PERFORMANCE REPORT FOR FEDERAL AID GRANT F-63-R, SEGMENT 10

2019

MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

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

September, 2020

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Fishing and Boating Services
Chesapeake Finfish Program

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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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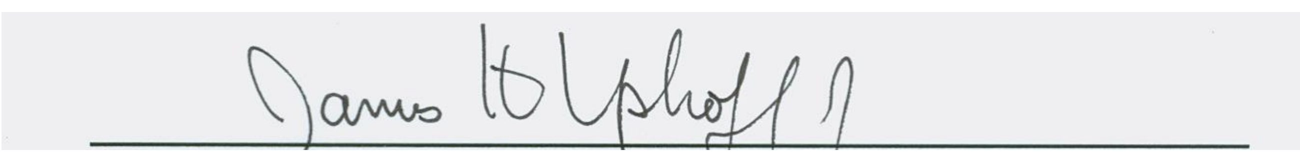

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Report Organization

This report was completed during November, 2020. It consists of summaries of activities for Jobs 1–4 under this grant cycle. All pages are numbered sequentially; there are no separate page numbering systems for each Job. Job 1 activities are reported in separate numbered sections. For example, Job 1, section 1 would cover development reference points (Job 1) for stream spawning habitat of anadromous fish (Section 1). Tables in Job 1 are numbered as section number – table number (1-1, 1-2, etc). Figures are numbered in the same fashion. Throughout the report, multiple references to past annual report analyses are referred to. The complete PDF versions of many past annual reports can be found under the Publications and Report link on the Fisheries Habitat and Ecosystem (FHEP) website page on the Maryland DNR website. The website address is http://dnr.maryland.gov/fisheries/Pages/FHEP/pubs.aspx . Table 1 provides the page number for each job and section.

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

Changes to Planned Activities due to Coronavirus

Advent of the coronavirus pandemic during early 2020 prevented continuing some activities during 2020 and required modification of others. Home offices allowed for analyses to continue, but not always with the full complement of resources that were previously available. We could not sample for Job 1 during spring, 2020. Activities impacted were sampling anadromous fish stream spawning habitat (Job 1, Section 1), and sampling of fresh-tidal estuarine habitat for presence-absence of Yellow Perch larvae and Striped Bass eggs during March-May, 2020 (Job 1, Section 2). An extensive analysis of Striped Bass egg and larval habitat using data from the 1950s through 2019 was added to Job 1, Section 2 for this report and this analysis should continue into the next report cycle. Sampling of summer habitat began in July under coronavirus prevention protocols. Activities under Job 2 were curtailed, but some were continued using virtual meetings and email exchanges. Job 3 was not impacted other than through difficulty in running GIS from home. Sampling of Striped Bass condition and diets for Job 4 is scheduled for fall 2020, but is dependent on whether the Fish and Wildlife Health Program will be able to sample during October-November.

<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 DOP, 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. Regression analyses that included spawning stock categories (0 for low during 2005-2011 and 1 for high during 2012-2019), indicated significant and logical relationships among P_{herr} and C/ha (R^2 = 0.74) or conductivity (R^2 = 0.69) consistent with the hypothesis that development was detrimental to stream spawning. Predicted P_{herr} declined by 51% over the range of observed C/ha (0.07-1.52); and increased by 58% between the two spawning stock categories. Predicted P_{herr} declined by 46% over the range of observed conductivity standardized to its baseline (1.14-2.19) and increased by 58% between the two spawning stock categories. The high spawning stock category in the analysis of 2005-2019 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. 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} ($\leq$ 0.4) were much more frequent beyond the C/ha threshold or

when standardized conductivity was 1.8-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, provided 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 targets and thresholds. Higher DO and pH measurements in urbanized large subestuaries sampled since 2015 (Patuxent and Wicomico rivers) during L_p surveys indicate their water quality dynamics were different from the rural, agricultural Choptank River watershed.

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. 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.

*Section 2.1: Striped Bass spawning and larval habitat status - W*e developed techniques for assessing changes in spawning and larval habitat in Maryland's Striped Bass spawning areas. Striped bass are sensitive to egg and larval habitat perturbations in the first three weeks of life in spring in limited fresh-low salinity tidal reaches of 16 Chesapeake Bay tributaries that serve as spawning areas and larval nurseries. Maryland has measured year-class success of Striped Bass in four major Chesapeake Bay nursery areas with a shore zone seine survey since 1957 (JI). A long time-series of the proportions of ichthyoplankton samples with eggs (*Ep*; 1955-2019), equivalent to that of the JI, provided a means of understanding the role of spawning stock status on recruitment. The ratio of JI to *Ep* was used as an indicator of relative survival of early life stages (RLS) for analyses searching for shifts in RLS through time. Trends in year-class success of Striped Bass were compared with White and Yellow Perch trends, semi-anadromous fish that share a common larval nursery with Striped Bass but have different life histories and fisheries. Comparisons of Striped Bass JI's among areas offered insight on regional similarity of habitat conditions. Since 2014, we have collected basic water quality data (temperature, conductivity, dissolved oxygen or DO, and pH). Historical water quality measurements were available from paper records, reports, or old electronic files, making retrospective comparisons possible.

Survival of Striped Bass eggs and larvae, and subsequent recruitment in Maryland's portion of Chesapeake Bay exhibited time blocks of varying productivity during 1957-2019. The near collapse of the Striped Bass fishery in the 1980s was driven by a shift to low JIs in the early1970s that was followed by a decline in baywide *Ep* a decade later. Baywide *Ep* increased during 1955-1957, was high during 1961- 1979, low during 1982-1988, and recovered to 1961-1979 levels after 1988. Year-classes in the top quartile occurred frequently during 1958-1970 (31% of indices) and 1993-2019 (41% of indices). Juvenile indices between these periods were not present in the top quartile and year-classes in the bottom quartile were much more likely to occur. Recovery of Striped Bass spawning stock, indicated by high *Ep* after 1988, was accompanied and complemented by a recovery of egg-larval survival, indicated by RLS, a few years later. Estimates of high RLS have occurred every few years since 1993, with the exception of 2006-2010. Estimates of RLS indicated periods of fairly consistent higher or lower survival rather than random scattering throughout the time-series indicative of stationary influences on recruitment. Estimates of RLS in the bottom quartile were concentrated in the period spanning 1977-1991, while periods of RLS in the upper quartile occurred during 1961-1970 and 1993-2019. Use of *Ep* or spawning stock biomass estimates (1982-2017 available) from the current Striped Bass stock assessment as the denominator for determining relative larval survival produced different depictions of egg-larval survival dynamics and patterns of underlying productivity.

Maryland's Striped Bass JI was well correlated with JIs of White Perch and Yellow Perch. These two estuarine resident species differed enough in life history characteristics and fisheries that they should not have been simultaneously overfished, indicating common larval habitat conditions played a large role in determining their year-class success. Associations among Striped Bass JIs in adjacent spawning areas (Choptank and Nanticoke rivers in eastern Maryland or Potomac and Patuxent rivers in southern Maryland) were moderate to strong, and correlations were weaker when spawning areas were not adjacent. Conditions among these major spawning and larval nurseries occasionally aligned, resulting in a strong Striped Bass JI.

Striped Bass egg presence-absence in three infrequently sampled spawning areas (Patuxent, Wicomico, and Chester rivers) between the 1950s and 2015-2019 did not indicate major changes in spawning stock status in these spawning areas.

Basic water quality data with adequate sample sizes at the spawning and larval nursery spatial and temporal scale were surprisingly sparse. Comparisons of flow, water temperature, conductivity, and pH indicated conditions within Maryland's Striped Bass spawning and larval nursery areas have changed over time, but changes were variable among areas. Water quality conditions differed between spawning areas in rural and urbanizing watersheds. Dissolved oxygen during spawning and larval periods did not fall below the 5 mg / L target for Chesapeake Bay living resources over all the years and spawning areas available.

Long-term (1950s to present), concurrently collected water temperature and egg distribution data suggested that water temperature (21˚C) indicative of the end of spawning and-or poor survival of hatched larvae was occurring earlier in recent years. The scattershot nature of sampling during the 1950s makes this finding tenuous, but we hope to be able to investigate this further through the extensive Nanticoke River time-series.

Choptank River pH offered the clearest indication of change between 1986-1991 and 2014-2019, from largely acidic and highly variable conditions to neutral and more stable (and closer to those cited for productive hatcheries). The more acidic conditions in Choptank River surveys during the 1980s were consistent with descriptions of water quality described for in situ and on-site toxicity tests conducted in Choptank and Nanticoke rivers during 1984-1990. Acidic conditions and poor buffering coupled with concurrent elevated metals were associated with low survival of Striped Bass prolarvae during some trials. Distributions of pH during the 1990s in Nanticoke, Patuxent and Chester rivers' spawning areas were generally in the upper range of those found in the Choptank River during 1986-1991 and exhibited wide variability. During 2014-2019, pH conditions in spawning areas with urbanizing watersheds (Patuxent and Wicomico rivers) generally exhibited higher means and greater variation in measurements than rural watersheds (Choptank and Chester rivers). Patuxent River pH means and ranges appeared to change little between 1991 and 2015, while pH means increased and range contracted in Chester River between 1996 and 2019.

Conductivity distributions in spawning areas with urban watersheds exhibited higher minimums than spawning areas in rural watershed during 2014-2019. Minimum conductivity in the Patuxent River spawning area increased by a factor of 2.2-2.4 between 1991 and 2015-2016. Wicomico River minimum conductivity was 1.4-2.3 times higher than nearby Choptank River or Nanticoke River. Choptank River spawning area conductivity summaries offered little indication of change between 1986-1991 and 2014-2019. Minimum conductivity in Chester River was about 40% higher in 2019 than in 1996. Elevated salt levels by themselves in the upper spawning area should not be an issue for Striped Bass since they can be abundant in higher conductivity regions further downstream where freshwater is more mixed with intruding saltwater. However, elevated conductivity could indicate other stressors have increased as well.

What may have triggered periods of enhanced or depressed larval survival? Long-term climate patterns, long-term climate warming, deterioration and improvement in acidic deposition, concurrent increases in freshwater salinization and alkalinization (salinization syndrome), increasing addition of a suburbs to the Chesapeake Bay watershed, a shift to conservation agriculture, and watershed management practices associated with the Chesapeake Bay Program could result in detrimental or beneficial larval habitat changes. It is likely that combinations of these factors have shifted from period to period.

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 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 2018-2019, 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.

The effects of high precipitation in 2018 did not have a lingering impact on survey water quality measurements during 2019. Salinities in subestuaries sampled either increased or remained within bounds of what had been observed previously, remaining in their salinity class. Chester River has shown short-term improvement, although that could reflect it shifting to oligohaline; salinity increased during 2019, but remained oligohaline instead of returning to mesohaline. 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 2019 was most likely to be above the target level and measurements below the threshold were uncommon in Chester River and its two tributaries. Corsica River, one tributary to the Chester River, had a noticeable improvement in bottom DO during 2018-2019 compared to earlier years sampled; the increase may reflect the State's designation as a targeted restoration watershed in 2005 which provided additional funding for several restoration programs to occur, as well as an upgrade to the wastewater treatment plant that occurred in 2010. Most bottom DO measurements in Wye River fell between the target and threshold level, below threshold readings decreased slightly in 2019. Station 1 (upper site) in the Wye River during 2018 and 2019 showed substantially lower bottom DO readings than previous years, possibly due to increased precipitation that would increase run-off of nutrients and organic matter. We noted an increase in leaf litter in seine and trawl samples during the summer of 2018 and decomposition of this organic matter may have increased oxygen demand. Frequency of below threshold bottom DO continued to increase in 2019 in Tred Avon River (this watershed is approaching the development threshold) and below target DO became more frequent. Other water quality metrics (pH and Secchi depth) in the subestuaries sampled during 2019 were within previous years' ranges. Finfish catches in trawls sampling bottom water habitat remained steady or slightly increased among all subestuaries sampled. Species composition changed slightly, reflecting the reappearance of Bay Anchovy throughout most of the subestuaries sampled after largely disappearing in 2018. Spot, Channel Catfish, White Catfish, and Brown Bullhead also increased. Inshore seine catches were within a normal range. Modified proportional stock densities for trawl and seine samples for subestuaries sampled in 2019 indicated that mid-Bay subestuaries, Tred Avon River and Wye River, have greater population densities of White Perch of interest to anglers compared to the White Perch communities in upper-Bay subestuaries, Chester River, Corsica River, and Langford Creek. While it appears that heavy rainfall and high freshwater discharge into the Chesapeake Bay and its tributaries during 2018 may have slightly impacted the upper- and mid-Bay subestuaries with lower salinities, lower DO, and smaller finfish catches (GMs for 2018-2019 were among the lowest of the time-series for a majority of the subestuaries sampled), the effects of very wet conditions in 2018 caused quick changes that lingered during 2019. Overall, we saw increases in water quality parameters and increased finfish catches with increased species composition.

Common Background for Job 1, Sections 1-3

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

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

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

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

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

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

Impact of development on estuarine systems has not been well documented, but measurable adverse changes in physical and chemical characteristics and living resources have occurred at IS of 10-30% (Mallin et al. 2000; Holland et al. 2004; Uphoff et al. 2011). Habitat reference points based on IS have been developed (ISRPs) for Chesapeake Bay estuarine watersheds (Uphoff et al. 2011). They provide a quantitative basis for managing fisheries in increasingly urbanizing Chesapeake Bay watersheds and enhance communication of limits of fisheries resources to withstand development-related habitat changes to fishers, land-use planners, watershed-based advocacy groups, developers, and elected officials (Uphoff et al. 2011; Interagency Mattawoman Ecosystem Management Task Force 2012). These guidelines have held for Herring stream spawning, Yellow Perch larval habitat (they are incorporated into the current draft of Maryland's tidal Yellow Perch management plan), and summer habitat in tidal-fresh subestuaries (Uphoff et al. 2015). Preserving watersheds at or below 5% IS would be a viable fisheries management strategy. Increasingly stringent fishery regulation might compensate for habitat stress as IS increases from 5 to 10%. Above a 10% IS threshold, habitat stress mounts and successful management by harvest adjustments alone becomes unlikely (Uphoff et al. 2011; Interagency Mattawoman Ecosystem Management Task Force 2012; Uphoff et al. 2015). We have estimated that impervious surface in Maryland's portion of the Chesapeake Bay watershed will exceed 10% by 2020; a preliminary estimate of IS in 2018 equaled 9.3%. We expect adverse habitat conditions for important forage and gamefish to worsen with future growth. Managing this growth with an eye towards conserving fish habitat is important to the future of sportfishing in Maryland.

We now consider tax map derived development indices as the best source for standardized, readily updated, and accessible watershed development indicators in Maryland and have development targets and thresholds based on it that are the same as ISRPs (Uphoff et al. 2015; Topolski 2015). Counts of structures per hectare (C/ha) had strong relationships with IS in years when all were estimated (1999-2000; Uphoff et al. 2015). Tax map data can be used as the basis for estimating target and threshold levels of development in Maryland and these estimates can be converted to IS. Estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.37, 0.86, and 1.35 C/ha, respectively. Tax map data provide a development time-series that goes back to 1950, making retrospective analyses possible (Uphoff et al. 2015).

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

Agricultural pesticides and fertilizers were thought to be potential sources of toxic metals implicated in some episodic mortality of Striped Bass larvae in Bay spawning tributaries in the early 1980s (Uphoff 1989; 1992; Richards and Rago 1999; Uphoff 2008). A correlation analysis of Choptank River watershed agricultural best management practices (BMPs) and estimates of postlarval survival during 1980-1990 indicated that as many as four BMPs were positively associated with survival (Uphoff 2008). Two measures that accounted for the greatest acreage, conservation tillage and cover crops, were strongly associated with increased postlarval survival

(r = 0.88 and r = 0.80, respectively). These correlations cannot explain whether toxicity was lowered by BMPs, but it is possible that reduced contaminant runoff was a positive byproduct of agricultural BMPs aimed at reducing nutrients (Uphoff 2008).

Agriculturally derived nutrients have been identified as the primary driver of hypoxia and anoxia in the mainstem Chesapeake Bay (or Bay; Hagy et al. 2004; Kemp et al. 2005; Fisher et al. 2006; Brush 2009). Hypoxia is also associated with transition from rural to suburban landscapes in brackish Chesapeake Bay subestuaries (Uphoff et al. 2011). Hypoxia's greatest impact on gamefish habitat occurs during summer when its extent is greatest, but hypoxic conditions are present at lesser levels during spring and fall (Hagy et al. 2004; Costantini et al. 2008). Episodic hypoxia may elevate catch rates in various types of fishing gears by concentrating fish at the edges of 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; Smith 2020). 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 $r^2 \leq 0.25$; and moderate relationships fell in between. 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, rejection of the model was considered. 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).

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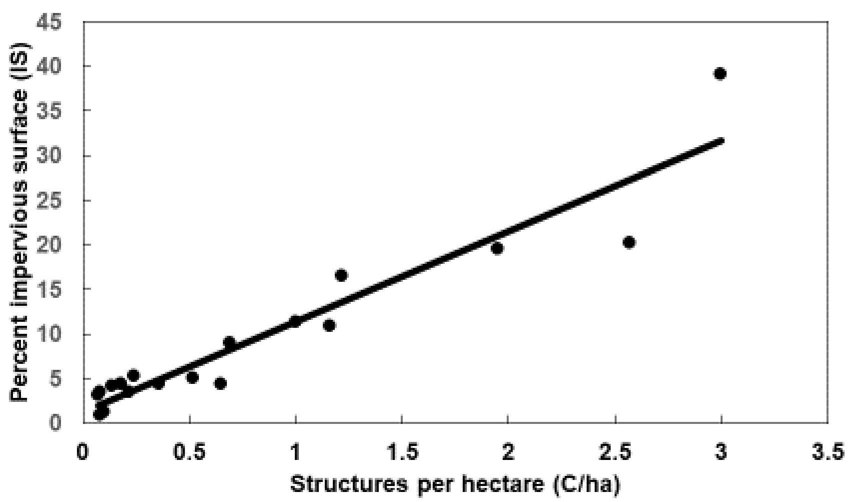
Table 1. Structures per hectare (C/ha) and percent impervious surface estimates (IS) used to estimate the relationship for predicting IS from C/ha. Old C/ha were estimates used previous to this report and New C/ha were revised estimates used to estimate the current relationship.

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

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Introduction

Urbanization associated with increased population growth became a factor in the decline of diadromous fishes in the late 20th century (Limburg and Waldman 2009). 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), Patapsco River (2013-2017) and Chester River (2019; 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 eastern shore tributary of the Chesapeake Bay within the Coastal Plain and has a watershed dominated by agriculture. The Patapsco River watershed is located within Coastal Plain and Piedmont 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). Fluvial Patapsco River meets the Chesapeake Bay and forms the port of Baltimore. The Chester River, located on the eastern shore, is a fluvial-tidal system located in the Coastal Plain. Agriculture is predominant in its watershed (O'Dell et al. 1975; Table 1-1; Figure 1-1).

We developed two indicators of anadromous fish spawning in a watershed based on presence-absence of eggs and larvae: occurrence at a site (a spatial indicator) and proportion of samples with eggs and larvae (a spatial and temporal indicator). Occurrence of eggs or larvae of an anadromous fish group (White Perch, Yellow Perch, or Herring) at a site recreated the indicator developed by O'Dell et al. (1975; 1980). This spatial indicator was compared to the extent of development in the watershed (counts of structures per hectare or C/ha; 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; Moore et al. 2017; Bird et al. 2018).

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 (or other large scale factors such as decreased predation or increased at-sea survival due to improved feeding and-or environmental conditions) 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 2005-2019 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 2005-2019, with the exception of Deer Creek during 2012-2015, Choptank River and Tuckahoe Creek during 2016-2017, Patapsco River during 2013-2017, and Chester River during 2019 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. Chester River (CH19001-CH19016; Figure 1-9) was sampled during 2019 to provide up-to-date information for the Queen Anne County comprehensive growth plan. Table 1-2 summarizes sites, dates, and sample sizes in Mattawoman, Piscataway, Deer, and Tuckahoe Creeks, and Bush, Choptank, Patapsco, and Chester Rivers during 2005-2019.

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 with 10% buffered formalin after collection by MD DNR staff, or were placed in a cooler with ice for transport and preserved 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 in a solution of 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 in a solution of 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.

Watershed area draining into the Herring spawning areas (hereafter, watershed), land use, and C/ha in those Herring spawning areas were estimated. Mattawoman Creek's watershed was 24,430 ha and estimated C/ha increased from 0.87 to 0.97 during 2008-2018; 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; and the Chester River drainage (watershed area = 77,751 and developmental level = 0.13 C/ha) was sampled in 2019 (Table 1-1; Figure 1-1). These three systems are all spawning streams with high agricultural influence and low watershed development. Deer Creek, Choptank River and Tuckahoe Creek, and Chester River collections were made by MD 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 1.15 in 2017. 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 were collected for each date and stream site (mainstem and tributaries) during 2008-2018 from Mattawoman Creek, but only 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. 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, in the Patapsco River in 2013-2017, and in the Chester River drainage in 2019.

Presence of eggs and-or larvae of White Perch, Yellow Perch, and Herring at each station was compared to past surveys to determine which sites still supported spawning. The only exception was Mattawoman Creek in 2018 when only presence of Herring eggs and-or larvae was determined. 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), Tuckahoe Creek in 1976-77 (O'Dell et al. 1980), and Chester River in 1975-1977 (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), Patapsco River (2013-2017), and Chester River drainage (2019). Counts of Herring eggs and larvae were available for Mattawoman in 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). All sites sampled within the Chester River drainage correspond to sites sampled by O'Dell et al. (1980; Figure 1-9).

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, Patapsco, and Chester Rivers. Thirty-six sets of estimates of C/ha, percent agriculture, percent forest, and P_{herr} were available (1991 estimates for Mattawoman Creek could be included), while 35 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 (both had two parameters) model was considered the best description if a moderate or strong relationship was suggested, 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 for physiographic province 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 (Morgan et al. 2012). 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 2005-2019 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 lower spawning stock prior to the full implementation of river closures and bycatch reductions (2005-2011) and 1 indicating higher spawning stock following these measures (2012-2019). 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 and Chester Rivers, 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 that time (Figure 1-10). Surveys conducted by O'Dell et al. (1975, 1980) in the 1970s, sampled largely rural watersheds (C/ha < 0.28) 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 (~1.36 C/ha) than Mattawoman Creek (~0.81 C/ha) and Patapsco River (~1.02 C/ha). Deer Creek (zoned for agriculture and preservation) and the Choptank and Chester River drainages (predominantly agricultural) remained rural through 2019 (0.24, 0.18, and 0.13 C/ha, respectively; Figure 1-10).

Conductivities were usually elevated beyond background levels in all streams studied during 2008-2019 and median conductivities ranged from 1.14- to 2.4-times expected background levels (Table 1-3). 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 (1.39- to 1.69-times), Tuckahoe Creek (~1.46-times), and Chester River (1.66-times) were similarly elevated, even though Tuckahoe Creek and Chester River were much more rural (Table 1-3).

Herring spawning was detected at all mainstem stations sampled in Mattawoman Creek (MC1-MC4) during 1971 and 1991 (Table 1-4). 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 only been detected at MC1 in all surveys conducted since 1971, with the exceptions of 2009 and 2012 when spawning was not detected (Table 1-4). Presence of White Perch and Yellow Perch spawning in Mattawoman Creek was not determined in 2018 due to time and staffing limitations.

Herring spawning was detected at all mainstem sites in Piscataway Creek in 2012-2014 (Table 1-5). 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 1-5).

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 1-6). Herring eggs and larvae were present at three to five stations (not necessarily the same ones) in any given year sampled. There were far less occurrences of White and Yellow Perch eggs and larvae during 2005-2008 than 1973 and 2014 (Table 1-6).

O’Dell et al. (1975) reported that Herring, White Perch, and Yellow Perch spawned in Deer Creek during 1972 (Table 1-7). Three sites were sampled during 1972 in Deer Creek and one of these sites was located upstream of an impassable dam near Darlington (a fish passage was installed there in 1999). During 1972, Herring spawning was detected at both sites below the dam (SU01 and SU03), while White and Yellow Perch spawning were detected at the mouth (SU01). During 2012-2015, Herring spawning was detected at all sites sampled in each year. White Perch spawning was not detected in Deer Creek in 2012 but was detected at three sites each in 2013 and 2014, and two sites in 2015. Yellow Perch spawning detection has been intermittent; evidence of spawning was absent in 2013 and 2015, while spawning was detected at two and three sites in 2012 and 2014, respectively (Table 1-7).

While the Choptank River itself had not been sampled prior to 2016 (Table 1-8), O’Dell et al. (1980) reported Herring, White Perch, and Yellow Perch spawned in its drainage (Tuckahoe Creek) during 1976-1977 (Table 1-9). 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 1-9). 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 1-9).

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 1-10). 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 1-10).

Sites sampled in 2019 in the Chester River drainage match a subset of those sampled from 1975-1977 by O’Dell et al. (1980). Herring spawning was detected at a larger number of sites in 2019 than historically, while White Perch spawning was detected at roughly the same number of sites, although locations differed, and Yellow Perch spawning detection decreased (Figure 1-9; Table 1-11).

The 90% confidence intervals of P_{herr} (Figure 1-11) 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), Choptank River (2016-2017), and Chester River (2019) are the least developed watersheds and were characterized by the highest level of Herring spawning (level 3) in all years sampled (Figure 1-11).

Estimates of P_{herr} increased in Bush River, and Mattawoman and Piscataway creeks during 2005-2018 (Figure 1-12). 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 (P_{herr} as high as 0.78), 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 (to P_{herr} as high as 0.47; Figure 1-12). Remaining systems were sampled after 2011. Estimates of P_{herr} in Choptank and Chester rivers, and Deer and Tuckahoe creeks were high and steady through 2019 (0.62 to 0.87), while estimates for Patapsco River were lower and more variable (Figure 1-12).

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 = 0.34, P = 0.0002, N = 35; Table 1-12; Figure 1-13). Estimates of P_{herr} were linearly, moderately, and negatively related to C/ha (r^2 = 0.52, P < 0.0001, N = 36; Figure 1-14). Negative linear and curvilinear (power function) regressions similarly described weak relationships of P_{herr} and standardized median conductivity (r^2 = 0.20, P = 0.0064; or approximate r^2 = 0.18, P < 0.0001, respectively), with linear regression explaining only slightly more variability (N = 35; Figure 1-14). Low estimates of P_{herr} ($\leq$ 0.4) were much more frequent beyond the C/ha threshold (0.86 C/ha) or when standardized conductivity was 1.8-times or more than the baseline level (Figure 1-14). Estimates of P_{herr} were consistently above 0.6 in the four watersheds dominated by agriculture (Deer Creek, Tuckahoe Creek, Choptank River, and Chester River; Figure 1-14). 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 1971 estimate of P_{herr} was above 0.6 and was consistent with watersheds dominated by agriculture. Remaining estimates for Mattawoman Creek were represented by 53.9% forest cover with C/ha increasing from 0.87 in 2008 to 0.97 in 2018. Samples were not collected in Mattawoman Creek in 2019, but it is the system with the longest data set. Additional analyses have been performed on these data in previous years; see Uphoff et al. 2019 for more information.

Plots of residuals against year for linear regressions of C/ha or standardized conductivity and P_{herr} indicated an increasing trend (Figure 1-15); 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 = 0.52, SE = 0.08) and both coefficients (C/ha slope = -0.28, SE = 0.05; spawning stock slope = 0.30, SE = 0.06) were estimated with reasonable precision (CV < 20%). Predicted P_{herr} declined by 51% over the range of observed C/ha (0.07-1.52; Figure 1-16). Predicted P_{herr} increased by 58% between the two spawning stock categories (Table 1-13). Only the high spawning stock category contained estimates from the three land use types.

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 = 0.71, 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 46% over the range of observed standardized conductivity (1.14-2.19; Figure 1-16). Predicted P_{herr} increased by 58% between the two spawning stock categories (Table 1-14). Only the high spawning stock category contained estimates from all three land use types (Figure 1-16). Standardized median conductivities in excess of 1.75 were exclusively from watersheds categorized as urban. Higher conductivity (up to about 1.60) in agricultural and forested watersheds did not appear to be associated with distinctly lower P_{herr}; declines appeared concurrent with higher conductivity associated with urban development (Figure 1-16). 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 a spawning stock size time category (Figure 1-17). Linear regressions of residuals from the multiple regressions and year in Figure 1-17 indicated a slight increasing trend over time was possible for standardized conductivity ($r^2 = 0.11$, $P = 0.05$) but unlikely for C/ha ($r^2 = 0.03$, $P = 0.34$). 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 2012-2019, indicated significant and logical relationships among P_{herr} and C/ha consistent with the hypothesis that urbanization was detrimental to stream spawning. Predicted P_{herr} declined by 51% over the range of observed C/ha (0.07-1.52). 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. Higher standardized conductivity (up to about 1.60) in agricultural and forested watersheds did not appear to be associated with distinctly lower P_{herr}. Declines in P_{herr} appeared with higher conductivity in developing watersheds, suggesting that other urban stressors accompanied increasing conductivity. 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; Moore et al. 2017; Bird et al. 2018). Salt pollution and human-accelerated weathering have shifted the chemical composition of major ions in fresh water and increased salinization and alkalinization (freshwater salinization syndrome) across North America (Kaushal et al 2018). Coupled changes in conductivity, major ions, and pH began in the early and middle twentieth century and have influenced the water quality of most of the streams in the eastern United States. Densities of urban and agricultural land within a watershed can be strong predictors of base cations and pH in streams and rivers. In developed areas with colder climates, road salt is an important source of salinization. Agriculture can contribute significant bicarbonate and base cations from liming, potash, and fertilizer applications (Kaushal et al. 2018).

Uphoff et al. (2017) 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 we did not pursue statistical analyses with land uses other than C/ha. Our preference for using C/ha in analyses was two-fold: we have already done considerable work using C/ha, and C/ha provides a continuous rather than episodic time-series. However, we did note when these other land uses were predominant for particular P_{herr} outcomes. Estimates of P_{herr} were consistently high in 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 in Mattawoman Creek's watershed (our only forested watershed).

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 province effects 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 (0.46; Table 1-13) than for standardized conductivity (0.37; Table 1-14), possibly reflecting the wider coverage of stressors by C/ha.

Liess et al. (2016) developed a stress addition model for meta-analysis of toxicants that combined 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 standardized conductivity, with conductivity accounting for water quality and C/ha accounting for multiple stressors. This concept may also warn against 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 described 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 our 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. Maryland closed its Herring fisheries in 2011, and most other in-river fisheries along the Atlantic Coast were closed by 2012 (AFMFC 2019). Caps on Herring bycatch in Atlantic Herring and Atlantic Mackerel fisheries were also implemented in 2014 (MAFMC 2019), and estimates of P_{herr} in 2005-2019 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 (Bourdon and Jarzynski 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. 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). Urbanization affects the quality and quantity of organic matter (detritus) 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 phosphorus 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.

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 indicated 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 to the O’Dell et al. (1975; 1980) estimates 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 2005-2019 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 these 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		
Chester	2019	2010	0.13	65.9	24.8	77,751	Agriculture
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	0.94	9.3	53.9		
Mattawoman	2016	2010	0.95	9.3	53.9		
Mattawoman	2017	2010	0.96	9.3	53.9		
Mattawoman	2018	2010	0.97	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	1.13	24.4	30.4		
Patapsco	2016	2010	1.14	24.4	30.4		
Patapsco	2017	2010	1.15	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
Chester	2019	14	18-Mar	7-May	8	93
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 statistics of conductivity (µS/cm) for mainstem stations in Deer, Mattawoman, Piscataway, and Tuckahoe Creeks, and Bush, Chester, Choptank, and Patapsco Rivers during 2005-2019. 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	2019
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					
Chester															
Mean															175.8
Standard Error															4.0
Median															181.5
Kurtosis															-0.40
Skewness															-0.37
Range															164
Minimum															85
Maximum															249
Count															93
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 1-3 cont.

	Year														
Conductivity	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
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 1-3 cont.

	Year														
Conductivity	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2019
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 1-4. 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 1-5. 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 1-6. 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 1-7. 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 1-8. 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 1-9. Site-specific presence-absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Tuckahoe Creek during 1976-1977 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 1-10. 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 1-11. Site-specific presence-absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Chester River during 1975-1977 and 2019. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-9.

	Year	
Station	1975-77	2019
	Herring	
CH19001	0	1
CH19002	0	1
CH19003	1	1
CH19004	0	0
CH19005	1	1
CH19006	1	1
CH19007	0	1
CH19008	0	1
CH19009	1	1
CH19010	1	1
CH19011	1	1
CH19012	1	1
CH19014	1	1
CH19015	1	1
	White Perch	
CH19001	0	1
CH19002	0	1
CH19003	1	1
CH19004	1	0
CH19005	1	1
CH19006	1	1
CH19007	0	0
CH19008	0	0
CH19009	1	1
CH19010	1	1
CH19011	1	1
CH19012	1	0
CH19014	0	1
CH19015	1	1
	Yellow Perch	
CH19001	1	1
CH19002	1	0
CH19003	1	1
CH19004	0	0
CH19005	1	0
CH19006	1	0
CH19007	0	0
CH19008	0	0
CH19009	0	0
CH19010	0	0
CH19011	1	0
CH19012	0	0
CH19014	0	0
CH19015	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	1.4179	1.4179	17.31	0.0002	
Residual	33	2.70339	0.08192			
Total	34	4.12129				
$r^2 = 0.3440$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.20807	0.10374	11.64	<.0001	0.997	1.41914
C / ha	0.41969	0.10088	4.16	0.0002	0.21445	0.62493

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Structure density (C/ha)					
ANOVA	df	SS	MS	F	P	
Regression	1	1.30305	1.30305	36.66	<.0001	
Residual	34	1.20846	0.03554			
Total	35	2.51151				
$r^2 = 0.5188$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.84405	0.06685	12.63	<.0001	0.7082	0.9799
C / ha	-0.39792	0.06572	-6.05	<.0001	-0.53148	-0.26436

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Standardized conductivity					
ANOVA	df	SS	MS	F	P	
Regression	1	0.5045	0.5045	8.47	0.0064	
Residual	33	1.96454	0.05953			
Total	34	2.46904				
$r^2 = 0.2043$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.03726	0.19548	5.31	<.0001	0.63955	1.43497
Standardized conductivity	-0.34988	0.12019	-2.91	0.0064	-0.5944	-0.10535

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	1.81647	0.90824	44.54	<.0001	
Residual	32	0.65257	0.02039			
Total	34	2.46904				
$r^2 = 0.7357$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.52366	0.07992	6.55	<.0001	.	.
C / ha	-0.28455	0.05468	-5.2	<.0001	0.51114	0.45838
Time category	0.30267	0.05805	5.21	<.0001	0.45935	0.45935

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	1.71489	0.85744	36.38	<.0001	
Residual	32	0.75415	0.02357			
Total	34	2.46904				
$r^2 = 0.6946$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.71286	0.13106	5.44	<.0001	.	.
Standardized conductivity	-0.33089	0.07567	-4.37	0.0001	0.20433	0.37406
Time category	0.4119	0.05748	7.17	<.0001	0.61612	0.61612

Figure 1-1. Watersheds sampled for stream spawning anadromous fish eggs and larvae during 2005-2019. Coastal Plain and Piedmont Regions are indicated.

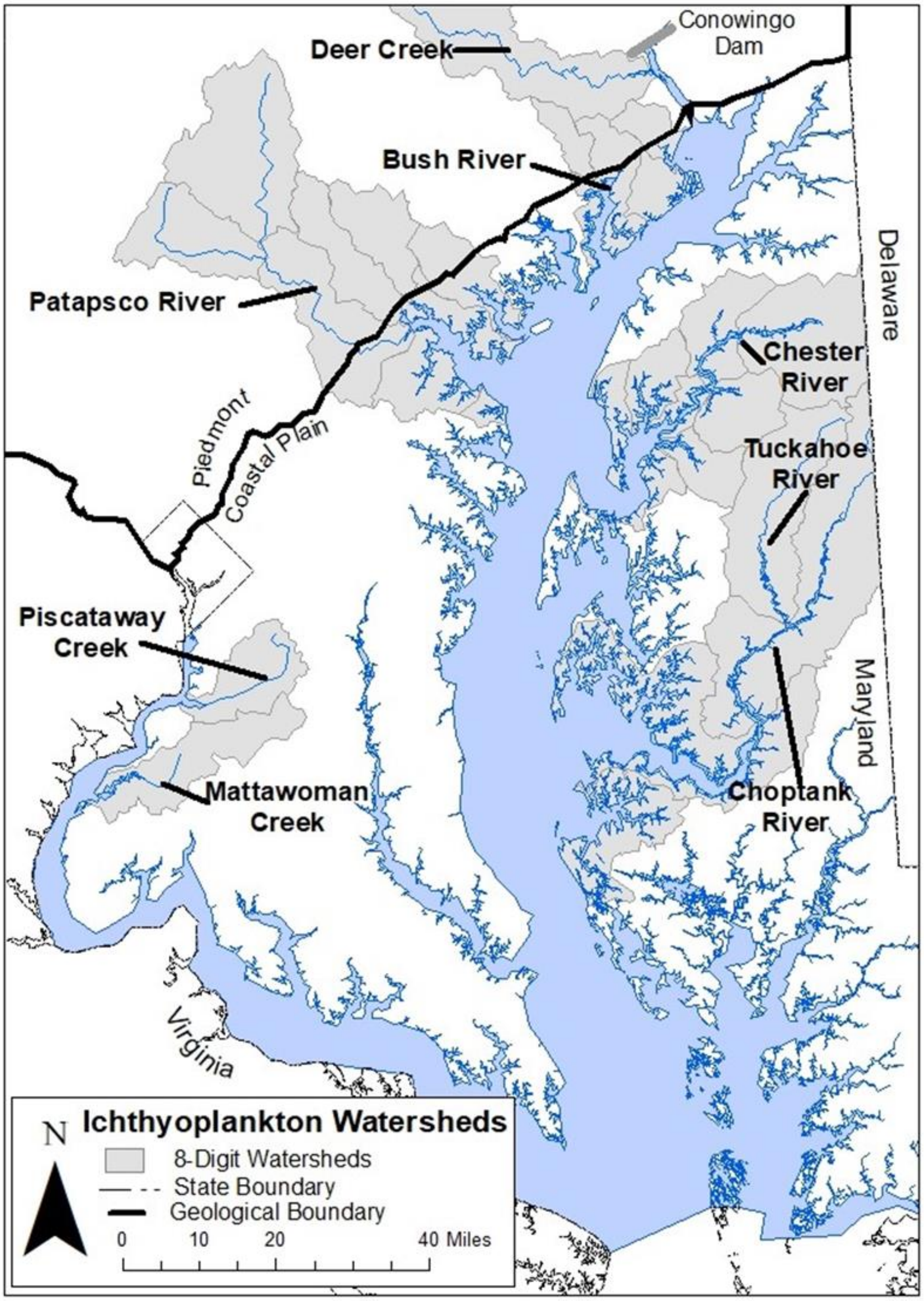


Figure 1-2. Mattawoman Creek's 1971 (O'Dell et al. 1975) and 2008-2018 sampling stations. Bar approximates lower limit of development associated with the town of Waldorf.

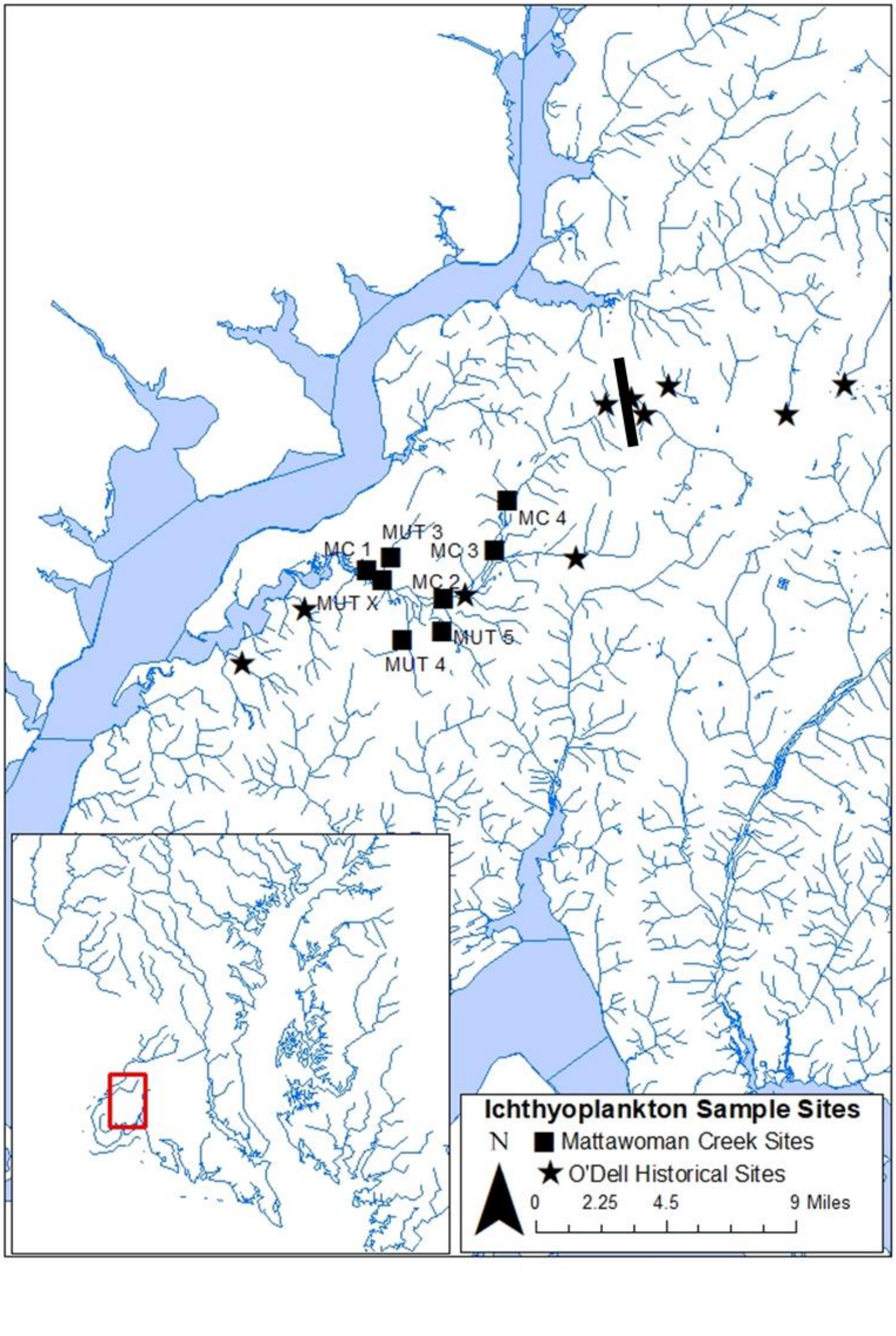


Figure 1-3. Piscataway Creek's 1971 (O'Dell et al. 1975), 2008-2009, and 2012-2014 sampling stations.

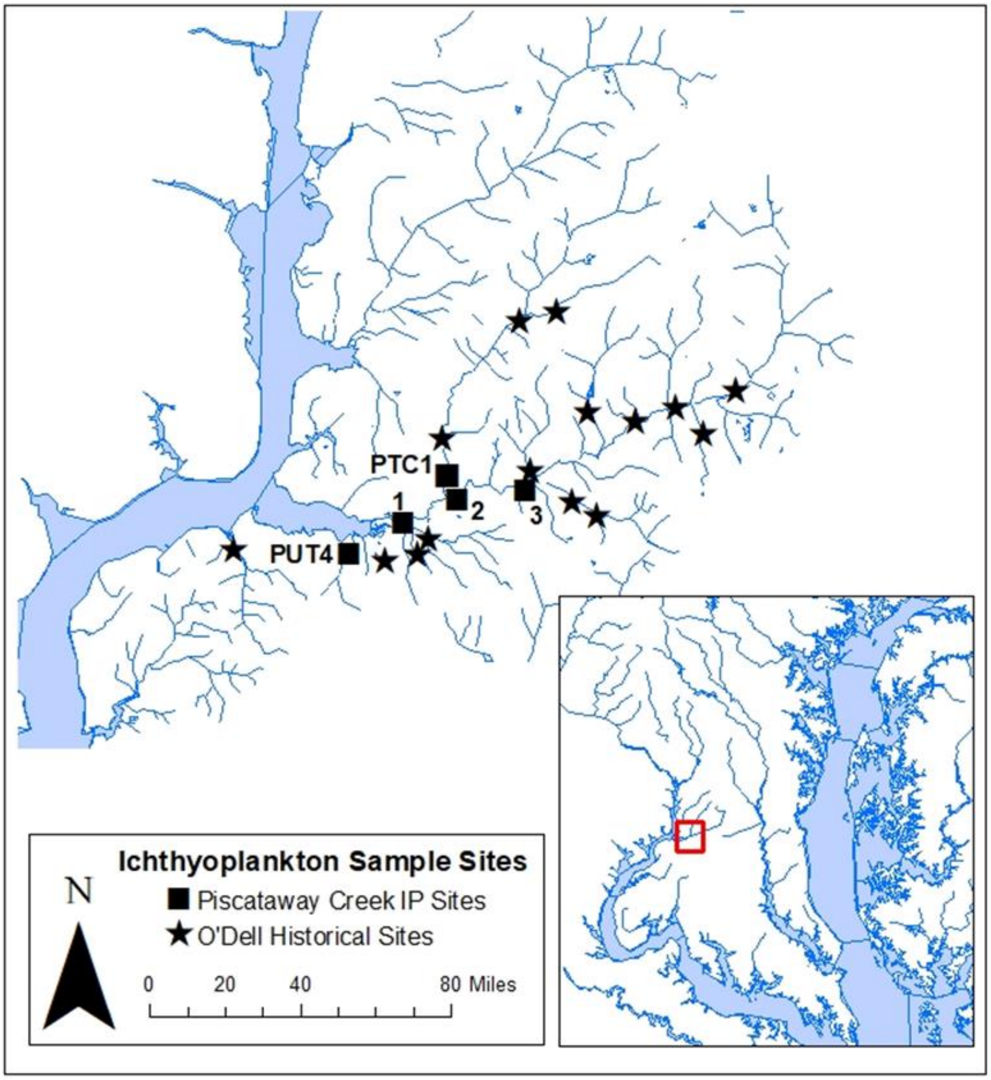


Figure 1-4. Bush River's 1973 (O'Dell et al. 1975), 2005-2008, and 2014 sampling stations. Stations in Aberdeen Proving Grounds (APG) have been separated from other Bush River stations. Line delineates APG streams that were excluded.

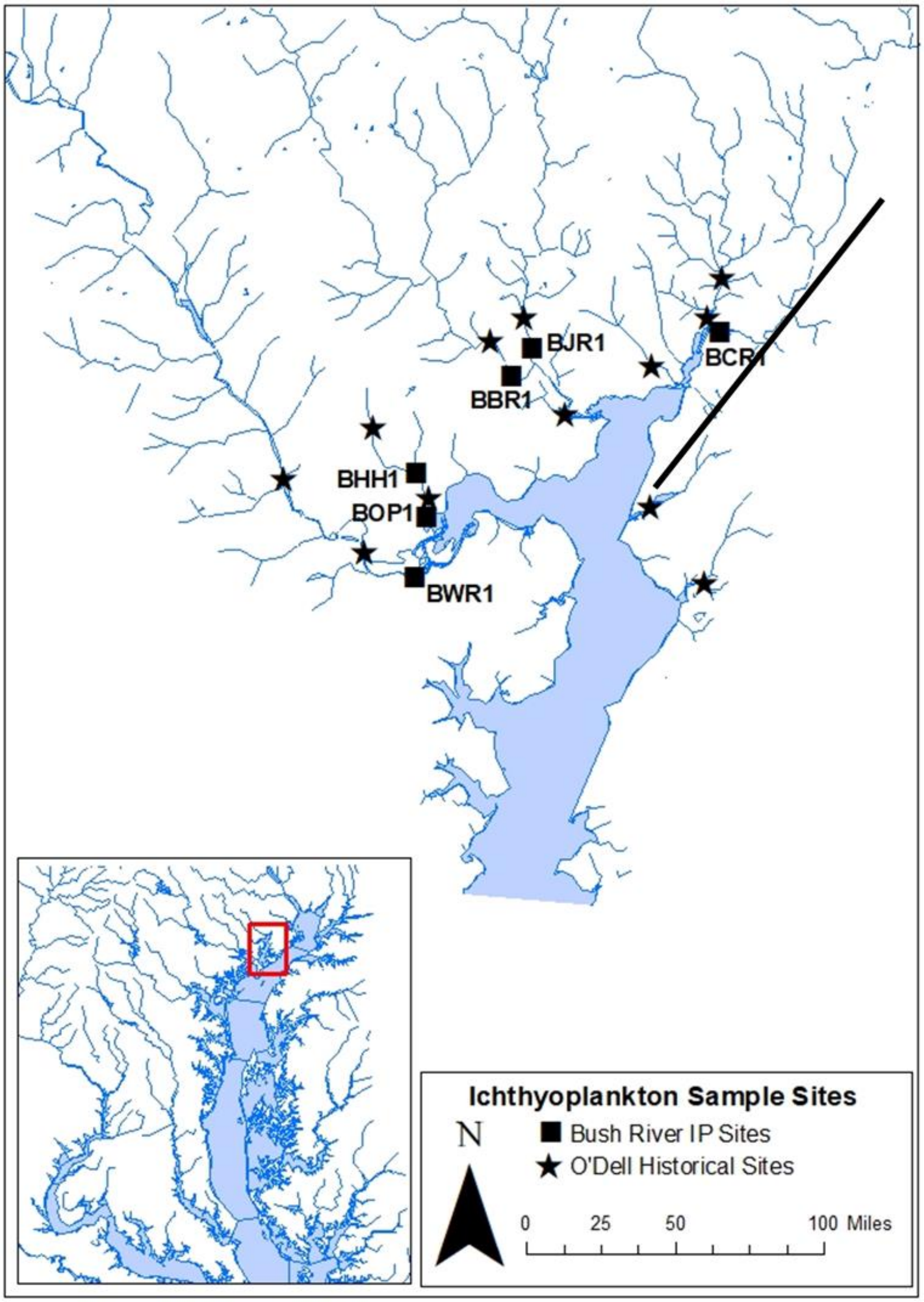
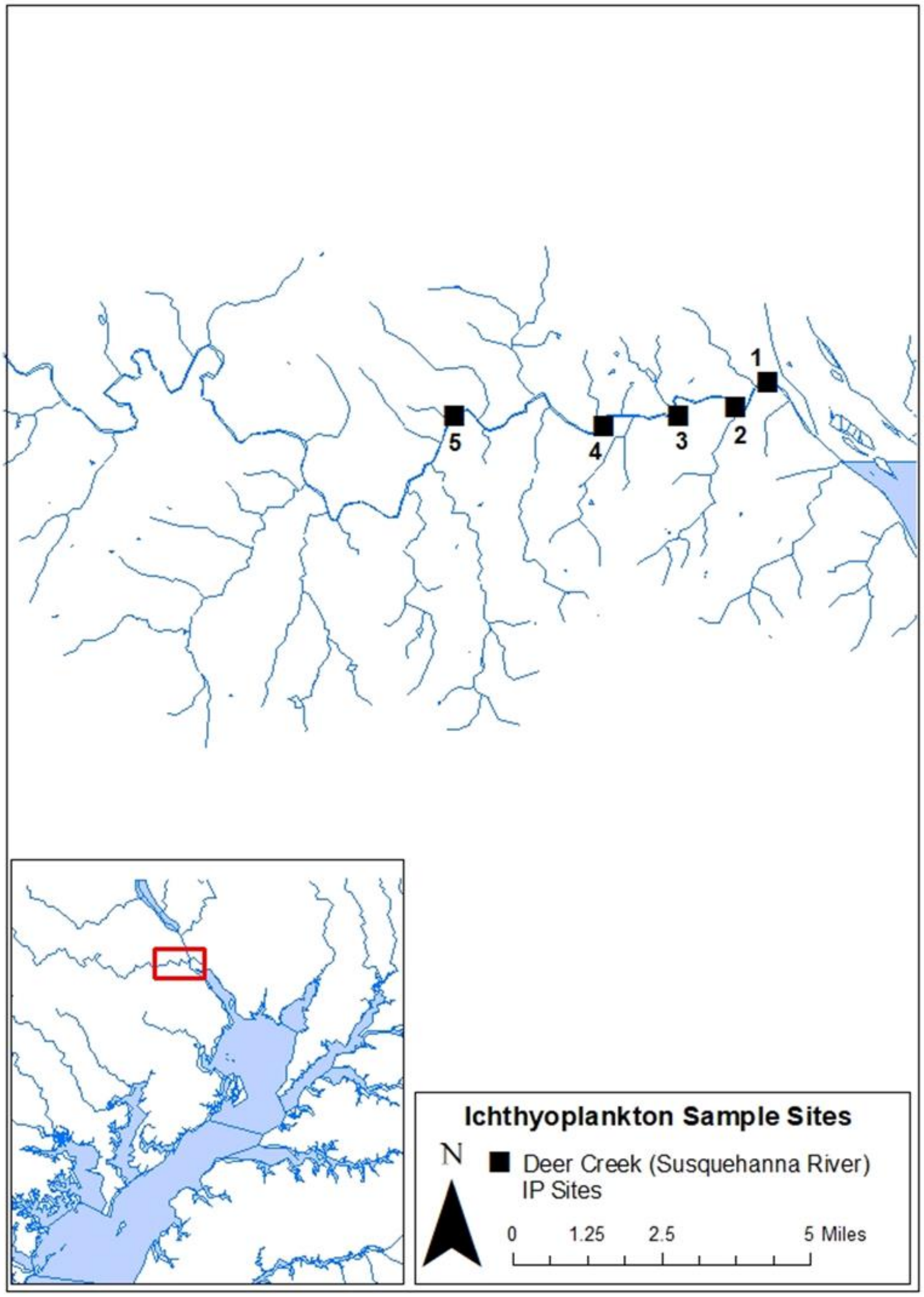


Figure 1-5. Deer Creek's 1972 (O'Dell et al. 1975) and 2012-2015 sampling stations.



Figures 1-6 and 1-7. Choptank River and Tuckahoe Creek's 2016-2017 sampling stations. Stars indicate sites only sampled by O'Dell et al. (1980). D = drift net and T = towed 0.5 m diameter net. stations TUC101, TUC102, TUC103, TUC108, TUC109, and TUC110 correspond to O'Dell et al. (1980) sites

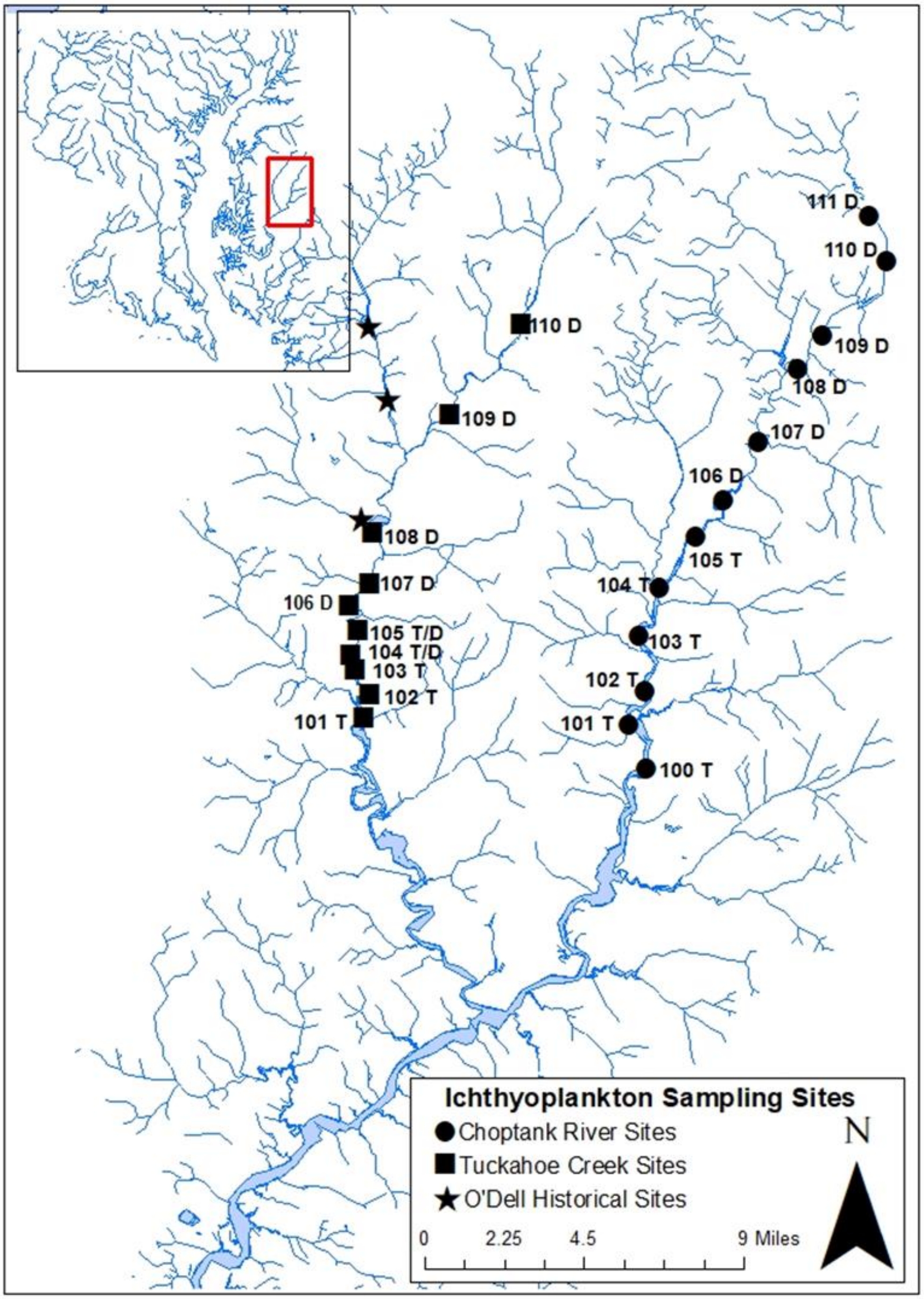


Figure 1-8. Patapsco River's 1973 (O'Dell et al. 1975) and 2013-2017 sampling stations.

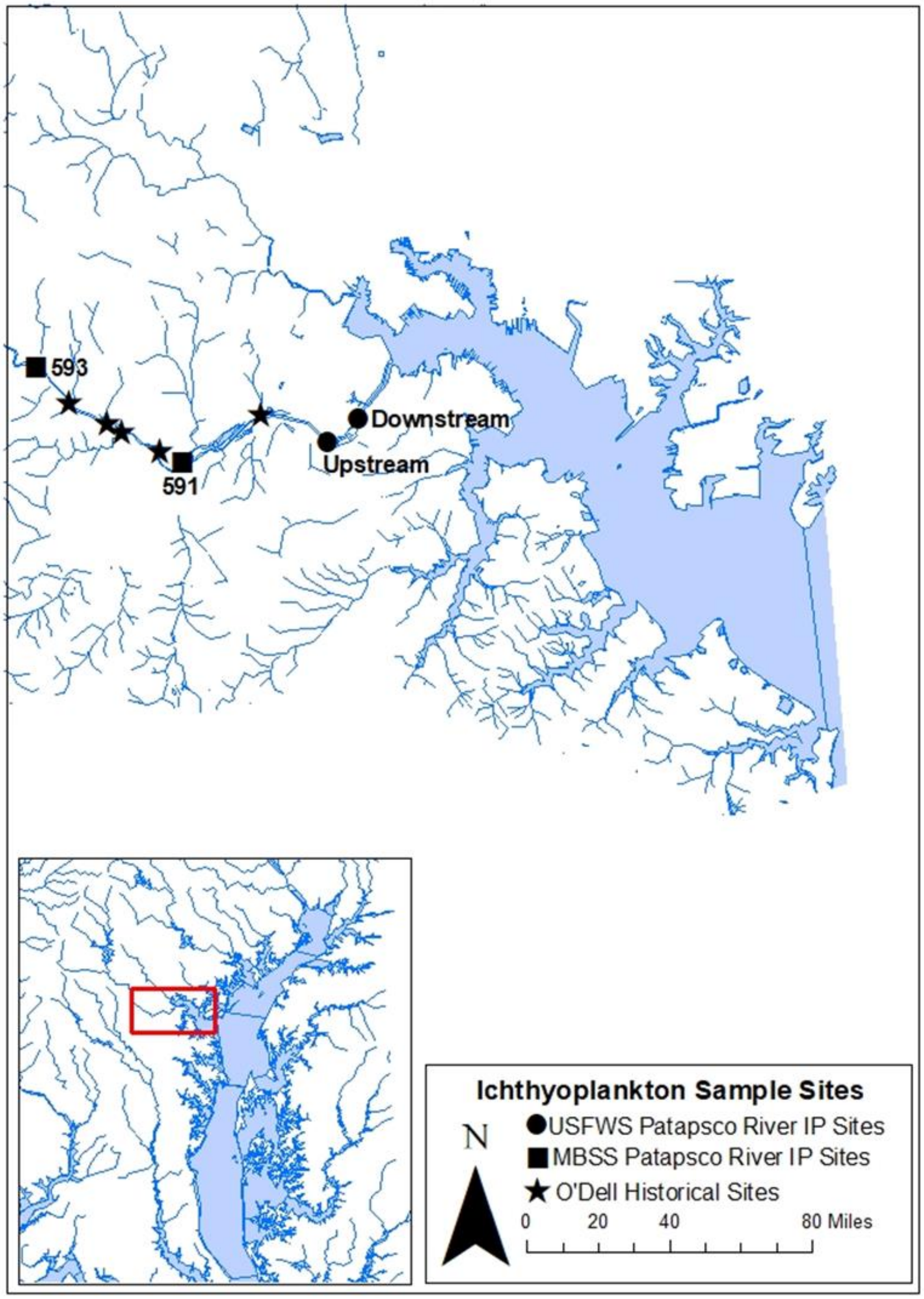


Figure 1-9. Chester River's 1975-1977 (O'Dell et al. 1980) and 2019 sampling stations.

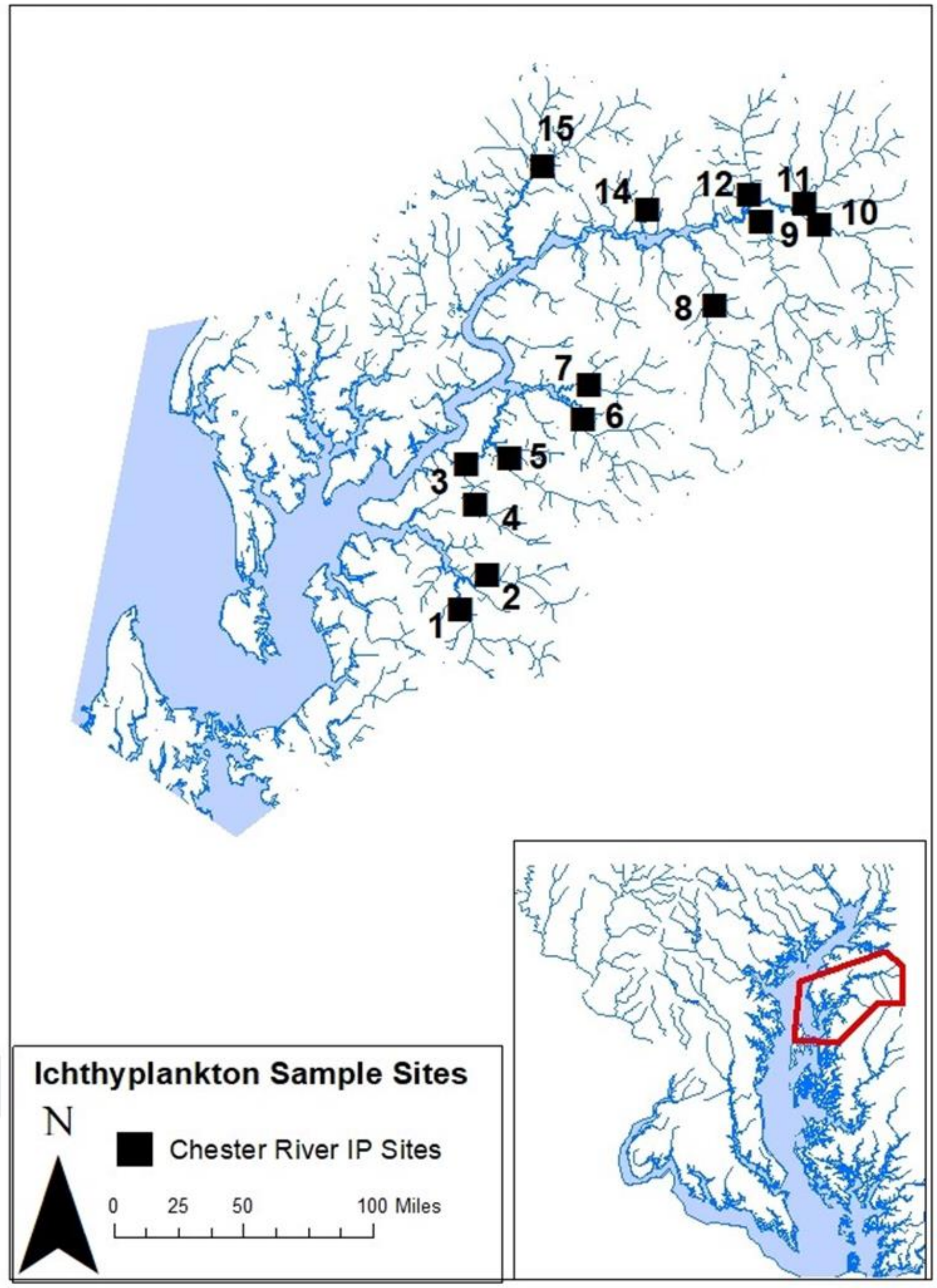


Figure 1-10. Trends in counts of structures per hectare (C/ha) during 1950-2019 in Deer, Mattawoman, and Piscataway Creeks, the Bush and Patapsco Rivers, and the Chester and Choptank River drainages. Estimates of C/ha were only available to 2017 or 2018, depending on Department of Planning data updates. Large symbols indicate years when stream ichthyoplankton was sampled.

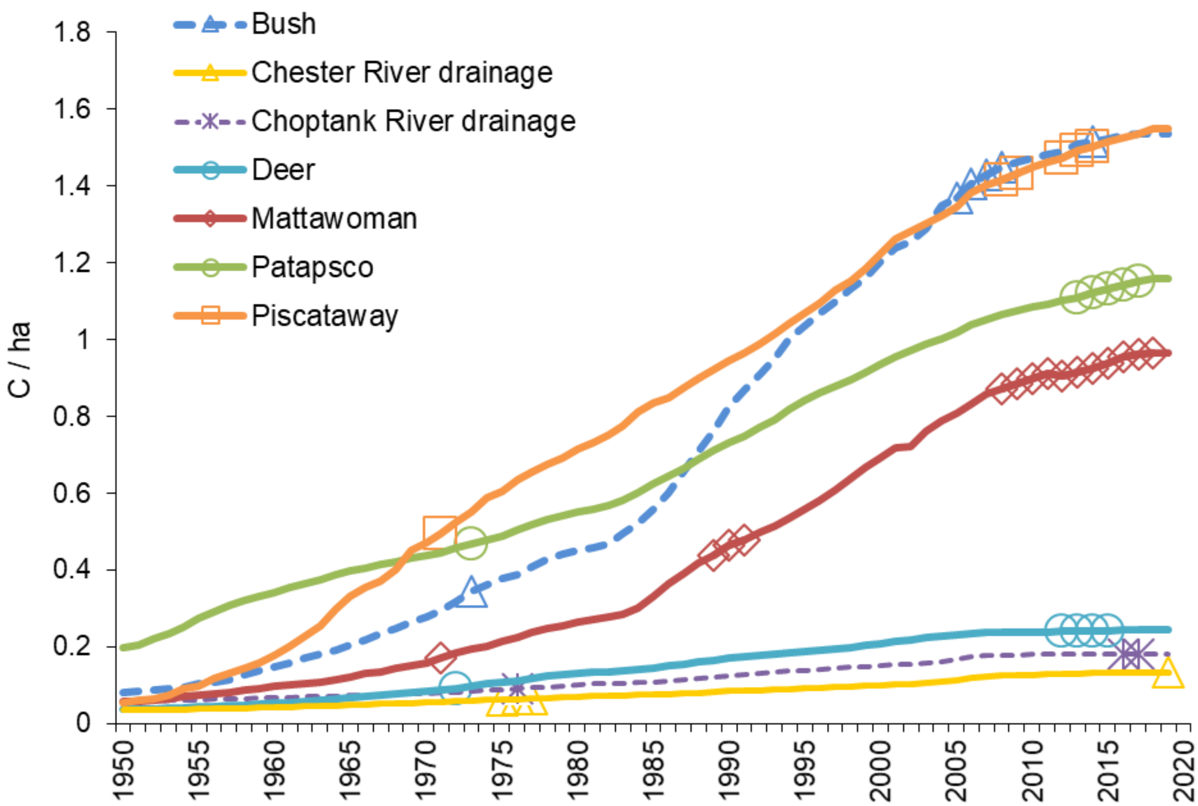


Figure 1-11. Proportion of samples (P_{herr}) with Herring and their 90% confidence intervals for stream ichthyoplankton surveys in Mattawoman, Piscataway, Deer, and Tuckahoe Creeks, and Bush, Choptank, Patapsco, and Chester Rivers.

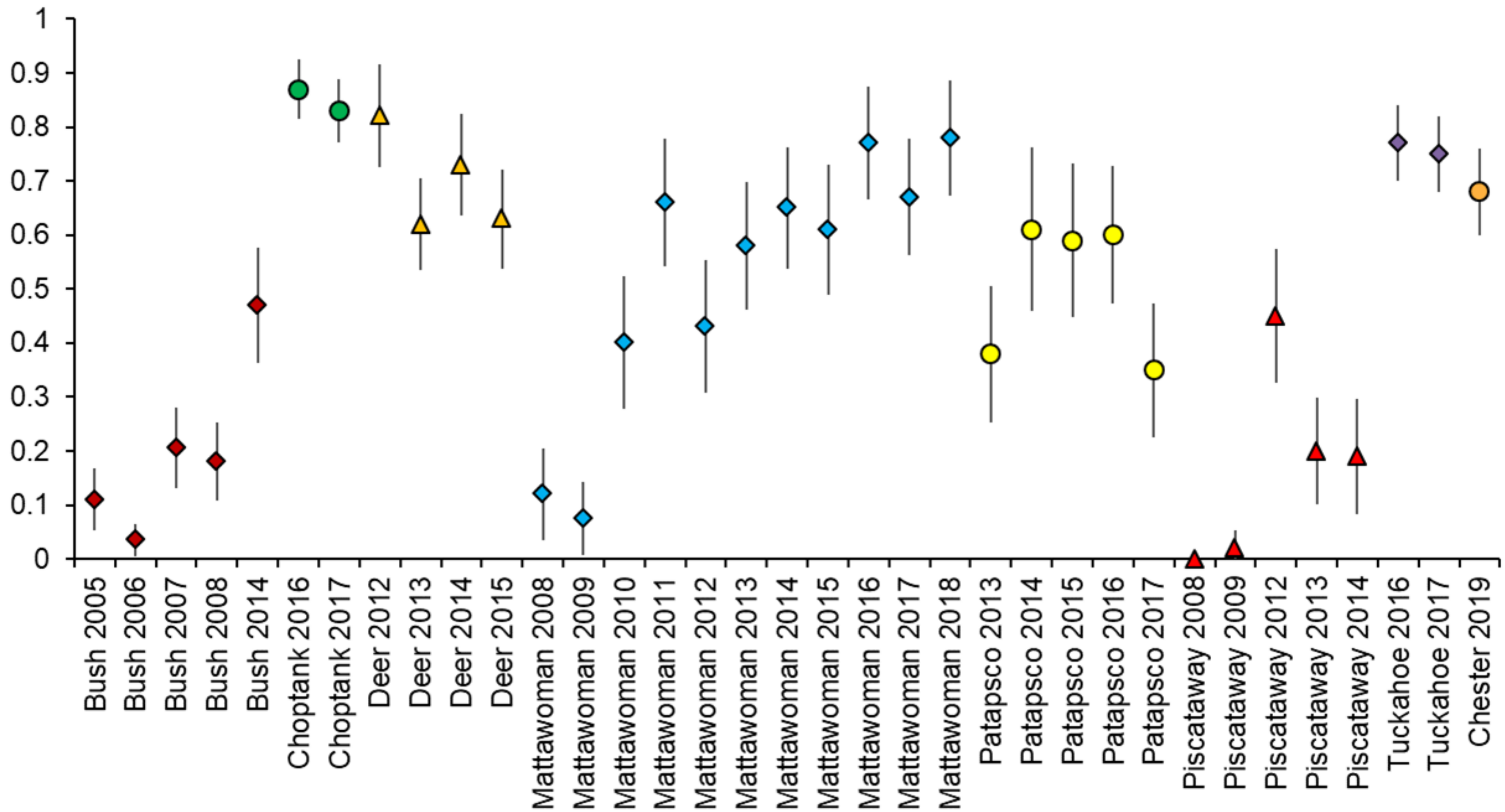


Figure 1-12. Trends in P_{herr} (proportion of stream samples with Herring eggs and-or larvae) by watershed. Watersheds sampled in both early (2005-2011) and late (2012-2019) spawning periods are indicated by large triangles.

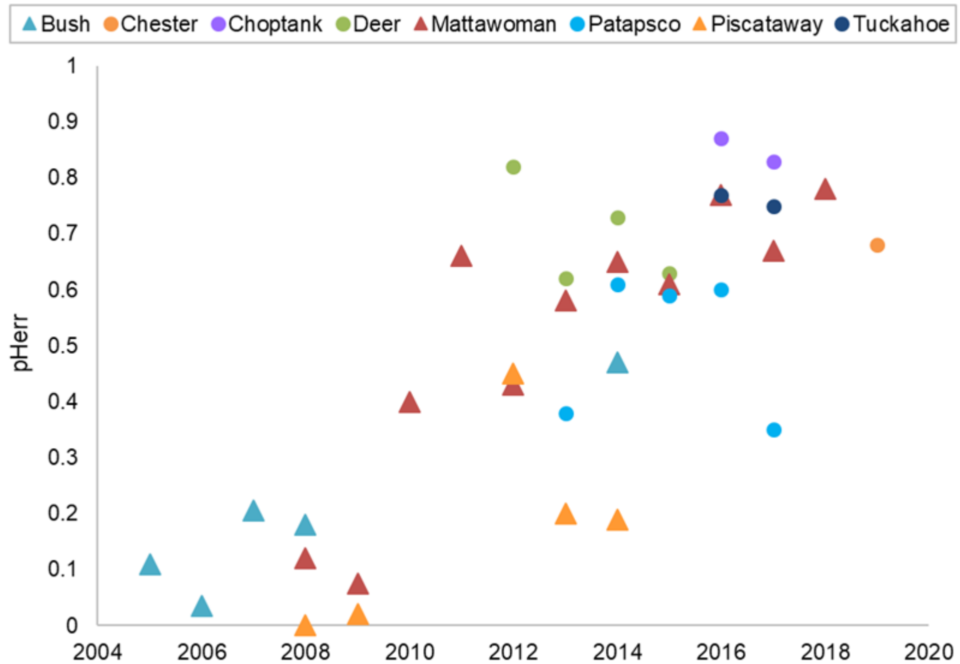


Figure 1-13. Standardized median conductivity during spring spawning surveys and level of development (C/ha). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

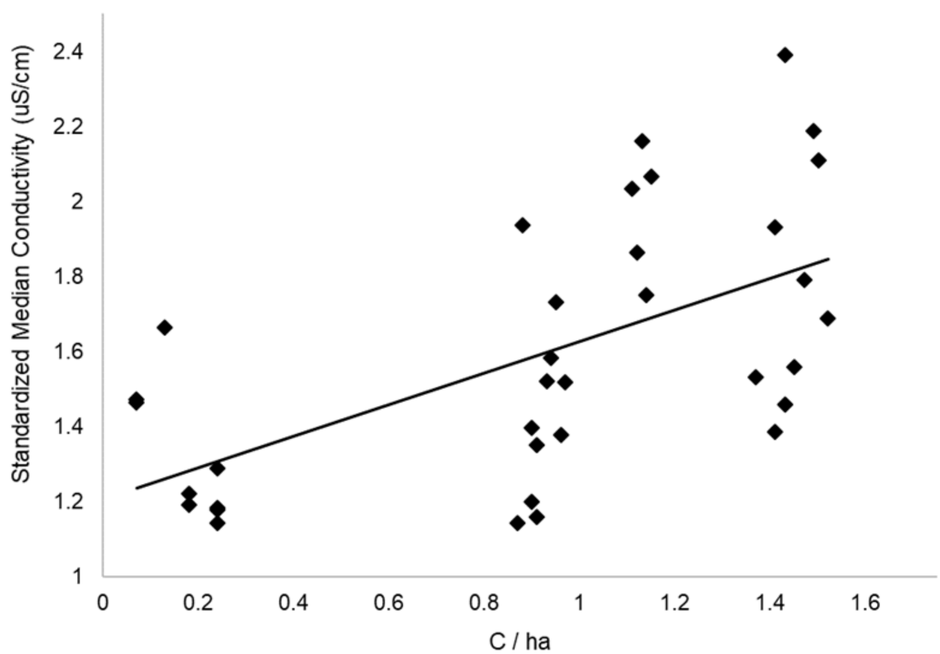


Figure 1-14. (A) Proportion of stream samples with Herring eggs and-or larvae (P_{herr}) and level of development (C/ha) with dominant Department of Planning land use designations. (B) P_{herr} and standardized median spawning survey conductivity (uS/cm). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

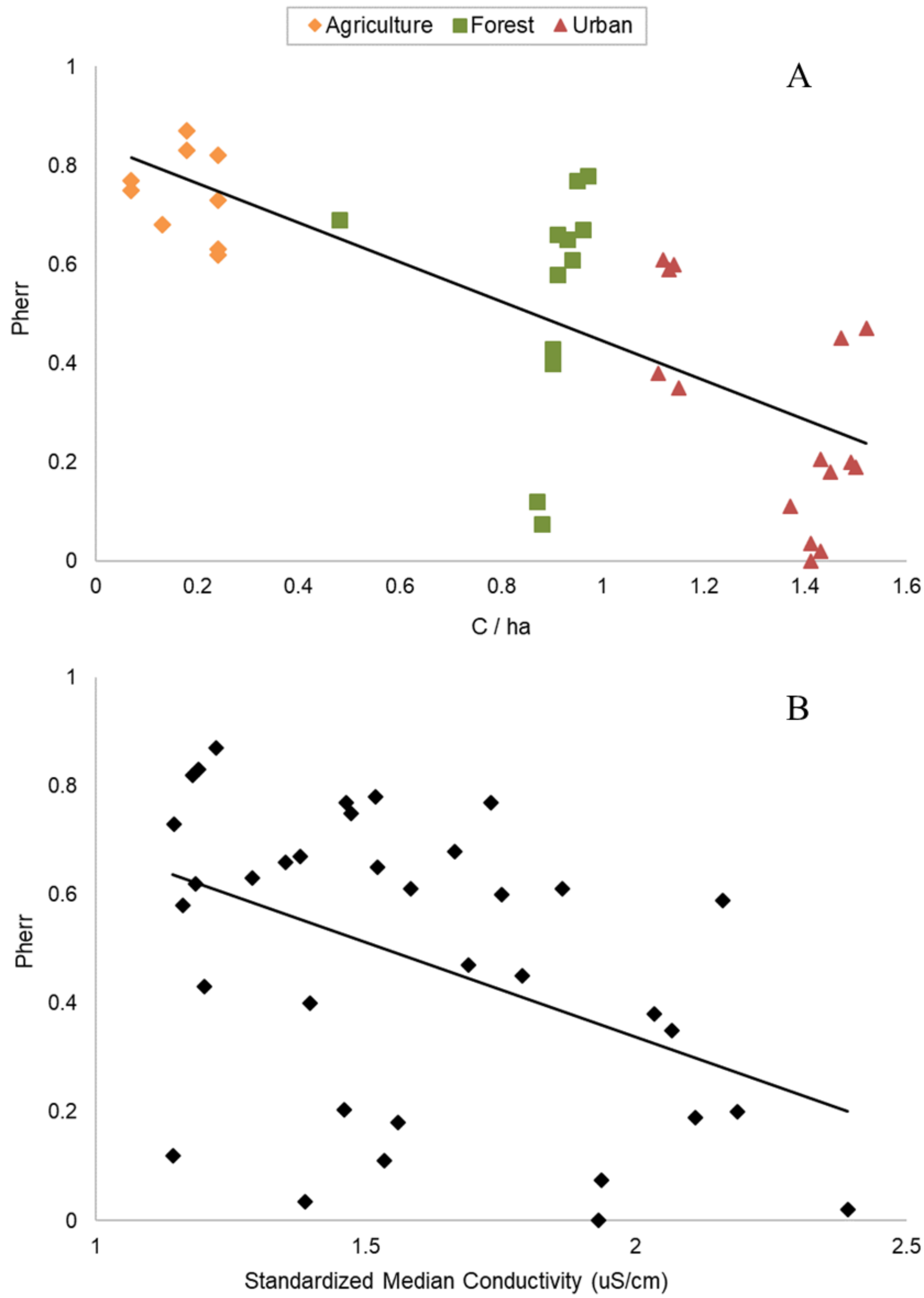


Figure 1-15. Serial patterns of residuals versus year for regressions of P_{herr} (proportion of stream samples with Herring eggs and-or larvae) and (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on Morgan et al. (2012).

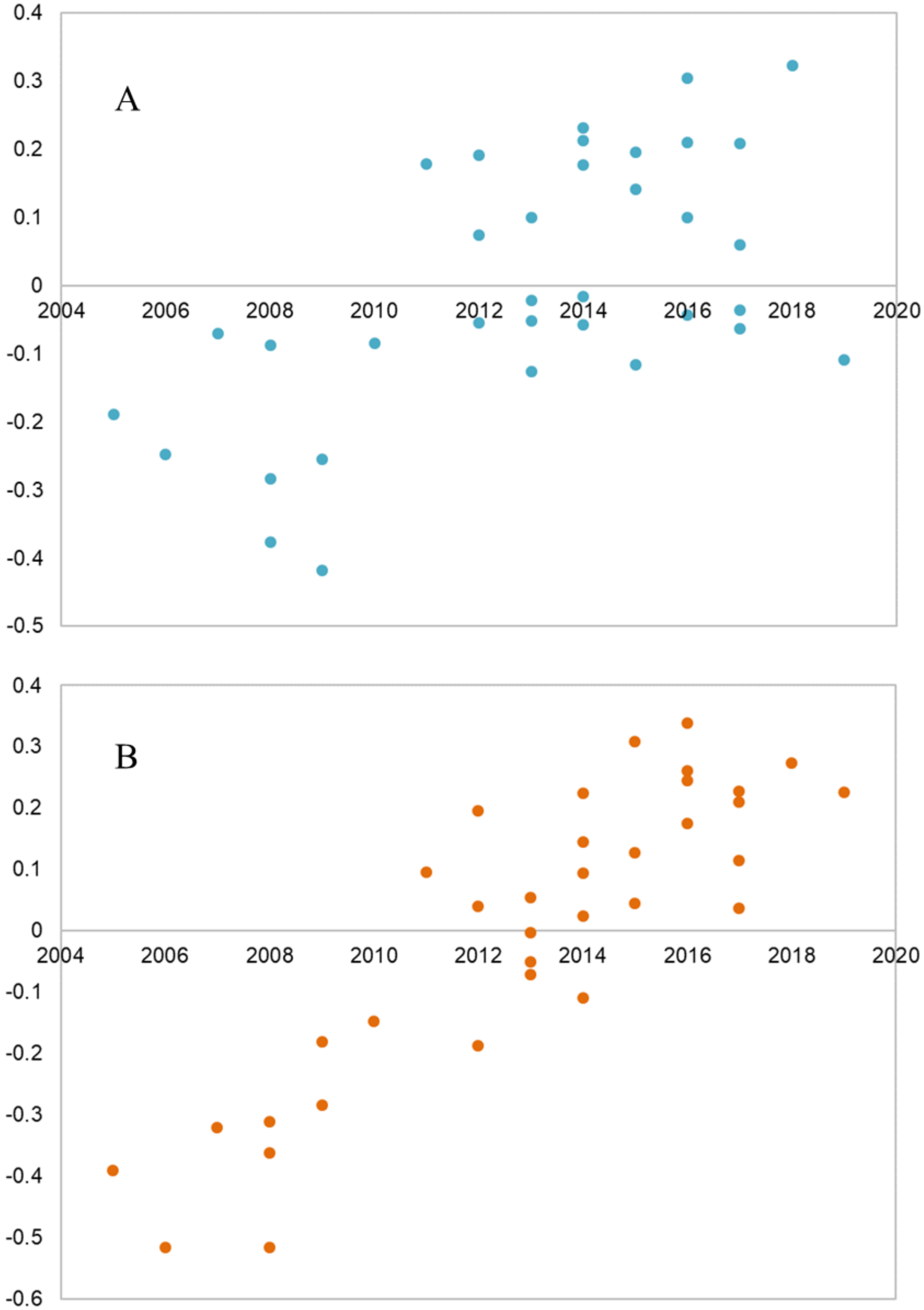


Figure 1-16. Plots of regressions of P_{herr} (proportion of stream samples with Herring eggs and-or larvae) against (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm) with spawning stock time categories (0 = 2005-2011; 1 = 2012-2019) included. Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

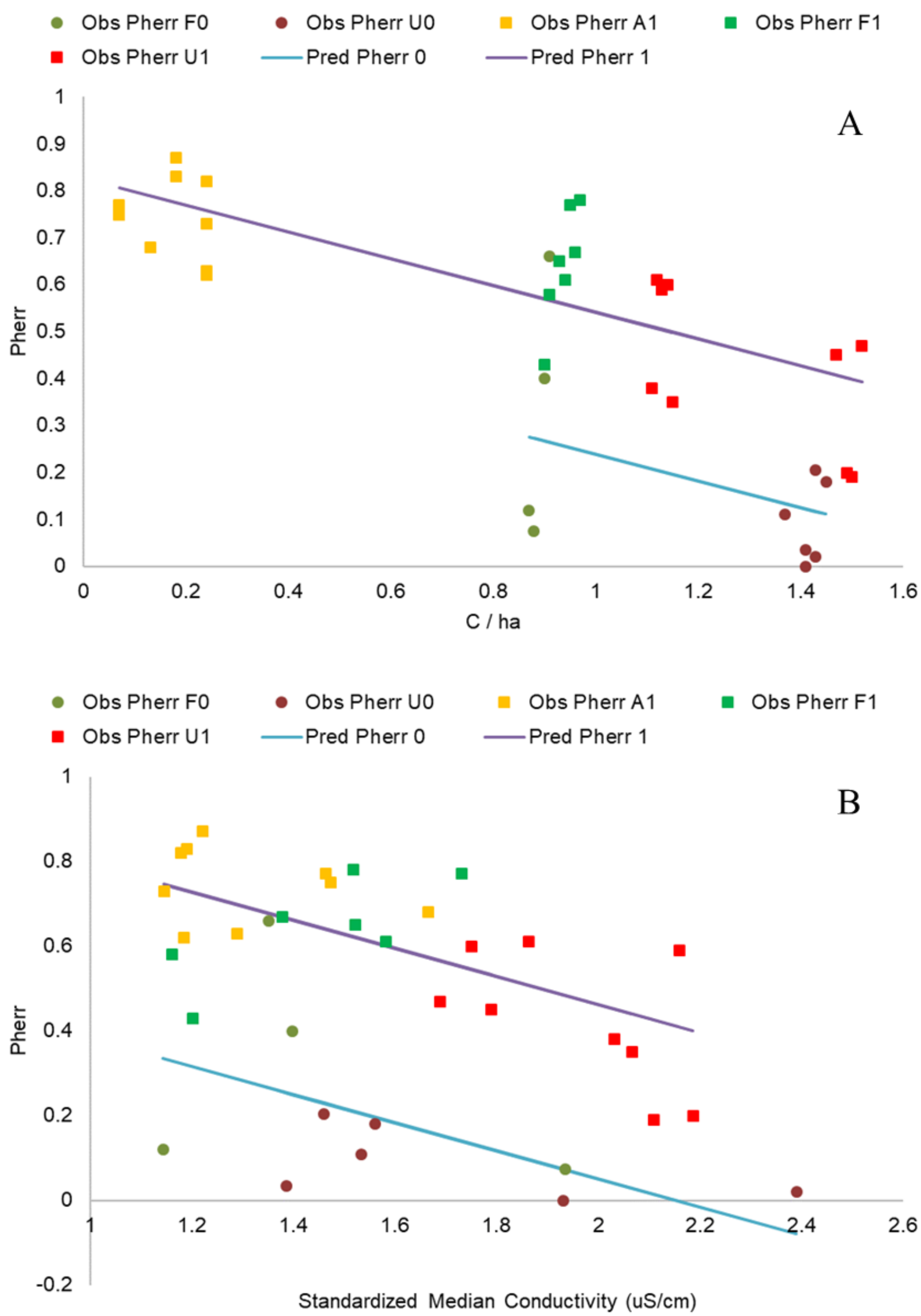
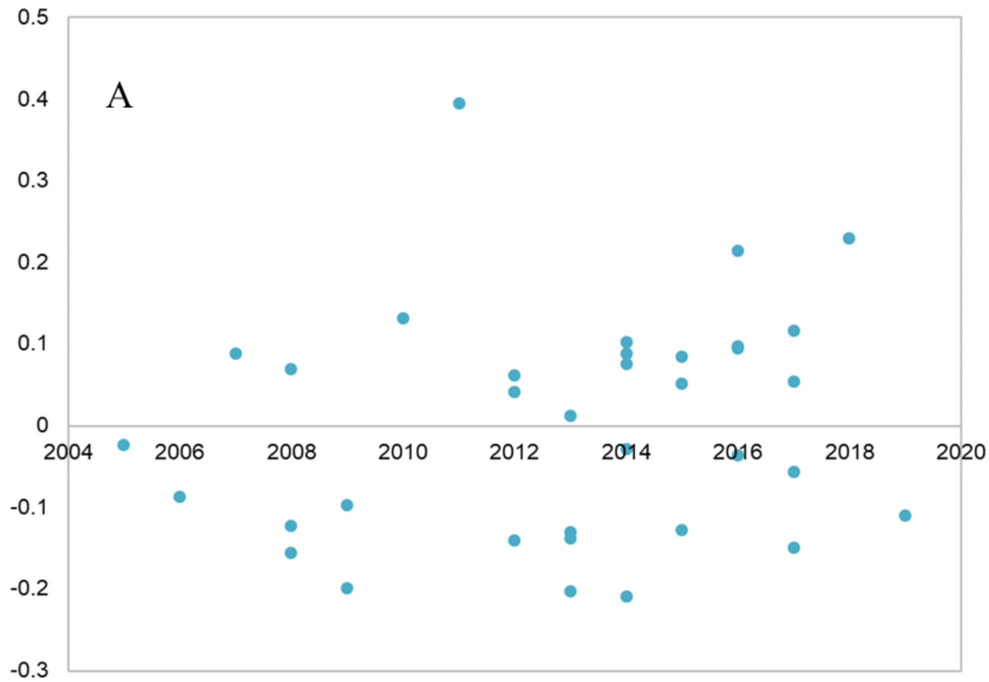
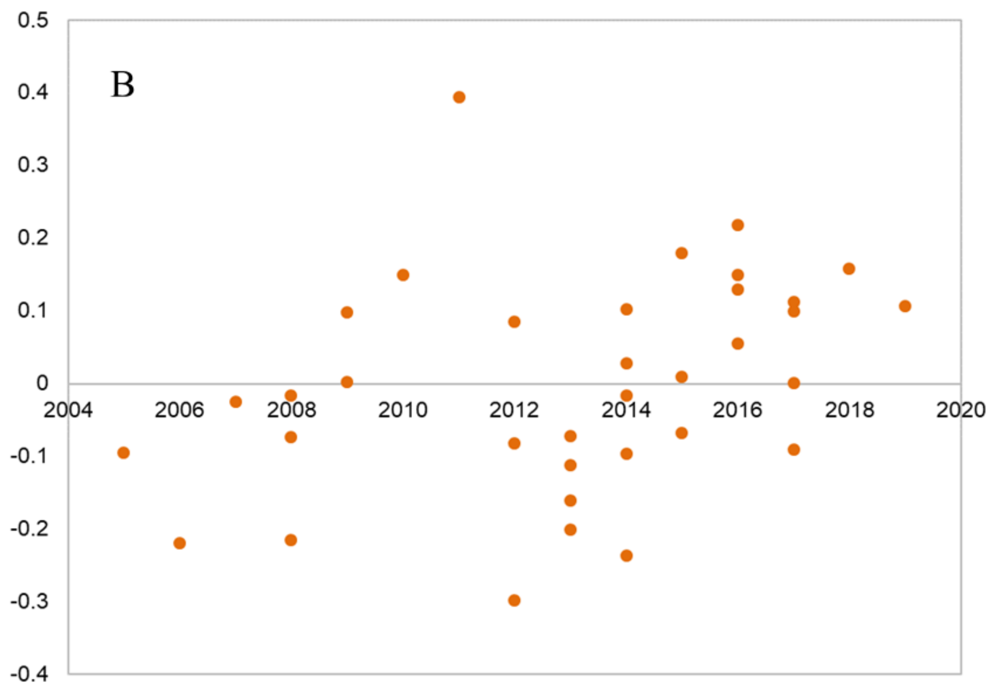


Figure 1-17. Residuals versus year for multiple regressions of spawning stock size time category and (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm) against proportion of stream samples with Herring eggs and-or larvae (P_{herr}). Median conductivity was 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 Seth Dawson

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 2019 was conducted in the upper tidal reaches of the Choptank, Nanticoke, and Chester rivers (Figure 2-1). Sampling started the first week of April in all three systems and continued through the end of the month.

In 2019 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

Choptank and Chester Rivers were sampled by program personnel in 2019. 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.

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 (DO), conductivity, pH, and salinity were measured at each site on each sample date. Each sample was collected in a glass jar which was then emptied into a dark pan to check for Yellow Perch 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.

Ten sites were sampled twice weekly in all systems (Figure 2-1). Boundaries of areas sampled were determined from Yellow Perch larval presence in estuarine surveys conducted during the 1970s and 1980s (O'Dell 1987) when this information was available. In larger subestuaries with designated Striped Bass areas (Choptank, Nanticoke, Patuxent, Wicomico, and Chester rivers), boundaries were the same as the legal Striped Bass spawning areas. Estimates of L_p were initially developed from historical surveys conducted for Striped Bass eggs and larvae in the Choptank and Nanticoke rivers (Uphoff 1993) and continuity with past surveys was maintained by sampling Striped Bass spawning areas.

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, usually mid-to-late April, depending on larval presence and catchability. 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; see also Section 2.1), 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).

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) :

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

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:

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

The 95% confidence intervals were constructed as:

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

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, Chester, Wicomico (eastern shore region of Maryland or 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 2017 or 2018, whichever year the most recent data was available for (M. Topolski, MD DNR, personal communication).

Estimates of C/ha for 2017 or 2018 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 within the same time span (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 (1986-2019; N = 19) and Nanticoke River (1965-2019; N = 20). Neither time-series was continuous; Choptank River was sampled during 1986-2004 and 2013-2019, while the Nanticoke River estimates were available for 1965, 1967, 1968, 1970, 1971, 2004-2009, and 2011-2019.

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 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; Freund and Littel 2006):

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

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 2019). 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 OM0 estimates were available for 2011-2019. 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 2019, sampling on Choptank River began on April 2 and lasted until May 2. Sampling on Chester River began on April 3 and concluded on May 1. Samples through April 18 and April 22 were used to estimate L_p in Choptank and Chester Rivers, respectively. Sampling began on April 5 and ended on April 26 in the Nanticoke River, and all samples were used for estimating L_p.

Estimates of mean L_p were above the brackish threshold (0.40; 95% CI's did not overlap the threshold) in Chester River L_p = 0.73 and Choptank River L_p = 0.69 during 2019 (Figure 2-2). Estimate of mean L_p in the Nanticoke River (L_p = 0.41) in 2019 did, however, overlap the brackish subestuary threshold based on 95% CIs (Figure 2-2).

Comparisons of L_p during 2019 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 0.05-2.84 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 1986-2019 exhibited little indication of decline (r^2 = 0.004; P = 0.80), while a decline of L_p of about 0.005 per year was detected during 1965-2019 (predicted L_p declined from 0.63 to 0.38) in the Nanticoke River (r^2 = 0.17; P = 0.07; Figure 2-3). Both of these subestuaries are rural, land use is dominated by agriculture, and they are closed to commercial fishing.

Separate linear regressions of C/ha and L_p by salinity category 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 ($P \leq 0.0005$; Table 2-2; Figure 2-5). 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 different between tidal-fresh (mean = 0.95, SE = 0.09) and brackish (mean = 0.58, SE = 0.03) subestuaries. Mean slope for C/ha estimated for tidal-fresh subestuaries (mean = -0.28, 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 moderate 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 0.58, 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 0.13 in brackish subestuaries and from 0.82 to 0 in tidal-fresh subestuaries (Figure 2-5).

Estimates of L_p were poorly related to agriculture ($r^2 = 0.12$, $P = 0.0041$) and forest ($r^2 = 0.03$, $P = 0.1762$) 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 a relationship 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, $r^2 = 0.14$, $P = 0.0001$; 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 $R^2 = 0.32$; salinity, $P = 0.0060$ and size, $P = 0.1306$). 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, Wicomico (ES), and Chester 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 2015-2019, 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 2-12; See Section 2.1 also). In 2015, urban and rural water quality measurements were similar, with the exception of elevated median conductivity 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. In 2019, water quality measurements were similar in the Choptank and Chester Rivers, both of which are rural agricultural systems (Table 2-6; Figure 2-12). The exception was median conductivity, which was significantly higher in the Chester River, although salinity values were higher overall there as well (Choptank mean = 0.16 and median = 0.07 ppt; Chester mean = 0.45 and median = 0.19 ppt). Spring rainfall was high in 2019 and salinity in Chesapeake Bay was much lower than normal (USGS 2019), so it seems contradictory that salinity in the upper Chester River was higher. 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 lower 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 = 0.46-0.95; 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; 43 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 = 0.68). 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 ≥ 1.24; 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 provided a moderate fit to the data (approximate $r^2 = 0.47$, $P < .0001$; $N = 42$), depicting OM0 increasing towards 1.0 at a decreasing rate as C/ha approached 1.50 (Figure 2-13). The relationship was described by the equation:

$$^{(5)}\ \mathrm{OM0} = 0.78 \cdot ((\mathrm{C/ha})^{0.26}).$$

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, and OM0 and wetland percentage were negatively related. An inverse power function provided a moderate fit of C/ha and percent wetland (approximate $r^2 = 0.46$, $P < .0001$, $N = 42$; Figure 2-14), while the relationship of OM0 and wetland percentage was linear ($r^2 = 0.53$, $P < .0001$, $N = 42$; Figure 2-14). These relationships 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 analyses: 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 $L_p = 0.52$) 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 Yellow Perch 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. These changes 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 to extremely wet (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) and hypoxic conditions are well developed by the time we begin our summer habitat assessments in early July (see Section 3).

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 with rural watersheds 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). Winter water temperature has also been found to have an influence on peak abundances of an important prey species of larval Striped Bass, which could affect recruitment in the spring (Millette et al. 2020). Millette et al. (2020) found that low temperature delayed development timing and increased the size of peak spring abundance of copepod nauplii in Chesapeake Bay Striped Bass larval nurseries. Results suggest that cold winters, in conjunction with freshwater discharge, explained up to 78% of annual recruitment variability in Striped Bass due to larvae occurring at the same time as high concentrations of their prey (Millette et al. 2020). Yellow Perch and Striped Bass larvae are found in the same regions of large tidal rivers in Chesapeake Bay (Uphoff 1991).

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 > 1.32; 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 western shore subestuaries below the Head-of-Bay: Rock and Curtis creeks and Severn, South, and Magothy rivers (Piavis and Uphoff 1999). 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). Development associated with increased human population growth in the Chesapeake Bay watershed converts natural sources of organic matter (forests and welands) to agricultural, residential and industrial uses that alters and lessens the supply of watershed organic matter.

