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**MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT
INVESTIGATIONS**



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Sampling	Volunteer / organization
Mapping	Marek Topolski MD DNR

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Report Organization

This report was completed during October, 2023. It consists of summaries of activities for Projects 1–4 under this grant cycle. All pages are numbered sequentially; there are no separate page numbering systems for each Project. Project 1 activities are reported in separate numbered sections. For example, Project 1, section 1 would cover development reference points (Project 1) for stream spawning habitat of anadromous fish (Section 1). Tables in Project 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 Project and section.

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

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MD – Marine and estuarine finfish ecological and habitat investigations
Project 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern

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Executive Summary

Purpose - Project 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.).

Spatial Analyses - We used property tax map-based counts of structures (C) in a watershed, standardized to hectares (C/ha), as our indicator of watershed development. Estimates of C/ha are converted to percent impervious surface (%IS) using an equation estimated by regression. Recent improvements to spatial resolution of land cover necessitated revisiting the relationship between C/ha and %IS. Land cover estimates became available at 1 m x 1 m resolution for the entire Chesapeake Bay watershed; resolution of land use data used in past reports to estimate %IS from C/ha had 30m x 30m resolution. A non-linear power function provided a very good fit to the high-resolution data and was used to predict %IS from C/ha. Estimates of C/ha that were equivalent to 5% IS, 10% IS, and 15% IS) were estimated as 0.31, 0.84, and 1.51 C/ha, respectively. A target level of development (C/ha \leq 0.31 or 5% IS) supports desirable production and habitat needed for recreational fisheries. We considered 0.84 C/ha or 10% IS as threshold level of development beyond which increasing fishery problems related to habitat will occur.

Section 1, Stream Ichthyoplankton - 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 ostensibly accounted for increased spawner abundance between

2005-2011 and 2012-2021, 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 50% over the range of observed C/ha (0.07-1.52).

Final data from Patuxent River sampling during 2021 (C/ha = 1.39) suggested localized impact of development on Herring spawning. Drift net stations, located above the tide line at Route 214 that were within or just below the developed Laurel-Bowie area had much lower P_{herr} (0.17; N = 36) than drift stations below Route 214 (P_{herr} = 0.58; N = 12), and collections in the nearby tidal, more rural portion of the watershed that were made by boat tows (P_{herr} = 0.90; N = 52). The Choptank River (C/ha = 0.18) and Tuckahoe Creek (C/ha = 0.07; both were agricultural watersheds), sampled in 2016 and 2017, had a similar sampling design as Patuxent River (drift nets upstream and boat samples downstream; Uphoff et al. 2017; 2018). Estimated P_{herr} in Choptank River was 0.74 during 2016 (N = 46) and 2017 (N = 43) where upstream drift nets were employed, and 1.00 (N = 54) in 2016 and 0.88 (N = 66) in 2017 where samples were taken by boat. Estimated P_{herr} in Tuckahoe Creek was 0.67 (N = 57) during 2016 and 0.72 (N = 58) in 2017 where upstream drift nets were used for collections, and 0.92 (N = 40) in 2016 and 0.77 (N = 44) in 2017 where boat tows were used for collections. While upstream drift net collections in these streams always had lower P_{herr} estimates, the difference was more extreme in Patuxent River than in Choptank River and Tuckahoe Creek. This speaks to greater stress from urbanization upstream in the Patuxent River. Since P_{herr} estimates of stream drift collections were more similar to boat tow collections in the area below Route 214 (Figure 1-10), it is possible there are also impediments that don't allow as many fish to get up to those reaches. Median conductivity of drift sites above Route 214 was much higher (414.2 $\mu\text{S}/\text{cm}$) than drift sites below (238.9 $\mu\text{S}/\text{cm}$), indicating higher anthropogenic impact and potential for greater harm and reduced survival. Higher standardized conductivity (up to about 1.6-times higher) 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.

Section 2: Estuarine Yellow Perch Larval Presence-Absence Sampling - Annual L_p , or the proportion of tows containing 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 was conducted in the upper tidal reaches of the Choptank and Sassafras Rivers in 2022. Sampling started the third week of March and the last week of March, respectively, and continued through the second week of May in the Choptank River, and the third week of April in the Sassafras River. Both watersheds are rural and agricultural; Sassafras River represents the only tidal-fresh, agricultural watershed in the L_p time-series.

Estimated L_p was determined annually from dates spanning the first day Yellow Perch larvae were caught up until the 18°C water temperature cutoff criterion was met. Previous did not have a temperature cutoff, so estimates were recalculated. Approximately 84% of old estimates were within $\pm 20\%$ of the new ones, median was 0%, and there was a slight tail out at positive values.

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. However, wetlands may be an important source of organic matter that influences Yellow Perch larval feeding success in some subestuaries.

Rural features (agriculture, forest, and wetlands) were negatively correlated with development in the watersheds monitored for L_p . 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 ≤ 0.59 . 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 $L_p = 0.74$) than agriculture (median $L_p = 0.50$) or development (median $L_p = 0.35$), but these differences may have also reflected dynamics unique to brackish or tidal-fresh subestuaries since all but one agricultural watershed had brackish subestuaries, and nearly all forested watersheds had tidal-fresh subestuaries.

Section 2.1: Investigation of Striped Bass spawning and larval habitat status in Maryland
– This survey did not cover the general hypotheses about development. Rather, it described general habitat conditions for Striped Bass and reflected increasing concern about their status after a series of poor year-classes in Maryland spawning areas. Water temperature and flow conditions are important influences on year-class success of Striped Bass and we examined changes in these parameters. Toxic water quality conditions encountered by Striped Bass larvae were implicated in episodic mortalities in some spawning areas (Choptank River, Nanticoke River, and possibly Potomac River) in the 1980s. During 2014-2021, we collected basic water quality data (temperature, conductivity, dissolved oxygen or DO, pH, and alkalinity) and contrasted them with conditions during the 1980s.

We updated the proportion of tows with Striped Bass eggs (Ep) index and the Maryland baywide juvenile index (JI), and then estimated relative larval survival (RLS, baywide JI / Ep) through 2022. Estimated baywide Ep , based on Choptank River Ep (0.69), was not in the top tier of estimates (roughly 0.80 or greater), but there was a high chance it was above levels during 1982-1988 when spawning stock was depleted enough to affect year-class success. Estimates of RLS in 2022 were similar to 2019-2021 and slightly above a poor survival criterion.

We examined Striped Bass egg and larvae survey data during 1954-2022 for four spawning milestones in the Choptank River and Nanticoke River time-series that were reasonably straightforward to interpret: date that the first egg was collected, and the dates when 12°C, 16°C, and 20°C (low temperature, peak temperature, and high temperature for spawning and egg-larval survival) were consistently met. The first egg was collected on March 22, 2022 - the earliest date on record. Water temperatures were already above 12°C (13.4-13.8°C) and remained there on March 25. Two substantial temperatures drops were evident between March 25 and April 20. The 16°C milestone was reached on April 16, 2022, three days shy of the time-series median. Temperatures rebounded after April 20, but did not reach the 21°C cutoff during the survey. Eggs were only present once during surveys on May 11 and 13 and these dates were not included.

We updated average annual 2-month flows estimated for spawning periods for the Head-of-Bay, Potomac River, Choptank River, and Nanticoke River. Flows standardized to average 1957-2020 flows (standardized flow = 1.0) were above average in Choptank and Nanticoke rivers during 2021 (1.35 and 1.25, respectively), and below average in Head-of-Bay (0.73) and Potomac River (0.58). It was near average in Head-of-Bay (1.04) and Choptank (0.98) rivers during 2022 and below average in Potomac (0.58) and Nanticoke rivers (0.64). Spawning area standardized flows appear to have shifted downward after 2011; above average flows have been lower during 2012-2022 than during 1993-2011 while below average flows were similar during the two periods. Above average flows resulted in a higher chance that strong year-classes would be formed and a modest reduction in occurrence of poor year-classes.

Measurements of pH in Choptank River during April 1-May 8 between 1986-1991 and 2013-2022 indicated improvement (higher averages and less variability of individual measurements) that would have lowered toxicity of metals implicated in high larval mortality in some Striped Bass spawning areas during the 1980s. Average alkalinity was at least 3-times higher in 2021-2022 compared to 1986-1991. It seems unlikely that poor year-class success during 2019-2022 could be attributed to a return of toxic water quality conditions implicated in poor recruitment during the 1980s.

Section 3 - Estuarine Fish Community Sampling – Sampling of juvenile and adult habitat occurred during summer and dissolved oxygen (DO) was the primary environmental response variable to land use. Sampling during 2003-2022 has resulted in 162 subestuary and year combinations; 91 of these combinations have been in mesohaline subestuaries, 18 have been in oligohaline, and 53 have been tidal-fresh.

Correlation analyses of bottom DO with temperature and C/ha in subestuaries sampled since 2003 indicated that bottom 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 (threshold level) when development was beyond its threshold; occupation of bottom channel habitat diminishes at or below threshold DO. Mean bottom DO did not decline in oligohaline or tidal-fresh subestuaries. The extent of bottom channel habitat that can be occupied by fish does not diminish with development in tidal-fresh and oligohaline subestuaries due to low DO. However, more localized, or episodic habitat issues such as harmful algal blooms, ammonia toxicity, and depletion of DO in thick SAV beds seem to be important.

Median bottom DO in mesohaline subestuaries increased as agricultural coverage went from 3 to 39% and the DO trend appeared to be stable or slightly declining when agricultural coverage was 43-72%. A dome-shaped quadratic model of median bottom DO and agricultural coverage that did not account for regional differences fit the data well. Below threshold median bottom DO was predicted when agricultural coverage fell below 18%. Median bottom DO was predicted to peak at about 50% agricultural coverage and modest declines in bottom DO would occur through 72% of their watershed covered in agriculture. Predicted median bottom DO at the highest level of agriculture observed would equal 4.3 mg/L, between the DO target and threshold. Residuals suggested that the predications at the highest coverage ($\geq 65\%$) may have been negatively biased. Agricultural coverage and C/ha were strongly and inversely correlated, so the positive trend of DO at low agricultural coverage was likely to reflect development's negative impact.

Occupation of bottom channel habitat by fish was influenced by watershed development and subestuary salinity type. Mesohaline subestuaries over the threshold level of development exhibited depletion of bottom DO below 3 mg/L and abundance and species richness in bottom trawl samples exhibited a negative threshold response. DO did not exhibit noticeable change with development in shore zone seine samples for all three salinity types nor were there negative changes DO in bottom channel habitat of tidal-fresh and oligohaline subestuaries. Episodes of abnormally low abundance of fish or fish kills occurred in tidal-fresh and oligohaline subestuaries, but they were not related to low DO.

In 2022, we evaluated summer nursery and adult habitat for recreationally important finfish in Tred Avon River (mesohaline; C/ha = 0.79), Mattawoman Creek (tidal-fresh; C/ha = 1.00), Northeast River (tidal-fresh; C/ha = 0.51), and South River (mesohaline; C/ha = 0.79). All subestuaries sampled in 2022 had a lower summer median bottom DO than their time-series

median. Tred Avon River median summer bottom DO in 2022 fell below the target level of 5 mg/L, while South River median summer bottom DO fell below the threshold level of 3 mg/L. In 2022, Tred Avon River had the greatest variation in bottom DO measurements, ranging from 1.33 mg/L to 6.33 mg/L; Northeast River, ranged from 4.25 mg/L to 8.76 mg/L; Mattawoman Creek, ranged from 4.56 mg/L to 8.17 mg/L; and South River, ranged from 0.31 mg/L to 3.96 mg/L.

Tred Avon River bottom channel trawl catches fell to their 5th lowest level for all years sampled, while inshore seine catches were lower than previous years but not the lowest. South River, sampled in 2022, had the lowest bottom trawl GM rank out of all four years sampled and ranked 81st out of 90 mesohaline subestuaries. Northeast River, a tidal-fresh system, had its lowest ranking trawl GM estimate in 2022 and ranked 29th out of 38 tidal-fresh systems sampled since 2023. Mattawoman Creek, a tidal-fresh system, had its lowest ranking trawl GM estimate in 2022 since 2009, and ranked 32nd out of 38 tidal-fresh systems. To some extent, these rankings depend on whether the varying time-series contain good year-classes of non-marine target species; longer time-series (5 – 12 years) had higher rankings.

Common Background for Project 1, Sections 1-3

Jim Uphoff

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

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

Forests and wetlands in the Chesapeake Bay watershed have been converted to agriculture and residential areas to accommodate increased human populations since colonial times (Brush 2009). These watershed alterations have affected major ecological processes and have been most visibly manifested in Chesapeake Bay eutrophication, hypoxia, and anoxia (Hagy et al. 2004; Kemp et al. 2005; Fisher et al. 2006; Brush 2009). Human population growth since the 1950s added a suburban landscape layer to the Chesapeake Bay watershed (Brush 2009) that has been identified as a threat (Chesapeake Bay Program or CBP 1999). Land in agriculture has been relatively stable, but fertilizer and pesticide use 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 in attempts 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 this land use on sportfish habitat.

Project 1 investigates two general alternative hypotheses relating recreationally important species to development and 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 life 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; Hughes et al. 2014a; 2014b). 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; Kaushal et al. 2018; Baker et al. 2019) 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.

Measurable adverse changes in physical and chemical characteristics and living resources of estuarine systems have occurred at IS of 10-30% (Mallin et al. 2000; Holland et al. 2004; Uphoff et al. 2011; Seitz et al. 2018; Uphoff et al. 2022). 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 Maryland's tidal Yellow Perch management plan; MD DNR 2017), and summer habitat in tidal-fresh subestuaries (Uphoff et al. 2015). Conserving 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. 2022). A preliminary estimate of IS in Maryland's portion of the Chesapeake Bay watershed in 2020 equaled 9.6%. 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 (Topolski 2015; Uphoff et al. 2020; see **General Spatial and Analytical Methods used in Project 1, Sections 1-3**). Counts of structures per hectare (C/ha) had strong relationships with IS (Uphoff et al. 2022). 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 0.31, 0.84, and 1.51 C/ha, respectively (Uphoff et al. 2022). Tax map data provide a development time-series that goes back to 1950, making retrospective analyses possible (Uphoff et al. 2020). Development in Maryland's portion of the Chesapeake Bay watershed, approximately 0.17 C/ha in 1950, reached 0.81 C/ha in 2020 (the latter is a preliminary estimate; M. Topolski, MD DNR, personal communication).

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; Uphoff et al. 2020). 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 suburban landscapes in mesohaline Chesapeake Bay subestuaries (Uphoff et al. 2011; Uphoff et al. 2022). 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; Roman et al. 2019; Slater et al. 2020). 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).

A hypoxia based hypothesis, originally formed to explain die-offs of large adult Striped Bass in southeastern reservoirs, links increased natural mortality and deteriorating condition in Chesapeake Bay through a temperature-oxygen squeeze (mismatch of water column regions of desirable temperature and dissolved oxygen in stratified Chesapeake Bay during summer; Coutant 1985; Price et al. 1985; Coutant 1990; Coutant 2013). Constantini et al. (2008), Kraus et al. (2015), and Itakura et al. (2021) examined the impact of hypoxia on 2 year-old and older Striped Bass in Chesapeake Bay through bioenergetics modeling and acoustic tagging and concluded that a temperature-oxygen squeeze by itself was not limiting for Striped Bass. However, Groner et al (2018) suggested that Striped Bass are living at their maximum thermal tolerance and that this is driving increased mycobacteriosis and associated mortality. Adequate levels of Striped Bass prey can offset negative effects of warm temperatures and suboptimal dissolved oxygen in reservoirs (Thompson et al. 2010; Coutant 2013).

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General Spatial and Analytical Methods used in Project 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 for us by M. Topolski (MD DNR). Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MD DOP 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. 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 a 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 through 2017.

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 (Uphoff et al. 2020). 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 (Uphoff et al. 2020). The same watersheds and years used to estimate the original nonlinear relationship were used in the update to maintain continuity. These estimates were used from 2018-2021.

The requirements of the Environmental Protection Agency's Chesapeake Bay total maximum daily load require precision planning not possible using the coarse resolution (30m • 30m) of Landsat TM data (Uphoff et al. 2022). Chesapeake Conservancy's Conservation Innovation Center was contracted by the Chesapeake Bay Program to develop high-resolution, 1m • 1m, land cover data for the Chesapeake Bay watershed. We updated our estimates of C/ha that were equivalent to 5%, 10%, and 15% impervious surface benchmarks for fisheries management advice in Maryland's portion of the Chesapeake Bay using this high resolution data set. The land cover raster created was a composite of LiDAR, imagery, and land cover data having varied spatial resolutions for the years 2013/2014. Models developed from the high-resolution Chesapeake Conservancy 2013/2014 data indicated that the C/ha to %IS relationship was best described by a nonlinear power function across a broad range of land development. The equation that best describe the relationship was

$$\%IS = 11.255 \cdot (C/ha)^{0.698}.$$

The C/ha equivalents for 5%, 10%, and 15% IS were 0.31, 0.84, and 1.51, respectively (Uphoff et al. 2022). These estimates of C/ha are now used as development reference points for fisheries management advice in Maryland's portion of Chesapeake Bay.

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 commonly 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 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 Littell 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 Littell 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 Littell 2006) was used for single variable linear regressions. Multiple linear and quadratic regressions were analyzed with Proc REG in SAS (Freund and Littell 2006).

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

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

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

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

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

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

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

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MD – Marine and estuarine finfish ecological and habitat investigations
Project 1: Development of habitat-based reference points for recreationally important
Chesapeake Bay fishes of special concern
Section 1: Stream Ichthyoplankton Sampling

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

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 impervious surface reference points (ISRPs; Uphoff et al. 2011) or analogous structure per hectare (C/ha) reference points (Uphoff et al. 2022) developed for Chesapeake Bay subestuaries 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; O'Dell and Mowrer 1984; Mowrer and McGinty 2002; Uphoff et al. 2020). 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 sampling methods of O'Dell et al. (1975; 1980) and O'Dell and Mowrer (1984) 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), Chester River (2019), and Patuxent River (2021; 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). The Patuxent River, located within the Piedmont and Coastal Plain provinces, is a major tributary of the Chesapeake Bay and is the largest river that is located entirely within the state of Maryland. The upper portion of the drainage (MD Route 214 north,

including the Little Patuxent River drainage) is located between Washington, DC and Baltimore, while the middle portion of the drainage (MD Route 214 south to Hall Creek) extends through Anne Arundel, Prince George's, and Calvert counties (O'Dell and Mowrer 1984; Figure 1-1). The Patuxent River is urbanized, with extensive development that has affected water quality and physical characteristics of the system (O'Dell and Mowrer 1984; Uphoff et al. 2018; Table 1-1; Figure 1-1).

We developed two indicators of anadromous fish spawning in a watershed based on presence-absence of eggs and-or larvae: occurrence at a site (a spatial indicator) and proportion of samples with eggs and-or 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) and O'Dell and Mowrer (1984). 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's eggs and-or larvae was compared to level of development (C/ha) and conductivity, an indicator of freshwater 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 eggs and-or larvae 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 2005-2021 were typically at road crossings that O'Dell et al. (1975; 1980) and O'Dell and Mowrer (1984) determined were anadromous fish spawning sites during the 1970s and 1980s. O'Dell et al. (1975; 1980) and O'Dell and Mowrer (1984) summarized spawning activity as the presence of an anadromous fish species (White Perch, Yellow Perch, or Herring) group's egg, larva, or adult at a site sampled with stream drift ichthyoplankton nets and wire traps.

All collections during 2005-2019 and 2021, with the exception of Deer Creek during 2012-2015, Choptank River and Tuckahoe Creek during 2016-2017, Patapsco River during 2013-2017, Chester River during 2019, and Patuxent River during 2021 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. Patuxent River (12 sites; Figure 1-10) was sampled during 2021 to provide information for the Anne Arundel County comprehensive growth plan. Table 1-2 summarizes sites, dates, and sample sizes in Mattawoman, Piscataway, Deer, and Tuckahoe creeks, and Bush, Choptank, Patapsco, Chester, and Patuxent rivers during 2005-2019 and 2021.

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, and in Patuxent River in 2021, were made using stream drift nets at wadable 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). These techniques mimic collections made by O'Dell et al. (1980) within the Choptank River drainage, specifically Tuckahoe Creek, and by O'Dell and Mowrer (1984) within the Patuxent River drainage. For both types of collections, 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 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 ($\mu\text{S}/\text{cm}$), and dissolved oxygen (DO, mg/L) were recorded at each site using either a hand-held YSI Model 85 meter or YSI Pro2030 meter. Meters were calibrated for DO each day prior to use. All data were recorded on standard field data sheets and double-verified at the site during volunteer collections. Approximately 2-ml of rose bengal dye was added to each sample in order to stain the organisms pink to aid sorting.

Ichthyoplankton samples were sorted in the laboratory by project personnel. All samples were rinsed with water to remove formalin and placed into a white sorting pan. Samples were sorted systematically (from one end of the pan to another) under a 10x bench magnifier. With the exception of 2018, all eggs and-or larvae were removed and retained in a small vial with a label (site, date, and time) and stored with 20% ethanol for later identification under a microscope. Each sample was systematically sorted a second time for quality assurance (QA).

Any additional eggs and-or larvae found were removed and placed in a vial with a label (site, date, time, and QA) and stored with 20% ethanol for identification under a microscope. All eggs and larvae found during sorting (both in original and QA vials) were identified as either Herring (Blueback Herring, Alewife, and Hickory Shad), Yellow Perch, White Perch, unknown (eggs and-or larvae that were too damaged to identify) or other (indicating another fish species) and the presence or absence of each of the above was recorded. The three Herring species' eggs and larvae are very similar (Lippson and Moran 1974) and identification to species can be problematic. American Shad eggs and larvae would be larger at the same stages of development than those identified as Herring (Lippson and Moran 1974) and none have been detected in our previous surveys.

Collections and sample processing were adjusted in 2018 due to 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 that 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 Project 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 24,329 ha and estimated C/ha increased from 0.87 to 0.97 during 2008-2018; Piscataway Creek's watershed was 17,536 ha and estimated C/ha increased from 1.41 to 1.51 during 2008-2014; Bush River's watershed was 35,956 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 37,612 ha and estimated C/ha was 0.24 during 2012-2015 (Table 1-1). The upper portion of the Choptank River (watershed area = 38,263 ha and developmental level = 0.18 C/ha) and a tributary of the Choptank River, Tuckahoe Creek (watershed area = 39,272 ha and developmental level = 0.07 C/ha), were added in 2016-2017; and the Chester River drainage (watershed area = 77,357 and developmental level = 0.14 C/ha) was sampled in 2019. These three systems are all spawning streams with high agricultural influence and low watershed development. The Patuxent River (watershed area = 99,960 ha and developmental level = 1.39 C/ha) was sampled in 2021 (Table 1-1; Figure 1-1). Deer Creek, Choptank River and Tuckahoe Creek, Chester River, and Patuxent River (all upper river, and two middle river, stream drift net sites) were sampled by DNR biologists from the Fishery Management Planning and Fish Passage Programs. Four middle Patuxent River collections were made by boat by DNR biologists from the Fish Health and Hatcheries, Anadromous Species Division, at no charge to this grant. Patapsco River's watershed equaled 93,728 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 was measured 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 (although they can provide within watershed low versus high development comparisons for spawning streams in years when all were sampled). Conductivity was measured with each sample and summarized in Deer Creek in 2012-2015, in the Choptank River and Tuckahoe Creek in 2016-2017, in the Patapsco River in 2013-2017, in the Chester River drainage in 2019, and in the Patuxent River in 2021 (Table 1-3).

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 (Tables 1-4 to 1-12). 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; O'Dell and Mowrer 1984) as evidence of spawning. Raw data from early 1970s and 1980s 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 (Table 1-4). Hall et al. (1992) collected ichthyoplankton with 0.5 m diameter plankton nets (3:1 length to opening ratio and 363 μ 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 (Table 1-5; O'Dell et al. 1975), Deer Creek in 1972 (Table 1-7; O'Dell et al. 1975), Bush and Patapsco Rivers in 1973 (Tables 1-6 and 1-10, respectively; O'Dell et al. 1975), Tuckahoe Creek in 1976-77 (Table 1-9; O'Dell et al. 1980), Chester River in 1975-1977 (Table 1-11; O'Dell et al. 1980), and Patuxent River in 1980-1983 (Table 1-12; O'Dell and Mowrer 1984).

The proportion of samples where Herring eggs and-or larvae were present (P_{herr} ; described below) 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), Chester River drainage (2019), and Patuxent River (2021). 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 proportion of sample estimates with reasonable precision. Mainstem stations (PC1-PC3) and Tinkers Creek (PTC1) were used in Piscataway Creek (Table 1-5; Figure 1-3). Only sites in streams that were sampled in all years (sites in common) in the Bush River drainage were analyzed (Table 1-6; 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 (Table 1-7; 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 (going from downstream to upstream, CH100-CH111; Table 1-8; Figure 1-6) were added in that system for analysis. Tuckahoe Creek stations TUC101, TUC102,

TUC103, TUC108, TUC109, and TUC110 (going from downstream to upstream) correspond to O'Dell et al. (1980) sites 4, 5, 6, 8, 11, and 12 respectively (Table 1-9; 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 (Table 1-10; Figure 1-8). All sites sampled within the Chester River drainage correspond to sites sampled by O'Dell et al. (1980; Table 1-11; Figure 1-9). Eight of the twelve sites sampled within the Patuxent River correspond to sites sampled by O'Dell and Mowrer (1984; Table 1-12; Figure 1-10).

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:

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

The 90% confidence intervals were constructed as:

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

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 independent 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}), and 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, Chester and Patuxent rivers. Thirty-seven sets of estimates of C/ha, percent agriculture, percent forest, and P_{herr} were available (1991 estimates for Mattawoman Creek could be included), while 36 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 for additional information and results). 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, and 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 $\mu\text{S}/\text{cm}$) when development was relatively low (Hall et al. 1992). Background conductivity used to standardize median conductivities was 109 $\mu\text{S}/\text{cm}$ in Coastal Plain streams and 150 $\mu\text{S}/\text{cm}$ in Piedmont streams (Morgan et al. 2012). For Bush, Patapsco, and Patuxent rivers, watersheds that run through both physiographic provinces, conductivities were standardized using the 150 $\mu\text{S}/\text{cm}$ 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 2005-2021 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 (2012-2021). 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).

Upstream (drift) and downstream (boat tow) P_{herr} estimates in urbanized Patuxent River (2021; this report), and rural, agricultural Choptank River and Tuckahoe Creek (2016-2017; Uphoff et al. 2017; 2018), were also compared to look at sensitivity to collection method. Location was modeled as an indicator variable in the multiple regression of C/ha against P_{herr} . These analyses were initially done in Excel and run again in SAS (Proc Reg) to confirm the estimates.

Results

Development level of Piscataway, Mattawoman, and Deer creeks, Bush, Chester, and Patuxent 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 this time (Figure 1-11). Surveys conducted by O'Dell et al. (1975, 1980) in the 1970s, and O'Dell and Mowrer (1984) in the 1980s, sampled largely rural watersheds (C/ha < 0.28) except for Piscataway Creek (C/ha = 0.47), Patapsco River (C/ha = 0.44), and Patuxent River (C/ha = 0.33). By 1991, C/ha in Mattawoman Creek was similar to that of Piscataway in 1970. By the mid-2000s, Bush and Patuxent Rivers 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 1-11).

Conductivities were usually elevated beyond background levels in all streams studied during 2008-2021 and standardized 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, Patapsco River and Piscataway Creek had consistently high conductivity, and Patuxent River had the highest conductivities seen to date. 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 stream 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 fluvial 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 the furthest downstream stations TUC101-TUC103 and the furthest upstream stations 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 that 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 during 1975-1977, 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).

Herring spawning was detected in the upper portion of Patuxent River at two of the six sites that matched O'Dell and Mowrer (1984) collections in 1980 (AFC10-4 and AFC3-165), and at one site (AFC3-163) where Herring spawning previously was not detected (Figure 1-10; Table 1-12). White Perch spawning was only detected in the upper portion of Patuxent River at half the number of sites as collections in 1980, and no Yellow Perch spawning was detected in this area in 2021, even though presence was noted at two sites by O'Dell and Mowrer (1984). Yellow Perch larvae were frequently encountered, however, during sampling of the downstream estuarine Patuxent River Striped Bass spawning area during 2015-2016 (Uphoff et al. 2017). Only two stations (AFC3-161 and site 1) in the middle portion of Patuxent River matched sampling locations of O'Dell and Mowrer (1984; Figure 1-10; Table 1-12). Herring spawning was detected in 2021 at these two locations, as well as the other four sites in the middle portion of Patuxent River. It should be noted, however, that 2021 ichthyoplankton collections were made at the same time DNR biologists were monitoring for, and making collections of, Hickory Shad adults in middle Patuxent River (MD DNR 2022). White Perch spawning was detected at five out of six middle Patuxent River sites in 2021, two of which matched presence detected in 1982 by O'Dell and Mowrer (1984). Yellow Perch spawning was not detected in any of the

middle Patuxent River sites in 2021, but was found at the two stations sampled by O'Dell and Mowrer (1984; Table 1-12).

Estimates of P_{herr} increased in Bush River, and Mattawoman and Piscataway creeks during 2005-2018 (Figure 1-12). Counterintuitively, increases coincided with increased development for watersheds sampled before and after 2011; 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. The estimate of P_{herr} in Patuxent River was high (Figure 1-12), however ichthyoplankton collections there were concurrent with monitoring and collections of spawning adult Hickory Shad (MD DNR 2022).

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 1-13). The relationship of C/ha with standardized median conductivity was linear, moderate, and positive ($r^2 = 0.36$, $P = 0.0001$, $N = 36$; Table 1-13; Figure 1-13). Estimates of P_{herr} were linearly, moderately, and negatively related to C/ha ($r^2 = 0.48$, $P < 0.0001$, $N = 37$; Figure 1-14). Negative linear and curvilinear (power function) regressions similarly described weak relationships of P_{herr} and standardized median conductivity ($r^2 = 0.15$, $P = 0.0181$; or approximate $r^2 = 0.15$, $P < 0.0001$, respectively), with linear regression explaining only slightly more variability ($N = 36$; Figure 1-14). Low estimates of P_{herr} (≤ 0.4) were much more frequent beyond the C/ha threshold (0.86 C/ha) or when standardized conductivity was 1.8-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 1991 estimate of P_{herr} was above 0.6 and was consistent with watersheds dominated by agriculture. Remaining estimates for Mattawoman Creek were represented by 53.9% forest cover with C/ha increasing from 0.87 in 2008 to 0.97 in 2018. Samples were not collected in Mattawoman Creek since 2018, 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 73% of variation in P_{herr} ($P < 0.0001$; Table 1-14). The intercept (mean = 0.51, SE = 0.08) and both coefficients (C/ha slope = -0.27, SE = 0.05; spawning stock coefficient = 0.32, SE = 0.06) were estimated with reasonable precision (CV < 30%). Predicted P_{herr} declined by 49% over the range of observed C/ha (0.07-1.52; Figure 1-16). Predicted P_{herr} increased by 62% between the two spawning stock categories (Table 1-14). 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 67% of variation in P_{herr} ($P < 0.0001$; Table 1-15). The intercept (mean = 0.64, SE = 0.13) and both coefficients (standardized conductivity slope = -0.29, SE = 0.07; spawning stock coefficient = 0.42, SE = 0.06) were estimated with reasonable precision (CV < 33%). Predicted P_{herr} declined by 49% over the range of observed standardized conductivity (1.14-2.42; Figure 1-16). Predicted P_{herr} increased by 66% between the two spawning stock categories (Table 1-15). 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 standardized median 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 ($r^2 = 0.14$, $P = 0.03$) but unlikely for C/ha ($r^2 = 0.04$, $P = 0.21$). 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. Improved survival to maturity in response to declines in undescribed non-fishing related sources of at-sea losses (predation and feeding) could have contributed to increased spawning stock or supplied an alternative hypothesis to harvest reductions for the increase.

Examination of the plots of P_{herr} estimates and their 90% confidence intervals seemed to indicate differences between upstream drift net and downstream boat tow collections (Figures 1-18 and 1-19). Differences in highly developed Patuxent River collections (2021) were much more evident, with no overlap of 90% confidence intervals, however middle Patuxent in-stream drift collections were more similar to middle Patuxent boat tows than to upstream drifts (Figure 1-19). Choptank River and Tuckahoe Creek (2016-2017; Uphoff et al. 2017; 2018) drifts and tows did not have overlapping CI's in both systems in 2016, but did in 2017 (Figure 1-18). P_{herr} estimates were higher for boat tow collections in all systems and years, but these differences were not as extreme in Choptank River and Tuckahoe Creek as they were in Patuxent River.

The C/ha and location category multiple regression was strong, explaining 65% of variation in P_{herr} ($P = 0.01$; Table 1-16). The intercept (mean = 0.68, SE = 0.07) and both coefficients (C/ha slope = -0.17, SE = 0.08; location coefficient = 0.30, SE = 0.09) were estimated with reasonable precision (CV < 20%).

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 ostensibly accounted for shifting spawner abundance between 2005-2011 and 2012-2021, 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 50% 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 even at low levels of development.

Data from Patuxent River suggested a localized impact of development on Herring spawning. Drift net stations, located above the tide line at Route 214 that were within or just below the developed Laurel-Bowie area had much lower P_{herr} (0.17; N = 36) than drift stations below Route 214 (P_{herr} = 0.58; N = 12), and collections in the nearby tidal, more rural portion of the watershed that were made by boat tows (P_{herr} = 0.90; N = 52). The Choptank River, and Tuckahoe Creek (both with low development, agricultural watersheds), sampled in 2016 and 2017, had a similar sampling design as Patuxent River (drift nets upstream and boat samples downstream; Uphoff et al. 2017; 2018). Estimated P_{herr} in Choptank River was 0.74 during 2016 (N = 46) and 2017 (N = 43) where upstream drift nets were employed, and 1.00 (N = 54) in 2016 and 0.88 (N = 66) in 2017 where samples were taken by boat. Estimated P_{herr} in Tuckahoe Creek was 0.67 (N = 57) during 2016 and 0.72 (N = 58) in 2017 where upstream drift nets were used for collections, and 0.92 (N = 40) in 2016 and 0.77 (N = 44) in 2017 where boat tows were used for collections. While upstream drift net collections in these streams always had lower P_{herr} estimates, the difference was more extreme in Patuxent River than in Choptank River and Tuckahoe Creek. This seems to speak to greater stress from higher urbanization upstream in the Patuxent. Since P_{herr} estimates of stream drift collections were more similar to boat tow collections in the area below Route 214 (Figure 1-10), it is possible there are also impediments that don't allow as many fish to get up to those reaches. Median conductivity of drift sites above Route 214 was much higher (414.2 $\mu\text{S}/\text{cm}$) than drift sites below (238.9 $\mu\text{S}/\text{cm}$), indicating higher anthropogenic impact and potential for greater harm and reduced survival.

Higher standardized conductivity (up to about 1.6-times higher) 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; Kaushal et al. 2018; Baker et al. 2019). 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. Different mixtures of salt ions (such as sodium, bicarbonate, magnesium, sulfate, etc.) produce differential toxicity to aquatic life (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. The MD Department of Planning 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).

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 (0.44; Table 1-14) than for standardized conductivity (0.32; Table 1-15), 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 spawning stream 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 spatial and temporal distribution of earliest life 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 did not indicate that increased spawning stock overcame 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. Increasing variability, however, was likely an artifact of increasing spawning stock size with time. Once a time category term, that we assumed 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} 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 did not experience a 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 2020), but Herring may have increased in the Head-of-Bay region (Bourdon 2022).

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 selected spawning habitat based on macrohabitat features (Harris and Hightower 2011) and spawned 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 do not clean substrate during spawning or provide protection to eggs and larvae in nests (Kemp 2014). Urbanization affects the quality and quantity of organic matter, another source of spawning substrate (detritus) in streams (Pardue 1983; 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; Kaushal et al. 2018; Baker et al. 2019). Use of salt as a deicer may lead to both “shock loads” of salt that may be acutely toxic to freshwater biota, as well as elevated baselines (increased average concentrations) of chloride that have been associated with decreased fish and benthic diversity (Kaushal et al. 2005; Wheeler et al. 2005; Morgan et al. 2007; 2012). Commonly used anti-clumping agents for road salt (ferro- and ferricyanide) that are not thought to be directly toxic are of concern because they can break down into toxic cyanide under exposure to ultraviolet light. Although the degree of breakdown into cyanide in nature is unclear (Pablo et al. 1996; Transportation Research Board 2007), these compounds have been implicated in fish kills (Burdick and Lipschuetz 1950; Pablo et al. 1996; Transportation Research Board 2007). Heavy metals and phosphorous may also be associated with road salt (Transportation Research Board 2007). Salt concentration, salt type, and, to a lesser extent, stormwater BMP (best management practice) type determines the composition and concentrations of chemical cocktails released from stormwater BMPs (Galella et al. 2023). Salt ion amounts had significant effects on mobilization of a wide variety of contaminants including nutrients, other salt ions, and metals. Though NaCl is often the most cost-effective deicer available, the use of brines, MgCl₂ and CaCl₂ should be considered as an alternative, especially in environmentally fragile areas or where fisheries are affected by Cu pollution (Galella et al. 2023).

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. Sodium chloride is the dominant form of salt pollution with freshwater salinization syndrome, but increases in different mixtures of salt ions such as bicarbonate, magnesium, sulfate, etc., are part of the syndrome (Kaushal et al. 2018).

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, however 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 assumed that spawning sites detected in the 1970s and 1980s indicated the extent of habitat. O'Dell et al. (1975; 1980) and O'Dell and Mowrer (1984) 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 and early 1980s 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 with Herring (P_{herr}) incorporated spatial and temporal presence-absence and provided an economical and precise alternative to the O'Dell et al. (1975; 1980) and O'Dell and Mowrer (1984) 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 2005-2018 used only stream drift nets, while O'Dell et al. (1975; 1980), O'Dell and Mowrer (1984), 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, Bush River, and Patuxent 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 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		
Bush (w/o APG)	2006	2002	1.41	25.4	35		
Bush (w/o APG)	2007	2010	1.43	18	29.9	35,956	Urban
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	0.14	65.9	24.8	77,357	Agriculture
Choptank	2016	2010	0.18	55	27.8	38,263	Agriculture
Choptank	2017	2010	0.18	55	27.8		
Deer	2012	2010	0.24	44.6	28.4		
Deer	2013	2010	0.24	44.6	28.4	37,612	Agriculture
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		
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	24,329	Forest
Mattawoman	2013	2010	0.92	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	0.96	9.3	53.9		
Mattawoman	2017	2010	0.97	9.3	53.9		
Mattawoman	2018	2010	0.97	9.3	53.9		

Table 1-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	Watershed Size (ha)	Primary Land Use
Patapsco	2013	2010	1.11	24.4	30.4		
Patapsco	2014	2010	1.12	24.4	30.4		
Patapsco	2015	2010	1.13	24.4	30.4	93,728	Urban
Patapsco	2016	2010	1.14	24.4	30.4		
Patapsco	2017	2010	1.15	24.4	30.4		
Patuxent	2021	2010	1.39	20.5	35.1	99,960	Urban
Piscataway	2008	2010	1.41	10	40.4		
Piscataway	2009	2010	1.43	10	40.4		
Piscataway	2012	2010	1.47	10	40.4	17,536	Urban
Piscataway	2013	2010	1.50	10	40.4		
Piscataway	2014	2010	1.51	10	40.4		
Tuckahoe	2016	2010	0.07	66.6	25.4	39,272	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
Patuxent	2021	12	18-Mar	9-Jun	18	100
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 ($\mu\text{S}/\text{cm}$) for mainstem stations in Deer, Mattawoman, Piscataway, and Tuckahoe creeks, and Bush, Chester, Choptank, Patapsco, and Patuxent rivers during 2005-2021. Unnamed tributaries were excluded from analysis. Tinkers Creek was included with mainstem stations in Piscataway Creek.

<u>Conductivity</u>	<u>Bush</u>					<u>Chester</u>	<u>Choptank</u>		<u>Deer</u>			
	<u>2005</u>	<u>2006</u>	<u>2007</u>	<u>2008</u>	<u>2014</u>	<u>2019</u>	<u>2016</u>	<u>2017</u>	<u>2012</u>	<u>2013</u>	<u>2014</u>	<u>2015</u>
Mean	269.5	206.3	262.5	236.5	276.7	175.8	130.7	129.7	174.9	175.6	170.3	191.8
Standard Error	25.4	5.0	16.0	6.1	15.0	4.0	1.4	1.0	1.0	1.5	1.4	0.9
Median	229.5	208.1	218.7	233.9	253.4	181.5	133.2	129.8	176.8	177.7	171.7	193.5
Kurtosis	38.2	2.3	22.5	6.5	3.2	-0.40	2.41	-0.05	17.22	13.88	9.21	7.43
Skewness	5.8	-0.7	3.8	0.1	1.6	-0.37	-1.07	-0.07	-3.78	-2.25	-2.42	-1.97
Range	1861	321	1083	425	606	164	89	49	39.3	122	66	51
Minimum	79	0	105	10	107	85	74	107	140	93	116	156
Maximum	1940	321	1187	435	713	249	163	156	180	215	183	207
Count	81	106	79	77	60	93	101	109	44	87	60	75

<u>Conductivity</u>	<u>Mattawoman</u>										
	<u>2008</u>	<u>2009</u>	<u>2010</u>	<u>2011</u>	<u>2012</u>	<u>2013</u>	<u>2014</u>	<u>2015</u>	<u>2016</u>	<u>2017</u>	<u>2018</u>
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

Table 1-3 cont.

<u>Conductivity</u>	Patapsco					Patuxent	Piscataway					Tuckahoe	
	<u>2013</u>	<u>2014</u>	<u>2015</u>	<u>2016</u>	<u>2017</u>	<u>2021</u>	<u>2008</u>	<u>2009</u>	<u>2012</u>	<u>2013</u>	<u>2014</u>	<u>2016</u>	<u>2017</u>
Mean	406.2	282.5	346.8	310.4	340.3	354.5	218.4	305.4	211.4	245	249.4	152.2	155.9
Standard Error	48.7	8.0	18.2	30.6	15.1	8.6	7.4	19.4	5.9	6.9	11.1	2.4	1.7
Median	304.9	279.5	324	262.7	310	360.5	210.4	260.6	195.1	238.4	230	159.6	160.5
Kurtosis	12.13	-0.24	5.04	17.97	2.22	1.08	-0.38	1.85	0.11	-0.29	2.56	-0.29	-0.18
Skewness	3.33	0.42	1.97	3.99	1.36	-0.99	0.75	1.32	0.92	0.73	1.50	-0.68	-0.61
Range	1554	166	487	1055	432	391	138	641	163	173	274	103	82
Minimum	245	219	216	188	175	103	163	97	145	181	174	85	103
Maximum	1799	385	703	1243	607	494	301	737	308	354	449	188	185
Count	40	28	32	40	40	96	29	50	44	44	36	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.

Station	Year														
	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.

Station	Year					
	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.

Station	Year					
	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.

Station	Year				
	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.

Station	Year					
	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.

Station	Year		
	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)		Year				
Station	Herring	2013	2014	2015	2016	2017
Inland 1	0	Herring				
Inland 2	1	USFWS Down River	1	1	1	1
Inland 3	1	USFWS Upriver	1	1	1	1
Inland 4	1	MBSS 591	1	1	1	1
Inland 5	0	MBSS 593	1	1	0	1
White Perch		White Perch				
Inland 1	1	USFWS Down River	0	1	1	1
Inland 2	1	USFWS Upriver	1	1	1	1
Inland 3	0	MBSS 591	0	1	1	1
Inland 4	1	MBSS 593	0	0	0	0
Inland 5	0	Yellow Perch				
Yellow Perch		USFWS Down River	1	1	1	1
Inland 1	1	USFWS Upriver	1	0	1	0
Inland 2	0	MBSS 591	0	0	1	0
Inland 3	0	MBSS 593	0	0	1	0
Inland 4	0					
Inland 5	1					

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.

Station	Year	
	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

Station	Year	
	1975-77	2019
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

Station	Year	
	1975-77	2019
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. Site-specific presence-absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Patuxent River during 1980-1982 and 2021. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-10.

Station	Patuxent	
	1980-1982	2021
	Herring	
AFC3-163	0	1
AFC3-164	0	0
AFC3-165	1	1
AFC3-114	1	0
AFC10-4	1	1
AFC10-8	1	0
AFC3-188		1
AFC3-161	1	1
1	1	1
2		1
3		1
4		1
	White Perch	
AFC3-163	1	1
AFC3-164	1	1
AFC3-165	1	1
AFC3-114	1	0
AFC10-4	1	0
AFC10-8	1	0
AFC3-188		0
AFC3-161	1	1
1	1	1
2		1
3		1
4		1
	Yellow Perch	
AFC3-163	1	0
AFC3-164	1	0
AFC3-165	0	0
AFC3-114	0	0
AFC10-4	0	0
AFC10-8	0	0
AFC3-188		0
AFC3-161	1	0
1	1	0
2		0
3		0
4		0

Table 1-13. 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	1.71041	1.71041	19.04	0.0001
Residual	34	3.05422	0.08983		
Total	35	4.76463			

$r^2 = 0.3590$

	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.19225	0.10849	10.99	<.0001	0.97177	1.41272
C / ha	0.45443	0.10414	4.36	0.0001	0.24279	0.66607

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Structure density (C/ha)				
ANOVA	df	SS	MS	F	P
Regression	1	1.22258	1.22258	32.88	<.0001
Residual	35	1.30139	0.03718		
Total	36	2.52397			

$r^2 = 0.4844$

	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.83652	0.06826	12.25	<.0001	0.69794	0.9751
C / ha	-0.37981	0.06624	-5.73	<.0001	-0.51428	-0.24534

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Standardized conductivity				
ANOVA	df	SS	MS	F	P
Regression	1	0.38127	0.38127	6.17	0.0181
Residual	34	2.10153	0.06181		
Total	35	2.48281			

$r^2 = 0.1536$

	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94044	0.18827	5.00	<.0001	0.55783	1.32306
Standardized conductivity	-0.28288	0.1139	-2.48	0.0181	-0.51435	-0.05141

Table 1-14. 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	1.80526	0.90263	43.96	<.0001	
Residual	33	0.67755	0.02053			
Total	35	2.48281				
$r^2 = 0.7271$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.50716	0.07879	6.44	<.0001	.	.
C / ha	-0.27086	0.05345	-5.07	<.0001	0.47638	0.43764
Time category	0.31516	0.05724	5.51	<.0001	0.47883	0.47883

Table 1-15. 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	1.66788	0.83394	33.77	<.0001	
Residual	33	0.81493	0.02469			
Total	35	2.48281				
$r^2 = 0.6718$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.64075	0.12604	5.08	<.0001	.	.
Standardized conductivity	-0.28607	0.07199	-3.97	0.0004	0.15357	0.32362
Time category	0.42208	0.05848	7.22	<.0001	0.61222	0.61222

Table 1-16. Summary statistics of the multiple regression of proportion of samples with Herring eggs and-or larvae (P_{herr}) versus location category (upstream = 0 and downstream = 1) and development level (C/ha).

Linear Model		Multiple Regression				
ANOVA	df	SS	MS	F	P	
Regression	2	0.3139	0.15695	7.46	0.0149	
Residual	8	0.16839	0.02105			
Total	10	0.48229				
$r^2 = 0.6509$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.67527	0.07155	9.44	<.0001	0.51027	0.84027
C / ha	-0.16986	0.07618	-2.23	0.0563	-0.34554	0.00581
Location	0.30259	0.0886	3.42	0.0092	0.09829	0.5069

Figure 1-1. Watersheds sampled for stream spawning anadromous fish eggs and larvae during 2005-2021. Coastal Plain and Piedmont Regions are indicated.

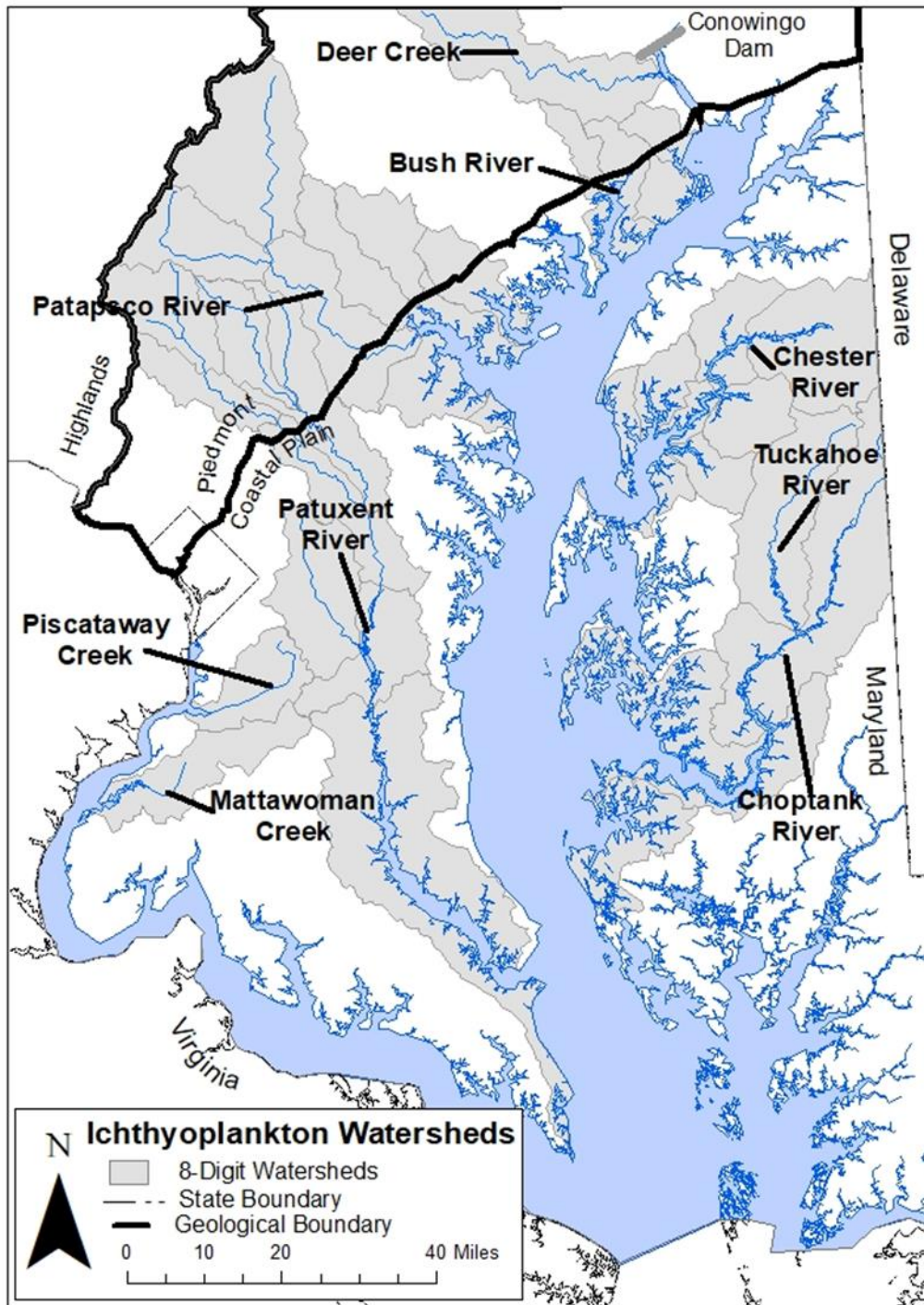


Figure 1-2. Mattawoman Creek's 1971 (O'Dell et al. 1975) and 2008-2018 sampling stations. Bar approximates lower limit of development associated with the town of Waldorf.

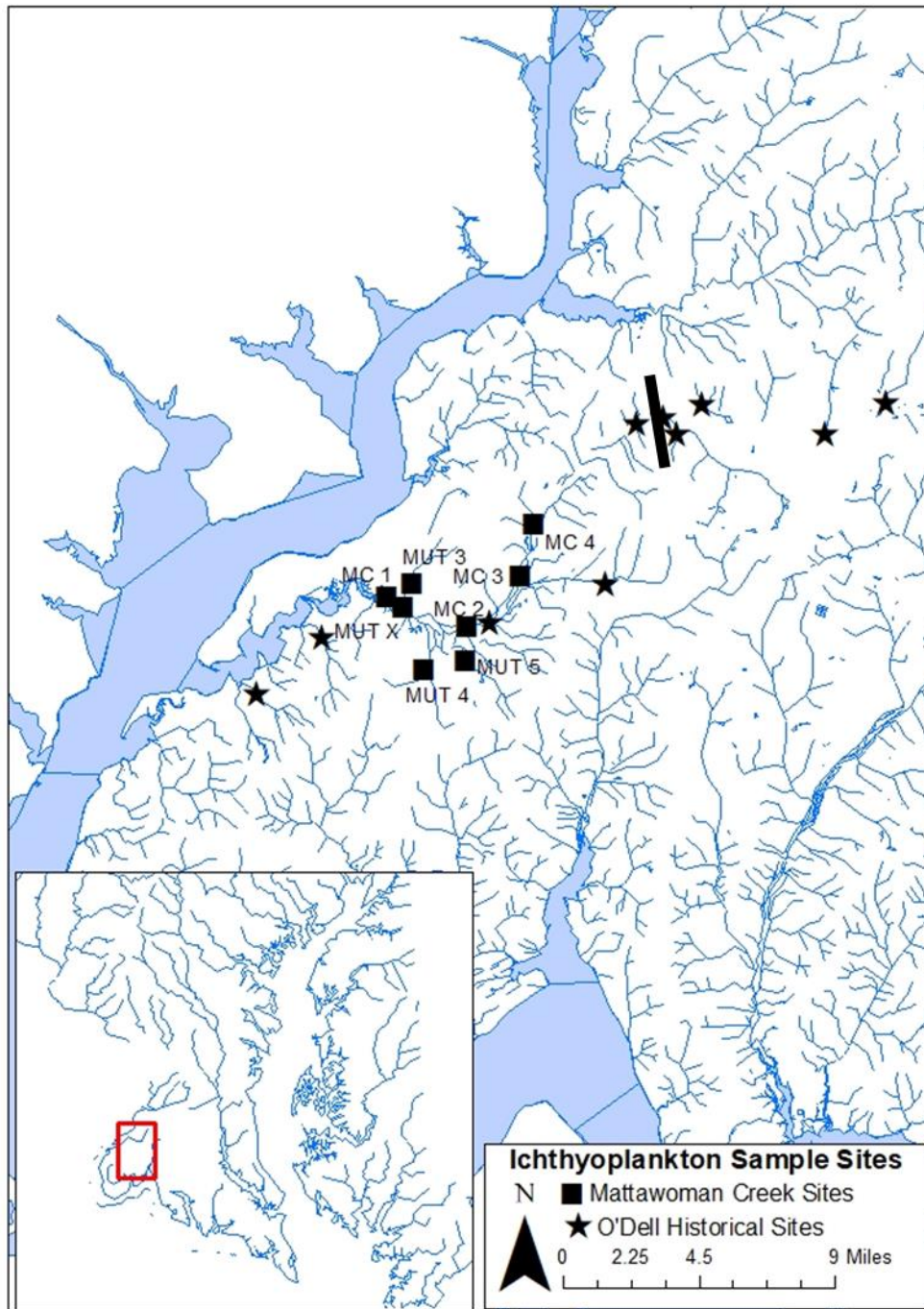


Figure 1-3. Piscataway Creek's 1971 (O'Dell et al. 1975), 2008-2009, and 2012-2014 sampling stations.

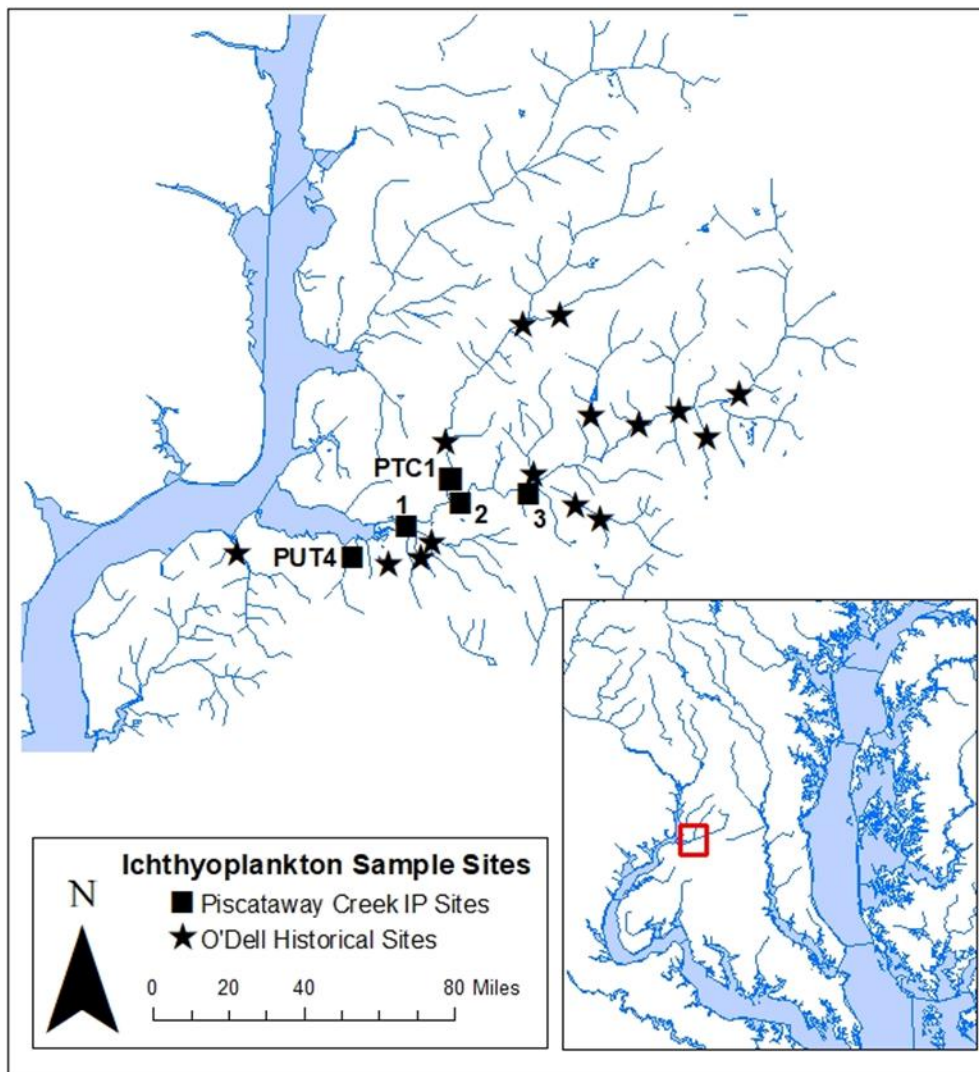


Figure 1-4. Bush River's 1973 (O'Dell et al. 1975), 2005-2008, and 2014 sampling stations. Stations in Aberdeen Proving Grounds (APG) have been separated from other Bush River stations. Line delineates APG streams that were excluded.

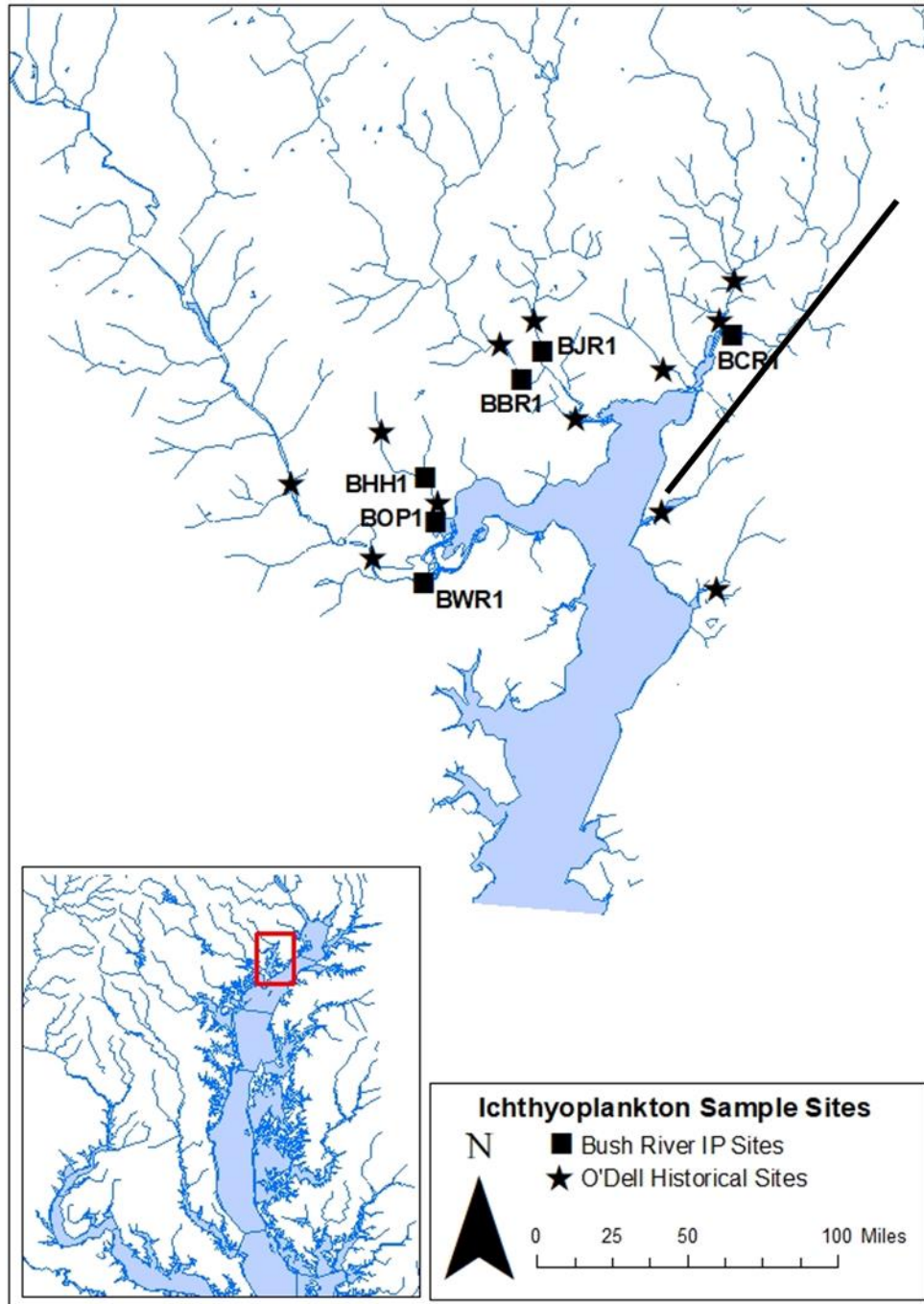
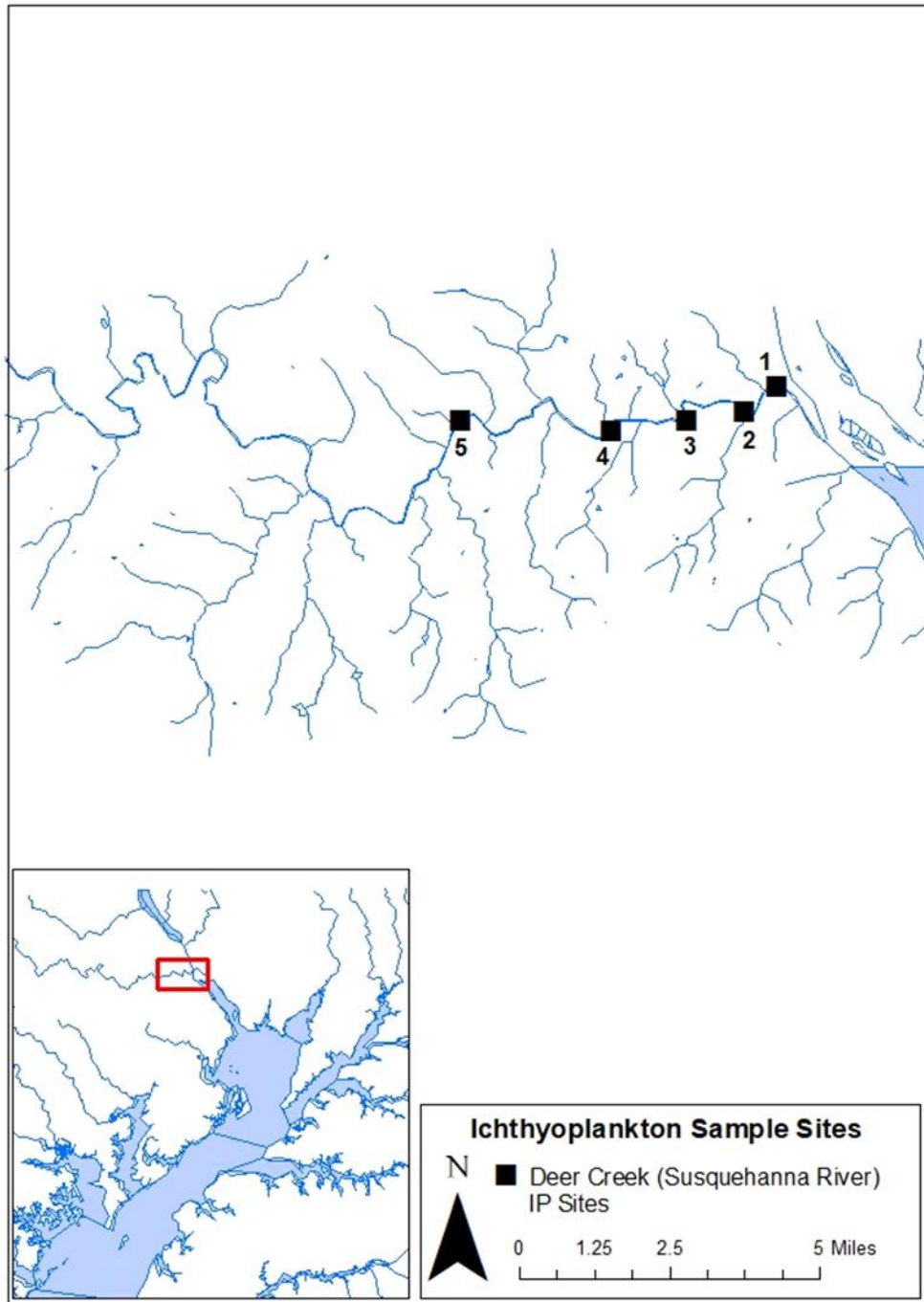


Figure 1-5. Deer Creek's 1972 (O'Dell et al. 1975) and 2012-2015 sampling stations.



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 collections and T = boat tow collections. Stations TUC101, TUC102, TUC103, TUC108, TUC109, and TUC110 correspond to O'Dell et al. (1980) sites

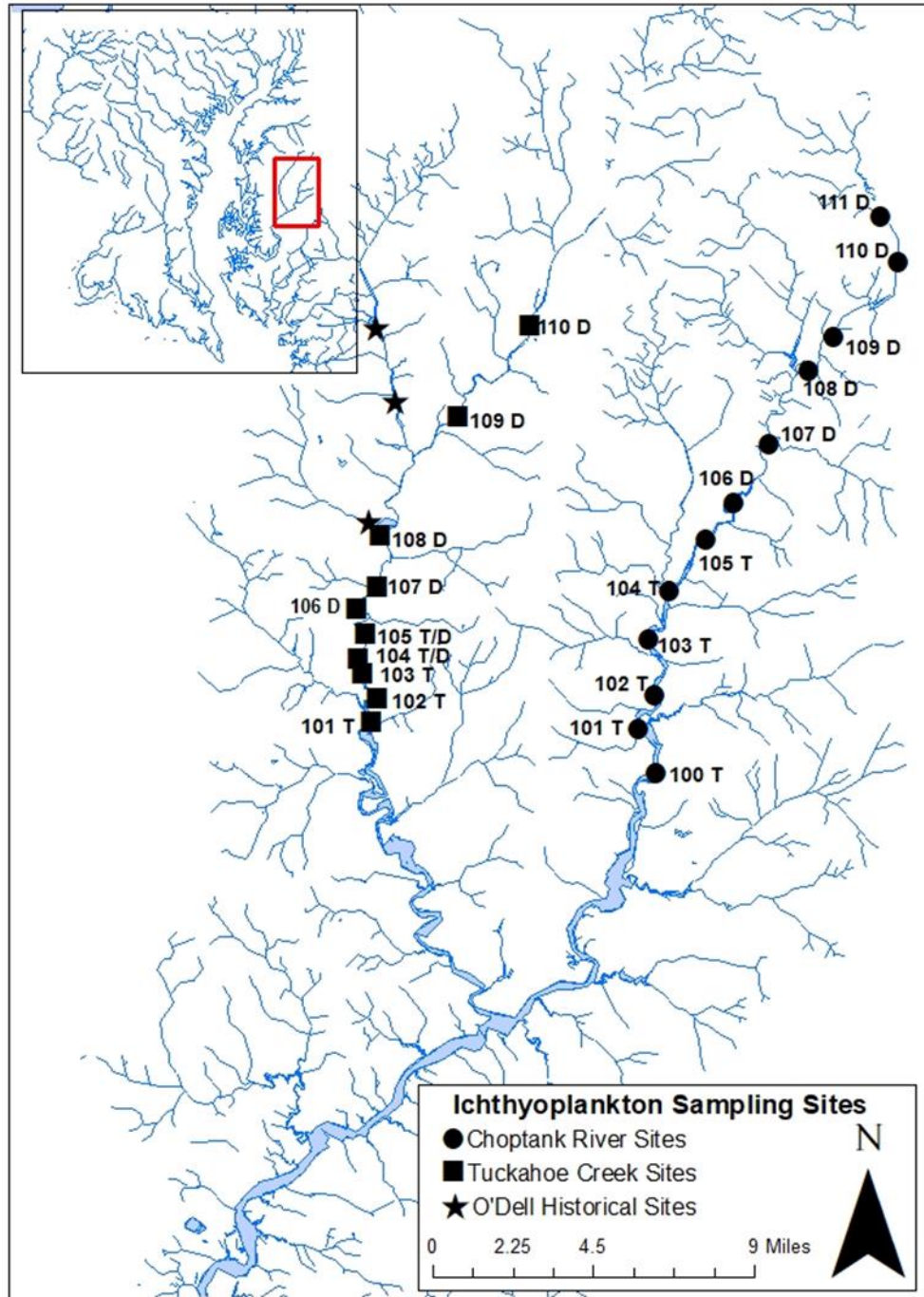


Figure 1-8. Patapsco River's 1973 (O'Dell et al. 1975) and 2013-2017 sampling stations.

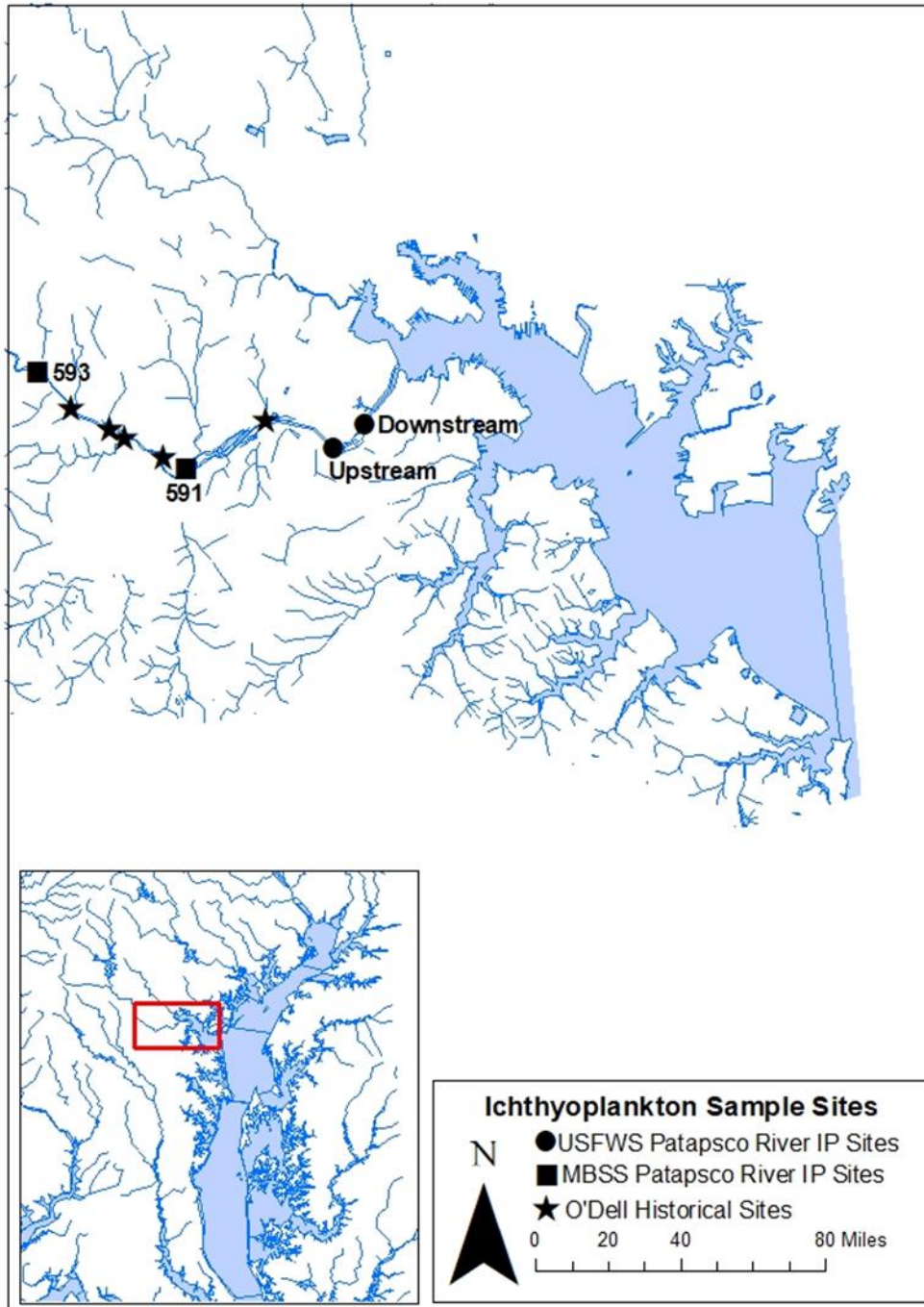


Figure 1-9. Chester River's 1975-1977 (O'Dell et al. 1980) and 2019 sampling stations.

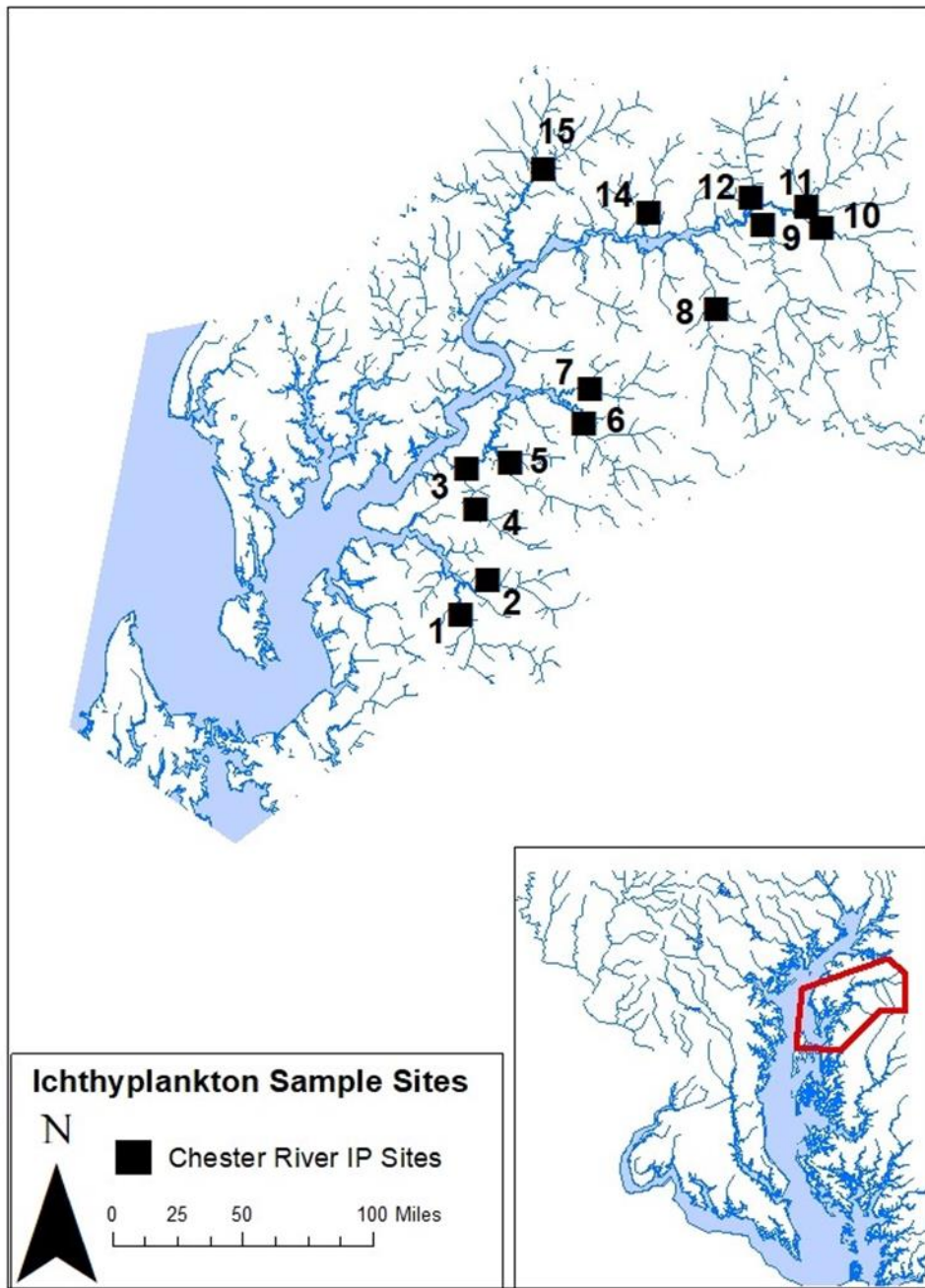


Figure 1-10. Patuxent River's 1980-1982 (O'Dell and Mowrer 1984) and 2021 sampling stations. Upstream sample sites correspond to drift net collections, while mainstem sample sites are boat tow collections. Approximate location of Route 214 provided for reference.

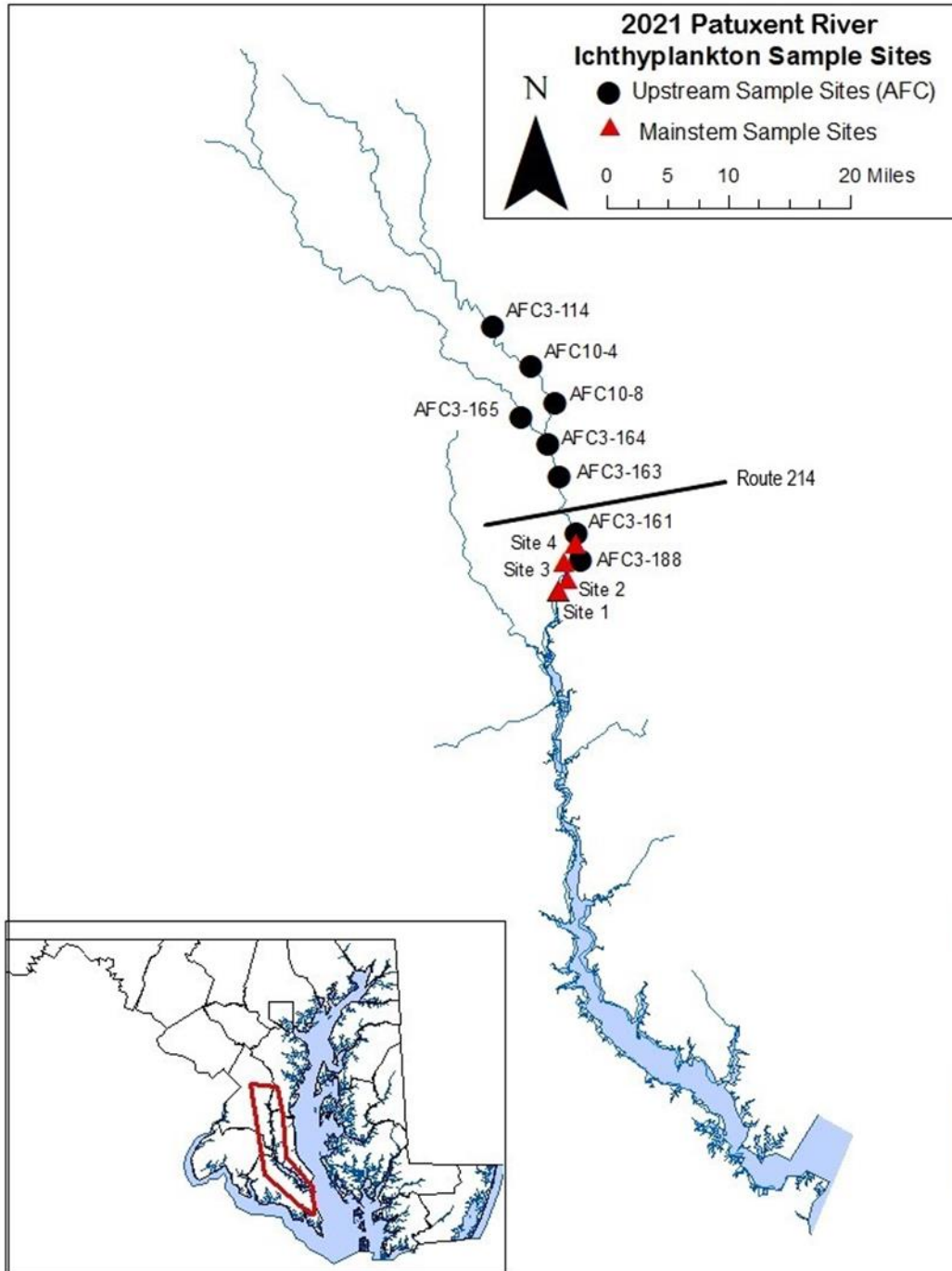


Figure 1-11. Trends in counts of structures per hectare (C/ha) during 1950-2021 in Deer, Mattawoman, and Piscataway creeks; Bush and Patapsco rivers; and Chester, Choptank, and Patuxent river drainages. Estimates of C/ha were only available to 2020. Large symbols indicate years when stream ichthyoplankton was sampled.

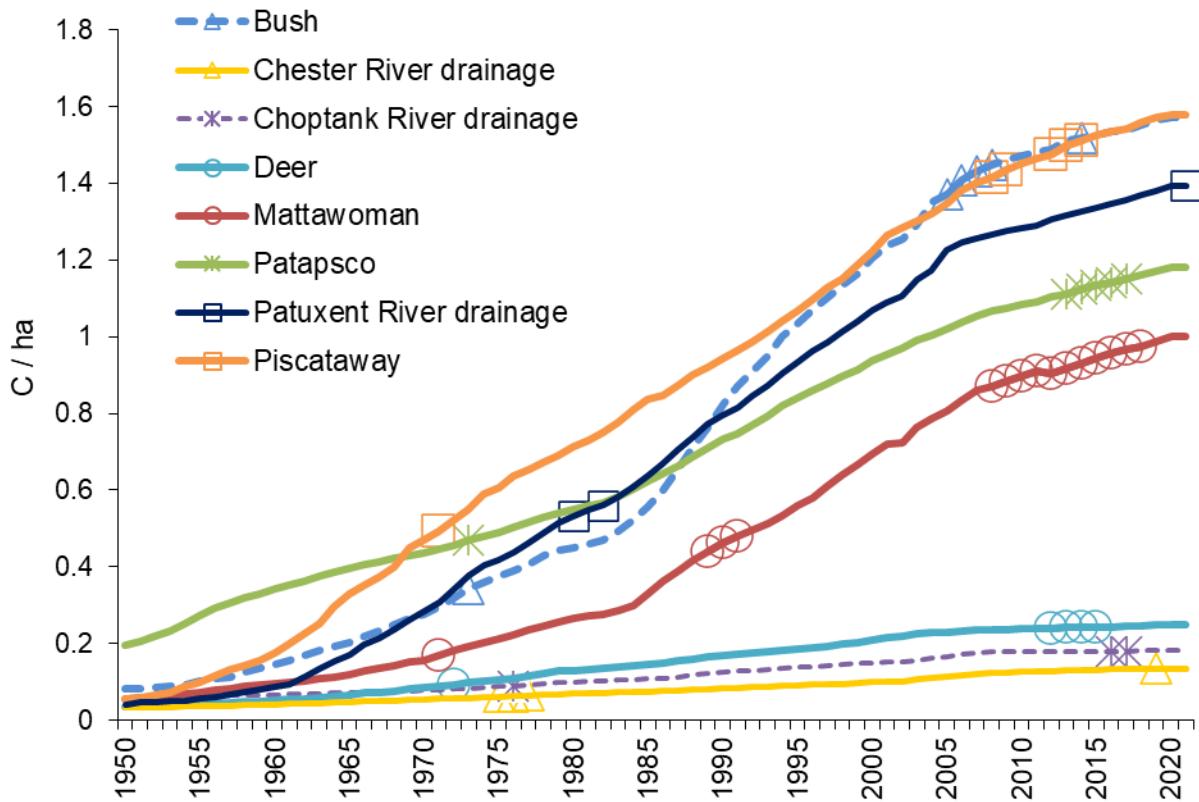


Figure 1-12. Trends in P_{herr} (proportion of stream samples with Herring eggs and-or larvae) by watershed. Watersheds sampled in both early (2005-2011) and late (2012-2021) spawning periods are indicated by large triangles.

