PERFORMANCE REPORT FOR FEDERAL AID GRANT F-63-R, SEGMENT 9

2018

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
December, 2019

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Boyd Rutherford
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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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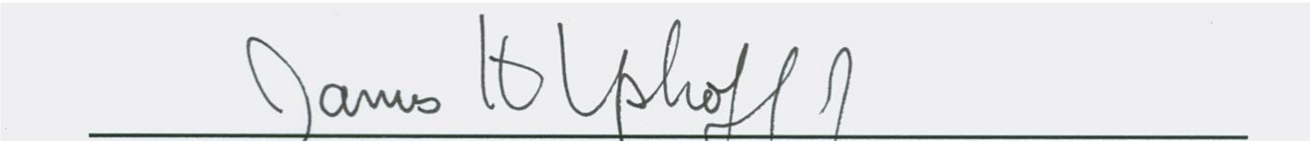

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Mattawoman Creek	Mattawoman Watershed Society
Bush River	Anita Leight Estuary Center
Mapping	Marek Topolski MD DNR

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Report Organization

This report was completed during December, 2019. 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 habitat 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.

<u>STATE:</u> MARYLAND

<u>SURVEY TITLE:</u> MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

<u>PROJECT 1</u>: 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: 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. Recalculation of the equation previously used to convert annual estimates of C/ha to estimates of impervious surface (IS) was necessary in 2018 due to a new time-series provided by MD Department of Planning, as well as inconsistencies found in the data for some watersheds up to 2002. New 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 0.37, 0.86, and 1.35 C/ha, respectively. Previous C/ha estimates corresponding to 5%, 10%, and 15% IS were 0.27, 0.83, and 1.59, respectively (Uphoff et al. 2018). Percent of watershed in agriculture, forest, and wetlands were estimated from Maryland Department of Planning spatial data.

Section 1, Stream Ichthyoplankton - Proportion of samples with Herring eggs and-or larvae (P_{herr}) provided a reasonably precise indicator of habitat occupation based on encounter rate. Regression analyses that included spawning stock categories (0 for low during 2005-2011 and 1 for high during 2012-2018), indicated significant and logical relationships among P_{herr}, C/ha, and conductivity consistent with the hypothesis that urbanization was detrimental to stream spawning. Predicted P_{herr} declined by 53% over the range of observed C/ha (0.07-1.52; and increased by 55% between the two spawning stock categories. Predicted P_{herr} declined by 47% over the range of observed conductivity standardized to its baseline (1.14-2.19) and increased by 57% between the two spawning stock categories. The high spawning stock category in the analysis of 2005-2018 corresponded with closure of Maryland's River Herring fisheries in 2011, closure of most other in-river fisheries along the Atlantic Coast by 2012, and caps on River Herring bycatch in coastal Atlantic Herring and Atlantic Mackerel fisheries.

Herring spawning declined in streams as watersheds developed and conductivity increased. Conductivity was positively related with C/ha in our analysis, and with urbanization in other studies. 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. Estimates of P_{herr} were consistently high in the three watersheds dominated by agriculture, while importance of forest cover could not be assessed with confidence since it was possible that forest cover estimates included residential tree cover. General development targets and limits for C/ha or IS worked reasonably well in

characterizing habitat conditions for stream spawning of Herring. Low estimates of P_{herr} (≤ 0.4) were much more frequent beyond the C/ha threshold or when standardized conductivity was 1.5-times or more than the baseline level. Estimates of P_{herr} were consistently above 0.6 when development was less than the C/ha target.

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 large scale land use and L_p emerged from the expanded analyses conducted for this report: L_p was negatively related to development and positively associated with forest and agriculture. Development was an important influence on Yellow Perch egg and larval dynamics and negative changes generally conformed to development reference points. Higher DO and pH measurements in urbanized large subestuaries sampled since 2015 (Patuxent and Wicomico rivers) during L_p surveys indicated their water quality dynamics were different from the rural, agricultural Choptank River watershed.

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 in the subestuaries we studied.

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.

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 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.

Median bottom DO in mesohaline subestuaries increased as agricultural coverage went from 3 to 39%; these watersheds were located on the western shore. Median DO measurements beyond this level of agricultural coverage (43-72% agriculture) were from eastern shore subestuaries and the DO trend appeared to be stable or slightly declining. 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 43%-72% of their watershed covered in agriculture. 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. Predicted median bottom DO at the highest level of agriculture observed would equal 4.3 mg/L, between the DO target and threshold.

Section 3: Estuarine Community Sampling in Summer: Subestuary Surveys – We continued to examine and Tred Avon River, a tributary of Choptank River located in Talbot County. We contrasted Tred Avon River with two adjacent subestuaries: Broad Creek (sampled during 2012-2017) and Harris Creek, (2012-2016). Broad and Harris creeks have just passed the target level of development, while Tred Avon River is approaching the development threshold. In

2018, we returned to previously sampled middle Bay subestuaries: Chester River, Corsica River, Langford Creek, and Wye River. These subestuaries are located in Queen Anne's County and we sampled them to support the County's pending comprehensive growth plan. We examined associations among relative abundance of all finfish from Choptank River and the Head of Bay with Chester and Tred Avon Rivers to evaluate potential contributions of the two large outside regions to the abundance in subestuaries in our study. High to record rainfall in the Chesapeake Bay watershed and runoff preceded and continued during summer 2018 sampling. We added an evaluation of precipitation patterns to our analysis in order to better understand how increased precipitation conditions may have impacted our evaluations of watersheds during 2018.

High precipitation in 2018 did not have an overwhelming impact on survey water quality measurements. The increase in rainfall in 2018 caused a decline in salinities, possibly altering the composition of finfish and shifting the migratory range finfish are known to inhabit. Salinities in most subestuaries sampled were at the lower bounds of what had been observed during previously, but remained within their salinity class. Chester River was an exception; salinity dropped enough in 2018 for it to be classified as oligohaline.

Bottom DO conditions conformed to the expected relationship with development and salinity class. Frequency of below threshold bottom DO (3 mg / L) held steady at the level estimated (13% -14%) since 2015 in Tred Avon River (this watershed is approaching the development threshold), but below target DO (5 mg / L) became more frequent. Queen Anne's County watersheds all were at or below the target level of development. Bottom DO in 2018 was most likely to be above the target level and measurements that breached the threshold were uncommon in Chester River, Corsica River, and Langford Creek. Most bottom DO measurements in Wye River fell between the target and threshold level, but below threshold readings were much more common in 2018 than previous surveys. Other water quality metrics (pH and Secchi depth) in all subestuaries sampled during 2018 were within previous years' ranges.

Finfish catches in trawls sampling bottom water habitat declined among all subestuaries sampled. A drop in trawl relative abundance was common among the subestuaries sampled and did not reflect bottom DO (except in the upper most station in Tred Avon River). Species composition changed, reflecting of a drastic drop in Bay Anchovy and a concurrent substitution of species that would have fallen into the "other species" category had Bay Anchovy abundance not fallen. Inshore seine catches were within a normal range. Correlation analyses suggested that relative abundance of finfish in the subestuaries sampled during 2018 could be influenced by production from larger adjacent regions.

Common Background for Job 1, Sections 1-3

Jim Uphoff

"*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 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. 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 normoxic 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 was estimated 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 2019). 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 GIS database. Files were managed and geoprocessed in ArcGIS 10.3.1 from Environmental Systems Research Institute (ESRI 2015). 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, although there is typically a two-year lag in processing by MD DOP. Tax data through 2014 or 2016 were available for the 2018 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.

During 2003-2010, we used impervious and watershed area estimates made by Towson University from Landsat, 30-meter pixel resolution satellite imagery (eastern shore of Chesapeake Bay in 1999 and western shore in 2001) as our measure of development for each watershed (Barnes et al. 2002). They became outdated and C/ha provided a readily updated substitute. Uphoff et al. (2012) developed an nonlinear power function to convert annual estimates of C/ha during 1999-2000 for watersheds sampled during 2003-2009 (Table 1) to estimates of percent impervious surface (IS) calculated by Towson University from 1999-2000 satellite imagery. This equation was used to convert each year's C/ha estimates to IS.

Recalculation of this conversion equation was necessary in 2018 due to a new time-series provided by MD DOP, as well as inconsistencies found in the data for some watersheds up to 2002 (M. Topolski, MD DNR, personal communication). Historic data were recalculated using 2002 MdProperty View data (previously 1999 data had been used) which corrected data deficiencies in the 2000 and 2001 data, as well as errors in the 1999 data (Table 1; M. Topolski, MD DNR, personal communication). The same watersheds and years used to estimate the original nonlinear relationship (Uphoff et al. 2012) were used in the update to maintain continuity.

A linear regression described the updated relationship well:

$$IS = (10.129 \cdot C/ha) + 1.286; (r^2 = 0.905; P < 0.0001; \text{Figure 1}).$$

New 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 0.37, 0.86, and 1.35 C/ha, respectively. The previous C/ha estimates, based on a nonlinear power function, corresponding to 5%, 10%, and 15% IS were 0.27, 0.83, and 1.59, respectively (Uphoff et al. 2018).

Percent of watershed in agriculture, forest, and wetlands were estimated from MD DOP 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. Urban land consisted of high and low density residential, commercial, and institutional acreages and was not a direct measure of IS.

Land use and land cover (LULC) shapefiles were available for 1973, 1994, 1997, 2002, and 2010 for each Maryland jurisdiction and as an aggregated statewide file. Metadata for the LULC categories is available for download from MD DOP. 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 its land area in hectares was calculated. The land use total was divided by the watershed total to the nearest tenth of a hectare and multiplied by 100%.

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. Typical 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, fish presence-absence or relative abundance, and fish diversity in summer (Section 3).

Correlations among watershed estimates of C/ha and percent of watershed estimated in urban, agriculture, forest, and wetland based on MD DOP spatial data were used to describe associations among land cover types. 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 MD DOP 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 0.80; 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 reported, but potential management and biological significance took precedence over significance at $P < 0.05$ (Anderson et al. 2000). We classified correlations as strong, based on $r \geq 0.80$; weak correlations were indicated by $r < 0.50$; and moderate correlations fell in between. Relationships indicated by regressions were considered strong at $r^2 \geq 0.64$; weak relationships were indicated by $r < 0.25$; and moderate relationships fell in between. Residuals of regressions 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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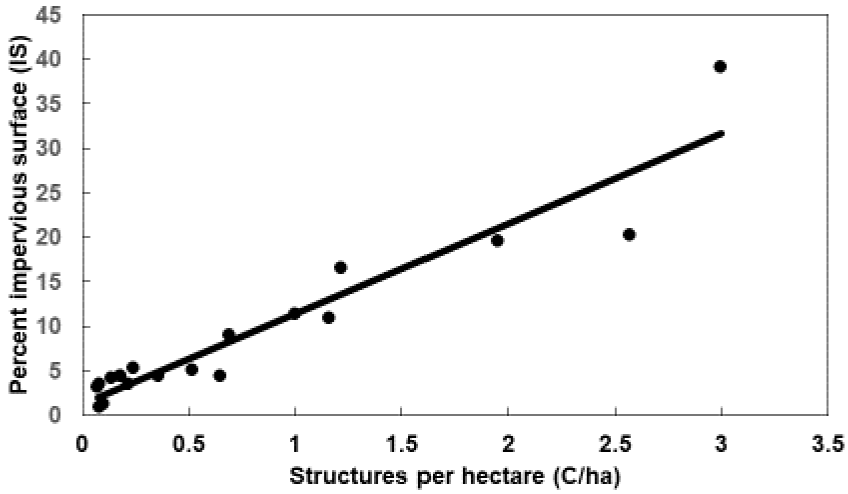
Table 1. Structures per hectare (C/ha) and percent impervious surface estimates (IS) used to estimate the relationship for predicting IS from C/ha. Old C/ha were estimates used previous to this report and New C/ha were revised estimates used to estimate the current relationship.

Watershed	Old C/ha	New C/ha	IS
Nanjemoy Creek	0.08	0.08	0.9
Bohemia River	0.10	0.10	1.2
Langford Creek	0.07	0.07	3.1
Wye River	0.08	0.08	3.4
Miles River	0.23	0.22	3.4
Corsica River	0.14	0.14	4.1
Wicomico River west	0.29	0.18	4.3
Northeast River	0.36	0.36	4.4
Gunpowder River	0.03	0.65	4.4
St Clements Bay	0.19	0.18	4.4
West River Rhode River	0.55	0.52	5.0
Breton Bay	0.25	0.24	5.3
Mattawoman Creek	0.71	0.69	9.0
South River	1.23	1.16	10.9
Bush River	0.98	1.00	11.3
Piscataway Creek	1.34	1.22	16.5
Severn River	2.14	1.95	19.5
Magothy River	3.01	2.57	20.2
Middle River	7.39	3.00	39.1

Figure 1. Relationship of structures per hectare (C/ha) and percent impervious surface (IS).



Section 1: Stream Ichthyoplankton Sampling

Carrie Hoover, Alexis Park, Margaret McGinty, Jim Uphoff, and Marcus Patton

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). Increases in impervious surface 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 (2008-2018), 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 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 drainage, which includes Tuckahoe Creek, is a major tributary of the Chesapeake Bay that has an agricultural watershed and is entirely within the eastern shore's Coastal Plain. 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). Fluvial Patapsco River meets the Chesapeake Bay and forms the port of Baltimore.

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; Topolski 2015) between the 1970s and the present. 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).

In addition, we attempted to address the possibility that proportion of samples with anadromous Herring may have been impacted by spawning stock abundance increases due to more restrictive coast-wide regulatory measures implemented over the past decade. Closures of most in-river fisheries along the Atlantic Coast were in place by 2012 (including Maryland in 2011; ASMFC 2019) and caps on River Herring bycatch in Atlantic Herring and Atlantic Mackerel fisheries that started in 2014 (MAFMC 2019) could have boosted Herring spawning stock. Increases in presence of Herring eggs and-or larvae due to regulatory measures should potentially have been evident across three watersheds studied before and after regulatory measures were put in place. Increases in spawning stock abundance over time would have the potential to bias estimated relationships of C/ha and conductivity with indicators of anadromous Herring stream spawning intensity.

Methods

Stream sites sampled for anadromous fish eggs and larvae during 2005-2018 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 2005-2018, 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 2016-2018, 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 during 2013-2017 by U.S. Fish and Wildlife Service 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 2005-2018.

Ichthyoplankton samples were collected in all systems and years using stream drift nets constructed of 360-micron mesh. Nets were attached to a square frame with a 300 • 460 mm opening. The stream drift net configuration and techniques were the same as those used by O'Dell et al. (1975). The 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 (Uphoff et al. 2017; 2018). 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 YSI Pro2030 meter. Meters were calibrated for DO each day prior to use. All data were recorded on standard field data sheets 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. With the exception of 2018, all eggs and-or larvae were removed and 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 systematically sorted 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 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.

Collections and sample processing were adjusted in 2018 due to anticipated time and staffing limitations. Mattawoman Creek volunteers received training on field identification of Herring eggs and larvae prior to the start of the season, and if they were able to determine presence in the field the sample was not retained. Samples which they could not determine conclusively contained Herring, or ones in which no eggs or larvae were observed in the field, were preserved for laboratory examination. In the lab, samples were sorted only for presence of Herring eggs and-or larvae. Once a Herring egg or larvae was encountered, processing of the sample was considered complete, regardless of how much of it had been gone through.

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 24,430 ha and estimated C/ha increased from 0.87 to 0.93 during 2008-2014 (the most recent year C/ha data is available); Piscataway Creek's watershed was 17,634 ha and estimated C/ha increased from 1.41 to 1.50 during 2008-2014; Bush River's watershed was 36,009 ha and estimated C/ha increased from 1.37 to 1.52 during 2005-2014; and Deer Creek, a spawning stream with low development, had a watershed of 37,724 ha and estimated C/ha was 0.24 during 2012-2015 (Table 1-1). The upper portion of the Choptank River (watershed area = 38,285 ha and developmental level = 0.18 C/ha) and a tributary of the Choptank River, Tuckahoe Creek (watershed area = 39,364 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 93,730 ha and estimated C/ha was 1.11 in 2013 and 1.12 in 2014. Collections in the Patapsco River were made by U.S. Fish and Wildlife Service and were provided at no charge to this grant.

Conductivity measurements collected for each date and stream site (mainstem and tributaries) during 2008-2018 from Mattawoman Creek were plotted and mainstem measurements were summarized for each year. Mainstem sites would be influenced by development in Waldorf, the major urban influence on the watershed, while the monitored tributaries would not (Figure 1-2). 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 2008-2018 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 2008-2018 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). Historical monthly median conductivities at each mainstem Mattawoman Creek non-tidal site and 2008-2018 spawning season median conductivities were plotted together.

Presence of eggs and-or larvae of White Perch and Yellow Perch at each station through 2017, and Herring in 2018, 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), Deer Creek in 1972 (O'Dell et al. 1975), Bush and Patapsco Rivers in 1973 (O'Dell et al. 1975), and Tuckahoe Creek in 1976-77 (O'Dell et al. 1980).

The proportion of samples where Herring eggs and-or larvae were present (P_{herr}) was estimated for Mattawoman Creek mainstem stations (MC1-MC4) during 1991 and 2008-2018, 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 = 0.48) 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 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, TUC108, TUC109, and TUC110 correspond to O'Dell et al. (1980) sites 4, 5, 6, 8, 11, and 12 respectively (Figure 1-7). Four 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 and-or 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:

$$^{(3)}\ P_{herr} \pm (1.645 \bullet SD).$$

Two regression approaches were used to examine possible linear relationships between C/ha or standardized conductivity and P_{herr}: simple linear regression and multiple regression using two dependent variables: a categorical variable to indicate two levels of spawning stock (low and high) and C/ha or standardized conductivity. Simple linear 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-five sets of estimates of C/ha, percent agriculture, percent forest, and P_{herr} were available (1991 estimates for Mattawoman Creek could be included), while 34 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} (see Uphoff et al. 2018). Nonlinear power functions were used to fit curvilinear models. Simple 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, watersheds that run through both physiographic provinces, conductivities were standardized using the 150 μS/cm of Piedmont streams since sampling locations were solely within that region.

Multiple regression of C/ha or standardized conductivity and spawning stock class against P_{herr} assumed slopes were equal for two stock size categories, but intercepts were different (Neter and Wasserman 1974; Rose et al. 1986; Freund and Littell 2006). This common slope would describe the relationship of C/ha or standardized conductivity to P_{herr}, while the intercept would indicate the effect of high or low spawning stock size. This analysis was conducted for the continuous 2005-2018 time-series and excluded 1991. These analyses were initially done in Excel and run again in SAS (Proc Reg) to confirm the estimates. Spawning stock size was modeled as an indicator variable in the multiple regression with 0 indicating low spawning stock prior to the full implementation of river closures and bycatch reductions (2005-2011) and 1 indicating higher spawning stock following these measures (2012-2018). Categorizing spawning stock was necessary because P_{herr} would be the indicator of spawning stock size for each watershed and the dependent variable in the analysis if used as a continuous variable. None of the watersheds studied had independent indicators of spawning stock size. Rose et al. (1986) presented the use of categorized variables and linear regression as an alternative to Box-Jenkins models and time-series regression. In addition to standard regression output, we also used the type II sums of squared partial correlation coefficients to examine the amounts of variation in P_{herr} explained by each independent term in the multiple regression models after holding the other constant (Ott 1977; Sokal and Rohlf 1981; Afifi and Clark 1984).

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 0.20 C/ha at this 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 = 0.47) and Patapsco River (C/ha = 0.44). 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 (~1.35 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 2018 (0.24 and 0.18 C/ha, respectively; Figure 1-9).

With the exception of one date in 2018 (May 19th – during a period of heavy flooding), conductivity measurements in mainstem Mattawoman Creek during 2008-2018 were always above the range observed during 1991 (Figure 1-10). Conductivity in Mattawoman Creek tributaries sampled during 2008-2018 often fell within the range observed during 1991 (Figure 1-10).

In 2018, conductivity measurements in mainstem Mattawoman Creek were elevated from March through May (≥ 138 μS/cm), with only one date (5/19/18) falling below the 1991 maximum at all sites (114 μS/cm; Figure 1-10). Conductivity measurements in tributary MUT3 in 2018 were below the 1991 maximum the entire time, and on one date (5/19/18) fell below the 1991 minimum (61 μS/cm; Figure 1-10). Conductivities in 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 2018, with a conductivity pattern similar to 2010-2013 and 2016-2017 (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 2008-2018 and median conductivities ranged from 1.14- to 2.4-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 0.16 to 0.44. 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 2008-2018 were at or below the Coastal Plain stream background criterion (109 μS/cm).

Herring spawning was detected at all mainstem stations sampled 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 2010-2018. Herring spawning was not detected at tributary site MUT3 during 2008-2010, but was consistently present during 2011-2016. Herring spawning was not detected in 2017 at MUT3, but was in 2018. 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, at MC2 during 2013-2014 and 2016-2017, and at MC3 during 1971 and 2016. Yellow Perch spawning in Mattawoman Creek has been detected at MC1 in all surveys conducted since 1971, with the exceptions of 2009 and 2012 (Table 1-5). Presence of White Perch and Yellow Perch spawning could not be determined in 2018 due to time and staffing limitations.

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 2015, 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 1976-1977 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 1976-1977, 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-TUC110 (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 2013-2018). 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). Patapsco River was characterized by spawning at level 2 (2013 and 2017) and 3 (2014-2016). 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).

Estimates of P_{herr} increased in Bush River, and Mattawoman and Piscataway creeks during 2005-2018 (Figure 1-13). The degree of increase appeared to reflect development status: P_{herr} in Mattawoman Creek (C/ha increasing from 0.87 to 0.93) approached levels exhibited in streams in rural watersheds, while P_{herr} in developed Bush River and Piscataway Creek watershed streams (C/ha increasing from 1.37 to 1.52 and 1.41 to 1.50, respectively) increased to a lesser extent (Figure 1-13). Remaining systems were sampled after 2011. Estimates of P_{herr} in Choptank River, and Deer and Tuckahoe creeks were high and steady through 2018, while estimates for Patapsco River were lower and more variable (Figure 1-13).

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, moderate, and positive ($r^2 = 0.38$, $P < 0.0001$, N = 34; Figure 1-14). Estimates of P_{herr} were linearly, moderately, and negatively related to C/ha ($r^2 = 0.53$, $P < 0.0001$, N = 35). Negative linear and curvilinear (power function) regressions similarly described the relationship of P_{herr} and standardized median conductivity ($r^2 = 0.21$, $P = 0.0061$; or approximate $r^2 = 0.19$, $P < 0.0001$, respectively), with linear regression explaining only slightly more variability (N = 34; Figure 1-15). Low estimates of P_{herr} (≤ 0.4) were much more frequent beyond the C/ha threshold (0.86 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 = 0.48) represented development below the C/ha threshold. The 1991 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).

Plots of residuals against year for linear regressions of C/ha or standardized conductivity and P_{herr} indicated an increasing trend (Figure 1-16); residuals were all negative prior to 2011 and nearly all positive afterwards for either model. Predictions based on these models were likely to be biased.

The C/ha and spawning stock time category multiple regression explained 74% of variation in P_{herr} ($P < 0.0001$; Table 1-13). The intercept (mean = 0.54, SE = 0.08) and both coefficients (C/ha slope = -0.30, SE = 0.06; spawning stock slope = 0.30, SE = 0.06) were estimated with reasonable precision (CV < 20%). Predicted P_{herr} declined by 53% over the range of observed C/ha (0.07-1.52; Figure 1-17). Predicted P_{herr} increased by 55% between the two spawning stock categories (Table 1-13).

The standardized conductivity and spawning stock time category multiple regression explained 69% of variation in P_{herr} ($P < 0.0001$; Table 1-14). The intercept (mean = 0.72, SE = 0.13) and both coefficients (standardized conductivity slope = -0.33, SE = 0.08; spawning stock coefficient = 0.41, SE = 0.06) were estimated with reasonable precision (CV < 23%). Predicted P_{herr} declined by 47% over the range of observed standardized conductivity (1.14-2.19; Figure 1-17). Predicted P_{herr} increased by 57% between the two spawning stock categories (Table 1-14).

An increasing trend in residuals, evident in the simple linear regressions of P_{herr} against C/ha or standardized conductivity, was eliminated (or nearly so) for the multiple regressions that added spawning stock size time category (Figure 1-18). Linear regressions of residuals from the multiple regressions and year in Figure 1-18 indicated a slight increasing trend over time was possible for standardized conductivity ($r^2 = 0.11$, $P < 0.06$) but unlikely for C/ha ($r^2 = 0.04$, $P = 0.27$). Cook's distance statistics identified 2011 as an outlier in both multiple regressions; the 2011 estimate of P_{herr} was more consistent with the high spawning stock (2012-2018) period than the low. This may have indicated some benefit by regulatory actions prior to the in-river fisheries deadline (2012; ASMFC 2019), including Atlantic coast bycatch reduction, or improved survival to maturity in response to declines in undescribed non-fishing related sources of at-sea losses (predation and feeding).

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 that accounted for shifting spawner abundance between 2005-2011 and 2012-2018, indicated significant and logical relationships among P_{herr}, C/ha, and conductivity consistent with the hypothesis that urbanization was detrimental to stream spawning. Herring spawning declined in streams as watersheds developed and conductivity increased. 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).

Maryland closed its River Herring fisheries in 2011, and most other in-river fisheries along the Atlantic Coast were closed by 2012 (AFMFC 2019). Caps on River Herring bycatch in Atlantic Herring and Atlantic Mackerel fisheries were also implemented in 2014 (MAFMC 2019), and estimates of P_{herr} in 2005-2018 increased concurrently with these reductions.

The 2017 ASMFC River Herring stock assessment update indicated that 16 stocks experienced increasing abundance, two experienced decreasing abundance, eight experienced stable abundance, and 10 experienced no discernable trend in abundance over the final 10 years of the times series (2006-2015; ASMFC 2019). Long-term monitoring of adult Blueback Herring and Alewife during spawning runs in the Nanticoke River, however, has not indicated an increase in recent years (Lipkey and Jarzynski 2015; Durell and Weedon 2018; McClair and Jarzynski 2018; ASMFC 2019).

Urbanization and physiographic province both affect discharge and sediment supply of streams (Paul and Meyer 2001; Cleaves 2003). These, in turn, could affect location, substrate composition, and extent and success of spawning. 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.

Processes such as flooding, riverbank erosion, and landslides vary by geographic province (Cleaves 2003) and influence physical characteristics of anadromous fish spawning streams. Coastal Plain streams have slow flows and sand or gravel bottoms (Boward et al. 1999). Unconsolidated layers of sand, silt, and clay underlie the Coastal Plain, with broad plains of low relief and wetlands characterizing the natural terrain (Cleaves 2003). Most Piedmont streams are of moderate slope with rock or bedrock bottoms (Boward et al. 1999), and the region is underlain by metamorphic rocks and characterized by narrow valleys and steep slopes, with regions of higher land between streams in the same drainage. The Piedmont 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.

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), and these can be impacted by development. Strong impacts of urbanization on lithophilic spawners include 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. While organic matter may be positively impacted by nutrients, it can also be negatively impacted by fine sediment from agriculture (Piggot et al. 2015).

Elevated conductivity, related primarily to chloride from road salt (although it includes 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, as well as 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 disrupt 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.

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). The MD DOP forest cover estimate mixes 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. Estimates of P_{herr} were consistently high in the three watersheds dominated by agriculture, while importance of forest cover could not be assessed with confidence since it was possible that forest cover estimates included residential tree cover.

An unavoidable assumption of regression analyses of P_{herr}, C/ha, and standardized 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. Squared type II partial correlation coefficients for regressions of C/ha with P_{herr} were higher (0.47; Table 1-13) than for standardized conductivity (0.38; Table 1-14).

Liess et al. (2016) developed a stress addition model for meta-analysis of toxicants, combined with 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. This concept may also serve as a caveat for expectations of overcoming Herring habitat deterioration, due to development, through stringent management of directed fisheries and bycatch. An underlying negative relationship of P_{herr} with C/ha was present, but only describes how the abundance of earliest live stages of Herring may be impacted. Increasingly frequent poor juvenile indices of Blueback Herring and Alewife in the urbanizing Patuxent River after the late 1990s do not indicate that increased spawning stock (assuming the trend seen in studied systems occurred there as well) has overcome deterioration of habitat (Uphoff et al. 2018).

Based on a simple plot and linear regression of C/ha and P_{herr}, it appeared that spawning both declined and became more variable as development increased. However, increasing variability likely was an artifact of increasing spawning stock size with time. Once a time category term that accounted for changing spawner abundance was added to the P_{herr} and C/ha regression, the variability about the predicted slopes was reduced considerably.

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. 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.

Volunteer-based sampling of stream spawning during 2005-2018 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).

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, Department of Planning (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	36,009	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	1.52	18	29.9		
Choptank	2016	2010	0.18	55	27.8	38,285	Agriculture
Choptank	2017	2010	0.18	55	27.8		
Deer	2012	2010	0.24	44.6	28.4	37,724	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	0.48	13.8	62.6	24,430	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		
Mattawoman	2018	2010	0.93	9.3	53.9		
Patapsco	2013	2010	1.11	24.4	30.4	93,730	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	17,634	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	39,364	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
Mattawoman	2018	5	11-Mar	19-May	11	55
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	2017	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 2005-2018. 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	2018
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	2018
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	160.7
Standard Error				3.8	19.2	38	2.8	1.9	2.4	9.1	6.5	4.1	3.7	4.4
Median				124.6	211	152.3	147.3	130.9	126.5	165.8	172.5	188.8	150.2	165.5
Kurtosis				2.1	1.41	1.3	8.29	-0.26	5.01	0.33	1.49	-0.80	-0.55	2.99
Skewness				-1.41	1.37	0.03	1.72	-0.67	-1.70	1.00	1.33	-0.68	-0.36	-1.70
Range				102	495	111	117	49	96	261	185	93	102	120
Minimum				47	115	99	109	102	63	88	130	121	91	79
Maximum				148	610	210	225	151	158	350	315	214	193	198
Count				39	40	43	44	44	48	48	44	44	52	44
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.

	Year													
Conductivity	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018
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 2008-2018. 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	2018
	Herring														
MC1	1	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	1
MC3	1			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	1
MUT3	1				0	0	0	1	1	1	1	1	1	0	1
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	0	1	0
TUC110	0	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	0
TUC110	0	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	0
TUC110	0	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	1.58284	1.58284	20	<.0001	
Residual	32	2.53261	0.07914			
Total	33	4.11545				
$r^2 = 0.3846$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.1584	0.10743	10.78	<.0001	0.93958	1.37723
C / ha	0.46259	0.10344	4.47	<.0001	0.25189	0.67328

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Structure density (C/ha)					
ANOVA	df	SS	MS	F	P	
Regression	1	1.30391	1.30391	36.8	<.0001	
Residual	33	1.16921	0.03543			
Total	34	2.47312				
$r^2 = 0.5272$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.86055	0.07015	12.27	<.0001	0.71783	1.00326
C / ha	-0.41443	0.06831	-6.07	<.0001	-0.55342	-0.27544

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Standardized conductivity					
ANOVA	df	SS	MS	F	P	
Regression	1	0.51608	0.51608	8.64	0.0061	
Residual	32	1.9122	0.05976			
Total	33	2.42827				
$r^2 = 0.2125$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.03736	0.19585	5.3	<.0001	0.63843	1.4363
Standardized conductivity	-0.35412	0.1205	-2.94	0.0061	-0.59957	-0.10867

Table 1-13. Summary statistics of the multiple regression model for development level (C/ha) and spawning stock time category versus proportion of samples with Herring eggs and-or larvae (P_{herr}).

ANOVA			Multiple Regression			
Source	df	SS	MS	F	P	
Regression	2	1.79803	0.89902	44.22	<.0001	
Residual	31	0.63024	0.02033			
Total	33	2.42827				
$r^2 = 0.7405$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.54261	0.08201	6.62	<.0001	.	.
C / ha	-0.30026	0.0568	-5.29	<.0001	0.51885	0.47409
Time category	0.29914	0.05814	5.14	<.0001	0.46058	0.46058

Table 1-14. Summary statistics of the multiple regression model for standardized conductivity (annual median/province background) and spawning stock time category versus proportion of samples with Herring eggs and-or larvae (P_{herr}).

ANOVA			Multiple Regression			
Source	df	SS	MS	F	P	
Regression	2	1.68589	0.84295	35.2	<.0001	
Residual	31	0.74238	0.02395			
Total	33	2.42827				
$r^2 = 0.6943$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.71645	0.13221	5.42	<.0001	.	.
Standardized conductivity	-0.33312	0.07634	-4.36	0.0001	0.21253	0.38051
Time category	0.40741	0.05829	6.99	<.0001	0.61177	0.61177

Figure 1-1. Watersheds sampled for stream spawning anadromous fish eggs and larvae during 2005-2018. Coastal Plain and Piedmont Regions are indicated.

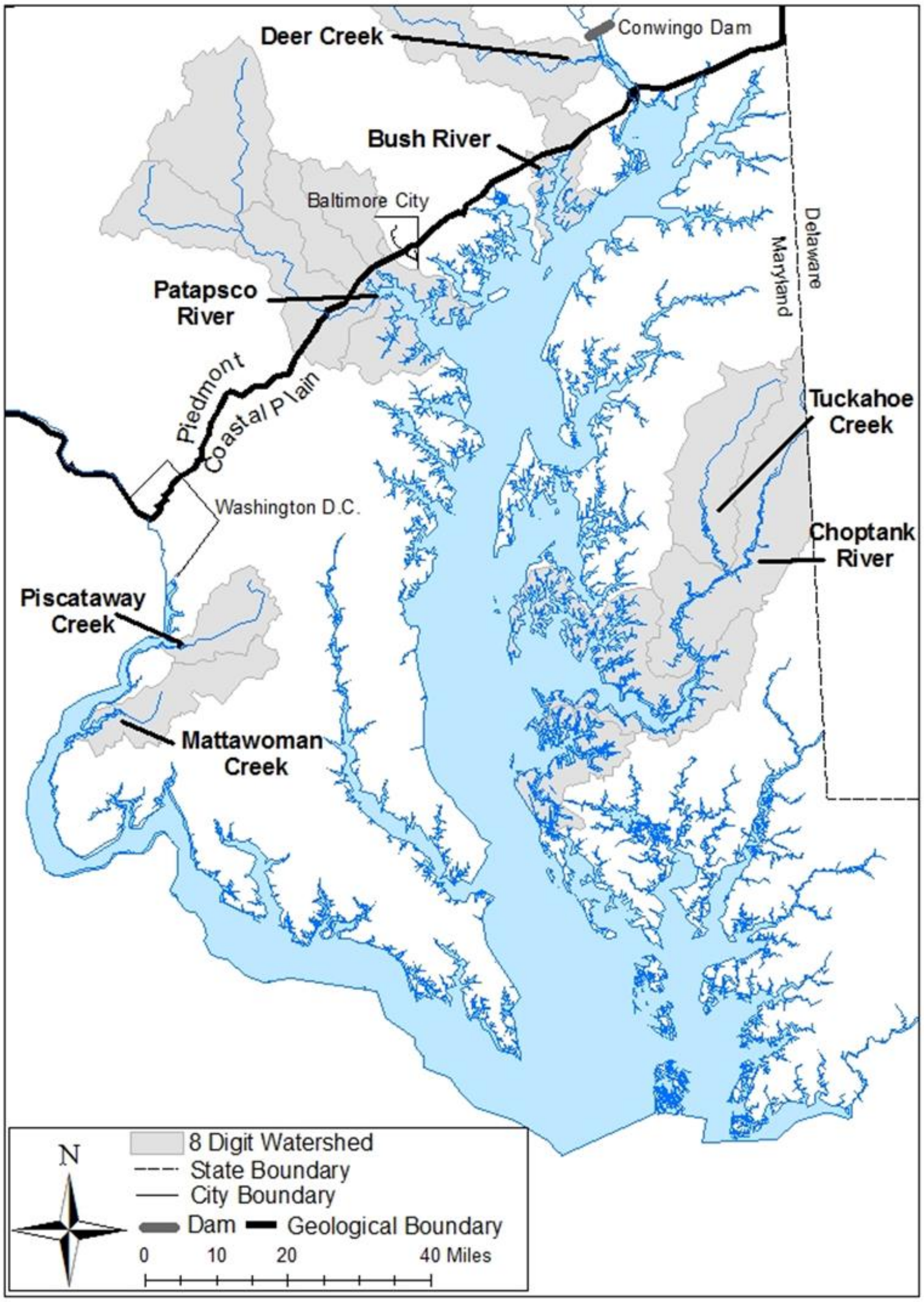


Figure 1-2. Mattawoman Creek's 1971 and 2008-2018 sampling stations. Bar approximates lower limit of development associated with the town of Waldorf.

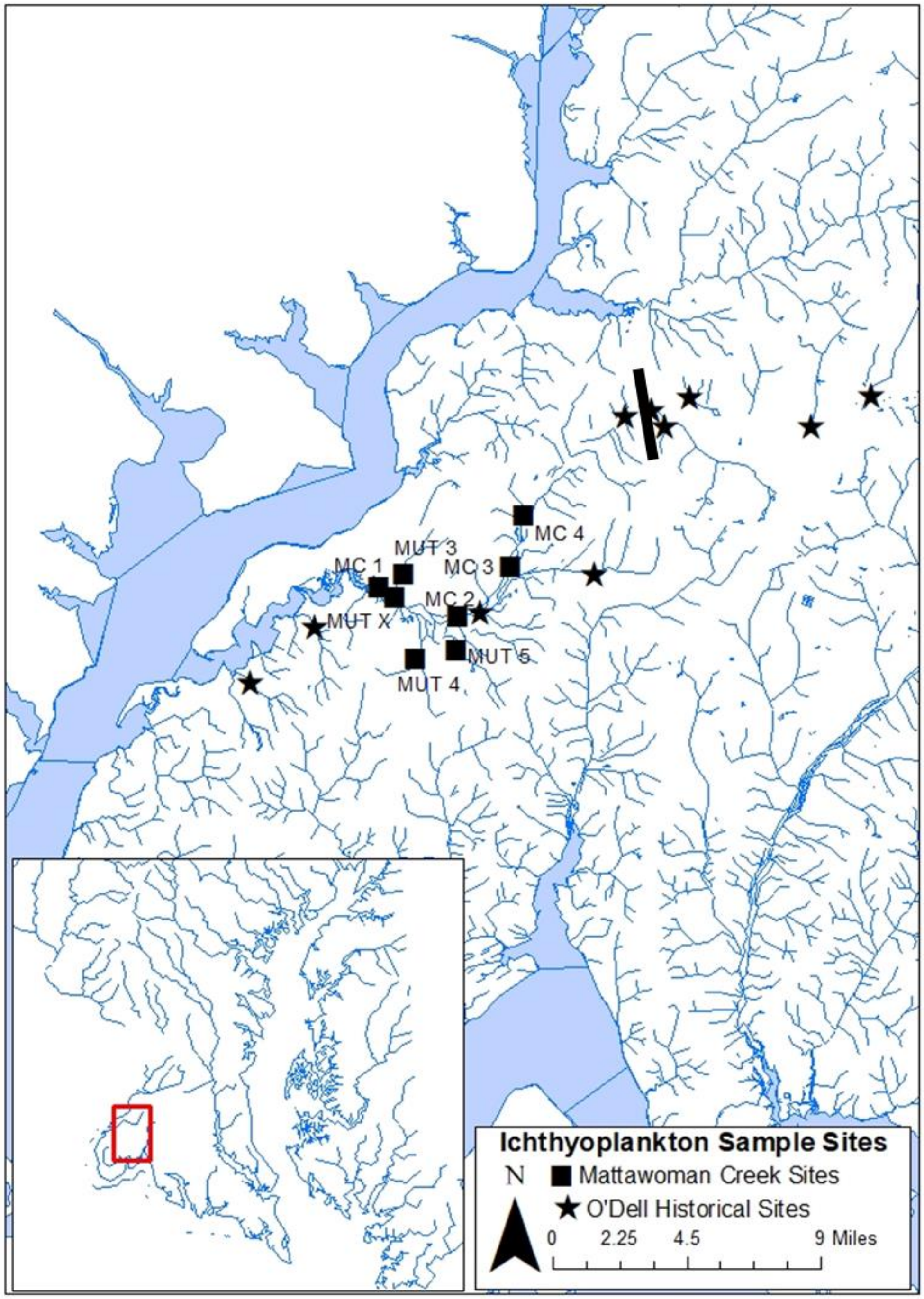


Figure 1-3. Piscataway Creek's 1971, 2008-2009, and 2012-2014 sampling stations.

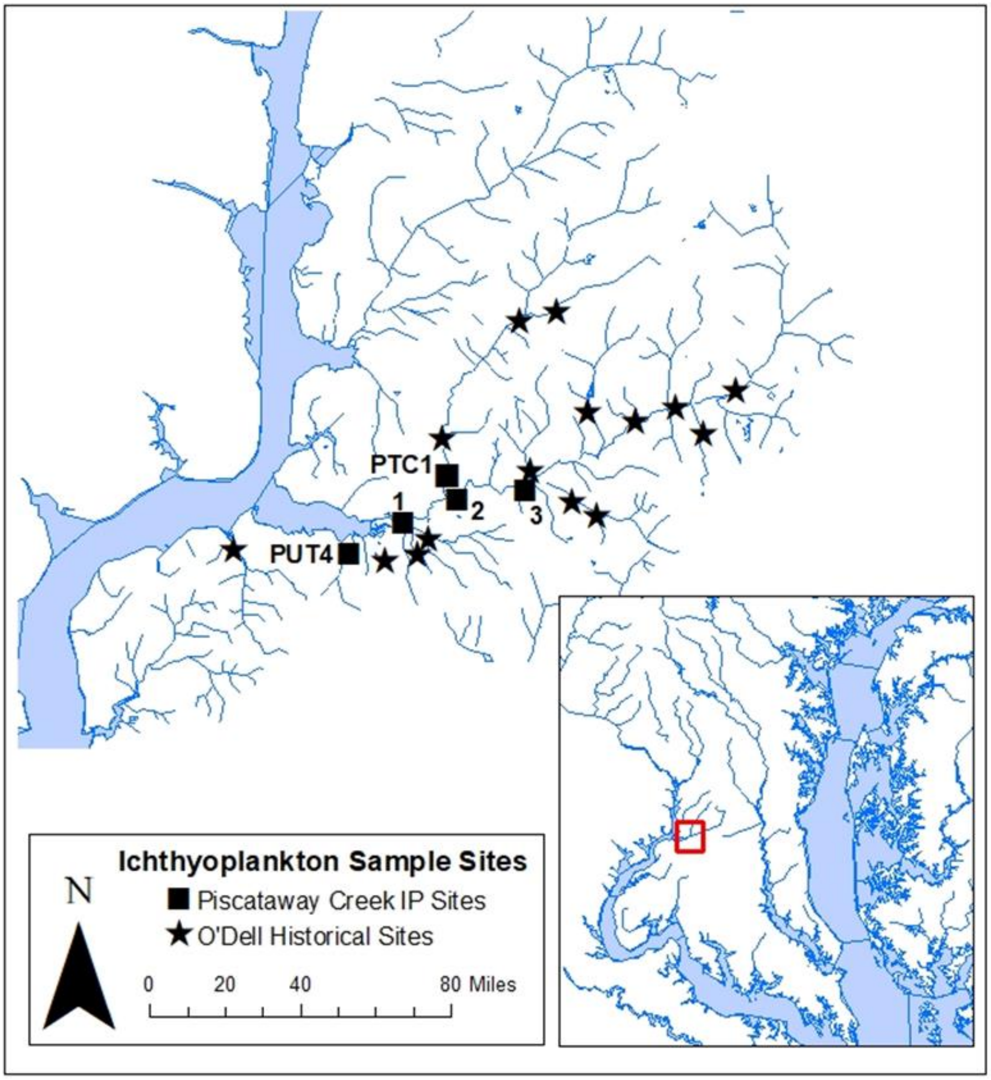


Figure 1-4. Bush River's 1973, 2005-2008, and 2014 sampling stations. Stations in Aberdeen Proving Grounds (APG) have been separated from other Bush River stations. Line delineates APG streams that were excluded.

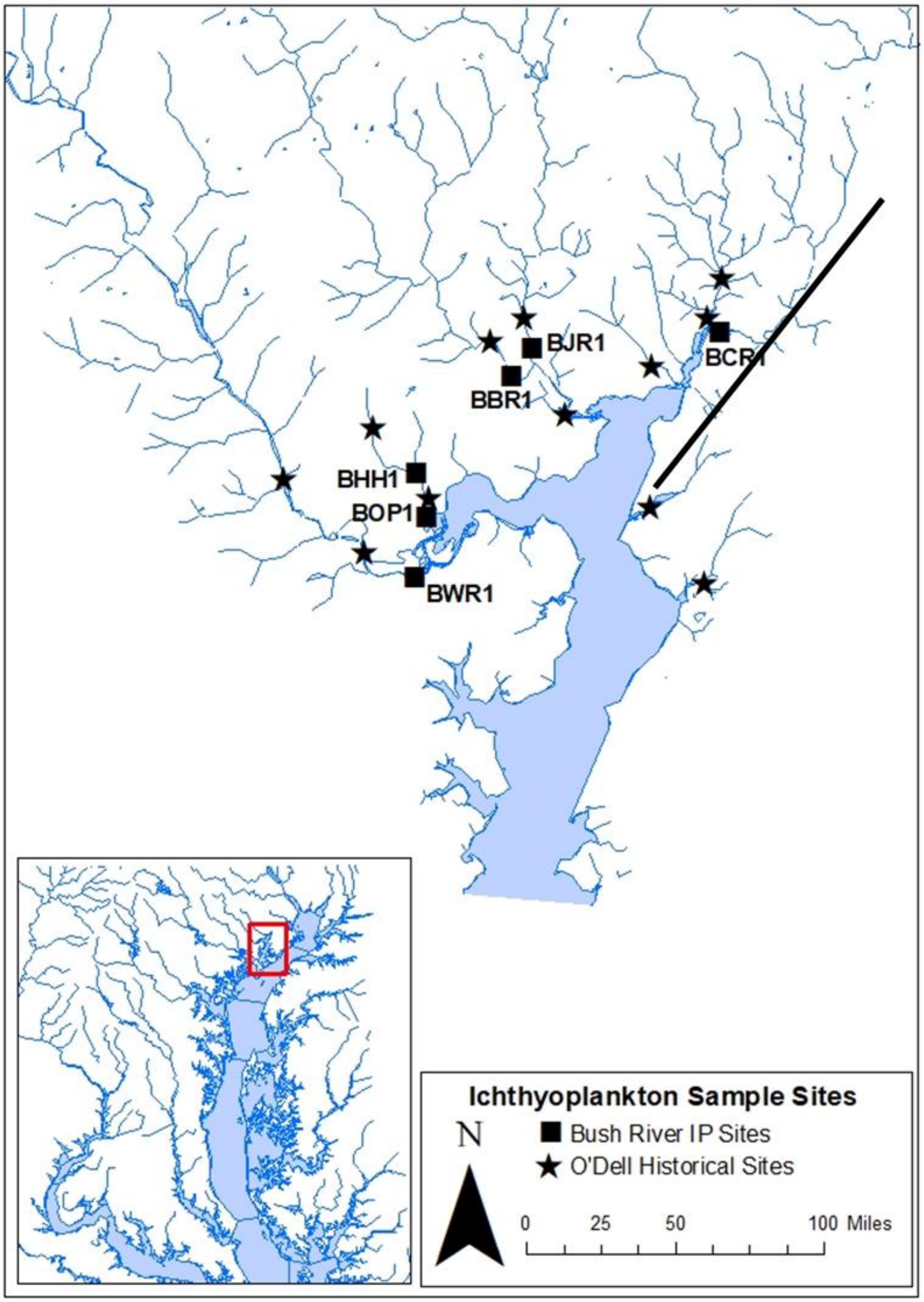
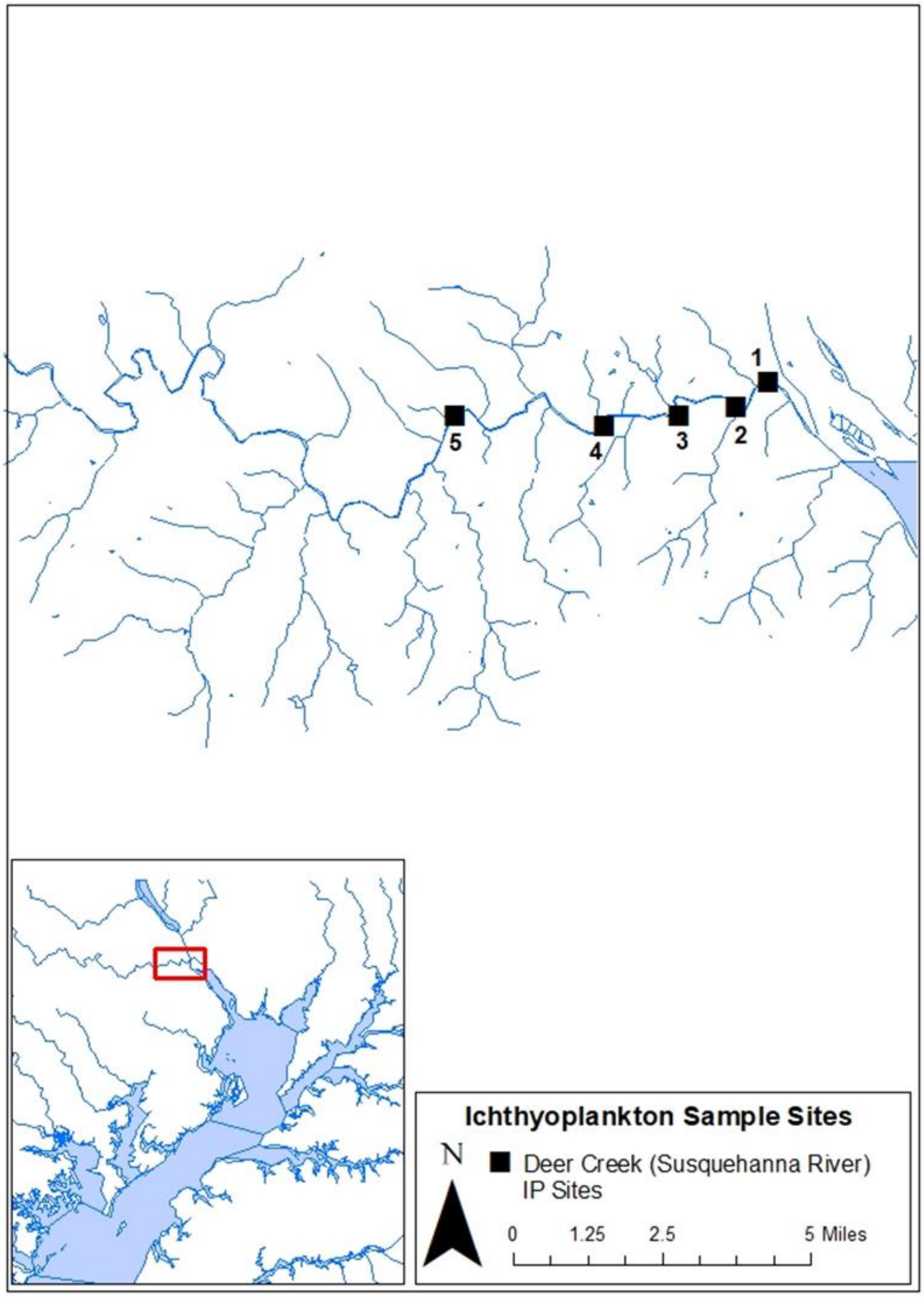


Figure 1-5. Deer Creek's 1972 and 2012-2015 sampling stations.



Figures 1-6 and 1-7. Choptank River and Tuckahoe Creek's 2016-2017 sampling stations.

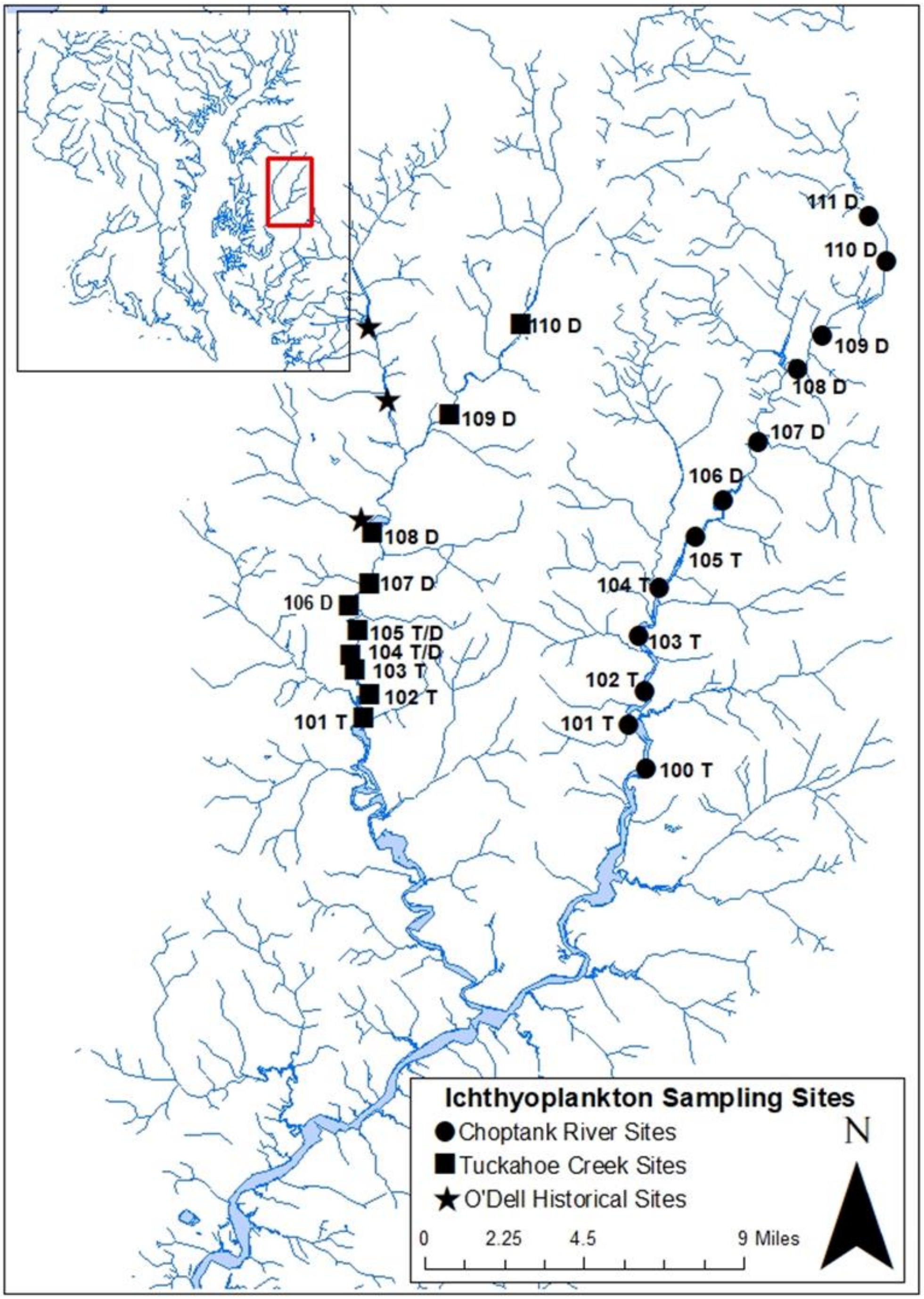


Figure 1-8. Patapsco River's 1973 and 2013-2017 sampling stations.

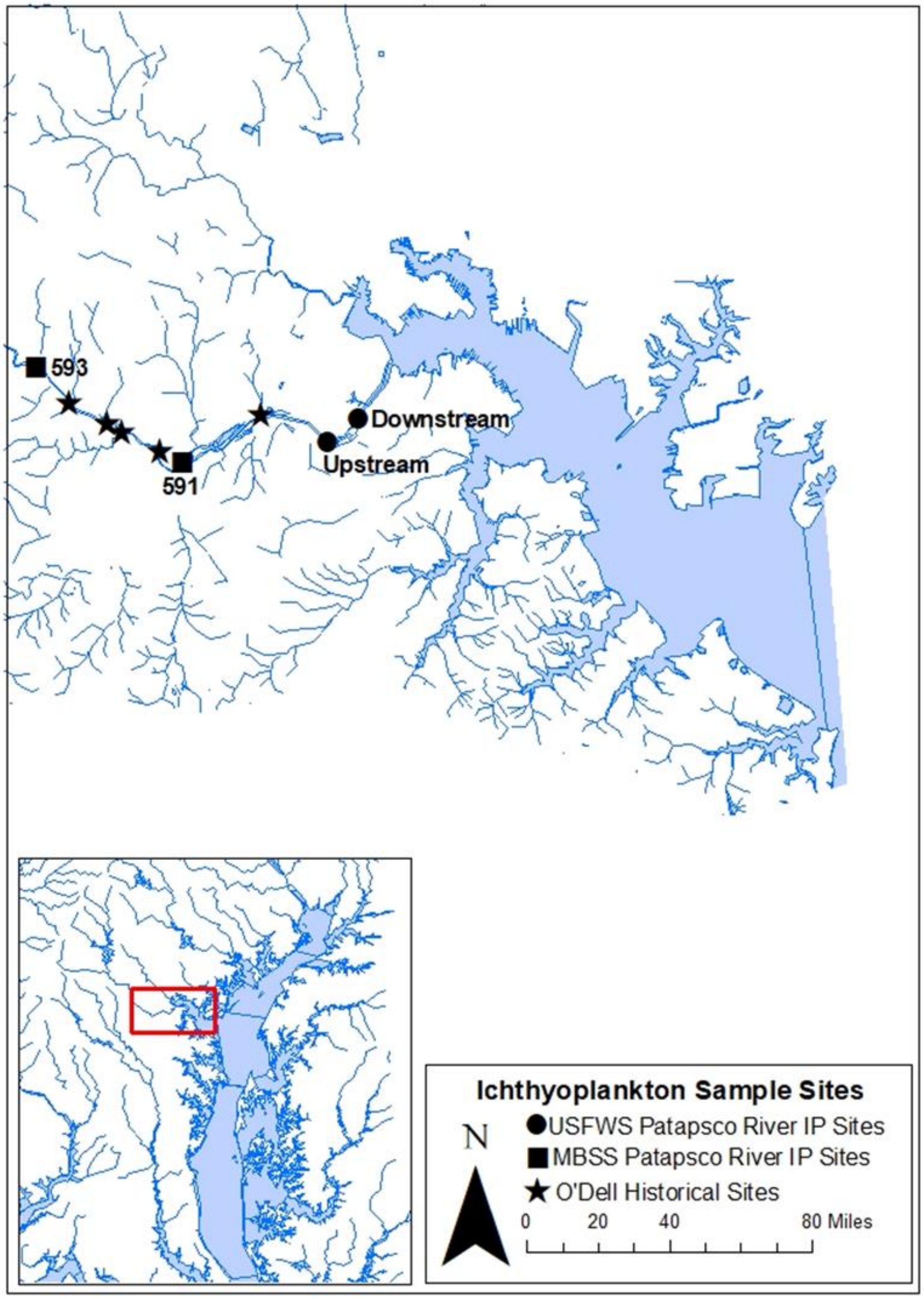


Figure 1-9. Trends in counts of structures per hectare (C/ha) during 1950-2018 in Deer, Mattawoman, and Piscataway Creeks, the Bush and Patapsco Rivers, and the Choptank River drainage watersheds. Estimates of C/ha were only available to 2014 or 2016, depending on Department of Planning data updates. Large symbols indicate years when stream ichthyoplankton was sampled.

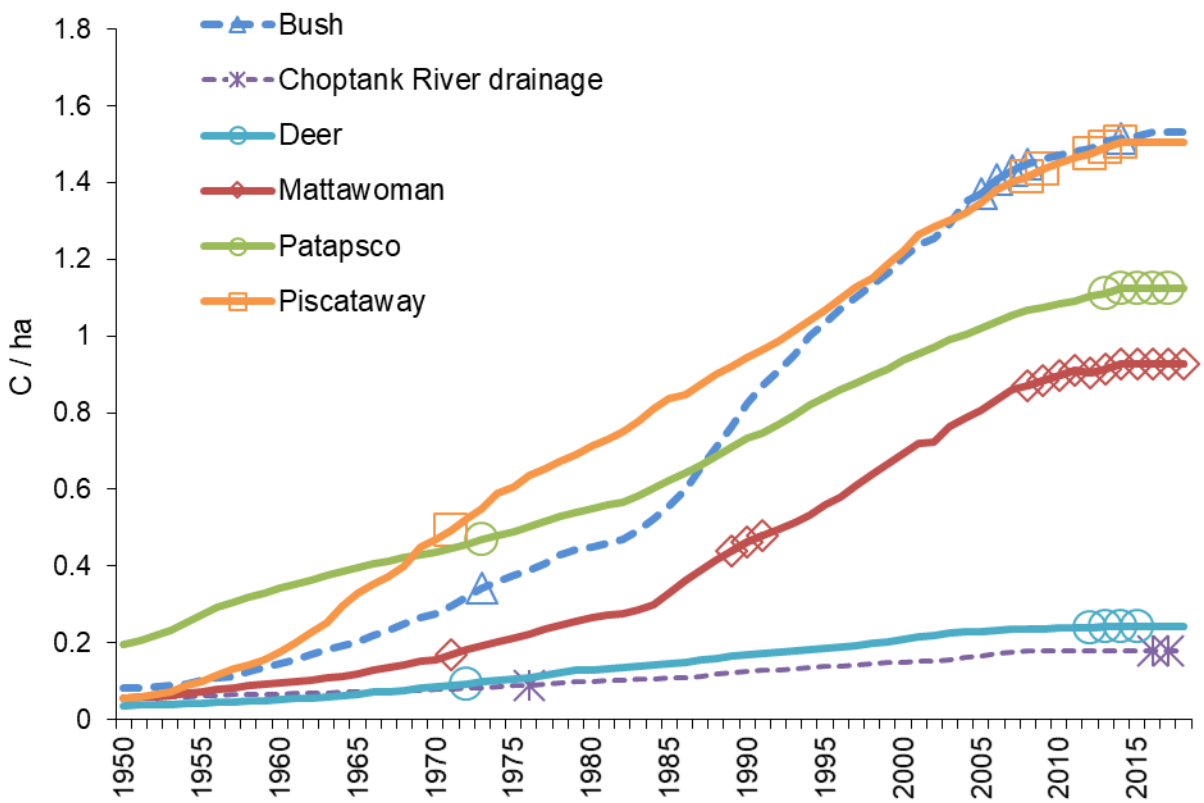


Figure 1-10. Stream conductivity measurements (µS/cm), by station and date, in Mattawoman Creek during (A) 2009, (B) 2010, (C) 2011, (D) 2012, (E) 2013, (F) 2014, (G) 2015, (H) 2016, (I) 2017 and (J) 2018. Lines indicate conductivity range measured at mainstem sites (MC1 – MC4) during 1991 by Hall et al. (1992). Note changes in axis scale among years.

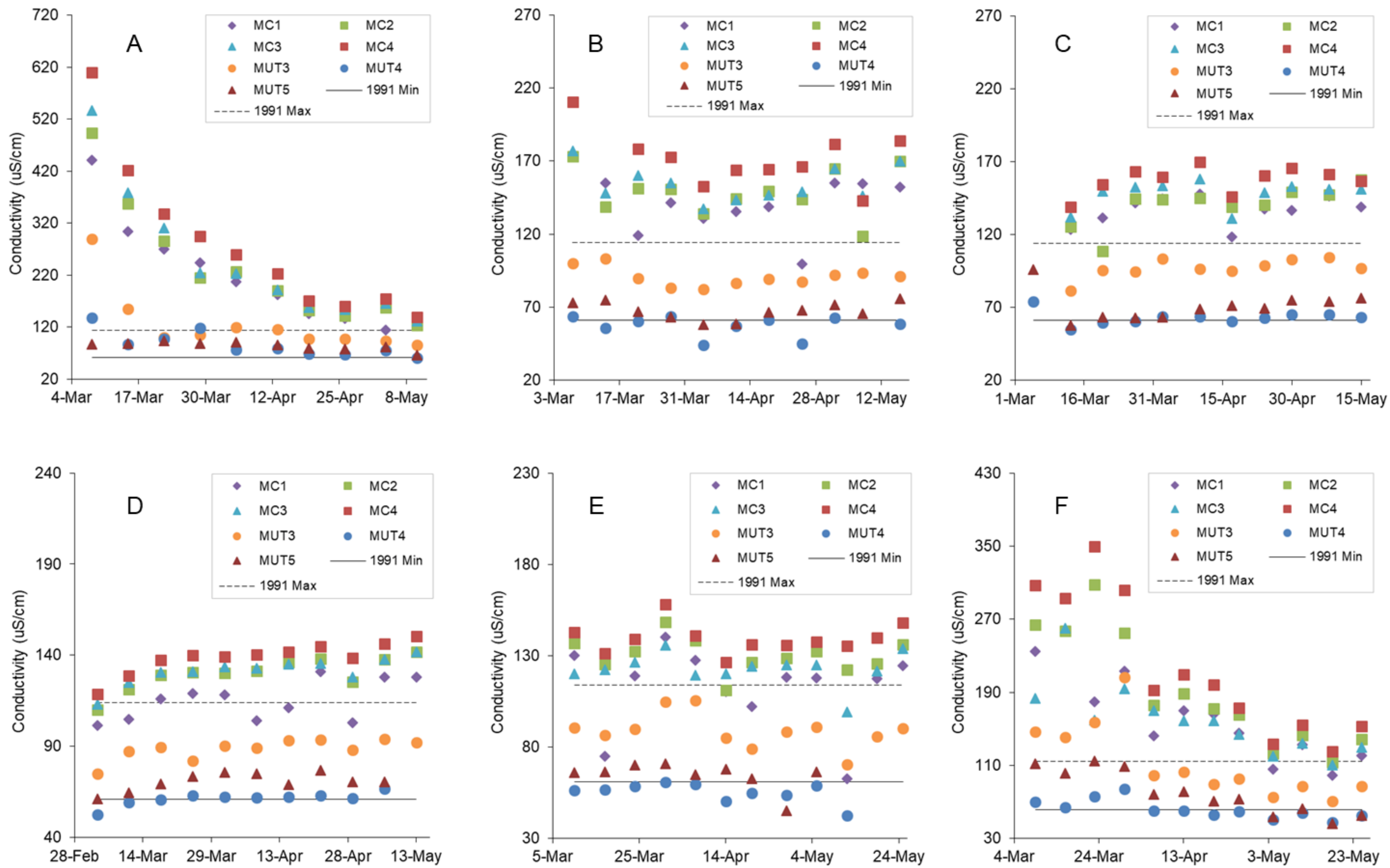


Figure 1-10 cont.

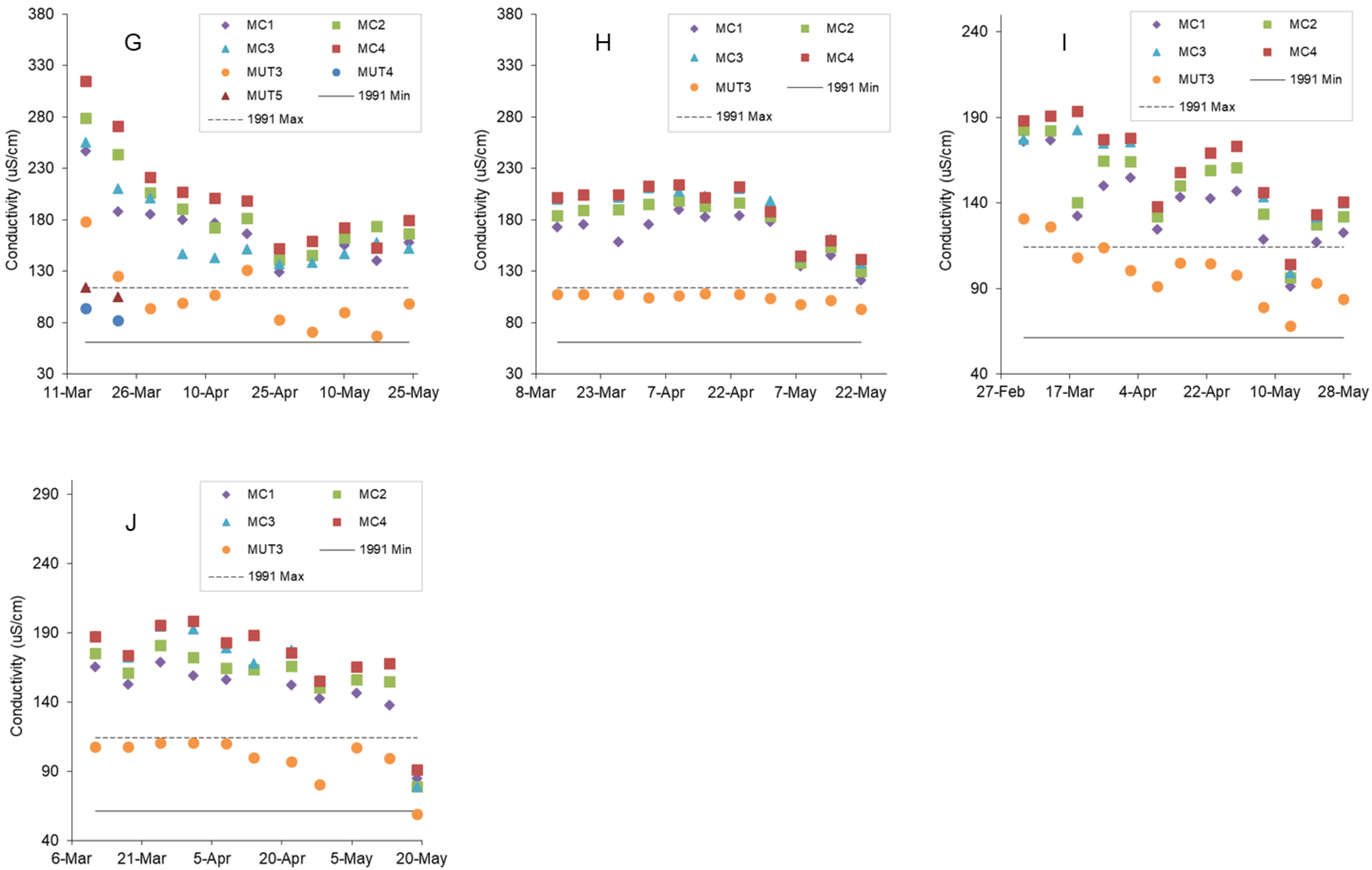


Figure 1-11. Historical (1970-1989) median conductivity measurements and current (2008-2018) anadromous spawning survey median conductivity in non-tidal Mattawoman Creek (between the junction with the subestuary and Waldorf) plotted against distance from the mouth. The two stations furthest upstream are nearest Waldorf. Median conductivity was measured during March-May, 2008-2018, and varying time periods (see Table 1-2) during 1970-1989.

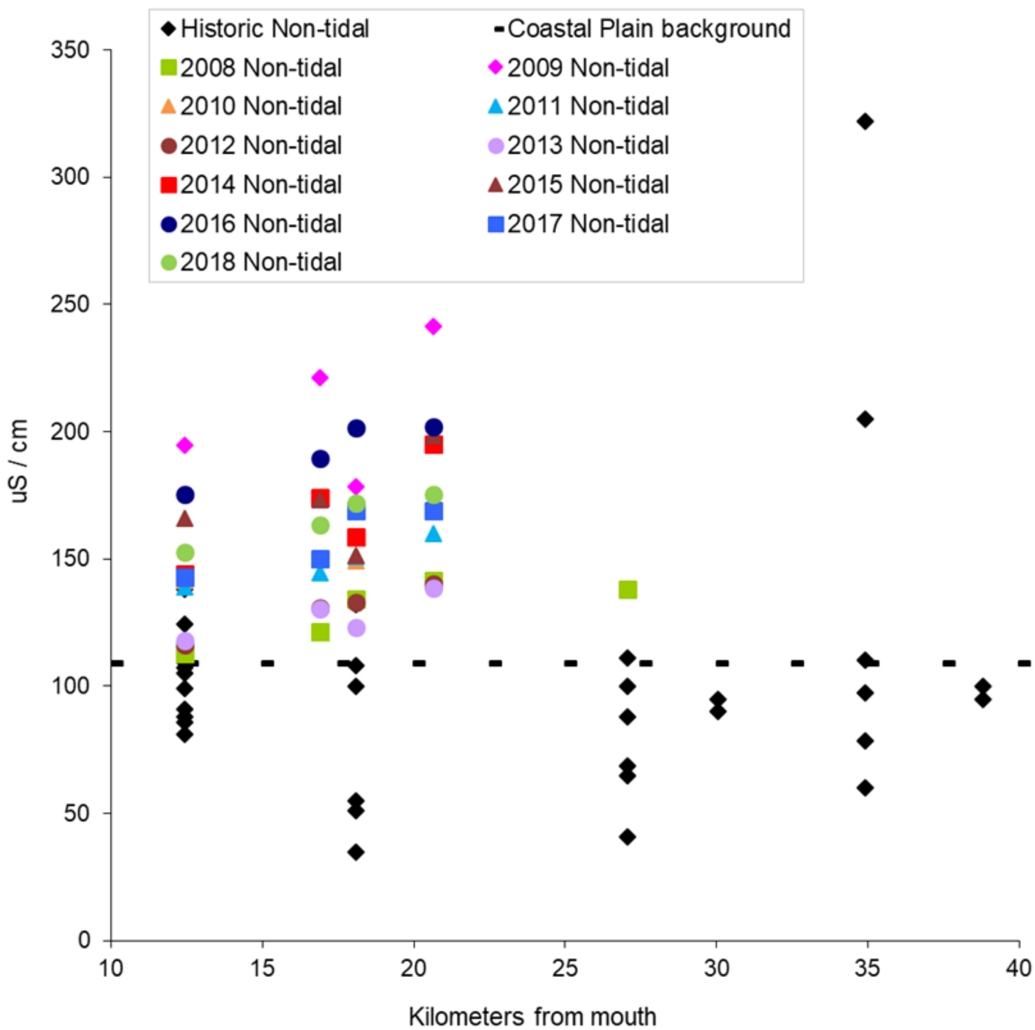


Figure 1-12. Stream conductivity measurements (μS/cm), by station and date, in Mattawoman Creek during (A) 2009, (B) 2010, (C) 2011, (D) 2012, (E) 2013, (F) 2014, (G) 2015, (H) 2016, (I) 2017 and (J) 2018. Lines indicate conductivity range measured at mainstem sites (MC1 – MC4) during 1991 by Hall et al. (1992). Note changes in axis scale among years.