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) rivers than rural Choptank River 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 when compared with rural Choptank River; however, presence of OM and subsequent feeding success of first-feeding Yellow Perch was negatively influenced by development in multiple subestuaries (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 OM), 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 decade. 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 DO, 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 1965-2019 and data used for regressions 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
Chester	2019	2010	0.13	66.6	24.5	0.8	7.8	Agriculture	1	0.73
Choptank	1986	1994	0.07	64	29.2	2.3	4.4	Agriculture	1	0.53
Choptank	1987	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.73
Choptank	1988	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.80
Choptank	1989	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.71
Choptank	1990	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.66
Choptank	1998	1997	0.10	63.6	27.7	2.2	6.4	Agriculture	1	0.60
Choptank	1999	1997	0.11	63.6	27.7	2.2	6.4	Agriculture	1	0.76
Choptank	2000	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.25
Choptank	2001	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.21
Choptank	2002	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.38
Choptank	2003	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.52
Choptank	2004	2002	0.12	63.9	27.1	2.1	6.9	Agriculture	1	0.41
Choptank	2013	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.47
Choptank	2014	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.68
Choptank	2015	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.82
Choptank	2016	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.90
Choptank	2017	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.40
Choptank	2018	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.44
Choptank	2019	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.69
Corsica	2006	2002	0.21	64.3	27.4	0.4	7.9	Agriculture	1	0.47
Corsica	2007	2010	0.22	60.4	25.5	0.1	13.2	Agriculture	1	0.83
Elk	2010	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.75
Elk	2011	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.79
Elk	2012	2010	0.60	28	38.7	1.1	31.2	Forest	0	0.55

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Langford	2007	2010	0.07	70.2	20.4	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	2.84	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	0.94	9.3	53.9	2.8	34.2	Forest	0	1.00
Mattawoman	2016	2010	0.95	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
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

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
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
Nanticoke	2019	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.41
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	1.24	20.5	35.1	1	41.7	Urban	1	0.72
Patuxent	2016	2010	1.25	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	0.68	30.1	36.8	2.3	29.9	Forest	1	0.53
Wicomico (ES)	2018	2010	0.68	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	1.04301	1.04301	19.84	<.0001	
Error	62	3.25869	0.05256			
Total	63	4.3017				
r^2	0.2425					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57783	0.03401	16.99	<.0001	0.50984	0.64581
C / ha	-0.15669	0.03517	-4.45	<.0001	-0.227	-0.08638

ANOVA			(A) Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.77146	0.77146	15.45	0.0005	
Error	30	1.49814	0.04994			
Total	31	2.2696				
r^2	0.3399					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94871	0.08622	11	<.0001	0.77263	1.12479
C / ha	-0.28495	0.07364	-3.93	0.0005	-0.43985	-0.13905

ANOVA			(A) Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	2.16912	1.08456	20.63	<.0001	
Error	93	4.89021	0.05258			
Total	95	7.05933				
r^2	0.3073					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.83517	0.05239	15.94	<.0001	0.73113	0.9392
C / ha	-0.18034	0.03189	-5.65	<.0001	-0.24368	-0.117
Salinity	-0.24503	0.05234	-4.68	<.0001	-0.34897	-0.14108

Table 2-2 cont.

ANOVA			(B) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.53955	0.53955	8.89	0.0041	
Error	62	3.76215	0.06068			
Total	63	4.3017				
r^2	0.1254					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.32727	0.06449	5.07	<.0001	0.19836	0.45619
% Ag	0.00415	0.00139	2.98	0.0041	0.00137	0.00692

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	1.17819	0.5891	9.32	0.0002	
Error	93	5.88114	0.06324			
Total	95	7.05933				
r^2	0.1669					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57536	0.04953	11.62	<.0001	0.47701	0.67371
% Ag	0.00452	0.00137	3.30	0.0014	0.0018	0.00724
Salinity	-0.26334	0.06415	-4.1	<.0001	-0.39074	-0.13595

Table 2-2 cont.

ANOVA			(C) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.12605	0.12605	1.87	0.1762	
Error	62	4.17565	0.06735			
Total	63	4.3017				
r^2	0.0293					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.36579	0.10073	3.63	0.0006	0.16443	0.56715
% Forest	0.00361	0.00264	1.37	0.1762	-0.00166	0.00887

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	0.77817	0.38908	5.76	0.0044	
Error	93	0.62812	0.06754			
Total	95	7.05933				
r^2	0.1102					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.44379	0.10849	4.09	<.0001	0.22834	0.65924
% Forest	0.00482	0.00233	2.07	0.041	0.0002021	0.00945
Salinity	-0.12208	0.058	-2.1	0.038	-0.23726	-0.00691

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	0.05258	96	4	2.94541946	8	40	91	11.4	2.04	1.53
Fresh	0.04994	32	3	2.996932994	6	24	28	9.9		
Brackish	0.05256	64	3	2.945799905	6	24	60	9.3		

Model (B)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.06324	96	4	2.760818267	8	40	91	11.2	2.00	1.66
Fresh	0.06856	32	3	2.680046005	6	24	28	9.5		
Brackish	0.06068	64	3	2.802141125	6	24	60	9.2		

Model (C)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.06754	96	4	2.695035264	8	40	91	11.1	2.04	1.59
Fresh	0.06803	32	3	2.687806495	6	24	28	9.5		
Brackish	0.06735	64	3	2.697852376	6	24	60	9.1		

Table 2-4. Summary of results of regressions of proportions of tows with Yellow Perch larvae (L_p) and (A) small system counts of structures per hectare (C/ha), or (B) large system 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	1.21497	1.21497	22.9	0.0001	
Error	18	0.95513	0.05306			
Total	19	2.1701				
r^2	0.5599					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.70463	0.07948	8.87	<.0001	0.53765	0.8716
C / ha	-0.22859	0.04777	-4.79	0.0001	-0.32896	-0.12823

ANOVA			(A) Small Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.77146	0.77146	15.45	0.0005	
Error	30	1.49814	0.04994			
Total	31	2.2696				
r^2	0.3399					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94871	0.08622	11	<.0001	0.77263	1.12479
C / ha	-0.28945	0.07364	-3.93	0.0005	-0.43985	-0.13905

ANOVA			(B) Large Brackish			
Source	df	SS	MS	F	P	
Model	1	0.02282	0.02282	0.62	0.4399	
Error	22	0.81136	0.03688			
Total	23	0.83418				
r^2	0.0274					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.5662	0.04887	11.58	<.0001	0.46484	0.66756
C / ha	0.09109	0.1158	0.79	0.4399	-0.14906	0.33124

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 2015-2019.

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	0.3846	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	0.5714	0.0836
Up-river	9-12	8	31	0.2581	0.0786

Choptank 2019	Stations	Presence	N	Lp	SD
Down-river	1-5	4	9	0.4444	0.1656
Mid-river	6-11	15	20	0.7500	0.0968
Up-river	12-17 and 18-20	22	30	0.7333	0.0807

Chester 2019	Stations	Presence	N	Lp	SD
Down-river	1-4	6	8	0.75	0.1531
Mid-river	5-8	15	20	0.75	0.0968
Up-river	9-12	16	23	0.6957	0.0959

Table 2-6. Summary of annual water quality parameter statistics for large systems sampled in 2015-2019. Mean pH was calculated from H^+ concentrations, then converted to pH.

System/Year		*Temp C*	*DO (mg/L)*	*Cond (umhols)*	*pH*
Choptank 2015	Mean	14.87	8.05	585.5	7.41
	Standard Error	0.30	0.12	111.62	
	Median	14.41	8.33	193.5	7.43
	Mode	12.5	8.7	172	7.6
	Kurtosis	-0.99	-0.04	5.13	0.09
	Skewness	0.51	-0.76	2.42	0.66
	Minimum	11.9	5.77	137	7.1
	Maximum	19	9.5	3780	8.07
	Count	56	56	56	56
Patuxent 2015	Mean	15.58	8.18	682.08	7.49
	Standard Error	0.19	0.12	82.01	
	Median	15.39	8.2	420	7.5
	Mode	13.50	8.2	416	7.5
	Kurtosis	-0.61	-0.67	7.6	4.49
	Skewness	0.51	0.32	2.84	1.02
	Minimum	13.5	6.48	317	7.22
	Maximum	18.66	10.44	3341	8.12
	Count	65	65	65	65
Choptank 2016	Mean	13.25	8.77	829.24	7.20
	Standard Error	0.12	0.09	149.73	
	Median	13.42	8.73	295.5	7.21
	Mode	13.26	8.21	238	7.29
	Kurtosis	0.33	1.79	2.51	1.11
	Skewness	-1.09	0.73	1.84	0.68
	Minimum	10.96	7.67	148	7.04
	Maximum	14.53	10.87	4389	7.6
	Count	50	50	50	50
Patuxent 2016	Mean	13.01	9.60	1137.23	7.56
	Standard Error	0.14	0.08	144.1	
	Median	12.75	9.34	695	7.56
	Mode	13.27	9.32	381	7.55
	Kurtosis	-0.78	-0.79	5.32	-0.29
	Skewness	0.61	0.62	2.27	0.14
	Minimum	11.33	8.82	378	7.41
	Maximum	15.14	11	5623	7.75
	Count	61	61	61	61

Table 2.6 cont.

System/Year		*Temp C*	*DO (mg/L)*	*Cond (umhols)*	*pH*
Choptank 2017	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
	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
Wicomico ES 2017	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
	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
Choptank 2018	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
	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
Wicomico ES 2018	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
	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

Table 2.6 cont.

System/Year		*Temp C*	*DO (mg/L)*	*Cond (umhols)*	*pH*
Choptank 2019	Mean	14.97	8.20	336.68	7.19
	Standard Error	0.32	0.11	63.97	
	Median	15.85	8.26	155	7.28
	Mode	11.74	8.61	142	7.31
	Kurtosis	-1.55	0.16	11.44	3.58
	Skewness	-0.45	0.24	3.34	-0.05
	Minimum	10.96	6.48	124	6.56
	Maximum	17.72	10.43	2484	8.10
	Count	59	59	59	59
Chester 2019	Mean	15.54	8.38	878.71	7.42
	Standard Error	0.41	0.12	131.98	
	Median	16.04	8.30	398	7.41
	Mode	10.42	9.34	.	7.35
	Kurtosis	-0.69	-0.78	0.74	0.64
	Skewness	-0.50	0.04	1.38	0.62
	Minimum	10.2	6.58	140	7.15
	Maximum	19.7	9.94	3471	7.84
	Count	51	51	51	51

Figure 2-1. Subestuaries sampled for Yellow Perch larval presence-absence studies, 2006-2019. Watersheds of subestuaries sampled in 2019 are highlighted in green and have bolded text labels; watershed delineation is for Maryland only.

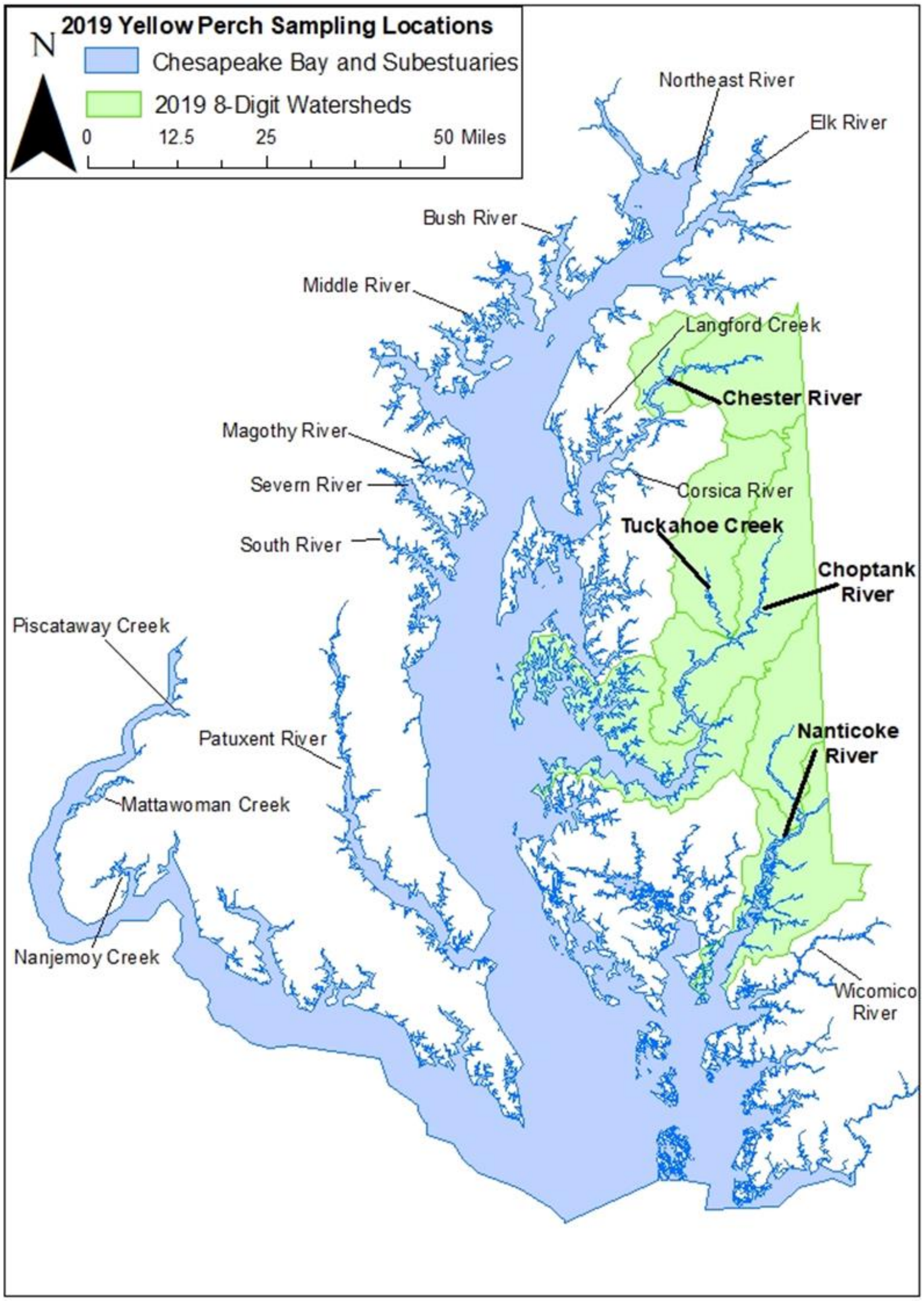


Figure 2-2. Proportion of tows with larval Yellow Perch (*L*p) and its 95% confidence interval in systems studied during 2019. Mean *L*p of brackish tributaries indicated by green triangle, and brackish subestuary *L*p threshold is indicated by dotted line.

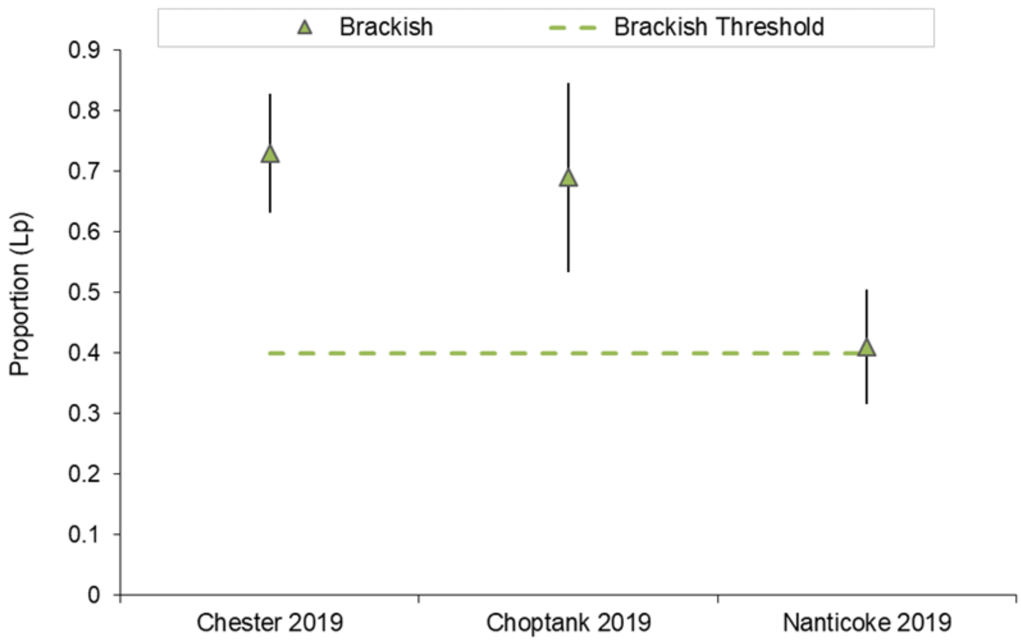


Figure 2-3. Proportion of tows with Yellow Perch larvae (*Lp*) for brackish subestuaries, during 1965-2019. Dotted line provides threshold for persistent poor *Lp* exhibited in developed brackish subestuaries. Dominant land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

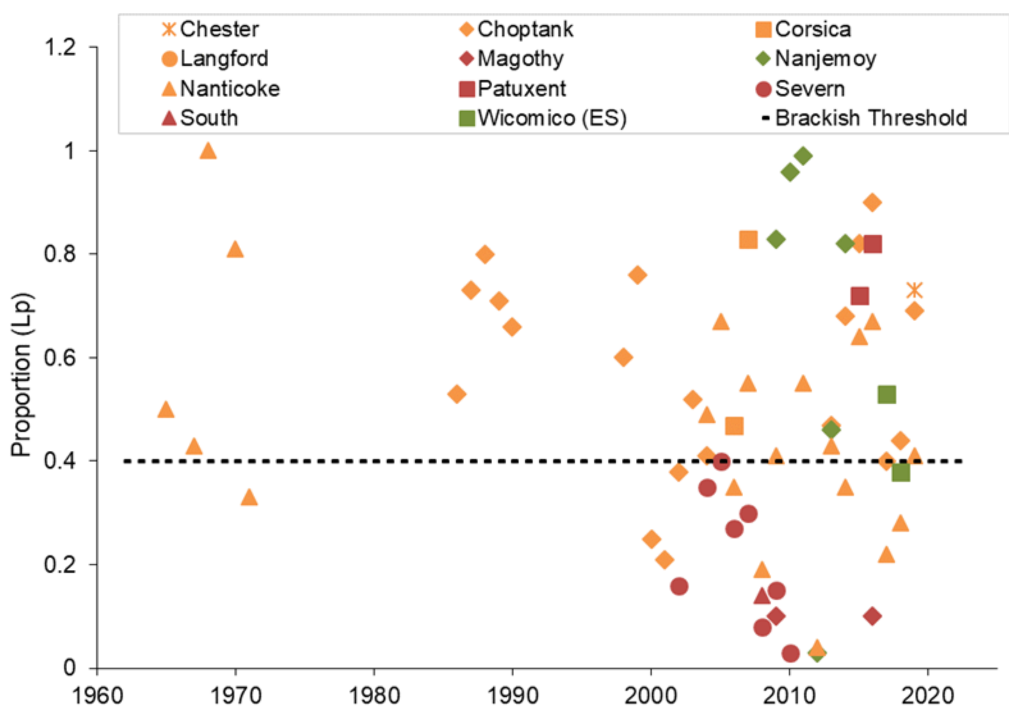


Figure 2-4. Proportion of tows with Yellow Perch larvae (*Lp*) for tidal-fresh subestuaries, during 1990-2019. Dotted line provides reference for consistent poor *Lp* exhibited in a more developed tidal-fresh subestuary (Piscataway Creek). Dominant Department of Planning land use is indicated by symbol color (green = forest and red = urban).

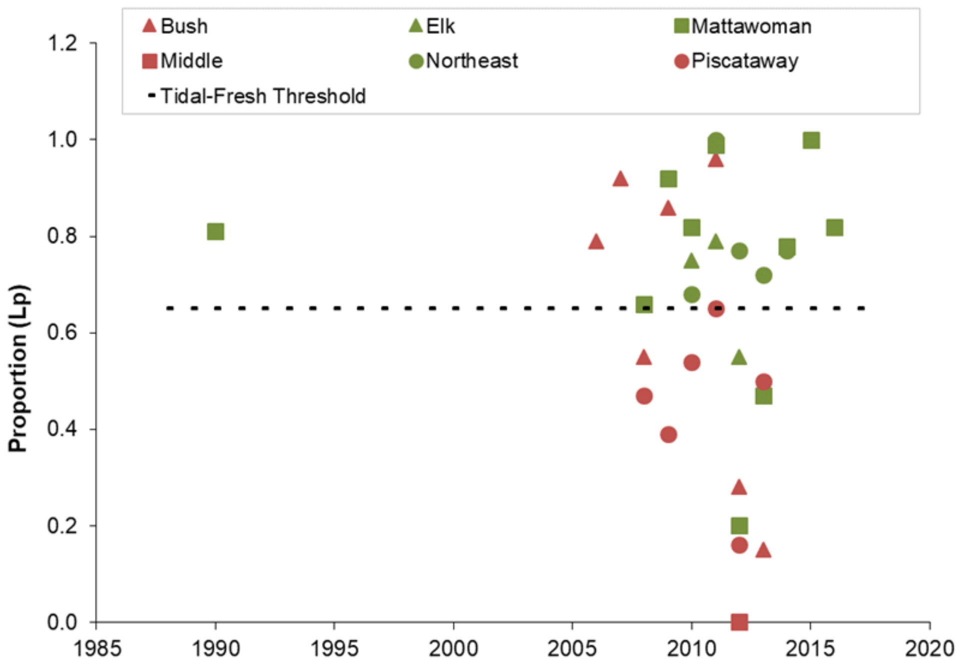


Figure 2-5. Relationship of proportion of plankton tows with Yellow Perch larvae and (A) development (structures per hectare or C/ha), (B) percent agriculture, and (C) percent forest, indicated by multiple regression of fresh and brackish subestuaries combined (prediction = MR) and separate linear regressions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

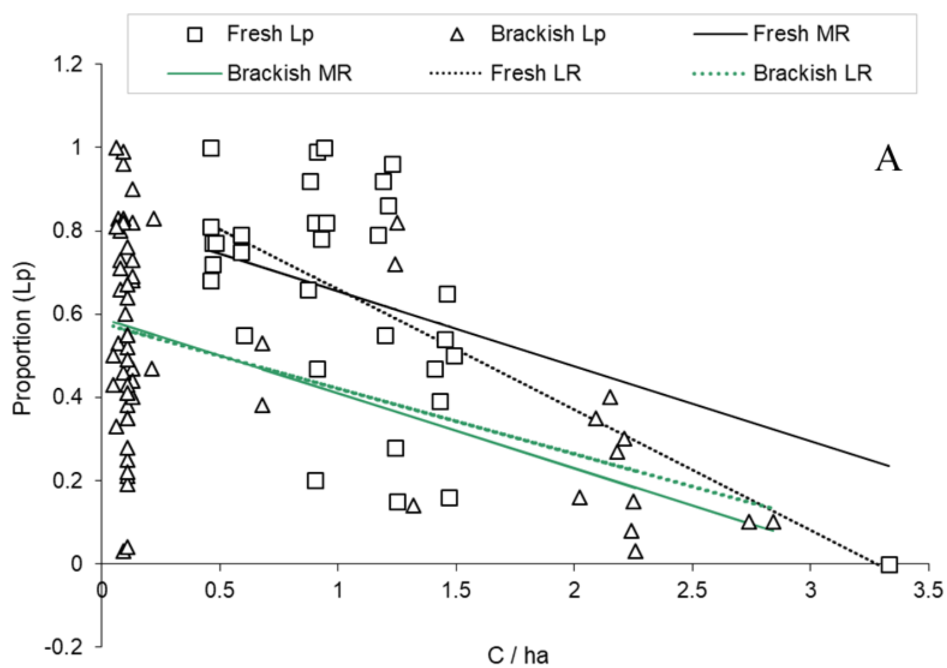
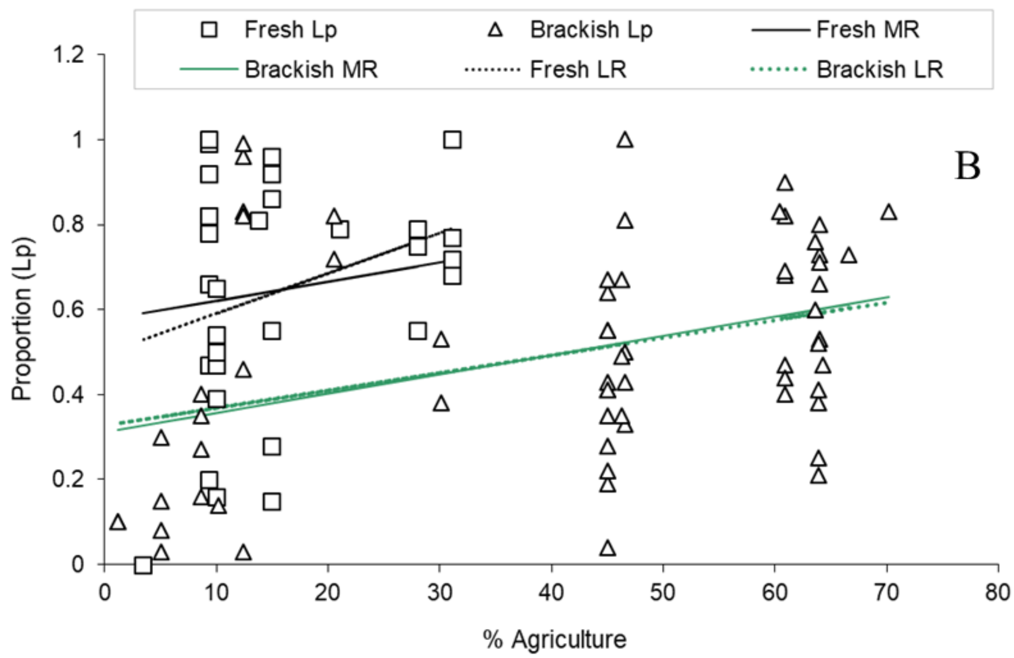


Figure 2-5 cont.



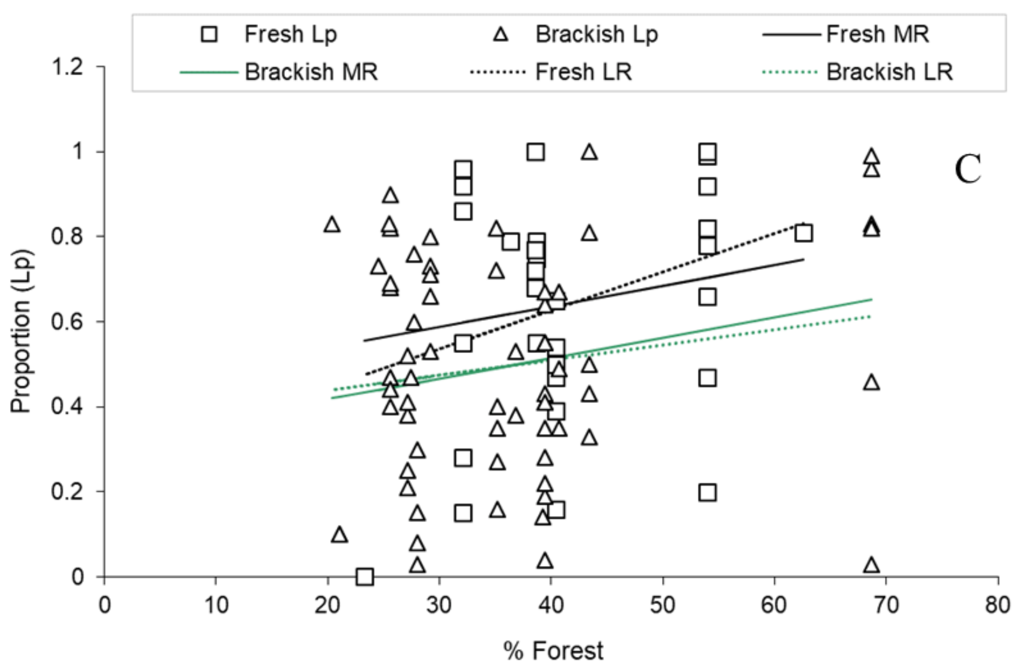


Figure 2-6. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations and salinity class indicated by symbols. Diamonds and a “1” behind land use in the key indicate brackish subestuaries, while squares and a “0” indicate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

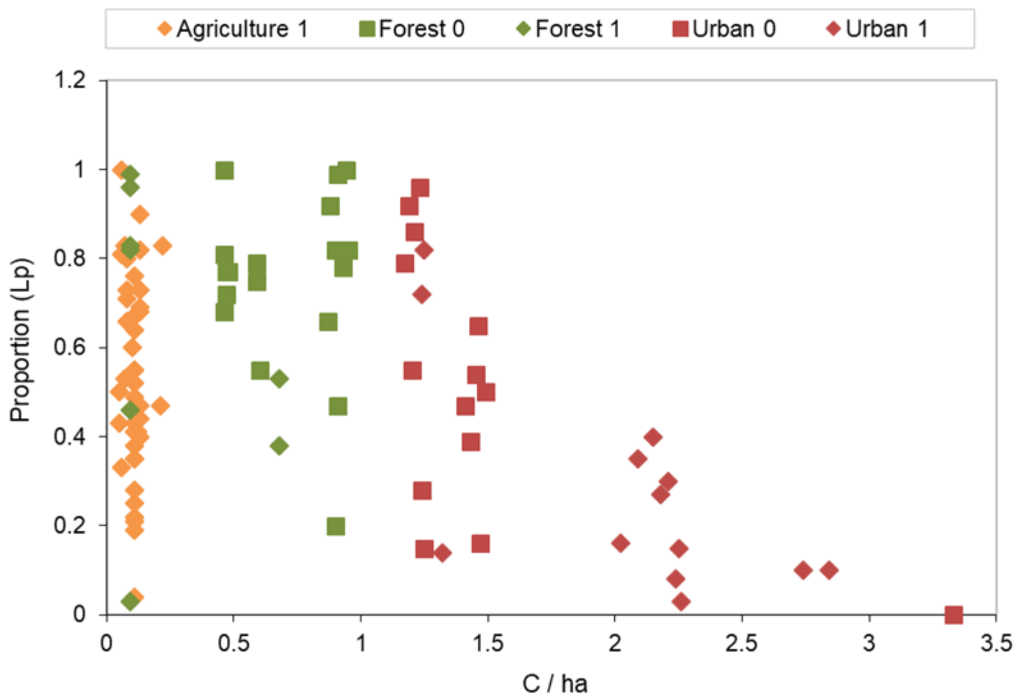


Figure 2-7. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations for large systems.

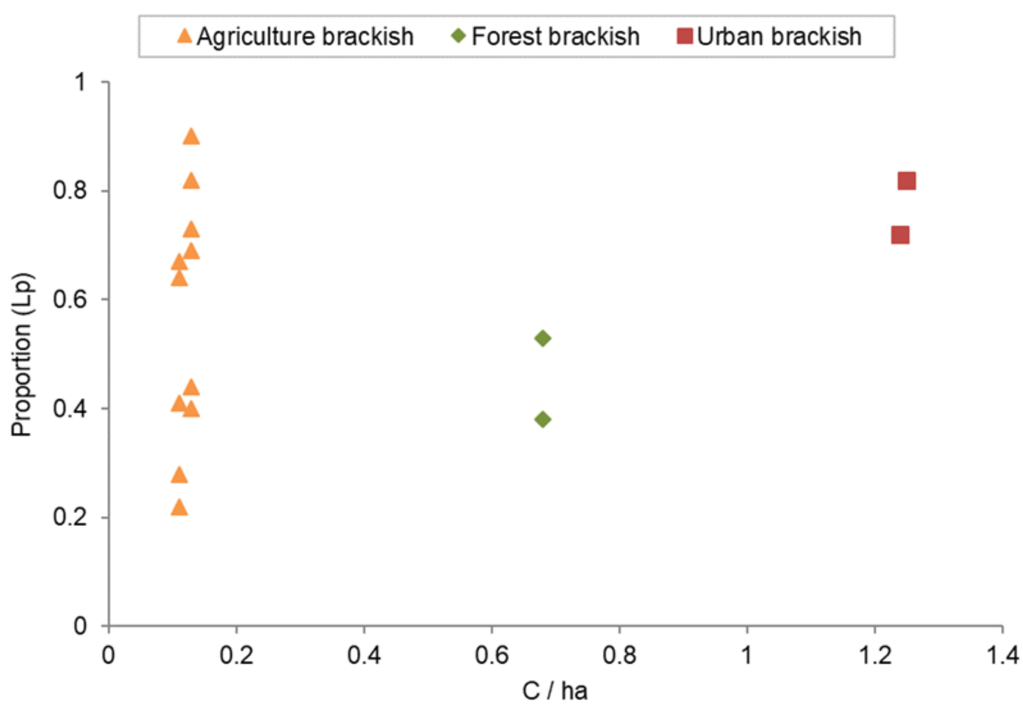


Figure 2-8. Individual values of water temperature, DO, conductivity, and pH, by date sampled, in Choptank and Patuxent rivers during 2015.

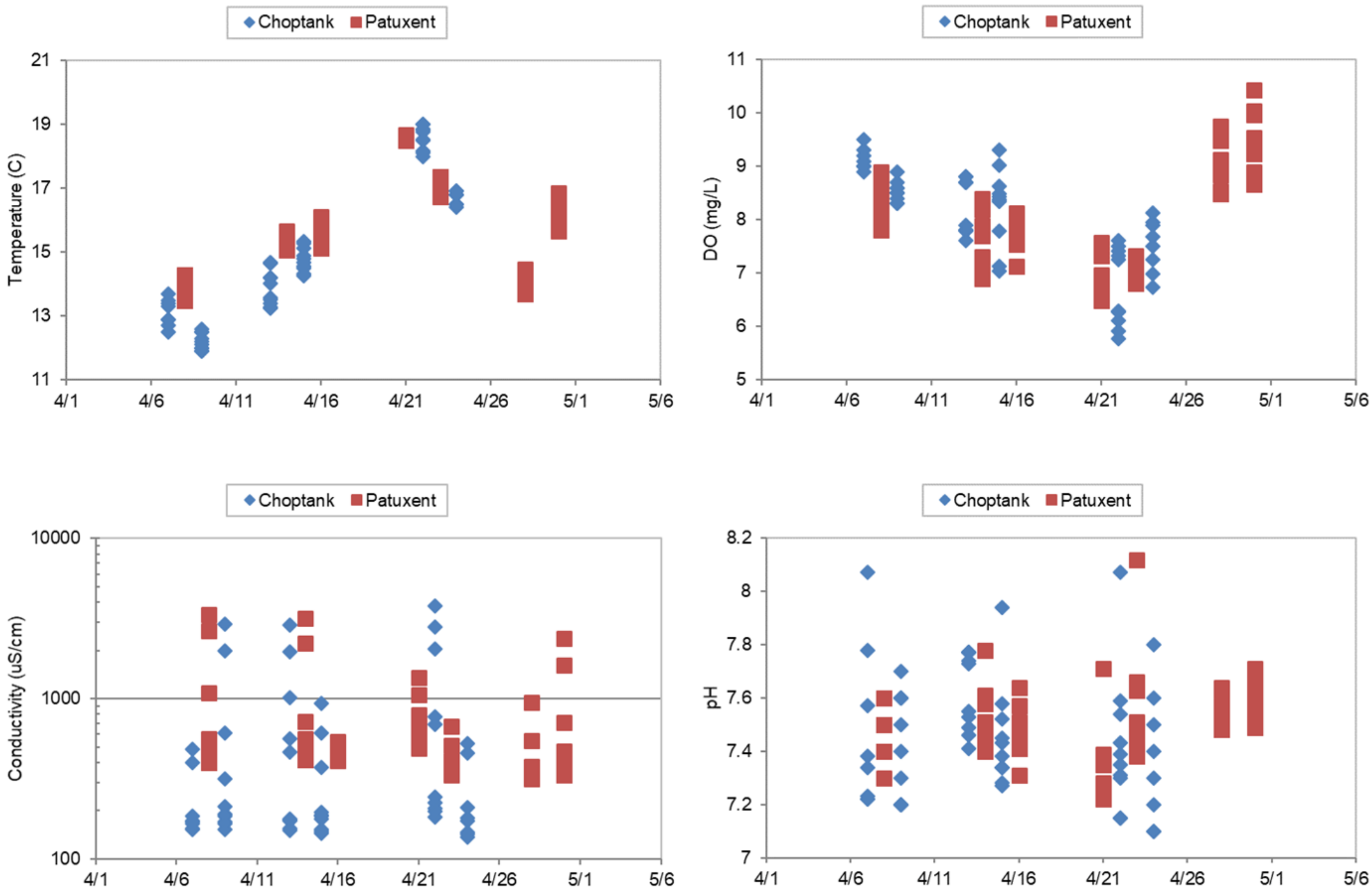


Figure 2-9. Individual values of water temperature, DO, conductivity, and pH, by date sampled, in Choptank and Patuxent rivers during 2016.

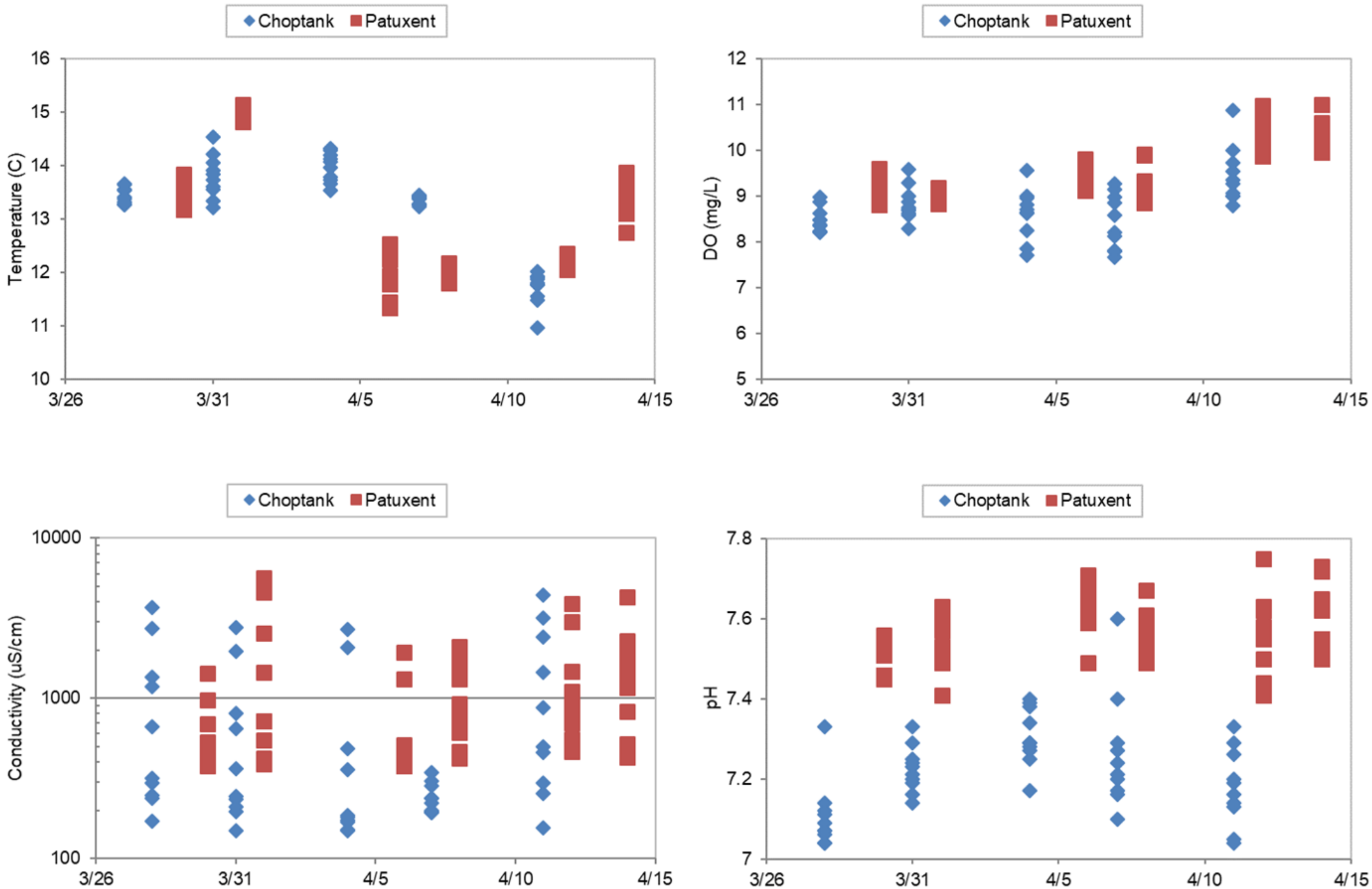


Figure 2-10. Individual values of water temperature, DO, conductivity, and pH, by date sampled, in Choptank and Wicomico (ES) rivers during 2017.

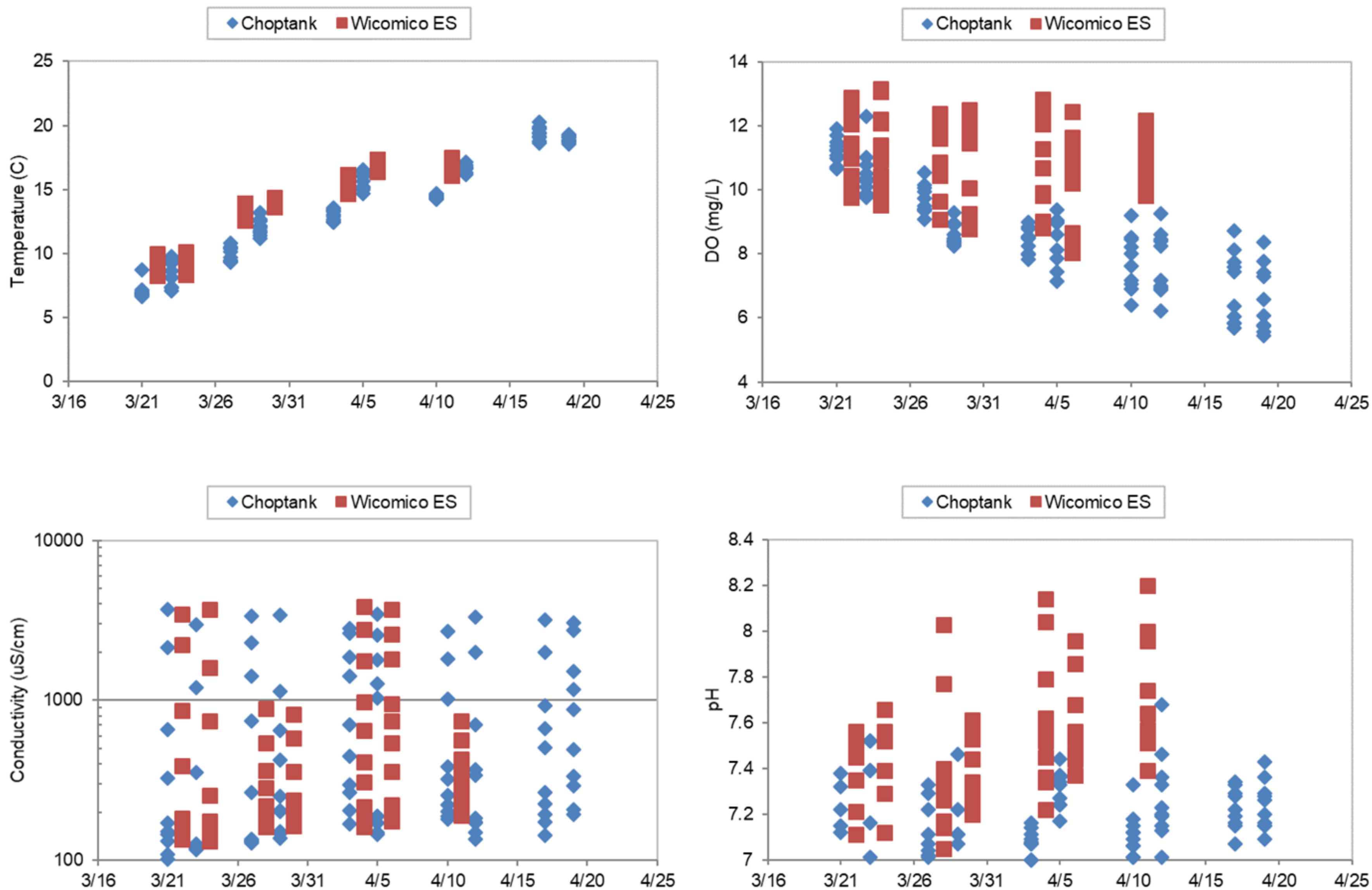


Figure 2-11. Individual values of water temperature, DO, conductivity, and pH, by date sampled, in Choptank and Wicomico (ES) rivers during 2018.

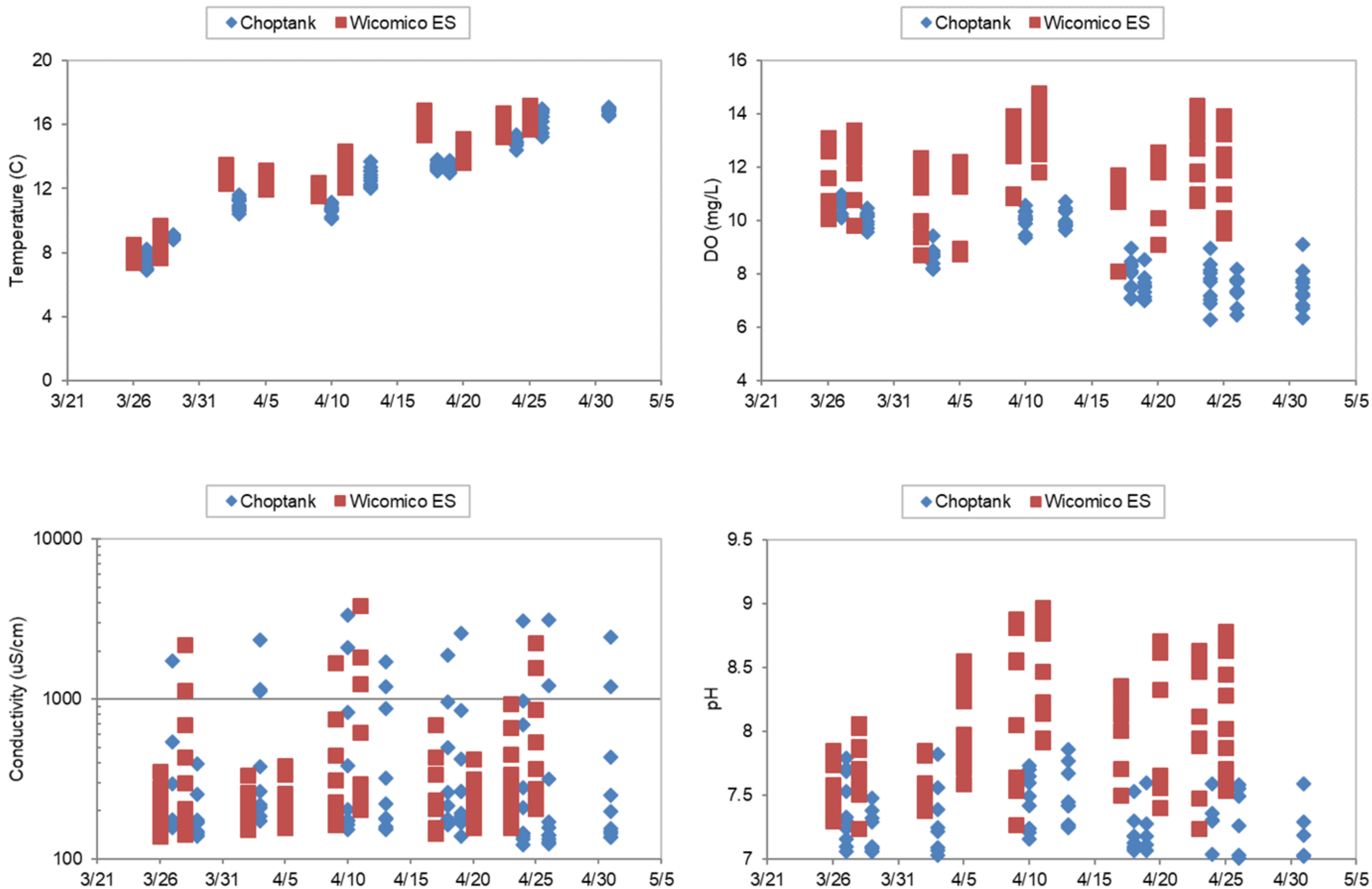


Figure 2-12. Individual values of water temperature, DO, conductivity, and pH, by date sampled, in Choptank and Chester rivers during 2019.

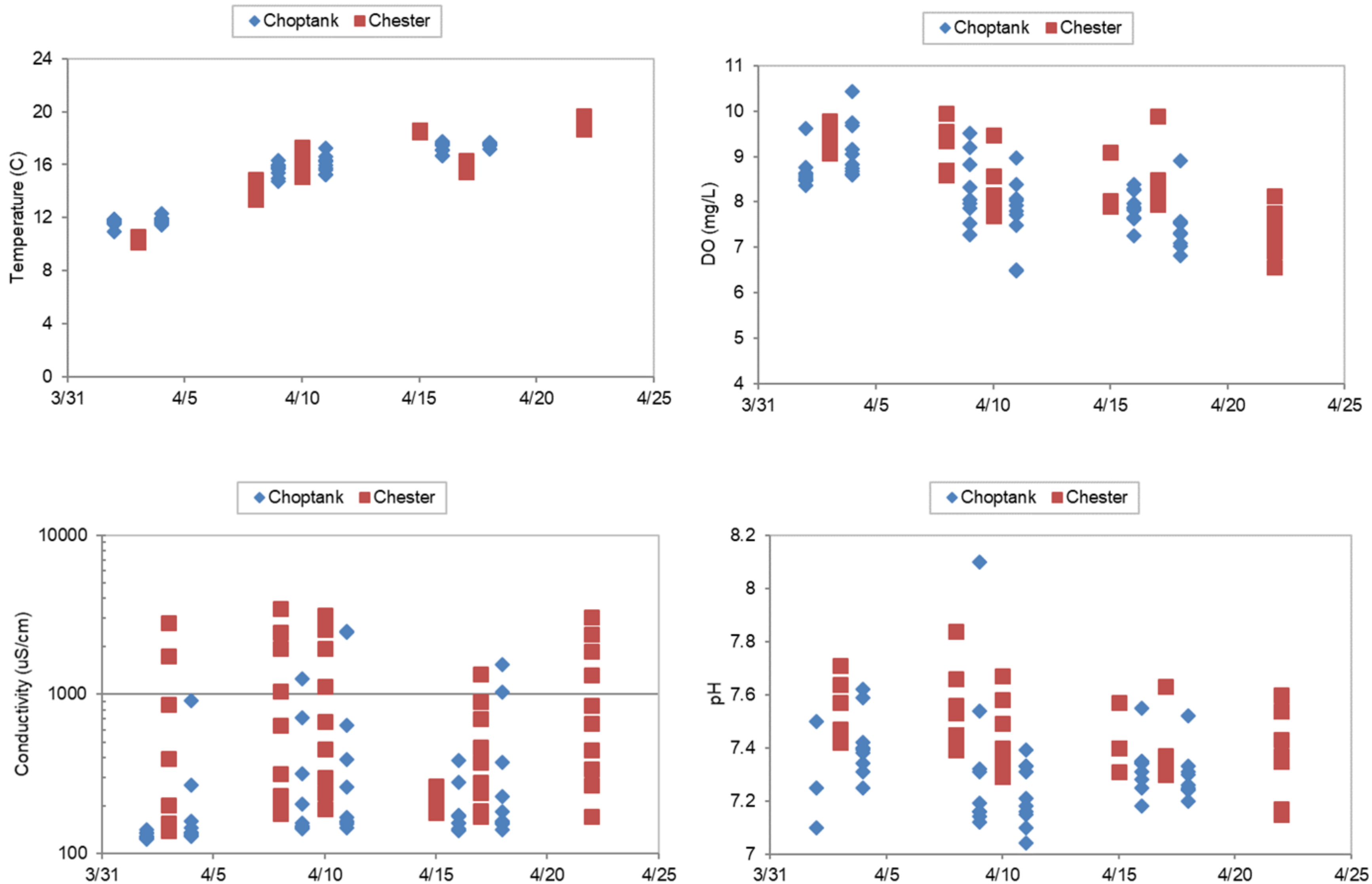


Figure 2-13. Relationship of proportion of plankton tows without detritus (OM0) and development (structures per hectare or C/ha). Dominant Department of Planning land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

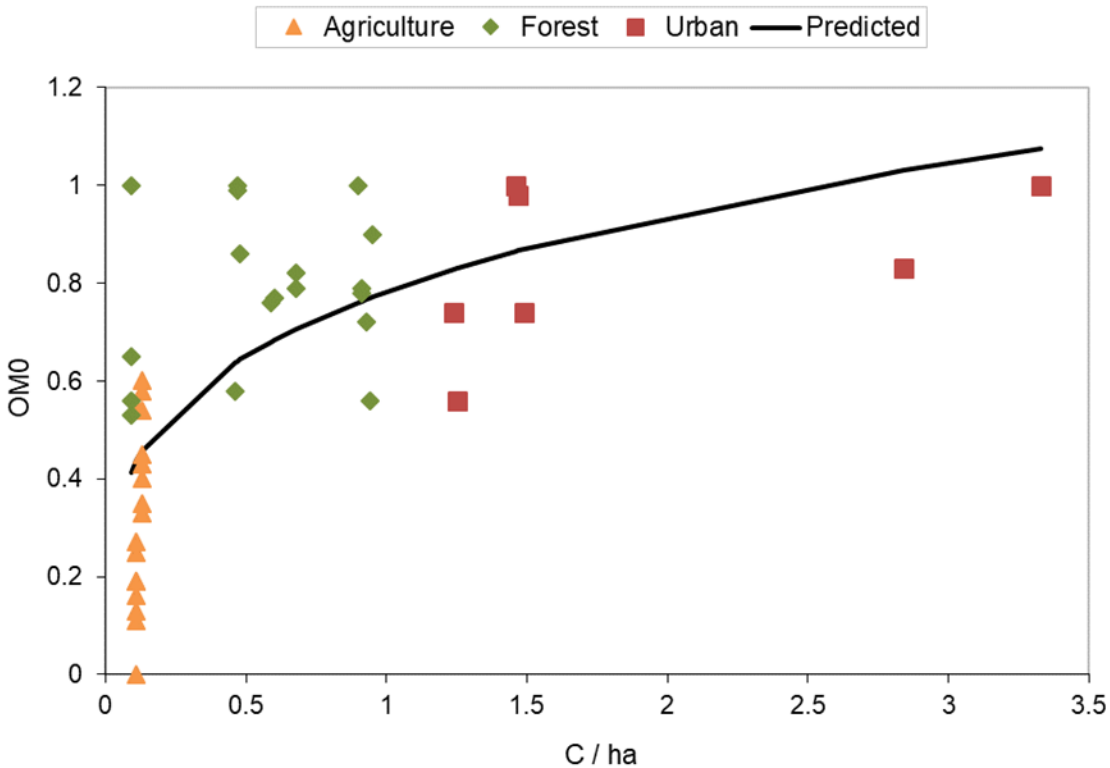
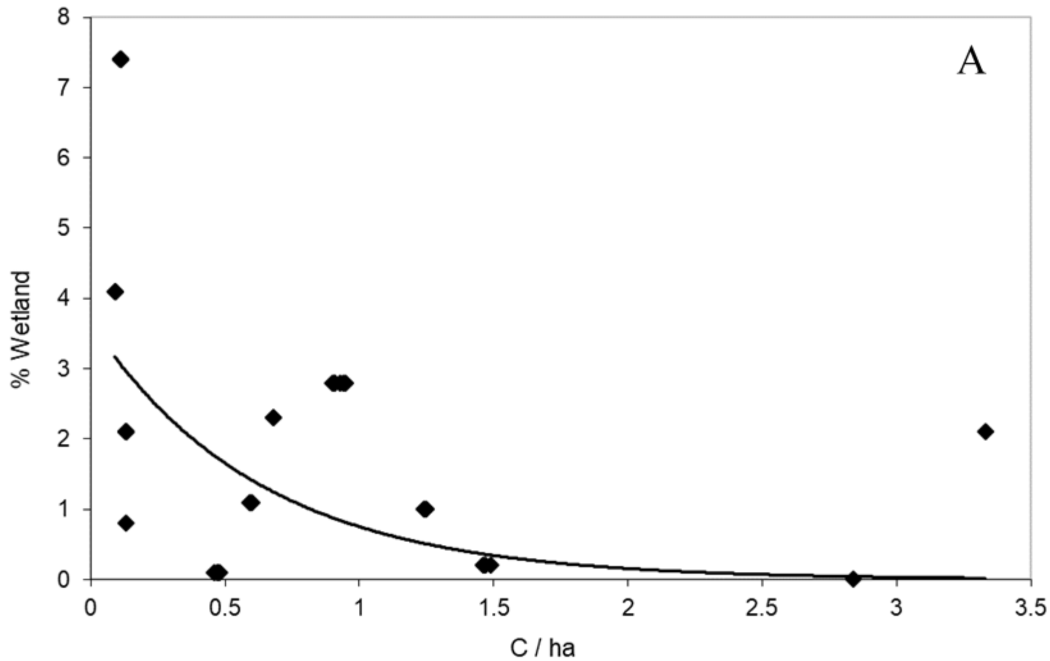
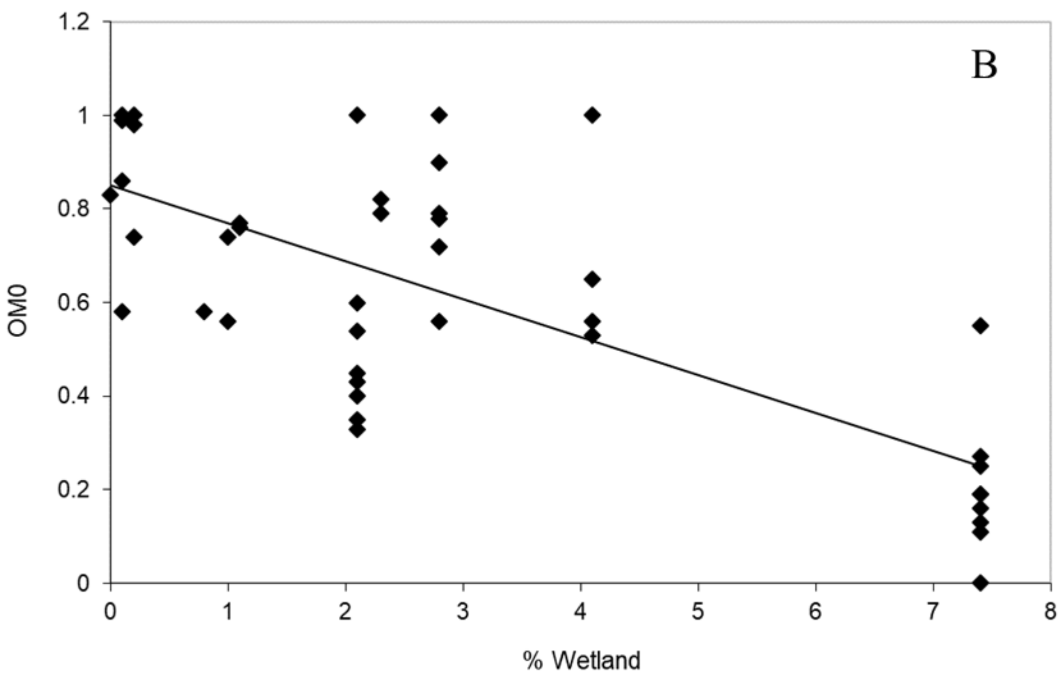


Figure 2-14. (A) Relationship of percent wetlands per watershed and level of development (C/ha). (B) Proportion of samples without organic material (OM0) and percent wetlands per watershed.





Section 2.1: Investigation of Striped Bass spawning and larval habitat status in Maryland

Jim Uphoff, Alexis Park, Carrie Hoover, Margaret McGinty, and Marek Topolski

Executive Summary

In this report we developed techniques from existing data for inexpensively assessing changes in spawning and larval habitat in Maryland's Striped Bass spawning areas based on changes in life stage (juvenile per egg) relative abundance. Striped bass may be sensitive to egg and larval habitat perturbations because the potential year-class is concentrated during early spring in limited fresh-low salinity tidal reaches of 16 Chesapeake Bay tributaries. Year-class success of striped bass is largely determined by the first three weeks of life and is a product of egg abundance and survival through the postlarval stage that is driven by water temperature and flow under normal conditions.

Maryland has measured year-class success of Striped Bass in four major Chesapeake Bay nursery areas with a shore zone seine survey since 1957 (juvenile index or JI). A long time-series of the proportions of ichthyoplankton samples with Striped Bass eggs (*Ep*; 1955-2019), equivalent in length to that of the JI, provided a means of understanding the role of habitat and spawning stock status on recruitment. The ratio of JI to *Ep* was used as an indicator of relative survival of early life stages (RLS) for analyses searching for shifts in RLS through time. Trends in year-class success of Striped Bass were compared with White and Yellow Perch trends, semi-anadromous fish that share a common larval nursery with Striped Bass but have different life histories and fisheries. Comparisons of Striped Bass JI's among areas offered insight on regional similarity of habitat conditions. Since 2014, we have collected basic water quality data (temperature, conductivity, dissolved oxygen or DO, and pH). Historical water quality measurements of sufficient intensity were available from paper records, reports, or old electronic files, making retrospective comparisons possible.

Survival of Striped Bass eggs and larvae, and subsequent recruitment in Maryland's portion of Chesapeake Bay exhibited time blocks of varying productivity during 1957-2019. Baywide *Ep* increased during 1955-1957, was high during 1961- 1979, low during 1982-1988, and recovered to 1961-1979 levels after 1988. The near collapse of the Striped Bass fishery in the 1980s was driven by a shift to low JIs in the early1970s that was followed by a decline in baywide *Ep* a decade later. Year-classes in the top quartile occurred frequently during 1958-1970 (31% of indices) and 1993-2019 (41% of indices). Juvenile indices between these periods were not present in the top quartile and year-classes in the bottom quartile were frequent. Recovery of Striped Bass spawning stock, indicated by high *Ep* after 1988, was complemented by a recovery of egg-larval survival, indicated by RLS, a few years later. Estimates of high RLS have occurred every few years since 1993, with 2006-2010 representing the longest period since 1993 without high RLS. Estimates of RLS indicated periods of fairly consistent higher or lower survival rather than random scattering throughout the time-series indicative of stationary influences on recruitment. Estimates of RLS in the bottom quartile were concentrated in the period spanning 1977-1991, while periods of RLS in the upper quartile occurred during 1961-1970 and 1993-2019.

The criterion in the ASMFC Striped Bass Management Plan for determining recruitment failure in a spawning region (3 consecutive years of lowest quartile juvenile indices) appeared to be an insensitive trigger when applied to the JI or RLS time-series. When applied to the historic time-series as indicators, RLS triggered the criterion earlier than the JI, but both were well after the actual management response. Forming a recruitment trigger around the absence of a strong

year-class for some extended period of time (> 5 years) may lead to a more timely response in tune with the needs of the fishery and its management. Use of *Ep* or SSB from the Striped Bass stock assessment as the denominator for determining RLS or SSB RLS, respectively, produced different depictions of egg-larval survival dynamics and patterns of underlying productivity. Estimates of RLS were depressed during 1982-1992 (SSB was not available earlier) and then shifted upwards. The pattern of SSB RLS during 1982-1989 appeared similar to much of the remaining time-series, indicating stable habitat.

Maryland's Striped Bass JI was well correlated with JIs of White Perch and Yellow Perch. These two estuarine resident species differed enough in life history characteristics and fisheries that they should not have been simultaneously overfished, indicating common larval habitat conditions played a large role in determining their year-class success. Associations among Striped Bass JIs in adjacent spawning areas (Choptank and Nanticoke rivers or Potomac and Patuxent rivers) were moderate to strong, and weak between these two regions and with the Head-of-Bay. Conditions among these major spawning and larval nurseries occasionally aligned favorably, resulting in a strong Striped Bass year-class.

Striped Bass egg presence-absence in three infrequently sampled spawning areas (Patuxent, Wicomico, and Chester rivers) between the 1950s and 2015-2019 did not indicate major changes in spawning stock status in these spawning areas.

Basic water quality data with adequate sample sizes at the spawning and larval nursery spatial and temporal scale were surprisingly sparse. Comparisons of flow, water temperature, conductivity, and pH indicated conditions within Maryland's Striped Bass spawning and larval nursery areas have changed over time, but changes varied among areas. Water quality conditions differed between spawning areas in rural and urbanizing watersheds. Dissolved oxygen during spawning and larval development periods did not fall below the 5 mg / L target for Chesapeake Bay living resources over all the years and spawning areas available.

Long-term (1950s to present), concurrently collected water temperature and egg distribution data suggested that water temperature (21˚C) indicative of the end of spawning and-or poor survival of hatched larvae was occurring earlier in recent years. The scattershot nature of sampling during the 1950s makes this finding tenuous, but we hope to be able to investigate this further through the extensive Nanticoke River time-series.

Choptank River pH offered the clearest indication of habitat change between 1986-1991 and 2014-2019, from largely acidic and highly variable conditions to neutral and more stable (and closer to those cited for productive hatcheries). Acidic conditions in Choptank River surveys during the 1980s were consistent with descriptions of water quality described for in situ and on-site toxicity tests conducted in Choptank and Nanticoke rivers during 1984-1990. Acidic conditions and poor buffering coupled with concurrent elevated metals were associated with low survival of Striped Bass prolarvae during some trials; Potomac River and Head-of-Bay trials did not share these characteristics. Distributions of pH during the 1990s in Nanticoke, Patuxent and Chester rivers' spawning areas were generally in the upper range of those found in the Choptank River during 1986-1991 and exhibited wide variability. During 2014-2019, pH in spawning areas with urbanizing watersheds (Patuxent and Wicomico rivers) generally exhibited higher means and greater variation in measurements than rural watersheds (Choptank and Chester rivers). Patuxent River pH means and ranges appeared to change little between 1991 and 2015, while pH means increased and range contracted in Chester River between 1996 and 2019.

Conductivity distributions in spawning areas with urban watersheds exhibited higher minimums than spawning areas in rural watershed during 2014-2019. Minimum conductivity in the Patuxent River spawning area increased by a factor of 2.2-2.4 between 1991 and 2015-2016. Wicomico River minimum conductivity was 1.4-2.3 times higher than nearby Choptank River or Nanticoke River. Choptank River spawning area conductivity summaries offered little indication of change between 1986-1991 and 2014-2019. Minimum conductivity in Chester River was about 40% higher in 2019 than in 1996. Elevated salt levels by themselves in the upper spawning area should not be an issue for Striped Bass larvae since they can be abundant in higher conductivity regions further downstream where freshwater is more mixed with intruding saltwater. However, elevated conductivity could indicate other chemical stressors have increased as well.

Long-term climate patterns, long-term climate warming, deterioration and improvement in acidic deposition, concurrent increases in freshwater salinization and alkalinization (salinization syndrome), increasing addition of a suburbs to the Chesapeake Bay watershed, a shift to conservation agriculture, and watershed management practices associated with the Chesapeake Bay Program could have resulted in detrimental or beneficial larval habitat changes. It is likely that combinations of these factors have shifted from period to period.

Introduction

The Atlantic States Marine Fisheries Commission (ASMFC) has determined that Atlantic coast Striped Bass are overfished (defined in this report as fishing that has caused a reduction in spawning stock beyond a point where recruitment is reduced) based on its most current stock assessment (ASMFC 2019). This finding has generated concern in the angling community. Although much of this concern has focused on the abundance of spawning stock, there has been unease expressed about degradation of Striped Bass spawning and larval nursery habitat in Chesapeake Bay (J. Uphoff, personal observation). In this report we develop techniques for assessing changes in spawning and larval habitat to address these concerns.

Striped bass are anadromous, long-lived, late maturing, highly fecund, and exhibit complex migrations (by age and sex; Boreman and Lewis 1987; Rago and Goodyear 1987; Rago 1992; Dorazio et al. 1994; Secor and Piccoli 2007; Maryland Sea Grant 2009; Secor et al. 2020). Population dynamics of striped bass are driven by dominant year-classes that strongly reflect environmental conditions encountered by eggs and larvae: longevity (in the absence of heavy fishing) ensures that these strong year-classes will reproduce over many years and dampen the effects of environmental variation (Rago and Goodyear 1987; Rago 1992; Richards and Rago 1999; Maryland Sea Grant 2009). Production from Chesapeake Bay (or Bay) spawning areas has been estimated to account for up to 90% of landings along the entire Atlantic Coast (Richards and Rago 1999). On an egg production basis, Maryland's spawning areas were estimated to produce approximately 69% of the Chesapeake Bay total (Uphoff 2008).

Striped bass may be sensitive to egg and larval habitat perturbations in spring because this spawning and nursery habitat is concentrated in limited reaches of 16 Bay tributaries (Hollis et al. 1967; Grant and Olney 1991; Schaaf et al. 1993; Uphoff 2008; Maryland Sea Grant 2009). Striped bass spawning and larval nursery areas are located in the fresh-low salinity tidal reaches within the coastal plain and the estuarine turbidity maximum is particularly important (North and Houde 2003; Maryland Sea Grant 2009). Year-class success of striped bass is largely determined by the first three weeks of life and is a product of egg abundance and survival through the postlarval stage (Uphoff 1989; 1993; Houde 1996; Maryland Sea Grant 2009). Generally, these differences in survival rates of the early life stages are believed to result from environmental factors, particularly temperature and freshwater flow (Maryland Sea Grant 2009;

Martino and Houde 2010; Millette et al. 2020). Temperature could impact recruitment through direct mortality of eggs and larvae due to lethally low or high temperatures; and-or via its influence on the timing of zooplankton blooms (match-mismatch hypothesis), while flow could also influence zooplankton dynamics, nursery volume, water quality and toxicity of contaminants (Uphoff 1989; 1992; Secor and Houde 1995; Maryland Sea Grant 2009; Martino and Houde 2010; Secor et al. 2017; Millette et al. 2020). Relationships and associations of Chesapeake Bay tributary flow to striped bass early life stage survival and year-class success have been explored and both positive and negative relationships and associations have been detected (Kernehan et al. 1981; Uphoff 1989; Uphoff 1992; Rutherford et al. 1997; Martino and Houde 2010; Millette et al. 2020).

Maryland has measured year-class success of Striped Bass in four major Chesapeake Bay nursery areas (Head-of-Bay, Potomac River, Nanticoke River, and Choptank River) with a shore zone seine survey since 1954 (Figure 2.1.1; Hollis 1967; Maryland Sea Grant 2009; Durell and Weedon 2019) and the baywide juvenile index has proven to be a reliable indicator of recruitment to Atlantic coast fisheries (Schaefer 1972; Goodyear 1985a; Richards and Rago 1999; Maryland Sea Grant 2009). Between 1954 and 1970, strong year-classes were produced in Chesapeake Bay every 2-4 years (Richards and Rago 1999). Strong year-classes failed to appear after 1970 and year-class success declined steadily into the early 1980s and generally remained depressed until after 1988. A pattern of strong year-classes appearing every few years returned in 1993 and has continued to the present (Figure 2.1.1).

Synthesis of Striped Bass related contaminants and water quality studies, funded under the Emergency Striped Bass Act and conducted during 1984-1990, suggested that water quality problems in some Chesapeake Bay spawning rivers contributed to declining year-class success, but were not the sole problem (Hall et al. 1993; Richards and Rago 1999). Toxic water quality conditions (low conductivity, alkalinity, hardness, and pH, and high levels of trace metals) and low water temperatures (< 12 °C) encountered by striped bass larvae were implicated in episodic mortalities in some tributaries in the 1980s (Uphoff 1989; 1992; Hall et al. 1993; Richards and Rago 1999). Unfortunately, retrospective data on factors identified in experiments was insufficient to address whether conditions in the spawning rivers had changed (Richards and Rago 1999).

Simulations indicated that decreased survival of Chesapeake Bay Striped Bass due to excessive larval mortality or overfishing could have reduced the population in the 1970s (Goodyear 1985b; Richards and Rago 1999). Available fishing mortality estimates in the 1970s were very high and in 1985 states imposed moratoria or much more conservative fishing restrictions (Richards and Rago 1999; Maryland Sea Grant 2009). Overfishing may have acted synergistically with increased larval mortality (Richards and Rago 1999). In either case, reduction of fishing mortality was a viable management strategy for restoring the stock (Goodyear 1985b) and recovery was largely attributed to reduced fishing mortality (Field 1997; Richards and Rago 1999).

Detecting changes in first year survival of Striped Bass was difficult in the presence of excessive fishing mortality (Rago 1992). Duration of studies to detect changes in first year survival attributable environmental conditions also has an impact; longer studies generally have a better chance (Rago 1992). The longest Striped Bass ichthyoplankton survey of early life stage survival, 1980-1989, was conducted in Choptank River, Maryland (Uphoff 1989; 1992; 1993) and concluded that temperature and water quality operated independently in Choptank River. Egg-prolarval survival was reflective of water temperature and postlarval mortality was associated with water quality conditions (low pH and conductivity) that would have influenced toxicity of metals (Uphoff 1989; 1992; 1993). Poor conditions at either or both stages produced a poor year-class, while optimal conditions were needed at both stages for a strong year-class (Uphoff 1989; 1992). Instantaneous daily mortality rates of postlarvae shifted from higher (0.096-0.222) during 1980-1984 to lower (0.065-0.110) during 1985-1989, while egg-prolarvae survival varied without trend (Uphoff 1993). Secor et al. (2017) generally did not observe low pH and high concentrations of metals during larval cohort mark-capture experiments in the Nanticoke River during 1992-1993. However, the disappearance of a larval cohort in 1993 after a storm and rainfall event was followed by a sudden drop in temperature and pH. Associated elevated toxicity due to metals was a viable explanation, as was loss through downstream advection (Secor et al. 2017).

If a period of environmental forcing is longer than the period of internal population dynamics, there will be periods when a stock may appear overfished even though lower productivity is the underlying cause (Szuwalski et al. 2014). Long-term patterns of the Maryland Striped Bass juvenile index and spawning stock could shed light on whether the decline in recruitment during the 1970s and 1980s was concurrent with or preceded by a decline in spawning stock, or whether spawning stock declined before recruitment. This in turn, would give insight on the importance of habitat (environmental) conditions on the dynamics of the decline and recovery of one of the Atlantic coast’s premier gamefish. It also provides a basis for using techniques developed to evaluate overfishing and spawning habitat hypotheses in current management.

Atlantic coast Striped Bass spawning stock biomass (SSB; MT) has been estimated as part of the Striped Bass stock assessment since 1982 (NEFSC 2019) and is management’s preferred estimate of spawning stock condition. This estimate contains the Delaware River and Hudson River stocks, but SSB is dominated by the Chesapeake Bay stock. Current management uses SSB targets and limits that reflect the status of the stock in 1995 when the stock was considered recovered (NEFSC 2019). The SSB time-series does not extend back far enough to address whether the JI decline in the 1970s was an immediate response to low spawning stock or whether a decline in the JI preceded a decline in spawning stock.

Uphoff (1993; 1997) used historical data to develop Striped Bass egg presence-absence (*Ep* or proportion of samples with eggs) as an indicator of spawning stock status in six Maryland spawning areas during 1955-1995. Striped bass ichthyoplankton surveys have collected eggs in Maryland’s Chesapeake Bay spawning areas (defined here as Maryland's portion of Chesapeake Bay) since the 1950s to delineate spawning areas, assess power plant impacts, identify habitat problems and their impact on recruitment, describe egg and larval population dynamics, and determine temporal and spatial distribution of eggs and larvae (Uphoff 1997). In general, proportion of positive samples provides an estimate of habitat occupation based on encounter rate that is readily related to the probability of detecting a population; higher encounter rates are related to higher population densities (Strayer 1999). Trends in *Ep* matched perceived changes in the Chesapeake Bay Striped Bass population (Uphoff 1993; 1997) and provided an estimate that could be referenced to a period of time when stocks were considered to be at a desirable or undesirable level (Uphoff 1993; 1997). In addition to *Ep* estimates in Uphoff (1997), a time-series has largely been maintained through 2019 (described in Methods).

Striped bass eggs are semibuoyant, pelagic, distributed throughout the water column, and do not have an escape response (Bulak et al. 1993; Uphoff 1993), making them ideally suited for collection with plankton nets. The eggs are easily identified in the field because of their large size (3-4 mm), thin transparent egg membrane, large perivitelline space, and single distinctively large oil globule (Uphoff 1997). Net diameter did not have a discernable effect on estimating *Ep* and longer tow durations with single nets did not result in consistently higher *Ep* during years where more than one tow time was employed among different spawning areas (Uphoff 1997). Estimation of Striped Bass stock status using presence-absence sampling was relatively simple and straightforward (Uphoff 1993; 1997). The normal distribution approximation of the binomial distribution could be used to describe stock status under the range of observed stock sizes. The location and extent of spawning areas were well defined, so few samples were "wasted" defining unsuitable habitat, and, because Striped Bass eggs were nonmotile, *Ep* could be interpreted as reflecting abundance. It was not necessary to adjust for extrusion, daily differences in spawning activity, and mortality that are required for estimates of egg production from egg surveys (Uphoff 1993).

As a step towards ecosystem-based fisheries management, a team of experts was assembled in the late 2000s to develop background documents for a broad array of issues that will confront current and future managers of Chesapeake Bay Striped Bass. The *Striped Bass Species Team Background and Issue Brief* (Maryland Sea Grant 2009) was developed as a guide to the array of ecological problems that managers may face as Striped Bass management's focus moves beyond managing fishers as its fundamental response to nearly all issues. Multiple issue statements involved Striped Bass spawning and larval nursery habitat: climate change, flow, contaminants, and watershed development. Striped Bass *Ep* and the JI were listed as indicators for some of these issues (Maryland Sea Grant 2009).

The long time-series of *Ep*, equivalent to that of the JI, can provide a means of understanding the role of spawning stock status on recruitment. A linear regression of Choptank River JI against *Ep* during 1980-1989 (using estimated survival of larvae as a weight for *Ep*) indicated that spawning stock had an influence on recruitment; there was strong contrast in *Ep* for this analysis (Uphoff 1993). Uphoff (1997) used *Ep* and spawning area specific juvenile indices in a stock-recruitment table to describe the influence of spawning stock on recruitment in Maryland's major Striped Bass spawning areas. An underlying assumption of Uphoff (1997) was that the average stock-recruit relationship did not change over time; analyses to detect changes were recommended (Uphoff 1997).

The ratio of JI to *Ep* has been proposed as an indicator of relative survival of early life stages for retrospective analyses searching for nonstationary patterns (Uphoff 2008; Maryland Sea Grant 2009). Patterns in this ratio can provide an inexpensive indication of changes in egg and larval habitat conditions without specification of the myriad factors (water quality, food availability, water temperature, etc.) that are needed to determine habitat suitability. These latter data are not commonly collected from the spawning and nursery areas. Comparing trends in year-class success of Striped Bass with other anadromous fish that share a common larval nursery with Striped Bass, but have different life histories and fisheries, may be useful for exploring whether early life stage habitat conditions or fishing influenced population dynamics of Striped Bass. Comparisons of Striped Bass JI's among areas could supply insight on regional similarity of habitat conditions.

Striped Bass along the Atlantic Coast are managed under Amendment 6 to the Interstate Striped Bass Management Plan (ASMFC 2003) and this plan specifies monitoring of recruitment success (juvenile indices) in seven regions of the Atlantic coast (including Maryland Chesapeake Bay spawning areas). Recruitment failure of any area-specific is defined as a juvenile index lower than 75% of all other values in the dataset for three consecutive years (ASMFC 2003). We compared how a management response might be triggered by using the JI, the ratio of JI to *Ep*, or the ratio of the JI to SSB as an indicator of habitat stress in Maryland's Striped Bass spawning areas. The actual response of management to the determination of a recruitment problem described in Richards and Rago (1999; initiation of the Emergency Striped Bass Study in 1979 was considered the first response) was contrasted with the results from the three indicators.

Striped bass spawn in the large subestuaries that we have sampled for yellow perch larvae under F-63 during 2013-2019 (Nanticoke, Choptank, Chester, Patuxent, and Wicomico rivers). Striped Bass eggs and Yellow Perch larvae were collected concurrently during several weeks, but Yellow Perch larvae outgrow efficient sampling by our 0.5 m plankton nets before significant Striped Bass spawning is over. We extended sampling in these systems during 2013-2019 to meet criteria for estimating *Ep*. Since 2014, we have collected basic water quality data (temperature, conductivity, dissolved oxygen or DO, and pH) by date and station. In addition, MD DNR has a history of conducting or sponsoring surveys of Striped Bass spawning and larval nursery habitat. Basic water quality measurements were or may be available from paper records, reports, or old electronic files, making retrospective comparisons possible. Water temperature and salinity were routinely collected, while other water quality parameters of interest were collected in some years and locations. Current and past sampling provide multiple paths for exploring water quality issues in Striped Bass spawning and larval nursery areas. Choptank River allowed for long-term comparisons of flow, water temperature, conductivity, and pH during 1986-1991 and 2014-2019 to examine whether conditions have changed or remained relatively constant. Water quality conditions can be compared between spawning areas in rural (Nanticoke, Choptank, and Chester rivers) and urbanizing watersheds (Patuxent and Wicomico rivers). Extent of spawning habitat in spawning rivers and changes in timing or duration that have been sampled infrequently can be confirmed (Wicomico and Chester rivers). Long-term temperature data could provide insight on whether timing of spawning has been impacted by global warming. Data on current and past spawning can be provided for county comprehensive development plans. This portion of Job 1, Section 2 describes progress to date on addressing these issues.

Methods

Study area **-** Maryland's portion of Chesapeake Bay contains 11 Striped Bass spawning areas, comprising an estimated 57,448 ha (Figure 2.1.2; Hollis et al. 1967). They range in size from 23 to 27,225 ha. Two spawning areas, Patuxent and Potomac rivers are located to the west of the Susquehanna River, the Head-of-Bay spawning area is to the south, and the remaining spawning areas are to the east. The two largest spawning areas, Head-of-Bay (27,225 ha) and Potomac River (22,162 ha), dwarf the remaining spawning areas in Maryland (3,034 ha or less; Table 2.1.1). The entire Chesapeake Bay has a surface area of $1.16 \cdot 10^6$ ha.

Proportion of positive tows (Ep) – Previously analyzed Striped Bass ichthyoplankton surveys (1955-1995; summarized in Tables 1 and 2 in Uphoff 1997) were used as a starting basis for the *Ep* time-series. Corrections were made to these data and studies conducted since 1995 were added to this time-series.

With the exception of the Chester River during 1996 (Burton et al. 1996), all Striped Bass ichthyoplankton sampling after 1995 was conducted by MD DNR to estimate *Ep*. A full sampling program for *Ep* was conducted during 1996-2003 under Federal Aid to Sportfishing (Investigations of Striped Bass reports F42-R-8 to F42-R-16) using protocols developed in Uphoff (1993; 1997). This monitoring was suspended under this grant after 2003, but was continued by other projects for presence-absence of Yellow Perch larvae (see **Section 2:**

Estuarine Yellow Perch Larval Presence-Absence Sampling); Striped Bass egg presence-absence continued to be recorded. Striped Bass egg presence-absence was recorded in the Choptank River in 1994, Elk River during 1995-1996, and Nanticoke River during 2004-2019. These collections were made only during April in the Nanticoke River due to the need for staff to transfer to other monitoring projects in May.

We sampled the Choptank River under F-63 during 2013-2019, Patuxent River in 2015-2016, Wicomico River in 2017-2018, and Chester River in 2019 (Figure 2.1.2). These surveys were added into the time-series.

Spawning areas sampled during 2014-2019 were divided into 1.61-km (1-mile) segments and 10 distinct segments were sampled once a trip (single tow at each). Choptank River and Nanticoke River sampling areas were large and a stratified random design was employed (Uphoff 1997). Ten sites were drawn from 21 potential sites in the Choptank River spawning area (Figure 2.1.3) during 1997-2004 and 20 during 2013-2019 (the uppermost station in Tuckahoe Creek was dropped). There were 18 sites in the Nanticoke River spawning area (Figure 2.1.3). These spawning areas were divided into upriver, midriver, and lower river subareas containing 5-6 segments and segments were selected randomly in proportion to subarea size. Tributary spawning areas of the Choptank (Tuckahoe Creek) and Nanticoke River (Marshyhope Creek) were also subareas and had an additional 4 (reduced to 3 after 2004) and 2 segments, respectively. The smaller Patuxent River, Wicomico River, and Chester River spawning areas each had 12 segments (Figure 2.1.3) and 10 stations were randomly selected on each day sampled. Sample trips were usually made twice per week, spaced 2-3 days apart (barring poor weather or equipment breakdowns). Basic water quality (water temperature, pH, conductivity, salinity, and DO) was measured at the surface at each site; pH was not measured in the Nanticoke (the meter used did not measure it). Sampling was conducted until the 21˚C water temperature cutoff criterion was met (Uphoff 1993; 1997) or was likely to be met based on whether water temperature and forecast air temperatures made hitting the cutoff very likely in the interim before the next scheduled sampling visit. In some years, persistent cool temperatures during late spring sometimes did not allow water temperatures to rise above 21˚ C even though egg catches tapered off greatly and a judgement was made to discontinue sampling before the cutoff was reached.

A 2.0‰ salinity cut-off for sampling a site with replacement by lower salinity sites was applied (Uphoff 1997). More than 99% of Striped Bass eggs collected (and counted) in Choptank River during 1980-1985 were collected at 2.0‰ salinity or less (Uphoff 1989). However, high salinity sites in Elk River during 1995-1996 and Nanticoke River after 1995 were not replaced with low salinity sites when *Ep* sampling was piggybacked onto other activities.

Two minute tows were made against the current at the surface with a 0.5-m plankton net made of 0.5 mm Nitex mesh and a 3:1 length-to-mouth diameter ratio. If eggs were readily seen in a sample during or after processing, the sample was discarded, and presence of eggs was recorded. If a sample was fully rinsed and the sampler was confident that eggs were absent, the jar was discarded and absence of eggs was recorded. In these cases, the net was rinsed thoroughly without a jar before taking the next sample. If a sample had been completely processed and the sampler was unsure if eggs were present or not, the sample was preserved in 5-10% buffered formalin and rose bengal stain was added to aid detection. Samples were processed in the laboratory.

The proportion of tows with one or more eggs and its 90% confidence interval was calculated using the normal distribution to approximate the binomial probability distribution (Ott

1977). This approximation can be used when the sample size is greater than or equal to 5 divided by the smaller of the proportion of positive or zero tows (Ott 1977). We restricted analysis to collection dates between the first sample containing an egg and when water temperature reached 21°C. Sites with salinity greater than 2.0‰ and stations past outer boundaries where eggs were not collected during an entire season were excluded to minimize zeros representing non-spawning periods or unsuitable habitat (Uphoff 1993). A station or stations where eggs were not collected located between stations where eggs were present were included in analyses. In cases where cool temperatures persisted, we calculated overall mean *Ep* for all dates sampled, recalculated each mean (*j*) with each sample date (*i*) excluded, Ep_{ji}, and then examined the distribution of Ep_{ji} to judge influence of a single date. A late sample date that represented an outlier was expected to noticeably depress Ep_{ji} lower than all other combinations of sample dates and the date prior was used as the terminal date. If late dates did not represent an outlier, estimates of Ep_{ji} were expected to be distributed evenly above and below *Ep* and all dates would be included.

The proportion of tows with eggs was estimated for each spawning area and year or for the baywide estimate as

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

where $N_{present}$ equaled the number of samples with Striped Bass eggs present and N_{total} equaled the total number of samples during the *Ep* period. The SD of *Ep* was estimated as:

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

Ninety percent confidence intervals were constructed as:

$$(3)\ Ep \pm 1.645 \cdot SD; \text{ (Ott 1977).}$$

We plotted all available estimates of *Ep* as spawning area specific estimates to view individual spawning area trends. Choptank River, Head-of-Bay, Potomac River, and Nanticoke River were sampled for the JI and we pooled data from these spawning areas, when available, to estimate baywide *Ep*. Uphoff (1997) determined that *Ep* in one or several systems could represent baywide spawning stock status.

Linear regression was used to estimate the relationship of *Ep* and SSB during 1982-2017 (this time span represents temporal coverage of SSB estimates; NEFSC 2019). Estimates of SSB were $\log_e$-transformed. The general linear regression model for these analyses was

$$(4)\ \log_e SSB = (m \cdot Ep) + b;$$

where m is the slope and b is the intercept. Residuals were examined for outliers and serial trends. Relationships indicated by regressions were considered strong at $r^2 \geq 0.64$ (this is the "strong correlation" recommendation of Ricker (1975) squared); weak relationships were indicated by $r^2 \leq 0.25$; and moderate relationships fell in between. Level of significance was reported, but potential management and biological significance took precedence over significance at $P < 0.05$ (Anderson et al. 2000; Smith 2020). Strong and moderate relationships were considered of interest for management.

Relative Larval Survival (RLS) - We used the geometric mean catch of Striped Bass juveniles per standard seine haul at permanent stations in Head-of-Bay, and Potomac, Choptank, and Nanticoke rivers as our index of baywide recruitment (baywide JI; Durell and Weedon 2019) and baywide *Ep* (described above) to estimate relative larval survival (RLS) as

$$(5)\ RLS = JI / Ep.$$

Amendment 3 to the Interstate Striped Bass Management Plan specified recruitment failure as three consecutive years of a juvenile index lower than 75% of all other values in the dataset for three consecutive years (ASMFC 2003). We used JIs or estimates of RLS in the lower quartile of their time-series (depending on which was being analyzed) as our criteria for a poor year-class. Conversely, we used the upper quartile as an indicator of a strong year-class. We determined how often RLS and JIs, when available for the same years, were classified into the same category (upper quartile, lower quartile, or the combined mid-quartiles) or were classified into a different quartile. We applied the recruitment failure criterion to the RLS and JI time-series to see when an evaluation would have been triggered and compared that to when the Emergency Striped Bass Act was enacted (1979; Richards and Rago 1999).

Confidence intervals (90%) were developed for RLS using an Excel add-in, @Risk, to simulate distributions reported for numerators and denominators. @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).

Each annual set of estimates was simulated 5,000-times. Annual means and SDs of *Ep* were used to generate simulations. Juvenile indices, based on geometric means, were back-transformed into the mean of $\log_e$-transformed catches (+1) and its SE was derived from the 95% CI reported by Durell and Weedon (2020). Geometric means were recreated by exponentiating the simulated mean of $\log_e$-transformed catches (+1).

We constructed deterministic estimates of relative survival during 1982-2017 based on SSB (SSB RLS) as

$$(6)\ \text{SSB RLS} = \text{JI} / \text{SSB}.$$

Estimates of RLS (based on *Ep*) and SSB RLS were standardized to their respective 1982-2017 means to place them on the same scale and were plotted to assess similarities in trends.

Exploration of (1) associations among spawning area recruitment indices and (2) year-class success of Striped Bass with other anadromous fish – Correlation analysis was used to explore the strength of associations of spawning area specific Striped Bass JIs (Head-of-Bay, Potomac River, Nanticoke River, Choptank River, and Patuxent River) during 1957-2019 (Patuxent River time-series started in 1983; Durell and Weedon 2020). Strength of associations of JI's among areas could supply insight on regional similarity of habitat conditions.

Correlation was also used to examine associations among the baywide Striped Bass JI and baywide JI's for two semi-anadromous species (White Perch, and Yellow Perch). Yellow and White perch share the larval nursery of Striped Bass (Uphoff 1991; North and Houde 2001; 2003) and the degree of correspondence of their baywide JI trends with the Striped Bass JI could provide insight on the importance of common larval nursery habitat conditions on year-class success.

We classified correlations as strong, based on $r \geq 0.80$ (Ricker 1975); weak correlations were indicated by $r < 0.50$; and moderate correlations fell in between. Level of significance was reported, but as with linear regression, potential management and biological significance took precedence over significance at $P \leq 0.05$ (Anderson et al. 2000; Smith 2020). Strong and moderate correlations were considered of interest for management.

Egg Distribution in Infrequently Sampled Spawning Areas - Striped bass egg presence-absence (by spawning area, date, site, and time) from surveys conducted by MD DNR since the early 1950s (through 1981) were summarized in a spreadsheet by Marcus Patton in 2018 from tables in federal aid reports, existing paper summaries, or original data sheets; yellow perch larval presence, water temperature, and salinity were also transcribed. These data were used to compare changes in extent of spawning in time and space determined from sporadic surveys during the 1950s (Patuxent, Wicomico, and Chester rivers) with later surveys, including those we conducted during 2014-2019. Tables were constructed to view the presence or absence of eggs in samples by date and location for Patuxent, Wicomico, and Chester rivers. Each table summarized this information for a year, from the date a first egg was collected (surveys in the 1950s may not have started soon enough to capture early spawning) to when water temperature reached 21°C. Multiple surveys from a spawning area during the 1950s were combined into a single table when sampling was sparse. Surveys conducted in the Patuxent River during 1991 (Secor et al. 1994) and Chester River during 1996 (Burton et al. 1996) used combined tows and were not included in *Ep* estimates; however, they were included to examine distribution.

Land Use and Water Quality - We sampled Choptank River under F-63 during 2013-2019, Patuxent River in 2015-2016, Wicomico River in 2017-2018, and Chester River in 2019 to explore the influence of land use on spawning and basic water quality conditions within the spawning and larval nursery area. Patuxent and Wicomico rivers were considered urbanizing watersheds, and the remaining watershed were categorized as rural.

We used property tax map-based counts of structures in a watershed, standardized to hectares (C/ha), as a continuous indicator of development during 1950-2016 (Topolski 2015). We also used Maryland Department of Planning (MD DOP 2019) estimates of percent of watershed in urban, agriculture, forest, and wetlands that were available for 1973, 1994, 1997, 2002, and 2010. Current estimates of C/ha were compared with development targets (C/ha = 0.37) and thresholds (C/ha = 0.86) for Chesapeake Bay fish habitat developed under F-63. Watersheds at or below target C/ha generally support productive habitat for spawning and larval development based on F-63 monitoring of Herring and Yellow Perch. With C/ha at or above the threshold, risk is high that habitat problems in a watershed will impede traditional fisheries management. See **Common Background for Job 1, Sections 1-3** and **General Spatial and Analytical Methods used in Job 1, Sections 1-3** for detailed descriptions of how land use estimates and development reference points were derived.

We constructed annual conductivity, pH, and water temperature summaries over a standard time period (measurements available during April 1-May 8) and relative area (salinity $\leq$ 2.0 ‰) for spawning areas surveyed during 2014-2019. Choptank River provided long-term comparisons of water quality between 1986-1991 (Uphoff 1989; 1992) and 2014-2019 to examine whether conditions changed or remained relatively constant between the two time periods. Historical summaries for the standard period were developed for Patuxent River in 1991, Nanticoke River in 1992 and 1993, and Chester River in 1996 from data in report appendices (Secor et al. 1994; Houde et al. 1996; Burton et al. 1996). Choptank River data for 1980-1990 existed in a MD DNR data base in a format that has not been supported for years; documentation for the data base was scanty but we were able to extract water quality data from it. M. McGinty and J. Uphoff had used this system in the past, but retrieving data was a challenge.

Historical surveys analyzed had sampled the span of an entire spawning area; water quality surveys that did not span the spawning area were not used. Choptank River (1986), Patuxent River, Nanticoke River, and Chester River surveys employed fixed station designs (remaining surveys were described previously or in Uphoff 1997). Summary water quality statistics included mean, median, minimum, maximum, and the interval encompassing 90% of measurements. Analyses of Choptank River water temperature, pH, and conductivity in Uphoff

(1989; 1992) featured both mean and minimum conditions and these summary statistics, along with their maximums, were emphasized in Choptank River comparisons. Means, minimums, and maximums would be relevant biologically; means would provide some indication of chronic conditions, while maximums and minimums would capture the extent of extreme events likely to influence acute conditions. Estimates of pH were converted to H+ concentration to estimate the mean and then converted to mean pH.

Water quality conditions were compared among time periods and among spawning areas in rural (Choptank, Nanticoke, and Chester rivers) and urbanizing watersheds (Patuxent and Wicomico rivers). These summaries were not likely to provide a means of determining whether these changes were directly related to year-class success since they were not directly tuned to time periods of egg or larval mortality.

Dissolved oxygen data were not summarized because they did not fall below the 5 mg / L target for Chesapeake Bay living resources over all the years and spawning areas available. Water temperatures were cool and turbulent flow within the spawning area provided for thorough mixing. Dissolved oxygen data were not summarized further in tables or graphs; however, DO measurements were used in some correlation analyses (described below).

Correlation analysis was used to explore associations among temperature, DO, pH, and conductivity during the standard time period within each spawning area and time period; correlations were based on data rather than data summaries. Of particular interest were associations of DO and pH. Strong to moderate positive correlations of DO and pH would indicate that photosynthesis by phytoplankton may be an important source of pH change in addition to atmospheric deposition, discharges, and watershed runoff.

Correlation analysis was used to determine the strength of associations of river flow, and mean water temperature, conductivity, and pH in Choptank River during 1986-1990 and 2013-2019 (combined). Average Choptank River flow (discharge in ft^3/s) at the USGS gauging station at Greensboro, Maryland, was averaged for March (month prior to spawning) and April (month when most spawning occurs; Uphoff 1989; 1992), 1980-2019. Strong correlations among river discharge, conductivity, and pH during 1983-1988 suggested these variables were not independent influences (Uphoff 1992) and we were interested in whether these strong associations had changed over time. Average March-April flow during 1957-2019 was plotted against $\log_e$-transformed Striped Bass Choptank River JIs to view potential relationships. The $\log_e$-transformation linearized data and reduced variability among years (particularly the extremely large 2001 JI) for linear regression analyses that followed from examination of plots. Residuals were examined for serial patterning that could indicate nonstationarity of habitat conditions. If serial patterning was evident, the time periods suggested could be compared to *Ep* and water quality summaries for coinciding years.

Results

Proportion of tows with Striped Bass eggs (Ep) - Sufficient data to estimate *Ep* were available for seven spawning areas: Chester River, Choptank River, Head-of-Bay and Elk River (latter was considered a proxy for Head-of-Bay when the whole Head-of-Bay was not sampled), Nanticoke River, Patuxent River, Potomac River, and Wicomico River (Table 2.1.2). Surveys spanned 1955-2019, but sample sizes were insufficient for estimates of *Ep* during 1958-1960. We eliminated nine *Ep* estimates based on samples from paired or triplicate tows that were combined or averaged (seven were from spawning areas with JIs); examination of time-series plots with these estimates suggested they were positively biased. Elimination of surveys with combined tows during created a gap during 1992-1993 (Table 2.1.2).

Data sets retained had 3-21 stations and number of tows analyzed ranged between 26 and 352 (Table 2.1.3). There were 109 surveys where *Ep* could be estimated; surveys of JI rivers comprised 98 of these. The Nanticoke River (N = 42) and Choptank River (N = 28) were the most frequently surveyed JI spawning systems. Ichthyoplankton surveys on the largest spawning areas, Head-of-Bay (N = 7) and Potomac River (N = 10) were primarily during the 1970s and 1980s. Elk River, representing a portion of the Head-of-Bay, had 11 surveys, but 7 of these were redundant with Head-of-Bay surveys and were not used to estimate Baywide *Ep*.

Estimates of *Ep* for all individual spawning areas ranged from 0.36 to 1.00. Sixteen percent were less than 0.65, and 77% were fairly evenly distributed between 0.65 and 0.95 (Table 2.1.3; Figure 2.1.4). Coefficients of variation of individual *Ep* estimates varied from 1.5% to 21%; 81% of CV estimates were 12.5% or less (Table 2.1.3).

Estimated spawning area specific *Ep* based on JI rivers, could be estimated for 59 years during 1955-2019 (91 surveys; Figure 2.1.5). Thirty nine years had a single spawning area survey, 15 had 2 spawning area surveys, and 6 had 3 spawning area surveys. Estimates of *Ep* ranged from 0.36 to 1.0 and 12% were 0.65 or less; all of the latter occurred during 1982-1988 (Figure 2.1.5). Coefficients of variation were between 2% and 12% (Table 2.1.3).

Spawning area specific *Ep* for spawning areas sampled for the JI grew to its zenith during the 1960s, from 0.60-0.74 during 1955-1957 to 0.74-0.94 (median = 0.86) during 1961-1971 (Figure 2.1.5). Estimates of *Ep* decreased during 1971-1974. This decline reflected low *Ep* in the Potomac River (0.44-0.54), while it was higher in the Nanticoke and Elk rivers (0.65-0.76). Estimates of *Ep* then returned to near peak levels during 1975-1979 (0.63-0.91; estimates from Nanticoke River, Potomac River, and Head-of-Bay and Elk River). A steady decline in *Ep* to the nadir of 0.36-0.39 during 1983-1984 was followed by a return to higher levels starting in 1989 (0.74) and spawning area specific *Ep* has varied between 0.57 and 1.0 through 2019 (median = 0.77). Only one estimate of *Ep* (Choptank River in 2015) has been below 0.60. Variability of *Ep* was lower during 1989-2000 (0.72-0.86) than afterward (0.60–1.0). Estimates of *Ep* were similar among spawning areas during any given year, except during 1970-1975, when Potomac River *Ep* was noticeably lower (Figure 2.1.5).

When available surveys from JI rivers were combined to form a single baywide annual estimate, 90% CI overlap of baywide *Ep* indicated increasing spawning stock during 1955-1957, high spawning stock from 1961 to 1979 (although somewhat lower in 1973-1974), low *Ep* during 1982-1988 (with a nadir during 1983-1984), and recovery to 1961-1979 levels after 1988 (Table 2.1.4; Figure 2.1.6). The plot of baywide *Ep* and JIs indicated that recruitment shifted to a low level nearly a decade sooner (early 1970s) than spawning stock (early 1980s; Figure 2.1.7). Consistently poor baywide JI's, among the lowest of the time-series, coincided with depleted *Ep* during 1983-1988. Estimates of *Ep* returned to high levels in 1989, but a strong year-class was not produced until 1993; *Ep* was not estimated in 1992-1993, but was presumed to be similar to high values before and after. A pattern of high and low JI's similar to that of 1958-1970 has persisted since 1993. Year-classes in the top quartile (JI $\geq$ 5.6) occurred in 5 of 13 years during 1958-1970 and 10 of 27 years during 1993-2019. JI's between these periods were all below the top quartile. Conversely, year-classes in the bottom quartile (JI $\leq$ 1.6) were more likely to have occurred during 1971-1992 (11 years below), than during 1958-1970 (2 years below) or 1993-2019 (3 years below; Figure 2.1.7).