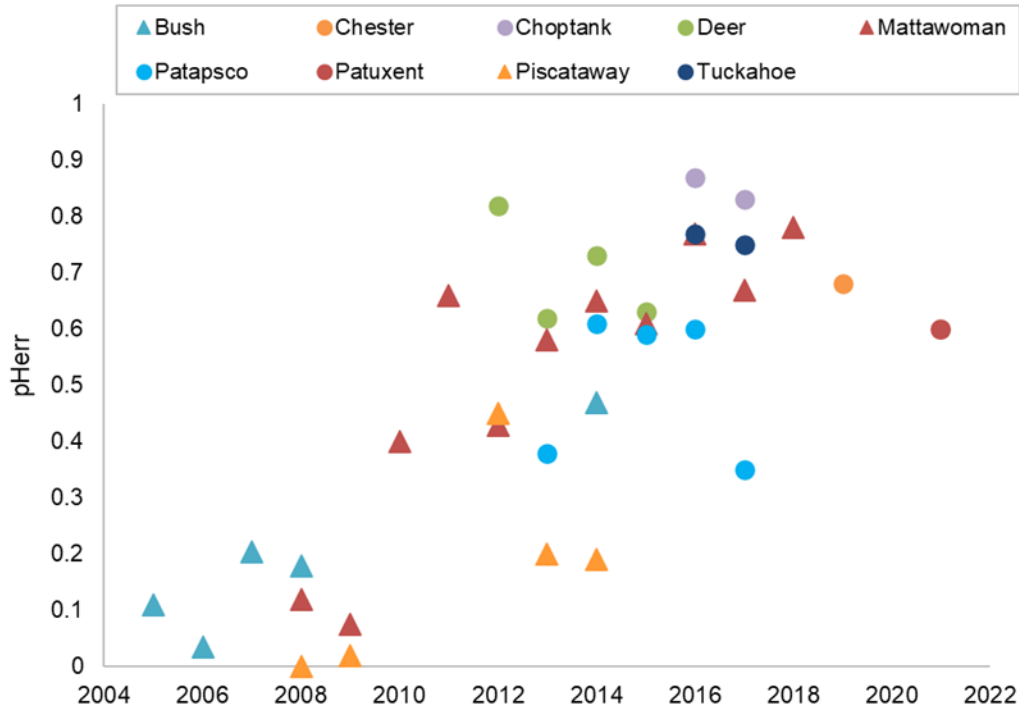


Figure 1-13. Standardized median conductivity during spring spawning surveys and level of development (C/ha) with dominant Department of Planning land use designations. Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et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

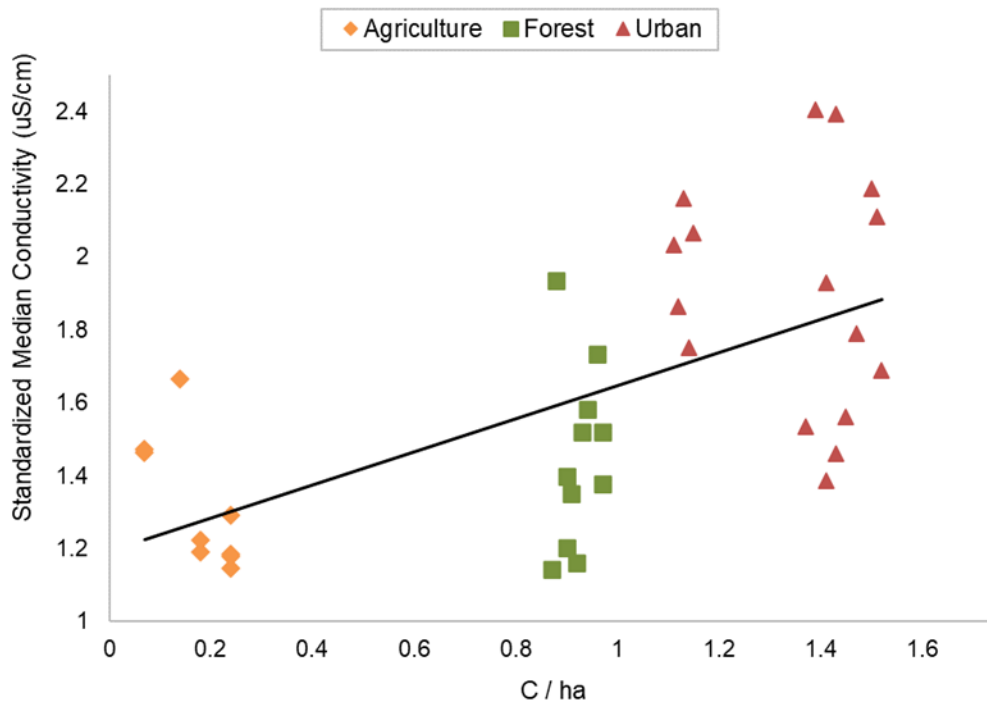


Figure 1-14. (A) Proportion of stream samples with Herring eggs and-or larvae (P_{herr}) and level of development (C/ha) with dominant Department of Planning land use designations. (B) P_{herr} and standardized median spawning survey conductivity (uS/cm). Median conductivity was standardized to background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

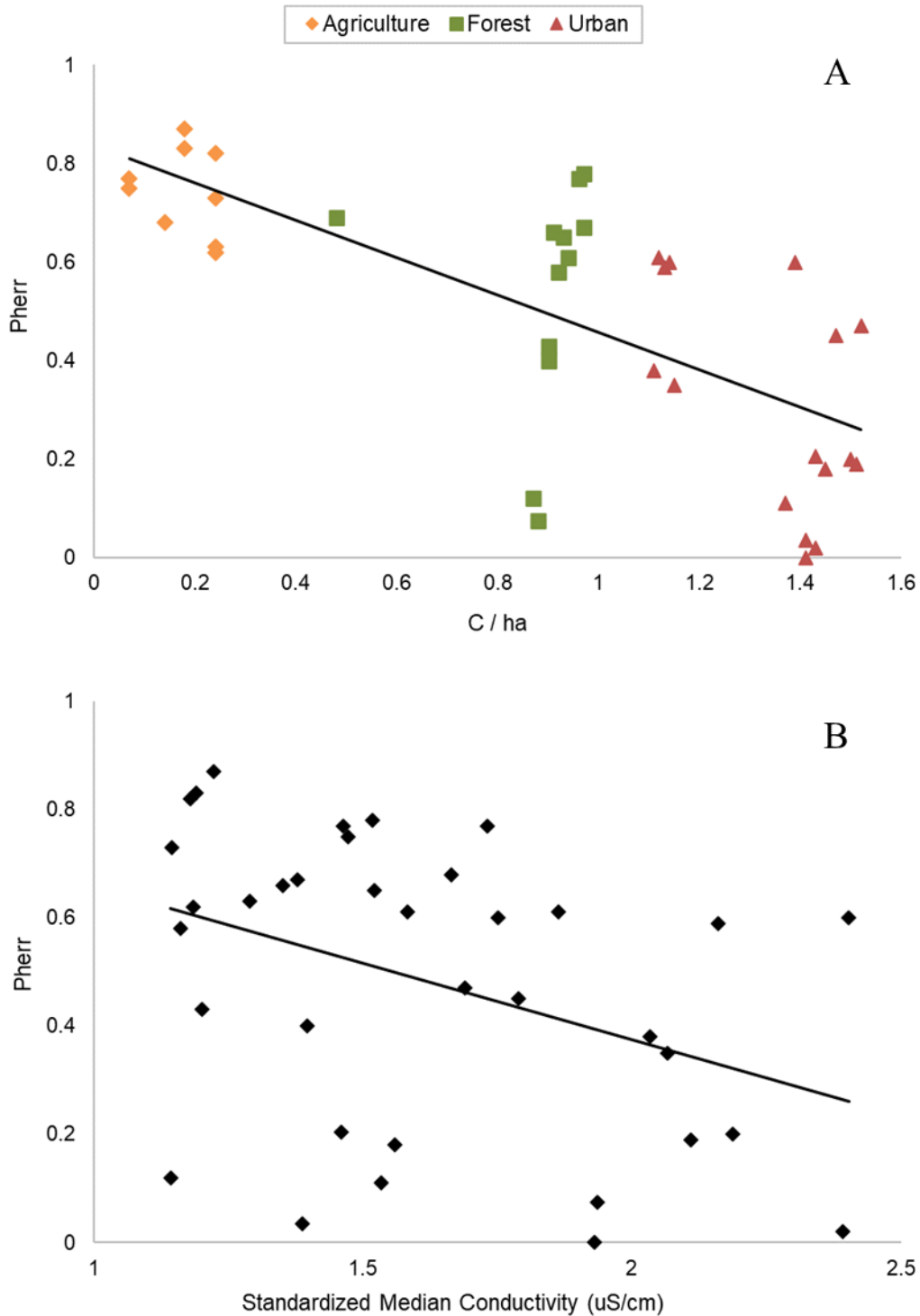


Figure 1-15. Serial patterns of residuals versus year for regressions of P_{herr} (proportion of stream samples with Herring eggs and-or larvae) and (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm). Median conductivity was standardized to background estimates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

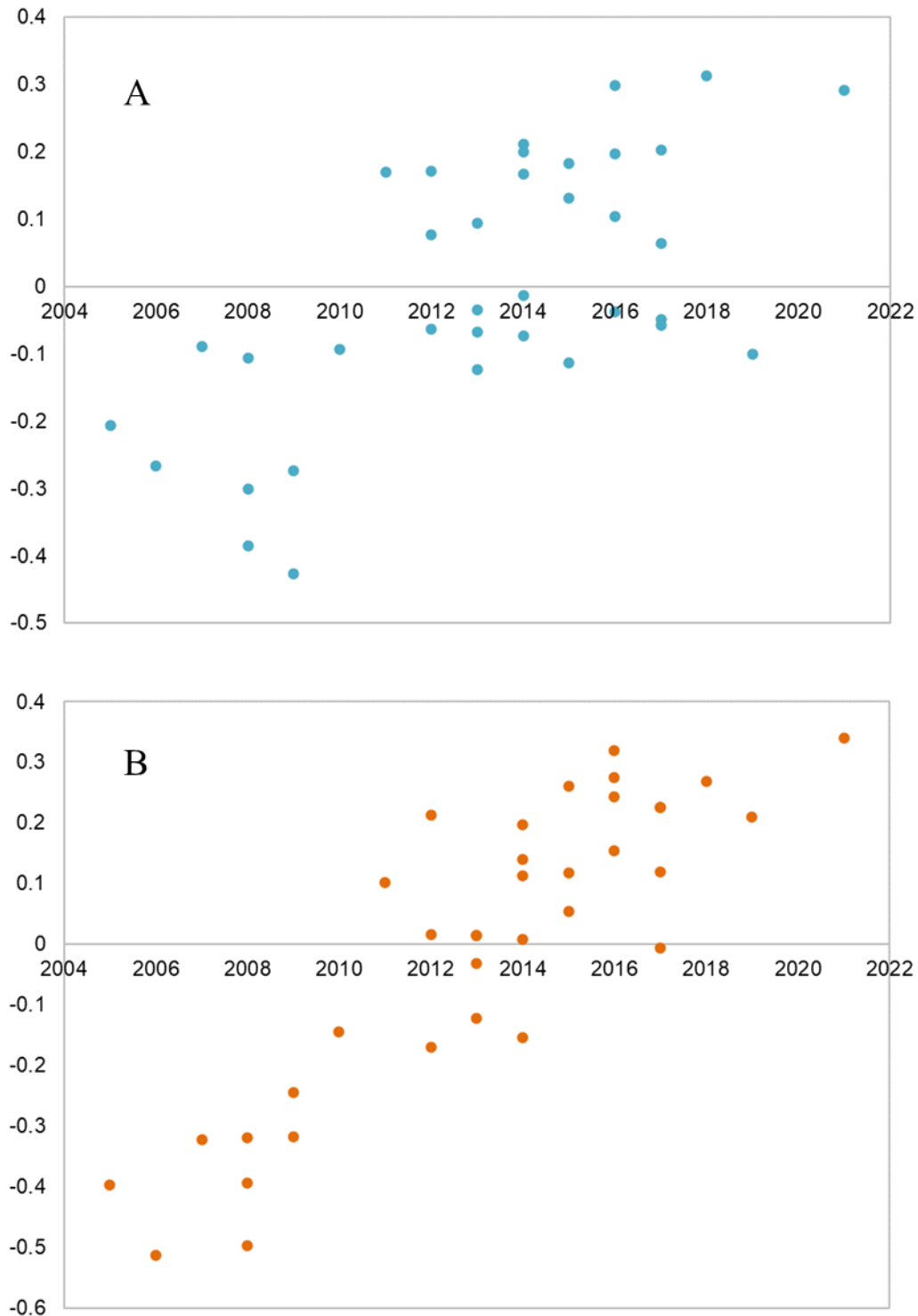


Figure 1-16. Plots of regressions of P_{herr} (proportion of stream samples with Herring eggs and-or larvae) against (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm) with spawning stock time categories (0 = 2005-2011; 1 = 2012-2021) included. Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

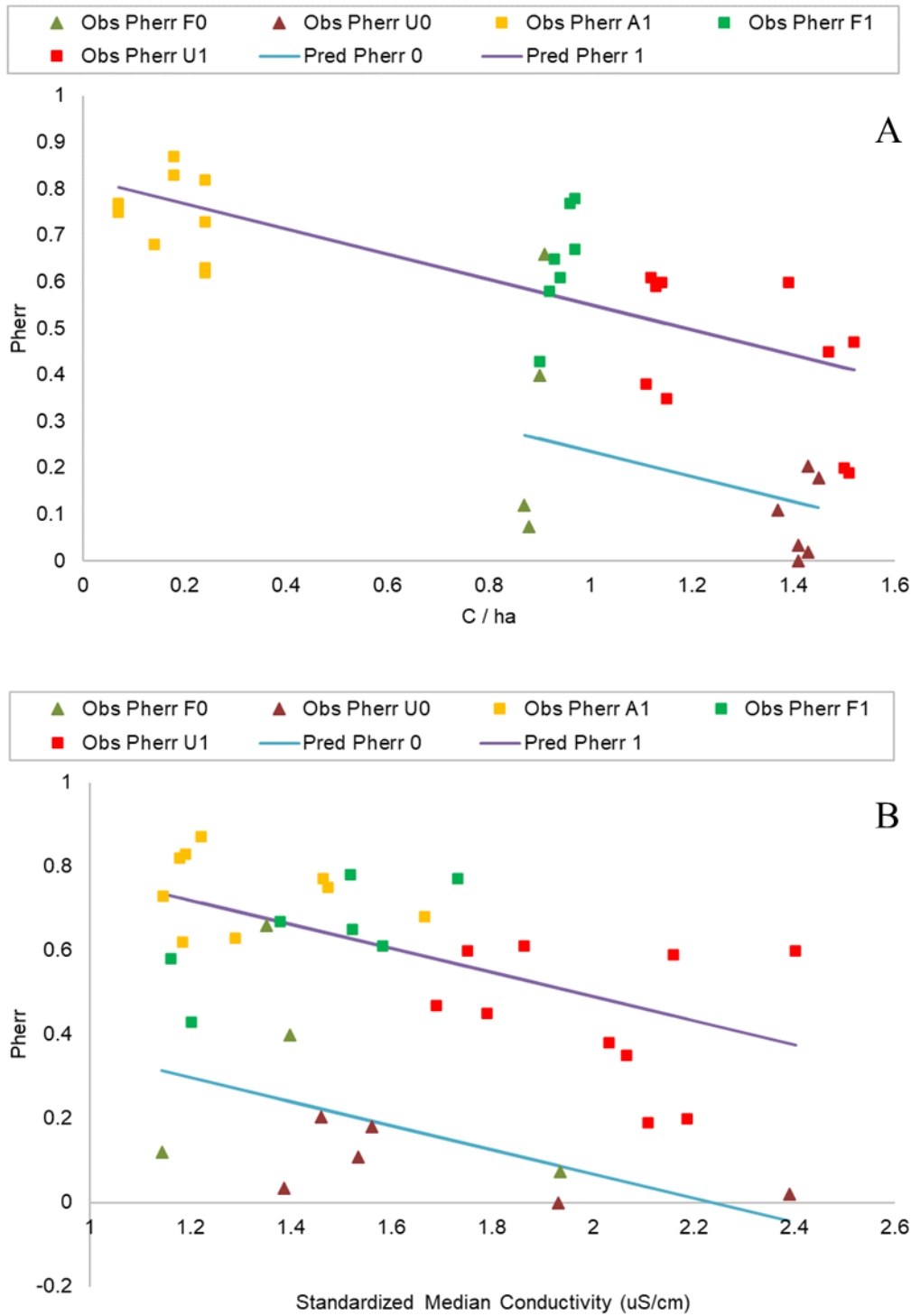


Figure 1-17. Residuals versus year for multiple regressions of spawning stock size time category and (A) level of development (C/ha) or (B) standardized median spawning survey conductivity (uS/cm) against proportion of stream samples with Herring eggs and-or larvae (P_{herr}). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

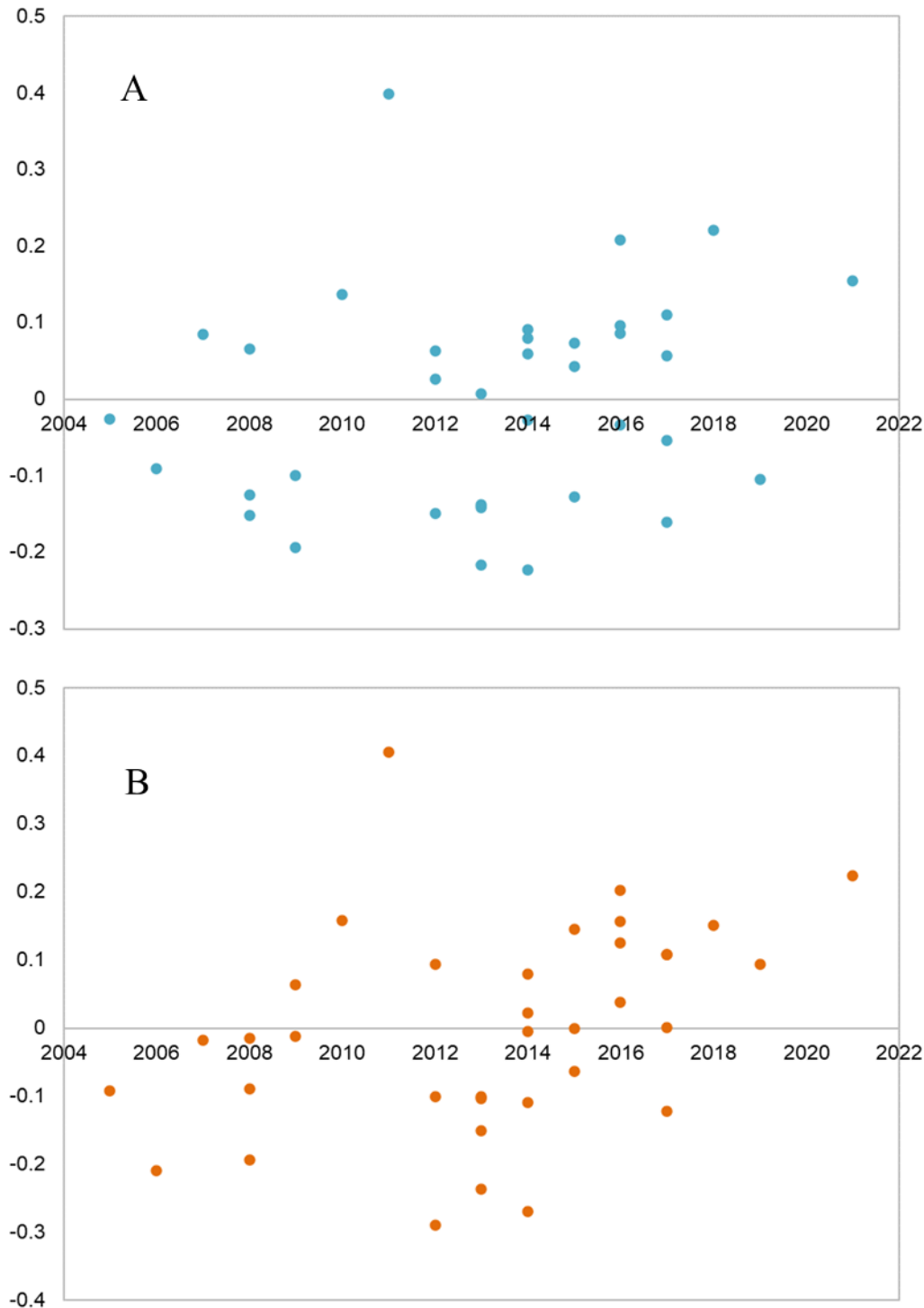


Figure 1-18. Plots of proportion of samples with Herring (P_{herr}) and their 90% confidence intervals for all collections combined, and upstream drift net, and downstream boat tow collections separated for 2016 and 2017 in Choptank River and Tuckahoe Creek.

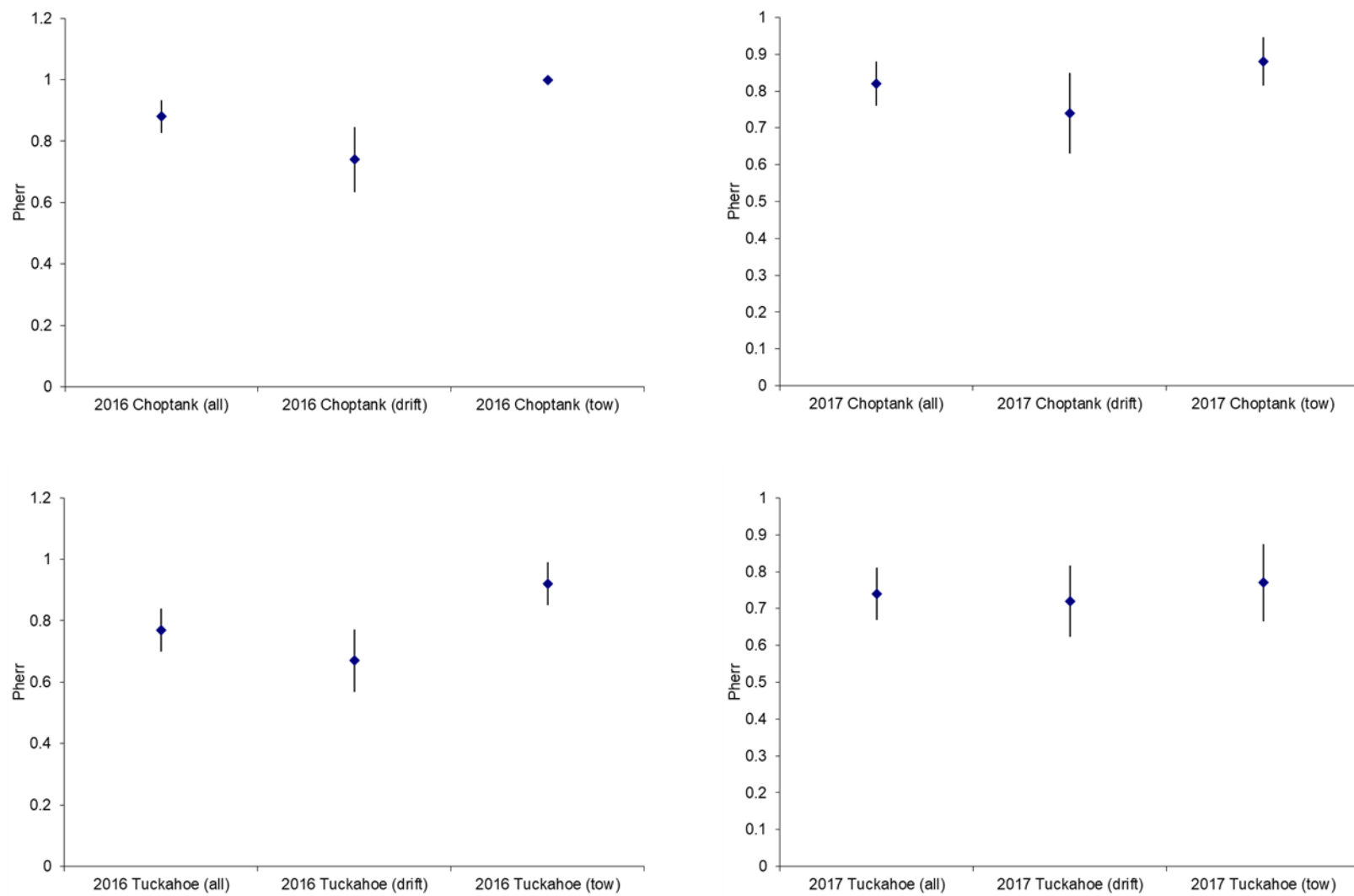
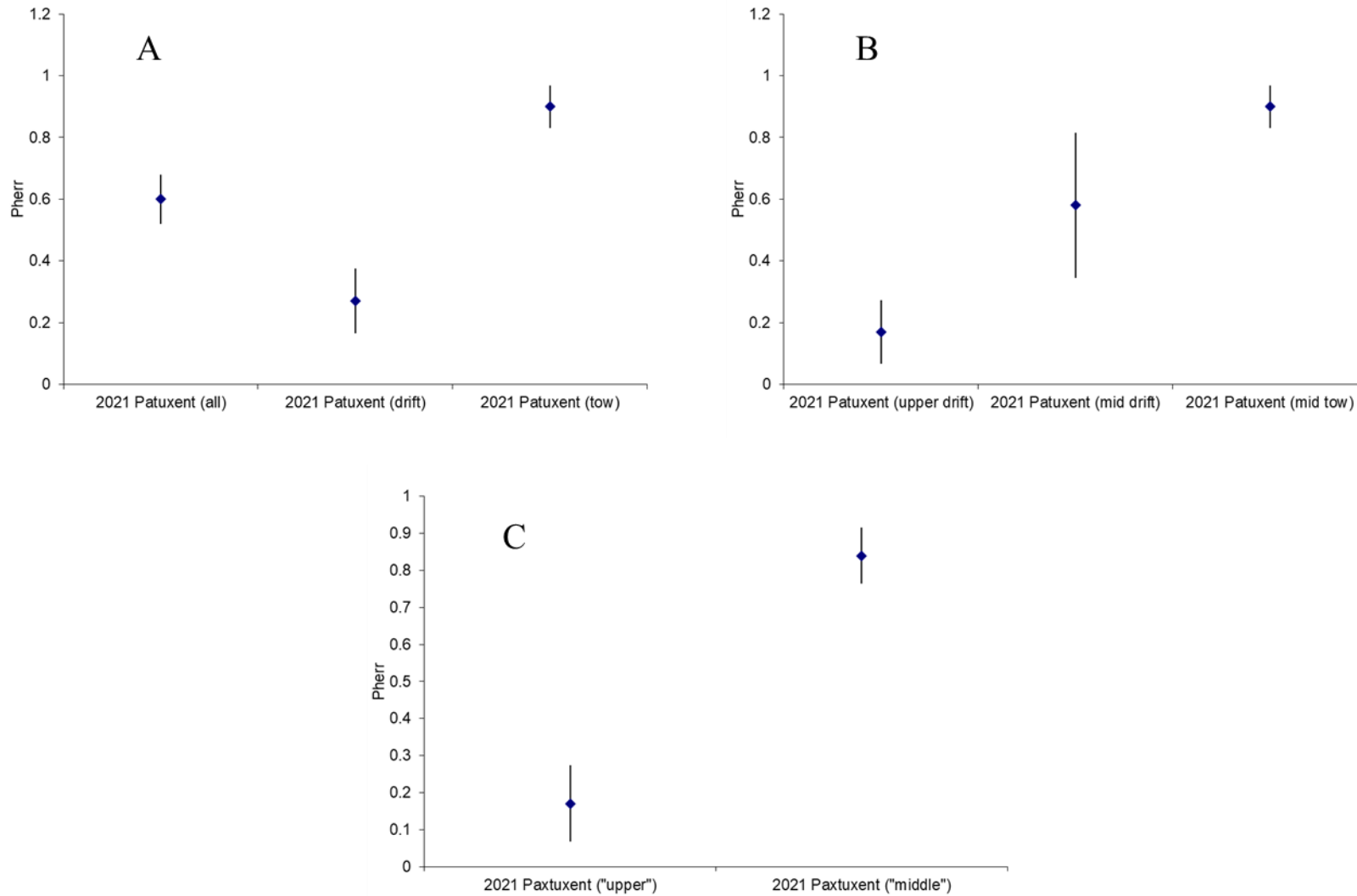


Figure 1-19. Plots of proportion of samples with Herring (P_{herr}) and their 90% confidence intervals for Patuxent River in 2021. (A) all collections combined, and upstream drift net, and downstream boat tow collections separated. (B) upper six drift collections only, middle two drift collections only, and middle four boat tow collections. (C) six “upper” drift collections only, and two “middle” drift collections and four middle boat tows combined.



MD – Marine and estuarine finfish ecological and habitat investigations
Project 1: Development of habitat-based reference points for recreationally important
Chesapeake Bay fishes of special concern
Section 2: Estuarine Yellow Perch Larval Presence-Absence Sampling

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

Introduction

Annual L_p , or the proportion of tows containing 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 was conducted in the upper tidal reaches of the Choptank and Sassafras Rivers in 2022 (Figure 2-1). Sampling started the third week of March and the last week of March, respectively, and continued through the second week of May in the Choptank River, and the third week of April in the Sassafras River.

Choptank and Sassafras Rivers are located in the Coastal Plain on the eastern side of Chesapeake Bay (Figure 2-1). The Sassafras River is a small tributary of Chesapeake Bay with a 19,501 Ha watershed. Agriculture is the primary land use (64% of the watershed) and development is low ($C/ha = 0.11$; Table 2-1). The subestuary is classified as oligohaline (salinity $< 0.5\text{‰}$) based on two Chesapeake Bay Program (CBP) monitoring stations; one within Sassafras River (ET 3.1) and another just below the mouth in the mainstem Bay (CB 2.2; MD DNR 2022a). Sassafras River represented the only low development, low salinity watershed with agriculture as its dominant land use with Yellow Perch spawning in our dataset (Table 2-1).

Choptank River is a large tributary of Chesapeake Bay with a watershed of 109,478 Ha. Land use is similar to Sassafras River (Table 2-1). Salinity classification runs from mesohaline at the mouth (CBP site EE2.1), to oligohaline at Ganey's Wharf, in the Yellow Perch larval nursery and Striped Bass spawning area (CBP site ET5.1; MD DNR 2022a). Nursery conditions for Yellow Perch larvae and Striped Bass eggs and larvae (see Section 2.1) could be surveyed concurrently in Choptank River and that influenced it being chosen for monitoring. An overfishing declaration and successive years of poor recruitment of Striped Bass have generated concern in the fisheries management and angling community. There has been unease expressed about degradation of Striped Bass spawning and larval nursery habitat in Chesapeake Bay. We assembled historical data (Uphoff et al. 2020; 2022a) and reoriented some of our spring monitoring in 2020 to respond to Striped Bass habitat concerns while maintaining Yellow Perch larval monitoring. See section 2.1 of this report for further details on the 2022 investigation of Striped Bass egg and larval habitat.

In 2022 we used regression analyses to examine relationships among land use types (development, agriculture, forest, and wetlands), L_p , and watershed size. We also examined a hypothesis that L_p has been declining in the Choptank River over the available time-series (1980-1990, 1998-2004, and 2013-2022).

Methods

Choptank and Sassafras Rivers were sampled by program personnel in 2022. 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). 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 that they could be confused with (Lippson and Moran 1974). Contents of the jar were allowed to settle and then the amount of settled organic material (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 (see Uphoff et al. 2022b for more information). If a pan contained enough OM to obscure seeing larvae, it was 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. Temperature, dissolved oxygen (DO), conductivity, pH, and salinity were measured at each site on each sample date. In 2022 some alkalinity samples were also collected in Choptank River (see Section 2.1 of this report for additional information and results).

Ten sites were sampled twice weekly in all systems (Figure 2-1) unless weather or salinity did not allow. 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. Historical estimates of L_p were initially developed from 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 these Striped Bass spawning areas.

In general, sampling to determine L_p began during the last week of March or first week 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. This sampling schedule has been maintained for tributaries sampled by program personnel since 2006. Sampling for L_p conducted 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 with a 1:3 mouth to length ratio 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, 1:8 mouth to length ratio) mounted in the cod-end were used in the Choptank River during 1980-1990 (Uphoff 1997; Uphoff et al. 2005; Uphoff et al. 2022a). 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 recalculated in 2022 based on an 18°C temperature maximum sampling cutoff (Table 2-1; Uphoff et al. 2022b). Uphoff et al. (2022b) estimated cumulative frequency of presence by temperature increment. They determined that the cumulative catch distribution showed the

greatest increase between 12°C and 18°C (full time series cumulative proportion equaled 0.93) and 18°C was adopted as a sampling and analysis cutoff (Uphoff et al. 2022b). Estimated L_p was determined annually from dates spanning the first day Yellow Perch larvae were caught up until the 18°C water temperature cutoff criterion was met (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. The SD of L_p was estimated as

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

The 95% confidence intervals were constructed as

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

The percent difference between old and new (based on the temperature cutoff) L_p estimates was estimated as

$$(4) ((\text{old } L_p - \text{new } L_p) / \text{old } L_p) \cdot 100\%.$$

In a few years, persistent cool temperatures during late spring did not allow water temperatures to reach 18°C even though catches of Yellow Perch larvae had tapered off. A judgement was then made on cutoff date based on current catches, other systems in the area, and previous years sampling within that system. Water quality data were also unavailable for several systems/years and revised L_p estimates could not be calculated. L_p estimates for the following systems were then left as previously calculated: 1990 Mattawoman Creek; 2002 and 2004 Severn River; 2005 and 2006 Nanticoke River; and 2009 Bush River (Table 2-1). In 2020, historical Maryland DNR data sheets were mined to create georeferenced data on distribution of anadromous fish eggs and larvae (Striped Bass, White Perch, Yellow Perch, and Alosids), and water quality, in estuarine reaches (see Uphoff et al. 2022a). Using this data, we were also able to calculate L_p estimates for several years not previously done (Table 2-1) and include those in this report's updated analyses.

Methods used to estimate development (C/ha) and land use indicators (percent of watershed in agriculture, forest, wetlands, and urban land use) are explained in **General Spatial and Analytical Methods used in Job 1, Sections 1-3**. Development targets and limits, and general statistical methods (analytical strategy and equations) are described 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 2020 for Yellow Perch analyses (M. Topolski, MD DNR, personal communication).

Uphoff et al. (2012) developed L_p thresholds for brackish (salinity > 2.0‰ in the subestuary outside of the larval nursery) and tidal-fresh systems (salinity always ≤ 2.0‰). Choptank River was classified as brackish, while Sassafras River was classified as tidal-fresh. 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 15th out of 15 estimates). The maximum for Piscataway Creek's four estimates, $L_p = 0.59$ (updated from 0.65 after the recalculation of L_p using the 18°C cutoff; Table 2-1), 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 in decline, 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 (1980-2022; $N = 27$) and Nanticoke River (1963-2019; $N = 32$). Neither time-series was continuous; Choptank River estimates were available for 1980-1990, 1998-2004, and 2013-2022, while the Nanticoke River estimates were available for 1963-1968, 1970-1977, 1979, 1981, and 2004-2019.

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, leading to poor egg and larval viability (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, AIC_c , to evaluate the models that describe hypotheses that related changes in L_p to either C/ha for each salinity category (separate slopes), or to C/ha and salinity category (common slopes, separate intercepts; Burnham and Anderson 2001; Freund and Littell 2006):

$$(5) AIC_c = -2(\log\text{-likelihood}) + 2K + [(2K \cdot (K+1)) / (n-K-1)];$$

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 , ($AIC_{ci} - \text{minimum } AIC_c$), 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

2020). Urban land consisted of high and low density residential, commercial, and institutional acreages (MD DNR 1999).

We were also interested in whether L_p has been declining in Choptank River and Nanticoke River over the time-series, and if there might be an influence of increasing temperatures in Choptank River. We used linear and quadratic regressions to explore the relationship of L_p with year and L_p with mean temperature (data used for temperature estimates span the same dates used to estimate L_p). Linear and quadratic regressions explored this relationship for all data, with linear regression describing a hypothesis about steady change, while the dome-shaped quadratic relationship could indicate an optimum temperature for L_p (Freund and Littell 2006).

Results

The proportion of tows with Yellow Perch larvae (L_p) for each subestuary was recalculated in 2022 based on an 18°C temperature maximum (Table 2-1). Approximately 84% of old estimates were within $\pm 20\%$ of the new ones, median was 0%, and there was a slight tail out at positive values (Figure 2-2).

Sampling in 2022 began on Choptank River on March 22 and lasted until May 13, while sampling on Sassafra River began on March 30 and concluded on April 22. Samples through April 18 and April 22 were used to estimate L_p in Choptank and Sassafra Rivers, respectively.

The estimate of mean L_p was above the tidal-fresh threshold (0.59) in the Sassafra River ($L_p = 0.82$), and just above the brackish threshold (0.40) in the Choptank River ($L_p = 0.46$), during 2022 (Figure 2-3). Comparisons of L_p during 2022 with historical estimates for brackish subestuaries are plotted in Figure 2-4 and for tidal-fresh estimates in Figure 2-5. The range of C/ha values available for analysis with L_p was 0.05-2.86 for brackish subestuaries and 0.11-3.33 for tidal-fresh (Table 2-1).

Separate linear regressions of C/ha and L_p by salinity category indicated that C/ha was modestly and 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 2-6). Estimates of C/ha accounted for 19% of variation of L_p in brackish subestuaries and 32% in tidal-fresh subestuaries. Based on 95% CI overlap, intercepts were different between tidal-fresh (mean = 0.90, SE = 0.07) and brackish (mean = 0.53, SE = 0.03) subestuaries. The mean slope for C/ha estimated for tidal-fresh subestuaries (mean = -0.25, SE = 0.06) was steeper, but 95% CI's overlapped CI's estimated for the slope of brackish subestuaries (mean = -0.14, SE = 0.03; Table 2-2). Both regressions indicated that L_p would be extinguished between 3.0 and 3.5 C/ha (Figure 2-6).

Overall, the multiple regression approach offered a similar moderate fit of L_p with C/ha ($r^2 = 0.29$; Table 2-2) as separate regressions for each salinity type. Intercepts of tidal-fresh and brackish subestuaries equaled 0.90 and 0.53, respectively; the common slope was -0.16. Predicted L_p over the observed ranges of C/ha available for each salinity type (the range for tidal-fresh was smaller than for brackish) would decline from 0.53 to 0.14 in brackish subestuaries and from 0.87 to 0.07 in tidal-fresh subestuaries (Figure 2-6).

Estimates of L_p were weakly related to agriculture ($r^2 = 0.14$, $P = 0.0005$) and unrelated to forest ($r^2 = 0.005$, $P = 0.51$) in brackish tributaries (Table 2-2; Figure 2-6). Regressions of L_p and agriculture or forest in tidal-fresh subestuaries indicated no relationship (Table 2-2), while the multiple regression approach indicated weak relationships with C/ha (agriculture, $r^2 = 0.20$, $P < 0.0001$; and forest, $r^2 = 0.11$, $P = 0.0009$; Table 2-2; Figure 2-6). 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 9.4 for the regression of C/ha and L_p for brackish subestuaries, 9.9 for tidal-fresh estuaries, and 11.4 for the multiple regression that included salinity category (Table 2-3). Calculations of Δi for brackish or tidal-fresh versus multiple regressions were approximately 2.04 and 1.54 (respectively), indicating that either hypothesis (different intercepts for tidal-fresh and brackish subestuaries with different or common slopes describing the decline of L_p with C/ha) were plausible (Table 2-3).

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, $r^2 = 0.09$, $P = 0.001$; size, $R^2 = 0.22$, $P < 0.0001$; and salinity, $R^2 = 0.29$, $P < 0.0001$), but combining them into a single model did not improve the fit and size was no longer significant (combined $R^2 = 0.31$; salinity, $P = 0.0004$ and size, $P = 0.1553$). Considering size separately, all tidal-fresh systems are within the small-system size category, so fit did not change from previous analyses ($r^2 = 0.32$, $P = 0.0005$; Tables 2-2 and 2-4, respectively). The relationship between C/ha and L_p in small, brackish systems was better explained, however ($r^2 = 0.52$, $P = 0.0001$; Table 2-4). A relationship between C/ha and L_p was not detected for large systems (Table 2-4).

In 2022, temperatures were similar in rural, agricultural Choptank and Sassafras Rivers, but DO and pH values were significantly higher in the Sassafras (Table 2-5; Figure 2-7). Harmful algal blooms have been documented several times in the Sassafras River (see Section 3 of Uphoff et al. 2022a; MD DNR 2022b) and while a phytoplankton bloom was not noted during sampling, this could contribute to the high DO and pH values observed. While these differences are not likely to be fatal to Yellow Perch larvae, they do point to differences in dynamics and conditions among 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 with below threshold development (at or below C/ha target) in tidal-fresh subestuaries were dominated by forest, with only a single low development, low salinity watershed with agricultural as its dominant land use available (Figure 2-8). Dominant land cover estimated by MD DOP for watersheds of tidal-fresh subestuaries was split between forest (C/ha = 0.46-0.96; 18 observations) and urban (C/ha \geq 1.17; 14 observations). Nearly all rural land in brackish subestuary watersheds was in agriculture (C/ha \leq 0.22; 63 observations), while forest land cover was represented by six observations from Nanjemoy Creek (C/ha = 0.09) and two from Wicomico River (eastern shore; 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 \geq 1.24; Table 2-1; Figure 2-8). 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.

Regression analyses indicated that mean temperature may have limited influence on L_p in the Choptank River. A linear regression of mean temperature and L_p was not significant, but indicated a potential relationship ($r^2 = 0.12$, $P = 0.08$; Table 2-6), and examination of the plot of

L_p and mean temperature suggested a dome-shaped quadratic relationship (Figure 2-9). Quadratic regression fit the data better and suggested a modest relationship between L_p and mean temperature ($R^2 = 0.25$, $P = 0.03$; Table 2-6). The quadratic model suggested an optimum temperature of about 14.85°C that would produce an $L_p = 0.67$ (Figure 2-9).

Previously, strong relationships of L_p with year were not evident in the Choptank or Nanticoke Rivers. After L_p was recalculated using the 18°C cutoff (Table 2-1), a decline of L_p of about 0.005 per year was detected during 1980-2022 (predicted L_p declined from 0.70 to 0.47) in the Choptank River ($r^2 = 0.16$; $P = 0.04$; Table 2-7; Figure 2-10). A quadratic model appeared to fit the data better ($R^2 = 0.29$; $P = 0.02$; Table 2-7; Figure 2-10) so AICc was used to evaluate the models that described changes to L_p related to year. The declining linear model fit best, but the quadratic (no decline) model was also reasonably supported ($\Delta AICc = 2.9$), suggesting either was plausible (Table 2-7). Estimates of L_p in Nanticoke River during 1963-2019 exhibited no indication of decline ($r^2 = 0.02$; $P = 0.46$; Figure 2-10). These subestuaries are both rural, land use is dominated by agriculture, and they have been closed to commercial fishing since 1989 (Piavis 2005). Their different trends in L_p could reflect the use of plankton trawls in Choptank River in the 1980s that would have been more efficient at capturing larvae (see Section 2.1). The highest L_p in the 1980s also corresponded to a fixed station design rather than the stratified random one employed since 1987. With years that used a fixed station design removed (1980-1986), the Choptank River trend is a weak linear ($r^2 = 0.02$; $P = 0.56$; Figure 2-10) like the trend for Nanticoke River. All in all, we cannot conclude that there has been a decline in L_p over time in either river.

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. However, wetlands may be an important source of organic matter that influences Yellow Perch larval feeding success (Uphoff et al. 2016; 2022b).

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 ≤ 0.59 . 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 $L_p = 0.74$) than agriculture (median $L_p = 0.50$) or development (median $L_p = 0.35$), but these differences may have also reflected dynamics unique to brackish or tidal-fresh subestuaries since all but one agricultural watershed 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. Some of 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, and an expanded analysis using data from 1980-1990 found that 93.5% were from 1‰ or less (C. Hoover, MD DNR, unpublished analysis). Approximately 85% of Yellow Perch larvae

collected by Dovel (1971) from Magothy and Patuxent rivers, and Head-of-Bay, during 1963-1967 were collected at salinity 1‰ or less.

Severn River offers the most extensive evidence of salinity changes in a subestuary that were concurrent with development from 0.35 to 2.30 C/ha. During 2001-2003 salinity within Severn River's estuarine Yellow Perch larval nursery ranged between 0.5 and 13‰ (C/ha was ~ 2.0); 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 (C/ha was 0.35 in 1950 and rose to 1.01 by 1976), 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 in fish due to endocrine disruption may follow inadequate DO the previous summer (Wu et al. 2003; 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; Breitbart 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).

Yellow Perch and Striped Bass larvae are found in the same regions of large tidal rivers in Chesapeake Bay (Uphoff 1991; 2020). Copepods, typically of the genus *Eurytemora*, were important prey of Striped Bass and Yellow Perch larvae (MD Sea Grant 2009; Uphoff et al. 2016). Winter water temperature has also been found to have an influence on peak abundances of an important zooplankton prey (*Eurytemora carolleeae*) 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). Given the high correlation of Striped Bass and Yellow Perch juvenile indices in Maryland's portion of Chesapeake Bay and high concurrence of their larvae in their nursery (Uphoff et al. 2020), we would expect these same factors would impact Yellow Perch recruitment.

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 \geq 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 through 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 eleven estimates of L_p have been below the threshold in the three western shore subestuaries with well-developed watersheds during 2002-2016, while estimates from Head-of-Bay subestuaries have typically been above the threshold (5 of 7 Bush River estimates, 3 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).

Higher DO and pH values in rural Sassafras River likely reflected 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 during 2015 and 2016 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. 2016).

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 (<https://www.epa.gov/chesapeake-bay-tmdl/chesapeake-bay-tmdl-fact-sheet>). 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.

Annual L_p (proportion of tows with Yellow Perch larvae during a standard period of time, 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 an ecosystem approach to 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. 2016) required laboratory analysis. These latter two analyses represented separate studies rather than a requirement for estimating L_p (Uphoff et al. 2016).

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). 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 two samples) 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.

Hilborn (2016) reviewed the use of correlation in fisheries and ecosystem management and this advice should apply to regression analyses that we used since the underlying math is very similar. Ideally, manipulative experiments and formal adaptive management should be employed. In large-scale aquatic ecosystems these opportunities are limited and are not a possibility for us. Correlations may not be causal, but they represent 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).

Development was an important influence on Yellow Perch egg and early 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 1963-2022 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 ($\leq 2.0\text{‰}$) and 1 = brackish ($> 2.0\text{‰}$). 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. Old L_p are estimates previously reported, while New L_p are recent calculations using an 18°C cutoff. % Diff is percent difference between the two methods, (---) indicates data was unavailable to calculate new L_p estimates so no percent difference was calculated, and (*) indicates systems that did not have L_p reported previously.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Old L_p	New L_p	% Diff
Bush (w/ APG)	2006	2002	1.17	21	36.3	5.5	37	Urban	0	0.79	0.78	0.01
Bush (w/ APG)	2007	2010	1.19	14.9	32.1	5.5	46.4	Urban	0	0.92	0.90	0.02
Bush (w/ APG)	2008	2010	1.20	14.9	32.1	5.5	46.4	Urban	0	0.55	0.69	-0.25
Bush (w/ APG)	2009	2010	1.21	14.9	32.1	5.5	46.4	Urban	0	0.86	0.86	---
Bush (w/ APG)	2011	2010	1.23	14.9	32.1	5.5	46.4	Urban	0	0.96	0.96	0.00
Bush (w/ APG)	2012	2010	1.24	14.9	32.1	5.5	46.4	Urban	0	0.28	0.34	-0.21
Bush (w/ APG)	2013	2010	1.25	14.9	32.1	5.5	46.4	Urban	0	0.15	0.15	0.00
Chester	2019	2010	0.13	66.6	24.5	0.8	7.8	Agriculture	1	0.73	0.82	-0.12
Choptank	1980	1973	0.07	65.2	30.6	2	2.1	Agriculture	1	0.59	0.71	-0.20
Choptank	1981	1973	0.07	65.2	30.6	2	2.1	Agriculture	1	0.82	0.86	-0.05
Choptank	1982	1973	0.07	65.2	30.6	2	2.1	Agriculture	1	0.80	0.89	-0.11
Choptank	1983	1973	0.07	65.2	30.6	2	2.1	Agriculture	1	0.33	0.32	0.03
Choptank	1984	1994	0.07	64	29.2	2.3	4.4	Agriculture	1	0.64	0.71	-0.11
Choptank	1985	1994	0.07	64	29.2	2.3	4.4	Agriculture	1	0.85	1.00	-0.18
Choptank	1986	1994	0.07	64	29.2	2.3	4.4	Agriculture	1	0.80	0.73	0.09
Choptank	1987	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.76	0.75	0.01
Choptank	1988	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.63	0.70	-0.11
Choptank	1989	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.64	0.64	0.00
Choptank	1990	1994	0.08	64	29.2	2.3	4.4	Agriculture	1	0.66	0.62	0.06
Choptank	1998	1997	0.10	63.6	27.7	2.2	6.4	Agriculture	1	0.61	0.57	0.07
Choptank	1999	1997	0.11	63.6	27.7	2.2	6.4	Agriculture	1	0.75	0.60	0.20
Choptank	2000	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.27	0.19	0.30
Choptank	2001	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.26	0.25	0.04
Choptank	2002	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.38	0.32	0.16
Choptank	2003	2002	0.11	63.9	27.1	2.1	6.9	Agriculture	1	0.52	0.54	-0.04
Choptank	2004	2002	0.12	63.9	27.1	2.1	6.9	Agriculture	1	0.41	0.50	-0.22

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Old Lp	New Lp	% Diff
Choptank	2013	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.47	0.58	-0.23
Choptank	2014	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.68	0.68	0.00
Choptank	2015	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.82	0.81	0.01
Choptank	2016	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.90	0.59	0.34
Choptank	2017	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.40	0.43	-0.07
Choptank	2018	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.44	0.44	0.00
Choptank	2019	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.69	0.68	0.01
Choptank	2021	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	0.41	0.44	-0.07
Choptank	2022	2010	0.13	60.9	25.6	2.1	11.2	Agriculture	1	*	0.46	*
Corsica	2006	2002	0.21	64.3	27.4	0.4	7.9	Agriculture	1	0.47	0.47	0.00
Corsica	2007	2010	0.22	60.4	25.5	0.1	13.2	Agriculture	1	0.83	0.83	0.00
Elk	2010	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.75	0.75	0.00
Elk	2011	2010	0.59	28	38.7	1.1	31.2	Forest	0	0.79	0.79	0.00
Elk	2012	2010	0.60	28	38.7	1.1	31.2	Forest	0	0.55	0.66	-0.20
Langford	2007	2010	0.07	70.2	20.4	1.5	8	Agriculture	1	0.83	0.54	0.35
Magothy	2009	2010	2.74	1.2	21	0	76.8	Urban	1	0.10	0.10	0.00
Magothy	2016	2010	2.86	1.2	21	0	76.8	Urban	1	0.10	0.10	0.00
Mattawoman	1990	1994	0.46	13.8	62.6	0.9	22.5	Forest	0	0.81	0.81	---
Mattawoman	2008	2010	0.87	9.3	53.9	2.8	34.2	Forest	0	0.66	0.58	0.12
Mattawoman	2009	2010	0.88	9.3	53.9	2.8	34.2	Forest	0	0.92	0.90	0.02
Mattawoman	2010	2010	0.90	9.3	53.9	2.8	34.2	Forest	0	0.82	0.82	0.00
Mattawoman	2011	2010	0.91	9.3	53.9	2.8	34.2	Forest	0	0.98	0.92	0.06
Mattawoman	2012	2010	0.90	9.3	53.9	2.8	34.2	Forest	0	0.20	0.20	0.00
Mattawoman	2013	2010	0.92	9.3	53.9	2.8	34.2	Forest	0	0.47	0.64	-0.36
Mattawoman	2014	2010	0.93	9.3	53.9	2.8	34.2	Forest	0	0.78	0.67	0.14
Mattawoman	2015	2010	0.94	9.3	53.9	2.8	34.2	Forest	0	1.00	1.00	0.00
Mattawoman	2016	2010	0.96	9.3	53.9	2.8	34.2	Forest	0	0.82	0.90	-0.10
Middle	2012	2010	3.33	3.4	23.3	2.1	71	Urban	0	0.00	0.00	0.00
Nanjemoy	2009	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.83	0.74	0.11
Nanjemoy	2010	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.96	0.90	0.06
Nanjemoy	2011	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.98	0.92	0.06

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Old Lp	New Lp	% Diff
Nanjemoy	2012	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.03	0.03	0.00
Nanjemoy	2013	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.46	0.52	-0.13
Nanjemoy	2014	2010	0.09	12.4	68.7	4.1	14.7	Forest	1	0.82	0.88	-0.07
Nanticoke	1963	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	*	0.65	*
Nanticoke	1964	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	*	0.50	*
Nanticoke	1965	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	0.50	0.34	0.32
Nanticoke	1966	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	*	0.39	*
Nanticoke	1967	1973	0.05	46.6	43.4	8.1	1.9	Agriculture	1	0.43	0.29	0.33
Nanticoke	1968	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	1.00	0.40	0.60
Nanticoke	1970	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	0.81	0.65	0.20
Nanticoke	1971	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	0.33	0.24	0.27
Nanticoke	1972	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	*	0.26	*
Nanticoke	1973	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	*	0.53	*
Nanticoke	1974	1973	0.06	46.6	43.4	8.1	1.9	Agriculture	1	*	0.35	*
Nanticoke	1975	1973	0.07	46.6	43.4	8.1	1.9	Agriculture	1	*	0.48	*
Nanticoke	1976	1973	0.07	46.6	43.4	8.1	1.9	Agriculture	1	*	0.30	*
Nanticoke	1977	1973	0.07	46.6	43.4	8.1	1.9	Agriculture	1	*	0.72	*
Nanticoke	1979	1973	0.07	46.6	43.4	8.1	1.9	Agriculture	1	*	0.30	*
Nanticoke	1981	1973	0.08	46.6	43.4	8.1	1.9	Agriculture	1	*	0.39	*
Nanticoke	2004	2002	0.11	46.3	40.7	7.4	5.5	Agriculture	1	0.49	0.49	0.00
Nanticoke	2005	2002	0.11	46.3	40.7	7.4	5.5	Agriculture	1	0.67	0.67	---
Nanticoke	2006	2002	0.11	46.3	40.7	7.4	5.5	Agriculture	1	0.35	0.35	---
Nanticoke	2007	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.55	0.69	-0.25
Nanticoke	2008	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.19	0.11	0.42
Nanticoke	2009	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.41	0.32	0.22
Nanticoke	2010	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	*	0.39	*
Nanticoke	2011	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.55	0.55	0.00
Nanticoke	2012	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.04	0.04	0.00
Nanticoke	2013	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.43	0.48	-0.12
Nanticoke	2014	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.35	0.35	0.00
Nanticoke	2015	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.64	0.59	0.08

Table 2-1 cont.

River	Sample Year	DOP Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Old Lp	New Lp	% Diff
Nanticoke	2016	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.67	0.38	0.43
Nanticoke	2017	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.22	0.22	0.00
Nanticoke	2018	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.28	0.28	0.00
Nanticoke	2019	2010	0.11	45	39.4	7.4	8.1	Agriculture	1	0.41	0.41	0.00
Northeast	2010	2010	0.46	31.1	38.6	0.1	28.9	Forest	0	0.68	0.68	0.00
Northeast	2011	2010	0.46	31.1	38.6	0.1	28.9	Forest	0	1.00	1.00	0.00
Northeast	2012	2010	0.47	31.1	38.6	0.1	28.9	Forest	0	0.77	0.66	0.14
Northeast	2013	2010	0.48	31.1	38.6	0.1	28.9	Forest	0	0.72	0.72	0.00
Northeast	2014	2010	0.48	31.1	38.6	0.1	28.9	Forest	0	0.77	0.77	0.00
Patuxent	2015	2010	1.24	20.5	35.1	1	41.7	Urban	1	0.72	0.74	-0.03
Patuxent	2016	2010	1.25	20.5	35.1	1	41.7	Urban	1	0.82	0.72	0.12
Piscataway	2008	2010	1.41	10	40.4	0.2	47	Urban	0	0.47	0.41	0.13
Piscataway	2009	2010	1.43	10	40.4	0.2	47	Urban	0	0.39	0.39	0.00
Piscataway	2010	2010	1.45	10	40.4	0.2	47	Urban	0	0.54	0.54	0.00
Piscataway	2011	2010	1.46	10	40.4	0.2	47	Urban	0	0.65	0.59	0.09
Piscataway	2012	2010	1.47	10	40.4	0.2	47	Urban	0	0.16	0.18	-0.13
Piscataway	2013	2010	1.50	10.00	40.40	0.20	47.00	Urban	0	0.50	0.59	-0.18
Sassafras	2021	2010	0.11	64.10	25.90	1.30	8.30	Agriculture	0	0.60	0.60	0.00
Sassafras	2022	2010	0.11	64.10	25.90	1.30	8.30	Agriculture	0	*	0.82	*
Severn	2002	2002	2.02	8.60	35.20	0.20	55.80	Urban	1	0.16	0.16	---
Severn	2004	2002	2.09	8.60	35.20	0.20	55.80	Urban	1	0.35	0.35	---
Severn	2005	2002	2.15	8.60	35.20	0.20	55.80	Urban	1	0.40	0.40	0.00
Severn	2006	2002	2.18	8.60	35.20	0.20	55.80	Urban	1	0.27	0.24	0.11
Severn	2007	2010	2.21	5.00	28.00	0.20	65.10	Urban	1	0.30	0.35	-0.17
Severn	2008	2010	2.24	5.00	28.00	0.20	65.10	Urban	1	0.08	0.08	0.00
Severn	2009	2010	2.25	5.00	28.00	0.20	65.10	Urban	1	0.15	0.13	0.13
Severn	2010	2010	2.26	5.00	28.00	0.20	65.10	Urban	1	0.03	0.03	0.00
South	2008	2010	1.32	10.20	39.20	0.50	48.80	Urban	1	0.14	0.12	0.14
Wicomico (ES)	2017	2010	0.68	30.10	36.80	2.30	29.90	Forest	1	0.53	0.46	0.13
Wicomico (ES)	2018	2010	0.68	30.10	36.80	2.30	29.90	Forest	1	0.38	0.34	0.11

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	0.87629	0.87629	19.08	<.0001	
Error	82	3.76698	0.04594			
Total	83	4.64327				
r^2	0.1887					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.53471	0.02683	19.93	<.0001	0.48133	0.58808
C / ha	-0.13858	0.03173	-4.37	<.0001	-0.2017	-0.07546

ANOVA		(A) Tidal-Fresh				
Source	df	SS	MS	F	P	
Model	1	0.67804	0.67804	14.83	0.0005	
Error	32	1.46301	0.04572			
Total	33	2.14105				
r^2	0.3167					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.90191	0.07386	12.21	<.0001	0.75146	1.05236
C / ha	-0.25015	0.06495	-3.85	0.0005	-0.38245	-0.11784

ANOVA		(A) Multiple Regression				
Source	df	SS	MS	F	P	
Model	2	2.20995	1.10498	23.8	<.0001	
Error	115	5.33897	0.04643			
Total	117	7.54893				
r^2	0.2928					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.81292	0.04655	17.46	<.0001	0.72072	0.90512
C / ha	-0.15999	0.02867	-5.58	<.0001	-0.21679	-0.10319
Salinity	-0.26934	0.04677	-5.76	<.0001	-0.36199	-0.17669

Table 2-2 cont.

ANOVA		(B) Brackish				
Source	df	SS	MS	F	P	
Model	1	0.63665	0.63665	13.03	0.0005	
Error	82	4.00662	0.04886			
Total	83	4.64327				
r^2	0.1371					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.29139	0.05686	5.12	<.0001	0.17827	0.4045
% Ag	0.00425	0.00118	3.61	0.0005	0.00191	0.00659

ANOVA		(B) Tidal-Fresh				
Source	df	SS	MS	F	P	
Model	1	0.11218	0.11218	1.77	0.1929	
Error	32	2.02887	0.0634			
Total	33	2.14105				
r^2	0.0524					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57838	0.07199	8.03	<.0001	0.43174	0.72502
% Ag	0.00408	0.00306	1.33	0.1929	-0.00217	0.01032

ANOVA		(B) Multiple Regression				
Source	df	SS	MS	F	P	
Model	2	1.51327	0.75664	14.42	<.0001	
Error	115	6.03565	0.05248			
Total	117	7.54893				
r^2	0.2005					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57571	0.04455	12.92	<.0001	0.48747	0.66395
% Ag	0.00422	0.00112	3.78	0.0003	0.00201	0.00643
Salinity	-0.28313	0.05429	-5.22	<.0001	-0.39067	-0.1756

Table 2-2 cont.

ANOVA		(C) Brackish				
Source	df	SS	MS	F	P	
Model	1	0.02493	0.02493	0.44	0.5077	
Error	82	4.61834	0.05632			
Total	83	4.64327				
r^2	0.0054					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.42162	0.08755	4.82	<.0001	0.24746	0.59579
% Forest	0.00153	0.00229	0.67	0.5077	-0.00304	0.00609

ANOVA		(C) Tidal-Fresh				
Source	df	SS	MS	F	P	
Model	1	0.13861	0.13861	2.22	0.1465	
Error	32	2.00244	0.06258			
Total	33	2.14105				
r^2	0.0647					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.38652	0.18543	2.08	0.0452	0.00882	0.76422
% Forest	0.00651	0.00437	1.49	0.1465	-0.0024	0.01541

ANOVA		(C) Multiple Regression				
Source	df	SS	MS	F	P	
Model	2	0.86592	0.43296	7.45	0.0009	
Error	115	6.68301	0.05811			
Total	117	7.54893				
r^2	0.1147					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.54395	0.09372	5.8	<.0001	0.35832	0.72959
% Forest	0.00269	0.00204	1.32	0.1893	-0.00135	0.00673
Salinity	-0.16486	0.04996	-3.3	0.0013	-0.26382	-0.06589

Table 2-3. Summary of Akaike's Information Criteria for small N from regressions of proportions of tows with Yellow Perch larvae (L_p) and counts of structures per hectare (C/ha) for each salinity category, and a multiple regression using salinity as a class variable.

Model	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.04643	118	4	3.069809477	8	40	113	11.4	2.04	1.54
Fresh	0.04572	34	3	3.08521944	6	24	30	9.9		
Brackish	0.04594	84	3	3.080419082	6	24	80	9.4		

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 \leq 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	0.98767	0.98767	21.83	0.0001	
Error	20	0.90488	0.04524			
Total	21	1.89255				
r^2	0.5219					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.64168	0.0696	9.22	<.0001	0.4965	0.78686
C / ha	-0.2031	0.04347	-4.67	0.0001	-0.29378	-0.11243

Table 2-4 cont.

ANOVA		(B) Small Tidal-Fresh				
Source	df	SS	MS	F	P	
Model	1	0.67804	0.67804	14.83	0.0005	
Error	32	1.46301	0.04572			
Total	33	2.14105				
r^2	0.3167					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.90191	0.07386	12.21	<.0001	0.75146	1.05236
C / ha	-0.25015	0.06495	-3.85	0.0005	-0.38245	-0.11784

ANOVA		(C) Large Brackish				
Source	df	SS	MS	F	P	
Model	1	0.02024	0.02024	0.53	0.4719	
Error	28	1.06566	0.03806			
Total	29	1.0859				
r^2	0.0186					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.59341	0.04202	14.12	<.0001	0.50733	0.67948
C / ha	0.09084	0.12457	0.73	0.4719	-0.16433	0.346

Table 2-5. Summary of water quality parameter statistics for Choptank and Sassafras Rivers sampled 2022. Mean pH was calculated from H⁺ concentrations, then converted to pH.

System/Year		Temp C	DO (mg/L)	Cond (umhols)	pH
Choptank 22	Mean	13.48	8.02	732.15	6.96
	Standard Error	0.21	0.18	98.42	
	Median	13.49	8.24	340	7.11
	Mode	13.48	8.65	310	7.18
	Kurtosis	-1.05	-0.69	1.53	-0.66
	Skewness	0.17	-0.34	1.60	-0.22
	Minimum	10.80	4.33	99	6.58
	Maximum	16.78	11.12	3419	7.66
	Count	74	74	74	74
Sassafras 22	Mean	11.37	11.92	473.47	8.13
	Standard Error	0.34	0.10	11.85	
	Median	11.60	11.98	434	8.13
	Mode	6.56	12.56	599	8.03
	Kurtosis	-0.91	1.75	-0.99	-0.02
	Skewness	-0.48	0.73	0.35	0.92
	Minimum	6.46	10.48	320	7.80
	Maximum	14.94	14.70	667	9.05
	Count	60	60	60	60

Table 2-6. Summary of results of regressions of proportions of tows with Yellow Perch larvae (L_p) and Mean Temperature (data used for temperature estimates span the same dates used to estimate L_p) in Choptank River from 1980-2022. (A) linear regression, or (B) multiple regression using a quadratic.

(A) Choptank L_p						
Source	df	SS	MS	F	P	
Model	1	0.11756	0.11756	3.31	0.0808	
Error	25	0.88741	0.0355			
Total	26	1.00496				
r^2	0.1170					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	-0.33012	0.50853	-0.65	0.5221	-1.37746	0.71721
Mean Temperature	0.06395	0.03514	1.82	0.0808	-0.00842	0.13632

Table 2-6 cont.

(B) Choptank Lp					
Source	df	SS	MS	F	P
Model	2	0.25175	0.12587	4.01	0.0314
Error	24	0.75322	0.03138		
Total	26	1.00496			
r^2	0.2505				
	Coefficients	SE	t Stat	P-value	
Intercept	-12.47388	5.89229	-2.12	0.0448	
Mean Temperature	1.76735	0.82445	2.14	0.0424	
Mean Temp sq	-0.05942	0.02874	-2.07	0.0496	

Table 2-7. Summary of results, and Akaike’s Information Criteria for small N, from regressions of proportions of tows with Yellow Perch larvae (L_p) and Sample Year in Choptank River from 1980-2022. (A) linear regression, or (B) multiple regression using a quadratic, and (C) AICc.

(A) Choptank Lp						
Source	df	SS	MS	F	P	
Model	1	0.1596	0.1596	4.72	0.0395	
Error	25	0.84536	0.03381			
Total	26	1.00496				
r^2	0.1588					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	11.50118	5.02107	2.29	0.0307	1.1601	21.84227
Sample Year	-0.00545	0.00251	-2.17	0.0395	-0.01063	-0.000284

Table 2-7 cont.

(B) Choptank Lp					
Source	df	SS	MS	F	P
Model	2	0.28843	0.14421	4.83	0.0173
Error	24	0.71653	0.02986		
Total	26	1.00496			
r^2	0.2870				
	Coefficients	SE	t Stat	P-value	
Intercept	1933.5850	925.3099	2.09	0.0474	
Sample Year	-1.92687	0.92498	-2.08	0.0481	
Sample Year sq	0.0004802	0.000231	2.08	0.0486	

(C) Model	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	$\Delta AICc$
Year linear	0.03381	27	3	3.386998662	6	24	23	10.4	0.00
Year quad	0.02986	27	4	3.511235487	8	40	22	13.3	2.90

Figure 2-1. Subestuaries sampled for Yellow Perch larval presence-absence studies, 2006-2022. Watersheds of subestuaries sampled during 2006-2021 are indicated by labels and those sampled in 2022 are highlighted in green and have bolded text labels; watershed delineation is for Maryland only.

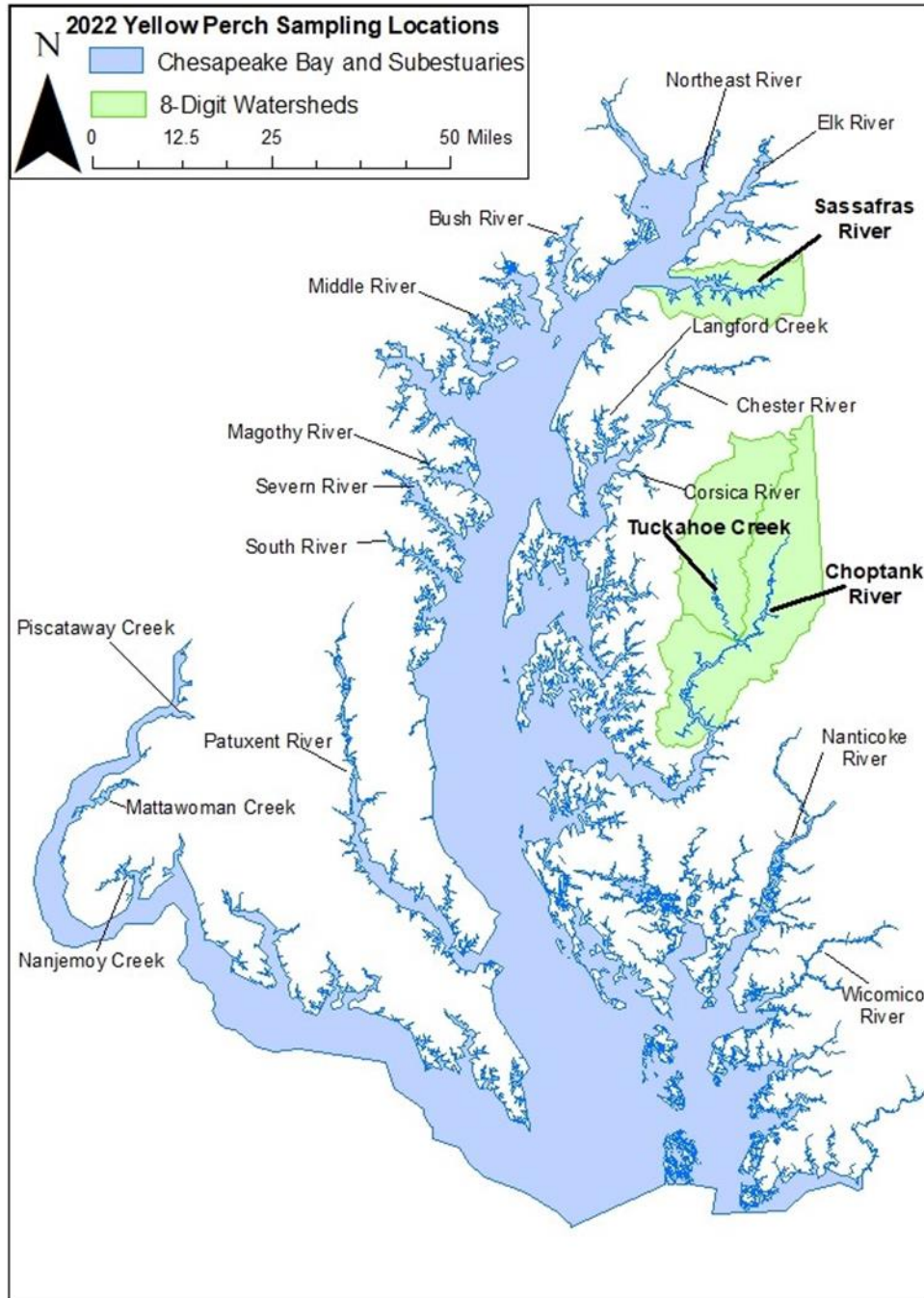


Figure 2-2. Distribution of percent difference between previous estimates of proportion of tows with larval Yellow Perch (*Lp*) and new estimates based on an 18°C temperature maximum.

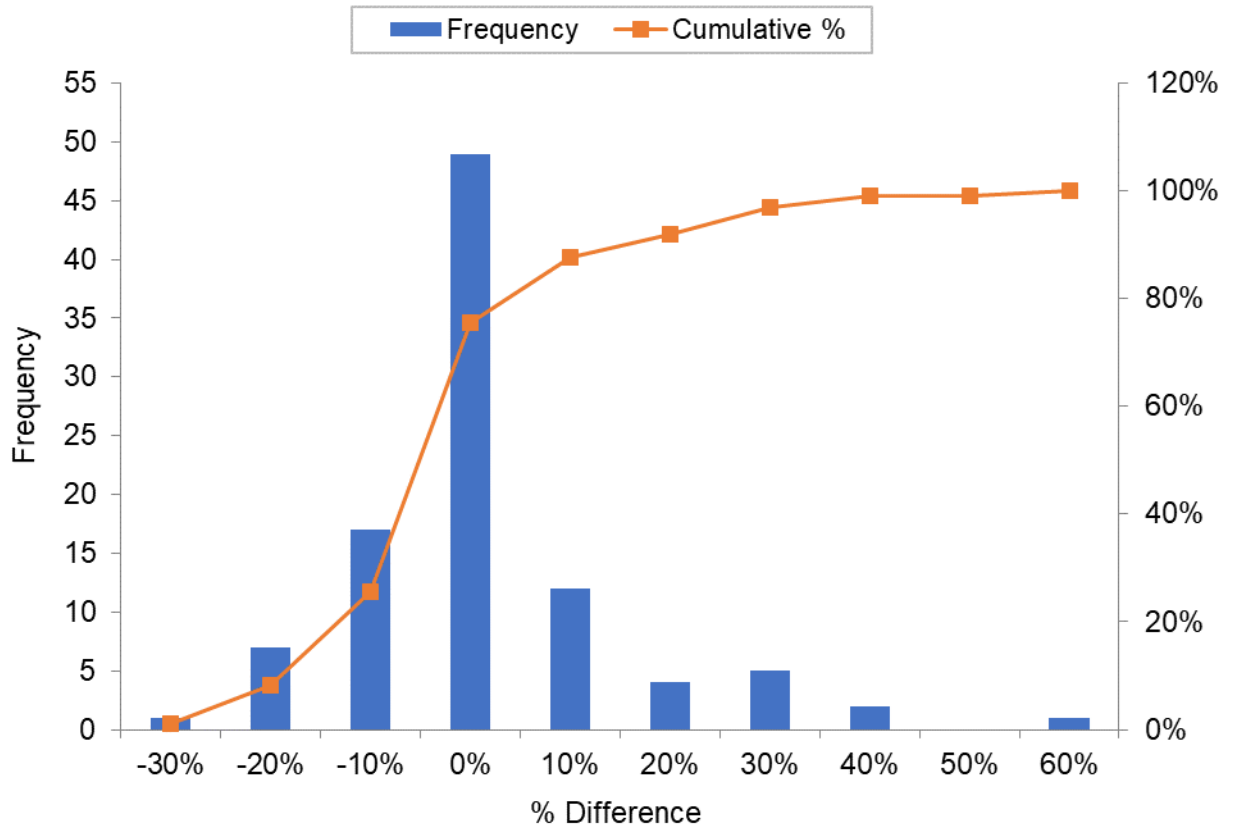


Figure 2-3. Proportion of tows with larval Yellow Perch (*Lp*) and their 95% confidence intervals in systems studied during 2022. Mean *Lp* of brackish Choptank River, and tidal-fresh Sassafras River, is indicated by green triangle and blue circle, respectively. Brackish subestuary *Lp* threshold is indicated by a green dotted line, and tidal-fresh threshold is a blue dotted line.

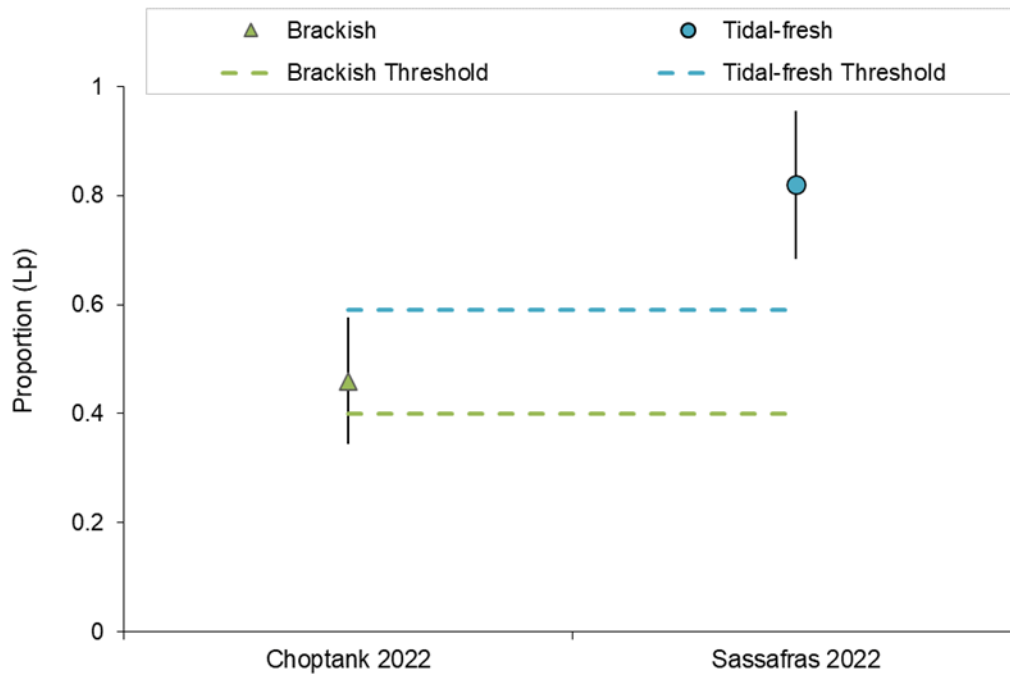


Figure 2-4. Proportion of tows with Yellow Perch larvae (*Lp*) for brackish subestuaries, during 1963-2022. Dotted line provides threshold for persistent poor *Lp* exhibited in developed brackish subestuaries. Dominant land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

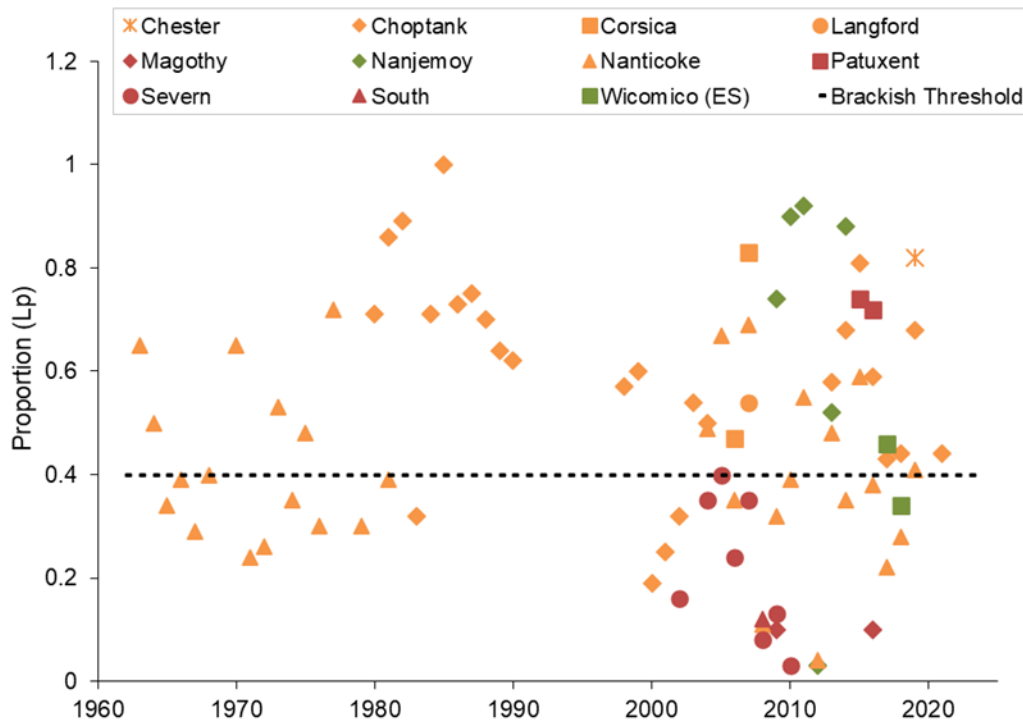


Figure 2-5. Proportion of tows with Yellow Perch larvae (*Lp*) for tidal-fresh subestuaries, during 1990-2022. Dotted line provides reference for consistent poor *Lp* exhibited in a more developed tidal-fresh subestuary (Piscataway Creek). Dominant Department of Planning land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

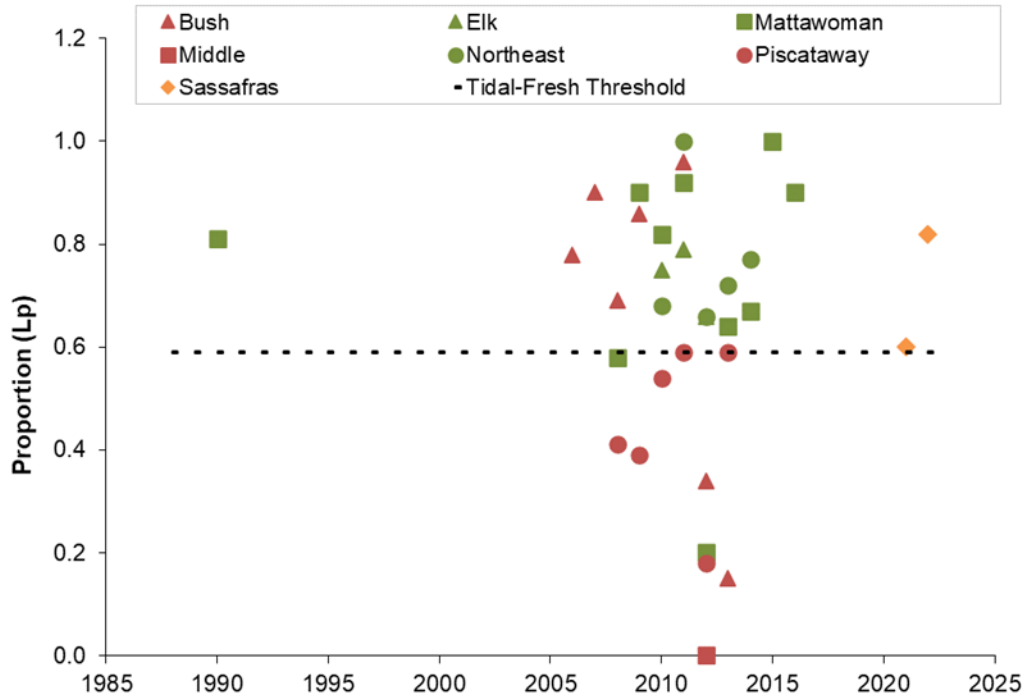


Figure 2-6. Relationship of proportion of plankton tows with Yellow Perch larvae (*Lp*) and (A) development (structures per hectare or C/ha), (B) percent agriculture, and (C) percent forest, indicated by multiple regression of fresh and brackish subestuaries combined (prediction = MR) and separate linear regressions for both (prediction = LR).

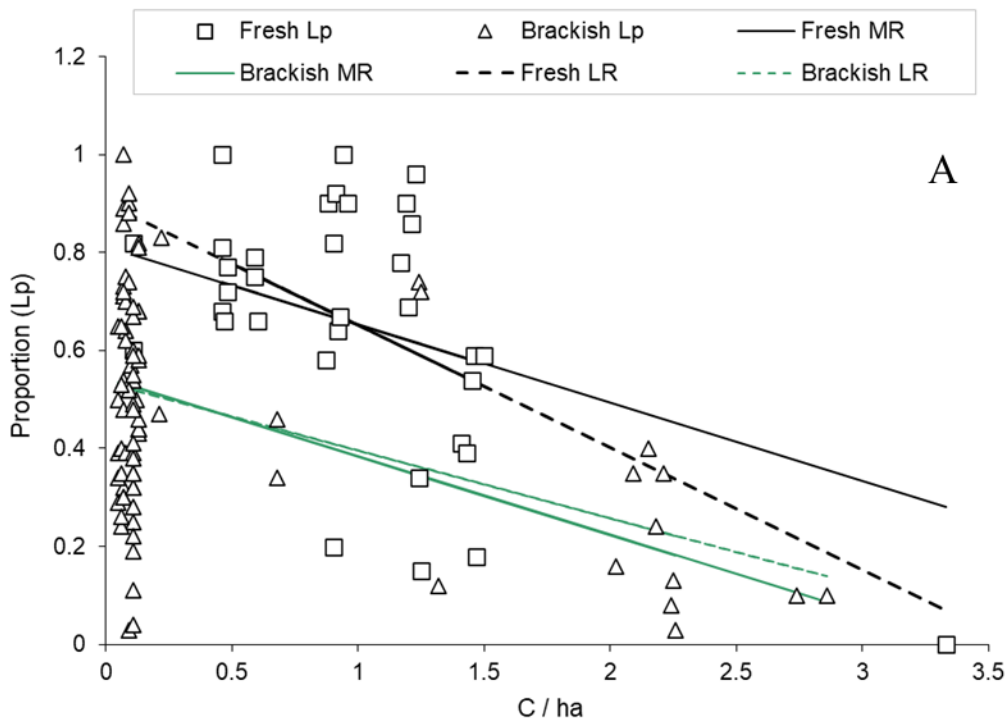


Figure 2-6 cont.

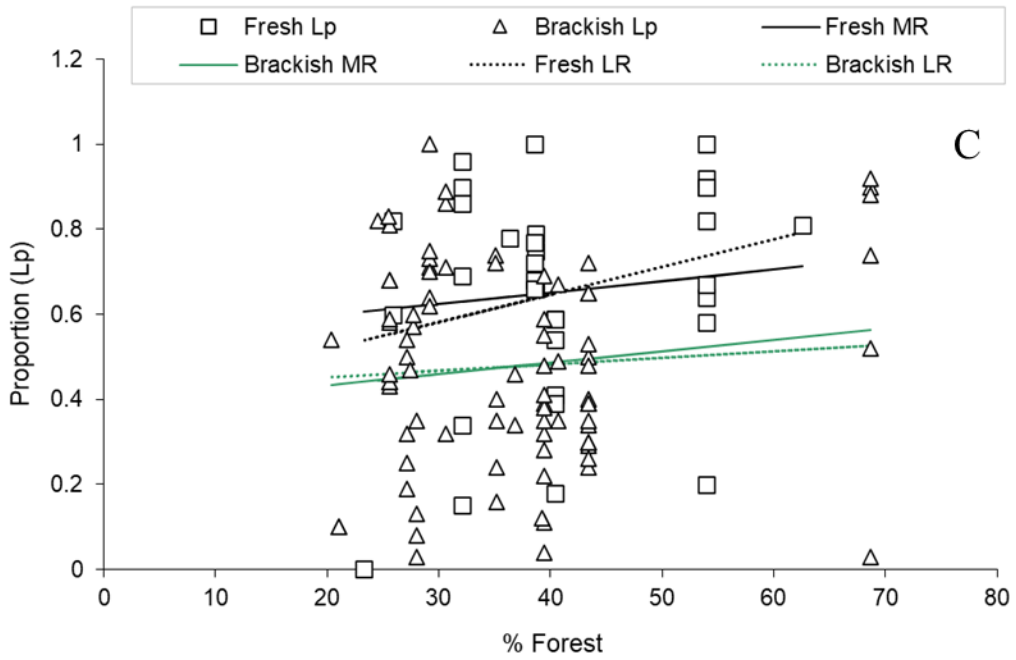
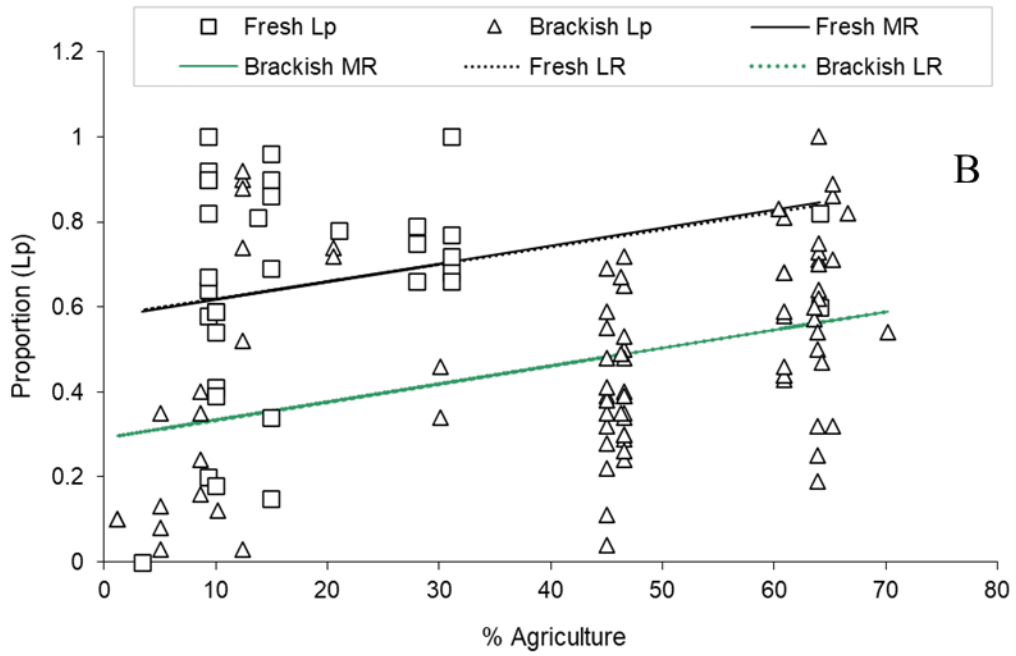


Figure 2-7. Individual values of water temperature, DO, conductivity, and pH, by date sampled, in Choptank and Sassafras Rivers during 2022.

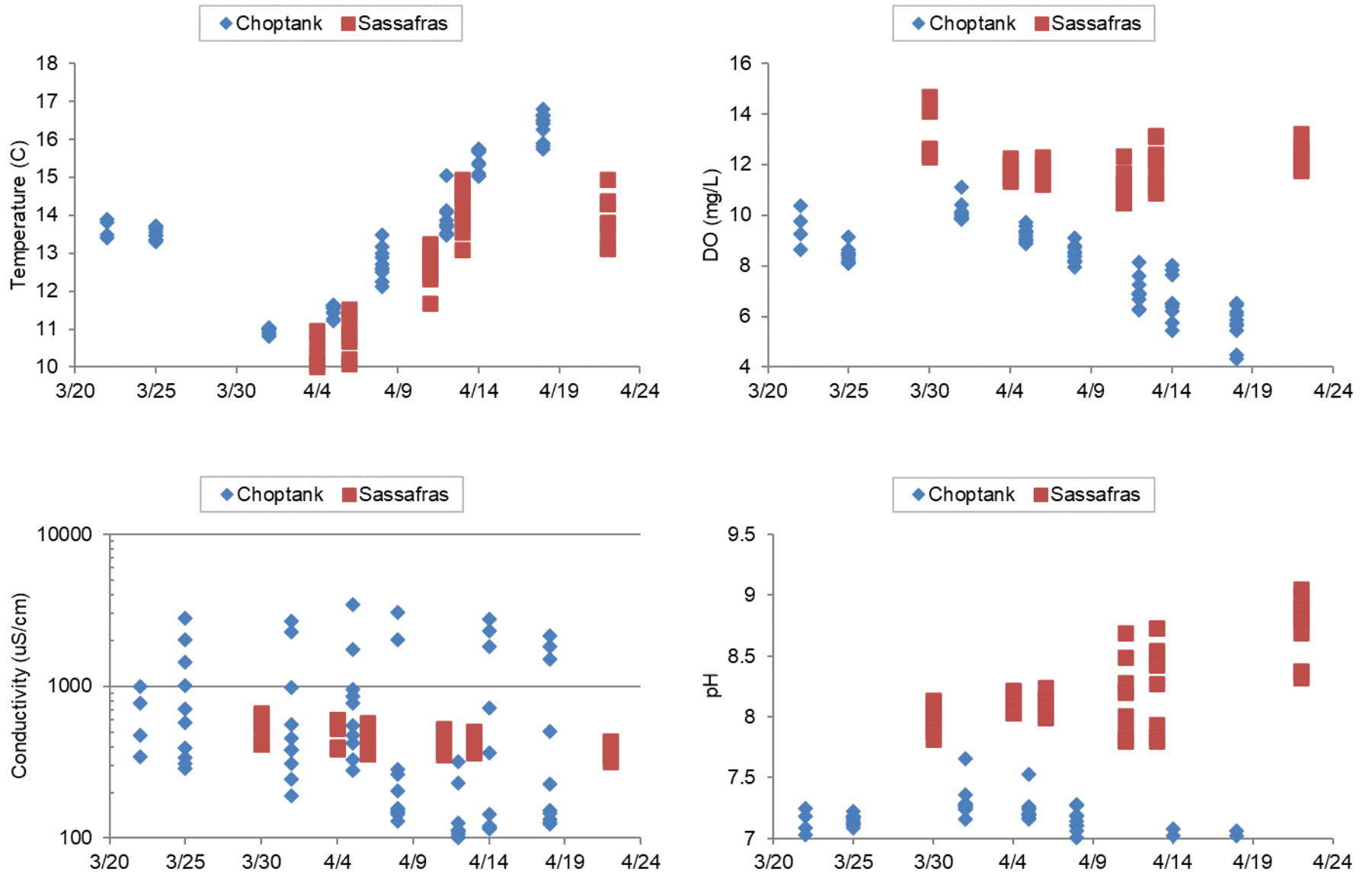


Figure 2-8. Proportion of plankton tows with Yellow Perch larvae (*Lp*) plotted against development (C/ha) with Department of Planning land use designations and salinity class indicated by symbols. Squares and a “0” behind land use indicate tidal-fresh subestuaries (A), while diamonds and a “1” indicate brackish subestuaries (B).

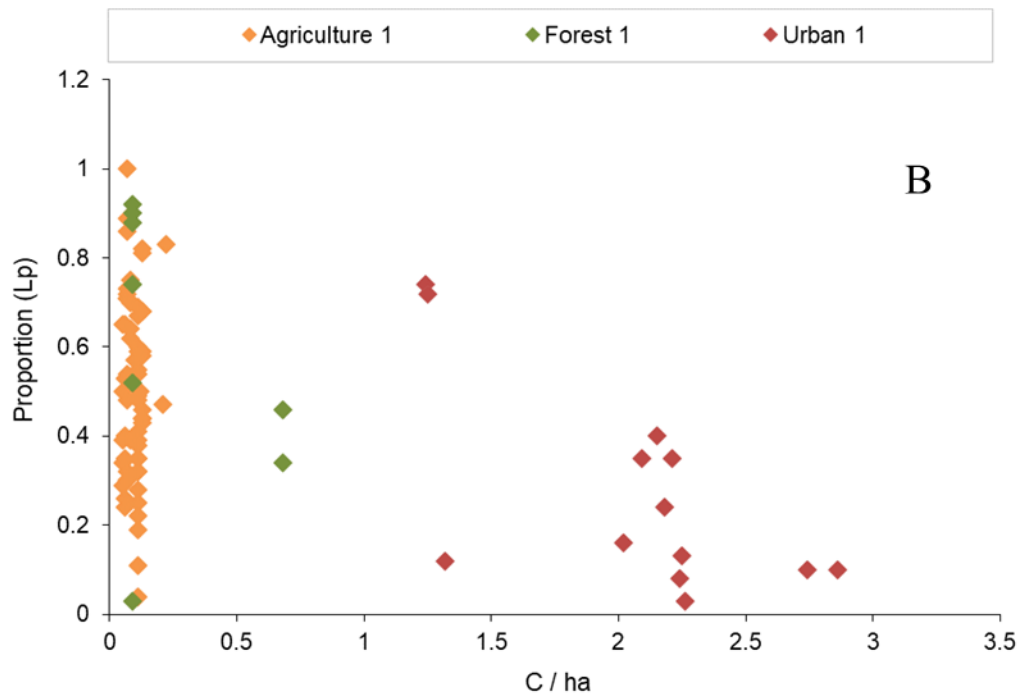
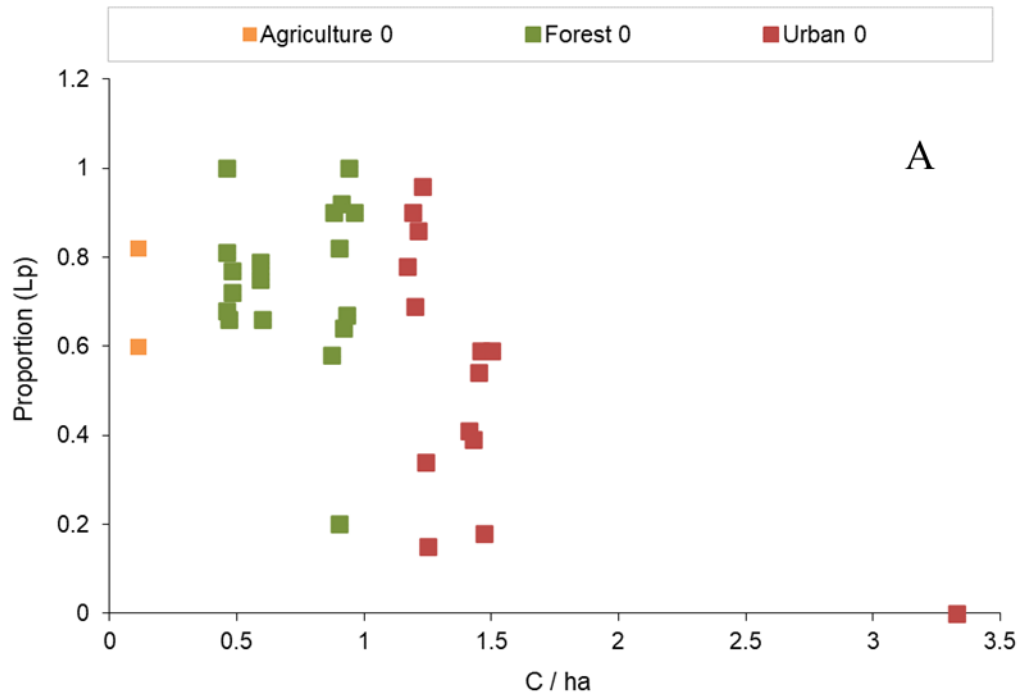


Figure 2-9. Relationship of proportion of tows with Yellow Perch larvae (Lp) and mean temperature (data used for temperature estimates span the same dates used to estimate Lp) in Choptank River from 1980-2022.

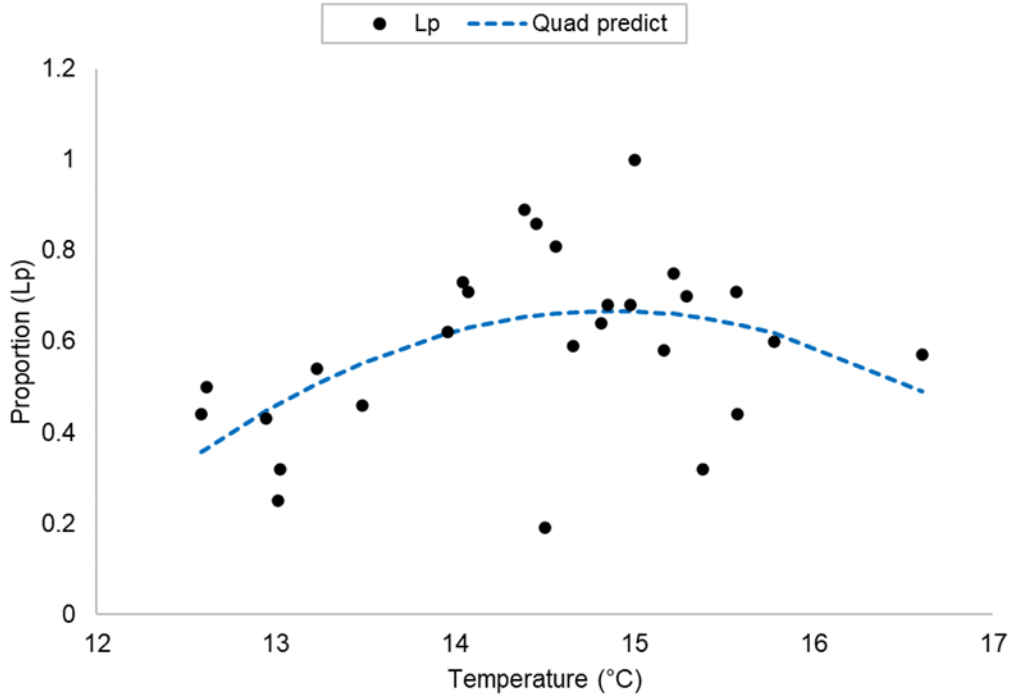


Figure 2-10. Relationship of proportion of tows with larval Yellow Perch (Lp) and (A) all years sampled in Choptank River, 1980-2022 (linear regression = blue, quadratic = red); (B) all years sampled in Nanticoke River, 1963-2019; and (C) fixed station design years (1980-1986) removed in Choptank River.

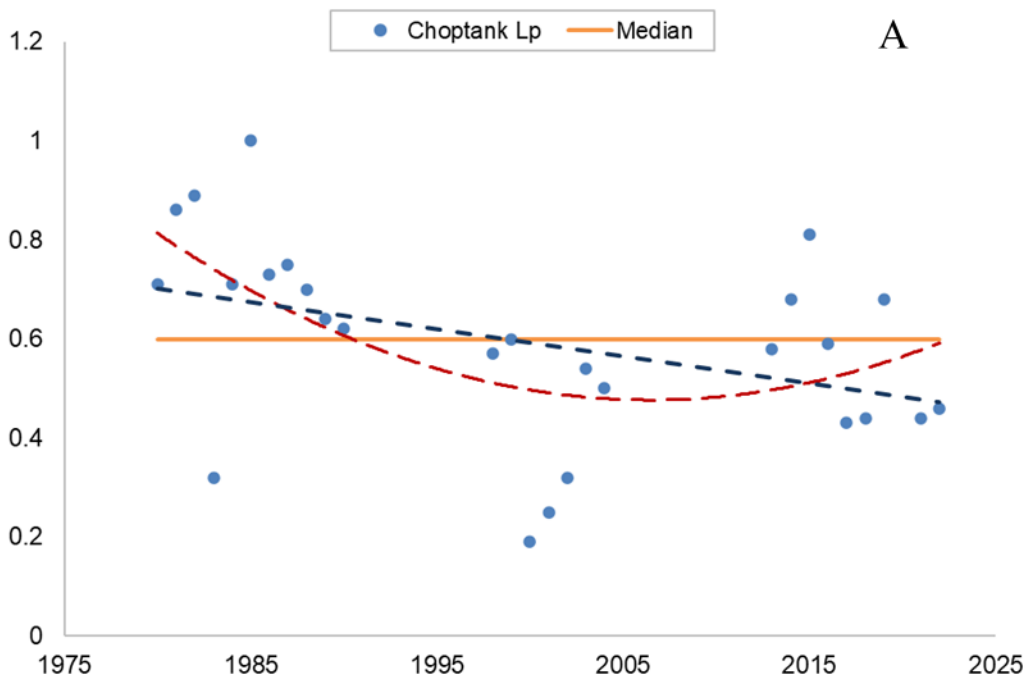
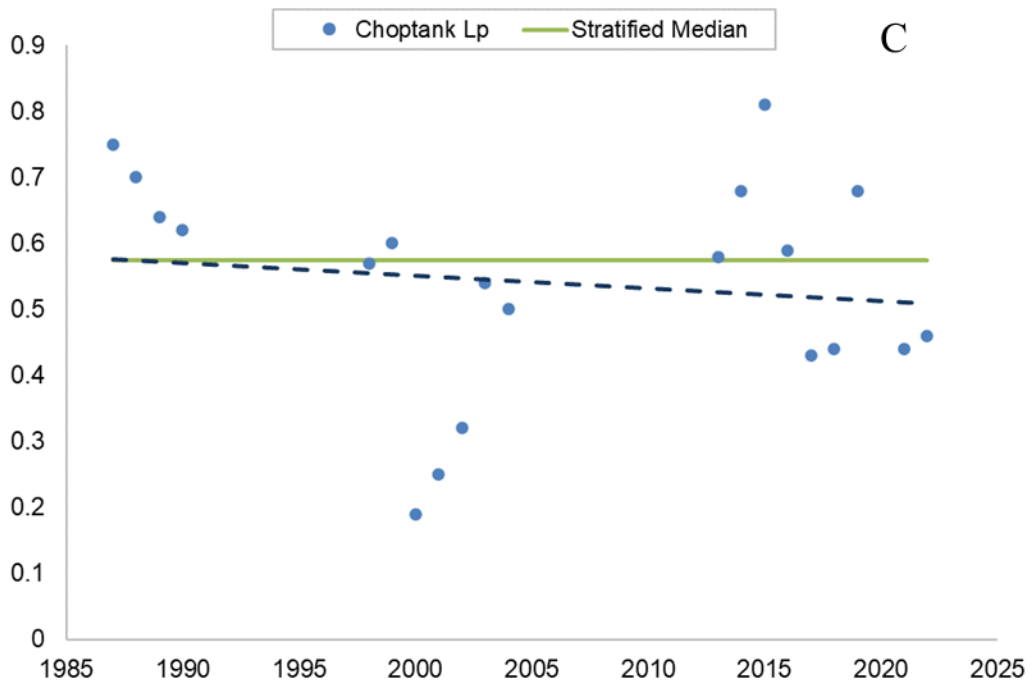
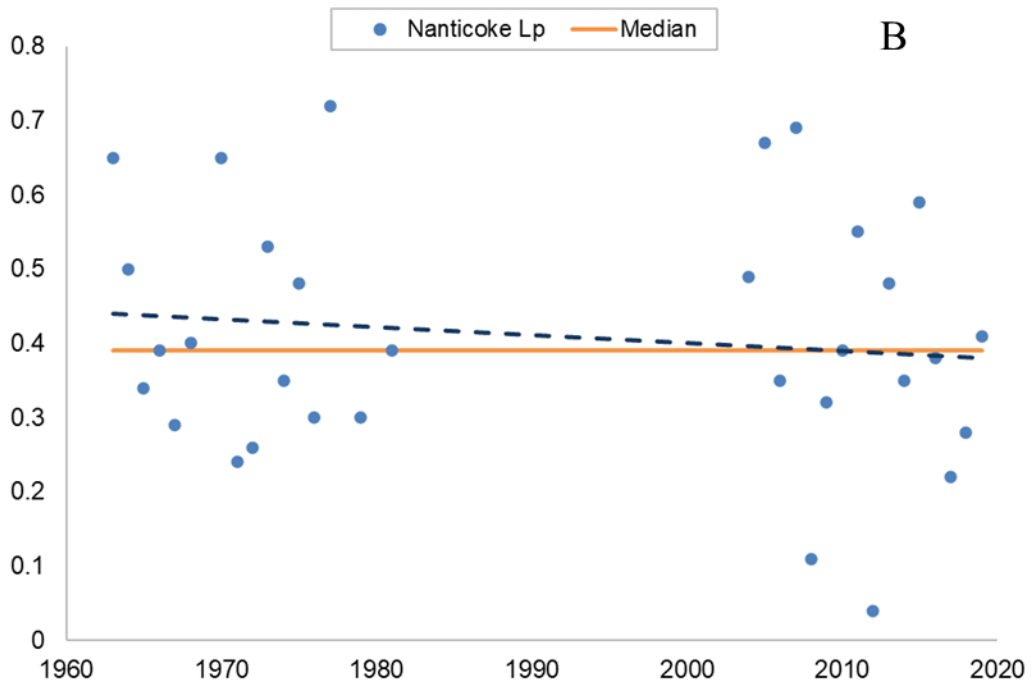


Figure 2-10 cont.



