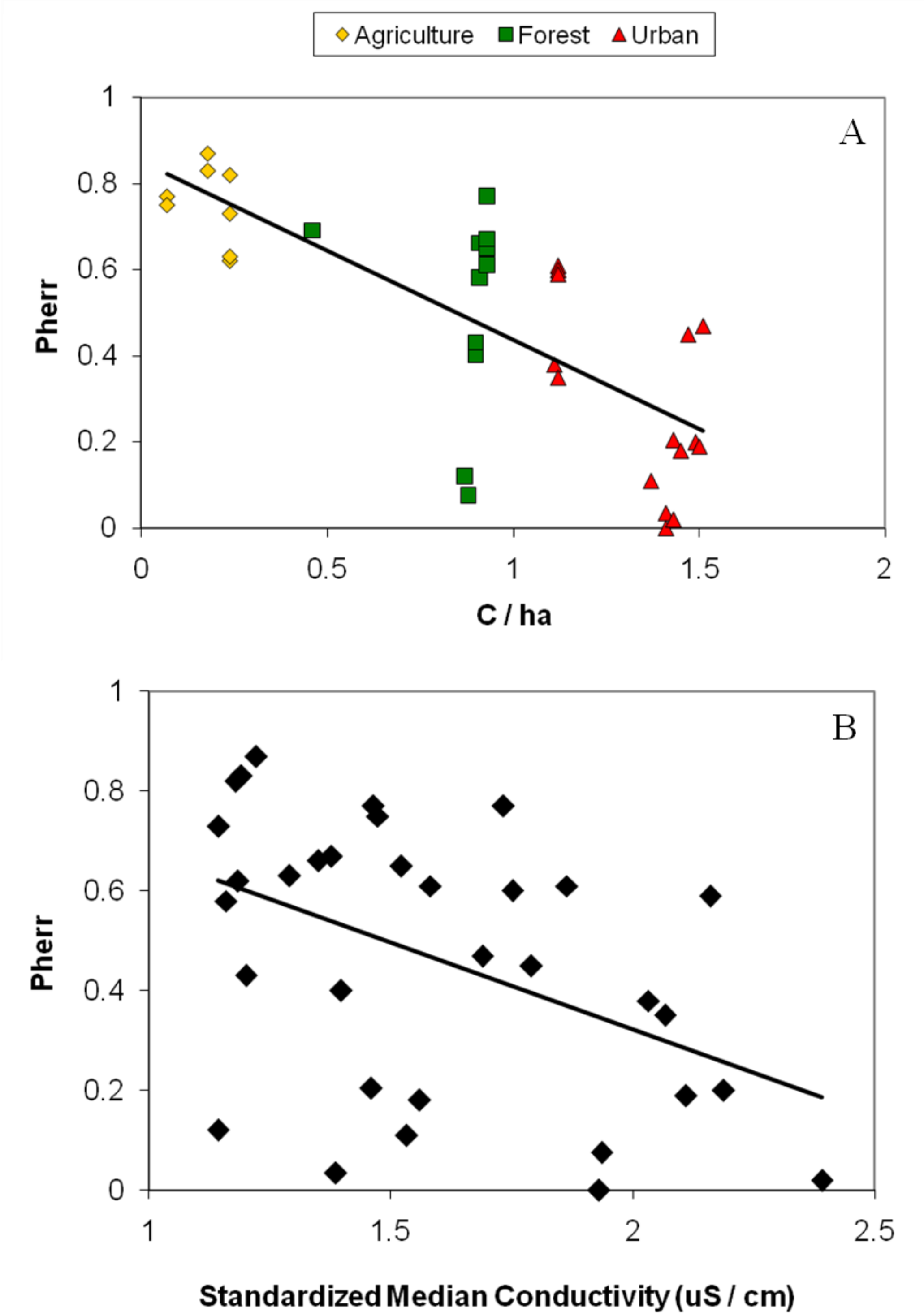


Figure 1-15. (A) Proportion of stream samples with Herring eggs and-or larvae (P_{herr}) and level of development (C/ha) with Department of Planning land use designations. (B) P_{herr} and standardized median spawning survey conductivity (uS/cm). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).



Section 2: Estuarine Yellow Perch Larval Presence/Absence Sampling

Carrie Hoover, Alexis Park, Jim Uphoff, Margaret McGinty, and Ben Wahle

Introduction

Annual L_p, the proportion of tows with Yellow Perch larvae during a standard time period and where larvae would be expected, provides a cost-effective measure of the product of egg production and survival through the early postlarval stage. Presence/absence sampling for Yellow Perch larvae in 2017 was conducted in the upper tidal reaches of the Choptank, Nanticoke, and Wicomico (eastern shore; ES from here on; there is a Wicomico River on the western shore as well) rivers (Figure 2-1). Sampling started the third week of March in the Choptank and Wicomico (ES) rivers, and the first week of April in the Nanticoke River. Sampling continued through the end of April.

In 2017 we used regression analyses to examine associations and relationships among land use types (development, agriculture, forest, and wetlands), L_p, organic matter availability, and watershed size. We also examined a hypothesis that watershed land use impacted related organic matter (OM) dynamics. Urbanization was expected to be negatively associated with extent of wetlands in Chesapeake Bay subestuary watersheds (Uphoff et al. 2011).

Methods

Conical plankton nets were towed from boats in upper portions of subestuaries to collect Yellow Perch larvae. Nets were 0.5-m in diameter, 1.0-m long, and constructed of 0.5 mm mesh. Nets were towed with the current for two minutes at a speed that maintained the net near the surface (approximately 2.8 km per hour). Temperature, dissolved oxygen, conductivity, and salinity were measured at each site on each sample date.

Ten sites were sampled twice weekly in the Choptank, Wicomico (ES), and Nanticoke rivers (Figure 2-1). In general, boundaries of areas sampled were determined from Yellow Perch larval presence in estuarine surveys conducted during the 1970s and 1980s (O'Dell 1987). However, the larger watersheds sampled in 2017 were not sampled by O'Dell (1987) and boundaries used were the same as the legal Striped Bass spawning areas. Uphoff (1991) found that the Choptank River Striped Bass spawning area and Yellow Perch larval nursery were very similar. Larval sampling usually occurs during late March through mid-to-late April, depending on larval presence and catchability.

Each sample, collected in a glass jar, was emptied into a dark pan and checked for larvae. Yellow Perch larvae can be readily identified in the field since they are larger and more developed than Striped Bass and White Perch larvae with which they could be confused (Lippson and Moran 1974). Contents of the jar were allowed to settle and then the amount of settled OM was assigned a rank: 0 = a defined layer was absent; 1 = defined layer on bottom; 2 = more than defined layer and up to ¼ full; 3 = more than ¼ to ½ and; 4 = more than ½ full. If a jar contained enough OM to obscure seeing larvae, it was emptied into a pan with a dark background and observed through a 5X magnifying lens. Organic matter was moved with a probe or forceps to free larvae for observation. If OM loads, wave action, or collector uncertainty prevented positive identification, samples were preserved and taken back to the lab for sorting.

Choptank and Wicomico (ES) Rivers were sampled by program personnel in 2017, while Nanticoke River was voluntarily sampled by the Maryland Fishing and Boating Services Shad and Herring program during its normal operations without charge to this grant.

The proportion of tows with Yellow Perch larvae (L_p) for each subestuary was determined annually for dates spanning the first catch through the last date that larvae were consistently present (L_p period) for as:

$$^{(1)}\ L_p = N_{present} / N_{total};$$

where $N_{present}$ equaled the number of samples with Yellow Perch larvae present during the L_p period and N_{total} equaled the total number of samples during the L_p period. Sites used to estimate L_p did not include downstream or upstream sites beyond the range where larvae were found. The SD of L_p was estimated as:

$$^{(2)}\ SD = [(L_p \cdot (1 - L_p)) / N_{total}]^{0.5} \text{ (Ott 1977).}$$

The 95% confidence intervals were constructed as:

$$^{(3)}\ L_p \pm 1.96 \cdot SD; \text{ (Ott 1977).}$$

In general, sampling to determine L_p began during the last days of March or first days of April and ended after larvae were absent (or nearly so) for two consecutive sampling rounds. In years where larvae disappeared quickly, sampling rounds into the third week of April were included in analysis even if larvae were not collected. Inclusion of these zeros reflected expectation (based on previous years) that larvae would be available to the sampling gear had they been there. This sampling schedule has been maintained for tributaries sampled by program personnel since 2006. Sampling by other Fisheries Service projects and volunteers sometimes did not adhere as strictly to this schedule.

Historical collections in the Choptank and Nanticoke Rivers targeted Striped Bass eggs and larvae (Uphoff 1997), but Yellow Perch larvae were also common (Uphoff 1991). Uphoff et al. (2005) reviewed presence/absence of Yellow Perch larvae in past Choptank and Nanticoke River collections and found that starting dates during the first week of April or early in the second week were typical and end dates occurred during the last week of April through the first week of May. Larval presence/absence was calculated from data sheets (reflecting lab sorting) for surveys through 1990. During 1998-2004, L_p in the Choptank River was determined directly in the field and recorded on data sheets (P. Piavis, MD DNR, personal communication). All tows were made for two minutes. Standard 0.5 m diameter nets were used in the Nanticoke River during 1965-1971 (1.0 • 0.5 mm mesh) and after 1998 in the Choptank River (0.5 mm mesh). Trawls with 0.5 m nets (0.5 mm mesh) mounted in the cod-end were used in the Choptank River during 1980-1990 (Uphoff 1997; Uphoff et al. 2005). Survey designs for the Choptank and Nanticoke Rivers were described in Uphoff (1997).

Methods used to estimate development (C/ha) and land use indicators (percent of watershed in agriculture, forest, wetlands, and urban land use) are explained in **General Spatial and Analytical Methods used in Job 1, Sections 1-3.** Development targets and limits and general statistical methods (analytical strategy and equations) are described there as well. Specific spatial and analytical methods for Section 2 are described below.

Estimates of C/ha and MD DOP land cover (agriculture, forest, and wetland) percentages were used as measures of watershed land use for analyses (Table 2-1). Whole watershed estimates were used with the following exceptions: Nanticoke, Choptank, Wicomico (ES), and Patuxent River watersheds were truncated at the lower boundaries of their striped bass spawning areas, and estimates for Choptank and Nanticoke River watersheds stopped at the Delaware border (latter due to lack of comparable land use data). Estimates of C/ha were available from 1950 through 2014 (M. Topolski, MD DNR, personal communication). Estimates of C/ha for 2014 were used to represent 2015, 2016, and 2017 in analyses for all systems.

Uphoff et al. (2012) developed L_p thresholds for brackish and tidal-fresh systems. Three brackish subestuaries with C/ha > 1.59 (10 estimates from Severn, South, and Magothy Rivers) exhibited chronically depressed L_p and their maximum L_p (0.40) was chosen as a threshold indicating serious deterioration of brackish subestuary larval nursery habitat. Similarly, tidal-fresh Piscataway Creek's four estimates of L_p (2008-2011) consistently ranked low when compared to other tidal-fresh subestuaries sampled (13th to 17th out of 17 estimates). The maximum for Piscataway Creek's four estimates, L_p = 0.65, was chosen as a threshold indicating serious deterioration of tidal-fresh larval habitat. Estimates of L_p would need to be consistently at or below this level to be considered "abnormal" as opposed to occasional depressions (Uphoff et al. 2012).

Two regression approaches were used to examine possible linear relationships between C/ha and L_p. First, separate linear regressions of C/ha against L_p were estimated for brackish and tidal-fresh subestuaries. If 95% CIs of slopes overlapped and 95% CIs of the intercepts did not overlap, we used the multiple regression of C/ha and salinity class against L_p. This latter approach assumed slopes were equal for two subestuary salinity categories, but intercepts were different (Freund and Littell 2006). Salinity was modeled as an indicator variable in the multiple regression with 0 indicating tidal-fresh subestuaries and 1 indicating brackish subestuary conditions. High salinity has been implicated in contributing to low L_p in Severn River (Uphoff et al. 2005). The association of mean salinity and impervious surface (IS) can be significant and strong (Uphoff et al. 2010), and salinity is important to formation of stressful dissolved oxygen (DO) conditions in summer in mesohaline tributaries that may cause endocrine disruption (Wu et al. 2003; see Section 3). Ricker (1975) warned against using well correlated variables in multiple regressions, so categorizing salinity for multiple or separate regressions of C/ha against L_p minimized confounding salinity with level of development. These same analyses were repeated using percent agriculture and percent forest land cover estimates in place of C/ha in regressions with L_p. Regression analyses were also used to examine relationships between C/ha, watershed size and salinity, and their effects on L_p

We used Akaike Information Criteria adjusted for small sample size, AICc, to evaluate the models that describe hypotheses that related changes in L_p to either C/ha, percent agriculture, or percent forest, for each salinity category (separate slopes) or to C/ha (percent agriculture or percent forest) and salinity category (common slopes, separate intercepts; Burnham and Anderson 2001):

$$^{(4)}\ AIC_c = -2(\text{log-likelihood}) + 2K + [(2K \cdot (K+1)) / (n-K-1)];$$

where n is sample size and K is the number of model parameters. Model parameters for the least squares regressions consisted of their mean square error estimates (variance), intercepts, slopes, and salinity category in the case of the multiple regression. We rescaled AIC_c values to Δ_i, (AIC_{ci} – minimum AIC_c), where i is an individual model, for the tidal-fresh or brackish regression compared to the multiple regression. The Δ_i values provided a quick "strength of evidence" comparison and ranking of models and hypotheses. Values of $\Delta_i \leq 2$ have substantial support, while those > 10 have essentially no support (Burnham and Anderson 2001).

An additional view of the relationship of L_p and C/ha was developed by considering dominant land use classification (land use type that predominated in the watershed) when interpreting plots of salinity classification (brackish or tidal-fresh), C/ha, and L_p. Dominant land use (agriculture, forest, or urban) was determined from Maryland Department of Planning estimates for 1973, 1994, 1997, 2002, or 2010 that fell closest to a sampling year (MD DOP

2013). Urban land consisted of high and low density residential, commercial, and institutional acreages (MD DNR 1999).

We used OM0 (proportion of samples without organic material, i.e., rank = 0) as our indicator of detritus availability, and proportions of samples without OM were estimated during 2011-2017. The distribution of OM ranks assigned to samples were highly skewed towards zero, and few ranks greater than one were reported. We regressed OM0 against C/ha, and were specifically interested in the relationship of the amount of organic matter to development. Examination of the plot of OM0 and C/ha suggested that the relationship could be nonlinear, with OM0 increasing at a decreasing rate with C/ha. We fit power and logistic growth functions to these data.

We were interested in links among OM0, percent wetlands in a watershed, and C/ha. Examination of the plot of percent wetlands and C/ha suggested that the relationship was nonlinear, with percentage of wetlands decreasing at a decreasing rate with C/ha, and appeared to be a mirror image of the plot of OM0 and C/ha. Examination of the plot of OM0 and percent wetlands suggested a linear relationship, with proportion of samples without organic material decreasing as percent wetlands per watershed increased. We fit power, logistic growth, or a linear function to these data sets, respectively.

Results

During 2017, sampling on Choptank River began on March 8, but due to weather and cold temperatures sampling was put on hold until March 21, and lasted until May 3. Sampling on Wicomico (ES) River began on March 9, was halted until March 22, and concluded on April 25. Samples through April 19 and April 11 were used to estimate L_p in Choptank and Wicomico (ES) rivers, respectively. Sampling began on April 3 in the Nanticoke River and ended on April 28, with all dates used for estimating L_p. Sites 1 to 6 were not used to estimate L_p in the Nanticoke River in 2017 because Yellow Perch larvae were continuously absent in these samples, indicating they were not nursery habitat during that year.

Based on 95% CIs, estimates of L_p during 2017 were sufficiently precise to separate them from thresholds (Figure 2-2). Estimates of L_p for Choptank and Wicomico (ES) rivers (L_p = 0.40 and 0.53, respectively) were above the brackish threshold (L_p = 0.33) and higher than Nanticoke River in 2017 (L_p = 0.22). The estimate of L_p for the Nanticoke River was below the brackish threshold, but did overlap with the lower portion of the Choptank River confidence interval (Figure 2-2).

Comparisons of L_p during 2017 with historical estimates for brackish subestuaries is plotted in Figure 2-3 and for tidal-fresh values in Figure 2-4. The range of C/ha values available for analysis with L_p was 0.05-2.78 for brackish subestuaries and 0.45-3.33 for tidal-fresh (Table 2-1). Estimates of L_p in 2017 were among the lowest historical values for Choptank and Nanticoke rivers; historical values were not available for Wicomico River (ES).

Separate linear regressions of C/ha and L_p by salinity category were significant at $P \leq 0.0005$; Table 2-2; Figure 2-5). These analyses indicated that C/ha was negatively related to L_p and L_p was, on average, higher in tidal-fresh subestuaries than in brackish subestuaries. Estimates of C/ha accounted for 26% of variation of L_p in brackish subestuaries and 34% in tidal-fresh subestuaries. Based on 95% CI overlap, intercepts were significantly different between tidal-fresh (mean = 0.95, SE = 0.09) and brackish (mean = 0.59, SE = 0.04) subestuaries. Mean slope for C/ha estimated for tidal-fresh subestuaries (mean = -0.29, SE = 0.07) were steeper, but 95% CI's overlapped CI's estimated for the slope of brackish subestuaries (mean = -0.16, SE =

0.04; Table 2-2). Both regressions indicated that L_p would be extinguished between 3.0 and 3.5 C/ha (Figure 2-5).

Overall, the multiple regression approach offered a similar fit (r^2 = 0.32; Table 2-2) to separate regressions for each salinity type. Intercepts of tidal-fresh and brackish subestuaries equaled 0.95 and 0.59, respectively; the common slope was -0.18. Predicted L_p over the observed ranges of C/ha would decline from 0.58 to 0.14 in brackish subestuaries and from 0.82 to 0 in tidal-fresh subestuaries (Figure 2-5).

Estimates of L_p were positively and weakly related to agriculture (r^2 = 0.09, P = 0.0208) or forest (r^2 = 0.10, P = 0.0185) in brackish tributaries (Table 2-2; Figure 2-5). Regressions of L_p and agriculture and forest in tidal-fresh subestuaries were very similar to that found in brackish ones, but sample sizes were lower so their level of significance was slightly above 0.05 (Table 2-2). Regression analysis did not suggest an association of wetlands with L_p in subestuaries of either salinity type so additional analyses were not conducted.

Additional regressions examining effects of watershed size and salinity on the relationship between C/ha and L_p indicated that considering either separately improved the regression fit similarly (overall, r^2 = 0.16, P <.0001; size, R^2 = 0.28, P <.0001; and salinity, R^2 = 0.32, P <.0001), but combining them into a single model did not improve the fit and watershed size was no longer significant (combined R^2 = 0.34; salinity, P = 0.0078 and size, P = 0.1433). Considering size separately, all tidal-fresh systems are within the small-system size category, so fit did not change from previous analyses (see Tables 2-2 and 2-3, respectively). The relationship between C/ha and L_p in small, brackish systems was better explained, however (r^2 = 0.57, P = 0.0001; Table 2-3). A relationship between C/ha and L_p was not detected for large systems (Table 2-3), so additional analyses were performed to explore their differences deeper.

Choptank, Patuxent, and Wicomico (ES) rivers were designated as large systems for additional analyses, and were defined as those watersheds which, overall, are considered brackish, but also have a large, distinct, tidal-fresh area. Analyses of these systems were limited to 2015-2017, where urban verses rural comparisons were available within the same year. Nanticoke River, also a large system, was excluded from analyses because sampling in this river either started later or ended earlier (collections were only made during the month of April) and level of effort was not comparable. Differences in L_p between up-river, mid-river, and down-river sections of large systems were not noted, even though the upriver portion of the Wicomico (ES) is in a high-development area, and upper sites in the Patuxent have elevated conductivity (an indication of possible water quality change due to development; Table 2-4; Figure 2-7).

Water quality parameters in large systems showed differences in some years among DO, pH, and conductivity between urban and rural systems (Table 2-5; Figures 2-8, 2-9, and 2-10). 2015 urban and rural water quality measurements were similar, with the exception of median conductivity which was significantly higher in urban Patuxent River (Table 2-5; Figure 2-8). In 2016, urban Patuxent River had higher DO, conductivity, and pH values than rural Choptank River (Table 2-5; Figure 2-9), and this was also true in 2017, when rural Choptank River had lower DO and pH values compared to more developed Wicomico (ES) River (Table 2-5; Figure 2-10). Conductivity has been consistently higher in the Patuxent, but surprisingly this is not the case in the Wicomico (ES) even though it passes through the city of Salisbury and upper sites are in a highly developed area. While these differences are not likely to be fatal, they do point to differences in dynamics among larger tributaries and between years.

Akaike's Information Criteria values equaled 9.4 for the regression of C/ha and L_p for brackish subestuaries, 9.9 for tidal-fresh estuaries, and 11.4 for the multiple regression that included salinity category. Calculations of Δi for brackish or tidal-fresh versus multiple regressions were approximately 2.04 and 1.54 (respectively), indicating that either hypothesis (different intercepts for tidal-fresh and brackish subestuaries with different or common slopes describing the decline of L_p with C/ha) were plausible (Table 2-6). These same calculations were performed from the regressions of percent agriculture or percent forest and L_p and results were almost identical to AIC values of C/ha and L_p (Table 2-6).

Although we have analyzed these data in terms of tidal-fresh and brackish subestuaries, inspection of Table 2-1 indicated an alternative interpretation based on primary land use estimated by MD DOP. Predominant land use at lowest levels of development may be influencing the intercept estimates. Rural watersheds (at or below C/ha target) were absent for tidal-fresh subestuaries analyzed and the lowest levels of development in tidal-fresh subestuary watersheds were dominated by forest (Figure 2-6). Dominant land cover estimated by MD DOP for watersheds of tidal-fresh subestuaries was split between forest (C/ha = 0.45-0.93; 18 observations) and urban (C/ha > 1.17; 14 observations). Nearly all rural land in brackish subestuary watersheds was in agriculture (C/ha < 0.22; 38 observations), while forest land cover was represented by six observations from Nanjemoy Creek (C/ha = 0.09) and one from Wicomico River (ES; C/ha = 0.67). The range of L_p was similar in brackish subestuaries with forest and agricultural cover, but the distribution shifted towards higher L_p in the limited sample from Nanjemoy Creek. Urban land cover predominated in 13 observations of brackish subestuaries (C/ha > 1.22; Table 2-1; Figure 2-6). Tidal-fresh subestuary intercepts may have represented the intercept for forest cover and brackish subestuary intercepts may have represented agricultural influence. If this is the case, then forest cover provides for higher L_p than agriculture. Increasing suburban land cover leads to a significant decline in L_p regardless of rural land cover type.

Estimates of C/ha and OM0 were significantly related. A non-linear power function fit the data (approximate $r^2 = 0.50$, $P < 0.0001$; $N = 33$), depicting OM0 increasing towards 1.0 at a decreasing rate as C/ha approached 1.50 (Figure 2-11). The relationship was described by the equation:

$$^{(5)}\ \mathrm{OM0} = 0.87 \cdot ((\mathrm{C/ha})^{0.14}).$$

Approximate standard errors were 0.04 and 0.05 for parameters a and b, respectively. A logistic growth function fit these data similarly, but one term was not significantly different from zero, so the model was rejected.

Percent wetlands (determined from the most recent MD DOP estimates in 2010) and development were negatively related. An inverse power function fit the relationship of C/ha and percent wetland well (approximate $r^2 = 0.50$, $P = 0.005$, $N = 10$; Figure 2-12). This relationship suggested that wetlands could be the main source of organic material in our study areas. We do not know whether lower wetland percentages were normal for more developed watersheds or if wetlands were drained and filled during development prior to wetland conservation regulations.

Discussion

General patterns of land use and L_p emerged from the expanded analyses conducted for this report: L_p was negatively related to development, positively associated with forest and agriculture, and not associated with wetlands. Wetlands appeared to be an important source of organic matter for subestuaries.

Rural features (agriculture, forest, and wetlands) were negatively correlated with development in the watersheds monitored for L_p (Uphoff et al. 2017). A broad range of L_p (near 0 to 1.0) was present up to 1.3 C/ha. Beyond 1.3 C/ha, estimates of L_p values were less than

0.65. A full range of L_p values occurred in subestuaries with agricultural watersheds (C/ha was $\leq$ 0.22). A forest cover classification in a watershed was associated with higher L_p (median L_p = 0.78) than agriculture (median L_p = 0.53) or development (median L_p = 0.35), but these differences may have also reflected dynamics unique to brackish or tidal-fresh subestuaries since all agricultural watersheds had brackish subestuaries and nearly all forested watersheds had tidal-fresh subestuaries.

At least five factors can be identified that potentially contribute to variations in L_p: salinity, summer hypoxia, maternal influence, winter temperature, and watershed development. These factors may not be independent and there is considerable potential for interactions among them.

Salinity may restrict L_p in brackish subestuaries by limiting the amount of available low salinity habitat over that of tidal-fresh subestuaries. Uphoff (1991) found that 90% of larvae collected in Choptank River (based on counts) during 1980-1985 were from 1‰ or less. Approximately 85% of Yellow Perch larvae collected by Dovel (1971) from Magothy and Patuxent rivers, and Head-of-Bay, during 1963-1967 were collected at salinity 1‰ or less.

Severn River offers the most extensive evidence of salinity changes in a subestuary that were concurrent with development from 0.35 to 2.29 C/ha. During 2001-2003, salinity within Severn River's estuarine Yellow Perch larval nursery ranged between 0.5 and 13‰ and 93% of measurements were above the salinity requirement for eggs and larvae of 2‰ (Uphoff et al. 2005). Muncy (1962) and O'Dell's (1987) descriptions of upper Severn River salinity suggested that the nursery was less brackish in the 1950s through the 1970s than at present, although a single cruise by Sanderson (1950) measured a rise in salinity with downstream distance similar to what Uphoff et al. (2005) observed. Most Yellow Perch spawning in Severn River during 1958 occurred in waters of 2.5‰ or less (Muncy 1962). Mortality of Yellow Perch eggs and prolarvae in experiments generally increased with salinity and was complete by 12‰ (Sanderson 1950; Victoria et al. 1992). Uphoff et al. (2005) estimated that nearly 50% of the historic area of estuarine nursery for Yellow Perch was subject to salinities high enough to cause high mortality. Salinity in the estuarine nursery of Severn River varied without an annual pattern even though conditions went from extremely dry (2001-2002) to extremely wet (2003; Uphoff et al. 2005).

As development increases, rainfall flows faster across the ground and more of it reaches fluvial streams rather than recharging groundwater (Cappiella and Brown 2001; Beach 2002). In natural settings, very little rainfall is converted to runoff and about half is infiltrated into underlying soils and the water table (Cappiella and Brown 2001). These pulses of runoff in developed watersheds alter stream flow patterns and could be at the root of the suggested change in salinity at the head of the Severn River estuary where the larval nursery is located (Uphoff et al. 2005).

In our studies, suburban mesohaline subestuaries commonly exhibit summer hypoxia in bottom channel waters, but it is less common in agricultural watersheds (see Section 3). Stratification due to salinity is an important factor in development of hypoxia in mesohaline subestuaries, while hypoxia is rarely encountered in tidal-fresh and oligohaline subestuaries (see Section 3). Depressed egg and larval viability due to endocrine disruption may follow inadequate DO the previous summer (Wu et al. 2003; Uphoff et al. 2005; Thomas and Rahman 2011; Tuckey and Fabrizio 2016). Ovaries of Yellow Perch are repopulated with new germs cells during late spring and summer after resorptive processes are complete (Dabrowski et al. 1996, Ciereszko et al. 1997).

Hypoxia in coastal waters reduces fish growth and condition due to increased energy expenditures to avoid low DO and compete for reduced food resources (Zimmerman and Nance 2001; Breitburg 2002; Stanley and Wilson 2004). Reproduction of mature female fish is higher when food is abundant and condition is good (Marshall et al. 1999; Lambert and Dutil 2000; Rose and O'Driscoll 2002; Tocher 2003), but stress may decrease egg quality (Bogevik et al. 2012). A female Yellow Perch's energetic investment provides nutrition for development and survival of its larvae until first feeding (Heyer et al. 2001) and differences in Yellow Perch larval length, yolk volume, and weight were attributed to maternal effects in Lake Michigan (Heyer et al. 2001).

Widespread low L_p occurs sporadically in Chesapeake Bay subestuaries that appears to be linked to high winter temperatures (Uphoff et al. 2013). During 1965-2012, estimates of L_p less than 0.5 did not occur when average March air temperatures were 4.7°C or less (N = 3), while average March air temperatures of 9.8°C or more were usually associated with L_p estimates of 0.5 or less (7 of 8 estimates). Estimates of L_p between this temperature range exhibited high variation (0.2 – 1.0, N = 27; Uphoff et al. 2013). In Yellow Perch, a period of low temperature is required for reproductive success (Heidinger and Kayes 1986; Ciereszko et al. 1997). Recruitment of Yellow Perch continuously failed in Lake Erie during 1973-2010 following short warm winters (Farmer et al. 2015). Subsequent lab and field studies indicated reduced egg size, energy and lipid content, and hatching success followed short winters even though there was no reduction in fecundity. Whether this reduced reproductive success was due to metabolic or maternal endocrine pathways could not be determined (Farmer et al. 2015).

Yellow Perch egg viability declined in highly developed suburban watersheds of Chesapeake Bay (C/ha above threshold level; Uphoff et al. 2005; Blazer et al. 2013). Abnormalities in ovaries and testes of adult Yellow Perch during spawning season were found most frequently in subestuaries with suburban watersheds and these abnormalities were consistent with contaminant effects (Blazer et al. 2013). Blazer et al. (2013) offered an explanation for low egg viability observed by Uphoff et al. (2005) in Severn River during 2001-2003 and persistently low L_p detected in three western shore subestuaries with highly developed suburban watersheds (C/ha > 1.59; Severn, South, and Magothy Rivers). Endocrine disrupting chemicals were more likely to cause observed egg hatching failure in well-developed tributaries than hypoxia and increased salinity (Blazer et al. 2013). It is unlikely that low L_p has always existed in well-developed Magothy, Severn, and South rivers since all supported well known recreational fisheries into the 1970s (the C/ha thresholds were met during the late 1960s-1970s). Severn River supported a state hatchery through the first half of the twentieth century and hatching rates of eggs in the hatchery were high up to 1955, when records ended (Muncy 1962). News accounts described concerns about fishery declines in these rivers during the 1980s and recreational fisheries were closed in 1989 (commercial fisheries had been banned many years earlier; Uphoff et al. 2005). A hatchery program attempted to raise Severn River Yellow Perch larvae and juveniles for mark-recapture experiments, but egg viability declined drastically by the early 2000s and Choptank River brood fish had to be substituted (Uphoff et al. 2005). Estimates of L_p from Severn River were persistently low during the 2000s. Yellow Perch egg per recruit (EPR) analyses incorporating Severn River egg hatch ratios or relative declines in L_p with C/ha indicated that recovery of Yellow Perch EPR in Severn River (and other developed tributaries) by managing the fishery alone would not be possible (Uphoff et al. 2014). Angler reports indicated that viable recreational fisheries for Yellow Perch returned to Severn River and similarly impacted western shore subestuaries (Magothy and South rivers) in the mid-to-late 1990s.

These reconstituted fisheries were likely supported by juvenile Yellow Perch that migrated from the upper Bay nursery rather than internal production (Uphoff et al. 2005). A sudden upward shift in both Yellow Perch juvenile indices and mesozooplankton relative abundance occurred in the early 1990s in the Head-of-Bay region which coincided with a downward shift in annual chlorophyll a averages at two Head-of-Bay monitoring stations (Uphoff et al. 2013). This shift in Head-of-Bay productivity was followed by reports of increased angling success in Severn, South, and Magothy rivers. Trends in L_p between the Magothy, Severn, and South rivers indicate a loss of productivity. All estimates of L_p have been below the threshold in the three western shore subestuaries with well-developed watersheds during 2001-2016 (11 of 11 estimates), while estimates from Head-of-Bay subestuaries have typically been above the threshold (4 of 7 Bush River estimates, 2 of 3 Elk River estimates, and 5 of 5 Northeast River estimates). Trends in volunteer angler catch per trip in Magothy River matched upper Bay estimates of stock abundance during 2008-2014 (P. Piavis, MD DNR, personal communication). Recreational fisheries in these three subestuaries were reopened to harvest in 2009 to allow for some recreational benefit of fish that migrated in and provided a natural “put-and-take” fishery. The term “regime shift” has been used to suggest these types of changes in productivity are causally connected and linked to other changes in an ecosystem (Steele 1996; Vert-pre et al. 2013).

Amount of organic matter present was negatively influenced by development. Estimates of C/ha and OM0 were significantly related and a non-linear power function depicted OM0 increasing towards 1.0 at a decreasing rate with C/ha. Riparian zones and floodplains that are sources of OM become disconnected from stream channels by stormwater management in suburban and urban watersheds (Craig et al. 2008; Kaushal et al. 2008; Elmore and Kaushal 2008; Brush 2009; NRC 2009), altering quantity and transport of OM (Paul and Meyer 2001; McClain et al. 2003; Stanley et al. 2012).

Years of high spring discharge favor anadromous fish recruitment in Chesapeake Bay (Hoffman et al. 2007; Martino and Houde 2010) and may represent episodes of hydrologic transport of accumulated OM from watersheds (McClain et al. 2003) that fuel zooplankton production and feeding success. Under natural conditions in York River, Virginia, riparian marshes and forests would provide OM subsidies in high discharge years (Hoffman et al. 2007), while phytoplankton would be the primary source of OM in years of lesser flow. Stable isotope signatures of York River American Shad larvae and zooplankton indicated that terrestrial OM largely supported one of its most successful year-classes. Lesser year-classes of American Shad on the York River were associated with low flows, OM based on phytoplankton, and lesser zooplankton production (Hoffman et al. 2007). The York River watershed, with large riparian marshes and forest, was largely intact relative to other Chesapeake Bay tributaries (Hoffman et al. 2007). Multiple regression models provided evidence that widespread climate factors (March precipitation as a proxy for OM transport and March air temperature) influenced year-class success of Head-of-Bay Yellow Perch (Uphoff et al. 2013).

Higher DO and pH values in urbanized Patuxent and Wicomico (ES) indicate these rivers could have a different OM source than rural Choptank, and likely reflect higher primary production by phytoplankton. The possibility exists that this could lead to lower zooplankton production and lower juvenile abundance, although these mechanisms are not clearly understood. RNA/DNA analyses did not indicate reduced larval condition in urbanized Patuxent River, however overall amount of organic matter present and subsequent feeding success of first-feeding Yellow Perch was negatively influenced by development (see Uphoff et al. 2017).

Urbanization reduces quantity and quality of OM in streams (Paul and Meyer 2001; Gücker et al. 2011; Stanley et al. 2012). Riparian zones and floodplains that are sources of OM become disconnected from stream channels by stormwater management in suburban and urban watersheds (Craig et al. 2008; Kaushal et al. 2008; Brush 2009; NRC 2009). Small headwater streams in the Gunpowder River and Patapsco River watersheds (tributaries of Chesapeake Bay) were sometimes buried in culverts and pipes, or were paved over (Elmore and Kaushal 2008). Decay of leaves occurred much faster in urban streams, apparently due to greater fragmentation from higher stormflow rather than biological activity (Paul and Meyer 2001). Altered flowpaths associated with urbanization affect timing and delivery of OM to streams (McClain et al. 2003). Organic matter was transported further and retained less in urban streams (Paul and Meyer 2001). Uphoff et al. (2011) and our current analysis found that the percentage of Maryland's Chesapeake Bay subestuary watersheds in wetlands declined as C/ha increased, so this source of OM diminishes with development.

Management for organic carbon is nearly non-existent despite its role as a great modifier of the influence and consequence of other chemicals and processes in aquatic systems (Stanley et al. 2012). It is unmentioned in the Chesapeake Bay region as reductions in nutrients (N and P) and sediment are pursued for ecological restoration (http://www.epa.gov/reg3wapd/pdf/pdf_chesbay/BayTMDLFactSheet8_6.pdf). However, most watershed management and restoration practices have the potential to increase OM delivery and processing, although it is unclear how ecologically meaningful these changes may be. Stanley et al. (2012) recommended beginning with riparian protection or re-establishment and expand outward as opportunities permit. Wetland management represents an expansion of effort beyond the riparian zone (Stanley et al. 2012).

Agriculture also has the potential to alter OM dynamics within a watershed and has been associated with increased, decreased, and undetectable changes in OM that may reflect diversity of farming practices (Stanley et al. 2012). In our study, agricultural watersheds (all eastern shore) had most of the lower OM0 scores (indicating more detritus), while OM0 levels were higher and distributed similarly among watersheds that were predominately in development (all western shore) or forest (eastern and western shore).

Annual L_p (proportion of tows with Yellow Perch larvae during a standard time period and where larvae would be expected) provided an economically collected measure of the product of egg production and egg through early postlarval survival. We used L_p as an index to detect "normal" and "abnormal" larval dynamics. We considered L_p estimates from subestuaries that were persistently lower than those measured in other subestuaries indicative of abnormally low survival. Remaining levels were considered normal. Assuming catchability does not change greatly from year to year, egg production and egg through early postlarval survival would need to be high to produce strong L_p, but only one factor needed to be low to result in lower L_p.

High estimates of L_p that were equal to or approaching 1.0 have been routinely encountered in the past, and it is likely that counts would be needed to measure relative abundance if greater resolution was desired. Mangel and Smith (1990) indicated that presence/absence sampling of eggs would be more useful for indicating the status of depleted stocks and count-based indices would be more accurate for recovered stocks. Larval indices based on counts have been used as a measure of year-class strength of fishes generally (Sammons and Bettoli 1998) and specifically for Yellow Perch (Anderson et al. 1998). Tighter budgets necessitate development of low cost indicators of larval survival and relative abundance in order to pursue ecosystem-based fisheries management. Characterizations of larval survival and relative abundance normally are derived from counts requiring labor-intensive sorting and processing. Estimates of L_p were largely derived in the field and only gut contents and RNA/DNA in previous years (see Uphoff et al. 2017) required laboratory analysis. These latter two analyses represented separate studies rather than a requirement for estimating L_p (Uphoff et al. 2017).

We have relied on correlation and regression analyses to judge the effects of watershed development on Yellow Perch larval dynamics (see Uphoff et al. 2017). Ideally, manipulative experiments and formal adaptive management should be employed (Hilborn 2016). In large-scale aquatic ecosystems these opportunities are limited and are not a possibility for us. Correlations are often not causal, but may be all the evidence available. Correlative evidence is strongest when (1) correlation is high, (2) it is found consistently across multiple situations, (3) there are not competing explanations, and (4) the correlation is consistent with mechanistic explanations that can be supported by experimental evidence (Hilborn 2016).

Interpretation of the influence of salinity class or major land cover on L_p needs to consider that our survey design was limited to existing patterns of development. All estimates of L_p at or below target levels of development (forested and agricultural watersheds) or at the threshold or beyond high levels of development (except for one sample) were from brackish subestuaries; estimates of L_p for development between these levels were from tidal-fresh subestuaries with forested watersheds. Larval dynamics below the target level of development primarily reflected eastern shore agricultural watersheds. Two types of land use would be needed to balance analyses: (1) agricultural, tidal-fresh watersheds with below target development and (2) forested, brackish watersheds with development between the target and threshold. MD DOP land use estimates from 2010 (most recent year available) indicate that the Wicomico River (ES) would fall into the latter category. Estimates of three land use categories (agriculture, forest, and urban) were almost evenly divided at that time, with forest being marginally dominant (Table 2-1), however it is unlikely that this is still the case. Salisbury, MD, a city, is located on the Wicomico (ES), and it is expected that increased development has occurred in this area over past seven years. We do not believe that any other of these combinations exist where Yellow Perch spawning occurs in Maryland's portion of Chesapeake Bay. The MD DOP forest cover estimates have a minimum mapping unit of 10 acres that mixes forest cover in residential areas (trees over lawns) with true forest cover, clouding interpretation of forest influence (R. Feldt, MD DNR Forest Service, personal communication).

Development was an important influence on Yellow Perch egg and larval dynamics and negative changes generally conformed to impervious surface reference points developed from distributions of dissolved oxygen, and juvenile and adult target fish in mesohaline subestuaries (Uphoff et al. 2011). Hilborn and Stokes (2010) advocated setting reference points related to harvest for fisheries (stressor) based on historical stock performance (outcome) because they were based on experience, easily understood, and not based on modeling. We believe applying IS or C/ha watershed development reference points (stressor) based on L_p (outcome) conforms to the approach advocated by Hilborn and Stokes (2010).

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Table 2-1. Estimates of proportions of ichthyoplankton net tows with Yellow Perch larvae (L_p) during 1965-2017 and data used for regression with counts of structures per hectare (C/ha), percent agriculture, percent forest, and percent wetland. Salinity class 0 = tidal-fresh (≤ 2.0‰) and 1 = brackish (> 2.0‰). Land use percentages and overall primary land use were determined from Maryland Department of Planning estimates for 1973, 1994, 1997, 2002, or 2010 that were closest to a sampling year.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Bush (w/ APG)	2006	2002	1.17	21	36.3	5.5	37	Urban	0	0.79
Bush (w/ APG)	2007	2010	1.19	14.9	32.1	5.5	46.4	Urban	0	0.92
Bush (w/ APG)	2008	2010	1.2	14.9	32.1	5.5	46.4	Urban	0	0.55
Bush (w/ APG)	2009	2010	1.21	14.9	32.1	5.5	46.4	Urban	0	0.86
Bush (w/ APG)	2011	2010	1.23	14.9	32.1	5.5	46.4	Urban	0	0.96
Bush (w/ APG)	2012	2010	1.24	14.9	32.1	5.5	46.4	Urban	0	0.28
Bush (w/ APG)	2013	2010	1.25	14.9	32.1	5.5	46.4	Urban	0	0.15
Choptank	1986	1994	0.07	58.5	32.4	1.3	7.7	Agriculture	1	0.53
Choptank	1987	1994	0.08	58.5	32.4	1.3	7.7	Agriculture	1	0.73
Choptank	1988	1994	0.08	58.5	32.4	1.3	7.7	Agriculture	1	0.8
Choptank	1989	1994	0.08	58.5	32.4	1.3	7.7	Agriculture	1	0.71
Choptank	1990	1994	0.08	58.5	32.4	1.3	7.7	Agriculture	1	0.66
Choptank	1998	1997	0.1	57.9	31.3	1.2	9.5	Agriculture	1	0.6
Choptank	1999	1997	0.1	57.9	31.3	1.2	9.5	Agriculture	1	0.76
Choptank	2000	2002	0.1	58.2	30.8	1.1	9.9	Agriculture	1	0.25
Choptank	2001	2002	0.1	58.2	30.8	1.1	9.9	Agriculture	1	0.21
Choptank	2002	2002	0.11	58.2	30.8	1.1	9.9	Agriculture	1	0.38
Choptank	2003	2002	0.11	58.2	30.8	1.1	9.9	Agriculture	1	0.52
Choptank	2004	2002	0.12	58.2	30.8	1.1	9.9	Agriculture	1	0.41
Choptank	2013	2010	0.13	55	27.8	1.4	15.8	Agriculture	1	0.47
Choptank	2014	2010	0.13	55	27.8	1.4	15.8	Agriculture	1	0.68
Choptank	2015	2010	0.13	55	27.8	1.4	15.8	Agriculture	1	0.82
Choptank	2016	2010	0.13	55	27.8	1.4	15.8	Agriculture	1	0.9
Choptank	2017	2010	0.13	55	27.8	1.4	15.8	Agriculture	1	0.4
Corsica	2006	2002	0.21	64.3	27.4	0.4	7.9	Agriculture	1	0.47
Corsica	2007	2010	0.22	60.4	25.5	0.1	13.2	Agriculture	1	0.83
Elk	2010	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.75
Elk	2011	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.79

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Elk	2012	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.55
Langford	2007	2010	0.07	20.4	70.2	1.5	8	Agriculture	1	0.83
Magothy	2009	2010	2.73	1.2	21	0	76.8	Urban	1	0.1
Magothy	2016	2010	2.78	1.2	21	0	76.8	Urban	1	0.1
Mattawoman	1990	1994	0.45	13.8	62.6	0.9	22.5	Forest	0	0.81
Mattawoman	2008	2010	0.87	9.3	53.9	2.8	34.2	Forest	0	0.66
Mattawoman	2009	2010	0.88	9.3	53.9	2.8	34.2	Forest	0	0.92
Mattawoman	2010	2010	0.9	9.3	53.9	2.8	34.2	Forest	0	0.82
Mattawoman	2011	2010	0.91	9.3	53.9	2.8	34.2	Forest	0	0.99
Mattawoman	2012	2010	0.9	9.3	53.9	2.8	34.2	Forest	0	0.2
Mattawoman	2013	2010	0.91	9.3	53.9	2.8	34.2	Forest	0	0.47
Mattawoman	2014	2010	0.93	9.3	53.9	2.8	34.2	Forest	0	0.78
Mattawoman	2015	2010	0.93	9.3	53.9	2.8	34.2	Forest	0	1
Mattawoman	2016	2010	0.93	9.3	53.9	2.8	34.2	Forest	0	0.82
Middle	2012	2010	3.33	3.4	23.3	2.1	71	Urban	0	0
Nanjemoy	2009	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.83
Nanjemoy	2010	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.96
Nanjemoy	2011	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.99
Nanjemoy	2012	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.03
Nanjemoy	2013	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.46
Nanjemoy	2014	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.82
Nanticoke	1965	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	0.5
Nanticoke	1967	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	0.43
Nanticoke	1968	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	1
Nanticoke	1970	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	0.81
Nanticoke	1971	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	0.33
Nanticoke	2004	2002	0.11	46.3	40.7	7.4	5.5	Agriculture	1	0.49
Nanticoke	2005	2002	0.11	46.3	40.7	7.4	5.5	Agriculture	1	0.67
Nanticoke	2006	2002	0.11	46.3	40.7	7.4	5.5	Agriculture	1	0.35
Nanticoke	2007	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.55
Nanticoke	2008	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.19
Nanticoke	2009	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.41

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Nanticoke	2011	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.55
Nanticoke	2012	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.04
Nanticoke	2013	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.43
Nanticoke	2014	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.35
Nanticoke	2015	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.64
Nanticoke	2016	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.67
Nanticoke	2017	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.22
Northeast	2010	2010	0.46	31.1	38.6	0.1	28.9	Forest	0	0.68
Northeast	2011	2010	0.46	31.1	38.6	0.1	28.9	Forest	0	1
Northeast	2012	2010	0.47	31.1	38.6	0.1	28.9	Forest	0	0.77
Northeast	2013	2010	0.47	31.1	38.6	0.1	28.9	Forest	0	0.72
Northeast	2014	2010	0.48	31.1	38.6	0.1	28.9	Forest	0	0.77
Patuxent	2015	2010	1.22	20.5	35.1	1	41.7	Urban	1	0.72
Patuxent	2016	2010	1.22	20.5	35.1	1	41.7	Urban	1	0.82
Piscataway	2008	2010	1.41	10	40.4	0.2	47	Urban	0	0.47
Piscataway	2009	2010	1.43	10	40.4	0.2	47	Urban	0	0.39
Piscataway	2010	2010	1.45	10	40.4	0.2	47	Urban	0	0.54
Piscataway	2011	2010	1.46	10	40.4	0.2	47	Urban	0	0.65
Piscataway	2012	2010	1.47	10	40.4	0.2	47	Urban	0	0.16
Piscataway	2013	2010	1.49	10	40.4	0.2	47	Urban	0	0.5
Severn	2002	2002	2.02	8.6	35.2	0.2	55.8	Urban	1	0.16
Severn	2004	2002	2.09	8.6	35.2	0.2	55.8	Urban	1	0.35
Severn	2005	2002	2.15	8.6	35.2	0.2	55.8	Urban	1	0.4
Severn	2006	2002	2.18	8.6	35.2	0.2	55.8	Urban	1	0.27
Severn	2007	2010	2.21	5	28	0.2	65.1	Urban	1	0.3
Severn	2008	2010	2.24	5	28	0.2	65.1	Urban	1	0.08
Severn	2009	2010	2.25	5	28	0.2	65.1	Urban	1	0.15
Severn	2010	2010	2.26	5	28	0.2	65.1	Urban	1	0.03
South	2008	2010	1.61	10.2	39.2	0.5	48.8	Urban	1	0.14
Wicomico (ES)	2017	2010	0.67	30.1	36.8	2.3	29.9	Forest	1	0.53

Table 2-2. Summary of results of regressions of proportions of tows with Yellow Perch larvae (L_p) and (A) counts of structures per hectare (C/ha), (B) percent agriculture, and (C) percent forest. Separate regressions by salinity (tidal-fresh ≤ 2.0 ‰ and brackish > 2.0 ‰) and a multiple regression using salinity as a class variable (tidal-fresh = 0 and brackish = 1) are presented.

ANOVA	(A) Brackish					
Source	df	SS	MS	F	P	
Model	1	1.07793	1.07793	19.72	<.0001	
Error	56	3.06067	0.05465			
Total	57	4.1386				
r^2	0.2605					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.5861	0.03666	15.99	<.0001	0.51265	0.65955
C / ha	-0.16061	0.03617	-4.44	<.0001	-0.23306	-0.08816

ANOVA	(A) Tidal-Fresh					
Source	df	SS	MS	F	P	
Model	1	0.77363	0.77363	15.51	0.0005	
Error	30	1.49597	0.04987			
Total	31	2.2696				
r^2	0.3409					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94825	0.08595	11.03	<.0001	0.7727	1.12379
C / ha	-0.28944	0.07348	-3.94	0.0005	-0.43951	-0.13937

ANOVA	(A) Multiple Regression					
Source	df	SS	MS	F	P	
Model	2	2.19272	1.09636	20.37	<.0001	
Error	87	4.68215	0.05382			
Total	89	6.87487				
r^2	0.3189					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.83862	0.05311	15.79	<.0001	0.73306	0.94418
C / ha	-0.18393	0.03248	-5.66	<.0001	-0.24849	-0.11938
Salinity	-0.23959	0.05346	-4.48	<.0001	-0.34584	-0.13334

Table 2-2 cont.

ANOVA			(B) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.37961	0.37961	5.66	0.0208	
Error	56	3.75899	0.06712			
Total	57	4.1386				
r^2	0.0917					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.3518	0.06992	5.03	<.0001	0.21174	0.49187
% Ag	0.00391	0.00164	2.38	0.0208	0.00061663	0.00721

ANOVA			(B) Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.21286	0.21286	3.1	0.0883	
Error	30	2.05674	0.06856			
Total	31	2.2696				
r^2	0.0938					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.49679	0.09725	5.11	<.0001	0.29818	0.69541
% Ag	0.00944	0.00536	1.76	0.0883	-0.0015	0.02038

ANOVA			(B) Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	0.99254	0.49627	7.34	0.0011	
Error	87	5.88233	0.06761			
Total	89	6.87487				
r^2	0.1444					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57731	0.05241	11.02	<.0001	0.47314	0.68147
% Ag	0.0044	0.00158	2.79	0.0065	0.00126	0.00753
Salinity	-0.24353	0.06628	-3.67	0.0004	-0.37527	-0.11179

Table 2-2 cont.

ANOVA			(C) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.39378	0.39378	5.89	0.0185	
Error	56	3.74482	0.06687			
Total	57	4.1386				
r^2	0.0951					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.24578	0.10898	2.26	0.028	0.02747	0.46409
% Forest	0.00654	0.0027	2.43	0.0185	0.00114	0.01195

ANOVA			(C) Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.22878	0.22878	3.36	0.0766	
Error	30	2.04082	0.06803			
Total	31	2.2696				
r^2	0.1008					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.26392	0.21418	1.23	0.2274	-0.1735	0.70134
% Forest	0.00908	0.00495	1.83	0.0766	-0.00103	0.0192

ANOVA			(C) Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	1.07551	0.53775	8.07	0.0006	
Error	87	5.79937	0.06666			
Total	89	6.87487				
r^2	0.1564					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.34627	0.10963	3.16	0.0022	0.12838	0.56416
% Forest	0.00713	0.00236	3.02	0.0033	0.00244	0.01182
Salinity	-0.12308	0.05757	-2.14	0.0353	-0.23751	-0.00865

Table 2-3. Summary of results of regressions of proportions of tows with Yellow Perch larvae (L_p) and (A) small systems with counts of structures per hectare (C/ha), or (B) large systems counts of structures per hectare (C/ha). Separate regressions by salinity (tidal-fresh ≤ 2.0 ‰ and brackish > 2.0 ‰) are presented for small systems only as all large systems are brackish.

ANOVA			(A) Small Brackish			
Source	df	SS	MS	F	P	
Model	1	1.24697	1.24697	24.31	0.0001	
Error	18	0.92313	0.05129			
Total	19	2.1701				
r^2	0.5746					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.71161	0.07863	9.05	<.0001	0.54642	0.87681
C / ha	-0.23209	0.04707	-4.93	0.0001	-0.33098	-0.13321

ANOVA			(A) Small Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.77363	0.77363	15.51	0.0005	
Error	30	1.49597	0.04987			
Total	31	2.2696				
r^2	0.3409					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94825	0.08595	11.03	<.0001	0.7727	1.12379
C / ha	-0.28944	0.07348	-3.94	0.0005	-0.43951	-0.13937

ANOVA			(B) Large Brackish			
Source	df	SS	MS	F	P	
Model	1	0.05018	0.05018	1.31	0.2666	
Error	18	0.68732	0.03818			
Total	19	0.7375				
r^2	0.068					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.55982	0.0534	10.48	<.0001	0.44764	0.672
C / ha	0.14389	0.12552	1.15	0.2666	-0.11981	0.40759

Table 2-4. Estimates of proportions of ichthyoplankton net tows with Yellow Perch larvae (L_p) and their standard deviations (SD) within up-river, mid-river, and down-river sections of large systems sampled in 2015-2017.

Choptank 2015	Stations	Presence	N	Lp	SD
Down-river	1-5	4	6	0.6667	0.1925
Mid-river	6-11	13	19	0.6842	0.1066
Up-river	12-17 and 18-20	29	31	0.9355	0.0441

Patuxent 2015	Stations	Presence	N	Lp	SD
Down-river	1-2	11	14	0.7857	0.1097
Mid-river	3-6	23	27	0.8519	0.0684
Up-river	7-12	13	24	0.5417	0.1017

Choptank 2016	Stations	Presence	N	Lp	SD
Down-river	1-5	2	2	1	0.0000
Mid-river	6-11	15	18	0.8333	0.0878
Up-river	12-17 and 18-20	28	30	0.9333	0.0455

Patuxent 2016	Stations	Presence	N	Lp	SD
Down-river	1-2	5	10	0.5	0.1581
Mid-river	3-6	20	25	0.8	0.0800
Up-river	7-12	25	26	0.9615	0.0377

Choptank 2017	Stations	Presence	N	Lp	SD
Down-river	1-5	4	10	0.4	0.1549
Mid-river	6-11	12	38	0.3158	0.0754
Up-river	12-17 and 18-20	24	52	0.4615	0.0691

Wicomico 2017	Stations	Presence	N	Lp	SD
Down-river	1-4	10	24	0.4167	0.1006
Mid-river	5-8	16	25	0.64	0.0960
Up-river	9-12	11	21	0.5238	0.1090

Table 2-5. Summary of annual water quality parameter statistics for large systems sampled in 2015-2017. Mean pH was calculated from H^+ concentrations and back-converted for reporting here.

System/Year		*Temp C*	*DO (mg/L)*	*Cond (umhols)*	*pH*
Choptank 15	Mean	14.87	8.05	585.5	7.41
	Standard Error	0.30	0.12	111.62	
	Median	14.41	8.33	193.5	7.43
	Mode	12.5	8.7	172	7.6
	Kurtosis	-0.99	-0.04	5.13	0.09
	Skewness	0.51	-0.76	2.42	0.66
	Minimum	11.9	5.77	137	7.1
	Maximum	19	9.5	3780	8.07
	Count	56	56	56	56
Patuxent 15	Mean	15.58	8.18	682.08	7.49
	Standard Error	0.19	0.12	82.01	
	Median	15.39	8.2	420	7.5
	Mode	13.50	8.2	416	7.5
	Kurtosis	-0.61	-0.67	7.6	4.49
	Skewness	0.51	0.32	2.84	1.02
	Minimum	13.5	6.48	317	7.22
	Maximum	18.66	10.44	3341	8.12
	Count	65	65	65	65
Choptank 16	Mean	13.25	8.77	829.24	7.20
	Standard Error	0.12	0.09	149.73	
	Median	13.42	8.73	295.5	7.21
	Mode	13.26	8.21	238	7.29
	Kurtosis	0.33	1.79	2.51	1.11
	Skewness	-1.09	0.73	1.84	0.68
	Minimum	10.96	7.67	148	7.04
	Maximum	14.53	10.87	4389	7.6
	Count	50	50	50	50
Patuxent 16	Mean	13.01	9.60	1137.23	7.56
	Standard Error	0.14	0.08	144.1	
	Median	12.75	9.34	695	7.56
	Mode	13.27	9.32	381	7.55
	Kurtosis	-0.78	-0.79	5.32	-0.29
	Skewness	0.61	0.62	2.27	0.14
	Minimum	11.33	8.82	378	7.41
	Maximum	15.14	11	5623	7.75
	Count	61	61	61	61

Table 2.5 cont.

System/Year		*Temp C*	*DO (mg/L)*	*Cond (umhols)*	*pH*
	Mean	13.55	8.60	840.29	7.12
	Standard Error	0.41	0.16	101.73	
	Median	13.9	8.51	279.5	7.15
	Mode	8.14	8.26	132	7.15
Choptank 17	Kurtosis	-1.12	-0.45	0.82	-0.13
	Skewness	-0.12	0.08	1.45	0.10
	Minimum	6.62	5.45	102	6.70
	Maximum	20.24	12.31	3688	7.68
	Count	100	100	100	100
	Mean	13.56	11.01	678.61	7.37
	Standard Error	0.37	0.15	111.48	
	Median	14.19	11.20	255	7.46
	Mode	16.94	10.46	182	7.53
Wicomico ES 17	Kurtosis	-1.18	-0.67	4.56	0.27
	Skewness	-0.50	-0.38	2.32	0.25
	Minimum	8.28	8.05	131	6.83
	Maximum	17.52	13.17	3846	8.2
	Count	70	70	70	70

Table 2-6. Summary of Akaike's Information Criteria from regressions of proportions of tows with Yellow perch larvae (Lp) and (A) counts of structures per hectare (C/ha), (B) percent agriculture, and (C) percent forest for each salinity category and a multiple regression using salinity as a class variable.

Model (A)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.05382	90	4	2.92211	8	40	85	11.4	2.04	1.54
Fresh	0.04987	32	3	2.99834	6	24	28	9.9		
Brackish	0.05465	58	3	2.90681	6	24	54	9.4		

Model (B)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.06712	90	4	2.70127	8	40	85	11.2	2.03	1.63
Fresh	0.06856	32	3	2.68005	6	24	28	9.5		
Brackish	0.06761	58	3	2.69400	6	24	54	9.1		

Model (C)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.06687	90	4	2.70500	8	40	85	11.2	2.02	1.63
Fresh	0.06803	32	3	2.68781	6	24	28	9.5		
Brackish	0.06666	58	3	2.70815	6	24	54	9.2		

Figure 2-1. Areas sampled for Yellow Perch larval presence-absence studies, 2006-2017. Areas sampled in 2017 are highlighted in green. Nanticoke River watershed delineation was unavailable for Delaware and Northeast and was unavailable for Pennsylvania.

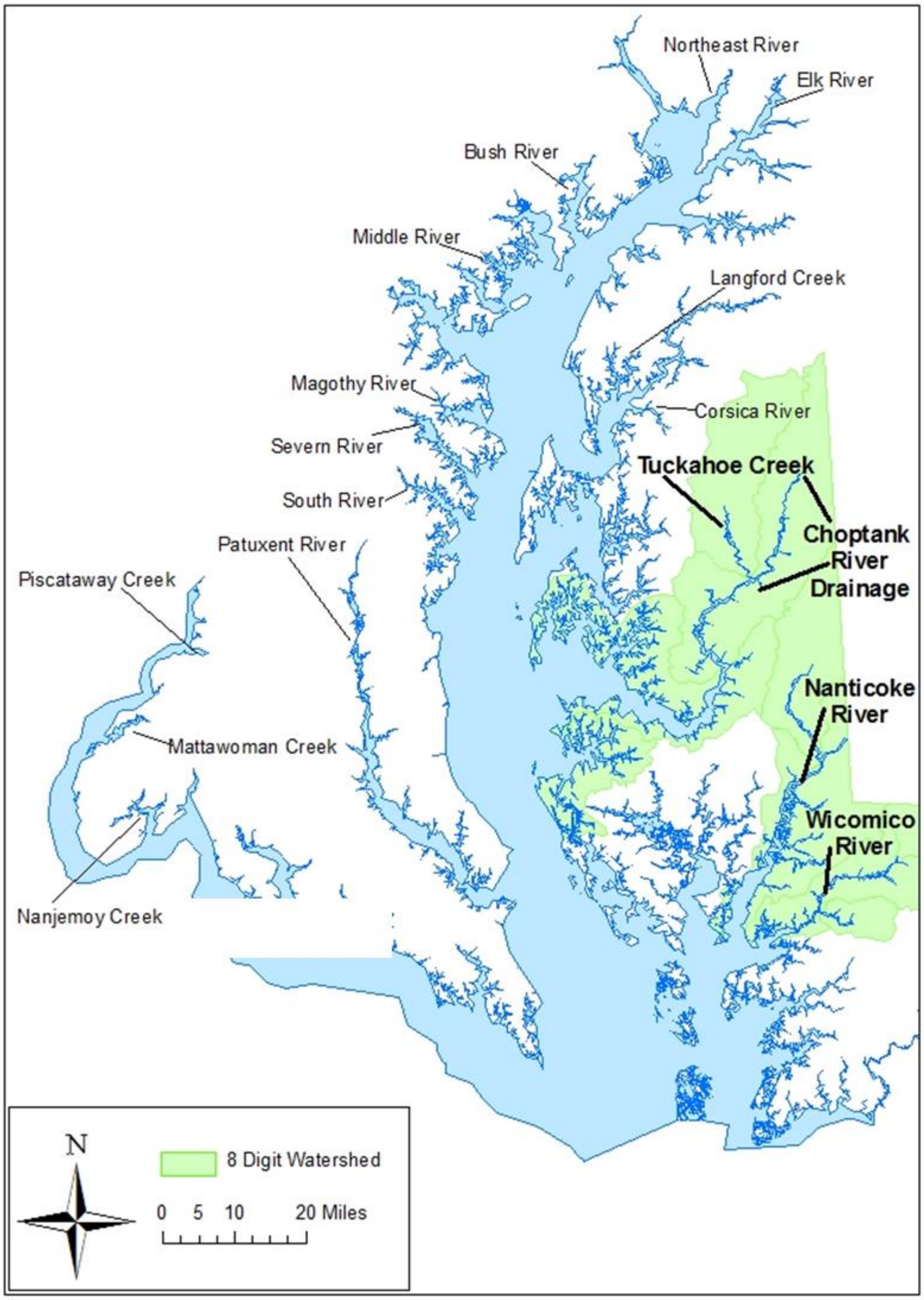


Figure 2-2. Proportion of tows with larval Yellow Perch (*Lp*) and its 95% confidence interval in systems studied during 2017. Mean *Lp* of brackish tributaries indicated by green diamond.

