PERFORMANCE REPORT FOR FEDERAL AID GRANT F-63-R, SEGMENT 14

2023 Activities

MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

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Fishing and Boating Services
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September 2024

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This grant was funded by the State of Maryland Fisheries Research and Development Fund

and

Federal Aid in Sport Fish Restoration Acts (Dingell-Johnson/Wallop-Breaux)

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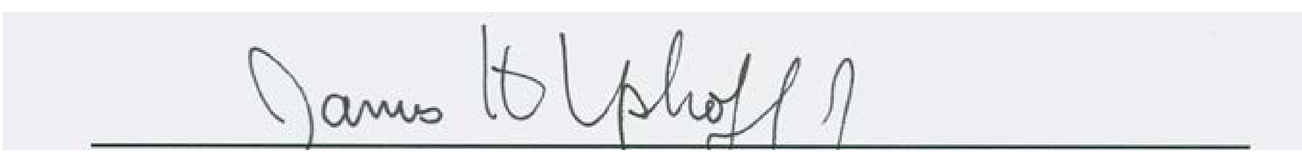

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Acknowledgements

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The Fisheries Habitat and Ecosystem Program, funded by F-63, was elevated into a division within Maryland Department of Natural Resource's Fishing and Boating Services in October 2023 and renamed the Fisheries Ecosystem Assessment Division (FEAD). Alexis Park and Carrie Hoover left FEAD during the report cycle. Shannon Moorhead was hired to replace Margaret McGinty (retired in 2022); Marisa Ponte replaced Carrie Hoover; and Jeffrey Horne replaced Alexis Park. Marek Topolski was added to the federal aid grant.

Report Organization

This report was completed during September 2024. It consists of summaries of activities under this grant cycle. All pages are numbered sequentially; there are no separate page numbering systems for each objective. Objectives are reported in separate numbered sections. Objective 1 is broken into different sections that encompass a particular subject. Throughout the report, multiple references to past annual report analyses are made. The complete PDF versions of many 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. Objective and section number, topic covered, and page number.

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

Objective 1 - Assess land use and aquatic habitat change effects on recreationally important fish populations in tidal tributaries to Chesapeake Bay and establish and confirm habitat reference points.

Sections 1-3 Executive Summary

Purpose - Objective 1 primarily 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 Maryland 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 can be converted to percent impervious surface (%IS) using a regression equation. 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 Chesapeake Bay 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. Severe degradation would be expected at 1.51 C/ha (15% IS) and beyond.

Section 1: Stream Ichthyoplankton – We did not sample spawning streams in this report cycle because of insufficient staff and the need to investigate Striped Bass spawning and nursery habitat after five years of poor year-class success.

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 River and Mattawoman Creek in 2023 during late winter – early spring. 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. Choptank River has a rural, agricultural watershed while Mattowman Creek is forested with development surpassing a threshold level. Estimates of L_p from 2023 were compared to thresholds from brackish or tidal-fresh subestuaries based on a time-series of surveys from subestuaries with rural to urban watersheds stretching back to 1963.

The estimate of mean L_p for Mattawoman Creek in 2023 (L_p = 0.68, SD = 0.08) was above the tidal-fresh threshold (0.59). Choptank River (L_p = 0.64, SD = 0.08) was above the brackish threshold (0.40), during 2023.

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 Choptank River survey did not cover the general hypotheses about development. It investigated habitat conditions for Striped Bass eggs and larvae, reflecting concern about 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 2014-2023, we collected basic water quality data (temperature, conductivity, dissolved oxygen or DO, pH, and alkalinity) from Choptank River 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 2023. Estimated baywide *Ep*, based on Choptank River *Ep* in 2023 (0.74), was within the range of baywide values exhibited in the 1960s-1970s and since 1989. It 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. Estimated RLS in 2023 was below a poor survival criterion.

We examined four spawning milestone dates that bracketed when spawning began, peaked, and ended: date that the first egg was collected, and the dates when 12ºC, 16 ºC, and 20ºC were consistently met. The first egg was collected on March 30, 2023, the third earliest date that spawning has been detected in Choptank River ichthyoplankton surveys and the second earliest date that 12ºC was reached. The mid-milestone, 16ºC, was reached between April 4 and April 7. The 12-16°C span was breached between April 5 and 6 and lasted no more than 8 days. The 20ºC milestone was reached on April 17 and 2023 is tied with 2002 for the earliest this milestone was reached. These milestones have been reached earlier and more often since 2017. The 2023 spawning season, based on the dates that 12°C and 21°C were reached, ran less than 19 days. This was a short, temperature driven spawning season.

We updated average annual 2-month flows estimated for periods immediately before and during spawning for the Head-of-Bay, Potomac River, Choptank River, and Nanticoke River. Standardized flows were below average baseline flow of 1957-2020 during 2023 in Choptank River (0.67), Nanticoke River (0.41), Head-of-Bay (0.66), and Potomac River (0.46; Figure

2.1.15). The spawning area was smaller than normal in 2023, reflecting low winter-spring precipitation and flow.

Measurements of pH in Choptank River during April 1-May 8 between 1986-1991 and 2013-2023 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 2021-2023 compared to 1986-1991. It seems unlikely that poor year-class success during 2019-2023 could be attributed to a return of toxic water quality conditions (metals, low pH, and low buffering) implicated in poor recruitment during the 1980s.

Section 2.2 - Derivation of Criteria for Postlarval Striped Bass Feeding Using Historic Choptank River Estimates - We examined Choptank River Striped Bass and White Perch feeding and larval dynamics data during 1981-1986 and 1989-1990 to establish baselines for comparison with 2023 collections. These feeding data existed in a hard to access database and data were reentered. Daily instantaneous mortality and growth of postlarvae (Z and G, respectively) had been estimated for 1980-1990. Feeding success metrics were estimated and compared to Z and G to form criteria for successful and unsuccessful feeding. The 2023 sampling program was not intended to be a full early life history study and we developed the annual ratio of feeding sample sizes (N_{ratio}) for 8-10 mm larvae with that for 5-7 mm larvae as a proxy estimate of survival to consider for 2023 samples. White perch larvae were included because they shared larval habitat and diet with Striped Bass larvae. Contrast in White Perch and Striped Bass feeding success would provide insight on the extent that zooplankton abundance and-or larval feeding ability might be reflected by feeding success.

Feeding incidence of first-feeding Striped Bass larvae (5-7 mm, TL) on copepods (FI_{cope}) was positively linked to G and negatively linked to Z, while feeding incidence on cladocerans (FI_{clad}) were negatively linked. Negative linkage of FI_{clad} and positive linkage of FI_{cope} with G and Z suggested that nutrition from consumption of copepods was greater than from cladocerans. However, positive correlation of 6 mm larval abundance with FI_{clad} suggested that cladocerans may have provided a supplement when first-feeding *Morone* larvae were abundant and higher intraspecific and interspecific competition for copepods was likely. A power function reasonably described the relationship of Z to the N_{ratio}. High estimates of Z occurred when N_{ratio} was 0.36 or less and low Z occurred when it was 0.44 or more. Copepod feeding incidences for White Perch and Striped Bass postlarvae 5-7 mm in Choptank River during the 1980s were similar, but White Perch were more likely than Striped Bass to feed on cladocerans.

Lower Striped Bass postlarval growth rates (G < 0.034) occurred when FI_{cope} was less than 0.38 (Table 2.2.6). High postlarval Z occurred exclusively when FI_{cope} was 0.11 or less; FI_{cope} between 0.22 and 0.24 exhibited one estimate of high Z and two of low Z; and low estimates of Z occurred when FI*cope* was 0.38 or more. However, two of four years exhibiting low Z (1982 and 1989) in Choptank River exhibited a combination of moderate FI_{cope} (0.26 and 0.22, respectively) and high FI_{clad} (> 0.57). Anomalously high Z in 1984 occurred under moderate FI_{cope} (0.24) coupled with the third lowest estimate of F_{clad} (0.34). High Z occurred when high to low FI_{clad} (0.55-0.12) was coupled with low FI_{cope}.

Section 2.3: Striped Bass Larval Feeding – Evaluation of 2023 Data - We collected Striped Bass and White Perch larvae from the Choptank River in 2023 to evaluate their gut contents and metrics of feeding success. We then compared these metrics to criteria developed from historic

Choptank River data in Section 2.2 to assess 2023 larval Striped Bass feeding success and its potential implications for larval growth and mortality.

When assessed against the feeding success and mortality criteria developed in Section 2.2, the FI_{cope} values and the N_{ratio} survival proxy estimated in 2023 suggested that feeding on copepods was successful and postlarval mortality was low. FI_{cope} estimates for both first-feeding (i.e., 5-7 mm TL) and larger (i.e., 8-10 mm TL) Striped Bass postlarvae were higher in 2023 than all historic years (1981-1985, 1987-1990) except 1985. In Section 2.2, we established that high estimates of Z occurred when $FI_{cope} < 0.38$; FI_{cope} of 2023 first-feeding larvae was 1.7-fold higher than this threshold. The N_{ratio} in 2023 was 2.9-fold higher than the minimum N_{ratio} threshold ($N_{ratio} \geq 0.44$) associated with low estimates of Z and 3.5-fold higher than the maximum N_{ratio} threshold ($N_{ratio} \leq 0.36$) associated with high estimates of Z in the historic dataset. The 2023 N_{ratio} was higher than the N_{ratio} calculated from all but one year of historical data (1989, $N_{ratio} = 1.55$).

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 2003-2023 has resulted in 167 subestuary and year combinations; 94 of these combinations have been in mesohaline subestuaries, 18 have been in oligohaline, and 55 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 patches of depleted DO in thick SAV beds seemed important.

Median bottom DO in mesohaline subestuaries increased as agricultural coverage of a watershed 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 (≥ 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 bottom DO below 3 mg/L and abundance and species richness in bottom trawl samples declined. Surface DO did not exhibit noticeable change with development 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 2023, 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 = 1.03), Northeast River (tidal-fresh; C/ha = 0.52), Miles River (mesohaline; C/ha = 0.26) and South River (mesohaline; C/ha = 1.43). Dissolved oxygen was most frequently below the 5 mg/L target or 3 mg/L threshold in bottom channel waters of mesohaline subestuaries. Frequency of below threshold DO increased with development in mesohaline subestuaries and the most severe conditions were observed in South River (100%), followed by Tred Avon River (21%) and Miles Creek (13%). Below target and threshold DO frequency did not reflect level of development in tidal-fresh Mattawoman Creek (0% of measurements were below the target or threshold) and Northeast River (25% and 4%, respectively).

Mattawoman Creek bottom trawl geometric means (GMs) of all species and White Perch GMs from 4.9m bottom trawl catches increased between 2022 and 2023. Bottom trawl GMs were low in 2004–2009, 2011–2012, and 2014–2016; the 2023 trawl GM was greater than those for 2006–2010, 2012, and 2022. Spottail Shiners reappeared in the top 90% of trawl catches in 2023 after disappearing in 2022, a notable change since they have comprised part of the top 90% since sampling started. Species richness in bottom trawl catches increased in 2023 (19) from 2022 (12).

Additional analysis was conducted to see if increased SAV coverage in Mattawoman Creek negatively affected trawl catchability. Linear regressions of SAV coverage estimates and bottom trawl GMs of finfish from 2003 to 2022 for Mattawoman and Piscataway Creeks were compared; SAV coverages were similar over time for the two subestuaries. A relationship was not detected in Piscataway Creek; a strong negative relationship was detected in Mattawoman Creek. Estimated SAV coverage affected catches differently. Changes in catches were more likely to reflect subestuary habitat conditions and overall year-class success of species caught.

Northeast River finfish trawl catches increased in 2023 but remained low compared to 2007–2017; beach seine catches declined in 2023 and were the third lowest GM in the time-series. White Perch GMs for juveniles and adults decreased slightly in 2023. White Perch (juvenile and adult) remained dominant in the top 90% of species in the catch during all sampling years. Seining indicated that Northeast River was an important nursery for Alosids and Gizzard Shad.

Miles River was previously sampled in 2003–2005 and 2020. Little change in C/ha has occurred in this rural watershed but bottom DO has been low in 2005 and 2020 and better in remaining years. Miles River finfish bottom trawl and beach seine GMs increased compared to 2020 GMs; trawl and seine GMs in 2023 were similar to 2003 and 2004. There was an increase in the number of finfish caught and species richness in bottom trawls between 2020 and 2023; trawl species composition changed with greater presence of Spot and absence of White Perch (juvenile and adult); Bay Anchovy remained within the top 90% and Atlantic Croaker appeared for the first time in 2023.

Tred Avon River finfish trawl GMs have increased since 2020 and were at a moderate level, but beach seine GMs were relatively low in 2022–2023. Juvenile White Perch have not been present in trawl catches since 2019; adult White Perch abundance in trawl catches has increased since 2021. Bay Anchovy increased noticeably in 2023, while White Perch (juvenile and adult) disappeared from the top 90%.

South River was previously sampled in 2003–2005 and 2022; C/ha increased from 1.27 to 1.43 C/ha. Low or absent finfish catches in the bottom channel within the upper- and mid-river reflects high development and low bottom DO measurements. Juvenile or adult White Perch were not caught in bottom trawls in 2023. South River's poor habitat is similar to other western shore mesohaline subestuaries with suburbanized watersheds (Severn and Magothy Rivers).

Modified Proportional Stock Density (PSD) time-series indicated that the tidal-fresh subestuaries sampled in 2023 (Mattawoman Creek and Northeast River) were primarily nursery habitat for White Perch too small to be of interest to anglers. Mesohaline subestuaries with extensive low bottom channel DO measurements (South and Miles Rivers) had highly variable PSDs from year to year and their abundance appeared unstable. White Perch of a size of interest to anglers were more likely to be found in subestuaries with rural or transition watersheds and least likely to be found in subestuaries with suburban to urban watersheds.

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

Background for 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; 2023). 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.

Objective 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 live stages (usually eggs or larvae) or limitations on use of habitat for spawning or growth (eggs-adults). In either case, we would expect that stress from habitat would be reflected by dynamics of critical life stages (abundance, survival, growth, condition, etc.).

Development associated with increased population growth converts land use typical of rural areas (farms, wetlands, and forests) to residential and industrial uses (Wheeler et al. 2005; National Research Council or NRC 2009; Brush 2009) that have ecological, economic, and societal consequences (Szaro et al. 1999). Ecological stress from development of the Bay watershed conflicts with demand for fish production and recreational fishing opportunities from its estuary (Uphoff et al. 2011; Uphoff et al 2023). 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) 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. 2023). Habitat reference points based on IS have been developed (ISRPs) for Chesapeake Bay estuarine watersheds (Uphoff et al. 2011; 2023). 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. 2023). 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, page 19**). 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 0.37, 0.86, and 1.35 C/ha, respectively (Uphoff et al. 2023). 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 0.82 C/ha in 2023.

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 (Uphoff 2023). Trends in juvenile indices of these species are similar, indicating similar influences on year-class success (Uphoff 2008; 2023).

Agricultural pesticides and fertilizers were thought to be potential sources of toxic metals implicated in some episodic mortalities of Striped Bass larvae in Bay spawning tributaries in the early 1980s (Uphoff 1989; 1992; Richards and Rago 1999; Uphoff 2008; Uphoff 2023). 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; 2023).

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 2023). 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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Objective 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern

General Spatial and Analytical Methods used in Sections 1-3

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Spatial Methods - We used property tax map-based counts of structures in a watershed, standardized to land hectares (C/ha), as our indicator of development (Uphoff et al. 2012; Topolski 2015). This indicator was estimated by M. Topolski. Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning [MD DOP] 2019). MdProperty View tax data are annually updated by each Maryland jurisdiction to monitor parcel development for tax assessment purposes, although there is typically a two-year lag in processing by MD DOP. Maryland's tax maps are organized by county and updated, maintained, and available electronically in point shapefile format as part of MD DOP's GIS MdProperty View database. Files were managed and geoprocessed using software developed by Environmental Systems Research Institute (ESRI); initially ArcMap 10.1 (ESRI 2012) and beginning in 2019 with ArcGIS Pro 2.4 (ESRI 2019) and newer. 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. Geoprocessing models were developed using Model Builder to automate assembly of statewide tax maps, query tax map data, and assemble summary data.

Watersheds straddle political boundaries; therefore, one statewide tax map was created for each year of available tax data, and then spatially joined to MD DNR 12-digit subwatersheds (herein 12-digit; MD DNR 1998). Records lacking coordinates could not be joined to a subwatershed and were excluded. These subwatershed tax maps were queried for all parcels having a structure built from 1700 to the tax data year. Parcels which did not have year built recorded due to either lack of a primary structure or incomplete data were excluded. Consistent undercounts should not have presented a problem since we were interested in the trend and not absolute magnitude. Estuarine and fresh waters were erased from 12-digit subwatershed polygons to calculate land area for C/ha estimates and joined to the watershed data; shoreline change was accommodated by use of estuarine and riverine shoreline data (MD DNR and MD SHA 2003) and lake/pond data (≥ 1 acre; MD DNR 2006) for historic years through 2012 and land use/land cover (LULC) data developed by Chesapeake Conservancy from 2013/2014 (2018) for years 2013-2016 and from 2017/2018 (2022) for years 2017 on. All watersheds selected for study were mapped by dissolving the constituent 12-digit subwatersheds into one polygon; tax data and land area were summed for each of these watersheds. During 2003-2010 (prior to tax index development), we used percent impervious surface (%IS) and watershed land area estimates made by Towson University from Landsat 30m • 30m 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 to the estimates of %IS calculated by Towson University. 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 time-series revision that addressed inconsistencies found in the data for some watersheds prior 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.

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 used by Towson University (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 (LC) and land use land cover (LULC) data in raster format for the Chesapeake Bay watershed. The LC and LULC rasters created were each a composite of parcel, LiDAR, imagery, and land cover data having varied spatial resolutions for the years 2013/2014. Difference between the LU and LULC rasters is restricted to classifications assigned to pixels identified as land. These data allowed for revised estimates of %IS per 12-digit watershed. Specifically, LC categories Impervious Roads, Impervious Structures, Other Impervious, Tree Canopy over Impervious Roads, Tree Canopy over Impervious Structures, and Tree Canopy over Other Impervious were reclassified to a single Impervious Surface category then summarized by 12-digit 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 these high-resolution raster data sets (Uphoff et al. 2022). The revised model (approximate $R^2 = 0.982$, $P < 0.0001$) indicated that the C/ha to %IS relationship was best described by a nonlinear power function across a broad range of land development (Figure 1). The equation that best described the relationship was

$$\%IS = 11.255 \bullet C/ha^{0.698}.$$

The C/ha equivalents for 5%, 10%, and 15% IS were 0.31, 0.84, and 1.51, respectively (Uphoff et al. 2022). These C/ha estimates are now used as development reference points for fisheries management advice in Maryland's portion of Chesapeake Bay. Recalibration of this relationship was particularly relevant as these high-resolution land cover data have become the authoritative source of current and future on-the-ground conditions. The C/ha conversion allows for retrospective estimates back to 1950. Chesapeake Conservancy's Conservation Innovation Center has since completed a 2017/2018 update and is contracted by the Chesapeake Bay Program to produce high resolution land cover datasets for the years 2021/2022 (Walker et al. 2022). Each modeled watershed %IS estimate was then calibrated using %IS estimates derived from Chesapeake Conservancy LC for that watershed (Table 1):

$$\%IS_c = \%IS_m \bullet \left(\frac{\sum (\%IS_{a,i} / \%IS_{m,i})}{n} \right)$$

where *m* was modeled %IS from tax data, *a* was corresponding estimated %IS from Chesapeake Conservancy LC data, *i* was the tax data year, *n* was the number of *i* tax data years, and *c* represents the calibrated %IS. The general calibration for Maryland watersheds 12-digit or larger is 0.961. The current calibrations are based on two tax data years (2013 and 2018) which align with the Chesapeake Conservancy LC data years.

Generalized LULC polygon shapefiles were available from MD DOP for the years 1973, 1994, 1997, 2002, and 2010 for each Maryland jurisdiction and as aggregated statewide shapefiles. Percent of watershed in agriculture, forest, wetlands, and urban (including commercial, industrial, institutional, and density-based residential classifications; Table 2) categories were estimated for each year of MD DOP spatial data to track broad patterns of LULC. The statewide LULC shapefiles were clipped for each watershed of interest. Once clipped, polygon geometry was recalculated and water polygons were omitted when calculating watershed area; that is only land area was considered when calculating the percentage of each LULC category. A planned 2020 update to MD DOP LULC had not been developed as of December 1, 2023.

The Chesapeake Conservancy high resolution LULC datasets (2013/2014 and 2017/2018) were used to generate comparable categorical (agriculture, forest, wetland, and urban) estimates of LULC to those for MD DOP data. Chesapeake Conservancy LULC rasters for Maryland were comprised of 53 classifications grouped into 18 general classifications (Chesapeake Conservancy 2022) which allowed the data to be directly aggregated into three of the LULC categories: agriculture, forest, and wetland. A developed category comprised of impervious surfaces (excluding roads) and developed land was created and treated as comparable to the urban category (Table 2). Three caveats are worth mentioning. First, the MD DOP forest cover estimates have a minimum mapping unit of 10 acres that mixes tree cover in residential areas (such as trees over turf grass) with true forest cover, clouding interpretation of forest influence (R. Feldt, MD DNR Forest Service, personal communication). In contrast, the Chesapeake Conservancy forest classification was applied to contiguous patches of trees ≥ 1 acre having a patch width of ≥ 240 ft (Chesapeake Conservancy 2022) which lessened the classification of residential tree cover as forested. Second, the urban category used for MD DOP data aligned as much with zoning classifications as implemented land development. Third, urban and developed land classifications were not direct measures of IS but they are closely associated (Uphoff et al. 2011, 2023).

Watersheds used to model %IS underwent modest increase in C/ha from 2010 (last MD DOP LULC data) through 2014 (first Chesapeake Conservancy LULC dataset's end date); %IS increase ranged from 0.008 – 0.35% (median = 0.09). Based on the lack of substantial development across the watersheds during these five years, the quotient of 2010 MD DOP LULC and 2013/2014 Chesapeake Conservancy LULC estimates for each category in each watershed were used to calculate correction factors (Table 3). Percent cover of each category for both Chesapeake Conservancy datasets (2013/2014 and 2017/2018) were calibrated with these correction factors. While the tabular area estimates for broad LULC categories are comparable between the 2010 and calibrated 2013/2014 datasets, there are spatial inconsistencies when the data are overlayed due in part to the differing methodologies for their development. Maryland DOP used a combination of imagery and parcel zoning from tax maps to delineate polygons that were categorized by majority LULC (MD DOP 2004, 2010a); whereas, Chesapeake Conservancy incorporated the spectral characteristics of land, water, and objects along with parcel characteristics, existing land cover datasets, and hydrography to categorize LULC on a pixel by pixel basis (Chesapeake Conservancy 2022).

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 Littel 2006). Multiple regression models accommodated an additional variable (Z):

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

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

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

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

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

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

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 • K; and b was a shape factor (Pielou 1981; Prager et al. 1989).

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Table 1. Conversion factors used to tune modeled %IS for 46 distinct watershed areas. Constraints to watershed area (exclusions and downstream extent) are provided parenthetically. Limited LC data was available for the military installation Aberdeen Proving Grounds (APG).

Watershed	Conversion
Big Annemessex River	0.90957
Blackwater River	1.11667
Bohemia River	1.19891
Breton Bay	1.13059
Broad Creek	1.00455
Broad Creek (Lower Choptank)	0.97328
Bush River (No Swan Creek, No APG)	0.99245
Bush River (No Swan Creek, With APG)	1.1489
Chester River (No Langford Creek, Ichthyoplankton Sampling)	1.029
Chester River	0.98305
Chester River (Striped Bass Spawn)	1.071
Choptank River (No Tuckahoe Creek, River Herring Spawn)	1.0354
Choptank River (Striped Bass Spawn)	1.06834
Corsica River	0.88718
Deer Creek	0.92214
Elk River (Upstream of C&D Canal)	1.03391
Fishing Bay	1.19184
Gunpowder River (No Middle River-Browns)	0.92291
Harris Creek	0.8686
Langford Creek	1.12285
Magothy River	0.9176
Mattawoman Creek	0.87744
Middle River-Browns	1.18443
Miles River	1.14381
Nanjemoy Creek	0.7506
Nanticoke River (Striped Bass Spawn)	1.19312
Northeast River	1.14249
Patapsco River (River Herring Spawn)	1.00286
Patuxent River (River Herring Spawn)	1.01627
Patuxent River (River Herring Spawn Rt214)	1.01888
Patuxent River (River Herring Spawn Rt50)	1.01486
Patuxent River (Striped Bass Spawn)	0.99279
Piscataway Creek	1.05377
Rhode River	0.96043
Sassafras River	1.11536
Severn River	1.0462
South River	1.01575

Watershed	Conversion
St. Clements Bay	1.19942
Transquaking River	1.60934
Tred Avon River	1.0667
Tuckahoe Creek	1.23179
West River & Rhode River	0.93309
West River (No Rhode River)	0.91043
Wicomico River Eastern Shore (Striped Bass Spawn)	1.13332
Wicomico River Western Shore (With Zekiah and Gilbert Swamps)	0.86627
Wye River	1.21716

Table 2. Grouping of MD DOP and Chesapeake Conservancy data into common categories for tracking LULC change over time.

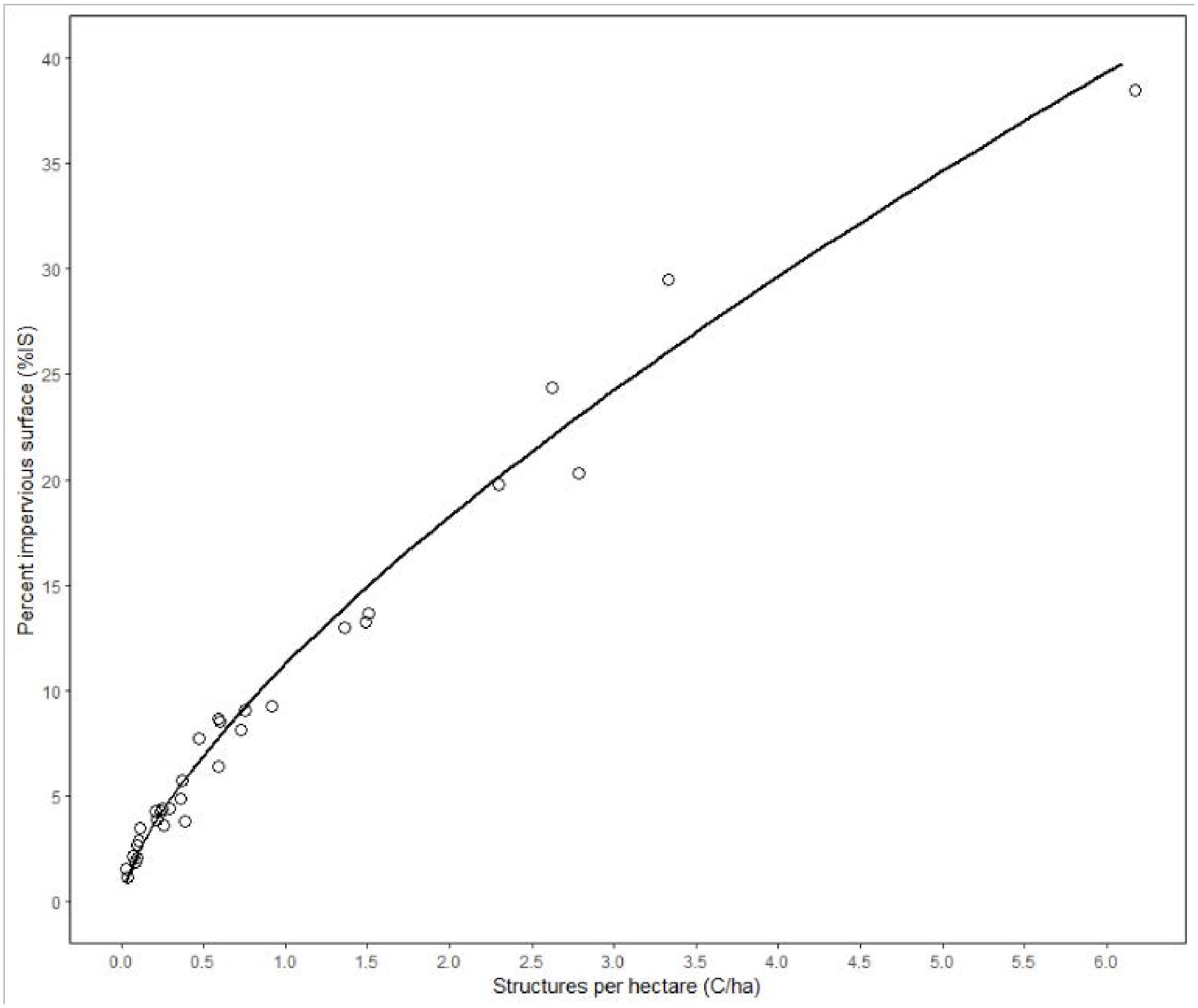
Category	Maryland Department of Planning (MD DOP 2010b)	Chesapeake Conservancy (Chesapeake Conservancy 2022)
Agriculture	21 Cropland 22 Pasture 23 Orchards/vineyards/horticulture 24 Feeding operations 25 Row and garden crops 241 Feeding operations 242 Agricultural building	Cropland Pasture/Hay
Forest	41 Deciduous forest 42 Evergreen forest 43 Mixed forest 44 Brush	Forest Harvested Forest
Urban/Developed	11 Low-density residential 12 Medium-density residential 13 High-density residential 14 Commercial 15 Industrial 16 Institutional 17 Extractive 18 Open urban land 191 Large lot subdivision (agriculture) 192 Large lot subdivision (forest)	Impervious Structures Impervious Other Tree Canopy Over Impervious Turf Grass Tree Canopy Over Turf Grass Extractive Pervious Developed Other
Wetland	60 Wetlands	Riverine Non-Forested Wetlands Terrene Non-Forested Wetlands Tidal Non-Forested Wetlands

Table 3. Conversion factors used to adjust Chesapeake Conservancy LULC based %IS estimates to the same scale as MD DOP LULC based %IS estimates for 46 distinct watershed areas. Land use/land cover category “Ag” refers to agricultural land and “Develop” refers to urban/developed land. Constraints to watershed area (exclusions and downstream extent) are provided parenthetically. Limited LULC data was available for the military installation Aberdeen Proving Grounds (APG).

Watershed	Ag	Forest	Wetland	Develop
Big Annemessex River	1.15721	0.9639	1.01918	1.27206
Blackwater River	1.15904	1.08311	0.90894	0.91757
Bohemia River	1.04254	1.04815	0.60684	1.0929
Breton Bay	1.14962	0.91302	0.65273	1.4022
Broad Creek	1.11331	0.72983	1	1.63247
Broad Creek (Lower Choptank)	1.29679	0.64558	0.13361	1.75333
Bush River (No Swan Creek, No APG)	1.02476	0.9502	5.35677	1.11857
Bush River (No Swan Creek, With APG)	0.84899	1.02911	9.3062	1.0838
Chester River (No Langford Creek, Ichthyoplankton Sampling)	1.09035	0.87969	0.60056	1.17546
Chester River	1.09768	0.87049	0.76906	1.20285
Chester River (Striped Bass Spawn)	1.08273	0.90053	0.4713	1.16819
Choptank River (Striped Bass Spawn)	1.06588	0.84259	0.85203	1.5227
Choptank River (River Herring Spawn)	1.06021	0.80812	0.91886	1.7326
Corsica River	1.07675	0.86071	0.52542	1.28946
Deer Creek	1.09653	0.7542	0.03752	1.56024
Elk River (Upstream of C&D Canal)	1.13515	0.94174	0.84449	1.17471
Fishing Bay	1.08574	0.98541	1.02748	0.97328
Gunpowder River (No Middle River-Browns)	1.06398	0.94303	1.02366	1.24032
Harris Creek	1.20605	0.76685	0.77232	1.38915
Langford Creek	1.08959	0.86335	0.75926	1.34737
Magothy River	1.73238	0.71787	0.01929	1.26263
Mattawoman Creek	1.52274	0.9659	0.6285	1.17981
Middle River-Browns	2.53083	1.06829	1.19737	1.12892
Miles River	1.14355	0.8112	0.38766	1.38233
Nanjemoy Creek	1.32187	0.91048	1.4457	1.83922
Nanticoke River (Striped Bass Spawn)	1.07619	0.89349	1.46076	1.34358
Northeast River	1.05534	0.98377	0.27299	1.20284
Patapsco (River Herring Spawn)	0.97738	0.97734	0.55328	1.24132
Patuxent River (River Herring Spawn)	1.08937	0.88284	0.14474	1.27944
Patuxent River (River Herring Spawn Rt214)	1.06805	0.89106	0.15363	1.27339
Patuxent River (River Herring Spawn Rt50)	1.06129	0.88999	0.1617	1.27771
Patuxent River (Striped Bass Spawn)	1.18023	0.88038	0.81565	1.25417
Piscataway Creek	2.30099	0.9528	0.39971	1.07835

Watershed	Ag	Forest	Wetland	Develop
Rhode River	1.45683	0.86617	0.81097	1.32358
Sassafras River	1.0362	1.00295	0.63273	1.27881
Severn River	1.68073	0.77826	0.29798	1.27277
South River	1.51572	0.82924	0.50116	1.29717
St. Clements Bay	1.10885	0.96659	0.54083	1.25793
Transquaking River	1.07944	0.86527	1.37593	0.83778
Tred Avon	1.27883	0.74908	0.34779	1.22726
Tuckahoe Creek	1.0805	0.88426	0.31396	1.39419
West River & Rhode River	1.46687	0.8501	0.50423	1.25923
West River (No Rhode River)	1.47507	0.82697	0.08662	1.1977
Wicomico River Eastern Shore (Striped Bass Spawn)	1.12147	0.91567	0.83765	1.25528
Wicomico River Western Shore (With Zekiah and Gilbert Swamps)	1.17536	0.92301	0.77014	1.33936
Wye River	1.12845	0.84447	0.35486	1.19005

Figure 1. Relationship of structures per hectare (C/ha) and percent impervious surface (%IS; adapted from “Marine and estuarine finfish ecological and habitat investigations”, Uphoff et al. 2022, p. 40).



Note. Adapted from “Marine and estuarine finfish ecological and habitat investigations”, Uphoff, J. H., Jr., M. McGinty, A. Park, and C. Hoover. 2022, Performance Report for Federal Aid Grant F-63-R, Segment 12, 2021, Maryland Department of Natural Resources, Fishing and Boating Services, Annapolis, Maryland, p. 40.

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

Section 1: Stream Ichthyoplankton

We did not sample spawning streams in this report cycle because of insufficient staff and the need to investigate Striped Bass spawning and nursery habitat after five years of poor year-class success. Mattawoman Creek will be sampled in spring 2024.

Objective 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 Marek Topolski

Introduction

Annual L_p (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 River and Mattawoman Creek in 2023 (Figure 2-1).

Choptank River and Mattawoman Creek are both Coastal Plain watersheds. Choptank River is on the eastern side of Chesapeake Bay. Mattawoman Creek is on the western side and lies along an urban gradient emanating from Washington, DC (Figure 2-1). Mattawoman Creek is a tributary of the Potomac River with a 24,329 Ha watershed. Forest is the primary land use (54% of the watershed), but development is high (C/ha = 1.03; Table 2-1). The subestuary is classified as tidal-fresh (0–0.5 ‰) based on two Chesapeake Bay Program (CBP) monitoring stations; one near the mouth (MAT0016) and another near the confluence of Mattawoman Creek's stream and estuary (MAT0078; MD DNR 2022).

Choptank River is a large tributary of Chesapeake Bay with a watershed of 109,478 Ha. Agriculture is the primary land use (61% of the watershed) and development is low (C/ha = 0.13; Table 2-1). Salinity is mesohaline (5.0 – 18.0 ‰) at the mouth (CBP site EE2.1) and oligohaline (0.5–5.0 ‰) at Ganey's Wharf, in the Yellow Perch larval nursery and Striped Bass spawning area (CBP site ET5.1; MD DNR 2022). 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. After assembling historical data (Uphoff et al. 2020; 2022a), in 2020 we reoriented some of our spring monitoring 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 2023 investigation of Striped Bass egg and larval habitat.

Methods

Conical plankton nets were towed from boats in upper portions of subestuaries to collect Yellow Perch larvae. Nets were 0.5-m in diameter, 1.0-m long, and constructed of 0.5 mm mesh. Nets were towed with the current for two minutes at a speed that maintained the net near the surface (approximately 2.8 km per hour). 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 2023 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 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, Patuxent, and Chester rivers), boundaries were the same as the legal Striped Bass spawning areas. Stratified random designs were used in large rivers with 18 or more sites (Choptank and Nanticoke rivers) and in rivers with 12 sites (Patuxent and Chester rivers), 10 sites were sampled randomly. Sampling was confined to sites with 2.0‰ or less salinity. 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.

During 2023, the Choptank River spawning area was divided into 19 1.61-km segments, starting at km 47.2 and proceeding upstream (Figure 2-2). We could not access two of the furthest upstream stations (stations 17 and 21) because of shallow depths in the last several years. Three segments, 18-20, were in Tuckahoe Creek (starting at the mouth). Segments were aggregated into four subareas. The lower Choptank area consisted of the first 5 segments; the middle, segments 6-11; the upper, segments 12-16; and Tuckahoe Creek, segments 18-20. Barring unsuitable weather and equipment issues, 10 stations were visited during a sampling day. A stratified random design without replacement was used to select three stations each from the lower, middle, and upper mainstem stations and two stations from Tuckahoe Creek.

Mattawoman Creek had 10 stations during 2008-2016 and ideally all were sampled in a visit (Figure 2-3). In 2023, increased sediment had lowered water depths in upper Mattawoman Creek enough that some upper sites could no longer be consistently accessed. There were six survey visits conducted between March 20 and April 11. Of the original ten sites, only sites 6-10 could be consistently sampled; sites 1-2 could be sampled on one visit and sites 3-5 could be sampled on three. Two sites (11 and 12) below sites 1-10 were added on April 3 (Figure 2-3).

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

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

Uphoff et al. (2022b) estimated cumulative frequency of presence by temperature increment and determined that the cumulative catch distribution showed the greatest increase between 12ºC and 18ºC (full time series cumulative proportion equaled 0.93) and 18ºC was adopted as a sampling and analysis cutoff (Uphoff et al. 2022b). In the past, 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. The proportion of tows with Yellow Perch larvae (L_p) for each subestuary and year were recalculated in 2022 based on an 18°C temperature maximum sampling cutoff (Uphoff et al. 2022b).

Methods used to estimate development (C/ha) and land use indicators (percent of watershed with agriculture, forest, wetlands, and urban land uses) are explained in **General Spatial and Analytical Methods used in Job 1, Sections 1-3** (page 19)**.** Development targets and limits, and general statistical methods (analytical strategy and equations) are described there as well.

Estimates of C/ha and MD Department of Planning (DOP) land cover (agriculture, forest, and wetland) percentages were used as measures of watershed land use for analyses from 1973-2010 (MD DOP 2015), while estimates from 2013 on were made from newer Chesapeake Conservancy data (Table 2-1; Chesapeake Conservancy 2023). Updates to MD DOP land use estimates have not been released since 2010 (MD DOP 2015), however in 2013 high resolution land use estimates became available from the Chesapeake Conservancy (Chesapeake Conservancy 2023). Conversion factors were calculated which allowed Chesapeake Conservancy data to correspond to previous MD DOP estimates, effectively making it one continuous data set. Land use estimates for 2010 (MD DOP) and 2013 (Chesapeake Conservancy) are the same as those were the years the conversion factors were created from. 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 2023 for Yellow Perch analyses. Specific spatial and analytical methods for Section 2 are described below. 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 Mattawoman Creek 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 (Figure

2-2). 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, was chosen as a threshold indicating serious deterioration of tidal-fresh larval habitat (Figure 2-2). 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).

Two regression approaches were used to examine possible linear relationships between C/ha and L_p during 1964-2023. 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; Severn River has a suburban level of development (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) and result in low L_p. Ricker (1975) warned against using well correlated variables in multiple regressions, so categorizing salinity for multiple or separate regressions of C/ha against L_p minimized confounding salinity with level of development. These same analyses were repeated using percent agriculture and percent forest land cover estimates in place of C/ha in regressions with L_p. Regression analyses were also used to examine relationships between C/ha, watershed size and salinity, and their effects on L_p.

We used Akaike Information Criteria adjusted for small sample size (AICc) to evaluate the models that describe hypotheses that related changes in L_p to either C/ha for each salinity category (separate slopes), or to C/ha and salinity category (common slopes, separate intercepts; Burnham and Anderson 2001; Freund and Littel 2006):

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

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

Correlation analysis was used to explore associations among temperature, DO, pH, and conductivity during the period Lp was estimated. Of particular interest were associations of DO and pH. Strong to moderate positive correlations of DO and pH would indicate that photosynthesis by phytoplankton may be an important source of pH change in addition to atmospheric deposition, discharges, and watershed runoff.

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 uses (agriculture, forest, or urban), that fell closest to a sampling year, were determined from

Maryland Department of Planning data for all estimates up to 2010 (MD DOP 2020), while Chesapeake Conservancy data was used for estimates from 2013 on (Chesapeake Conservancy 2023). Urban land consisted of high and low density residential, commercial, and institutional acreages (MD DNR 1999).

Results

Sampling in 2023 began on March 13 in Choptank River and lasted until April 26, while sampling on Mattawoman Creek began on March 10 and concluded on April 11. Samples between March 27 and April 11 and March 21 and April 7 were used to estimate L_p in Choptank River and Mattawoman Creek, respectively.

The estimate for Mattawoman Creek of mean L_p in 2023 (L_p = 0.68, SD = 0.08) was above the tidal-fresh threshold (0.59), and above the brackish threshold (0.40) in Choptank River (L_p = 0.64, SD = 0.08), during 2023 (Figure 2-2). The chance that L_p in Mattawoman Creek fell below the tidal-fresh threshold was 11.4% and the chance that L_p fell below the brackish threshold in Choptank River was 0%. Comparisons of L_p during 2023 with historical estimates for brackish subestuaries are plotted in Figure 2-3 and for tidal-fresh estimates in Figure 2-4. 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 ($P \leq 0.0004$; Table 2-2; Figure 2-5). 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 = 0.54, 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 2-5).

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 0.54, 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 0.88 to 0.07 in tidal-fresh subestuaries (Figure 2-5).

Estimates of L_p were weakly and positively related to agriculture (r^2 = 0.14, P = 0.0004) and unrelated to forest (r^2 = 0.004, P = 0.57) in brackish tributaries (Table 2-2; Figure 2-5). Regressions of L_p and agriculture or forest in tidal-fresh subestuaries did not indicate a relationship (Table 2-2), while the multiple regression approach indicated weak positive relationships with L_p (agriculture, r^2 = 0.20, P < 0.0001; and forest, r^2 = 0.11, P = 0.0009; Table 2-2; Figure 2-5). Previous regression analyses did not suggest a relationship of wetland area with L_p in subestuaries of either salinity type so updated 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 Di for brackish or tidal-fresh versus multiple regressions were approximately 2.05 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, $P = 0.0003$ and size, $P = 0.16$). Considering size separately, all tidal-fresh systems are within the small-system size category, so fit did not change from previous analyses ($r^2 = 0.32$, $P = 0.0004$; 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 2023, temperatures were similar between Choptank River and Mattawoman Creek, but DO and pH values were much higher in Mattawoman Creek (Table 2-5; Figure 2-6). Temperatures increased during the larval periods in both subestuaries. Measurements of pH increased in Mattawoman Creek and held steady in Choptank River. Dissolved oxygen declined in Choptank River and held steady in Mattawoman Creek. Conductivity was much higher in Choptank River than Mattawoman Creek, but both were steady (Table 2-5; Figure 2-6).

Correlation analysis of these parameters in Mattawoman Creek during the L_p period (March 27 – April 11; N = 34) indicated a weak association of temperature and DO ($r = 0.075$, $P = 0.67$) and moderate correlations for temperature and pH ($r = 0.55$, $P = 0.0009$) and pH and DO ($r = 0.58$, $P = 0.0003$). Correlation analysis of these parameters in Choptank River during the larval period (March 21 – April 7; N = 504) indicated a strong negative association of temperature and DO ($r = -0.82$, $P < 0.0001$), a poor correlation of temperature and pH ($r = -0.13$, $P = 0.38$) and a moderate correlation of pH and DO ($r = 0.57$, $P < 0.0001$). This correlation analysis suggested that pH and DO were primarily influenced by photosynthesis in Mattawoman Creek. Temperature appeared to play a major role in Choptank River, but there was support for photosynthesis influencing dynamics as well.

Choptank River is a brackish system, while Mattawoman Creek is tidal-fresh, and salinity and conductivity differences between the two are expected, but conductivity was much higher than normal in Choptank River in 2023. The Maryland State Climatologist Office reported that winter 2022-2023 was considerably warmer and drier than the 30-year average (1991-2020), with minimum, mean, and maximum temperatures also being warmer than normal by at least 2.6ºF, 3ºF, and 3.8ºF, respectively (Ruiz-Barradas 2023). Precipitation was also lower over most of the state, averaging approximately 1.4 inches less than normal, with Maryland's 2022-23 winter being the second warmest, and 49th driest on record (1896-2023; Ruiz-Barradas 2023). The lack of winter precipitation brought the salt wedge in Choptank River further upstream than normal, with salinities as high as 2.1 ppt at Station 11 (Ganey's Wharf) and 3.98 ppt at Station 8 (Kingston Landing ramp). In spring 2023 sampling was not conducted below Station 7 due to high salinity. The nursery grounds for Yellow Perch larvae may have been confined to a smaller area than usual.

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. Rural watersheds with below threshold development (at or below C/ha target) in tidal-fresh subestuaries were dominated by forest, and only a single low development, low salinity watershed with agricultural as its dominant land was available (Figure 2-7). Dominant land cover estimates for watersheds of tidal-fresh subestuaries were split between forest (C/ha = 0.46-1.03; 19 observations) and urban (C/ha ≥ 1.17; 14 observations). Nearly all rural land in brackish subestuary watersheds was in agriculture (C/ha ≤ 0.22; 64 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 ≥ 1.24; Table 2-1; Figure 2-7). Tidal-fresh subestuary intercepts may have represented the intercept for forest cover while 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 led to a significant decline in L_p regardless of rural land cover type.

Discussion

Estimates of L_p for Mattawoman Creek and Choptank River in 2023 were above their respective tidal-fresh and brackish thresholds.

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 1.35 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 2.44 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 2001-2003 (Uphoff et al. 2005). 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 Severn River's 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 bottom channel waters of 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 summer habitat assessments began in early July (see Section 3).

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

Widespread low L_p occurs sporadically in Chesapeake Bay subestuaries with rural watersheds and appears to be linked to high winter temperatures (Uphoff et al. 2013). During 1965-2012, estimates of L_p less than 0.5 did not occur in rural subestuaries 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 fecundity was not reduced. 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 *Eurytemora carolleeae*, are important prey of Striped Bass and Yellow Perch larvae (MD Sea Grant 2009; Uphoff et al. 2016), and it has been found that winter water temperatures also have an influence on peak abundances of these zooplankton. *Eurytemora carolleeae* are a particularly important prey item of larval Striped Bass, and their absence may negatively 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 the nursery areas (Uphoff 2023), we would expect these same factors would impact Yellow Perch recruitment.

Yellow Perch egg viability declined in highly developed suburban watersheds of brackish Chesapeake Bay subestuaries (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). Results from Blazer et al. (2013) offered an explanation for low egg viability observed by Uphoff et al. (2005) in Severn River during 2001-2003, as well as 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 Yellow Perch 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 (rural watershed) 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 in western shore subestuaries 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 Mattawoman Creek in 2023 possibly reflect higher primary production, as well as pH-buffering mechanisms (Su et al. 2020), produced by SAV. Beach seining has not been conducted in Mattawoman Creek since 2003 due to high SAV density. Coverage estimates of SAV in Mattawoman Creek ranged from 1.5% to 46. 5% from 1989 to 2021, and estimates were above the time-series median (30.1%) in 2002–2011, 2015–2017, and 2019–2021, with coverage estimates being on the upswing since 2019 (Uphoff et al. 2023). Mattawoman Creek is a shallow tributary of the Potomac River, and SAV has been found to play an important role in shallow lakes, acting as a refuge for zooplankton, and providing diverse ecological niches for these organisms (Bolduc et al. 2016). Bolduc et al. (2016) reported that SAV abundance explained 41% of the variation in the zooplankton community structure and 25% of the variation in zooplankton functional diversity. Their findings suggest that a loss of SAV biomass and complexity can affect both community structure and functional diversity of zooplankton in shallow fluvial lakes. While estimated L_p in Mattawoman Creek was lower in 2023 (0.68) than the median L_p estimated from all years when sampling was conducted there (0.81), it was still above the tidal-fresh threshold (0.59). This could indicate that primary production levels, and therefore primary consumer (zooplankton) levels, still support a functioning nursery habitat.

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). However, most watershed management and restoration practices have the potential to increase organic matter 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 then expanding 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), provides an economical 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 appears to influence 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 1963-2023 and data used for regressions with counts of structures per hectare (C/ha), percent agriculture, percent forest, and percent wetland. Salinity class 0 = tidal-fresh (≤ 2.0‰) and 1 = brackish (> 2.0‰). Land use percentages and overall primary land use were determined from Maryland Department of Planning estimates for 1973-2010 and Chesapeake Conservancy estimates for 2013-present that were closest to a year.

River	Sample Year	LULC Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Bush (w/ APG)	2006	2002	1.17	20.99	36.26	5.55	36.99	Urban	0	0.78
Bush (w/ APG)	2007	2010	1.19	14.94	32.14	5.54	46.44	Urban	0	0.90
Bush (w/ APG)	2008	2010	1.20	14.94	32.14	5.54	46.44	Urban	0	0.69
Bush (w/ APG)	2009	2010	1.21	14.94	32.14	5.54	46.44	Urban	0	0.86
Bush (w/ APG)	2011	2010	1.23	14.94	32.14	5.54	46.44	Urban	0	0.96
Bush (w/ APG)	2012	2010	1.24	14.94	32.14	5.54	46.44	Urban	0	0.34
Bush (w/ APG)	2013	2013	1.25	14.94	32.14	5.54	46.44	Urban	0	0.15
Chester	2019	2018	0.13	66.39	24.46	0.84	8.11	Agriculture	1	0.82
Choptank	1980	1973	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.71
Choptank	1981	1973	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.86
Choptank	1982	1973	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.89
Choptank	1983	1973	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.32
Choptank	1984	1994	0.07	64.03	29.16	2.3	4.15	Agriculture	1	0.71
Choptank	1985	1994	0.07	64.03	29.16	2.3	4.15	Agriculture	1	1.00
Choptank	1986	1994	0.07	64.03	29.16	2.3	4.15	Agriculture	1	0.73
Choptank	1987	1994	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.75
Choptank	1988	1994	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.70
Choptank	1989	1994	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.64
Choptank	1990	1994	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.62
Choptank	1998	1997	0.10	63.6	27.72	2.2	6.44	Agriculture	1	0.57
Choptank	1999	1997	0.11	63.6	27.72	2.2	6.44	Agriculture	1	0.60
Choptank	2000	2000	0.11	63.55	27.5	2.12	6.79	Agriculture	1	0.19
Choptank	2001	2000	0.11	63.55	27.5	2.12	6.79	Agriculture	1	0.25
Choptank	2002	2002	0.11	63.85	27.14	2.02	6.94	Agriculture	1	0.32
Choptank	2003	2002	0.11	63.85	27.14	2.02	6.94	Agriculture	1	0.54
Choptank	2004	2002	0.12	63.85	27.14	2.02	6.94	Agriculture	1	0.50
Choptank	2013	2013	0.13	61.02	25.58	2.11	11.19	Agriculture	1	0.58
Choptank	2014	2013	0.13	61.02	25.58	2.11	11.19	Agriculture	1	0.68
Choptank	2015	2013	0.13	61.02	25.58	2.11	11.19	Agriculture	1	0.81

Table 2-1 cont.

River	Sample Year	LULC Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Choptank	2016	2018	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.59
Choptank	2017	2018	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.43
Choptank	2018	2018	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.44
Choptank	2019	2018	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.68
Choptank	2021	2018	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.44
Choptank	2022	2018	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.46
Choptank	2023	2018	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.64
Corsica	2006	2002	0.21	64.32	27.36	0.43	7.88	Agriculture	1	0.47
Corsica	2007	2010	0.22	60.37	25.51	0.39	13.16	Agriculture	1	0.83
Elk	2010	2010	0.59	28.02	38.7	1.13	31.15	Forest	0	0.75
Elk	2011	2010	0.59	28.02	38.7	1.13	31.15	Forest	0	0.79
Elk	2012	2010	0.60	28.02	38.7	1.13	31.15	Forest	0	0.66
Langford	2007	2010	0.07	70.19	20.35	1.46	7.97	Agriculture	1	0.54
Magothy	2009	2010	2.74	1.24	21.03	0.01	76.77	Urban	1	0.10
Magothy	2016	2018	2.86	1.2	20.42	0.01	77.87	Urban	1	0.10
Mattawoman	1990	1994	0.46	13.76	62.56	0.87	22.52	Forest	0	0.81
Mattawoman	2008	2010	0.87	9.33	53.88	1.13	34.18	Forest	0	0.58
Mattawoman	2009	2010	0.88	9.33	53.88	1.13	34.18	Forest	0	0.90
Mattawoman	2010	2010	0.90	9.33	53.88	1.13	34.18	Forest	0	0.82
Mattawoman	2011	2010	0.91	9.33	53.88	1.13	34.18	Forest	0	0.92
Mattawoman	2012	2010	0.90	9.33	53.88	1.13	34.18	Forest	0	0.20
Mattawoman	2013	2013	0.92	9.33	53.88	1.13	34.18	Forest	0	0.64
Mattawoman	2014	2013	0.93	9.33	53.88	1.13	34.18	Forest	0	0.67
Mattawoman	2015	2013	0.94	9.33	53.88	1.13	34.18	Forest	0	1.00
Mattawoman	2016	2018	0.96	8.63	52.83	1.14	35.65	Forest	0	0.90
Mattawoman	2023	2018	1.03	8.63	52.83	1.14	35.65	Forest	0	0.68
Middle	2012	2010	3.33	3.41	23.32	2.12	70.98	Urban	0	0.00
Nanjemoy	2009	2010	0.09	12.38	68.7	4.09	14.74	Forest	1	0.74
Nanjemoy	2010	2010	0.09	12.38	68.7	4.09	14.74	Forest	1	0.90
Nanjemoy	2011	2010	0.09	12.38	68.7	4.09	14.74	Forest	1	0.92
Nanjemoy	2012	2010	0.09	12.38	68.7	4.09	14.74	Forest	1	0.03
Nanjemoy	2013	2013	0.09	12.38	68.7	4.09	14.74	Forest	1	0.52

Table 2-1 cont.

River	Sample Year	LULC Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Nanjemoy	2014	2013	0.09	12.38	68.7	4.09	14.74	Forest	1	0.88
Nanticoke	1963	1973	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.65
Nanticoke	1964	1973	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.50
Nanticoke	1965	1973	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.34
Nanticoke	1966	1973	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.39
Nanticoke	1967	1973	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.29
Nanticoke	1968	1973	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.40
Nanticoke	1970	1973	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.65
Nanticoke	1971	1973	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.24
Nanticoke	1972	1973	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.26
Nanticoke	1973	1973	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.53
Nanticoke	1974	1973	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.35
Nanticoke	1975	1973	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.48
Nanticoke	1976	1973	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.30
Nanticoke	1977	1973	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.72
Nanticoke	1979	1973	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.30
Nanticoke	1981	1973	0.08	46.57	43.38	8.06	1.92	Agriculture	1	0.39
Nanticoke	2004	2002	0.11	46.3	40.73	7.4	5.54	Agriculture	1	0.49
Nanticoke	2005	2002	0.11	46.3	40.73	7.4	5.54	Agriculture	1	0.67
Nanticoke	2006	2002	0.11	46.3	40.73	7.4	5.54	Agriculture	1	0.35
Nanticoke	2007	2010	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.69
Nanticoke	2008	2010	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.11
Nanticoke	2009	2010	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.32
Nanticoke	2010	2010	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.39
Nanticoke	2011	2010	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.55
Nanticoke	2012	2010	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.04
Nanticoke	2013	2013	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.48
Nanticoke	2014	2013	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.35
Nanticoke	2015	2013	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.59
Nanticoke	2016	2018	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.38
Nanticoke	2017	2018	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.22
Nanticoke	2018	2018	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.28

Table 2-1 cont.