Trends in *Ep* and SSB both indicated recovery from a depleted state prior to 1989, but *Ep* indicated quicker recovery (Figure 2.1.8). Spawning stock biomass declined after 2012, but *Ep* did not. During recovery (1982-1995), *Ep* increased from 0.36 to 0.72-0.86, while SSB estimates increased from 15,000 to 91,000 MT. After recovery (1996-2017), *Ep* fluctuated between 0.63 and 0.85, while SSB estimates were between 68,000 and 114,000 MT. Estimates of SSB varied from 86,000-99,000 MT during 1996-2012, while *Ep* varied from 0.63-0.85. As estimated SSB declined from 89,000 to 68,000 MT during 2013-2017, *Ep* fluctuated between 0.67 and 0.82 with one higher estimate, 0.95 (Figure 2.1.8).

Two types of analytical approaches were suggested by examination of the trends in *Ep* and SSB: (1) a continuous linear relationship of SSB and *Ep* over the time-series and (2) separate relationships for the recovery period (1982-1995) and post-recovery (1996-2017). The linear regression of *Ep* and $\log_e$-transformed SSB during 1982-2017 explained 54% of the variation ($P < 0.0001$). Residuals plotted against *Ep* exhibited a "hook" pattern (two parallel relationships; one for 1982-1991, one for 2000-2017, and joined by years between; Figure 2.1.9) and a serial pattern in residuals was evident that did not support use of this model. Breaking the time-series into a recovery (1982-1995) and post-recovery (1996-2017) time-blocks resulted in a strong fit for the recovery time block ($r^2 = 0.82$; $P < 0.0001$) and an extremely poor fit ($r^2 \sim 0.00$; $P = 0.95$) to the post-recovery time block. The relationship of *Ep* with $\log_e$-transformed SSB (MT) during the recovery period was described by the equation

$$(7)\ \log_e \mathrm{SSB} = (3.35 \cdot Ep) + 8.30;$$

The SE of the slope equaled 0.50 and the SE of the intercept equaled 0.31. Residuals did not exhibit serial patterning.

Relative larval survival (RLS) – Estimates of RLS indicated periods of higher and lower survival rather than random scattering throughout the time-series. Ninety percent CI overlap indicated that two years of higher RLS between 1961 and 1970 (10 years available) and seven years during 1994-2019 (26 years) could be clearly separated from the estimates of RLS during 1971-1991 (21 years; Figure 2.1.10). Estimates of RLS in the bottom quartile ($\geq$ 2.03) were concentrated in the period spanning 1977-1991. Nine of these years were in the bottom quartile, with 1977-1985 having six of them. There were two years of RLS in the bottom quartile during 1961-1970 and three years during 1994-2019. Three years were in the top quartile ($\geq$ 6.87) during 1961-1970 and 11 years during 1994-2019. Only a single year (1982) fell in the top quartile between 1971 and 1991 (Figure 2.1.10).

Over the years where pairs of RLS and JI were available, concordance in assigning these indices into upper or lower quartiles or into the mid-quartiles (second and third quartiles) was good. There were 13 years where both indices fell in the upper quartile, 13 where both fell into the lowest quartile, and 32 years in the mid-quartiles. There were nine pairs where indices were assigned into an adjacent quartile and no instances where an index in the top quartile was assigned into the bottom or vice versa. Based on the Amendment 6 criterion (three consecutive years of juvenile indices in the bottom quartile in a spawning region), only one period (1985-1988) would have triggered an evaluation of recruitment in Maryland spawning areas based on the JI. Substituting RLS for the JI, a single, but earlier period (1979-1981) would have met the evaluation criterion. Both of these hypothetical triggering events were after the actual initiation of the Emergency Striped Bass Act (1979).

Use of *Ep* or SSB as the denominator for determining relative larval survival produced different impressions of changes in relative larval survival (Figure 2.1.11). Standardized estimates of RLS indicated that on a relative basis, RLS was lower than SSB RLS during 1982-1991 and higher afterward. The pattern of SSB RLS during 1982-1992 was not visually dissimilar from the rest of the time-series, whereas RLS was depressed during 1982-1992. Estimates of SSB RLS during 2004-2011 appeared worse than during 1982-1992 (Figure 2.1.11).

Exploration of (1) associations among spawning area recruitment indices and (2) year-class success of Striped Bass with other anadromous fish – Area Striped Bass JI's exhibited similar general patterns during 1959-2019 (Figure 2.1.12). Correlations of area JI's were strong for Potomac and Patuxent rivers ($r = 0.86$, $P < 0.0001$) and moderate for Choptank and Nanticoke rivers ($r = 0.66$, $P < 0.0001$); these pairs of areas were adjacent to one another (Table 2.1.5). Correlations of remaining combinations were weaker ($r = 0.32$-0.56; Table 2.1.5) and none of these areas were in as close proximity as Potomac River to Patuxent River or Choptank River to Nanticoke River. Strong baywide year-classes in Maryland reflected occasional synchronization of conditions (environmental and spawning stock) among regions (Head-of-Bay (northern Maryland), Potomac and Patuxent rivers (southern Maryland), and Choptank and Nanticoke rivers (eastern Maryland) that were weakly associated.

Baywide Striped Bass, White Perch, and Yellow Perch JIs were synchronous. Striped Bass were strongly correlated with White Perch ($r = 0.86$, $P < 0.0001$) and moderately correlated with Yellow Perch ($r = 0.73$, $P < 0.0001$; Figure 2.1.13). Yellow and White Perch were strongly correlated with one another ($r = 0.83$, $P < 0.0001$; Figure 2.1.13). These correlations supported the hypothesis that larval nursery conditions played a large part in the dynamics of Maryland Striped Bass during 1957-2019.

Egg Distribution in Infrequently Sampled Spawning Areas – The Patuxent River Striped Bass spawning area was determined by MD DNR surveys conducted during 1953-1955 and 1959 (technically, MD DNR did not exist before 1969, but we use its abbreviation to indicate Maryland natural resource agencies that existed earlier as well). It was surveyed again in 1978-1979 (Setzler et al. 1979; Mihursky et al. 1980), 1991 (Secor et al. 1994), and by FHEP in 2015-2016; reports for 1978 and 1979 were located too late to be summarized as tables in this report. Surveys in the 1950s began later than those of subsequent years and we do not know if that reflects later spawning or late scheduling of samples (Table 2.1.6). Presence or absence of eggs indicated that spawning was detected well upstream in the 1950s (FHEP site 11 and above 12). These sites were silted in during 2015-2016 and could not be sampled without significant risk of grounding (J. Uphoff, personal experience). Sampling of Patuxent River in 1991 detected eggs at site 10 and sites above there were not sampled; we do not know whether the absence of the uppermost sites reflected shallowness or not (duplicate tows were combined, so ability to detect spawning may have been enhanced; Secor et al. 1994). The core area for spawning was approximated by FHEP sites 6-11 in the 1950s, 1-8 in 1991, 1-6 in 2015, and 1-9 in 2016. Dates when the 21˚C temperature cut-off was met ran later (mid-to-late May) in some of the surveys during the 1950s (as late as May 25), 1978-1979 (May 23 and May 10, respectively; Setzler et al. 1979; Mihursky et al. 1980) than during 1991 (April 30) or 2015-2016 (May 5; Table 2.1.6).

The Wicomico River Striped Bass spawning area was designated by MD DNR surveys conducted during 1954, 1957, and 1959 and was resampled by FHEP during 2017-2018 (Table 2.1.7). Eggs were most often present during the 1950s between FHEP sites 3 and 5, but were present between sites 1 and 8; eggs were present at a site well below the lowest site sampled by FHEP in 1959. Eggs were not collected above site 8 during the 1950s and 2017, but were present at sites 8 and 9 during 2018. Eggs were present in similar temporal spans (mid-April to late April – early May) during the 1950s and 2018, but spawning was shifted earlier in 2017 (March 30-April 18; Table 2.1.7).

The Chester River Striped Bass spawning area was established by a MD DNR survey conducted during 1955, sampled in 1996 by Burton et al. (1996) to determine egg production for MD DNR, and sampled by FHEP in 2019 to supply information for the Queen Anne County comprehensive growth plan. Egg presence was most consistent in mid- and upstream stations during 1955 and 1996, but was oriented downstream during 2019 (Table 2.1.8). Eggs were present between FHEP sites 3 and 11 in 1955 and were most prevalent upstream at sites 7, 9, and 11. During 1996, eggs were present from FHEP site 2 to a site above site 12 (this area was too shallow to sample in 2019); they were most prevalent at sites 5-11 (duplicate tows were combined, so ability to detect spawning may have been enhanced). Spawning was spread throughout sites 3 to 10 during 2019, but eggs were not detected above site 7 after April 10, 2019. The spawning season was earlier and much shorter during 2019 than the other two years. Initial major spawning activity (eggs present at multiple sites) was detected on April 14 in 1955, April 15 in 1996, and April 8 in 2019 (Table 2.1.8). Water temperatures below 21˚C occurred through May 15, 1955; May 9, 1996; and April 22, 2019.

Land Use and Water quality – Patuxent River's watershed is located on the western shore of Chesapeake Bay in the suburbs of Washington, DC (Figure 2.1.14). Patuxent River originates in Maryland's Piedmont, but the tidal portion (including the Striped Bass spawning area) is located in the Coastal Plain. Wicomico, Choptank, Chester, and Nanticoke rivers are located on the eastern shore of Maryland, entirely within the Coastal Plain (Figure 2.1.14).

The portion of the Patuxent River watershed draining into the Striped Bass spawning area has undergone extensive development since 1950, climbing from 0.04 C/ha to 1.28 C/ha in 2018; rapid growth occurred during 1962-2005 (Figure 2.1.14). The C/ha target was breached in 1978 and the threshold was breached in 1997. Most recent MD DOP estimates of land use (2010) characterizes the watershed draining into the spawning area as primarily urban (42%; low and high density residential) and forest (35%; forest may include low density residential; Table 2.1.9).

MD DOP estimates of Wicomico River land use in 2010 indicated that forest was the largest land use (37%) in the area draining into the spawning area, but both urban and agriculture were not far behind (both near 30%; Table 2.1.9). The city of Salisbury is located on the upper tidal area and much of its non-tidal drainage is encompassed in the city and its suburbs. Estimates of C/ha in the portion of the watershed draining into the Striped Bass spawning area started higher in 1950, 0.16 C/ha, than the other Striped Bass spawning areas sampled by FHEP (Figure 2.1.14). Growth was steady through 1983, reaching 0.39 C/ha; growth then increased and reached 0.65 C/ha in 2007. Estimates of C/ha increased very slowly through 2018 (0.68; Figure 2.1.14). Wicomico River watershed development is above the C/ha target (0.36 C/ha, breached in 1978) and is at about 78% of the threshold.

Choptank and Chester rivers' spawning area drainages are rural with very similar patterns of land use and development history. Agriculture is by far the predominant land use (> 60%) in the portions of their watersheds draining into the Striped Bass spawning areas (Table 2.1.9).

Development trends were very similar, increasing from ~ 0.03 in 1950 to ~ 0.13 in 2018 (Figure 2.1.14). Development was well below the target.

Nanticoke River has a rural watershed. Agriculture is the predominant land use (> 45%), but forest comprises a major portion (39%), and it has the highest fraction in wetland of the study watersheds (7%) in the portion draining into the Striped Bass spawning areas (Table 2.1.9). Development increased from ~ 0.04 in 1950 to ~ 0.11 in 2018 (Figure 2.1.14). Development is well below the target.

Of the three Choptank River water quality parameters summarized, pH offered the clearest indication of change between 1986-1991 and 2014-2019. Estimated means of pH during the standardized period were consistently higher and measurements were more stable (less variable) during 2014-2019 (Table 2.1.10; Figure 2.1.15). Means during 1986-1991 ranged from 6.0 to 6.8, minimums fell between 5.8 and 6.5, and maximums varied from 6.5 to 9.2. During 2014-2019, the range was 7.1 to 7.4 for means; 6.7 to 7.0 for minimums, and 7.8 to 8.1 for maximums. The lowest and also most stable pH conditions were found in 1989, but 1987 and 1990 pH measurement variability was comparable to 2014-2019 (Table 2.1.10; Figure 2.1.15).

Distributions of pH during the 1990s in spawning areas other than Choptank River (Patuxent River, 1991; Nanticoke River, 1992-1993; and Chester River, 1996; Secor et al. 1994; Houde et al. 1996; Burton et al. 1996) were generally in the upper range of the Choptank River during 1986-1991 (Table 2.1.10). Means for the non-Choptank River spawning areas were between 7.0 and 7.6 and exhibited similar wide annual ranges (0.79-1.03; range = maximum-minimum). These ranges were wider than those exhibited by half of the 1986-1991 Choptank River surveys (Table 2.1.10).

During 2014-2019, pH conditions in spawning areas with urbanizing watersheds (Patuxent and Wicomico rivers) generally exhibited higher means (7.5-7.8) and greater variation in measurements (range = 1.1-1.6) than rural watersheds (Choptank and Chester rivers; means varied from 7.1 to 7.4 and ranges were 0.7-1.3; Table 2.1.10). Patuxent River pH means and ranges appeared to change little between 1991 and 2015, while pH range in Chester River contracted from 2.4 in 1996 to 0.7 in 2019 (Table 2.1.10).

Choptank River spawning area conductivity summaries offered little indication of change (Table 2.1.11; Figure 2.1.16). During 1986-1991, mean conductivity ranged from 426-910 uS/cm and minimums were between 94 and 177 uS/cm. During 2014-2019, means ranged between 463 and 990 uS/cm and minimums from 93 to 135 uS/cm (Table 2.1.11; Figure 2.1.16). Maximums were similar because of the 2 ‰ salinity cut-off.

Comparisons of Patuxent River and Nanticoke River mean and maximum conductivity between 1991-1993 and 2014-2019 were confounded by different survey designs and maximum cutoffs, 2,300 uS/cm in the former and approximately 3,500 uS/cm in the latter. Comparisons of minimum conductivities were not likely to have been impacted and it was considered the most meaningful summary statistic for these surveys. Comparisons of maximum conductivities were fairly meaningless since cutoffs were applied. Mean conductivities, in turn would be impacted by the different maximum cut-offs. Minimums were generally tracked by the 5th percentiles. The 5th percentile would remove potentially extreme values.

Conductivity distributions in spawning areas with urban watersheds exhibited higher minimums than spawning areas in rural watershed during 2014-2019 (Table 2.1.11). Minimum conductivity in the Patuxent River spawning area was 142 uS/cm in 1991 and it rose to 317-378 uS/cm in 2015-2016. Minimum conductivities in Wicomico River were 217 uS/cm in 2017 and 199 uS/cm in 2018. Minimum conductivities were similar between (rural) Nanticoke and Choptank rivers during 2014-2019 (93-141 uS/cm). Minimum conductivity in rural Chester River during 1996 (97 uS/cm) and 2019 (140 uS/cm) fell within the same range as Choptank and Nanticoke rivers (Table 2.1.11).

There appeared to be a general upward shift in Choptank River spawning area water temperature between 1986-1991 and 2014-2019 during the standard period (April 1 – May 8) used for comparisons (Table 2.1.12; Figure 2.1.17). Mean temperatures ranged from 14.0 to 15.9 ˚C during 1986-1991 and 14.3-17.8 ˚C during 2014-2019, with three of the means during the latter period exceeding those of the earlier period. Minimum water temperatures during 1986-1991 were 9.3 to 11.7 ˚C and were 10.1 to 12.4 ˚C during 2014-2019; two minimums during 2014-2019 were higher than any detected in the earlier period. Maximum temperatures ranges from 17.7 to 21.8 ˚C during 1986-1991 and 18.5-23.1 ˚C during 2014-2019; water temperature during 2017 was well above any other maximum temperatures measured (Table 2.1.12; Figure 2.1.17).

Water temperature patterns within years were similar among spawning areas sampled during 2014-2019 (Table 2.1.12); 2018 was an exception with Choptank River exhibiting much greater variation and a lower mean (14.3 ˚C) than Wicomico River (mean = 16.3 ˚C). Means were typically near or below 16 ˚C during 2014-2016 and above 16 ˚C during 2017-2019 (Table 2.1.12).

Correlations among DO, pH, water temperature, and conductivity revealed three patterns among spawning areas and time periods (Table 2.1.13). In the rural Choptank River, none of the correlations were strong enough to be of interest for either time period (1986-1991 or 2014-2019), suggesting lower potential for phytoplankton influence on pH. In the remaining spawning areas, pH was moderately and positively correlated with DO, suggesting greater phytoplankton influence. In the rural Chester River during 1996, there were moderate correlations of pH with temperature (-) and DO (+), and a strong correlation of temperature with DO (-). During 2019, pH in Chester River was moderately correlated with temperature (+), DO (+), and conductivity (+), and DO was strongly correlated with temperature (-). In the urban Patuxent River (1991 and 2015-2016) and Wicomico River (2017-2018), pH and DO were moderately (+) correlated. None of the other combinations were correlated well enough to be of interest (Table 2.1.13).

March and April mean flows in Choptank River during 1986-1991 and 2014-2019 (combined) were moderately correlated with one another (+) and both were moderately correlated (-) with mean conductivity (Table 2.1.14). Remaining variables (mean water temperature and pH) exhibited weak correlations with flow, conductivity, and each other (Table 2.1.14).

In the long-term (1957-2019), there appeared to be a weak influence of Choptank River March-April flow on $\log_e$-transformed JI. A quadratic function explained a moderate amount of variation ($r^2 = 0.25$, $P = 0.0002$; Figure 2.1.18) and suggested there could be an optimum range of flow associated with a higher frequency of strong year-classes. The plot of residuals against flow did not suggest patterning; however, serial patterning of residuals was indicated (Figure 2.1.19). As many as six different periods were suggested: 1957-1965, positive and negative residuals; 1966-1972, positive residuals; 1973-1981, negative residuals; 1982-1990, positive and negative; 1991-2007, positive residuals; and 2008-2019, positive and negative residuals (Figure

2.1.19). The relationship was not stable over time; some sets of years had stronger or weaker responses to flow. The lower bounds of points in the regression mostly reflected 1973-1990, while the upper bounds primarily reflected 1966-1972 and 1991-2007.

Discussion

Survival of Striped Bass eggs and larvae, and subsequent recruitment in Maryland's portion of Chesapeake Bay exhibited time blocks of varying productivity during 1957-2019. Recovery of Striped Bass spawning stock, indicated by high *Ep* after 1988, was accompanied and complemented by a recovery of egg-larval survival, indicated by RLS, a few years later. Estimates of high RLS have occurred every few years since 1993. Comparisons of flow, water temperature, conductivity, and pH indicated conditions within Maryland's Striped Bass spawning and larval nursery areas have changed over time and among areas. Water quality conditions differed between spawning areas in rural and urbanizing watersheds. Long-term (1950s to present), concurrently collected water temperature and egg distribution data suggested that water temperature (21˚C) indicative of the end of spawning and-or poor survival of hatched larvae was occurring earlier in recent years.

The near collapse of the Striped Bass fishery in the 1980s was driven by a shift to low JIs in the early1970s that was followed by a decline in baywide *Ep* a decade later. Baywide *Ep* increased during 1955-1957, was high during 1961- 1979, low during 1982-1988, and recovered to 1961-1979 levels after 1988. Year-classes in the top quartile occurred frequently during 1958-1970 (31% of indices) and 1993-2019 (41% of indices). Juvenile indices between these periods were not in the top quartile and year-classes in the bottom quartile were concentrated between those two periods.

Maryland's baywide Striped Bass JI was well correlated with baywide JIs of White Perch and Yellow Perch, two estuarine resident species that shared common larval nurseries with Striped Bass, indicating larval conditions were the primary factor influencing their year-class success. White and Yellow Perch differ from Striped Bass in size attained as adults; maturation, migrations, spawning locations, egg types, adult trophic levels, fisheries, and management (Lippson and Moran 1974; Piavis 1991; Setzler-Hamilton 1991; Setzler-Hamilton and Hall 1991; Maryland Sea Grant 2009; Kerr and Secor 2012; Yellow Perch Workgroup 2002; Maryland Fishing and Boating Services 2015a; 2015b; NEFSC 2019), making simultaneous overfishing an unlikely explanation for similarly timed recruitment trends.

Associations among Striped Bass JIs in adjacent spawning areas (Choptank and Nanticoke rivers in eastern Maryland or Potomac and Patuxent rivers in southern Maryland) were moderate to strong, and correlations were weaker when spawning areas were not adjacent. Conditions among these major spawning and larval nurseries occasionally aligned favorably, resulting in a strong Striped Bass JI.

Estimates of RLS indicated periods of fairly consistent higher or lower survival rather than random scattering throughout the time-series indicative of stationary influences on recruitment. Estimates of RLS in the bottom quartile were concentrated in the period spanning 1977-1991, while periods of RLS in the upper quartile occurred during 1961-1970 and 1994-2019. The second period of higher RLS likely began in 1993 when a strong year-class was produced baywide. However, *Ep* was not sampled during 1992-1993. Relative larval survival in 1993 can be approximated by the 1993 MDJI (14.0) divided by mean *Ep* in 1990-1991 (0.76 and 0.82) and 1994-1995 (0.74 and 0.86). This estimate of RLS, 17.6, places at third highest of the time-series.

The criterion in Amendment 6 (ASMFC 2003) for determining recruitment failure in a spawning region (3 consecutive years of lowest quartile juvenile indices) appeared to be an insensitive trigger when applied to the JI or RLS time-series. The JI did not breach this criterion until 1987. When compared to actual actions adopted in response to the decline (Richards and Rago 1999), the trigger would have fired 17 years after the last strong year-class had appeared, 9 years after the Emergency Striped Bass study was initiated, and 2 years after moratoria and other much more conservative measures were adopted. Applying the Amendment 3 criterion to RLS would have triggered evaluation 6 years earlier than applying the criterion to the JI, but it would still have been delayed from the actual schedule. It is likely the conservative management of the fishery in place since 1995 would not allow for the precipitous population decline experienced in the 1970s-1980s, but it might lead to critical delay in evaluating habitat conditions that might be impairing formation of successful year-classes.

Inspection of either time-series of indicators (JI or RLS) indicated that the strong 1970 year-class was followed by 5-6 years of mid-quartile indices and then 15-16 years of mixed mid-quartile and lower quartile indices. Year-classes in the lowest quartile became frequent during the late 1970s into the early 1990s, but 3 years of consecutive poor indices occurred relatively late in both time-series. The most recent period of lower recruitment during 2006-2010 (four baywide JIs in the mid-quartile and one in the bottom quartile) was followed by a return to a mix of high, mid, and lower indices. Forming a recruitment trigger around the absence of a strong year-class and-or high RLS for some extended period of time (more than 5 years) may lead to a more timely response in tune with the needs of the fishery and its management.

Uphoff (1997) created a Striped Bass stock-recruitment table based on increments of *Ep* and Maryland spawning area JIs to describe the influence of spawning stock on average recruitment. Detection of periods of different RLS indicated that the average recruitment assumption was not met and the relationship detected was not valid. Stock assessments often assume the number of fish recruiting to a population is related to the biomass of spawning adults and that recruitment dynamics are stable over time (Walters 1987; Szuwalski et al. 2014). Szuwalski et al. (2014) found that recruitment was not positively related to spawning stock for 61% of 224 stocks examined and 85% of the stocks not found related to spawning stock exhibited shifts in average recruitment. Environment appears to more strongly influence recruitment than spawning biomass for most stocks, but recruitment driven by spawning biomass is central to fisheries management. Nonrandom, unaccounted for shifts in productivity can bias estimates of important management parameters (Szuwalski et al. 2014).

Use of *Ep* or SSB as the denominator for determining RLS or SSB RLS, respectively, produced different depictions of egg-larval survival dynamics and patterns of underlying productivity. Estimates of RLS were depressed during 1982-1992 (SSB was not available for earlier years to estimate SSB RLS, but RLS was depressed during 1971-1992) and then shifted upwards and remained high, except for a cluster of years of depressed survival during 2006-2010. The pattern of SSB RLS during 1982-1989 appeared similar to much of the remaining time-series, indicating that habitat conditions were similar. Estimates of SSB RLS during 2004-2011 appeared worse than during 1982-1989 and the stretch of poor survival estimates during 2004-2011 was two years longer than indicated by RLS.

A stronger density-dependent response of egg production than that modeled by the Striped Bass statistical catch-at-age model's estimates of SSB may be reflected by the rapid, earlier recovery of *Ep* and variability around a constant level of post-recovery *Ep* estimates. A continuous, sharp decline in SSB that began in 2012 was not evident with *Ep*. While there was a strong relationship of baywide *Ep* to $\log_e$-transformed SSB during recovery (1982-1995), the fit during post-recovery was very poor. There was strong contrast of values of the two spawning stock size indicators in the recovery period, but not the post-recovery period (all were fairly high).

While SSB is in wide use in fisheries management, its use as a proxy for total egg production (TEP) has been challenged (Kell et al. 2016; Marshall 2016; Barneche et al. 2018). SSB underestimates uncertainty and can be an insensitive index of stock reproductive potential (Kell et al. 2016; Marshall 2016; Barneche et al. 2018). Omitted processes and simplifying assumptions may result in the characteristics of the SSB time-series being determined by the model used to generate them rather than descriptive of underlying ecological phenomena (Kell et al. 2016). Dynamics based on TEP may be different from those based on SSB derived /from commonly applied stock assessment models since SSB ignores biological phenomenon such as cohort effects and assumes condition, growth, maturity, fecundity, and natural mortality are time invariant. A constant relationship of fish weight with fecundity is the fundamental assumption for SSB as a proxy for TEP (Kell et al. 2015; Marshall 2016; Barneche et al. 2018). A meta-analysis of the relationship of body mass on total reproductive energy (fecundity × egg volume × egg energy) of 342 marine species indicated hyperallometry (scaling described by a power function with an exponent greater than 1) in 79.1% of species examined, suggesting that larger mothers contribute disproportionately to population replenishment (Barneche et al. 2018). Treating stock assessment model output as data in subsequent modeling and analyses may overlook uncertainty in the assessment output, potential bias of estimates, correlation between estimates, and structural assumptions of the original assessment model (Brooks and Deroba 2014).

Retrospective patterns are common in catch-at-age modeling and make it difficult to decide on appropriate stock size estimates (Walters and Martell 2004; Rothschild et al. 2014). Five years of additional data were needed in the current Striped Bass stock assessment before the 2010 estimate of SSB stabilized about 15% higher (NEFSC 2019). If this percentage change persisted through the most recent years, the decline in SSB would be less pronounced than indicated. Presence of a retrospective pattern suggests that some structural assumption(s) of the model could be invalid and could be related to time-varying dynamics that are insufficiently specified, but absence of a retrospective pattern does not necessarily imply that a model has little or no structural uncertainty (Brooks and Deroba 2014).

The Striped Bass statistical catch-at-age model accommodates changes in weight-at-age when estimating SSB, but other processes such as condition (weight is not a robust indicator of Striped Bass condition; Jacobs et al. 2013), natural mortality, and maturity are assumed constant (NEFSC 2019). Striped Bass have exhibited multiple signs that biological processes important to TEP have not remained constant after they were declared recovered in 1995. A shift to poor condition, changes in length- and weight-at-age, a disease outbreak (mycobacteriosis), increased natural mortality, and a substantial drop in forage per Striped Bass occurred in the late 1990s (summarized in Uphoff 2003; Uphoff and Sharov 2018). Skipped spawning by Striped Bass has been documented (Secor 2008). First age that female Striped Bass appeared in experimental gill net surveys on the Potomac River and Head-of-Bay spawning grounds (Versak 2019), an indicator of changing maturity, has shifted from primarily ages 3-5 during 1985-1994, to ages 5-7 during 1995-2010, and to ages 4-5 during 2011-2018.

The number of eggs is an ideal measurement of spawning stock (Hilborn and Walters 1992; Marshall 2016). Egg production was estimated from egg and larval surveys in some

Maryland spawning areas during 1989 to 1996 (Olney et al. 1991; Houde and Rutherford 1992; Secor et al. 1994; Burton et al. 1996; Houde et al. 1996; Rutherford et al. 1997) but this technique was not adopted for stock assessment. Unfortunately, egg presence-absence information was not recoverable from these intensive surveys due to the use of combined paired or triplicate tows that appeared to positively bias estimates of *Ep*.

Empirical indicators of TEP can be particularly important when data-rich stocks become 'data poor' through inability of models to describe processes (Kell et al. 2016). Baywide *Ep* during the latter years of recovery (0.72-0.86 during 1988-1995) was within the range exhibited during recovery (0.63-0.85), but it is possible that baywide *Ep* during post-recovery was insensitive to trends at high stock size. 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. However, if a species is moderately common or a large number of sites are sampled, statistical power of presence-absence surveys should be adequate (Stayer 1999). There is the possibility of some bias between estimates of *Ep* from fixed and stratified random station designs, especially fixed station surveys that appeared to contract to a few core stations that may have minimized zeroes (Nanticoke River 1961-1981).

Live and dead eggs were used as indicators of presence, but *Ep* estimates may have been negatively impacted by extensive episodic egg mortality between sampling dates if dead eggs dissolved between sample visits to the extent that detection was hindered. The proportion of tows with eggs (or SSB and TEP) would not reflect egg quality issues (i.e., larger females producing eggs with higher energy content).

Uphoff (1997) concluded that *Ep* in one or several systems could represent baywide spawning stock status since estimates from different spawning areas within a year were usually similar. However, *Ep* in the Potomac River during 1973-1975 was noticeably lower than other spawning areas sampled. Lower *Ep* in Potomac River at that time likely reflected Maryland's increasingly intense commercial fisheries that targeted Striped Bass on their spawning run (Maryland Sea Grant 2009) that could have depleted local spawning populations. Exploitation rates were at their highest during 1972-1984 (Maryland Sea Grant 2009). Maryland began restricting commercial Striped Bass fishing on the spawning grounds during spawning season in 1978 and permanently ended it in 1982 (Speir et al. 1999).

Estimates of *Ep* from Nanticoke River during 2004-2019 were only based on sampling during April due to the need for staff to transfer to other monitoring projects in May (*Ep* sampling was an add-on rather than part of their regular sampling). This may have introduced positive bias in some years since spawning in May is usually not as intense as April in eastern shore rivers. During 2013-2019, when both rivers were sampled concurrently, Nanticoke River *Ep* was higher than Choptank River *Ep* nearly every year. In 2013, 2014, and 2016, Nanticoke River *Ep* was 6-8% higher than Choptank River *Ep*, but 90% CIs overlapped; Choptank River *Ep* was 2% higher than Nanticoke River during 2019 (90% CIs overlapped). Two years had more substantial differences (90% CIs did not overlap) - 2015 (Nanticoke River was 30% higher) and 2017 (Nanticoke River was 24% higher). Lack of sampling into May could have been associated with higher Nanticoke River *Ep* in 2015, but only one additional date was added to the *Ep* estimate for Choptank River. The ending date was a day later in Nanticoke River than Choptank River during 2017. Choptank River generally had higher sample sizes during 2013-2019 and the influence of Nanticoke River on *Ep* would be lessened when calculating a combined estimate.

Striped Bass egg presence-absence in three infrequently sampled spawning areas (Patuxent, Wicomico, and Chester rivers) between the 1950s and 2015-2019 did not indicate major changes in spawning stock status (surveys with single tows only) in these spawning areas. Chester River *Ep* estimates were similar between 1955 (*Ep* = 0.50) and 2019 (*Ep* = 0.48), and Patuxent River *Ep* was 0.62 in 1955, 0.67 in 1979, 0.64 in 2015, and 0.68 in 2016. Patuxent River and Wicomico River estimates fell near those of other systems sampled in the same year, but Chester River estimates were consistently lower. Chester River is a small spawning area adjacent to the Head-of-Bay, the largest spawning area on the Atlantic coast, and spawners may be siphoned away from Chester River once Head-of-Bay spawning begins. It appears from these two years that there was a pulse of spawning (nearly all sites had eggs) that would have occurred earlier than Head-of-Bay spawning, followed by low presence when Head-of-Bay spawning would have been expected.

Examination of spatial and temporal trends of egg presence-absence in three infrequently sampled spawning areas (Patuxent, Wicomico, and Chester rivers) suggested two types of changes between the 1950s and 2015-2019: loss of spawning (or ability to sample spawning) in the upper reaches and earlier timing of the 21˚C cutoff. Uppermost stations in Chester River (agricultural watershed) and Patuxent River (suburban watershed) had become too shallow to sample between the 1950s and 2015-2019 due to siltation. There was a less consistent presence of eggs in sites located in the upper quarter-to-third (approximately) of these spawning areas that could be sampled during 2015-2019. The uppermost stations in Wicomico River, located within Salisbury, were accessible by boat since they were located in the port where a channel was maintained, but spawning was not indicated there in the 1950s or 2017-2018. Spawning in the Wicomico River occurred below the city in all years.

There appeared to be a general upward shift in spawning area water temperature in Choptank River during the standard period (April 1 – May 8), with three of the means, two minimums, and one maximum during 2014-2019 exceeding 1986-1991. The 21 ˚C cutoff was sometimes breached later in the 1950s and 1978-1979 than the 1990s or 2015-2019 in Patuxent River and Chester River, but not in Wicomico River. The scattershot nature of sampling during the 1950s makes those temperature related findings tenuous, but we hope to be able to investigate this further through the extensive Nanticoke River time-series. However, if this pattern holds then the window of optimal temperature conditions between too cold and too warm could be narrowing unless offset by compensating shift of early spawning with favorable temperatures. Temperatures above 21˚C fall on a rapidly ascending limb of instantaneous daily mortality rates that would negate benefit from later spawning (Secor and Houde 1995).

Of the three Choptank River water quality parameters summarized, pH offered the clearest indication of change between 1986-1991 and 2014-2019, from largely acidic and highly variable conditions to neutral and more stable. Latter pH levels were closer to those cited for productive hatcheries (Uphoff 1989). Stable conditions would have been beneficial for larvae due to absence of lethal high rate of pH change events (Hall et al. 1993). The successful 1989 Choptank River year-class was associated with the lowest mean pH of the available time-series, but it was also the least variable (Hall et al. 1993). The more acidic conditions in Choptank River surveys during the 1980s were consistent with descriptions of water quality described for in situ and on-site toxicity tests conducted in Choptank and Nanticoke rivers during 1984-1990 (acidic conditions and poor buffering coupled with concurrent elevated metals were associated with low survival of Striped Bass prolarvae during some trials; Hall et al. 1993; Richards and Rago 1999).

Distributions of pH during the 1990s in spawning areas other than Choptank River were generally in the upper range of those found in the Choptank River during 1986-1991 and exhibited wide variability. During 2014-2019, pH conditions in spawning areas with urbanizing watersheds (Patuxent and Wicomico rivers) generally exhibited higher means and greater variation in measurements than rural watersheds (Choptank and Chester rivers). Patuxent River pH means and ranges appeared to change little between 1991 and 2015, while pH means increased and range contracted in Chester River between 1996 and 2019.

Conductivity distributions in spawning areas with urban watersheds exhibited higher minimums than spawning areas in rural watershed during 2014-2019. Minimum conductivity in the Patuxent River spawning area increased by a factor of 2.2-2.4 between 1991 and 2015-2016. Wicomico River minimum conductivity was 1.4-2.3 times higher than Choptank River or Nanticoke River. Choptank River spawning area conductivity summaries offered little indication of change between 1986-1991 and 2014-2019. Minimum conductivity in Chester River was about 40% higher in 2019 than in 1996. Elevated salt levels by themselves in the upper spawning area should not be an issue for Striped Bass or (White Perch, and Yellow Perch) since they can be abundant in higher conductivity regions further downstream where freshwater is more mixed with intruding saltwater. However, elevated conductivity could indicate other stressors have increased as well. Higher conductivity in developed watersheds appeared to have a greater negative effect on anadromous Herring stream spawning (*Pherr*) than elevated conductivity in agricultural watersheds (see Job 1, Section 1). Correlations among DO, pH, water temperature, and conductivity exhibited different patterns among spawning areas and time periods, suggesting higher potential for phytoplankton influence on pH in some areas.

Late winter-early spring mean flows in Choptank River were moderately and negatively correlated with mean conductivity, but not with mean water temperature or pH. In the long-term (1957-2019), there appeared to be a weak influence of Choptank River March-April flow on $\log_e$-transformed JIs. However, patterning of residuals indicated the relationship was not stable over time with sets of years having stronger or weaker responses to flow. As many as six different periods were suggested and basic water quality data existed for all or portions of two of them (1982-1990 and 2008-2019). During 1986-1991, water temperatures during the standard period were usually lower and pH was lower and less stable than during 2014-2019; spawning stock was also at its nadir during 1982-1988. A particularly positive shift in the relationship of flow and the Choptank River JI resulted in frequent strong year-classes during 1991-2007. The most recent period that started in 2008 seems associated with lower flows in April and, while strong year-class have occurred (2011 and 2015), they appear to be less frequent than in the preceding period.

What may have triggered periods of enhanced or depressed larval survival? Specific factors and their combinations that were important are beyond the scope of this report, but a general description of major long-term natural and anthropogenic factors that could result in detrimental or beneficial larval habitat changes can be offered. It is possible, perhaps likely, that combinations of these factors have shifted from period to period.

Szuwalski et al. (2014) offered synchronous shifts in long-term climate patterns (such as the North Atlantic Oscillation or El Nino in the Atlantic) and recruitment as “low hanging fruit” in a search for environmental drivers of shifts in fish production. Wood and Austin (2009) detected a pattern of antagonistic recruitment trends in Chesapeake Bay recruitment (multiple Maryland and Virginia indices, 1968-2004) between anadromous species and shelf-spawners.

Shifts in recruitment success would last for a decade or more and the last shift detected, 1992, favored anadromous fish recruitment. Shifts in timing were similar with those reported here for the JI (a component of the data analyzed by Wood and Austin (2009)) and RLS. Winter-spring climate variability was considered a prime candidate as an environmental driver (Wood and Austin 2009) and multiple studies of Striped Bass recruitment have cited cooler and wetter winters and springs as favorable (Maryland Sea Grant 2009; Martino and Houde 2010; Millette et al. 2020).

Long-term warming could disrupt the timing of spawning and survival of eggs and early larvae (Maryland Sea Grant 2009). During the past 70 years the Chesapeake Bay has experienced nearly a 2˚C rise in mean surface water temperature (Maryland Sea Grant 2009). Comparisons of intermittent egg survey data that spanned the 1950s to the present in this report provided tenuous evidence that the late portion spawning season has contracted and high temperatures more lethal to larvae are appearing earlier (Secor and Houde 1995; Maryland Sea Grant 2009).

Acidic deposition, pesticides, and fertilizers could have been sources of toxic inorganic metals implicated in episodic mortalities of larvae in bioassays and surveys in some Chesapeake Bay spawning areas during the 1980s (May and McKinney 1981; Peterson et al. 1982; Uphoff 1992; Hall et al. 1993; Richards and Rago 1999). Extended life cycle tests with several species of fish found that early life stages were most sensitive and larvae were extremely sensitive to a variety of toxicants (McKim 1977; Peterson et al. 1982; Bengtson et al. 1993). Hall et al. (1993) identified multiple inorganic contaminants (aluminum, copper, zinc, cadmium, chromium, lead, and arsenic) associated with excessive mortality of Striped Bass prolarvae in bioassays conducted during 1984-1990 in Choptank, Nanticoke, and Potomac rivers spawning areas. Adverse water quality and contaminant conditions were not detected in Head-of-Bay experiments. These experiments were conducted after the majority of low RLS estimates had occurred; RLS estimates in the bottom quartile were concentrated in the period spanning 1977-1991 and most (7 of 9) occurred during 1977-1985. Acidic conditions and toxic metals were associated high mortality in bioassays conducted in Choptank and Nanticoke rivers, while low temperatures and toxic metals were identified with potentially stressful conditions in Potomac River. Job 3 in Uphoff et al. (2018) described aluminum, lead, and zinc as ubiquitous in Chesapeake Bay Program sediment data throughout Maryland's portion of Chesapeake Bay and these metals were often found in Striped Bass spawning and nursery areas. Ausili et al. (2020) identified marine sediments as a conservative means to assess whether contaminants had been present.

Improvement in rainfall pH, concentrations of sulphates and nitrates, and atmospheric contaminant deposition associated with the implementation of the federal Clean Air Act (1970) and its amendments (1977 and particularly 1990; US EPA 2020) have been documented along the Atlantic coast. Acid precipitation has been a major concern in eastern North America, especially in regions where the geological terrain and soils contain little acid buffering capacity (Clair et al. 2011). North American emissions peaked in the mid-1970s, though large-scale sampling for environmental effects did not begin in most regions until the early 1980s when the scope of environmental problems became most evident (Clair et al. 2011). Precipitation in Maryland in the early 1980s was as acidic as many regions of the northeastern United States and the spawning rivers of the eastern shore were susceptible to acidification (Hall et al. 1993). Increases in rainfall pH from ~4.0-4.4 during the late 1970s to early 2000s to ~4.8-5.1 by 2013-2018, and large decreases in sulfate and nitrate concentrations in rainfall have been reported from Vermont (Nevins et al. 2018), New Hampshire (Nelson and Neils 2015), New Jersey and Pennsylvania (Elkin et al. 2016; New Jersey Department of Environmental Protection 2016), and Maryland (Eshelman et al. 2013). Since wet Sulphur deposition sampling started in the early 1980s, measured values in Canada's Atlantic Provinces have decreased by approximately 50% (Clair et al. 2011).

Salt pollution and human-accelerated weathering have shifted the chemical composition of major ions in fresh water and increased salinization and alkalinization across North America (Kaushal et al 2018). Concurrent increases in conductivity, base cations, alkalinity, and pH across large geographic areas of North America began in the early and middle twentieth century. The concept of freshwater salinization syndrome links salinization and alkalinization. Coupled changes in conductivity, major ions, and pH related to freshwater salinization syndrome have influenced water quality of most of the stream flow in the eastern United States, including increased alkalinity in tidal waters of Potomac and Susquehanna rivers since 1985 (when monitoring began) and pH and conductivity the non-tidal Patuxent River. Larger rivers are prone to alkalinization and pH increases downstream due to cumulative effects of weathering (due to anthropogenic and natural causes) and biological processes. Densities of urban and agricultural land within a watershed can be strong predictors of base cations and pH in streams and rivers. In developed areas with colder climates, road salt is an important source of salinization. Anaerobic metabolism and decomposition can further increase alkalinity and pH in urbanized and agricultural watersheds. Agriculture can contribute significant bicarbonate and base cations from liming, potash, and fertilizer applications. Fertilizers can stimulate aquatic primary production, nitrification, and accelerated weathering of agricultural soils which increases pH and alkalinity. Elevated pH and base cations, such as calcium and magnesium, may reduce the bioavailability and toxicity of trace metals (Kaushal et al. 2018).

Human population growth since the 1950s added a suburban landscape layer to the Chesapeake Bay watershed (Brush 2009; Ator et al. 2020) that has been identified as a threat to Chesapeake Bay and its Striped Bass spawning areas (Chesapeake Bay Program or CBP 1999; Uphoff 2008; Maryland Sea Grant 2009). Three Maryland spawning areas, Wicomico River, Potomac River and Patuxent River, have urbanizing watersheds (Uphoff 2008; Maryland Sea Grant 2009). Urban stormwater is a major source of nutrients, sediment, pesticides, and trace metals in the Bay, and is a leading impediment to meeting water quality standards (Majcher et al. 2020; Chesapeake Bay Stormwater Network 2020). Although polluted urban stormwater has been recognized as a problem for two decades, federal, state, and local land development regulations have had little impact (Chesapeake Bay Stormwater Network 2020).

Striped bass spawning areas 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. Use of commercial fertilizers grew dramatically after the 1950s (Kemp et al. 2005) and other changes in the character of farming occurred between 1959 and 1974 (USDA 1978). On the Delmarva Peninsula, number of farms decreased, average farm size increased, grain and poultry farming greatly increased, and dairy and general farming declined (USDA 1978). No-till farming and cover crops (conservation agriculture) began to be widely adopted in the U.S. in the 1980s (Islam and Reeder 2014). Records from the Choptank River watershed indicated that their adoption was underway in the early 1980s (Uphoff 2008). These practices were widely adopted in the Chesapeake Bay watershed by the late 2000s (USDA NRCS 2013). Conservation agriculture is considered the primary strategy in modern farming that adequately protects the soil from erosion, while providing solid economic returns and enhanced environmental benefits (Islam and Reeder 2014).

Since 1983, a multi-state and federal multi-billion dollar effort, the Chesapeake Bay Program (CBP), has attempted to manipulate the Chesapeake Bay to a more desirable state largely by managing nutrients (Ator et al. 2020; CBP 2020). Agricultural nutrient management tied to the watershed-wide Chesapeake Bay Program led to downward trends in flow-adjusted nutrient concentrations in the watersheds of the major rivers of the Chesapeake Bay after 1985 (Sprague et al. 2000; Kemp et al. 2005), all of which were also Striped Bass spawning and larval nursery areas. Agricultural sediment and nutrient management practices in place in the Chesapeake Bay watershed were generally considered effective at reducing contaminants as well, although potential exists for improvement (Majcher et al 2020). An increasing trend in survival of Striped Bass postlarvae in Choptank River during 1980-1990 was strongly correlated with growth of agricultural best management practices (BMPs) that were designed to conserve soil and reduce nutrient runoff (Uphoff 2008). A positive byproduct of agricultural BMPs in Choptank River watershed may have been reduced contaminant runoff, even though BMPs were aimed at reducing nutrients. These associations suggested that agricultural BMPs were beneficial for larval Striped Bass survival (Uphoff 2008).

Presence-absence of Striped Bass eggs, when combined with JIs, provides an inexpensive, added value means of evaluating habitat conditions in Maryland's major spawning and larval nursery areas. At a minimum, *Ep* provides an indication that low JIs may or may not reflect low spawning stock independent of model based estimates. Estimates of RL*S* during 1955-2019 allowed for detection of patterns and trends in egg and larval survival that suggested underlying productivity shifts. A long-term pattern required for managing on a stock-recruit relationship basis was not supported by these indicators. These shifts were not always apparent in similar analyses based on SSB. The cathartic recovery of Atlantic Coast Striped Bass has largely been attributed to eliminating overfishing, but trends in *Ep* and RL*S* indicated that recruitment failure and recovery preceded overfishing and stock recovery, respectively. Water quality programs with sufficient stressor, spatial, and temporal scale oriented towards monitoring water quality and zooplankton conditions on the major spawning grounds are not in place nor are they likely to be. A few stations within larval nurseries are monitored for water quality on a monthly basis as part of the Chesapeake Bay Program and zooplankton monitoring was eliminated after 2002. Multiple factors that do not share the same trajectories may be influencing larval habitat conditions. First warning of deteriorating larval nursery area conditions in major spawning areas are likely to come from fish data analyses (JIs, *Ep*, RLS) rather than water quality monitoring. If an *Ep* sampling program is maintained or revived from time to time, upgrading water quality monitoring by programs routinely sampling the spawning areas (adding conductivity, pH, and alkalinity) to temperature and salinity should be considered. These variables may provide retrospective insight should problems arise.

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Table 2.1.1. Size of Striped Bass spawning areas in Maryland's portion of Chesapeake Bay (Hollis et al. 1967).

Spawning area	Hectares
Head-of-Bay	27,225
Potomac	22,162
Nanticoke	3,034
Choptank	1,734
Patuxent	1,011
Chester	786
Wicomico	649
Pocomoke	417
Blackwater	238
Transquaking	170
Manokin	23
Total	57,449

Table 2.1.2. Summary of methods used in historical and present studies to sample Striped Bass eggs. Time = tow duration; ≈ indicates tows standardized to distance. Tow Type: B = bottom, M = midwater, S = surface, O = oblique, I = inshore. Dia. = net diameter. Rigging equals net configuration; cone = conical net with bridle; and cone/trawl = conical net placed in trawl cod end. Set = number of each tow type made at a station; S = two single tows; C = combined samples; and P = paired tows. MD DNR is the Maryland Department of Natural Resources.

Year	Time (min)	Tow type	Dia. (m)	Rigging	Set	Mesh (mm)	Reference
Choptank River spawning area							
1980-1986	2	M,B,I	0.5	Cone in Trawl	2,S	0.5	Uphoff 1993
1987-1991	2	M,B,I	0.5	Cone in Trawl	1	0.5	Uphoff 1993
1994	2	S	0.5	Cone	1	0.5	Uphoff 1997
1997-2003	2	S	0.5	Cone	1	0.5	MD DNR 1997-2003
2004, 2013-2019	2	S	0.5	Cone	1	0.5	This report
Nanticoke River spawning area							
1955-1962	5	S	0.5	Cone	1	1.0*0.5	Hollis 1967
1963-1981	2	S	0.5	Cone	1	1.0*0.5	Hollis 1967
1992	5	O	0.6	Cone	2,P	0.5	Houde et al. 1996
1993	6	O	1.6	Cone	2,P	1.5	Houde et al. 1996
1994	2	S	0.5	Cone	1	0.5	Uphoff 1997
2004-2019	2	S	0.5	Cone	1	0.5	This report
Potomac River spawning area							
1956	5	S	0.5	Cone	1	1.0*0.5	Hollis 1967
1971	5	S	0.5	Cone	1	1.0*0.5	Unpublished MD DNR
1973	5	S	0.5	Cone	1	1.0*0.5	Unpublished MD DNR
1974	2.5	O	1	Cone	1	0.5	Setzler-Hamilton et al. 1981
1975	4	O	1	Cone	1	0.5	Setzler-Hamilton et al. 1981
1976-1977	6	O	1	Cone	1	0.5	Setzler-Hamilton et al. 1981
1987	5	O	0.6	Cone	2,C	0.5	Houde and Rutherford 1992
1988-1989	5	O	0.6	Cone	2,P	0.5	Houde and Rutherford 1992

Table 2.1.2 (continued)

Year	Time (min)	Tow type	Dia. (m)	Rigging	Set	Mesh (mm)	Reference
Elk River spawning area							
1956	5	S	0.5	Cone	1	1.0*0.5	Hollis 1967
1962	5	S	0.5	Cone	1	1.0*0.5	Hollis 1967
1974-1977	5	S,M,B	0.5	Cone	1	0.5	Kernehan et al. 1981
1984-1985	5	S,M,B	0.5	Cone	1	0.5	DEL 1986
1989	5	M,B,I	0.5	Cone in Trawl	1	0.5	Takacs 1990
1995	2,5	S	0.5	Cone	1	0.5	Uphoff 1997
1996	5	S	0.5	Cone	1	0.5	MD DNR 1996
Head of Chesapeake Bay							
1975-1977	5	S,M,B	0.5	Cone	1	0.5	Kernehan et. al. 1981
1984-1985	5	S,M,B	0.5	Cone	1	0.5	DEL 1986
1988	5	O	0.6	Cone	2,P	0.5	Houde and Rutherford 1992
1989	5	O	0.6	Cone	2,P	0.3	Houde and Rutherford 1992
Patuxent River spawning area							
1955	5	S	0.5	Cone	1	1.0*0.5	Hollis 1967
1978	6	O	1	Cone	3,P	0.5	Setzler et al. 1979
1979	2	O	1	Cone	1	0.5	Mihursky et al. 1980
1991	5	O	0.6	Cone	2,P	0.5	Secor et al. 1994
2015-2016	2	S	0.5	Cone	1	0.5	This report
Wicomico River spawning area							
2017-2018	2	S	0.5	Cone	1	0.5	This report
Chester River spawning area							
1955	5	O	0.5	Cone	1	1.0*0.5	Hollis 1967
1996	5	O	0.5	Cone	2,P	0.5	Burton et al. 1996
2019	2	S	0.5	Cone	1	0.5	This report

Table 2.1.3. Summary of data used to estimate the proportion of tows with eggs (E_p). Stations = number of stations used in presence-absence analysis; Tows with eggs = number of tows with eggs; and N = total number of tows.

Year	Stations	Tows with eggs	N	Ep	SD	CV
Choptank River spawning area						
1980	4	53	82	0.65	0.053	0.082
1981	3	74	112	0.66	0.045	0.068
1982	3	33	64	0.52	0.062	0.121
1983	4	47	132	0.36	0.042	0.117
1984	4	47	132	0.36	0.042	0.117
1985	3	54	110	0.49	0.048	0.097
1986	3	40	94	0.43	0.051	0.12
1987	16	49	87	0.56	0.053	0.094
1988	21	57	128	0.45	0.044	0.099
1989	21	103	133	0.77	0.036	0.047
1990	20	87	115	0.76	0.04	0.053
1991	20	78	95	0.82	0.039	0.048
1994	21	63	90	0.7	0.048	0.069
1997	20	89	112	0.79	0.038	0.048
1998	20	75	99	0.76	0.043	0.057
1999	19	81	99	0.82	0.039	0.047
2000	20	65	90	0.72	0.047	0.065
2001	12	31	47	0.66	0.069	0.105
2002	21	51	60	0.85	0.046	0.054
2003	20	70	92	0.76	0.044	0.058
2004	19	48	68	0.71	0.055	0.078
2013	19	73	92	0.79	0.042	0.053
2014	18	57	87	0.66	0.051	0.078
2015	18	47	83	0.57	0.054	0.096
2016	20	77	96	0.8	0.041	0.051
2017	17	65	90	0.72	0.047	0.065
2018	15	48	73	0.66	0.056	0.084
2019	17	51	72	0.71	0.054	0.076

Table 2.1.3 (continued)

Year	Stations	Tows with eggs	N	Ep	SD	CV
Nanticoke River spawning area						
1955	7	24	40	0.6	0.077	0.129
1956	9	41	53	0.77	0.057	0.074
1957	11	34	44	0.77	0.063	0.082
1961	4	53	61	0.87	0.043	0.05
1962	3	68	79	0.86	0.039	0.045
1963	3	92	101	0.91	0.028	0.031
1964	3	65	85	0.76	0.046	0.06
1965	3	53	59	0.9	0.039	0.044
1966	3	67	68	0.99	0.015	0.015
1967	3	70	92	0.76	0.044	0.058
1968	3	53	65	0.82	0.048	0.059
1969	3	48	65	0.74	0.055	0.074
1970	3	68	79	0.86	0.039	0.045
1971	3	32	34	0.94	0.04	0.043
1972	3	39	53	0.74	0.061	0.082
1973	3	27	41	0.66	0.074	0.112
1974	3	25	33	0.76	0.075	0.098
1975	3	24	37	0.65	0.078	0.121
1976	3	44	51	0.86	0.048	0.056
1977	3	27	35	0.77	0.071	0.092
1978	3	31	42	0.74	0.068	0.092
1979	3	40	44	0.91	0.043	0.048
1980	3	25	36	0.69	0.077	0.111
1981	3	31	51	0.61	0.068	0.112
1992	9	71	79	0.9	0.034	0.038
1993	11	55	63	0.87	0.042	0.048
1994	18	63	80	0.79	0.046	0.058
2004	14	45	57	0.79	0.054	0.068
2005	15	44	66	0.67	0.058	0.087
2006	18	54	77	0.7	0.052	0.074
2007	15	47	61	0.77	0.054	0.07
2008	17	60	96	0.63	0.049	0.079
2009	15	61	76	0.8	0.046	0.057
2010	18	58	69	0.84	0.044	0.052
2011	17	39	47	0.83	0.055	0.066
2012	18	34	54	0.63	0.066	0.104
2013	18	37	43	0.86	0.053	0.061
2014	18	43	62	0.69	0.059	0.084
2015	17	50	62	0.81	0.05	0.062
2016	18	43	50	0.86	0.049	0.057
2017	16	38	40	0.95	0.034	0.036
2019	18	39	56	0.7	0.061	0.088

Table 2.1.3 (continued)

Year	Stations	Tows with eggs	N	Ep	SD	CV
Potomac River spawning area						
1956	13	40	58	0.69	0.061	0.088
1971	10	20	37	0.54	0.082	0.152
1973	10	35	80	0.44	0.055	0.127
1974	11	35	66	0.53	0.061	0.116
1975	14	37	56	0.66	0.063	0.096
1976	9	68	108	0.63	0.046	0.074
1977	6	29	32	0.91	0.052	0.057
1987	8	16	32	0.5	0.088	0.177
1988	14	52	76	0.68	0.053	0.078
1989	9	173	256	0.68	0.029	0.043
Elk River spawning area						
1956	12	44	68	0.65	0.058	0.09
1962	5	22	26	0.85	0.071	0.084
1973	3	111	155	0.72	0.036	0.051
1974	5	139	210	0.66	0.033	0.049
1975	9	269	317	0.85	0.02	0.024
1976	8	227	289	0.79	0.024	0.031
1977	10	222	271	0.82	0.023	0.029
1985	4	95	167	0.57	0.038	0.067
1989	20	39	63	0.62	0.061	0.099
1995	20	59	69	0.86	0.042	0.05
1996	19	71	90	0.79	0.043	0.055
Head-of-Bay spawning area						
1975	12	300	350	0.86	0.019	0.022
1976	13	269	346	0.78	0.022	0.029
1977	16	293	352	0.83	0.02	0.024
1984	3	88	225	0.39	0.033	0.083
1985	6	109	202	0.54	0.035	0.065
1988	10	23	43	0.53	0.076	0.142
1989	9	125	167	0.75	0.034	0.045
Patuxent River spawning area						
1955	6	16	26	0.62	0.095	0.155
1978	5	32	35	0.91	0.047	0.052
1979	6	24	36	0.67	0.079	0.118
1991	7	35	43	0.81	0.059	0.073
2015	11	47	73	0.64	0.056	0.087
2016	10	68	100	0.68	0.047	0.069
Wicomico River spawning area						
2017	7	26	37	0.7	0.075	0.107
2018	9	38	47	0.81	0.057	0.071
Chester River spawning area						
1955	5	15	30	0.5	0.091	0.183
1996	12	89	123	0.72	0.04	0.056
2019	11	12	25	0.48	0.1	0.208

Table 2.1.4. Baywide proportion of tows with Striped Bass eggs (*Ep*) estimated for spawning areas sampled for the Maryland juvenile index. N with eggs = number of samples with eggs; N Total = total number of samples. High CI and Low CI refer to 90% confidence interval boundaries.

Year	N with eggs	N Total	*Ep*	SD	CV	High CI	Low CI
1955	25	40	0.63	0.08	0.12	0.75	0.5
1956	128	179	0.72	0.03	0.05	0.77	0.66
1957	35	44	0.8	0.06	0.08	0.9	0.7
1958							
1959							
1960							
1961	54	61	0.89	0.04	0.05	0.95	0.82
1962	92	105	0.88	0.03	0.04	0.93	0.82
1963	93	101	0.92	0.03	0.03	0.97	0.88
1964	66	85	0.78	0.05	0.06	0.85	0.7
1965	54	59	0.92	0.04	0.04	0.98	0.86
1966	68	68	1	0		1	1
1967	71	92	0.77	0.04	0.06	0.84	0.7
1968	54	65	0.83	0.05	0.06	0.91	0.75
1969	49	65	0.75	0.05	0.07	0.84	0.67
1970	69	79	0.87	0.04	0.04	0.94	0.81
1971	54	71	0.76	0.05	0.07	0.84	0.68
1972	40	53	0.75	0.06	0.08	0.85	0.66
1973	176	276	0.64	0.03	0.05	0.69	0.59
1974	202	309	0.65	0.03	0.04	0.7	0.61
1975	364	443	0.82	0.02	0.02	0.85	0.79
1976	384	505	0.76	0.02	0.02	0.79	0.73
1977	352	419	0.84	0.02	0.02	0.87	0.81
1978	32	42	0.76	0.07	0.09	0.87	0.65
1979	41	44	0.93	0.04	0.04	0.99	0.87
1980	80	118	0.68	0.04	0.06	0.75	0.61
1981	107	163	0.66	0.04	0.06	0.72	0.6
1982	34	64	0.53	0.06	0.12	0.63	0.43
1983	48	132	0.36	0.04	0.12	0.43	0.29
1984	137	357	0.38	0.03	0.07	0.43	0.34
1985	165	312	0.53	0.03	0.05	0.58	0.48
1986	41	94	0.44	0.05	0.12	0.52	0.35
1987	50	87	0.57	0.05	0.09	0.66	0.49
1988	58	128	0.45	0.04	0.1	0.53	0.38
1989	144	196	0.73	0.03	0.04	0.79	0.68
1990	88	115	0.77	0.04	0.05	0.83	0.7
1991	79	95	0.83	0.04	0.05	0.89	0.77
1992							
1993							
1994	128	170	0.75	0.03	0.04	0.81	0.7
1995	60	69	0.87	0.04	0.05	0.94	0.8
1996	72	90	0.8	0.04	0.05	0.87	0.73
1997	90	112	0.8	0.04	0.05	0.87	0.74

Table 2.1.4 (continued)

Year	N with eggs	N Total	*Ep*	SD	CV	High CI	Low CI
1998	76	99	0.77	0.04	0.06	0.84	0.7
1999	82	99	0.83	0.04	0.05	0.89	0.77
2000	66	90	0.73	0.05	0.06	0.81	0.66
2001	32	47	0.68	0.07	0.1	0.79	0.57
2002	52	60	0.87	0.04	0.05	0.94	0.79
2003	71	92	0.77	0.04	0.06	0.84	0.7
2004	95	125	0.76	0.04	0.05	0.82	0.7
2005	45	66	0.68	0.06	0.08	0.78	0.59
2006	55	77	0.71	0.05	0.07	0.8	0.63
2007	48	61	0.79	0.05	0.07	0.87	0.7
2008	61	96	0.64	0.05	0.08	0.72	0.55
2009	62	76	0.82	0.04	0.05	0.89	0.74
2010	59	69	0.86	0.04	0.05	0.92	0.79
2011	40	47	0.85	0.05	0.06	0.94	0.77
2012	35	54	0.65	0.06	0.1	0.76	0.54
2013	112	135	0.83	0.03	0.04	0.88	0.78
2014	102	149	0.68	0.04	0.06	0.75	0.62
2015	99	145	0.68	0.04	0.06	0.75	0.62
2016	122	146	0.84	0.03	0.04	0.89	0.79
2017	105	130	0.81	0.03	0.04	0.86	0.75
2018	49	73	0.67	0.05	0.08	0.76	0.58
2019	92	128	0.72	0.04	0.06	0.78	0.65

Table 2.1.5. Correlations (r) among Striped Bass spawning area juvenile indices.

Spawning area JI	Statistic	Choptank	Head-of-Bay	Nanticoke	Potomac
	r	0.32			
Head-of-Bay	P	0.0113			
	N	63			
	r	0.66	0.41		
Nanticoke	P	<0.0001	0.0008		
	N	63	63		
	r	0.47	0.47	0.35	
Potomac	P	<0.0001	<0.0001	0.0046	
	N	63	63	63	
	r	0.46	0.56	0.41	0.86
Patuxent	P	0.0038	0.0003	0.0109	<0.0001
	r	37	37	37	37

Table 2.1.6. Summary of Striped Bass egg presence or absence in Patuxent River ichthyoplankton surveys. Sum is the number of times eggs were present at a site across all dates. Dates end when water temperature reached 21˚C and sites reflect 2.0 ‰ salinity or less. Location = site names used by Secor et al. (1994). FHEP stations are sites sampled in 2015-2016 (see Figure 2.1.3 for locations).

Location	FHEP site	4/21/1954	4/22/1953	4/22/1955	4/24/1955	4/29/1954	4/29/1955
Magruder’s Landing	3	1	0				
White's Landing	6	1	0	1		1	
Nottingham	8	1	1	1	1	1	1
Ferry's Landing	9	1	0	1	1	1	1
Above Lyon's Creek	11	0		1	1		1
Above Jug Bay		0	1		0	0	

Location	FHEP site	4/30/1953	4/30/1959	5/5/1955	5/7/1953	5/7/1954	5/12/1955
Magruder’s Landing	3	1			0		
White's Landing	6	1	0	1	1	0	
Nottingham	8	0	0	0	1	1	1
Ferry's Landing	9	0	0	1	1	1	0
Above Lyon's Creek	11		0	0	1	1	0
Above Jug Bay		0		0	0	0	0

Location	FHEP site	5/17/1954	5/25/1954	Sum
Magruder’s Landing	3	0		2
White's Landing	6	1	1	8
Nottingham	8	1	0	9
Ferry's Landing	9	0	1	9
Above Lyon's Creek	11	0	1	6
Above Jug Bay			0	1

Table 2.1.6 (continued)

Location	FHEP site	4/4/1991	4/8/1991	4/11/1991	4/15/1991	4/18/1991	4/22/1991
Truman's Point		0	0				
Deep Landing		1	0	0	0	1	0
Power Cable	1	1	1	1	1	1	0
Magruder's Landing	3						
Lower Marlboro	4	0	1	1	1	1	0
White's Landing	6	0	1	1	1	1	0
Nottingham	8	0	1	1	1	1	1
Ferry's Landing	9			0	0	1	1
Lyon's Cr	10						1

Location	FHEP site	4/27/1991	4/29/1991	Sum
Truman's Point				0
Deep Landing		0	1	3
Power Cable	1	1	1	7
Magruder's Landing	3			
Lower Marlboro	4	1	1	6
White's Landing	6	1	1	6
Nottingham	8	1	1	7
Ferry's Landing	9	1	1	4
Lyon's Cr	10	1		2

Table 2.1.6 (continued)

Location	FHEP site	4/8/2015	4/14/2015	4/16/2015	4/21/2015	4/23/2015	4/28/2015
Power Cable	1	0	1	1	1	1	0
	2	0	1	1	1	1	1
Magruder's Landing	3	1		1	1	1	1
Lower Marlboro	4	1	1	1	1	1	1
	5	1	1	1	1	1	1
White's Landing	6	1	1	1	0	1	1
	7	1		1	0	1	0
Nottingham	8	1	1				0
Ferry's Landing	9	1	1	0			0
Lyon's Creek	10	0	1	0			0

Location	FHEP site	4/30/2015	5/5/2015	Sum
Power Cable	1	0	1	5
	2	0	0	5
Magruder's Landing	3	1	0	6
Lower Marlboro	4	1	0	7
	5	1	0	7
White's Landing	6	1	1	7
	7	0	0	3
Nottingham	8	0	0	2
Ferry's Landing	9	0	0	2
Lyon's Creek	10	0	0	1

Table 2.1.6 (continued)

Location	FHEP site	3/30/2016	4/1/2016	4/6/2016	4/8/2016	4/12/2016	4/14/2016
Power Cable	1	0		1	1	0	
	2	1		0	1	0	
Magruder's Landing	3	1	1	1	0	1	
Lower Marlboro	4	0	1	0	1	1	0
	5	1	1	1	1	0	1
White's Landing	6	1	1	0	1	1	1
	7	1	1	1	1	1	1
Nottingham	8	0	0	0	0	1	1
Ferry's Landing	9	0	1	0	0	0	1
Lyon's Creek	10	0	0	0	0	0	1

Location	FHEP site	4/14/2016	4/19/2016	4/21/2016	4/26/2016	4/29/2016	5/5/2016	Sum
Power Cable	1		0				0	2
	2		1	1			1	5
Magruder's Landing	3		1	1			1	7
Lower Marlboro	4		1	1	0		0	5
	5	1	1	1	1	0	1	10
White's Landing	6	1	1	1	1	0	1	10
	7	1	1	1	1	0	1	11
Nottingham	8		1	1	1	0	1	6
Ferry's Landing	9		1	1	1	1	1	7
Lyon's Creek	10		0	1	1	1	1	5

Table 2.1.7. Summary of Striped Bass egg presence or absence in Wicomico River ichthyoplankton surveys during 1954, 1957, and 1959, 2017, and 2018. Sum is the number of times eggs were present at a site across all dates. Dates end when water temperature reached 21 ˚C and sites reflect 2.0 ‰ salinity or less. NM = nautical miles; station locations were estimated in NM from mouth of river. FHEP stations are sites used in 2017-2018 (see Figure 2.1.3 for locations).

FHEP station	NM	4/12/1954	4/17/1959	4/19/1954	4/23/1957	4/23/1957	4/27/1954	4/30/1959	Sum
	2		1						1
	3								0
1	4				1				1
3	5	1			1	1	0	1	3
5	6	1		1	1			1	3
8	7	1		0	1			0	1
12	8	0		0	0			0	0
	9	0				0		0	0

FHEP station	3/30/2017	4/4/2017	4/6/2017	4/11/2017	4/13/2017	4/18/2017	Sum
1	1			1	1	0	3
2	1	1	1	1	1	1	6
3	1	1	1	1	1	1	6
4		0	1	1			2
5	0	1	1	1	0	0	3
6	0	1	1	1	0	0	3
7	0	1	1	1	0	0	3
8	0	0	0		0	0	0
9		0		0	0	0	0
10	0		0	0		0	0
11	0	0	0	0	0	0	0
12	0	0	0		0		0

Table 2.1.7 (continued)

FHEP station	4/17/2018	4/20/2018	4/23/2018	4/25/2018	4/30/2018	5/2/2018	Sum
1	1	1	1		1	1	5
2	1	1	1	0	1	1	5
3	1	1	1	1	1	1	6
4			1	1	1	1	4
5	1	1	1	1	1	1	6
6	1	1	1	1	1	0	5
7	1	0	1	1		1	4
8	0	0		1	1		2
9	0	0	0	1		0	1
10		0		0	0		0
11	0	0	0	0	0	0	0
12	0		0		0	0	0

Table 2.1.8. Summary of Striped Bass egg presence or absence in Chester River ichthyoplankton surveys during 1955, 1996, and 2019. Sum is the number of times eggs were present at a site across all dates. Dates end when water temperature reached 21 ˚C and sites reflect 2.0 ‰ salinity or less. FHEP stations are sites used in 2017-2018 (see Figure 2.1.3 for locations). NM = nautical miles; station locations were estimated in NM from mouth of river. River RKM = river kilometers from river mouth and was used by Burton et al. (1996) to designate sites.

FHEP site	NM	4/14/1955	4/20/1955	4/27/1955	5/4/1955	5/13/1955	5/19/1955	Sum
3	22	0	1	0	0	0	0	1
5	24	0	1	1	0	0	0	2
7	26	0	1	0	1	0	1	3
9	28	1	1	1	0	1	1	5
11	30	1	1	1	1	0	0	4

FHEP site	River KM	4/10/1996	4/12/1996	4/15/1996	4/17/1996	4/19/1996	4/22/1996	4/24/1996	4/29/1996
1	12	0	0	0					
2	14	0	0					1	1
3	16	0	0	0			1	1	1
4	18	0	0	0	0		1	1	1
5	20	0	0	1	1	0	1	1	1
6	22	0	0	1	1	1	1	1	1
7	24	0	0	1	1	0	1	1	1
8	26	0	0	1	1	1	1	1	1
9	28	0	0	1	1	1	1	1	1
10	30	1	0	1	1	1	1	1	1
11	32	0	0	1	1	1	1	1	1
12	34	0	0	1	0	0	1	1	0
	36			0	1	0	1	1	1

Table 2.1.8 (continued)

FHEP site	Station	5/1/1996	5/3/1996	5/6/1996	Sum
1	12				0
2	14	1	1	1	5
3	16	1	1	0	5
4	18	1	1	1	6
5	20	1	1	1	8
6	22	1	1	1	9
7	24	1	1	1	8
8	26	1	1	1	9
9	28	1	1	1	9
10	30	1	1	1	10
11	32	1	1	1	9
12	34	1	0	1	5
	36	1	0	1	6

FHEP site	Station	4/8/2019	4/10/2019	4/15/2019	4/17/2019	4/22/2019	Sum
1	12				0		0
2	14				0		0
3	16				1	0	1
4	18	0	1		1	0	2
5	20	0	1		0	1	2
6	22	1	1		0	1	3
7	24	1	1		0	1	3
8	26	1	1		0	0	2
9	28	1	1	0	0	0	2
10	30	1	1	0	0	0	2
11	32	0	0	0	0	0	0
12	34	0	0	0	0	0	0

Table 2.1.9. Maryland Department of Planning estimates of percent major land use for portion of watersheds draining into the Striped Bass spawning areas monitored for development impacts during 2013-2019. Estimates were for land use in 2010 and are the most recent available.

River	% Agriculture	% Forest	% Urban	% Wetland
Chester	66.6	24.5	7.8	0.8
Choptank	60.9	25.6	11.2	2.1
Nanticoke	45	39.4	8.1	7.4
Patuxent	20.5	35.1	41.7	1
Wicomico	30.1	36.8	29.9	2.3

Table 2.1.10. Summary statistics for pH measurements made in the Striped Bass spawning areas during April 1 – May 8, by year.

Choptank River						
Year	1986	1987	1988	1989	1990	1991
Mean	7.04	6.76	6.93	6.17	6.97	6.74
Median	7.15	6.78	7.02	6.18	7.03	7.02
95th%	7.76	7.07	8.01	6.39	7.19	7.51
5th%	6.71	6.54	6.53	6	6.78	6.13
Minimum	5.75	6.3	6.45	5.78	6.5	5.86
Maximum	9.15	7.45	8.4	6.46	7.34	8.2
Diff 90% CI	1.05	0.53	1.48	0.39	0.41	1.38
N	628	249	122	139	150	222

Choptank River						
Year	2014	2015	2016	2017	2018	2019
Mean	7.09	7.39	7.22	7.23	7.12	7.18
Median	7.19	7.42	7.27	7.27	7.15	7.25
95th%	7.8	7.83	7.68	7.55	7.68	7.55
5th%	6.8	7.11	6.92	7.01	6.83	6.92
Minimum	6.7	7.05	6.68	6.87	6.71	6.56
Maximum	8	8.07	7.85	7.76	7.86	8.1
Diff 90% CI	1	0.72	0.76	0.54	0.85	0.63
N	96	96	88	100	90	100

Nanticoke River		
Year	1992	1993
Mean	7.25	7.03
Median	7.28	7.15
95th%	7.78	7.59
5th%	6.99	6.61
Minimum	6.9	6.3
Maximum	7.98	7.64
Diff 90% CI	0.79	0.98
N	61	63

Patuxent River			
Year	1991	2015	2016
Mean	7.57	7.52	7.6
Median	7.6	7.51	7.61
95th%	8.5	7.96	8.56
5th%	7.3	7.29	7.27
Minimum	7.2	7.22	7.21
Maximum	8.7	8.29	8.78
Diff 90% CI	1.2	0.67	1.29
N	36	75	93

Wicomico River		
Year	2017	2018
Mean	7.55	7.9
Median	7.55	8.13
95th%	8.2	8.83
5th%	7.27	7.44
Minimum	7.22	7.37
Maximum	8.27	8.97
Diff 90% CI	0.93	1.39
N	40	89

Chester River		
Year	1996	2019
Mean	7.17	7.37
Median	7.3	7.37
95th%	8.13	7.66
5th%	6.8	7.17
Minimum	6.6	7.1
Maximum	9	7.84
Diff 90% CI	1.33	0.49
N	175	83

Table 2.1.11. Summary statistics for conductivity (uS/cm) measurements made in the Striped Bass spawning areas during April 1 – May 8, by year.

Year	1986	1987	1988	1989	1990	1991	1992	1993	2014	2015	2016	2017	2018	2019
Choptank														
Mean	858	893	910	426	650	603			669	673	963	991	619	464
Median	560	372	363	194	161	217			177	208	416	535	207	166
95th%	2480	3175	3686	1824	3053	3092			3101	2956	3538	3054	2652	2185
5th%	126	144	186	132	136	147			118	137	150	149	135	128
Minimum	94	132	177	93	129	126			111	126	93	135	122	124
Maximum	3950	4410	4390	3750	3660	4090			4881	3934	4389	3664	3770	3496
Diff 90% CI	2354	3031	3500	1692	2917	2945			2983	2819	3388	2905	2517	2057
N	628	250	122	148	144	212			96	96	88	100	90	100
Nanticoke														
Mean							523	434	827	624	991	1242		634
Median							310	165	228	237	390	525		201
95th%							1766	1750	3245	2421	3648	3494		2311
5th%							100	95	135	139	145	148		136
Minimum							96	90	132	117	141	119		100
Maximum							2300	2299	3709	3662	3849	3695		3184
Diff 90% CI							1666	1655	3110	2281.35	3503.36	3346		2175
N							64	73	61	64	57	44		67
Patuxent														
Mean						536				670	1210			
Median						340				422	890			
95th%						1595				2268	3132			
5th%						150				327	386			
Minimum						142				317	378			
Maximum						2300				3341	4299			
Diff 90% CI						2158				1941	2746			
N						35				73	91			

Table 2.1.11 (continued)

Year	1996	2019
Chester		
Mean	1046	875
Median	411	449
95th%	3870	2802
5th%	128	161
Minimum	97	140
Maximum	4500	3660
Diff 90% CI	3742	2641
N	175	83

Year	2017	2018
Wicomico		
Mean	792	429
Median	406	293
95th%	2809	926
5th%	240	209
Minimum	217	199
Maximum	3846	2235
Diff 90% CI	2569	717
N	40	42

Table 2.1.12. Summary statistics for water temperature (˚C) measurements made in the Striped Bass spawning areas during April 1 – May 8, by year.

Year	1986	1987	1988	1989	1990	1991	1992	1993	1996	2014	2015	2016	2017	2018	2019
Choptank															
Mean	15.2	14.11	14.31	15.63	15.94	15.83				15.06	16.16	15.56	17.77	14.33	16.83
Median	14.8	14.1	14.29	16.09	15.93	15.84				15.4	16.12	15.81	17.44	13.59	17.49
95th%	19.2	16.98	16.65	18.23	20.69	20.05				18.84	20.91	19.42	22.61	19.63	20.51
5th%	12.31	11.3	11.92	11.76	10.07	10.02				13.2	12.18	11.77	13.05	10.58	11.61
Minimum	11.66	10.5	11.29	10.68	9.71	9.27				10.7	11.9	10.96	12.41	10.11	10.96
Maximum	21.8	17.74	18	18.71	21	21.26				18.5	21.86	19.88	23.09	20.4	20.98
Diff 90% CI	6.89	5.68	4.73	6.48	10.62	10.03				5.65	8.74	7.65	9.56	9.05	8.9
N	645	249	122	148	150	222				96	96	88	100	90	100
Nanticoke															
Mean							15.68	17.03		15.12	15.4	15.26	17.04		16.69
Median							15.4	16.5		15.4	15.55	15.3	17.8		17.7
95th%							18.4	20.5		16.3	17.9	18.82	19.6		19.77
5th%							12.62	14.2		13	12.23	12.44	12.8		11.96
Minimum							9.7	12.5		12	11.9	11.9	12.5		11.2
Maximum							18.9	21.9		17.8	18.1	19	20		20.1
Diff 90% CI							5.79	6.3		3.3	5.67	6.38	6.8		7.81
N							64	73		61	64	57	44		67

Table 2.1.12 (continued)

Year	1986	1987	1988	1989	1990	1991	1992	1993	1996	2014	2015	2016	2017	2018	2019
							Patuxent								
Mean						16.17					16.12	15.1			
Median						16.3					15.67	15.04			
95th%						18.73					19.68	18.99			
5th%						13.14					13.66	11.84			
Minimum						12.8					13.5	11.33			
Maximum						18.9					20.04	19.22			
Diff 90% CI						5.59					6.02	7.16			
N						35					75	91			

Year	1996	2019
	Chester	
Mean	14.22	16.57
Median	14.58	17.26
95th%	19.38	20.3
5th%	8.72	10.42
Minimum	7.56	10.09
Maximum	20.59	20.66
Diff 90% CI	10.66	9.88
N	175	83

Year	2017	2018
	Wicomico	
Mean	17.42	16.35
Median	17.04	15.71
95th%	20.26	19.31
5th%	15	13.76
Minimum	14.69	13.65
Maximum	20.34	19.58
Diff 90% CI	5.27	5.55
N	40	42

Table 2.1.13. Correlations (r) among dissolved oxygen (DO), conductivity, pH, and water temperature measured during April 1–May 8, aggregated over survey years and by spawning area. Gray shading of r indicates a correlation moderate to strong enough to be of interest.

Choptank River		2014-2019			1986-1991		
		DO	Conductivity	pH	DO	Conductivity	pH
Temperature	r	-0.4521	0.06379	0.10397	-0.3658	0.05108	0.20615
	P	<.0001	0.1249	0.0122	<.0001	0.0477	<.0001
	N	580	580	580	1427	1503	1509
DO	r		-0.10511	0.43494		-0.14121	0.14285
	P		0.0113	<.0001		<.0001	<.0001
	N		580	580		1414	1420
Conductivity	r			0.05284			0.07324
	P			0.2039			0.0046
	N			580			1494
Patuxent River		2015-2016			1991		
Temperature	r	0.15187	0.13928	0.48278	-0.0578	0.00514	0.46524
	P	0.0442	0.0652	<.0001	0.716	0.9763	0.0028
	N	176	176	176	42	36	39
DO	r		-0.03497	0.77663	1	-0.62501	0.6383
	P		0.6449	<.0001		<.0001	<.0001
	N			176	42	36	39
Conductivity	r			-0.0493			-0.29046
	P			0.5157			0.1011
	N			176			33
Chester River		2019			1996		
Temperature	r	-0.84027	-0.16501	-0.58664	-0.87489	0.00574	-0.6149
	P	<.0001	0.136	<.0001	<.0001	0.94	<.0001
	N	83	83	83	175	175	175
DO	r		0.30857	0.72869		-0.02412	0.6748
	P		0.0045	<.0001		0.7514	<.0001
	N		83	83		175	175
Conductivity	r			0.67605			0.40746
	P			<.0001			<.0001
	N			83			175
Wicomico River		2017-2018					
Temperature	r	-0.3391	-0.06907	-0.1168			
	P	0.0001	0.4516	0.2021			
	N	121	121	121			
DO	r		-0.41834	0.7235			
	P		<.0001	<.0001			
	N		121	121			
Conductivity	r			-0.3053			
	P			0.0007			
	N			121			

Table 2.1.14. Correlations of water quality variables summarized over a standardized time period (April 1–May 8) and monthly average Choptank River discharge (ft^3 / second) for March and April. Gray shading indicates correlations strong enough for consideration. Water quality was summarized for 1986-1990 and 2014-2019.

Variable	Statistic (N = 12)	April flow	Mean pH	Mean conductivity	Mean temperature
March flow	r	0.56951	0.00492	-0.61756	0.15407
	P	0.0532	0.9879	0.0324	0.6326
April flow	r		-0.39556	-0.72873	0.24027
	P		0.2031	0.0072	0.4519
Mean pH	r			0.39133	0.28238
	P			0.2084	0.3739
Mean conductivity	r				-0.10195
	P				0.7525

Figure 2.1.1. Baywide Striped Bass juvenile indices (geometric mean catch per standard seine haul; diamonds) and their 90% confidence interval (line) estimated for Maryland's major spawning areas during 1957-2019 (Durrell and Weedon 2019).

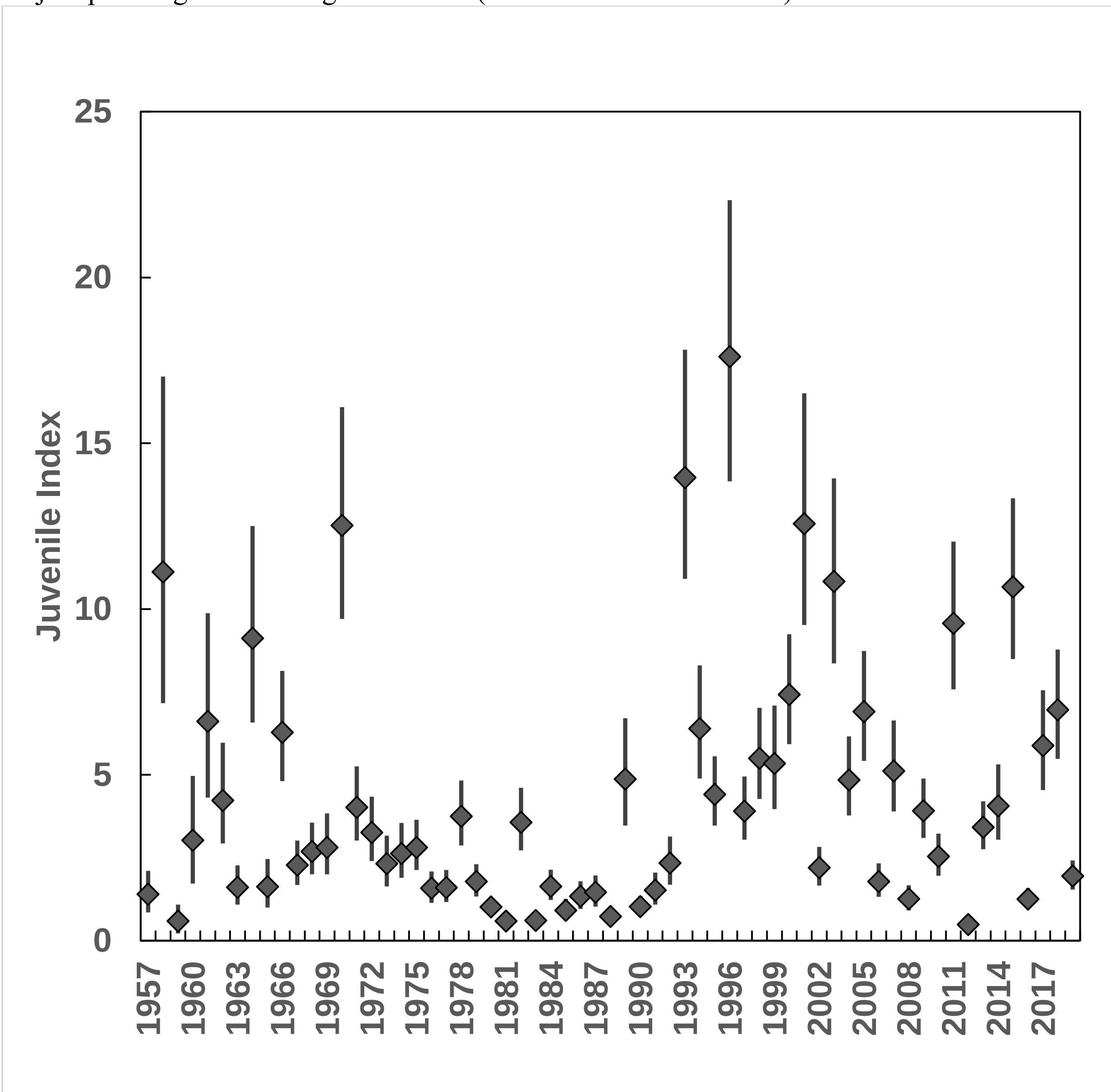


Figure 2.1.2. Location of Striped Bass spawning and larval nursery habitat in MD's portion of Chesapeake Bay based on average salinity less than 2 ppt (see Job 3, Uphoff et al. 2017). These areas encompass spawning areas described in (Hollis 1967), but do not exactly duplicate them.

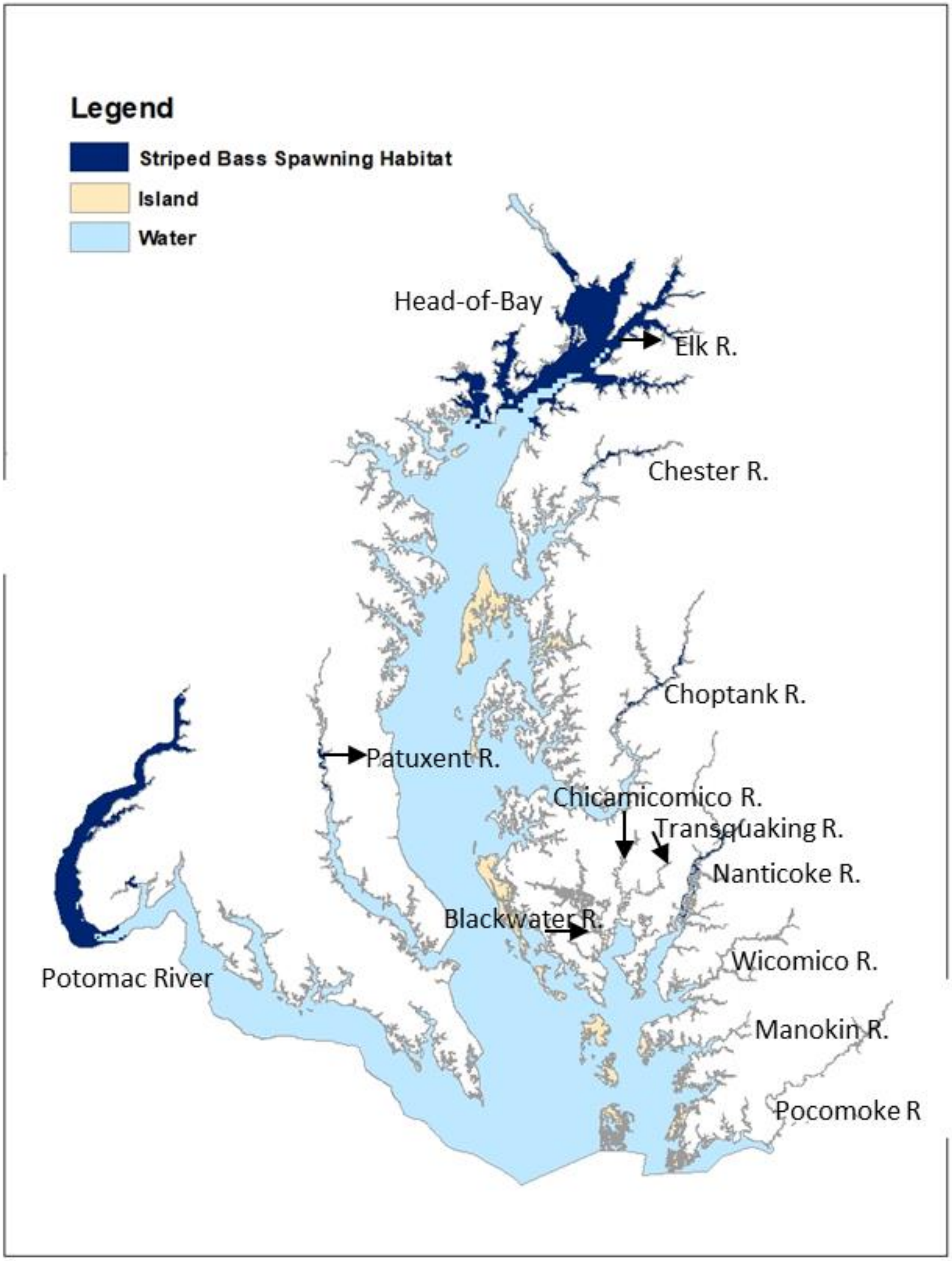


Figure 2.1.3. Location of sampling stations within Striped Bass spawning areas surveyed during 2013-2019. Yellow Perch sites are the same as sites sampled for Striped Bass eggs.

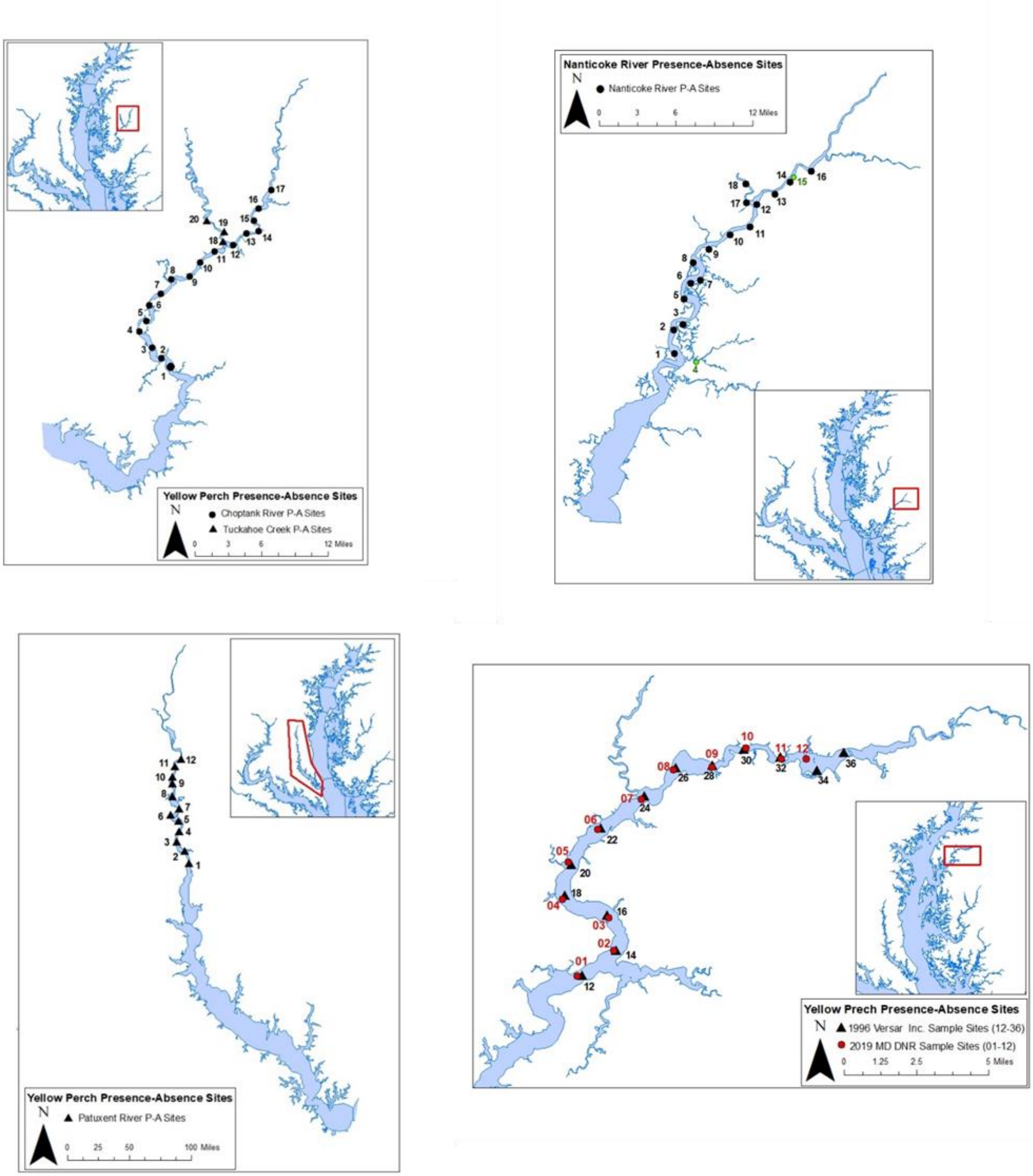


Figure 2.1.4. Spawning area proportions of tows with Striped Bass eggs (*Ep*) estimated from all surveys conducted during 1955-2019. Elk River represents a portion of the Head-of-Bay. Estimates include surveys using single and combined multiple tows.

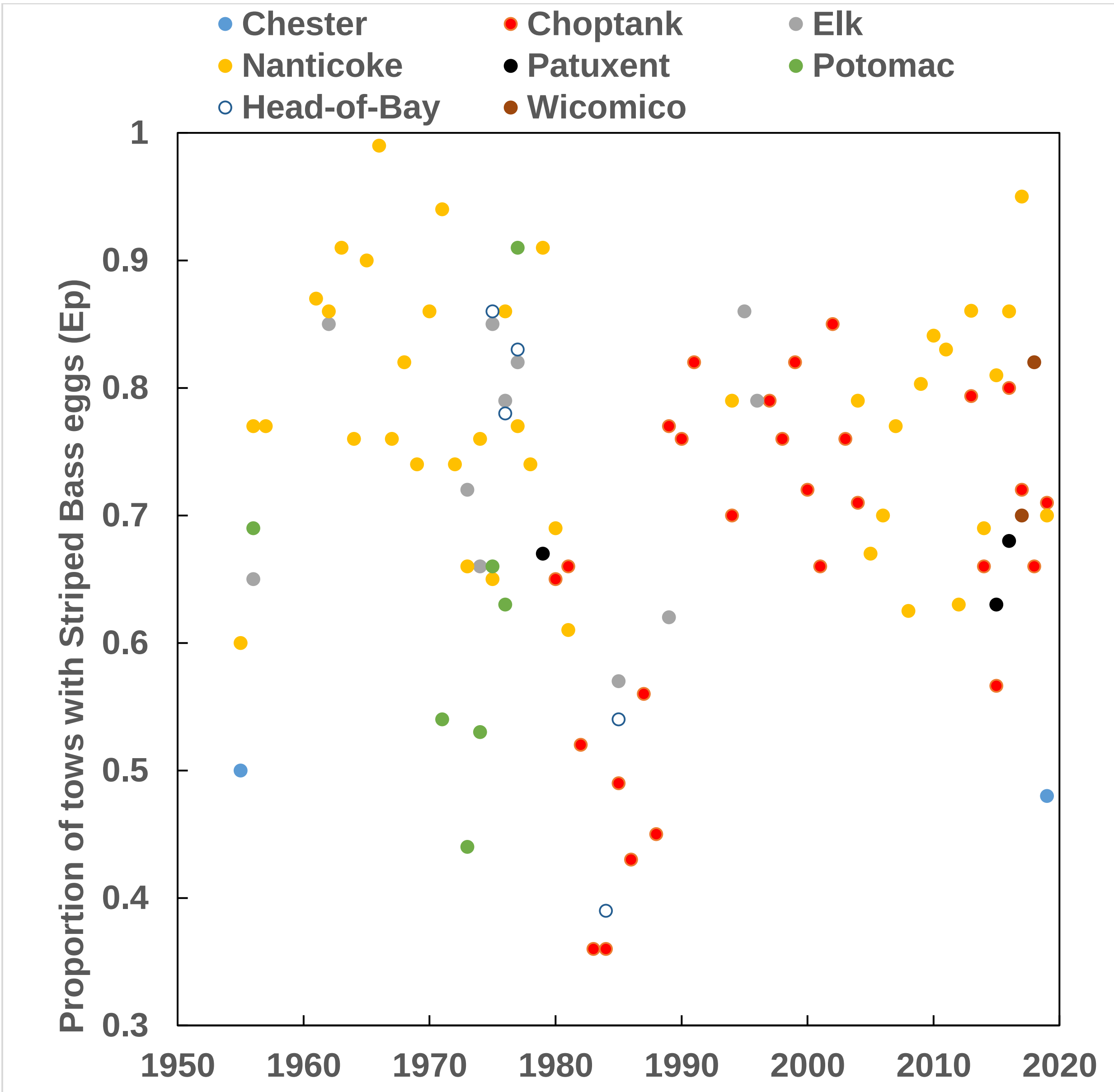


Figure 2.1.5. Spawning area specific proportion of tows with Striped Bass eggs (*Ep*) estimated from surveys in juvenile index rivers conducted during 1955-2019. Elk River represents a portion of the Head-of-Bay. Single tow surveys only.

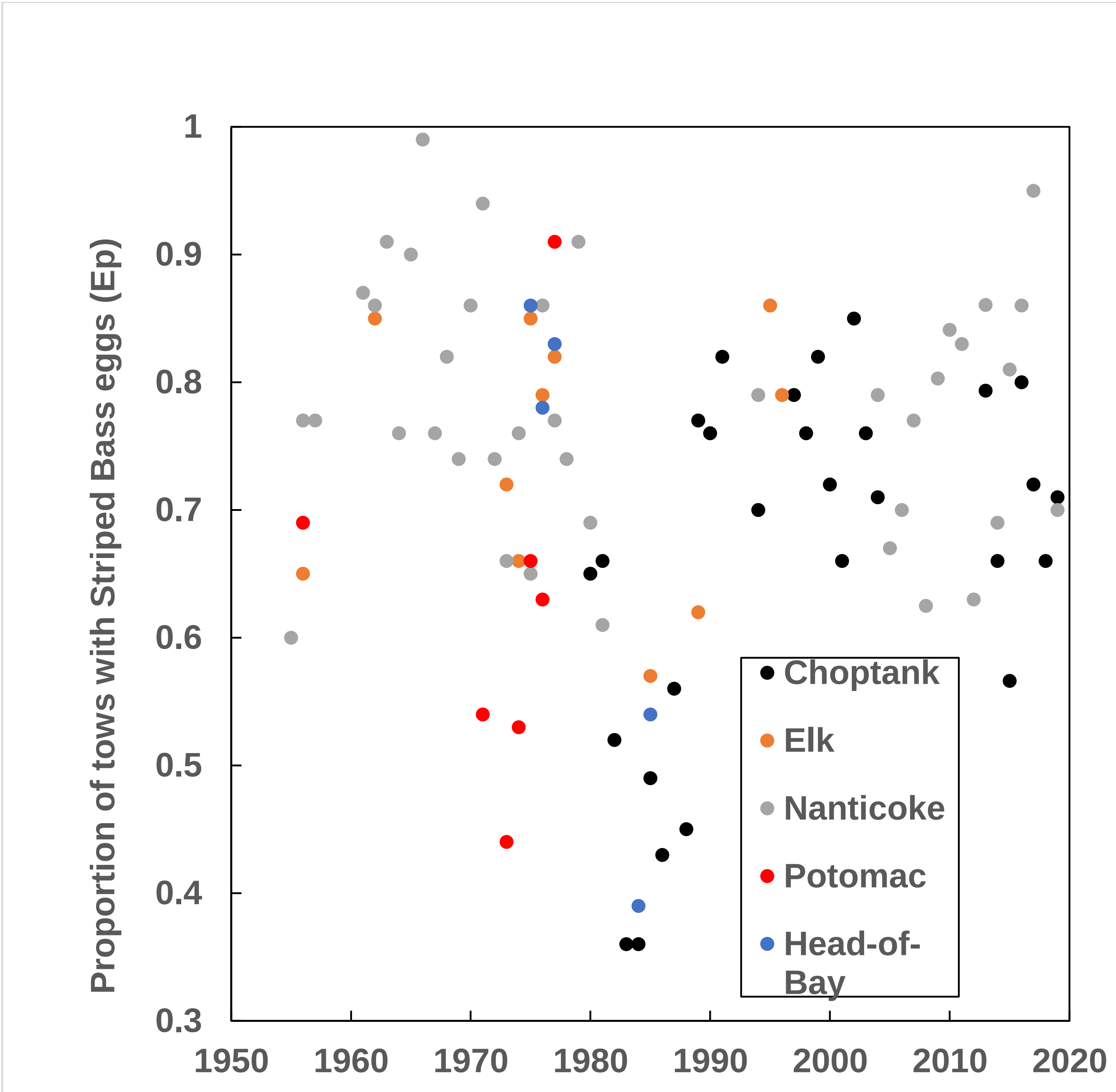


Figure 2.1.6. Baywide (Maryland's spawning areas) proportion of tows with Striped Bass eggs (*Ep;* diamond) and its 90% CI (line) estimated from surveys in juvenile index rivers conducted during 1955-2019. Baywide estimate pools available data from spawning surveys conducted in four areas surveyed for the juvenile index: Head-of-Bay, Potomac River, Nanticoke River, and Choptank River.