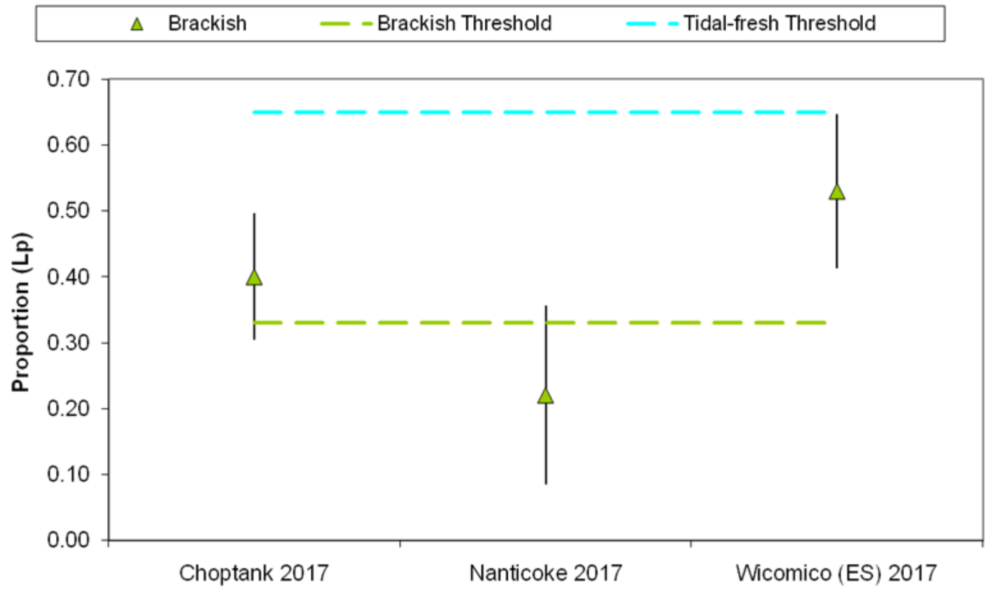


Figure 2-3. Proportion of tows with Yellow Perch larvae (*Lp*) for brackish subestuaries, during 1965-2017. Dotted line provides reference for persistent poor *Lp* exhibited in developed brackish subestuaries. Dominant Department of Planning land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

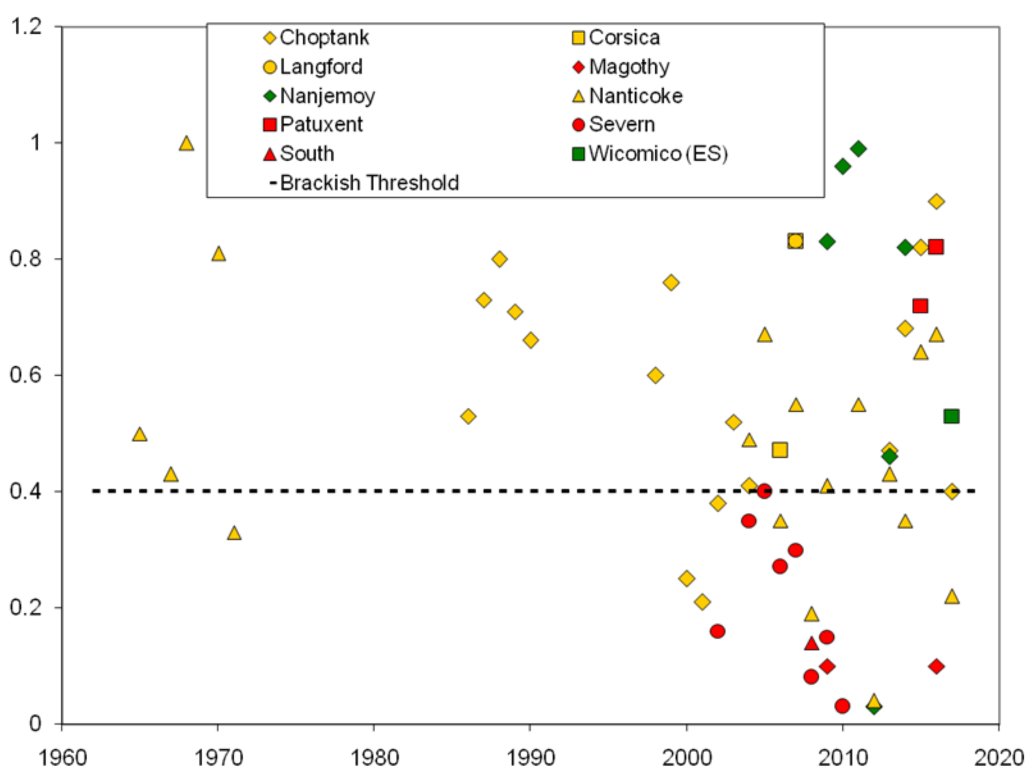


Figure 2-4. Proportion of tows with Yellow Perch larvae (*Lp*) for tidal-fresh subestuaries, during 1990-2017. Dotted line provides reference for consistent poor *Lp* exhibited in a more developed tidal-fresh subestuary (Piscataway Creek). Dominant Department of Planning land use is indicated by symbol color (green = forest and red = urban).

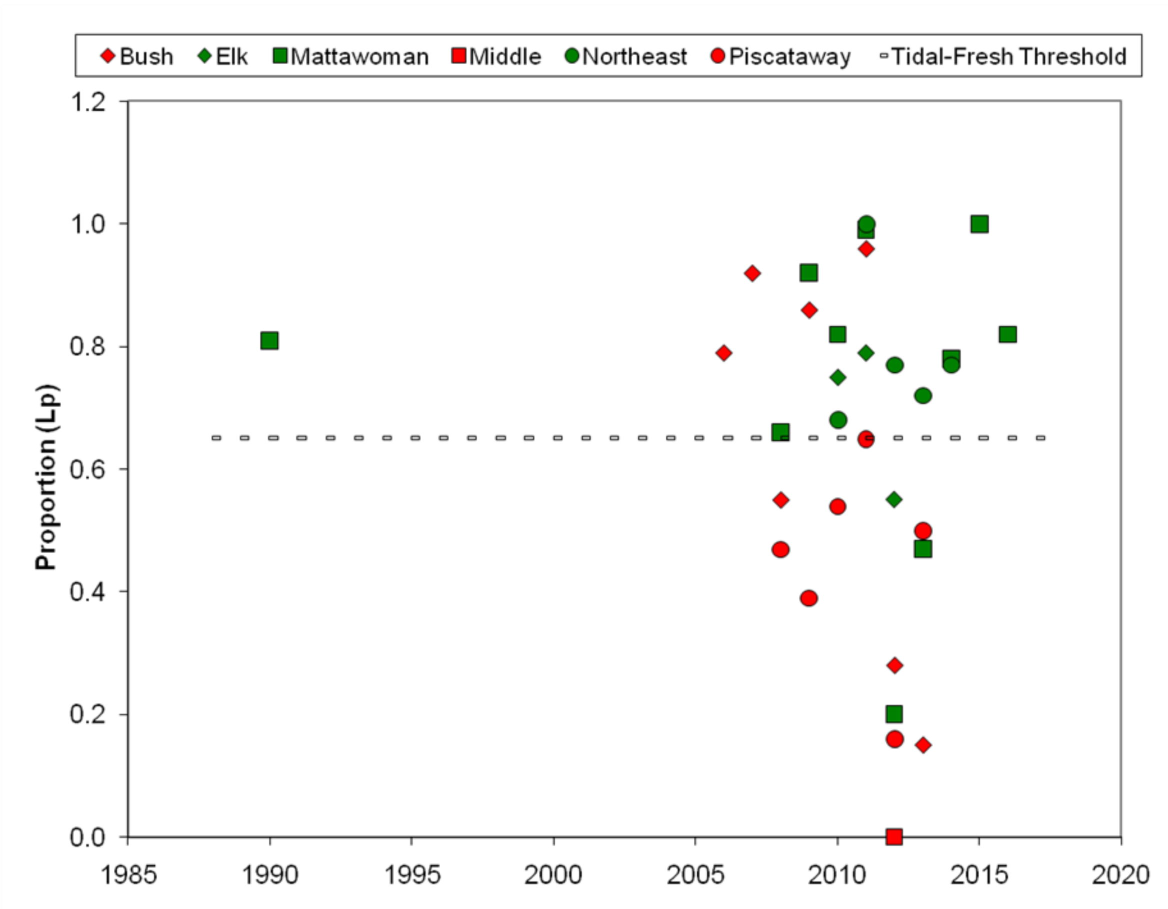


Figure 2-5. Relationship of proportion of plankton tows with Yellow Perch larvae and (A) development (structures per hectare or C/ha), (B) percent agriculture, and (C) percent forest, indicated by multiple regression of fresh and brackish subestuaries combined (prediction = MR) and separate linear regressions for both (prediction = LR).

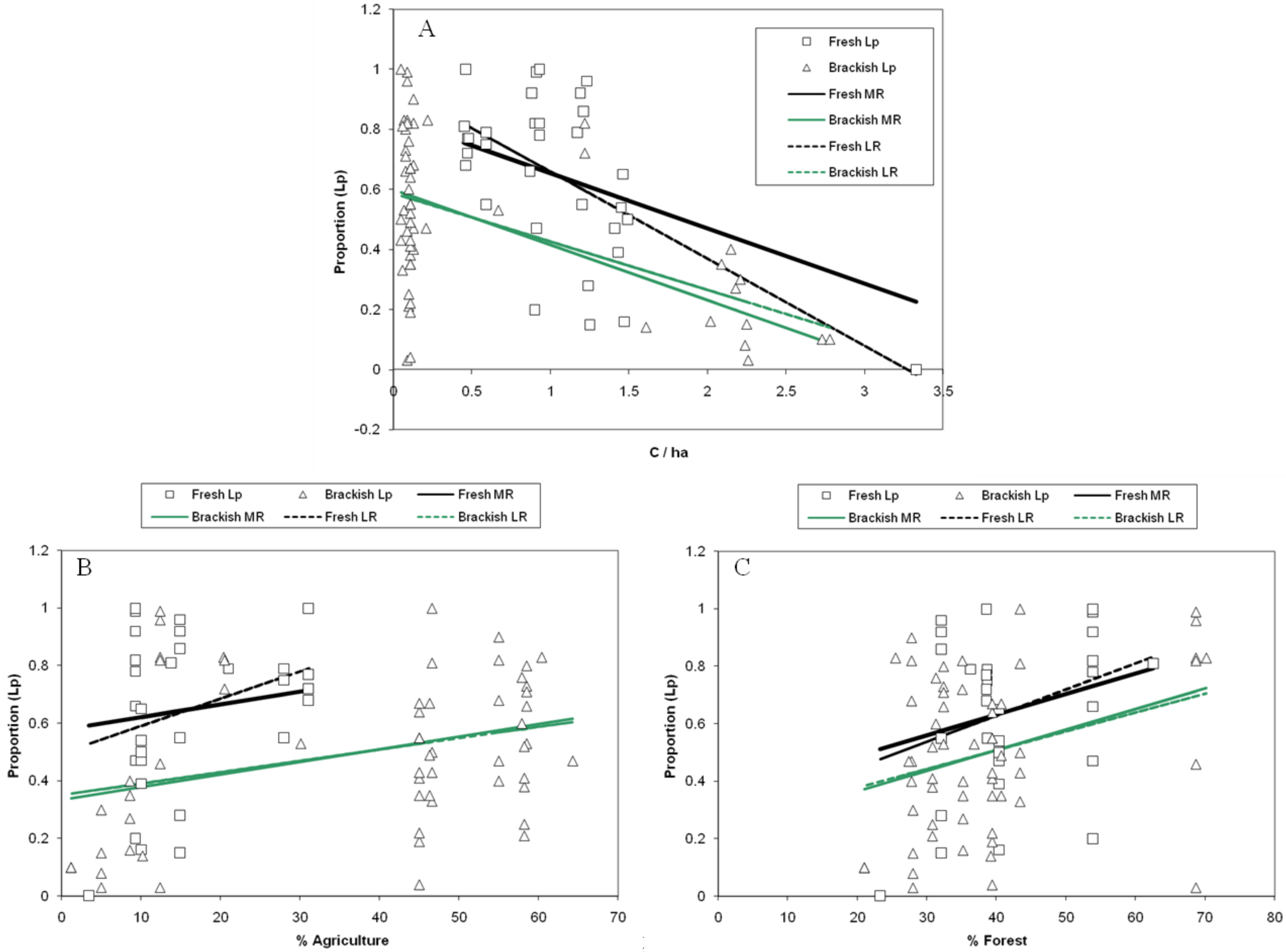


Figure 2-6. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations and salinity class indicated by symbols. Diamonds and a "1" behind land use in the key indicate brackish subestuaries, while squares and a "0" indicate tidal-fresh.

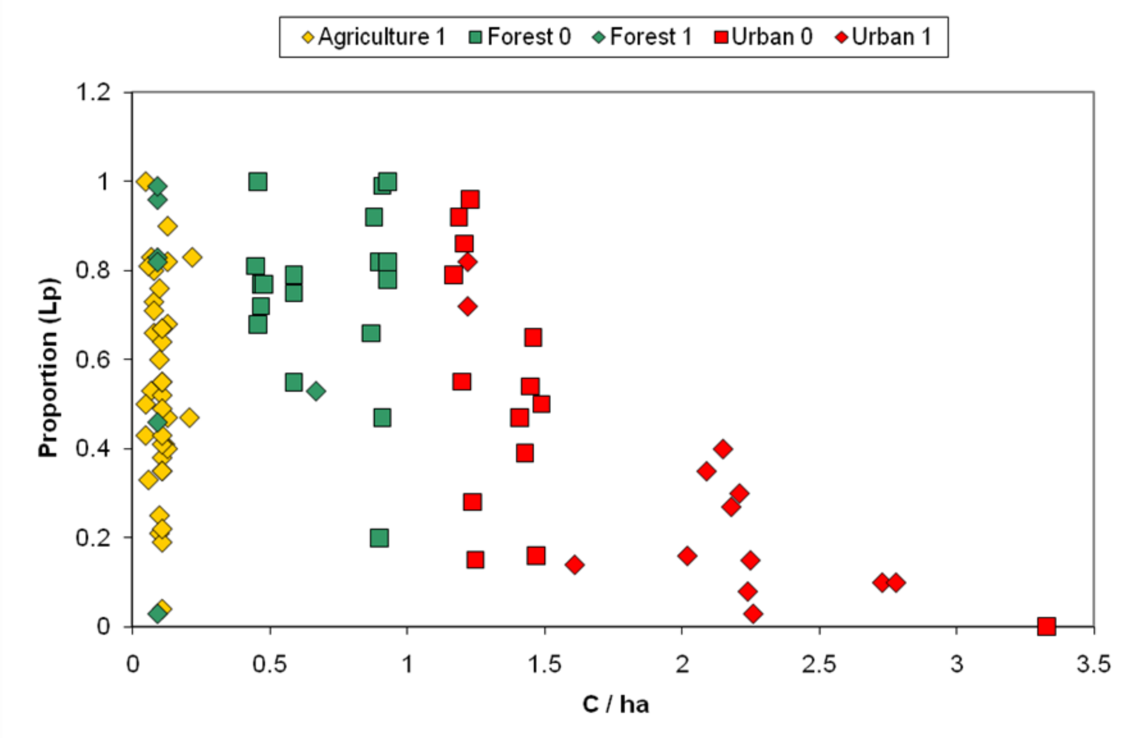


Figure 2-7. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations for large systems.

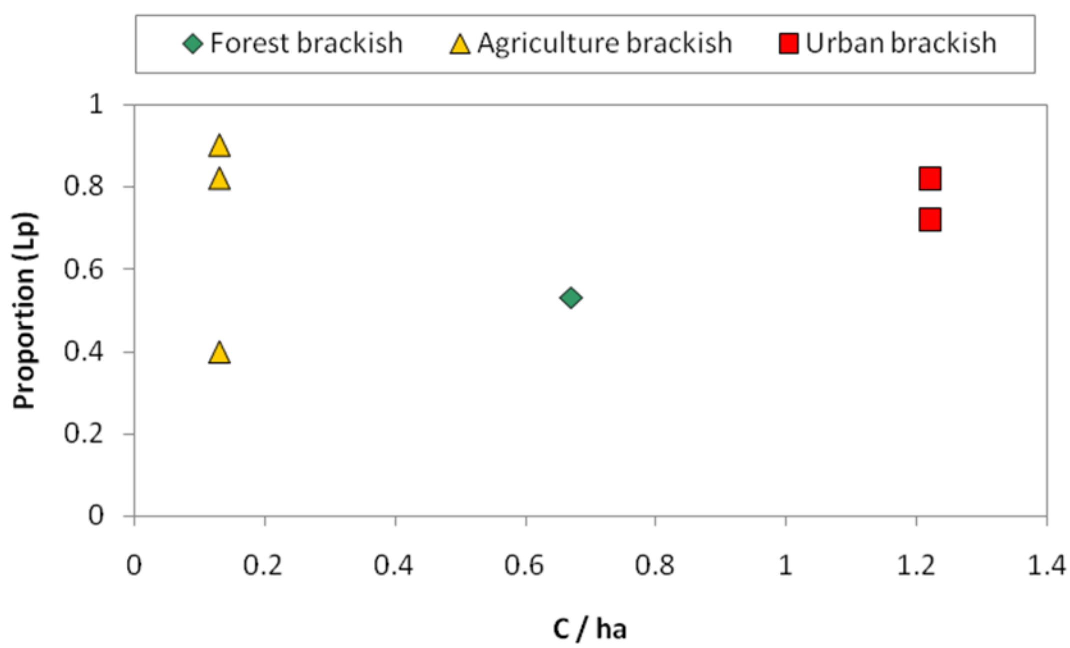


Figure 2-8. Water quality parameters sampled in Choptank and Patuxent rivers during 2015.

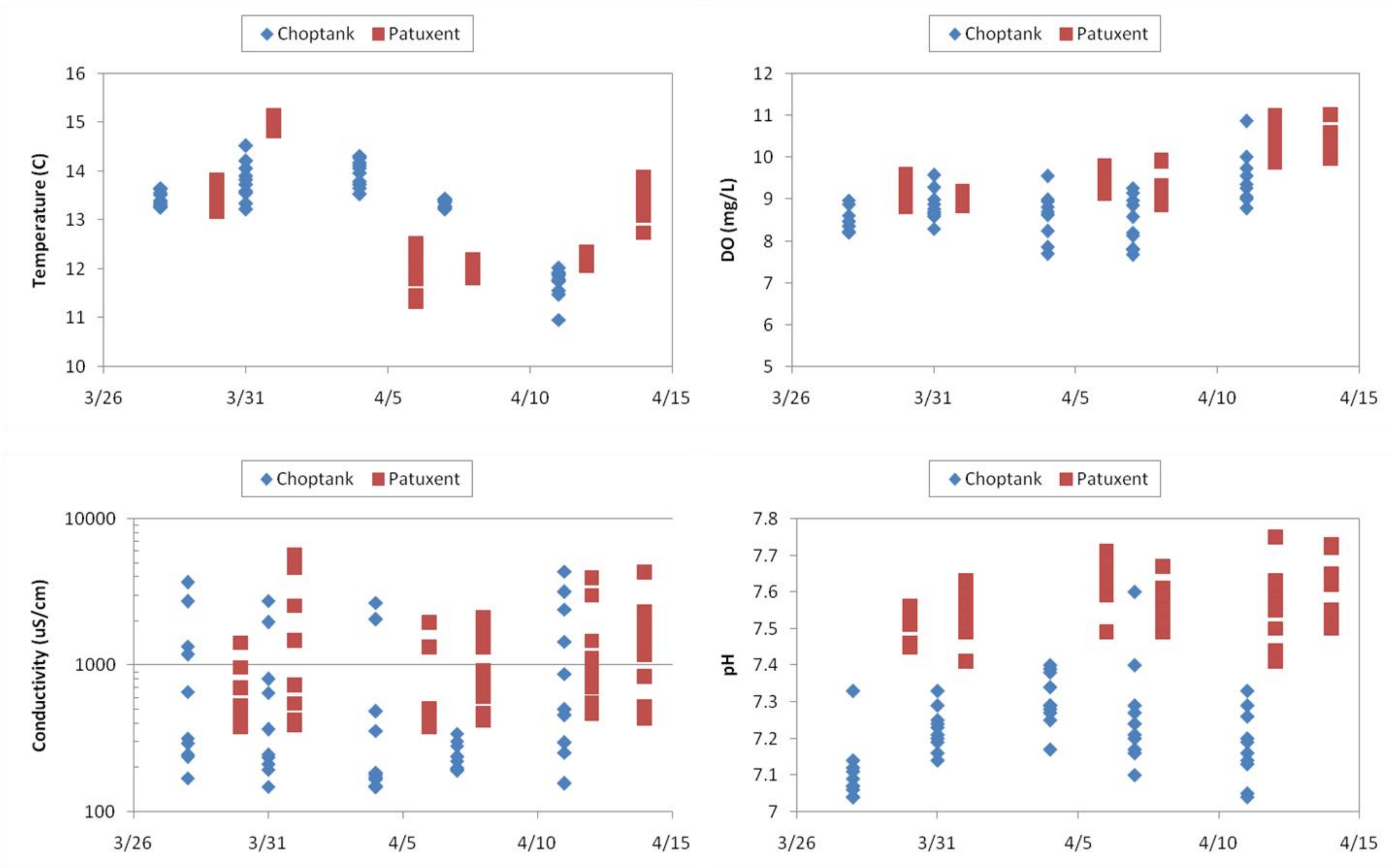


Figure 2-9. Water quality parameters sampled in Choptank and Patuxent rivers during 2016.

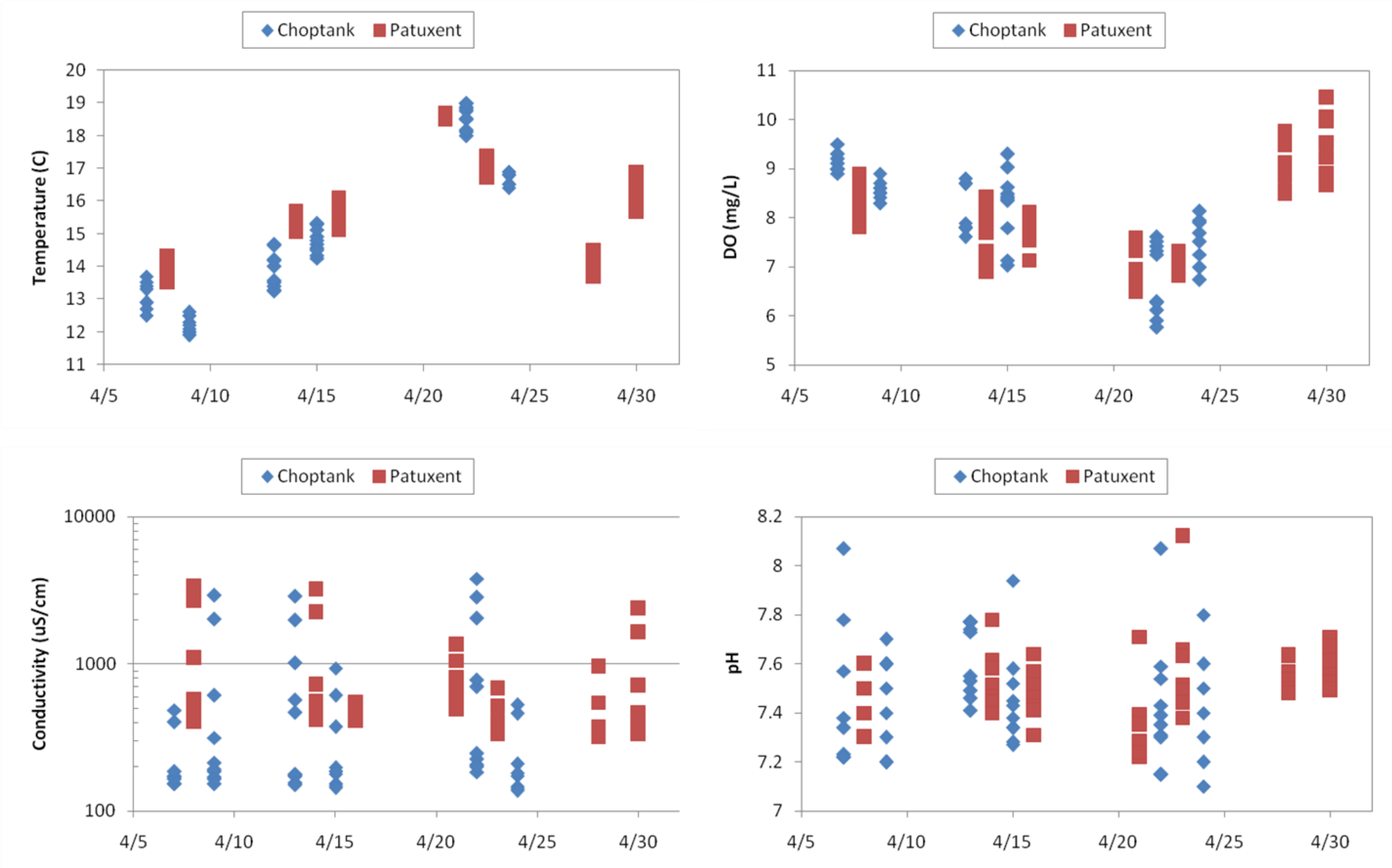


Figure 2-10. Water quality parameters sampled in Choptank and Wicomico (ES) rivers during 2017.

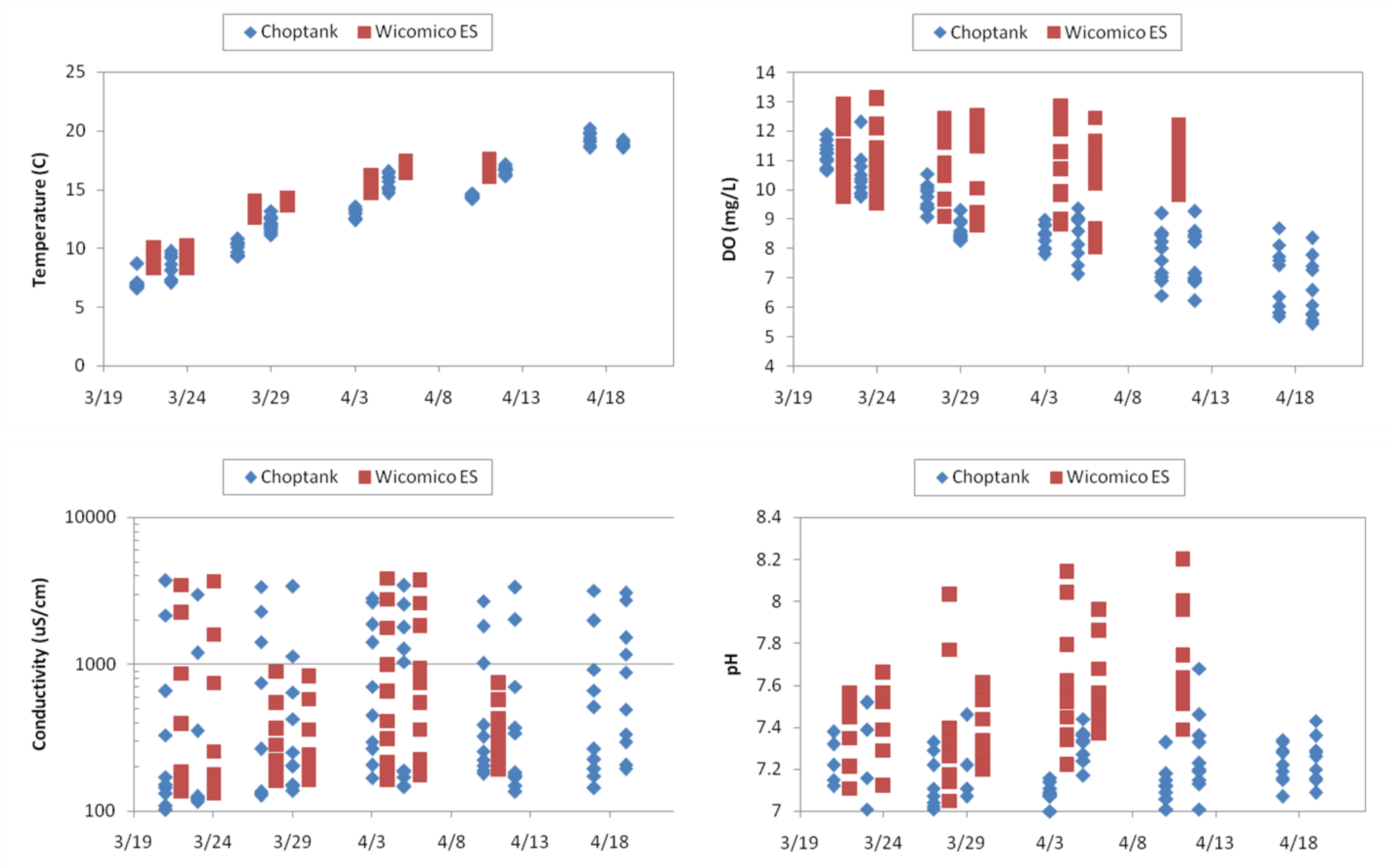


Figure 2-11. Relationship of proportion of plankton tows without detritus (OM0) and development (structures per hectare or C/ha). Dominant Department of Planning land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

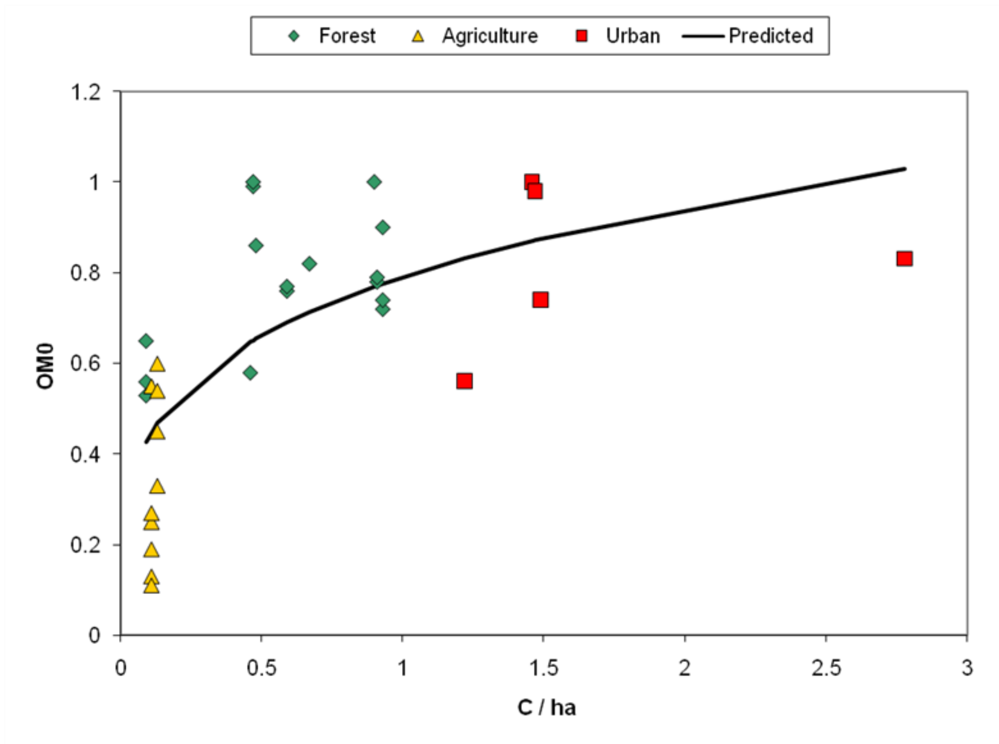
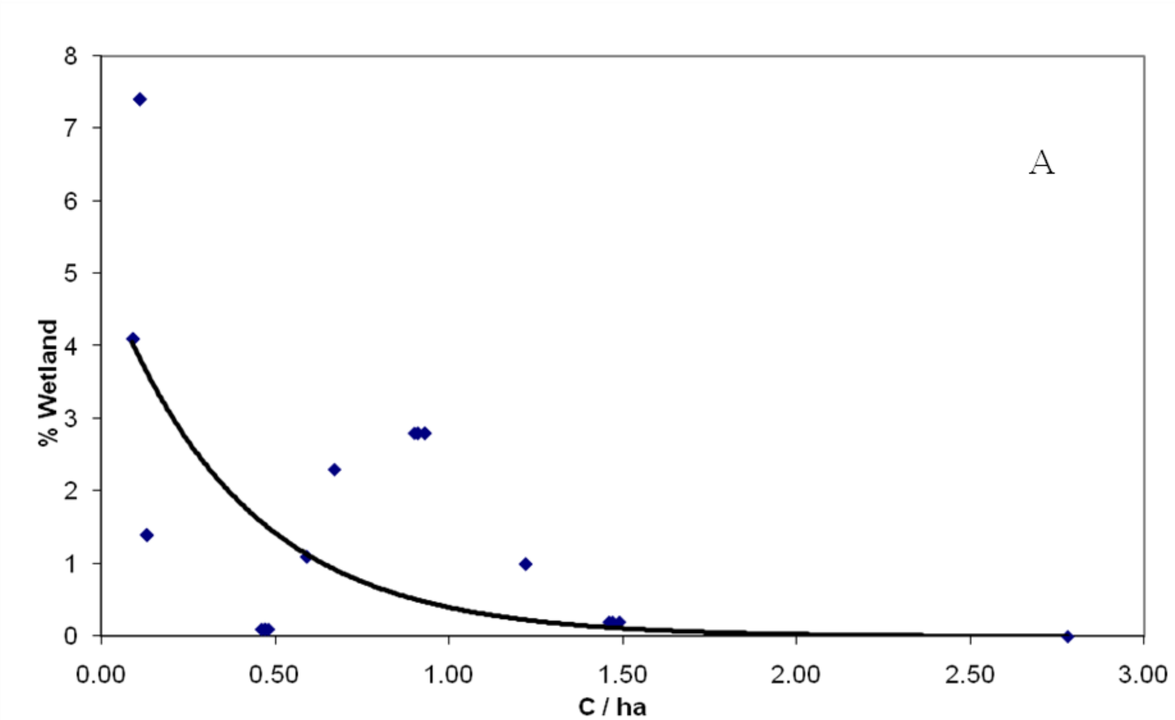
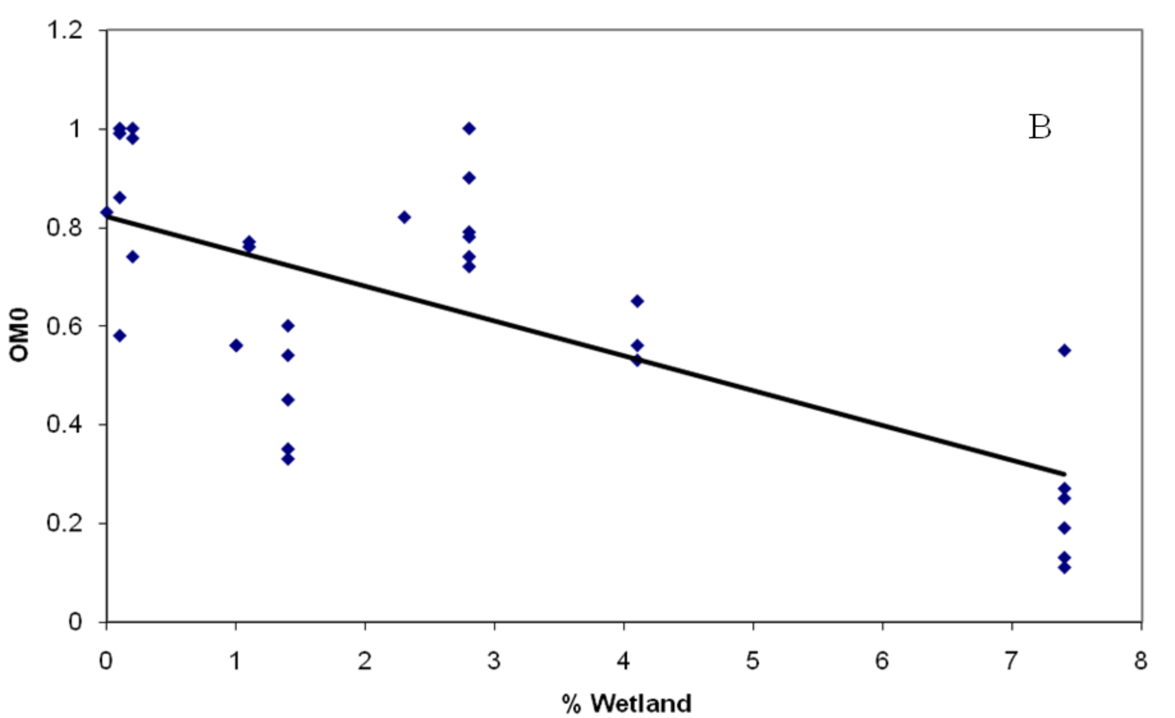


Figure 2-12. (A) Relationship of percent wetlands per watershed obtained from 2010 Department of Planning estimations and level of development (C/ha). (B) Proportion of samples without organic material (OM0) and percent wetlands per watershed.





Section 3 - Estuarine Fish Community Sampling

Alexis Park, Carrie Hoover, Margaret McGinty, Jim Uphoff, Benjamin Wahle

Introduction

Human population growth since the 1950s added a suburban landscape layer to the Chesapeake Bay (or Bay) watershed (Brush 2009) that has been identified as a threat (Chesapeake Bay Program or CBP 1999). Development converts land use typical of rural areas (farms, wetlands, and forests) to residential and industrial uses (Wheeler et al. 2005; National Research Council or NRC 2009; Brush 2009; Meals et al. 2010; Sharpley et al. 2013; Zhang et al. 2016). These are the basic trade-off in land use facing Maryland as its population grows (Maryland Department of Planning 2015) and they have ecological, economic, and societal consequences (Szaro et al. 1999).

Water quality and aquatic habitat is altered by agricultural activity and urbanization within watersheds. Both land-uses include pesticide and fertilizer application. Agriculturally derived nutrients have been identified as the primary driver of hypoxia and anoxia in the mainstem of the Bay (Hagy et al. 2004; Kemp et al. 2005; Fisher et al. 2006; Brush 2009; Zhang et al. 2016). Land in agriculture has been relatively stable but farming itself has become much more intensive (fertilizer and pesticide use has increased) to support crop production and population growth (Fisher et al. 2006; Brush 2009).

Urbanization may introduce additional industrial wastes, contaminants, stormwater runoff, and road salt (Brown 2000; NRC 2009; Benejam et al. 2010; McBryan et al. 2013; Branco et al. 2016) that act as ecological stressors. Extended exposure to biological and environmental stressors affect fish condition and survival (Rice 2002; Barton et al. 2002; Benejam et al. 2008; Benejam et al. 2010; Branco et al. 2016). Reviews by Wheeler et al. (2005), the National Research Council (NRC 2009) and Hughes et al. (2014a; 2014b) documented deterioration of non-tidal stream habitat with urbanization.

Development of the Bay watershed brings with it ecologically stressful factors that conflict with demand for fish production and recreational fishing opportunities from its estuary (Uphoff et al. 2011a; Uphoff et al 2016). Uphoff et al. (2011a) estimated target and limit impervious surface reference points (ISRPs) for productive juvenile and adult fish habitat in brackish (mesohaline) Chesapeake Bay subestuaries based on dissolved oxygen (DO) criteria, and associations and relationships of watershed impervious surface (IS), summer DO, and presence/absence of recreationally important finfish in bottom waters. Watersheds of brackish subestuaries at a target of 5.5% IS (expressed as IS equivalent to that estimated by the methodology used by Towson University for 1999-2000) or less (rural watershed) maintained mean bottom DO above 3.0 mg / L (threshold DO), but mean bottom DO was only occasionally at or above 5.0 mg / L (target DO). Mean bottom DO seldom exceeded 3.0 mg / L above 10% IS (suburban threshold; Uphoff et al. 2011a). Although bottom DO concentrations were influenced by development (indicated by IS) in brackish subestuaries, Uphoff et al. (2011b; 2012; 2013; 2014; 2015; 2016) have found adequate concentrations of DO in bottom channel habitat of tidal-fresh and oligohaline subestuaries with watersheds at suburban and urban levels of development. They suggested these bottom channel waters were not succumbing to low oxygen because stratification due to salinity was weak or absent, allowing for more mixing.

In 2017, we continued to evaluate summer nursery and adult habitat for recreationally important finfish in tidal-fresh (0-0.5 ‰), oligohaline (0.5-5.0 ‰) and meshohaline (5.0-18.0 ‰; Oertli, 1964) subestuaries of the Chesapeake Bay. In this report, we evaluated the influence of watershed development on target species presence/absence and abundance, total abundance of finfish, and finfish species richness. We analyzed the associations of land use (i.e., agriculture, forest, urban, and wetlands) and C/ha (structures per hectare) on the annual median bottom DO (mg/L) among subestuaries sampled during 2003-2017 using correlation analysis (Pearson correlation coefficients). We continue to evaluate and summarize the Choptank River subestuaries: Tred Avon River and Broad Creek in this annual report; upper Bay subestuaries, Northeast River and Bush River; and middle Bay subestuary, Middle River. In 2017, we also returned to re-evaluate previously sampled subestuaries, Wicomico River (Potomac River tributary, western shore) and Severn River.

Methods

Each subestuary sampled in the past or the present was classified into a salinity category based on the Venice System for Classification of Marine Waters (Oertli1964). Salinity influences distribution and abundance of fish (Hopkins and Cech 2003; Cyrus and Blaber, 1992; Allen 1982) and dissolved oxygen (DO; Kemp et al. 2005). Uphoff et al. (2012) calculated an arithmetic mean of all bottom salinity measurements over all years available to determine salinity class of each subestuary. Tidal-fresh ranged from 0-0.5 ‰; oligohaline, 0.5-5.0 ‰; and meshohaline, 5.0-18.0 ‰ (Oertli 1964). We grouped data by these classifications when examining effects of development throughout the sampled subestuaries.

We sampled six Chesapeake Bay subestuaries during 2017: Broad Creek and Tred Avon River, mesohaline subestuaries of the Choptank River; Wicomico River, mesohaline tributary of the lower Potomac River; and Northeast River (fresh-tidal) and Middle River (oligohaline) subestuaries located in the upper Chesapeake Bay (Table 3-1; Figure 3-1). This is the sixth year of sampling of Broad Creek, located downstream of the Tred Avon River (sampled since 2006) on the Choptank River, representing a rural to near suburban development within a single major watershed. Northeast River has been sampled since 2007 and the Middle River since 2009. We returned to the Wicomico River, previously sampled in 2003 and 2011-2012, and the Severn River previously sampled in 2003-2005.

We obtained compatible data from Bush River monitoring by citizen volunteers and staff from the Anita C. Leight Estuary Center (Table 3-1; Figure 3-1). The Bush River has been sampled since 2006; the Estuary Center and its citizen volunteers, trained in 2011 by the Fisheries Service staff, have taken over sampling. We verified and included their data in this report.

We used property tax map-based counts of structures in a watershed (C), standardized to hectares (C/ha), as our indicator of development (Table 3-1; Uphoff et al. 2012; Topolski 2015). Estimates of C/ha and MD DOP land use percentages were used for analyses of data from mesohaline subestuaries sampled during 2003-2017 (Table 3-2). Estimates were available through 2014 and 2014 estimates were used to represent 2015-

2017 in analyses. Methods used to estimate development (C/ha) and land use indicators (percent of watershed in agriculture, forest, wetlands, and urban land use) are explained in **General Spatial and Analytical Methods used in Job 1, Sections 1-3.** Development targets and limits and general statistical methods (analytical strategy and equations) are described in this section as well. Specific spatial and analytical methods for this section of the report are described below.

Tidal water surface area of each subestuary was estimated using the planimeter function on MDMerlin satellite photographs and maps (www.mdmerlin.net; Table 3-1). Shorelines were traced five-times for each system, and an average area was calculated. The lower limit of each water body was determined by drawing a straight line between the lowest downriver points on opposite shores (the mouth of each system) and the upper limits were to include all waters influenced by tides.

Surveys focused on eleven target species of finfish that fell within four broad life history groups: anadromous (American Shad, Alewife, Blueback Herring, Striped Bass), estuarine residents (semi-anadromous White Perch and Yellow Perch), marine migrants (Atlantic Menhaden and Spot), and tidal-fresh forage (Spottail Shiner, Silvery Minnow, and Gizzard Shad). With the exception of White Perch, adults of the target species were rare, and juveniles were common. Use of target species is widespread in studies of pollution and environmental conditions (Rice 2003). These species are widespread and support important recreational fisheries in the Bay (directly or as forage); they are well represented in commonly applied seine and-or trawl techniques (Bonzek et al. 2007); and the Bay serves as an important nursery for them (Lippson 1973; Funderburk et al. 1991; Deegan et al. 1997). Gear specifications and techniques were selected to be compatible with past and present MD DNR Fishing and Boating Service surveys (Carmichael et al. 1992; Bonzek et al. 2007; Durell 2007).

Ideally, four evenly spaced haul seine and bottom trawl sample sites were located in the upper two-thirds of each subestuary. Broad Creek lacked shoreline for a fourth seine site; the system has four trawl sites and three seine sites. Sites were not located near a subestuary's mouth to reduce influence of mainstem waters on fish habitat. We used GPS to record latitude and longitude at the middle of the trawl site, while latitude and longitude at seining sites were taken at the seine starting point on the beach.

Sites were sampled once every two weeks during July-September, totaling six annual visits per a system. The number of total samples collected from each system varied due to number of sites, SAV, and weather/tidal influences, and equipment issues. All sites on one river were sampled on the same day, usually during morning through mid-afternoon. Sites were numbered from upstream (site 1) to downstream (site 4). The crew leader flipped a coin each day to determine whether to start upstream or downstream. This coin-flip somewhat randomized potential effects of location and time of day on catches and DO. However, sites located in the middle would not be as influenced by the random start location as much as sites on the extremes because of the bus-route nature of the sampling design. If certain sites needed to be sampled on a given tide then the crew leader deviated from the sample route to accommodate this need. Trawl sites were generally in the channel, adjacent to seine sites. At some sites, seine hauls could not be made because of permanent obstructions, dense SAV beds, or lack of beaches. Seine and trawl sites were conducted one right after the other to avoid time of day or tidal influences.

Water quality parameters were recorded at all sites. Temperature (ºC), DO (mg / L), conductivity (mS / cm), salinity (‰), and pH were recorded at the surface, middle, and bottom of the water column at the trawl sites and at the surface of the seine site. Mid-depth measurements were omitted at sites with less than 1.0 m difference between surface and bottom. Secchi depth was measured to the nearest 0.1 m at each trawl site. Weather, tide state (flood, ebb, high or low slack), date, and start time were recorded for all sites.

A 4.9 m headrope semi-balloon otter trawl was used to sample fish in mid-channel bottom habitat. The trawl was constructed of treated nylon mesh netting measuring 38 mm stretch-mesh in the body and 33 mm stretch-mesh in the cod-end, with an untreated 12 mm stretch-mesh knotless mesh liner. The headrope was equipped with floats and the footrope was equipped with a 3.2 mm chain. The net used 0.61 m long by 0.30 m high trawl doors attached to a 6.1 m bridle leading to a 24.4 m towrope. Trawls were towed in the same direction as the tide. The trawl was set up tide to pass the site halfway through the tow, allowing the same general area to be sampled regardless of tide direction. A single tow was made for six minutes at 3.2 km / hr (2.0 miles / hr) per site on each visit. The contents of the trawl were emptied into a tub for processing.

A 30.5 m × 1.2 m bagless beach seine, constructed of untreated knotted 6.4 mm stretch mesh nylon, was used to sample inshore habitat. The float-line was rigged with 38.1 mm by 66 mm floats spaced at 0.61 m intervals and the lead-line rigged with 57 gm lead weights spaced evenly at 0.55 m intervals. One end of the seine was held on shore, while the other was stretched perpendicular from shore as far as depth permitted and then pulled with the tide in a quarter-arc. The open end of the net was moved towards shore once the net was stretched to its maximum. When both ends of the net were on shore, the net was retrieved by hand in a diminishing arc until the net was entirely pursed. The section of the net containing the fish was then placed in a washtub for processing. The distance the net was stretched from shore, maximum depth of the seine haul, primary and secondary bottom type, and percent of seine area containing aquatic vegetation were recorded.

All fish captured were identified to species and counted. Striped Bass and Yellow Perch were separated into juveniles and adults. White Perch were separated into three categories (i.e., juvenile, small adults, and harvestable size adults) based on size and life stage. The small adult White Perch category consisted of ages 1+ White Perch smaller than 200 mm.

2017 Sampling Summary - Three basic metrics of community composition were estimated for subestuaries sampled: geometric mean catch of all species, total number of species (species richness), and species comprising 90% of the catch. The geometric mean (GM) was estimated as the back-transformed mean of $\log_e$-transformed catches (Ricker 1975; Hubert and Fabrizio 2007). The GM is a more precise estimate of central tendency of fish catches than the arithmetic mean but is on a different scale (Ricker 1975; Hubert and Fabrizio 2007). We noted which target species were within the group that comprised 90% of fish collected. We summarized these metrics by salinity type since some important ecological attributes (DO and high or low SAV densities) appeared to reflect salinity class (Uphoff et al. 2012).

We plotted species richness collected by seine and by 4.9 m trawl against C/ha by salinity class. A greater range of years (1989-2017) was available for seine samples than the 4.9 m trawl (2004-2017) due to a change from the 3.1 m trawl used during 1989-2002 (Carmichael et al. 1992). We set a minimum number of samples (15) for a subestuary in a year to include estimates of species richness based on species accumulation versus sample size analyses in Uphoff et al. (2014). This eliminated years where sampling in a subestuary ended early due to site losses, typically from SAV growth.

Dissolved Oxygen Dynamics - Dissolved oxygen concentrations were evaluated against a target of 5.0 mg / L and a threshold of 3.0 mg / L (Batiuk et al. 2009; Uphoff et al. 2011a). The target criterion was originally derived from laboratory experiments but was also associated with asymptotically high presence of target species in trawl samples from bottom channel habitat in mesohaline subestuaries (Uphoff et al. 2011a). Target DO was considered sufficient to support aquatic life needs in Chesapeake Bay (Batiuk et al. 2009) and has been used in a regulatory framework to determine if a water body is meeting its designated aquatic life uses. Presence of target species in bottom channel trawls declined sharply when bottom DO fell below the 3.0 mg / L threshold (Uphoff et al. 2011a). We estimated the percentages of DO samples in each subestuary that did not meet the target or threshold for all samples (surface to bottom) and for bottom waters alone. The percentages of DO measurements that met or fell below the 5 mg / L target (Vtarget) or fell at or below the 3 mg / L threshold (Vthreshold) were estimated as:

$$Vtarget = (Ntarget / Ntotal)*100;$$

and

$$Vthreshold = (Nthreshold / Ntotal)*100;$$

where Ntarget was the number of measurements meeting or falling below 5 mg / L, Nthreshold was the number of measurements falling at or below 3 mg / L, and Ntotal was total sample size.

Separate Pearson correlation analyses were conducted for surface or bottom temperature or C/ha with surface or bottom DO for all subestuaries sampled since 2003. This analysis explored multiple hypotheses related to DO conditions. Structures per hectare estimates were considered proxies for nutrient loading and processing due to development in the subestuaries in this analysis (Uphoff et. al 2011a). Water temperature would influence system respiration and stratification (Kemp et al. 2005; Murphy et al. 2011; Harding et al. 2016). Conducting correlation analyses by salinity classification provided a means of isolating the increasing influence of salinity on stratification from temperature. Our primary interest was in associations of C/ha to DO in surface and bottom channel waters. Temperature and salinity were potential influences on DO because of their relationships with DO saturation and stratification (Kemp et al. 2005; Murphy et al. 2011; Harding et al. 2016). We correlated mean surface temperature with mean surface DO, mean bottom temperature with mean bottom DO, and C/ha with surface and bottom DO for each salinity class. We chose annual means of surface or bottom DO and water temperature in summer at all sites within a subestuary for analyses to match the geographic scale of C/ha estimates (whole watershed) and characterize chronic conditions.

Land Use Categories, C/ha, and Mesohaline Subestuary Bottom Dissolved Oxygen - We analyzed the associations of land use (i.e., agriculture, forest, urban, and wetlands) and C/ha (structures per hectare) on the annual median bottom DO (mg/L) among mesohaline systems sampled during 2003-2017 using correlation analysis (Pearson correlation coefficients). We obtained land use estimates for our watersheds from the Maryland Department of Planning for 2002 and 2010 (MD DOP 2002 and 2010). The MD DOP provides agriculture, forest, urban, and wetlands estimates periodically rather than annually, but C/ha is estimated annually. Median summer bottom DO estimates made before 2010 were compared with 2002 MD DOP land use estimates and those made for 2010-2017 were matched with 2010 MD DOP estimates. Four categories of land use were present for all mesohaline tributaries, agriculture, forest, urban, and wetlands were estimated based on the land portion of the watershed (water area was excluded).

Choptank River Subestuaries - The trajectories of C/ha since 1950 were plotted for each of the two Choptank River tributaries. Bottom DO measurements during 2006-2017 were plotted against C/ha for each Choptank River subestuary. The percentage of target and threshold violations (violations meant that target or threshold criteria were not met; they did not have a legal meaning) were estimated using all DO measurements combined (surface, middle, and bottom) and for bottom DO measurements alone. Annual mean bottom DO in Tred Avon River at each station for 2006-2017 was estimated and plotted by year.

Middle, Northeast, Severn, and Wicomico Rivers - We analyzed water quality and fish community data from the Middle, Northeast, Severn, and Wicomico Rivers' subestuaries in more detail. We assembled our time-series of Secchi depth, SAV area, bottom DO (mg/L), pH, and salinity (ppt). Geometric means (GM) of total fish abundance and their 95% CI's were estimated for 4.9 m trawl for sampled time-series. Compositions of all finfish species caught by seine for all time-series were graphed. The top 90% of finfish species occurring in annual trawl and seine catches were estimated for each subestuary time-series. Middle River had notable fish kills in 2015 and 2016.

Errata - Conductivity measurements in 2012-2013 were recorded incorrectly. The raw conductivity was recorded instead of the specific conductivity, which compensates for temperature. An equation was used to correct the error and convert the raw conductivity measurements that were recorded to specific conductivity (Fofonoff and Millard 1983):

$$\text{Specific Conductivity} = \text{Conductivity} / (1 + 0.02 \cdot T) - 25));$$

for each ºC change in water temperature (T) there was a 2% change in conductivity.

During database restructuring in 2017-2018 of summer estuarine fish and water quality data, older data was discovered to be entered incorrectly (i.e., entered twice, skipped, or disorderly). Data entry could have been incorrect due to untrained outside agencies entering data and lack of quality control. Data found to be entered incorrectly was corrected; quality control is ongoing due to size of database. Corrected data is used throughout the analyses in this report.

Results and Discussion

2017 Sampling Summary - Northeast River and Broad Creek were the only subestuaries that did not have any DO readings less than the target level (5.0 mg / L) during 2017 (Table 3-3). Eleven percent of all DO measurements in 2017 from Middle River were below the target; Bush River had 12%; Wicomico River, 15%; Tred Avon River, 17%; and Severn River, 47%. In 2017, only two subestuaries did not have any bottom DO estimates below the 3 mg / L threshold; Northeast River and Broad Creek. The remaining five subestuaries had threshold bottom DO violations: Middle River, 8%; Wicomico River, 9%; Tred Avon River, 13%; Bush River, 33%; and Severn River, 88%. The Severn River, a highly developed (C/ha = 2.29) mesohaline system, had a higher frequency of target bottom and threshold bottom DO violations than the more developed system, oligohaline Middle River (C/ha = 3.34; Table 3-3).

During 2017, seine CPUE's and species richness were not calculated in Broad Creek and Middle River since less than 15 samples were collected. Dense SAV prevented seining in Middle River. Seining in Broad Creek was very restricted because of high tides that limited beach availability and dense SAV in one seine site (BROS04).

Geometric mean catch per seine haul ranged from 149 to 223 among the seven subestuaries sampled during 2017, with little indication that salinity class or development level exerted an influence (Table 3-4). The GM of the one tidal-fresh subestuary ranked first out of the five remaining subestuaries, oligohaline ranked fourth, and mesohaline subestuaries ranked second, third, and fifth (Table 3-4).

Between 18 and 24 species were encountered in seine samples from mesohaline subestuaries. Oligohaline Bush and Middle Rivers' subestuaries 10 and 20 species. Tidal-fresh Northeast River had 24 species (Table 3-4).

A plot of species richness in seine samples and C/ha did not suggest a relationship in tidal-fresh or oligohaline, or mesohaline subestuaries (Figure 3-2). Tidal-fresh subestuary watersheds were represented by a limited range of C/ha (0.42 - 0.66). Oligohaline subestuary watersheds were represented by the widest range of C/ha (0.08 - 3.32, rural to urban) of the three salinity classes (Figure 3-2). Mesohaline subestuary watersheds were represented by the largest number of samples within a range (55; 0.07 – 2.67) compared to the number of samples for tidal-fresh and oligohaline subestuaries (22 and 32; Figure 3-2).

A total of 33,854 fish representing 48 species were captured by beach seine in 2017 (Table 3-4). Eight species comprised 90% of the total fish caught in 2017, including (from greatest to least) Atlantic Silverside, Gizzard Shad, Striped Killifish, Bay Anchovy, Banded Killifish, White Perch (juveniles), Alewife, and White Perch (adults). Gizzard Shad and White Perch (juvenile and adult) represented target species among the species comprising 90% of the total catch. Six target species were present among species comprising 90% of the seine catch throughout all subestuaries: Gizzard Shad were present in this category in two of the seven subestuaries; White Perch (juveniles and/or adults) in three; Spottail Shiner in one; Alewife in one; Striped Bass in one. Target species comprised 90% of the catch in the Bush and Northeast Rivers.

Geometric mean trawl catches during 2017 were between 15 and 132 (Table 3-5). Subestuaries had 22–24 samples. Bush River had only 6 samples; no GM CPUE was calculated due to limited number of samples and was not included in rankings. Broad Creek had the greatest GM (132) and Severn River had the lowest (15). Oligohaline Middle River ranked low, fifth out of six subestuaries. Mesohaline subestuaries' GM's were interspersed throughout, ranking first, third, and sixth. The only tidal-fresh subestuary ranked second.

Number of species captured by trawl in subestuaries sampled during 2017 (9 – 18) overlapped for all three salinity classifications (Table 3-5). Number of species in Bush River was only calculated based on the six samples collected and not included in analysis. A plot of species richness in trawl samples against C/ha (2003-2017) did not indicate a relationship of development and number of species for tidal-fresh (species richness ranging from 14 to 25) or oligohaline subestuaries (species richness ranging from 12 to 26; Figure 3-3). Species richness (ranging from 3 to 23) declined in mesohaline subestuaries as C/ha advanced beyond the threshold (C/ha = 0.83; Figure 3-3). Trawl species richness ranged from 9 to 26 when development was below the threshold. Fifteen of the 129 species richness estimates for all tidal-fresh, oligohaline, and mesohaline subestuaies were below the median species richness (17) when development was above the threshold. Eight of the 15 estimates of species richness were in mesohaline subestuaries and fell steadily when C/ha was over the threshold, from 11-16 species at South River (C / ha ~1.25) and 3-9 species at Magothy and Severn Rivers (C / ha ranging from 2.06 to 2.29; Figure 3-3). Five of the 15 estimates of species richness estimates were in oligohaline subestuaries, Bush River 15-16 species (C / ha ~1.45) and Middle River 12-15 species (C / ha ~3.35; Figure 3-3). Two of the 12 estimates of species richness estimates were in fresh-tidal subestuaries, Piscataway Creek 14 species (C / ha ~1.38) and Mattawoman Creek 15 species (C / ha ~0.88; Figure 3-3).

Sampling with a 4.9 m headrope bottom trawl was conducted in all seven subestuaries in 2017 (Table 3-5). Unlike seining, all trawl sites could be sampled (except in bad weather). A total of 22,735 fish and 31 fish species were captured. Three species comprised 90% of the total catch for 2017 (from most to least): Bay Anchovy, White Perch (juveniles and adults), and Atlantic Croaker; all three were target species. Target species comprising 90% of the catch in all of the seven subestuaries sampled during 2017 were White Perch (juveniles and/or adults) and Bay Anchovy in five subestuaries; Atlantic Croaker in three; and Weakfish in one (Table 3-5).

Dissolved Oxygen Dynamics - Correlation analyses of DO with temperature and C/ha in subestuaries sampled since 2003 (Table 3-6) indicated that DO responded to temperature and C/ha differently depending on salinity classification (Table 3-7). Mean bottom DO in summer surveys declined below the threshold level in mesohaline tributaries, but did not in oligohaline or tidal-fresh (Figure 3-4). There were a few years where mean survey bottom DO fell below the target in oligohaline subestuaries, but remained above 4.0 mg / L; these below target conditions would not affect the use of this habitat (Uphoff et al. 2011a). Mean surface DO in summer surveys did not fall below the threshold for oligohaline and tidal-fresh subestuaries, but one mesohaline subestuary fell below the target in one year (Figure 3-5).

Negative associations of surface and bottom DO with corresponding mean water temperatures at depth were detected for oligohaline subestuaries by correlation analyses (surface: $r = -0.33$, $P = 0.04$, $N = 40$; bottom: $r = -0.60$, $P < .0001$, $N = 40$; Table 3-7), suggesting respiration was a factor in oligohaline subestuaries. Oligohaline subestuaries were shallower than most subestuaries of the other salinity categories, making them more likely to be warmer throughout than the other categories. Associations of temperature and DO were not detected in mesohaline or tidal-fresh subestuaries. The strongest and only negative association between bottom DO and C/ha was found in mesohaline subestuaries (r = -0.62, $P < 0.0001$, N= 68); mesohaline subestuaries were where strongest stratification was expected. Positive associations of surface DO with development were suggested for oligohaline subestuaries (r = 0.34, $P = 0.03$, N= 40). A positive association between bottom DO and C/ha in fresh-tidal subestuaries was significant (r = 0.38, $P = 0.02$, N= 34). Given that multiple comparisons were made, correlations that were significant at $P < 0.02$ might be considered spurious if one rigorously adheres to significance testing (Nakagawa 2004). However, oligohaline and tidal-fresh subestuaries were less likely to stratify because of low or absent salinity and the biological consequences of no or positive relationships would be similar (i.e., a negative impact on habitat would be absent). Sample sizes of mesohaline subestuaries (N = 68) were greater than oligohaline (N = 40) or tidal-fresh subestuaries (N = 34), so ability to detect significant associations in mesohaline subestuaries was greater.

Levels of bottom DO were not negatively associated with development in tidal-fresh or oligohaline subestuaries, but were in mesohaline subestuaries (Table 3-7). Depletion of bottom DO in mesohaline subestuaries to hypoxic or anoxic levels represented a direct loss of habitat that could be occupied. Uphoff et al. (2011a) determined that the odds of adult and juvenile White Perch, juvenile Striped Bass, Spot, and Blue Crabs being present in shore zone seine samples from mesohaline subestuaries were not influenced by development, but odds of target species being present in bottom channel trawl samples were negatively influenced by development.

The extent of bottom channel habitat that can be occupied does not appear to diminish with development in tidal-fresh and oligohaline subestuaries due to low DO. Sampling of DO in dense SAV beds in tidal-fresh Mattawoman Creek in 2011 indicated that shallow water habitat could be negatively impacted by low DO within the beds (Uphoff et al. 2012; 2013; 2014; 2015; 2016); it was not feasible for us to routinely monitor fish within the beds, so the impact on target finfish could not be estimated. During November, 2015, the oligohaline Middle River subestuary (the most heavily developed watershed in our study) experienced an extensive fish kill attributable to harmful algal blooms (MDE 2016). Middle River has exhibited diverse and abundant fish communities over the course of our monitoring.

Land Use Categories, C/ha, and Mesohaline Subestuary Bottom Dissolved Oxygen - We correlated percent of watershed in MD DOP land use categories (agriculture, forest, urban, and wetlands), and C/ha (structures per hectare) to explore associations among land uses. Correlations of agriculture with C/ha and urban cover were negative and strong (r = -0.76, P <0.0001 and r = -0.81, P <0.0001, respectively); the correlation of urban land cover with C/ha was positive and strong (r = 0.93, P < 0.0001); and forest cover was moderately and positively correlated with wetland (r = 0.62, P = 0.006; Table 3-8). Remaining pairings of categories were not well correlated (Table 3-8).

After inspection of scatter plots, forest cover was further divided into regional categories, East (eastern shore of Chesapeake Bay: Broad Creek, Corsica River, Harris Creek, Langford Creek, Miles River, Tred Avon River, and Wye River) and West (western shore of Chesapeake Bay: Breton Bay, Magothy River, Rhode/West Rivers, Severn River, South River, St. Clements River, and Wicomico River) reflecting lower percentages of forest cover on the eastern Shore (Figure 3-6). Additional regression analyses were done in 2016 (see Table 3-9 in Uphoff et al. 2016).

Inspection of the scatter plot of percent of watershed in agriculture versus median bottom DO in mesohaline subestuaries indicated an ascending limb of median DO when agricultural coverage went from 2.6 to 40.9% that was comprised entirely of western shore subestuaries (Figure 3-6). Median DO measurements beyond this level of agricultural coverage (42.6-71.6% agriculture) were from eastern shore subestuaries and the DO trend appeared to be stable or declining (Figure 3-6). Development was predominant at low levels of agriculture (< 20%). Agricultural coverage and C/ha were strongly and inversely correlated, so the positive trend of DO with agriculture when agricultural coverage was low was likely to reflect development's negative impact. We split agricultural coverage and median DO data into western and eastern regions and used a linear regression for each region to describe regional changes in DO with agriculture (model equation was median DO = % agriculture). The relationship was positive for the western shore (slope = 0.127, SE = 0.019; r^2 = 0.703, P < 0.0001, N = 21; Table 3-9) and negative for the eastern shore (slope = -0.047, SE = 0.012; r^2 = 0.278, P = 0.0005, N = 39; Table 3-9). Predictions of median DO for mesohaline western shore subestuaries rose from 0.8 mg / L at 6.0% agricultural coverage to 5.5 mg / L at 40.9% (Figure 3-7). Predictions of median DO for mesohaline eastern shore subestuaries fell from 5.4 mg / L at 42.6% agricultural coverage to 4.2 mg/L at 71.6% (Figure 3-6). Additional regional DO regression analyses were completed in 2016 (see Table 3-11 and Figure 3-8 in Uphoff et al. 2016).

Choptank River Subestuaries - In 2017, we continued to assess the trends in mesohaline subestuaries Broad Creek and Tred Avon River (Figure 3-7); monitoring is being conducted in Harris Creek by others to evaluate large scale oyster restoration. In 2015 and 2016, we assessed the Choptank River subestuaries, Broad Creek, Harris Creek, and Tred Avon River, detailed analyses of how these three subestuaries compare are in the 2015 and 2016 annual reports (see Uphoff et al. 2015 and 2016). Broad Creek and Tred Avon River watersheds have undergone development at different levels, with the former approaching the development threshold and the latter having just passed the target level of development. We have monitored the Tred Avon in anticipation of measuring DO and fish community changes in a mesohaline subestuary as its watershed develops over time and contrast it with less developed watersheds in the same region.

Percentages of land in agriculture (42-43%), forest (21-25%), and urban (31-33%) categories were similar among the three Choptank River subestuaries (MD DOP 2010; Table 3-11; Figure 3-8). However, wetlands varied among the three systems, comprising 0.4% of Broad Creek's watershed, 5.6% of Harris Creek's, and 0.8% of Tred Avon's watershed (Table 3-12). Water comprised a larger fraction of the area in Broad and Harris Creeks (57% and 61%, respectively) than Tred Avon River (27%, respectively; i.e., water to watershed ratios were higher in the former; MD DOP 2010).