River	Sample Year	LULC Year	C / ha	% Ag	% Forest	% Wetland	% Urban	Primary Land Use	Salinity	Lp
Nanticoke	2019	2018	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.41
Northeast	2010	2010	0.46	31.08	38.65	0.11	28.86	Forest	0	0.68
Northeast	2011	2010	0.46	31.08	38.65	0.11	28.86	Forest	0	1.00
Northeast	2012	2010	0.47	31.08	38.65	0.11	28.86	Forest	0	0.66
Northeast	2013	2013	0.48	31.08	38.65	0.11	28.86	Forest	0	0.72
Northeast	2014	2013	0.48	31.08	38.65	0.11	28.86	Forest	0	0.77
Patuxent	2015	2013	1.24	20.51	35.07	1.02	41.67	Urban	1	0.74
Patuxent	2016	2018	1.25	20.21	33.98	1.07	43.17	Urban	1	0.72
Piscataway	2008	2010	1.41	9.98	40.37	0.24	47.01	Urban	0	0.41
Piscataway	2009	2010	1.43	9.98	40.37	0.24	47.01	Urban	0	0.39
Piscataway	2010	2010	1.45	9.98	40.37	0.24	47.01	Urban	0	0.54
Piscataway	2011	2010	1.46	9.98	40.37	0.24	47.01	Urban	0	0.59
Piscataway	2012	2010	1.47	9.98	40.37	0.24	47.01	Urban	0	0.18
Piscataway	2013	2013	1.50	9.98	40.37	0.24	47.01	Urban	0	0.59
Sassafras	2021	2018	0.11	63.98	25.8	1.28	8.55	Agriculture	0	0.60
Sassafras	2022	2018	0.11	63.98	25.8	1.28	8.55	Agriculture	0	0.82
Severn	2002	2002	2.02	8.57	35.18	0.18	55.84	Urban	1	0.16
Severn	2004	2002	2.09	8.57	35.18	0.18	55.84	Urban	1	0.35
Severn	2005	2002	2.15	8.57	35.18	0.18	55.84	Urban	1	0.40
Severn	2006	2002	2.18	8.57	35.18	0.18	55.84	Urban	1	0.24
Severn	2007	2010	2.21	4.97	27.97	0.2	65.07	Urban	1	0.35
Severn	2008	2010	2.24	4.97	27.97	0.2	65.07	Urban	1	0.08
Severn	2009	2010	2.25	4.97	27.97	0.2	65.07	Urban	1	0.13
Severn	2010	2010	2.26	4.97	27.97	0.2	65.07	Urban	1	0.03
South	2008	2010	1.32	10.24	39.15	0.47	48.82	Urban	1	0.12
Wicomico (ES)	2017	2018	0.68	29.07	37.68	2.03	30.68	Forest	1	0.46
Wicomico (ES)	2018	2018	0.68	29.07	37.68	2.03	30.68	Forest	1	0.34

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	0.88747	0.88747	19.48	<.0001	
Error	83	3.78198	0.04557			
Total	84	4.66944				
r^2	0.1901					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.53647	0.02654	20.21	<.0001	0.48367	0.58927
C / ha	-0.13934	0.03157	-4.41	<.0001	-0.20214	-0.07654

ANOVA			(A) Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.67741	0.67741	15.27	0.0004	
Error	33	1.46425	0.04437			
Total	34	2.14166				
r^2	0.3163					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.90279	0.07257	12.44	<.0001	0.75515	1.05044
C / ha	-0.25001	0.06398	-3.91	0.0004	-0.38019	-0.11983

ANOVA			(A) Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	2.23023	1.11511	24.37	<.0001	
Error	117	5.35352	0.04576			
Total	119	7.58375				
r^2	0.2941					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.81438	0.0458	17.78	<.0001	0.72368	0.90509
C / ha	-0.16055	0.02845	-5.64	<.0001	-0.21689	-0.10421
Salinity	-0.26919	0.04599	-5.85	<.0001	-0.36027	-0.17811

Table 2-2 cont.

ANOVA			(B) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.65953	0.65953	13.65	0.0004	
Error	83	4.00991	0.04831			
Total	84	4.66944				
r^2	0.1412					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.29029	0.05641	5.15	<.0001	0.17809	0.40248
% Ag	0.0043	0.00116	3.69	0.0004	0.00199	0.00662

ANOVA			(B) Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.10753	0.10753	1.74	0.1957	
Error	33	2.03413	0.06164			
Total	34	2.14166				
r^2	0.0502					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.5824	0.06959	8.37	<.0001	0.44082	0.72398
% Ag	0.00397	0.003	1.32	0.1957	-0.00214	0.01008

ANOVA			(B) Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	1.53906	0.76953	14.89	<.0001	
Error	117	6.04469	0.05166			
Total	119	7.58375				
r^2	0.2029					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57718	0.0435	13.27	<.0001	0.49104	0.66332
% Ag	0.00425	0.0011	3.85	0.0002	0.00206	0.00643
Salinity	-0.28452	0.05357	-5.31	<.0001	-0.39062	-0.17842

Table 2-2 cont.

ANOVA			(C) Brackish			
Source	df	SS	MS	F	P	
Model	1	0.01818	0.01818	0.32	0.5705	
Error	83	4.65126	0.05604			
Total	84	4.66944				
r^2	0.0039					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.43212	0.08651	5.00	<.0001	0.26006	0.60419
% Forest	0.00129	0.00227	0.57	0.5705	-0.00322	0.00581

ANOVA			(C) Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.13501	0.13501	2.22	0.1457	
Error	33	2.00665	0.06081			
Total	34	2.14166				
r^2	0.063					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.3928	0.1813	2.17	0.0376	0.02393	0.76166
% Forest	0.00633	0.00424	1.49	0.1457	-0.00231	0.01496

ANOVA			(C) Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	0.86067	0.43033	7.49	0.0009	
Error	117	6.72308	0.05746			
Total	119	7.58375				
r^2	0.1135					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.55234	0.09283	5.95	<.0001	0.36849	0.73619
% Forest	0.00249	0.00201	1.24	0.2183	-0.00149	0.00647
Salinity	-0.1636	0.04927	-3.32	0.0012	-0.26117	-0.06603

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 (C/ha)	MSE	n	K	neg2loge(MSE)	2K	2K(K+1)	(n-K-1)	AICc	Delta brackish	Delta fresh
Categorical	0.04576	120	4	3.084344932	8	40	115	11.4	2.05	1.54
Fresh	0.04437	35	3	3.115191714	6	24	31	9.9		
Brackish	0.04557	85	3	3.088505674	6	24	81	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 ≤ 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			(A) Small Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.67741	0.67741	15.27	0.0004	
Error	33	1.46425	0.04437			
Total	34	2.14166				
r^2	0.3163					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.90279	0.07257	12.44	<.0001	0.75515	1.05044
C / ha	-0.25001	0.06398	-3.91	0.0004	-0.38019	-0.11983

ANOVA			(B) Large Brackish			
Source	df	SS	MS	F	P	
Model	1	0.01996	0.01996	0.54	0.4673	
Error	29	1.06683	0.03679			
Total	30	1.08679				
r^2	0.0184					
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.59465	0.04072	14.6	<.0001	0.51136	0.67794
C / ha	0.09016	0.12241	0.74	0.4673	-0.1602	0.34052

Table 2-5. Summary of water quality parameter statistics for Choptank River and Mattawoman Creek sampled in 2023. Mean pH was calculated from H^+ concentrations, then converted to pH.

System/Year		*Temp C*	*DO (mg/L)*	*Cond (umhols)*	*pH*
Choptank 23	Mean	12.82	9.05	1101.78	7.26
	Standard Error	0.44	0.17	119.70	
	Median	12.62	9.01	804.5	7.26
	Mode	13.60	8.41	-	7.22
	Kurtosis	-0.57	0.90	0.55	2.64
	Skewness	0.21	-0.70	1.17	0.82
	Minimum	8.13	5.58	198	7.05
	Maximum	18.56	11.01	3553	7.63
	Count	50	50	50	50
Mattawoman 23	Mean	14.53	10.99	270.29	8.10
	Standard Error	0.29	0.10	3.93	
	Median	14.51	11.02	266	8.16
	Mode	12.69	10.27	258	8.07
	Kurtosis	-1.73	-1.45	-1.21	-0.48
	Skewness	0.08	0.15	-0.13	0.23
	Minimum	12.06	10.18	226	7.68
	Maximum	17.18	11.99	306	8.64
	Count	34	34	34	34

Figure 2-1. Subestuaries sampled for Yellow Perch larval presence-absence studies, 2006-2023. Watersheds of subestuaries sampled during 2006-2022 are indicated by labels and those sampled in 2023 are highlighted in green and have bolded text labels; watershed delineation is for Maryland only.

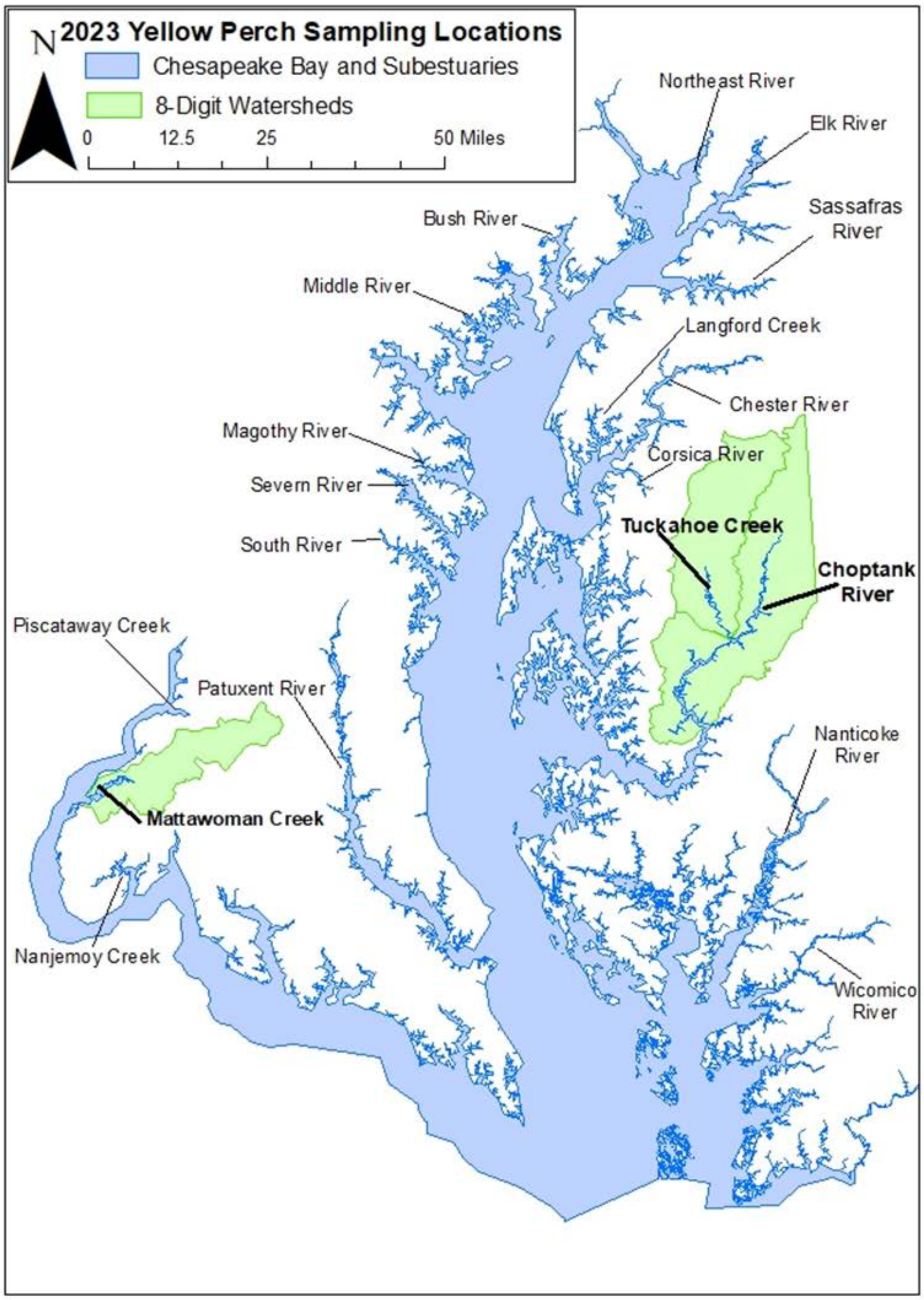


Figure 2.2. Location of Choptank River stations sampled for larval Yellow Perch presence-absence.

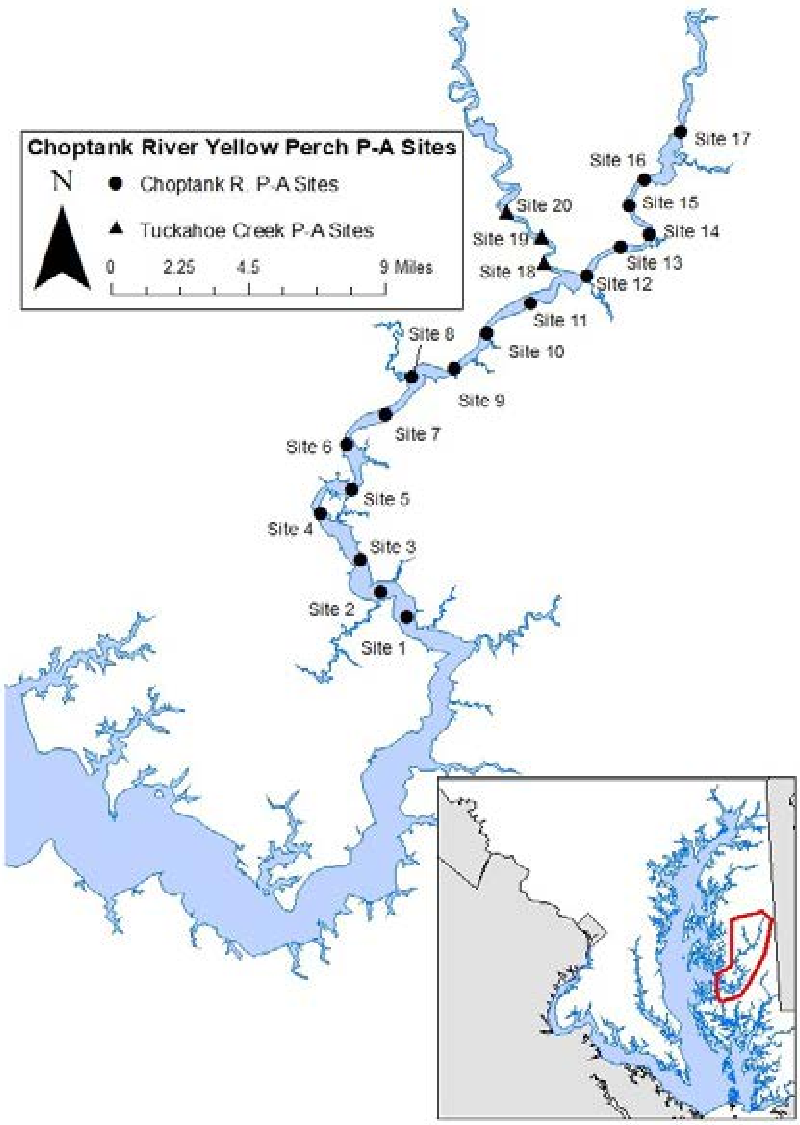


Figure 2.3. Location of sampling stations for larval Yellow Perch presence-absence in upper Mattawoman Creek. Stations 11 and 12 were added in 2023 to replace loss of sites 1 and 2 due to shallowing.

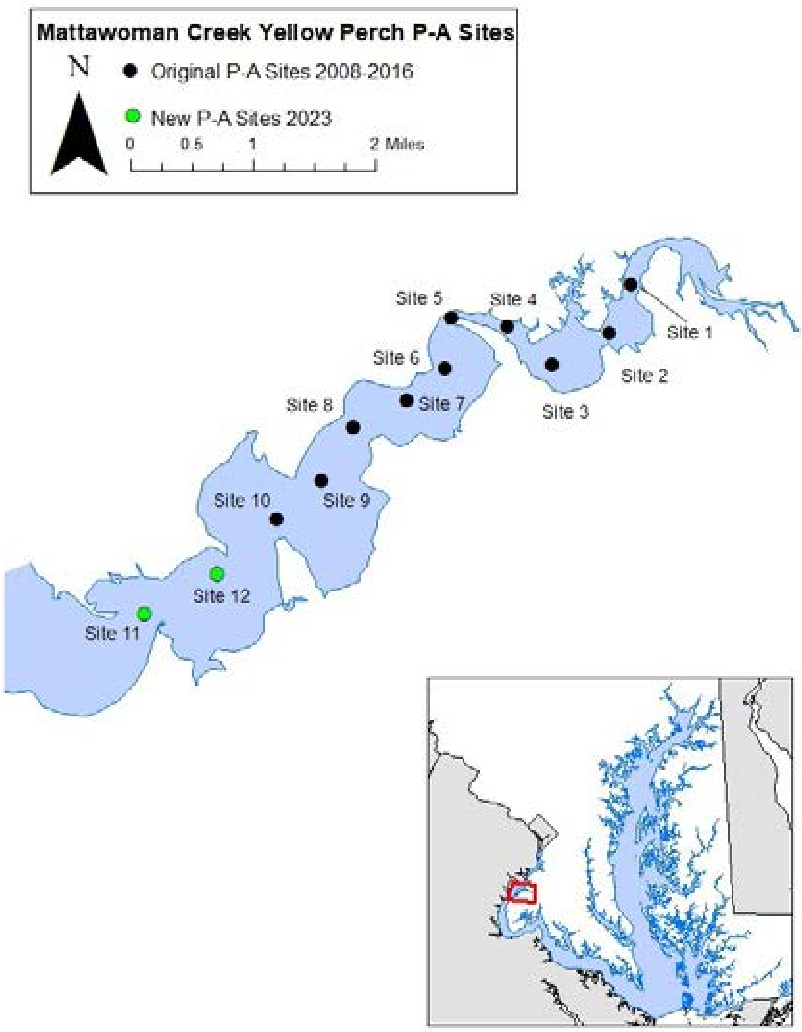


Figure 2-4. Proportion of tows with larval Yellow Perch (*Lp*) and their 95% confidence intervals in systems studied from 2006 -2023. Mean *Lp* of (A) brackish, and (B) tidal-fresh systems, are indicated by green triangles and blue circles, respectively. Brackish subestuary *Lp* threshold is indicated by a green dotted line, and tidal -fresh threshold is a blue dotted line.

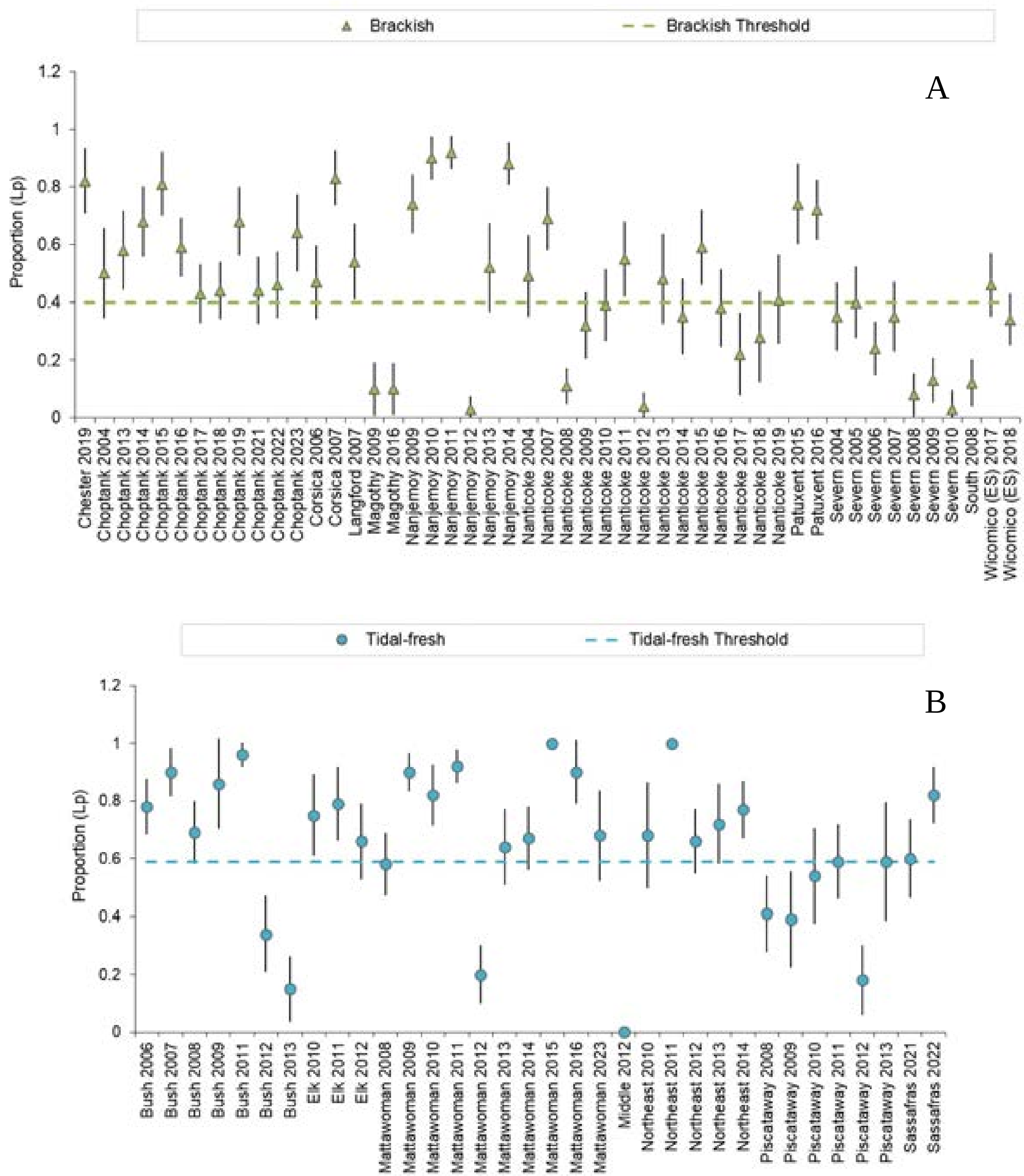


Figure 2-5. Proportion of tows with Yellow Perch larvae (Lp) for brackish subestuaries, during 1963-2023. Dotted line provides threshold for persistent poor Lp exhibited in developed brackish subestuaries. Dominant land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

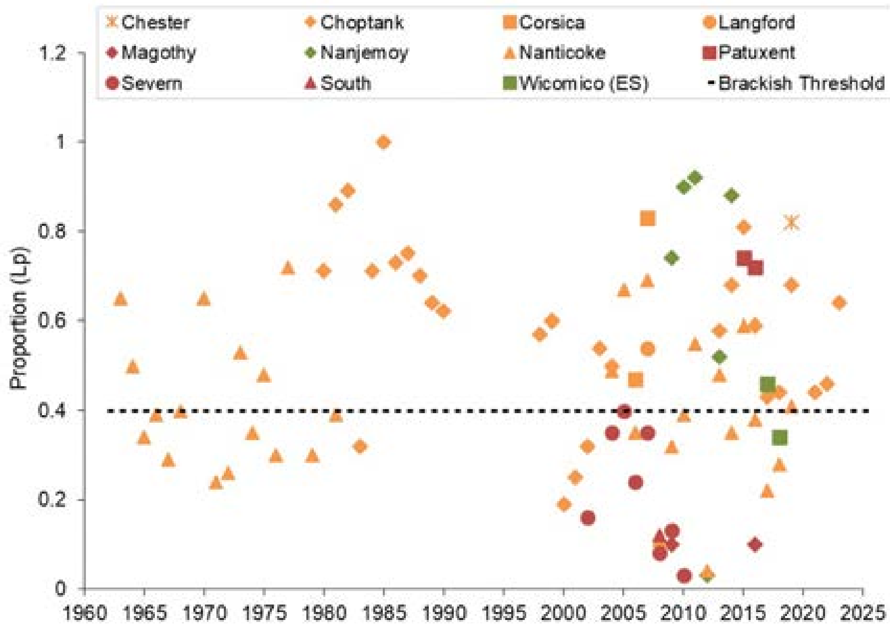


Figure 2-6. Proportion of tows with Yellow Perch larvae (Lp) for tidal-fresh subestuaries, during 1990-2023. Dotted line provides reference for consistent poor Lp exhibited in a more developed tidal-fresh subestuary (Piscataway Creek). Dominant Department of Planning land use is indicated by symbol color (gold = agriculture, green = forest, and red = urban).

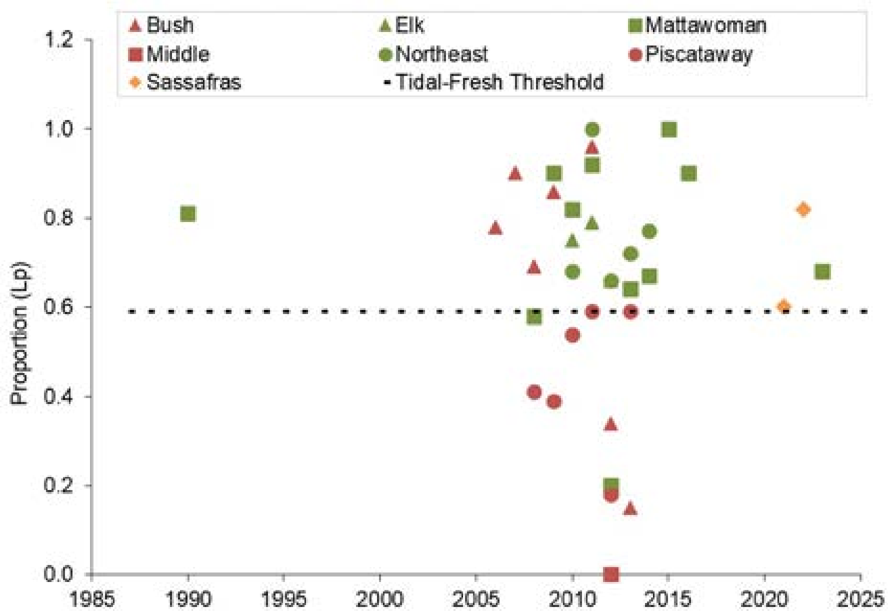
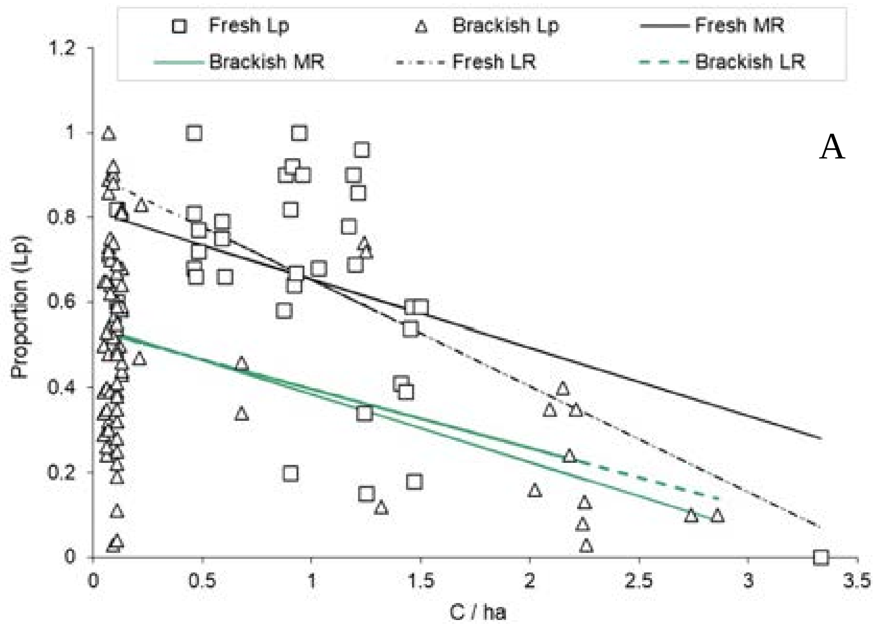


Figure 2 -7. Relationship of proportion of plankton tows with Yellow Perch larvae (*Lp*) and (A) development (structures per hectare or C/ha), (B) percent agriculture, and (C) percent forest, indicated by multiple regression of fresh and brackish subestuaries combined (prediction = MR) and separate linear regressions for both (prediction = LR).



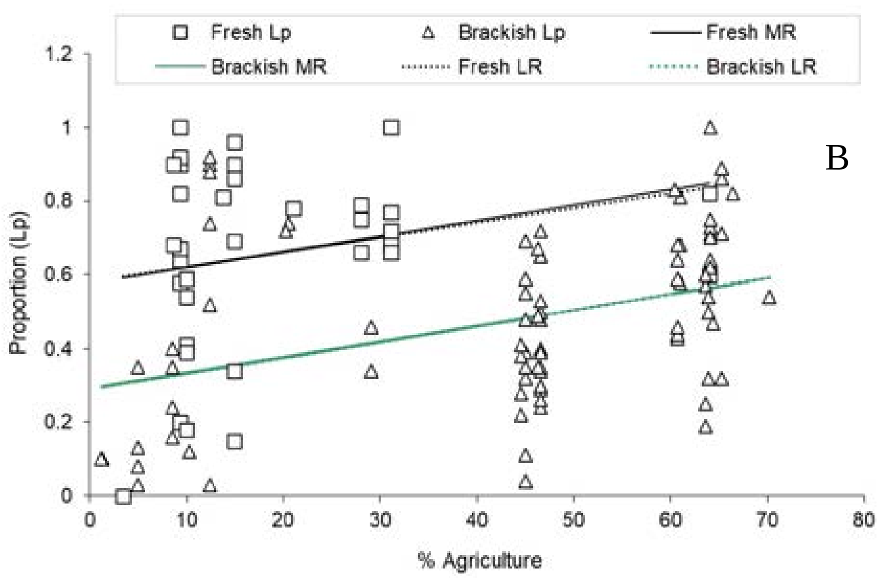


Figure 2-7 cont.

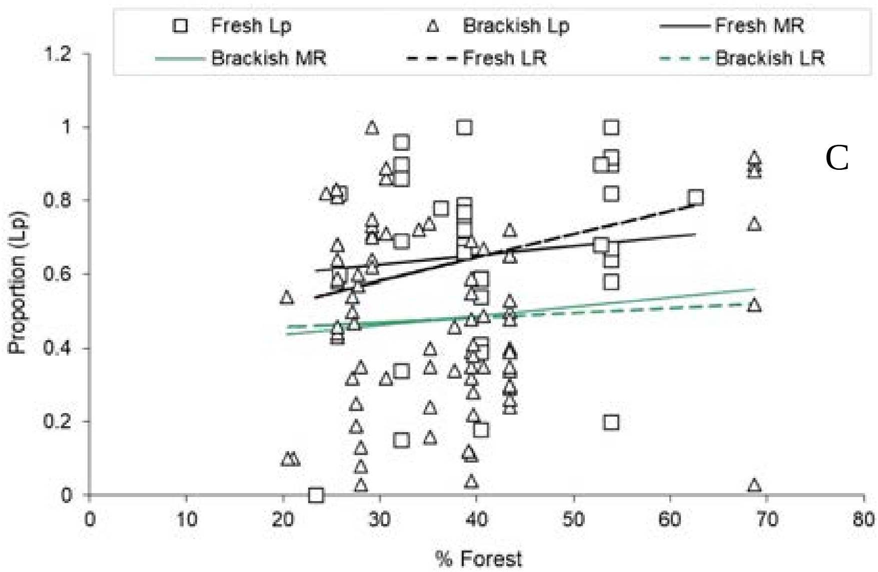
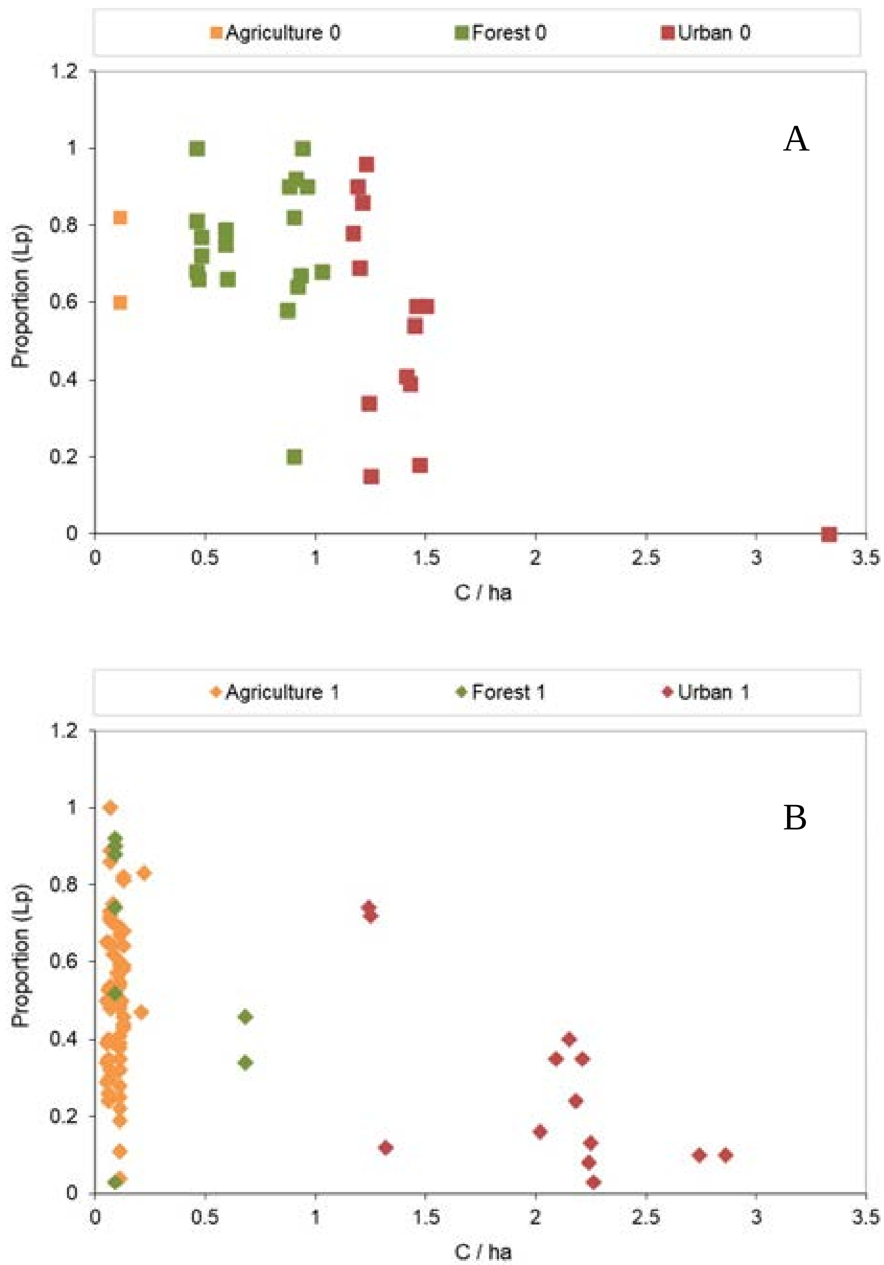


Figure 2-9. Proportion of plankton tows with Yellow Perch larvae (*Lp*) plotted against development (C/ha) with Department of Planning land use designations and salinity class indicated by symbols. Squares and a "0" behind land use indicate tidal-fresh subestuaries (A), while diamonds and a "1" indicate brackish subestuaries (B).



Objective 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, and Marek Topolski

Introduction

An overfishing declaration and successive poor year-classes of Striped Bass in Maryland spawning areas during 2019-2023 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 initiated in the last four annual reports (Uphoff et al. 2020; 2022a; 2022b; 2023) to assess spawning and larval habitat. Uphoff et al. (2020;2022b; 2023) 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; Uphoff 2023). 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; Uphoff 2023).

Water temperature and flow conditions are important influences on year-class success of Striped Bass (Maryland Sea Grant 2009; Uphoff 2023). 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 has been 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 (Kernehan 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). Recent analyses of spawning season temperatures on Chesapeake Bay spawning grounds have provided limited evidence of earlier spawning, but have confirmed that duration has shortened (Guiliano 2023; Uphoff et al 2022a; 2022b; 2023).

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 (Uphoff 2023). 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 limit dispersion (spatial and temporal distribution) and recruitment during 1982-1988 (Uphoff 2023).

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) since 1954 with a shore zone seine survey of young-of-year juveniles (Hollis et al. 1967; Durell and Weedon 2023) 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). Recent concerns about poor recruitment voiced in ASMFC technical and management meetings have focused on the Maryland JI because of its strong influence on the fishery (J. Uphoff, MD DNR, personal observation).
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 2023). 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 high productivity during 1958-1970 (Uphoff 2023). Year-class success has been low during 2019-2023 (Durell and Weedon 2023) and fell below an ASMFC (2003; 2010) criterion defining poor year-class success in 2023.

Uphoff (1993; 1997; 2023) used historical ichthyoplankton survey data to develop a Striped Bass egg presence-absence index (*Ep* or proportion of samples with eggs) of spawning dispersion during 1955-2022 for Maryland's spawning areas. An *Ep* time-series has been maintained, although it became a low priority in the 2000s as catch-at-age modeling became the primary stock assessment method (Uphoff 2023). 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. 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 2023).

Toxic water quality conditions encountered by Striped Bass larvae were implicated in episodic mortalities in some spawning areas (Choptank River, Nanticoke River, and 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 baywide recruitment rose. This river served as a rural reference system for our investigations of development's effect on Striped Bass egg and larval habitat and there were records of basic water quality conditions and egg-larval mortality during 1980-1991 for comparisons with current conditions (Uphoff 1989; 1992; Uphoff 2023; Uphoff et al. 2023). We added alkalinity to the suite of water quality variables sampled on the Choptank River spawning grounds in 2021. 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).

Carrie 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 (˚C) and salinity (‰), until the early 1980s. During the 1980s and after, dissolved oxygen (DO; mg/L), pH, and conductivity (µS/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 through 2023 for the two spawning areas with the most extensive time-series: the Choptank and Nanticoke rivers.

We examined four spawning milestones that were reasonably straightforward to interpret: date that the first egg was collected, and the dates when 12ºC, 16 ºC, and 20ºC were consistently met. Spawning in Chesapeake Bay rivers generally occurs between 12ºC and 23 ºC (Peer and Miller 2014), but temperatures above 21ºC 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 ºC (Uphoff 1989; Rutherford and Houde 1995; Rutherford et al. 1997; 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 ºC) and late (>21 ºC) cohorts experiencing higher mortality (Secor and Houde 1995; Peer and Miller 2014). We selected 20ºC as an upper temperature boundary since egg presence-absence surveys sometimes cut off sampling just prior to when 21ºC was anticipated to occur; 16ºC represented the midpoint of the range and was a temperature where larval cohort survival was expected to be high based on 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).

We updated the following metrics developed in Uphoff et al. (2020; 2022b) through 2023 in this report: *Ep*, JI, RLS. Temperature, DO, pH, salinity, conductivity, and alkalinity comparisons were updated. We updated the occurrence of spawning temperature 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 2.1.1; 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 2.1.1).

Proportion of ichthyoplankton tows with Striped Bass eggs (Ep) 2023 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 2023). 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 2023) 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).

During 2023, the Choptank River spawning area was divided into 19 1.61-km segments, starting at km 47.2 and proceeding upstream (Figure 2.1.2). We could not access two of the furthest upstream historic stations (stations 17 and 21) sampled during 1987-1990 because of shallow depths in the last several years. Three segments, 18-20, were in Tuckahoe Creek (starting at the mouth). Segments were aggregated into four subareas. The lower Choptank area consisted of the first 5 segments; the middle, segments 6-11; the upper, segments 12-16; and Tuckahoe Creek, segments 18-20. Barring unsuitable weather and equipment issues, 10 stations were visited during a sampling day.

Surveys prior to 1994 varied in tow durations, net configuration, and mesh sizes (Uphoff 1993; 1997; Uphoff 2023). Surveys to estimate *Ep* during 1994-2023 were standardized to techniques of the longest running early time-series (Nanticoke River, 1955-1981). 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 (Uphoff 2023).

Sample trips during 1994-2023 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 2023) or was very likely to be met before the next scheduled sampling visit based on water temperature and forecast air temperatures. In a few years, 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 2023). 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 (Uphoff 2023). 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. Stations where eggs were not collected located between stations where eggs were present were included in analyses (Uphoff 2023).

We also assigned a qualitative summary of egg density to a day's sampling to better discriminate spawning intensity: very low, light, modest, and high egg density. Very low density was indicated when eggs were not readily detectable in samples; they were found by letting the sample settle or while sorting and a defined layer was absent. Light density described when most samples had very few eggs, but they were abundant enough to be seen without sorting or allowing samples to settle. Modest densities were indicated by a layer of eggs in multiple sample jars that occupied up to a half a sample jar. High density was indicated when half or more of multiple sample jars were filled with eggs.

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 (Uphoff 2023). 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 2023). 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;2023b). 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 (Uphoff 2023).

Juvenile index 2023 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 2023). 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 2023). 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 2023).

Relative Larval Survival (RLS) update - We used the JI and baywide *Ep* to estimate annual relative larval survival (RLS) during 1957-2023 as:

(4) RLS = JI / baywide *Ep* (Uphoff 2023).

Estimates of the JI concurrent with Ep were available for 1957-2023 (Durell and Weedon 2023). The baywide *Ep* time-series started in 1955 and continued through 2023; 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; Uphoff 2023). 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 2023).

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). Uphoff (2023) used the same series of years to develop criteria for low and high RLS and we adopted these criteria. The lowest quartile of RLS during 1957-2009 was the criterion for poor egg-larval survival. Conversely, Uphoff (2023) 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 (Uphoff 2023).

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 2023. Measurements of water temperature (ºC), pH, dissolved oxygen (mg/L), conductivity (µS/cm), and salinity (‰) were made at the surface during each site visit with a YSI model 556 water quality multimeter during 2014-2023 (Uphoff 2023). The meter was calibrated frequently using YSI protocols. 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 (Uphoff 2023).

During 2021-2023, 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 the 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 $\leq$ 2.0 ‰). Means and medians would provide some indication of chronic conditions, while maximums and minimums would capture acute conditions (Uphoff 2023). 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 (Uphoff 2023).

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 12ºC, 16 ºC, and 20ºC 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 early in the spawning season were sometimes rejected since these areas warm quickly before detectable spawning activity. Dates indicating when the first egg was detected or 12ºC or 20ºC 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 2021-2023. 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 in the Coastal Plain of Maryland's eastern shore. We estimated the median date for a milestone for each year through 1999 and then examined the frequency that dates exceeded or fell below the median after 1999.

Flow – We updated the standardized flows developed in Uphoff et al. (2022b) through 2023. 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) 2023 update – Sample size was sufficient for estimating *Ep* in the Choptank River (N = 46) during 2023; Nanticoke River was not sampled in 2023. Samples used to estimate *Ep* began on March 30 and ended on April 12. The temperature cutoff was reached on April 17 and we sampled through April 26 because it was reached much earlier than expected.

The estimate of *Ep* in Choptank River during 2023 was 0.74 with a 90% CI of 0.61-0.87 (SD = 0.06; Figure 2.1.3) and this estimate served as the baywide *Ep* estimate (Appendix; Figure 2.1.4) as well. Estimated *Ep* in 2023 was within the range of baywide values exhibited in the 1960s-1970s and since 1989. 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. 2023b). There was a 1.7% chance that *Ep* was low enough that spawning dispersion impacted recruitment (*Ep* less than 0.60); *Ep* was consistently below 0.60 during 1982-1988 when low juvenile indices reflected effects of both low spawning stock and poor habitat (Uphoff 2023). There was a 33.9% chance that *Ep* was above the median since *Ep* recovered (0.77; 1989-2019 baseline). The 2023 sample size (N = 46) was low for the baywide series and ranked 61st out of 65; this low N_{total} reflected rapid warming rather than lower than normal sampling effort (see below). When all dates were included, *Ep* was 0.80 (SD = 0.042; N_{total} = 86).

Based on 2023 observations, high egg density was noted on the April 7 survey; modest densities on April 4, April 10, and April 12; light density on April 17; and very low density on April 24 and April 26. Temperatures dips were not detected in the Choptank River in 2023 (Figure 2.1.10). A chronology of spawning by date would be as follows: low intensity spawning began on March 30 (first egg collected); spawning intensity increased through April 7 (peak spawning); intensity moderated through the 12th; and nearly ceased after the 17th.

The spawning area was smaller than normal in 2023, likely reflecting low winter-spring precipitation and flow. Salinity at lower sites 1-6 was always above our 2‰ sampling cutoff. Stations 7-9 were above 2‰ on some days. Eggs were present at stations 7-12 on every date sampled (station 12 is at the intersection of Choptank River and Tuckahoe Creek) and on most dates up to station 15. In Tuckahoe Creek, 4 of 5 samples upstream of the mouth of Tuckahoe Creek (station 18) had eggs present as did 2 of 5 samples at station 19. Spawning has been detected at all sites (1-21) over the years, although spawning at uppermost Choptank River and Tuckahoe Creek sites (stations 13-16 and 19-21) has been less consistent. Sites 17 and 21 are no longer accessible due to channel shoaling.

Juvenile index 2023 update – The Baywide JI was 0.57 in 2023 (Figure 2.1.5; Durell and Weedon 2023). This was the second lowest in the time-series (Durell and Weedon 2023). *Relative larval survival 2023 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 0.78 in 2023 (Figure 2.1.6). The simulated mean was 0.79 and the SD was 0.13. The probability of falling below the poor RLS criterion in 2023 was 1.0 and the probability that survival was above the high RLS criterion was zero.

With the exception of 1982-1988, deviations between standardized RLS and standardized JIs during 1957-2023 fell between -0.21 and 0.23 (hereafter, the normal range; Figure 2.1.7). During 1982-1988, larger negative deviations occurred, -0.38 to -1.12; these larger negative deviations were interpreted as an indication of the effect of low spawning stock. The deviation for 2022, -0.03, was within the normal range (Figure 2.1.7).

Water quality update - During 2023, median pH during April 1-May 8 (a standard period across years) in Choptank River was 7.27 and measurements ranged between 7.01 and 7.63 (Table 2.2.1; Figure 2.1.8). This continued the pattern of above neutral median pH measurements since pH measurements became routine in 2014. Medians during 2014-2022 ranged from 7.03-7.42, 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 to 6.50, and maximum pH measurements were between 6.46 and 9.15 (Table 2.1.1; Figure 2.1.8).

Standard period total alkalinity measurements in Choptank River during 2021-2023 were much higher than during 1986-1991 (Table 2.1.1; Figure 2.1.8). Median total alkalinity was 82.5 mg/L and ranged between 60 and 115 mg/L during 2023. 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 indicated a trend of stable low measurements (5th percentile) and an increase in higher measurements (95th percentile) in the latter two years (Table 2.1.1; Figure 2.1.8). Alkalinity measurements began earlier in 2021 (March 30) and 2023 (April 4) than 2022 (April 20), so 2022 measurements may have missed the period prior to application of lime on agricultural fields.

We could not discern potential patterns in conductivity summary statistics from the Choptank River spawning area during the standard period that would suggest differences between 1986-1991 and 2014-2023 (Table 2.1.1). Standard period median, minimum, and maximum conductivity measurements during 2023 were 1,088, 198, and 4,418 µS/cm^2, respectively. These 2023 summary statistics and the 5th and 95th percentiles were the highest or second highest estimated during 1986-1991 and 2014-2023 (Table 2.1.1).

Water temperature –The first egg was collected on March 30, 2023. This was the third earliest date that spawning has been detected in Choptank River ichthyoplankton surveys (Figure 2.1.9). It was also the second earliest date that 12ºC was reached (Figure 2.1.11). These two early spawning milestones seem to be occurring sooner more often since 2017. The mid-milestone, 16ºC, was reached between April 4 and April 7; temperatures rose from around 13.5-13.9 ºC to 17.0-18.6ºC during this interval. The 12-16°C span was breached between April 5 and 6 and lasted no more than 8 days. This milestone has been reached earlier and more consistently since 2017 (Figure 2.1.12). The 20ºC milestone was reached on April 17 and 2023 is tied with 2002 for the earliest this milestone was reached in Choptank River (Figure 2.1.13).
Since 2000, 5 surveys have detected the first egg at a date the same as or later than the median date for 1954-1999 and 8 have detected it earlier; all have been earlier since 2017. Five surveys since 2000 have detected the 12˚C milestone the same date or earlier than surveys during 1954-1999 and 12 have been earlier. The 16˚C milestone was detected on the same date or later 7 times and earlier 20 times and the 20˚C milestone was on the same date or later 3 times and earlier on 11.

It appears that the 2023 spawning season, based on the dates that 12°C and 21°C were reached, ran less than 19 days, (Figure 2.1.14). This was a short, temperature driven spawning season. There are other short seasons earlier in the time-series, but they appear to have become more common since the early 2000s. In the Choptank River during 1954-1987, there were 2 of 7 years with complete sets of temperature milestones present that had spawning seasons 20 days or less. Two of the 19 years had 20 day or less spawning seasons in the Nanticoke River during 1954-1992. Since 2001, 5 of 10 surveys in Choptank River had spawning seasons of 20 days or less (Figure 2.1.14).

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 below average baseline flow of 1957-2020 during 2023 in Choptank River (0.67), Nanticoke River (0.41), Head-of-Bay (0.66), and Potomac River (0.46; Figure 2.1.15).

Discussion

The estimate of baywide *Ep* (0.74) for 2023 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 1993-2023 when *Ep* was within the top tier (1993, 1996, 2003, and 2011), while 5 were present when *Ep* was at a similar level to 2023 (2000, 2001, 2005, 2015, and 2018). Estimated RLS in 2023 was below 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. Five consecutive years of low baywide JIs have occurred since 2019 and presumably 5 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). Five consecutive years of low year-class success is worrisome and will impact the fishery in the future.

Comparisons of pH and alkalinity in Choptank River between 1986-1991 and 2013-2023 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 2023). Average pH was generally lower during 1986-1991 and more variable in half the years available than during 2014-2023 in Choptank River. Average alkalinity was at least 3-times higher in 2021-2023 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 across the U.S. as well (Uphoff 2023). While recent measurements of metals are unavailable, it seems unlikely that poor survival of larvae during 2019-2023 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 9 surveys during 2014-2023. Disparities between time periods suggest change in underlying dynamics.

There were considerable differences in total alkalinity measurements in Choptank River during 1986-1991 and 2021-2023. Alkalinity during 1986-1991 was measured by titration and with a photometer during 2021-2023. Measurements during 2021-2023 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. We do not intend on measuring alkalinity in 2024 and beyond since three years of monitoring indicated it is stable. A correlation analysis indicated that minimum pH tracked all of the alkalinity summary statistics (r range = 0.72-0.80). However (and as an example), while the correlation with mean alkalinity bordered on strong ($r = 0.77$, $P = 0.015$), the bivariate plot (not shown) suggested a threshold response. Any minimum pH during 1986-1991 (5.8-6.5) reflected mean alkalinity less than 25 mg/L and minimum pH during 2021-2023 (6.6-7.0) reflected mean alkalinity between 67 and 84 mg/L. A minimum pH less than 6.5 could be low enough to consider resumption of alkalinity monitoring.

Means or medians of days between 12ºC and 20ºC 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 have changed the least. 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 12ºC in 2000-2021 (about 2-3 days) was not fully supported. As the milestones progressed in magnitude, average dates of occurrence shortened between 1954-1992

and 2000-2021 (7 days earlier at 16ºC and 10 days for 20ºC). The analysis of milestones in Uphoff et al. (2022b) has been updated through 2023 and the conclusions have not changed. The portion of the spawning period when most eggs were historically collected (days from 12ºC to 16ºC) has shortened and high temperatures (indicated by days to 20ºC) were being reached earlier. In addition to these general changes, 3 years during 2000-2023 (of 10 available) had very short spans between 12ºC and 16ºC (2 days), another had a span of 5 days, and 2021 had the earliest date that 12ºC was reached in the entire time-series. During 1954-1992, the transition from 12ºC to 16ºC took a week or less with 5 of 19 Nanticoke River surveys and 2 of 7 Choptank River surveys (Uphoff et al. 2022b); after 2000, 4 of 10 Choptank River surveys exhibited a transition of a week or less. The transition from 16ºC to 20ºC 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, 4 of 10 Choptank River surveys made this transition in a week or less.
Water temperature and spawning changes were similar to expectations described by MD Sea Grant (2009) for Chesapeake Bay and Nack et al. (2019) for the Hudson River. 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 12ºC and 16 ºC 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 14ºC and 17ºC, followed by a lesser, but steady, increase to 20ºC. 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; 2022b) 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 1ºC 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 (Guiliano 2023).

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). Inclusion of upper sites where early spawning was not likely could have negatively biased dates when 12ºC 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.

Temperature 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 most 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 20ºC 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.

Spawning area standardized flows appear to have shifted downward after 2011; above average flows have been lower during 2012-2023 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). Poor year-class success of Chesapeake Bay Striped Bass was highly likely when flows were below average (Gross et al. 2022; 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 (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.

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 (µS/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
2023	7.23	7.24	7.50	7.04	7.01	7.63	90
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
2023	1348	1,088	3309	270	198	4418	90

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
2023	83.9	82.5	110	60	60	115	48

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	12916	237	97
2023	41,566	10,101	163	62
1957-2020 Average	62,616	22,128	242	143

Figure 2.1.1. 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.

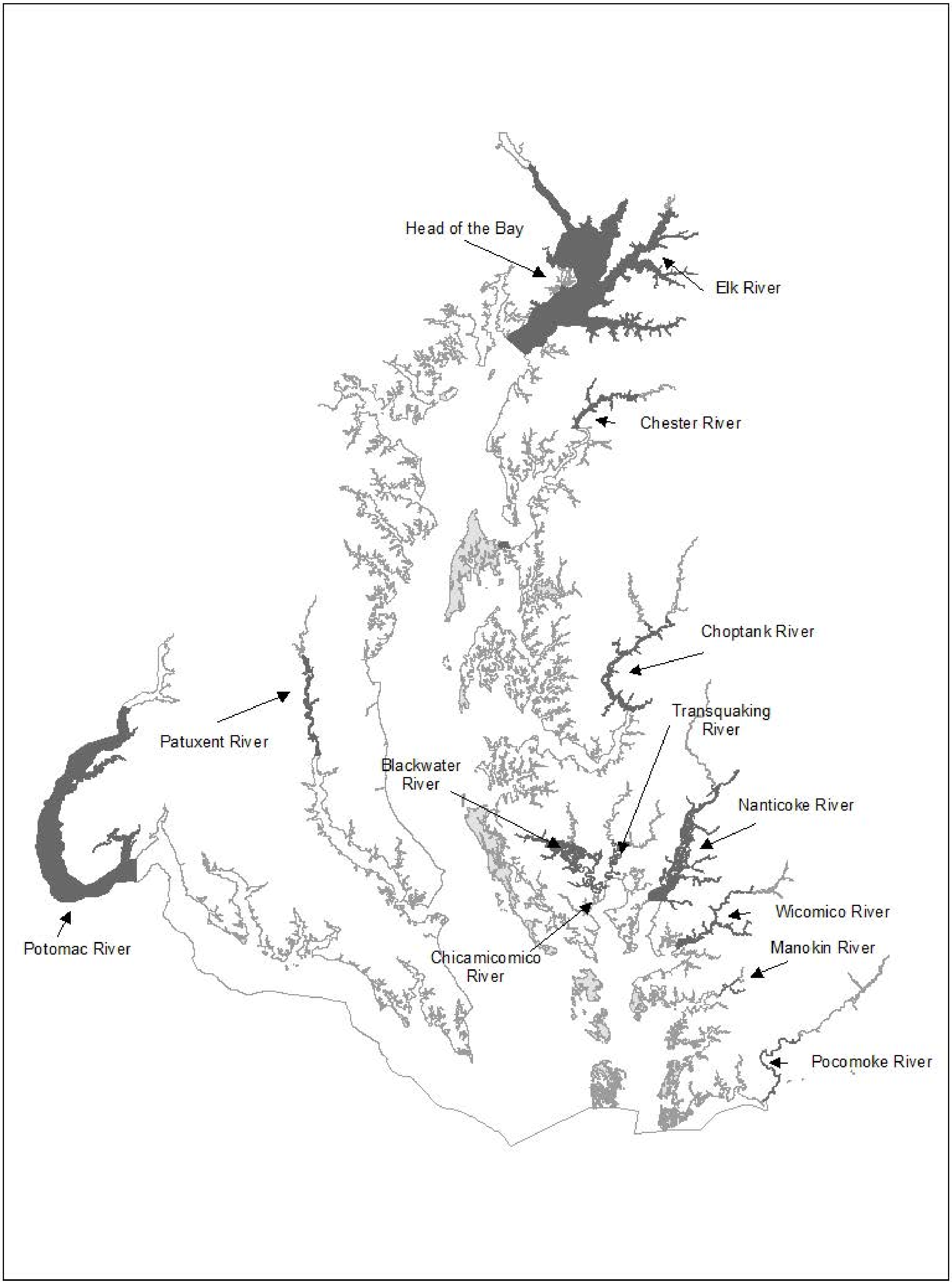


Figure 2.1.2. Location of historic and sites sampled in Choptank River during 2023 and mainstem sites or within Tuckahoe Creek (triangles). Stations 17 and 21 were not sampled in 2023 because access was too shallow. Inset shows location of Choptank River within Chesapeake Bay.