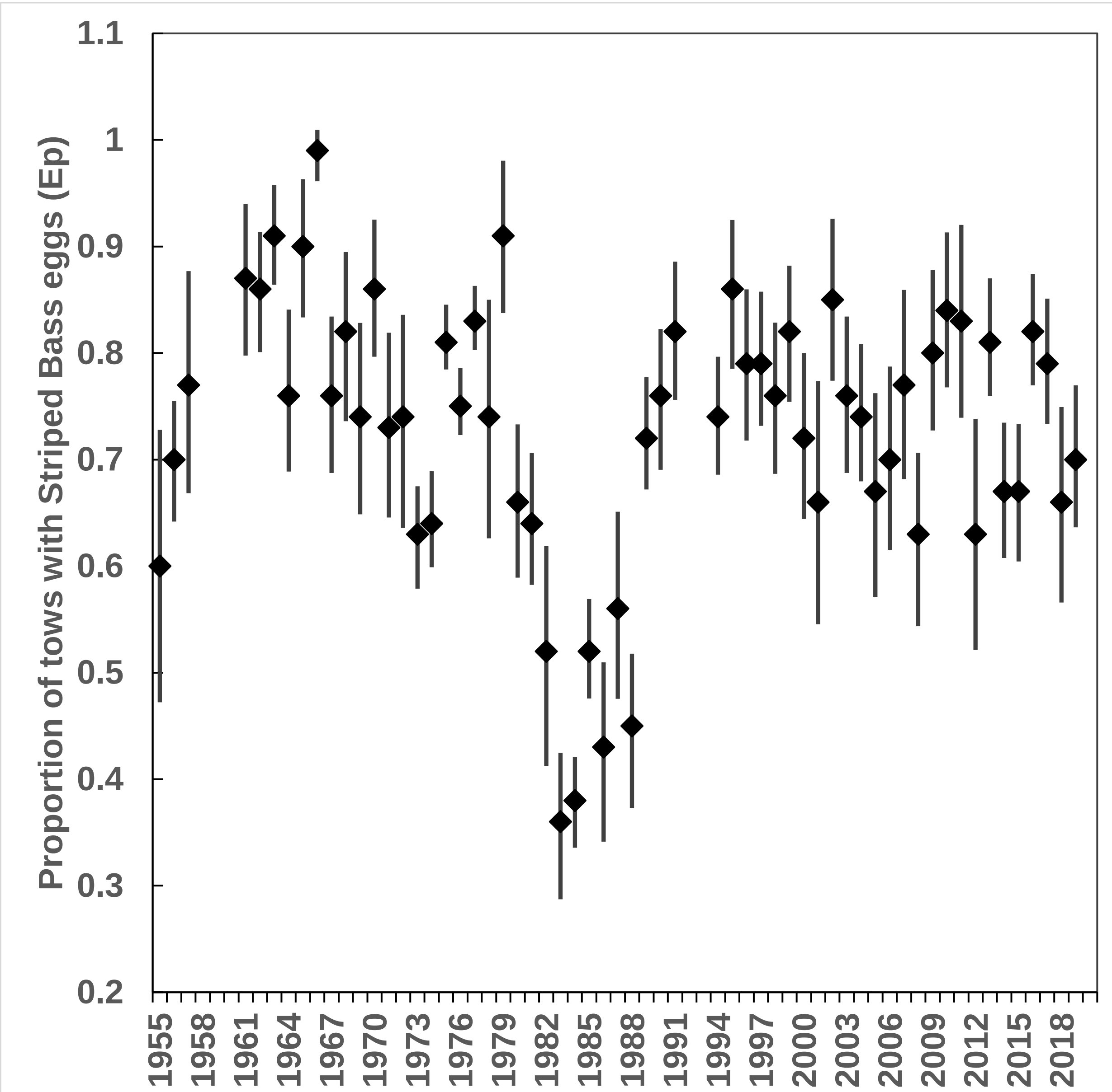


Figure 2.1.7. Baywide (Maryland spawning areas) proportion of tows with Striped Bass eggs (baywide *Ep*) and the baywide juvenile index (JI) time-series.

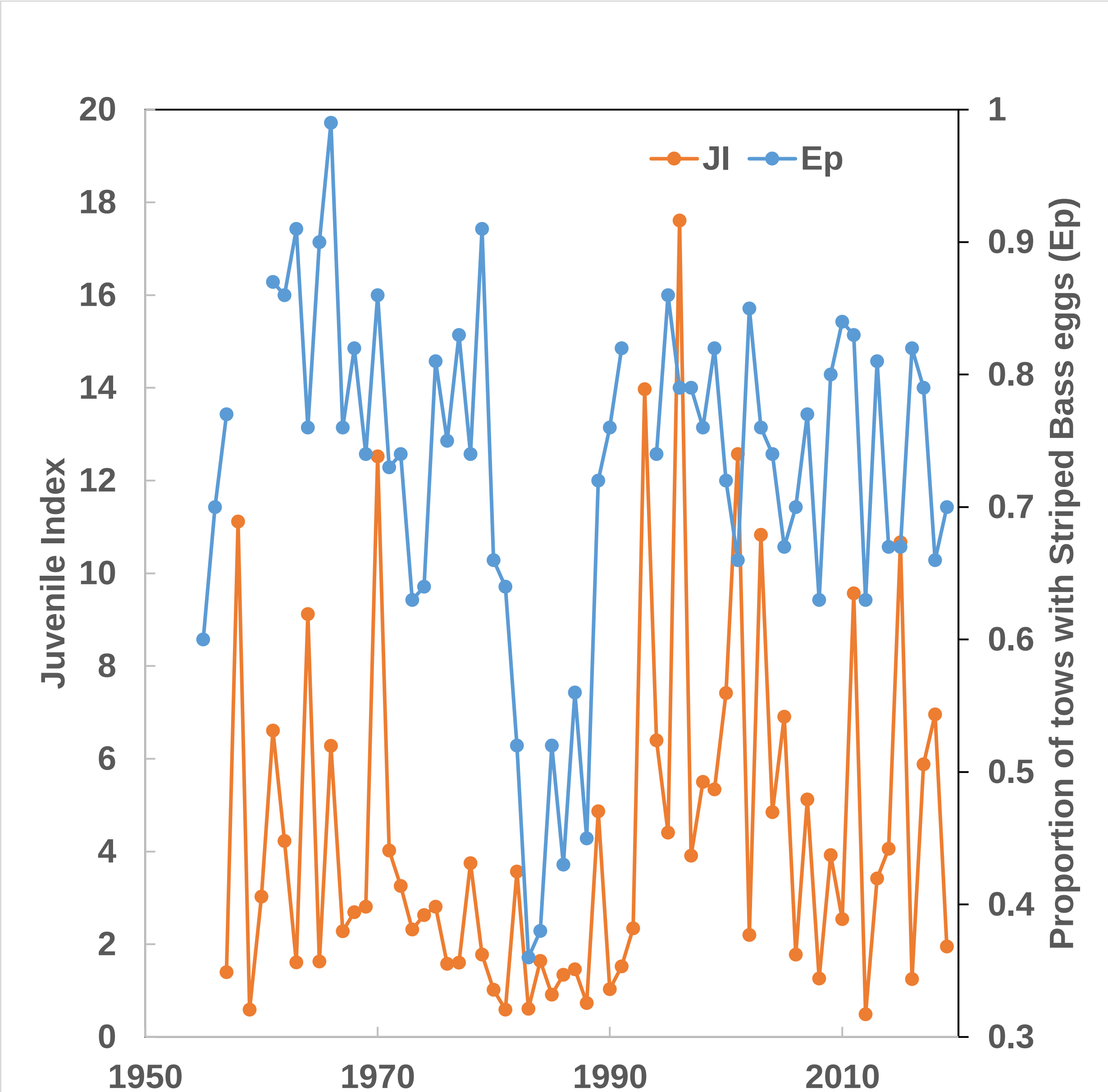


Figure 2.1.8. Trends in Maryland's baywide proportion of tows with Striped Bass eggs (baywide *Ep*) and spawning stock biomass (SSB) of Atlantic coast Striped Bass (NEFSC 2019) estimated from a statistical catch-at-age model. The time-series is restricted to years with SSB estimates available.

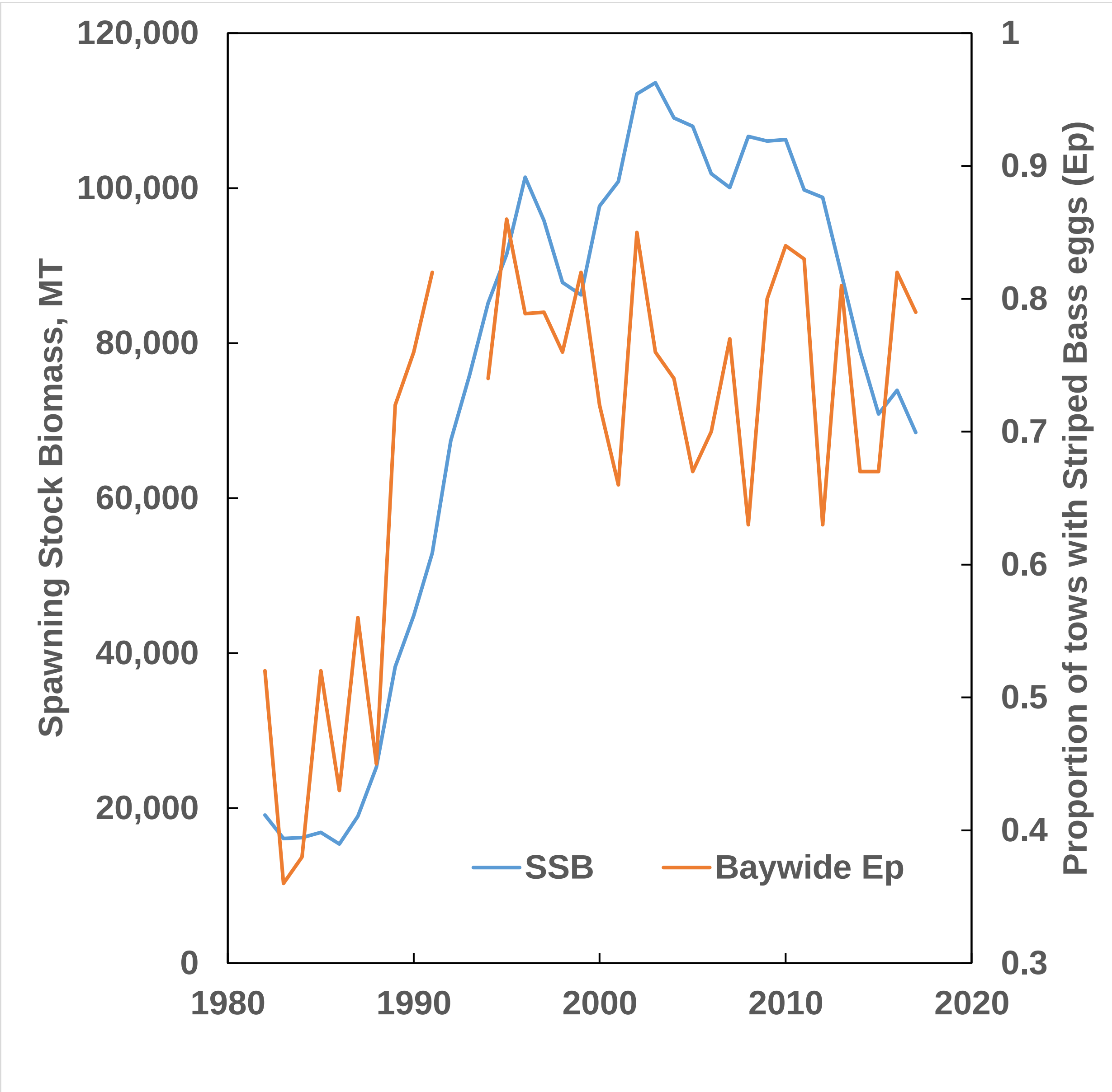


Figure 2.1.9. “Hook” pattern of residuals from the linear regression of $\log_e$-transformed Striped Bass spawning stock biomass (SSB; NEFSC 2019) and the baywide proportion of tows with Striped Bass eggs (baywide *Ep*).

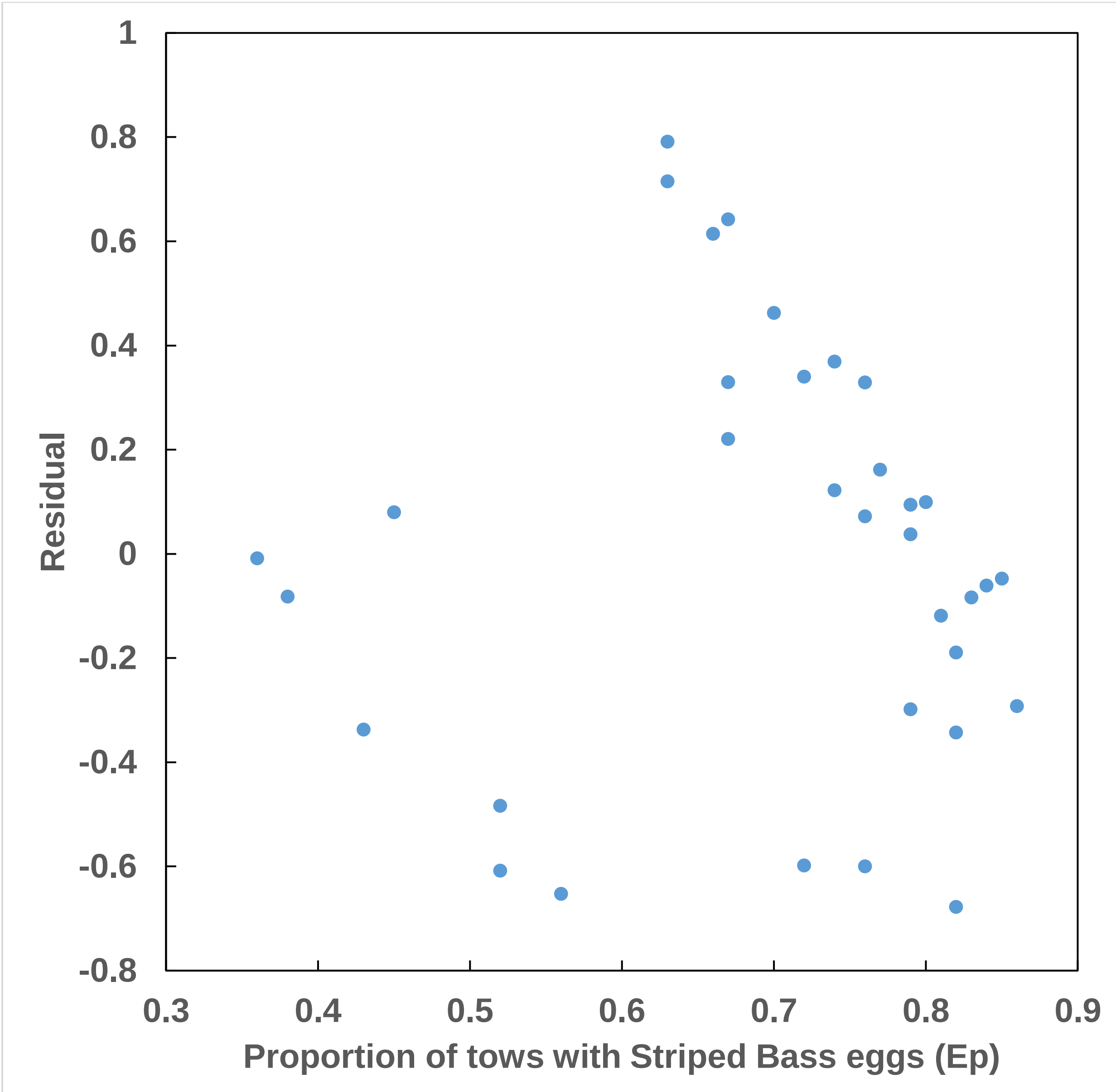


Figure 2.1. 10. Relative larval survival (baywide JI / baywide Ep) mean and 90% CIs, 1957-2019.

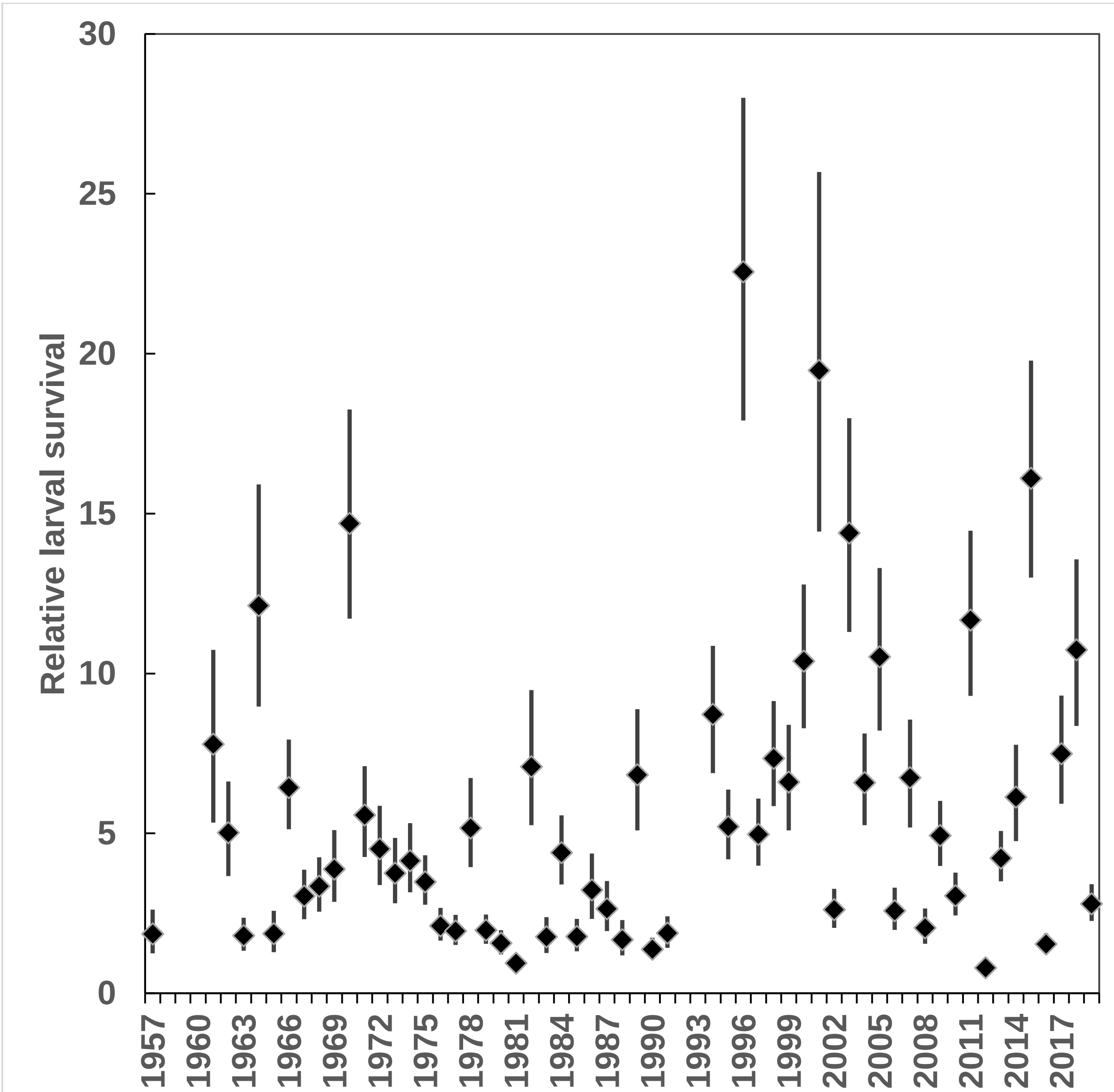


Figure 2.1.11. Relative survival during 1982-2017 based on Ep (RLS) and SSB (SSB RLS). Estimates are standardized to their means for years in common.

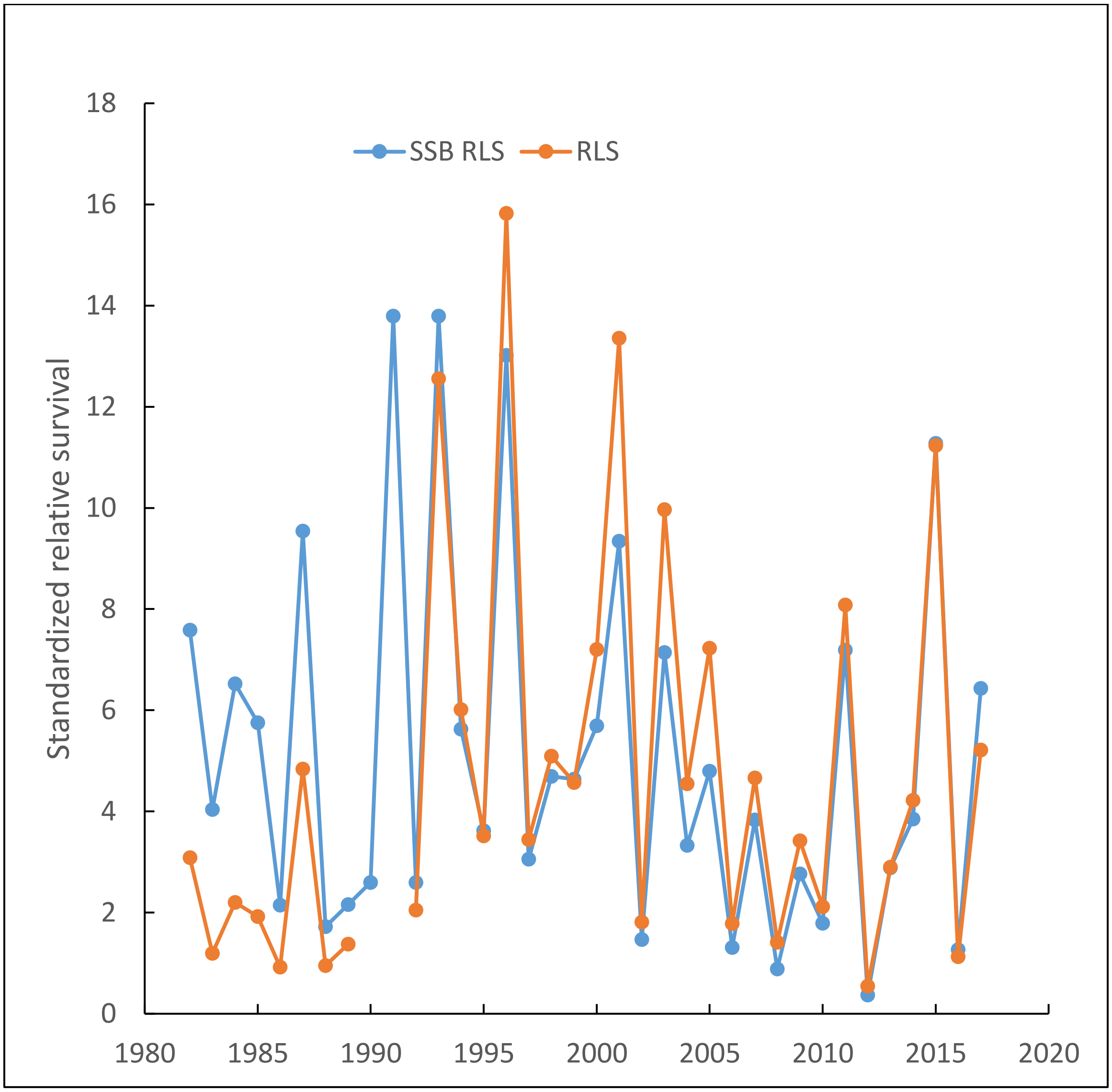


Figure 2.1.12. Spawning area specific Striped Bass juvenile indices. Note $\log_{10}$ scale on y-axis.

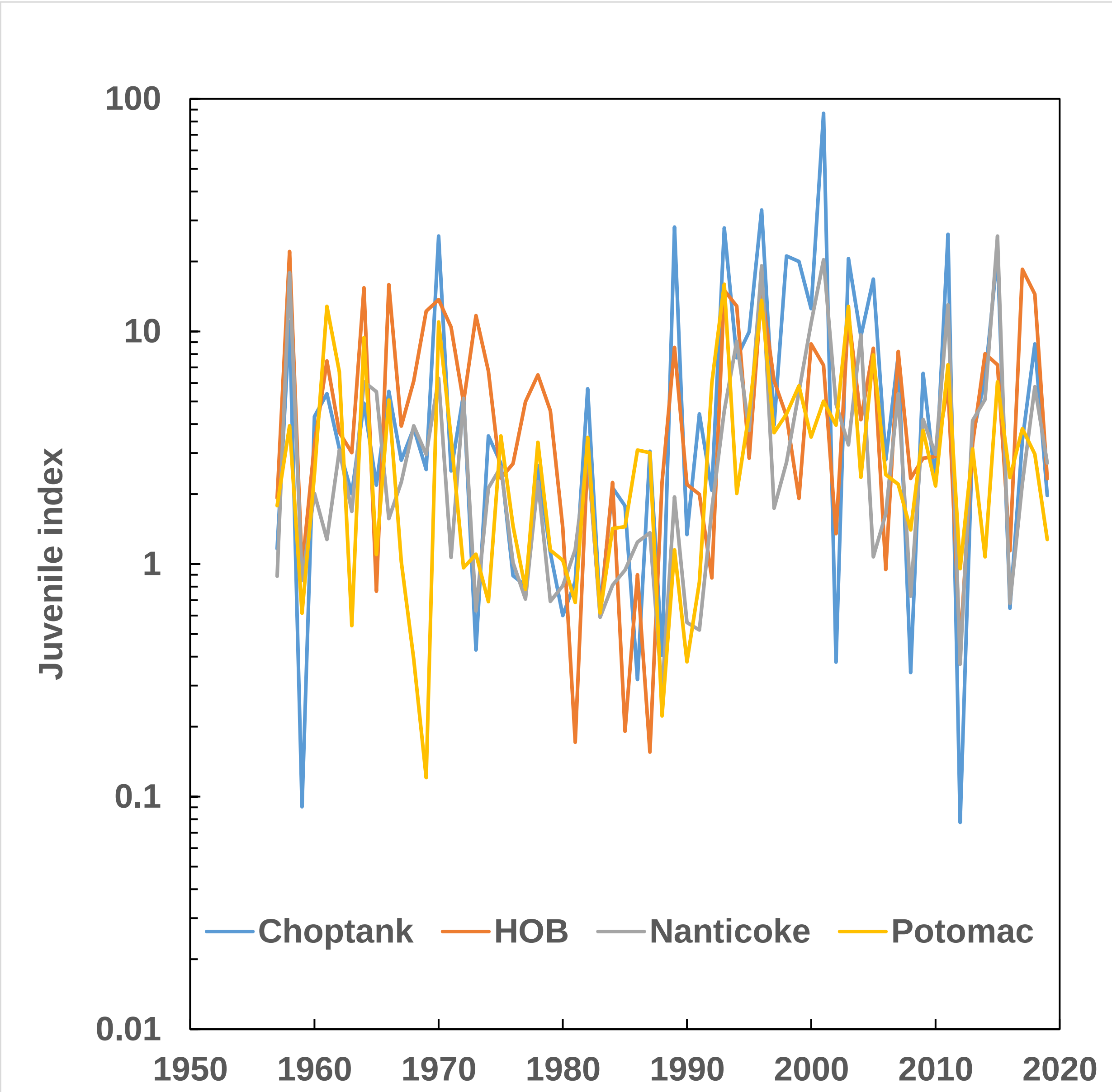


Figure 2.1.13. Striped Bass, White Perch, and Yellow Perch baywide juvenile indices standardized to their common time period means.

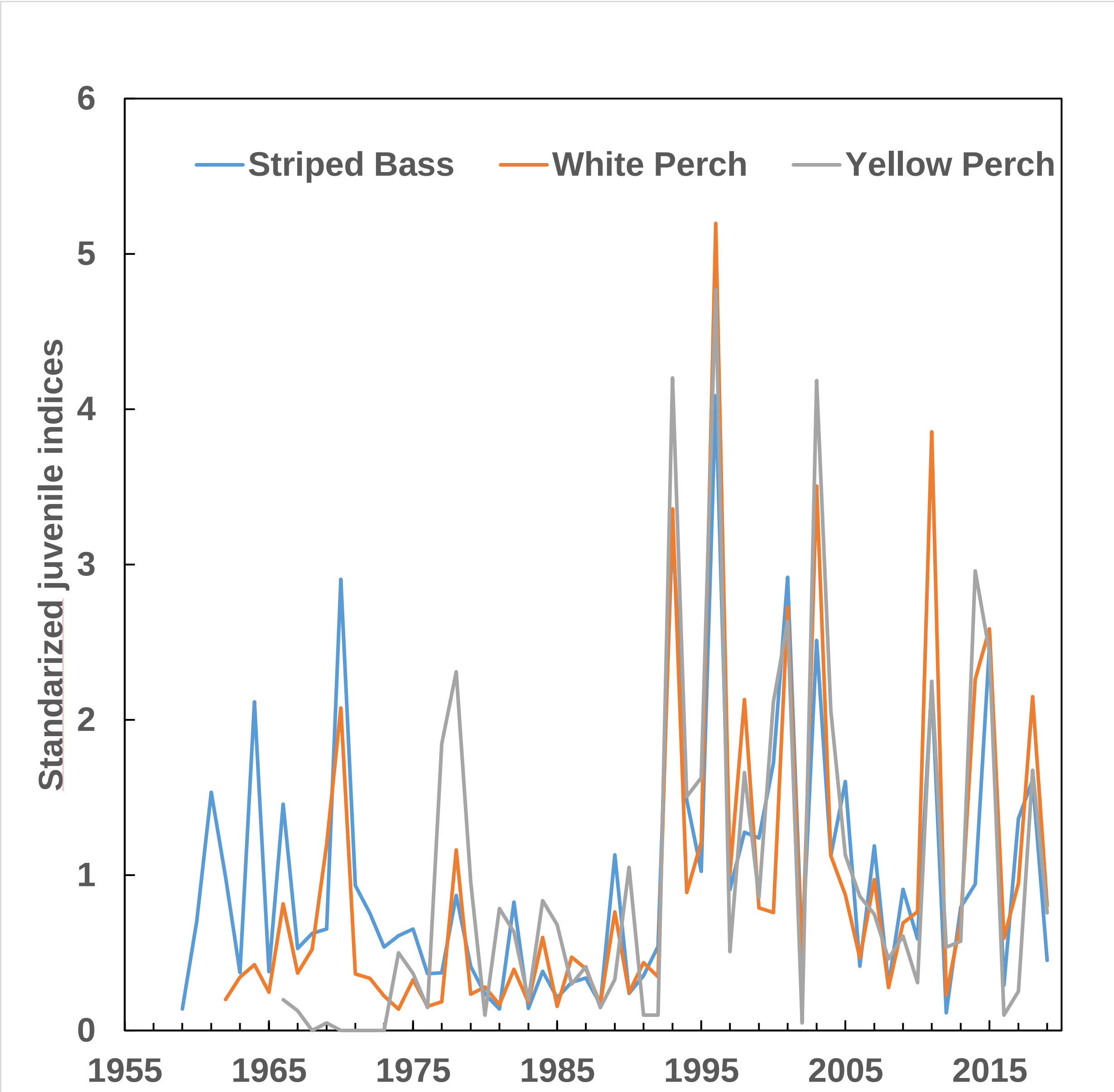


Figure 2.1.14. Trends in development (structures per hectare, C / ha) since 1950 in portions of watersheds draining into five Striped Bass spawning areas sampled during 2013-2019.

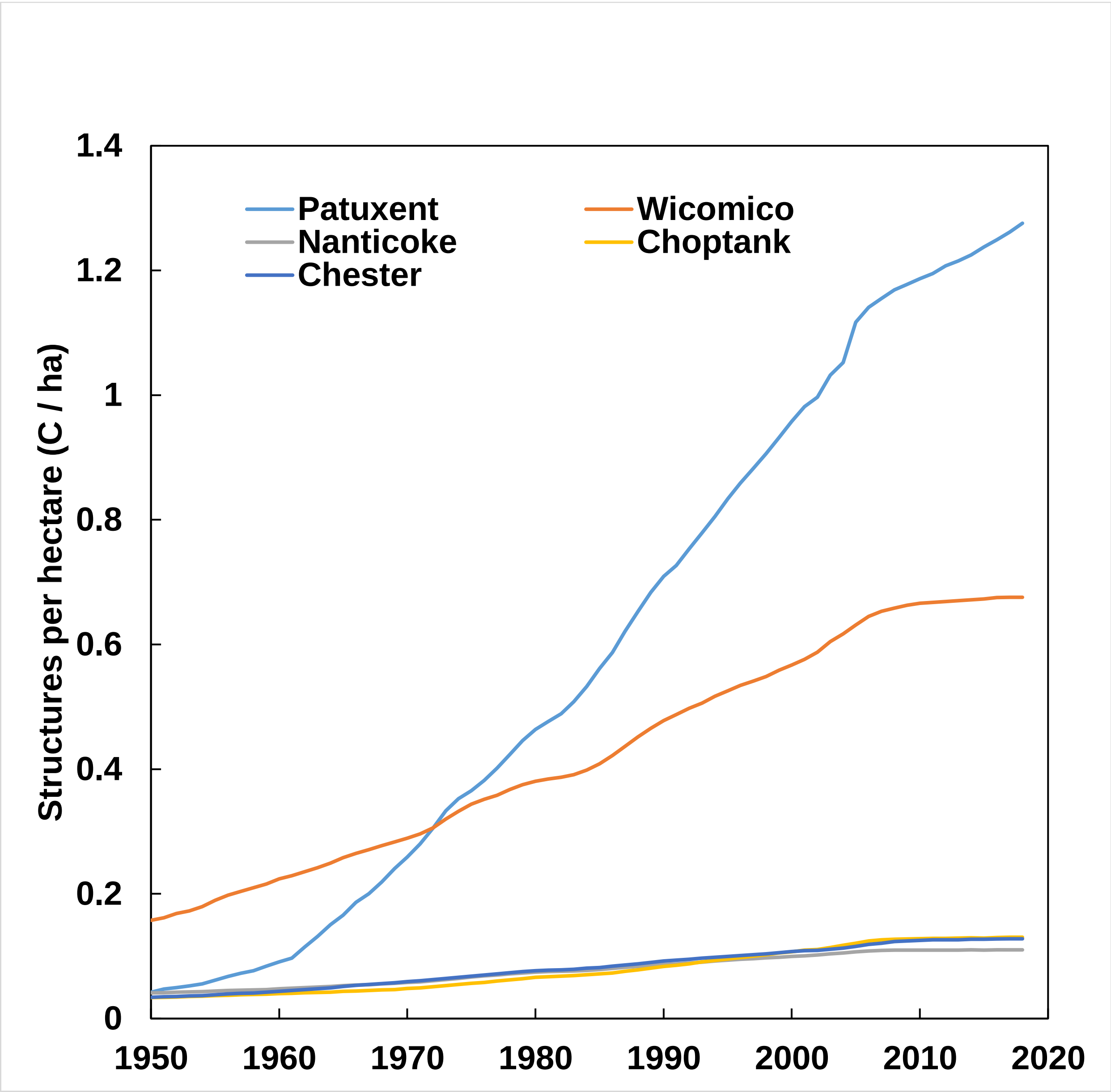


Figure 2.1.15. Choptank River pH mean (diamond), minimum, and maximum during a standard period (April 1 – May 8), 1986-1991 and 2014-2019.

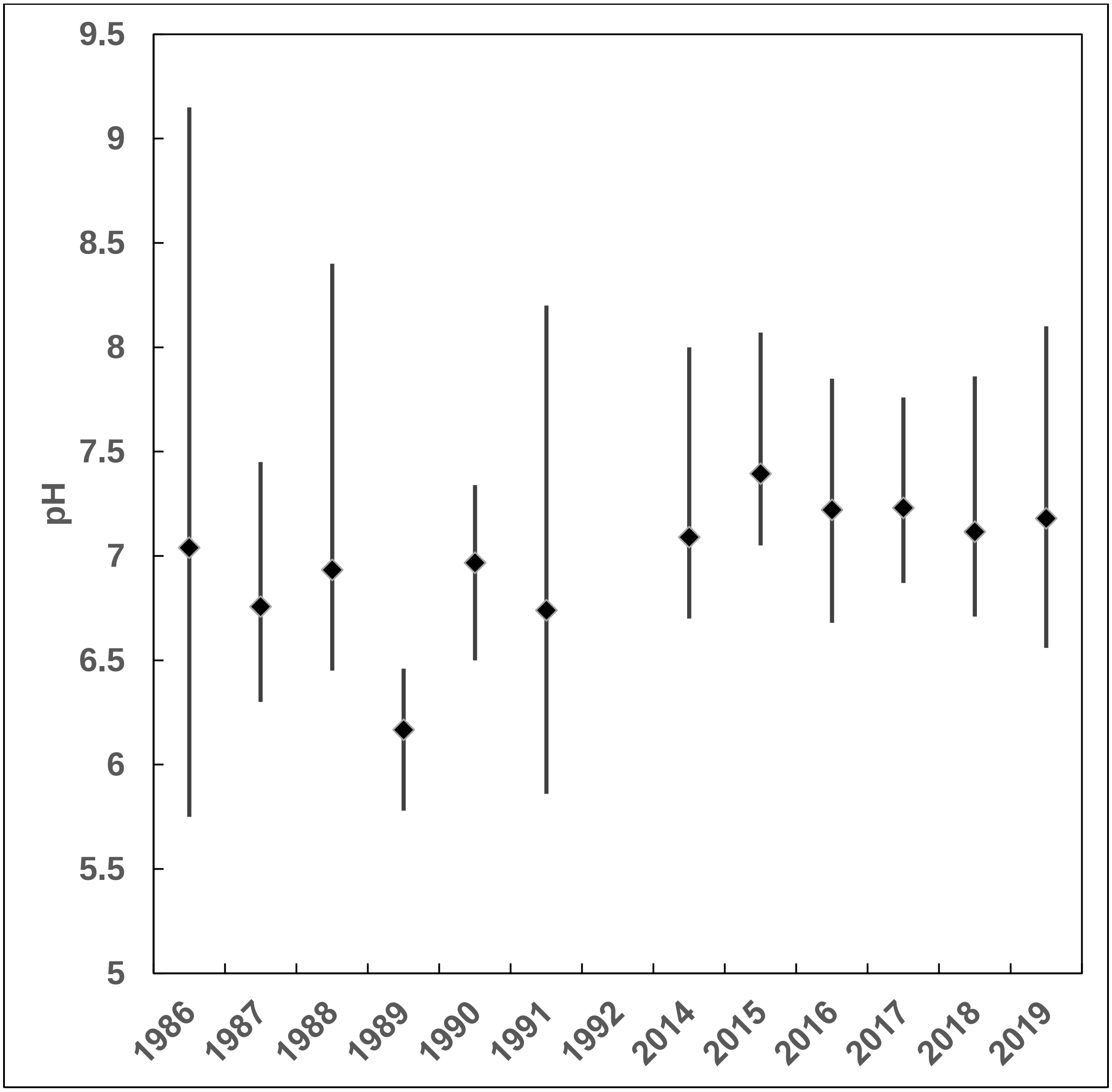


Figure 2.1.16. Choptank River maximum, mean (diamond), and minimum conductivity during a standard period (April 1 – May 8), 1986-1991 and 2014-2019.

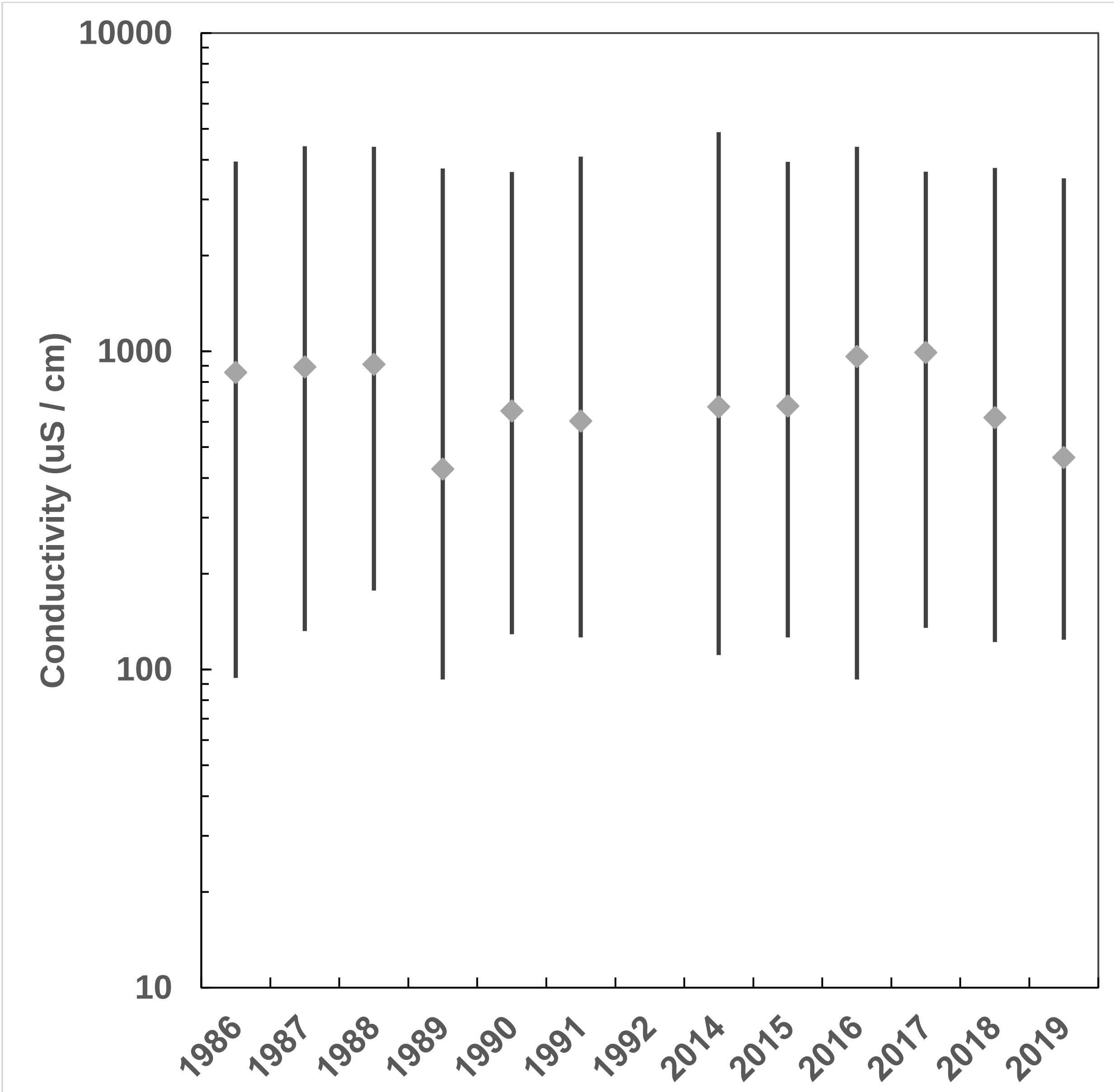


Figure 2.1.17. Choptank River water temperature mean (diamond), minimum, and maximum during a standard period (April 1 – May 8), 1986-1991 and 2014-2019.

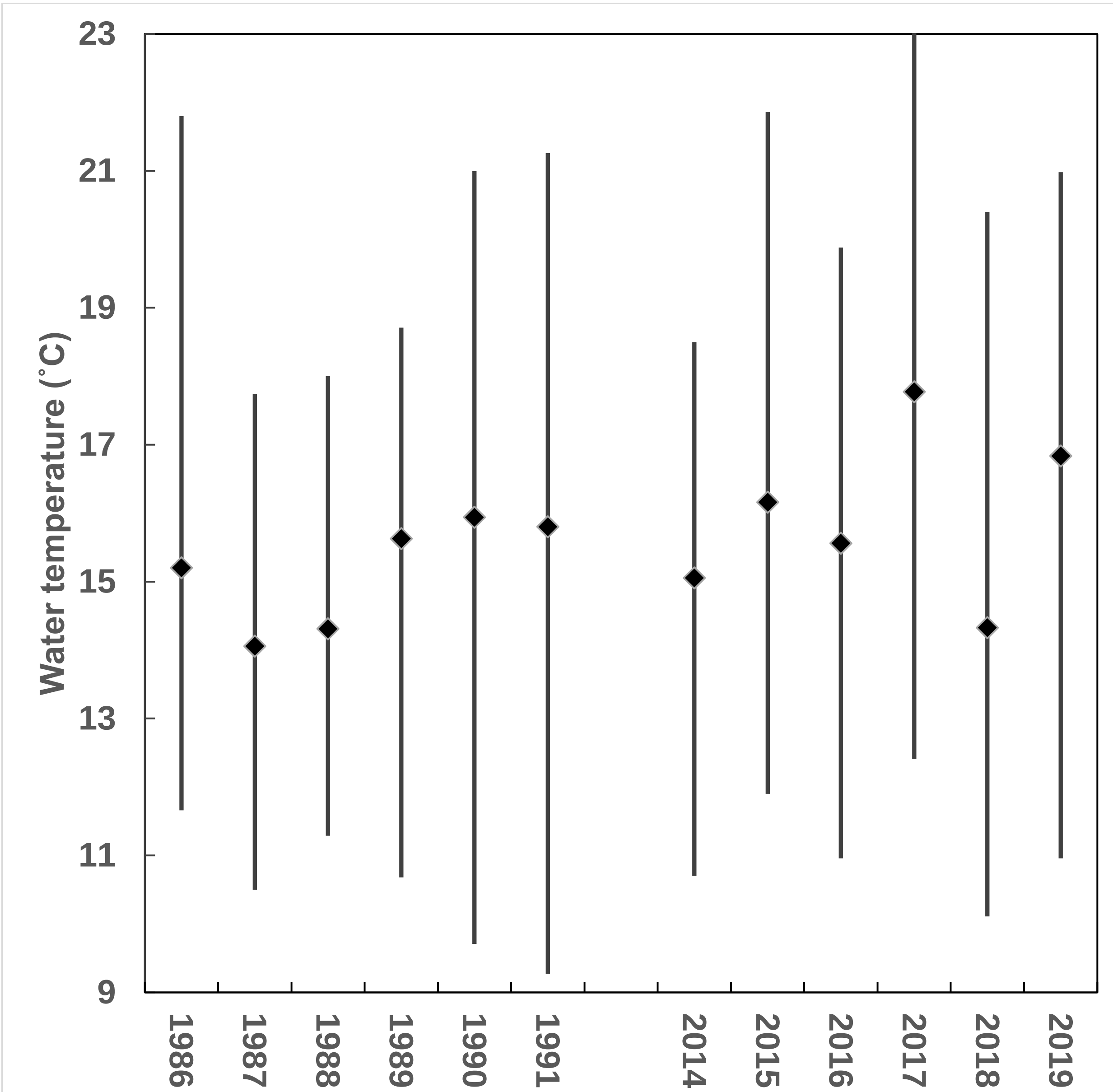


Figure 2.1.18. March and April Choptank River flow (cubic feet per second), 1957-2019.

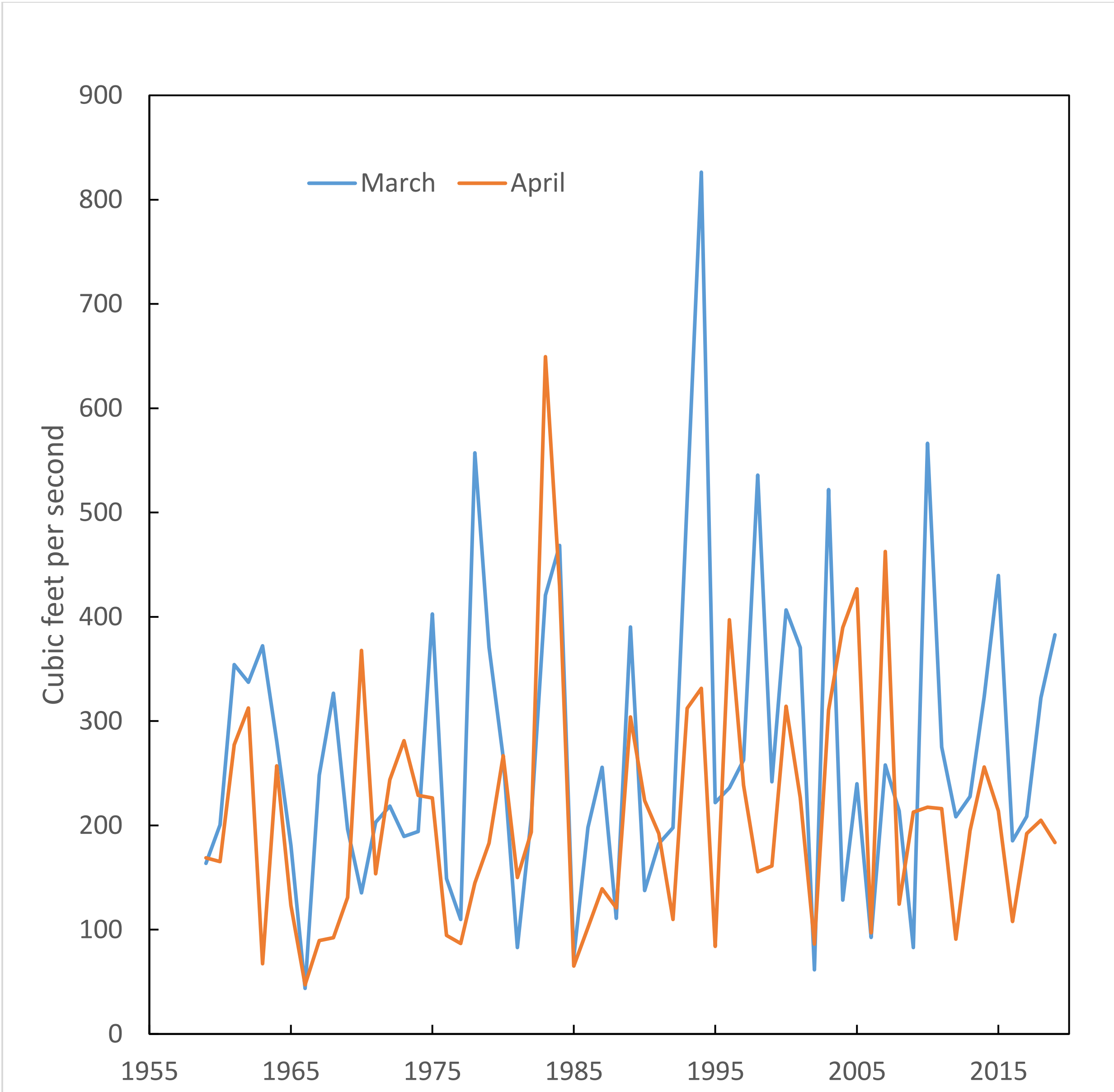


Figure 2.1.19. Relationship of the $\log_e$-transformed Choptank River Striped Bass juvenile index and average March-April flow (cubic feet per second) during 1957-2019.

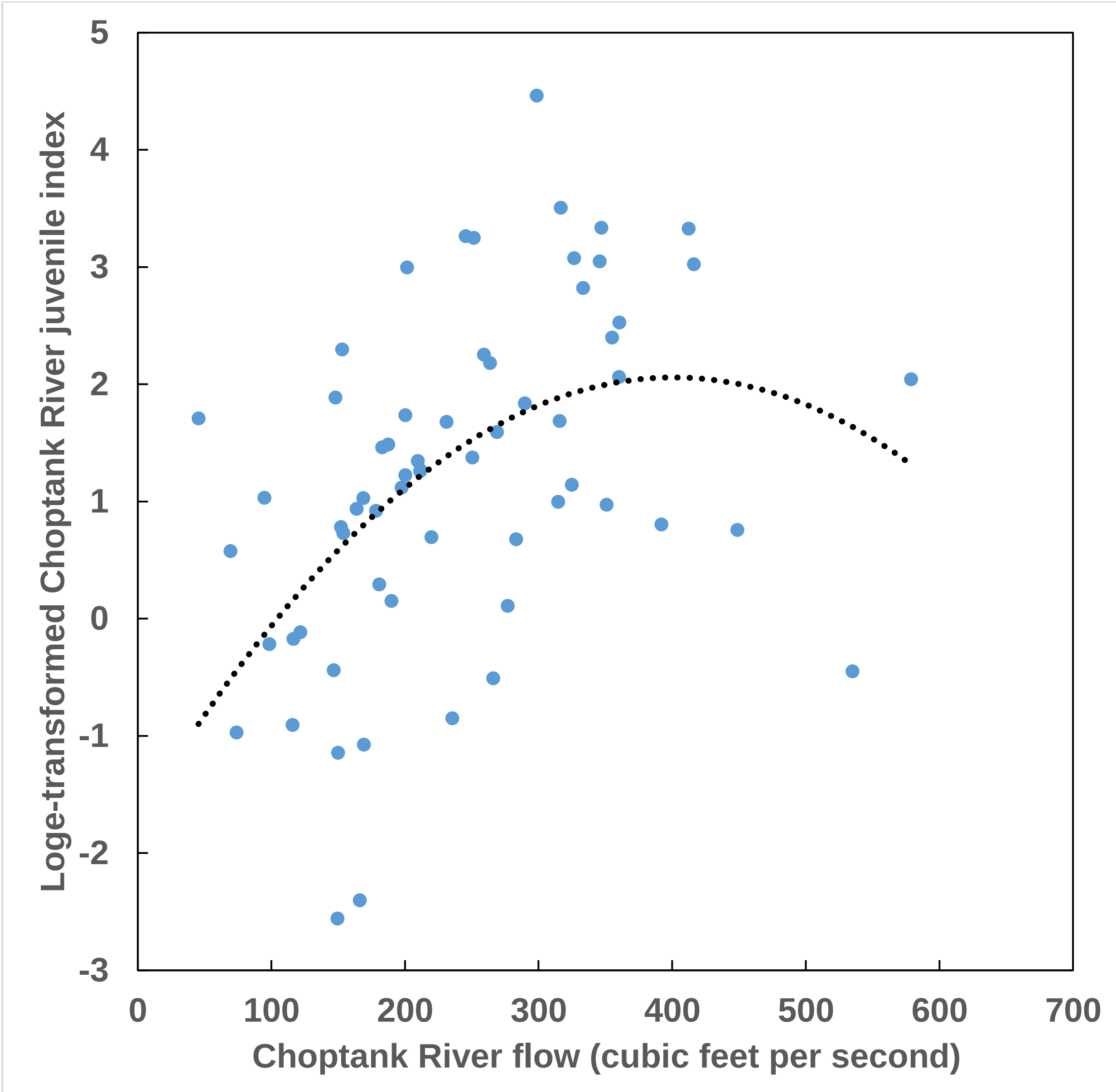
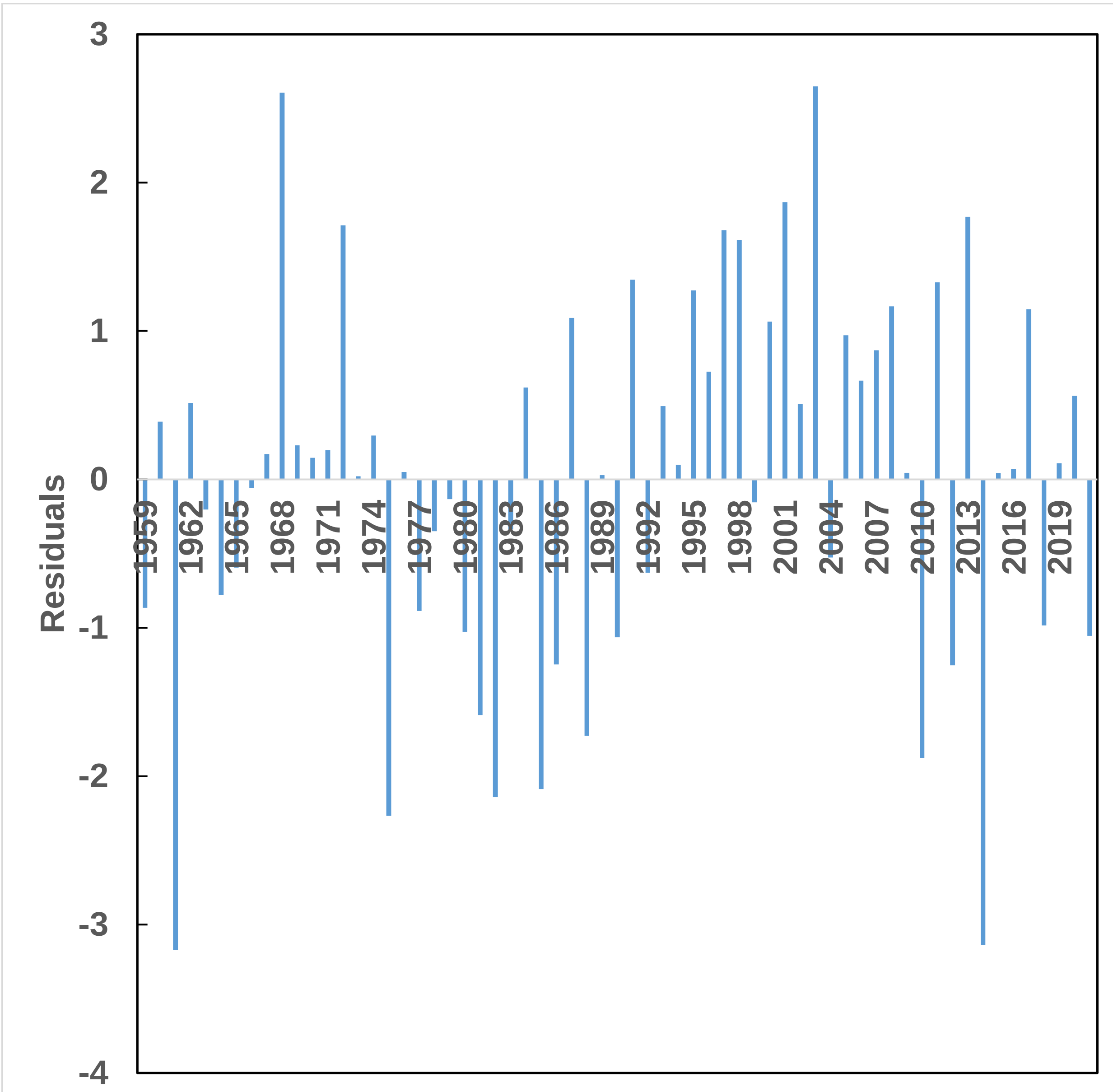


Figure 2.1.20. Residuals of the quadratic fit to $\log_e$-transformed Choptank River Striped Bass juvenile index and average March-April flow (cubic feet per second) during 1957-2019.



Section 3 - Estuarine Fish Community Sampling

Alexis Park, Carrie Hoover, Margaret McGinty, Jim Uphoff, Seth Dawson

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; MD DOP 2019) and they have ecological, economic, and societal consequences (Szaro et al. 1999).

Water quality and aquatic habitat are 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. Todd et al. (2019) reviewed impacts of three interacting drivers of marine urbanization (resource exploitation, pollution, and proliferation of manmade marine structures) and described negative impacts that were symptomatic of urban marine ecosystems.

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; 2019) have found adequate concentrations of DO in bottom channel habitat of tidal-fresh (0-0.5 ‰), oligohaline (0.5-5.0 ‰) and mesohaline (5.0-18.0 ‰; Oertli, 1964) 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 2019, we continued to evaluate summer nursery and adult habitat for recreationally important finfish in oligohaline and mesohaline 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 2003-2019 using correlation analysis. We continue to examine and Tred Avon River, a tributary of Choptank River located in Talbot County (Table 3-1; Figure 3-1). In 2019, we returned for a second year 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. We added a more detailed evaluation of species composition and richness to our analysis in order to better understand the possible changes occurring within the Chester River.

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). We calculated an arithmetic mean of all bottom salinity and measurements for all years available through 2019 to determine salinity class of each subestuary, grouping data by the mean into the 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 2019 to support their comprehensive growth plan: Corsica River and Langford Creek (mesohaline tributaries of the Chester River), Chester River, and Wye River. We previously sampled Corsica River, from 2003 to 2012 and in 2018; Langford Creek was previously sampled from 2006 to 2008 and in 2018; and Wye River was previously sampled from 2007 to 2008 and in 2018. 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. We, FHEP, returned to the Chester River in 2018 and 2019.

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 3-1): 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 2003-2019 (Table 3-2). Maryland DOP only has structure estimates available through 2017; 2017 estimates were used to represent 2017-2019 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, 2019, 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). Impervious surface estimates were made by Towson University from Landsat, 30-meter pixel resolution satellite imagery (eastern shore of Chesapeake Bay in 1999 and western shore in 2001; Barnes et al. 2002). Development targets and limits, and general statistical methods (analytical strategy and equations) are described in **General Spatial and Analytical Methods used in Job 1, Sections 1-3** as well. Specific spatial and analytical methods for this section of the report are described below.

Surveys focused on twelve 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 estuarine 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 Services' surveys (Carmichael et al. 1992; Bonzek et al. 2007; Durell and Weedon 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 2019 unless they were no longer accessible. The 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 Chesapeake Bay (in the case of Chester River) or Chester River waters on subestuary fish habitat (in the cases of Corsica River and Langford Creek). 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. The Chester River had six seine and trawl sites in 2019 throughout the river based on previous sites sampled in 2012. In 2018, we did not sample with bottom trawls in the Chester River due to limited staff. We did not seine in the Corsica River in 2019 and only trawled instead. Four sites (based on previous locations) were sampled by trawling in the Corsica River during 2019; beach seines were not used in 2019 due to lack of suitable shoreline access.

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 and was based on the 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 sites due to its larger size. 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 dissolved oxygen, 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 (μS / 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. Previously in 2018, Chester River bottom water quality parameters were recorded in the channel at three locations (upper, middle, and lower seine sites). Water quality data for the Chester River from 1995 to 1998 were recovered and added to the 2019 analyses.

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 then 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 measured and the measurements were recorded for a modified proportional stock density analysis (Willis et al. 1993).

Seining in Langford Creek and Wye River was very restricted because of high tides that limited beach availability during 2019; only 3 of the 4 seine sites could be sampled in each of these subestuaries. Seining was not conducted in the Corsica River in 2019 due to limited beach availability and increased amounts of large woody debris blocking shorelines. Eliminating seines sites from the Corsica River allowed time for trawl sampling in the Chester River. 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 sampled during 2019. Unlike seining sites, all trawl sites could be sampled during 2019.

2019 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 (1989-2019) was available for beach seine samples than the 4.9 m bottom trawl (2003-2019) 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 1989-2019 by salinity class, then ranked their bottom trawl GMs by year for all species combined to find where the 2019 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 correlation analyses by salinity classification provided a means of isolating the increasing influence of salinity on stratification from the influence of 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 2019). 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 2010-2019 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. Newer land use estimates have not been released by MD DOP at this time.

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 2003-2019 using correlation analysis. We further examined the influence of percent of land in agriculture on median bottom DO using linear, multiple linear, and quadratic regression models.

Tred Avon River - In 2019, we sampled four stations in Tred Avon River (Figure 3-2). We contrasted Tred Avon River to Broad Creek (sampled during 2012-2017 and Harris Creek (2012-2016). Trajectories of C/ha since 1950 were plotted for the three Choptank River subestuaries (Figure 3-3). Bottom DO measurements during 2006-2019 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 2006-2019 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 within the three subestuaries. We estimated GMs of trawl and seine catches, modified PSD of White Perch, and species composition.

We used a percent similarity index to evaluate variation in finfish species composition among Tred Avon River trawl stations 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 percent similarity index varies from 0% (no species in common) to 100% (all species in common) and is considered a robust measure (Kwak and Peterson 2007).

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 stations within each subestuary were significantly different from one another. An overall median DO was calculated for all time-series data available for each system and used to detect how annual station DO compared with the time-series median. Correlation analysis of annual median DO measurements was used among the three systems.

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

Queen Anne's County Subestuaries - In 2019, 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.

We assembled time-series of Secchi depth, SAV area, bottom DO, pH, and salinity. 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 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 compared with annual mean station DO. Correlation analysis was used to examine associations of annual median DO among the four systems. We also used correlation analysis to examine associations among subestuaries of each water quality variable: Secchi depth, GM catches, DO, pH, or salinity.

White Perch Modified Proportional Stock Density - A modified Proportional Stock Density (PSD; Anderson 1980; Anderson and Neumann 1996; Neumann and Allen 2007) was calculated for White Perch in each Choptank River and Queen Anne's County subestuary for all years available to compare relative proportions of the adult population that would be of interest to anglers. Low PSD percentages indicate highly variable densities of year-classes where higher densities of stock length fish exist due to possibly overharvest or habitat issues (Anderson 1980; Neumann and Allen 2007). Proportional stock density is calculated using length-frequency data and provides population dynamics information (Anderson and Neumann 1996; Neumann and Allen 2007). Normally, a PSD is calculated as:

$$PSD = ((N \geq L_{\text{Quality}}) / (N \geq L_{\text{Stock}})) \times 100;$$

where N is the number of White Perch caught in each subestuary that were stock length or greater. Stock length (L_{Stock}) refers to the number of White Perch at the minimum length of fish that provides a recreational value (≥ 125 mm TL; Piavis and Webb 2019); however, we substituted stock length with the total number of small adults plus harvestable length White Perch for stock length to estimate a modified PSD since we did not measure small adults. Quality length (L_{Quality}) refers to the number of fish at the minimum length most anglers like to catch (≥ 200 mm TL; Piavis and Webb 2019).

White Perch greater than or equal to 130 mm TL is 20 - 26% of the world record length TL (Gablehouse et al 1984) is considered stock length category minimum; 125 mm TL is used as the length cut-off for White Perch in Chesapeake Bay recruitment and length-frequency assessments (Piavis and Webb 2019). Modified stock length category included small adults under 200 mm TL and could have fish as small as 90 mm TL.

White Perch greater than or equal to 200 mm TL were considered to be of harvestable size and all captured were measured to the nearest millimeter. White Perch of this size or larger corresponded to the quality length category minimum (36-41% of the world record TL) proposed by Gablehouse et al (1984); 200 mm TL is used as the quality length category minimum length cut-off for White Perch in Chesapeake Bay (Piavis and Webb 2019). These data provided an opportunity to evaluate the influence of development on the availability of fish for anglers to harvest.

Modified PSDs were calculated for the Choptank River subestuaries and Queen Anne's County subestuaries by sampling year for 4.9m trawl samples and for beach seine samples. Beach seine sampling was variable due to tides, SAV, beach obstructions, and people; therefore, the preferred PSDs for White Perch was based on trawl samples. White Perch were collected from the Chester River from 1994 to 2000, but no indication of age or length were recorded; therefore, these earlier years of sampling could not be included in the modified PSD analysis. In 2018, trawl sampling was not conducted in Chester River, no bottom water White Perch community composition was included in the PSD. In 2019, no seine sampling was conducted in Corsica River, no shallow water White Perch community composition was included in the PSD.

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 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 seine data 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.

In the summer of 2019, we noticed that pH measurements were off due to a faulty pH probe. The probe was replaced, but pH readings for two weeks from 8/5/2019 to 8/15/2019 were higher than other recorded readings in 2019. Those two weeks of pH readings (one sampling round) were removed from analysis.

During restructuring of summer estuarine fish and water quality data in 2017-2019, older data (before 2006) were found to be 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

2019 Sampling Summary – Table 3-3 provides summary statistics for surface and bottom water quality for each subestuary sampled in 2019. All tributaries and subestuaries sampled had DO readings less than the target level (5.0 mg / L) during 2019 (Table 3-4). Eleven percent of all DO measurements (surface and bottom) from Chester River were below the target; Langford Creek had 17%; Tred Avon River, 30%; Corsica River, 34%; and Wye River, 33%. In 2018 and 2019, only two subestuaries did not have any bottom DO estimates below the 3 mg / L threshold; Chester River and Langford Creek (Table 3-4). The remaining subestuaries had threshold bottom DO violations: Corsica River, 5%; Tred Avon River, 17%; and Wye River, 13%.

Salinities were lower than their long-term averages for all subestuaries sampled during summer 2018-2019 due to large amounts of spring and summer precipitation; Chester River salinities were less than 5.0‰, shifting the normally mesohaline subestuary to oligohaline during 2018 and 2019. Salinities in remaining subestuaries were close to or within mesohaline bounds (Table 3-1).

Geometric mean catch per seine haul ranged from 98 to 139 among the four subestuaries sampled during 2019 (Table 3-5). Geometric mean seine catches in 2019 ranked Tred Avon River, 1st; Chester River, 2nd; Langford Creek, 3rd; and Wye River ranked 4th. Between 20 and 25 species were encountered in mesohaline tributary seine samples (Table 3-5).

A plot of species richness in seine samples and C/ha during 1989-2019 did not suggest a strong relationship in tidal-fresh, oligohaline, or mesohaline subestuaries (Figure 3-4). Tidal-fresh subestuary watersheds were represented by a limited range of C/ha (0.43 – 0.67). Oligohaline subestuary watersheds were represented by the widest range of C/ha (0.08 – 3.33) of the three salinity classes. Mesohaline subestuary watersheds were represented by a larger number of samples (N = 71; C/ha range = 0.07 – 2.68) than tidal-fresh and oligohaline subestuaries (N = 22 and 35, respectively; Figure 3-4).

A total of 11,683 fish representing 35 species were captured by beach seining in 2019 (Table 3-5). Eight species comprised 90% of the total fish caught in 2019, including (from greatest to least) Atlantic Silverside, White Perch (adults), Atlantic Menhaden, Mummichog, Striped Killifish, Spottail Shiner, Banded Killifish, Bay Anchovy, and White Perch (juveniles). White Perch (juveniles and adults), Spottail Shiner, Bay Anchovy, and Atlantic Menhaden represented target species among the species comprising 90% of the total catch. Five target species were present among species comprising 90% of the seine catch throughout all subestuaries: White Perch (juveniles and/or adults) were present in this category in all four subestuaries; Atlantic Menhaden and Bay Anchovy in three; Spottail Shiner and Striped Bass (juveniles) in two.

Geometric mean catches per trawl were between 19 and 77 during 2019 (Table 3-6). All subestuaries had 24 samples in 2019, except for the Chester River which had 34 samples (6 stations). Chester River had the greatest GM (77) and Wye River had the lowest (19) for the second year in a row; Langford Creek ranked 2nd; Corsica River ranked, 3rd; and Tred Avon River ranked, 4th.

Number of species captured by trawl in subestuaries sampled during 2019 ranged from 8 to 16 (Table 3-6). A plot of species richness in trawl samples against C/ha (all subestuaries during 2003-2019) did not indicate a relationship of development and number of species for tidal-fresh (species richness ranging from 14 to 25) or oligohaline subestuaries (species richness ranging from 12 to 26; Figure 3-5). 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 8,059 fish and 25 fish species were captured by trawling during 2019. Five species comprised 90% of the total catch for 2019 (from most to least): White Perch (adult), Spot, White Perch (juvenile), Bay Anchovy, Channel Catfish, and White Catfish; Bay Anchovy, Spot, and White Perch were the only target species. Target species comprising 90% of the catch in each of the five subestuaries sampled during 2019 were White Perch (adult) in five subestuaries; Spot in four; White Perch (juveniles) and Bay Anchovy each in two subestuaries (Table 3-6).

Dissolved Oxygen Dynamics – Correlation analyses of DO with temperature and C/ha in subestuaries sampled since 2003 (Table 3-7) indicated that DO responded to temperature and C/ha differently depending on salinity classification (Table 3-8). Mean bottom DO in summer surveys declined below the threshold level in mesohaline tributaries, but did not in oligohaline or tidal-fresh (Figure 3-6). 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 occupation of this habitat (Uphoff et al. 2011a). Mean surface DO in summer surveys did not fall below the threshold, but two mesohaline subestuaries (Chester River, 2011-2012; Corsica River, 2012; Table 3-7) fell below the target conditions (Figure 3-7).

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 3-8), 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 and 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. 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). 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). Sample sizes of mesohaline subestuaries (N = 88) were over twice as high as oligohaline (N = 33) or tidal-fresh subestuaries (N = 48), so ability to detect significant associations in mesohaline subestuaries was greater (Table 3-8).

Depletion of bottom DO in mesohaline subestuaries with suburban-urban watersheds to below target levels resulted in lost habitat. 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 development through its negative influence on DO.

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 land cover were negative and moderate to strong (r = -0.759; P <0.0001 and r = -0.812; P <0.0001, respectively); the correlation of urban land cover with C/ha was positive and strong (r = 0.898; P < 0.0001; Table 3-9). Correlation between forest cover and agriculture cover was negative and moderate (r = -0.578; P < 0.0001). Wetland cover and C/ha were negatively and weakly correlated (r = -0.263; P = 0.02). Remaining pairings of categories were not well correlated (Table 3-9).

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 3-8). 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 3-8). 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 bottom DO data into western and eastern regions and used a linear regression for each region to describe regional changes in annual median subestuary bottom DO with percent agriculture. The relationship was strongly positive for the western shore (slope = 0.13; SE = 0.02; $r^2 = 0.73$; $P < 0.0001$; N = 21; Table 3-10) and weakly negative for the eastern shore (slope = -0.03; SE = 0.01; $r^2 = 0.18$; $P = 0.0011$; N = 55; Table 3-10). 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 %. 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 well ($R^2 = 0.61$, $P < 0.001$; Table 3-11; Figure 3-8).

Mesohaline subestuaries sampled with bottom trawl in 2019 ranked relatively low compared to earlier years. The 2019 Chester River GM ranked the highest out of the other 2019 GMs, 57th out of 83; Langford Creek, 62nd; Corsica River, 69th; Tred Avon River, 71st; and Wye River, 74th (Table 3-12). Correlation between mesohaline subestuary GMs and C/ha was weak and negative ($r = -0.21$; $P = 0.05$; Table 3-13; Figure 3-9). Remaining pairings of categories were negative and not well correlated. Tidal-fresh and oligohaline subestuaries had limited samples, so ability to detect significant associations in mesohaline subestuaries was greater.

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 2019; Table 3-14; Figure 3-1); 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 3-14). 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 2019).

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 3-3) and more than indicated by the Maryland Department of Planning urban category (Table 3-14; Figure 3-1). 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 = 0.38 in 2016), while Broad Creek is still under the rural development target (C/ha was 0.29 in 2016). More growth occurred in Tred Avon River's watershed (C/ha = 0.76 in 2016; Figure 3-3). 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 2019, 71 % of bottom DO samples were below the target and 17% were below the threshold in Tred Avon River (Table 3-15). During 2006-2019, 9 % of bottom DO measurements from Tred Avon River were below the DO threshold and 37 % were below the DO target (Figure 3-10). Less than 1% of Broad Creek bottom DO measurements during 2012- 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 (Figure 3-10).

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 4.5 mg / L (2019) to 6.3 mg / L (2009; Figure 3-11). Median bottom DO in Broad Creek ranged from 5.6 mg / L (2012) to 6.6 mg / L (2015) and in Harris Creek it ranged from 5.7 mg / L (2013) to 6.3 mg / L (2015; Figure 3-11). Correlations of median bottom DO among Choptank subestuaries were modest to low and trends were not considered meaningful (Table 3-16).

An ANOVA of Tred Avon River stations and bottom DO during 2006-2019 indicated significant differences among stations (F = 50.63; DF = 3; $P < 0.0001$; N = 335). 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 for all years were 5.23 mg / L and 0.08, respectively. Mean and SE for bottom DO at station 1 were 3.80mg / L and 0.18; station 2 was 5.64 mg / L and 0.11; station 3 was 5.74 mg / L and 0.11; and station 4 was 5.75 mg / L and 0.11. Deterioration of DO at the uppermost station (station 1; Figure 3-12) since 2012 indicated that increased watershed development around Easton was the source of poor water quality rather than water intruding from downstream. During 2019, 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 2 and 3 had mean bottom DO above the target value during 2019, but were below the overall median for the time-series. Station 4 fell below the overall median DO and target level (= 5.0 mg / L; Figure 3-13).

An ANOVA of Broad and Harris Creeks station bottom DO measurements did not indicate significant differences among stations in either subestuary during sampling years. Annual station means in both subestuaries varied without trend around the time-series median for all sites (Figure 3-13). Additional information and analysis of Broad and Harris Creeks are described in **Estuarine Fish Community Sampling in Job 1, Section 3** (Uphoff et al. 2018).

We ranked the bottom trawl GMs for all species combined in each of the Choptank subestuaries sampled during 2006-2019 (Table 3-17). Tred Avon River was the only Choptank River subestuary sampled in 2019 and it ranked at the very bottom, 25th out of 25 surveys (Table 3-17). The GMs for Broad Creek, Harris Creek, and Tred Avon River in 2015 all ranked in the top 25 % and 2016 ranked in the top 32 %. The remaining years were scattered with no real pattern (Table 3-17).

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 among the three Choptank River subestuaries did suggest coherence in annual relative abundance of finfish (Table 3-18). Strong correlations of GMs were present between Broad Creek and Harris Creek (r = 0.87, P = 0.05, N = 5); Broad Creek and Tred Avon River (r = 0.83, P = 0.04, N = 6); a moderate correlation was present between Tred Avon River and Harris Creek (r = 0.69, P = 0.19, N

= 5; Table 3-18). Correlations of beach seine GMs with bottom trawl GMs for Broad Creek, Harris Creek and Tred Avon River were not significant, but sample sizes were small (Table 3-19).

Five species were in the top 90 % of finfish caught in the Tred Avon River during 2006-2019: Bay Anchovy (57.9 %), Spot (16.5 %), White Perch (adults and juveniles; 7.4 %), Hogchoker (7.1 %), and Striped Bass (adults and juveniles; 3.6 %; Figure 3-15); all except Hogchoker, were target species. An additional 33 species comprised the 'other species' category (Figure 3-15).

Species richness in the top 90 % of species collected in Tred Avon River trawl samples increased in 2019 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, but reappeared in 2019. Percent similarity in finfish species composition among stations 1-4 in the Tred Avon River decreased substantially after 2016, reaching 7 % in 2019 (Figure 3-17). 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 just at or above 50 % from 2007 to 2017. During 2006 and 2018-2019, the similarity index was below 25 %, reflecting rainfall and low salinity (Figure 3-17). Previous analyses conducted in 2018, suggested wet years with lower salinity would have species composition dissimilar to dry years with higher salinity. Spring 2019 was considerably wetter than summer.

We analyzed finfish species composition of bottom trawls in all mesohaline subestuaries sampled during 2003-2019 to see if changes in Tred Avon River in 2019 were unique. A similar change in finfish composition for all mesohaline systems was observed the last couple of years; Bay Anchovy dropped out of the top 90 % of species during 2018 and reappeared in 2019 (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 2019 changed over the years and some differences could reflect these changes.

The Tred Avon River adult White Perch trawl GM fell below the median time-series GM (6) in 2009-2011 and 2014-2016 (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 medians for the time-series in Broad Creek (2012-2017) and Harris Creek (2012-2016) were 4. 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 3-20).

Finfish seine GMs in the three Choptank subestuaries were highest during 2015, (Figure 3-20); 2012-2016 represented years in common among these three subestuaries. Seine GMs for all finfish in Tred Avon River samples were lowest in 2008 (77). Broad Creek and Harris Creek had their lowest GMs in 2012 (106 and 131, respectively). Tred Avon River seine GMs have decreased since 2015 and remained steady since 2017 (Figure 3-20).

Seven species were in the top 90 % of finfish in beach seines for all years caught in the Tred Avon River: Atlantic Silverside (37.8 %), Atlantic Menhaden (19.0 %), White Perch (15.1 %), Striped Killifish (7.7 %), Mummichog (6.7%), Bay Anchovy (3 %), and Striped Bass (2.9%), (Figure 3-21). An additional 40 species (7.8 %) were in the other species category (Figure 3-21). All species in the top 90 % of all the subestuaries, except Atlantic Silverside, Mummichog, and Striped Killifish were target species.

Modified PSD for White Perch in Choptank River subestuaries (Broad Creek, Harris Creek, and Tred Avon River) by year for 4.9m trawl samples varied greatly among subestuaries and years, but were generally lower in Tred Avon River (Table 3-21; Figure 3-22). The increase in Tred Avon River modified PSD increased during 2012-2018 (from 4.7% to 53.3%) reflects the size progression of the strong 2011 year-class (juvenile index = 35.2, highest of the time-series; Durrell and Weedon 2019) into harvestable size. The 2011 year-class followed a stretch of lesser year-classes during the 2000s. Tred Avon River fell below 30% in 2019 and this may reflect dilution as two good year-classes (2014 and 2015; juvenile indices ~ 14.4 and 14.8, respectively; Durrell and Weedon 2019) become increasingly available in trawl samples. Less developed Choptank subestuary, Harris Creek, had higher modified PSDs for trawl samples than Tred Avon River during corresponding sampling years (2012-2016). Modified PSDs for trawl samples in Broad Creek fluctuated above and below modified PSDs in Tred Avon River for trawl samples during matching sampling years (2012-2017; Table 3-21; Figure 3-22). Modified PSDs for seine sample of White Perch for Choptank River subestuaries indicated that smaller White Perch were more prevalent inshore and more prevalent in Tred Avon River than Broad and Harris Creeks. Modified PSDs were often less than 10% (Table 3-22).

Size quality of White Perch directly aligned with the percentage of all DO measurements below the target level (5.0 mg / L) although this may not indicate cause and effect. Tred Avon River is both the most developed watershed of the three Choptank River subestuaries and is closer to the Choptank River spawning area. Presence or absence of adult White Perch in trawl samples was negatively influenced by development and distance from their spawning area (Uphoff et al. 2011a). Sample sizes observed indicate that White Perch were more abundant in Tred Avon River, especially during 2012-2016 when all three Choptank subestuaries were sampled at the same time and diminished size quality may reflect density-dependence.

Tred Avon River median Secchi depths ranged from 0.4 m to 0.75 m during 2006-2019; 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 3-23). The three Choptank subestuaries Secchi depths were strongly correlated with each other (Table 3-23).

Tred Avon River, Broad Creek, and Harris Creek SAV coverage were included in the mouth of the Choptank River region (VIMS 2019). SAV coverage increased substantially from 1% in 2012 to 11.8% in 2017 (Figure 3-24) and was far above the time-series median (4%) in 2017 (Figure 3-24). The 2018 survey was only partially mapped. An estimate for 2019 was not available.

Median pH in Tred Avon River ranged from 7.4 (2007) to 8.1 (2019; Figure 3-25). Broad Creek median pH ranging from 7.8 (2014) to 8.1 (2015). Harris Creek median pH ranging from 7.7 (2013-2014) to 8.0 (2015; Figure 3-25). Median pH in Broad Creek and Harris Creek were strongly correlated, but remaining combinations were not (Table 3-24).

Tred Avon River had its second lowest median salinity in 2019 (7.5 ‰; Figure 3-26). 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 3-18). 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 3-26). Median salinities of all three Choptank subestuaries were positively and strongly correlated among each other; these strong correlations among these subestuaries reflected their proximity to one another (Table 3-25).

In 2019, finfish trawl catches in the Tred Avon River bottom channel were only slightly above their lowest level in 2018, while inshore seine catches were average and remain steady since 2017. 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; however, in 2019, Bay Anchovy reappeared. 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 and 2019 was notable for Tred Avon River and other mesohaline systems saw a dramatic shift in species composition in bottom trawl catches in 2018 and 2019 as well. Tred Avon River seine GM in 2019 was similar to previous years. Anecdotally, during 2018 and 2019 sampling 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.

Queen Anne's County Subestuaries - Estimated percentages of watershed 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 2019; Table 3-26; Figure 3-1). 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 2019) had the lowest fraction of water coverage (Table 3-26).

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 3-2; Figure 3-1). All Queen Anne County subestuaries were below the rural development target (IS 5 % = 0.37); however, Corsica River is the closest to breaching that target (C/ha = 0.27 in 2018). Time-series for all subestuaries started at a rural level of development in 1950 (C/ha ranged from 0.01 to 0.05; 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).

In 2019, bottom DO readings breaching the threshold (3.0 mg / L) and target (5.0 mg / L) were most frequent in the Wye River (13% and 52%, respectively; Table 3-27). Chester River and Langford Creek did not have threshold violations, and Corsica River had 5% of bottom DO readings violate the threshold (Table 3-27). Bottom DO target violations during 2019 for the Chester River were 24 %; Corsica River, 74 %; and Langford Creek, 50 %. Corsica River had threshold and target violations every year bottom DO was sampled, 2003-2008, 2010-2012, and 2018-2019; 66% of bottom DO measurements in Corsica River for all years sampled were below the DO target and 22% were below the DO threshold (Figure 3-28). Chester River had threshold violations 5 years out of 12 years and target violations every year; 40% were below the target and 4% were below the threshold (Figure 3-28). Langford Creek only had threshold violations in 2007 and target violations every year. Overall in Langford Creek, 36% of bottom DO measurements were below the target and 1% were below the threshold (Figure 3-28). Wye River had threshold violations in 2018 and 2019 and target violations every year; 47% were below the target and 8% were below the threshold (Figure 3-28).

Median bottom DO estimates ranged from 4.3 mg / L (2008) to 6.8 mg / L (1996) for the Chester River (Figure 3-29). Corsica River had the greatest change in median bottom DO from 2.9 mg / L (2012) to 5.3 mg / L (2018). Langford Creek median bottom DO estimates ranged from 5.8 mg / L (2018) to 6.1 mg / L (2006). Median bottom DO estimates ranged from 4.6 mg / L (2018) to 6.1 mg / L (2007) for the Wye River. Correlation analyses suggested a moderate, positive association of median bottom DO estimates between Corsica and Chester rivers (Table 3-28). Remaining correlations were weak (Table 3-28).

In 2019, Corsica River had the greatest percentage of all DO measurements (surface to bottom) below target (5.0 mg / L), 34 %; followed by Wye River, 33 %; Langford Creek, 17 %; and Chester River, 11 % (Table 3-27). Frequency of all DO violations were higher in 2019 than in 2018 for all subestuaries. Chester River had 4 of 12 years with target violations above 50 % for all DO measurements; Corsica River had 2 of 11 years above 50 %, all years were greater than or equal to 26 %. Langford Creek and Wye River did not have any target violations above 50 %; highest violation for Langford Creek, 29 %, and Wye River, 40 % (Table 3-27).

In 2019, mean bottom DO estimates at all stations of the Chester River were above the median of all years sampled (Figure 3-30). Chester River bottom DO measurements were recorded at all six trawl sites during 2019, and only recorded at three site locations in 2018: sites 01, 03, and 06 (N = 19; Figure 3-2). Corsica River mean bottom DO at stations 1, 2, and 3 were above or at the median of all years sampled, station 4 fell below (Figure 3-30). All stations in the Corsica River had a sizeable decline in median DO from 2018. Langford Creek mean bottom DO at all stations fell below the overall median DO (Figure 3-30). In 2018 and 2019, Wye River station 1 mean bottom DO fell below earlier sampling years, stations 2, 3, and 4 had a similar mean bottom DO. Only station 2 in 2019 in the Wye River was above the overall median of all years sampled (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 and 2019; 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 median and SE for bottom DO in Chester River during 2007-2012 and 2018-2019 were 4.96 mg / L and 0.08, respectively. The overall median and SE for bottom DO in Corsica River for years 2003-2012 and 2018-2019 were 4.26 mg / L and 0.12, respectively. The overall median and SE for bottom DO in Langford Creek for years 2006-2008 and 2018-2019 were 5.46 mg / L and 0.19, respectively. The overall median and SE for bottom DO in Wye River for years 2007-2008 and 2018-2019 were 5.05 mg / L and 0.16, 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 2003-2019 (Table 3-29). Chester River had the highest ranked GM in 2019, ranked 20th out of 34th; bottom trawls were not conducted in 2018 for Chester River (Table 3-29). Corsica River had the highest ranked GM (378 in 2003), followed by Langford Creek at 273 (2007), and Chester River at 259 (2011). The four 2019 GMs were grouped together and a majority ranked higher than 2018 GMs (Table 3-29). 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 (Figure 3-31). Correlations among Chester River (2007-2012, 2019), Corsica River, Langford Creek, or Wye River, annual GMs were positive and strong, but sample sizes were low for comparisons (Table 3-30).

Chester River bottom trawl catches for all sampling years were composed mostly of White Perch (adults and juveniles; 70%), Bay Anchovy (12%), Spot (10%), and other species (28 species; 7%; Figure 3-32). Three species defined the top 90% of finfish caught in the Corsica River for all sampling years, White Perch (adults and juveniles; 71%), Bay Anchovy (16%), and Spot (7%). The other species category included 24 additional species, comprising of 4% of the finfish catch. Langford Creek bottom trawl catches for all sampling years were composed of White Perch (adults and juveniles; 70%), Bay Anchovy (19%), and other species (21 species; 10%; Figure 3-32). Three species comprised the top 90% of finfish in the Wye River for all sampling years, Bay Anchovy (45%), Spot (27%), and White Perch (adults and juveniles; 18%). The other species category included 16 species (9%). Every subestuary had the same three finfish species that dominated the top 90%; however, Wye River was the only subestuary where White Perch was not the dominate finfish present.

Annual finfish composition for Chester River, Corsica River, and Langford Creek bottom trawl catches did not indicate a drastic shift in species composition during 2019 (Figure 3-33). Annual finfish composition for Wye River bottom trawl catches did undergo a shift in species composition; Spot is the primary species in 2019, whereas White Perch was the primary species in 2018. White Perch (juveniles and adults) make up the top 90 % of species present in Chester and Corsica Rivers, and Langford Creek. Chester and Corsica Rivers also experienced an increase in the *Ictaluridae* family in the top 90% of species; Chester River, Channel and White Catfishes, and Corsica River, Brown Bullhead. Corsica River also had Spot present in the top 90 % of species. Bay Anchovy reappeared in the top 90% of species during 2019 in Langford Creek (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 its greatest finfish seine GM in 2018 (237) and lowest in 2006 (60). Seine catch GMs for the Wye River ranged from 79 (2008) to 182 (2018; Figure 3-33). Seine catch GMs in Langford Creek and Wye River decreased by half from 2018 to 2019 (Figure 3-34). Chester River had a slight decline in seine catch GM between 2018 and 2019. Seine sampling was not conducted in the Corsica River in 2019 due to limited beach sites and increased debris. Additional correlation analysis for Chester River between seine and trawl GMs for all years sampled from 1994 to 2019 did not indicate meaningful associations.

Chester River seine catches had 9 species in the top 90 %: Atlantic Silverside (32 %), White Perch (adults and juveniles; 29 %), Blueback Herring (7 %), Bay Anchovy (5 %), Mummichog (4 %), Spottail Shiner (4 %), Atlantic Menhaden (3 %), Gizzard Shad (3 %), Striped Bass (adults and juveniles; 2 %), and other species (44 species; 8 %). Langford Creek seine catches were comprised of Atlantic Silverside (36 %), Atlantic Menhaden (17 %), White Perch (adults and juveniles; 16 %), Striped Killifish (8 %), Blueback Herring (5 %), Alewife (3 %), Mummichog (2 %), and other species (26 species; 8 %). Wye River seine finfish catches included Atlantic Silverside (31 %), White Perch (adults and juveniles; 20 %), Atlantic Menhaden (18 %), Mummichog (10 %), Striped Killifish (8 %), Bay Anchovy (2 %), and other species (24 species; 9 %; Figure 3-35). Seine sampling was not conducted in Corsica River in 2019, for species composition see Uphoff et al. (2018). A majority of species in the top 90% of all the subestuaries were considered target species, with Atlantic Silverside, Mummichog, and Striped Killifish being exceptions. Chester River and Langford Creek, both had Blueback Herring in the top 90 %; upper Chester River is a spawning area for anadromous Herring. Bay Anchovy was only present in the top 90 % of species in the Wye River.

Modified PSDs for White Perch sampled in trawl samples in Queen Anne's County subestuaries, Chester River, Corsica River, Langford Creek, and Wye River, by year were low (< 4%) for all subestuaries and years with the exception of Wye River during 2018 and 2019 (modified PSDs = 14.3% and 47.5%, respectively; Table 3-31; Figure 3-36). Wye River modified PSD increased substantially in 2018 (14.3%) and 2019 (47.5%), similar to White Perch modified PSDs estimated within Choptank River subestuaries (described previously). During early sampling years, 2007-2008, Wye River trawl samples corresponded more with Queen Anne's County subestuaries, Chester River, Corsica River, and Langford Creek (Figure 3-36). Modified PSDs of seine samples for Queen Anne's County subestuaries, Chester River, Corsica River, and Langford Creek, varied slightly more than trawl samples, but were ≤ 5% for all years sampled except for Wye River during 2018 and 2019 (8.4% and 21.6%, respectively; Table 3-32; Figure 3-36).

Development and dissolved oxygen violations did not align with modified PSDs in these subestuaries. The Chester River and its tributaries, Corsica River and Langford Creek, had the lowest PSDs compared to the lower mid-Bay subestuary, Wye River, which was the only system within Queen Anne's County to achieve a PSD greater than 30% in trawl samples (Table 3-31; Figure 3-36). Location within the mid-Bay may be more influential than development and dissolved oxygen violations.

There was little evidence of long-term change in Secchi depths in Queen Anne County watersheds since the mid-to-late 2000s (Figure 3-37). There is a suggestion of a downshift in Secchi depths from higher levels in years prior in Chester River. Chester River was the only subestuary where the median Secchi depth increased by 0.05 m in 2019 (Figure 3-37). Langford Creek's median Secchi depth remained constant at 0.5 m from 2018 to 2019. Both Corsica and Wye Rivers decreased in median Secchi depth in 2019. Wye River had the largest decrease between 2018 (0.5 m) to 0.3 in 2019; whereas, Corsica River decreased from 0.5 (2018) to 0.4 (2019).

Coverage of water area in SAV varied among subestuaries; 2019 SAV data was not available at the time of this report. Chester River SAV coverage included all segments (upper, middle, and lower) of the river. Chester River SAV coverage ranged between 0 % and 2.3 % during 1989-2018 (Figure 3-38). Coverage in 2018 (2 %) was above the median of the time-series (0.4 %). Coverage data for 2003 was partial and not included in the median time-series. Coverage of SAV in Eastern Bay included Miles and Wye Rivers and varied between 0% and 8% from 1989 to 2018 (Figure 3-38). In 2018, SAV coverage (2.6 %) was at the median of the time-series (2.6 %; Figure 3-38). 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 fluctuated over the years; pH was not collected in some years because it was not a component of the water quality equipment used. Chester River pH data was available from 1995-1998 and 2018-2019; in 2019 median pH was 7.4 and annual median pH ranged from 6.5 (1997) to 7.4 (2018; Figure 3-39). Corsica River median pH for 2019 was 7.5 and annual median pH ranged from 7.5 (2018, 2019) to 7.7 (2006). Langford Creek annual median pH ranged from 7.4 (2019) to 7.9 (2006). Wye River annual median pH ranged from 7.6 (2018) to 7.9 (2019). Correlations of annual median pH among Queen Anne County subestuaries were inconsistent, indicating different dynamics. Moderate or near moderate positive correlations were found for Corsica River and Langford Creek and Corsica River and Wye River. A strong negative correlation was detected for Corsica River and Langford Creek. Sample sizes were low for these comparisons (Table 3-33).

Median salinity fluctuated substantially among years and subestuaries. Corsica River, Langford Creek, and Wye River had similar median salinities in 2018 and 2019 (Figure 3-40). Chester River median salinity increased from 1.4 ‰ (2018) to 3.5 ‰ (2019), still well below the mesohaline salinity range. The Chester River is normally mesohaline, but was oligohaline in 1996, 2011, 2018, and 2019. 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, 2019) to 11.7 ‰ (2007; Figure 3-40).

Correlations of median salinity estimates among the Queen Anne's County subestuaries were positive and strong ($r = 0.85$-0.97; Table 3-34). These strong correlations indicated similar influences could be present, but sample sizes were small for some comparisons. Additionally, due to the extreme change in salinity within the Chester River in 2011, 2018, and 2019, we examined the correlation of salinity (‰) and bottom DO (mg / L) measurements from Chester River. The correlation was weak ($r = -0.18$; $P = 0.0005$; $N = 260$; Figure 3-41).

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 = 0.24; N = 61; Table 3-35). Plots of the annual GM of catches of all species combined in Head of Bay and Choptank River indicated an interesting switch around the early 1980s; 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 very weak for GMs of all finfish in the Head of Bay and Chester River (r = -0.06; P = 0.78; N = 22; Table 3-35). Annual GM of the Chester River (all species combined) was not coherent with the Head of Bay system during 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 2007-2019, 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 2007-2019 in the Chester River indicated a moderate positive association with the Head of Bay system (r = 0.77; P = 0.07; N = 8). This moderate 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.

A modest correlation between Head of Bay and Tred Avon River annual GM of catches of all finfish (r = 0.55; P = 0.04; N = 14) was found (Table 3-35). A strong, positive association was present between the Choptank system and Tred Avon River annual GM (r = 0.80; P = 0.0006; N = 14; Table 3-35). Tred Avon River GMs (all species) likely reflected abundance in the Choptank River. Seine GMs in the Tred Avon River and Choptank River appeared fairly similar until 2019, when Tred Avon River GM dropped drastically and Choptank River GM remained steady (Figure 3-42c).

Summary – The effects of high precipitation in 2018 did not have a lingering impact on survey water quality measurements during 2019. Salinities in subestuaries sampled either increased or remained within bounds of what had been observed previously, remaining in their salinity class. Chester River has shown short-term improvement, although that could reflect it shifting to oligohaline; salinity increased during 2019, but remained oligohaline instead of returning to mesohaline. 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 2019 was most likely to be above the target level and below threshold measurements were uncommon in Chester River and its two tributaries. Corsica River, one tributary to the Chester River, had a noticeable improvement in bottom DO during 2018-2019 compared to earlier years sampled; the increase may reflect the State's designation as a targeted restoration watershed in 2005 which provided additional funding for several restoration programs to occur, as well as an upgrade to the wastewater treatment plant that occurred in 2010 (CRC 2012). Most bottom DO measurements in Wye River fell between the target and threshold level, below threshold readings decreased slightly in 2019. Station 1 (upper site) in the Wye River during 2018 and 2019 showed substantially lower bottom DO readings than previous years, possibly due to increased precipitation that would increase run-off of nutrients and organic matter. We noted an increase in leaf litter in seine and trawl samples during the summer of 2018 and decomposition of this organic matter may have increased oxygen demand. Frequency of below threshold bottom DO continued to increase in 2019 in Tred Avon River (this watershed is approaching the development threshold) and below target DO became more frequent. Other water quality metrics (pH and Secchi depth) in the subestuaries sampled during 2019 were within previous years' ranges. Finfish catches in trawls sampling bottom water habitat remained steady or slightly increased among all subestuaries sampled. Species composition changed slightly, reflecting the reappearance of Bay Anchovy throughout most of the subestuaries sampled; Spot, Channel Catfish, White Catfish, and Brown Bullhead also increased in presence. Inshore seine catches were within a normal range. Modified PSDs for trawl and seine samples for subestuaries sampled in 2019 indicated that mid-Bay subestuaries, Tred Avon River and Wye River, have greater population densities of White Perch of interest to anglers compared to the White Perch communities in upper-Bay subestuaries, Chester River, Corsica River, and Langford Creek. While it appears that heavy rainfall and high freshwater discharge into the Chesapeake Bay and its tributaries during 2018 may have slightly impacted the upper- and mid-Bay subestuaries with lower salinities, lower DO, and smaller finfish catches (GMs for 2018-2019 were among the lowest of the time-series for a majority of the subestuaries sampled), the effects of very wet conditions in 2018 caused quick changes that lingered during 2019. Overall, we saw increases in water quality parameters and increased finfish catches with increased species composition. Our assessment of habitat, particularly the subestuaries sampled for the Queen Anne's County comprehensive growth plan, provided additional insight into the subestuaries and what can be expected during dry and wet years.

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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 2019.

2019 Sampled Subestuaries						
Area	Subestuary	IS	C/ha	Land Hectares	Water Hectares	Salinity Class
Mid-Bay	Chester River	2.8	0.15	99,542	21,084	Oligohaline*
Mid-Bay	Corsica River	4.0	0.27	9,671	565	Mesohaline
Mid-Bay	Langford Creek	2.0	0.07	9,631	1,306	Mesohaline
Mid-Bay	Tred Avon River	9.1	0.77	9,561	3,086	Mesohaline
Mid-Bay	Wye River	2.3	0.10	20,395	2,670	Mesohaline

* Chester River is typically a mesohaline subestuary. Salinity data taken in 2019 indicated salinity levels less than 5.0 ‰ for oligohaline.

Table 3-2. Estimates of structures per hectare (C / ha) and land use percentages from Maryland Department of Planning (2002 and 2010) for subestuaries sampled 2003-2019.