MD – Marine and estuarine finfish ecological and habitat investigations
Project 1: Development of habitat-based reference points for recreationally important
Chesapeake Bay fishes of special concern
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

Introduction

An overfishing declaration and successive poor year-classes of Striped Bass in Maryland spawning areas during 2019-2022 have generated concern in the fisheries management and 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). We have assembled historical data and oriented some of our spring monitoring to respond to these concerns. This report updates efforts begun in the last three annual reports (Uphoff et al. 2020; 2022a; 2022b) to assess spawning and larval habitat. Uphoff et al. (2020;2022b) provide extensive background for this report and Uphoff et al. (2022a) provide detail on the data set assembled for those analyses and this report.

Year-class success of Chesapeake Bay Striped Bass is largely determined within the first three weeks of life in early spring and is a product of egg abundance and highly variable survival through the postlarval stage (Uphoff 1989; 1993; Houde 1996; Maryland Sea Grant 2009; Shideler and Houde 2014; Martino and Houde 2010; Secor et al. 2017). Spawning and larval nursery habitat (both are basically the same) is concentrated in limited fresh to low salinity tidal reaches of 16 Chesapeake Bay tributaries within the Coastal Plain; the estuarine turbidity maximum is particularly important (Hollis et al. 1967; Grant and Olney 1991; Schaaf et al. 1993; North and Houde 2001; 2003; Secor 2007; Uphoff 2008; Maryland Sea Grant 2009; Martino and Houde 2010).

Water temperature and flow conditions are important influences on year-class success of Striped Bass. Temperature may directly impact recruitment through mortality of eggs and larvae due to lethally low or high temperatures and indirectly via its influence on the timing of zooplankton blooms for first-feeding larvae (match-mismatch hypothesis), while flow may be associated with zooplankton dynamics, nursery volume, location of the nursery, advection from the nursery, and water quality and toxicity of contaminants (Hollis et al. 1967; Uphoff 1989; 1992; Secor and Houde 1995; North and Houde 2001; 2003; Maryland Sea Grant 2009; Martino and Houde 2010; Shideler and Houde 2014; Secor et al. 2017; Millette et al. 2020). Positive and negative relationships and associations of Chesapeake Bay tributary flow to Striped Bass early life stage survival and year-class success have been detected (Kernehhan et al. 1981; Uphoff 1989; 1992; Rutherford et al. 1997; Martino and Houde 2010; Millette et al. 2020).

Winter-spring climate variability was considered a prime environmental driver of Striped Bass recruitment (Wood and Austin 2009) and multiple studies have cited cooler and wetter winters and springs as favorable (Maryland Sea Grant 2009; Martino and Houde 2010; Millette et al. 2020). During the past 70 years the Chesapeake Bay has experienced nearly a 2°C rise in mean surface water temperature and long-term warming could alter timing of spawning and survival of eggs and early larvae (Maryland Sea Grant 2009; Peer and Miller 2014). Hinson et al. (2022) determined that warming in Chesapeake Bay was occurring at a more rapid rate during May-October than November-April. The seasonal split during April-May coincides with Striped

Bass spawning and larval development in the Chesapeake Bay region. Modeling of the effect of likely temperature increase scenarios on Striped Bass spawning in the Hudson River from 2010 to the 2090s indicated spawning will occur earlier and be of shorter duration (Nack et al. 2019). The Atlantic States Marine Fisheries Commission (ASMFC) has determined that Atlantic coast Striped Bass spawning stock biomass (SSB) is overfished but is not now experiencing overfishing based on a stock assessment update covering 1982-2021 (ASMFC 2022). Based on updated SSB estimates from a statistical catch at age model, Striped Bass have been overfished since 2013 and target SSB was only achieved briefly in the early 2000s (ASMFC 2022). These SSB estimates contain Delaware River and Hudson River stocks, but are dominated by the Chesapeake Bay stock (NEFSC 2019). High SSB reference points currently in use are not a product of stock-recruitment analysis, but appear to reflect an expectation that higher spawning stock will positively influence recruitment. Management of Striped Bass along the Atlantic Coast strives to achieve high SSB levels through targets and limits that reflect SSB when it was considered recovered (1995) after the period of depletion (Richards and Rago 1999; ASMFC 2003; NEFSC 2019). An egg index independent of this model, based on egg presence-absence in Chesapeake Bay ichthyoplankton surveys during 1957-2019, indicated that stock levels were low enough to impact recruitment during 1982-1988 (Uphoff et al. 2020).

Maryland has measured year-class success (recruitment) of Striped Bass in four major Chesapeake Bay spawning and nursery areas (Head-of-Bay, Potomac River, Nanticoke River, and Choptank River) with a shore zone seine survey of young-of-year juveniles since 1954 (Hollis et al. 1967; Durell and Weedon 2022) and the juvenile index (JI) has proven to be a reliable indicator of recruitment to Atlantic coast fisheries (Schaefer 1972; Goodyear 1985; Richards and Rago 1999; Maryland Sea Grant 2009). Strong year-classes failed to appear during 1971-1992, but a pattern of strong year-classes appearing every few years returned to Maryland's portion of Chesapeake Bay in 1993 (Maryland Sea Grant 2009; Durell and Weedon 2022). Notably, poor year-classes did not occur during 1993-2001. Occasional poor year-classes reappeared during 2002-2018. Year-class success during 1993-2018 was a mix of poor to strong year-classes reminiscent of a high productivity that spanned 1958-1970 (Uphoff et al. 2020). Year-class success has been low during 2019-2022 (Durell and Weedon 2022), narrowly avoiding an ASMFC (2003; 2010) criterion defining poor year-class success.

Uphoff (1993; 1997) used historical ichthyoplankton survey data to develop a Striped Bass egg presence-absence index (Ep or proportion of samples with eggs) of spawning stock status during 1955-1995 for Maryland's spawning areas. An Ep time-series has been maintained through 2022, although it became a low priority in the 2000s as catch-at-age modeling became the primary stock assessment method (Uphoff et al. 2020). An index of relative larval survival, the ratio of the juvenile index to Ep ($RLS = JI / Ep$), was used for retrospective examination of the relative importance of egg and larval habitat on Striped Bass year-class success (Uphoff et al. 2020). Patterns in this ratio provided an indication of changes in egg and larval habitat conditions without specification of the myriad factors (water quality variables, food availability, water temperature, etc.) that determined habitat suitability (Uphoff et al. 2020).

Toxic water quality conditions encountered by Striped Bass larvae were implicated in episodic mortalities in some spawning areas (Choptank River, Nanticoke River, and possibly Potomac River) in the 1980s and 1990s (Uphoff 1989; 1992; Hall et al. 1993; Richards and Rago 1999). During 2014-2019, we collected basic water quality data (temperature, conductivity, dissolved oxygen or DO, and pH) on the spawning grounds of several Striped Bass spawning areas as we investigated the impact of urbanization (Uphoff et al. 2020). During 2021, we began

to shift focus to habitat conditions on the Choptank River as our concern about poor recruitment rose. This river served as a rural control for our investigations of development effect and there was an extensive record of basic water quality conditions and egg-larval mortality during 1980-1991 for comparisons with current conditions (Uphoff 1989; 1992; Uphoff et al. 2020). We added alkalinity to the suite of water quality variables sampled on the Choptank River spawning grounds. Low survival of Striped Bass postlarvae during 1980-1988 in the Choptank River estimated from ichthyoplankton surveys was associated with low pH, alkalinity, and conductivity that could have influenced toxicity of metals (Uphoff 1989; 1992). Water quality in Choptank River ichthyoplankton surveys (Uphoff 1992) was consistent with descriptions for in situ toxicity tests conducted in Choptank and Nanticoke rivers during 1984-1990 (Hall et al. 1993). Acidic conditions, low buffering, and toxic metals (Al, Cu, Zn, Cd, Cr, Pb, and As) were associated with high mortality of Striped Bass larvae in bioassays conducted during 1984-1990 in Choptank and Nanticoke rivers (Hall et al. 1993; Richards and Rago 1999).

C. Hoover mined historical reports and Maryland DNR data sheets to create a spreadsheet with georeferenced data on distribution of anadromous fish eggs and larvae (Striped Bass, White Perch, Yellow Perch, and Alosids) and water quality in Maryland's Striped Bass spawning areas (Uphoff et al. 2022a). Most of this information was focused on Striped Bass. Water quality parameters available varied, but were generally confined to temperature ($^{\circ}\text{C}$), salinity (‰), and tide stage until the early 1980s. During the 1980s and after, dissolved oxygen (DO; mg/L), pH, and conductivity ($\mu\text{S}/\text{cm}$) were monitored more routinely (Uphoff et al. 2022a).

Uphoff et al. (2020) examined long-term (1950s to present), concurrently collected water temperature and egg distribution data from some, but not all spawning areas contained in the data set compiled for Uphoff et al. (2022a). This examination suggested that water temperature (21°C) indicative of the end of spawning and-or poor survival of recently hatched larvae was occurring earlier in recent years. Temperatures approaching and exceeding 21°C fall on a rapidly ascending limb of instantaneous daily mortality rates of larvae that would negate benefit from late spawning (Secor and Houde 1995). There appeared to be a general upward shift in Choptank River spawning area average water temperature between 1986-1991 and 2014-2019 during a standard period (April 1 – May 8) used for comparisons. The 21°C cutoff was sometimes breached later in the 1950s and 1978-1979 than during the 1990s or 2015-2019 in Patuxent River and Chester River, but not in Wicomico River (Uphoff et al. 2020). In this report, we updated temperature patterns for the two spawning areas with the most extensive time-series: the Choptank and Nanticoke rivers.

We examined four spawning milestones that we felt were reasonably straightforward to interpret: date that the first egg was collected, and the dates when 12°C , 16°C , and 20°C were consistently met. Spawning in Chesapeake Bay rivers generally occurs between 12°C and 23°C (Peer and Miller 2014), but temperatures above 21°C are generally not suitable (Uphoff 1993). Secor and Houde (1995) found temperature oscillations had an important influence on egg production. Episodic mortalities of eggs and newly hatched larvae occurred when temperatures fell below 12°C (Uphoff 1989; Rutherford and Houde 1995; Peer and Miller 2014). Olney et al. (1991) reported that for most years, peak egg production in the Pamunkey and Rappahannock rivers occurred with rising temperatures between 15°C and 18°C . Cohort-specific mortality rates of early Striped Bass larvae were strongly temperature dependent, with both early ($<14^{\circ}\text{C}$) and late ($>21^{\circ}\text{C}$) cohorts experiencing higher mortality (Secor and Houde 1995; Peer and Miller 2014). We selected 20°C as an upper temperature boundary since egg presence-absence surveys sometimes cut off sampling just prior to when 21°C was anticipated to occur; 16°C represented

the midpoint of the range and was a temperature where larval cohort survival was expected to be high (Secor and Houde 1995).

Cumulative distributions of egg counts for the Nanticoke and Choptank rivers during 1954-1993 (105,336 and 113,503 eggs, respectively) indicated that 99.3% of eggs were collected by 20 °C and 99.9% by 21 °C (Uphoff et al. 2022b). These cumulative distributions indicated that most egg deposition would occur between 12 and 16 °C (83.2% for Nanticoke River and 89.2% for Choptank River; Uphoff et al. 2022b).

Ichthyoplankton studies and modeling of Striped Bass egg and larval dynamics in Chesapeake Bay spawning areas have linked recruitment success to higher river discharge (Secor and Houde 1995; North and Houde. 2001; 2003; North et al. 2005; Martino and Houde 2010; Secor et al. 2017; Millette et al. 2020). Uphoff et al. (2020) explored long-term (1957-2019) influence of Choptank River March-April flow on \log_e -transformed JIs and a weak relationship was found. Patterning of residuals indicated the relationship was not stable over time with sets of years having stronger or weaker responses to flow. A particularly positive shift in the relationship of flow and the Choptank River JI was reflected by frequent strong year-classes during 1993-2007. The period that followed (2008-2019) coincided with lower flows in April and, while strong year-class have occurred (2011 and 2015), they were less frequent than in 1993-2007 (Uphoff et al. 2020). Uphoff et al. (2022b) expanded this analysis to include all four spawning areas with JIs and explored relationships for both long-term (1957-2020) and the most recent period of high productivity (1993-2020). The long-term data set would be subject to extra variability due to shifts in productivity and low spawning stock; these impacts would be minimized during 1993-2020 (Uphoff et al. 2022b).

Below average flow conditions during 1957-2020 were less conducive to formation of strong year-classes and poor year-classes were more likely (Uphoff et al. 2022b). Above average flows resulted in a higher chance that strong year-classes would be formed and a modest reduction in occurrence of poor year-classes. When all spawning areas were combined during the recent period of high productivity, 1993-2020 (N = 112 area and year combinations), there were 4 strong year-classes when flows were below average and 24 strong year-classes when flow was at or above average. There were 17 poor year-classes when flow was below average and 13 when it was at or above average. When the 1993-2020 was split in half (14 years each), below average flows were less common during the first half (1993-2006) than the second in the Potomac (7 in the first half and 10 in the second), Choptank (4 and 7), and Nanticoke (5 and 8) rivers. There was no change in the Head-of-Bay (5 years of below average flow in each half). We updated the following metrics developed in Uphoff et al. (2020; 2022b) through 2022 in this report: *Ep*, JI, RLS. Temperature, DO, pH, salinity, conductivity, and alkalinity comparisons were updated. We updated the occurrence of spawning milestones and flow patterns in the four major spawning areas.

Methods

Study area - Maryland's portion of Chesapeake Bay contains 12 Striped Bass spawning areas (4 more are in Virginia; Olney et al. 1991), comprising an estimated 57,448 ha (Figure 2.2.1; Hollis et al. 1967). The entire Chesapeake Bay has a surface area of 1,160,000 ha (Malmquist 2009). On an egg production basis, Maryland's spawning areas were estimated to produce approximately 69% of the Chesapeake Bay total (Uphoff 2008). The four largest Maryland spawning areas are sampled for the MD JI: Head-of-Bay (drowned river valley of the Susquehanna River, 27,225 ha), Potomac River (22,162 ha), Nanticoke River

(3,034 ha), and Choptank River (1,734 ha); remaining spawning areas in Maryland are 23-1,011 ha (Hollis et al. 1967). These four largest spawning areas comprise 94% of Maryland's total surface area (Hollis et al. 1967). Two Maryland spawning areas, Patuxent and Potomac rivers, are located on the west side of Chesapeake Bay, the Head-of-Bay is in the center and is furthest north, and remaining spawning areas are on the east side (Figure 2.2.1).

Proportion of ichthyoplankton tows with Striped Bass eggs (Ep) 2022 update – Surveys included in the time-series were considered to have covered most to all of the spawning season and spawning area through multiple sampling events. We confined analysis to spawning areas sampled for the JI to view status and trends (Choptank River, Head-of-Bay, Potomac River, and Nanticoke River; Hollis et al. 1967; Durell and Weedon 2022). Elk River was considered a proxy for the Head-of-Bay if the latter was not sampled. Previously summarized Striped Bass ichthyoplankton surveys (1955-2019; Uphoff 1997; Uphoff et al. 2020; 2022b) were used as a starting basis for the *Ep* time-series. Stratified random sampling designs for Choptank, and Nanticoke rivers used to sample for *Ep* since 1987 were described in Uphoff (1997) and these designs were used during 2022.

Surveys prior to 1994 varied in tow durations, net configuration, and mesh sizes (Uphoff 1993; 1997; Uphoff et al. 2020). Surveys to estimate *Ep* during 1994-2022 were standardized to techniques of the longest running early time-series (Nanticoke River, 1955-1981; Uphoff 1997). These surveys used 2-minute tows made with the current at the surface with a 0.5-m diameter 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, it 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, rose bengal stain was added to aid detection, and it was sorted in the laboratory.

Sample trips during 1994-2022 were usually made twice per week, spaced 2-4 days apart. Sampling was conducted until a 21°C water temperature cutoff criterion was met (Uphoff 1993; 1997; Uphoff et al. 2020) or was very likely to be met before the next scheduled sampling visit based on water temperature and forecast air temperatures. In a few previous years and in 2022, persistent cool temperatures during late spring did not allow water temperatures to rise above 21°C even though egg presence had tapered off and a judgement was made to discontinue sampling. Sites with greater than 2.0‰ salinity usually were randomly replaced within the same sample strata (if possible) by lower salinity sites during sampling to minimize including non-spawning habitat (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). Based on egg counts, 99.5% of eggs in Choptank River (113,313 eggs during 1954-1991) and 94.1% of eggs in Nanticoke River (79,023 eggs during 1954-1985) were collected at salinity less than 2‰ (Uphoff et al. 2022b). Historic field collections were not subject to these criteria and they were applied during analysis when estimating *Ep*.

We restricted *Ep* estimation 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 habitat (Uphoff 1993; 1997). Stations where eggs were not collected located between stations where eggs were present were included in analyses.

The proportion of tows with one egg or more and its 90% confidence interval were estimated 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). Surveys that did not meet this sample size requirement were not included. The proportion of tows with eggs was estimated for each spawning area and year, and for an annual baywide estimate (described below) as:

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

where $N_{present}$ equaled the number of qualifying samples with Striped Bass eggs present and N_{total} equaled the total number of qualifying samples. 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:

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

In cases where cool temperatures persisted and sampling ended before 21°C, 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 combinations of sample dates preceding it 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 these dates would be included.

Uphoff (1997) concluded that Ep in one or more spawning areas could represent baywide spawning stock status since consistent differences in tow times, net diameters, and spawning areas were not detected (Uphoff 1997). We pooled available annual data from these spawning areas to estimate baywide Ep using equation 1, its SD using equation 2, and its 90% CI using equation 3. Five Elk River surveys were redundant with Head-of-Bay surveys and were not used to estimate baywide Ep .

Juvenile index 2022 update - We used annual geometric mean catches of Striped Bass juveniles per standard seine haul at permanent stations in Head-of-Bay, and Potomac, Choptank, and Nanticoke rivers (combined) as the juvenile index (JI; Durell and Weedon 2022). Baywide (Maryland's portion of Chesapeake Bay) and spawning area specific JI's were available online from the MD DNR *Juvenile Striped Bass Survey* website <https://dnr.maryland.gov/fisheries/pages/striped-bass/juvenile-index.aspx> ; we converted the 95% CI's provided to 90% CI's.

The JI was derived annually from sampling at 22 fixed stations within Maryland's portion of Chesapeake Bay (Durell and Weedon 2022). There were seven stations each in the Potomac River and Head-of-Bay and four each in the Nanticoke and Choptank Rivers. Two seine hauls, a minimum of thirty minutes apart, were taken at each site on each sample round. Sampling occurred during July prior to 1962 (44 samples per year), during July and August during 1962-1965 (88 samples), and during July, August, and September after 1965 (132 samples; Durell and Weedon 2022).

Relative Larval Survival (RLS) update - We used the JI and baywide Ep to estimate annual relative larval survival (RLS) during 1957-2022 as:

$$(4) RLS = JI / \text{baywide } Ep \text{ (Uphoff et al 2020).}$$

Estimates of the JI concurrent with Ep were available for 1957-2022 (Durell and Weedon 2022). The baywide Ep time-series started in 1955 and continued through 2022; estimates were not available for 1958-1960 and 2020.

Confidence intervals (90%) were developed for RLS ratios using an Excel add-in, @Risk, to simulate distributions reported for numerators and denominators using Latin Hypercube sampling to recreate input distributions (Palisade Corporation 2016). Each annual RLS estimate was simulated 5,000-times. Annual means and SDs of Ep were used for the denominator in simulations. Juvenile indices, based on geometric means, were back-transformed into the mean of \log_e -transformed catches (+1) and its SE was used. Geometric means were recreated for the numerator for each simulation (Uphoff et al. 2020).

The Striped Bass management plan specifies a criterion for recruitment failure as three consecutive years of Baywide juvenile indices lower than 75% of all other values in the dataset during 1957-2009 (lowest quartile; ASMFC 2003; 2010). We used the same series of years to develop criteria for low and high RLS. We adopted the lowest quartile of RLS during 1957-2009 as a criterion for poor egg-larval survival. Conversely, we chose the upper quartile as an indicator of high egg-larval survival; a strong year-class criterion is not suggested in the management plan. The probability of falling below the poor larval survival criterion was estimated by using the RLS mean and SD to estimate its cumulative probability distribution and the proportion below the criterion was an estimate of risk. The probability of meeting or exceeding the high larval survival criterion was estimated by using the RLS mean and SD to estimate its cumulative probability distribution and the proportion above the criterion was an estimate of this probability.

We expressed deviations between the relative status indicated by the JI and RLS by standardizing each variable by their common time-series mean. This deviation was expressed for each year as:

$$(5) (SJI_t - SRLS_t) / SJI_t;$$

where SJI_t is the standardized juvenile index in year t and $SRLS_t$ is standardized RLS in year t . *Water quality update* – Choptank River was sampled during 2022. Measurements of water temperature (°C), pH, dissolved oxygen (mg/L), conductivity (μ S/cm), and salinity (‰) were made at the surface during each site visit with a YSI model 556 water quality multimeter during 2014-2022. The meter was calibrated frequently. The Choptank River is turbulent and did not show signs of stratification during 1983-1991 surveys when surface, mid-depth, and bottom measurements or measurements at 2-m increments were taken (Uphoff 1992), so surface measurements should have been comparable to those at multiple depths (J. Uphoff, personal observation).

During 2021-2022, total alkalinity (mg/L CaCO_3), was measured in Choptank River using a YSI 9500 Photometer. The Photometer was calibrated for use with YSI photometer color standards, and the transmittance test (program 000), just prior to the beginning of the season. The color standards came with a sheet which provided certified transmittance values, and as long as the photometer produced a similar result (within a specific +/- margin of error), it was working properly. Water samples were collected just below the surface (0.5 m) in Nalgene bottles that were triple rinsed on location before the final collection was made. Bottles were kept in a small cooler while sampling was being conducted, and total alkalinity was measured within 24 hours after collection. Bottles were shaken prior to removing a 10 ml sample, which was then added to a round glass test tube for processing. All collections were free of debris and particulates, so “blanks” were made using the same water from each site just prior to the reagent being added. After reading the blank, one total alkalinity tablet (Alkaphot) was crushed and mixed into a sample until all particles had dissolved. Samples were allowed to stand for exactly one minute before remixing and were then read immediately using the Phot 2 program on the

Photometer. The YSI Photometer 9500 has a minimum detection limit of 10 mg/L, a working range of 0-500 mg/L, and a tolerance of ± 7 mg/L at 200 mg/L for the total alkalinity test.

Water quality analyses were split into two categories. The first examined changes in pH, total alkalinity, and conductivity. These variables were associated with toxic conditions encountered by larvae in the 1980s in the Nanticoke and Choptank rivers during the 1980s (Uphoff 1992; Hall et al. 1993). The second looked at long-term changes in water temperature on the spawning grounds of these two rivers.

Water quality surveys were conducted in Choptank River spawning area during 1983-1985, but they focused on fewer fixed stations that did not span the spawning area (Uphoff 1989; 1992). After 1985, sampling spanned the entire spawning area. Four fixed stations were sampled in Choptank River during 1986 and the stratified random design described in Uphoff (1997) was employed afterwards (Uphoff 1992). Choptank River data for 1980-1991 existed in a database in a format that had not been supported for years; documentation for the database was scanty but water quality data was extracted from it.

Summary water quality statistics included mean, median, minimum, maximum, and the interval encompassing 90% of measurements over a standard time period relevant to eggs, prolarvae, and postlarvae (measurements available during April 1-May 8; Uphoff 1989; 1992; Houde et al. 1996) and relative area (salinity ≤ 2.0 ‰). Means and medians would provide some indication of chronic conditions, while maximums and minimums would capture acute conditions. The 90% data interval would provide an indication of how extreme minimums and maximums were. Estimates of pH were converted to H⁺ concentration to estimate the mean and then converted to mean pH.

We examined four spawning milestones in the Choptank River and Nanticoke River time-series that were reasonably straightforward to interpret: date that the first egg was collected, and the dates when 12°C, 16 °C, and 20°C were consistently met. All dates were expressed as days from April 1 (day 0). To be considered consistent, temperatures could not be single, isolated measurements; a date with multiple readings at milestone would be selected. Intervals between sampling visits had to be no more than weekly for a survey to be included. In some cases, sampling from a single site was all that was available (a few years in the Choptank River), but most surveys had multiple sites spanning most or all of the spawning area. Measurements from the upper reaches of the spawning grounds were sometimes rejected since these areas warm quickly before detectable spawning activity. Dates indicating when the first egg was detected or 12°C or 20°C were consistently met had to be preceded by one day without eggs detected or lower temperatures, respectively. These criteria were not met for each milestone in all years, so time-series varied among milestones.

Surveys from the Nanticoke River during 1954-1981, 1985, 1989, 1992-1994, 2004-2019, and 2021 were used (Uphoff et al. 2022a). The Choptank River time-series consisted of 1954, 1957-1962, 1980-1989, 1994, 1997-2004, 2013-2019, and 2021-2022. J. Uphoff carefully examined spreadsheets containing either Nanticoke River or Choptank River time-series and determined the first eligible date for each criterion. These dates were plotted against year to view trends. Choptank River and Nanticoke River data were combined for summaries and plots. These two spawning areas are adjacent to each other in the Coastal Plain. We estimated the median date for a milestone for each year through 2021 and then examined the frequency that dates exceeded or fell below the median prior to 2001 and after 2000.

Flow – We updated the standardized flows developed in Uphoff et al. (2022b) through 2022. Monthly average flow for each year (in cubic feet per second or CFS) were obtained from

the US Geological Survey gauging stations at Marietta, PA (Susquehanna River), for the Head-of-Bay; Little Falls, MD, for the Potomac River; Greensboro, MD, for the Choptank River; and Bridgeville, DE, for the Nanticoke River from the National Water Information System: Web Interface (<https://waterdata.usgs.gov/>). Uphoff et al. (2022b) identified two-month periods that were likely to precede and be concurrent with spawning and egg and early larval development for each spawning area: March-April for the Choptank, Nanticoke, and Potomac rivers, and April-May for Head-of-Bay. Flows were standardized to 1957-2020 means. The update concentrated on flow conditions since 1993, the beginning of the most recent high productivity period.

Results

Proportion of ichthyoplankton tows with Striped Bass eggs (Ep) 2022 update – Sample size was sufficient for estimating *Ep* in the Choptank River (N = 118) during 2022; Nanticoke River was not sampled in 2022. The estimate of *Ep* in Choptank River during 2022 was 0.69 (SD = 0.04; Figure 2.1.2) and this estimate served as the baywide *Ep* estimate (Figure 2.1.3) as well. As baywide *Ep*, the 2022 estimate was within the bounds exhibited since *Ep* recovered in 1989 (*Ep* bounds = 0.57-0.90), although its 90% CI did not overlap some of the higher estimates (Figure 2.1.3). It was clearly separated from the 90% CIs of lower baywide *Ep* estimates during 1982-1988; estimates during this period were reflected by JIs lower than expected given their estimates of relative survival (Uphoff et al. 2020).

Juvenile index 2022 update – The Baywide JI was 1.78 in 2021 (Figure 2.1.4; Durell and Weedon 2022).

Relative larval survival 2022 update – We adopted the lowest quartile of RLS (<2.07) during 1957-2009 as a criterion for poor egg-larval survival and the upper quartile (>6.73) as an indicator of high egg-larval survival. Estimated RLS was 2.58 in 2022 (Figure 2.1.5). The simulated mean was 2.60 and the SD was 0.37. The probability of falling below the poor RLS criterion in 2022 was 0.07 and the probability that survival was above the high RLS criterion was 0.

With the exception of 1982-1988, deviations between standardized RLS and standardized JIs during 1957-2022 fell between -0.21 and 0.23 (hereafter, the normal range; Figure 2.1.6). During 1982-1988, larger negative deviations occurred, -0.38 to -1.12; these large negative deviations were interpreted as an indication of the effect of low spawning stock. The deviation for 2022, -0.12, was within the normal range (Figure 2.1.6).

Water quality update - During 2022, median pH during the April 1-May 8 standard time period in Choptank River was 7.08 and measurements ranged between 6.58 and 7.66 (Table 2.2.1; Figure 2.1.7). This continued the pattern of above neutral median pH measurements since 2014 in Choptank River. Medians during 2014-2021 ranged from 7.05-7.42, minimums ranged between 6.56 and 7.05, and maximums were between 7.50 and 8.10. Measurements of pH during 1986-1991 were generally acidic and exhibited higher annual and interannual variation. Median pH during 1986-1991 ranged from 6.18 to 7.15, minimums ranged from 5.75-6.50, and maximum pH measurements were between 6.46 and 9.15 (Table 2.1.1; Figure 2.1.7).

Standard period Choptank River total alkalinity measurements during 2021-2022 were much higher than measurements during 1986-1991 (Table 2.1.1; Figure 2.1.7). During 2022, median total alkalinity was 70 mg/L and ranged between 60 and 100 mg/L. During 2021, median total alkalinity was 76 mg/L and ranged between 30 and 110 mg/L. During 1986-1991, median total alkalinity varied from 19 to 23 mg/L, and minimums ranged from 7 to 20 mg/L.

Maximum total alkalinity was lower during 1986-1989 (22-32 mg/L) and rose during 1990-1991 (37 and 45 mg/L, respectively); the 5th and 95th percentile of annual measurements during 1986-1991 confirmed the trend of stable low measurements (5th percentile) throughout the period and an increase in higher measurement (95th percentile) in the latter two years (Table 2.1.1; Figure 2.1.7). Alkalinity measurements began earlier in 2021 (March 30) than 2022 (April 20), so 2022 measurements were likely to have missed the period prior to application of lime on agricultural fields. In 2021, alkalinity was 25-45 mg/L during March 30-April 6, 40-80 mg/L during April 9-14, and 45-110 mg/L during April 20-May 6.

We could not discern potential patterns in conductivity summary statistics from Choptank River during the standard period that would suggest differences between 1986-1991 and 2014-2022 (Table 2.1.1). Standard period median, minimum, and maximum conductivity measurements during 2022 were 281, 99, and 3,419 $\mu\text{S}/\text{cm}^2$. The range for median, minimum, and maximum measurements during 1986-2019 were 161-560 $\mu\text{S}/\text{cm}^2$, 93-135 $\mu\text{S}/\text{cm}^2$, and 3,496-4,881 $\mu\text{S}/\text{cm}^2$, respectively (Table 2.1.1).

Water temperature – The first egg was collected on March 22, 2022 (day -10), the earliest date on record. The plot of days from April 1 that the first egg was collected in the Choptank and Nanticoke rivers (combined) indicated that date was later during 1954-1986 (span of years that first egg capture date could be estimated) than 2000-2022 (Figure 2.1.8). Water temperatures were already above 12 °C (13.4-13.8°C) and remained there on March 25 (Figure 2.1.9). Substantial temperatures drops were evident between March 25 and March 31, and April 18 and April 20. Temperatures rebounded after April 20, but did not reach the 21°C cutoff during the survey (Figure 2.1.9).

Criteria were not met for the 12°C milestone in 2022. The plot of when 12 °C was reached in the Choptank and Nanticoke rivers (combined) did not indicate an obvious decline over time; however, the date it was reached in 2021 was the earliest for the time-series (Figure 2.1.10).

The 16 °C milestone was reached on April 16, 2022, three days shy of the time-series median (Figure 2.1.11). The plot of days from April 1 indicated that 16°C was reached in the Choptank and Nanticoke rivers (combined) later during 1954-1999 (span of years that surveys could address this criterion) than 2000-2021 (Figure 2.1.11).

The 21 °C milestone was not reached through May 13, 2022. However, eggs were present only once during surveys on May 11 and 13 and these dates were not included when *Ep* was estimated. The plot of days from April 1 that 20°C was reached in the Choptank and Nanticoke rivers (combined) indicated it was later during 1954-1998 (span of years that surveys could address this criterion) than 2000-2021 (Figure 2.1.12). Only Choptank River estimates of days since April 1 were available for 2000-2021; Nanticoke River sampling ended earlier than when 21°C criteria were met as personnel left for other monitoring projects.

Flow – We updated average annual 2-month flows (cubic feet per second or CFS) estimated for periods immediately before and during spawning for the Head-of-Bay, Potomac River, Choptank River, and Nanticoke River (Table 2.1.2). Standardized flows were above average baseline flow during 1957-2020 in Choptank and Nanticoke rivers during 2021 (1.35 and 1.25, respectively), and below average in Head-of-Bay (0.73) and Potomac River (0.58). It was near average in Head-of Bay (1.04) and Choptank (0.98) rivers during 2022 and below average in Potomac (0.58) and Nanticoke rivers (0.64; Figure 2.1.13).

Discussion

Baywide *Ep* (0.69) estimated for 2022, was not in the top tier of estimates since 1993 (roughly 0.80 or greater), but there was a high chance it was above levels during 1982-1988 when spawning stock was depleted enough to affect year-class success. Four top quartile baywide JIs were present during 1993-2022 when *Ep* was within the top tier (1993, 1996, 2003, and 2011), while 5 were present when *Ep* was at a similar level to 2022 (2000, 2001, 2005, 2015, and 2018). Estimated RLS in 2022 was just above the poor survival criterion; most of the poor RLS estimates were concentrated in 1980-1991. Estimates of RLS near or below the poor survival criterion were absent during 1993-2001, but returned afterward and occurred intermittently through 2019. Four consecutive years of low baywide JIs have occurred since 2019 and presumably 4 years of low RLS have occurred as well; *Ep* (denominator for RLS) was not estimated in 2020 due to Covid restrictions on sampling, but *Ep* is assumed to be in the same mid-range as 2019 and 2021 (0.70 and 0.67, respectively). Four consecutive years of low JIs is worrisome and will impact the fishery.

Measurements of pH and alkalinity in Choptank River between 1986-1991 and 2013-2021 indicated improvement (higher averages) that would have lowered toxicity of metals implicated in poor recruitment in some Striped Bass spawning areas during the 1980s (Uphoff 1989; 1992; Hall et al. 1993; Richards and Rago 1999; Uphoff et al. 2020). Average alkalinity was at least 3-times higher in 2021-2022 than 1986-1991. Low survival of Striped Bass larvae during the 1980s in the Choptank River estimated from ichthyoplankton surveys and in situ bioassays were associated with low pH, alkalinity, and conductivity that could have influenced toxicity of metals (Uphoff 1989; 1992; Hall et al. 1993; Richards and Rago 1999). Increases in pH, alkalinity, and RLS coincided with actions that reduced acidity and deposition of toxic metals in acid rain, increased implementation of conservation agriculture that reduced use of inorganic fertilizers and pesticides (a potential source of metals) and decreased erosion (sediment is a vector for contaminants); alkalinity of freshwater increased in the U.S. as well (Uphoff et al. 2020). While recent measurements of metals are unavailable, it seems unlikely that poor survival of larvae during 2019-2022 could be attributed to a return of toxic water quality conditions implicated in poor recruitment during the 1980s.

Moderate to strong positive correlations among DO and pH may indicate potential for phytoplankton influence on pH (Uphoff et al. 2020). In the rural Choptank River, none of the correlations were strong enough to be of interest ($r > 0.50$) during 1986-1991 ($r = 0.01-0.42$), but correlations of interest were present during 5 of 8 surveys during 2014-2021. Disparities between time periods suggest change in underlying dynamics.

There were considerable differences in total alkalinity measurements in Choptank River during 1986-1991 and 2021-2022. Alkalinity during 1986-1991 was measured by titration and with a photometer during 2021-2022. Measurements during 2021-2022 were well above the minimum tolerance of the photometer and were within the working range and it seems reasonable to conclude the differences were real and unrelated to different methods.

The date that the first egg was present in 2022 in the Choptank River was the earliest on record for the two adjacent spawning areas and the date 12°C was reached in 2021 was also earliest. Sixteen degrees (°C) was reached 3 days earlier in 2022 than during 1954-2020 and 12 days earlier in 2021. The 20°C milestone was not reached during the 2022 survey even though sampling extended for two weeks into a period when eggs were rare (one presence during two surveys).

Means or medians of days between 12°C and 20°C milestones during 2000-2021 were 10 days to 12 days shorter (respectively) than during 1954-1992 in Choptank and Nanticoke rivers (Uphoff et al. 2022b). Changes were not uniform among temperature milestones. Early milestones appeared to be the least affected. While analysis of the average first date that eggs were collected indicated that date had shifted about 3 days earlier between time periods, but earlier attainment of 12°C in 2000-2021 (about 2-3 days) was not fully supported. As the milestones progressed in magnitude, average dates of occurrence progressed between 1954-1992 and 2000-2021 (7 days earlier at 16°C and 10 days for 20°C). The portion of the spawning period when most eggs were collected (days from 12°C to 16°C) has shortened and high temperatures (indicated by days to 20°C) were being reached earlier. In addition to these general changes, 3 years during 2000-2021 (of 9 available) had very short spans between 12°C and 16°C (2 days) and 2021 had the earliest date that 12°C was reached in the entire time-series. During 1954-1992, the transition from 12°C to 16°C took a week or less with 5 of 19 Nanticoke River surveys and 2 of 7 Choptank River surveys; after 2000, 4 of 9 Choptank River surveys exhibited a transition of a week or less. The transition from 16°C to 20°C took a week or less in 2 of 19 Nanticoke River surveys during 1954-1992 and in 1 of 7 Choptank River surveys; after 2000, 3 of 9 Choptank River surveys made this transition in a week or less (Uphoff et al. 2022b). Water temperature and spawning changes were similar to expectations described by MD Sea Grant (2009) and Nack et al. (2019). Mismatches between the occurrence of larvae and environmental conditions favorable for their survival were considered likely under projected warming scenarios (MD Sea Grant 2009). Higher temperatures during spring could have a negative effect on larval survival due to a more rapid spring to summer transition that reduces when temperatures are most favorable for larval survival (MD Sea Grant 2009).

Our temperature milestones generally captured most Striped Bass egg and larval production based on counts in historic datasets (1950s to 1990s). Cumulative catch distributions of Striped Bass eggs increased rapidly between 12°C and 16 °C in the Choptank and Nanticoke rivers, indicating most eggs were collected when these temperatures prevailed (Uphoff et al. 2022b). Eggs do not have an escape response (Bulak 1993) and changes would reflect hatching or death. The larval cumulative catch distribution gained most rapidly between 14°C and 17°C, followed by a lesser, but steady, increase to 20°C. Changes in larval distribution would have been related to growth and its effect on increasing mobility of larvae and changes in catchability with size, as well as mortality (Uphoff et al. 2022b).

Survival of striped bass larvae is highest at 18°C (Secor and Houde, 1995; MD Sea Grant 2009). In the past, average springtime temperatures in Chesapeake Bay typically fell near 18°C for approximately 2 to 3 weeks during April and May before consistently remaining above 20°C at the onset of summer (MD Sea Grant 2009). Warming in Chesapeake Bay now occurs at a more rapid rate and duration of suitable temperatures for larval development became shorter by 10 days on average after 2000.

Water temperature analyses presented here and in Uphoff et al. (2020) have not covered Head-of-Bay and Potomac River Striped Bass spawning areas. Peer and Miller (2014) analyzed catches from Maryland's spring gill net monitoring of adult Striped Bass on these two spawning grounds during 1985-2010 and found that females moved onto Head-of-Bay and Potomac River spawning grounds approximately 3 d earlier for every 1°C increase in spring water temperature. Further analysis of spring gill net data (1985-2020) indicated that timing of a 14°C milestone was about 3-5 days earlier and that the date that cumulative catch of females reached 100% was

8-9 days earlier, but date that 25% of catch was reached had not changed (A Guiliano, MD DNR, personal communication).

Water temperature milestones were conceptually straightforward, but a bit ambiguous in practice at times. Sites in the upper reaches of the spawning areas appear to warm quicker than downstream, but early spawning was typically downstream (J. Uphoff, MD DNR, personal observation). Use of upper sites where early spawning was not likely could have negatively biased dates when 12°C was relevant to spawning dynamics. There were also instances that impacted all three temperature milestones individually when they were reached at multiple stations considered relevant, followed by a sustained decrease and an interval before they were reached again. The initial occurrence at multiple stations was used for the milestone temperature. Sampling interval could have an impact as well. None of the surveys were conducted daily and most were conducted several days a week with a maximum interval of a week for inclusion in analysis. Spawning season temperatures can be volatile and longer intervals are more likely to miss important events than shorter ones.

Spawning area standardized flows appear to have shifted downward after 2011; above average flows have been lower during 2012-2022 than during 1993-2011 while below average flows were similar during the two periods. Above average flows resulted in a higher chance that strong year-classes would be formed and a modest reduction in occurrence of poor year-classes (Uphoff et al. 2022b). When all spawning areas were combined during 1993-2020 (N = 112 area and year combinations), there were 4 strong year-classes when flows were below average and 24 strong year-classes when flow was at or above average. There were 17 poor year-classes when flow was below average and 13 when it was at or above average. When the 1993-2020 high productivity period was split in half (14 years each), below average flows were less common during the first half (1993-2006) than the second in the Potomac (7 in the first half and 10 in the second), Choptank (4 and 7), and Nanticoke (5 and 8) rivers. There was no change in the Head-of-Bay (5 years of below average flow in each half; Uphoff et al. 2022b).

Frequency of below average flow conditions during 1993-2022 increased since 2006 in 3 of the 4 spawning areas (no change in Susquehanna River), increasing odds that a lesser year-class will be formed and decreasing the odds that strong baywide year-class will form (Uphoff et al. 2022b). General timing of spawning season flows associated with JIs were similar (March-April) for Potomac River, Choptank River, and Nanticoke River, and later (April- May) for Susquehanna River. The watersheds of the three rivers with higher frequency of low flows fall roughly along similar latitudes, while the Susquehanna River drains to the north (Uphoff et al. 2022b). Average winter water temperatures were lower in Head-of-Bay than in Choptank River (Millette et al. 2020), indicating these latitude differences could reflect local climate. Flow and year-class patterns detected here also suggested differences between the large fluvial rivers draining three geographic provinces and smaller spawning rivers located on the Coastal Plain (Uphoff et al. 2022b). The Susquehanna and Potomac rivers flow through the Coastal Plain, Piedmont, and Appalachian geographic provinces while Choptank and Nanticoke rivers are adjacent Coastal Plain rivers on the eastern shore of Chesapeake Bay. Strongest correlations among spawning period flows were indicated for rivers draining similar provinces (Uphoff et al. 2022b).

Years of high spring discharge favor anadromous fish recruitment in Chesapeake Bay and may represent episodes of hydrologic transport of accumulated terrestrial carbon (organic matter or OM) from watersheds that fuel zooplankton production and feeding success (McClain et al. 2003; Hoffman et al. 2007; Martino and Houde 2010; Shideler and Houde 2014). Under natural

conditions in York River, Virginia, riparian marshes and forests would provide OM subsidies in high discharge years, while phytoplankton would be the primary source of OM in years of lesser flow (Hoffman et al. 2007). Differences in watershed characteristics of land draining into the Striped Bass spawning areas may influence their sources of OM. Choptank and Nanticoke rivers are largely agricultural watersheds (40-49% of watershed non-water area) with modest forest cover (18-25%) and extensive non-tidal and tidal wetlands (18-19%); wetlands would be an important source of OM (Uphoff et al. 2022b). Potomac and Susquehanna rivers have proportionally less agriculture (21-23%), more forest cover (57-60%) and less wetlands (1-2%; Uphoff et al. 2022b); OM would more likely be derived from upland forest sources. Our investigation of temperature and flow conditions lead to a general conclusion that these two important influences on year-class success have changed. Hypotheses relating these influences to a downturn in year-class success are viable, but require specific investigations as to how. Relating specific changes, mechanisms, or episodes detected within a survey to year-class success requires directed research.

This analysis was constrained to the Choptank and Nanticoke River spawning areas (both watersheds located in the Coastal Plain) because of their long time-series and more current sampling. These areas were sampled more frequently because their size made them tractable for small boats used by DNR surveys that made up the vast majority of available data. None of these surveys were specifically designed to monitor for long-term temperature changes and they represent “targets of opportunity” for investigating effects of climate warming on Striped Bass spawning and year-class success. Head-of-Bay and Potomac River have not had ichthyoplankton surveys that qualified for *Ep* analysis since 1996 (Uphoff 1997; Uphoff et al. 2020). The absence of information on the 20°C milestone from Nanticoke River beyond 1993 was not anticipated and the dynamics of all three milestones since 2000 were based the Choptank River alone. Nanticoke and Choptank rivers were combined to understand pre-2000 dynamics under an assumption that spawning season temperatures were not likely to be different.

Use of juvenile index quartiles to designate poor and strong year-classes was convenient and use of the lower quantile as a poor year-class marker was based on criteria of ASMFC (2003; 2010). Time periods used for quartiles should reflect similar underlying dynamics (spawning stock and environmental forcing), although that may be difficult to determine with confidence, particularly during periods of transition. For Striped Bass in Maryland’s portion of the Chesapeake Bay, RLS and *Ep* can be used to identify periods of productivity (Uphoff et al. 2020). However, quartiles may not align with the needs of the fishery. The fishery has been generally described as driven by strong year-classes (Florence 1980; Rago and Goodyear 1987; Rago 1992; Richards and Rago 1999; Secor 2000; Uphoff et al. 2020), but some of the lesser year-classes within the upper quartile may not meet expectations of the fishery.

Magnitude of an upper quartile JI may not translate directly into fish available to the fishery due to changing natural mortality. Martino and Houde (2012) detected density-dependent mortality of age 0 Striped Bass in Chesapeake Bay, supporting a hypothesis that density dependence in the juvenile stage can contribute significantly to regulation of year-class strength. Tagging models indicated that annual instantaneous natural mortality rates (*M*) of legal sized Striped Bass in Chesapeake Bay increased substantially during the mid-1990s while annual instantaneous fishing mortality rates (*F*) remained low (Kahn and Crecco 2006; Jiang et al. 2007; NEFSC 2013; NEFSC 2019). The rise in *M* in the mid-to-late 1990s was consistent with a compensatory response to high Striped Bass abundance, low forage, and poor condition (Uphoff et al. 2022a).

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Table 2.1.1. Summary of pH, conductivity ($\mu\text{S}/\text{cm}^2$), and total alkalinity (mg/L) during a standard period (April 1 – May 8), 1986-1991 and 2014-2022. Surveys had similar geographic scales.

pH							
Year	Mean	Median	95th%	5th%	Minimum	Maximum	N
1986	7.04	7.15	7.76	6.71	5.75	9.15	628
1987	6.76	6.78	7.07	6.54	6.30	7.45	249
1988	6.93	7.02	8.01	6.53	6.45	8.40	122
1989	6.17	6.18	6.39	6.00	5.78	6.46	139
1990	6.97	7.03	7.19	6.78	6.50	7.34	150
1991	6.74	7.02	7.51	6.13	5.86	8.20	222
2014	7.09	7.19	7.80	6.80	6.70	8.00	96
2015	7.39	7.42	7.83	7.11	7.05	8.07	96
2016	7.22	7.27	7.68	6.92	6.68	7.85	88
2017	7.23	7.27	7.55	7.01	6.87	7.76	100
2018	7.12	7.15	7.68	6.83	6.71	7.86	90
2019	7.18	7.25	7.55	6.92	6.56	8.10	100
2021	7.05	7.07	7.38	6.86	6.83	7.50	100
2022	6.99	7.08	7.28	6.64	6.58	7.66	110
Conductivity							
Year	Mean	Median	95th%	5th%	Minimum	Maximum	N
1986	858	560	2480	126	94	3950	628
1987	893	372	3175	144	132	4410	250
1988	910	363	3686	186	177	4390	122
1989	426	194	1824	132	93	3750	148
1990	650	161	3053	136	129	3660	144
1991	603	217	3092	147	126	4090	212
2014	669	177	3101	118	111	4881	96
2015	673	208	2956	137	126	3934	96
2016	963	416	3538	150	93	4389	88
2017	991	535	3054	149	135	3664	100
2018	619	207	2652	135	122	3770	90
2019	464	166	2185	128	124	3496	100
2021	636	186	2703	133	115	3695	100
2022	720	281	2666	112	99	3419	110

Total alkalinity (mg/L)							
	Mean	Median	95th%	5th%	Minimum	Maximum	Count
1986	22	23	26	15	13	26	155
1987	24	24	31	19	17	32	99
1988	21	22	23	20	20	23	21
1989	20	22	22	13	13	22	42
1990	20	19	30	12	11	37	146
1991	20	20	28	10	7	45	173
2021	67	70	116	43	30	110	80
2022	75.8	75	95.5	62.5	60	100	30

Table 2.1.2. Average annual flow during two-month periods used in correlation analyses with spawning area JIs, 1957-2022. Average = 1957-2020 mean flow used to standardize spawning area flows.

Spawning area:	Head-of-Bay	Potomac	Choptank	Nanticoke
Flow months:	April-May	March-April	March-April	March-April
Year	Average Flow (CFS)			
1957	67,575	18,229	191	153
1958	108,466	29,206	356	337
1959	67,856	10,887	166	93
1960	77,964	24,448	183	152
1961	103,887	32,207	316	234
1962	98,648	35,007	325	208
1963	78,189	27,046	222	168
1964	103,173	24,567	269	210
1965	50,680	22,720	153	108
1966	56,618	12,130	45	67
1967	74,053	22,738	170	89
1968	46,059	16,339	212	139
1969	45,407	7,732	164	108
1970	96,811	25,193	250	155
1971	84,439	16,172	179	154
1972	103,426	26,152	231	153
1973	73,217	26,074	235	137
1974	78,047	16,015	211	112
1975	64,807	22,773	316	194
1976	53,559	11,695	122	94
1977	105,910	23,412	99	65
1978	99,422	29,709	354	202
1979	100,419	28,290	278	211
1980	86,123	27,082	266	174
1981	35,393	10,277	116	90
1982	79,995	21,339	200	127
1983	88,097	36,577	533	223
1984	88,910	41,035	449	245
1985	51,850	12,268	70	55
1986	77,920	18,670	151	87
1987	72,447	30,639	198	144
1988	40,483	9,970	116	93
1989	50,739	15,266	348	213
1990	44,690	9,792	180	120

Table 2.1.2 (continued).

Spawning area:	Head-of-Bay	Potomac	Choptank	Nanticoke
Flow months:	April-May	March-April	March-April	March-April
Year	Average Flow (CFS)			
1991	61,383	21,045	187	110
1992	63,902	18,685	155	109
1993	157,282	60,335	414	235
1994	145,038	47,900	583	354
1995	40,000	8,295	154	92
1996	74,468	26,262	315	164
1997	57,667	21,333	251	177
1998	93,633	38,132	349	250
1999	58,209	15,009	202	136
2000	88,025	16,878	361	182
2001	69,919	18,843	300	182
2002	43,577	9,154	74	70
2003	91,707	37,750	418	241
2004	80,247	26,067	257	133
2005	86,598	24,551	332	176
2006	30,021	7,730	95	89
2007	85,882	27,951	359	183
2008	91,886	20,571	170	81
2009	48,301	10,822	147	97
2010	63,776	30,040	395	285
2011	155,230	39,021	246	119
2012	34,200	12,898	151	69
2013	48,655	16,987	212	137
2014	69,046	18,500	290	171
2015	70,654	21,031	329	171
2016	38,148	10,093	147	114
2017	75,359	12,015	200	95
2018	67,873	17,559	265	108
2019	71,674	26,581	285	156
2020	62,062	12,719	200	77
2021	45,554	18,728	328	189
2022	65,085	12,916	237	97
1957-2020 Average	62,616	22,128	242	143

Figure 2.2.1. Location of Striped Bass spawning and larval nursery habitat in MD's portion of Chesapeake Bay based on average salinity less than 2 ppt. These areas encompass spawning areas described in (Hollis 1967), but do not exactly duplicate them.

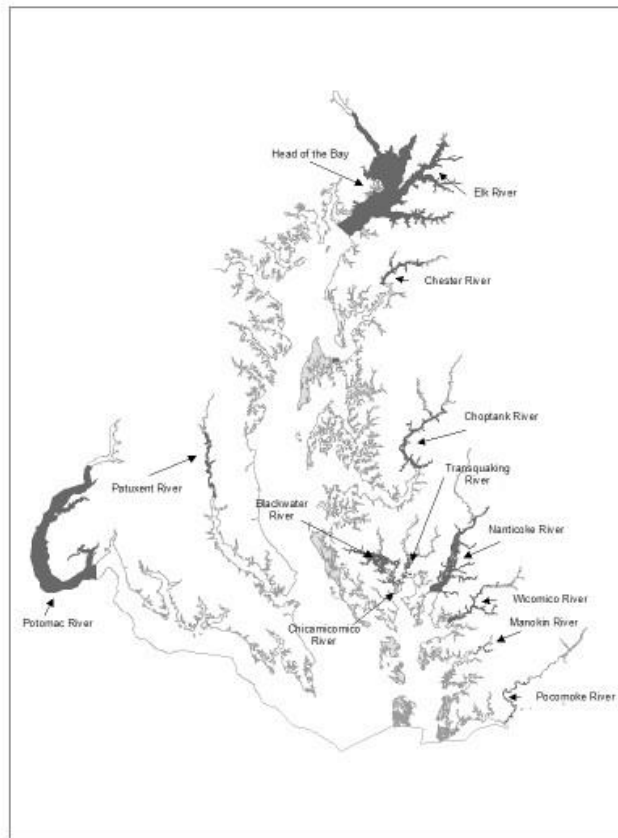


Figure 2.1.2. Spawning area specific proportion of tows with Striped Bass eggs (*Ep*) estimated from surveys in juvenile index rivers conducted during 1955 -2022. Elk River represents a portion of the Head-of-Bay.

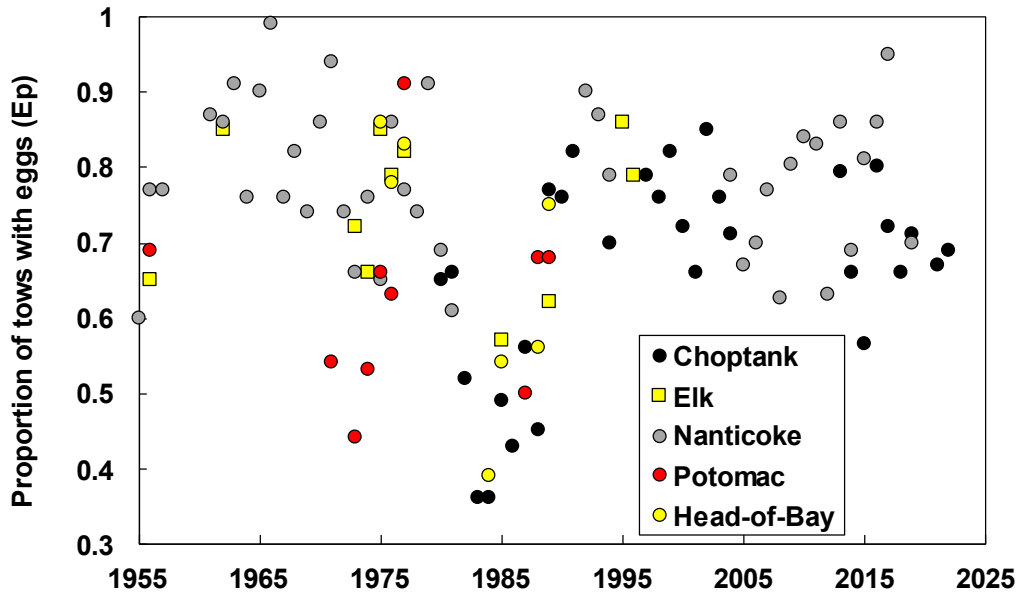


Figure 2.1.3. Baywide (Maryland's spawning areas) proportion of tows with Striped Bass eggs (*Ep*; diamond) and its 90% CI (vertical line) estimated from surveys in juvenile index rivers conducted during 1955 -2022. Baywide estimate pools available data from spawning surveys conducted in four areas surveyed for the juvenile index: Head-of-Bay, Potomac River, Nanticoke River, and Choptank River.

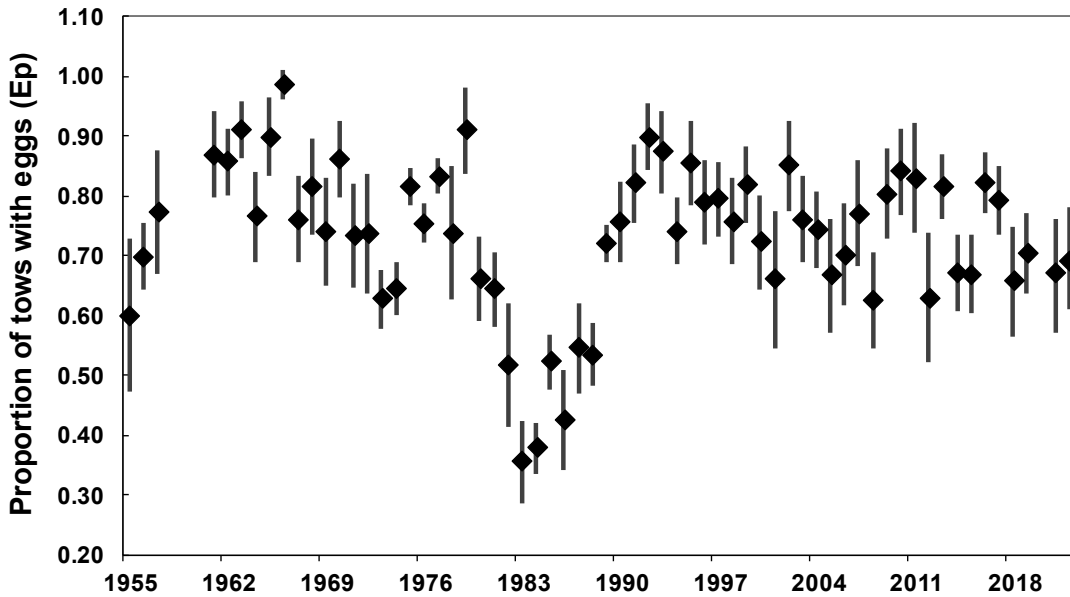


Figure 2.1.4. Baywide Striped Bass juvenile indices (geometric mean catch per standard seine haul; diamonds) and their 90% confidence interval (vertical line) estimated for Maryland's major spawning areas during 1957 -2022 (Durrell and Weedon 2022).

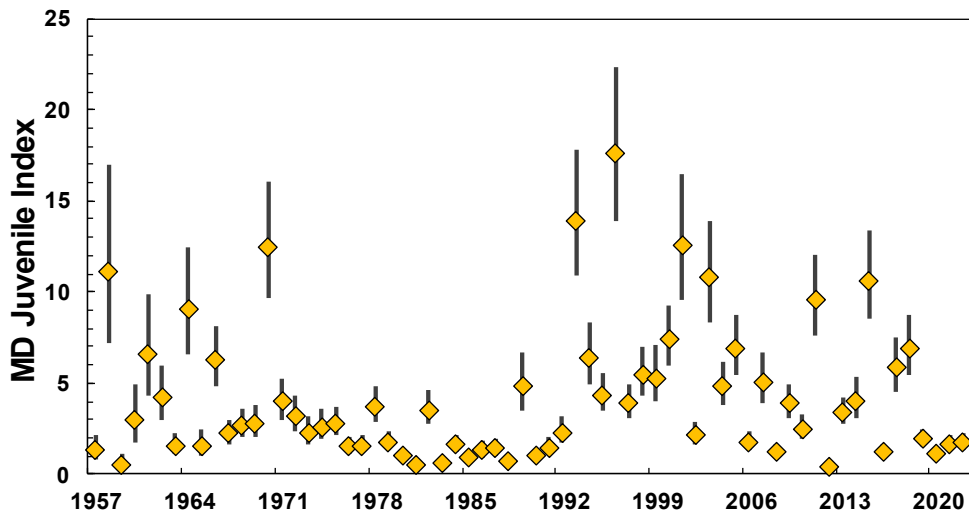


Figure 2.1.5. Relative larval survival(baywide JI / baywide *Ep*) mean and 90% CIs, 1957 -2022. Quartiles (green and grey dashed lines) based on 1957 -2009 (ASMFC JI base years); upper quartile = green dashed line and lowest quartile = grey dashed line.

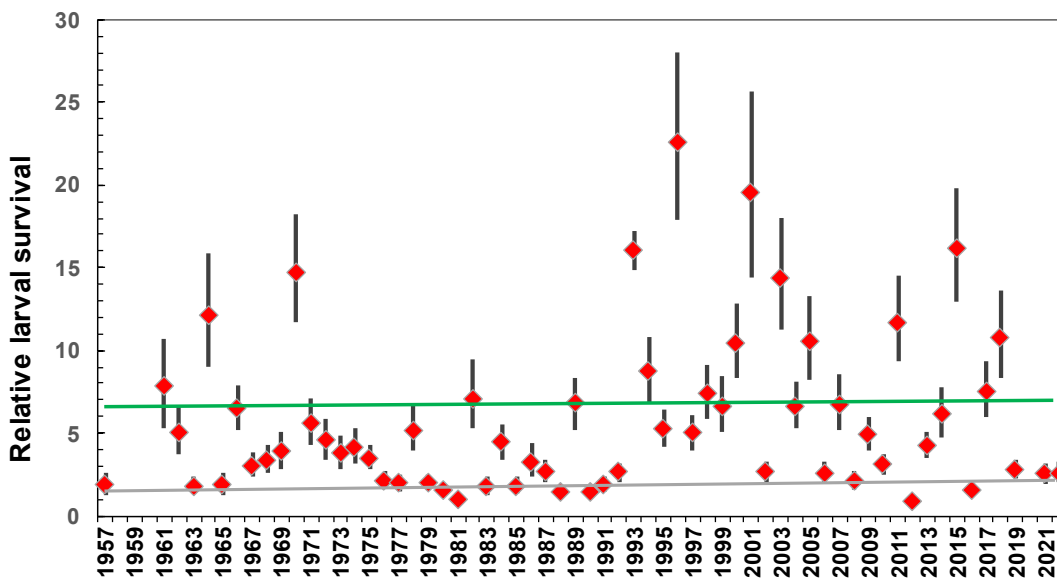


Figure 2.1.6. Difference of standardized juvenile index (Std JI) and standardized relative larval survival (Std RLS) as proportion of standardized JI during 1957 -2022. Large negative deviations indicate overfishing in 1982 -1988. Indices standardized to mean of common years (same scale).

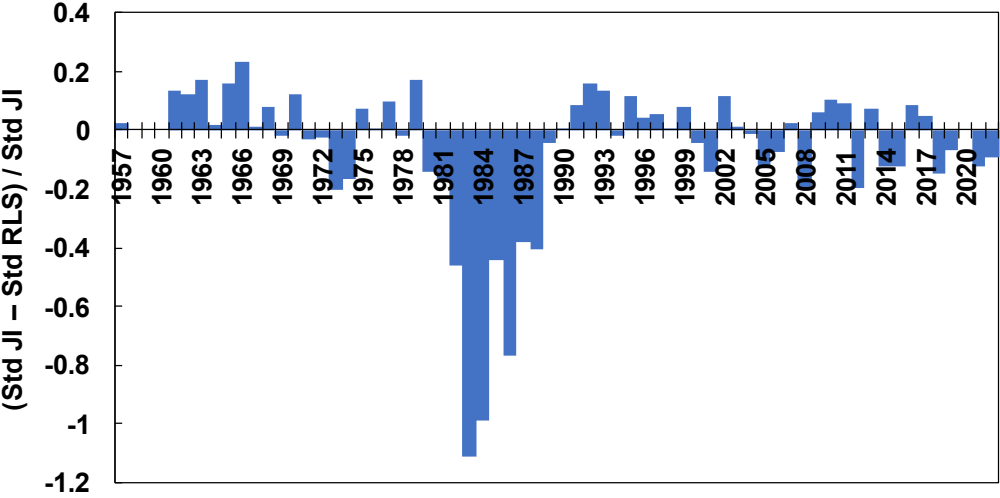


Figure 2.1.7. Choptank pH and alkalinity mean and range during April 1 - May 7, 1986-1991 and 2014-2022; 2022 estimates did not span the full period and began on April 20.

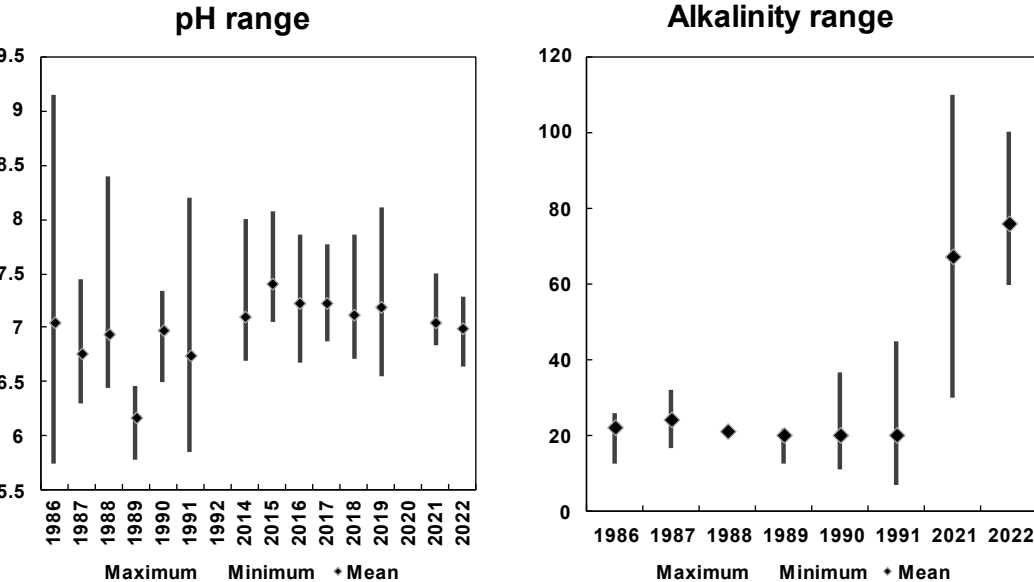


Figure 2.1.8. Days from April 1 (day = 0) that the first egg was collected in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2022. Median = median day for both rivers combined (day 7) during 1954 - 2021.

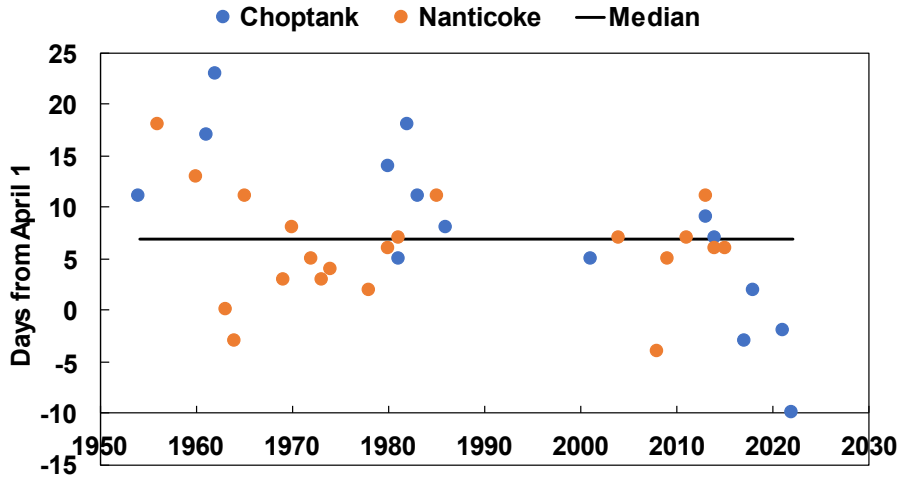


Figure 2.1.9. Water temperature (°C) measurements during the 2022 Choptank River Striped Bass egg presence survey.

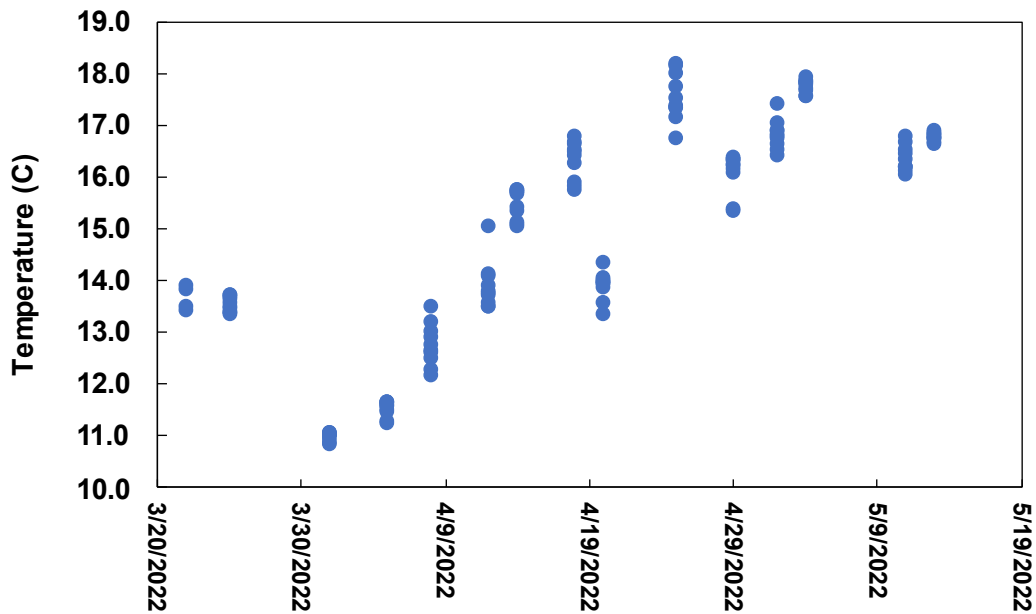


Figure 2.1.10. Days from April 1 (day = 0) that 12°C was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954 -2022. Median = median day for both rivers combined (day 11). This milestone was not present in 2022; temperatures exceeded the milestone on the first day of sampling (March 22, 2022)

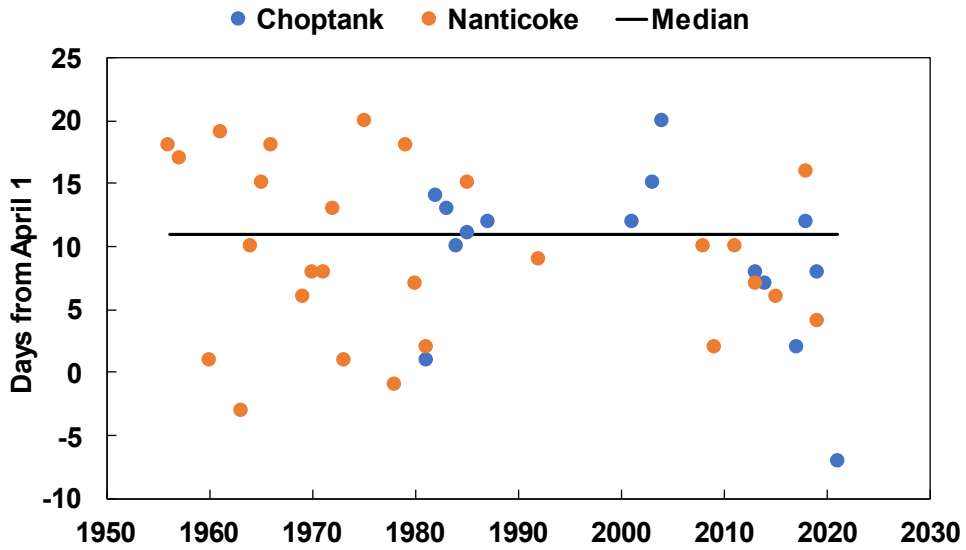


Figure 2.1.11. Days from April 1 (day = 0) that 16°C was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954 - 2021. Median = median day for both rivers combined (day 19) during 1954 -2021.

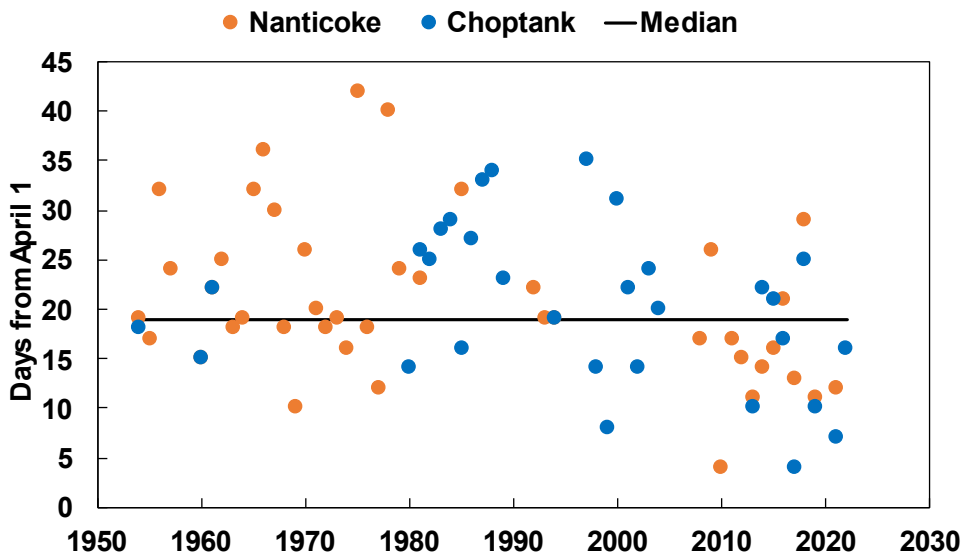


Figure 2.1.12. Days from April 1 (day = 0) that 20°C was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954 -2022. Median = median day for both rivers combined (day 41) during 1954 -2021. This milestone was not reached during 2022 sampling (eggs were present once during May 11 and 13 surveys and sampling was discontinued).

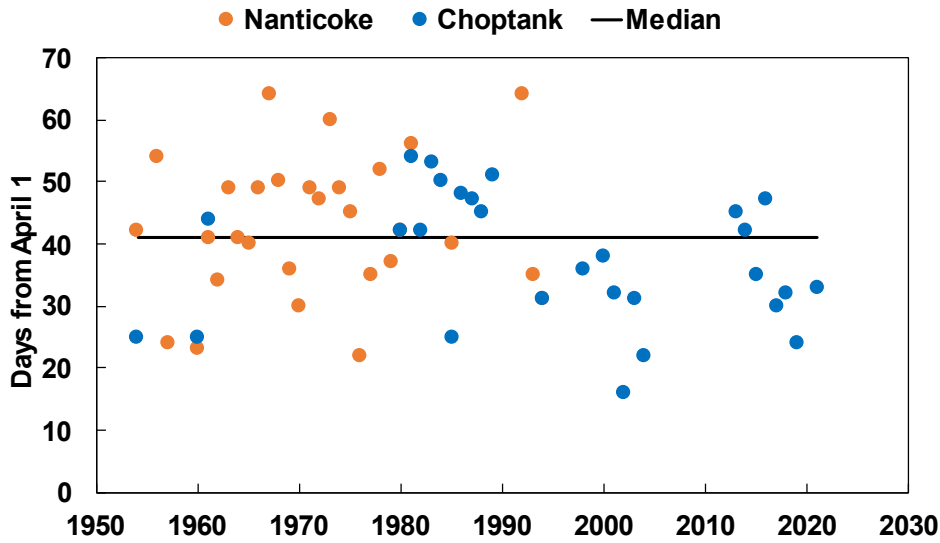
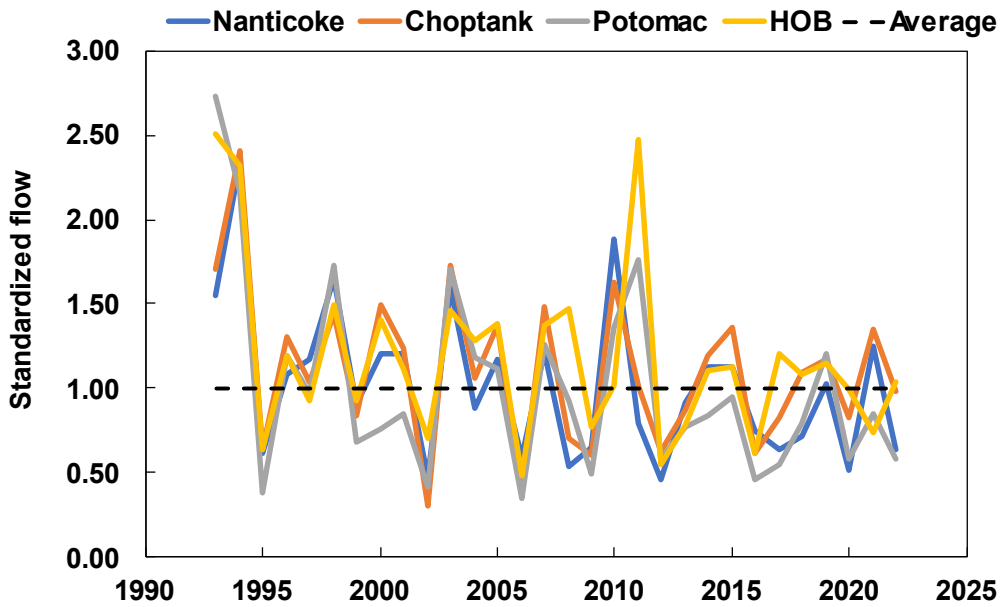


Figure 2.1.13. Two -month average flows for months prior to and including spawning season during 1993 -2022 standardized to their averages for years in common during 1957 -2020.



MD - Marine and estuarine finfish ecological and habitat investigations
Project 1: Development of habitat-based reference points for recreationally important
Chesapeake Bay fishes of special concern
Project 1, Section 3 - Estuarine Fish Community Sampling

Alexis Park, Carrie Hoover, Tyson Johnson, Margaret McGinty, Jim Uphoff

Changes to Project 1 Activities

The choice of subestuaries sampled during summer 2022 was restricted by limited personnel. The subestuaries chosen had resource issues of interest and continued long-term data sets.

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 2020a) and they have ecological, economic, and societal consequences (Szaro et al. 1999).

Water quality and aquatic habitat are altered by agricultural activity and urbanization. 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 and alter fish production. Extended exposure to biological and environmental stressors affects 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. Taylor and Suthers (2021) outlined how urban estuarine fisheries management was defined by unique ecological attributes of urbanized estuaries, the socio-economic objectives of anglers, and bottlenecks to productivity of exploited species.

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. 2020). Uphoff et al. (2011) estimated target and limit impervious surface reference points (ISRPs) for productive juvenile and adult fish habitat in brackish (mesohaline; 5.0 – 18.0 ‰; Oertli, 1964) 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 mesohaline 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 exceed 3.0 mg/L above 10 % IS (suburban threshold; Uphoff et al. 2011a). Although bottom DO concentrations were negatively influenced by development (indicated by IS) in mesohaline subestuaries, Uphoff et al. (2022a) have found adequate concentrations of DO in bottom channel habitat of tidal-fresh (0–0.5 ‰) and oligohaline (0.5–5.0 ‰) 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 2022, we continued to evaluate summer nursery and adult habitat for recreationally important finfish in tidal-fresh, oligohaline, and mesohaline subestuaries of Chesapeake Bay. In this section, 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 2003–2022 (Table 3-1). We evaluated the influence of watershed development on target species presence-absence and abundance, total abundance of finfish, and finfish species richness. We continued to examine Tred Avon River; a tributary of the Choptank River located in Talbot County (Table 3-2; Figure 3-1). We returned to three previously sampled systems, Mattawoman Creek, a tributary of the Potomac River, previously sampled from 1989 to 2016; Northeast River, located at the Head-of-Bay, previously sampled from 2007 to 2017; and South River, located mid-Bay, previously sampled from 2003 to 2005 (Table 3-2; Figure 3-1). We added a more detailed evaluation of species composition, abundance, and richness to our analysis to better understand the possible changes occurring throughout the subestuaries of the Chesapeake Bay sampled in 2022.

Methods

Land Use - 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 (MD DOP) land use and water percentages were used for analyses of data from mesohaline subestuaries sampled during 2003–2022 (Table 3-2). Maryland DOP only has structure estimates available through 2020; 2020 estimates were used to represent 2021 and 2022 in analyses (Table 3-3). 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 Project 1, Sections 1-3**. Land use estimates newer than 2010 have not been released by MD DOP. Chesapeake Conservancy’s Conservation Innovation Center developed high-resolution, 1m • 1m, land cover data for the Chesapeake Bay watershed for 2013/2014 and 2017/2018. Unfortunately, these land cover estimates do not appear to maintain continuity with the MD DOP estimates, so an updated time-series is not possible (M. Topolski, MD DNR, personal communication).

Development targets and limits, and general statistical methods (analytical strategy and equations) are described in **General Spatial and Analytical Methods used in Project 1**,

Sections 1 – 3 as well. Specific spatial and analytical methods for this section of the report are described below.

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 2003–2022 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. We focused this analysis on mesohaline subestuaries because bottom DO does not exhibit a negative response to development in the other salinity categories (Uphoff et al. 2022b).

Sampling Design - Ideally, four evenly spaced haul seine and bottom trawl sample sites were in the upper two-thirds of each subestuary. Lower portions of a subestuary were not sampled to minimize the impact of mainstem water and maximize subestuary watershed influence. We used GPS to record latitude and longitude at the beginning and end of each trawl site, while latitude and longitude at seining sites were taken at the seine starting point on the beach. We focused on using previously sampled historical sites in 2022 at each of the previously sampled subestuaries unless they were no longer accessible. Sites were sampled once every two weeks during July – September, totaling six visits per system during 2022. The number of total samples collected from each system varied based on the number of sites available, SAV interference, weather and 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 (station 01) to downstream (station 04). 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, as well as assisted the crew with seine 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 due to availability, then the crew leader deviated from the sample route to accommodate this need. Bottom trawl sites were generally in the channel, adjacent to haul seine sites. At some sites, beach seines could not be made because of permanent obstructions, dense SAV beds, or lack of beaches. Bottom trawl and beach seine sampling was conducted one right after the other at a site to minimize time of day or tidal influences between samples.

Water Quality Sampling – 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 2022 to determine salinity class of each subestuary. Water quality parameters were recorded at all stations for every individual sampling event in 2022. Temperature (°C), 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 each seine site. Mid-water 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.

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 in mesohaline subestuaries (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 alone. 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 ($V_{\text{threshold}}$) were estimated as:

$$V_{\text{target}} = (N_{\text{target}} / N_{\text{total}}) \cdot 100;$$

and

$$V_{\text{threshold}} = (N_{\text{threshold}} / N_{\text{total}}) \cdot 100;$$

where N_{target} was the number of measurements meeting or falling below 5 mg/L, $N_{\text{threshold}}$ 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 sampled since 2003. This analysis explored multiple hypotheses related to DO conditions. Structures per hectare estimates were considered proxies for nutrient loading and processing due to development in the subestuaries in this analysis (Uphoff et al. 2011). 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.

Trajectories of C/ha since 1950 were plotted for all tributaries and subestuaries sampled. Bottom DO measurements during 2003–2022 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 mesohaline subestuaries) at each station was estimated for tributaries sampled in 2022 and plotted by year sampled. We examined correlations of Secchi depths, SAV coverage, DO, pH, and salinity within all the tributaries and subestuaries. No pH measurements were acquired prior to 2006. See Uphoff et al. (2018) for additional water quality data and past analysis for Mattawoman Creek.

An ANOVA was used to examine differences in mean bottom DO among stations in 2022 subestuaries. 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 2022

subestuary and used to detect how annual station DO compared with the time-series median. Correlation analysis of annual median DO measurements was used among all the subestuaries sampled.

Finfish Community Sampling – 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 and Yellow Perch, and estuarine Bay Anchovy), marine migrants (Atlantic Menhaden and Spot), and tidal-fresh forage (Spottail Shiner, Silvery Minnow, and Gizzard Shad). Except for White Perch, adult sportfish of the target species were rare, but 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 2022).

Striped Bass and Yellow Perch were separated into two age categories, juveniles (JUV; 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 (PSD; Willis et al. 1993).

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 offshore in the same direction as the tide in the same general area as the seine site. 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 3.1 m box trawl made of 12.7 mm stretch-mesh nylon, referred to as the historical trawl, was towed for five minutes in Mattawoman Creek during 1989–2002 (Carmichael et al. 1992). Starting in 2003, the 4.9 m trawl mentioned above was introduced and used to sample Mattawoman Creek. During 2009–2016, both the historical 3.1 m trawl and 4.9 m trawl were used on the same day sampling was conducted in Mattawoman Creek to create a catch-effort time-series directly comparable to monitoring conducted during 1989–2002 (Carmichael et al. 1992). The net size at the start of a sampling day in Mattawoman Creek alternated between visits. Geometric means of adult White Perch abundance and their 95% confidence intervals were estimated for the 3.1 m and 4.9 m trawls for samples from Mattawoman Creek. We predicted 3.1 m GM for the years 2003–2008 and 2022 based on a linear regression of 3.1 m and 4.9 m GMs. Additional gear comparisons between the 3.1 m and 4.9 m trawls can be reviewed in Uphoff et al. (2016).

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. Beach seining could not be conducted in Mattawoman Creek during 2022 due to high SAV density.

Bottom trawl sites were generally in the channel, adjacent to haul seine sites. Bottom trawls and beach seines were conducted one right after the other at a site to minimize time of day or tidal influences between samples.

Three basic metrics of finfish 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 and 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). We correlated GMs of trawl by year and amongst each other for all subestuaries using correlation analysis to observe the relationship over time and between subestuaries. In addition, we noted which target species were within the group that comprised 90% of fish collected, grouping the remaining 10% of species into the “other species” category. 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 for all years in our database. A greater range of years (1989–2022) was available for beach seine samples than the 4.9 m bottom trawl (2003–2022) due to a change from the 3.1 m trawl used during 1989–2002 (Carmichael et al. 1992). We set a minimum number of samples (15 for 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 by salinity class, then ranked their 2003–2022 bottom trawl GMs by year for all species combined to find where the 2022 subestuaries sampled ranked when compared to other subestuaries in their respective salinity classes.

A modified Proportional Stock Density (PSD; Anderson 1980; Anderson and Neumann 1996; Neumann and Allen 2007) was calculated using trawl catch data for White Perch in subestuaries sampled each year (and in 2022) to estimate an annual proportion of the adult population of interest to anglers. Low PSD percentages indicate higher densities of small fish (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:

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

where N is the number of White Perch caught in each subestuary that were quality length or stock length or greater. Quality length (L_{Quality}) refers to the number of White Perch at the minimum length most anglers like to catch (≥ 200 mm TL; Piavis and Webb 2021). 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 2021). We substituted for stock length with the total number of small adults plus harvestable length White Perch to estimate a modified PSD since we did not measure small adults. 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 2021). 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 measured to the nearest millimeter. White Perch greater than or equal to 200 mm TL 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 2021). These data provided an opportunity to evaluate whether a subestuary served as a nursery, adult habitat, or both and to assess the influence of development on the availability of fish for anglers to harvest.

Species composition was calculated separately for bottom trawls and beach seines for all sampling years combined, as well as by year, for subestuaries sampled in 2022.

We used a percent similarity index to evaluate variation in finfish species composition among bottom trawl stations by year for subestuaries sampled in 2022 (Kwak and Peterson 2007). Finfish species abundances at 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).

In addition to our standard fish metrics, we also estimated adult and juvenile White Perch trawl GMs from subestuaries sampled in 2022. White Perch juveniles and adults were consistently abundant and represented the only gamefish that routinely appeared in samples.

Regional Time-Series - Regional comparisons for subestuaries sampled throughout the Chesapeake Bay from 2003-2022 are divided into three regions, Head-of-Bay, mid-Bay, and Potomac River. These regional comparisons focus on time-series of water quality (temperature, bottom dissolved oxygen, Secchi depths, pH, and salinity) and basic metrics of the finfish community. These comparisons can be found in the Appendices: A = Head-of-Bay, B = Mid-Bay, and C = Potomac River.

Results and Discussion

2022 Sampling Locations – Northeast River (Figure 3-1), a tidal-fresh subestuary located at the Head-of-Bay, was sampled previously from 2007 to 2017 (Uphoff et al. 2018; see Appendix A). The Northeast River watershed reached the target level of rural development in 1995 (0.31 C/ha). Development in 2007 was 0.44 structures per hectare increasing slightly to 0.49 C/ha in 2017 and is currently at 0.51 structures per hectare. Historical sampling stations were used in 2022, except seine station 02 which was roped off, preventing access for beach seining (Figure 3-2); therefore, no data was collected for seine station 02 in 2022.

Tred Avon River (0.79 C/ha in 2022; Figure 3-1) reached the target for rural development (C/ha = 0.31) in 1972 and remains under the 10% IS (C/ha = 0.84) threshold for suburban watersheds. Our sampling began in 2006, 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 (Figure 3-2). Talbot County and the town of Easton (located at the upper Tred Avon River) have active programs to mitigate runoff that provided 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: Broad Creek (through 2017 and in 2020) and Harris Creek (through 2016; *see* Uphoff et al. reports 2012–2017 for additional analyses of Choptank subestuaries; *see* Appendix B).

South River (Figure 3-1) sampling took place during 2003–2005, when C/ha ranged from 1.24 to 1.27. In 1966, South River passed the target for rural development (0.31 C/ha) and in 1990, surpassed the threshold for suburban development (0.84 C/ha). South River’s development has increased to 1.41 structures per hectare. In 2022, we returned to sample South River, the highly urbanized mid-Bay subestuary. Historical sampling stations were used in 2022 (Figure 3-2); however, seine station 02 was difficult to sample effectively with a beach seine due to the amount of the benthic algae *Ulva* present, only water quality data is available for this sampling station. For additional analyses of mid-Bay subestuaries, *see* Appendix B.

Mattawoman Creek (Figure 3-1) was “considered to have near to ideal conditions as can be found in northern Chesapeake Bay” in the early 1990s (Carmichael et al. 1992). During 1989–2020, development more than doubled (from C/ha = 0.44 to 1.00). The watershed surpassed the target (0.31 C/ha) for rural development in 1985 and the threshold (0.84 C/ha) for suburban watersheds in 2007. All previous trawl stations were sampled in 2022 (Figure 3-2). We last summarized all the work that had previously been done on Mattawoman Creek in the 2018 F-63-R-9 report (Uphoff et al. 2019; *see* Appendix C) as part of Maryland DNRs’ effort to assist Charles County with its comprehensive growth plan to conserve natural resources of its watershed, including its recreational fisheries (*see* Interagency Mattawoman Ecosystem Management Task Force 2012). Revisiting Mattawoman Creek in 2022, was based on continued development along the headwaters of Mattawoman Creek and a proposal to remove a portion of the Watershed Conservation District (land zoned at low density to conserve water quality) for expansion of the airport at Indianhead and further development.

2022 Water Quality Summary – Table 3-4 provides summary statistics for surface and bottom water quality for each tributary and subestuary sampled in 2022. Salinity in South River and Tred Avon River were within mesohaline bounds in 2022, and Mattawoman Creek and Northeast River were within tidal-fresh bounds. Bottom temperatures ranged from 24°C to 31°C in 2022, with the largest bottom temperature range occurring in Mattawoman Creek. Surface temperatures ranged from 22°C to 32°C in 2022, with the largest surface temperature swing in South River (Table 3-4).

All subestuaries sampled in 2022 had bottom DO readings less than the target level (5.0 mg/L): Mattawoman Creek, 4%; Northeast River, 13%; South River, 100%; and Tred Avon River, 54% (Table 3-5). Four percent of all DO measurements (surface, middle, and bottom) from Mattawoman Creek were below the target; 5%, Northeast River; 48%, South River; and 23%, Tred Avon River. In 2022, only the tidal-fresh systems, Mattawoman Creek and Northeast River, did not have bottom DO estimates below the 3 mg/L threshold; mesohaline systems, South River and Tred Avon River, had 83% and 13% of bottom DO estimates were below 3

mg/L threshold (Table 3-5). Estimated annual mean bottom DO measurements indicated that South River in 2022 had the lowest mean bottom DO (1.76 mg/L; Table 3-6). Tred Avon River annual mean bottom DO was usually above 5.0 mg/L during 2006-2017 and has been below 5.0 after 2018 (Table 3-6). Mattawoman Creek annual mean bottom DO fluctuated without an apparent trend during 2003-2016 and 2022. Northeast River annual mean bottom DO has fluctuated without trend during 2007-2017 and 2022 (Table 3-6).

Dissolved Oxygen Dynamics – Pearson correlation analyses of DO with temperature and C/ha in subestuaries sampled since 2003 indicated that DO responded to temperature and C/ha differently depending on salinity classification (Table 3-7). Mean bottom DO in summer surveys declined below the threshold level in mesohaline subestuaries but did not in oligohaline or tidal-fresh (Figure 3-3). A decline was indicated among mesohaline subestuaries mean bottom DO and C/ha. There were a few years in summer surveys where mean 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 in tidal-fresh and oligohaline subestuaries, but two mesohaline subestuaries (Chester River, 2011 and 2012; Corsica River, 2012) fell below the target conditions (Figure 3-4).

A moderate negative association of surface DO and a strong negative association of bottom DO with corresponding mean water temperatures were detected for oligohaline subestuaries by correlation analyses (Table 3-7), suggesting respiration was a factor in oligohaline subestuaries. Oligohaline subestuaries were shallower than most subestuaries of the other salinity categories, making them more likely to be warmer throughout. Associations of temperature and DO were weak in mesohaline and tidal-fresh subestuaries. A strong 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 strongly 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. Given that multiple comparisons were made, correlations that had a significant *P* might be considered spurious if one rigorously adheres to significance testing (Nakagawa 2004; Anderson et al. 2000; Smith 2020). Sample sizes of mesohaline subestuaries ($N = 90$) were over twice as high as oligohaline ($N = 33$) or tidal-fresh subestuaries ($N = 38$), so the ability to detect significant associations in mesohaline subestuaries was greater (Table 3-7).