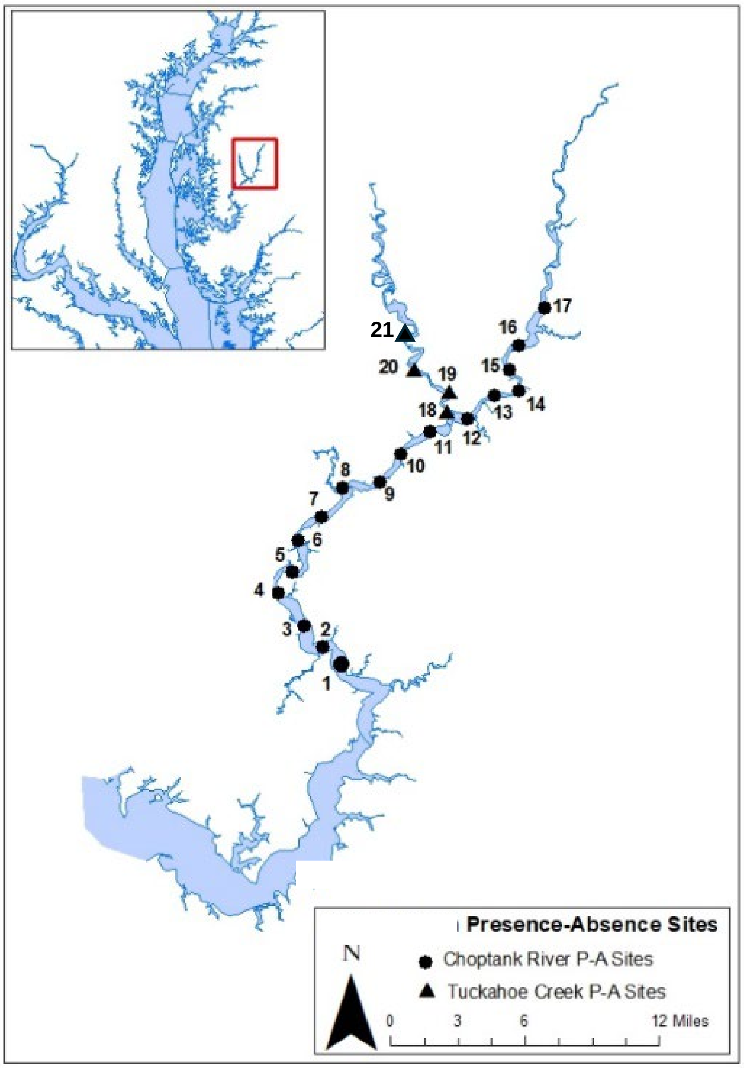


Figure 2.1.3. Spawning area specific proportion of tows with Striped Bass eggs (*Ep*) estimated from surveys in juvenile index rivers conducted during 1955-2023. Elk River represents a portion of the Head-of-Bay.

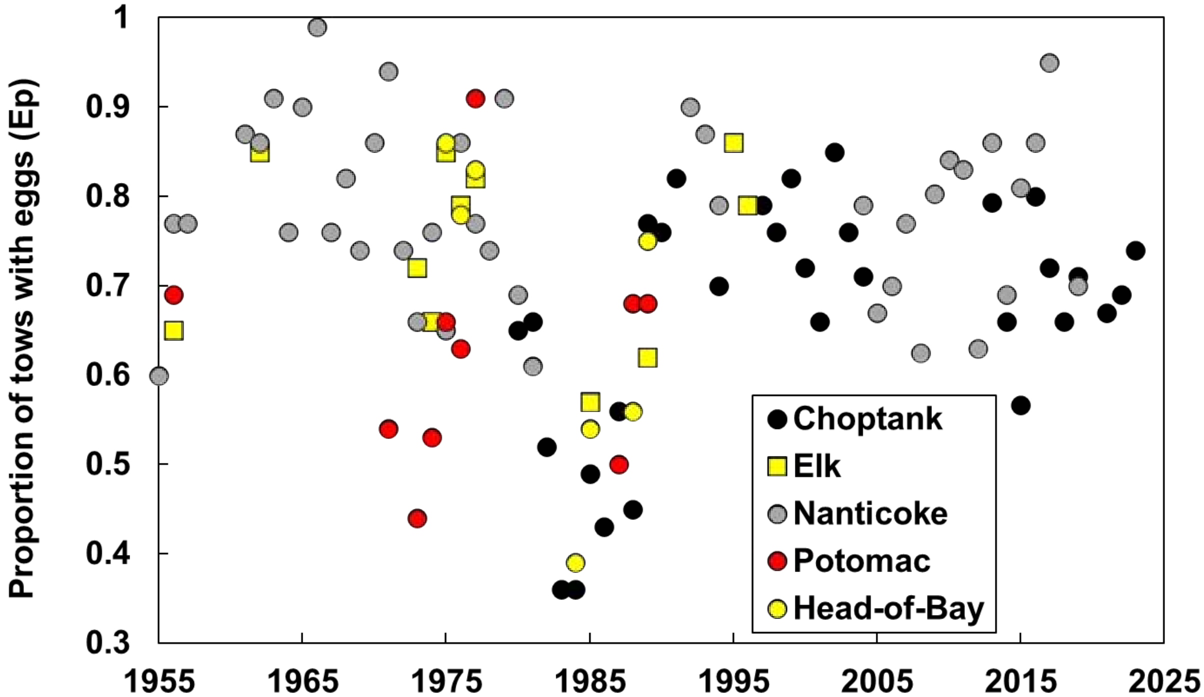


Figure 2.1.4. Baywide (Maryland's spawning areas) proportion of tows with Striped Bass eggs (*Ep;* diamond) and its 90% CI (vertical line) estimated from surveys in juvenile index rivers conducted during 1955-2023. Baywide estimate pools available data from spawning surveys conducted in four areas surveyed for the juvenile index: Head-of-Bay, Potomac River, Nanticoke River, and Choptank River.

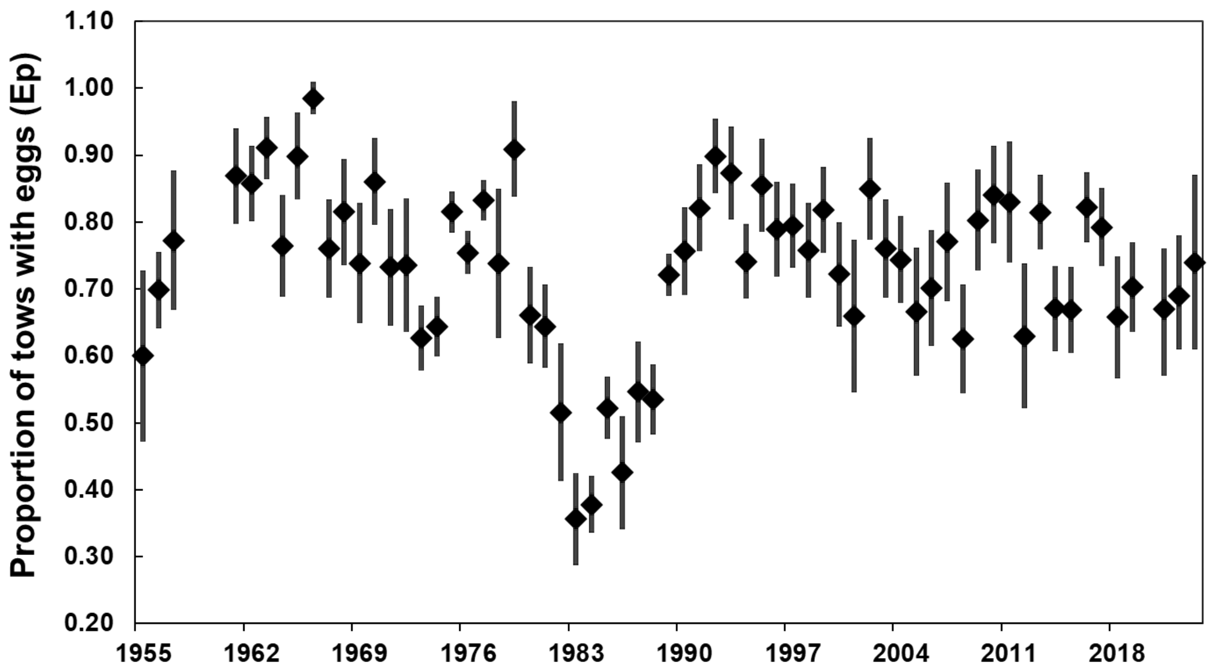


Figure 2.1.5. Baywide Striped Bass juvenile indices (geometric mean catch per standard seine haul; diamonds) and their 90% confidence interval (vertical line) estimated for Maryland's major spawning areas during 1957-2023 (Durell and Weedon 2023).

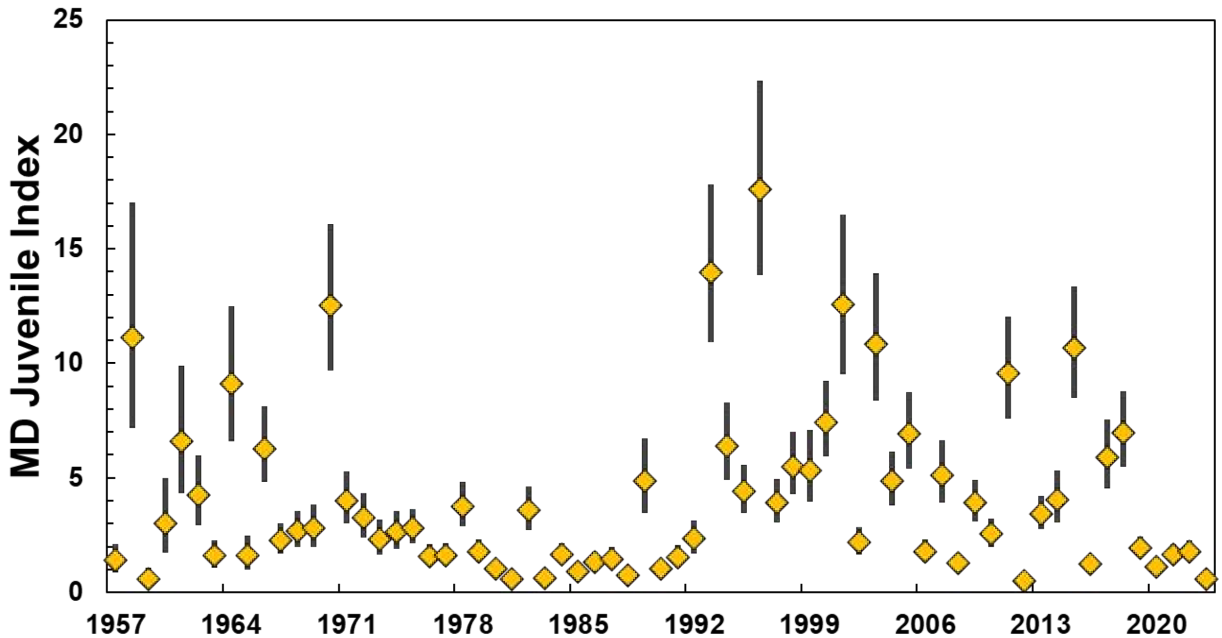


Figure 2.1.6. Relative larval survival (baywide JI / baywide *Ep*) mean and 90% CIs, 1957-2023.

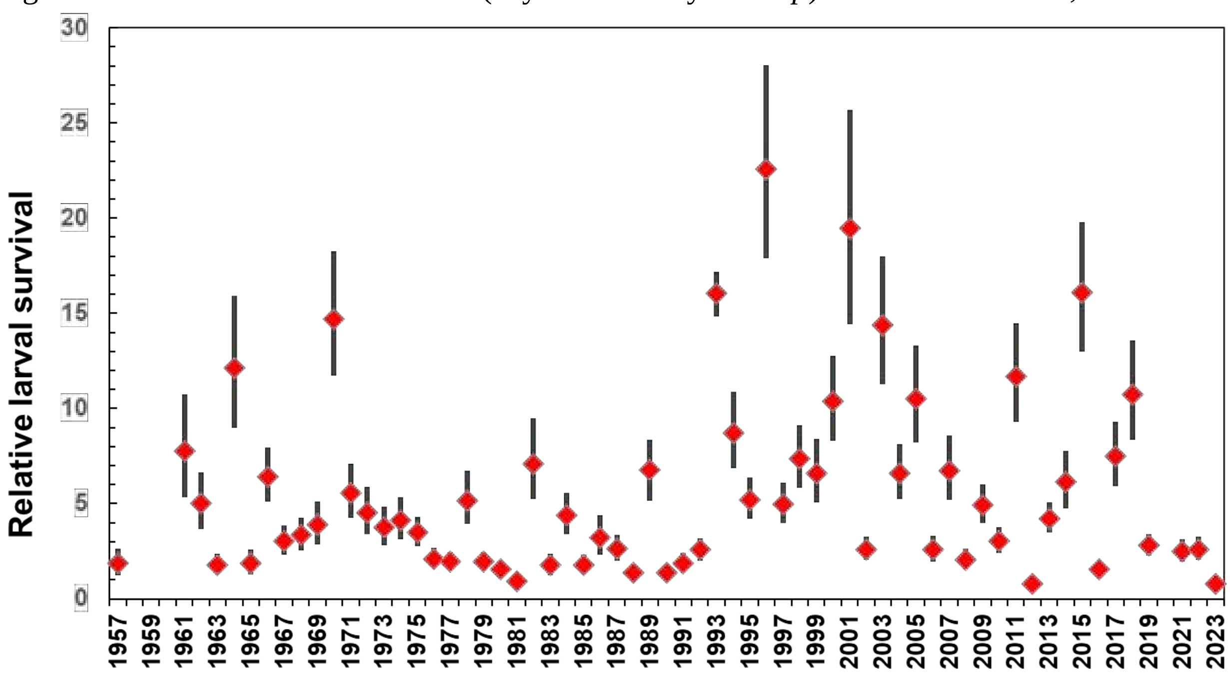


Figure 2.1.7. Difference of standardized juvenile index (Std JI) and standardized relative larval survival (Std RLS) as proportion of standardized JI during 1957-2023. Large negative deviations indicate overfishing in 1982-1988. Indices standardized to mean of common years (same scale).

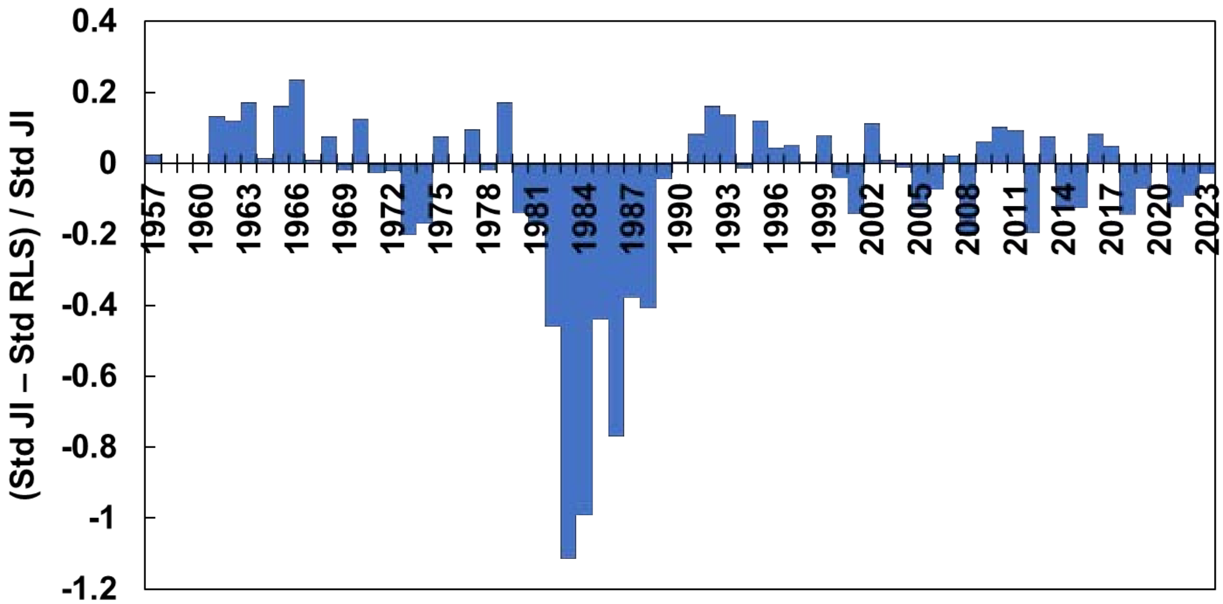


Figure 2.1.8. Choptank pH and alkalinity mean and range during April 1 – May 7, 1986-1991 and 2014-2023; 2022 alkalinity estimates did not span the full period and began on April 20.

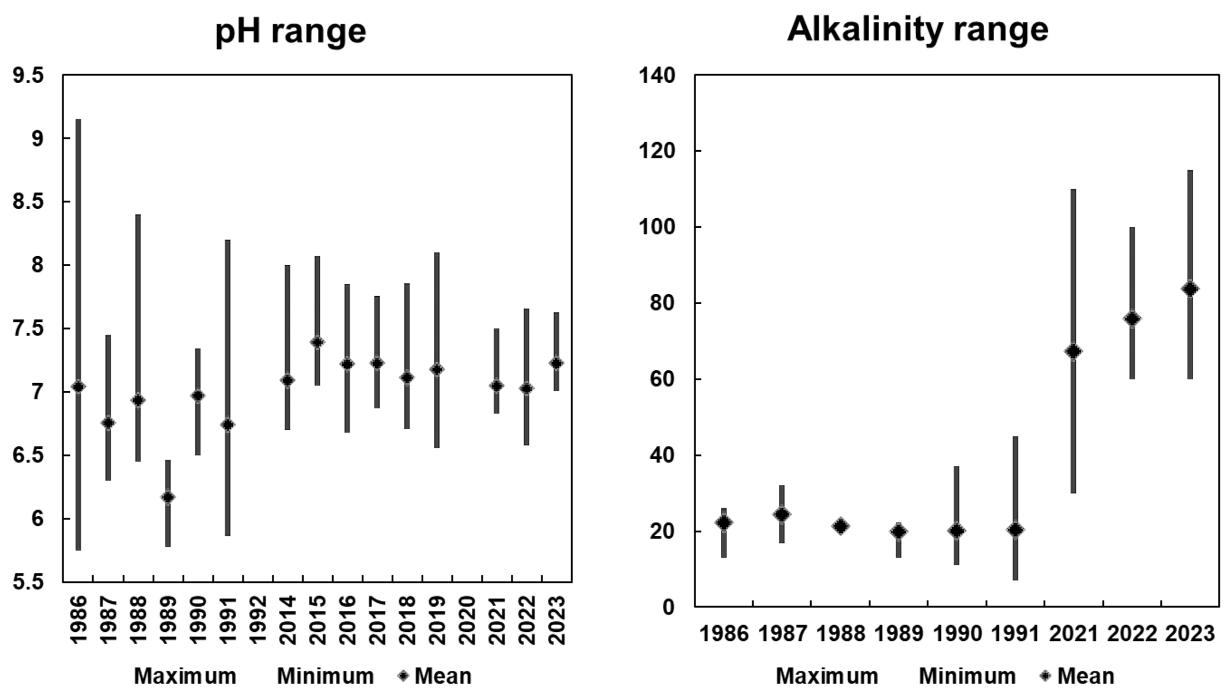


Figure 2.1.9. Days from April 1 (day = 0) that the first egg was collected in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2023. Median = median day for both rivers combined (day 7) during 1954-1999.

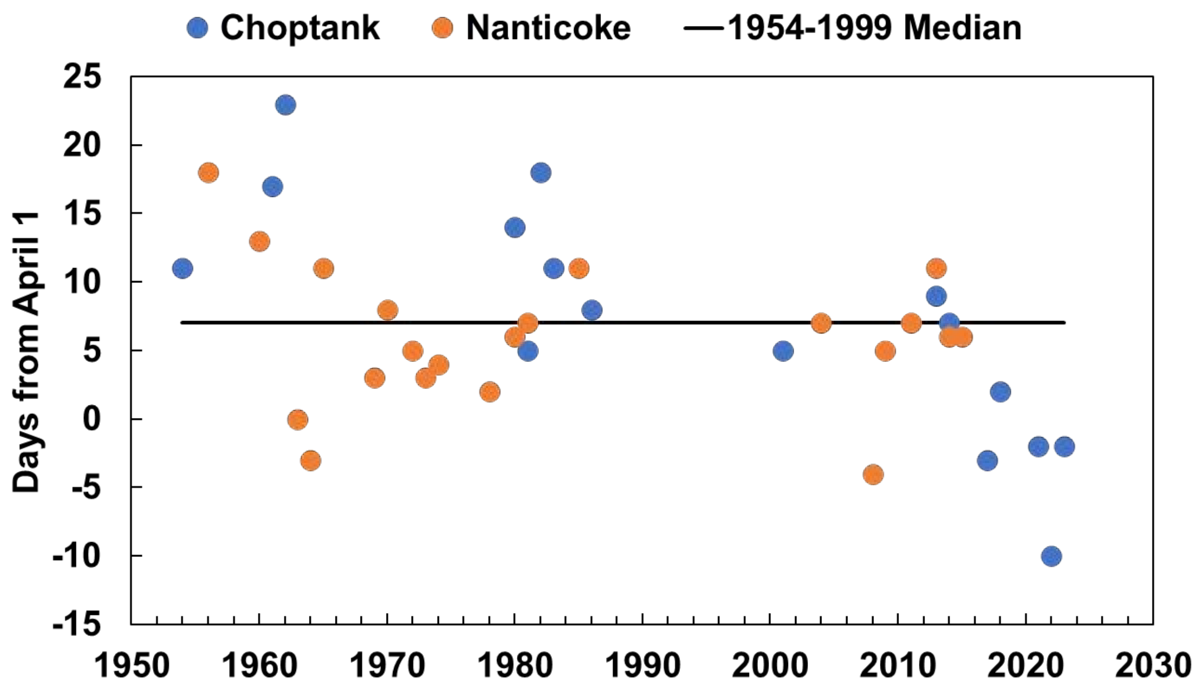


Figure 2.1.10. Water temperature (°C) measurements during the 2023 Choptank River Striped Bass egg survey.

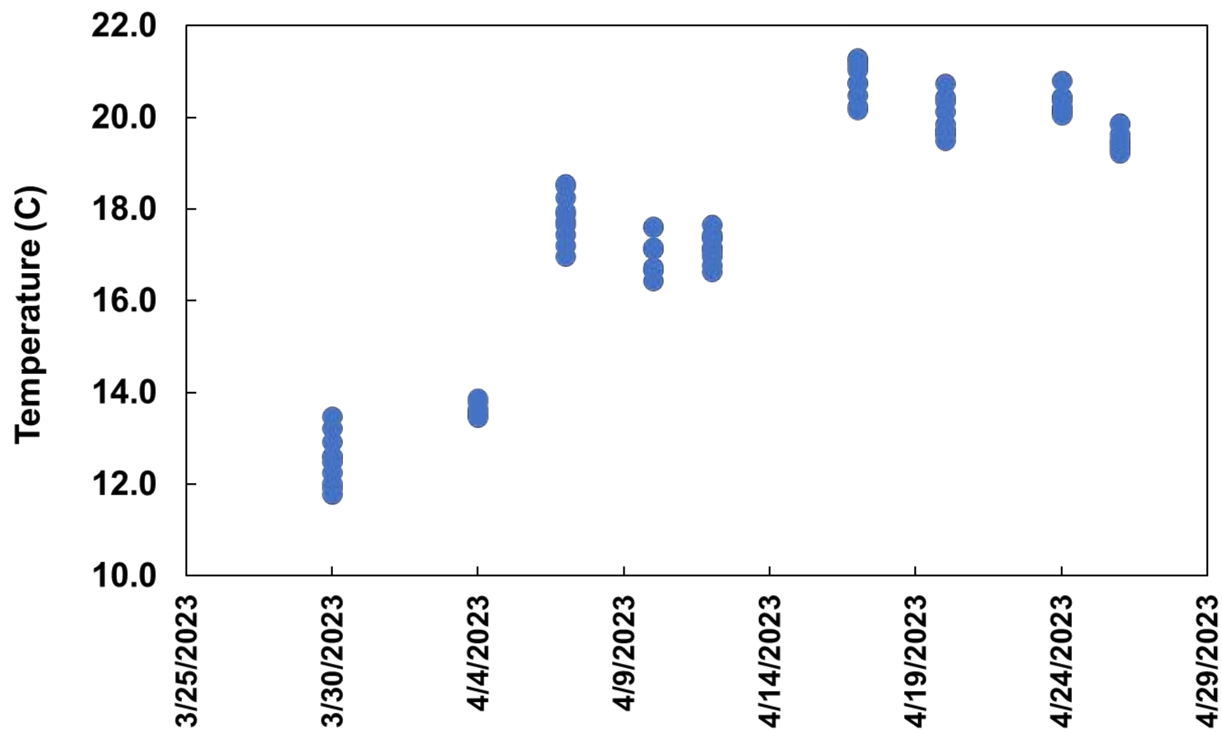


Figure 2.1.11. Days from April 1 (day = 0) that 12ºC was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-1999. Median = median day for both rivers combined (day 11).

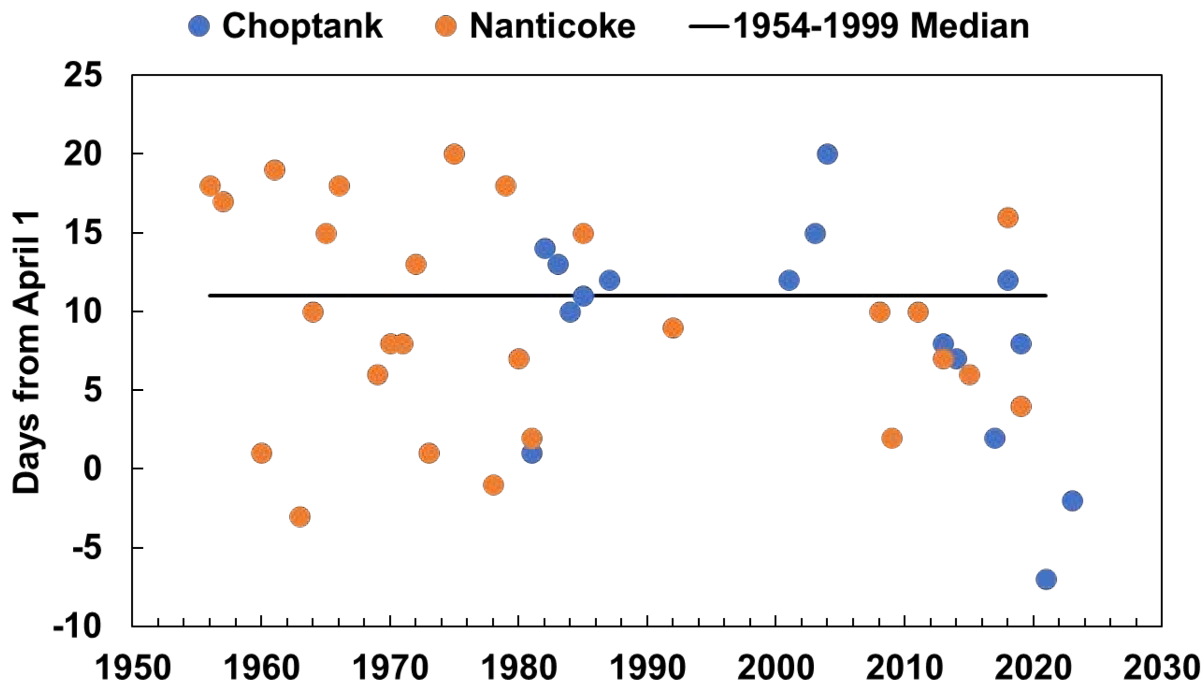


Figure 2.1.12. Days from April 1 (day = 0) that 16ºC was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2023. Median = median day for both rivers combined during 1954-1999.

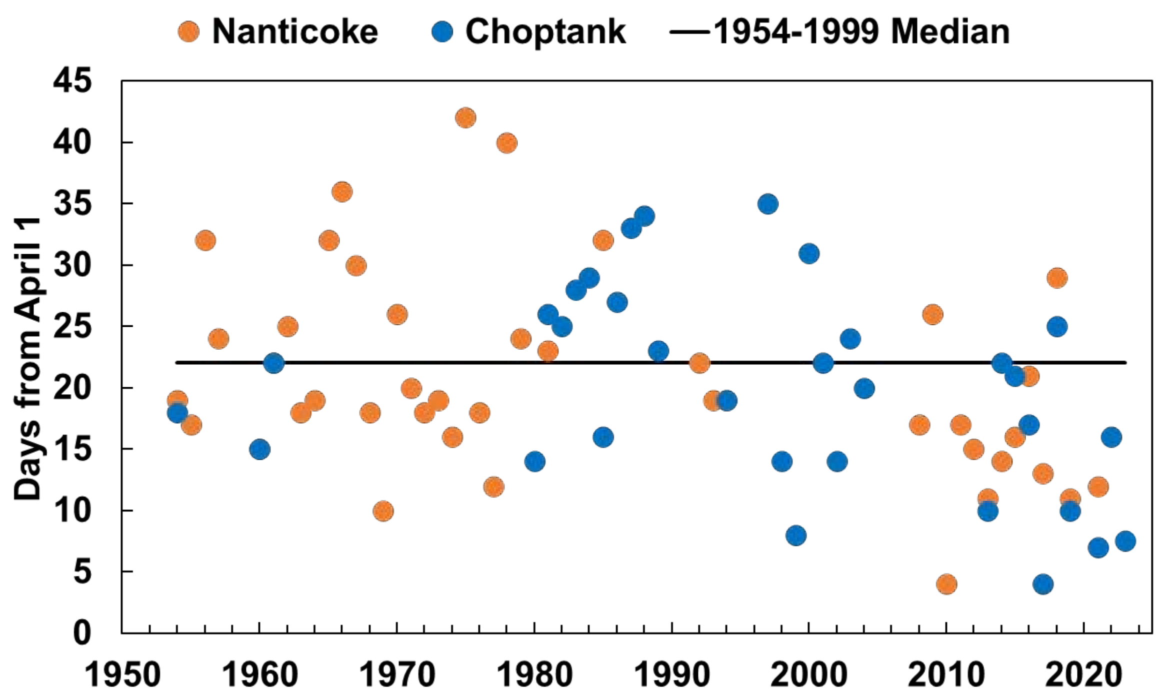


Figure 2.1.13. Days from April 1 (day = 0) that 20ºC was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2023. Median = median day for both rivers combined (day 41) during 1954-1999.

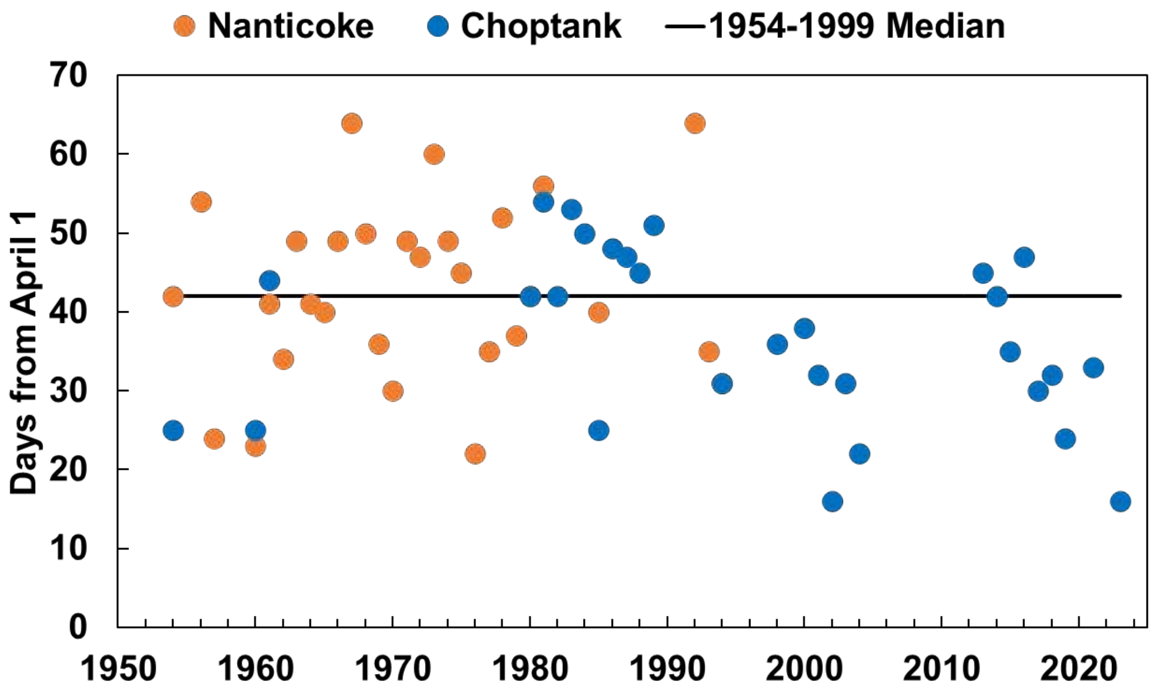


Figure 2.1.14. Days from April 1 (day = 0) that 12ºC, 16ºC, and 20ºC were reached in the Choptank River during 1954-2023.

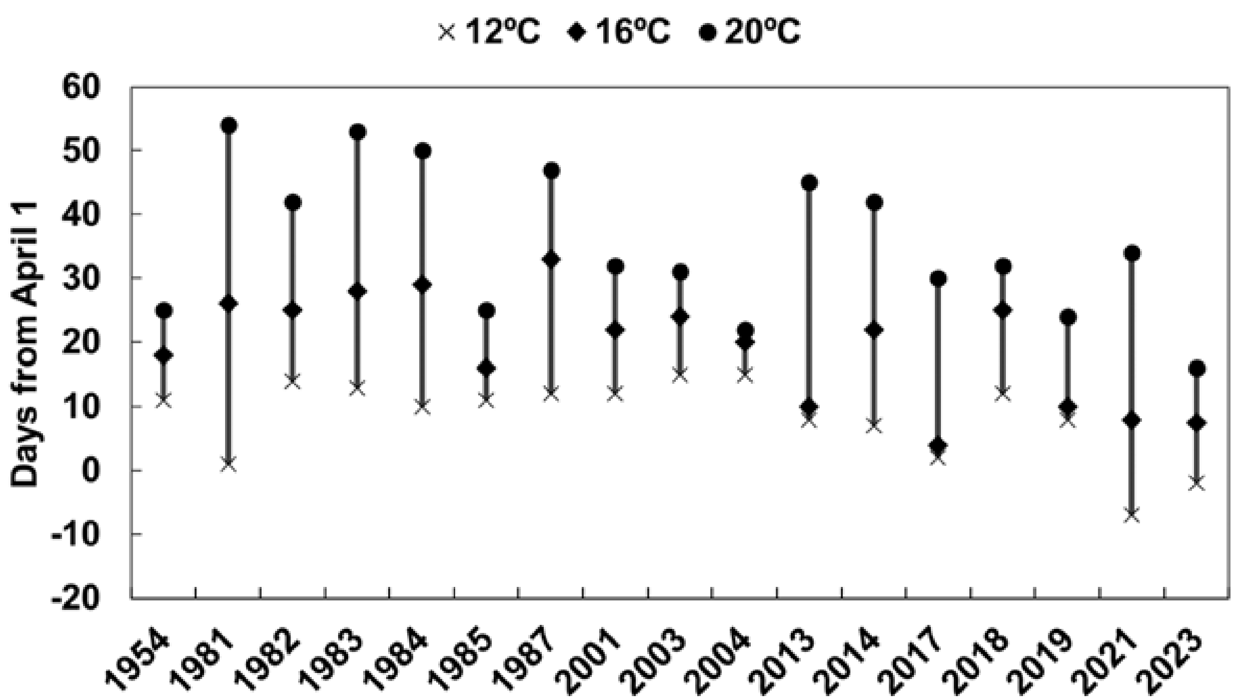
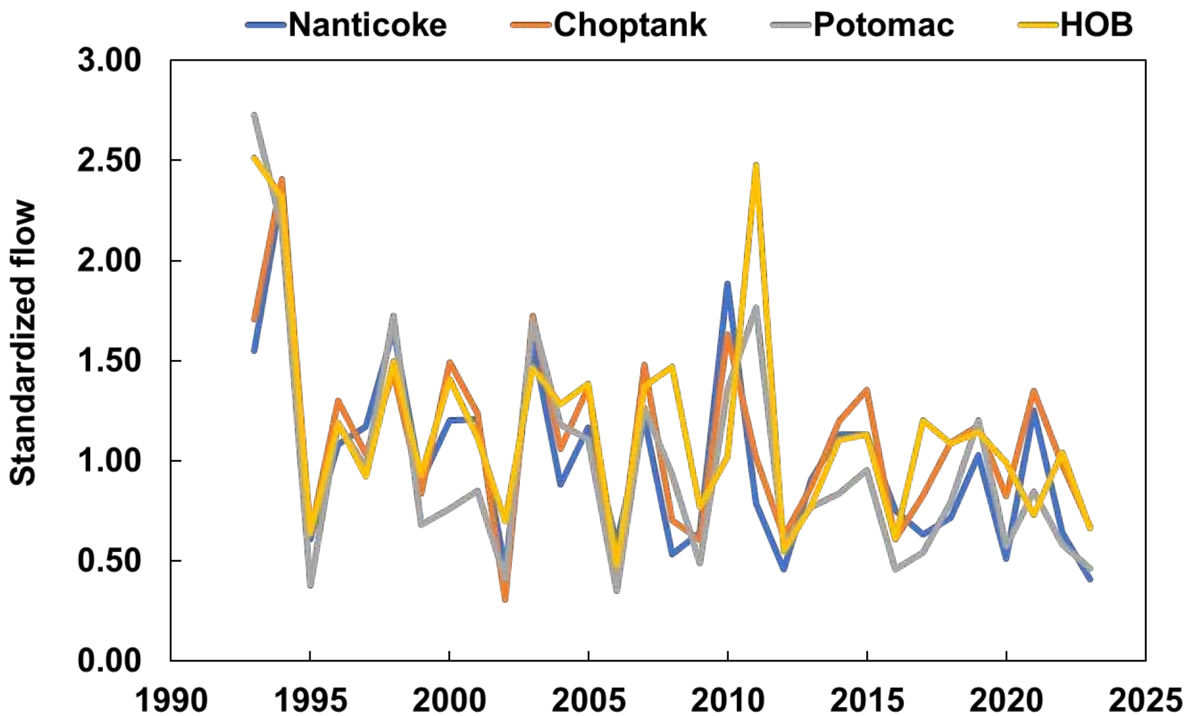


Figure 2.1.15. Two-month average flows in cubic feet per second for months prior to and including spawning season during 1993-2022, standardized to their averages for years in common during 1957-2020.


Nanticoke
Choptank
Potomac
HOB
Standardized flow
3.00
2.50
2.00
1.50
1.00
0.50
0.00
1990
1995
2000
2005
2010
2015
2020
2025

Appendix. Baywide *Ep* (proportion of tows with Striped Bass eggs) estimated for spawning areas sampled for the Maryland juvenile index combined into a baywide annual index. $N_{Present}$ = number of eligible samples with eggs; N_{Total} = total number of eligible samples. High CI and Low CI refer to 90% confidence interval boundaries.

Year	$N_{Present}$	N_{Total}	*Ep*	SD	CV	High CI	Low CI
1955	25	40	0.63	0.08	0.12	0.75	0.50
1956	128	179	0.72	0.03	0.05	0.77	0.66
1957	35	44	0.8	0.06	0.08	0.90	0.70
1958							
1959							
1960							
1961	54	61	0.89	0.04	0.05	0.95	0.82
1962	92	105	0.88	0.03	0.04	0.93	0.82
1963	93	101	0.92	0.03	0.03	0.97	0.88
1964	66	85	0.78	0.05	0.06	0.85	0.7
1965	54	59	0.92	0.04	0.04	0.98	0.86
1966	68	68	1.00	0		1.00	1.00
1967	71	92	0.77	0.04	0.06	0.84	0.7
1968	54	65	0.83	0.05	0.06	0.91	0.75
1969	49	65	0.75	0.05	0.07	0.84	0.67
1970	69	79	0.87	0.04	0.04	0.94	0.81
1971	54	71	0.76	0.05	0.07	0.84	0.68
1972	40	53	0.75	0.06	0.08	0.85	0.66
1973	176	276	0.64	0.03	0.05	0.69	0.59
1974	202	309	0.65	0.03	0.04	0.7	0.61
1975	364	443	0.82	0.02	0.02	0.85	0.79
1976	384	505	0.76	0.02	0.02	0.79	0.73
1977	352	419	0.84	0.02	0.02	0.87	0.81
1978	32	42	0.76	0.07	0.09	0.87	0.65
1979	41	44	0.93	0.04	0.04	0.99	0.87
1980	80	118	0.68	0.04	0.06	0.75	0.61
1981	107	163	0.66	0.04	0.06	0.72	0.6
1982	34	64	0.53	0.06	0.12	0.63	0.43
1983	48	132	0.36	0.04	0.12	0.43	0.29
1984	137	357	0.38	0.03	0.07	0.43	0.34
1985	165	312	0.53	0.03	0.05	0.58	0.48
1986	41	94	0.44	0.05	0.12	0.52	0.35
1987	65	119	0.55	0.05	0.08	0.62	0.47
1988	132	247	0.53	0.03	0.06	0.59	0.48
1989	401	556	0.72	0.02	0.03	0.75	0.69
1990	88	115	0.77	0.04	0.05	0.83	0.70
1991	79	95	0.83	0.04	0.05	0.89	0.77
1992	71	79	0.90	0.03	0.04	0.95	0.84
1993	55	63	0.87	0.04	0.05	0.94	0.80
1994	128	170	0.75	0.03	0.04	0.81	0.70
1995	59	69	0.86	0.04	0.05	0.94	0.77
1996	72	90	0.8	0.04	0.05	0.87	0.73
1997	90	112	0.8	0.04	0.05	0.87	0.74

Appendix. (Continued).

Year	$N_{Present}$	N_{Total}	*Ep*	SD	CV	High CI	Low CI
1998	76	99	0.77	0.04	0.06	0.84	0.70
1999	82	99	0.83	0.04	0.05	0.89	0.77
2000	66	90	0.73	0.05	0.06	0.81	0.66
2001	32	47	0.68	0.07	0.10	0.79	0.57
2002	52	60	0.87	0.04	0.05	0.94	0.79
2003	71	92	0.77	0.04	0.06	0.84	0.7
2004	95	125	0.76	0.04	0.05	0.82	0.7
2005	45	66	0.68	0.06	0.08	0.78	0.59
2006	55	77	0.71	0.05	0.07	0.8	0.63
2007	48	61	0.79	0.05	0.07	0.87	0.7
2008	61	96	0.64	0.05	0.08	0.72	0.55
2009	62	76	0.82	0.04	0.05	0.89	0.74
2010	59	69	0.86	0.04	0.05	0.92	0.79
2011	40	47	0.85	0.05	0.06	0.94	0.77
2012	35	54	0.65	0.06	0.10	0.76	0.54
2013	112	135	0.83	0.03	0.04	0.88	0.78
2014	102	149	0.68	0.04	0.06	0.75	0.62
2015	99	145	0.68	0.04	0.06	0.75	0.62
2016	122	146	0.84	0.03	0.04	0.89	0.79
2017	105	130	0.81	0.03	0.04	0.86	0.75
2018	49	73	0.67	0.05	0.08	0.76	0.58
2019	92	128	0.72	0.04	0.06	0.78	0.65
2020							
2021	60	90	0.67	0.05	0.07	0.76	0.57
2022	82	118	0.69	0.04	0.06	0.78	0.61
2023	34	46	0.74	0.06	0.08	0.87	0.61

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

Section 2.2: Striped Bass Larval Feeding – Derivation of Criteria for Successful and Unsuccessful Feeding Using Historic Choptank River Estimates

Jim Uphoff and Carrie Hoover

Striped Bass in Maryland's Chesapeake Bay spawning areas have experienced five years of poor year-class success in a row (2019-2023; Durell and Weedon 2023). Changed patterns in water temperature during winter and spawning season, and changes in flow patterns were offered as leading hypotheses for this dearth of year-class success by MD DNR https://news.maryland.gov/dnr/2023/10/12/chesapeake-bay-2023-young-of-year-striped-bass-survey-results-announced/. These conditions were further linked to mistiming of zooplankton availability to first-feeding postlarvae (mismatch hypothesis). If mistiming of zooplankton blooms with first-feeding larvae is important, its density-dependent nature may limit successful management based on high spawning stock size. It becomes possible that larval survival and subsequent recruitment will be capped at a low level due to inadequate larval foraging regardless of egg production if zooplankton production is misaligned with first-feeding larvae.

Factors affecting larval survival are critical to recruitment success of Striped Bass because year-class strength is largely determined by the end of the larval stage (Uphoff 1989; Rutherford et al. 1997; Maryland Sea Grant 2009). Availability of zooplankton prey affects larval Striped Bass nutritional condition, growth, size, and survival (Martin et al. 1985; Rutherford et al. 1997; North and Houde 2001; 2003; Houde 2008; Maryland Sea Grant 2009; Martino and Houde 2010; Shideler and Houde 2014). Growth and mortality are linked processes in larval fish and variability in either during early life will be expressed in variable sizes-at-age, stage durations, and stage-specific cumulative mortality (Houde 2008).

Water temperature and flow conditions are important influences on year-class success of Striped Bass that may be linked to zooplankton availability (Hollis et al. 1967; Uphoff 1989; 1992; Secor and Houde 1995; Rutherford et al. 1997; 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). Temperature can impact recruitment through direct 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, while flow may be associated with zooplankton dynamics, nursery volume, 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; Kimmel and Roman 2004, Kimmel et al. 2006; Maryland Sea Grant 2009; Martino and Houde 2010; Shideler and Houde 2014; Secor et al. 2017; Millette et al. 2020). In Chesapeake Bay spawning and larval nursery areas, a combination of high spring flows and cooler temperatures may extend or delay spring bloom conditions of zooplankton prey until when most feeding Striped Bass larvae are present (Wood 2000, Martino and Houde 2010; Millette et al. 2020). High spring discharge that favors anadromous fish recruitment in Chesapeake Bay could represent episodes of hydrologic transport of accumulated terrestrial carbon (organic matter) from watersheds that fuel zooplankton production and feeding success (North and Houde 2001; McClain et al. 2003; Hoffman et al. 2007; Martino and Houde 2010; Shideler and Houde 2014). Freshwater flow also controls the spatial distribution of larval Striped Bass and zooplankton prey, and high springtime flows are associated with increased spatial overlap of larvae and zooplankton prey at the salt front and turbidity maximum (North and Houde 2003; Martino and Houde 2010). Poor year-class success of Chesapeake Bay Striped Bass was highly likely when flows were below average (Gross et al. 2022; Uphoff et al. 2022).

Hypotheses relating temperature and flow to this recent downturn in year-class success are viable, but relating these to zooplankton mismatch requires directed research. In 2023, we collected larvae in Choptank River to explore the zooplankton mismatch hypothesis. We reexamined historic Choptank River Striped Bass and White Perch feeding and larval dynamics data from 1980s ichthyoplankton surveys (Uphoff 1989; 1992; 1993) to establish baselines for comparison with 2023 collections.

Postlarvae were collected during Choptank River ichthyoplankton surveys and examined for feeding success during 1981-1986 and 1989-1990. The only published analysis of these feeding data was a summary of feeding incidence on all items during 1981-1985 (Uphoff 1989). These feeding data existed in a hard to access database that had not been supported for many years. Carrie Hoover reentered these data from data sheets. Daily instantaneous mortality and growth of postlarvae (Z and G, respectively) had been estimated for 1980-1989 for Striped Bass (Uphoff 1993) and these estimates were updated through 1990. Feeding success metrics were estimated and compared to Z and G to form criteria for successful and unsuccessful feeding. White Perch larvae were included because they shared larval habitat and diet with Striped Bass larvae (North and Houde 2001; 2003; Campfield and Houde 2011; Uphoff 2023). Contrast in White Perch and Striped Bass feeding success would provide insight on the extent that zooplankton abundance and-or larval feeding ability might be reflected by feeding success. Striped Bass and White Perch juvenile indices (JI, an indicator of year-class success; Durell and Weedon 2023) in Maryland's portion of Chesapeake Bay during 1957-2019 were well correlated, indicating larval habitat conditions were a major factor influencing their year-class success (Uphoff 2023).

Methods

Striped Bass eggs and larvae were sampled weekly from April to mid-June 1980-1990 with midwater and bottom trawls fitted with plankton nets in their cod ends. Bottom channel and inshore bottom habitats were sampled with a 3.05-m box trawl made of 1.27-cm-stretch-mesh knotless nylon. A 1.53 • 1.53-m midwater trawl was used to sample upper to mid-depth habitat in the channel; the mouth of the mid-water trawl (approximately 1-m deep) was made of 3.18-cm-stretch-mesh nylon, and the remainder of the net was 1.27-cm-stretch-mesh nylon. These gears were similar to those deployed by Kerhnehan et al. (1981) to survey Striped Bass early life stages in the vicinity of the Chesapeake and Delaware Canal during 1973-1977. The midwater trawl had two large floats added to each metal door to prevent diving during deployment. Codends of both nets had 0.5-mm mesh liners (8:1 length to opening ratio) kept open with 0.5-m-diameter hoops. All trawls were towed at 1.03 m/s for 2 min in the same direction as the tidal current. Sampling was conducted during daylight hours, usually in the morning. In 1980-1986, four stations were sampled within the Choptank River Striped Bass spawning area at km 47.2, 56.8, 67.0, and 79.0 (Figure 2.2.1). One site per day was sampled, and all samples collected during a weekly sample interval were taken within 5 d of each other. The uppermost site was moved from km 79.9 to 79.0 after the 1980 season to avoid excessive detrital concentrations. Two bottom (5-8 m deep) and two midwater (0.5-2.0 m below surface) tows were made in the channel at all four stations. Two inshore bottom trawl tows (1-2 m deep) per station were made at the three lower stations. Inshore bottom tows could not be made above km 67.0 because shallow areas were not extensive enough. During 1987-1990, the Choptank River spawning area was divided into 21 1.61-km segments, starting at km 47.2 and proceeding upstream (Figure 2.2.1); four of these segments were in Tuckahoe Creek (starting at the mouth). Segments were aggregated into four subareas: Tuckahoe Creek and three subareas covering the spawning area of the mainstem. These subareas maintained the geographic coverage of the fixed stations used during 1980-1986 and incorporated an important tributary. The lower Choptank area consisted of the first 5 segments; the middle, segments 6-11; the upper, segments 12-17; and Tuckahoe Creek, segments 18-21. Eight to 12 stations were visited randomly within a week, and each subarea had a minimum of one visit. A bottom and a midwater trawl tow were made at each visit prior to the last week of May to sample eggs, prolarvae, and postlarvae. An additional inshore tow was made, where possible, from the last week of May through early to mid-June to sample early juveniles (Uphoff 1992).

Samples were stained with rose bengal, preserved in 5-10% formalin, and sorted in the laboratory. Organisms were identified to the lowest practical taxonomic level and life stage. The structural definitions of Rogers et al. (1977) were used to differentiate prolarvae (larvae with yolk), postlarvae (larvae that had absorbed their yolks), or juveniles (post-metamorphic stages). Larvae of Striped Bass and White Perch were abundant in samples; *Morone* specimens between 11 and 14 mm TL were cleared and stained to aid identification (Fritsche and Johnson 1981). The total lengths of Striped Bass and White Perch prolarvae, postlarvae, and juveniles were measured to the nearest millimeter during 1980-1986 and to the nearest tenth of a millimeter during 1987-1990. All Striped Bass and White Perch larvae and juveniles in a sample were measured if there were fewer than 30. Otherwise, a subsample of 30 was selected. Measurements in tenths of a millimeter were rounded to the nearest millimeter when classifying them into length bins (for example, larvae in the 5 mm bin would consist of those between 4.6 and 5.5 mm). All measurements were total length (TL).

The presence and type of food (cladocerans, copepods, and miscellaneous) in the gut were recorded for each Striped Bass or White Perch larva or juvenile measured; cladocerans were most likely to be *Bosmina longirostris* and copepods *Eurytemora carolleeae* based on other Chesapeake Bay studies (Martin et al. 1985; North and Houde 2003; Martino and Houde 2010; Campfield and Houde 2011; Shideler and Houde 2014; Millette et al. 2020). The copepod category included both adults and copepodites. The miscellaneous category included unidentifiable contents, debris, and other organisms. Each food type was assigned a score of 0-4. If food was not present the gut was assigned a score of 0. If up to ¼ of the gut contained an item, it was assigned a score of 1. If ¼ up to ½ of the gut contained an item, it was assigned a score of 2. If ½ up to ¾ of the gut contained an item, it was assigned a score of 3. If the gut was filled by an item, the score was 4. The sum of scores for all items in a gut could not exceed 4. We calculated feeding incidence (frequency of presence) of all items, cladocerans, copepods, and miscellaneous items for each postlarva as well (described below).

Our reexamination of feeding was confined to *Morone* larvae 10 mm and smaller. This limited the sizes to those we were going to be able to identify in our 2023 collection without clearing and staining and would represent the lengths prior to when year-class success was set (Uphoff 1989; 1992). Relative abundances of Striped Bass larvae or early juveniles in 1-mm length increments between 10 and 20 mm were strongly correlated with the Choptank River

Striped Bass year-class as measured by the juvenile index (Durell and Weedon 2023) and those increments representing smaller lengths were not (Uphoff 1989; 1992). Larvae 10 mm and less would represent sizes where most mortality occurred. Striped Bass and White Perch larvae of this size could be identified based on external characteristics. White Perch would have more advanced development until about 7-8 mm (Lippson and Moran 1974) and then the presence of an oil globule or oil droplets in Striped Bass were an important characteristic (J. Uphoff, personal observation). Ventral pigment patterns just below the gills would be considered when presence of oil was uncertain; White Perch often had well defined dark pigment "dashes" that were either aligned or in a "box" pattern (four dashes evenly spaced) while Striped Bass had irregular pigment in the same area (J. Uphoff, personal observation).

We classified 5-7 mm postlarvae as first-feeding larvae, matching the lengths used in Uphoff (1989). A second group of larger Striped Bass postlarvae, 8-10 mm, was also created for comparisons of feeding metrics of later postlarvae with first-feeding postlarvae. Uphoff (1989) estimated that approximately 18% of larvae at 5 mm would be classified as postlarvae (the remainder would be prolarvae), 42% at 6 mm, and 89% at 7mm; all 8-10 mm larvae were classified as postlarvae. Correlations of larval abundance at length and the Striped Bass juvenile index strengthened rapidly after 7 mm TL and were strong ($r > 0.90$) by 10 mm TL in the Choptank River (Uphoff 1989; 1992). Year-class success was determined by 8 mm standard length in the Potomac River (Rutherford et al. 1997).

For each size group of Striped Bass postlarvae, we estimated the following feeding metrics based on food category scores: mean score of all food items, mean score of cladocerans, mean score of copepods, and mean score for miscellaneous items.