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
Bush River	2019	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
Chester River	2019	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
Corsica River	2019	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	0.74	30.6	1.0	32.1	35.6
Gunpowder River	2016	0.74	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
Langford Creek	2019	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	0.55	34.1	0.8	45.3	19.8
Rhode/West Rivers	2004	0.56	34.1	0.8	45.3	19.8
Rhode/West Rivers	2005	0.56	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	2.36	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
Tred Avon River	2019	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	2010	0.21	31.6	4.6	44.9	18.7
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
Wye River	2019	0.10	64.9	0.6	23.0	10.9

Table 3-3. Summary of water quality parameter statistics for subestuaries sampled in 2019. Measurements for pH were calculated from H+ concentrations and back-converted for reporting here.

		Surface Measurements					Bottom Measurements					
System	Statistics	Temp ©	DO (mg / L)	Cond (umhols)	Salinity	pH	Temp ©	DO (mg / L)	Cond (umhols)	Salinity	pH	Secchi
Chester River	Mean	27.19	6.82	6881.09	3.82	7.31	26.73	5.77	7371.75	4.11	7.32	0.59
	Standard Error	0.34	0.14	552.46	0.32	8.38	0.45	0.14	794.51	0.47	8.27	0.04
	Median	25.66	6.85	6873.00	3.74	7.35	25.52	5.90	6870.00	3.74	7.26	0.50
	Mode	25.66	7.70	.	0.67	7.94	.	4.98	.	.	7.06	0.50
	Kurtosis	-1.80	-0.21	-0.72	-0.61	-0.34	-1.64	-1.47	-0.67	-0.57	.	4.28
	Skewness	0.29	0.36	0.42	0.51	-0.05	0.47	-0.08	0.45	0.54	.	1.80
	Minimum	24.04	4.90	1143.00	0.56	6.79	24.18	4.53	1277.00	0.63	7.01	0.30
	Maximum	30.96	9.25	16032.00	9.38	8.38	30.36	6.96	16038.00	9.38	8.32	1.30
	Count	55	55	55	55	55	28	28	28	28	28	28
Corsica River	Mean	26.21	7.26	10874.56	6.21	7.70	26.11	4.66	12031.00	6.92	7.46	0.43
	Standard Error	0.56	0.35	1157.60	0.70	8.40	0.57	0.29	1193.00	0.73	8.28	0.03
	Median	25.36	6.98	11811.50	6.73	7.68	25.39	4.96	12664.00	7.47	7.47	0.40
	Mode	29.82	.	.	.	.	.	.	.	.	.	0.40
	Kurtosis	-0.56	-0.92	-1.15	-1.20	.	-0.16	-0.12	-1.09	-1.14	.	1.43
	Skewness	1.13	0.38	-0.39	-0.34	0.41	1.17	-0.94	-0.47	-0.45	.	0.56
	Minimum	24.15	5.14	2786.00	1.43	7.31	23.98	2.58	4972.00	2.64	7.18	0.25
	Maximum	30.01	9.84	17184.00	10.12	8.96	29.77	5.94	17359.00	10.22	8.42	0.70
	Count	16	16	16	16	16	13	13	13	13	13	16
Langford Creek	Mean	27.33	6.75	11154.19	6.34	7.51	27.12	5.20	11382.80	6.48	7.38	0.62
	Standard Error	0.34	0.18	503.66	0.31	8.70	0.42	0.17	737.78	0.45	8.45	0.04
	Median	26.80	6.84	10757.50	6.07	7.49	26.64	5.14	12006.50	6.74	7.39	0.55
	Mode	25.43	.	.	7.85	7.64	29.35	4.96	.	.	7.35	0.50
	Kurtosis	-1.14	-0.95	-1.55	-1.56	.	-1.58	3.61	-1.39	-1.36	0.28	0.07
	Skewness	0.59	-0.16	0.05	0.06	0.42	0.29	-0.45	0.14	0.18	0.07	0.71
	Minimum	25.13	5.06	6680.00	3.63	7.24	24.78	3.07	6683.00	3.63	7.11	0.30
	Maximum	31.10	8.42	15197.00	8.84	7.84	30.36	6.87	16962.00	9.94	7.72	1.00
	Count	32	32	32	32	32	20	20	20	20	20	20
Tred Avon River	Mean	28.64	6.94	13599.55	7.82	8.22	28.18	4.54	14161.75	8.15	7.96	0.38
	Standard Error	0.33	0.15	420.17	0.27	9.12	0.43	0.32	624.77	0.40	8.59	0.02
	Median	28.94	6.99	12771.50	7.29	8.32	28.67	4.52	13154.00	7.53	8.08	0.40
	Mode	29.29	7.58	.	6.45	8.66	.	.	.	.	8.05	0.40
	Kurtosis	-0.82	2.13	-1.05	-1.03	0.36	-0.93	0.12	-0.74	-0.74	-0.55	0.24
	Skewness	0.08	0.35	0.51	0.53	-0.03	-0.01	0.11	0.63	0.61	-0.24	-0.61
	Minimum	25.51	4.52	9696.00	5.38	7.75	25.23	1.87	10637.00	5.93	7.33	0.20
	Maximum	32.81	9.91	18602.00	11.06	9.03	31.63	7.40	20300.00	12.10	9.03	0.50
	Count	40	40	40	40	40	20	20	20	20	20	20
Wye River	Mean	27.47	6.44	14744.21	8.56	7.99	27.46	4.71	15022.47	8.74	7.87	0.47
	Standard Error	0.43	0.25	490.32	0.31	8.85	0.60	0.32	715.12	0.46	8.66	0.16
	Median	25.86	6.73	14261.50	8.28	8.04	25.67	4.74	14434.00	8.39	7.95	0.30
	Mode	25.17	7.14	.	6.30	7.84	.	3.93	.	.	7.75	0.30
	Kurtosis	-1.63	-0.86	-1.33	-1.33	-0.56	-1.83	-1.17	-1.52	-1.52	.	19.57
	Skewness	0.47	-0.21	0.17	0.17	-0.22	0.42	-0.01	0.13	0.14	0.44	4.40
	Minimum	24.66	3.68	10631.00	5.99	7.43	24.76	2.41	10844.00	6.09	7.49	0.20
	Maximum	31.64	8.88	19496.00	11.59	9.18	31.61	6.86	20136.00	12.01	9.00	3.50
	Count	34	34	34	34	34	19	19	19	19	19	20

Table 3-4. 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 2019. 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**
Chester River	Mesohaline	0.15	133	11%	34	24%	0%
Corsica River	Mesohaline	0.27	53	34%	19	74%	5%
Langford Creek	Mesohaline	0.07	85	17%	24	50%	0%
Tred Avon River	Mesohaline	0.77	96	30%	24	71%	17%
Wye River	Mesohaline	0.10	73	33%	23	52%	13%

Table 3-5. Beach seine catch summary, 2019. C / ha = structures per hectare. GM CPUE = geometric mean catch per seine sample. Italics designate target species. Young of the year or juveniles = JUV.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Chester River	6	31	24	Atlantic Silverside Mummichog *White Perch (Adults)* *Spottail Shiner* Banded Killifish *White Perch (JUV)* *Bay Anchovy* Striped Killifish	0.15	3,994	112
Langford Creek	3	15	20	Atlantic Silverside Striped Killifish *White Perch (Adults)* *Atlantic Menhaden* *Bay Anchovy*	0.07	1,829	109
Tred Avon River	4	24	25	Atlantic Silverside *Atlantic Menhaden* *White Perch (Adults)* Mummichog Striped Killifish *Bay Anchovy* *Striped Bass (JUV)*	0.77	4,041	139
Wye River	3	17	20	*White Perch (Adults)* *Atlantic Menhaden* Atlantic Silverside Mummichog Striped Killifish *Spottail Shiner* Pumpkinseed *Striped Bass (JUV)*	0.10	1,819	98
Grand Total	16	87	35	Atlantic Silveside *White Perch (Adults)* *Atlantic Menhaden* Mummichog Striped Killifish *Spottail Shiner* Banded Killifish *Bay Anchovy* *White Perch (JUV)*		11,683	

Table 3-6. Bottom trawl catch summary, 2019. C / ha = structures per hectare. GM CPUE = geometric mean catch per trawl sample. Italics designate target species. Young of the year or juveniles = JUV.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Chester River	6	34	16	*White Perch (Adults)* *White Perch (JUV)* Channel Catfish White Catfish	0.15	3,248	77
Corsica River	4	24	15	*White Perch (Adults)* *White Perch (JUV)* *Spot* Brown Bullhead	0.27	1,398	32
Langford Creek	4	24	9	*White Perch (Adults)* *Bay Anchovy* *Spot*	0.07	1,370	42
Tred Avon River	4	24	14	*White Perch (Adults)* *Bay Anchovy* *Spot* Hogchoker	0.77	1,301	31
Wye River	4	24	8	*Spot* *White Perch (Adults)*	0.10	742	19
Grand Total	22	130	25	*White Perch (Adults)* *Spot* *White Perch (JUV)* *Bay Anchovy* Channel Catfish White Catfish		8,059	

Table 3-7. Subestuaries sampled during 2003–2019, 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 (ºC)		Dissolved Oxygen (mg / L)	
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	1995	0.11	26.70	26.32	7.58	6.28
	1996	0.11	23.86	23.63	7.62	6.71
	1997	0.11	27.02	26.32	7.66	6.37
	1998	0.11	28.00	27.49	7.40	6.26
	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
	2019	0.15	27.45	27.05	6.75	5.77
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
	2019	0.27	27.24	27.04	6.82	4.39
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
	2019	0.07	27.77	27.51	6.69	5.07
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

Table 3-7 (Cont.)

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	2.36	26.93	26.07	6.86	1.78
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
	2019	0.77	28.62	28.22	6.79	4.49
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	0.22	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
	2019	0.10	27.68	27.67	6.33	4.68
		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

Table 3-7 (Cont.)

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	0.74	27.51	27.65	8.02	6.63
	2016	0.74	27.70	26.46	7.43	6.18
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	1989	0.44	26.84	26.72	8.95	8.48
	1990	0.46	26.82	26.39	10.15	9.56
	1991	0.48	27.27	27.28	11.01	10.41
	1992	0.50	27.49	26.73	11.53	10.41
	1993	0.51	27.59	26.72	11.20	9.43
	1994	0.53	26.84	26.53	11.67	10.86
	1995	0.56	27.79	27.29	10.86	9.35
	1996	0.58	25.25	24.80	11.43	10.51
	1997	0.61	26.85	26.59	9.17	8.28
	1998	0.64	27.55	27.38	10.35	9.65
	1999	0.67	27.40	26.83	9.65	8.52
	2000	0.69	24.82	24.46	8.85	8.36
	2001	0.72	26.41	26.36	7.58	7.48
	2002	0.72	26.75	26.57	7.95	7.70
	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

Table 3-7 (Cont.)

	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 3-8. 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 2003-2019, by salinity class. Level of significance = *P*. N = sample size.

DO Depth	Statistics	Temperature	C / ha
		Mesohaline	
Surface	r	-0.039	0.196
	P	0.719	0.068
	N	88	88
Bottom	r	0.040	-0.608
	P	0.709	<.0001
	N	88	88
		Oligohaline	
Surface	r	-0.307	0.432
	P	0.082	0.012
	N	33	33
Bottom	r	-0.601	-0.062
	P	0.0002	0.733
	N	33	33
		Tidal Fresh	
Surface	r	0.020	-0.140
	P	0.891	0.342
	N	48	48
Bottom	r	0.071	-0.129
	P	0.630	0.381
	N	48	48

Table 3-9. Pearson correlations (r) among Maryland Department of Planning (DOP) land use categories and with C / ha for mesohaline subestuaries sampled during 2003-2019. Land cover estimates were estimated by MD DOP for 2002 and 2010. *P* = level of significance. N = sample size.

			Land Use Categories			
	Statistics	**C/ha**	**Agriculture**	**Forest**	**Wetland**	**Urban**
C/ha	r					
	P	1				
	N					
Agriculture	r	-0.759				
	P	<.0001	1			
	N	78				
Forest	r	0.076	-0.578			
	P	0.509	<.0001	1		
	N	78	78			
Wetland	r	-0.263	0.013	0.002		
	P	0.020	0.908	0.983	1	
	N	78	78	78		
Urban	r	0.898	-0.812	0.000	-0.116	
	P	<.0001	<.0001	0.998	0.311	1
	N	78	78	78	78	

Table 3-10. 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	52.61	52.61	53.84	<0.0001	
Residual	20	19.54	0.98			
Total	21	72.15				
r^2 = 0.7292						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.50	0.48	1.04	0.31	-0.50	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	7.02	7.02	11.81	0.0011	
Residual	54	32.10	0.59			
Total	55	39.12				
r^2 = 0.1795						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	7.05	0.55	12.80	<0.0001	5.95	8.16
Agriculture (%)	-0.03	0.01	-3.44	0.0011	-0.05	-0.01

Table 3-11. Statistics and parameter estimates for a quadratic regression of median bottom dissolved oxygen (DO) versus percent agricultural coverage (western and eastern shore combined).

Linear Model	**Median Bottom DO = Agriculture (%) Coverage**					
ANOVA	df	SS	MS	F	Significance F	
Regression	2	91.04	45.52	59.37	<0.0001	
Residual	75	57.50	0.77			
Total	77	148.54				
$r^2 = 0.6129$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	-0.11	0.46	-0.25	0.80	-1.03	0.80
Agriculture (%)	0.23	0.02	9.85	<0.0001	0.18	0.27
Agriculture (%)^2	-0.002	0.0003	-8.43	<0.0001	-0.003	-0.002

Table 3-12. Subestuaries sampled during 2003-2019, 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	2015	448	4
Broad Creek	2016	422	5
Tred Avon River	2015	419	6
Harris Creek	2015	408	7
Harris Creek	2016	406	8
Corsica River	2003	380	9
Harris Creek	2014	359	10
Chester River	2011	296	11
Langford Creek	2007	274	12
Langford Creek	2006	259	13
Corsica River	2004	252	14
Corsica River	2011	246	15
Broad Creek	2017	231	16
Tred Avon River	2016	219	17
Tred Avon River	2011	207	18
Broad Creek	2014	204	19
Harris Creek	2013	201	20
Chester River	2007	185	21
Broad Creek	2013	177	22
Corsica River	2006	176	23
Chester River	2010	174	24
Wye River	2007	171	25
Tred Avon River	2010	171	26
Corsica River	2012	168	27
Tred Avon River	2014	164	28
Corsica River	2010	163	29
Rhode River	2005	163	30
Langford Creek	2008	162	31
Tred Avon River	2017	157	32
Tred Avon River	2018	138	33
Tred Avon River	2012	135	34
Chester River	2012	135	35
Corsica River	2007	133	36
Fishing Bay River	2006	131	37
Harris Creek	2012	131	38
Transquaking River	2006	131	39
West River	2005	125	40
Chester River	2008	123	41
Tred Avon River	2006	121	42
Wicomico River	2010	120	43
Wye River	2008	115	44

Table 3-12 (Cont.)

Corsica River	2005	111	45
South River	2003	110	46
Wicomico River	2012	110	47
Broad Creek	2012	106	48
Tred Avon River	2007	103	49
Tred Avon River	2009	103	50
Tred Avon River	2013	96	51
Corsica River	2008	90	52
Miles River	2004	82	53
Wicomico River	2017	81	54
Chester River	2009	79	55
Tred Avon River	2008	77	56
Chester River	2019	77	57
Miles River	2005	72	58
Wicomico River	2011	65	59
Wicomico River	2003	59	60
St. Clements River	2005	54	61
Langford Creek	2019	42	62
Rhode River	2004	38	63
South River	2005	35	64
Blackwater River	2006	35	65
Breton Bay	2005	34	66
West River	2004	34	67
Magothy River	2003	33	68
Corsica River	2019	32	69
St. Clements River	2003	31	70
Tred Avon River	2019	31	71
Langford Creek	2018	29	72
South River	2004	21	73
Wye River	2019	19	74
Breton Bay	2003	18	75
Corsica River	2018	18	76
St. Clements River	2004	17	77
Breton Bay	2004	16	78
Severn River	2017	16	79
Wye River	2018	14	80
Severn River	2003	9	81
Severn River	2004	5	82
Severn River	2005	3	83
Oligohaline			
Bush River	2011	666	1
Nanjemoy Creek	2013	576	2

Table 3-12 (Cont.)

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
Tidal-Fresh			
Mattawoman Creek	2014	580	1
Northeast River	2010	392	2
Piscataway Creek	2011	320	3
Northeast River	2014	291	4

Table 3-12 (Cont.)

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 3-13. Pearson correlations of annual 4.9 m trawl geometric means (GM) of subestuaries sampled from 2003-2019 with intensity of watershed development (C / ha), by salinity class.

Geometric Mean (GM)	Statistics	C / ha
Tidal-Fresh	r	-0.13
	P	0.46
	N	34
Oligohaline	r	-0.08
	P	0.64
	N	40
Mesohaline	r	-0.21
	P	0.05
	N	83

Table 3-14. 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 3-15. 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%
	2019	0.77	96	30%	24	71%	17%

Table 3-16. 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	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	-0.07		
	P	0.91	1	
	N	5		
Tred Avon River	r	0.39	0.73	
	P	0.45	0.16	1
	N	6	5	

Table 3-17. Choptank subestuaries sampled during 2006-2019, ranked by annual 4.9 m trawl catch geometric mean (GM).

River	Year	GM	Rank
Broad Creek	2015	448	1
Broad Creek	2016	422	2
Tred Avon River	2015	419	3
Harris Creek	2015	408	4
Harris Creek	2016	406	5
Harris Creek	2014	359	6
Broad Creek	2017	231	7
Tred Avon River	2016	219	8
Tred Avon River	2011	207	9
Broad Creek	2014	204	10
Harris Creek	2013	201	11
Broad Creek	2013	177	12
Tred Avon River	2010	171	13
Tred Avon River	2014	164	14
Tred Avon River	2017	157	15
Tred Avon River	2018	138	16
Tred Avon River	2012	135	17
Harris Creek	2012	131	18
Tred Avon River	2006	121	19
Broad Creek	2012	106	20
Tred Avon River	2007	103	22
Tred Avon River	2009	103	22
Tred Avon River	2013	96	23
Tred Avon River	2008	77	24
Tred Avon River	2019	31	25

Table 3-18. 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.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.96		
	P	0.01	1	
	N	5		
Tred Avon River	r	0.95	0.84	
	P	0.003	0.07	1
	N	6	5	

Table 3-19. Pearson correlations (r) of annual beach seine GM against annual 4.9 m trawl catch GM for Choptank subestuaries, Broad Creek, Harris Creek, and Tred Avon River. Level of significance of Pearson correlation = P. Sample size (N) for the number of GM measurements for each subestuary sampled.

		Seine Geometric Mean		
Trawl Geometric Mean	**Statistics**	**Broad Creek**	**Harris Creek**	**Tred Avon River**
Broad Creek	r	-0.58		
	P	0.23		
	N	6		
Harris Creek	r		-0.49	
	P		0.40	
	N		5	
Tred Avon River	r			-0.07
	P			0.82
	N			14

Table 3-20. 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.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.02		
	P	0.98	1	
	N	5		
Tred Avon River	r	0.63	0.45	
	P	0.18	0.45	1
	N	6	5	

Table 3-21. Modified proportional stock density (PSD) of White Perch in Choptank River subestuaries is the proportion of 4.9m trawl samples with quality length or greater White Perch. N_{TOTAL} is the total number of White Perch (all juveniles and adults) captured in both seine and trawl catches. Number of L_{STOCK} is the number of all adult White Perch (adults age +1). Number of $L_{QUALITY}$ is the number of harvestable adults (≥ 200 mm).

Subestuary	Year	N_{TOTAL}	N L_{STOCK}	N $L_{QUALITY}$	Modified PSD
Broad Creek	2012	86	86	4	4.7%
	2013	42	42	3	7.1%
	2014	38	38	14	36.8%
	2015	214	21	1	4.8%
	2016	60	51	15	29.4%
	2017	16	16	5	31.3%
Harris Creek	2012	106	106	45	42.5%
	2013	244	237	26	11.0%
	2014	52	51	11	21.6%
	2015	39	39	27	69.2%
	2016	96	96	41	42.7%
Tred Avon River	2006	368	366	45	12.3%
	2007	426	397	22	5.5%
	2008	265	265	31	11.7%
	2009	150	150	30	20.0%
	2010	27	21	6	28.6%
	2011	828	95	19	20.0%
	2012	570	570	27	4.7%
	2013	225	225	11	4.9%
	2014	62	60	4	6.7%
	2015	282	80	18	22.5%
	2016	102	102	6	5.9%
	2017	126	118	39	33.1%
	2018	109	92	49	53.3%
	2019	554	553	147	26.6%

Table 3-22. Modified proportional stock density (PSD) of White Perch in Choptank River subestuaries is the proportion of seine samples with quality length or greater White Perch. N_{TOTAL} is the total number of White Perch (all juveniles and adults) captured in both seine and trawl catches. Number of L_{STOCK} is the number of all adult White Perch (adults age +1). Number of $L_{QUALITY}$ is the number of harvestable adults (≥ 200 mm).

Subestuary	Year	N_{TOTAL}	N L_{STOCK}	N $L_{QUALITY}$	Modified PSD
Broad Creek	2012	410	410	34	8.3%
	2013	128	107	19	17.8%
	2014	146	13	4	30.8%
	2015	536	152	13	8.6%
	2016	78	72	2	2.8%
	2017	0	0	0	0.0%
Harris Creek	2012	179	179	7	3.9%
	2013	48	48	2	4.2%
	2014	63	21	9	42.9%
	2015	517	70	7	10.0%
	2016	73	73	17	23.3%
Tred Avon River	2006	1,513	1,470	72	4.9%
	2007	924	729	40	5.5%
	2008	529	529	15	2.8%
	2009	171	157	15	9.6%
	2010	216	210	6	2.9%
	2011	1,480	146	10	6.8%
	2012	2,338	2,338	92	3.9%
	2013	394	393	21	5.3%
	2014	260	188	4	2.1%
	2015	753	141	4	2.8%
	2016	700	700	49	7.0%
	2017	92	68	4	5.9%
	2018	369	145	11	7.6%
	2019	810	745	80	10.7%

Table 3-23. Pearson correlations (r) of annual survey median Secchi depths for Broad Creek, Harris Creek, and Tred Avon River among subestuaries. *P* = level of significance. N = number of annual survey median Secchi depths.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.972		
	P	0.006	1	
	N	5		
Tred Avon River	r	0.864	0.928	
	P	0.027	0.023	1
	N	6	5	

Table 3-24. Pearson correlations (r) of annual median pH for Broad Creek, Harris Creek, and Tred Avon River among subestuaries. *P* = level of significance. N = number of annual survey median pH estimates.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.937		
	P	0.019	1	
	N	5		
Tred Avon River	r	0.396	0.174	
	P	0.437	0.779	1
	N	6	5	

Table 3-25. Pearson correlations (r) of annual survey median salinity (‰) for Broad Creek, Harris Creek, and Tred Avon River among subestuaries. *P* = level of significance. N = number of annual survey median salinity estimates.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.990		
	P	0.001	1	
	N	5		
Tred Avon River	r	0.995	0.979	
	P	<0.0001	0.004	1
	N	6	5	

Table 3-26. Percent of major land use categories estimated by Maryland Department of Planning (DOP 2010) in each of the Queen Anne's County subestuaries. 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 3-27. 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	1995	0.11	88	8%	30	10%	0%
Chester River	1996	0.11	84	6%	30	13%	0%
Chester River	1997	0.11	86	3%	30	10%	0%
Chester River	1998	0.11	89	2%	30	7%	0%
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%
Chester River	2019	0.15	133	11%	34	24%	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%
Corsica River	2019	0.27	53	34%	19	74%	5%
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%
Langford Creek	2019	0.07	85	17%	24	50%	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%
Wye River	2019	0.10	73	33%	23	52%	13%

Table 3-28. 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.

	Statistics	Chester River	Corsica River	Langford Creek	Wye River
Chester River	r				
	P	1			
	N				
Corsica River	r	0.665			
	P	0.103	1		
	N	7			
Langford Creek	r	0.326	0.013		
	P	0.674	0.984	1	
	N	4	5		
Wye River	r	-0.525	-0.004	0.236	
	P	0.475	0.996	0.764	1
	N	4	4	4	

Table 3-29. Chester River, Corsica River and Langford Creek, and Wye River sampled, ranked by annual 4.9 m trawl catch geometric mean (GM) during 2003-2019. 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
Langford Creek	2008	161	12
Corsica River	2010	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	2019	77	20
Chester River	2009	76	21
Langford Creek	2019	42	22
Corsica River	2019	32	23
Langford Creek	2018	27	24
Wye River	2019	19	25
Corsica River	2018	16	26
Wye River	2018	12	27

Table 3-30. 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 GMs for each subestuary.

	Statistics	Chester River	Corsica River	Langford Creek	Wye River
Chester River*	r				
	P	1			
	N				
Corsica River	r	0.927			
	P	0.008	1		
	N	6			
Langford Creek	r	0.997	0.959		
	P	0.047	0.010	1	
	N	3	5		
Wye River	r	0.999	0.994	0.995	
	P	0.034	0.006	0.006	1
	N	3	4	4	

*Only years 2007-2012, 2019 were used in correlations due to different sampling gear used in previous years.

Table 3-31. Modified proportional stock density (PSD) of White Perch in Queen Anne's County subestuaries is the proportion of 4.9m trawl samples with quality length or greater White Perch; Wye River is located in both Queen Anne's and Talbot Counties. N_{TOTAL} is the total number of White Perch (all juveniles and adults) captured in both seine and trawl catches. Number of L_{STOCK} is the number of all adult White Perch (adults age +1). Number of $L_{QUALITY}$ is the number of harvestable adults (≥ 200 mm). No trawl samples were conducted during 2018 in Chester River.

Subestuary	Year	N_{TOTAL}	N L_{STOCK}	N $L_{QUALITY}$	Modified PSD
Chester River	2007	4,800	3,247	5	0.2%
	2008	5,944	4,758	15	0.3%
	2009	4,018	1,953	6	0.3%
	2010	4,543	3,533	27	0.8%
	2011	13,881	2,580	3	0.1%
	2012	6,680	6,673	8	0.1%
	2018	.	.	.	.
	2019	2,123	1,416	23	1.6%
Corsica River	2003	8,914	2,721	44	1.6%
	2004	5,778	4,663	27	0.6%
	2005	2,887	1,633	20	1.2%
	2006	5,823	5,381	28	0.5%
	2007	2,550	1,775	10	0.6%
	2008	1,776	1,756	3	0.2%
	2009	6,652	4,214	9	0.2%
	2010	2,374	1,726	3	0.2%
	2011	11,491	1,508	2	0.1%
	2012	5,382	5,330	0	0.0%
	2018	822	280	9	3.2%
	2019	1,167	1,040	7	0.7%
Langford Creek	2006	6,426	6,424	42	0.7%
	2007	4,972	4,564	21	0.5%
	2008	2,692	2,691	17	0.6%
	2018	1,034	613	8	1.3%
	2019	1,117	1,042	19	1.8%
Wye River	2007	699	607	8	1.3%
	2008	415	415	4	1.0%
	2018	334	252	36	14.3%
	2019	238	236	112	47.5%

Table 3-32. Modified proportional stock density (PSD) of White Perch in Queen Anne's County subestuaries is the proportion of seine samples with quality length or greater White Perch; Wye River is located in both Queen Anne's and Talbot Counties. N_{TOTAL} is the total number of White Perch (all juveniles and adults) captured in both seine and trawl catches. Number of L_{STOCK} is the number of all adult White Perch (adults age +1). Number of $L_{QUALITY}$ is the number of harvestable adults (≥ 200 mm). No seine samples were conducted during 2019 in Corsica River.

Subestuary	Year	N_{TOTAL}	N L_{STOCK}	N $L_{QUALITY}$	Modified PSD
Chester River	2007	226	110	1	0.9%
	2008	1,217	1,018	1	0.1%
	2009	2,071	1,017	2	0.2%
	2010	2,091	1,351	0	0.0%
	2011	1,311	165	2	1.2%
	2012	2,746	2,707	8	0.3%
	2018	1,600	383	18	4.7%
	2019	992	698	3	0.4%
Corsica River	2003	2,566	604	14	2.3%
	2004	501	488	1	0.2%
	2005	1,326	864	12	1.4%
	2006	1,184	1,173	20	1.7%
	2007	994	623	12	1.9%
	2008	1,453	1,364	10	0.7%
	2009	1,457	589	11	1.9%
	2010	877	866	2	0.2%
	2011	1,231	110	0	0.0%
	2012	649	649	0	0.0%
	2018	396	178	7	3.9%
	2019	.	.	.	.
Langford Creek	2006	658	657	14	2.1%
	2007	574	395	20	5.1%
	2008	1,087	1,086	16	1.5%
	2018	279	169	1	0.6%
	2019	277	274	2	0.7%
Wye River	2007	1,103	1,058	36	3.4%
	2008	655	655	16	2.4%
	2018	346	239	20	8.4%
	2019	357	352	76	21.6%

Table 3-33. 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.

	Statistics	Chester River	Corsica River	Langford Creek	Wye River
Chester River	r				
	P	1			
	N				
Corsica River	r	.			
	P	.	1		
	N	2			
Langford Creek	r	.	0.591		
	P	.	0.294	1	
	N	2	5		
Wye River	r	.	0.477	-0.925	
	P	.	0.523	0.075	1
	N	2	4	4	

Table 3-34. 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.

	Statistics	Chester River	Corsica River	Langford Creek	Wye River
Chester River	r				
	P	1			
	N				
Corsica River	r	0.869			
	P	0.011	1		
	N	7			
Langford Creek	r	0.925	0.847		
	P	0.075	0.070	1	
	N	4	5		
Wye River	r	0.939	0.958	0.967	
	P	0.061	0.042	0.033	1
	N	4	4	4	

Table 3-35. 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.

	Statistics	Head of Bay	Choptank River
Choptank River	r	0.152	
	P	0.243	1
	N	61	
Chester River	r	-0.064	0.225
	P	0.777	0.314
	N	22	22
Tred Avon River	r	0.554	0.801
	P	0.040	0.0006
	N	14	14

Figures

Figure 3-1. Map illustrating subestuaries sampled in summer 2019, 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, and their land use categories. Land use data is based on Maryland Department of Planning (DOP) 2010 land use land cover data. Figure includes previously sampled subestuaries, Broad Creek (6; 2012-2017) and Harris Creek (7; 2012-2016), referenced throughout this report.

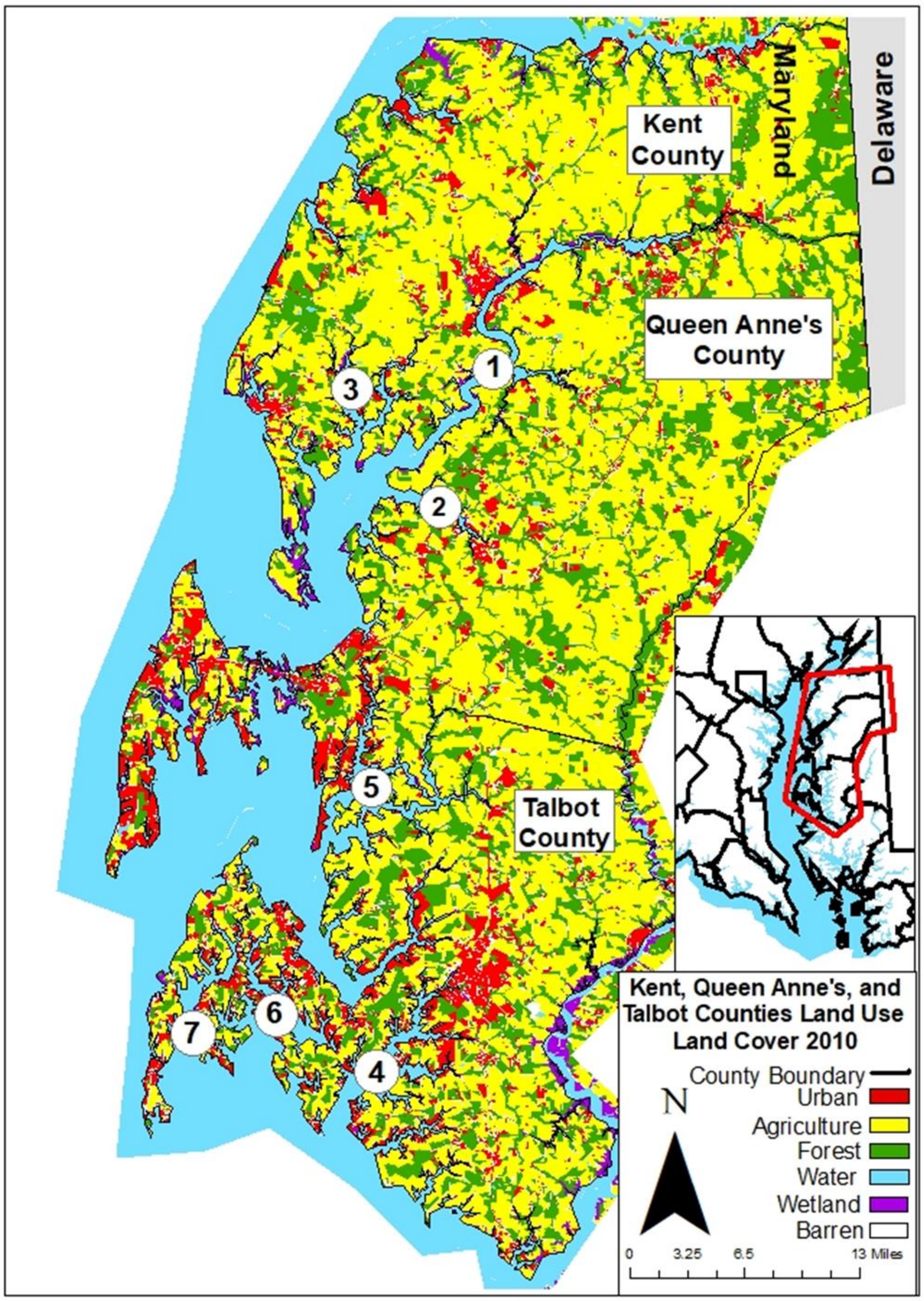


Figure 3-2. Map indicating current locations of 2019 sampling sites for subestuaries, Chester River, Corsica River, and Langford Creek located in Queen Anne's County, and Tred Avon River and Wye River located in Talbot County.

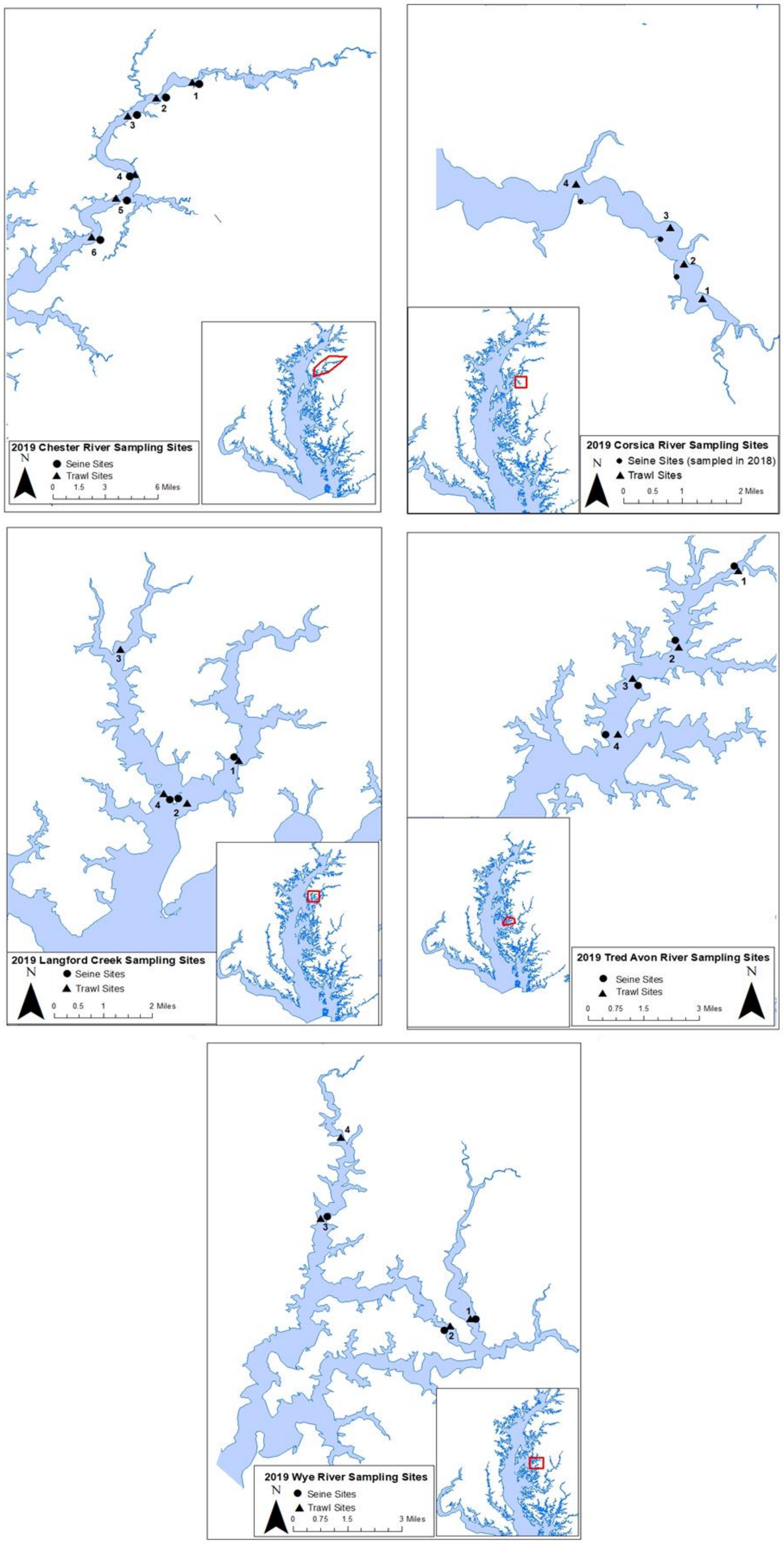


Figure 3-3. Trends in development (structures per hectare = C / ha) from 1950 to 2018 of watersheds of three subestuaries surveyed in the Choptank River, Broad Creek, Harris Creek, and Tred Avon River. Black diamond markers indicate the years that subestuaries were sampled. Development data was not available for 2019.

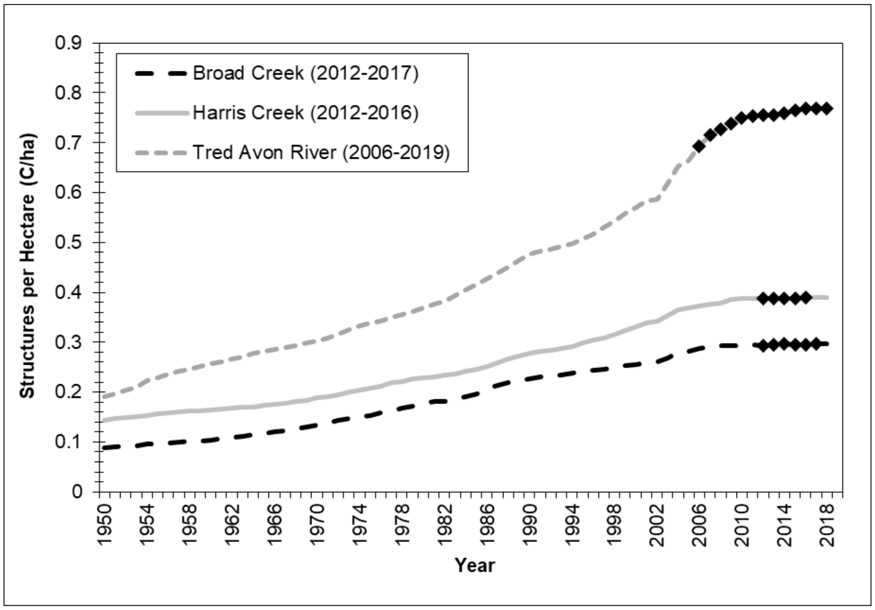


Figure 3-4. Number of finfish species (richness) collected by beach seines in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of watershed development (C / ha = structures per hectare). Points were omitted if beach seine effort (number of samples) < 15 samples.

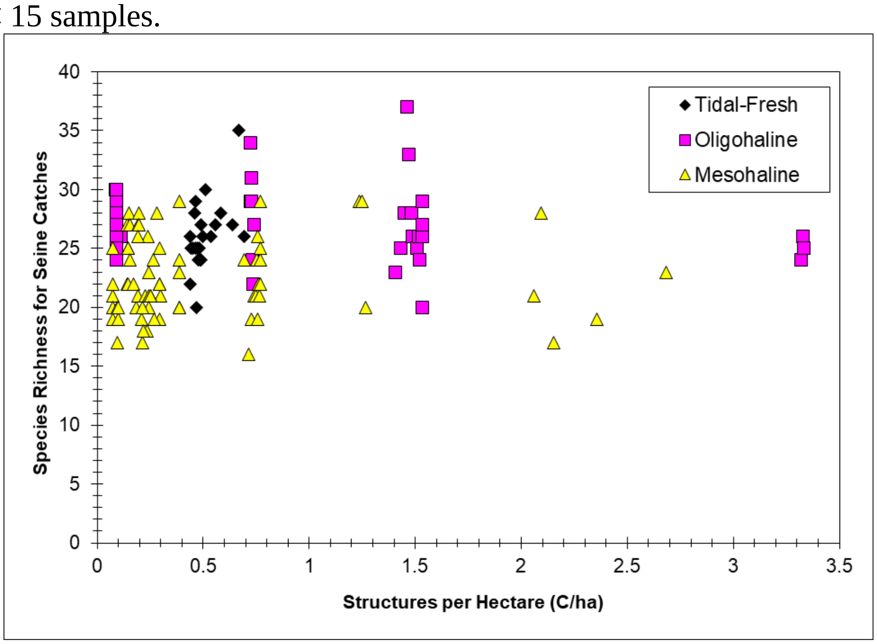


Figure 3-5. Number of finfish species (richness) collected by 4.9 m bottom trawl in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of development (C / ha = structures per hectare). Points were omitted if number of samples was less than 15.

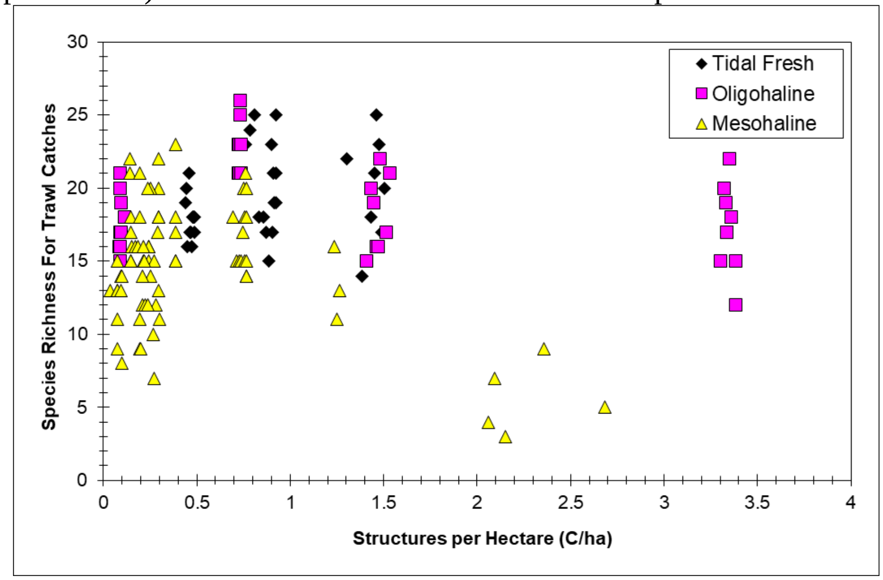


Figure 3-6. Mean subestuary bottom dissolved oxygen during summer sampling, 2003-2019, plotted against level of development (C / ha or structures per hectare).

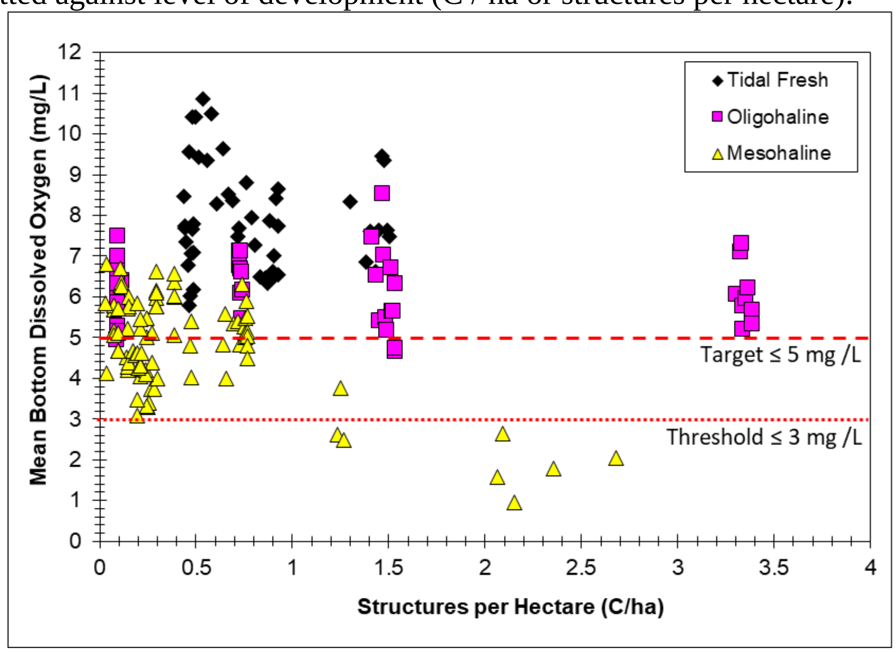


Figure 3-7. Mean subestuary surface dissolved oxygen during summer (July-October) sampling, 2003-2019, plotted against level of development (C / ha or structures per hectare).

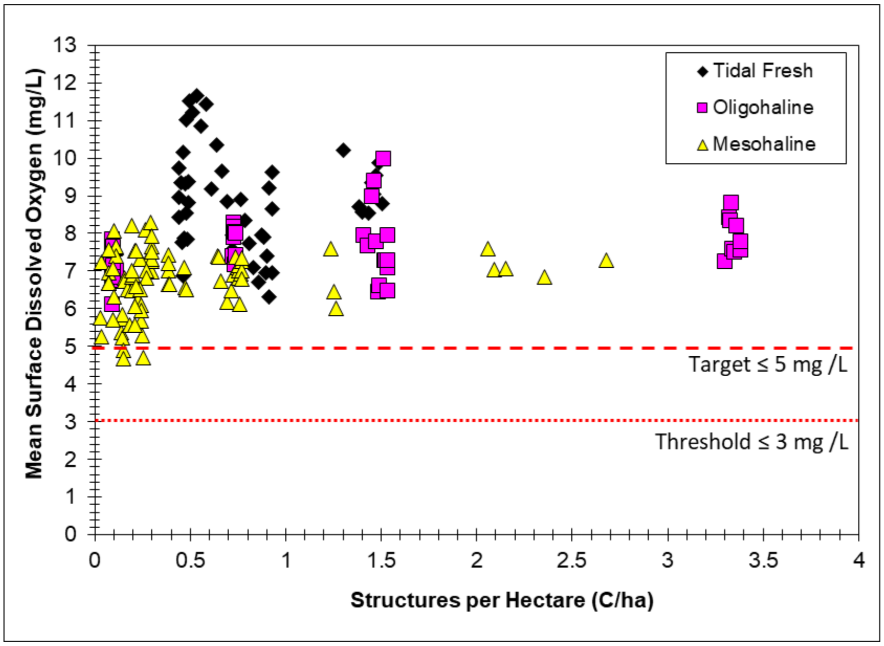


Figure 3-8. Maryland Department of Planning (DOP) estimates agricultural land coverage (% watershed land area) by region (western or eastern shore) versus median bottom dissolved oxygen (DO) in mesohaline subestuaries (2003-2019). Quadratic model predicts median bottom DO and agricultural coverage (%) using data from both regions.

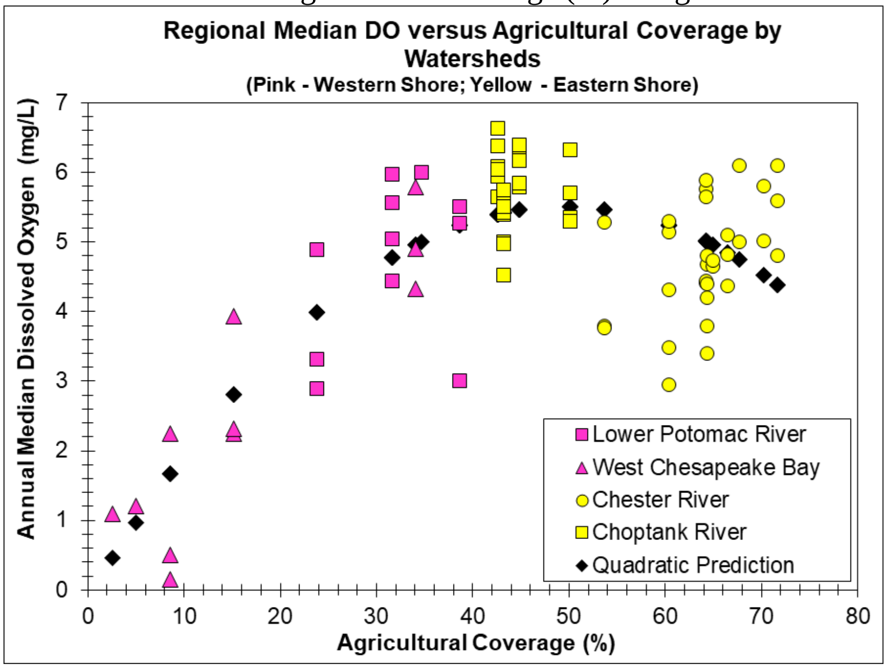


Figure 3-9. Annual 4.9m trawl geometric mean (GM) catches plotted against C / ha Subestuaries sampled during 2003-2019 and separated by salinity class.

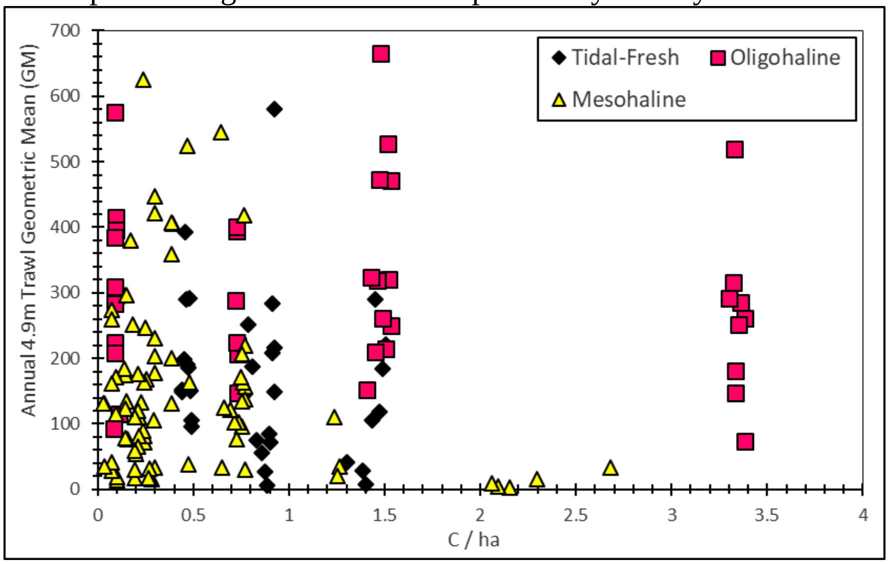


Figure 3-10. Bottom dissolved oxygen (DO; mg / L) readings (2006-2019) versus intensity of development (C / ha = structures per hectare) in Choptank subestuaries, Broad Creek, Harris Creek, and Tred Avon River. Target (5 mg / L) and threshold (3 mg / L) boundaries are indicated by red dashed lines. See legend for years subestuaries were sampled.

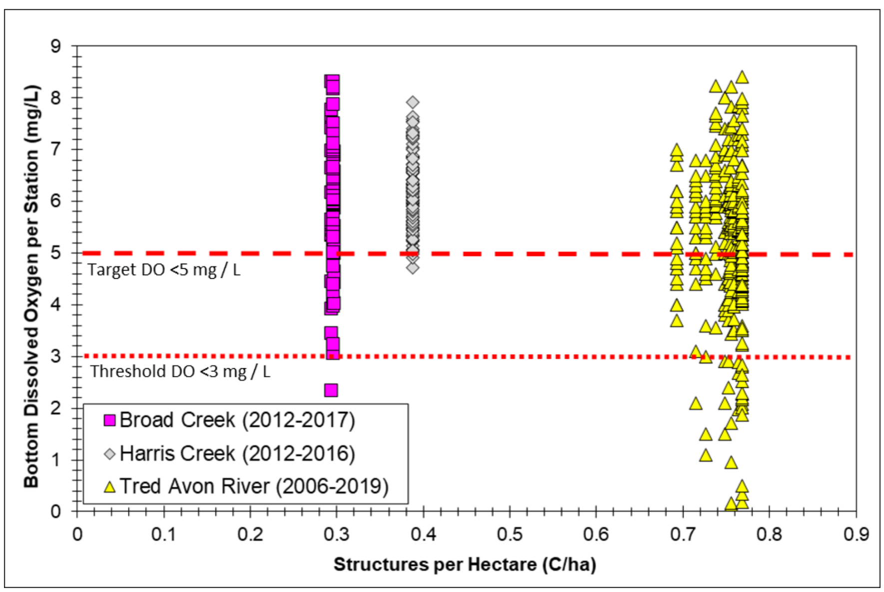


Figure 3-11. Median bottom dissolved oxygen (DO; red squares; mg / L) year's sampled for Broad Creek, Harris Creek, and Tred Avon River. Solid black bars indicate range of all bottom DO measurements for that year.

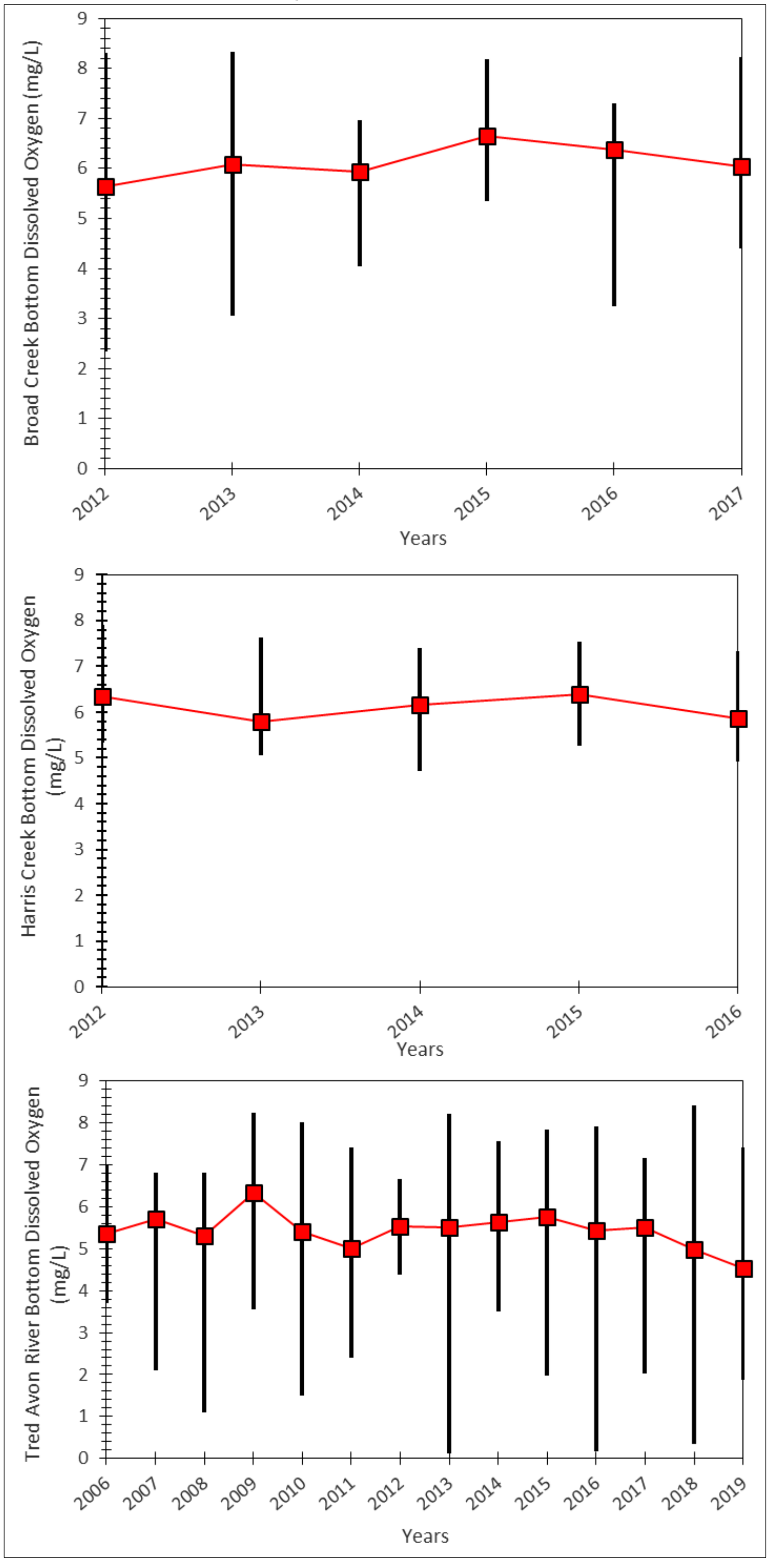


Figure 3-12. Map indicating the locations of seine and bottom trawl sites for the lower Choptank River subestuaries, Broad Creek (2012-2017), Harris Creek (2012-2016), and Tred Avon River (2006-2019).

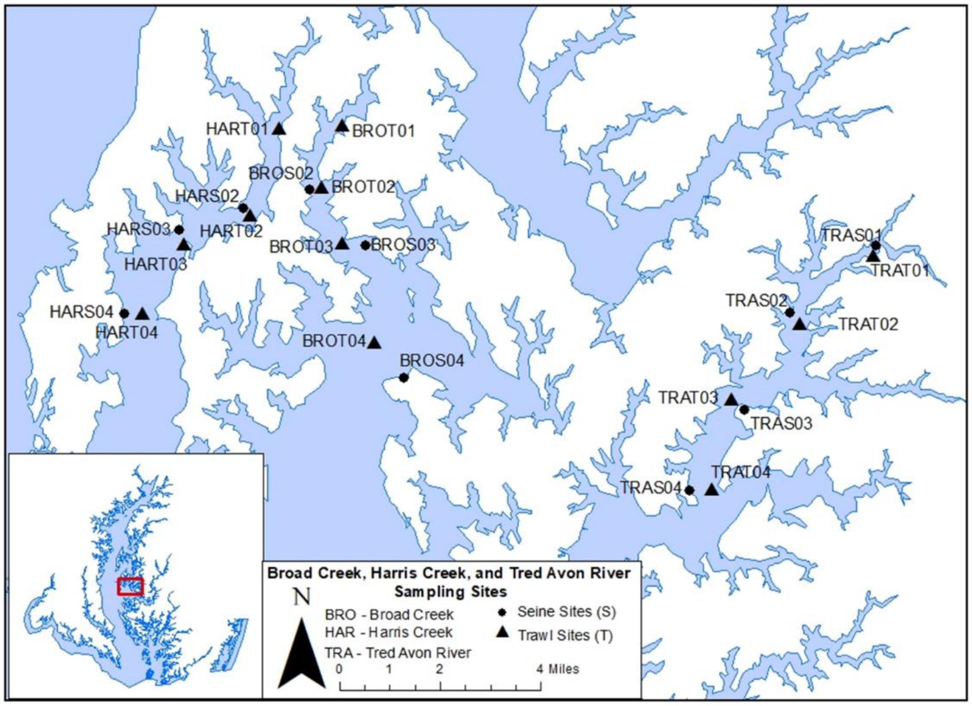


Figure 3-13. Mean bottom dissolved oxygen (DO; mg / L) for all years surveyed for Broad Creek, Harris Creek, and Tred Avon River, by sampling station. Dotted line indicates the median of all DO measurement data for the time-series available.

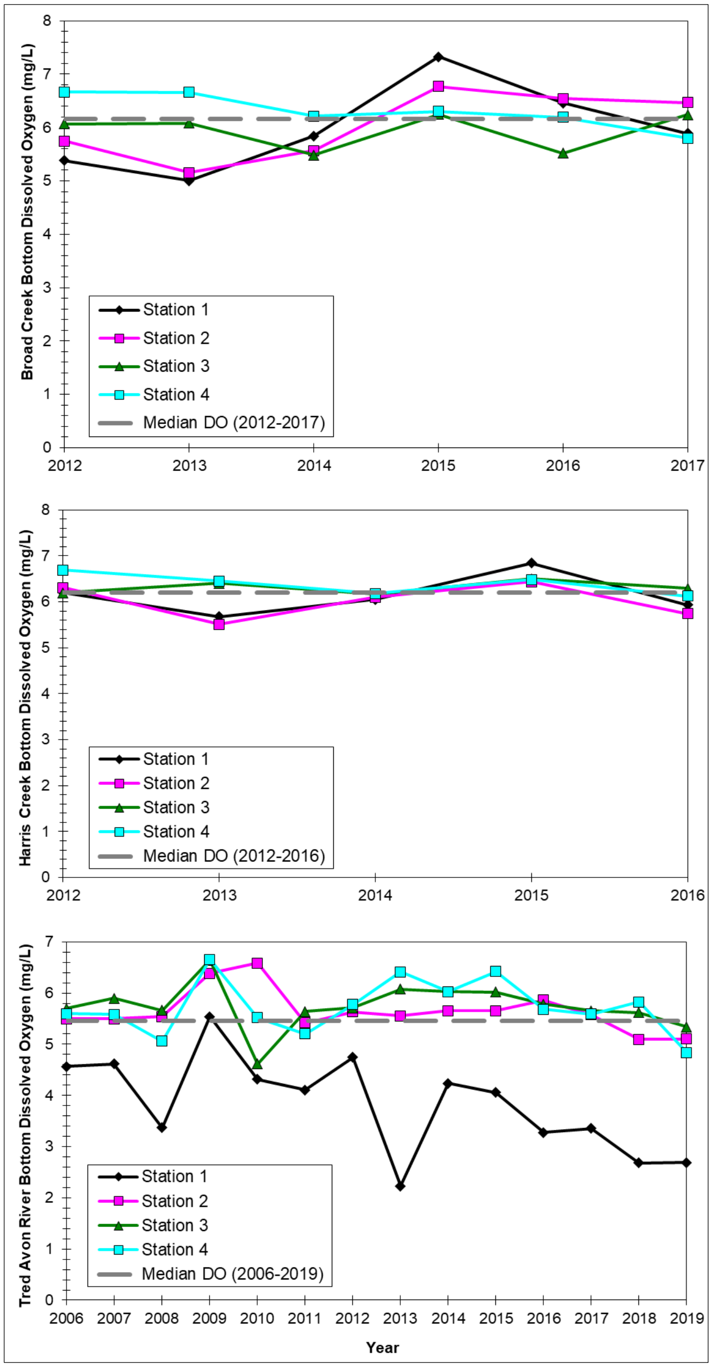


Figure 3-14. Annual 4.9m bottom trawl catch geometric mean (GM) per of all finfish species (red squares) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Black bars indicate the 95% confidence intervals.

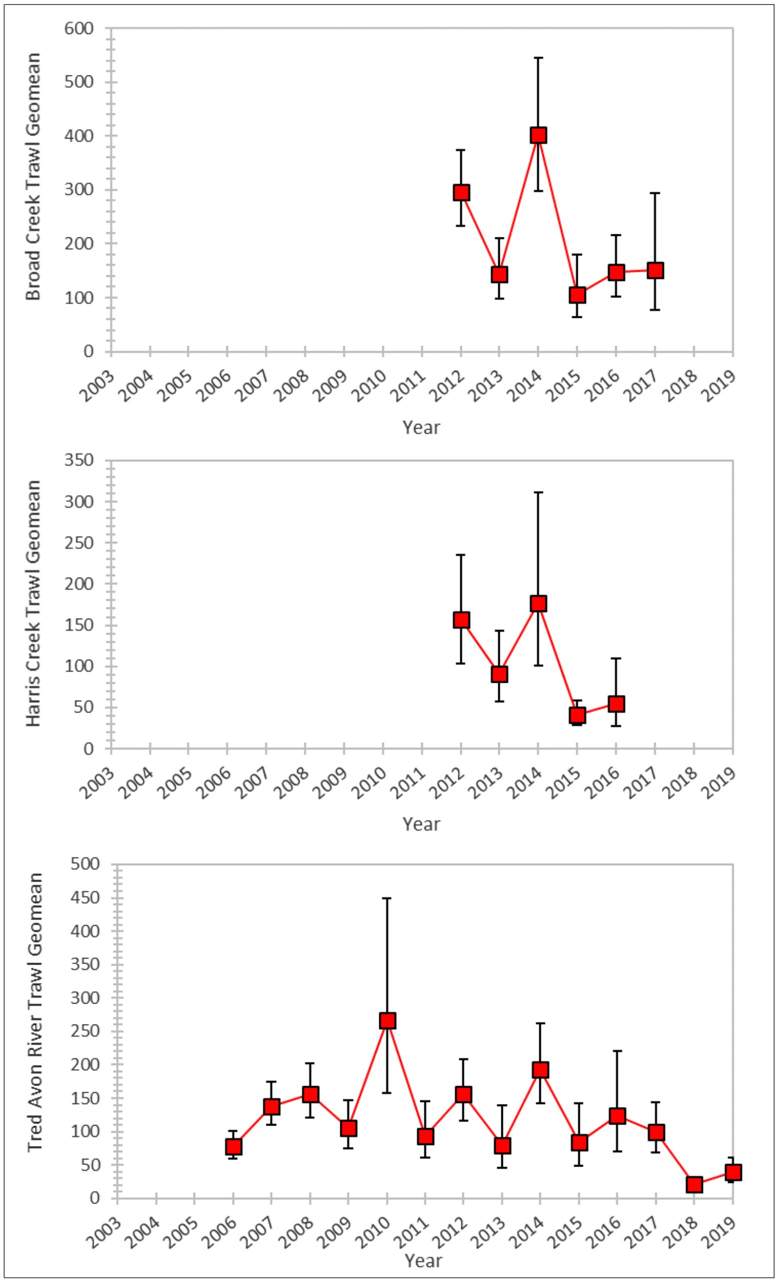


Figure 3-15. Finfish species composition for 4.9 m bottom trawl catch in Tred Avon River for all sampling years combined (2006-2019). Species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

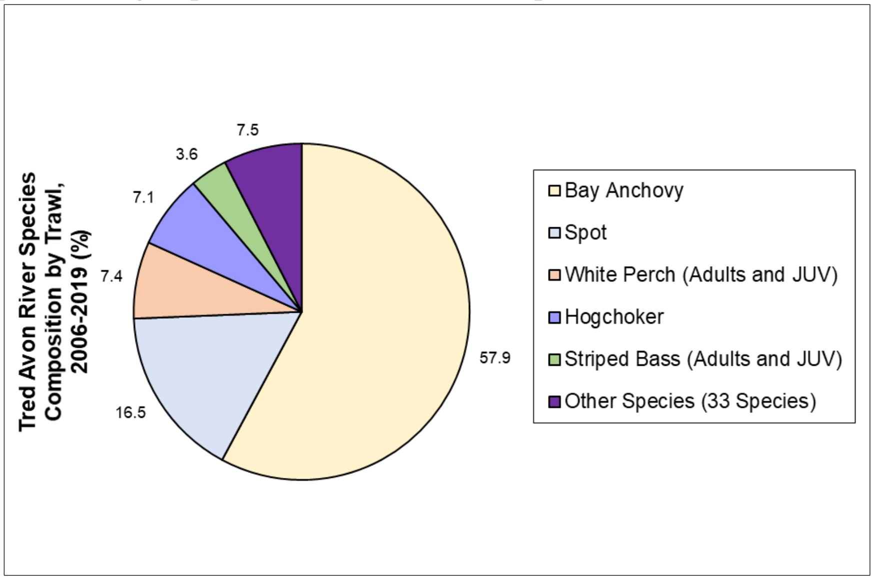


Figure 3-16. Finfish species composition for 4.9 m bottom trawl catch in Tred Avon River for each year sampled. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

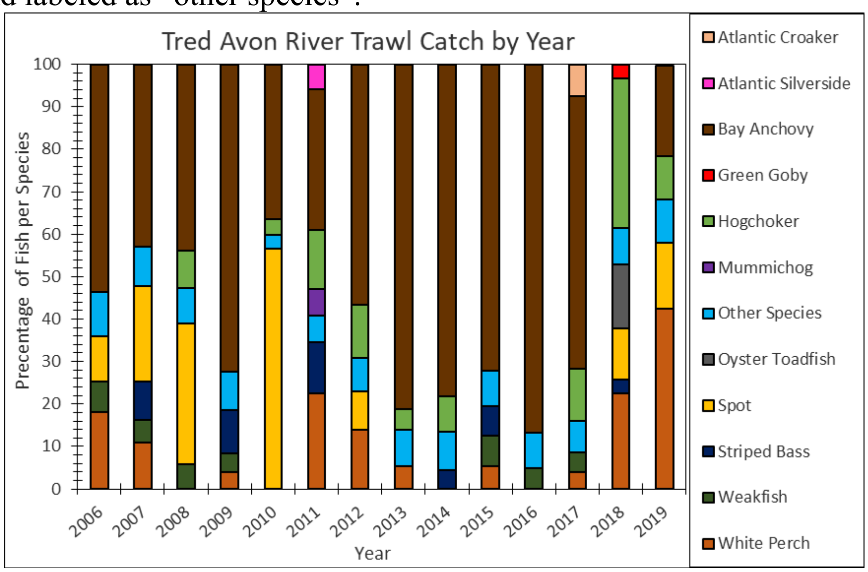


Figure 3-17. Percent similarity index (%) for 4.9 m bottom trawl stations 1-4 in Tred Avon River by year. The greater the similarity value, the more finfish species there are in common throughout all four bottom trawl stations.

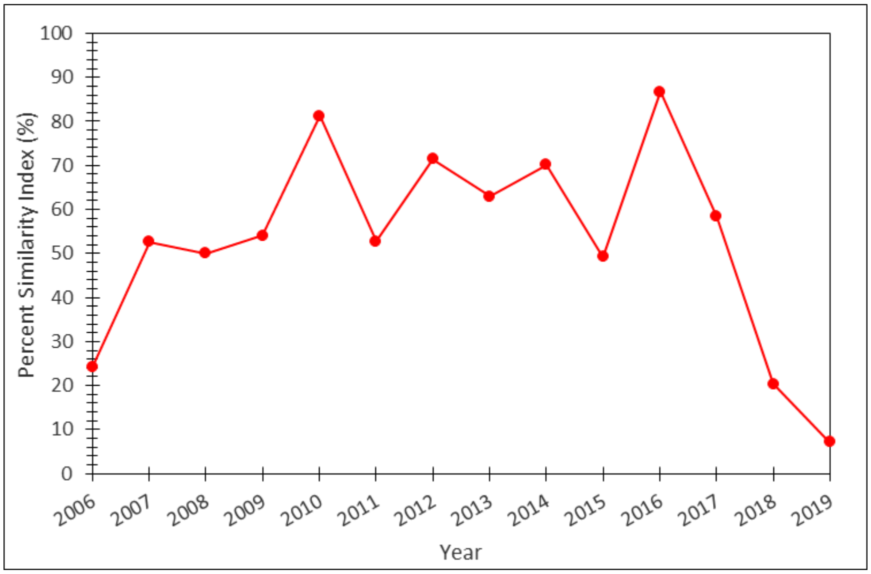


Figure 3-18. Finfish species composition for 4.9 m bottom trawl catch in all mesohaline subestuaries sampled during 2003-2019, by year. Finfish species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

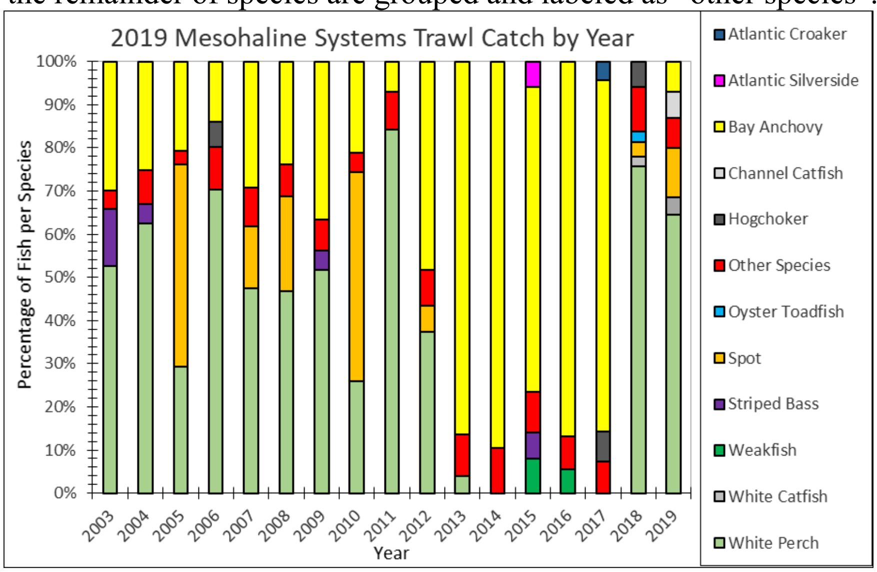


Figure 3-19. Geometric mean (GM) per 4.9 m bottom trawl catch for adult White Perch in Broad Creek (blue triangles), Harris Creek (red squares), and Tred Avon River (black circles), by sampling year.

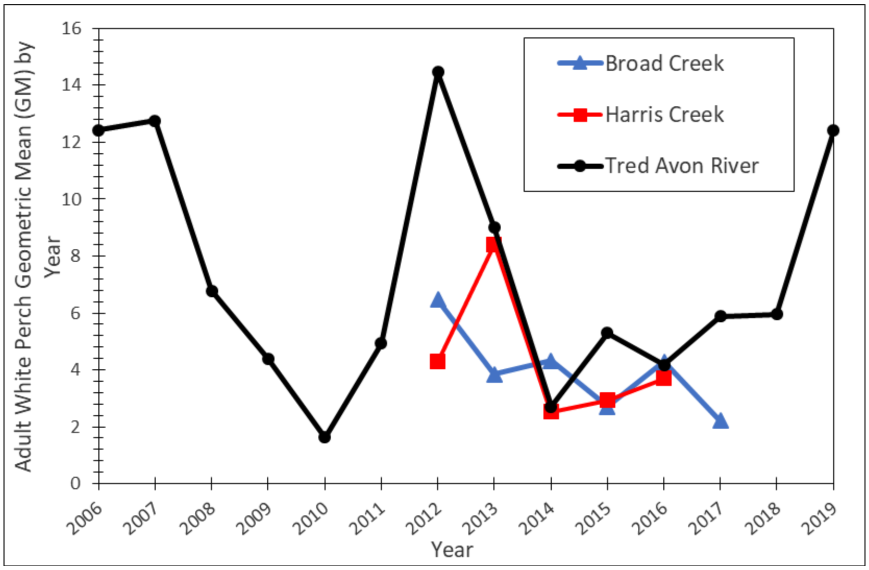


Figure 3-20. Annual beach seine catch geometric mean (GM) per of all finfish species (red squares) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Black bars indicate the 95% confidence intervals.

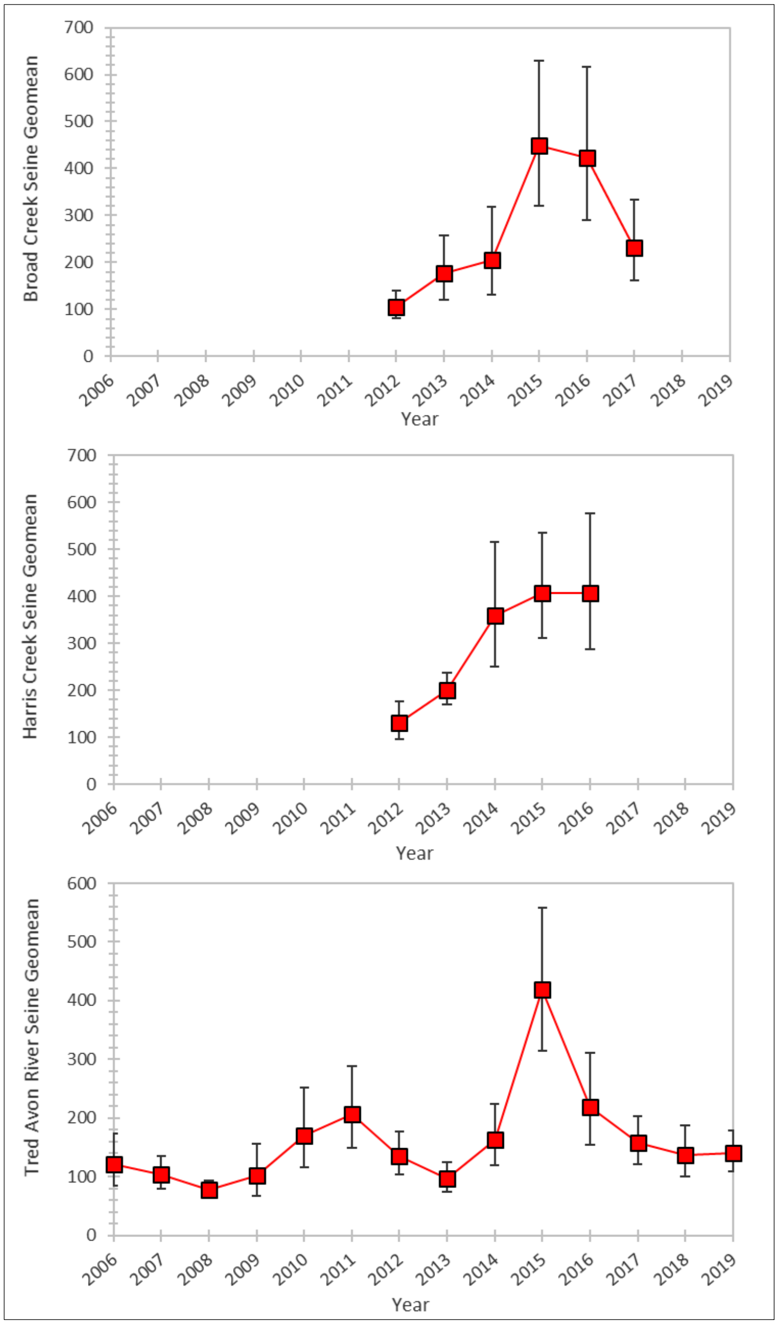


Figure 3-21. Finfish species composition for beach seine catch in Tred Avon River for all years combined (2006-2019). Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

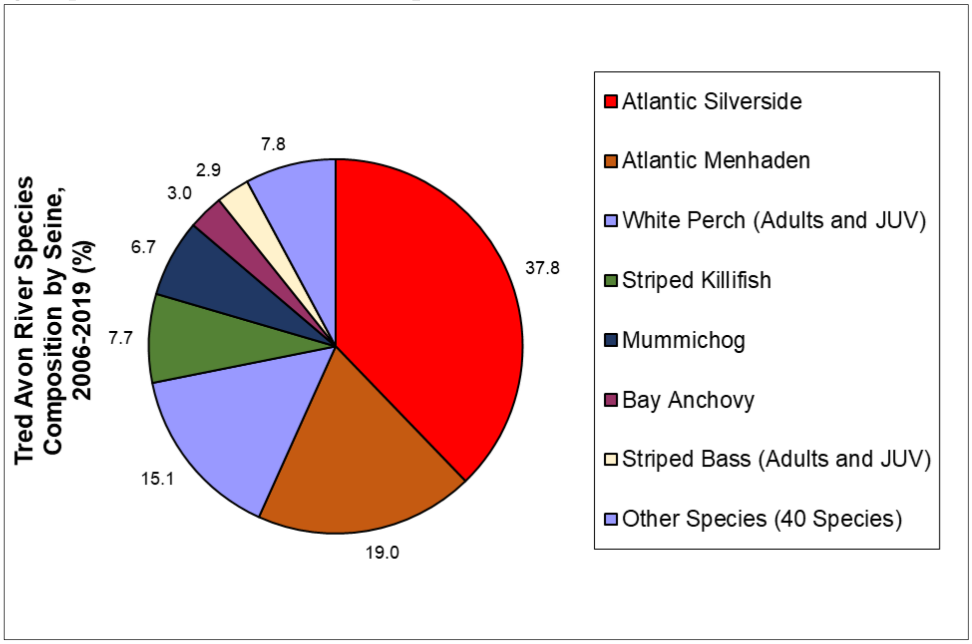


Figure 3-22. Modified proportional stock density (PSD) of White Perch in Choptank River subestuaries, Broad Creek, Harris Creek, and Tred Avon River, is the proportion of 4.9m trawl samples with quality length or greater White Perch.

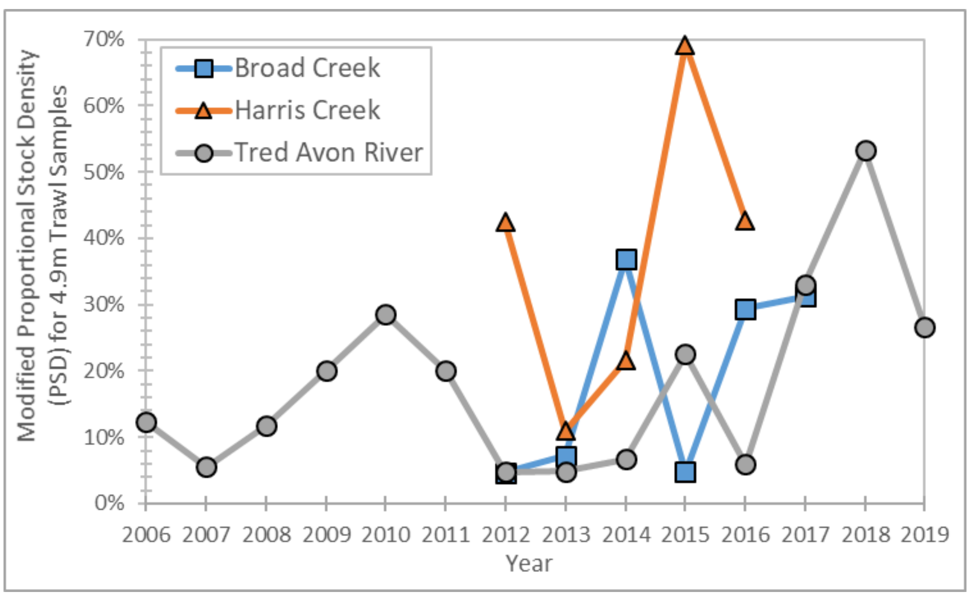


Figure 3-23. Median Secchi depth (m) for Broad Creek, Harris Creek, and Tred Avon River (red squares), by year. Solid black bars indicate the range of Secchi depth (m) measurements by year.

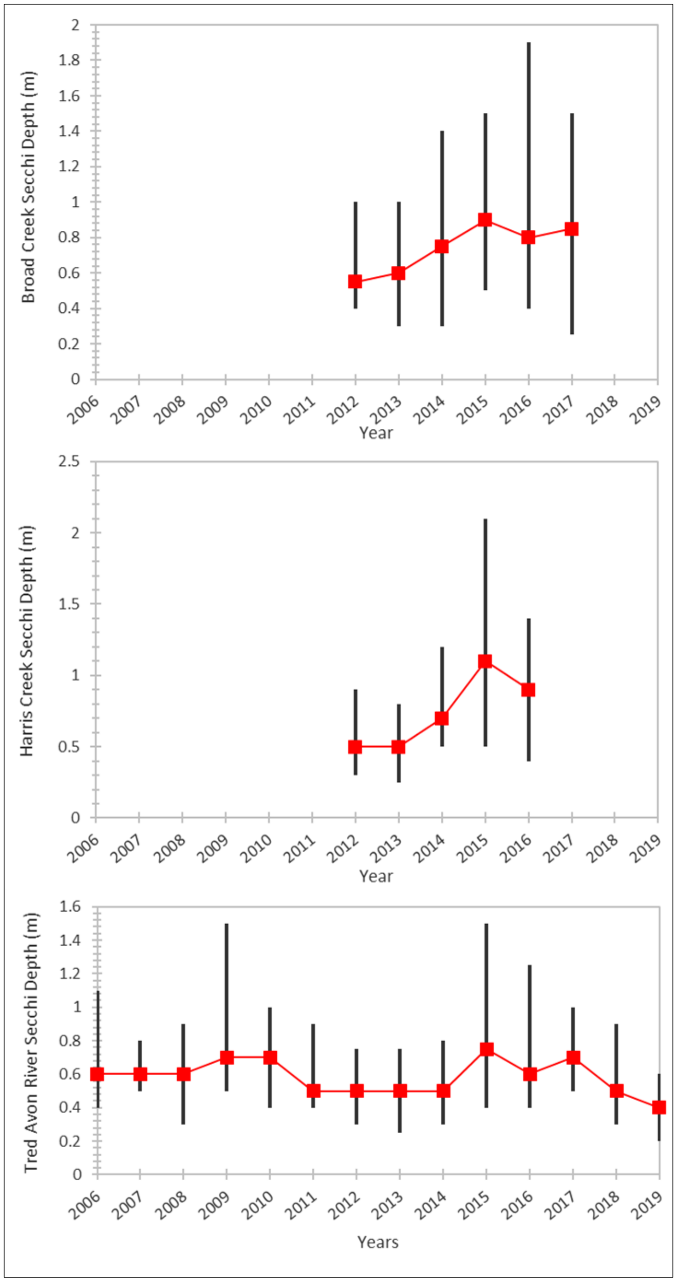


Figure 3-24. Coverage of SAV (percent of coverage in water area) for the mouth of the Choptank (containing Broad Creek, Harris Creek, and Tred Avon) during 1989-2018. Median of only fully mapped years (1989-2017) for the time-series is indicated by the dashed line. Data for 2019 was not available at the time of this report.

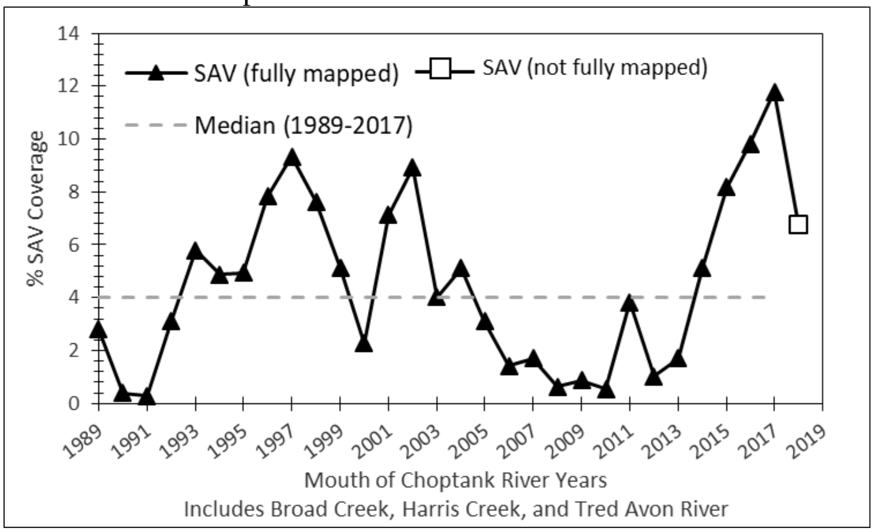


Figure 3-25. Median bottom pH (red squares) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Solid black bars indicate the range of pH measurements by year.

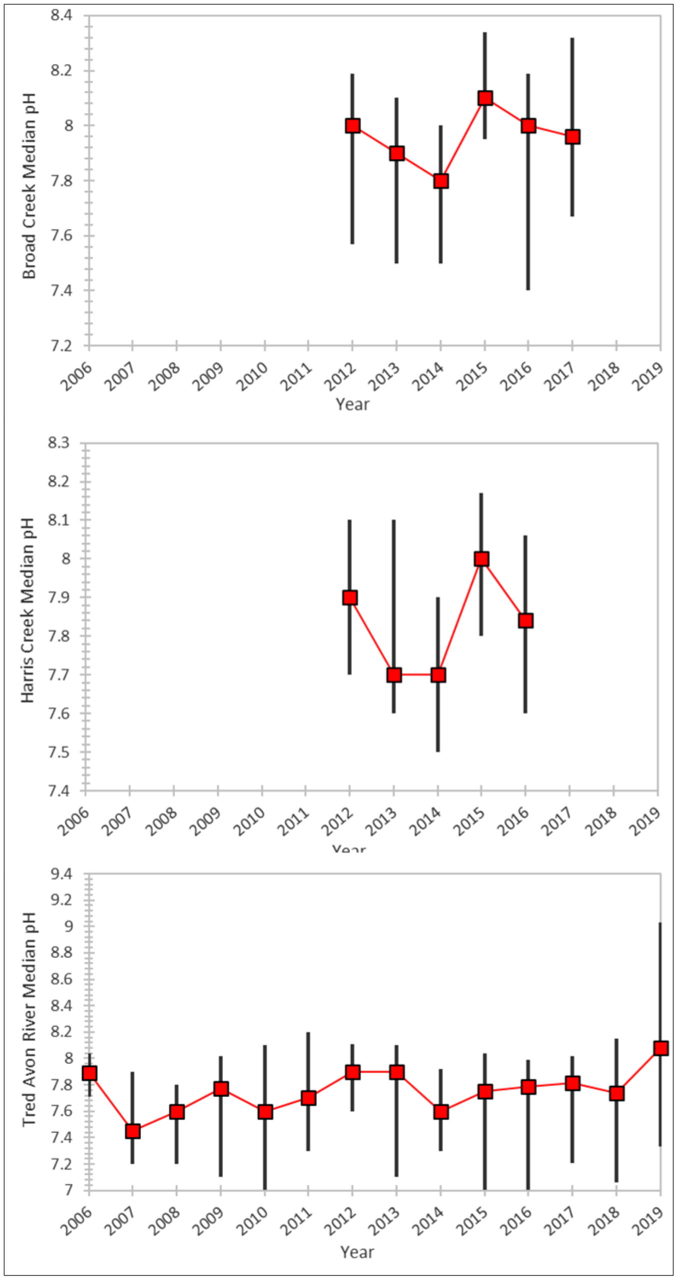


Figure 3-26. Median bottom salinity (red squares; ppt = ‰) for Broad Creek, Harris Creek, and Tred Avon River, by sampling year. Solid black bars indicate the range of pH measurements by year.

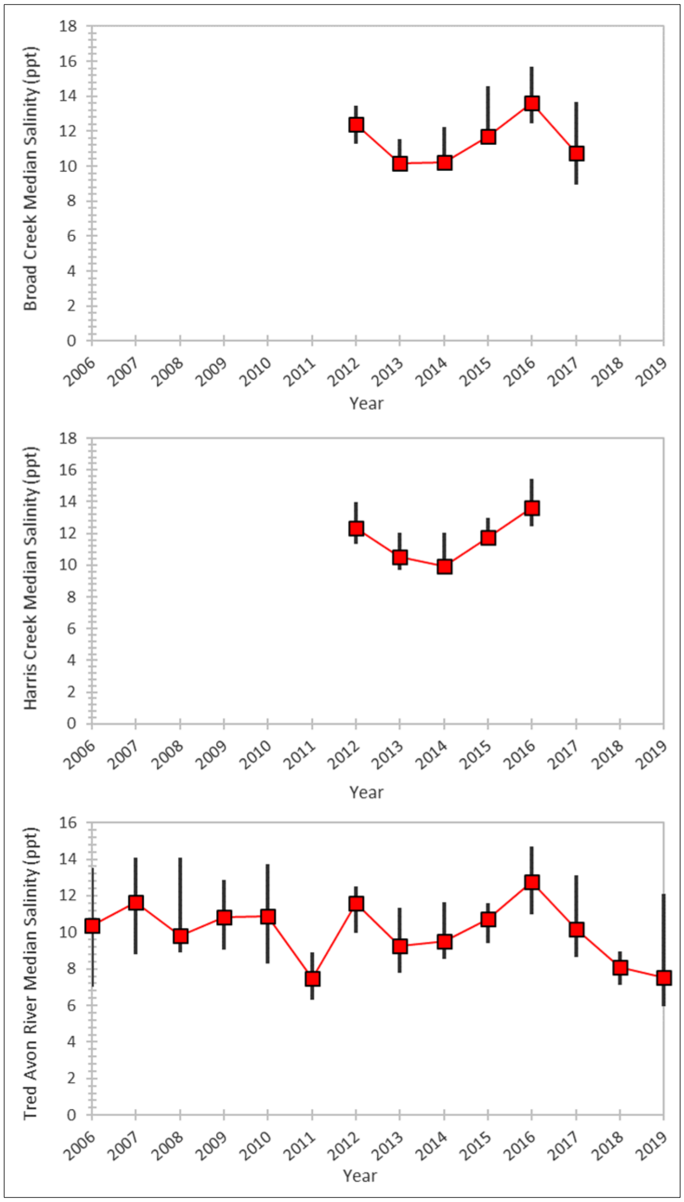


Figure 3-27. Trends in levels of development (structures per hectare = C / ha) during 1950-2018 in watersheds of two subestuaries surveyed, the Chester River and its tributaries, Corsica River and Langford Creek, and the Wye River. Black diamond markers indicate the years that subestuaries were sampled. Development data was not available for 2019.

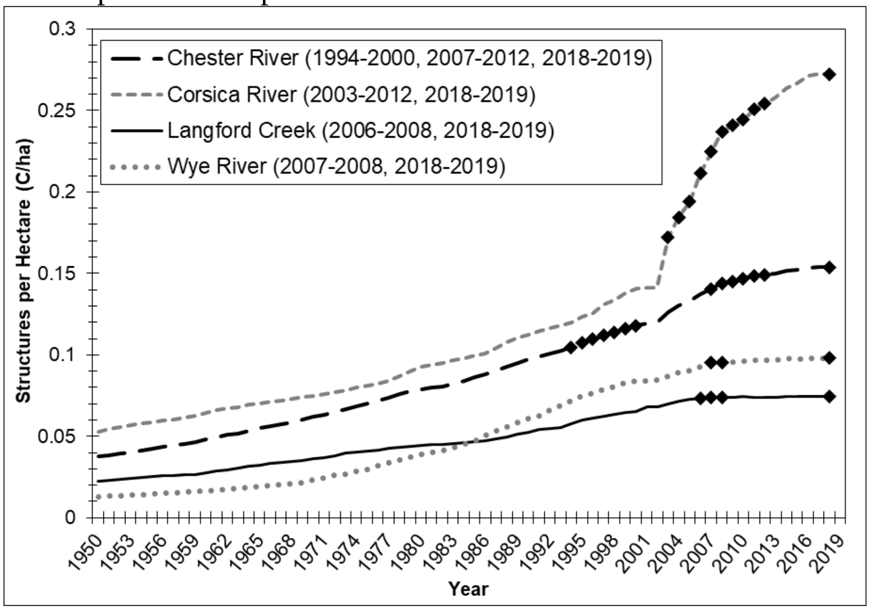


Figure 3-28. Bottom dissolved oxygen (DO; mg / L; 1995-2019) versus intensity of development (C / ha = structures per hectare) in Chester River, Corsica River, Langford Creek, and the Wye River. Target (= 5 mg / L) and threshold (= 3 mg / L) boundaries are indicated (red dashed lines).

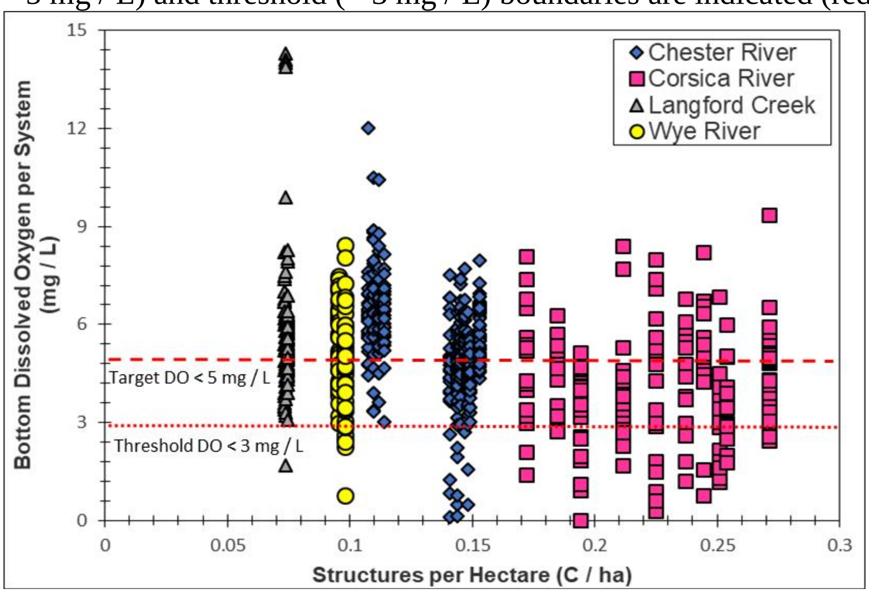


Figure 3-29. Median bottom dissolved oxygen (DO; red squares; mg / L) for Chester River, Corsica River, Langford Creek, and Wye River surveys. Solid black bars indicate range of all bottom DO measurements for that year.

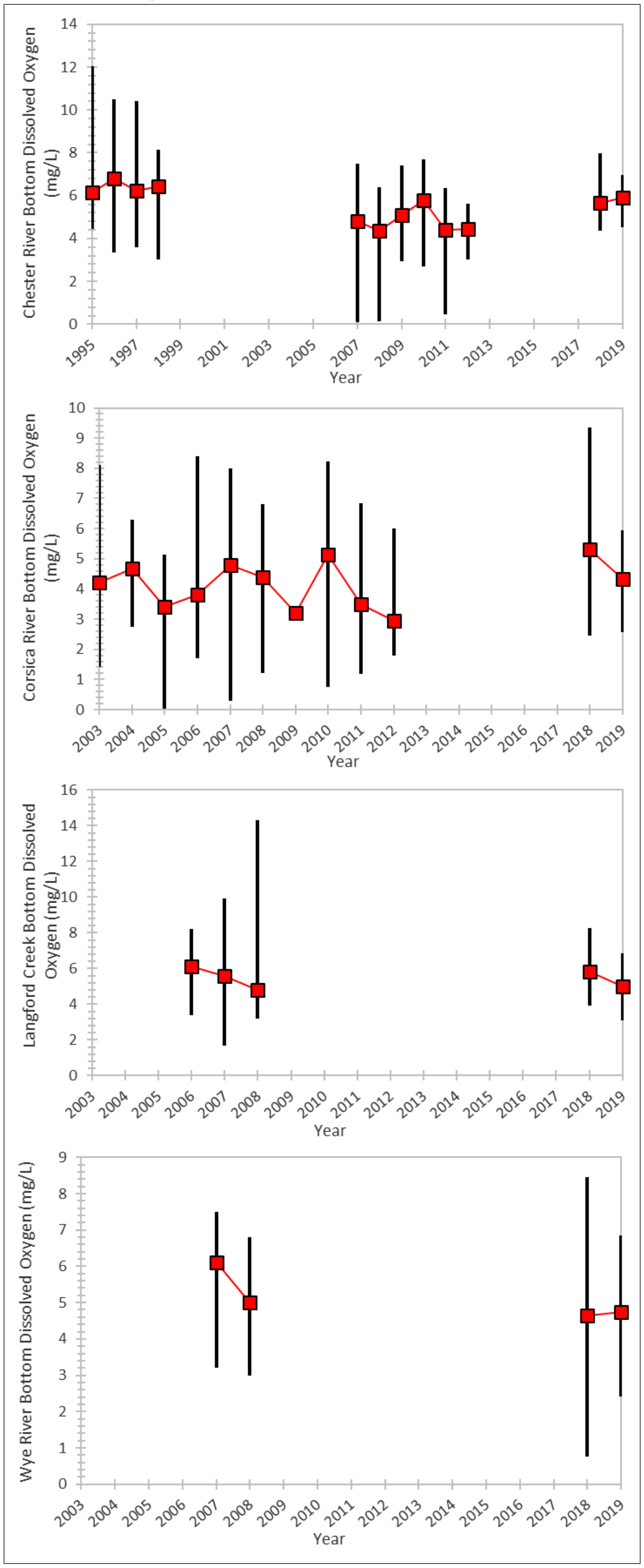


Figure 3-30. Mean bottom dissolved oxygen (DO; mg / L) for all years surveyed for Chester River, Corsica River, Langford Creek, and Wye River, by sampling station. Dotted line indicates the median of all DO measurement data for the time-series available.