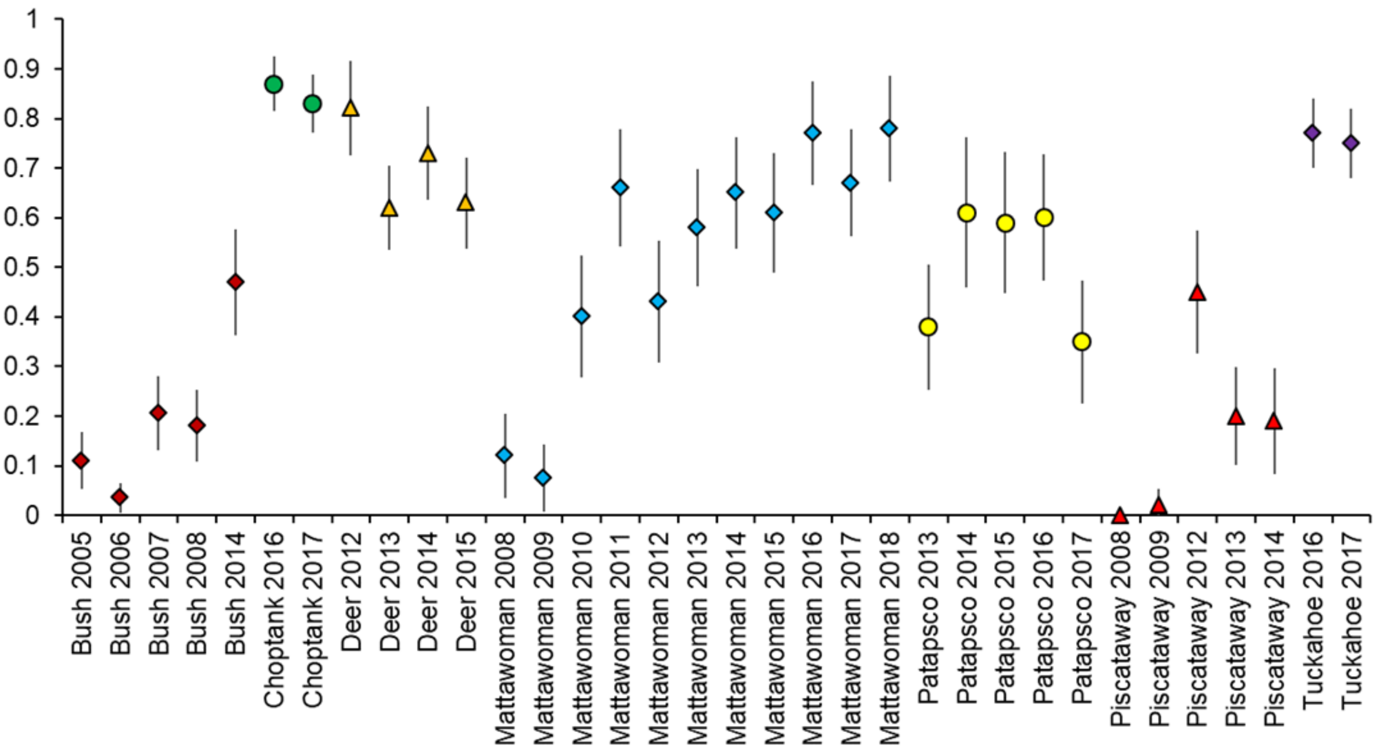


Figure 1-13. Trends in P_{herr} (proportion of stream samples with Herring eggs and-or larvae) by watershed. Watersheds sampled in both early (2005-2011) and late (2012-2018) spawning periods are indicated by large triangles.

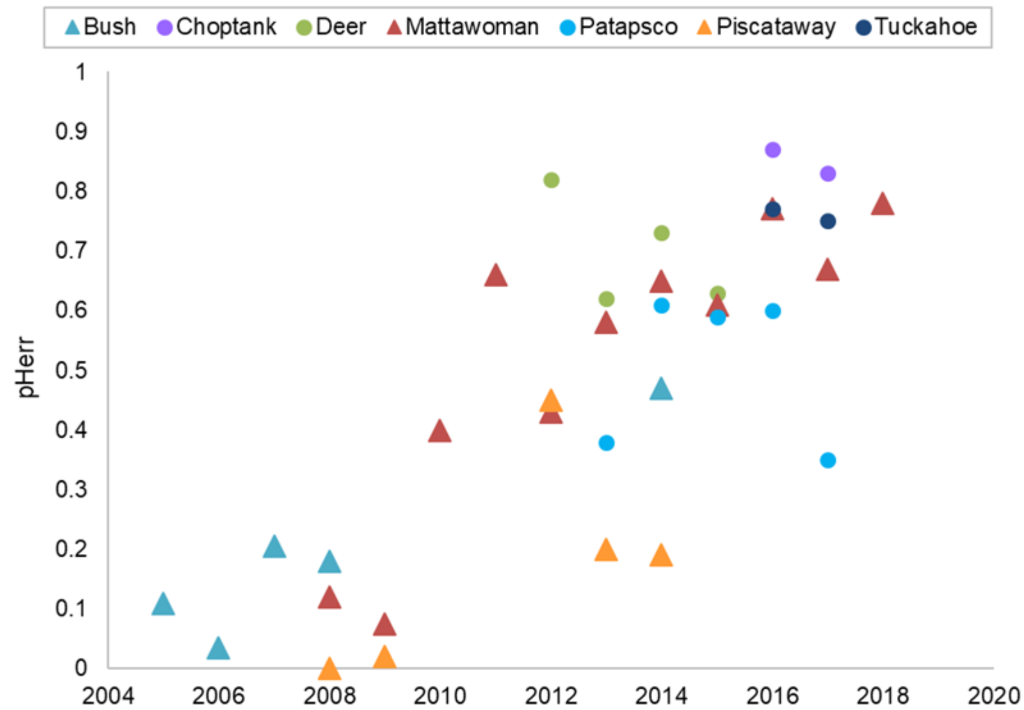


Figure 1-14. Standardized median conductivity during spring spawning surveys and level of development (C/ha). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

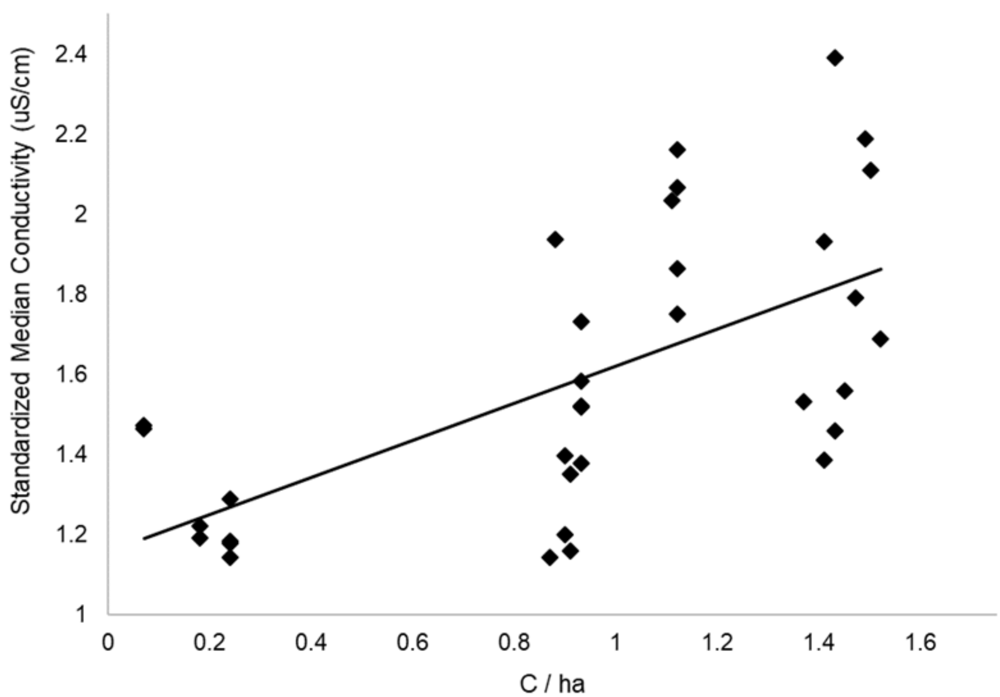


Figure 1-15. (A) Proportion of stream samples with Herring eggs and-or larvae (P_{herr}) and level of development (C/ha) with Department of Planning land use designations. (B) P_{herr} and standardized median spawning survey conductivity (uS/cm). Median conductivity was standardized to background estimates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

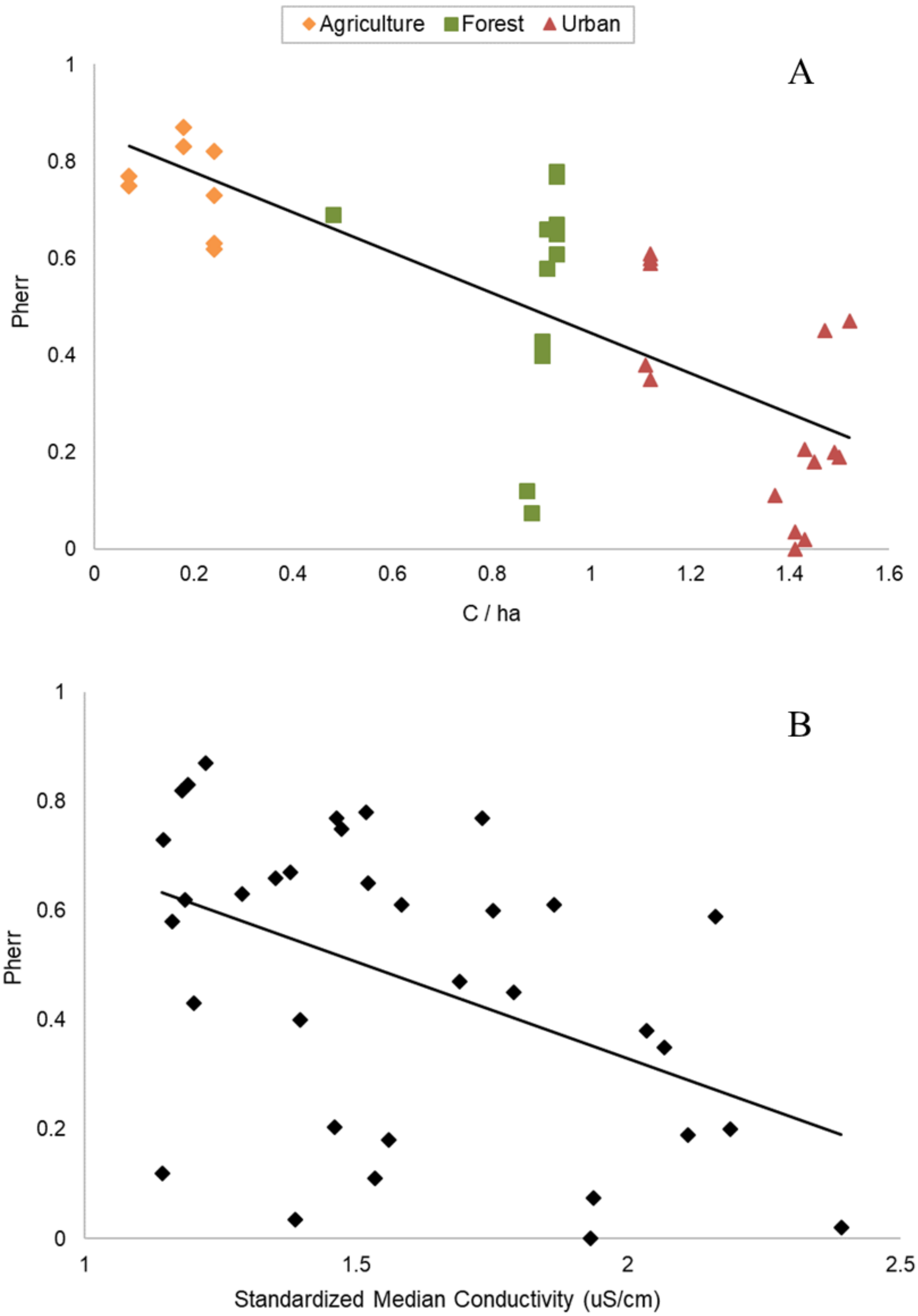


Figure 1-16. Serial patterns of residuals versus year for regressions of P_{herr} (proportion of stream samples with Herring eggs and-or larvae) and (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

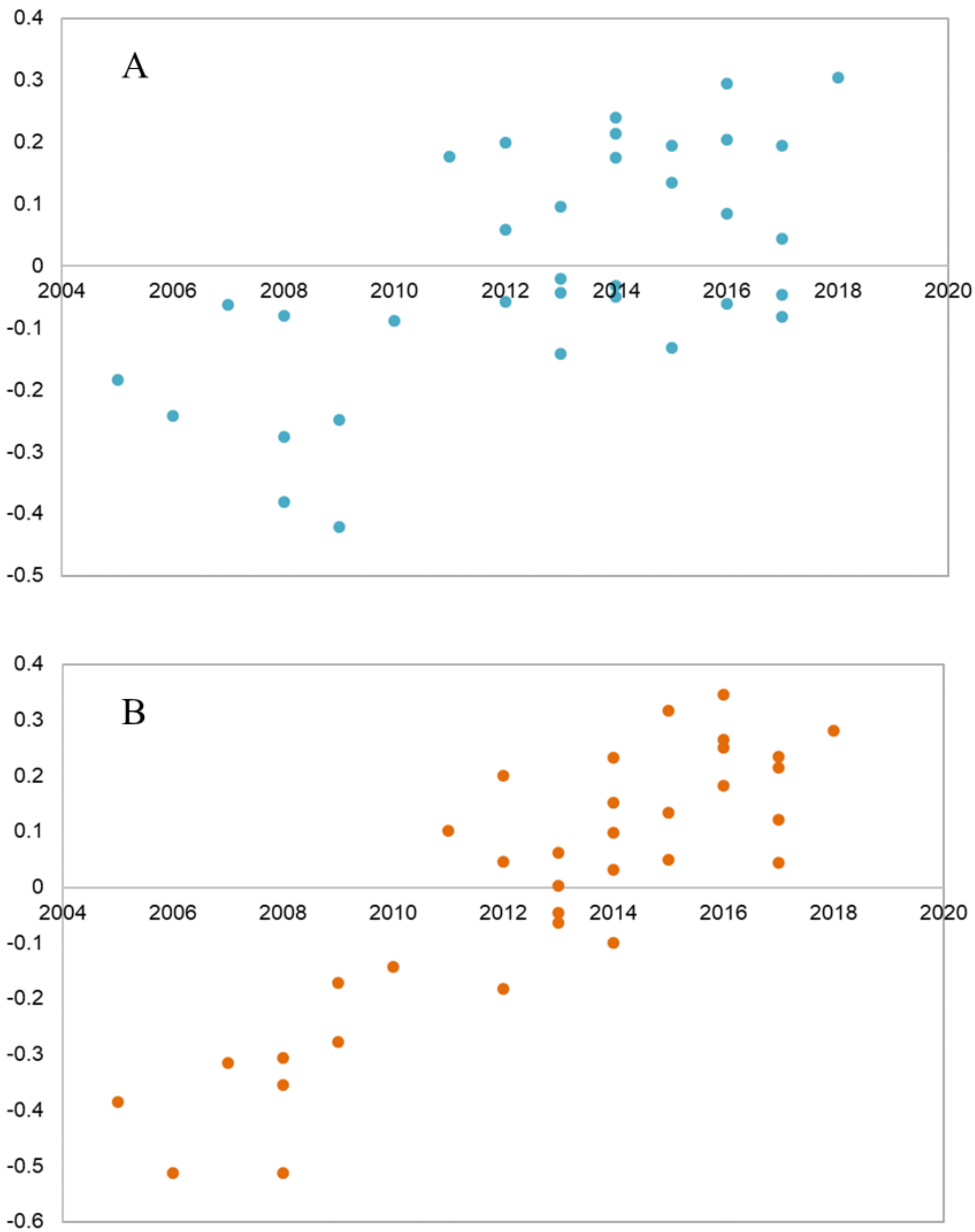
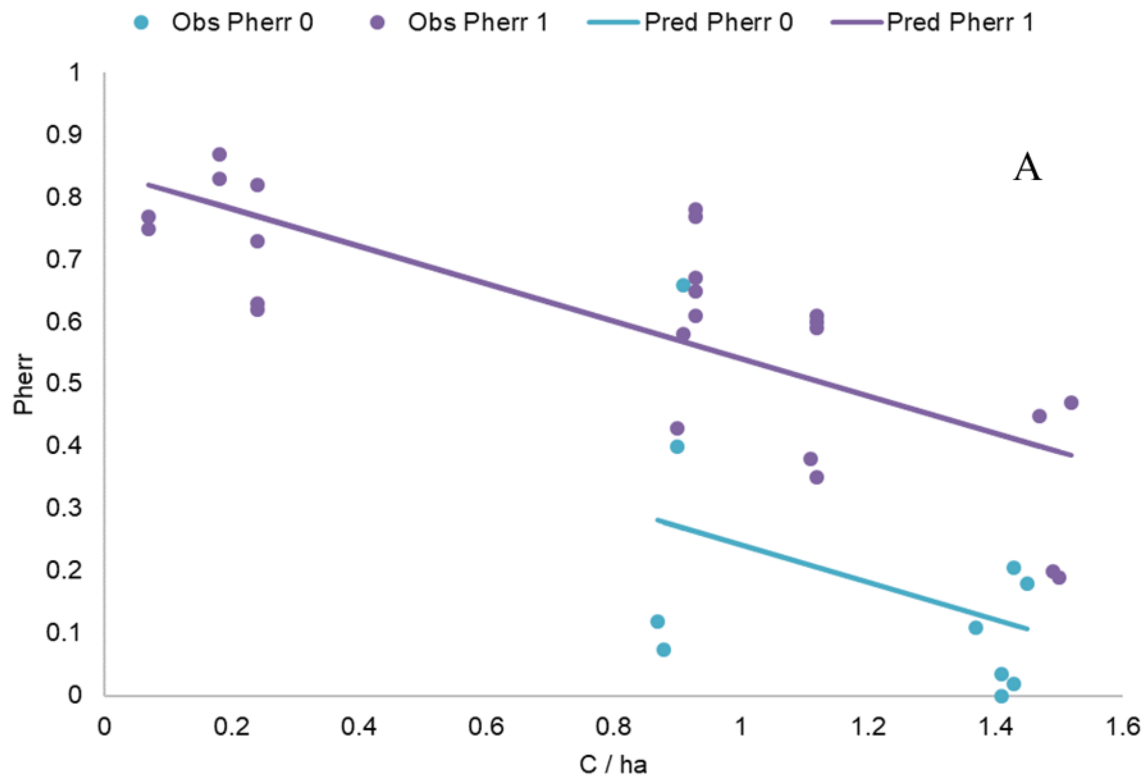


Figure 1-17. Plots of regressions of P_{herr} (proportion of stream samples with Herring eggs and-or larvae) against (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm) with spawning stock time categories (0 = 2005-2011; 1 = 2012-2018) included. Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).



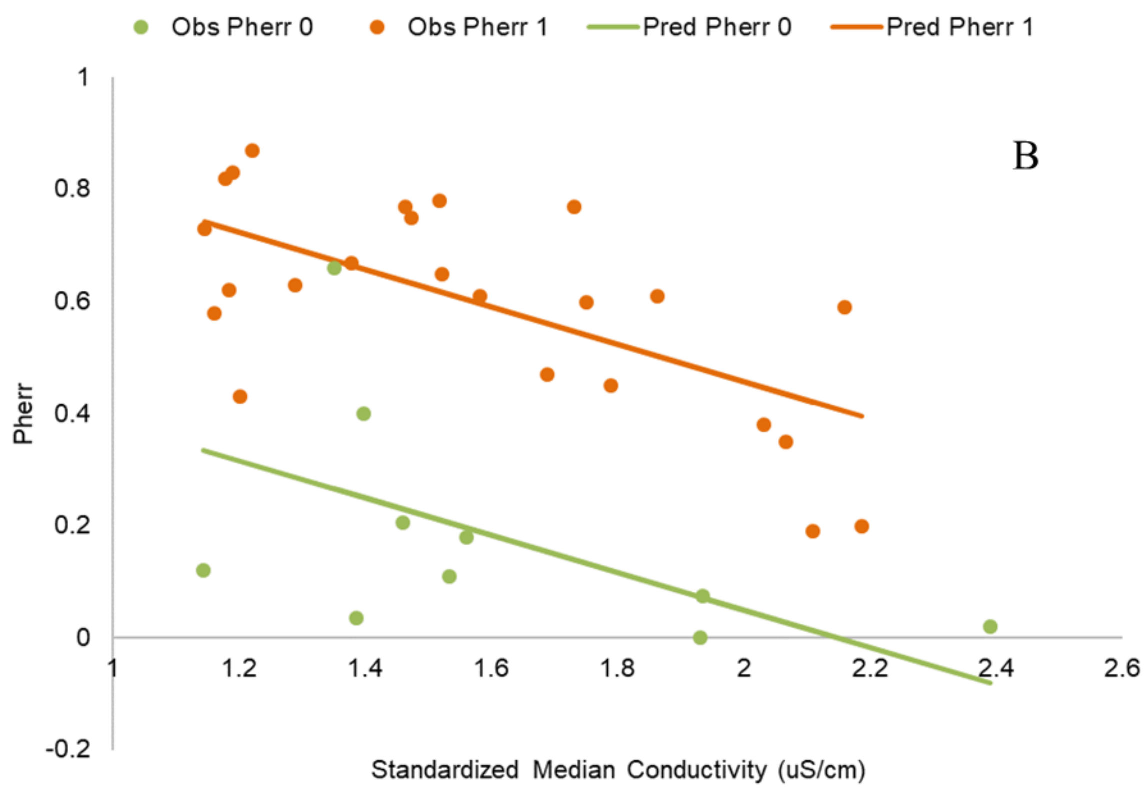
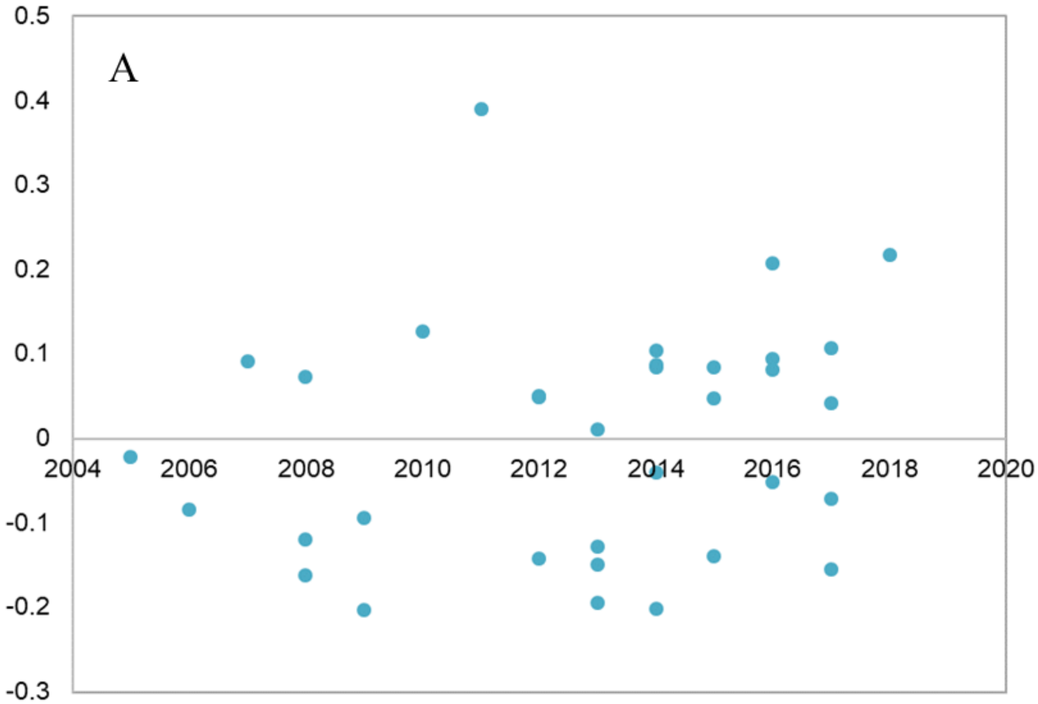
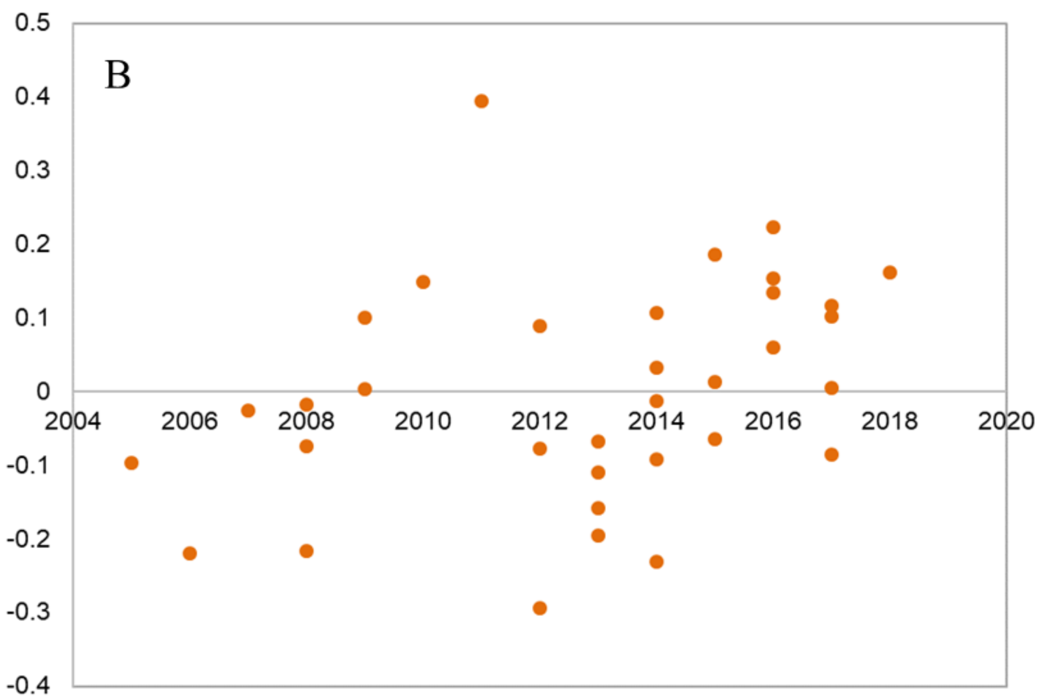


Figure 1-18. Residuals versus year for multiple regressions of spawning stock size time category and (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm) against proportion of stream samples with Herring eggs and-or larvae (P_{herr}). Median conductivity was 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 Marcus Patton

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 2018 was conducted in the upper tidal reaches of the Choptank, Nanticoke, and Wicomico (eastern shore; ES hereafter; 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 2018 we used regression analyses to examine 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.

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, pH, and salinity were measured at each site on each sample date.

Ten sites were sampled twice weekly in the Choptank and Wicomico (ES) Rivers, and once weekly in the Nanticoke River (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 2018 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 areas 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 2018, 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, or early in the second week, of April 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 or 2016, whichever the most recent data was available for (M. Topolski, MD DNR, personal communication). Estimates of C/ha for 2014 or 2016 were used to represent that year forward 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).

Linear regression was used to evaluate time trends in L_p in two large subestuaries with extended time-series: Choptank River (1986-2018; N = 18) and Nanticoke River (1965-2018; N = 19). Neither time-series was continuous; Choptank River was sampled during 1986-2004 and 2013-2018, while the Nanticoke River estimates were available for 1965, 1967, 1968, 1970, 1971, 2004-2009, and 2011-2018.

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 2011-2018. 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 2018, sampling on Choptank River began on March 27 and lasted until May 3. Sampling on Wicomico (ES) River began on March 26 and concluded on May 2. Samples through May 1 and April 25 were used to estimate L_p in Choptank and Wicomico (ES) Rivers, respectively. Sampling began on April 2 in the Nanticoke River and ended on April 30, with samples from April 10 on used for estimating L_p.

Based on 95% CIs, estimates of L_p during 2018 overlapped the brackish subestuary threshold (0.40) in the Choptank, Wicomico, and Nanticoke rivers (Figure 2-2). Estimate of mean L_p for the Choptank River (L_p = 0.44)) was above the brackish threshold, while mean L_p was below this threshold in the Nanticoke and Wicomico (ES) rivers r (L_p = 0.28 and 0.38, respectively; Figure 2-2).

Comparisons of L_p during 2018 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 0.46-3.33 for tidal-fresh (Table 2-1). Strong relationships of L_p with year were not evident in the Choptank River or Nanticoke River. Estimate of L_p in Choptank River during 1986-2018 exhibited little indication of decline (r^2 = 0.01; P = 0.64), while a marginal decline of L_p was detected during 1965-2018 in the Nanticoke River (r^2 = 0.16; P = 0.08; Figure 2-4). Both of these subestuaries are rural and land use is dominated by agriculture.

Separate linear regressions of C/ha and L_p by salinity category were significant at P ≤ 0.0004; 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 24% 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 = 0.57, 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 ($r^2 = 0.31$; Table 2-2) as separate regressions for each salinity type. Intercepts of tidal-fresh and brackish subestuaries equaled 0.95 and 0.57, respectively; the common slope was -0.18. Predicted L_p over the observed ranges of C/ha available for each salinity type would decline from 0.57 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 = 0.0221) or forest ($r^2 = 0.09$, P = 0.0162) 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.

Akaike's Information Criteria values equaled 9.3 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 1.53 (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 2-3). 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 2-3).

Additional regressions examining the 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, $r^2 = 0.14$, P = 0.0002; size, $R^2 = 0.27$, P <.0001; and salinity, $R^2 = 0.31$, P <.0001), but combining them into a single model did not improve the fit and size was no longer significant (combined $R^2 = 0.33$; salinity, P = 0.0063 and size, P = 0.1165). 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 2-4, respectively). The relationship between C/ha and L_p in small, brackish systems was better explained, however ($r^2 = 0.56$, P = 0.0001; Table 2-4). A relationship between C/ha and L_p was not detected for large systems (Table 2-4), so additional analyses were performed to explore their differences.

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 2015-2018, 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 2-5; Figure 2-7).

Water quality parameters in large systems exhibited differences in some years among DO, pH, and conductivity between urban and rural systems (Table 2-6; Figures 2-8 through 2-11). In 2015, urban and rural water quality measurements were similar, with the exception of median conductivity which was significantly higher in urban Patuxent River (Table 2-6; Figure 2-8). In 2016, urban Patuxent River had higher DO, conductivity, and pH values than rural Choptank River (Table 2-6; Figure 2-9). This was also true in 2017 and 2018, when rural Choptank River had lower DO and pH values compared to more developed Wicomico (ES) River (Table 2-6; Figures 2-10 and 2-11). Conductivity was consistently higher in the Patuxent River than the Choptank River, but surprisingly, this is not the case in the Wicomico (ES) River 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 to Yellow Perch larvae, they do point to differences in dynamics and conditions among larger tributaries and years.

Although we have analyzed these data by distinguishing 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 influence 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 = 0.46-0.93; 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; 40 observations), while forest land cover was represented by six observations from Nanjemoy Creek (C/ha = 0.09) and two 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 $r^2 = 0.47$, $P < .0001$; $N = 39$), depicting OM0 increasing towards 1.0 at a decreasing rate as C/ha approached 1.50 (Figure 2-12). The relationship was described by the equation:

$$^{(5)}\ \mathrm{OM0} = 0.79 \cdot ((\mathrm{C/ha})^{0.25}).$$

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 $r^2 = 0.44$, $P < .0001$, $N = 39$; Figure 2-13). 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 ≤ 0.65. A full range of L_p values occurred in subestuaries with agricultural watersheds (C/ha was ≤ 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 = 0.51) 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 2.30 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 germ 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 and 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 > 1.36; 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. Declines in L_p in the Magothy, Severn, and South rivers indicated 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 (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. Declines in survival for older Yellow Perch life stages would not be detected using L_p alone. We used L_p as an index to detect "normal" and "abnormal" egg and early larvae 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). Counts coupled with gear efficient at collecting larger, older larvae would be needed to estimate mortality rates. 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 (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. DOP land use estimates from 2010 (most recent year available) indicate that the Wicomico River (ES) would fall into the latter category. Estimates of these three land use categories (agriculture, forest, and urban) in the Wicomico River (ES) watershed 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 upper tidal portion of the Wicomico River (ES), and it is likely that increased development has occurred in this area over the 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 1965-2018 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	1.20	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	0.80
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	0.10	57.9	31.3	1.2	9.5	Agriculture	1	0.60
Choptank	1999	1997	0.11	57.9	31.3	1.2	9.5	Agriculture	1	0.76
Choptank	2000	2002	0.11	58.2	30.8	1.1	9.9	Agriculture	1	0.25
Choptank	2001	2002	0.11	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	0.90
Choptank	2017	2010	0.13	55	27.8	1.4	15.8	Agriculture	1	0.40
Choptank	2018	2010	0.13	55	27.8	1.4	15.8	Agriculture	1	0.44
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

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Elk	2011	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.79
Elk	2012	2010	0.60	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	2.74	1.2	21	0	76.8	Urban	1	0.10
Magothy	2016	2010	2.78	1.2	21	0	76.8	Urban	1	0.10
Mattawoman	1990	1994	0.46	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	0.90	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	0.90	9.3	53.9	2.8	34.2	Forest	0	0.20
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	1.00
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	0.00
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	0.50
Nanticoke	1967	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	0.43
Nanticoke	1968	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	1.00
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

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Nanticoke	2009	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.41
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
Nanticoke	2018	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.28
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	1.00
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	0.50
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	0.40
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	0.30
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	1.32	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
Wicomico (ES)	2018	2010	0.67	30.1	36.8	2.3	29.9	Forest	1	0.38

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	1.01739	1.01739	18.86	<.0001	
Error	59	3.18278	0.05395			
Total	60	4.20017				
r^2	0.2422					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57476	0.03549	16.2	<.0001	0.50375	0.64577
C / ha	-0.15631	0.03599	-4.34	<.0001	-0.22834	-0.08429

ANOVA			(A) Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.77498	0.77498	15.56	0.0004	
Error	30	1.49462	0.04982			
Total	31	2.2696				
r^2	0.3415					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94902	0.08603	11.03	<.0001	0.77332	1.12473
C / ha	-0.29001	0.07353	-3.94	0.0004	-0.44019	-0.13984

ANOVA			(A) Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	2.17385	1.08693	20.33	<.0001	
Error	90	4.81226	0.05347			
Total	92	6.98611				
r^2	0.3112					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.83521	0.05299	15.76	<.0001	0.72994	0.94047
C / ha	-0.18054	0.03243	-5.57	<.0001	-0.24496	-0.11612
Salinity	-0.24741	0.05303	-4.67	<.0001	-0.35276	-0.14206

Table 2-2 cont.

ANOVA			(B) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.35963	0.35963	5.52	0.0221	
Error	59	3.84054	0.06509			
Total	60	4.20017				
r^2	0.0856					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.3493	0.06844	5.1	<.0001	0.21236	0.48625
% Ag	0.00378	0.00161	2.35	0.0221	0.00056136	0.00699

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	1.01879	0.5094	7.68	0.0008	
Error	90	5.96732	0.0663			
Total	92	6.98611				
r^2	0.1458					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57942	0.0518	11.18	<.0001	0.4765	0.68233
% Ag	0.00427	0.00155	2.75	0.0071	0.00119	0.00734
Salinity	-0.24846	0.06532	-3.8	0.0003	-0.37822	-0.11869

Table 2-2 cont.

ANOVA			(C) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.39548	0.39548	6.13	0.0162	
Error	59	3.80469	0.06449			
Total	60	4.20017				
r^2	0.0942					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.24156	0.10571	2.29	0.0259	0.03004	0.45309
% Forest	0.00652	0.00263	2.48	0.0162	0.00125	0.01179

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	1.12655	0.56328	8.65	0.0004	
Error	90	5.85956	0.06511			
Total	92	6.98611				
r^2	0.1613					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.34735	0.10792	3.22	0.0018	0.13296	0.56175
% Forest	0.00711	0.00232	3.06	0.0029	0.0025	0.01172
Salinity	-0.12829	0.05647	-2.27	0.0255	-0.24047	-0.0161

Table 2-3. Summary of Akaike's Information Criteria from 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 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	0.05347	93	4	2.92863	8	40	88	11.4	2.04	1.53
Fresh	0.04982	32	3	2.99934	6	24	28	9.9		
Brackish	0.05395	61	3	2.91970	6	24	57	9.3		

Model (B)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.0663	93	4	2.713565382	8	40	88	11.2	2.02	1.63
Fresh	0.06856	32	3	2.680046005	6	24	28	9.5		
Brackish	0.06509	61	3	2.731984351	6	24	57	9.2		

Model (C)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.06511	93	4	2.731677132	8	40	88	11.2	2.02	1.64
Fresh	0.06803	32	3	2.687806495	6	24	28	9.5		
Brackish	0.06449	61	3	2.741245106	6	24	57	9.2		

Table 2-4. 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	1.21604	1.21604	22.94	0.0001	
Error	18	0.95406	0.053			
Total	19	2.1701				
r^2	0.5604					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.70523	0.07951	8.87	<.0001	0.53819	0.87227
C / ha	-0.22961	0.04794	-4.79	0.0001	-0.33032	-0.1289

ANOVA	(A) Small Tidal-Fresh					
Source	df	SS	MS	F	P	
Model	1	0.77498	0.77498	15.56	0.0004	
Error	30	1.49462	0.04982			
Total	31	2.2696				
r^2	0.3415					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94902	0.08603	11.03	<.0001	0.77332	1.12473
C / ha	-0.29001	0.07353	-3.94	0.0004	-0.44019	-0.13984

ANOVA	(B) Large Brackish					
Source	df	SS	MS	F	P	
Model	1	0.02906	0.02906	0.75	0.396	
Error	20	0.77247	0.03862			
Total	21	0.80153				
r^2	0.0363					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.55072	0.05252	10.49	<.0001	0.44116	0.66027
C / ha	0.10564	0.12178	0.87	0.396	-0.1484	0.35967

Table 2-5. 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 2015-2018.

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

Choptank 2018	Stations	Presence	N	Lp	SD
Down-river	1-5	5	13	0.38462	0.1349
Mid-river	6-11	13	36	0.3611	0.0801
Up-river	12-17 and 18-20	26	50	0.5200	0.0707

Wicomico 2018	Stations	Presence	N	Lp	SD
Down-river	1-4	10	34	0.2941	0.0781
Mid-river	5-8	20	35	0.57143	0.0836
Up-river	9-12	8	31	0.2581	0.0786

Table 2-6. Summary of annual water quality parameter statistics for large systems sampled in 2015-2018. Mean pH was calculated from H^+ concentrations and back-converted for reporting here.

System/Year		*Temp C*	*DO (mg/L)*	*Cond (umhols)*	*pH*
	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
Choptank 15	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
	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
Patuxent 15	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
	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
Choptank 16	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
	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
Patuxent 16	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 2.6 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
	Mean	12.59	8.73	514.53	7.15
	Standard Error	0.29	0.13	71.98	
	Median	13.12	8.60	178.5	7.19
	Mode	13.56	10.13	173	6.96
Choptank 18	Kurtosis	-0.94	-1.38	5.48	-0.66
	Skewness	-0.20	0.01	2.45	0.41
	Minimum	6.92	6.28	122	6.71
	Maximum	17.08	10.98	3366	7.86
	Count	100	100	100	100
	Mean	12.87	12.04	412.82	7.80
	Standard Error	0.28	0.14	53.71	
	Median	12.76	12.19	219	7.97
	Mode	8.20	13.39	216	7.58
Wicomico ES 18	Kurtosis	-0.77	0.23	18.85	-1.20
	Skewness	-0.42	-0.67	3.96	0.23
	Minimum	7.41	8.10	138	7.24
	Maximum	17.21	14.80	3847	8.97
	Count	100	100	100	100

Figure 2-1. Areas sampled for Yellow Perch larval presence-absence studies, 2006-2018. Areas sampled in 2018 are highlighted in green. Nanticoke River watershed delineation was unavailable for Delaware and Northeast and was unavailable for Pennsylvania.

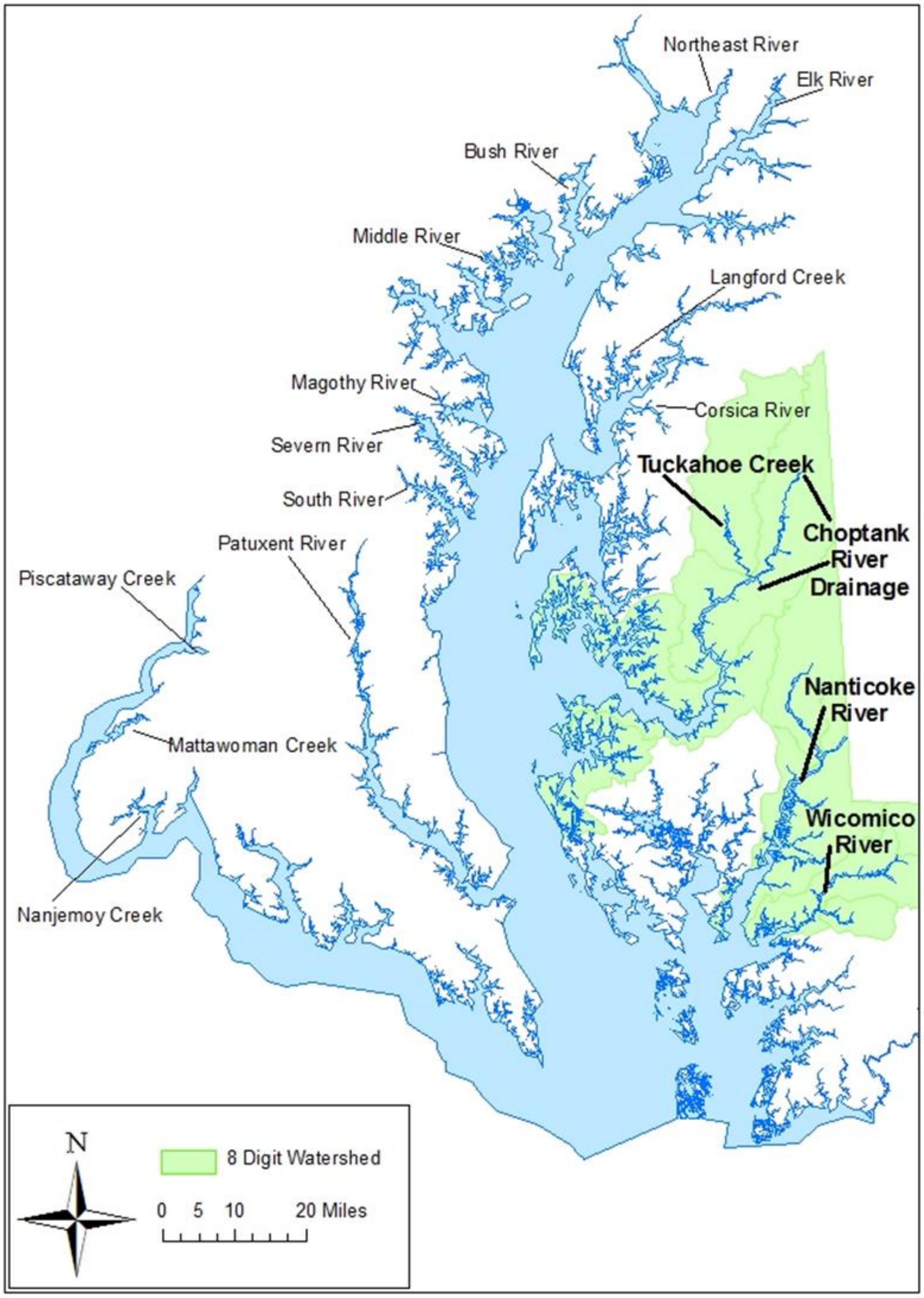


Figure 2-2. Proportion of tows with larval Yellow Perch (*Lp*) and its 95% confidence interval in systems studied during 2018. Mean *Lp* of brackish tributaries indicated by green diamond, and brackish subestuary threshold indicated by dotted line.

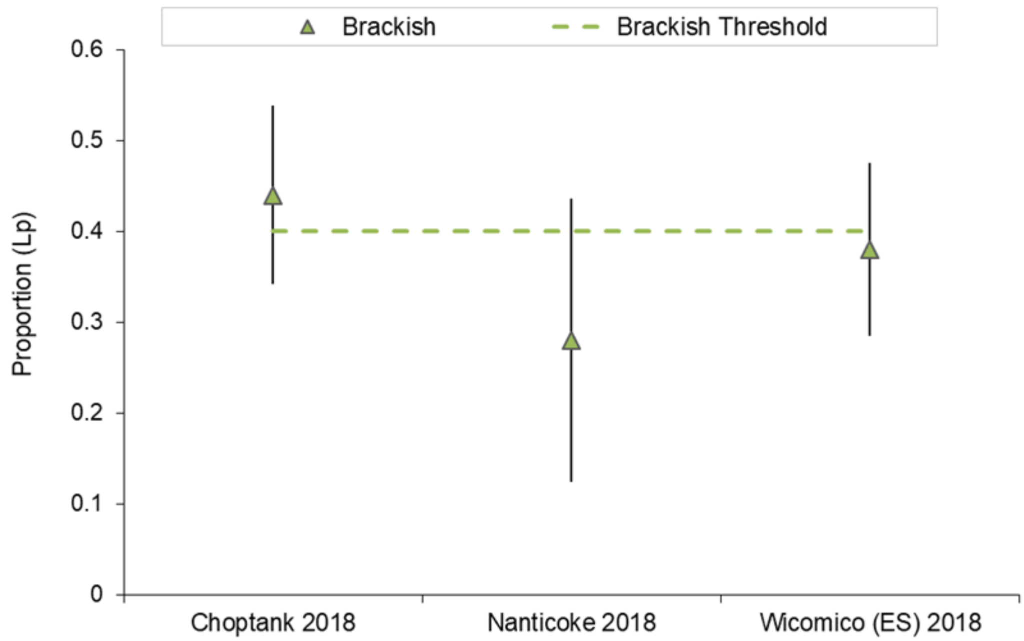


Figure 2-3. Proportion of tows with Yellow Perch larvae (*Lp*) for brackish subestuaries, during 1965-2018. Dotted line provides threshold for persistent poor *Lp* exhibited in developed brackish subestuaries. Dominant Department of Planning land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

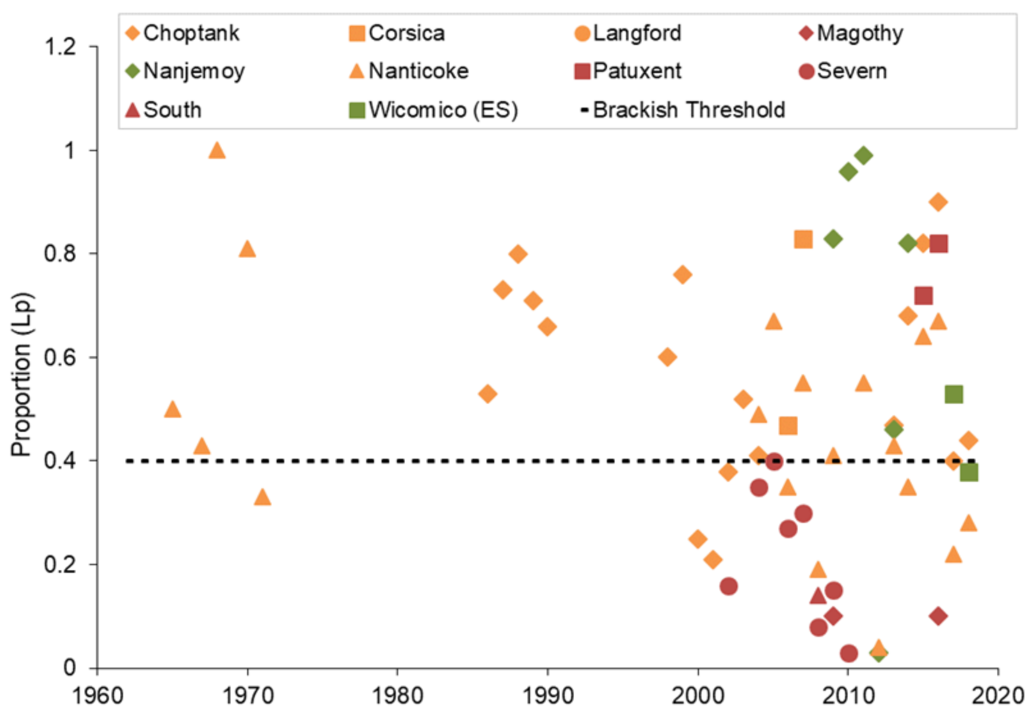


Figure 2-4. Proportion of tows with Yellow Perch larvae (*Lp*) for tidal-fresh subestuaries, during 1990-2018. Dotted line provides reference for consistent poor *Lp* exhibited in a more developed tidal-fresh subestuary (Piscataway Creek). Dominant Department of Planning land use is indicated by symbol color (green = forest and red = urban).

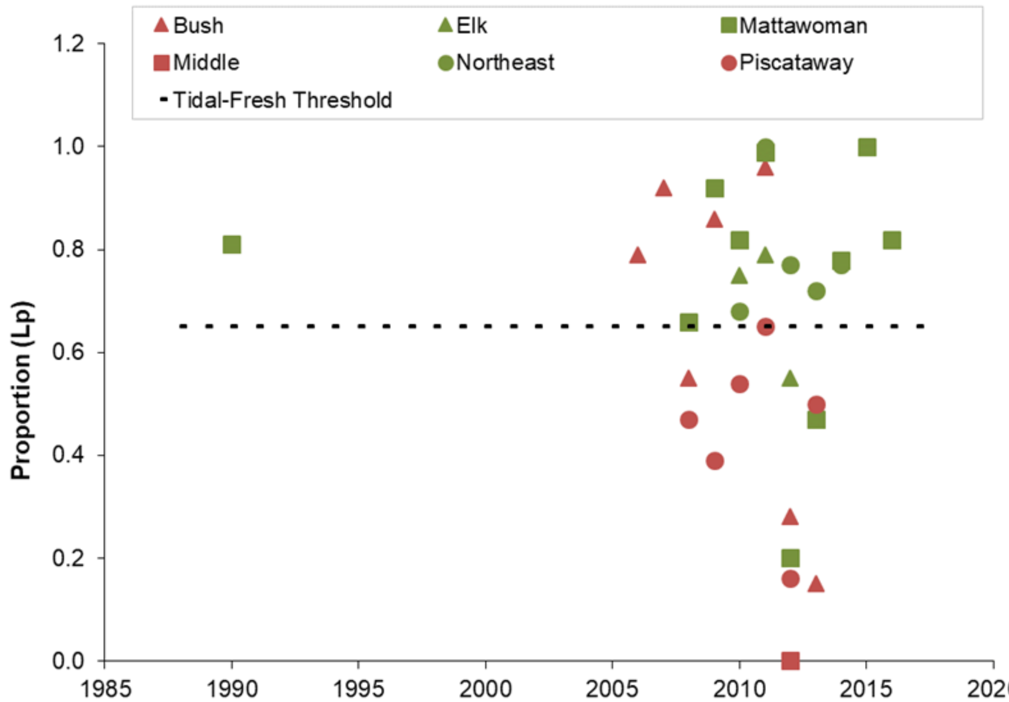


Figure 2-5. Relationship of proportion of plankton tows with Yellow Perch larvae and (A) development (structures per hectare or C/ha), (B) percent agriculture, and (C) percent forest, indicated by multiple regression of fresh and brackish subestuaries combined (prediction = MR) and separate linear regressions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

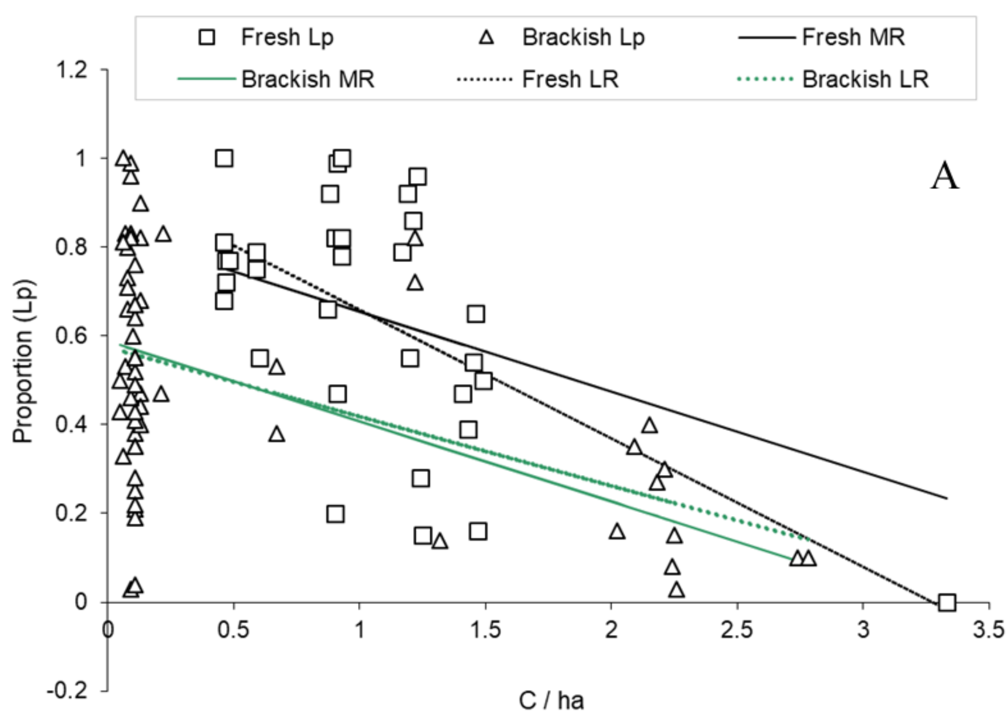
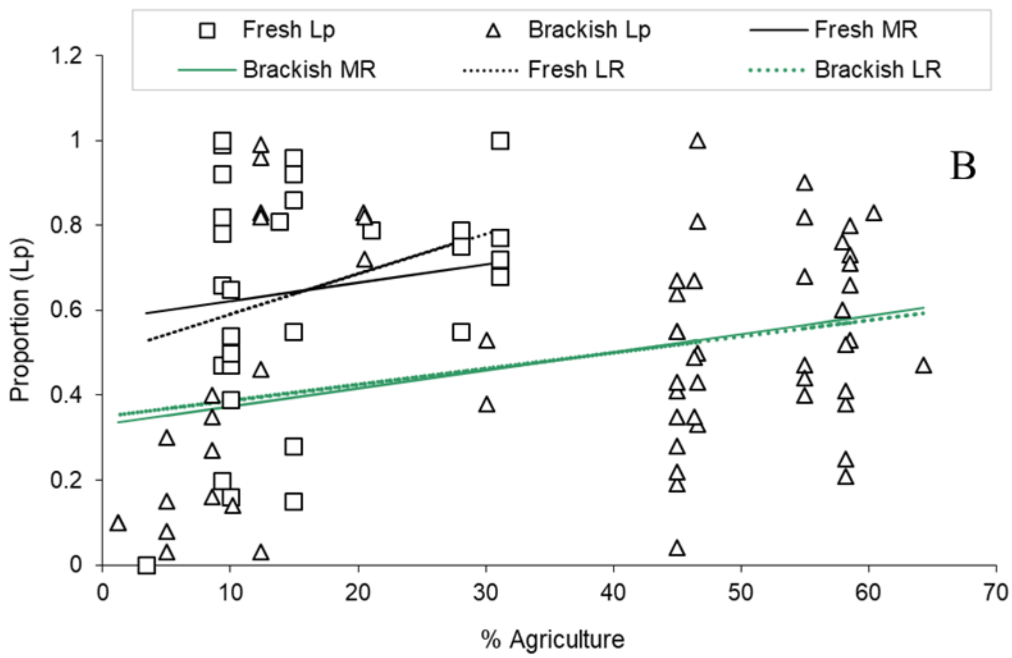


Figure 2-5 cont.



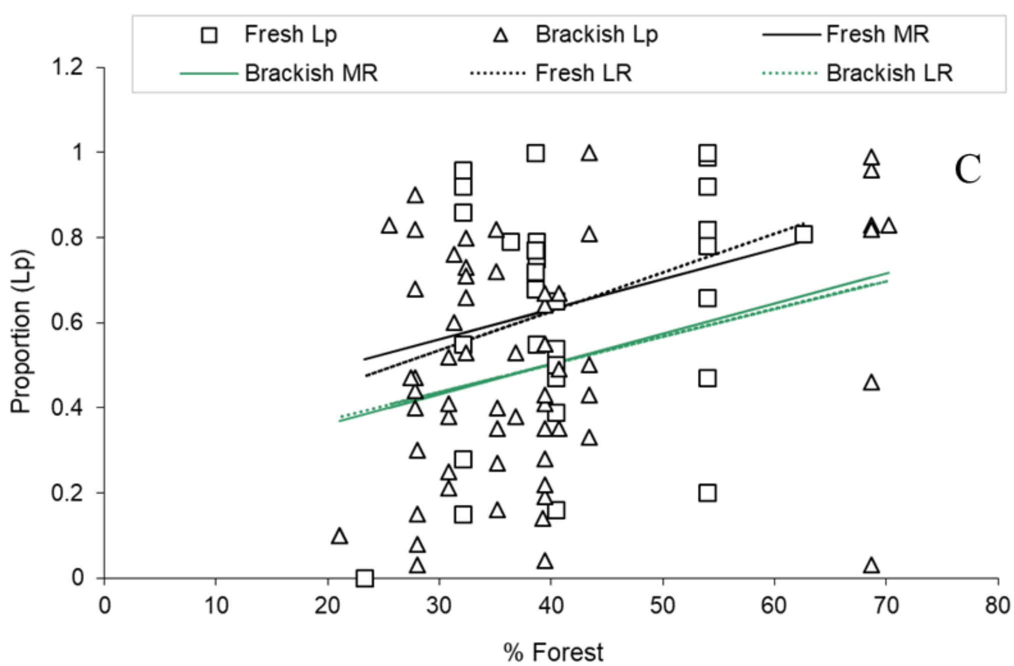


Figure 2-6. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations and salinity class indicated by symbols. Diamonds and a "1" behind land use in the key indicate brackish subestuaries, while squares and a "0" indicate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

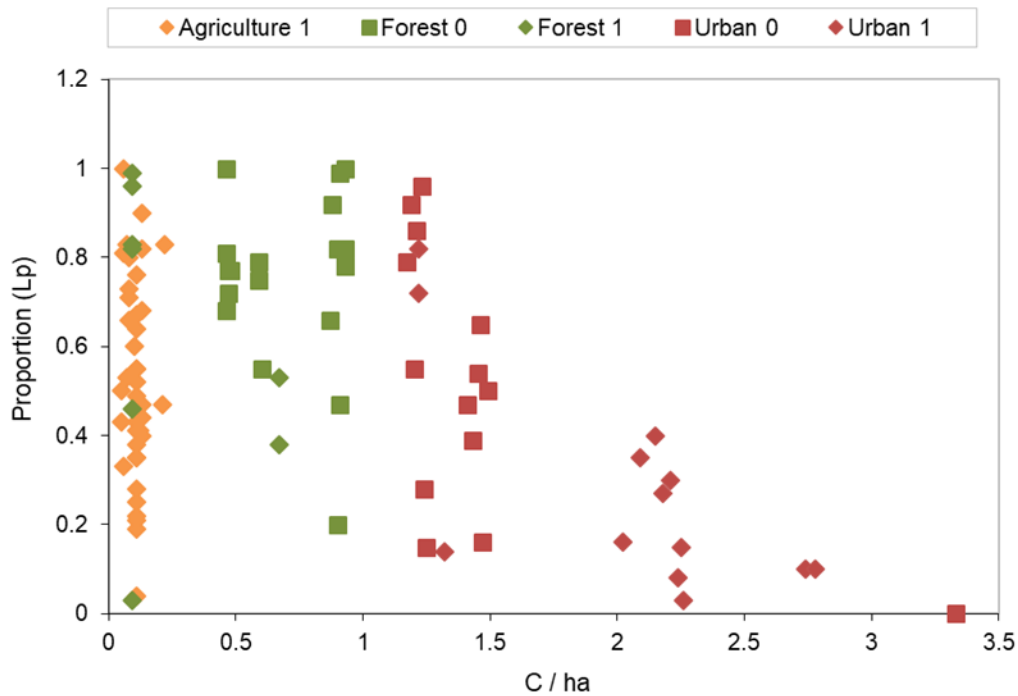


Figure 2-7. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations for large systems.

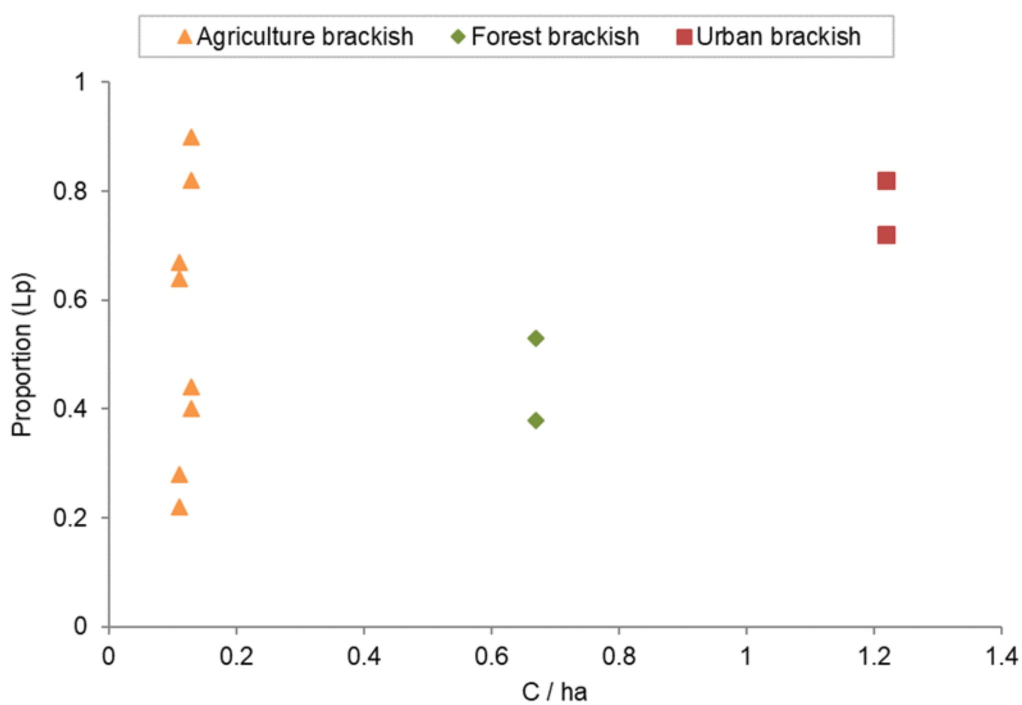


Figure 2-8. Water quality parameters sampled in Choptank and Patuxent rivers during 2015.

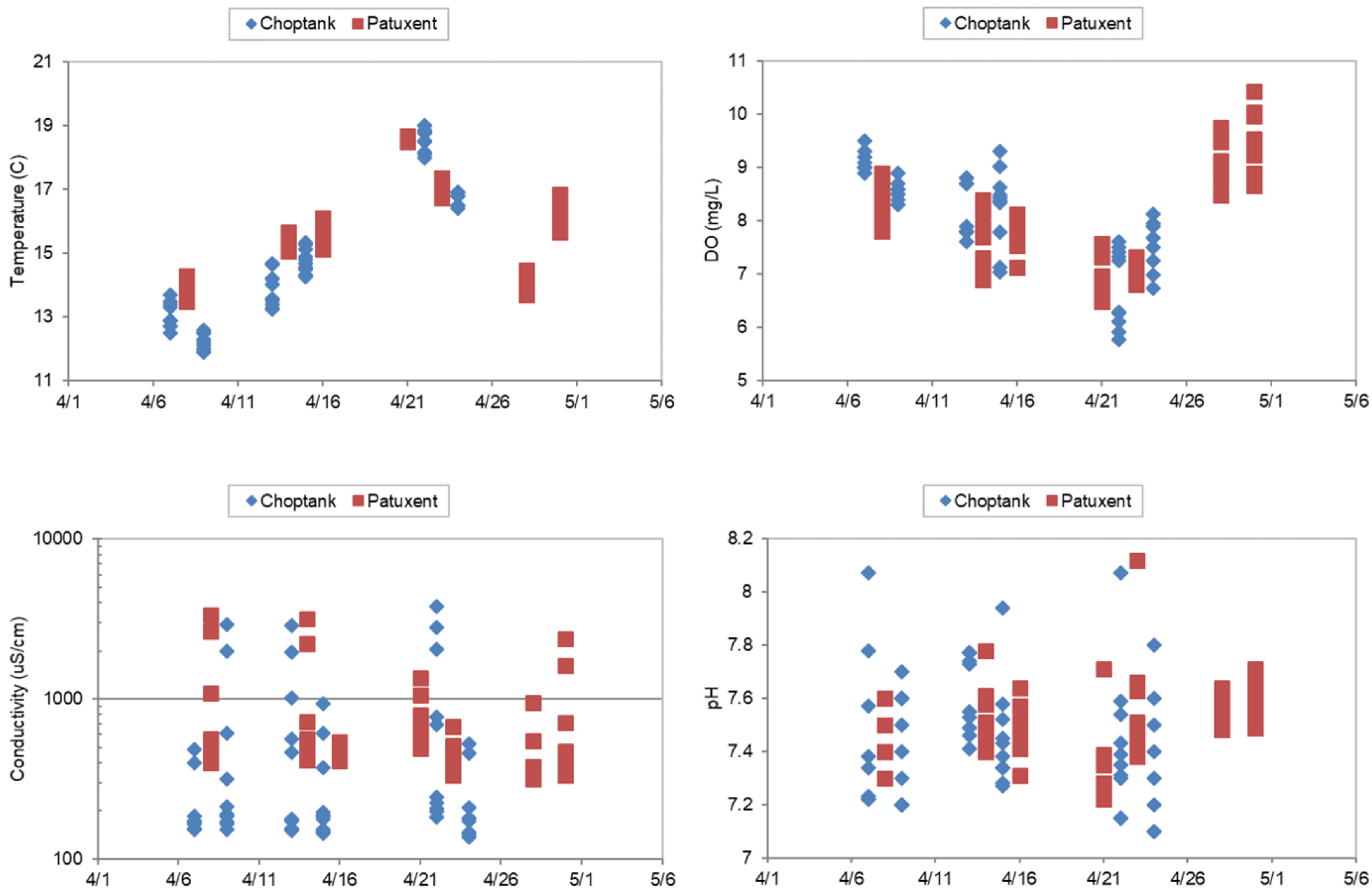


Figure 2-9. Water quality parameters sampled in Choptank and Patuxent rivers during 2016.

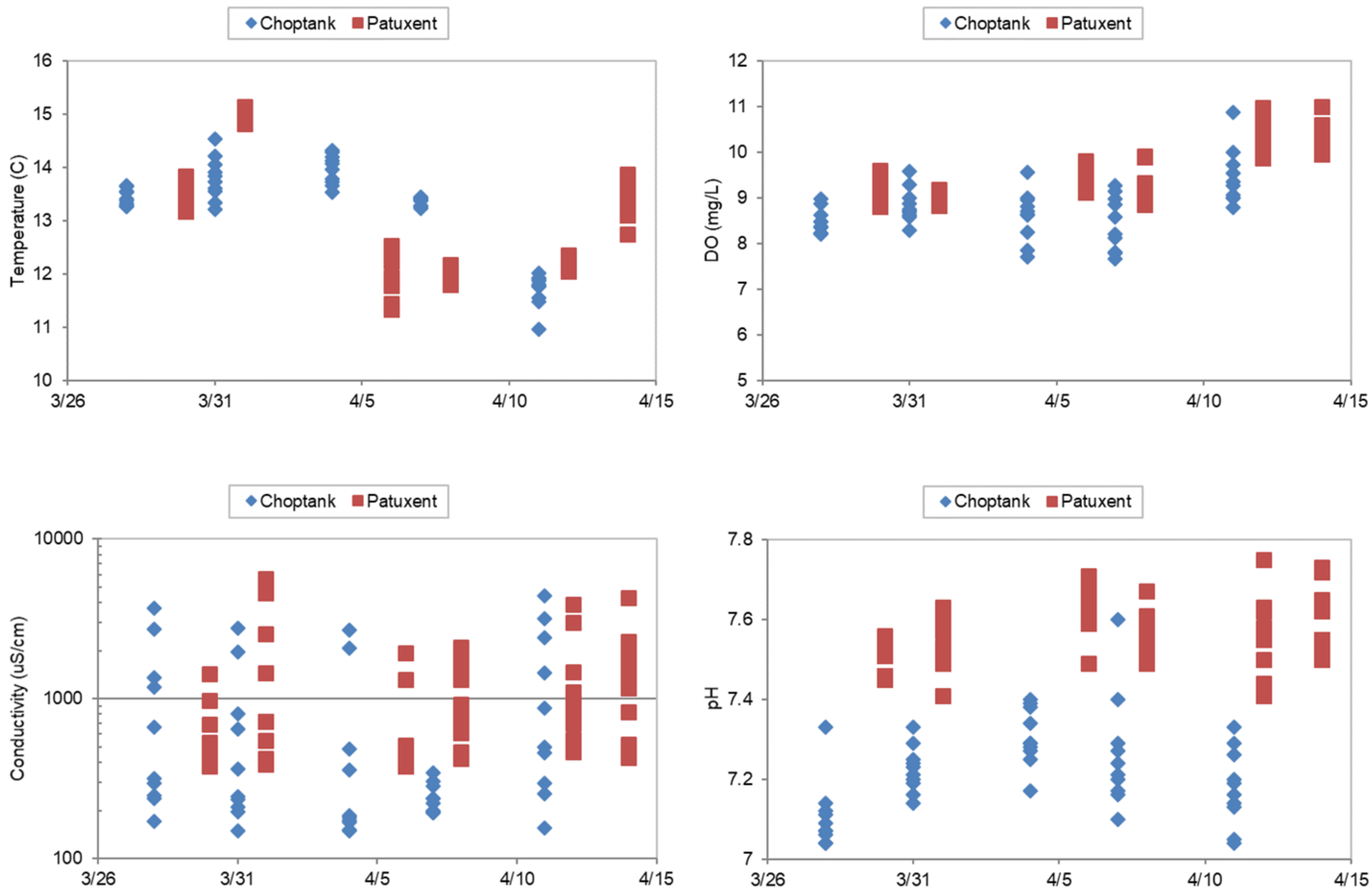


Figure 2-10. Water quality parameters sampled in Choptank and Wicomico (ES) rivers during 2017.

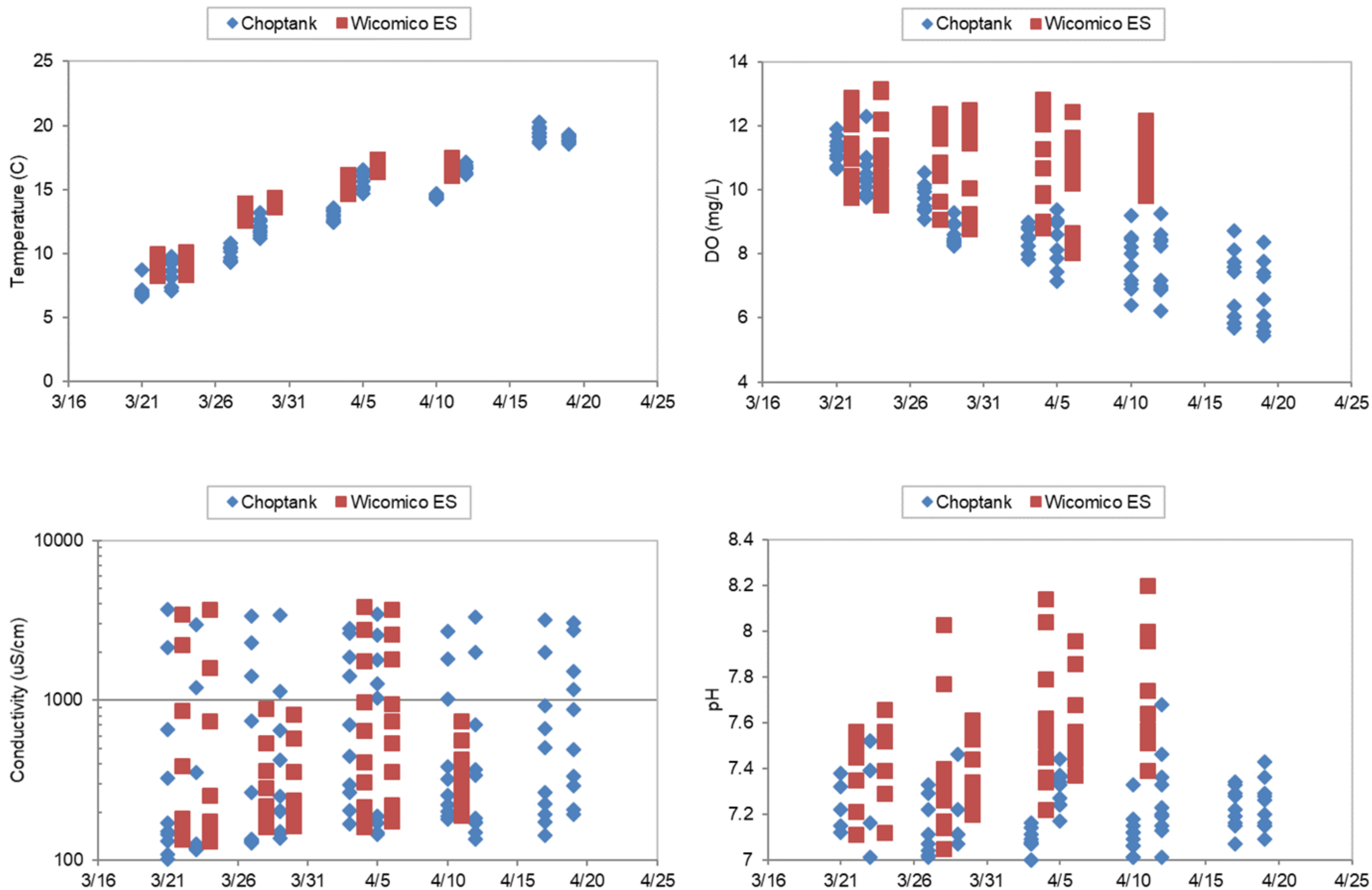


Figure 2-11. Water quality parameters sampled in Choptank and Wicomico (ES) rivers during 2018.