The feeding success scoring system was amenable to estimating feeding incidence (proportion with food) as well since it had a zero category. We estimated feeding incidence on all items, feeding incidence on cladocerans, feeding incidence on copepods, and feeding incidence on miscellaneous items. For any given category of food item (FI_x = feeding incidence on a given item category *x* and *x* was all items, cladocerans, copepods, or miscellaneous) feeding incidence was estimated as

$$FI_x = Nx_{present} / N_{total};$$

where $Nx_{present}$ equaled the number of qualifying samples with a food item present and N_{total} equaled the total number of qualifying samples. The SD of FI_x was estimated from the normal distribution approximation of the binomial distribution as:

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

Ninety-five percent confidence intervals were constructed as:

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

We reported the mean score and feeding incidence for miscellaneous items but did not include them in subsequent analyses because copepods and cladocerans have been identified in nearly all studies as main food items and because miscellaneous encompassed food and non-food items in unknown proportions. The all-item categories included miscellaneous items.

We used correlation analysis to determine the strength of linear associations among feeding categories (except miscellaneous) for first-feeding larvae for each year sampled ($N = 7$ for all feeding comparisons). If the metrics from feeding scores and feeding incidence were strongly correlated ($r \leq 0.80$; Ricker 1975) then one set was chosen to reduce redundancy and minimize reductions in statistical power by making more comparisons than necessary

(Nakagawa 2004). Our preference was to use feeding incidence since interpretation was straightforward and robust (Baker et al. 2014) and confidence intervals could easily be constructed using the normal distribution approximation of the binomial distribution.

Candidate metrics for first-feeding postlarvae were compared with each other, estimates of Striped Bass postlarval Z, postlarval G, and comparable feeding metrics of 8-10 mm postlarvae (all items, cladocerans, or copepods). These exploratory analyses used correlation analysis, linear regression, and nonlinear regression (N = 7 for all feeding comparisons). The relationships and distributions of feeding metrics with Z and G were of particular interest for developing criteria for successful and unsuccessful feeding. Methods used to estimate Z and G are presented in the Appendix and are also described in Uphoff (1989; 1992; 1993).

Once a set of suitable feeding metrics was developed, we compared feeding success to relevant zooplankton densities (organisms per liter) reported for Choptank River during 1983-1985. Uphoff (1989) used zooplankton data collected by Wright et al. (1987) at sites 5 and 10 (Figure 1) to estimate average densities of copepods and cladocerans during the postlarval periods. The estimated postlarval period began one week after the midpoint of the peak spawning period (time span to collect 85% of eggs) and lasted for the time estimated for growth in length from 6 mm to 12 mm (Uphoff 1989). The relevant Choptank River feeding metrics and zooplankton densities were compared to examine how well feeding tracked zooplankton availability.

We also tested for density-dependent effects of initial Striped Bass larval abundance on FI_{cope} and FI_{clad} using correlation analysis. Total larval catches in the annual length-frequency distributions peaked at 6 mm and Uphoff (1989; 1992; 1993) used the ratio of 6-mm larvae catches to egg catches to estimate prolarval survival. We log_e-transformed catches of 6 mm larvae for our index of initial abundance of first-feeding larvae to linearize these estimates in this analysis. Total annual survey catches were adjusted for differences in effort among surveys. Estimates of adjusted catches were available for 1980-1990 from printouts and notes. Ideally, we would have used 5-7 mm larval catches, but these estimates were not available and the data could not be re-analyzed to produce new estimates.

Bivariate plots were viewed to be sure that linear analyses would capture basic dynamics. If curvilinear relationships were suspected, non-linear regression was used (SAS Proc NLIN). Residuals of regression analyses were examined for outliers and serial trends.

Nonlinear models were fit in Excel using Solver to obtain starting parameter estimates. Then the default Gauss-Newton algorithm was used in Proc NLIN (SAS) to fit the model and obtain approximate SEs for the parameters. An r^2 and P for nonlinear regressions were approximated from linear regressions of observed and predicted dependent variables.

Feeding success metrics of Striped Bass and White Perch postlarvae were of interest and the same set of metrics that were moderately or strongly associated or related to G and Z for Striped Bass postlarvae were estimated for 5-7 mm White Perch postlarvae. Contrast or similarity in feeding metrics of these two species could provide insight on zooplankton abundance and feeding dynamics.

The 2023 sampling program was not intended to be a full early life history study and we developed the ratio of feeding sample sizes for the 5-7 mm (N_{5-7}) and 8-10 mm (N_{8-10}) size groups for 1981-1990 as a proxy estimate of survival to consider for 2023 samples. The ratio was estimated as

$$N_{ratio} = N_{8\text{-}10} / N_{5\text{-}7}.$$

The N_{ratio} time-series was compared to Z for years in common using linear and non-linear regression to determine if it might be a reasonable proxy for Z.

We classified correlations as strong, based on r > $\pm$ 0.80 (Ricker 1975). Weak correlations were indicated by r < $\pm$ 0.50; and moderate correlations fell in between. Relationships indicated by regressions were categorized using the values for correlations squared: a strong relationship was indicated at $r^2 > 0.64$; weak relationships were indicated by $r^2 < 0.25$; and moderate relationships fell in between. We considered strong and moderate correlations or relationships to be of interest. Levels of significance were reported, but potential for biological significance took precedence over $P \leq 0.05$ (Anderson et al. 2000; Smith 2020).

Results

Feeding metrics could be estimated for 1981-1985 and 1989-1990; annual sample sizes for 5-7 mm Striped Bass postlarvae ranged between 104 and 390 and between 19 and 314 for 8-10 mm postlarvae (Table 2.2.1). Sample size for first-feeding larvae was very low in 1986 (N = 10) and it was dropped from analysis.

Mean feeding scores of 5-7 mm Striped Bass postlarvae ranged from 1.10 to 1.90 for all items, 0.20 to 1.20 for cladocerans, 0.20 to 1.70 for copepods, and 0.05-0.37 for miscellaneous items (Table 2.2.1). Feeding incidence ranged from 0.55 to 0.80 for all items, 0.12 to 0.63 for cladocerans, 0.09 to 0.70 for copepods, and 0.05-0.019 for miscellaneous items (Table 2.2.1; Figure 2.2.2). Mean scores and FI between all-item, cladoceran, or copepod metrics were all strongly correlated (r = 0.97-0.99, $P \leq 0.0002$). Based on this strong agreement and preference for FI metrics, we dropped feeding scores from further analysis.

All-item and miscellaneous FI varied less than FI for cladocerans (FI_{clad}) or copepods (FI_{cope}; Table 2.2.1; Figure 2.2.2). Higher FI_{clad} (> 0.50) occurred during 1981, 1982, and 1989; lower FI_{clad} (<0.35) occurred during the remaining years. Higher FI_{cope} (> 0.35) occurred during 1985 and 1990; mid-range F_{cope} (0.22-0.26) occurred in 1982, 1984, and 1989; and lower FI_{cope} (<0.11) occurred during the remaining years. Miscellaneous FI was usually low relative to the other two item-specific categories and only exceeded one of them (FI_{cope}) in 1983; it was close to FI_{clad} in 1985 and 1990 (Table 2.2.1; Figure 2.2.2).

All-item FI of 5-7 mm postlarvae was poorly correlated with FI_{clad} (r = 0.07, P = 0.87) and modestly correlated with FI_{cope} (r = 0.55, P = 0.20). Feeding incidences of 5-7 mm postlarvae for cladocerans and copepods were moderately and negatively correlated (r = -0.68, P = 0.095). Inspection of the bivariate plot of FI_{clad} and FI_{cope} suggested a negative curvilinear relationship and a power function provided a better fit ($r^2 = 0.82$, P = 0.005; Figure 2.2.3) than indicated by linear correlation. This strong negative relationship was described by the equation:

$$FI_{cope} = 0.0812 \bullet (FI_{clad}^{-0.9876})$$

The approximate SE was 0.0315 for the scale coefficient and approximate SE = 0.2099 for the exponent. The residual plot did not suggest an outlier.

During 1980-1990, estimates of G ranged from 0.030 to 0.044 and the three highest estimates (1985, 1986, and 1990) were between 0.038 to 0.044 and could be differentiated from five of the lower estimates (range = 0.030–0.034) based on 95% CI overlap (Figure 2.2.4). Three estimates of G (1984, 1987, and 1988) could not be differentiated as high or low (Table 2.2.1; Figure 2.2.4).

Estimates of G were weakly correlated with all-item FI ($r = 0.41$, $P = 0.357$) and strongly correlated with item-specific estimates of FI. The correlation was negative for FI_{clad} ($r = -0.83$, $P = 0.02$) and positive for FI_{cope} ($r = 0.94$, $P = 0.002$). Lower growth rates ($G \leq 0.034$) occurred when FI_{clad} was greater than 0.32 and FI_{cope} was less than 0.38 (Table 2.2.1).

During 1980-1990, estimates of Z ranged from 0.04 to 0.22 (Figure 2.2.5). There were four estimates of higher Z (0.16-0.22) and seven lower estimates (0.04-0.11). Four of the lower estimates could be separated from the higher estimates based on 95% CI overlap. Three lower estimates could not be separated from 0 based on 95% CIs. Two estimates of low mean Z (1986 and 1988) could not be separated from high or low estimates. There were three higher estimates of Z (1981, 1983, and 1984) within the years with FI estimates and the available lower estimates did not overlap the high ones (Table 2.2.1; Figure 2.2.5).

Estimates of Z were moderately and negatively correlated with all-item FI ($r = -0.50$, $P = 0.250$), weakly correlated with FI_{clad} ($r = 0.27$, $P = 0.560$), and modestly correlated with FI_{cope} ($r = -0.66$, $P = 0.103$). Inspection of the bivariate plot of Z and FI_{cope} suggested a negative curvilinear relationship and a power function provided a better, but moderate fit ($r^2 = 0.56$, $P = 0.054$; Figure 2.2.6). This relationship was described by the equation:

$$Z = 0.0433 \bullet (FI_{cope}^{-0.6226})$$

Variation of parameter estimates was high (approximate SE = 0.0237 for the scale coefficient and approximate SE = 0.2746 and for the exponent). The residual for 1984 was over 1.9-times greater than the next largest and we suspected it was a highly influential outlier. Removal of 1984 from the regression greatly improved the fit ($r^2 = 0.91$, $P = 0.003$; Figure 2.2.6). This relationship was described by the equation:

$$Z = 0.0240 \bullet (FI_{cope}^{-0.8656})$$

Precision of the parameter estimates improved (approximate SE = 0.0084 for the scale coefficient and approximate SE = 0.1625 for the exponent). The residual plot did not suggest an outlier.

Points representing higher Z occurred exclusively when FI_{cope} was 0.11 or less (Figure 2.2.6). When FI_{cope} was between 0.22 and 0.24, there was one estimate of higher Z and two of lower Z. Lower estimates of Z occurred exclusively when FI_{cope} was 0.38 or more (Figure 2.2.6).

All-item FI of 5-7 mm Striped Bass postlarvae was poorly correlated with all-item FI of 8-10 mm postlarvae ($r = -0.03$, $P = 0.948$). On an item-specific basis, feeding incidence of first-feeding larvae and more advanced postlarvae was moderately associated for cladocerans ($r = 0.68$, $P = 0.092$) and strongly associated for copepods ($r = 0.83$, $P = 0.022$). Two years (1981 and 1985) exhibited lower FI_{clad} as length group progressed from smaller to larger (difference in FI_{clad}

was -0.12 and -0.13 within a year) but the remaining years exhibited an increase in FI_{clad} (+0.05 to +0.42; Table 2.2.1). Copepod FI always increased as postlarval size class increased (difference in FI_{cope} between the smaller and larger size classes was +0.05 to +0.36; Table 2.2.1).

High estimates of Z occurred when the N ratio was 0.36 or less and low Z occurred when it was 0.44 or more (Figure 2.2.7). The bivariate plot of the N ratio (N_r) and Z suggested a negative, curvilinear relationship and a power function provided a moderate fit when all years were included (approximate $r^2 = 0.50$, $P = 0.074$). The relationship was described by the equation:

$$Z = 0.0661 \bullet N_r^{-0.6395} \text{ (Figure 2.2.6).}$$

The SE for the scale parameter was 0.0209 and 0.2311 for the exponent. The largest residual, 1984, was 1.9-times higher than the next and was a possible outlier. The power function was rerun without 1984 and the fit improved ($r^2 = 0.88$, $P = 0.017$). The relationship was described by the equation:

$$Z = 0.0535 \bullet Nr^{-0.7289} \text{ (Figure 2.2.6).}$$

The SE for the scale parameter was 0.0141 and 0.1816 for the exponent. Examination of residuals did not suggest an outlier.

Mean densities of copepods in Choptank River were 41.2/L, 50.9/L, and 285.1/L during the postlarval periods of 1983 (May 4-14), 1984 (May 8-18), and 1985 (April 22-30); respective cladoceran densities were 29.8/L, 45.1/L, and 2.9/L. Estimates of FI_{cope} for 5-7 mm postlarvae were 0.106, 0.240, and 0.702 during 1983, 1984, and 1985, respectively. Estimates of FI_{clad} were 0.317, 0.338, and 0.115 during 1983, 1984, and 1985, respectively.

Annual sample sizes for 5-7 mm White Perch postlarvae ranged between 130 and 753 (Table 2.2.2). Analyses for White Perch were confined to feeding incidence on cladocerans ($WPFI_{clad}$) and copepods ($WPFI_{cope}$) since these were the two 5-7 mm Striped Bass metrics that were reasonably linked to G and Z. Feeding incidence of 5-7 mm White Perch ranged from 0.04 to 0.75 for cladocerans and 0.10 to 0.71 for copepods (Table 2.2.2; Figure 2.2.8). Coefficients of variation for $WPFI_{clad}$ were less than 0.10 except for 1985 (CV = 0.44); CVs for $WPFI_{cope}$ were between 0.05 and 0.08, except for 1981(0.15; Table 2.2.2).

The linear relationship relating $WPFI_{clad}$ and Striped Bass FI (FI_{clad}) was strong ($r^2 = 0.72$, $P = 0.0146$) and described by the equation:

$$FI_{clad} = (0.649 \bullet WPFI_{clad}) + 0.102.$$

The SE of the slope equaled 0.177 and the SE of the intercept equaled 0.090. First feeding Striped Bass postlarvae were less likely to feed on cladocerans than similarly sized White Perch postlarvae.

The equation relating $WPFI_{cope}$ (White Perch) and FI_{cope} (Striped Bass) was strong ($r^2 = 0.92$, $P = 0.0006$) and was described by the equation:

$$FI_{cope} = (1.034 \bullet WPFI_{cope}) - 0.079.$$

The SE of the slope equaled 0.135 and the SE of the intercept equaled 0.053. The 95% CI of the slope overlapped 1.0 and the intercept CIs overlapped 0, indicating that it was likely that FI_{cope} would the similar between 5-7 mm Striped Bass and White Perch in a given year.

Effort adjusted log_e-transformed catches of 6 mm Striped Bass larvae ranged from 4.16 (64 in 1983) to 8.06 (3,176 in 1989; Table 2.2.3, Figure 2.2.9). Estimates of FI_{cope} were poorly correlated with log_e-transformed catches of 6 mm larvae ($r = 0.33$, $P = 0.47$) and this association offered little support for a hypothesis linking abundance of first-feeding larvae and their success feeding on copepods. Estimates of FI_{clad} were positively and moderately correlated with log_e-transformed catches of 6 mm larvae ($r = 0.72$, $P = 0.07$; Figure 2.2.9). First feeding Striped Bass larval diets included more cladocerans when 6 mm larval abundance was high. This could reflect cladocerans as an important diet supplement with increased competition among other Striped Bass larvae and White Perch larvae for copepods, or increased cladoceran abundance due to favorable environmental conditions.

Discussion

Both FI_{clad} and FI_{cope} from 1980s surveys appeared to be useful candidates for developing criteria to judge feeding success of first-feeding Striped Bass postlarvae in samples collected during 2023. Associations and relationships were strong enough to be of interest and high and low values could be discerned. Feeding success by first-feeding Striped Bass larvae on copepods was positively linked to G and negatively linked to Z, while FI_{clad} linkages were opposite. Negative linkage of FI_{clad} and positive linkage of FI_{cope} with G and Z suggested that nutrition from consumption of copepods was greater than from cladocerans. However, positive correlation of 6 mm larval abundance with FI_{clad} suggests that cladocerans may have provided a supplement when first-feeding larvae were abundant and higher intraspecific and interspecific competition for copepods was likely. Abundance of 6 mm larvae was a positive function of water temperature during the peak spawn period and it may also have been lower during 1982-1988 due to lower spawning stock biomass, egg production, and spatial-temporal distribution (Uphoff 1989;1992; 2023).

Positive linkages of first-feeding Striped Bass FI_{cope} with G or Z were stronger than for FI_{clad}. Lower growth rates ($G \leq 0.034$) occurred when FI_{cope} was less than 0.38 (Table 2.2.6). High Z occurred exclusively when FI_{cope} was 0.11 or less; FI_{cope} between 0.22 and 0.24 exhibited one estimate of high Z and two of low Z; and low estimates of Z occurred when FI_{cope} was 0.38 or more. However, two of four years exhibiting low Z (1982 and 1989) in Choptank River exhibited a combination of moderate FI_{cope} (0.26 and 0.22, respectively) and high FI_{clad} (≥ 0.57). Anomalously high Z in 1984 occurred under moderate FI_{cope} (0.24) coupled with the third lowest estimate of F_{clad} (0.34). High Z occurred when high to low FI_{clad} (0.55-0.12) was coupled with low FI_{cope} (Table 2.2.6).

We reviewed papers with compatible copepod and cladoceran density estimates from other Chesapeake Bay larval nurseries and compared those densities with the 1983-1985 Choptank River densities. Comparable estimates of copepods/L (adults, copepodites, and nauplii) and cladocerans/L existed for the Potomac River (1976, 1977, 1980- 1982, and 1987-1989) and Head-of-Bay (1988 and 1989) for two time periods: April-May and the last two weeks of May (peak densities; Rutherford et al. 1997; Table 2.2.4). Estimates of mean April-May densities of copepods in these two spawning areas were less variable (31.1/L – 71.1/L) than then late May (6.0/L – 169.4/L). Choptank River copepod densities for 1983 (41.2/L) and 1984 (50.9/L) fell within both ranges, but 1985 density was much greater (285.1/L). Cladoceran densities reported in Rutherford et al. (1997) for April-May were higher in the Potomac River (14.6/L – 187.1) than Head-of-Bay (0.6/L and 3.1/L) or Choptank River (29.8/L, 45.1/L, and 2.9/L during 1983-1985); estimates of cladoceran density for the last two weeks of May were only available for Potomac River (3.2/L – 252.0/L) and half of these were greater than those reported for Choptank River (Table 2.2.4).

Copepods (*Eurytemora*) were an important larval Striped Bass diet item in Head-of-Bay studies during 2001, 2003, 2007, and 2008 (Martino and Houde 2010; Shideler and Houde 2014). The role of cladocerans (*Bosmina*) was more variable and their inclusion in the diet supported growth and survival of larvae above the estuarine turbidity maximum and later in the spawning season (Martino and Houde 2010; Shideler and Houde 2014). Striped Bass larvae may have selected for *Eurytemora* in upper Bay during 2007, but positive selection was not indicated during 2008. There was a negative preference for *Bosmina* in 2007 and no selection was evident in 2008 (Shideler and Houde 2014). Striped Bass larvae positively selected for *Eurytemora* and negatively selected for *Bosmina* during 2001-2002 in Patuxent River (Campfield and Houde 2011). The lowest level of starvation of Striped Bass larvae in Potomac River during 1981, as measured by morphometry, occurred when cladocerans were most abundant (Martin et al. 1985).

Both FI_{clad} and FI_{cope} in Choptank River exhibited considerable variation during the 1980s that provided contrast for the analyses. The highest FI_{clad} was 5.5-times greater than the lowest and there was a 7.6-times difference for FI_{cope}. When all items were considered, FI exhibited little variation (1.4-times) and was poorly linked to G and Z. The range of our all-item FI estimates, 0.55-0.80, were within estimates for the upper Bay during 2001-2003 (>0.50 to ~0.90; Martino and Houde 2010) and 2007-2008 (0.63 and 0.55, respectively; Shideler and Houde 2014). Item-specific FI and densities were not reported for these studies.

Copepod FI always increased as Striped Bass postlarval size class increased from 5-7 mm to 8-10 mm. Most, but not all years, indicated an increase in FI_{clad} between the size classes.

Copepod feeding incidences for White Perch and Striped Bass postlarvae 5-7 mm in Choptank River during the 1980s were not different based on overlap of linear regression parameters, but White Perch were more likely than Striped Bass to feed on cladocerans. If sample size of first-feeding Striped Bass is insufficient in 2023 (or subsequent years), pooling 5-7 mm White Perch and Striped Bass copepod feeding data could provide an option for at least partially addressing the feeding mismatch hypothesis. Further analysis would be needed to confirm if pooling into a single *Morone* category would produce acceptable results.

Analysis of Choptank River data from the 1980s indicated that the N ratio could serve as a proxy for Z. However, sample sizes and survey designs were different during the 1980s than in 2023 and some caution should be considered with comparisons. During the 1980s, feeding samples were drawn from all sites where feeding larvae were encountered over a sampling season designed to cover eggs through early juveniles. In 2023, sampling for feeding analysis began a week after significant spawning had occurred (see 2023 section). Sampling in 2023 was stratified by temperature and conductivity based on cumulative catch distributions of larvae during 1980s sampling to maximize the chance of catching larvae. Three sites were sampled each day within three conductivity zones that accounted for about 60% of larvae in the 1980s.

Sampling started the first week after major spawning and continued weekly for two more weeks until it appeared that there was little chance of catching first-feeding larvae.

Variation in FI_{cope} in the 1980s did not conform to the hypotheses linking long-term Striped Bass year-class success and low winter temperature and high spring flow and their effect on spring copepod production in the spawning rivers based on spawning area specific estimates spanning the mid-1990s to 2016 (Millete et al. 2020). We used long-term winter air temperature average (˚F) at Baltimore (www.weather.gov/media/1wx/climate/bwitemps.pdf) as a regional indicator of winter severity and March-April flows from the USGS gauging station at Greensboro, MD, as our indicator of spawning period discharge (choice of flow indicator is described in Uphoff et al. 2022; Table 2.2.5). These two environmental metrics were compared to FI_{cope} with correlation analysis. The correlation with winter severity was poor ($r = 0.36$, $P = 0.42$) and it was moderate, but negative, for flow ($r = -0.55$, $P = 0.20$; Table 2.2.5).

A major assumption that accompanies using feeding information from the 1980s to judge the present is that feeding itself and not underlying habitat conditions that may have affected feeding or zooplankton abundance are reflected in the associations and relationships with G and Z, i.e., habitat was stable. A general description of trends in major long-term natural and anthropogenic factors that could have resulted in larval habitat changes provided little indication of stable habitat conditions within the spawning and larval nursery areas of Chesapeake Bay since the 1950s (Uphoff 2023).

Acidic conditions and toxic metals were associated with high mortality of Striped Bass prolarvae in bioassays conducted during 1984 and 1987-1990 in Choptank River (Hall et al. 1993; Richards and Rago 1999). Low survival of Striped Bass postlarvae during 1980-1988 in the Choptank River estimated from ichthyoplankton surveys was associated with high flow and low pH, alkalinity, and conductivity that could have influenced toxicity of metals (Uphoff 1989; 1992). Associations of Z or G with pH and flow weakened considerably with the addition of 1989-1990 to the time-series, but associations of Z and conductivity or alkalinity and G with mean pH remained strong enough to be of interest. (J. Uphoff, unpublished analysis). Water quality monitoring during April – early May (approximating when larvae were most abundant in the 1980s) indicated that pH and alkalinity (alkalinity measured during 2021-2023) had increased substantially between 1986-1991 and 2014-2022 and were closer to those cited for productive hatcheries in the latter period (Uphoff 1989; 1992; 2023a; Uphoff et al. 2023). Conductivity did not appear to change. It seems unlikely that poor year-class success during 2019-2023 could be attributed to a return of toxic water quality conditions implicated in poor recruitment during the 1980s.

While water conditions associated with suspected metals toxicity in Choptank River improved during 2000-2023, water temperature patterns during spawning season became less favorable. Means or medians of days between 12ºC and 20ºC milestones during 2000-2021 were 10 days to 12 days shorter (respectively) than during 1954-1992 in Choptank and Nanticoke rivers (Uphoff et al. 2022). Changes were not uniform among temperature milestones. Early milestones appeared to be the least affected. In addition to these general changes, very short, asymmetric spans between 12ºC and 16ºC (when most eggs are collected) have become more common (Uphoff et al. 2022).

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Table 2.2.1. Choptank River Striped Bass postlarval feeding metrics and statistical characteristics, by size class, during 1981-1985 and 1989-1990. Clad = cladoceran; mean = mean feeding score; FI = feeding incidence, SD = standard deviation, CV = coefficient of variation (proportion), Cope = copepod; All = all items; Misc = miscellaneous items; N ratio = ratio of N between 5-7 mm and 8-10 mm length classes; G = instantaneous daily growth rate of postlarvae; Z = instantaneous daily mortality rate of postlarvae.

	Year						
	1981	1982	1983	1984	1985	1989	1990
				5-7 mm TL Postlarvae			
Clad mean	1.100	1.200	0.600	0.400	0.200	1.000	0.300
FI_{clad}	0.551	0.632	0.317	0.338	0.115	0.569	0.217
FI SD	0.030	0.030	0.050	0.030	0.030	0.030	0.020
FI_{clad} CV	0.054	0.047	0.158	0.089	0.260	0.053	0.092
Cope mean	0.200	0.500	0.200	0.600	1.700	0.400	0.700
FI_{cope}	0.092	0.259	0.106	0.240	0.702	0.218	0.380
FI_{cope} SD	0.010	0.020	0.030	0.030	0.040	0.030	0.030
FI_{cope} CV	0.108	0.077	0.284	0.125	0.057	0.138	0.079
All mean	1.300	1.700	1.200	1.100	1.900	1.500	1.200
All FI	0.640	0.750	0.620	0.550	0.800	0.690	0.580
All FI SD	0.020	0.020	0.050	0.030	0.040	0.030	0.030
All FI CV	0.031	0.027	0.081	0.055	0.050	0.043	0.052
Misc mean	0.060	0.050	0.370	0.070	0.110	0.080	0.220
Misc FI	0.05	0.05	0.16	0.06	0.11	0.15	0.19
Misc SD	0.01	0.02	0.08	0.02	0.04	0.02	0.02
Misc CV	0.20	0.40	0.50	0.33	0.36	0.13	0.11
N	390	321	104	287	104	202	276
				8-10 mm TL Postlarvae			
Clad mean	0.783	1.510	1.632	1.291	0.000	1.151	0.340
FI_{clad}	0.422	0.713	0.737	0.612	0.000	0.643	0.269
FI SD	0.054	0.038	0.101	0.048	0.000	0.027	0.030
FI_{clad} CV	0.129	0.053	0.137	0.079		0.042	0.111
Cope mean	0.735	1.238	0.526	0.932	1.600	0.843	1.129
FI_{cope}	0.398	0.622	0.158	0.456	0.747	0.462	0.580
FI_{cope} SD	0.054	0.041	0.084	0.049	0.050	0.028	0.033
FI_{cope} CV	0.135	0.065	0.530	0.108	0.067	0.061	0.058
All mean	1.723	2.797	1.446	2.330	1.707	2.157	1.822
All FI	0.759	0.888	1.000	0.816	0.827	0.857	0.872
All FI SD	0.047	0.026	0.000	0.038	0.044	0.020	0.023
All FI CV	0.062	0.030	0.000	0.047	0.053	0.023	0.026
Misc mean	0.200	0.050	0.210	0.110	0.110	0.160	0.350
Misc FI	0.160	0.050	0.160	0.060	0.110	0.150	0.250
Misc SD	0.040	0.020	0.080	0.020	0.040	0.020	0.030
Misc CV	0.250	0.400	0.500	0.333	0.364	0.133	0.120
N	83	143	19	103	75	314	219
N ratio	0.213	0.445	0.183	0.359	0.721	1.554	0.793
G	0.030	0.034	0.034	0.034	0.044	0.031	0.038
Z	0.199	0.096	0.160	0.196	0.055	0.065	0.044

Table 2.2.2. Choptank River White Perch postlarval (5-7 mm) feeding metrics and statistical characteristics during 1981-1985 and 1989-1990. Clad = cladoceran; mean = mean feeding score; FI = feeding incidence, SD = standard deviation, CV = coefficient of variation (proportion), and Cope = copepod.

	Year						
	1981	1982	1983	1984	1985	1989	1990
	5-7 mm TL Postlarvae						
FI_{clad}	0.753	0.588	0.430	0.206	0.038	0.686	0.415
FI SD	0.022	0.024	0.024	0.015	0.017	0.024	0.037
FI_{clad} CV	0.030	0.042	0.055	0.072	0.439	0.036	0.090
Cope mean	0.102	0.301	0.276	0.332	0.708	0.262	0.483
FI_{cope}	0.016	0.023	0.021	0.017	0.040	0.023	0.038
FI_{cope} SD	0.154	0.076	0.078	0.052	0.056	0.088	0.078
N	373	405	435	753	130	363	176

Table 2.2.3. Sample size (N, sample effort), adjustment factor (proportion = N in year t / N in year with lowest effort), and $\log_e$-transformed abundance of 6 mm Striped Basslarvae in year t ($\log_e$ N6).

Year	N	Adjustment	$\log_e$ N6
1980	151	0.87	7.24
1981	161	0.82	7.09
1982	132	1.00	6.89
1983	132	1.00	4.16
1984	154	0.86	5.93
1985	154	0.86	4.55
1986	194	0.68	2.48
1987	186	0.71	5.6
1988	201	0.66	4.47
1989	180	0.73	8.06
1990	194	0.68	6.64

Table 2.2.4. Mean densities (organisms/L) estimated from Potomac River, Head-of-Bay (HOB), and Choptank River Striped Bass ichthyoplankton surveys during 1976-1989 for copepods (includes copepodites) and cladocerans. Potomac River and Head-of-Bay estimates were from (Rutherford et al. 1997 Tables 2A and 2B). Choptank River densities were from Uphoff (1989) based on data from Wright et al. (1987). Postlarval period = 1983 (May 4-14), 1984 (May 8-18), and 1985 (April 22-30).

		Copepods	Copepods	Copepods	Cladoceran	Cladoceran	Cladoceran
Year	Survey	April-May	Last 2 weeks May	Postlarval period	April-May	Last 2 weeks May	Postlarval period
1976	Potomac	58.3	126.5		30.9	3.2	
1977	Potomac	71.2	42.9		187.1	24.0	
1980	Potomac		6.0			78.5	
1981	Potomac		16.0			95.5	
1982	Potomac		34.0			252.0	
1987	Potomac	66.9	169.4		29.2	81.1	
1988	Potomac	46.9	66.5		14.6	32.4	
1989	Potomac	31.1	23.8		6.5	4.8	
1988	HOB	52.2			0.6		
1989	HOB	67.5			3.1		
1983	Choptank			41.2			29.8
1984	Choptank			50.9			45.1
1985	Choptank			285.1			2.9

Table 2.2.5. Correlation of mean winter air temperature (Mean temp, ˚F), Choptank River March-April mean flow (March-April CFS), and copepod feeding incidence (FI_{cope}) of first-feeding (5-7 mm TL) Striped Bass postlarvae during 1981-1985 and 1989-1990. Parameter r is the correlation coefficient and P is level of significance. Winter air temperature is from long-term records for Baltimore (available from www.weather.gov/media/1wx/climate/bwitemps.pdf) and flow is at the Greensboro, MD, USGS gauging station.

Year	Mean temp	Mar-Apr CFS	FI_{cope}
1981	34.1	116	0.092
1982	31.9	200	0.259
1983	37.1	533	0.106
1984	34.5	449	0.240
1985	37.4	70	0.702
1989	36.9	348	0.218
1990	36.6	180	0.380
FI_{cope} correlation			
r	-0.55	-0.03	
P	0.20	0.95	

Table 2.2.6. Classification of daily instantaneous growth and mortality rates (G and Z, respectively) based on copepod feeding incidence or the combination of copepod and cladoceran feeding derived from 1981-1985 and 1989-1990 surveys.

	G and Z Classification	
Feeding	Low	High
Incidence	G	G
Copepod	$\leq$ 0.38	> 0.38
	Z	Z
Copepod	$\geq$ 0.38	$\leq$ 0.11
Copepod and	0.22-0.26	0.24
Cladoceran	$\geq$ 0.57	< 0.34

Appendix

Methods to estimate daily instantaneous mortality (Z) of postlarvae for each year during 1980-1990 adhered to those described in Uphoff (1989, 1992, 1993). To estimate annual postlarval mortality during 1980-1989, weekly length-abundance distributions were multiplied by the weekly proportion of fish in each 1-mm length interval (pi) of the length-frequency distribution by weekly total catch (ni). Weekly catch totals of fish in each length interval (pi • ni = Ni) were then summed across the entire sampling season (∑Ni) to create a yearly length-abundance distribution that was adjusted to lowest seasonal effort (132 trawls from the beginning of each survey to the first week of June). Each year's instantaneous daily growth rate (G) was estimated and applied as a time scale to the length-abundance distribution to estimate Z (Uphoff 1989, 1992, 1993). Estimates of G and Z in 1990 were added to estimates from Uphoff (1993) in Tables 1 and 2.

Annual estimates of G and their SEs were obtained by fitting the exponential growth model for G for each year during 1980-1990:

$$\text{Log}_e \text{ Lt} = \log_e L_0 + G \cdot (\text{Xt});$$

where Lt was mean total length at during week t, Xt = number of days from the first week to week i. By using mean length to estimate growth, it described growth of individuals under average conditions, but not growth variations that individual larvae might have exhibited under different conditions (Kaufmann1981).

The natural logarithm of abundance in the catch at each millimeter size-class was modeled against relative age for each year to estimate annual Z of postlarvae (average for all cohorts within a year) and its SE. For convenience, 6-12-mm fish were categorized as postlarvae, although late prolarvae and early juveniles were also present in this length-group (Uphoff 1989; 1993). Between 6 and 12 mm, each 1-mm length interval was assigned a relative age (in days) according to that year's G by back-calculating the time needed to grow from 6 mm to a given length. Relative age was calculated as

$$X_t = (\log_e L_m - \log_e 6) / G$$

where X_t = relative age or days past 6 mm, L_m = 1-mm length interval from 6-12 mm, and G = instantaneous growth rate.

Instantaneous mortality rates were calculated for each year as

$$\text{Log}_e N_{12} = \text{loge } N_6 - Z_t;$$

where Z = instantaneous daily mortality rate for the entire length interval, N_{12} = predicted number of 12-mm larvae at age t, and N_6 = number of 6-mm larvae at the beginning of the time interval.

Figure 2.2.1. Location of fixed sites (arrows) sampled in Choptank River during 1980-1986 and mainstem sites sampled with a stratified-random design (dots) or within Tuckahoe Creek (triangles) during 1987-1990. Inset shows location of Choptank River within Chesapeake Bay.

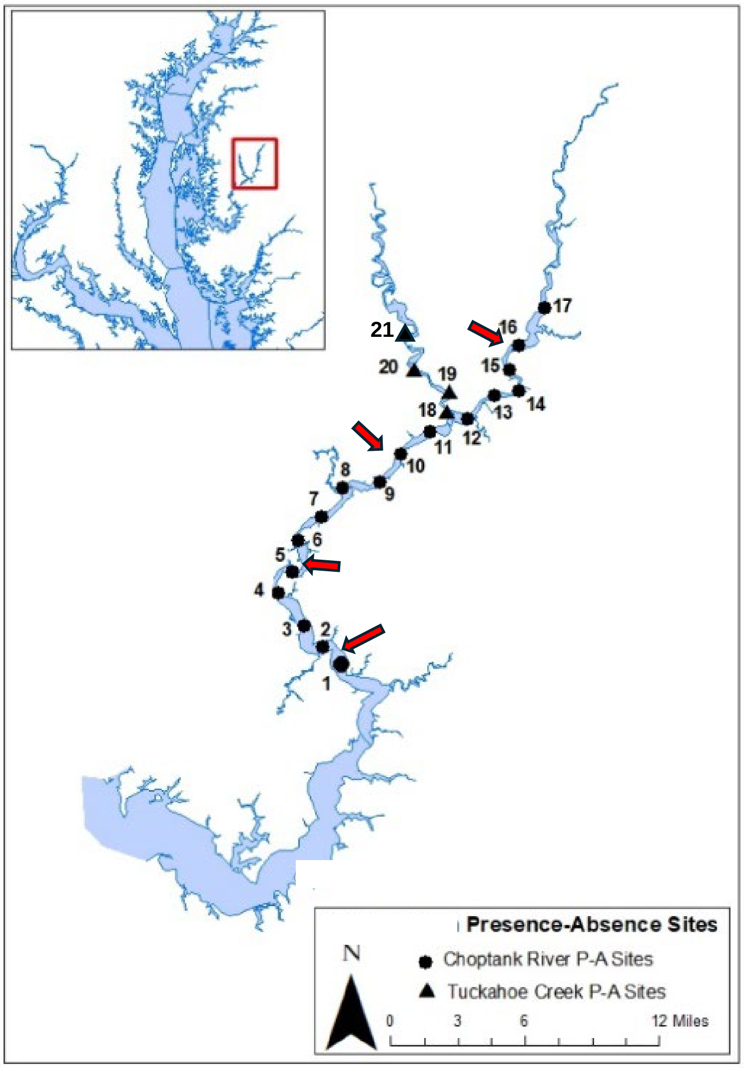


Figure 2.2.2. Feeding incidence (FI) of Striped Bass postlarvae (5-7 mm TL) for all items, cladocerans (Clad), copepods (Cope), and miscellaneous items (Misc) during 1981-1985 and 1989-1990.

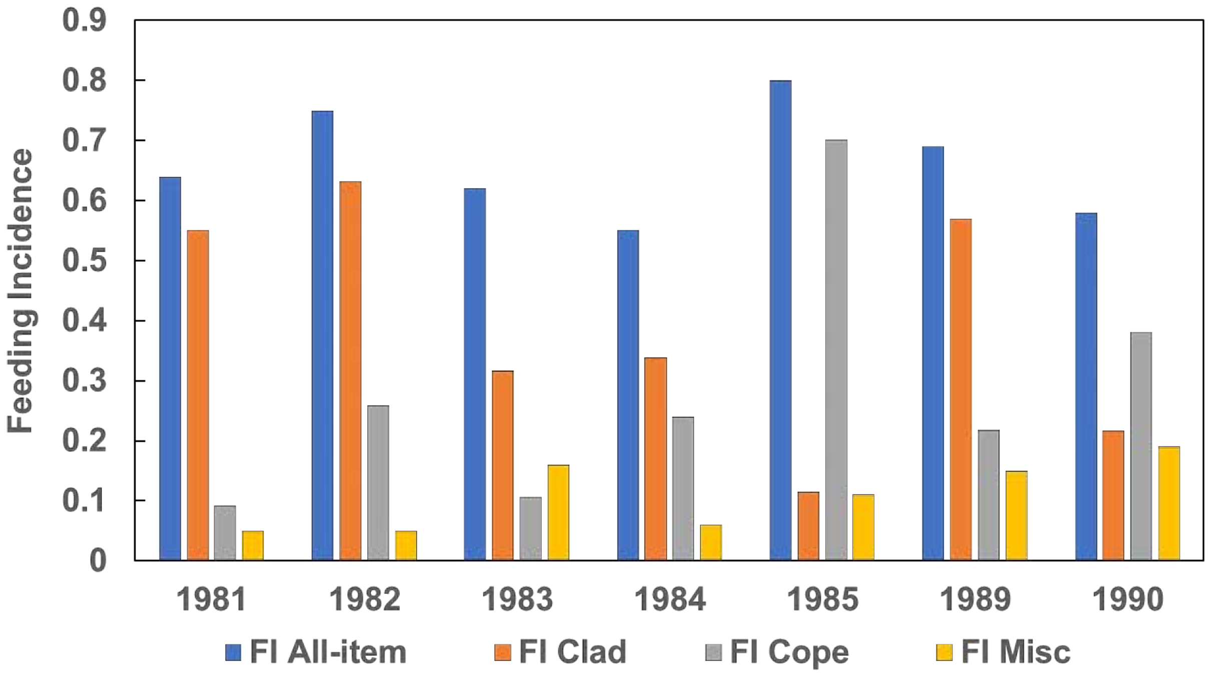


Figure 2.2.3. Relationship of 5-7 mm TL Striped Bass postlarval feeding incidence for cladocerans and copepods during 1981-1985 and 1989-1990 in Choptank River.

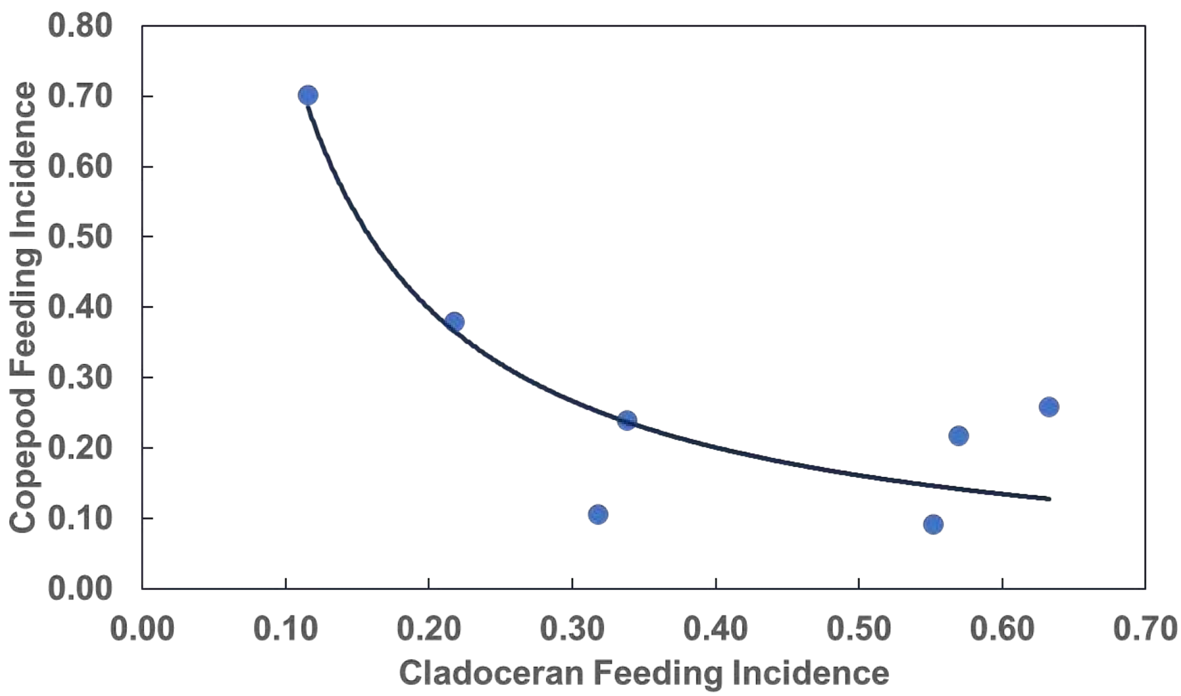


Figure 2.2.4. Annual estimates of daily instantaneous growth rate (G, diamonds) of Striped Bass postlarvae between 6 and 12 mm TL and their 95% CI (lines) during 1980-1990. A star indicates that feeding incidence was not estimated in that year.

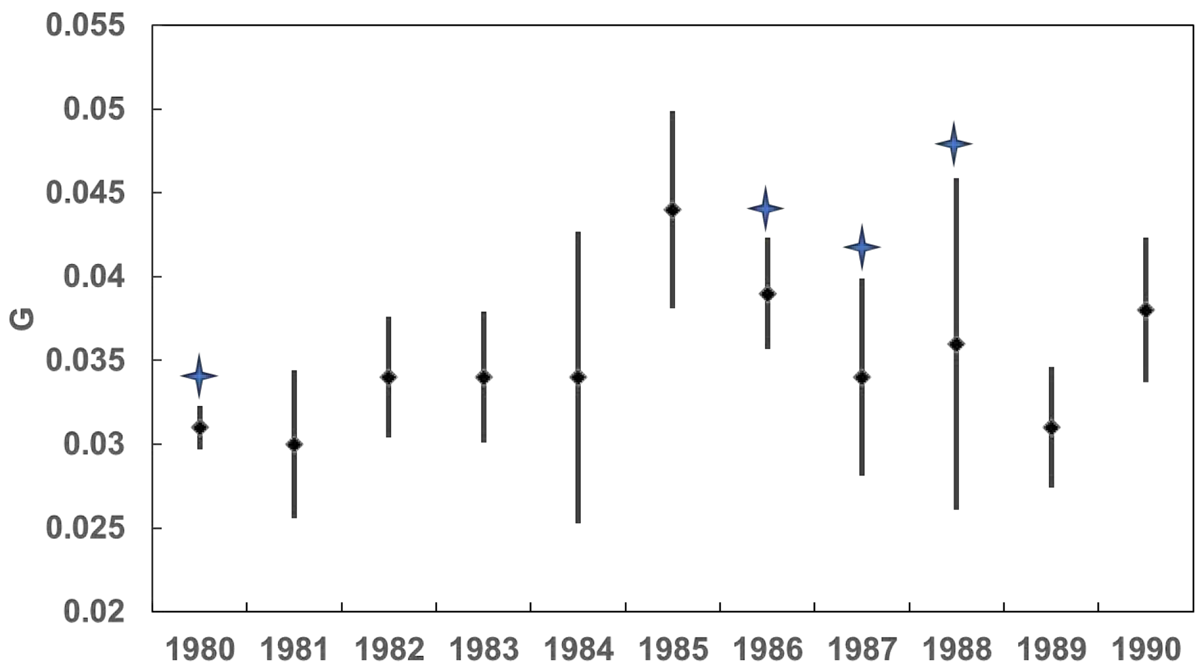


Figure 2.2.5. Annual estimates of daily instantaneous mortality rate (Z, diamonds) of Striped Bass postlarvae between 6 and 12 mm TL and their 95% CI (lines) during 1980-1990. A star indicates that feeding incidence was not estimated in that year.

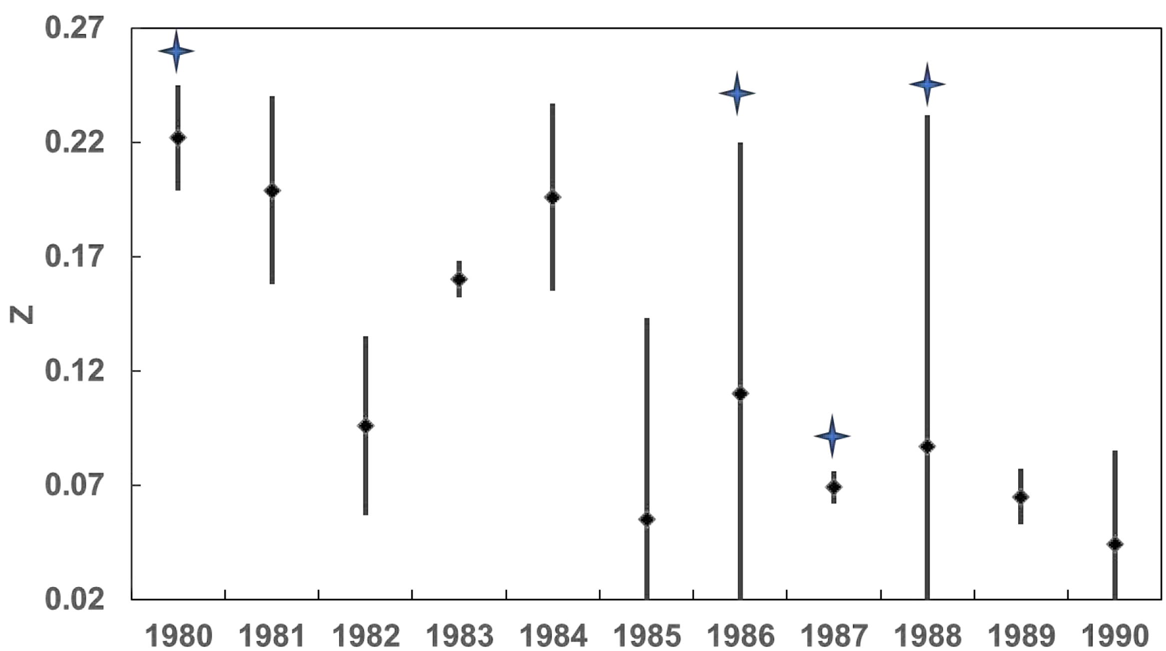


Figure 2.2.6. Relationship of feeding incidence on copepods for 5-7 mm TL Striped Bass postlarvae and Z for postlarvae during 1981-1985 and 1989-1990. Grey line indicates relationship for all years and black line indicates the relationship with 1984 (orange point) removed.

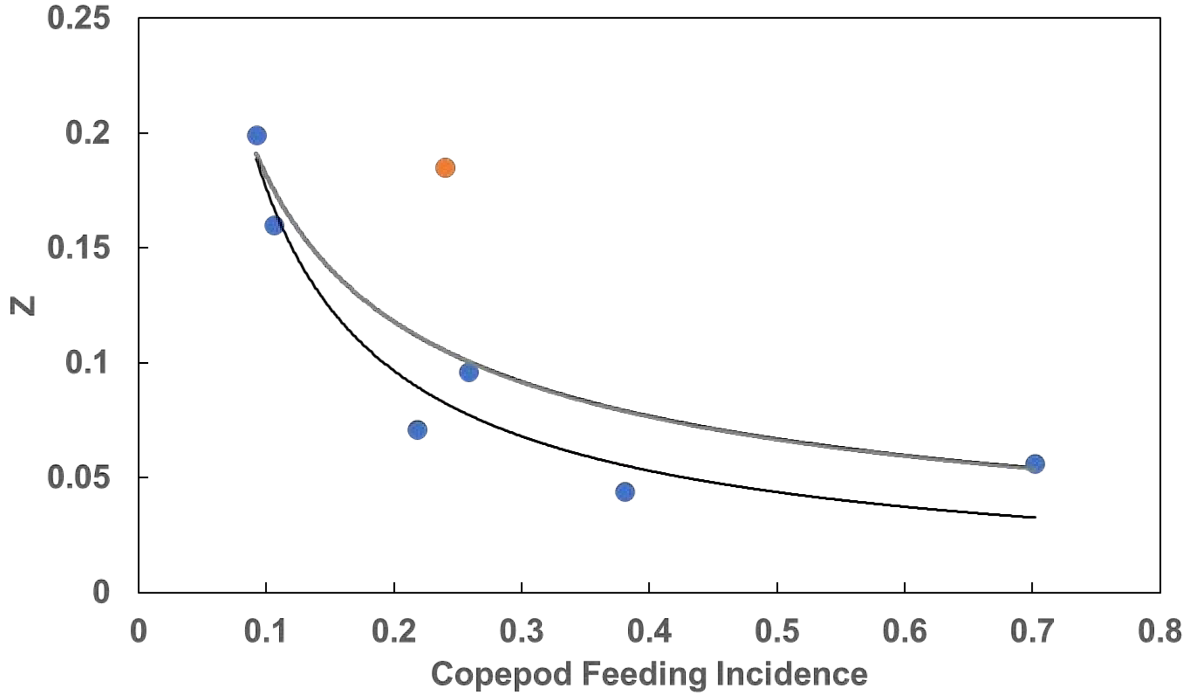


Figure 2.2.7. Relationship of the postlarvae N ratio to Z. N ratio = N of 8-10 mm TL Striped Bass postlarvae examined for feeding ÷ N of 5-7 mm TL postlarvae examined for feeding during 1981-1985 and 1989-1990. Grey line indicates relationship for all years and black line indicates the relationship with 1984 (orange point) removed.

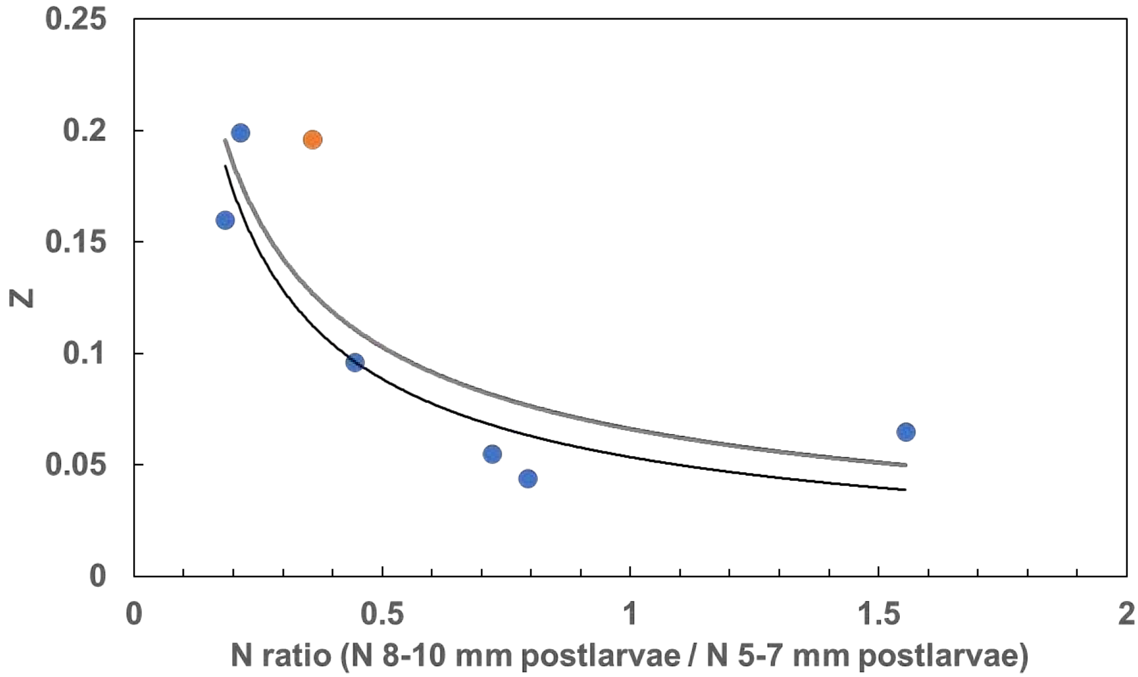


Figure 2.2.8. Feeding incidence (FI) of 5-7 mm TL Striped Bass (SB) and White Perch (WP) postlarvae on cladocerans (Clad) and copepods (Cope) in Choptank River during 1981-1985 and 1989-1990.