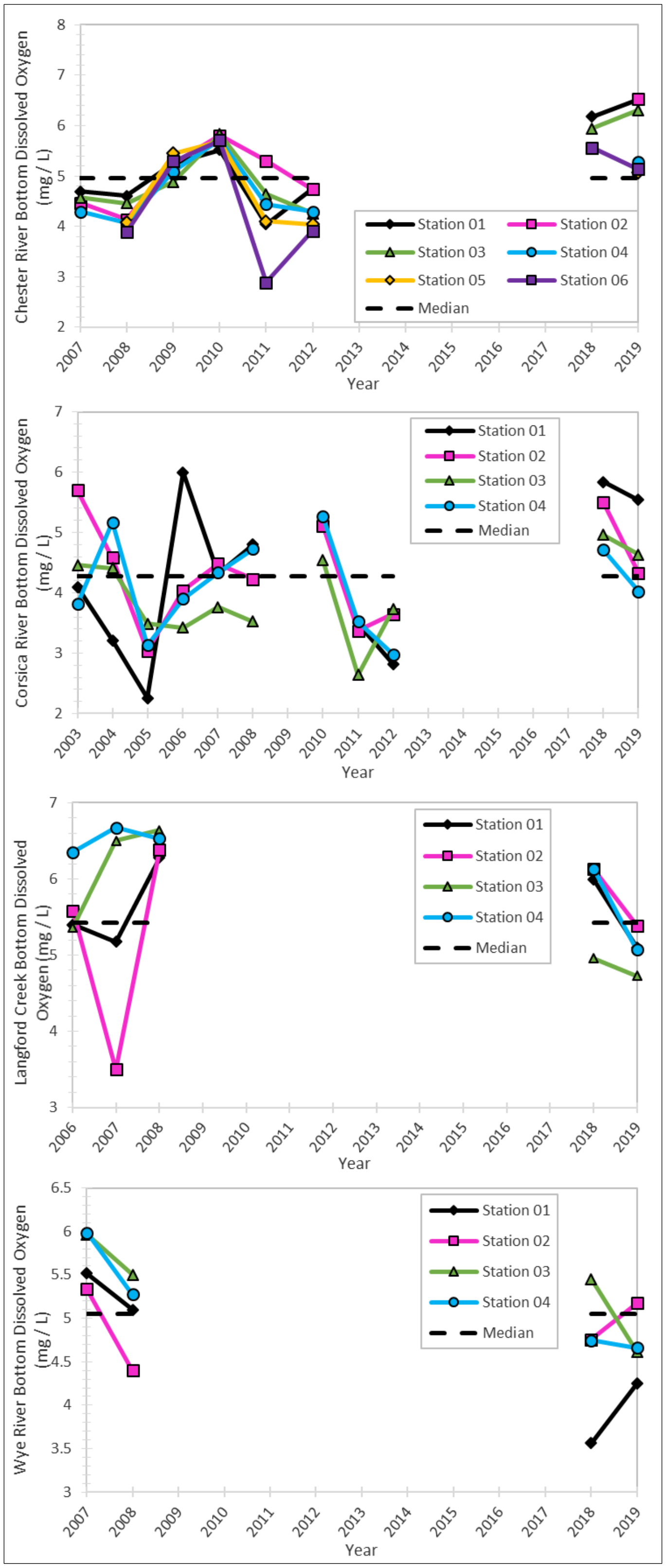


Figure 3-31. Annual 4.9m bottom trawl catch geometric mean (GM) per of all finfish species (red squares) for Chester River, Corsica River, Langford Creek, and Wye River, by sampling year. Black bars indicate the 95% confidence intervals. Chester River includes annual 3.1 m bottom trawl catch GM data for 1994-2000 (grey squares).

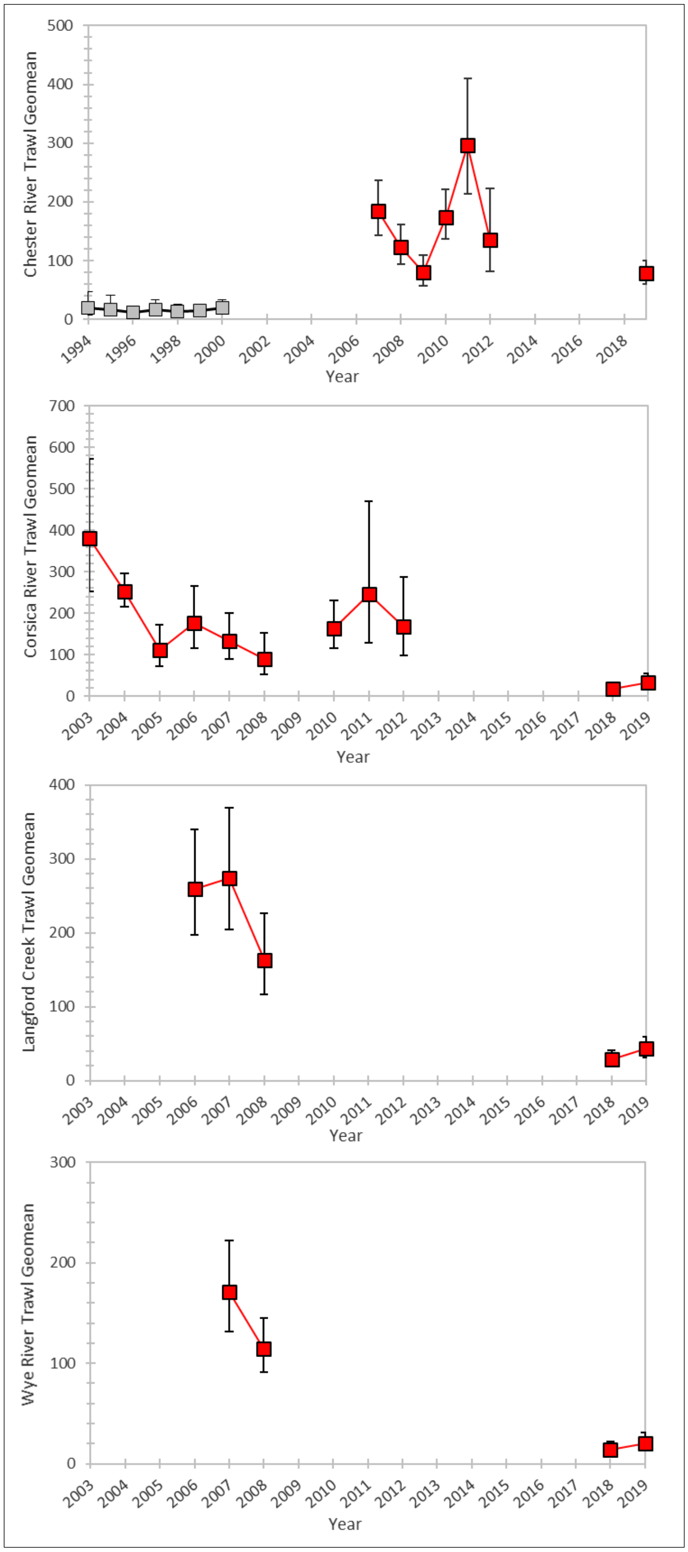


Figure 3-32. Finfish species composition for 4.9 m bottom trawl catch in Chester River (2007-2012, 2019), Corsica River (2003-2012, 2018), Langford Creek (2006-2008, 2018-2019), and Wye River (2007-2008, 2018-2019) for all sampling years combined. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

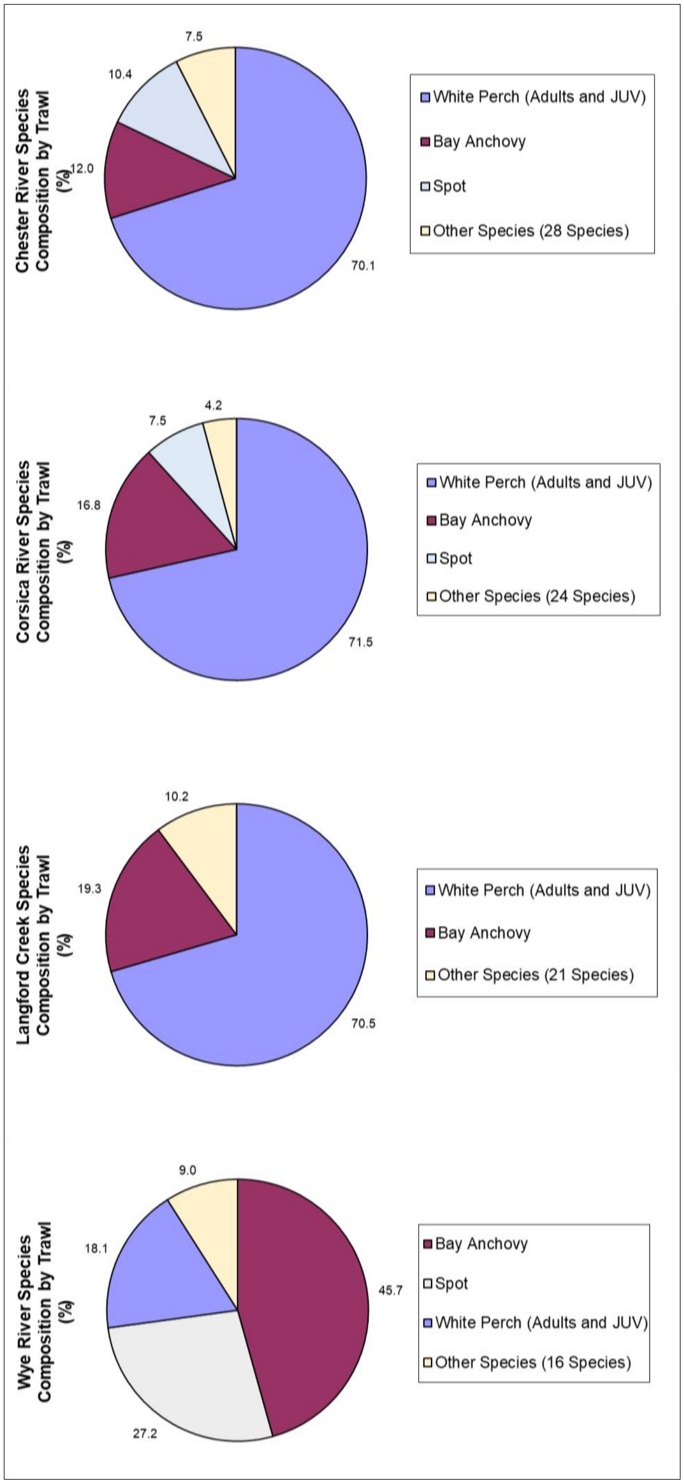


Figure 3-33. Finfish species composition for 4.9 m bottom trawl catch in Chester River (2007-2012, 2019), Corsica River, Langford Creek, and Wye River, by year. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”. Chester River includes annual 3.1 m bottom trawl catch data (1994-2000).

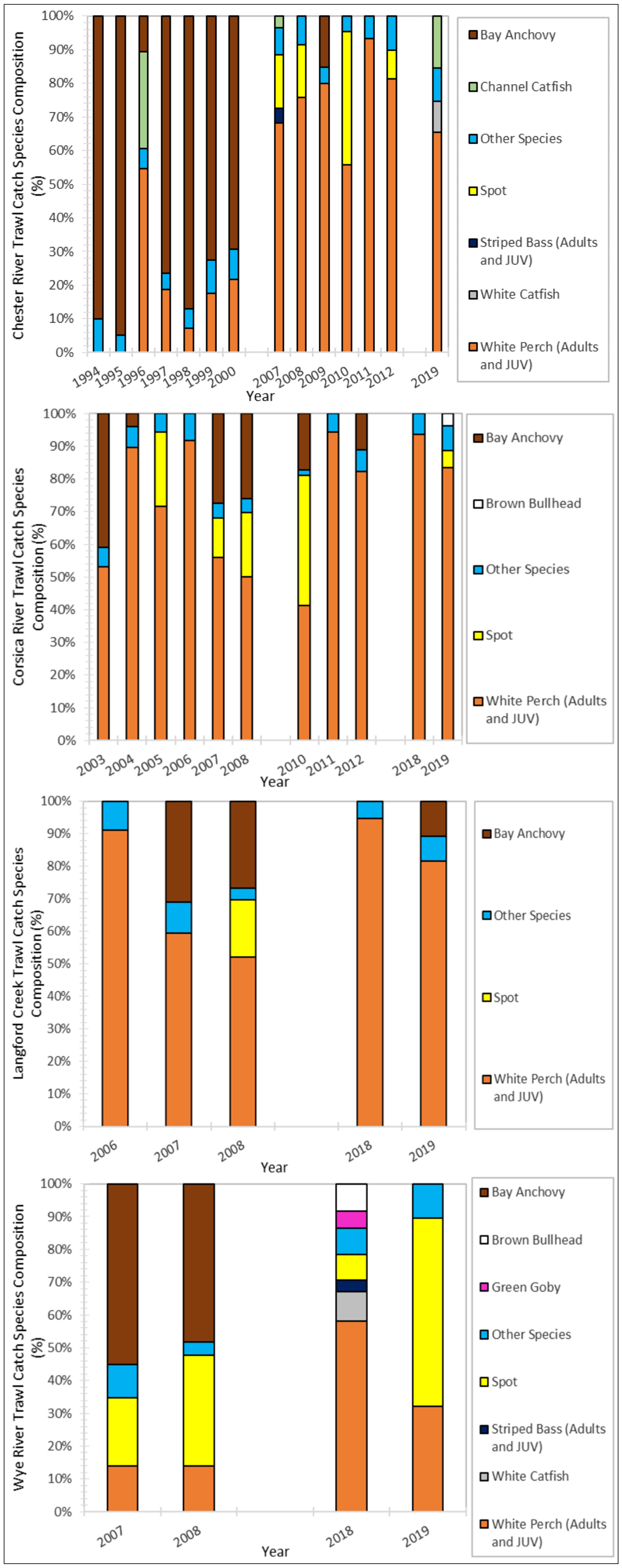


Figure 3-34. Annual beach seine catch geometric mean (GM) per of all finfish species (red squares) for Chester River, Corsica River, Langford Creek, and Wye River, by year. Black bars indicate the 95% confidence intervals.

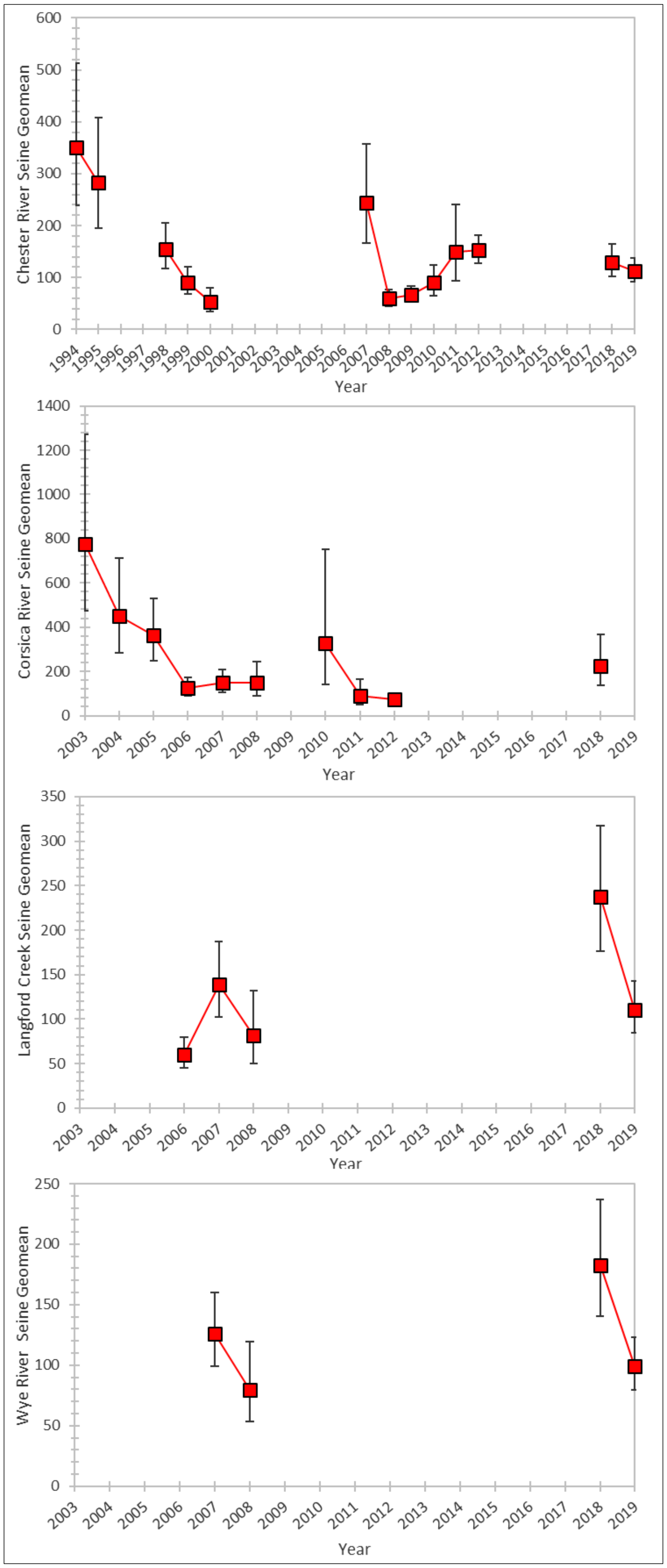


Figure 3-35. Finfish species composition for beach seine catch in Chester River (2007-2012, 2018-2019), Corsica River (2003-2012, 2018), Langford Creek (2006-2008, 2018-2019), and Wye River (2006-2007, 2018-2019) for all years combined. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

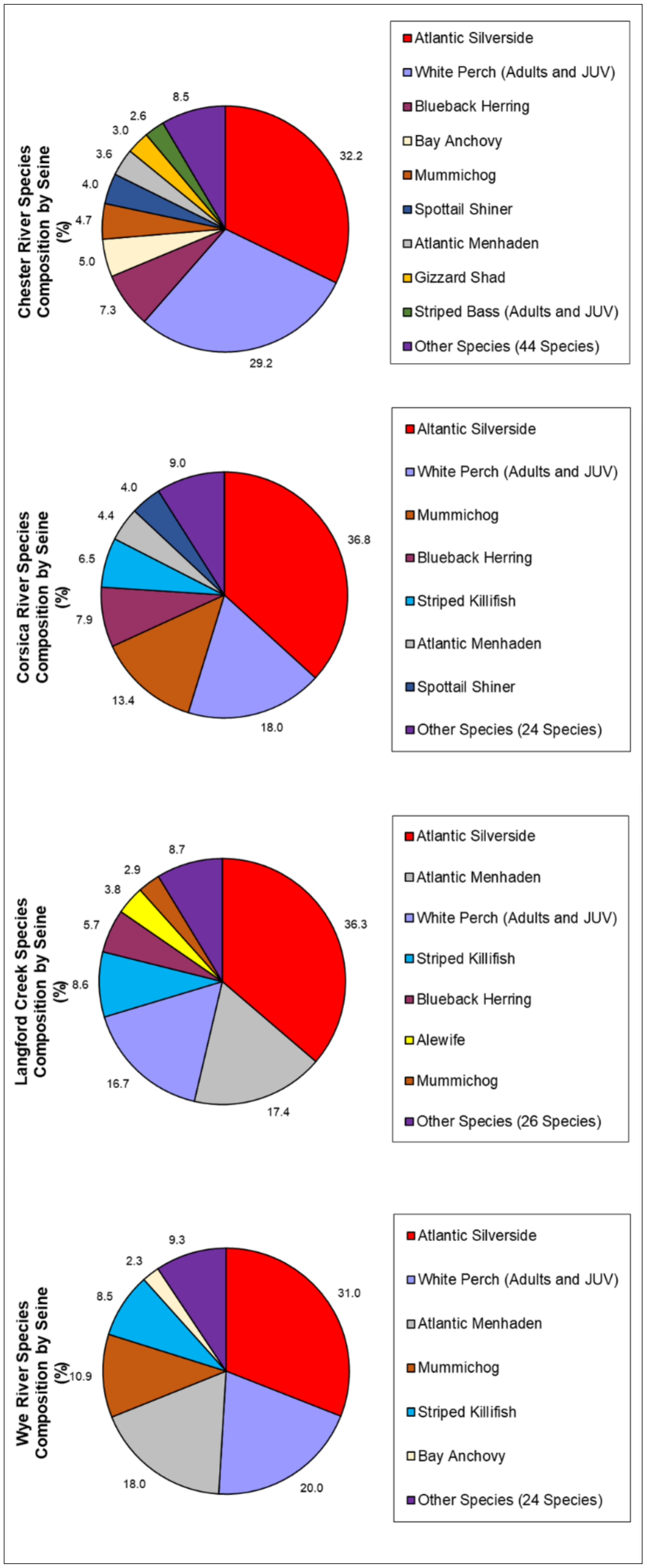


Figure 3-36. Modified proportional stock density (PSD) of White Perch in Queen Anne's County subestuaries, Chester River, Corsica River, Langford Creek, and Wye River, is the proportion of 4.9m trawl samples with quality length or greater White Perch.

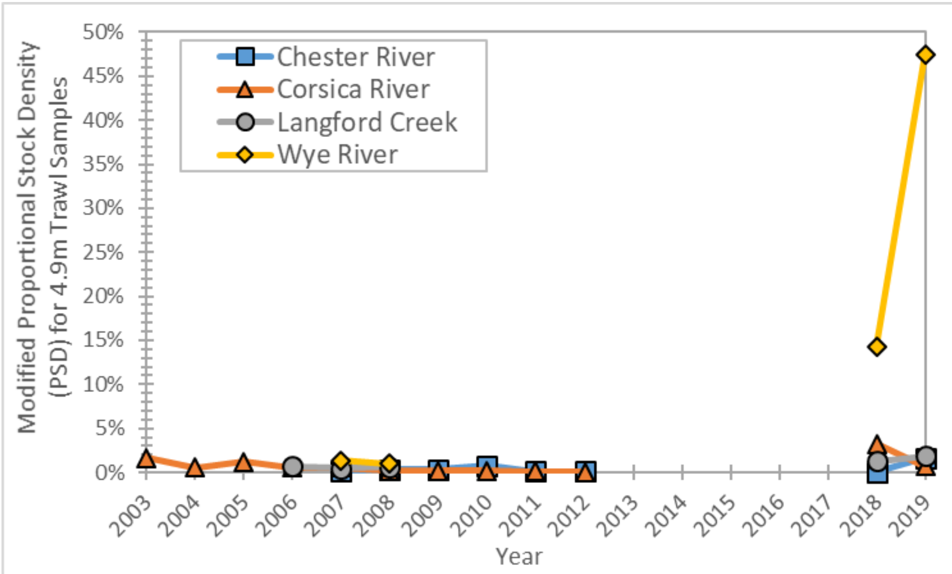


Figure 3-37. Median Secchi depth (m) for Corsica River, Langford Creek, and Wye River (red squares), by year. Solid black bars indicate the range of Secchi depth (m) measurements by year. Secchi depths (m) were not available for Chester River.

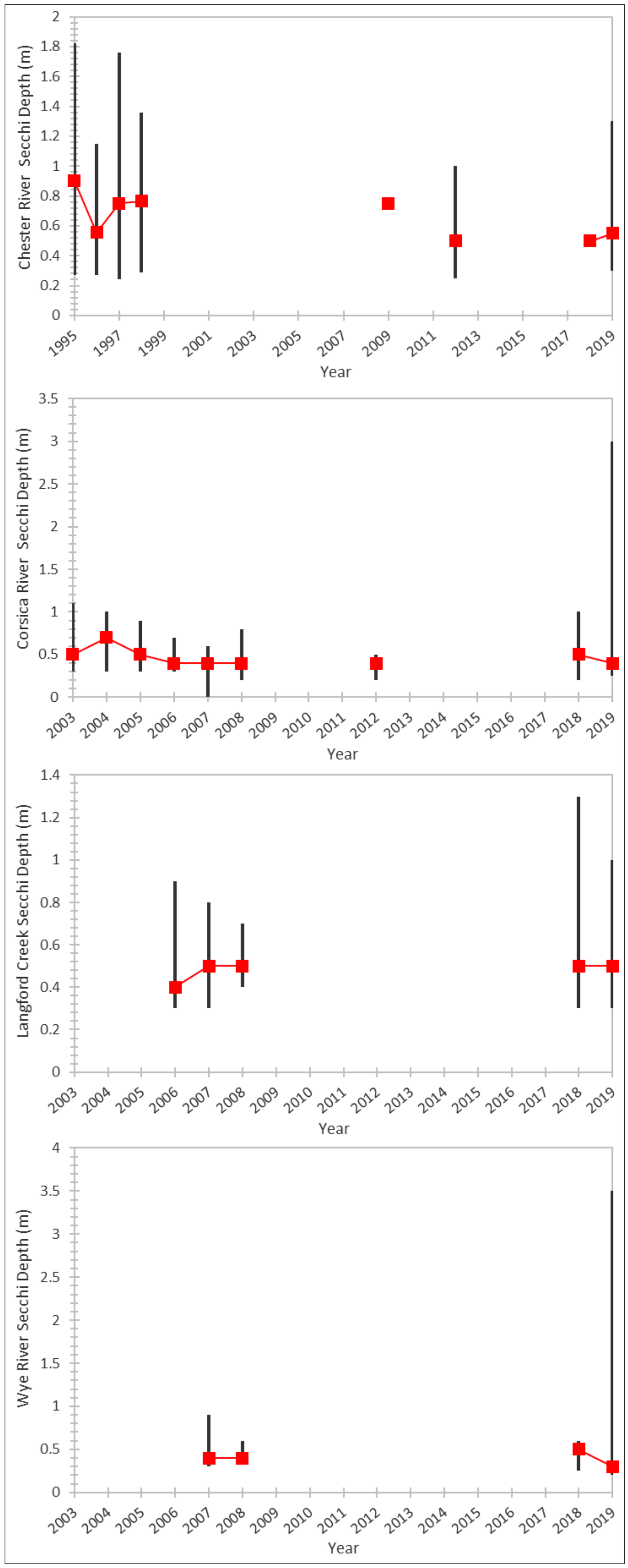


Figure 3-38. Coverage of SAV (percent of water covered) for the Chester River, Corsica River, Langford Creek, and for the Eastern Bay area, including the Wye River, for years, 1989-2018. Several years were excluded due to inadequate mapping. Median of only fully mapped years for the time-series is indicated by the dashed line. Data for 2019 was not available at the time of this report.

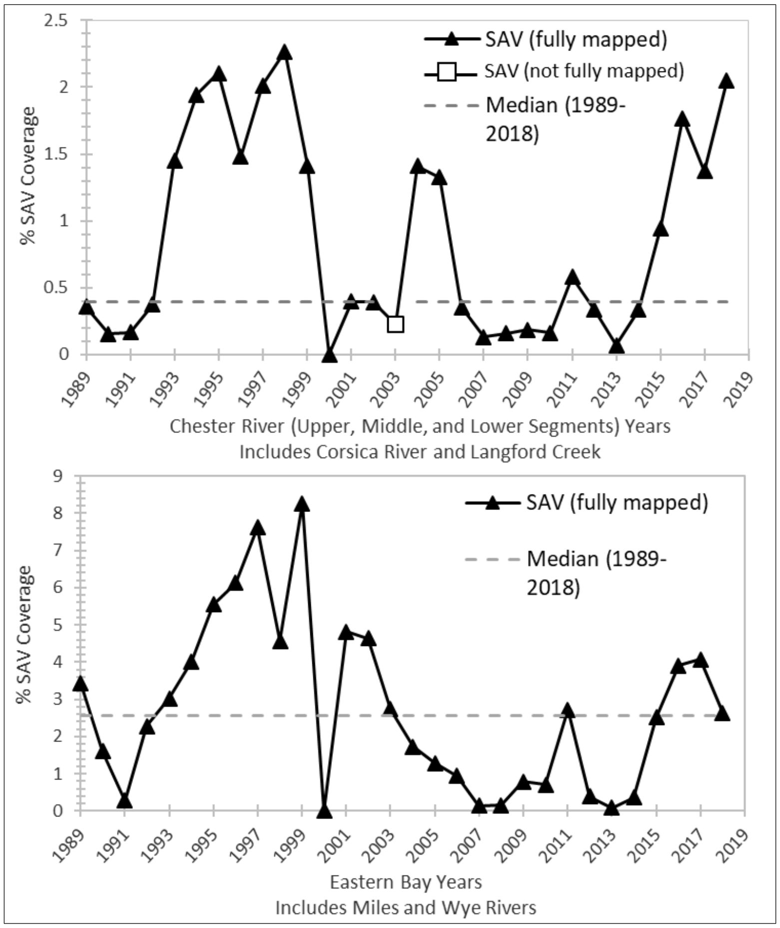


Figure 3-39. Median bottom pH (red squares) for Chester River and its tributaries, Corsica River and Langford Creek, and the Wye River, by sampling year. Solid black bars indicate the range of pH measurements by year.

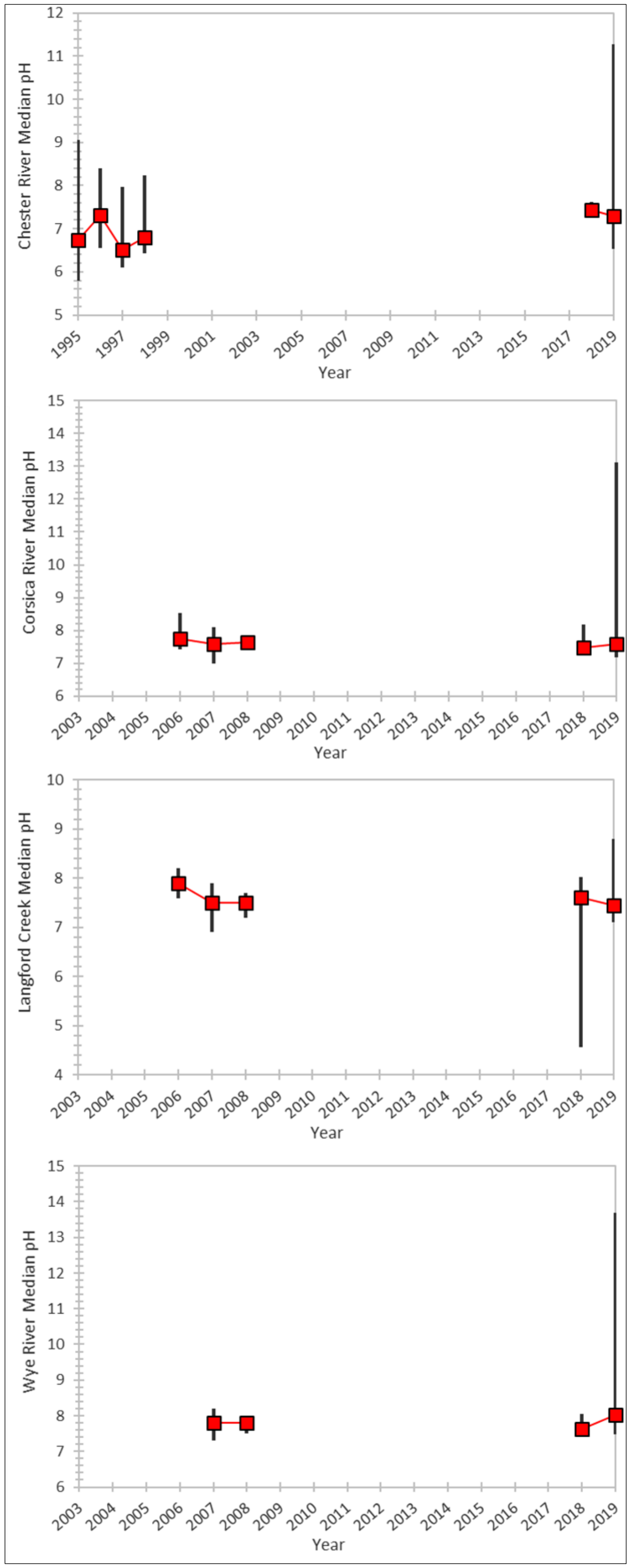


Figure 3-40. Median bottom salinity (red squares; ppt = ‰) for Chester River, Corsica River, Langford Creek, and Wye River, by sampling year. Solid black bars indicate the range of salinity measurements by year.

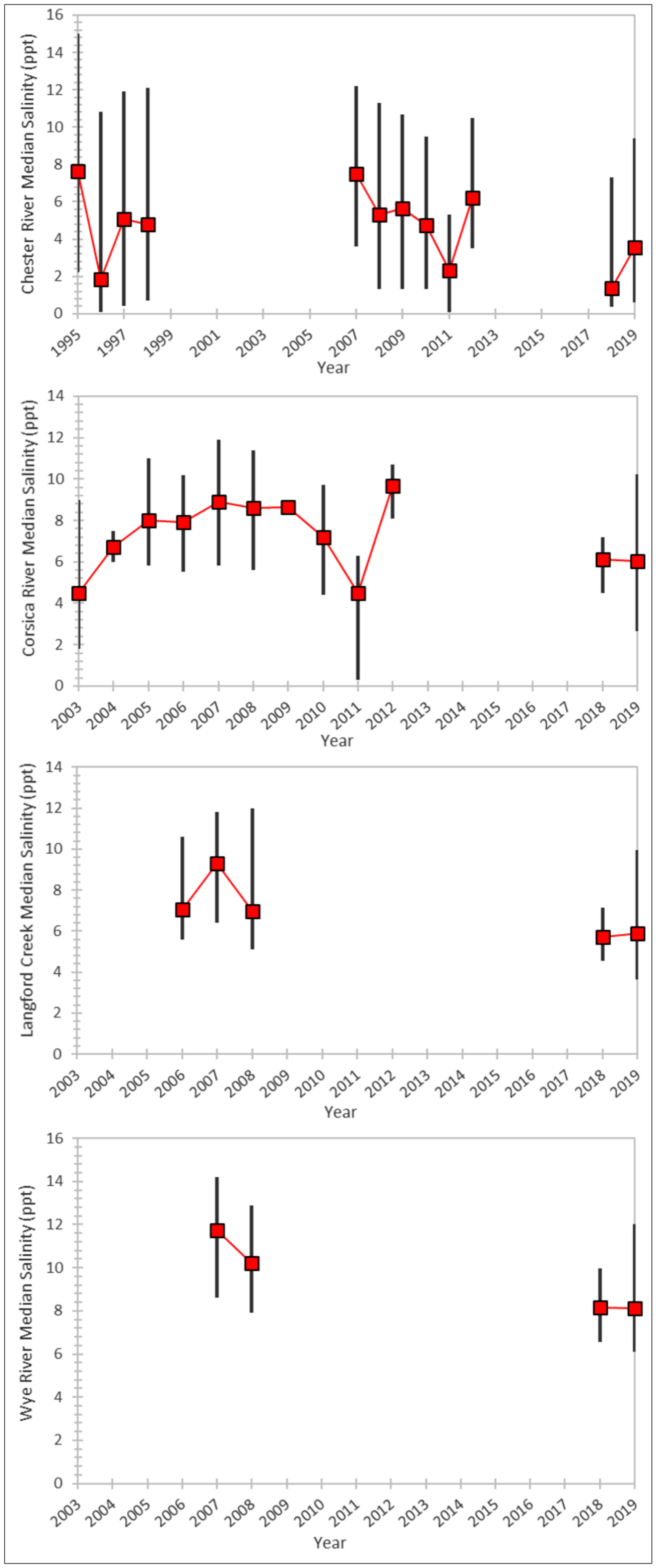


Figure 3-41. Chester River bottom dissolved oxygen (DO; mg / L) measurements versus bottom salinity measurements (‰) during 2007-2012, and 2018-2019. Red dashed lines indicate DO target (5 mg / L) and threshold (3 mg / L). Black line indicates the linear trend of the bottom DO occurring within the Chester River.

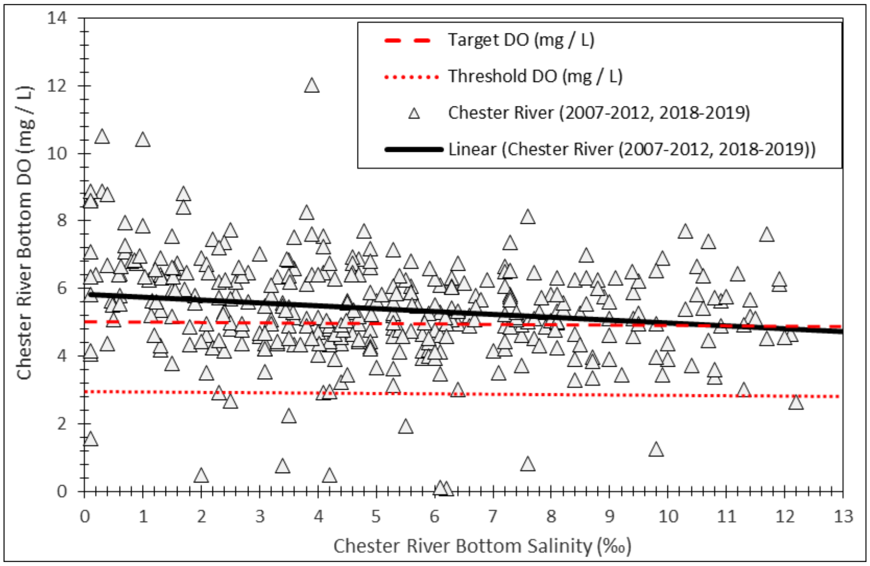
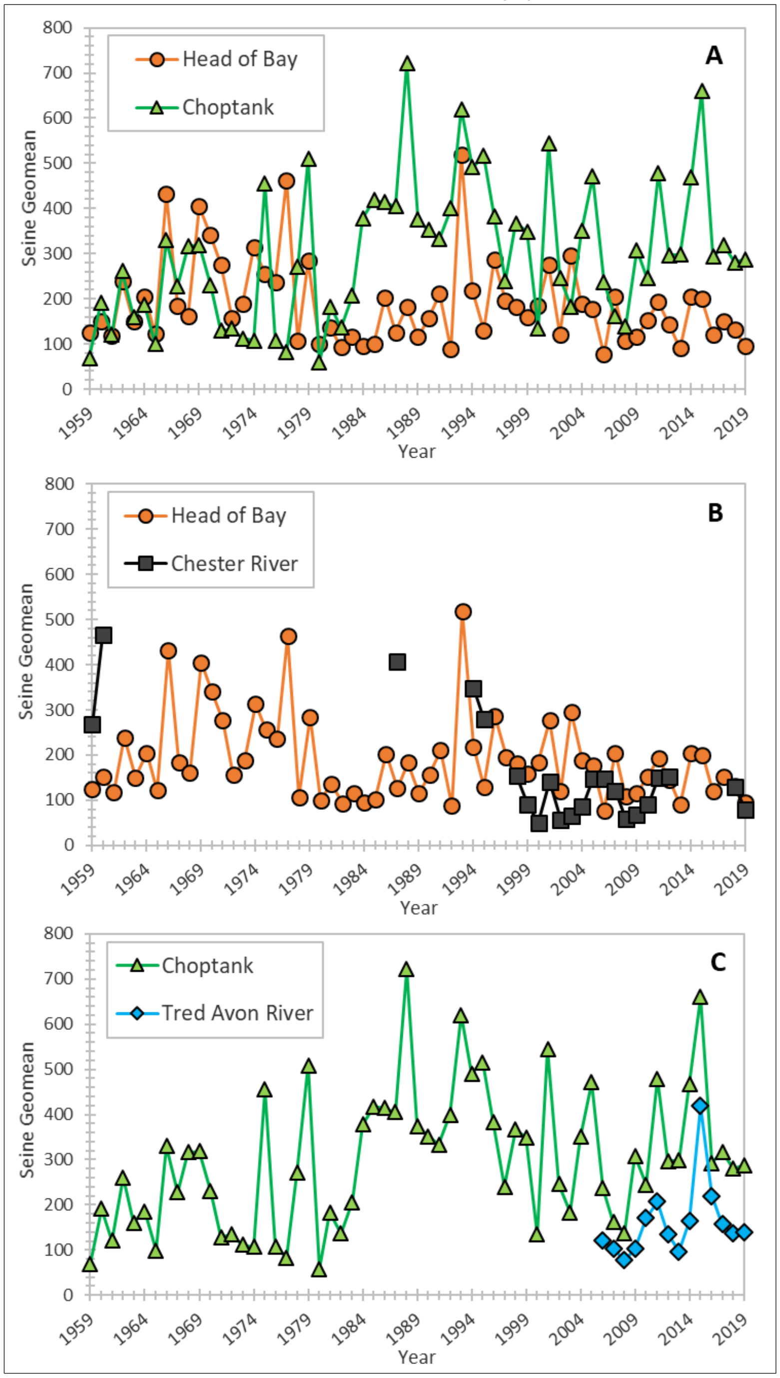


Figure 3-42. Geometric means (GM) of annual beach seine catch during 1959-2019 for all finfish species in the Chester River (black squares), Head of Bay (orange circles), Choptank River (green triangles), and Tred Avon River (blue diamond), by year.



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 Seth Dawson

Introduction

The objective of Job 2 was to document participation of the Fisheries Habitat and Ecosystem Program (FHEP) from July 1st, 2019 to June 30th, 2020 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 FHEP staff through data collection and compilation, analysis, and expertise are vital if Maryland is to successfully add 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 and publications. The web site was redesigned in April 2015 to help with navigation. The website can be found at https://dnr.maryland.gov/fisheries/Pages/FHEP/index.aspx.

Publications – J. Uphoff is a coauthor on two invited publications that are being written on ecological reference points (ERPs; forage reference points) for Atlantic Menhaden.

Environmental Review Unit Bibliography Database – We maintain an Environmental Review Unit database, adding additional literature when it becomes available. Older reports that are not in electronic format are scanned in to preserve the reports and data for future use. Striped Bass reports from 1955 to 1963 were electronically scanned in for long-term storage and protection.

Review of County Comprehensive Growth Plans – We reviewed comprehensive growth plans for Trappe and the Town of Port Deposit, 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 stakeholders, planning staff, and implementers group to highlight the importance of fishing in the county and offer assistance to incorporate fish habitat needs in future planning activities. .

Cooperative Research – M. McGinty participated in a Bay Program Fisheries GIT meeting where habitat was discussed.

M. McGinty participated in the Chesapeake Bay Trust proposal review process, evaluating proposals to develop habitat suitability indices for striped bass juvenile habitat. J. Uphoff and M. McGinty are on the project steering committee.

M. McGinty met with the Smithsonian Estuarine Research Center (SERC) to share information about the oyster fouling data set that has been vetted and analyzed. Our program has developed a hard bottom benthic index from these data to serve as an indicator of benthic forage for Chesapeake Bay gamefish for this important benthic habitat. SERC was looking at oyster fouling and population data to examine correlations with disease dynamics. These data were shared with SERC.

J. Uphoff participated in a Fishing and Boating Services exercise to refine a catfish management plan for Maryland.

J. Uphoff contacted Dr. Jeremy Testa at Chesapeake Biological Laboratory to discuss reasons for a marked increase in average Choptank River pH and a decrease in its variability in the striped bass spawning area between 1986-1991 and 2014-2019. These changes represent improved habitat conditions for early life stage survival.

M. McGinty worked with Environmental Review to refine Yellow Perch Spawning maps and clarify their application for determining where to apply time of use restrictions. This was part of an effort to streamline permit reviews. We recommended using our maps with the various management priorities as a triage tool to determine when a review merits stronger scrutiny.

M. McGinty shared information about anadromous spawning data with the Interagency Review Team. This team is staffed with representatives from NOAA, ACOE, USFWS, MDE and MD DNR and is tasked with coordinating projects that require permit review. At the request of the team M. McGinty committed to provide a mapping tool that can help them assess habitat quality and streamline the permitting process. She updated historical spawning maps to refine spawning habitat maps and tools and developed a data base to support mitigation and restoration siting. This tool is described in Job 2 of this report.

A. Park and C. Hoover provided field support to additional MD DNR programs, Coastal Bays, Resident Species, and Hatcheries.

J. Uphoff, M. Margaret, A. Park, and C. Hoover collaborated with other MD DNR Fish Health, Shellfish, Fish Passage, Alosines, and Hatcheries programs and projects via data, research findings and advice.

Presentations and Outreach – A. Park attended *Water Quality and Agriculture in the Choptank Watershed* at Washington College. The presentation discussed the use of BMPs throughout the Choptank watershed on three different spatial scales: farm, watershed, and an intermediate scale.

M. McGinty reviewed final draft of the Atlantic Coast Fish Habitat Partnership's Southeast Prioritization Mapping report.

M. McGinty, A. Park, and C. Hoover participated in a NOAA webinar titled *Fisheries in a New Era of Offshore Wind Development.*

M. McGinty drafted a brief document describing a plan that Fisheries will pursue to examine potential impacts of saltwater intrusion and other episodic climate impacts.

M. McGinty attended the joint American Fisheries Society and The Wildlife Society national annual meeting. She presented on the HBBI she developed as an indicator of forage conditions for benthic gamefish provided by the epibenthic community on hard bottoms (oyster bars). This index was developed from previously unused data.

M. McGinty, A. Park, and C. Hoover participated in a NOAA webinar titled *Improving Microplastics Research* given by Judith Weis of Rutgers University.

J. Uphoff, M. McGinty, A. Park, and C. Hoover attended the Maryland Water Monitoring Council conference at the Maritime Institute on December 6, 2019. J. Uphoff presented *Declining Status of Anadromous Fish Spawning Habitat in Patuxent River* and M. McGinty presented *Foul Play: Long Term Data Trends in the Epibenthic Community of Maryland Oyster Bars* at the conference.

J. Uphoff, M. McGinty, A. Park, and C. Hoover participated in a NOAA webinar titled *Tidal Wetland Loss, Restoration, and Fish Response: Tales from the Pacific Coast.*

J. Uphoff responded to several inquiries regarding stocking Yellow Perch in a restored reach of Bacon Ridge Branch on the South River. Stocking was not recommended, however, staff did agree to review the restoration plan and provide comments and support for elements that would benefit fish habitat in the future.

M. McGinty, A. Park, and C. Hoover participated in the AFS Virtual Spring Conference where multiple AFS Chapters gave presentations regarding fisheries across the United States virtually due to cancelled AFS Chapter meetings because of the pandemic.

A. Park provided a recorded presentation on the Bush River estuarine fish community for Anita C. Leight Estuary Center (ACLEC) for their volunteer training. Data has been collected since 2006 (MD DNR 2006-2010) and by the ACLEC volunteers since 2011. FHEP provides an annual updated presentation on their data during ACLEC's volunteer training workshop.

J. Uphoff submitted an abstract on the link between watershed urbanization and decline of anadromous herring in Patuxent River for consideration as part of a symposium, *Confronting Present and Emerging Stressors in Rivers for Global Fisheries Conservation*, for the upcoming 2020 virtual annual meeting of the American Fisheries Society. The talk was accepted and a presentation was developed.

Atlantic States Marine Fisheries Commission (ASMFC) – J. Uphoff provided a supporting Steele-Henderson Striped Bass-Atlantic Menhaden model for Ecological Reference Point (ERP) workgroup as one of a suite of models that were developed to provide management advice on the forage role of Menhaden. These models went to peer-review and an approach that mixed a single species Beaufort Assessment model and an Ecopath with Ecosim model of intermediate complexity were approved for giving management advice. Maintaining the forage role of Atlantic Menhaden was adopted as a primary management goal. The full ERP report is (560 pp.) available online at: http://www.asmfc.org/files/Meetings/2020WinterMeeting/AtlMenhadenERPAssmt_PeerReviewReports.pdf

A. Park and C. Hoover participated in ASMFC's *Introduction to Stock Assessment* on-line course.

M. McGinty reviewed the initial draft of ASMFC Habitat Committee's document *Fish Habitats of Concern Designations for Fish and Shellfish Species Managed by the Atlantic States Marine Fisheries Commission.*

Chesapeake Bay Program (CBP) – M. McGinty participated in the CBP Fish Habitat Action Team (FHAT) meeting and in follow up discussions to update the workplan. She also participated in a webinar to share updates from Chesapeake Bay Fisheries Funded Research, and reviewed and submitted comments to a CBP FHAT fact sheet to communicate the value of fishing to localities.

J. Uphoff and M. McGinty gave a webinar presentation on its fish habitat studies and application of these analyses to the Chesapeake Bay Program's Integrated Trends Analysis Team (ITAT). The ITAT analyzes water quality trends and is interested in relating its analyses to fish and fish habitat. The information presented was well received and may lead to coordinated analyses in the future.

M. McGinty participated in two Community Based Social Marketing meetings to support the Fisheries Habitat Workgroup's effort to develop an outreach and marketing approach to motivate landowners to consider using living shorelines when practicable.

J. Uphoff and M. McGinty participated in conference call to define roles and responsibilities of steering team members associated with a CBP funded project to develop refined Striped Bass juvenile habitat suitability indices.

J. Uphoff monitored activities of the Forage Action Team as they slowly develop forage inidicators.

Envision the Choptank – J. Uphoff and M. McGinty, representing MD DNR fishery management concerns, participated in meetings geared toward assisting local government to incorporate natural resource needs into county comprehensive growth plans. A description of Envision can be found at https://www.envisionthechoptank.org/.

M. McGinty reviewed NOAA's final report that evaluated land use impacts on the Tred Avon River. The Tred Avon is one of our treatment rivers, chosen to track changes in habitat over time as urbanization increases in the watershed. M. McGinty met with one of the report authors to discuss findings and share information.

M. McGinty developed a list of available fisheries information to inform localities of fisheries resources in their jurisdictions in order to promote local awareness of the value of conserving fisheries habitat. This information can be provided to any county, not just those in the Choptank River watershed.

JOB 3: Developing Priority Fish Habitat Spatial Tools

Margaret McGinty and Jim Uphoff

Abstract

This report describes updated mapping of historical anadromous fish spawning data to support requests for habitat maps for all life stages of anadromous fish in Maryland. We applied historical spawning data and impervious surface target and thresholds at three different watershed scales (8 digit, 12 digit, and catchment; large, medium, and small scale, respectively) to explore the potential to target small scale restoration and conservation projects. We developed a composite habitat rank (Hrank) for a station that combined watershed condition scores at the three scales; the three scales reflected dependency of small scale watershed on larger scale watershed condition as well as their local condition. There was a total of 1,239 stations sampled to assess anadromous spawning areas in studies conducted between 1967 and 1990. Proportion of sites sampled with a species present was highest for White Perch (0.51), followed by Herring and Shad (0.45), and Yellow Perch (0.32). There was a notable increase in impervious cover between 1970 (adopted as a baseline for comparisons) and 2018, with land area above the impervious surface threshold in Maryland nearly increasing from 15.3% in 1970 to 27.4% in 2018. This change was most pronounced in the Baltimore-Washington corridor. Scale was an important consideration when characterizing anadromous fish spawning habitat potential in a watershed. The percentage of watersheds with anadromous fish present decreased with decreasing watershed scale and habitat condition scores were not redundant among scales. On a percentage basis, declines in preferred habitat were greater at the 8-digit scale (-28%) than the 12-digit or catchment scale (both ~ -11%). Gains in marginal habitat were greatest and similar (+39%) at the 8- and 12-digit scale, and less at the catchment scale (+22%). Changes in acceptable habitat were similar among scales (-17% to -23%).

Introduction

Recently, an interagency workgroup that reviews projects to mitigate environmental impacts (interagency review team or IRT) requested habitat maps for all life stages of anadromous fish in Maryland. This prompted a review of historical surveys that revealed a need to refine existing maps and develop a supporting data base.

In response to concerns over declining stocks of anadromous species, O’Dell et al. (1972) established the first study to inventory anadromous habitat in Maryland. This initial study applied various methods to identify spawning habitat and laid the groundwork for subsequent studies by region to inventory all anadromous spawning habitat in Maryland. Focal species included the five anadromous species in Maryland (Alewife, Blueback Herring, American, Shad Hickory Shad, and Striped Bass) and two semi-anadromous species (White and Yellow Perch). Data from these studies was computerized and housed in a central state database. However, these data were lost during a system failure. Fortunately, data are still partially available in reports and computer printouts, and have been used to develop mapping tools for permit reviews and guiding management priorities (Mowrer and McGinty 2002; Uphoff et al. 2013). However, these previous efforts were limited in the information recorded and only included species presence by location. Recent requests for mapping tools that can inform small scale planning and provide historical context for habitat management prompted us to enhance existing mapping capability.

O'Dell et al. (1972) recognized anthropogenic stressors would limit habitat and partnered with sister agencies to investigate Patuxent River (suspected to have impacts from water quality) and the Chester River (selected as a pilot area for developing methods). Besides noting presence of stream blockages, field crews also reported suspected pollutant sources such as pipes, sewage effluent, riparian disturbances (such as livestock access), or unusual water quality conditions. This information was relayed to the Water Resources Administration for further investigation and corrective action. This reflected a concern that multiple stressors (physical and chemical) could be impairing habitat and limiting its use. Subsequent studies continued to record limited water quality information and inventory blockages, with biological sampling becoming the focus. This change in focus was presumably driven by limited resources and heightened concerns over declining stocks of anadromous species. Hindsight shows the misfortune of abandoning water quality collections, because it does not allow us to assess changes in water quality over time to evaluate these as a source of stress or at the very least an indication of habitat change.

The first step for updating anadromous fish habitat maps was to review old reports to mine additional data. This effort made us aware that additional historical information was available, but in disparate locations. This prompted us to develop an inventory of historical reports and one comprehensive database to preserve the information in one central location. The following describes this effort and is intended to serve as a summary of the historical surveys that can be cited as metadata for spatial files.

The inventory of spawning areas in Maryland, collected primarily in the 1970s to mid - 1980s (Table 1), has been applied to designate anadromous spawning habitat for special protection. However, land use has significantly changed, mostly due to development, since these data were collected and when mapped do not account for modern stressor impact (See **Common Background for Job 1, Sections 1-3)**.

Since our program's inception, we have conducted studies to examine the impact of land use change (focusing on suburban sprawl) on fish habitat (McGinty et al. 2006-2009; Uphoff et al. 2005; 2011a; 2010-2018). One outcome of this work was the development of impervious surface targets and thresholds for fisheries and habitat management (Uphoff et al. 2011). In 2014, we applied these thresholds in a mapping exercise to establish habitat management priorities for anadromous spawning areas in Maryland (Uphoff et al. 2014). This effort used historical data from O'Dell (1967-1990; Table 3-1) to indicate historic spawning habitat and applied impervious surface targets and thresholds to prioritize watersheds based on historical habitat use and contemporary land use condition. These maps were intended to serve as a modern snapshot of habitat condition with impervious surface representing multiple stressors associated with development.

Current maps have all species sampled at a station mapped individually (Mowrer and McGinty 2002). In some cases, though a single station at a location was sampled and multiple species were present, maps developed showed these species present at two different (albeit very close) locations. For example, a single ichthyoplankton station was sampled on a stream and found to have eggs of White Perch and Herring. Yet when mapped, these data had two different station locations (one for Herring and one for White Perch). Additionally, only presence of a species was mapped and we were interested in knowing stations where species were not observed.

This report describes mapping to support new data requests using the updated data and the database and associated projects from which the data were derived. This effort provides additional information about the sampling stations, pins all species observations at a station to the same station, and provides a database that serves as an archive for these studies. This new database also includes sites where fish were absent, allowing us to better represent key spawning habitat in Maryland. This new database and accompanying map will fulfill data requests to provide accurate locations of spawning areas to assess potential impacts on anadromous spawning areas in Maryland for environmental review, species management plans that require historical and present metrics quantifying habitat by life stage, and to target specific management action to promote conservation of viable habitat while directing potential restoration in areas where it is more likely to be effective.

We also updated original maps to examine present conditions. While agriculture is a large scale land-use in Maryland, we have found that it supports anadromous fish spawning while development has been detrimental and a source of stressors (see Job 1, Sections 1 and 2, for descriptions of how these two land uses related to anadromous fish spawning). These two human based land uses are strongly and inversely correlated (Uphoff et al. 2019), making use of one or the other necessary analytically. Since tax map data is updated annually, we have a continuous record of development available for 1950-2018 (see **General Spatial and Analytical Methods used in Job 1, Sections 1-3**). Estimates of agricultural land use are produced intermittently by the Maryland Department of Planning and the last estimate was made for 2010.

We examined different watershed scales to explore the potential to use these data to target effective management to small scale projects. The rationale for multiple scales reflects the influence of larger scale watershed conditions on success of stream restoration at a smaller scale (Wang et al. 2001; Walsh et al. 2005; Palmer et al. 2010; Simenstad and Cordell 2010; Stoll et al. 2016). With these results in mind, we assessed impervious cover at three watershed scales and overlaid these layers to demonstrate the potential to use these overlays to score stations based on land use influences at these three different scales. While this approach does not identify specific stressors, it targets management to watersheds and sites where the cumulative effects of disturbances at a large scale are less pronounced.

Methods

We gathered all available reports that were associated with the anadromous spawning surveys. These were grey literature and were not housed in one central library or location. These reports documented results from six studies conducted by the Department of Natural Resources, a study by University of Maryland, and reports produced by Coastal Conservation Association that recorded information on location of Yellow Perch spawning (Table 1). All reports were reviewed for content and evaluated to determine the utility of data contained within. These reports, except O'Dell (1972), contained hand drawn maps of sampling stations and attendant information on species observed at each station.

Initially, stations were identified as anadromous habitat if eggs, larvae or adults were collected through sampling in the spring, or juveniles or adults were observed present by wildlife officers, anglers or biologists (Table 1). This set the stage for successive studies which employed adult fish traps and plankton net sampling to identify spawning areas (Table 1). Streams were identified as candidates for sampling if they were at least one mile in length, salinity was < 3.5 ppt and stream barriers were absent (O'Dell 1972). Each candidate stream was investigated by locating a station near the mouth of the stream that had an access point (typically a road crossing). Additional sites were located upstream at approximately one-mile intervals. If a barrier was encountered and determined to preclude upstream migration, a sampling site was established on the downstream side of the barrier (O'Dell 1972).

Once reports were reviewed, we developed a new spatial dataset with a standardized supporting database that contained a specific station identifier, report number, source of the data within the report (page or appendix number), basin name, stream name, location (road crossing or other landmark associated with the site), county, latitude and longitude, habitat type sampled (spawning or juvenile), investigator (MD DNR staff or other as indicated), blockages present when indicated, approximate year of sampling, presence by species and life stage, and comments. These attributes were established from a table of locations sampled with a description of the station and species observed in the first study (O'Dell 1972) since it did not have them mapped. Additional attributes were appended to accommodate new key variables (species by life stage) in latter studies. This exercise produced maps with a supporting database with all species observed at a station linked to a single latitude and longitude. We plotted all stations sampled, stations with each anadromous species or species group, as well as a plot summarizing whether anadromous fish were present at a site as a combined group. These breakdowns were requested by the IRT.

We applied impervious surface target and thresholds at three different watershed scales to assess the potential to use historical presence with contemporary land use to target management action at smaller scales. Impervious surface was estimated by clipping Maryland property tax data to each watershed scale and estimating housing density by watershed (number of dwellings per hectare). These estimates were converted to impervious cover by applying the equation developed from associating impervious cover with housing density estimate (see **General Spatial and Analytical Methods used in Job 1, Sections 1-3**).

$$IS = 10.129 (C/ha) + 1.286;$$

where IS = impervious surface and C/ha = structures per hectare. We assigned 1970 as a reference year for land use since the first study was completed then and subsequent studies were initiated in 1970. We compared impervious surface estimates in 1970 and 2018.

We assigned previously designated habitat categories (Uphoff et al. 2014) to targets and thresholds: watersheds with impervious cover less than 5% were considered preferred habitat areas (at or below the development target; potential high productivity), watersheds greater than 10% impervious surface (above the development threshold; potential low productivity) were assigned as marginal habitat, and impervious cover between 5 and 10% was considered acceptable habitat (potential moderate productivity). We then compared impervious cover in 1970 to 2018 using these categories at the three watershed scales to determine the number or watersheds that changed in priority classification over the time frame.

Large scale data were estimated from Department of Natural Resources 8 Digit Watersheds (MDE8Digit; MD DNR 2008a). This is a statewide digital data set delineating 138 watersheds uniquely identified by an eight-digit watershed code. These data were developed by identifying watershed boundaries for third order streams according to contours delineated on U.S. Geological Survey 7.5 minute quadrangle maps with larger watersheds developed from these boundaries. Average area of large scale watersheds was 24,728 hectares and ranged from 2,753 to 83,991 hectares (Figure 1). This was the scale used to develop impervious surface reference points (Uphoff et al. 2011). We originally chose this scale, because we thought it best represented the cumulative watershed impacts influencing tidal summer habitat. It was also the scale used by Maryland Department of the Environment in developing water monitoring programs (MD DNR 2008a).

Medium scale data were derived from Maryland Department of Natural Resources 12 Digit Watershed (DNR12Digit; MD DNR 2008b). This statewide digital data file represents third order watersheds in Maryland. They were delineated using U.S. Geological Survey 7.5 minute quadrangle maps (MD DNR 2008a). There are 1,136 watersheds ranging from 265 to 763 hectares with an average watershed size of 3,295 hectares (Figure 2).

Small scale watersheds (catchments) were estimated using National Hydrography Dataset Watersheds (USGS 2012) which were developed by defining the watersheds associated with small streams and stream segments. Watershed boundaries were delineated based only on hydrologic principle and did not incorporate political boundaries, therefore in some case they extended beyond jurisdictional boundaries. There were a total of 15,398 catchments delineated for Maryland ranging from 2 to 4,372 hectares with a mean of 438 hectares (Figure 3). There is a notable range in watershed area among and between the three scales used to delineate watersheds. This is driven by topography of the landscape which dictates the watershed boundaries. For example, mountainous regions with a number of ridges would likely have more and smaller catchments than a stream valley that is expansive but contained within a single watershed boundary.

We applied an impervious surface target and a threshold to assess the level of development in each watershed at each scale. We then scored each watershed according to the target and threshold where watersheds below the target of 5% received a score of 5, watersheds between the target (5%) and below the threshold (10% impervious surface) received a score of 3, and watersheds above the threshold of 10% received a score of 1. We combined individual station data with the three watershed scales and evaluated changes in land use from 1970 (when the first study was completed) and 2018 (the latest land cover dataset available). We also assigned the watershed score for each scale to each station and summed these scores to explore the potential for developing a station ranking approach.

$$Hrank = W8score + W12score + Wcatchscore;$$

where Hrank is the habitat rank, W8score is the 8 digit score, W12score is the 12 digit score, and Wcatchscore is the catchment score. We assessed the distribution of these ranks and assigned priorities based on the data distribution, where ranks falling below the 25th percentile were assigned as low priority, ranks between the 25th percentile and the median were assigned moderate priority, and ranks exceeding the median were given high priority. The rationale of the three scale Hrank reflected that small scale watershed functions were dependent on larger scale watershed condition as well as their local condition (Walsh et al. 2005; Palmer et al. 2010). The IRT requested that maps provide guidance on locating suitable areas for restoration and mitigation that are generally small scale. We selected the Patuxent River watershed to provide an example demonstrating how these categories might be applied in habitat management.

Results

There were a total of 1,239 stations sampled to assess anadromous spawning areas in studies conducted between 1967 and 1990 (Table 1). The original study (1968-1970; AFC-3) reported data from a total of 293 stations (Table 1; Figure 4). Subsequent studies (1970-1990) reported data from 1,143 stations. Coastal Conservation Association surveyed 98 stations (2001-2008) to document Yellow Perch spawning (Table 1; Figure 4).

Juvenile sampling was conducted at 654 stations in the 1970s (Table 1). We did not evaluate juvenile data, but intend to revisit it to update juvenile habitat maps and determine its utility to assess changes in habitat occupation over time. There are additional data to be mined and evaluated which could potentially be useful further examining the extent of habitat change since the original surveys were conducted (Table 1).

We pooled all the spawning data to calculate proportion presence (Pp) by species. White Perch were present most often (Pp = 0.512; Figure 5), Herring and Shad Pp equaled 0.446 (Figure 6), Yellow Perch Pp was 0.318 (Figure 7), and Striped Bass Pp was 0.054 (Figure 8). All species except for Striped Bass were widely distributed. Striped Bass distribution was limited to lower areas of tributaries in tidal habitat consistent with previous work that delineated Striped Bass spawning areas (Hollis et al. 1967; Figure 9). American Shad presence was generally limited to tidal areas above sites where Striped Bass were present (Figure 9). Data for individual species were combined and plotted as anadromous fish spawning presence (one or more anadromous species present) as requested by the IRT (Figure 10). These maps illustrate the distribution of spawning habitat as it was historically inventoried.

The percentage of watersheds with anadromous fish present decreased with decreasing watershed scale. Of the 84 watersheds sampled, only 5 (6%) watersheds indicated absence of spawning at the MDE8Digit watershed scale (Figure 11). At the DNR12Digit watershed scale, 17% (71) of watersheds sampled indicated absence (Figure 12), while 32% of Catchments sampled indicated absence (Figure 13).

There was a notable change in impervious cover between 1970 and 2018 at the 8 digit watershed scale (Table 2). The greatest increase occurred in land area with greater than 10% impervious surface category; Maryland-wide estimates almost doubled from 15.3% in 1970 to 27.4% in 2018. (Table 2). We consider 10% impervious surface a tipping point beyond which habitat limitations become difficult to address with traditional management strategies and we have applied this threshold to predict losses to fish habitat. The change in areas with greater than 10% impervious were most pronounced on the western shore in the Baltimore-Washington corridor (Figure 14). Rural land with impervious cover between 5% (target) and 10% (threshold) showed the smallest change, increasing from 9.2 – 10.6% of the land area, while land meeting the target of 5% decreased from 75.5% in 1970 to 62.2% in 2018. The eastern shore showed less change with most of the area meeting the target of 5% impervious surface (Figure 14).

When we evaluated these changes between 1970 and 2018 at the three watershed scales, we found that twelve watersheds at the MDE8Digit scale declined in priority status, twelve at the DNR12Digit scale declined and 55 at the Catchment level declined in status (Table 3). On a percentage basis, declines in preferred habitat were greater at the 8-digit scale (-28%) than the 12-digit or catchment scale (both ~ 11%). Gains in marginal habitat were greatest and similar (+39%) at the 8- and 12-digit scale, and less at the catchment scale (+22%). Changes in acceptable habitat were similar among scales (-17% to -23%; Table 3). Changes among all three categories were not redundant, indicating each scale provides its own perspective on habitat conditions. Lack of redundancy was important for the use of the Hrank score to avoid double counting of the same habitat conditions.

Changes at the MDE8Digit scale were most pronounced on the Western Shore (Figure 15). Compared to 1970, marginal habitat in 2018 increased along much of the middle to upper Western shore (Figure 15). Changes in the DNR12 Digit watersheds show a similar band of marginal habitat along the Western shore, while also exposing areas on both shores that migrated from the preferred category to the acceptable category (Figure 16). The picture was patchy at the catchment scale because it was closer to the station scale (Figure 17) rather than integrated over larger scales.

At the MDE8Digit scale (Figure 18), most stations with Hrank at low or medium priority occurred in watersheds designated as marginal habitat (>10% impervious), while high priority stations were associated with preferred habitat (watersheds less than 5% impervious cover). Watersheds that fell in the acceptable category had a mix of medium and high priority stations.

When Patuxent River station priorities were compared to watershed impervious surface at MDE8Digit, the upper watershed area was classified as marginal habitat (>10% impervious surface) and stations within this area were for the most part low and medium priority (Figure 19). However, when we examined these stations at MDE12Digit scale, some smaller watersheds were classified at acceptable (Figure 20). For example, the area in the box in panel a of Figure 20 show stations scoring as low and medium priority while the MDE8Digit watershed was categorized as marginal habitat. Examination of this same area at the DNR12Digit scale (Figure 20b) showed some medium priority stations in watersheds categorized as acceptable and preferred. Honing in on this same area at the catchment level allows for delineation of condition at small scale based on management priorities (Figure 21). The area at the tip of the arrow in Figure 21 (panel a) shows the DNR12Digit watershed falls into marginal habitat category, while at the catchment level this areas is classified as preferred habitat.

To demonstrate the potential use of these overlays, we considered a scenario where a county agency may want to target restoration or conservation to benefit anadromous spawning habitat. Figure 22 shows the Anne Arundel County boundary imposed on the Patuxent River with station priorities designated and catchments categorized by anadromous management priority. The box on the figure indicates an area a management agency might focus on if interested in conservation and or restoration of habitat. With this focus area in mind, they could hone in on these habitats and choose catchments for management action based on their objectives. In this case, if they were targeting watersheds for conservation, they might choose to focus on catchments classified as preferred habitat where stations were designated as high priority to promote conservation (Figure 23). If restoration was the objective, they could hone in on catchments classified as acceptable with stations designated as medium or even high priority if they were present (Figure 23). These overlays provide the benefit of considering a station within the context of the watershed condition at various scales.

Discussion

These historical studies contained a wealth of data. This present effort focused on examining and digitizing spawning habitat data, providing a fairly robust database to assess historical distribution of spawning habitat in Maryland. There is more untapped potential in these reports for comparing present and historical habitat conditions.

Historical studies reported very low occurrence of American and Hickory Shad and Striped Bass and these spawning habitats are likely underrepresented. The low occurrence of American and Hickory Shad was attributed to declining populations, while low presence of Striped Bass was related to the paucity of sampling in their known habitat (O’Dell et al. 1985). It is possible that American and Hickory Shad are under-represented in the sampling for the same reason. Bilkovic et al. (2002) found that American Shad spawning habitat in Virginia overlapped Striped Bass which use the upper tidal fresh reaches of large tributaries for spawning. Maryland DNR and its predecessor agencies have sampled Striped Bass spawning areas since the early 1950s (see Job 1, Section 2.1) and our program is in the process of creating a georeferenced data set with as much of this information as possible. However, these spawning areas were defined in Hollis et al. (1967) and our experience has been that these boundaries have held up well over time.

In addition to the digitized data, there are several data tables and print outs containing water quality information and observations of other species that could provide historical context for additional studies. We plan to digitize these data in the future to examine changes in condition against this baseline period. These additional resources are documented in Table 1 under the "additional data sources" heading.

Our approach to prioritize stations based on impervious cover at various scales is a first attempt to use these data to guide management decisions. This approach attempts to account for the larger scale watershed condition when considering management at the local scale. Simenstad and Cordell (2000) advocated taking a broad landscape perspective to promote successful Salmon restoration in the Pacific Northwest; it was fundamental to restoration planning and implementation. By focusing management on watersheds with lower impervious cover, we believe there is a better chance of seeing habitat improvements.

Examining presence by watershed scale could be useful in future applications to prioritize habitat, especially if there is an initiative to examine historical connectivity and clusters of watersheds historically supporting spawning. For example, fisheries managers can apply habitat-based reference points to examine changes in production related to habitat changes. They can then use this information to look at projected land use change and develop estimates of potential losses to production based on habitat loss. Land managers that typically work at a smaller watershed scale (catchment or stream reach) may be interested in assessing resource condition on a smaller scale, to target small watersheds for conservation or restoration.

We do plan to continue to refine this tool. The more complete data base lends itself to developing metrics that may be useful in developing habitat tools by species. We will explore the potential in the coming year. Additionally, we plan to continue to mine data to determine if we can reconstruct historical habitat, particularly for juvenile life stages as there appears to be potential to develop data from historical reports. If so, we can compare historical proportion presence to today's measures. We also would like to explore the smaller scale data to increase our understanding of effects of impervious cover. We could explore the potential to incorporate additional years of land use data that correspond with the specific year data were collected and then compare presence and land use at the time samples were collected to presence and land use during the first study. Additionally, we could examine data collected over the last fifteen years to assess their utility in assessing changes in biological scores compared to historical data.

We also hope to conduct sampling on the Patuxent River to re-assess spawning habitat to examine changes in spawning habitat use from historical status. This information will inform our understanding of impacts of land use change while also providing the county with more data to target conservation and enhancement activities. Finally, we will continue to mine data and use it to refine indices and mapping tools. We will use the present information to work with various user groups to assess utility of these tools and make refinements to support needs related to targeting habitat for conservation and sound management.

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Table1. Key information regarding studies conducted to inventory anadromous spawning habitat in Maryland.

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
AFC- 3	O,Dell, C. J. 1972. Stream Improvement Program for Anadromous Fish Management, Federal Aid Report. Submitted to US DOC, NOAA, NMFS	June 1, 1967- August 31, 1970	Through biological sampling and stakeholder interviews streams supporting anadromous spawning were identified allowing managers to better assess potential threats that watershed development could impose on anadromous spawning habitat. Stream barriers were inventoried and some habitat improvements were made. A cooperative program was established with state and local partners to promote pollution abatement practices.	Chesapeake Bay tributaries and streams Coastal Bays streams Chester River used as pilot to develop sampling approach (see report for specific methods and approach); Patuxent River focal study on water quality and anadromous fish spawning areas conducted	Inland and Tidal habitats were sampled to identify potential anadromous spawning areas	293 spawning stations	Report Appendices with inventory of species collected by river system and location. These data were mapped by locating the approximate location on the map.	Anadromous Species Presence, Water Quality Data taken: water temp (F), DO (ppm), pH, Turbidity (ppm), alkalinity (ppm), Conductivity (umhos/cm, salinity (ppt)	Water Quality: Hach Kits and conductivity meters Biological: Interviews, electrofishing, seining, visual observation, explosives, traps (in 1970 the primary sampling was conducted with wire traps)	Alewife, Blueback, American Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	Pilot Study of Chester River was conducted. The report describes the study and provides summary data on ranges of water quality parameters and a table of species observed by tributary. Appendices include data tables. Intensive investigation of the Patuxent River was conducted with regular water quality and fish sampling. Results are reported in report appendices.

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
AFC-8	O'Dell, C.J., J. Gabor, and R. Dintaman. 1975. Survey of Anadromous Fish Spawning Areas Completion Report, Project AFC-8. Maryland DNR Fisheries Administration, Annapolis, MD.	July 1970-January 1975	Conducted a 4.5 year study to identify spawning streams in the Potomac River Drainage and the Upper Bay (North of the Bay Bridge on the W. Shore to North of Chester River on the E. Shore).	Potomac, Elk, Lower Susquehanna, Bush, Bird, Gunpowder, Northeast, Elk, Bohemia, Sassafras, West Chesapeake Bay Tributaries north of the Bay Bridge, Chesapeake Bay Proper , Magothy, Patapsco, Back	Inland and Tidal habitats were sampled to identify potential anadromous spawning and nursery areas	864 (487 spawning stations; 377 juvenile stations)	Maps in various sections of the report. Data were mapped by approximating the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	Anadromous Species Presence, Water Quality Data taken: water temp (F), Conductivity (umhos/cm), salinity (ppt)	Biological: Fish Traps, Plankton Nets, Seines Each sites was sampled twice a week (one trap and one plankton sample each week) for a period of 12 weeks for a total of 12 samples per site. Investigators deemed probability of documenting presence was near 100%; Seining was conducted July-September to document juvenile habitat use	Alewife, Blueback, American Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	A supplementary study on American eel in the Potomac River and Upper Chesapeake Bay to document occurrence and abundance of eel in these areas. Results of this survey were summarized in a report titled, "A Preliminary Study of the Occurrence of American Eel and Other Finfish Species in Maryland. Limited water quality data for the Potomac River are contained in a table in the report. Additional data was to be computerized in latter years. It is unclear if these data can be mined at this time.

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
AFC-9	O'Dell, C.J., J. Mowrer and J. Gabor. 1980. Survey of Anadromous Fish Spawning Areas Completion Report, Project AFC-98. Maryland DNR Tidewater Administration, Tidal Fisheries Division Annapolis, MD.	January 1, 1975-December 31, 1979	Conducted a multi-year study to identify spawning habitat in the Chester River and West Chesapeake Bay Drainage. Additional studies were conducted in Choptank, South, Patapsco and Anacostia Rivers to assess effectiveness of addressing fish barriers.	Chester River mainstem and tributaries, West Chesapeake Streams draining to the Bay, Severn, South, West, Rhode Rivers	Inland and Tidal habitats were sampled to identify potential anadromous spawning and nursery areas	538 (261 spawning stations; 277 juvenile stations)	Maps in various sections of the report. Data were mapped by approximating the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	Anadromous Species Presence, Water Quality Data taken: water temp (F), Conductivity (umhos/cm), salinity (ppt)	Biological: Fish Traps and Plankton nets were employed to document anadromous spawning habitat occupation; seines were used to identify juvenile nursery areas for anadromous species	Alewife, Blueback, American Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	A limited number of sites were sampled related to newly established fish passage. Data were collected to assess efficacy of passage and use of upstream passage. Results are reported in Section D. Additional sampling for American Eel was conducted and data were published in a report, "A Preliminary Study of the Occurrence of American Eel and Other Finfish Species in Maryland, Volume II." Data tables are included in the Appendix with information on species presence by site and date. There is potential to use these to estimate proportion presence by site.

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
AFC-10 is represented by two reports containing sampling methods and results for Patuxent River Surveys.	AFC-10-1 O'Dell, C. J. and J. Mowrer. 1981. Survey and Inventory of Anadromous Fish Spawning and Nursery Areas, Segment Report, Project AFC-10-1 for Patuxent River Drainage Upper Chesapeake Bay. Maryland DNR, Tidewater Administration, Frihseries Division. Annapolis, MD.	January 1980-June 1981	Upper Patuxent River and Upper Bay in Harford and Cecil Counties were sampled for anadromous fish using traps and plankton nets in the spring and push trawls in summer to access nursery habitat. Data were also collected to characterize herring stocks in the Patuxent and Upper Bay and to estimate juvenile abundance	Upper Patuxent (above Central Avenue), Upper Bay in Harford and Cecil Counties and Upper Patuxent to assess summer juvenile habitat.	Inland and Tidal habitats were sampled to identify potential anadromous spawning and nursery areas	AFC 10-1 and AFC10-2 sampled a total of 73 spawning habitat stations combined	Maps in various sections of the report. Data were mapped by approximating the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	Species present by life stage, temperature and salinity	Biological: Fish Traps and Plankton nets were employed to document anadromous spawning habitat occupation; push trawls were used to sample juveniles. Adult fish were subsampled from commerical catches collected from gill and pound nets.	Alewife, Blueback, American Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	There are several estimates for herring species that could serve as a baseline for assessing changes in stock conditions in the Patuxent and Upper Bay areas. Additionally, there are tables with station information reporting species and life stages observed during spring sampling. These data could be evaluated to estimate proportion presence and establish a stronger baseline to assess changes.
AFC-10 is represented by two reports containing sampling methods and results for Patuxent River Surveys.	AFC-10-2 O'Dell, C. J. and J. Mowrer. 1983. Survey of Anadromous fish Spawning and Nursery Areas, Segment Report, Project AFC-10-2. Maryland DNR< Tidewater Administration	July 1981-June 1982	Middle Patuxent River was sampled in the spring to assess anadromous spawning habitat. Data collection to assess herring stock	Middle Patuxent River	Inland and Tidal habitats were sampled to identify potential anadromous spawning and nursery areas	AFC 10-1 and AFC10-2 sampled a total of 73 spawning habitat stations combined	Maps in various sections of the report. Data were mapped by approximating the location using county ADC maps and stream files. Most	species present, temperature and salinity	Biological: Fish Traps and Plankton nets were employed to document anadromous spawning habitat occupation; push trawls were used to	Alewife, Blueback, American Shad, Hickory Shad, Striped Bass, White Perch, Yellow	Contains baseline data for contemporary comparisons

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
	Fisheries Division, Annapolis, MD		continued as well as summer push trawl sampling to estimate juvenile abundance.				stations were located at road crossings as indicated in report methods.		sample juveniles. Adult fish were subsampled from commercial catches collected from gill and pound nets.	Perch	
AFC-14 is represente d by three reports, including two interim reports and one final.	AFC-14-1 O'Dell, C.J., J. Mowrer, R. Dintaman. 1984. Survey of Anadromous Fish Spawning Areas and Stream Barriers in the Upper Choptank River Drainage, Segment Report, Project AFC-14-1. Maryland DNR Tidewater Administration, Fisheries Division, Annapolis, MD	July 1983-June1984	The Upper Choptank River was sampled to assess anadromous spawning habitat and collect data to characterize adult spawning stocks of river herring.	Upper Choptank River	Inland Habitat was sampled to identify potential spawning habitat	A total of 105 spawning habitat stations on the Choptan k River were sampled	Maps in various sections of the report. Data were mapped by approximatin g the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	species present, surface temperatur e and salinity summary data by station sampled	Biological: Fish Traps and Plankton nets were used to document anadromous spawning habitat; subsamples of commercial catches were obtained for evaluation of adult fish	Alewife, Blueback , America n Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	Tables in the report contain information on surface water temperature and salinity. Summary statistics are included on Herring adult catch and length information.
AFC-14 is represente d by three reports, including two interim reports and one final.	AFC-14-1 MD DNR, 1985. Survey of Anadromous Fish Spawning Areas and Stream Barriers in the Lower Choptank River Drainage, Segment Report, Project AFC-14-2. Maryland DNR Tidewater Administration,	July 1984-June 1985	The Lower Choptank River was sampled to assess anadromous spawning habitat and collect data to characterize adult spawning	Lower Choptank River	Inland Habitat was sampled to identify potential spawning habitat	A total of 105 spawning habitat stations on the Choptan k River were sampled	Maps in various sections of the report. Data were mapped by approximatin g the location using county ADC maps and stream files. Most stations were	species present, ranges of surface salinity by station	Biological: Fish Traps and Plankton nets were used to document anadromous spawning habitat; midwater trawls were used to sample	Alewife, Blueback , America n Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	Tables in the report (section II) contain catch data by station for Herring with EPUE and salinity ranges recorded. This can be digitized to map juvenile habitat. Summary statistics are included on Herring adult catch and length

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
	Fisheries Division, Annapolis, MD		stocks of river herring. Midwater trawls were used to sample juvenile fish to identify anadromous spawning areas in Pocomoke, Nanticoke, Choptank, Chester and Patuxent Rivers. Adult Herring were sampled to describe stock structure.				located at road crossings as indicated in report methods.		juveniles to estimate abundance and defined nursery habitat; subsamples of commercial catches were obtained for evaluation of adult fish		information.
AFC-14 is represented by three reports, including two interim reports and one final.	AFC-14-3 Speir, H. and J. Mowrer.1987. Survey of Anadromous Fish Spawning Areas and Stream Barriers in the Choptank River Drainage, Final Completion Report (Not for Publication), Project AFC-14-1. Maryland DNR Tidewater Administration, Fisheries Division, Annapolis, MD	July 1983-March 1986	This is a Summary Report describing sampling conducted in AFC-14-1 and AFC-14-2	Choptank River and other Eastern Shore Rivers	Inland Habitat was sampled to identify potential spawning habitat	A total of 105 spawning habitat stations on the Choptank River were sampled	Maps in various sections of the report. Data were mapped by approximating the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	species present, ranges of surface water quality by station	Biological: Fish Traps and Plankton nets were used to document anadromous spawning habitat; midwater trawls were used to sample juveniles to estimate abundance and defined nursery habitat; subsamples of	Alewife, Blueback, American Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	Table in the report with water quality ranges for stations sampled on the Choptank. This information can be used to compare changes in habitat to historical habitat condition.

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
									commercial catches were obtained for evaluation of adult fish		
FR-37-R	2 Annual Reports F-37-R Wienrich, D.R., N. Butowski, W. Franklin, J. Mowrer. 1987. Investigation of Anadromous Alosids, USFWS Federal Aid Annual Report. MD DNR Tidewater Administration, Fisheries Division. Annapolis, MD	February, 1, 1986 - January 1, 1987; February 1, 1987- January 1, 1988	Annual Report describing data collected to characterize Herring Populations, including information on spawning habitat, juvenile and adult life stages.	Upper Nanticoke River and other areas on the Eastern Shore and the Patuxent River; Spawning Habitat was assessed on the Nanticoke, Miles, Tred Avon, Wye East and Wye River	Inland Habitat was sampled to identify potential spawning habitat	89 spawning habitat stations	Maps in various sections of the report. Data were mapped by approximatin g the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	species present, summary water quality information	Biological: Fish Traps and Plankton nets were used to document anadromous spawning habitat; midwater trawls were used to sample juveniles to estimate abundance and defined nursery habitat; subsamples of commercial catches were obtained for evaluation of adult fish	Alewife, Blueback , America n Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	Juvenile presence is included and could be used to map juvenile habitat on various tributaries on the E. Shore

Study	Source	Period Covered	Study Description	Systems Sampled	Habitat Sampled	Number of Stations Mapped	Source of Mapped Data	Parameters	Methods	Focal Species	Additional Data Sources
CZM 1990	Jesien, R., T. Hopkins, C Counts, R. Tackas. 1990. Anadromous Fish Survey of Somerset County Streams Final Report. University of Maryland, Eastern Shore, Princess Anne, MD.		Report describing anadromous spawning sampling in Somerset County	Wicomico, Pocomoke, Manokin and Big Annemessex Rivers	Inland Habitat was sampled to identify potential spawning habitat	21 spawning habitat stations	Maps in various sections of the report. Data were mapped by approximating the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	species present	Biological: Electrofishing and Plankton nets were used to document anadromous spawning habitat	Alewife, Blueback , American Shad, Hickory Shad, Striped Bass, White Perch, Yellow Perch	Table with all species captured are included and could be useful in assessing fish community changes.
CCA Yellow Perch	Coastal Conservation Association (CCA). 2008. 2008 Yellow Perch Spawning Habitat, CCA MD.	Spring sampling period 2001-2008	Describes yellow perch egg chain sampling conducted at various stations throughout Maryland's Tidal watersheds	Various tidal watersheds known to support Yellow Perch on the Eastern and Western Shores	Streams were sampled to document presence of Yellow Perch egg chains	98 yellow perch spawning stations	Maps in various sections of the report. Data were mapped by approximating the location using county ADC maps and stream files. Most stations were located at road crossings as indicated in report methods.	egg chains observed	visual observation and egg chain counts on standard segments of streams	Yellow Perch	

Table 2. Percentage of land area in Maryland by each category, calculated at the 8 digit watershed scale.

		Percent Impervious Land	
Management Category	Impervious Cover	1970	2018
Preferred	<5%	75.5	62.2
Acceptable	5-10%	9.2	10.6
Marginal	>10%	15.3	27.4

Table 3. Change between 1970 and 2018 in number of watersheds falling in spawning habitat categories due to development at three watershed scales. N Change = number of stations lost or gained and % Change expresses these as a percentage of 1970.

Category	1970	2018	N Change	% Change
		8-digit		
Preferred	21	15	-6	-28.6%
Acceptable	26	20	-6	-23.1%
Marginal	31	43	12	38.7%
		12-digit		
Preferred	36	32	-4	-11.1%
Acceptable	48	40	-8	-16.7%
Marginal	31	43	12	38.7%
		Catchment		
Preferred	191	170	-21	-11.0%
Acceptable	186	152	-34	-18.3%
Marginal	256	311	55	21.5%

Figure 1. Maryland MDE 8 Digit Watershed delineations.

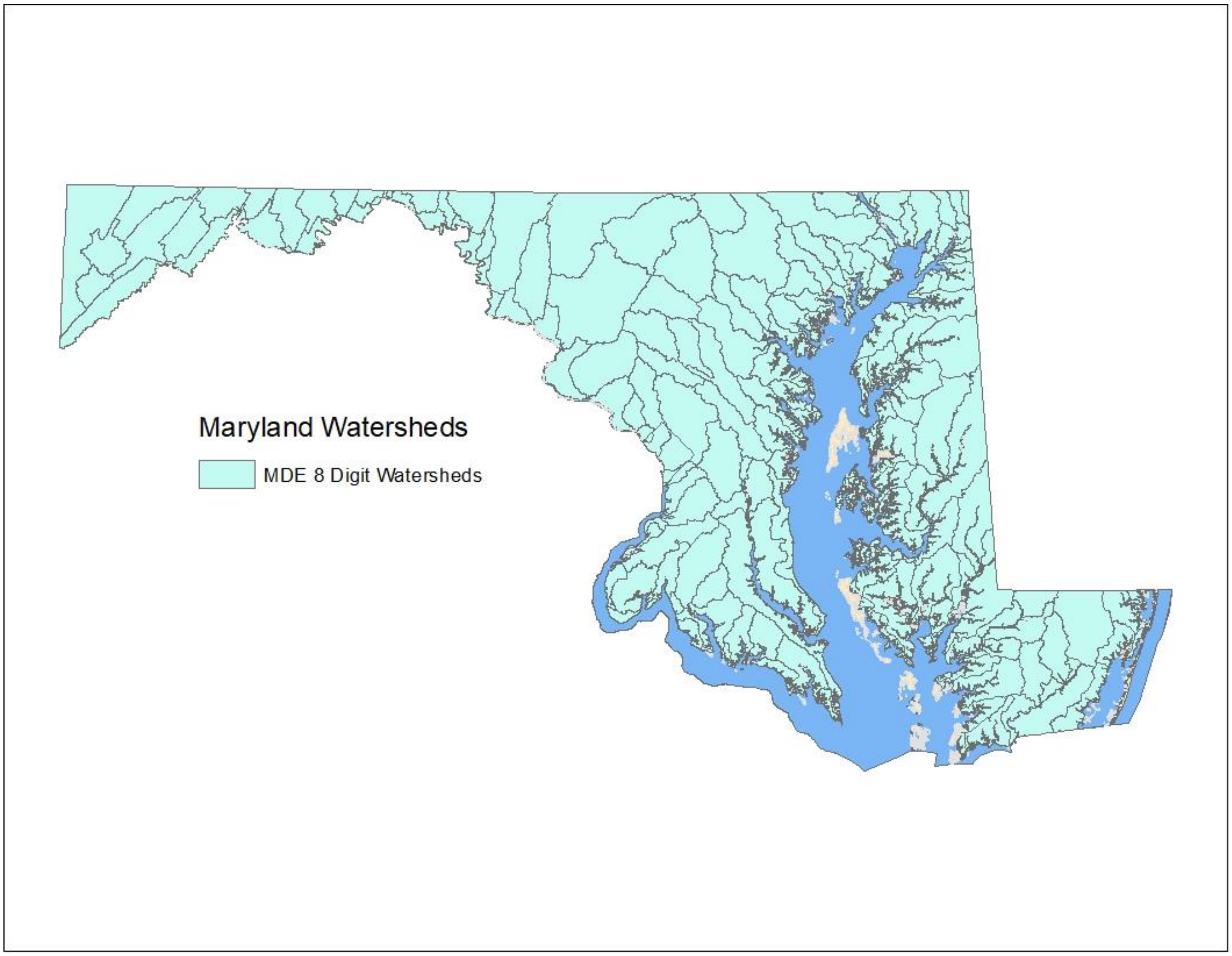


Figure 2. Maryland MDE 12 Digit Watershed delineations.

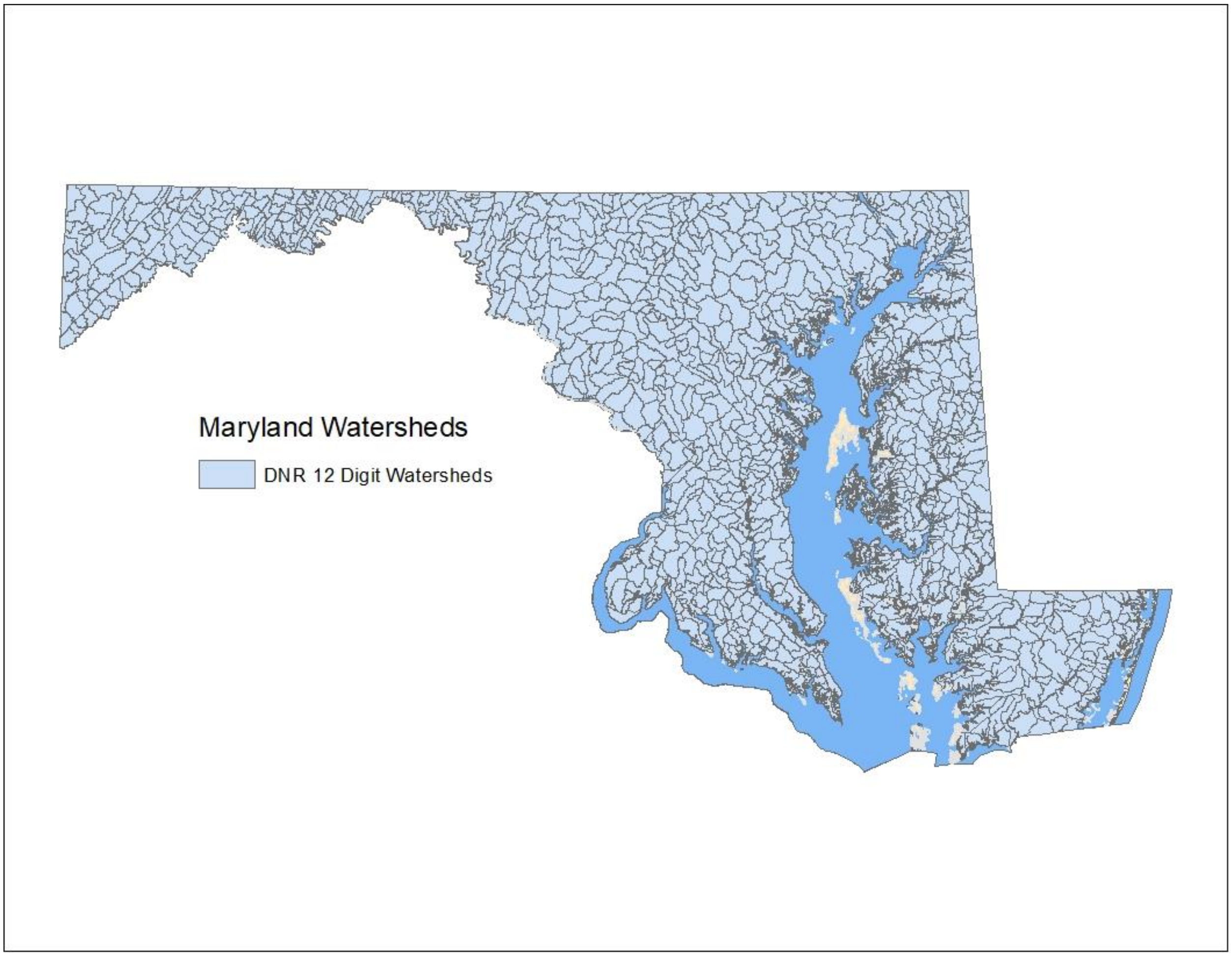


Figure 3. NHD Catchments in Maryland.

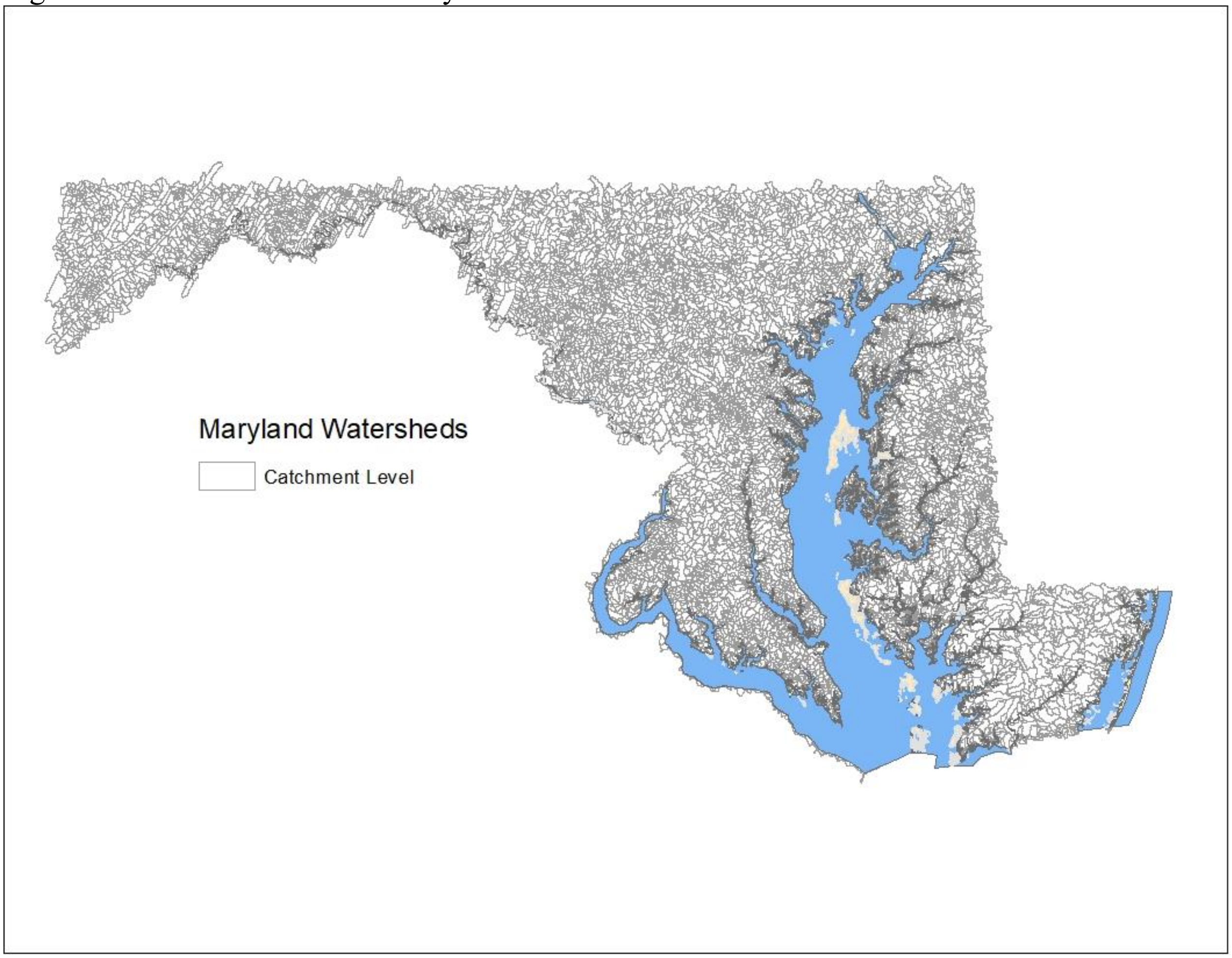


Figure 4. Stations evaluated for anadromous spawning presence in Maryland by study number (See Table 1 for study details).

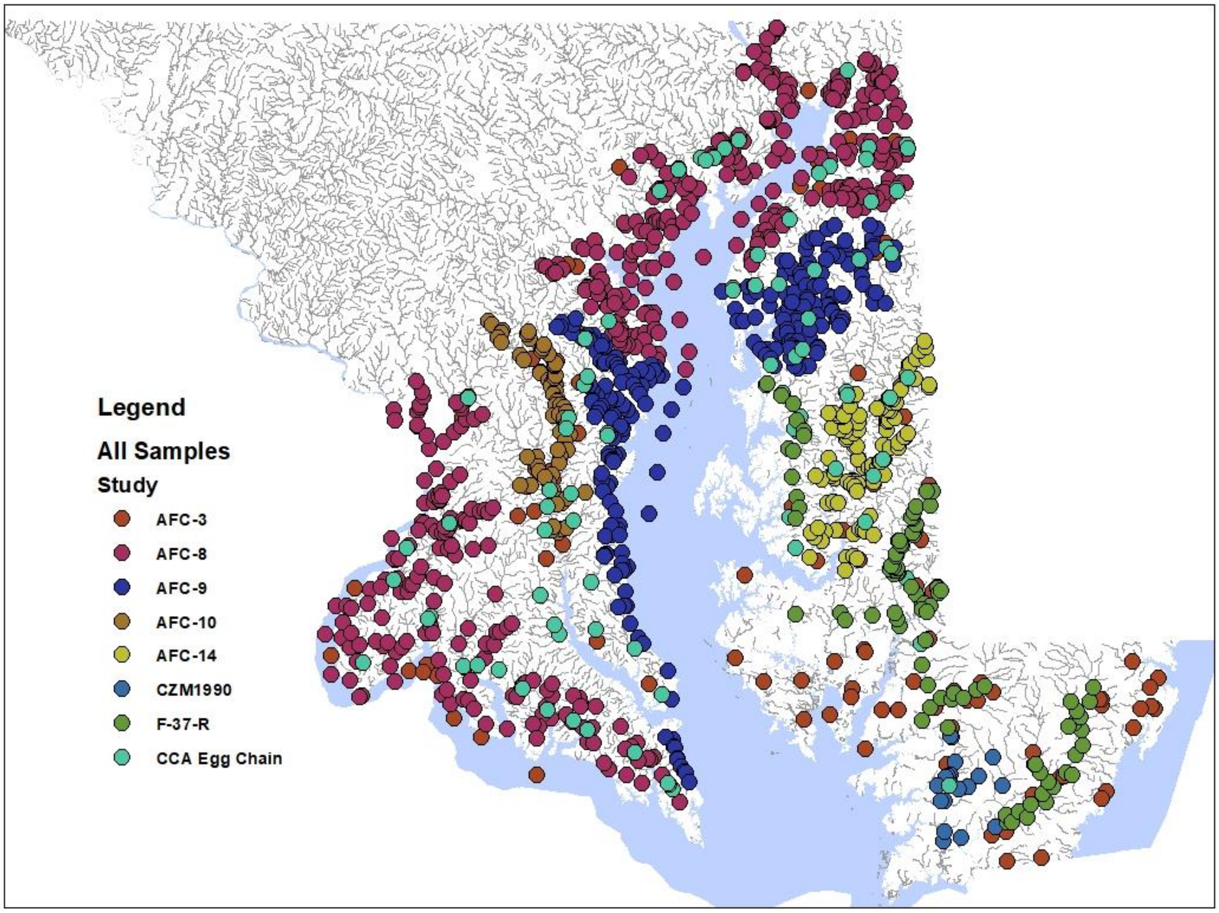


Figure 5. Spawning stations sampled with White Perch presence indicated based on presence of eggs, larvae or adults.