Tax map estimates of C/ha indicated that the Tred Avon River watershed has been subject to more development than Broad Creek watersheds and more than indicated by the MD DOP urban category (Figure 3-7). Time-series for both watersheds started at a rural level of development (C/ha ranged from 0.1 to 0.2) in 1950 (Figure 3-7). Broad Creek's watershed experienced low growth (C/ha = 0.29 in 2014), while more growth occurred in Tred Avon River's watershed (C/ha = 0.76 in 2014; Figure 3-7). Development accelerated noticeably in the Tred Avon watershed during 1999-2007 and then slowed. Tred Avon River's watershed has been approaching the suburban threshold, C/ha > 0.87). Broad Creek watershed has passed the rural development target (C / ha = 0.27).

During 2017, bottom DO readings below the threshold (DO < 3.0 mg / L) were more frequent in the more developed Tred Avon River than Broad Creek (Figure 3-9). Seven percent of bottom DO measurements during 2006-2017 in Tred Avon River were below the DO threshold; 32% were below the DO target; in Broad Creek, 1% were below the threshold and 14% of all DO values were below the target (Figure 3-9; Table 3-12).

An ANOVA of Tred Avon River stations and bottom DO indicated significant differences were present (F = 38.33; DF = 3; $P < 0.0001$; N = 283). Tukey Studentized Range and Tukey Honestly Significant Difference (HSD) tests indicated that bottom DO at station 1 (station at Easton, Maryland) was significantly lower than downstream stations 2, 3, and 4 (critical value of studentized range = 3.66). This result was consistent with other mesohaline tributaries with high impervious surface; DO declines as you move upstream (Uphoff et al. 2011a). The mean and SE for bottom DO at all stations in Tred Avon River were 5.33 mg / L and 0.08, respectively. Mean and SE for bottom DO at station 1 were 4.01 mg / L and 0.20; station 2 was 5.74 mg / L and 0.12; station 3 was 5.79 mg / L and 0.12; and station 4 was 5.81 mg / L and 0.12. Deterioration of DO at the uppermost station (station 1; Figure 3-10 and 3-11) during 2006-2017 indicated that watershed development around Easton was the source of poor water quality rather than water intruding from downstream. During 2017, mean bottom DO at station 1 was above threshold value but below the target value and the overall median for the time-series. Stations 2, 3, and 4 had mean bottom DO above the overall median for the time-series during 2017and mean DO was above the target (Figure 3-10). ANOVAs of Broad Creek (F = 0.9; DF = 3; $P = 0.4434$; N = 138) stations and bottom DO concentrations did not indicate significant differences among stations. The overall mean and SE for bottom DO in Broad Creek were 6.06 mg / L and 0.09, respectively. In 2017, stations 1 and 4 dropped below the overall median for the time-series; stations 2 and 3 were at or above the overall median for the time-series (Figure 3-10 and 3-11). Mean bottom DO's were above the target at each Broad Creek station.

Middle, Northeast, Severn, and Wicomico Rivers –The oligohaline Middle River was routinely sampled during 2009-2017; tidal-fresh Northeast River was sampled 2007-2010 and 2012-2017; and mesohaline subestuaries, Severn (2003-2005, 2017) and Wicomico (2003, 2012, and 2017) Rivers, were sampled sporadically. In the fall of 2015, a fish kill occurred in Middle River. The Maryland Department of the Environment reported that the fish kill were caused by high amounts of toxic algae, *Karlodinium veneficum*, whose toxin causes gill damage to fish when in high concentrations (MDE 2016; 2017). In 2015, MD DNR biologists discovered and confirmed zebra mussel presence in the Middle River. Nogaro and Steinman (2014) indicated that invasive mussels can change a system's ecosystem and promote toxic algae blooms. These events triggered a review of our data collected during 2009-2017 for the Middle River. A comprehensive analysis of the Middle and Gunpowder Rivers was reported in Uphoff et al. (2016).

In Middle, Northeast, Severn, and Wicomico Rivers we examined trends in Secchi depth to see if water clarity was increasing; especially due to the colonization of Zebra Mussels in the Middle River (Figure 3-12). During 2009-2017, median Secchi measurements in Middle River ranged from 0.5m to 1.1m during 2009-2017; 2015, the year Zebra Mussels appeared, had the greatest Secchi depth (Figure 3-12). Zebra Mussels were not observed in 2016-2017 and Secchi depths declined and appear to be returning to pre-Zebra Mussel levels. Northeast River median Secchi depth measurements ranged from 0.3m to 0.5m during sampling years; Severn River ranged from 1.0 m to 1.2 m; and Wicomico River remained steady at 0.5 m (Figure 3-12).

Middle, Northeast, Severn, and Wicomico Rivers SAV coverage varied. Data that was only partially mapped or not mapped at all were not included in this assessment. Middle River has SAV coverage data from 1989 to 2000 and 2002-2017; Northeast River, 1994 to 2000 and 2002-2017; Severn River, 1994-2000 and 2002-2017; and Wicomico River, 1989-2002, 2004-2015, and 2017 (Figure 3-13). Coverage of SAV in 2017, was below the overall time-series median in Middle River (14%; 1989-1999, 2006, and 2011-2012) and Severn River (3%; 1994-1995, 2000, 2003, 2009-2014, and 2017; Figure 3-13). In 2017, SAV coverage was higher than the overall time-series median in Northeast River (2%; 1994-2000, 2003, and 2012-2014) and Wicomico River (6%; 1989-1995, 2008-2011, and 2014; Figure 3-13). Middle and Severn Rivers' SAV coverage appears erratic over the years, while the Northeast and Wicomico Rivers SAV coverage appears to form a pattern with highs and lows (Figure 3-13).

Bottom DO (mg/L) did not appear to fluctuate dramatically from year to year in Middle, Northeast, Severn, and Wicomico Rivers (Figure 3-14). Median bottom DO estimates ranged from 5.3 mg / L to 7.2 mg / L for Middle River; from 5.8 mg / L to 8.0 mg / L for Northeast River; from 0.1 mg / L to 2.2 mg / L for Severn River; and from 4.4 mg / L to 6.0 mg / L for Wicomico River (Figure 3-13). Measurements of pH for Middle, Northeast, Severn, and Wicomico Rivers were typically between 7 and 8 (Figure 3-15), but Northeast River pH measurements appeared higher than the others.

During 2009-2017, median salinity fluctuated over a wider range in the Middle River (1.2-5.5 ‰), Severn River (5.7-8.7 ‰), and Wicomico River (5.7-12.0 ‰) than Northeast River (0.09-0.13 ‰; Figure 3-16). Salinity was the highest for Middle River in 2016, with salinity reaching 7.36 ‰; Northeast River in 2007 and 2010, reaching 0.2 ‰; Severn River in 2005 reaching, 10.1 ‰; and Wicomico River in 2010, reaching 13 ‰ (Figure 3-16). Increased salinity could have negative or lethal effects on the Zebra Mussels found in Middle River in 2015, depending on variability of salinity measurements and exposure time. USGS (2017) indicated Zebra Mussels in North America can tolerate salinity up to 4 ‰ and Strayer and Smith (1993), Spidle (1994), and Walton (1996) indicated repeated salinity levels greater than 5 ‰ were lethal. Zebra Mussels within the Chesapeake Bay subestuaries are usually found at 3 ‰ or less (M. Ashton, MD DNR *personal communication*).

Middle, Northeast, Severn, and Wicomico Rivers GM's of all finfish (GM) and their 95% CI's were plotted for all sampling years (Figure 3-17). Relative abundance was highest during 2009-2011 for Middle River; 2010-2011 and 2014 for Northeast River; 2017 for Severn River; and 2010 and 2012 for Wicomico River (Figure 3-17). Since 2015, both Middle and Northeast Rivers exhibited lower finfish catches (Figure 3-17). The sporadic sampling within the Severn River indicates that the Severn River had a slight increase in finfish catches between 2003-2005 and 2017; Wicomico River's GM in 2017 was in the middle of the available estimates (Figure 3-17).

We graphed composition of finfish in trawl catches for all years that Middle, Northeast, Severn, and Wicomico Rivers were sampled (Figure 3-18); species that defined the top 90% were identified and the remainder were grouped as “other species”. Middle River trawl catches were composed of White Perch juveniles (46%), Bay Anchovy (30%), White Perch adults (12%), Pumpkinseed (6%), and other species (32 species; 4.8%; Figure 3-18). Northeast River trawl data was composed of White Perch juveniles (45%), White Perch adult (38%), Bay Anchovy (6%), and other species (31 species; 10%; Figure 3-18). Severn River trawl data was composed of Bay Anchovy (73%), White Perch (adult; 21%), and other species (10 species; 6%; Figure 3-18). Wicomico River trawl data was composed of Bay Anchovy (39%), White Perch juveniles (20%), Spot (19%), White Perch adults (11%), and other species (28 species; 11%; Figure 3-18).

We separated all subestuaries sampled from 1989-2017 by salinity class, then ranked GMs for all species combined to find where the Middle, Northeast, Severn, and Wicomico Rivers ranked when compared to other subestuaries in their respective salinity classes (Table 3-13). Middle River had one GM within the top ten oligohaline subestuary GMs; five GMs in the middle; and three GMs in the bottom ten GMs (Table 3-13). Northeast River had three GMs within the top ten tidal-fresh subestuary GMs; eight in the middle; and one in the bottom ten GMs (Table 3-13). Severn River had the last four ranked GMs for mesohaline subestuaries and Wicomico River had all five GMs ranked in the middle (Table 3-13).

Middle, Northeast, Severn, and Wicomico Rivers GM seine catches of all finfish and their 95% CI’s were plotted for all years sampled (Figure 3-19). Relative abundance was highest during 2011 and 2015 for Middle River; 2016-2017 for Northeast River; 2003 for Severn River; and 2003 for Wicomico River (Figure 3-19). Both the Severn and Wicomico Rivers exhibited lower finfish catches than previous years (Figure 3-19). Middle River declined in 2017 but the 2017 GM is still greater than 2012-2014 GMs (Figure 3-19). Northeast River declined slightly from 2016 to 2017 (Figure 3-19); 2016-2017 GMs are greater than 2007-2015 GMs.

Composition of finfish in seine catches for all years sampled for Middle, Northeast, Severn, and Wicomico Rivers were graphed (Figure 3-20); species that define the top 90% were identified and the remainder of species were grouped and labeled as “other species”. Composite Middle River seine catches were comprised of Gizzard Shad (28%), White Perch juveniles (23%), Pumpkinseed (11%), White Perch adults (7%), Banded Killifish (7%), Blueback Herring (7%), Spottail Shiner (5%), Atlantic Silverside (3%), and other species (25 species; 10%; Figure 3-20). Northeast River seine catches were composed of Gizzard Shad (38%), Blueback Herring (18%), White Perch juveniles (17%), White Perch adults (7%), Alewife (4%), Bay Anchovy (4%), Pumpkinseed (3%), and other species (34 species; 10%; Figure 3-20). Severn River seine collections were composed of White Perch juveniles (46%), Atlantic Silverside (25%); Striped Bass juveniles (9%), Striped Killifish (5%), White Perch adults (3%), Banded Killifish (3%), and other species (31 species; 9%; Figure 3-20). Composition of Wicomico River seine catches was Atlantic Silverside (58%), White Perch juveniles (12%), White Perch adults (9%), Atlantic Menhaden (5%), Bay Anchovy (4%), Striped Killifish (3%), and other species (30 species; 10%; Figure 3-20).

Overall, the relative conditions at Northeast and Wicomico Rivers have been fairly stable over the available time-series. Middle River conditions have been declining. Severn River conditions remained poor and were the worst among the four subestuaries analyzed.

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Tables

Table 3-1. Percent impervious cover (IS), structures per hectare (C/ha), watershed area, area of tidal water, and salinity class for the subestuaries sampled in 2017.

Area	Subestuary	IS	C/ha	Total Hectares	Water Hectares	Salinity Class
Mid-Bay	Broad Creek	5.1	0.29	4,730	3,148	Mesohaline
Mid-Bay	Tred Avon River	9.2	0.76	9,563	2,429	Mesohaline
Mid-Bay	Severn River	18.5	2.29	17,921	2,940	Mesohaline
Lower-Bay	Wicomico River	4.2	0.21	19,936	4,310	Mesohaline
Upper-Bay	Bush River	14.3	1.51	36,038	2,962	Oligohaline
Mid-Bay	Middle River	23.5	3.34	2,753	982	Oligohaline
Upper-Bay	Northeast River	6.9	0.48	16,342	1,579	Tidal Fresh

Table 3-2. Estimates of C/ha and land use percentages from Maryland Department of Planning (2002 and 2010) for subestuaries sampled 2003-2017.

River	Year	C/ha	Agriculture	Wetland	Forest	Urban
Breton Bay	2003	0.265021	26.3	0.5	61.5	11.3
Breton Bay	2004	0.281742	26.3	0.5	61.5	11.3
Breton Bay	2005	0.298533	26.3	0.5	61.5	11.3
Broad Creek	2012	0.293475	42.6	0.4	25.4	31.5
Broad Creek	2013	0.2958	42.6	0.4	25.4	31.5
Broad Creek	2014	0.296435	42.6	0.4	25.4	31.5
Broad Creek	2015	0.296435	42.6	0.4	25.4	31.5
Broad Creek	2016	0.296435	42.6	0.4	25.4	31.5
Broad Creek	2017	0.296435	42.6	0.4	25.4	31.5
Bush River	2006	1.406067	25.4	3.2	35.0	36.2
Bush River	2007	1.429209	25.4	3.2	35.0	36.2
Bush River	2008	1.447218	25.4	3.2	35.0	36.2
Bush River	2009	1.461508	25.4	3.2	35.0	36.2
Bush River	2010	1.470554	18.0	3.2	29.9	47.8
Bush River	2011	1.479684	18.0	3.2	29.9	47.8
Bush River	2012	1.487564	18.0	3.2	29.9	47.8
Bush River	2013	1.506516	18.0	3.2	29.9	47.8
Bush River	2014	1.51473	18.0	3.2	29.9	47.8
Bush River	2015	1.51473	18.0	3.2	29.9	47.8
Bush River	2016	1.51473	18.0	3.2	29.9	47.8
Bush River	2017	1.51473	18.0	3.2	29.9	47.8
Corsica River	2003	0.171949	64.3	0.4	27.4	7.9
Corsica River	2004	0.184452	64.3	0.4	27.4	7.9
Corsica River	2005	0.193959	64.3	0.4	27.4	7.9
Corsica River	2006	0.211423	64.3	0.4	27.4	7.9
Corsica River	2007	0.224649	64.3	0.4	27.4	7.9
Corsica River	2008	0.23705	64.3	0.4	27.4	7.9
Corsica River	2011	0.250586	60.4	0.1	25.5	13.2
Corsica River	2012	0.254	60.4	0.1	25.5	13.2
Gunpowder River	2009	0.720501	30.6	1.0	32.1	35.6
Gunpowder River	2010	0.722787	30.6	1.0	32.1	35.6
Gunpowder River	2011	0.724747	30.6	1.0	32.1	35.6
Gunpowder River	2012	0.726953	30.6	1.0	32.1	35.6
Gunpowder River	2013	0.729485	30.6	1.0	32.1	35.6
Gunpowder River	2014	0.731964	30.6	1.0	32.1	35.6
Gunpowder River	2015	0.731964	30.6	1.0	32.1	35.6
Gunpowder River	2016	0.731964	30.6	1.0	32.1	35.6
Harris Creek	2012	0.387979	44.9	5.6	19.7	29.8
Harris Creek	2013	0.387708	44.9	5.6	19.7	29.8
Harris Creek	2014	0.387979	44.9	5.6	19.7	29.8
Harris Creek	2015	0.387979	44.9	5.6	19.7	29.8
Harris Creek	2016	0.387979	44.9	5.6	19.7	29.8
Langford Creek	2006	0.072884	71.6	1.5	23.0	3.9
Langford Creek	2007	0.073608	71.6	1.5	23.0	3.9
Langford Creek	2008	0.073504	71.6	1.5	23.0	3.9
Magothy River	2003	2.678242	6.0	0.0	32.8	61.1

Table 3-2 (Cont).

Mattawoman Creek	2003	0.762374	11.9	1.2	59.4	27.4
Mattawoman Creek	2004	0.786923	11.9	1.2	59.4	27.4
Mattawoman Creek	2005	0.807012	11.9	1.2	59.4	27.4
Mattawoman Creek	2006	0.83238	11.9	1.2	59.4	27.4
Mattawoman Creek	2007	0.858197	11.9	1.2	59.4	27.4
Mattawoman Creek	2008	0.871208	11.9	1.2	59.4	27.4
Mattawoman Creek	2009	0.883196	11.9	1.2	59.4	27.4
Mattawoman Creek	2010	0.897762	9.3	2.8	53.9	34.2
Mattawoman Creek	2011	0.910569	9.3	2.8	53.9	34.2
Mattawoman Creek	2012	0.902877	9.3	2.8	53.9	34.2
Mattawoman Creek	2013	0.914415	9.3	2.8	53.9	34.2
Mattawoman Creek	2014	0.925789	9.3	2.8	53.9	34.2
Mattawoman Creek	2015	0.925789	9.3	2.8	53.9	34.2
Mattawoman Creek	2016	0.925789	9.3	2.8	53.9	34.2
Middle River	2009	3.300754	4.5	2.2	27.9	63.9
Middle River	2010	3.320004	3.4	2.1	23.3	71.0
Middle River	2011	3.329084	3.4	2.1	23.3	71.0
Middle River	2012	3.333079	3.4	2.1	23.3	71.0
Middle River	2013	3.335258	3.4	2.1	23.3	71.0
Middle River	2014	3.348333	3.4	2.1	23.3	71.0
Middle River	2015	3.348333	3.4	2.1	23.3	71.0
Middle River	2016	3.348333	3.4	2.1	23.3	71.0
Middle River	2017	3.348333	3.4	2.1	23.3	71.0
Miles River	2003	0.23851	56.1	1.4	30.4	12.1
Miles River	2004	0.243382	56.1	1.4	30.4	12.1
Miles River	2005	0.244374	56.1	1.4	30.4	12.1
Nanjemoy Creek	2003	0.084899	15.1	4.1	73.1	7.6
Nanjemoy Creek	2008	0.091092	15.1	4.1	73.1	7.6
Nanjemoy Creek	2009	0.091197	15.1	4.1	73.1	7.6
Nanjemoy Creek	2010	0.091568	12.4	4.1	68.7	14.7
Nanjemoy Creek	2011	0.091568	12.4	4.1	68.7	14.7
Nanjemoy Creek	2012	0.091727	12.4	4.1	68.7	14.7
Nanjemoy Creek	2013	0.091885	12.4	4.1	68.7	14.7
Nanjemoy Creek	2014	0.092256	12.4	4.1	68.7	14.7
Nanjemoy Creek	2015	0.092256	12.4	4.1	68.7	14.7
Nanjemoy Creek	2016	0.092256	12.4	4.1	68.7	14.7
Northeast River	2007	0.43979	36.7	0.1	42.7	20.1
Northeast River	2008	0.443095	36.7	0.1	42.7	20.1
Northeast River	2009	0.449642	36.7	0.1	42.7	20.1
Northeast River	2010	0.459127	31.1	0.1	38.6	28.9
Northeast River	2011	0.464818	31.1	0.1	38.6	28.9
Northeast River	2012	0.467816	31.1	0.1	38.6	28.9
Northeast River	2013	0.473507	31.1	0.1	38.6	28.9
Northeast River	2014	0.479688	31.1	0.1	38.6	28.9
Northeast River	2015	0.479688	31.1	0.1	38.6	28.9
Northeast River	2016	0.479688	31.1	0.1	38.6	28.9
Northeast River	2017	0.479688	31.1	0.1	38.6	28.9

Table 3-2 (Cont.)

Piscataway Creek	2003	1.300181	12.8	0.3	45.8	40.6
Piscataway Creek	2006	1.38186	12.8	0.3	45.8	40.6
Piscataway Creek	2007	1.401642	12.8	0.3	45.8	40.6
Piscataway Creek	2009	1.433215	12.8	0.3	45.8	40.6
Piscataway Creek	2010	1.448746	10.0	0.2	40.4	47.0
Piscataway Creek	2011	1.462066	10.0	0.2	40.4	47.0
Piscataway Creek	2012	1.472495	10.0	0.2	40.4	47.0
Piscataway Creek	2013	1.49035	10.0	0.2	40.4	47.0
Piscataway Creek	2014	1.503274	10.0	0.2	40.4	47.0
Rhode/West Rivers	2003	0.5484	36.4	1.0	44.9	17.7
Rhode/West Rivers	2004	0.5549	36.4	1.0	44.9	17.7
Rhode/West Rivers	2005	0.5611	36.4	1.0	44.9	17.7
Severn River	2003	2.058995	11.1	0.2	41.2	47.3
Severn River	2004	2.09118	11.1	0.2	41.2	47.3
Severn River	2005	2.148981	11.1	0.2	41.2	47.3
Severn River	2017	2.292926	5.0	0.2	28.0	65.1
South River	2003	1.234149	19.9	0.4	50.5	29.0
South River	2004	1.2497	19.9	0.4	50.5	29.0
South River	2005	1.26471	19.9	0.4	50.5	29.0
St. Clements River	2003	0.192976	40.9	0.8	51.3	7.0
St. Clements River	2004	0.19621	40.9	0.8	51.3	7.0
St. Clements River	2005	0.198532	40.9	0.8	51.3	7.0
Tred Avon River	2006	0.691286	50.1	1.0	21.6	27.2
Tred Avon River	2007	0.713035	50.1	1.0	21.6	27.2
Tred Avon River	2008	0.724433	50.1	1.0	21.6	27.2
Tred Avon River	2009	0.736144	50.1	1.0	21.6	27.2
Tred Avon River	2010	0.74681	43.2	0.9	21.6	33.6
Tred Avon River	2011	0.750993	43.2	0.9	21.6	33.6
Tred Avon River	2012	0.75298	43.2	0.9	21.6	33.6
Tred Avon River	2013	0.754025	43.2	0.9	21.6	33.6
Tred Avon River	2014	0.757267	43.2	0.9	21.6	33.6
Tred Avon River	2015	0.757267	43.2	0.9	21.6	33.6
Tred Avon River	2016	0.757267	43.2	0.9	21.6	33.6
Tred Avon River	2017	0.757267	43.2	0.9	21.6	33.6
Wicomico River	2003	0.193906	34.7	4.6	48.5	12.0
Wicomico River	2011	0.212462	31.6	4.6	44.9	18.7
Wicomico River	2012	0.213493	31.6	4.6	44.9	18.7
Wicomico River	2017	0.216379	31.6	4.6	44.9	18.7
Wye River	2007	0.095131	67.7	0.7	23.5	8.1
Wye River	2008	0.095424	67.7	0.7	23.5	8.1

Table 3-3. Percentages of all DO measurements and bottom DO measurements that did not meet target (= 5.0 mg/L) and threshold (= 3.0 mg/L) conditions during July-September, 2017, for each subeastuary. C/ha = structures per hectare.

				All DO		Bottom DO	
Subestuary	Salinity Class	C/ha	N	% < 5.0 mg/L	N	% < 5.0 mg/L	% < 3.0 mg/L
Bush River	Tidal Fresh	1.51	25	12%	6	50%	33%
Northeast River	Tidal Fresh	0.48	93	0%	24	0%	0%
Middle River	Oligohaline	3.34	72	11%	24	33%	8%
Broad Creek	Mesohaline	0.29	72	0%	22	0%	0%
Severn River	Mesohaline	2.29	87	47%	24	96%	88%
Tred Avon River	Mesohaline	0.76	89	17%	24	42%	13%
Wicomico River	Mesohaline	0.21	84	15%	22	50%	9%

Table 3-4. Beach seine catch summary for 2017. C/ha = structures per hectare. GM = Geometric mean. Italicized species are target species.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Broad Creek	2	12	18*	Atlantic Silverside	0.29	3,129	**
				Mummichog			
				Banded Killifish			
				Striped Killifish			
Bush River	4	15	20	*Gizzard Shad*	1.51	3,660	151
				Spottail Shiner			
				White Perch (Adult)			
				White Perch (JUV)			
Middle River	1	2	10*	Banded Killifish	3.34	364	**
				Pumpkinseed			
				White Perch (JUV)			
Northeast River	4	23	24	*Gizzard Shad*	0.48	8,880	223
				Alewife			
				White Perch (JUV)			
				Blueback Herring			
				White Perch (Adult)			
Severn River	3	18	19	Atlantic Silverside	2.29	3,531	159
				Striped Killifish			
				Bay Anchovy			
Tred Avon River	4	18	24	Atlantic Silverside	0.76	3,085	149
				Bay Anchovy			
				Mummichog			
				Striped Killifish			
				Banded Killifish			
				Striped Bass (JUV)			
Wicomico River	4	24	18	Atlantic Silverside	0.21	11,205	157
Grand Total	22	112	48	Atlantic Silverside		33,854	
				Gizzard Shad			
				Striped Killifish			
				Bay Anchovy			
				Banded Killifish			
				White Perch (JUV)			
				Alewife			
				White Perch (Adult)			

*Number of species for Broad Creek and Middle River is only calculated based on the number of samples, not included in analysis due to <15 samples.

**Broad Creek and Middle River GM CPUE is not calculated due to due to <15 samples.

Table 3-5. Bottom trawl (4.9m) catch summary, 2017. C/ha = structures per hectare. GM is the geometric mean catch of all fish per seine. Italicized species are considered target species.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Broad Creek	4	22	13	*Bay Anchovy*	0.29	8,318	132
Bush River	3	6	11*	*White Perch (JUV)*	1.51	3,057	**
				White Perch (Adult)			
Middle River	4	24	12	*White Perch (JUV)*	3.34	2,144	73
				White Perch (Adult)			
				Bay Anchovy			
				Atlantic Croaker			
Northeast River	4	24	18	*White Perch (JUV)*	0.48	3,462	105
				White Perch (Adult)			
Severn River	4	24	9	*Bay Anchovy*	2.29	722	15
Tred Avon River	4	24	14	*Bay Anchovy*	0.76	2,900	78
				Hogchoker			
				Atlantic Croaker			
				Weakfish			
				White Perch (Adult)			
Wicomico River	4	24	15	*Bay Anchovy*	0.21	2,132	77
				Hogchoker			
				Atlantic Croaker			
				White Perch (Adult)			
Grand Total	27	148	31	*Bay Anchovy*		22,735	
				White Perch (JUV)			
				White Perch (Adult)			
				Atlantic Croaker			

*Number of species for Bush River is only calculated based on the 6 samples, not included in analysis due to <15 samples.

**Bush River GM CPUE is not calculated due to due to <15 samples.

Table 3-6. Subestuaries sampled during 2003–2017 divided by salinity class with mean annual surface and bottom temperatures, mean annual dissolved oxygen (mg/L), and C / ha.

			Temperature		Dissolved Oxygen	
River	Year	C / ha	Surface	Bottom	Surface	Bottom
Mesohaline						
Blackwater River	2006	0.037667	28.14444	27.98333	5.266667	4.116667
Breton Bay	2003	0.265021	26.4	25.6875	8.1	3.745833
	2004	0.281742	27.00625	25.95417	7.360417	3.725
	2005	0.298533	28.62143	27.50833	6.978571	3.990417
Broad Creek	2012	0.293475	27.49537	26.60167	8.30439	5.965417
	2013	0.2958	27.29952	26.48913	7.257857	5.756957
	2014	0.296435	27.62292	26.64167	7.64625	5.77625
	2015	0.296435	28.04902	27.0487	7.930244	6.631304
	2016	0.296435	29.16368	28.33	7.304324	6.163182
	2017	0.296435	27.00514	26.29409	7.504	6.106364
Corsica River	2003	0.171949	25.9	26.13043	6.5	4.669565
	2004	0.184452	27.17561	26.875	5.568293	4.571
	2005	0.193959	28.5381	28.14286	6.483333	3.08
	2006	0.211423	27.38571	26.84118	7.548571	4.047059
	2007	0.224649	25.94146	25.81818	6.2425	4.218182
	2008	0.23705	26.20488	25.21538	7.319512	4.207692
	2010	0.244283	34.35789	26.61818	5.688148	5.01125
	2011	0.250586	27.00233	27.01	5.295455	3.280556
	2012	0.253996	27.79286	27.46875	4.7125	3.403333
Fishing Bay	2006	0.033932	26.22963	25.27857	7.240741	6.792857
Harris Creek	2012	0.387979	26.55429	26.41783	7.438571	6.354348
	2013	0.387708	26.39146	26.05292	7.015366	6.01
	2014	0.387979	27.61383	26.68357	6.842397	4.838725
	2015	0.387979	26.61634	26.62125	7.193902	6.564583
	2016	0.387979	27.8235	27.75348	6.64975	6.016522
Langford Creek	2006	0.072884	27.05111	26.52083	6.946667	5.675
	2007	0.073608	26.23261	25.47895	6.691892	5.684615
	2008	0.073504	27.46738	26.64689	6.852975	5.051951
Magothy River	2003	2.678242	25.7	25.31429	7.3	2.035714
Miles River	2003	0.23851	25.5	25.6	6.5	4.092
	2004	0.243382	25.7525	25.63913	6.0825	5.466087
	2005	0.244374	28.03333	27.44167	5.961905	3.308333
Rhode River	2003	0.466475	25	24.69286	7.1	4.8
	2004	0.473539	27	26.94545	6.578261	5.389091
	2005	0.476647	27.77917	27.15833	6.5	4.025833
Severn River	2003	2.058995	26.3	24.75185	7.6	1.574074
	2004	2.09118	27.41667	26.17917	7.05	2.636667
	2005	2.148981	28.01489	26.22917	7.07234	0.96375
	2017	2.292926	26.9269	26.07042	6.856429	1.779583
South River	2003	1.234149	25.4	24.56429	7.6	2.610714
	2004	1.2497	25.7875	25.47917	6.45625	3.77375
	2005	1.26471	27.57083	26.67083	6.016667	2.49125

Table 3-6 (Cont.)

St. Clements River	2003	0.192976	26	25.28519	8.2	3.481481
	2004	0.19621	26.07917	25.775	6.8375	4.608333
	2005	0.198532	27.12154	26.36158	6.854435	4.420257
Transquaking River	2006	0.028893	26.675	22.75	5.75	5.85
Tred Avon River	2006	0.691286	27.11591	26.72083	6.181818	5.341667
	2007	0.713035	26.85	26.59167	6.485106	5.391304
	2008	0.724433	26.27708	25.6087	6.895833	4.833333
	2009	0.736144	26.15417	26.03333	7.370833	6.305833
	2010	0.74681	27.46957	26.92917	7.084783	5.258333
	2011	0.750993	28.48409	28.18095	6.815909	5.10619
	2012	0.75298	27.27106	27.15833	7.022292	5.46625
	2013	0.754025	26.78808	26.39038	7.15	4.998077
	2014	0.754025	26.65875	26.50542	6.116667	5.902917
	2015	0.754025	27.99688	27.59542	6.924792	5.537083
	2016	0.754025	28.88563	28.43958	7.273958	5.150417
	2017	0.754025	26.48929	26.13375	7.011905	5.044167
West River	2003	0.642769	24.9	24.31429	7.4	4.835714
	2004	0.64863	26.83333	26.59167	7.366667	5.583333
	2005	0.658398	27.96111	27.15	6.722222	3.986667
Wicomico River	2003	0.193906	25.4	23.83043	7	5.852174
	2010	0.212462	25.425	25.30476	6.057353	5.214706
	2011	0.213493	27.07826	26.8913	5.567609	4.301739
	2012	0.215297	27.56833	27.382	6.585417	5.444667
	2017	0.216379	26.70146	25.72545	7.546042	4.622727
Wye River	2007	0.095131	26.75417	26.45	7.075	5.7
	2008	0.095424	26.98444	26.21875	5.702222	5.113333
Oligohaline						
Bohemia River	2006	0.111575	26.7881	26.02	7.009524	6.41
Bush River	2006	1.406067	25.47632	24.28	7.957895	7.472727
	2007	1.429209	27.02222	26.41818	7.677778	6.536364
	2008	1.447218	26.58571	24.2	9	5.433333
	2009	1.461508	25.88095	24.3375	9.409524	8.54
	2010	1.470554	27.71944	23.8	7.791667	7.04
	2011	1.479684	26.98205	26.94	6.465641	5.496667
	2012	1.487564	26.79	26.16667	6.6275	5.200833
	2013	1.506516	25.11071	24.725	9.98	6.7275
	2014	1.51473	26.52255	25.64399	7.297698	5.727833
	2015	1.51473	26.52255	25.64399	7.297698	5.727833
	2016	1.51473	27.97865	27.47647	7.971176	6.34
	2017	1.51473	26.07619	29.13333	7.097895	4.661667
Gunpowder River	2009	0.720501	25.7093	26.05	7.390698	6.789444
	2010	0.722787	25.16889	25.90769	7.893182	7.130769
	2011	0.724747	25.08581	25.55556	8.283871	7.144444
	2012	0.726953	26.48444	25.93133	8.193778	6.708667
	2013	0.729485	25.85282	27.45667	8.047949	6.1
	2014	0.731964	26.65354	26.14882	7.276846	5.756238
	2015	0.731964	27.51024	27.65	8.016341	6.631538
	2016	0.731964	27.70429	26.46375	7.42619	6.175

Table 3-6 (Cont.)

Middle River	2009	3.300754	26.4963	25.78182	7.266667	6.067727
	2010	3.320004	24.64865	24.2	8.437838	7.113636
	2011	3.329084	27.13095	26.41667	8.354286	7.333333
	2012	3.333079	28.05	26.59565	8.817105	5.209167
	2013	3.335258	27.1203	26.45545	7.58303	5.786364
	2014	3.348333	26.55717	26.01033	7.553754	6.044032
	2015	3.348333	28.46556	27.2	8.195556	6.232083
	2016	3.348333	28.87435	27.81727	7.559565	5.69
	2017	3.348333	25.53786	25.16917	7.804643	5.360417
Nanjemoy Creek	2003	0.084899	25.9	28.8	7.3	4.96
	2008	0.091092	27.52571	26.575	7.851429	6.65
	2009	0.091197	26.30556	24.6375	7.05	7.49375
	2010	0.091568	26.49658	24.79857	7.664474	7.015714
	2011	0.091568	29.33625	28.54962	6.1275	5.303077
	2012	0.091727	26.17875	25.9245	6.73075	5.978
	2013	0.091885	26.87944	26.295	6.759722	5.86
	2014	0.092256	26.78102	26.36262	7.660629	6.249737
	2015	0.092256	27.39889	27.098	7.159722	6.324
	2016	0.092256	28.49389	28.20857	6.856667	5.162857
			Tidal Fresh			
Mattawoman Creek	2003	0.762374	26	25.74545	9	8.813636
	2004	0.786923	27.32581	27.14348	8.341935	7.95087
	2005	0.807012	28.772	28.0875	7.736	7.266875
	2006	0.83238	27.04762	26.44	7.095	6.495455
	2007	0.858197	26.8875	26.85417	6.704167	6.475
	2008	0.871208	26.39583	24.51538	7.968182	6.325
	2009	0.883196	26.20478	26.63846	7.920217	7.858846
	2010	0.897762	26.20625	26.09524	6.945833	6.62381
	2011	0.910569	27.07938	27.45794	6.327083	6.511765
	2012	0.902877	26.69563	26.81977	7.39875	6.999535
	2013	0.914415	26.35125	25.93733	9.218958	8.404
	2014	0.925789	26.72971	26.23991	7.48062	6.173849
	2015	0.925789	27.91396	26.84419	8.6575	7.742791
	2016	0.925789	28.46917	28.02571	6.961667	6.542143
Northeast River	2007	0.43979	26.82708	26.42727	9.733333	7.747826
	2008	0.443095	25.34792	24.98421	8.429167	7.7
	2009	0.449642	26.33061	25.54783	9.35102	7.361304
	2010	0.459127	25.90426	26.20588	7.761702	6.782353
	2011	0.464818	25.96739	25.7125	6.872826	5.792083
	2012	0.467816	27.78333	27.59167	7.877083	6.033333
	2013	0.473507	26.6137	26.10957	9.333696	7.055217
	2014	0.479688	26.9435	26.52358	7.716936	6.805324
	2015	0.479688	26.65521	26.23	7.835833	6.17375
	2016	0.479688	27.94563	26.86333	8.809167	7.099583
	2017	0.479688	26.38404	25.67708	9.378511	7.800833

Table 3-6 (Cont.)

Piscataway Creek	2003	1.300181	25.6	24.63333	10.2	8.333333
	2006	1.38186	28.155	24.96667	8.7	6.85
	2007	1.401642	27.46667	26	8.566667	7.6
	2009	1.433215	26.71667	27.06667	8.555556	6.622857
	2010	1.448746	27.06667	25.075	9.355556	7.625
	2011	1.462066	28.24611	30.06667	9.05	9.466667
	2012	1.472495	27.92	25.50875	9.532105	9.3425
	2013	1.49035	27.18706	26.22111	9.87	7.648889
	2014	1.503274	26.97934	26.28079	8.662952	7.334941

Table 3-7. Correlations of 2003-2016 mean annual surface and bottom DO (mg/L) with like water temperatures at depth (surface and bottom) or watershed development (C/ha = structures per hectare), by salinity class. Bold numbers indicate a significant relationship.

DO Depth	Statistics	Temperature	C / ha
	Mesohaline		
Surface	r	-0.21081	0.1803
	P	0.0844	0.1412
	N	68	68
Bottom	r	0.08169	**-0.61736**
	P	0.5078	**<.0001**
	N	68	**68**
	Oligohaline		
Surface	r	**-0.32528**	**0.33963**
	P	**0.0406**	**0.032**
	N	**40**	**40**
Bottom	r	**-0.60429**	-0.07511
	P	**<.0001**	0.6451
	N	**40**	40
	Tidal Fresh		
Surface	r	-0.04909	0.26539
	P	0.7828	0.1293
	N	34	34
Bottom	r	0.04591	**0.37617**
	P	0.7966	**0.0283**
	N	34	**34**

Table 3-8. Correlations (r) among land use categories and C/ha, level of significance (P), and sample size (N) for mesohaline subestuaries sampled during 2003-2017. Land cover estimates were estimated by Maryland Department of Planning for 2002 and 2010. Bold numbers indicate a significant relationship at α < 0.05.

			Land Use Categories			
	Statistics	**C/ha**	**Agriculture**	**Forest**	**Wetland**	**Urban**
C/ha	r	1				
	P					
	N					
Agriculture	r	**-0.76478**	1			
	P	**<.0001**				
	N	**52**				
Forest	r	-0.06003	-0.32326	1		
	P	0.6725	0.1907			
	N	52	18			
Wetland	r	-0.21699	-0.04892	**0.61657**	1	
	P	0.1223	0.8471	**0.0064**		
	N	52	18	**18**		
Urban	r	**0.92656**	**-0.81485**	-0.28032	-0.31599	1
	P	**<.0001**	**<.0001**	0.2599	0.2015	
	N	**52**	**18**	18	18	

Table 3-9. Statistics and parameter estimates for regional (western and eastern shore) linear regressions of median bottom DO versus percent agricultural coverage.

Linear Model	**Western Shore: Median DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	51.20737	51.20737	47.29	<.0001	
Residual	20	21.65828	1.08291			
Total	21	72.86565				
r^2 = 0.7028						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.48592	0.51285	0.95	0.3547	-0.58386	1.5557
Agriculture (%)	0.12779	0.01858	6.88	<.0001	0.08902	0.16655
Linear Model	**Eastern Shore: Median DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	9.34299	9.34299	14.6	0.0005	
Residual	38	24.30918	0.63972			
Total	39	33.65217				
r^2 = 0.2776						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	7.76303	0.66317	11.71	<.0001	6.4205	9.10555
Agriculture (%)	-0.04713	0.01233	-3.82	0.0005	-0.07209	-0.02216

Table 3-10. Statistics and parameter estimates for a quadratic regression of median bottom DO versus agricultural (%) coverage.

Linear Model	Median Bottom DO = Agriculture (%) Coverage					
ANOVA	df	SS	MS	F	Significance F	
Regression	2	92.47526	46.23763	53.68	<.0001	
Residual	59	50.8186	0.86133			
Total	61	143.29386				
$r^2 = 0.6454$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	-0.45103	0.51249	-0.88	0.3824	-1.47653	0.57447
Agriculture (%)	0.24772	0.02639	9.39	<.0001	0.19491	0.30054
Agriculture (%)^2	-0.00256	0.00032304	-7.93	<.0001	-0.00321	-0.00191

Table 3-11. Distribution of major land use category for Broad Creek and Tred Avon subestuaries' watershed estimated by Maryland Department of Planning for 2010. Numbers for the four land use categories are percentage estimates for the land portion of the watershed only; water area is removed from each of these categories. Water is the percent of land and water area represented by water.

	Subestuary	
Land Use Category	**Broad Creek**	**Tred Avon River**
Agriculture	42.55	43.2
Forest	25.39	21.63
Urban	31.47	33.57
Wetlands	0.36	0.85
Water	57.28	24.4

Table 3-12. Percentages of all DO measurements (surface, middle, and bottom) and bottom DO measurements that did not meet target (= 5.0 mg/L) and threshold (= 3.0 mg/L) conditions during July - September for all sampling years, for Broad Creek and Tred Avon River subestuaries. N = total DO measurements for each system by year for all and bottom DO measurements.

				All DO		Bottom DO	
Subestuary	Year	C / ha	N	% < 5.0 mg/L	N	% < 5.0 mg/L	% < 3.0 mg/L
Broad Creek	2012	0.29	83	7%	24	17%	4%
	2013	0.30	78	10%	23	30%	0%
	2014	0.30	81	6%	24	21%	0%
	2015	0.30	82	1%	23	0%	0%
	2016	0.29	76	4%	22	9%	0%
	2017	0.29	72	0%	22	0%	0%
Tred Avon River	2006	0.69	91	19%	24	38%	0%
	2007	0.71	93	11%	23	26%	4%
	2008	0.72	89	24%	21	48%	14%
	2009	0.74	95	6%	24	13%	0%
	2010	0.75	89	20%	24	38%	13%
	2011	0.75	82	22%	21	48%	10%
	2012	0.75	94	10%	24	29%	0%
	2013	0.75	103	15%	26	31%	15%
	2014	0.75	96	11%	24	21%	0%
	2015	0.75	96	8%	24	21%	13%
	2016	0.76	96	13%	24	38%	13%
	2017	0.76	89	17%	24	42%	13%

Table 3-13. Subestuaries sampled during 2003-2017, by salinity class and ranked by annual 4.9m trawl geometric mean (GM) catches.

River	Year	GM	Rank
Mesohaline			
Miles River	2003	626	1
West River	2003	545	2
Rhode River	2003	524	3
Broad Creek	2014	401	4
Corsica River	2003	378	5
Broad Creek	2012	294	6
Langford River	2007	273	7
Tred Avon River	2010	264	8
Langford River	2006	258	9
Corsica River	2004	251	10
Corsica River	2011	238	11
Tred Avon River	2014	192	12
Harris Creek	2014	174	13
Corsica River	2006	174	14
Wye River	2007	170	15
Rhode River	2005	163	16
Corsica River	2012	162	17
Langford River	2008	161	18
Corsica River	2010	161	19
Tred Avon River	2008	155	20
Tred Avon River	2012	155	21
Harris Creek	2012	155	22
Broad Creek	2017	148	23
Broad Creek	2016	147	24
Broad Creek	2013	142	25
Tred Avon River	2007	137	26
Corsica River	2007	131	27
Fishing Bay	2006	131	28
Trasnquaking River	2006	131	29
West River	2005	125	30
Tred Avon River	2016	121	31
Wicomico River	2010	120	32
Wye River	2008	114	33
South River	2003	110	34
Wicomico River	2012	110	35
Corsica River	2005	109	36
Tred Avon River	2009	104	37
Broad Creek	2015	103	38
Tred Avon River	2017	98	39
Tred Avon River	2011	92	40
Harris Creek	2013	89	41

Table 3-13 (Cont.)

Corsica River	2008	86	42
Miles River	2004	82	43
Wicomico River	2017	81	44
Tred Avon River	2015	80	45
Tred Avon River	2013	77	46
Tred Avon River	2006	76	47
Miles River	2005	72	48
Wicomico River	2011	65	49
Wicomico River	2003	59	50
St. Clements River	2005	54	51
Harris Creek	2016	51	52
Harris Creek	2015	40	53
Rhode River	2004	38	54
South River	2005	35	55
Blackwater River	2006	35	56
Brenton Bay	2005	34	57
West River	2004	34	58
Magothy River	2003	33	59
St. Clements River	2003	31	60
South River	2004	21	61
Brenton Bay	2003	18	62
St. Clements River	2004	17	63
Brenton Bay	2004	16	64
Severn River	2017	16	65
Severn River	2003	9	66
Severn River	2004	5	67
Severn River	2005	3	68
Oligohaline			
Bush River	2011	666	1
Nanjemoy Creek	2013	576	2
Bush River	2014	528	3
Middle River	2011	520	4
Bush River	2010	473	5
Bush River	2017	471	6
Nanjemoy Creek	2015	416	7
Gunpowder River	2010	401	8
Nanjemoy Creek	2014	396	9
Gunpowder River	2011	394	10
Nanjemoy Creek	2011	385	11
Bush River	2007	324	12
Bush River	2015	321	13
Bush River	2009	319	14

Table 3-13 (Cont.)

Middle River	2010	315	15
Nanjemoy Creek	2010	309	16
Nanjemoy Creek	2016	297	17
Middle River	2009	292	18
Gunpowder River	2009	289	19
Middle River	2015	286	20
Nanjemoy Creek	2009	284	21
Middle River	2016	261	22
Bush River	2012	261	23
Middle River	2014	251	24
Bush River	2016	250	25
Nanjemoy Creek	2012	224	26
Gunpowder River	2012	224	27
Gunpowder River	2014	219	28
Gunpowder River	2015	218	29
Bush River	2013	215	30
Bush River	2008	210	31
Nanjemoy Creek	2008	209	32
Gunpowder River	2016	206	33
Middle River	2013	181	34
Bush River	2006	152	35
Middle River	2012	148	36
Gunpowder River	2013	147	37
Bohemia River	2006	115	38
Nanjemoy Creek	2003	93	39
Middle River	2017	74	40
Tidal-Fresh			
Mattawoman Creek	2014	580	1
Northeast River	2010	392	2
Piscataway Creek	2011	320	3
Northeast River	2014	291	4
Northeast River	2011	290	5
Piscataway Creek	2010	290	6
Mattawoman Creek	2013	283	7
Mattawoman Creek	2004	252	8
Piscataway Creek	2014	221	9
Mattawoman Creek	2015	217	10
Mattawoman Creek	2011	208	11
Northeast River	2009	198	12
Northeast River	2012	191	13
Mattawoman Creek	2005	187	14
Northeast River	2013	186	15

Table 3-13 (Cont.)

Piscataway Creek	2013	184	16
Northeast River	2008	152	17
Northeast River	2015	150	18
Northeast River	2007	149	19
Mattawoman Creek	2016	149	20
Mattawoman Creek	2003	144	21
Piscataway Creek	2012	119	22
Northeast River	2017	105	23
Piscataway Creek	2009	105	24
Northeast River	2016	96	25
Mattawoman Creek	2010	84	26
Mattawoman Creek	2006	75	27
Mattawoman Creek	2012	72	28
Mattawoman Creek	2007	56	29
Piscataway Creek	2003	42	30
Piscataway Creek	2006	28	31
Mattawoman Creek	2008	27	32
Piscataway Creek	2007	8	33
Mattawoman Creek	2009	6	34

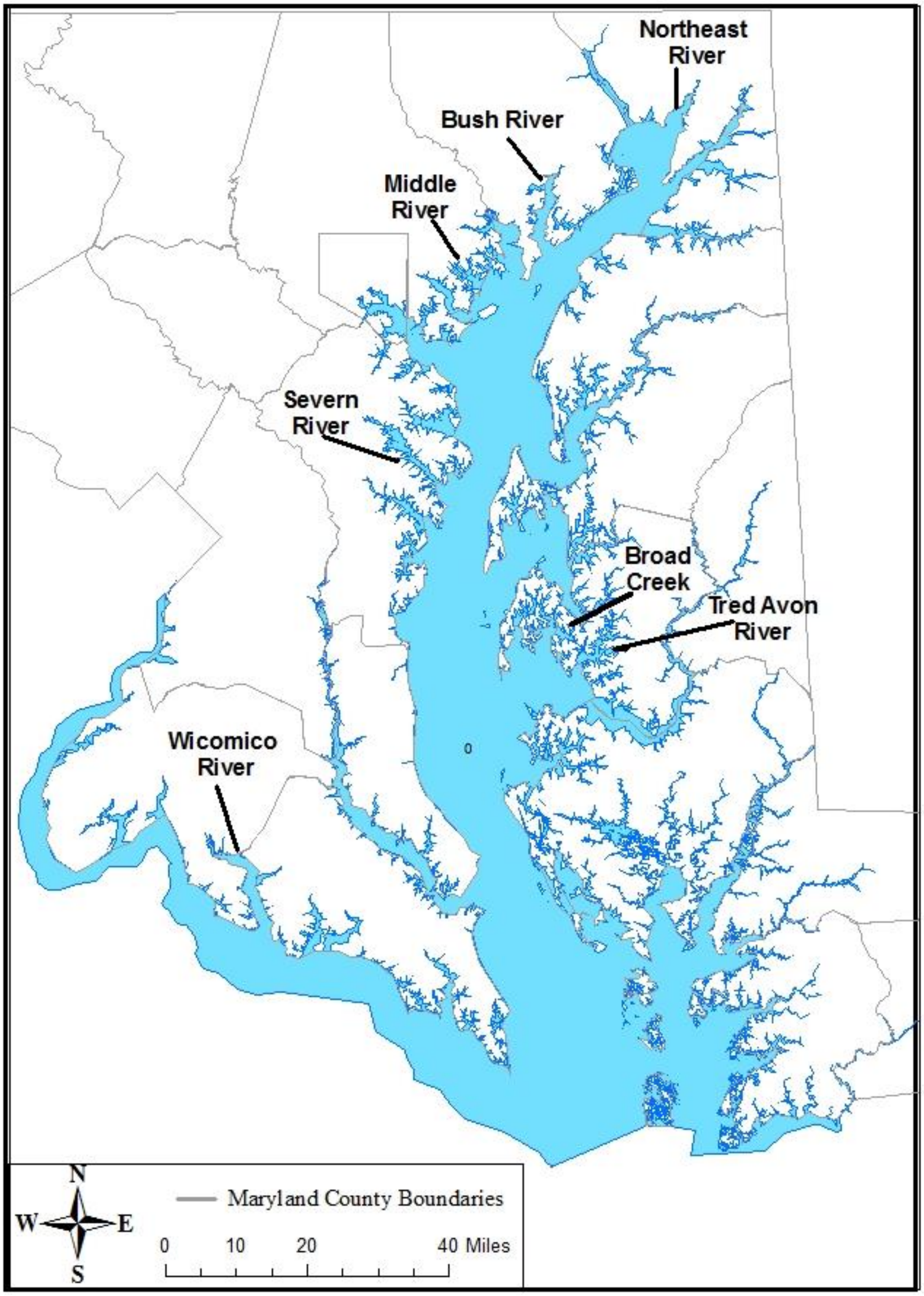


Figure 3-1. Subestuaries sampled in 2017.

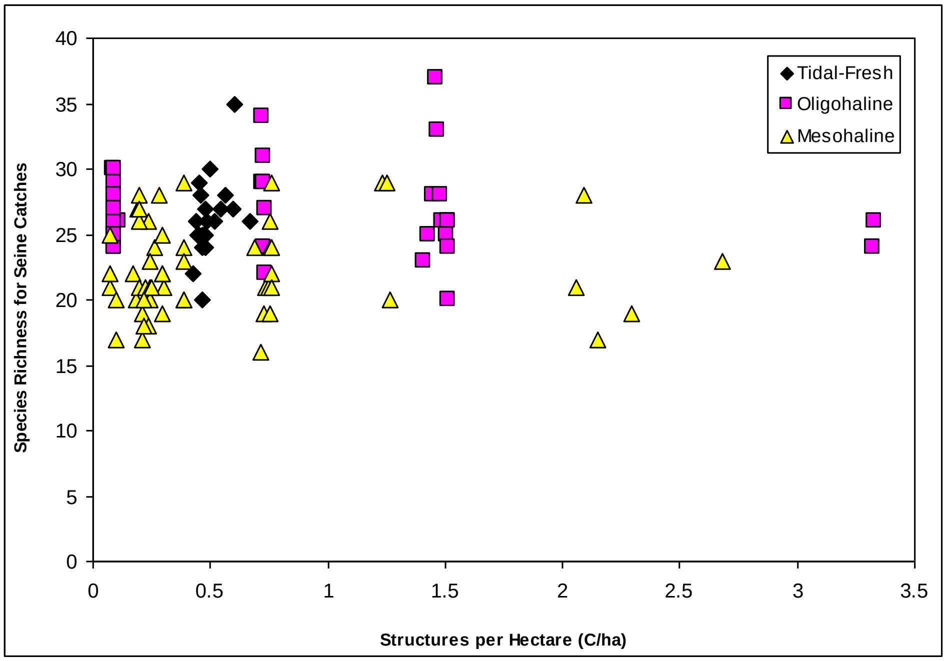


Figure 3-2. Number of finfish species collected by seining in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of watershed development (C/ha = structures per hectare). Points were omitted if seine effort <15 hauls.

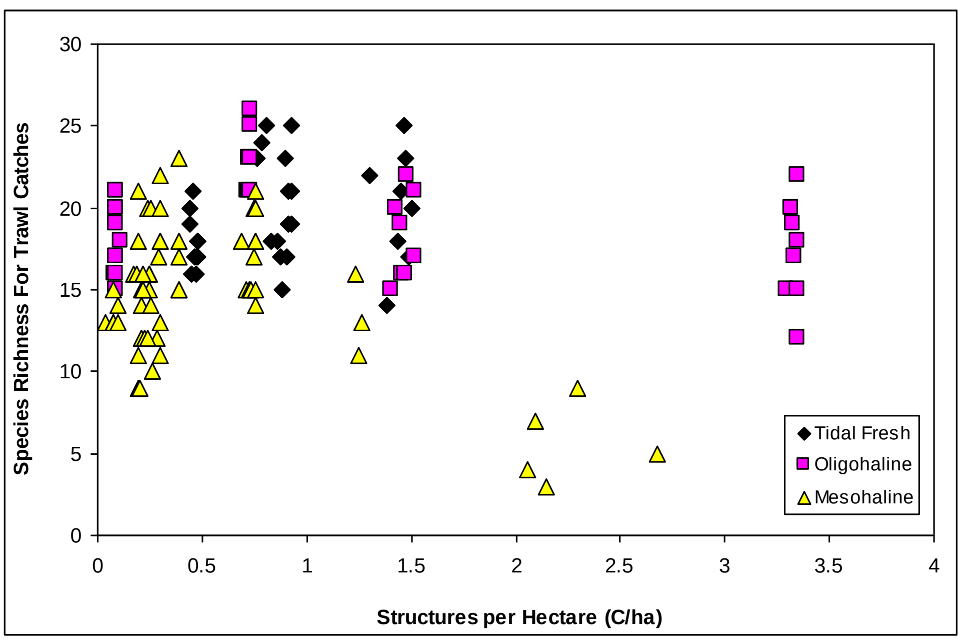


Figure 3-3. Number of finfish species collected by 4.9m trawl in tidal-fresh or oligohaline subestuaries versus intensity of development (C/ha = structures per hectare). Points were omitted if trawl effort < 15 hauls.

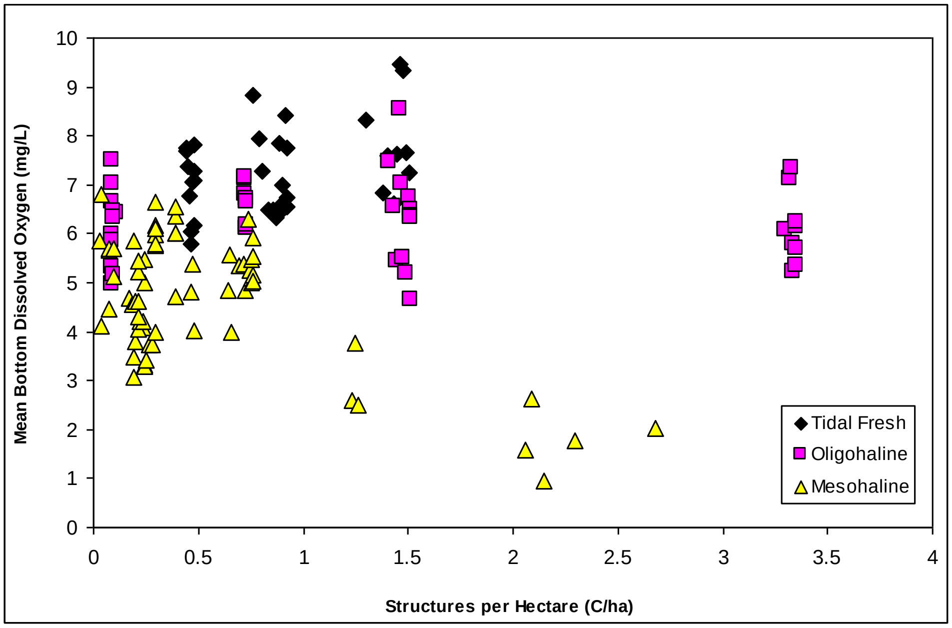


Figure 3-4. Mean subestuary bottom DO during summer sampling, 2003-2017, plotted against C/ha (structures per hectare).

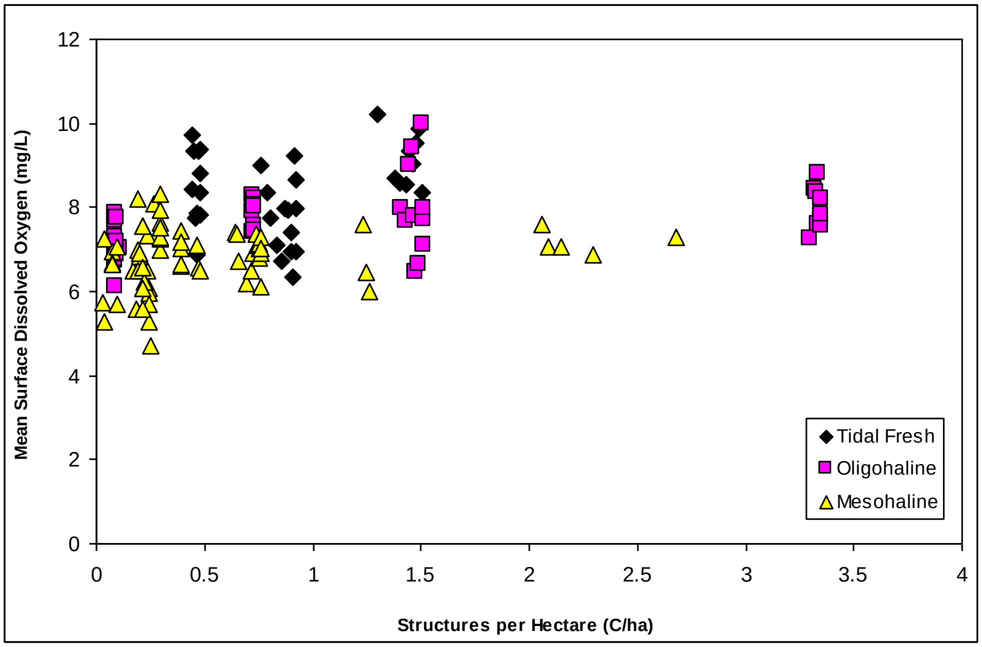


Figure 3-5. Mean subestuary surface DO during summer sampling, 2003-2017, plotted against C/ha (structures per hectare).

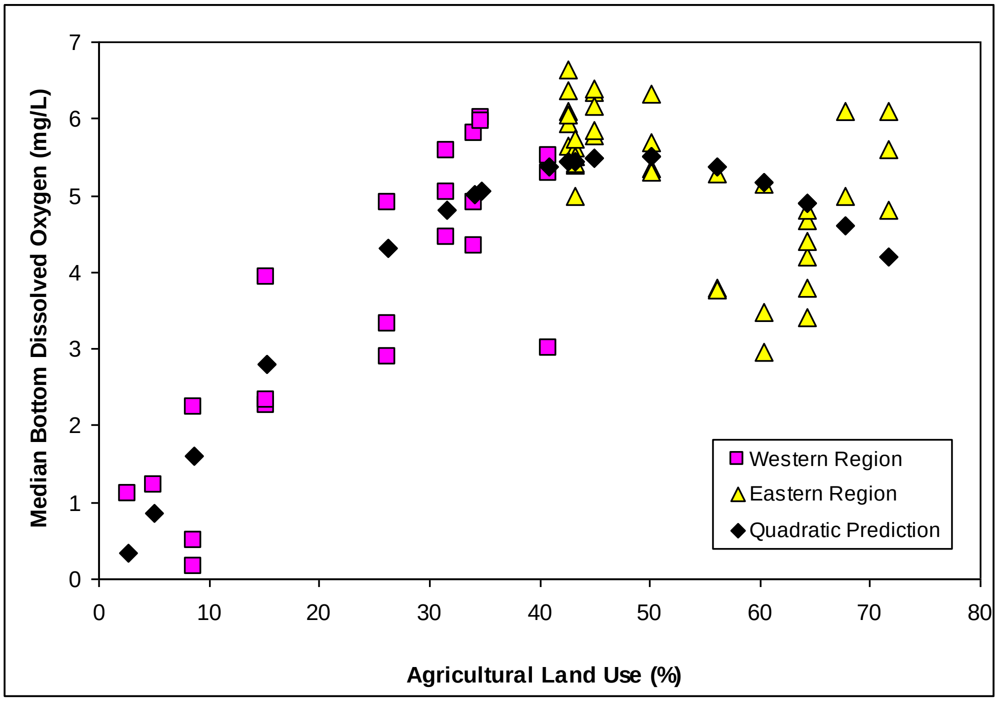


Figure 3-6. The percentage of agriculture land used by region (i.e., Western shore and Eastern shore) versus median bottom dissolved oxygen in mesohaline subestuaries (2003-2017) of the Chesapeake Bay. Quadratic model predictions of median bottom DO and agricultural coverage (%) for all data.

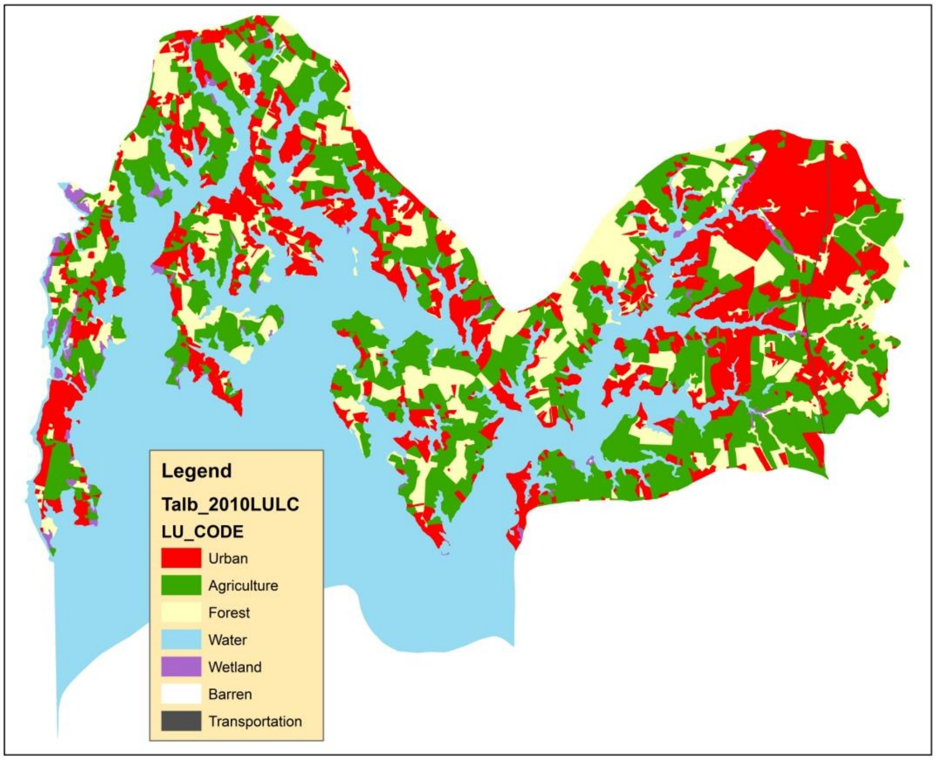


Figure 3-7. Map illustrating land use categories for the lower Choptank River subestuaries, Broad Creek and Tred Avon River.