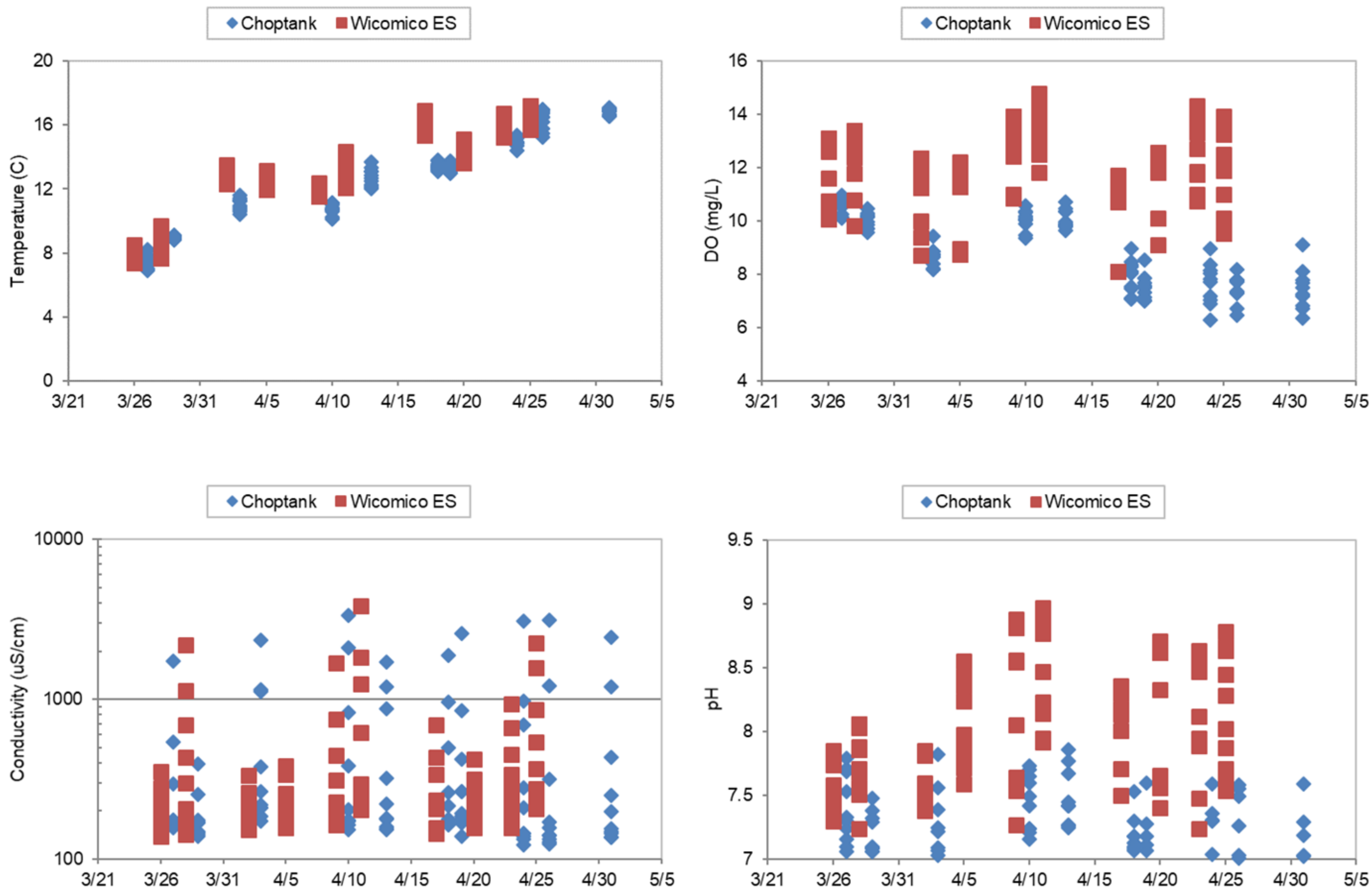


Figure 2-12. Relationship of proportion of plankton tows without detritus (OM0) and development (structures per hectare or C/ha). Dominant Department of Planning land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

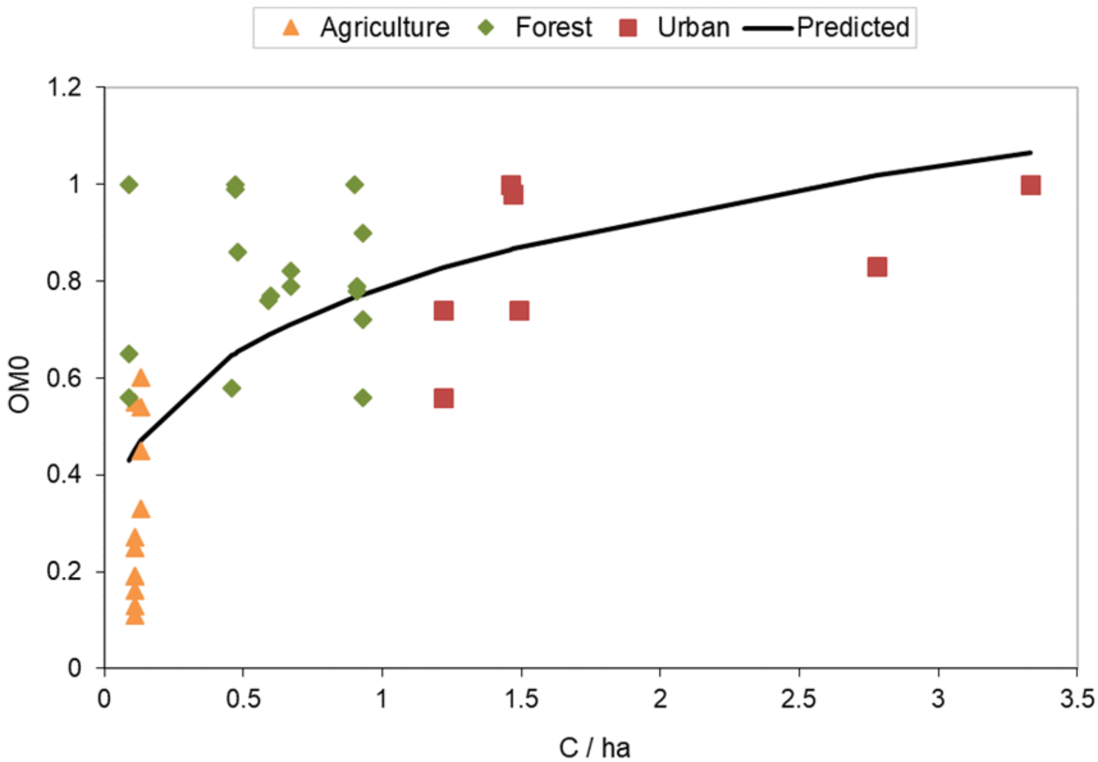
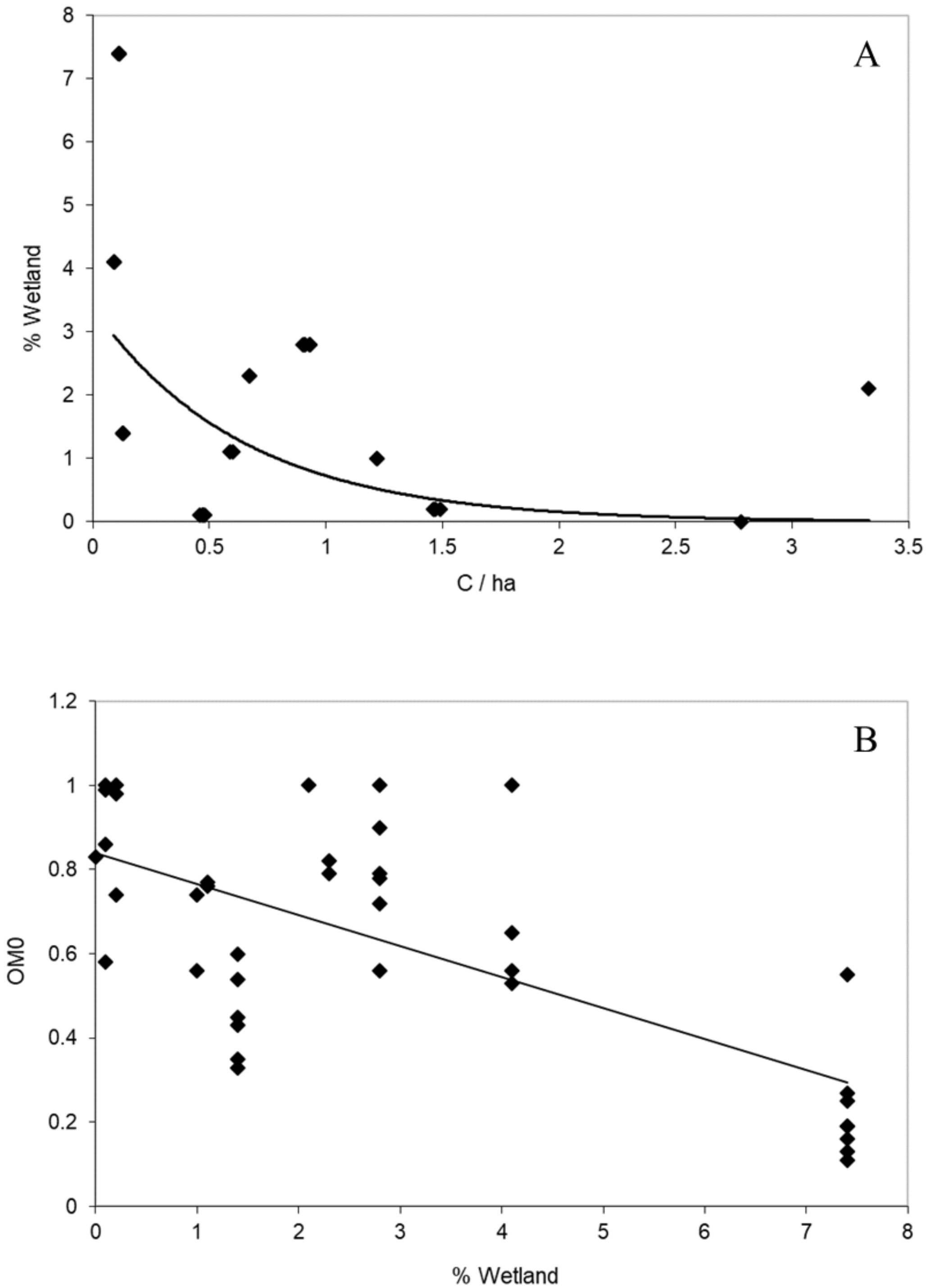


Figure 2-13. (A) Relationship of percent wetlands per watershed obtained from 2010 Department of Planning estimations and level of development (C/ha). (B) Proportion of samples without organic material (OM0) and percent wetlands per watershed.



Section 3 - Estuarine Fish Community Sampling

Alexis Park, Carrie Hoover, Margaret McGinty, Jim Uphoff, Marcus Patton

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; 2017; 2018) 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 2018, 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 mesohaline (5.0-18.0 ‰; Oertli, 1964) subestuaries of the Chesapeake Bay. In this section, 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 among subestuaries sampled during 2003-2018 using correlation analysis (Pearson correlation coefficients). We continue to examine and Tred Avon River, a tributary of Choptank River located in Talbot County (Table 3-1; Figure 3-1). In 2018, we returned to previously sampled middle Bay subestuaries, Chester River, Corsica River, Langford Creek, and Wye River in Queen Anne's County to support the County's pending comprehensive growth plan (Table 3-1; Figure 3-1). We examined associations among relative abundance of all finfish from Choptank River and the Head of Bay with Chester and Tred Avon Rivers to evaluate potential contributions of the two large outside regions to the abundance in subestuaries in our study. High to record rainfall in the Chesapeake Bay watershed and runoff preceded and continued during summer 2018 sampling. We added an evaluation of precipitation patterns to our analysis in order to better understand how increased precipitation conditions may have impacted our evaluations of watersheds during 2018.

Methods

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

We sampled four Chesapeake Bay mesohaline subestuaries in Queen Anne's County during 2018 to support their Comprehensive growth plan: Corsica River and Langford Creek (mesohaline tributaries of the Chester River), Chester River, and Wye River. We returned to the Corsica River, previously sampled from 2003 to 2012; Langford Creek, previously sampled from 2006 to 2008; and the Wye River, previously sampled from 2007 to 2008. The Chester River was previously sampled by other MD DNR programs, Resource Assessment Service from 1994 to 2000 and the Shad and Herring Program from 2007 to 2012.

The Tred Avon River, a mesohaline subestuary of the Choptank River in Talbot County, has been sampled since 2006 (Figure 3-1), one year ahead of a substantial development project. We have continued monitoring Tred Avon River in anticipation of DO and fish community changes as its watershed continues to develop and contrasted it with less developed Harris Creek and Broad Creek watersheds in the same region (Figure 3-1). Talbot County and the town of Easton (located at the upper Tred Avon River) have active programs to mitigate runoff and this provides an opportunity to evaluate how well up-to-date stormwater management practices maintain subestuary fish habitat. Starting in 2012, we assessed adjacent subestuaries that were less developed (Figure 3-2): Broad Creek (through 2017) and Harris Creek (through 2016; Uphoff et al. 2015; 2016; 2017).

We used property tax map-based counts of structures in a watershed (C), standardized to hectares (C / ha), as our indicator of development (Uphoff et al. 2012; Topolski 2015). Estimates of C / ha and Maryland Department of Planning land use and water percentages were used for analyses of data from mesohaline subestuaries sampled during 2003-2018 (Table 3-2). Estimates were available through 2016; 2016 estimates were used to represent 2016-2018 in analyses. Methods used to estimate development (C / ha) and land use indicators (percent of agriculture, forest, wetlands, urban land use, and water in the watershed) are explained in **General Spatial and Analytical Methods used in Job 1, Sections 1-3.** The C / ha to impervious surfaces (IS) conversion based on 1999-2000 property tax map estimates and subestuaries was revised this year, 2018, to reflect updates and led to revised C / ha levels for IS reference points (5% IS = 0.37; 10% IS = 0.86; and 15% IS = 1.35). 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.

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, Yellow Perch, and Bay Anchovy), marine migrants (Atlantic Menhaden and Spot), and tidal-fresh forage (Spottail Shiner, Silvery Minnow, and Gizzard Shad). With the exception of White Perch, adult sportfish 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 2019).

Ideally, four evenly spaced haul seine and bottom trawl sample sites were located in the upper two-thirds of each subestuary. We focused on using previously sampled historical sites in each of the subestuaries sampled in 2018 unless they were no longer accessible. The Corsica River, Langford Creek, and Wye River lacked shoreline for a fourth seine site; each system has four bottom trawl sites and three beach 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 beginning and end of the trawl site, while latitude and longitude at seining sites were taken at the seine starting point on the beach. Only beach seines were conducted on the Chester River in 2018, six seine sites were selected throughout the river based on previous sites sampled in 2012. During preceding years, bottom trawls were used to sample the Chester River but were not used during 2018 due to limited staff. Beach seining allowed for comparisons of relative abundance of target species with previous years in the Chester River and adequacy of its channel bottom habitat was assessed from DO surveys. Omitting trawling allowed for the Chester River sampling to be conducted in one day and freed up limited project personnel to sample additional subestuaries.

Sites were sampled once every two weeks during July-September, totaling six annual visits per 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); Chester River was the only system with 6 seine sites. The crew determined whether to start upstream or downstream based on tidal direction; this helped randomized potential effects of location and time of day on catches and DO and assisted the crew with site availability. 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 sampling was conducted one right after the other at a site to minimize time of day or tidal influences between samples.

Water quality parameters were recorded at both seine and trawl sites. Temperature (ºC), DO (mg / L), conductivity (mS / cm), salinity (parts per thousand; ppt = ‰), and pH were recorded at the surface, middle, and bottom of the water column at the trawl sites depending on depth 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. In Chester River, bottom water quality parameters were recorded in the channel at three locations (upper, middle, and lower seine 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 than emptied into a tub for processing.

A 30.5 m × 1.2 m bag-less 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 tub for processing. The distance the net was stretched from shore, maximum depth of the seine haul, primary and secondary bottom types (i.e., gravel, sand, mud, and shell), and percent of seine area containing submerged aquatic vegetation were recorded.

All fish captured were identified to species and counted. Striped Bass and Yellow Perch were separated into two age categories, juveniles (young of year = YOY) and adults (ages 1+). White Perch were separated into three age categories based on size and life stage, juveniles, small adults (ages 1+ fish measuring < 200 mm), and harvestable size adults (fish measuring > 200 mm). Harvestable size adult White Perch were also measured and the measurements were recorded for proportional stock density analysis.

Seining in Corsica River, Langford Creek, and Wye River was very restricted because of high tides that limited beach availability during 2018; only 3 of the 4 seine sites could be sampled in each of these subestuaries. Higher than normal high tides have become increasingly common and prevent the seine from being stretched the whole 30.5 meters (m) in length. Dense submerged aquatic vegetation (SAV), previously an issue in other systems during preceding years, was not an issue in the subestuaries during the 2018 sampling season. Unlike seining sites, all trawl sites could be sampled during 2018.

2018 Sampling Summary - Three basic metrics of community composition were estimated for subestuaries sampled: geometric mean (GM) catch of all species, total number of species (species richness), and species comprising 90 % of the catch. The GM of seine or trawl catches were 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). In addition, 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.

We plotted species richness in seine and trawl collections against C / ha by salinity class. A greater range of years (1989-2018) was available for beach seine samples than the 4.9 m bottom trawl (2003-2018) due to a change from the 3.1 m trawl used during 1989-2002 (Carmichael et al. 1992). Gear comparison analysis between the 3.1 m and 4.9 m trawls can be reviewed in Uphoff et al. (2016). We set a minimum number of samples (15 for seine and trawl) 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) or high tides. We separated all subestuaries sampled during 1989-2018 by salinity class, then ranked their bottom trawl GMs by year for all species combined to find where the 2018 subestuaries sampled ranked when compared to other subestuaries in their respective salinity classes.

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 DO samples (surface, middle, and bottom DO) and for bottom DO. Percentages not meeting target or threshold conditions were termed "violations", but the term did not have a regulatory meaning. The percentages of DO measurements that met or fell below the 5 mg / L target (V_{target}) or fell at or below the 3 mg / L threshold ($V_{threshold}$) were estimated as:

$$V_{target} = (N_{target} / N_{total})*100;$$

and

$$V_{threshold} = (N_{threshold} / N_{total})*100;$$

where N_{target} was the number of measurements meeting or falling below 5 mg / L, $N_{threshold}$ was the number of measurements falling at or below 3 mg / L, and N_{total} 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 sample 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 Pearson 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 survey 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 and Bottom Dissolved Oxygen – 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 2010-2018 were matched with 2010 MD DOP estimates (the most current available). Four categories of land use (percent in agriculture, forest, urban, and wetlands) were estimated based on the land portion of the watershed (water area was excluded from these categories). A fifth category, percent in water, was estimated based on the water plus land area of the watershed.

We analyzed the associations of land use (i.e., agriculture, forest, urban, and wetlands) and C / ha (structures per hectare) with annual median bottom DO among mesohaline systems sampled during 2003-2018 using correlation analysis (Pearson correlation coefficients). We further examined the influence of percent of land in agriculture on median bottom DO using linear, multiple linear, and quadratic regression models.

Precipitation – Precipitation during summer 2018 was unusually high and salinities at some of our stations that should have been in the mesohaline range were far lower, indicating potential for dramatic habitat change. Anecdotally, we observed unusual loads of dead leaves and empty clam shells in many of our samples. This prompted us to add analyses to understand how this unusual precipitation pattern may have impacted our surveys.

We analyzed mean monthly precipitation for years 2014-2018, including near-term 5 year (2014-2018) and long-term 25 year (1993-2018) mean monthly trends to observe changes in precipitation patterns over a short (5 years) and long (25 years) periods. We obtained mean monthly precipitation estimates from NOAA's National Climatic Data Center (NCDC 2019) for Kent, Queen Anne's, and Talbot Counties. The number of stations within a county recording precipitation varied and changed over the years. The long-term trend for Queen Anne's County actually consisted of 13 years since monthly precipitation data was only available for 2006 to 2018.

Tred Avon River - In 2018, we sampled four stations in Tred Avon River (Figure 3-3). We contrasted Tred Avon River to Broad Creek (sampled during 2012-2017 and Harris Creek (2012-2016; Figure 3-3). Trajectories of C / ha since 1950 were plotted for the three Choptank River subestuaries. Bottom DO measurements during 2006-2018 were plotted against C / ha and percent of target and threshold DO violations were estimated using all measurements combined (surface, middle, and bottom) and for bottom DO only. Annual mean bottom DO (depth most sensitive to violations) in Tred Avon River at each station for 2006-2018 was estimated and plotted by year. We examined correlations of Secchi depths, 4.9 m bottom trawl geometric mean catches of all finfish or adult White Perch, SAV coverage, DO, pH, and salinity among the three subestuaries. We estimated GMs of trawl or seine catches, and species composition.

We used a percent similarity index to calculate percent similarity of the finfish species composition among Tred Avon River trawl stations 1, 2, 3, and 4 by year (Kwak and Peterson 2007). Finfish species abundances per a trawl station were standardized to percentages by dividing the abundance of each finfish species in a trawl station by the total number of fish collected at that trawl station, by year. The similarity among stations, P_{jklm} was calculated as:

$$\sum \text{minimum } (p_{ji},\ p_{ki},\ p_{li}\ p_{mi});$$

where p_{ji}, p_{ki}, p_{li}, and p_{mi} refers to the finfish species abundance of one particular finfish species *i* in trawl stations *j*, *k*, *l*, and *m*, by year, and the minimum indicates that the smallest of the four relative abundances was used in the summation (Kwak and Peterson 2007). The greater the similarity value, the more finfish species present and abundant throughout all four bottom trawl stations; lower values indicate finfish species are uncommon and/or scarce throughout all four trawl stations.

An ANOVA was used to examine differences in mean bottom DO among stations in Broad Creek, Harris Creek, or Tred Avon River. Tukey Studentized Range and Tukey Honestly Significant Difference (HSD) tests examined whether station(s) within each subestuary were significantly different from other station(s). An overall median DO was calculated for all time-series data available for each system and used to detect how annual DO per a station fell against the time-series median DO. Pearson correlation coefficients analyzed annual median DO measurements to observe the relationship among the three systems, as well as annual median DO trends.

In addition to our standard fish metrics, we also compared adult White Perch trawl GMs from Broad Creek, Harris Creek, and Tred Avon River. White perch adults were consistently abundant and represented the only adult gamefish that routinely appeared in samples using Pearson correlation coefficients.

Queen Anne's County Subestuaries - In 2018, we sampled mainstem Chester River, Corsica River, Langford Creek, and Wye River (Figure 3-1) to provide information on fish habitat status for the pending Queen Anne's County's comprehensive growth plan. These subestuaries had been monitored in the past; Chester River in 2007-2012, Corsica River in 2003-2012, Langford Creek in 2006-2008, and Wye River in 2007-2008 (Figure 3-4).

We assembled time-series of Secchi depth, SAV area, bottom dissolved oxygen (DO; mg / L), pH, and salinity (ppt = ‰). Annual GMs of total fish relative abundance and their 95 % CIs were estimated for 4.9 m trawl. Annual compositions of all finfish species caught by seine were graphed. The top 90 % of finfish species occurring in annual trawl and seine catches was estimated for each subestuary time-series.

An ANOVA was used to evaluate the station differences in mean bottom DO; Tukey Studentized Range and Tukey Honestly Significant Difference (HSD) tests examined which station(s) within each subestuary were significantly different from others. An overall median DO was calculated for all time-series data available for each system and used to detect how annual DO per a station fell against the time-series median DO. Pearson correlation coefficients analyzed annual median DO measurements to observe the relationship among the four systems, as well as annual median DO trends. We also used correlation analysis to examine associations among subestuaries of each water quality variable: Secchi depth, 4.9 m bottom trawl GM catches, DO, pH, or salinity.

Exploration of relative abundance of finfish in Chester River, Tred Avon River, Choptank River, and Head-of-Bay seine samples - We compared relative abundance of all fish collected in our seine catches to the same metric from adjacent regions sampled by the Juvenile Index Striped Bass survey (JI survey; Durell and Weedon 2019). Annual geometric means (GM) of all finfish sampled in Head of Bay and Choptank River during the JI survey (Durell and Weedon 2019) were compared to available data collections from Chester River and Tred Avon River using correlation analysis to see how coherent trends were. If trends were coherent, then there was some chance that the Chester River and-or Tred Avon River finfish populations could be significantly supplemented by adjacent, larger subestuaries.

Catch data from the first seine haul at both permanent and auxiliary sites for Head of Bay and Choptank River were used in these analyses. Using the first haul duplicated the single haul used in our work; GMs included all finfish present in catches. The Chester River annual GM was based on data collected at various times by the Striped Bass Program, Resource Assessment Services, Shad and Herring Program, and Fisheries Habitat and Ecosystem Program.

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 (before 2006) e entered incorrectly (i.e., entered twice, skipped, or disorderly). Incorrect data were corrected; quality control is ongoing due to size of database. Corrected data is used throughout the analyses in this report.

Results and Discussion

2018 Sampling Summary – All tributaries and subestuaries sampled had DO readings less than the target level (5.0 mg / L) during 2018 (Table 3-3). Three percent of all DO measurements (surface and bottom) in 2018 from Chester River were below the target; Langford Creek had 6 %; Bush River, 13 %; Tred Avon River, 18 %; Corsica River, 26 %; and Wye River, 27 %. In 2018, only two subestuaries did not have any bottom DO estimates below the 3 mg / L threshold; Chester River and Langford Creek (Table 3-3). The remaining three subestuaries had threshold bottom DO violations: Corsica River, 9 %; Tred Avon River, 14 %; and Wye River, 15 %. Bottom DO was not measured in Bush River during 2018.

Geometric mean catch per seine haul ranged from 129 to 237 among the six subestuaries sampled during 2018 (Table 3-4). Bush River, considered an oligohaline subestuary, had a GM catch per seine haul of 279. Salinities were lower than average for all subestuaries sampled during the 2018 season due to large amounts of precipitation during the spring and summer seasons; Chester River salinities were less than 5 ‰, shifting the normally mesohaline subestuary to oligohaline during 2018. Salinities in remaining subestuaries were close to or within mesohaline bounds.

Normal monthly precipitation totals from March to September were 28.4 inches for Harford County; 26.5 inches in Queen Anne's County; 26.8 inches for Talbot County; and 26.9 inches in Kent County (MDE 2018). Median monthly precipitation totals in 2018 from March to September for Hartford County was 35.0 inches; Queen Anne's County, 37.1 inches; Talbot County, 33.8 inches; and Kent County, 37.8 inches (NCDC 2019).

Geometric mean seine catches in 2018 ranked Bush River 1st; Langford Creek, 2nd; Corsica River, 3rd; Wye River, 4th; Tred Avon River, 5th; and Chester River ranked 6th. Between 19 and 27 species were encountered in mesohaline tributary seine samples (Table 3-4). Oligohaline Bush River had 27 species present in seine catches, including a Northern Snakehead.

A plot of species richness in seine samples and C / ha during 1989-2018 did not suggest a strong relationship in tidal-fresh, oligohaline, or mesohaline subestuaries (Figure 3-5). 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) of the three salinity classes. Mesohaline subestuary watersheds were represented by a larger number of samples (N = 66; C / ha range = 0.07 – 2.67) than tidal-fresh and oligohaline subestuaries (N = 22 and 32, respectively; Figure 3-5).

A total of 28,420 fish representing 42 species were captured by beach seining in 2018 (Table 3-4). Ten species comprised 90% of the total fish caught in 2018, including (from greatest to least) Atlantic Silverside, Mummichog, White Perch (juveniles), Striped Killifish, Gizzard Shad, Spottail Shiner, White Perch (adults), Inland Silverside, Blueback Herring, and Alewife. White Perch (juveniles and adults), Gizzard Shad, Spottail Shiner, Blueback Herring, and Alewife 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: Spottail Shiner were present in this category in two of the five subestuaries; White Perch (juveniles and/or adults) in five of the subestuaries; Striped Bass in two; Blueback Herring, Gizzard Shad, and Alewife in one. Notably absent from the top 90 % in 2018 were the usually common Bay Anchovy and Atlantic Menhaden.

Geometric mean catch for trawls were between 12 and 27 during 2018 (Table 3-5). All subestuaries had 24 samples in 2018. Langford Creek had the greatest GM (29) and Wye River had the lowest (14); Tred Avon River ranked 2nd; and Corsica River ranked, 3rd.

Number of species captured by trawl in subestuaries sampled during 2018 ranged from 7 to 15 (Table 3-5). A plot of species richness in trawl samples against C / ha (all subestuaries during 2003-2018) 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 ranging from 12 to 26; Figure 3-6). Species richness (ranging from 3 to 23) declined in mesohaline subestuaries as C / ha advanced beyond the threshold (C / ha = 0.86 = 10 % IS; Figure 3-6).

A total of 2,336 fish and 22 fish species were captured by trawling during 2018. Six species comprised 90% of the total catch for 2018 (from most to least): White Perch (adult and juvenile), Hogchoker, Spot, Oyster Toadfish, and Green Goby; White Perch and Spot were the only target species. Target species comprising 90% of the catch in each of the four subestuaries sampled during 2018 were White Perch (juvenile and-or adult) in four subestuaries; Spot in two; and Striped Bass (juvenile) and Bay Anchovy each in one subestuary (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 3-7). 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 (Chester River) fell below the target in two years (Table 3-6; Figure 3-8).

Moderate negative associations of surface and bottom dissolved oxygen (DO) with corresponding mean water temperatures at depth were detected for oligohaline subestuaries by correlation analyses (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. Associations of temperature and DO were weak in mesohaline or tidal-fresh subestuaries.

A moderate negative association between bottom DO and C / ha was found in mesohaline subestuaries; mesohaline subestuaries were where strongest stratification was expected. Remaining correlations were weak, although some were significant at $P < 0.03$. Given that multiple comparisons were made, correlations that were significant at $P < 0.03$ might be considered spurious if one rigorously adheres to significance testing (Nakagawa 2004; Anderson et al. 2000). 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 = 79) were over twice as high as oligohaline (Surface N = 41; Bottom N = 40) or tidal-fresh subestuaries (N = 34), so ability to detect significant associations in mesohaline subestuaries was greater (Table 3-7).

Bottom DO was negatively associated with development in mesohaline subestuaries (Table 3-7); remaining correlations were weak. Depletion of bottom DO in mesohaline subestuaries to below target 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 these target species being present in bottom channel trawl samples were negatively influenced by below target DO concurrent with development.

The extent of bottom channel habitat that can be occupied does not appear to diminish due to low DO with increasing watershed development in tidal-fresh and oligohaline subestuaries. However, more localized or episodic habitat issues appear to be important. 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). Unfortunately, it was not feasible for us to routinely monitor fish within the beds and the impact on target finfish could not be estimated. Ammonia toxicity that was potentially associated with high SAV coverage was suspected as a cause of boom and bust dynamics of trawl GMs in Mattawoman Creek during the 2000s (Uphoff et al. 2016). During November, 2015, the oligohaline Middle River subestuary experienced an extensive fish kill attributed to harmful algal blooms (MDE 2016).

Land Use Categories, C / ha, and Mesohaline Subestuary Bottom Dissolved Oxygen - Correlations of agriculture with C / ha or urban cover were negative and moderate to strong (r = -0.75; 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 = 0.90; P < 0.0001; Table 3-8). Correlation between forest cover and agriculture cover was negative and moderate (r = -0.58; P < 0.0001). Wetland cover and C / ha were negatively and weakly correlated (r = -0.27; P = 0.02). Remaining pairings of categories were not well correlated (Table 3-8).

After inspection of scatter plots, agricultural cover was further divided into regional categories reflecting lower percentages of forest cover on the eastern shore, east and west of Chesapeake Bay, for analyses with DO in mesohaline subestuaries (Figure 3-9). Two western shore sub-regions reflected agricultural coverage: subestuaries located on the western shore of Chesapeake Bay (Magothy, Rhode, Severn, and South rivers) fluctuated between 2.6 % to 34.1 % agricultural coverage, while lower Potomac River watersheds (Breton Bay, St. Clements, and Wicomico Rivers) ranged from 31.6 % to 38.6 % agricultural coverage. Eastern shore watersheds in the Choptank River drainage (Broad and Harris creeks, and Tred Avon River) ranged from 42.6 % to 50.1 % agricultural coverage. Mid-eastern shore watersheds (Chester, Corsica, Miles, Wye Rivers, and Langford Creek) ranged from 53.7 % to 71.6 % agricultural coverage.

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 % comprised entirely of western shore subestuaries (Figure 3-9). 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. Development was predominant at low levels of agriculture (< 20 %). Agricultural coverage and C / ha were 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 annual median subestuary DO with percent agriculture. The relationship was positive for the western shore (slope = 0.132; SE = 0.018; $r^2 = 0.729$; $P < 0.0001$; N = 21; Table 3-9) and negative for the eastern shore (slope = -0.037; SE = 0.010; $r^2 = 0.213$; $P = 0.0007$; N = 51; Table 3-9). Predictions of median DO for mesohaline western shore subestuaries rose from 0.42 mg / L at 2.6 % agricultural coverage to 5.27 mg / L at 38.6 % (Figure 3-2). Predictions of median DO for mesohaline eastern shore subestuaries fell from 5.43 mg / L at 42.6 % agricultural coverage to 4.34 mg / L at 71.6 %. A quadratic regression of median bottom DO versus agricultural coverage described the relationship of median bottom DO with agricultural coverage ($R^2 = 0.63$, $P < 0.001$; Table 3-10; Figure 3-9).

Mesohaline subestuaries sampled with bottom trawl in 2018 ranked relatively low compared to earlier years. The 2018 Corsica River GM ranked 70th out of 78 GMs; Langford Creek 67th; Tred Avon River 69th; and Wye River 75th (Table 3-11). The Chester River (2018) was not included because bottom trawls were not conducted.

Tred Avon River – Percentages of land in agriculture (42-45%), forest (19–25%), and urban (29–34%) categories were similar among the three Choptank River subestuaries (MD DOP 2010; Table 3-12; Figure 3-2); 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 Creek and Tred Avon River (57% and 62%, respectively) than Broad Creek (24%; 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 was subjected to more development than Broad Creek and Harris Creek watersheds (Figure 3-10) and more than indicated by the Maryland Department of Planning urban category (Table 3-12; Figure 3-2). Time-series for both watersheds started at a rural level of development (C / ha ranged from 0.1 to 0.2) in 1950. Harris Creek watershed has passed the rural development target (C / ha = 0.37), while Broad Creek is still under the rural development target (C / ha was 0.29 in 2014). More growth occurred in Tred Avon River's watershed (C / ha = 0.76 in 2014; Figure 3-10). Development accelerated noticeably in the Tred Avon River watershed during 1999-2007 and then slowed. Tred Avon River's watershed has been approaching the suburban threshold, C / ha = 0.86;

During 2018, bottom DO readings below the threshold (3.0 mg / L) and target (5.0 mg / L) were frequent in the Tred Avon River; 50% of bottom DO samples were below the target and 14% were below the threshold (Table 3-13). During 2006-2018, 7% of bottom DO measurements from Tred Avon River were below the DO threshold and 34% were below the DO target (Figure 3-11). Less than 1% of Broad Creek bottom DO measurements during 2012 to 2017 were below the threshold and 14% were below the target. During 2012-2016, Harris Creek did not have bottom DO measurements below the threshold and 2.5% were below the target (Table 3-13; Figure 3-11).

Median bottom DO did not fluctuate substantially from year to year in the three Choptank River subestuaries. Median bottom DO in the Tred Avon River ranged from 4.9 mg / L (2018) to 6.3 mg / L (2009; Figure 3-12). Median bottom DO in Broad Creek ranged from 5.65 mg / L (2012) to 6.64 mg / L (2015). Median bottom DO in Harris Creek ranged from 5.79 mg / L (2013) to 6.39 mg / L (2015; Figure 3-12). Correlations of median bottom DO with year and among Choptank subestuaries were modest to low and trends were not considered meaningful (Table 3-14).

An ANOVA of Tred Avon River stations and bottom DO during 2006-2018 indicated significant differences among stations (F = 44.93; DF = 3; $P < 0.0001$; N = 311). 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 2-4. This decline in bottom DO with upstream distance was consistent with other mesohaline tributaries with high impervious surface (Uphoff et al. 2011a). The mean and SE for bottom DO at all stations in Tred Avon River were 5.28 mg / L and 0.08, respectively. Mean and SE for bottom DO at station 1 were 3.89 mg / L and 0.19; station 2 was 5.68 mg / L and 0.12; station 3 was 5.78 mg / L and 0.11; and station 4 was 5.82 mg / L and 0.11. Deterioration of DO at the uppermost station (station 1; Figure 3-3) since 2012 indicated that increased watershed development around Easton was the source of poor water quality rather than water intruding from downstream. During 2018, mean bottom DO at station 1 was below the threshold and target values and the overall median for the Tred Avon River time-series. Stations 3 and 4 had mean bottom DO above the overall median for the time-series during 2018 and mean DO was above the target. Station 2 fell slightly below the overall median DO, but remained above the target level (= 5.0 mg / L; Figure 3-13).

An ANOVA of Broad Creek (F = 0.9; DF = 3; $P = 0.4434$; N = 138) station bottom DO measurements did not indicate significant differences among stations during 2012-2017. The overall mean and SE for bottom DO in Broad Creek were 6.06 mg / L and 0.09, respectively. Annual station means varied without trend around the time-series median for all sites (Figure 3-13).

An ANOVA of Harris Creek (F = 1.63; DF = 3; $P = 0.187$; N = 117) station bottom DO measurements did not indicate significant differences among stations during 2012-2016. The overall mean and SE for bottom DO in Harris Creek were 6.21 mg / L and 0.07, respectively. Similar to Broad Creek, annual station mean DO varied without trend around the time-series median for all sites (Figure 3-13).

We ranked the bottom trawl GMs for all species combined in each of the Choptank subestuaries sampled during 2006-2018 (Table 3-15). Tred Avon River was the only Choptank River subestuary sampled in 2018 and it ranked at the very bottom, 24th out of 24 surveys (Table 3-15). The GMs for Broad Creek, Harris Creek, and Tred Avon

River in 2014 all ranked in the top 25 % and 2012 ranked in the top 30 %. The remaining years were scattered with no real pattern (Table 3-15).

Annual GMs of catches of all species of finfish in 4.9 m bottom trawls in Broad Creek, Harris Creek, and Tred Avon River for all sampling years and their 95 % CIs were plotted (Figure 3-14). The greatest GM (266) in Tred Avon River occurred in 2010 and 2018 had the lowest GM (20). Broad Creek GMs ranged from 106 (2015) to 402 (2014) and Harris Creek GMs ranged from 41 (2015) to 176 (2014; Figure 3-14).

Correlations of trawl GMs with year did not suggest meaningful time trends in relative abundance for the three Choptank River subestuaries (Table 3-16). Correlations of trawl GMs among the three Choptank River subestuaries did suggest coherence in annual relative abundance of finfish (Table 3-16). Strong correlations of GMs were present between Broad Creek and Harris Creek (r = 0.956, *P* = 0.011, N = 5); Broad Creek and Tred Avon River (r = 0.951, *P* = 0.004, N = 6); and Tred Avon River and Harris Creek (r = 0.842, *P* = 0.07, N = 5; Table 3-16).

Five species were in the top 90 % of finfish caught in the Tred Avon River during 2006-2018: Bay Anchovy (58.8 %), Spot (16.5 %), Hogchoker (7 %), White Perch (adults; 4.6 %), and Weakfish (3.5 %; Figure 3-15). An additional 36 species comprised the other category (Figure 3-15). All species in the top 90 %, except Hogchoker, were target species.

Species richness in the top 90 % of species collected in Tred Avon River trawl samples increased in 2018 concurrent with the large drop in relative abundance of all finfish (Figure 3-16). The usually common Bay Anchovy dropped out of the top 90 % during 2018. Percent similarity in finfish species composition among stations 1 – 4 in the Tred Avon River was lowest in 2018 (20 %; Figure 3-16) indicating a shift in finfish composition among years. In 2016, Tred Avon River had the greatest percent similarity index in finfish species in bottom trawls among stations 1–4 (87 %). The similarity index was above 50 % from 2007 to 2017. During 2006 and 2018, the similarity index was below 50 %, reflecting rainfall and salinity (Figure 3-17). The correlation of percent similarity and median precipitation was negative (r = -0.65; *P* = 0.0154; N = 13) and stronger than the positive association of similarity and median salinity (r = 0.52; *P* = 0.07; N = 13), suggesting wet years with lower salinity would have species composition dissimilar to dry years with higher salinity.

We analyzed finfish species composition of bottom trawls in all mesohaline subestuaries sampled during 2003-2018 to see if changes in Tred Avon River in 2018 were unique. A similar change in finfish composition for all mesohaline systems was observed; Bay Anchovy dropped out of the top 90 % of species during 2018 (Figure 3-18). There was an increase in the number of species in the top 90 % that reflected the scarcity of this usually common forage fish (Figure 3-18). Mesohaline subestuaries sampled from 2003 to 2018 changed by year and some differences could reflect these changes.

The Tred Avon River adult White Perch trawl GM fell below the median time-series GM (4) in 2010, 2014, and 2016 (Figure 3-19). The greatest White Perch GM in Tred Avon River was in 2012 (14) and the least was in 2010 (2). During 2016, White Perch GMs in Broad and Harris Creeks and Tred Avon River were similar (4; Figure 3-19). The median for the time-series in Broad Creek was 3 and 2.5 in Harris Creek. A moderate negative correlation of White Perch GMs with year was present for Broad

Creek (Table 3-17); correlation analysis indicated weak negative correlations with year in Harris Creek and Tred Avon River. White Perch GMs in Broad Creek were moderately and positively correlated with adjacent Tred Avon River's GMs. Remaining correlations of White Perch GMs among subestuaries were weakly positive (Table 3-17).

Finfish seine GMs in the three Choptank subestuaries were highest during 2015, indicating an increase in finfish throughout the Choptank system (Figure 3-20). Seine GMs for all finfish in Tred Avon River samples were lowest in 2008 (77). Broad Creek and Harris Creek had their lowest GMs 2012 (106 and 131, respectively). The 2012 GM in Tred Avon River was also low in 2012 (Figure 3-20); 2012-2016 represented years in common among these three subestuaries.

Seven species, Atlantic Silverside (38.3 %), Atlantic Menhaden (18.8 %), White Perch (adults; 10.9 %), Striped Killifish (7.7 %), White Perch (juveniles; 6.5 %), and Bay Anchovy (3 %), were in the top 90 % of finfish caught in the Tred Avon River (Figure 3-21). An additional 41 species were considered other species (Figure 3-21). All species in the top 90 % of all the subestuaries, except Atlantic Silverside, Mummichog, and Striped Killifish were target species.

Tred Avon River median Secchi depths ranged from 0.5 m to 0.7 m during 2006-2018; from 0.6 m to 0.9 m in Broad Creek during 2012-2017; and from 0.5 m to 1.1 m in Harris Creek during 2012-2016 (Figure 3-22). Median Secchi depths were strongly and positively correlated with years for Broad Creek and Harris Creek (r = 0.86 and 0.85, respectively) and weakly correlated for Tred Avon River (Table 3-18). This analysis indicated that clarity had improved over the Broad Creek and Harris Creek time-series, but improvement was not suggested for Tred Avon River. The three Choptank subestuaries Secchi depths were strongly correlated with each other (Table 3-18).

Tred Avon River, Broad Creek, and Harris Creek SAV coverage were included in the mouth of the Choptank River region (VIMS 2018). SAV coverage increased substantially from 1% in 2012 to 11.8% in 2017 (Figure 3-23) and was far above the time-series median (4%) in 2017 (Figure 3-23). An estimate for 2018 was not available.

Median pH in Tred Avon River ranged from 7.4 (2007) to 7.9 (2012 and 2013; Figure 3-24). Broad Creek median pH ranging from 7.8 (2014) to 8.1 (2015). Harris Creek median pH ranging from 7.7 (2013-2014) to 8 (2015; Figure 3-24). Correlations of pH with year for the three subestuaries did not suggest a trend (Table 3-19). Median pH in Broad Crek and Harris Creek were strongly correlated, but remaining combinations were not (Table 3-19).

Tred Avon River had its second lowest median salinity in 2018 (Figure 3-25). Tred Avon River had its highest median salinity in 2016 (12.8 ‰) and the lowest in 2011 (7.5 ‰). Low salinity in 2011 was not accompanied by the complete loss of Bay Anchovy as it was in 2018 (see Figure 3-16). Broad Creek (2012-2017) had the greatest median salinity in 2016 (13.6 ‰) and the lowest in 2013 (10.2 ‰). Harris Creek (2012-2016) had the greatest median salinity in 2016 (13.6 ‰) and the lowest in 2014 (10.0 ‰; Figure 3-25). Correlations of median salinity with year for Broad Creek, Harris Creek, or Tred Avon River were weak did not suggest meaningful trends (Table 3-20). However, median salinities of all three Choptank subestuaries were positively and strongly correlated among each other; these strong correlations among these subestuaries reflect their proximity to each other (Table 3-20).

Mean monthly precipitation in Talbot County was high during late spring and summer in 2018 (NCDC 2019; Figure 3-26) and was above the 5-year mean for 8 of 12 months. Mean monthly precipitation in the most recent 5 years was not very different than the 25-year average (Figure 3-26). Talbot County received less precipitation than other regions around Chesapeake Bay.

In 2018, finfish trawl catches in the Tred Avon River bottom channel were at their lowest level, while inshore seine catches were average to good. There was little indication that low DO was more widespread than usual, nor did the other water quality measurements offer an obvious connection to changes in finfish abundance. Typically, low finfish catches in the bottom channel within mesohaline systems are associated with development and low DO measurements. Salinity was lower, but not the lowest that has been recorded for the time-series available for Tred Avon River. The Tred Avon River trawl GM in 2018 was the lowest and reflected a large decline in Bay Anchovy. A similar decline in Bay Anchovy presence appeared in mesohaline systems sampled during 2004-2005 (mesohaline systems sampled during 2004-2005 are listed in Table 3-2). An extreme change in the species present and richness in bottom trawl catches in 2018 was notable for Tred Avon River; other mesohaline systems saw a dramatic shift in species composition in bottom trawl catches in 2018 as well. Tred Avon River seine GM in 2018 was average. Anecdotally, we noted lots of small, empty clam shells were present in bottom trawls throughout Tred Avon River, as well as un-decayed leaves in both trawls and seines that may suggested episodic ecosystem disruption may have occurred in 2018.

Queen Anne's County Subestuaries - Estimated percentages of watersheds in agriculture (60% - 70%), forest (20% - 25%), urban (8% - 13%), and wetlands (0.1% - 2%) were similar for the Queen Anne's County subestuaries (MD DOP 2010; Table 3-21; Figure 3-2). Water comprised a larger fraction of the Chester River drainage (17.5%) than in Langford Creek and Wye River (11.9% and 11.6%). Corsica River (5.5%; MD DOP 2010) had the lowest fraction of water coverage (Table 3-21).

Tax map estimates of C / ha indicated that the Corsica River has been subject to more development than Chester River, Wye River, and Langford Creek (Figure 3-27) and more than indicated by the Maryland Department of Planning urban category (Table 3-21; Figure 3-2). Time-series for all subestuaries started at a rural level of development (C / ha ranged from 0.01 to 0.05) in 1950 (Figure 3-27). Langford Creek's watershed has experienced the lowest growth (C / ha = 0.07 in 2014), while the most growth occurred in Corsica River's watershed (C / ha = 0.27 in 2014). Wye River development steadily increased until the mid-2000s and has hovered at 0.10 since then. Development accelerated noticeably in the Corsica River watershed in 2002, and still appears to be increasing. Both the Chester River and Wye River showed increasing development until 2007 when development may have stabilized, possibly reflecting the Great Recession (Figure 3-27). All subestuaries are below the rural development target (IS 5 % = 0.37); however, Corsica River is the closest to breaching that target (C / ha = 0.27 in 2016).

In 2018, bottom DO readings breaching the threshold (3.0 mg / L) and target (5.0 mg / L) were most frequent in the Wye River (15% and 59%, respectively; Table 3-22). Chester River and Langford Creek did not have threshold violations, and Corsica River had 9 % of bottom DO readings violate the threshold (Figure 3-28). Bottom DO target violations during 2018 for the Chester River were 5%; Corsica River, 35%; and Langford

Creek, 18%. Corsica River had threshold and target violations every year bottom DO was sampled; 64% of bottom DO measurements in Corsica River (2003-2012, 2018) were below the DO target and 24% were below the DO threshold. Chester River had threshold violations 5 years out of 7 years and target violations every year; 58% were below the target and 7% were below the threshold. Langford Creek (2006-2008, 2018) had threshold violations only in 2007 and target violations every year. Overall in Langford Creek, 32% of bottom DO measurements were below the target and 1% were below the threshold. Wye River (2007-2008 and 2018) had threshold violations only in 2018 and target violations every year; over all three years, 45% were below the target and 6% were below the threshold (Table 3-22; Figure 3-28).

Median bottom DO estimates ranged from 4.4 mg / L to 5.7 mg / L for the Chester River during 2007-2012 and 2018 (Figure 3-29). Corsica River had the greatest change in median bottom DO (2003-2008, 2010-2012, and 2018), from 2.9 mg / L in 2012 to 5.3 mg / L in 2018. Langford Creek median bottom DO estimates ranged from 6.1 mg / L in 2006 to 4.8 mg / L in 2008, and was 5.8 in 2018. Median bottom DO estimates ranged from 4.6 mg / L to 6.1 mg / L for the Wye River during 2007-2008 and 2018. Wye River was the only Queen Anne's County subestuary to not exhibit an increase in bottom DO between 2008 and 2018 (Figure 3-29). Correlation analyses did not indicate meaningful trends over time (Table 3-23). Strong correlations of median DO between Chester River and Corsica River and Chester River and Langford Creek were present that may have indicated an influence of mainstem waters on DO conditions in these tributaries. Corsica River and Langford Creek are longitudinally aligned and separated by the mainstem of the Chester River. Remaining pairings were weakly correlated (Table 3-23).

In 2018, Wye River had the greatest percentage of all DO measurements (surface to bottom) below target (5.0 mg / L), 27%; followed by Corsica River, 26%; Langford Creek, 6%; and Chester River, 3% (Table 3-22). Frequency of all DO violations were lower in 2018 than previous years for all subestuaries. Chester River had 4 of 7 years with target violations above 50% for all DO measurements; Corsica River had 2 of 10 years above 50%. Langford Creek and Wye River did not have above 50% target violations (Table 3-22).

In 2018, mean bottom DO at all stations of the Chester River and Corsica River were above the median of all years sampled (Figure 3-30). Chester River bottom DO measurements were only recorded in the middle of the channel at three site locations in 2018: sites 01, 03, and 06 (N = 19; Figure 3-4). Langford Creek and Wye River mean bottom DO at all stations were near or below the overall median DO (Figure 3-30).

ANOVAs were used to detect differences in mean bottom DO among stations in the each of the Queen Anne's County subestuaries. Chester River ANOVAs contained only bottom DO data for stations sampled from 2007 to 2012; 2018 was omitted due to its different sampling routine. The ANOVAs for site comparisons for each subestuary were not significant; site differences in mean bottom DO were not detected in Chester River, Corsica River, Langford Creek, or Wye River.

The overall mean and SE for bottom DO in Chester River during 2007-2012 and 2018 were 4.80 mg / L and 0.08, respectively. The overall mean and SE for bottom DO in Corsica River for years 2003-2012 and 2018 were 4.18 mg / L and 0.13, respectively. The overall mean and SE for bottom DO in Langford Creek for years 2006-2008 and

2018 were 5.92 mg / L and 0.24, respectively. The overall mean and SE for bottom DO in Wye River for years 2007-2008 and 2018 were 5.14 mg / L and 0.19, respectively.

We ranked the 4.9 m bottom trawl GMs of all species combined from Chester River mainstem, Corsica River, Langford Creek, and Wye River during 2003-2018 (Table 3-24). Chester River was not sampled by bottom trawl in 2018 and was not included. Corsica River had the highest ranked GM (378 in 2003), followed by Langford Creek at 273 (2007), and Chester River at 259 (2011). The three 2018 GMs ranked at the bottom (Table 3-24). Annual GM catches per 4.9 m bottom trawl of all species of finfish in the Chester River, Corsica River, Langford Creek, and Wye River and their 95 % confidence intervals (CI) were plotted on Figure 3-31. Correlations of GMs for subestuaries sampled through 2018 (Corsica River, Langford Creek, and Wye River) against year were moderately to strongly negative ($r = -0.62$ to -0.96), but sample sizes were low for some comparisons. These trends were heavily influenced by poor catches in 2018 (Table 3-25). Given the aberrantly high rainfall in 2018, these declining correlations with time may not reflect a true trend. Analysis of Chester River sampling through 2012 did not indicate a trend. Correlations among Chester River, Corsica River, Langford Creek, or Wye River; Corsica River annual GMs were positive and strong, but sample sizes were low for some comparisons (Table 3-25).

Chester River bottom trawl catches were composed of White Perch adults (44.4%), White Perch juveniles (33.5%), Spot (12.2%), Bay Anchovy (3.7%), and other species (27 species; 6.1%; Figure 3-32). Four species defined the top 90% of finfish caught in the Corsica River, White Perch (adults; 39.8%), White Perch (juveniles; 31.4%), Bay Anchovy (17.1%), and Spot (7.6%). The other species category included 23 additional species, comprising of 4.1% of the finfish catch. Langford Creek bottom trawl catches were composed of White Perch (adults; 65.9%), Bay Anchovy (19.9%), Spot (8.0%), and other species (21 species; 6.3%; Figure 3-32).

Annual finfish composition for Chester and Corsica Rivers, and Langford Creek bottom trawl catches did not indicate a shift in species composition in 2018 in response to high rainfall (Figure 3-33). Annual finfish composition for Wye River bottom trawl catches did undergo a shift in species composition; adult white perch dropped out of the top 90% in 2018 and were replaced by Brown Bullhead, Green Goby, and White Catfish. White Perch (juveniles and adults) make up the top 90 % of species present in Chester and Corsica Rivers, and Langford Creek. Chester River had adult White Perch and Spot present in the top 90 % of species. Wye River had Bay Anchovy, Brown Bullhead, Green Goby, Spot, Striped Bass, White Catfish, and juvenile White Perch in the top 90 % of species in 2018 (Figure 3-33).

Beach seine catch GMs for the Chester River ranged from 52 (2000) to 350 (1994; Figure 3-34). Corsica River had its lowest finfish seine GM in 2012 (74) and the greatest finfish GM in 2003 (775). Langford Creek exhibited greatest finfish seine GM most in 2018 (237) and lowest in 2006 (60). Seine catch GMs for the Wye River ranged from 79 (2008) to 182 (2018; Figure 3-34). Seine catch GMs in 2018 were high in Langford Creek and Wye River, and were mid-range in Chester River and Corsica River (Figure 3-34).

Chester River seine catches had 8 species in the top 90%, Atlantic Silverside (32.8%), White Perch (adults; 15.3%), White Perch (juveniles; 13.5%), Bay Anchovy (6.4%), Atlantic Menhaden (5.9%), Mummichog (5.2%), Spottail Shiner (4.7%), Blueback Herring (3.3%), Striped Killifish (3%), and other species (41 species; 10%). Eight species defined the top 90% of finfish caught in the Corsica River: Atlantic Silverside (36.8%), Mummichog (13.4%), White Perch (adults; 11.1%), Blueback Herring (7.9%), White Perch (juveniles; 6.9 %), Striped Killifish (6.5%), Atlantic Menhaden (4.4%), Spottail Shiner (4%). The other species category included 24 additional species, comprising of 9% of the finfish catch. Langford Creek seine catches were comprised of Atlantic Silverside (36.4%), Atlantic Menhaden (18.4%), White Perch (adults; 15%), Striped Killifish (6.8%), Blueback Herring (6.4%), Alewife (4.2%), Mummichog (3.3%), and other species (25 species; 9.5%). Wye River seine finfish catches included Atlantic Silverside (33.9%), White Perch adults (18.6%), Atlantic Menhaden (18.1%), Mummichog (10.4%), Striped Killifish (8.0%), Bay Anchovy (2.3%), and other species (24 species; 8.8%; Figure 3-35). All species in the top 90% of all the subestuaries, except Atlantic Silverside, Mummichog, and Striped Killifish, were considered target species. One notable difference between Wye River and Chester River, Corsica River, and Langford Creek was the consistent inclusion of Blueback Herring in the top 90% in the latter three subestuaries; upper Chester River is a spawning area for anadromous herring.

Corsica River had the greatest median Secchi depth range among the Queen Anne's County subestuaries, 0.4 m to 0.7 m; median Secchi depths for all but one year (2004) ranged between 0.4 and 0.5 m (Figure 3-36). Median Secchi depths in the remaining subestuaries were between 0.4-0.5 (Figure 3-36). Secchi data was not available for the Chester River. Correlation analyses of median Secchi depth against year did not suggest strong linear trends, except in Wye River; sample size was low for this comparison (N = 3) and the correlation was not likely meaningful (Table 3-26). The two Chester River tributaries, Corsica River and Langford Creek, were not significantly correlated. Corsica River and Wye River had a positive, strong correlation, but the sample size was too low (N = 3) to support a conclusion (Table 3-26).

Coverage of water area in SAV varied among subestuaries. Chester River SAV coverage included all segments (upper, middle, and lower) of the river. Chester River SAV coverage ranged between 0.4% and 2.3% during 1989-2017 (Figure 3-37). Coverage in 2017 (1.4%) was above the median of the time-series (0.4%). Coverage data were not available for 2003. Coverage of SAV in Eastern Bay included Miles and Wye Rivers, and varied between 0% and 8% from 1989 to 2017. In 2017, SAV coverage (4%) was above the median of the time-series (2.5%; Figure 3-37). Data that were only partially mapped or not mapped at all were not included in this assessment.

Estimates of pH for Chester River, Corsica River, Langford Creek, and Wye River were limited due to limitations of water quality equipment used before 2006. Only one year of pH data was available for the Chester River (2018); median pH was 7.4 and ranged from 7.4 to 7.5 (Figure 3-38). Corsica River median pH ranged from 7.5 (2018) to 7.7 (2006). Langford Creek median pH ranged from 7.5 (2007 and 2008) to 7.9 (2006). Wye River median pH ranged from 7.6 (2018) to 7.8 (2007 and 2008; Figure 3-38). Median pH estimates in Corsica River and Wye River were strongly and negatively associated with years, but sample sizes were low (N = 3-4; Table 3-27). Median pH in Langford Creek and Corsica River were strongly correlated with Wye River (although the signs of the correlations were opposite), but sample sizes were too low to consider these correlations meaningful (Table 3-27).

Median salinity fluctuated substantially among years and subestuaries. All subestuaries sampled had a lower median salinity in 2018, reflecting greater precipitation. Chester River median salinity ranged from 1.38 ‰ (2018) to 7.5 ‰ (2007; Figure 3-39). The Chester River is normally a mesohaline system (5.0 ‰ – 18.0 ‰), but oligohaline median salinities were estimated in 2011 and 2018. Median salinity was greatest for Corsica River in 2012 (9.6 ‰) and lowest in 2003 and 2011 (4.5 ‰). Langford Creek median salinity ranged from 5.7 ‰ (2018) to 9.3 ‰ (2007). Wye River annual median salinity ranged from 8.1 ‰ (2018) to 11.7 ‰ (2007; Figure 3-39).

Correlations of median salinity with year indicated moderate to strong declines in Chester River, Langford Creek, and Wye River, but not in Corsica River (Table 3-28). Correlations of median salinity estimates among the four Queen Anne's County subestuaries were positive and strong, but sample sizes were small for some comparisons. These strong correlations indicated similar influences could be present.

Due to the extreme change in salinity within the Chester River in 2011 and 2018, we examined the correlation of salinity and DO measurements from Chester River. The correlation was very weak ($r = -0.02$; $P = 0.74$; $N = 226$; Figure 3-40) and changes in DO were unlikely to reflect a change in salinity.

Mean monthly precipitation for Kent and Queen Anne's Counties increased during late spring and summer, 2018; monthly precipitation in May, July, and September ranked the highest over a 5 year span (NCDC 2019; Figure 3-41). The 5 year mean trend for summer for Kent and Queen Anne's Counties was at or above the 25 year mean trend from May to September, indicating that the last 5 years on average have experienced greater amounts of precipitation (Figure 3-41).

High precipitation occurred during spring and summer 2018 in Kent, Queen Anne's, and Talbot Counties. Finfish trawl catches from the bottom channel were at their lowest level, while inshore beach seine catches were average to good. There was little indication that DO was a serious issue throughout the water column. Typically, low finfish catches in the bottom channel within mesohaline systems are associated with development and low DO measurements. Salinity was lower, but not necessarily the lowest that was been recorded for the time-series available for some subestuaries.

Exploration of relative abundance of finfish in Chester River, Tred Avon River, Choptank River, and Head-of-Bay seine samples – Correlations of Head of Bay and Choptank River annual beach seine catch GMs of all finfish were weak ($r = 0.15$; $P = 0.25$; $N = 60$; Table 3-29). Plots of the annual GM of catches of all species combined in Head of Bay and Choptank River indicated an interesting switch around 1980; magnitude of Head of Bay GMs and Choptank River GMs were similar prior to the switch and Choptank River GMs were higher afterward (Figure 3-42a). Correlations were not significant for GMs of all finfish in the Head of Bay and Chester River ($r = -0.11$; $P = 0.63$; $N = 21$). Annual GM of the Chester River (all species combined) was not coherent with the Head of Bay system during years, 1959, 1960, and 1987 (Figure 3-42b). Trends in Chester River appeared to represent internal production rather than spillover from adjacent, major subestuaries. However, during 2007-2018, the annual GM appeared to rise and fall in unison with the Head-of-Bay. An additional correlation analysis using only the annual GMs for 2007-2018 in the Chester River indicated a strong positive association with the Head of Bay system ($r = 0.85$; $P = 0.02$; $N = 7$). This strong correlation could indicate greater synchrony of conditions influencing finfish production between the two systems or supplementation of Chester River production from the larger Head-of-Bay region.

The correlation between Head of Bay and Tred Avon River annual GM of catches of all finfish (r = 0.50; P = 0.08; N = 13) was marginally moderate. A strong positive association was present between the Choptank system and Tred Avon River annual GM (r = 0.84; P = 0.0003; N = 13; Table 3-29). Tred Avon River GMs (all species) likely reflected abundance in the Choptank River. Seine GMs in the Tred Avon River and Choptank River appeared very similar (Figure 3-42c). Future analysis of Juvenile Index seine survey data will focus primarily on anadromous fish within the systems.

Summary – High precipitation in 2018 did not have an overwhelming impact on survey water quality measurements. The increase in rainfall in 2018 caused a decline in salinities, possibly altering the composition of finfish and shifting the migratory range finfish are known to inhabit. Salinities in most subestuaries sampled were at the lower bounds of what had been observed during previously, but remained within their salinity class. Chester River was an exception; salinity dropped enough in 2018 for it to fall into the oligohaline class. Bottom DO conformed to their expected relationships to level of development and salinity class. Queen Anne's County watersheds all were at or below the target level of development. Bottom DO in 2018 was most likely to be above the target level and below threshold measurements were uncommon in Chester River and its two tributaries. Most bottom DO measurements in Wye River fell between the target and threshold level, but below threshold readings were much more common in 2018 than previous surveys. Frequency of below threshold bottom DO held steady at the level estimated (13% -14%) since 2015 in Tred Avon River (this watershed is approaching the development threshold), but below target DO became more frequent. Other water quality metrics (pH and Secchi depth) in the subestuaries sampled during 2018 were within previous years' ranges. Finfish catches in trawls sampling bottom water habitat declined among all subestuaries sampled. A drop in trawl GMs was common among the subestuaries sampled and did not reflect bottom DO (except in the upper most station in Tred Avon River). Species composition changed, reflecting of a drastic drop of Bay Anchovy and a concurrent substitution of species that would have fallen into the "other species" category had Bay Anchovy abundance not fallen. Inshore seine catches were within a normal range. While it appears that heavy rainfall and high freshwater discharge into the Chesapeake Bay and its tributaries did not have an uniformly aberrant impact during 2018, we plan to re-sample these systems in 2019 to be sure that our assessment of habitat, particularly the subestuaries sampled for the Queen Anne's County comprehensive growth plan, was not inadvertently biased.

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Tables

Table 3-1. Percent impervious cover (IS), structures per hectare (C / ha), watershed area (land hectares), area of tidal water (water hectares), and salinity class for the subestuaries sampled in 2018.

Area	Subestuary	IS	C/ha	Land Hectares	Water Hectares	Salinity Class
Upper-Bay	Bush River	14.4	1.53	36,002	3,640	Tidal-Fresh*
Mid-Bay	Chester River	3.3	0.15	99,542	21,084	Oligohaline*
Mid-Bay	Corsica River	4.8	0.27	9,671	565	Mesohaline
Mid-Bay	Langford Creek	2.1	0.07	9,631	1,306	Mesohaline
Mid-Bay	Tred Avon River	9.3	0.77	9,561	3,086	Mesohaline
Mid-Bay	Wye River	2.5	0.10	20,395	2,670	Mesohaline

* Bush River is typically classified as a oligohaline subestuary and Chester River is typically a mesohaline subestuary. Salinity data taken in 2018 indicated salinity levels less than 0.5 ‰ for tidal-fresh and less than 5.0 ‰ for oligohaline.

Table 3-2. Estimates of C / ha and land use percentages from Maryland Department of Planning (2002 and 2010) for subestuaries sampled 2003-2018.

River	Year	C/ha	Agriculture	Wetland	Forest	Urban
Breton Bay	2003	0.27	23.8	0.8	56.1	18.7
Breton Bay	2004	0.28	23.8	0.8	56.1	18.7
Breton Bay	2005	0.30	23.8	0.8	56.1	18.7
Broad Creek	2012	0.29	42.6	0.4	25.4	31.5
Broad Creek	2013	0.30	42.6	0.4	25.4	31.5
Broad Creek	2014	0.30	42.6	0.4	25.4	31.5
Broad Creek	2015	0.30	42.6	0.4	25.4	31.5
Broad Creek	2016	0.30	42.6	0.4	25.4	31.5
Broad Creek	2017	0.30	42.6	0.4	25.4	31.5
Bush River	2006	1.41	25.4	3.2	35.0	36.2
Bush River	2007	1.43	25.4	3.2	35.0	36.2
Bush River	2008	1.45	25.4	3.2	35.0	36.2
Bush River	2009	1.46	25.4	3.2	35.0	36.2
Bush River	2010	1.47	18.0	3.2	29.9	47.8
Bush River	2011	1.48	18.0	3.2	29.9	47.8
Bush River	2012	1.49	18.0	3.2	29.9	47.8
Bush River	2013	1.51	18.0	3.2	29.9	47.8
Bush River	2014	1.52	18.0	3.2	29.9	47.8
Bush River	2015	1.52	18.0	3.2	29.9	47.8
Bush River	2016	1.53	18.0	3.2	29.9	47.8
Bush River	2017	1.53	18.0	3.2	29.9	47.8
Bush River	2018	1.53	18.0	3.2	29.9	47.8
Chester River	2007	0.14	66.5	2.0	25.8	5.8
Chester River	2008	0.14	66.5	2.0	25.8	5.8
Chester River	2009	0.15	66.5	2.0	25.8	5.8
Chester River	2010	0.15	64.2	2.0	24.7	8.9
Chester River	2011	0.15	64.2	2.0	24.7	8.9
Chester River	2012	0.15	64.2	2.0	24.7	8.9
Chester River	2018	0.15	64.2	2.0	24.7	8.9
Corsica River	2003	0.17	64.3	0.4	27.4	7.9
Corsica River	2004	0.18	64.3	0.4	27.4	7.9
Corsica River	2005	0.19	64.3	0.4	27.4	7.9
Corsica River	2006	0.21	64.3	0.4	27.4	7.9
Corsica River	2007	0.22	64.3	0.4	27.4	7.9
Corsica River	2008	0.24	64.3	0.4	27.4	7.9
Corsica River	2010	0.24	60.4	0.1	25.5	13.2
Corsica River	2011	0.25	60.4	0.1	25.5	13.2
Corsica River	2012	0.25	60.4	0.1	25.5	13.2
Corsica River	2018	0.27	60.4	0.1	25.5	13.2
Gunpowder River	2009	0.72	30.6	1.0	32.1	35.6
Gunpowder River	2010	0.72	30.6	1.0	32.1	35.6
Gunpowder River	2011	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2012	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2013	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2014	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2015	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2016	0.73	30.6	1.0	32.1	35.6

Table 3-2 (Cont).

Harris Creek	2012	0.39	44.9	5.6	19.7	29.8
Harris Creek	2013	0.39	44.9	5.6	19.7	29.8
Harris Creek	2014	0.39	44.9	5.6	19.7	29.8
Harris Creek	2015	0.39	44.9	5.6	19.7	29.8
Harris Creek	2016	0.39	44.9	5.6	19.7	29.8
Langford Creek	2006	0.07	71.6	1.5	23.0	3.9
Langford Creek	2007	0.07	71.6	1.5	23.0	3.9
Langford Creek	2008	0.07	71.6	1.5	23.0	3.9
Langford Creek	2018	0.07	70.2	1.5	20.4	8.0
Magothy River	2003	2.68	2.6	0.0	27.8	69.5
Mattawoman Creek	2003	0.76	11.9	1.2	59.4	27.4
Mattawoman Creek	2004	0.79	11.9	1.2	59.4	27.4
Mattawoman Creek	2005	0.81	11.9	1.2	59.4	27.4
Mattawoman Creek	2006	0.83	11.9	1.2	59.4	27.4
Mattawoman Creek	2007	0.86	11.9	1.2	59.4	27.4
Mattawoman Creek	2008	0.87	11.9	1.2	59.4	27.4
Mattawoman Creek	2009	0.88	11.9	1.2	59.4	27.4
Mattawoman Creek	2010	0.90	9.3	2.8	53.9	34.2
Mattawoman Creek	2011	0.91	9.3	2.8	53.9	34.2
Mattawoman Creek	2012	0.90	9.3	2.8	53.9	34.2
Mattawoman Creek	2013	0.91	9.3	2.8	53.9	34.2
Mattawoman Creek	2014	0.93	9.3	2.8	53.9	34.2
Mattawoman Creek	2015	0.93	9.3	2.8	53.9	34.2
Mattawoman Creek	2016	0.93	9.3	2.8	53.9	34.2
Middle River	2009	3.30	4.5	2.2	27.9	63.9
Middle River	2010	3.32	3.4	2.1	23.3	71.0
Middle River	2011	3.33	3.4	2.1	23.3	71.0
Middle River	2012	3.33	3.4	2.1	23.3	71.0
Middle River	2013	3.34	3.4	2.1	23.3	71.0
Middle River	2014	3.35	3.4	2.1	23.3	71.0
Middle River	2015	3.36	3.4	2.1	23.3	71.0
Middle River	2016	3.38	3.4	2.1	23.3	71.0
Middle River	2017	3.38	3.4	2.1	23.3	71.0
Miles River	2003	0.24	53.7	0.9	27.2	18.1
Miles River	2004	0.24	53.7	0.9	27.2	18.1
Miles River	2005	0.24	53.7	0.9	27.2	18.1
Nanjemoy Creek	2003	0.08	15.1	4.1	73.1	7.6
Nanjemoy Creek	2008	0.09	15.1	4.1	73.1	7.6
Nanjemoy Creek	2009	0.09	15.1	4.1	73.1	7.6
Nanjemoy Creek	2010	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2011	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2012	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2013	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2014	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2015	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2016	0.09	12.4	4.1	68.7	14.7
Northeast River	2007	0.44	36.7	0.1	42.7	20.1
Northeast River	2008	0.44	36.7	0.1	42.7	20.1
Northeast River	2009	0.45	36.7	0.1	42.7	20.1
Northeast River	2010	0.46	31.1	0.1	38.6	28.9
Northeast River	2011	0.46	31.1	0.1	38.6	28.9

Table 3-2 (Cont.)