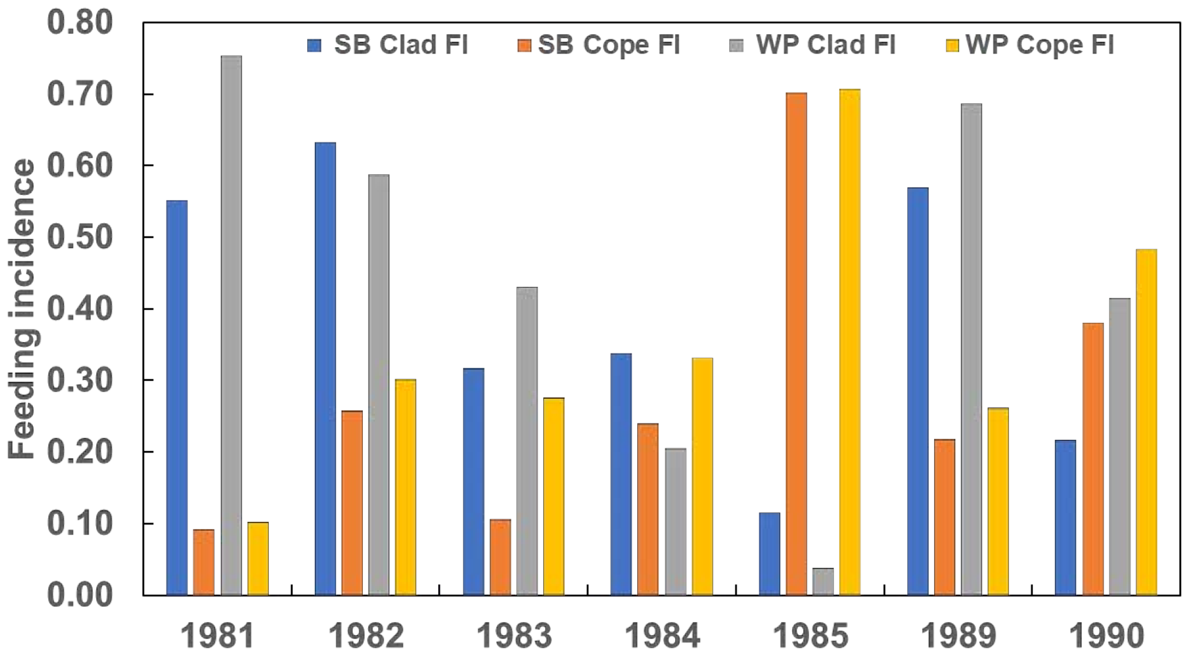
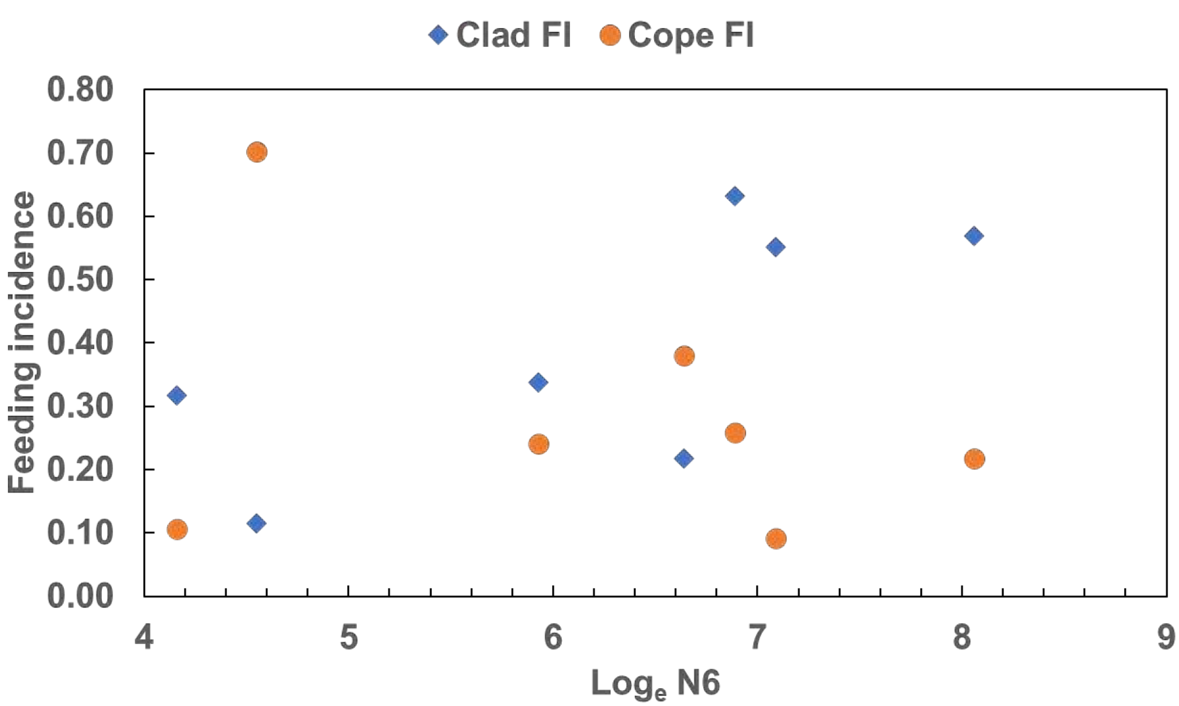


Figure 2.2.9. Bivariate plot of feeding incidence (FI) of first-feeding Striped Bass on cladocerans (Clad FI, blue diamonds) or copepods (Cope FI, orange circles) against initial abundance of 6 mm TL larvae (N6).



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

Section 2.3: Evaluation of 2023 Striped Bass Larval Feeding

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Introduction

Maryland's portion of the Chesapeake Bay contains many of the Striped Bass spawning areas along the North American Atlantic coast (Richards and Rago 1999; Maryland Sea Grant 2009). Over the last five years (2019-2023), these spawning areas have yielded poor year-classes (Durell and Weedon 2023). There are many potential hypotheses for these year-class failures that warrant investigation. This section specifically addresses the mismatch hypothesis (mistiming of first-feeding larvae and zooplankton they feed on; Cushing 1990; Houde 2008). Millette et al. (2020) hypothesize that climactic changes (warmer winters) lead to a mistiming of Striped Bass spawning with spring zooplankton blooms (Houde 2008). This results in reduced prey availability for first-feeding postlarvae that may be followed by reduced larval survival and recruitment. To begin exploration of the zooplankton mismatch hypothesis for the current decline, we collected Striped Bass and White Perch larvae from the Choptank River in 2023 to evaluate their gut contents and metrics of feeding success. We then compared these metrics to criteria developed from historic Choptank River data in Section 2.2 to assess 2023 larval Striped Bass feeding success and its potential implications for larval growth and mortality.

Methods

We evaluated available Choptank River temperature (1981-1989) and conductivity (1987-1989) distributions from years during which larvae were sampled with midwater trawls to plan when and where to sample larval Striped Bass in 2023. Temperature was measured during 1981-1989, but conductivity was not routinely recorded until 1987. Temperature distributions indicated that collection of Striped Bass larvae should commence once water temperatures were consistently greater than 14.5°C and conclude when temperatures reached 21°C, or when larvae appeared to have grown beyond sizes of interest (Uphoff et al. 2022). Historic conductivity distributions revealed that Striped Bass larvae were most prevalent (i.e., 60% of total Striped Bass larvae collected) between conductivities of 150-900 µmhos/cm (Uphoff et al. 2022); this range was divided into three increments (i.e., 150-375 µmhos/cm, 376-625 µmhos/cm, 626-900 µmhos/cm), each containing ~20% of larvae collected in the evaluated timeframe. The sampling team selected one site within each conductivity increment during each sampling date (i.e., stratified sampling, three sites per date) using observed *in situ* water conductivity measurements.

In 2023, Striped Bass larvae in the Choptank River were collected weekly via the same midwater trawl sampling procedures described in Section 2.2. At each sampling site, water temperature (°C), dissolved oxygen (mg/L), conductivity (µmhos/cm), and salinity (ppt) were measured with a YSI 556 Multiparameter Instrument (Xylem Inc., Washington, D.C., USA). Collected samples were stored in glass mason jars, preserved with 5-10% formalin, and stained with rose Bengal dye. Samples were sorted twice in the laboratory; during sorting, all fish present were separated from the organic matter collected with the sample and stored in a glass vial with 20% alcohol for later examination. Afterwards, fish were identified to the lowest practical taxonomic level and life stage using structural definitions outlined in Rogers et al. (1977) to differentiate prolarvae (i.e., larvae with yolk), postlarvae (i.e., larvae that have absorbed their yolk), and juveniles (i.e., post-metamorphic stages). These criteria were used with the historical collections (Section 2.2).

Each *Morone* spp. (i.e., Striped Bass and White Perch) with a total length (TL) ≤ 10 mm was measured to the nearest millimeter using a stereomicroscope. If the *Morone* spp. was determined to be a postlarvae between 5-10 mm TL, its digestive tract was carefully teased away with a probe or forceps to examine the gut contents. The presence or absence of any food in the gut were recorded and food items were classified as one of the following: cladocerans (typically *Bosmina longirostris*), copepods (including both adults and copepodites; typically *Eurytemora carolleeae*), or miscellaneous (debris, other organisms, unidentifiable matter). Justification for limiting our examination of feeding to target species and size class groups (i.e., 5-7 mm TL Striped Bass postlarvae, 8-10 mm TL Striped Bass postlarvae, 5-7 mm TL White Perch postlarvae) is outlined in Section 2.2.

Metrics developed in Section 2.2 were used to compare the feeding success of Striped Bass larvae in 2023 to those sampled in the 1980s. The feeding incidence (frequency of presence, FI_x) of all items (FI_{all}), cladocerans (FI_{clad}), copepods (FI_{cope}), and miscellaneous items (FI_{misc}) was calculated for 5-7 mm TL Striped Bass, 8-10 mm TL Striped Bass, and 5-7 mm TL White Perch as:

$$FI_x = N_{x\,present}/N_{total}$$

where x is a given item category (all items, cladocerans, copepods, or miscellaneous), $N_{x\ present}$ is the number of qualifying individuals with x food item present and N_{total} is the total number of qualifying individuals. The standard deviation (SD) of FI_x, estimated from the normal distribution approximation of the binomial distribution, was calculated as:

$$SD = \left[\left(FI_x \cdot (1 - FI_x)\right)/N_{total}\right]^{0.5}; \text{ (Ott 1977).}$$

Ninety-five percent confidence intervals were constructed as:

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

Additionally, the N_{ratio}, developed from the historic 1980s data as a potential proxy estimate of survival, was calculated as:

$$N_{ratio} = N_{8-10}/N_{5-7}$$

where N_{8-10} and N_{5-7} are the total numbers of 8-10 mm TL and 5-7 mm TL Striped Bass postlarvae collected in 2023, respectively.

Feeding incidence metrics derived from 2023 sampling for Striped Bass were compared among and between the target species and size class groups. Estimated FI_x for copepods and cladocerans were compared to criteria developed for successful and unsuccessful feeding from the 1980s (See Table 2.2.6). The 2023 N_{ratio} was compared to 1980s estimates to judge whether Z was high or low. In the historic dataset, high estimates of Z occurred when the N_{ratio} was 0.36 or less and low Z occurred when it was 0.44 or more (See Section 2.2). Criteria were not available for White Perch postlarvae.

Results

In 2023, Choptank River water temperatures reached the threshold for sampling on April 10th and sampling for feeding larvae commenced later that same week. We collected nine samples from the Choptank River across three dates (13th, 19th, and 25th) in April 2023 (Table 2.3.1; Figure 2.3.1); sampling concluded when the likelihood of catching first-feeding larvae diminished in the weeks following spawning.

Extremely dry conditions resulted in higher than anticipated conductivity and our sampling scheme had to be altered. Observed water conductivity during sampling events ranged from 246-2,580 µmhos/cm. Conductivity measurements below 700 µmhos/cm occurred during four samples and exceeded 1,000 µmhos/cm during five samples (Table 2.3.1); due to these highly variable conductivity observations, we were unable to collect samples from each of the target conductivity ranges, as originally intended.

A total of 31 5-7 mm TL Striped Bass postlarvae, 39 8-10 mm TL Striped Bass postlarvae, and 67 5-7 mm TL White Perch postlarvae were collected in 2023. First-feeding (5-7 mm TL) Striped Bass were present in six of nine samples, while 8-10 mm TL Striped Bass were present in three of nine; 5-7 mm TL White Perch were present in all but one sample (Table 2.3.1). A full list of species and life stages collected during each sampling event is available in Appendix A (Table A.2.3.1).

For 5-7 mm TL Striped Bass larvae, FI_{all} was estimated as 0.806 ± 0.071 (FI_x ± SD), FI_{clad} was estimated as 0.258 ± 0.078, and FI_{cope} was estimated as 0.645 ± 0.086 (Table 2.3.2; Figure 2.3.2). For 8-10 mm TL Striped Bass larvae, FI_{all} was estimated as 0.872 ± 0.054, FI_{clad} was estimated as 0.512 ± 0.080, and FI_{cope} was estimated as 0.746 ± 0.070. For 5-7 mm TL White Perch, FI_{all} was estimated as 0.896 ± 0.037, FI_{clad} was estimated as 0.552 ± 0.061, and FI_{cope} was estimated at 0.702 ± 0.056. Miscellaneous food items were only observed in the gut contents of 5-7 mm TL White Perch; FI_{misc} for this group was estimated as 0.029 ± 0.021. Cladocerans and copepods were prevalent in larval gut contents across target postlarval species and size classes. Based on 95% CI overlap, we determined: (1) FI_{all} was similar among species and size groups, (2) FI_{cope} was similar among species and size groups, and (3) FI_{clad} was likely lower (small overlap of CIs) for 5-7 mm TL Striped Bass than the other species and size groups (Table 2.3.2; Figure 2.3.2).

Based on criteria developed from 1980s Choptank River feeding analysis, FI_{cope} of 5-7 mm TL Striped Bass postlarvae (0.642) was well above the successful feeding criterion (0.38). The N_{ratio} in 2023 (1.26) was above the criterion (≥ 0.44) and indicated low Z.

Discussion

When assessed against the feeding success and mortality criteria developed in Section 2.2, the FI_{cope} values and the N_{ratio} survival proxy estimated in 2023 suggested that feeding on copepods was successful and mortality was low. FI_{cope} estimates for both first-feeding (i.e., 5-7 mm TL) and larger (i.e., 8-10 mm TL) Striped Bass postlarvae were higher in 2023 than all historic years (1981-1985, 1987-1990) except 1985. In Section 2.2, we established that high estimates of Z occurred when FI_{cope} < 0.38; FI_{cope} of 2023 first-feeding larvae was 1.7-fold higher than this threshold. The N_{ratio} in 2023 was 2.9-fold higher than the minimum N_{ratio} threshold (N_{ratio} ≥ 0.44) associated with low estimates of Z and 3.5-fold higher than the maximum N_{ratio}

threshold ($N_{ratio} \leq 0.36$) associated with high estimates of Z in the historic dataset. The 2023 N_{ratio} was higher than the N_{ratio} calculated from all but one year of historical data (1989, $N_{ratio} = 1.554$).

Estimates of FI_{cope} in 2023 were similar among species and size class groups. Estimates of FI_{cope} for 5-7 mm TL Striped Bass in 2023 ranged from 1.7 to 7-fold higher than all historic FI_{cope} values (except 1985, $FI_{cope} = 0.702$). Similarly, the FI_{cope} estimates for 8-10 mm TL Striped Bass and 5-7 mm TL White Perch were also higher in 2023 than all but 1985. It should be noted that, while estimated FI_{cope} for these groups in 2023 was lower than in 1985 (1985 8-10 mm TL Striped Bass $FI_{cope} = 0.747$, 1985 5-7 mm TL White Perch $FI_{cope} = 0.708$), 2023 estimates were within one standard deviation of 1985 estimates.

Estimates of FI_{clad} in 2023 were lower for all size groups than most estimates from historic years. Estimated FI_{clad} for 5-7 mm TL White Perch in 2023 was greater than that for 5-7 mm TL Striped Bass, but similar to 8-10 mm TL Striped Bass.

When compared to 1981-1990 estimates (Section 2.2), FI_{all} for 5-7 mm TL Striped Bass postlarvae in 2023 (0.81) was higher than in historic years (1981-1985, 1989-1990). Similarly, the 2023 FI_{all} estimate for 8-10 mm TL Striped Bass postlarvae was higher than four of seven annual historic estimates. It was also higher than estimates from the Upper Bay in 2007-2008 (0.63 and 0.55, respectively; Shideler and Houde 2014) and within range of 2001-2003 estimates (>0.50 to ~0.90; Martino and Houde 2010). FI_{misc} for both Striped Bass groups was less than calculated in all historic years, as miscellaneous food items were not detected in Striped Bass guts in 2023.

High and variable conductivity observed in the Choptank River in 2023 (see Section 2.1) reflected low freshwater flow. In March 2023, the Choptank River produced decade-low discharge (102.5 cfs, 2014-2023 mean = 271.45 cfs). April 2023 discharge (226.0 cfs) was above the ten-year average (2014-2023 mean = 216.3 cfs) but, was largely driven by a storm event that occurred after sampling concluded. Prior to that event, April 2023 discharge was also at a decade-low (100.8 cfs, USGS 2024). The highest FI_{cope} in historic years occurred in 1985 (0.702), which was also a year of low spring discharge (March-April mean flow = 70 cfs) that resulted in extremely high and variable conductivity (Uphoff 1989). Low freshwater discharge in Chesapeake Bay subestuaries has been associated with low copepod densities, poor feeding success, and poor Striped Bass larval survival in some studies (e.g., North and Houde 2001, 2003; Kimmel and Rogers 2006; Martino and Houde 2010).

The Choptank River 2023 Striped Bass year-class was the fourth smallest since the time-series began in 1957 (Durell and Weedon 2023), although feeding success and postlarval survival were classified as good. Working backward, we are left with low spawning stock and poor egg hatching success as possible explanations for the poor year-class. Year-class success of Striped Bass in the Chesapeake Bay is a function of spawning stock size and behavior, water temperatures encountered by eggs and prolarvae, and mortality of early postlarvae (Uphoff 1989, 1992, 1993, 2023; Secor and Houde 1995; Rutherford et al. 1997; Maryland Sea Grant 2009). Estimated *Ep* was not low in 2023 and there was little chance that the combination of spawning stock and behavior (i.e., dispersion in space and time) indicated by this index fell below a threshold where they were low enough to negatively impact recruitment (see Section 2.1). Based on observations, spawning began on March 30 (first egg collected); spawning intensity increased through April 7 (peak spawning); moderated through the 12th; and nearly ceased after the 17th. Water temperatures were low through April 4th, warmed very rapidly by the 7th, and were at the upper limit for eggs and prolarvae by April 17th. This indicated a very short window for successful egg hatching and prolarval development.

Given the low 2023 Striped Bass juvenile index, modest sample sizes for first-feeding (N = 31) and more advanced Striped Bass postlarvae (N = 39) were not particularly surprising. White Perch postlarval FI_{cope} (N = 67) was statistically indistinguishable from the two Striped Bass postlarval size classes, and the combined picture across sizes and species indicated that copepods were abundant enough for successful feeding.

Analysis in Section 2.2 indicated that FI_{cope} could be negatively influenced by larval abundance. Striped Bass larval abundance was low, and the 2023 Choptank River White Perch juvenile index (indicating low larval abundance) was the lowest of the time-series, so competition at these stages may have been low. The poor 2023 Choptank River White Perch juvenile index and general high correlation of White Perch and Striped Bass juvenile indices argue for some common habitat factor as the primary determinant of year-class success (Uphoff 2023).

As discussed in Section 2.2, comparisons between historic and current N_{ratio} values may be influenced by differences in sampling methodology. Given the differences in sampling intensity between historic surveys and the 2023 feeding survey, conclusions from these comparisons should be considered conditional.

Though definitive conclusions should not be drawn from a single year, 2023 collections of gut contents confirmed that a study comparing historic and modern Striped Bass larval feeding success was feasible. It is our hope that continuing data collection across multiple years with varying environmental conditions will lend further insight into the connections among larval Striped Bass survival, feeding success, and climate variability and, ultimately, the validity of the mismatch hypothesis.

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Table 2.3.1. Summary of sampling by station and date. Cond = conductivity, µS/cm. N_{SB} (5-7 mm) is the number of 5-7 mm TL Striped Bass postlarvae collected per sampling event, N_{SB} (8-10 mm) is the number of 8-10 mm TL Striped Bass postlarvae collected per sampling event, N_{WP} (5-7 mm) is the number of 5-7 mm TL White Perch postlarvae collected per sampling event, and N_{other} is the number of other fish (i.e., outside the target postlarval species and size classes) collected per sampling event.

Station	Date	Cond (µS/cm)	N_{SB} (5-7 mm)	N_{SB} (8-10 mm)	N_{WP} (5-7 mm)	N_{other}
14	04/13/23	667	7	0	4	42
11	04/13/23	1685	10	0	27	143
9	04/13/23	1991	0	0	2	16
16	04/19/23	246	5	1	10	343
13	04/19/23	519	0	0	1	99
10	04/19/23	1387	0	0	5	36
11	04/25/23	2580	1	0	0	43
13	04/25/23	1581	3	14	2	79
16	04/25/23	438	5	24	16	188

Table 2.3.2. Feeding incidences and summary statistics for Striped Bass and White Perch postlarvae, collected from the Choptank River in 2023, by size class. FI_{all} is the feeding incidence of all items combined, FI_{clad} is the feeding incidence of cladocerans, FI_{cope} is the feeding incidence of copepods, FI_{misc} is the feeding incidence of miscellaneous items, N is the sample size. SD is the standard deviation and CV is the coefficient of variation (proportion). N_{ratio} is the ratio of the sample sizes for 5-7 mm TL and 8-10 mm TL Striped Bass postlarvae size classes, a proxy for survival.

	5-7 mm TL Striped Bass Postlarvae		
Parameter	Value	SD	CV
FI_{all}	0.806	0.071	0.088
FI_{clad}	0.258	0.079	0.305
FI_{cope}	0.645	0.086	0.133
FI_{misc}	0.000	0.000	NA
N	31		
	8-10 mm TL Striped Bass Postlarvae		
Parameter	Value	SD	CV
FI_{all}	0.872	0.054	0.061
FI_{clad}	0.513	0.080	0.156
FI_{cope}	0.743	0.069	0.094
FI_{misc}	0.000	0.000	NA
N	39		
N_{ratio}	1.258		
	5-7 mm TL White Perch Postlarvae		
Parameter	Value	SD	CV
FI_{all}	0.896	0.037	0.042
FI_{clad}	0.552	0.061	0.110
FI_{cope}	0.701	0.056	0.079
FI_{misc}	0.029	0.021	0.696
N	67		

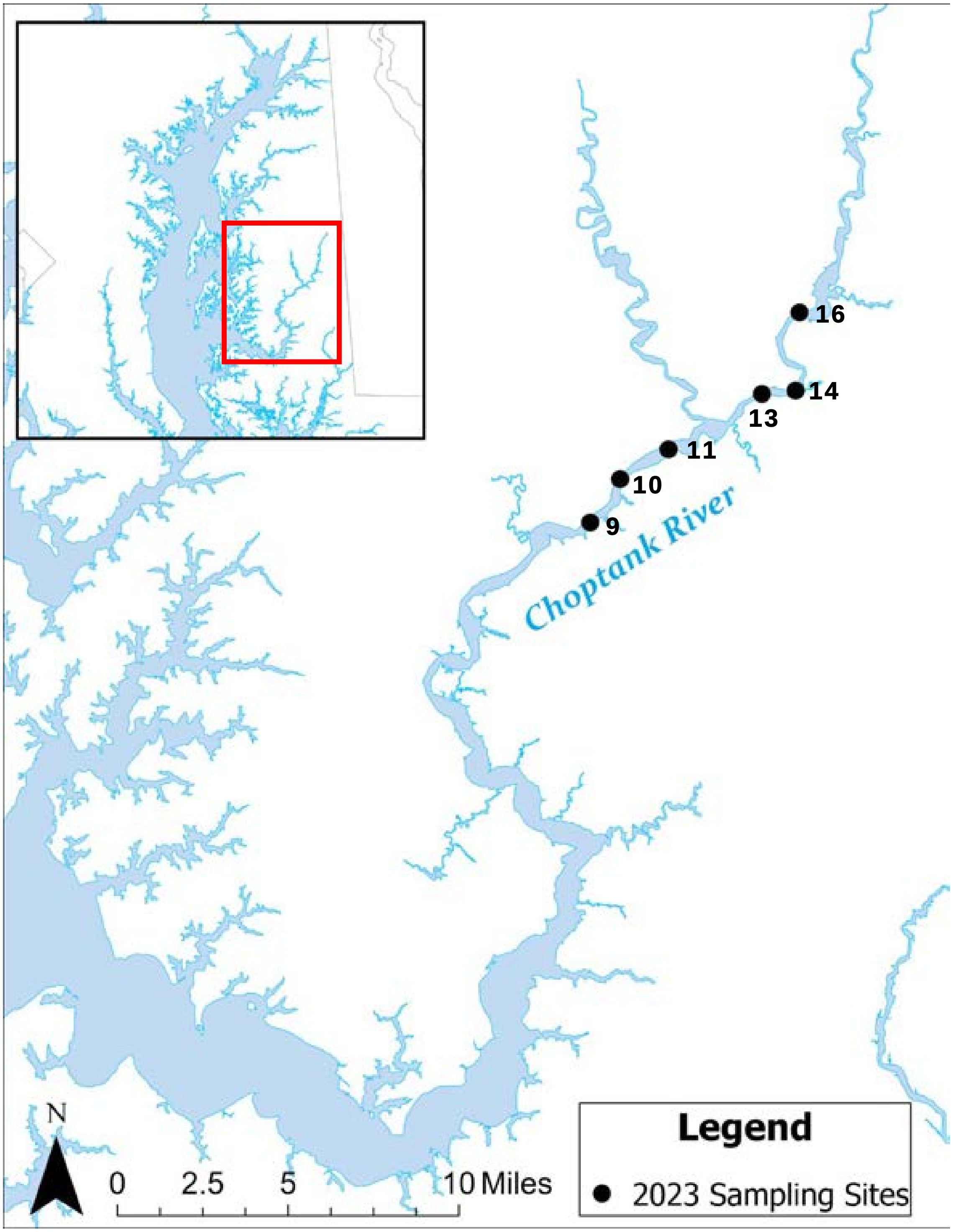

Figure 2.3.1. Locations of 2023 midwater trawl sampling sites (black dots) in the Choptank River. Map inset shows location of Choptank River within Chesapeake Bay.

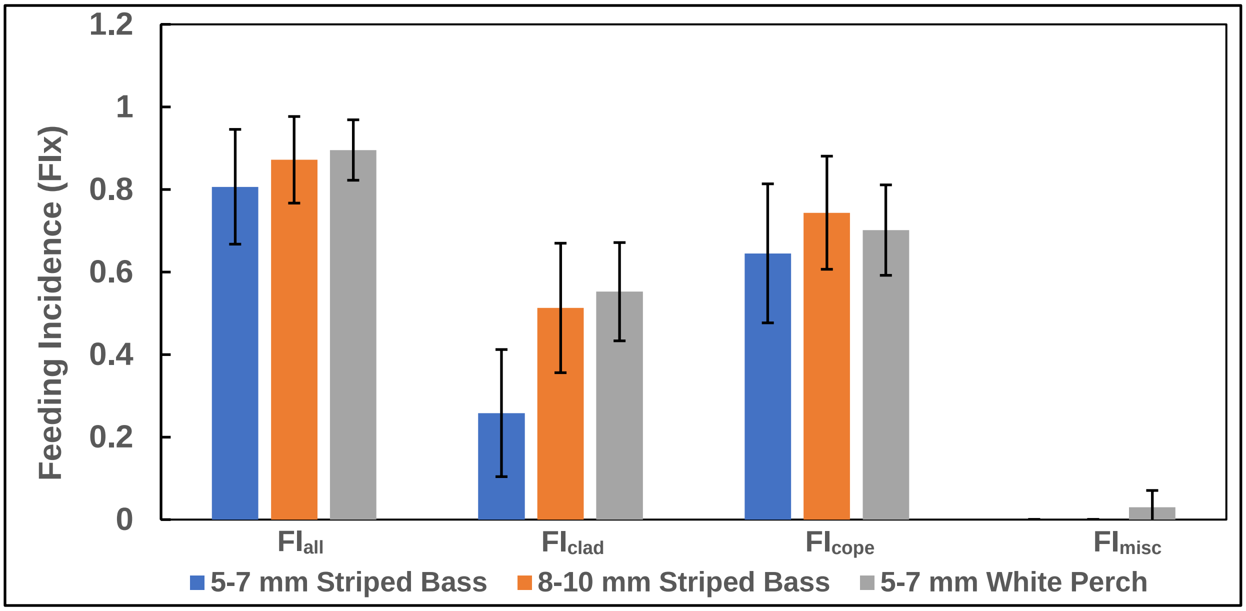

Figure 2.3.2. Bar plot depicting the feeding incidences of all items (FI_{all}), cladocerans (FI_{clad}), copepods (FI_{cope}), and miscellaneous items (FI_{misc}) for 5-7 mm TL Striped Bass (N = 31), 8-10 mm TL Striped Bass (N = 39), and 5-7 mm TL White Perch postlarvae (N = 67) collected from the Choptank River in 2023. Error bars represent the 95% confidence intervals for calculated FI_x values.

Appendix A

Table A.2.3.1. Counts of all fish species and life stages collected at each station during each sampling date. “Pro.” is prolarvae, “Post.” is postlarvae, and “Juv.” is juvenile. “Morone spp.” refers to any Striped Bass or White Perch that could not be confidently identified as either species; “Unknown” is any individual that could not be confidently identified and assigned to a group. Numbers beneath the dates are the stations sampled that day.

Species Common Name	Life	04/13/23			04/19/23			04/25/23		
(*Genus species*)	Stage	9	11	14	10	13	16	11	13	16
Atlantic Croaker	Pro.	0	0	0	0	0	0	0	0	0
M. undulatus	Post.	0	0	0	0	0	0	0	0	0
	Juv.	8	20	0	0	0	0	7	3	0
Atlantic Menhaden	Pro.	0	0	0	0	0	0	0	0	0
B. tyrannus	Post.	0	0	0	0	0	0	0	0	0
	Juv.	0	0	0	0	6	37	0	1	0
Bay Anchovy	Pro.	0	0	0	0	0	0	0	0	0
A. mitchilli	Post.	0	0	0	0	0	0	0	0	0
	Juv.	0	0	0	0	1	5	32	0	0
Blue Catfish	Pro.	0	0	0	0	0	0	0	0	0
I. furcatus	Post.	0	0	0	0	0	0	0	0	0
	Juv.	0	0	0	0	0	0	1	0	0
Gizzard Shad	Pro.	0	0	0	8	11	28	0	7	37
D. cepedianum	Post.	0	0	0	0	0	0	0	0	0
	Juv.	0	0	0	0	0	0	0	0	0
Herring/Shad spp.	Pro.	0	0	4	0	2	1	0	26	4
(Order: Clupeiformes)	Post.	0	14	7	15	66	262	2	13	35
	Juv.	0	1	0	0	0	0	0	0	0
Morone spp.	Pro.	2	0	1	0	0	0	0	0	0
	Post.	0	4	1	1	1	0	0	14	29
	Juv.	0	0	0	0	0	0	0	0	0
Striped Bass	Pro.	1	11	4	2	2	0	1	6	14
M. saxatilis	Post.	0	10	7	0	0	6	1	17	29
	Juv.	0	0	0	0	0	0	0	0	0
White Perch	Pro.	1	0	0	1	2	0	0	1	9
M. americana	Post.	3	54	14	14	1	18	0	10	75
	Juv.	0	0	0	0	0	0	0	0	0
Yellow Perch	Pro.	0	0	0	0	0	0	0	0	1
P. flavescens	Post.	2	58	15	0	7	1	0	0	0
	Juv.	0	0	0	0	0	0	0	0	0
Unknown	Pro.	0	0	0	0	0	0	0	0	0
	Post.	1	8	0	0	1	0	0	0	0
	Juv.	0	0	0	0	0	0	0	0	0

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

Section 3 - Estuarine Fish Summer Habitat and Community Sampling

Alexis Park, Carrie Hoover, Tyler Fowler, and Jim Uphoff

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-offs 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). Compacted soils at both the top- and sub-soil levels can be found in both altered (farmed lands, harvested forests, urbanized lands) and unaltered lands (forests, wetlands) have shown a decrease in permeability and an increase in degradation due to changes in climate and weather, causing a rise in run-off of sediment, pesticides, and nutrients (Batey 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. (2011a) 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 2023, 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 2003–2023 (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, which has been sampled consistently since 2006; a tributary of the Choptank River located in Talbot County (Table 3-1; Figure 3-1). We returned to four previously sampled systems, Mattawoman Creek, a tidal-fresh tributary of the Potomac River, previously sampled from 1989 to 2016 and in 2022; Miles River, a mesohaline tributary of the Eastern Bay located on the Eastern Shore, previously sampled from 2003 to 2005 and in 2020; Northeast River, a tidal-fresh subestuary located at the Head-of-Bay, previously sampled from 2007 to 2017 and in 2022; and South River, a mesohaline subestuary located mid-Bay on the Western Shore, previously sampled from 2003 to 2005 and in 2022 (Table 3-1; 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 2023.

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 were used for analyses of data from subestuaries sampled during 2003–2023 (Table 3-2). Maryland Department of Planning (MD DOP) only has structure estimates available through 2020; 2021–2023 estimates are extracted from MD DOP property sales data; 2023 data is only through September 2023. 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 (Table 3-3) for 1973–2010 use MD DOP data. Land use estimates for 2013 and 2018 were estimated using a conversion factor and Chesapeake Conservancy (high resolution) data to correspond to previous MD DOP land use estimates, allowing for a continuous data set. 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. Conversion factors were implemented for each land use type within each subestuary.

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 2003–2023 surveys 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 seine sites were taken at the seine starting point on the beach. We focused on using previously sampled historical sites 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 2023. The number of total trawl and seine 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 randomize 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, seines hauls could not be made because of permanent obstructions, dense submerged aquatic vegetation (SAV), or lack of shore/beach availability. Bottom trawl and beach seine sampling were 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 annual median of all bottom salinity measurements for all years available to determine salinity class of each subestuary. Water quality parameters were recorded at all stations for every individual sampling event in 2023. Temperature (ºC), DO (mg/L), conductivity (µS/cm), salinity (parts per thousand; ppt = ‰), and pH were recorded at the surface, middle, and bottom of the water column at the trawl sites depending on channel depth, and just below the water's surface (0.5 m) at 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 cm at each trawl site; Secchi depths were not recorded at seine sites. 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. The 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 measurements) and for bottom DO measurements alone. The percentage of DO measurements not meeting target or threshold conditions were termed "violations", but the term did not have a regulatory meaning. The percentage of DO measurements that met or fell below the 5 mg/L target (Vtarget) or fell at or below the 3 mg/L threshold (Vthreshold) were estimated as:

$$Vtarget = (Ntarget / Ntotal) \bullet 100;$$

and

$$Vthreshold = (Nthreshold / Ntotal) \bullet 100;$$

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

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 watersheds of subestuaries sampled. Bottom DO measurements during 2003–2023 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 2023 and plotted by year sampled. We examined plots of Secchi depths, SAV coverage, DO, pH, and salinity within subestuaries by year. Measurements of pH were not made prior to 2006; during 2019, pH readings taken from 8/5/2019 to 8/15/2019 were omitted to pH probe issues.

An ANOVA examined differences in mean bottom DO among stations in 2023 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 2023 subestuary and used to detect how annual station DO compared with the time-series median.

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, and 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, Eastern 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 common 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 2023).

Striped Bass and Yellow Perch were separated into two age categories, juveniles (JUV) 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, described below; 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 a 3.1-m trawl GM for each year during 2003–2008 and 2022–2023 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. Seining was not conducted in Mattawoman Creek after 2005 due to high SAV density that caused the seine to roll up.

Bottom trawl sites were generally located in the channel, adjacent to haul seine sites. Bottom trawls and beach seines were conducted one right after the other in no particular order 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 the top 90% of the catch. The GM of seine and trawl catches was the back-transformed mean of loge-transformed catches (Ricker 1975; Hubert and Fabrizio 2007; Durell and Weedon 2023). 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; Durell and Weedon 2023). 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 (1989–2023) was available for beach seine samples than the 4.9 m bottom trawl (2003–2023) 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 2003–2023 bottom trawl GMs by year for all species combined to find where the 2023 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 2023) 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:

$$PSD = ((N \geq L\ Quality) / (N \geq L\ Stock)) \bullet 100;$$

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 2022). 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 2022). We substituted the total number of small adults plus harvestable length White Perch for stock length with to estimate a modified PSD since we did not measure small adults. The stock length category minimum for White Perch is130 mm TL (20–26% of the world record length TL;Gablehouse et al 1984) ; 125 mm TL is used as the length cut-off for White Perch in Chesapeake Bay recruitment and length-frequency assessments (Piavis and Webb 2022). 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 2022). 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 2023.

The diversity within a subestuary may be affected by land use and environmental conditions (Radinger et al 2016). We used a percent similarity index to evaluate variation in finfish species composition among bottom trawl stations by year for subestuaries sampled in 2023 (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, Pjklm was calculated as

$$\Sigma \text{minimum (pji, pki, pli pmi);}$$

where pji, pki, pli, and pmi 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 2023. White Perch juveniles and adults were consistently abundant and represented the only gamefish that routinely appeared in samples.

Results and Discussion

2023 Sampling Locations – 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 (Table 3-3). We returned to sample Mattawoman Creek in 2022 and 2023 after a six-year hiatus, and C/ha was estimated at 1.03. All historical trawl stations were sampled in 2023 (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 and 2023). This monitoring was an important 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 and 2023 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 development.

Miles River (Figure 3-1) a mesohaline tributary located mid-Bay, was previously sampled in 2003–2005 when C/ha ranged from 0.23 to 0.24 (Table 3-3). In 2020, we returned to sample the Miles River; C/ha slightly increased to 0.26. We once again returned to sample Miles River in 2023 due to concerns within the local fishing community about decreasing observations of adult White Perch during 2022. Development remained at 0.26 C/ha. Historical trawl stations were sampled, and two of the three historical seine stations were sampled; seine station 01 was not sampled because a beach to land the seine was not available (Figure 3-2; see Uphoff et al. 2023 for additional analyses of mid-Bay subestuaries).

Northeast River (Figure 3-1), a tidal-fresh subestuary located at the Head-of-Bay, was sampled previously from 2007 to 2017 and in 2022 (Uphoff et al. 2018 and 2023). The Northeast River watershed reached the target level of rural development in 1995 (0.31 C/ha; Table 3-3). Development in 2007 was 0.44 C/ha and increased during the sampling period to 0.49 C/ha in 2017. In 2022 and 2023, when we returned to sample the Northeast River, development had increased to 0.52 structures per hectare (Table 3-3). Historical sampling stations were used in 2023. Seine station 02 was not available in 2022 but became available again in 2023 (Figure 3-2).

South River (Figure 3-1) sampling took place during 2003–2005 when C/ha ranged from 1.24 to 1.27 (Table 3-3). In 1966, South River passed the target for rural development (0.31 C/ha) and in 1989, surpassed the threshold for suburban development (0.84 C/ha). In 2022 and 2023, we returned to sample South River and C/ha was 1.43 (Table 3-3). Historical sampling stations were used in 2022 and 2023 (Figure 3-2). Seine station 02 was difficult to sample effectively with a beach seine in 2022 due to the high amount of the benthic algae *Ulva* present. We dropped seine station 02 altogether in 2023 due to sampling difficulty (see Uphoff et al. 2023 for additional analyses of mid-Bay subestuaries).

Tred Avon River (0.79 C/ha in 2023; 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 (Table 3-3). Our sampling began in 2006 with 0.69 structures per hectare, 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 (Table 3-3). 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 and 2023 for additional analyses of Choptank subestuaries).

2023 Water Quality Summary – Table 3-4 provides summary statistics for surface and bottom water quality for each tributary and subestuary sampled in 2023. Salinity in Miles, South, and Tred Avon Rivers were within mesohaline bounds in 2023. Mattawoman Creek and Northeast River salinity ranged from the tidal-fresh (<0.5 ‰) to oligohaline (0.5–5.0 ‰) during 2023 due to a multi-year trend of below-average rainfall and flow (Ruiz-Barradas 2023; NOAA NCBO 2023). Mattawoman Creek's median bottom salinity for 2023 was 0.7 ‰, while the median bottom salinity for all years sampled was 0.2 ‰; Northeast River, median bottom salinity for 2023 was 0.3 ‰ and for all years sampled was 0.1 ‰. Bottom temperatures ranged from 21.0°C to 30.3°C in 2023 for all subestuaries. Surface temperatures ranged from 19.4°C to 30.9°C during the 2023 survey (Table 3-4).

In 2023, Mattawoman Creek was the only subestuary sampled that did not have any DO target or threshold violations (Table 3-5). The other four subestuaries sampled in 2023 had bottom DO readings less than the target level (<5.0 mg/L): Miles River, 75%; Northeast River, 25%; South River, 100%; and Tred Avon River, 46%. Thirty-seven percent of all DO measurements (surface, middle, and bottom) from Miles River were below the target level of 5 mg/L; 9%, Northeast River; 53%, South River; and 24%, Tred Avon River. In 2023, threshold level (<3.0 mg/L) violations occurred in four of the five subestuaries sampled. Miles River, Northeast River, South River, and Tred Avon River had 13%, 4%, 71%, and 21% of bottom DO estimates below the threshold level, respectively (Table 3-5). South River had the lowest mean bottom DO (2.30 mg/L; Table 3-6); it was only above the threshold level in 2004. Tred Avon River annual mean bottom DO was usually above 5.0 mg/L during 2006–2007 and 2009–2017 but has been below 5.0 mg/L since 2018 (Table 3-6). Miles River mean annual bottom DO was the only year above target level in 2004; remaining years fell between the target and threshold levels. Mattawoman Creek annual mean bottom DO fluctuated without an apparent trend during 2003–2016 and 2022–2023. Northeast River annual mean bottom DO fluctuated without trend during 2007–2017; however, in 2022 and 2023, the annual mean bottom DO declined with 2023 having the lowest annual mean bottom DO of all sampling years (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 was detected between bottom DO and C/ha in mesohaline subestuaries ($r = -0.61$, $P < 0.0001$; Table 3-7). Remaining correlations were not strong enough to be of interest (Table 3-7). 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). 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=94) were over twice as high as oligohaline (N=33) or tidal-fresh subestuaries (N=40), 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. (2011a) determined that the odds of adult and juvenile White Perch, juvenile Striped Bass, Spot, and Blue Crabs being present in shore zone seine samples from mesohaline subestuaries were not influenced by development, but odds of these target species being present in bottom channel trawl samples were negatively influenced by development through its negative influence on DO.

The extent of bottom channel habitat that can be occupied does not appear to diminish due to low DO with increasing watershed development in tidal-fresh and oligohaline subestuaries. However, more localized, or episodic habitat issues 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 shallow water SAV beds and the impact on target finfish could not be estimated. Episodic ammonia toxicity 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, oligohaline Middle River 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 = -0.61; P < 0.0001); urban cover with agriculture was negative and considered moderate, bordering on strong (r = -0.74; P < 0.0001). Weak correlations were found for wetland cover with C/ha (r = -0.21; P = 0.04) and wetland cover with urban (r = -0.27; P = 0.01). The remaining pairings of categories were poorly correlated. Correlations among land uses and year were weak, although annual forest land use estimates and year bordered on moderate (r = -0.46; P <0.0001; 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 29.0% 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 four divisions: Chester River, Choptank River, Miles River, and Wye River. Choptank River included Broad and Harris Creeks and Tred Avon Rivers, which ranged from 42.1% to 50.1% agricultural coverage. Chester River included the Chester River, Corsica River, and Langford Creek, which ranged from 59.9% to 71.6% agricultural coverage. Agricultural coverage in Miles River ranged from 48.7% to 53.7% and in Wye River ranged from 64.3% to 67.7% (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; $r^2 = 0.56$; $P < 0.0001$; N = 23; Table 3-9) and negative and weak for the eastern shore (slope = -0.03; SE = 0.01; $r^2 = 0.13$; $P = 0.004$; N = 63; Table 3-9). Predictions of median bottom DO for mesohaline western shore subestuaries rose from 0.76 mg/L at 2.6% agricultural coverage to 5.19 mg/L at 38.6% (Figure 3-5). Predictions of median bottom DO for mesohaline eastern shore subestuaries started at 5.34 mg/L at 42.6% agricultural coverage, increased to 5.42 mg/L at 50.1%, and then decreased to 4.28 mg/L at 71.6%. A quadratic regression of median bottom DO versus agricultural coverage (eastern and western regions combined) described the moderately strong relationship of median bottom DO with agricultural coverage ($r^2 = 0.55$, $P < 0.001$; 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 (≥65%) may have been negatively biased. Additionally, a piecewise regression indicated a similar trend in the separate regions that both the linear and quadratic regressions observed.

Water Quality for Mattawoman Creek, Miles River, Northeast River, South River, and Tred Avon River – Percentages of land in agriculture ranged from 10.1% to 48.7%; forest, 21.7–52.8%; wetlands, 0.1–1.1%; and urban, 23.4–49.7%. (Table 3-11; Figure 3-1). Water comprised a larger fraction of the area in Tred Avon River (34.1%) than in the other subestuaries sampled in 2023: Mattawoman Creek (3.5%), Miles River (28.6%), Northeast River (10.9%), and South River (16.6%; Table 3-11).

The western shore of Maryland has substantially more development than the eastern shore 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 2023 (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 1985; and Northeast River, in 1995. Miles River was the only subestuary sampled in 2023 that remains under the rural development target (C/ha = 0.26). 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 2023. 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; Figure 3-6).

During 2023, the mesohaline subestuaries we sampled experienced more DO violations than tidal-fresh ones. In 2023, 46% of Tred Avon River bottom DO samples were below the target and 21% were below the threshold (Table 3-12; Figure 3-7). During 2006–2023, 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 21% (2023) in Tred Avon River. All DO measurements below target level ranged from 6% (2009) to 30% (2019) in Tred Avon River. South River in 2023 had all bottom DO measurements under the target for the second year in a row and 71% were below the threshold. During 2003–2005 and 2022, South River's percent of bottom DO measurements below target ranged from 75% (2003) to 100% (2022 and 2023) and bottom DO measurements below threshold ranged from 25% (2004) to 83% (2022). All DO measurements below target level ranged from 28% (2003) to 53% (2023) 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. Miles River in 2023 had 75% of bottom DO measurements under the target and 13% were below the threshold. During 2003–2005 and 2020, the percentage of bottom DO measurements below target ranged from 48% (2004) to 81% (2020) and bottom DO measurements below threshold ranged from 0% (2004) to 48% (2020) in the Miles River. All DO measurements below target level ranged from 26% (2004) to 46% (2003) in Miles River (Table 3-12; Figure 3-7).

Tidal-fresh Northeast River during 2023 had 29% of bottom DO samples below the target and 4% were below the threshold (Table 3-12; Figure 3-7). During 2007–2023, the percentage of bottom DO measurements below target ranged from 0% (2010 and 2016) to 33% (2011) and bottom DO only fell below threshold in three different years, 2011 (13%), 2015 (4%), and 2023 (4%). The percentage of all DO measurements below target ranged from 0% (2016) to 14% (2011) in Northeast River. Mattawoman Creek in 2023 did not have bottom DO below the target or threshold. During 2003–2016 and 2022–2023, Mattawoman Creek bottom DO measurements below target ranged from 0% (2003–2005, 2009–2010, and 2013–2016, and 2023) 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, 2013–2016, and 2023) to 14% (2011) (Table 3-12; Figure 3-7).

The median and range of bottom DO measurements for the time-series of each subestuary sampled during 2023 are depicted in Figure 3-8. Median DO estimates were above the target level in the tidal-fresh subestuaries' time-series and below the target in mesohaline subestuaries. South River's medians were all below the threshold. Miles River annual median DO fell between the target and threshold. Tred Avon River medians were above the target (Figure 3-8).

Chesapeake Bay Program monitoring stations were in or near subestuaries that we sampled during 2023. Station EE2.1 (CBP 2024), located at the mouth of the Choptank River, showed less variation in annual summer (July–September) median bottom DO during 2018–2023 than 1989–2017 and there was little indication of a trend over time (Figure 3-9). Annual summer median bottom DO range from 4.3 mg/L (2005 and 2017) to 6.9 mg/L (2003). In 2023, the summer median bottom DO of 5.7 mg/L was slightly above the time-series mean of 5.9 mg/L. Monitoring station ET1.1 (CBP 2024), located at the mouth of the Northeast River, showed moderate levels of annual variation with little indication of a trend over time. Annual summer median bottom DO range from 5.4 mg/L (2023) to 10.0 mg/L (1990) from 1989 to 2023. In 2023, the median summer bottom DO of 5.4 mg/L was below the time-series median of 8.0 mg/L. Monitoring station EE1.1 (CBP 2024), located in Eastern Bay near the mouth of Miles River, exhibited annual summer median bottom DO ranging from 0 mg/L (1998, 2004, 2011, 2016, 2017, and 2020) to 6.2 mg/L (2009) during 1989 to 2023. Monitoring station WT8.1 (CBP 2024) was located within the South River near our station 3. Annual median summer bottom DO was often below the threshold and varied at a low level without trend during 1989–2023. The greatest annual summer median bottom DO was 4.7 mg/L was in 2023 and the lowest was 0.1 mg/L in 1998. In 2023, the summer median bottom DO was greater than the time-series median of 1.3 mg/L (Figure 3-8). Monitoring station MAT0016 (CBP 2024; M. Trice, MD DNR, personal communication), located within Mattawoman Creek's channel between our trawl stations 03 and 04 (Figure 3-2), recorded annual summer median bottom DO ranging from 5.3

mg/L (2010) to 10.3 mg/L (1998; Figure 3-8). In 2023, the summer median bottom DO was 7.6 mg/L just slightly below the time-series median of 7.7 mg/L (Figure 3-9).

Trends in bottom DO among stations within the subestuaries sampled were generally coherent within a year across their time-series except for station 01 in Tred Avon River (Figure 3-10). Tred Avon River's station 01 exhibited a marked decline since 2007 while the remaining stations in that subestuary varied nearer the long-term median. Deterioration of DO at the uppermost station at Easton (station 01) since 2009 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.

There was little improvement or deterioration of water clarity over time in the subestuaries sampled during 2023 except for Mattawoman Creek (Figure 3-11). Mattawoman Creek underwent a transition in the early 2000s from a phytoplankton dominated subestuary to extensive SAV coverage and Secchi depths improved concurrently. Annual median Secchi depths in Mattawoman Creek ranged from 0.5–1.3 m; Miles River, 0.45–0.65 m; Northeast River, 0.3–0.5 m; South River, 0.7–0.9 m; and Tred Avon River, 0.4–0.9 m (Figure 3-11).

Estimates of SAV coverage (% of water area) through 2022 were reviewed for subestuaries sampled in 2023 (VIMS 2023). Coverage estimates for 2023 were not available at the time of this report (Figure 3-12). Coverage estimates of SAV in Mattawoman Creek ranged from 4.7% to 37.0% from 1989 to 2022. Coverage estimates were above the time-series median (28.9%) in 2002–2011, 2015–2017, and 2019–2022. Coverage estimates have been on the upswing since 2019. The greatest SAV coverage was in 2008 (42.3%) and the lowest coverage was in 1995 (1.3%). Coverage in 2001 was only partially mapped.

Miles River was combined in the Eastern Bay region SAV coverage estimates that included Wye River, Cox Creek, and Prospect Bay (VIMS 2023; Figure 3-12). Coverage estimates of SAV in Miles River ranged from 14.1% to 3.9% from 1989 to 2022 (Figure 3-12). Coverage estimates were above the time-series median (9.9%) in 1989, 1993–1999, 2001–2003, 2011, and 2016–2019. Coverage estimates have been on the downswing since 2017. The greatest SAV coverage was in 1999 (34.1%) and the lowest coverage was in 2013 (0.33%); SAV was absent in 2000 (Figure 3-12).

Coverage estimates of SAV in Northeast River ranged from 0% to 2.3% from 1989 to 2022 (VIMS 2023; Figure 3-12). Coverage estimates were above the time-series median (1.7%) in 2002, 2004–2011, and 2014–2022. Greatest SAV coverage was in 2009 (5.5%) and the lowest coverage was in 1997 (0.28%); SAV was absent in 1989–1993 and in 2001 (Figure 3-12).

Submerged aquatic vegetation was absent from South River for 15 of 34 years; SAV coverage was below 1% in the remaining years (VIMS 2023; 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 2023; Figure 3-12). The SAV coverage increased substantially from 3.24% in 2013 to 22.4% in 2017 and then declined to 9.4% in 2021; lowest SAV coverage was recorded in 1991 at 0.5%. Coverage estimates were above the time-series median (9.4%) in 1993–1999, 2001–2002, 2004, and 2014–2022. Coverage estimates were only partially mapped in 2018, but still above the time-series median (Figure 3-12).

Measurements of pH were not available prior to 2006. Annual median pH measurements in Mattawoman Creek ranged from 7.3 to 8.6 for all years sampled; Miles River, 7.5–7.6; Northeast River, 7.6–8.6; South River, 7.3–7.4; and Tred Avon River, 7.4–8.0 (Figure 3-13).

Below average rainfall and freshwater flow since 2020 resulted in high salinities during the summer of 2023 (Ruiz-Barradas 2023; NOAA NCBO 2024; Figure 3-14). Most of the salinity measurements in 2023 remained in the appropriate salinity classification assigned to the subestuaries sampled, Mattawoman Creek salinity measurements fluctuated between tidal-fresh and oligohaline classes in 2023 (Figure 3-14). Annual median salinity measurements in Mattawoman Creek ranged from 0.1 to 1.0‰ for all years sampled; Miles River, 9.1–14.3‰; Northeast River, 0.1–0.3‰; South River, 7.5–12.4‰; and Tred Avon River, 7.5–13.2‰. High summer precipitations in 2011, 2018, and 2019 lowered salinity in Tred Avon River considerably.

2023 Finfish Community Summary – Geometric mean catch per seine haul ranged from 76 to 298 among the four subestuaries sampled during 2023 (Table 3-13). Miles River had 12 seine samples at 2 stations; Northeast River, 24 seine samples at 4 stations; South River, 18 seine samples at 3 stations; and Tred Avon River, 24 seine samples at 4 stations (Table 3-13; Figure 3-2). The number of samples varied among subestuaries due to the number of stations established and availability; Miles River lost seine station 01 because a beach was not available. South River seine station 02 had too much *Ulva* present to seine. Extensive SAV has precluded seining in Mattawoman Creek since 2003. In 2022, Northeast River seine station 02 was roped off and could not be sampled in 2022 but was available to sample again in 2023.

Northeast River seine catch species richness (N=26) was greater than in Miles (N=22), South (N=23), and Tred Avon (N=22) Rivers (Table 3-13). Miles River did not meet the minimum criterion of 15 seine samples so the low richness partially or wholly reflected sampling effort. Logistic regression of seine catches of four finfish species (three were among our target species) common to Chesapeake Bay were not influenced by the level of development in mesohaline subestuaries (Uphoff et al. 2011a).

A plot of species richness in seine samples against C/ha during 1989–2023 did not suggest a strong relationship in tidal-fresh, oligohaline, or mesohaline subestuaries (Figure 3-15); points were omitted (based on past analysis of rarefication curves) if the seine sample effort was less than 15 samples. 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=78; C/ha range = 0.07–2.68) than tidal-fresh and oligohaline subestuaries (N=25 and 26, respectively; Figure 3-15).

A total of 31,822 fish representing 45 species were captured by beach seines in 2023 (Table 3-13). Five species comprised 90% of the total fish caught in 2023, including (from greatest to least) Atlantic Menhaden (target species), Atlantic Silverside, Striped Killifish, Gizzard Shad (target species), and White Perch (juvenile and adult; target species). Target species collected in 2023 were Atlantic Menhaden, Bay Anchovy, Blueback Herring, Gizzard Shad, Spottail Shiner, Spot, and White Perch. Different target species comprised the top 90% of catches in the four subestuaries that could be seined. Atlantic Menhaden in the top 90% of catches in all four subestuaries sampled; Bay Anchovy and White Perch were in the top 90% of catches in two subestuaries; and Blueback Herring, Gizzard Shad, Spot, and Spottail Shiner were in the top 90% of catches in one subestuary (Table 3-13).

Geometric mean (GM) of bottom trawl catches of all finfish were between 30 and 91 during 2023 (Table 3-14). Mattawoman Creek had the greatest GM (91), and South River had the lowest GM (30; Table 3-14). All subestuaries had 4 trawl stations and 6 tows were made at each site during sampling season (Figure 3-2). Bottom trawl stations sampled had been sampled in previous surveys; Mattawoman Creek trawl station 01 was moved slightly downstream due to increased siltation and lack of a defined channel.

The number of species captured by trawl during 2023 ranged from 13 to 19 (Table 3-14). A plot of species richness in trawl samples against C/ha (all subestuaries during 2003–2023; points were omitted if effort was less than 15 samples) did not indicate a relationship of development and number of species for oligohaline subestuaries (species richness ranging from 12 to 26; Figure 3-16). 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 = -3.07; SE = 0.75; r^2 = 0.17; P = 0.0001; N = 84), 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 3-16). Linear regression analysis of tidal-fresh subestuaries did indicate a weak positive relationship of development and trawl species richness (slope=3.56; SE=1.39; r^2=0.15; P=0.015; N=39), 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.

Trawl GMs declined with C/ha during 2003–2023 in mesohaline subestuaries and negative threshold response was suggested at C/ha between 0.8 and 1.2 (Figure 3-17); this change reflects the change to consistent low DO conditions in mesohaline bottom channel waters in Figure 3-3. A linear regression analysis of mesohaline subestuaries sampled during 2003–2023 indicated a weak negative linear relationship of development (C/ha) on trawl GM catches (slope=-59.89; SE=16.38; r^2=0.13; P=0.0004; N=93, respectively); a linear model may not be a good candidate for describing threshold changes. Median trawl GM calculated for mesohaline subestuaries with below target watershed development (C/ha=0.31) was 104 (N=50); it was 54 (N=43) with watershed development between the target and threshold (C/ha=0.84); and 9 (N=10) when development was greater than threshold. Trawl GMs did not exhibit an obvious decline with C/ha in tidal-fresh and oligohaline subestuaries (Figure 3-17).

A total of 11,158 fish and 35 fish species were captured by bottom trawl during 2023 (Table 3-14). Five species comprised 90% of the total catch for 2023 (from greatest to least): White Perch (juvenile and adult), Spot, Bay Anchovy, Atlantic Croaker, and Hogchoker; the first three were target species. Mattawoman Creek target species within the top 90% consisted of White Perch, Spot, and Spottail Shiner. Northeast River target species within the top 90% were White Perch and Spot. Bay Anchovy and Spot were target species target species within the top 90% in South River, Miles River, and Tred Avon River. Atlantic Croaker, Weakfish, Brown Bullhead and Hogchoker were non-target species present in the top 90% of the total catch (Table 3-14).