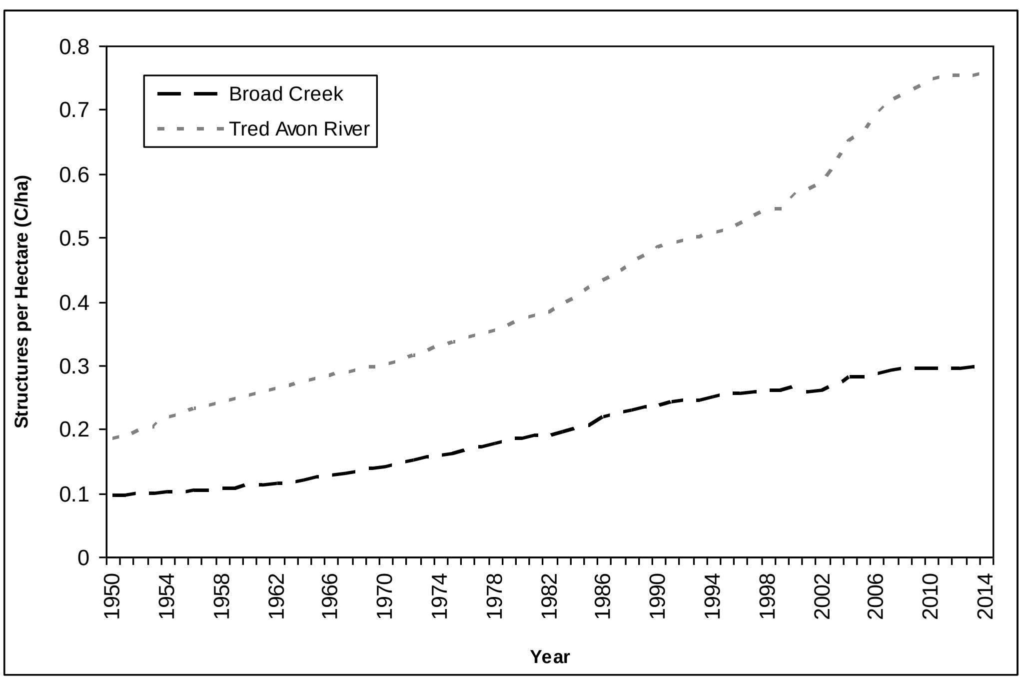


Figure 3-8. Trends in development (structures per hectare) in watersheds of two subestuaries in the Choptank River drainage, 1950-2014.

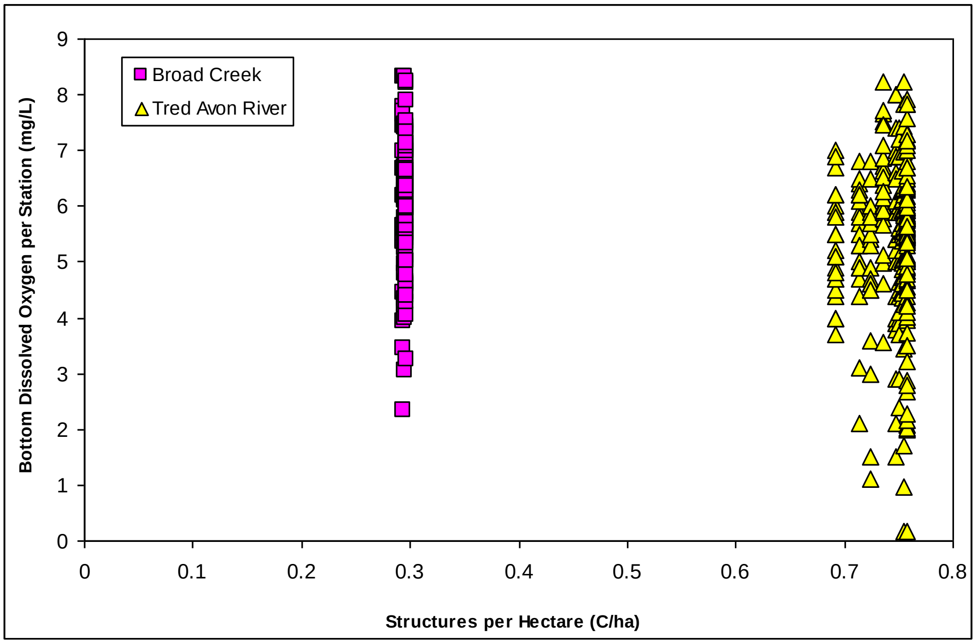


Figure 3-9. Bottom dissolved oxygen (mg/L) readings (2006-2017) versus intensity of development (C/ha = structures per hectare) in Choptank subestuaries, Broad Creek and Tred Avon River.

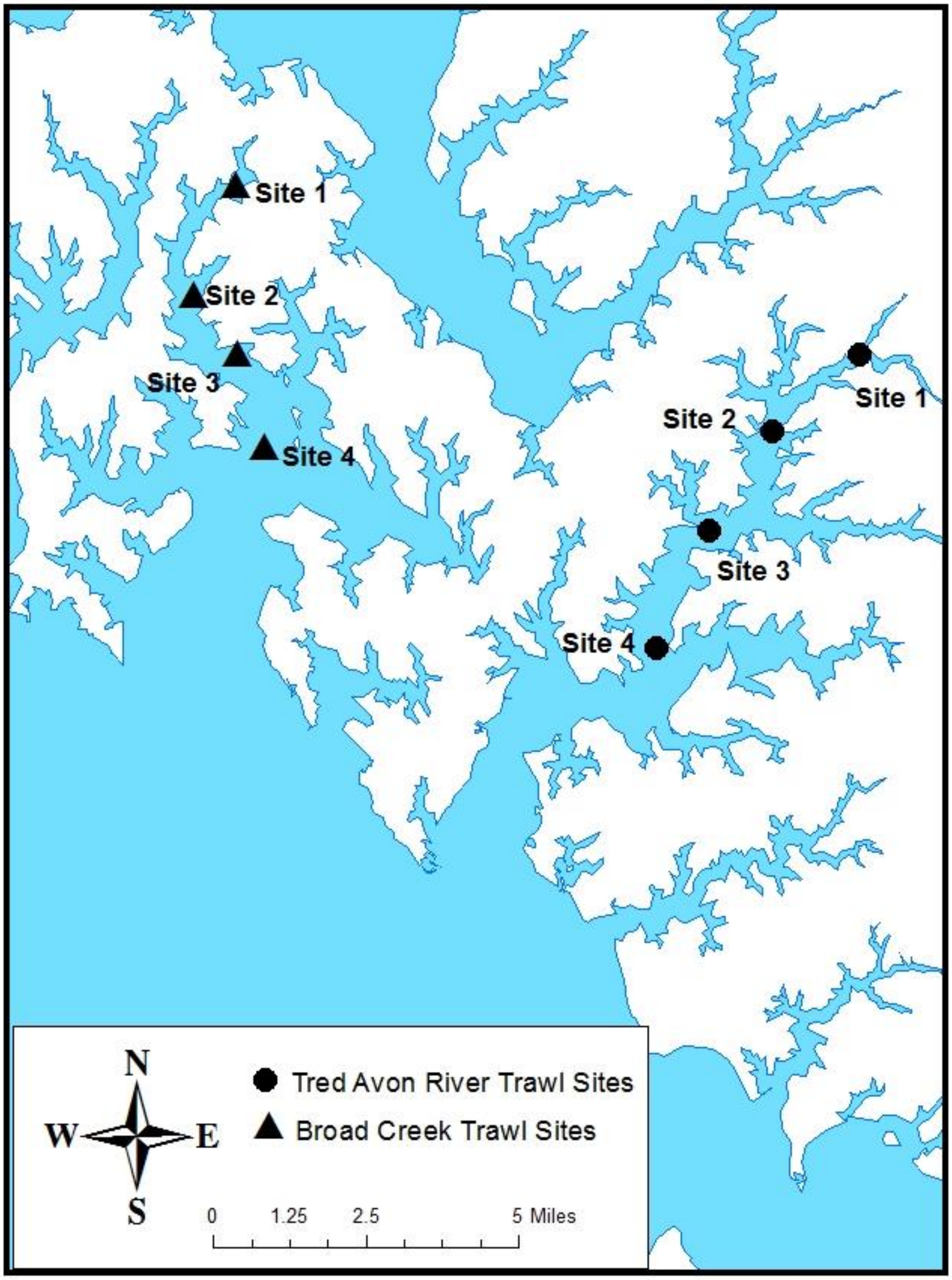


Figure 3-10. Map indicating the 2017 locations of bottom trawl sites for the lower Choptank River subestuaries, Broad Creek and Tred Avon River.

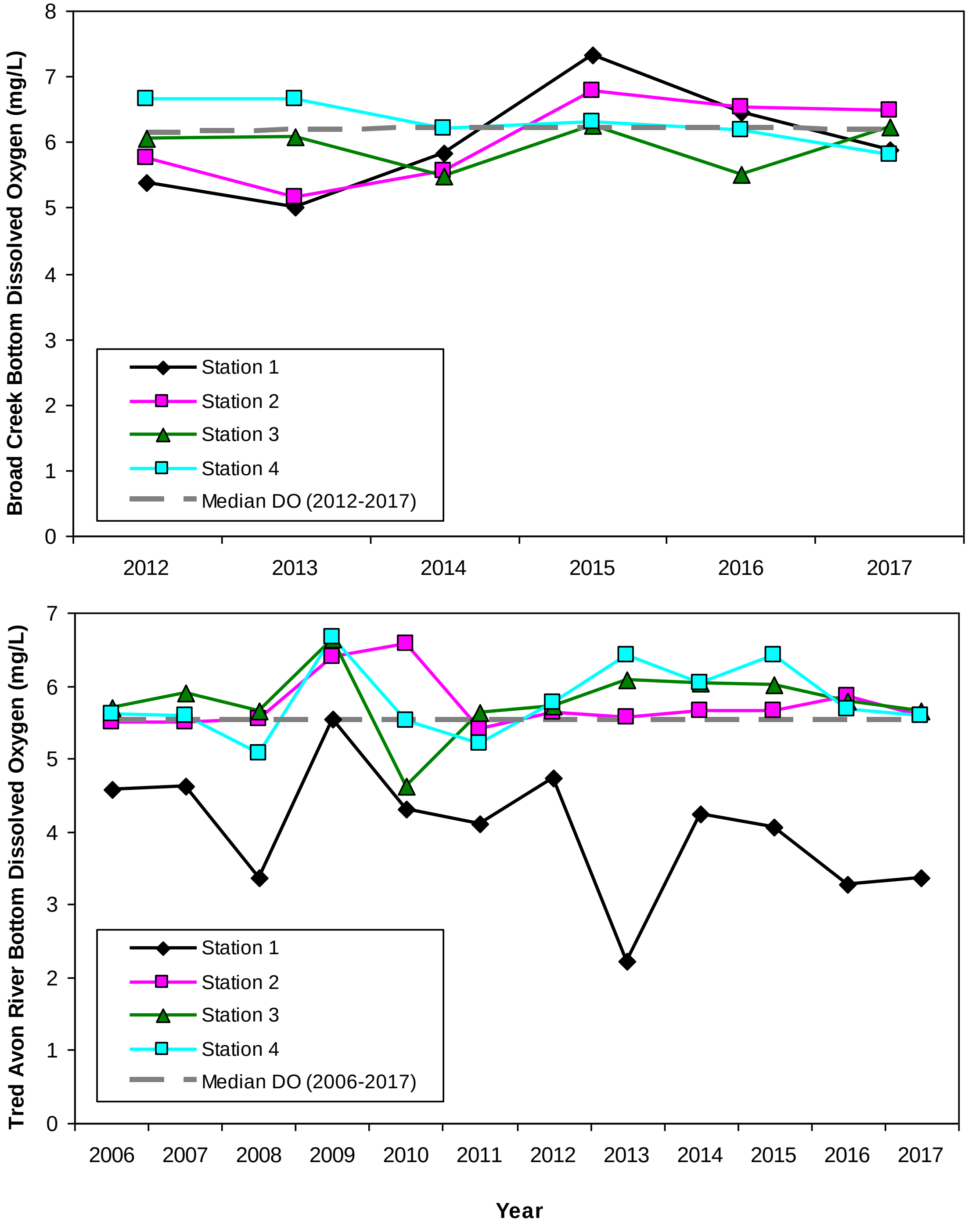
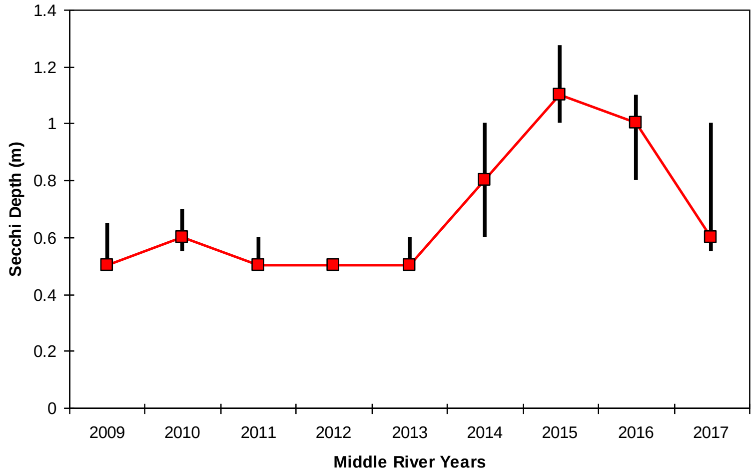
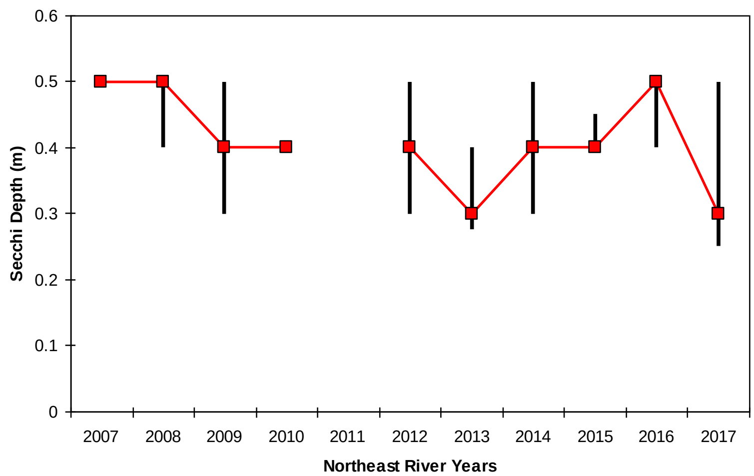
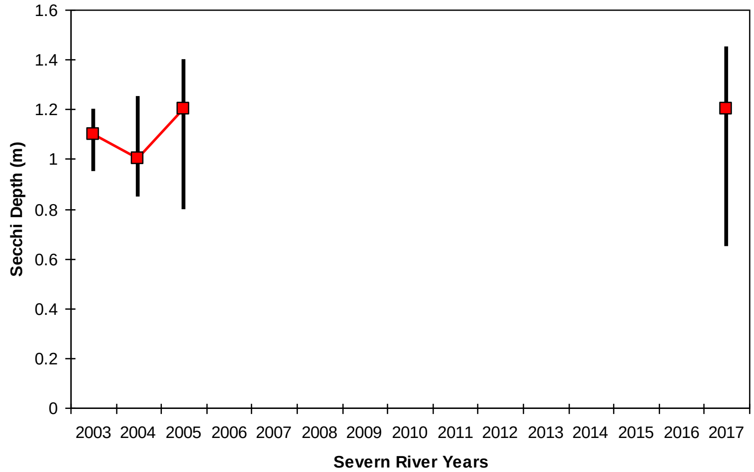


Figure 3-11. Mean bottom DO (mg/L) in Broad Creek's and Tred Avon River's subestuary by station, 2006-2017. Dotted line indicates the median for the time-series.


1.4
1.2
1
0.8
0.6
0.4
0.2
0
Secchi Depth (m)
2009
2010
2011
2012
2013
2014
2015
2016
2017
Middle River Years


0.6
0.5
0.4
0.3
0.2
0.1
0
Secchi Depth (m)
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
Northeast River Years



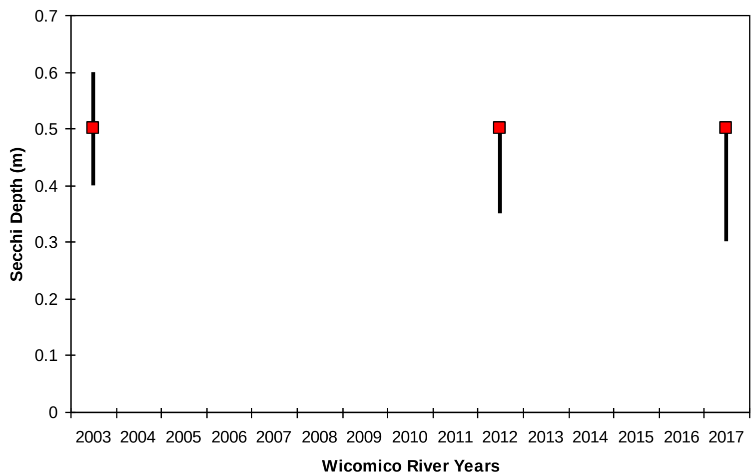
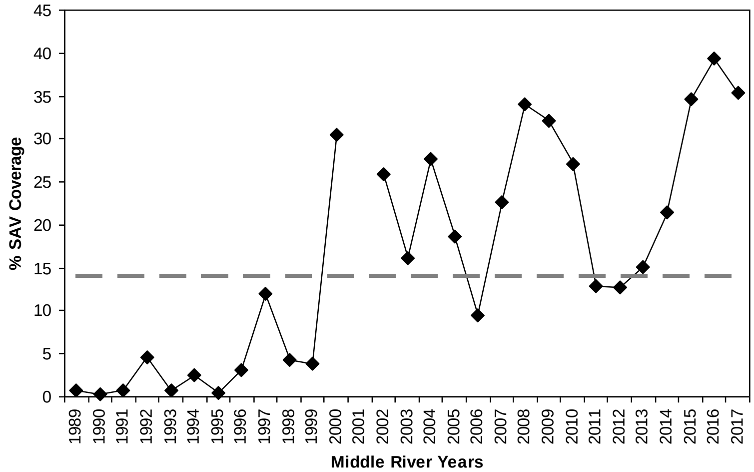
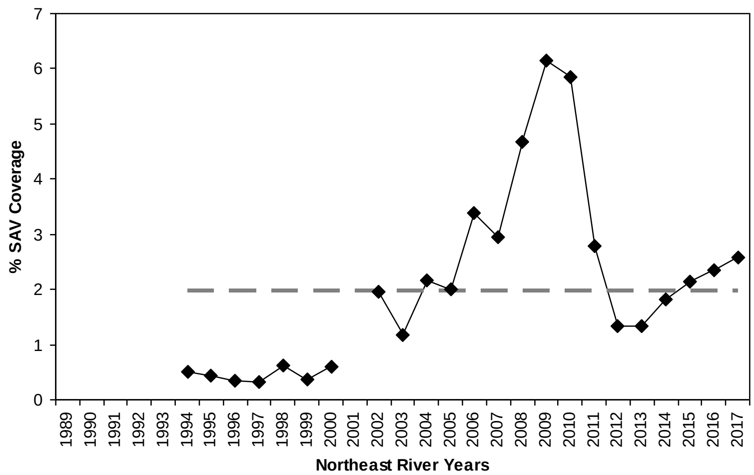
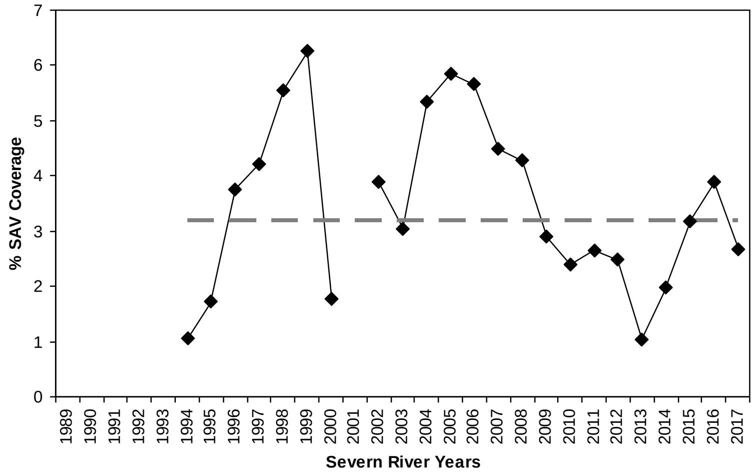


Figure 3-12. Median Secchi depth (m) by sampling year for Middle River, Northeast River, Severn River, and Wicomico River. Solid black bars indicate range of Secchi depth measurements.


45
40
35
30
25
20
15
10
5
0
% SAV Coverage
1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017
Middle River Years


7
6
5
4
3
2
1
0
% SAV Coverage
1989 1990 1991 1992 1993 1994 1995 1996 1997 1998 1999 2000 2001 2002 2003 2004 2005 2006 2007 2008 2009 2010 2011 2012 2013 2014 2015 2016 2017
Northeast River Years



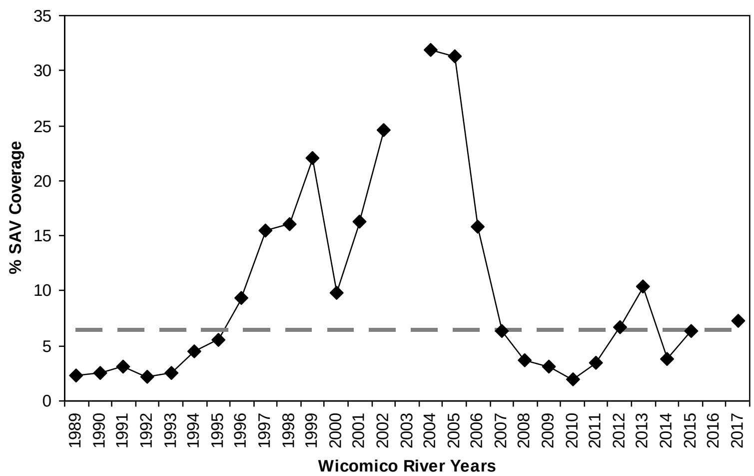
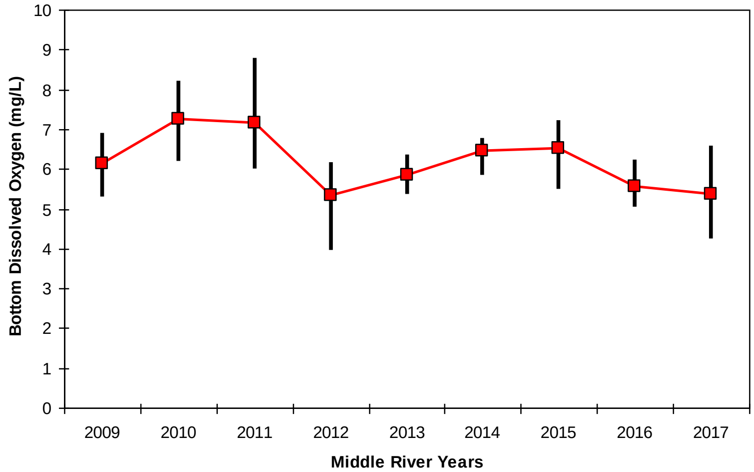
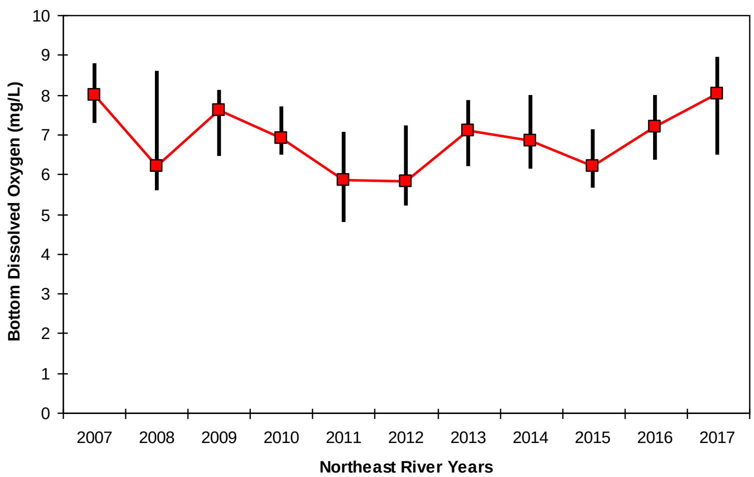


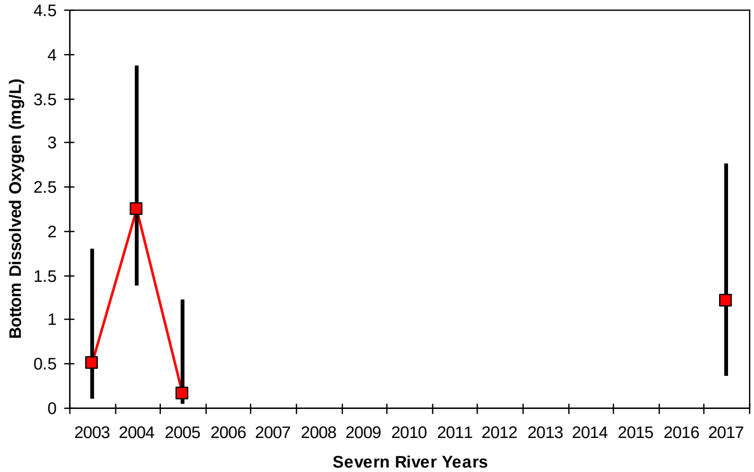
Figure 3-13. Percent of subestuaries, Middle River, Northeast River, Severn River, and Wicomico River, covered by SAV during 1989-2017 (several years were excluded due to no mapping or only partially mapping for each subestuary). Median of time-series is indicated by the dashed line.



Bottom Dissolved Oxygen (mg/L)
0
1
2
3
4
5
6
7
8
9
10
2009
2010
2011
2012
2013
2014
2015
2016
2017
Middle River Years



Bottom Dissolved Oxygen (mg/L)
0
1
2
3
4
5
6
7
8
9
10
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
Northeast River Years



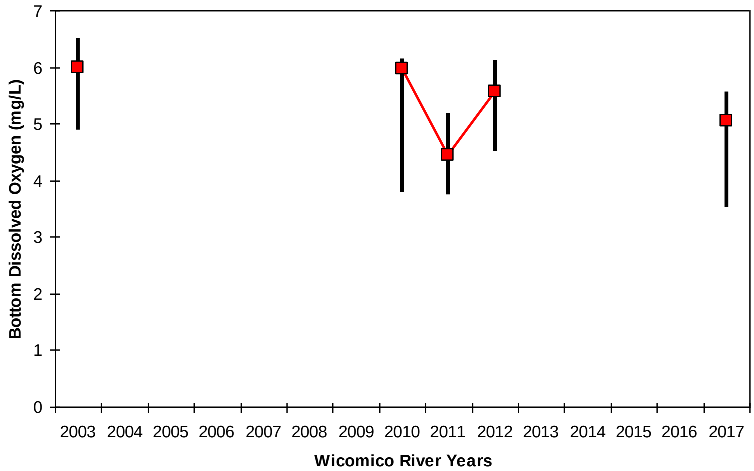
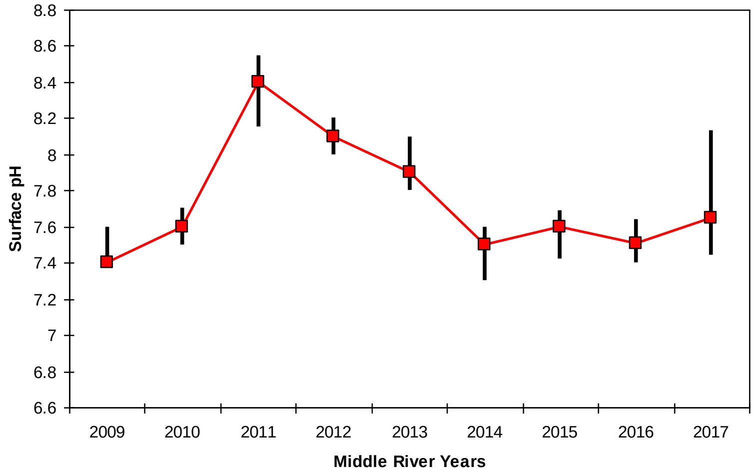
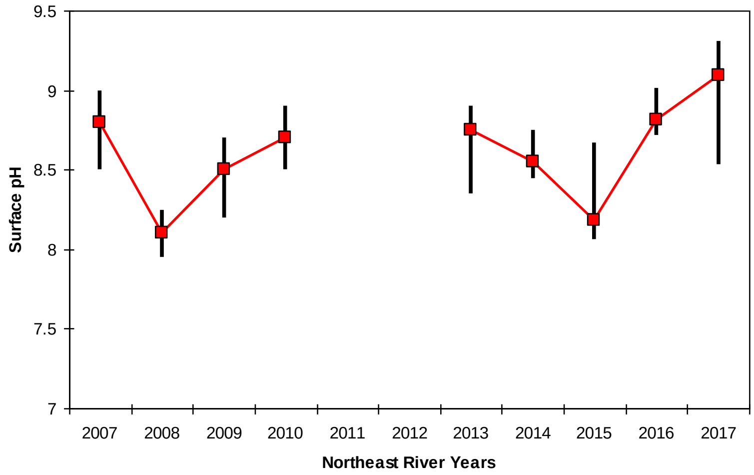
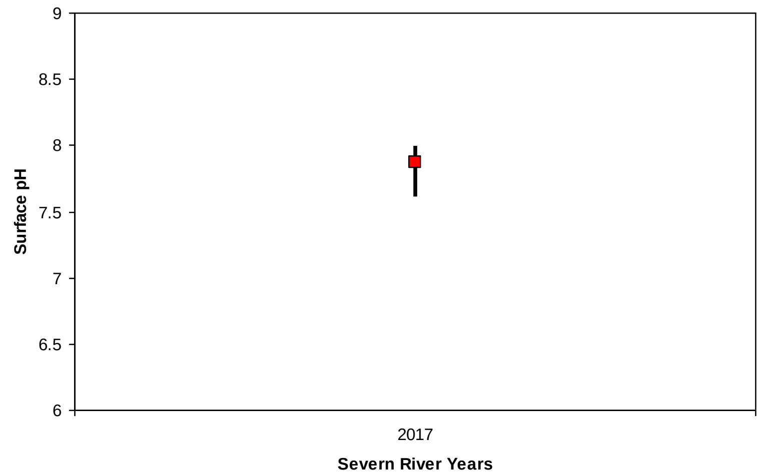


Figure 3-14. Median bottom DO (red squares and line; mg/L) by sampling year for Middle River, Northeast River, Severn River, and Wicomico River. Solid black bars indicate range of bottom DO measurements.


Surface pH
8.8
8.6
8.4
8.2
8
7.8
7.6
7.4
7.2
7
6.8
6.6
2009
2010
2011
2012
2013
2014
2015
2016
2017
Middle River Years


Surface pH
9.5
9
8.5
8
7.5
7
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
Northeast River Years



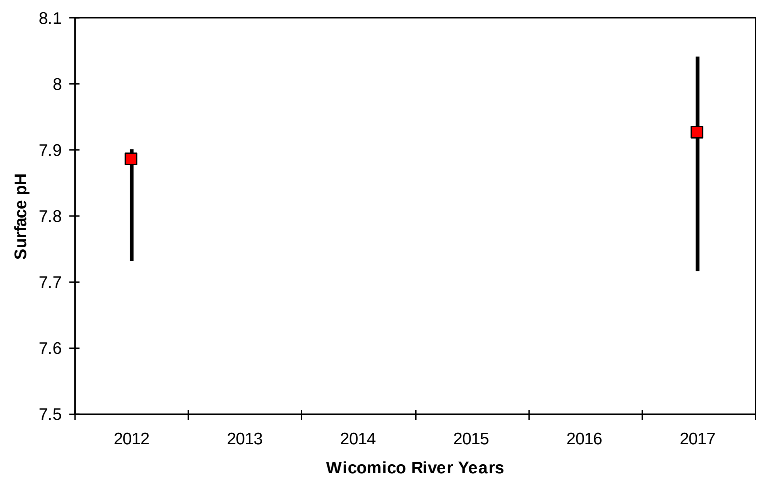
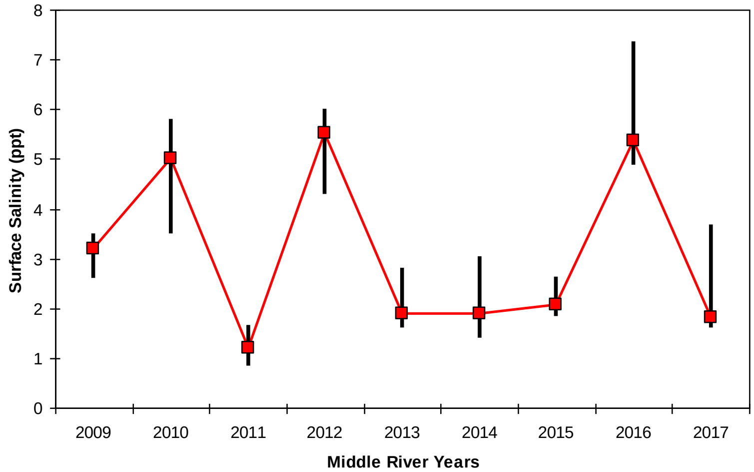
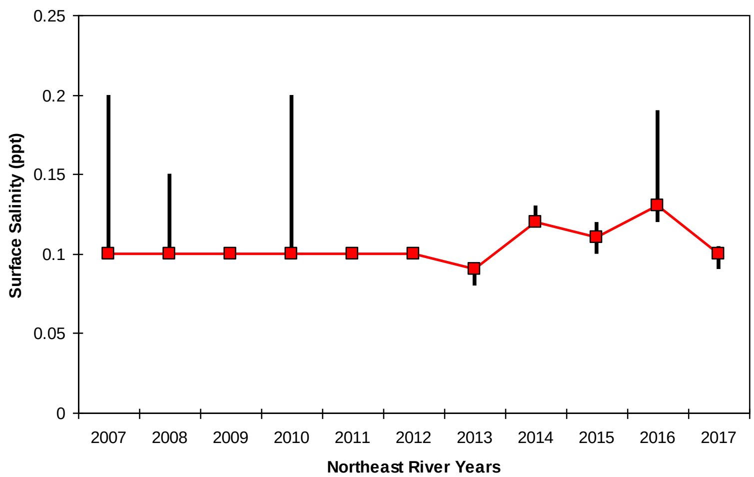


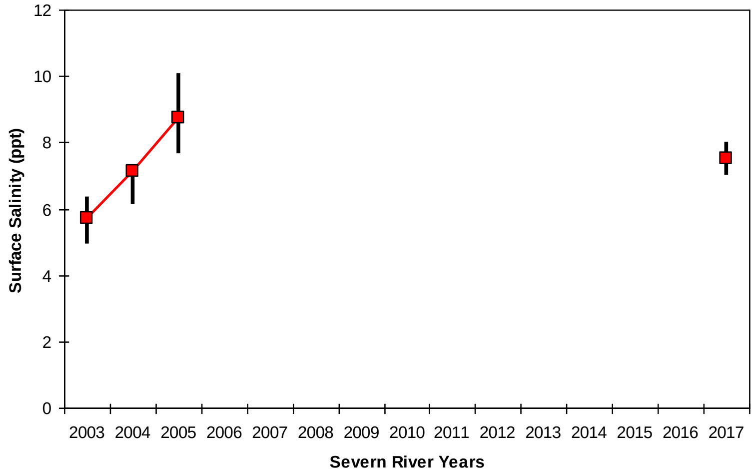
Figure 3-15. Middle River, Northeast River, Severn River, and Wicomico River median surface pH (red squares and line) by sampling year. Solid black bars indicate range of pH measurements.



8
7
6
5
4
3
2
1
0
Surface Salinity (ppt)
2009
2010
2011
2012
2013
2014
2015
2016
2017
Middle River Years



0.25
0.2
0.15
0.1
0.05
0
Surface Salinity (ppt)
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
Northeast River Years



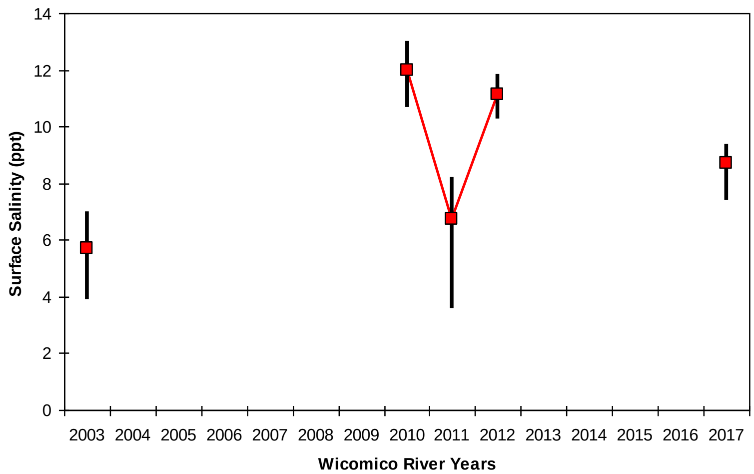
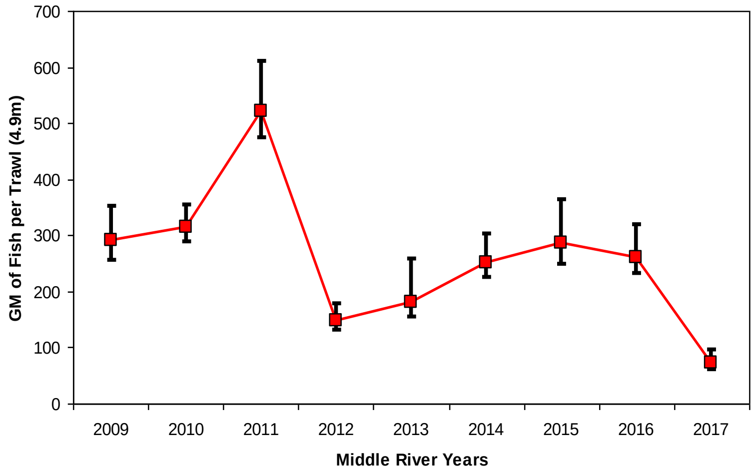
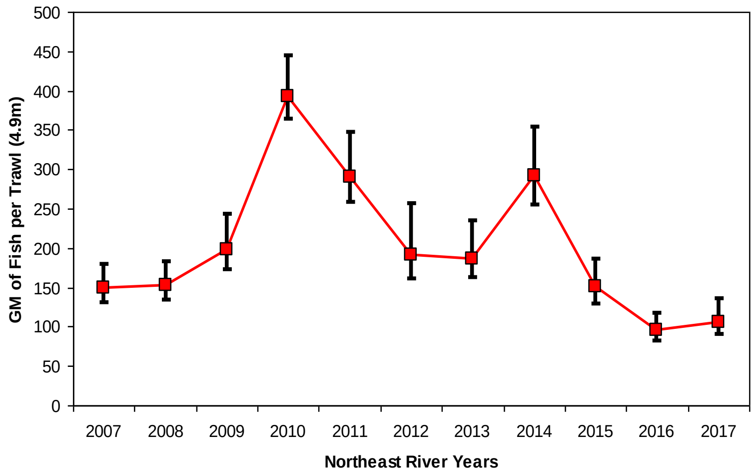


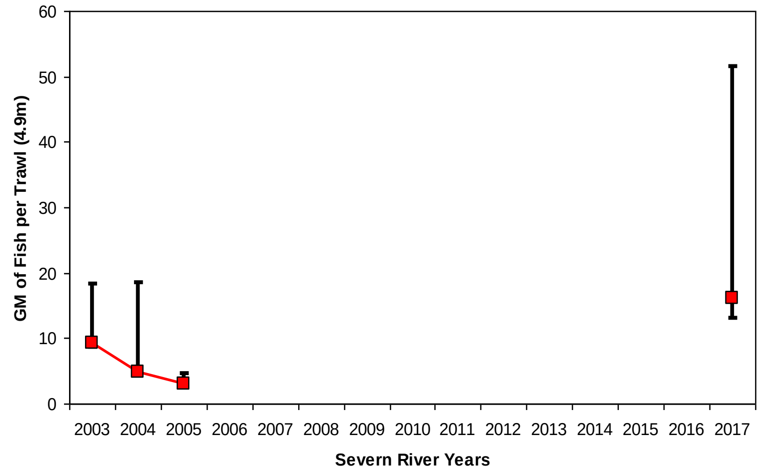
Figure 3-16. Median surface salinity (red squares and line; ppt = ‰) by year for Middle River, Northeast River, Severn River, and Wicomico River by sampling year. Solid black bars indicate range of salinity measurements.



700
600
500
400
300
200
100
0
GM of Fish per Trawl (4.9m)
2009
2010
2011
2012
2013
2014
2015
2016
2017
Middle River Years



500
450
400
350
300
250
200
150
100
50
0
GM of Fish per Trawl (4.9m)
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
Northeast River Years



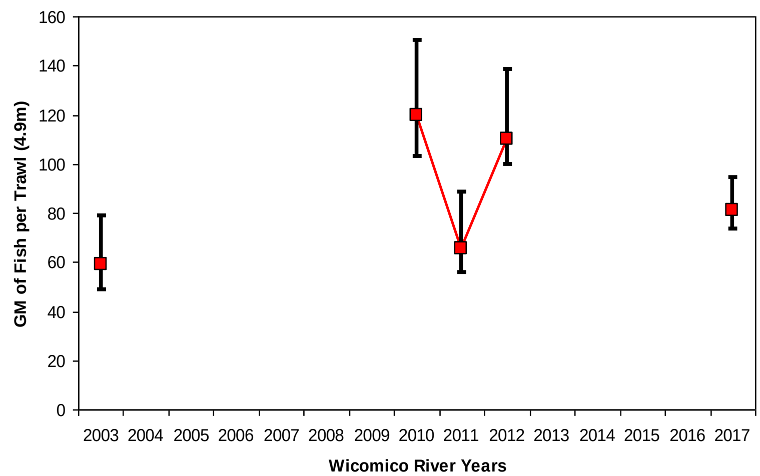
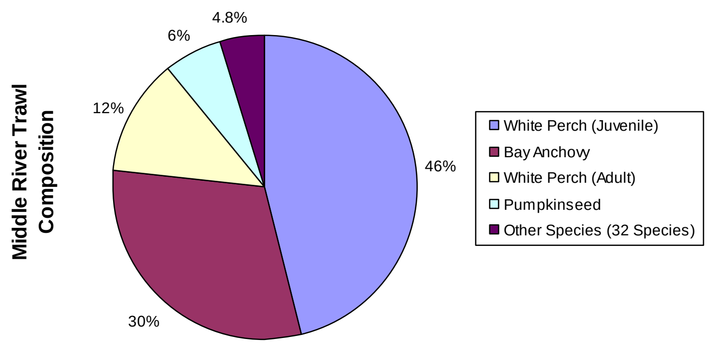
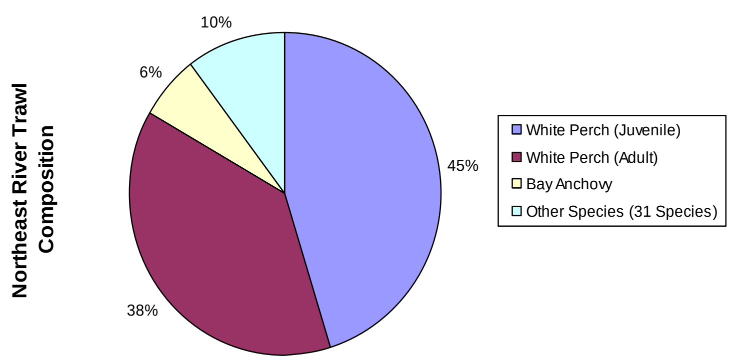
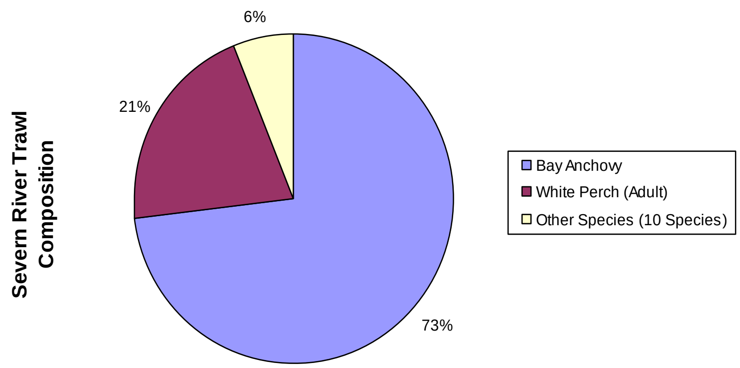


Figure 3-17. Annual geometric mean catches per 4.9m trawl of all species of finfish (GM; red squares and line) in Middle River, Northeast River, Severn River, and Wicomico River by sampling year. Black bars indicate the 95% confidence intervals.


Middle River Trawl Composition
4.8%
6%
12%
46%
30%
White Perch (Juvenile)
Bay Anchovy
White Perch (Adult)
Pumpkinseed
Other Species (32 Species)


Northeast River Trawl Composition
10%
6%
45%
38%
White Perch (Juvenile)
White Perch (Adult)
Bay Anchovy
Other Species (31 Species)



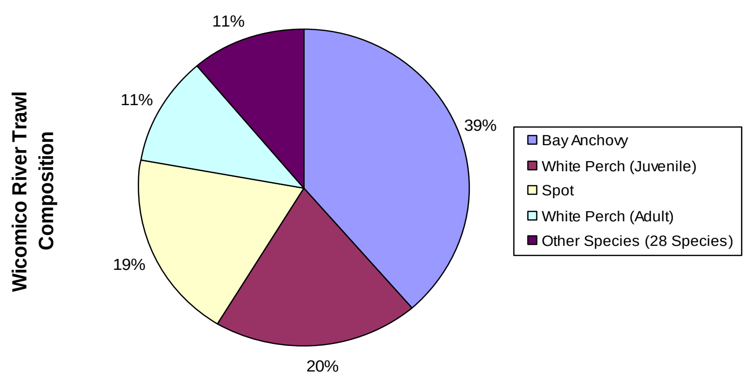
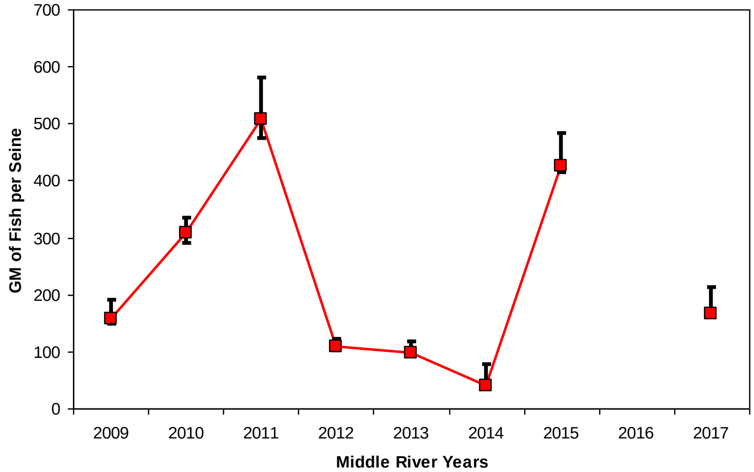
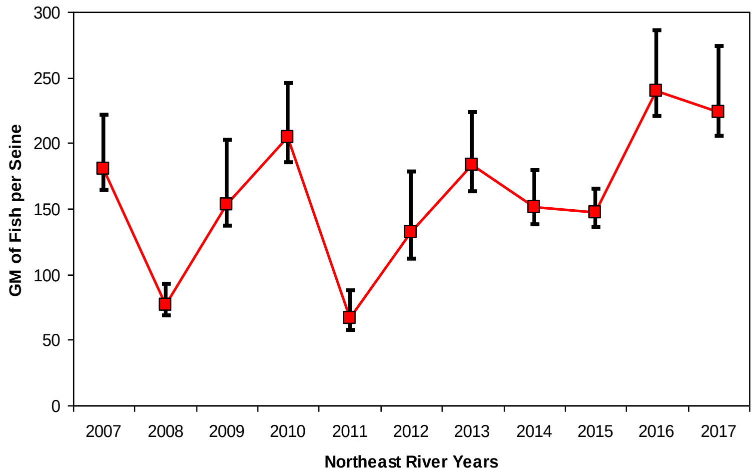
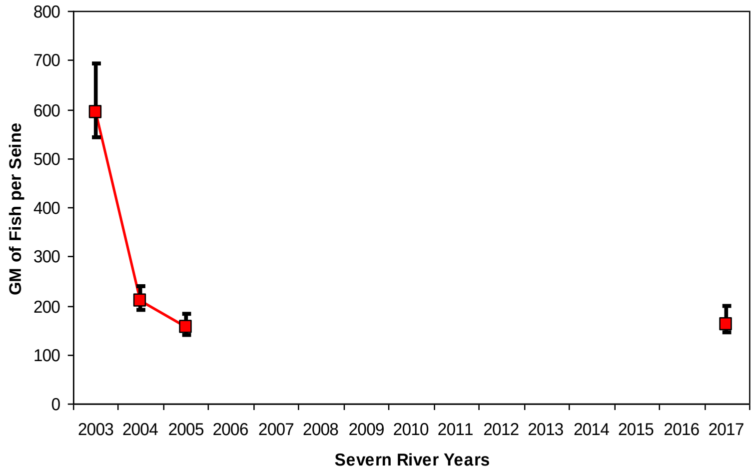


Figure 3-18. Species composition for trawl catches of all species of finfish in Middle River (2009-2017), Northeast River (2007-2017), Severn River (2003-2005, 2017), and Wicomico River (2003, 2010-2012, 2017) during sampling years. Species that define the top 90% are identified and the remainder of species are grouped and labeled as “other species”.


700
600
500
400
300
200
100
0
GM of Fish per Seine
2009
2010
2011
2012
2013
2014
2015
2016
2017
Middle River Years


300
250
200
150
100
50
0
GM of Fish per Seine
2007
2008
2009
2010
2011
2012
2013
2014
2015
2016
2017
Northeast River Years



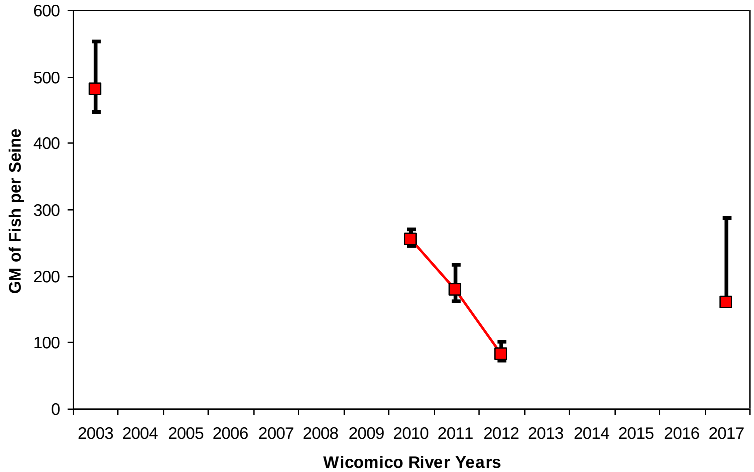
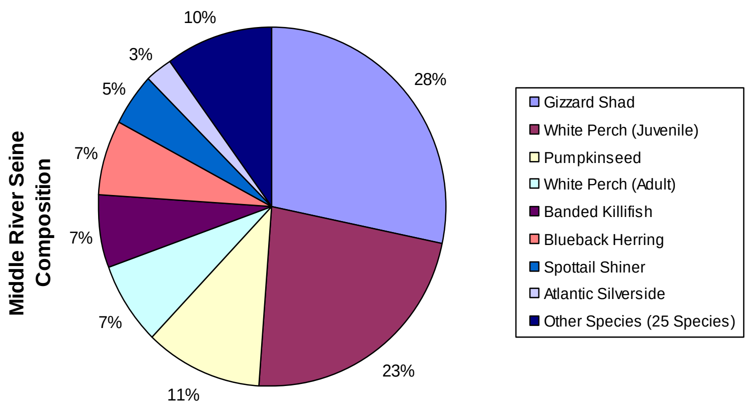
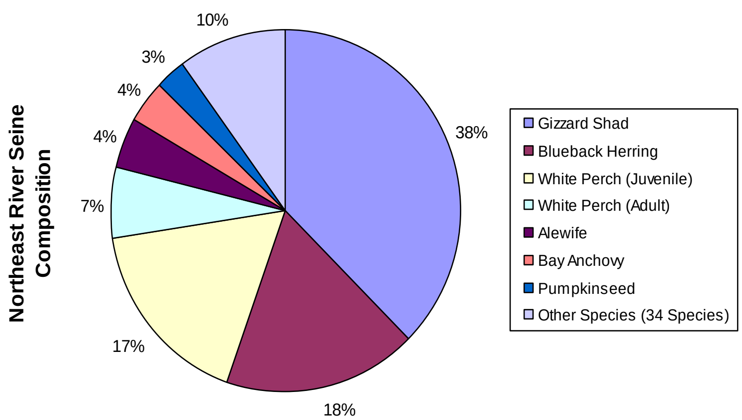
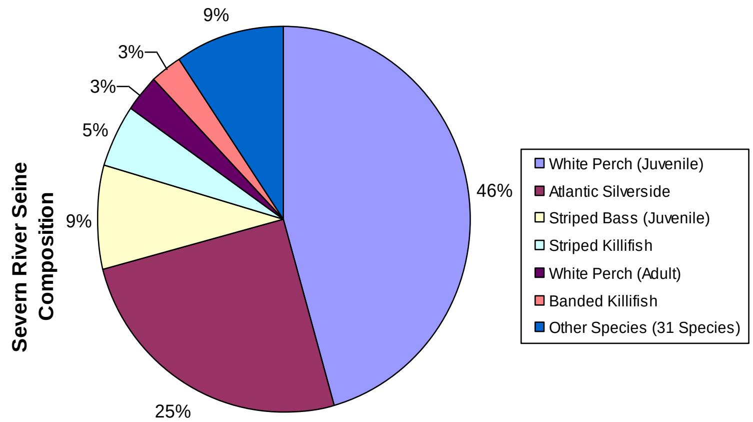


Figure 3-19. Annual geometric mean catches per seine of all species of finfish (GM; red squares and line) in Middle River, Northeast River, Severn River, and Wicomico River by sampling year. Black bars indicate the 95% confidence intervals.


Middle River Seine Composition
10%
3%
5%
7%
7%
7%
11%
23%
28%
Gizzard Shad
White Perch (Juvenile)
Pumpkinseed
White Perch (Adult)
Banded Killifish
Blueback Herring
Spottail Shiner
Atlantic Silverside
Other Species (25 Species)


Northeast River Seine Composition
10%
3%
4%
4%
7%
17%
18%
38%
Gizzard Shad
Blueback Herring
White Perch (Juvenile)
White Perch (Adult)
Alewife
Bay Anchovy
Pumpkinseed
Other Species (34 Species)



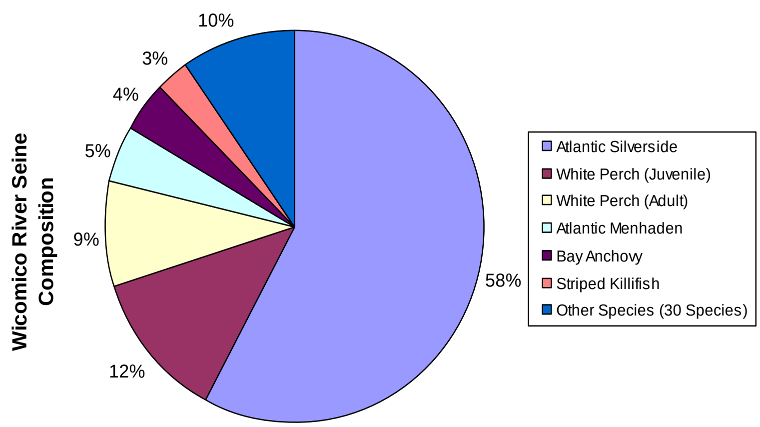


Figure 3-20. Species composition for seine catches of all species of finfish in Middle River (2009-2015, 2017), Northeast River (2007-2017), Severn River (2003-2005, 2017), and Wicomico River (2003, 2010-2012, 2017) during sampling years. Species that define the top 90% are identified and the remainder of species are grouped and labeled as “other species”.

JOB 2: Support multi-agency efforts to assess finfish habitat and ecosystem-based fisheries management.

Jim Uphoff, Margaret McGinty, Alexis Park, Carrie Hoover, and Benjamin Wahle

Introduction

The objective of Job 2 was to document participation of the Fisheries Habitat and Ecosystem Program (FHEP) in habitat, multispecies, and ecosystem-based management approaches important to recreationally important finfish in Maryland's Chesapeake Bay and Atlantic coast. Activities in this job used information generated by F-63 in communication and fisheries management or were consistent with the goals of F-63.
Contributions to various research and management forums by Program staff through data collection and compilation, analysis, and expertise are vital if Maryland is to successfully develop an ecosystem approach to fisheries management.

Fisheries Habitat and Ecosystem Program Website – We continued to populate the website with new reports to keep it up to date with project developments. The web site was redesigned in April 2015 to help with navigation. Currently, we are working on compiling reports, maps, and presentations to add to the FHEP website.

Publications – Uphoff, J. H., and A. Sharov. 2018. Striped Bass and Atlantic Menhaden predator–prey dynamics: model choice makes the difference. Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science 10:370-385.

Environmental Review Unit Bibliography Database – We maintain an Environmental Review Unit database, adding additional literature when it becomes available.

Review of County Comprehensive Growth Plans – We reviewed and commented on three county and two municipal comprehensive growth plans (Harford, Kent, and Frederick Counties; Town of Manchester and Ocean City Municipalities), providing recommendations consistent with maintaining viable fish habitat. These efforts included an assessment of local fisheries resources that represent recreational opportunities and the importance to consider fish habitat protection in planning. We continue to meet with Queen Anne’s County planning staff to highlight the importance of fishing in the county and offer assistance to incorporate fish habitat needs in future planning activities.

Cooperative Research – M. McGinty, A. Park, C. Hoover, and B. Wahle supported field sampling efforts of various state and federal projects including: the DNR’s Coastal Bays Program, Resident Fish Species, Fish Passage Program, the Alosid Project, Oyster Program, Resource Assessment Services, Artificial Reefs Initiative, Hatcheries Division Program, Inland Fisheries Division, Striped Bass Program, APAIS, and the Fish Health Program.

J. Uphoff, A. Park, B. Wahle collaborated with the Fish Health Program at the Oxford Lab assessing Striped bass stomach contents collected from the upper, middle, and lower Chesapeake Bay.

A. Park assisted the Coastal Bays Program with the highly migratory species tagging and data collection at the Mid-Atlantic and the White Marlin Fishing Tournaments held in Ocean City, Maryland.

Presentations and Outreach – A. Park presented on 2016 Bush River summer sampling results and illustrated what summer juvenile fish sampling involved for the Anita C. Leight Estuary Center staff and volunteers. The Bush River is one of FHEP’s sampling areas and has been sampled since 2006 by staff and volunteers. The volunteer group samples the Bush River and provides data to FHEP staff.

M. McGinty conducted a seining demonstration at a community youth summer camp in Round Bay and discussed the habitat needs of the fish species present.

M. McGinty presented at the quarterly Planners Meeting of the Critical Areas Commission held in Annapolis.

J. Uphoff presented a paper on the roles of predation and fishing on the biomass dynamics of Atlantic Menhaden at the National AFS Conference in Tampa, Florida.

M. McGinty attended a meeting at the Critical Areas Commission to discuss mapping Habitat Protected Areas, habitat changes observed in urban environments, and suggested a mapping approach that considers ecosystem services in relation to ecosystem regimes would be a positive approach to help the public understand the value of conserving key ecological resources, while establishing realistic expectations for ecological services.

M. McGinty and J. Uphoff presented information on development's effect on Chesapeake Bay fish habitat to a delegation of fisheries scientists from Yunnan Province, China.

J. Uphoff presented "Menhaden 101' to interested citizens at Chesapeake College and at Horn Point Laboratory. This presentation covered basic life history and management.

J. Uphoff, M. McGinty, A. Park, and C. Hoover participated in several webinars hosted by AFS and NOAA concerning fish passage, overfishing, and conversing with the public about conservation.

M. McGinty participated in the steering committee to develop the Scientific and Technical Advisory Committee (STAC) Fish Habitat Workshop.

M. McGinty contributed to the Fish Habitat Workshop and is contributing to the final report.

M. McGinty participated in a stream walk with the Magothy River Association to learn about their plans to attempt to restore Yellow Perch in the Magothy River.

M. McGinty continues to serve on the Chesapeake Bay Program Fish Habitat Action Team.

J. Uphoff and M. McGinty presented on the relationship of agriculture and fisheries at the Maryland Water Monitoring Council conference.

C. Hoover, J. Uphoff, M. McGinty, and A. Park presented a poster at the Maryland Water Monitoring Council conference on whether RNA/DNA of yellow perch larvae could indicate effects of development.

A. Park submitted a narrated presentation about the FHEP program for the Fishing and Boating Annual Summit.

A. Park and C. Hoover attended a Chesapeake Bay River Herring Workshop to discuss past/future regional work and data findings.

Park and Hoover attended the fisheries index standardization class using R.

Uphoff and McGinty developed an analysis of Patuxent River anadromous fish recruitment that indicated that spawning and larval nursery habitat of anadromous Herring has deteriorated to the point that recruitment has declined. This analysis was presented to Fisheries and other DNR leadership and there was agreement to move this issue forward to the Patuxent River Commission as a next step. A copy of the briefing document can be found in Appendix 1 of Job 2.

ASMFC - J. Uphoff participated in an ASMFC conference call with Dr. Ray Hilborn regarding his team's journal paper on the flaws of the Lenfest forage reference points.

J. Uphoff commented on forage reference points that will be sent to the ASMFC's Atlantic Menhaden management board.

J. Uphoff participated in the ASMFC workgroup developing forage reference points for Atlantic Menhaden. J. Uphoff has developed a biomass dynamic (Steele-Henderson) predatory Prey model for Menhaden that is one of the models being considered for management.

A. Park participated in the ASFMC Shad and River Herring meeting in Baltimore to observe and determine if FHEP can provide and help with any future data requests.

Chesapeake Bay Agreement – J. Uphoff presented an update on progress on the Maryland oriented forage fish indicator to DNR's Bay Policy group and participated in meetings of the Forage Action Team.

Envison the Choptank – J. Uphoff participated in an Envision the Choptank meeting on fisheries and habitat in the Choptank River. The Chesapeake Conservancy is working with NOAA to facilitate this collaborative initiative. The initiative is bringing together nonprofits, government agencies, scientists, and community groups with the purpose of identifying collaborative solutions that will provide a swimmable, fishable Choptank and enhance the health and productivity of native oysters.

Job 2, Appendix 1.

Briefing Document: Status of Anadromous Fish Spawning Habitat in Patuxent River
Fish Habitat and Ecosystem Program
February, 2018

Subject of concern –Year-class success of anadromous Alewife, Blueback Herring, and American Shad (collectively, "Herring") has declined over the past decade as Patuxent River's watershed has transformed into a suburban landscape.

Area of concern – The portion of the river containing the anadromous fish spawning and Striped Bass spawning areas (from Laurel to several miles upstream of the Benedict Bridge; Figure 1). The Patuxent River drainage is entirely in Maryland and drains the Piedmont and Coastal Plain regions. Anadromous fish and Striped Bass spawning and larval nurseries are located in the Coastal Plain. Two WSSC reservoirs are located in the Piedmont. The non-tidal boundary is at the Route 214 Bridge. Location of the Striped Bass spawning area was determined in the 1950s and the spawning and nursery area of other anadromous fish was determined in the early 1980s (Figure 1).

Water quality has been continuously monitored at eight stations in the Patuxent River watershed (Figure 1). Stations at Unity (near Tridelphia Reservoir), Rocky Gorge, and the USGS gauging station (at Bowie) provide data on the non-tidal portion of the river. Five additional stations in the tidal-fresh region (TF1.3 to TF1.7) of the area of concern are spread from above the Route 4 Bridge to just above Benedict (Figure 1).

Development has surpassed the impervious surface threshold for anadromous fish eggs and larvae – Impervious surface is an indicator of development that reflects cumulative ecological deterioration of a watershed. Development increases flow extremes (lower lows and more flooding), erosion, and sediment that negatively alters channel depth and width, substrate characteristics and cover for fish. As trees are lost, runoff temperature increases. Nutrients from developed lands occur at levels as high as from agriculture and cause algae blooms that deplete oxygen. In winter, more roads require more salt that pollutes streams and kills freshwater organisms, including fish. Other pollutants such as toxic metals (lead for example) and organic pollutants (oil, grease, and pesticides) enter waterways in urban runoff and wastewater. Compounds, such as pharmaceuticals and personal hygiene products, may not be removed by wastewater treatment facilities and septic systems; some alter hormones that are important for successful sperm and egg development (endocrine disruptors). Food webs may become altered. Fish become less abundant, diverse, and safe to eat in polluted waters resulting from high development.

The Fish Habitat and Ecosystem Program (http://dnr.maryland.gov/fisheries/Pages/FHEP/index.aspx) estimated impervious surface targets and limits for watershed development to judge whether watersheds will be able to provide healthy habitat for fish sought by Chesapeake Bay fisheries. Rural watersheds with 5% or less impervious surface support productive fisheries, while suburban watersheds at 10% or more are increasingly likely to develop serious problems.

Impervious surface in the Patuxent River Watershed above the Striped Bass spawning area was 1.5% in 1950, rising to 12.5% by 2014 (latest estimate; Figure 2). Impervious surface increased fastest between 1962 (2.8% impervious surface) and 2004 (11.3% impervious surface) and then slowed. The 5% impervious surface target for keeping a watershed rural was exceeded in 1972. The 10% threshold for a suburban watershed was breached in 1997 (Figure 2). Plotting cumulative percent impervious surface for the watershed draining water quality stations from Unity (furthest upstream) to TF 1.7 (near the lower boundary of the Striped Bass spawning and larval nursery) during 1977-2014 (this time period will match water quality time-series presented later) provides an indication of the pattern of development (Figure 3). The portion of the watershed draining into Tridelphia Reservoir has remained rural (from 2.5% impervious surface in 1977 to 4.2% in 2014), but has moved above the impervious surface target at Rocky Gorge (from 3.8% to 7.0%). The greatest change occurred as the river passed through in the Laurel-Bowie area; watershed impervious surface jumped to 13.6% in 2014 for the portion above the Bowie gauging station (it was 7.0% in 1977). Percent impervious surface becomes diluted somewhat with the addition of less developed land associated with the watershed draining TF 1.3 and TF 1.7 (Figure 3).