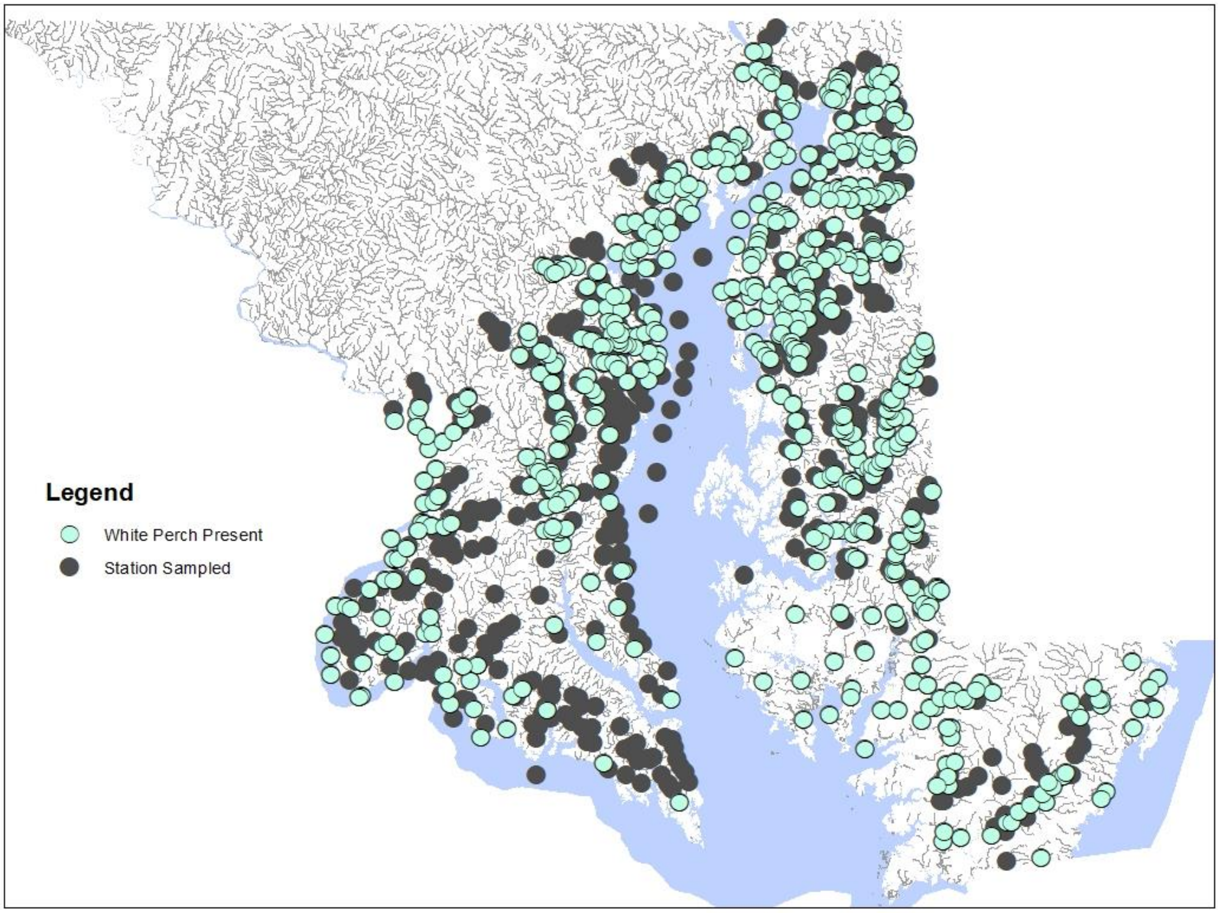


Figure 6. Spawning stations sampled with Herring and or Shad presence indicated based on presence of eggs, larvae or adults.

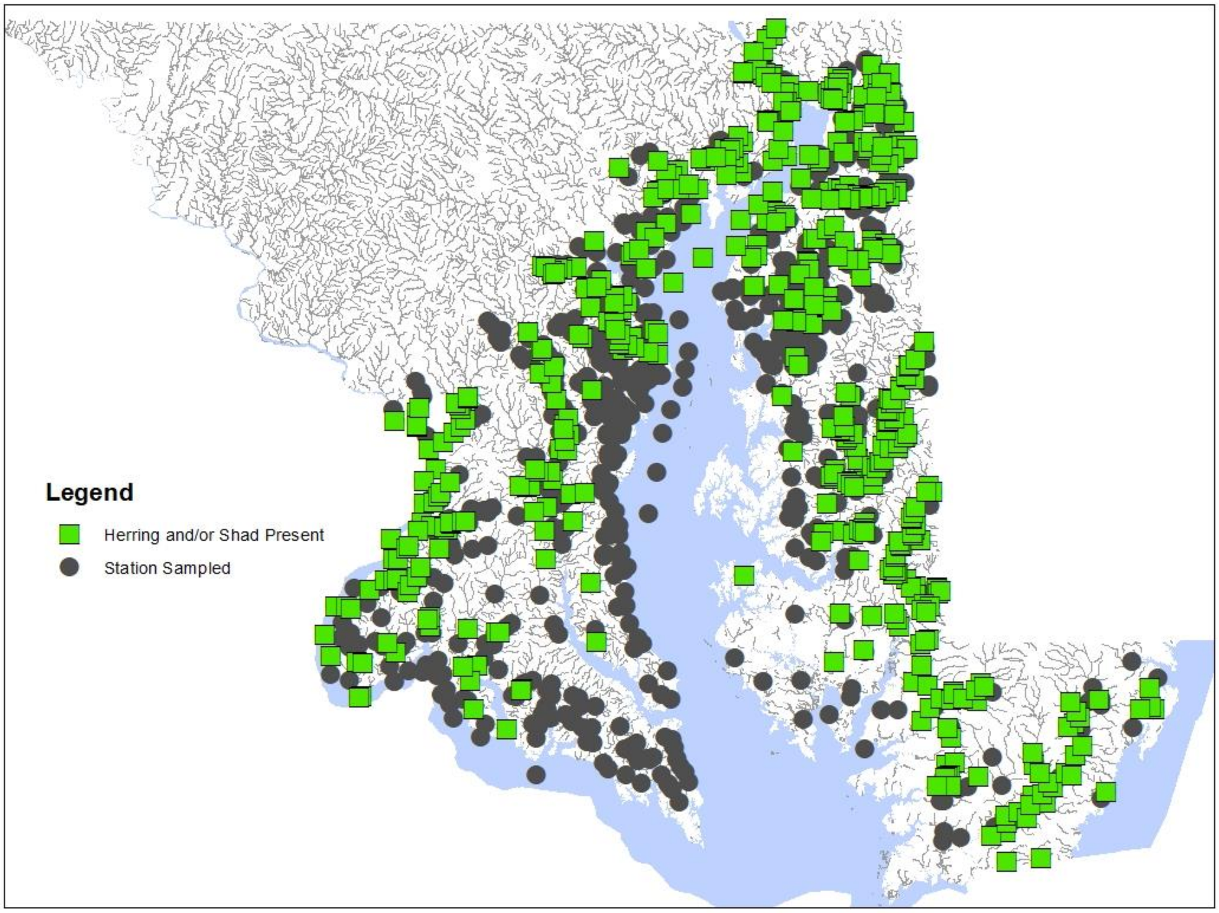


Figure 7. Spawning stations sampled with Yellow Perch presence indicated based on presence of eggs, larvae or adults.

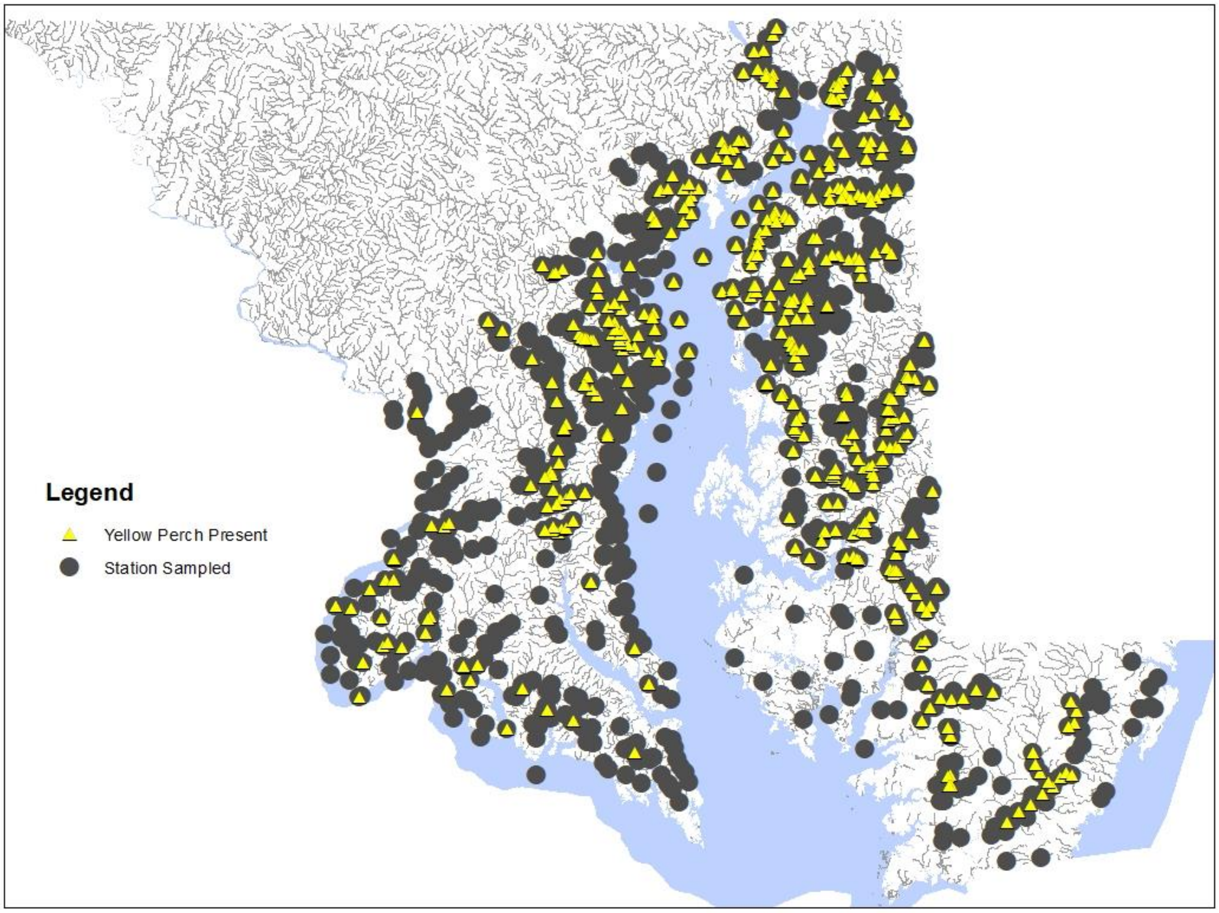


Figure 8. Spawning stations sampled with Striped Bass presence indicated based on presence of eggs, larvae or adults.

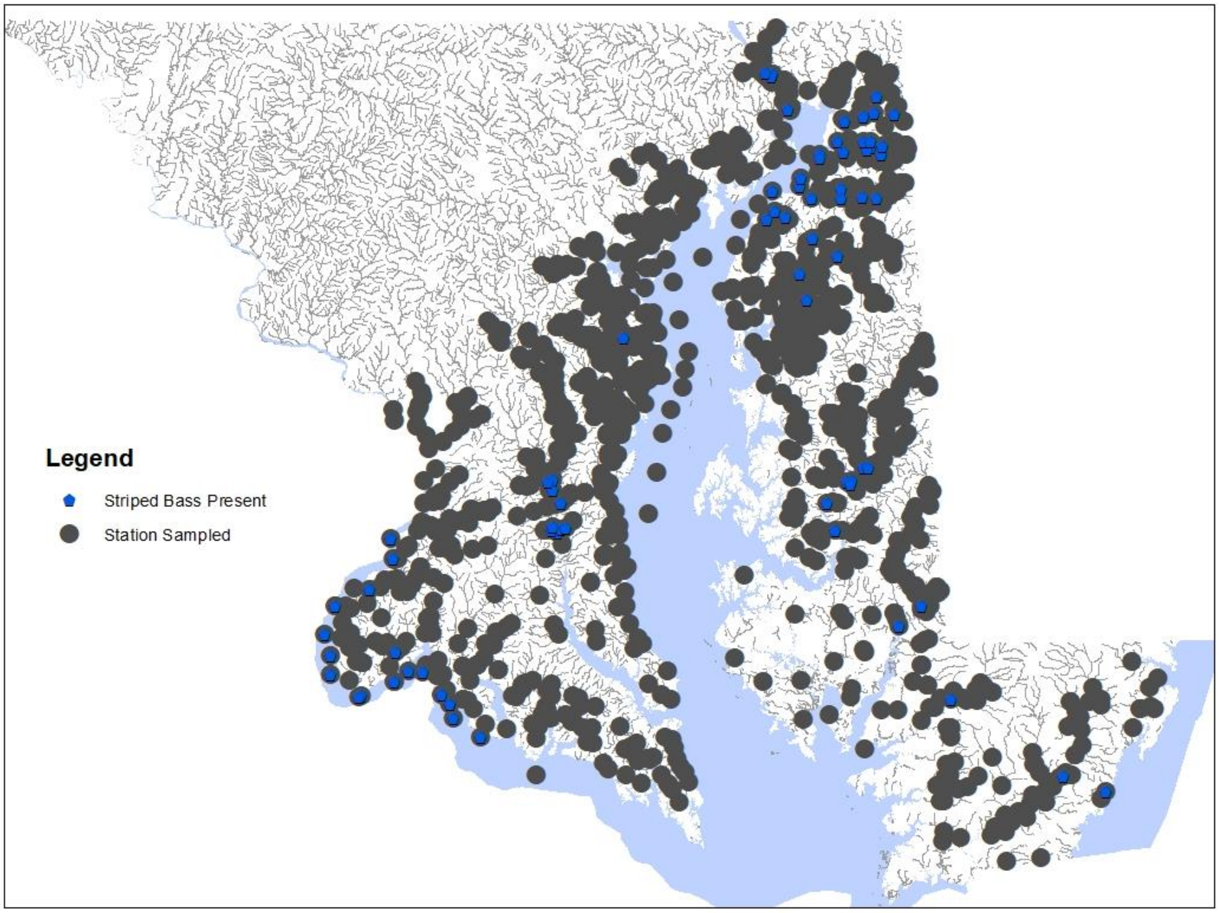


Figure 9. Anadromous spawning stations with presence for American Shad and Striped Bass eggs or larvae for all studies combined (1967-1990) and Striped Bass designated spawning areas (Hollis et al. 1967).

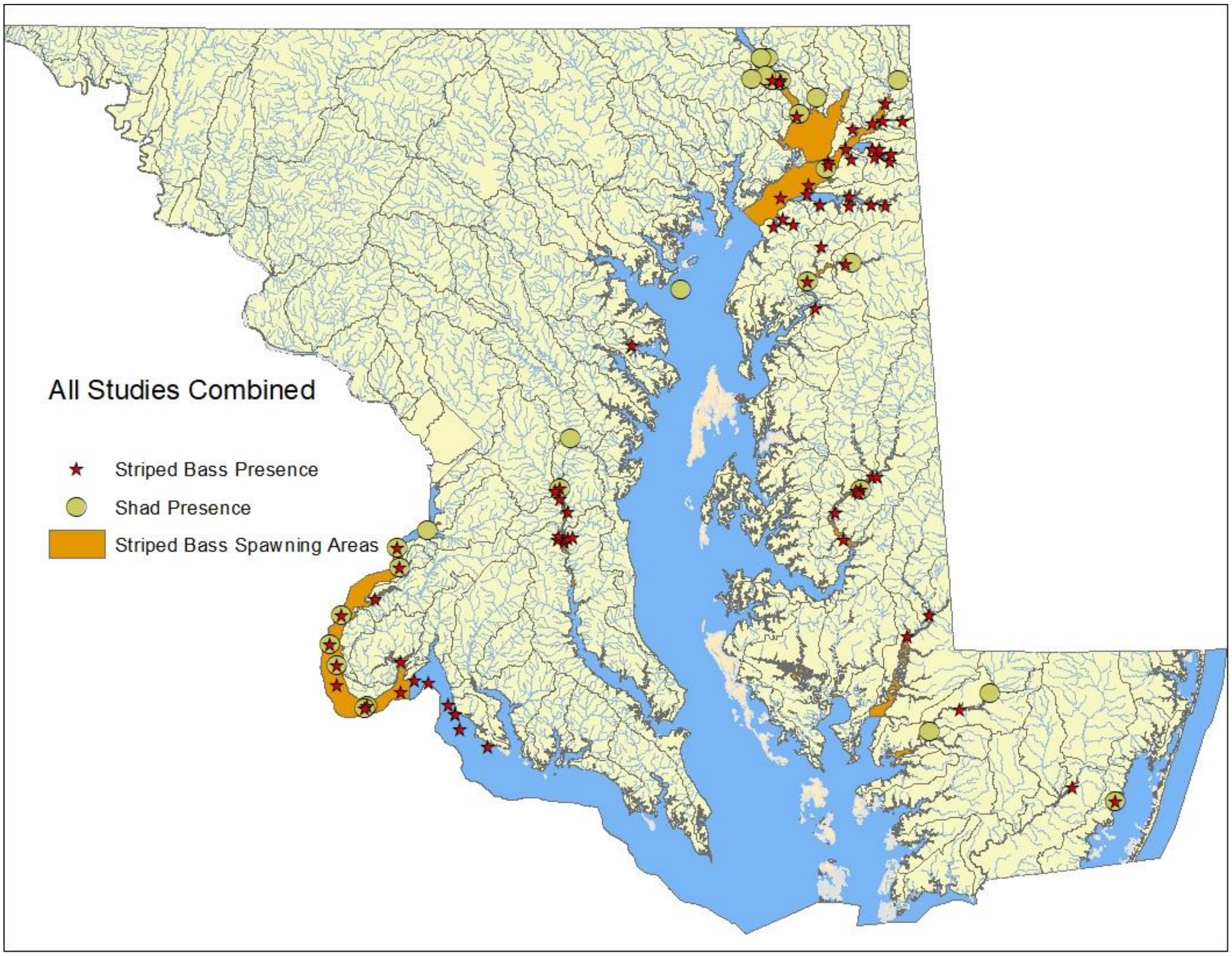


Figure 10. Spawning stations sampled where one or more anadromous species were present.

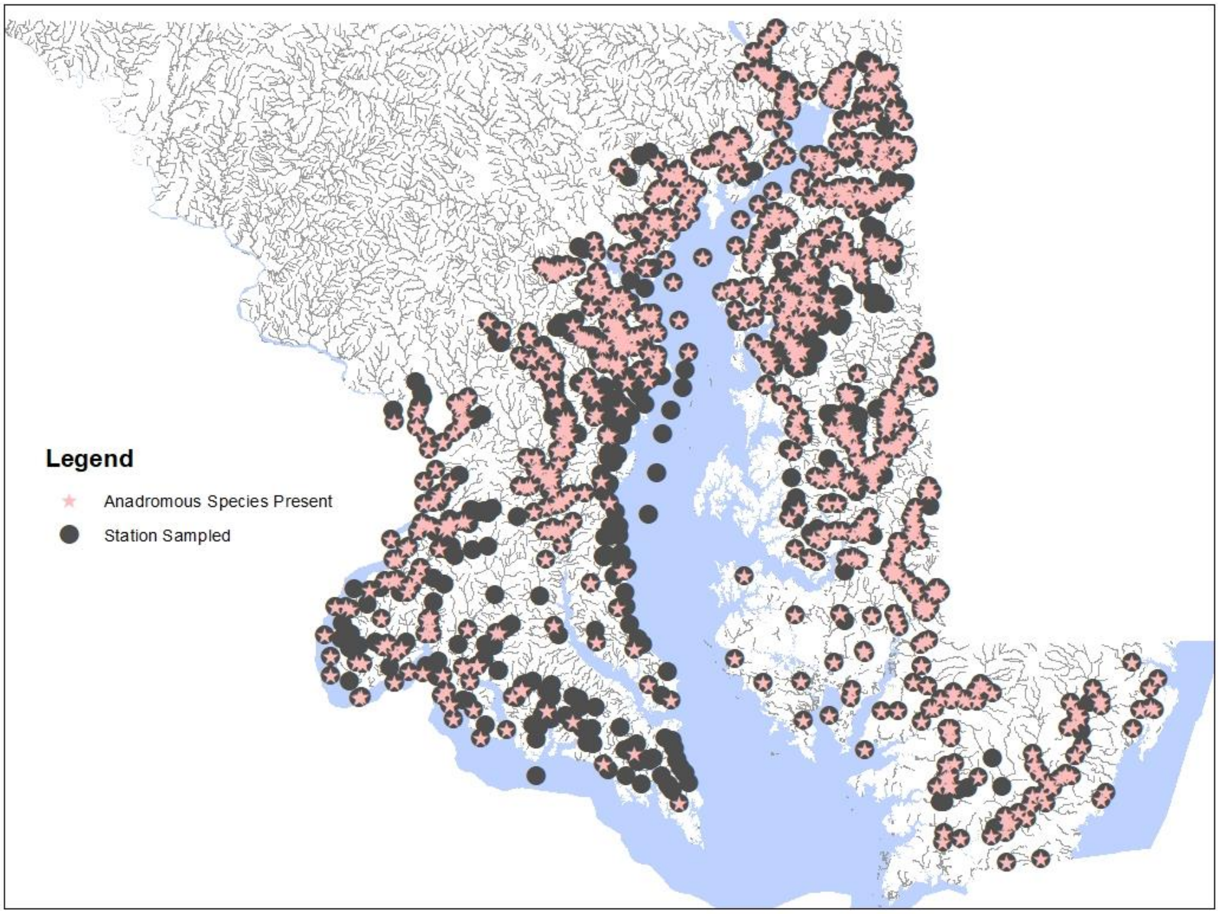


Figure 11. MDE8Digit watersheds sampled indicating presence and absence of anadromous species.

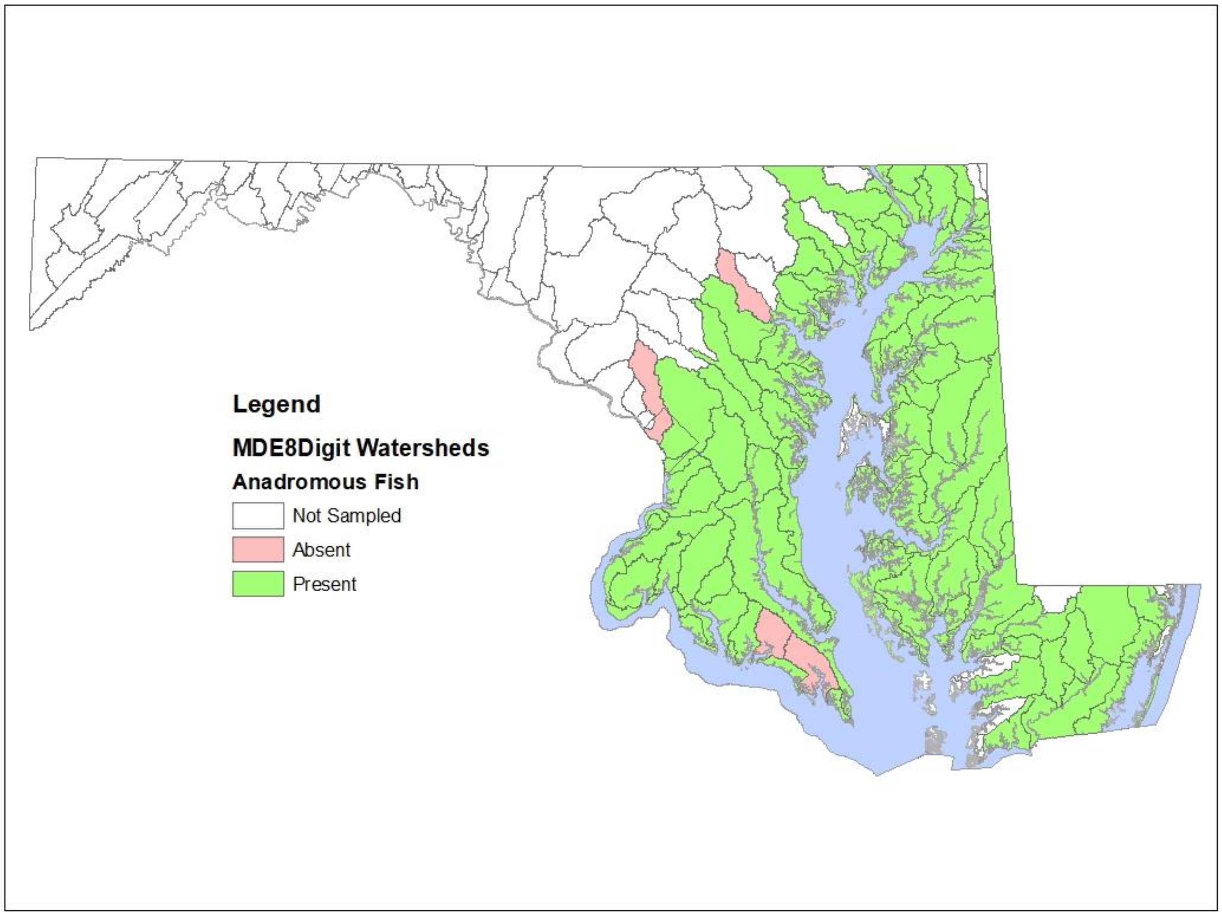


Figure 12. MDE12Digit watersheds sampled indicating presence and absence of anadromous species.

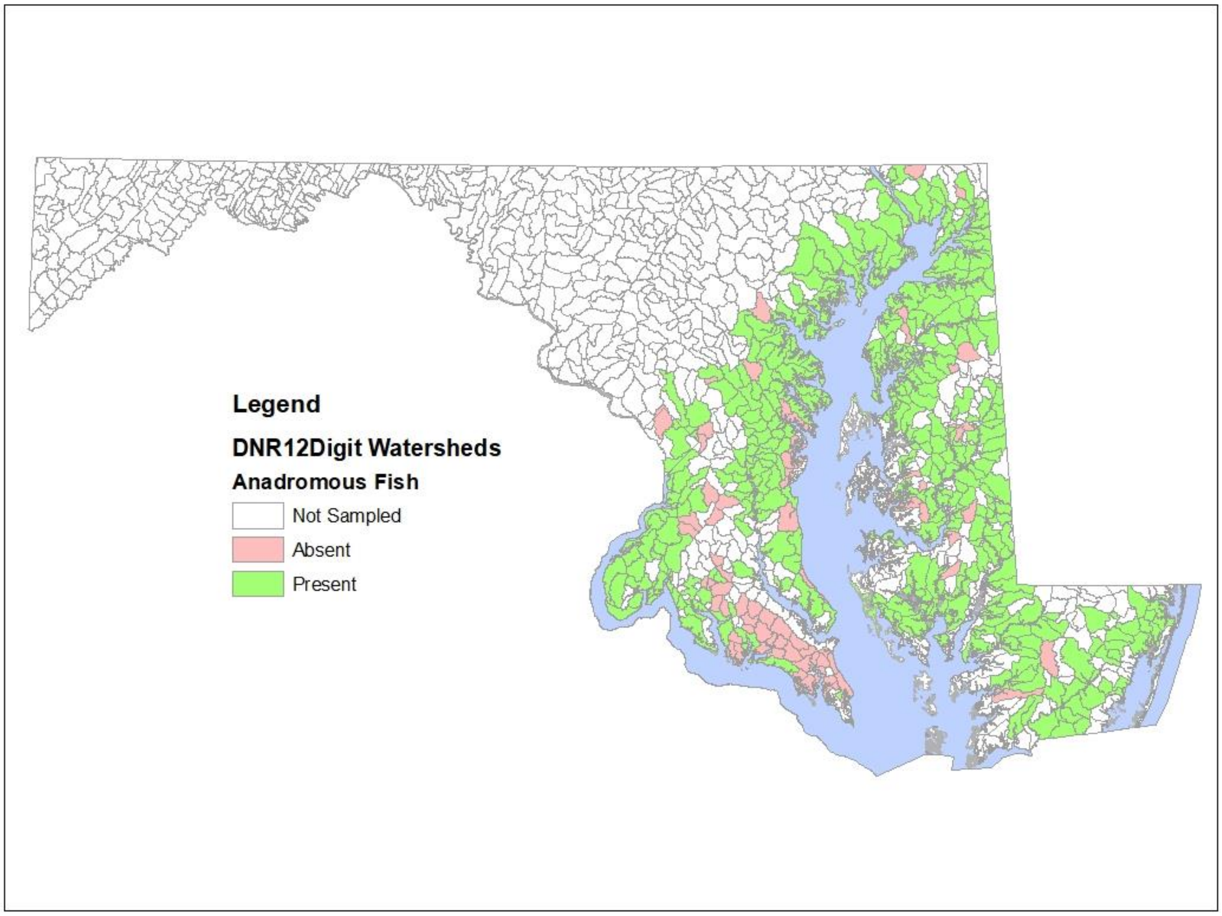


Figure 13. Catchments sampled indicating presence and absence of anadromous species.

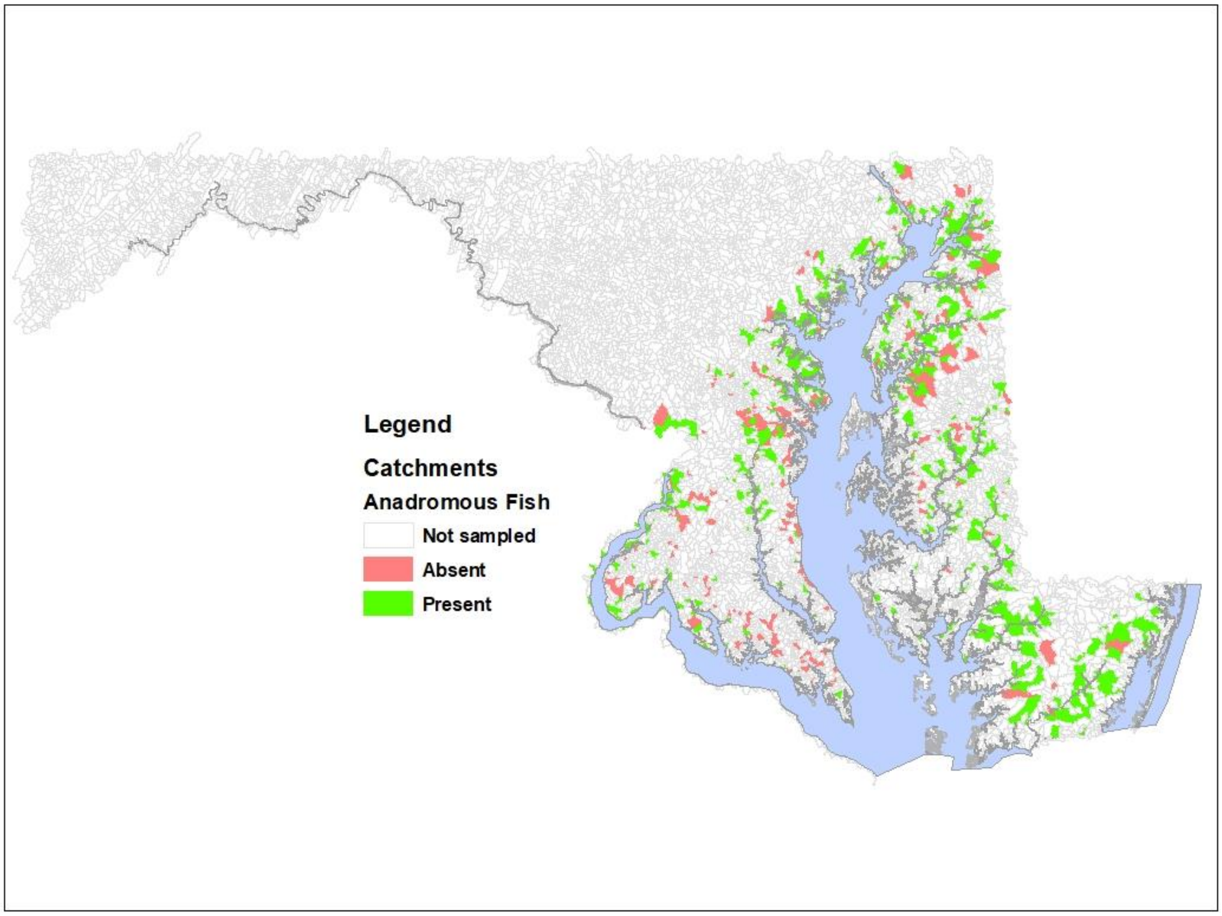
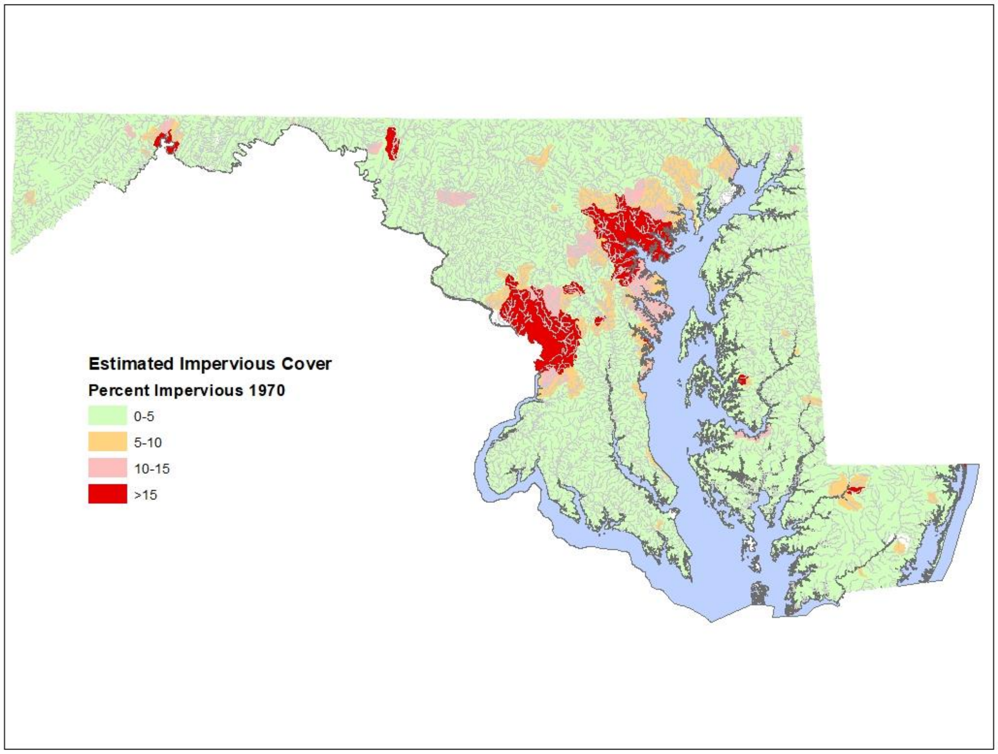


Figure 14. Change in percent impervious cover from 1970 to 2018 calculated at the MDE8 Digit watershed scale for Maryland to demonstrate the changes in impervious cover statewide.



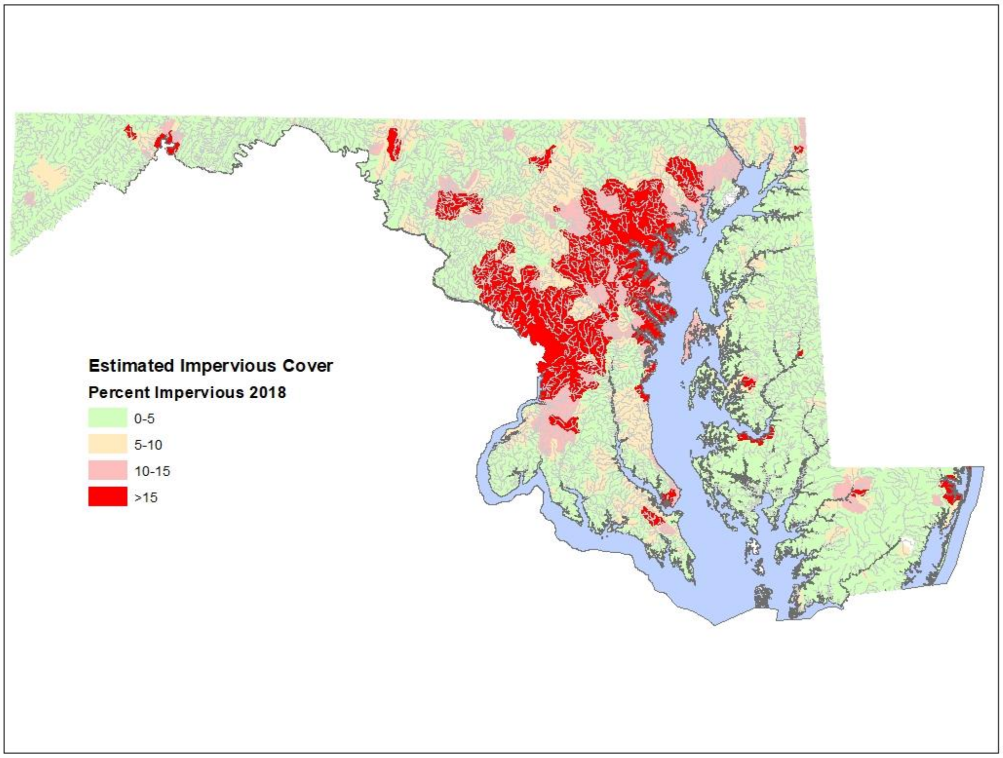


Figure 15. Extent of spawning habitat in preferred, acceptable, and marginal categories at the MDE8 Digit Watershed Scale during 1970 and 2018.

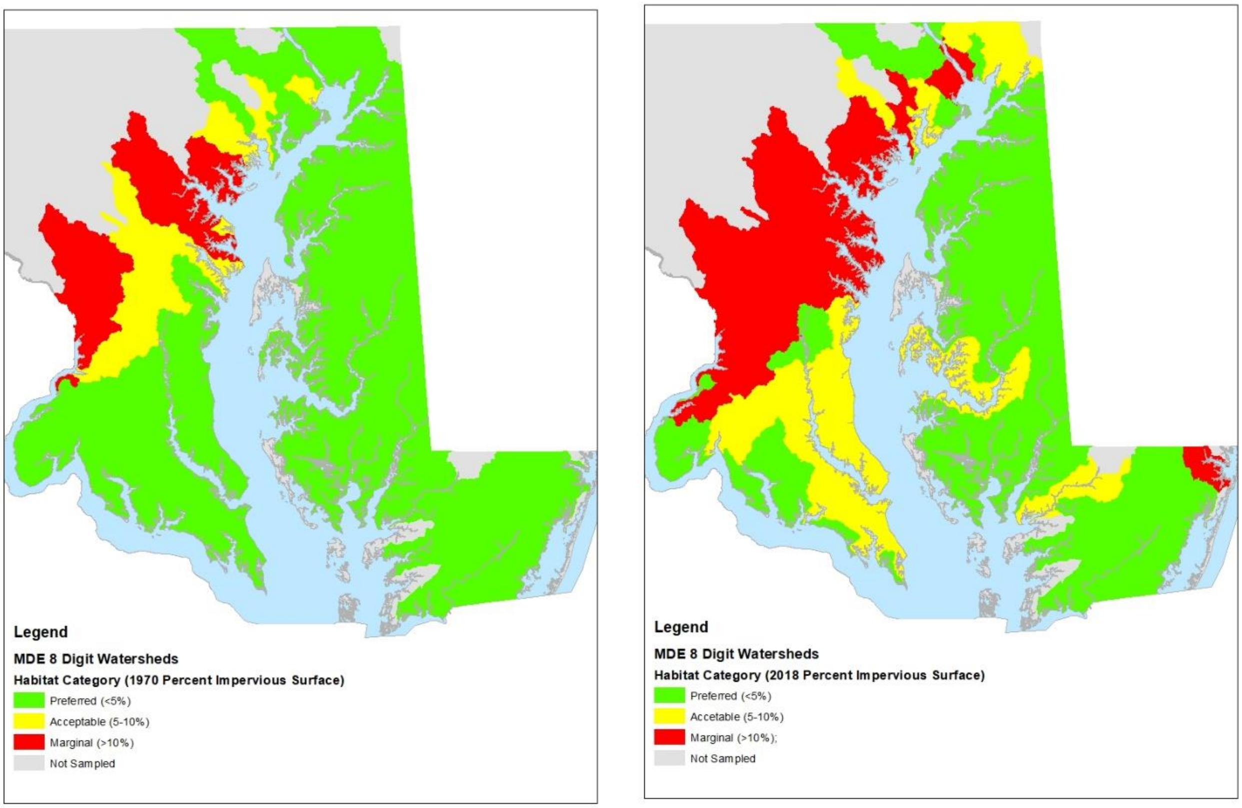
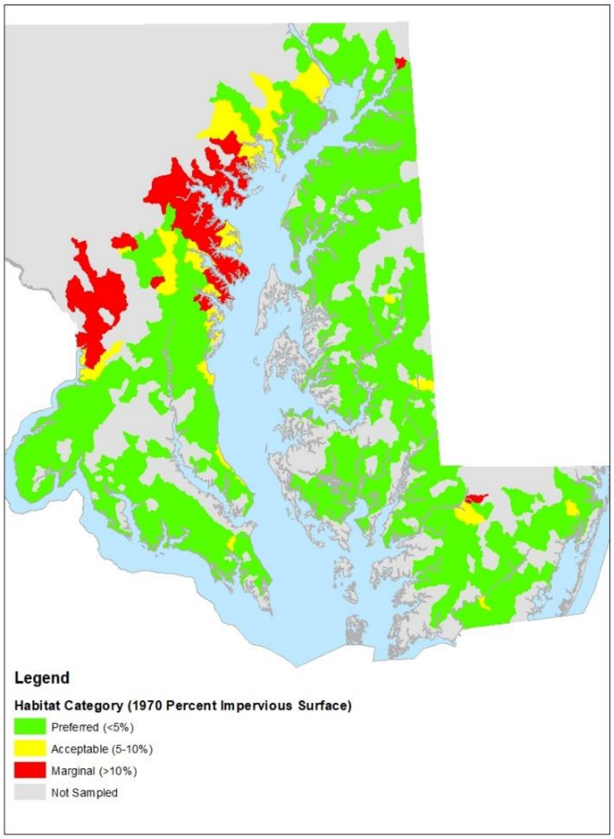


Figure 16. Extent of spawning habitat in preferred, acceptable, and marginal categories at the 12 Digit Watershed Scale during 1970 and 2018.



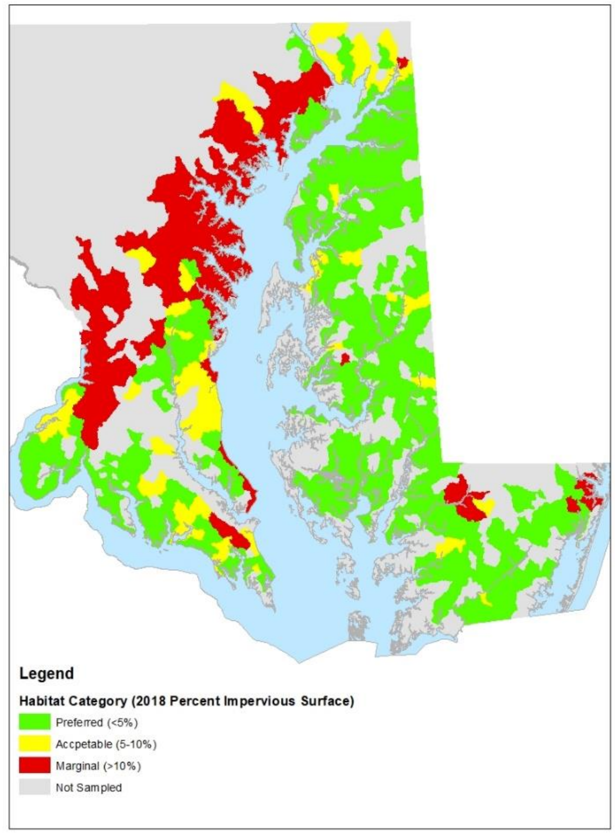


Figure 17. Extent of spawning habitat in preferred, acceptable, and marginal categories at the Catchment level during 1970 and 2018.

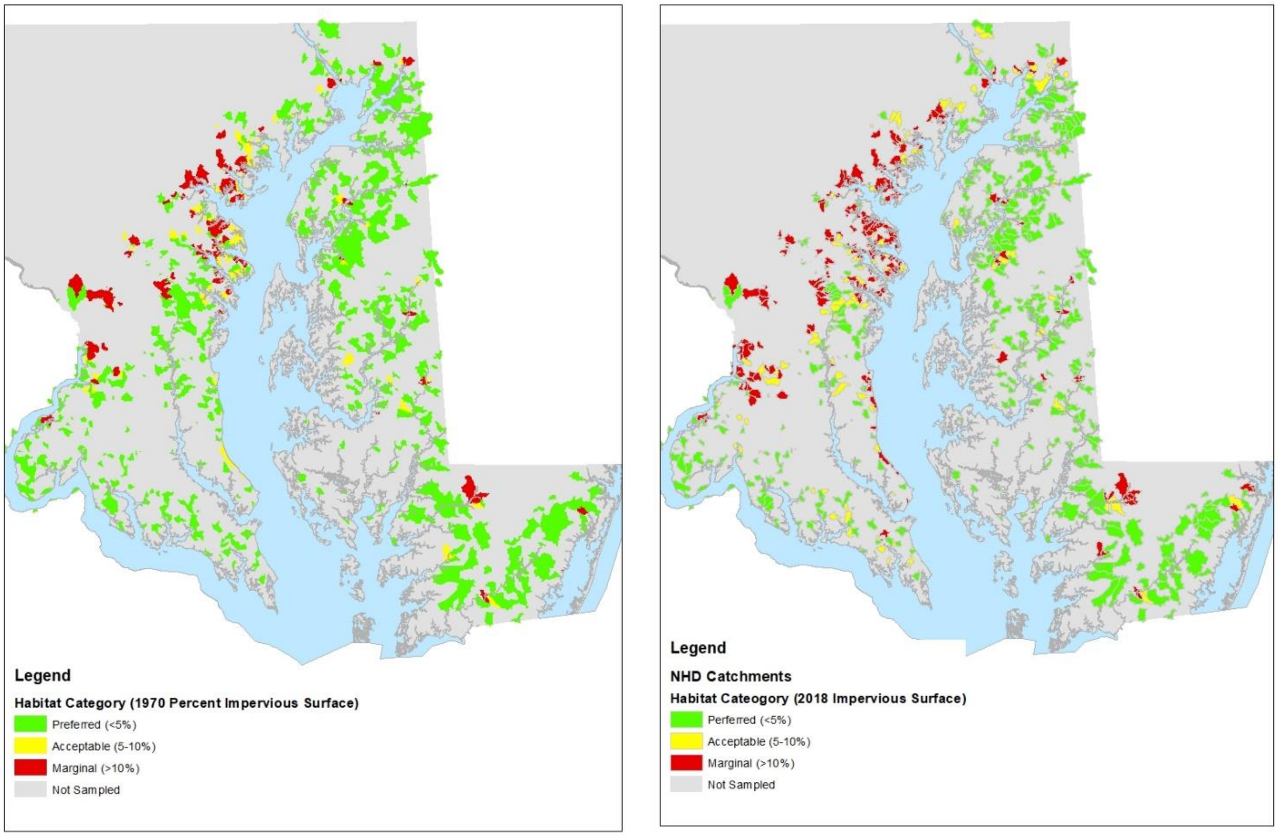


Figure 18. Habitat score by station plotted against watershed priorities at the MDE8Digit watershed scale.

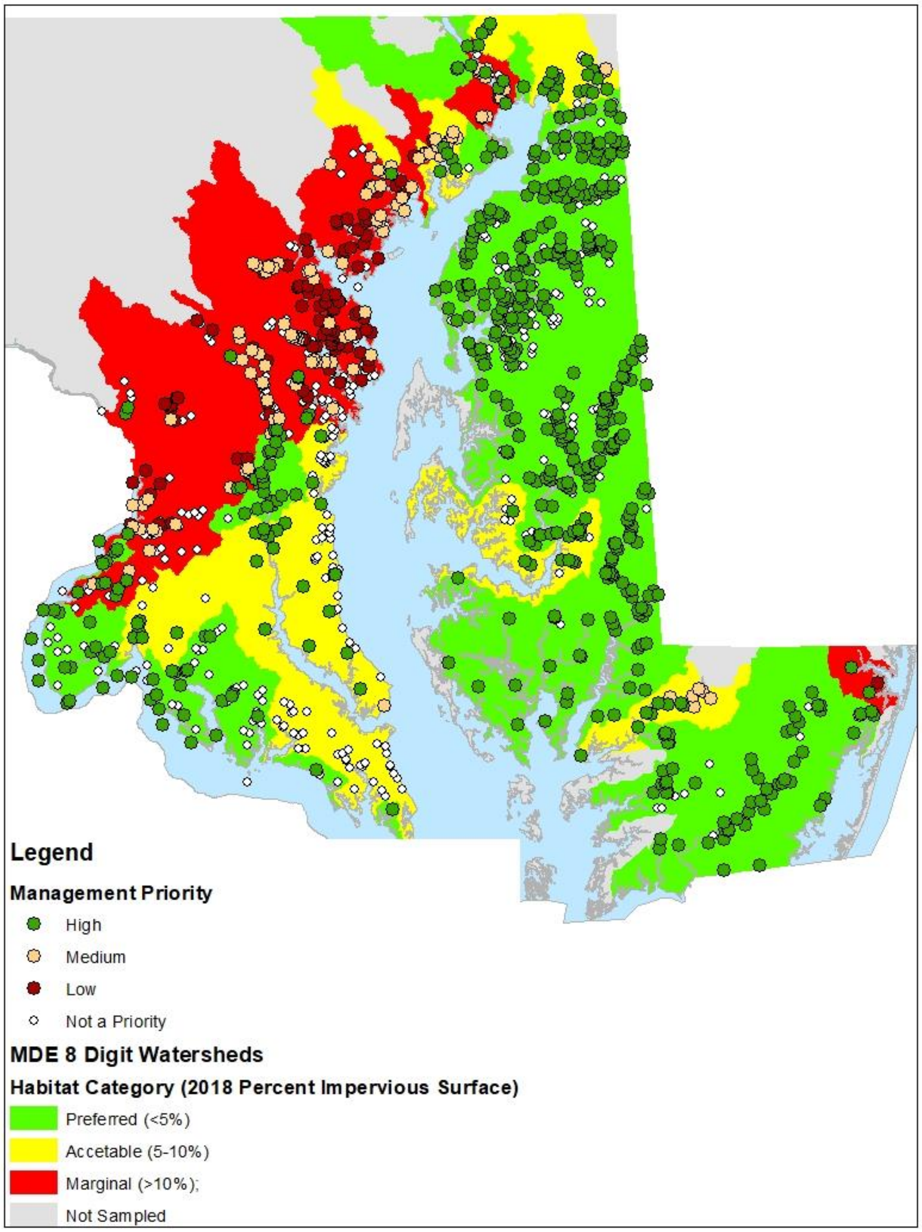


Figure 19. Anadromous spawning stations in the Patuxent River Watershed by management priorities at the three watershed scales based on 2018 impervious surface.

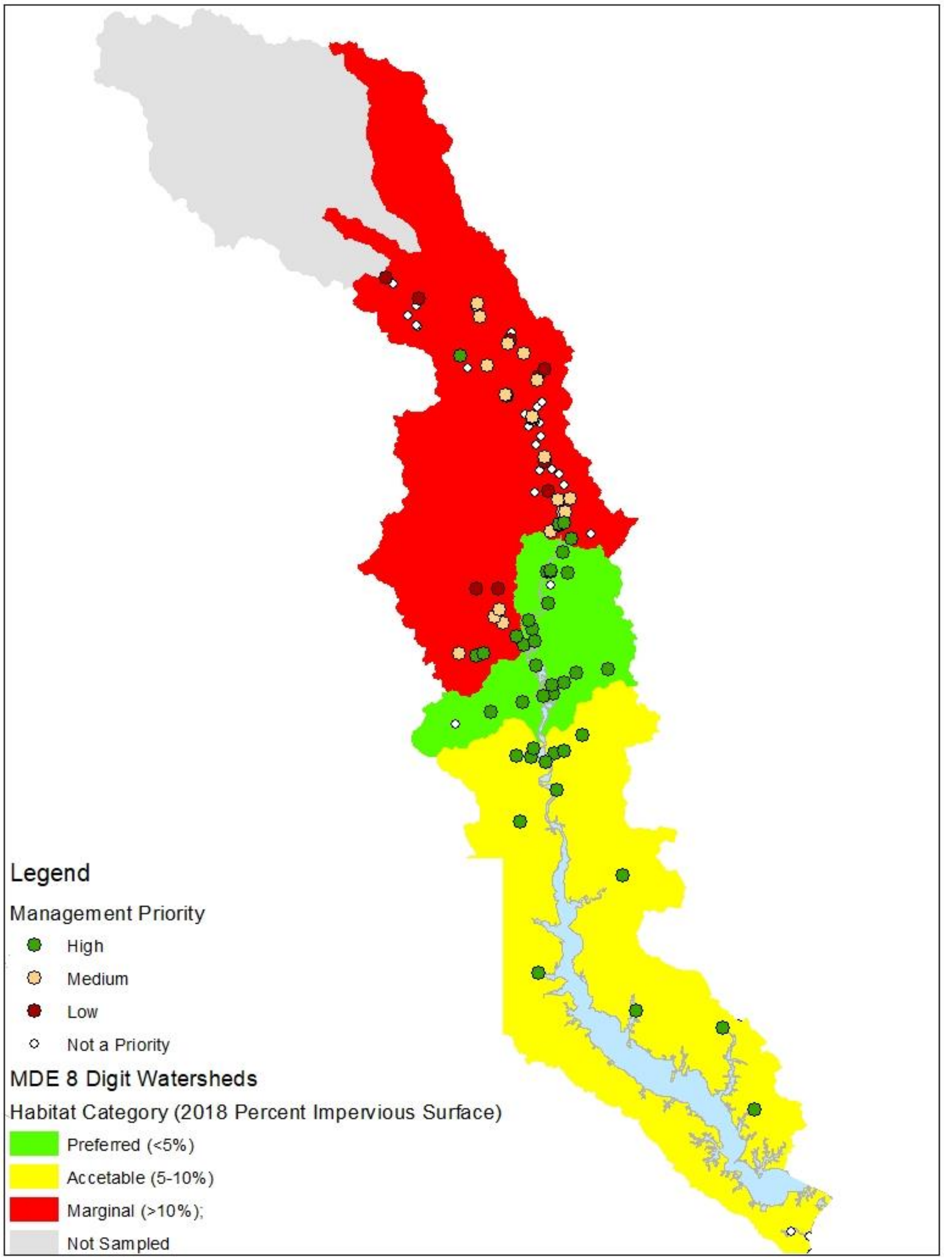


Figure 20. Anadromous spawning stations in the Patuxent River Watershed by management priorities at the MDE8Digit (a) and DNR12Digit (b) scales.

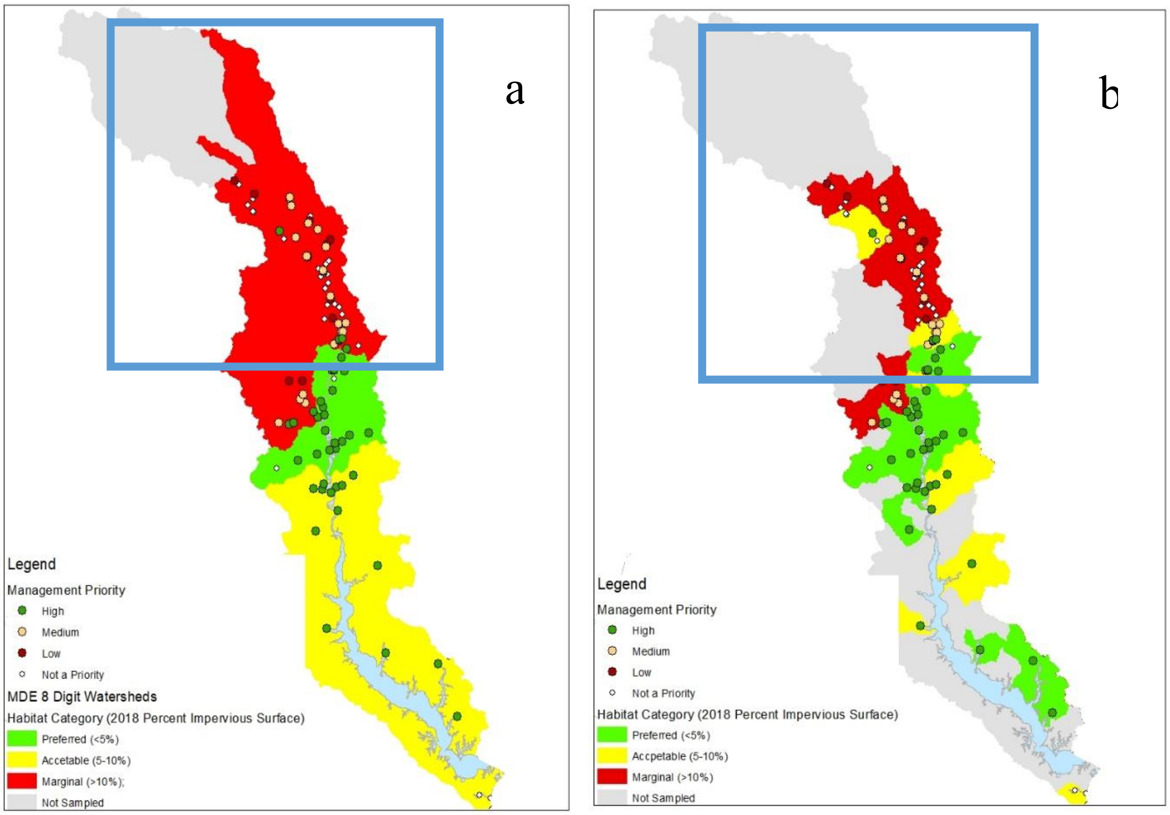


Figure 21. Anadromous spawning stations in the Patuxent River Watershed by management priorities at the DNR12Digit (a) and Catchment (b) scales.

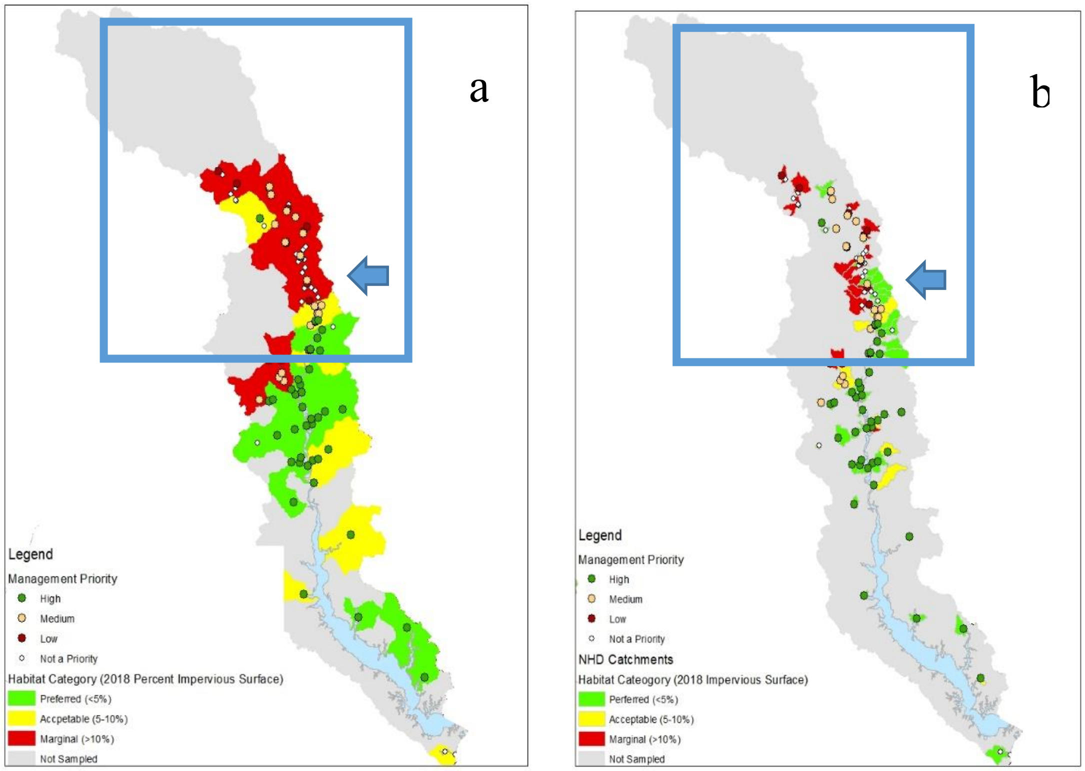


Figure 22. Anadromous spawning stations by management priorities at the Catchment level in the Patuxent River with the Anne Arundel County boundary indicated by the light blue line.

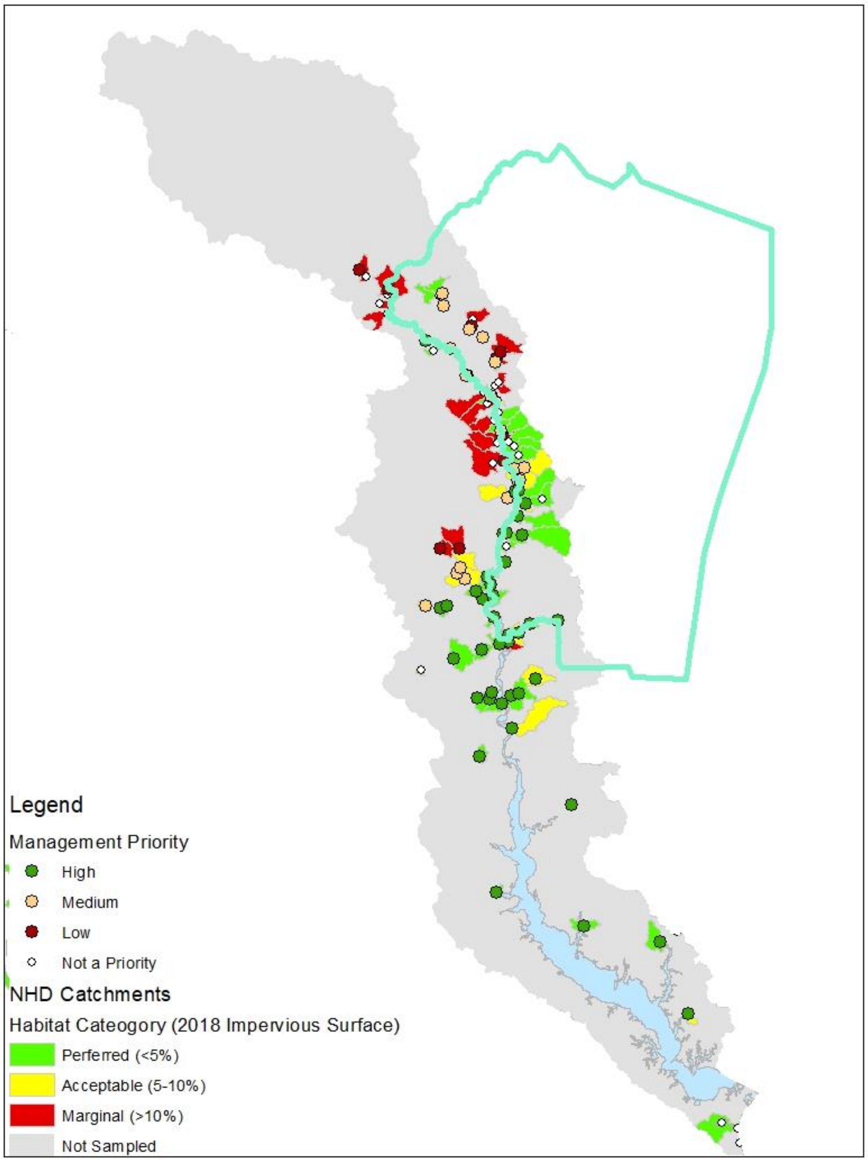
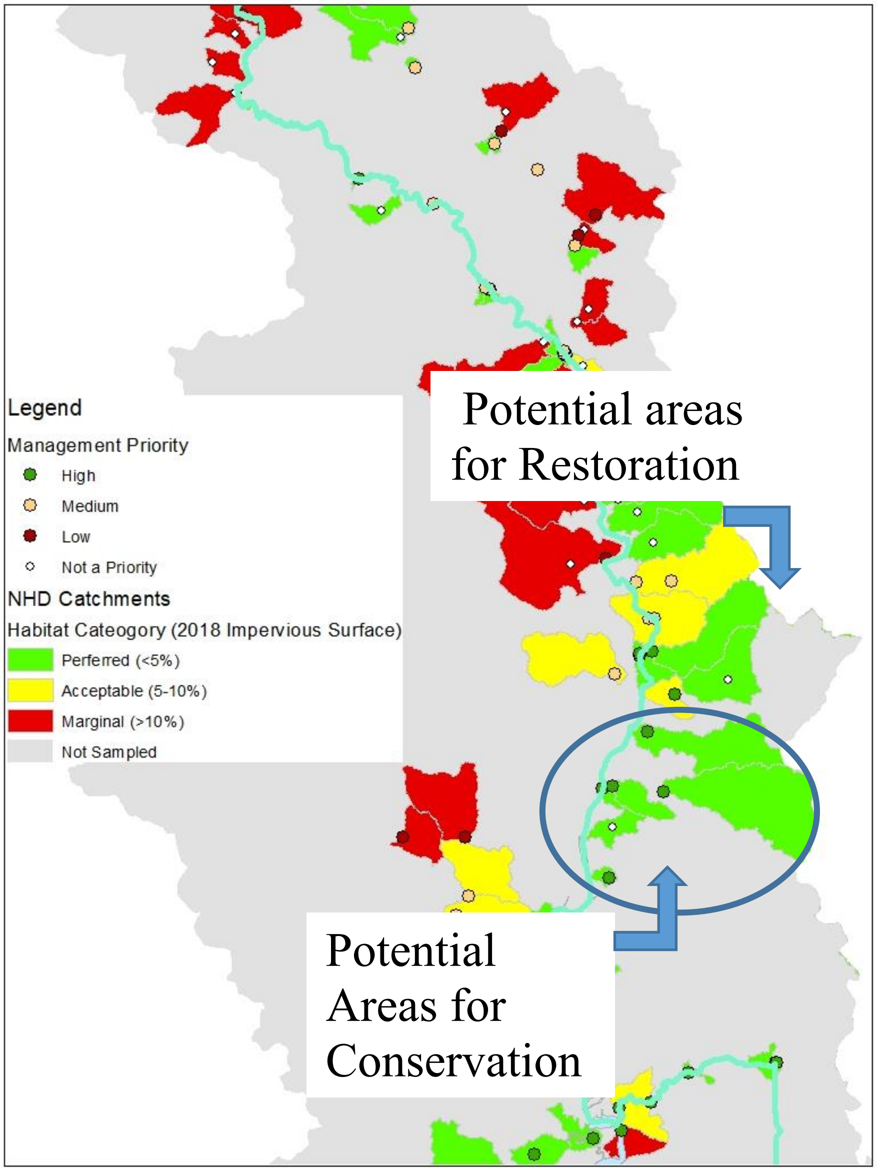


Figure 23. Anadromous spawning stations by management priorities at the Catchment level in the Patuxent River magnified to demonstrate opportunity for management in Anne Arundel County.



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

Jim Uphoff, Alexis Park, and Carrie Hoover

Executive Summary

An index-based (Index of Forage or IF) approach was developed to 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.

Indices of Striped Bass health (1998-2019), relative abundance (1983-2019), natural mortality (1986-2019), and forage relative abundance in surveys (1959-2019) and fall diets of Striped Bass (1998-2000 and 2006-2019) provided metrics (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 (≥ 457 mm TL), due to sampling considerations. The small class was most sensitive to forage and indicators were mostly based on it.

Targets and thresholds were then developed for each of these indicators to assign them scores. A score of 1 indicated threshold (poorest) conditions; a score of 3 indicated target (best) conditions; and a score of 2 indicated conditions between. Time-periods where body fat indicators 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 index of forage 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. Index of forage scores were elevated beyond the threshold after 2004. Index of forage scores during 2008-2011 were near or at the target (best foraging conditions), then fell into an intermediate region (1.4-2.4). It has been near 2.0 (does not breach threshold or target) during 2017-2019, indicating some recovery from poorer foraging conditions during 2015-2016.

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 lower 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. Striped Bass returned to noticeably higher abundance after 2014, but major forage did not; however, condition has not declined to threshold conditions. It appears that slight increases in Atlantic Menhaden relative abundance, while not statistically significant, 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 during 2006-2019. Condition (proportion of Striped Bass without body fat) of small and large Striped Bass diverged recently (2016 and 2018-2019), improving for large fish and worsening for small ones. The ratio of larger major prey (Spot and Atlantic Menhaden) length to Striped Bass length for small fish has been consistently high since 2015. Small Striped Bass would have more difficulty in catching and handling these larger prey than large Striped Bass in any given year.

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; see Job 1, Section 2.1). 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 (Richards and Rago 1999).

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 saltwater 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; 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 annual instantaneous fishing mortality rates (F) remained low (Jiang et al. 2007; ASMFC 2013; NEFSC 2019). Prevalence of mycobacteriosis and high M appear to be less outside Chesapeake Bay (Matsche et al. 2010; NEFSC 2019), but abundance, condition, and M of the coastal migration contingent appears linked to ages 1+ Atlantic Menhaden (Buchheister et al. 2017; Uphoff and Sharov 2018; ASMFC 2020a).

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) has adopted ecological (forage) reference points for Atlantic Menhaden's forage role along the Atlantic coast and Striped Bass is a predator of concern because of its sensitivity to Atlantic Menhaden population size (SEDAR 2015; ASMFC 2020a; 2020b). 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."

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 for Striped Bass in Chesapeake Bay (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 approach used here (an integrated index of forage or IF) was based on a suite of five indicators (metrics) that could be inexpensively and easily developed from existing MD DNR sampling programs. In addition to providing a basis for judging whether the forage base is adequate to support Striped Bass in Maryland's portion of Chesapeake Bay, two additional objectives of the IF were low cost and tractability for available staff. Proportion of resident Striped Bass in fall without visible body fat (P0) and an index of survival (SR) reflecting natural mortality of pre-recruits were indicators of Striped Bass well-being. A Striped Bass recreational catch per trip index (RI) provided an index of relative predator abundance (demand). Forage-to-Striped Bass ratios for major forage species (FR) and proportion of empty guts (PE) in fall provided trends in supply relative to demand based on relative abundance and diet sampling, respectively.

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; Buchheister and Houde 2016). We selected these species as focal prey (major prey) for forage indices. Forage ratios (FR) of species-specific indices of major prey relative abundance from fishery-independent surveys to RI were examined for each focal prey. Forage species 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).

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).

Proportion of empty guts (PE) was used as a consumption based indicator of prey availability. A consumption indicator based on weight consumed per weight of Striped Bass that consumed them (C) and its species composition, and composition of prey consumed by number were useful for interpreting PE.

The ratio of age-3 relative abundance of male Striped Bass in spring spawning ground gill net surveys (Versak 2019) 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. We expected SR to vary without trend if natural mortality (M) remained constant.

Forage 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 were at target or threshold levels 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).

This report provides a complete set of indicators through 2019. All were summarized into a single score to serve as a quick reference for managers and the public.

Methods

Abbreviations - Definitions of abbreviations can be found in Table 1.

Striped Bass condition, feeding success, and diet composition indices – Indicators of condition, feeding success, and diet composition during October-November were developed from Striped Bass caught by hook-and-line. A citizen-science based Striped Bass diet monitoring program was conducted by Chesapeake Bay Ecological Foundation (CBEF) during 2006-2015 and 2006-2013 collections were used to estimate feeding success and diet composition. Diet samples from a Fish and Wildlife Health Program (FWHP) Striped Bass health survey were used after 2013. Methods for CBEF and FWHP collections have been described in Uphoff et al. (2014; 2015; 2016) and will be briefly repeated below.

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 when the permit was issued) and 15 fish 457 mm TL or larger (large Striped Bass or fish) per trip during 2006-2014. 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).

Striped Bass diet collections by CBEF 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). 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 upon return to shore or held on ice for processing the next day. Collections of large sized Striped Bass were supplemented by sampling charter boat hook-and-line catches at a fish cleaning business. 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.

During 2014-2019, Striped Bass collected for health samples by Fish and Wildlife Health Program (FWHP) were processed by Fish Habitat and Ecosystem Program personnel for diet information. Collections by FWHP were not constrained by collector's permit conditions like CBEF collections. Fish were collected by hook-and-line from varying locations during fall, 1998-2019, between Baltimore, Maryland (northern boundary) and the Maryland-Virginia state line (southern boundary; Figure 1). 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). Bait was not included in diet data.

Condition indices were estimated from an existing Striped Bass health survey (FWHP) that began in 1998. Nutritional status (condition) for upper Bay Striped Bass was estimated as the proportion of fish without visible body fat during October-November in FWHP samples (P0; Jacobs et al. 2013). Estimates of P0 were made for the two size classes of Striped Bass separately and combined. Estimates of P0 for 1998–2013 were provided by FWHP and remaining years were estimated from FWHP data by FHEP. 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, a period of consistently poor condition). Other indicators of condition were described in Jacobs et al. (2013), but P0 was chosen because it could be applied to data collected by CBEF; P0 estimates from CBEF collections were similar to those estimated for FWHP collections for years in common (Uphoff et al. 2018).

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.

Collections by CBEF were processed by James Price and identification of items were aided (when needed) by J. Uphoff and Joseph Boone (a retired MD DNR fisheries biologist). Guts were removed from the Striped Bass and emptied. Total length of intact fish and shrimp, carapace width of crabs, and shell length of intact bivalves were measured; some food items were weighed with a calibrated digital scale. Non-linear allometry equations for converting diet item length to weight (Hartman and Brandt 1995a) were used for items that were only measured. In a few cases, equations for a similar species were substituted when an equation was not available. These equations, originally developed and used by Hartman and Brandt (1995a), had been used to reconstruct diets for Overton et al. (2009) and Griffin and Margraf (2003).

Fish Habitat and Ecosystem Program staff identified, measured, and weighed diet items from FWHP sampling (2014 to present) as FWHP staff processed Striped Bass in the lab. All organisms were blotted as dry as possible before weighing. Three broad data categories of diet data were formed for processing. The first category was composed of fish and invertebrates where information from individual organisms was desired. Lengths (TL for fish, CW or carapace width for crabs, and maximum length of shell for intact bivalves) and weights were measured. Bay Anchovy were a special case since Striped Bass sometimes consumed large numbers. Up to ten Bay Anchovies were measured and weighed per Striped Bass and the remainder were weighed together. Total weight of partially intact fish in a gut was recorded. The second category were data from larger invertebrates that may be present as whole individuals or identifiable with inspection as parts. If these items were in good condition, they were recorded as counts and individual lengths and mass recorded with the same procedure as Bay Anchovy. Otherwise, a count and combined mass were recorded. In some cases, it was only possible to record that these organisms were present (lots of parts, not many whole). The third category was soft invertebrates such as amphipods or polychaetes that were likely to be pretty broken up or digested. Presence was the only numerical descriptor possible. Empirical relationships developed by Stobberup et al. (2009) for general taxonomic categories were used to estimate relative weight from frequency of occurrence of these soft invertebrates. These soft items were uncommon in our fall collections, but were more common during other seasons (J. Uphoff, personal observation).

Diets were analyzed separately for small and large Striped Bass for both CBEF and FWHP collections. 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 very 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 in numbers during 2006-2019 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 (Hyslop 1980). Standard deviations and 90% CI's of PE were estimated using the normal distribution approximation of the binomial distribution (Ott 1977). Estimates of PE from Overton et al. (2009) were available to estimate threshold conditions during 1998-2000 (Uphoff et al. 2017). In addition, this indicator could be derived from published diet information from the 1930s (Hollis 1952) and the 1950s (Griffin and Margraf 2003).

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 of large major prey 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.

Relative abundance indices of prey and Striped Bass - 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 since 1959 provided indices for Atlantic Menhaden, Bay Anchovy, and Spot (Durell and Weedon 2019). Additional indices for Spot and Bay Anchovy since 1989 were estimated from a Blue Crab trawl survey conducted during summer (Uphoff 1998; Rickabaugh and Messer 2019; MD DNR 2020a; the most current 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 May-October.

Density of juvenile Blue Crabs in a stratified random winter dredge survey that has sampled Chesapeake Bay-wide (Maryland and Virginia) since 1989 was our indicator of Blue Crab relative abundance (Sharov et al. 2003; Jensen et al. 2005; MD DNR 2020b). 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 (1989-2019) to place their time-series on the same scale for graphical comparisons of trends among surveys.

A fishery-independent index of relative abundance of upper Bay resident Striped Bass was not available and we used estimates of Maryland Striped Bass catch-per-private boat trip (released and harvested fish; RI) from the National Marine Fisheries Service's (NMFS) Marine Recreational Information Program (MRIP; NMFS Fisheries Statistics Division 2020) database as an index. Online estimates of catch and effort are available for 1981 and onwards. 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* (NEFSC 2019; NEFSC 2012; ASMFC 2013). 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 2020). 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 2020). 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 2-5 year-old Striped Bass estimated by the statistical catch at age model used in the recent stock assessment in this report (NEFSC 2019).

We used forage indices divided by RI (forage index-to-Striped Bass index ratios, i.e., forage ratio 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 1989-2019, a time-period in common among all data; FR covered 1983-2019.

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 over all species, 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 averaged for the year to calculate the annual weighted FR. Targets and limits for annual weighted FR were drawn from periods of three or more years when they coincided with target or limit P0, respectively. The annual weighted FR target for major forage ratios was estimated as the lowest standardized ratio that coincided with P0 meeting its target. The annual weighted FR threshold was estimated as the highest 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 2019 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; nearly all 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; this was 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 (stock went from severely depleted to recovered) and the original estimates from Versak (2018) afterwards. The 2019 estimate was supplied directly by Beth Versak. First we estimated a catchability coefficient (q) for age Striped Bass by dividing CPUE3 by the estimated abundance at age 3 from NEFSC (2019 during 1985-2017); 2017 was the last year in the assessment. 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 CPUEs were used (Uphoff et al. 2019).

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});

$$(1)\ SR_t = HI_t / JI_{t-3}.$$

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.

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 2020; Versak 2019; MD DNR 2020b). 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 original catch 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).

Index of Forage or IF – Examination of 90% confidence intervals of IF metrics in Uphoff (2019) indicated a scoring system reduced from previous years (from scores of 1-5 to 1-3) better matched a generalization of separation indicated by 90% confidence intervals (Uphoff et al. 2019). Based on the variation of IF indices using the "leave one out" approach (described below), Uphoff et al. (2019) placed an upper boundary for threshold (poorest conditions indicated at 1.0-1.5) that captured all of the threshold period. Similarly, a lower bound for target conditions that captured the target period (or best conditions) was 2.5 and target conditions were indicated by IF scores between 2.5 and 3.0. This 1-3 scoring system was used for all metrics included in the IF. Annual scores for each metric were averaged for a combined annual IF score (Uphoff et al. 2019).

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.

Analyses - Statistical analyses provided evidence that forage and Striped Bass abundance and well-being were inter-related. 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:

$$(2)\ Y = (m \cdot X) + b;$$

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) was used to remove time-series bias (Rose et al. 1986):

$$(3)\ Y = (m \cdot X) + (n \cdot T) + b;$$

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):

$$(4)\ Y = (m \cdot X) + (n \cdot X^2) + b.$$

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:

$$(5)\ Y = a \bullet (X)^b;$$

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:

$$(6)\ Y = b / ((1 + ((b – c) / c) \bullet (\exp (-a \bullet X)));$$

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:

$$(7)\ Y = K\{1 - \exp [-(Y / S)^b]\};$$

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).

Level of significance was reported, but potential management and biological significance took precedence over significance at $P < 0.05$ (Anderson et al. 2000; Smith 2020). 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 $r^2 \leq 0.25$; and moderate relationships fell in between. Moderate to strong correlations and relationships were considered biologically significant and of interest to management. 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, rejection of the model was considered.

Results

Striped Bass condition, feeding success, and diet composition Indices - During 1998-2019, 10,672 Striped Bass were sampled to estimate P0 during October-November (Table 2). Annual sample sizes ranged from 224 to 1,202 with a median sample size of 372 (Table 2).

Striped Bass in the upper Bay during fall were usually in poor condition (P0 ≥ threshold; threshold = 0.68) during 1998-2004 and at or near the target level of condition (P0 ≤ target; target = 0.30) during 2008-2010, 2014-2015, and 2017; P0 was 0.44 in 2019 (Figure 2). The 90% confidence intervals of P0 allowed for separation of years meeting the target or threshold conditions from remaining estimates. An 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; P0 in 2019 was assigned a score of 2. Condition shifted away from threshold to intermediate IF scores during 1998-2007 and to intermediate to target IF scores afterwards (Figure 2).

A combined P0 index for all sizes of Striped Bass was adopted in Uphoff (2016) based on 1998-2014 data; however, in 2016 a pronounced difference in condition was evident between small (small P0 = 0.83) and large sized fish (P0 = 0.25; Figure 3). This phenomenon was not repeated in 2017, but was present in 2018-2019 (2018 small fish P0 = 0.40 and large fish P0 = 0.05 and 2019 large fish P0 = 0.06 and small fish P0 = 0.70; Figure 3). This recent divergence in P0 between small and large Striped Bass may indicate a prey bottleneck existed for small fish but not large fish.

Samples from 2,047 small and 2,617 large sized Striped Bass were analyzed for diet composition during October-November, 2006-2019. This fall diet sampling was used to index availability of major forage items to compare to relative abundance indices. Numbers examined each year ranged from 47 to 330 small fish and 49 to 327 large fish (Table 3). 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 per trips in fall by FWHP during 2014-2019 versus 11-22 trips by CBEF during 2006-2013). Starting dates for surveys analyzed were similar between those conducted by CBEF and FWHP (October 1-9), but samples taken on September 24, 2015 were included in that year's analysis because the earliest date sampled in October would have been October 21, 2015. End dates tended to be earlier in November for FWHP surveys, reflecting when size categories needed were filled out (Table 3).

In combination and by number, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (major forage items) accounted for 95.9% of diet items encountered in small Striped Bass collected from upper Bay during fall, 2006-2019 (Figure 4). Bay Anchovy accounted for the highest percentage by number when all years were combined (63.7%, annual range = 19.1-87.9%); Atlantic Menhaden, 13.5% (annual range = 0-48.8%); Spot 5.8% (annual range = 0-70.7%); Blue Crab, 13.0% (annual range = 0.8-34.6%); and other items accounted for 4.1% (annual range = 0-24.5%; Figure 4). The vast majority of major prey in small Striped Bass diet samples during fall were YOY (Uphoff et al. 2016).

Major prey accounted for 93.3% of diet items, by number, encountered in large Striped Bass diet samples during fall 2006-2019 (Figure 5). Atlantic Menhaden accounted for 45.5% when all years were combined (annual range =12.4-82.0%); Bay Anchovy, 15.8% (annual range = 3.4-32.5%); Spot, 8.2% (annual range = 0-52.4%); Blue Crab, 22.8% (annual range = 2.6-59.4%); and other items, 7.7% (annual range = 0-40.0%). 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 5) 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 2006-2019 (combined) were dominated by Atlantic Menhaden (71.1%), followed by Spot (11.2%), Bay Anchovy (9.8%), Blue Crab (2.1%) and other items (5.7%; Figure 6). Estimates of C (total grams of prey consumed per gram of Striped Bass) for small Striped Bass varied as much as 8.7-times during 2006-2019. 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 2019 estimate of C of small fish was eighth highest of the 14 year time-series (Figure 6).

By weight, Atlantic Menhaden predominated in large fish sampled (86.5% of diet weight during fall, 2006-2019, combined); Bay Anchovy accounted for 1.2%; Spot, 3.7%; Blue Crab, 3.8%; and other items, 4.7% (Figure 7). Estimates of C for large Striped Bass varied as much as 3.8-times among years sampled. The 2019 estimate of C of large fish was seventh highest of the 14 year time-series (Figure 7).

Estimates of PE (proportion of empty stomachs) of small Striped Bass during fall, 2006-2019, ranged between 0.10 and 0.57 (Figure 8). Estimated PE was 0.24 in 2019. Lowest estimates of PE for small fish (2009-2011, 2014, 2017, and 2019) could be separated from remaining estimates (except 2008) based on 90% confidence interval overlap. 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. The highest PE point estimate during 2008-2010 (PE ranged from 0.19 to 0.31 and was highest in 2008) when P0 was at its target was selected as the PE target (PE $\leq$ 0.31 is assigned a score of 3). Estimated PE in 2019 was below the target and was assigned a score of 3. Estimates of PE steadily fell for small fish during 2006-2011. Since then, two years have met the target (2014 and 2019), three years have been between the target and threshold, and three years have been at the threshold (Figure 8).

Estimates of PE of large Striped Bass during fall, 2006-2013, ranged between 0.40 and 0.63, fell to 0.10-0.29 during 2014-2016, rose to 0.60 in 2017, and fell again to 0.18 in 2018 and 0.25 in 2019 (Figure 9). Lowest estimates of PE for large fish (2013-2016 and 2018-2019) 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 9).

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 2006-2019 (Figure 10). Median PPLRs for large Striped Bass were much closer to the optimum (0.21 based on Overton et al. 2009) than for small fish. Median PPLRs for small fish were particularly high (0.34-0.38) during 2012 and 2015-2019 and were close to optimum in 2010 when Spot constituted a large fraction of their diet. Median PPLRs have been higher for both size groups since 2016 (Figure 10).

Correlation analyses among C, PE, median PPLR for large major prey, and P0 for each year and size class (Table 4) 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 each size class of Striped Bass. For small fish correlations of large major prey PPLR with PE and C with P0 were strong enough for consideration ($r = 0.58$, $P = 0.037$ and $r = -0.69$, $P = 0.008$, respectively). Feeding success of small Striped Bass in fall was positively associated with availability of major prey in a smaller size range. Body fat scores of small fish reflected years of higher consumption that reflected higher consumption of Atlantic Menhaden and Spot. For large fish, correlations of PE with P0 and PE with C were strongly correlated enough for consideration ($r = 0.71$, $P = 0.004$ and $r = -0.56$, $P = 0.038$, respectively). Better body fat scores for large Striped Bass reflected a higher frequency of empty guts and the higher frequency of empty guts was negatively associated with consumption which, like with small fish, was higher due to consumption of more Atlantic Menhaden and Spot. Low PE reflected prevalence of small and alternative prey when Atlantic Menhaden and Spot were not available.

Relative abundance indices of prey and Striped Bass – Major pelagic prey were generally much more abundant during 1959-1994 than afterward (Figure 11). Bay Anchovy seine indices 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 2015-2019. 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 low abundance points). Atlantic Menhaden seine indices were high during 1971-1994 and much lower during 1959-1970 and 1995-2019 (Figure 11).

Major benthic forage indices were low after the 1990s, but years of higher relative abundance were interspersed during the 2000s (Figure 12). Seine (1959-2019) and trawl (1989-2019) 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 (1989-2019) were generally at or above the time-series median during 1989-1998, and 2009-2015 (Figure 12).

In general, relative abundance of Striped Bass (RI) during 1981-2018 was lowest prior to 1994 (mean RI < 0.4 fish per trip; Figure 13). 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), rose to 2.4-3.0 during 2014-2018.and was 3.6 in 2019. 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 1994-2019. Ninety percent CIs of periods of threshold P0 (1998-2004) and target P0 (2008-2010) indicated some overlap of RI; median RI estimates during 2008-2010 did not or barely overlapped the lower 90% CIs of 1998-2004 (Figure 13).

Threshold conditions of P0 were most often breached when RI was at or greater than 2.0 (score = 1; 5 of 15 years versus 1 of 7 when RI was less than 2.0). Target conditions were more often met when RI was less than 2.0 (score = 3; 3 of 7 years when RI was less than 2.0 versus 2 of 15 when RI was at or greater than 2.0). RI has been in excess of 2.0 since 2014 (Figure 13) and 2019 was assigned an IF score of 1.

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 14). Overall, the estimates were well correlated ($r = 0.79$, $P < 0.001$).

Species-specific standardized FRs exhibited similar patterns during 1983-2019 (Figures 15-20). 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. The 90% CIs for prey to Striped Bass ratios indicated these ratios were high prior to 1994 and lower afterward (Atlantic Menhaden, Figure 15; Bay Anchovy, Figures 16 and 17; Spot, Figures 18 and 19; Blue Crab, Figure 20).

In the first step for estimating weighted FR, correlations among species-specific FRs (1998-2019) indicated that the two indices for Spot were closely correlated ($r = 0.98$). 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 ($r = -0.41$, $P = 0.06$), followed by Blue Crab ($r = -0.32$; $P = 0.15$), Spot (seine index, $r = -0.25$, $P = 0.26$), and Bay Anchovy (seine index, $r = -0.11$, $P = 0.61$). These correlations corresponded to a weight of 1.00 for Atlantic Menhaden, 0.77 for Blue Crab, 0.61 for Spot, and 0.28 for Bay Anchovy. These weights would apply to importance in fall diets and should not be interpreted as indicating importance to year-round Striped Bass diets. Trends in candidate species-specific standardized FRs for all years available (1989-2019) are depicted in Figure 21 (standardized species-specific FRs are on a $\log_{10}$-scale). Trends in standardized FRs for years used for weighting FR (1998-2019, years with P0 estimates; on an arithmetic scale) are depicted in Figure 22.

Weighted FR was lowest during the threshold period for P0, 1998-2004 (except 2001;

Figure 23). Threshold FR was 0.20 or less (score =1). Threshold conditions were also breached during 2006, 2015-2017, and 2019. 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; Figure 23).

The hybrid age 3 gill net index of male relative abundance (HI_3) on the spawning grounds indicated a dearth of high indices during 1985-1995 (Figure 24). These low HI_3 year-classes were followed by 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 24).

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 25). Estimated SR was consistently high during 1986-1996, shifted to consistently low during 1999-2004, and varied afterwards. Low survival in 1985 reflected the effect of the fishery prior to imposition of a harvest moratorium in Maryland (Figure 25).

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 26). After 2004, threshold conditions were met in 2007, 2008, 2012, and 2016 (Figure 26). The relative survival estimate for 2019, 31.7, was assigned an IF score of 2.

Targets and thresholds scores for P0, PE, RI, FR, and SR are summarized in Table 5.

Index of Forage or IF – The IF varied from 1.0 to 3.0 during 1998-2018 (Figure 27). During 1998-2004, the IF was low, between 1.0 and 1.25 (threshold forage conditions). The IF increased to 2.25 in 2005, fell below 2 in 2006-2007, and then increased to 2.5 to 3.0 (target forage conditions) during 2008-2011. After 2011, it varied from above 1.4 to 2.4. The IF was 2.0 during 2019. 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 27).

Estimates of mean IF with each component removed indicated little variation from the overall IF (Figure 28). 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 28). This approach suggested that IF means could be separated into high, medium, and low categories.

Discussion

Striped Bass condition, feeding success, and diet composition Indices - When both size categories were combined, estimates of P0 after 2014 did not decline to threshold conditions. Estimates of P0 for small and large Striped Bass tracked each other through 2015. However, P0 diverged between small fish (P0 has been high) and large fish (P0 has been low) during 2016, 2018, and 2019. Small fish breached the P0 threshold in 2016 and 2019, while large fish achieved target P0 during 2014-2019. This divergence may indicate relief from a prey bottleneck for large fish since it coincides with slightly elevated Atlantic Menhaden FR, higher prey availability indicated by lower PE, and higher consumption of Atlantic Menhaden.

Estimates of PE improved for large fish, but not small ones since 2014. Estimates of the PPLR provided supplemental information for evaluating PE. Higher PPLRs (indicating larger sized major prey) were positively associated with higher PE for small Striped Bass, but not large ones, suggesting that PE of small fish would be influenced by size of larger prey. Median PPLR of large major prey for Striped Bass, primarily Atlantic Menhaden, has been higher since 2015 than previous years (except 2012) for both size classes. Median PPLRs were smaller for large Striped Bass in some years and were much closer to their optimum, particularly during the target P0 period (2008-2010).

Consumption of Atlantic Menhaden during fall by small and large Striped Bass since 2013 has been higher, more frequently ranking in the top half of estimates of C. Low consumption of Atlantic Menhaden by small Striped Bass during 2016-2017 was not offset by other prey. Spot, a major prey that had contributed to lower PPLR of large prey and achievement of target P0 and PE for small fish in 2010, have become rare in diets of both size classes. Bay Anchovy, while dominant by number in small Striped Bass diets, made up a low fraction of fall diet weight in all but the worst years; Blue Crab were a minor component as well.

Small Striped Bass would have more difficulty in catching and handling the same sized large major prey than large Striped Bass in any given year. 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 larger size makes them more difficult to retain if caught (Lundvall et al. 1999). With high size limits and low fishing mortality in place since restoration, intraspecific competition for limited forage should be greater for small Striped Bass because they compete with one another and large Striped Bass. All things being equal, large striped bass should forage more efficiently and outcompete small fish through greater vision, swimming speed, and experience (Ward et al. 2006). Below threshold P0 of small fish in 2016 and 2019 coincides with two large year-classes of Striped Bass having reached the large size category (2011 year-class in 2016 and 2015 year-class in 2019).

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 on occasion White Perch made a contribution to large Striped Bass diet biomass.

Hook-and-line samples collected by CBEF (2006-2013) and FWHP (2014-2019) were treated as a single time-series. Sampling by CBEF stopped in 2015 due to failing health of Mr. Price (CBEF President and organizer of the CBEF diet survey). Samples were collected by both programs during 2014, providing an opportunity for comparison (Uphoff 2018). Size of Striped Bass sampled by the two programs appeared comparable and estimates of P0 were similar. Fall diets were dominated by Atlantic Menhaden and Spot were absent in both cases. Differences arose in smaller major prey, particularly Bay Anchovy, and in the importance of “Other” prey (Uphoff et al. 2018).

CBEF conducted a year-round diet sampling program useful to MD DNR free of charge, but this level of sampling could not be maintained by FHEP staff due to existing duties. Piggybacking diet sampling onto the existing fall FWHP Striped Bass health survey provided a low-cost alternative that would provide information on Striped Bass condition and relative availability of major prey, but would not characterize the annual diet or condition changes within a year. Consumption based indices of prey availability in fall (PE and C) appeared to be more sensitive and biologically significant (i.e., were reflected by P0) than FR based on relative abundance indices.

We chose to treat hook-and-line samples in fall as random samples (Chipps and Garvey

2007) rather than as cluster samples (Rudershausen et al. 2005; Hansen et al. 2007; Overton 2009; Nelson 2014),i.e., individual fish rather than a school were considered the sampling unit. This choice reflected changing feeding behavior of Striped Bass in fall. Fall is a period of active feeding and growth for resident Striped Bass and forage fish biomass is at its peak (Hartman and Brandt 1995c; Walter and Austin 2003; Overton et al. 2009). Striped Bass leave the structures they occupied during summer-early fall and begin mobile, aggressive, open water feeding. Forage begins to migrate out of the Bay and its tributaries (and refuges therein) or to deeper water at this time and are much more vulnerable to predation. Both major forage and Striped Bass schools are constantly moving and changing. Schools of Striped Bass and their prey no longer have a fixed nature, presenting well mixed populations (J. Uphoff, MD DNR, personal observation) that made a random sampling assumption a reasonable choice.

Relative abundance indices of prey and Striped Bass – Forage to Striped Bass ratios reflected availability of major prey to both small and large size classes of Striped Bass since RI was used in the denominator, while PE of the small size class was used as an indicator of forage availability (Uphoff et al. 2017). Although size classes could not be specified for RI, Uphoff et al. (2014) found that a multiple regression using Maryland Striped Bass juvenile indices lagged to ages 2-5 (corresponding to both size classes) predicted trends in the RI.

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 as a weight (resulting in an index dominated by Atlantic Menhaden for most years).

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 22 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.

Multiple lines of evidence suggest that survival of both small and large Striped Bass has decreased in the Chesapeake Bay since the late 1990s. The 40% percent reduction in median SR of small (sublegal) Striped Bass between 1986-1996 and 1997-2019 was very close to changes in conventional tag-based estimates of survival in Chesapeake Bay based on converting M of large (457-711 mm Striped Bass, equivalent to our large category) fish, from 77% annual survival (1987-1996) to 44% (1997-2017), a 43% reduction (based on Table B8.25 in NEFSC 2019). Estimates of F in Chesapeake Bay of a size range corresponding to our large fish from tagging have been low and estimates of M have been high (NEFSC 2019). Secor et al. (2020) implanted a size-stratified sample of Potomac River Striped Bass with acoustic transmitters and recorded their migrations during 2014-2018 with telemetry receivers throughout the Mid-Atlantic Bight and Southern New England. Analysis of the last day of transmission indicated that Chesapeake

Bay resident Striped Bass experienced 53% lower survival (30% per year) than coastal shelf emigrants (63% per year; Secor et al. 2020).

Decreased survival estimates from conventional tags during 1987-1996 and 1997-2017 in NEFSC (2019) was attributed to mycobacteriosis. Mycobacteriosis alone would not necessarily be the only source of increased M of Chesapeake Bay Striped Bass. Jacobs et al. (2009) were able to experimentally link the progression of mycobacterial disease in Striped Bass to their diet: inadequate diet led to more severe disease progression compared with a higher ration. Abundant individuals competing for limited prey may hinder one another's feeding activities, leading to starvation (Yodzis 1994). Two of 8 estimates of P0 since 1998 breached the body fat threshold when RI was below its median and 5 of 9 estimates of P0 breached it when RI was above its median (5 estimates very near median RI were not included), indicating higher vulnerability to starvation at higher RI. Shifts from high survival during 1987-1996 to lower survival afterwards lagged two years behind downward shifts in forage-to-Striped Bass ratios. Dutil and Lambert (2000) found that the response Atlantic Cod (*Gadus morhua*) M could be delayed after unfavorable conditions. Similar to Striped Bass, some stocks of Atlantic Cod experienced forage fish declines, followed by declining body condition and increased M; starvation caused declines in energy reserves, physiological condition, and enzyme activity (Lilly 1994; Lambert and Dutil 1997; Dutil and Lambert 2000; Shelton and Lilly 2000; Rose and O'Driscoll 2002). Recovery of the northern stock of Atlantic Cod has paralleled recovery of Capelin (*Mallotus villosus*), its main prey (Rose and Rowe 2015); increases in size composition and fish condition and apparent declines in mortality followed. Mortality due to starvation is a size-dependent process that represents an alternative (albeit final) response to reduced growth and stunting during food shortages and may be more common than generally perceived (Ney 1990; Persson and Brönmark 2002).

Catch-and-release mortality different from that assumed in NEFSC (2019) could have been confounded with natural mortality. However, Striped Bass size was a significant factor in Chesapeake Bay catch-and-release mortality experiments (probability of dying increased with TL; Lukacovic and Uphoff 2007) and similar relative changes in survival between SR (small fish) and the tag-based estimates survival based on M (large fish) would not have been expected if a sizeable fraction of survival was influenced by catch-and-release mortality. Decreases in conventional tag-based estimates of mortality of legal-sized fish could also reflect misspecification of parameters such as tag reporting rates that make absolute estimates less reliable (NEFSC 2019); however, mortality estimates based on acoustic tags produced similar differences in mortality of coastal migrants and Chesapeake Bay residents (Secor et al. 2020).

The fall in survival was consistent with a compensatory response to high Striped Bass abundance, low forage, and poor condition. The degree that M compensates with F may reduce effectiveness of management measures since total mortality, Z, may not be reduced by harvest restrictions when M increases as F decreases (Hilborn and Walters 1992; Hansen et al. 2011; Johnson et al. 2014). Single species stock assessments typically assume that M is constant and additive with F to keep calculations tractable (Hilborn and Walters 1992). Animal populations may exhibit additive mortality at low abundance and compensatory mortality at high abundance or compensatory mortality that changes continuously with density (Hansen et al. 2011). Increased M over time may have serious implications for management since Chesapeake Bay is the main contributor to Atlantic coast fisheries (Richards and Rago 1999; NEFSC 2019). Management of Chesapeake Bay Striped Bass fisheries attempts to balance a trade-off of yield with escapement of females to the coastal migration by controlling fishing mortality, and compensatory mortality would undercut both objectives.

We developed the hybrid methodology for estimating SR over several years. 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. (2018) and used it to estimate a SR time-series that reflected changes in catchability based on the most recent ASMFC Striped Bass stock assessment (NEFSC 2019).

Confining the spring gill net relative abundance index to 3 year-old males makes it likely that trends in SR will reflect survival of resident Striped Bass before harvest (i.e., due to 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 for harvest (Versak 2019), but they could be subject to catch-and-release mortality. 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 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.

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 since shifts in migration can produce similar changes as M. Migration estimates based on 1988-1991 spawning survey 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). Secor et al. (2020) confirmed this general migration schedule by tracking acoustic tags. Kohlenstein (1981) determined that few young males leave the Chesapeake Bay.

The utility of estimates of biomass of invertebrates comprising a benthic IBI in Maryland's portion of the Bay used for water quality monitoring was explored in Uphoff (2018). A complementary index for hard (oyster) bottom was developed by M. McGinty (Uphoff et al. 2019; Job 3). 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 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*.

We used available estimates of central tendency and variability for 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.

Index of Forage or IF – 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 were elevated beyond the threshold after 2004 (with the exception of 2016). 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 or at 2.0 (does not breach threshold or target) during 2017-2019, 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 high abundance after 2014 was not shared by major forage, but it appears that slightly higher Atlantic Menhaden seine indices since 2007, while not always statistically distinguishable from indices during the 1998-2004 threshold period, may have biological significance for upper Bay Striped Bass.

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 could end up indicating threshold conditions even though Striped Bass were well supported by forage.

We have 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 (Hilborn 2016), but these are not possible 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).

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.

Two objectives of the IF are low cost and tractability for available staff. 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 used information from existing sampling programs and indices (i.e., convenience sampling and proxies for population level estimates, respectively; Falcy et al. 2016). This trade-off is very common in fisheries and wildlife management (Falcy et al. 2016).