Additional analysis was conducted to see if the increased SAV coverage in Mattawoman Creek negatively affected trawl catchability. Linear regressions of SAV coverage estimates and bottom trawl geometric means (GM) of finfish from 2003 to 2022 for Mattawoman and Piscataway Creeks were compared (Figure 3-18). Estimated SAV coverages were similar over time for the two subestuaries. If catchability was affected by SAV coverage, we would see a similar relationship for each system. Results indicated a relationship was unlikely between SAV coverage and catchability in Piscataway Creek; coverage ranged from 21% to 78% (slope = -0.99; SE = 1.86; r^2 = 0.05; P = 0.61; N = 8). A strong negative relationship was detected in

Mattawoman Creek (slope = -14.91; SE = 3.56; $r^2 = 0.57$; $P = 0.0011$; N = 15) where SAV coverage was between 23% and 46%. SAV coverage affected catches differently but changes in catches over the years in Mattawoman Creek were unlikely due to SAV's effect on catchability. Changes were more likely to reflect changes in subestuary habitat conditions and overall year-class success of species caught.

Fish Communities in Mattawoman Creek, Miles River, Northeast River, South River, and Tred Avon River – Three mesohaline subestuaries were sampled in 2023. Miles River bottom trawl GM in 2023 for all species ranked 48th out of 93 mesohaline subestuaries sampled since 2003; Tred Avon River, 58th; and South River, 66th (Table 3-15). Two tidal-fresh subestuaries were sampled in 2023. Mattawoman Creek bottom trawl GM ranked 26th out of 40 tidal-fresh subestuaries sampled since 2003, and Northeast River, ranked 28th (Table 3-15). To some extent, these rankings depend on whether the varying time-series contained good year-classes of anadromous, estuarine, and tidal-fresh forage target species; time-series consisting of more than 5 years generally 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 2023 and their 95% CIs were plotted; Mattawoman Creek, ranged from 7 (2009) to 582 (2014); Miles River, ranged from 315 (2003) to 10 (2020); Northeast River, ranged from 59 (2022) to 306 (2010); South River, ranged from 10 (2022) to 30 (2023); and Tred Avon River, ranged from 13 (2008) to 253 (2010; Figure 3-19). Noticeable declines in estimated GMs appeared after 2014 for subestuaries sampled (Figure 3-19).

Species composition for bottom trawl catches for all years combined indicated that in the mesohaline subestuaries sampled in 2023, Bay Anchovy was the most abundant species in South River and Tred Avon River; the second most abundant species in Miles River (Figure 3-20); Bay Anchovy made up greater than 50% of species present in Tred Avon River during 2006, 2009, and 2012–2017, and in South River during 2004 and 2022 (Figure 3-21). Four target species, Bay Anchovy, Spot, Striped Bass (juvenile and adult), and White Perch (juvenile and adult), make up 90% of species composition in trawl catches in Miles River for all years combined; there were 25 other species collected (Figure 3-20). Four target species (Bay Anchovy, Spot, Striped Bass, and White Perch) plus Atlantic Croaker made up 90% of species in trawl catches in South River for all years combined; there were 17 other species. Tred Avon River had five species for all years combined, Bay Anchovy (target), Hogchoker, Spot (target), Weakfish, and White Perch(target); there were 33 other species (Figure 3-20). In these comparisons of samples with all sampling years combined, the number of other species may be a positive function of how many years were sampled.

White Perch (juvenile and adult) were the most abundant species found in the tidal-fresh subestuaries, Mattawoman Creek and Northeast River, for all years combined (Figure 3-20); 13 out of 16 sampling years White Perch made up greater than 50% of species present in Mattawoman Creek and all 13 sampling years in Northeast River (Figure 3-21). Four species consisting of three target species, Bay Anchovy, Spottail Shiner, and White Perch, plus Tessellated Darter, make up the top 90% of species composition in trawl catches in Mattawoman Creek for all years combined; there were 36 other species present (Figure 3-20). Bay Anchovy (target), Brown Bullhead, and White Perch (target) make up the top 90% of species composition in trawl catches in Northeast River for all years combined; there were 31 other species present (Figure 3-20).

Annual Mattawoman Creek bottom trawl catches were largely comprised of White Perch (juvenile and adult) during all sampling years; Alewife, Bay Anchovy, Blueback Herring, Bluegill, Channel Catfish, Pumpkinseed, Eastern Silvery Minnow, Spot, Spottail Shiner, Striped Bass (juvenile and adult), and Tessellated Darter have been the only other species to be featured in the top 90% of species caught since 2003 (Figure 3-21). In 2023, White Perch (80.6%), Spot (6.6%) and Spottail Shiner (3.7%) were among the top 90% of fish caught and there were 16 other species (Figure 3-21).

Miles River bottom trawl catches in 2023 showed an increase in prevalence of Spot (72.0%) compared to 2005 and 2020, when they were near 44% of the catch, and 2003-2004 when they fell in the other species category (Figure 3-21). Bay Anchovy were abundant in 2003 (35.6%) and 2004 (47.8%), only to disappear from the top 90% in 2005 and reappear in 2020 (24.0%) and 2023 (15.0%). White Perch (juvenile and adult) were present in the top 90% of species sampled in 2003-2005 and 2020 but none were not caught in 2023. There were 12 other species present in 2023. During 2003 to 2023, Atlantic Croaker, Bay Anchovy, Hogchoker, Striped Bass (juvenile and adult), and White Perch have all been present in the top 90% of species; Atlantic Croaker was only present in 2023 (6.3%; Figure 3-21).

Northeast River bottom trawl catches were largely comprised of White Perch (juvenile and adult) during all sampling years; Bay Anchovy, Brown Bullhead, Gizzard Shad, and Spot have been the only other species to be featured in the top 90% of species caught (Figure 3-21). In 2023, White Perch (71.3%), Spot (15.6%) and Brown Bullhead (3.2%) were among the top 90% of fish caught; and other species consisted of 16 species. Invasive Blue Catfish were present in Northeast River (Figure 3-21).

South River, sampled in 2003–2005 and 2022–2023, only had one species in common in the top 90% among all sampling years, Bay Anchovy (Figure 3-21). Spot were present in the top 90% of species caught for three of the five years sampled, 2005 (72.7%), 2022 (41.2%), and 2023 (23.5%); Striped Bass (juvenile and adult; 17.2% in 2003 and 5.2% in 2004); and White Perch (juvenile and adult; 48.0% in 2003 and 23.5% in 2004) were present in the top 90% in two of the five years sampled; Atlantic Croaker (36.7% in 2023), Brown Bullhead (11.1% in 2004), and Weakfish (7.6% in 2023) were present for only one year of sampling. In 2023, South River trawl catches were comprised mostly of Atlantic Croaker (36.7%), followed by Bay Anchovy (27.6%), Spot (23.5%), Weakfish (7.6%), and there were nine other species (Figure 3-21).

Tred Avon River trawl samples usually had four species in the top 90% of species caught; 2010 and 2016 had the lowest frequency of species in the top 90% (2) and 2018 had the highest (6; Figure 3-21). 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 were at a noticeably reduced abundance in 2019, 2021, and 2022; Bay Anchovy abundance increased to pre-2018 levels in 2023 (38.4%). Spot increased each year from 2018 (11.3%) to 2022 (66.2%) but declined in 2023 (9.5%). White Perch have declined since 2019 and they were not in the top 90% of species caught in 2023 (6.6%). Hogchoker presence remained steady since 2018 in Tred Avon River. In 2023, only two other species made the top 90% of species in Tred Avon River, Atlantic Croaker (7.0%) and Weakfish (6.9%); Atlantic Croaker were also observed in 2017 (7.5%) and 2020 (6.5%); Weakfish were more commonly observed over the time-series and ranged from 4.2% (2009) to 6.9% (2015 and 2023; Figure 3-21).

The percent similarity among trawl stations 01–04 in Mattawoman Creek was at its lowest in 2009 (13.7%) concurrent with a large, continuous decline in abundance during 2003-2009 (Figure 3-22). Similarity indices varied within a high range during remaining years (Figure 3-22).

Miles River had its greatest percent similarity in 2023 (75.5%) among bottom trawl stations 01–04; least percent similarity occurred in 2004 (14.2%; Figure 3-22). Similarity indices were 55.5%, 54.3%, and 41.6% in 2003, 2005, and 2020. (Figure 3-22).

Percent similarity among Northeast River stations was at its lowest in 2009 (22.5%) and greatest in 2023 (80.8%; Figure 3-22). Similarity was between (29.4–40.3%) during 2007–2011, in a mid-range during 2013–2016 (45.3–51.7%) and was above 60% in 2012 (70.2%) and 2017 (66.4%). In 2023, percent similarity had increased substantially from 2022 (30.1%; Figure 3-22).

South River exhibited a narrower range in similarity indices than other subestuaries, 20.9% to 58.7% (Figure 3-22). Years sampled were low (N=4), but not much different than Miles River (N=5).

The percent similarity in Tred Avon River was at its lowest in 2019 (8.4%) but continued to increase until 2022 (51.3%) and then fell in 2023 (29.7%; Figure 3-22). During 2006 and 2018–2021, the similarity index was below 50%, reflecting possible impacts of heavy rainfall and subsequent low salinity during 2018–2019 on fish community composition. Tred Avon River was the only subestuary to have a decline in percent similarity in 2023 (Figure 3-22).

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 (Uphoff et al. 2018).

Prevalent species in bottom trawl in mesohaline subestuaries shifted annually during 2003–2023 (Figure 3-23). Bay Anchovy and White Perch (juvenile and adult) 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% completely in 2022–2023. In the last three years, 2020–2023, Spot has been the dominant species, followed by Bay Anchovy. Hogchoker have been steadily present in the top 90% since 2020. Atlantic Croaker returned to the top 90% in 2023, last observed in the top 90% in 2017 and 2020 (Figure 3-23).

Adult White Perch trawl GMs for Mattawoman Creek were at or above the time-series median GM (9) in 2003–2007, 2016, and 2023; adult White Perch GM was 19 in 2023 (Figure 3-24). 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), 2016, and 2023. Juvenile White Perch GMs were at or above the time-series median GM (44) in 2003–2005, 2011, and 2013–2016; juvenile White Perch GM was 36 in 2023. Juvenile White Perch trawl GMs ranged from 2 (2009) to 256 (2014); the only spike in juvenile White Perch abundance occurred in 2014 (Figure 3-24). Mattawoman Creek is considered more of a nursery area for juvenile White Perch, indicated by the larger abundance of juveniles present and low proportional stock densities (PSDs; see below).

Miles River adult White Perch trawl GMs were at or above the time-series median GM (2) in 2003–2005; adult White Perch GM was 0 in 2023 (Figure 3-24). The greatest adult White Perch GM in Miles River was in 2004 (11) and the lowest was in 2023. Juvenile White Perch GMs were at or above the time-series median GM (1) in 2003–2005; both 2022 and 2023 had

GMs of 0. Juvenile White Perch trawl GMs declined substantially between 2003 (83) and 2004 (1); adult GMs spiked in 2004 but have since gradually declined (Figure 3-24).

Northeast River adult White Perch trawl GMs were at or above the time-series median GM (50) in 2008–2010 and 2012–2015; adult White Perch GM was 39 in 2023 (Figure 3-24). 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 2012. Juvenile White Perch GMs were at or above the time-series median GM (45) in 2007–2011, 2014, and 2017; juvenile White Perch GM was 2 in 2023. Juvenile White Perch GMs ranged from 2 to 203 (2011); spikes in juvenile White Perch abundance occurred in 2011 and 2014 (151). The abundance of juvenile and adult White Perch were similar during sampling years, 2007–2009 and 2015–2017. In 2023, both juvenile and adult White Perch trawl GMs declined slightly since 2022 (Figure 3-24).

South River adult White Perch trawl GMs in 2003–2005 were at or above the median time-series GM (1); 2023 was 0 (Figure 3-24). Juvenile White Perch GMs in South River ranged from 0 (2004 and 2023) to 7 (2003). South River juvenile and adult White Perch trawl GMs varied little due to poor bottom DO throughout the subestuary.

Adult White Perch trawl GMs in Tred Avon River in 2006–2009, 2012–2013, 2017, 2019–2020 were at or below the median time-series GM (3); in 2023, adult White Perch GM was 2 (Figure 3-24). Juvenile White Perch GMs ranged from 0 (2008–2009, 2012–2013, 2016, and 2020–2023) to 13 (2011); juvenile White Perch GM was 0 in 2023. Tred Avon River adult White Perch trawl GMs declined during 2006–2010 and 2012–2014; they were highest during 2012 and 2013. Abundances of juvenile and adult White Perch had similar patterns throughout the time-series (Figure 3-24).

Modified proportional stock densities (PSDs) revealed White Perch primarily use Mattawoman Creek as nursery habitat. Modified PSDs for Mattawoman Creek fluctuated between 0% and 1.4% (Table 3-16; Figure 3-25). Mattawoman Creek's modified PSD in 2023 was 0%, which is also the long-term median (Table 3-16; Figure 3-25).

Adult White Perch were not present in Miles River in 2023 so a modified PSD could not be estimated. Modified PSDs for the four years White Perch were present have been either at or near 0% or 48.6% - 58.4% (Table 3-16; Figure 3-25).

Modified PSDs for Northeast River fluctuated between 0.2% and 1.6% (Table 3-16; Figure 3-25). Quality size White Perch comprised a greater portion of the samples prior to 2013. The modified PSD estimate was 0.3% in 2023. Modified PSDs were greater than the time-series median (0.5%) in 2007–2012 and 2017 (Table 3-16; Figure 3-25).

Adult White Perch were not present in South River in 2023 and a modified PSD could not be estimated. Boom and bust dynamics of modified PSDs, similar to Miles River, were observed in the time series (Table 3-16; Figure 3-25).

Tred Avon River modified PSDs have ranged from 4.7% to 52.1% (Table 3-16; Figure 3-25). They have been relatively high since 2017, reflecting the size progression of the stronger year-classes in 2011, 2014, 2015, and 2018 (Durell and Weedon 2023) into harvestable size. Modified PSDs were above the time-series median (21.8%) in 2010–2011, 2015, 2017–2020, and 2023 (Table 3-16; Figure 3-25).

Modified PSD time-series indicated that the tidal-fresh subestuaries sampled in 2023 (Mattawoman Creek and Northeast River) were primarily habitat for White Perch too small to be of interest to anglers. Mesohaline subestuaries with extensive low bottom channel DO measurements (South and Miles Rivers) had highly variable PSDs from year to year and their fisheries appeared unstable. White Perch of a size of interest to anglers were more likely to be found in subestuaries with rural or transition watersheds and least likely to be found in subestuaries with suburban-urban watersheds (Uphoff et al. 2013). In the Choptank River, a higher proportion of White Perch adults in Harris and Broad Creeks were of a size of interest to anglers than more developed Tred Avon River (Uphoff et al. 2016). Size quality of White Perch directly aligned with the percentage of all DO measurements below the target level; however, sample sizes indicated higher abundance in Tred Avon River, so diminished size quality may reflect density-dependent dynamics (Uphoff et al. 2016).

Geometric mean catches for all finfish in historical 3.1 m and 4.9 m bottom trawls in Mattawoman Creek were calculated for 1989–2016 and 2022–2023 (Figure 3-26). Uphoff et al. (2016) used 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 years when both were used. This analysis 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 1989–2023 (slope=0.34; SE=0.03; r^2=0.97; P<0.0001; N=8, respectively). 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 2022–2023 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 catches in 3.1 m and 4.9 m bottom trawls in Mattawoman Creek (1989–2016 and 2022–2023; Uphoff et al. 2016). Obvious changes in these catches were not suggested for early and later periods (Figure 3-26).

There was substantial CI overlap of seine GMs in Miles and South Rivers (Figure 3-27). The 2023 seine GM in Tred Avon River was low compared to most years. Northeast River seine GM in 2023 was low compared to most years. Due to high SAV presence in Mattawoman Creek, consistent beach seine sampling was not possible after 2004 (Figure 3-27).

Mattawoman Creek beach seine finfish composition for the top 90% of the catch during 2003–2005 were comprised of nine species: White Perch (juvenile and adult; 26.3%), Banded Killifish (18.6%), Golden Shiner (14.6%), Largemouth Bass (7.7%), Bluegill (7.1%), Spottail Shiner (5.2%), Tessellated Darter (5.0%), Blueback Herring (3.7%), and Alewife (2.8%; Figure 3-28). Twenty species comprised the Other Species (8.9%) category in Mattawoman Creek. Four of the eight species in the top 90% are considered target species, White Perch, Spottail Shiner, Blueback Herring, and Alewife (Figure 3-28).

Miles River's top species caught by seining for all sampling years were Atlantic Menhaden (33.9%), Atlantic Silverside (31.2%), White Perch (juvenile and adult; 13.6%), Striped Killifish (9.0%), and Mummichog (3.2%); an additional 31 species (9.0% of catch) were collected in Miles River. Two species in the top 90% were target species: Atlantic Menhaden and White Perch (Figure 3-28).

Seven species were in the top 90% of seine catches in the Northeast River when all years were combined (Figure 3-28). Gizzard Shad (34.9%) was the most abundant species, followed by White Perch (juvenile and adult; 23.1%), Blueback Herring (16.6%), Alewife (4.3%), Bay Anchovy (4.3%), Atlantic Menhaden (3.5%), and Threadfin Shad (3.4%); Clupeids were abundant in seine samples, but not bottom trawl samples. Threadfin Shad were not a target species. Thirty-seven species comprised the other species (9.9%) category in the Northeast River (Figure 3-28); Pumpkinseed and Spottail Shiner comprised 26.7% and 22.5% of the Other

Species category, respectively. Other notable species caught in Northeast River seine catches were two invasive species, Blue Catfish and Flathead Catfish; Blue Catfish were caught in seine catches in 2015, 2022, and 2023; only one Flathead Catfish was caught in a seine catch in 2022. Three species in the top 90% of beach seine catches were in common between the tidal-fresh subestuaries, Mattawoman Creek and Northeast River: Alewife, Blueback Herring, and White Perch (Figure 3-28).

South River's top species caught by seine when all sampling years were combined were Atlantic Menhaden (56.2%), White Perch (juvenile and adult; 14.4%), Atlantic Silverside (11.4%), Striped Bass (juvenile and adult; 6.7%), and Inland Silverside (3.7%); an additional 36 species (7.5%) were collected in South River (Figure 3-28). Three species in the top 90% were target species: Atlantic Menhaden, White Perch, and Striped Bass (Figure 3-28).

Seven species comprised the top 90% of finfish in beach seines when all years were combined in Tred Avon River (Figure 3-28). Tred Avon River's top species during 2006–2023 were Atlantic Silverside (35.6%), Atlantic Menhaden (18.8%), White Perch (juvenile and adult; 14.6%), Mummichog (7.8%), Striped Killifish (7.7%), Bay Anchovy (3.9%), and Banded Killifish (3.0%); an additional 42 other species (8.7%) were collected while seining in Tred Avon River. Only three species in the top 90% were target species: Atlantic Menhaden, White Perch, and Bay Anchovy. Three species in the top 90% of beach seine catches were common between the mesohaline subestuaries, Miles, South, and Tred Avon Rivers: Atlantic Menhaden, Atlantic Silverside, and White Perch (Figure 3-28).

More species appear in the top 90% of seine catches than trawl catches each year, 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.

Striped Killifish was the only species that was consistently present in the top 90% of Miles River annual beach seines, ranging from 4% (2003) to 26% (2020; Figure 3-29). Species composition was dominated by Atlantic Menhaden, Atlantic Silverside, and White Perch (juvenile and adult). Other species present in the top 90% of species caught annually were Bay Anchovy, 1 of 5 years, 15% in 2020; Mummichog, 2 of 5 years; Spot, 1 of 5 years; and Striped Bass (juvenile and adult), 1 of 5 years (Figure 3-29).

Northeast River annual beach seine collections were generally dominated by clupeids and White Perch (juvenile and adult) during 2007–2017 and 2022–2023 with Alewife, Blueback Herring, Gizzard Shad, and White Perch often present in the top 90%; 2023 was notable for the dominance of Gizzard Shad and White Perch (Figure 3-29). Alewife fluctuated between 3% and 11%); Blueback Herring, 4% and 35%; Gizzard Shad, 5% and 63%; and White Perch, 7% and 46%. Other species present in the top 90% were Bay Anchovy, 8 of the 13 years; Pumpkinseed, 4 of the 13 years; Spottail Shiner, 5 of 13 years; Striped Bass (juvenile and adult), 1 of 13 years; Threadfin Shad, 3 of 13 years; and Yellow Perch, 1 of 13 years (Figure 3-29).

South River annual beach seine composition primarily consisted of Atlantic Menhaden and Atlantic Silverside, both were present during all five sampling years; Atlantic Menhaden was the dominant species in 2004 (64%), 2005 (57%), 2022 (71%), and 2023 (87%), but only made up 5% of catch in 2003 (Figure 3-29). Atlantic Silverside ranged from 7% (2023) to 25% (2005). Other species present in the top 90% of species caught annually: Inland Silverside, 3 of 5 years; Spot, 1 of 5 years; Striped Bass (juvenile and adult), 2 of 5 years; and White Perch (juvenile and adult), 2 of 5 years (Figure 3-29).

Tred Avon River annual beach seine composition was similar among all seventeen years of sampling (Figure 3-29). One species, Atlantic Silverside, was in the top 90% during all years.

Striped Killifish was present in 17 of 18 sampling years; White Perch (juvenile and adult), 16 of 18 years,; Mummichog, 15 of 18 years; Atlantic Menhaden, 11 of 18 years; Bay Anchovy, 7 of 18 years; Banded Killifish, 6 of 18 years; Striped Bass (juvenile and adult), 6 of 18 years; Spot, 5 of 18 years; Inland Silverside, 1 of 18 years; and Sheepshead Minnow, 1 of 18 years (Figure 3-29).

Mattawoman Creek bottom trawl GMs of all species and White Perch GMs from 4.9m bottom trawl catches increased between 2022 and 2023. Both adult White Perch and juvenile White Perch GMs increased twofold since 2022 without changes in modified PSDs. Mattawoman Creek bottom trawl GMs were low in 2004–2009, 2011–2012, and 2014–2016; the 2023 trawl GM was greater than those for 2006–2010, 2012, and 2022. Spottail Shiners reappeared in the top 90% of trawl catches in 2023 after disappearing in 2022 in Mattawoman Creek, a notable change since Spottail Shiners have comprised part of the top 90% since sampling started. Species richness in bottom trawl catches increased in 2023 (19) from 2022 (12). Total catch of adult White Perch in trawl catches increased substantially in 2023 and juvenile White Perch declined. The changes in species composition could reflect changes in flows and water quality, 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 (N) 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 in the early 2000s. 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. Sediment loads in Mattawoman Creek from construction and stream bank erosion were high (Gellis et al. 2009) and they increased nutrient loading.

In 2023, there was little indication that low DO was more widespread in Mattawoman Creek than usual. Salinity was noticeably greater in 2023 and more marine species were present in trawl catches. Bottom DO at all stations remained above the target level, but station 04 bottom DO declined below the time-series median, joining the other stations. Other water quality measurements did not offer an obvious connection to changes in finfish abundance. 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 SAV coverage, 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.

Miles River was previously sampled in 2003–2005 and 2020. Little change in C/ha has occurred between sampling years 2020 and 2023 in the rural watershed. Miles River finfish bottom trawl and beach seine GMs increased compared to 2020 GMs; trawl and seine GMs in 2023 were similar to 2003 and 2004. There was an increase in the number of finfish caught and species richness in bottom trawls between 2020 and 2023; trawl species composition changed indicating greater presence of Spot and absence of White Perch (juvenile and adult); Bay Anchovy remained within the top 90% and Atlantic Croaker appeared for the first time in 2023. Violations for all DO measurements increased minimally while bottom DO violations decreased in 2023. Overall, the decline in White Perch abundance in Miles River is following suit with other mesohaline systems sampled in 2023, where juvenile and adult White Perch have disappeared from the top 90% of species caught.

Northeast River finfish trawl catches increased in 2023 but remained low compared to 2007–2017; beach seine catches declined in 2023 and were the third lowest GM in the time-series. White Perch GMs for juveniles and adults decreased slightly in 2023, while there was a minimal increase in modified PSDs. White Perch (juvenile and adult) remained dominant in the top 90% of species in the catch during all sampling years. Bay Anchovy and Brown Bullhead are other species consistently present in the top 90% of trawl catches. Spot appeared for the first time in the top 90% in 2023. Seining indicated that clupeids were abundant in Northeast River.

South River was previously sampled in 2003–2005. Returning to South River in 2022 and 2023, C/ha increased from 1.27 to 1.43 C/ha within the suburban watershed. South River finfish bottom trawl and beach seine GMs increased in 2023; the total number of finfish caught in seine catches more than doubled in 2023 compared to 2022, respectively. There was little change in the total finfish caught and species richness in bottom trawls in 2023. Juvenile or adult White Perch were not caught in bottom trawls in 2023. Lower finfish catches and absence in the bottom channel within the upper- and mid-river reflects high development and low DO measurements. South River habitat is similar to other western shore mesohaline subestuaries with suburbanized watersheds (Severn and Magothy Rivers.

Tred Avon River finfish trawl GMs have increased since 2020 and were at a moderate level, but beach seine GMs were relatively low in 2022–2023. Adult White Perch GMs CIs have overlapped since 2014 at a moderate level and modified PSDs have been at the mid-range for this subestuary since 2021. Juvenile White Perch have not been present in trawl catches since

2019; adult White Perch abundance in trawl catches has increased since 2021. Bay Anchovy increased noticeably in 2023, while White Perch (juvenile and adult) disappeared from the top 90%.

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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		
Broad Creek	Mid-Bay (Choptank)	2012	x	3	x	4		
		2013	x	3	x	4		
		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		
		2019			x	4		
Fishing Bay	Lower-Bay	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		
		2016	x	3	x	4		
Harris Creek	Mid-Bay (Choptank)	2012	x	3	x	4		
		2013	x	3	x	4		
		2014	x	3	x	4		
		2015	x	3	x	4		
		2016	x	3	x	4		
Langford Creek	Mid-Bay (Chester)	2006	x	4	x	4		
		2007	x	4	x	4		
		2008	x	4	x	4		
		2018	x	3	x	4		
		2019	x	3	x	4		
Magothy River	Mid-Bay	2003	x	4	x	4		
Mattawoman Creek	Potomac	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		
		2023			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
		2023	x	2	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
		2023	x	4	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
Rhode River	Mid-Bay	2003	x	2	x	2
		2004	x	2	x	2
		2005	x	2	x	2
Sassafras River	Upper-Bay	2020			x	4
		2021	x	4	x	4
Severn River	Mid-Bay	2003	x	5	x	4
		2004	x	5	x	4
		2005	x	5	x	4
		2017	x	3	x	4
South River	Mid-Bay	2003	x	4	x	4
		2004	x	4	x	4
		2005	x	4	x	4
		2022	x	4	x	4
		2023	x	3	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
		2023	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
		2017	x	4	x	4
Wye River	Mid-Bay	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 2023.

2023 Sampled Subestuaries						
Area	**Subestuary**	**IS**	**C/ha**	**Land Hectares**	**Water Hectares**	**Salinity Class**
Potomac	Mattawoman Creek	11.70	1.03	24,331	840	Tidal-Fresh
Mid-Bay	Miles River	3.95	0.26	10,968	3,141	Mesohaline
Head-of-Bay	Northeast River	6.57	0.52	16,214	1,772	Tidal-Fresh
Mid-Bay	South River	15.76	1.43	14,681	2,436	Mesohaline
Mid-Bay	Tred Avon River	9.31	0.79	9,433	3,216	Mesohaline

Table 3-3. Estimates of structures per hectare (C/ha) and land use percentages for subestuaries sampled 2003–2023.

River	Year	C/ha	Agriculture	Forest	Wetland	Urban
Breton Bay	2003	0.27	23.81	56.08	0.81	18.72
Breton Bay	2004	0.28	23.81	56.08	0.81	18.72
Breton Bay	2005	0.30	23.81	56.08	0.81	18.72
Broad Creek	2012	0.29	42.55	25.39	0.36	31.47
Broad Creek	2013	0.30	42.55	25.39	0.36	31.47
Broad Creek	2014	0.30	42.55	25.39	0.36	31.47
Broad Creek	2015	0.30	42.55	25.39	0.36	31.47
Broad Creek	2016	0.30	42.55	25.39	0.36	31.47
Broad Creek	2017	0.30	42.55	25.39	0.36	31.47
Broad Creek	2020	0.31	42.05	25.47	0.35	31.89
Bush River (no APG)	2006	1.41	25.39	34.95	3.23	36.17
Bush River (no APG)	2007	1.43	25.39	34.95	3.23	36.17
Bush River (no APG)	2008	1.45	25.39	34.95	3.23	36.17
Bush River (no APG)	2009	1.46	25.39	34.95	3.23	36.17
Bush River (no APG)	2010	1.47	17.99	29.91	3.21	47.76
Chester River	2007	0.14	66.47	25.76	1.98	5.77
Chester River	2008	0.14	66.47	25.76	1.98	5.77
Chester River	2009	0.15	66.47	25.76	1.98	5.77
Chester River	2010	0.15	64.21	24.67	1.96	8.90
Chester River	2011	0.15	64.21	24.67	1.96	8.90
Chester River	2012	0.15	64.21	24.67	1.96	8.90
Chester River	2018	0.16	63.93	24.62	1.93	9.21
Chester River	2019	0.16	63.93	24.62	1.93	9.21
Corsica River	2003	0.17	64.32	27.36	0.43	7.89
Corsica River	2004	0.18	64.32	27.36	0.43	7.89
Corsica River	2005	0.19	64.32	27.36	0.43	7.89
Corsica River	2006	0.21	64.32	27.36	0.43	7.89
Corsica River	2007	0.22	64.32	27.36	0.43	7.89
Corsica River	2008	0.24	64.32	27.36	0.43	7.89
Corsica River	2010	0.24	60.37	25.51	0.39	13.17
Corsica River	2011	0.25	60.37	25.51	0.39	13.17
Corsica River	2012	0.25	60.37	25.51	0.39	13.17
Corsica River	2018	0.28	59.94	25.71	0.35	13.62
Corsica River	2019	0.28	59.94	25.71	0.35	13.62
Gunpowder River	2009	0.72	35.74	36.15	1.03	26.84
Gunpowder River	2010	0.72	30.63	32.09	1.02	35.58
Gunpowder River	2011	0.73	30.63	32.09	1.02	35.58
Gunpowder River	2012	0.73	30.63	32.09	1.02	35.58
Gunpowder River	2013	0.73	30.63	32.09	1.02	35.58
Gunpowder River	2014	0.73	30.63	32.09	1.02	35.58
Gunpowder River	2015	0.74	30.63	32.09	1.02	35.58
Gunpowder River	2016	0.74	30.63	32.09	1.02	35.58
Harris Creek	2012	0.39	44.87	19.72	5.61	29.80
Harris Creek	2013	0.40	44.87	19.72	5.61	29.80
Harris Creek	2014	0.40	44.87	19.72	5.61	29.80
Harris Creek	2015	0.40	44.87	19.72	5.61	29.80
Harris Creek	2016	0.40	44.87	19.72	5.61	29.80

Tabel 3-3. Continued.

Langford Creek	2006	0.07	71.63	23.04	1.48	3.85
Langford Creek	2007	0.07	71.63	23.04	1.48	3.85
Langford Creek	2008	0.07	71.63	23.04	1.48	3.85
Langford Creek	2018	0.08	70.14	20.31	1.43	8.11
Langford Creek	2019	0.08	70.14	20.31	1.43	8.11
Magothy River	2003	2.68	2.57	27.82	0.00	69.51
Mattawoman Creek	2003	0.76	11.88	59.37	1.18	27.38
Mattawoman Creek	2004	0.79	11.88	59.37	1.18	27.38
Mattawoman Creek	2005	0.81	11.88	59.37	1.18	27.38
Mattawoman Creek	2006	0.83	11.88	59.37	1.18	27.38
Mattawoman Creek	2007	0.86	11.88	59.37	1.18	27.38
Mattawoman Creek	2008	0.87	11.88	59.37	1.18	27.38
Mattawoman Creek	2009	0.88	11.88	59.37	1.18	27.38
Mattawoman Creek	2010	0.90	9.33	53.88	1.13	34.18
Mattawoman Creek	2011	0.91	9.33	53.88	1.13	34.18
Mattawoman Creek	2012	0.90	9.33	53.88	1.13	34.18
Mattawoman Creek	2013	0.92	9.33	53.88	1.13	34.18
Mattawoman Creek	2014	0.93	9.33	53.88	1.13	34.18
Mattawoman Creek	2015	0.94	9.33	53.88	1.13	34.18
Mattawoman Creek	2016	0.96	9.33	53.88	1.13	34.18
Mattawoman Creek	2022	1.03	8.63	52.83	1.14	35.65
Mattawoman Creek	2023	1.03	8.63	52.83	1.14	35.65
Middle River	2009	3.30	4.55	27.87	2.16	63.91
Middle River	2010	3.32	3.41	23.32	2.13	70.98
Middle River	2011	3.33	3.41	23.32	2.13	70.98
Middle River	2012	3.33	3.41	23.32	2.13	70.98
Middle River	2013	3.37	3.41	23.32	2.13	70.98
Middle River	2014	3.39	3.41	23.32	2.13	70.98
Middle River	2015	3.40	3.41	23.32	2.13	70.98
Middle River	2016	3.42	3.41	23.32	2.13	70.98
Middle River	2017	3.41	3.41	23.32	2.13	70.98
Miles River	2003	0.24	53.71	27.21	0.89	18.14
Miles River	2004	0.24	53.71	27.21	0.89	18.14
Miles River	2005	0.24	53.71	27.21	0.89	18.14
Miles River	2020	0.26	48.70	26.52	0.86	23.39
Miles River	2023	0.26	48.70	26.52	0.86	23.39
Nanjemoy Creek	2003	0.08	15.13	73.07	4.12	7.62
Nanjemoy Creek	2008	0.09	15.13	73.07	4.12	7.62
Nanjemoy Creek	2009	0.09	15.13	73.07	4.12	7.62
Nanjemoy Creek	2010	0.09	12.38	68.70	4.09	14.74
Nanjemoy Creek	2011	0.09	12.38	68.70	4.09	14.74
Nanjemoy Creek	2012	0.09	12.38	68.70	4.09	14.74
Nanjemoy Creek	2013	0.09	12.38	68.70	4.09	14.74
Nanjemoy Creek	2014	0.09	12.38	68.70	4.09	14.74
Nanjemoy Creek	2015	0.09	12.38	68.70	4.09	14.74
Nanjemoy Creek	2016	0.09	12.38	68.70	4.09	14.74
Northeast River	2007	0.44	36.66	42.65	0.13	20.13
Northeast River	2008	0.44	36.66	42.65	0.13	20.13
Northeast River	2009	0.45	36.66	42.65	0.13	20.13
Northeast River	2010	0.46	31.08	38.65	0.11	28.86
Northeast River	2011	0.46	31.08	38.65	0.11	28.86

Table 3-3. Continued.

Northeast River	2012	0.47	31.08	38.65	0.11	28.86
Northeast River	2013	0.48	31.08	38.65	0.11	28.86
Northeast River	2014	0.48	31.08	38.65	0.11	28.86
Northeast River	2015	0.49	31.08	38.65	0.11	28.86
Northeast River	2016	0.49	31.08	38.65	0.11	28.86
Northeast River	2017	0.49	31.08	38.65	0.11	28.86
Northeast River	2022	0.52	30.63	38.07	0.11	29.92
Northeast River	2023	0.52	30.63	38.07	0.11	29.92
Piscataway Creek	2003	1.30	12.76	45.76	0.25	40.57
Piscataway Creek	2006	1.38	12.76	45.76	0.25	40.57
Piscataway Creek	2007	1.40	12.76	45.76	0.25	40.57
Piscataway Creek	2009	1.43	12.76	45.76	0.25	40.57
Piscataway Creek	2010	1.45	9.98	40.37	0.24	47.01
Piscataway Creek	2011	1.46	9.98	40.37	0.24	47.01
Piscataway Creek	2012	1.47	9.98	40.37	0.24	47.01
Piscataway Creek	2013	1.50	9.98	40.37	0.24	47.01
Piscataway Creek	2014	1.51	9.98	40.37	0.24	47.01
Rhode/West Rivers	2003	0.55	34.07	45.30	0.79	19.84
Rhode/West Rivers	2004	0.56	34.07	45.30	0.79	19.84
Rhode/West Rivers	2005	0.56	34.07	45.30	0.79	19.84
Sassafras River	2020	0.11	63.98	25.80	1.28	8.55
Sassafras River	2021	0.11	63.98	25.80	1.28	8.55
Severn River	2003	2.06	8.57	35.18	0.18	55.84
Severn River	2004	2.09	8.57	35.18	0.18	55.84
Severn River	2005	2.15	8.57	35.18	0.18	55.84
Severn River	2017	2.38	4.97	27.97	0.20	65.07
South River	2003	1.24	15.25	45.62	0.36	38.76
South River	2004	1.25	15.25	45.62	0.36	38.76
South River	2005	1.27	15.25	45.62	0.36	38.76
South River	2022	1.43	10.11	38.60	0.48	49.67
South River	2023	1.43	10.11	38.60	0.48	49.67
St. Clements River	2003	0.19	38.61	48.62	0.85	11.84
St. Clements River	2004	0.20	38.61	48.62	0.85	11.84
St. Clements River	2005	0.20	38.61	48.62	0.85	11.84
Tred Avon River	2006	0.69	50.08	21.58	1.00	27.23
Tred Avon River	2007	0.71	50.08	21.58	1.00	27.23
Tred Avon River	2008	0.73	50.08	21.58	1.00	27.23
Tred Avon River	2009	0.74	50.08	21.58	1.00	27.23
Tred Avon River	2010	0.75	43.20	21.63	0.85	33.57
Tred Avon River	2011	0.75	43.20	21.63	0.85	33.57
Tred Avon River	2012	0.75	43.20	21.63	0.85	33.57
Tred Avon River	2013	0.76	43.20	21.63	0.85	33.57
Tred Avon River	2014	0.77	43.20	21.63	0.85	33.57
Tred Avon River	2015	0.77	43.20	21.63	0.85	33.57
Tred Avon River	2016	0.78	43.20	21.63	0.85	33.57
Tred Avon River	2017	0.77	43.20	21.63	0.85	33.57
Tred Avon River	2018	0.78	42.63	21.67	0.86	33.96
Tred Avon River	2019	0.79	42.63	21.67	0.86	33.96

Table 3-3. Continued.

Tred Avon River	2020	0.79	42.63	21.67	0.86	33.96
Tred Avon River	2021	0.79	42.63	21.67	0.86	33.96
Tred Avon River	2022	0.79	42.63	21.67	0.86	33.96
Tred Avon River	2023	0.79	42.63	21.67	0.86	33.96
Wicomico River	2003	0.30	27.72	54.11	1.66	15.89
Wicomico River	2010	0.34	24.40	49.52	1.65	23.49
Wicomico River	2011	0.35	24.40	49.52	1.65	23.49
Wicomico River	2012	0.35	24.40	49.52	1.65	23.49
Wicomico River	2017	0.39	24.40	49.52	1.65	23.49
Wye River	2007	0.10	64.93	22.98	0.63	10.92
Wye River	2008	0.10	64.93	22.98	0.63	10.92
Wye River	2018	0.10	64.34	23.26	0.63	11.21
Wye River	2019	0.10	64.34	23.26	0.63	11.21

Table 3-4. Summary of water quality parameter statistics collected during both seine and trawl samples for subestuaries in 2023. Summary statistics for pH were calculated from H+ concentrations and converted back to pH.

		Surface Measurements					Bottom Measurements					
System	Statistics	Temp ©	DO (mg / L)	Cond (umhols)	Salinity	pH	Temp ©	DO (mg / L)	Cond (umhols)	Salinity	pH	Secchi
Mattawoman Creek	Mean	27.54	7.44	1787.71	0.91	8.12	27.44	6.84	1833.42	0.94	8.04	0.80
	Standard Error	0.41	0.10	252.05	0.13	8.88	0.36	0.10	254.73	0.14	8.85	0.04
	Median	27.63	7.50	1331.00	0.66	8.31	27.50	6.82	1339.50	0.67	8.18	0.80
	Mode	27.49	7.65	.	0.25	7.95	27.40	7.13	.	0.25	8.43	1.00
	Kurtosis	0.28	-1.07	-1.51	-1.50	-1.47	1.31	-0.49	-1.53	-1.51	-1.58	-1.32
	Skewness	-0.99	-0.16	0.53	0.56	0.03	-1.31	0.19	0.49	0.52	0.44	-0.03
	Minimum	23.53	6.48	509.00	0.24	8.89	23.52	6.00	511.00	0.24	8.83	0.50
	Maximum	30.22	8.34	3736.00	1.96	7.65	29.45	7.80	3765.00	1.98	7.68	1.10
	Count	24	24	24	24	24	23	24	24	24	24	24
Miles River	Mean	27.51	6.35	23213.22	13.98	7.78	27.10	4.29	23382.17	14.02	7.57	0.46
	Standard Error	0.50	0.15	209.35	0.14	9.00	0.57	0.26	265.02	0.17	8.66	0.03
	Median	28.58	6.22	23758.50	14.31	7.77	28.17	4.33	23770.50	14.27	7.62	0.45
	Mode	29.61	6.80	24040.00	14.55	7.73	27.81	.	.	14.56	7.67	0.30
	Kurtosis	1.23	0.87	3.05	2.93	0.90	1.48	-0.42	1.59	1.56	-0.18	-0.82
	Skewness	-1.69	0.93	-1.80	-1.77	0.48	-1.74	-0.20	-1.20	-1.36	-0.11	0.43
	Minimum	20.89	4.96	19076.00	11.26	8.22	20.99	1.84	19812.00	11.73	7.86	0.25
	Maximum	29.94	8.83	24470.00	14.78	7.51	29.74	6.36	25533.00	14.97	7.26	0.75
	Count	36	36	36	36	36	24	24	24	24	24	24
Northeast River	Mean	27.27	8.21	591.27	0.28	8.08	26.14	5.30	589.67	0.28	7.62	0.35
	Standard Error	0.34	0.15	42.14	0.02	8.90	0.47	0.17	59.69	0.03	8.70	0.02
	Median	27.33	8.41	498.50	0.24	8.33	26.39	5.15	546.50	0.26	7.62	0.30
	Mode	.	8.63	342.00	0.13	8.60	#N/A	5.06	284.00	0.14	7.72	0.30
	Kurtosis	0.17	1.25	-1.01	-0.93	-0.68	0.59	2.58	-0.92	-0.92	0.94	-0.85
	Skewness	-0.81	-0.76	0.61	0.65	-0.33	-1.22	-0.78	0.63	0.64	1.50	0.39
	Minimum	21.38	4.69	259.00	0.12	8.84	21.25	2.84	259.00	0.12	8.22	0.20
	Maximum	30.87	9.86	1116.00	0.55	7.39	28.80	6.87	1106.00	0.54	7.37	0.50
	Count	48	48	48	48	48	24	24	24	24	24	24
South River	Mean	26.86	6.00	19272.74	11.41	7.69	26.93	2.30	21221.25	12.69	7.38	0.83
	Standard Error	0.49	0.20	245.44	0.16	8.77	0.48	0.32	334.01	0.22	8.48	0.05
	Median	27.98	6.27	19890.00	11.84	7.75	27.56	2.31	20760.00	12.37	7.36	0.90
	Mode	28.81	6.70	.	11.40	7.75	27.26	.	.	12.36	7.30	1.10
	Kurtosis	1.23	-0.33	0.64	0.43	0.02	1.13	-0.93	0.99	1.10	-0.86	-0.97
	Skewness	-1.65	-0.38	-1.20	-1.12	-0.02	-1.55	0.26	1.15	1.20	0.41	-0.40
	Minimum	19.37	3.09	14957.00	8.68	8.20	21.86	0.07	18816.00	11.10	7.75	0.30
	Maximum	30.03	8.46	21066.00	12.56	7.27	29.29	4.99	25026.00	15.26	7.13	1.20
	Count	43	43	43	43	43	24	24	24	24	24	24
Tred Avon River	Mean	28.17	6.26	21400.79	12.87	7.71	27.85	4.35	22091.50	13.22	7.53	0.62
	Standard Error	0.30	0.14	196.19	0.16	8.88	0.40	0.32	157.92	0.11	8.53	0.03
	Median	28.96	6.43	21767.50	13.05	7.75	28.44	5.09	22158.50	13.23	7.63	0.60
	Mode	28.70	5.28	.	13.66	7.75	23.91	5.14	.	13.17	7.72	0.60
	Kurtosis	0.41	0.77	0.57	5.16	-0.73	0.48	0.72	0.09	-0.08	0.03	-0.41
	Skewness	-1.30	0.21	-1.07	0.77	-0.30	-1.24	-1.35	-0.43	-0.21	-0.17	0.42
	Minimum	23.60	4.02	17660.00	10.29	8.06	23.91	0.50	20326.00	12.08	7.72	0.40
	Maximum	30.13	9.31	22961.00	17.40	7.26	30.32	6.30	23547.00	14.28	7.19	0.90
	Count	48	48	48	48	48	24	24	24	24	24	24

Table 3-5. Dissolved oxygen (DO) violations, percentages of all DO (surface, middle, and bottom) measurements that did not meet target (5.0 mg/L) and bottom DO measurements that did not meet target (5.0 mg/L) or threshold (3.0 mg/L) conditions for each subestuary sampled in 2023. C/ha=structures per hectare. N=number of samples.

				All DO		Bottom DO	
Subestuary	Salinity Class	C/ha	N	% < 5.0 mg/L	N	% < 5.0 mg/L	% < 3.0 mg/L
Mattawoman Creek	Tidal-Fresh	1.03	57	0%	24	0%	0%
Miles River	Mesohaline	0.26	78	37%	24	75%	13%
Northeast River	Tidal-Fresh	0.52	87	9%	24	25%	4%
South River	Mesohaline	1.43	85	53%	24	100%	71%
Tred Avon River	Mesohaline	0.79	96	24%	24	46%	21%

Table 3-6. Subestuaries sampled during 2003–2023, by salinity class, with C/ha (watershed structures per hectare), arithmetic mean annual surface and bottom temperatures, and arithmetic mean annual surface and bottom dissolved oxygen (mg/L).

			Temperature (ºC)		Dissolved Oxygen (mg / L)	
River	Year	C / ha	Surface	Bottom	Surface	Bottom
Mesohaline						
Blackwater River	2006	0.04	28.14	27.98	5.27	4.12
Breton Bay	2003	0.27	26.69	25.68	7.94	3.52
	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	27.76	26.64	7.69	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
	2020	0.31	27.94	27.57	7.55	5.57
Chester River	2007	0.14	25.59	24.18	5.38	4.53
	2008	0.14	25.06	25.29	5.25	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	171.38	6.83	6.00
	2019	0.16	27.45	27.05	6.75	5.77
Corsica River	2003	0.17	25.74	26.13	6.63	4.61
	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
	2009	0.24	25.35	32.14	5.42	3.57
	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
	2019	0.28	27.24	27.04	6.82	4.39
Fishing Bay	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	26.61	26.43	6.95	6.12
	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	26.33	25.85	7.70	6.46
	2018	0.08	26.86	26.58	7.61	5.80
	2019	0.08	27.77	27.51	6.69	5.07
Magothy River	2003	2.68	25.68	25.46	7.31	2.00
Miles River	2003	0.24	25.60	25.60	6.61	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
	2023	0.26	27.51	27.10	6.35	4.29

Table 3-6. Continued.

Rhode River	2003	0.47	25.08	24.69	7.58	4.80
	2004	0.47	26.88	26.72	6.59	5.49
	2005	0.48	27.78	27.16	6.50	4.03
Severn River	2003	2.06	26.18	24.75	7.56	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	25.52	24.56	7.70	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
	2023	1.43	26.86	26.93	6.00	2.30
St. Clements River	2003	0.19	26.11	25.36	7.87	3.39
	2004	0.20	26.08	25.78	6.84	4.61
	2005	0.20	27.94	27.14	6.05	4.39
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	26.63	26.51	6.32	5.49
	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
	2023	0.79	28.17	27.85	6.26	4.35
West River	2003	0.64	25.04	24.31	7.59	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	25.39	23.83	7.11	5.85
	2010	0.34	25.39	23.83	7.11	5.85
	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	28.40	27.78	7.99	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	25.26	25.73	8.47	7.14
	2015	0.74	27.51	27.65	8.02	6.63
	2016	0.74	27.70	26.46	7.43	6.18
Middle River	2009	3.30	26.50	25.78	7.27	6.07
	2010	3.32	24.65	24.20	8.44	7.11
	2011	3.33	27.13	26.42	8.35	7.33
	2012	3.33	28.05	26.60	8.82	5.21
	2013	3.37	27.05	26.35	7.57	5.80
	2014	3.39	25.99	25.17	7.87	6.45
	2015	3.40	28.47	27.20	8.20	6.23
	2016	3.42	28.87	27.82	7.56	5.69
	2017	3.41	25.54	25.17	7.80	5.36
Nanjemoy Creek	2003	0.08	26.21	28.03	7.43	5.59
	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	5.33
	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	25.30	24.20	7.27	6.99
	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	26.58	25.75	8.89	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	26.08	26.81	8.09	7.85
	2010	0.90	26.17	26.04	7.07	6.55
	2011	0.91	27.11	27.28	6.47	6.63
	2012	0.90	26.68	27.00	7.47	7.08
	2013	0.92	26.32	25.96	9.14	8.40
	2014	0.93	27.22	26.73	9.71	8.66
	2015	0.94	27.97	26.33	8.63	7.64
	2016	0.96	28.32	27.97	6.88	6.56
	2022	1.03	27.69	27.46	7.59	6.64
	2023	1.03	27.54	27.29	7.44	6.84

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	9.37	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	26.57	26.04	9.29	6.95
	2014	0.48	25.93	25.10	9.13	6.91
	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	0.52	27.60	26.88	8.95	6.62
	2023	0.52	27.27	26.14	8.21	5.30
Piscataway Creek	2003	1.30	26.02	24.63	10.54	8.33
	2006	1.38	28.16	24.97	8.70	6.85
	2007	1.40	27.47	26.00	8.57	7.60
	2009	1.43	26.72	27.07	8.56	6.62
	2010	1.45	27.07	25.08	9.36	7.63
	2011	1.46	28.25	30.07	9.05	9.47
	2012	1.47	27.92	25.51	9.53	9.34
	2013	1.50	27.19	26.22	9.87	7.65
	2014	1.51	26.08	26.41	8.46	7.34
Sassafras River	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 arithmetic mean surface and bottom dissolved oxygen (DO; mg/L) with water temperatures at depth (surface or bottom) collected during surveys or watershed development (C/ha=structures per hectare) for subestuaries sampled during 2003–2023, by salinity class. *P*=level of significance. N=sample size.

DO Depth	Statistics	Temperature	C / ha
Mesohaline			
Surface	r	-0.036	0.180
	P	0.731	0.082
	N	94	94
Bottom	r	0.108	-0.613
	P	0.301	<0.0001
	N	94	94
Oligohaline			
Surface	r	-0.315	0.402
	P	0.074	0.021
	N	33	33
Bottom	r	-0.578	-0.143
	P	0.0004	0.427
	N	33	33
Tidal Fresh			
Surface	r	-0.029	0.121
	P	0.859	0.457
	N	40	40
Bottom	r	0.021	0.417
	P	0.898	0.007
	N	40	40

Table 3-8. Pearson correlations (r) of C/ha for mesohaline subestuaries sampled during 2003–2023 with Maryland Department of Planning (DOP; 2002 and 2010) and Chesapeake Conservancy (2013 and 2018) land use categories. Pearson correlations (r) between land use categories. *P*=level of significance. N=sample size. Duplicate entries of C/ha for mesohaline subestuaries from 2003 to 2023 were not included in analysis.

				Land Use Categories			
	Statistics	**Year**	**C/ha**	**Agriculture**	**Forest**	**Wetland**	**Urban**
C/ha	r	-0.01					
	P	0.91	1				
	N	91					
Agriculture	r	0.12	-0.72				
	P	0.28	<0.0001	1			
	N	91	91				
Forest	r	-0.46	0.07	-0.61			
	P	<0.0001	0.52	<0.0001	1		
	N	91	91	91			
Wetland	r	-0.11	-0.21	-0.13	0.14		
	P	0.28	0.04	0.22	0.19	1	
	N	91	91	91	91		
Urban	r	0.21	0.90	-0.74	-0.02	-0.27	
	P	0.04	<0.0001	<0.0001	0.87	0.01	1
	N	91	91	91	91	91	

Table 3-9. Statistics and parameter estimates of agricultural land cover by region (western and eastern shores) versus median bottom dissolved oxygen (DO; mg/L) in mesohaline subestuaries within major drainages (2003–2023).

Linear Model	**Western Shore: Median Bottom DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	44.15	44.15	28.15	<.0001	
Residual	22	34.50	1.57			
Total	23	78.65				
$r^2 = 0.5613$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.80	0.57	1.40	0.18	-0.38	1.97
Agriculture (%)	0.13	0.02	5.31	<.0001	0.08	0.18
Linear Model	**Eastern Shore: Median Bottom DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	6.00	6.00	8.92	0.004	
Residual	62	41.71	0.67			
Total	63	47.71				
$r^2 = 0.1258$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	6.68	0.54	12.45	<.0001	5.61	7.76
Agriculture (%)	-0.03	0.01	-2.99	0.0040	-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	94.27	47.14	51.35	<.0001	
Residual	85	78.03	0.92			
Total	87	172.30				
$r^2 = 0.5471$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.23	0.45	0.5	0.62	-0.67	1.13
Agriculture (%)	0.21	0.02	9.04	<.0001	0.17	0.26
Agriculture (%)^2	-0.002	0.0003	-7.69	<.0001	-0.003	-0.002

Table 3-11. Percent of watershed in major land use cover categories for each subestuary sampled in 2023. 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.

	Head-of-Bay Subestuary	Mid-Bay Subestuaries			Potomac River Tributary
Land Use Category (%)	Northeast River	Miles River	South River	Tred Avon River	Mattawoman Creek
Agriculture	30.63	48.70	10.11	42.63	8.63
Forest	38.07	26.52	38.60	21.67	52.83
Wetlands	0.11	0.86	0.48	0.86	1.14
Urban	29.92	23.39	49.67	33.96	35.65
Water	10.93	28.64	16.59	34.10	3.45

Table 3-12. Percentages of all dissolved oxygen (DO; mg/L) measurements (surface, middle, and bottom) and 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 2023. N=sample size.

				All DO		Bottom DO	
Subestuary	Year	C / ha	N	% < 5.0 mg/L	N	% < 5.0 mg/L	% < 3.0 mg/L
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	1	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	46	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	1.03	52	4	24	4	0
	2023	1.03	57	0	24	0	0
Miles River	2003	0.24	96	46	25	72	24
	2004	0.24	81	26	23	48	0
	2005	0.24	86	45	24	71	46
	2020	0.26	76	36	21	81	48
	2023	0.26	78	37	24	75	13
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	88	5	24	13	4
	2016	0.49	84	0	24	0	0
	2017	0.49	93	1	24	4	0
	2022	0.52	77	5	24	13	0
	2023	0.52	87	9	24	29	4
South River	2003	1.24	112	28	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
	2023	1.43	85	53	24	100	71
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	10
	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	35	15
	2014	0.77	96	11	24	21	0
	2015	0.77	96	8	24	21	13
	2016	0.78	96	15	24	38	13
	2017	0.77	90	17	24	42	13
	2018	0.78	110	18	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	13
	2022	0.79	95	23	24	54	13
	2023	0.79	96	24	24	46	21

Table 3-13. Beach seine catch summary, 2023. C/ha=structures per hectare. GM CPUE=geometric mean catches per seine sample. Italics designate target species.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Miles River	2	12	22	*Atlantic Menhaden* Striped Killifish	0.26	7,462	298
Northeast River	4	24	26	*Gizzard Shad* *White Perch* *Atlantic Menhaden* Threadfin Shad *Spottail Shiner* *Blueback Herring* *Bay Anchovy* Pumpkinseed	0.52	2,445	88
South River	3	18	23	*Atlantic Menhaden* Atlantic Silverside	1.43	19,572	173
Tred Avon River	4	24	22	Atlantic Silverside *Atlantic Menhaden* Striped Killifish *Bay Anchovy* Mummichog Banded Killifish *White Perch* *Spot* Atlantic Croaker	0.79	2,343	76
Grand Total	13	78	45	*Atlantic Menhaden* Atlantic Silverside Striped Killifish *Gizzard Shad* *White Perch*		31,822	

Table 3-14. Bottom trawl catch summary, 2023. C/ha=structures per hectare. GM CPUE=geometric mean catches per trawl sample. Italics designate target species.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Mattawoman Creek	4	24	19	*White Perch* *Spot* *Spottail Shiner*	1.03	3,187	91
Miles River	4	24	15	*Spot* *Bay Anchovy* Atlantic Croaker	0.26	2,231	79
Northeast River	4	24	19	*White Perch* *Spot* Brown Bullhead	0.52	2,241	80
South River	4	24	13	Atlantic Croaker *Bay Anchovy* *Spot* *Weakfish*	1.43	1,560	30
Tred Avon River	4	24	16	*Bay Anchovy* *Spot* Hogchoker Atlantic Croaker Weakfish	0.79	1,939	52
Grand Total	20	120	35	*White Perch* *Spot* *Bay Anchovy* Atlantic Croaker Hogchoker		11,158	

Table 3-15. Subestuaries sampled during 2003–2023, grouped by salinity class and ranked by annual 4.9 m trawl geometric mean (GM) for catches 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
Tred Avon River	2007	120	31
Wicomico River	2010	118	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	2006	105	38
Corsica River	2008	103	39
Transquaking River	2006	95	40
Broad Creek	2015	94	41
Broad Creek	2017	90	42

Table 3-15. Continued.