Maryland Department of Planning has estimated land use across Maryland in 1973, 1994, 1997, 2002, and 2010 (Table 1, below). Agriculture was the major human-related land use in the watershed in 1973 and 1994, but was surpassed by urban land in subsequent surveys. Forest (includes tree cover on residential land) was the overall largest land use through 2002, but was exceeded by urban use in 2010. Although wetlands occupy a small fraction of the Patuxent River watershed, their area is relatively high for a developed watershed. Wetlands are important sources of organic matter that support zooplankton that larval fish feed on and they absorb substantial amounts of nutrients and other chemicals. While the decline in wetlands since 1994 seems small (if it isn't an artifact of methodology), it represents a 17% decline in wetland area between 1994 and 2010.

Table 1. Estimates of major land use for the portion of the Patuxent River's watershed that feeds into anadromous fish and Striped Bass spawning and nursery areas.

Year	% Agriculture	% Forest	% Urban	% Wetland
1973	39.8	45.9	13.1	1.2
1994	30.9	40.6	26.6	1.2
1997	29.4	39.2	30.0	1.1
2002	25.5	39.5	33.5	1.0
2010	20.5	35.1	41.7	1.0

Maryland initiated a *Patuxent River Policy Plan* (http://planning.maryland.gov/PDF/OurWork/PRC/OriginalPolicyPlan.pdf) in 1984 (impervious surface was 7.3%) to deal with point and nonpoint nutrient and sediment pollution associated with development of the watershed of its largest native river. Stormwater and wastewater are now the largest source of nitrogen, phosphorus, and sediment loads according to MD DNR Eyes on the Bay (http://eyesonthebay.dnr.maryland.gov/eyesonthebay/documents/PatuxentRiverupdatesummary2015.pdf). During 1999-2014, nitrogen and phosphorus loads decreased at the fall-line station at Bowie, but sediment increased. In the tidal portion of the upper river (where the anadromous fish and Striped Bass spawning and nurseries are), nitrogen, phosphorus, and sediment levels are improving, but still too high (MD DNR Eyes on the Bay).

Anadromous fish spawning success is negatively impacted by development – Eggs and larvae of fish are life stages most sensitive to pollution. The relative impact of development on eggs and larvae appears greater for Herring than other anadromous fish native to Maryland. The percentage of plankton drift net samples with Herring eggs and larvae (an index of their early life stage abundance) in Maryland's streams declines, on average, from 80% at 3% impervious surface to 20% at 14% impervious surface (Figure 4). The percent of towed plankton net samples with Yellow Perch larvae in Maryland's tidal nursery areas downstream of Herring spawning streams declined from 68% to 37% over the same span of development. The percentage of towed plankton net samples with Striped Bass eggs in Patuxent River during 2015 and 2016 were similar to those from less developed Choptank and Nanticoke rivers. In general, Herring eggs and larvae are found furthest upstream and Striped Bass furthest downstream. Yellow and White Perch overlap both.

Freshwater inputs have become saltier - Conductivity, a measure of how much salt is in water, is a freshwater quality indicator of urbanization. Conductivity measures dissolved salts in water based on how well water conducts electricity (measured as micro-Siemens per centimeter or µS/cm). Increases beyond a natural base level are usually from road salt, but eroding concrete, sewage, and industrial discharges may contribute. In non-tidal waters of an undeveloped watershed, salts that compose the base level come from weathering of rocks and soils. Base levels are higher for non-tidal streams in Maryland's Piedmont than its Coastal Plain (150 versus 109 µS/cm). Conductivity is expected to increase downstream in tidal waters due to intrusion of saltwater from the Bay.

Correlation analyses explored associations of annual mean conductivity at each water quality station with year (time trend), impervious surface (development), and annual mean flow (freshwater input). Correlation was also used to examine associations of mean annual conductivity among the stations themselves and there was a geographic pattern in the results. Correlations were different in significance and sign between upstream (Unity to TF1.3) and downstream stations, but not across regions. Conductivity increased with each successive year upstream ($r = 0.69$ to 0.89, $P < 0.0001$; Figure 5), where mean annual conductivity was well correlated with impervious surface coverage ($r = 0.65$ to 0.69, $P \leq 0.0001$; Figure 5). Meaningful associations of mean annual conductivity with these two variables were not apparent downstream (Figure 5). Flow was not associated with conductivity upstream, but was downstream ($r = -0.51$ to -0.87, $P = 0.005$ to < 0.0001); this negative correlation was not unexpected, since saltwater from the Bay would increasingly dilute freshwater input with downstream distance.

The upstream region identified in the correlation analysis encompassed much of the freshwater mainstem Patuxent River used by anadromous Herring for spawning. Mean conductivity increased from about 200 μS/cm in 1978 to 350-400 μS/cm by 2017. Conductivity measurements during spawning season in the American Shad spawning area just below Queen Anne's Bridge by the American Shad Restoration Project during 2001-2014 (indicated by the "Hatchery WQ" symbol on the map) exhibited the same increase (from about 250 to 330μS/cm) as the other stations (Figure 5). Conductivity readings at Laurel during July, 1963, to April, 1964 (N = 24), averaged 76 μS/cm. Twenty-four measurements during July, 1986, to April, 1987, averaged 125.7 μS/cm.

Mean annual conductivity was near base level at Unity and Rocky Gorge throughout the time-series and became elevated once water passed through the urbanized Laurel to Bowie region. Increasing conductivity over time indicated that elevated conductivity was not normal for the portion of upper Patuxent River used by anadromous Herring for spawning. Conductivity increases were strongly associated with increased impervious surface (development; Figure 5). Note that impervious surface growth slowed after 2004, but conductivity increases did not. The percent of samples with Herring eggs and larvae from Maryland spawning streams declines steadily with standardized median conductivity during spawning season (median conductivity is standardized to region by dividing by region base level; Figure 6). A breakpoint of about 1.6-times base level conductivity divides where normal levels of Herring eggs and larvae will be found in spawning streams (at less than 1.6) and where levels of eggs and larvae will be depressed (above 1.6; Figure 6). The non-tidal and fresh-tidal waters of the Patuxent River anadromous fish spawning area are well above this breakpoint regardless of whether Piedmont or Coastal Plain base levels are used

Several hypotheses can be formed to relate decreased anadromous Herring spawning to increased conductivity. Eggs and larvae may die in response to changes in salinity that challenge poorly developed salt regulation ability (osmoregulation) of early life stages. Toxic amounts of contaminants and additives may be associated with road salt. Changing stream chemistry may disorient spawning adults and disrupt upstream migration. Very recent research on stormwater ponds suggests that freshwater zooplankton that early larvae feed on decline with increased conductivity. Increasing conductivity may also be a symptom of cumulative deterioration of the watershed rather than a specific cause like increased salinity.

The Patuxent River Striped Bass spawning area never exhibited conductivity lower than 317 μS/cm in surveys during spring 2015-2016. Lower conductivities were common in comparable portions of Choptank, Nanticoke, and Wicomico River Striped Bass spawning areas during 2015-2016. Elevated salt levels by themselves at the head of the Patuxent River spawning area should not be an issue for Striped Bass, White Perch, and Yellow Perch since they can be abundant in higher conductivity regions further downstream where freshwater is more mixed with intruding saltwater. Elevated conductivity does suggest that water quality has been influenced by development in this region.

Runoff characteristics may have changed - The ratio of mean annual flow (Bowie gauging station) to annual precipitation (Reagan National Airport) may indicate change in runoff with precipitation (Figure 7). This ratio was much more variable during 1978-2008 than afterwards (Figure 7). The more stable ratio for recent years could indicate a shift in flow source away from groundwater (where surface runoff is absorbed and released long-term) to surface runoff (most water moves as short-term surface runoff). Previous analyses of Mattawoman and Piscataway creeks found changes in relationships of flow variability to the ratio of mean annual flow / annual precipitation occurred at around 9% impervious surface. The percent of samples with Herring eggs and larvae were predominantly low once this change occurred.

Juvenile indices of Patuxent River anadromous Herring have deteriorated – Year-class success of anadromous fish in Maryland's portion of the Bay has been measured by a standardized summer seine survey targeting juveniles (Juvenile Seine Survey) in four major spawning systems: Head-of-Bay, Potomac River, Nanticoke River, and Choptank River (http://dnr.maryland.gov/fisheries/Pages/striped-bass/juvenile-index.aspx). Patuxent River monitoring was added in 1983 to support the *Patuxent River Policy Plan*.

Anadromous fish year-class success across the four major nursery areas in Maryland's portion of Chesapeake Bay (Patuxent River not included) shared similar trajectories during 1959 to the mid-1990s. Good year-classes were common into the mid-1960s to early 1970s, followed by an extended period of depressed year-class success that led to collapse of traditional fisheries. The extent of the depressed period varied by species and system monitored. This depression, shared among different anadromous species across a broad geographic area, indicated a common habitat issue may have been in play. Recovery in various degrees was evident for all anadromous fishes in Head-of-Bay, Potomac River, Nanticoke River, and Choptank River by the early 1990s. Year-class success of anadromous fish stabilized at higher levels in the 1990s-2000s than during the depressed period (with the exception of Blueback Herring in the Nanticoke River). Agriculture was and is the main human land use in all watersheds surveyed by the Juvenile Seine Survey.

Patuxent River anadromous fish shared year-class trends of other survey regions until its watershed transformed from a rural to suburban. Since the mid-2000s, Alewife (Figure 8) and Blueback Herring (Figure 9) indices have been low when compared to the general trend for the major spawning areas; a high Patuxent River Blueback Herring index in 2011was the sole exception. American Shad juveniles have not been detected in Patuxent River by the Juvenile Seine Survey since 2009 (Figure 10).

Other Patuxent River anadromous fish (not shown) have exhibited lesser or no signs of recruitment decline. White Perch year-class success appears unimpacted. Striped Bass year-class success in Patuxent River has been similar to other systems, but the 2015 year-class may not have been as strong as elsewhere. Yellow Perch trends are difficult to judge from seine surveys, but juveniles were not detected in 5 of 23 annual surveys during 1983-2005 and in 6 of 12 surveys afterward. The percent of plankton net samples from Patuxent River with Yellow Perch larvae and initial feeding success of those larvae during 2015-2016 were comparable to those of less developed watersheds of similar scale (Nanticoke and Choptank rivers). Yellow Perch juveniles were detected in Patuxent River by the Juvenile Seine Survey during 2015-2016.
American Shad stocking – Stocking of American Shad by the Hatcheries Division Program (http://dnr.maryland.gov/fisheries/Pages/hatcheries/Shad.aspx) as larvae and juveniles provides additional information on habitat conditions within the Patuxent River anadromous fish nursery area.

American Shad were only present in the Juvenile Seine Survey during the years that heavy stocking occurred (1995-2009). Even during the stocking period, a breakpoint was evident in the ratio of the Juvenile Seine Survey indices to numbers stocked of each life stage category (i.e., juvenile index per larvae, early juvenile, or late juvenile stocked) – ratios were lower after 2001 (Figure 11). Lower ratios indicated fewer Shad per unit stocked were surviving to be captured later. The 2001 breakpoint corresponds to watershed development at 10.8% impervious surface.

During 1994-2009, the American Shad Restoration Project stocked large numbers of larvae (feeding larvae 6-12 days old), early juveniles (30 days old), and late juveniles (75 days old) with otolith (earbone) marks indicating hatchery origin. They intensively monitored the river for juvenile American Shad over 10 sites during August-October to evaluate the impact of stocking. The American Shad Restoration Project seine survey, with about 100 seine hauls per year, is more likely to detect low numbers of American Shad juveniles than the more general Juvenile Seine Survey (18 seine hauls per year) since it takes more samples, employs a longer net, and concentrates sampling in the Shad nursery.

Wild Shad accounted for 1-17% of Shad sampled during the stocking period. The catch of juvenile Shad in the American Shad Fish Restoration Project seine survey was well related to numbers stocked during 2004-2014, particularly for Shad stocked as larvae and early juveniles. Numbers of juveniles captured plummeted after stocking was discontinued in 2009. It appears that a (presumably) boosted spawning population from these stockings did not translate into increased juvenile production. Impervious surface rose from 9.5% to 12.5% from the initiation of stocking in 1994 to the latest impervious surface estimate available, 2014.

Recaptures of stocked larvae as juveniles later in summer-fall (15-50% of recaptures) suggested a habitat-related bottleneck during egg through first-feeding (stages that precede those stocked) that limited survival in Patuxent River. These earliest life stages (eggs, yolk sac and early postlarvae) undergo rapid physiological change and are sensitive to contaminants and water quality; food (zooplankton) availability is important to first-feeding larvae. These factors are negatively influenced by development. Endocrine disrupting compounds absorbed as juveniles may impact egg formation as adults (B. Richardson, MD DNR, personal communication).

Implications – Habitat changes in Patuxent River fit the general schedule indicated by impervious surface reference points that were largely based on monitoring subestuaries at the next smallest scale. Smaller subestuaries such as Middle, Magothy, Severn, and South rivers (15% or more impervious surface) have exhibited serious habitat issues with development that have precluded success of traditional fisheries management measures. Our hope that watersheds of larger scale (such as the Patuxent, Choptank, or Nanticoke rivers) would be more resistant to development appears unfounded. Management of Maryland’s Bay fisheries resources will be increasingly challenged by development.

Potential Next Steps – Opening lines of communication with county governments upstream and adjacent to the spawning to inform them about land use impacts to anadromous fish habitat allows them an opportunity to conserve healthy portions of the watershed and mitigate detrimental effects elsewhere. A couple of general principles can be highlighted: 1) Rural lands tend to maintain healthy habitat for fisheries resources and 2) Harmful contaminants other than nutrients (road salt, toxic metals, endocrine disrupters) may be present in surface run off or sewage effluent, but methods may be available to deal with them and nutrients as well.

Figure 1. Location of major features in the anadromous fish spawning area of the Patuxent River.

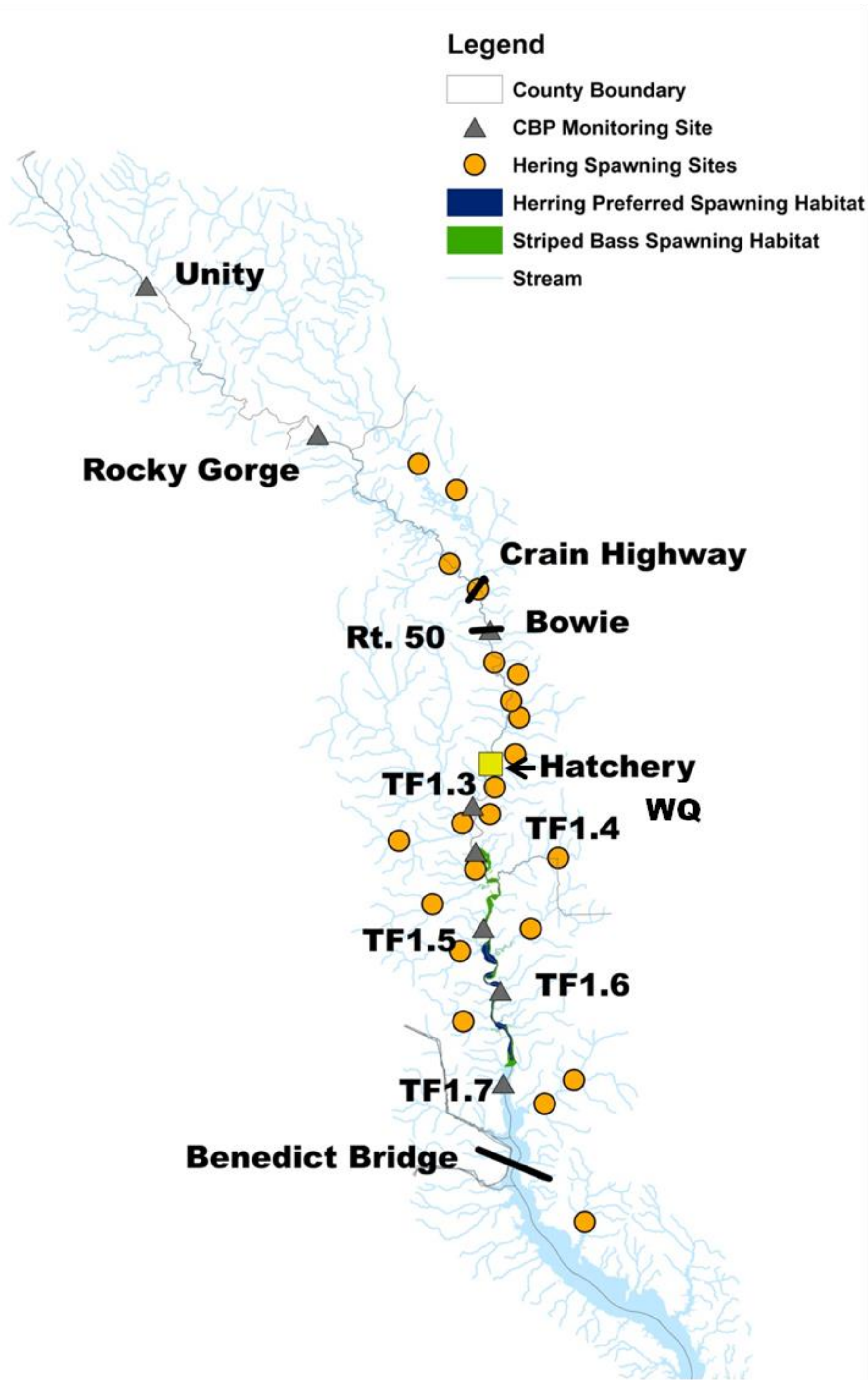


Figure 2. Trend in impervious surface coverage of the portion of Patuxent River watershed at and above the anadromous fish spawning and nursery area.

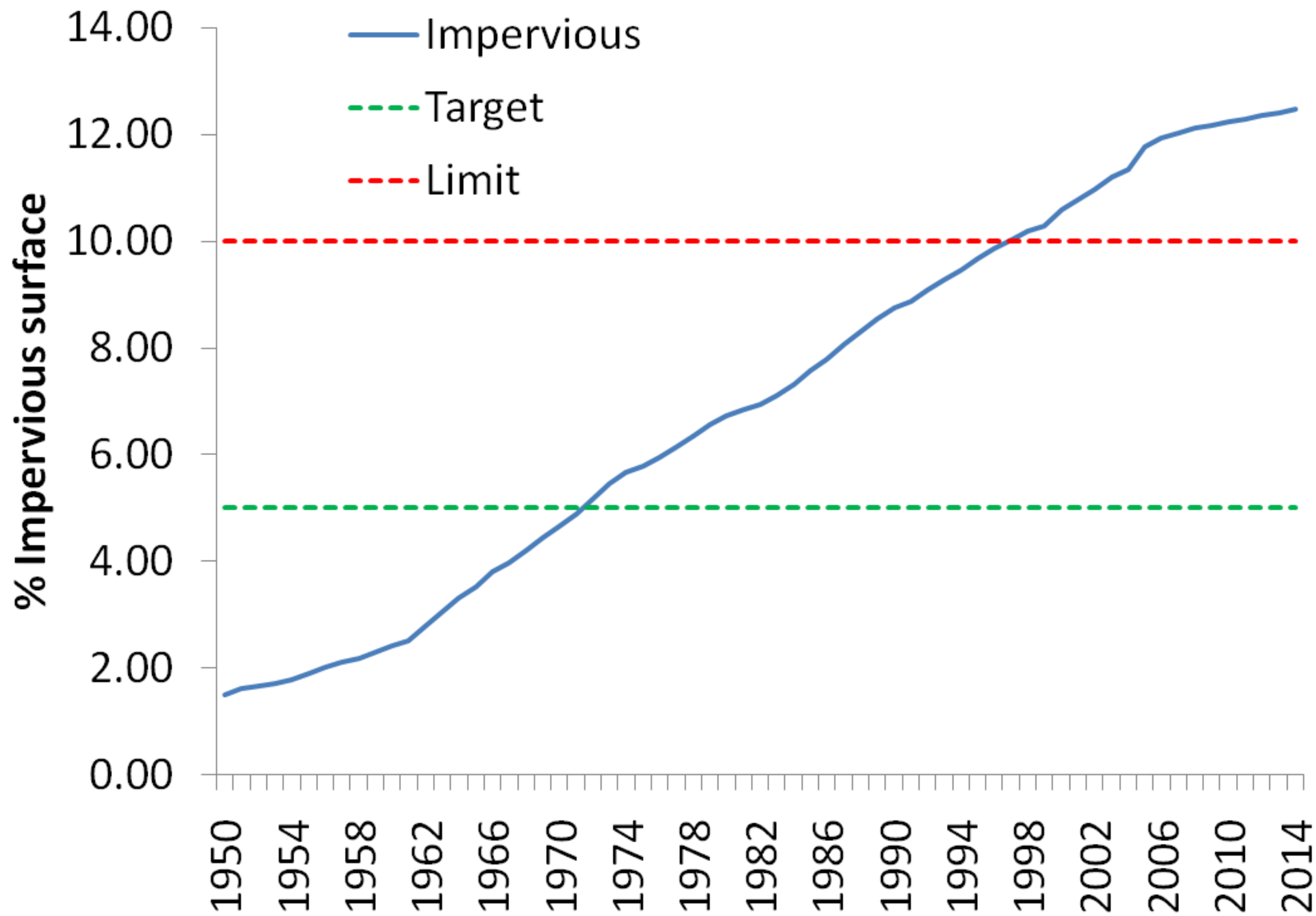


Figure 3. Trends in development (percent impervious surface) for segments of the Patuxent watershed. Name indicates the portion of the watershed above a water quality station. See Figure 1 for locations.

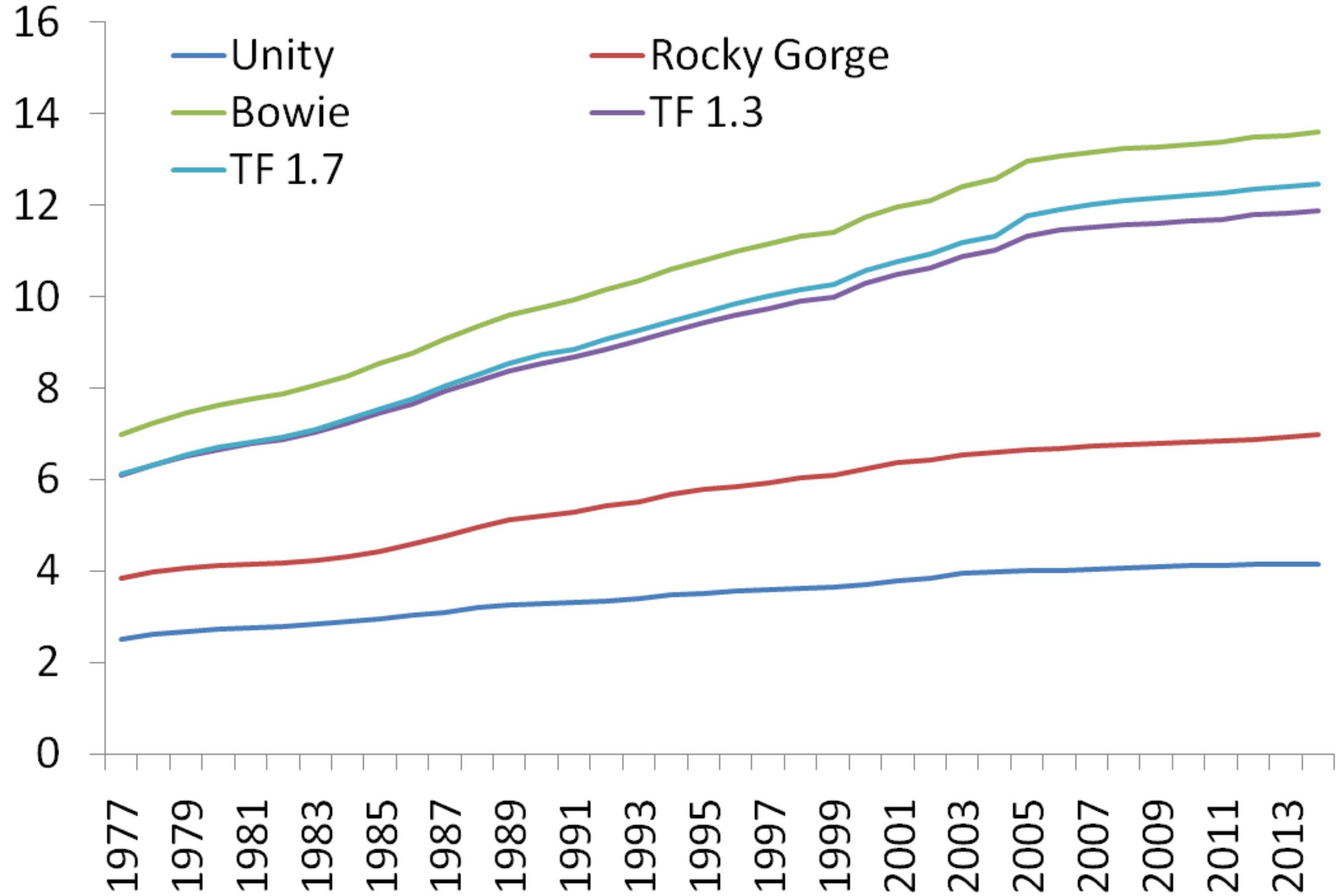


Figure 4. Relationship of level of development (impervious surface) to percent of samples with Herring eggs and larvae for six spawning streams in Maryland.

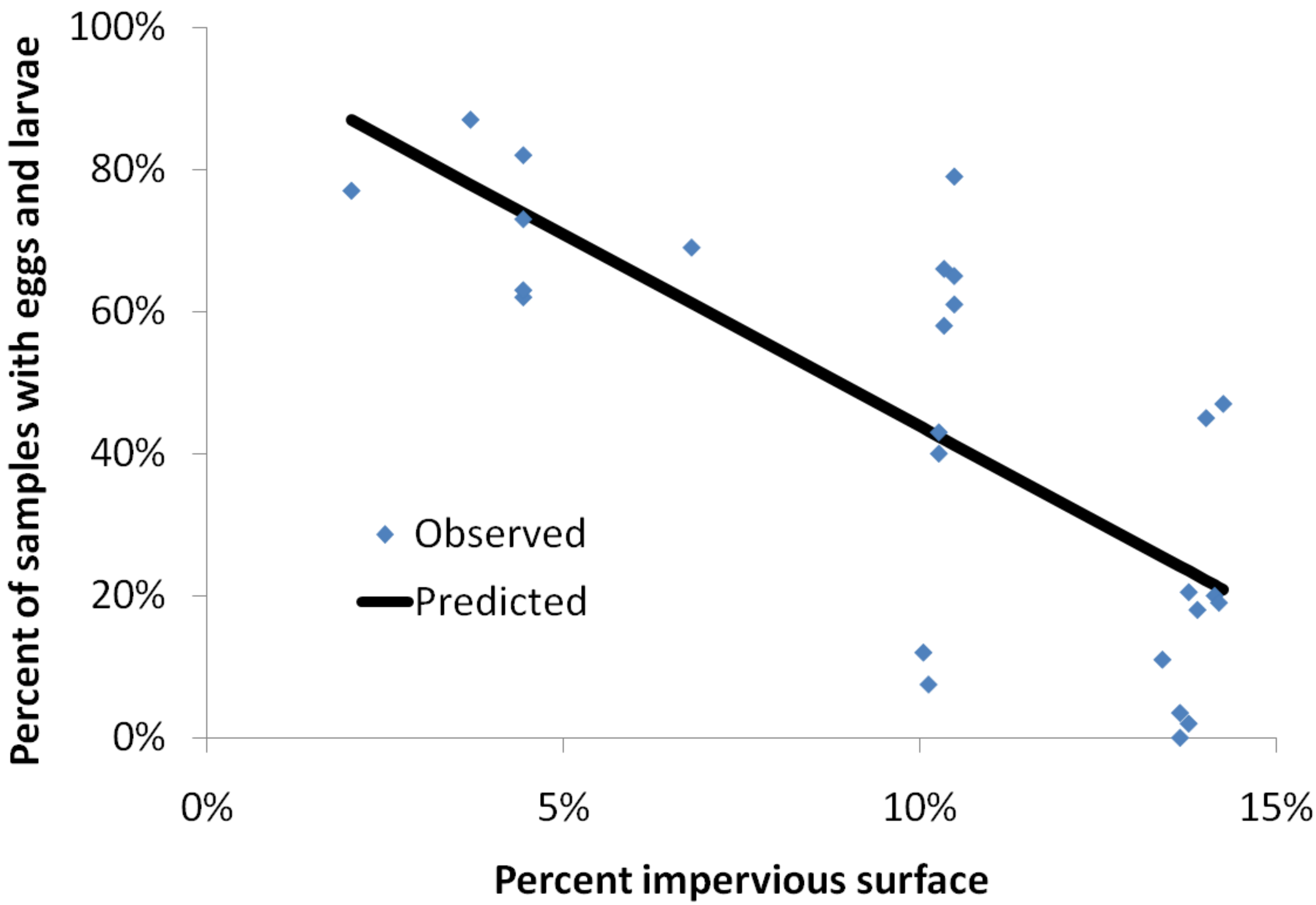


Figure 5. Annual trends in conductivity in (A) upriver and downriver (B) water quality stations and trends in conductivity with development (impervious surface) upriver (C) and (D) downriver.

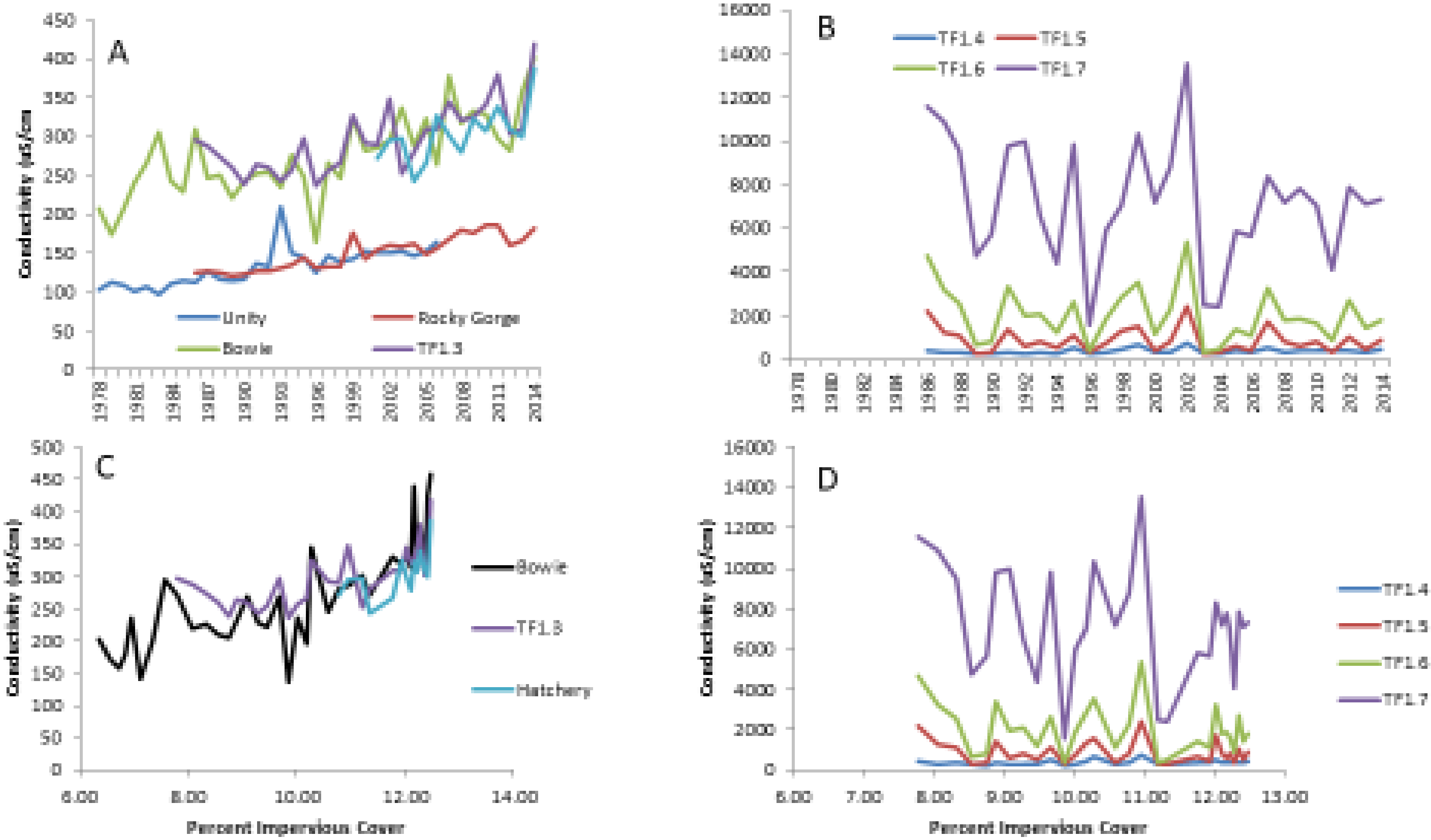


Figure 6. Relationship of standardized median conductivity (median divided by its regional base level) and the proportion of stream samples with Herring eggs and larvae (Pherr) in six Maryland spawning streams.

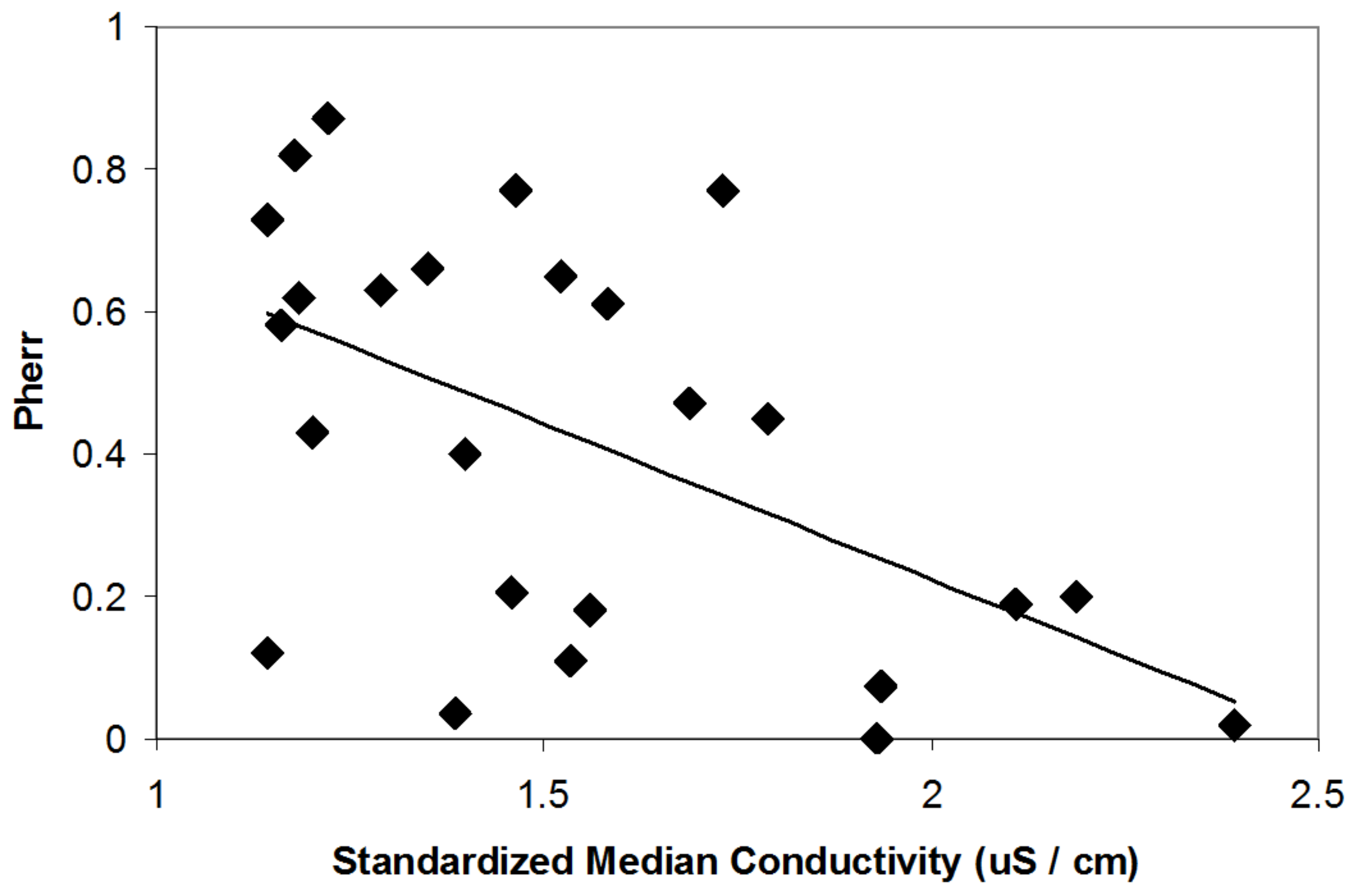


Figure 7. Ratio of mean annual flow at USGS gauge and annual precipitation at Reagan National Airport. Lines indicate range since 2009.

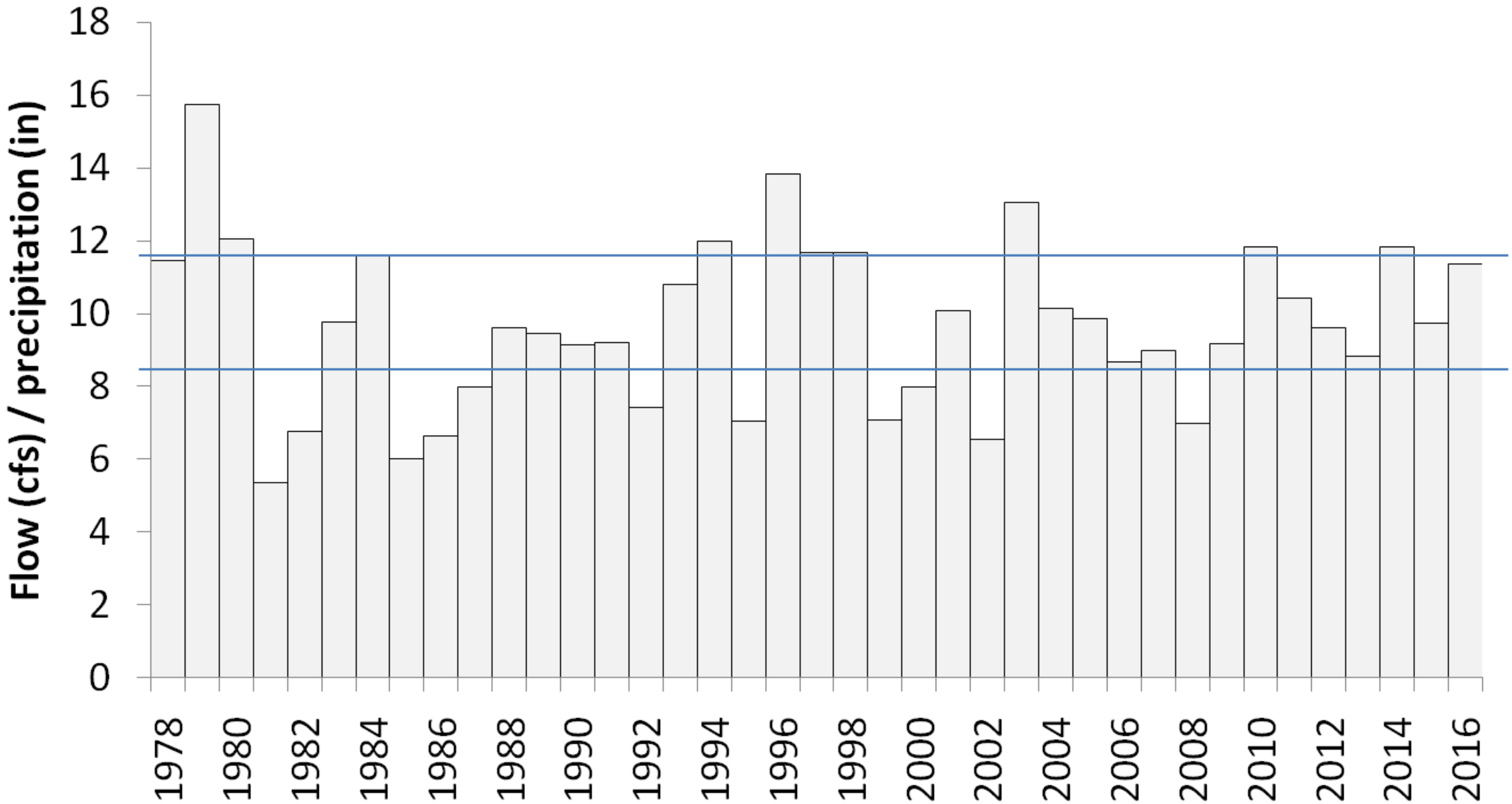


Figure 8. Comparison of relative abundance of Alewife juveniles in Patuxent River with relative abundance averaged across four other major nursery areas.

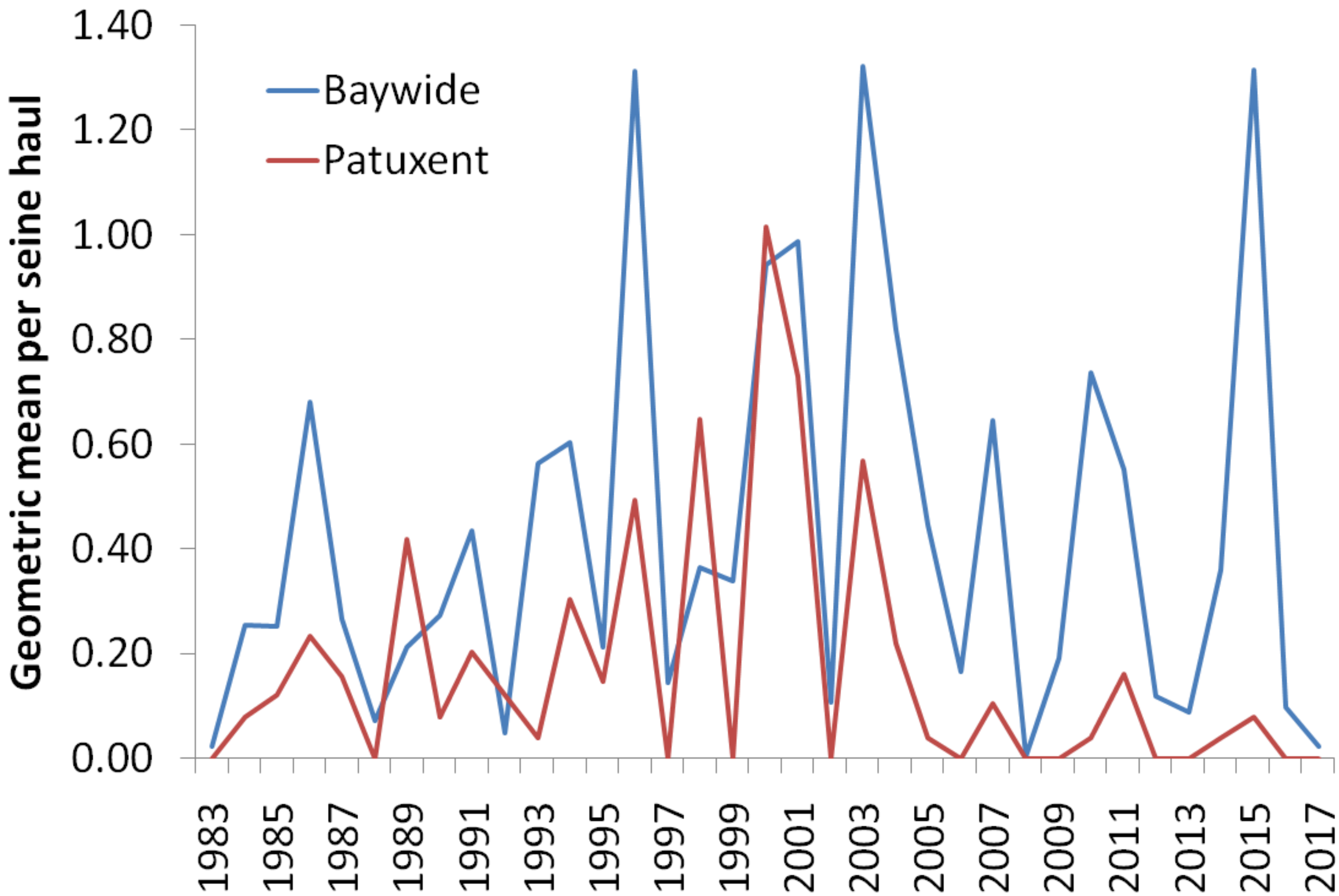


Figure 9. Comparison of relative abundance of Blueback Herring juveniles in Patuxent River with relative abundance averaged across four other major nursery areas.

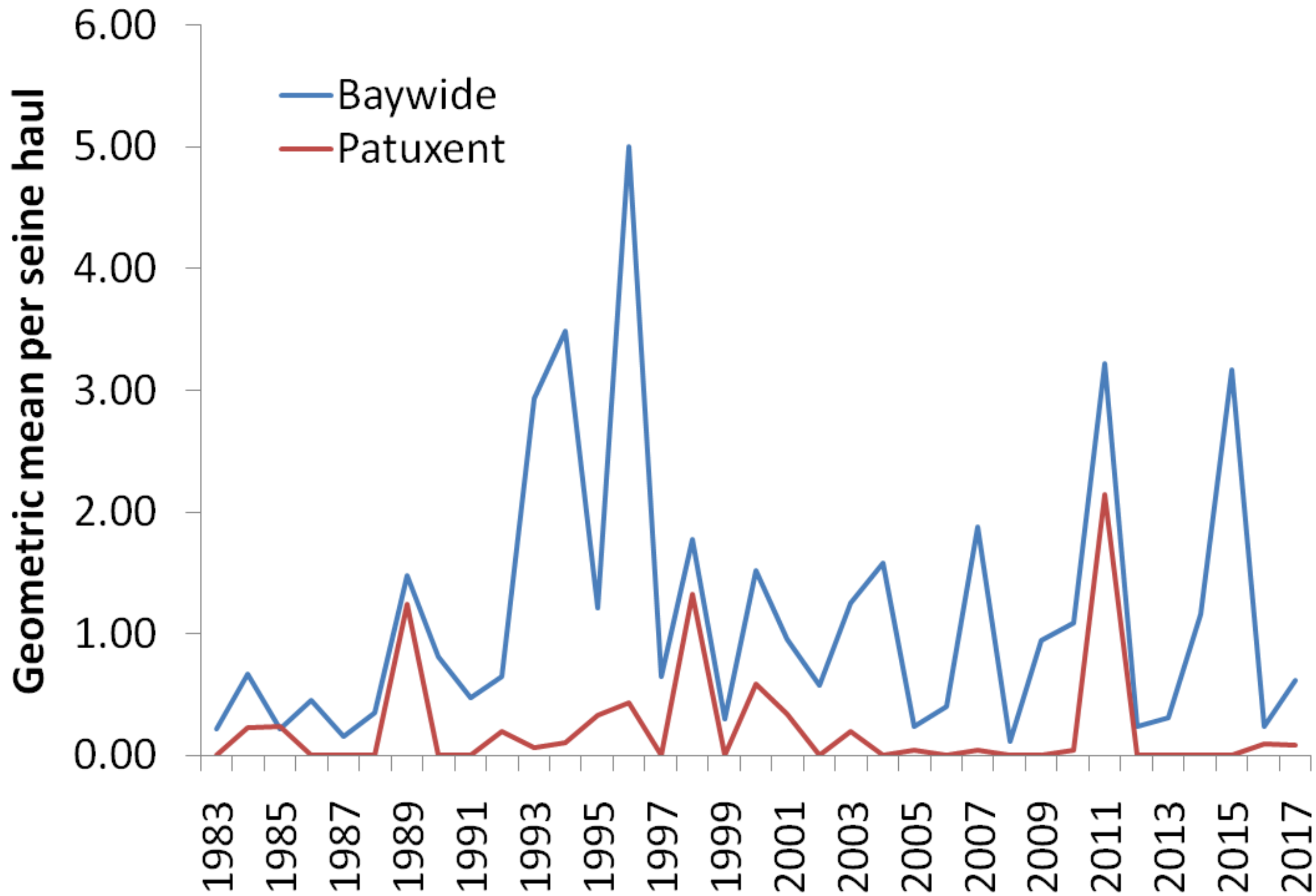


Figure 10. Comparison of relative abundance of American shad juveniles in Patuxent River with relative abundance averaged across four other major nursery areas

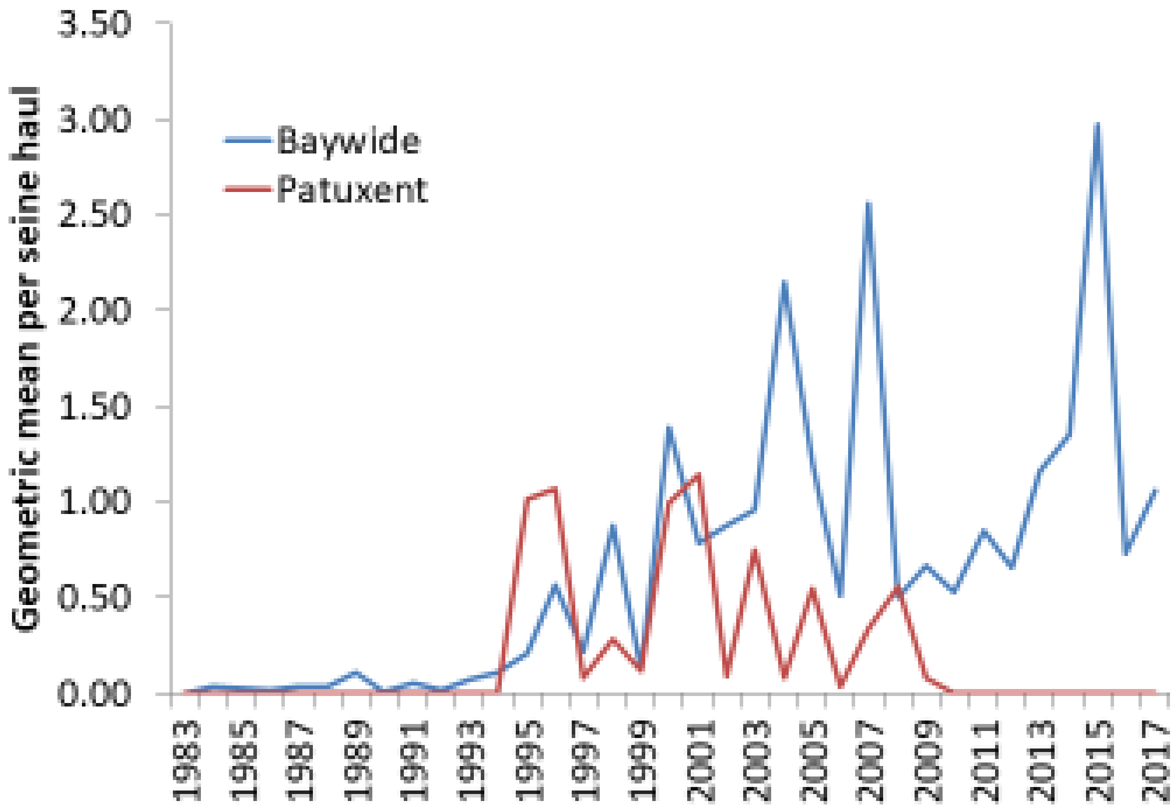
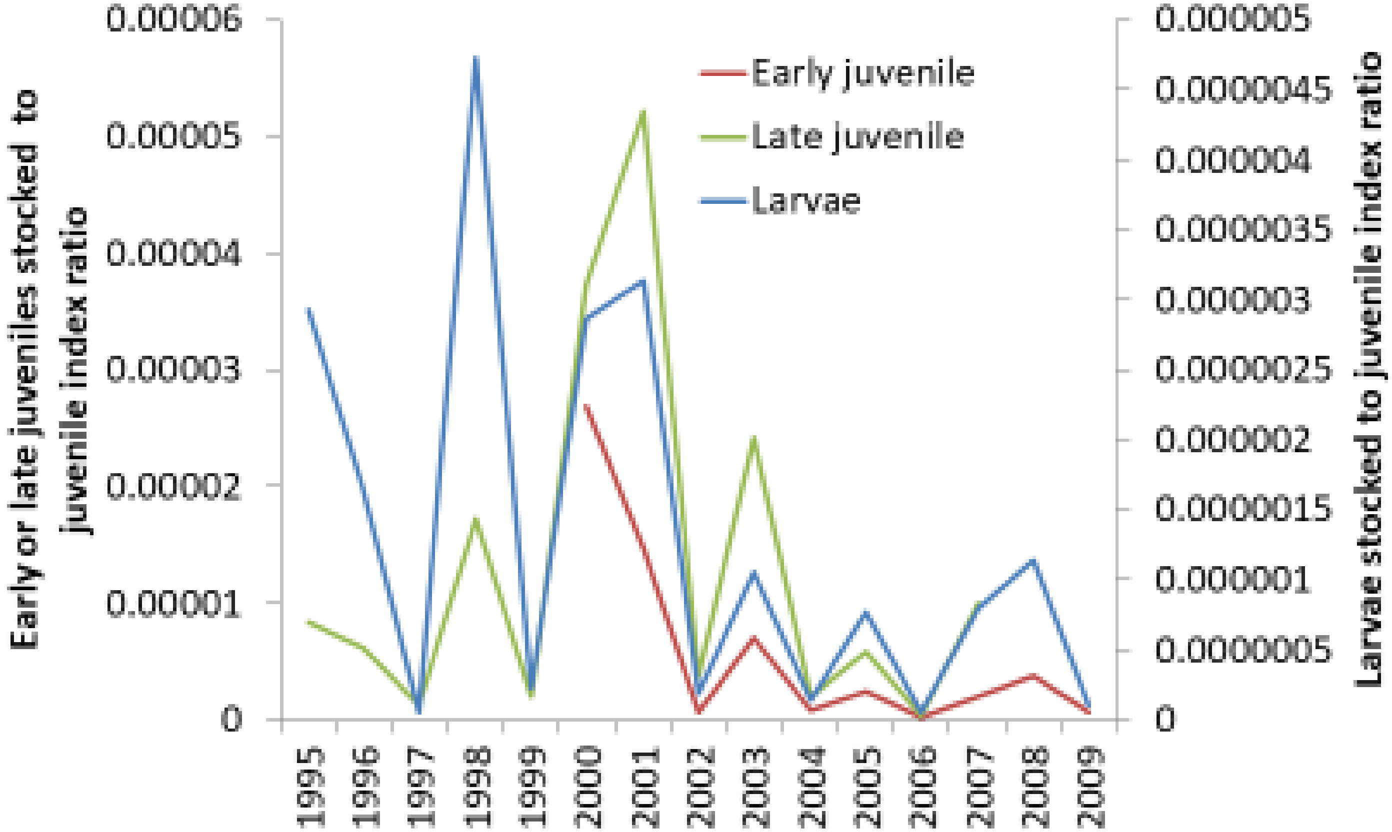


Figure 11. Ratios of the American shad Juvenile Seine Survey index to numbers stocked at three life stages



JOB 3: Developing Priority Fish Habitat Spatial Tools

Margaret McGinty and Jim Uphoff

Abstract

We compiled information on eggs and larvae of American Shad, Alewife and Blueback Herring (alosines) and pH, total suspended solids, aluminum, iron, lead, and zinc to map these potential stressors in relation to regions with life stages that would be most sensitive to them. Variables mapped were chosen based on Chesapeake Bay habitat requirements. Species-specific criteria for evaluating toxicity were sparse. While several Chesapeake Bay Program and USEPA databases were available, spatial and temporal resolution was poor for evaluating toxicity in spawning and larval nursery habitat of American Shad and Herring. Contaminants were ubiquitous in sediment; zinc and lead were more prevalent in urban areas. These data should be considered when planning restoration and management to develop effective strategies to restore habitat that supports all life cycles. In absence of such focus, an alternative approach would be to identify regionally productive watersheds and focus on conservation of viable habitats.

Introduction

This installment focuses on compiling information on all life stages of American Shad, Alewife and Blueback Herring to demonstrate how well we can mine and evaluate other data sources to visualize potential stressors when prioritizing areas for restoration. In this report, we reviewed factors other than dissolved oxygen (DO) and impervious surface (these were described in Uphoff et al. 2015; 2016). Using the collective information, we developed a conceptual diagram to visualize how species life history was potentially impaired by a given stressor.

In our previous report (Uphoff et al. 2017), we described how spatial tools depict fish habitat suitability to guide management policy. This is a timely pursuit in light of a new effort underway to develop a Chesapeake Bay fish habitat assessment (Hunt et al. 2018; M. McGinty, MDDNR, personal observation). We are participating in that effort to assure consideration of stressors other than nutrients and sediment that are the focus of Chesapeake Bay restoration efforts. Without consideration of the suite of factors influencing fish habitat, investments in restoration may not lead to improved fish habitat and production.

(Uphoff et al. 2014; 2017) identified and mapped impervious surface and low dissolved oxygen as stressors that could limit production of target species. This year we identify other stressors, primarily related to, but not limited to, potential toxicity of inorganic contaminants. We have chosen American Shad, Alewife, and Blueback Herring (alosines) as demonstration species and have focused on habitat where eggs and larvae would be found. These early life stages are most sensitive to anthropogenic habitat perturbations; larvae and eggs are most sensitive to contaminants (Bengston et al. 1993; Brooks et al. 2012).

Anadromous fish (American Shad, Blueback Herring, Alewife, Striped Bass, White Perch, and Yellow Perch) in Maryland’s portion of Chesapeake Bay have exhibited similar trends in baywide juvenile indices since 1959 (Figure 1; Durell and Weedon 2017). Indices were generally high into the early 1970s, fell to a sustained nadir that lasted until the early 1990s, and then recovered (Uphoff 2008). These species differed in management strategies, maturation, migration, spawning locations, egg types, and adult trophic levels but shared common larval nurseries within these tributaries (Uphoff 2008). Although this decline was across multiple species, the vast majority of work to understand the impact of water quality and inorganic contaminants was confined to Striped Bass during the 1980s (Richards and Rago 1999). Bioassays (Hall et al. 1993) and field surveys (Uphoff 1989; 1992) within Maryland's four major Striped Bass spawning and larval nursery areas indicated that survival of Striped Bass larvae was negatively influenced by trace metals (aluminum, cadmium, copper, lead, and zinc) and water quality (pH, hardness, and alkalinity) that affected metals toxicity. Striped Bass larval survival improved steadily in the 1980s (Uphoff 2008) and work to understand the influence of water quality and toxic metals was suspended once juvenile indices noticeably improved and emphasis shifted towards managing reopening fisheries. Given the correspondence of trends in juvenile of Striped Bass, American Shad, Blueback Herring, and Alewife (Figure 1), it is not unreasonable to infer that conditions impacting Striped Bass impacted the latter three species. Inorganic contaminants and water quality parameters that influence toxicity have potential to limit habitat quality for eggs, larvae, and juveniles of target alosine species (Klauda et al. 1991a; 1991b).

American Shad Life History - American Shad were once among the most esteemed species in Chesapeake Bay (Uphoff et al. 2017). Hildebrand and Schroeder (1928) reported a decline in Chesapeake Bay landings from historical high levels. Commercial landings in Maryland were high during the early 1930s and again during the late 1940s through the early 1970's (Bonzek and Jones 1984). The fishery collapsed in the 1970s (Bonzek and Jones 1984) following a decline of year-class success (indicated by juvenile indices; Figure 1). Stocks reached an all-time low in the 1970's, prompting a moratorium in 1980. Aggressive stocking efforts had some success, but the Chesapeake Bay fishery is still closed and the ocean intercept fishery is as well. Klauda, et al. (1991b) cited overfishing, migratory impediments and degradation of habitat as key causes of the decline. They proposed that management toward recovery should include managing to prevent overfishing and protection of spawning and nursery habitats, citing that eggs, larvae and juveniles as critical life stages (Klauda et al. 1991b).

American Shad display true anadromy, residing in oceanic waters as adults and return to their natal rivers to spawn in tidal-fresh and freshwater stream habitat (Klauda et al. 1991b). Spawning occurs in the Chesapeake Bay region, during April to June, with specific timing dictated by temperature (16-19°C). American Shad select spawning habitat based on macrohabitat features (Harris and Hightower 2011) and spawn in moderate to swift flows (Hightower and Sparks 2003). Egg incubation time varies (2-9 days) depending on water temperature, but egg mortality increases with prolonged exposure to temperatures below 16°C (Klauda et al. 1991b). After hatching, prolarvae quickly absorb the yolk sac and are feeding within four to seven days. Postlarvae remain near the surface and disperse downstream with ebbing tides (Klauda et al. 1991b). Mortality of eggs and larvae is high, with a 1-2% survival rate from egg to juvenile stage (Crecco et al. 1987). Juveniles spend their first summer in tidal-fresh and low salinity regions of their natal rivers. They begin their seaward migration in the fall when temperatures begin to drop below 20°C and are thought to remain at sea for about five years until they begin their migratory trek back to their natal rivers to spawn (Klauda et al. 1991b).

Alewife and Blueback Herring Life History - Alewife and Blueback Herring (collectively Herring) have long been a favored fish in Chesapeake Bay (Hildebrand and Schroeder 1928) and one of the oldest documented fisheries in the United States (Read et al. 2011). Like American Shad, Herring landings along the Atlantic Coast have been declining for well over the last century leaving the population severely depressed compared to historical levels (Limburg and Waldman 2009). Hildebrand and Schroeder (1928) documented declines in Chesapeake Bay landings in the 1920's. Landings in Maryland declined about 3-fold (on average) between 1929 and 1975, before collapsing entirely by the late 1970's (Bonzek and Jones 1984). The collapse in Maryland landings was preceded by a drastic decline in Blueback Herring juvenile indices (Figure 1); landings were typically reported for both species combined, but Blueback Herring were more valuable commercially and made up most of the harvest. Declines in alosine commercial and recreational catch rates prompted the Atlantic States Marine Fisheries Commission (ASMFC) to develop an inter-jurisdictional fisheries management plan in 1985 (ASMFC 1985), but stocks continued to decline (ASMFC 2018). In 2009, the ASMFC required states to restrict fishing in state waters by 2012, unless they had an approved management plan in place. Maryland imposed a harvest moratorium in 2012 (Starks et al. 2017). Though overfishing has been implicated as a factor prompting these declines, ASMFC elected to apply the depleted rather than overfished status, citing additional factors including habitat loss, predation and climate change have also contributed to observed declines (ASMFC 2018).

Alewife and Blueback Herring have very similar life histories, with habitat occupation overlapping for all life stages. Like American Shad, Herring are anadromous species, entering the Bay and its tributaries, in the spring from March through May (Murdy et al. 1997). Alewife and Blueback Herring are thought to home to natal rivers to spawn (ASMFC 2009a; ASMFC 2009b). Historical surveys of Alewife and Blueback Herring spawning habitat found spawning extended into fluvial areas (O'Dell et al. 1975; 1980; Figure 2). Where the two species are sympatric, they are reported to partition when spawning. Blueback Herring show preferences for waters with higher flows and avoidance of lentic areas (Loesch 1987). Alewife spawn in sluggish flows, while Blueback Herring spawn in sluggish to swift flows (Pardue 1983). Spawning substrates for Herring include gravel, sand, and detritus (Pardue 1983). Both species respond to temperature cues to prompt spawning with adults returning to marine waters when spawning is complete (Klauda et al. 1991a). In an extensive review of Herring habitat requirements, Klauda et al. (1991a) reported Alewife eggs hatch between 2 and 15 days depending on temperature and Blueback Herring eggs hatch between 2 and 4 days. Alewife spawn earlier in the spring (Plough et al. 2018) at lower temperatures that would slow egg development, resulting in a wider range of hatching dates. We assume Blueback Herring hatch rates are also temperature dependent. Blueback Herring transition from yolk-sac larvae to feeding larvae in about four days post-hatch when their yolk is absorbed, but no timeframe was reported for Alewife (Klauda et al. 1991a). No timelines were identified for larval transformation to the juvenile stage, though Klauda (1991a) cited one study by Nordon (1967) that found warmer temperatures promoted faster larval growth rates. Juveniles of both species remain in fresher reaches of the Bay throughout the summer (Klauda et al. 1991a). Fall temperature declines initiate emigration of both species with one study suggesting 21°C as a trigger and migration peaking at 14-15°C (Klauda et al. 1991a). Loesch et al. (1982) observed juvenile Alewife and Blueback Herring practice diel migratory patterns during their seaward migration with Blueback Herring orienting higher in the water column than Alewife at night. Loesch (1987) suggested these differences reduced competition among the species as they emigrate.