The IF represents a framework for condensing complex ecological information so that it can be communicated simply to decision makers and stakeholders. The science of decision making has shown that too much information can lead to objectively poorer choices (Begley 2011). The brain's working memory can hold roughly seven items and any more causes the brain to struggle with retention. Proliferation of choices can create paralysis when the stakes are high and information is 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). Although it is intrinsic to integration that some information is lost, the loss is not necessarily of practical importance. 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.

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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
FHEP	Fish Habitat and Ecosystem Program
FWHP	Fish and Wildlife Health Program
HI_3	Hybrid gill net index of relative abundance of age-3 male Striped Bass that has been adjusted for catchability change with population size.
IF	Index of Forage. 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	Prey length to predator length ratio.
q	Catchability (efficiency of a gear).
RI	Index of relative abundance of resident Striped Bass estimated from MRIP as private / rental boat catch per trip during September-October.
SR	Relative survival from late age 0 to age 3.

Table 2. Estimates of proportion of Striped Bass without body fat (P0; both size categories combined), sample size (N), and the standard deviation (SD) of the estimate of P0 during 1998-2019.

Year	P0	N	SD
1998	0.749	338	0.024
1999	0.779	344	0.022
2000	0.773	290	0.025
2001	0.745	224	0.029
2002	0.605	316	0.028
2003	0.700	237	0.030
2004	0.746	414	0.021
2005	0.596	524	0.021
2006	0.600	863	0.017
2007	0.500	662	0.019
2008	0.137	629	0.014
2009	0.312	1107	0.014
2010	0.270	693	0.017
2011	0.531	1202	0.014
2012	0.658	333	0.026
2013	0.576	441	0.024
2014	0.312	398	0.023
2015	0.124	347	0.018
2016	0.476	429	0.024
2017	0.237	325	0.024
2018	0.403	330	0.027
2019	0.442	226	0.033

Table 3. Number of dates sampled and number of small (<457 mm, TL) and large sized Striped Bass collected for October-November diet information in each size category, by year. Diet collections were made by Chesapeake Bay Ecological Foundation during 2006-2013 and MD DNR Fish and Wildlife Health Program during 2014-2019. Start date indicates first date included in estimates of P0, PE, C, and diet composition and end date indicates the last.

Year	N dates	Small N	Large N	Start date	End date
2006	19	118	49	2-Oct	26-Nov
2007	20	76	203	4-Oct	29-Nov
2008	15	29	207	4-Oct	25-Nov
2009	17	99	240	3-Oct	25-Nov
2010	22	112	317	9-Oct	29-Nov
2011	19	74	327	1-Oct	26-Nov
2012	11	47	300	7-Oct	30-Nov
2013	14	191	228	3-Oct	18-Nov
2014	7	277	108	2-Oct	12-Nov
2015	8	174	173	24-Sep	17-Nov
2016	12	169	260	3-Oct	16-Nov
2017	9	272	52	2-Oct	13-Nov
2018	6	330	87	3-Oct	28-Nov
2019	8	135	90	1-Oct	19-Nov

Table 4. Correlations among median major prey prey-predator length ratios (PPLR), feeding metrics (proportion of empty stomachs, PE, and grams of all prey consumed per gram of Striped Bass, C) and condition (proportion without visible body fat, P0) for small (< 457 mm, TL) and large ($\geq$ 457 mm, TL) Striped Bass.

Small

Variable	Statistic	C	PPLR median	PE
PPLR median	r	-0.35735		
	P	0.2306		
PE	r	-0.23439	0.5814	
	P	0.4408	0.0371	
P0	r	-0.69428	0.4019	0.10744
	P	0.0085	0.1734	0.7268

Large

Variable	Statistic	C	PPLR median	PE
PPLR median	r	0.33288		
	P	0.2449		
PE	r	-0.55774	0.01132	
	P	0.0382	0.9694	
P0	r	-0.33112	0.11608	0.70895
	P	0.2475	0.6927	0.0045

Table 5. 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.

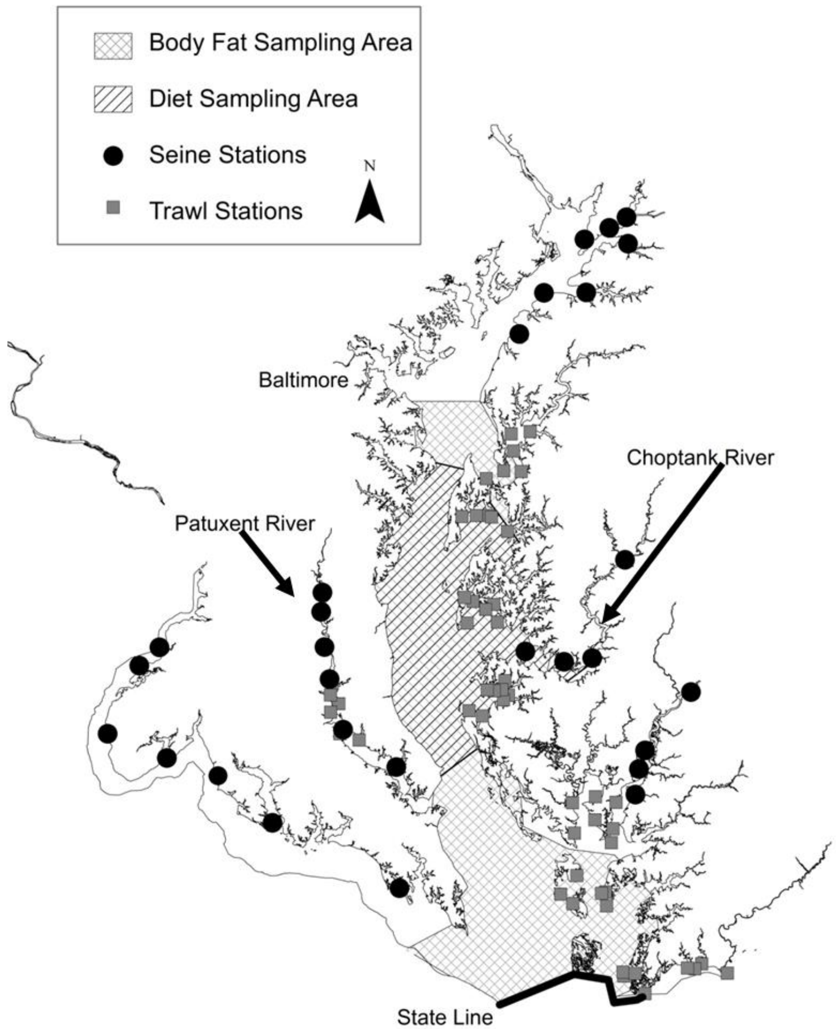

Figure 2. Proportion of Striped Bass without body fat (P0) during October-November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

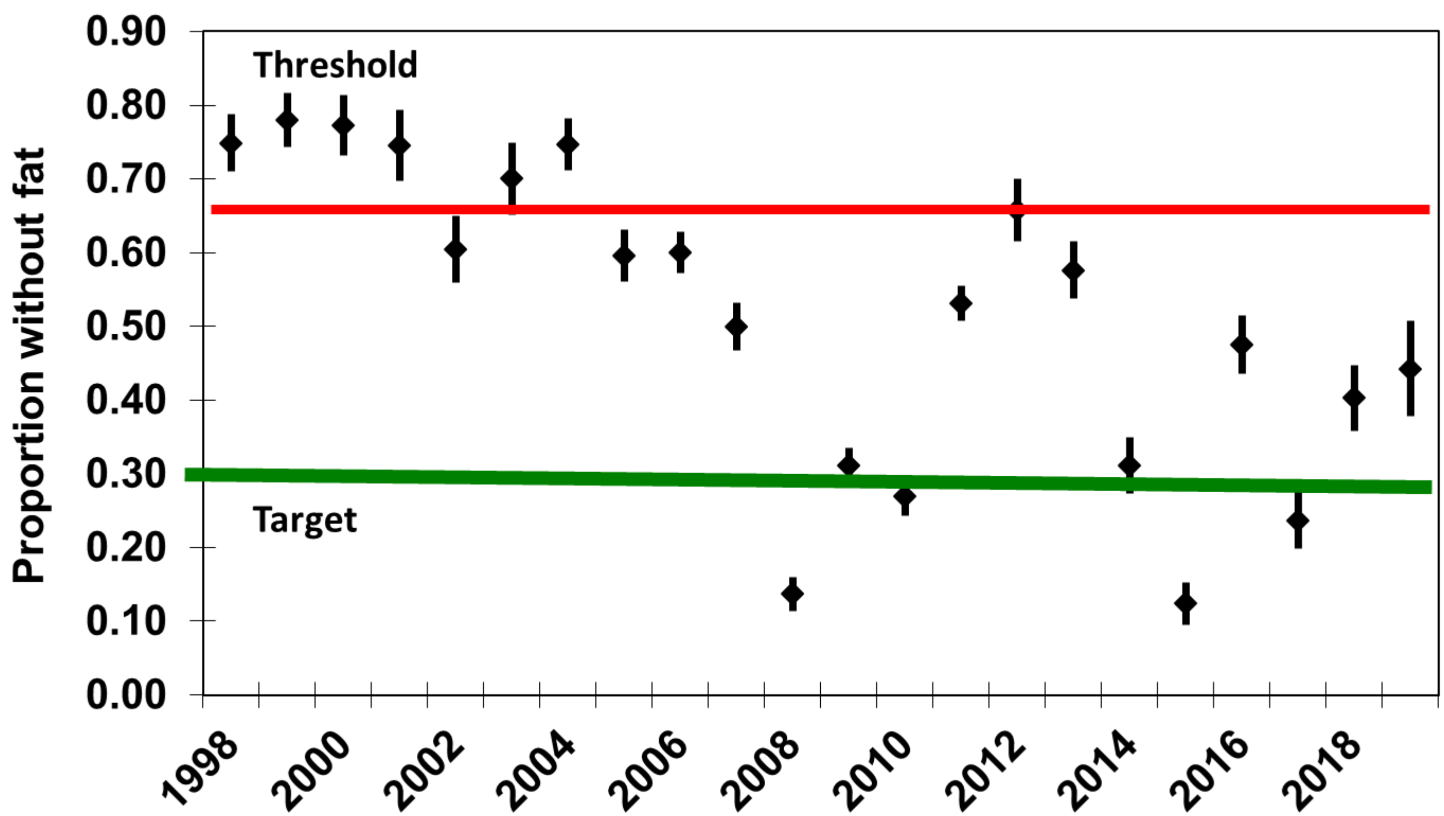

Figure 3. Trends in fall body fat indices (P0) for small (280-456 mm) and large striped bass

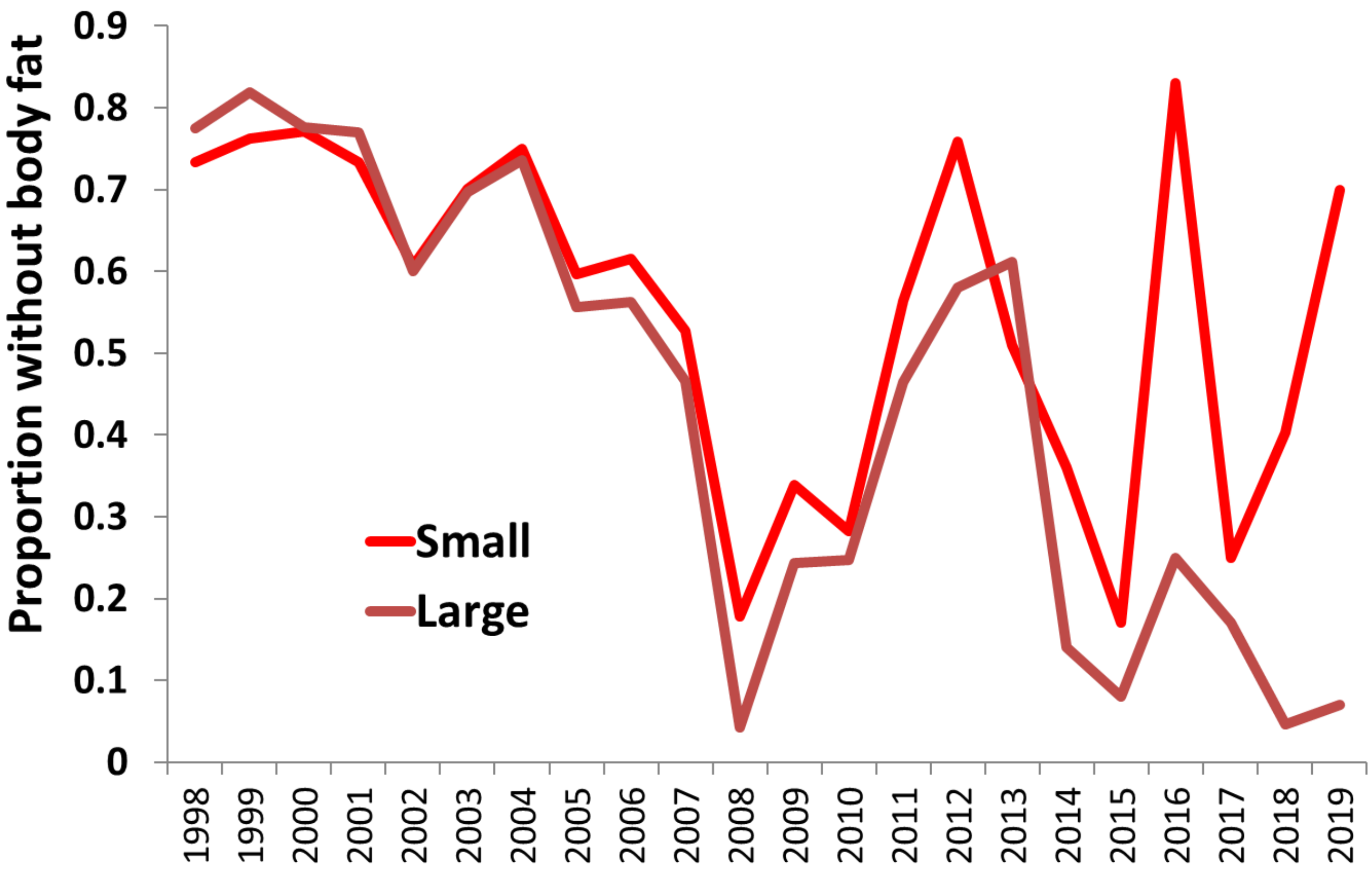

Figure 4. Percent, by number (counts of individuals plus presence of parts), of identifiable (excludes unknown) major forage groups in small Striped Bass (< 457 mm TL) guts, in fall.

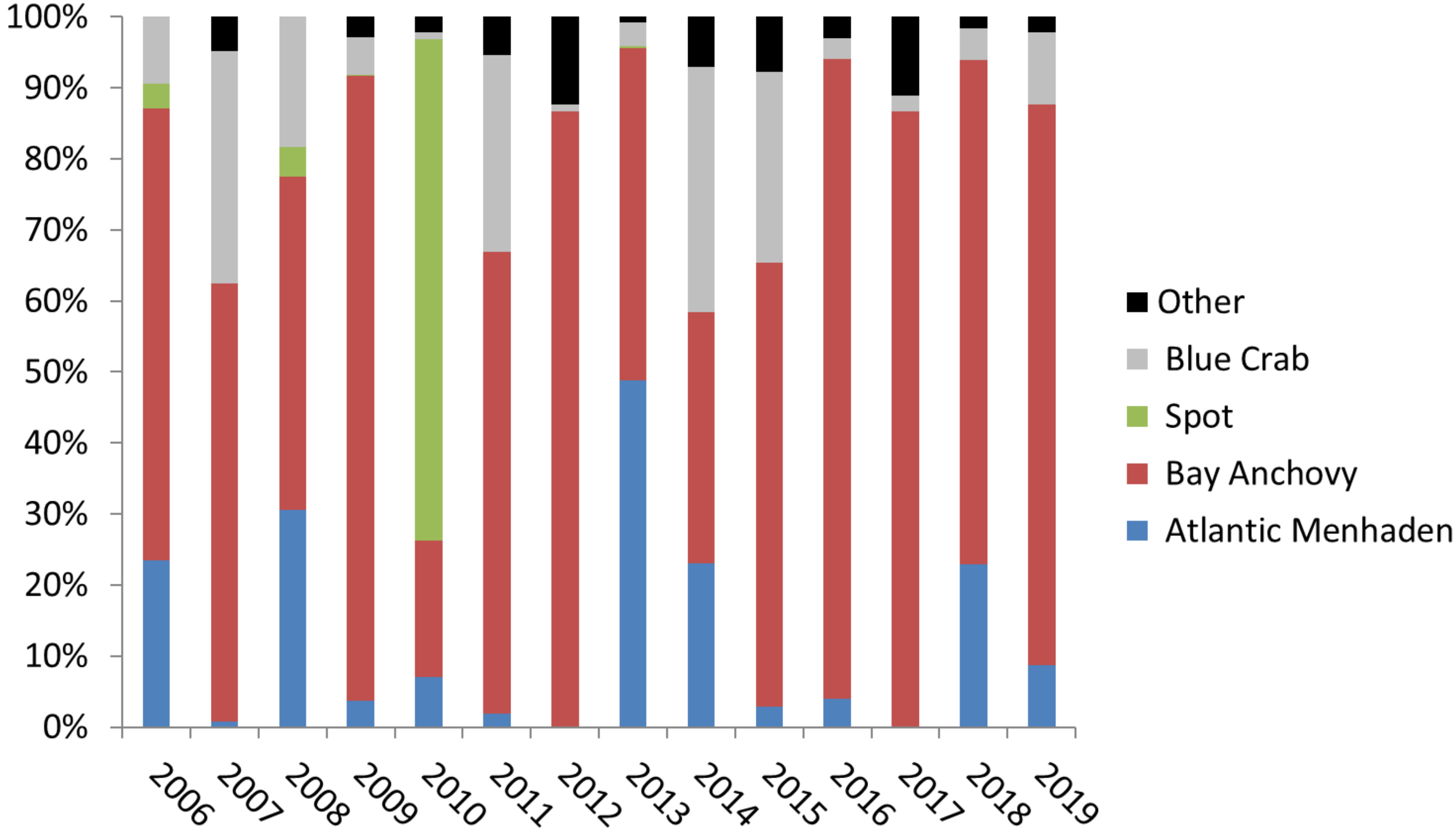

Figure 5. Percent of large Striped Bass (≥ 457 mm TL) identifiable diet represented by major forage groups, by number, in fall.

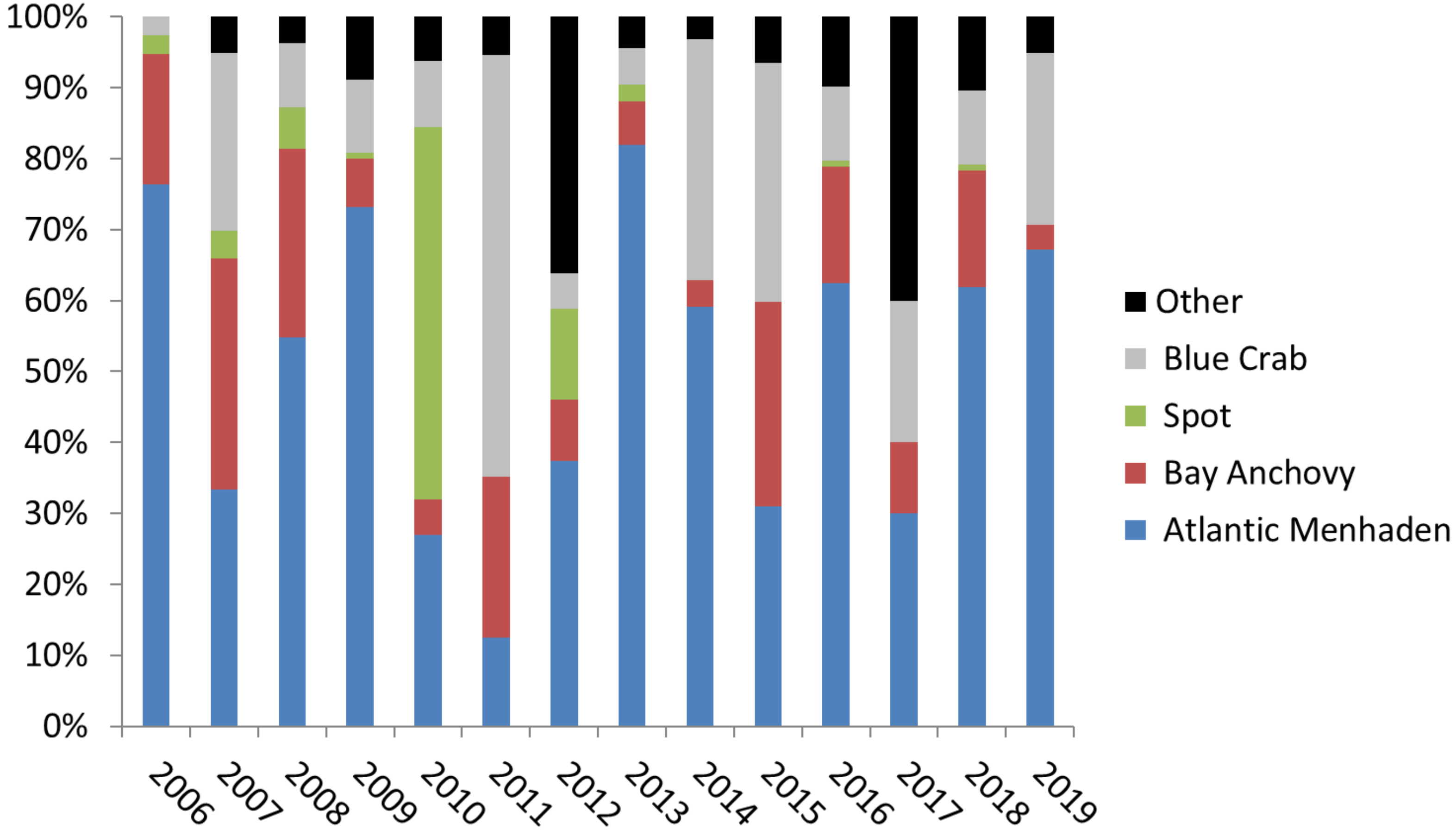

Figure 6. Gram prey consumed per gram (C) of small (< 457 mm TL) Striped Bass in fall hook-and-line samples. Age-0 forage dominate the diet. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

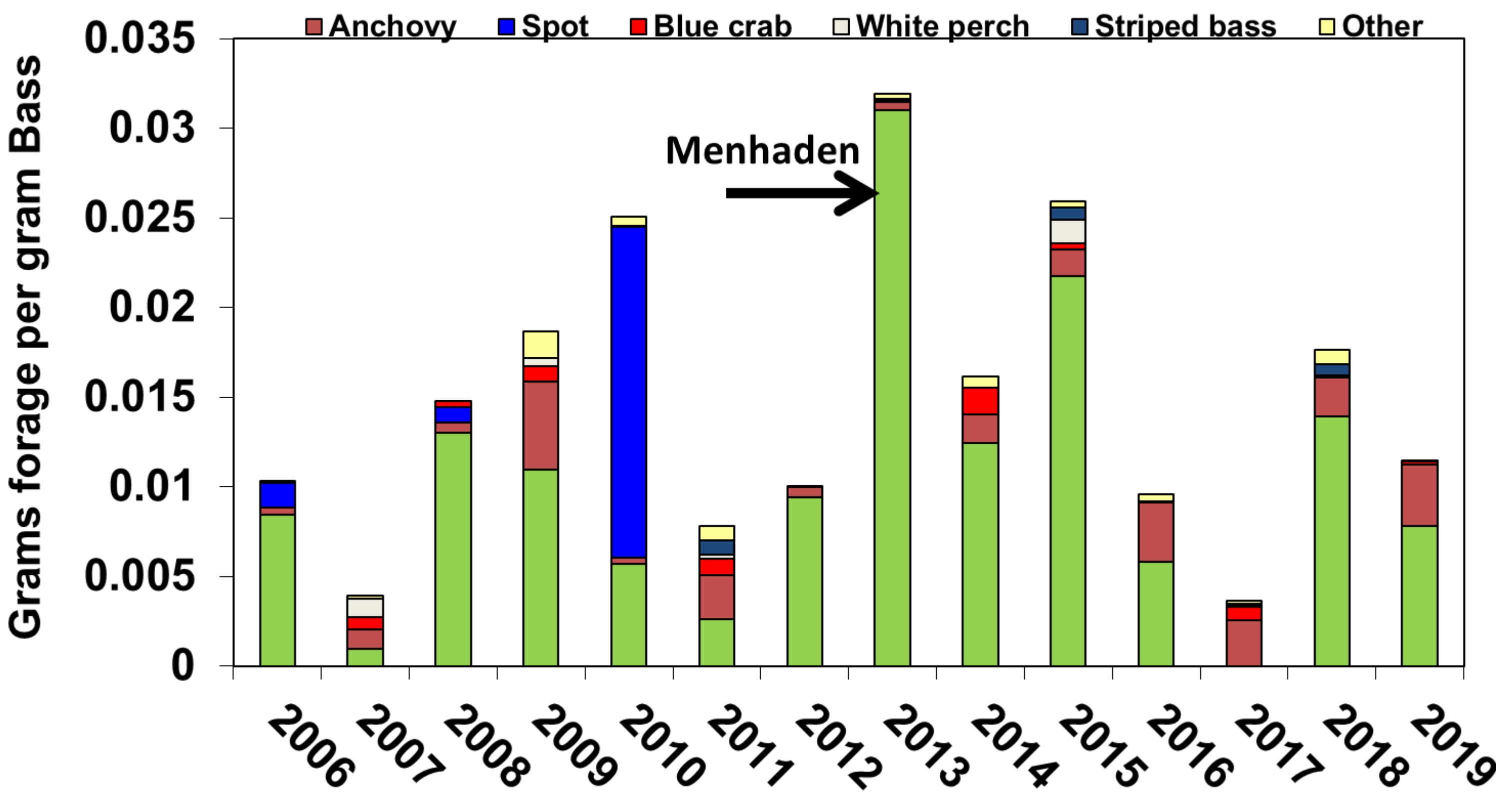

Figure 7. Grams of prey consumed per gram (C) of large (≥ 457 mm TL) Striped Bass during October-November. Fall consumption dominated by age 0 forage. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

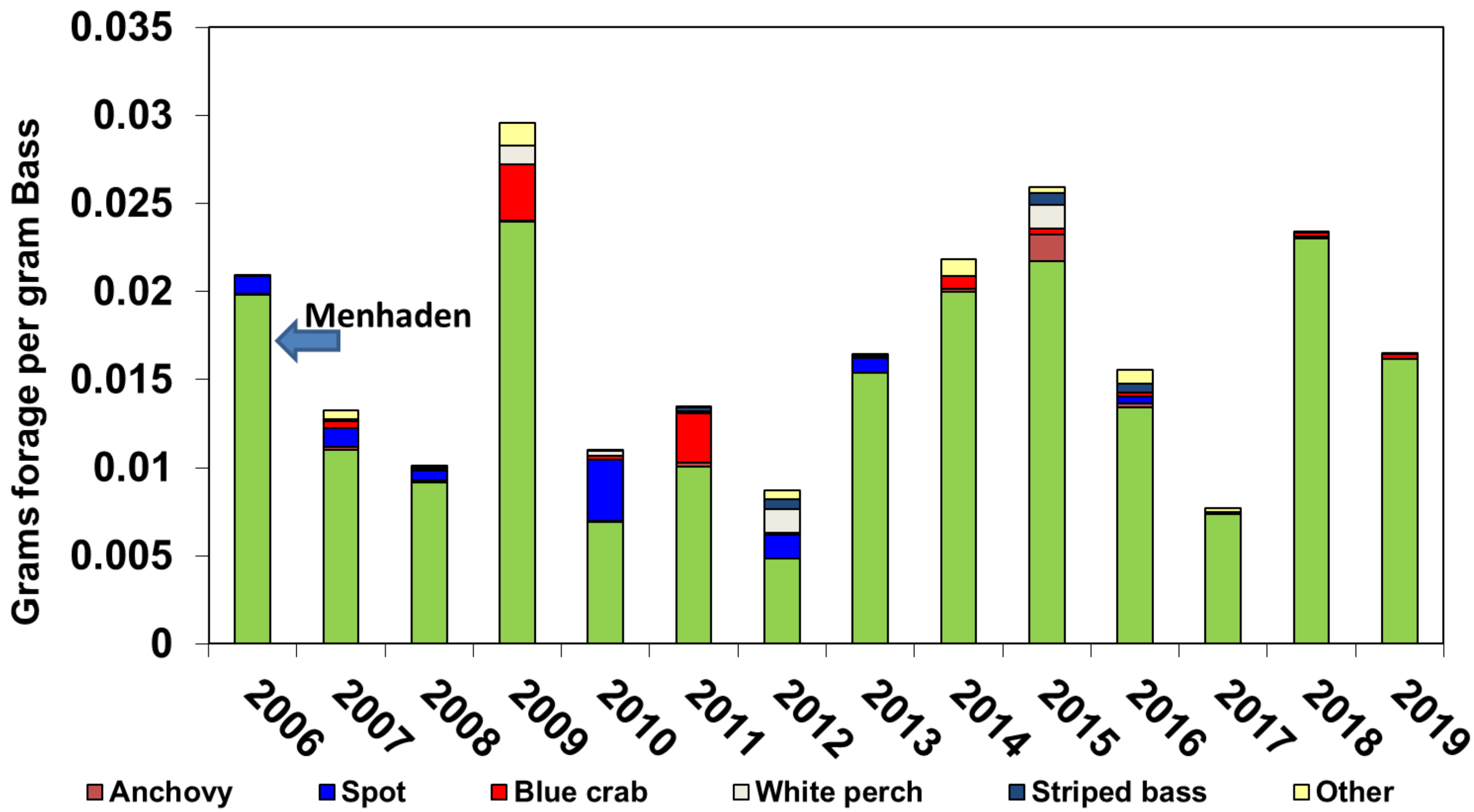

Figure 8. Proportion of small Striped Bass guts without food (PE) in fall and its 90% confidence interval. Red diamond represents threshold PE and green diamond indicates the PE target.

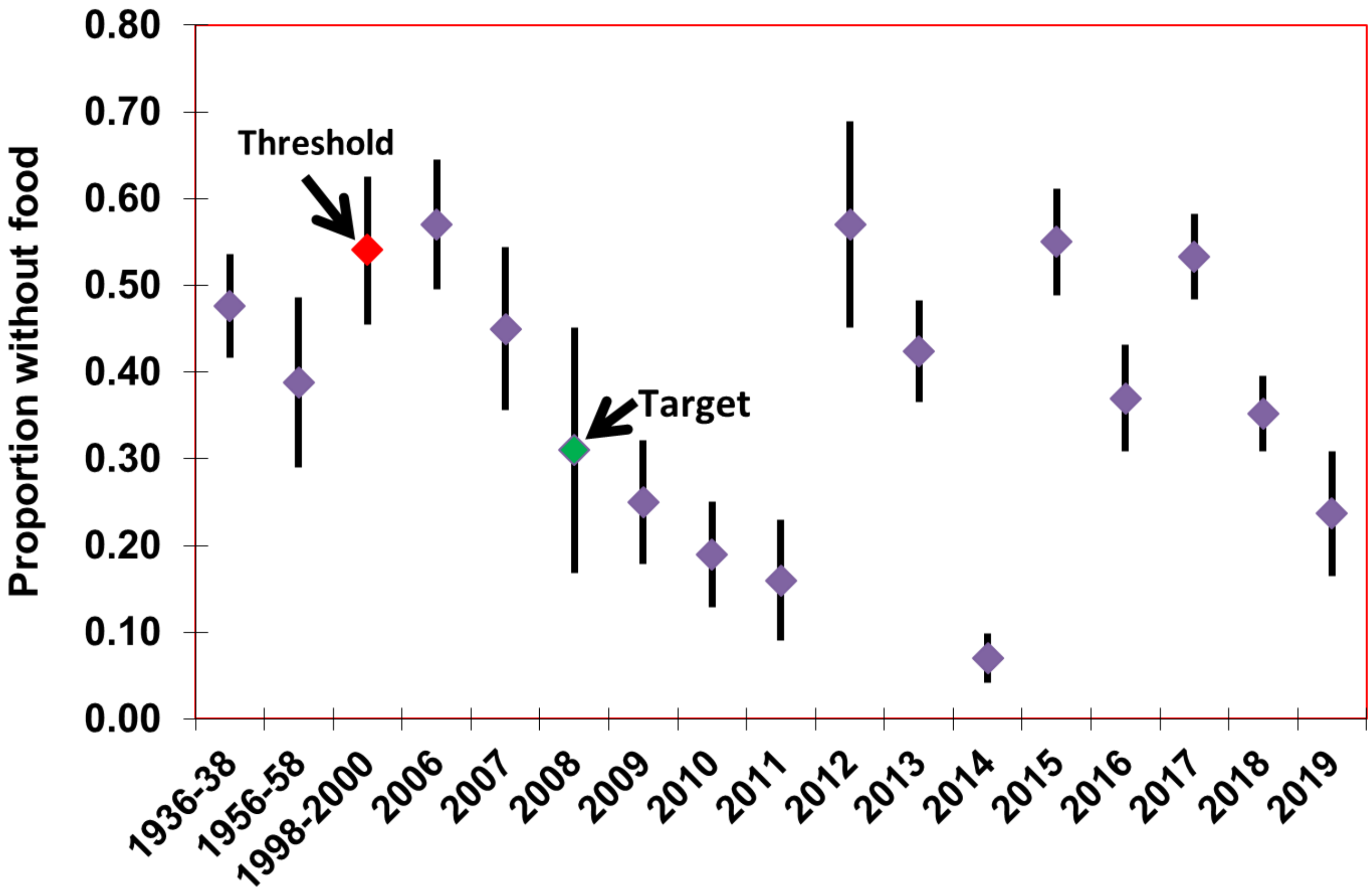

Figure 9. Proportion of large Striped Bass (> 456 mm or 18 in, TL) guts without food (PE) in fall and its 90% confidence interval.

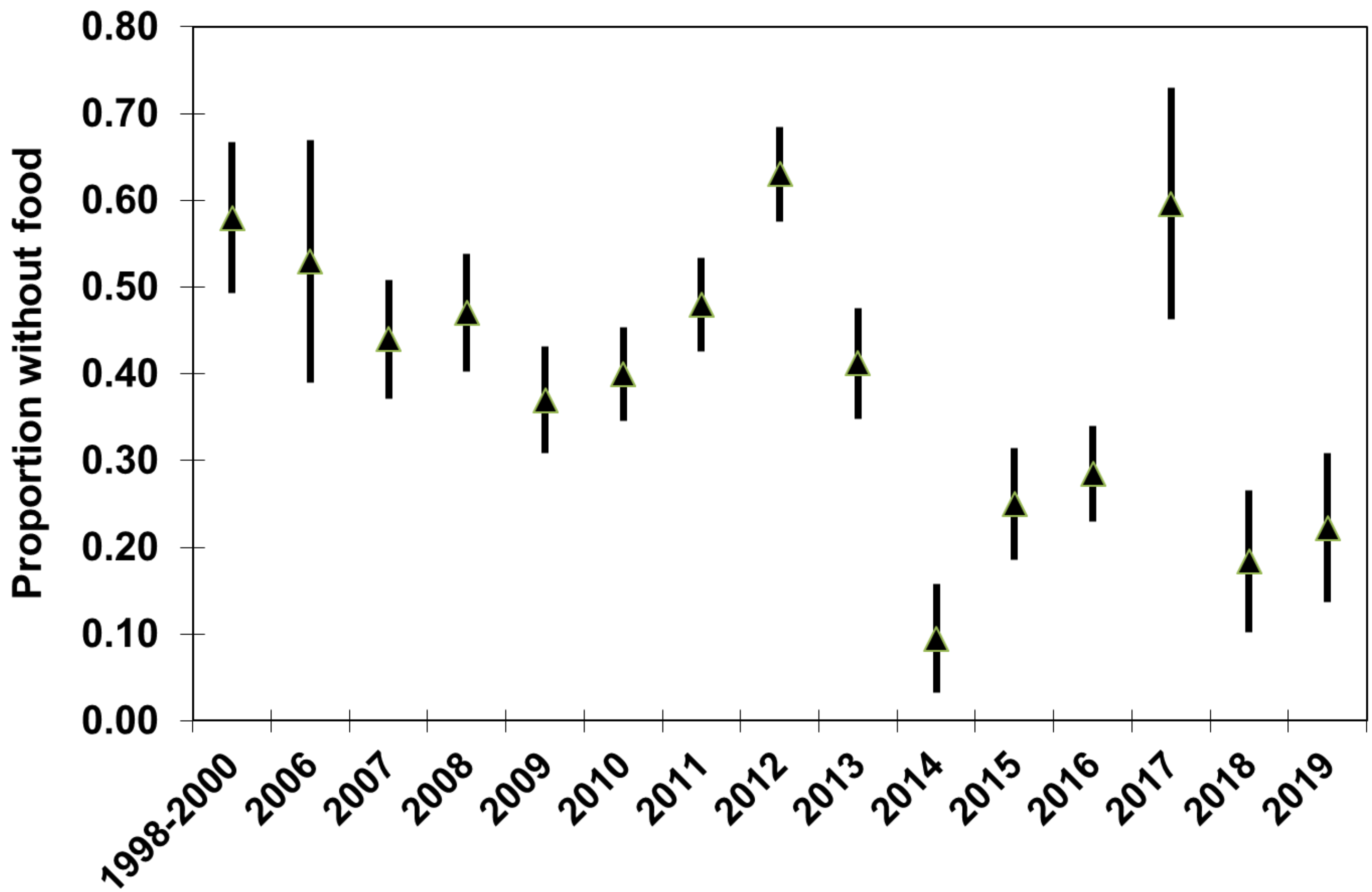

Figure 10. Median prey-predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for small (< 457 mm) and large Striped Bass. Optimum ratio was estimated by Overton et al. (2009).

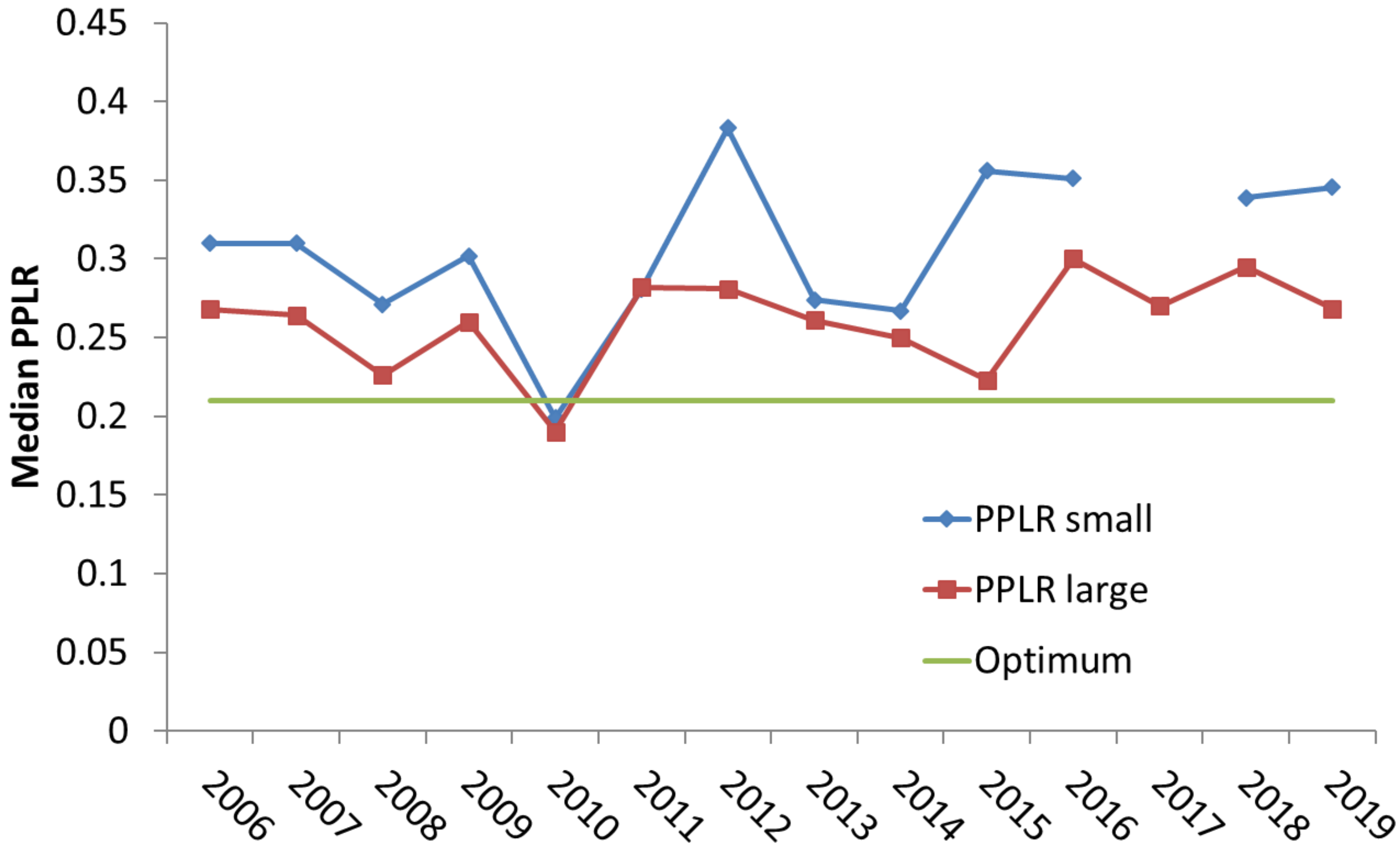

Figure 11. Trends in major pelagic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2019. Indices were standardized to their 1989-2019 means (years in common). Menhaden = Atlantic Menhaden and Anchovy = Bay Anchovy.

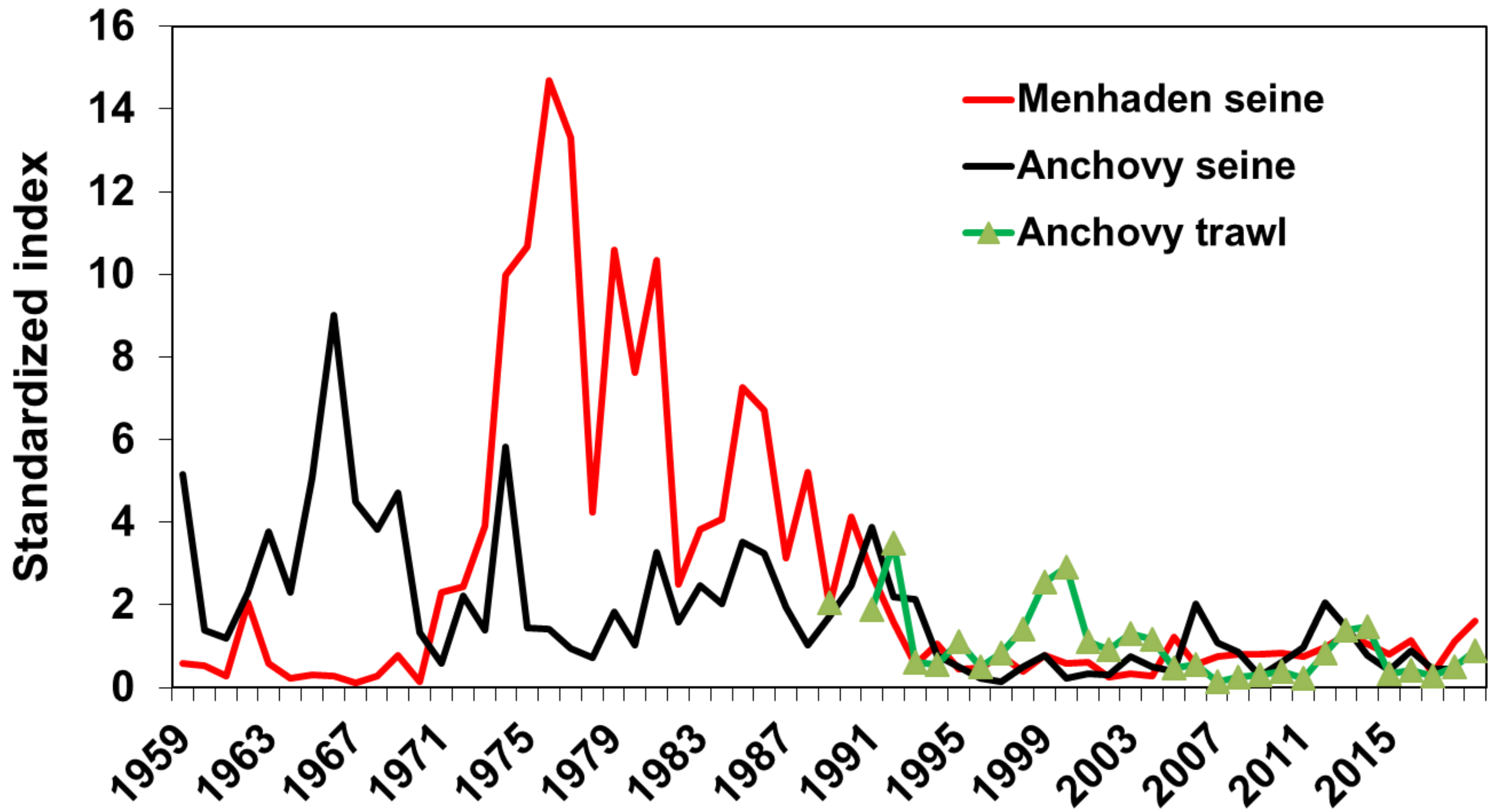

Figure 12. Trends in major benthic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2019. Indices were standardized to their 1989-2019 means (years in common).

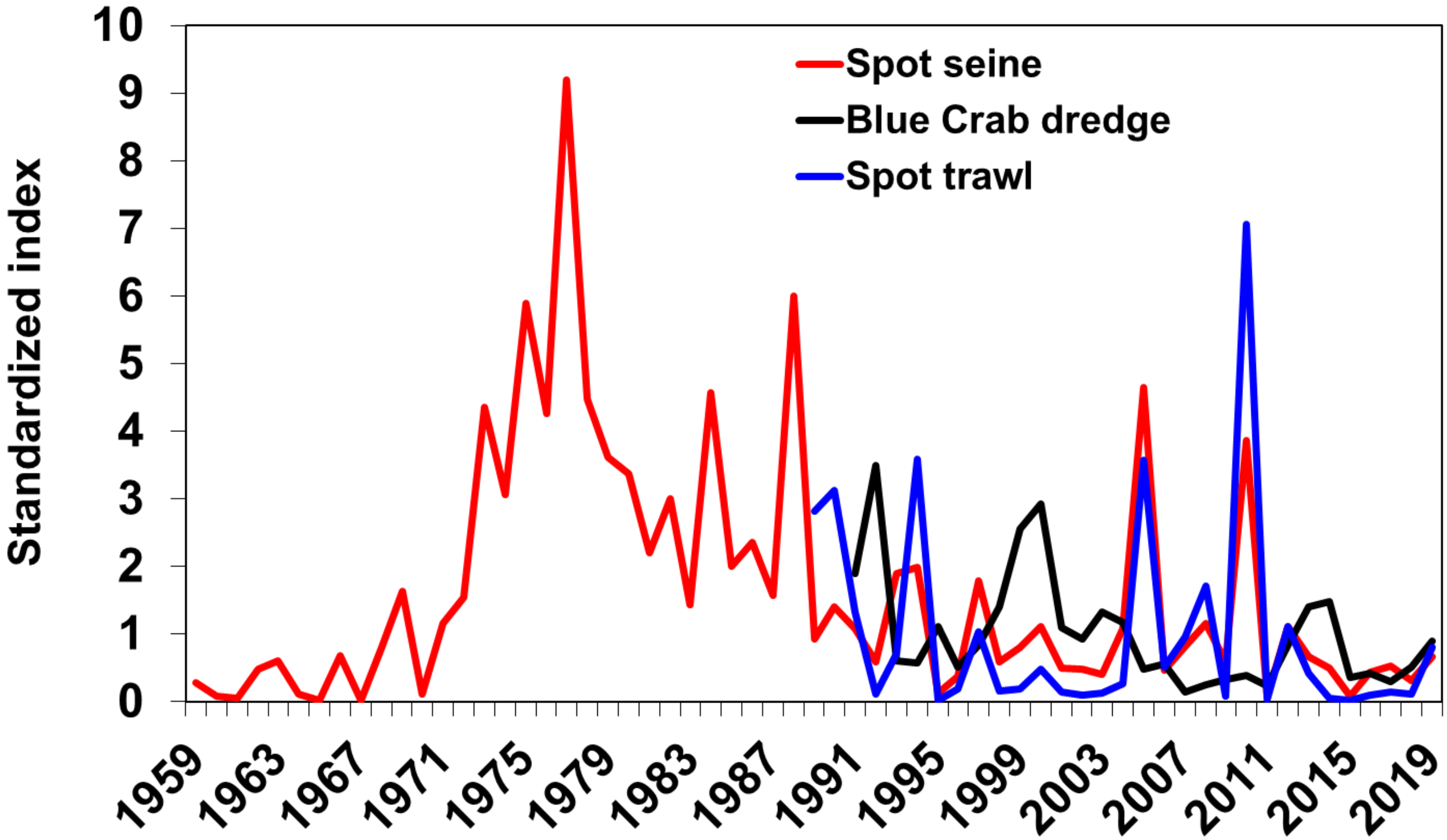

Figure 13. Maryland resident Bay Striped Bass annual abundance index (RI; MD MRIP recreational catch per private boat trip; mean = black line) during 1981-2019 and its 90% confidence intervals based on @Risk simulations of catch and effort distributions. Catch = number harvested and released.

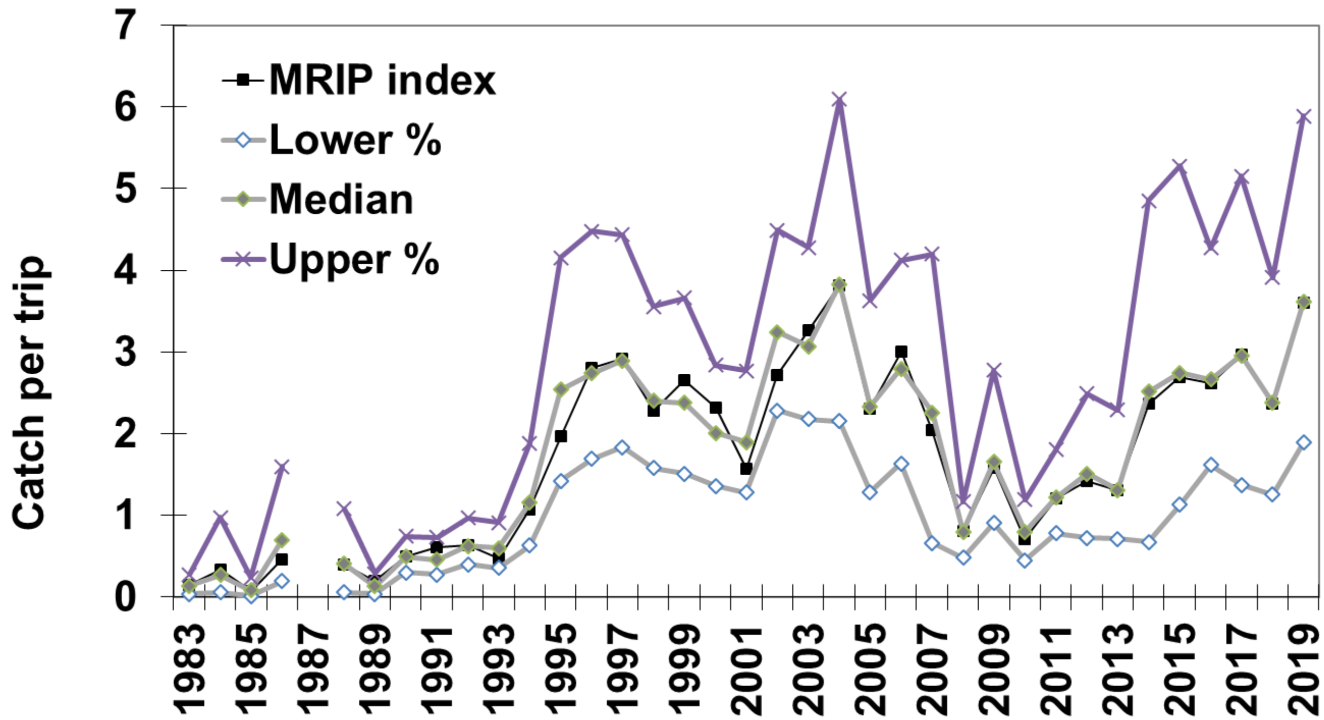

Figure 14. Comparison of trends of aggregate abundance of ages 2-5 Striped Bass estimated by the current stock assessment (N ages 2-5; NEFSC 2019) and the RI index (September-October catch per private / rental boat trip).

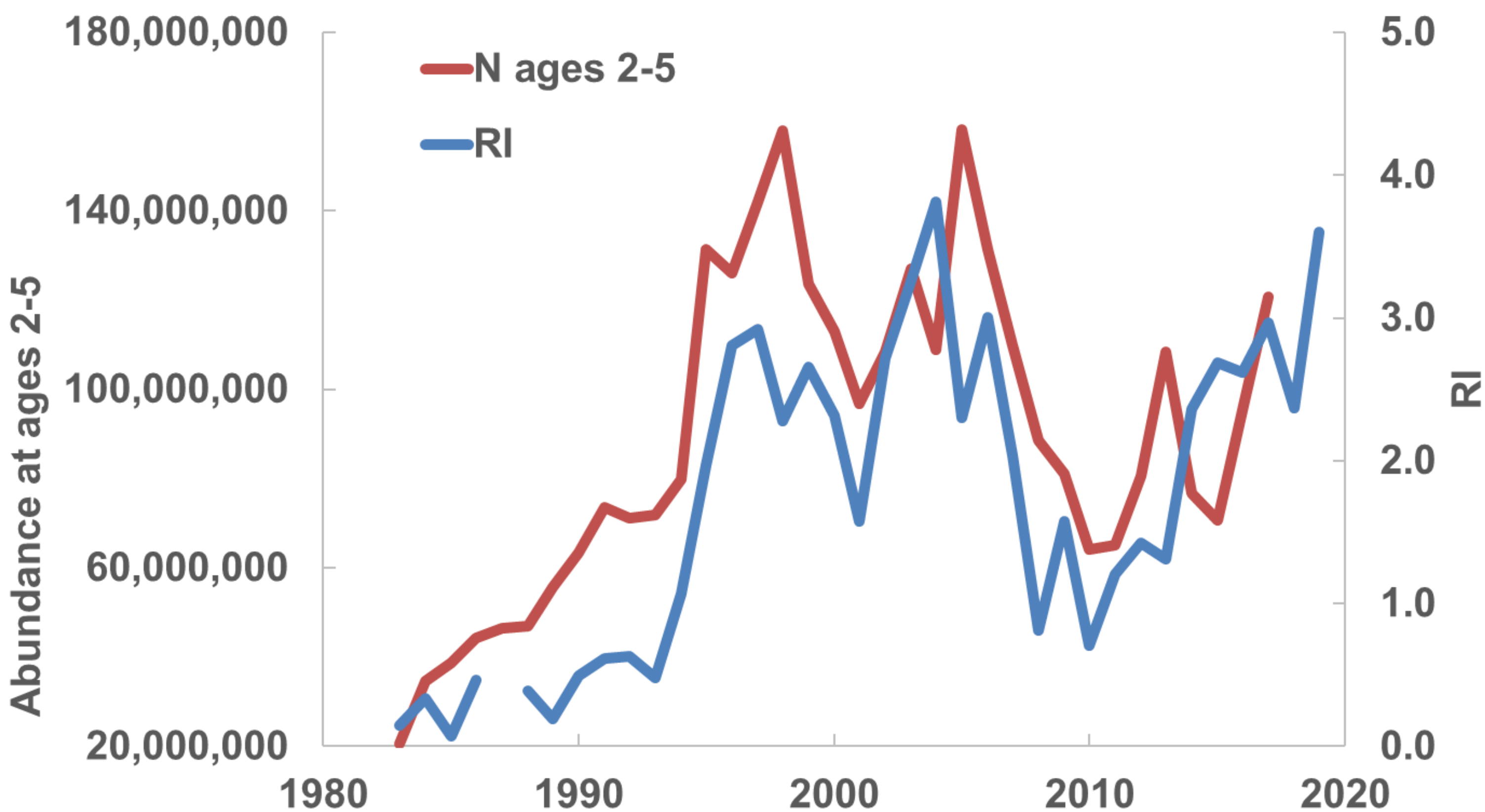

Figure 15. Atlantic Menhaden index to Striped Bass index (RI) ratios during 1983-2019 and their 90% confidence intervals based on @Risk simulations of Atlantic Menhaden seine indices and RI distributions. Note $\log_{10}$ scale on Y-axis.

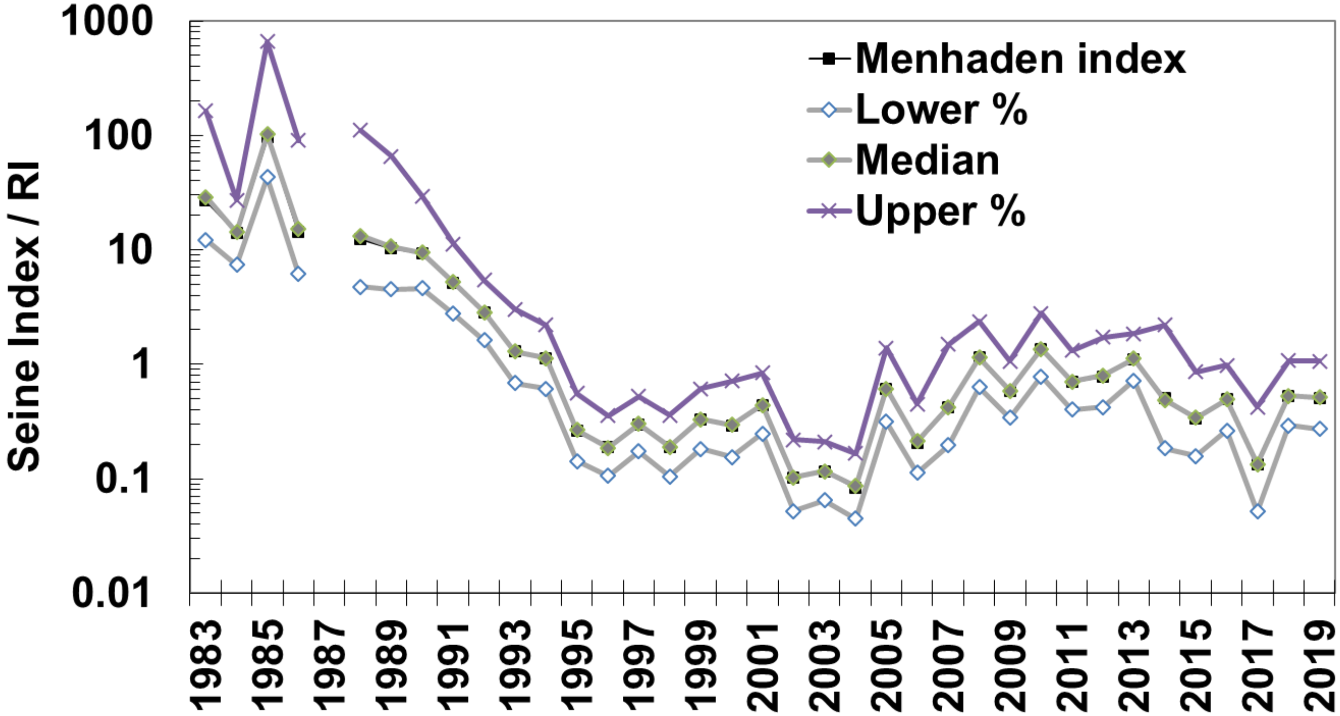

Figure 16. Bay Anchovy seine index to Striped Bass index (RI) ratios during 1983-2019 and their 90% confidence intervals based on @Risk simulations of Bay Anchovy seine indices and RI distributions. Note $\log_{10}$ scale on the Y-axis.

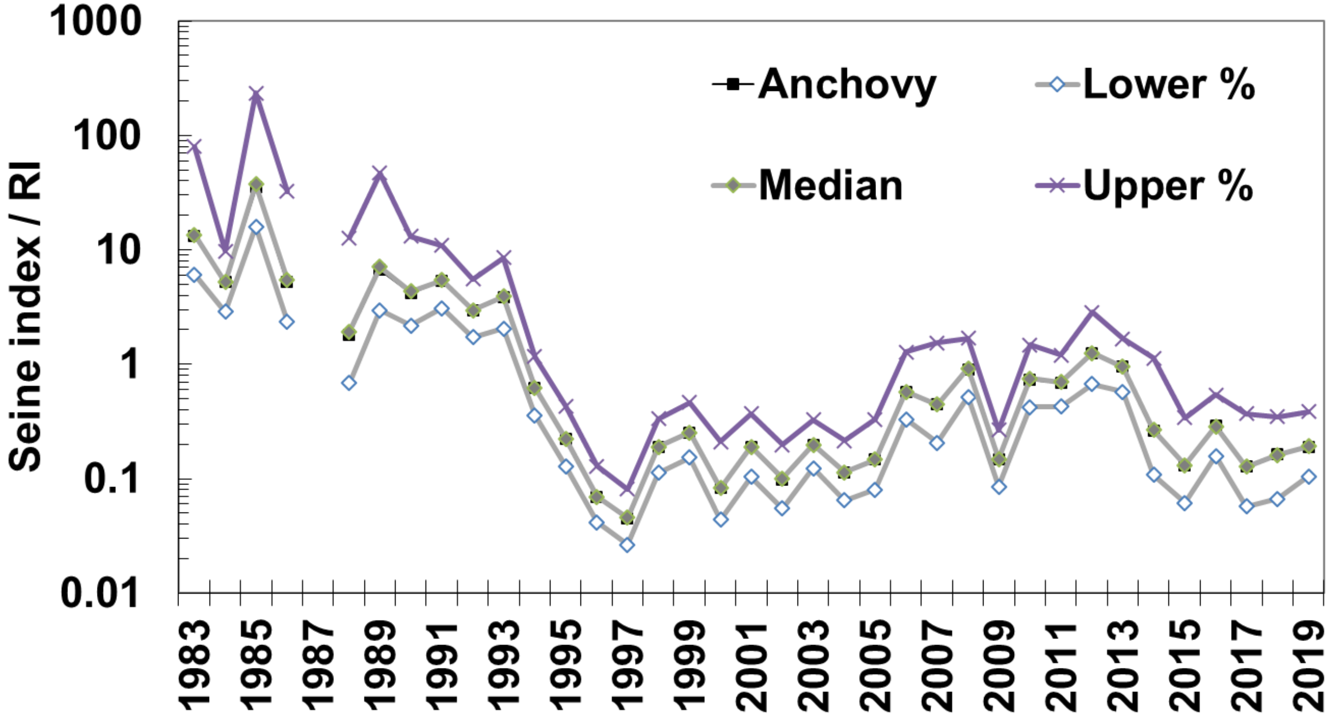

Figure 17. Bay Anchovy trawl index to Striped Bass index (RI) ratios during 1989-2019 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale on Y-axis.

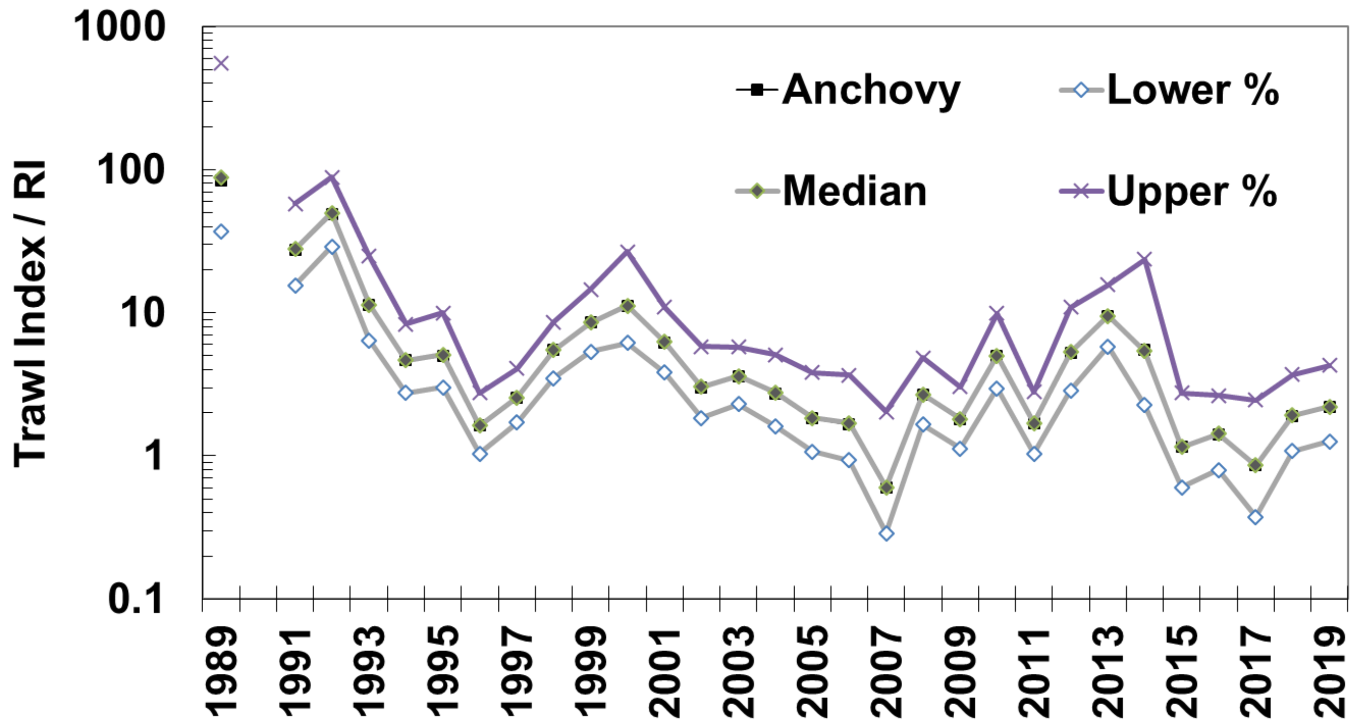

Figure 18. Spot seine index to Striped Bass index (RI) ratios during 1983-2019 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Spot seine indices and RI. Note $\log_{10}$ scale on Y-axis

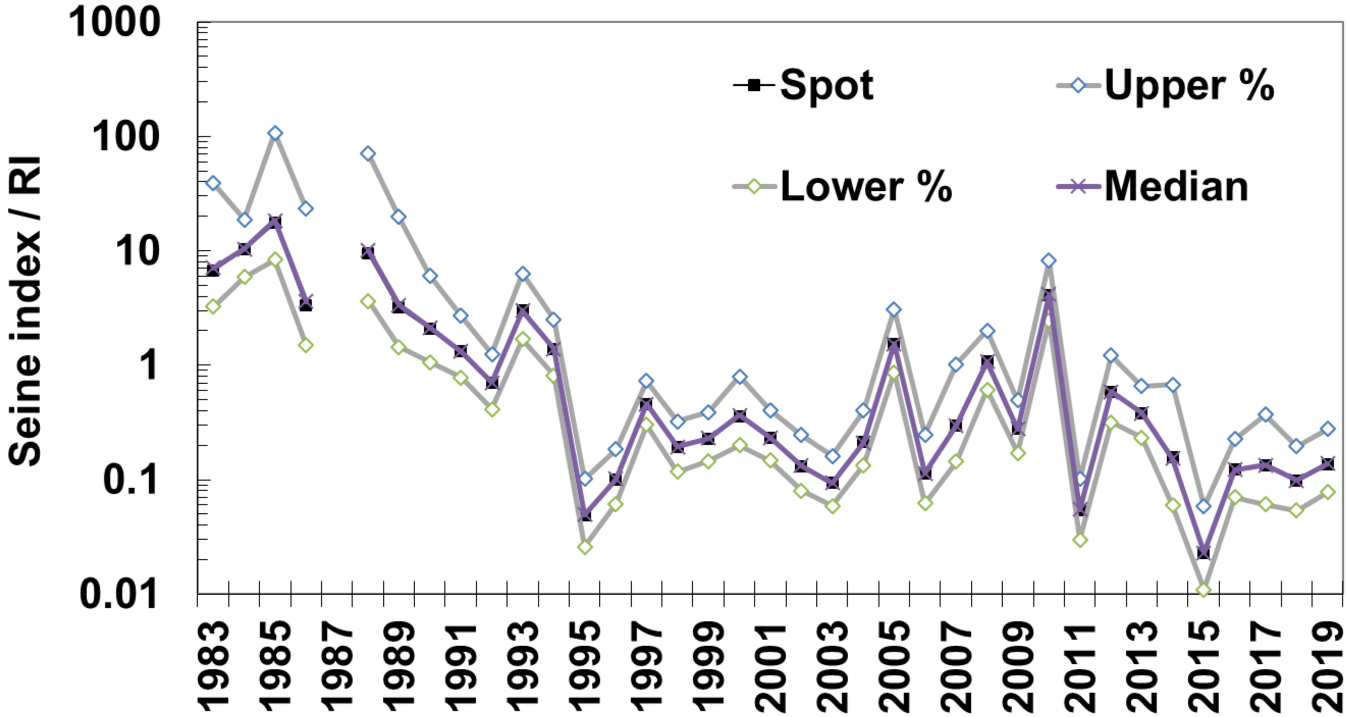

Figure 19. Spot trawl index to Striped Bass index (RI) ratios during 1989-2019 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale on Y-axis.

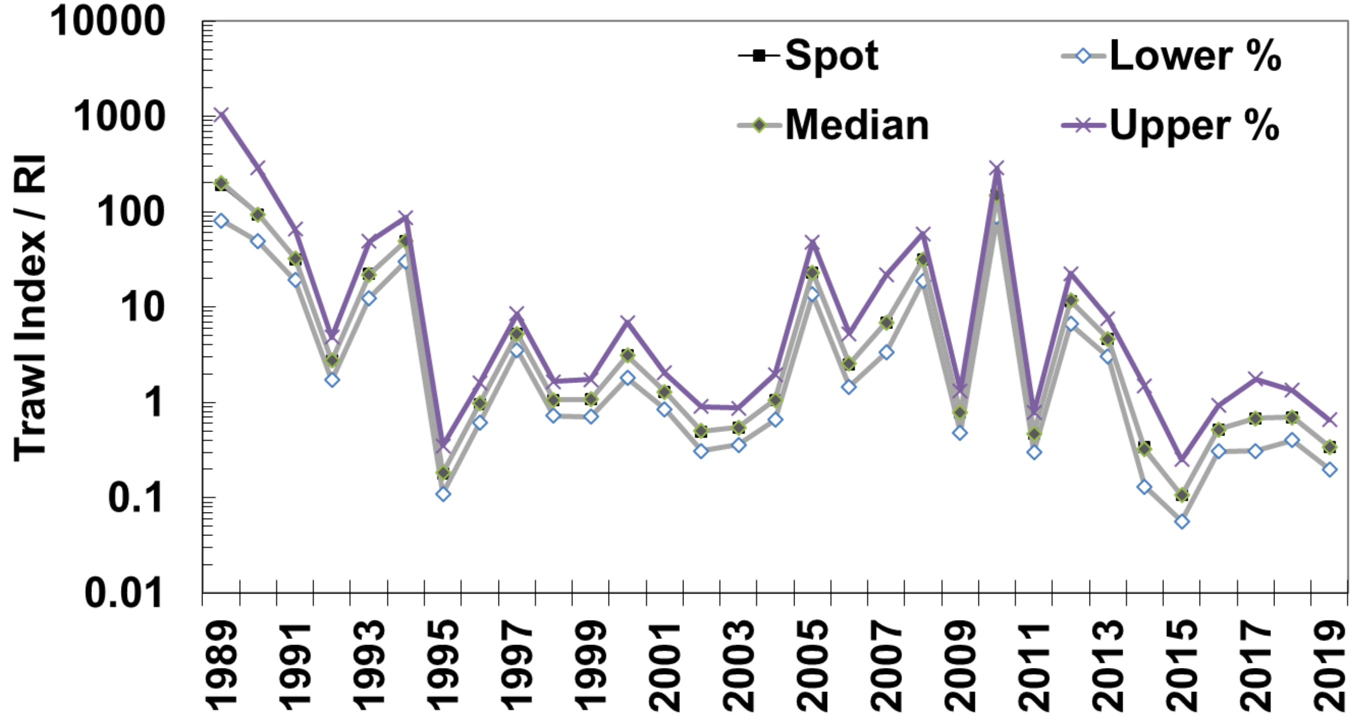

Figure 20. Blue Crab index to Striped Bass index (RI) ratios during 1989-2019 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Blue Crab (age 0) winter dredge densities and RI. Note $\log_{10}$ scale on Y-axis.

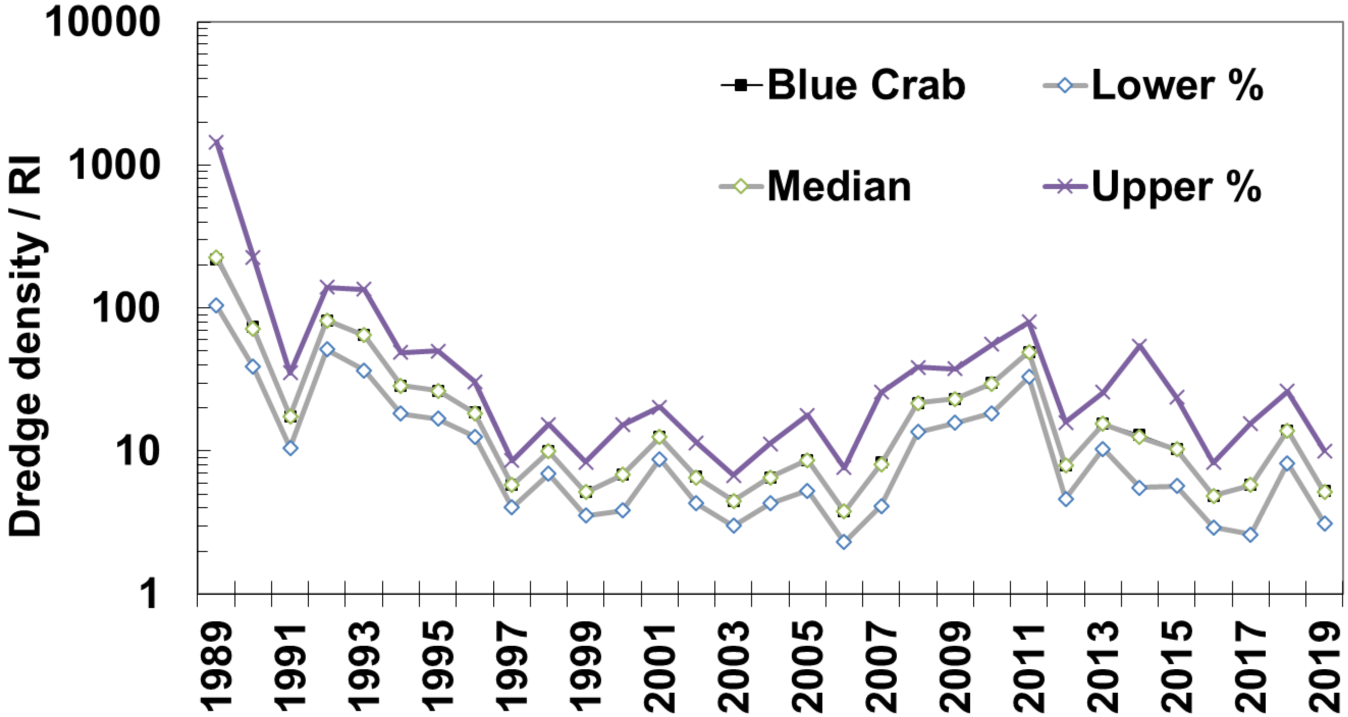

Figure 21. Trends of standardized ratios major upper Bay forage species indices to Striped Bass relative abundance (RI). Forage ratios have been standardized to their 1989-2019 mean to place them on the same scale. S indicates a seine survey index; T indicates a trawl survey index; and D indicates a dredge index. Note the $\log_{10}$ scale on Y-axis.

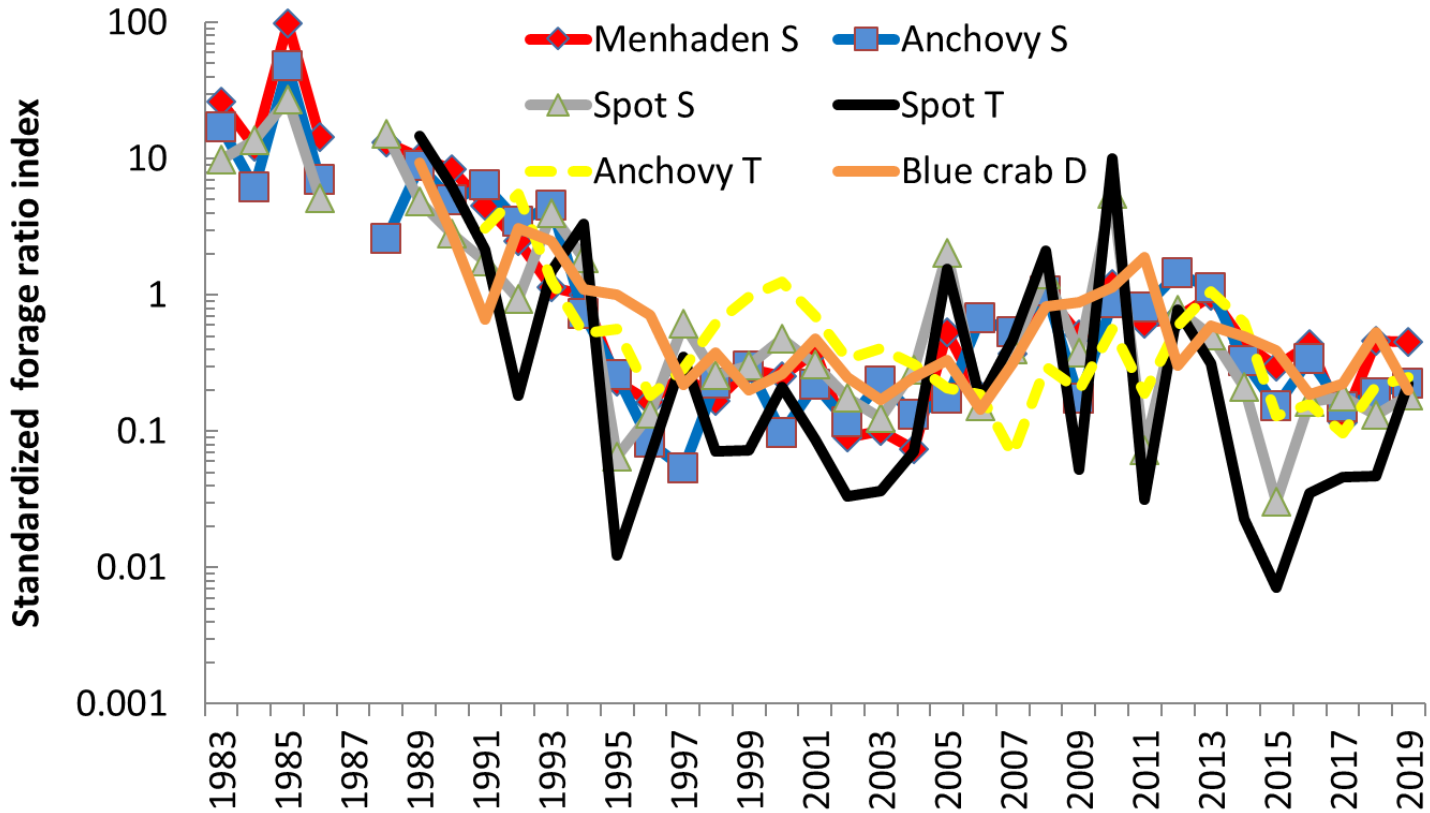

Figure 22. Standardized ratios of major forage indices / Striped Bass RI during the time period when body fat (P0) indices were available.

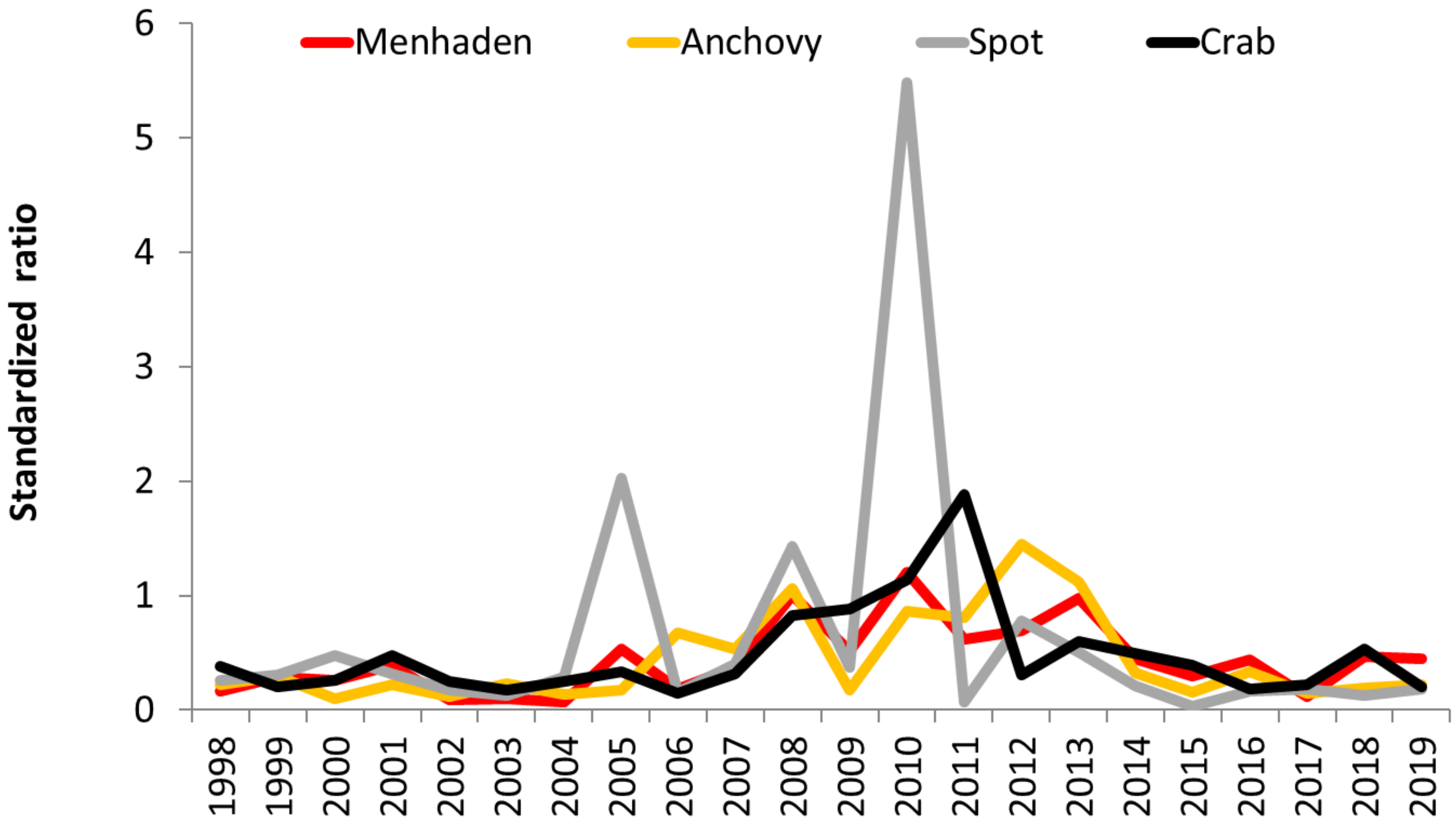

Figure 23. Mean of weighted standardized ratios of major forage indices and associated targets and thresholds during the time period when body fat (P0) indices were available.

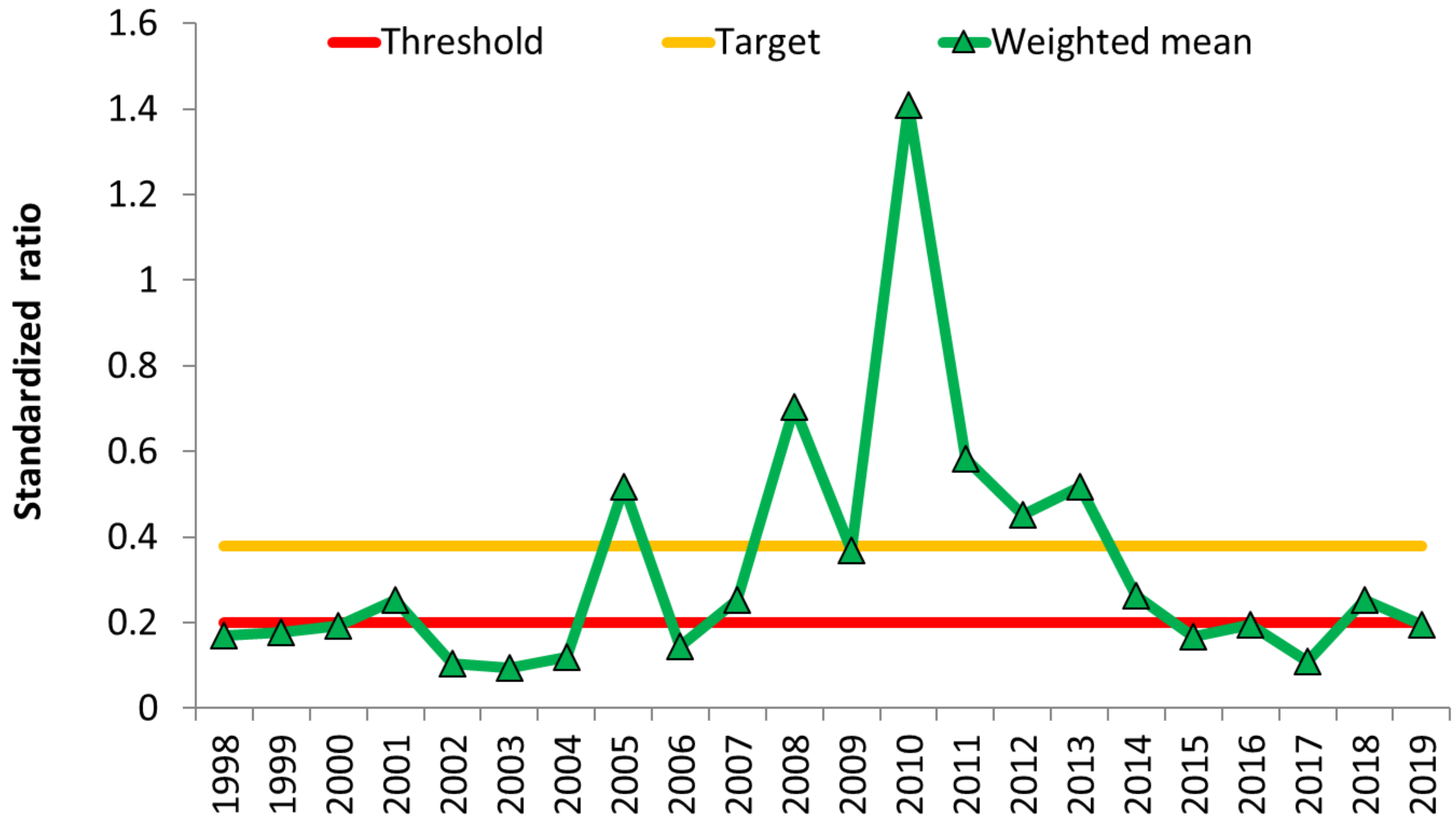

Figure 24. Time-series of age 3 Striped Bass relative abundance on two major Maryland spawning areas (hybrid index = index adjusted for changing catchability during 1985-1995; units = number of fish captured in 1000 square yards of net per hour) and abundance of age 3 Striped Bass along the Atlantic Coast estimated by the NEFSC(2019) statistical catch-at-age model. Hybrid index time series =1985-2019; Statistical catch-at-age model time-series = 1985-2017.

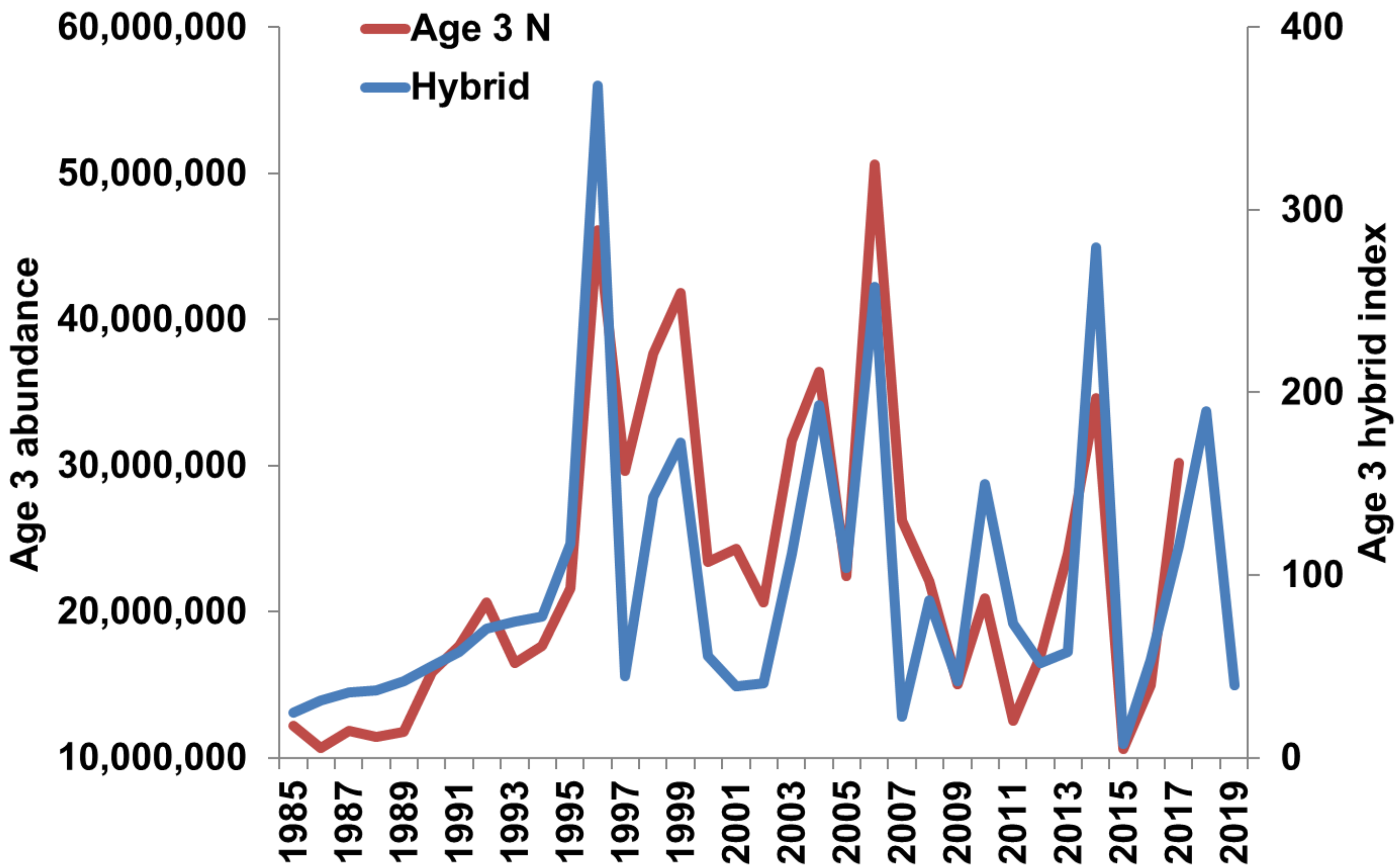

Figure 25. Relative survival (SR) of Striped Bass during 1985-2019 and 90% confidence intervals based on @Risk simulations of age 3 hybrid gill net indices divided by juvenile index (year-3) distributions.

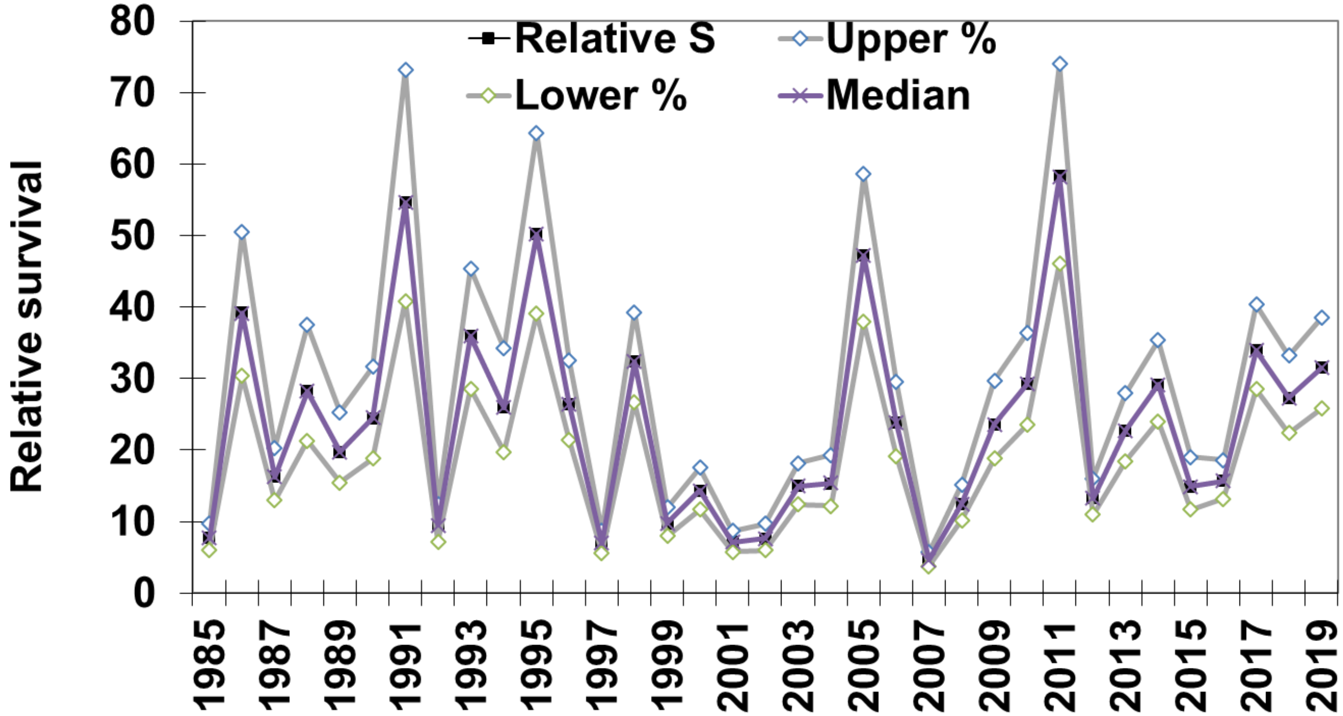

Figure 26. Relative survival of Striped Bass during 1985-2018 with targets and limits. Target = highest point of target P0 period (2008-2010). Threshold = highest point consistent with other points during threshold P0 period (1998-2004).

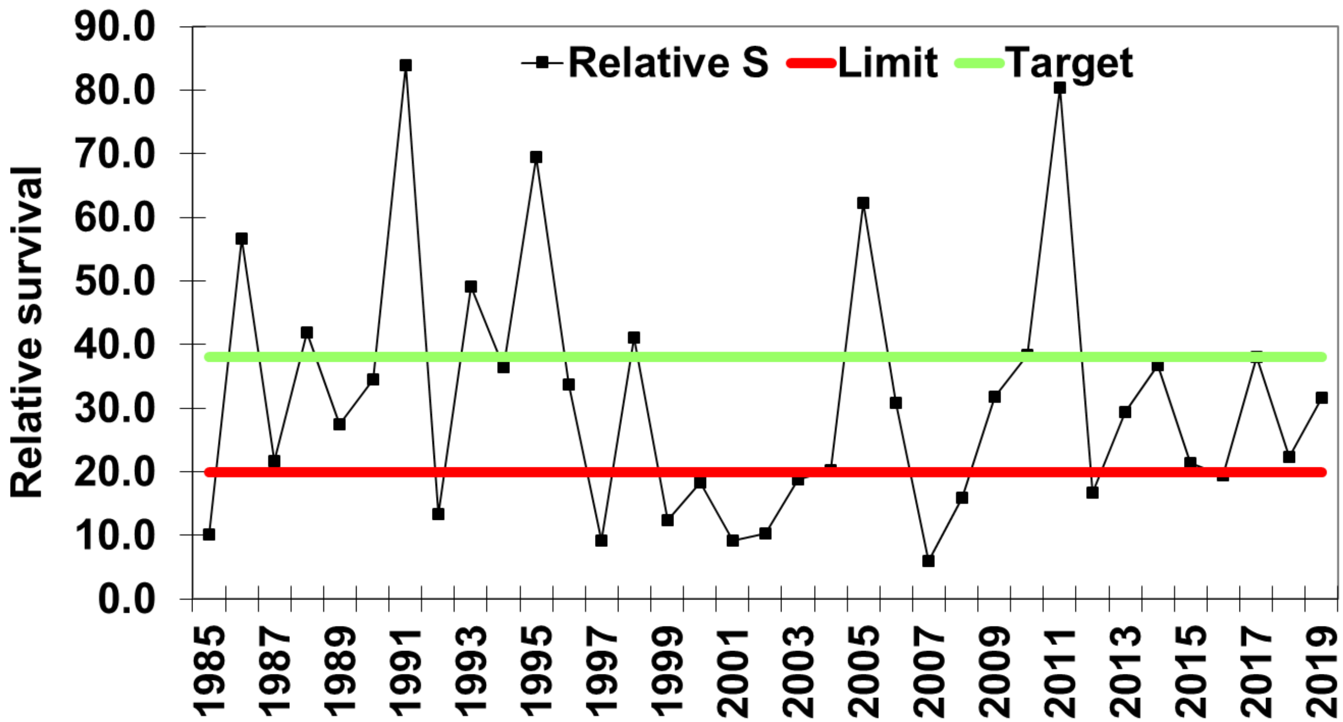

Figure 27. Index of Forage (IF) and its component scores. IF averages scores given to five indicators of forage status in upper Bay. A score of 3 indicates target conditions were met; 1 indicates threshold conditions; 2 indicates status in between. RI = index of relative abundance of resident Striped 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.

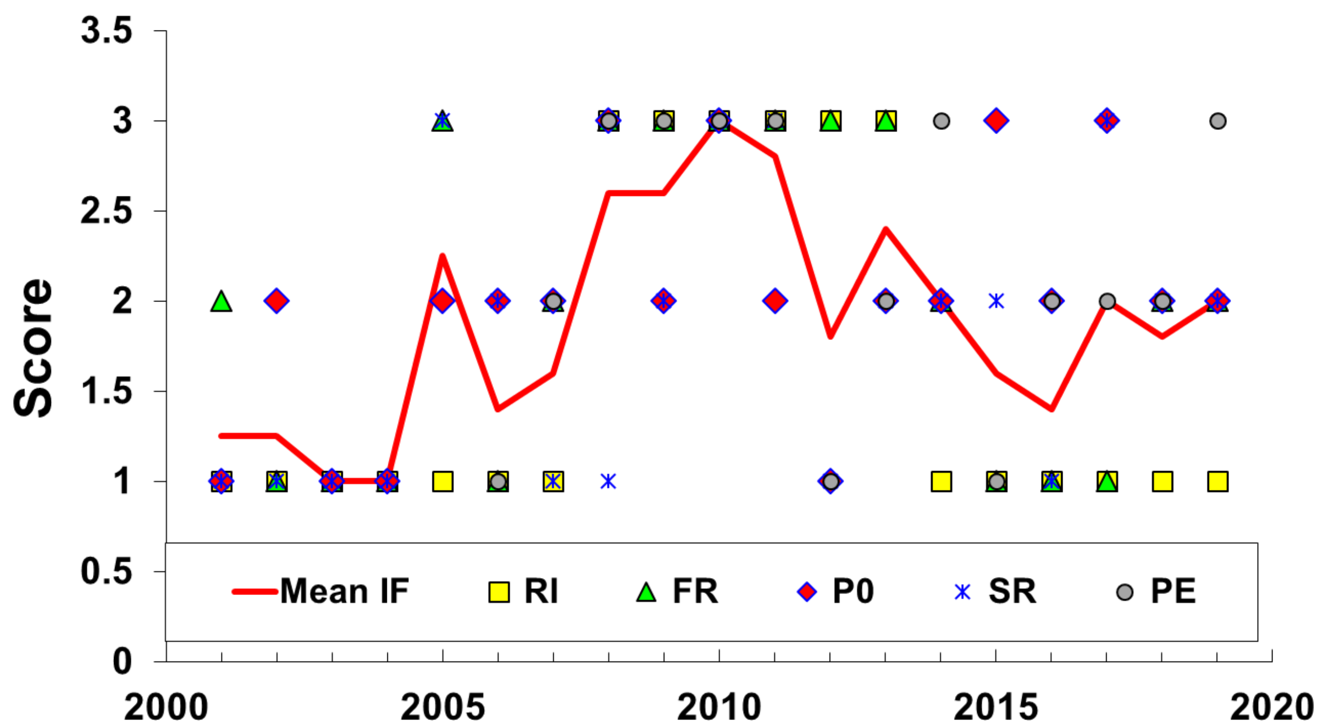

Figure 28. Forage index with all components averaged (Mean IF) and averaged with each component removed (leave one out average). Dashed lines indicate proposed IF target (at or above green dashed line) and threshold (at or below red dashed line) based on this approach. See Figure 30 for explanation of scores and abbreviations.