Depletion of bottom DO to below target levels in mesohaline subestuaries with suburban-urban watersheds resulted in lost habitat. Uphoff et al. (2011) 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 seem 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. Episodic 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 2015, the oligohaline Middle River subestuary experienced an extensive fish kill attributed to harmful algal blooms (HABs; MDE 2016). The distribution and magnitude of HAB events may increase due to increased nutrient pollution from point and non-point sources and increased water temperatures throughout the Bay (Kemp et al. 2005; MDE 2016).

Land Use Categories, C/ha, and Bottom Dissolved Oxygen in Mesohaline Subestuaries – Correlation of agriculture with C/ha was negative and considered moderate, bordering on strong ($r = -0.72$; $P < 0.0001$); the correlation of urban land cover with C/ha was positive and considered strong ($r = 0.90$; $P < 0.0001$; Table 3-8). Correlation between forest cover with agriculture cover was negative and considered moderate ($r = -0.60$; $P < 0.0001$); urban cover with agriculture was negative and considered strong ($r = -0.72$; $P < 0.0001$). Weak correlations were found for wetland cover with C/ha ($r = -0.21$; $P = 0.06$), wetland cover with agriculture ($r = -0.22$; $P = 0.04$), and wetland cover with forest cover ($r = 0.21$; $P = 0.06$). Remaining pairings of categories were poorly correlated (Table 3-8).

After inspection of scatter plots, agricultural cover was further divided into regional categories (east and west of Chesapeake Bay), reflecting lower percentages of forest cover on the eastern shore, for analyses with DO in mesohaline subestuaries (Figure 3-5). Two western shore sub-regions reflected agricultural coverage: subestuaries located on the western shore of Chesapeake Bay (Magothy, Rhode, Severn, South, and West Rivers) fluctuated between 2.6% to 34.1% agricultural coverage, while lower Potomac River watersheds (Breton Bay, St. Clements, and Wicomico Rivers) ranged from 23.8% to 38.6% agricultural coverage. Eastern shore watersheds were divided into two divisions: Choptank River and Chester River. Choptank River included Broad and Harris Creeks and Miles and Tred Avon Rivers, which ranged from 42.6% to 53.7% agricultural coverage. Chester River included the Chester, Corsica, and Wye Rivers, and Langford Creek, which ranged from 60.4% to 71.6% agricultural coverage (Figure 3-5).

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 38.6%, comprised entirely of western shore subestuaries (Figure 3-5). Median DO measurements beyond this level of agricultural coverage (42.6% – 71.6% agriculture) were from eastern shore subestuaries and the DO trend appeared to be stable or declining (Figure 3-5). Development was predominant at low levels of agriculture (< 20% agricultural coverage). 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 positive and considered strong for the western shore (slope = 0.13; SE = 0.02; $r^2 = 0.74$; $P < 0.0001$; N = 23; Table 3-9) and negative and weak for the eastern shore (slope = -0.03; SE = 0.01; $r^2 = 0.14$; $P = 0.003$; N = 62; Table 3-9). Predictions of median bottom DO for mesohaline western shore subestuaries rose from 0.50 mg/L at 2.6% agricultural coverage to 5.15 mg/L at 38.6%. Predictions of median bottom DO for mesohaline eastern shore subestuaries started at 5.31 mg/L at 42.6% agricultural coverage, increased to 5.51 mg/L at 50.1%, and then decreased to 4.33 mg/L at 71.6% (Figure 3-5). A quadratic regression of median bottom DO versus agricultural coverage (eastern and western regions combined) described the relationship of median bottom

DO with agricultural coverage well ($r^2 = 0.59$, $P < 0.001$; Table 3-10; Figure 3-5). Median bottom DO residuals were inspected and then plotted against agricultural coverage; residuals did not indicate substantial bias. However, residuals suggested that the predications at the highest coverage ($\geq 65\%$) may have been negatively biased.

Water Quality for Mattawoman Creek, Northeast River, South River, and Tred Avon River – Percentages of land in agriculture ranged from 9% to 43%; forest, 22 – 54%; wetlands, <1 – 3%; and urban, 28 – 49 (Table 3-11; Figure 3-1). Water comprised a larger fraction of the area in Tred Avon River (24%) than in the other subestuaries sampled in 2022: South River (14%), Northeast River (9.2%), and Mattawoman Creek (3%; Table 3-11).

The western shore of Maryland has substantially more development than the eastern shore of Maryland which is reflected in tax map estimates of C/ha. South River, a western shore subestuary, was subjected to more development than other subestuaries sampled in 2022 (Figure 3-6). However, eastern shore Tred Avon River watershed was subjected to more development than South River from 1950 to 1964 and greater development than the Potomac River tributary Mattawoman Creek from 1950 to 1992 (Figure 3-6). Time-series for all watersheds started at a rural level of development (C/ha ranged from 0.06 to 0.19) in 1950. South River passed the rural development target (C/ha = 0.31) in 1966; Tred Avon River, in 1972; Mattawoman Creek, in 1982; and Northeast River, in 1995. The development threshold (C/ha = 0.84 or 10% IS) was passed by South River in 1989 and by Mattawoman Creek in 2007; Tred Avon and Northeast Rivers remain under the threshold. Faster growth occurred in Tred Avon River's watershed, reaching 0.79 C/ha in 2022. Development accelerated noticeably in the Tred Avon River watershed during 1996–2011 and then slowed. Tred Avon River's watershed has been approaching the suburban threshold (C/ha = 0.84). South River watershed developed rapidly until the early 2000s, reaching 1.43 C/ha in 2022 (1.51 C/ha or 15% IS; Figure 3-6).

During 2022, 54% of Tred Avon River bottom DO samples were below the target and 13% were below the threshold (Table 3-12; Figure 3-7). During 2006–2022, bottom DO measurements below target level ranged from 13% (2009) to 71% (2019) and bottom DO measurements below threshold level ranged from 0% (2006, 2009, 2012, and 2014) to 17% (2019–2021) in Tred Avon River. All DO measurements below target level ranged from 6% (2009) to 30% (2019) in Tred Avon River.

South River in 2022 had 100% of bottom DO measurements under the target and 83% were below the threshold (Table 3-12; Figure 3-10). During 2003–2005 and 2022, percent of bottom DO measurements below target ranged from 75% (2003) to 100% (2022) and bottom DO measurements below threshold ranged from 25% (2004) to 83% (2022) in the South River. All DO measurements below target level ranged from 29% (2003) to 48% (2022) in South River. The percentage of target level violations increased annually for each year sampled in South River for all DO and bottom DO measurements (Table 3-12).

During 2022, 13% of Northeast River bottom DO samples were below the target, but none were below the threshold (Table 3-12; Figure 3-7). During 2007–2022, percent of bottom DO measurements below target ranged from 0% (2010 and 2016) to 33% (2011) and bottom DO only fell below threshold in two different years, 2011 (13%) and 2015 (4%). Percent of all DO measurements below target ranged from 0% (2016) to 14% (2011) in Northeast River (Table 3-12; Figure 3-7).

During 2022, 4% of Mattawoman Creek bottom DO samples were below the target, and none were below the threshold (Table 3-12; Figure 3-7). During 2003–2016, and 2022, Mattawoman Creek bottom DO measurements below target ranged from 0% (2003–2005, 2009–

2010, and 2013–2016) to 18% (2006) and bottom DO only fell below threshold in 2006 (5%); percent of all DO measurements below target ranged from 0% (2003–2005, 2009–2010, and 2013–2016) to 15% (2006) (Table 3-12; Figure 3-7).

Chesapeake Bay Program’s monitoring station EE2.1 (CBP 2022), located at the mouth of the Choptank River, showed less variation in annual summer (July–September) median bottom DO during 2008-2020 than 1989-2007 and there was little indication of a trend over time (Figure 3-8). Annual summer median bottom DO for 1993 had the highest variation of 7.1. In 2022, the summer median bottom DO was slightly below the time-series median of 5.3 mg/L.

Chesapeake Bay Program’s monitoring station ET1.1 (CBP 2022), located at the mouth of the Northeast River, showed annual summer median bottom DO ranged from 5.9 mg/L to 10.0 mg/L from 1989 to 2022 and varied without trend (Figure 3-8). The greatest summer median bottom DO record was in 1990 and the lowest was in 2021. Station ET1.1 showed greatest variation in annual summer bottom DO measurements in 2021 (5.0 – 10.4 mg/L) and least variation in 2002 (7.9 – 8.2 mg/L).

Chesapeake Bay Program monitoring station CB4.1E (CBP 2022), located south of Kent Point in the mouth of Eastern Bay, the closest monitoring station to Miles River that had bottom DO available (Figure 3-8). Monitoring station EE1.1 had no water quality data available. Annual summer median bottom DO ranged from 0 mg/L (1991) to 1.0 mg/L (2012). The greatest variation in annual summer bottom DO measurements was in 1993 (0 – 6.3 mg/L) and the lowest was in 2001 (0.1 – 0.2 mg/L) and 2005 (0.1 – 0.2 mg/L). The time-series bottom DO median of 0.275 mg/L was greater than 2022 bottom DO median of 0.235 mg/L.

Chesapeake Bay Program monitoring station MAT0016 (CBP 2022), located in Mattawoman Creek’s channel between our stations 3 and 4 (Figure 3-2), recorded annual summer median bottom DO between 5.3 mg/L and 10.25 mg/L from 1989 to 2022 and median bottom DO appeared to decline over the long-term (Figure 3-8). The greatest range at CBP station MAT0016 in summer bottom DO was in 1996 and the lowest was in 2012.

Chesapeake Bay Program’s monitoring station WT8.1 (CBP 2022) was located within the South River. Annual median summer bottom DO was often below the threshold and varied at a low level without trend during 1989-2022 (Figure 3-8). The greatest annual summer median bottom DO was 4.2 mg/L in 2004 and the lowest was 0.1 mg/L in 1998. In 2022, the summer median bottom DO was slightly below the time-series median of 1.3 mg/L.

All subestuaries sampled in 2022 had a lower summer median bottom DO than the time-series median of all years sampled. Tred Avon River had a summer median bottom DO of 4.87 mg/L in 2022, which is below the time-series median of 5.35 mg/L (Figure 3-9); South River, 1.70 mg/L in 2022 and a time-series median of 2.57 mg/L; Northeast River, 6.84 mg/L in 2022 and a time-series median of 6.91 mg/L; and Mattawoman Creek, 6.63 mg/L in 2022 and a time-series median of 7.73 mg/L (Figure 3-9). Tred Avon River median summer bottom DO in 2022 fell below the target level of 5 mg/L, while South River median summer bottom DO fell below the threshold level of 3 mg/L. Time-series median bottom DO measurements for all subestuaries, except South River, remained above the target; South River time-series median bottom DO was below the threshold. In 2022, Tred Avon River has the greatest variation in bottom DO measurements, ranging from 1.33 mg/L to 6.33 mg/L; Northeast River, ranged from 4.25 mg/L to 8.76 mg/L; Mattawoman Creek, ranged from 4.56 mg/L to 8.17 mg/L; and South River, ranged from 0.31 mg/L to 3.96 mg/L (Figure 3-9). The greatest variation for all years sampled for annual summer bottom DO occur in Tred Avon River during 2013 (0.13 – 8.22 mg/L); South

River, in 2003 (0.10 – 6.70 mg/L); Northeast River, in 2008 (3.60 – 13.70 mg/L); and Mattawoman Creek, in 2012 (3.76 – 11.12 mg/L; Figure 3-9).

The mean and SE for bottom DO at all stations in Mattawoman Creek for all years were 7.29 mg/L and 0.08, respectively. Mattawoman Creek means and SE for bottom DO for all years sampled at station 01 were 6.91 mg/L and 0.20; station 02 were 7.18 mg/L and 0.16; station 03 were 7.49 mg/L and 0.15; station 04 were 7.49 mg/L and 0.14, respectively. Mean bottom DO at station 01 was the lowest of the time-series in 2012 (5.77 mg/L), closely followed by 2006 (5.85 mg/L; Figure 3-10). All Mattawoman Creek stations (Figure 3-2) were below the time-series median (7.73 mg/L) in 2022. In 2003, 2013, and 2014, all station means were above the time-series median; all other years sampled, at least one station falls below the time-series median (Figure 3-10).

The mean and SE for bottom DO at all stations in Northeast River for all years were 6.9 mg/L and 0.10, respectively. Northeast River means and SE for bottom DO for all years sampled at station 01 were 6.51 mg/L and 0.22; station 02 were 6.97 mg/L and 0.21; station 03 were 6.95 mg/L and .19; station 04 were 7.17 mg/L and 0.21, respectively. Mean bottom DO at station 01 was the lowest of the time-series in 2011 (5.24 mg/L), closely followed by 2015 (5.59 mg/L; Figure 3-10). Northeast River stations 01, 02, and 03 (Figure 3-2) were all below the time-series median (6.91 mg/L) in 2022; station 04 remained above the time-series median. In 2008 and 2017, all stations mean were above the time-series median; all other years sampled, at least one station falls below the time-series median. Stations 01, 02, and 04 have a negative linear trend over the years sampled, and station 03 has a positive linear trend (Figure 3-10). Northeast River's low percentage of water hectares per area of water and land (9%, respectively) may influence low DO. Furthermore, proximity to Conowingo Dam can influence the amount of nutrient loading that occurs each year when numerous gates are opened to control flooding. In 2011, more than 40 flood gates were opened at Conowingo Dam following Hurricane Irene and Tropical Storm Lee which caused sediment plumes to occur as far down the Bay as the Potomac River; the most recent event occurred in 2018, when 20 flood gates were opened due to increased precipitation throughout the mid-Atlantic region. Numerous studies done by multiple agencies have indicated that effects from major storms have short term effects (Palinkas et al 2019).

The mean and SE for bottom DO at all stations in South River for all years were 2.66 mg/L and 0.19, respectively. South River means and SE for bottom DO for all years sampled at station 01 were 2.50 mg/L and 0.36; station 02 was 1.98 mg/L and 0.37; station 03 was 2.58 mg/L and 0.39; and station 04 was 3.57 mg/L and 0.32, respectively. In 2022, all four stations in South River had the lowest mean bottom DO of the time-series, and only station 04 was above the time-series median (2.57 mg/L; Figure 3-10). Measurements below the 3 mg/L threshold comprised 59% of all bottom measurements during 2003-2005 and 2022. South River's high level of development may influence the increased internal nutrient loading and dampen processing, resulting in low DO within the subestuary.

The mean and SE for bottom DO at all stations in Tred Avon River for all years were 5.10 mg/L and 0.07, respectively. Tred Avon River means and SE for bottom DO for all years sampled at station 01 were 3.59 mg/L and 0.17; station 02 were 5.55 mg/L and 0.10; station 03 were 5.65 mg/L and 0.10; station 04 were 5.65 mg/L and 0.09, respectively. Mean bottom DO at station 01 was the lowest of the time-series in 2020 (2.13 mg/L), closely followed by 2013 (2.23 mg/L; Figure 3-10). Tred Avon River stations 01, 03, and 04 (Figure 3-2) were all below the time-series median (5.35 mg/L) in 2022. Deterioration of DO at the uppermost station at Easton (station 01; Figure 3-10) since 2012 indicated that stormwater from increased watershed

development around Easton was the source of poor water quality rather than runoff from the whole watershed or water intruding from downstream. Station 02 mean bottom DO has declined since 2018 and was at the lowest of the time-series in 2021 (4.8 mg/L); however, during 2022, station 02 was the only station to improve (Figure 3-10). Tred Avon River was the subestuary with the most developed watershed on the eastern shore that we surveyed. It had a low percentage of water hectares per area of water and land (24%, respectively). A low percentage of water hectares may limit intrusion of “good” mainstem water into a subestuary and increase the importance of internal nutrient loading and processing.

Median Secchi depths fluctuated slightly from year to year in subestuaries sampled in 2022, while annual ranges of Secchi depths revealed larger fluctuations within each system (Figure 3-11). Annual median Secchi depths in Tred Avon River ranged from 0.4 – 0.75 m; Northeast River, 0.3 – 0.5 m; Mattawoman Creek, 0.5 – 1.3 m; and South River, 0.8 – 0.9 m (Figure 3-11). South River had the least difference in annual median Secchi depths for all years sampled (0.1 m). Mattawoman Creek had the greatest difference in annual median Secchi depths for all years sampled (0.8 m). Northeast River had the least variation in Secchi depth range over the entire time-series, ranging from 0.2 to 1.1 m; Mattawoman Creek had the greatest variation, ranging from 0.3 to 1.9 m (Figure 3-11).

Coverage estimates (%) of SAV were reviewed for subestuaries sampled in 2022 (Figure 3-12). The Tred Avon River was combined in the Choptank River region estimates for the mouth of the Choptank River, which included subestuaries Broad Creek, Harris Creek, and Tred Avon River (VIMS 2022). The SAV coverage increased substantially from 3.53% in 2013 to 24.4% in 2017 and then declined to 11.7% in 2021; since mapping started, the least SAV coverage was recorded in 1991 at 0.6%. The percentage of SAV coverage has remained above the time-series median of 10.2% since 2014, and displayed a similar trend present in the 1990s (Figure 3-12). The 2018 survey was only partially mapped.

Coverage in South River for fifteen of thirty-three years indicated SAV was absent; greatest SAV coverage was 0.95% in 1998 (Figure 3-12). The coverage of SAV has declined since 2019 (0.31%) to 0.07% in 2020 to 0% in 2021. The time-series median was 0.07% (Figure 3-12).

Coverage estimates of SAV in Northeast River were consistently <1% from 1989 to 2001 (Figure 3-12). Coverage estimates were above the time-series median (1.7%) during 2004–2011. A gradual increase in coverage estimates occurred in 2002 and lasted till 2010 when coverage plummeted. The greatest SAV coverage estimate was in 2009 (5.9%). Since 2011, SAV coverage estimates have fluctuated between 1.3% (2013) and 2.6% (2020; Figure 3-12).

Coverage estimates of SAV in Mattawoman Creek ranged from 1.45% to 46.35% from 1989 to 2021 (Figure 3-12). Coverage estimates were above the time-series median (30.08%) in 2002–2011, 2015–2017, and 2019–2021. Coverage estimates have been on the upswing since 2019. The greatest SAV coverage was in 2008 and the lowest coverage was in 1995. Coverage estimates fluctuated a little over 44% from 2002–2021 (Figure 3-12). Coverage estimates for SAV were not available for 2022 at the time of this report. Correlation between annual SAV coverage estimates and median bottom DO for Mattawoman Creek was negative and strong ($r = -0.62$; $P = 0.02$; $N = 14$), respectively.

Coverage estimates for SAV were not available for 2022 for subestuaries sampled in 2022 at the time of this report.

Measurements of pH were not available prior to 2006 for subestuaries. We did not have meters with pH probes prior to 1996, but did afterwards.

Annual median pH measurements in Tred Avon River ranged from 7.4 to 8.0 for all years sampled; Northeast River, 7.8 – 8.7; and Mattawoman Creek, 7.3 – 8.5 (Figure 3-13). South River median pH was 7.46 and ranged from 7.17 to 7.72 in 2022. The greatest pH measurement in Northeast River was 9.45 in 2017; Mattawoman Creek, 9.5 in 2014; Tred Avon River, 9.03 in 2019; and South River, 7.72 in 2022. The lowest pH measurement in Northeast River was 7.2 in 2008; Mattawoman Creek, 6.8 in 2008; Tred Avon River, 5.69 in 2016; and South River, 7.17 in 2022. The greatest difference in pH range occurred in Tred Avon River in 2016 (2.3; Figure 3-13). A majority of subestuaries had a difference in annual pH range <1.0.

All salinity measurements in 2022 remained in the appropriate salinity classification assigned to the subestuaries sampled (Figure 3-14). Annual median salinity measurements in South River ranged from 7.5 to 11.4‰ for all years sampled; Tred Avon River, 7.5 – 12.8‰; Northeast River, 0.09 – 0.13‰; and Mattawoman Creek, 0.1 – 1.0‰. Median salinity was greatest in Northeast River in 2008 (3.3‰); Tred Avon River, in 2016 (12.78‰); South River, in 2022 (14.5‰); and Mattawoman Creek, 0.9‰ in 2010. Median salinity was least in Northeast River in 2009 (0.09‰); Tred Avon River, in 2019 (5.93‰); South River, in 2005 (1.2‰); and Mattawoman Creek, 0.1‰ in 2003–2006. South River had years where the salinity range fluctuated between oligohaline and mesohaline categories. Salinity in Northeast River and Mattawoman Creek fluctuated between tidal-fresh and oligohaline. Greater amounts of precipitation, such as in 2018 and 2019, lowered salinity conditions (Figure 3-14).

2022 Finfish Community Summary – Geometric mean catch per seine haul ranged from 65 to 210 among the three subestuaries sampled during 2022 (Table 3-13). Tred Avon River was the only station to have the full number of seine samples conducted (N = 24; Figure 3-2). The number of samples varied among subestuaries due to seine station 02 being roped off in the Northeast River (N = 18), and seine station 02 in the South River having too much *Ulva* present to properly seine the station (N = 19; Figure 3-2). Between 22 and 27 species were encountered in seine samples. South River seine catch was greater than the seine catches in Northeast and Tred Avon Rivers, indicating the shore zone represented the majority of suitable habitat (Uphoff et al. 2011a; Table 3-13). Extensive SAV has precluded seining in Mattawoman Creek since 2003.

A plot of species richness in seine samples against C/ha during 1989–2022 did not suggest a strong relationship in tidal-fresh, oligohaline, or mesohaline subestuaries (Figure 3-15). Tidal-fresh subestuary watersheds were represented by a limited range of C/ha (0.11 – 0.69). Oligohaline subestuary watersheds were represented by the widest range of C/ha (0.08 – 3.41) of the three salinity classes. Mesohaline subestuary watersheds were represented by a larger number of surveys (N = 81; C/ha range = 0.07 – 2.68) than tidal-fresh and oligohaline subestuaries (N = 25 and 33, respectively; Figure 3-15).

A total of 16,249 fish representing 38 species were captured by beach seines in 2022 (Table 3-13). Eleven species comprised 90% of the total fish caught in 2022, including (from greatest to least) Atlantic Menhaden, Atlantic Silverside, White Perch (adults), Inland Silverside, Bay Anchovy, Threadfin Shad, Striped Killifish, Gizzard Shad, Blueback Herring, Alewife, and Mummichog. Target species collected were Alewife, Atlantic Menhaden, Bay Anchovy, Blueback Herring, Gizzard Shad, Spottail Shiner, Spot, Striped Bass, and White Perch. Different target species comprised the top 90% of catches in the three subestuaries that could be seined. Atlantic Menhaden, Bay Anchovy, and White Perch were observed in 2 of the 3 subestuaries sampled in the top 90% of catches; Alewife, Blueback Herring, Gizzard Shad, Spot, and Striped Bass were in the top 90% of catches in one subestuary (Table 3-13).

Geometric mean catches per trawl were between 10 and 59 during 2022 (Table 3-14). All subestuaries had 24 samples from four stations in 2022. Northeast River had the greatest GM (59), and South River had the lowest (10; Table 3-14). A plot of trawl GMs against C/ha during 2003–2022 declined with development in mesohaline subestuaries with a possible negative threshold response at C/ha between 0.8 and 1.2 (Figure 3-16); this change reflects the change to consistent low DO conditions in Figure 3-3. A linear regression analysis of mesohaline subestuaries sampled during 2003–2022 indicated a weak negative linear relationship of development (C/ha) on trawl GM catches (slope = -56.93; SE = 17.07; $r^2 = 0.112$; P = 0.001; N = 90, respectively); a linear model may not be a good candidate for describing threshold changes. Trawl GM catches did not exhibit an obvious decline with C/ha in tidal-fresh and oligohaline subestuaries (Figure 3-16).

Number of species captured by trawl in subestuaries sampled during 2022 ranged from 12 to 15 (Table 3-14). A plot of species richness in trawl samples against C/ha (all subestuaries during 2003–2022) did not indicate a relationship of development and number of species for oligohaline subestuaries (species richness ranging from 12 to 26; Figure 3-17). Species richness (ranging from 3 to 23) declined in mesohaline subestuaries as C/ha advanced beyond the threshold (C/ha = 0.86 = 10% IS). Linear regression analysis of mesohaline subestuaries indicated a weak negative relationship of development and trawl species richness (slope = -3.11; SE = 0.78; $r^2 = 0.168$; P = 0.0001; N = 81, respectively); linear models may not be suitable for describing threshold changes. Tidal-fresh species richness (ranging from 10 to 25) increased as development approached the 10% IS threshold for suburban watersheds; once the threshold was breached species richness remained relatively steady (Figure 3-17). Linear regression analysis of tidal-fresh subestuaries did indicate a weak positive relationship of development and trawl species richness (slope = 3.69; SE = 1.44; $r^2 = 0.159$; P = 0.015; N = 37, respectively). It is possible that subestuary location may be exerting an influence since the two tidal-fresh subestuaries with lowest development were in the Head-of-Bay region while the two most developed were in the Potomac River region.

A total of 8,110 fish and 28 fish species were captured by bottom trawl during 2022 (Table 3-14). Five species comprised 90% of the total catch for 2022 (from greatest to least): White Perch (juvenile), White Perch (adult), Spot, Bay Anchovy, Brown Bullhead, and Tessellated Darter; three of the five species were target species; Brown Bullhead and Tessellated Darter were the exception. At least one target species comprising 90% of the catch was present in all subestuaries sampled in 2022. Mattawoman Creek target species consisted of Spottail Shiner and White Perch (juvenile) in 2022; Northeast River, White Perch (adult); South River, Bay Anchovy and Spot; and Tred Avon River, Bay Anchovy, Spot, and White Perch (adult; Table 3-14).

Subestuaries in 2022 had lower species richness than the previously sampled year. Trawl catch GMs in 2022 were lower in subestuaries than the last previously sampled year, except for Tred Avon River which had a greater GM than the last four sampling years (2018–2021). Tred Avon River has a localized upper-subestuary DO issue that has worsened since 2014. South River has a subestuary-wide DO issue; the upper two stations have notably lower DO than the lower two stations. Although sampling has not occurred in South River since 2005, DO was not improved in 2022. Trawl catch GMs appeared to bottom out in Tred Avon and South Rivers following decreases in Bay Anchovies and White Perch (adult and juvenile).

Fish Communities in Mattawoman Creek, Northeast River, South River, and Tred Avon River - Mesohaline Tred Avon River, sampled in 2022, had the 5th lowest bottom trawl GM

ranking out of Tred Avon River's seventeen sampling years, and ranked 60th out of 90 mesohaline subestuaries sampled since 2003 (Table 3-15). South River, sampled in 2022, had the lowest bottom trawl GM rank out of all four years sampled and ranked 81st out of 90 mesohaline subestuaries (Table 3-15). Northeast River, a tidal-fresh system, had its lowest ranking trawl GM estimate in 2022 (59) and ranked 29th out of 38 tidal-fresh systems sampled since 2023 (Table 3-15). Mattawoman Creek, a tidal-fresh system, had its lowest ranking trawl GM estimate in 2022 (28) since 2009 (5), and ranked 32nd out of 38 tidal-fresh systems (Table 3-15). To some extent, these rankings depend on whether the varying time-series contain good year-classes of non-marine target species; longer time-series (5 – 12 years) had higher rankings.

Annual GMs of catches of all species of finfish in 4.9 m bottom trawls for all years sampled in subestuaries sampled in 2022 and their 95% CIs were plotted; Mattawoman Creek, ranged from 5 (2009) to 582 (2014); Northeast River, ranged from 59 (2022) to 306 (2010); South River, ranged from 10 (2022) to 20 (2003); and Tred Avon River, ranged from 13 (2008) to 253 (2010; Figure 3-18). Noticeable declines in estimated GMs appeared after 2013 (Figure 3-18).

Bay Anchovy was the most abundant species in South River and Tred Avon River bottom trawl catches during all years sampled; Bay Anchovy made up greater than 50% of species present during 2006–2022 in Tred Avon River and 30% of species in South River during 2003–2005 and 2022 (Figure 3-19). Four target species, Bay Anchovy, Spot, Striped Bass, and White Perch (adult and juvenile), make up 90% of species composition in trawl catches in South River (all years); there were 16 other species. Tred Avon River had 4 target species that made up 90% of species composition in trawl catches (all years): Bay Anchovy, Spot, Striped Bass, and White Perch (adult and juvenile), and one additional species, Hogchoker. There were 33 other species collected (Figure 3-19). In these comparisons of samples with years combined, the number of other species may be a positive function of how many years were sampled.

White Perch was the most abundant species found in Mattawoman Creek and Northeast River during all years sampled. Northeast River bottom trawl catches for all years (2007–2017 and 2022) were composed of White Perch (adults and juveniles; 83.1%), Bay Anchovy (6.2%), and 32 other species (10.7%; Figure 3-19). Bottom trawl catches for all sampling years, 2003–2017 and 2022, in Mattawoman Creek, were composed of White Perch (adults and juveniles; 53.0%), Spottail Shiner (32.7%), Tessellated Darter (3.6%), and Bay Anchovy (2.0%) and 33 other species (8.7%; Figure 3-19).

Annual species composition in Tred Avon River trawl samples were similar during 2019–2022 (4 species: Bay Anchovy, Hogchoker, Spot, and White Perch); 2018 had the highest species richness (7 species; Figure 3-20). The usually common Bay Anchovy, observed in the top 90% during most sampling years, were missing from the top 90% during 2018 and 2020 and was at noticeably reduced abundance in 2019, 2021, and 2022. Spot presence has increased each year since 2018 in Tred Avon River, while White Perch presence has declined since 2019. Hogchoker presence remains relatively steady since 2018 in Tred Avon River. Within the last five sampling years, only a few other species have made the top 90% of species annually in Tred Avon River, in 2018, American Eel (5%), Green Goby (3%), Oyster Toadfish (14%), and Striped Bass (3%); in 2020, Atlantic Croaker (6%); and in 2021, Summer Flounder (2%); American Eel, Green Goby, Mummichog, Oyster Toadfish, and Summer Flounder were only observed in the 90% of species one year since sampling started in 2006; Atlantic Croaker were only observed two years since 2006 (Figure 3-20).

South River sampled in 2003–2005 and 2022, only had 2 species in common in the top 90% among all sampling years, Bay Anchovy and Spot (Figure 3-20). Striped Bass (17% in 2003 and 5% in 2004) and White Perch (48% in 2003 and 23% in 2004) were present in two of the four years sampled, and Brown Bullhead were present for one year (11% in 2004) of sampling. In 2022, South River trawl catches were comprised mostly of Bay Anchovy (51%), followed by Spot (41%), and the remaining “Other Species” comprised of ten species (Atlantic Croaker, Atlantic Menhaden, Brown Bullhead, Green Goby, Hogchoker, Northern Pipefish, Striped Bass, Summer Flounder, White Perch, and Weakfish; Figure 3-20). There were no significant changes between 2005 and 2022 bottom trawl GMs and trawl composition.

Northeast River bottom trawl catches were largely comprised of White Perch (adults and juveniles) during all sampling years; Bay Anchovy, Brown Bullhead, and Gizzard Shad have been the only other species to be featured in the top 90% of species caught (Figure 3-20). In 2022, Brown Bullhead (24%) were among the top 90% of fish caught; White Perch (73%) declined in abundance slightly compared to 2017 (90%); and other species consisted of 13 species and made up 3% of the bottom trawl catch (Figure 3-20). Invasive Blue Catfish were present in Northeast River during sampling years but not in the top 90% of trawl catch (Figures 3-20); Blue Catfish outnumbered White and Channel Catfish species in Northeast River during 2022.

Annual Mattawoman Creek bottom trawl catches were largely comprised of White Perch (adults and juveniles) during all sampling years; Alewife, Bay Anchovy, Blueback Herring, Bluegill, Channel Catfish, Pumpkinseed, Silvery Minnow, Spot, Spottail Shiner, Striped Bass, and Tessellated Darter have been the only other species to be featured in the top 90% of species caught since 2003 (Figure 3-20). In 2022, Tessellated Darter (11%) were among the top 90% of fish caught; White Perch (79%) increased slightly compared to 2016 (63%); and other species consisted of 10 species (Figure 3-20).

Percent similarity in trawl sample finfish species composition among stations 01–04 in the Tred Avon River was at its lowest in 2019 (7%) but has since been increasing (51% in 2022); percent similarity was at or above 50% during 2007–2017 (Figure 3-21). During 2006 and 2018–2021, the similarity index was below 50%, reflecting possible impacts of heavy rainfall during 2018–2019 and subsequent low salinity on fish community composition.

Percent similarity in South River among stations 01–04 was lowest in 2022 (21%; Figure 3-21). Sampling years, 2003 and 2005, percent similarity among stations was over 50%; during 2004 percent similarity was 25%.

Percent similarity in Northeast River among stations 01–04 was at its lowest in 2009 (22.5%) and greatest in 2012 (70.2%; Figure 3-21). Percent similarity among stations fluctuated, but appeared to have been increasing during 2007–2017; 2012–2017, were above the time-series median. In 2022, percent similarity (30.2%) had declined from 2017, the last year trawl sampling was conducted.

Percent similarity in Mattawoman Creek among stations 01–04 was at its lowest in 2009 (13.7%) and greatest in 2013 (76.7%; Figure 3-21). Similarity indices varied within a high range during 2003–2007 and 2010–2017 and 2022; 2005–2006, 2012–2014, 2016 and 2022, were above the time-series median. They were much lower in 2008–2009 coincidentally with a large drop in the trawl GM that started in 2004. In 2022, percent similarity (70%) did increase slightly from 2016 (66.6%), the last year trawl sampling was conducted (Figure 3-21).

(Uphoff et al. (2018) examined percent similarity in multiple subestuaries and suggested wet years with lower salinity had species composition dissimilar to dry years with higher

salinity. Large drops in similarity reflected large habitat disruptions. The large drop in similarity in Mattawoman Creek during 2007-2009 corresponded with increased total ammonia nitrogen that was believed to indicate possible ammonia toxicity that greatly reduced finfish abundance and diversity (Uphoff et al. 2017). A sharp drop in similarity in Tred Avon River occurred simultaneously with extraordinary rainfall amounts in 2018 and 2019.

Prevalent species in bottom trawl in mesohaline subestuaries shifted annually during 2003-2022 (Figure 3-22). Bay Anchovy and White Perch were predominant during 2003–2010; Spot, along with Bay Anchovy and White Perch, were also predominant in 2005, 2007–2008, and 2010. Bay Anchovy were more abundant than all other species in 2013–2017, but completely disappeared from the top 90% in 2018. White Perch predominated during 2018–2019, declined substantially in 2020–2021 and disappeared from the top 90% in 2022. In the last three years, 2020–2022, Spot has been the dominant species, followed by Bay Anchovy (Figure 3-22).

Adult White Perch GMs for Mattawoman Creek were at or below the time-series median GM (8) in 2008–2015, and 2022; adult White Perch GM was 4 in 2022 (Figure 3-23). The greatest adult White Perch GM in Mattawoman Creek was in 2016 (21) and the lowest was in 2009 (2). The only noticeable spikes in adult White Perch GMs occurred in 2006 (19) and 2016. Juvenile White Perch GMs were at or below the time-series median GM (52) in 2006–2010, 2012, 2016, and 2022; juvenile White Perch GM was 14 in 2022. Juvenile White Perch GMs ranged from 2 (2009) to 256 (2014); the only spike in juvenile White Perch abundance occurred in 2014 (Figure 3-23). Mattawoman Creek is considered more of a nursery area for juvenile White Perch, indicated by the larger abundance of juveniles present and low PSDs.

Northeast River Adult White Perch GMs were at or below the time-series median GM (54) in 2007–2008, 2011, and 2022; adult White Perch GM was 42 in 2022 (Figure 3-23). The greatest adult White Perch GM in Northeast River was in 2012 (119) and the lowest was in 2017 (19); a substantial decline in adult White Perch abundance occurred during this period. The only noticeable increase in adult White Perch GMs occurred in 2011 (43) to 2012. Juvenile White Perch GMs were at or below the time-series median GM (49) in 2008, 2012–2013, 2015–2016, and 2022; juvenile White Perch GM was 4 in 2022. Juvenile White Perch GMs ranged from 4 to 203 (2011); spikes in juvenile White Perch abundance occurred in 2011 and 2014. Abundance of juvenile and adult White Perch were similar during sampling years, 2007–2009 and 2015–2016. In 2022, juvenile White Perch GM declined substantially, while adult White Perch abundance increased slightly (Figure 3-23).

Adult White Perch trawl GMs in Tred Avon River in 2010–2011, 2014–2018, and 2020–2022 were at or fell below the median time-series GM (2); in 2022, adult White Perch GM was 1 (Figure 3-23). The greatest adult White Perch GM in Tred Avon River was in 2012 (12) and the lowest was in 2021 and 2022 (1). Tred Avon River adult White Perch GMs declined during 2006–2010 and 2014–2016; they were highest during 2012-2013. Juvenile White Perch GMs ranged from 0 (2008–2009, 2012–2013, 2016, and 2020–2022) to 13 (2011); juvenile White Perch GM was 0 in 2022. Abundances of juvenile and adult White Perch had similar patterns throughout the time-series (Figure 3-23).

South River adult White Perch GMs in 2003, 2005, and 2022 were at or below the median time-series GM (1); 2004 was 2 (Figure 3-23). Juvenile White Perch GMs in South River ranged from 0 (2004) to 7 (2003); a large decline occurred from 2003 to 2004. Juvenile White Perch GMs in 2004–2005 and 2022 were at or below the time-series median (1; Figure 3-23).

South River juvenile and adult White Perch GMs varied little to poor bottom DO throughout the subestuary that affected availability of habitat.

Modified proportional stock densities (PSDs) revealed White Perch primarily use Mattawoman Creek as nursery habitat. Modified PSDs for Mattawoman Creek fluctuated between 0% and 1.7 % (Table 3-16; Figure 3-24). Mattawoman Creek, sampled in 2022, had a modified PSD estimate of 0%, indicating no change since last sampled in 2016, and remained at the time-series median (0.0%) for all sampling years.

Modified PSDs for Northeast River fluctuated between 0.2% and 1.6% (Table 3-16; Figure 3-24). Earlier years indicated larger numbers of quality size White Perch were present; in the last few years quality size White Perch numbers have declined. Northeast had a modified PSD estimate of 0.3% in 2022, a decline from 2017 (0.7%) and fell below the time-series median (0.6%) for all sampling years (Table 3-16; Figure 3-24).

Tred Avon River modified PSDs have decline since 2018; modified PSD ranged from 4.7% (2012) to 52.1% (2018), an increase after 2016 reflected the size progression of the strong 2011 year-class (juvenile index = 35.2, respectively; Durell and Weedon 2022) into harvestable size (Table 3-16; Figure 3-24). The decline after 2018 may indicate recruitment of two top quartile year-classes (2014 and 2015 juvenile indices = 14.4 and 14.8, respectively) into the stock category. The 2011 year-class followed a stretch of lesser year-classes during the 2000s (Durell and Weedon 2022).

South River had a modified PSD of 82.4% in 2022 that was greater than the modified PSD of 2003 (60.7%; Table 3-16; Figure 3-24). Modified PSD during 2004 was 0% and 15.2% in 2005 (Table 3-16; Figure 3-24).

Geometric mean catches for all finfish in historical 3.1m and 4.9m bottom trawls in Mattawoman Creek were calculated for 1989 to 2016 and 2022 (Figure 3-25). The linear regression of GM catches of all fish combined during 2009-2016 to predict the GM for the 3.1 m trawl from the 4.9 m trawl during sampling years indicated that their trends were strongly and linearly related and we used this relationship to scale the 4.9 m trawl catches to those of the 3.1 m trawl to have a full time-series spanning 1989-2022. The full 3.1 m bottom trawl GM time-series (observations and predictions) suggested abundance of all species became more variable after 2001. Observed and predicted 3.1 m trawl catches during 2002, 2006-2009 and 2022 were lower than observed catches during 1989–2002, catches during 2013 were higher, and remaining catches were similar. A similar approach based on linear regression was used to create extended time-series of GMs for juvenile White Perch (YOY) catches in 3.1 m and 4.9 m bottom trawls in Mattawoman Creek (1989–2016 and 2022; Uphoff et al. 2016). Obvious changes in these catches were not relevant for early and later periods (Figure 3-25).

Seine GMs (all species combined) for subestuaries sampled in 2022 showed similar or lower GMs compared to previous years (Figure 3-26).

Due to high SAV presence in Mattawoman Creek, beach seines were not possible after 2004 and even then the sample size was less than 15 needed for comparisons of species richness. Seven hauls were possible in 2003 and six in 2004. Seine GMs for Mattawoman Creek were 154 (2003) and 274 (2004; Figure 3-26).

Seine GMs for Northeast River indicated minimal swings in relative abundance of all species combined during all sampling years (Figure 3-26). The seine GM was 210 in Northeast River in 2022. The lowest seine GM was 68 in 2011 and the greatest occurred in 2016 (241).

In 2022, South River seine GM (110) was equal to the last sampling year of 2005 (111; Figure 3-26); absence of sampling between 2005 and 2022, makes it hard to assess the trend in

relative abundance of all species combined. Low DO present throughout the subestuary channel limits habitat available to shallow, inshore habitat (Uphoff et al. 2011a).

Tred Avon River has been consistently sampled since 2006 and the minimum, maximum, and median GMs of the time-series are 44, 419, and 111 (Figure 3-26). The greatest GM occurred in 2015 (419) and the lowest GM was in 2017 (44); noticeable peaks in GMs occurred in 2011, 2015, and 2020. Seine GMs have trended downwards since 2020, with seine GM of 69 in 2022 (Figure 3-26).

Mattawoman Creek beach seine finfish composition for 2003–2005 were comprised of eight species: White Perch (adult and juvenile; 27.2%), Banded Killifish (19.2%), Tessellated Darter (15.1%), Largemouth Bass (8%), Bluegill (7.3%), Spottail Shiner (5.4%), Golden Shiner (5.2%), and Alewife (2.9%). Twenty-one species comprised the Other Species (9.6%) category in Mattawoman Creek. Three of the eight species in the top 90% are considered target species, White Perch, Spottail Shiner, and Alewife (Figure 3-27).

Finfish composition for beach seine catches for all sampling years combined in the Northeast River comprised of seven species in the top 90% of seine catches (Figure 3-27). Gizzard Shad (35.3%) was the most abundant species in the Northeast River during all sampling years, followed by White Perch (adult and juvenile; 23.2%), Blueback Herring (16.9%), Alewife (4.4%), Bay Anchovy (4.3%), Atlantic Menhaden (3.2%), and Threadfin Shad (3.1%); Clupeids were abundant in seine samples, but not bottom trawl samples. Six of the seven species in the top 90% were target species; only Threadfin Shad were not a target species. Thirty-four species comprised the other species (9.6%) category in the Northeast River (Figure 3-53); Pumpkinseed and Spottail Shiner comprised 27.6% and 21.1% of the Other Species category, respectively. Other notable species caught in Northeast River seine catches in 2022 were invasive Blue Catfish and Flathead Catfish.

South River's top species during 2003–2005, and 2022 were Atlantic Menhaden (41.5%), Atlantic Silverside (13.5%), Inland Silverside (4.8%), Striped Bass (10%), and White Perch (21.4%); an additional 32 species (8.8%) were collected in South River (Figure 3-27). Three species in the top 90% were target species: Atlantic Menhaden, Striped Bass, and White Perch.

Seven species comprised the top 90% of finfish in beach seines when all years were combined in Tred Avon River (Figure 3-27). Tred Avon River's top species sample by seine during 2006–2022 were Atlantic Menhaden (18.7%), Atlantic Silverside (35.7%), Banded Killifish (2.8%), Bay Anchovy (3.8%), Mummichog (7.8%), Striped Killifish (7.7%), and White Perch (14.9%); an additional 41 other species (8.5%) were collected while seining in Tred Avon River. Only three species in the top 90% were target species: Atlantic Menhaden, Bay Anchovy, and White Perch. Three species in the top 90% of beach seine catches were common between South and Tred Avon Rivers: Atlantic Menhaden, Atlantic Silverside, and White Perch (Figure 3-27).

More species are in the top 90% of catches when seines are used than bottom trawls and species richness is higher in beach seine samples (Tables 3-13 and 3-14). Sample sizes were insufficient in Mattawoman Creek during 2003-2005 to compare species richness by year.

Northeast River annual beach seine collections were generally dominated by clupeids and White Perch during 2007-2017 with Blueback Herring, Gizzard Shad, and White Perch often present in the top 90%; 2022 was notable for the dominance of Atlantic Menhaden (Figure 3-28). Blueback Herring fluctuated between 5% (2014) and 35% (2009); Gizzard Shad, 5% (2008) and 63% (2012); and White Perch, 7% (2013) and 46% (2010). Alewife was present in the top 90% of species during 6 of the 12 sampling years, ranging from 3% (2014) to 11% (2017); Bay

Anchovy, 7 of the 12 years, ranging from 3% (2008) to 17% (2013); Spottail Shiner, 4 of 12 years, ranging from 3% (2008) to 5% (2015); Pumpkinseed, 3 of the 12 years, ranging from 2% (2015) to 9% (2010); and Threadfin Shad, 2 of 12 years, ranging from 10% (2022) to 21% (2013). Atlantic Menhaden (36% in 2022), Striped Bass (2% in 2008), and Yellow Perch (2% in 2008) were only present in the top 90% during one year of sampling. Species richness in the top 90% was greatest in 2008 (8) and lowest in 2012 (2; Figure 3-28).

South River annual beach seine composition primarily consisted of Atlantic Menhaden in 2004 (64%), 2005 (58%), and 2022 (71%; Figure 3-28). Although present in 2003, Atlantic Menhaden only made up 5% of catch. Atlantic Silverside were present during all four sampling year, ranging from 8% (2022) to 25% (2005); Inland Silverside, 3 of the 4 years, ranging from 2% (2005) to 9% (2022); Striped Bass, 2 of the 4 years, ranging from 3% (2005) to 24% (2003); and White Perch, 2 of the 4 years, ranging from 9% (2004) to 47% (2003). Spot (3% in 2005) was only present in seine samples during one year of sampling. Species richness was greatest in 2003 (5) and lowest in 2004 and 2022 (3).

Tred Avon River annual beach seine composition was similar among all seventeen years of sampling (Figure 3-28). One species, Atlantic Silverside, was in the top 90% during all years, ranging from 10% (2010) to 71% (2015). Striped Killifish was present 16 out of 17 sampling years, ranging from 5% (2006) to 18% (2008); White Perch, 15 out of 17 years, ranging from 4% (2010) to 63% (2012); Mummichog, 14 out of 17 years, ranging from 4% (2015) to 20% (2020); Atlantic Menhaden, 10 out of 17 years, ranging from 3% (2012) to 53% (2010); Bay Anchovy, 6 out of 17 years, ranging from 4% (2019) to 14% (2013); Striped Bass, 6 out of 17 years, ranging from 3% (2006) to 6% (2018); and Spot, 4 out of 17 years, ranging from 6% (2007) to 13% (2010). Inland Silverside (3% in 2010) and Sheepshead Minnow (4% in 2010) were present in the top 90% during one year of sampling. The greatest species richness occurred in 2020 and 2022, both had 7 species in the top 90%; the lowest species richness occurred in 2009 and 2015, only 4 species were in the top 90% (Figure 3-28).

Finfish trawl catches in Mattawoman Creek during 2022 were low compared to previous years (Figure 3-18); juvenile and adult White Perch GMs from 4.9m bottom trawl catches declined compared to previous sampling years (Figure 3-23). Adult White Perch abundance declined twice as much as juvenile White Perch abundance (Figure 3-23) with no changes in modified PSDs (Table 3-16). Mattawoman Creek bottom trawl catches have been declining during 2014-2017; the 2022 trawl GM was lower than 2017 and comparable to lowest catches of 2007-2009 (Figure 3-18). Spottail Shiners completely disappeared from the top 90% of trawl catches in 2022 in Mattawoman Creek, a notable change since Spottail Shiners have comprised part of the top 90% since sampling started (Figure 3-20); species richness in bottom trawl catches declined in 2022 (12) from 2016 (19), respectively. The changes in species composition could reflect changes in flows, increased development, changes in sedimentation and siltation, increased SAV presence, increased invasive fish species, and conditions outside of Mattawoman Creek where other processes important to year-class strength occur.

The Scientific and Technical Advisory Committee of the Chesapeake Bay Program or STAC (2023) cited estuarine Mattawoman Creek as an example of a dramatic restoration in recent decades based on reduced nutrient loads, improved water clarity, and SAV restoration. In the mid- to late-1990s, nitrogen reductions began in earnest, and an extended drought period in 1999–2002 contributed to drops in N loads. This extended period of reduced nutrient loads produced a decline in algal biomass and a correlated increase in water clarity. The increase in water clarity supported the resurgence of SAV, assisted by the presence of an invasive

introduced species (*Hydrilla*) which can take advantage of short-term periods of water clarity for establishment (STAC 2023).

Uphoff et al. (2016) described Mattawoman Creek's ecosystem status as shifting between ecosystem states. A similar shift within the same timeframe to a clear, SAV dominated state due to lowered nutrients has been described for Gunston Cove, a tidal-fresh subestuary located nearby on the Virginia side of Potomac River (Jones 2020). The term "regime shift" has been used to suggest jumps between alternative equilibrium states are nonlinear, causally connected, and linked to other changes in an ecosystem (Steele 1996; Duarte et al. 2009; Kemp et al. 2009). Eutrophication is one of these forcing mechanisms (Duarte et al. 2009), while urbanization creates a set of stream conditions (urban stream syndrome; Hughes et al. 2014a; 2014b; Mackintosh et al. 2016) that qualifies as a shift as well. Both processes (eutrophication and urban stream syndrome) are interrelated products of development in Mattawoman Creek's watershed. Sediment loads in Mattawoman Creek from construction and stream bank erosion were high (Gellis et al. 2009) and increased nutrient loading.

In 2022, there was little indication that low DO was more widespread in Mattawoman Creek than usual, nor did the other water quality measurements offer an obvious connection to changes in finfish abundance. Bottom DO at all stations remained above the target level. Coverage of SAV was on an upward swing. Changes in stream hydrology and water quality have been concurrent with the approaching and breaching of the development threshold in Mattawoman Creek's watershed, increased sediment and nutrient loading from stream erosion and construction, decreased chlorophyll a (a powerful indicator of ecosystem response to nutrients; Duarte et al. 2009) and DO. Boyton et al. (2014) modeled nutrient inputs and outputs in Mattawoman Creek and found that nutrients were not exported out of the subestuary, suggesting that wetlands, emergent vegetation, and SAV in Mattawoman Creek were efficiently metabolizing and sequestering nutrients. Uphoff et al. (2011b) found low DO patches were not uncommon within an extensive SAV bed in Mattawoman Creek and DO conditions were generally worse within the SAV bed than in bottom channel waters. The SAV may have higher respiration than the phytoplankton it has replaced or provides more organic biomass that fuels respiration of decomposers, lowering DO. During 2014, we further explored a hypothesis that water quality dynamics in Mattawoman Creek's extensive SAV beds (low DO, high pH, and high organic matter) may be creating episodes of ammonia toxicity for fish (Uphoff et al. 2014). A 24-hour study in a single SAV bed suggested that fish could be caught in a habitat squeeze in SAV from high ammonia at the surface and low DO at the bottom (Uphoff et al. 2014). Clear evidence of fish community recovery associated with recovery of this subestuary's SAV has not revealed itself.

Northeast River finfish trawl catches declined in 2022 compared to previous sampling years (Figure 3-18); beach seine catches also indicated a slight decline since last sampled in 2017 in the number of finfish caught (Figure 3-26). Adult White Perch GMs increased slightly higher than the time-series median in Northeast River (Figure 3-23), coinciding with a minimal decrease in modified PSDs (Table 3-16). Northeast River bottom channel trawl catches fell to their lowest level for all years sampled (Figure 3-18), while inshore seine catches ranked among the highest of all sampling years (Figure 3-26). Brown Bullhead appeared in the top 90% of trawl catches in 2022 in Northeast River, a notable change in the species present from 2017 when only White Perch comprised the top 90% (Figure 3-20); species richness in bottom trawl catches declined in 2022, respectively. The changes in species composition could reflect changes in flows, increased

development, DO violations, and conditions outside of Northeast River where other processes important to year-class strength occur.

Tred Avon River finfish trawl catches declined in 2022 compared to previous sampling years; beach seine catches also indicated a decline in the number of finfish caught. Adult White Perch GMs increased slightly higher than the time-series median in Tred Avon River (Figure 3-23), coinciding with a minimal increase in modified PSDs (Table 3-16; Figure 3-24). Tred Avon River bottom channel trawl catches fell to their 5th lowest level for all years sampled (Figure 3-18), while inshore seine catches were lower than previous years but not the lowest (Figure 3-26). Tred Avon River trawl catches in 2022 were slightly higher than the last few years but remained below the time-series median GM (Figure 3-18). There was little change in the species present and richness in bottom trawl catches in 2022 for Tred Avon River (Figure 3-20). Spot increased noticeably again in 2022 and has continued to increase in abundance over the last five years, while White Perch has been declining in abundance since 2018 (Figure 3-20). The changes in species composition could reflect changes in salinity, increased development, DO violations, and conditions outside of Tred Avon River where other processes important to year-class strength occur.

South River was previously sampled in 2003–2005. During the sixteen-year hiatus, C/ha increased from 1.27 to 1.43 C/ha within the suburban watershed. South River finfish bottom trawl (Figure 3-18) and beach seine (Figure 3-26) catches fell to their lowest levels. There was little change in the species present and richness in bottom trawls. Adult White Perch GMs declined (Figure 3-23), while modified PSDs increased substantially in 2022 (Table 3-16; Figure 3-24). Bottom trawl finfish counts indicate a number of bottom trawls did not catch finfish; in 2003, seven bottom trawls caught no finfish (station 02 = 4; station 03 = 3); in 2004, two bottom trawls caught no finfish (station 01 = 1; station 03 = 1); in 2005, five bottom trawl caught no finfish (station 02 = 2; station 03 = 3); and in 2022, six bottom trawl caught no finfish (station 01 = 2; station 02 = 3; station 03 = 1), respectively. For all sampling years, South River trawl station 02 has the greatest number of trawls with zero finfish catch (9), followed by station 03 (8), and then station 01 (3), respectively; station 04 trawls all caught finfish (Figure 3-2). Lower finfish catches in the bottom channel and inshore within the upper- and mid-river could be associated with the increased development and low DO measurements. Overall, South River is following suit with other developed mesohaline systems (Severn and Magothy Rivers), where low finfish catches in the bottom channel reflect increased development and low DO.

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Tables

Table 3-1. Summary of all subestuaries and their location, year sampled, number of stations, and sampling gear used. An 'x' indicates sampling was conducted with the gear labeled.

Subestuary	Area	Year	Beach Seines (30.5 m)	Number of Seine Stations	Bottom Trawls (4.9 m)	Number of Trawl Stations	Bottom Trawls (3.1 m)	Number of Trawl Stations		
Blackwater River	Lower-Bay	2006	x	1	x	5				
Bohemia River	Upper-Bay	2006	x	4	x	4				
Breton Bay	Potomac	2003	x	4	x	4				
		2004	x	4	x	4				
		2005	x	4	x	4				
		2012	x	3	x	4				
		2013	x	3	x	4				
Broad Creek	Mid-Bay (Choptank)	2014	x	3	x	4				
		2015	x	3	x	4				
		2016	x	3	x	4				
		2017	x	3	x	4				
		2020	x	2	x	4				
		Bush River	Upper-Bay	2006	x	4	x	3		
				2007	x	4	x	3		
2008	x			4	x	3				
2009	x			4	x	3				
2010	x			4	x	3				
Chester River	Mid-Bay (Chester)	1994	x	4			x	4		
		1995	x	10			x	10		
		1996	x	10			x	10		
		1997	x	10			x	10		
		1998	x	10			x	10		
		1999	x	10			x	10		
		2000	x	10			x	10		
		2007	x	6	x	6				
		2008	x	6	x	6				
		2009	x	6	x	6				
		2010	x	6	x	6				
		2011	x	6	x	6				
		2012	x	6	x	6				
		2018	x	6						
		2019	x	6	x	6				
Corsica River	Mid-Bay (Chester)	2003	x	3	x	4				
		2004	x	3	x	4				
		2005	x	3	x	4				
		2006	x	3	x	4				
		2007	x	3	x	4				
		2008	x	3	x	4				
		2009	x	3	x	4				
		2010	x	3	x	4				
		2011	x	3	x	4				
		2012	x	3	x	4				
		2018	x	3	x	4				
Fishing Bay	Lower-Bay	2019			x	4				
		2006	x	4	x	4				

Table 3-1. Continued.

Gunpowder River	Upper-Bay	2009	x	4	x	4		
		2010	x	4	x	4		
		2011	x	4	x	4		
		2012	x	4	x	4		
		2013	x	4	x	4		
		2014	x	3	x	4		
		2015	x	3	x	4		
Harris Creek	Mid-Bay (Choptank)	2016	x	3	x	4		
		2012	x	3	x	4		
		2013	x	3	x	4		
		2014	x	3	x	4		
Langford Creek	Mid-Bay (Chester)	2015	x	3	x	4		
		2016	x	3	x	4		
		2006	x	4	x	4		
		2007	x	4	x	4		
		2008	x	4	x	4		
Magothy River Mattawoman Creek	Mid-Bay Potomac	2018	x	3	x	4		
		2019	x	3	x	4		
		2003	x	4	x	4		
		1989	x	5			x	5
		1990	x	5			x	5
		1991	x	5			x	5
		1992	x	5			x	5
		1993	x	5			x	5
		1994	x	5			x	5
		1995	x	5			x	5
		1996	x	5			x	5
		1997	x	5			x	5
		1998	x	5			x	5
		1999	x	5			x	5
		2000	x	5			x	5
2001	x	5			x	5		
2002	x	4			x	5		
2003	x	4		x	4			
2004	x	4		x	4			
2005	x	4		x	4			
2006				x	4			
2007				x	4			
2008				x	4			
2009	x	1		x	4	x	4	
2010				x	4	x	4	
2011				x	4	x	4	
2012				x	4	x	4	
2013				x	4	x	4	
2014				x	4	x	4	
2015				x	4	x	4	
2016				x	4	x	4	
2022				x	4			

Table 3-1. Continued.

Middle River	Upper-Bay	2009	x	2	x	4		
		2010	x	3	x	4		
		2011	x	3	x	4		
		2012	x	3	x	4		
		2013	x	2	x	4		
		2014	x	2	x	4		
		2015	x	1	x	4		
		2016			x	4		
		2017			x	4		
		Miles River	Mid-Bay	2003	x	3	x	4
2004	x			3	x	4		
2005	x			3	x	4		
2020	x			3	x	4		
Nanjemoy Creek	Potomac	2003	x	3	x	3		
		2008	x	3	x	3		
		2009	x	3	x	3		
		2010	x	3	x	3		
		2011	x	4	x	4		
		2012	x	4	x	4		
		2013	x	3	x	3		
		2014	x	3	x	3		
		2015	x	3	x	3		
		2016	x	3	x	3		
Northeast River	Upper-Bay	2007	x	4	x	4		
		2008	x	4	x	4		
		2009	x	4	x	4		
		2010	x	4	x	4		
		2011	x	4	x	4		
		2012	x	4	x	4		
		2013	x	4	x	4		
		2014	x	4	x	4		
		2015	x	4	x	4		
		2016	x	4	x	4		
		2017	x	4	x	4		
		2022	x	3	x	4		
		Piscataway Creek	Potomac	2003	x	3	x	3
				2006	x	2	x	3
2007					x	3		
2009					x	3		
2010					x	3		
2011					x	3		
2012					x	3		
2013					x	3		
2014					x	3		
2015					x	3		
Rhode River	Mid-Bay	2003	x	2	x	2		
		2004	x	2	x	2		
		2005	x	2	x	2		
		2021	x	4	x	4		
Sassafas River	Upper-Bay	2020			x	4		
		2003	x	5	x	4		
		2004	x	5	x	4		
		2005	x	5	x	4		
Severn River	Mid-Bay	2017	x	3	x	4		
		2003	x	4	x	4		
		2004	x	4	x	4		
		2005	x	4	x	4		
South River	Mid-Bay	2022	x	4	x	4		
		2003	x	4	x	4		
		2004	x	4	x	4		
		2005	x	4	x	4		
St. Clements River	Potomac	2003	x	4	x	4		
		2004	x	4	x	4		
		2005	x	4	x	4		
Transquaking River	Lower-Bay	2006			x	1		

Table 3-1. Continued.

Tred Avon River	Mid-Bay (Choptank)	2006	x	4	x	4		
		2007	x	4	x	4		
		2008	x	4	x	4		
		2009	x	4	x	4		
		2010	x	4	x	4		
		2011	x	4	x	4		
		2012	x	4	x	4		
		2013	x	4	x	4		
		2014	x	4	x	4		
		2015	x	4	x	4		
		2016	x	4	x	4		
		2017	x	4	x	4		
		2018	x	4	x	4		
		2019	x	4	x	4		
		2020	x	4	x	4		
		2021	x	4	x	4		
		2022	x	4	x	4		
		West River	Mid-Bay	2003	x	1	x	2
				2004	x	1	x	2
				2005	x	1	x	2
		Wicomico River	Potomac	2003	x	4	x	4
				2010	x	4	x	4
2011	x			4	x	4		
2012	x			4	x	4		
Wye River	Mid-Bay	2017	x	4	x	4		
		2007	x	4	x	4		
		2008	x	4	x	4		
		2018	x	3	x	4		
		2019	x	3	x	4		

Table 3-2. 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 2022.

2022 Sampled Subestuaries						
Area	Subestuary	IS	C/ha	Land Hectares	Water Hectares	Salinity Class
Potomac	Mattawoman Creek	11.41	1.00	24,403	765	Tidal-Fresh
Head-of-Bay	Northeast River	6.41	0.51	16,328	1,661	Tidal-Fresh
Mid-Bay	South River	15.72	1.43	14,792	2,324	Mesohaline
Mid-Bay	Tred Avon River	9.26	0.79	9,562	3,087	Mesohaline

Table 3-3. Estimates of structures per hectare (C/ha) and land use percentages from Maryland Department of Planning (2002 and 2010) for subestuaries sampled 2003 – 2022.

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
Broad Creek	2020	0.31	42.6	0.4	25.4	31.5
Bush River	2006	1.17	25.4	3.2	35.0	36.2
Bush River	2007	1.19	25.4	3.2	35.0	36.2
Bush River	2008	1.2	25.4	3.2	35.0	36.2
Bush River	2009	1.21	25.4	3.2	35.0	36.2
Bush River	2010	1.22	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	0.16	64.2	2.0	24.7	8.9
Chester River	2019	0.16	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	0.28	60.4	0.1	25.5	13.2
Corsica River	2019	0.28	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
Harris Creek	2012	0.39	44.9	5.6	19.7	29.8
Harris Creek	2013	0.40	44.9	5.6	19.7	29.8
Harris Creek	2014	0.40	44.9	5.6	19.7	29.8
Harris Creek	2015	0.40	44.9	5.6	19.7	29.8
Harris Creek	2016	0.40	44.9	5.6	19.7	29.8

Table 3-3. Continued.

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	0.08	70.2	1.5	20.4	8.0
Langford Creek	2019	0.08	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	0.92	9.3	2.8	53.9	34.2
Mattawoman Creek	2014	0.93	9.3	2.8	53.9	34.2
Mattawoman Creek	2015	0.94	9.3	2.8	53.9	34.2
Mattawoman Creek	2016	0.96	9.3	2.8	53.9	34.2
Mattawoman Creek	2022	1.00	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	3.37	3.4	2.1	23.3	71.0
Middle River	2014	3.39	3.4	2.1	23.3	71.0
Middle River	2015	3.40	3.4	2.1	23.3	71.0
Middle River	2016	3.42	3.4	2.1	23.3	71.0
Middle River	2017	3.41	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
Miles River	2020	0.26	49.0	0.8	26.7	23.1
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 3-3. Continued.

Northeast River	2012	0.47	31.1	0.1	38.6	28.9
Northeast River	2013	0.48	31.1	0.1	38.6	28.9
Northeast River	2014	0.48	31.1	0.1	38.6	28.9
Northeast River	2015	0.49	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
Northeast River	2022	0.51	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	1.50	10.0	0.2	40.4	47.0
Piscataway Creek	2014	1.51	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
Sassafras River	2020	0.11	64.1	1.3	25.9	8.3
Sassafras River	2021	0.11	64.1	1.3	25.9	8.3
Seyern River	2003	2.06	8.6	0.2	35.2	55.8
Seyern River	2004	2.09	8.6	0.2	35.2	55.8
Seyern River	2005	2.15	8.6	0.2	35.2	55.8
Seyern River	2017	2.38	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
South River	2022	1.43	10.2	0.5	39.2	48.8
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	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2015	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2016	0.78	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	0.78	43.2	0.8	21.6	33.6
Tred Avon River	2019	0.79	43.2	0.8	21.6	33.6
Tred Avon River	2020	0.79	43.2	0.8	21.6	33.6
Tred Avon River	2021	0.79	43.2	0.8	21.6	33.6
Tred Avon River	2022	0.79	43.2	0.8	21.6	33.6
Wicamico River	2003	0.30	34.7	4.6	48.5	12.0
Wicamico River	2010	0.34	31.6	4.6	44.9	18.7
Wicamico River	2011	0.35	31.6	4.6	44.9	18.7
Wicamico River	2012	0.35	31.6	4.6	44.9	18.7
Wicamico River	2017	0.39	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 3-4. Summary of water quality parameter statistics collected during both seine and trawl samples for subestuaries in 2022. Summary statistics for pH were calculated from H+ concentrations and converted back to pH.

System	Statistics	Surface Measurements					Bottom Measurements					Secchi
		Temp (°C)	DO (mg / L)	Cond (umhols)	Salinity	pH	Temp (°C)	DO (mg / L)	Cond (umhols)	Salinity	pH	
Mattawoman Creek	Mean	27.69	7.59	472.00	0.22	7.87	27.46	6.64	483.67	0.23	7.77	0.79
	Standard Error	0.42	0.27	58.57	0.03	8.59	0.42	0.22	63.52	0.03	8.54	0.05
	Median	27.03	7.73	293.00	0.14	8.24	26.88	6.63	295.00	0.14	8.08	0.78
	Mode	.	.	.	0.13	8.52	.	5.56	296.00	0.14	8.24	1.00
	Kurtosis	-1.19	-1.06	-1.37	-1.35	-1.76	-1.02	-1.28	-1.03	-0.90	-1.61	1.56
	Skewness	0.35	-0.22	0.78	0.80	0.03	0.31	-0.17	0.87	0.90	0.02	0.74
	Minimum	24.54	4.96	211.00	0.10	8.55	24.22	4.56	212.00	0.10	8.42	0.40
	Maximum	30.89	9.85	962.00	0.47	7.35	30.94	8.17	1153.00	0.58	7.28	1.50
	Count	24	24	24	24	24	24	24	24	24	24	24
	Northeast River	Mean	27.60	8.95	264.55	0.13	8.44	26.88	6.62	264.96	0.12	7.96
Standard Error		0.21	0.16	6.59	0.00	9.50	0.23	0.24	9.06	0.00	8.81	0.02
Median		27.88	8.72	265.50	0.13	8.49	26.88	6.84	265.50	0.13	8.05	0.40
Mode		29.01	8.42	296.00	0.10	8.44	26.33	.	221.00	0.12	7.77	0.40
Kurtosis		-0.98	-0.07	-1.16	-1.18	-0.69	-1.28	-0.20	-1.20	-1.11	0.63	-0.06
Skewness		-0.27	0.55	0.03	0.01	0.08	-0.22	-0.24	-0.07	-0.09	0.02	0.64
Minimum		25.30	6.96	189.00	0.09	9.05	24.91	4.25	195.00	0.09	8.64	0.25
Maximum		30.53	11.57	343.00	0.16	8.07	28.43	8.76	339.00	0.16	7.52	0.60
Count		42	42	42	42	42	24	24	24	24	24	24
South River		Mean	27.44	6.71	16855.38	9.89	-0.89	26.77	1.76	19374.67	11.52	7.43
	Standard Error	0.24	0.22	494.79	0.31	1.42	0.21	0.23	617.14	0.40	8.55	0.05
	Median	27.64	6.79	17146.50	10.05	-0.89	26.77	1.70	19217.00	11.37	7.45	0.80
	Mode	27.98	6.78	.	7.75	-0.89	26.76	0.52	.	.	7.36	1.00
	Kurtosis	0.57	0.29	-0.46	-0.55	0.23	-0.29	-1.21	-1.19	-1.24	-1.05	-0.34
	Skewness	-0.56	0.20	-0.39	-0.33	1.82	-0.14	0.30	-0.13	-0.07	0.23	-0.23
	Minimum	22.45	3.62	8507.00	4.72	-0.86	24.95	0.31	13817.00	7.96	7.72	0.25
	Maximum	30.69	10.96	22169.00	13.32	-0.93	28.94	3.96	23956.00	14.50	7.17	1.25
	Count	48	48	48	48	48	24	24	24	24	24	24
	Tred Avon River	Mean	28.34	6.62	17635.42	30.67	7.73	28.16	4.61	18125.83	10.66	7.59
Standard Error		0.24	0.15	194.82	20.31	9.02	0.28	0.26	260.80	0.17	8.67	0.02
Median		28.58	6.54	17524.50	10.31	7.77	28.51	4.87	17851.50	10.50	7.63	0.50
Mode		29.88	6.38	17271.00	9.66	7.80	7.62	0.50
Kurtosis		-0.44	0.16	0.61	48.00	-0.48	-0.49	1.32	1.87	2.05	-0.79	0.06
Skewness		-0.06	0.17	0.24	6.93	-0.20	-0.69	-1.35	1.26	1.30	-0.36	0.20
Minimum		25.46	4.38	14367.00	8.30	8.05	25.67	1.33	16438.00	9.59	7.79	0.30
Maximum		31.94	9.02	20830.00	985.00	7.39	30.23	6.33	21811.00	13.11	7.20	0.70
Count		48	47	48	48	48	24	24	24	24	24	24

Table 3-5. Percentages of all dissolved oxygen (DO; surface, middle, and bottom) 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 2022. C/ha = structures per hectare. N = number of samples.

Subestuary	Salinity Class	C/ha	N	All DO		Bottom DO	
				% < 5.0 mg/L	N	% < 5.0 mg/L	% < 3.0 mg/L
Mattawoman Creek	Tidal-Fresh	1.00	52	4%	24	4%	0%
Northeast River	Tidal-Fresh	0.51	77	5%	24	13%	0%
South River	Mesohaline	1.43	90	48%	24	100%	83%
Tred Avon River	Mesohaline	0.79	95	23%	24	54%	13%

Table 3-6. Subestuaries sampled during 2003 – 2022, 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).

River	Year	C / ha	Temperature (°C)		Dissolved Oxygen (mg / L)	
			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	2020	0.31	27.94	27.57	7.55	5.57
	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	0.16	27.54	26.90	6.83	6.00
Corsica River	2019	0.16	27.45	27.05	6.75	5.77
	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	0.28	27.23	26.71	7.02	5.12
Fishing Bay	2019	0.28	27.24	27.04	6.82	4.39
	2006	0.03	26.23	25.28	7.24	6.79
Harris Creek	2012	0.39	26.55	26.42	7.44	6.35
	2013	0.40	26.39	26.05	7.02	6.01
	2014	0.40	27.61	26.68	6.84	4.84
	2015	0.40	26.62	26.62	7.19	6.56
	2016	0.40	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	0.08	27.08	31.78	6.40	5.10
	2019	0.08	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
	2020	0.26	27.88	26.90	6.50	3.42

Table 3-6. Continued.

Rhode River	2003	0.47	25.00	24.69	7.10	4.80
	2004	0.47	27.00	26.95	6.58	5.39
	2005	0.48	27.78	27.16	6.50	4.03
Severn River	2003	2.06	26.30	24.75	7.60	1.57
	2004	2.09	27.42	26.18	7.05	2.64
	2005	2.15	28.01	26.23	7.07	0.96
	2017	2.38	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
	2022	1.43	27.44	26.77	6.71	1.76
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	0.77	26.66	26.51	6.12	5.90
	2015	0.77	28.00	27.60	6.92	5.54
	2016	0.78	28.89	28.44	7.27	5.15
	2017	0.77	26.49	26.13	7.01	5.04
	2018	0.78	27.79	27.34	7.34	4.81
	2019	0.79	28.62	28.22	6.79	4.49
	2020	0.79	28.29	28.11	6.91	4.35
2021	0.79	28.72	28.04	6.61	4.64	
2022	0.79	28.34	28.16	6.62	4.61	
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	0.30	25.40	23.83	7.00	5.85
	2010	0.34	25.43	25.30	6.06	5.21
	2011	0.35	27.08	26.89	5.57	4.30
	2012	0.35	27.57	27.38	6.59	5.44
	2017	0.39	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

Table 3-6. Continued.

Oligohaline						
Bohemia River	2006	0.11	26.79	26.02	7.01	6.41
Bush River	2006	1.17	25.48	24.28	7.96	7.47
	2007	1.19	27.02	26.42	7.68	6.54
	2008	1.20	26.59	24.20	9.00	5.43
	2009	1.21	25.88	24.34	9.41	8.54
	2010	1.22	27.72	23.80	7.79	7.04
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
Middle River	2016	0.74	27.70	26.46	7.43	6.18
	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	3.37	27.12	26.46	7.58	5.79
	2014	3.39	26.56	26.01	7.55	6.04
	2015	3.40	28.47	27.20	8.20	6.23
Nanjemoy Creek	2016	3.42	28.87	27.82	7.56	5.69
	2017	3.41	25.54	25.17	7.80	5.36
	2003	0.08	25.90	28.80	7.30	4.96
	2008	0.09	27.53	26.58	7.85	6.65
	2009	0.09	26.31	24.64	7.05	7.49
	2010	0.09	26.50	24.80	7.66	7.02
	2011	0.09	29.34	28.55	6.13	5.30
	2012	0.09	26.18	25.92	6.73	5.98
	2013	0.09	26.88	26.30	6.76	5.86
	2014	0.09	26.78	26.36	7.66	6.25
2015	0.09	27.40	27.10	7.16	6.32	
2016	0.09	28.49	28.21	6.86	5.16	
Tidal Fresh						
Mattawoman Creek	2003	0.76	26.00	25.75	9.00	8.81
	2004	0.79	27.33	27.14	8.34	7.95
	2005	0.81	28.77	28.09	7.74	7.27
	2006	0.83	27.05	26.44	7.10	6.50
	2007	0.86	26.89	26.85	6.70	6.48
	2008	0.87	26.40	24.52	7.97	6.33
	2009	0.88	26.20	26.64	7.92	7.86
	2010	0.90	26.21	26.10	6.95	6.62
	2011	0.91	27.08	27.46	6.33	6.51
	2012	0.90	26.70	26.82	7.40	7.00
	2013	0.92	26.35	25.94	9.22	8.40
	2014	0.93	26.73	26.24	7.48	6.17
	2015	0.94	27.91	26.84	8.66	7.74
	2016	0.96	28.47	28.03	6.96	6.54
	2022	1.00	27.69	27.46	7.59	6.64

Table 3-6. Continued.

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	0.48	26.61	26.11	9.33	7.06
	2014	0.48	26.94	26.52	7.72	6.81
	2015	0.49	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
	2022	0.51	27.60	26.88	8.95	6.62
	Piscataway Creek	2003	1.30	25.60	24.63	10.20
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		1.50	27.19	26.22	9.87	7.65
Sassafras River	2014	1.51	26.98	26.28	8.66	7.33
	2020	0.11	28.14	27.27	9.83	7.06
	2021	0.11	27.65	27.27	7.83	6.30

Table 3-7. Pearson correlations (*r*) of mean survey surface and bottom dissolved oxygen (DO; mg/L) with water temperatures at depth (surface or bottom) or watershed development (C/ha = structures per hectare) for subestuaries sampled during 2003 – 2022, by salinity class. Level of significance = *P*. N = sample size.

DO Depth Statistics		Temperature	C / ha
Mesohaline			
Surface	<i>r</i>	-0.006	0.229
	<i>P</i>	0.952	0.030
	N	90	90
Bottom	<i>r</i>	0.063	-0.602
	<i>P</i>	0.557	<.0001
	N	90	90
Oligohaline			
Surface	<i>r</i>	-0.310	0.402
	<i>P</i>	0.080	0.020
	N	33	33
Bottom	<i>r</i>	-0.602	-0.085
	<i>P</i>	0.0002	0.637
	N	33	33
Tidal Fresh			
Surface	<i>r</i>	-0.008	0.151
	<i>P</i>	0.960	0.364
	N	38	38
Bottom	<i>r</i>	-0.013	0.397
	<i>P</i>	0.937	0.014
	N	38	38

Table 3-8. Pearson correlations (r) of C/ha for mesohaline subestuaries sampled during 2003–2022 with Maryland Department of Planning (DOP) land use categories. Pearson correlations (r) between land use categories estimated by MD DOP for 2002 and 2010. *P* = level of significance. *N* = sample size. Duplicate entries of C/ha for mesohaline subestuaries from 2003 to 2022 were not included in analysis.

	Statistics	C/ha	Land Use Categories			
			Agriculture	Forest	Wetland	Urban
C/ha	r					
	<i>P</i>	1				
	<i>N</i>					
Agriculture	r	-0.715				
	<i>P</i>	<0.0001	1			
	<i>N</i>	86				
Forest	r	0.076	-0.597			
	<i>P</i>	0.486	<0.0001	1		
	<i>N</i>	86	86			
Wetland	r	-0.206	-0.223	0.207		
	<i>P</i>	0.057	0.039	0.056	1	
	<i>N</i>	86	86	86		
Urban	r	0.896	-0.717	-0.040	-0.263	
	<i>P</i>	<0.0001	<0.0001	0.713	0.015	1
	<i>N</i>	86	86	86	86	

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

Linear Model Western Shore: Median Bottom DO = Agriculture (%)						
ANOVA	df	SS	MS	F	Significance F	
Regression	1	56.04	56.04	60.21	<0.0001	
Residual	21	19.55	0.93			
Total	22	75.58				
$r^2 = 0.741$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.49	0.45	1.09	0.29	-0.44	1.41
Agriculture (%)	0.13	0.02	7.76	<0.0001	0.10	0.17
Linear Model Eastern Shore: Median Bottom DO = Agriculture (%)						
ANOVA	df	SS	MS	F	Significance F	
Regression	1	6.38	6.38	9.36	0.0033	
Residual	60	40.90	0.68			
Total	61	47.28				
$r^2 = 0.135$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	6.78	0.55	12.27	<0.0001	5.67	7.88
Agriculture (%)	-0.03	0.01	-3.06	0.0033	-0.05	-0.01

Table 3-10. Statistics and parameter estimates for a quadratic regression of median bottom dissolved oxygen (DO; mg/L) 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	96.20	48.10	59.14	<0.0001	
Residual	82	66.69	0.81			
Total	84	162.89				
$r^2 = 0.5906$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	-0.06	0.45	-0.14	0.89	-0.96	0.83
Agriculture (%)	0.22	0.02	9.8	<0.0001	0.18	0.27
Agriculture (%)^2	-0.002	0.0003	-8.34	<0.0001	-0.003	-0.002

Table 3-11. Percent of watershed in major land use categories estimated by Maryland Department of Planning (DOP 2010) for each subestuary sampled in 2022. 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.

Land Use Category	Head-of-Bay Subestuary	Mid-Bay Subestuaries		Potomac River Tributary
	Northeast River	Tred Avon River	South River	Mattawoman Creek
Agriculture	31.1	43	10	9.3
Forest	38.6	22	39	53.9
Urban	28.9	34	49	34.2
Wetlands	0.1	1	<1	2.8
Water	9.2	24	14	3

Table 3-12. Percentages of all dissolved oxygen (DO; mg/L) 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 subestuaries and tributaries sampled in 2022. N = sample size.

Subestuary	Year	C / ha	N	All DO		Bottom DO	
				% < 5.0 mg/L	N	% < 5.0 mg/L	% < 3.0 mg/L
Mattawoman Creek	2003	0.76	59	0%	22	0%	0%
	2004	0.79	63	0%	23	0%	0%
	2005	0.81	46	0%	16	0%	0%
	2006	0.83	46	15%	22	18%	5%
	2007	0.86	60	13%	24	17%	0%
	2008	0.87	35	3%	12	8%	0%
	2009	0.88	34	0%	11	0%	0%
	2010	0.90	50	0%	23	0%	0%
	2011	0.91	51	14%	19	11%	0%
	2012	0.90	47	6%	21	5%	0%
	2013	0.92	50	0%	23	0%	0%
	2014	0.93	50	0%	20	0%	0%
	2015	0.94	47	0%	21	0%	0%
	2016	0.96	48	0%	21	0%	0%
	2022	1.00	52	4%	24	4%	0%
Northeast River	2007	0.44	86	3%	23	9%	0%
	2008	0.44	74	7%	19	11%	0%
	2009	0.45	78	1%	23	4%	0%
	2010	0.46	71	1%	17	0%	0%
	2011	0.46	88	14%	24	33%	13%
	2012	0.47	82	7%	24	21%	0%
	2013	0.48	85	2%	24	8%	0%
	2014	0.48	80	1%	24	4%	0%
	2015	0.49	85	5%	24	13%	4%
	2016	0.49	84	0%	24	0%	0%
2017	0.49	93	1%	24	4%	0%	
2022	0.51	77	5%	24	13%	0%	
South River	2003	1.24	112	29%	28	75%	57%
	2004	1.25	93	34%	24	79%	25%
	2005	1.27	91	42%	24	92%	63%
2022	1.43	90	48%	24	100%	83%	
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	0.77	96	11%	24	21%	0%
	2015	0.77	96	8%	24	21%	13%
	2016	0.78	96	13%	24	38%	13%
	2017	0.77	89	17%	24	42%	13%
	2018	0.78	110	17%	28	50%	14%
2019	0.79	96	30%	24	71%	17%	
2020	0.79	96	27%	24	63%	17%	
2021	0.79	96	23%	24	46%	17%	
2022	0.79	95	23%	24	54%	13%	

Table 3-13. Beach seine catch summary, 2022. C/ha = structures per hectare. GM CPUE = geometric mean catches 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
Northeast River	3	18	27	<i>Atlantic Menhaden</i> <i>White Perch (Adults)</i> Threadfin Shad <i>Blueback Herring</i> Gizzard Shad Alewife Bay Anchovy Spottail Shiner	0.51	5,610	210
South River	4	19	24	<i>Atlantic Menhaden</i> Inland Silverside Atlantic Silverside Striped Killifish	1.43	8,843	110
Tred Avon River	4	24	22	<i>Atlantic Silverside</i> <i>White Perch (Adults)</i> Spot Mummichog Banded Killifish <i>Striped Bass (Juv)</i> Bay Anchovy Striped Killifish	0.79	1,796	65
Grand Total	11	61	38	<i>Atlantic Menhaden</i> Atlantic Silverside <i>White Perch (Adults)</i> Inland Silverside Bay Anchovy Threadfin Shad Striped Killifish Gizzard Shad <i>Blueback Herring</i> Alewife Mummichog		16,249	

Table 3-14. Bottom trawl catch summary, 2022. C/ha = structures per hectare. GM CPUE = geometric mean catches 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
Mattawoman Creek	4	24	12	<i>White Perch (Juv)</i> Tessellated Darter <i>Spottail Shiner</i>	1.00	2,797	28
Northeast River	4	24	15	<i>White Perch (Adults)</i> Brown Bullhead	0.51	2,329	59
South River	4	24	12	<i>Bay Anchovy</i> <i>Spot</i>	1.43	1,460	10
Tred Avon River	4	24	13	<i>Spot</i> <i>Bay Anchovy</i> Hogchoker <i>White Perch (Adults)</i>	0.79	1,524	43
Grand Total	16	96	28	<i>White Perch (Juv)</i> <i>White Perch (Adults)</i> <i>Spot</i> <i>Bay Anchovy</i> Brown Bullhead Tessellated Darter		8,110	

Table 3-15. Subestuaries sampled during 2003 – 2022, grouped by salinity class and ranked by annual 4.9 m trawl catch geometric mean (GM) of all species combined.

River	Year	GM	Rank
Mesohaline			
Broad Creek	2014	385	1
Corsica River	2003	379	2
Miles River	2003	315	3
West River	2003	290	4
Broad Creek	2012	275	5
Langford Creek	2007	273	6
Chester River	2011	258	7
Tred Avon River	2010	253	8
Corsica River	2004	251	9
Corsica River	2011	245	10
Corsica River	2009	217	11
Langford Creek	2006	204	12
Tred Avon River	2014	182	13
Corsica River	2006	175	14
Rhode River	2003	174	15
Wye River	2007	168	16
Corsica River	2012	166	17
Harris Creek	2014	163	18
Corsica River	2010	157	19
Langford Creek	2008	157	20
Chester River	2010	157	21
Chester River	2007	149	22
Rhode River	2005	149	23
Chester River	2012	144	24
Tred Avon River	2008	141	25
Broad Creek	2013	139	26
Harris Creek	2012	133	27
Corsica River	2007	133	28
Broad Creek	2016	130	29
Tred Avon River	2012	128	30
Wicomico River	2010	124	31
Tred Avon River	2007	120	32
Chester River	2008	117	33
Corsica River	2005	114	34
Tred Avon River	2016	108	35
Wye River	2008	107	36
West River	2005	106	37
Fishing Bay River	2006	105	38
Corsica River	2008	103	39
Transquaking River	2006	95	40
Broad Creek	2015	94	41
Wicomico River	2012	93	42
Broad Creek	2017	90	43
Harris Creek	2013	88	44
Tred Avon River	2011	84	45

Table 3-15. Continued.

Chester River	2009	83	46
Tred Avon River	2009	82	47
Wicomico River	2017	82	48
Tred Avon River	2015	77	49
Miles River	2005	70	50
Chester River	2019	69	51
Wicomico River	2011	67	52
Tred Avon River	2017	66	53
Miles River	2004	65	54
Wicomico River	2003	61	55
Tred Avon River	2013	54	56
Tred Avon River	2006	48	57
Harris Creek	2016	45	58
Langford Creek	2019	43	59
Tred Avon River	2022	43	60
Corsica River	2019	34	61
Tred Avon River	2019	33	62
Harris Creek	2015	31	63
Langford Creek	2018	28	64
Broad Creek	2020	25	65
Rhode River	2004	21	66
Wye River	2019	20	67
South River	2003	20	68
St. Clements River	2005	20	69
Tred Avon River	2021	19	70
Breton Bay	2005	18	71
Tred Avon River	2020	17	72
South River	2005	17	73
Corsica River	2018	16	74
West River	2004	15	75
South River	2004	14	76
Tred Avon River	2018	13	77
Wye River	2018	12	78
Blackwater River	2006	10	79
Miles River	2020	10	80
South River	2022	10	81
St. Clements River	2004	9	82
Breton Bay	2004	9	83
Magothy River	2003	7	84
Breton Bay	2003	7	85
St. Clements River	2003	6	86
Severn River	2017	3	87
Severn River	2004	3	88
Severn River	2003	2	89
Severn River	2005	1	90
Oligohaline			
Nanjemoy Creek	2013	578	1

Table 3-15. Continued.

Middle River	2011	520	2
Bush River	2010	467	3
Nanjemoy Creek	2011	451	4
Nanjemoy Creek	2015	418	5
Nanjemoy Creek	2014	397	6
Gunpowder River	2011	394	7
Gunpowder River	2010	393	8
Bush River	2007	325	9
Bush River	2009	320	10
Nanjemoy Creek	2010	310	11
Middle River	2010	310	12
Nanjemoy Creek	2016	299	13
Middle River	2009	292	14
Middle River	2015	290	15
Gunpowder River	2009	288	16
Nanjemoy Creek	2009	286	17
Middle River	2016	262	18
Middle River	2014	254	19
Nanjemoy Creek	2012	226	20
Gunpowder River	2012	223	21
Gunpowder River	2014	220	22
Gunpowder River	2015	218	23
Bush River	2008	211	24
Nanjemoy Creek	2008	211	25
Gunpowder River	2016	206	26
Middle River	2013	183	27
Bush River	2006	153	28
Gunpowder River	2013	148	29
Middle River	2012	147	30
Bohemia River	2006	112	31
Nanjemoy Creek	2003	95	32
Middle River	2017	74	33
Tidal-Fresh			
Mattawoman Creek	2014	582	1
Northeast River	2010	306	2
Northeast River	2014	292	3
Piscataway Creek	2010	292	4
Northeast River	2011	291	5
Mattawoman Creek	2013	287	6
Piscataway Creek	2011	281	7
Mattawoman Creek	2004	252	8
Piscataway Creek	2014	223	9
Mattawoman Creek	2015	218	10
Mattawoman Creek	2011	210	11
Northeast River	2009	200	12
Northeast River	2012	195	13
Northeast River	2013	187	14

Table 3-15. Continued.

Piscataway Creek	2013	185	15
Northeast River	2008	153	16
Northeast River	2015	152	17
Mattawoman Creek	2005	150	18
Mattawoman Creek	2016	147	19
Mattawoman Creek	2003	145	20
Northeast River	2007	121	21
Piscataway Creek	2012	121	22
Piscataway Creek	2009	107	23
Northeast River	2017	107	24
Northeast River	2016	97	25
Mattawoman Creek	2010	81	26
Mattawoman Creek	2006	75	27
Mattawoman Creek	2012	72	28
Northeast River	2022	59	29
Mattawoman Creek	2007	57	30
Piscataway Creek	2003	45	31
Mattawoman Creek	2022	28	32
Piscataway Creek	2006	25	33
Sassafras River	2020	25	34
Mattawoman Creek	2008	21	35
Sassafras River	2021	13	36
Mattawoman Creek	2009	5	37
Piscataway Creek	2007	5	38

Table 3-16. Annual modified proportional stock density (PSD) of White Perch for subestuaries sampled in 2022. Number of N_{TOTAL} is the total number of White Perch (all juveniles and adults) in 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	Years	N_{TOTAL}	$N L_{STOCK}$	$N L_{QUALITY}$	Modified PSD (%)
Mattawoman Creek	2003	3661	382	0	0
	2004	2791	355	2	0.6
	2005	3916	471	1	0.2
	2006	1978	567	0	0
	2007	1385	442	1	0.2
	2008	716	366	1	0.3
	2009	299	58	0	0
	2010	2375	382	0	0
	2011	2860	239	4	1.7
	2012	2049	211	0	0
	2013	5243	94	1	1.1
	2014	9122	84	0	0
	2015	4411	420	0	0
	2016	3690	697	0	0
2022	2223	109	0	0	
Northeast River	2007	2961	1222	15	1.2
	2008	2967	1468	11	0.7
	2009	4681	2841	15	0.5
	2010	7929	2104	16	0.8
	2011	6694	1487	24	1.6
	2012	6699	4962	63	1.3
	2013	4781	3875	13	0.3
	2014	6929	2254	5	0.2
	2015	3828	2247	4	0.2
	2016	2073	968	2	0.2
	2017	3123	557	4	0.7
2022	1708	1571	5	0.3	
South River	2003	1508	28	17	60.7
	2004	165	165	0	0.0
	2005	34	33	5	15.2
	2022	19	17	14	82.4
Tred Avon River	2006	364	360	45	12.5
	2007	404	375	22	5.9
	2008	234	234	31	13.2
	2009	120	120	30	25.0
	2010	21	15	6	40.0
	2011	809	76	17	22.4
	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	111	94	49	52.1
2019	554	553	147	26.6	
2020	165	165	56	33.9	
2021	52	52	11	21.2	
2022	104	104	22	21.2	

Figures

Figure 3-1. Map illustrating subestuaries sampled in summer 2022: Northeast River (1), Tred Avon River (2), South River (3), Mattawoman Creek (4), and their land use categories. Land use data is based on Maryland Department of Planning (DOP) 2010 land use land cover data. Stars indicate previously sampled subestuaries mentioned throughout the report.

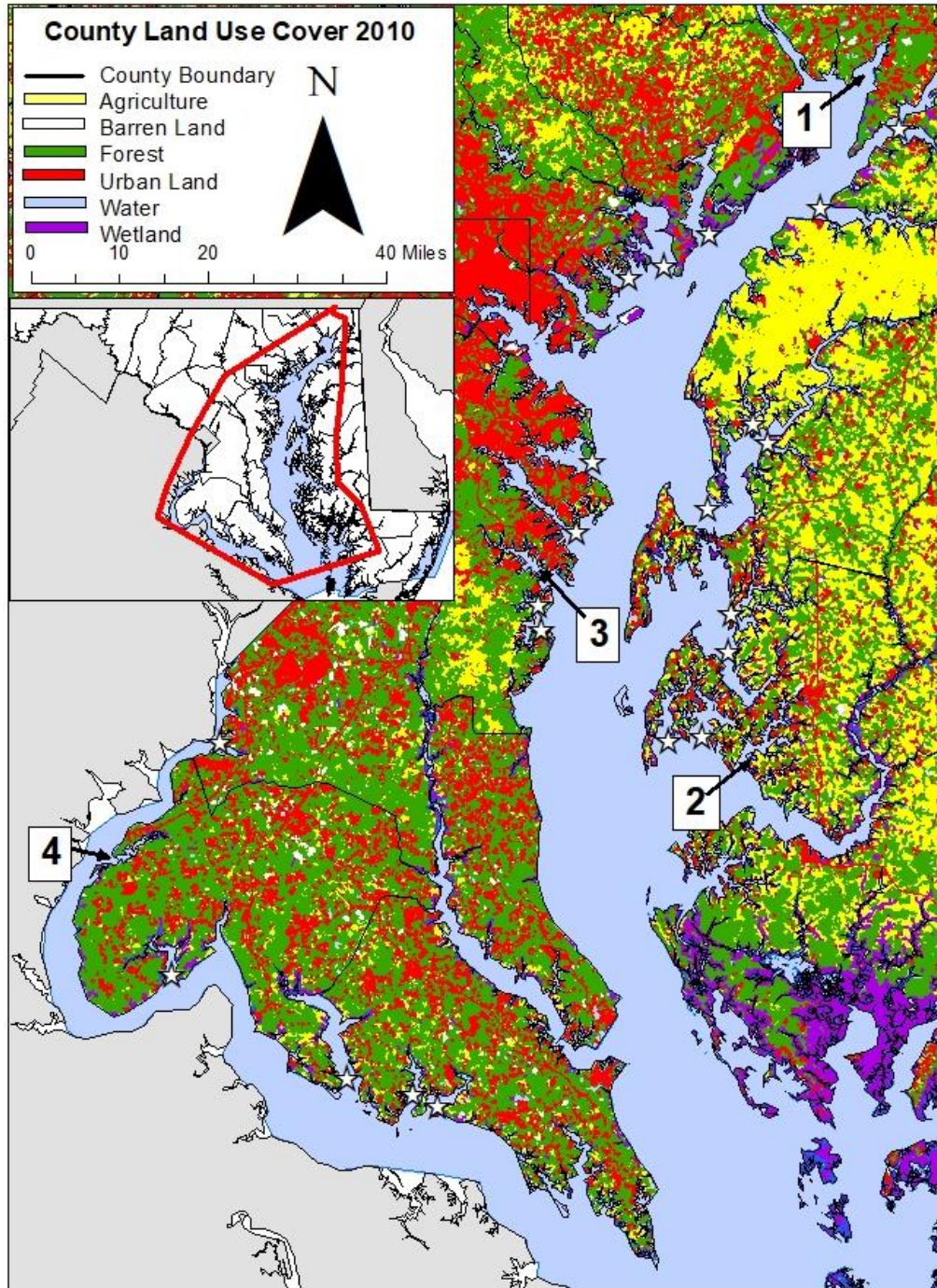


Figure 3-2. Map indicating locations of sampling stations sampled in 2022 located within Mattawoman Creek, Northeast River, South River, and Tred Avon River.