Methods

Estimating Egg and Larval Habitat - Uphoff et al. (2014) described the general approach applied to delineate natural limits to distribution and develop habitat criteria for eggs and larvae of target species occurring in Maryland's tidal water. This methodology was used to define habitat for life stages of American Shad, Alewife, and Blueback Herring. Salinity influences distribution and abundance of fish in estuaries (Hopkins and Cech 2003; Cyrus and Blaber 1992; Allen 1982) and was used to estimate the extent of tidal habitat typically available to a species and life stage (eggs, larvae, and juveniles) in the absence of stressors. We classified habitat of a species' life stage into four categories: preferred (area of high occurrence of a species' life stage), acceptable (area of moderate occurrence), marginal (area of low occurrence) and not suitable (life stage absent). For eggs and larvae, we used historical data from a longitudinal survey of eggs and larvae in Chesapeake Bay with a wide range of salinities (Dovel 1971) to develop cumulative distributions. Lines were fitted to three segments of the cumulative percentage distributions to determine salinity categories by species. Lines with highest slopes were categorized as preferred habitat; the next, more moderate slope indicated acceptable habitat; and marginal habitat was represented by slope with the least amount of change (starting at the end of the acceptable habitat segment) until 100% of the cumulative distribution was reached (Uphoff et al. 2014; 2017). We used interpolated average seasonal bottom salinity data obtained from the Chesapeake Bay Program data to map salinity in the Bay (Tom Parham, MD DNR Resource Assessment Service, personal communication) and categorized habitat for each cell for each species and life stage (Uphoff et al. 2014; 2017).

We used presence data from historical anadromous fish spawning habitat surveys (O'Dell et al. 1975; 1980) to identify nontidal streams where spawning was observed in the 1970s. O'Dell et al. (1975; 1980) combined alosids into a single category (Herring) because identification of eggs and larvae to species can be difficult (Uphoff et al. 2014; 2017). Presence of Herring was based on a single occurrence of an egg, larva, or adult at a site during a survey (O'Dell et al. 1975; 1980)

Water Quality - We accessed Chesapeake Bay Program water quality data to evaluate total suspended solids and pH. We evaluated frequency of samples with values exceeding reported biological limits for Shad and Herring (Klauda et al. 1991a; 1991b). These data, available through the Chesapeake Bay Program's Data Hub (http://datahub.chesapeakebay.net/WaterQuality), were collected by partners in the Chesapeake Bay jurisdiction, and compiled and managed by the Chesapeake Bay Program Data Center. The database contains reported measurements of various physicochemical and nutrient parameters. Sampling frequency has changed over the years, but is generally conducted at each fixed station at least monthly. Data evaluated covered tidal and nontidal habitat in Maryland and District of Columbia to evaluate stressors influencing habitat mainly in Maryland's watersheds and jurisdiction (Figure 3).

Contaminants - We accessed the Chesapeake Bay Program, Toxics Database (https://www.chesapeakebay.net/what/downloads/cbp_toxics_database; Figure 4) to examine distributions of metals (aluminum, iron, lead and zinc), identified as potentially toxic to alosines. In addition to reported biological concentrations of these metals that demonstrated toxicity to our target species in Klauda et al. (1991a; 1991b), we also compared levels of these metals to US EPA water quality criteria (US EPA 1986) and Sediment Quality Guidelines. US EPA issued ambient water quality criteria guidelines for a large suite of contaminants, defining acute and chronic limits for freshwater and saltwater. US EPA issued criteria for the four metals identified as potentially toxic to alosines (Table 1; US EPA 1986).

National Oceanographic and Atmospheric Administration (NOAA) developed sediment quality guidelines for lead and zinc in tidal waters, which were meant to aid interpretation of potential toxicity of sediments (Long and MacDonald 1998). They evaluated a set of toxicity tests results to examine the probability of a toxicant causing a biological effect. They established two effect levels, Effects Range-Median (ER-M) and Effects Range-Low (ER-L). These represent the median and tenth percentile concentrations where toxicity was observed. While these values are not meant to be applied as thresholds or criteria, they can indicate potential for toxicity (Long and MacDonald 1998). We investigated sediment concentrations based on these guidelines to explore frequency of stations exceeding the criteria for lead and zinc to determine if management attention is warranted.

The Chesapeake Bay Program Toxics database is a compilation of data from various studies conducted between 1980 and 2002. There are over 400 chemicals evaluated representing various classes including metals, organic compounds, inorganic compounds, pesticides, PCB's and PAH's.

We evaluated the data by media sampled: fish and shellfish tissue samples were classified as biological, sediment samples as sediment, and various fractions of water as water media. Because these data cover an expanse of time with various sampling frequencies and methodologies, we present the data to only represent potential stressors and evaluate their spatial distribution. We converted all reported values to standard units of µg/L. Because water sampling was limited to a few locales, we also examined sediment and biological data to assess the spatial extent of a metal in the Bay watershed. When toxicity criteria for saltwater or sediments did not exist, we applied the EPA freshwater acute criteria as a relative measure of concentration. Presence in sediments did not imply toxicity, however it did indicate a legacy of load to the watershed. Biological samples indicate a toxic metal was bioavailable since it was assimilated into fish and shellfish tissues.

We also accessed EPA's Toxics Release Inventory Data set (https://www.epa.gov/toxics-release-inventory-tri-program/tri-data-and-tools) to identify release areas of toxics identified as limiting to American Shad and Herring life stages (Figure 5). These data are provided by industry permit holders to track and identify release volumes and occurrences of toxic chemicals.

Finally, we examined EPA's superfund inventory data (https://www.epa.gov/superfund/search-superfund-sites-where-you-live#map) and calculated the number of superfund sites by county in Maryland. Superfund sites are in various stages of completion from design to implementation. Unless a site was declared completed, we included it as an active site and consider it a potential source of contaminants.

Results

pH – Klauda et al. (1991b) cited a paucity of information regarding pH effects on American Shad early life stages, but recommended pH less than 6.0 as a suitable threshold for American Shad egg survival and 6.7 for larvae. Klauda et al. (1991a) did not find sufficient studies evaluating effects of pH on early life stages on Alewife, so they combined the few study results they found with conclusions reported for Blueback Herring to propose suitable pH criteria for Alewife eggs of 5.0-8.5 and 5.5-8.5 for prolarvae. For Blueback Herring, Klauda et al. (1991a) reported newly spawned eggs were sensitive to low pH, while 1 day old eggs were resistant. They proposed suitable criteria of pH ranging from 5.7-8.5 and optimum of 6.0-8.0.

The pH criteria for all species and life stages were generally met most of the time (Table 2). When examining stations where pH was <5.7 (Herring limit), there were only a few stations that low. These were in the tidal Potomac River, tidal Patuxent River and two stations on the lower Eastern Shore that overlap or are in close proximity to documented Herring habitat (Figure 6).

Klauda et al. (1991a) reported that increased exposure time to low pH in bioassays decreased egg and larval survival and presence of monomeric aluminum influenced mortality. Mortality rates accelerated when monomeric aluminum concentrations ranging from 200-400 µg/L were added to the treatment. We investigated the presence of aluminum in the Bay watershed and report our findings in the following section evaluating contaminants that have been implicated as toxic to early life stages of Shad and Herring.

Inorganic Contaminants - Aluminum toxicity in fish was the focus of many studies in the early 1980's as research centered on understanding the impacts of acid rain and the attendant release of metals from soils. Aluminum is naturally present in the environment in background levels, but can become water soluble in low acid conditions and have a toxic effect on fish. Aluminum is not an essential element to aquatic life. It has been identified as a stressor to fish because it inhibits ion regulation and respiration (US EPA 1988a). Baker et al. (1982) examined effects of aluminum on White Sucker *Catostomus commersonii* and Brook Trout *Salvelinus fontinalis* in streams and found beneficial effects of low concentrations of aluminum in low pH environments, but concentrations above saturation resulted in increased mortality of eggs and larvae. Slaninova et al. (2014) attributed a fish kill in a Czech Republic pond to acid induced release of aluminum and iron. Aluminum and iron bind to the gills of fish causing reduced oxygen uptake. Histological sampling of affected fish also showed damage to the liver, kidney and spleen. Species responses vary based on acid tolerances. Keinanen et al. (2000) documented tolerance to aluminum in Pike *Esox lucius*, while Roach *Rutilus rutilus* showed various responses to aluminum in low pH environments. Kroglund and Finstad (2003) observed declines in the Atlantic Salmon *Salmo salar* populations associated with reduced marine survival of smolts chronically exposed to aluminum in low pH waters. Monette et al. 2008 also found reduced osmoregulatory function in Atlantic Salmon smolts exposed to aluminum in low pH environments. Moore (1994) documented reduced ability of male Atlantic Salmon to cue in on spawning females in acidic habitats. No studies have evaluated potential effects of aluminum exposure in acidic environments on juvenile or adult life stages of alosine species. Klauda, et al. (1991b) reported increased mortality of American Shad eggs in low pH (5.7) bioassay treatments with 50-400 µg/L monomeric aluminum and reduced survival of larvae when 50 µg/L monomeric aluminum was added to bioassay test chambers. Klauda et al. (1991a) reported Blueback Herring larvae succumbed more quickly to pH of 5.7 with addition of 200-400 µg/L of monomeric aluminum.

We found that aluminum was present at all sites sampled in the water column at concentrations above the US EPA (1986) acute criteria for aluminum (750 µg/L; Figure 7). When we mapped the low pH sites with sites where aluminum was present, we saw an overlap in stations where low pH was observed and water column aluminum concentrations were above the US EPA (1986) acute criteria. However, we cannot infer toxicity since these measurements may not have been concurrent. Low pH and elevated aluminum concentrations were prevalent in the upper tidal Potomac River where there has been a considerable focus on recovering American Shad (Figure 7; Cummins 2016). Figure 8 shows concentrations of aluminum were above the freshwater acute criteria in sediment and biological tissues.

Klauda et al. (1991b) citing Bradford et al. (1968), indicated that iron, lead and zinc were toxic to American Shad eggs; we did not find additional reports of toxicity. However, US EPA reviewed potential toxicity of these metals to fish and invertebrates to develop freshwater and saltwater acute and chronic limits (US EPA 1986, 1988b).

Iron is the fourth most abundant element by weight in the earth's crust and naturally occurs in surface waters. In a comprehensive literature review, Vuori (1995) reported large variability in spatial-temporal iron concentrations and iron speciation in receiving waters, with concentrations mediated by redox and light conditions, pH, flow conditions and the type of dissolved organic matter present. In riverine environments, iron dominates in the particulate form and settles to the sediments, but can be resuspended in turbulent conditions, or oxidized with some fractions diffusing into the water column and others being mineralized and deposited in sediments (Davidson and DeVitre 1992). Vuori (1995) reported information on aquatic toxicity of iron is scarce, but studies that have been conducted implicate dissolved iron as a toxic agent. Peuranen et al. (1994) observed gill damage in young Brown Trout *Salmo trutta* when exposed to total iron concentrations of 2.0 mg/L in water with pH ranging from 5.0-6.0. In summarizing several bioassay studies, Vuori (1995) cited formation of iron precipitates on gills and eggs as the main factor contributing to mortality of Rainbow Trout *Oncorhynchus mykiss* and Fathead Minnow *Pimephales promelas* eggs in laboratory bioassays. Beyond effects on fish, Vuori (1995) also cited impacts of iron precipitate on macroinvertebrates and periphyton in streams. Bradford et al. (1968) observed mortality of American Shad eggs at iron concentrations exceeding 100 µg/L. US EPA issued a single ambient criterion of 1000 µg/L.

We evaluated the Chesapeake Bay program using the US EPA 1000 µg/L acute concentration for freshwater, choosing this value because it represented the highest water criterion. All water quality samples evaluated for iron exceeded the 1000 µg/L criterion (Figure 9). Several stations were in egg and larval habitat areas (Figure 9). In recent post construction monitoring of stream restoration projects, researchers observed localized formation of iron flocculate that could limit habitat quality in restored stream segments by clogging gills of macroinvertebrates and fish (Williams et al. 2016). Citizen scientists reported iron flocculate in anadromous spawning areas of Mattawoman Creek (Jim Long, Mattawoman Watershed Society, personal communication).

Lead is a naturally occurring element naturally found in small quantities. Because lead has been used in many applications, there are numerous anthropogenic sources of lead in the environment including combustion of leaded gasoline, mining, steel production, improper battery disposal, and crop enhancers (Michigan DEQ 2018). Deposition is generally through lead dust accumulation over time and concentrations from past industrial uses on a site can persist indefinitely. Lead is probably the best known toxic metal because of widespread public service campaigns to alert people to the dangers of lead based paints (Michigan DEQ 2018). We found few studies examining lead impacts on fish. Davies et al. (1976) demonstrated lead was toxic to Rainbow Trout at various life stages, with eggs being most sensitive. Hodson et al. (1976) observed increased levels of lead in the blood of Rainbow Trout as water pH decreased. Martinez et al. (2004) investigated sublethal effects of lead on the Lined Prochilod *Prochilodus lineatus* in South America and observed abrasions on the gills at low and high sublethal concentrations of lead. They also observed what they classified as adaptation at low sublethal lead concentrations and exhaustion at high concentrations. We could not find recent studies evaluating aquatic effects of lead toxicity, perhaps because implementation of regulatory measures resulted in declines in lead concentrations between 1976 and 1984 (Schmitt and Brumbaugh 1990). Lead concentrations greater than 100 µg/L were associated with high egg mortality in American Shad (Bradford et al. 1968). We examined the lead concentrations exceeding the highest US EPA (1985) water criteria (saltwater acute criterion of 210 µg/L); this concentration was exceeded at all stations sampled for water toxicity (Figure 10).

We examined sediment samples to determine if lead exceeded sediment quality guidelines recommended by Long and MacDonald (1998). All sediment sites exceeded the higher ERM guideline, with several sites overlapping egg and larval habitat (Figure 11). While lead is likely to be bound in the sediments, Figure 11 indicated that a contaminant source influenced sediments in these areas. There were handful of sites with elevated concentrations of lead in tissue. A few sites overlapped spawning habitat (Figure 11).

Zinc naturally occurs in surface waters as a byproduct of bedrock weathering. While background concentrations of zinc are low in fresh and saltwater environments (Wallace et al. 1983), it is the fourth most widely used metal for manufacturing of products such as paints, cosmetics, pharmaceuticals, inks, soaps and batteries (Cammarota 1980); it is highly mobile and elevated levels of zinc have been detected in receiving waters. It is an essential nutrient, so small concentrations of zinc are necessary to maintain the health of aquatic organisms. Given the use of zinc and its mobility, studies have focused on examining toxicity of zinc to fish and other aquatic organisms. In developing water quality standards for zinc, US EPA (1987), provided an extensive evaluation of studies that had been conducted to derive acute and chronic criteria for aquatic life. There was a large range of concentration related acute toxicity of zinc to aquatic organisms (US EPA 1987), likely attributable to a large variation in water hardness and biological tolerances. Nonetheless, using standard methods, US EPA adopted water quality criteria for zinc in freshwater and saltwater. American Shad egg mortality was attributed to zinc concentrations greater than 300 µg/L (Bradford et al. 1968). We examined the CBP toxics data and applied the highest effects concentration (300 µg/L limit identified by Bradford et al. 1968) and examined where the limit was exceeded. Like the other metals examined, the zinc criterion was exceeded at all sites (Figure 12). There were stations with high zinc concentrations in areas identified as spawning habitat (Figure 12). We also evaluated data according NOAA Sediment Quality guidelines (Long and MacDonald. 1998) and zinc was detected throughout the Bay at concentrations higher than the higher ERM guideline (Figure 13).

Sources of lead and zinc identified in the Toxic Release Inventory data were most prevalent on the western shore near alosine spawning areas (Figure 14). Sources of zinc were also located at several eastern shore spawning areas. Counties with greater number of Superfund sites were also more prevalent on the western shore in watersheds identified as alosine egg and larval habitat (Figure 14).

Total Suspended Solids - Citing a study conducted by Auld and Schubel (1978), Klauda et al. (1991a; 1991b) reported increased yolk-sac larval mortality of American Shad in bioassays where organisms were exposed to 100 mg/L total suspended solids for four days. Using Chesapeake Bay Program water quality data, we calculated the proportion of samples at a station exceeding the 100mg/L standard in the spring months of March to May (Figure 15). Exceedances of the TSS limit in alosine egg and larval habitat occurred at low frequency on the Western Shore (Figure 15).

Discussion

We compiled information on water quality and contaminant stressors that potentially limit American Shad and River Herring early life stages and evaluated available water quality and toxics data to assess the potential of various stressors to limit habitat. While several databases were available, spatial and temporal resolution was poor for evaluating spawning and larval nursery habitat of American Shad and Herring. Nonetheless, these data should be considered when planning restoration and management. Contaminants were ubiquitous in sediment; zinc and lead were more prevalent in urban areas. We emphasize this so that restoration planning carefully considers all potential stressors to develop effective strategies to restore habitat that supports all life cycles. In areas where stressors may have previously limited habitat and where present stressors like toxic releases could continue to limit habitat, studies should be conducted to assess the degree and persistence of a stressor before restoration is initiated. Without this, investments in restoration may not result in better fish habitat.

While we only presented data on four contaminants that bioassays had been conducted with American Shad and Herring, there are many studies documenting effects of other contaminants on other fish that should be considered in evaluating habitat. Though many of these studies may not be directly related to our target species, they may be all that is available to illustrate potential for toxics to limit target species. Dwyer et al. (2000), attempted to conduct bioassay studies on American Shad to evaluate toxicity of various contaminants. Controls experienced excessive mortality and could not offer statistically viable estimates, but results from bioassays on Fathead Minnows and Rainbow Trout could be applied until bioassay methods were improved for American Shad (Dwyer et al. 2000).

In an extensive review to develop habitat criteria for American Shad and Herring, Klauda et al. (1991a; 1991b) assessed stressors by life stage that could limit habitat suitability. Maryland Sea Grant (2011) developed ecosystem based fisheries management approaches for alsoines in Chesapeake Bay. Changes in flow, contaminants, low pH, rapid temperature change, increased impervious surface, sediments and blockages as factors that could limit spawning, egg and larval habitat. Harvest and bycatch were identified as factors limiting adult populations (Klauda et al. 1991a; 1991b; Maryland Sea Grant 2011). Fishing is probably the only anthropogenic source of mortality that is density-dependent (Boreman 1997). Remaining stressors, such as pollution from development, flow changes, and sediment, are insensitive to the abundance of the animals they are killing (fishery-independent; Boreman 1997) and can overwhelm the intended benefits of fishing reductions. An egg-per-recruit model for Chesapeake Bay yellow perch that included an impact of development on eggs and larvae indicated that fishery-independent aspects of mortality overwhelmed density-dependent compensation from reducing F with the progression of suburban development (Uphoff et al. 2015). A similar exercise with Blueback Herring indicated they were more sensitive to disturbance of spawning and larval habitat than Yellow Perch (J. Uphoff, MD DNR, personal communication).

We constructed a conceptual diagram representing alosine life history with stressors by life stage to visualize the potential pressures by life stage these populations face (Figure 16). Stressors of juvenile and adults were harvest, bycatch, predation, and dams and blockages to migration and emigration of juveniles and adults. Stressors of eggs and larvae were development, pH, contaminants, flow, rapid temperature changes, and sediment (Figure 16).

We have documented declines in spawning habitat occupation by herring as development increased beyond 10% impervious cover (Uphoff et al. 2017). Limburg and Schmidt (1990) had documented a strong effect of urbanization on anadromous fish spawning in the Hudson River. Uphoff et al. (2017; also see Job 1, Section 1) related declines in presence of herring eggs and larvae with increased development. Analyses in Job 1, Section 1 indicated significant and logical relationships among the percent of plankton net samples with eggs and larvae (P_{herr}), watershed development, and conductivity (salinization of freshwater) consistent with the hypothesis that urbanization was detrimental to stream spawning. Conductivity was positively related with development in our analysis and with urbanization in other studies. Herring spawning declined and became more variable in streams as watersheds developed. General development reference points (structures per hectare or impervious surface) worked reasonably well in characterizing habitat conditions for stream spawning of Herring and we have used these to develop maps to guide habitat management.

Approximately 38% of anadromous fish stream spawning habitat in eastern Maryland (east of Chesapeake Bay) was high priority, 43% was mid-priority, and 19% was low priority at 12-digit resolution (Uphoff et al. 2014). West of Chesapeake Bay, 21% of anadromous fish stream spawning habitat was in the high priority category, 21% was mid-priority, and 58% was identified as low priority. In the western section, much of the low and mid-priority anadromous fish spawning habitat (37% at 12- digit) coincided with impervious surface of 10% or more. This was not the case in the eastern section (6%

at 12-digit with > 10% IS) and high salinity gradients would be the main spawning habitat limitation (Uphoff et al. 2014).

Flow change may stress early life stages through excessive velocity that flushes eggs and larvae into hostile habitats and through reduced velocity that can promote low oxygen or cause eggs and prolarvae to sink and become smothered by sediment (Maryland Sea Grant 2011). Tuckey (2009) observed exponential growth of juveniles related to spring flow during early life stages. Maryland Sea Grant (2011) cited additional studies that highlighted timing of flows as key factors in survival of early life stages of alosines with high flow and low temperature potentially impairing larval survival, while high summer flows can enhance juvenile survival. Leach and Houde (1999) and Crecco and Savoy (1995) reported lower pH and temperature driven by high flow can also reduce survival of early life stages. Tuckey and Sadzinski (2011) attributed flow effects to climate driven changes affecting flow timing and magnitude, direct withdrawal for water use functions, dam regulation of flows and watershed development. Uphoff et al. (2010) found changes in flow in Piscataway and Mattawoman creeks were correlated with watershed urbanization and documented an attendant decline in Herring spawning.

Years of high spring discharge favored American Shad recruitment in Chesapeake Bay and may have represented episodes of hydrologic transport of accumulated organic matter from watersheds that fuel zooplankton production and feeding success (McClain et al. 2003; Hoffman et al. 2007). Under natural conditions, riparian marshes and forests would provide organic matter subsidies in high discharge years (Hoffman et al. 2007), while phytoplankton would be the primary source of organic matter in years of lesser flow. Stable isotope signatures of York River American Shad larvae and zooplankton indicated that terrestrial organic matter largely supported one of its most successful year-classes. Lesser year-classes of American Shad on the York River were associated with low flows, organic matter based on phytoplankton, and lesser zooplankton production (Hoffman et al. 2007). The presence of organic matter was negatively related to development in eight Chesapeake Bay subestuaries (Uphoff et al. 2017).

Notably absent from the list of fishery-independent stressors was low oxygen as a limiting factor for American Shad and Herring larvae and juveniles in Maryland (Uphoff et al. 2017). Low dissolved oxygen measurements (< 3 mg/L) were not encountered in larval habitat of American Shad. More than 98% of dissolved oxygen measurements in juvenile habitat were greater than 3 mg/L (marginal habitat; Uphoff et al. 2017). Reducing nutrients to restore main-Bay dissolved oxygen is the major habitat restoration strategy of the Chesapeake Bay Program.

Management of American Shad and Herring in Maryland, under Chesapeake Bay Program and ASMFC guidelines, consists of harvest closures, stocking, and removing physical stream blockages (http://dnr.maryland.gov/fisheries/Documents/Section_2_American_Shad_Herring.pdf). Harvest closures and stocking attempt to bolster abundance, while physical removal of blockages presumes that habitat quantity for spawning is limiting to the spawning population.

Directed harvest of American Shad and Herring has been greatly reduced by elimination or restriction of in-river and ocean intercept fisheries. Maryland's bay-wide juvenile indices of American Shad and Herring suggest at least partial recovery in the four large subestuaries (primarily rural watersheds) they are derived from. Returning adults do not appear able to support fisheries. This leaves bycatch and predation on immature fish as they leave the Chesapeake Bay and reside in the ocean, or reductions in productive ocean conditions as the leading candidates for removals that short-circuit fishery recovery.

Physical barriers encountered in spawning migrations include dams, culverts and poorly constructed road crossings that obviously restrict upstream movement of adult migration to spawning grounds (Maryland Sea Grant 2011). In a study to evaluate the impact of blockages in Maine, Hall et al. (2011) attributed effects of blockages to drastic Herring population declines. Beyond impeding spawning migration, they cited dams as an indicator of large scale landscape changes that fragmented habitat and altered landscape ecology affecting terrestrial and aquatic species. Recognition of the effects of migratory barriers prompted management initiatives to construct passageways that restore migration corridors. However, Castro-Santos and Letcher (2010) reported a need to also address adult downstream migration to conserve iteroparity of the species. There has been significant management action in Chesapeake Bay to address migratory barriers by providing passage, with over 1,200 miles of stream opened since 2011 (Chesapeake Bay Program 2018). Burdick and Hightower (2006) reported an increase in spawning habitat use in a reach of the Neuse River, North Carolina after a dam was removed, noting that successful recovery of spawning habitat can occur as long as flow is adequate to provide access to previously blocked habitat. However, they did recognize that recovery will be successful when all aspects that limit early life stage survival are addressed (Burdick and Hightower 2006). A recent study conducted by Brown et al. (2013) evaluated American Shad recovery in response to fish passage at stream impediments. They observed a very low percentage of American Shad using the passageway and concluded that blockage removal is the approach to meet stock recovery goals.

There is limited understanding of chemical barriers that could limit spawning habitat use by adult Herring and American Shad. Maryland Sea Grant (2011) identified physicochemical blockages to alosine spawning, including acute or chronic chemical, thermal or flow alterations. Changing stream chemistry may cause disorientation of spawning adults and disrupted upstream migration. Physiological details of spawning migration are not well described for our target species, but homing migrations in anadromous American Shad and Salmon have been connected with chemical composition, smell, and pH of spawning streams (Royce-Malmgren and Watson 1987; Dittman and Quinn 1996; Carruth et al. 2002). Fish passages may not work if chemical cues are badly altered.

While these mapping exercises can be helpful to visualize stressors and associated risks related to habitat management we urge caution in overly relying on them to promote restoration without understanding underlying mechanism responsible for declines, particularly in highly altered suburban and urban watersheds. To date, focus on nutrient and sediment reductions has dominated stormwater best management practices (BMP's) designed to restore stream functions. More recently, these practices have been evaluated to assess their ability to entrain contaminants (Schueler and Youngk 2015). While we endorse practices that remove stressors, we only have a limited understanding of what stressors are present, how they act to limit habitat and production, and how effective BMP's are at removing them. While BMP technology is ever evolving, we do issue caution in overly relying on them to sequester nutrients and contaminants. Both processes rely on availability of organic matter (carbon) to mediate chemical processes that reduce chemical and nutrient loads. Uphoff (2017; also see Job 1, Section 2) has shown that watershed based organic matter decreases with development in subestuaries and that lack organic matter may impact first-feeding fish larvae through zooplankton production, a process not considered in nutrient oriented BMPs. While new technologies are designed to subsidize organic matter, we ask if these BMPs can become saturated and lose their effectiveness? We are also concerned with long term stability of these restored features. If they effectively sequester toxics, are they armored against large storms that could potentially mobilize them, making them bioavailable in the future? And finally, we ask, if these challenges are addressed, what is the expected time frame for recovery, assuming these projects are not overwhelmed by the rest of the watershed and future changes to accommodate growth and human needs?

In our review, we have only considered stressors that have been directly implicated in limiting survival of various life stages of alosine species, but the potential for contaminants to be acting in the background to affect bioenergetics and habitat suitability through various mechanisms needs to be considered (Brooks et al. 2012). We could be spending money to restore habitat function for little to no return if we do not understand and address the suite of impaired ecological functions that are important for fish production. Urbanizing habitats present a perfect research laboratory for such studies. Looking at several watersheds representing a gradient of development and focusing on evaluating contaminants, could yield valuable insight into specific stressors operating to limit habitat and production that preclude successful recovery of key ecological functions. The toxics data base, though dated, can give insight into the level of stress the Bay has absorbed in the past. Yet without contemporary monitoring and data, we are left to guess if a watershed has recovered from these stresses or absorbed the insult through shifting ecological states. We recommend an aggressive review of literature in tandem with evaluation of data to examine potential for toxicity not just to fish, but to important ecological features to better understand cumulative impacts of stressors to the Bay and to target species.

In absence of such focus, an alternative approach would be to identify regionally productive watersheds and focusing aggressive management to maintain viable habitats (Bowden 2014). This is similar to our recommendation to prioritize conservation of productive habitats. This would eliminate the need for extensive habitat surveys that often lack resources for sound spatial and temporal monitoring needed to identify sources of stress (Bowden 2014). Monitoring presence of early life stages has allowed us to identify areas we believe to remain productive and resilient to natural variation, at least at the present level of stress (Uphoff et al. 2017).

Finally, we offer some recommendations that could improve alosine habitat assessment:

1. Use information from hatchery collections of American Shad to further define habitat. Records of occurrence of running ripe females could help refine spawning habitat in space and time.
2. Examine hatchery water quality criteria for alosines to support criteria development.
3. Examine hardness, alkalinity, and conductivity data to further define potentially toxic conditions. Hardness and alkalinity affect toxicity of trace

metals. High conductivity (beyond base levels; see Job 1, Section 1) may indicate habitat that has become less suitable.

4. Access other water quality data bases (USGS stream data).
5. Routinely monitor contaminants in nursery areas in the spring.

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Table 1. Toxicity criteria or guidelines reported for four media. Water criteria are US EPA Water Quality criteria standards for freshwater and saltwater toxicity (US EPA 1986). Biological concentrations are derived from Klauda et al. (1991a: 1991b). Sediment Quality Concentrations were adapted from NOAA guidelines (Long and MacDonald 1998).

	Freshwater		Saltwater		Biological	Sediment Quality	
	Acute (µg/L)	Chronic (µg/L)	Acute (µg/L)	Chronic (µg/L)	(µg/L)	ERL (ppm dry weight)	ERM (ppm dry weight)
Aluminum	*750*	87			50, 200		
Iron		*1000*			100		
Lead	65	2.5	*210*	8.1	100	46.7	*218*
Zinc	120	120	90	81	*300*	150	*410*

Table 2. Percentage of pH records meeting criteria for Shad and Herring eggs and larvae. Tidal data were evaluated for all stations with salinity less than 3‰ during the spring (March- May). Nontidal samples represent all stations sampled.

Species	**Life Stage**	**Criteria (pH)**	**Tidal Percentage Meeting**	**Nontidal Percentage Meeting**	**Criteria Source**
American Shad	Egg	> 6.0	98.4	100.0	Klauda et al. 1991b
	Larvae	> 6.7	93.4	98.0	Klauda et al. 1991b
Alewife	Egg	5.0-8.5	80.8	76.0	Klauda et al. 1991a
	Larvae	5.5-8.5	91.7	93.6	Klauda et al. 1991a
Blueback Herring	Egg	5.7-8.5 (suitable)	91.7	93.6	Klauda et al. 1991a
		6.0-8.0 (optimum)	80.5	76.0	Klauda et al. 1991a
	Larvae	6.2-8.5 (suitable)	91.1	93.6	Klauda et al. 1991a
		6.5-8.0 (optimum)	78.4	75.4	Klauda et al. 1991a

Figure 1. Baywide trends in juvenile indices of American Shad (A. shad), Alewife, Blueback Herring (B. Herring), and Striped Bass (S. Bass) in four major tributaries in Maryland's portion of Chesapeake Bay. Juvenile indices (geometric means) are standardized to their 1959-2018 means.

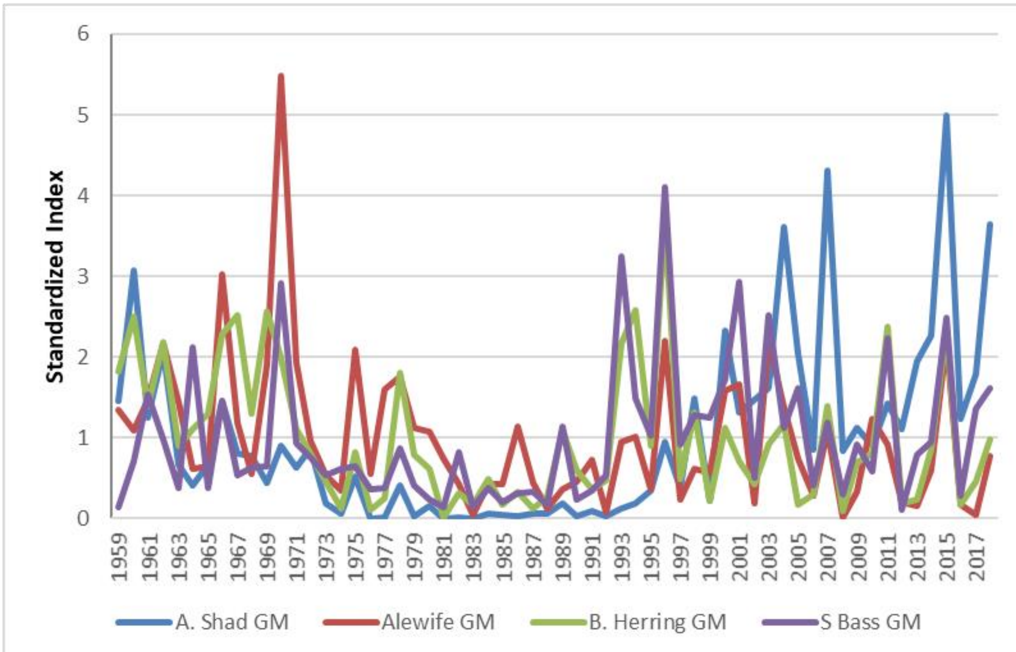


Figure 2. Alosine egg and larval habitat based on salinity distributions (Uphoff et al. 2014) and historical distributions reported by O'Dell et al. (1975; 1980).

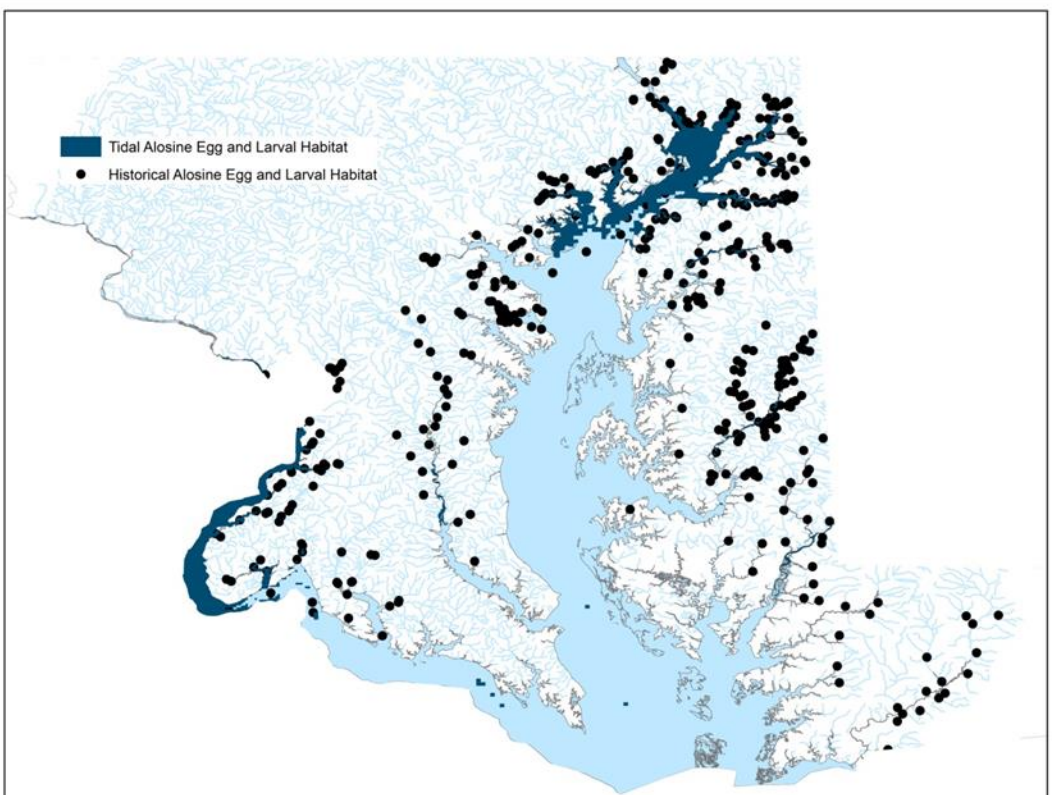


Figure 3. Chesapeake Bay Program water quality sampling stations in Maryland and the District of Columbia (http://datahub.chesapeakebay.net/WaterQuality).

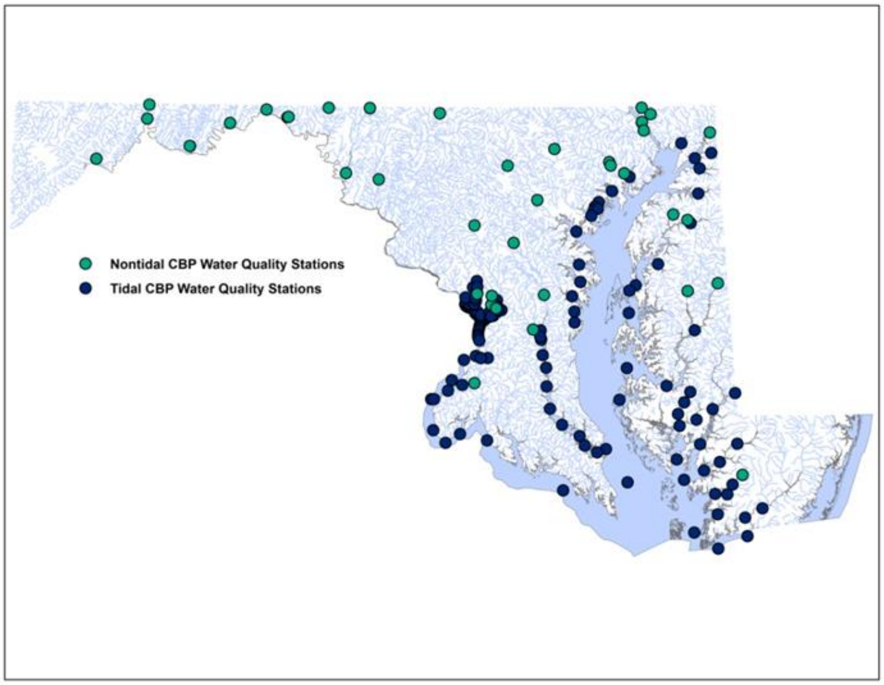


Figure 4. Chesapeake Bay Program toxic data base station locations (https://www.chesapeakebay.net/what/downloads/cbp_toxics_database).

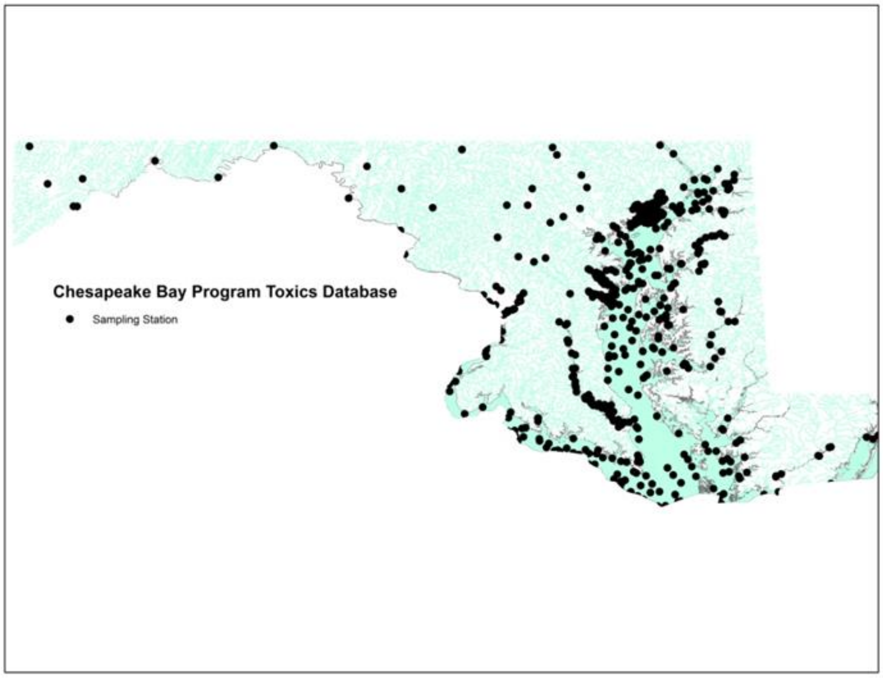


Figure 5. Environmental Protection Agency (EPA) Toxic Release Inventory locations (https://www.epa.gov/toxics-release-inventory-tri-program/tri-data-and-tools).

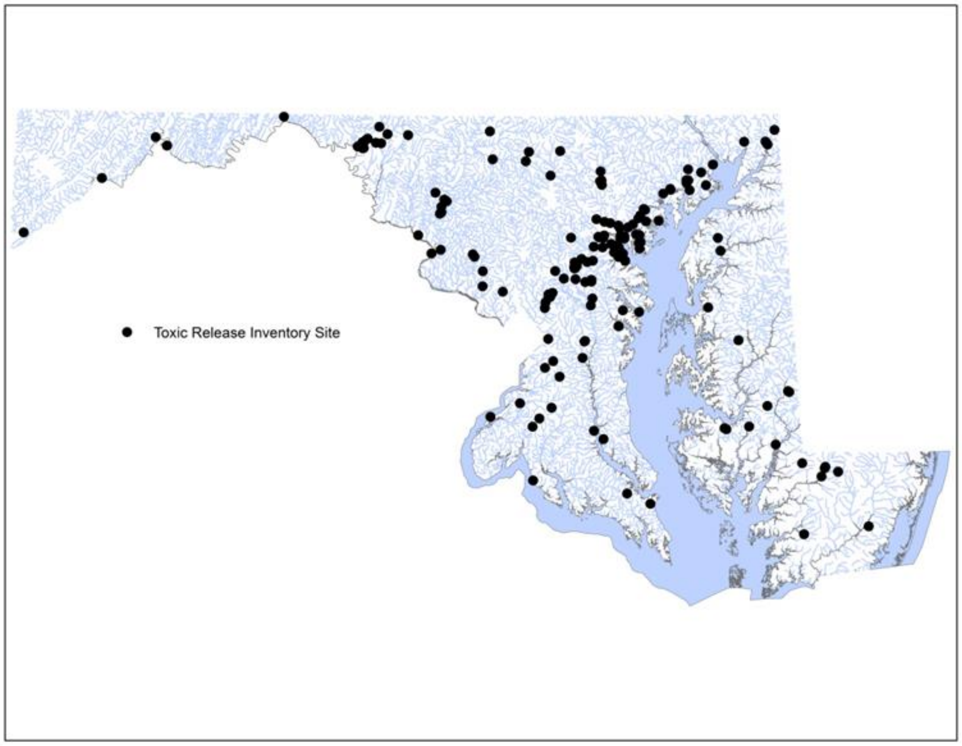


Figure 6. Alosine spawning and nursery areas with Chesapeake Bay Program water quality stations where pH was less than 5.7 at anytime during the spring (March-May), 1985 -2017.

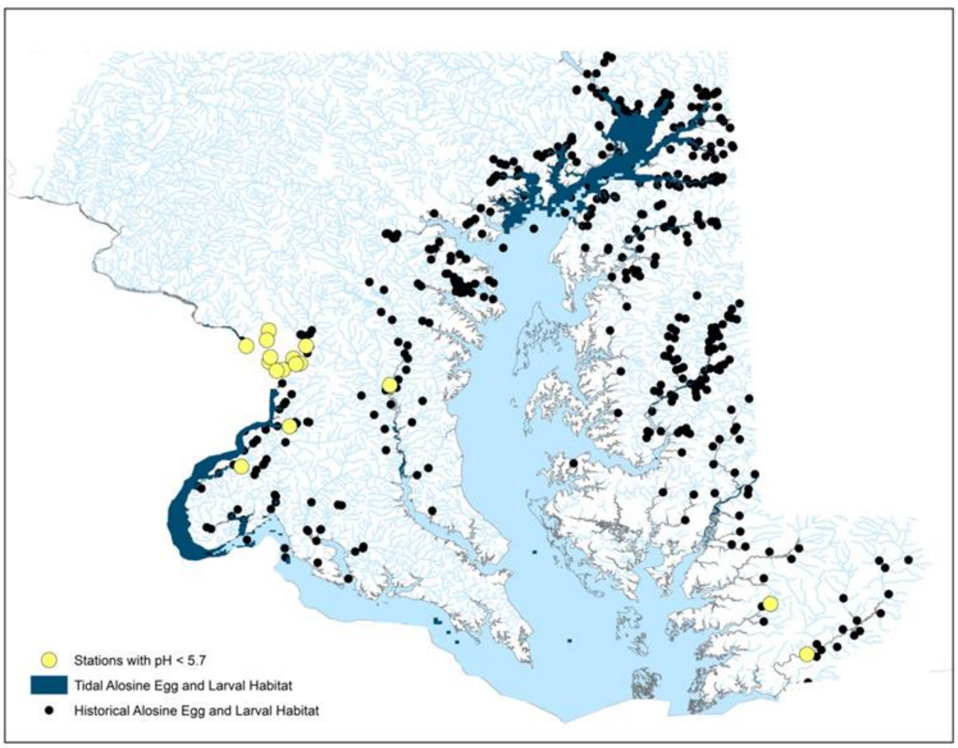


Figure 7. Chesapeake Bay Toxicity data stations where water column concentrations for aluminum exceeded the highest criterion proposed, the EPA water quality acute criteria of 750 ugl/L (Table 1) and alosine egg and larval habitat

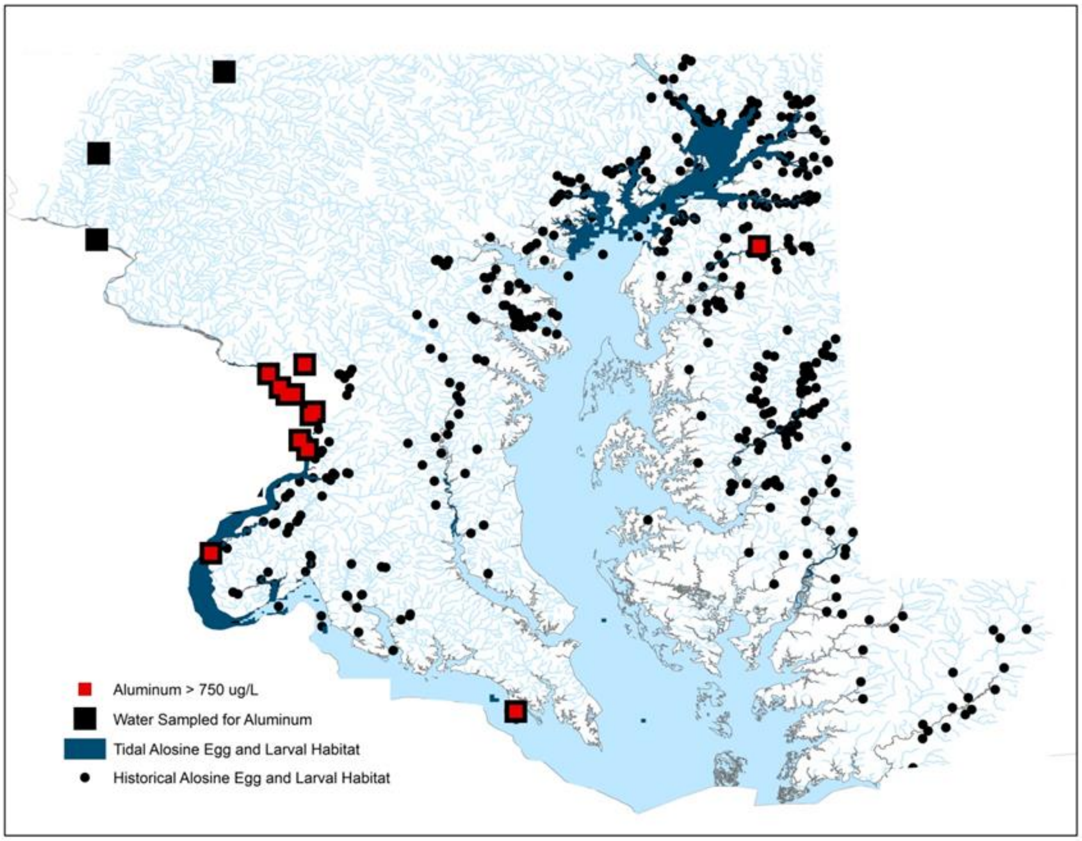
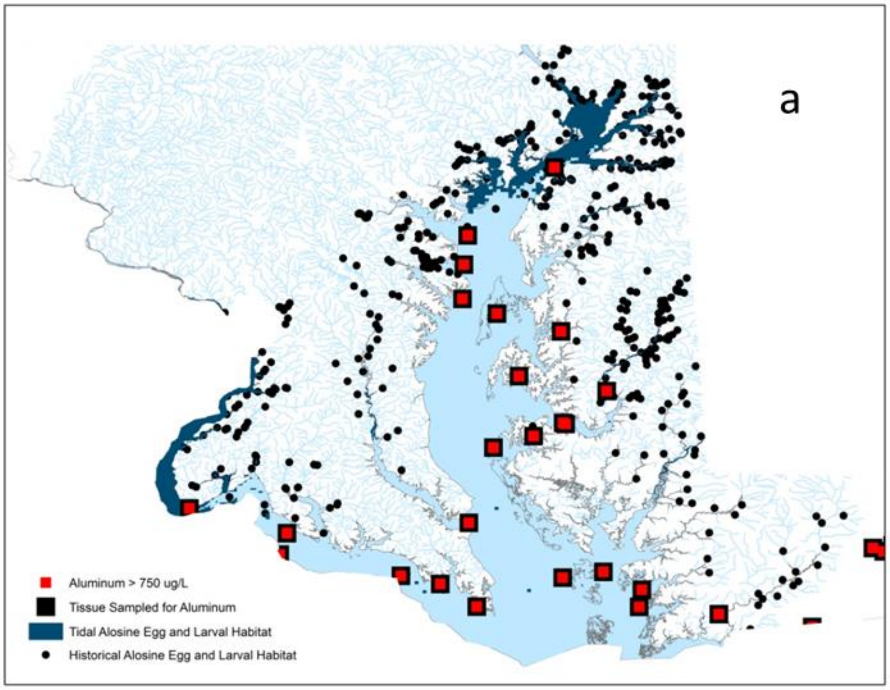


Figure 8. Alosine spawning and nursery areas with Chesapeake Bay Program Toxicity stations where sediment aluminum concentrations (a) and biological tissue samples (b) exceeded the USEPA Water Quality Chronic criterion of 750 ug/l (USEPA, 1986).



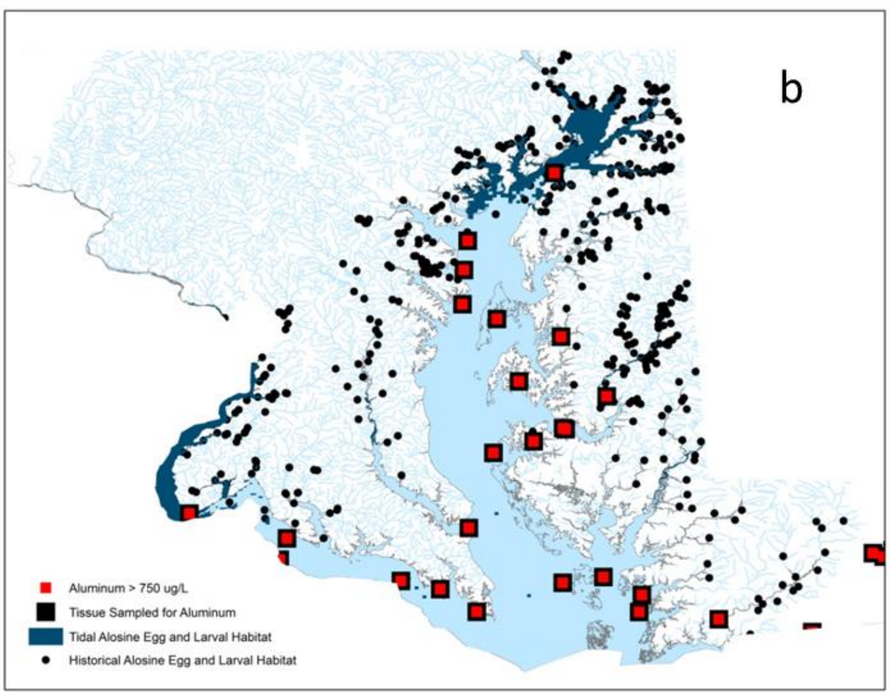


Figure 9. Alosine spawning and nursery areas with Chesapeake Bay Program water quality stations where iron concentration exceeded the EPA Water Quality Chronic criterion of 1000 ug/l (USEPA, 1986).

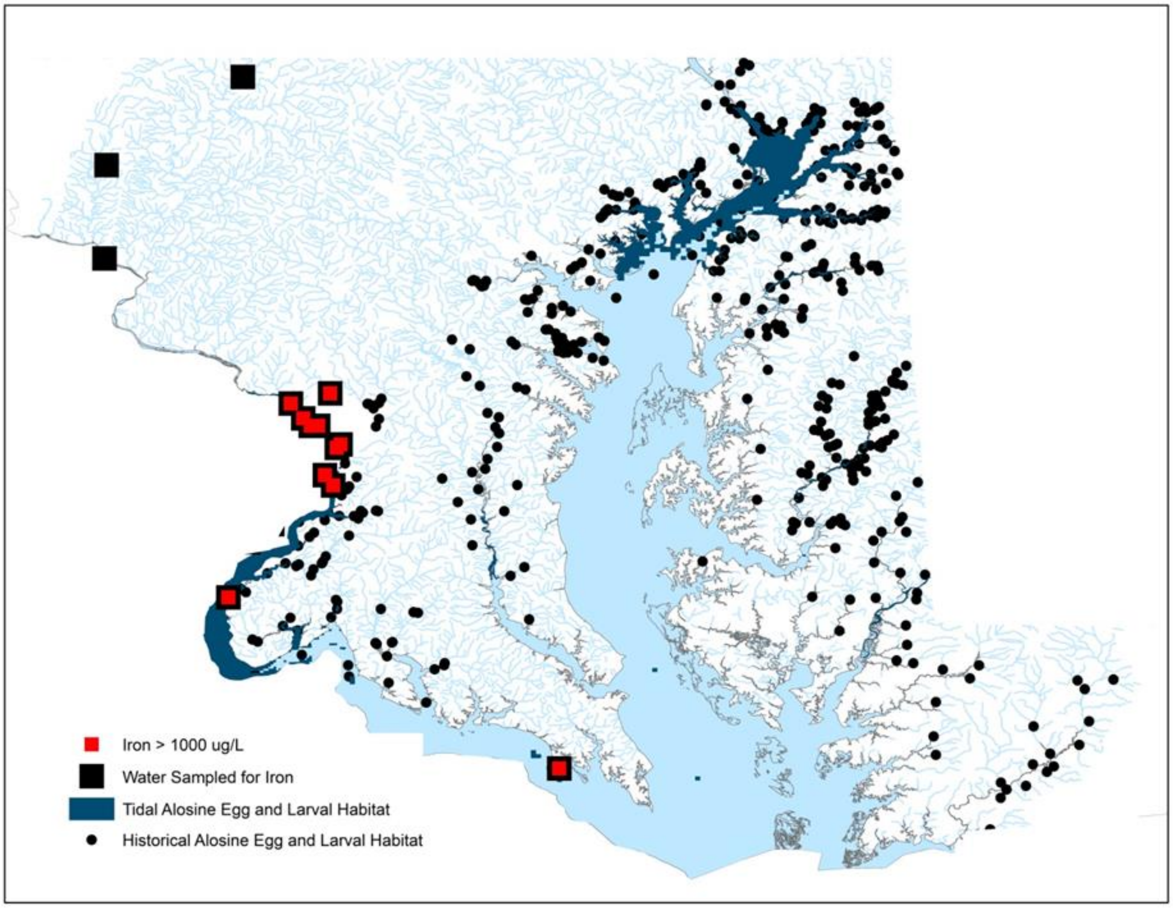


Figure 10. Alosine spawning and larval habitat, and stations where water concentrations of lead exceeded USEPA (1986) highest water criterion (saltwater acute criterion of 210 ug/L).

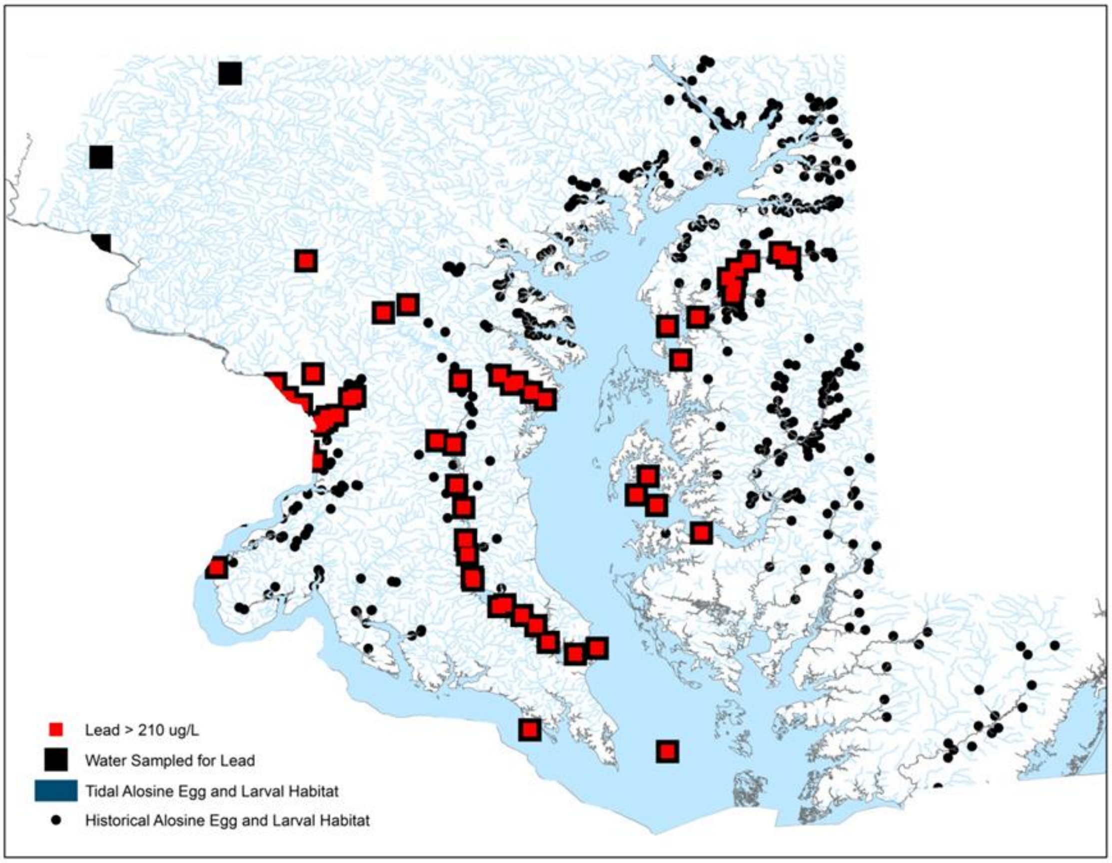
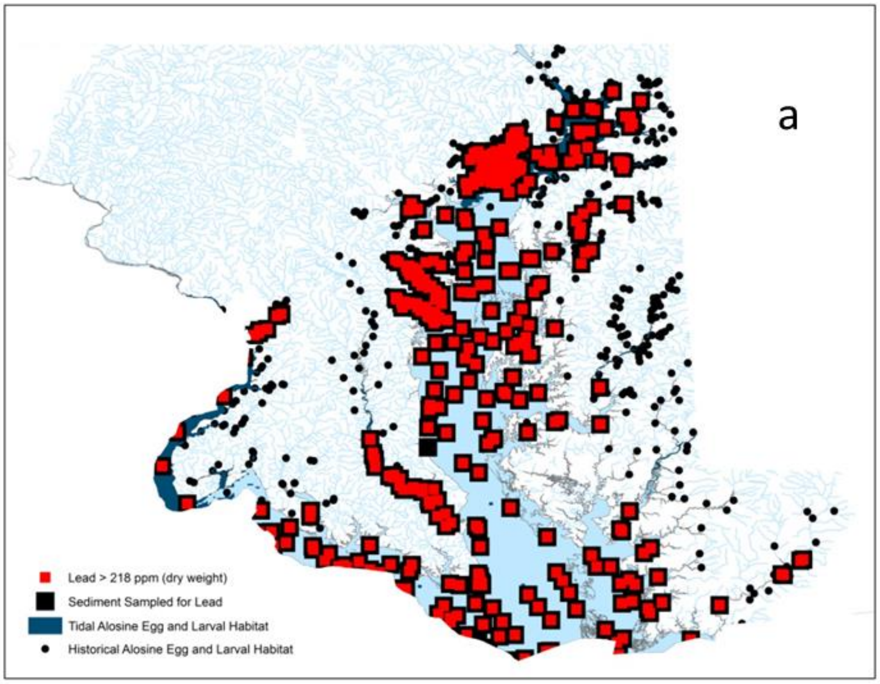


Figure 11. Alosine spawning and larval habitat, and stations where sediment concentrations of lead exceeded NOAA (Long et al. 1998) ERM guidelines for toxicity. (a) All stations sampled exceeded the guidelines with several stations overlapping spawning habitat. (b) Stations where biological tissue samples had concentrations of lead greater than the 100 ug/L USEPA water quality acute concentration show that lead was present in all fish and shellfish sampled.



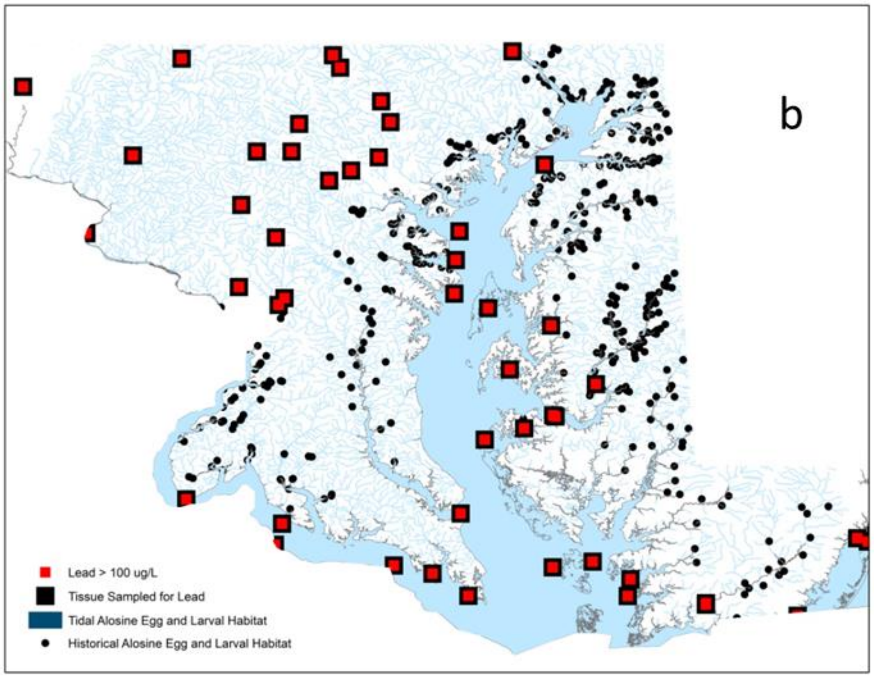


Figure 12. Alosine spawning and larval habitat, and stations where water concentrations of zinc exceeded the highest water criterion (300 ug/L limit reported by Bradford et al. (1968) that was toxic to American Shad eggs). All stations sampled exceeded the criteria, with numerous stations overlapping Alosine egg and larval habitat.

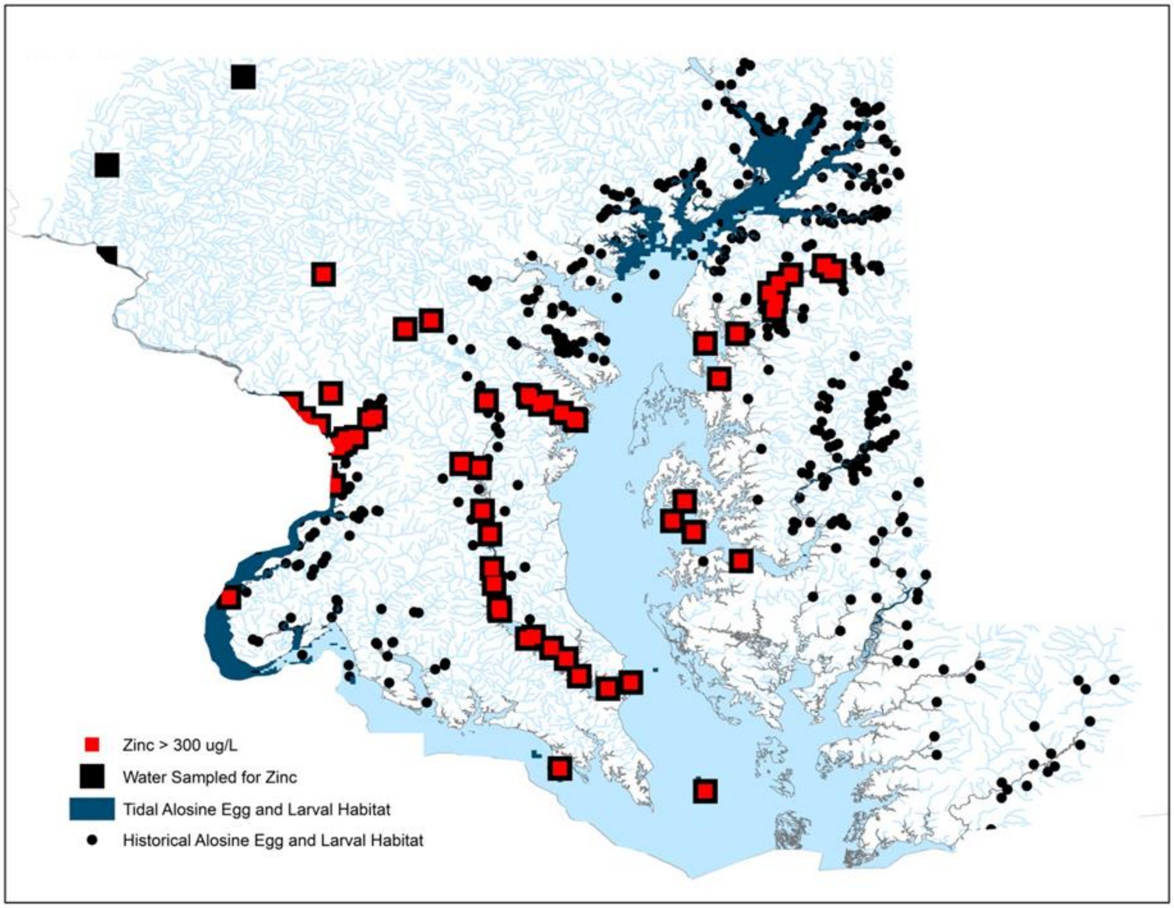


Figure 13. Alosine spawning and larval habitat, and (a) stations where sediment concentrations of zinc exceeded NOAA (1998) ERM guidelines for toxicity and (b) stations where biological tissue samples had concentrations of zinc greater than the 300 ug/L USEPA water quality acute concentration.