Northeast River	2012	0.47	31.1	0.1	38.6	28.9
Northeast River	2013	0.47	31.1	0.1	38.6	28.9
Northeast River	2014	0.48	31.1	0.1	38.6	28.9
Northeast River	2015	0.48	31.1	0.1	38.6	28.9
Northeast River	2016	0.49	31.1	0.1	38.6	28.9
Northeast River	2017	0.49	31.1	0.1	38.6	28.9
Piscataway Creek	2003	1.30	12.8	0.3	45.8	40.6
Piscataway Creek	2006	1.38	12.8	0.3	45.8	40.6
Piscataway Creek	2007	1.40	12.8	0.3	45.8	40.6
Piscataway Creek	2009	1.43	12.8	0.3	45.8	40.6
Piscataway Creek	2010	1.45	10.0	0.2	40.4	47.0
Piscataway Creek	2011	1.46	10.0	0.2	40.4	47.0
Piscataway Creek	2012	1.47	10.0	0.2	40.4	47.0
Piscataway Creek	2013	1.49	10.0	0.2	40.4	47.0
Piscataway Creek	2014	1.50	10.0	0.2	40.4	47.0
Rhode/West Rivers	2003	0.47	34.1	0.8	45.3	19.8
Rhode/West Rivers	2004	0.47	34.1	0.8	45.3	19.8
Rhode/West Rivers	2005	0.48	34.1	0.8	45.3	19.8
Severn River	2003	2.06	8.6	0.2	35.2	55.8
Severn River	2004	2.09	8.6	0.2	35.2	55.8
Severn River	2005	2.15	8.6	0.2	35.2	55.8
Severn River	2017	2.30	5.0	0.2	28.0	65.1
South River	2003	1.24	15.2	0.4	45.6	38.8
South River	2004	1.25	15.2	0.4	45.6	38.8
South River	2005	1.27	15.2	0.4	45.6	38.8
St. Clements River	2003	0.19	38.6	0.9	48.6	11.8
St. Clements River	2004	0.20	38.6	0.9	48.6	11.8
St. Clements River	2005	0.20	38.6	0.9	48.6	11.8
Tred Avon River	2006	0.69	50.1	1.0	21.6	27.2
Tred Avon River	2007	0.71	50.1	1.0	21.6	27.2
Tred Avon River	2008	0.73	50.1	1.0	21.6	27.2
Tred Avon River	2009	0.74	50.1	1.0	21.6	27.2
Tred Avon River	2010	0.75	43.2	0.8	21.6	33.6
Tred Avon River	2011	0.75	43.2	0.8	21.6	33.6
Tred Avon River	2012	0.75	43.2	0.8	21.6	33.6
Tred Avon River	2013	0.76	43.2	0.8	21.6	33.6
Tred Avon River	2014	0.76	43.2	0.8	21.6	33.6
Tred Avon River	2015	0.76	43.2	0.8	21.6	33.6
Tred Avon River	2016	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2017	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2018	0.77	43.2	0.8	21.6	33.6
Wicomico River	2003	0.19	34.7	4.6	48.5	12.0
Wicomico River	2011	0.21	31.6	4.6	44.9	18.7
Wicomico River	2012	0.21	31.6	4.6	44.9	18.7
Wicomico River	2017	0.22	31.6	4.6	44.9	18.7
Wye River	2007	0.10	67.7	0.7	23.5	8.1
Wye River	2008	0.10	67.7	0.7	23.5	8.1
Wye River	2018	0.10	64.9	0.6	23.0	10.9

Table 3-3. Percentages of all dissolved oxygen (DO) measurements and all bottom DO measurements that did not meet target (5.0 mg / L) or threshold (3.0 mg / L) conditions for each subestuary sampled in 2018. C / ha = structures per hectare. N = number of samples.

				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	1.53	8	13%	.	.	.
Chester River**	Mesohaline	0.15	61	3%	19	5%	0%
Corsica River	Mesohaline	0.27	77	26%	23	35%	9%
Langford Creek	Mesohaline	0.07	100	6%	28	18%	0%
Tred Avon River	Mesohaline	0.77	110	18%	28	50%	14%
Wye River	Mesohaline	0.10	94	27%	27	59%	15%

* Bush River only contains surface DO readings. ACLEC volunteers did not collect bottom DO measurements in 2018.

** Chester River includes bottom DO readings taken in the channel; bottom DO measurments were recorded in channel at only 3 sites (Sites 1, 3, and 6).

Table 3-4. Beach seine catch summary, 2018. C / ha = structures per hectare. GM CPUE = geometric mean catch per seine sample. Italics designate target species. Young of the year or juveniles = YOY.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Bush River	4	16	27	*Gizzard Shad*	1.53	5,176	279
				Spottail Shiner			
				Blueback Herring			
				Inland Silverside			
				White Perch (YOY)			
				Alewife			
				Pumpkinseed			
Chester River	6	36	27	Atlantic Silverside	0.15	6,025	129
				Mummichog			
				White Perch (YOY)			
				White Perch (Adult)			
				Striped Killifish			
				Banded Killifish			
				Inland Silverside			
				Spottail Shiner			
				Striped Bass (YOY)			
Corsica River	3	16	19	Atlantic Silverside	0.27	5,393	225
				Mummichog			
				Striped Killifish			
				Spottail Shiner			
				White Perch (YOY)			
				Inland Silverside			
Langford Creek	3	16	19	Atlantic Silverside	0.07	4,403	237
				Alewife			
				Striped Killifish			
				White Perch (Adult)			
				Mummichog			
Tred Avon River	4	22	22	Atlantic Silverside	0.77	3,731	137
				Striped Bass (YOY)			
				Striped Killifish			
				White Perch (YOY)			
				Mummichog			
				Inland Silverside			
Wye River	3	18	19	Atlantic Silverside	0.10	3,692	182
				Mummichog			
				Striped Killifish			
				White Perch (Adult)			
				Inland Silverside			
Grand Total	23	124	42	Atlantic Silverside		28,420	
				Mummichog			
				White Perch (YOY)			
				Striped Killifish			
				Gizzard Shad			
				Spottail Shiner			
				White Perch (Adult)			
				Inland Silverside			
				Blueback Herring			
				Alewife			

Table 3-5. Bottom trawl catch summary, 2018. C / ha = structures per hectare. GM CPUE = geometric mean catch per trawl sample. Italics designate target species. Young of the year or juveniles = YOY.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Corsica River	4	24	7	*White Perch (YOY)*	0.27	586	16
				White Perch (Adult)			
Langford Creek	4	24	11	*White Perch (Adult)*	0.07	919	27
				White Perch (YOY)			
Tred Avon River	4	24	15	Hogchoker	0.77	383	20
				White Perch (Adult)			
				Oyster Toadfish			
				Spot			
				American Eel			
				Green Goby			
				Striped Bass (YOY)			
Wye River	4	24	14	*White Perch (Adult)*	0.10	448	12
				White Perch (YOY)			
				Spot			
				Green Goby			
				Brown Bullhead			
				White Catfish			
				Bay Anchovy			
Grand Total	16	96	22	*White Perch (Adult)*		2,336	
				White Perch (YOY)			
				Hogchoker			
				Spot			
				Oyster Toadfish			
				Green Goby			

Table 3-6. Subestuaries sampled during 2003–2018, by salinity class, with C / ha (watershed structures per hectare), mean annual surface and bottom temperatures, and mean annual surface and bottom dissolved oxygen (mg / L).

			Temperature		Dissolved Oxygen	
River	Year	C / ha	Surface	Bottom	Surface	Bottom
		Mesohaline				
Blackwater River	2006	0.04	28.14	27.98	5.27	4.12
Breton Bay	2003	0.27	26.40	25.69	8.10	3.75
	2004	0.28	27.01	25.95	7.36	3.73
	2005	0.30	28.62	27.51	6.98	3.99
Broad Creek	2012	0.29	27.50	26.60	8.30	5.97
	2013	0.30	27.30	26.49	7.26	5.76
	2014	0.30	27.62	26.64	7.65	5.78
	2015	0.30	28.05	27.05	7.93	6.63
	2016	0.30	29.16	28.33	7.30	6.16
	2017	0.30	27.01	26.29	7.50	6.11
Chester River	2007	0.14	25.59	24.18	5.38	4.53
	2008	0.14	25.09	25.35	5.24	4.20
	2009	0.15	25.79	25.77	5.74	5.21
	2010	0.15	26.12	24.97	5.84	5.71
	2011	0.15	25.31	25.41	4.90	4.28
	2012	0.15	27.12	27.12	4.67	4.39
	2018	0.15	27.54	26.90	6.83	6.00
Corsica River	2003	0.17	25.90	26.13	6.50	4.67
	2004	0.18	27.18	26.88	5.57	4.57
	2005	0.19	28.54	28.14	6.48	3.08
	2006	0.21	27.39	26.84	7.55	4.05
	2007	0.22	25.94	25.82	6.24	4.22
	2008	0.24	26.20	25.22	7.32	4.21
	2010	0.24	34.36	26.62	5.69	5.01
	2011	0.25	27.00	27.01	5.30	3.28
	2012	0.25	27.79	27.47	4.71	3.40
	2018	0.27	27.23	26.71	7.02	5.12
Fishing Bay	2006	0.04	26.23	25.28	7.24	6.79
Harris Creek	2012	0.39	26.55	26.42	7.44	6.35
	2013	0.39	26.39	26.05	7.02	6.01
	2014	0.39	27.61	26.68	6.84	4.84
	2015	0.39	26.62	26.62	7.19	6.56
	2016	0.39	27.82	27.75	6.65	6.02
Langford Creek	2006	0.07	27.05	26.52	6.95	5.68
	2007	0.07	26.23	25.48	6.69	5.68
	2008	0.07	27.47	26.65	6.85	5.05
	2018	0.07	27.08	31.78	6.40	5.10
Magothy River	2003	2.68	25.70	25.31	7.30	2.04
Miles River	2003	0.24	25.50	25.60	6.50	4.09
	2004	0.24	25.75	25.64	6.08	5.47
	2005	0.24	28.03	27.44	5.96	3.31
Rhode River	2003	0.47	25.00	24.69	7.10	4.80
	2004	0.47	27.00	26.95	6.58	5.39
	2005	0.48	27.78	27.16	6.50	4.03
Severn River	2003	2.06	26.30	24.75	7.60	1.57
	2004	2.09	27.42	26.18	7.05	2.64
	2005	2.15	28.01	26.23	7.07	0.96
	2017	2.30	26.93	26.07	6.86	1.78

Table 3-6 (Cont.)

South River	2003	1.24	25.40	24.56	7.60	2.61
	2004	1.25	25.79	25.48	6.46	3.77
	2005	1.27	27.57	26.67	6.02	2.49
St. Clements River	2003	0.19	26.00	25.29	8.20	3.48
	2004	0.20	26.08	25.78	6.84	4.61
	2005	0.20	27.12	26.36	6.85	4.42
Transquaking River	2006	0.03	26.68	22.75	5.75	5.85
Tred Avon River	2006	0.69	27.12	26.72	6.18	5.34
	2007	0.71	26.85	26.59	6.49	5.39
	2008	0.73	26.28	25.61	6.90	4.83
	2009	0.74	26.15	26.03	7.37	6.31
	2010	0.75	27.47	26.93	7.08	5.26
	2011	0.75	28.48	28.18	6.82	5.11
	2012	0.75	27.27	27.16	7.02	5.47
	2013	0.76	26.79	26.39	7.15	5.00
	2014	0.76	26.66	26.51	6.12	5.90
	2015	0.76	28.00	27.60	6.92	5.54
	2016	0.77	28.89	28.44	7.27	5.15
	2017	0.77	26.49	26.13	7.01	5.04
	2018	0.77	27.79	27.34	7.34	4.81
West River	2003	0.64	24.90	24.31	7.40	4.84
	2004	0.65	26.83	26.59	7.37	5.58
	2005	0.66	27.96	27.15	6.72	3.99
Wicomico River	2003	0.19	25.40	23.83	7.00	5.85
	2010	0.21	25.43	25.30	6.06	5.21
	2011	0.21	27.08	26.89	5.57	4.30
	2012	0.21	27.57	27.38	6.59	5.44
	2017	0.22	26.70	25.73	7.55	4.62
Wye River	2007	0.10	26.75	26.45	7.08	5.70
	2008	0.10	26.98	26.22	5.70	5.11
	2018	0.10	28.36	27.78	8.07	4.67
		Oligohaline				
Bohemia River	2006	0.11	26.79	26.02	7.01	6.41
Bush River	2006	1.41	25.48	24.28	7.96	7.47
	2007	1.43	27.02	26.42	7.68	6.54
	2008	1.45	26.59	24.20	9.00	5.43
	2009	1.46	25.88	24.34	9.41	8.54
	2010	1.47	27.72	23.80	7.79	7.04
	2011	1.48	26.98	26.94	6.47	5.50
	2012	1.49	26.79	26.17	6.63	5.20
	2013	1.51	25.11	24.73	9.98	6.73
	2014	1.52	26.52	25.64	7.30	5.73
	2015	1.52	26.52	25.64	7.30	5.73
	2016	1.53	27.98	27.48	7.97	6.34
	2017	1.53	26.08	29.13	7.10	4.66
	2018	1.53	28.18	.	7.30	.
Gunpowder River	2009	0.72	25.71	26.05	7.39	6.79
	2010	0.72	25.17	25.91	7.89	7.13
	2011	0.73	25.09	25.56	8.28	7.14
	2012	0.73	26.48	25.93	8.19	6.71
	2013	0.73	25.85	27.46	8.05	6.10
	2014	0.73	26.65	26.15	7.28	5.76
	2015	0.73	27.51	27.65	8.02	6.63
	2016	0.73	27.70	26.46	7.43	6.18

Table 3-6 (Cont.)

Middle River	2009	3.30	26.50	25.78	7.27	6.07
	2010	3.32	24.65	24.20	8.44	7.11
	2011	3.33	27.13	26.42	8.35	7.33
	2012	3.33	28.05	26.60	8.82	5.21
	2013	3.34	27.12	26.46	7.58	5.79
	2014	3.35	26.56	26.01	7.55	6.04
	2015	3.36	28.47	27.20	8.20	6.23
	2016	3.38	28.87	27.82	7.56	5.69
	2017	3.38	25.54	25.17	7.80	5.36
Nanjemoy Creek	2003	0.08	25.90	28.80	7.30	4.96
	2008	0.09	27.53	26.58	7.85	6.65
	2009	0.09	26.31	24.64	7.05	7.49
	2010	0.09	26.50	24.80	7.66	7.02
	2011	0.09	29.34	28.55	6.13	5.30
	2012	0.09	26.18	25.92	6.73	5.98
	2013	0.09	26.88	26.30	6.76	5.86
	2014	0.09	26.78	26.36	7.66	6.25
	2015	0.09	27.40	27.10	7.16	6.32
	2016	0.09	28.49	28.21	6.86	5.16
Tidal Fresh						
Mattawoman Creek	2003	0.76	26.00	25.75	9.00	8.81
	2004	0.79	27.33	27.14	8.34	7.95
	2005	0.81	28.77	28.09	7.74	7.27
	2006	0.83	27.05	26.44	7.10	6.50
	2007	0.86	26.89	26.85	6.70	6.48
	2008	0.87	26.40	24.52	7.97	6.33
	2009	0.88	26.20	26.64	7.92	7.86
	2010	0.90	26.21	26.10	6.95	6.62
	2011	0.91	27.08	27.46	6.33	6.51
	2012	0.90	26.70	26.82	7.40	7.00
	2013	0.91	26.35	25.94	9.22	8.40
	2014	0.93	26.73	26.24	7.48	6.17
	2015	0.93	27.91	26.84	8.66	7.74
	2016	0.93	28.47	28.03	6.96	6.54
Northeast River	2007	0.44	26.83	26.43	9.73	7.75
	2008	0.44	25.35	24.98	8.43	7.70
	2009	0.45	26.33	25.55	9.35	7.36
	2010	0.46	25.90	26.21	7.76	6.78
	2011	0.46	25.97	25.71	6.87	5.79
	2012	0.47	27.78	27.59	7.88	6.03
	2013	0.47	26.61	26.11	9.33	7.06
	2014	0.48	26.94	26.52	7.72	6.81
	2015	0.48	26.66	26.23	7.84	6.17
	2016	0.49	27.95	26.86	8.81	7.10
	2017	0.49	26.38	25.68	9.38	7.80
Piscataway Creek	2003	1.30	25.60	24.63	10.20	8.33
	2006	1.38	28.16	24.97	8.70	6.85
	2007	1.40	27.47	26.00	8.57	7.60
	2009	1.43	26.72	27.07	8.56	6.62
	2010	1.45	27.07	25.08	9.36	7.63
	2011	1.46	28.25	30.07	9.05	9.47
	2012	1.47	27.92	25.51	9.53	9.34
	2013	1.49	27.19	26.22	9.87	7.65
	2014	1.50	26.98	26.28	8.66	7.33

Table 3-7. Pearson correlations (r) of mean survey surface and bottom dissolved oxygen (DO; mg / L) with water temperatures at depth (surface and bottom) and with watershed development (C / ha = structures per hectare) from subestuaries sampled during 2003-2018, by salinity class. Level of significance = *P*. N = sample size. Bold numbers indicate a significant relationship ($P \leq 0.05$).

DO Depth	Statistics	Temperature	C / ha
	Mesohaline		
Surface	r	-0.012	**0.240**
	P	0.913	**0.033**
	N	79	**79**
Bottom	r	0.079	**-0.598**
	P	0.491	**<0.0001**
	N	79	**79**
	Oligohaline		
Surface	r	**-0.340**	**0.339**
	P	**0.030**	**0.030**
	N	**41**	**41**
Bottom	r	**-0.608**	-0.076
	P	**<0.0001**	0.643
	N	**40**	40
	Tidal Fresh		
Surface	r	-0.051	0.267
	P	0.773	0.127
	N	34	34
Bottom	r	0.044	**0.377**
	P	0.807	**0.028**
	N	34	**34**

Table 3-8. Pearson correlations (r) among Maryland Department of Planning (DOP) land use categories and with C / ha for mesohaline subestuaries sampled during 2003-2018. Land cover estimates were estimated by MD DOP for 2002 and 2010. *P* = level of significance. N = sample size. Bold numbers indicate a significant relationship ($\leq$ 0.05).

			Land Use Categories			
	Statistics	**C/ha**	**Agriculture**	**Forest**	**Wetland**	**Urban**
C/ha	r					
	P	1				
	N					
Agriculture	r	**-0.752**				
	P	**<0.0001**	1			
	N	**70**				
Forest	r	0.062	**-0.581**			
	P	0.612	**<0.0001**	1		
	N	70	**70**			
Wetland	r	**-0.270**	0.019	-0.020		
	P	**0.024**	0.875	0.872	1	
	N	**70**	70	70		
Urban	r	**0.904**	**-0.810**	0.0003	-0.110	
	P	**<0.0001**	**<0.0001**	0.998	0.363	1
	N	**70**	**70**	70	70	

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

Linear Model	**Western Shore: Median Bottom DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	53.12	53.12	53.82	<0.0001	
Residual	20	19.74	0.99			
Total	21	72.87				
r^2 = 0.7291						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.49	0.48	1.02	0.32	-0.51	1.50
Agriculture (%)	0.13	0.02	7.34	<0.0001	0.09	0.17
Linear Model	**Eastern Shore: Median Bottom DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	7.91	7.91	13.26	0.0007	
Residual	49	29.22	0.60			
Total	50	37.13				
r^2 = 0.213						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	7.28	0.57	12.67	<0.0001	6.12	8.43
Agriculture (%)	-0.04	0.01	-3.64	0.0007	-0.06	-0.02

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

Linear Model	**Median Bottom DO = Agriculture (%) Coverage**					
ANOVA	df	SS	MS	F	Significance F	
Regression	2	92.28	46.14	58.79	<0.0001	
Residual	70	54.94	0.78			
Total	72	147.22				
$r^2 = 0.627$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	-0.17	0.47	-0.36	0.72	-1.10	0.76
Agriculture (%)	0.23	0.02	9.82	<0.0001	0.18	0.28
Agriculture (%)^2	-0.002	0.0003	-8.39	<0.0001	-0.003	-0.002

Table 3-11. Subestuaries sampled during 2003-2018, grouped by salinity class and ranked by annual 4.9 m trawl catch geometric mean (GM).

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 Creek	2007	273	7
Tred Avon River	2010	264	8
Chester River	2011	259	9
Langford Creek	2006	258	10
Corsica River	2004	251	11
Corsica River	2011	238	12
Tred Avon River	2014	192	13
Harris Creek	2014	174	14
Corsica River	2006	174	15
Chester River	2010	172	16
Wye River	2007	170	17
Rhode River	2005	163	18
Corsica River	2012	162	19
Langford Creek	2008	161	20
Corsica River	2010	161	21
Tred Avon River	2008	155	22
Tred Avon River	2012	155	23
Harris Creek	2012	155	24
Chester River	2007	152	25
Broad Creek	2017	148	26
Broad Creek	2016	147	27
Broad Creek	2013	142	28
Tred Avon River	2007	137	29
Corsica River	2007	131	30
Fishing Bay River	2006	131	31
Transquaking River	2006	131	32
Chester River	2012	130	33
West River	2005	125	34
Tred Avon River	2016	121	35
Chester River	2008	120	36
Wicomico River	2010	120	37
Wye River	2008	114	38
South River	2003	110	39

Table 3-11 (Cont.)

Wicomico River	2012	110	40
Corsica River	2005	109	41
Tred Avon River	2009	104	42
Broad Creek	2015	103	43
Tred Avon River	2017	98	44
Tred Avon River	2011	92	45
Harris Creek	2013	89	46
Corsica River	2008	86	47
Miles River	2004	82	48
Wicomico River	2017	81	49
Tred Avon River	2015	80	50
Tred Avon River	2013	77	51
Chester River	2009	76	52
Tred Avon River	2006	76	53
Miles River	2005	72	54
Wicomico River	2011	65	55
Wicomico River	2003	59	56
St. Clements River	2005	54	57
Harris Creek	2016	51	58
Harris Creek	2015	40	59
Rhode River	2004	38	60
South River	2005	35	61
Blackwater River	2006	35	62
Breton Bay	2005	34	63
West River	2004	34	64
Magothy River	2003	33	65
St. Clements River	2003	31	66
Langford Creek	2018	27	67
South River	2004	21	68
Tred Avon River	2018	20	69
Corsica River	2018	16	70
Breton Bay	2003	18	71
St. Clements River	2004	17	72
Breton Bay	2004	16	73
Severn River	2017	16	74
Wye River	2018	12	75
Severn River	2003	9	76
Severn River	2004	5	77
Severn River	2005	3	78

Table 3-11 (Cont.)

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
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

Table 3-11 (Cont.)

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
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

Table 3-12. Percent of major land use categories estimated by Maryland Department of Planning (DOP) in each of the Choptank River subestuaries. Land use estimates are determined from MD DOP 2010 data. The first four land use categories contain only land area (hectares) of the watershed; water area (hectares) is removed from each of these categories. Water is the percent of water hectares per area of water and land.

	Subestuary		
Land Use Category	**Broad Creek**	**Tred Avon River**	**Harris Creek**
Agriculture	42.6	43.2	44.9
Forest	25.4	21.6	19.7
Urban	31.5	33.6	29.8
Wetlands	0.4	0.8	5.6
Water	57.3	62.0	24.4

Table 3-13. Percentages of all dissolved oxygen (DO) measurements (surface, middle, and bottom) and all bottom DO measurements that did not meet target (5.0 mg / L) or threshold (3.0 mg / L) conditions during July-September for years sampled. N = sample size.

				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	0.30	76	4%	22	9%	0%
	2017	0.30	72	3%	22	9%	0%
Harris Creek	2012	0.39	82	0%	23	0%	0%
	2013	0.39	83	0%	24	0%	0%
	2014	0.39	84	1%	23	4%	0%
	2015	0.39	85	0%	24	0%	0%
	2016	0.39	79	3%	23	9%	0%
Tred Avon River	2006	0.69	91	19%	24	38%	0%
	2007	0.71	93	11%	23	26%	4%
	2008	0.73	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	0.76	103	15%	26	31%	15%
	2014	0.76	96	11%	24	21%	0%
	2015	0.76	96	8%	24	21%	13%
	2016	0.77	96	13%	24	38%	13%
	2017	0.77	89	17%	24	42%	13%
	2018	0.77	110	17%	28	50%	14%

Table 3-14. Pearson correlations (r) of annual median bottom dissolved oxygen (DO; mg / L) for Broad Creek, Harris Creek, and Tred Avon River with year and among subestuaries. *P* = level of significance. N = number of annual median DO measurements for each subestuary sampled.

	Statistics	Year	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r	0.547			
	P	0.261	1		
	N	6			
Harris Creek	r	-0.22	-0.07		
	P	0.725	0.911	1	
	N	5	5		
Tred Avon River	r	-0.22	0.388	0.732	
	P	0.474	0.447	0.16	1
	N	13	6	5	

Table 3-15. Choptank subestuaries sampled during 2006-2018, ranked by annual 4.9 m trawl catch geometric mean (GM).

River	Year	GM	Rank
Broad Creek	2014	401	1
Broad Creek	2012	294	2
Tred Avon River	2010	264	3
Tred Avon River	2014	192	4
Harris Creek	2014	174	5
Harris Creek	2012	155	7
Tred Avon River	2008	155	7
Tred Avon River	2012	155	7
Broad Creek	2017	148	9
Broad Creek	2016	147	10
Broad Creek	2013	142	11
Tred Avon River	2007	137	12
Tred Avon River	2016	121	13
Tred Avon River	2009	104	14
Broad Creek	2015	103	15
Tred Avon River	2017	98	16
Tred Avon River	2011	92	17
Harris Creek	2013	89	18
Tred Avon River	2015	80	19
Tred Avon River	2013	77	20
Tred Avon River	2006	76	21
Harris Creek	2016	51	22
Harris Creek	2015	40	23
Tred Avon River	2018	20	24

Table 3-16. Pearson correlations (r) of annual 4.9 m trawl catch geometric mean (GM) for Broad Creek, Harris Creek, and Tred Avon River, with year and among subestuaries. *P* = level of significance. N = number of annual GMs for each subestuary. Bold numbers indicate a significant relationship ($P \leq 0.05$).

	Statistics	Year	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r	-0.466			
	P	0.351	1		
	N	6			
Harris Creek	r	-0.659	**0.956**		
	P	0.226	**0.011**	1	
	N	5	**5**		
Tred Avon River	r	-0.307	**0.951**	0.842	
	P	0.307	**0.004**	0.074	1
	N	13	**6**	5	

Table 3-17. Pearson correlations (r) of annual 4.9 m trawl White Perch geometric mean (GM) for Broad Creek, Harris Creek, and Tred Avon River with year and among subestuaries. *P* = level of significance. N = number of adult White Perch GMs. Bold numbers indicate a significant relationship ($P \leq 0.05$).

	Statistics	Year	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r	-0.769			
	P	0.074	1		
	N	6			
Harris Creek	r	-0.449	0.020		
	P	0.449	0.975	1	
	N	5	5		
Tred Avon River	r	-0.397	0.632	0.447	
	P	0.179	0.178	0.450	1
	N	13	6	5	

Table 3-18. Pearson correlations (r) of annual survey median Secchi depths (m) for Broad Creek, Harris Creek, and Tred Avon River with year and among subestuaries. *P* = level of significance. N = number of annual survey median Secchi depths. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	Year	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r	**0.863**			
	P	**0.027**	1		
	N	**6**			
Harris Creek	r	0.849	**0.972**		
	P	0.069	**0.006**	1	
	N	5	**5**		
Tred Avon River	r	-0.081	**0.864**	**0.928**	
	P	0.792	**0.027**	**0.023**	1
	N	13	**6**	**5**	

Table 3-19. Pearson correlations (r) of annual median pH for Broad Creek, Harris Creek, and Tred Avon River with year and among subestuaries. *P* = level of significance. N = number of annual survey median pH estimates. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	Year	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r	0.210			
	P	0.690	1		
	N	6			
Harris Creek	r	0.219	**0.937**		
	P	0.724	**0.019**	1	
	N	5	**5**		
Tred Avon River	r	0.277	0.396	0.174	
	P	0.359	0.437	0.779	1
	N	13	6	5	

Table 3-20. Pearson correlations (r) of annual survey median salinity (‰) for Broad Creek, Harris Creek, and Tred Avon River with year and among subestuaries. *P* = level of significance. N = number of annual survey median salinity estimates. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	Year	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r	0.133			
	P	0.802	1		
	N	6			
Harris Creek	r	0.412	**0.990**		
	P	0.490	**0.001**	1	
	N	5	**5**		
Tred Avon River	r	-0.154	**0.995**	**0.979**	
	P	0.615	**<0.0001**	**0.004**	1
	N	13	**6**	**5**	

Table 3-21. Percent of major land use categories estimated by Maryland Department of Planning (DOP) in each of the Queen Anne's County subestuaries. Land use estimates are estimates from MD DOP (2010). The first four land use categories contain only land area (hectares) of the watershed; water area (hectares) is removed from each of these categories. Water is the percent of water hectares per area of water and land.

	Subestuary			
Land Use Category	**Chester River**	**Corsica River**	**Langford Creek**	**Wye River**
Agriculture	64.2	60.4	70.2	64.9
Forest	24.7	25.5	20.4	23.0
Urban	8.9	13.2	8.0	10.9
Wetlands	2.0	0.1	1.5	0.6
Water	17.5	5.5	11.9	11.6

Table 3-22. Percent of all dissolved oxygen (DO) measurements (surface, middle, and bottom) and all bottom DO measurements that did not meet target (5.0 mg / L) or threshold (3.0 mg / L) conditions during July-September, by year sampled, for Chester River, Corsica River, Langford Creek, and Wye River. N = number of DO measurements.

				All DO	Bottom DO		
Subestuary	**Year**	**C / ha**	**N**	**% < 5.0 mg/L**	**N**	**% < 5.0 mg/L**	**% < 3.0 mg/L**
Chester River	2007	0.14	133	50%	30	70%	13%
Chester River	2008	0.14	190	63%	48	81%	13%
Chester River	2009	0.15	168	27%	46	41%	2%
Chester River	2010	0.15	81	14%	26	15%	4%
Chester River	2011	0.15	107	67%	29	79%	10%
Chester River	2012	0.15	122	75%	31	84%	0%
Chester River	2018	0.15	61	3%	19	5%	0%
Corsica River	2003	0.17	82	26%	23	57%	9%
Corsica River	2004	0.18	78	42%	20	60%	5%
Corsica River	2005	0.19	76	37%	21	95%	38%
Corsica River	2006	0.21	62	42%	17	82%	29%
Corsica River	2007	0.22	78	41%	22	59%	32%
Corsica River	2008	0.24	64	28%	13	62%	23%
Corsica River	2010	0.24	43	26%	16	31%	13%
Corsica River	2011	0.25	57	74%	18	94%	33%
Corsica River	2012	0.25	59	69%	15	80%	60%
Corsica River	2018	0.27	77	26%	23	35%	9%
Langford Creek	2006	0.07	92	21%	24	33%	0%
Langford Creek	2007	0.07	63	22%	13	23%	8%
Langford Creek	2008	0.07	82	29%	22	59%	0%
Langford Creek	2018	0.07	100	6%	28	18%	0%
Wye River	2007	0.10	90	20%	24	29%	0%
Wye River	2008	0.10	67	40%	15	47%	0%
Wye River	2018	0.10	94	27%	27	59%	15%

Table 3-23. Pearson correlations (r) of annual median bottom dissolved oxygen (DO; mg / L) for Chester River, Corsica River, Langford Creek, and Wye River with year and among subestuaries. *P* = level of significance. N = number of annual survey median estimates. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	**Year**	**Chester River**	**Corsica River**	**Langford Creek**	**Wye River**
Chester River	r	0.429				
	P	0.337	1			
	N	7				
Corsica River	r	0.199	0.796			
	P	0.582	0.058	1		
	N	10	6			
Langford Creek	r	0.131	0.878	-0.0933		
	P	0.869	0.318	0.907	1	
	N	4	3	4		
Wye River	r	-0.744	-0.392	-0.293	0.096	
	P	0.466	0.743	0.811	0.939	1
	N	3	3	3	3	

Table 3-24. Chester River, Corsica River and Langford Creek, and Wye River sampled, ranked by annual 4.9 m trawl catch geometric mean (GM) during 2003-2018. Chester River was not sampled by trawl during 2018.

River	Year	GM	Rank
Corsica River	2003	378	1
Langford Creek	2007	273	2
Chester River	2011	259	3
Langford Creek	2006	258	4
Corsica River	2004	251	5
Corsica River	2011	238	6
Corsica River	2006	174	7
Chester River	2010	172	8
Wye River	2007	170	9
Corsica River	2012	162	10
Corsica River	2010	161	12
Langford Creek	2008	161	12
Chester River	2007	152	13
Corsica River	2007	131	14
Chester River	2012	130	15
Chester River	2008	120	16
Wye River	2008	114	17
Corsica River	2005	109	18
Corsica River	2008	86	19
Chester River	2009	76	20
Langford Creek	2018	27	21
Corsica River	2018	16	22
Wye River	2018	12	23

Table 3-25. Pearson correlations (r) of annual 4.9 m trawl catch geometric mean (GM) for Chester River, Corsica River, Langford Creek, and Wye River with year and among subestuaries. *P* = level of significance. N = number of annual survey GM estimates. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	Year	Chester River	Corsica River	Langford Creek	Wye River
Chester River	r	0.263				
	P	0.614	1			
	N	6				
Corsica River	r	-0.615	0.859			
	P	0.059	0.062	1		
	N	10	5			
Langford Creek	r	-0.94	1	0.95		
	P	0.06	.	0.051	1	
	N	4	2	4		
Wye River	r	-0.961	1	**1**	0.994	
	P	0.178	.	**0.011**	0.071	1
	N	3	2	**3**	3	

Table 3-26. Pearson correlations (r) of annual survey median Secchi depths (m) for Chester River, Corsica River, Langford Creek, and Wye River with year and among subestuaries. *P* = level of significance. N = number of annual survey median estimates. Bold numbers indicate significance at $P \leq 0.05$. Secchi measurements were not available for Chester River.

	Statistics	Year	Corsica River	Langford Creek	Wye River
Corsica River	r	-0.258			
	P	0.537	1		
	N	8			
Langford Creek	r	0.450	0.333		
	P	0.550	0.667	1	
	N	4	4		
Wye River	r	0.997	**1**	.	
	P	0.052	**<0.0001**	.	1
	N	3	**3**	3	

Table 3-27. Pearson correlations (r) of annual median pH measurements for Chester River, Corsica River, Langford Creek, and Wye River with year and among subestuaries. *P* = level of significance. N = number of annual survey median estimates. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	Year	Chester River	Corsica River	Langford Creek	Wye River
Chester River	r	.				
	P	.	1			
	N	1				
Corsica River	r	-0.869	.			
	P	0.131	.	1		
	N	4	1			
Langford Creek	r	-0.175	.	0.572		
	P	0.825	.	0.428	1	
	N	4	1	4		
Wye River	r	-0.997	.	0.952	**-1**	
	P	0.052	.	0.198	**<0.0001**	1
	N	3	1	3	**3**	

Table 3-28. Pearson correlations (r) of annual median salinity (‰) measurements for Chester River, Corsica River, Langford Creek, and Wye River with year and among subestuaries. *P* = level of significance. N = number of annual survey median estimates. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	Year	Chester River	Corsica River	Langford Creek	Wye River
Chester River	r	**-0.757**				
	P	**0.049**	1			
	N	**7**				
Corsica River	r	0.003	**0.860**			
	P	0.994	**0.028**	1		
	N	10	**6**			
Langford Creek	r	-0.681	0.964	0.815		
	P	0.319	0.170	0.185	1	
	N	4	3	4		
Wye River	r	-0.936	**1**	0.941	0.966	
	P	0.229	**0.003**	0.221	0.167	1
	N	3	**3**	3	3	

Table 3-29. Pearson correlations of annual beach seine catch geometric mean (GM) all species of finfish from Head of Bay or Choptank River with Chester River and Tred Avon River. *P* = level of significance. N = number of annual survey GMs. Bold numbers indicate significance at $P \leq 0.05$.

	Statistics	Head of Bay	Choptank River
Choptank River	r	0.152	
	P	0.247	1
	N	60	
Chester River	r	-0.113	0.223
	P	0.627	0.332
	N	21	21
Tred Avon River	r	0.498	**0.839**
	P	0.083	**0.0003**
	N	13	**13**

Figures

Figure 3-1. Chesapeake Bay subestuaries sampled in summer of 2018. Including previously sampled subestuaries, Broad Creek (2012-2017) and Harris Creek (2012-2016), referenced throughout this report.

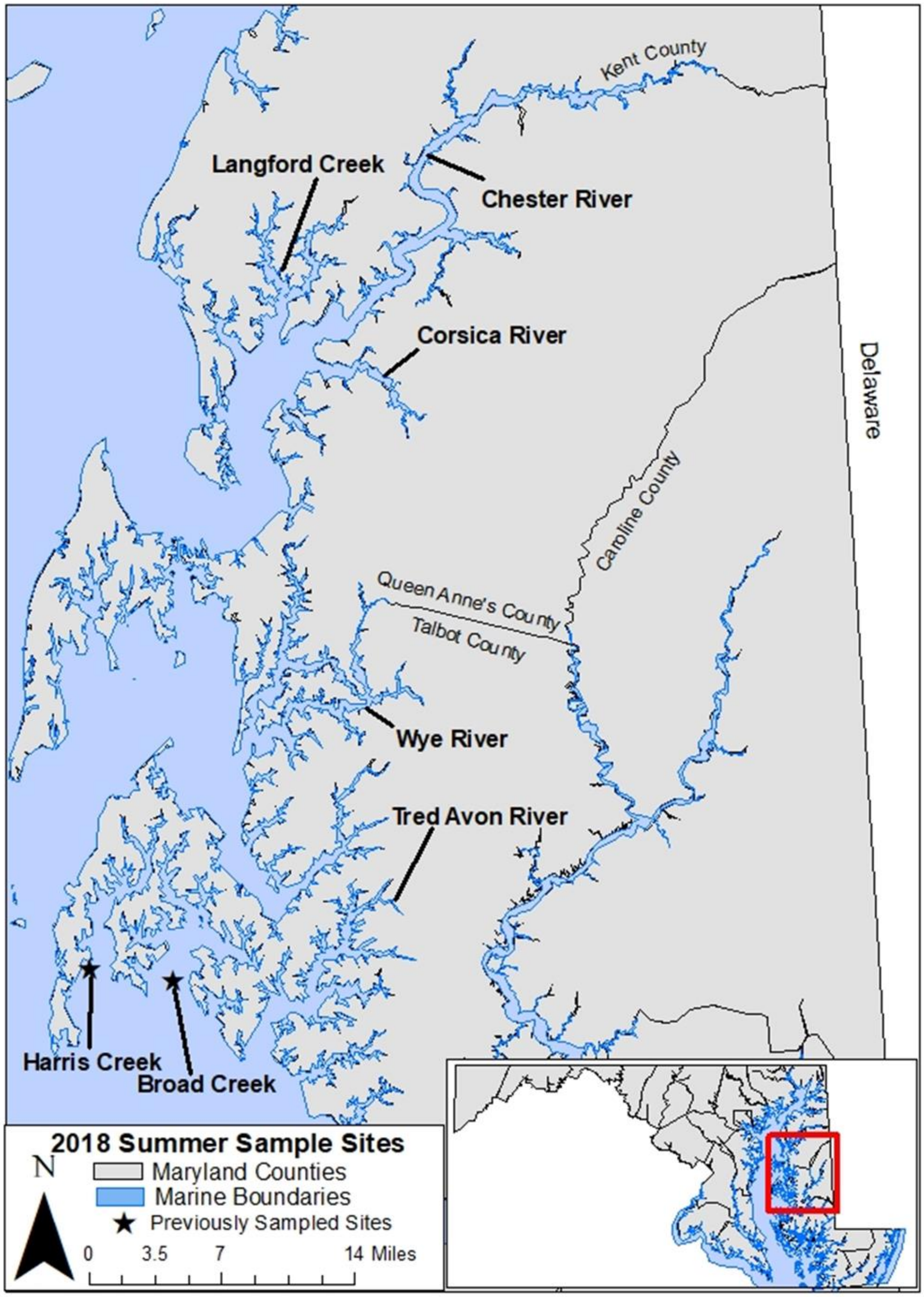


Figure 3-2. Map illustrating land use categories for the subestuaries Chester River (1), Corsica River (2), Langford Creek (3), Tred Avon River (4), and Wye River (5) in Queen Anne's, Kent, and Talbot Counties. Land use data is based on Maryland Department of Planning (DOP) 2010 land use land cover data. Including previously sampled subestuaries, Broad Creek (6) and Harris Creek (7), referenced throughout this report.

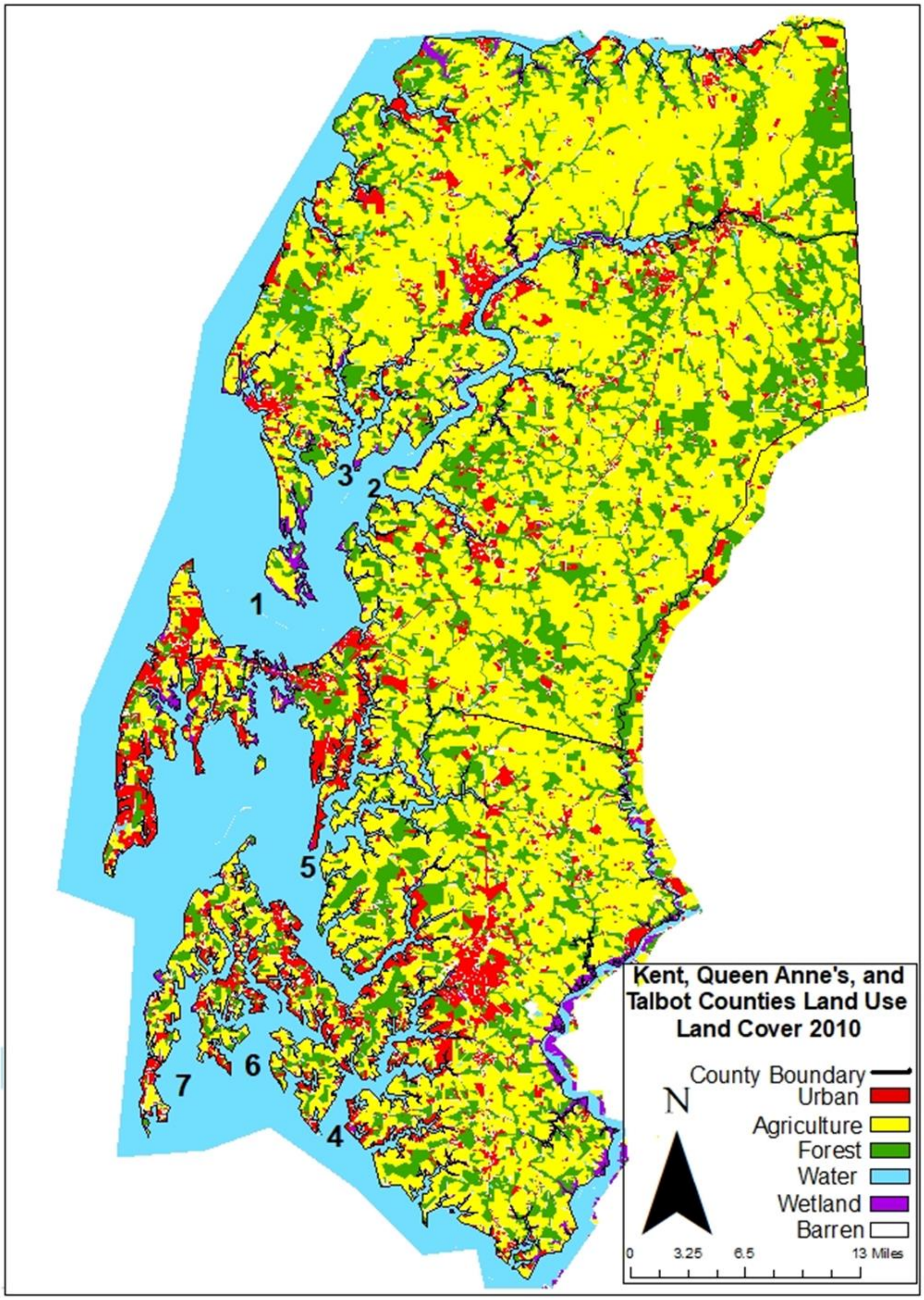


Figure 3-3. Map indicating the locations of beach seine and 4.9 m bottom trawl sampling sites for the lower Choptank River subestuaries, Broad Creek (2012-2017), Harris Creek (2012-2016), and Tred Avon River (2006-2018).

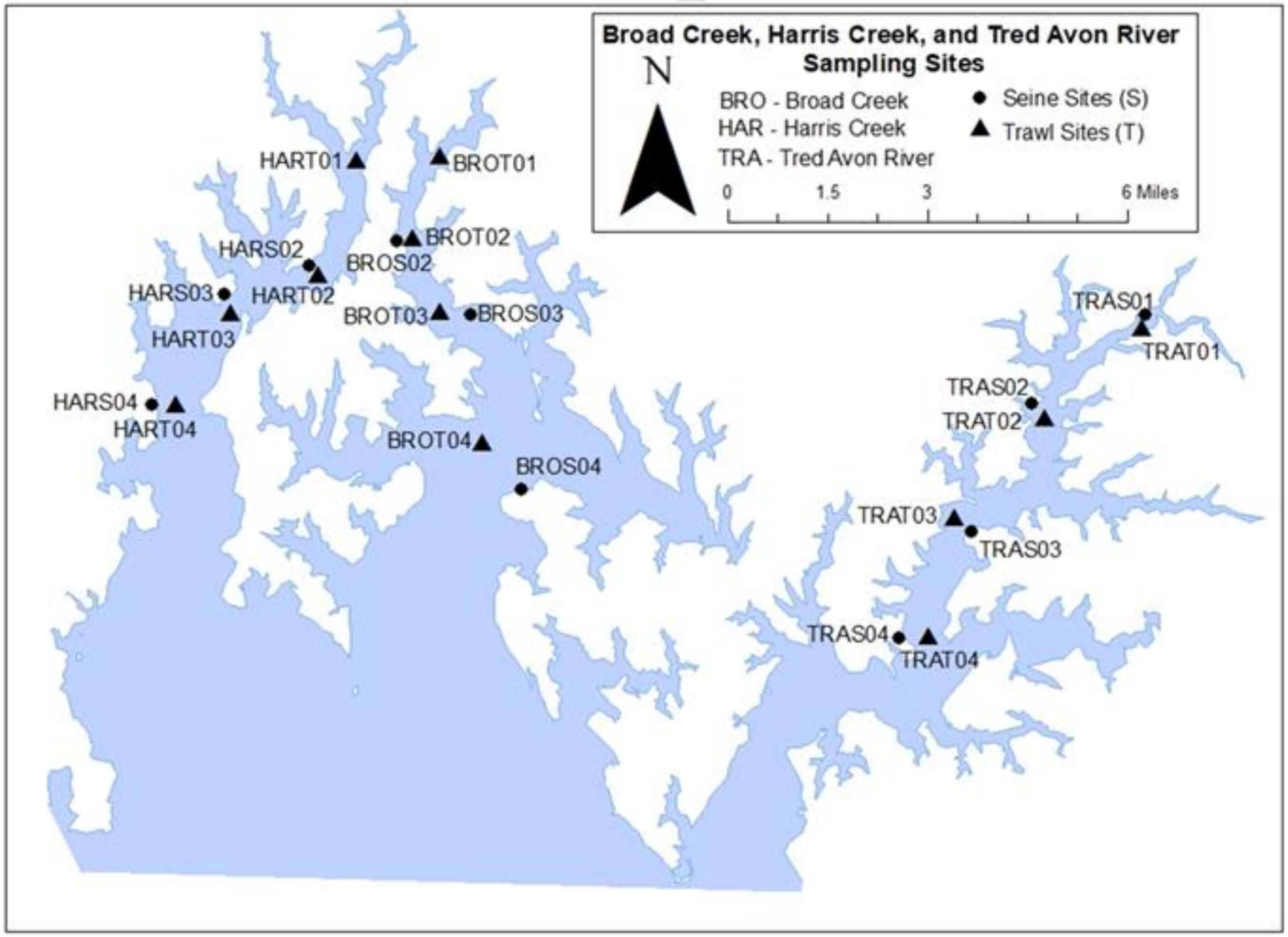


Figure 3-4. Map indicating the historical versus the present locations of sampling sites for subestuaries Chester River, Corsica River, and Langford Creek in Queen Anne’s County and the Wye River located in both Queen Anne’s and Talbot Counties.

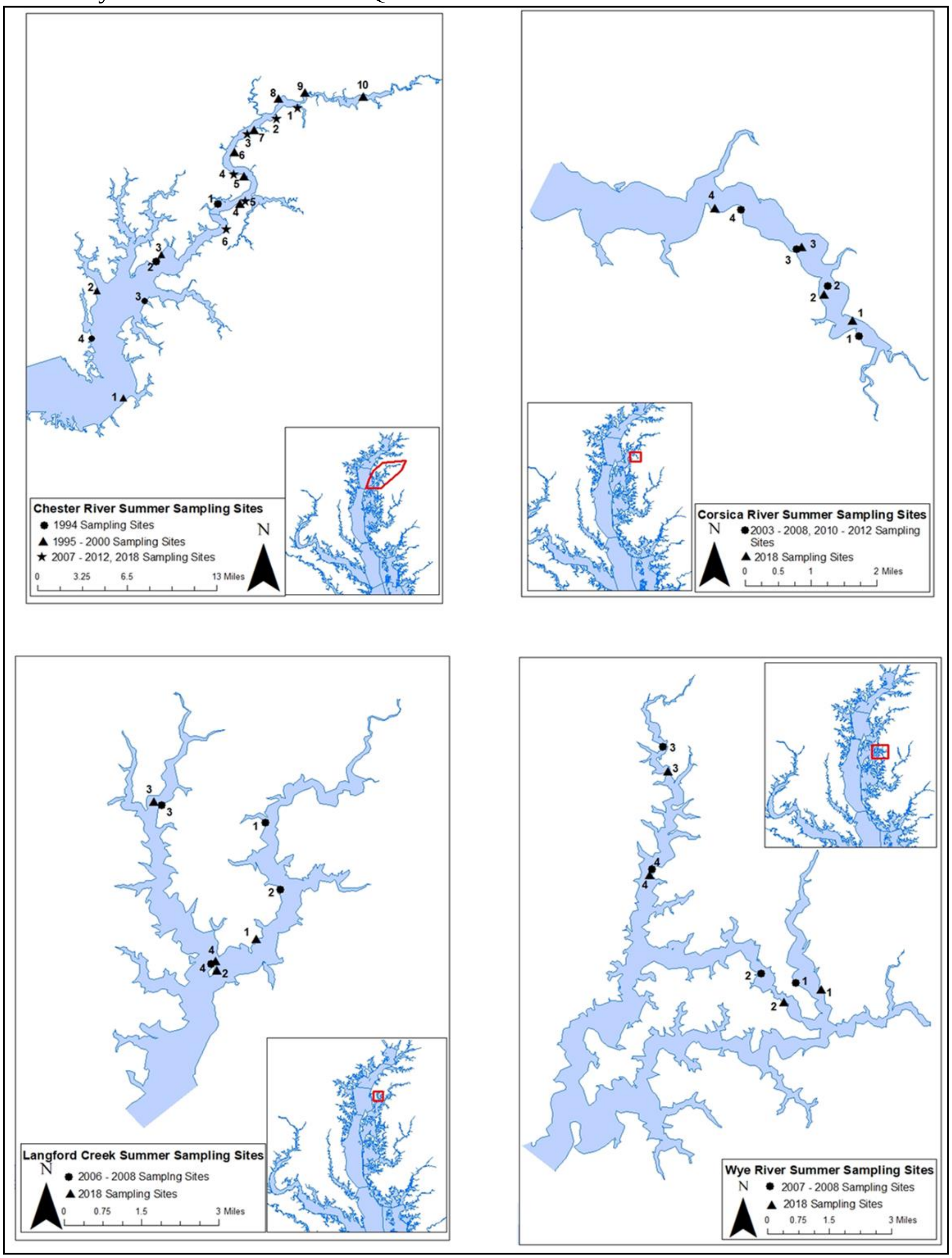


Figure 3-5. Number of finfish species (richness) collected by beach seines in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of watershed development (C / ha = structures per hectare). Points were omitted if beach seine effort (number of samples) < 15 samples.

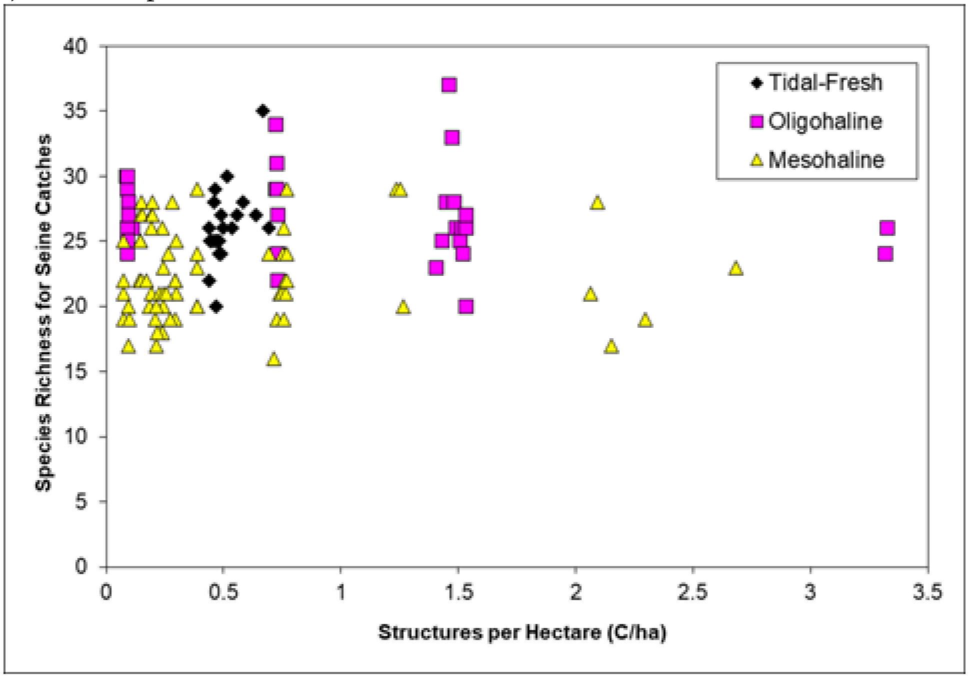


Figure 3-6. Number of finfish species (richness) collected by 4.9 m bottom trawl in tidal-fresh or oligohaline subestuaries versus intensity of development (C / ha = structures per hectare). Points were omitted if 4.9 m bottom trawl effort (number of samples) < 15 samples.

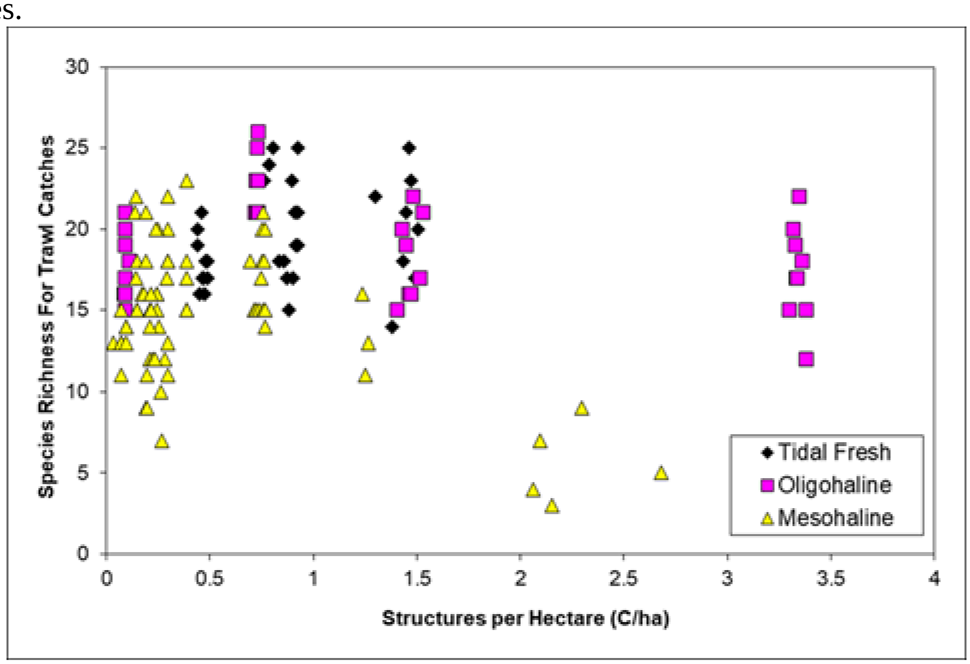


Figure 3-7. Mean subestuary bottom dissolved oxygen during summer sampling, 2003-2018, plotted against level of development (C / ha or structures per hectare).

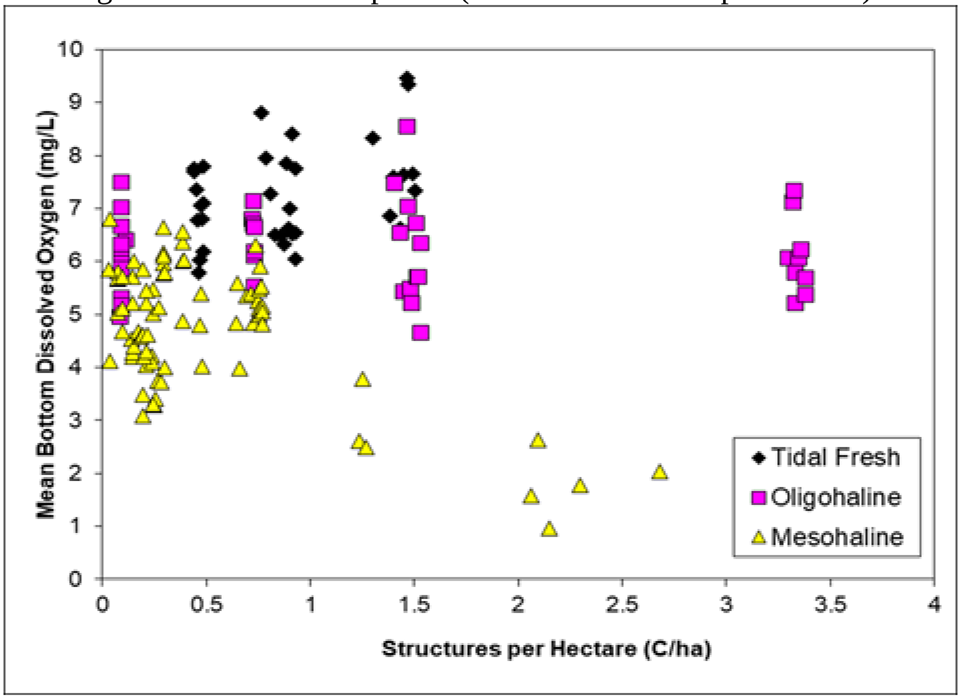


Figure 3-8. Mean subestuary surface dissolved oxygen during summer (July-October) sampling, 2003-2018, plotted against level of development (C / ha or structures per hectare).

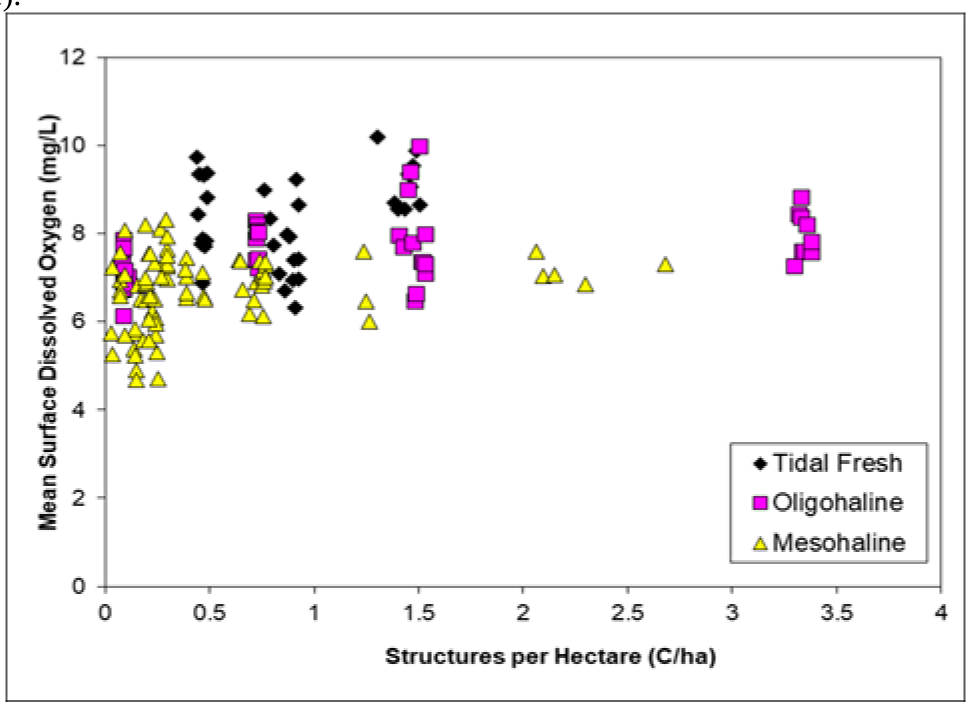


Figure 3-9. Maryland Department of Planning (DOP) estimates agricultural land coverage (% watershed land area) by region (western or eastern shore) versus median bottom dissolved oxygen (DO) in mesohaline subestuaries (2003-2018). Quadratic model predicts median bottom DO and agricultural coverage (%) using data from both regions.

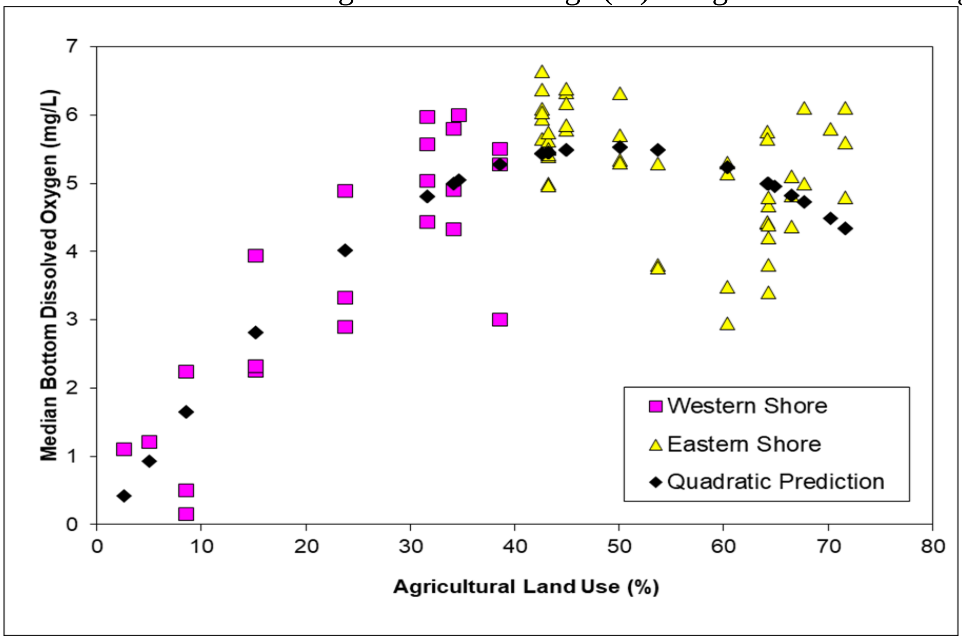


Figure 3-10. Trends in levels of development (structures per hectare = C / ha) during 1950-2016 of watersheds of three subestuaries surveyed in the Choptank River, Broad Creek, Harris Creek, and Tred Avon River, since 2006.

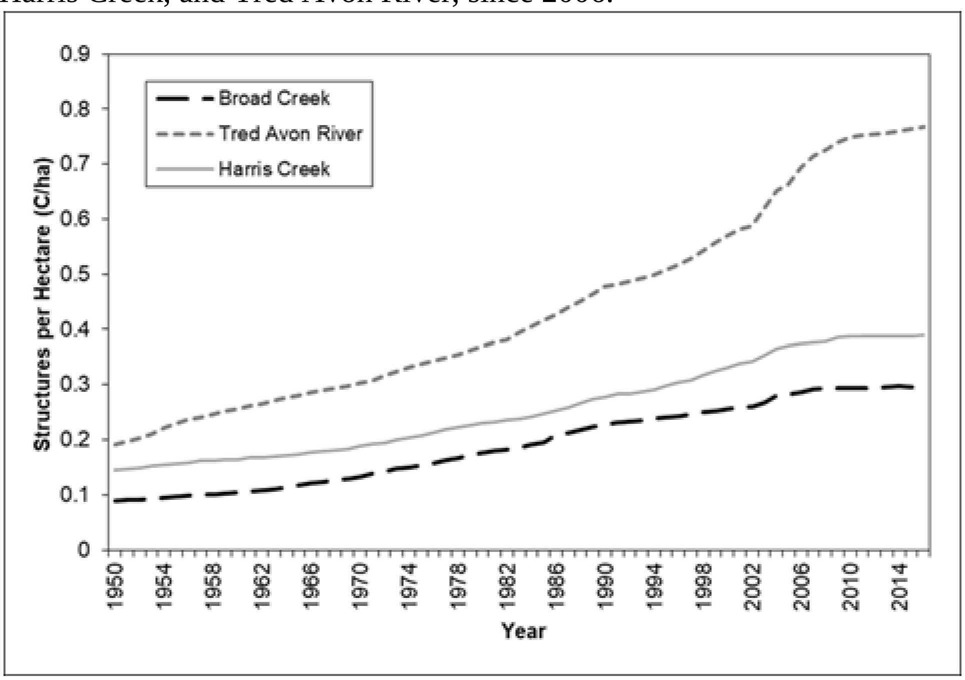


Figure 3-11. Bottom dissolved oxygen (DO; mg / L) readings (2006-2018) versus intensity of development (C / ha = structures per hectare) in Choptank subestuaries, Broad Creek, Harris Creek, and Tred Avon River. Target (5 mg / L) and threshold (3 mg / L) boundaries are indicated by red dashed lines. Harris Creek was sampled during 2012-2016 and Broad Creek was sampled during 2012-2017.

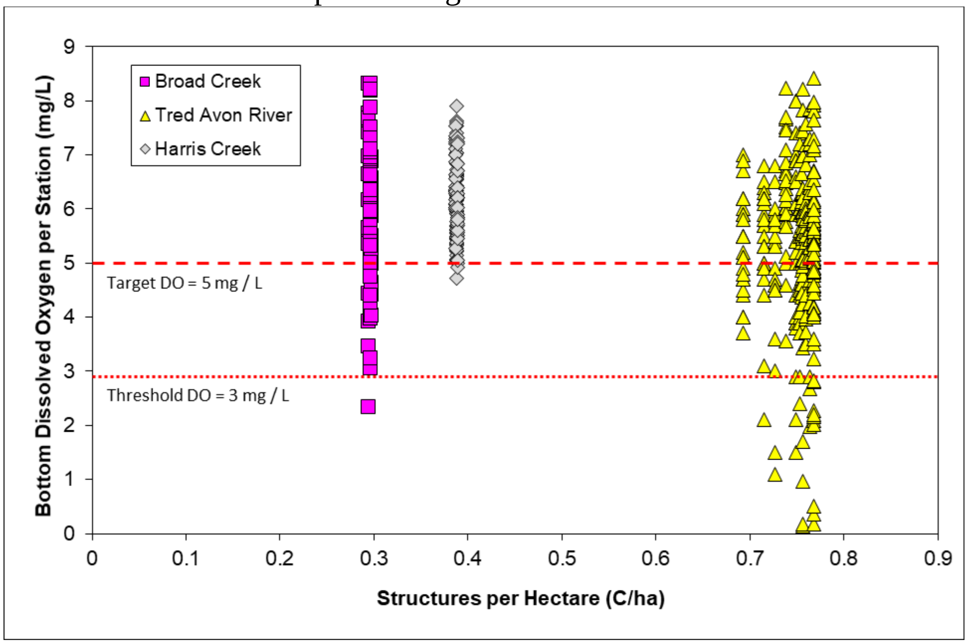


Figure 3-12. Median bottom dissolved oxygen (DO; red squares; mg / L) year's sampled for Broad Creek, Harris Creek, and Tred Avon River. Solid black bars indicate range of all bottom DO measurements for that year.

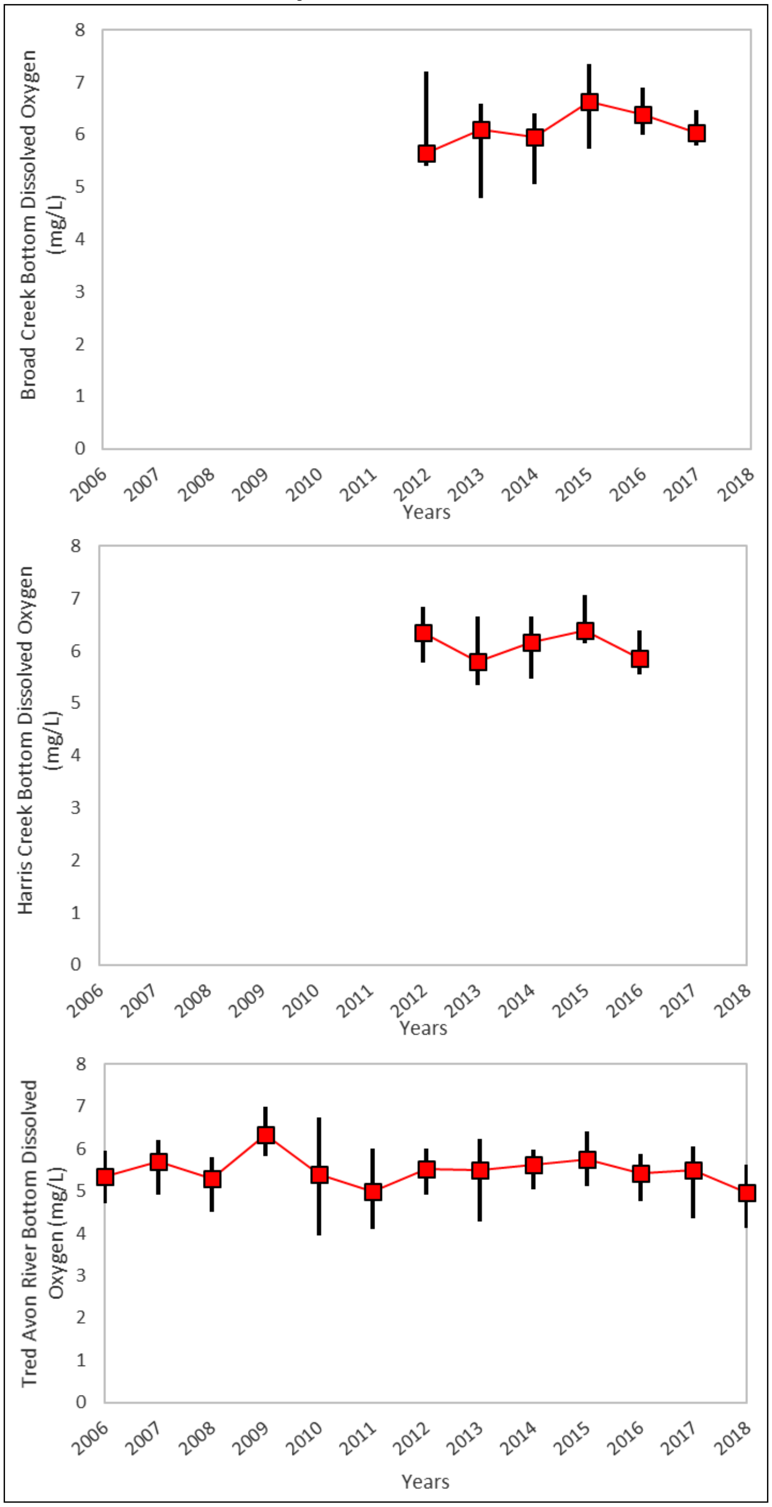


Figure 3-13. Mean bottom dissolved oxygen (DO; mg / L) for all years surveyed for Broad Creek, Harris Creek, and Tred Avon River, by sampling station. Dotted line indicates the median of all DO measurement data for the time-series available.

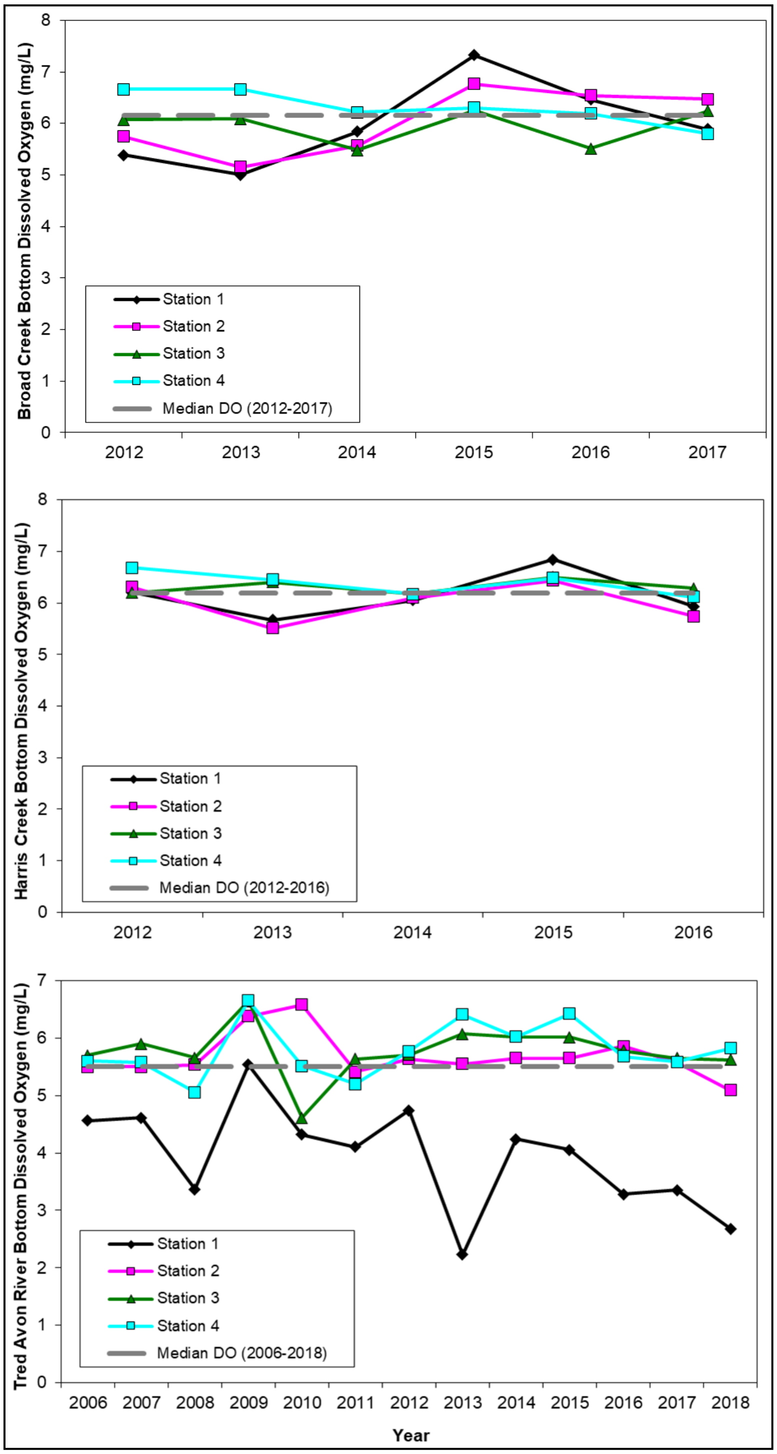


Figure 3-14. Annual 4.9m bottom trawl catch geometric mean (GM) per of all finfish species (red squares) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Black bars indicate the 95% confidence intervals.