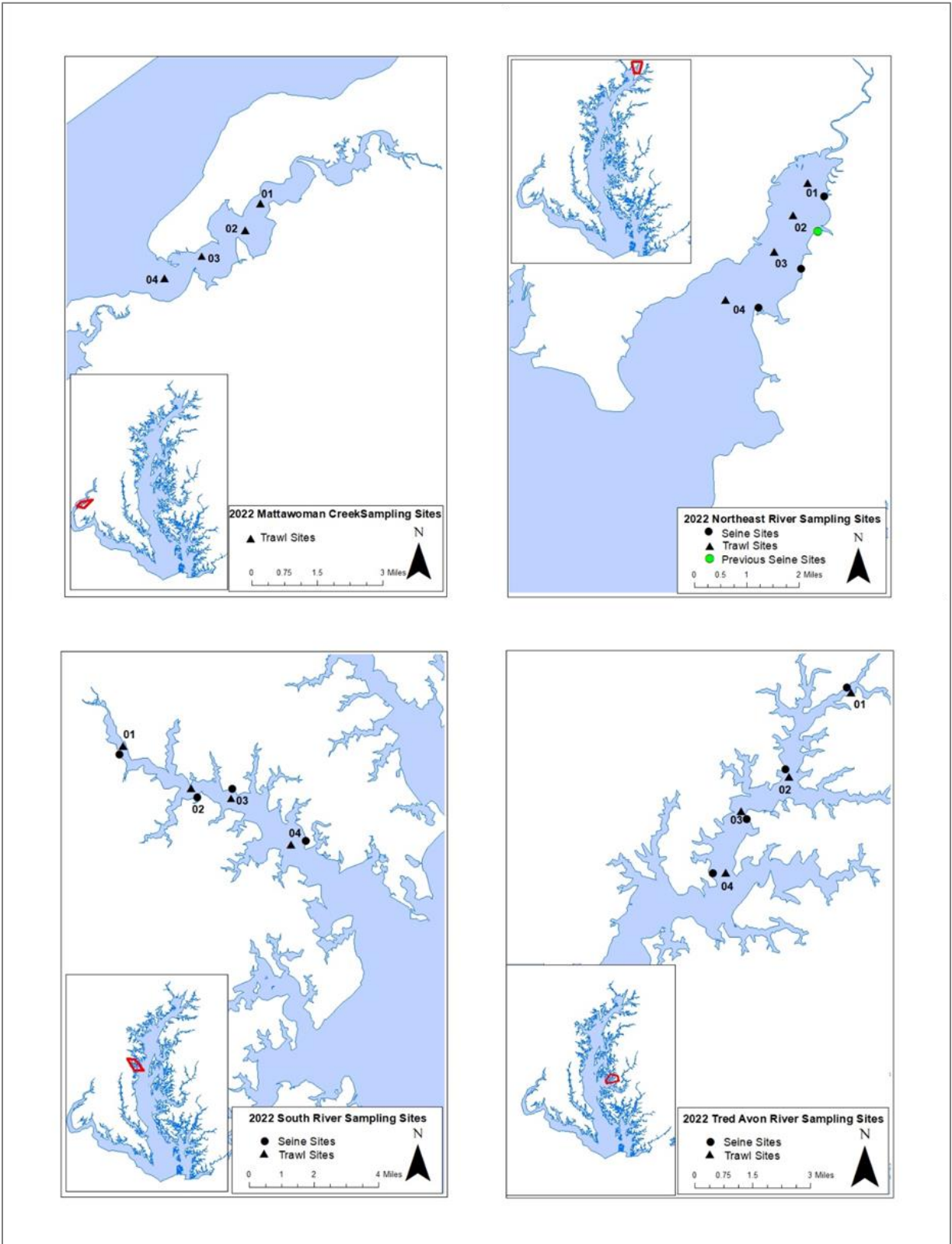


Figure 3-3. Mean subestuary bottom dissolved oxygen during summer (July – October) sampling, 2003 – 2022, plotted against level of development (C / ha or structures per hectare) and target and threshold dissolved oxygen.

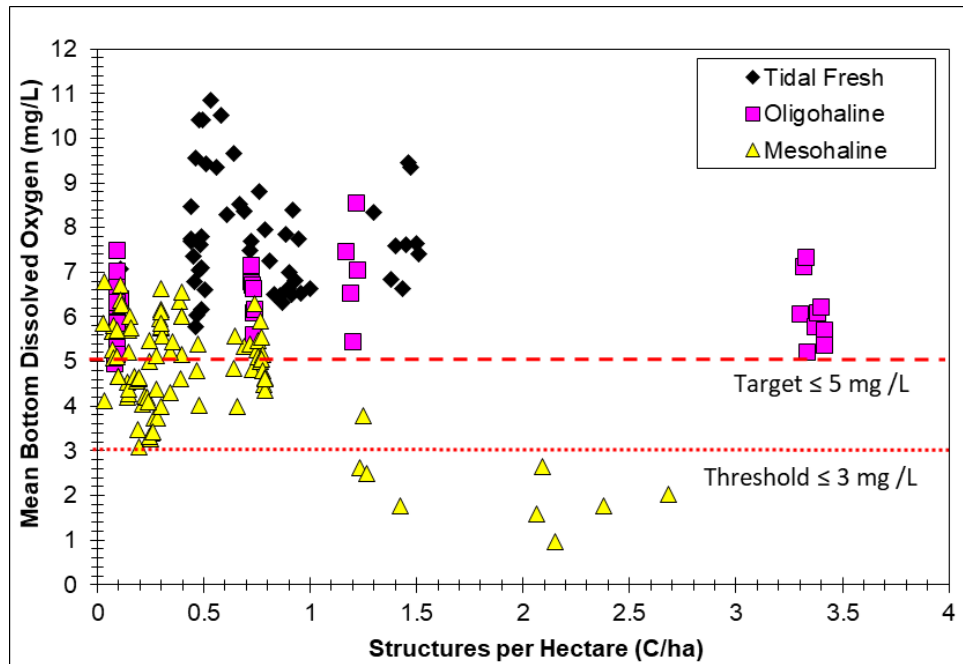


Figure 3-4. Mean subestuary surface dissolved oxygen during summer (July – October) sampling, 2003 – 2022, plotted against level of development (C / ha or structures per hectare) and target and threshold dissolved oxygen.

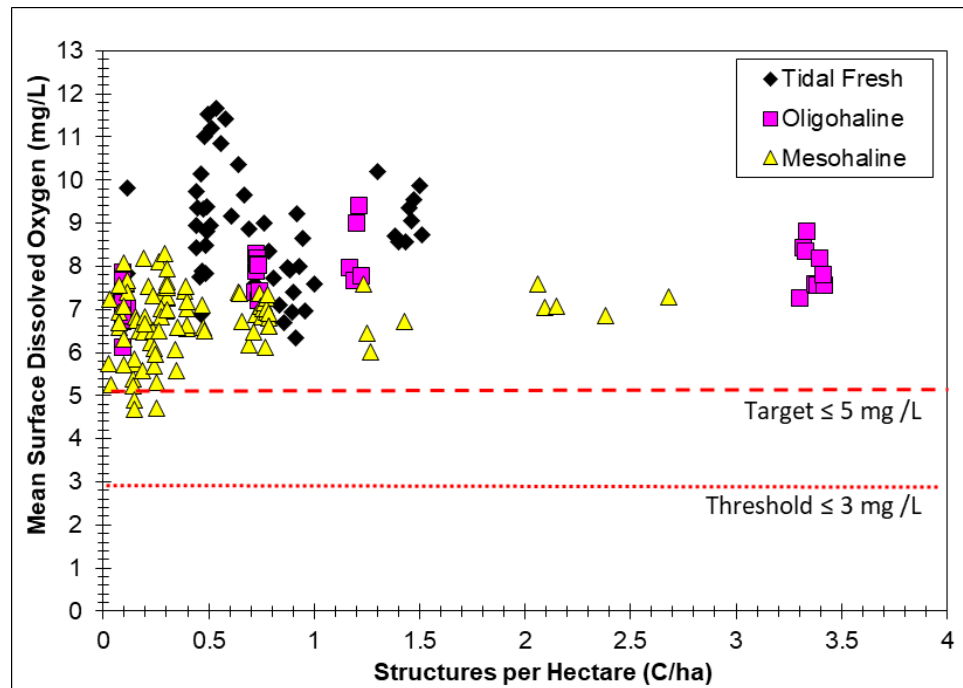


Figure 3-5. Maryland Department of Planning estimates of agricultural land coverage (% watershed land area) by region (**western** or **eastern** shore) versus median bottom dissolved oxygen (DO) in mesohaline subestuaries within major drainages (2003–2022). Quadratic model predicts median bottom DO and agricultural coverage (%) using data from both regions.

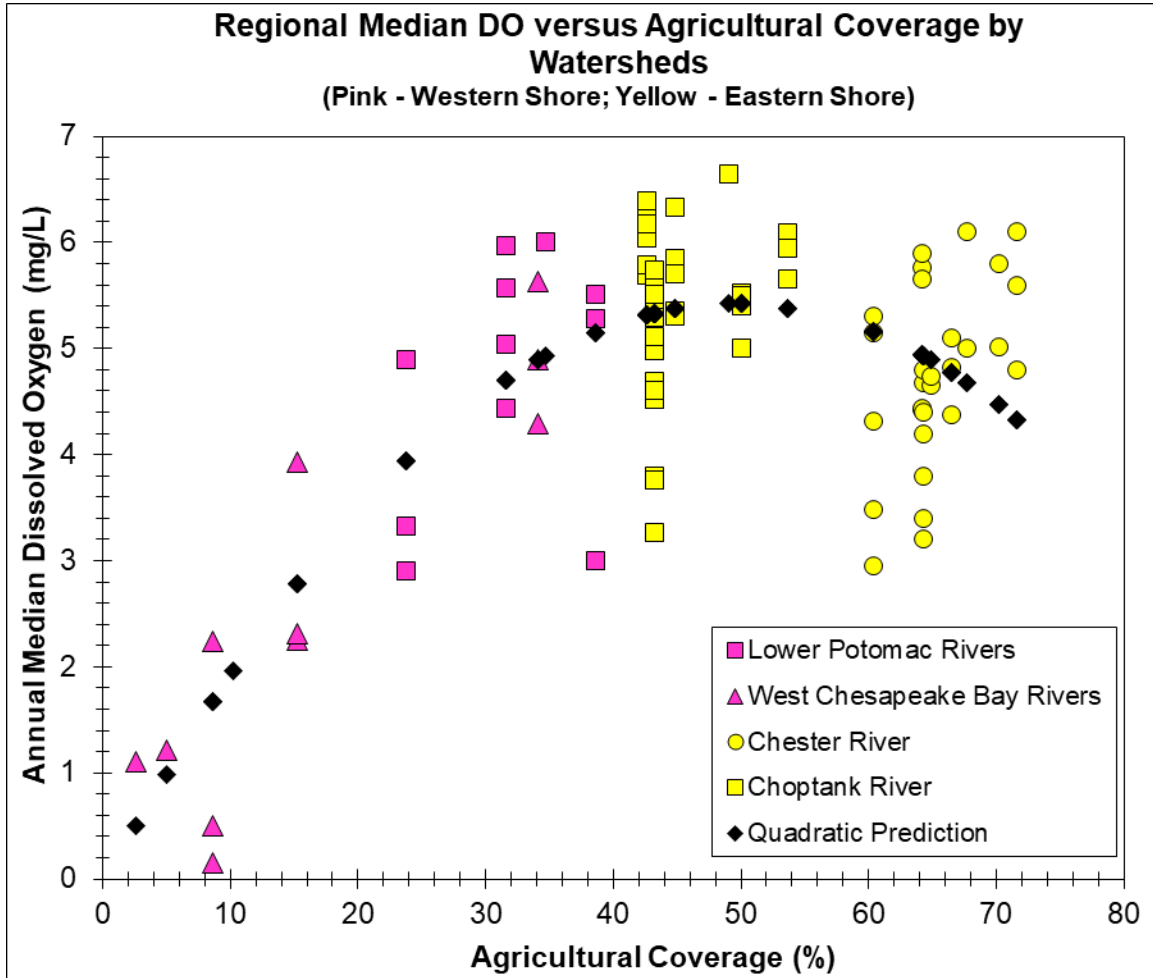


Figure 3-6. Trends in development (structures per hectare = C / ha) from 1950 to 2022 of watersheds of subestuaries sampled in 2022. Black diamond markers indicate the years that subestuaries were sampled. Development data was not available for 2021 and 2022; 2020 data was used for those years.

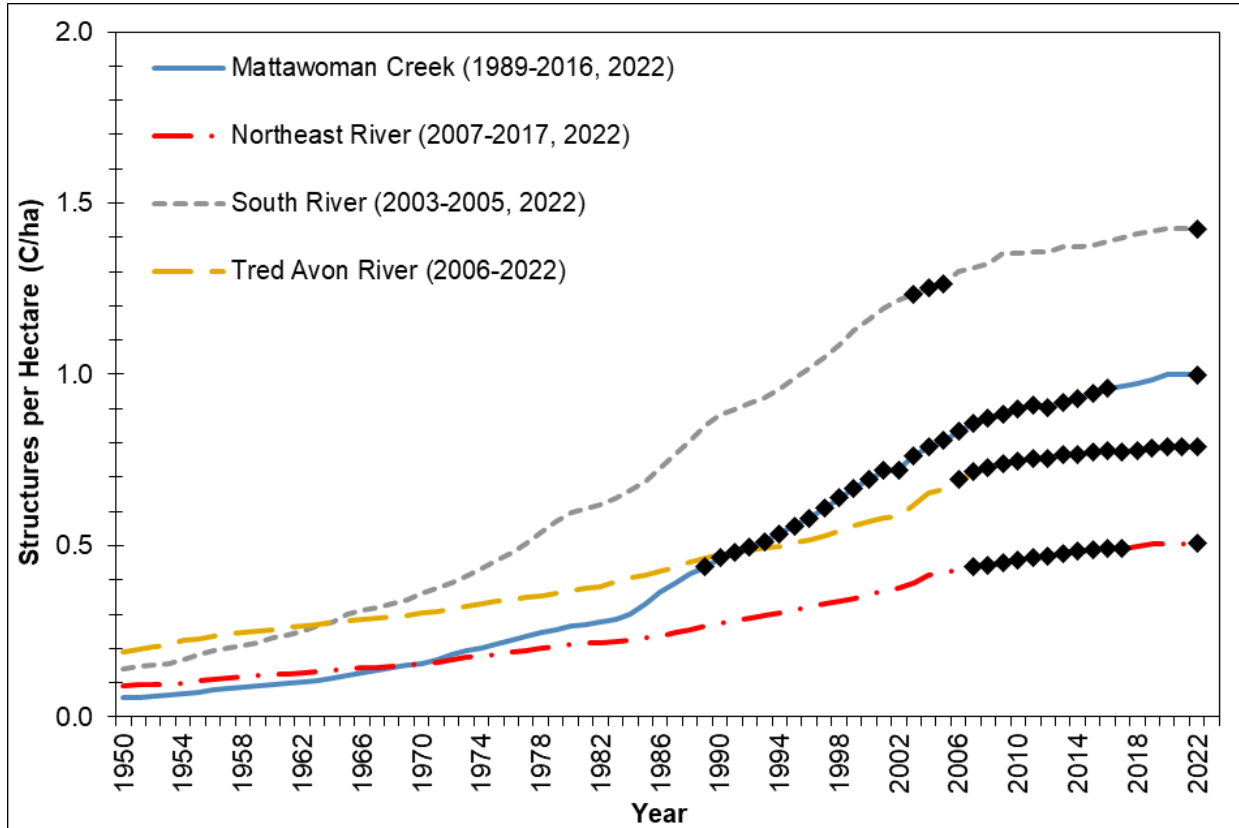


Figure 3-7. Bottom dissolved oxygen (DO; mg / L) readings (2003 – 2022) for subestuaries sampled in 2022 versus intensity of development (C / ha = structures per hectare). Target (5 mg / L) and threshold (3 mg / L) DO boundaries are indicated by red dashed lines.

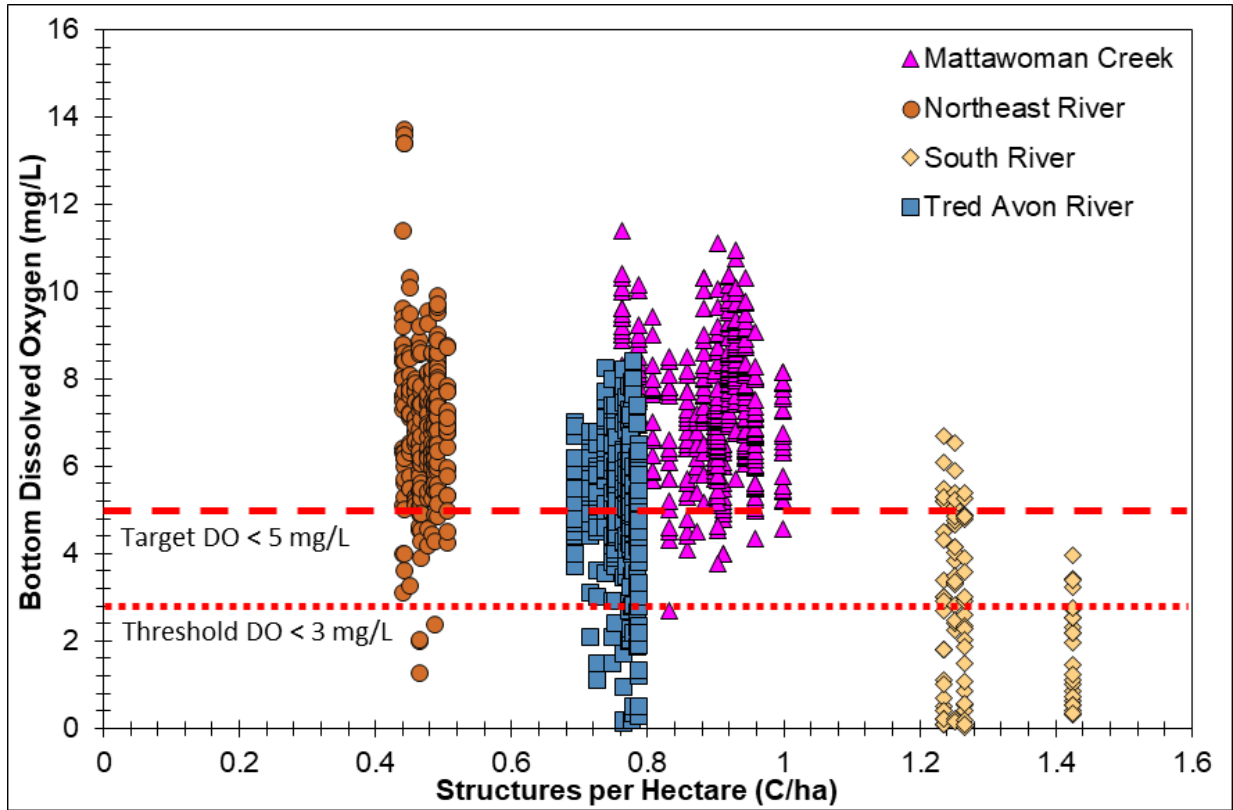


Figure 3-8. Summer (July – September) median bottom dissolved oxygen (DO; red squares; mg/L) for Chesapeake Bay Program Stations MAT0016 (Mattawoman Creek), ET1.1 (Northeast River), CB4.1E (Miles River), WT8.1 (South River), and EE2.1 (mainstem Choptank River) from 1989 to 2023. Solid black bars indicate range of bottom DO measurements for each year. Grey dashed line indicates time-series median. The y-axis ranges from 0 to 13 mg/L; x-axis ranges are years from 1988 to 2022.

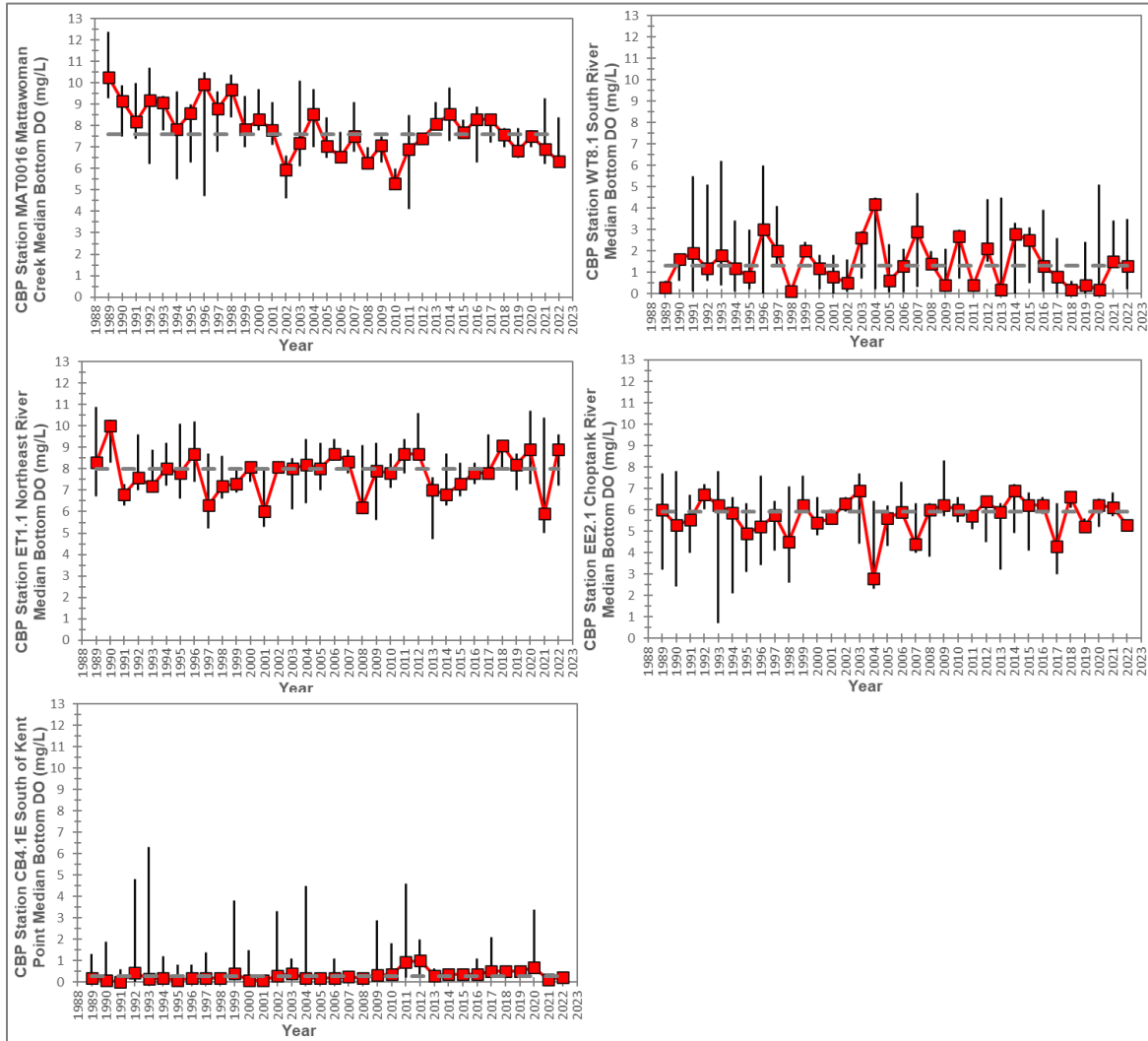


Figure 3-9. Median bottom dissolved oxygen (DO; red squares; mg / L) time-series for subestuaries sampled in 2022. Solid black bars indicate range of all bottom DO measurements for that year. The y-axes range from 0 to 16 mg / L; x-axes range are years from 2002 to 2023.

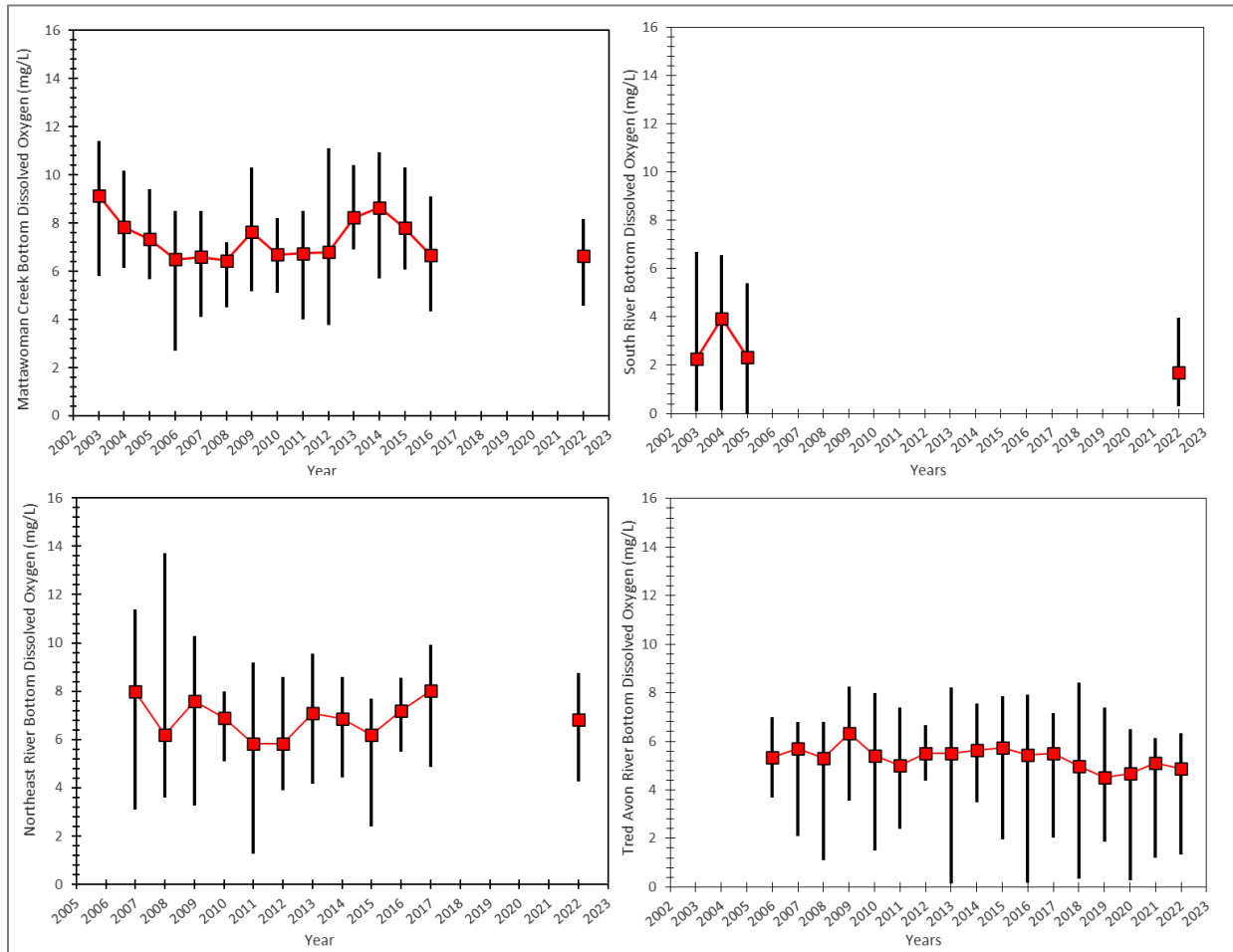


Figure 3-10. Mean bottom dissolved oxygen (DO; mg / L) time-series, by station, for subestuaries sampled in 2022. Dotted line indicates the median of all DO measurement data for the time-series. The y-axis range from 0 to 10 mg / L; x-axis range are years from 2002 to 2023.

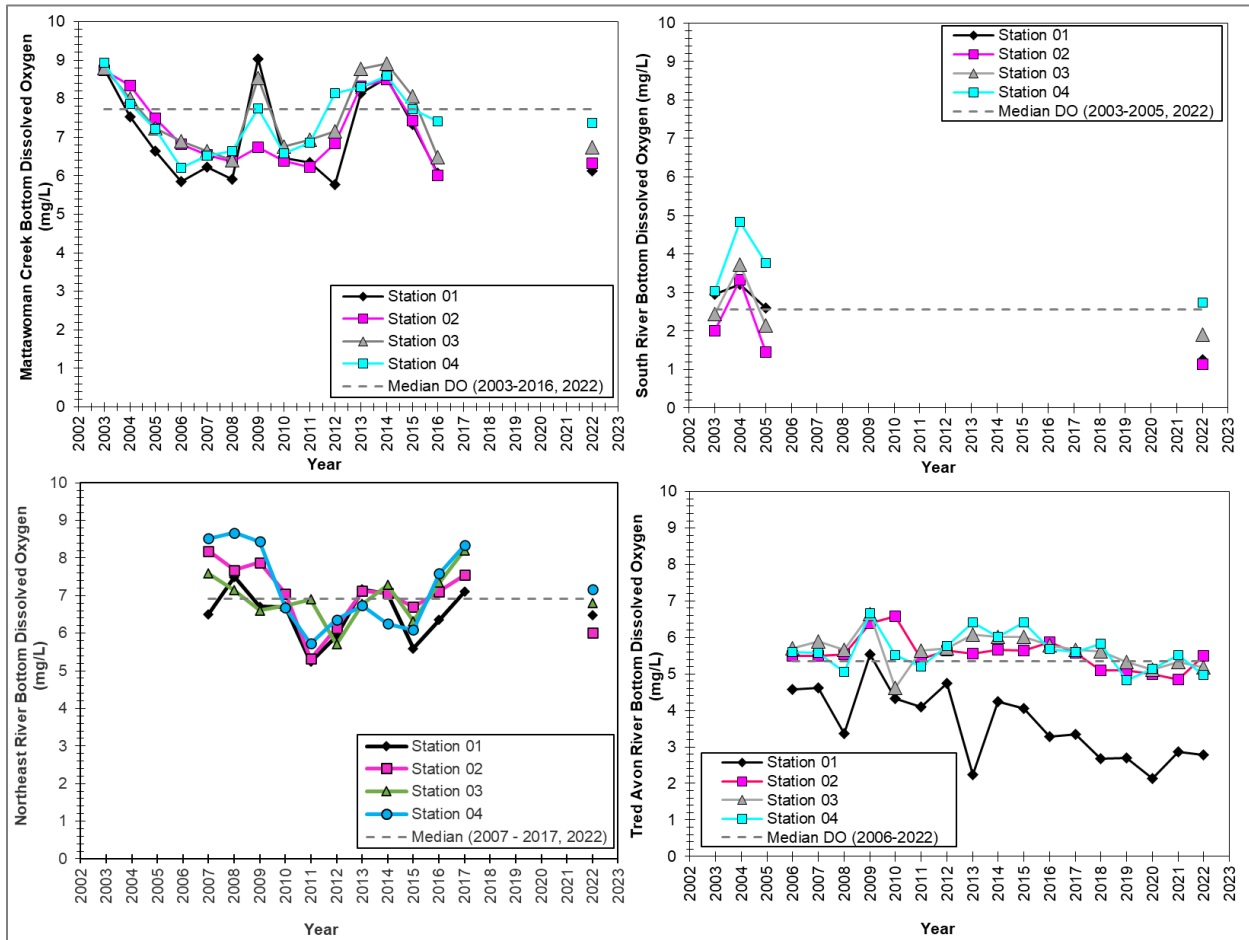


Figure 3-11. Median Secchi depth (m) time-series for subestuaries sampled in 2022. Solid black bars indicate the range of Secchi depth (m) measurements by year. The y-axes range from 0 to 2.5 m; x-axes range are years from 2002 to 2023.

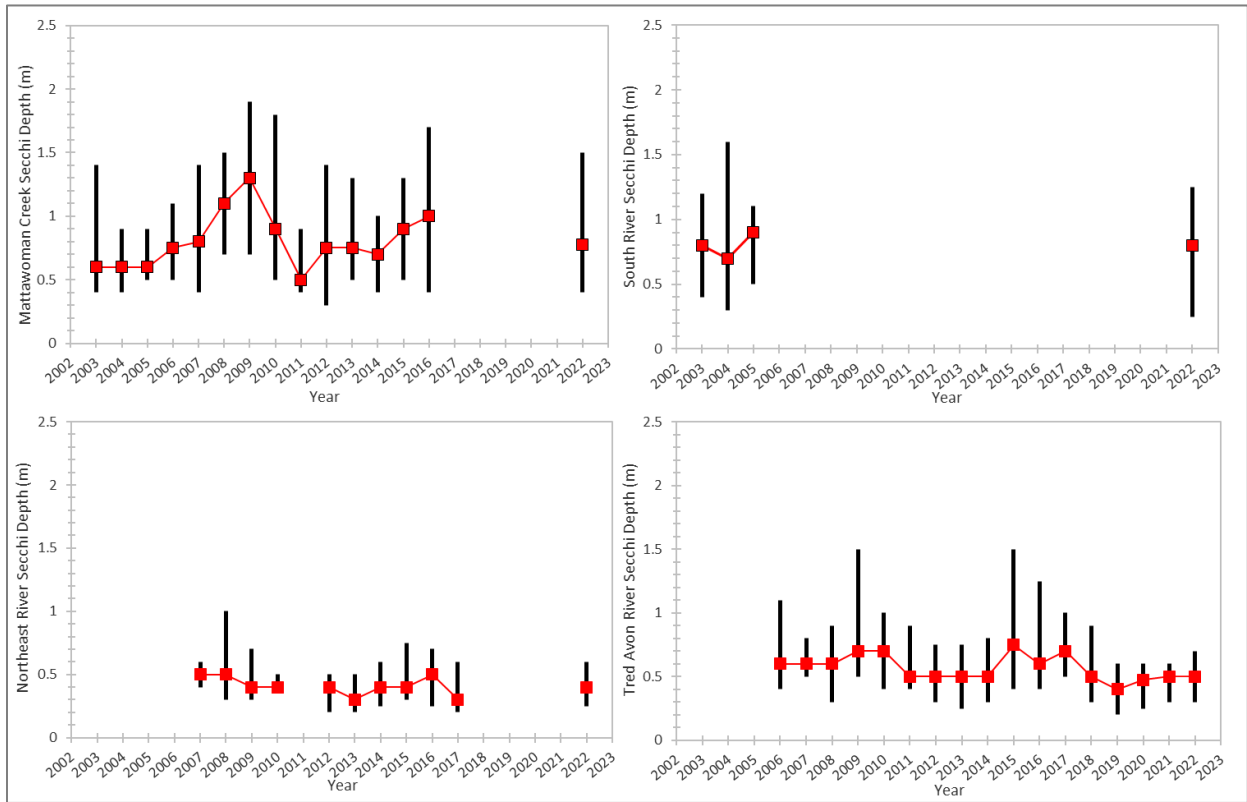


Figure 3-12. Time-series of coverage of SAV (percent of water area) for subestuaries sampled in 2022. Mouth of the Choptank River includes Broad Creek, Harris Creek, and Tred Avon River. during 1989 – 2021. Median of fully mapped years (filled triangles) for the time-series is indicated by the dashed line. Data for 2022 was not available at the time of this report.

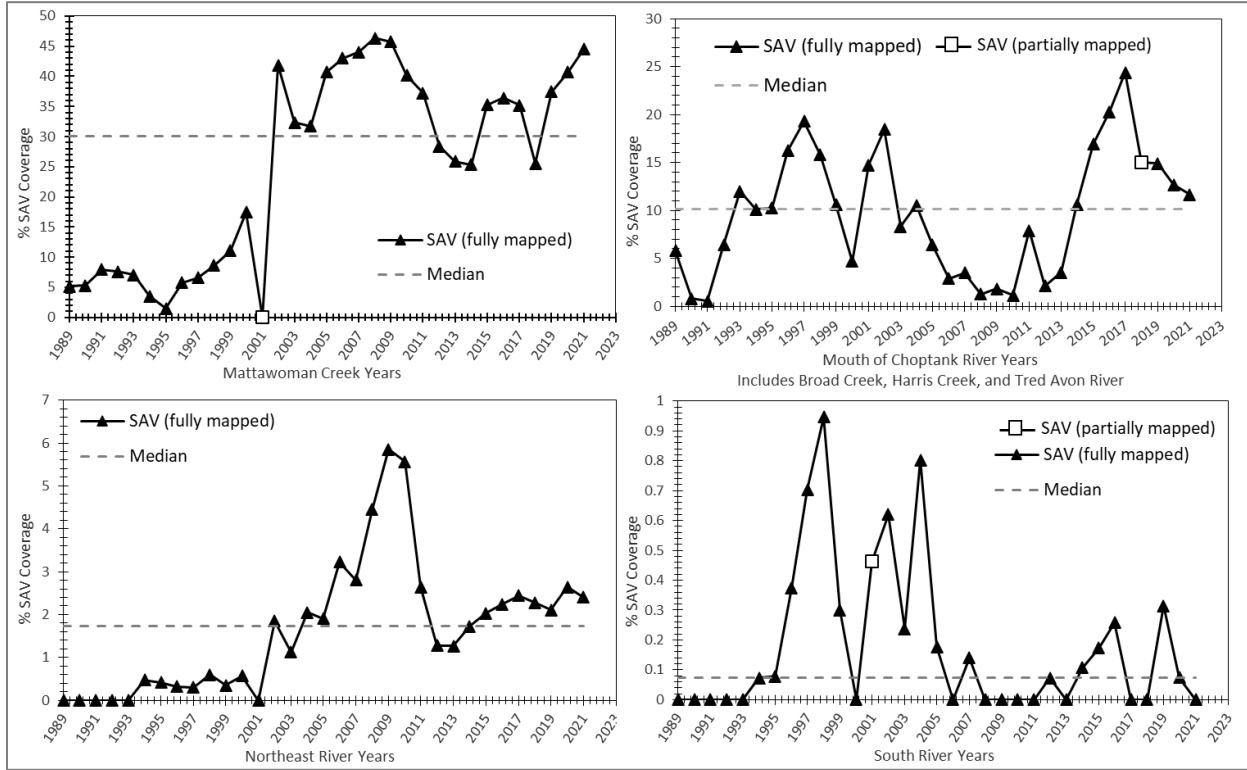


Figure 3-13. Median bottom pH (red squares) by year for subestuaries sampled in 2022. Measurements of pH were not made prior to 2006.

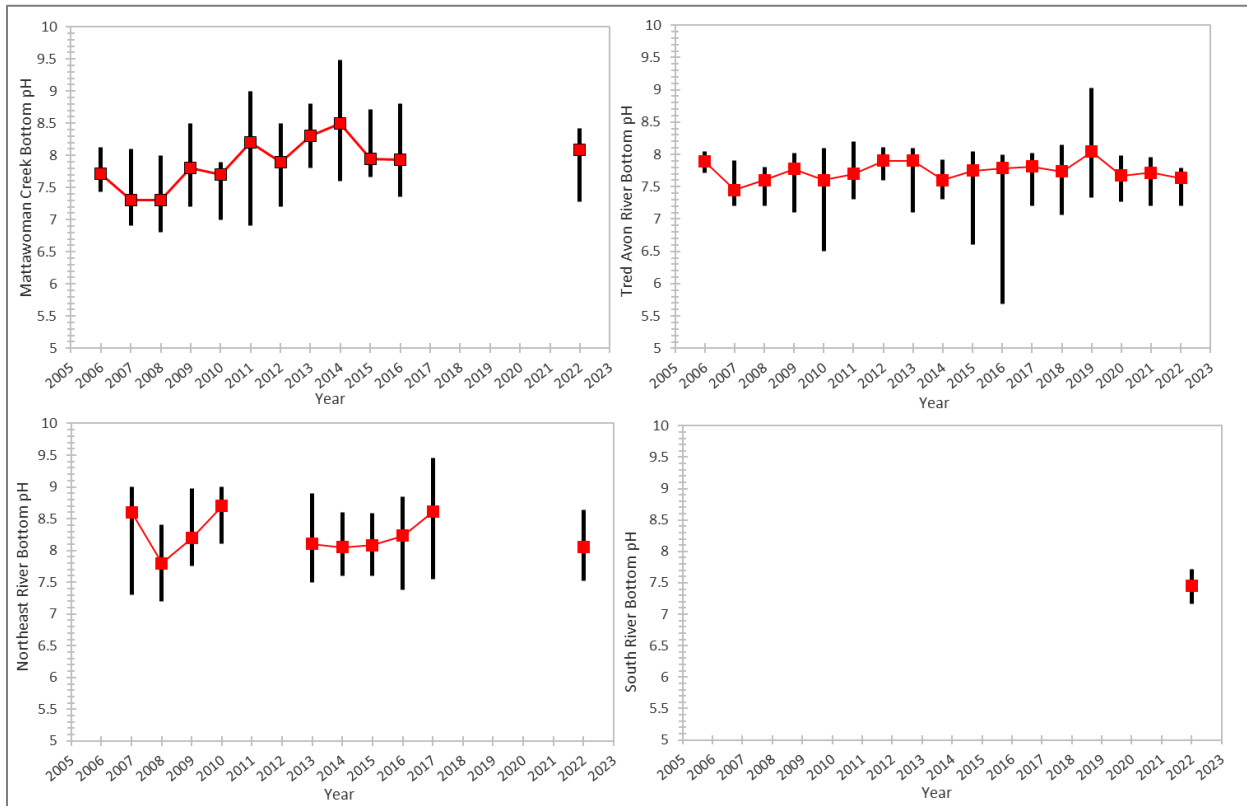


Figure 3-14. Median bottom salinity (red squares; ppt = ‰) time-series for subestuaries sampled in 2022. Solid black bars indicate the range of salinity measurements by year.

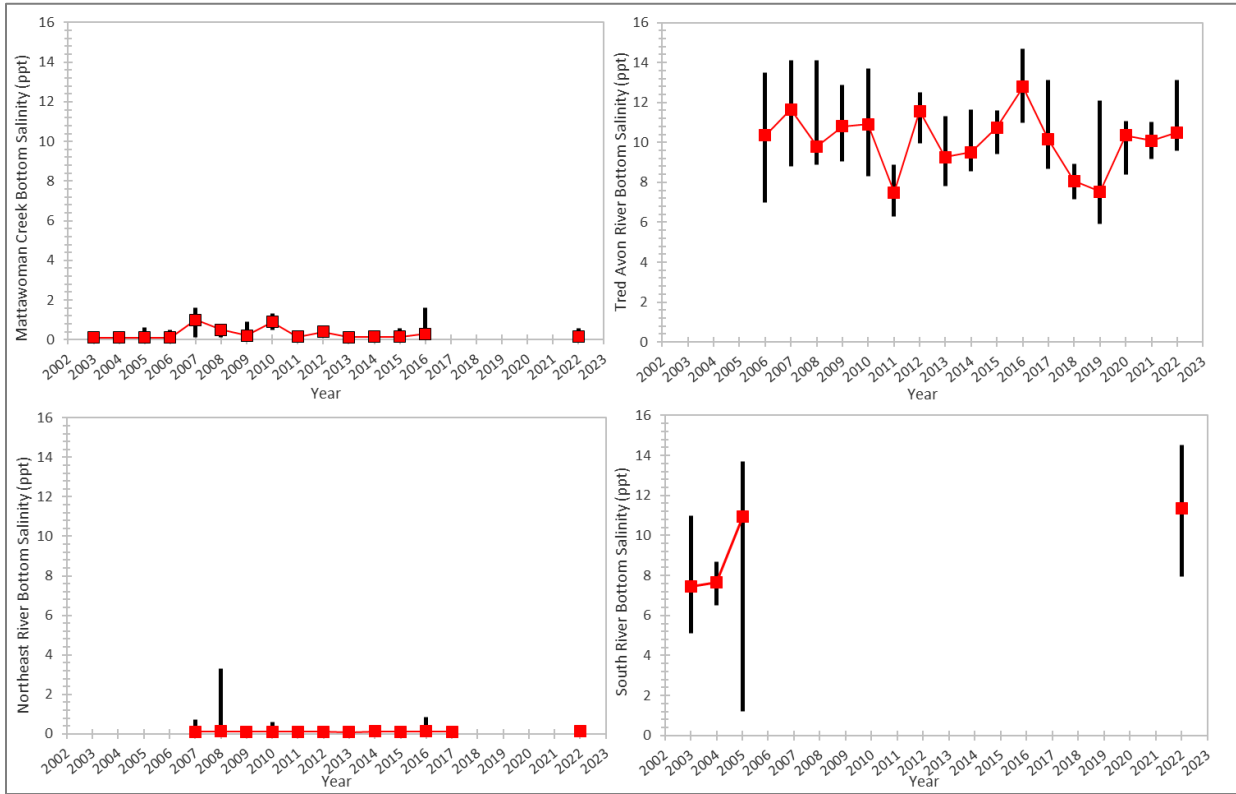


Figure 3-15. Annual number of finfish species (richness) collected by beach seines in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of watershed development (C/ha = structures per hectare) from 2003 to 2022. Points were omitted if beach seine effort (number of samples) < 15 samples.

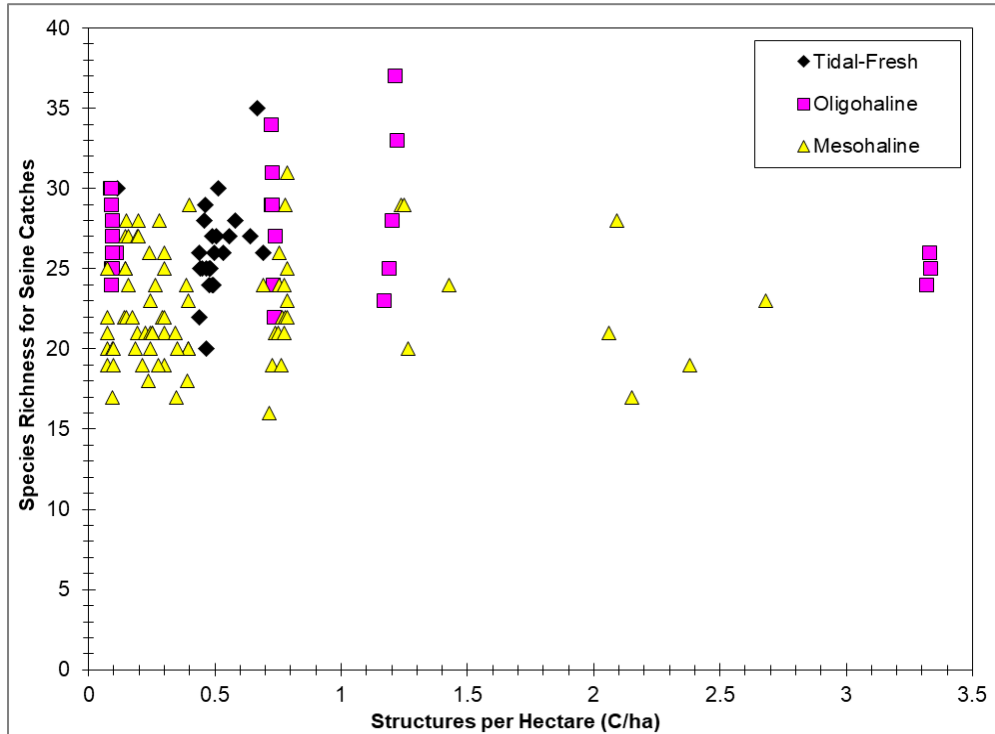


Figure 3-16. Annual 4.9m trawl geometric mean (GM) catches plotted against C/ha of subestuaries sampled during 2003 – 2022, by salinity class.

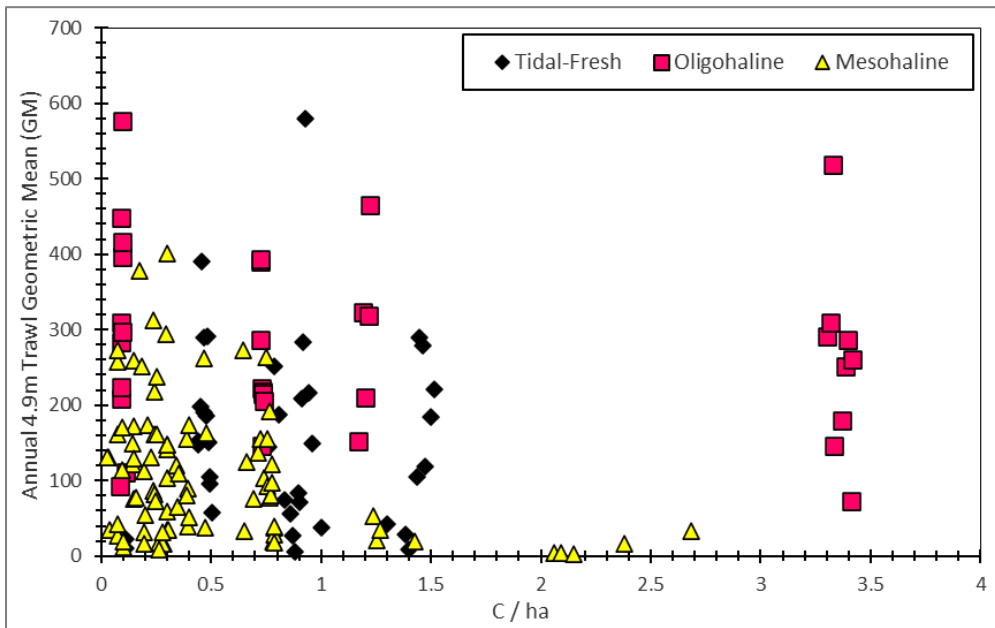


Figure 3-17. Annual number of finfish species (richness) collected by 4.9m bottom trawl in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of development (C/ha = structures per hectare) from 2003 to 2022. Points were omitted if trawl effort (number of samples) < 15 samples.

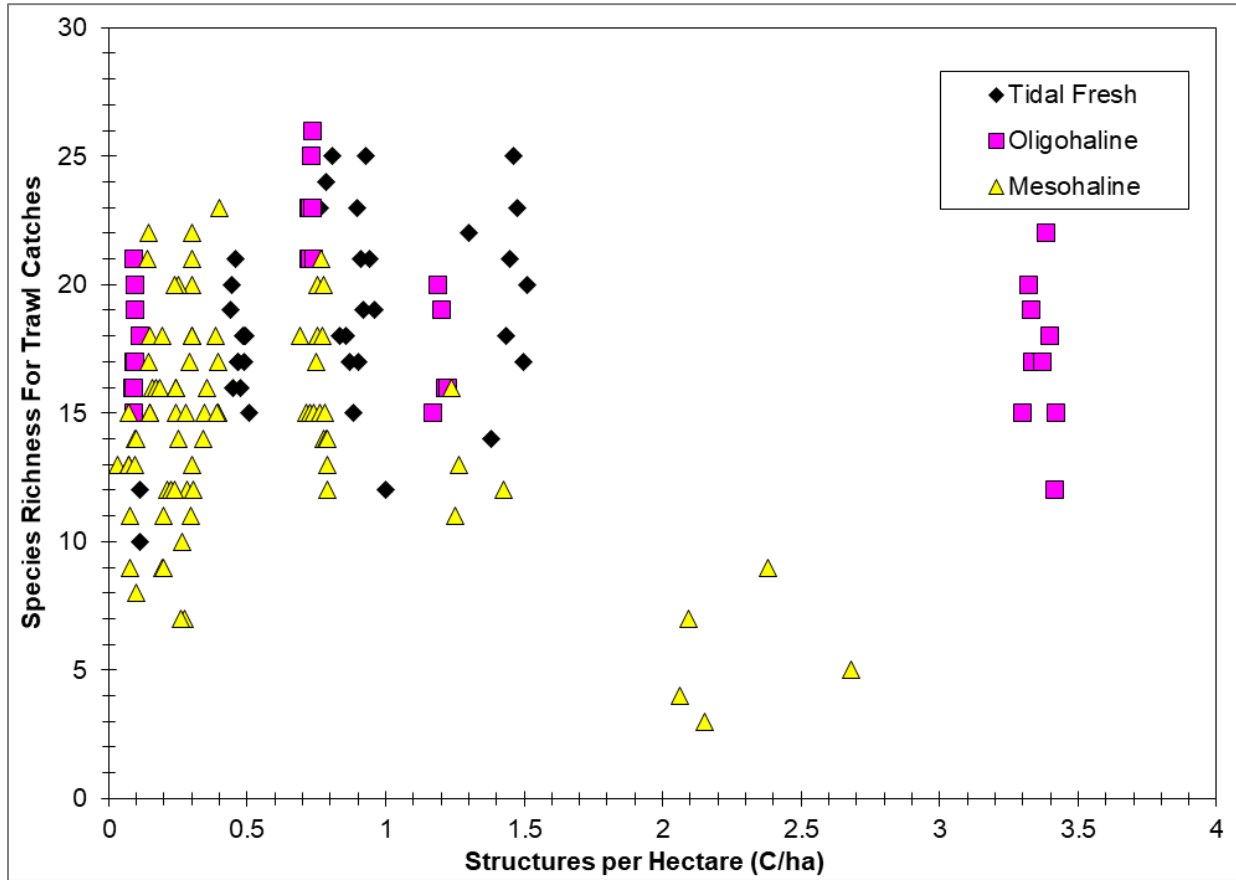


Figure 3-18. Time-series of 4.9 m bottom trawl catch geometric mean (GM) of all finfish species (red squares) for subestuaries sampled in 2022. Black bars indicate the 95% confidence intervals.

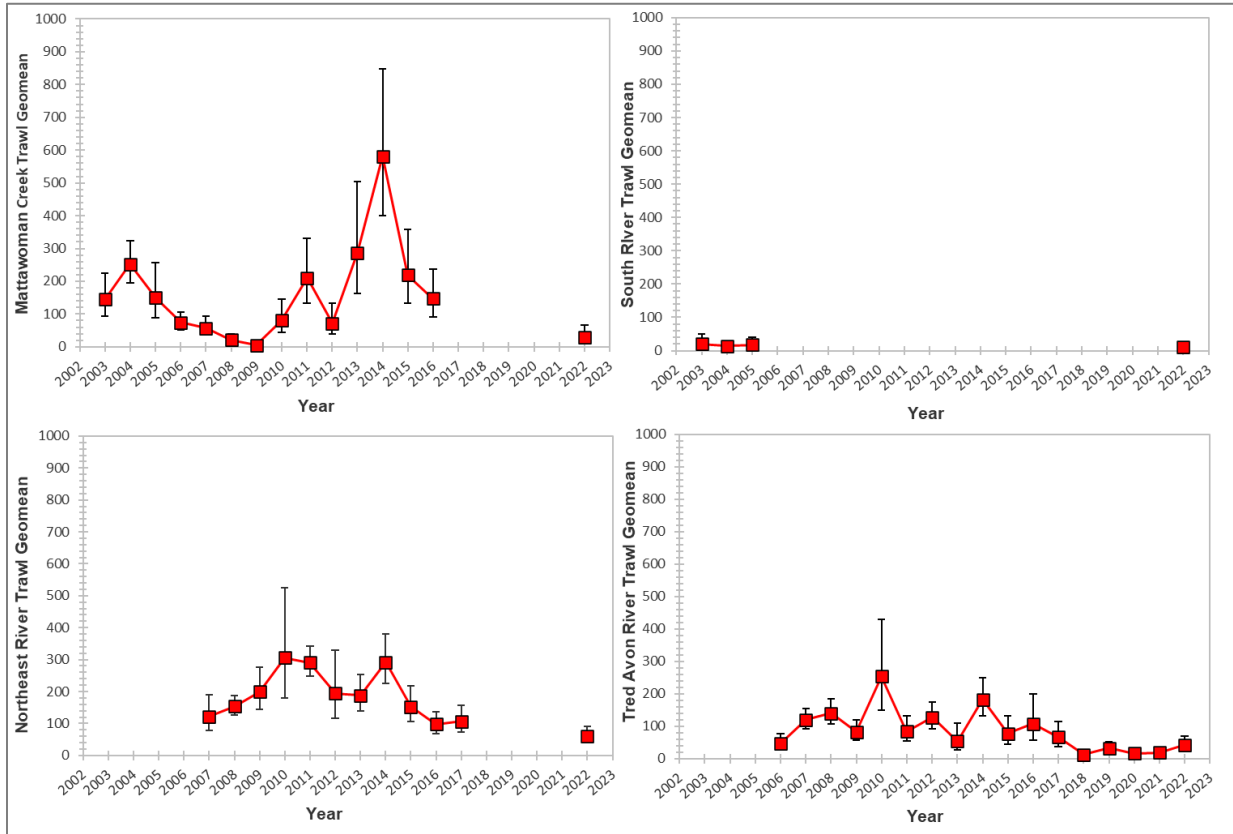


Figure 3-19. Finfish species composition for 4.9m bottom trawl catch subestuaries sampled during 2022, for all sampling years combined. Species that define the top 90% are identified, and the remainder are “other species”.

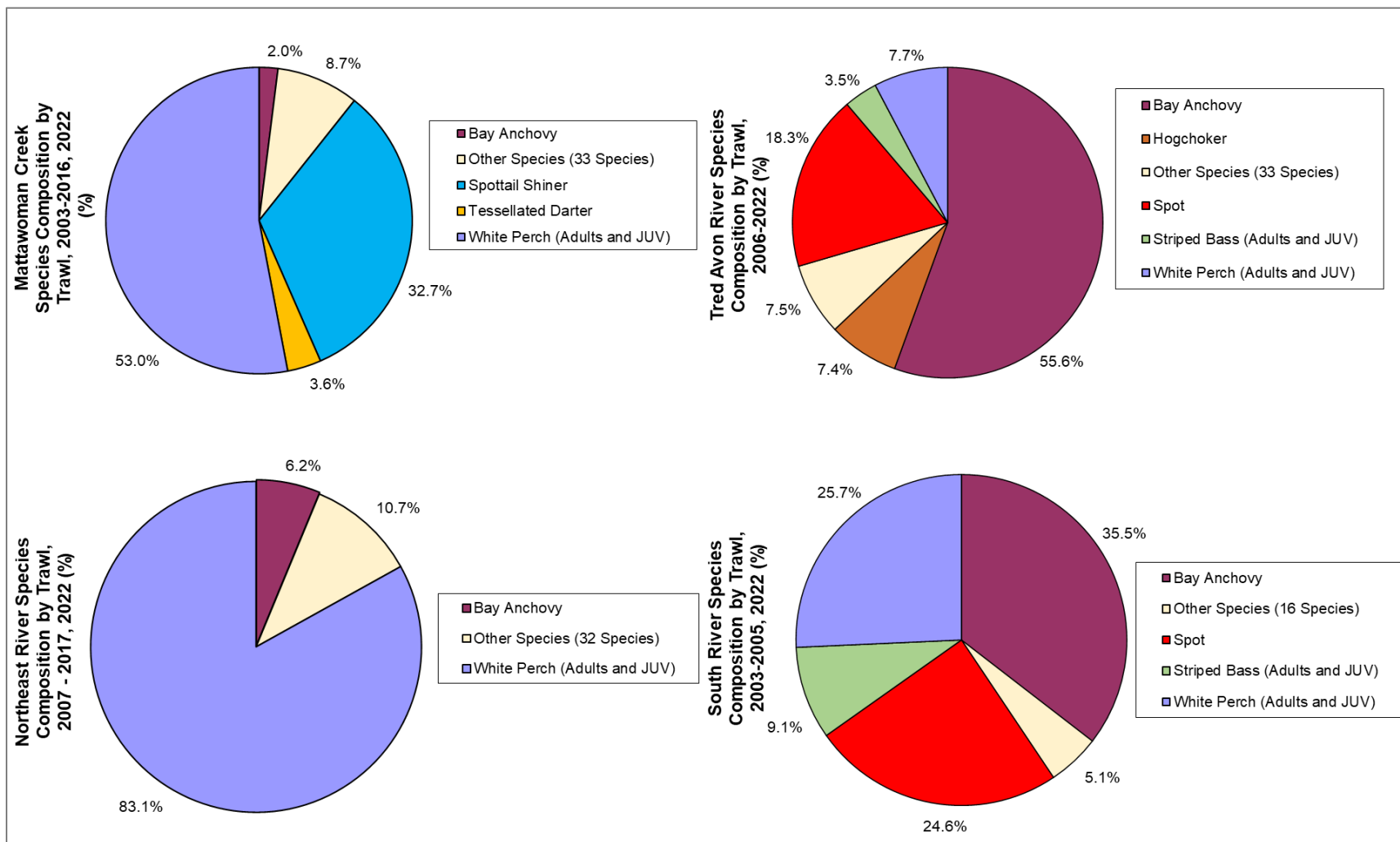


Figure 3-20. Time-series of finfish species composition for 4.9 m bottom trawl catch in subestuaries sampled during 2022,. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

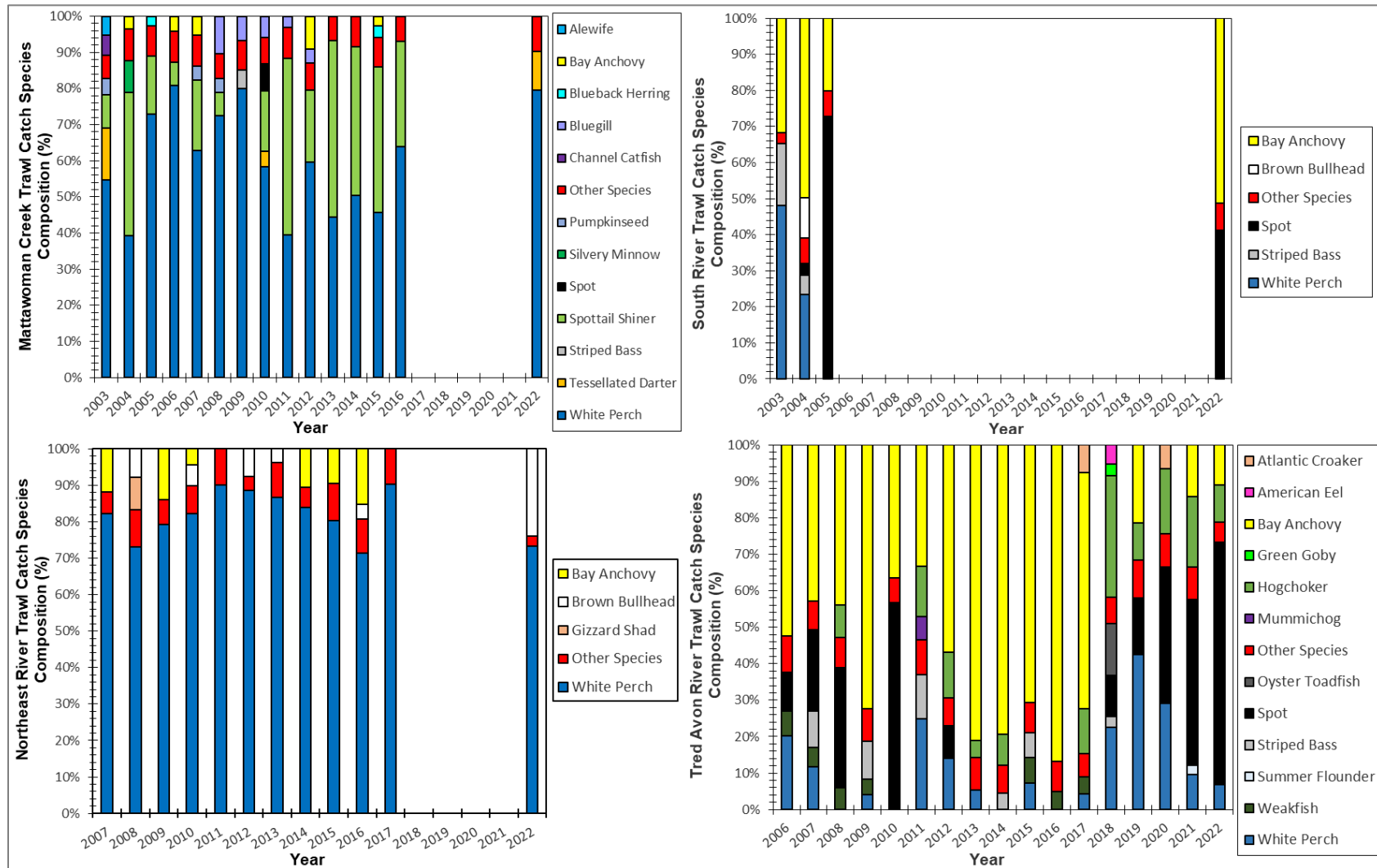


Figure 3-21. Time-series of a percent similarity index (%) for 4.9 m bottom trawl in subestuaries sampled during 2022. The greater the similarity value, the more finfish species there are in common throughout all bottom trawl stations (01 – 04).

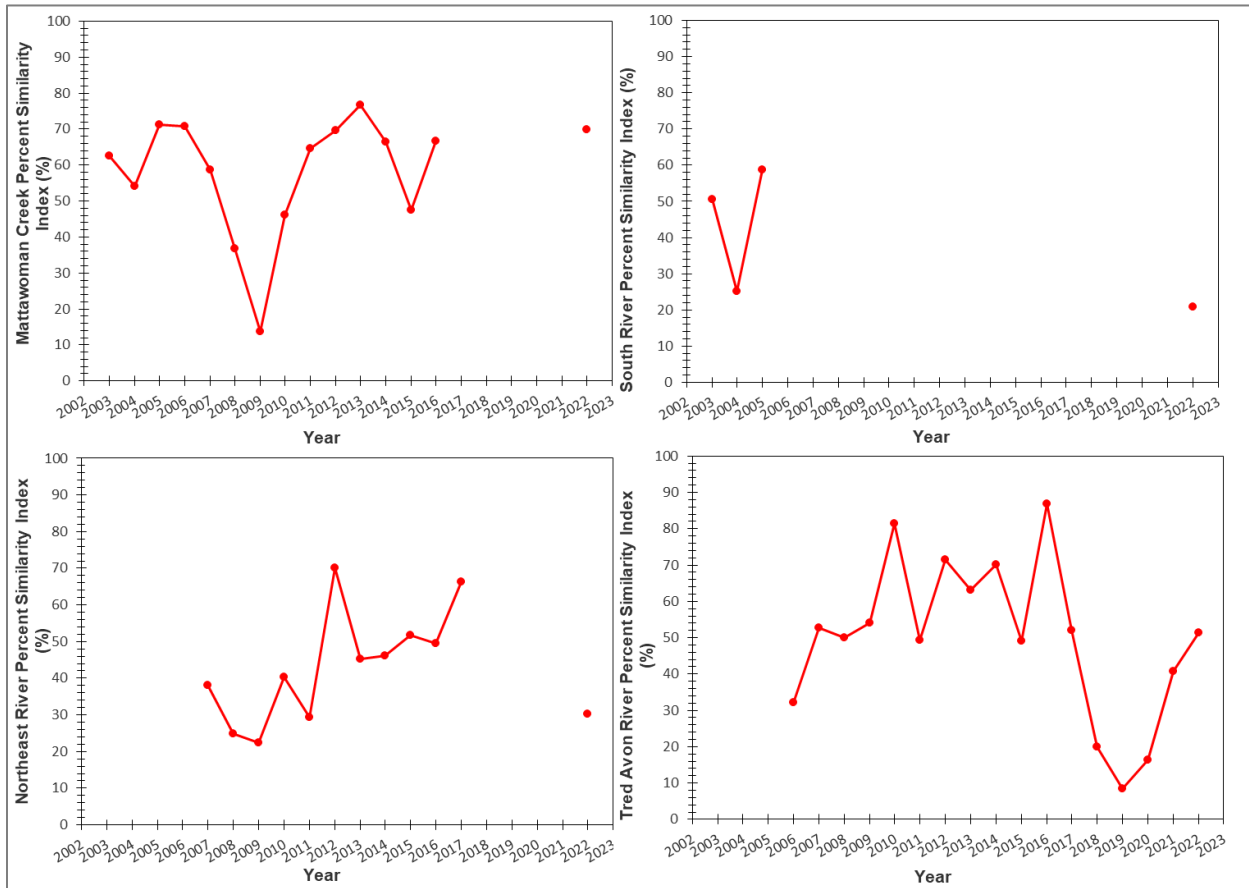


Figure 3-22. Time-series of finfish species composition for combined 4.9m bottom trawl catch in all mesohaline subestuaries sampled during 2003 – 2022, by year. Finfish species that define the top 90% are identified, and the remainder of species are “other species”.

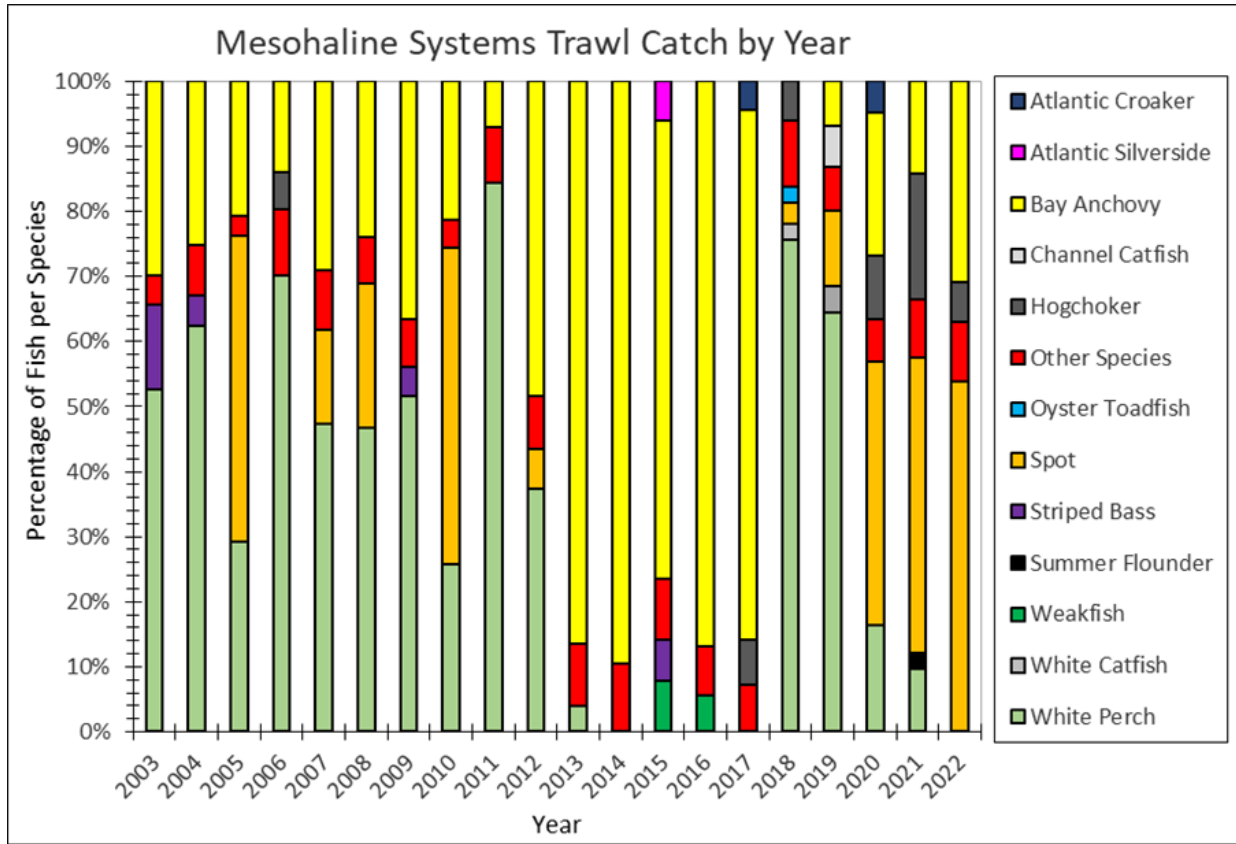


Figure 3-23. Time-series of geometric mean (GM) 4.9m bottom trawl catch of adult White Perch (primary vertical axis) and juvenile White Perch (secondary vertical axis) in subestuaries sampled during 2022. Black (Juvenile WP) and red (Adult WP) bars indicate the 95% confidence intervals.

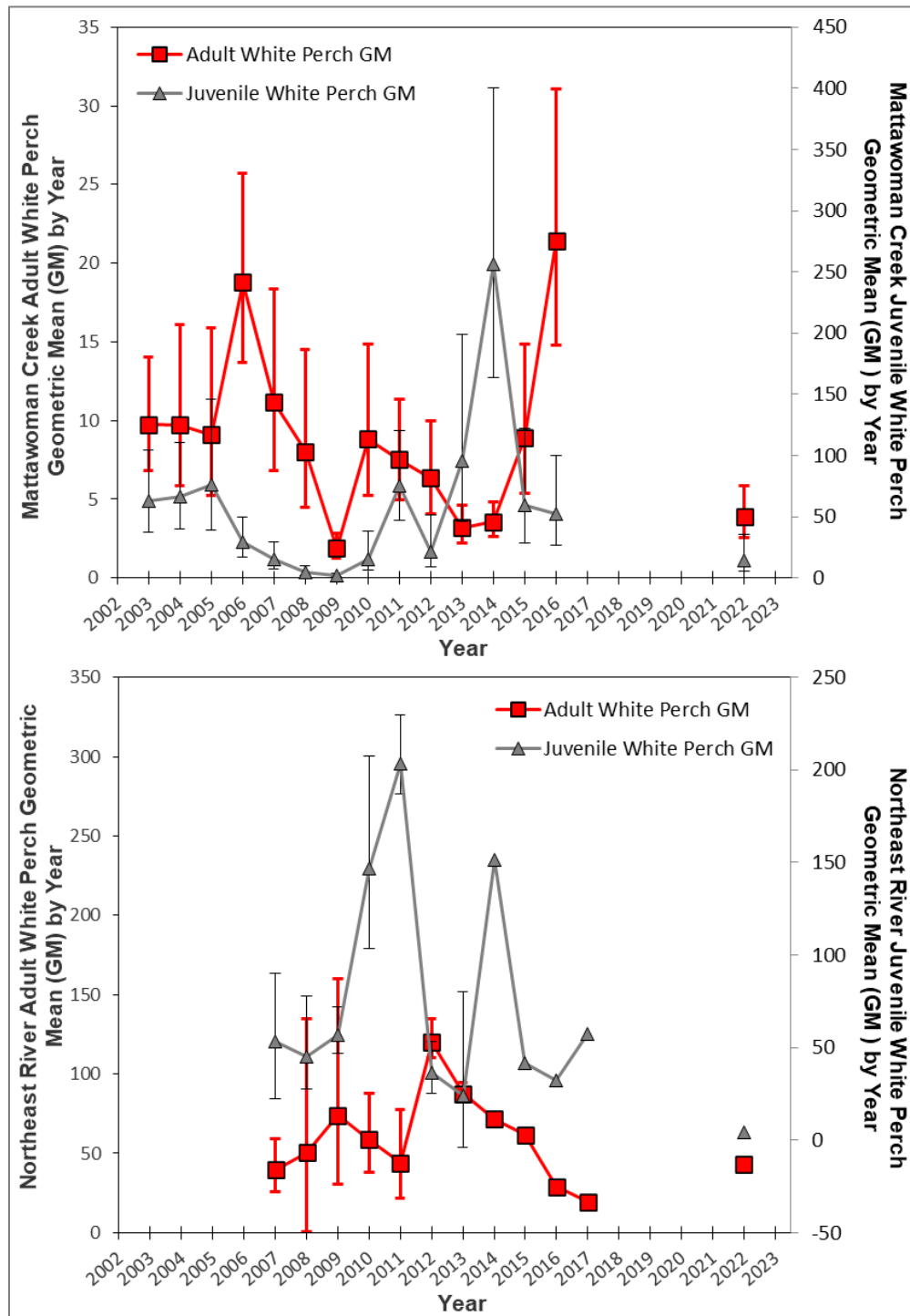


Figure 3-23. Continued.

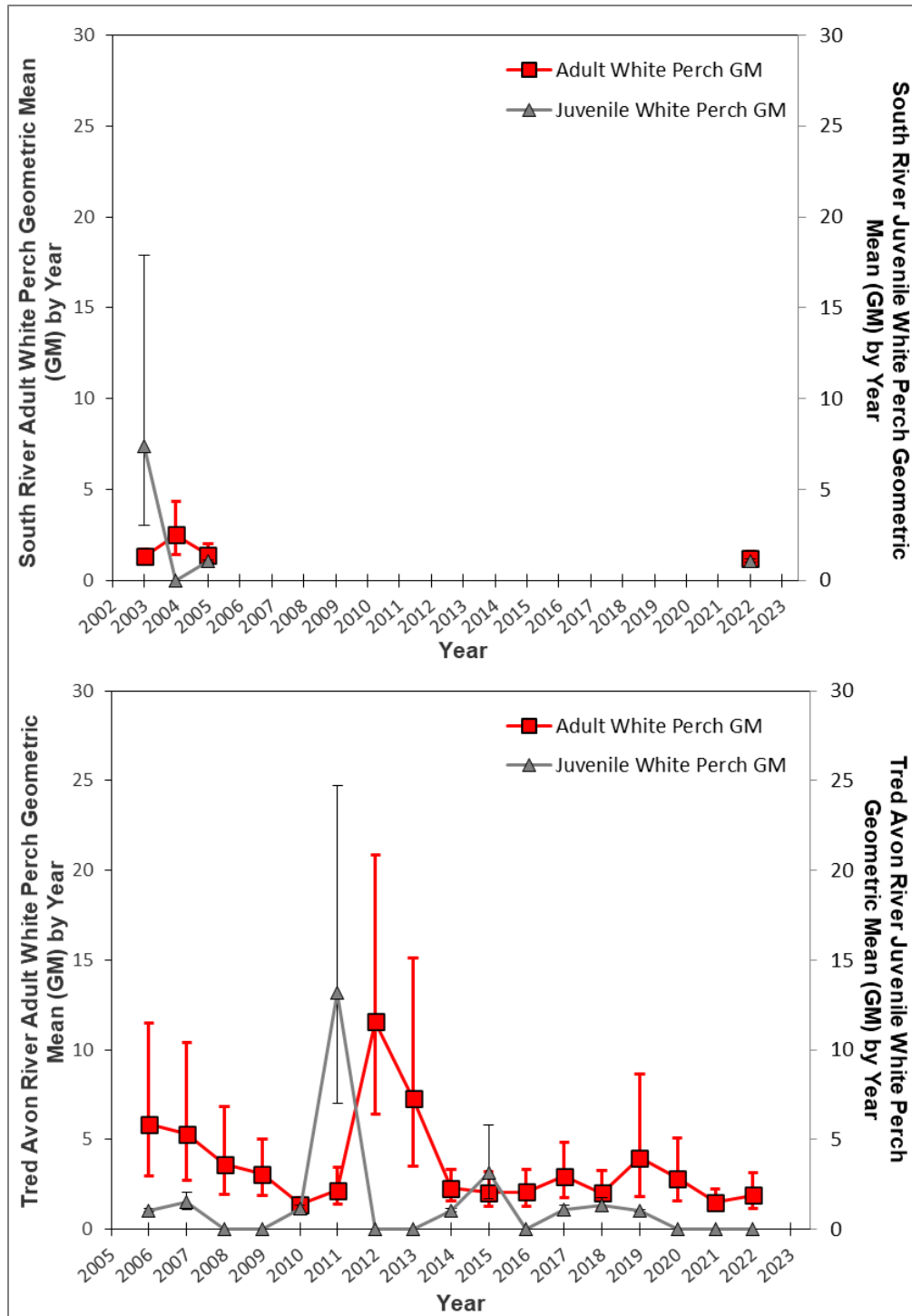


Figure 3-24. Time-series of the modified proportional stock density (PSD) of adult White Perch in subestuaries sampled during 2022.

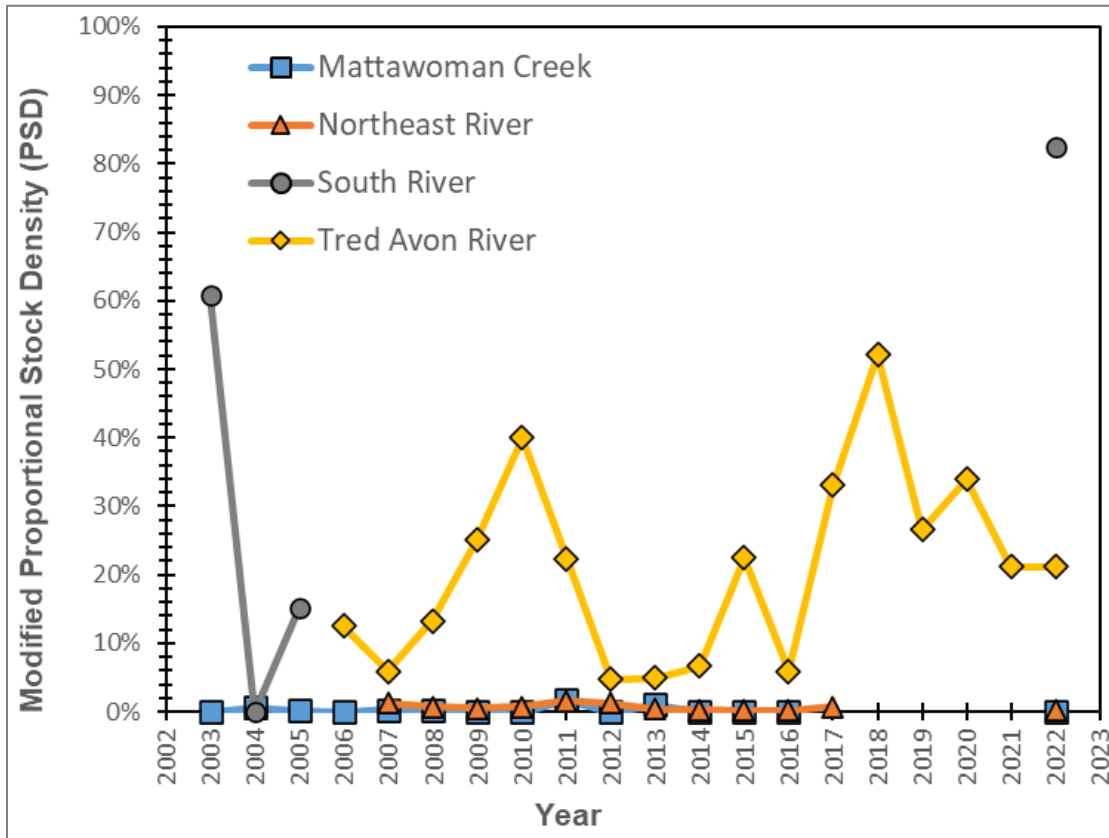


Figure 3-25. Geometric mean (GM) of bottom trawl catches of all finfish species and GM of juvenile White Perch for historical trawl 3.1 m and trawl 4.9 m in Mattawoman Creek, by sampling year. Predicted 3.1 m GM is based on a linear regression of 3.1 m and 4.9 m GMs. Black (trawl 3.1 m GM) and red (trawl 4.9 m GM) bars indicate the 95% confidence intervals.

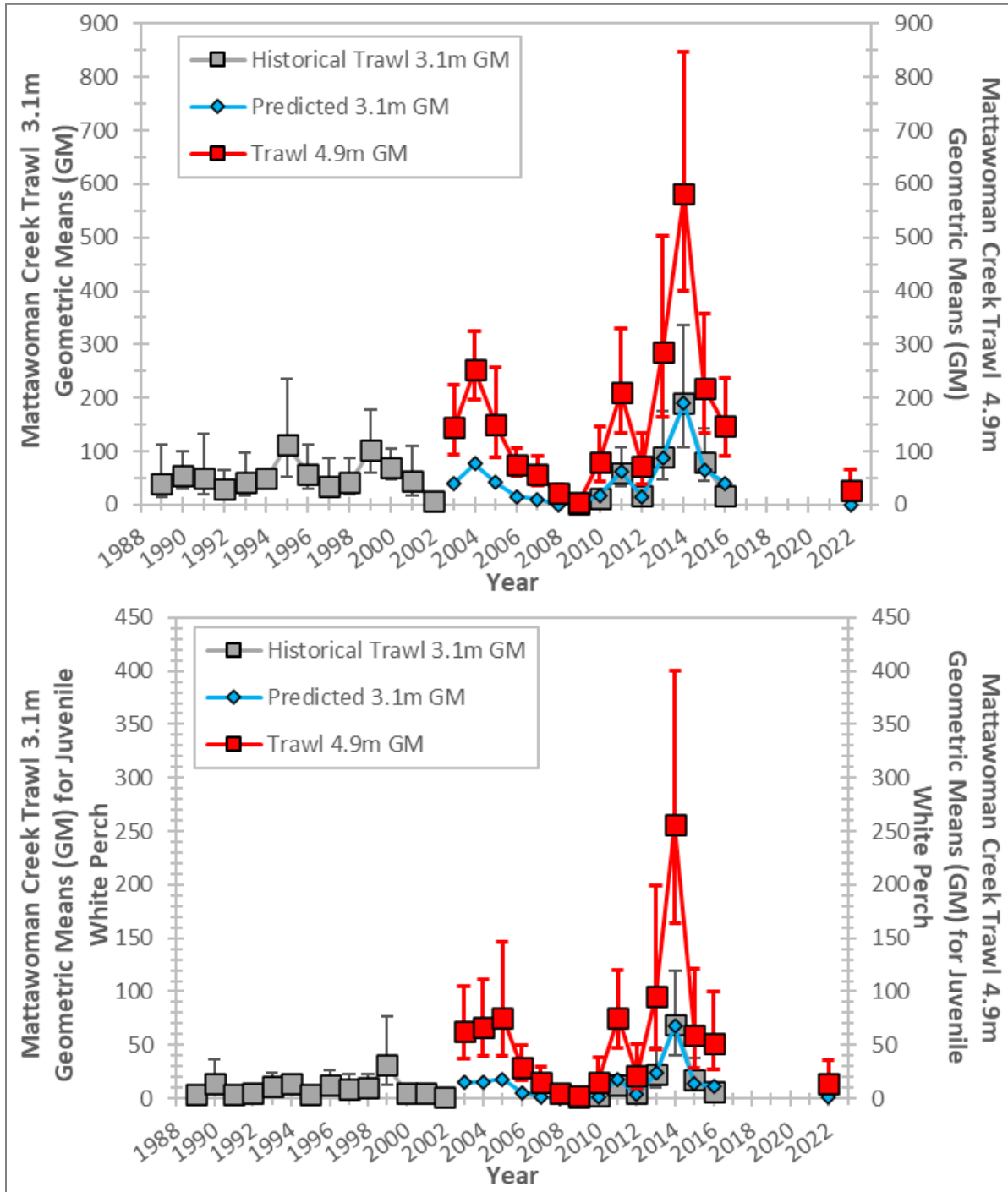


Figure 3-26. Time-series of beach seine catches geometric mean (GM) of all finfish species (red squares) for subestuaries sample in 2022.

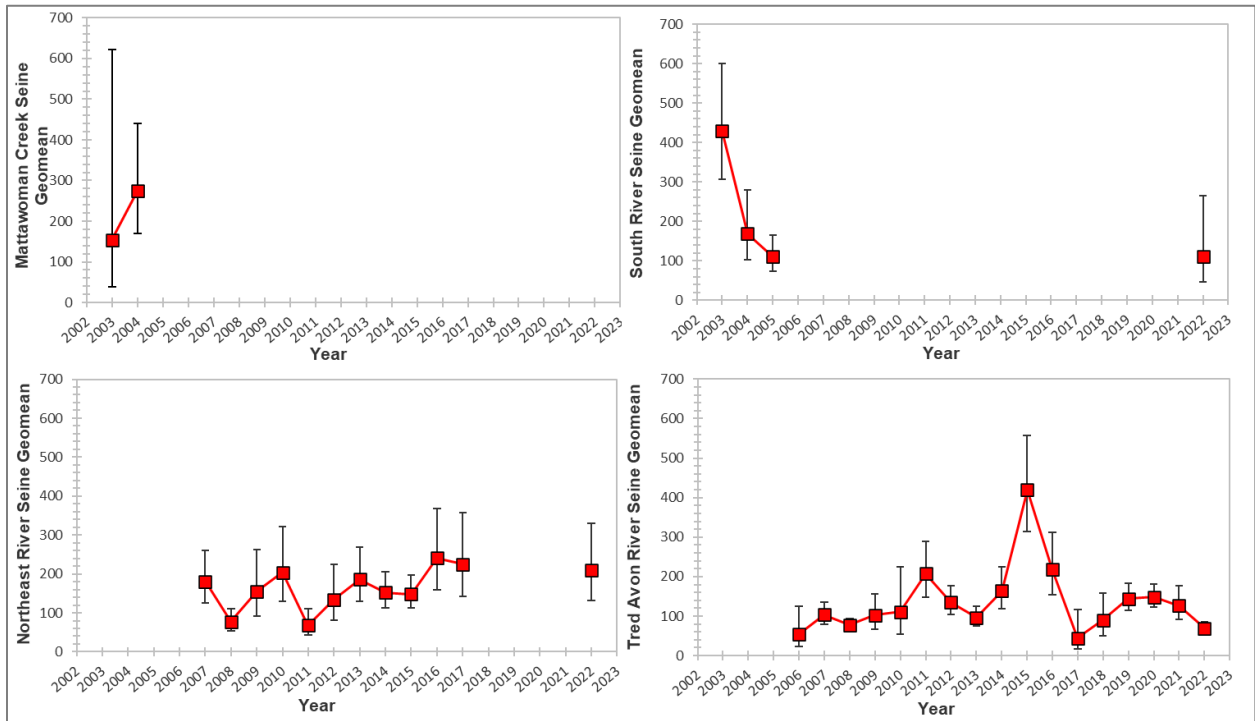


Figure 3-27. Finfish species composition for beach seine catches in subestuaries sampled during 2022 for all sampling years combined. Species that define the top 90% are identified, and the remainder are “other species”.

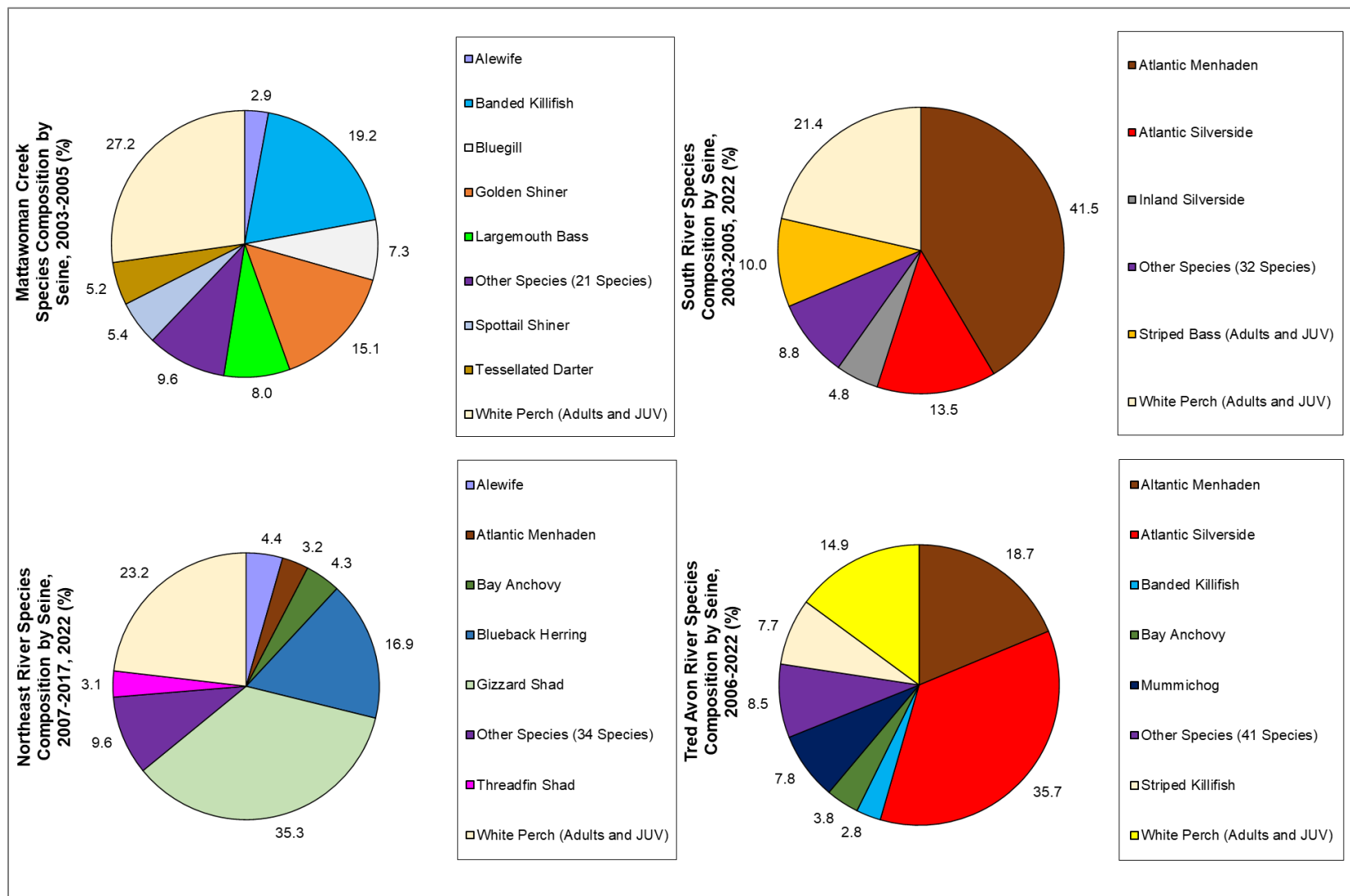
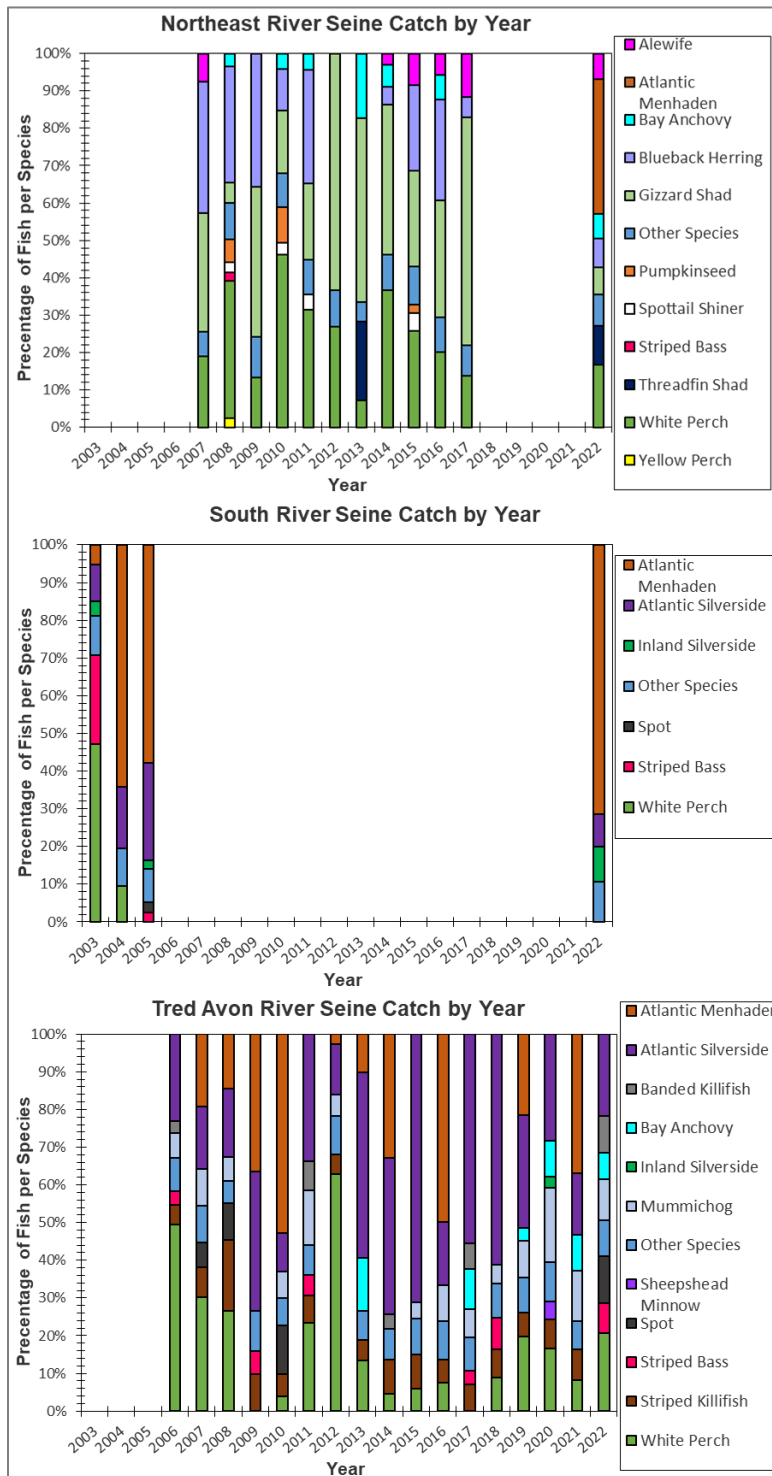


Figure 3-28. Time-series of finfish species composition for beach seine catches in subestuaries sampled during 2022. Species that define the top 90% are identified, and the remainder are “other species”.