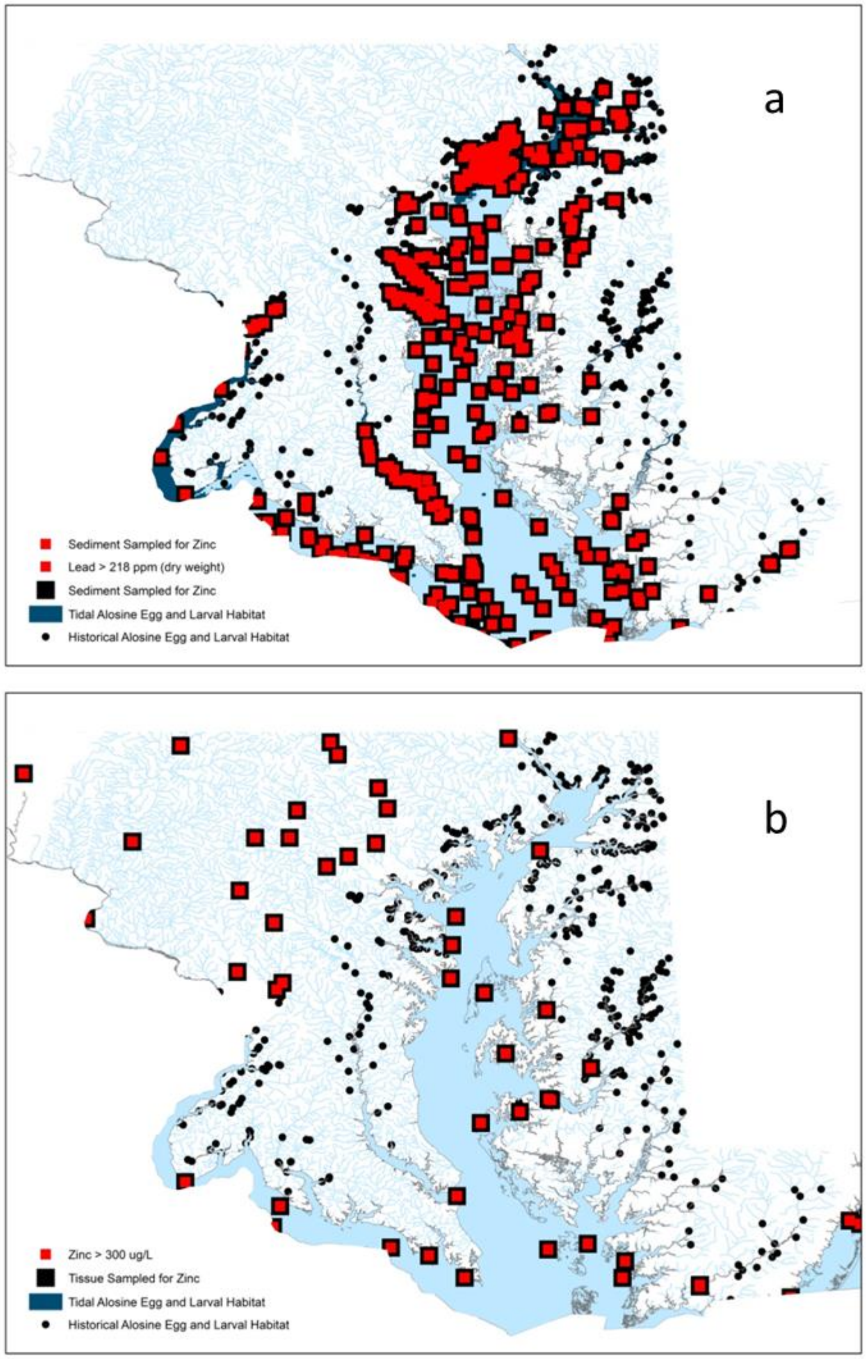


Figure 14. Alosine spawning and larvae habitat with potential sources of lead and zinc from industry releases along with number of superfund sites by county.

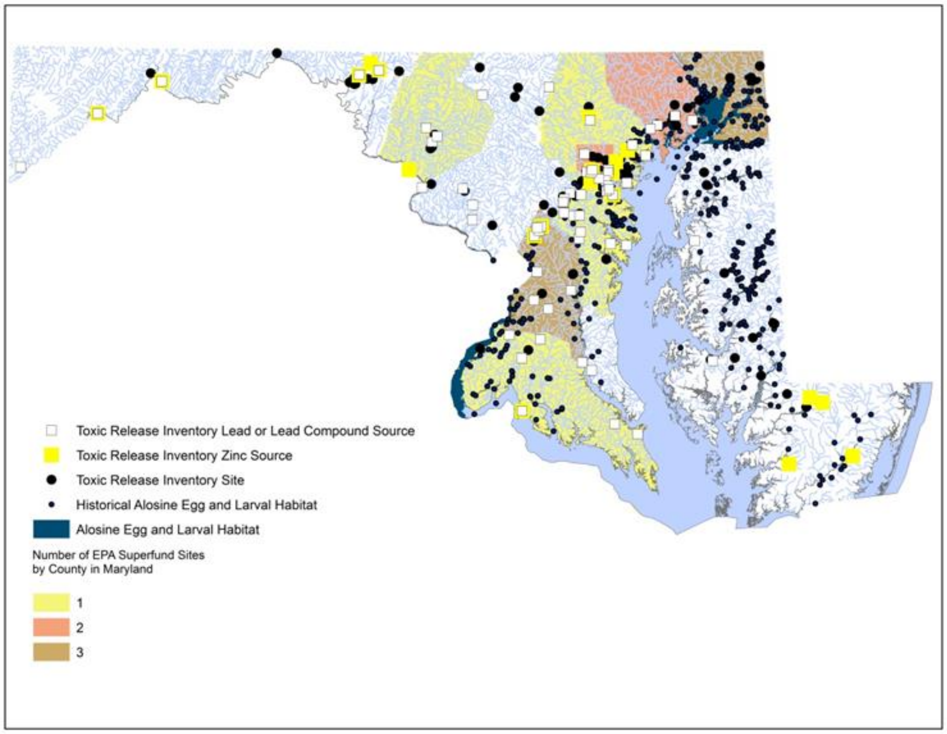


Figure 15. Tidal and nontidal proportion of spring samples with TSS > 100 mg/L in American Shad spawning and larvae habitat.

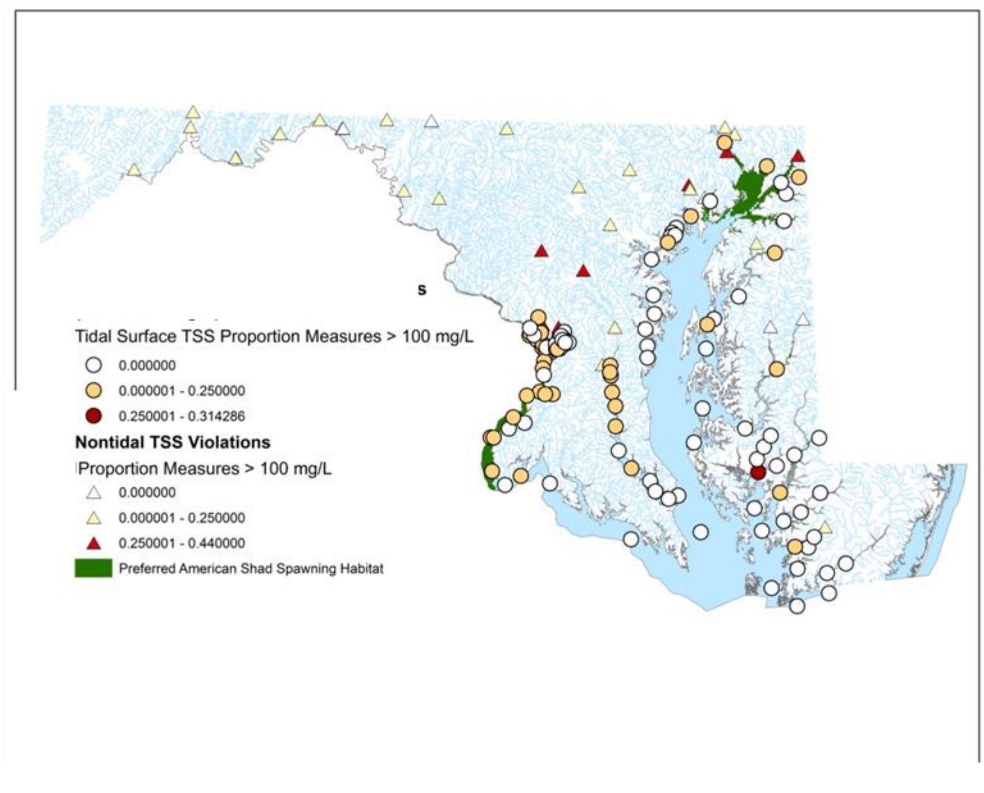
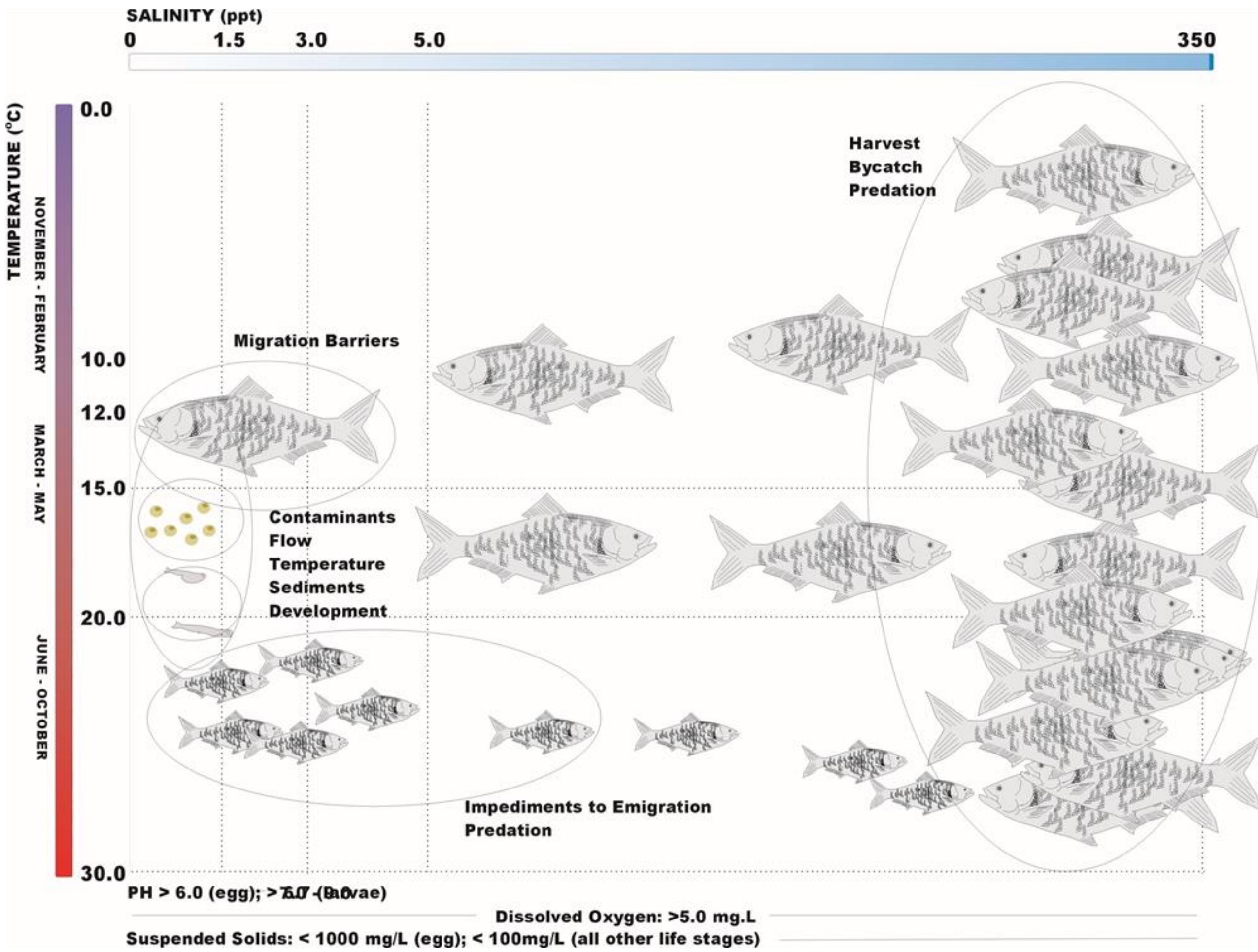


Figure 16. Alosine life history and habitat requirements with stressors identified by life stage.



Job 4: Development of ecosystem-based reference points for recreationally important Chesapeake Bay fishes of special concern: Striped Bass nutrition and forage availability benchmarks

Jim Uphoff, Alexis Park, Carrie Hoover, Ben Wahle

Executive Summary

Maryland's fisheries managers and stakeholders want to know whether there is enough forage to support Striped Bass in Maryland's portion of Chesapeake Bay (hereafter, upper Bay). Formal assessments of abundance and biomass of Striped Bass and most forage species in upper Bay are lacking due to cost and difficulty in mathematically separating migration from mortality. Past efforts to launch ecosystem based fisheries management in Chesapeake Bay have been comprehensive and complex, but have not resulted in integration into management. An index-based (Index of Forage or IF) approach could integrate forage into Maryland's resident Striped Bass management at low complexity and cost. The IF represents a framework for condensing complex ecological information so that it can be communicated simply to decision makers and stakeholders.

Monitoring of Striped Bass health (1998-2017), relative abundance (1983-2017), natural mortality (1986-2017), and forage relative abundance in surveys (1959-2017) and fall diets of Striped Bass (1998-2000 and 2006-2017) provided indicators to assess forage status and Striped Bass well-being in Maryland's portion of Chesapeake Bay. A Striped Bass recreational catch per trip index provided an index of relative abundance. Forage-to-Striped Bass ratios (focal prey species are Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab) and proportion of Striped Bass in fall with empty guts provided trends in prey supply relative to predator demand based on relative abundance and diet sampling, respectively. Proportion of resident Striped Bass without visible body fat and an index of natural mortality based survival were indicators of Striped Bass well-being. The proportion of Striped Bass without body fat, anchored our approach, providing a measure of condition and potential for starvation that was well-related to feeding of Striped Bass in the laboratory. Statistical analyses provided evidence that forage and Striped Bass abundance and well-being were inter-related. Analyses were split into two size classes, small (<457 mm TL) and large (≥ 457 mm TL), due to sampling considerations. The small class was most sensitive to forage and indicators were mostly based on it.

Targets and thresholds were then developed for each of these indicators to assign them scores. A score of 1 indicated threshold conditions; a score of 5 indicated target conditions; and scores of 2-4 indicated grades between. Time-periods where body fat indicators (1998-2017) were at target or threshold conditions provided a time-frame for evaluating other indicators. Annual scores for each variable were averaged for a combined annual IF score.

The IF indicated poor foraging conditions during 1998-2004 and improvement after. Best IF scores occurred during 2008-2010 (~4 and above) and all components reached their targets in 2010. The IF has fallen to between 2 and 3 (near or avoids threshold in 2015-2017. High variability in component scores was evident as IF improved after 2004 that may have reflected sampling issues, nonlinear relationships

among variables, lagged responses, potential insensitivity of some indices, behavioral changes that could increase feeding efficiency, influence of episodes of good foraging conditions outside of those monitored in fall, and combinations of the above.

Introduction

The Chesapeake Bay stock of Striped Bass *Morone saxatilis* supports major commercial and recreational fisheries within Chesapeake Bay and along the Atlantic coast of the United States (Richards and Rago 1999; Maryland Sea Grant 2009). Striped Bass, fueled by a series of strong year-classes in Chesapeake Bay, were abundant in the 1960s and early 1970s, then declined as recruitment faltered and fishing mortality rates increased (Richards and Rago 1999). A nadir for spawning stock was reached in the early 1980s (Uphoff 1997). Moratoria were imposed in several Mid-Atlantic States in the mid-to-late 1980s and conservative regulations were put in place elsewhere (Uphoff 1997; Richards and Rago 1999). Recovery of Atlantic coast Striped Bass was declared in 1995 after rapid stock growth between1982 and 1994 (Richards and Rago 1999; ASMFC 2016).

Striped Bass appear capable of limiting prey populations along the Atlantic coast and in its estuaries and concern emerged about the impact of high Striped Bass population size on its prey-base shortly after recovery (Hartman 2003; Hartman 2003; Uphoff 2003; Savoy and Crecco 2004; Heimbuch 2008; Davis et al. 2012; Overton et al. 2015). Major declines in abundance of important prey (Bay Anchovy *Anchoa mitchilli.* Atlantic Menhaden *Brevoortia tyrannus*, and Spot *Leiostomus xanthurus*) in Maryland's portion of Chesapeake Bay (hereafter upper Bay) coincided with recovery (Uphoff 2003; Overton et al. 2015). Maintaining a stable predator-prey base is a challenge for managing Striped Bass in lakes and poor condition is a common problem when supply decreases (Axon and Whitehurst 1985; Matthews et al. 1988; Cyterski and Ney 2005; Raborn et al. 2007; Sutton et al. 2013; Wilson et al. 2013).

A large contingent of Chesapeake Bay Striped Bass that do not participate in the Atlantic coast migration (mostly males along with some young, immature females; Setzler et al. 1980; Kohlenstein 1981; Dorazio et al. 1994; Secor and Piccoli 2007) constitute a year-round population of predators that provides Maryland's major recreational fishery and an important commercial fishery (Maryland Sea Grant 2009). Reports of Striped Bass in poor condition and with ulcerative lesions increased in Chesapeake Bay shortly after recovery and linkage of these phenomena and poor feeding success on Atlantic Menhaden and other prey was considered plausible (Overton et al. 2003; Uphoff 2003; Gauthier et al. 2008; Overton et al. 2015). Mycobacteriosis, a chronic wasting disease, became an epizootic in Chesapeake Bay in the late 1990s and was concurrent with lesions and poor condition (Overton et al. 2003; Jiang et al. 2007; Gauthier et al. 2008; Jacobs et al. 2009b). Challenge experiments with Striped Bass linked nutrition with progression and severity of the disease, and reduced survival (Jacobs et al. 2009a). Tagging models indicated that annual instantaneous natural mortality rate (M) of large sized Striped Bass in Chesapeake Bay increased substantially during the mid-1990s while instantaneous fishing mortality rate (F) remained low (Jiang et al. 2007; ASMFC 2013). Prevalence of mycobacteriosis and natural mortality appear to be less outside Chesapeake Bay (Matsche et al. 2010; ASMFC 2013).

Maryland's fisheries managers and stakeholders want to know whether there is enough forage to support Striped Bass in upper Bay. Formal assessments of abundance and biomass of Striped Bass and most forage species in upper Bay are lacking due to cost and difficulty in mathematically separating migration from mortality. The Atlantic States Marine Fisheries Commission (ASMFC) is moving to develop reference points for Atlantic Menhaden's forage role along the Atlantic coast and Striped Bass is a predator of concern (SEDAR 2015). In 2014, a forage fish outcome was included in the Chesapeake Bay Agreement (Chesapeake Bay Program): "By 2016, develop a strategy for assessing the forage fish base available as food for predatory species in the Chesapeake Bay." Resident Striped Bass offered an immediate opportunity to develop an indicator-based assessment approach based on existing monitoring.

Indicators based on monitoring, such as forage indices, prey-predator ratios, Striped Bass condition indices, and prey abundance in diet samples have been suggested as a basis for forage assessment (Maryland Sea Grant 2009; SEDAR 2015) and formed the foundation of our approach. Indicators are widely used for environmental reporting, research, and management support (Rice 2003; Jennings 2005; Fogarty 2014).

The IF approach was based on a suite of statistically linked indicators. Status would be judged by whether target or threshold reference points were met for each indicator. Time-periods where body fat indicators (1998-2017) were available provided a time-frame for developing targets and thresholds for other indicators. Targets and limits based on historical performance are desirable because they are based on experience and easily understood (Hilborn and Stokes 2010).

Uphoff et al. (2014) devised five annual forage indicators for resident Striped Bass in Maryland's portion of Chesapeake Bay. A Striped Bass recreational catch per trip index provided an index of relative abundance (demand). A forage-to-Striped Bass ratio (focal species combined) and grams of all forage consumed per gram of Striped Bass in fall provided trends in supply relative to demand based on relative abundance and diet sampling, respectively. Proportion of resident Striped Bass without visible body fat and an index of natural mortality based survival were indicators of Striped Bass well-being. Statistical analyses provided evidence that forage and Striped Bass abundance and well-being were inter-related (Uphoff et al. 2013; 2014; 2015; 2016). Targets and thresholds were then developed for each of these indicators to assign them scores. A score of 1 indicated threshold conditions; a score of 5 indicated target conditions; and scores of 2-4 indicate grades between (Uphoff et al. 2014). This report provides a complete set of indicators through 2017. Some indicators were revised and all were summarized into a single score to serve as a quick reference for managers and the public.

A nutritional indicator, proportion of Striped Bass without body fat, anchored our approach, providing a measure of condition and potential for starvation that was well-related to feeding of Striped Bass in the laboratory (Jacobs et al. 2013). Lipids are the source of metabolic energy for growth, reproduction, and swimming for fish and relate strongly to foraging success, subsequent fish health, and survival of individual fish and fish populations (Tocher 2003; Jacobs et al. 2013).

While upper Bay Striped Bass feed on a wide range of prey, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab *Callinectes sapidus* have consistently accounted for most annual diet biomass in Chesapeake Bay studies (Hartman and Brandt 1995c; Griffin and Margraf 2003; Walter et al. 2003; Overton et al. 2009; Overton et al. 2015). We selected these species as focal prey (major prey) for forage indices. Indices of major prey availability were estimated from fishery-independent surveys and fall diets of Striped Bass. Trends in prey index-to-Striped Bass index ratios were examined for each focal prey since forage indices alone would not consider the possibility of predator interference or the vulnerability exchange process of foraging arena theory (Ginzburg and Akçakaya 1992; Yodzis 1994; Ulltang 1996; Uphoff 2003; Walters and Martell 2004; Walters et al. 2016).

The ratio of age-3 relative abundance of male Striped Bass in spring spawning ground gill net surveys (Versak 2015) to their year-class-specific juvenile indices (Durell and Weedon 2017) were used as indicators of change in survival due to natural mortality (SR) prior to recruitment to the fishery (Uphoff et al. 2015; 2016). Confining the gill net relative abundance indices to 3 year-old males makes it likely that trends in SR will reflect resident Striped Bass survival before harvest (i.e., M). Males are completely mature at age-3 (nearly all females mature at older ages), so they would be fully recruited to the gill net survey (Maryland Sea Grant 2009). Age-3 males in the spring gill net survey were nearly always well below minimum length limits in years when harvest was allowed (Versak 2015), but they could be subject to catch-and-release mortality. We expected SR to vary without trend if natural mortality (M) remained constant. It became apparent that SR estimates used in Uphoff et al. (2015) were biased because age-3 gill net indices were not reflecting expected trends in abundance of age-3 fish indicated by the stock assessment, juvenile indices, and other indicators. Uphoff et al. (2016) developed adjusted gill net indices that reflected expected stock changes and used these as the numerator in the SR estimates. We have revisited the approach in Uphoff et al. (2016) in this report.

This year's report concentrated on developing CI's for ratio indicators using @Risk (Monte Carlo simulation software for Excel spreadsheets; Palisade Corporation 2016) to develop confidence intervals for these ratios. Uphoff et al. (2017) used a spreadsheet provided by Mary Christman (University of Florida**)** and Desmond Kahn (Fisheries Investigations Inc.**)** to estimate confidence intervals for the RI. Applying the spreadsheet to forage ratios was complicated by the use of geometric means for the forage indices and the simulations provided an alternative means of describing variability.

Methods

Definitions of abbreviations can be found in Table 1.

Nutritional status (condition) for upper Bay Striped Bass was estimated as the proportion of fish without visible body fat during October-November (P0; Jacobs et al. 2013). Body fat data have been collected by the Fish and Wildlife Health Program (FWHP) as part of comprehensive Striped Bass health monitoring in upper Bay initiated during an outbreak of lesions that began in the late 1990s. Fish were collected by hook-and-line from varying locations during fall, 1998-2016, between Baltimore, Maryland (northern boundary), and the Maryland-Virginia state line (southern boundary; Figure 1).

Estimates of P0 were made for two size classes of Striped Bass separately and combined: Striped Bass less than 457 mm total length (or TL; hereafter, small sized or small Striped Bass or fish) and fish 457 mm TL or larger (hereafter, large or large sized Striped Bass or fish). The small and large designations replace sublegal and legal sized designations used in previous reports; this change was made to prevent confusion that may arise due to length limit changes (the length limit was raised to 508 mm TL in 2015 and has been lowered to 483 mm TL in 2018). Standard deviations and confidence intervals (95%) of P0 were estimated using the normal distribution approximation of the binomial distribution (Ott 1977).

As Striped Bass experience starvation, lipids are replaced by water, conserving weight loss and hampering the interpretation of weight at length condition indices (Jacobs et al. 2013). Jacobs et al. (2013) presented a target for body moisture (25% or less of fish with starved status) as a surrogate for lipid content estimated from proximate composition of well-fed Striped Bass. This target was derived from fall 1990 field collections by Karahadian et al. (1995) - the only field samples available from favorable feeding conditions (high forage to Striped Bass ratios). A target for visible body fat was not presented in Jacobs et al. (2013) because the index was not applied in the 1990 collection. However, mean tissue lipid of Striped Bass without visible body fat was reported to be identical to that estimated from percent moisture in the remainder of the data set, meaning that P0 related strongly to the proportion exceeding the moisture criteria (Jacobs et al. 2013). A level of P0 of 0.30 or less was used to judge whether Striped Bass had fed successfully during October-November. Variation of tissue lipids estimated from body fat indices was greater than for moisture and the P0 target accounted for this additional variation plus a buffer for misjudging status (J. Jacobs, NOAA, personal communication). Jacobs et al. (2013) stressed that comparisons of Striped Bass body fat to a nutritional target or threshold in Chesapeake Bay should be based on October-November data since they were developed from samples during that time span. Uphoff et al. (2014) estimated the P0 threshold as 0.68 (average of the lower 95% CI of high Pf0 estimates during 1998-2004). Other indicators for condition were described in Jacobs et al. (2013), but P0 was chosen because it could be applied to data collected by Chesapeake Bay Ecological Foundation (CBEF) and CBEF P0 estimates were similar to those estimated from FWHP sampling (Uphoff et al. 2014).

We used geometric mean catches from fixed station seine and trawl surveys as indicators of relative abundance of most major prey species in upper Bay. A shoreline seine survey targeting age-0 Striped Bass during 1959-2016 provided indices for Atlantic Menhaden, Bay Anchovy, and Spot (Goodyear 1985; Richards and Rago 1999; Kimmel et al. 2012; Durell and Weedon 2016; Houde et al. 2016). Additional indices for Spot and Bay Anchovy were estimated from a Blue Crab trawl survey conducted during summer 1989-2017 (Uphoff 1998; Rickabaugh and Messer 2015; MD DNR 2017a; estimates were provided by H. Rickabaugh, MD DNR, personal communication). These surveys sampled major and minor tributaries, sounds adjacent to the mainstem upper Bay, but not the mainstem itself (Figure 1). Sampling occurred during summer through early fall. Density of juvenile Blue Crabs in a stratified random winter dredge survey (1989-2017) that sampled Chesapeake Bay-wide (Maryland and Virginia) was our indicator of Blue Crab relative abundance (Sharov et al. 2003; Jensen et al. 2005; MD DNR 2018b). Spot and Blue Crabs were classified as benthic forage, while Atlantic Menhaden and Bay Anchovy were pelagic (Hartman and Brandt 1995c; Overton et al. 2009). Each forage index was divided by its mean for years in common among all surveys (1989-2017) to place them on the same scale.

Indicators of feeding success and diet composition during October-November were developed using data from a citizen-science based Striped Bass diet monitoring program conducted by CBEF during 2006-2015. During 2014-2017, Striped Bass collected for health samples by FWHP were processed by Fish Habitat and Ecosystem Program personnel for diet information. Methods for CBEF and FWHP collections have been described in Uphoff et al. (2014; 2015; 2016) and will be briefly repeated here.

Striped Bass diet collections by CBEF and FWHP were made in a portion of upper Bay bounded by the William Preston Lane Bay Bridge to the north, the mouth of Patuxent River to the south, and into the lower Choptank River (Figure 1). Striped Bass were collected for diet samples by hook and line fishing.

Conditions of the collectors permit issued to CBEF allowed for samples of up to 15 Striped Bass less than 457 mm total length (or TL; small Striped Bass or fish; the minimum length limit for Striped Bass was 457 mm or 18 inches) and 15 fish 457 mm TL or larger (large Striped Bass or fish) per trip during 2006-2014. Most active trips by CBEF occurred in Choptank River, but some occurred in the mainstem Chesapeake Bay. These trips were our source of small sized fish, but large sized fish were caught as well. Striped Bass kept as samples during active trips were placed in a cooler and either processed immediately or held on ice for processing the next day. Large sized Striped Bass collections were supplemented by charter boat hook and line catches sampled at a fish cleaning business by CBEF. These fish were predominately from the mainstem Chesapeake Bay. These fish were iced immediately and cleaned at the station upon return to port. Fish, minus fillets, were held on ice over one to several days by the proprietor of the fish cleaning service and processed by CBEF at the check station.

Diet collections by FWHP during 2014 were not constrained by collectors permit conditions like CBEF collections. Sampling by FWHP was designed to fill size class categories corresponding to age-classes in an age-length key to assess Striped Bass health. Some trips occurred where fish in filled out length classes were discarded (typically small fish). Samples were usually obtained by fishing on a charter boat using the techniques considered most effective by the captain (bait or artificial lures).

Total length of each Striped Bass was recorded and whole fish were weighed on a calibrated scale for CBEF and FWHP samples. Striped Bass length-weight regressions based on that year's October-November samples were used to estimate missing weights from filleted fish in CBEF collections. Diet items of each fish were identified to the lowest taxonomic group. Contents were classified as whole or partially intact. In CBEF collections, total length of intact fish and shrimp, carapace width of crabs, and shell length of intact bivalves were measured. Non-linear allometry equations for converting diet item length to weight (Hartman and Brandt 1995a) were used. In a few cases, equations for a similar species were substituted when an equation was not available. These equations had been used to reconstruct diets for Overton et al. (2009) and Griffin and Margraf (2003), and were originally developed and used by Hartman and Brandt (1995a). Soft, easily digested small items such as amphipods or polychaetes that could not be weighed were recorded as present. Empirical relationships developed by Stobberup et al. (2009) were used to estimate relative weight from frequency of occurrence of their general taxonomic category. These soft items were not common in our fall collections, but were more common during other seasons (J. Uphoff, personal observation).

Striped Bass diets were analyzed separately for small and large sized fish. These categories accounted for ontogenic changes in Striped Bass diet, but also reflected unbalanced sample availability to CBEF (small fish could only be collected by fishing for them directly, while large sized fish were supplemented by cleaning station samples). The lower limit of fish analyzed in the small category, 286 mm, was the minimum length in common among years during 2006-2013. An upper limit of 864 mm avoided inclusion of large, migratory Striped Bass that reentered upper Bay in late fall.

We confined analysis of food items to those considered recently consumed in an attempt to keep odds of detection as even as possible. Items with “flesh”, including whole or partial fish and invertebrates, and intact crab carapaces were considered recently consumed. Hard, indigestible parts such as gizzards, mollusk shells, and backbones without flesh were excluded. Partially intact items with flesh were identified to lowest taxonomic group and assigned the mean weight estimated for intact items in the same group. Bait was excluded.

Percentage of food represented by an item (excluding bait) during 2006-2016 was estimated for each Striped Bass size class in numbers and weight based on fish with stomach contents (Pope et al. 2001). Two feeding metrics were calculated for each size class for each year. Relative availability of prey biomass (C) was estimated by dividing the sum of diet item weights by the sum of weight of all Striped Bass sampled (including those with empty stomachs; Pope et al. 2001). Estimates of C were subdivided by contribution of each major prey to overall diet mass (species-specific C). Proportion of Striped Bass with empty stomachs (PE) was also estimated as an indicator of total prey availability (Chipps and Garvey 2007). Standard deviations and 95% CI’s of PE were estimated using the normal distribution approximation of the binomial distribution (Ott 1977).

A fishery-independent index of relative abundance of upper Bay resident Striped Bass was not available; therefore, we developed a catch-per-private boat trip index (released and harvested fish) for 1981-2017 from the National Marine Fisheries Service’s (NMFS) Marine Recreational Information Program (MRIP; NMFS Fisheries Statistics Division 2018) database. Similar recreational catch per trip indices have been used as abundance indicators in Atlantic coast stock assessments of major pelagic finfish predators: Striped Bass, Bluefish *Pomatomus saltatrix*, and Weakfish *Cynoscion regalis* (ASMFC 2009; NEFSC 2012; ASMFC 2013).

This index was estimated as a catch-effort ratio for private and rental boat anglers in Maryland in the MRIP inland fishing area (inshore saltwater and brackish water bodies such as bays, estuaries, sounds, etc, excluding inland freshwater areas; NMFS Fisheries Statistics Division 2018). This Striped Bass recreational fishing index (RI) equaled September-October recreational private and rental boat catch divided by estimates of trips for the private and rental boat sector. Recreational survey estimates are made in two month waves and September-October constituted the fifth wave (NMFS Fisheries Statistics Division 2018). This wave was chosen because portions or the whole wave were continuously open for harvest of Striped Bass following the 1985-1990 moratorium, making it less impacted by regulatory measures than other waves that opened later. Recreational fishing by boat occurs over the entire portion of the upper Bay and this index would be as close to a global survey as could be obtained. Migratory fish were unlikely to have been present during this wave. The RI was related to juvenile indices 2-

5 years earlier (determined by multiple regression) and to Atlantic coast abundance estimates (Uphoff et al. 2014). MRIP data used in this analysis represent the old time-series and not the new, mail based estimates of catch and effort (see Discussion).

We used forage indices divided by RI (forage index-to-Striped Bass index ratios) as indicators of forage supply relative to Striped Bass demand (relative attack success). Ratios were standardized by dividing each year's estimate by the mean of ratios during 1989-2017, a time-period in common among all data. The ratios covered 1983-2017. A weighted grand mean of standardized ratios was used to generate a single trend in major forage-to-Striped Bass ratios (or major forage ratios). Two indices (seine and trawl) were available for Bay Anchovy and Spot, while Atlantic Menhaden and Blue Crab had one index each. When calculating the weighted grand mean of all standardized forage ratios, single standardized indices for Atlantic Menhaden and Blue Crab were multiplied by two to counterbalance two indices available for Spot and Bay Anchovy. Targets and limits for major forage ratios were drawn from periods of three or more years where forage ratios coincided with target or limit P0, respectively. A target for major forage ratios was estimated as the lowest average of the standardized ratio that coincided with P0 meeting its target. The limit was estimated as the highest major forage ratio during the P0 threshold period.

We estimated relative survival for age-3 Striped Bass in upper Bay as relative abundance at age-3 divided by age-0 relative abundance three years prior (juvenile index in year - 3). Striped Bass spawning season experimental gill net surveys have been conducted since 1985 in Potomac River and the Head-of-Bay (~39% and 47%, respectively, of Maryland's total spawning area; Hollis 1967) that provide age-specific indices of relative abundance (Versak 2017). Table 8 in Versak (2017) provided mean values of for annual, pooled, weighted, age-specific CPUEs (1985–2016) for the Maryland Chesapeake Bay striped bass spawning stock and we used the age-3 index (CPUE3) as the basis for an adjusted index. This table was updated with 2017 values (B. Versak, MD DNR, personal communication). Even though males and females were included, females were extremely rare on the spawning grounds at age 3; the vast majority of these fish would be resident males (Versak 2017). This CPUE3 index had the advantage of combining both spawning areas, a coefficient of variation (CV) estimate was provided, it was regularly updated in an annual report.

Gill net indices used in the numerator of SR in Uphoff et al. (2015) were suggesting either no change in abundance since 1985 or a decrease; Uphoff et al. (2016) considered both implausible when viewed against stock assessment estimates, juvenile indices, and harvest trends. Uphoff et al. (2016) determined that gill net survey catchability (q; estimated by dividing the catch per effort index by the stock assessment abundance estimate; rearrangement of equation 6.1 in Ricker (1975)) of 3 year-old male Striped Bass changed as an inverse nonlinear function of population size.

We created a "hybrid" gill net time-series that used indices adjusted for rapid changes in catchability during 1985-1995 and the original estimates from Versak (2017) afterwards. First we estimated a catchability coefficient (q) for age striped bass by dividing CPUE3 by the estimated abundance at age 3 from the Statistical Catch-at-Age model (ASMFC 2016) during 1985-2015. We averaged the q estimates for 1985-1995 (mean q) and used them to form a relative q as (annual q / mean q). An adjusted CPUE for each year from 1985-1995 was estimated as CPUE3 / relative q. After 1995, the values reported in Table 8 (Versak 2017) were used. We used a linear regression of relative q versus age 3+ abundance estimated by ASMFC (2016; ages representing mature males) to examine whether a trend was evident in relative q after 1995. If a trend was evident, we would repeat the process used for 1985-1995 on 1996-2017 estimates.

Relative survival (SR) in year t was estimated as the hybrid gill net index for age-3 in year t (HI_t) divided by its respective juvenile index three years earlier (JI_{t-3});

$$SR_t = HI_t / JI_{t-3}.$$

We considered two approaches for developing target and threshold SR values: one used means of SR for time periods of high or low stable SR (target or limit, respectively) and the other used medians. We adopted the approach using medians to estimate SR targets and limits. Means would be more susceptible to influence of anomalous values resulting from changes in catchability from either survey than the median.

Tag-based estimates of survival based on M for 457-711 mm Striped Bass from Chesapeake Bay in ASMFC stock assessment (ASMFC 2013) were compared to SR. Tag-based estimates of M were determined for two time periods in the ASMFC (2013) stock assessment (early period = 1987-1996 and late period = 1997-2011) and we converted the estimates of M in ASMFC (2013) to survival (S) using the equation $S = e^{-M}$ (Ricker 1975). The relative differences in survival (early period estimate / later period estimate) were compared for the two approaches.

Confidence intervals (90%) were developed for ratio based metrics using an Excel add-in, @Risk, to simulate distributions reported for numerators and denominators. Each annual set of estimates was simulated 1,000-times. Ratio metrics simulated were RI, SR, and prey to Striped Bass ratios (prey index / RI) for Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab. Annual means and standard errors reported for these indices were used to generate simulations. Numerators and denominators of the RI, SR, and the Blue Crab index were considered normally distributed since their distributions were characterized by means and SE's in their respective sources (NMFS Fisheries Statistics Division 2018; Versak 2018; MD DNR 2018b). Remaining indices for Atlantic Menhaden (seine), Bay Anchovy (seine and trawl), and Spot (seine and trawl) were based on geometric means (Durell and Weedon 2018). These indices were back-transformed into the mean of $\log_e$-transformed catches (+1) and its standard error was derived from the 95% CI. This transformation normalized the data. Geometric means were recreated by exponentiating the simulated mean of $\log_e$-transformed catches (+1). The type of distribution that best fit RI indices (only ratio without specified means and standard errors) was estimated through a data fitting module available in @Risk (Palisade Corporation 2016).

@Risk used Latin Hypercube sampling to recreate input distributions by stratifying their cumulative curves into equal intervals and then sampled each interval without replacement (Palisade Corporation 2016). Sampling was forced to represent values in each interval and recreated the original input distribution. Latin Hypercube sampling uses fewer iterations compared to random sampling employed by Monte Carlo simulations and is more effective when low probability outcomes are present (Palisade Corporation 2016).

We compared 90% confidence intervals estimated for RI with @Risk with another method for estimating the variance of ratios (Goodman 1960). Confidence intervals (90%) for RI during 1983-2016 were estimated using a spreadsheet developed by Desmond Kahn (Fisheries Investigations Inc.) and Mary Christman (University of Florida) based on Goodman's (1960) formula for the variance of two random variables (Uphoff et al. 2017). We used this comparison as a basis for judging the applicability of the simulation based technique for estimating confidence intervals. The RI estimate based on Goodman (1960) was already available and we did not make similar comparisons with other ratios.

There was some variation in which size classes were used for indicators. All size classes of Striped Bass were used to estimate P0 since Uphoff et al. (2016) did not detect meaningful differences in trend among size-specific estimates. While size classes could not be specified for RI, Uphoff et al. (2014) found that a multiple regression using Maryland Striped Bass juvenile indices for ages 2-5 (corresponding to both size classes) predicted trends in the RI. Forage to Striped Bass ratios would reflect availability to both size classes since RI was used in the denominator. Small classes of Striped Bass had a more varied diet than large sized fish (their fall diet was dominated by Atlantic Menhaden) and PE of the small size class was used as an indicator of forage availability (Uphoff et al. 2018). Estimates of SR reflected survival of small sized fish.

During 1998-2017, each indicator was assigned an annual score from 1 to 5 (bad to good); a score of 1 indicated bad or threshold conditions; a score of 5 indicated best or target conditions; and scores of 2-4 indicated grades of status in between. Scores between 2 and 4 were assigned by breaking the interval between a target and limit into 3 equal increments: a score of 3 represented the mid-increment (avoids the threshold); 2 represented the increment between the threshold and midpoint (approaching the threshold); and 4 indicated the increment between the midpoint and threshold (approaching the target).

Annual scores for each variable were averaged for a combined annual IF score. An average was necessary since five years were unavailable for the PE time-series. Two graphical depictions of uncertainty were developed for the IF. One presented the mean trend as a line and the scores for the individual components as points. This approach presented full variation of the component scores. The other used a "leave one out" approach where annual means were estimated by leaving one component out (i.e., a mean without P0, a mean without PE, etc.). Each set of means was compared to the overall mean and depicted variation in the means.

We conducted three additional analyses of factors that may influence metrics included in the IF. The first was an exploration of an index of benthic invertebrate biomass. Benthic invertebrates are not typically common in fall diets of Striped Bass, but may be important in spring and summer (Hartman and Brandt 1995b; Overton et al. 2009). Since P0 in fall may be influenced by condition in spring and summer (Uphoff et al. 2017), it seemed reasonable to explore whether benthic biomass might influence P0. Biomass per area of benthic invertebrates in Maryland's portion of Chesapeake Bay is a component of a benthic index of biotic integrity used to assess water quality (Lansó and Zaveta 2017); estimates in Figure 3-37 in Lansó and Zaveta (2017) for 1995-2016 were available for analysis. Ash free g/m^2 was estimated for 22 benthic species that accounted for 85% of abundance in samples. These included 7 polychaetes, 10 mollusks, 3 crustaceans (1 isopod and 2 amphipods), and 2 nemertina (ribbon worms; see Table 2-5 in Lansó and Zaveta 2017). Linear and multiple regression analysis (latter using time categories; Rose et al. 1986) were used to examine the relationship of P0 to benthic biomass.

The second analysis looked at whether prey size influenced PE and P0. Animal feeding in nature is composed of two distinct activities: searching for prey and handling prey (Yodzis 1994). Both can be influenced by prey size, with larger prey obtaining higher swimming speeds (typically a function of body length) that enable them to evade a predator and make them more difficult to retain (Ludvall et al. 1999). Overton et al. (2009) found that Striped Bass less than 300 mm TL consistently included prey less than 25% of their body lengths and that the optimum predator-prey length ratio (PPLR) was 21%. For this analysis we determined PPLRs for the two major (largest) prey that have contributed most biomass to the fall diets: Spot and Atlantic Menhaden. This analysis was based on ratios for whole prey and was split for small and large fish. We determined median PPLR for each year and size class and used correlation analysis to examine the associations of PE, median PPLR, and P0.

The third analysis was a comparison of P0, PE, C, diet composition, and characteristics of Striped Bass sampled in 2014 by CBEF and FWHP surveys; 2014 was the only year where data from both diet surveys were collected concurrently. Diet information from CBEF collections were used to characterize fall diets of Striped Bass during 2006-2014 and diet sampling was added to FWHP collection in 2014. Both CBEF and FWHP estimated body fat status during 2006-2014; estimates of annual P0 for both size classes combined were estimated and linear regression analysis of the relationship of P0 estimates by CBEF and FWHP in Uphoff et al. (2014; 2006-2012) was updated.

Correlation and regression were the primary means of analyzing data. For all analyses, scatter plots were examined for the need for data transformations and to identify candidate models. Residuals of regressions were inspected for outliers, trends, and non-normality. If a large outlier was identified, the data from that year was removed and the analysis was rerun. Levels of significance of correlations were not adjusted for multiple comparisons as there is no formal consensus as to when these adjustment procedures should be applied (Nakagawa 2004). A general description of equations used follows, while more specific applications will be described in later sections.

Linear regressions described continuous change in variable Y as X changed:

$$Y = (m \cdot X) + b;$$

where m is the slope and b is the Y-intercept (Freund and Littel 2006). When linear regression analyses exhibited serial patterning of residuals, a time category variable (T) that split the time-series into two time periods (T indicating time categories 0 and 1) were used to remove time-series bias (Rose et al. 1986):

$$Y = (m \cdot X) + (n \cdot T) + b;$$

Where m is the slope, n is a coefficient for the time-series, and b is the intercept.

Potential dome-shaped relationships were examined with quadratic models (Freund and Littell 2006):

$$Y = (m \cdot X) + (n \cdot X^2) + b.$$

The linear regression function in Excel or Proc REG in SAS (Freund and Littel 2006) was used for single variable linear regressions. Multiple linear and quadratic regressions were analyzed with Proc REG in SAS (Freund and Littell 2006).

Examination of scatter plots suggested that some relationships could be nonlinear, with the Y-axis variable increasing at a decreasing rate with the X-axis variable and we fit power, logistic growth, or Weibull functions to these data using Proc NLIN in SAS (Gauss-Newton algorithm). The power function described a relationship with a perceptible, but declining increase in Y with X by the equation:

$$Y = a \cdot (X)^b;$$

where a is a scaling coefficient and b is a shape parameter. The symmetric logistic growth function described growth to an asymptote through the equation:

$$Y = b / ((1 + ((b - c) / c) \cdot (\exp(-a \cdot X)));$$

where a is the growth rate of Y with X, b is maximum Y, and c is Y at X = 0 (Prager et al. 1989).

The Weibull function is a sigmoid curve that provides a depiction of asymmetric ecological relationships (Pielou 1981). A Weibull curve described the increase in Y as an asymmetric, ascending, asymptotic function of X:

$$Y = K\{1 - \exp[-(Y / S)^b]\};$$

where K was the asymptotic value of Y as X approached infinity; S was a scale factor equal to the value of Y where Y = 0.63 • K; and b was a shape factor (Pielou 1981; Prager et al. 1989).

Confidence intervals (90% or 95% CIs) of the model parameters for each indicator species were estimated to examine whether parameters were different from 0 (Freund and Littel 2006). If parameter estimates were not different from 0, the model was rejected.

Results

During 1998-2004, Striped Bass in the upper Bay during fall were usually in poor condition (P0 $\geq$ threshold; threshold = 0.68). They were at or near the target level of condition (P0 $\leq$ target; target = 0.30) during 2008-2010, 2014-2015, and 2017 (Figure 2). Condition has been much more variable since 2006 (target to threshold) than earlier (near threshold to threshold conditions). The 95% confidence intervals of P0 allowed for separation of years meeting the target condition from remaining estimates (Figure 2). A IF score of 1 was assigned to P0 at or more than 0.68; a score of 2 was assigned for P0 between 0.68 and 0.55; a score of 3 was assigned to P0 between 0.55 and 0.43; a score of 4 was assigned to P0 between 0.43 and 0.30; and a score of 5 was assigned when P0 was 0.30 or less (Table 2).

A combined P0 index for all sizes of Striped Bass was adopted in Uphoff (2016) based on 1998-2014 data; however, in 2016 a pronounced difference in condition was evident between small (small P0 = 0.83) and large sized fish (P0 = 0.25; Figure 3). This phenomenon was not repeated in 2017 (Figure 3).

Major pelagic prey were generally much more abundant during 1959-1994 than afterward (Figure 4). Bay Anchovy seine indices (1959-2017) following the early to mid-1990s were typically at or below the bottom quartile of indices during 1959-1993. Highest Bay Anchovy trawl indices (1989-2016) occurred in 1989-1992 and 2001-2002, while lowest indices occurred during 2006-2011 and 2015-2017. There was little agreement between the two sets of Bay Anchovy indices; however, there were few data points representing years of higher abundance and contrast may have been an issue (comparisons are of mostly similar low abundance points). Atlantic Menhaden seine indices (1959-2017) were high during 1971-1994 and much lower during 1959-1970 and 1995-2017 (Figure 4).

Benthic forage indices were low after the 1990s, but years of higher relative abundance were interspersed during the 2000s (Figure 5). Seine (1959-2017) and trawl (1989-2017) indices for Spot were similar in trend and indicated high abundance during 1971-1994 and low abundance during 1959-1970 and after 1995 (with 3 or 4 years of higher indices interspersed). Blue Crab densities (1989-2017) were highest during 1989-1996, 2009, and 2011 (Figure 5).

In general, relative abundance of Striped Bass (RI) during 1981-2017 was lowest prior to 1994 (mean RI < 0.7 fish per trip; Figure 6). Estimates of RI then rose abruptly to a high level and remained there during 1995-2006 (mean = 2.6). Estimates of RI fell to about half the 1995-2006 mean during 2008-2013 (mean = 1.2) and then rose to 2.5 in 2014, 2.7 in 2015-2016, and 3.0 in 2017; 2017 was assigned an IF score of 1. Simulated 90% confidence intervals indicated that RI was much lower during 1981-1993 than afterward and that there was some chance that RI during 2008-2013 was lower than other years during 1994-2017 (Figure 7). The 90% confidence intervals based on @Risk simulations (Figure 6) were very similar to those estimated using the technique of Goodman (1960; Figure 7).

Species-specific standardized forage-to-Striped Bass ratios exhibited similar patterns during 1983-2017 (Figures 8-13). Simulated 90% confidence intervals for prey to Striped Bass ratios indicated these ratios were high prior to 1994 and lower afterward (Atlantic Menhaden, Figure 8; Bay Anchovy, Figures 9 and 10; Spot, Figures 11 and 12; Blue Crab, Figure 13; trends in standardized indices since 1983; Figure 14; trends in standardized indices and the weighted grand mean or FR after 1997, when P0 was estimated, Figure 15). A nadir in the ratios appeared during 1995-2004, followed by occasional "spikes" of Spot and Blue Crab ratios and a slight rise in Atlantic Menhaden and Bay Anchovy ratios during 2005-2016.

Target P0 was met during 2008 and 2010 when FR was more than 1.14 (IF score for FR = 5; Table 2). Threshold P0 was met when FR was 0.66 or less (IF score for FR =1). An IF score of 2 was assigned for FR between 0.661 and 0.822; a score of 3 was assigned for FR between 0.823 and 0.981; and a score of 4 was assigned for FR between 0.981 and 1.14 (Table 2).

Samples from 1,582 small and 2,440 large sized Striped Bass were analyzed for diet composition during October-November, 2006-2017 (Table 3). Numbers examined each year ranged from 47 to 221 small fish and 49 to 327 large fish. Fewer dates were sampled within similar time spans after the FWHP became the platform for sampling in 2014 since number collected per trip was not confined by the terms of the CBEF collector's permit (7-12 trips by FWHP during 2014-2017 versus 11-22 trips by CBEF during 2006-2013; Table 3).

In combination and by number, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (major forage items) accounted for 95.3% of diet items encountered in small Striped Bass collected from upper Bay during fall, 2006-2017. Bay Anchovy accounted for the highest percentage by number (60.2%, annual range = 19.1-87.9%); Atlantic Menhaden, 13.1% (annual range = 0-48.8%); Spot 7.1% (annual range = 0-70.7%); Blue Crab, 14.8% (annual range = 0.8-34.6%); and other items accounted for 4.7% (annual range = 0-24.5%; Figure 16). The number of prey in the "Other" category represented a much larger fraction of the total for small Striped Bass during 2017 (24.5% of small diet by number and 40% of large) than any other year (next largest = 7.7%; Figure 16); tunicates represented 70% of “Other” items in small Striped Bass. The vast majority of major prey in small Striped Bass diet samples were YOY (Uphoff et al. 2016).

Major prey accounted for 92.4% of diet items, by number, encountered in large Striped Bass diet samples during fall 2006-2016 (Figure 17). Atlantic Menhaden accounted for 44.4% (annual range =12.4-76.4%); Bay Anchovy, 16.0% (annual range = 3.7-32.5%); Spot, 8.7% (annual range = 0-52.4%); Blue Crab, 23.3% (annual range = 2.6-59.4%); and other items, 7.6% (annual range = 0-36.2%). The “Other” category accounted for a noticeably higher fraction of large striped bass diets by number in 2012 and 2017 (36.2% and 40.0%, respectively; Figure 17) than remaining years (< 9.7%). The vast majority of major prey were young-of-year (Uphoff et al. 2016).

By weight, small Striped Bass diets in fall 2006-2017 (combined) were dominated by Atlantic Menhaden (70.6%), followed by Spot (11.3%), Bay Anchovy (10.0%), Blue Crab (2.3%) and other items (5.7%). Estimates of relative availability of prey biomass (C, total grams of prey consumed per gram of Striped Bass) for small Striped Bass varied as much as 8.7-times during 2006-2017 (Figure 18). During years of lowest C (2007, 2011, 2016, and 2017) varying items contributed to the diet of small fish. Bay Anchovy accounted for most diet weight during 2017 (the lowest year for C) and Atlantic Menhaden were absent. During years of higher C, either Spot (2010) or Atlantic Menhaden (remaining years) dominated diet mass (Figure 18).

By weight, Atlantic Menhaden predominated in large fish sampled (84.1% of diet weight during fall 2006-2017; all years combined); Bay Anchovy accounted for 1.3%; Spot, 4.5%; Blue Crab, 4.5%; and other items, 5.7% (Figure 19). Estimates of C for large Striped Bass varied as much as 3.8-times among years sampled; 2017 was the lowest estimate of C in the time-series. Atlantic Menhaden dominated diet weight of large sized fish during October-November (Figure 19).

Estimates of proportion of empty stomachs (PE) of small sized Striped Bass during fall, 2006-2017, ranged between 0.10 and 0.57 (Figure 20). Lowest estimates of PE for small fish (2009-2011, 2014 and 2017) could be separated from remaining estimates (except 2008) based on 90% confidence interval overlap. Estimates of PE steadily fell for small sized fish during 2006-2011 and have varied greatly between the target and threshold PE since then (Figure 20).

The estimate of PE during 1998-2000 (PE = 0.54) developed for small Striped Bass from Overton et al. (2009; Uphoff et al. 2016) was adopted as a threshold (IF score = 1) for small fish; annual estimates of P0 for small Striped Bass were at the threshold during 1998-2000. The highest PE estimate during 2008-2010 (PE ranged from 0.19 to 0.31) when P0 was at its target was selected as the PE target (PE $\leq$ 0.31 is assigned an IF score of 5; Table 2). IF scores for categories 2 and 4 were reported incorrectly in Uphoff et al. (2017) and the correction follows: “An IF score of 2 (replace with 4) was assigned to estimates greater than 0.310 and less than or equal to 0.387; a score of 3 was assigned to estimates greater than 0.387 and less than or equal to 0.463; and 4 (replace with 2) was assigned to estimates greater than 0.463 and less than 0.540.” (Table 2).

Estimates of proportion of empty stomachs (PE) of large sized Striped Bass during fall, 2006-2013, ranged between 0.40 and 0.63 (Figure 21). Estimates of PE of large sized fish fell to 0.10-0.29 during 2014-2016, then rose to 0.60 in 2017. Lowest estimates of PE for large fish (2013-2016) could be separated from remaining higher estimates based on 95% confidence interval overlap (Figure 21). Overton et al. (2009)

provided an estimate of the percent of Striped Bass in their large size class (501-700 mm, TL) with food during 1998-2000 and we used this estimate to derive a threshold PE for large sized fish (0.58). The 95% CI's during 2006-2008, 2011-2012, and 2017 overlapped this threshold.

The CPUE3 index was synchronous with the abundance of age 3 Striped Bass estimated by ASMFC (2016) assessment after 1992 (1985-2015 time-series; Figure 22), but earlier estimates of CPUE3 indicated a full range of abundance during this early period with some of the highest indices of the time-series occurring in 1985-1987. The ASMFC (2016) assessment indicated abundance of age 3 Striped Bass was very low during 1985-1989 (Figure 23). Estimated catchability continuously declined during 1985-1996 and appeared to stabilize afterward. A linear regression of CPUE3 against abundance at age 3 during 1996-2015 did not indicate a trend in catchability (r^2 = 0.04, P = 0.41) and observed CPUE3 values were used for this portion of the time-series. The hybrid age 3 gill net index of male relative abundance (HI_3) on the spawning grounds indicated a dearth of high indices during 1985-1995 (Figure 24). These low HI_3 year-classes were followed by the appearance of intermittent large year-classes at age 3 (1996, 1998, 1999, 2004, 2006, 2010, and 2014). The HI_3 indicated sharper changes in relative abundance of age 3 Striped Bass from year-to-year than the ASMFC (2016) assessment. Peaks generally aligned, but years of low abundance in the ASMFC (2016) assessment tended to be higher than would have been indicated by the hybrid gill net index (Figure 24).

Ninety percent confidence intervals of relative survival (SR; HI_3 / JI_{t-3}) allowed for separation of years of high and low survival, and some years where survival was in between (Figure 25). Estimated SR was consistently high during 1986-1996, shifted to consistently low during 1999-2004, and varied afterwards. Low survival in 1985 reflected the effect of the fishery prior to imposition of a harvest moratorium in Maryland (Figure 26). The 56% percent reduction in median relative survival to age 3 between1986-1996 (median SR = 60.0) and 1997-2016 (mean SR = 20.3) was greater than changes in tag-based estimates of survival of large-sized fish during the same period (from 77% annual survival to 44%, a 43% reduction).

The target for SR was derived as the median estimate for the period 1986-1996 (target SR = 60.0; IF score = 5) and the threshold was the median during 1999-2004 (threshold SR = 15.0; IF score = 1; Table 2). A score of 2 was assigned to SR between 15.1 and 30.0; a score of 3 was assigned when SR was between 30.1 and 45.0; and SR between 45.1 and 59.9 was given a score of 4 (Table 2). After 1998, target SR was reached in 2010, 2011, and 2014 (Figure 26). After 2004, threshold conditions were met in 2007 and 2008 (Figure 26).

Targets, thresholds, and increments of scores for P0, RI, FR, PE, and SR are summarized in Table 2.

The IF varied from 1.25 to 5 during 1998-2016 (Figure 27). During 1998-2004, the IF was low, between 1 and 2. The IF increased to 3 in 2005, fell back to around 2 in 2006-2007, and then increased to near 4 to 5 during 2008-2010. After 2010, it varied from above 2 to just below 4. IF was between 2 and 3 during 2015-2017. Spread of annual component scores was generally narrower (no more than 2 units) during 1998-2004 when the IF was consistently low. Spread was typically wider as scores improved after 2004 (Figure 27).

Estimates of mean IF with each component removed indicated little variation from the overall IF (Figure 29). The maximum deviation from the overall IF in any given year ranged between 0 and 0.75 (Figure 28). This approach suggested that IF means could be separated into high, medium, and low categories.

Benthic biomass estimates for Maryland's portion of the Bay varied by a factor of four during 1998-2016 (Figure 29). Density estimates were frequently at or above the time-series median prior to 2010 (13 estimates). During 2011-2016, two estimates were at or slightly above the median and the rest were below (Figure 29).

A simple linear relationship of benthic biomass with P0 was weak ($r^2 = 0.08$, P = 0.24; Table 5) and patterning in the residuals from positive to negative was evident. Addition of a categorical variable indicating two stanzas of time (1998-2007 and 2008-2016) to benthic biomass removed the patterning of residuals with time and all terms were significant ($R^2 = 0.66$, P = 0.0002; Table 3; Figure 30). This multiple regression indicated that P0 improved (declined) as benthic biomass increased, but time stanza shifted the relationship. Over the same range of benthic biomass, P0 was predicted to improve from above threshold (0.79) to below (0.53) during 1998-2007 and improve from above the target (0.49) to at target (0.22) during 2008-2016. Condition of Striped Bass in fall was best when benthic density was consistently low, a result that did not support classifying benthic invertebrates as consistently important prey. These time stanzas coincided with shifts in RI from high to low and FI from lower to higher.

We determined median PE, median PPLR, and P0 for each year and size class (Table 6). Median PPLR of Spot and Striped Bass (larger major prey) were generally lower for large fish (0.19-0.30) than small fish (0.27-0.38). Median PPLR were noticeably smaller for large than small fish during 2006-2009, 2012, and 2015-2017, and they tracked closely in remaining years (Figure 31). For small fish, only the correlation of PPLR and PE was significant (r = 0.68, P = 0.02), while for large fish only PE and P0 were significantly correlated (r = 0.64, P = 0.02). Higher PPLR ratios (indicating larger sized major prey) were positively associated with a higher frequency of empty guts for small Striped Bass, but not for large ones.

CBEF and FWHP estimates of P0 during 2006-2014 were comparable. There was a significant linear relationship between estimates of P0 made from data collected by CBEF and FWHP ($r^2 = 0.79$, P = 0.001; Figure 32). The intercept (mean = 0.049, SE = 0.111) was not different from zero and the slope (mean = 0.886, SE = 0.172) was not different than one, indicating a 1:1 relationship between the estimates.

CBEF diet samples were all drawn from Choptank River during 2014, while FWHP samples were drawn from the mid- and lower Bay. Summary statistics for lengths of Striped Bass in both size classes were fairly similar, although there were differences in kurtosis and skewness (Table 7). CBEF collections of small fish were 41% lower than FWHP collections, while CBEF collections of large fish were 55% of those take by CBEF. Estimates of C were lower in 2014 based on CBEF samples than for FWHP samples for both size classes (Table 7).

Fall diet samples of small Striped Bass from both sampling programs were dominated by Atlantic Menhaden during 2014 (94% of diet biomass in CBEF samples and 84% in FWHP samples; Table 8). Bay Anchovy were not detected in small fish in 2014 CBEF diet samples, but comprised nearly 11% of diet weight in FWHP samples. Spot were not detected in either diet survey. Blue Crabs represented a much lower fraction of diet weight in small fish in CBEF samples than FWHP samples, while the "Other" category was higher (Table 8).

Large Striped Bass diets in fall 2014 were dominated by Atlantic Menhaden (66% of CBEF diet weight and 92% of FWHP diet weight; Table 8). Spot were not detected in either diet survey; Blue Crab and Bay Anchovy made minor contributions. There was a large difference in the contribution of the "Other" category to diet biomass; "Other" diet items (comprised mostly of modest numbers of ages 1+ White Perch and Striped Bass) contributed 33% of diet biomass in CBEF samples compared to 4% of FWHP diet biomass (Table 8). White Perch and Striped Bass were not detected in large fish sampled by FWHP, but Herring (*Alosa* species), Atlantic Croaker, polychaetes, shrimp, and Mantis Shrimp were present in the "Other" category of FWHP samples collected in fall 2014.

CBEF based estimates of PE in 2014 were higher than those based on FWHP samples for small fish (CBEF PE = 0.40, SD = 0.039 and FWHP PE = 0.10, SD = 0.02). When Bay Anchovies were removed from consideration, the estimate of PE for small fish based on FWHP collections was closer to the CBEF estimate (FWHP PE without Bay Anchovy = 0.34, SD = 0.028).

Discussion

The IF has been between 2 and 3 (near or avoids threshold, respectively) during 2015-2017, reflecting demand for forage by the 2011 and 2015 dominant year-classes. The IF indicated threshold to near threshold foraging conditions (scores between 1 and 2) were typical during 1998-2007. IF scores during 2008-2010 (IF = 3.6-5.0) avoided the threshold and all scores reached their targets in 2010. Scores were in the avoid threshold to near target range (3-4) again in 2013-2014. High variability in component scores was evident as IF improved after 2004. This variability may have reflected sampling issues, nonlinear, asymptotic relationships among variables, lagged responses, potential insensitivity of some indices, behavioral changes that could increase feeding efficiency, episodes of good foraging conditions outside of those monitored in fall, larger major prey relative to size of Striped Bass and combinations of the above. Many of these issues were discussed in Uphoff et al. (2016; 2017) and the reader is referred to them. Issues not discussed in Uphoff et al. (2016; 2017) will be covered in this discussion.