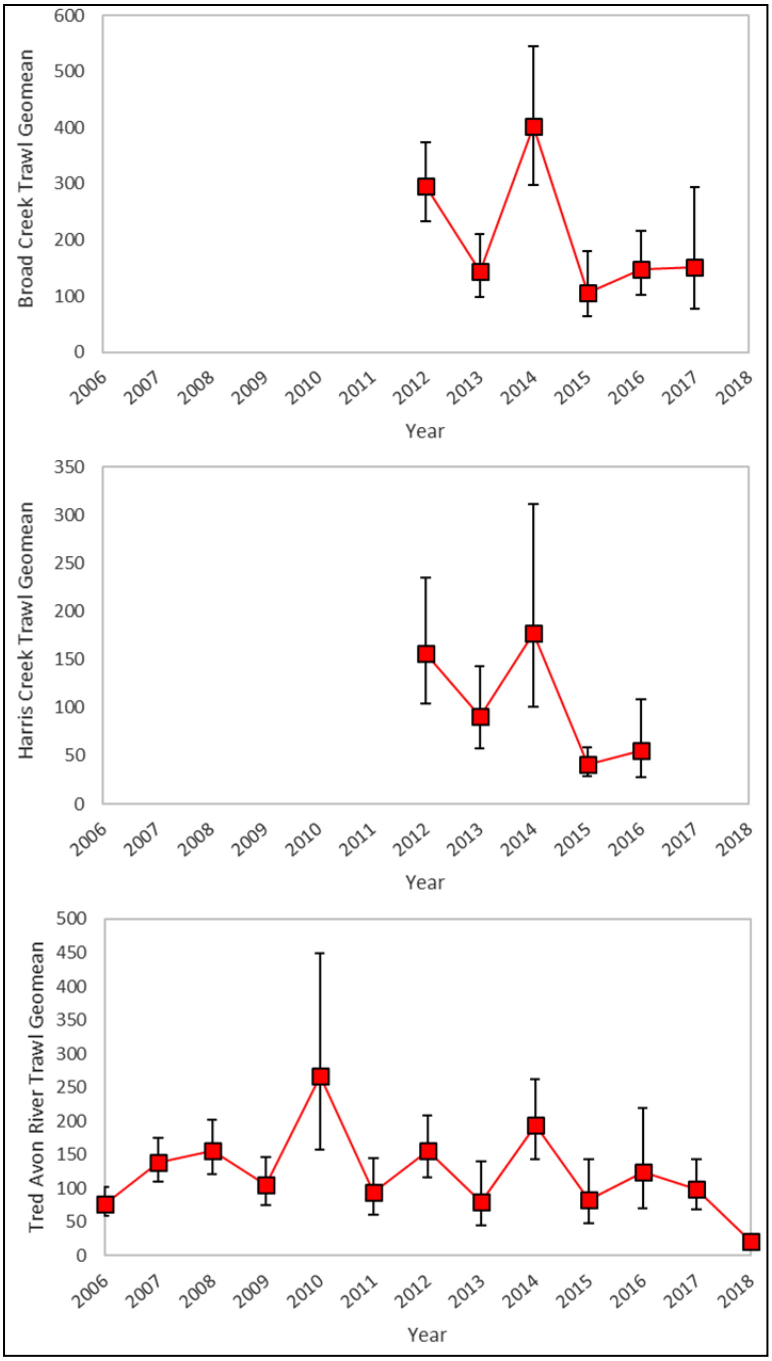


Figure 3-15. Finfish species composition for 4.9 m bottom trawl catch in Tred Avon River for all sampling years combined (2006-2018). Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

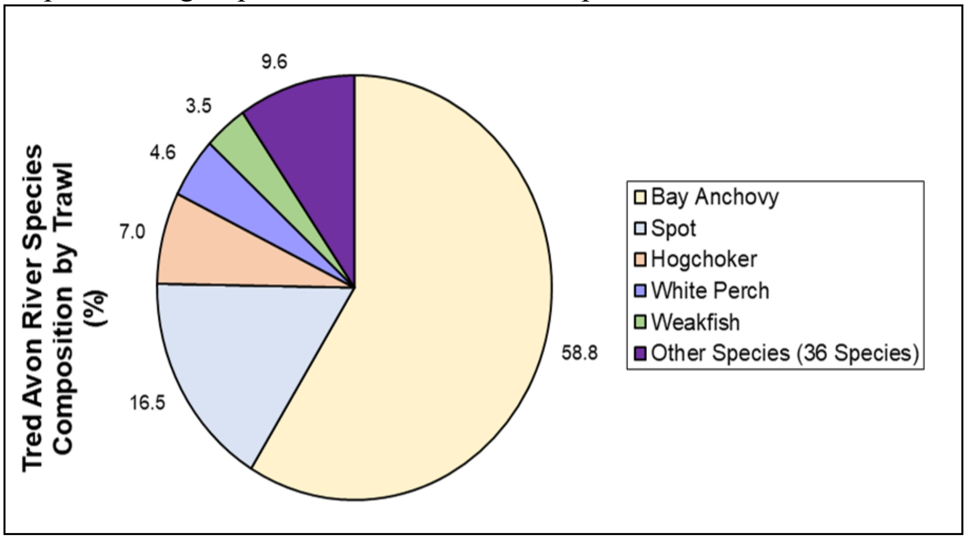


Figure 3-16. Finfish species composition for 4.9 m bottom trawl catch in Tred Avon River for each year sampled. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

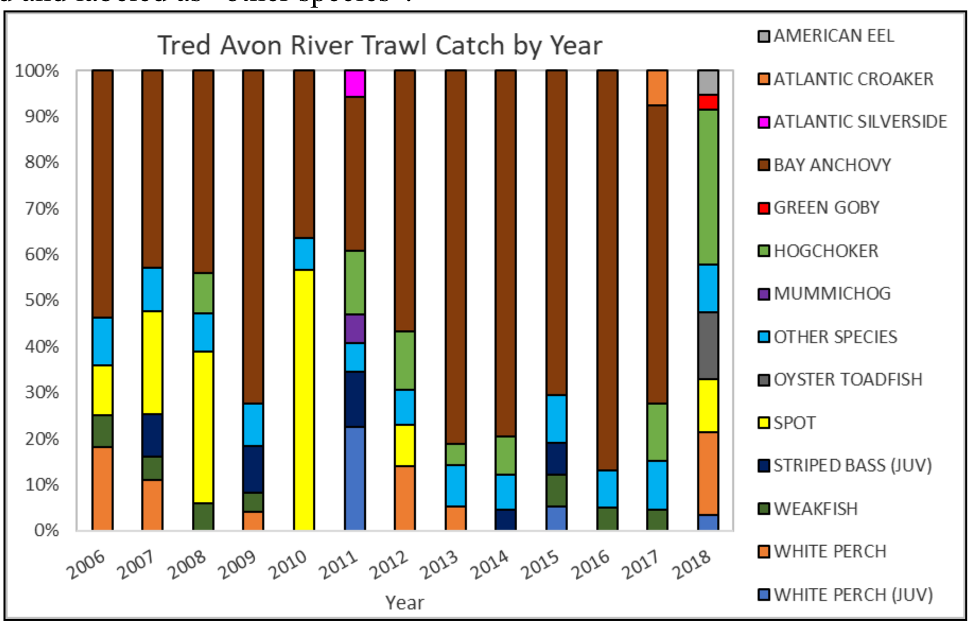


Figure 3-17. Percent similarity index (%) for 4.9 m bottom trawl stations 1-4 in Tred Avon River by year. The greater the similarity value is, the more finfish species there are present and abundant throughout all four bottom trawl stations; lower values indicate finfish species are uncommon and/or scarce throughout all four trawl stations.

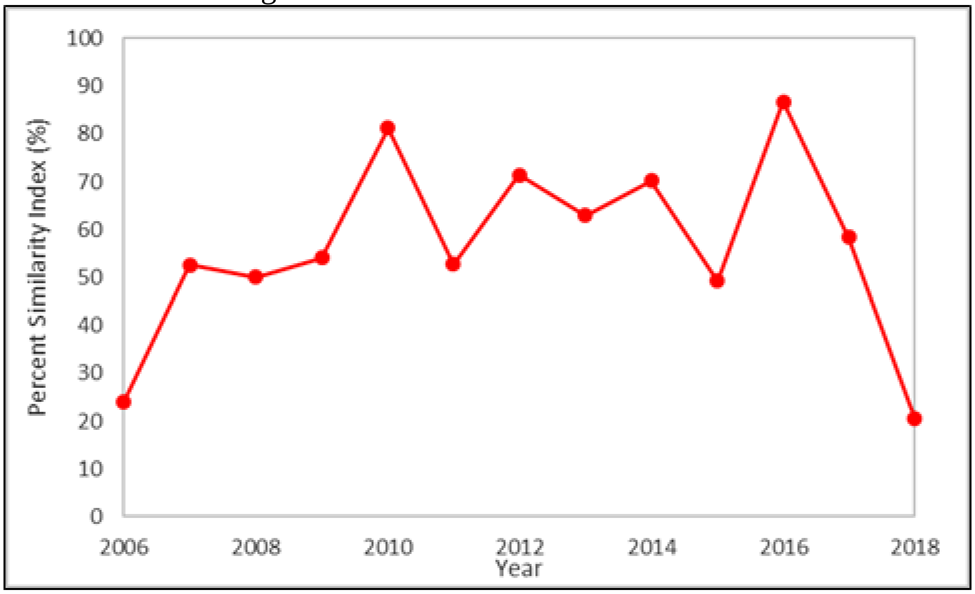


Figure 3-18. Finfish species composition for 4.9 m bottom trawl catch in all mesohaline systems sampled during 2003-2018, by year. Finfish species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

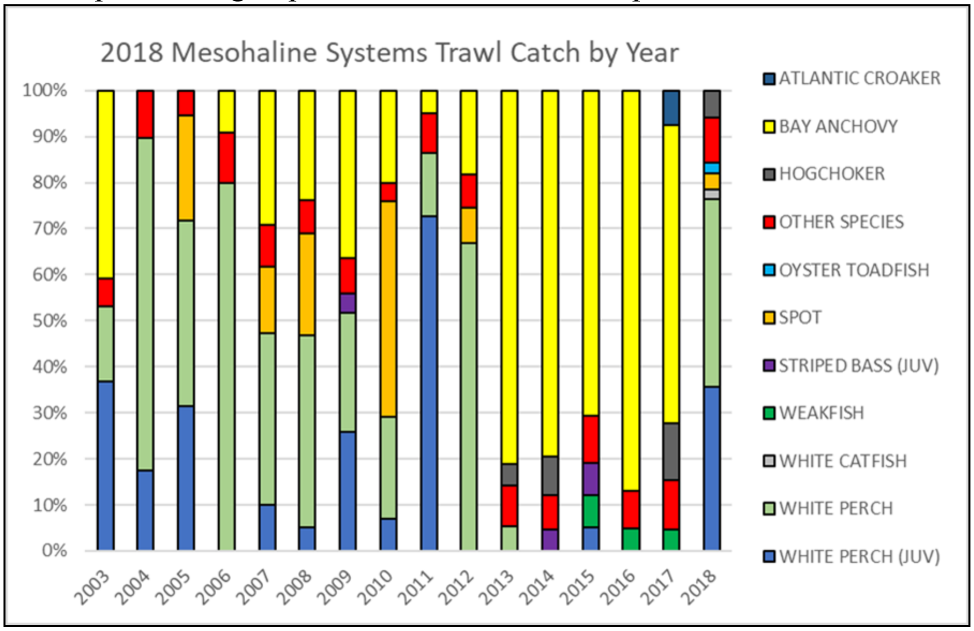


Figure 3-19. Geometric mean (GM) per 4.9 m bottom trawl catch for adult White Perch in Broad Creek (blue triangles), Harris Creek (red squares), and Tred Avon River (black circles), by sampling year.

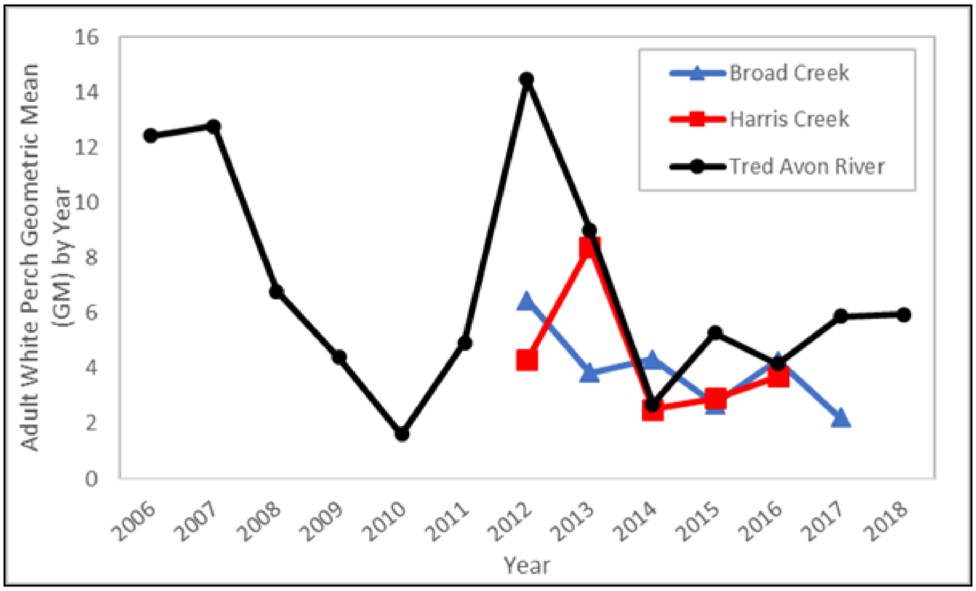


Figure 3-20. Annual beach seine catch geometric mean (GM) per of all finfish species (red squares) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Black bars indicate the 95% confidence intervals.

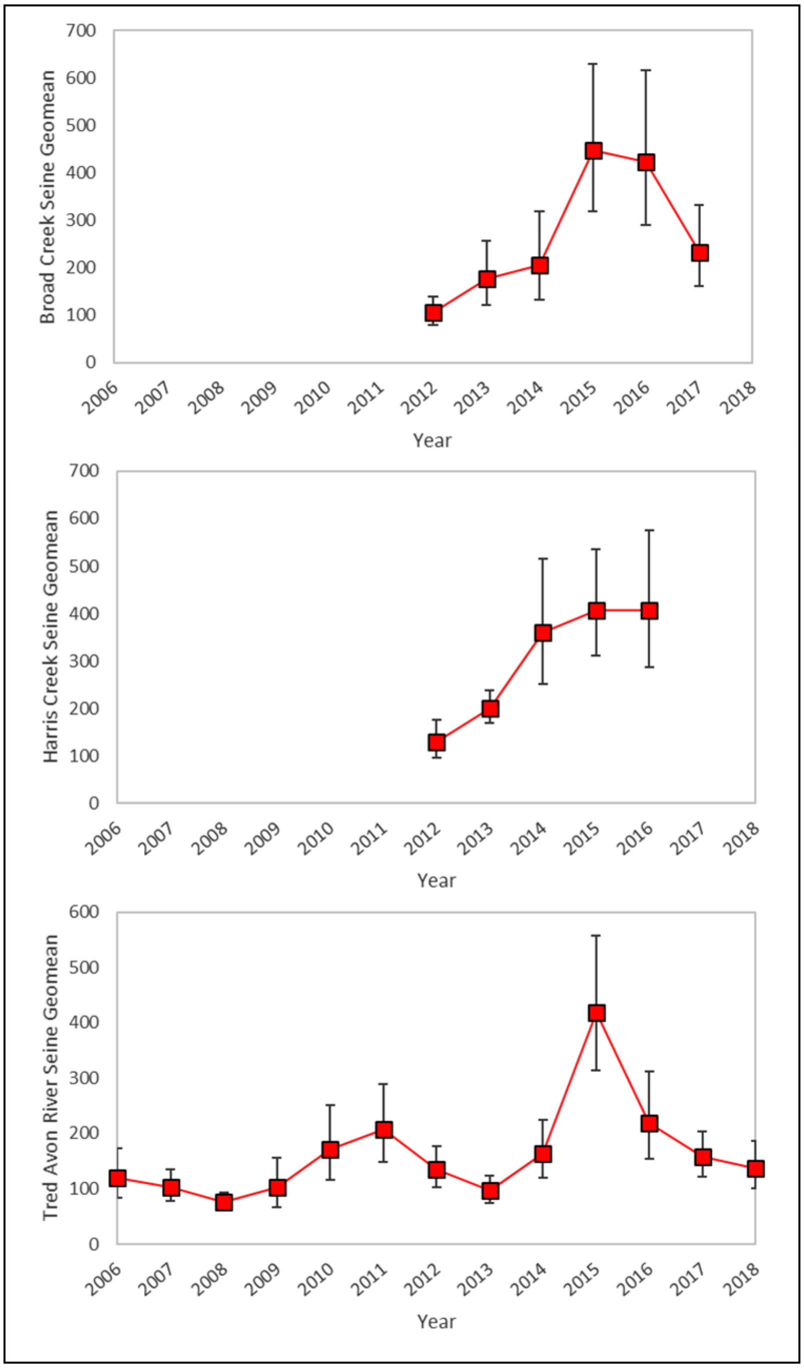


Figure 3-21. Finfish species composition for beach seine catch in Tred Avon River for all years combined (2006-2018). Species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

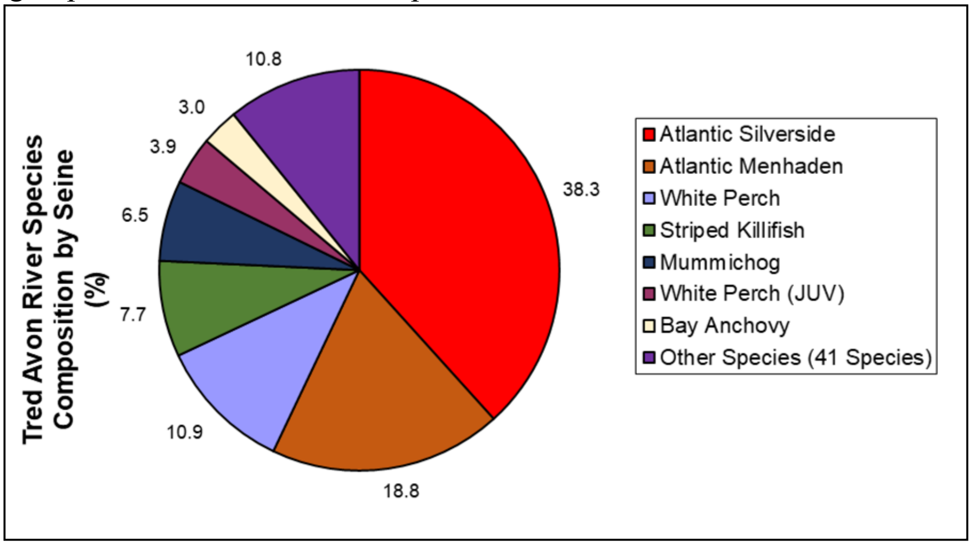


Figure 3-22. Median Secchi depth (m) for Broad Creek, Harris Creek, and Tred Avon River (red squares), by year. Solid black bars indicate the range of Secchi depth (m) measurements by year.

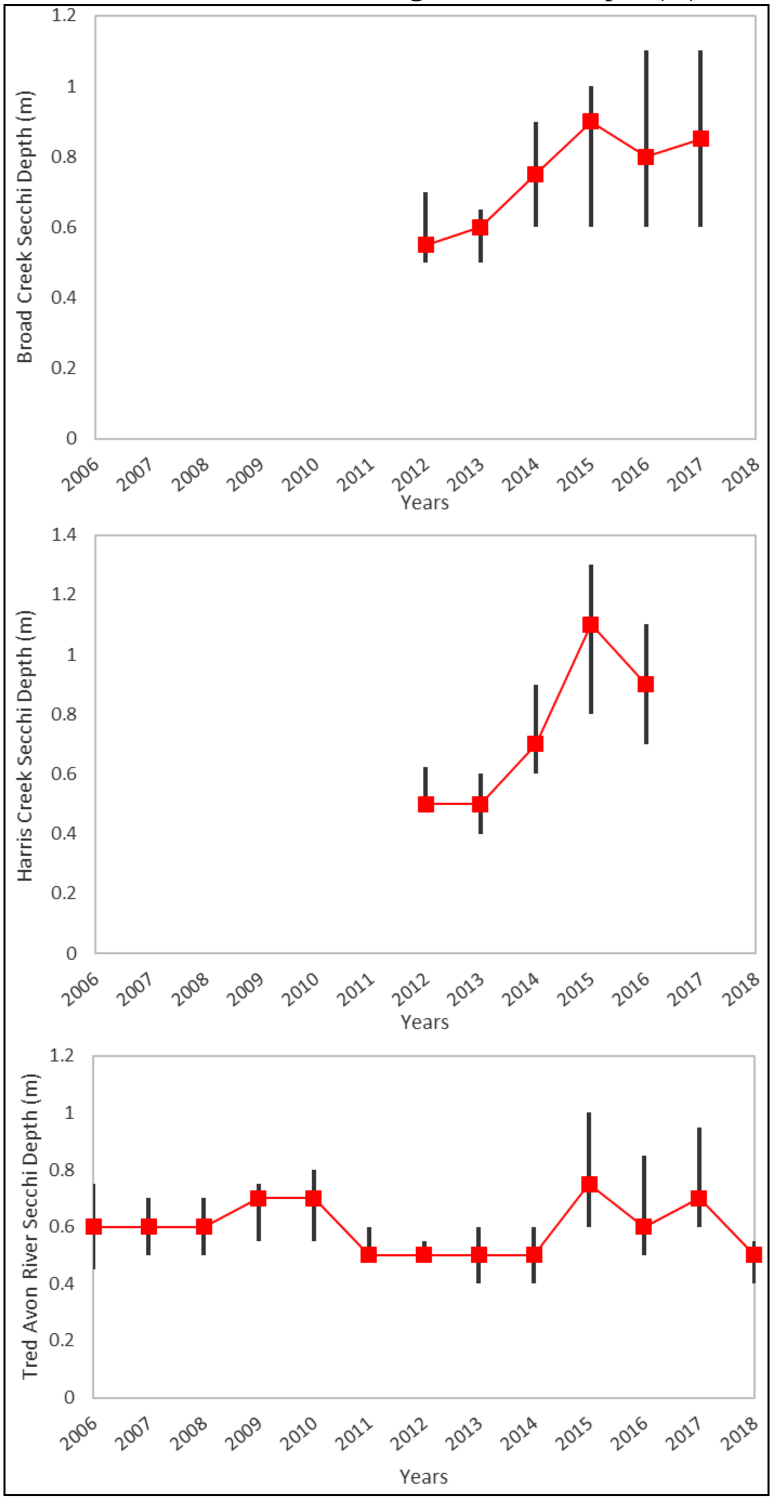


Figure 3-23. Coverage of SAV (percent of coverage in water area) for the mouth of the Choptank (containing Broad Creek, Harris Creek, and Tred Avon) during 1989-2017. Median for the time-series is indicated by the dashed line.

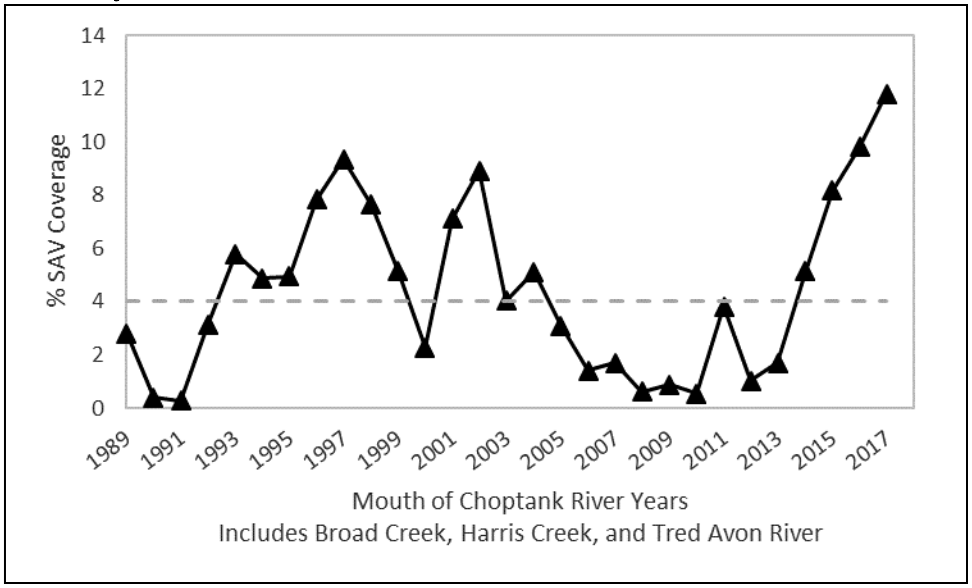


Figure 3-24. Median bottom pH (red squares) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Solid black bars indicate the range of pH measurements by year.

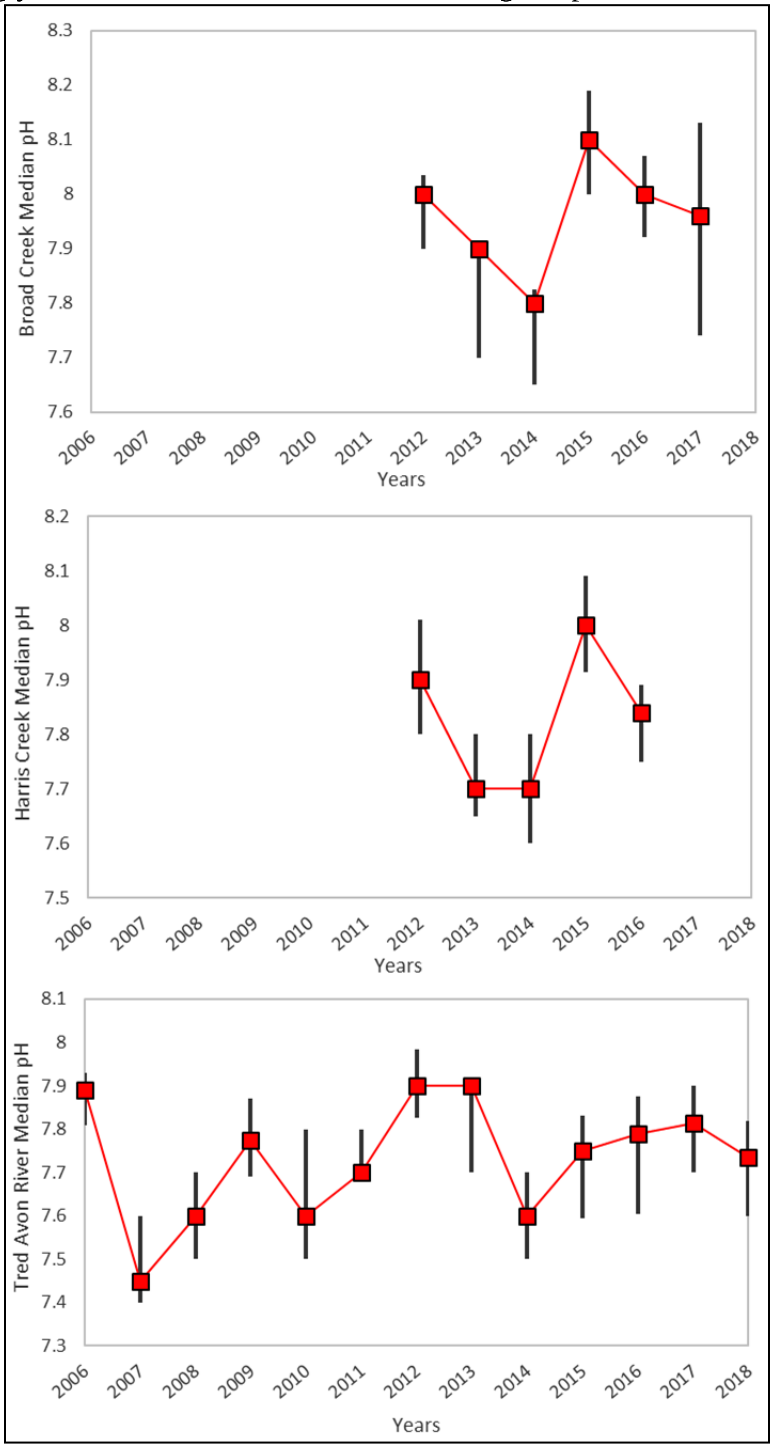


Figure 3-25. Median bottom salinity (red squares; ppt = ‰) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Solid black bars indicate the range of pH measurements by year.

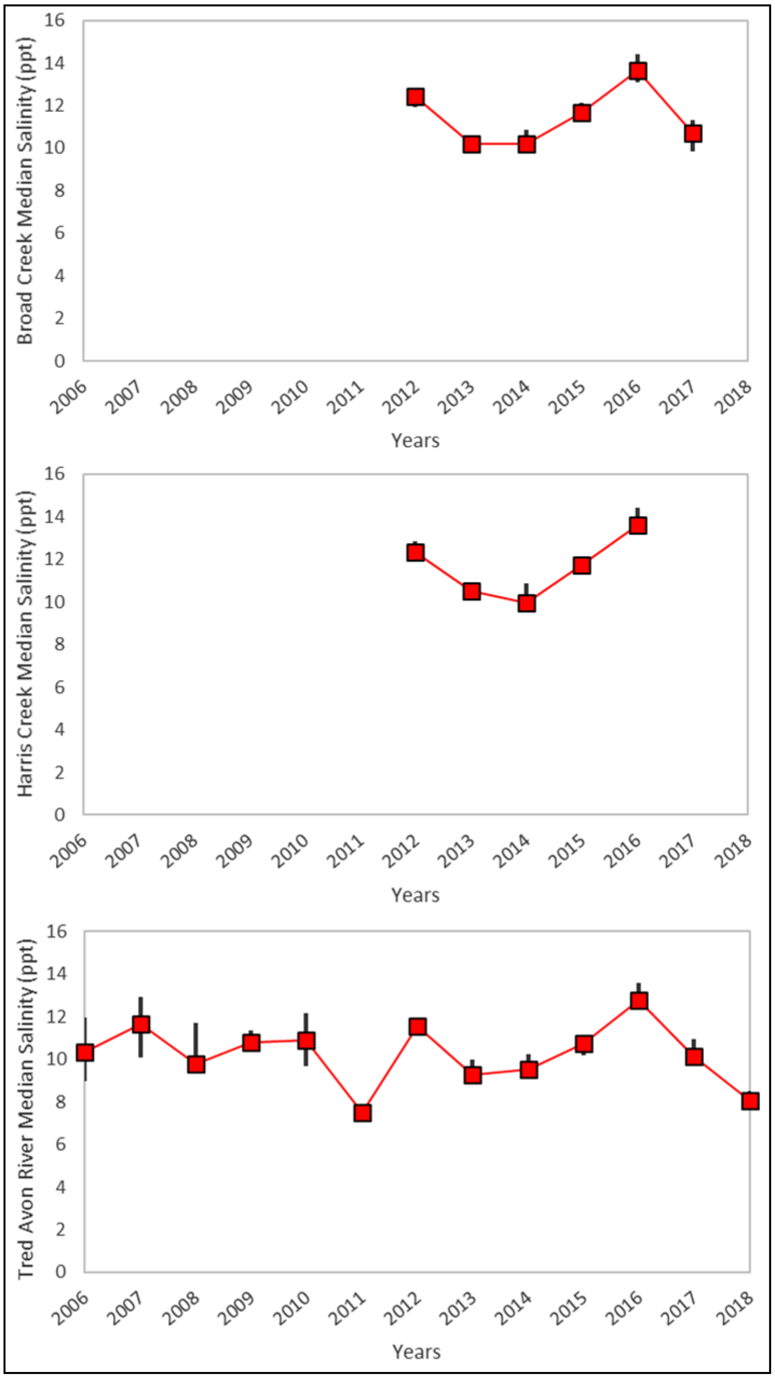


Figure 3-26. Mean monthly precipitation (inches) for Talbot County during 2014-2018. Solid black line indicates the 5 year mean (2014-2018). Dashed black line represents the 25 year mean (1993-2018; NCDC 2019).

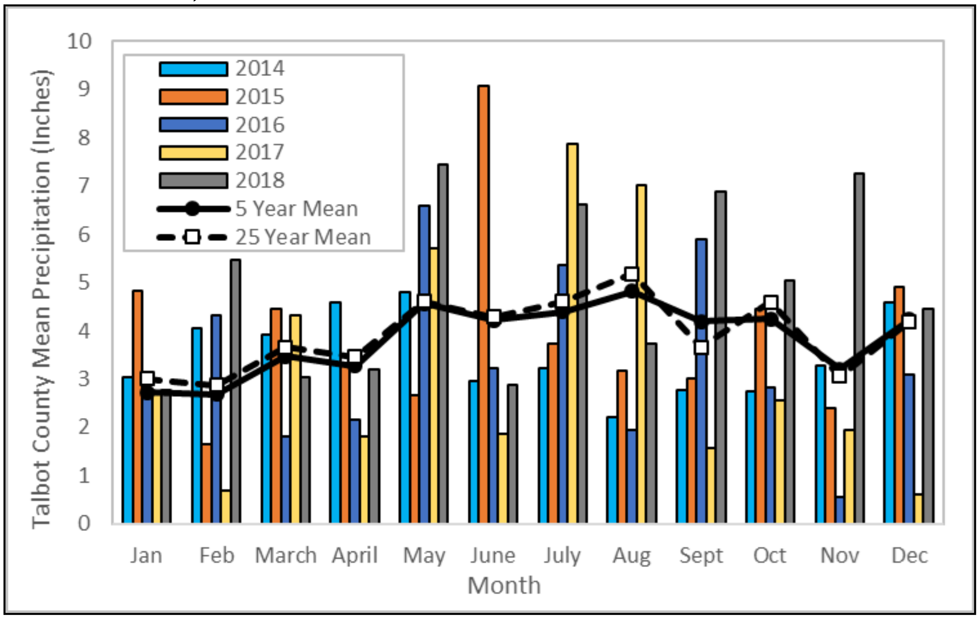


Figure 3-27. Trends in levels of development (structures per hectare = C / ha) during 1950-2016 in watersheds of two subestuaries surveyed, the Chester River and its tributaries, Corsica River and Langford Creek, and the Wye River.

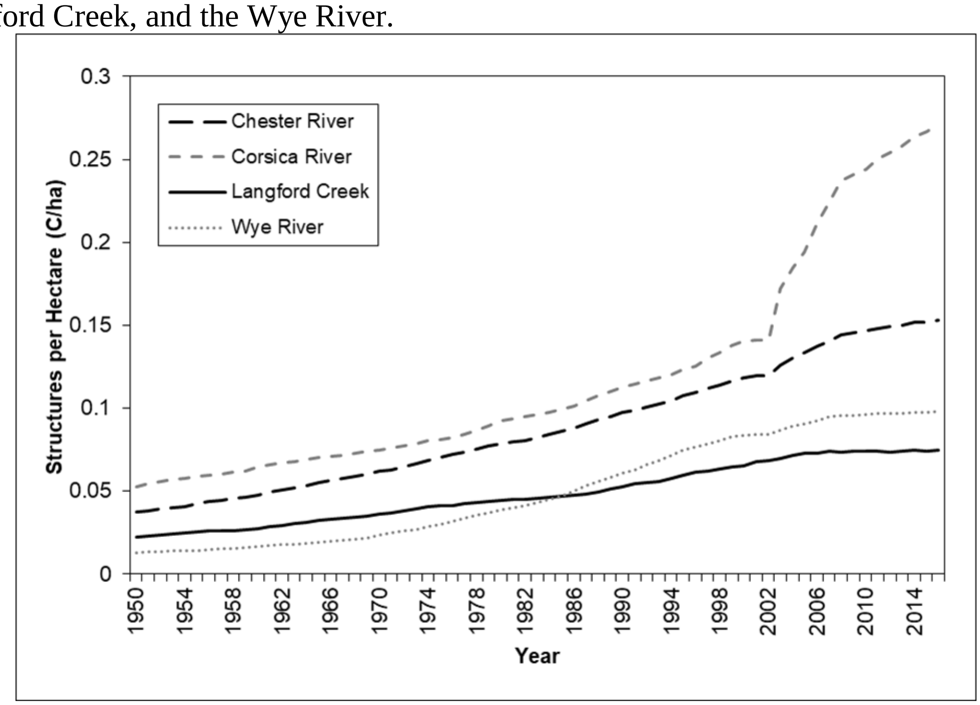


Figure 3-28. Bottom dissolved oxygen (DO; mg / L; 2003-2018) versus intensity of development (C / ha = structures per hectare) in Chester River, Corsica River, Langford Creek, and the Wye River. Target (= 5 mg / L) and threshold (= 3 mg / L) boundaries are indicated (red dashed lines).

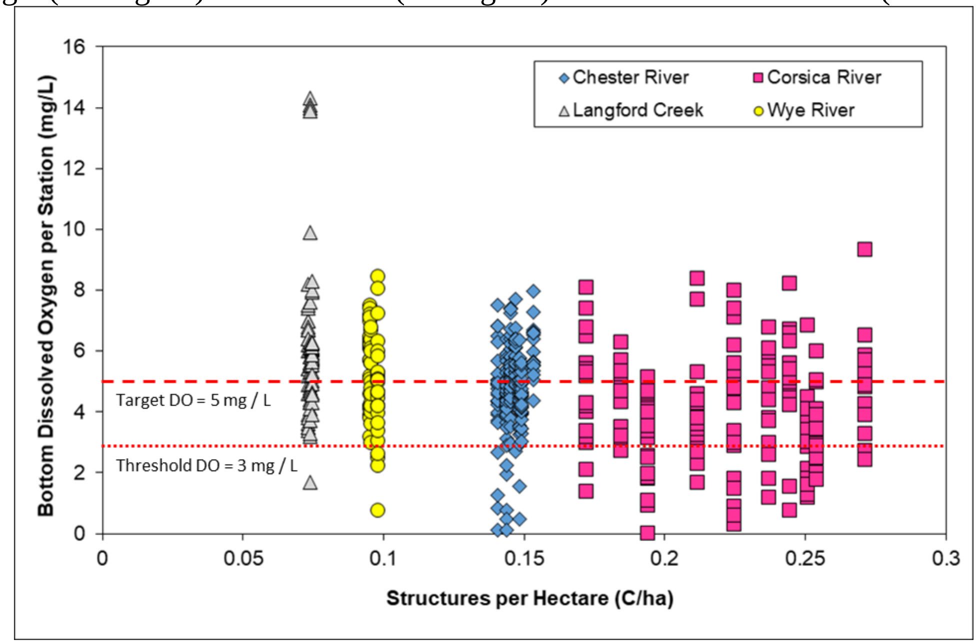


Figure 3-29. Median bottom dissolved oxygen (DO; red squares; mg / L) for Chester River, Corsica River, Langford Creek, and Wye River surveys. Solid black bars indicate range of all bottom DO measurements for that year.

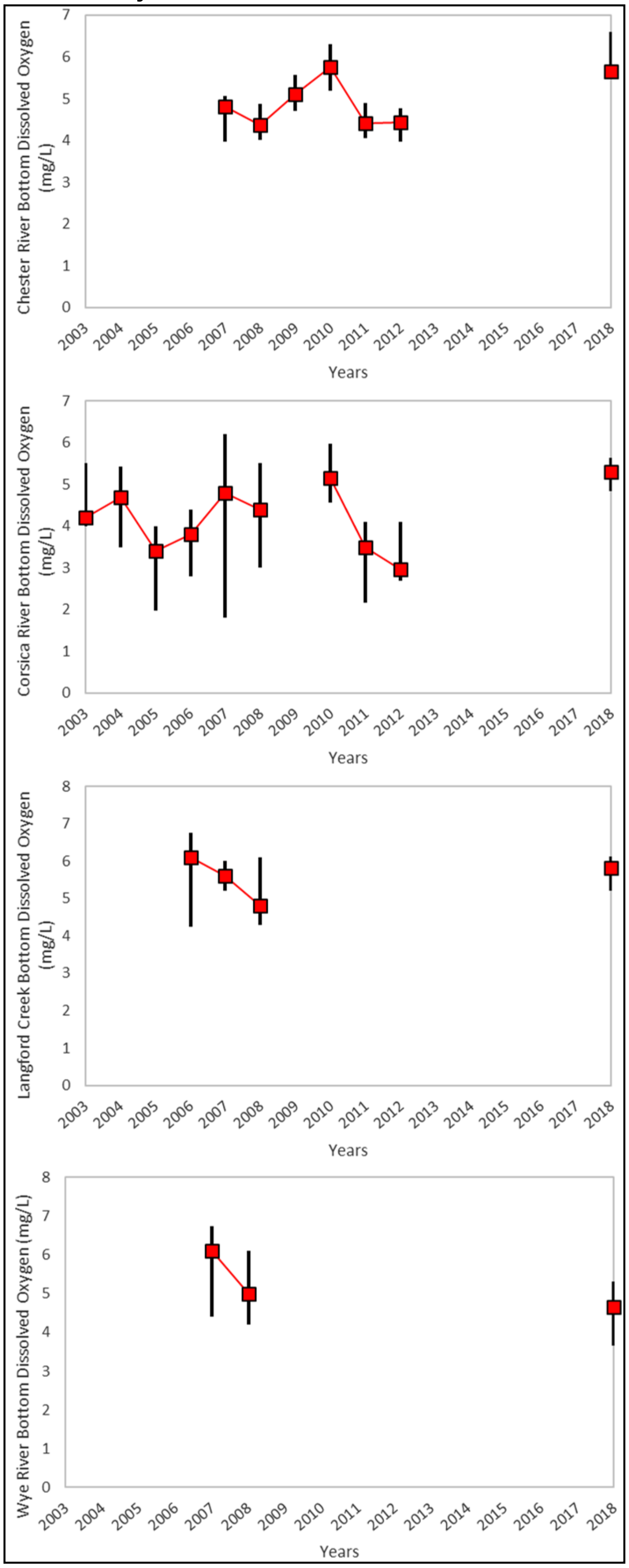


Figure 3-30. Mean bottom dissolved oxygen (DO; mg / L) for all years surveyed for Chester River, Corsica River, Langford Creek, and Wye River, by sampling station. Dotted line indicates the median of all DO measurement data for the time-series available.

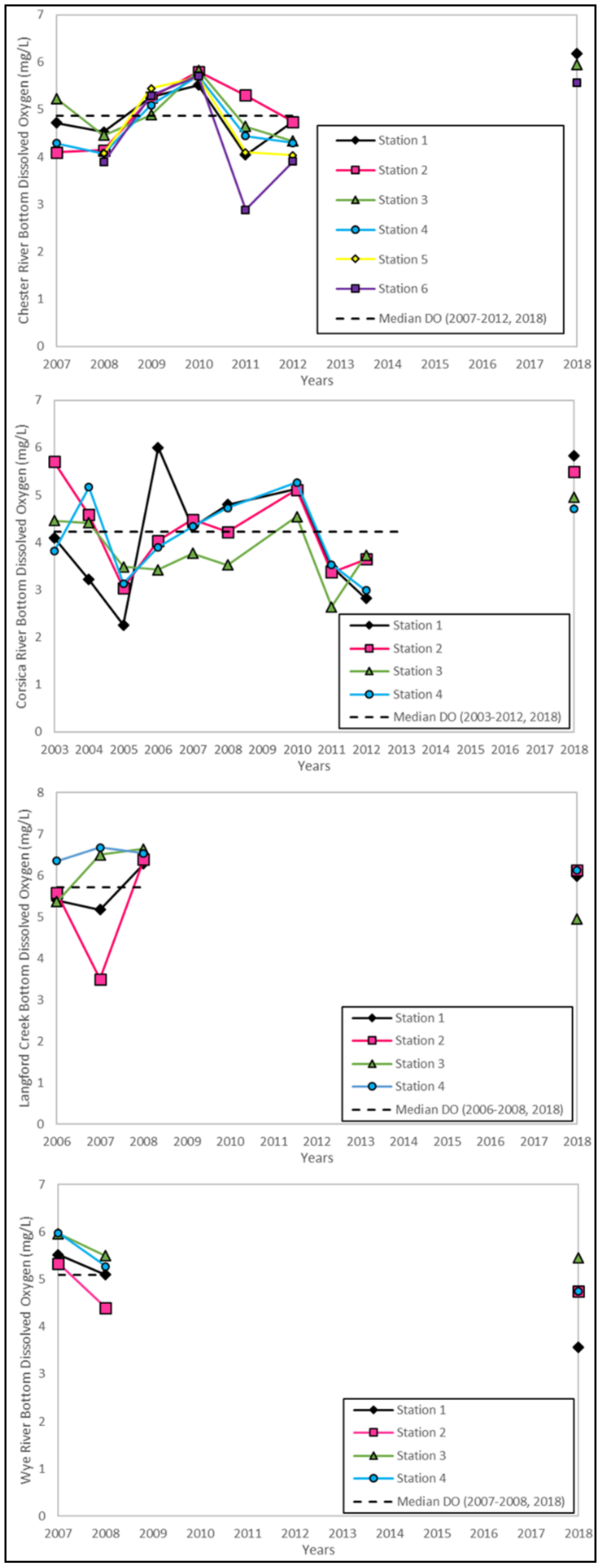


Figure 3-31. Annual 4.9m bottom trawl catch geometric mean (GM) per of all finfish species (red squares) for Chester River, Corsica River, Langford Creek, and Wye River, by sampling year. Black bars indicate the 95% confidence intervals.

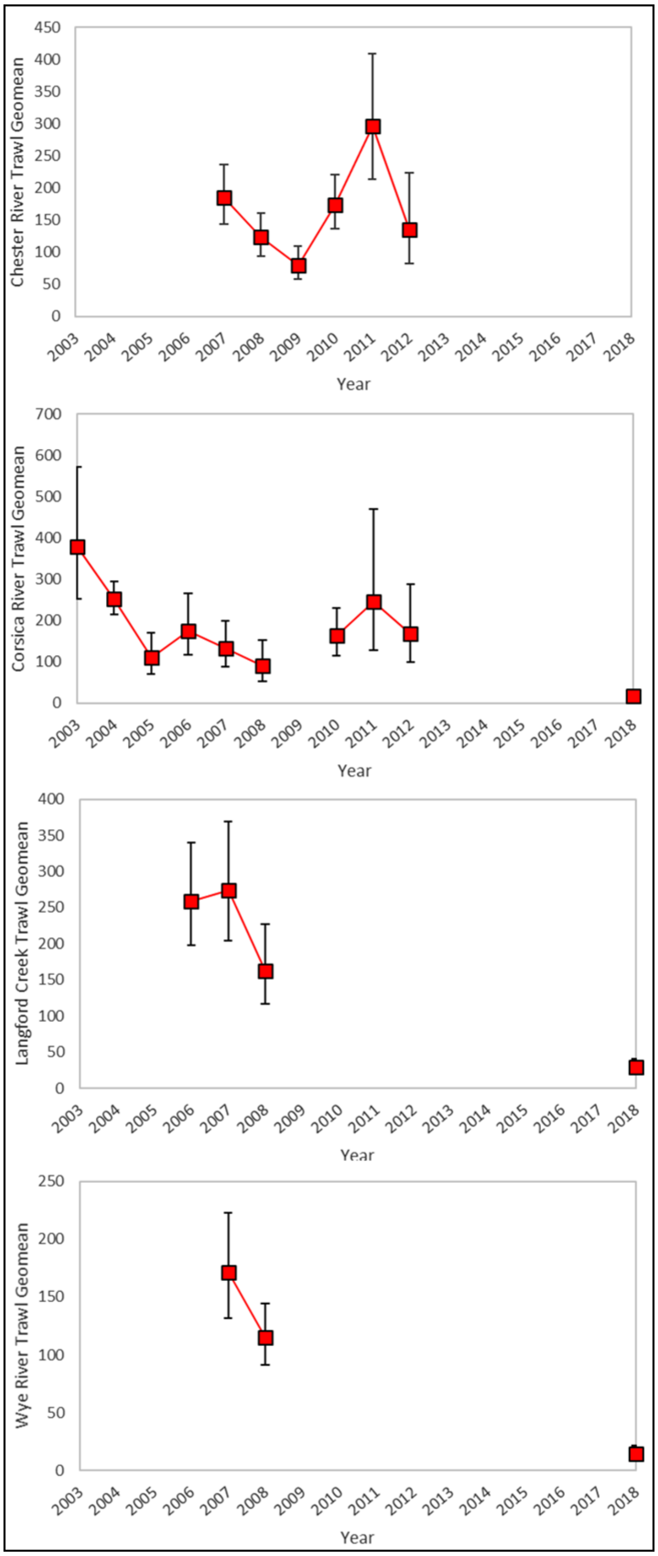


Figure 3-32. Finfish species composition for 4.9 m bottom trawl catch in Chester River (2007-2012), Corsica River (2003-2012, 2018), Langford Creek (2006-2008, 2018), and Wye River (2007-2008, 2018)for all sampling years combined. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

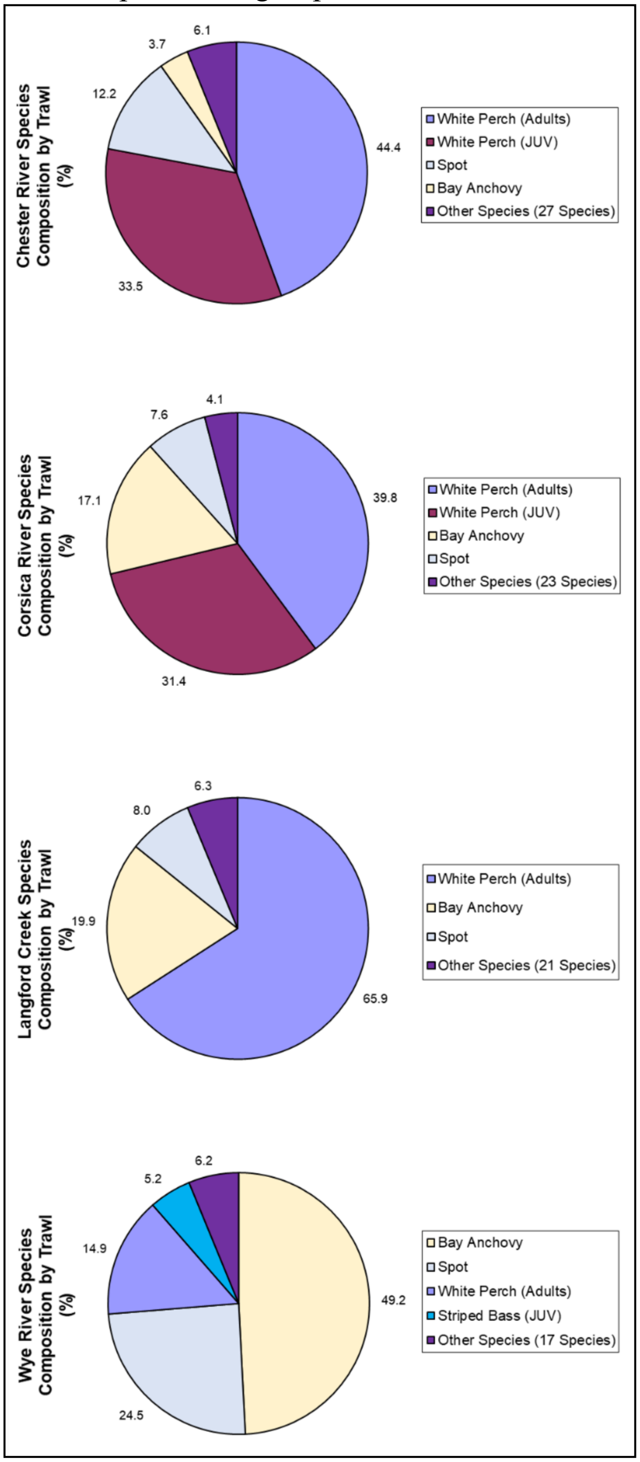


Figure 3-33. Finfish species composition for 4.9 m bottom trawl catch in Chester River, Corsica River, Langford Creek, and Wye River, by year. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

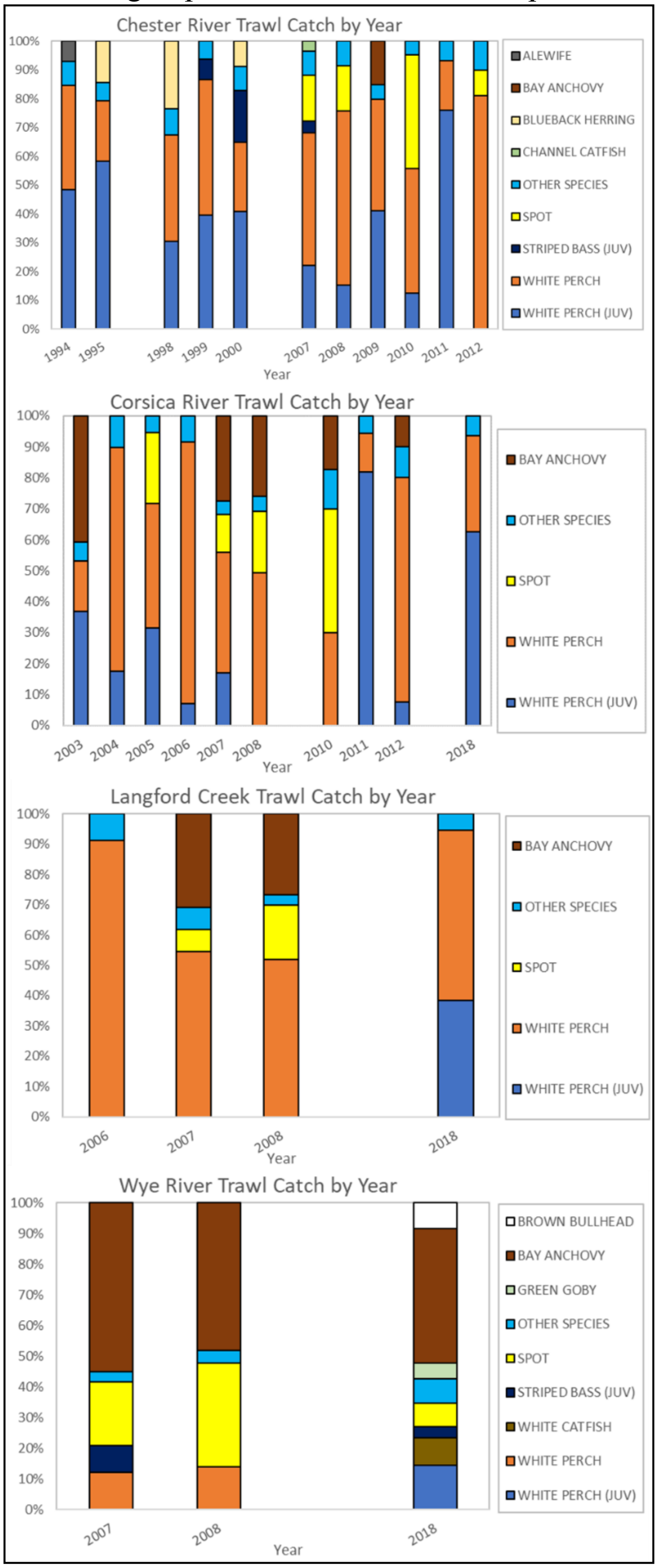


Figure 3-34. Annual beach seine catch geometric mean (GM) per of all finfish species (red squares) for Chester River, Corsica River, Langford Creek, and Wye River, by year. Black bars indicate the 95% confidence intervals.

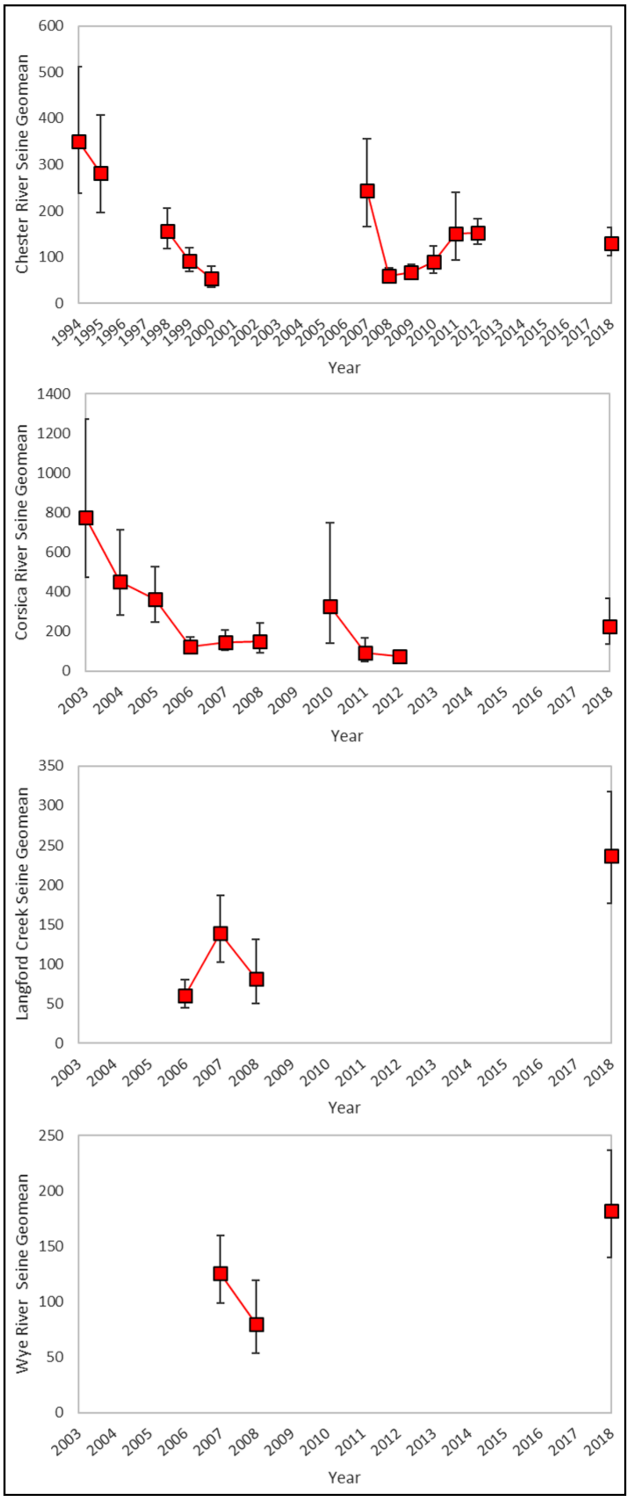


Figure 3-35. Finfish species composition for beach seine catch in Chester River (2007-2012, 2018), Corsica River (2003-2012, 2018), Langford Creek (2006-2008, 2018), and Wye River (2006-2007, 2018) for all years combined. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

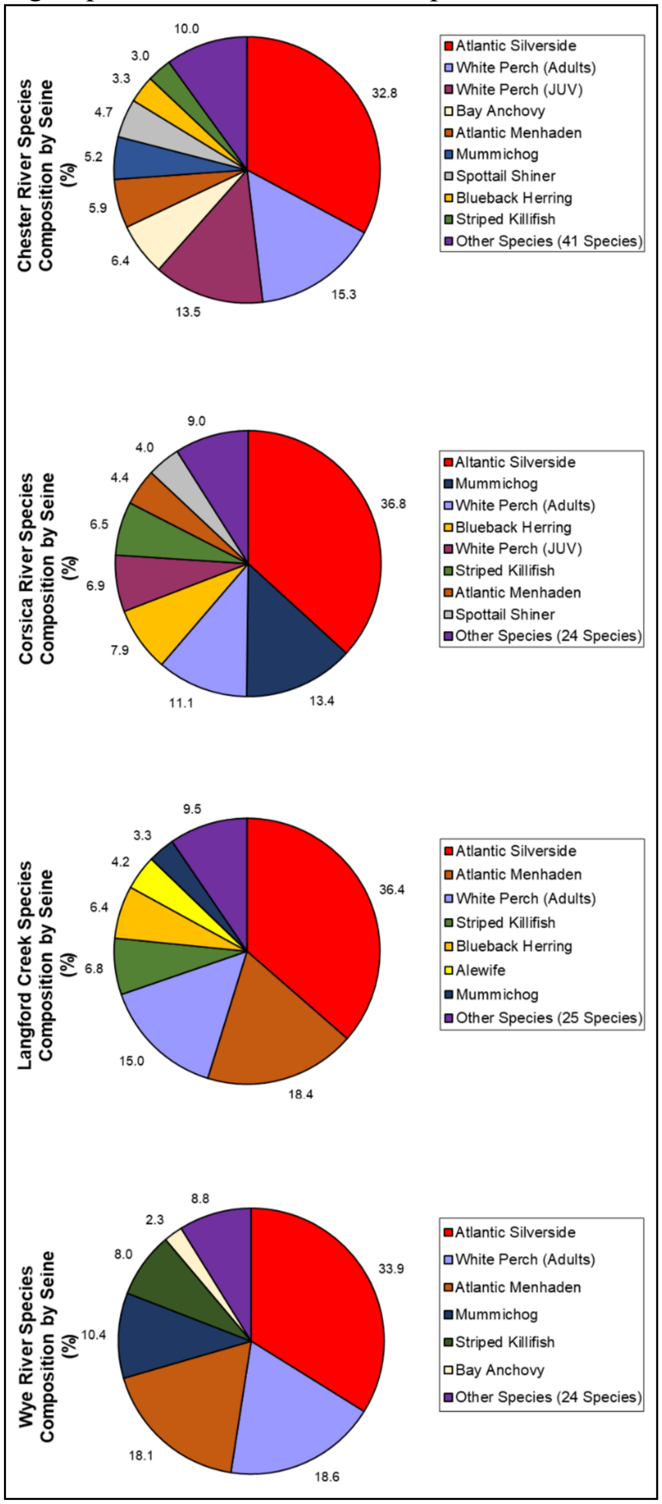


Figure 3-36. Median Secchi depth (m) for Corsica River, Langford Creek, and Wye River (red squares), by year. Solid black bars indicate the range of Secchi depth (m) measurements by year. Secchi depths (m) were not available for Chester River.

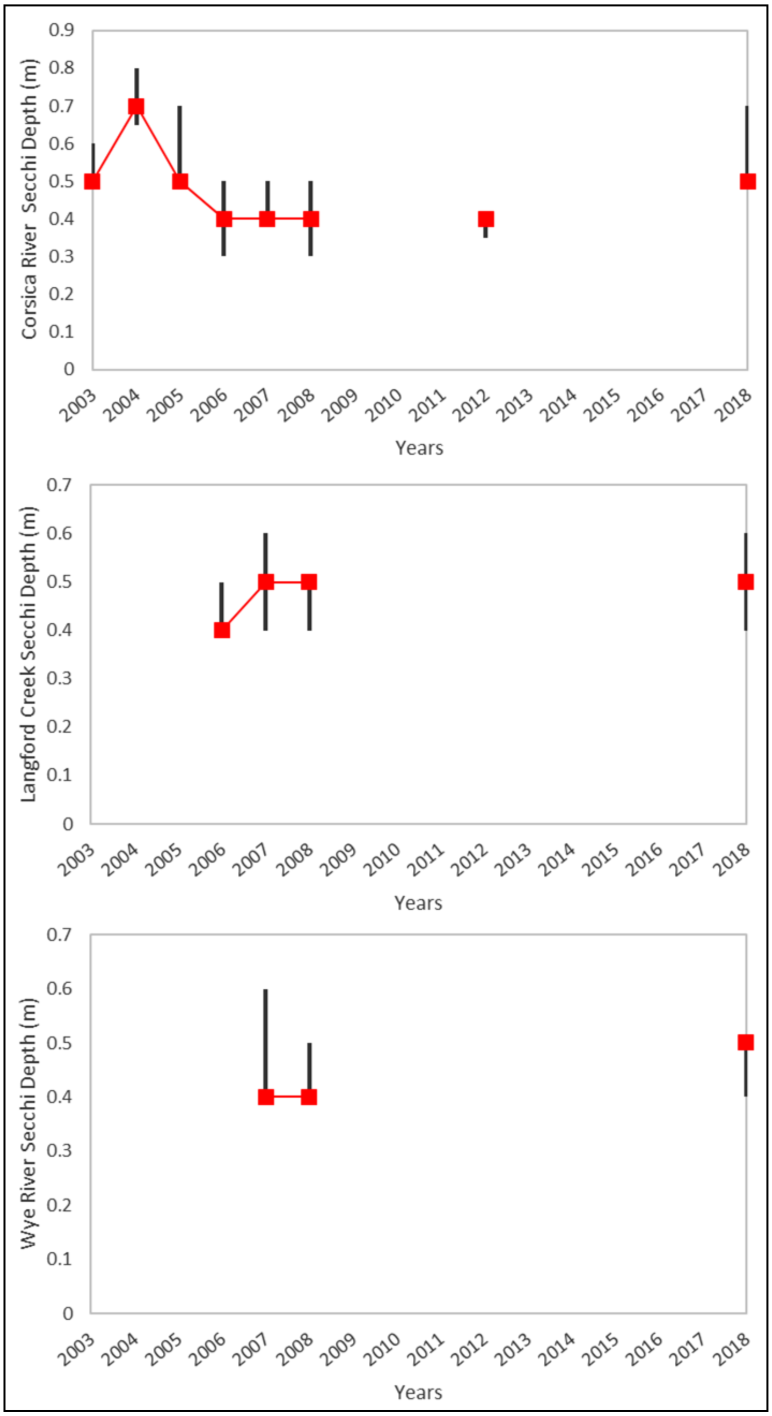


Figure 3-37. Coverage of SAV (percent of water covered) for the Chester River, Corsica River, Langford Creek, and for the Eastern Bay area, including the Wye River, for years, 1989-2017. Several years were excluded due to inadequate mapping. Median of all data available for that time-series is indicated by the dashed line.

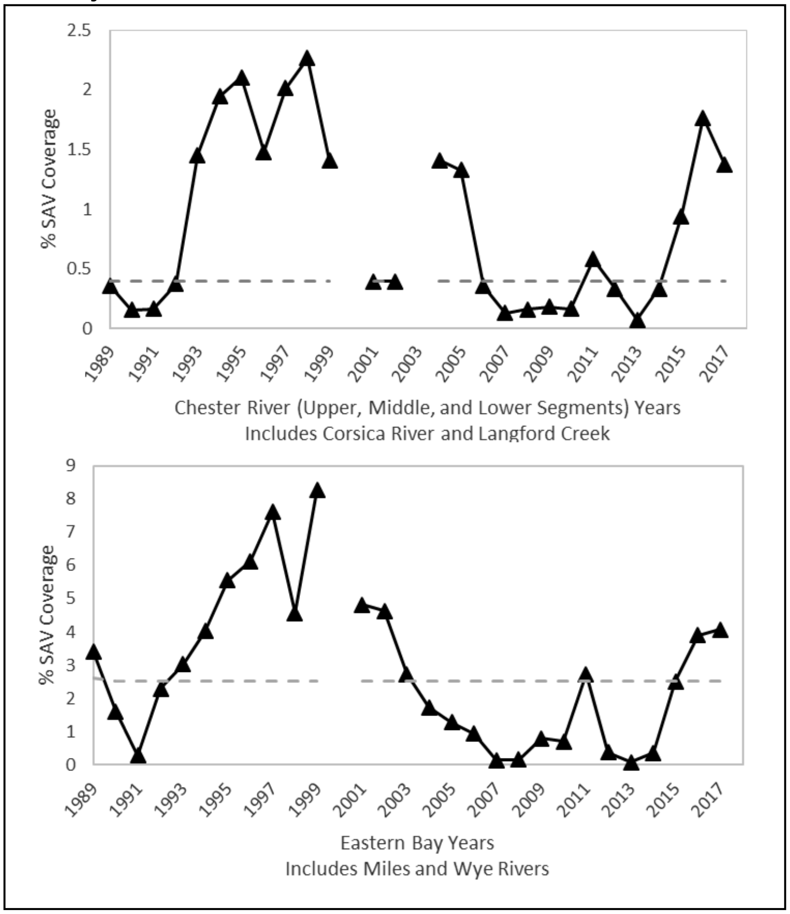


Figure 3-38. Median bottom pH (red squares) for Chester River and its tributaries, Corsica River and Langford Creek, and the Wye River, by sampling year. Solid black bars indicate the range of pH measurements by year.

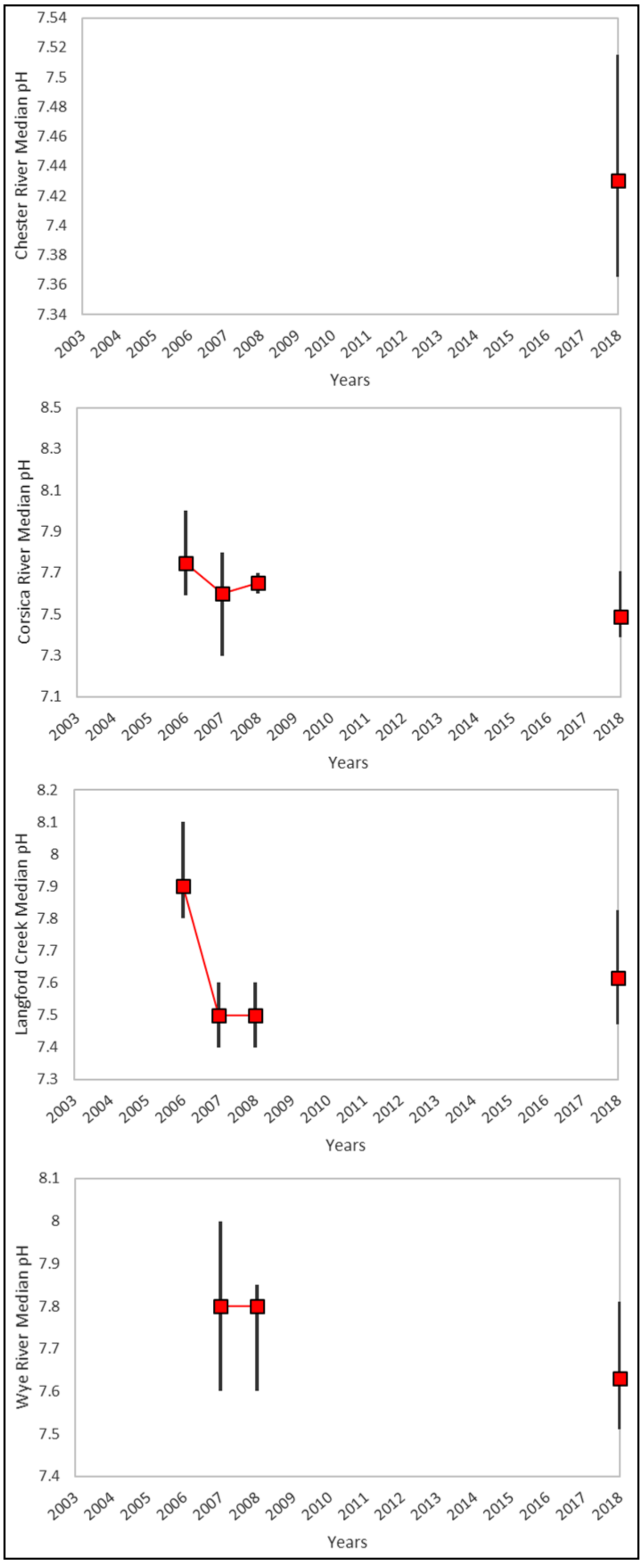


Figure 3-39. Median bottom salinity (red squares; ppt = ‰) for Chester River, Corsica River, Langford Creek, and Wye River, by sampling year. Solid black bars indicate the range of pH measurements by year.

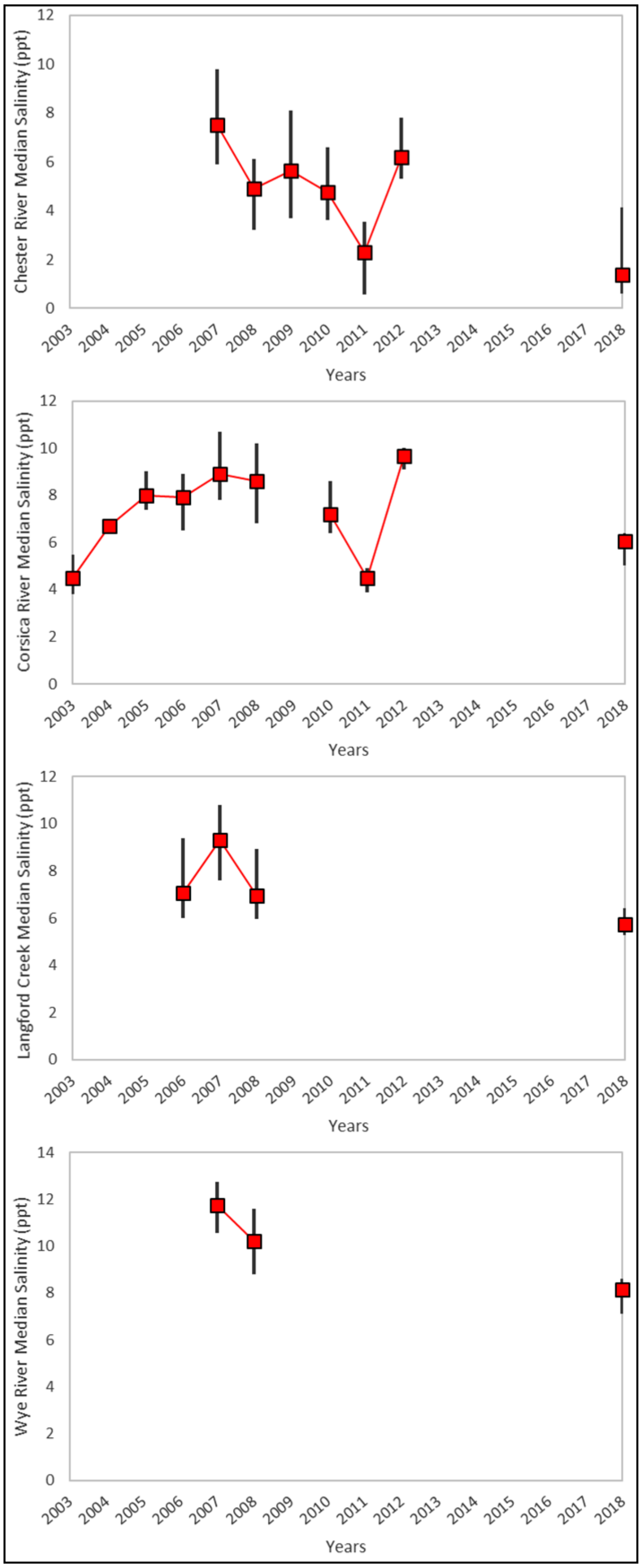


Figure 3-40. Chester River bottom dissolved oxygen (DO; mg / L) measurements versus bottom salinity measurements (‰) during 2007-2012, and 2018. Red dashed lines indicate DO target (5 mg / L) and threshold (3 mg / L).