Appendices

Regional comparisons for subestuaries and tributaries sampled throughout the Chesapeake Bay from 2003-2022 are divided into three regions, Head-of-Bay, mid-Bay, and Potomac River. These regional comparisons focus on time-series of water quality (temperature, bottom dissolved oxygen, Secchi depths, pH, and salinity) and basic metrics of the finfish community. These comparisons can be found in Appendices, A – Head-of-Bay Region, B – Mid-Bay Region, and C – Potomac River Region.

Appendix I – Head-of-Bay Region

Figure A1. Map indicating the locations of Head-of-Bay subestuaries sampled, Bohemia River, Bush River, Gunpowder River, Middle River, Northeast River, and Sassafras River, during 2003 – 2022.

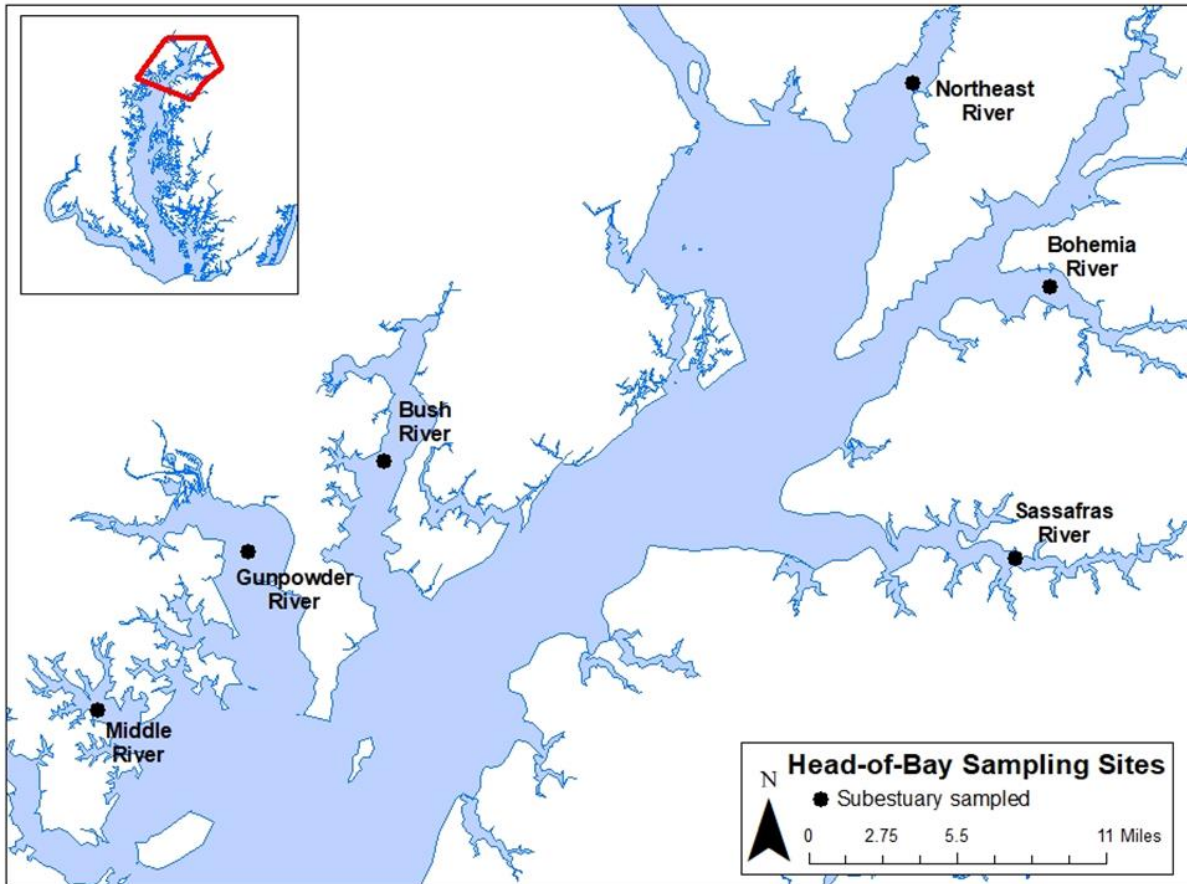


Figure A2. Median bottom temperature (°C; red squares) by year sampled for Head-of-Bay subestuaries. Solid black bars indicate range of all bottom temperature measurements for that year.

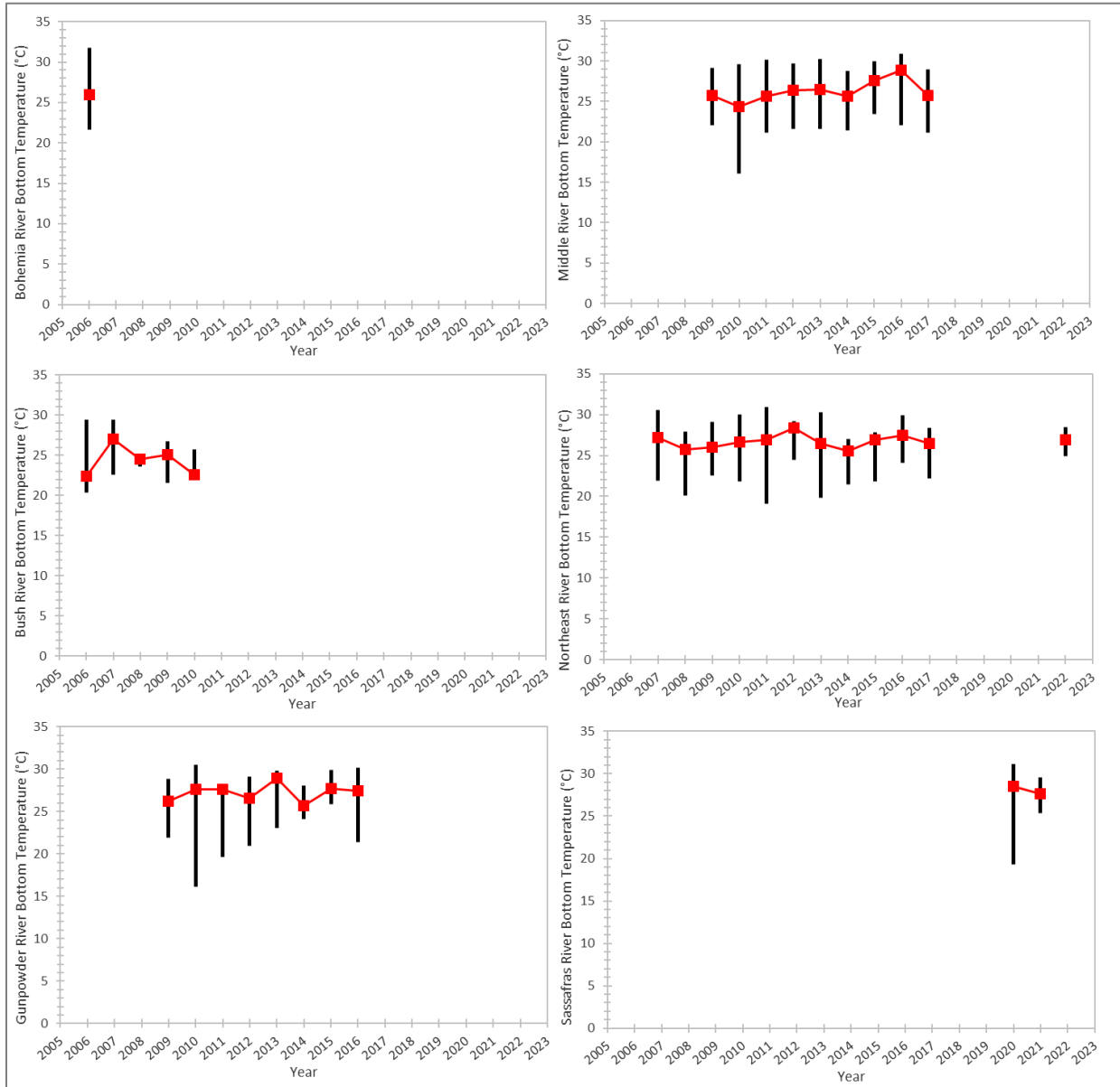


Figure A3. Median bottom dissolved oxygen (DO; red squares; mg / L) by year sampled for Head-of-Bay subestuaries. Solid black bars indicate range of all bottom DO measurements for that year.

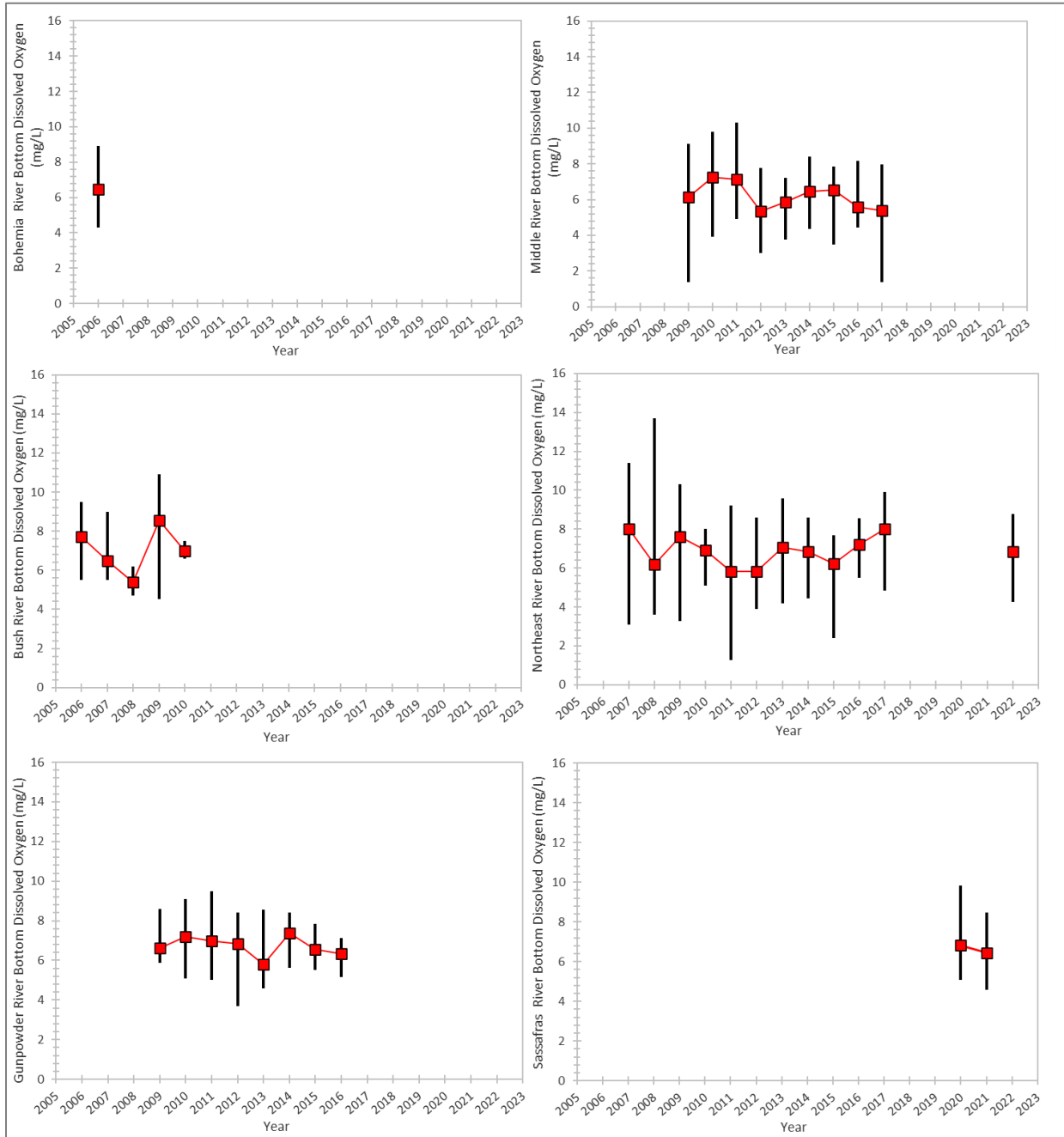


Figure A4. Median Secchi (m; red squares) depths by year sampled for Head-of-Bay subestuaries. Solid black bars indicate range of all Secchi measurements for that year.

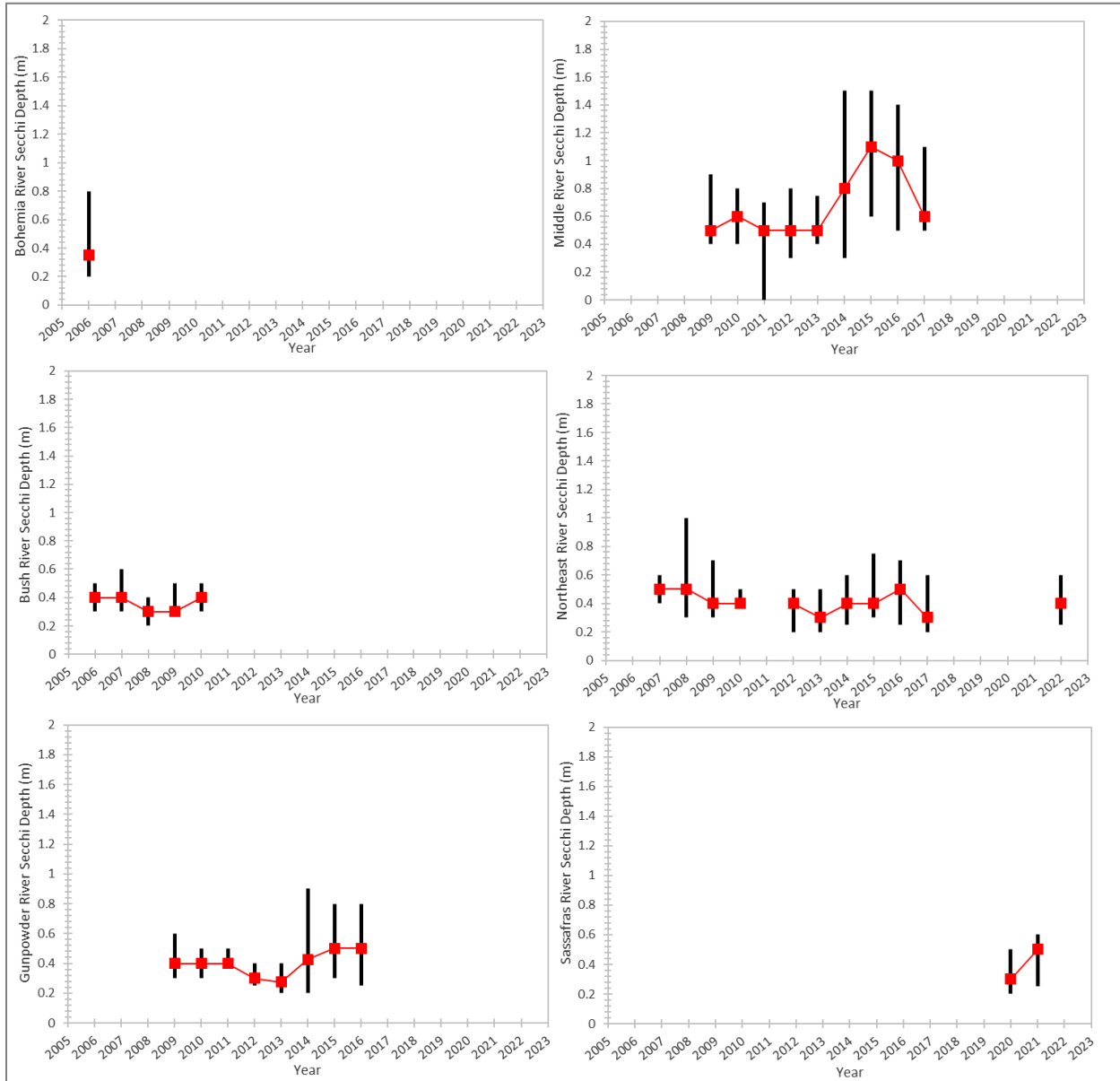


Figure A5. Median bottom pH (red squares) for Head-of-Bay subestuaries, by sampling year. Solid black bars indicate the range of pH measurements by year. No pH measurements were recorded prior to 2006.

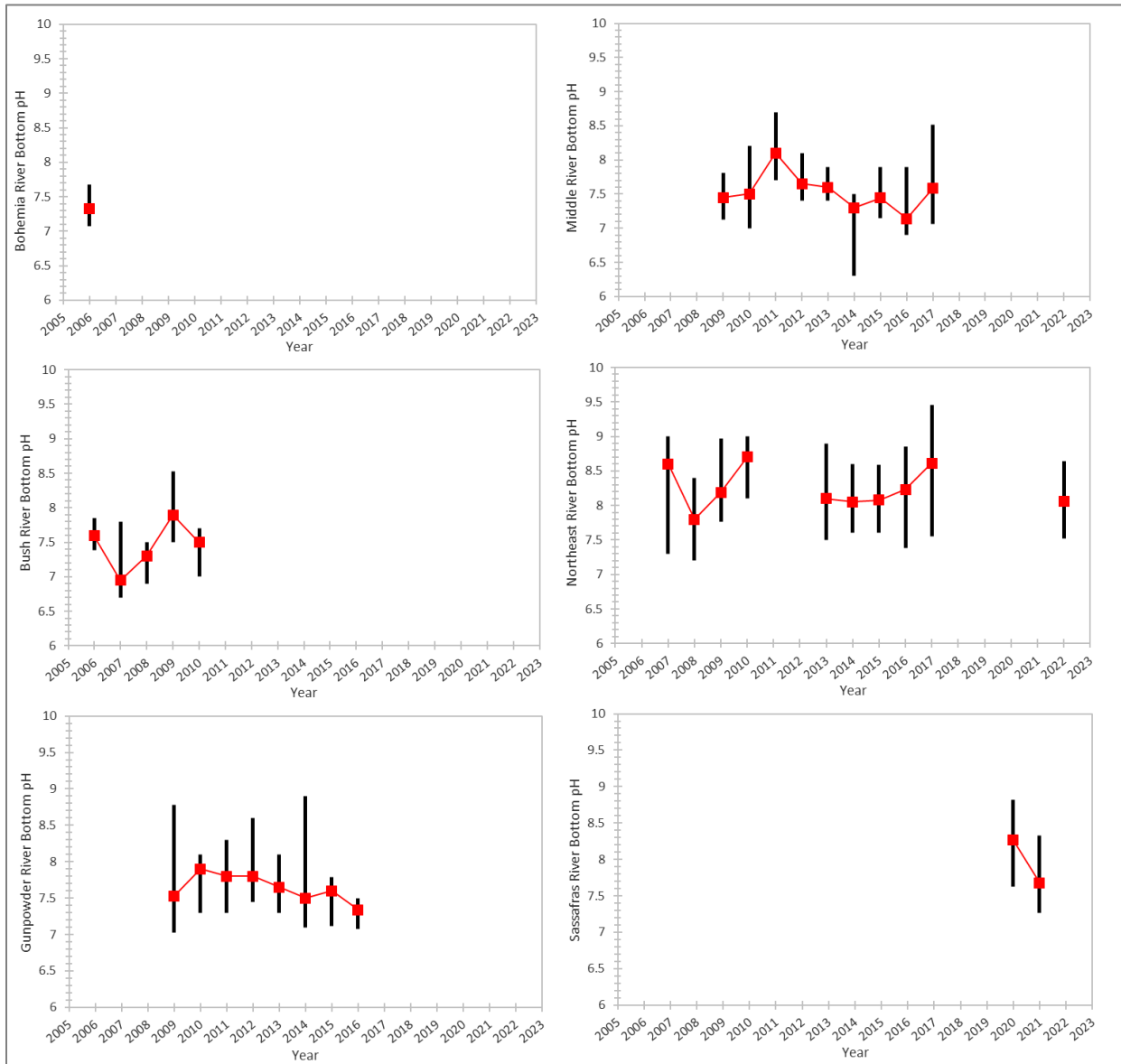


Figure A6. Median bottom salinity (ppt; red squares) by year sampled for Head-of-Bay subestuaries. Solid black bars indicate range of all bottom salinity measurements for that year.

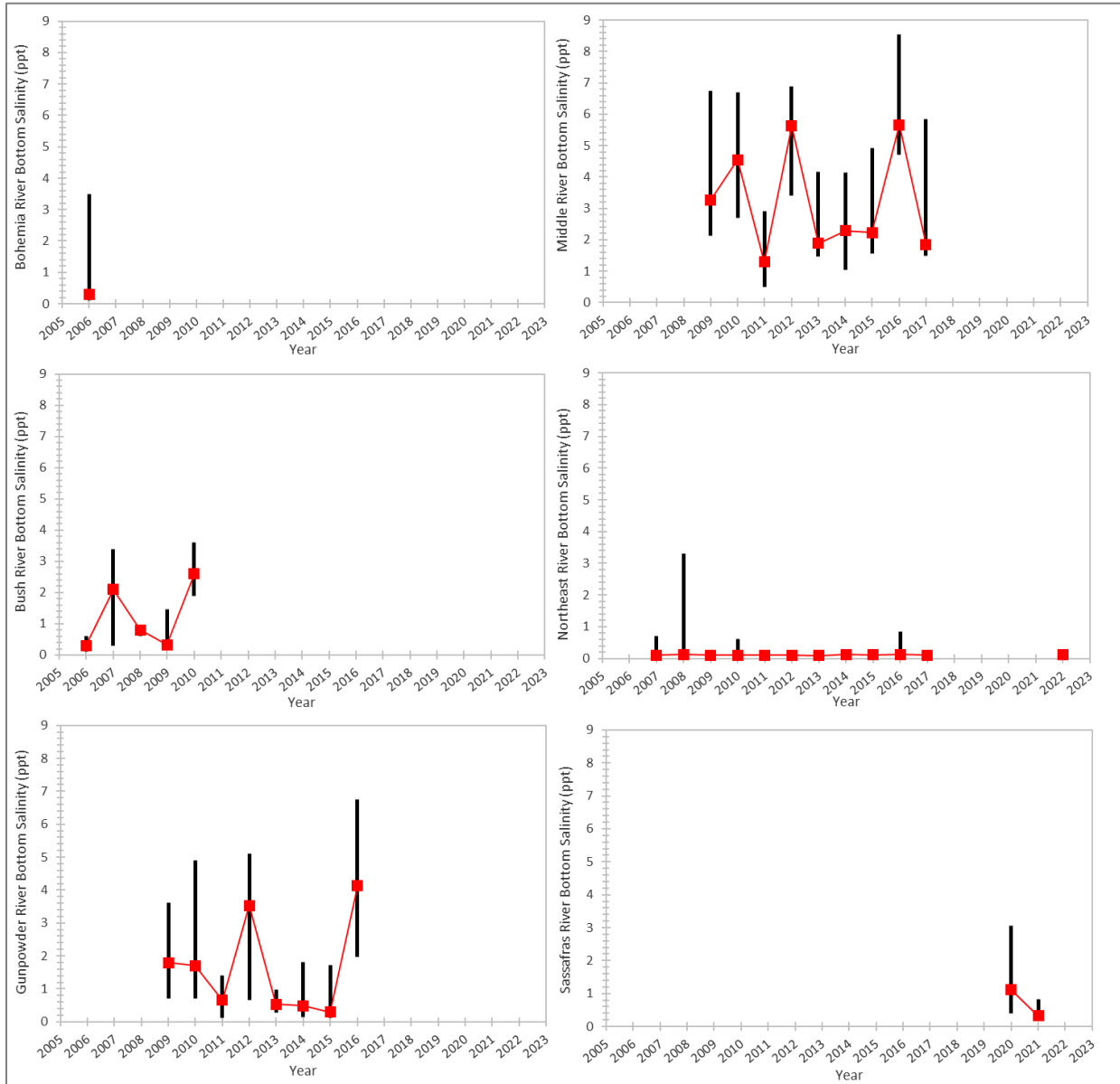


Figure A7. Bottom dissolved oxygen (DO; mg / L) readings (2003 – 2022) in Head-of-Bay subestuaries versus intensity of development (C / ha = structures per hectare). Target (5 mg / L) and threshold (3 mg / L) boundaries are indicated by red dashed lines.

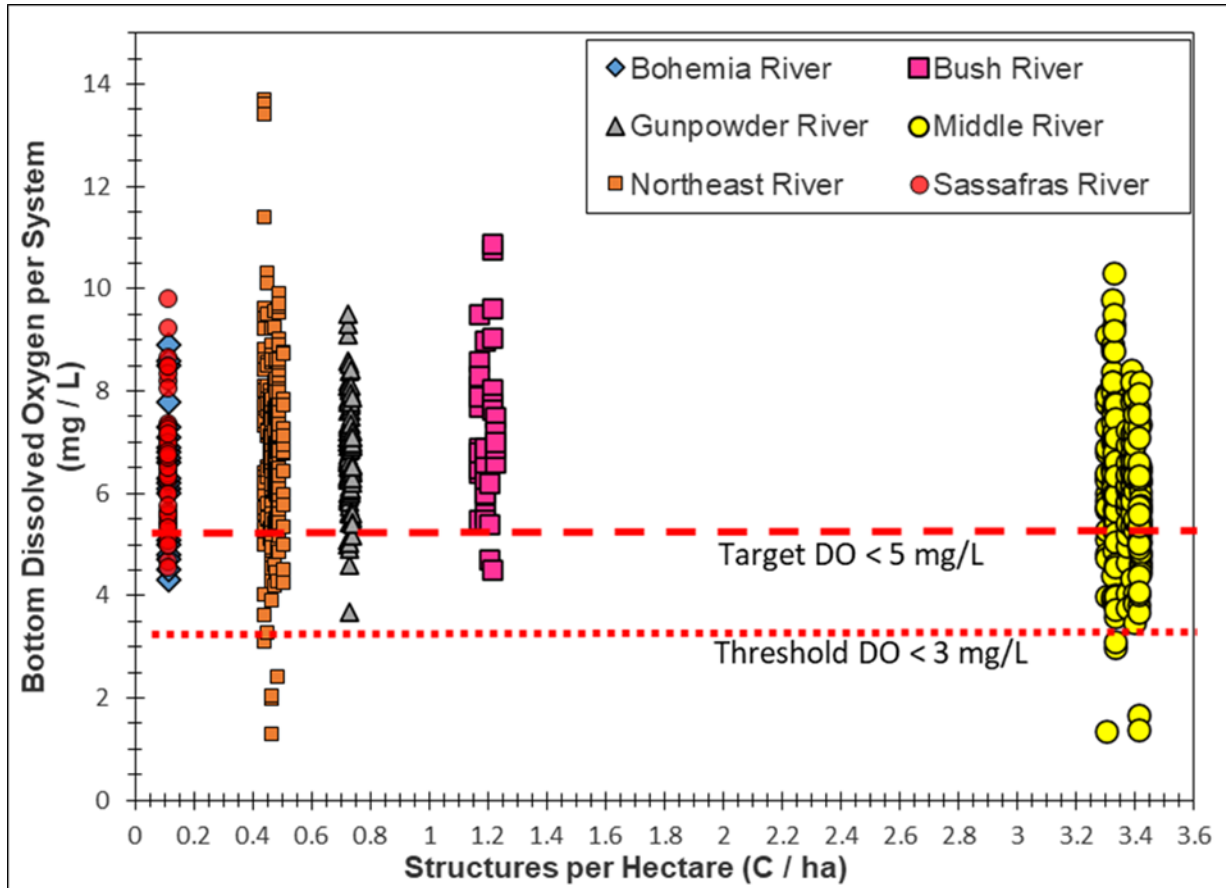


Figure A8. Annual 4.9 m bottom trawl catch geometric mean (GM) of all finfish species (red squares) for Head-of-Bay subestuaries, by sampling year. Black bars indicate the 95% confidence intervals.

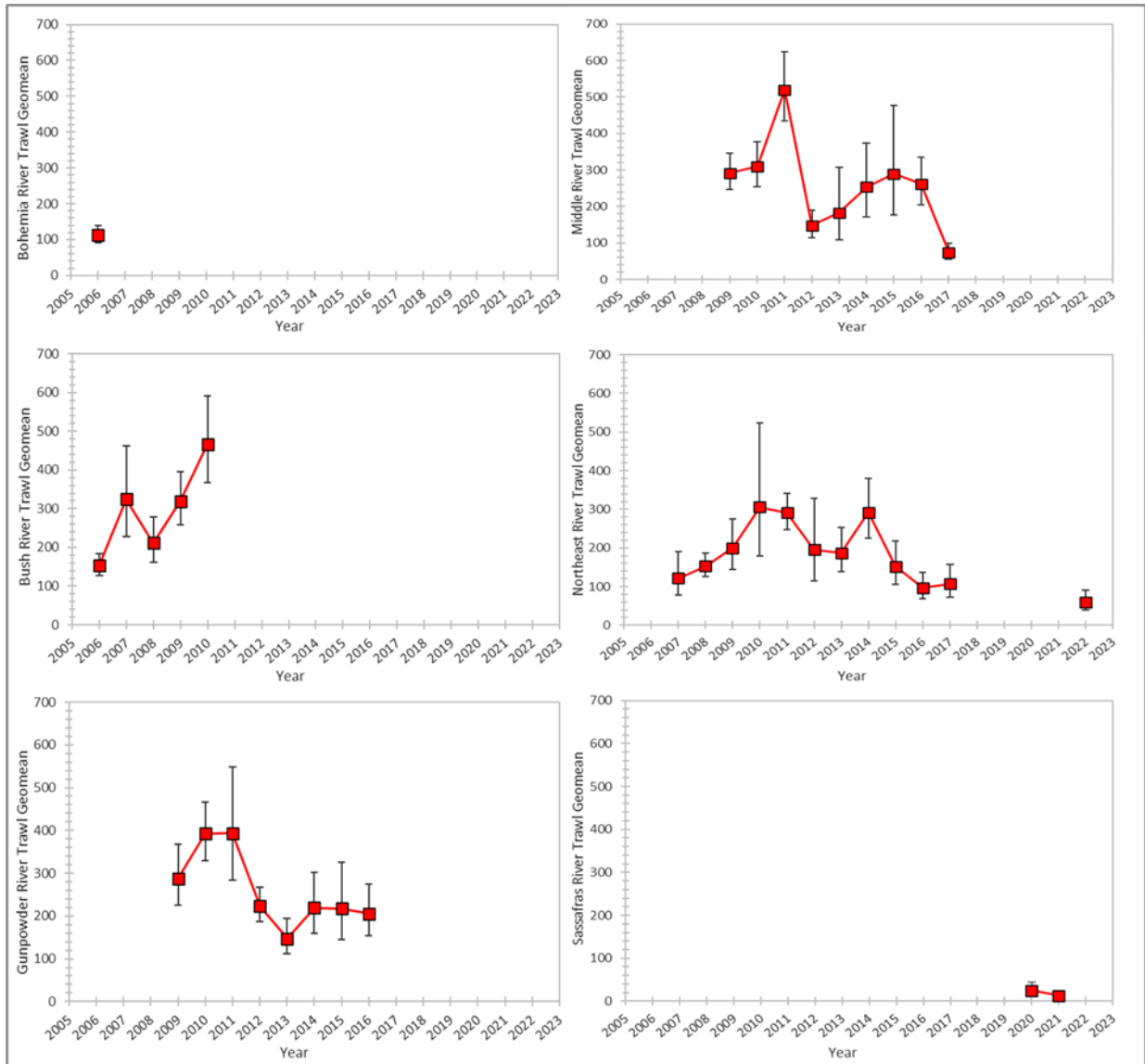
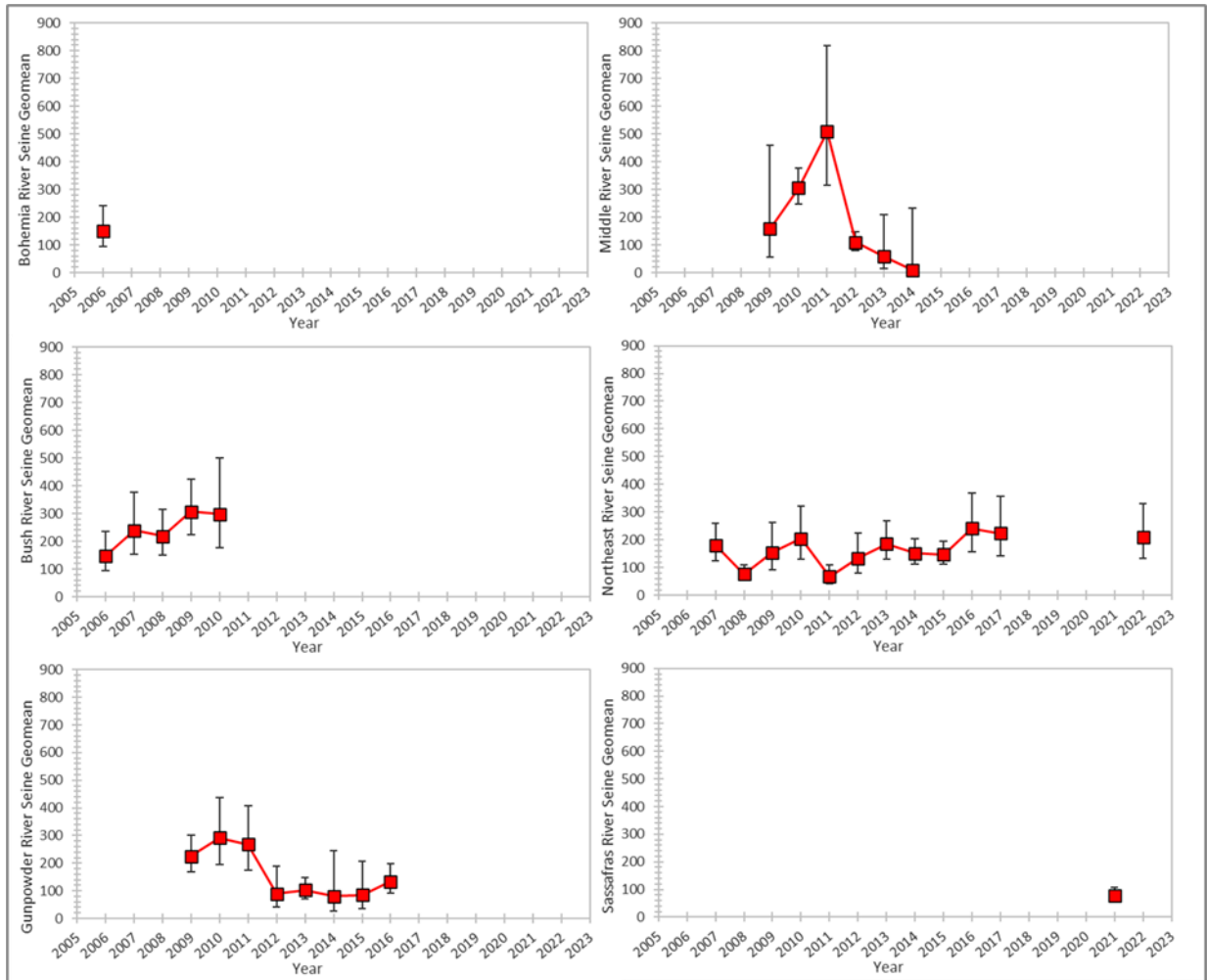


Figure A9. Annual beach seine catch geometric mean (GM) of all finfish species (red squares) for Head-of-Bay subestuaries, by sampling year. Black bars indicate the 95% confidence intervals.



Appendix B – Mid-Bay Region

Figure B1. Map indicating the locations of mid-Bay subestuaries sampled, Broad Creek, Chester River, Corsica River, Langford Creek, Magothy River, Miles River, Rhode River, Severn River, South River, Tred Avon River, West River, and Wye River, during 2003 – 2022.

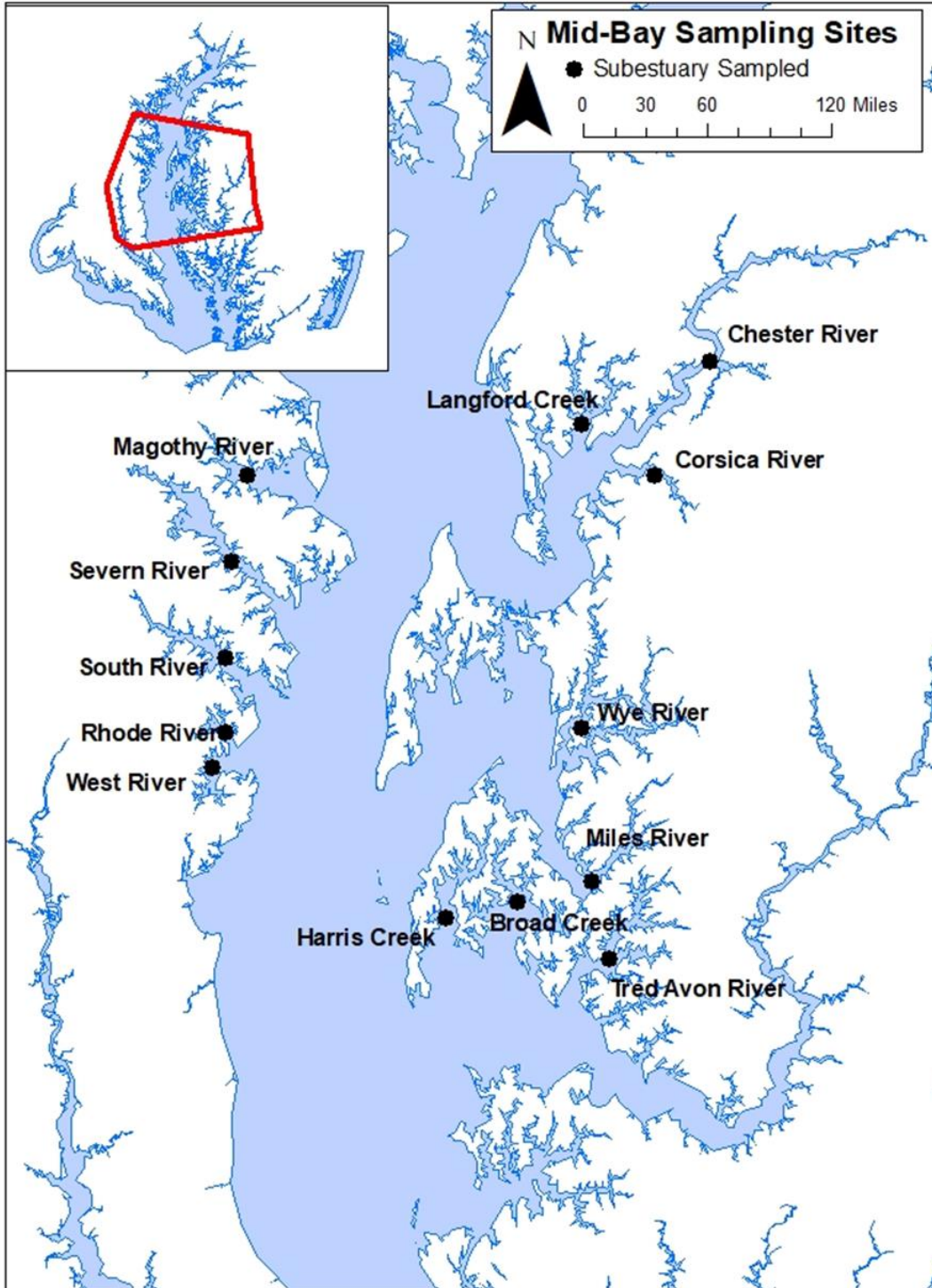


Figure B2. Median bottom temperature ($^{\circ}\text{C}$; red squares) by year sampled for mid-Bay subestuaries. Solid black bars indicate range of all bottom temperature measurements for that year.

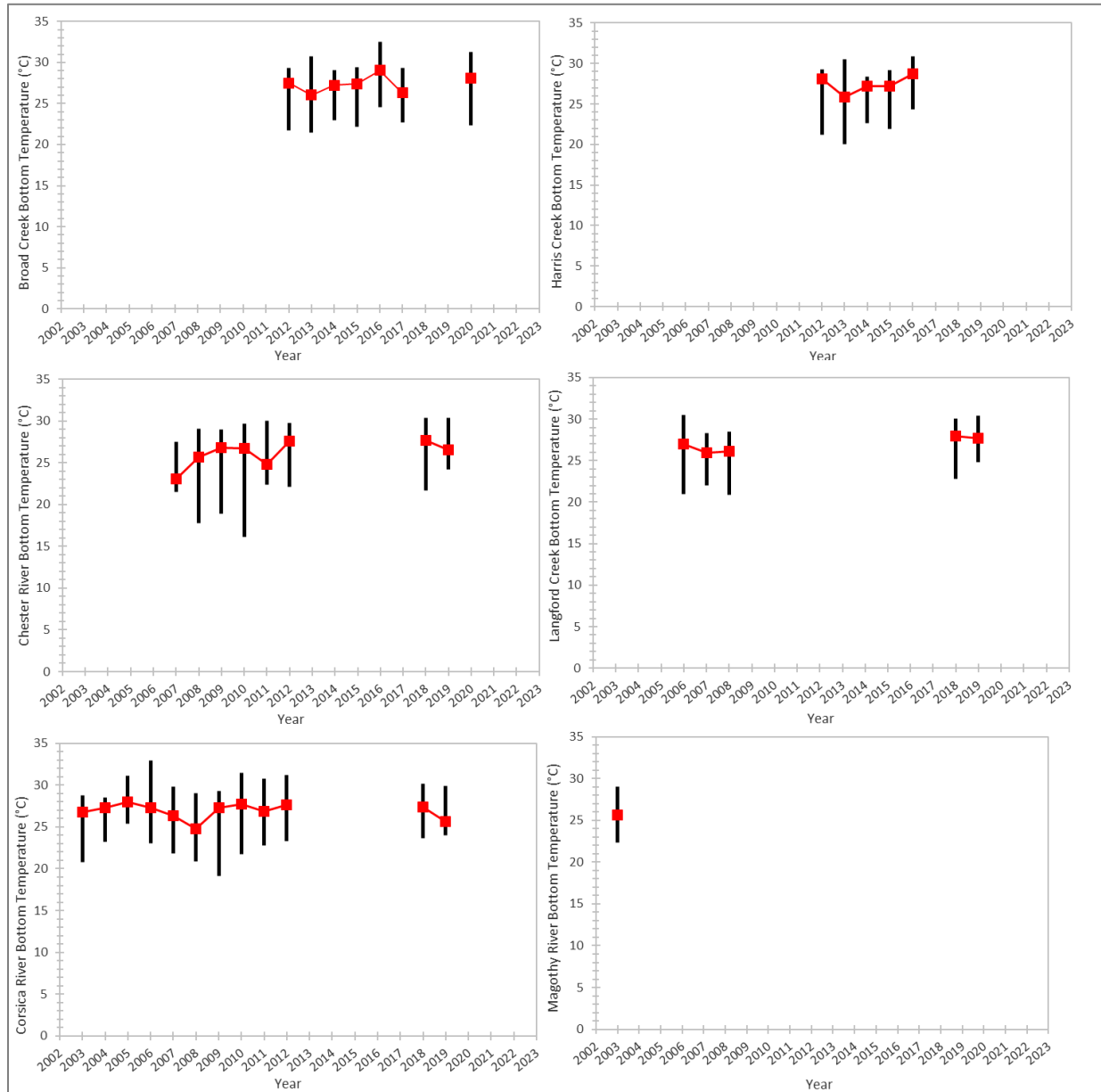


Figure B2. Continued.

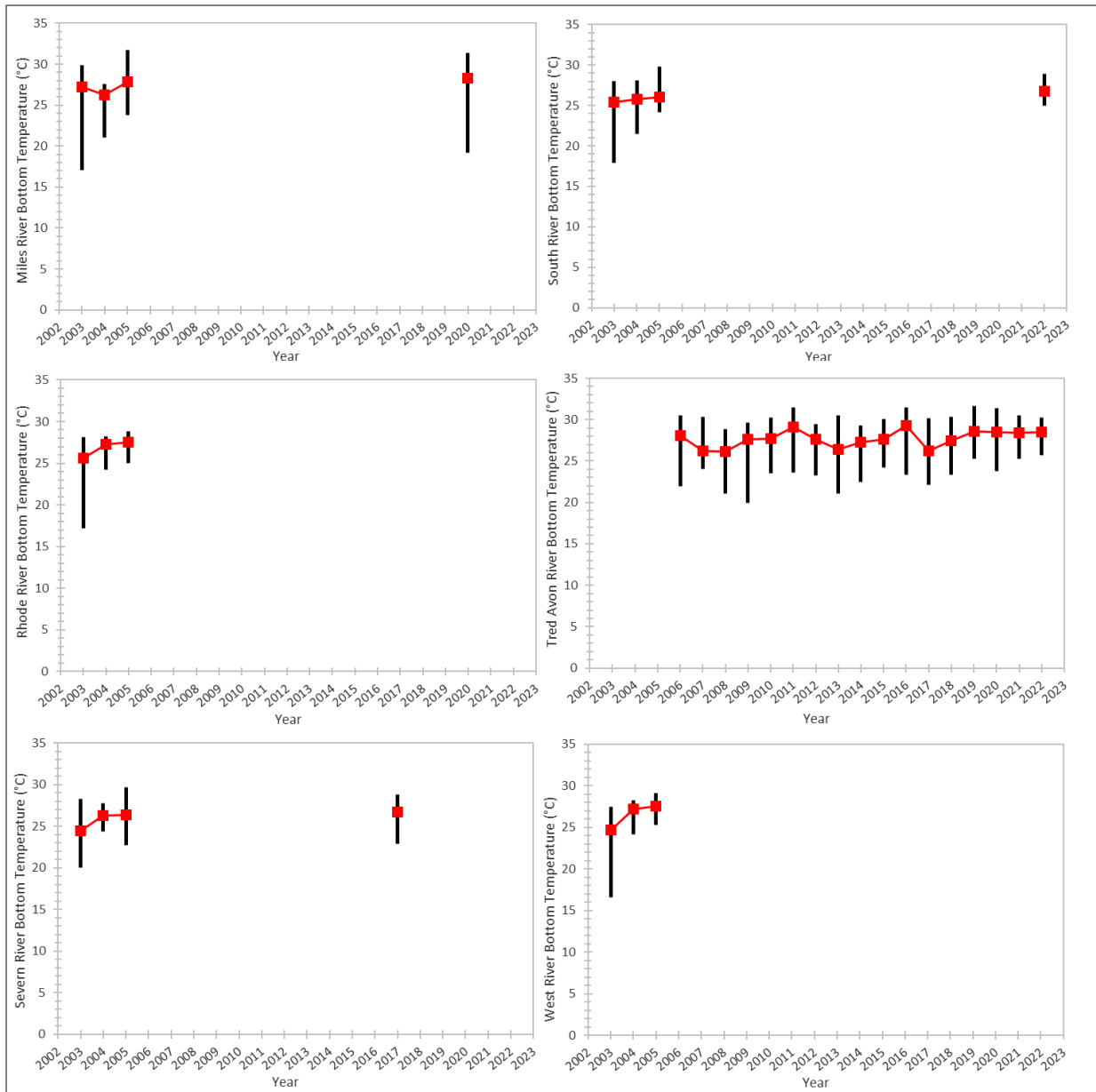


Figure B2. Continued.

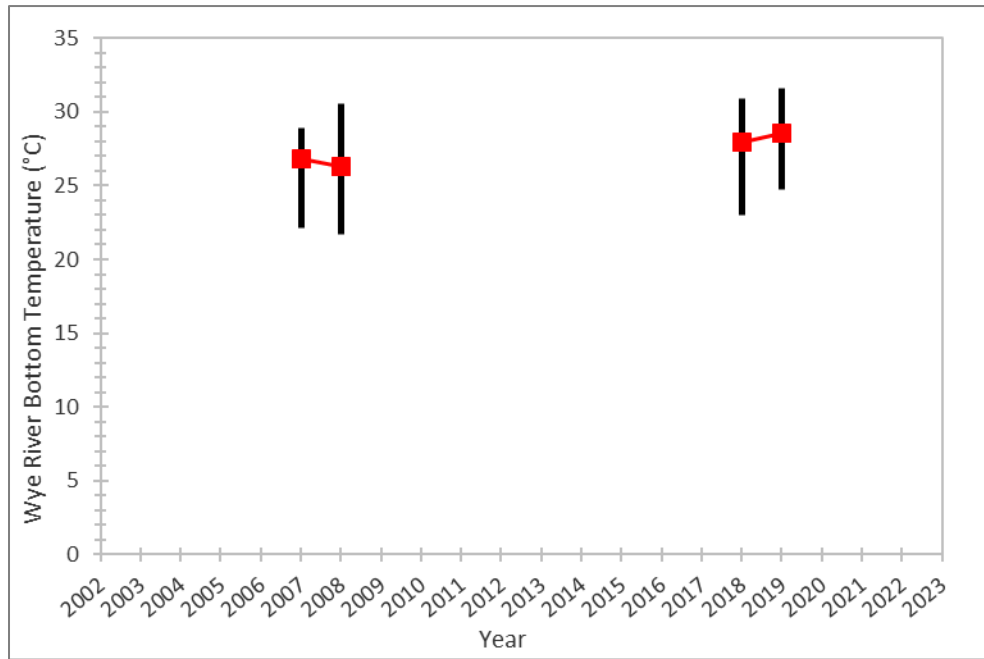


Figure B3. Median bottom dissolved oxygen (DO; red squares; mg / L) by year sampled for mid-Bay subestuaries. Solid black bars indicate range of all bottom DO measurements for that year.

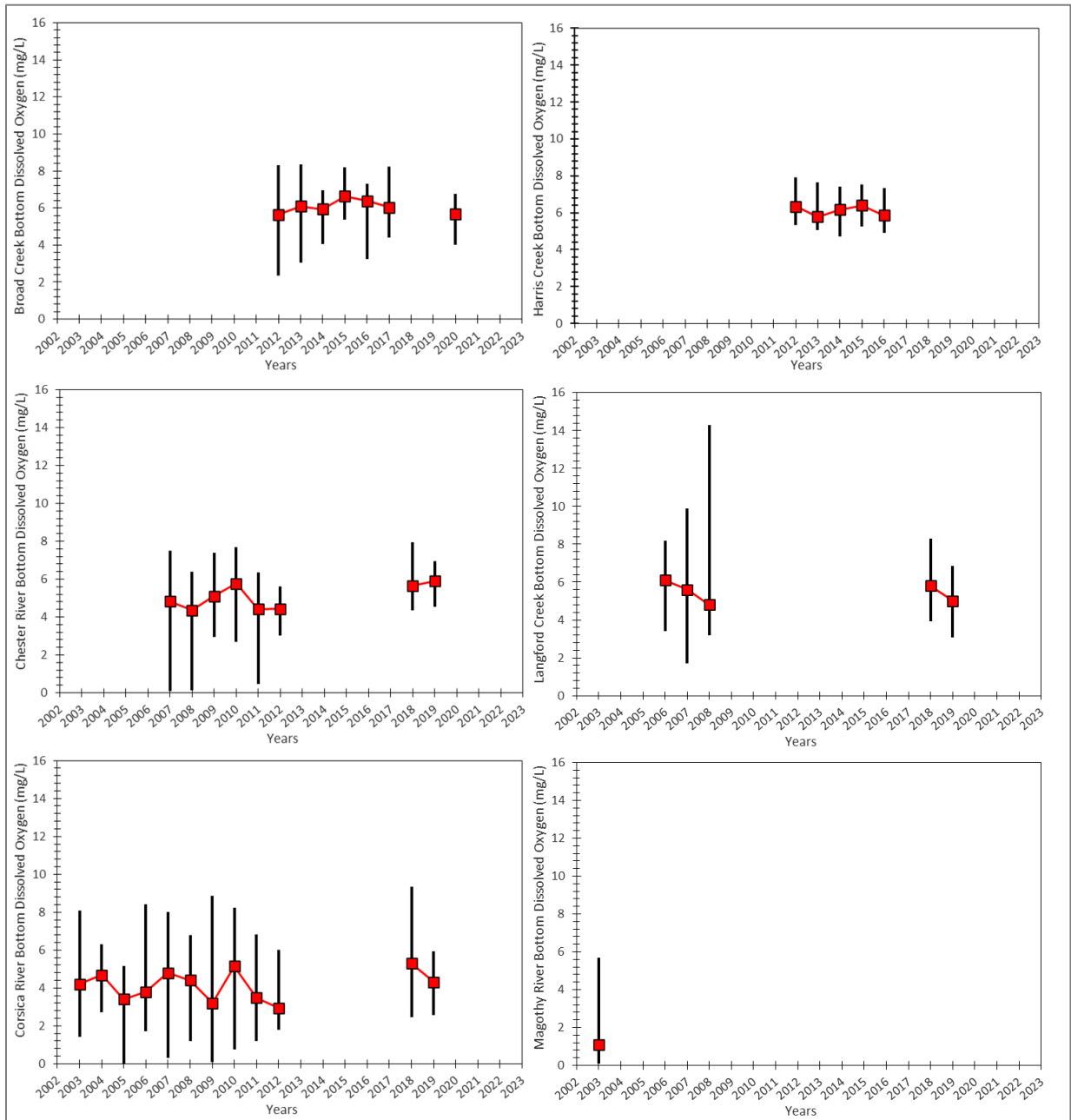


Figure B3. Continued.

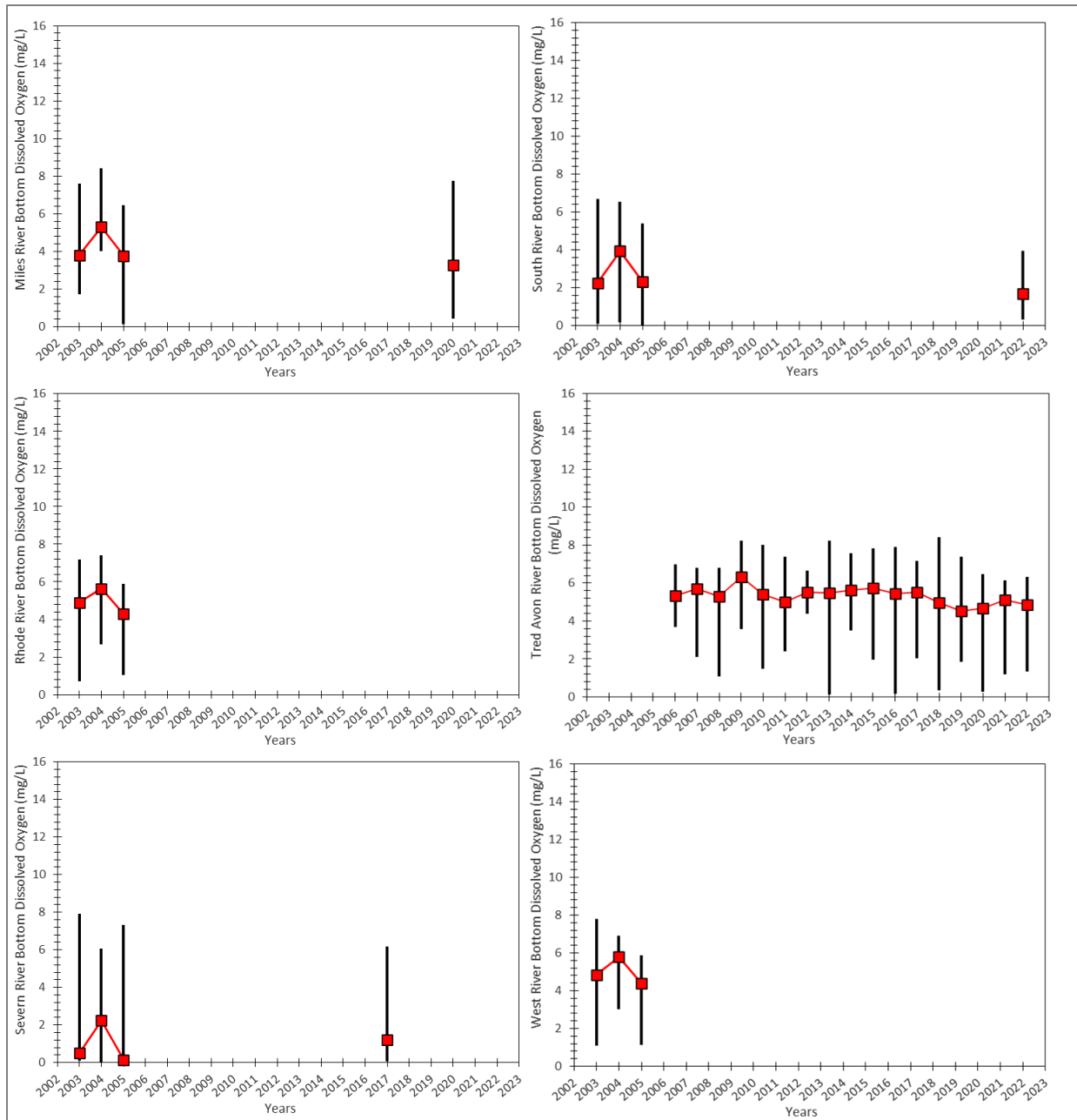


Figure B3. Continued.

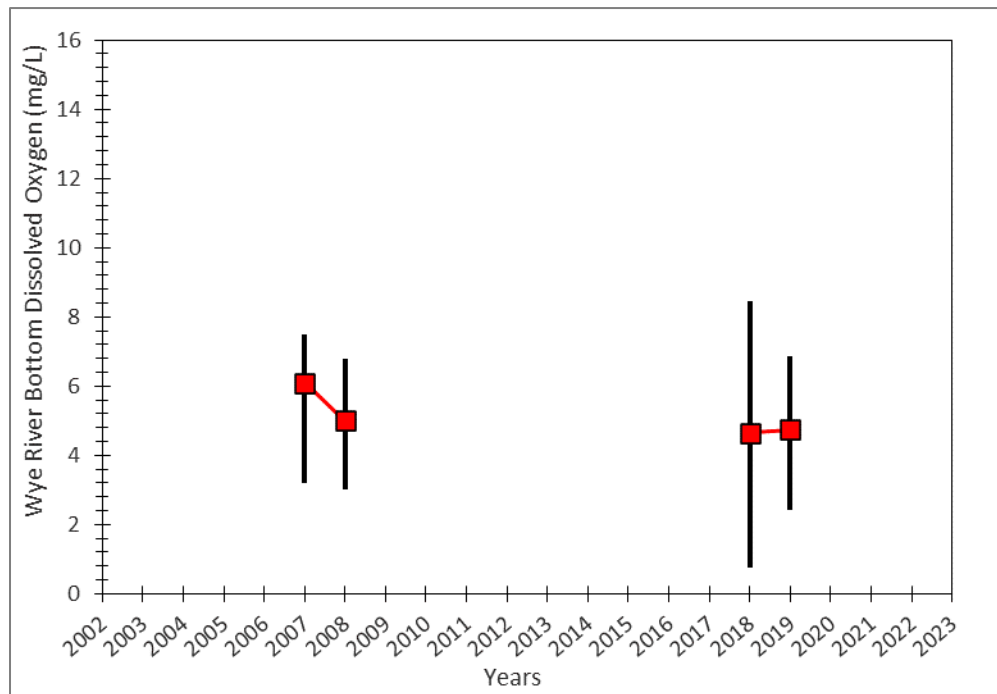


Figure B4. Median Secchi (m; red squares) depths by year sampled for mid-Bay subestuaries. Solid black bars indicate range of all Secchi measurements for that year.

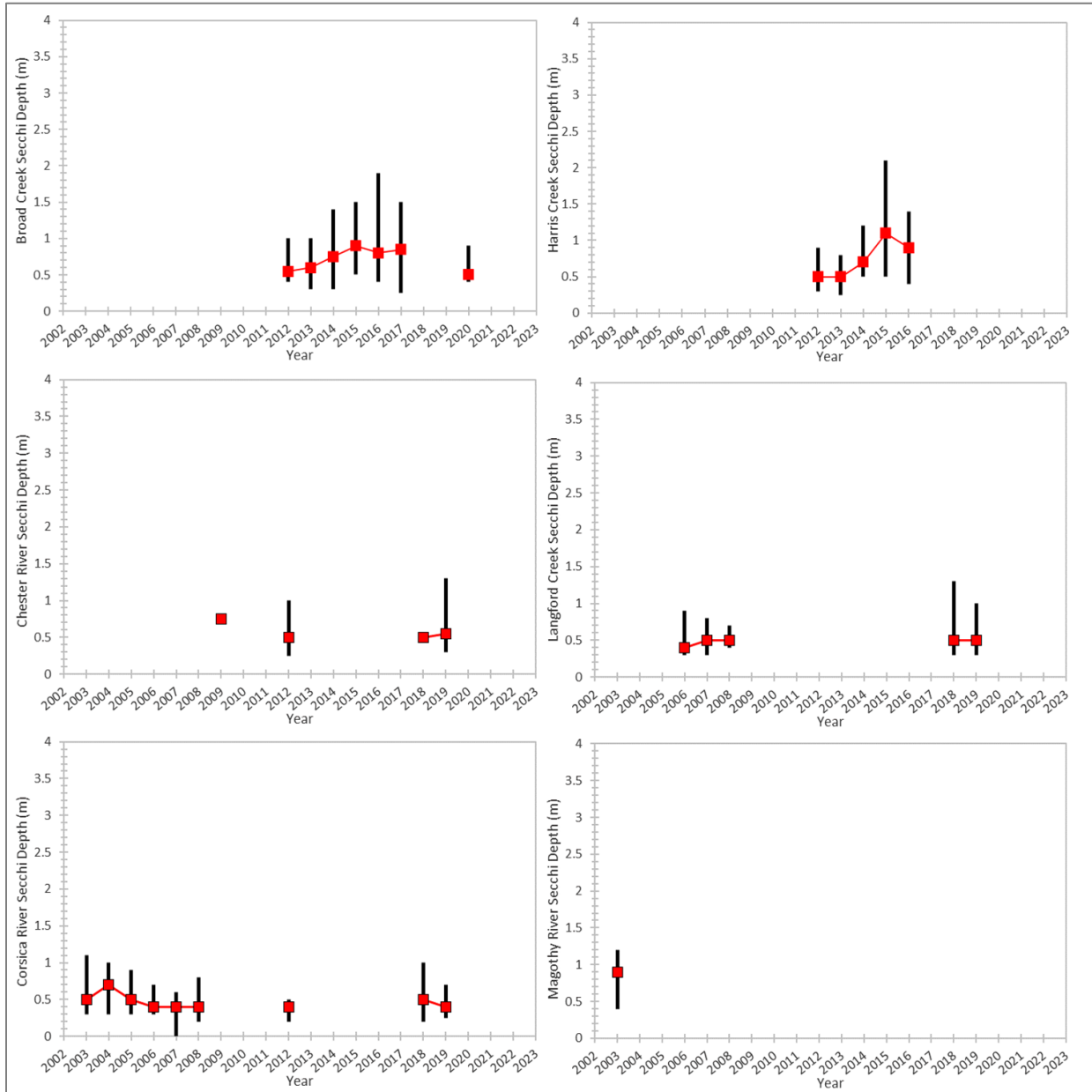


Figure B4. Continued.

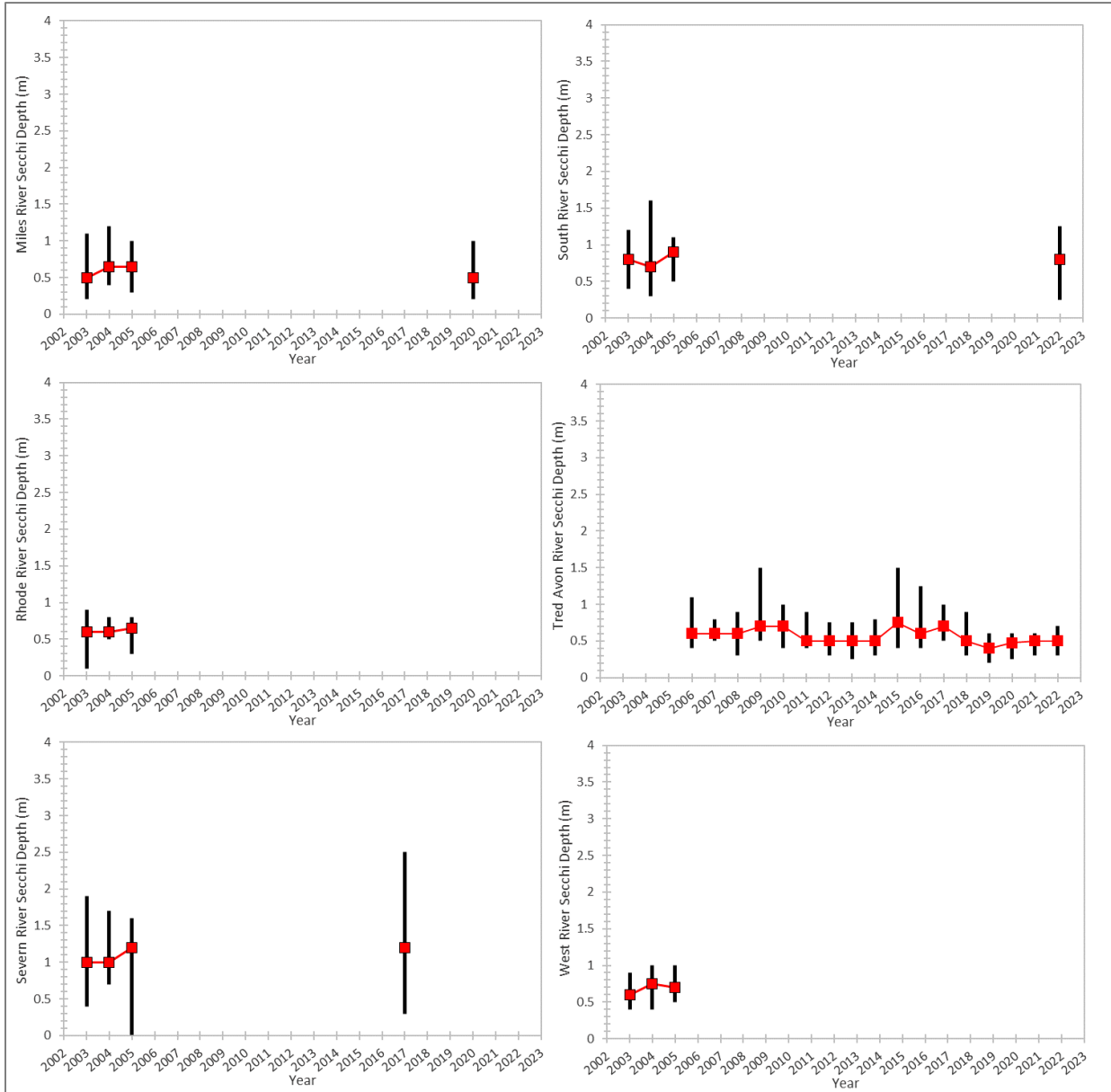


Figure B4. Continued.

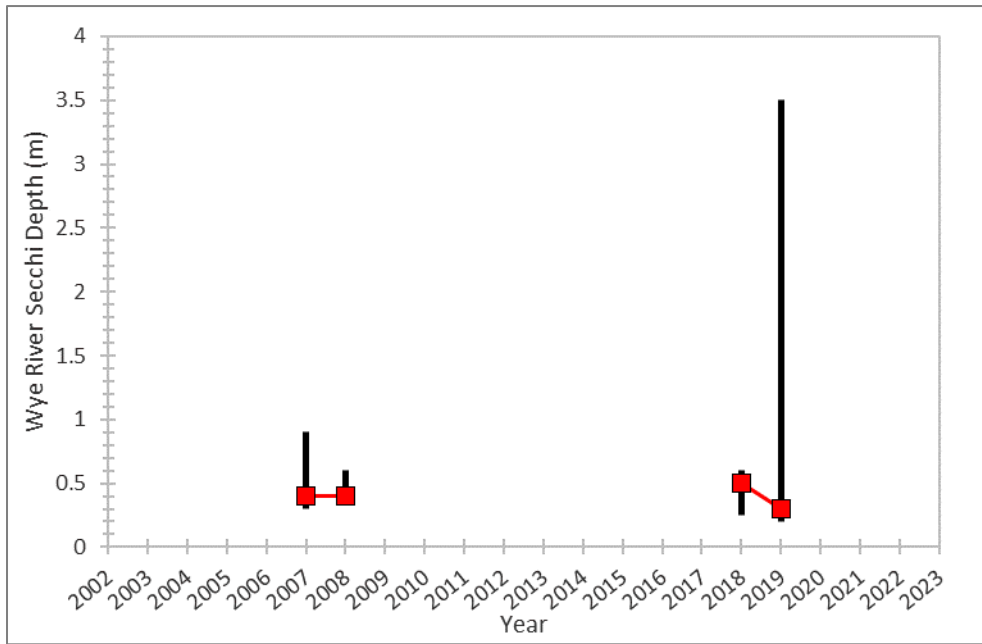


Figure B5. Median bottom pH (red squares) for mid-Bay subestuaries, by sampling year. Solid black bars indicate the range of pH measurements by year. No pH measurements were recorded prior to 2006.

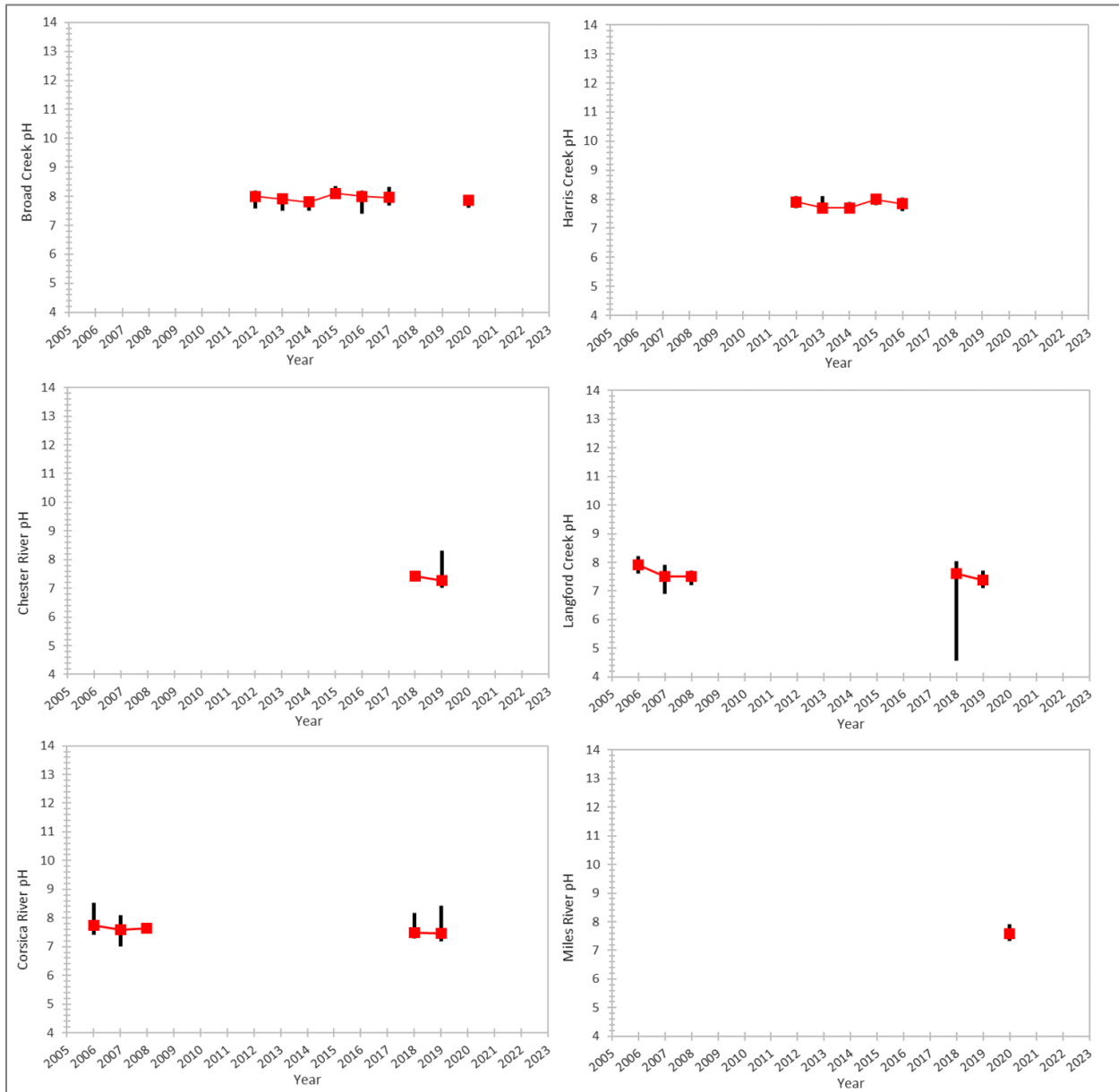


Figure B5. Continued.

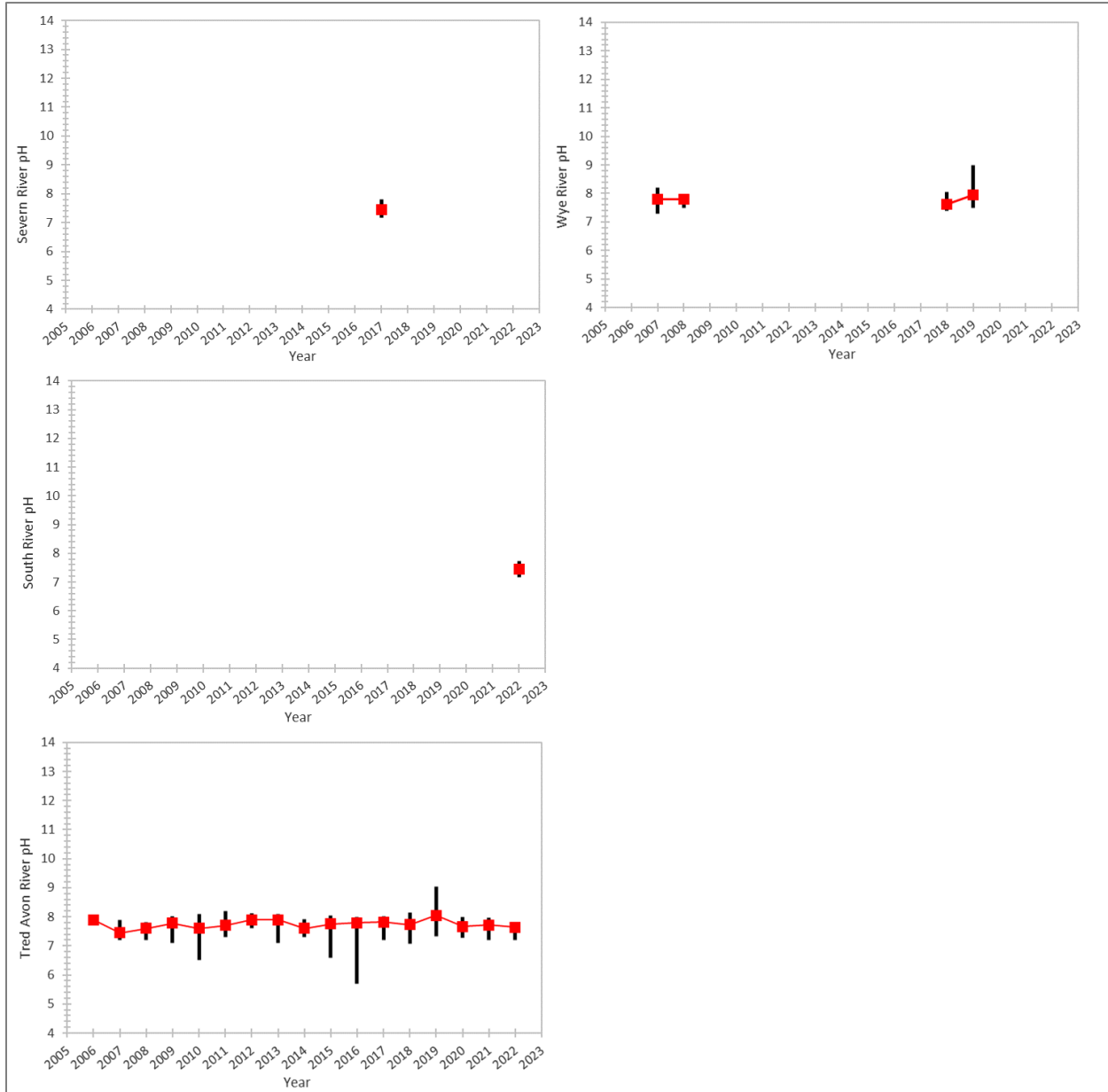


Figure B6. Median bottom salinity (ppt; red squares) by year sampled for mid-Bay subestuaries. Solid black bars indicate range of all bottom salinity measurements for that year.

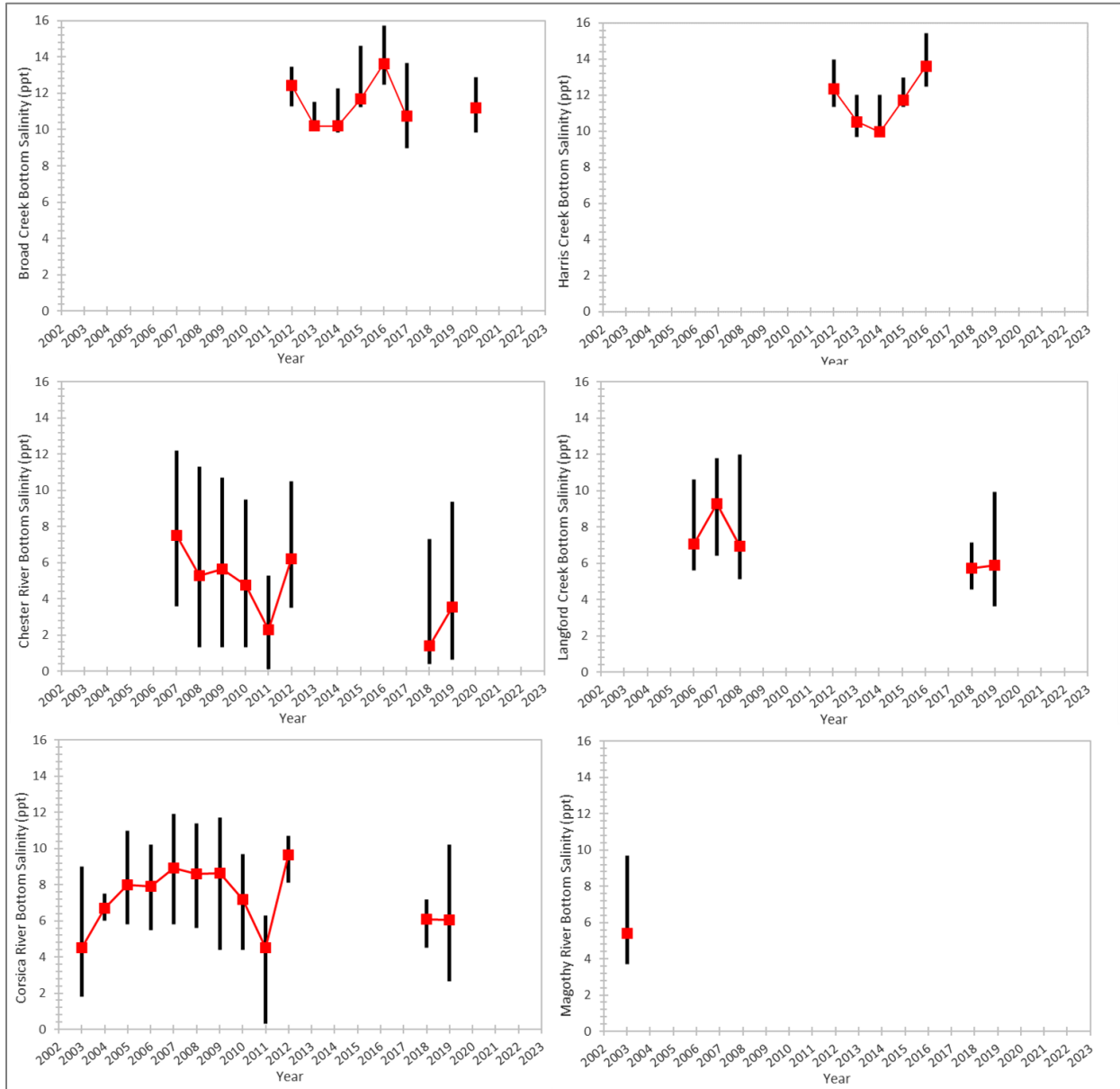


Figure B6. Continued.

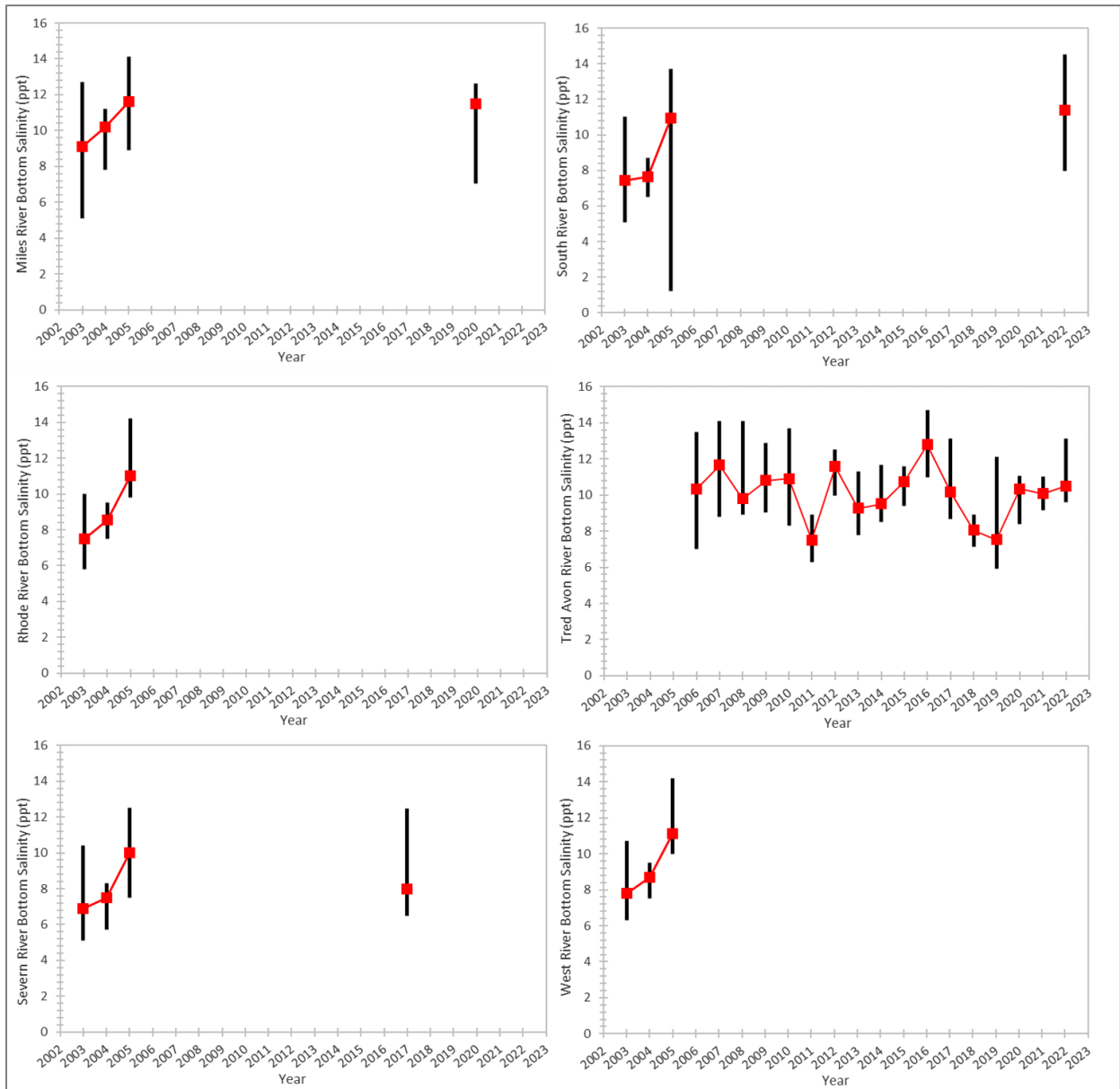


Figure B6. Continued.

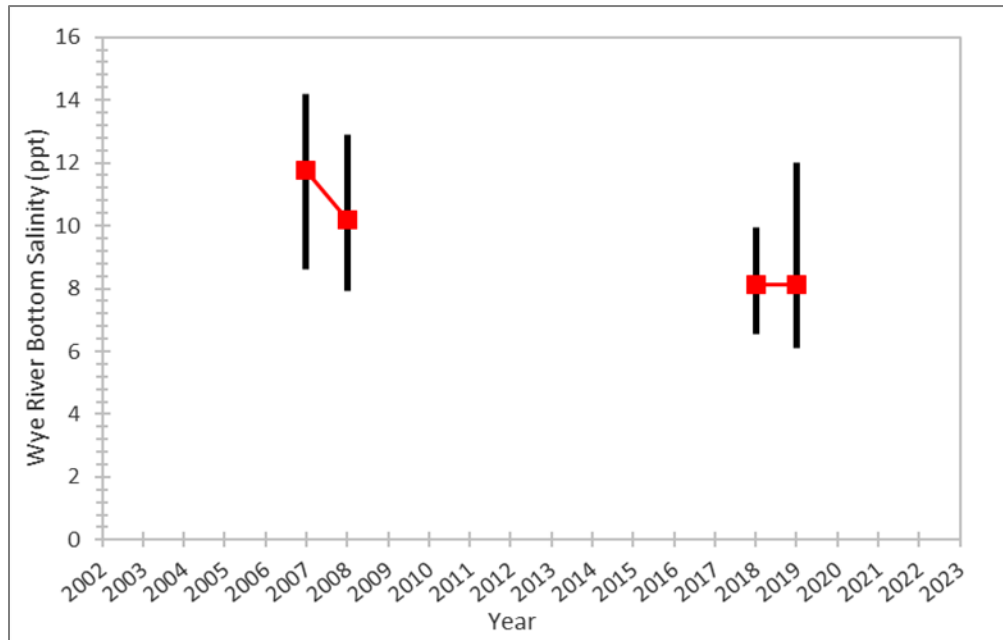


Figure B7. Bottom dissolved oxygen (DO; mg / L) readings (2003 – 2022) in mid-Bay subestuaries versus intensity of development (C / ha = structures per hectare). Target (5 mg / L) and threshold (3 mg / L) boundaries are indicated by red dashed lines.

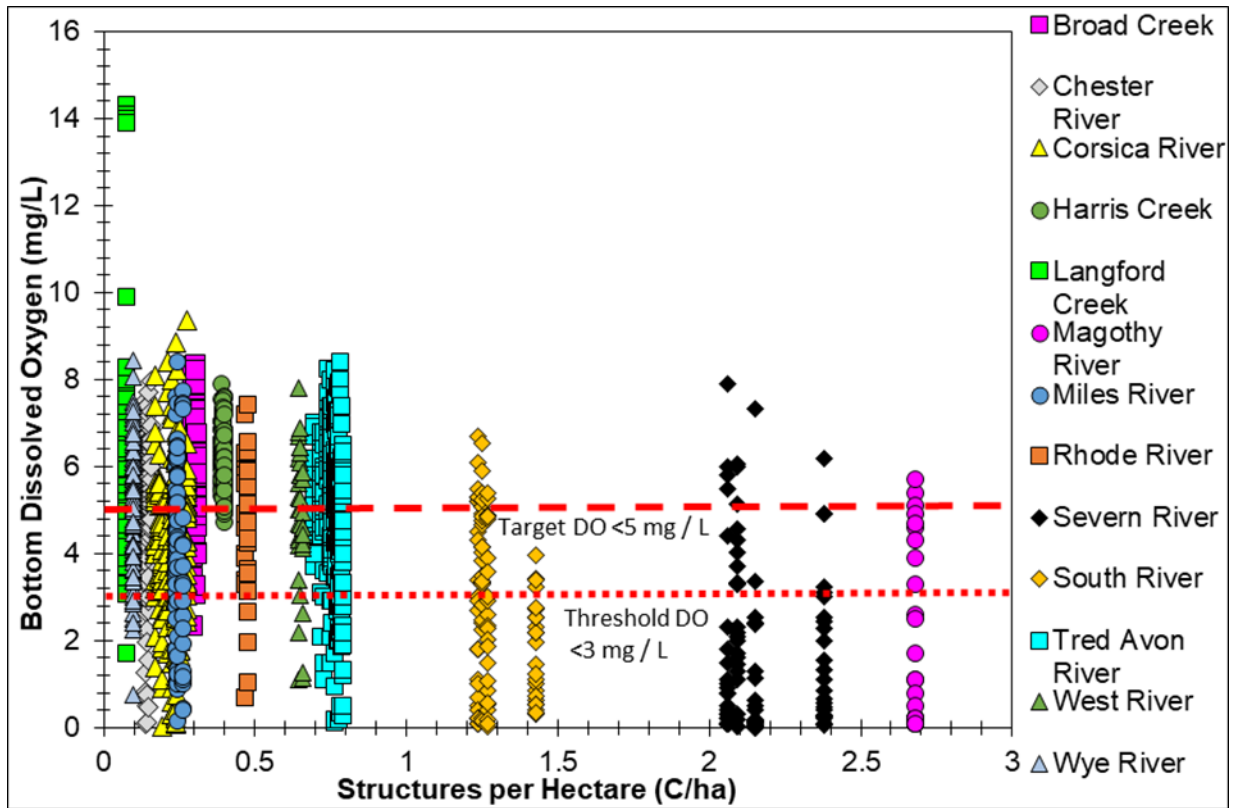


Figure B8. Annual 4.9 m bottom trawl catch geometric mean (GM) of all finfish species (red squares) for mid-Bay subestuaries, by sampling year. Black bars indicate the 95% confidence intervals.