A rapid rise in Striped Bass abundance in upper Bay during the mid-1990s followed by a dozen more years at high abundance coincided with declines in indices of Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (i.e., major pelagic and benthic prey) to low levels. Striped Bass were often in poor condition during fall 1998-2004 and vulnerable to starvation. Improvements in condition coincided with lower Striped Bass abundance after 2007, spikes or slight (statistically insignificant) increases in some major forage indices, and higher consumption of larger major prey (Spot and Atlantic Menhaden) in fall diets. Survival of small and large sized Striped Bass in upper Bay shifted downwards in the mid-1990s shortly after upper Bay major forage-to-Striped Bass ratios, an indicator of attack success, reached a nadir. Poor survival of age 3 Striped Bass persisted through 2004 and occasional years of above target survival occurred afterwards.

The IF approach was based on a suite of indicators that were statistically linked, but not linked so tightly that one would adequately represent another. Statistical analyses can provide insight into important processes related to predation (Whipple et al. 2000). Uphoff et al. (2016; 2017) used correlation and regression analyses to examine whether indicators of upper Bay Striped Bass abundance, forage abundance, consumption, and relative survival estimates were linked to the body fat condition indicator. Correlations and relationships may change over time it they do not reflect underlying ecological processes or the processes themselves shift over time (Skern-Mauritzen et al. 2016).

Simulated 90% confidence intervals for ratio based indicators (RI, forage index to RI, and SR) generally allowed for separation of high and low values and, in some cases, mid-level values could be determined. Most could be separated from zero and those that were not were usually very low. The 90% confidence intervals of RI based on @Risk simulations were very similar to those estimated using the technique of Goodman (1960; Uphoff et al. 2017), indicating that simulations were a reasonable approach.

On July 9, 2018, NMFS released revised Marine Recreational Information Program (MRIP) catch estimates (1981-2017) as part of its recent transition from the old Coastal Household Telephone Survey to a new, mail-based Fishing Effort Survey (FES). Our estimates of RI used the older catch and effort estimates. A July 11, 2018 comparison of catch and trip estimates used to estimate RI using new and old MRIP estimates indicated very little change in depiction of relative abundance by the RI (J. Uphoff, unpublished analysis). Expansion of catch and effort in the new MRIP estimates appeared to be very similar and dividing them produced very similar RI indices. RI estimates will be revised, but trends and variability depicted in this report should be reasonable.

One of the objectives for the IF is low cost and tractability for available staff. We used available estimates of central tendency and variability for the ratio simulations. We did not attempt to standardize indices to account for influences such as latitude, date, and temperature. Use of standardizing techniques that “account” for other influences have increased, but they require additional staff time and often have barely have a detectable effect on trends. Maunder and Punt (2004) described that their effect “can be disappointingly low” and they do not guarantee removal of biases that result in hyperstability.

Forage indices and forage to Striped Bass ratios were placed on the same scale by dividing them by arithmetic means over a common time period (ratio of means). Conn (2009) noted in several scenarios that the arithmetic mean of scaled indices performed as well as the single index estimated by a hierarchal Bayesian technique. Falcy et al. (2016) found that ratios of means provided a reasonable method for combining indices into a composite index to be calibrated with population estimates of Chinook Salmon *Oncorhynchus tshawytscha*, but there was no one optimal method among the four techniques applied.

Our concentration on fall diets did not directly consider some prey items in the “other” category that could be important in other seasons. White Perch (*Morone americana*) and invertebrates other than Blue Crab are important diet items during winter and spring, respectively (Walter et al. 2003; Hartman and Brandt 1995c; Overton 2009). These species did not usually make a large contribution to diet mass during fall, but White Perch from the 2011 dominant year-classes made a large contribution to large sized Striped Bass diet biomass in fall, 2012 and 2014 (CBEF collections for the latter).

Uphoff et al. (2017) identified outliers for comparisons of PE, RI, and forage ratios with P0 (2015 in all three cases) and SR with P0 (2004 and 2010). While similar analyses were not done for this report, it appears that P0 (score = 5) in 2017 contradicted remaining indicators (scores range from 1 to 3). Outliers occurred twice in 20 years, indicating a 10% chance of a non-conforming value in a given index. However, nonconformity of P0 scores is recent and may indicate change in dynamics beyond what has been experienced. If managers decide to use the IF for decision making, they should consider multiple years of IF scores to make a judgment rather than a single year to avoid false positives or negatives. Some of the analyses in Uphoff et al. (2017) relating P0 to other metrics need to be redone since methods for calculating metrics has changed somewhat.

A fairly constant migration schedule for male Striped Bass between when they are sampled as young-of-year and appear on the spawning ground at age 3 is an underlying assumption of the SR since shifts in migration can produce similar changes as M. Migration estimates based on 1988-1991 spawning area and season tagging (40-100 cm TL) indicated that larger striped bass were more likely to migrate from spawning areas of the Chesapeake Bay to coastal areas north of Cape May, NJ than were smaller fish (Dorazio et al. 1994). Fewer males participate in the northward migration, but this difference appeared to reflect differences in size of mature males and females (Dorazio et al. 1994). Kohlenstein (1981) determined that few young males leave the Chesapeake Bay. Observation error or change in catchabilities of the spring gill net and juvenile surveys can also produce changes in SR. Uphoff et al. (2016) determined that gill net survey catchability (q; estimated by dividing the index by the stock assessment abundance estimate) of 3 year-old male Striped Bass changed as an inverse nonlinear function of population size. While there is some year to year variation in age 3 catchability, major changes that would lead to bias would require a sustained drop in total abundance. The SR index has an added complication in that it is a measure of survival over about 2.5 years, while other IF indices are annual or have potential lags less than 2.5 years. The other IF indices would not be relevant to this whole SR period since fish less than about 2-years old were poorly represented in diet samples.

We chose PE as an indicator of feeding success over C because confidence intervals could be easily calculated for PE and estimates from Overton et al. (2009) were available to estimate threshold conditions during 1998-2000. In addition, this indicator could be derived from diet information from the 1930s (Hollis 1952) and the 1950s (Griffin and Margraf 2003). However, correlation analysis indicated PE could be sensitive to prey size and suggested that this may lead to positive bias if small items predominate and negative bias if large items predominate. Estimates of C and species specific C are useful for interpreting PE.

A transition from citizen-science diet sampling (CBEF) to use of an existing Striped Bass health platform (FWHP) to obtain diet samples in fall was undertaken in 2014. Both programs sampled Striped Bass diets separately and provided an opportunity to evaluate their compatibility. However, Mr. Price's (CBEF) health was beginning to fail and sampling was somewhat curtailed in frequency and geographic coverage (sampling did not occur in the Bay as in other years, only the Choptank River). Size of Striped Bass sampled by the two programs appeared comparable and estimates of P0 were similar. Fall diets were dominated by Atlantic Menhaden and Spot were absent in both cases. Differences arose in smaller major prey, particularly Bay Anchovy, and in the importance of "Other" prey. Bay Anchovy were absent in CBEF samples of small fish, but ranked second in biomass in FWHP samples. Large fish diet samples in

Choptank River (CBEF collections) indicated that “Other” food (primarily ages 1+ White Perch and Striped Bass by weight) was much more important than in Bay samples collected by FWHP. Absence of Bay Anchovy in CBEF samples resulted in much higher estimates of PE than in FWHP samples; elimination of Bay Anchovy from FWHP samples resulted in much closer estimates of PE.

Ecosystem based fisheries management has been criticized for poor tractability, high cost, and difficulty in integrating ecosystem considerations into tactical fisheries management (Fogarty 2014). It has been the principal investigator’s unfortunate experience that complex and comprehensive ecosystem based approaches to fisheries management for the entire Chesapeake Bay i.e., Chesapeake Bay Ecopath with Ecosim and MD Sea Grant’s Ecosystem Based Fisheries Management for Chesapeake Bay (Christensen et al. 2009; MD Sea Grant 2009) have not gained a foothold in Chesapeake Bay’s fisheries management. This is not surprising. While policy documents welcome ecosystem based approaches to fisheries management and a large number of studies that have pointed out the deficiencies of single-species management, a review of 1,250 marine fish stocks worldwide found that only 2% had included ecosystem drivers in tactical management (Skern-Mauritzen et al. 2016). The index-based IF approach represents a less complex, low cost attempt to integrate forage into Maryland’s fisheries management. Given the high cost of implementing new programs, we have combined effort with information from existing sampling programs and indices (i.e., convenience sampling and proxies for population level estimates, respectively; Falcy et al. 2016). This trade-off is very common in fisheries and wildlife management (Falcy et al. 2016).

The IF represents a framework for condensing complex ecological information so that it can be communicated simply to decision makers and stakeholders. The science of decision making has shown that too much information can lead to objectively poorer choices (Begley 2011). The brain’s working memory can hold roughly seven items and any more causes the brain to struggle with retention. Decision science has shown that proliferation of choices can create paralysis when the stakes are high and the information complex (Begley 2011). For this report, the IF condensed five elements into a combined score (sixth element) that, hopefully, can alert busy fisheries managers and stakeholders about the status of forage and whether forage merits further attention and action.

The IF is similar to traffic light style representations for applying the precautionary approach to fisheries management (Caddy 1999; Halliday et al. 2001). Traffic light representations can be adapted to ecosystem based fisheries management (Fogarty 2014). The strength of the traffic light method is its ability to take into account a broad spectrum of information, qualitative as well as quantitative, which might be relevant to an issue (Halliday et al. 2001). It has three elements – a reference point system for categorization of indicators, an integration algorithm, and a decision rule structure based on the integrated score (Halliday et al. 2001). In the case of the IF, it contains the first two elements, but not the last. Decision rules would need input and acceptance from managers and stakeholders.

Some form of integration of indicator values is required in the traffic light method to support decision making (Halliday et al. 2001). Integration has two aspects, scaling the indicators to make them comparable (ranking them from 1-5 in the IF) and applying an operation to summarize the results from many indicators (averaging the elements of the IF; Halliday et al. 2001). Although it is intrinsic to integration that some information is lost, the loss is not necessarily of practical importance (Haliday et al. 2001). The original indicators are still available for decision rules that might require more information than is contained in the characteristics. Simplicity and communicability are issues of over-riding importance (Haliday et al. 2001). Caddy (1998) presented the simplest case for single-species management where indicators were scaled by converting their values to traffic lights, and decisions were made based on the proportion of the indicators that were red. While the IF is numeric, it could easily be converted to a traffic light using the strict (three distinct colors) or fuzzy (blended colors) methods. A prototype of the IF used a traffic light color scheme (Uphoff et al. 2014).

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Table 1. Important abbreviations and definitions.

Abbreviation	Definition
@Risk	Software used to simulate confidence intervals of ratios
C	Grams of prey consumed per gram of Striped Bass, an indicator of feeding success and prey availability.
CBEF	Chesapeake Bay Ecological Foundation.
CI	Confidence interval.
CPUE3	Unmodified gill net index of relative abundance of age 3 male Striped
CV	Bass.
F	Coefficient of variation.
FI	Instantaneous annual fishing mortality rate.
	Forage index. Mean score for five indicators of forage status (FR, PE,
FR	P0, RI, and SR)
	Mean major forage ratio score (mean of scores assigned to
FWHP	standardized major prey to Striped Bass ratio
	Fish and Wildlife Health Program
HI	Hybrid gill net index of relative abundance of age 3 male Striped Bass that has been adjusted for catchability change with population size.
JI	Juvenile index of relative abundance of a species.
M	Instantaneous annual natural mortality rate.
MRIP	Marine Recreational Information Program
PE	Proportion of Striped Bass with empty stomachs, an indicator of feeding success and prey availability.
P0	Proportion of Striped Bass without visible body fat, an indicator of nutritional status (condition).
q	Catchability (efficiency of a gear).
RI	Catch (number harvested and released) of Striped Bass per private and rental boat trip, a measure of relative abundance.
SR	Relative survival index for small sized resident Striped Bass to age-3.

Table 2. Intervals used for assigning FR scores to metrics for P0, RI, FR, and PE. Blanks indicate a minimum or maximum cannot be defined.

		Score				
Metric	Interval	1	2	3	4	5
P0	Minimum	0.680	0.550	0.430	0.301	0.000
	Maximum	1.000	0.679	0.549	0.429	0.300
RI	Minimum		2.170	1.630	1.101	1.100
	Maximum	2.700	2.699	2.169	1.629	
FR	Minimum	0.000	0.249	0.441	0.631	0.820
	Maximum	0.250	0.440	0.630	0.820	
PE	Minimum	0.540	0.463	0.387	0.310	0.000
	Maximum	1.000	0.539	0.462	0.386	0.309
SR	Minimum	0.000	15.000	30.000	45.000	60.000
	Maximum	14.999	29.999	44.999	59.999	

Table 3. Proportion of Striped Bass without body fat (P0) during fall estimated from Fish and Wildlife Health Program sampling. N = sample size and SD is the standard deviation of P0.

Year	P0	N	SD
1998	0.749	338	0.024
1999	0.779	344	0.022
2000	0.773	290	0.025
2001	0.745	224	0.029
2002	0.605	316	0.028
2003	0.700	237	0.030
2004	0.746	414	0.021
2005	0.596	524	0.021
2006	0.600	863	0.017
2007	0.500	662	0.019
2008	0.137	629	0.014
2009	0.312	1107	0.014
2010	0.270	693	0.017
2011	0.531	1202	0.014
2012	0.658	333	0.026
2013	0.576	441	0.024
2014	0.312	398	0.023
2015	0.124	347	0.018
2016	0.476	429	0.024
2017	0.237	325	0.024

Table 4. Number of dates sampled and number of small (<457 mm, TL) and large sized Striped Bass collected in each size category, by year.

Year	N dates	Small N	Large N
2006	19	118	49
2007	20	76	203
2008	15	29	207
2009	17	99	240
2010	22	112	317
2011	19	74	327
2012	11	47	300
2013	14	191	228
2014	7	277	108
2015	8	174	173
2016	12	169	260
2017	9	272	52

Table 5. Results of linear regression models used to explore the relationship of biomass of benthic (gm / m^2) invertebrates and condition of Striped Bass (P0) in Maryland's portion of Chesapeake Bay. Time categories (Time cat) for the multiple regression were 1998-2007 (category = 0) and 2008-2015 (category = 1).

Linear regression

r Square 0.08

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	0.065421	0.065421	1.494078	0.238262
Residual	17	0.744379	0.043787		
Total	18	0.8098			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*
Intercept	0.687491	0.132793	5.177148	7.58E-05	0.407322	0.967661
g/m^2	-0.01342	0.010978	-1.22232	0.238262	-0.03658	0.009743

Multiple regression with time categories

R Square 0.66

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	2	0.536124777	0.268062	15.67185	0.00017
Residual	16	0.273675378	0.017105		
Total	18	0.809800155			

	Coefficients	*Standard Error*	*t Stat*	*P-value*	*Lower 95%*	*Upper 95%*
Intercept	0.877607811	0.090564447	9.690423	4.25E-08	0.68562	1.069596
g/m^2	-0.016971656	0.006894702	-2.46155	0.025574	-0.03159	-0.00236
Time cat	-0.316763307	0.060383597	-5.24585	8E-05	-0.44477	-0.18876

Table 6. Estimates of Striped Bass median prey-predator length ratio (PPLR) estimated for large major prey (Spot and Atlantic Menhaden combined), proportion of Striped Bass with empty guts (PE) and proportion of small Striped Bass without body fat (P0). Small Striped Bass were less than 457 mm and large were greater than or equal to 457 mm. Large major prey were not found in small Striped Bass in 2017 and could not be included in analysis.

Year	Size	PPLR median	PE	P0
2006	Small	0.31	0.57	0.60
2007	Small	0.31	0.45	0.50
008	Small	0.27	0.31	0.14
2009	Small	0.32	0.25	0.31
2010	Small	0.20	0.19	0.27
2011	Small	0.28	0.16	0.53
2012	Small	0.38	0.57	0.66
2013	Small	0.27	0.07	0.40
2014	Small	0.27	0.4	0.50
2015	Small	0.36	0.55	0.08
2016	Small	0.35	0.37	0.82
2006	Large	0.27	0.53	0.56
2007	Large	0.26	0.44	0.47
2008	Large	0.23	0.47	0.04
2009	Large	0.26	0.37	0.24
2010	Large	0.19	0.4	0.25
2011	Large	0.28	0.48	0.46
2012	Large	0.28	0.63	0.58
2013	Large	0.26	0.41	0.61
2014	Large	0.25	0.10	0.14
2015	Large	0.22	0.17	0.22
2016	Large	0.3	0.25	0.08
2017	Large	0.27	0.28	0.25

Table 7. Summary statistics for total length of Striped Bass, by size class, collected in fall 2014 for diet samples by Chesapeake Bay Ecological Foundation (CBEF) and DNR's Fish and Wildlife Health Program (FWHP).

	Total Length, mm			
Size Class	Small	Large	Small	Large
Source	CBEF	FWHP	CBEF	FWHP
	408	414	508	494
Standard Error	3.5	1.5	9.8	3.6
Median	419	418	483	479
Mode	432	427	483	461
Kurtosis	2.2	-0.6	7.3	1.8
Skewness	-1.5	-0.4	2.6	1.5
Minimum	279	339	457	458
Maximum	452	456	826	620
Count	111	277	60	108

Table 8. Estimates of weight of prey consumed per weight of Striped Bass (gm/gm) during concurrent fall 2014 sampling by Chesapeake Bay Ecological Foundation (CBEF) and DNR's Fish and Wildlife Health Program (FWHP). P = proportion of total diet weight represented by a food item.

Source	CBEF	CBEF	FWHP	FWHP
Metric	gm/gm	P	gm/gm	P
			Small	
Menhaden	0.017	0.941	0.026	0.838
Anchovy	0	0	0.003	0.106
Spot	0	0	0	0.000
Blue Crab	0.000	0.001	0.001	0.025
Other	0.001	0.057	0.001	0.030
Total	0.018		0.031	
			Large	
Menhaden	0.014	0.663	0.020	0.915
Anchovy	0.000	0.001	0.000	0.007
Spot	0	0	0	0
Blue Crab	0.000	0.004	0.001	0.035
Other	0.007	0.332	0.001	0.043
Total	0.020		0.022	

Figure 1. Upper Bay (Maryland's portion of Chesapeake Bay) with locations of forage index sites (black dots = seine site and grey squares = trawl site), and regions sampled for Striped Bass body fat and diet data. Patuxent River seine stations are not included in analyses.

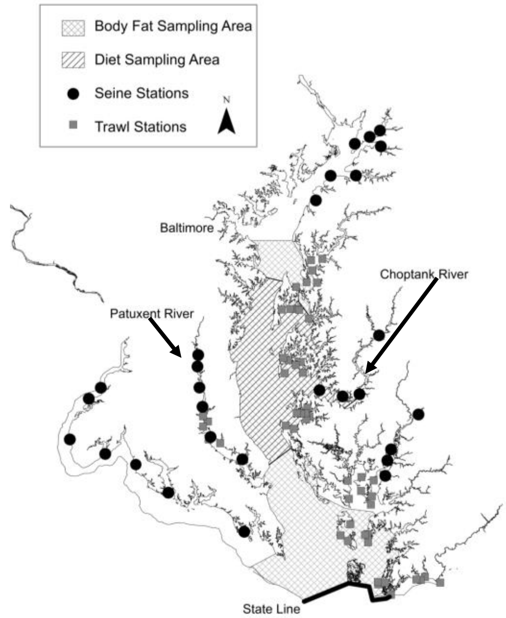

Figure 2. Proportion of Striped Bass without body fat (P0) during October-November (MD DNR Fish and Wildlife Health Program monitoring), with body fat targets and limits.

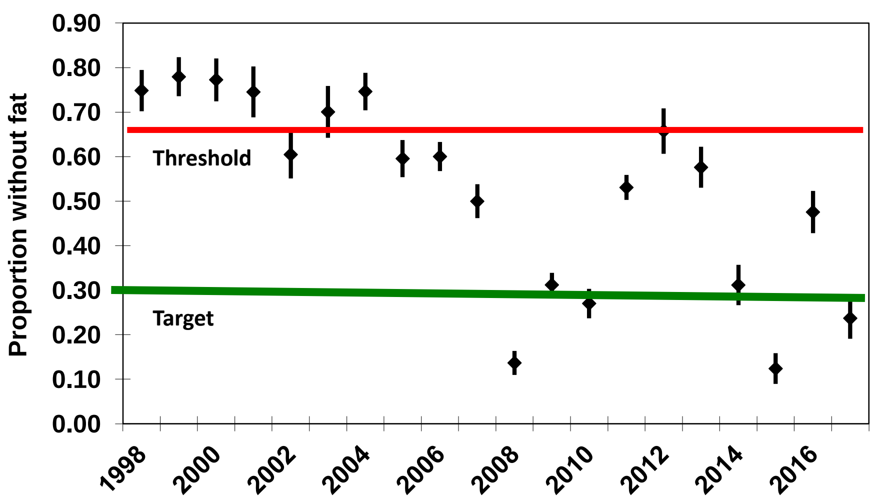

Figure 3. Trends in fall body fat indices (P0) for small (280-456 mm) and large striped bass

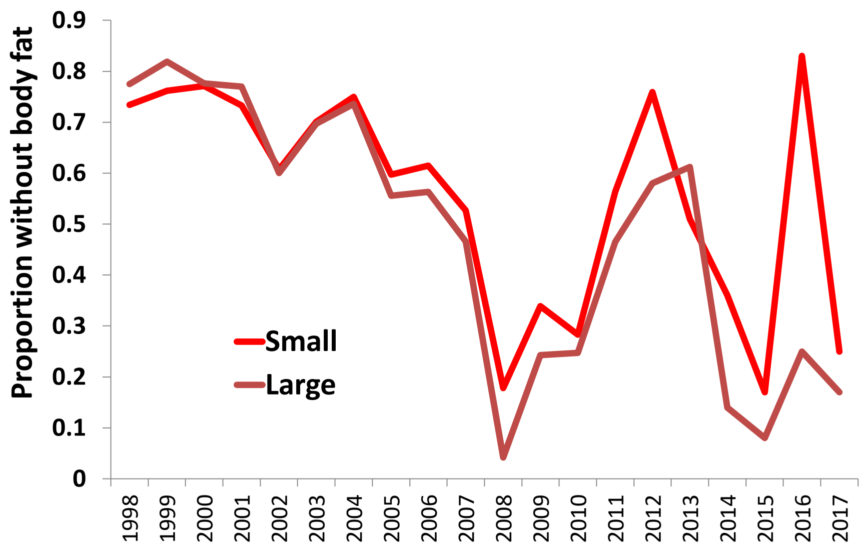

Figure 4. Trends in major pelagic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2017. Indices were standardized to their 1989-2017 means (time-series in common). Menhaden = Atlantic Menhaden and Anchovy = Bay Anchovy.

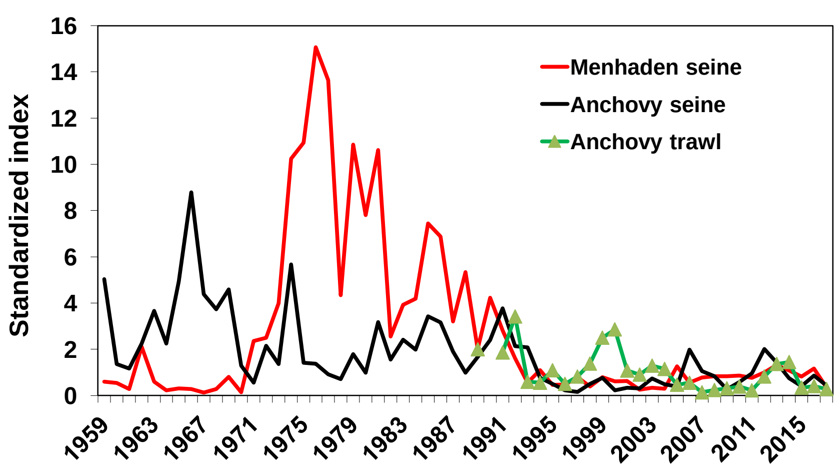

Figure 5. Trends in major benthic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2017. Indices were standardized to their 1989-2017 means (time-series in common).

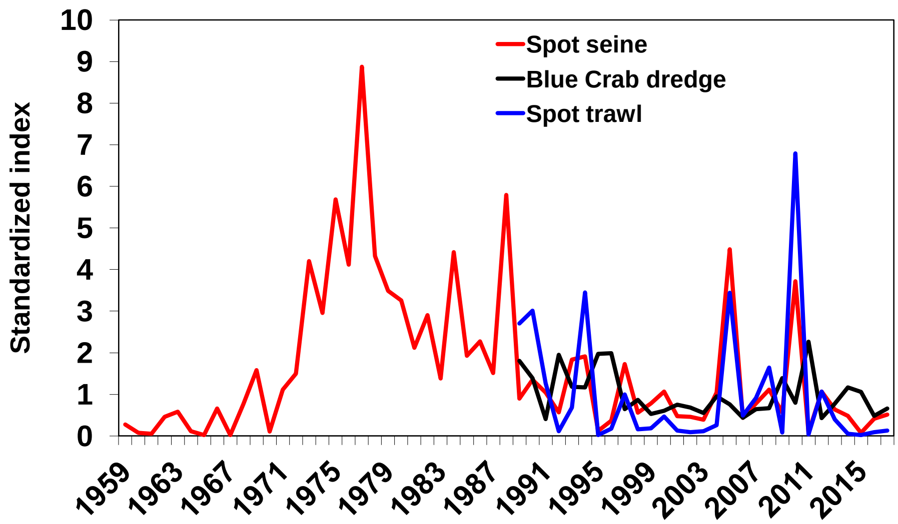

Figure 6. Maryland resident Bay Striped Bass abundance index (RI; MD MRIP recreational catch per private boat trip; mean = black line) during 1981-2017 and its 90% CI's based on @Risk simulations of catch and effort distributions. Catch = number harvested and released.

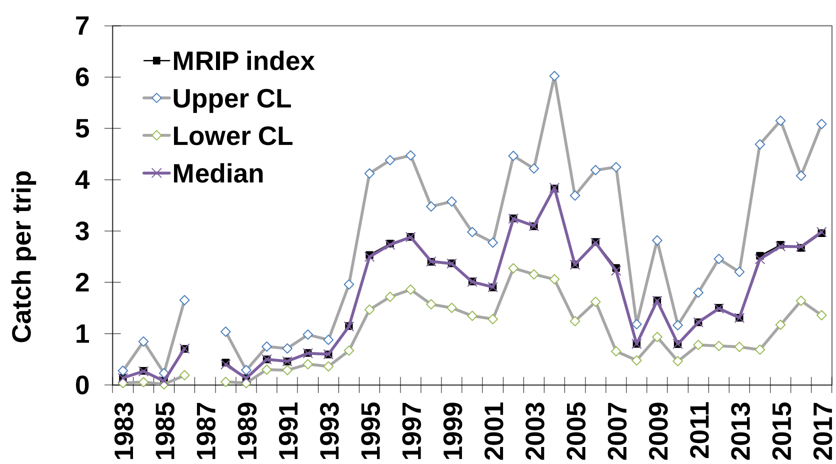

Figure 7. Maryland resident Striped Bass abundance index (RI; MD MRIP recreational catch per private boat trip; mean = black line) during 1981-2016 and its 90% CI's (grey lines). Catch = number harvested and released. CI's estimated using the technique of Goodman (1960).

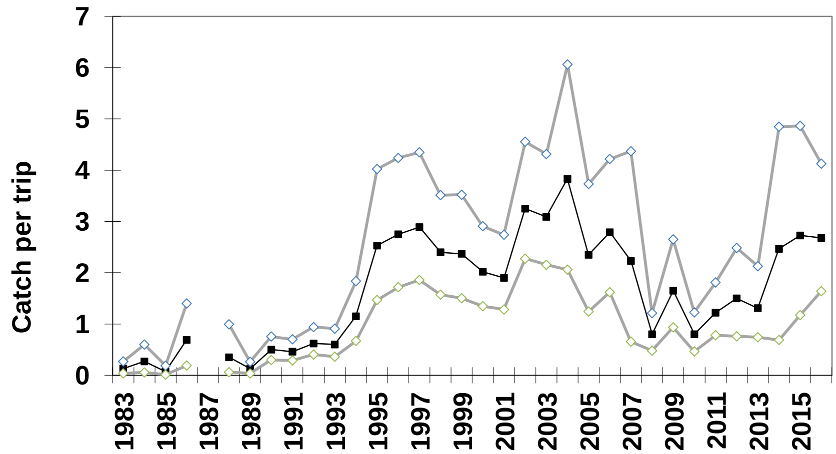

Figure 8. Atlantic Menhaden index to Striped Bass index (RI) ratios during 1983-2017 and their 90% CI's based on @Risk simulations of Atlantic Menhaden seine indices and RI distributions. Note $\log_{10}$ scale.

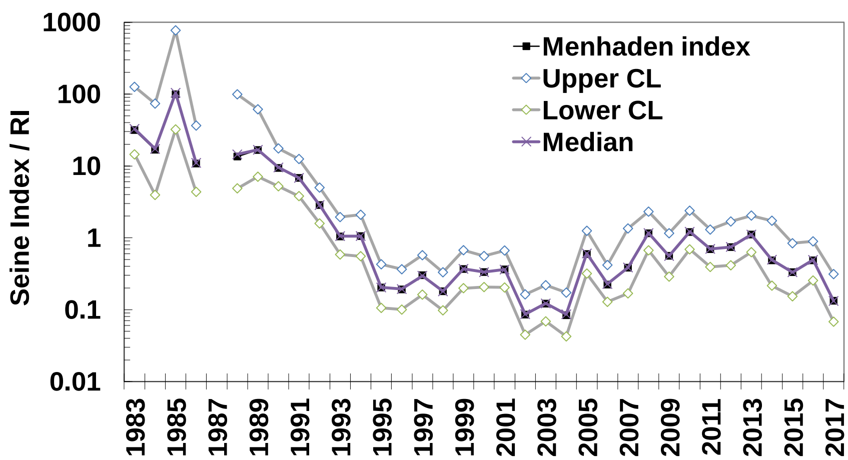

Figure 9. Bay Anchovy seine index to Striped Bass index (RI) ratios during 1983-2017 and their 90% CI's based on @Risk simulations of Bay Anchovy seine indices and RI distributions. Note log_{10} scale.

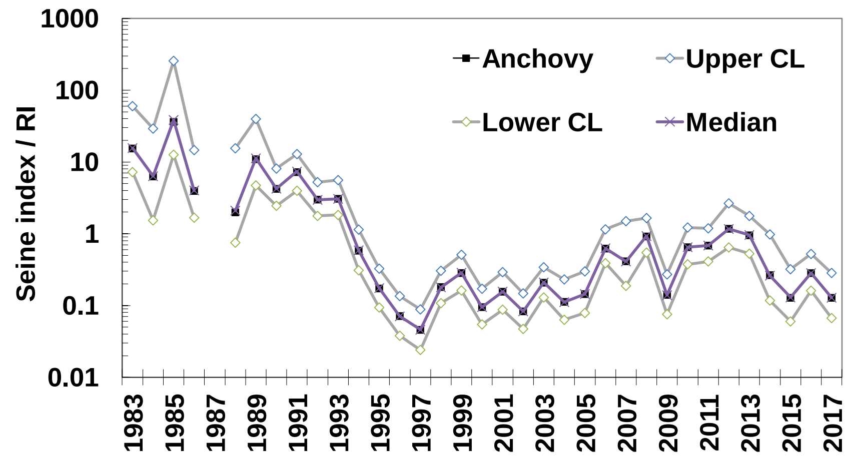

Figure 10. Bay Anchovy trawl index to Striped Bass index (RI) ratios during 1989-2017 and their 90% CI's based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note log_{10} scale.

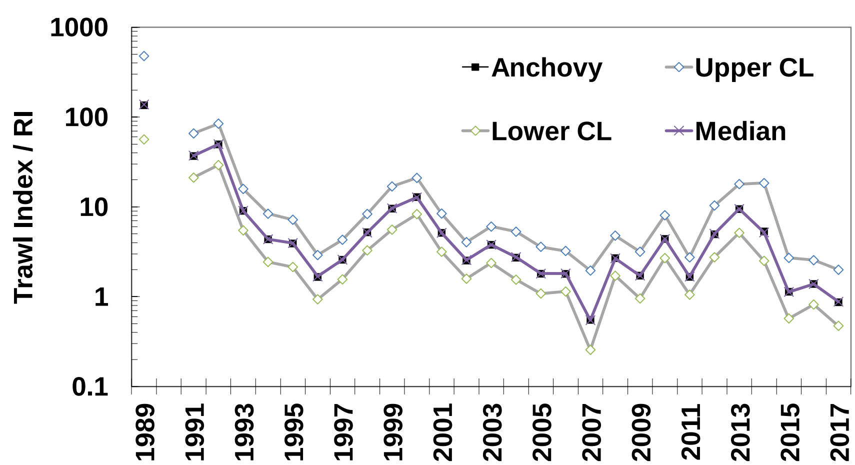

Figure 11. Spot seine index to Striped Bass index (RI) ratios during 1983-2017 and their 90% CI 's based on @Risk simulations of central tendency and estimated dispersion of data of Spot seine indices and RI. Note $\log_{10}$ scale.

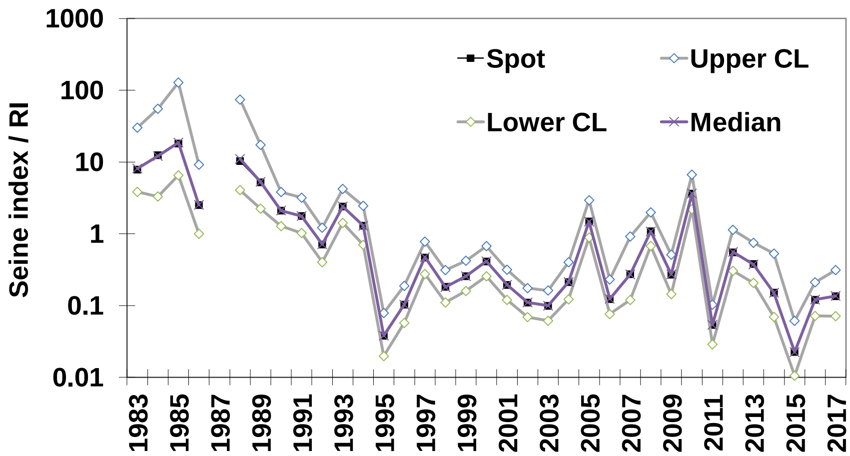

Figure 12. Spot trawl index to Striped Bass index (RI) ratios during 1989-2017 and their 90% CI's based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale.

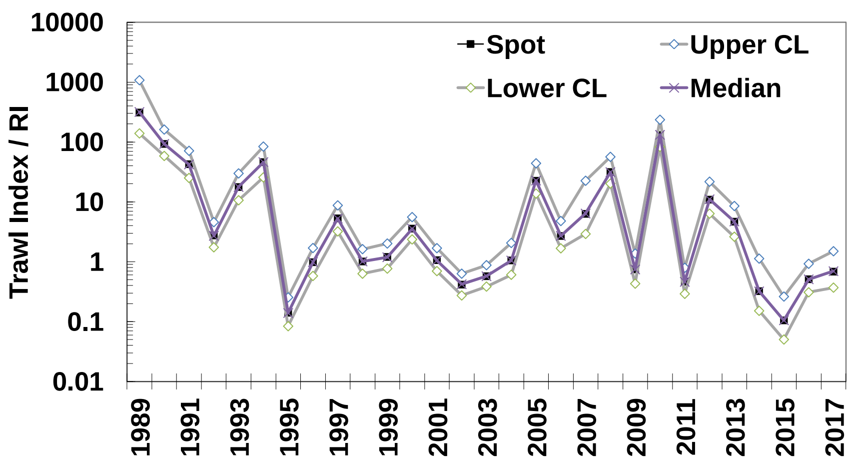

Figure 13. Blue Crab index to Striped Bass index (RI) ratios during 1989-2017 and their 90% CI's based on @Risk simulations of central tendency and estimated dispersion of data of Blue Crab (age 0) winter dredge densities and RI. Note log_{10} scale.

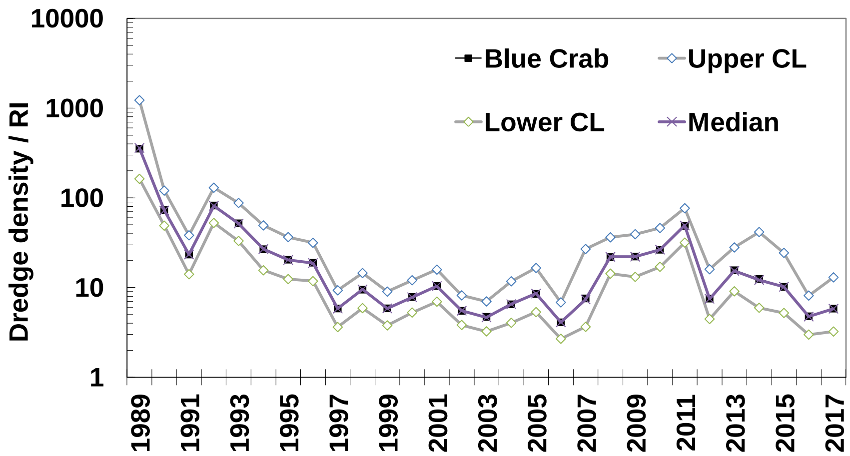

Figure 14. Trends of standardized ratios major upper Bay forage species indices to Striped Bass relative abundance (RI). Forage ratios have been standardized to their 1989-2017 mean to place them on the same scale. S indicates a seine survey index; T indicates a trawl survey index; and D indicates a dredge index. Note the log_{10} scale.

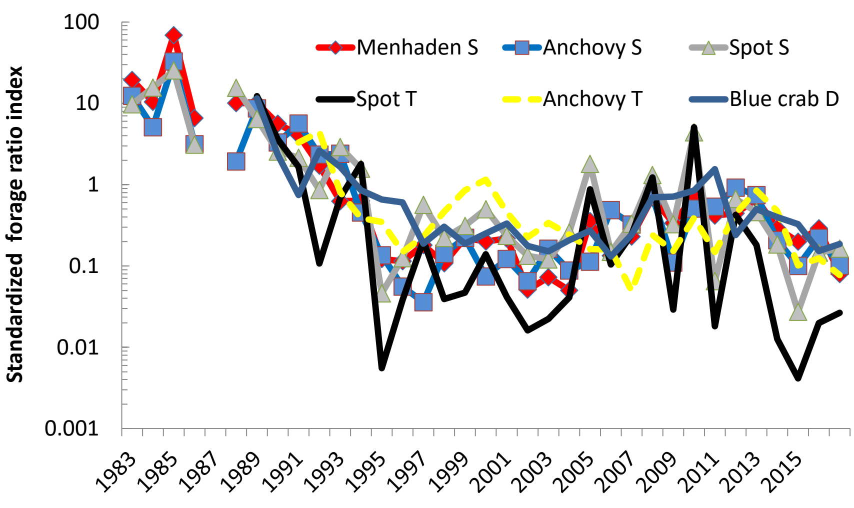

Figure 15. Standardized ratios of major forage indices / Striped Bass RI and their weighted mean during the time period when body fat (P0) indices were available. S indicates a seine survey index; T indicates a trawl survey index; and D indicates a dredge index.

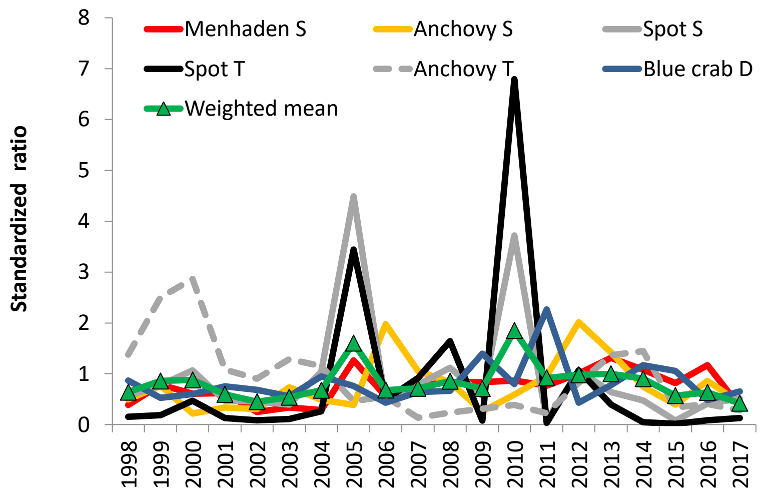

Figure 16. Percent of small Striped Bass (< 457 mm TL) diet represented by major forage groups, by number, in fall.

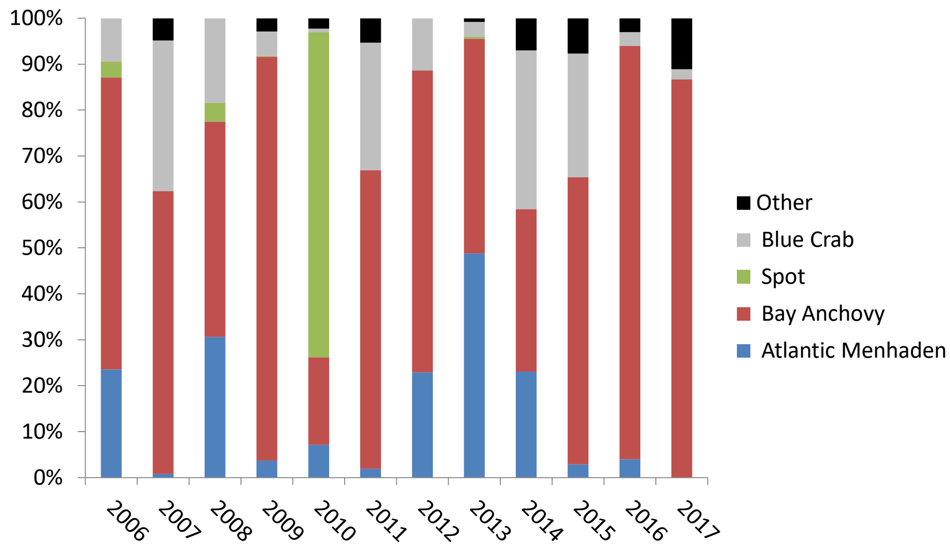

Figure 17 . Percent of large Striped Bass (≥ 457 mm TL) diet represented by major forage groups, by number, in fall.

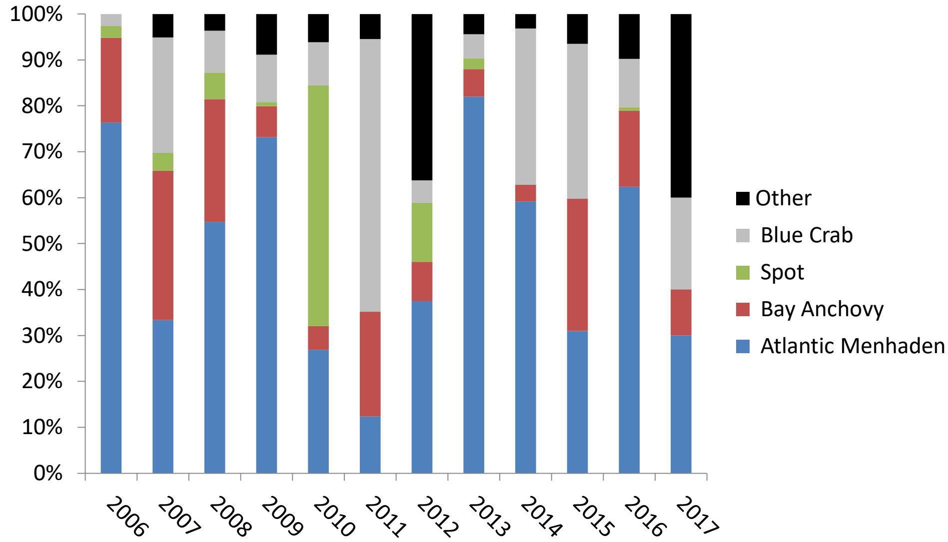

Figure 18. Gram prey consumed per gram (C) of small (< 457 mm TL) Striped Bass in fall hook-and-line samples. Age-0 forage dominate the diet. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

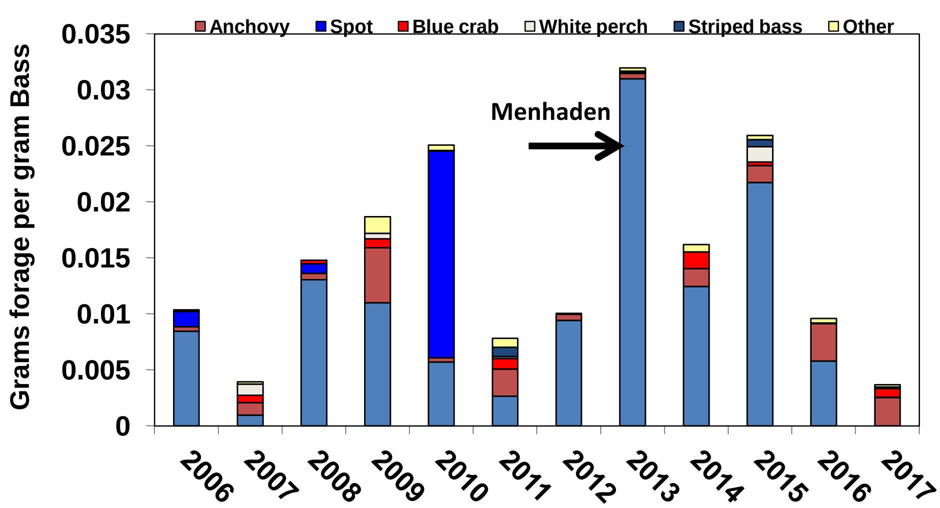

Figure 19. Grams of prey consumed per gram (C) of large (≥ 457 mm TL) Striped Bass during October-November. Fall consumption dominated by age 0 forage. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

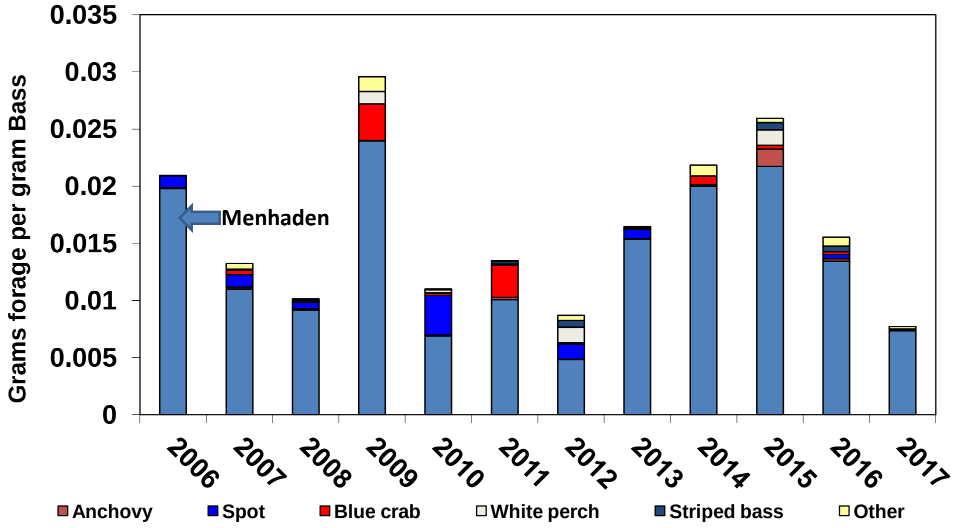

Figure 20. Proportion of small Striped Bass guts without food (PE) in fall. Red diamond represents threshold PE and green diamond indicates the PE target.

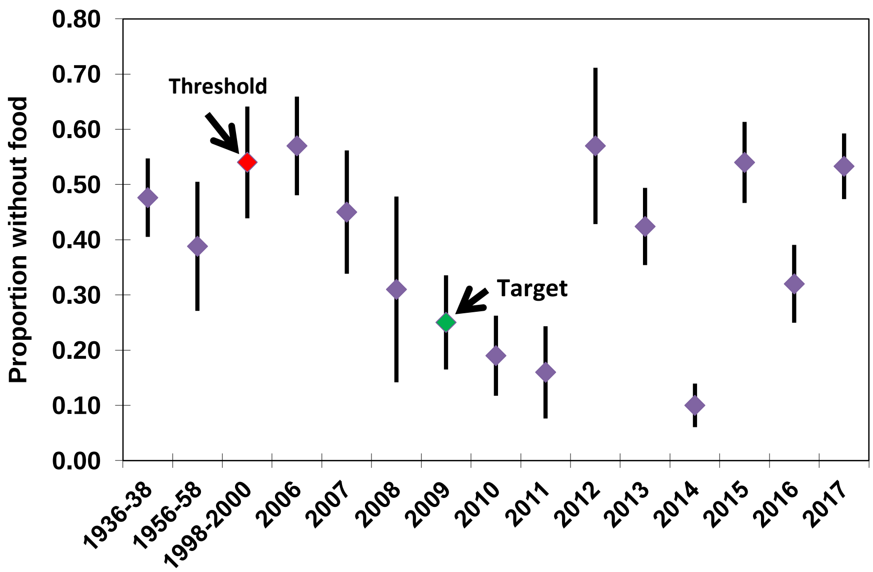

Figure 21. Proportion of large Striped Bass (> 456 mm or 18 in, TL) guts without food in fall.

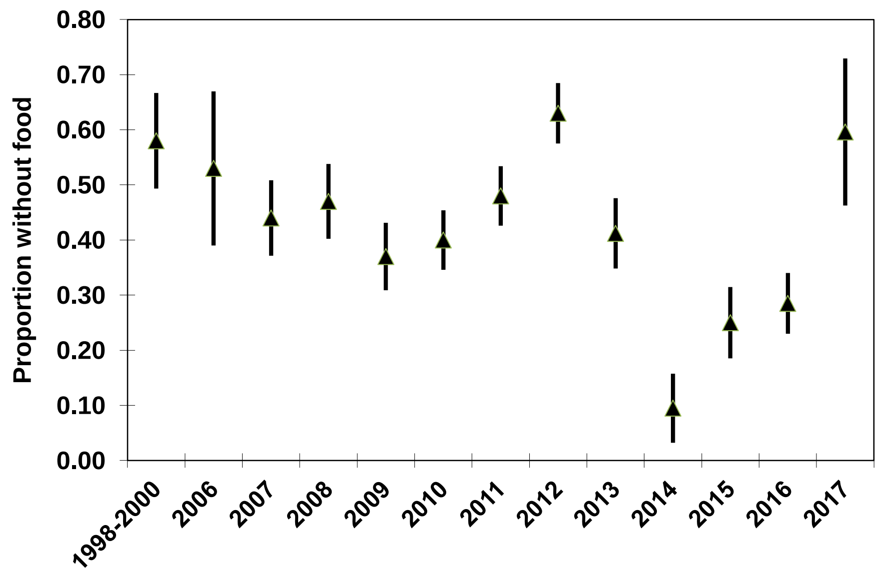

Figure 22. Time-series of age 3 male Striped Bass relative abundance on two major Maryland spawning areas (Age 3 CPUE; units = number of fish captured in 1000 square yards of net per hour) and abundance (N) of age 3 Striped Bass along the Atlantic Coast estimated by the ASMFC (2016) statistical catch-at-age model.

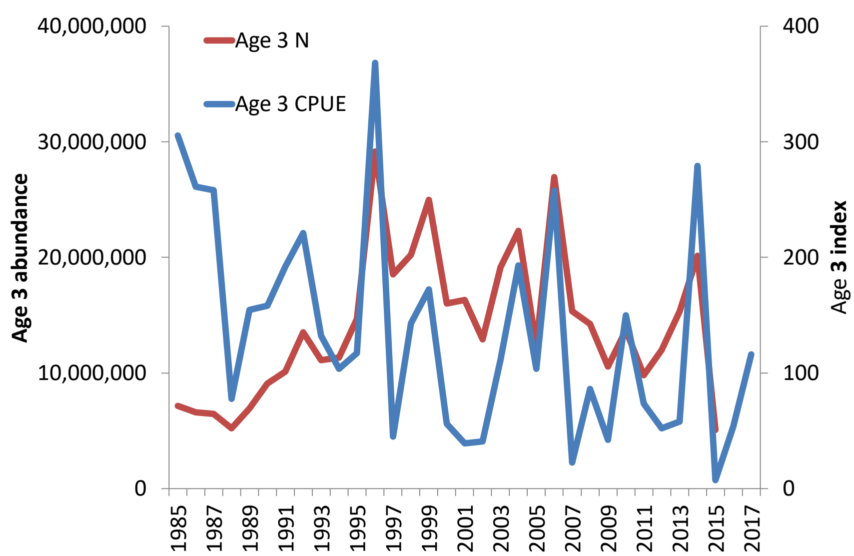

Figure 23. Trend in catchability of age 3 Striped Bass in Maryland's spring gill net survey.

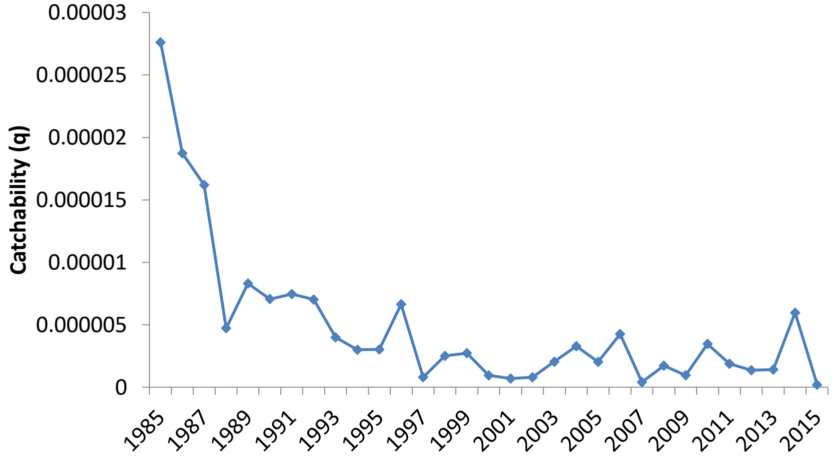

Figure 24. Time-series of age 3 Striped Bass relative abundance on two major Maryland spawning areas (Hybrid index = index adjusted for changing catchability during 1985-1995; units = number of fish captured in 1000 square yards of net per hour) and abundance (N) of age 3 Striped Bass along the Atlantic Coast estimated by the ASMFC (2016) statistical catch-at-age model.

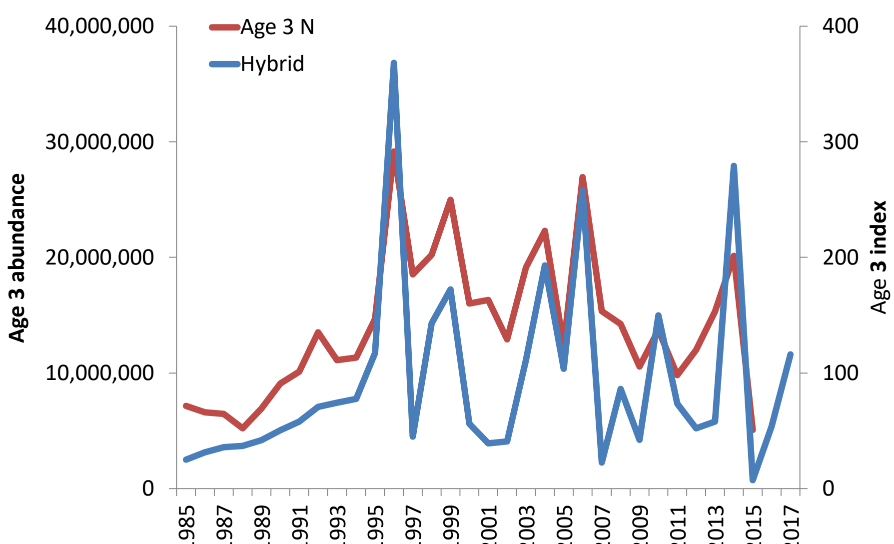

Figure 25. Relative survival (SR) of Striped Bass during 1985-2017 and 90% CI's based on @Risk simulations of age 3 hybrid gill net indices and juvenile index (year -3) distributions.

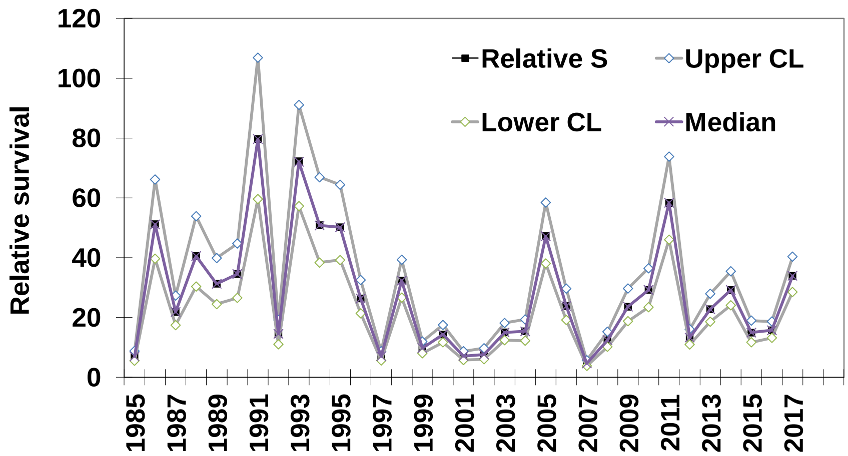

Figure 26. Relative survival of Striped Bass during 1985-2017 with targets and limits.

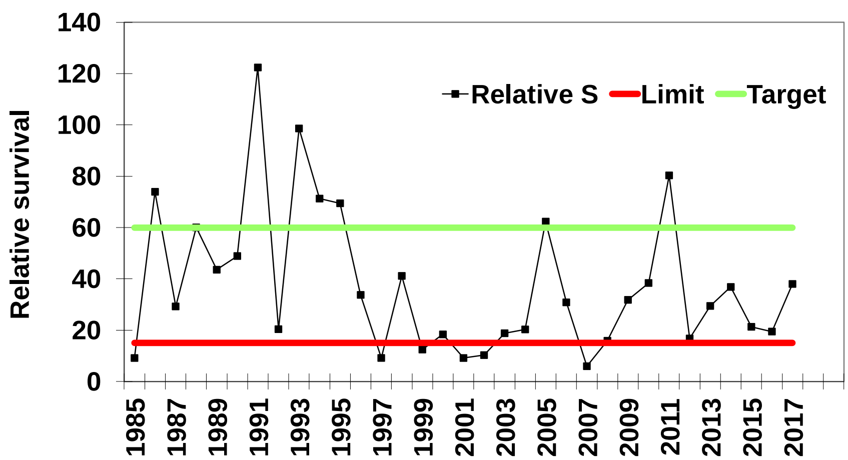

Figure 27. Index of Forage (IF) and its component scores. IF averages scores given to five indicators of forage status in upper Bay. A score of 5 indicates target conditions were met; 1 indicates threshold conditions; 4 indicates target was approached; 3 indicates threshold conditions were avoided; and 2 indicates threshold conditions were approached. RI = index of relative abundance of resident Striped Bass; FR = ratio of averaged major forage indices to RI; P0 = proportion of Striped Bass without body fat in fall; SR is relative survival of male Striped Bass to age 3; and PE = proportion of Striped Bass with empty guts in fall.

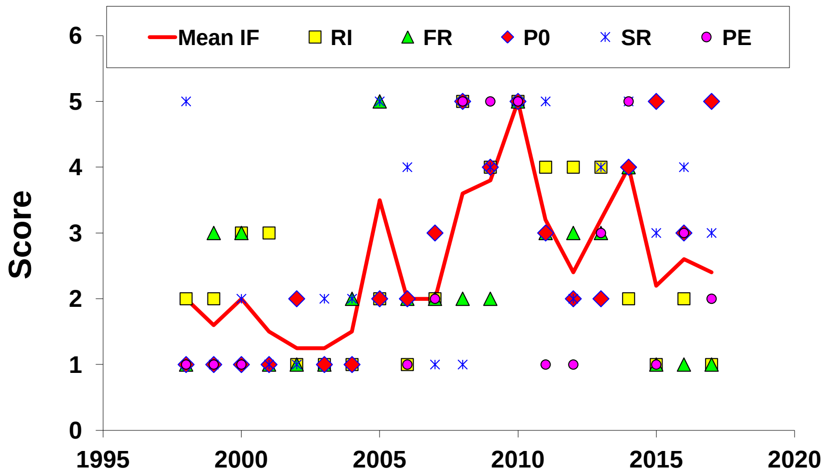

Figure 28. Forage index with all components averaged (Mean IF) and averaged with each component removed. See Figure 22 for explanation of scores and abbreviations.

Figure 29. Benthic organism biomass density (grams / m^2) and its median estimated for upper Bay for the Benthic Index of Biotic Integrity during 1998-2016 (Lansó and Zaveta 2017).

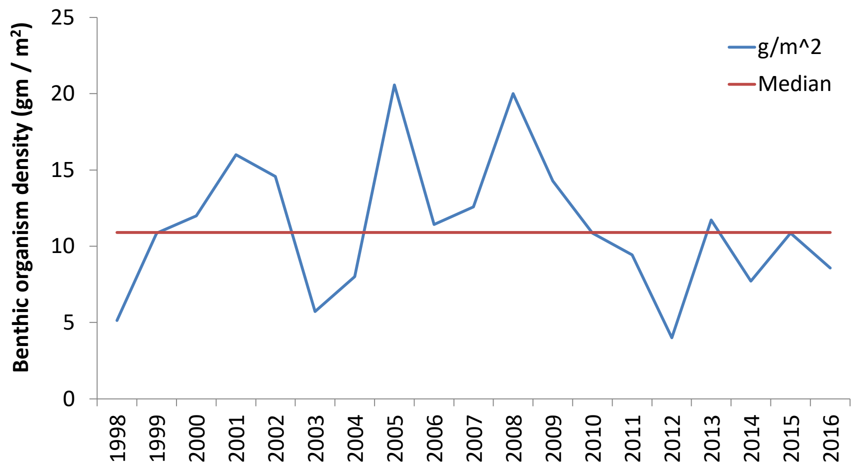

Figure 30. Relationship of the proportion of Striped Bass without body fat (P0) to benthic organism density for two time periods. Relationship was determined by multiple regression of time category (1998-2007 or 2008-2016) and benthic organism density with P0.

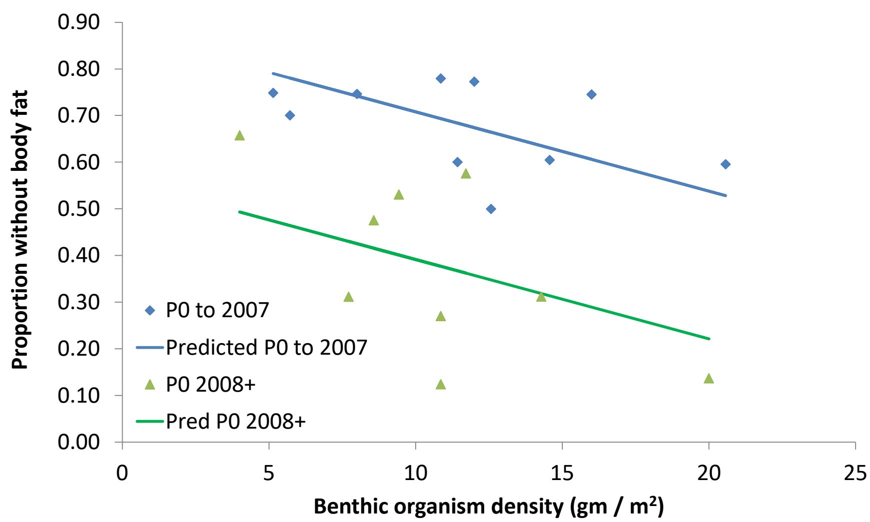

Figure 31. Median prey-predator length ratios (PPLR) for major prey (Spot and Atlantic Menhaden) for small (< 457 mm) and large Striped Bass.

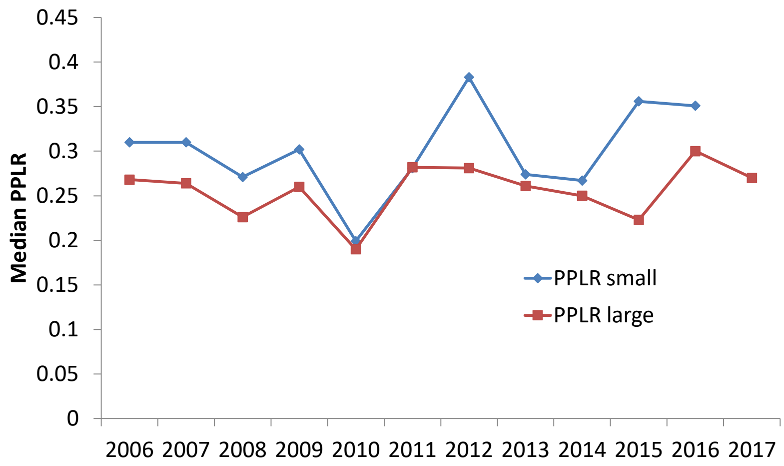

Figure 32. Relationship of estimates of proportion of Striped Bass without body fat during 2006-2014 made by two independent sampling programs. CBEF = Chesapeake Bay Ecological Foundation. FWHP = Fish and Wildlife Health Program.