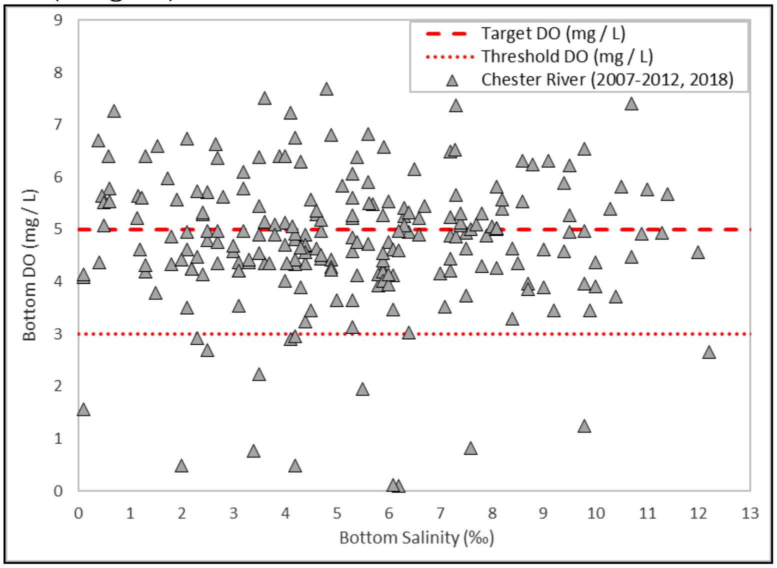


Figure 3-41. Mean monthly precipitation (inches) for Kent County and Queen Anne's County during 2014-2018. Solid black line indicates the 5 year mean (2014-2018). Dashed black line represents the 25 year mean (1993-2018; Queen Anne's County data only available for 2006-2018; NCDC 2019).

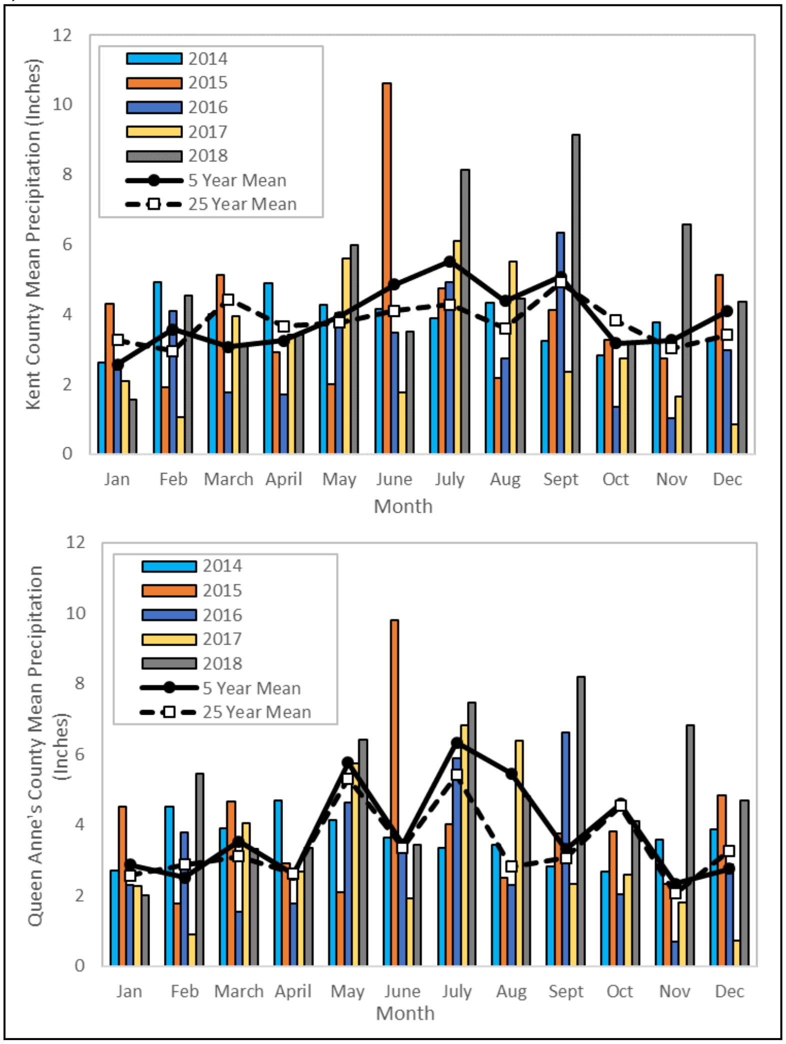
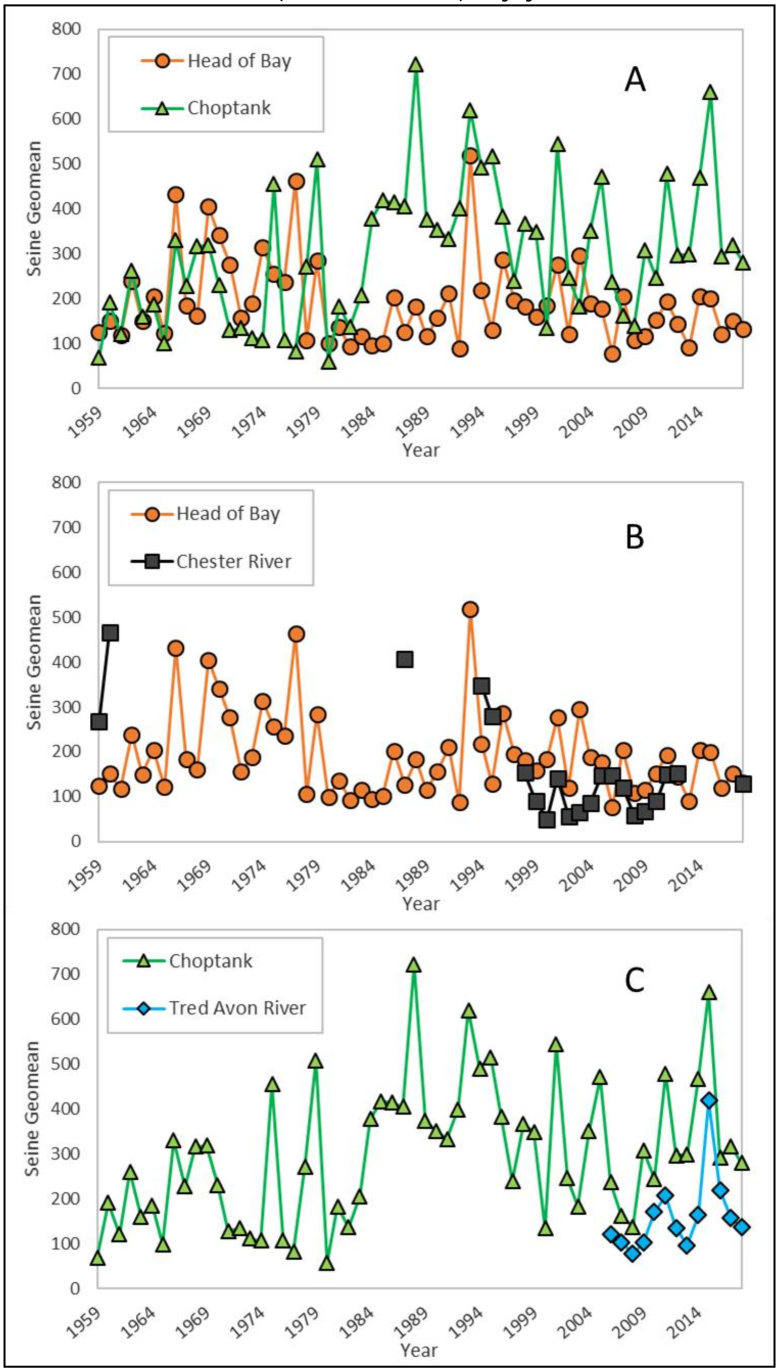


Figure 3-42. Geometric means (GM) of annual beach seine catch during 1959-2018 for all finfish species in the Chester River (black squares), Head of Bay (orange circles), Choptank River (green triangles), and Tred Avon River (blue diamond), by year.



Synthesis of Job 1 Findings

Jim Uphoff

The objectives of Job 1 were to (1) assess land use's effect on recreationally important fish populations and fish communities in tidal tributaries to Chesapeake Bay and use this information to aid fisheries and land-use management decisions. (2) Establish land-use reference points to prioritize watershed protection and restoration efforts for fish habitat, provide a planning tool for the governance of growth, and provide a quantitative basis for managing recreational fisheries in degraded environments. (3) Develop guidelines for conserving or restoring spawning and nursery areas and habitats supporting productive recreational fisheries.

Job 1 investigated two general alternative hypotheses relating recreationally important species to development and-or agriculture, the major human induced land-uses in Maryland's portion of the Chesapeake Bay watershed. The first hypothesis was that there was a target level of a particular land-use that did not significantly alter habitat suitability and the second was that there was a threshold level of land-use that significantly reduced habitat suitability, leading to diminished production. 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, distribution, etc.).

Habitat based reference points reflecting impervious surface (ISRPs) were proposed for Chesapeake Bay estuarine watersheds based on F-63 research which documented responses of DO and fish communities in mesohaline subestuaries (Uphoff et al. 2011). A target level of development for fisheries; a rural watershed, was indicated at 5% IS (0.37 C/ha). This was considered a "safe" level of development where ecological dysfunction was unlikely to impact productivity of fish habitat. Ten percent IS (suburban watershed; 0.86 C/ha) represented a development threshold where deterioration of ecological function supporting productive recreational fisheries was likely and fishery problems would result. Compensation for problems using traditional fishery management tools (harvest management and stocking) was increasing unlikely to overcome habitat issues by 10% IS.

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; MD Fishing and Boating Services 2017), and summer habitat in fresh-tidal, oligohaline, and mesohaline subestuaries (Uphoff et al. 2011; 2015; 2016; 2017; 2018). Through research under Job 1, we have identified negative consequences of watershed development on Bay habitat of sportfish beyond low bottom DO: altered flow and organic matter regimes, reduced Yellow Perch larval feeding success, disruption of normal Yellow Perch egg and larval development due to endocrine disruption (Blazer et al. 2013), increased Herring spawning stream conductivity (salinization of freshwater), and the possibility of chronic low DO and acute ammonia toxicity in extensive SAV beds in low salinity subestuaries.

Watershed restoration activities need to take into account degree of development to set realistic expectations for fish habitat function. Watershed development involves shifts in key ecological patterns in streams (urban stream syndrome; Walsh et al. 2005) and marine ecosystems (Todd et al. 2019) from effects of multiple stressors that are nonlinear, cumulative, poorly understood, detrimental, and difficult to overcome. Restoration activities driven by requirements for nutrient reductions necessitated by mandated Total Maximum Daily Loads may not encompass the full extent of stressors limiting fish habitat function. For example, stream restoration in urbanizing watersheds aimed at reviving anadromous fish production will need to consider restoring flow patterns and reducing conductivity close to background levels. Wetlands may have a positive impact on multiple stressors through the influence of organic matter and sequestration.

Agriculture was much more compatible with productive fish habitat than development in Maryland's Chesapeake Bay subestuary watersheds. Rural features (agriculture, forest, and wetlands) that are positive influences on fish habitat quality were negatively correlated with development. Agriculture is intensively managed in Maryland and predominantly agricultural watersheds have often been "best" from a tidal fish habitat perspective.

Proactive watershed based fisheries management involves engaging in comprehensive growth planning that is largely the jurisdiction of local government in Maryland. Managing this growth with an eye towards conserving fish habitat is important to the future of sportfishing in Maryland. ISRPs 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. We have used this information to guide planning and zoning (Interagency Mattawoman Ecosystem Management Task Force 2012) and Chesapeake Bay fisheries management (Uphoff et al. 2011; MD Fishing and Boating Services 2017). Information developed in Job 1 has provided the basis for continuous outreach efforts (described in Job 2 of our annual reports) with local and state agencies that manage growth and interested stakeholders that influence growth policies in their jurisdictions.

References

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

Jim Uphoff, Margaret McGinty, Alexis Park, Carrie Hoover, and Marcus Patton

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 and forums 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. We updated spawning habitat information to include historical maps of spawning habitat distribution in response to a constituent's request. The website can be found at https://dnr.maryland.gov/fisheries/Pages/FHEP/index.aspx.

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 comprehensive growth plans for Calvert County, 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 – J. Uphoff exchanged information with a North Carolina State researcher, Dr. Jacob Krause, on how increased predation depleted weakfish in the 2000s. Earlier staff analysis indicated that a rise in natural mortality led to a collapse of weakfish.

J. Uphoff and M. McGinty held discussions with Alex MacCleod on Yellow Perch egg hatching experiments and related research that was conducted during spring, 2019. An outreach article on this work can be found at https://agnr.umd.edu/news/doctoral-student-revives-multi-agency-research-collaboration-assess-why-yellow-perch-rivers.

Presentations and Outreach – J. Uphoff participated in three poster presentations focused on working with local government planning agencies to conserve fish habitat given to (1) NOAA's director, (2) Maryland DNR's Secretary, and to (3) Talbot County economic development staff at the Oxford Cooperative Research Laboratory.

J. Uphoff, M. McGinty and A. Park presented at the 148th American Fisheries Society Annual Meeting in Atlantic City, NJ. Uphoff presented on *Potential Time-Varying Forage Reference Points for Atlantic Menhaden*. Park presented McGinty's presentation on *Fish Habitat*

Management in Changing Chesapeake Bay Watersheds: Developing Sound Science to Guide Policy.

A. Park presented on 2017 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 participated in a workshop that explored research needs in the Baltimore Harbor Watershed.

J. Uphoff presented on striped bass forage reference points for a sport fishing club.

J. Uphoff presented on links among watershed development, organic matter, and yellow perch larval feeding success for a NOAA webinar.

J. Uphoff gave a presentation on F-63 herring spawning habitat work for a Chesapeake Bay Herring Workshop. M. McGinty, A. Park, and C. Hoover attended the Herring Workshop.

A. Park and C. Hoover attended the Hatchery's pre-production meeting to discuss yellow perch hatch experiments in partnership with the Mattawoman Watershed Society. Staff developed sampling protocol that would be required to move forward with the experiments. Unfortunately, the Mattawoman Watershed Society citizen scientists were unable to conduct the experiments.

M. McGinty reviewed and commented on the Phase III Waterway Improvement Program document.

J. Uphoff, M. McGinty, A. Park, and C. Hoover attended the AFS Tidewater Chapter 33rd Annual Meeting. J. Uphoff presented watershed development, organic matter, and yellow perch larval feeding success at the meeting.

At Fishing and Boating Services' director's request, J. Uphoff and M. McGinty began assembling information on Spot and Croaker to address lower Eastern Shore stakeholder concerns about declining catches of these species. This lead to the development of the oyster bottom benthic organism forage index described in Job 3.

J. Uphoff gave a presentation on declining habitat of anadromous shad and herring in Patuxent River to the Patuxent River Commission. The purpose of the presentation was to generate awareness of habitat deterioration and to ask local governments to consider stormwater and pollution control methods that might address a wider array of issues than just nutrients and sediment alone. A briefing document on this issue was included in the Job 2 appendix in Uphoff et al. (2018).

J. Uphoff and M. McGinty prepared information on fish habitat for NOAA at Oxford Lab for a NOAA poster on a proposed Chesapeake Bay fish habitat assessment. The poster was presented during a tour by two staffers of Maryland's U.S. senators. J. Uphoff attended the presentation, answered questions, and supplied comments.

M. McGinty presented *Hows That Habitat Working for You? Preserving and Conserving for Comprehensive and Environmental Plans* at the meeting of the Maryland Chapter of the American Planning Association. This presentation described the links between productive fish habitat and sound planning.

M. McGinty met with restoration staff to determine if debris dams placed in Cattail Creek would be an impediment for fish passage. MDE reviewers were concerned they would block passage. A recommendation for future restoration was to provide navigable channels to allow fish to pass under structure.

A. Park presented to Anita C. Leight Estuary Center volunteers on the 2018 summer seine data they collected in the Bush River. Each year the Estuary Center holds a training were FHEP presents on the previous year's data and explain what the data means.

M. McGinty gave a presentation to the Severn River Association on the Yellow Perch Case Study that was conducted in 2003-2005. This set the background for Alex MacLeod (Doctoral Student at UMD), to present his work which is focusing on evaluating potential problems with reproduction of Yellow Perch. The presentation was well received and folks were interested in continuing research to understand what is limiting Yellow Perch production in urban watersheds.

ASMFC – J. Uphoff was part of an ASMFC workgroup developing forage reference points for Atlantic menhaden. He developed an Altantic Menhaden - Striped Bass dynamic predator-prey model as part of an upcoming ASMFC assessment.

Chesapeake Bay Program – M. McGinty contributed to final edits of STAC workshop report *Factors Influencing the Headwaters, Nontidal, Tidal, and Mainstem Fish Habitat Function in the Chesapeake Bay Watershed: Application to Restoration and Management Decisions* http://www.chesapeake.org/pubs/397_Hunt2018.pdf.

M. McGinty reviewed and commented on the Chesapeake Bay Program's Fisheries Goal Implementation Team's Habitat Goal Workplan update.

M. McGinty participated in a Panel Discussion to examine linkages between water quality, habitat and fisheries at a meeting of the Chesapeake Bay Program's Fisheries Goal Implementation Team.

M. McGinty participated in a review of a project to establish thresholds for hardened shoreline. Results showed a range of thresholds that were discrete for specific species, therefore a single index could not be developed.

J. Uphoff and M. McGinty met with staff from NOAA CBP to guide research on microhabitat scales stressors on anadromous spawning and nursery habitat. The team received comments regarding our management needs, but did not develop the project accordingly.

J. Uphoff participated in Bay Program workshop *Integrating Science and Developing Approaches to Inform Management for Contaminants of Concern in Agricultural and Urban Settings*. The purpose of the workshop was to identify which BMPs used to meet nutrient and sediment TMDLs could pull double duty and reduce contaminants.

Envison the Choptank – J. Uphoff attended multiple Envision the Choptank meetings and has participated in this watershed exercise since its inception. Envision has the potential to provide a pathway for working with local government on planning that conserves recreational fisheries. A description of Envision can be found at https://www.envisionthechoptank.org/.

JOB 3: Developing Priority Fish Habitat Spatial Tools

Development of a Provisional Index of Hard Bottom Forage Taxa for Recreationally Important Finfish in Maryland's Portion of Chesapeake Bay

Margaret McGinty, Jim Uphoff and Mitch Tarnowski

Abstract

Angler concerns over reduced catch rates of key recreational species prompted investigations of potential habitat stressors. Initial efforts suggested hypoxia and low abundance of polychaetes in soft bottom could have contributed to changes in spatial distribution of resident Striped Bass in spring and summer. Additional concerns regarding declining catch rates for Spot and Atlantic Croaker, and a need to understand forage conditions on hard bottom habitat prompted us to evaluate available epibiotic data from Maryland's Fall Oyster Survey. Evaluation of data spanning 1995-2018 led to development of a provisional index of forage on hard bottom habitat (HBBI) that was weighted for Spot and Atlantic Croaker diets. Regression analysis showed the HBBI slightly declined with time. Comparisons between the HBBI and measures of biomass in soft bottom habitat showed positive significant relationships. Evaluation of the HBBI in relation to historical fishing areas for Spot and Atlantic Croaker showed half of the hard bottom habitats had below average HBBI scores compared to the time series mean. This evaluation shows promise in using these data to complete development of an index that can be used to measure one aspect of habitat change for key recreational species.

Introduction

Anglers have voiced concerns over reduced regional catch rates of resident Striped Bass, Spot, and Atlantic Croaker and have asked Fishing and Boating Services to look for habitat related explanations. This led to an investigation of potential causes related to habitat by the Resource Assessment Service (RAS) of MD DNR and our Fish Habitat and Ecosystem Program (FHEP). Initial concerns were related to declining catch rates of resident Striped Bass in the lower Potomac River and lower Chesapeake Bay during the spring and summer (Uphoff et al. 2016) which led to a coordinated effort between RAS and FHEP to examine habitat-based hypotheses. There was potential for poor dissolved oxygen (DO) to dislocate Striped Bass from the lower Potomac River and a regional decline in the biomass of polychaetes, a staple of Striped Bass diet in the spring (Overton et al. 2015), was concurrent with the regional changes in Striped Bass fishing. While the information available was not sufficient to reach firm conclusions, this effort was appreciated by DNR's recreational fishing advisors and provided them an entryway to ecosystem based fisheries management. Recent reports of declining Spot and Atlantic Croaker fishing in the Chesapeake Bay (both are benthic foragers) reinforced the need to understand the status of benthic forage.

A soft bottom benthic biomass index (invertebrates living in the sediment) has been a component of a Chesapeake Bay benthic index of biotic integrity (BIBI) and provides an accessible summary of status in this benthic habitat (Weisburg et al. 1997). The BIBI has been employed to monitor water quality since 1998. The benthic biomass component consists of 7 polychaetes, 10 mollusks, 1 isopod, 2 amphipods, and 2 ribbon worms (see Table 2-5 in Llansó and Zaveta 2017). Frequency of soft bottom benthic biomass below the time-series median became more frequent after 2009 (Llansó and Zaveta 2017). Uphoff et al. (2018) explored the relationship of this benthic biomass index on resident Striped Bass condition.

Eastern Oyster (hereafter, Oyster) reefs are a major, widespread habitat feature of Chesapeake Bay (Figure 1) and are considered “hard” bottom. One important ecological role that Oysters fill is habitat for commensal macrofauna (Tolley and Volety 2005). Secondary production associated with Oysters and their 3-dimensional reef structure attract numerous invertebrates and fishes (Tolley and Volety 2005) that may serve as a food source for benthic feeding gamefish. Rodney and Paytner (2006) suggested that Oyster bottom can be important to forage dynamics of multiple finfish species. Despite the widely acknowledged value of Oyster bottom as key finfish habitat, we found a paucity of studies documenting this value for gamefish. Acknowledging this, La Peyre et al. (2019) synthesized data from studies in the Gulf of Mexico to establish proposed gear based benchmarks for restoration of oysters in relation to benefits to specific community assemblages. La Peyre et al. (2019) identified several studies examining Spot and Atlantic Croaker associations with oysters. Our review of these studies did not yield strong species specific responses, but drew general conclusions that Spot and Atlantic Croaker may prefer Oyster habitat over adjacent low relief sand or mud bottoms (Peterson et al. 2003, Stunz et al. 2010, Robillard et al. 2010). Simonsen and Cowan (2013) showed diet variations in Atlantic Croaker when comparing fish caught over a restored Oyster Reef to those caught in adjacent mud habitat. Atlantic Croaker diets at reef sites were dominated by mud crabs and other crustaceans compared to mud bottom habitats where diets were devoid of mud crabs, but rich in detritus, bivalves and fish (Simonsen and Cowan 2013). The keystone work in the Chesapeake Bay was done by Brietburg (1999), where she conducted dive operations to record finfish species associations with an Oyster reef in the Patuxent River, Chesapeake Bay. She reported three types of associations between finfish and Oysters: residency, where species such as the Naked Goby, show a high dependency on Oysters; facultative, where species associated with structures, such as Black Seabass, will concentrate on Oyster bars; and transients, species such as Striped Bass and Spot that use a wide range of habitats, but do show periods of high association with Oyster bars. Additionally, Brietburg (1999) observed that larval Naked Goby provided significant forage for Striped Bass juveniles. Harding and Mann (2001) evaluated Oysters to determine if they met the criteria to be classified as “Essential Fish Habitat (EFH) according the Magnuson-Stevenson Fishery Conservation and Management Act of 1996 (Public Law 94-265)”. They sampled fish among three habitat types, a sand bar, a restored intertidal Oyster reef and a subtidal Oyster bar and observed fourteen transient species in their sampling, with Atlantic Croaker, Spot, and Striped Bass among the five most abundant species observed (Table 1 in Harding and Mann 2001). They observed ubiquity of dominant species among the three habitat types, citing that these species were generalists, opportunistically availing themselves of high quality habitat with attendant increased production. However, Harding and Mann (2001) reported a difference in size and abundance that appeared to relate to a gradient of “habitat productivity as enhanced by ecological and structural complexity”. Specifically, larger Atlantic Croaker and Striped Bass were associated with the intertidal Oyster habitat that was more structurally diverse, while the largest Spot were associated with the subtidal Oyster habitat. While they could not support defining Oyster reefs as essential fish habitat, they suggested Oyster habitats are of higher quality than other local habitat types because of the structural complexity associated with higher productivity. Pfirrman and Seitz (2019) compared fish community aspects on a restored Oyster reef with adjacent unstructured bottom and found no significant difference in the fish community among sites, but noted larger fish were associated with structured habitat. Spot and Atlantic Croaker were among the species they examined. The fact that Oyster bottom serves as an attractant to larger fish is likely why recreational anglers have long targeted them as fishing spots (Volety 2013).

Information was absent in Maryland portion of Chesapeake Bay on the status of benthic forage on "hard" (i.e., Oyster) bottom for Striped Bass, Spot, and Atlantic Croaker. The Forage Action Team of the Chesapeake Bay Program has identified a need for information on "hard" bottom" benthic forage. (J. Uphoff, MDDNR, personal communication). This prompted us to investigate unused hard bottom fouling data collected by the Maryland Oyster Fall Survey (Tarnowski 2018) as a potential index of benthic forage on Oyster (or hard) bottom (Hard Bottom Benthic Index or HBBI). We used diet information for Atlantic Croaker and Spot in Chesapeake Bay (Idhe et al. 2014) to examine the potential value of hard bottom habitat to these benthic feeding gamefish.

We used long term "fouling" organism data collected by Maryland's Fall Oyster Survey (Tarnowski 2018) to produce a provisional HBBI. Organisms such as worms, crustaceans, mollusks, sponges and tunicates are part of the Oyster Bar community and consist of many potential benthic food items. Many are attached to the surface of Oyster shells (fouling organisms), while others are mobile. This report describes progress so far in developing the HBBI.

Methods

Fishing Areas compared to Oyster Bars - Given the paucity of studies linking Spot, Atlantic Croaker, and Striped Bass fishing spots to Oyster habitat, we compared mapped fishing spots to mapped Oyster habitat. We used digitized maps of fishing spots in Maryland from *Fishing in Maryland* magazines published for 1960, 1968, 1989 and 2004 (Dillon 1960, 1968; de Russy 1989, 2004). We did not possess a complete series and these were the years available. These maps had been hand digitized from *Fishing in Maryland* paper maps of key fishing areas for each species as reported by guides, bait shops, and anglers. We considered these maps to represent "expert" opinion. We overlaid these maps on Oyster habitat maps (official Maryland Oyster Bar maps) and assessed the distance between fishing spots and the nearest Oyster bar using ArcView measuring tool to determine the relative importance of Oyster habitat to reported key fishing spots. We applied three distance categories: on the Oyster bar (0 m), close proximity to an Oyster bar (within 400 m), and distant from an Oyster bar (> 400 m). We selected 400 m as the outer limit of association with the Oyster bar based on two studies that examined changes in fish community measures related to distance from reef structures. One study reported a halo effect of the reef on distance, with no notable effects at 400m distance (Schultz et al. 2012). Another study evaluated fish dynamics in relation to reef balls, reporting pronounced changes at 300m (dos Santos et al. 2010). We chose the more liberal distance, recognizing that the fishing maps are approximate locations drawn according to angler reports of fishing spots. These in turn were digitized through visual approximation of the fishing spot. Applying a liberal distance should compensate for difference due to visual interpretation, although we have no way to measure error. We calculated percentage of fishing sites in each category for each year data were available for each species and took an overall average to assess angler affinity for fishing over hard bottom (Oyster habitat).

Developing Benthic Forage Indices - The Maryland Fall Oyster Survey (Fall Survey) has been conducted annually since 1939 to assess the condition of the Oyster stock for management (Tarnowski 2018). Over 250 discrete Oyster bars are sampled yearly with multiple samples taken at select bars (Figure 1). Samples were obtained using a 0.8 meter standard oyster dredge. The dredge was recovered and the sample released into a culling box (Tarnowski 2018). A ½ bushel (~17.6 liter) subsample of Oysters was taken for gathering data for population estimates. This subsample was then processed.

Prior to processing the Oysters, relative coverage on all Oysters in the subsample with a specific epibenthic organism was ranked: 0 indicated the organism was absent, 1 indicated few were present, 2 indicated moderate presence, 3 indicated organisms were numerous, and 4 signified the organism was very abundant. (Note, organisms were typically identified to genus level, but in some cases, to class level when field identification was too time consuming). For example, there are 4 species of barnacles in Chesapeake Bay: 3 in the genus *Balanus* and 1 in genus *Chthamalus*. For sampling purposes all 4 are identified as barnacles. These data are available in paper files dating back to the 1970s. For more detailed methodology, see Tarnowski, (2018).

The Fall Survey recorded information on 39 organisms associated with Oyster bars, of which 33 were benthic invertebrates (we did not include plants or fish in our evaluation). Temperature and salinity, were taken at surface and bottom at Key Bars. Key Bars are sampled annually and used in annual estimates of Oyster mortality and spat set (Tarnowski 2018).

We obtained hard bottom benthic community data from the survey for 1995 to 2018. The records were stored in several data bases, prompting the need to standardize and merge data sets. Merged data consisted of over 10,000 records containing duplicate and replicate records. Replicates were samples collected from the same oyster bar (or bar). Samples were labeled in the data base with a bar name and code. Replicate records had the same bar name and code. Multiple locations on the same bar were sampled when a bar had some type of management applied such as seeding, shell planting or sanctuary designation. We deleted duplicate records and then selected among replicate samples (multiple samples on the same bar with the same bar name and code, but on different locations on a bar), a single location from each bar using locational data and bar description information. Latitude, longitude and information on various management events were recorded in the data base and we used this information to identify the location on the bar with the least disturbance (determined by the dates when management action occurred). If a bar had a management action recorded, we chose the location with the oldest date, assuming that it would represent the location with the least disturbance from active planting or seeding and therefore represent the most "natural" condition. For example, if a bar had seed planted in 2000 on one location and shell placed in 2005 on another, we kept the sample representing the seed planting in 2000, because it was the older of the two. This allowed us to keep a single location from each bar, which we call sites, giving us one site per bar. (We recognize that a single site on an oyster bar may not represent the entire bar, but because they are fixed sites, they can indicate changes over time). We then examined frequency of sampling on each site and kept sites with 20 or more years of data.

We developed the HBBI from these data based on taxa presence. Since the ratings were qualitative and there were concerns that personnel changes could have introduced bias because of different interpretations of the levels of coverage, we chose to evaluate the data based on number of species present at a site.

Idhe et al. (2014) reported Spot and Atlantic Croaker diet information by four general invertebrate prey categories: crustaceans, mollusks, worms and miscellaneous (Table 1). We applied these categories to assign the hard bottom taxa by diet category and the HBBI was based on weighted mean presence of taxa by diet category for Spot and for Atlantic Croaker. We calculated annual mean presence by station and diet category, then weighted these averages to reflect forage importance for Spot and Atlantic Croaker by multiplying each category by the proportion each category contributed to Spot and Atlantic Croaker diet. We summed these weighted proportions to derive the HBBI, calculated as:

$$HBBI = p_c\,(\overline{X}\,(C_1+C_2\ldots C_n) + p_m\,(\overline{X}\,(M_1+M_2\ldots M_n) + p_w\,(\overline{X}\,(W_1+W_2\ldots W_n) + p_o\,(\overline{X}\,(O_1+O_2\ldots O_n)$$

Where p_c = proportion of diet comprising crustaceans,

$\overline{X}$ = sample mean;

C = presence of crustacean taxa;

p_m = proportion of diet comprising mollusks;

M = presence of taxa classified as mollusks;

p_w = proportion of diet comprising worms;

W = presence of taxa classified as worms;

p_o = proportion of diet categorized as miscellaneous taxa; and

O = presence of taxa classified as miscellaneous.

We used linear regression to examine changes in the index over time. We also calculated the time series mean HBBI at each site for Spot and Atlantic Croaker. We then categorized the annual mean HBBI for each site over the time series for Spot and Atlantic Croaker by assigning a score of 1 if the site met or exceeded the overall time series mean and a 0 if it did not. We mapped sites by score to visualize patterns.

In order to compare the HBBI with soft bottom indices, we obtained biomass data from the benthic monitoring program that produces the BIBI (Renee Karrh, MDDNR, personal communication) and calculated mean biomass by year for all stations sampled. Random and fixed stations are sampled (see LLanso et al. 2017 for further details of sampling protocols) and we used fixed station data for comparison with the hard bottom data which also employs a fixed station design. We calculated a mean of total sample biomass by year. We tested for linear changes over time in the soft bottom benthic biomass index during 1998-2018 with linear regression. Similar to what was done with the HBBI, we calculated the mean for the time series for the soft bottom biomass and compared the mean for the time series by annual mean, assigning a score of 1 if the annual biomass met or exceeded the mean time series biomass and 0 if it was below the mean.

Wells (1961) reported zonation of epibiotic taxa on oyster bars related to salinity with diversity declining as salinity decreased. We were concerned that the index might be biased with high salinity sites performing better. We examined the effect of salinity on the HBBI, by spatially plotting the HBBI score and examining if patterns of low or high scores suggested a salinity bias. We examined salinity preferences by taxa where available (Table 1) and binned stations based on general salinity limits. We defined low, mid, and high salinity categories: low salinity sites were less than 10‰; mid salinity sites, 10-15‰; and high salinity sites were greater than 15‰. We binned HBBI according to these categories and scored HBBIs by comparing annual mean at a site with the time series mean for each salinity category (score = 1 if annual mean was greater than the overall mean and 0 otherwise). We then mapped the HBBI score by salinity category.

To assess the potential effect of declines in hard bottom forage on Spot and Atlantic Croaker, we compared the salinity adjusted HBBI scores with historical fishing spots. We delineated historical fishing areas and then calculated the percentage of stations with HBBI scores below the time-series average within the delineated fishing areas.

Results

Fishing Areas compared to Oyster Bars - Historical fishing spots for Spot, Atlantic Croaker, and Striped bass strongly overlapped with Oyster bars (Table 2). Spot and Atlantic Croaker fishing spots showed similar concurrence with Oyster bars, with an average of 78.8% and 75.5% of fishing spots, respectively, found on Oyster bars. Striped Bass fishing spots were not as strongly associated with Oyster Bars (average 59.7% of fishing spots on Oyster bars). All three species had a low percentage of sites in close proximity (<400m) of the Oyster bars (Table 2). This could reflect a bias in digitizing sites if the analyst was using structure to identify locations. Nonetheless, over 75% of Spot and Atlantic Croaker fishing spots were associated with Oyster bars, suggesting that Oyster bars were potentially important habitat for the two benthivorous finfish. Striped Bass did not show as strong an affinity. The difference in overlap may be attributable to feeding preferences. Idhe et al. (2014) reported Spot and Atlantic Croaker as benthivorous, with diets primarily composed of invertebrates (~98%), while Striped Bass were highly piscivorous, with diet preferences including ~48% finfish and 52% invertebrates. Given this difference, we excluded Striped Bass from this present effort to examine focal species with diet preferences targeting invertebrates. Uphoff et al. (2019, Job 4) examined potential impacts of soft bottom forage changes on Striped Bass.

Developing Benthic Forage Indices - After eliminating replicate and duplicate samples, we used 4,821 records representing an average of 201 sites per year (Table 3). Thirty taxa were reported for the fall survey with frequency of occurrence varying from 0.001 to 0.952 (Table 4). The most common taxa based on proportion of samples with taxa present included, Barnacles, Ischadium, Mud Crabs, Anemones, Mud Tubes, Bryozoa, and Molgula (Table 4).

By diet category, the crustaceans had two taxa represented, Barnacle and Mud Crabs (Table 5). Mollusks had the greatest diversity within the diet categories with 14 taxa represented. Worms had 4 taxa and the miscellaneous category, 8 taxa (Table 5).

The HBBI for Spot ranged from 1.0 to 1.9 and the Atlantic Croaker HBBI ranged from 0.9 to 1.6 (Figure 2). Slight linear declines were indicated during 1995-2018 (Figure 2). Linear regression indicated that Atlantic Croaker and Spot HBBIs were strongly related (r^2 = 0.97; p = 0.001; Figure 3), reflecting strong overlap of their diets.

We plotted the annual HBBI for Spot and Croaker against the mean for the time-series and the annual mean soft bottom biomass against its time-series mean. All three indices followed similar patterns, with more frequent below mean observations in the last decade (Figure 4). Both HBBIs were moderately related to the soft bottom benthic organism biomass index (Spot, r^2 = 0.3860 ; P = 0.0012 and Atlantic Croaker, r^2 = 0.3978; P = 0.0010; Figure 5) .

We mapped site HBBI scores and observed that almost all high salinity sites had above average scores (Figure 6). When we applied the salinity adjustment, there were subtle changes in index scores for Spot and Atlantic Croaker (Figure 7). Whether or not we adjusted for salinity, the Potomac River and Western shore sites had low scores. This is interesting since the initial concern with poor Striped Bass catch came from anglers fishing in this area.

In assessing the potential impact declining HBBI could have on Spot and Atlantic Croaker fishing, we found approximately 50% of HBBI sites were below their time-series mean within the delineated fishing area. Thus hard bottom habitat, key foraging habitat for Spot and Atlantic Croaker, may have suffered a loss of ½ of the habitat, predominately in the Potomac River.

Discussion

In evaluating the interaction of Spot and Atlantic Croaker fishing locations and Oyster bars, we found that approximately ¾ of the fishing spots were on Oysters bars. While there has been a general notion that Oyster bars are prime locations for fishing, we were hard pressed to find studies that quantified the value of Oysters bars to fishing. With increased emphasis on restoring Oysters, more work is being done to assess the ecosystem services provided by Oysters beyond direct harvest (Coen et al. 2007; Grabowski and Peterson 2007; Peterson et al. 2003). Grabowski et al. (2012) reviewed multiple studies to predict the value of Oyster habitat to overall fish production, but did not specifically report on value to recreational fishing. Kroeger and Gaunnel (2014) estimated a net economic gain for recreational harvest of a suite of species at $28,005-33,625 (U.S.) on 3.6 miles of restored Oyster reefs. They also reported that studies showed recreational anglers are willing to pay ~ $13 (U.S.) per year in Louisiana to maintain the ability to fish over Oyster bars. To our knowledge, there are no direct studies to assess angler willingness to pay to fish Oyster bars in Chesapeake Bay, or assessments of the economic contribution of Oysters to the Spot and Atlantic Croaker recreational fisheries, so we cannot estimate the economic value of Oysters to these fisheries. The only indication we have that Oysters bars are preferred fishing habitat in Maryland is our comparison which showed the majority of Spot and Atlantic Croaker fishing spots were located over Oyster bars.

While we sense that these estimates are representative of common opinions held in the fishing community, we have to acknowledge the potential for error. Fishing location maps were hand digitized from hand drawn maps that were developed from anglers, bait shops and charter boat captain reports. Initial locations (hand drawn maps) are likely approximate locations of reported fishing spots. Likewise, digitized maps are approximate locations based on the analyst's best interpretation of location. It is impossible to estimate error in these locations.

Evaluation of the epibiotic taxa on Oyster bars suggest these data are useful to examine potential impacts to forage dynamics. Our index of mean presence, weighted for Spot and Atlantic Croaker diets showed significant, albeit slight, declines over time. This index should be considered provisional, as it includes all taxa present. Since many of the taxa have a low frequency of occurrence and salinity influences distribution of these taxa, we need to examine the effects of rare taxa and salinity on the index. Even so, the HBBI provided a snapshot of hard bottom forage condition and supports our systematic approach to address anglers concerns over loss of fishing quality, while also prompting development of hypotheses to explore cause of potential changes.

Comparisons between the HBBI and soft bottom fixed station biomass, showed reasonable agreement. Llanso and Zaveta. (2017) attributed low BIBI scores to hypoxia, but we do not expect that declines in the HBBI were entirely driven by hypoxia since not all sites were adjacent to deep water hypoxia. Sagasti et al. (2001) reported epifaunal communities in the York River thrived in spite of frequent exposure to hypoxic stress. An alternate hypothesis to hypoxic stress is increased forage pressure on hard bottom habitats due to displacement of benthic feeding fishes facing food limitations induced by hypoxic conditions in soft bottom habitats. Lenihan et al. (2001) demonstrated increased forage pressure on shallow water hard bottom when epibenthic fauna on adjacent low relief deep water hard bottom habitat was impaired by hypoxia. At present we have no means to do extensive evaluations to compare habitats and explore forage dynamics. However, we can take an incremental step and evaluate the percentage of hard bottom habitat overlapping hypoxic areas. This will give us insight into potential exposure of hard bottom fauna to hypoxia. We can also evaluate proximity of hard bottom habitat to hypoxic areas along with condition of the soft bottom community to determine the potential for fishes to seek Oyster habitat as areas of refuge.

Mapping the mean index the salinity adjustment showed subtle changes in sites status in high salinity areas in the lower eastern area of the Bay. The Potomac River and upper Western Shore had consistently below average HBBI's for both Spot and Atlantic Croaker with or without salinity adjustments. This suggests there is potential for habitat to be limiting, particularly since adjacent soft bottom habitat is showing stressed conditions (Llanso and Zaveta 2017). It is important to understand how this might impact fishing as our evaluation showed that 50% of the hard bottom sites within the historic fishing area were below the time series mean. While declines in hard bottom diversity may not have implications for overall Spot and Atlantic Croaker production, it could impact the quality of recreational fishing. Harding and Mann (2001) and Pfirrman and Seitz (2019) and found larger transient finfish, including Spot and Atlantic Croaker, were associated with Oyster habitat. The decline in habitat quality could be one among many factors contributing to perceived declines in fishing quality.

To date, we have demonstrated the potential to develop HBBI's. We limited this study to Spot and Altantic Croaker diet needs, but may expand our focus to other species, including White Perch and Striped Bass. We will also examine dynamics of hypoxia to determine the extent of potential direct and indirect impacts on hard bottom epibenthic fauna that could be important forage for key recreational species in Maryland.

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Table 1. Percentage of diet comprised by diet category for Spot and Atlantic Croaker from Idhe et al. (2014).

Diet Category	Spot	Atlantic Croaker
Crustaceans	7.6	15.4
Miscellaneous	47.1	27.3
Mollusks	11.3	14.3
Worms	32.6	41.3

Table 2. Percentage of fishing sites within (0 m), near (< 400 m), and away from an oyster bar.

Species	**Year**	**N**	**0 m**	**< 400 m**	**> 400 m**
Striped Bass	1960	*96*	57.3	7.3	35.4
	1968	*88*	60.2	0.0	39.8
	1989	*n/a*	n/a	n/a	n/a
	2004	*56*	62.5	0.0	37.5
	mean		60.0	2.4	37.6
Spot	1960	*44*	86.4	0.0	13.6
	1968	*24*	79.2	0.0	20.8
	1989	*24*	75.0	0.0	25.0
	2004	*24*	75.0	0.0	25.0
	mean		78.9	0.0	21.1
Atlantic Croaker	1960	*38*	68.4	0.0	31.6
	1968	*17*	82.4	0.0	17.6
	mean		75.4	0.0	24.6

Table 3. Number of samples collected and number of samples selected for analysis, by year.

Year	Number Collected	Number Selected
1995	257	192
1996	228	183
1997	219	176
1999	258	206
2000	249	200
2001	244	197
2002	261	208
2003	245	198
2004	261	208
2005	260	206
2006	260	207
2007	264	210
2008	265	206
2009	262	208
2010	236	189
2011	256	208
2012	258	206
2013	256	207
2014	249	206
2015	245	196
2016	265	206
2017	263	204
2018	262	202

Table 4. Taxa present at all fall survey sites used for analysis, by class, frequency of occurrence and available salinity limits (‰). (a. Andrews 1953; b. Lipson 1973; c.Steimle 1995; d. Jansson et al. 2013; e. Ma and Purcell 2005; f. Qui et al. 2002; g. Leamon and Fell 1990)

Organism	**Class**	**Proportion Present**	**Salinity Tolerance**
Anemones	Anthozoa	0.596	euryhaline[a]
Molgula	Ascidiacea	0.392	> 12-15[a]
Anomia	Bivalvia	0.006	
Blood Clam	Bivalvia	0.000	
Geukensia	Bivalvia	0.006	
Ischadium	Bivalvia	0.826	> 8-10[a]
Macoma	Bivalvia	0.003	> 5[d]
Mercenaria	Bivalvia	0.008	>15[b]
Mulinia	Bivalvia	0.004	
Mya	Bivalvia	0.004	>5[b]
Mytilopsis	Bivalvia	0.078	< 10-12[a]
Mytilus	Bivalvia	0.039	euryhaline[f]
Petricola	Bivalvia	0.001	
Rangia	Bivalvia	0.005	1-10[b]
Tagelus	Bivalvia	0.000	
Bryozoa	Cheilostomata	0.877	15-20[a]
BoringSponge	Demospongia	0.086	7-38[g]
Lissoden.	Demospongia	0.001	
Microciona	Demospongia	0.023	
OtherSponges	Demospongia	0.035	
Crepidula	Gastropoda	0.070	>15[a]
Eupleura	Gastropoda	0.000	>20[b]
EupleuraEgg	Gastropoda	0.000	>20[b]
MudSnails	Gastropoda	0.004	
Urosalpinx	Gastropoda	0.000	>18[b]
UrosalpinxEgg	Gastropoda	0.001	>18[b]
Hydroids	Hydrozoa	0.093	<9.3[e]
MudCrabs	Malacostraca	0.693	>5[c]
Barnacles	Maxillopoda	0.952	<15[a]
MudTubes	Polychaeta	0.578	
Sabellaria	Polychaeta	0.048	
Serpulids	Polychaeta	0.069	
Stylochus	Trepaxonemata	0.046	
ToadFish	Actinopterygii	0.004	
Grass	Angiospermae (Division)	0.073	
BrownAlgae	Phaeophyta (Division)	0.001	
RedAlgae	Rhodoaphyta (Division)	0.034	
Entemorpha	Ulvophyceae	0.000	

Ulva	Ulvophyceae	0.014

Table 5. Epibenthic taxa by diet category, with frequency of occurrence in all samples 1995-2018.

	Crustaceans		Mollusks		Worms		Miscellaneous	
Taxa	Barnacles	0.952	Ischadium	0.826	Mud Tube	0.578	Bryozoa	0.877
	Mud Crab	0.693	Mytilopsis	0.078	Serpulids	0.069	Anemone	0.596
			Crepidula	0.07	Sabellaria	0.048	Molgula	0.392
			Mytilus	0.039	Stylochus	0.046	Hydrozoa	0.093
			Mercenaria	0.008			Boring Sponge	0.086
			Anomia	0.006			Other Sponge	0.035
			Geukensia	0.006			Microcionia	0.023
			Rangia	0.005			Lissodendrix	0.001
			Mya	0.004				
			Mud Snail	0.004				
			Mulinia	0.004				
			Macoma	0.003				
			Urosalpinx	0.001				
			Pertricola	0.001				

Table 6. Percentage of sites within the historical fishing areas where Spot and Atlantic Croaker HBBI score were below average.

HBBI Score	Spot	Atlantic Croaker
Below Mean	50.41	51.24
Above Mean	49.59	48.76

Figure 1. Fall Oyster survey sites used to develop the Hard Bottom Benthic Indices (HBBIs) for Spot and Atlantic Croaker.

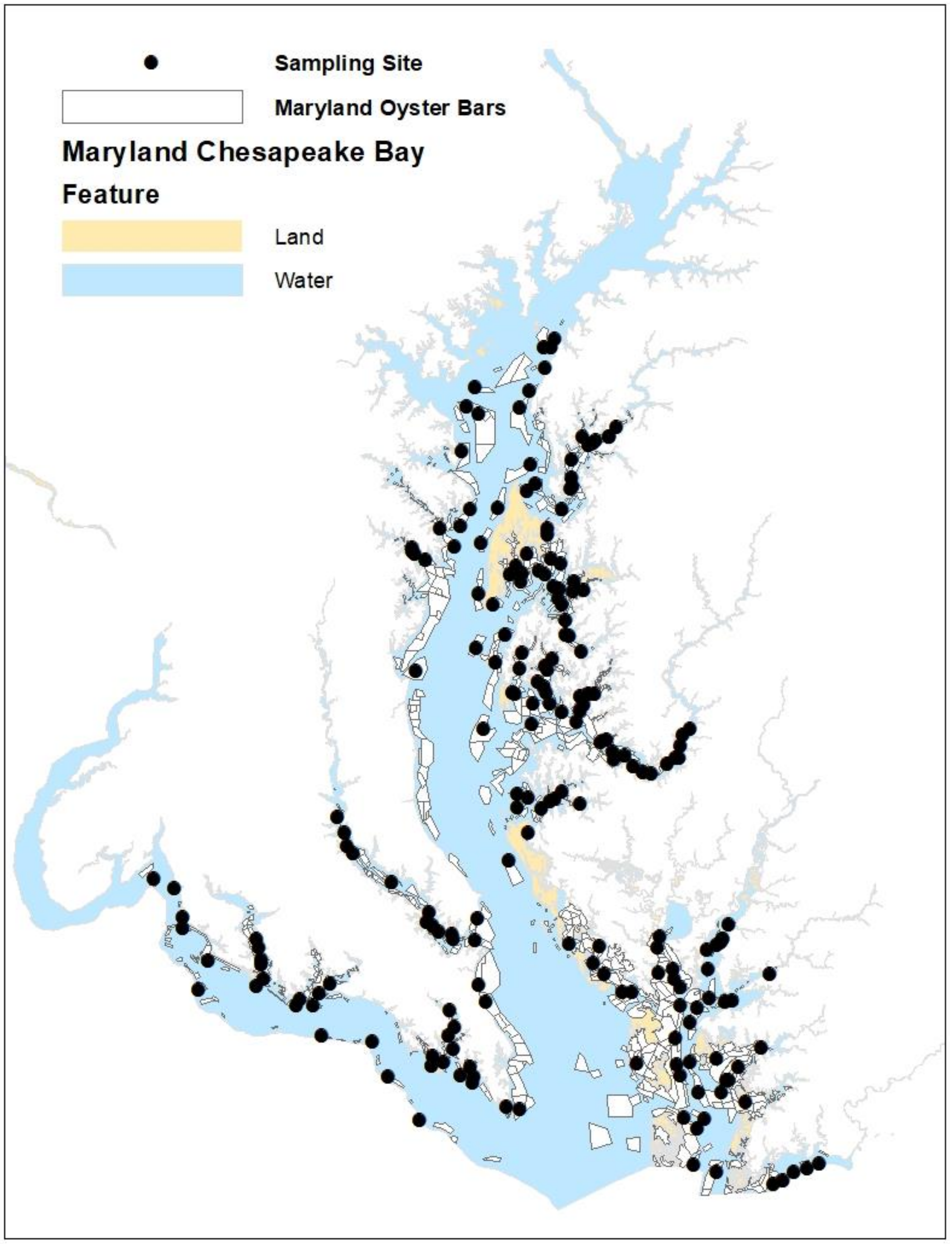


Figure 2. Spot and Atlantic Croaker HBBIs regressed against year.

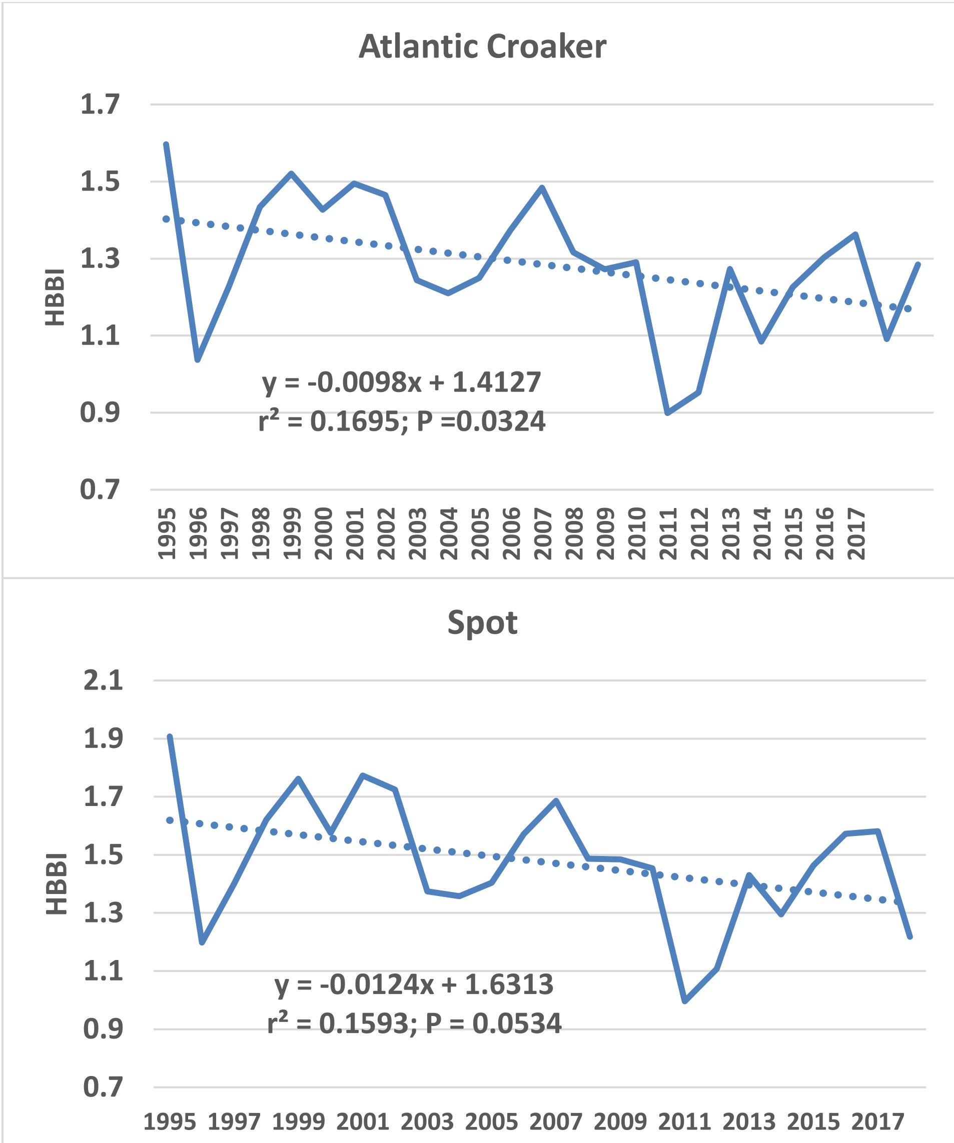


Figure 3. Linear relationship of Spot and Atlantic Croaker HBBIs.

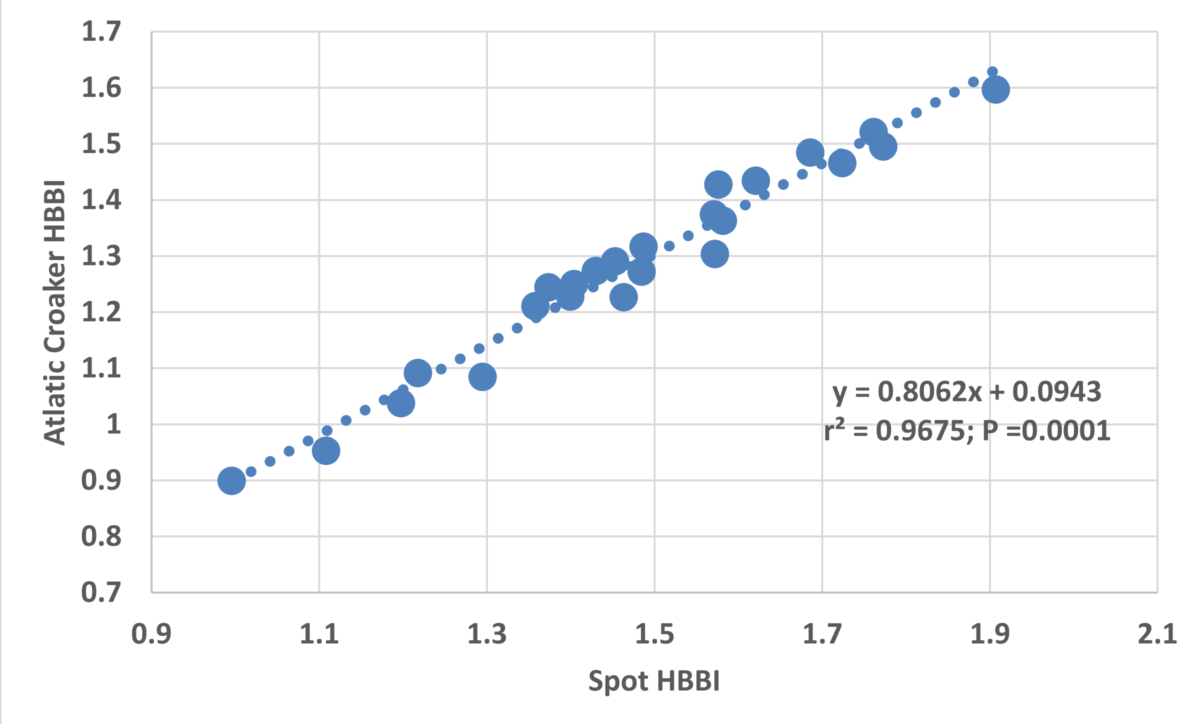


Figure 4. Spot and Atlantic Croaker HBBI and Soft bottom biomass annual means plotted against time-series means.

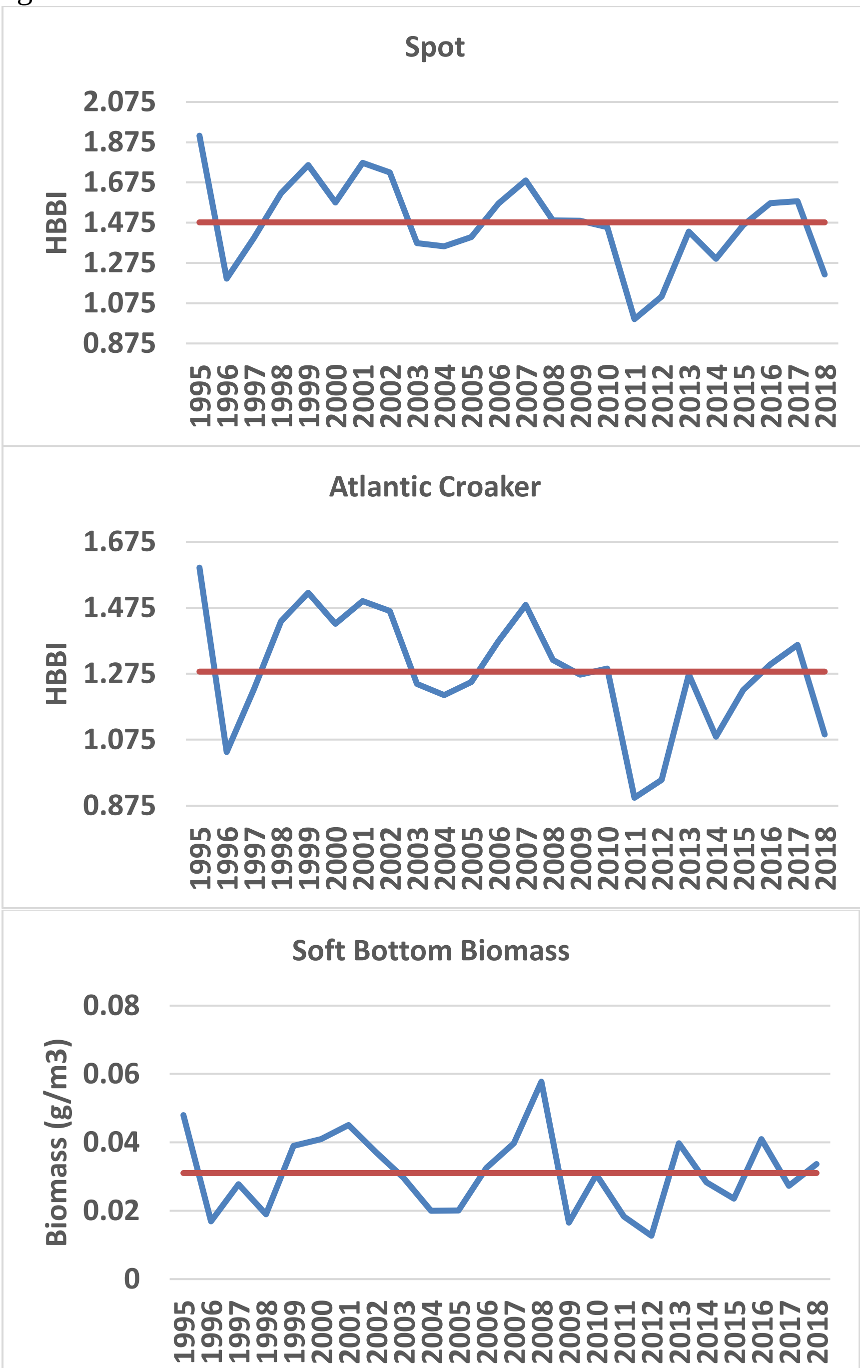


Figure 5. Regressions of HBBI with soft bottom biomass density.

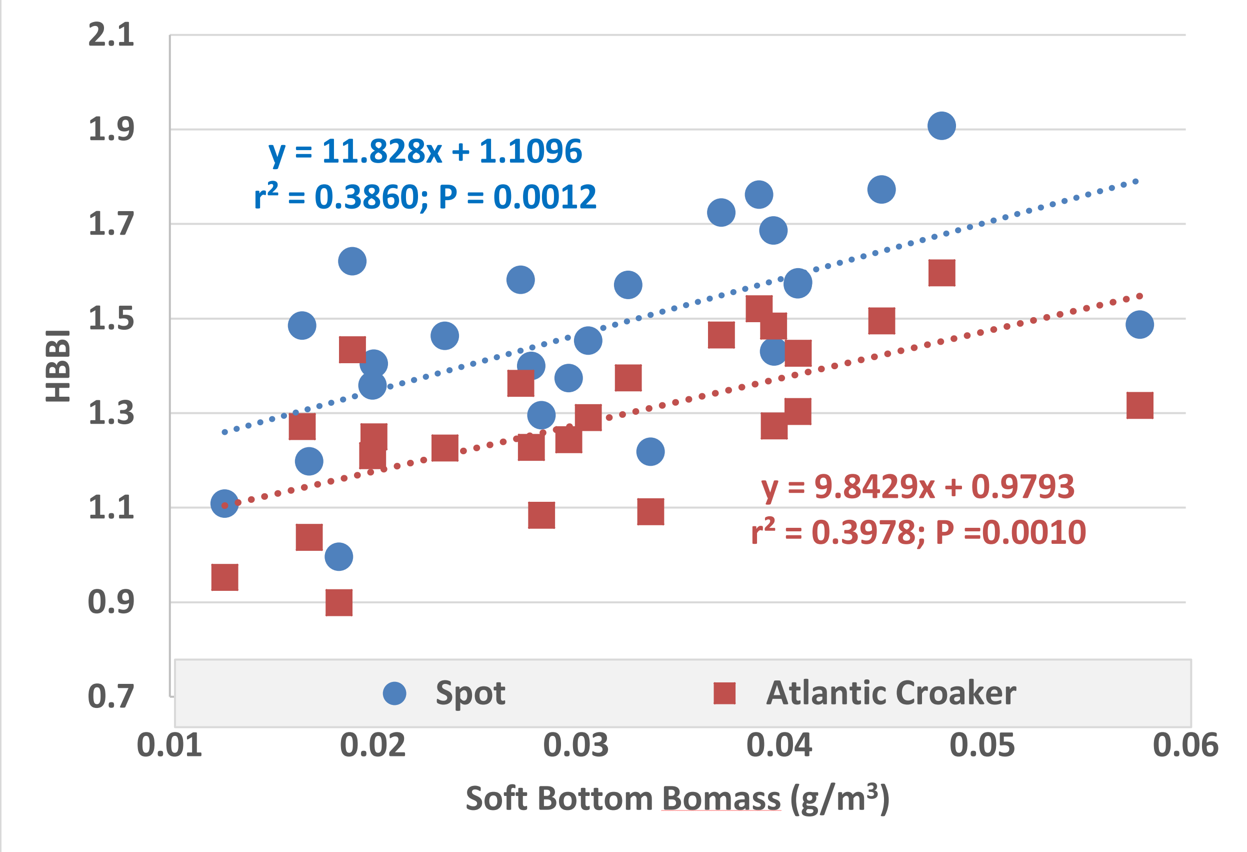


Figure 6. Mapped HBBI scores

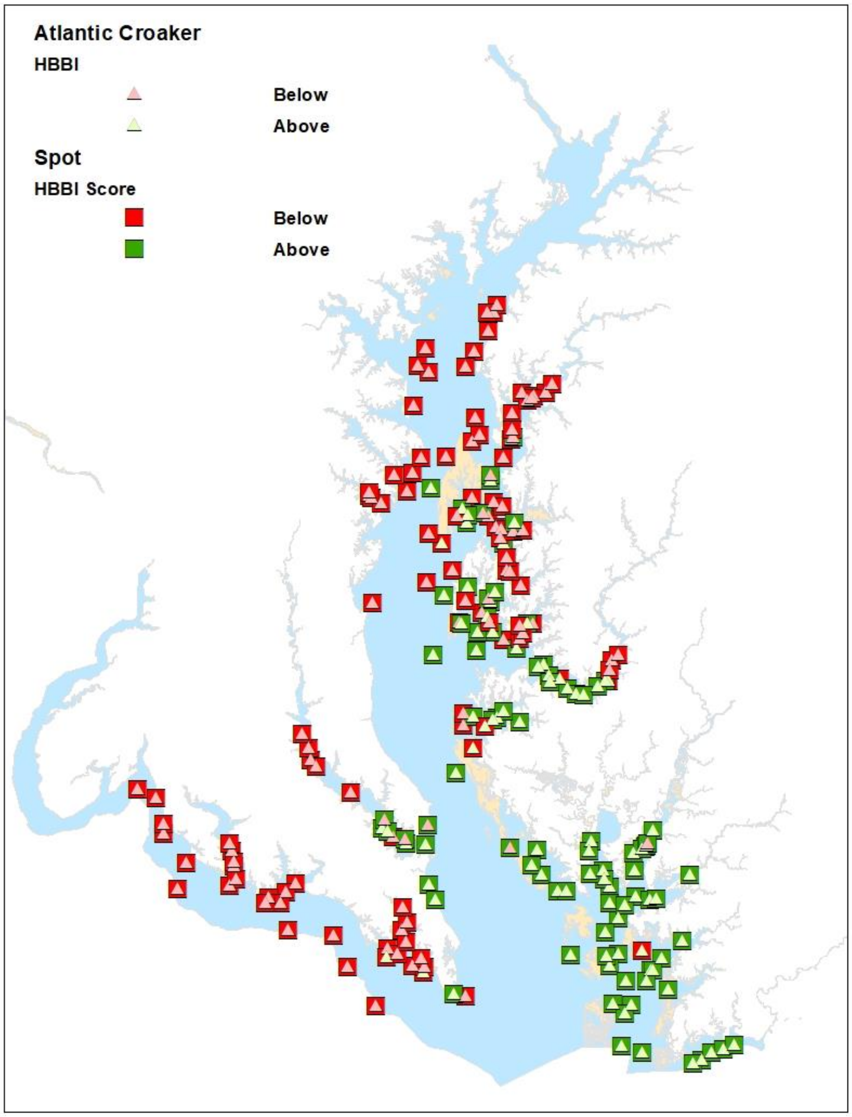
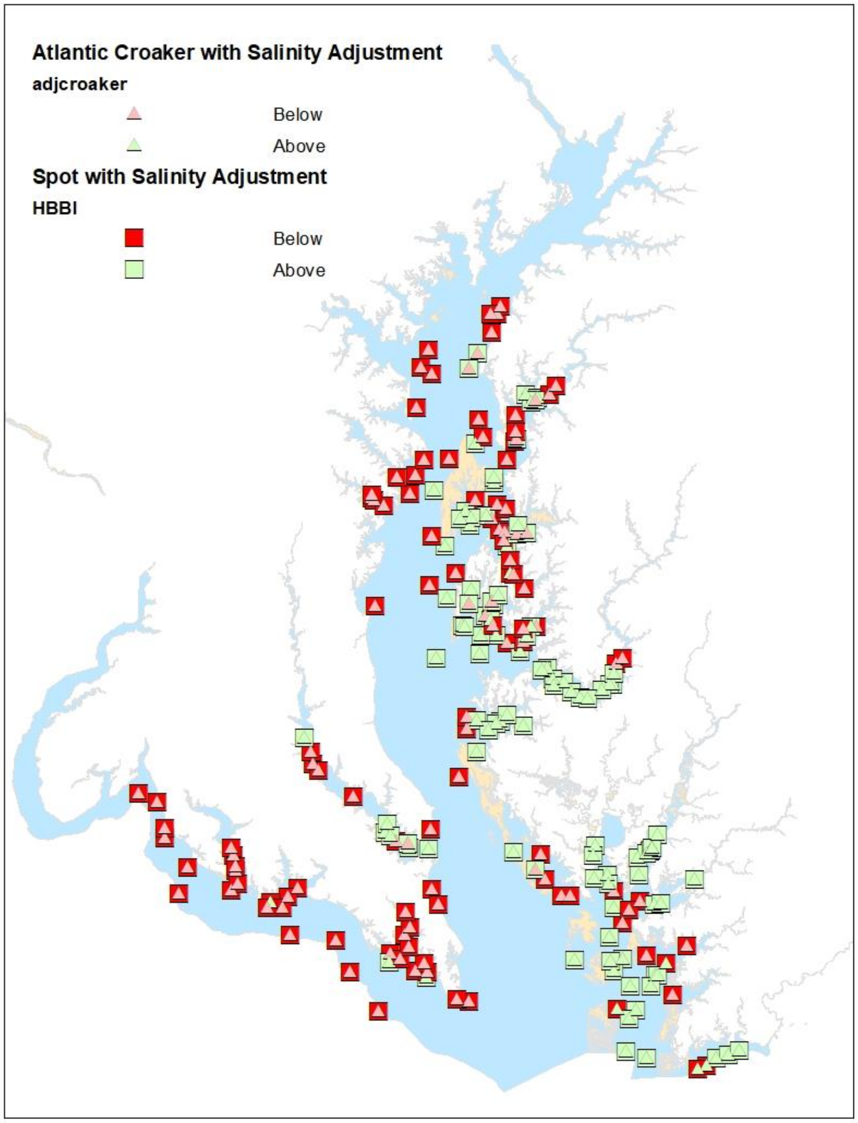


Figure 7. Mapped HBBI scores with salinity correction.



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

Jim Uphoff, Alexis Park, and Carrie Hoover

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). 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 (1998-2018), relative abundance (1983-2018), natural mortality (1986-2018), and forage relative abundance in surveys (1959-2018) and fall diets of Striped Bass (1998-2000 and 2006-2018) provided indicators to assess forage status and Striped Bass well-being in Maryland's portion of Chesapeake Bay (or upper 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 ($\geq$ 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. Examination of 90% confidence intervals of IF metrics indicated a scoring system reduced from previous years (scores from 1 to 5) better matched a generalization of separation indicated by percentile confidence intervals. A score of 1 indicated threshold (poorest) conditions; a score of 3 indicated target (best) conditions; and a score of 2 indicated conditions between. Time-periods where body fat indicators (1998-2018) were at target or threshold conditions provided a time-frame for assigning scores to other indicators. Annual scores for each metric were averaged for a combined annual IF score.

During 1998-2004, the IF indicated threshold to near threshold (poorest, i.e., scores near 1) foraging conditions for Striped Bass in Maryland's portion of Chesapeake Bay (upper Bay) were typical. IF scores (1.4 or above) were elevated beyond the threshold after 2004. IF scores during 2008-2011 (IF =2.6-3.0) were near or at the target (best foraging conditions), then IF fell into an intermediate region (1.4-2.4). It has been near 2.0 (does not breach threshold or target) during 2017-2018, indicating some recovery from poorer foraging conditions during 2015-2016 (Scores 1.4-1.6).

A rapid rise in Striped Bass abundance in upper Bay during the mid-1990s, followed by a dozen more years at high abundance after recovery was declared in 1995, coincided with declines in relative abundance of Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (i.e., major pelagic and benthic prey) to low levels. Survival of small and large sized Striped Bass in upper Bay shifted downwards in the mid-1990s and poor survival has persisted. Striped Bass were often in poor condition during fall 1998-2004 and vulnerable to starvation. Improvements in condition after 2007 coincided with lower Striped Bass abundance, spikes or slight increases in some major forage indices, and higher consumption of larger major prey (Spot and Atlantic Menhaden) in fall diets.

A return of Striped Bass to noticeably higher abundance after 2014 was not shared by major forage, but condition has not declined to threshold conditions. It appears that slight increases in Atlantic Menhaden relative abundance, while not statistically significant, appear to have biological significance for both size classes of upper Bay Striped Bass. Consumption of Atlantic Menhaden by small and large Striped Bass since 2013 has been higher, more frequently ranking in the top half of estimates during 2006-2018.

Recent divergence in condition (proportion of Striped Bass without body fat) between small and large Striped Bass (2016 and 2018) may indicate relief from a prey bottleneck for large fish. Estimates of the proportion without food improved for large fish, but not small ones, since 2014. The ratio of prey length to Striped Bass length for small fish has been consistently high since 2015, reflecting larger relative size of Atlantic Menhaden. Small Striped Bass would have more difficulty in catching and handling larger prey than large fish in any given year. Low consumption of Atlantic Menhaden by small Striped Bass during 2016-2017 was not offset by other prey.

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).