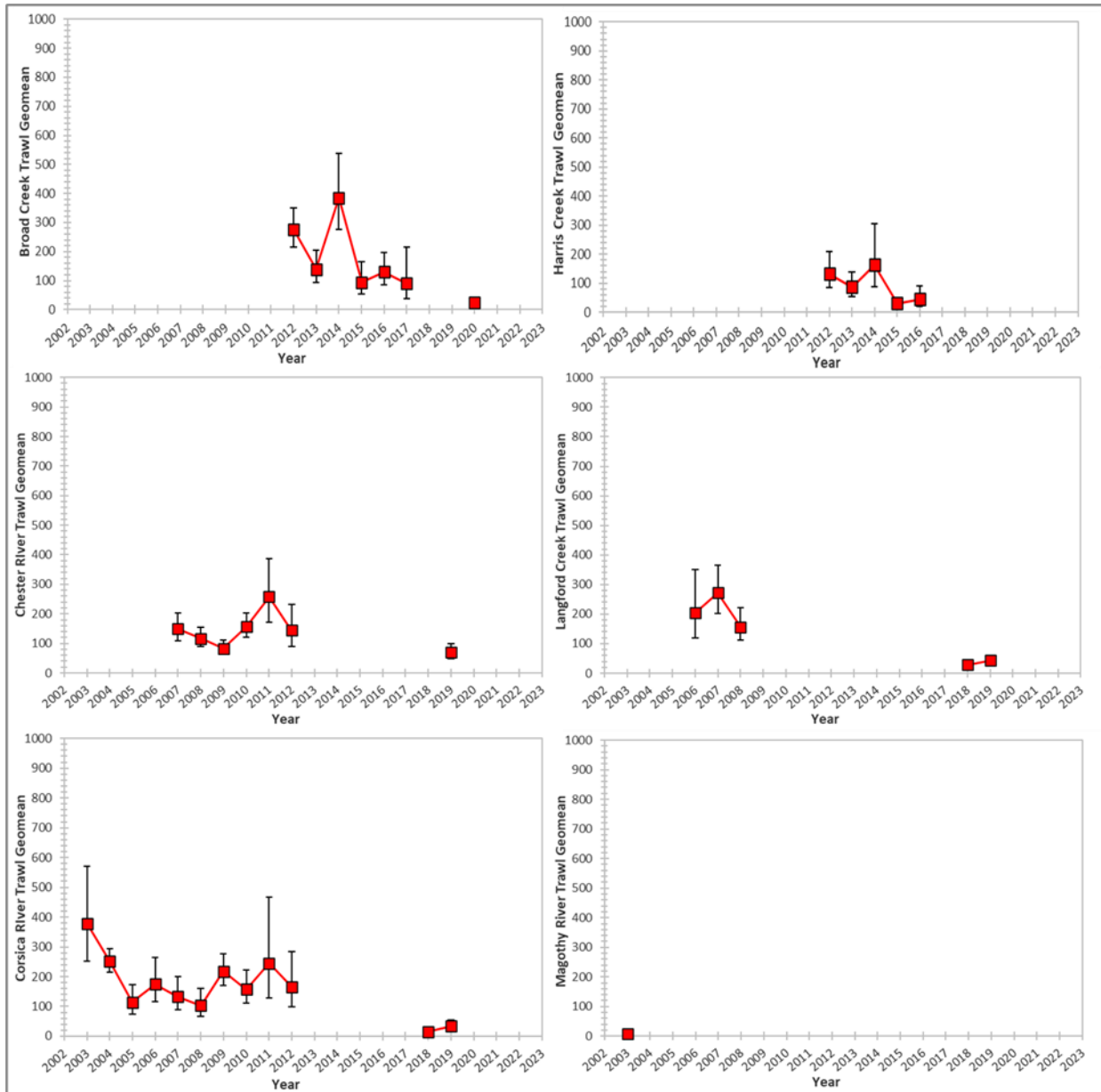


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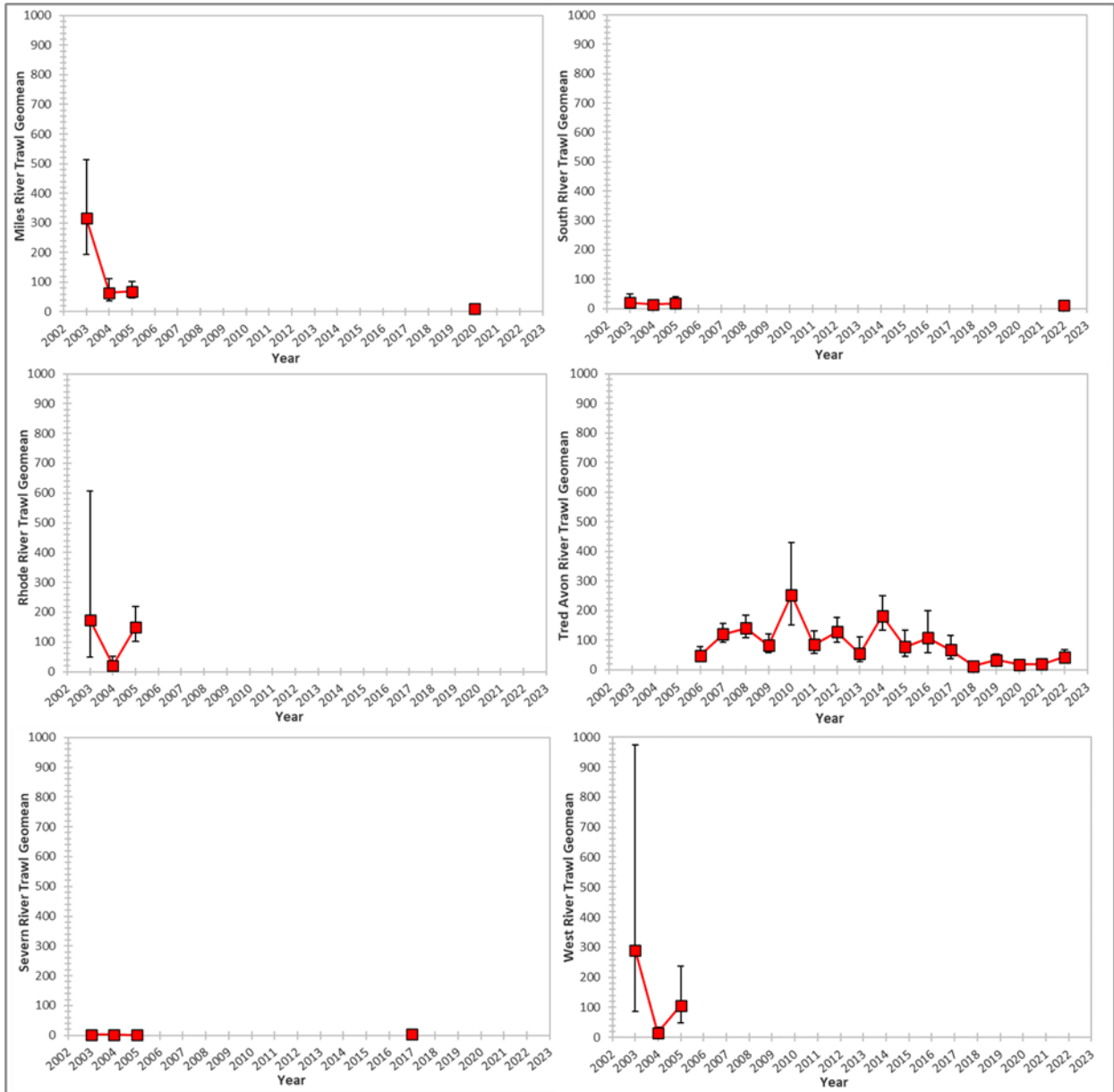


Figure B8. Continued.

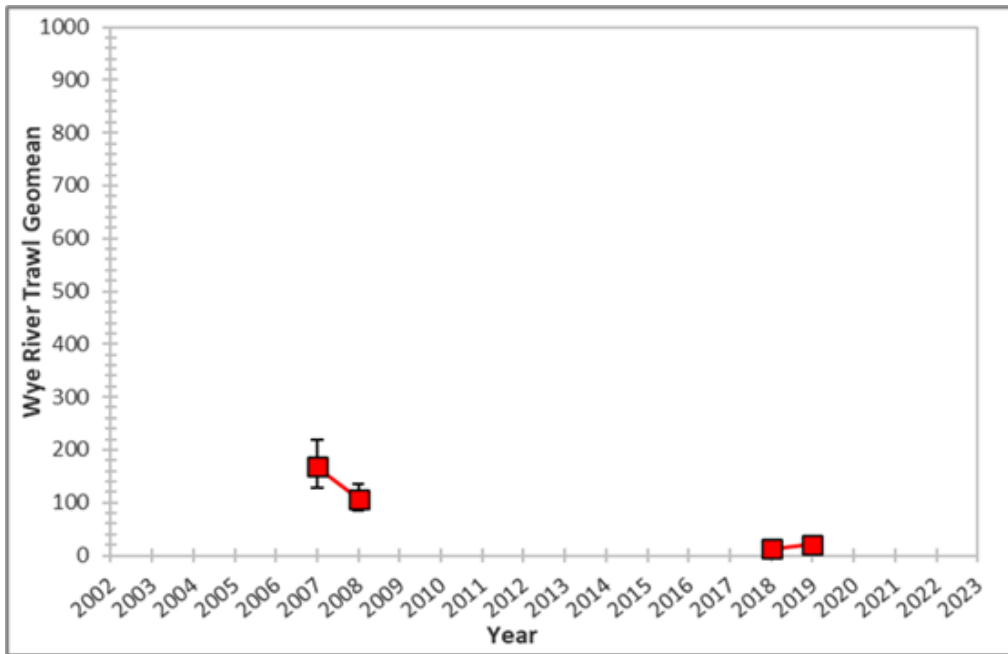


Figure B9. Annual beach seine catch geometric mean (GM) of all finfish species (red squares) for mid-Bay subestuaries, by sampling year. Black bars indicate the 95% confidence intervals.

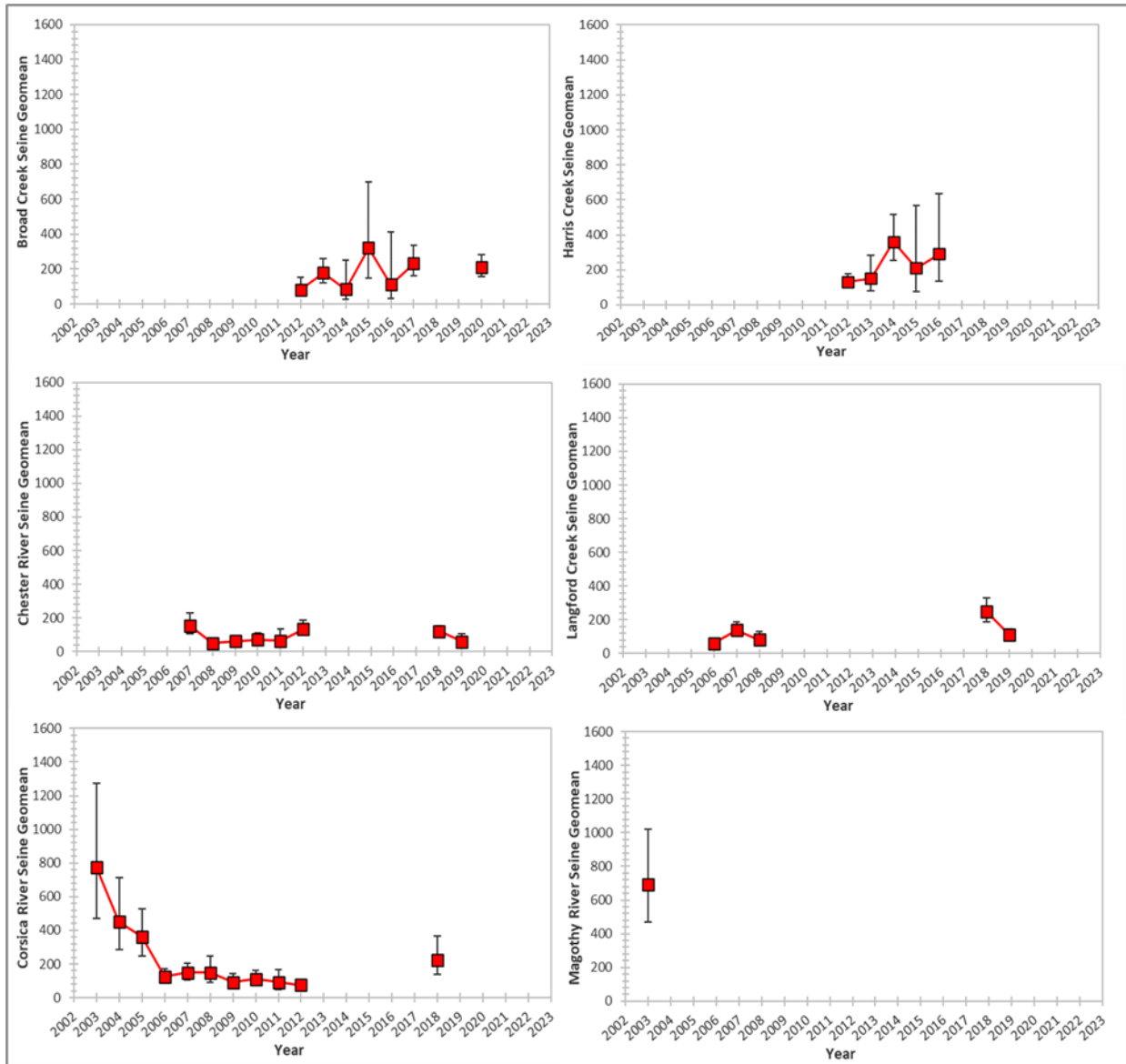


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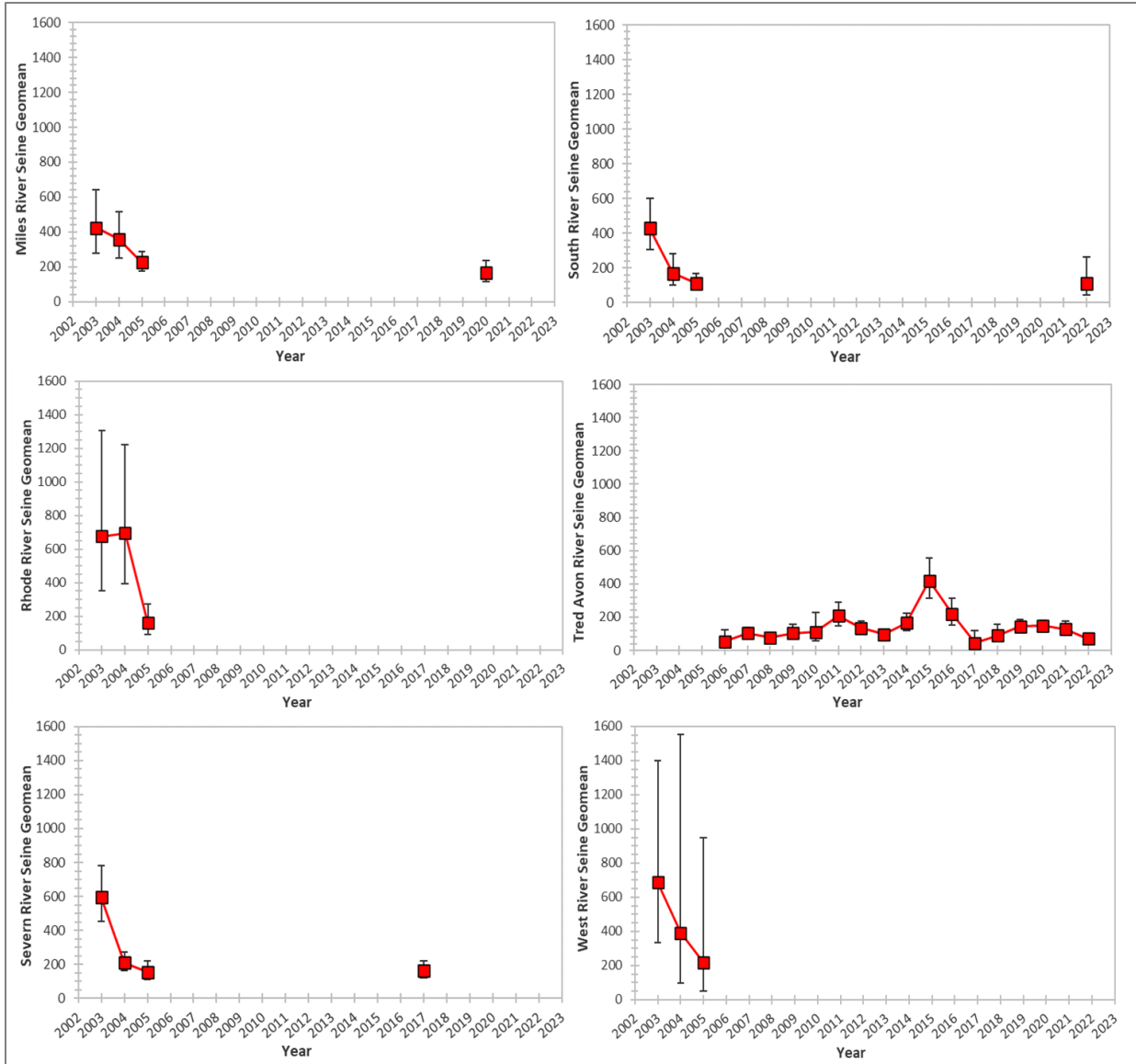
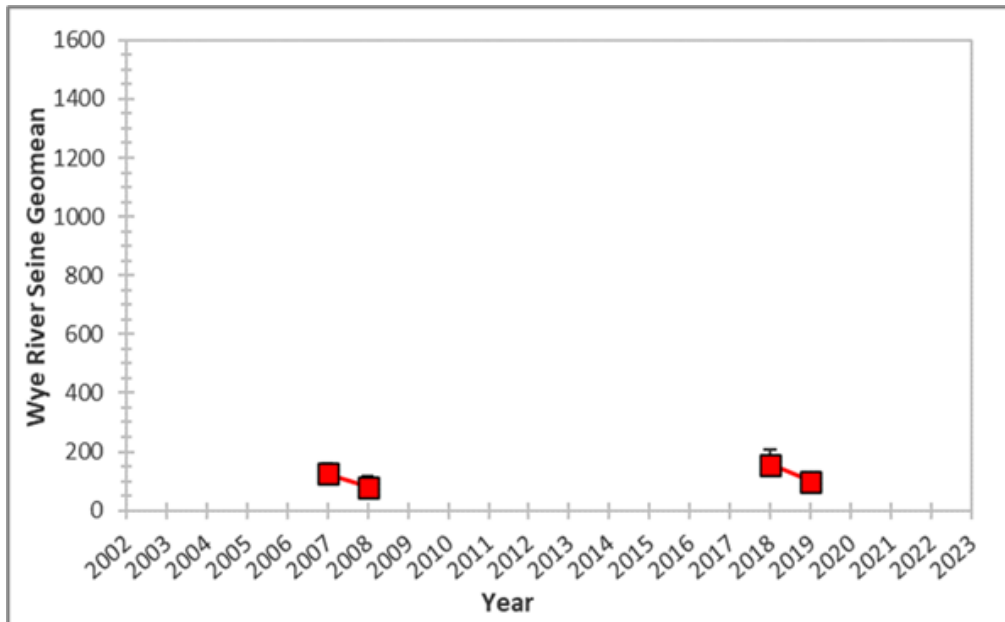


Figure B9. Continued.



Appendix C – Potomac River Region

Figure C1. Map indicating the locations of Potomac River tributaries sampled, Breton Bay, Mattawoman Creek, Nanjemoy Creek, Piscataway Creek, St. Clements River, and Wicomico River, during 2003 – 2022.

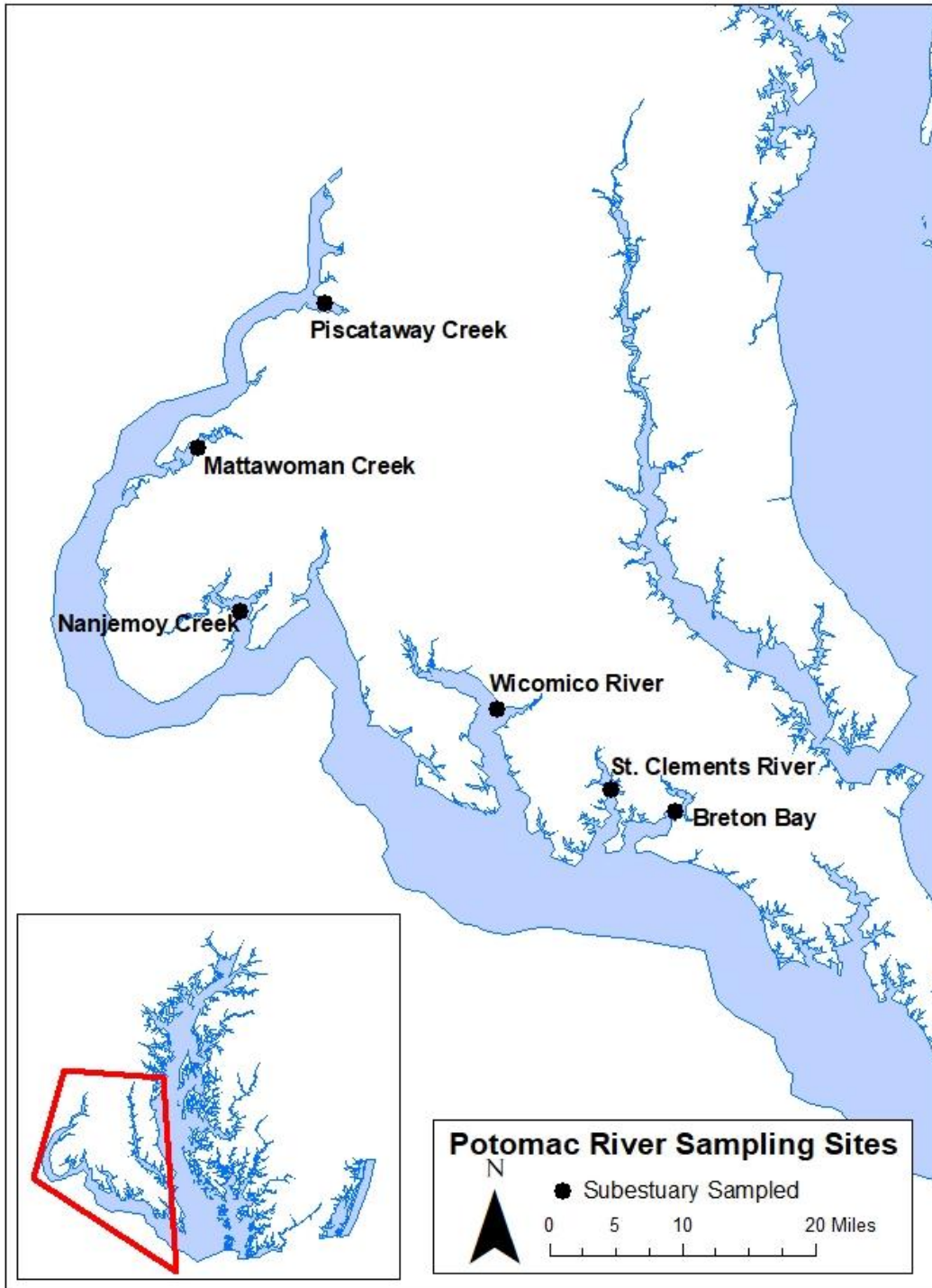


Figure C2. Median bottom temperature (°C; red squares) by year sampled for Potomac River tributaries. Solid black bars indicate range of all bottom temperature measurements for that year.

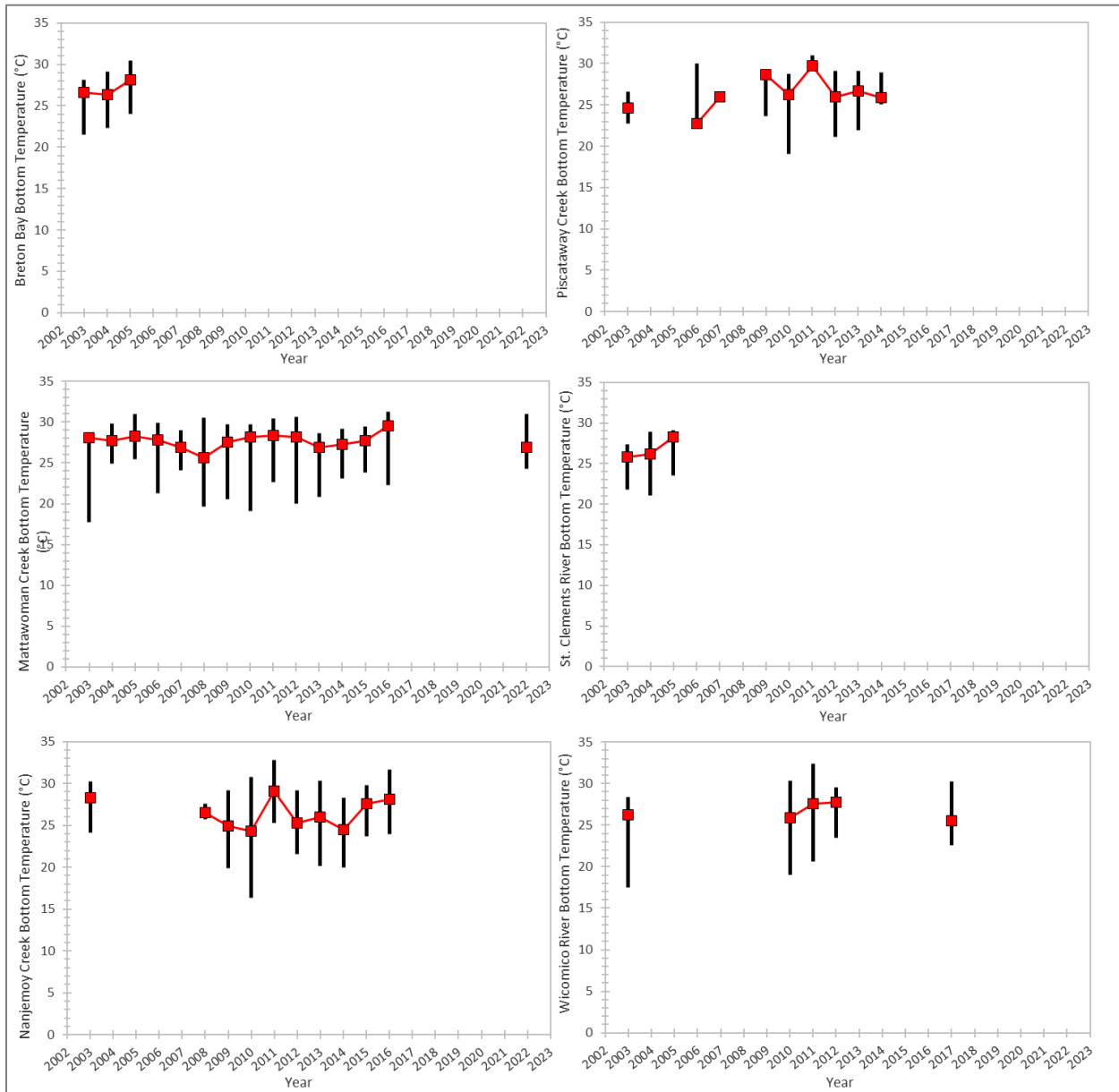


Figure C3. Median bottom dissolved oxygen (DO; red squares; mg / L) by year sampled for Potomac River tributaries. Solid black bars indicate range of all bottom DO measurements for that year.

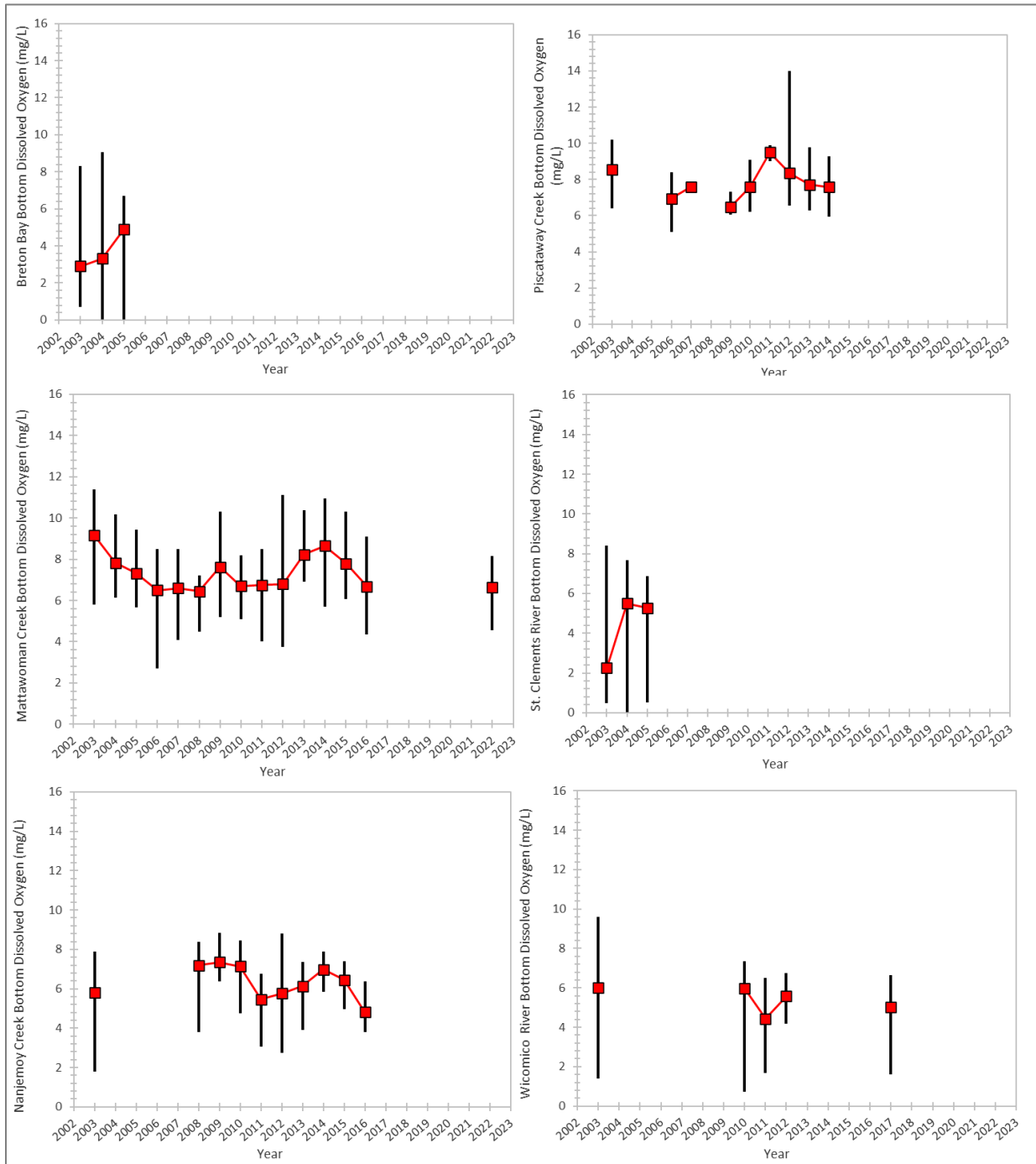


Figure C4. Median Secchi (m; red squares) depths by year sampled for Potomac River tributaries. Solid black bars indicate range of all Secchi measurements for that year.

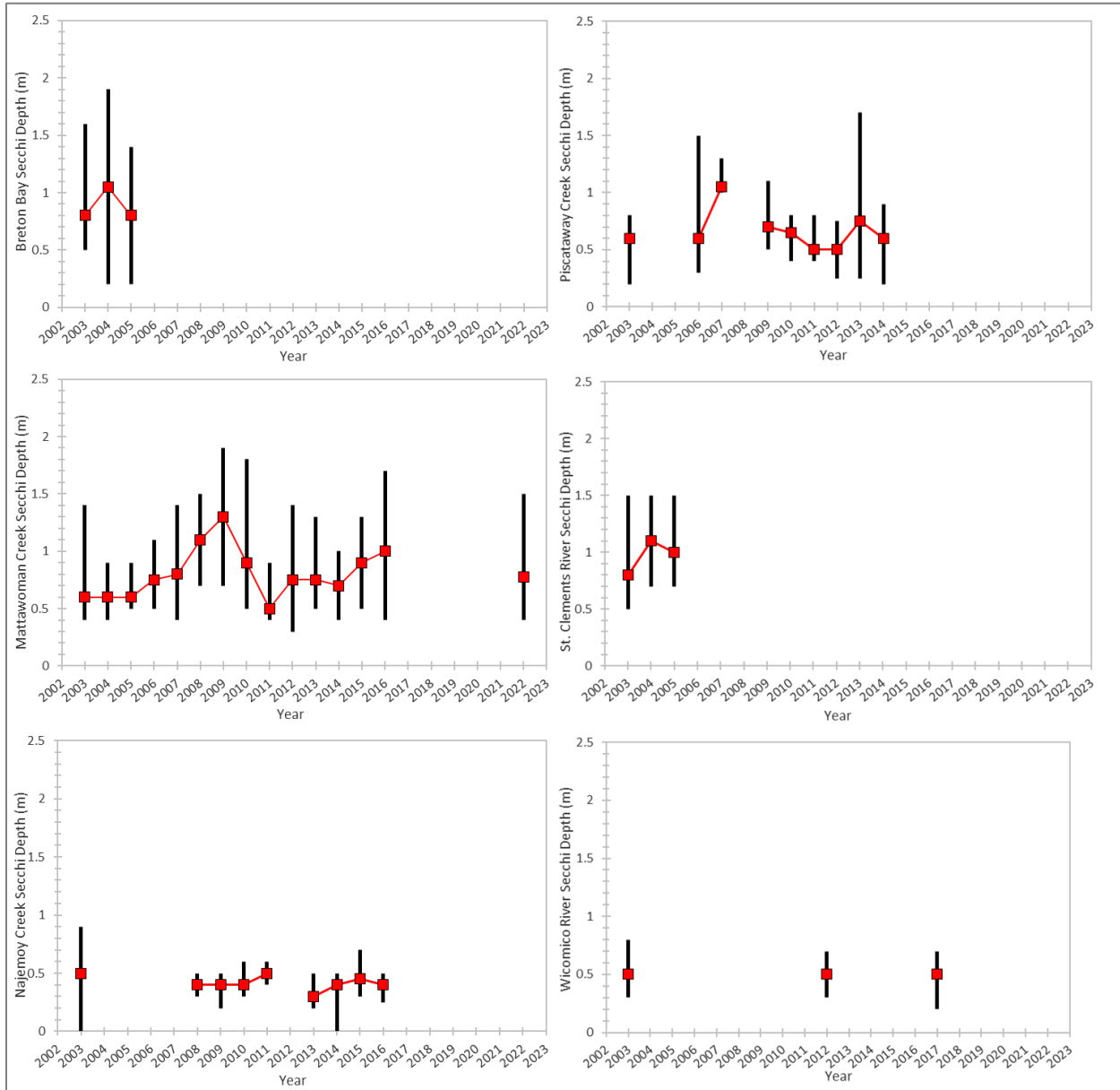


Figure C5. Median bottom pH (red squares) for Potomac River tributaries, by sampling year. Solid black bars indicate the range of pH measurements by year. No pH measurements were recorded prior to 2006.

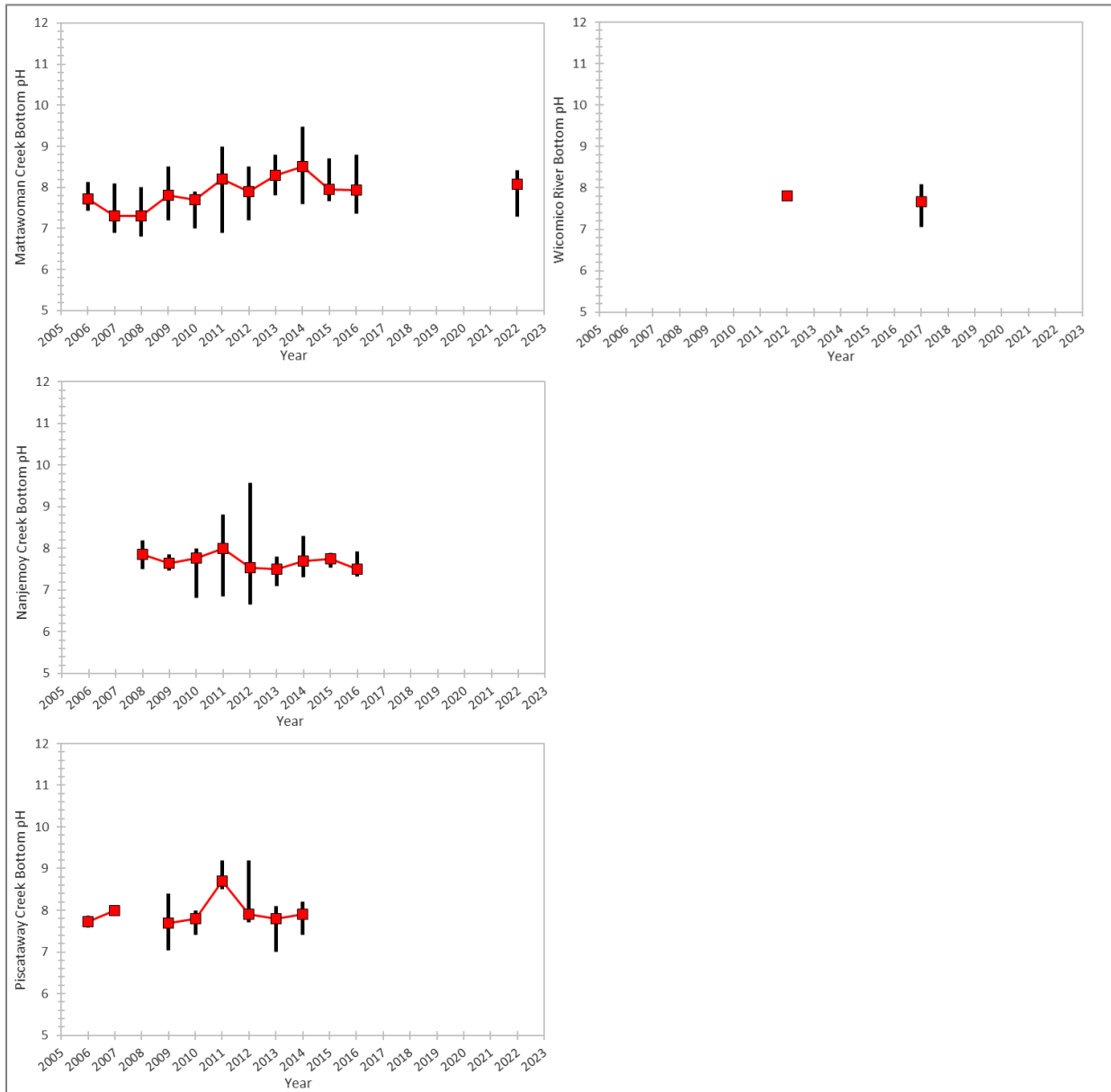


Figure C6. Median bottom salinity (ppt; red squares) by year sampled for Potomac River tributaries. Solid black bars indicate range of all bottom salinity measurements for that year.

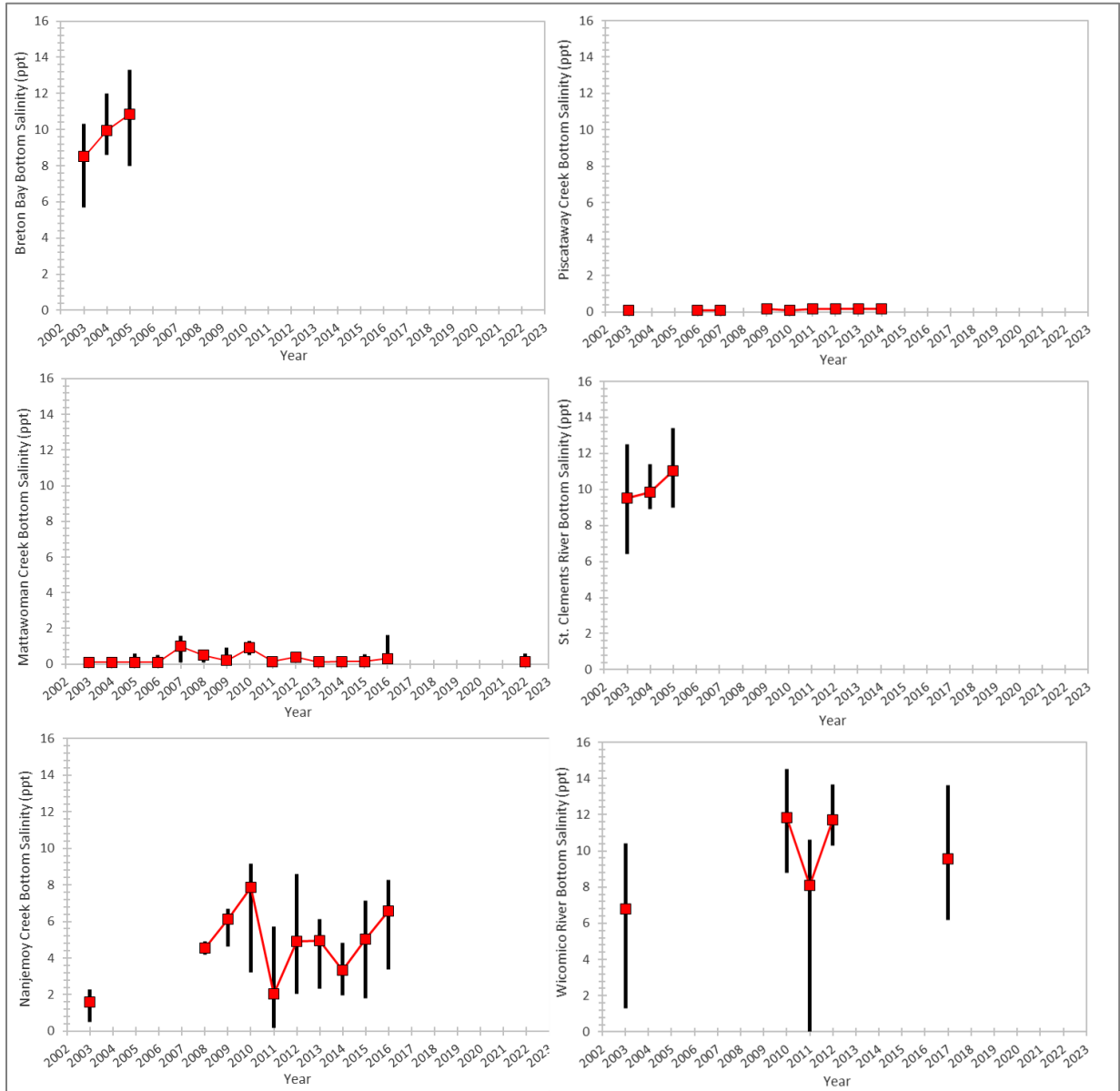


Figure C7. Bottom dissolved oxygen (DO; mg / L) readings (2003 – 2022) in Potomac River tributaries versus intensity of development (C / ha = structures per hectare). Target (5 mg / L) and threshold (3 mg / L) boundaries are indicated by red dashed lines.

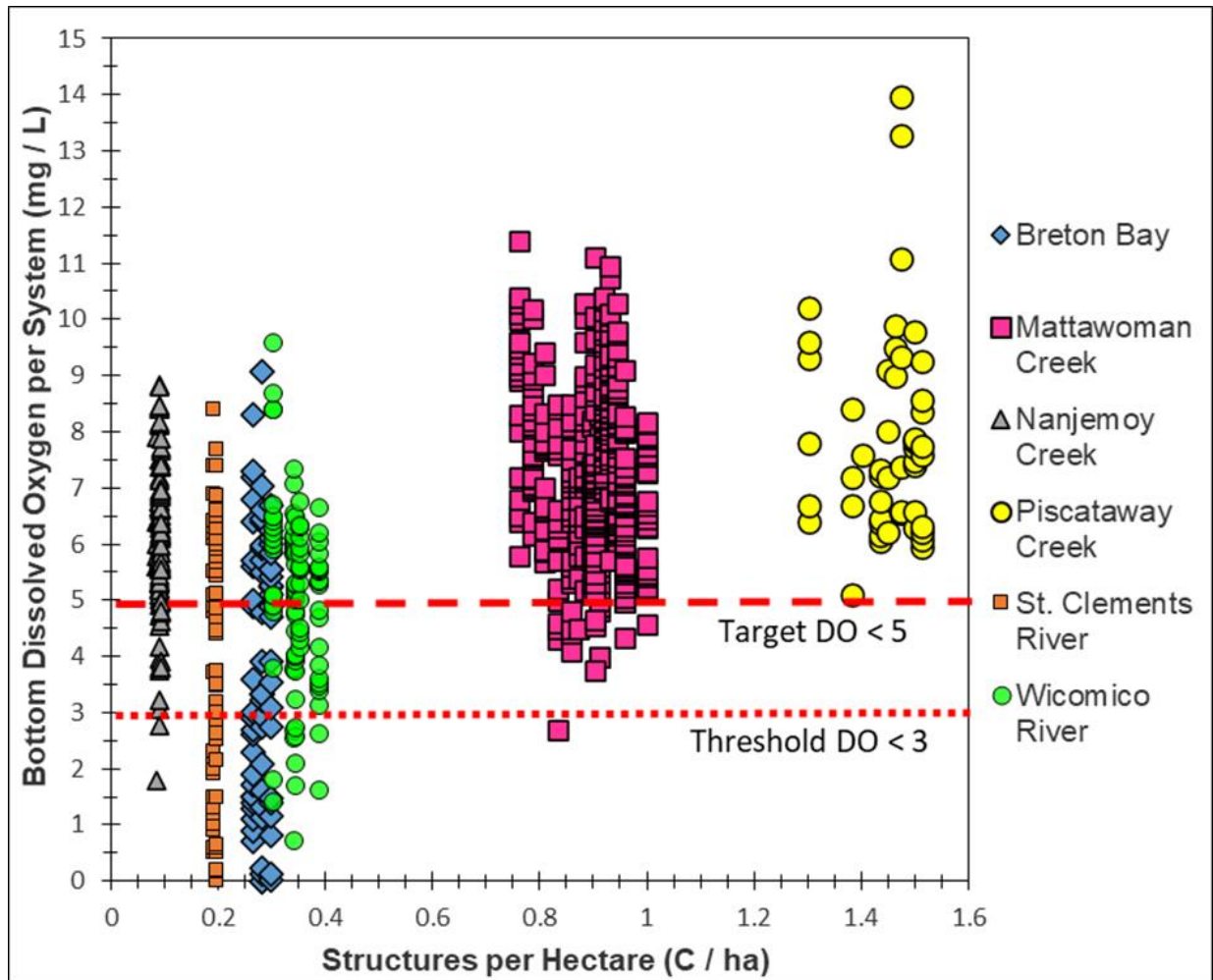


Figure C8. Annual 4.9 m bottom trawl catch geometric mean (GM) of all finfish species (red squares) for Potomac River tributaries, by sampling year. Black bars indicate the 95% confidence intervals.

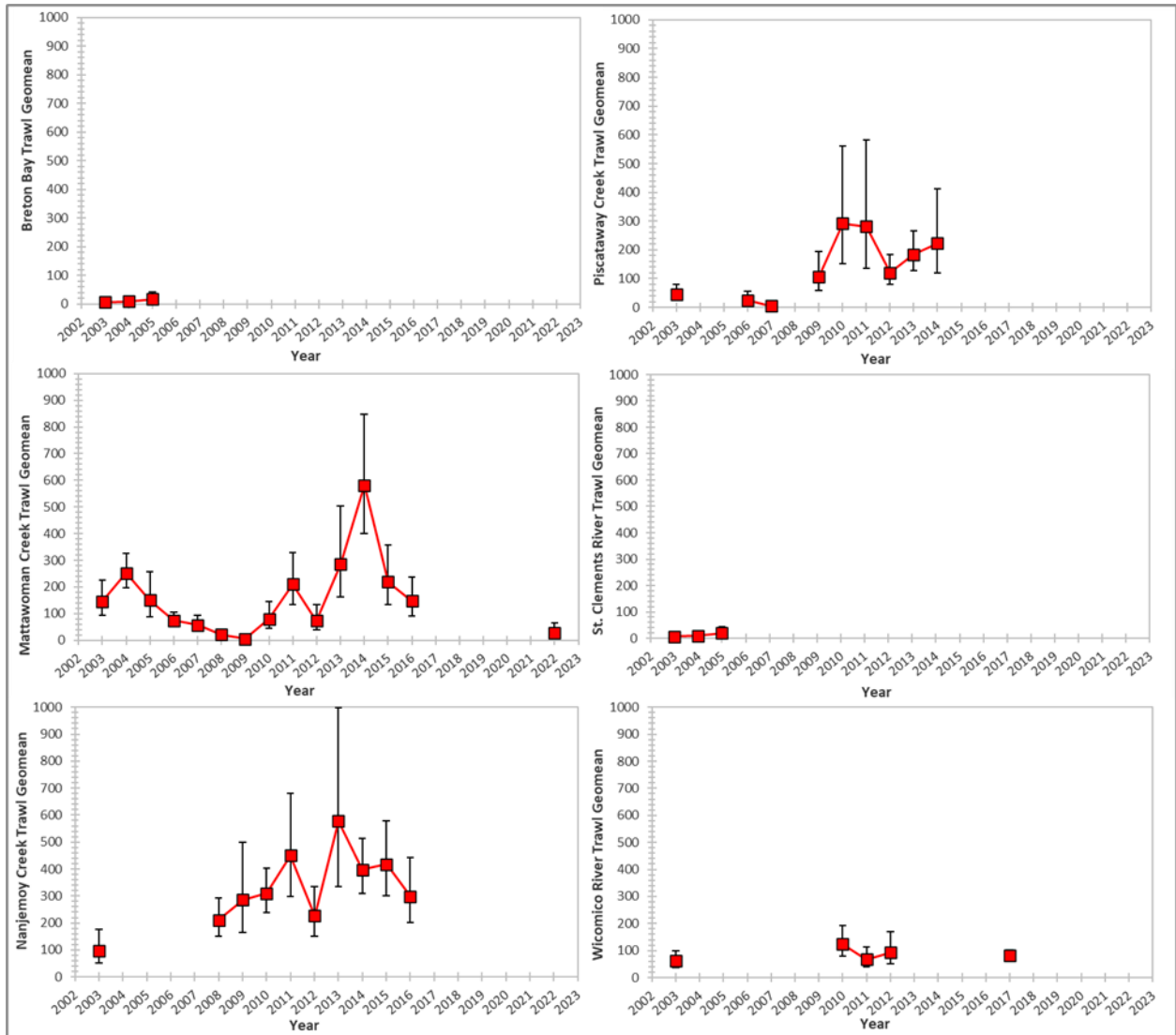
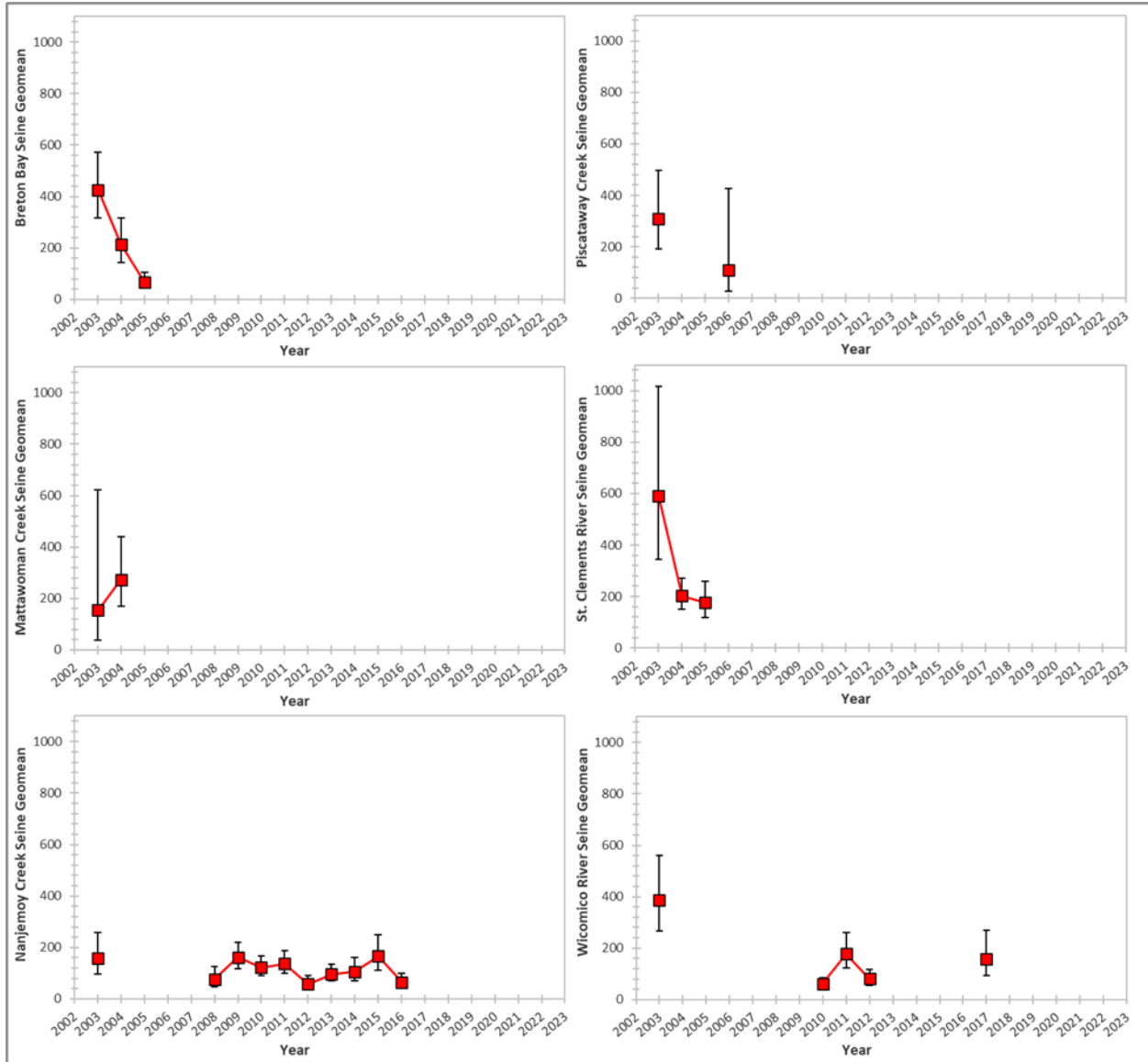


Figure C9. Annual beach seine catch geometric mean (GM) of all finfish species (red squares) for Potomac River tributaries, by sampling year. Black bars indicate the 95% confidence intervals.



MD - Marine and estuarine finfish ecological and habitat investigations

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

Project Staff

Jim Uphoff
Margaret McGinty
Alexis Park
Carrie Hoover
Tyson Johnson

Introduction

Project 2 documents participation by the Fisheries Habitat and Ecosystem Program (FHEP) in habitat, multispecies, and ecosystem-based management forums that relate to recreationally important finfish in Maryland's Chesapeake Bay and Atlantic coast during July 1, 2022 - June 30, 2023. These activities used information generated by F-63 or were consistent with the goals of F-63.

Margaret McGinty retired on December 1, 2022, and her activities under Project 2 and F-63 were reassigned for completion to remaining staff. We have been working to replace her position.

Fisheries Habitat and Ecosystem Program Website – We continued to update the website with project developments and publications. The website can be found at <https://dnr.maryland.gov/fisheries/pages/fhep/index.aspx>.

Publications – A paper entitled *Perspective Comes with Time: What Do Long-Term Egg and Juvenile Indices Say About Chesapeake Bay Striped Bass Productivity?* (J. Uphoff) was accepted for publication in *Marine and Coastal Fisheries*. This paper is based on work presented in F-63-R-10. Publication was delayed because it was part of a themed issue on Striped Bass and other papers were not ready. A link to the paper may be established by the time this annual report is published. This themed issue resulted from papers presented at the 2021 national meeting of the American Fisheries Society. J. Uphoff has served as an associate editor for this themed issue.

J. Uphoff reviewed a manuscript on decadal-scale changes in Hudson River Striped Bass spawning strategies for the *Marine Ecological Progress Series*.

Our segment 5 annual report (F63-R-5) has exceeded 100 reads according to ResearchGate and the segment 4 report has over 40.

Fish Habitat Conservation - We maintain an environmental review database, adding additional literature as it becomes available. Older reports that are not in electronic format are scanned and saved. Program staff continue to track research and literature regarding restoration effectiveness.

Comprehensive growth plans for the towns of St. Michaels, Aberdeen, and Centerville, and Howard County were reviewed by A. Park.

We provided comments on a recirculating salmon aquaculture facility proposed for Marshyhope Creek in Federalsburg. Marshyhope Creek supports anadromous fish spawning and spawning of Atlantic Sturgeon was of particular concern. Issues identified were potential temperature and salinity blockages from discharges and discharge of geosamin, a metabolic

waste product that imparts an off-flavor. M. McGinty with M. Topolski developed anadromous spawning maps for Marshyhope Creek to support the review process related to the proposed Salmon Hatchery. The company withdrew its application for the plant because of public opposition.

We reviewed proposed regulations changes for Critical Areas Commission Anadromous Spawning habitat protections. Regulations are periodically updated, and the last update was 2010. Updates involve modernizing terminology and information and providing updated references and research.

We supplied locations of anadromous fish spawning areas in non-tidal waters on state lands to define natural areas for the *Irreplaceable Natural Areas Bill* (HB 784). Specifically, river herring spawning tributary segments present on or downstream of state managed lands were identified.

We provided information on anadromous fish spawning and level of development in the Pomonkey Creek and Poorhouse Run watersheds for environmental reviews.

Cooperative Research and Monitoring - J. Uphoff met with researchers from University of Maryland on their Sea Grant pre-proposal to investigate what the fishery related ecological and habitat potential is for subestuaries in Baltimore. After the initial meeting, J. Uphoff wrote a letter of support for a UMCES proposal for research on the aquatic ecology of Baltimore Harbor. This proposal was not funded.

We developed land use/land cover estimates among various zoning categories within customized watersheds for use by Fish and Wildlife Health Program staff studying White Perch. Generalized zoning maps were completed for all watershed segments - Sassafras River, Marshyhope Creek, Wicomico River (eastern shore), Patapsco River, Potomac River, and James River systems. Accuracy of zoning data assembled was highly variable. Land use/land cover statewide datasets are currently being mosaiced prior to being overlaid on the zoning maps and summarized.

J. Uphoff met with University of Maryland researchers on their preproposal for combining automated sampling and machine learning for research on the effect of climate change on zooplankton and Striped Bass eggs and larvae. After the initial meeting, J. Uphoff reviewed and made comments on their proposal for sampling zooplankton and Striped Bass eggs and larvae simultaneously using a towed PlanktonScope in Choptank River and Patuxent River in 2024. Zooplankton, Striped bass eggs and larvae will be recognized and enumerated using an end-to-end deep learning algorithm and their sizes will be measured using a key-point detection based deep learning algorithm. This proposal was accepted for funding.

J. Uphoff met Catherine Fitzgerald, a PhD candidate at University of Maryland's Horn Point Lab, about analysis of Choptank River larval Striped Bass feeding data from 2017-2018 collections. These data, analyzed with dietary DNA could potentially be compared to feeding data collected in the 1980s and this year (2023) to investigate whether poor feeding success has been behind poor year-class success in recent years. C. Hoover rekeyed the 1980s feeding data so that comparisons could be made.

A. Park provided information and historical summer seine and trawl data from the Bush River to Anita C. Leight Estuary Center volunteers and staff to continue to develop their fish community database.

Presentations and Outreach – J. Uphoff took part in multiple presentations with Resource Assessment Service staff given to FABS leadership, the Striped Bass Program, and various Chesapeake Bay Program elements on the past, present, and future of temperature and dissolved oxygen conditions for resident striped bass during spring-fall. These presentations use Striped Bass temperature and dissolved oxygen criteria developed for F-63-R-11. While dissolved oxygen conditions have been relatively steady since 1986, water temperatures have increased since the 2000s. This has increased the frequency and duration of stressful habitat conditions. Loss of summer habitat for resident adult Striped Bass has been modest, but not trivial, up to this point. Judging the actual impact of these habitat changes on abundance and health of Striped Bass is difficult because of interactions with other factors influencing mortality such as catch-and-release, mycobacteriosis, fish condition, and fish size.

J. Uphoff attended the annual meeting of the Tidewater Chapter of the American Fisheries Society in Solomons and presented *Temperature and flow conditions associated with declining Striped Bass recruitment in Maryland's spawning areas*.

J. Uphoff gave two brief presentations on F-63 Program work to a group of state and federal congressional staffers at Oxford Lab.

Interjurisdictional Management – M. Toposki attended ASMFC meetings of the Habitat Committee and Atlantic Coast Fish Habitat Partnership (ACFHP). The Habitat Committee was given updates on ACFHP; the Northeast Regional Habitat Assessment; bluefish benchmark habitat section; Acoustics, Habitat Hotline, and Fish Habitats of Concern documents; and aquaculture efforts in the northeast among other topics. An ACFHP meeting was devoted to strategic and action planning for the next two and five years.

J Uphoff presented a summary on the Steele-Henderson prey-predator model to the ASMFC's Ecological Reference Point Workgroup. The Ecological Reference Point workgroup will prepare analyses of forage reference points for Menhaden along the Atlantic coast for the next benchmark stock assessment.

We provided comments on a Northeast Habitat Assessment tool developed for the Mid-Atlantic Fisheries Management Council. Four problem areas were identified: data accuracy, data interpretation, management utility, and the role of state agencies expertise in habitat assessment. There was a follow up phone conversation.

A. Park and M. Toposki participated in the Spring meeting of the Chesapeake Bay Program's Fish Habitat Action Team.

Traffic Light Index (TLI) - J. Uphoff worked with other FABS staff to develop a TLI for communicating Atlantic Menhaden and resident Striped Bass balance in Maryland's portion of Chesapeake Bay. It uses a three-color scheme patterned after familiar traffic lights to classify indicators as good or safe (green), intermediate or uncertain (yellow), and unacceptable or poor (red). A traffic light index for Menhaden's forage role in Maryland's portion of Chesapeake Bay based on indices can provide a cost-effective and timely alternative approach to communicate Menhaden's past and current forage status in Maryland's portion of the Bay. Despite the finding of a healthy coastal stock, there are concerns among stakeholders that harvest from the reduction fishery is too high within the Chesapeake Bay (or Bay), depleting Atlantic Menhaden (or Menhaden) locally and harming predators that rely on them. The TLI provides a means of

presenting indicators used by FABS to the public. J. Uphoff presented the Menhaden traffic light index for Maryland's portion of the Bay to one of Maryland's ASMFC commissioners. Development is ongoing and internal presentations and reports have been generated. Approval by other Bay jurisdictions and peer-review are needed for the TLI to become operational.

Envision the Choptank – Envision the Choptank (<https://www.envisionthechoptank.org/>) is a collaboration of conservation organizations, government agencies (town, county, state, and federal), and local citizens that work to maintain and improve the viability of the Choptank River's water quality and natural resources. We participated in virtual meetings of the Envision the Choptank Working with Local Government workgroup. Engaging in county and town comprehensive plan updates to strengthen natural resource components is a priority of this workgroup and we hope the Envision effort will lead to more consideration of fish habitat in county planning.

Training - A. Park attended and completed NOAA vessel training at Oxford Lab and a basic Yamaha outboard mechanics course at Chesapeake College.

MD - Marine and estuarine finfish ecological and habitat investigations
Project 3: Develop spatial data to assist in conserving priority fish habitat.
Refining anadromous fish non-tidal stream spawning maps for management applications

Marek Topolski

Abstract

Geospatial datasets (feature layers) depicting historical occurrences of anadromous fish spawning in Maryland tributaries were initially mapped in 2002 and iteratively revised by the Maryland Department of Natural Resources' Fish Habitat and Ecosystems Program (FHEP) over the past decade to improve site location accuracy and include species presence and absence in samples. Additionally, contributing streams and stream networks that support and/or influence anadromous fish spawning were identified. Utilization of these feature layers by state and federal staff who review environmental disturbance permit applications has been hampered by distribution issues and confusion with earlier versions of these data available. The historic anadromous fish spawning occurrence feature layer curated by the FHEP is the authoritative geospatial dataset and should be the primary basis of maps used by staff conducting environmental review of project permit applications. Currently, the data must be electronically distributed to each environmental review staff member and requires access to a desktop or web geographic information system (GIS). To ensure seamless distribution and use of these anadromous fish spawning feature layers (including future revisions), a simple web map is being created that can be accessed on desktop and mobile devices, centralizes the feature layers which obviates the need to electronically distribute files to environmental review staff, allows for existing feature layers to be seamlessly revised, and provides mechanisms to include supplemental ecological and environmental data. Supplemental data currently included in the web map are estimates of percent impervious surface within both Maryland 8-Digit and 12-Digit watersheds; percent impervious surface is negatively related to spawning success of anadromous fish. The web map, which will not require specialized training to use, will primarily focus on non-tidal anadromous fish spawning tributaries at this time.

Introduction

Print maps were created in the 1970s – 1980s to designate spawning areas of anadromous fish (American Shad, Alewife Herring, Blueback Herring, White Perch, and Yellow Perch; Uphoff et al. 2020). These maps were based on historical surveys data of eggs, larvae, or adults (and in some cases juveniles) present on spawning grounds during spawning season (Uphoff et al. 2020). The resulting maps represented historical (1970s and 1980s) and legal baseline spatial distributions of spawning habitat occupation.

Digital maps of Maryland anadromous fish spawning locations have undergone several iterations since they were first produced by Mowrer and McGinty (2002). Initial feature layer creation included only observations of species presence. An update in 2019 included the historic records of species absence at sampled stream stations and refined sample site locations (Uphoff et al. 2020).

Maryland Department of Natural Resources environmental reviews refer to these maps when applying restrictions (primarily time of year restrictions) to minimize impacts to anadromous fish spawning areas. Likewise, partner agencies (identified collectively as the Interagency Review Team or IRT) use these maps to limit habitat impacts, but also to identify potential locations to apply mitigation approaches. The IRT is a multi-agency team made up of members representing federal and state agencies including U.S. Army Corps of Engineers (ACOE), U.S. Environmental Protection Agency (EPA), U.S. Fish and Wildlife Service (USFWS), National Oceanic and Atmospheric Administration (NOAA), Maryland Department of Environment (MDE), Maryland Department of Natural Resources (MD DNR), Maryland Historical Trust (MHT) and Maryland's Critical Areas Commission (CAC). The team meets regularly to collaborate on permit reviews to promote agreement among agencies to streamline the review process, while assuring consistent use of conservation tools to promote minimal disturbance or mitigation for permitted projects.

Margaret McGinty worked with a subset of IRT participants (IRT mapping workgroup included NOAA, MDE, DNR, USFWS) during 2019 until her retirement in 2022 to develop updated maps that met their needs for information the IRT requested on non-tidal streams that support anadromous fish spawning (Uphoff et al. 2020; 2022). I have continued this effort and this report is an update on my collaboration with George Edmonds (Oxford Lab) to create an accessible, web-based mapping tool. In addition to updating spawning (egg, larval, and adult presence-absence) data, these maps depict watershed impervious surface levels to better reflect habitat potential.

Anadromous fish spawning tributary maps support resource protection efforts, via the permit review process, against adverse impacts that result from instream disturbances, shoreline and land development, and other landscape alterations. Anadromous fish spawning is influenced by urbanization (Limburg and Schmidt 1990; Limburg and Waldman 2009; Uphoff et al. 2022a) and the presence of early life stages can be predicted based on the extent of impervious cover, a commonly used metric of land development, in a watershed (Hare et al. 2021; Uphoff et al. 2022a; see **Common Background for Project 1, Sections 1-3**). The impact of watershed development on aquatic habitat quality is not limited to the non-tidal tributaries but extends downstream into the estuary (Uphoff et al. 2011, 2022a). Percent impervious surface in Chesapeake Bay drainages has increased since the collection of the historic anadromous fish spawning habitat occupation data (Hare et al. 2021; Uphoff et al. 2022a; see **Common Background for Project 1, Sections 1-3**). Existing development within a watershed is taken into consideration during the permit review process, so percent impervious surface estimates derived from housing density (see **Project 1, General spatial and analytical methods**) were calculated for different watershed scales: MDE 8-Digit and DNR 12-Digit.

Web Map Development

Feature layers that demarcate historical anadromous fish spawning areas were initially accessible only via a desktop GIS platform, such as ESRI's ArcGIS Pro or the open source QGIS, but were subsequently added as feature layers to the online MDDNR Aquatic Resources Pre-Screening Tool

<https://www.arcgis.com/apps/webappviewer/index.html?id=1c1095e641c541d8aa6588ef6c1b23>

c8) web map used to evaluate potential environmental impacts from proposed projects to aquatic resources. This management tool is heavily focused on tidal waters, so despite revisions (which most affect non-tidal sites) to the feature layer of historic anadromous fish spawning locations, there is no immediate plan to use the revised feature layer. Furthermore, this web map does not include feature layers that depict which non-tidal tributaries support and/or influence anadromous fish spawning nor estimates of watershed impervious surface. Exclusion of corrected non-tidal spawning distribution data and current estimates of watershed development may influence project review outcomes.

I chose to implement a web map interface to facilitate access to these feature layers. Web maps are configured to limit how data are accessed, visualized, and analyzed, so they are not equivalent to a web GIS which replicates the functionality of a desktop GIS environment (Veenendaal et al. 2017). Web maps, as described by Veenendaal et al. (2017), began with static maps having hyperlinks in the 1990s and have evolved into customizable interfaces that can integrate time enabled, crowd sourced, public, big data such as imagery, cloud services, analytics, and machine learning. Implementation of a web map provides an intuitive interactive map of documented anadromous fish spawning habitat occupancy, tributary networks that influence those spawning areas, and watershed development. The map would be accessible from a desktop or mobile device using internet browser software negating the need for a desktop or web GIS as is currently required. Data updates would be uploaded to the web map, rather than distributed direct to the user, thereby ensuring that project review uses the most up-to-date data. The platform being used for web map creation is ESRI's ArcGIS Online (AGOL); a web GIS that has the capacity to develop and deploy a variety of data and tools including web maps. Implementation is a two-step process: 1) develop map cartography and 2) design the user interface.

Map cartography encompassed all facets of the map design such as which feature layers to display, appropriate symbology, relevant attribute tables, and metadata. Feature layers included the historic survey data (point feature), anadromous fish spawning streams surveyed and impeded or unimpeded contributing streams (line features), and Maryland 8-Digit or 12-Digit watershed boundaries attributed with watershed development estimates (polygon features). All layers were symbolized with contrasting colors of equal weight that also contrasted with imagery.

An “out of the box” web map template was used to design the map which included several standard widgets (simple applications or programs that add functionality or display specific types of information) such as map documentation (**About**), **Zoom in/Zoom out**, **Default extent**, **Find address or place**, and **Legend** (Figure 1). Feature layers can be turned on and off with a checkbox in the **Layer List** (Figure 2A); for layers turned on individual features could be selected by clicking on the map. When selected, a feature's attribute table appeared in a pop-up window. When multiple features were selected, the pop-up windows were “stacked” and could be cycled through using forward and backward arrows. Only attributes relevant for project review were made visible. For example, only a watershed's numeric code, name, and estimated percent impervious surface were shown in the pop-up window. Alternately, watershed attribute tables are accessible by a tab at the bottom of the map. Functionality of the web map template was expanded with the addition of several widgets. The **Basemap Gallery** widget provides a

variety of basemaps for use such as Light Gray Canvas, OpenStreetMap, Terrain, and Imagery (Figure 2B). Additional spatial data can be imported from online sources or uploaded from a local file using the **Add Data** widget to create custom overlays (Figure 2C), although the map cannot be saved with the added data layers. Development of a custom widget to save the web map with added data layers is being explored. A **Chart** widget was added allowing visualization of the percent impervious surface trend within a watershed during the years 1935-2020 (Figure 3). The last two years are provisional estimates due to latency in the release of tax data which the estimates are derived from. The terminal year will be updated approximately every two years as new tax data are released. The web map also includes a hyperlink to the Maryland Department of Natural Resources homepage.

Spatial Data Development

A draft version of the web map was shared with a limited number of environmental review staff from Maryland Department of Natural Resources to confirm the usefulness of the web map cartography and design. The web map is intended to emphasize historic presence and absence of anadromous fish at sampled locations and the tributary network that influences those same locations. Tributary segments adjacent to the revised sample locations in the historic spawning feature layer were identified as supporting anadromous fish spawning (Uphoff et al. 2022b). The mapped flowlines depicting these tributaries were derived from the Maryland Stream Health – Md Biological Stream Survey Stream Reach File (Versar, Inc. and Maryland Biological Stream Survey 2004) feature layer developed at a 1:100,000 scale from the U.S. Geological Survey’s National Hydrography Dataset (NHD). Anadromous fish spawning tributaries are influenced by the upstream network of tributaries. Non-tidal tributaries upstream of these spawning reaches likely to support or influence spawning activity were identified; tributaries upstream of blockages were denoted as inaccessible (Uphoff et al. 2022b). These tributary feature layers were derived from the Maryland Stream Health - Md Biological Stream Survey Stream Reach File (1:24,000) MBSS24k (Versar, Inc. and Maryland Biological Stream Survey 2021) developed at a 1:24,000 scale from the U.S. Geological Survey’s National Hydrography Dataset Plus (NHDPlus). One caveat is the tributary datasets do not include segments passing through lentic waters. Combined, the spawning location and tributary datasets were the basis for identification of anadromous spawning areas and contributing streams.

Inclusion of a catchment (drainage area for a tributary segment) feature layer was requested to provide reviewers with a refined spatial context of the landscape surrounding a proposed project site. Catchments have been delineated nationwide by the U.S. Geological Survey and distributed as the National Hydrography Dataset Plus High Resolution (NHDPlus HR) edition (U.S. Geological Survey 2018). Catchments from the NHDPlus HR that intersect Maryland (N=104,203) were clipped to the Maryland state boundary and open water was erased leaving a total of 102,139 catchments. Median catchment land area was 5 ha and the largest catchment was 1,638 ha. For comparison, the 1,119 12-Digit Maryland watersheds had a median land area of 1,913 ha and a maximum of 9,775 ha. Eight-digit watersheds are fewer in number and larger in area than the 12-Digit watersheds.

A catchment feature layer with percent impervious surface estimates was not included in the web map because of uncertainty in the estimation of percent impervious surface from just tax

data at the catchment scale. Spatially, tax data are located at the land parcel centroid rather than the primary structure's footprint. At the catchment scale, the potential influence of the spatial mismatch between observed impervious surface location (primary structure's footprint), tax data (parcel centroid), and catchment boundary becomes relevant. Based on 2017/2018 Chesapeake Conservancy land cover data (Chesapeake Conservancy 2022), 59,490 (58%) of the catchments had impervious surface present, and forty-seven percent (N= 48,086) of catchments had $\geq 1\%$ impervious surface. Thirty-seven percent of catchments had observed impervious surface and 2017 tax data (N = 37,795).

Observed percent impervious surface for each catchment was compared to the predicted percent impervious surface derived using the tax-based model (see **Project 1, General spatial and analytical methods**). Prediction error is the difference between observed and predicted values. Percent impervious surface prediction error increased as catchment area decreased, particularly below 250 hectares (Figure 4A), which indicated a greater likelihood of spatial mismatch between tax data coordinates and catchment boundary when watershed area decreases. Tax model prediction error was biased to underprediction (positive error, Figure 4B). Median prediction error was positive above the 0-5% impervious surface category and increased nonlinearly. Most outlier estimates were overpredictions (negative error).

Tax model prediction error approximated a Gaussian distribution and was evaluated using root mean square error (RMSE, Chai and Draxler 2014):

$$RMSE = \sqrt{(X_{\text{observed}} - X_{\text{predicted}})^2}$$

where X represents percent impervious surface. Among all catchments the RMSE was considerable at 9.7% impervious surface. The RMSE for catchment subsets (see Figure 4B) were 1.9 (N=77,053; 0-5% observed IS), 5.4 (N=10,717; 5-10% observed IS), 9.3 (N=4,691; 10-15% observed IS), 13.6 (N=4,388; 15-25% observed IS), 30.9 (N=4,183; 25-50% observed IS), 49.0 (N=861; 50-75% observed IS), and 82.2 (N=246; > 75% IS) percent impervious surface, respectively. These measures of prediction error, compared to their group specific range of observed percent impervious surface, indicate that current tax-based estimates of percent impervious surface for the catchment scale are spurious regardless of the extent of land development. Additional metrics and-or sources of impervious surface need to be considered in addition to tax data when predicting catchment scale percent impervious surface.

The most prominent sources of impervious surface not explicitly accounted for in the tax model are roads and parking areas. Roads are a significant contributor of impervious surface in Maryland where they constitute an estimated one third of all impervious surfaces; impervious surface from roads increases as development increases (Goetz et al. 2004). Fifty percent (N = 51,039) of the 102,139 NHDPlus HR catchments in Maryland (previously described) were intersected by roads based on 2017 U.S. Census Bureau TIGER/Line road centerline data (U.S. Census Bureau 2017). Road categories included primary, secondary, local, ramp, service, alley, private, driveway, and parking designations. Thirty-five percent (N = 35,447) of catchments contained both tax and road centerline data. Eight percent (N=7,732) of catchments with observed impervious surface were not intersected by tax or road data; conversely 63 (0.0006%) catchments were intersected by tax and road data without any observed impervious surface. I am developing a multivariate model to estimate catchment percent impervious surface using both tax

and road data. In the interim, estimation of impervious surface at the NHDPlus HR catchment scale using the tax index developed for larger spatial extents (12-Digit watersheds or larger) is not advised.

Future Direction

Maps depicting locations of anadromous fish spawning activity and streams that influence them are valuable during evaluation of proposed projects that have the potential to affect stream ecosystems. Several modifications for the web map have been identified and will be implemented. 1) The existing spawning tributary feature layer is based on dated stream flowlines from the NHD (1:100,000 scale). The non-tidal tributary feature layers, based on the NHDPlus HR (1:24,000 scale), have been modified by removal of flowlines through lentic waters (ponds and lakes) which interferes with stream network analysis. All feature layers depicting tributary flowlines will be revised to match those from the NHDPlus HR (U.S. Geological Survey 2018). 2) Initial feature layers lacked precise geographic coordinates for sample sites and so locations were approximated; sample site locations were later corrected with the current feature layer but are not snapped to stream flowlines. Sample locations along nontidal tributaries will be snapped to the NHDPlus HR flowlines to facilitate stream network analyses. 3) Impeded and unimpeded classification of streams will be reviewed with an overlay of updated data on barriers (dams and culverts) to fish passage. 4) Pending software capabilities, spawning data will be symbolized to denote which species were present and absent at a sample site and the map user will be given the option to query the sites by species (symbolizing both presence and absence). 5) A NHDPlus HR catchment feature layer with impervious surface estimates will be developed and added to the map. 6) Add 8-Digit and 12-Digit watershed feature layers with “traffic light” symbology for percent impervious surface categories: 0-5% (green), 5-10% (yellow), 10-15% (orange), 15+% (red).

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Figure 1. Screen shot of web app depicting non-tidal anadromous fish spawning data for the state of Maryland when first opened (disclaimer splash screen is not shown). Hyperlinks to pertinent background information regarding the data are shown with the **About** widget. The **Legend** widget shows which feature layers are visible and how they are symbolized. Widgets that allow for navigation of the web map (**Zoom in/Zoom out**, **Default extent**, and **Find address or place**) are in the upper left corner.

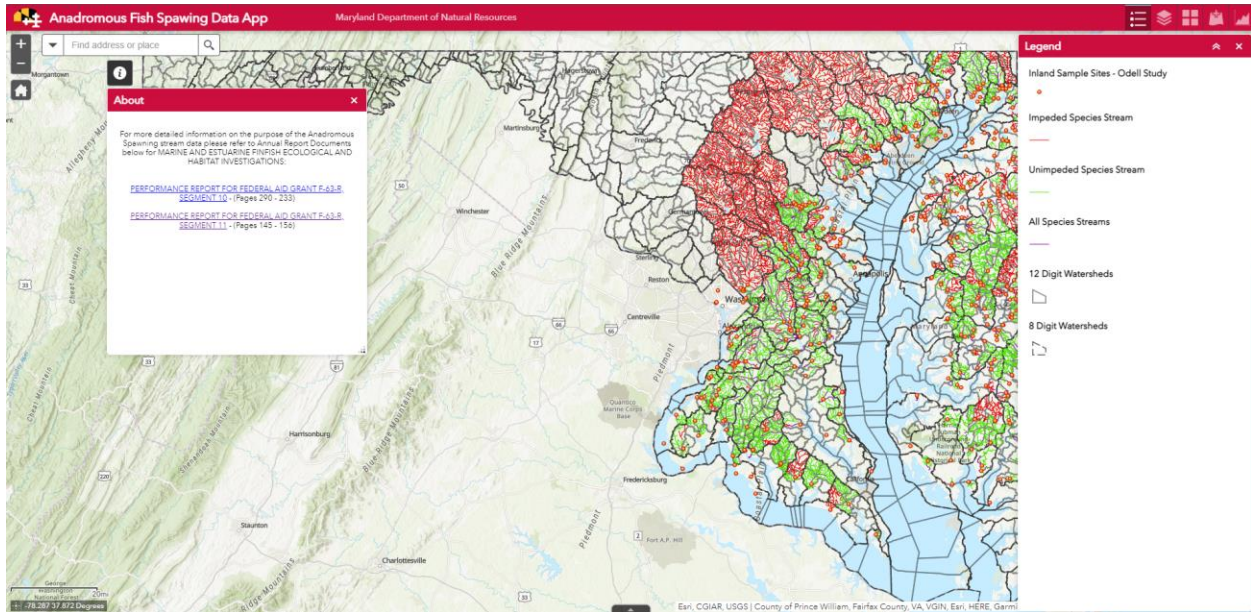


Figure 2. Screenshots of widgets that add interactive functionality to the web map. These widgets are located in the upper right corner of the web map. (A) **Layer List** widget that allows the user to toggle on/off each feature layer. (B) **Basemap Gallery** widget that allows the user to change the underlying map layer. (C) **Add Data** widget that enables external data to be uploaded into the web map for display.

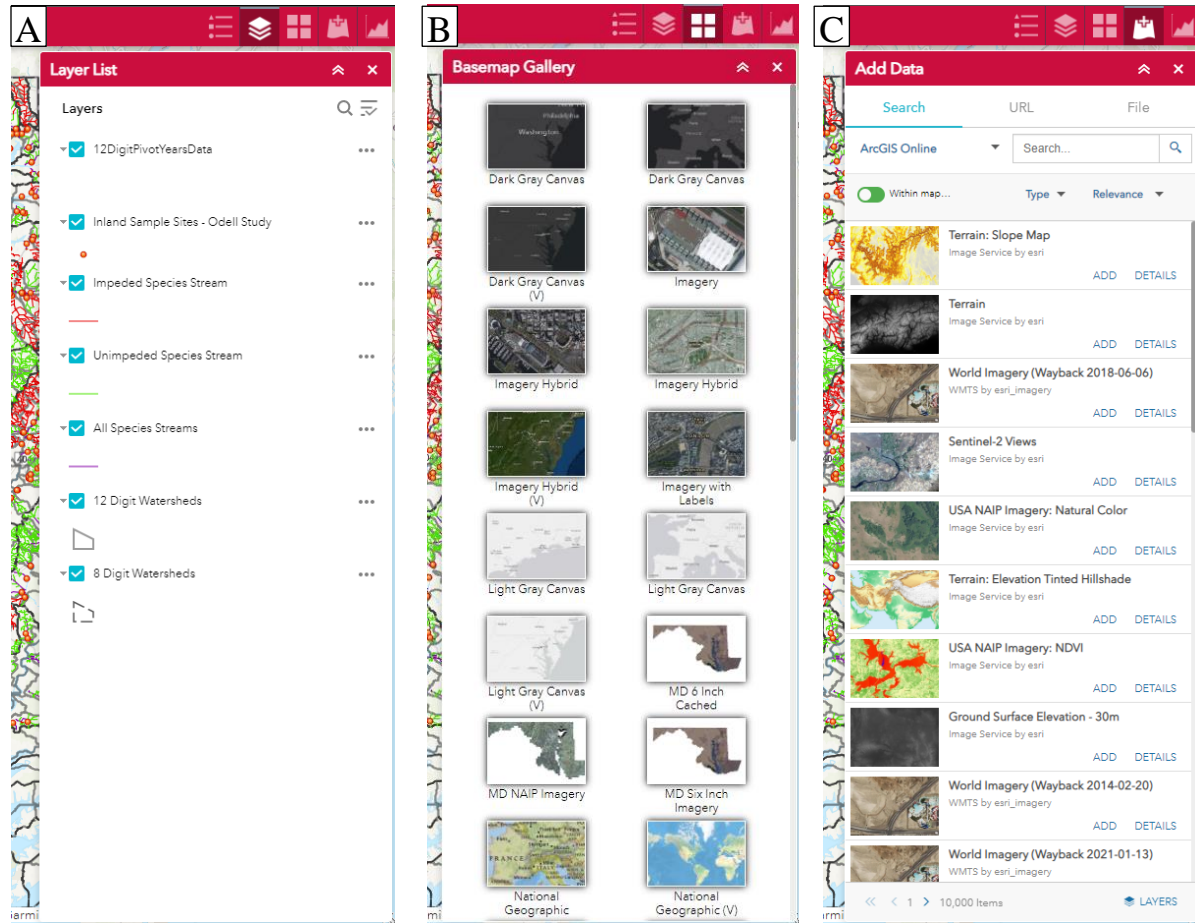


Figure 3. **Chart** widget depicting the percent impervious surface trend as estimated using the tax index. The slider at the top of chart allows the years of interest to be varied and the chart responds dynamically as the slider is adjusted. Percent impervious surface estimates can be viewed for each individual year by hovering the cursor over the chart trend line.

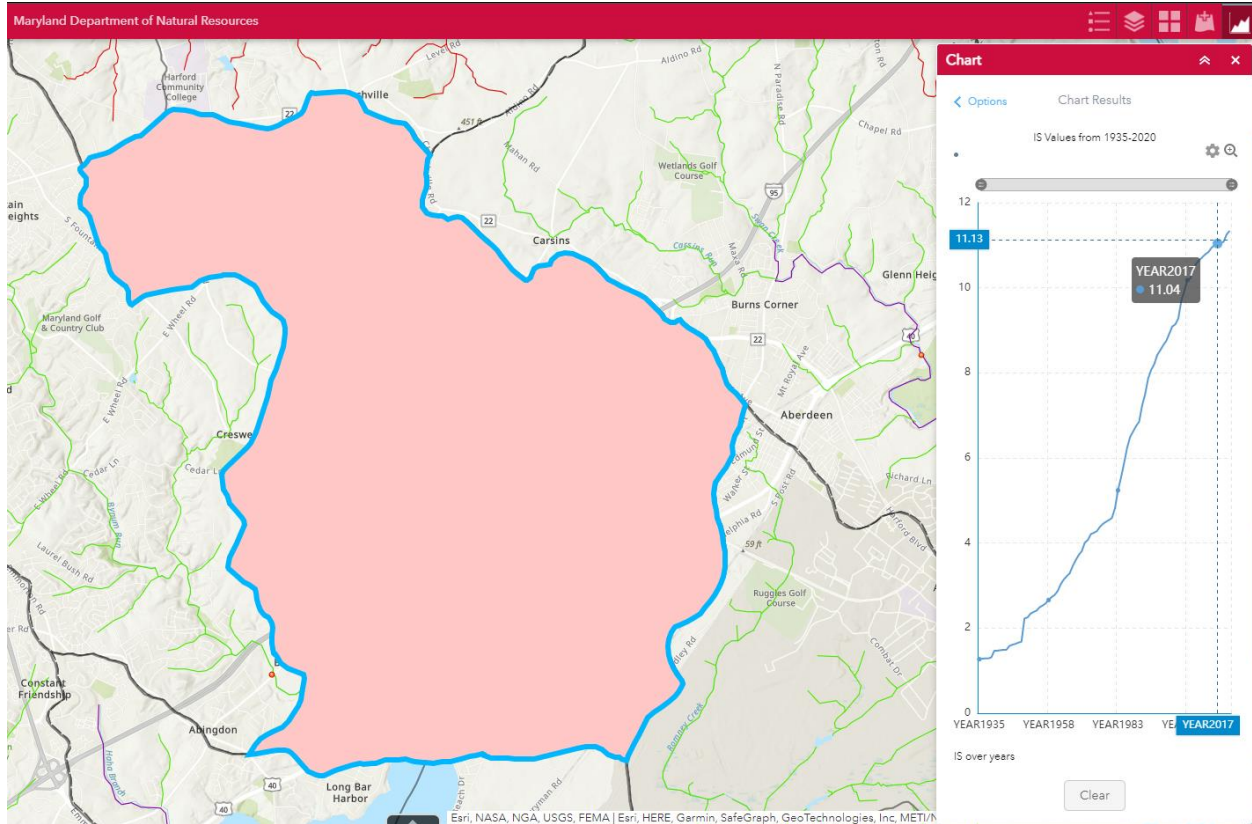
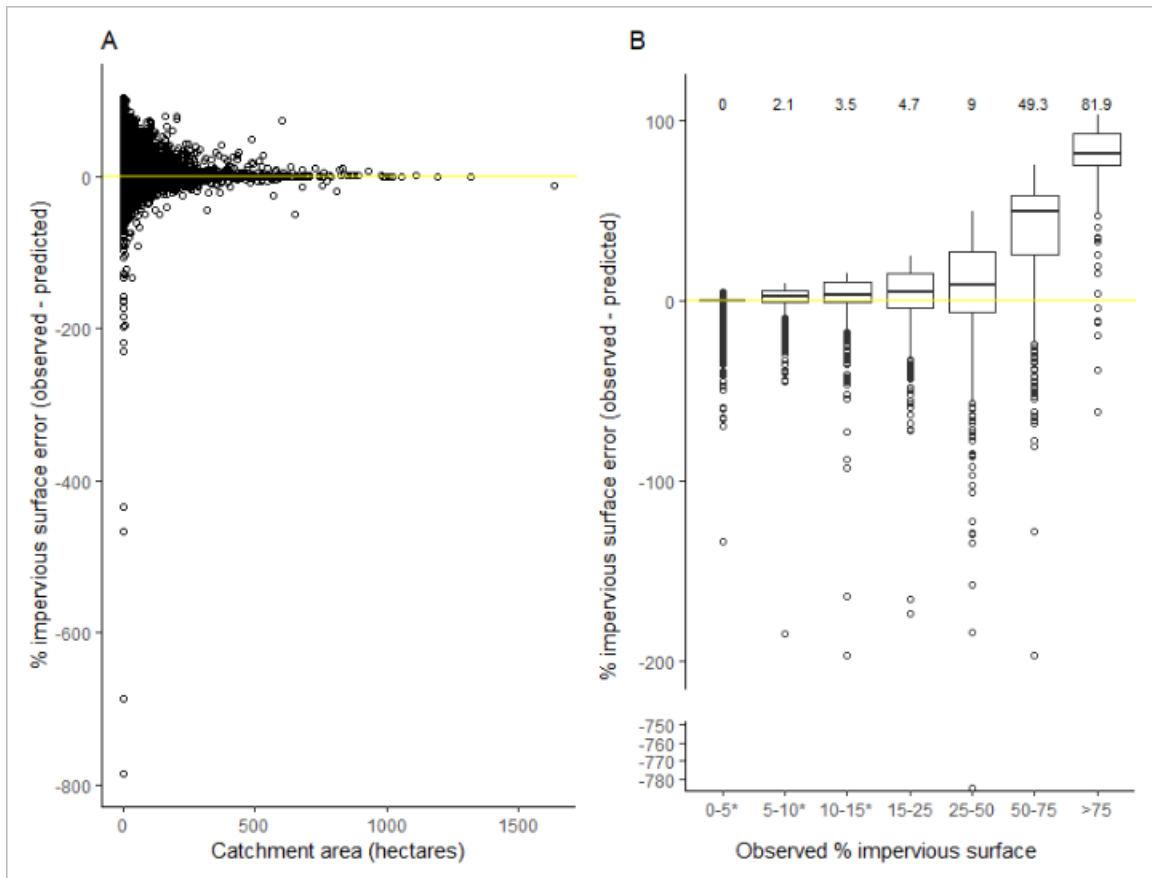


Figure 4. Plots of percent impervious surface error (observed - predicted) for NHDPlus HR catchments in Maryland relative to A) catchment land area in hectares and B) observed percent impervious surface where categories with an asterisk refer to habitat-based reference points for watershed development (Uphoff et al. 2022a) and median error is given above each box-and-whisker plot. Note the break in plot B's y-axis. Observed percent impervious surface was derived from Chesapeake Conservancy's 2017/2018 land use/land cover data (Chesapeake Conservancy 2022). Predicted percent impervious surface was derived using 2017 tax data (structures per hectare or C/ha) and the tax index (see **Project 1, General spatial and analytical methods**). All catchments were clipped to the Maryland state border and open water was erased prior to calculation of catchment area and percent impervious surface.



MD – Marine and estuarine finfish ecological and habitat investigations
Project 4: Resident Striped Bass forage benchmarks

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Executive Summary

Indices of Striped Bass condition, relative abundance, natural mortality, and forage relative abundance from annual surveys and fall diets provided metrics (indicators) to assess forage status and Striped Bass well-being in Maryland's portion of Chesapeake Bay. In addition to providing insight on forage status, these indicators were inexpensive and tractable for staff.

The 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. Proportion of Striped Bass in fall with empty guts (PE) provided trends in prey supply relative to predator demand based on relative abundance and diet sampling, respectively. The proportion of diet items by number and weight of prey per weight of Striped Bass (C) supplemented PE. Metrics based on examination of individual Striped Bass (P0, PE, and C) were split into two size classes (small, 260-456 mm TL and large, 457-864 mm TL) due to sampling considerations and divergence in trends in P0 between the size classes. The P0 and PE metrics had targets and thresholds. An index of survival (SR) that reflected natural mortality (M) from age 0 to 3 was developed for small Striped. Remaining metrics could not be split for size classes. A Striped Bass recreational catch per trip index (RI) that reflected ages 2-5 provided an index of relative abundance. Species specific forage-to-Striped Bass ratios were developed from relative abundance indices of major prey (FRs; focal prey species are Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab).

In 2022, the P0 indicators for both size classes were well below their targets (i.e., in a good range). Small Striped Bass condition was consistently poor (breaching the threshold) during 1998-2007 and shifted to a mix afterward. Condition of large Striped Bass was at its threshold in 6 of 7 years during 1998-2004 and has improved to only slightly missing its target once since 2014. Large Striped Bass P0 was the best of the time-series in 2021-2022.

Confidence intervals (90%) for PE of small Striped Bass overlapped the threshold in 2022; there is no target for this size class. This poor value of PE may have reflected a lack of Striped Bass in the low size range that feed on numerous, small items, particularly Bay Anchovy, in our sample. Estimated PE of small fish has been variable since 2012 and a trend is not evident. Correlation analysis of small Striped Bass diet composition indicated it may have been influenced by the length of fish sampled; frequency of Bay Anchovy was negatively associated with TL and Atlantic Menhaden were positively correlated. Small Striped Bass diets may be biased by the minimum sizes of Striped Bass available in annual samples.

Confidence intervals (90%) for PE of large Striped Bass exhibited considerable overlap with the target in 2022. Estimates of PE for large Striped Bass have improved from threshold conditions prior to 2007. Large Striped Bass have been mostly at target PE since 2014.

Atlantic Menhaden dominated small and large Striped Bass diets by weight during fall; C has been higher since 2013, more frequently ranking in the top half of estimates. Bay Anchovy

were dominant by number in small Striped Bass diets, but made up a low fraction of fall diet weight in all but the worst years. Small Blue Crabs were a minor component by weight as well, but were numerically abundant in some years. Spot, a major prey that had contributed to achievement of target P0 and PE for small fish in 2010 and largely absent in fall diets of both size classes since 2015, reappeared as a visible fraction of large fish diets in 2022. Bay Anchovy were consistently present in fall diets of both size classes of Striped Bass during 2006-2014, but have fallen substantially as a percent of large fish diet since 2015 as Atlantic Menhaden became more frequent. Bay anchovy represented a variable percentage of small fish diets during 2006-2015 and had a steadier, higher frequency until 2021-2022. Some to much of this change may be due to difficulty in collecting Striped Bass smaller than 334 mm, TL due to low year-class success. Diet changes since 2015 suggest the pelagic pathway is making a larger contribution to fall diets in recent years.

We may need to consider separating a smaller category of small fish that are not capable of feeding on Atlantic Menhaden from small Striped Bass transitioning to Atlantic Menhaden. Characteristics of large Striped Bass seem stable and are not subject to potential biases that are of concern for small fish.

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. Changes in FRs largely reflected decreasing prey during 1983-1994 since RI was low. After 1995, prey indices stayed relatively low and FR changes usually reflected fluctuations in RI. It appears that higher (but not always statistically different) Atlantic Menhaden indices since 2007 may have biological significance based on improvement in recent body fat and fall diet metrics.

Multiple lines of evidence suggest that survival of both small and large Striped Bass decreased in Chesapeake Bay due to higher M since the late 1990s. A sizeable increase in relative survival (SR) of small fish was evident in 2022. This estimate was from a poor Striped Bass year-class (2019) that was the first of a series of poor years through 2022 (Durell and Weedon 2022). If SR remains elevated through this series of poor year-classes, it may indicate lessening of density-dependent mortality up to age 3.

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). A large contingent of Chesapeake Bay Striped Bass that do not participate in the Atlantic coast migration (hereafter, resident Striped Bass) constitute a year-round population of predators that provide Maryland's major saltwater recreational fishery and an important commercial fishery; they are mostly males along with some young, immature females (Setzler et al. 1980; Kohlenstein 1981; Dorazio et al. 1994; Secor and Piccoli 2007; 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). 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 Chesapeake Bay stock growth (Richards and Rago 1999; ASMFC 2021). Management

since recovery has been based on much lower fishing mortality and much higher size limits than were in place into the early 1980s (Richards and Rago 1999; ASMFC 2021). An Atlantic Menhaden consumption per Striped Bass recruit analysis indicated that these conservative regulatory changes could have increased demand by approximately 2- to 5-times (Uphoff 2003).