Harris Creek	2013	88	43
Wicomico River	2012	87	44
Tred Avon River	2011	84	45
Chester River	2009	83	46
Tred Avon River	2009	82	47
Miles River	2023	79	48
Wicomico River	2017	78	49
Tred Avon River	2015	77	50
Miles River	2005	70	51
Chester River	2019	69	52
Tred Avon River	2017	66	53
Miles River	2004	65	54
Wicomico River	2011	62	55
Wicomico River	2003	58	56
Tred Avon River	2013	54	57
Tred Avon River	2023	52	58
Tred Avon River	2006	48	59
Harris Creek	2016	45	60
Langford Creek	2019	43	61
Tred Avon River	2022	43	62
Corsica River	2019	34	63
Tred Avon River	2019	33	64
Harris Creek	2015	31	65
South River	2023	30	66
Langford Creek	2018	28	67
Rhode River	2004	28	68
Broad Creek	2020	25	69
Wye River	2019	20	70
South River	2003	20	71
St. Clements River	2005	20	72
Tred Avon River	2021	19	73
Tred Avon River	2020	17	74
Breton Bay	2005	17	75
South River	2005	17	76
Corsica River	2018	16	77
West River	2004	15	78
South River	2004	14	79
Tred Avon River	2018	13	80
Wye River	2018	12	81
Blackwater River	2006	10	82
Miles River	2020	10	83
South River	2022	10	84
St. Clements River	2004	10	85
Breton Bay	2004	9	86

Table 3-15. Continued.

Breton Bay	2003	7	87
Magothy Ruver	2003	7	88
St. Clements River	2003	6	89
Severn River	2017	3	90
Severn River	2004	3	91
Severn River	2003	2	92
Severn River	2005	1	93
Oligohaline			
Nanjemoy Creek	2013	576	1
Middle River	2011	520	2
Bush River	2010	467	3
Nanjemoy Creek	2011	448	4
Nanjemoy Creek	2015	417	5
Nanjemoy Creek	2014	396	6
Gunpowder River	2011	394	7
Gunpowder River	2010	393	8
Bush River	2007	325	9
Bush River	2009	320	10
Middle River	2010	310	11
Nanjemoy Creek	2010	306	12
Nanjemoy Creek	2016	297	13
Middle River	2009	292	14
Middle River	2015	290	15
Gunpowder River	2009	288	16
Nanjemoy Creek	2009	282	17
Middle River	2016	262	18
Middle River	2014	254	19
Gunpowder River	2012	223	20
Nanjemoy Creek	2012	222	21
Gunpowder River	2014	220	22
Gunpowder River	2015	218	23
Bush River	2008	211	24
Nanjemoy Creek	2008	209	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	93	32
Middle River	2017	74	33
Tidal-Fresh			
Mattawoman Creek	2014	582	1
Northeast River	2010	306	2

Table 3-15. Continued.

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
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	120	22
Piscataway Creek	2009	107	23
Northeast River	2017	107	24
Northeast River	2016	97	25
Mattawoman Creek	2023	91	26
Mattawoman Creek	2010	81	27
Northeast River	2023	80	28
Mattawoman Creek	2006	75	29
Mattawoman Creek	2012	72	30
Northeast River	2022	59	31
Mattawoman Creek	2007	57	32
Piscataway Creek	2003	45	33
Mattawoman Creek	2022	28	34
Piscataway Creek	2006	25	35
Sassafras River	2020	25	36
Mattawoman Creek	2008	21	37
Sassafras River	2021	13	38
Mattawoman Creek	2009	7	39
Piscataway Creek	2007	5	40

Table 3-16. Annual modified proportional stock density (PSD) of White Perch for subestuaries sampled in 2023. Number of N_{TOTAL} is the total number of White Perch (all juvenile and adult) 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	0.0
	2004	2791	355	2	0.6
	2005	3916	471	1	0.2
	2006	1978	567	0	0.0
	2007	1385	442	1	0.2
	2008	716	366	1	0.3
	2009	346	61	0	0.0
	2010	2555	430	0	0.0
	2011	3460	282	4	1.4
	2012	2512	241	0	0.0
	2013	7026	223	1	0.4
	2014	12138	101	1	1.0
	2015	5774	466	0	0.0
	2016	4490	754	0	0.0
	2022	2223	109	0	0.0
	2023	2568	744	0	0.0
Miles River	2003	6704	185	108	58.4
	2004	941	798	4	0.5
	2005	1081	537	4	0.7
	2020	74	74	36	48.6
	2023	0	0	0	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
	2023	1599	1498	5	0.3
South	2003	1508	28	17	60.7
	2004	165	170	5	2.9
	2005	34	28	0	0.0
	2022	19	17	14	82.4
	2023	0	0	0	0.0
Tred Avon River	2006	364	362	45	12.4
	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
	2023	129	129	31	24.0

Figures

Figure 3-1. Map illustrating subestuaries sampled in summer 2023: Northeast River (1), Tred Avon River (2), South River (3), Miles River (4), Mattawoman Creek (5), and their estimated 2018 land use cover categories. Stars indicate previously sampled subestuaries mentioned throughout the report.

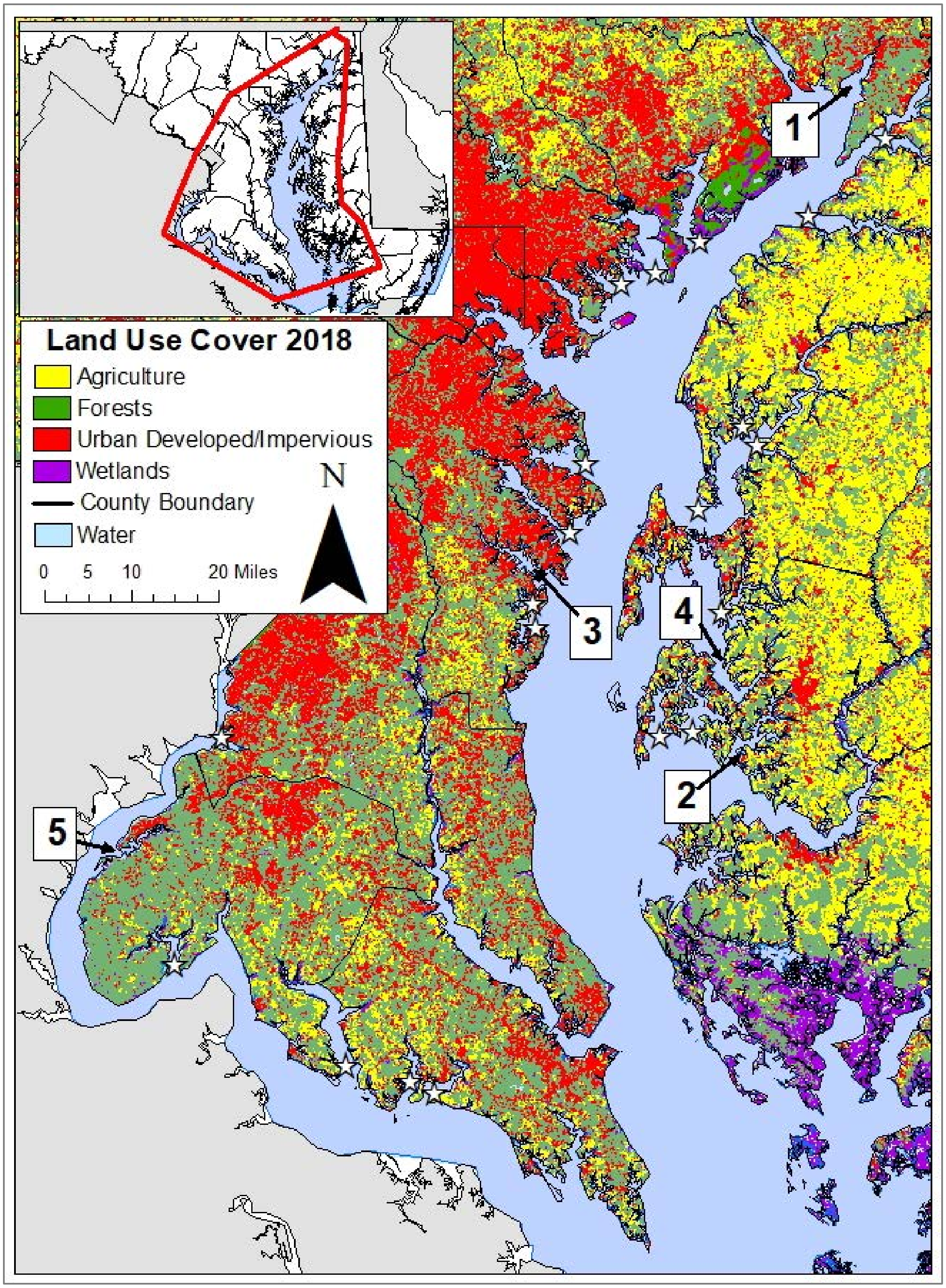

Figure 3-2. Map indicating locations of sampling stations sampled in 2023 located within Mattawoman Creek, Miles River, Northeast River, South River, and Tred Avon River.

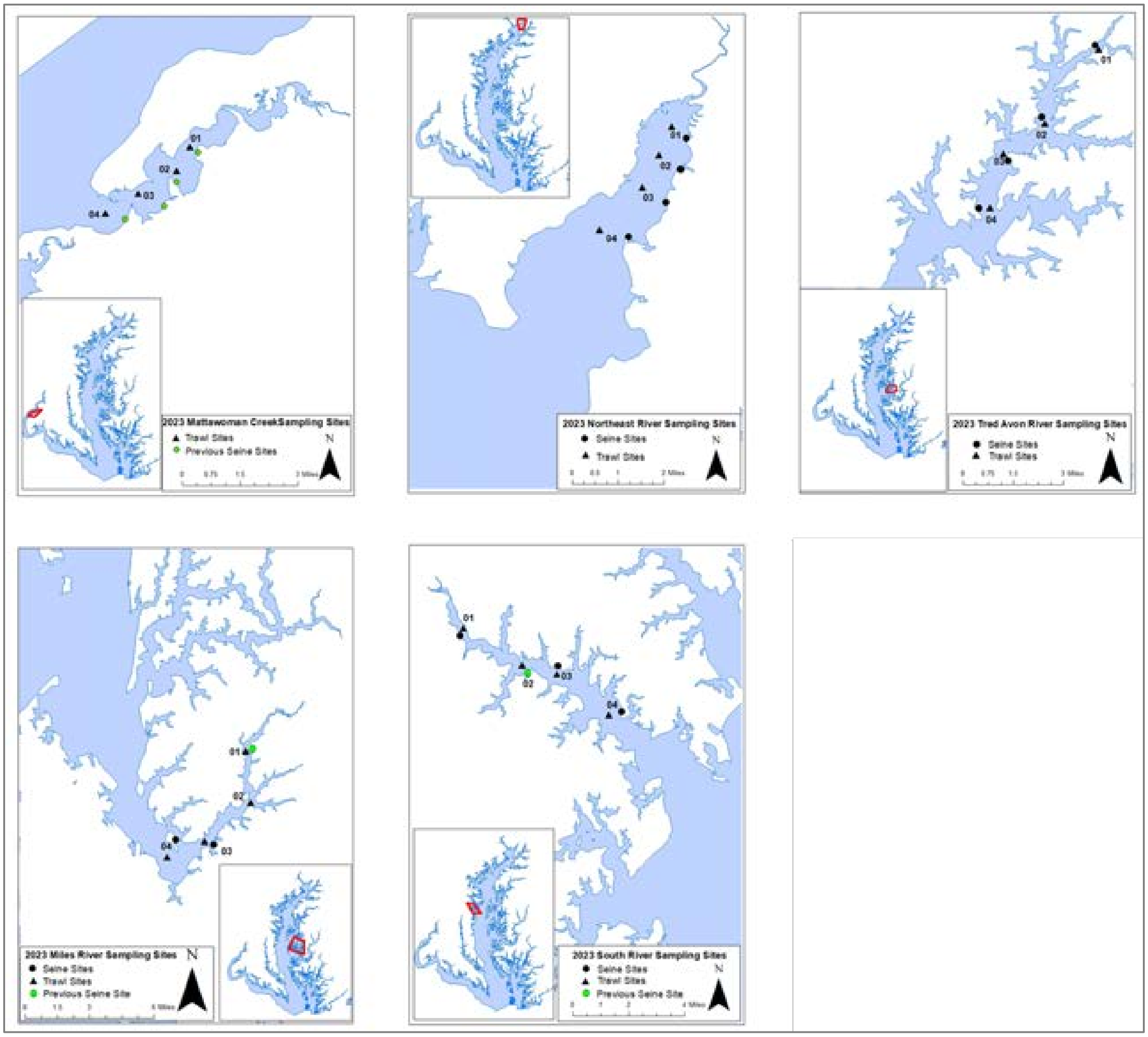

Figure 3-3. Mean subestuary bottom dissolved oxygen during summer (July–September) sampling, 2003–2023, plotted against level of development (C/ha or structures per hectare) and target and threshold dissolved oxygen.

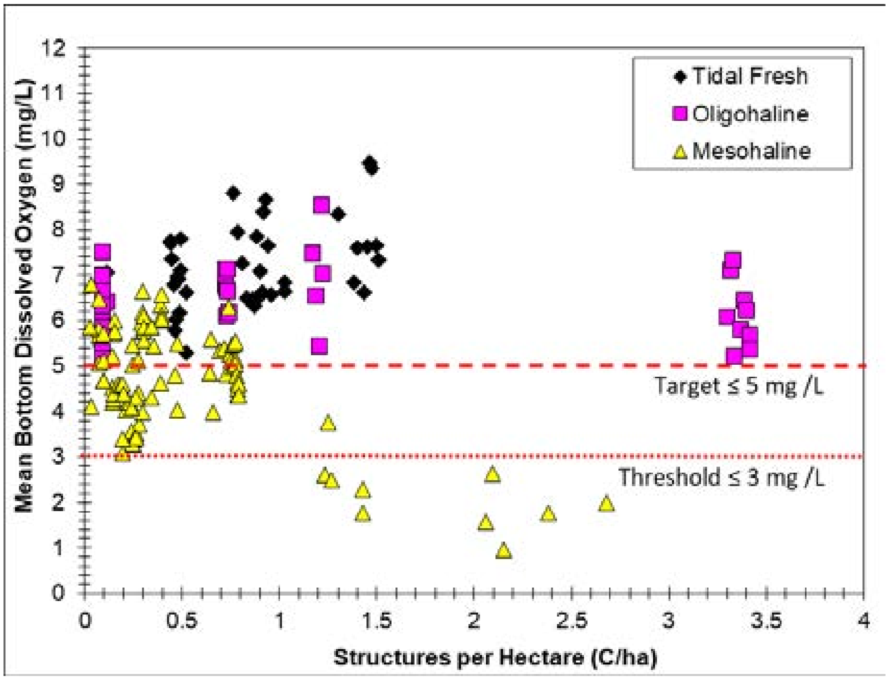

Figure 3-4. Mean subestuary surface dissolved oxygen during summer (July–October) sampling, 2003–2023, plotted against level of development (C/ha or structures per hectare) and target and threshold dissolved oxygen.

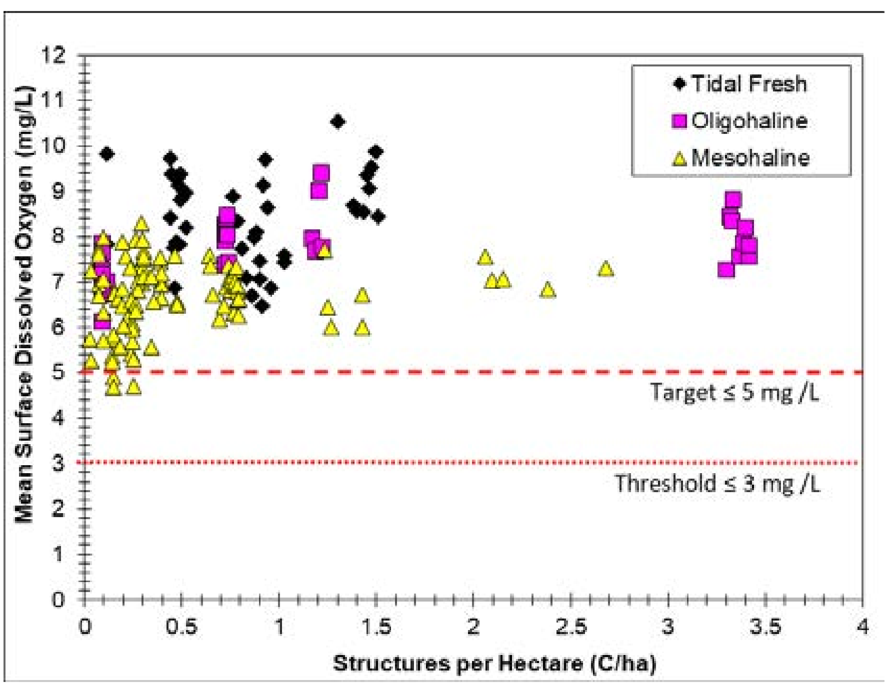

Figure 3-5. Estimates of agricultural land cover (% watershed land area) by region (western or eastern shore) versus median bottom dissolved oxygen (DO; mg/L) in mesohaline subestuaries within major drainages (2003–2023). The quadratic model predicts median bottom DO and agricultural coverage (%) using data from both regions.

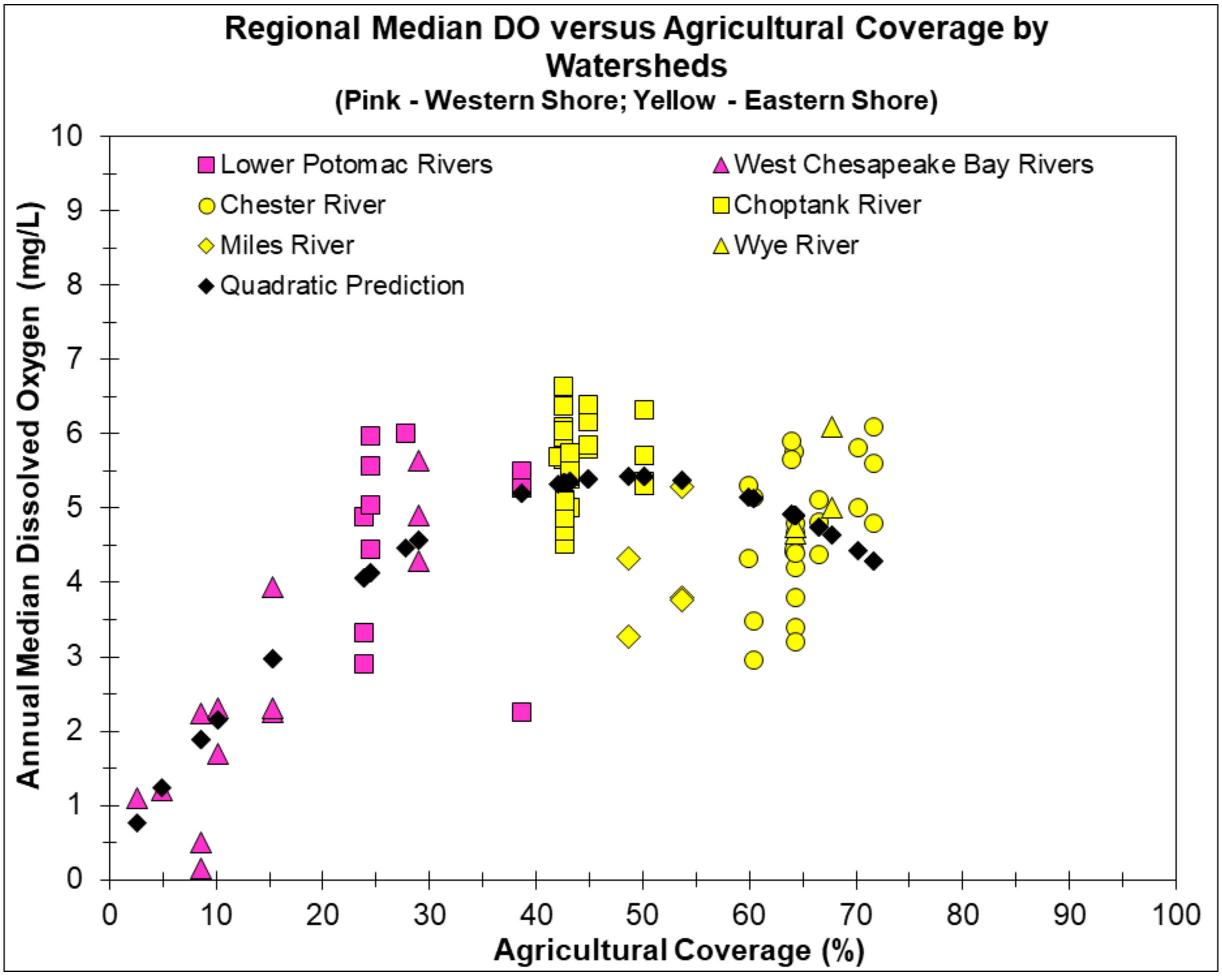

Figure 3-6. Trends in development (structures per hectare = C/ha) from 1950 to 2023 of watersheds of subestuaries sampled in 2023. Black diamond markers indicate the years that subestuaries were sampled.

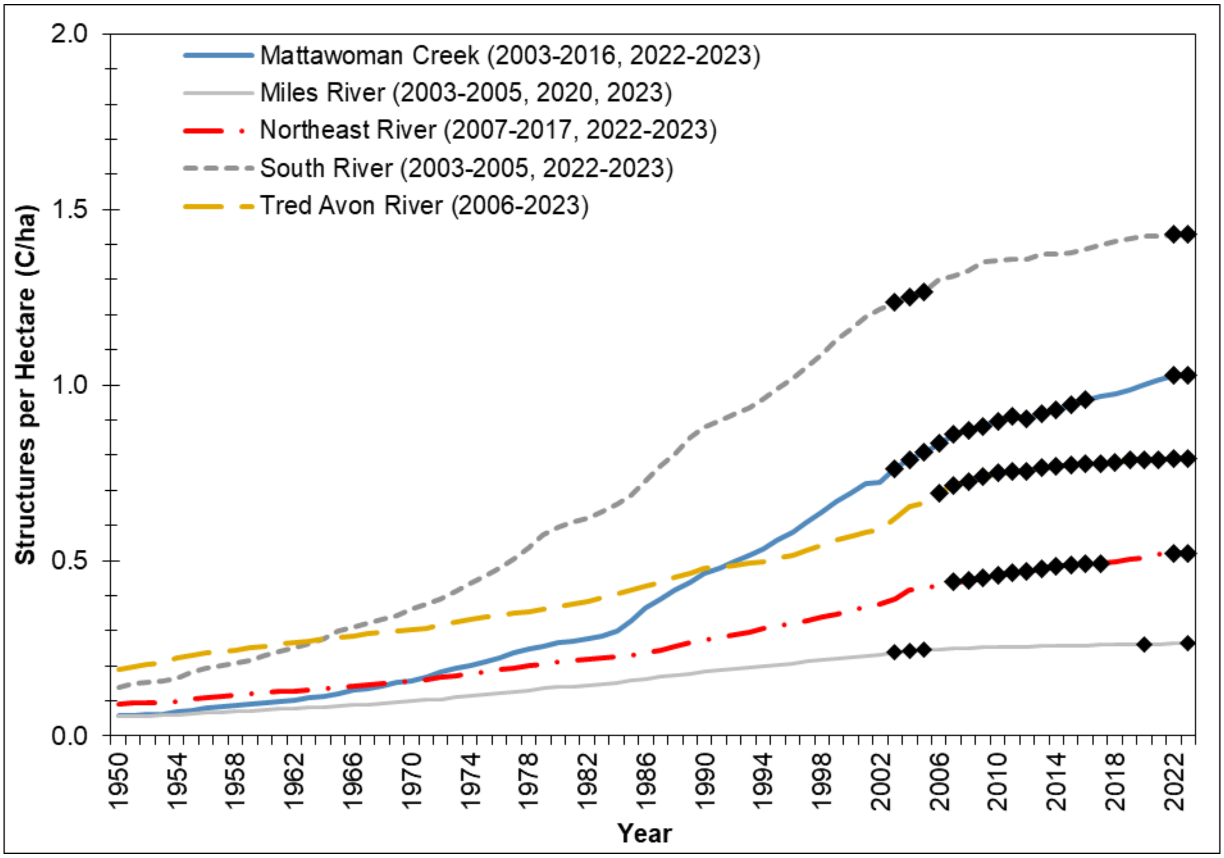

Figure 3-7. Bottom dissolved oxygen (DO; mg/L) readings (2003–2023) for subestuaries sampled in 2023 versus intensity of development (C/ha = structures per hectare). Target (5 mg/L) and threshold (3 mg/L) DO boundaries are indicated by red dashed lines.

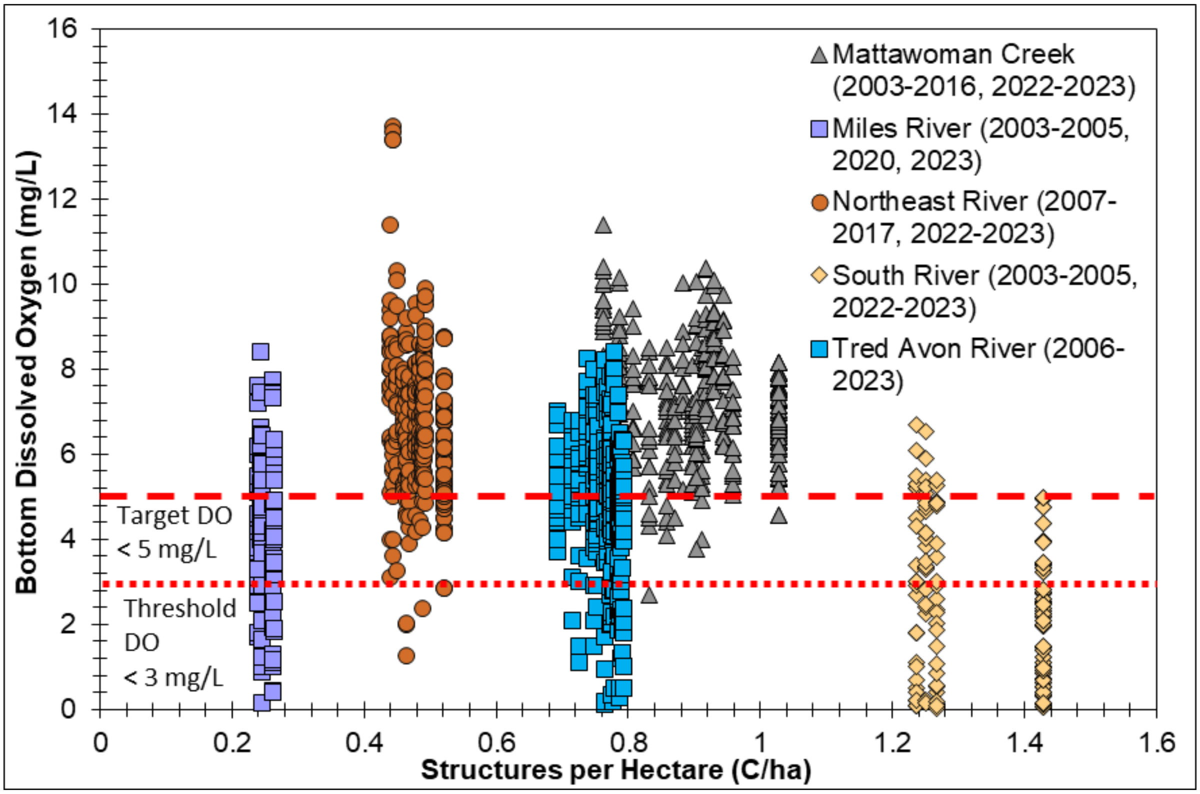

Figure 3-8. Median bottom dissolved oxygen (DO; red squares; mg/L) time-series for subestuaries sampled in 2023. Solid black bars indicate the range of bottom DO measurements for that year. The y-axes range from 0 to 15 mg/L; x-axes range are years from 2002 to 2024.

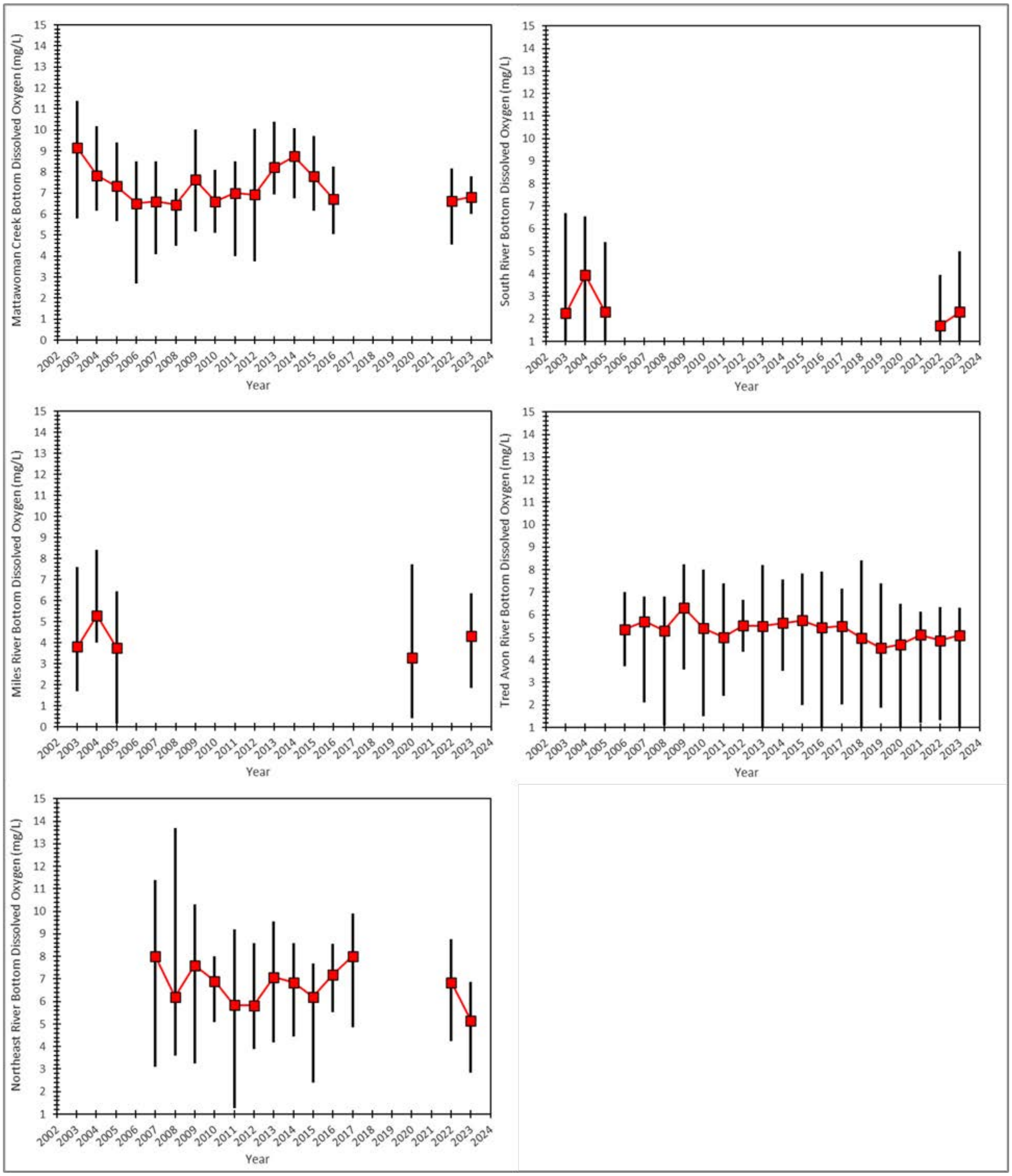

Figure 3-9. Summer (July–September) median bottom dissolved oxygen (DO; red squares; mg/L) for Chesapeake Bay Program Stations EE1.1 (Eastern Bay), MAT0016 (Mattawoman Creek), ET1.1 (Northeast River), WT8.1 (South River), and EE2.1 (Choptank River) from 1989 to 2023. Solid black bars indicate the range of bottom DO measurements for each year. Grey dashed line indicates time-series median. The y-axis ranges from 0 to 14 mg/L; x-axis ranges are years from 1988 to 2024.

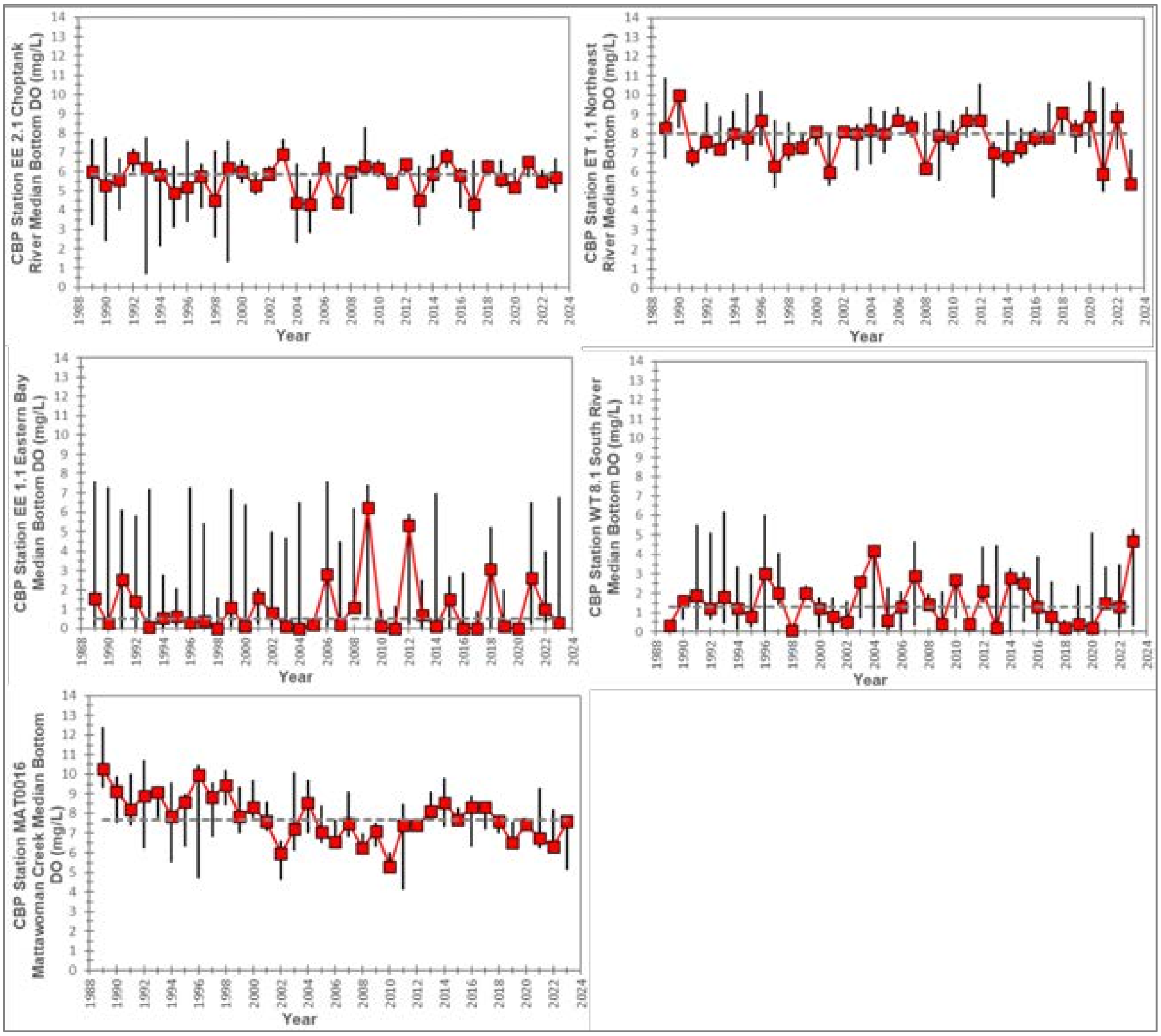

Figure 3-10. Mean bottom dissolved oxygen (DO; mg/L) time-series, by station, for subestuaries sampled in 2023. The dotted line indicates the median of all DO measurement data for the time-series. The y-axes range from 0 to 10 mg/L; x-axes range are years from 2002 to 2024.

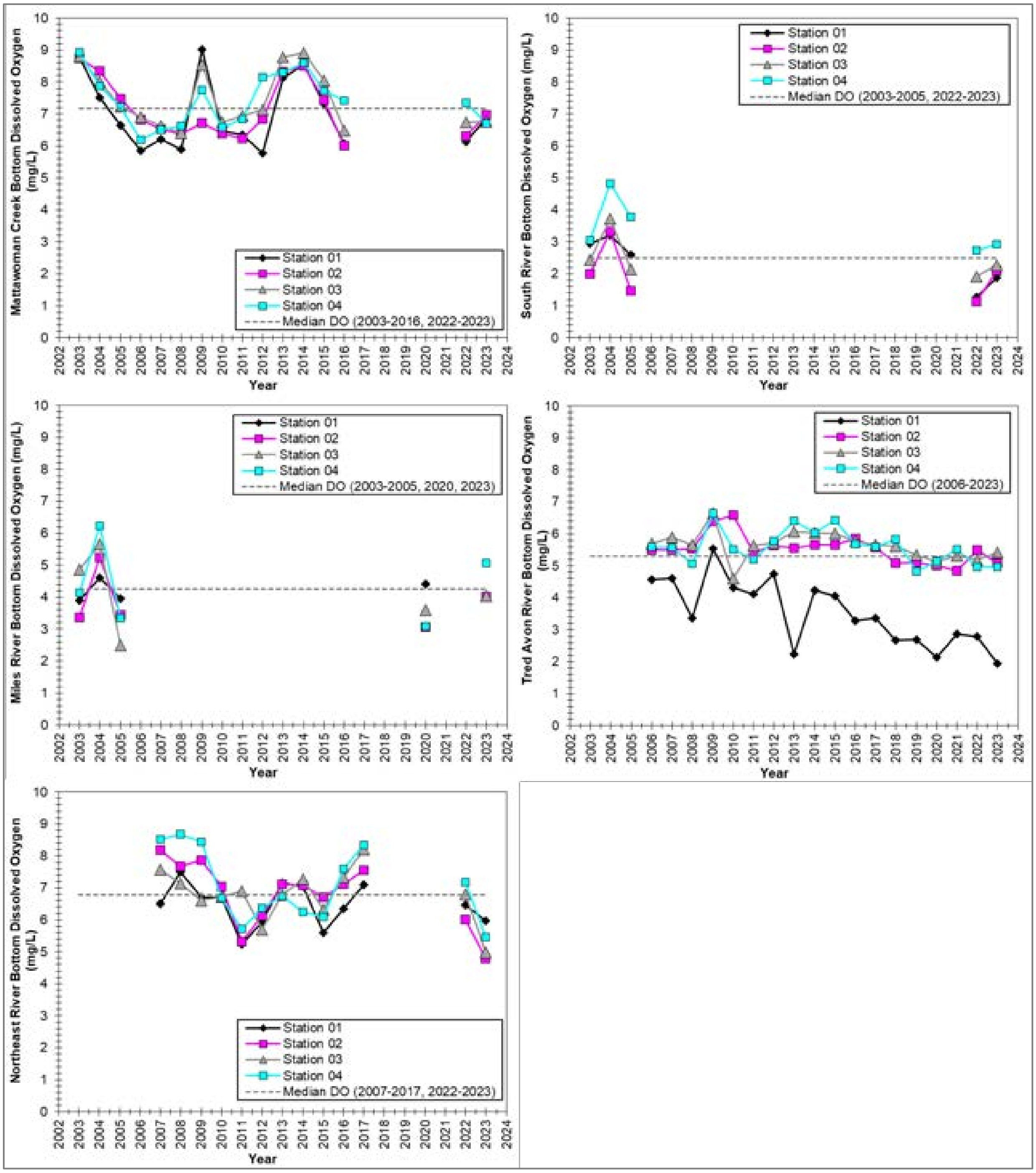

Figure 3-11. Median Secchi depth (m) time-series for subestuaries sampled in 2023. Solid black bars indicate the range of Secchi depth (m) measurements by year. The y-axes range from 0 to 2.0 m; x-axes range are years from 2002 to 2024.

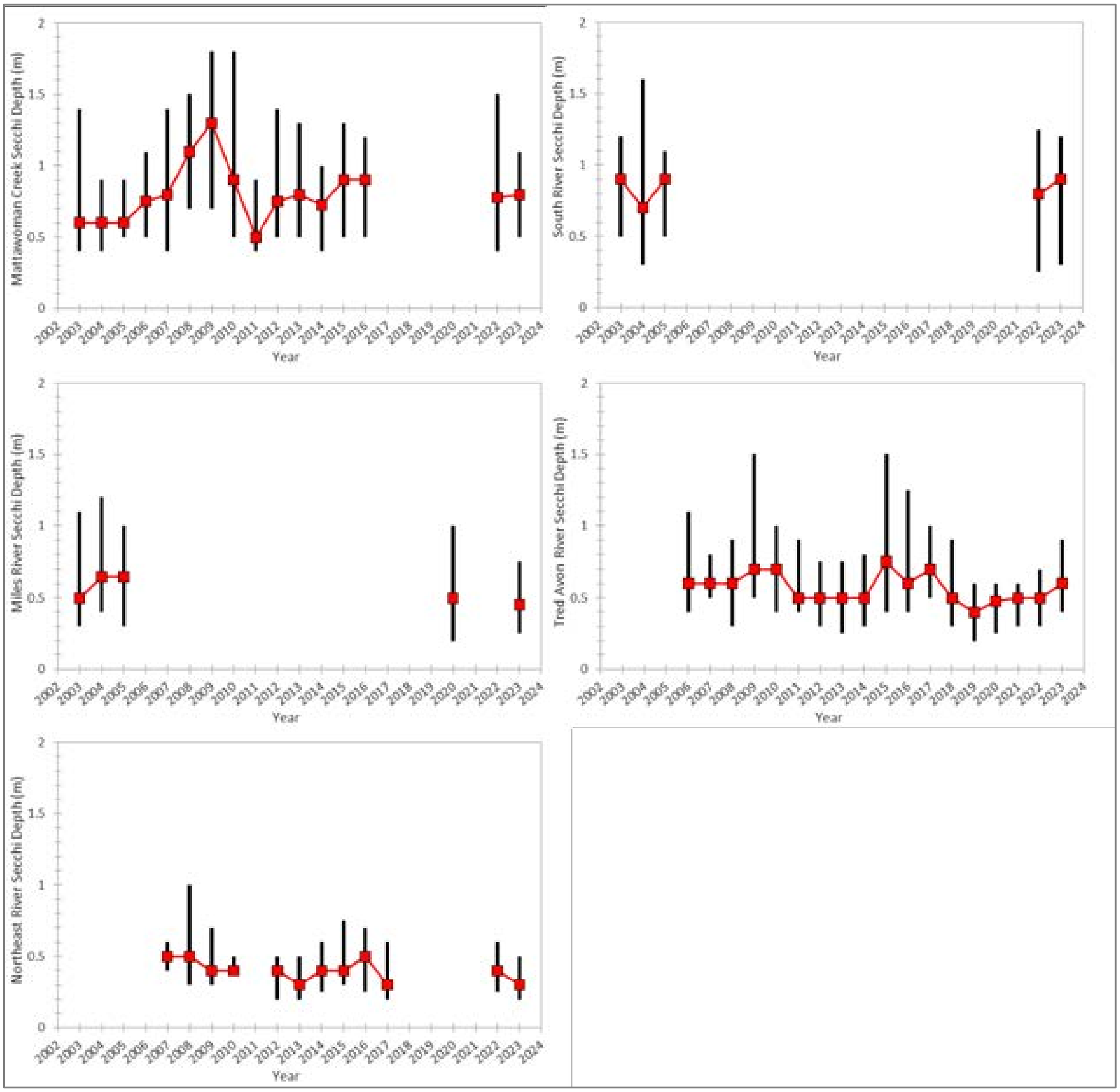

Figure 3-12. Time-series of SAV coverage (percent of water area) for subestuaries sampled in 2023. Eastern Bay includes Miles and Wye Rivers, and the Mouth of the Choptank River includes Broad Creek, Harris Creek, and Tred Avon River. Median of fully mapped years (black triangles) for the time-series is indicated by the dashed line. Partially mapped data is indicated by white squares. Data for 2023 was not available at the time of this report. The x-axis ranges from 1988 to 2024; y-axis varies for each system.

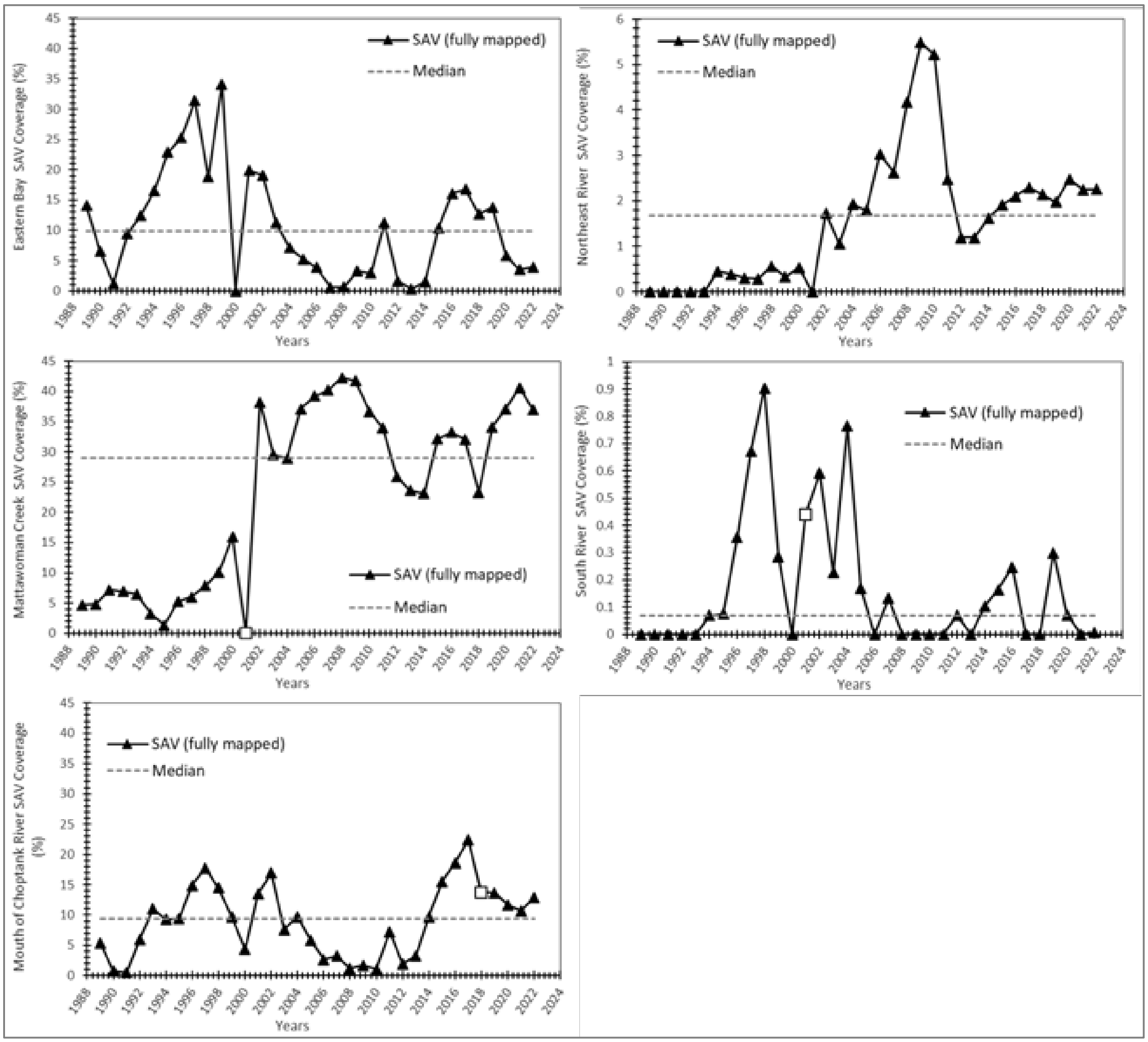

Figure 3-13. Median bottom pH (red squares) by year for subestuaries sampled in 2023. Measurements of pH were not made prior to 2006.

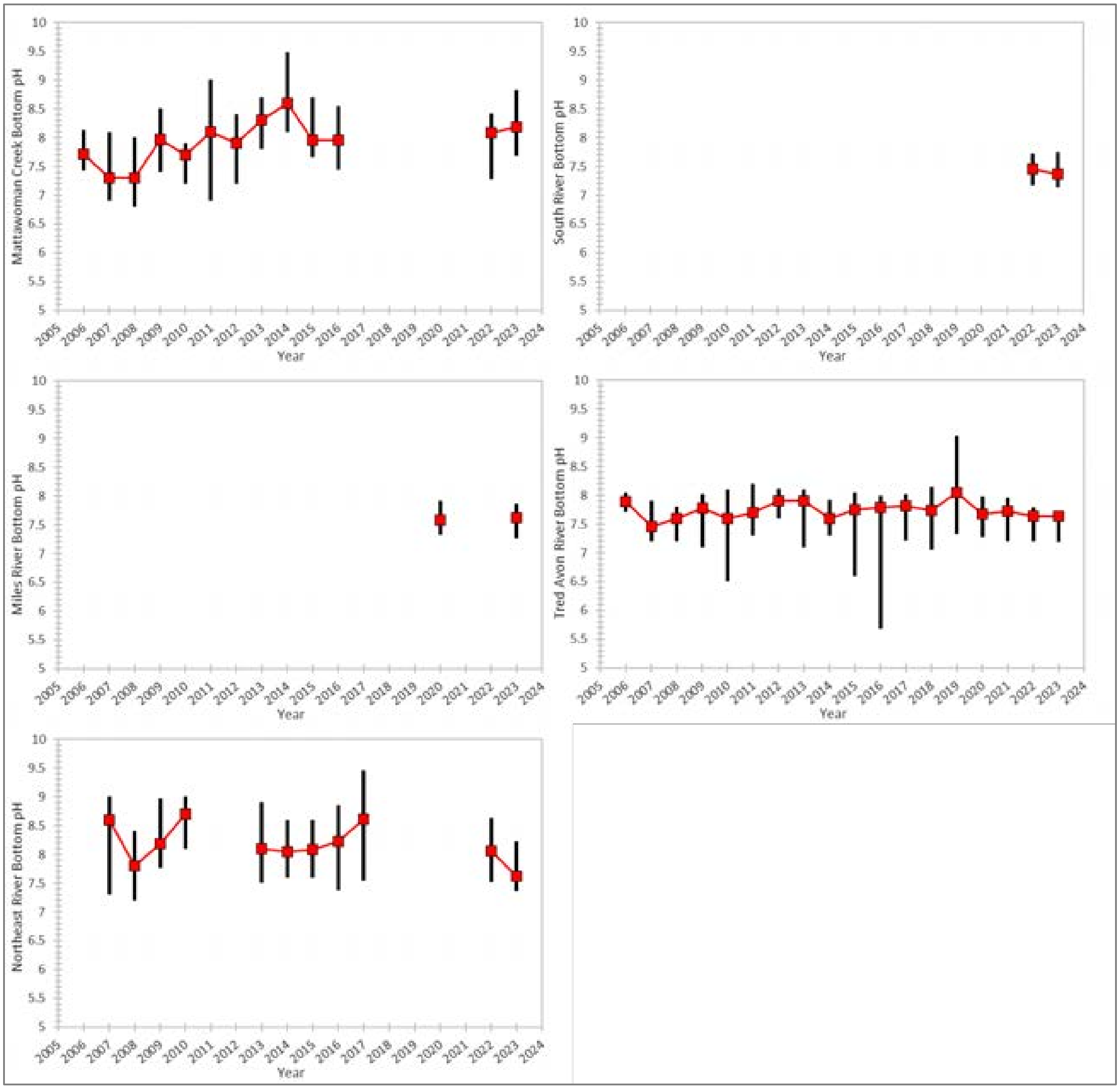

Figure 3-14. Median bottom salinity (red squares; ppt = ‰) time-series for subestuaries sampled in 2023. Solid black bars indicate the range of salinity measurements by year. Dashed line indicates tidal-fresh maximum and solid line indicates oligohaline maximum. The x-axis ranges from 2002 to 2024; y-axis vary depending on salinity classification.

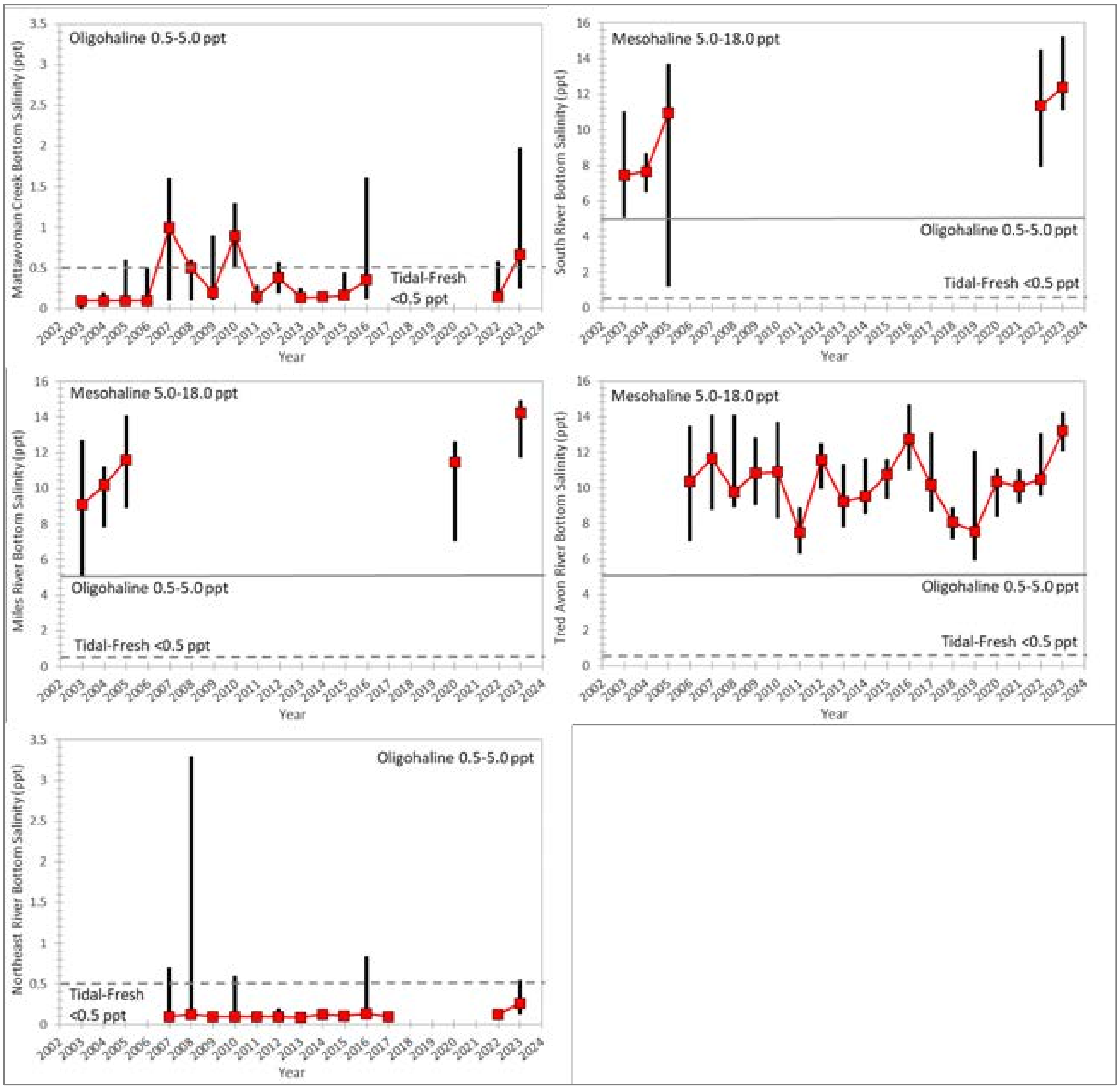

Figure 3-15. Annual number of finfish species (richness) collected by beach seines catches in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of watershed development (C/ha = structures per hectare) from 2003 to 2023. Points were omitted if beach seine effort (number of samples) < 15 samples.