Concern emerged about the impact of high Striped Bass population size on its prey-base shortly after recovery (Hartman 2003; Hartman and Margraf 2003; Uphoff 2003; Savoy and Crecco 2004; Heimbuch 2008; Davis et al. 2012; Overton et al. 2015; Uphoff and Sharov 2018). 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; Uphoff and Sharov 2018). 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 rates (M) of large sized Striped Bass in Chesapeake Bay increased substantially during the mid-1990s while instantaneous fishing mortality rates (F) remained low (Jiang et al. 2007; ASMFC 2013; 2019). Prevalence of mycobacteriosis and M appear to be less outside Chesapeake Bay (Matsche et al. 2010; ASMFC 2019), but condition and M of the coastal migration contingent appears linked to forage, particularly ages 1+ Atlantic Menhaden (Buccheister et al. 2017; Uphoff and Sharov 2018).

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 in Maryland 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; Dettmers et al. 2012; Fogarty 2014).

The IF approach was based on a suite of indicators (metrics). Status would be judged by whether target (indicating best forage conditions) or threshold (indicating poorest forage conditions) reference points were met for each metric. Time periods where body fat indicators (1998-2018) were available provided a time-frame for developing targets and thresholds for other metrics. 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 (C) in fall provided trends in supply relative to demand based on relative abundance and diet sampling, respectively. Proportion of resident Striped Bass in fall 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; 2017; 2018). 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).

A nutritional indicator, proportion of Striped Bass without body fat (P0), 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; Buccheister and Houde 2016). We selected these species as focal prey (major prey) for forage indices. Indices of major prey relative abundance and availability were estimated from fishery-independent surveys and fall diets of Striped Bass, respectively. Trends in prey index-to-Striped Bass index ratios (forage rations or FR) 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 consumption indicator based on weight consumed (C) was changed to proportion of empty guts (PE) in Uphoff et al. (2017) 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 published diet information from the 1930s (Hollis 1952) and the 1950s (Griffin and Margraf 2003). Baker et al. (2014) suggested that presence-absence of diet items (frequency of occurrence) provided the most robust and interpretable measure of diet composition. Estimates of C and its species composition were useful for interpreting PE (Uphoff et al. 2018).

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 2019) was used as an indicator of change in survival due to natural mortality (SR) prior to recruitment to the fishery (Uphoff et al. 2018). 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 (Versak 2018), 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 gill net indices adjusted for changes in catchability that reflected expected stock changes and used these as the numerator in the SR estimates. We revised the approach in Uphoff et al. (2016) in Uphoff et al. (2018) and used the latter to estimate a SR time-series through 2018 that reflected the changes in catchability that reflected the most recent ASMFC Striped Bass stock assessment (ASMFC 2019).

This report provides a complete set of indicators through 2018. Some indicators were revised and all were summarized into a single score to serve as a quick reference for managers and the public. We revised indicators to reflect a preference for indicators that had confidence intervals. The scoring system employed in Uphoff (2018) was evaluated to see if the gradation indicated by scores on a scale of 1 to 5 were a suitable match for the precision of the individual metrics.

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, 1998-2018, 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 457 mm TL during 1998-2014; it was raised to 508 mm TL in 2015 and lowered to 483 mm TL in 2018). Standard deviations and confidence intervals (90%) 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 condition target based on 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 were in good condition. 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 P0 estimates during 1998-2004, the period of consistently poor condition). 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); CBEF P0 estimates were similar to those estimated from FWHP sampling (Uphoff et al. 2018).

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 1959-2018 provided indices for Atlantic Menhaden, Bay Anchovy, and Spot (Durell and Weedon 2019). Additional indices for Spot and Bay Anchovy were estimated from a Blue Crab trawl survey conducted during summer 1989-2018 (Uphoff 1998; Rickabaugh and Messer 2018; MD DNR 2019a; 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 2019b). 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 (1989-2018) to place their time-series 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 2014-2018, 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 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) in numbers during 2006-2018 was estimated for each Striped Bass size class based on fish with stomach contents (Pope et al. 2001). Estimates included both counts of whole items and presence of partially intact prey (portions that were intact enough to identify a prey, but not intact enough to measure and weigh as individuals). The latter could include multiple individuals, so percent by number was negatively biased to some extent.

Relative availability of prey biomass (biomass consumed or 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 estimated as an indicator of total prey availability (Chipps and Garvey 2007). Standard deviations and 90% CI's of PE were estimated using the normal distribution approximation of the binomial distribution (Ott 1977).

To aid interpretation of PE, we examined the influence of prey-predator length ratios (PPLR) of the two size classes of Striped Bass. For this analysis we determined PPLRs for the two largest major prey in fall diets: Spot and Atlantic Menhaden. This analysis was based on ratios for whole prey and was split for small and large Striped Bass. We determined median PPLR for each year and size class. Optimum PPLR of Striped Bass was 0.21 (Overton et al.

2015) and we compared median PPLR for each size class to this estimate of optimum PPLR. Correlation analysis was used to examine the associations of PE, C, median PPLR, and P0. Influence of PPLR on differences in PE of large and small Striped Bass was further explored in two steps with linear regression and correlation process. The relationship of PE of large versus small fish was estimated by linear regression and then the association of the residuals of that relationship was examined by correlation analysis.

A fishery-independent index of relative abundance of upper Bay resident Striped Bass was not available and we used a Striped Bass catch-per-private boat trip index (released and harvested fish; RI) for 1981-2018 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 2019; NEFSC 2012; ASMFC 2013). On July 9, 2018, NMFS released revised Marine Recreational Information Program (MRIP) catch estimates as part of its recent transition from the old Coastal Household Telephone Survey to a new, mail-based Fishing Effort Survey. Uphoff et al. (2018) and previous F-63 reports used older catch and effort estimates to estimate RI. A 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 (Uphoff et al. 2018). Our RI estimates were based on revised MRIP estimates in this report.

The RI 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). The RI equaled September-October recreational private and rental boat catch of Striped Bass divided by estimates of trips for all species 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). We compared the RI to the abundance estimates for 3 year-old Striped Bass estimated by the statistical catch at age model used in the recent stock assessment in this report (ASMFC 2019).

We used forage indices divided by RI (forage index-to-Striped Bass index ratios or FR) as indicators of forage supply of major prey relative to Striped Bass demand (index of potential attack success). Ratios were standardized by dividing each year's estimate by the mean of ratios during 1989-2018, a time-period in common among all data; FR covered 1983-2018.

A weighted grand mean of FR was used to depict 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. Correlation analyses in two stages were used to judge indices for inclusion in the weighted FR. The first correlation analysis was among the species-specific FRs to determine if any were closely correlated enough that they were redundant. We used $r \geq 0.80$ suggested in Ricker

(1975) as an indication of close correlation and chose only one of the indices meeting that criterion. The second step was based on a correlations of species-specific FRs and P0. Correlation coefficients of negative associations between P0 and each FR provided the basis for weights. Positive correlations were considered illogical and were eliminated from consideration. Each correlation coefficient was standardized to the highest negative association among major prey as r_i / r_{max} ; where max indicates the highest negative correlation coefficient, r, and i indicates r for species, i. Annual FR for each major forage species was multiplied by its respective weight and these weighted FR values were summed for the year to calculate the annual weighted FR. Targets and limits for FR were drawn from periods of three or more years when FRs coincided with target or limit P0, respectively. The FR target for major forage ratios was estimated as the lowest standardized ratio that coincided with P0 meeting its target. The FR threshold was estimated as the highest major forage ratio coinciding with threshold P0 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 2018). Table 8 in Versak (2018) provided mean values of for annual, pooled, weighted, age-specific CPUEs (1985–2017) 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 2018 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 2018). This CPUE3 index had the advantage of combining both spawning areas, a coefficient of variation (CV) estimate was provided, and 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; 2017; 2018) considered both implausible when viewed against stock assessment estimates, juvenile indices, and harvest trends. Uphoff et al. (2016; 2017; 2018) 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 (2018) 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 2019; 2017 was the last year in the assessment) during 1985-2017. We averaged 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, reported values were used. We used a linear regression of relative q versus age 3+ abundance estimated by ASMFC (2019; 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 1996-2018 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});

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

The threshold for SR was estimated as the highest point of the threshold P0 period and the SR target was estimated as the highest point of the target P0 period that was consistent with the remaining points.

Tag-based estimates of survival based on M for 457-711 mm Striped Bass from Chesapeake Bay in ASMFC stock assessment (ASMFC 2019) were compared to SR. Tag-based estimates of M were determined for two time periods (early period = 1987-1996 and late period = 1997-2017) and we converted the estimates of M in ASMFC (2019) to survival (S) using the equation

$$S = e^{-M} \text{ (Ricker 1975).}$$

The relative differences in survival (early period survival / later period survival) 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 FR 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 2019b). Remaining indices for Atlantic Menhaden (seine), Bay Anchovy (seine and trawl), and Spot (seine and trawl) were based on geometric means (Durell and Weedon 2019). Geometric mean 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).

@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).

Ninety percent CIs for ratios provided a means for evaluating whether the system used to integrate the five forage indicators (ranking best or target conditions as 5, worst or threshold conditions as 1, and assigning scores of 2, 3, or 4 for conditions between) used previously (Uphoff et al. 2018) was depicting greater precision than warranted. We evaluated whether target metrics could be separated from threshold metrics and how fine a level of detail was justified for intermediate conditions.

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.

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 Littell 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 Littell 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 (95% Cis were standard output) of the model parameters for each indicator species were estimated to examine whether parameters were different from 0 (Freund and Littell 2006). If parameter estimates were often not different from 0, the model was rejected.

Results

Examination of 90% confidence intervals of IF metrics (described below) indicated a reduced scoring system (ranking best or target conditions as 3, worst or threshold conditions as 1, and assigning a score of 2 for intermediate conditions) better matched a generalization of separation indicated by percentile confidence intervals. This 1-3 scoring system was used for all metrics included in the IF.

Striped Bass in the upper Bay during fall were usually in poor condition (P0 $\geq$ threshold; threshold = 0.68) during 1998-2004 and at or near the target level of condition (P0 $\leq$ target; target = 0.30) during 2008-2010, 2014-2015, and 2017 (Figure 2). Condition shifted away from threshold to intermediate scores during 1998-2007 to intermediate to target afterwards. The 90% confidence intervals of P0 allowed for separation of years meeting the target or threshold conditions from remaining estimates (Figure 2). A IF score of 1 was assigned to P0 at or more than 0.68; a score of 3 was assigned for P0 less or equal to 0.30.

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, but was present in 2018 (small fish P0 = 0.40 and large fish P0 = 0.05; Figure 3). This recent divergence in P0 between small and large Striped Bass may indicate a prey bottleneck exists for small fish that large fish are no longer subject to.

Major pelagic prey were generally much more abundant during 1959-1994 than afterward (Figure 4). Bay Anchovy seine indices (1959-2018) following the early to mid-1990s were typically at or below the bottom quartile of indices during 1959-1993. Highest Bay Anchovy trawl indices occurred in 1989-1992 and 2001-2002, while lowest indices occurred during 2006-2011 and 2015-2018. There was little agreement between the two sets of Bay Anchovy indices; however, there were few data points representing years of higher abundance in the years in common 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 1995-2018 (Figure 4).

Benthic major forage indices were low after the 1990s, but years of higher relative abundance were interspersed during the 2000s (Figure 5). Seine (1959-2018) and trawl (1989-2018) 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 (1989-2018) were highest during 1989-1996, 2009, and 2011 (Figure 5).

In general, relative abundance of Striped Bass (RI) during 1981-2018 was lowest prior to 1994 (mean RI < 0.4 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 by about a third of the 1995-2006 mean during 2008-2013 (mean = 1.8) and then rose to 2.4 in 2014, 2.6-2.7 in 2015-2016, 3.0 in 2017, and 2.4 in 2018. The 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 1994-2018. Ninety percent CIs of periods of threshold P0 (1998-2004) and target P0 (2008-2010) indicated a dichotomous separation of RI with some overlap; median RI estimates during 2008-2018 did not or barely overlapped the lower 90% CI estimates of 1994-2018. Threshold conditions of P0 were generally breached when RI $\geq$ 2.0 (score = 1) and target conditions were breached when RI was < 2.0 (score = 3). RI has been in excess of 2.0 since 2014 (Figure 6). The trend in RI compared favorably to the trend in estimated aggregate abundance of 2- to 5-year old Striped Bass along the Atlantic Coast, particularly in the years after recovery was declared (1995; Figure 7). Overall, the estimates were well correlated ($r = 0.79$, $P < 0.001$).

Species-specific standardized forage-to-Striped Bass ratios exhibited a similar pattern during 1983-2018 (Figures 8-13). The 90% CIs 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 elevation in Atlantic Menhaden ratios after 2004 (Figure 15).

In the first step for estimating weighted FR, correlations among species-specific FRs during 1998-2018 indicated that the two indices for Spot were closely correlated (r = 0.97; Table 2). The seine index was chosen for inclusion in weighted FR because of its longer time-series. In the second step, the trawl based Bay Anchovy FR was positively correlated with P0, while remaining species-specific FRs were all negatively correlated. The trawl based FR for Bay Anchovy was eliminated from consideration. Atlantic Menhaden had the strongest correlation with P0 (r = -0.41), followed by Blue Crab (r = -0.33), Spot (seine index, r = -0.26), and Bay Anchovy (seine index, r = -0.13; Table 2). These correlations corresponded to a weight of 1.00 for Atlantic Menhaden, 0.81 for Blue Crab, 0.63 for Spot, and 0.30 for Bay Anchovy. Trends in the candidate species-specific and weighted FRs are depicted in Figure 15).

Weighted FR was lowest during the threshold period for P0, 1998-2004 (except 2001; Figure 16). Threshold FR was 0.20 or less (score =1). Threshold conditions were also breached during 2006 and 2015-2017. Target P0 was met during 2008 and 2010 when weighted grand mean FR was more than 0.38 (score = 3). Target conditions were met during 2005 and 2008-2013. Remaining years were intermediate (score = 2), including 2018 (Figure 16).

Samples from 1,912 small and 2,527 large sized Striped Bass were analyzed for diet composition during October-November, 2006-2018 (Table 3). Numbers examined each year ranged from 47 to 330 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 because numbers collected per trip were not confined by the terms of the CBEF collector's permit (6-12 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 95.8% of diet items encountered in small Striped Bass collected from upper Bay during fall, 2006-2018 (Figure 17). Bay Anchovy accounted for the highest percentage by number when all years were combined (62.9%, annual range = 19.1-87.9%); Atlantic Menhaden, 13.7% (annual range = 0-48.8%); Spot 6.1% (annual range = 0-70.7%); Blue Crab, 13.1% (annual range = 0.8-34.6%); and other items accounted for 4.2% (annual range = 0-24.5%; Figure 17). The vast majority of major prey in small Striped Bass diet samples during fall were YOY (Uphoff et al. 2016).

Major prey accounted for 91.9% of diet items, by number, encountered in large Striped Bass diet samples during fall 2006-2018 (Figure 18). Atlantic Menhaden accounted for 47.2% when all years were combined (annual range =12.4-76.4%); Bay Anchovy, 16.7% (annual range = 3.7-32.5%); Spot, 8.7% (annual range = 0-52.4%); Blue Crab, 23.8% (annual range = 2.6-59.4%); and other items, 8.1% (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 18) 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 2006-2018 (combined) were dominated by Atlantic Menhaden (71.3%), followed by Spot (10.3%), Bay Anchovy (10.2%), Blue Crab (2.1%) and other items (6.0%; Figure 19). 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 2006-2018. During years of lowest C (2007, 2011, 2016, and 2017) varying items contributed to the diet of small fish. During years of higher C, either Spot (2010) or Atlantic Menhaden (remaining years) dominated diet mass. The 2018 estimate of C of small fish was fourth highest of the time-series (Figure 19).

By weight, Atlantic Menhaden predominated in large fish sampled (85.6% of diet weight during fall 2006-2018 over all years combined); Bay Anchovy accounted for 1.3%; Spot, 4.0%; Blue Crab, 4.0%; and other items, 5.1% (Figure 20). Estimates of C for large Striped Bass varied as much as 3.8-times among years sampled. The 2018 estimate of C of small fish was third highest of the time-series (Figure 20).

Estimates of proportion of empty stomachs (PE) of small sized Striped Bass during fall, 2006-2018, ranged between 0.10 and 0.57 (Figure 21). PE was 0.35 in 2018. 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 (Figure 21).

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 (score = 1) for small fish; annual estimates of P0 for small Striped Bass were at the threshold during 1998-2000 (Figure 21). The highest PE point 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 a score of 3). PE in 2018 was between the target and threshold and was assigned a score of 2. Estimates of PE steadily fell for small sized fish during 2006-2011 and varied between the target and threshold PE afterward (Figure 21).

Estimates of PE of large sized Striped Bass during fall, 2006-2013, ranged between 0.40 and 0.63 (Figure 22). Estimates of PE of large sized fish fell to 0.10-0.29 during 2014-2016, then rose to 0.60 in 2017, and fell again to 0.18 in 2018. Lowest estimates of PE for large fish (2013-2016 and 2018) could be separated from remaining higher estimates based on 90% confidence interval overlap. 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 (within the period of threshold P0) and we used this estimate to derive a threshold PE for large sized fish (0.58). The 90% CI's during 2006, 2011-2012, and 2017 overlapped this threshold. Estimates of PE and their CI's have been substantially lower than the threshold since 2014 (except 2017; Figure 22).

Median PPLRs of large prey (Spot and Atlantic Menhaden) were noticeably smaller for large Striped Bass (0.19-0.30) than for small ones (0.21-0.38) during 2006-2009, 2012, and 2015-2017; they tracked closely in remaining years (Figure 23). During 2006-2018, median PPLRs for large Striped Bass were much closer to the optimum (0.21 based on Overton et al. 2009) than for small fish. The PPLRs for small fish were particularly high (0.34-0.38) during 2012, 2015, 2016, and 2018 (Figure 23). Large major prey were not found in the diets of small fish in 2017 and estimates of PPLR could not be made.

Correlation and regression analyses among C, PE, median PPLR for large major prey, and P0 for each year and size class indicated that small Striped Bass would have more difficulty in catching and handling large major prey than large fish in any given year and that at least one feeding metric was associated with P0. For small fish, only the correlation of large major prey PPLR with PE and C with P0 were strong enough for consideration ($r = 0.67$, $P = 0.019$ and $r = -0.69$, $P = 0.013$)), while for large fish correlations of PE with P0 and PE with C were strongly correlated ($r = 0.69$, $P = 0.009$ and $r = -0.59$, $P = 0.034$, respectively). The linear regression of PE of large fish against PE of small fish during 2006-2018 was positive and significant at $P = 0.08$ ($r^2 = 0.25$; slope = 0.53 with an SE = 0.27 and intercept = 0.16 with a SE = 0.12; See

Figures 21 and 22 for underlying data). Residuals were well correlated with small fish PPLR of large major prey ($r = 0.72$, $P < 0.0085$; Figure 24), suggesting that differences in PE of small versus large fish would be influenced by size of major prey.

The CPUE3 index was synchronous with the abundance of age 3 Striped Bass estimated by ASMFC (2019) assessment after 1992 (1985-2015 time-series; Figure 25), 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 (2019) assessment indicated abundance of age 3 Striped Bass was very low during 1985-1989 (Figure 25).

Estimated catchability continuously declined during 1985-1996 and appeared to stabilize afterward (Figure 26). A linear regression of CPUE3 against abundance at age 3 during 1996-2017 did not indicate a trend in catchability ($r^2 = 0.05$, $P = 0.33$) and the observed 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 27). These low HI_3 year-classes were followed by the appearance of intermittent appearances of large year-classes at age 3 in 1996, 1998, 1999, 2004, 2006, 2010, 2014, and 2018. The HI_3 indicated sharper changes in relative abundance of age 3 Striped Bass from year-to-year than the ASMFC (2019) assessment. Peaks generally aligned, but years of low abundance in the ASMFC (2019) assessment tended to be higher than would have been indicated by the hybrid gill net index (Figure 27).

Ninety percent CIs of relative survival (SR; HI_3 / JI_{t-3}) allowed for separation of years of high and low survival, and some years in between (Figure 28). 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 28). The 42% percent reduction in median SR between1986-1996 (median SR = 36.4) and 1997-2018 (median SR = 21.3) was very close to changes in tag-based estimates of survival of large-sized fish during the same period, from 77% annual survival (1987-1996) to 44% (1997-2017), a 43% reduction (based on Table B8.25 in ASMFC 2019).

The target for SR was $\geq$ 38.0 (score = 3) and the threshold was $\leq$ 20.0 (score = 1). After 1998, target SR was reached in 2010, 2011, and 2017 (Figure 29). After 2004, threshold conditions were met in 2007, 2008, 2012, and 2016 (Figure 29).

Targets and thresholds scores for P0, RI, FR, PE, and SR are summarized in Table 4.

The IF varied from 1.0 to 3.0 during 1998-2018 (Figure 30). During 1998-2004, the IF was low, between 1.0 and 1.25. The IF increased to 2.25 in 2005, fell below 2 in 2006-2007, and then increased to 2.5 to 3.0 during 2008-2011. After 2011, it varied from above 1.4 to 2.4. IF was 1.8 during 2018. Spread of annual component scores was narrower (no more than 1 unit during 1998-2004 when the IF was consistently low and 2008-2011 when IF was consistently high (Figure 30).

Estimates of mean IF with each component removed indicated little variation from the overall IF (Figure 31). The maximum deviation from the overall IF in any given year and metric ranged between -0.42 to 0.40 and averaged -0.06 to 0.03 (Figure 30). This approach suggested that IF means could be separated into high, medium, and low categories.

Discussion

The IF indicated threshold to near threshold foraging conditions for Striped Bass in upper Bay (scores at or near 1) were typical during 1998-2004. IF scores (1.4 or above) were elevated beyond the threshold after 2004. IF scores during 2008-2011 (IF =2.6-3.0) were near or at the target (best foraging conditions), then IF fell into an intermediate region (1.4-2.4). It has been near 2.0 (does not breach threshold or target) during 2017-2018, indicating some recovery from poorer foraging conditions during 2015-2016 (Scores 1.4-1.6).

A rapid rise in Striped Bass abundance in upper Bay during the mid-1990s, followed by a dozen more years at high abundance after recovery was declared in 1995, coincided with declines in relative abundance of Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (i.e., major pelagic and benthic prey) to low levels. Survival of small and large sized Striped Bass in upper Bay shifted downwards in the mid-1990s and poor survival has persisted. Striped Bass were often in poor condition during fall 1998-2004 and vulnerable to starvation. Improvements in condition after 2007 coincided with lower Striped Bass abundance, 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.

A return of Striped Bass to noticeably higher abundance after 2014 was not shared by major forage, but condition has not declined to threshold conditions. It appears that slight increases in Atlantic Menhaden relative abundance, while statistically insignificant, may have biological significance for both size classes of upper Bay Striped Bass. Consumption of Atlantic Menhaden by small and large Striped Bass since 2013 has been higher, more frequently ranking in the top half of estimates of C during 2006-2018.

Recent divergence in P0 between small and large Striped Bass (2016 and 2018) may indicate relief from a prey bottleneck for large fish. Estimates of PE improved for large fish, but not small ones, since 2014. The PPLR for small Striped Bass, reflecting larger relative size of Atlantic Menhaden, has been consistently high since 2015. Small Striped Bass would have more difficulty in catching and handling large major prey than large fish in any given year. Low consumption of Atlantic Menhaden by small Striped Bass during 2016-2017 was not offset by other prey

The IF approach was based on performance of the five metrics during periods of high and low P0; P0 provided a measure of condition and potential for starvation that was well-related to feeding of Striped Bass in the laboratory (Jacobs et al. 2013). The IF integrated four other metrics: RI provided an index of relative abundance Striped Bass (demand); weighted FR (focal species combined) and PE in fall provided trends in supply relative to demand based on relative abundance and diet sampling, respectively; and SR (along with P0) was an indicator of Striped Bass well-being.

The inclusion of RI in the IF may need to be reconsidered if there is a substantial rise in major prey FRs due to an increase in prey. Under the current low forage regime, the abundance of Striped Bass appears to be a major driver of foraging and well-being. If FRs increase because abundance of forage increases (and well-being increases with it), then RI may become a source of negative bias in the IF. The RI would end up indicating threshold conditions even though it has become well supported by forage.

Even though negative correlations used to estimate a weighted combined FR were not significant at $P \leq 0.05$, we felt the correlation coefficients provided the best available measure of the influence of each forage species relative to Striped Bass demand on condition of Striped Bass. Other possibilities considered were equal weighting (each item has the same relative value; used in Uphoff et al. 2018) or using prey average individual mass (resulting in an Atlantic Menhaden index for most years).

Simulated 90% confidence intervals for ratio based indicators (RI, FRs, and SR) generally allowed for separation of high and low values and, in most cases, mid-level values could be determined. We reduced the number of categories for metrics that comprised the scoring system from five to three based on this generalization. Nearly all years for the five metrics could be separated from zero, with the exception of the earliest years of the RI (reflecting Maryland's moratorium and restriction of catch and release fishing for Striped Bass). Poorest or threshold conditions of each metric were assigned a score of 1, best or target conditions were assigned a 3, and intermediate conditions were indicated by a score of 2. Based on the variation of IF indices using the "leave one out" approach (Figure 31), we believe it may be better to place an upper boundary for threshold conditions (1.5) that capture all of the threshold period (1998-2004), i.e., threshold conditions are indicated by IF scores between 1.0 and 1.5. Similarly, a lower boundary for target conditions that capture the target period (2008-2011) would bound target conditions between 2.5 and 3.0.

This report and previous ones used correlation and regression analyses to explore to what degree indicators of upper Bay Striped Bass abundance, forage abundance, consumption, and relative survival estimates were linked to the body fat condition indicator. Some metrics were statistically linked to one another, but not so tightly that one would adequately represent another. Statistical analyses can provide insight into important processes related to predation (Whipple et al. 2000), but 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). Ideally, manipulative experiments and formal adaptive management should be employed, but these are not possible for us (Hilborn 2016). 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).

High variability in component scores of the IF may reflect sampling issues, nonlinear, asymptotic relationships among variables, lagged responses, potential insensitivity of some indices, behavioral changes that 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; 2018) and the reader is referred to them.

Estimates of C and PPLR provided supplemental information for evaluating PE. Estimates of C indicated which prey were predominate in the diet and an indication of relative biomass consumed. 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 if caught (Lundvall et al. 1999). Median PPLRs were noticeably smaller for large than small fish in some years and were much closer to the optimum for large fish. PPLRs for small fish were generally higher during after 2011; this shift coincides with slightly elevated Atlantic Menhaden FR and low and declining Bay Anchovy FR. Higher PPLR ratios (indicating larger sized major prey) were positively associated with a higher PE for small Striped Bass, but not large ones. The linear regression of PE of large fish against PE of small fish and correlation analysis of the regression's residuals suggested that PE of small fish would be influenced by PPLR. Small fish were more likely to have more difficulty in catching and handling larger major prey than large fish in any given year.

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 1998; 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 1-3 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 (Halliday 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 (Halliday 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) method.

Two objectives of 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 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.

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.

There was some variation in size classes 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. However, recent divergence of P0 between size classes may warrant a size class specific approach. 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 in fall than large sized fish (latter was dominated by age 0 Atlantic Menhaden) and PE of the small size class was used as an indicator of forage availability (Uphoff et al. 2017). Estimates of SR reflected survival of small sized fish (between late age-0 to early 3 year-olds).

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 benthic invertebrates other than Blue Crab are important diet items during winter and spring, respectively (Walter et al. 2003; Hartman and Brandt 1995c; Overton et al. 2009; 2015). 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).

The utility of estimates of biomass of invertebrates comprising a benthic IBI in Maryland's portion of the by used for water quality monitoring was explored in Uphoff (2018). An update of benthic biomass component of this index was not available in time for this report. McGinty et al. (Job 3) developed a complementary index for hard (oyster) bottom in this year's report. These two benthic indices are considered supplemental information at this time that may provide clues on changes in fall condition that appear to be outliers. Uphoff et al. (2018) found that P0 the previous summer and the previous fall could influence P0; condition of small Striped Bass in summer may be influenced by benthic invertebrates since they can be a significant component of their spring diet (Overton et al. 2015). These benthic invertebrate indices will also be useful for forming hypotheses for exploring anglers concerns about changes in popular benthic gamefish such as Spot and Atlantic Croaker *Micropogon undulatus*.

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). During 2017, P0 (score = 3) contradicted remaining indicators (except SR). Conflict between SR and P0 might be expected since SR indicates survival of younger, smaller Striped Bass (1 and 2 year-olds) than many of the fish that make up the small category (typically in an ascending size range encompassing ages 1-4; Uphoff et al. 2014) and deviations of SR should not be considered true outliers (also see below). Outliers occurred twice in 21 years, indicating about 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.

An underlying assumption of the SR is 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 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 not always sufficiently represented in diet samples.

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).

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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 Bass.
CV	Coefficient of variation.
F	Instantaneous annual fishing mortality rate.
FR	Mean major forage ratio score (mean of scores assigned to standardized major prey to Striped Bass ratio
FWHP	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.
IF	Forage index. Mean score for five indicators of forage status (FR, PE, P0, RI, and SR)
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).
PPLR	Ratio of prey length to predator length.
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. Correlations among species and gear-specific forage ratios (FRs) and the correlations of FRs with proportion of Striped Bass without body fat (P0; last row of table) during 1998-2018. N = 21 for all comparisons.

Species	Gear	Statistic	Menhaden Seine	Anchovy Seine	Spot Seine	Spot Trawl	Anchovy Trawl	Blue Crab Dredge
Anchovy	Seine	r	0.7301					
		P	0.0002					
Spot	Seine	r	0.67466	0.3253				
		P	0.0008	0.1502				
Spot	Trawl	r	0.65966	0.34379	0.97986			
		P	0.0011	0.127	<.0001			
Anchovy	Trawl	r	0.16241	0.134	0.08295	0.05884		
		P	0.4818	0.5625	0.7207	0.8		
Blue Crab	Dredge	r	0.59888	0.37251	0.335	0.37159	-0.1007	
		P	0.0041	0.0963	0.1377	0.0972	0.664	
Body fat	P0	r	-0.4143	-0.1267	-0.2614	-0.3107	0.49592	-0.3338
		P	0.0619	0.5842	0.2525	0.1705	0.0222	0.1393

Table 3. 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
2018	6	330	87

Table 4. Criteria for assigning IF scores (1, 2, or 3) to metrics for P0, RI, FR, and PE. A score of 1 indicates threshold (poor) conditions and a score of 3 indicates target (good) conditions. Intermediate conditions (score = 2) fall between values for scores of 1 or 3.

	Score	
Metric	1	3
P0	≥ 0.68	≤ 0.30
RI	≥ 2.0	< 2.0
FR	≤ 0.20	≥ 0.38
PE	≥ 0.54	≤ 0.31
SR	≤ 20	> 38

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.

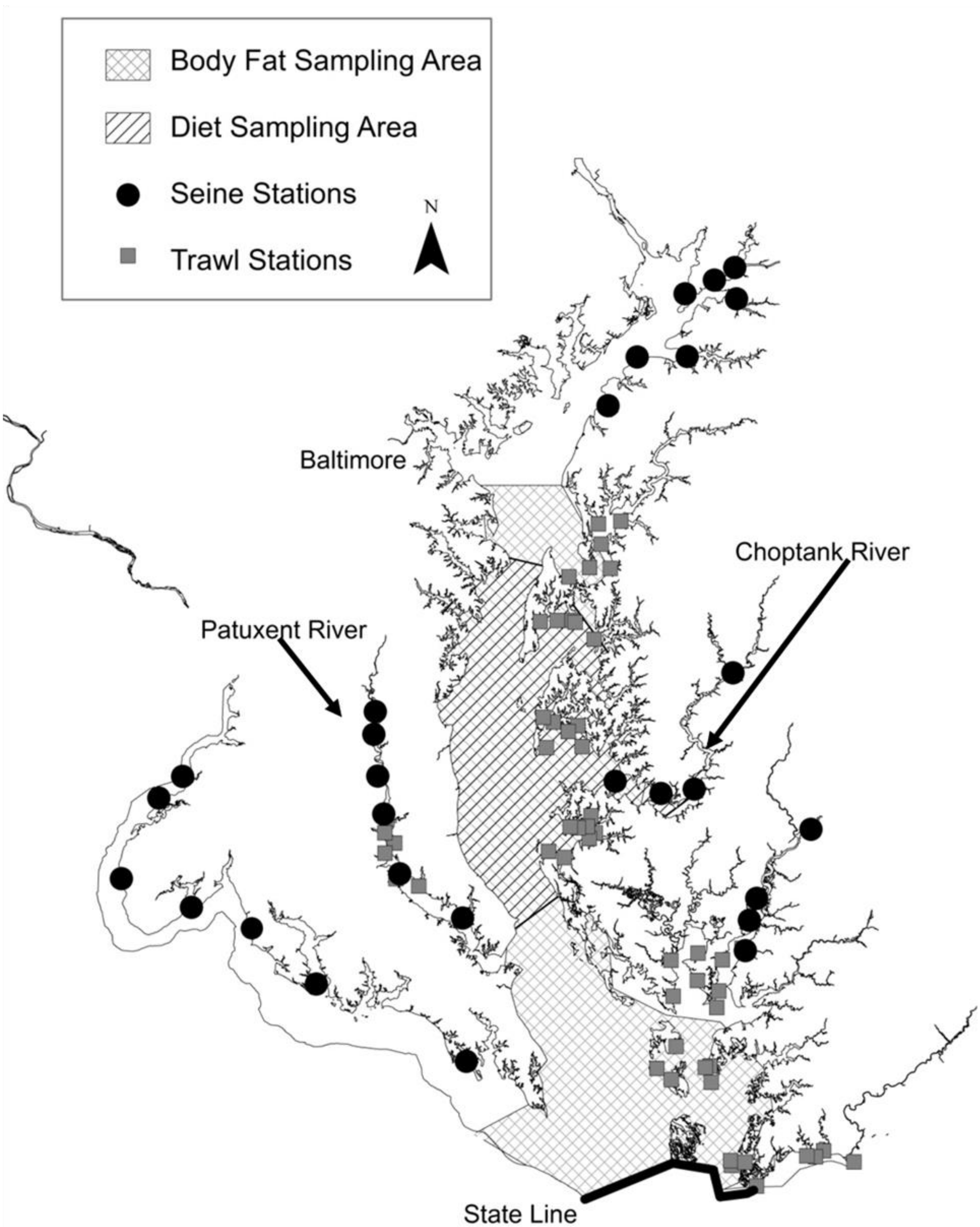

Figure 2. Proportion of Striped Bass without body fat (P0) during October-November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

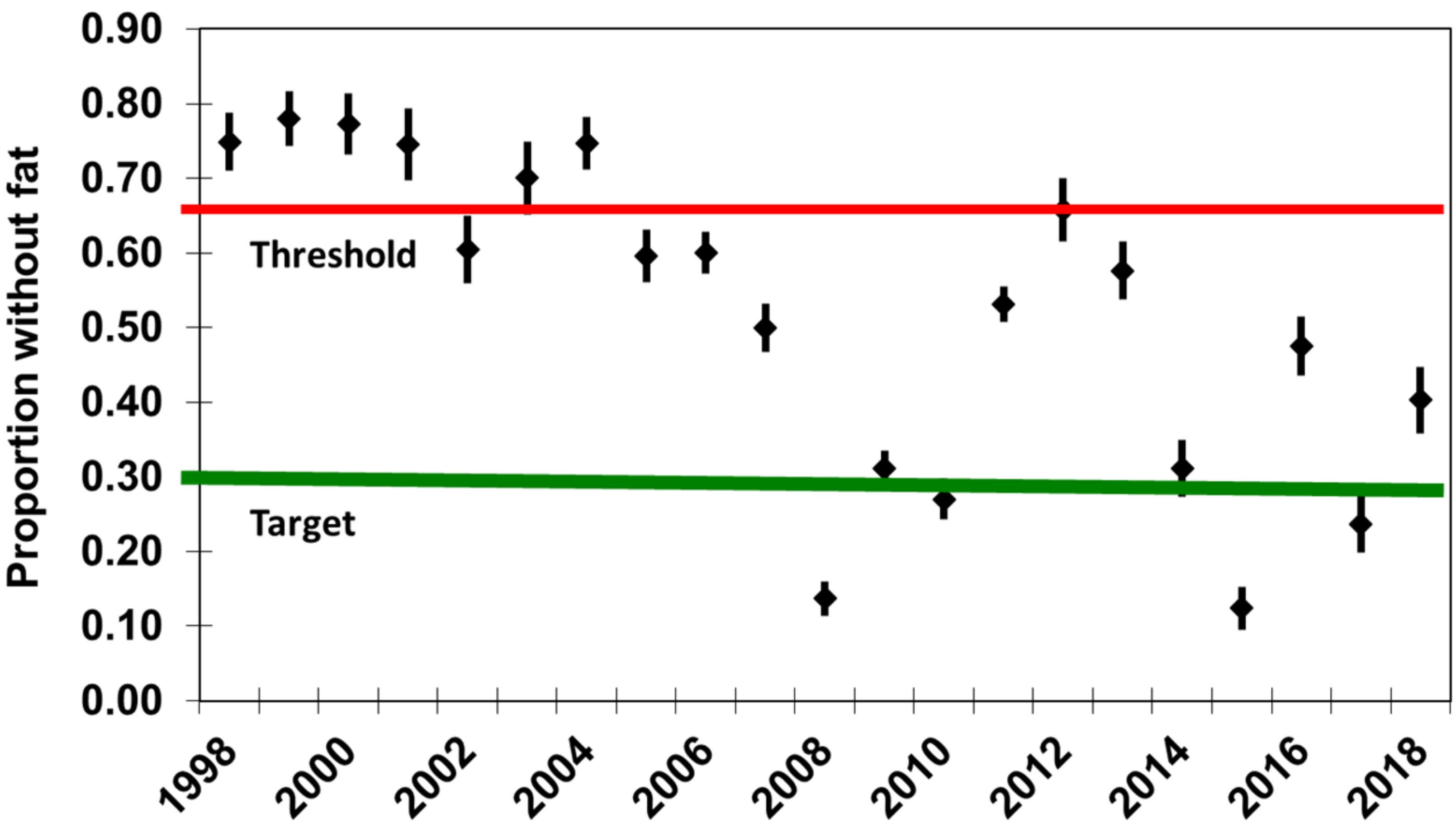

Figure 3. Trends in fall body fat indices (P0) for small (280-456 mm) and large striped bass

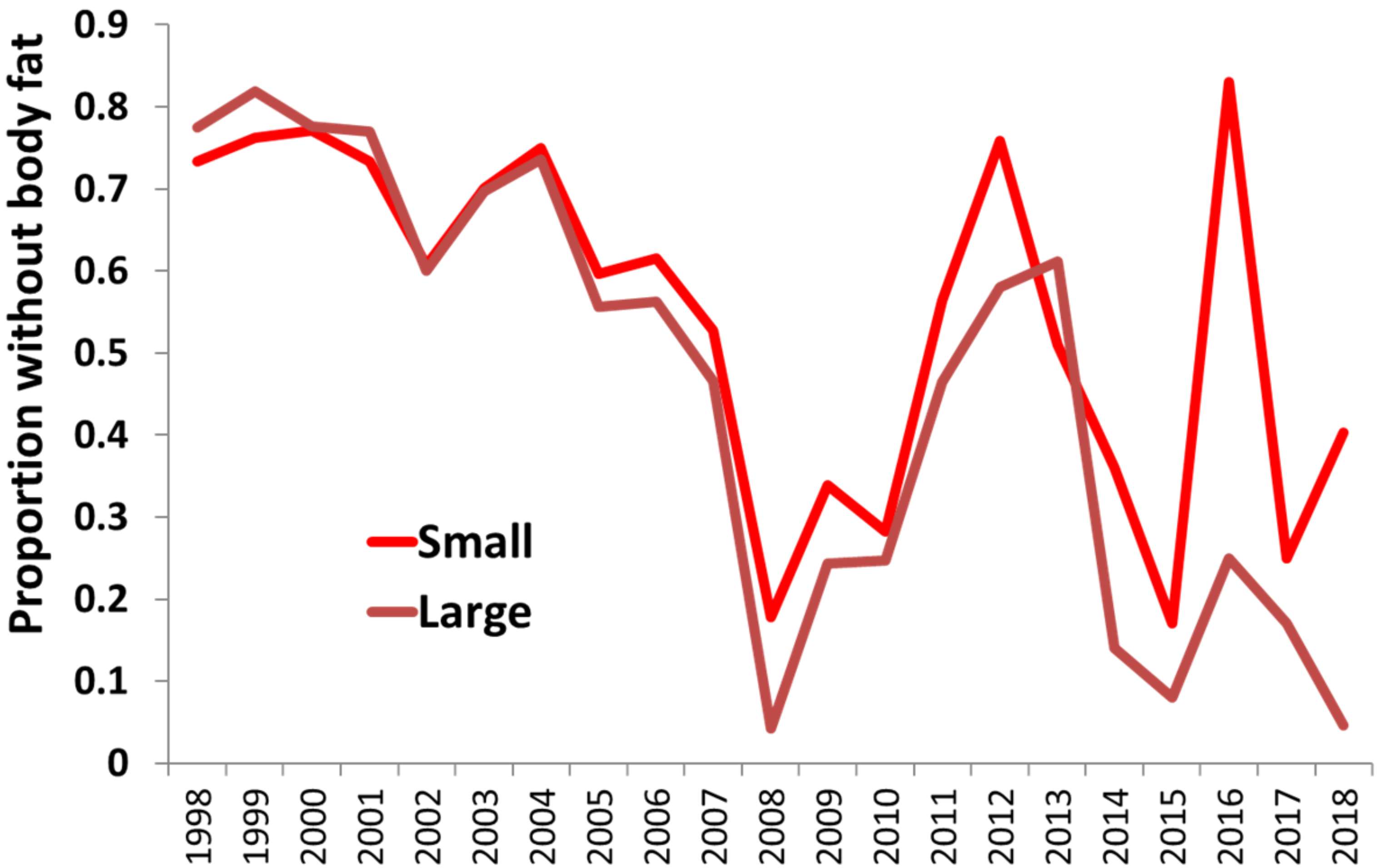

Figure 4. Trends in major pelagic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2018. Indices were standardized to their 1989-2018 means (years in common). Menhaden = Atlantic Menhaden and Anchovy = Bay Anchovy.

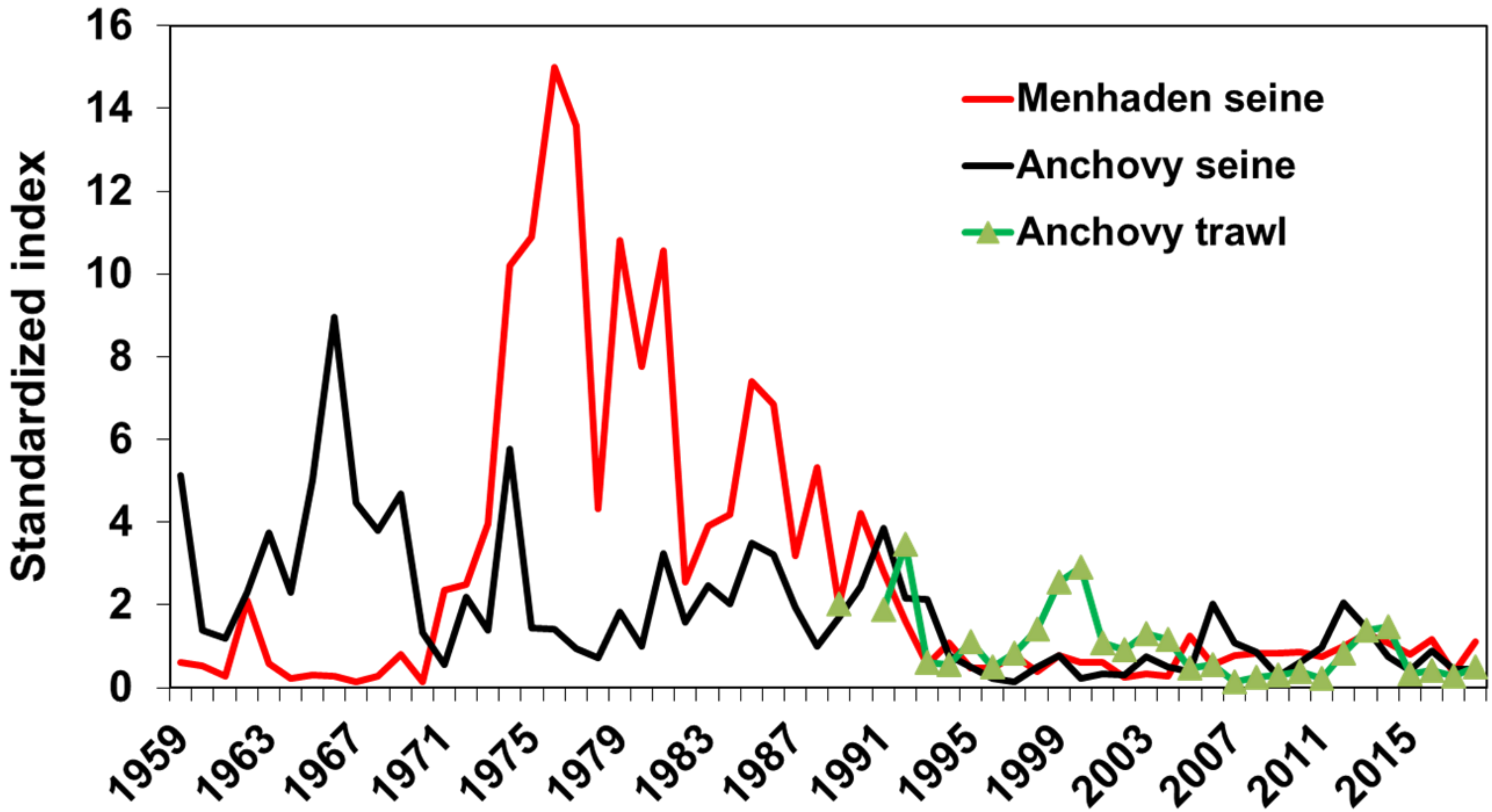

Figure 5. Trends in major benthic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2018. Indices were standardized to their 1989-2018 means (years in common).

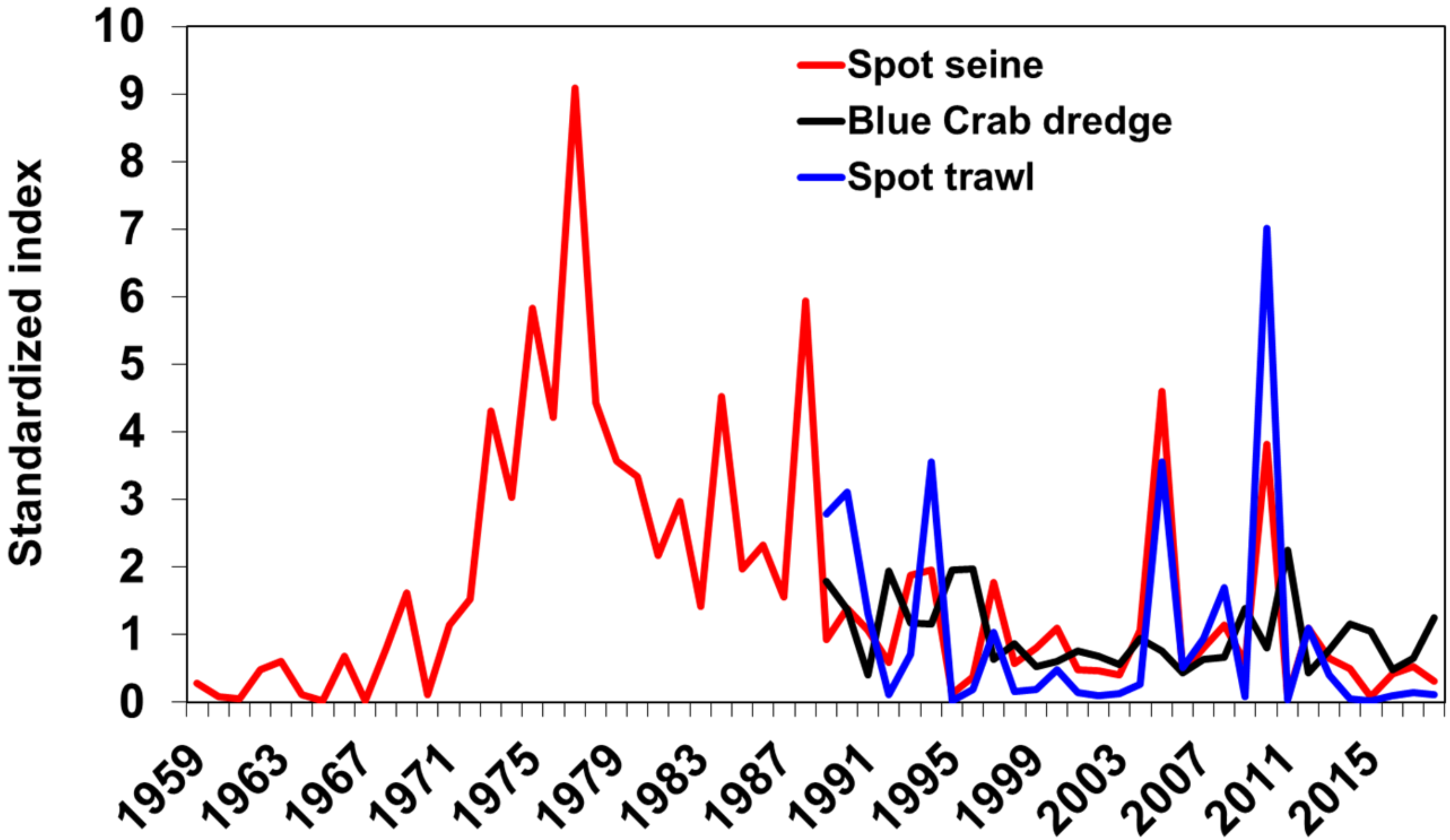

Figure 6. Maryland resident Bay Striped Bass annual abundance index (RI; MD MRIP recreational catch per private boat trip; mean = black line) during 1981-2018 and its 90% confidence intervals based on @Risk simulations of catch and effort distributions. Catch = number harvested and released.

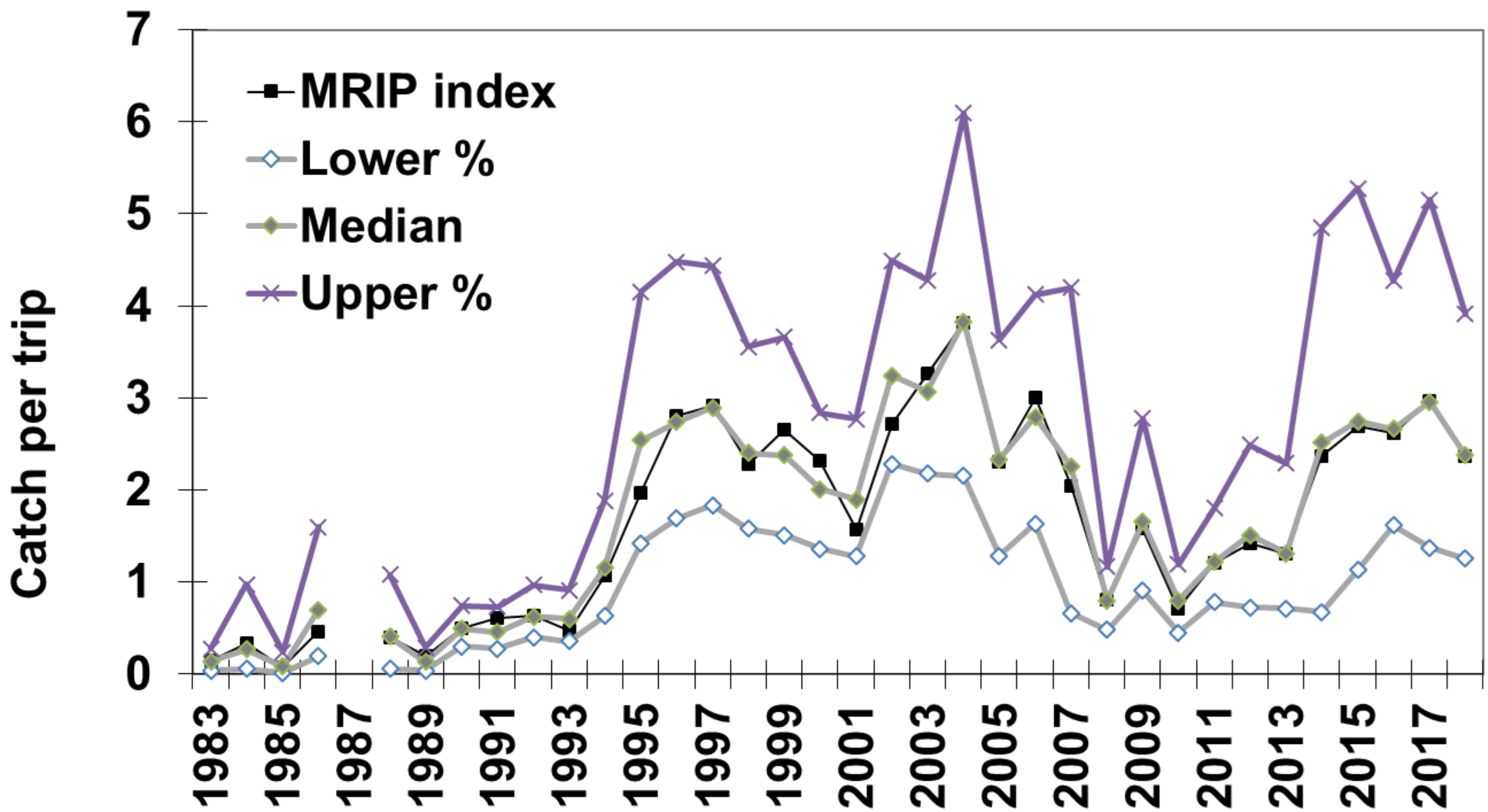

Figure 7. Comparison of trends of abundance of ages 2-5 Striped Bass estimated by the current stock assessment (N ages 2-5; ASMFC 2019) and the RI index (September-October catch per private / rental boat trip).

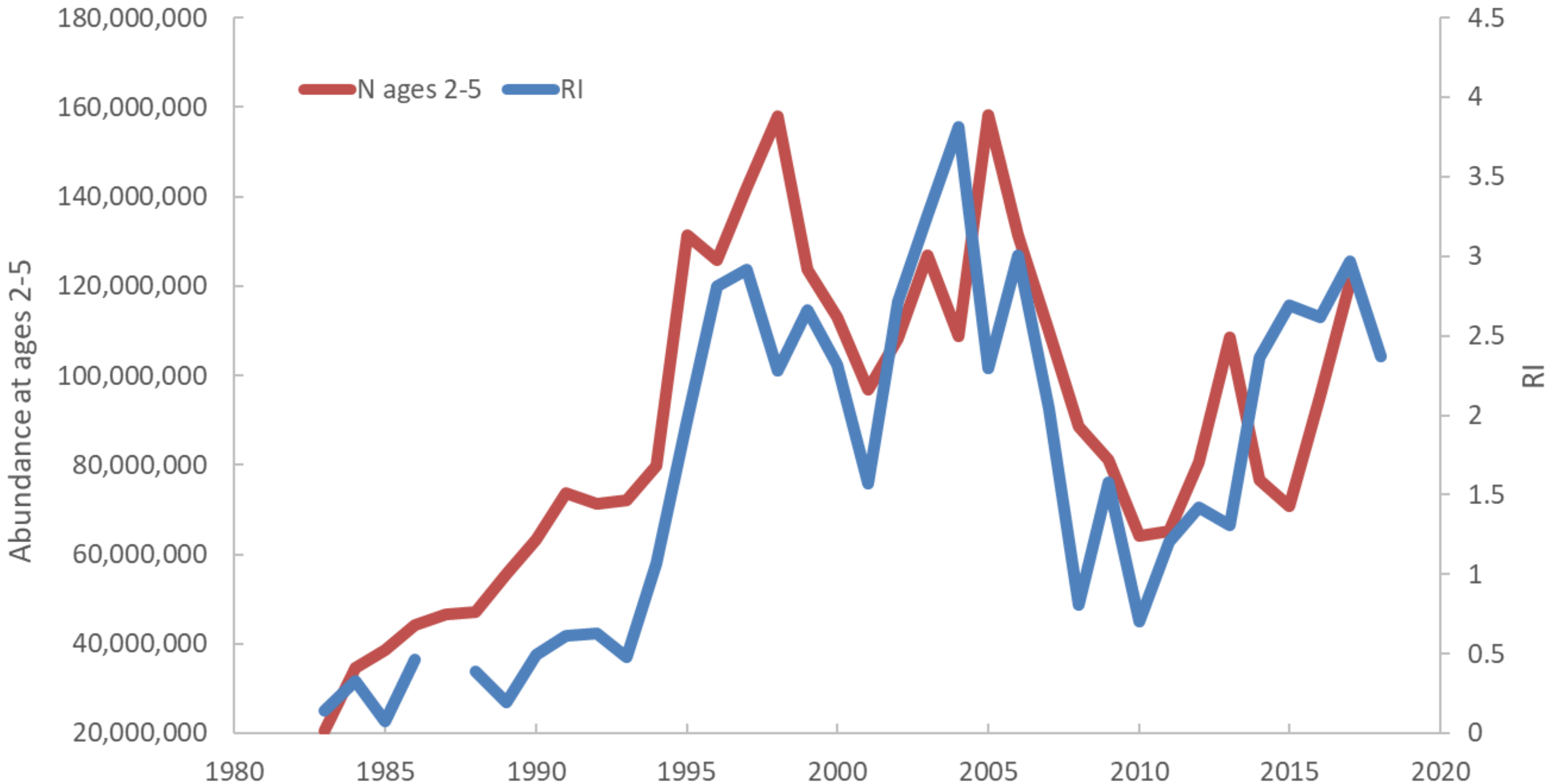

Figure 8. Atlantic Menhaden index to Striped Bass index (RI) ratios during 1983-2018 and their 90% confidence intervals based on @Risk simulations of Atlantic Menhaden seine indices and RI distributions. Note $\log_{10}$ scale on Y-axis.

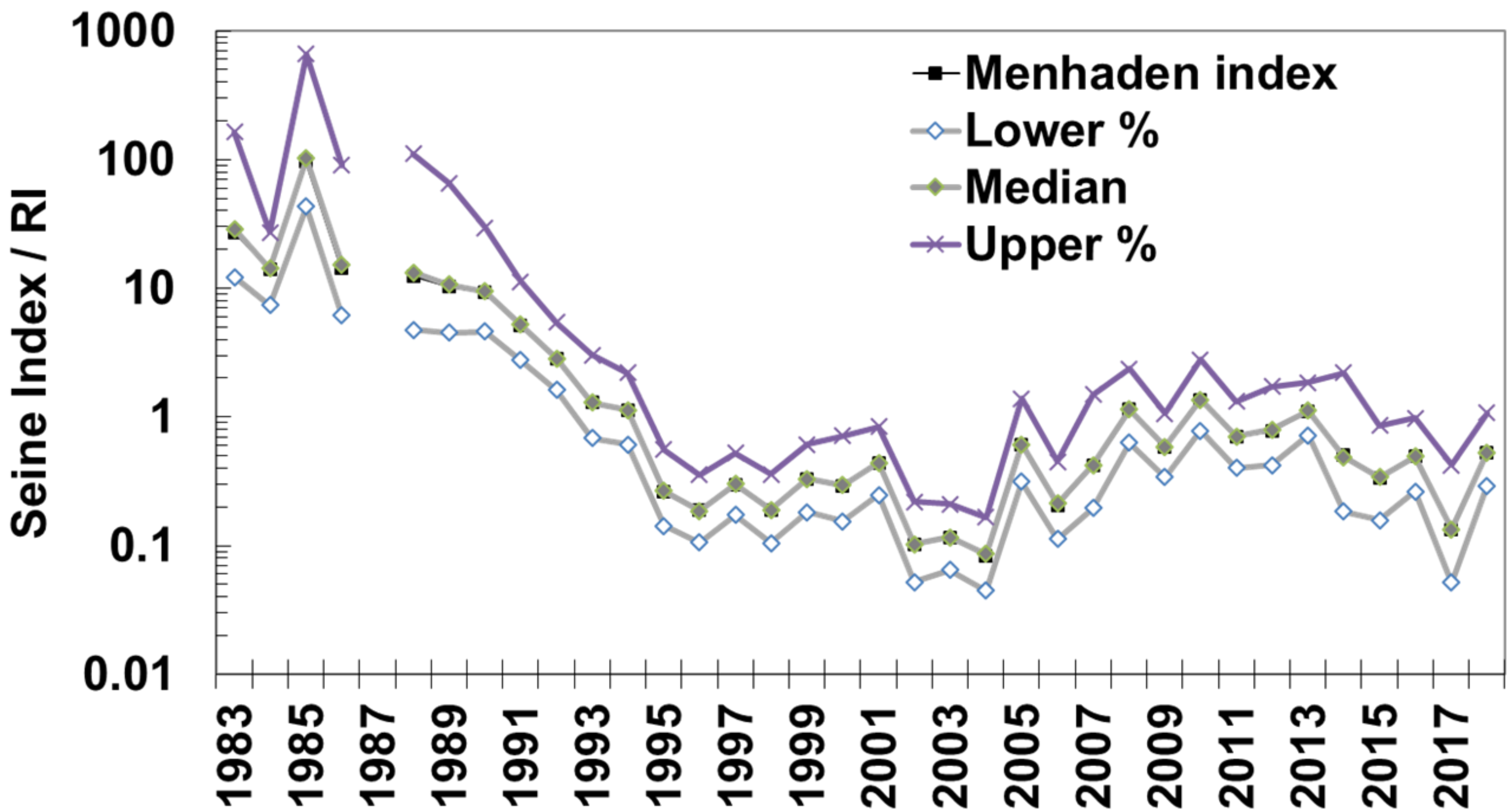

Figure 9. Bay Anchovy seine index to Striped Bass index (RI) ratios during 1983-2018 and their 90% confidence intervals based on @Risk simulations of Bay Anchovy seine indices and RI distributions. Note $\log_{10}$ scale on the Y-axis.

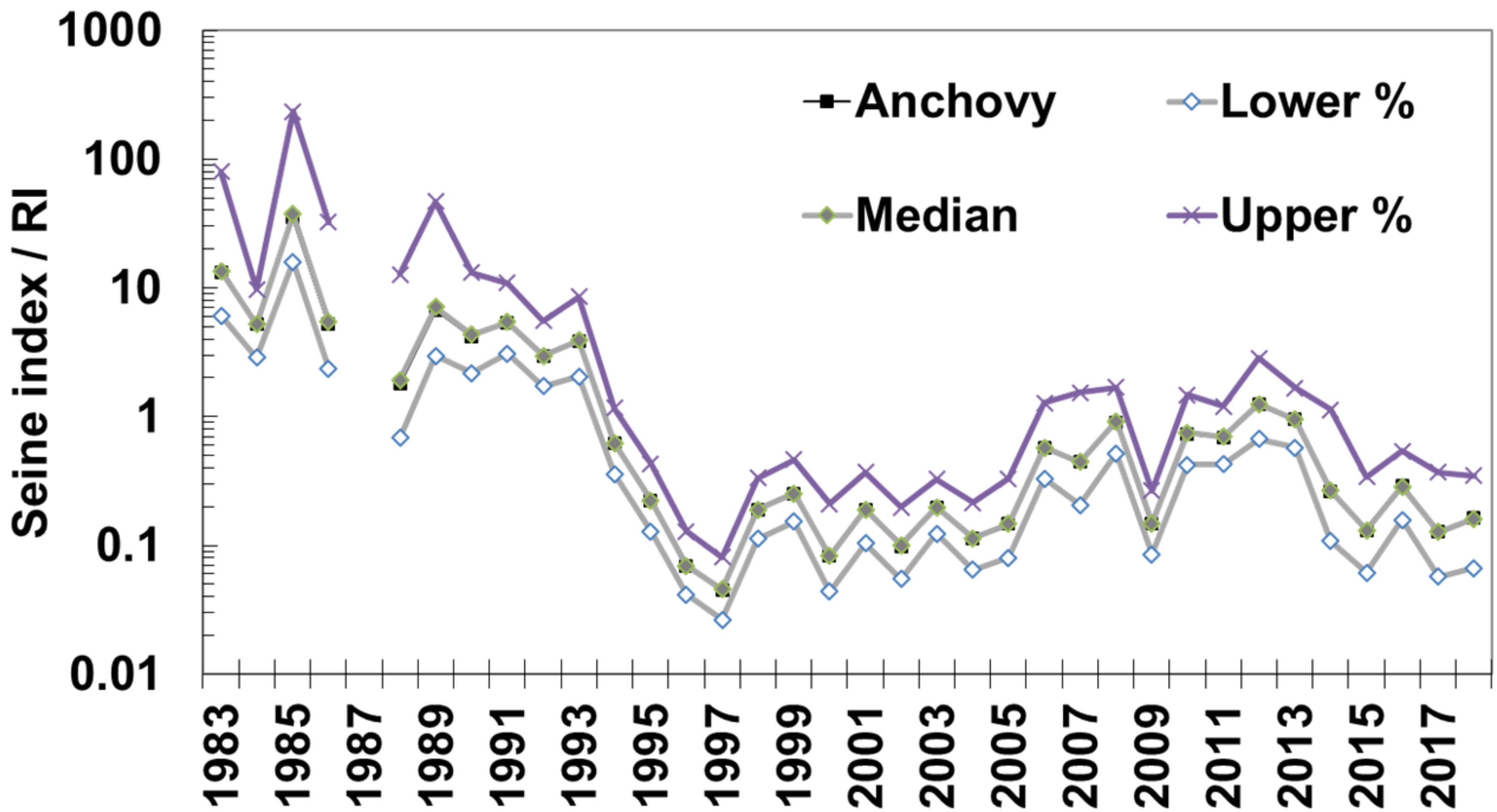

Figure 10. Bay Anchovy trawl index to Striped Bass index (RI) ratios during 1989-2018 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale on Y-axis.

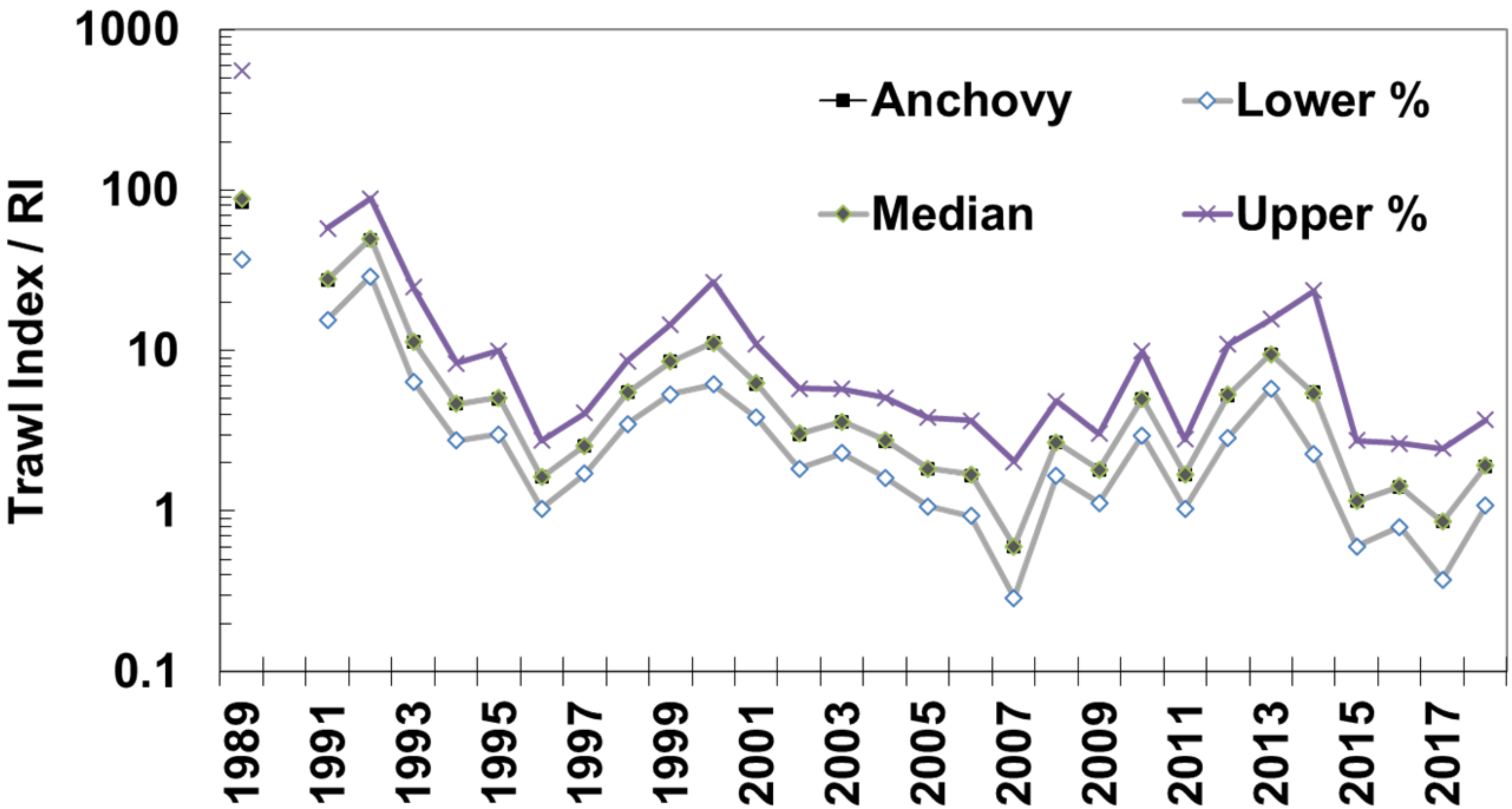

Figure 11. Spot seine index to Striped Bass index (RI) ratios during 1983-2018 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Spot seine indices and RI. Note $\log_{10}$ scale on Y-axis

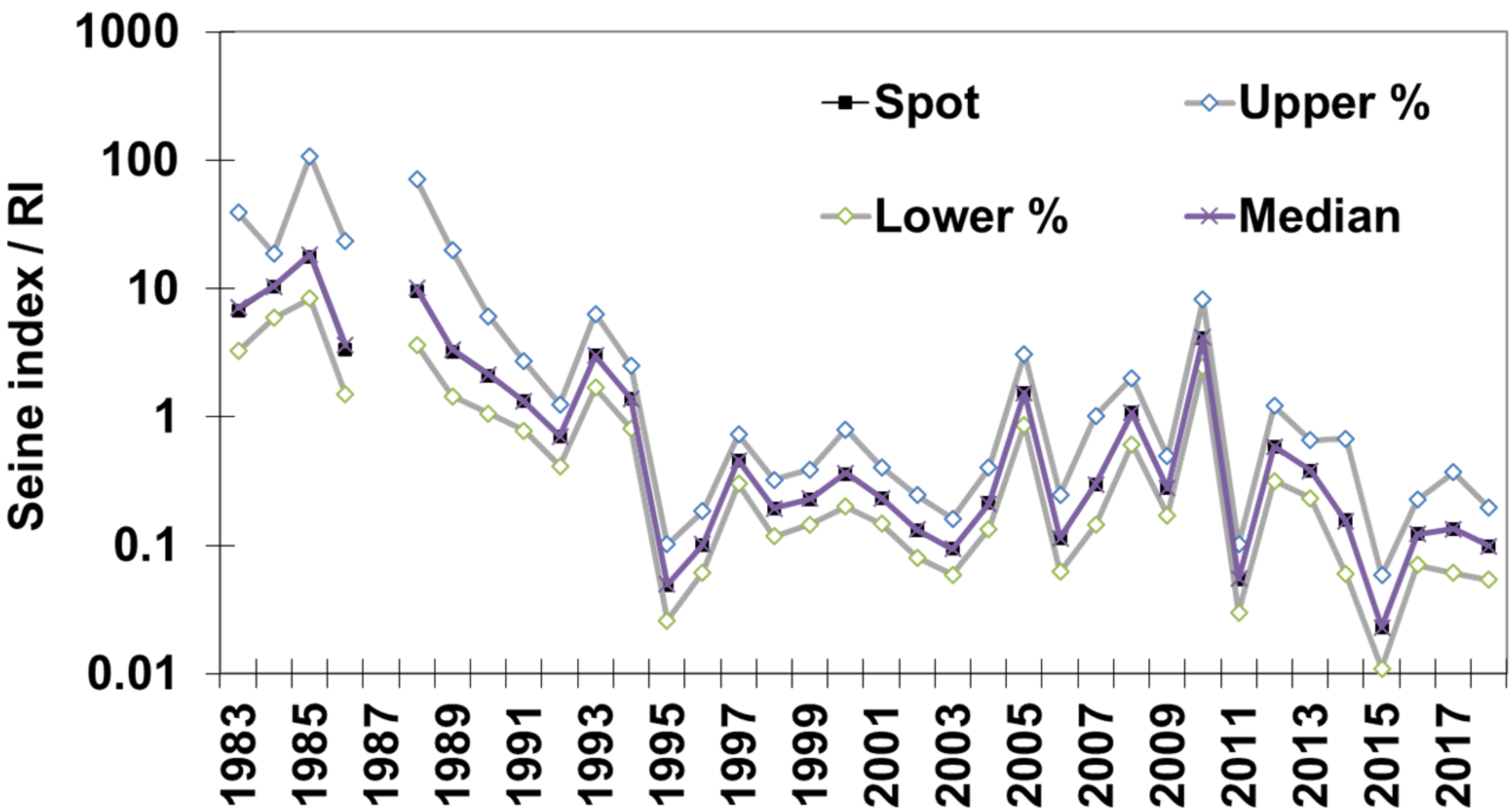

Figure 12. Spot trawl index to Striped Bass index (RI) ratios during 1989-2018 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale on Y-axis.