Concern emerged about the impact of high Striped Bass population size on its prey-base shortly after recovery from severe depletion in 1995 (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 Striped Bass recovery (Uphoff 2003; Overton et al. 2015). Reports of Striped Bass in poor condition and with ulcerative lesions increased in Chesapeake Bay shortly after recovery; linkage of these phenomena with 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 widespread 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 legal sized Striped Bass in Chesapeake Bay increased substantially during the mid-1990s while annual instantaneous fishing mortality rates (F) remained low (Kahn and Crecco 2006; Jiang et al. 2007; NEFSC 2013; NEFSC 2019). Prevalence of mycobacteriosis and M appeared to be lower outside Chesapeake Bay (Matsche et al. 2010; NEFSC 2019), but abundance, condition, and M of the coastal migratory contingent has been linked to abundance of ages 1+ Atlantic Menhaden (Buccheister et al. 2017; Uphoff and Sharov 2018; ASMFC 2020; Chagaris et al. 2020)

Maryland's fisheries managers and stakeholders want to know whether there is enough forage to support Striped Bass in Maryland's portion of the Bay. Maintaining a stable predator-prey base is a challenge for managing Striped Bass in lakes (Axon and Whitehurst 1985; Matthews et al. 1988; Cyterski and Ney 2005; Raborn et al. 2007; Sutton et al. 2013; Wilson et al. 2013). 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 along the Atlantic coast and Striped Bass is a predator of concern because of its high sensitivity to Atlantic Menhaden population size (ASMFC 2020; Chagaris et al. 2020; Drew et al. 2021; Anstead et al. 2021). 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." Project 4 is a direct response by MD DNR to this outcome.

Indicators based on monitoring, such as forage indices, prey-predator ratios, 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 is based on a suite of indicators (metrics) that are inexpensively

and easily developed from existing MD DNR sampling programs. This report provides indicators through 2022. In addition to providing information for judging whether the forage base is adequate to support Striped Bass in Maryland's portion of Chesapeake Bay, two additional objectives were low cost and tractability for available staff.

During 2014-2019, we developed an integrated index of forage or IF that was comprised of five metrics covering all sizes of Striped Bass within a defined size range (286-864 mm TL or 11.3-34.0 inches). Forage status was judged by whether target (indicating good forage conditions) or threshold (indicating poor 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.

Uphoff et al. (2020) expressed concern that divergences of some metrics between small (<457 mm TL; < 18 inches) and large (\geq 457 mm TL) Striped Bass were masked by the IF approach. In this report, we have split metrics developed from sampling individual Striped Bass (condition and feeding metrics) between large and small fish where possible. Targets and thresholds were possible for a reduced number of metrics that could be split into the two size classes. Results in this report will be organized into sections that describe metrics for small Striped Bass, metrics for large fish, and metrics for both sizes combined.

Poor condition is a common problem for Striped Bass in lakes when prey supply is inadequate (Axon and Whitehurst 1985; Matthews et al. 1988; Cyterski and Ney 2005; Raborn et al. 2007; Sutton et al. 2013; Wilson et al. 2013). The proportion of Striped Bass without body fat (P0), a nutritional indicator, anchors our approach, providing a measure of condition and potential for starvation for each size class that was well-related to proximate composition and feeding of Striped Bass in the laboratory (Jacobs et al. 2013). The target developed by Jacobs et al. (2013) has been retained for both size classes and thresholds developed in previous years were revisited in Uphoff et al. 2022). Lipids are the source of metabolic energy for growth, reproduction, and swimming for fish and energy reserves relate strongly to foraging success, reproductive success, potential prey density, habitat conditions, environmental stressors, and subsequent fish health and survival (Tocher 2003; Jacobs et al. 2013); P0 integrates these factors into a single measure. A reliable and easily applied indicator of nutritional state is critical for evaluating hypotheses related to nutrition, prey abundance, density, and the outcome of the management measures that may follow (Jacobs et al. 2013).

Proportion of empty guts (PE) was used as a consumption-based indicator of major prey availability for each size class. Supplemental metrics on weight of prey consumed per weight of Striped Bass that consumed them (C), and composition of prey consumed (by number) could be estimated for each size class as well.

While upper Bay Striped Bass feed on a wide range of prey, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab *Callinectes sapidus* have consistently accounted for most annual diet biomass in Chesapeake Bay studies (Hartman and Brandt 1995c; Griffin and Margraf 2003; Walter et al. 2003; Overton et al. 2009; Overton et al. 2015; Buccheister and Houde 2016). We selected these species as focal prey (major prey) for forage indices. Forage ratios of species-specific indices of major prey relative abundance from fishery-independent surveys to an indicator of resident Striped Bass relative abundance were examined for each focal prey as an indicator of potential attack success. These forage ratios could not be split into size categories. 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).

Benthic invertebrate indices (invertebrates other than Blue Crabs) are included in this report even though benthic invertebrates have not contributed much to fall diets. 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 - summer diet (Overton et al. 2015). The utility of estimates of biomass of invertebrates comprising a benthic IBI (BIBI) in Maryland's portion of the Bay used for water quality monitoring was explored in Uphoff et al. (2018). A complementary index for hard (oyster) bottom was developed by M. McGinty (Uphoff et al. 2018).

The ratio of age-3 relative abundance of male Striped Bass in spring spawning ground gill net surveys (Versak 2021) to their year-class-specific juvenile indices (Durell and Weedon 2022) since 1985 was used as an indicator of change in relative survival of small fish (SR) due to M prior to recruitment to the fishery. The SR was an index for small fish since it tracked survival trends between young-of-year and age 3. Martino and Houde (2012) detected density-dependent mortality of age 0 Striped Bass in Chesapeake Bay, supporting a hypothesis that density dependence in the juvenile stage can contribute significantly to regulation of year-class strength. We expected SR to vary without trend if M remained constant. Very general trends in the SR, an index of the effect of M on small Striped Bass, could be compared with trends in estimates of M for large fish developed from conventional (NEFSC 2019) and acoustic tags (Secor et al. 2020).

Methods

Abbreviations and definitions - Table 1 contains important abbreviations and definitions.

Striped Bass condition, feeding success, and diet composition indices – Indicators of condition, feeding success, and diet composition during October-November were developed for 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.

The collector's 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, lowered to 483 mm TL in 2018 and has remained there through 2022).

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. Active 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 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; they were iced immediately and cleaned 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.

Striped Bass collected for health samples by Fish and Wildlife Health Program (FWHP) have been processed since 2014 by Fish Habitat and Ecosystem Program (FHEP) biologists for diet information. Collections by FWHP were not constrained by collector's permit conditions like CBEF collections. Fish have been collected by hook-and-line from varying locations during fall since 1998 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 excluded from diet data.

Condition was estimated from an existing FWHP Striped Bass health survey that began in 1998. Nutritional status (condition) for upper Bay Striped Bass was estimated as the proportion of fish without visible body fat (P0) during October-November in FWHP samples. Estimates of P0 were made for the two size classes of Striped Bass. 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 FRs). 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 higher 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 for both size classes during 1998-2004, a period of consistently poor condition). Uphoff et al. (2022) revised this single P0 threshold for both species to 0.67 for small fish and 0.70 for large fish. 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 with aid on occasion from 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 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.

Proportion of food represented by an item in numbers was estimated for each Striped

Bass size class based on fish with stomach contents for each year since 2006 (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.

We attempted to base estimates of proportion of food represented by diet items for small fish on a standard TL range; however, it appeared that Striped Bass in the lower end of the size distribution were not always represented. We suspected that proportion of diet by number represented by Bay Anchovy and Atlantic Menhaden could be affected by size of Striped Bass present during a sample year. We estimated the mean, median, and minimum TL of Striped Bass in the small category for each year (2006-2022) and used correlation analysis to examine whether the annual proportions of the two diet items were associated with these size statistics.

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 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) for comparisons within our small fish category.

Overton et al. (2009) provided estimates of percent of Striped Bass stomachs with food during fall 1998-2000 (years combined) from a mid-Bay region that corresponded to our study area. We converted these estimates into PE; PE was 0.54 for fish between 301 and 500 mm, TL (approximating our small class) and 0.57 for Striped Bass between 501 and 700 mm (approximating our large class; Overton et al. 2009). These 1998-2000 estimates were comparable to our highest estimates of PE and were concurrent with high P0, high abundance of Striped Bass, and a nadir in major prey indices (except the Bay Anchovy trawl index). Target PE was estimated for small or large fish from periods when PE corresponded with target estimates of P0.

To aid interpretation of PE, we examined 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 of Striped Bass; we compared these estimates to optimum PPLR for Striped Bass (0.21; Overton et al. 2009).

We investigated a subset of years (2013-2022) to examine the length specific pattern of consumption of age 0 Atlantic Menhaden by small Striped Bass in fall. This analysis would provide insight on interpreting diet compositions based on number or weight of prey for this size class. We constructed a length-frequency distribution of small fish that had intact age 0 Atlantic Menhaden identified in their diet sample.

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 major prey in upper Bay. A shoreline seine survey targeting age-0 Striped Bass provided indices since 1959

for Atlantic Menhaden, Bay Anchovy, and Spot (Durell and Weedon 2022). 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 2020; MD DNR 2023a; 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 (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 2023b). 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-current) to place their time-series on the same scale for graphical comparisons of trends among surveys.

A soft bottom benthic biomass index (invertebrates living in the sediment) has been a component of a Chesapeake Bay benthic index of biotic integrity (BIBI); the BIBI provides an accessible summary of benthic habitat status (Weisburg et al. 1997). We used the biomass (grams / m²) of benthic invertebrates component for Maryland tidal waters as our index (Figure 3-38 in Llansó and Zaveta 2019; Versar Inc 2023). The BIBI has been employed to monitor water quality since 1995 and the latest indices are for 2021. The benthic biomass component consists of 7 polychaetes, 10 mollusks, 1 isopod, 2 amphipods, and 2 ribbon worms (see Table 2-5 in Llansó and Zaveta 2019). Uphoff et al. (2018) explored the relationship of this benthic biomass index on resident Striped Bass condition. This index was not incorporated into a forage ratio (described below for major prey).

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; <https://www.fisheries.noaa.gov/recreational-fishing-data/about-marine-recreational-information-program>) database as an index. The query tool provided 2-month wave-based estimates until 2022 when this webpage was discontinued. Angela Guiliano (MD DNR) provided a new catch per private boat trip and PSE time-series from the MRIP database for 1981-2022. The proportional standard error (PSE) calculated with these methods did not match the previous PSEs from the MRIP website due to new weighting variables in the public datasets that account for the cluster sampling as well as the effort survey information to expand the intercept values (A. Guiliano, MD DNR, personal communication).

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; NEFSC 2013).

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). 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. 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). Trends in the RI compared favorably to the abundance estimates for 2–5-year-old Striped Bass estimated by the statistical catch at age model used in the NEFSC (2019) stock assessment (Uphoff et al. 2020). Online estimates of catch and effort were available for 1981-2021.

This new RI time-series was compared to estimated abundance of ages 2-5 from the ASMFC (2022) update. Alexei Sharov (MD DNR) provided the Excel table output from the statistical catch-at-age model (SCAM). Ages 2-5 were likely to comprise most of the harvest and releases during wave 5. The SCAM estimated combined abundance for Chesapeake Bay, Hudson River, and Delaware River stocks, but mostly reflect abundance in the Bay (NEFSC 2019). A linear regression was used to determine the strength of the relationship of the RI to the to summed abundance of ages 2-5.

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, an index of potential attack success. Ratios were standardized by dividing each year's FR estimate by the mean of FR during 1989 to the present, a time-period in common among all data; FR estimates were available for every year since 1983 except 1987 (RI was not estimated).

We estimated relative survival as relative abundance at age-3 divided by age-0 relative abundance three years prior (juvenile index in year - 3) for 1985-2020 and 2022. We did not estimate relative survival (SR) for 2021 due to concerns about the validity of the gill net index for that year (B. Versak, MDDNR, personal communication). An outbreak of Covid in the Head-of-Bay crew caused two weeks during the main spawning period to be missed and it was feared the estimate would be negatively biased. 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 2021). Table 8 in Versak (2021) provided mean values of for annual, pooled, weighted, age-specific CPUEs since 1985 for the Maryland Chesapeake Bay Striped Bass spawning stock and we used the age-3 index (CPUE3) as the basis for an adjusted index. Typically, the most recent year's CPUE3 was unavailable on this table and was provided by Beth 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 fish would be resident males (Versak 2021). 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 (ASMFC 2022), juvenile indices (Durell and Weedon 2022), egg presence absence indices (Uphoff et al. 2022), 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 unaltered estimates afterwards. 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 $CPUE_3$ / 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 frequency of SR estimates above, below, and near the full time-series median was determined and trends in SR were compared to RI to examine whether density-dependent mortality was suggested.

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 5,000-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, HI, and the Blue Crab index were considered normally distributed since their distributions were characterized by means and SE's in their respective sources (Versak 2021; MD DNR 2020b). Remaining indices for Atlantic Menhaden (seine), Bay Anchovy (seine and trawl), and Spot (seine and trawl) and the JI for Striped Bass were based on geometric means (Durell and Weedon 2022). 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. The \log_e -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).

Results

Sample Size Summary - During 1998-2022, 2,202 small and 3,178 large Striped Bass were sampled during October-November (Table 2). Annual sample sizes for small fish in October-November ranged from 29 to 271 with a median of 118. Annual sample sizes for large fish ranged from 49 to 327 with a median of 203. 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 2014-2022 versus 11-22 trips by CBEF during 2006-2013). In most years, 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. The late start dates for 2021 and 2022 reflected a dearth of fish available until mid-October (J. Uphoff, MD DNR, personal observation). End dates during 2014-2020 tended to be earlier in November for

FWHP surveys, reflecting when size categories were filled out. End dates were later (November 30) during 2021 and 2022 (Table 2).

Small Striped Bass Condition, feeding success, and diet composition indices - Condition of small Striped Bass has transitioned from consistently poor during 1998-2007 to a mix of at or near target P0 interspersed with scattered years of poor P0 afterward (Figure 2). Small Striped Bass were at the target level of condition ($P0 \leq 0.30$) during 2008, 2015, 2017, 2021 and 2022 (the last 2 years have been the best of the time-series). Small fish in the upper Bay during fall were in poorest condition during 1998-2007, 2011-2012, 2016, and 2019; we adopted $P0 = 0.67$ (minimum during 1998-2007) as this size group's threshold (Uphoff et al. 2022). Estimates of P0 (0.36-0.46) were between the target and threshold during 2009-2010, 2013-2014, 2018, and 2020. The 90% confidence intervals of P0 allowed for separation of years at or near the threshold from remaining estimates (Figure 2).

Estimates of PE of small Striped Bass during fall, 2006-2022, ranged between 0.10 and 0.57 (Figure 3). Estimates of PE during 2006-2007, 2012, 2015, and 2022 could not be clearly separated from the threshold based on 90% CI overlap; PE during 1998-2000 (Overton et al. 2009), was the threshold for small fish ($PE = 0.54$; Uphoff et al. 2016). Lowest estimates of PE for small fish (2009-2011, 2014, 2017, and 2019) could be separated from remaining higher estimates (except 2008) based on 90% confidence interval overlap. Estimates of PE during 2008-2011, 2014, and 2016-2021 were clearly lower than the 90% CIs of years that breached the threshold. Estimated PE in 2022 (0.48) was below the threshold and above the time-series median (0.32). Confidence intervals overlapped a considerable portion of the threshold CI's and did not overlap the median (Figure 3).

In combination and by number, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab accounted for 94.4% of diet items encountered in small Striped Bass collected from upper Bay during fall, 2006-2022 (Figure 4). Bay Anchovy accounted for the highest percentage by number when all years were combined (62.6%, annual range = 19.1-87.9%); Atlantic Menhaden, 17.4% (annual range = 0-69.9%); Spot 5.4% (annual range = 0-74.1%); Blue Crab, 12.2% (annual range = 0.8-34.6%); and other items accounted for 4.0% (annual range = 0-12.9%; Figure 4). During 2022, Atlantic Menhaden accounted for 74.4% of the diet items; Bay Anchovy, 9.3%; Spot, 11.1%; Blue Crab, 0%; and other items (tunicates and shrimp) accounted for 5.6%. The vast majority of major prey in small Striped Bass diet samples during fall fell within young-of-year length cut-offs (Uphoff et al. 2016).

Annual minimum, mean, and median TL for small Striped Bass sampled during 2006-2022 ranged from 229-375 mm, 349-429 mm, and 335-437, respectively (Table 3). The minimum TL of Striped Bass in the small category that had an intact age 0 Atlantic Menhaden in its gut was 334 mm. The cumulative percent of menhaden in small Striped Bass guts gradually increased to about 20% by 395 mm and then increased rapidly, reaching 50% at 420 mm (Figure 5). The proportion of Atlantic Menhaden in small Striped Bass diets was moderately and positively correlated with annual small Striped Bass minimum TL ($r = 0.58$, $P = 0.0136$), mean TL ($r = 0.70$, $P = 0.0017$), and median TL ($r = 0.69$, $P = 0.0023$). The proportion of Bay Anchovy in small Striped Bass diets was moderately to strongly and negatively correlated with small Striped Bass minimum TL ($r = -0.60$, $P = 0.0115$), mean TL ($r = -0.79$, $P = 0.0001$), and median TL ($r = -0.69$, $P = 0.0023$). These associations indicated that small Striped Bass diet composition may have been influenced by the length of fish sampled.

By weight, small Striped Bass diets in fall 2006-2022 (combined) were comprised of Atlantic Menhaden (72.6%), Bay Anchovy (13.0%), Spot (8.5%), Blue Crab (1.8%) and other

items (4.1%; 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 2006-2022. During years of lowest C (2007, 2011, 2016, and 2017), varying items contributed to the diet of small fish. During years of when C was high (more than twice the 2006-2021 median) either Spot (2010) or Atlantic Menhaden (2013-2014) dominated diet mass. The 2022 estimate of C of small fish (0.0093) was below the median (0.0115) of the year time-series (Figure 6).

Median PPLRs of large prey of small Striped Bass (Spot and Atlantic Menhaden combined) were 0.20-0.38 during 2006-2022 (Figure 7). Median PPLRs for small fish were particularly high (0.34-0.38) during 2012 and 2015-2019. They were close to the optimum (0.21) described by Overton et al. (2009) in 2010 (2010 PPLR = 0.199) when Spot constituted a large fraction of their diet. The median PPLR was 0.27 for 2022 and it was the third lowest of the time-series (Figure 7). These PPLRs would not be affected by the relationships of Age 0 Atlantic Menhaden and small Striped Bass TL size since those ratios are based on Striped Bass with Age 0 Menhaden in their guts.

Large Striped Bass condition, feeding success, and diet composition indices - Condition of large Striped Bass has transitioned from mostly poor during 1998-2004 to a mix of at or near target P0 after 2013 (Figure 8). Large Striped Bass were at the target level of condition ($P0 \leq 0.30$) during 2008-2010, 2014-2015, and 2017-2022. Estimated P0 (0.035) in 2022 was the second lowest of the time-series. Large fish during fall were usually in poorest condition ($P0 \geq 0.70$) during 1998-2004 (except 2002) and we adopted $P0 = 0.70$ as this size group's threshold. The 90% confidence intervals of P0 allowed for separation of years at the target from remaining estimates and estimates at the threshold from those at the target. Five of six estimates were above the threshold during 1998-2001 and 2004, and could be separated from most (7 of 8) P0 estimates that fell between the target and threshold (Figure 8).

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 (0.58) as a threshold PE for large sized fish (Figure 9). Estimates of PE of large Striped Bass during fall were at the threshold level in 2006, 2012, and 2017 based on 90% CI overlap. There was a modest association of PE and P0 ($r = 0.63$, $P = 0.007$) during 2006-2022; review of the plot of these variables (not shown) indicated that P0 at the target level was more likely when PE was 0.40 or less (9 of 10 points) than above it (2 of 6). The PE target for large fish, 0.34, was met during 2014-2015 and 2018-2021. Estimated PE was 0.36 in 2022 (Figure 9).

Major prey accounted for 92.6% of diet items, by number, encountered in large Striped Bass diet samples during fall 2006-2022 (Figure 10). Atlantic Menhaden accounted for 49.9% by number when all years were combined (annual range = 12.4-97.0%); Bay Anchovy, 14.4% (annual range = 0-32.5%); Spot, 7.6% (annual range = 0-52.4%); Blue Crab, 20.7% (annual range = 0-59.4%); and other items, 7.4% (annual range = 0-40.0%). Spot were absent or very rare in fall diet items during 2014-2021, but reappeared in greater quantity in 2022. The "Other" category accounted for a higher fraction of large Striped Bass diets by number in 2012 and 2017 (36.2% and 40.0%, respectively) than remaining years ($< 9.7\%$). During 2022, Atlantic Menhaden accounted for 80.2% of October-November diet items; Bay Anchovy, 2.3%; Spot, 7.6%; Blue Crab, 2.3%; and other items (Atlantic Croaker, Mantis Shrimp, and Silver Perch) accounted for 7.6% (Figure 10). The vast majority of major prey fell within young-of-year length cut-offs (Uphoff et al. 2016).

By weight, Atlantic Menhaden predominated in large fish sampled (92.3% of diet weight

during fall, 2006-2022, combined); Bay Anchovy accounted for 1.1%; Spot, 3.5%; Blue Crab, 3.5%; and other items, 4.4% (Figure 11). Estimates of C for large Striped Bass varied as much as 3.8-times among years sampled. The 2022 estimate of C of large fish (0.013) was below the time-series median (0.014; Figure 11).

Median PPLRs of large prey (Spot and Atlantic Menhaden) for large Striped Bass were 0.19-0.30 during 2006-2022 (Figure 12). The median PPLR was 0.23 for 2022 (Figure 12). Median PPLRs for large Striped Bass were much closer to the optimum (0.21 based on Overton et al. 2009) than for small fish.

Relative abundance indices of Striped Bass and major prey – Relative abundance of Striped Bass (RI) during 1981-2020 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 2.4-3.6 during 2014-2019 (2019 was the second highest of the time-series), before falling to 1.8 in 2020, 1.4 in 2021, and 1.1 in 2022. 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 and 2020-2022 was lower than other years during 1995-2019 (Figure 13). The trend in RI tracked the trend in estimated aggregate abundance of 2- to 5-year-old Striped Bass along the Atlantic Coast fairly well through 2014 and less well after (Figure 14). A linear regression of ages 2-5 abundance estimated for the Atlantic coast (ASMFC 2022) was moderately related to RI from Maryland's portion of Chesapeake Bay during 1983-2022 ($r^2 = 0.52$, $P < 0.0001$).

Major pelagic prey were generally much more abundant during 1959-1994 than afterward (Figure 15). 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 (top quartile) occurred in 1989-1992, 1998-2000, 2013, and 2020-2021, while lowest quartile indices occurred during 2005, 2007-2011, and 2015-2018. There was little agreement between the two sets of Bay Anchovy indices; however, there were few data points representing years of higher abundance in the years in common and contrast may have been an issue (comparisons were of mostly low abundance points). Atlantic Menhaden seine indices were high during 1971-1994 and much lower during 1959-1970 and 1995-2022. There has been an upward shift in Atlantic Menhaden seine indices from mostly their lowest sustained level during 1995-2012 (Figure 15). There may be a need to take the influence of the Atlantic Multidecadal Oscillation into account when judging Atlantic Menhaden seine indices (Buccheister et al. 2016).

Major benthic forage indices were low after the 1990s, but years of higher relative abundance were interspersed during the 2000s (Figure 16). Seine (1959-2022) and trawl (1989-2022) indices for Spot had similar trends that indicated high abundance during 1971-1994 and low abundance during 1959-1970 and after 1995 (with 3 or 4 years of higher indices interspersed). Spot indices in 2020-2022 were much better than the previous nine years. Blue Crab densities (1989-2019) were generally at or above the time-series median during 1989-1998, and 2009-2015. Blue Crab densities in 2020-2022 were among the lowest of the time-series (Figure 16).

Most of the annual indices of biomass of soft bottom benthic invertebrates during 2000-2009 were well above the time-series median (Figure 19). Indices well below the median indices occurred during 1996, 1998, 2003-2004, 2012, 2021, and 2022. The biomass index for 2022 was the lowest of the 1995-2022 time-series (Figure 19).

Species-specific standardized FRs exhibited similar general patterns during 1983-2022

(Figure 18). Indices were at their highest in the early 1980s when Chesapeake Striped Bass were at their lowest level and fell steadily in the early 1990s as Striped Bass recovered and forage indices declined. A nadir in the ratios appeared during 1995-2004 (Striped Bass recovery was declared in 1995), followed by occasional “spikes” of Spot and Blue Crab ratios and a slight elevation in Atlantic Menhaden ratios after 2004. Forage ratios in 2022 of Blue Crab and trawl survey Bay Anchovy were below their 1989-2021 medians, while remaining species were near or slightly above this median (Figure 18). The Atlantic Menhaden FR has been generally elevated during 2005-2022 from its nadir during 1997-2004, but has been well below levels prior to the early 1990s (Figure 19). The Bay Anchovy seine FR was similar to years of higher FRs since 1995 (2006-2009 and 2010-2013; Figure 18). The Spot seine FR during 2020-2021 was in the higher portion of the range exhibited since 1995 (Figure 20). The Bay Anchovy trawl FR for 2022 was in the top third of the time-series and represented a considerable improvement over 2021 (second lowest; Figure 21). The Spot seine (Figure 22) and trawl FRs (Figure 23) for 2020-2022 were similar in magnitude and indicated considerable improvement over lows exhibited during 2014-2019. The Blue Crab FR was slightly above the time-series median (Figure 24).

Relative survival of small Striped Bass – The unadjusted age 3 gill net index of male relative abundance on the spawning grounds did not indicate the same trend as age 3 abundance in the assessment (NEFSC 2019) during 1985-1995; abundance during 1985-1995 was at least as high as any other period of the time-series through 2020 (Figure 25). The hybrid approach resulted in much better agreement with age 3 abundance trends in the ASMFC (2022) stock assessment update. 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. 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 NEFSC (2019) assessment tended to be higher than would have been indicated by the hybrid gill net index (Figure 25).

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 26). Estimated SR in 2022 was among the peak values of the time-series (Figure 26)

Estimated SR was more often high during 1986-1998 with 8 years above the median and 4 below; this time span coincided with consistently low RI estimates through 1994 and a rapid increase through 1998 (Figure 27). Low SR during this rapid increase of the RI may have indicated a lagged response. After 1998, SR shifted to consistently below the median during 1999-2004 and varied during 2005-2020 (9 years were at or above the median, 7 were below). Estimated SR in 2022 was the 5th highest of the time-series. Large oscillations in SR above and below the median were evident during 2005-2011 and they dampened after 2011. There was very general support for a density-dependence survival hypothesis. Estimates of RI were usually much higher after 1994, although there was a period (2009-2009) where relative abundance was between its lows and highs (Figure 27). Low survival in 1985 reflected the effect of the fishery (low length limits and high F) on the 1982 year-class prior to imposition of a harvest moratorium in Maryland, but SR in other years should have primarily reflected M since the fishery was closed during 1985-1990 and conservative management (high size limits and low creel limits) was in place after that (Richards and Rago 1999; ASMFC 2022).

Discussion

Average condition of small and large Striped Bass was good (met target conditions) during 2022 and represented the best body fat indices for both size classes for the whole time-series. Small Striped Bass condition was consistently poor (breaching the threshold) during 1998-2007 and shifted to a mix afterward. During 2008-2022, there were five years where P0 of small fish met the target, four years that the threshold was exceeded, and six years in between. Condition of large Striped Bass was at its threshold in 6 of 7 years during 1998-2004 and has improved, only slightly missing its target once since 2014.

The P0 metric represents an integration of multiple factors that affect condition into a single measure. Lipids are the source of metabolic energy for growth, reproduction, and swimming for fish and energy reserves relate strongly to foraging success, reproductive success, potential prey density, habitat conditions, environmental stressors, and subsequent fish health and survival (Tocher 2003; Jacobs et al. 2013). It is important to note that our condition and diet samples are mostly from survivors of two to five years (depending on size and age) of some combination of feeding success, growth, environmental conditions, mycobacteriosis, and catch-and-release and harvest mortality that reduce abundance and intraspecific competition among Striped Bass. The summer preceding our fall monitoring may be particularly stressful and potentially lethal. Summer represented a period of no to negative growth in weight for ages 3-6 during 1990-1992 (Hartman and Brandt 1995b), higher mortality of diseased and healthy Striped Bass (Groner et al. 2018), hypoxia and temperature stress (Constantini et al. 2008; Maryland Sea Grant 2009; Coutant 2013; LaPointe et al. 2014; Kraus et al. 2015; Itakura et al. 2021), and high catch-and-release mortality (Lukacovic and Uphoff 2007). Condition of Striped Bass in summer was a good predictor of fall condition, and condition in fall of the previous year appeared related to condition in the next fall (Uphoff et al. 2017). If fewer fish make it through these hurdles, the survivors may benefit from reduced intraspecific competition for forage. The RI is a rather blunt indicator of resident abundance since it aggregates both large and small size groups and seems likely to be dominated by the small size class. Improvement in condition due to greatly reduced abundance of Striped Bass is not likely to be comforting to fishermen or managers.

Large Striped Bass have been mostly at target PE associated with target P0 since 2014. A target was not readily suggested for PE of small fish, but PE was clearly below the threshold during 2008-2010, 2014, and 2016-2021.

The PE metric is a simple and robust indicator of overall feeding success (Baker et al. 2014), but it can be biased by high frequency of small items that may not have much nutritional value or low frequency of large items with higher nutritional value and digestion times (Hyslop 1980). Additional information (numeric frequency of diet items and estimates of C) aids interpretation of PE.

Atlantic Menhaden dominated small and large Striped Bass diets by weight during fall. Bay Anchovy were dominant by number in small Striped Bass diets, but made up a low fraction of fall diet weight in all but the worst years. Small Blue Crabs were a minor component by weight as well, but were abundant in some years. Spot, a major prey that had contributed to lower PPLR of large major prey and achievement of target P0 and PE for small fish in 2010, have been largely absent in fall diets of both size classes between 2014 and 2021. They reappeared as a visible fraction in 2022.

Small Striped Bass condition has improved since the mid-2000s, but not as consistently as for large fish. The transition from small to large major prey may represent a bottleneck for

small Striped Bass. 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 smaller predator and larger size makes prey more difficult to retain if caught (Lundvall et al. 1999). With high size limits and low fishing mortality in place for Striped Bass since restoration, intraspecific competition for limited forage should be greater for small Striped Bass because they compete with one another and large Striped Bass. Striped Bass in our large category were uncommon in Maryland's Bay prior to restoration because of higher F and lower length limits; pound net length-frequencies in the 1960s-1970s rarely contained large fish (J. Uphoff, MD DNR, personal observation). In addition to being able to handle a wider size range of prey, 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 coincided with two large year-classes of Striped Bass having approached or 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-early summer, respectively (Walter et al. 2003; Hartman and Brandt 1995c; Overton et al. 2009; 2015). These prey did not usually make a large contribution to diet mass during fall, but on occasion White Perch made a contribution to large Striped Bass C. The effect of other items consumed in other seasons would be incorporated into P0, but their contribution to P0 would be unknown, although it might be suspected from high P0 that seemed anomalous.

A rapid rise in Striped Bass abundance in Maryland's portion of the 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. Changes in FRs largely reflected decreasing prey during 1983-1994 since RI was low. After 1995, prey indices stayed relatively low and RI increased; FR changes usually reflected fluctuations in RI. 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 accompanied by greatly increased 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 when threshold P0 was predominant, may have biological significance based on improvement in recent body fat and fall diet metrics.

Forage to Striped Bass ratios indexed potential attack success on major prey (Uphoff 2003; MD Sea Grant 2009). Atlantic Menhaden FR reached its nadir during 1995-2004 and has risen just above it since. The FRs for Atlantic Menhaden, Spot, and Bay Anchovy since 2005 have been well below those that occurred in 1990, the year used to set target conditions for P0 (Jacobs et al. 2013). Condition of both size classes improved after 2004, but improvement was steadier and more pronounced for large Striped Bass. Bay Anchovy were consistently present in fall diets of both size classes of Striped Bass during 2006-2014, but have fallen substantially as a percent of large fish diet since 2015-2018 (10-29% by number) to 0-4% in 2019-2022 as Atlantic

Menhaden became frequent in their fall diet. Bay anchovy represented a variable percentage (22.7-87.9%) of small fish diets during fall 2006-2015 and had a steadier, higher frequency (65-90%) afterwards until 2021-2022. Spot have made an insignificant contribution to fall diets of both size classes of Striped Bass since 2011 and Blue Crab have made a consistently smaller contribution to small Striped Bass diets since 2015. These changes since 2015 suggest the pelagic pathway is making a larger contribution to fall diets. Overton et al. (2015) described shifting prey dependence over time in Chesapeake Bay based on bioenergetics analyses of annual Striped Bass diets in the late 1950s, early 1990s, and early 2000s. By the early 2000s, there was a greater dependence on Bay Anchovy by all ages of Striped Bass and older fish had a greater dependence on the benthic component as Atlantic Menhaden declined in the diet (Overton et al. 2015). Stable isotope analyses of archived Striped Bass scales from Maryland's portion of Chesapeake Bay indicated an increasing shift from pelagic to benthic food sources during 1982-1997 (Pruell et al. 2003).

Small Striped Bass diets may be biased by the minimum sizes available in samples. We may need to consider an additional smaller category of small fish that are not capable of feeding on Atlantic Menhaden and a "mid-sized" small category for Striped Bass transitioning to Atlantic Menhaden.

The soft bottom benthic index time-series covered 1995-2022 and changes prior to Striped Bass recovery could not be addressed. Benthic biomass has generally been lower since 2010 and has been very low in the last two years. Changes in benthic invertebrate populations have the potential to affect Striped Bass directly or through reductions in benthic major prey. There was little indication of correspondence of the soft bottom benthic index to P0 of either size class of Striped Bass. However, there may be years where consumption of benthic prey in spring and early summer (polychaete or "May worm" blooms) may help tide Striped Bass through late summer - early fall that may not be detected by an analysis of linear trends.

While top-down control of forage is suggested by opposing trends of major forage and Striped Bass, bottom-up processes may also be in play. A long-term decline Bay Anchovy in Maryland's portion of Chesapeake Bay (based on the seine index) was linked to declining abundance of the common calanoid copepod *Acartia tonsa* that, in turn, was linked to rising long-term water temperatures, eutrophication, and hypoxia (Kimmel et al. 2012; Roman et al. 2019; Slater et al. 2020). Copepod mortality not due to predation was higher under hypoxic conditions and implied a direct linkage between low dissolved oxygen and reduced copepod abundances (Slater et al. 2020). Houde et al. (2016) found Chl a and variables associated with freshwater flow, e.g. Secchi disk depth and zooplankton assemblages, were correlated with age-0 Menhaden abundance in the upper Bay. Variations in river flows to the Chesapeake Bay set up stratification, drive estuarine circulation, and cause fluctuations in inputs of freshwater, sediments, and nutrients, processes that greatly influence hypoxia (Hagy et al. 2004; Kemp et al. 2005; Maryland Sea Grant 2009). Woodland et al. (2021) demonstrated that bottom-up processes influenced fish and invertebrate forage in Chesapeake Bay (including our major forage species and benthic invertebrates included the BIBI based index; Blue Crabs were not examined). Annual abundance indices of many forage taxa were higher in years when spring water temperatures warmed slowly. Forage indices also were related (in taxon-specific ways) to winter-spring chlorophyll concentration and freshwater discharge, and to three summer water quality variables: dissolved oxygen, salinity, and water temperature, in addition to a broad-scale climate indicator (Atlantic Multidecadal Oscillation or AMO; Woodland et al. 2021). The AMO was the best single predictor of recruitment patterns of Atlantic Menhaden in Chesapeake Bay

and along the Atlantic coast, suggesting that broad-scale climate forcing was an important controller of recruitment dynamics, although the specific mechanisms were not identified (Buccheister et al. 2016). The MD Spot seine index was negatively and weakly correlated with the AMO (January-April mean; $r = -0.41$, $P = 0.001$, 1959-2020; J. Uphoff, unpublished analysis).

A hypoxia-based hypothesis, originally formed to explain die-offs of large adult Striped Bass in southeastern reservoirs, links increased M and deteriorating condition in Chesapeake Bay through a temperature-oxygen squeeze (mismatch of water column regions of desirable temperature and dissolved oxygen in stratified Chesapeake Bay during summer; Coutant 1985; Price et al. 1985; Coutant 1990; Coutant 2013). Constantini et al. (2008), Kraus et al. (2015), and Itakura et al. (2021) examined the impact of hypoxia on 2-year-old and older Striped Bass in Chesapeake Bay through bioenergetics modeling and acoustic tagging and concluded that a temperature-oxygen squeeze by itself was not limiting for Striped Bass. However, Groner et al. (2018) suggested that Striped Bass are living at their maximum thermal tolerance and that this is driving increased mycobacteriosis and associated mortality. Adequate levels of Striped Bass prey can offset negative effects of warm temperatures and suboptimal dissolved oxygen in reservoirs (Thompson et al. 2010; Coutant 2013). 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).

Multiple lines of evidence suggest that survival of both small and large Striped Bass decreased in Chesapeake Bay due to higher M since the late 1990s. A sizeable increase in SR was evident in 2022. This estimate was from a poor Striped Bass year-class (2019) that was the first of a series of poor years through 2022 (Durell and Weedon 2022). If SR remains elevated through this series of poor year-classes, it may indicate lessening of density-dependent mortality up to age 3.

Higher frequency of below time-series (1985-2020) median SR between ages 0 and 3 after 1996 was concurrent with declines in conventional tag-based estimates of survival of 457-711 mm of Striped Bass in Chesapeake Bay (based on time varying estimates of M ; Uphoff et al. 2022). Annual survival decreased from 77% during 1987-1996 to 44% during 1997-2017, a 43% reduction (based on Table B8.25 in NEFSC 2019); estimates of F in Chesapeake Bay 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 lower survival (30% per year) than coastal shelf emigrants (63% per year; Secor et al. 2020).

Decreased survival of large Striped Bass estimated 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. (2009b) 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. In addition, abundant individuals competing for limited prey may hinder one another's feeding activities, leading directly to starvation (Yodzis 1994). Shifts from high survival during 1987-1996 to lower survival afterwards (Kahn and Crecco 2006; Jiang et al. 2007; NEFSC 2013; NEFSC 2019) lagged two years behind downward shifts in forage-to-

Striped Bass ratios. Dutil and Lambert (2000) found that the response of M of Atlantic Cod (*Gadus morhua*) 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.

Condition of both size classes of resident Striped Bass has improved since the mid-2000s in concert with slight improvement in Atlantic Menhaden FR and consumption. No other major prey FR (or benthic invertebrate biomass) matches this timing. Recent improvement of Spot relative abundance may further enhance condition since they provided a major fraction of consumed biomass in spring-fall ahead of heavy feeding on Menhaden during 1990-1992 and 1998-2001 (Hartman and Brandt 1995b; Overton et al. 2009).

Decreased survival in the mid-to-late 1990s 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 may have serious implications for interstate 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 F , and compensatory M would undercut both objectives.

Long-term analyses of M based on conventional tags indicated survival of large Striped Bass decreased after stock recovery (NEFSC 2019), but the time blocks analyzed were large and only differentiated two periods (pre- and post-1997), the former of low M and latter of high M . A finer temporal resolution of M estimates is needed to relate forage or other conditions to survival of large fish. Survival of small Striped Bass in Chesapeake Bay has not been explored with conventional or acoustic tags.

Catch-and-release mortality different from that assumed in NEFSC (2019) could have confounded estimation of M from tagging experiments. Increases in conventional tag-based estimates of M of legal-sized fish over time could also reflect misspecification of parameters such as tag reporting rates that make absolute estimates less reliable (NEFSC 2019); however, M estimates based on acoustic tags (not subject to reporting rates) produced similar differences in mortality of coastal migrants and Chesapeake Bay residents (Secor et al. 2020).

Hook-and-line samples collected by CBEF (2006-2013) and FWHP (2014-2022) 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 et al. 2018). Sizes of Striped Bass sampled by the two programs were comparable and estimates of P_0 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). There has not been a readily discernable shift in patterns of PE, C, and frequency of diet items by number that would be readily attributed to changes from CBEF to FWHP sampling programs.

The 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, particularly age 0 Atlantic Menhaden, 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 FRs based on relative abundance indices (Uphoff et al. 2022).

We treated 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 and the nature of hook-and-line fishing for them. 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 and aggressive open water feeding. Forage begins to migrate out of the Bay and its tributaries (and refuges therein) or to deeper Bay waters 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 location, presenting well mixed populations (J. Uphoff, MD DNR, personal observation) that made a random sampling assumption reasonable. Treating hook-and-line samples as a cluster required a broad definition of a cluster in Overton et al. (2009), i.e., an entire day’s effort that assumed fish caught that day represented a non-independent sample. Neither assumption (random or cluster) provided a complete description of how hook-and-line sampling works and we believed that random sampling was a better fit.

Two additional objectives of this forage assessment 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 Maryland 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 forage assessment approach represents a less complex, low-cost attempt to integrate forage into Maryland’s Striped Bass 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).

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 arithmetic mean of scaled indices performed as well as the single index estimated by a hierarchical 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.

We have revised our original approach that integrated information for both size classes to one where each size class is evaluated separately. We felt important differences in forage dynamics between size classes were being lost by integrating them. The switch to size specific metrics complicated interpretation of other metrics that encompassed both size classes and could not be split. At this point, it is not apparent how to integrate these metrics, but they are reported and available for review. For this report, the two metrics with targets and thresholds (P0 and PE), hopefully, can alert busy fisheries managers and stakeholders about the status of forage and whether forage concerns merit further attention.

By splitting into small and large fish size classes, the P0 and PE metrics represent four pieces of information. 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).

The P0 and PE targets and thresholds represent a framework for condensing complex ecological information so that it can be communicated simply to decision makers and stakeholders. The target, threshold, or in-between status approach for P0 and PE was similar to traffic light style representations (but without the colors) 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 P0 and PE, 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 and simplicity and communicability are issues of over-riding importance (Halliday et al. 2001). Integration has two aspects, scaling the indicators to make them comparable (target, threshold, or in-between status in our case) and applying an operation to summarize the results from many indicators. Caddy (1998) presented the simplest case for single-species management where indicators were scaled by converting their values to traffic lights (red, yellow, and green), and decisions were made based on the proportion of the indicators that were red. In 2021, the P0 and PE indicators for both size classes would not have

been red; the three indicators with target and limit values would have been green.

Recent discussions with DNR fisheries managers and stock assessment scientists have indicated a preference for a stoplight approach for forage assessment in Maryland's portion of Chesapeake Bay based on time-series lower quartiles and medians. This approach has been developed for portraying the status of ages 1+ Atlantic Menhaden in Maryland's portion of the Bay and the P0 metric for large Striped Bass (J. Uphoff, MD DNR, personal communication).

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Table 1. Important abbreviations and definitions.

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

Table 2. 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 (CBEF) during 2006-2013 and MD DNR Fish and Wildlife Health Program (FWHP) after 2013. 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	1st date	Last date	Source
2006	19	118	49	2-Oct	26-Nov	CBEF
2007	20	76	203	4-Oct	29-Nov	CBEF
2008	15	29	207	4-Oct	25-Nov	CBEF
2009	17	99	240	3-Oct	25-Nov	CBEF
2010	22	112	317	9-Oct	29-Nov	CBEF
2011	19	74	327	1-Oct	26-Nov	CBEF
2012	11	47	300	7-Oct	30-Nov	CBEF
2013	14	191	228	3-Oct	18-Nov	CBEF
2014	7	121	84	2-Oct	12-Nov	FWHP
2015	8	174	173	24-Sep	17-Nov	FWHP
2016	12	165	260	3-Oct	16-Nov	FWHP
2017	9	271	52	2-Oct	13-Nov	FWHP
2018	6	260	87	3-Oct	28-Nov	FWHP
2019	8	135	90	1-Oct	19-Nov	FWHP
2020	10	116	120	7-Oct	19-Nov	FWHP
2021	8	126	185	14-Oct	30-Nov	FWHP
2022	7	88	256	17-Oct	30-Nov	FWHP

Table 3. Summary statistics for total lengths of Striped Bass in the small category (<457 mm, TL) during 2006-2022.

	Mean	Median	Mode	Minimum	Maximum
2006	400	410	432	302	455
2007	376	356	451	302	451
2008	396	406	445	305	451
2009	381	387	394	286	451
2010	392	397	445	289	451
2011	386	389	451	298	451
2012	381	368	356	298	451
2013	386	387	445	286	454
2014	414	418	427	339	456
2015	359	336	304	229	456
2016	365	364	320	272	452
2017	358	350	340	280	455
2018	380	398	306	286	456
2019	349	348	354	287	456
2020	366	368	401	286	456
2021	414	420	443	303	456
2022	429	437	441	375	456

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 during 2006 -2013 (these regions were one in the same after 2013). Patuxent River seine stations are not included in analyses.

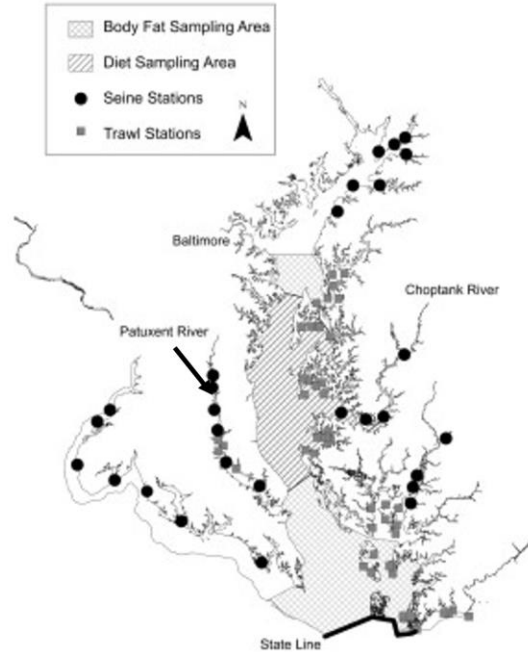


Figure 2. Proportion of small Striped Bass without body fat (P0) during October-November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

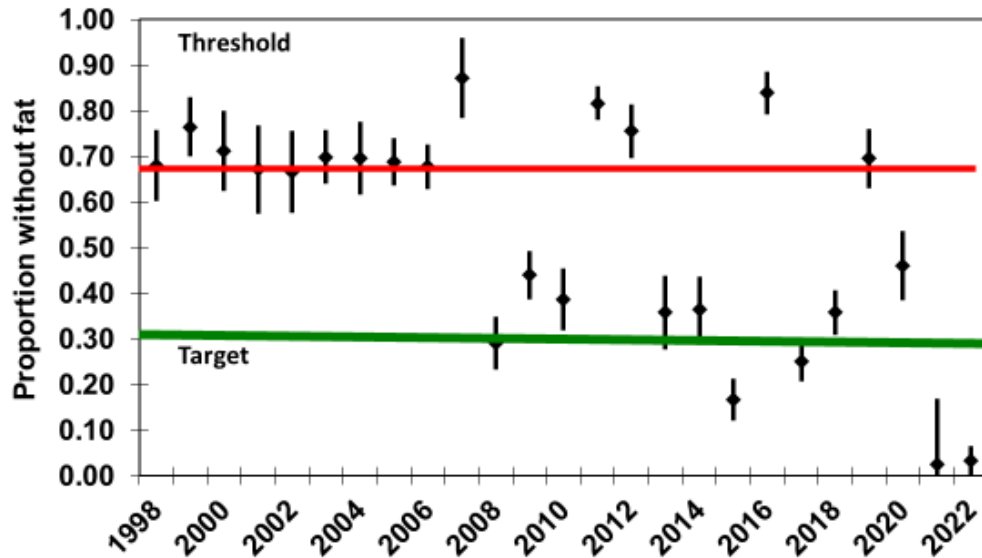


Figure 3. Proportion of small Striped Bass guts without food (PE) in fall and its 90% confidence interval. Red diamond represents threshold PE.

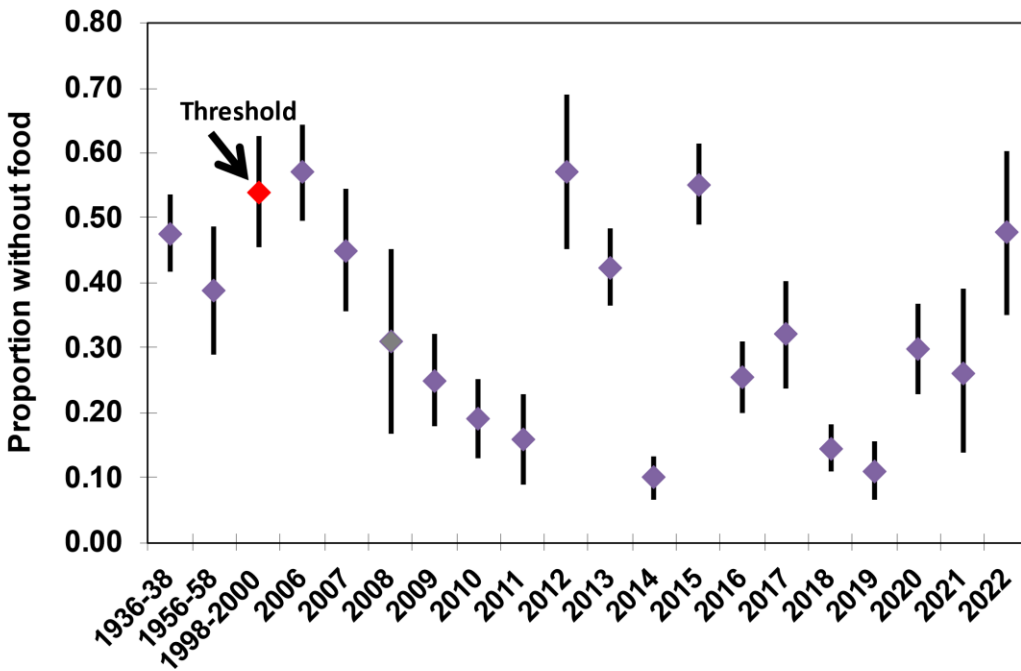


Figure 4. Percent, by number (counts of individuals plus presence of parts), of identifiable (excludes unknown) major forage groups in small Striped Bass (< 457 mm TL) guts, in fall.

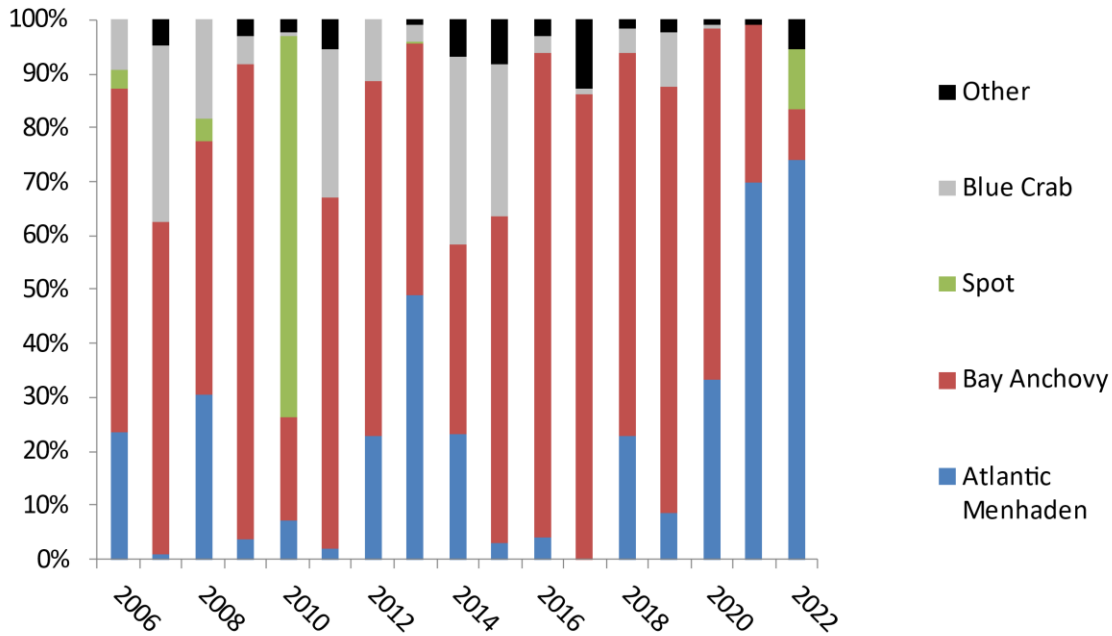


Figure 5. Frequency of occurrence and cumulative percentage of age 0 Atlantic Menhaden in guts of Striped Bass between 335 and 457 mm, TL (small Striped Bass), during 2013 - 2022. Graph starts at the smallest TL of small Striped Bass that consumed Atlantic Menhaden.

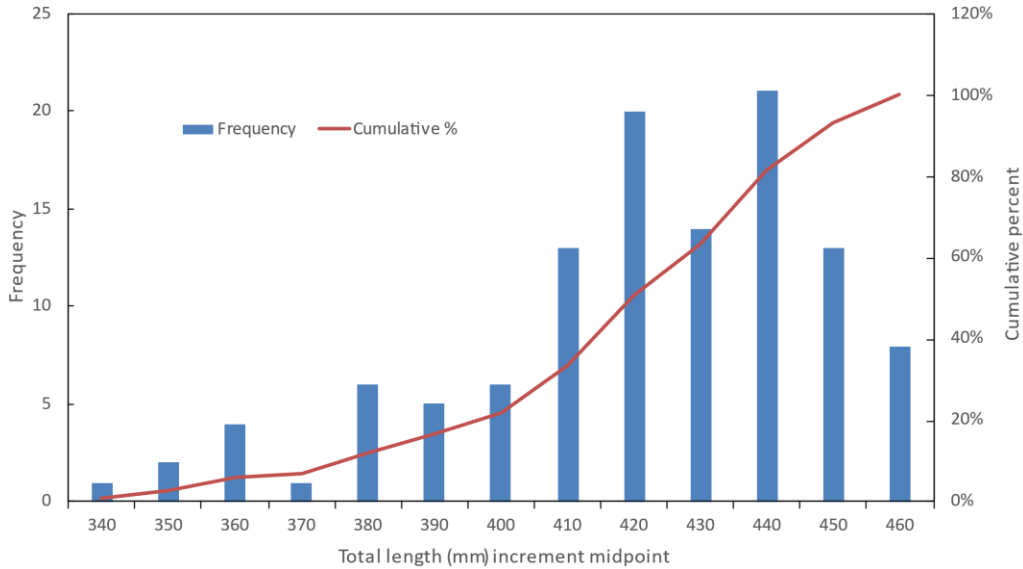


Figure 6. Gram prey consumed per gram (C) of small (< 457 mm TL) Striped Bass in fall hook and-line samples. Age-0 forage dominate the diet. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

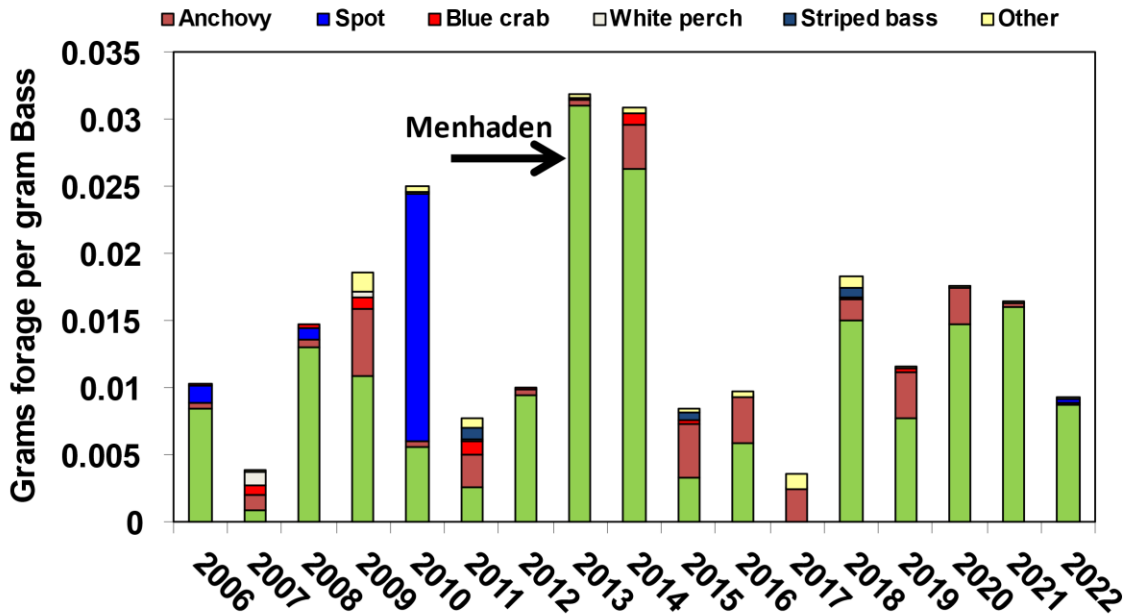


Figure 7. Median prey -predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for small (< 457 mm) Striped Bass. Optimum ratio was estimated by Overton et al. (2009).

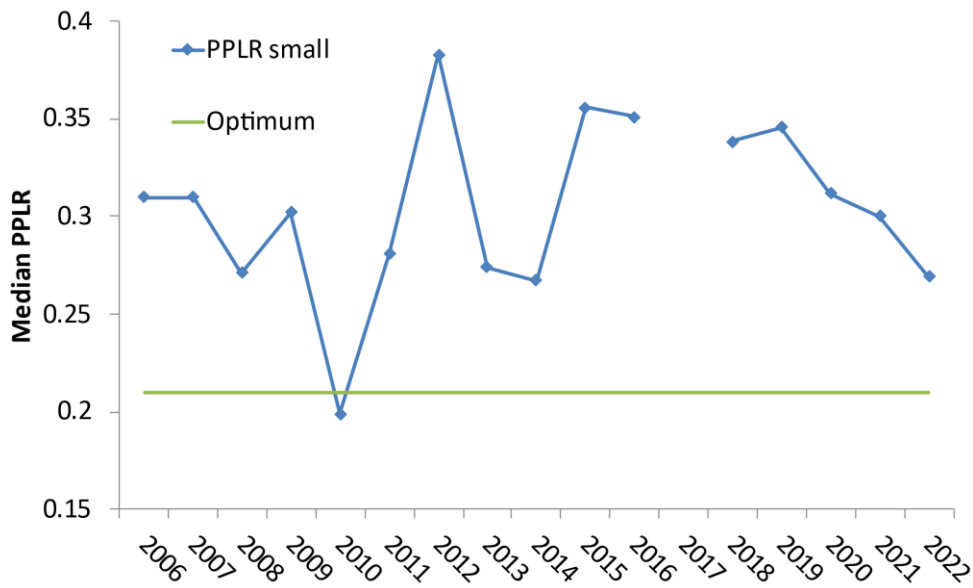


Figure 8. Proportion of large Striped Bass without body fat (P0) during October-November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

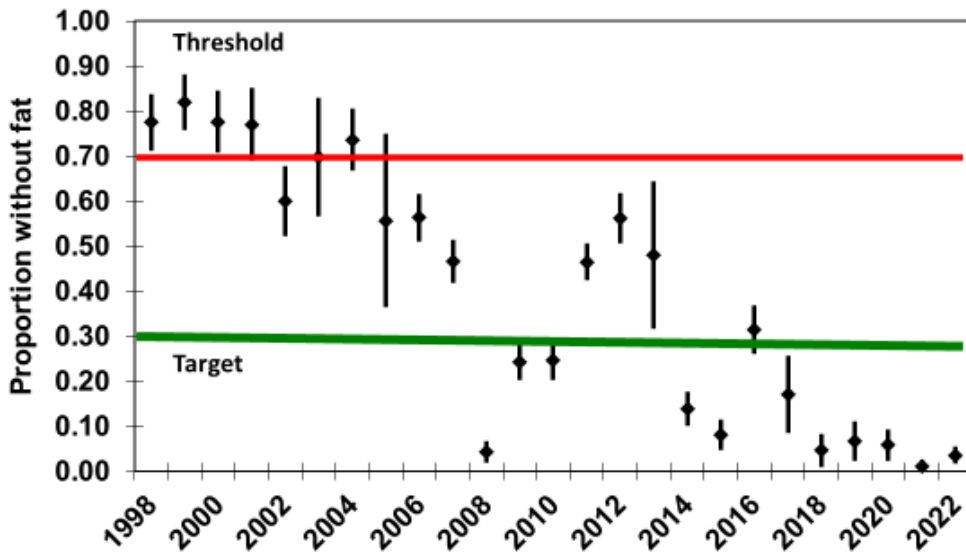


Figure 9. Proportion of large Striped Bass (≥ 457 mm or 18 in, TL) guts without food (PE) in fall and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

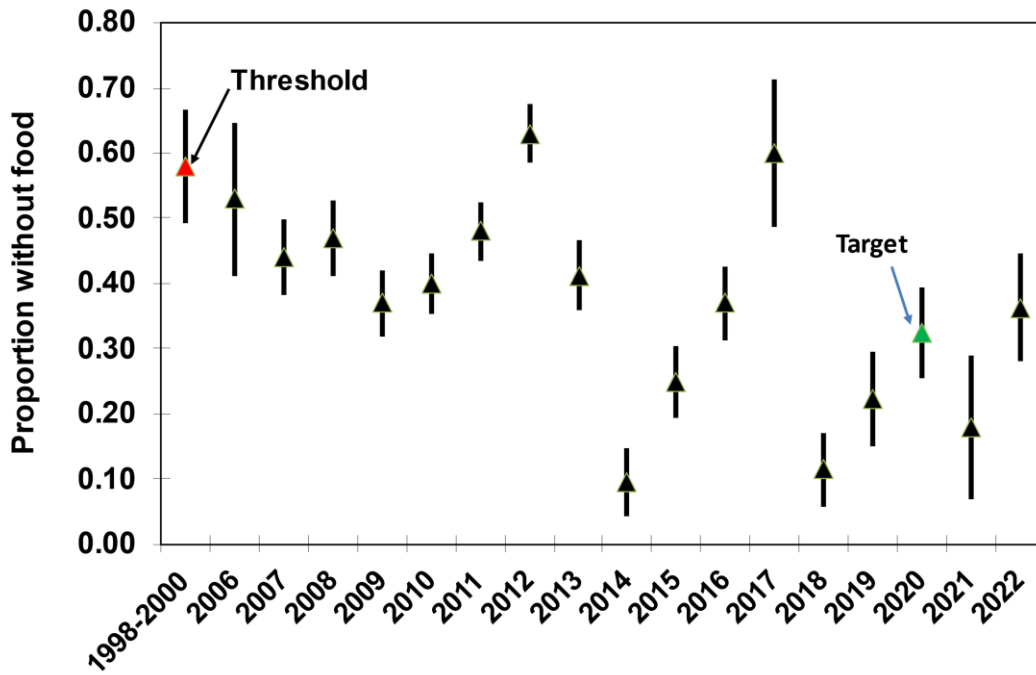


Figure 10. Percent of large Striped Bass (≥ 457 mm TL) identifiable diet represented by major forage groups, by number, in fall.

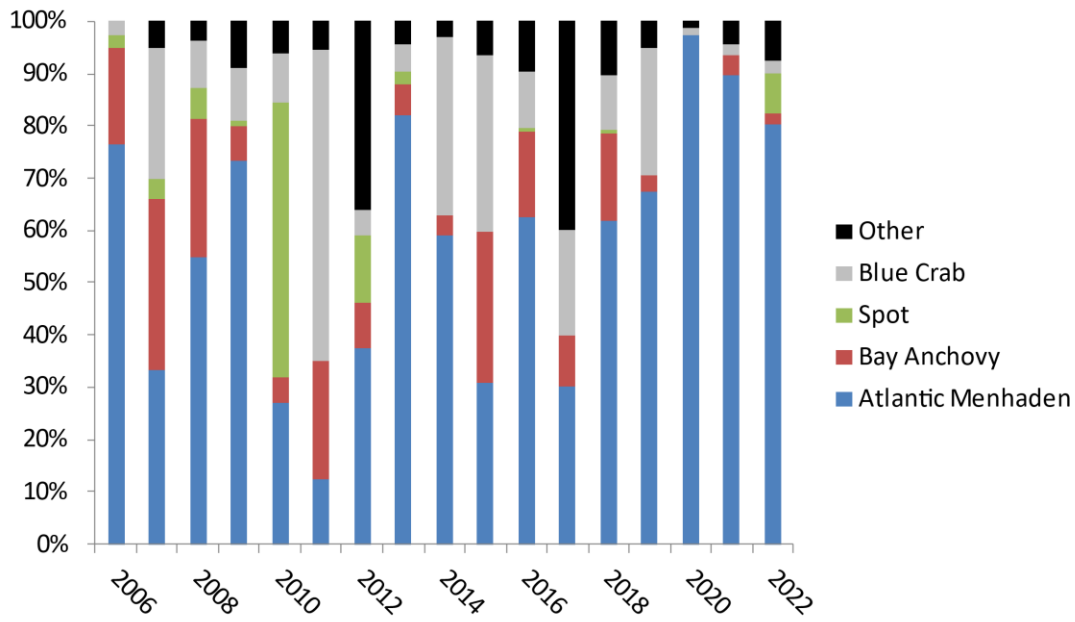


Figure 11. Grams of prey consumed per gram (C) of large (≥ 457 mm TL) Striped Bass during October -November. Fall consumption dominated by age 0 forage. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

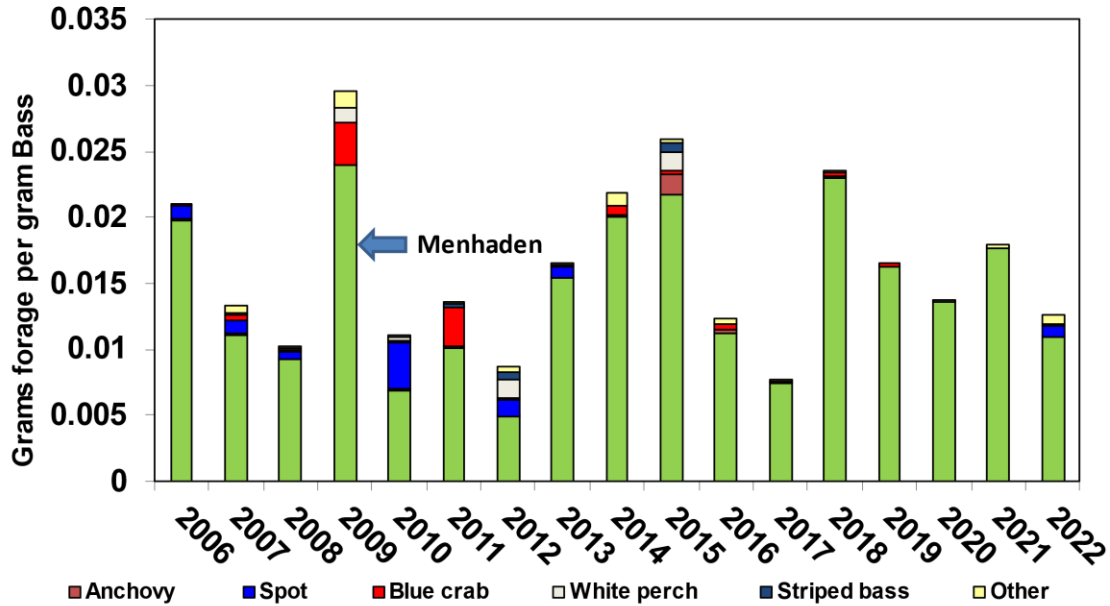


Figure 12. Median prey-predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for large Striped Bass (≥ 457 mm). Optimum ratio was estimated by Overton et al. (2009).

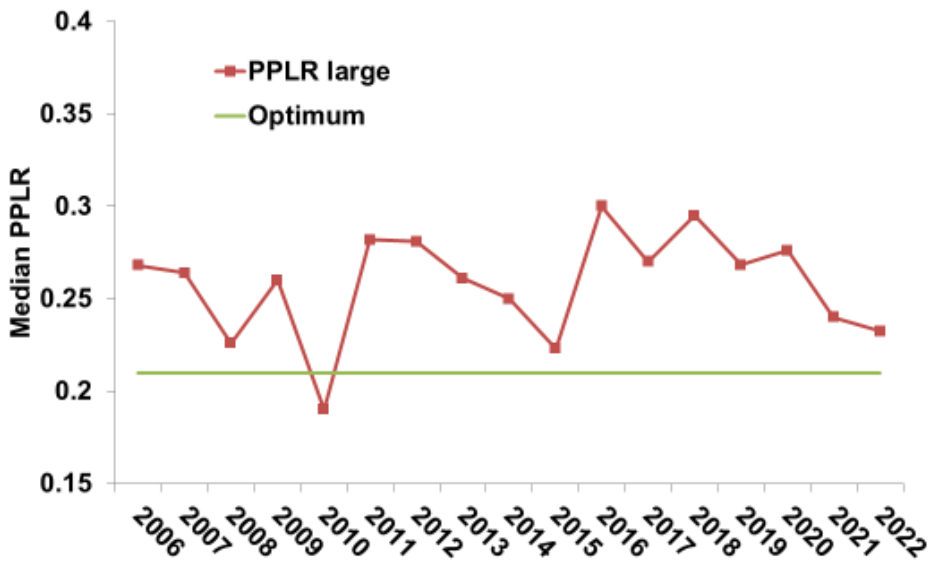


Figure 13. Maryland resident Bay Striped Bass annual abundance index (RI; MD MRIP inshore recreational catch per private boat trip during September-October; mean = black line) during 1983-2022 and its 90% confidence intervals based on @Risk simulations of catch and effort distributions. Catch = number harvested and released.

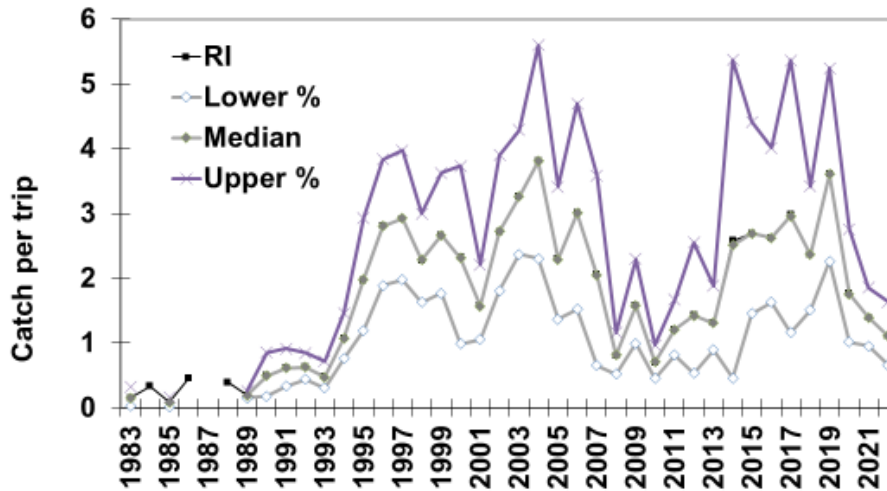


Figure 14. Maryland resident Bay Striped Bass annual abundance index (RI; MD MRIP inshore recreational catch per private boat trip during September-October; mean = black line) during 1983-2020 and estimated coastwide abundance of ages 2-5 from ASMFC (2022) stock assessment (SCAM) update. Catch = number harvested and released.

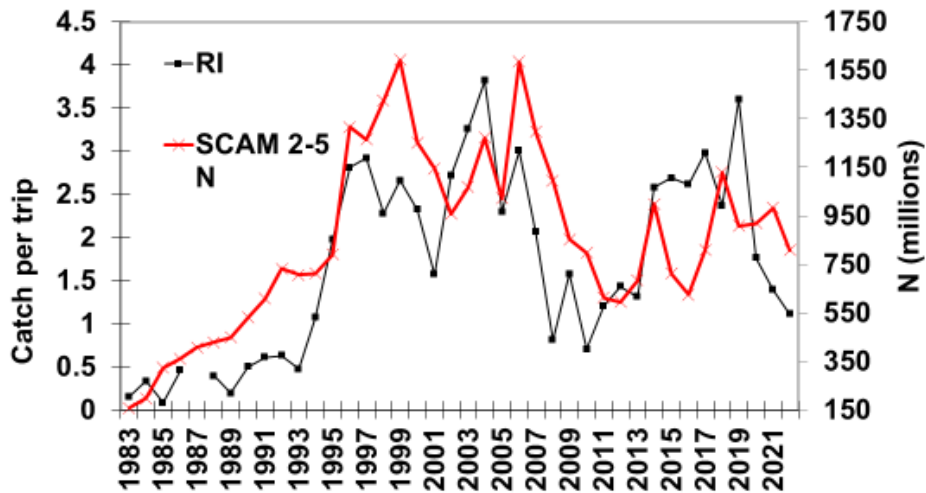


Figure 15. Trends in major pelagic prey of Striped Bass in Maryland Chesapeake Bay surveys since 1959. Indices were standardized to their 1989-2021 means (years in common). Menhaden = Atlantic Menhaden and Anchovy = Bay Anchovy.

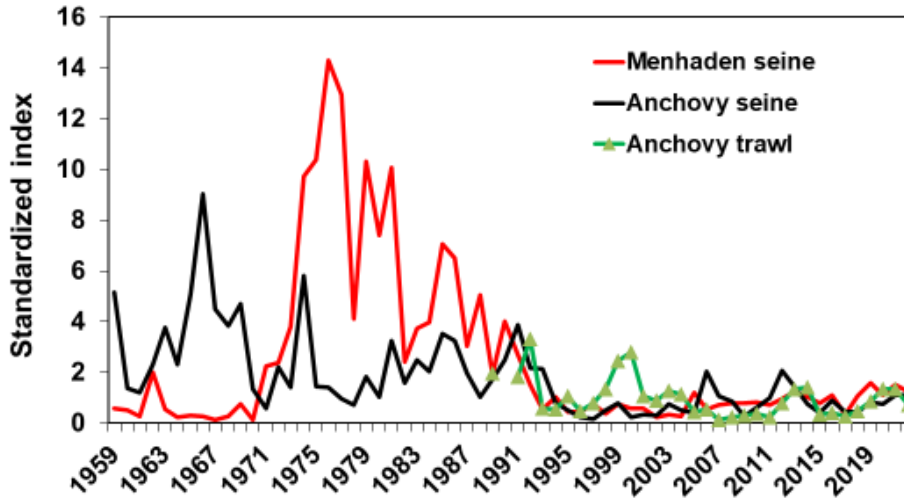


Figure 16. Trends in major benthic prey of Striped Bass in Maryland Chesapeake Bay surveys, since 1959. Indices were standardized to their 1989-2022 means (years in common).

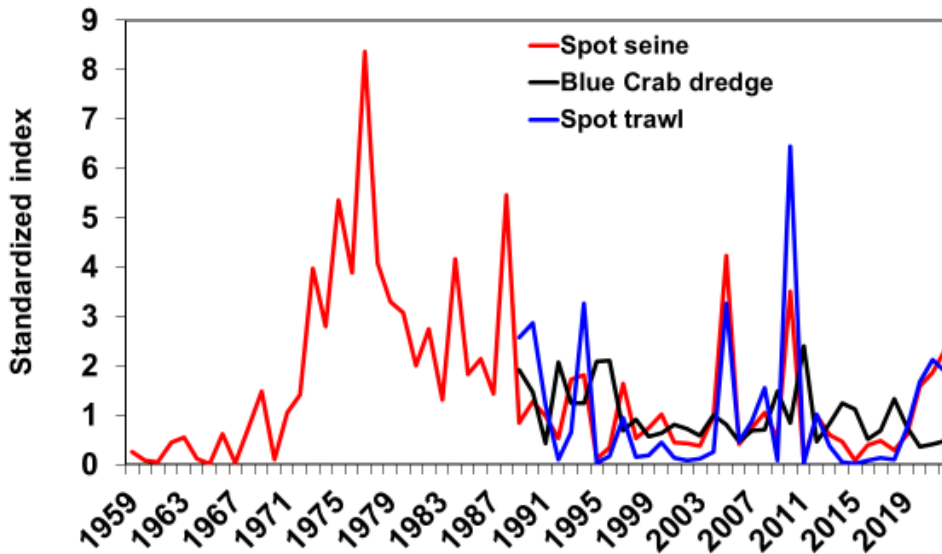


Figure 17. Trends in soft bottom benthic invertebrate biomass in Maryland waters (grams / m²) and its median during 1995-2022 (based on Figure 3-37 in Versar 2023).

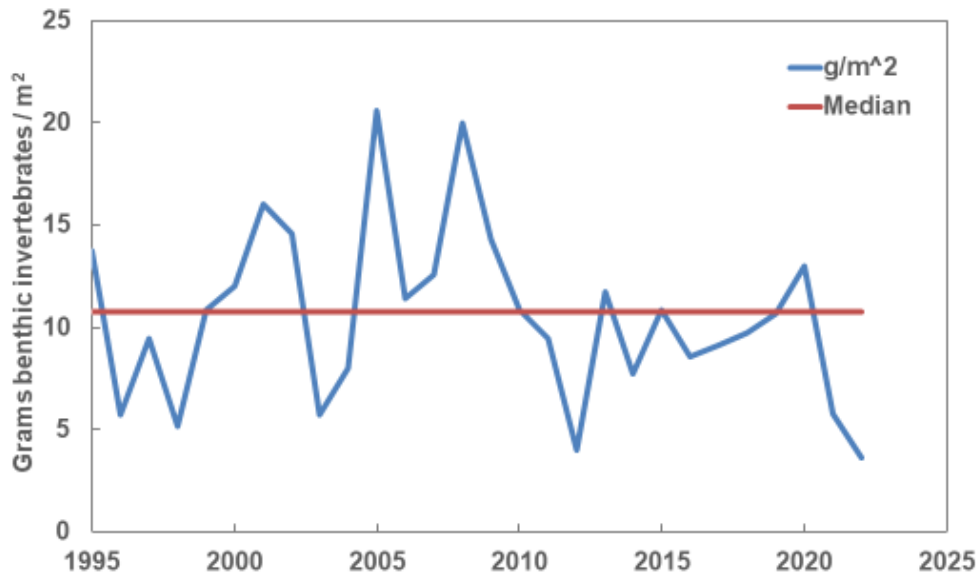


Figure 18. Trends of standardized ratios of major upper Bay forage species indices to Striped Bass relative abundance (RI). Forage ratios have been standardized to their 1989-2022 mean to place them on the same scale. S indicates a seine survey index; T indicates a trawl survey index; and D indicates a dredge index. Note the log₁₀ scale on Y-axis.

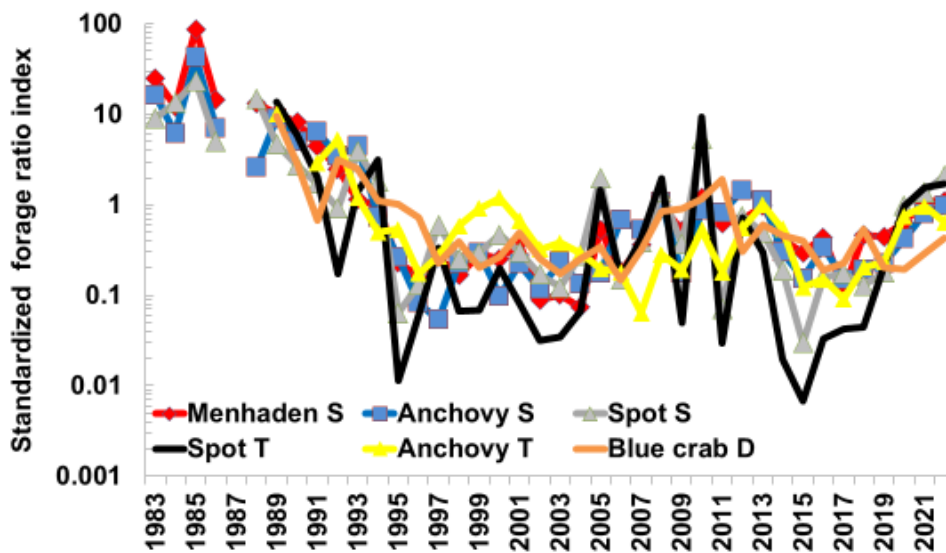


Figure 19. Atlantic Menhaden index to Striped Bass index (RI) ratios (Atlantic Menhaden FR) since 1983 and their 90% confidence intervals based on @Risk simulations of Atlantic Menhaden seine indices and RI distributions. Note \log_{10} scale on the Y-axis.

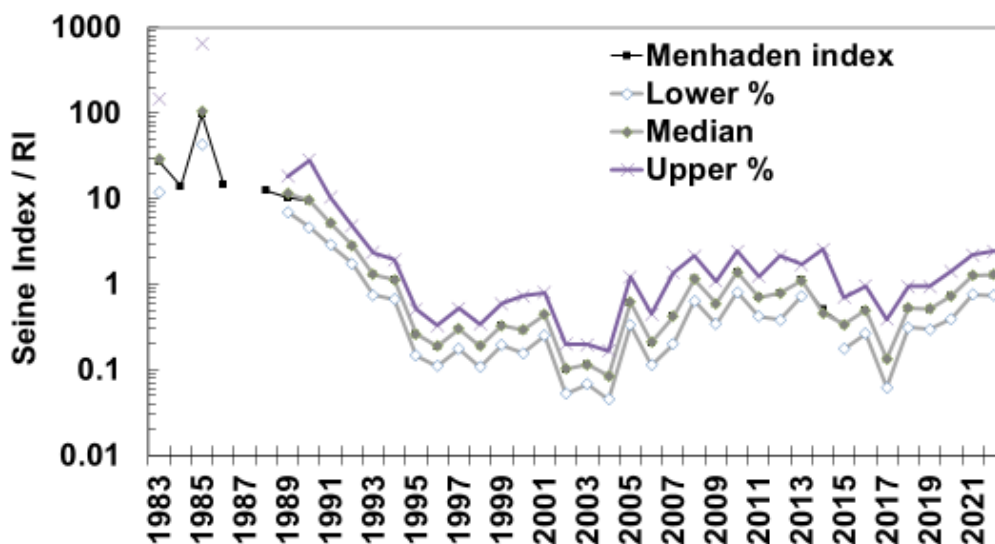


Figure 20. Bay Anchovy seine index to Striped Bass index (RI) ratios (Bay Anchovy seine FR) since 1983 and their 90% confidence intervals based on @Risk simulations of Bay Anchovy seine indices and RI distributions. Note \log_{10} scale on the Y-axis.

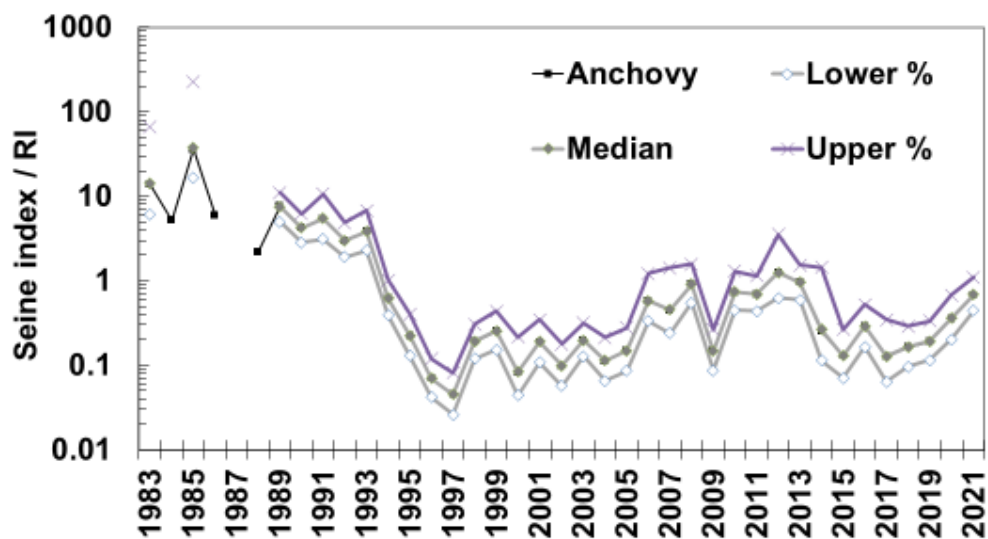


Figure 21. Bay Anchovy trawl index to Striped Bass index (RI) ratios (Bay Anchovy trawl FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note \log_{10} scale on the Y-axis.

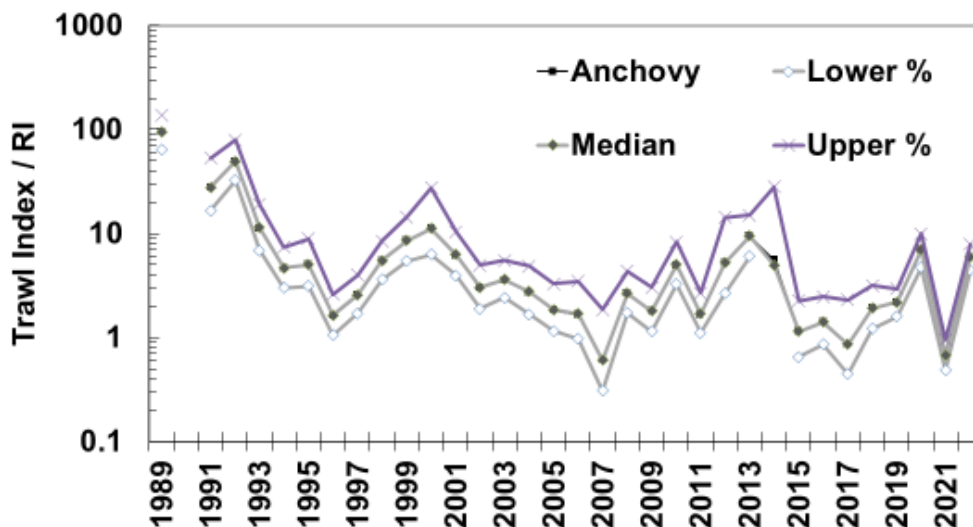


Figure 22. Spot seine index to Striped Bass index (RI) ratios (Spot seine FR) since 1983 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Spot seine indices and RI. Note \log_{10} scale on Y-axis.

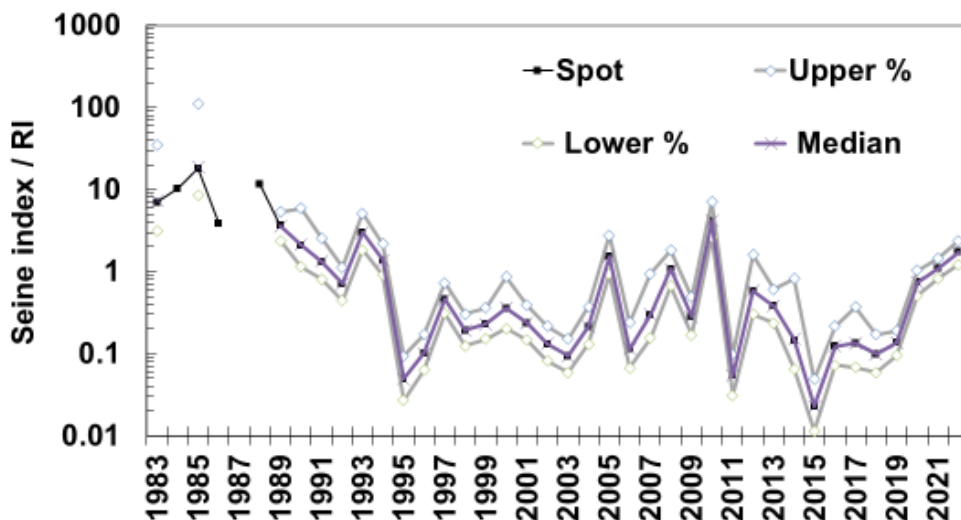


Figure 23. Spot trawl index to Striped Bass index (RI) ratios (Spot trawl FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note \log_{10} scale on Y-axis.

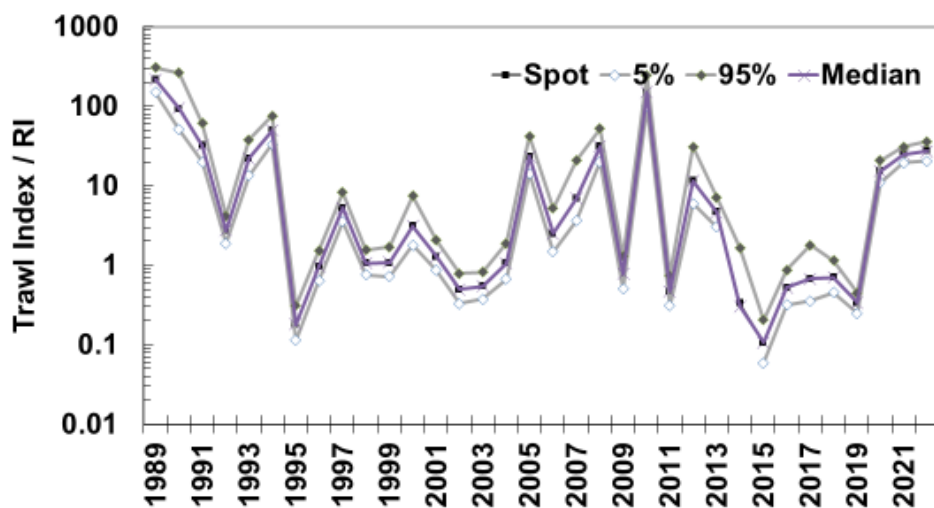


Figure 24. Blue Crab index to Striped Bass index (RI) ratios (Blue Crab FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Blue Crab (age 0) winter dredge densities and RI. Note the \log_{10} scale on Y-axis.

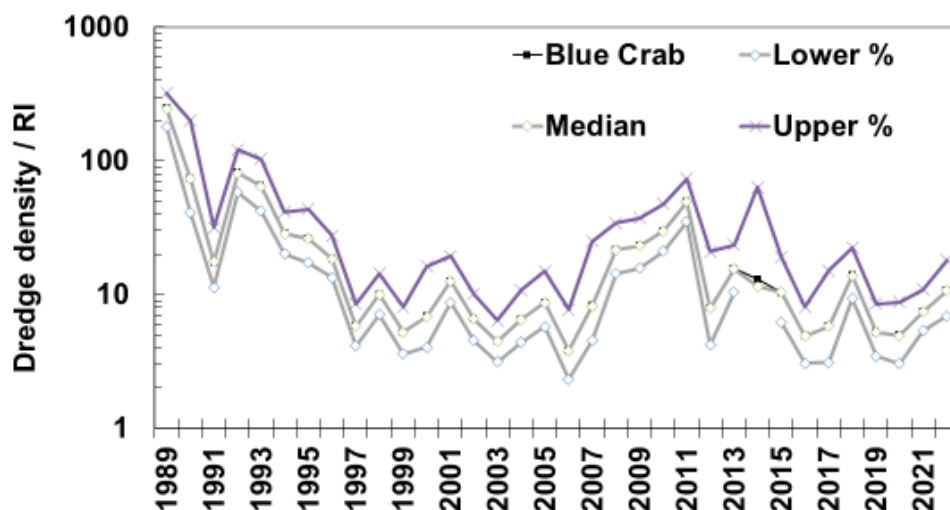


Figure 25. Time-series of age 3 Striped Bass relative abundance on two major Maryland spawning areas (hybrid index = gill net index adjusted for changing catchability during 1985-1995; units = number of fish captured in 1000 square yards of net per hour) and abundance of age 3 Striped Bass along the Atlantic Coast estimated by the ASMFC (2022) statistical catch-at-age model. Hybrid index time series = 1985-2022; Statistical catch-at-age model time-series = 1985-2021. Unadjusted = gill net index not adjusted for catchability during 1985-1995. An estimate was not made for 2021

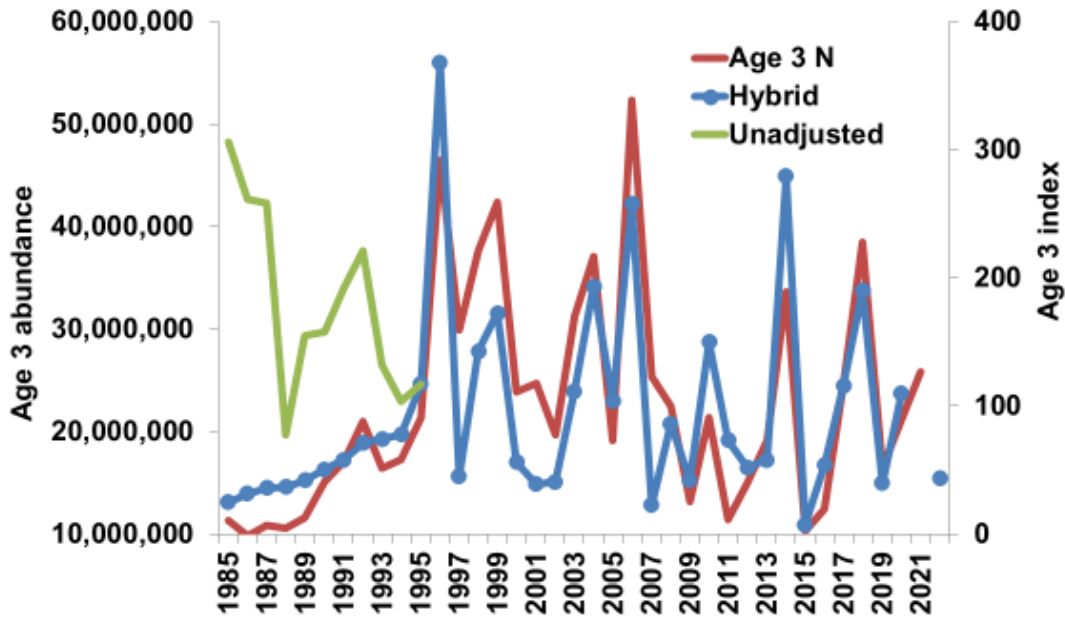


Figure 26. Relative survival (SR) of a Striped Bass year-class to approximately its third birthday during 1985-2022 and 90% confidence intervals based on @Risk simulations of age 3 hybrid gill net indices divided by juvenile index distributions. Year of estimate = year-class + 3. An estimate was not available for 2021 (2018 year-class).

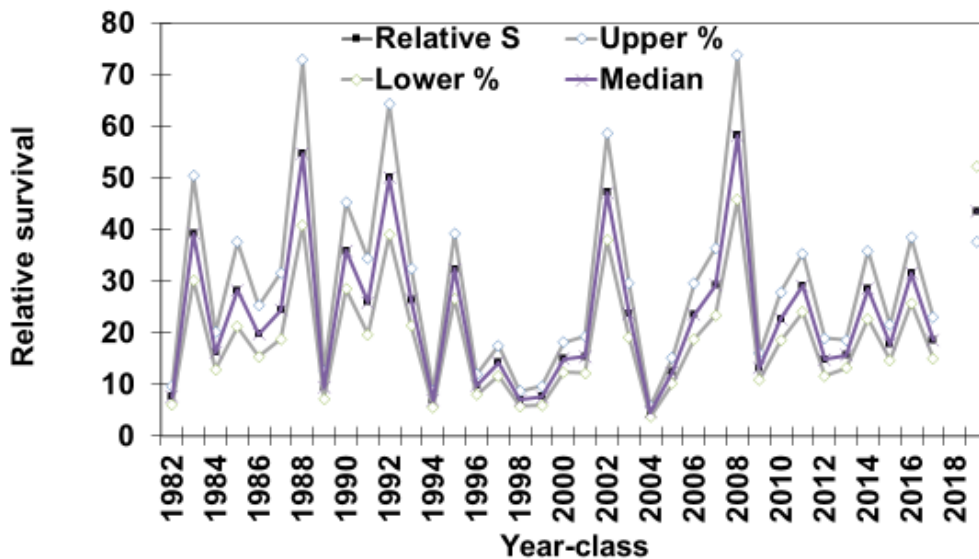


Figure 27. Relative survival (SR) of Striped Bass between ages 0 and 3, its median, and the relative abundance of resident Striped Bass (RI) in the previous year during 1985-2022 (year-class = year - 3). An estimate was not available for 2021.