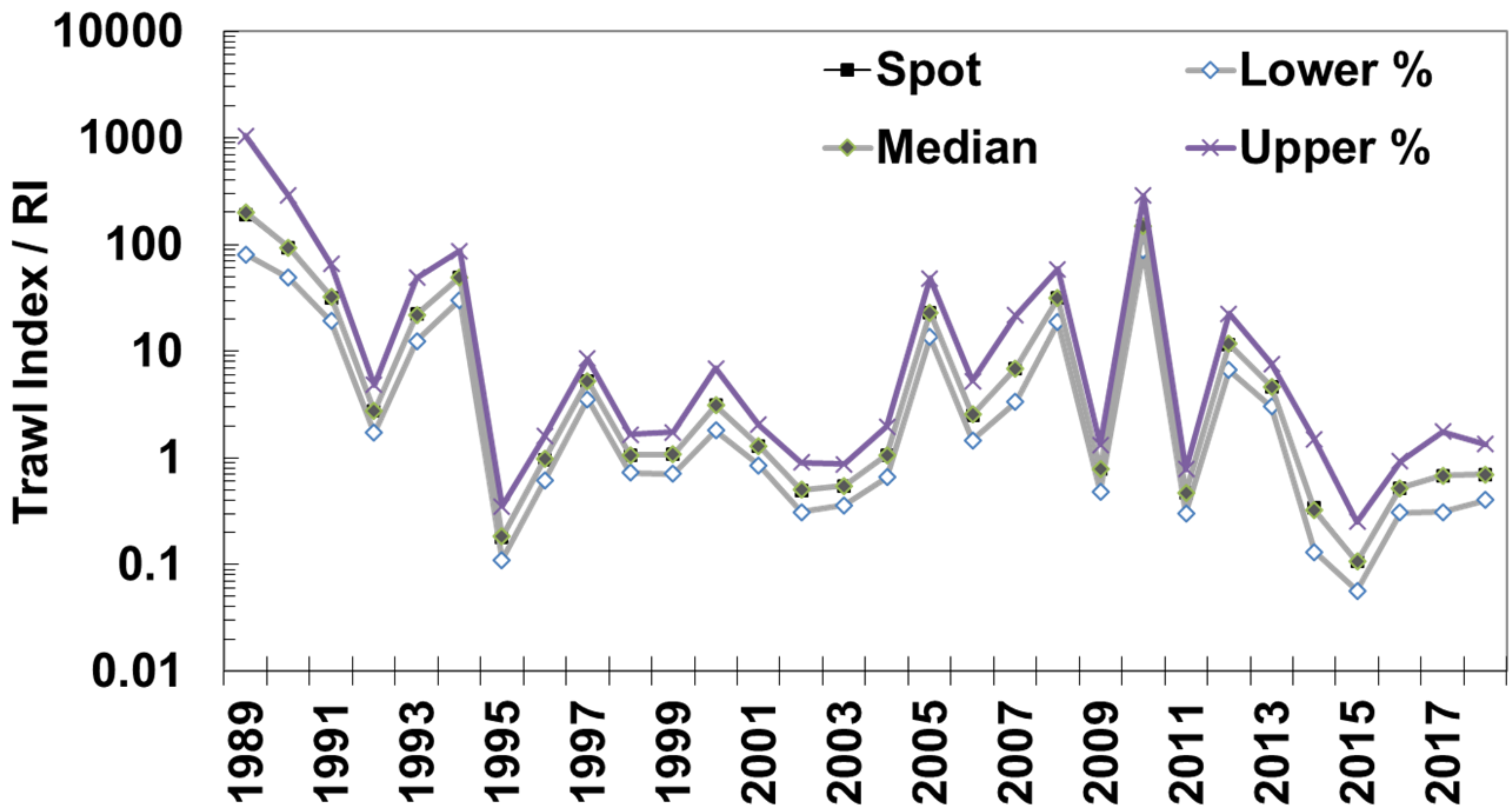

Figure 13. Blue Crab index to Striped Bass index (RI) ratios during 1989-2018 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Blue Crab (age-0) winter dredge densities and RI. Note $\log_{10}$ scale on Y-axis.

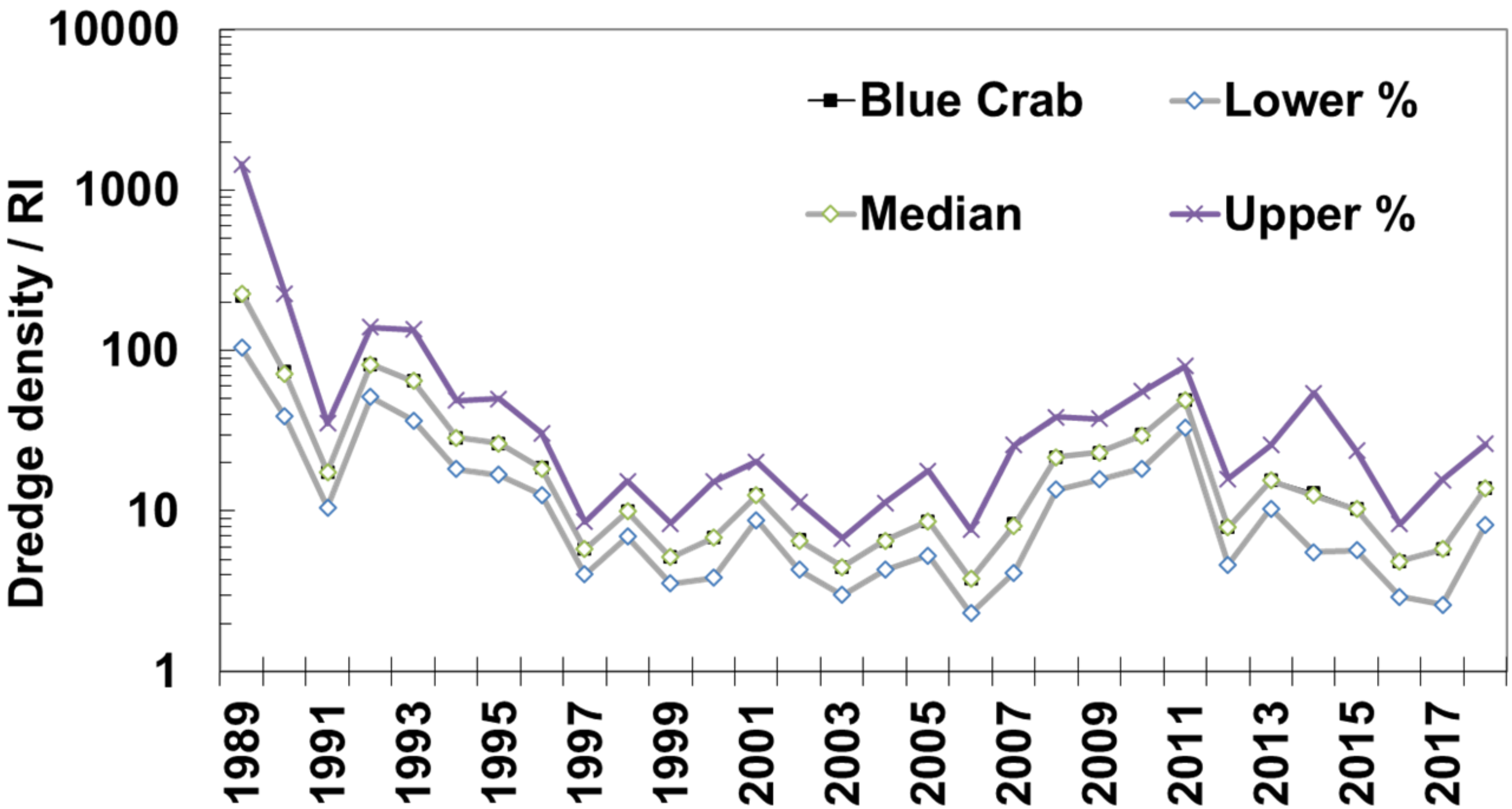

Figure 14. Trends of standardized ratios major upper Bay forage species indices to Striped Bass relative abundance (RI). Forage ratios have been standardized to their 1989-2018 mean to place them on the same scale. S indicates a seine survey index; T indicates a trawl survey index; and D indicates a dredge index. Note the $\log_{10}$ scale on Y-axis.

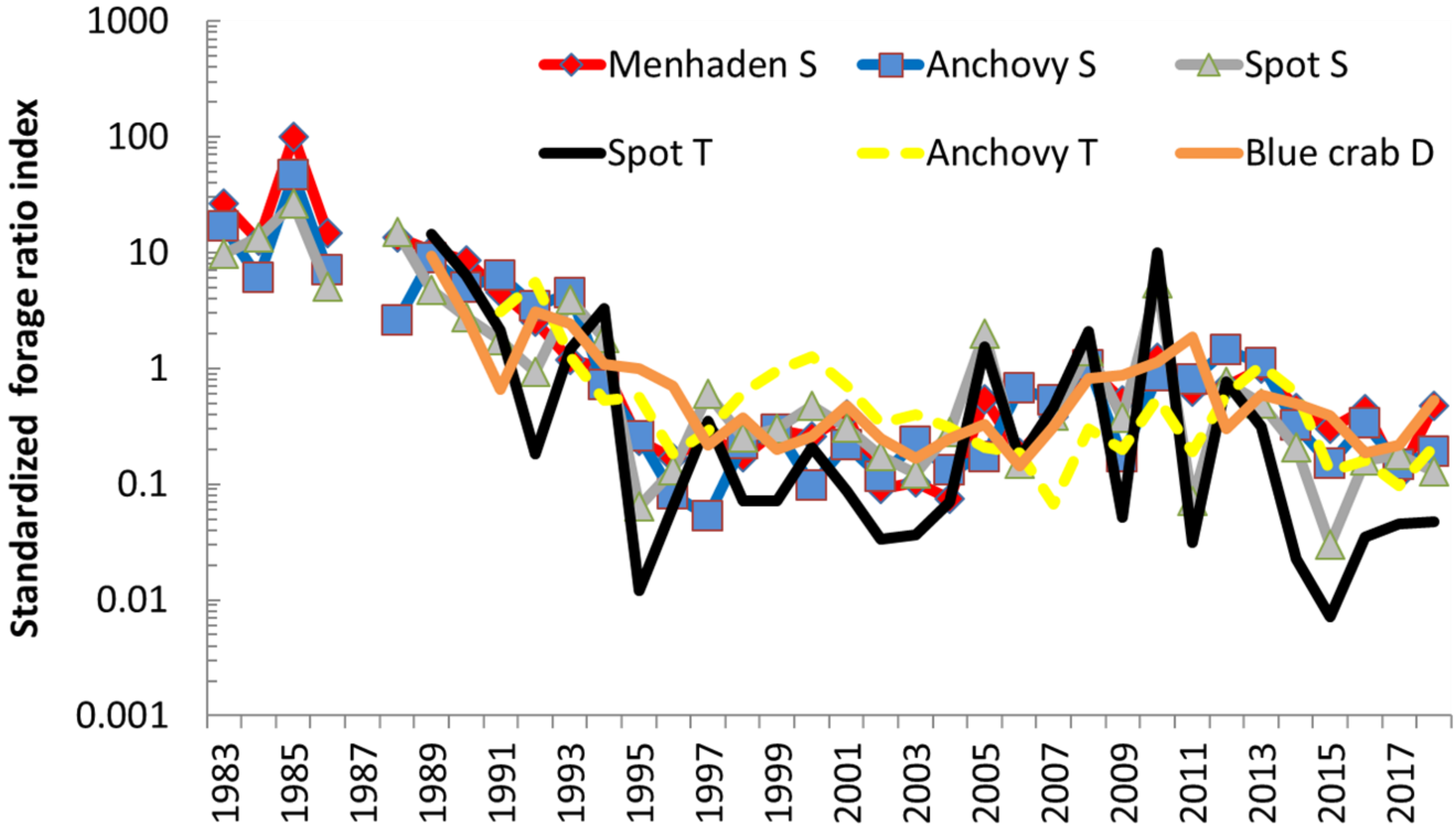

Figure 15. Standardized ratios of major forage indices / Striped Bass RI and their weighted mean during the time period when body fat (P0) indices were available.

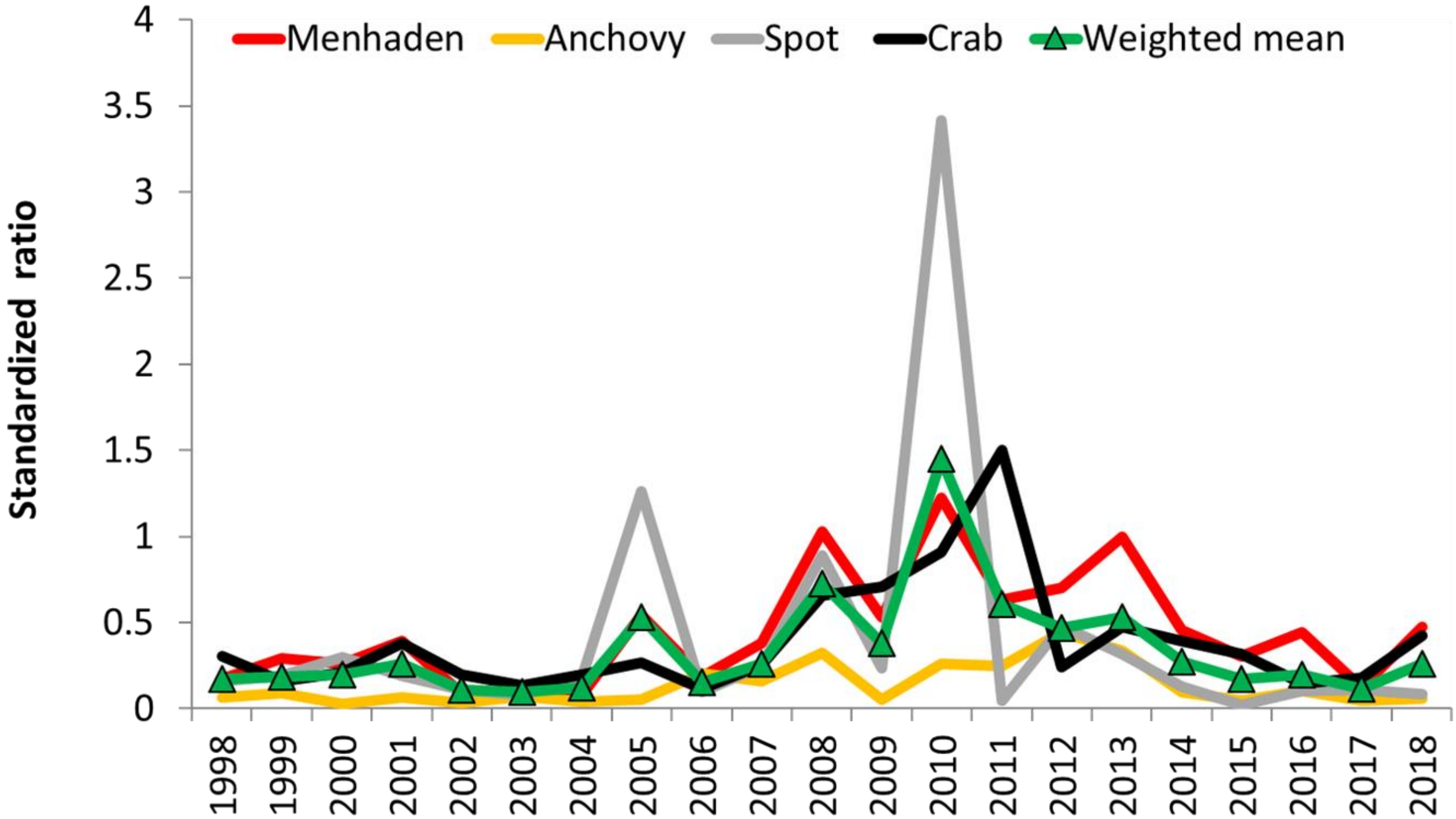

Figure 16. Weighed Standardized ratios of major forage indices (FR) / Striped Bass relative abundance (RI) and associated targets and thresholds during the time period when body fat (P0) indices were available.

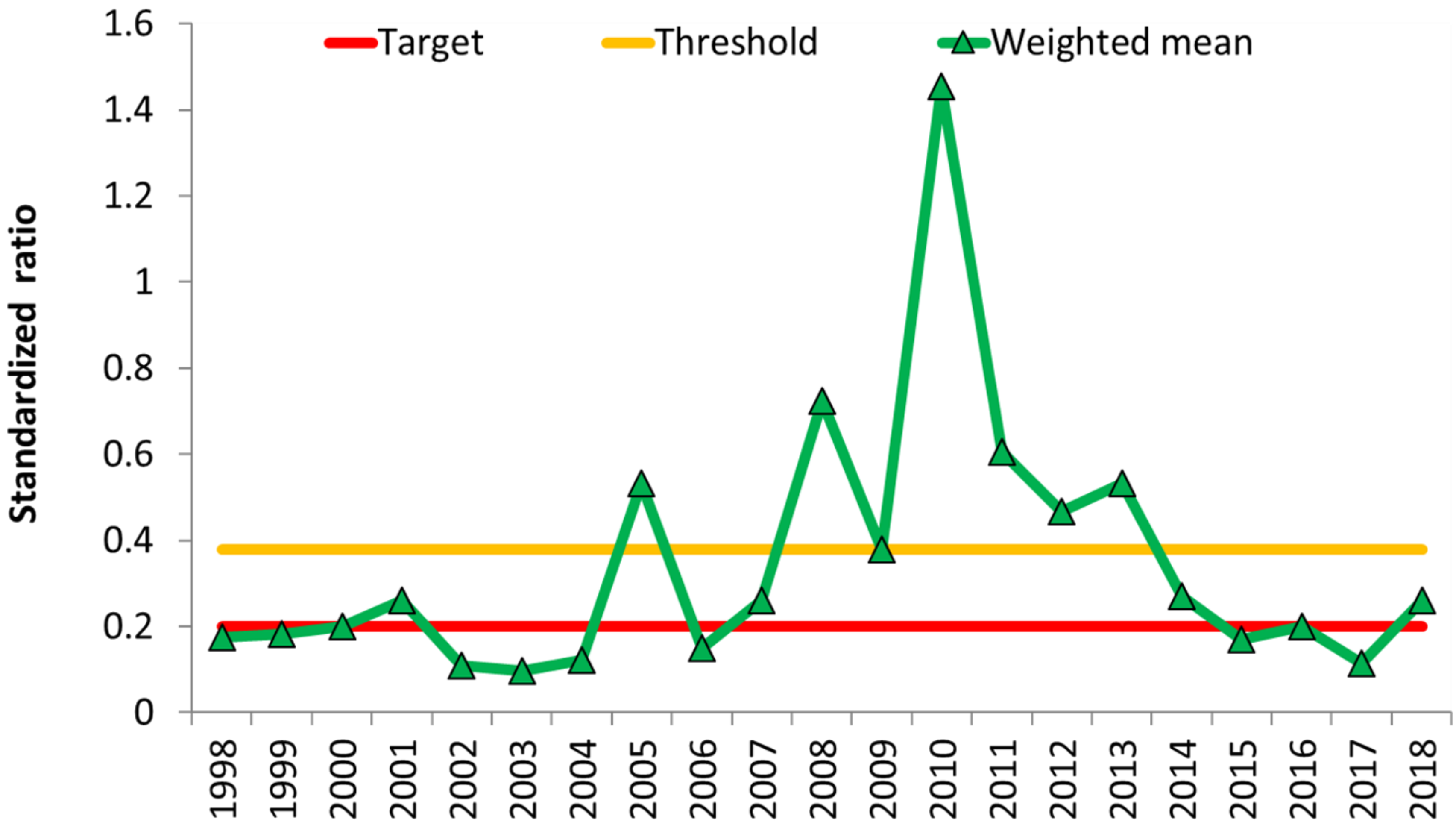

Figure 17. Percent, by number (counts of individuals plus presence of parts), of identifiable (excludes unknown) major forage groups in small Striped Bass (< 457 mm TL) guts, in fall.

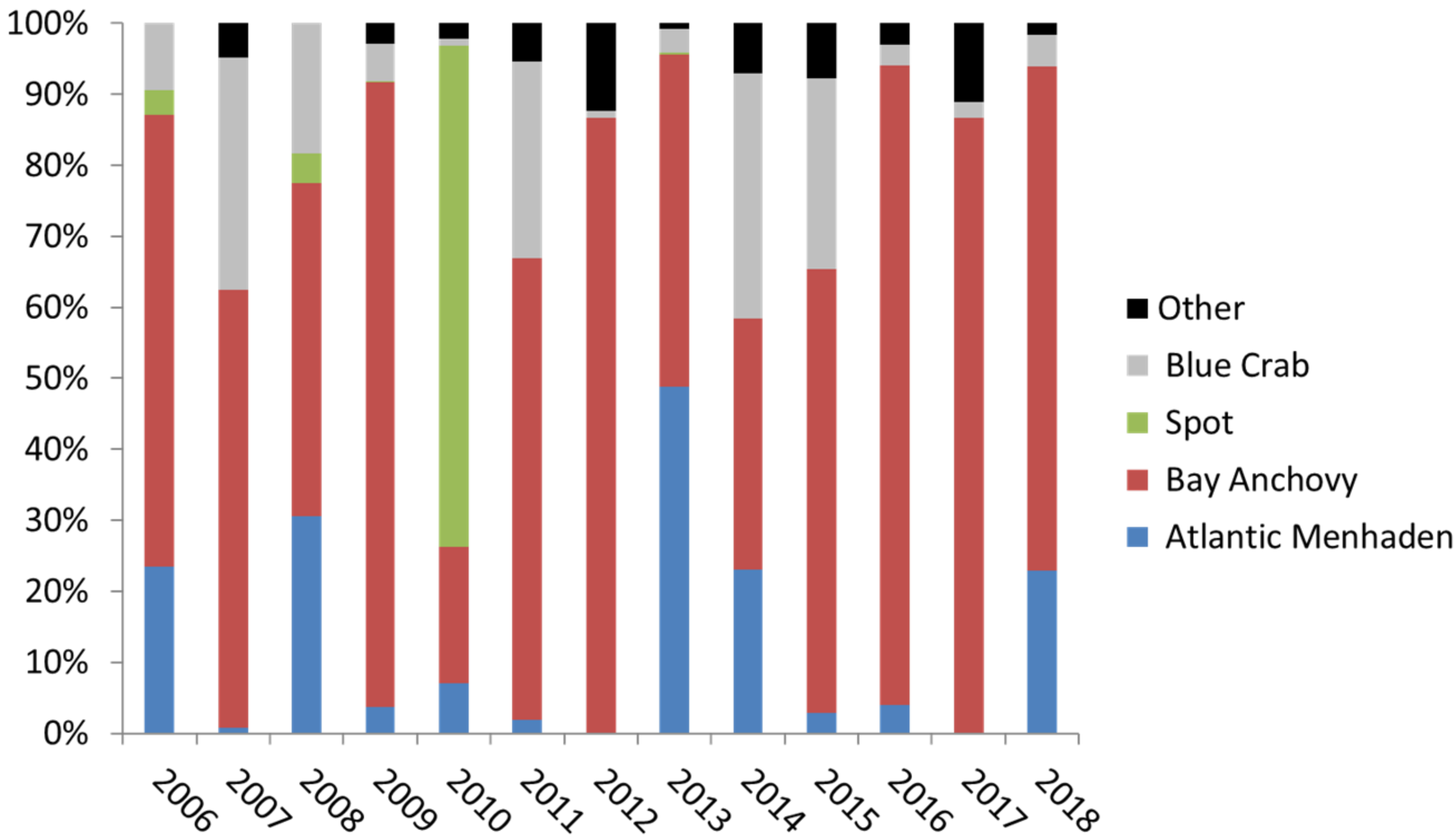

Figure 18. Percent of large Striped Bass (≥ 457 mm TL) diet represented by major forage groups, by number, in fall.

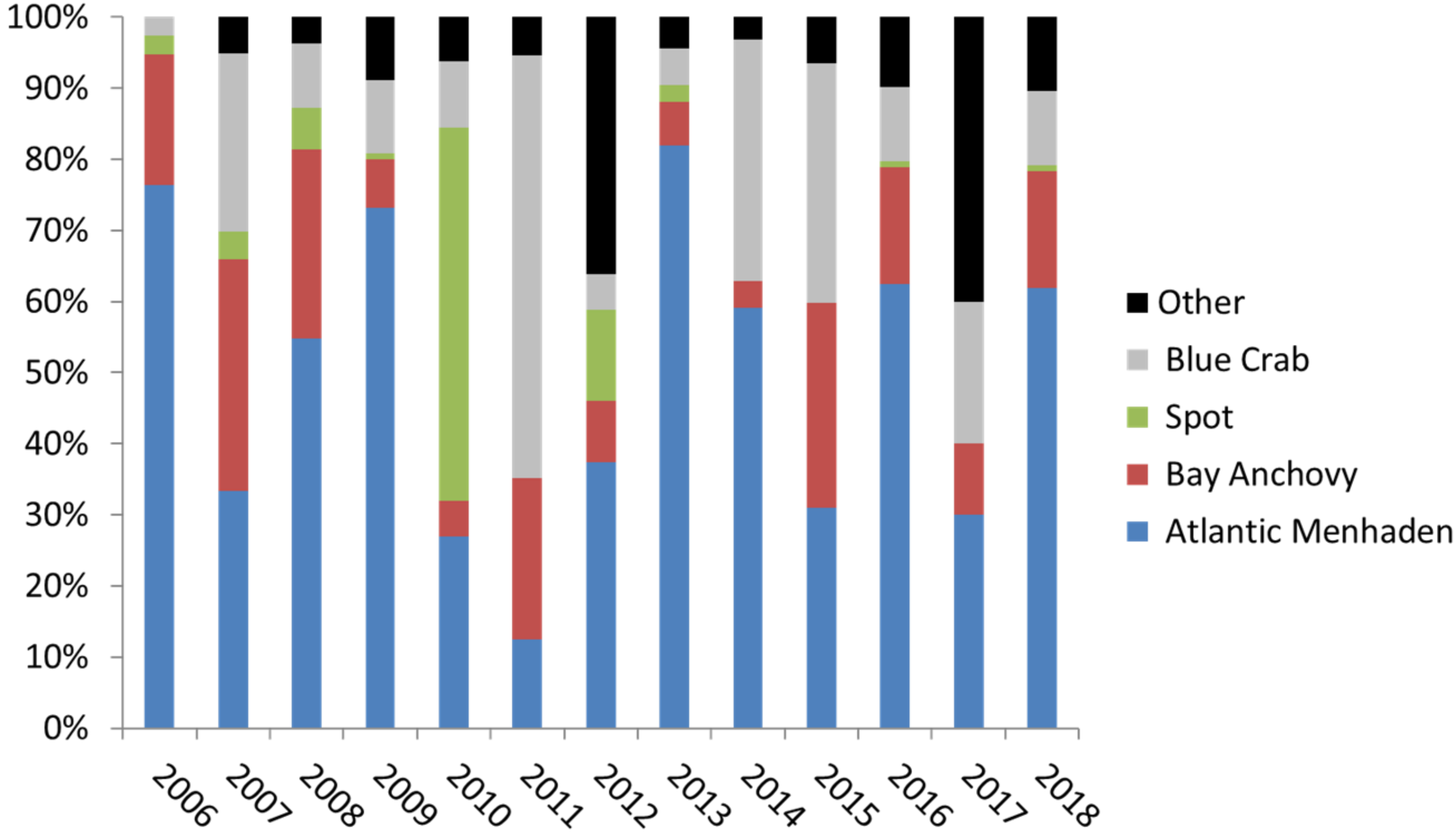

Figure 19. Gram prey consumed per gram (C) of small (< 457 mm TL) Striped Bass in fall hook-and-line samples. Age-0 forage dominate the diet. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

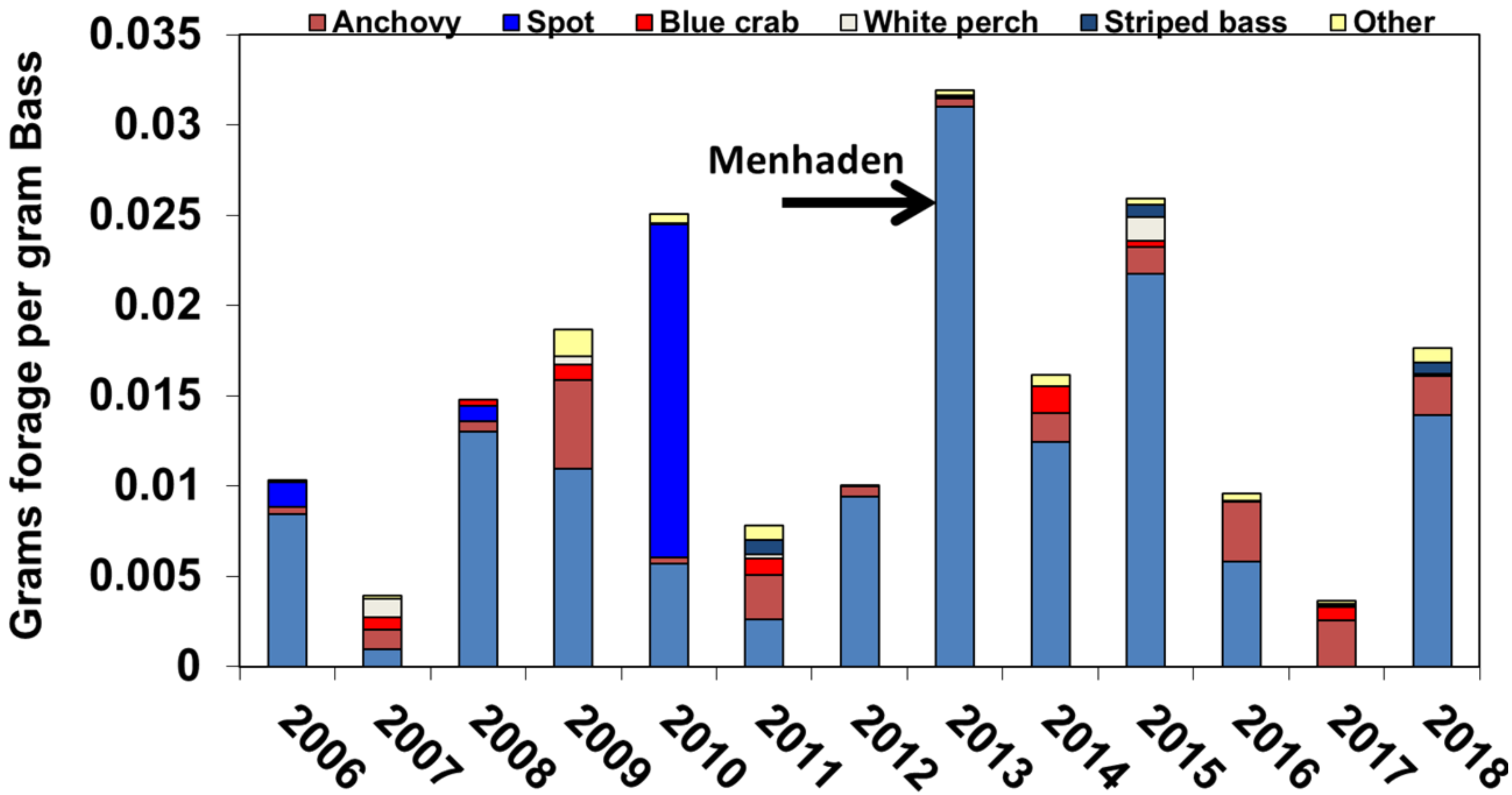

Figure 20. Grams of prey consumed per gram (C) of large (≥ 457 mm TL) Striped Bass during October-November. Fall consumption dominated by age 0 forage. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

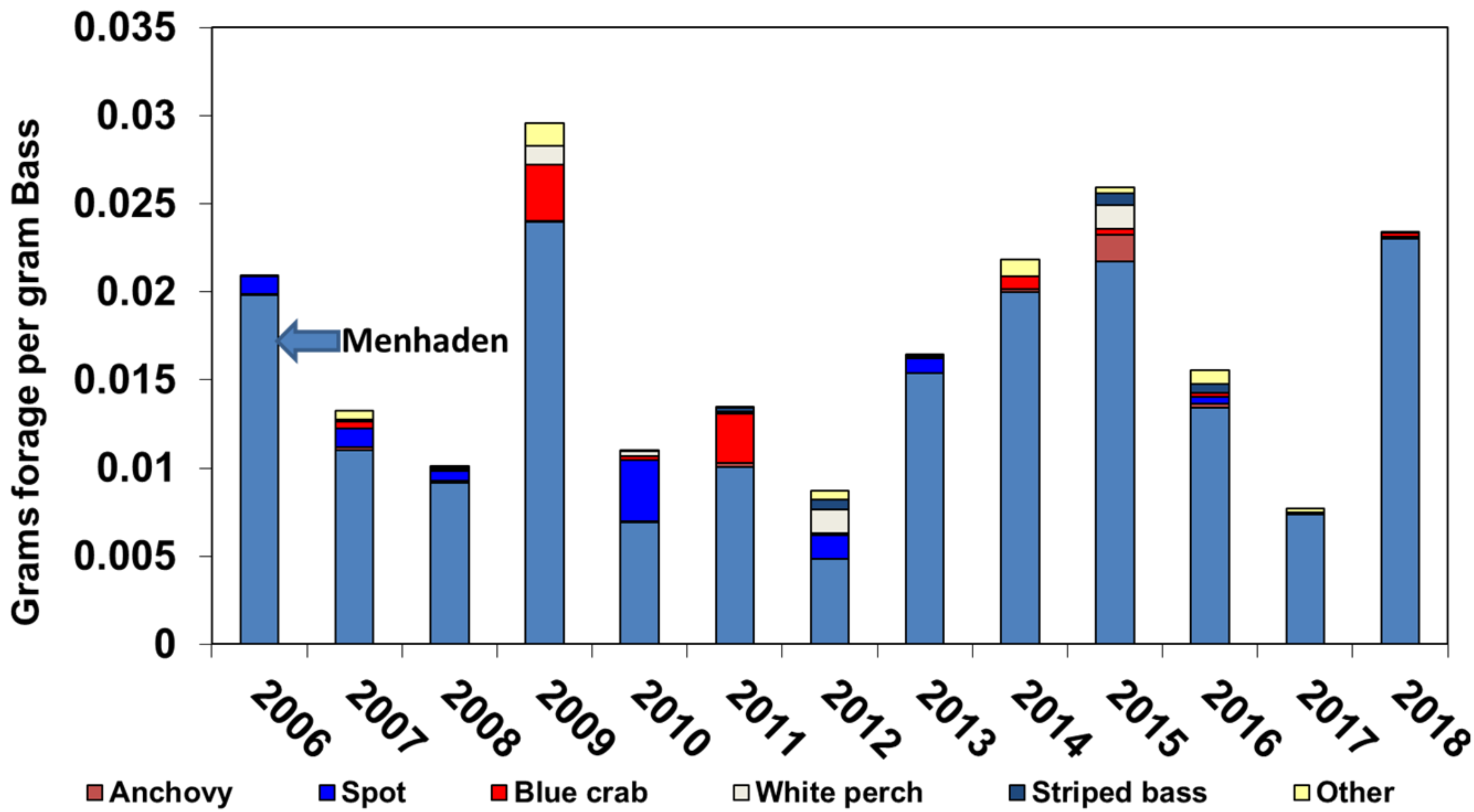

Figure 21. Proportion of small Striped Bass guts without food (PE) in fall and its 90% confidence interval. Red diamond represents threshold PE and green diamond indicates the PE target.

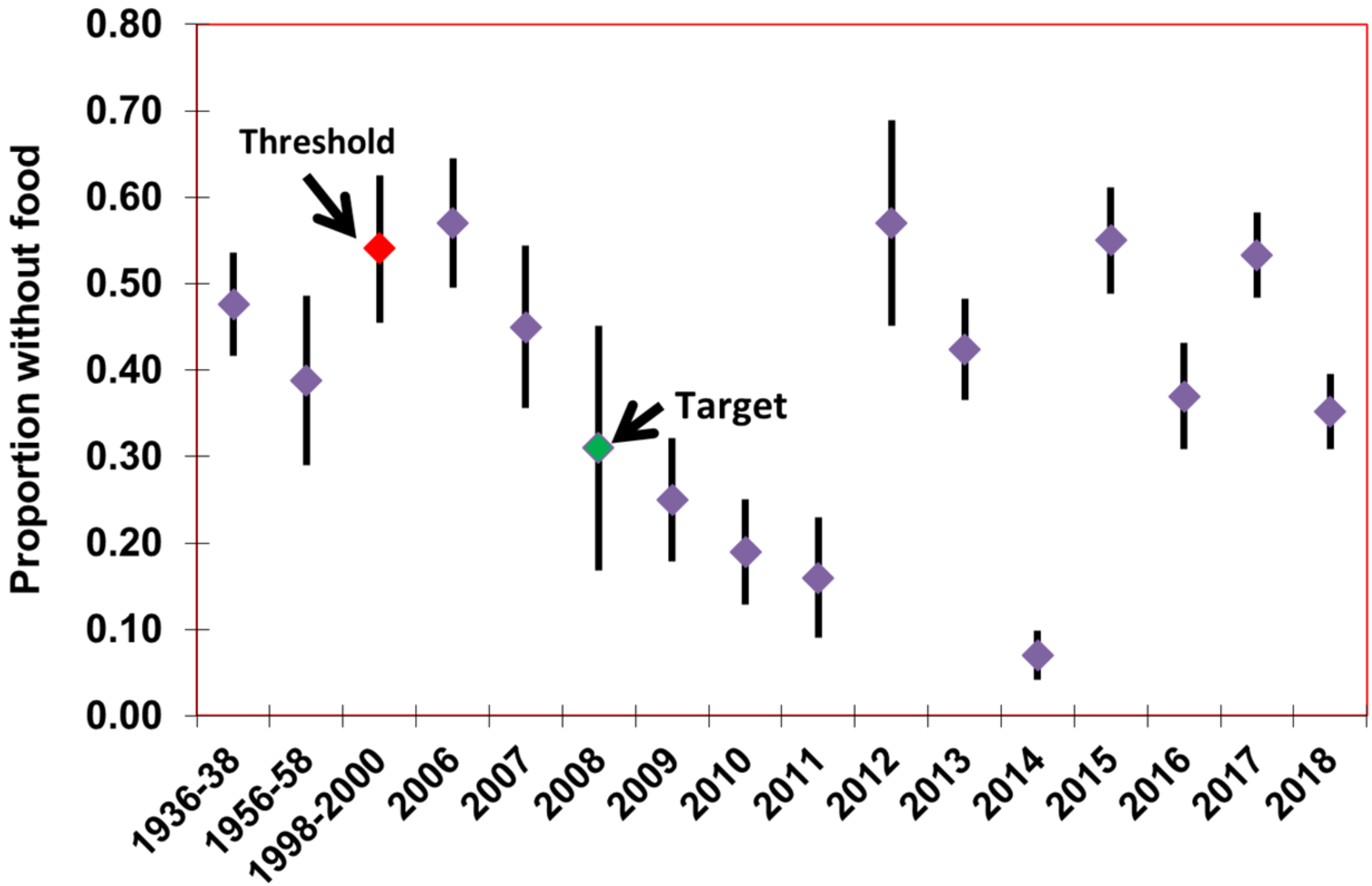

Figure 22. Proportion of large Striped Bass (> 456 mm or 18 in, TL) guts without food in fall.

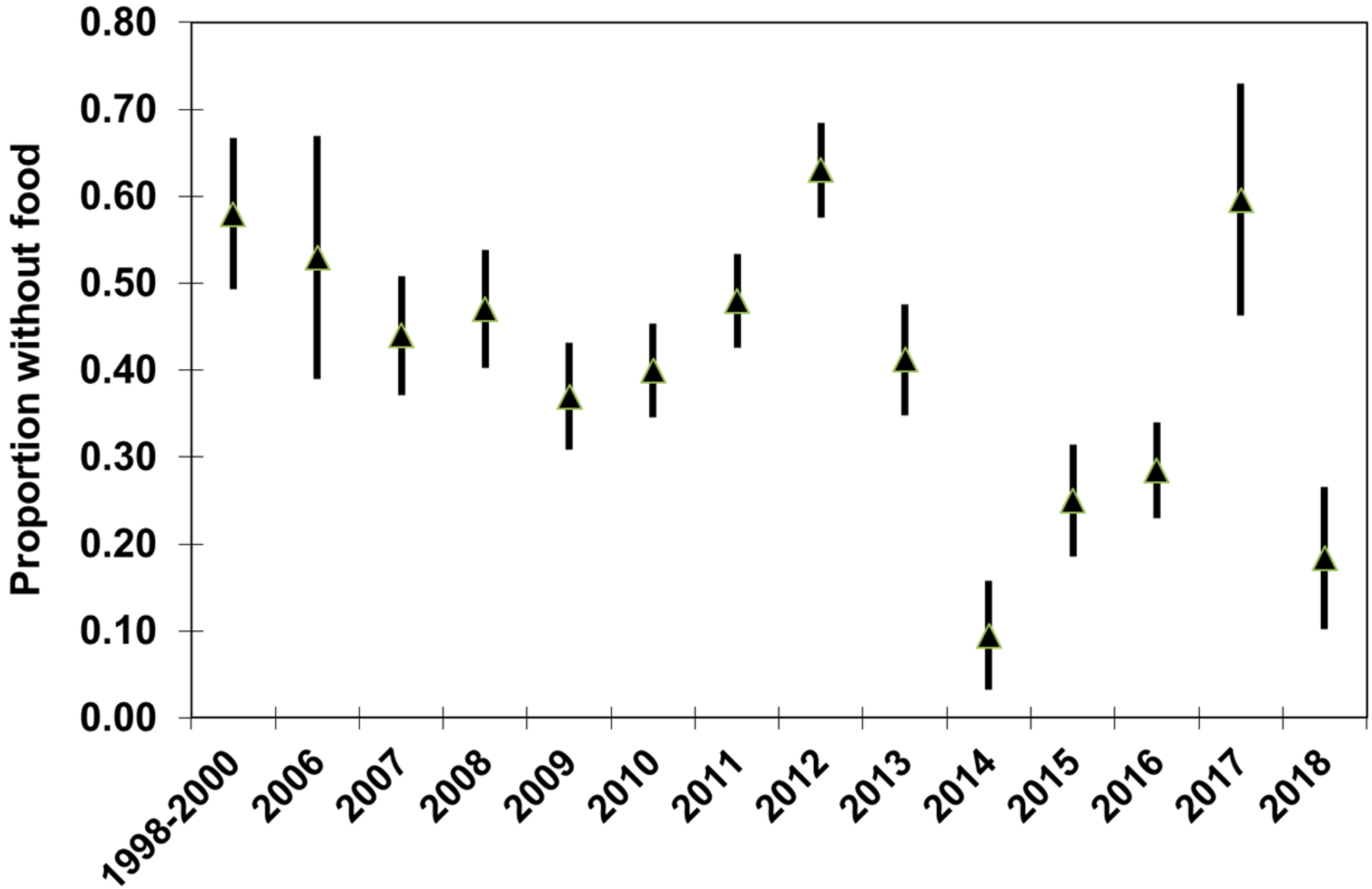

Figure 23. Median prey-predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for small (< 457 mm) and large Striped Bass. Optimum ratio was estimated by Overton et al. (2009).

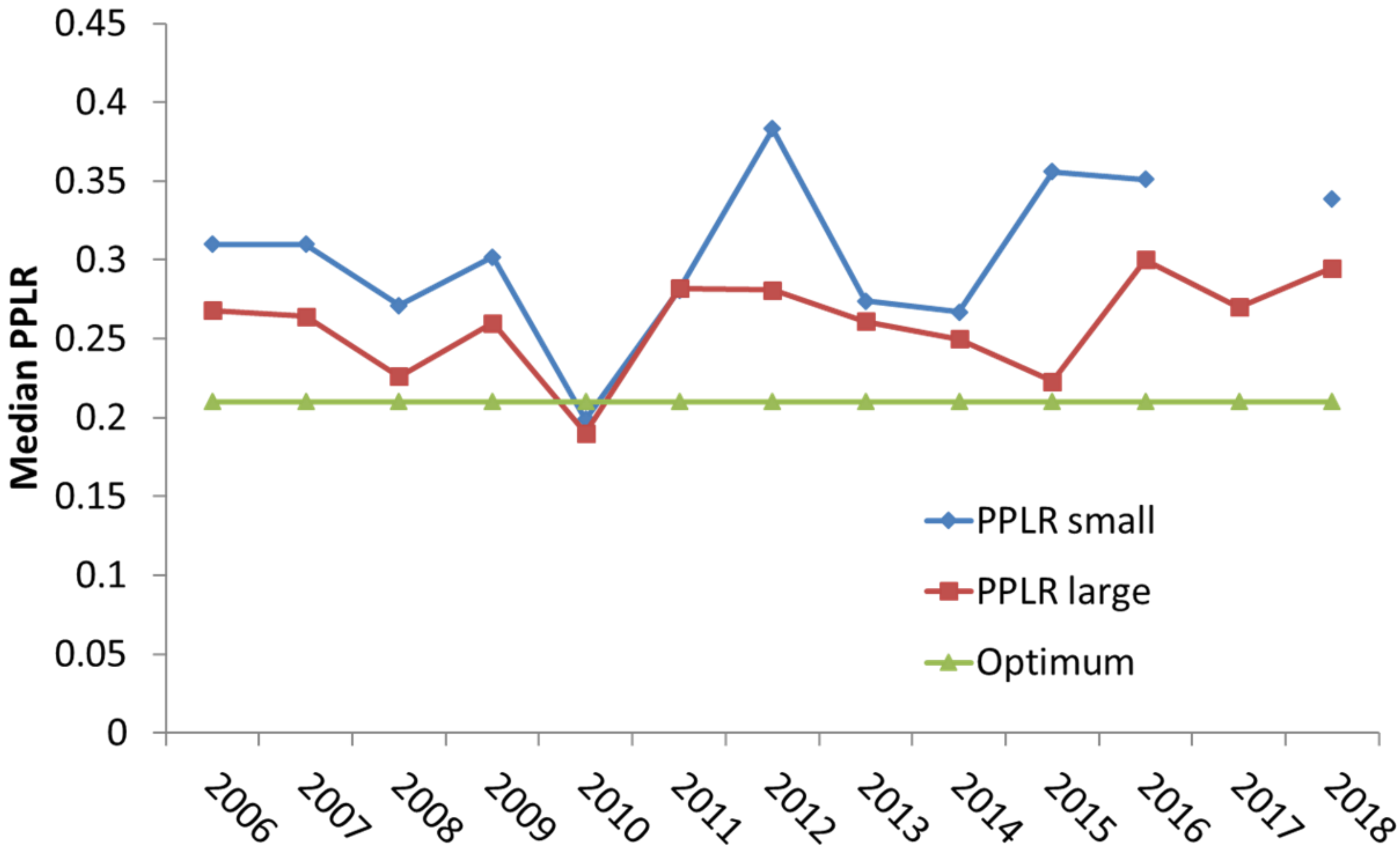

Figure 24. Scatter plot of the residuals of the regression of large Striped Bass PE with small Striped Bass PE against the ratio of prey length to Striped Bass length (PPLR). PE = proportion of sampled Striped Bass with empty guts.

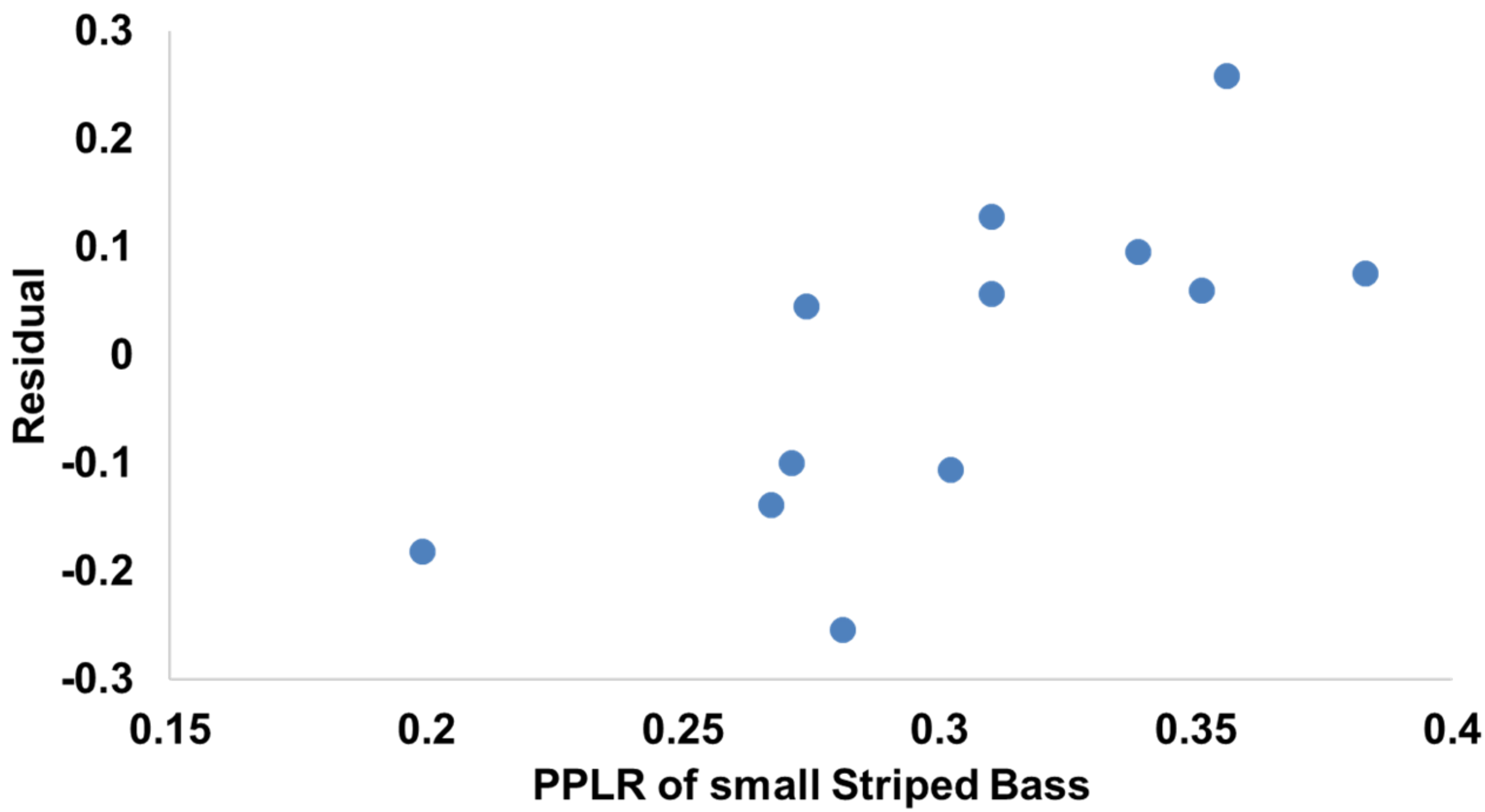

Figure 25. Time-series of age 3 male Striped Bass relative abundance on two major Maryland spawning areas (Age 3 CPUE; units = number of fish captured in 1000 square yards of net per hour) and abundance (N) of age 3 Striped Bass along the Atlantic Coast estimated by the ASMFC (2019) statistical catch-at-age model.

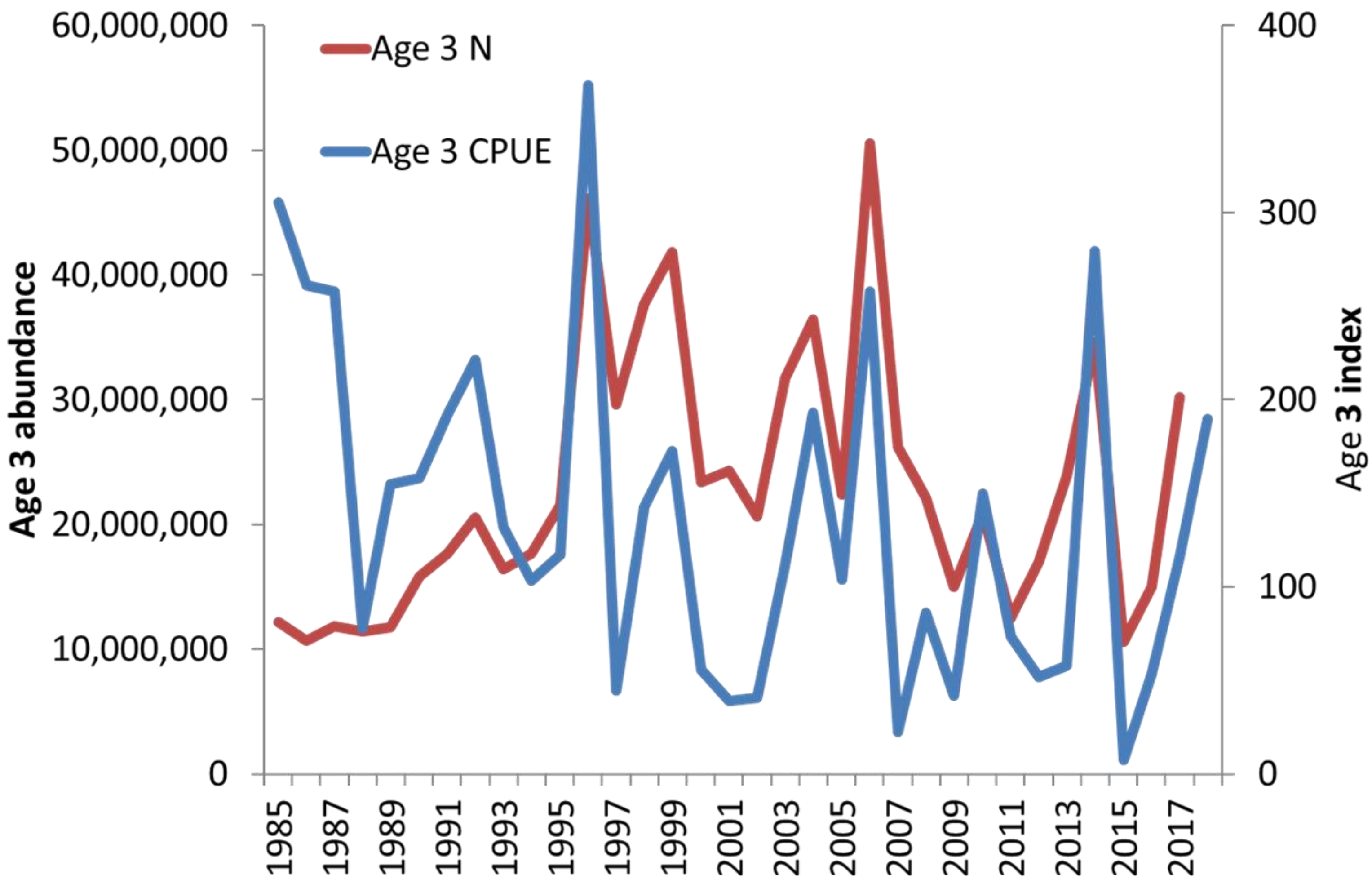

Figure 26. Trend in catchability of age 3 Striped Bass in Maryland's spring gill net survey.

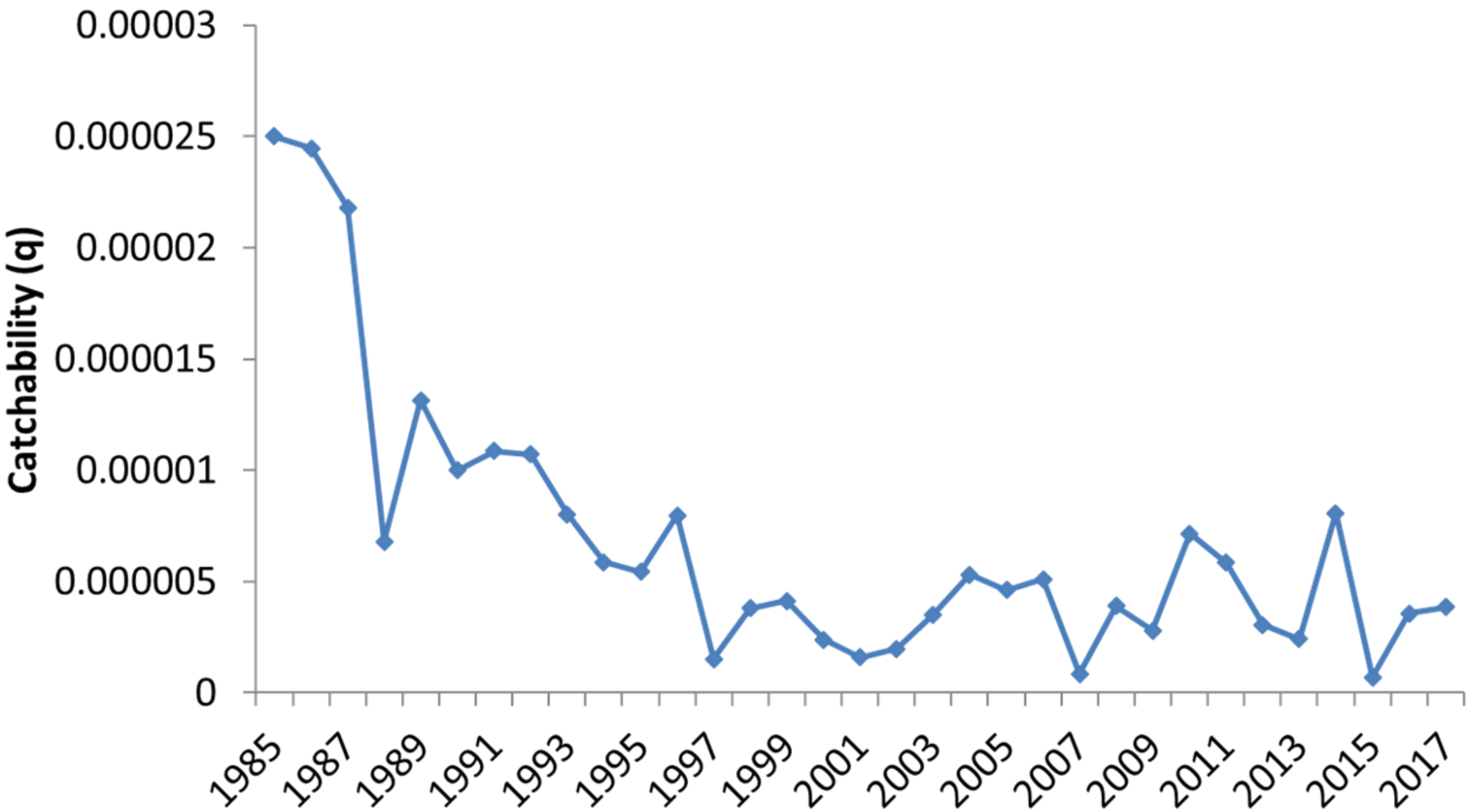

Figure 27. Time-series of age 3 Striped Bass relative abundance on two major Maryland spawning areas (Hybrid index = index adjusted for changing catchability during 1985-1995); units = number of fish captured in 1000 square yards of net per hour) and abundance (N) of age 3 Striped Bass along the Atlantic Coast estimated by the ASMFC (2019) statistical catch-at-age model.

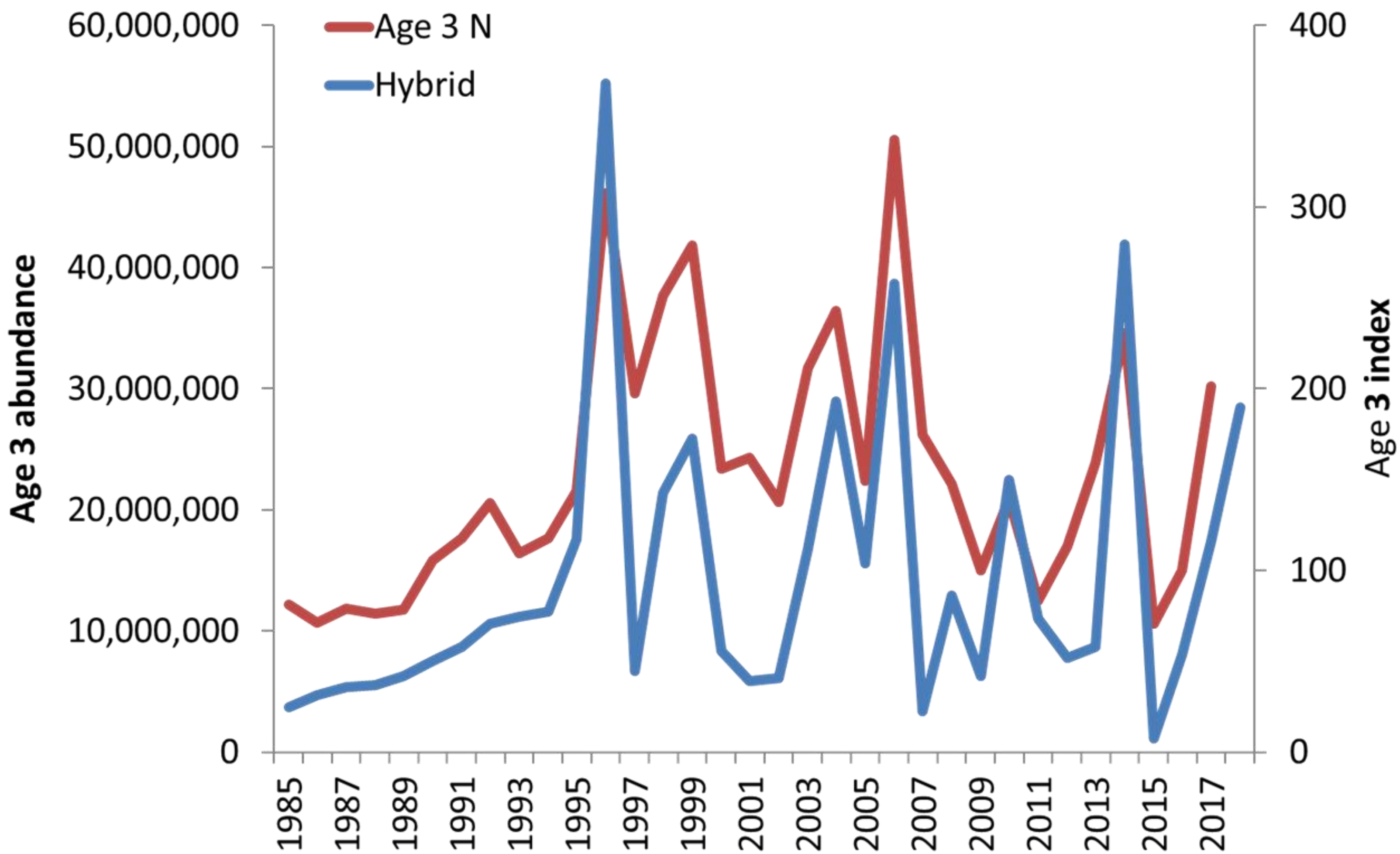

Figure 28. Relative survival (SR) of Striped Bass during 1985-2018 and 90% confidence intervals based on @Risk simulations of age 3 hybrid gill net indices divided by juvenile index (year -3) distributions.

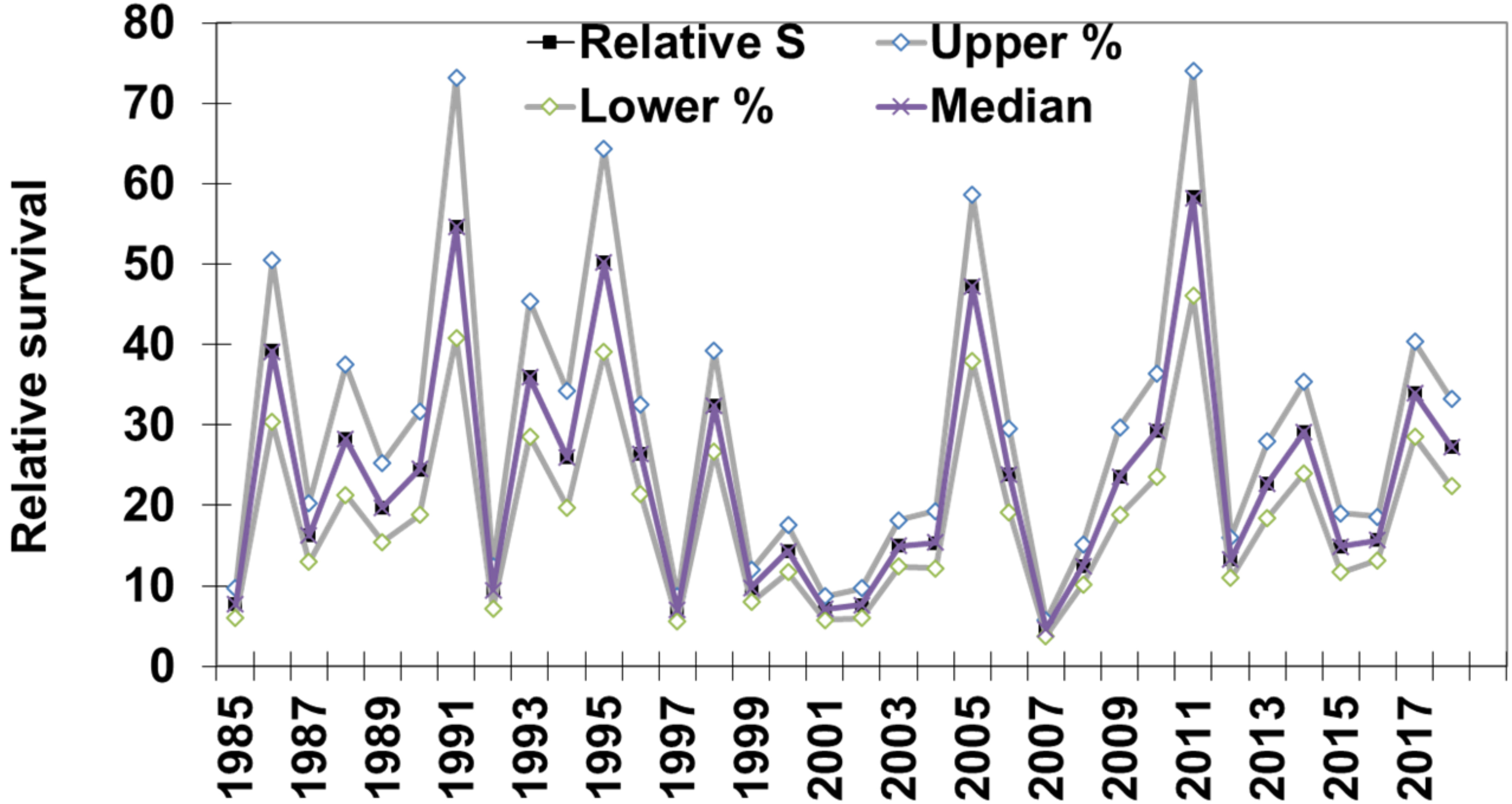

Figure 29. Relative survival of Striped Bass during 1985-2018 with targets and limits. Target = highest point of target P0 period (2008-2010). Threshold = highest point consistent with other points during threshold P0 period (1998-2004).

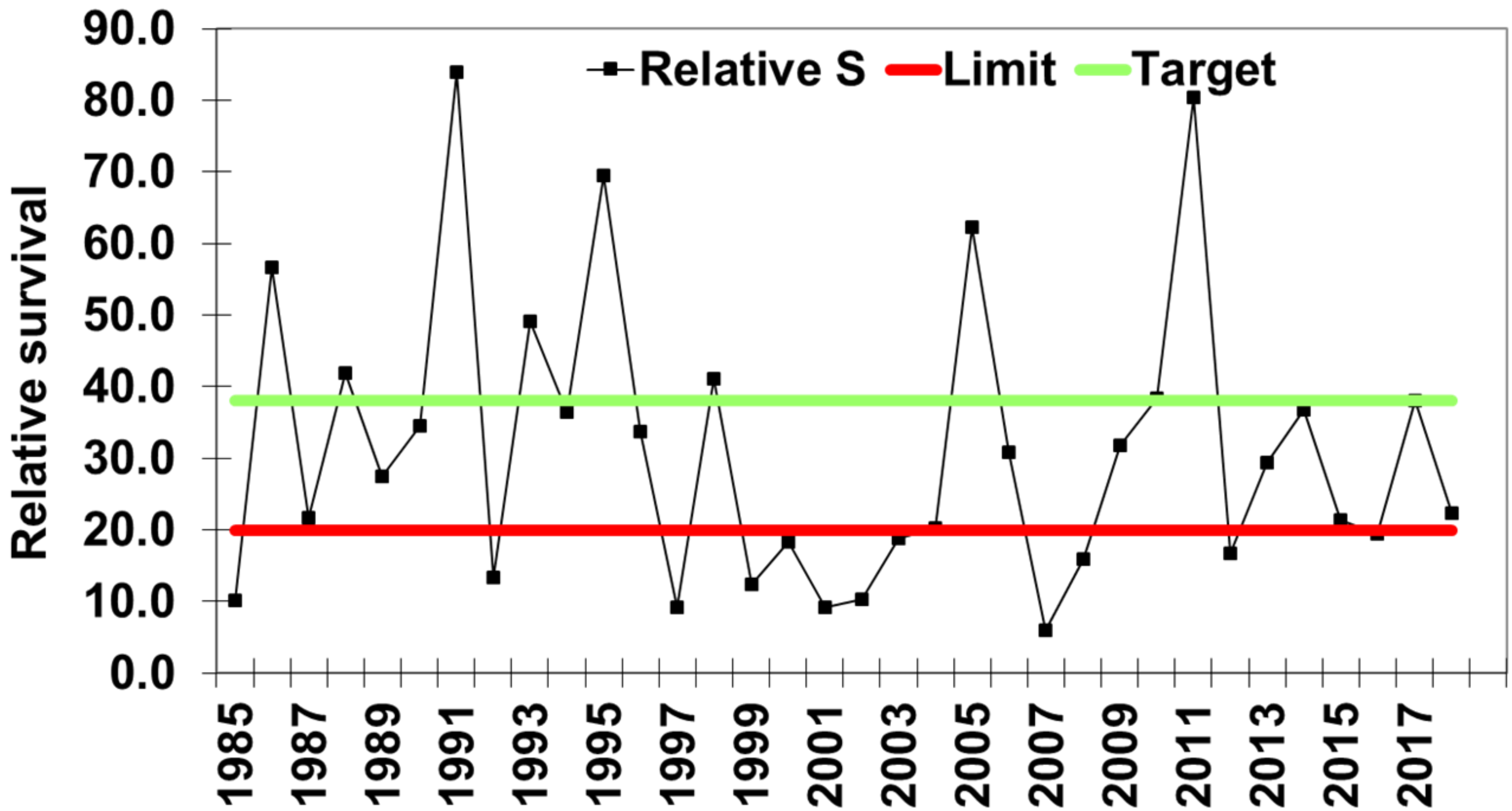

Figure 30. Index of Forage (IF) and its component scores. IF averages scores given to five indicators of forage status in upper Bay. A score of 3 indicates target conditions were met; 1 indicates threshold conditions; 2 indicates status in between. RI = index of relative abundance of resident Striped 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.

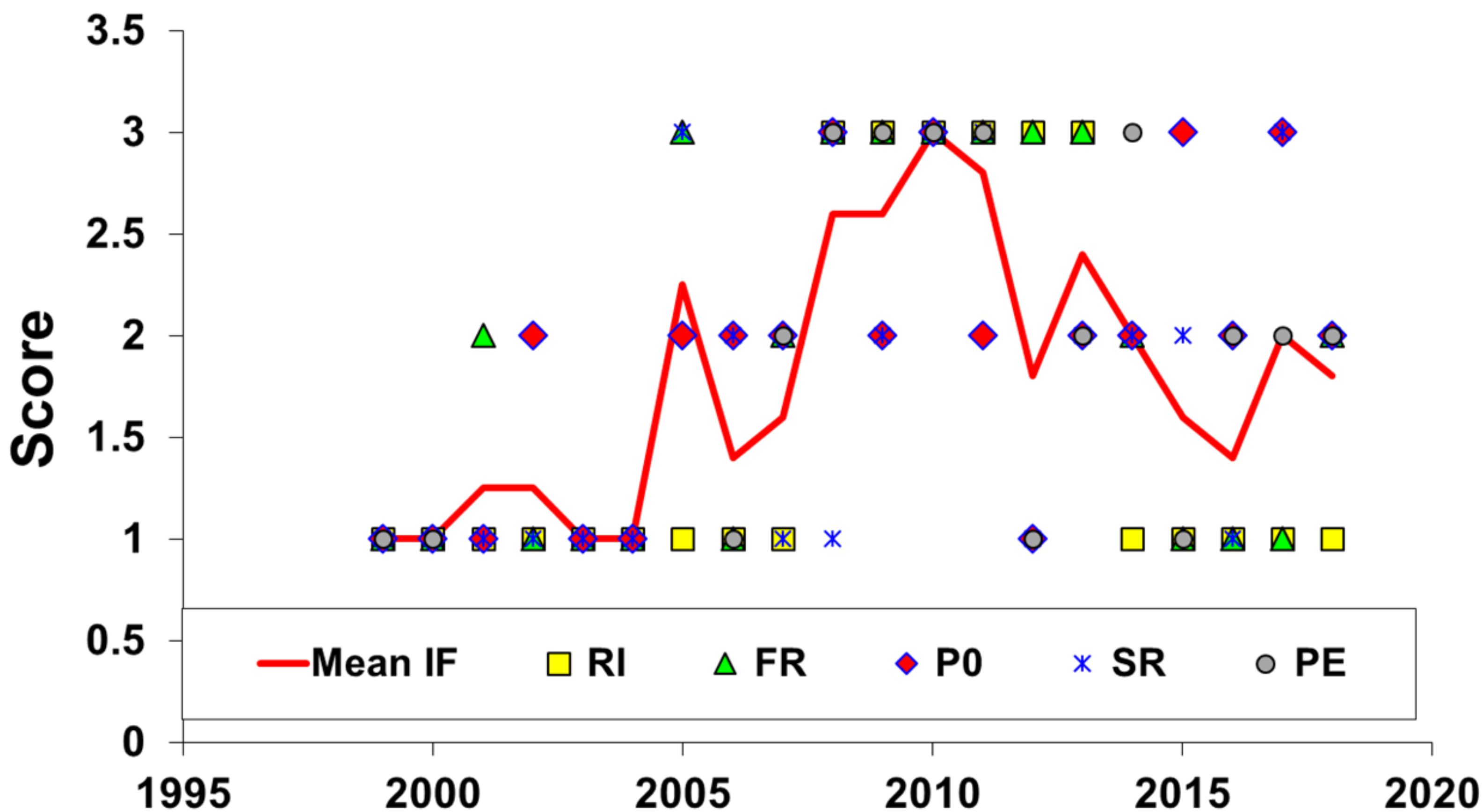

Figure 31. Forage index with all components averaged (Mean IF) and averaged with each component removed (leave one out average). Dashed lines indicate proposed IF target (at or above green dashed line) and threshold (at or below red dashed line) based on this approach. See Figure 30 for explanation of scores and abbreviations.