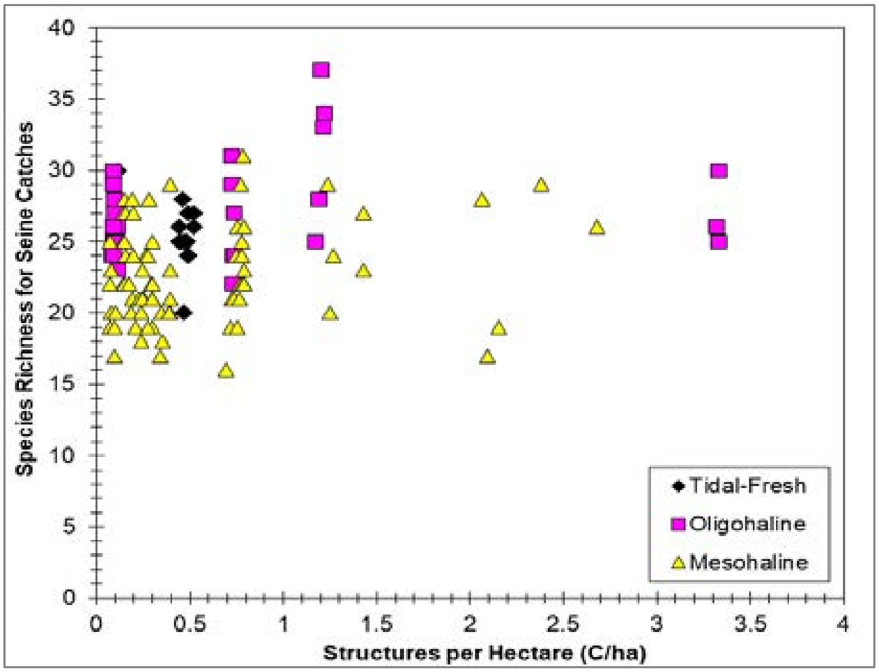

Figure 3-16. Annual number of finfish species (richness) collected by 4.9m trawl catches in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of watershed development (C/ha = structures per hectare) from 2003 to 2023. Points were omitted if beach seine effort (number of samples) < 15 samples.

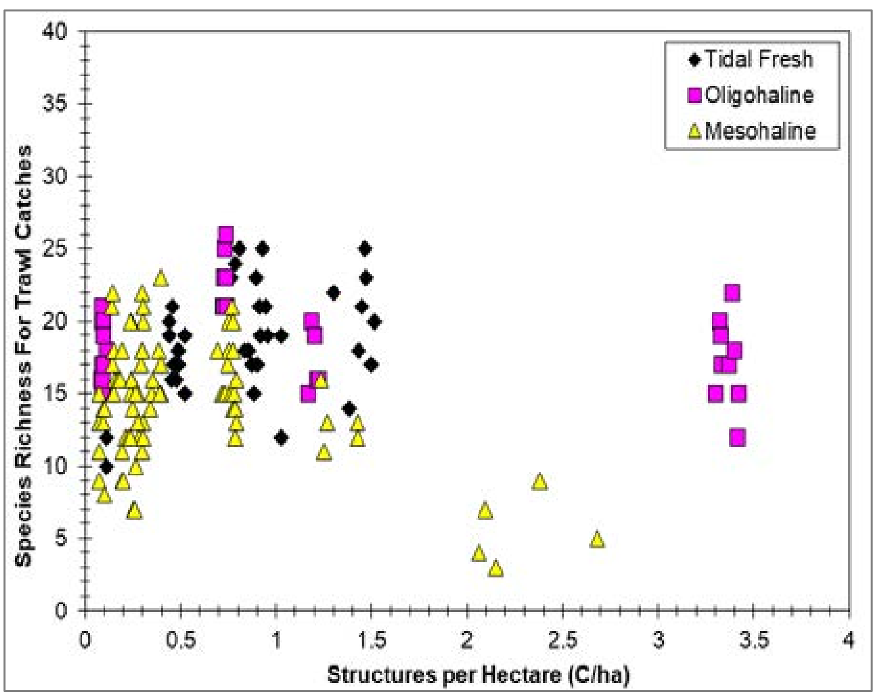

Figure 3-17. Annual 4.9m trawl geometric mean (GM) catches plotted against C/ha of subestuaries sampled during 2003–2023, by salinity class.

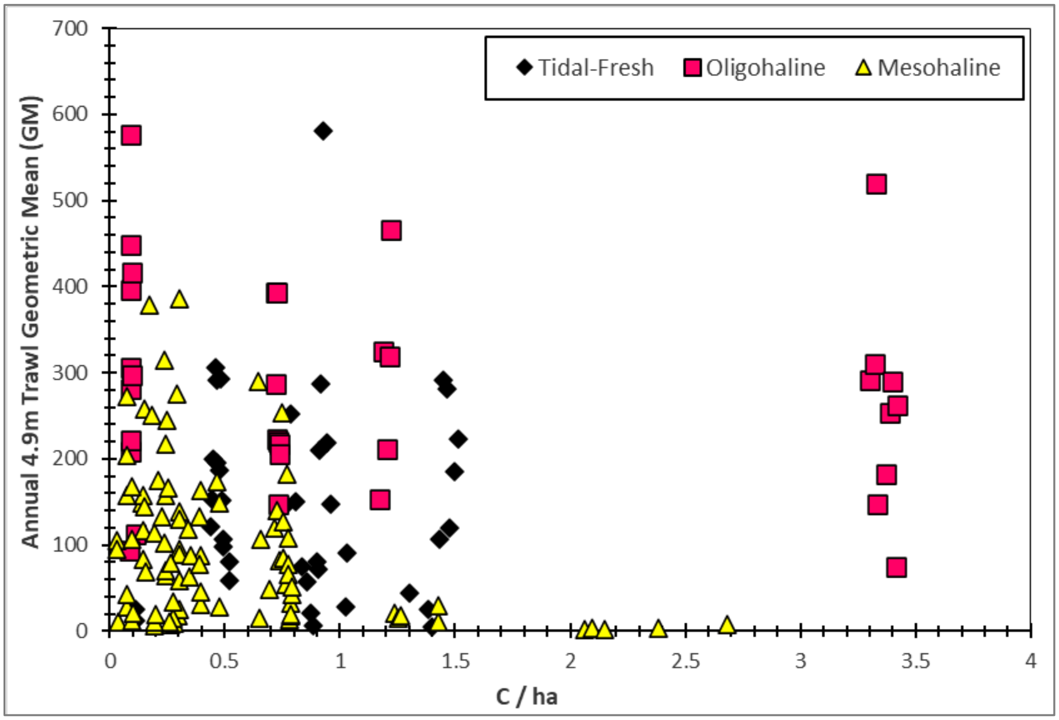

Figure 3-18. Estimates of SAV coverage (%) versus annual 4.9 m bottom trawl geometric mean (GM) catch of all species combined in Mattawoman and Piscataway Creeks for 2003–2022.

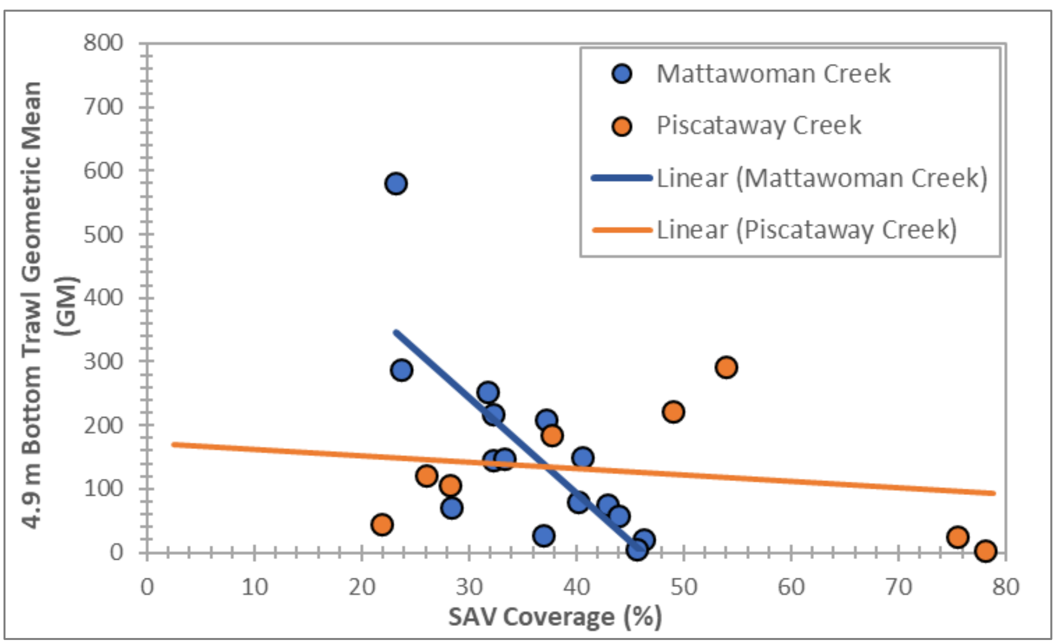

Figure 3-19. Time-series of 4.9 m bottom trawl geometric mean (GM) of catches for all finfish species (red squares) for subestuaries sampled in 2023. Black bars indicate the 95% confidence intervals.

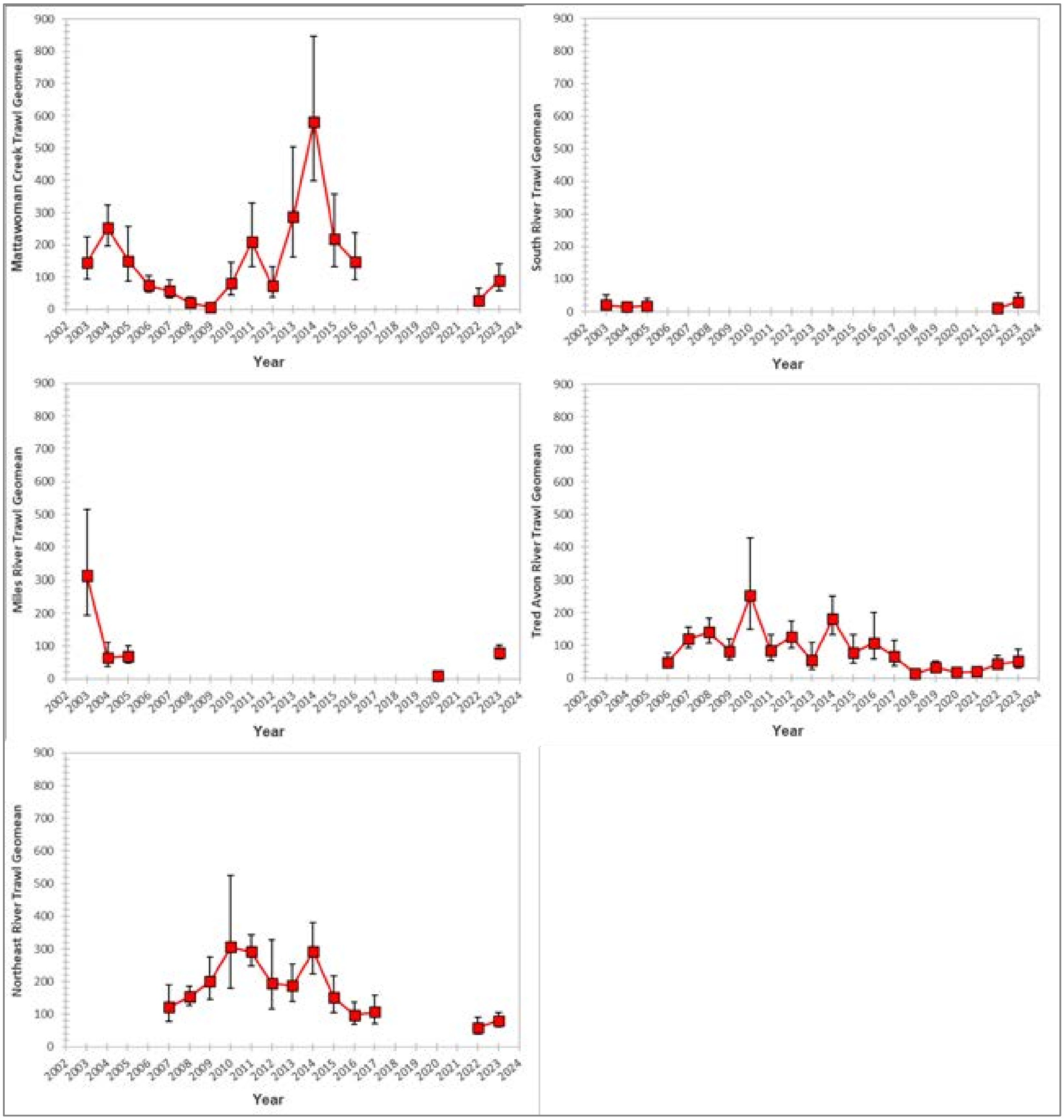

Figure 3-20. Finfish species composition for 4.9 m bottom trawl catch subestuaries sampled during 2023, for all sampling years combined. Species that define the top 90% are identified, and the remainder species are grouped and labeled as “other species”.

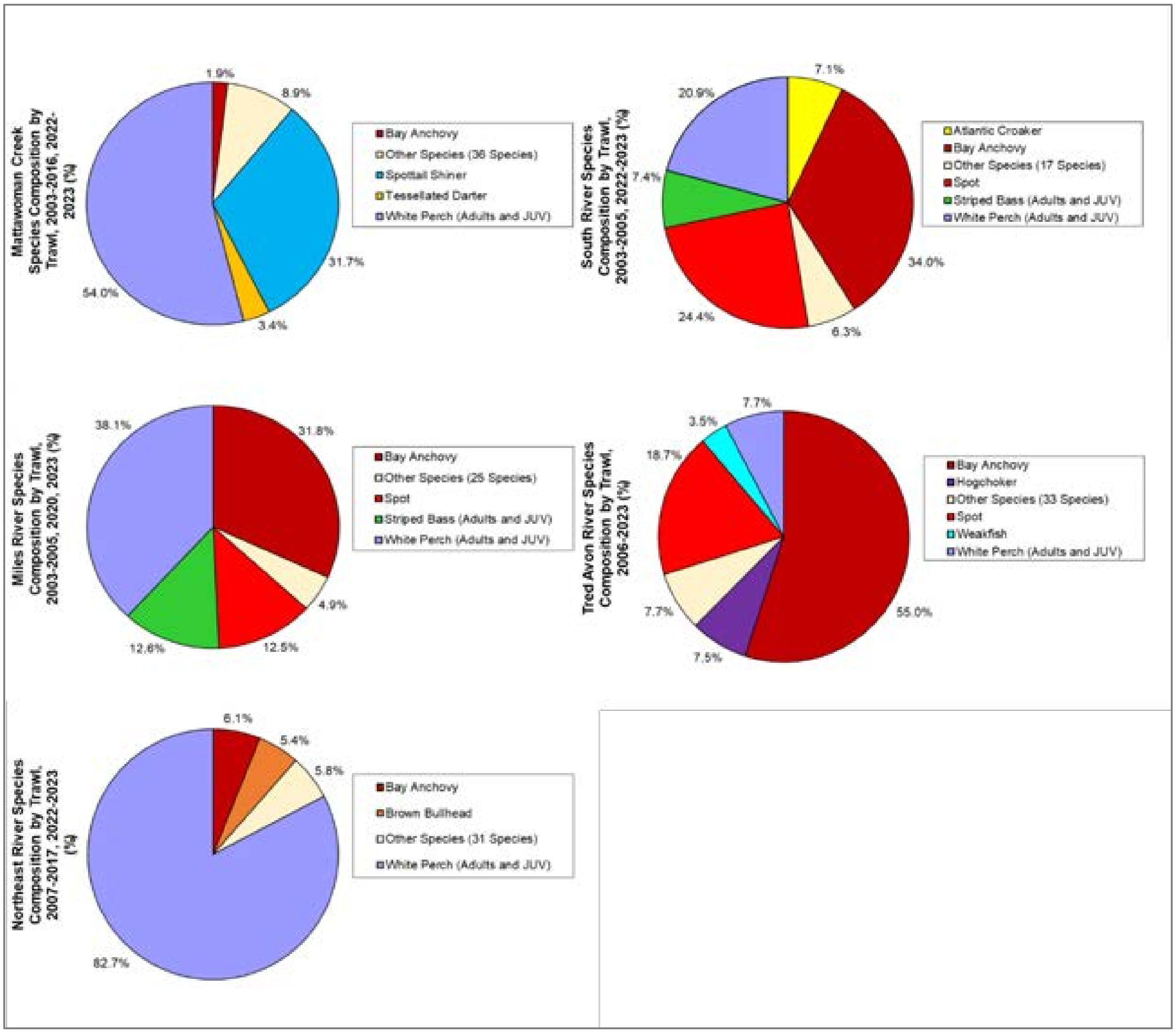

Figure 3-21. Time-series of finfish species composition for 4.9 m bottom trawl catch in subestuaries sampled during 2023. Species that define the top 90% are identified, and the remainder of species are grouped and labeled as "other species".

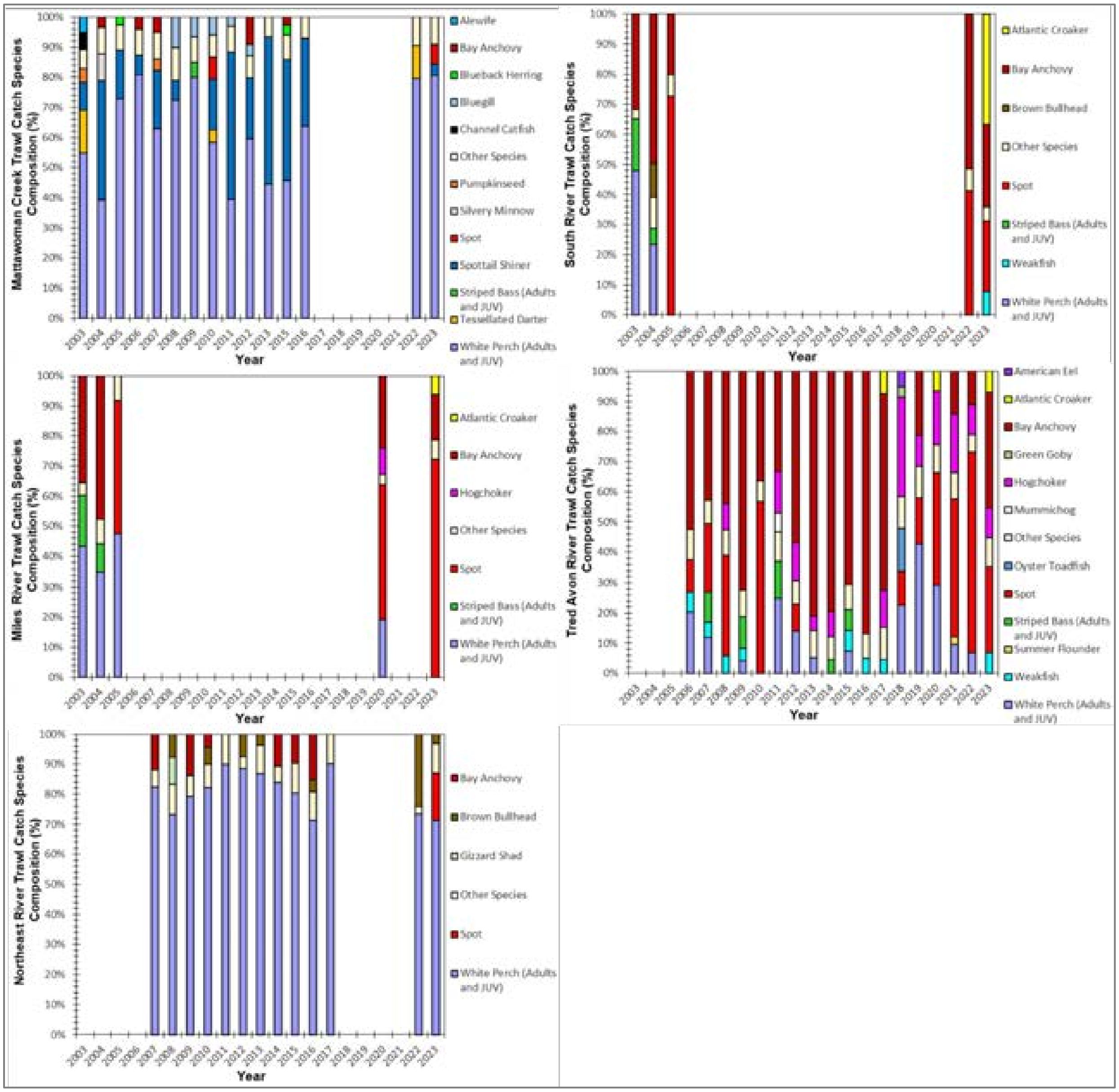

Figure 3-22. Time-series of a percent similarity index (%) for 4.9 m bottom trawl in subestuaries sampled during 2023. The greater the similarity value, the more finfish species there are in common throughout all bottom trawl stations (01–04) within the subestuary.

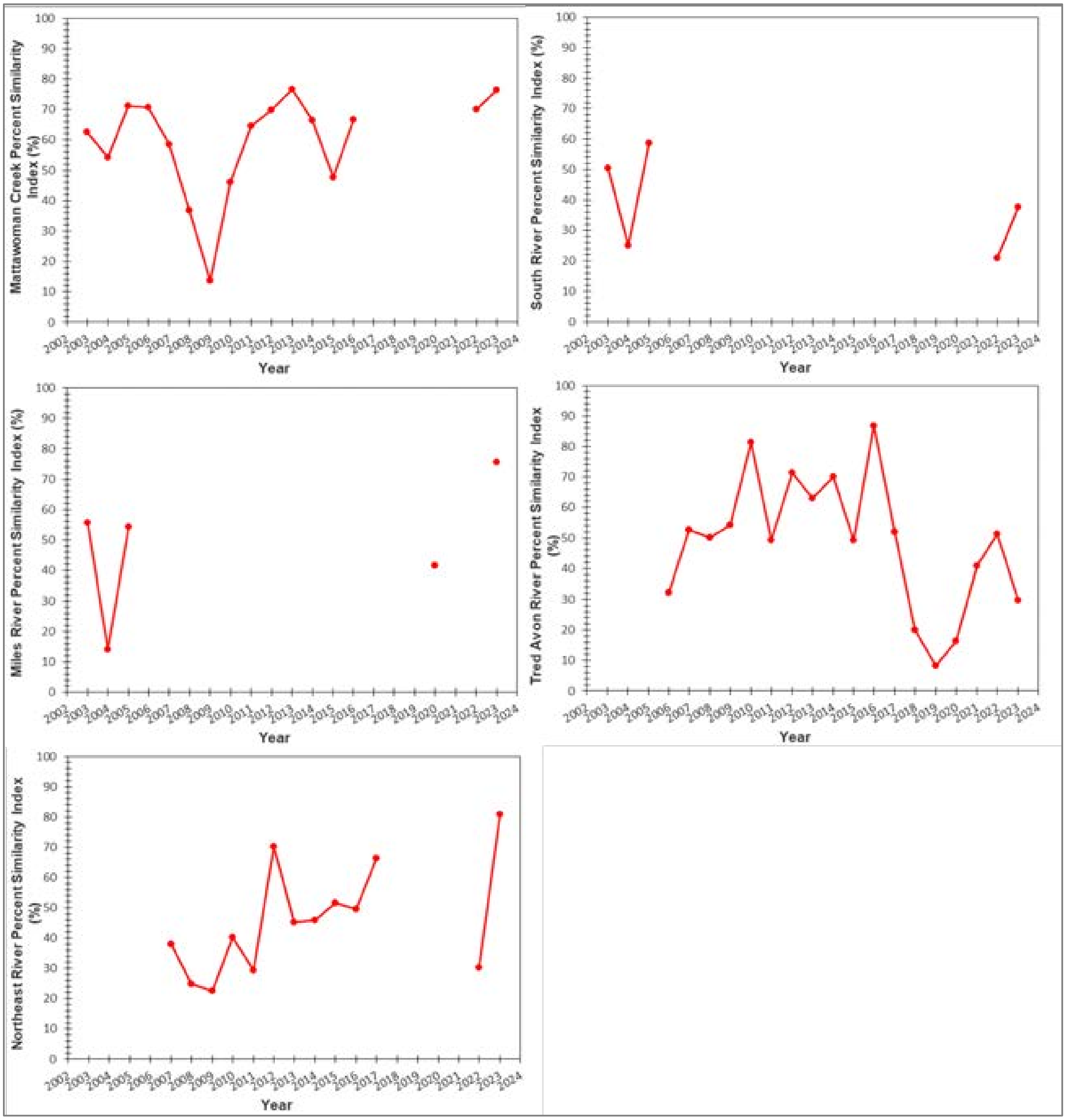

Figure 3-23. Time-series of finfish species composition for combined 4.9 m bottom trawl catch in all mesohaline subestuaries sampled during 2003–2023, by year. Finfish species that define the top 90% are identified, and the remainder of species are grouped and labeled as “other species”.

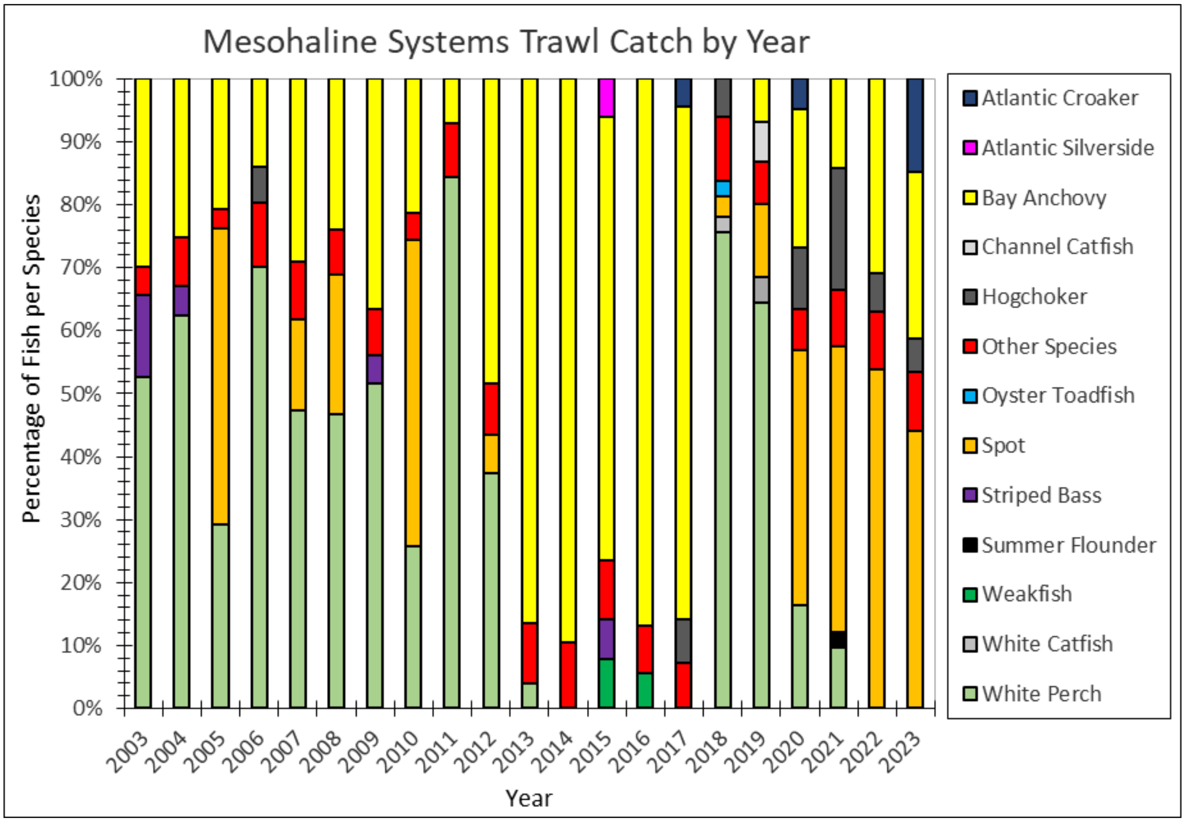

Figure 3-24. Time-series of geometric mean (GM) 4.9m bottom trawl catch of adult White Perch (primary vertical axis) and juvenile White Perch (secondary vertical axis) in subestuaries sampled during 2023. Black (juvenile WP) and red (adult WP) bars indicate the 95% confidence intervals.

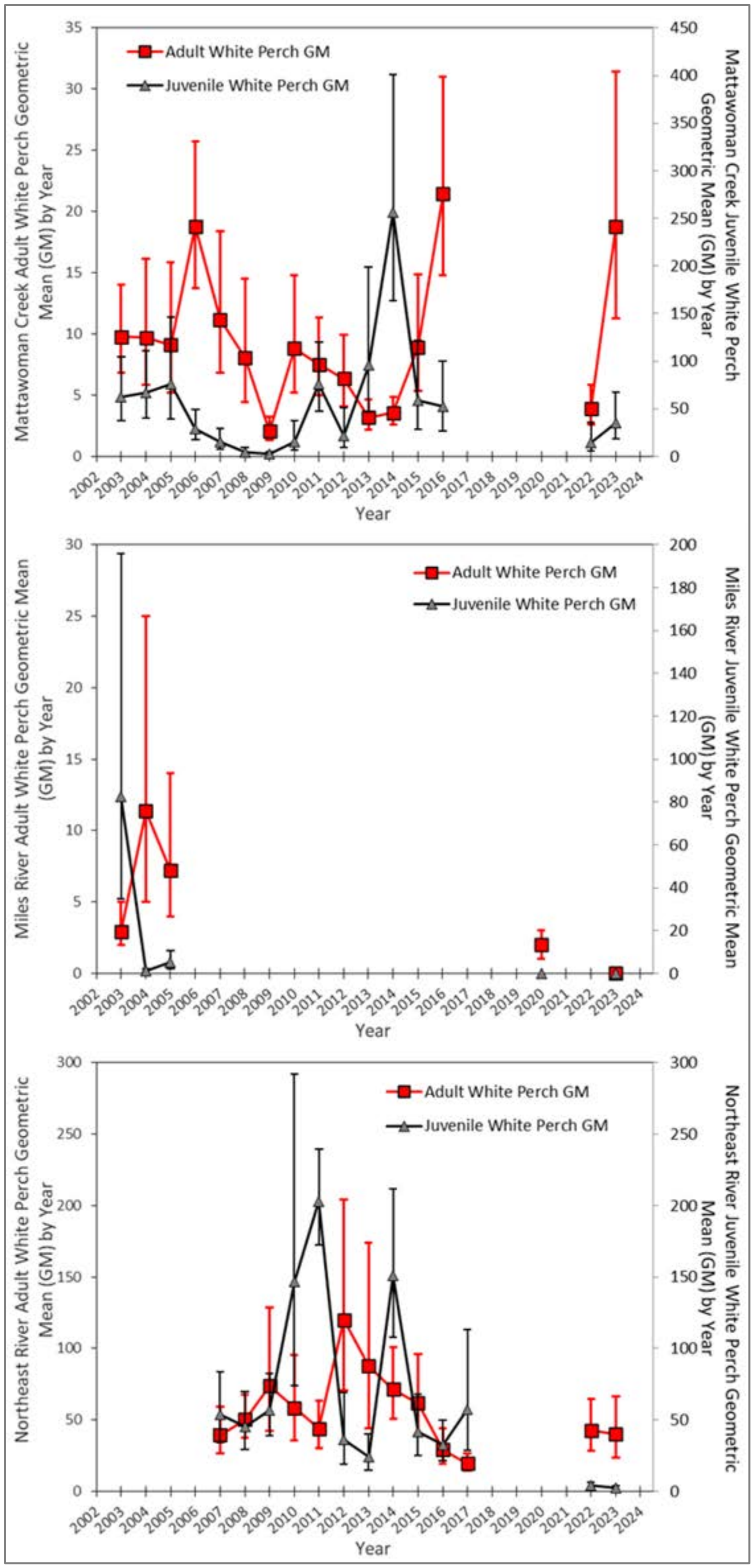

Figure 3-24. Continued.

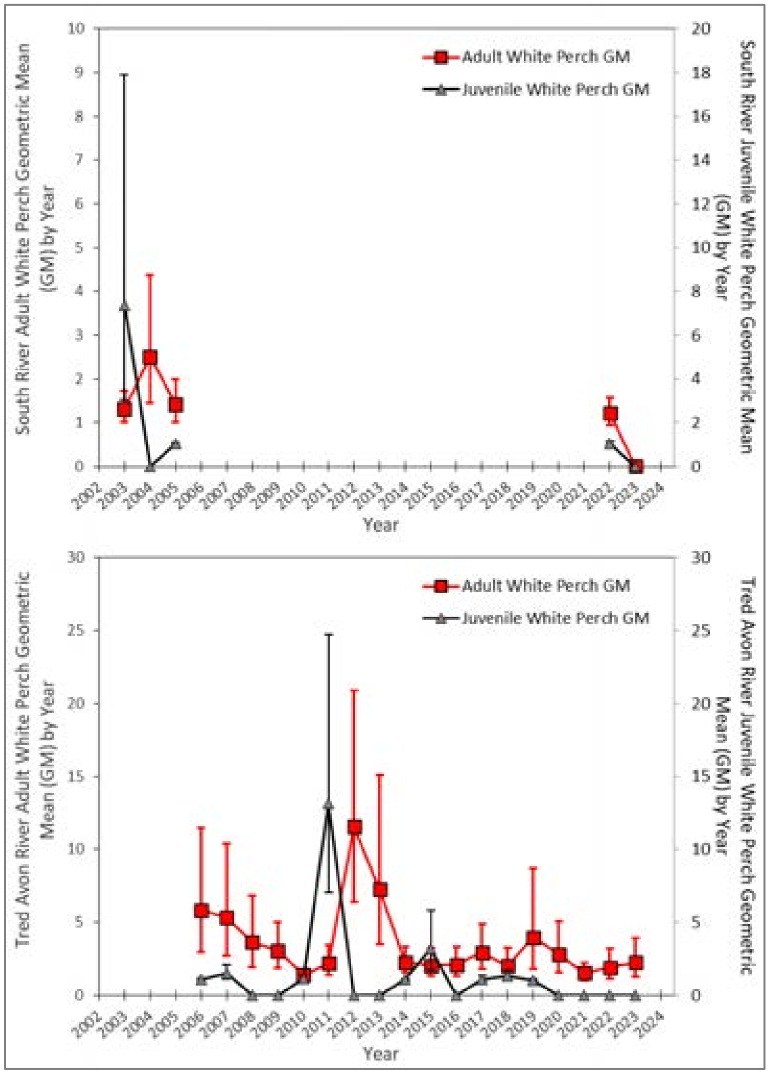

Figure 3-25. Time-series of the modified proportional stock density (PSD; %) of White Perch in tidal-fresh subestuaries, Mattawoman Creek and Northeast River, and in mesohaline subestuaries, Miles River, South River, and Tred Avon River, sampled during 2023.

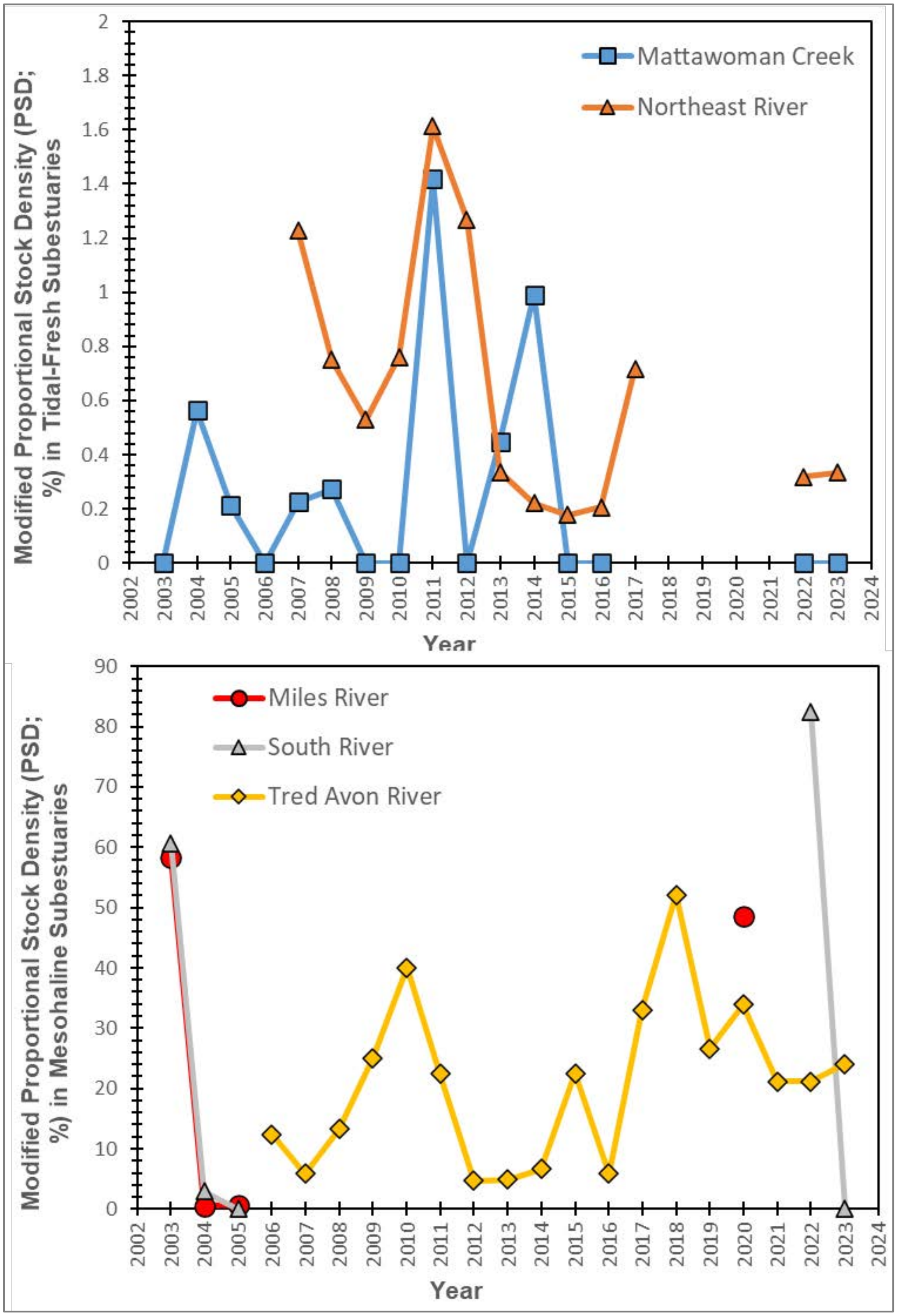

Figure 3-26. The geometric mean (GM) of bottom trawl catches of all finfish species (top) and GM of juvenile White Perch for historical trawl 3.1 m and trawl 4.9 m in Mattawoman Creek (bottom), by sampling year. Predicted 3.1 m GM is based on a linear regression of 3.1 m and 4.9 m GMs. Black (trawl 3.1 m GM) and red (trawl 4.9 m GM) bars indicate the 95% confidence intervals.

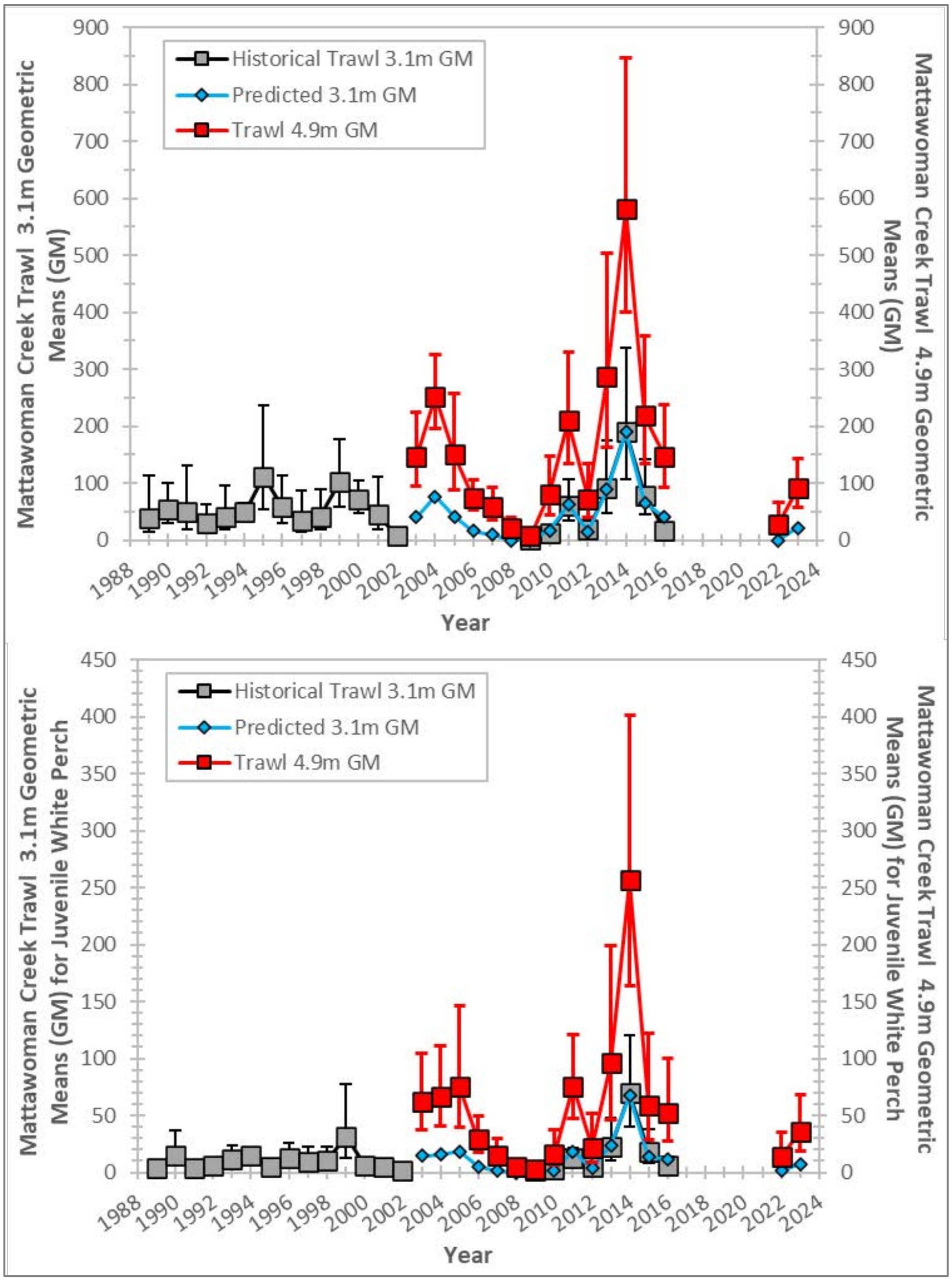

Figure 3-27. Time-series of beach seine geometric mean (GM) catches of all finfish species (red squares) for subestuaries sample in 2023.

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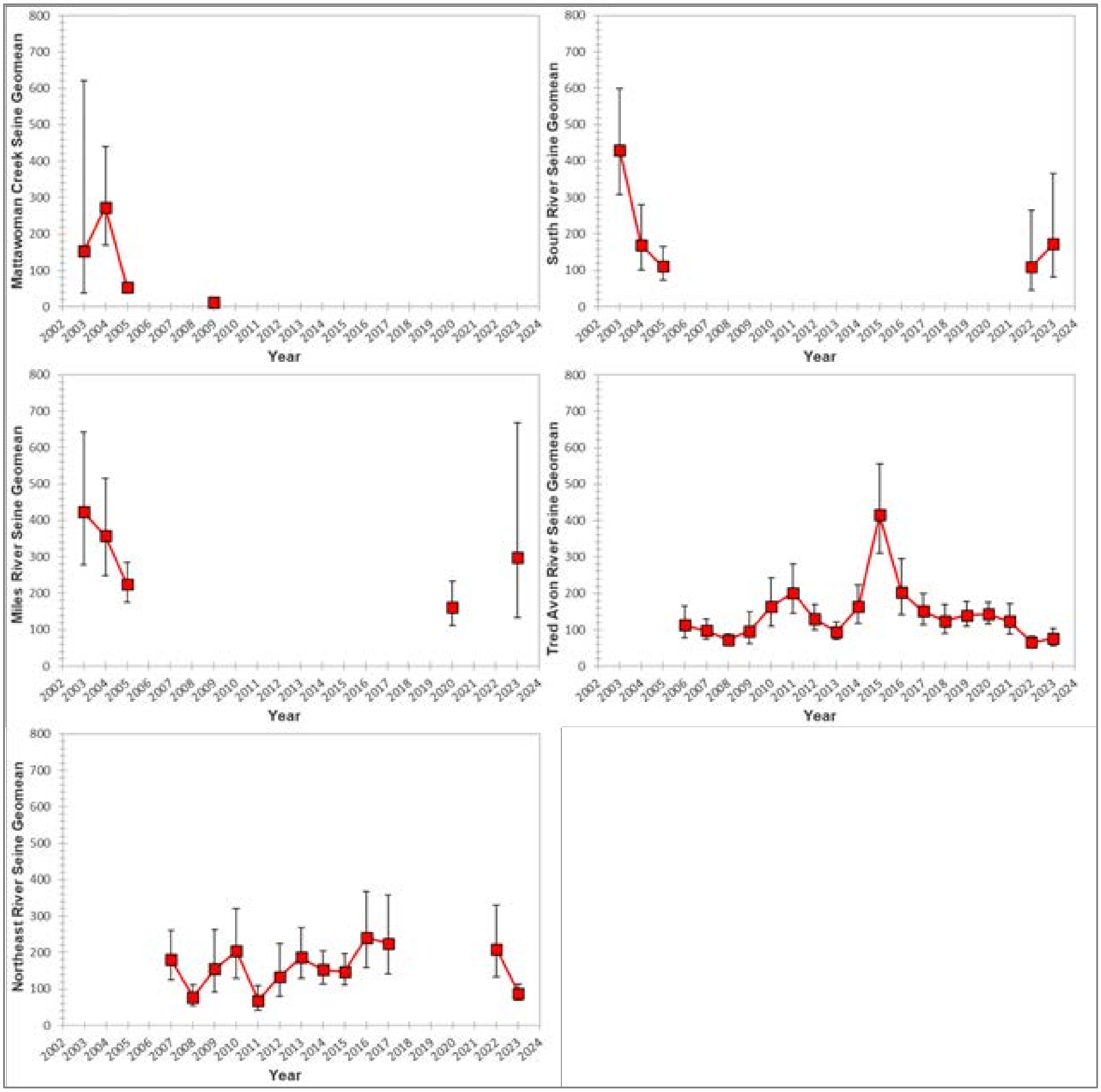


Figure 3-28. Finfish species composition for beach seine catches in subestuaries sampled during 2023 for all sampling years combined. Species that define the top 90% are identified, and the remainder species are grouped and labeled as “other species”. Upper left panel is South River.

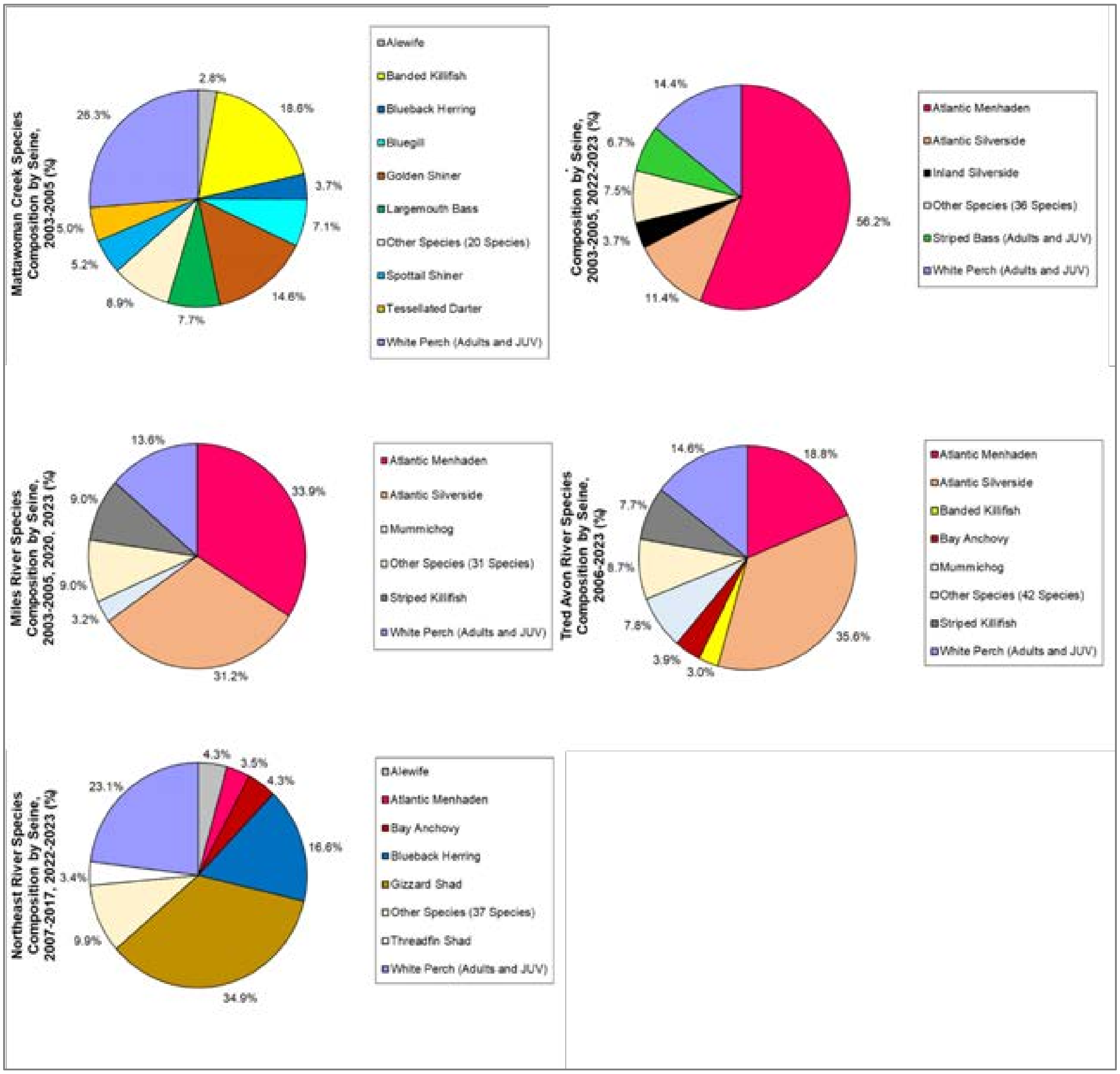
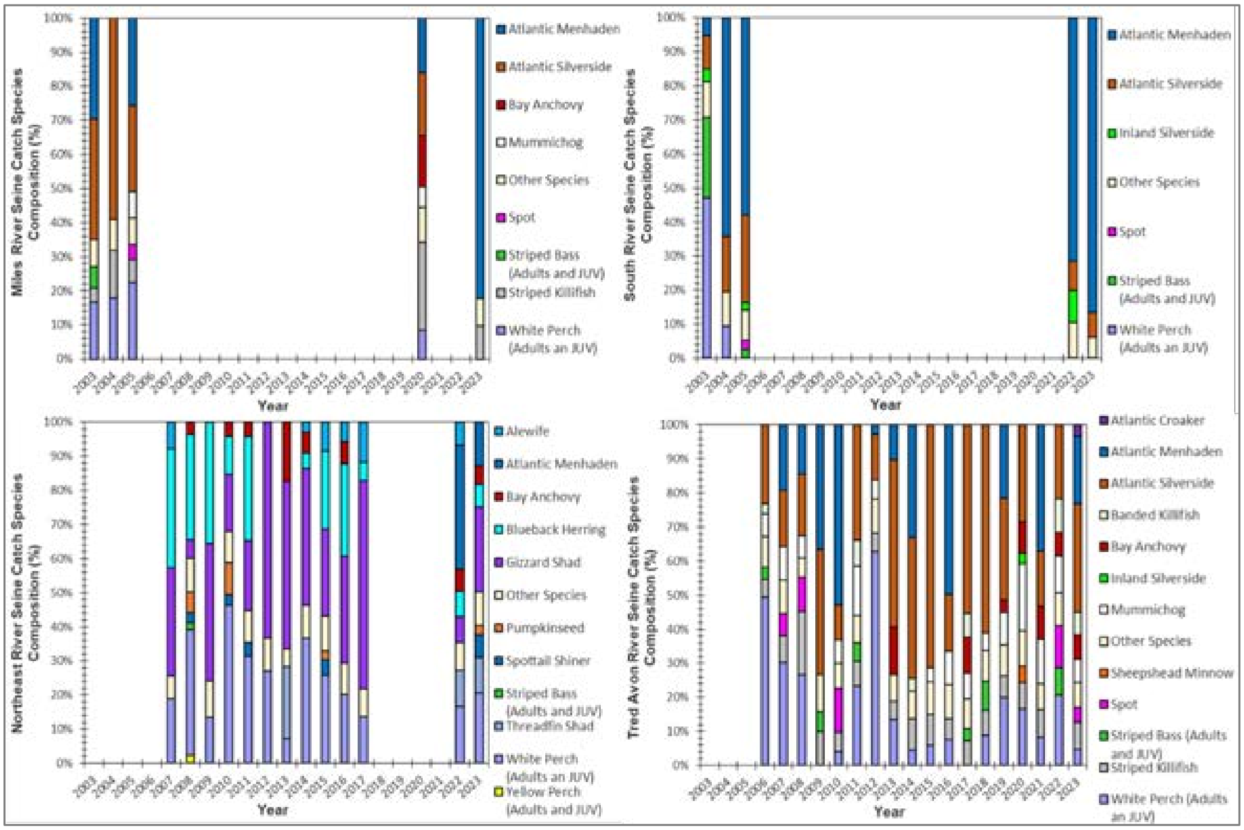


Figure 3-29. Time-series of finfish species composition for beach seine catches in subestuaries sampled during 2023. Species that define the top 90% are identified, and the remainder of the species are grouped and labeled as “other species”.



Objective 2 - Support multi-agency habitat, multispecies, and ecosystem-based fisheries management efforts by participating in multi-agency research, management, and communication forums for recreationally important finfish species found in Maryland's Chesapeake Bay and Atlantic coast

Project Staff

Jim Uphoff, Alexis Park, Carrie Hoover, Marek Toposki, Tyler Fowler, Shannon Moorhead, Marisa Ponte, and Jeffrey Horne

Introduction

Ecosystem-based fisheries management approaches require multidisciplinary expertise and coordination with local, state, and interstate agencies, nongovernmental organizations (NGOs), university researchers, and stakeholders. Contributions by Fisheries Ecosystem Assessment Division (FEAD) staff through data collection and participation with various research and management forums are vital if Maryland is to successfully develop and implement an ecosystem approach to fisheries management.

Objective 2 documents participation by FEAD 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, 2023 - June 30, 2024. These activities used information generated by F-63 or were consistent with the goals of F-63.

Fisheries Ecosystem Assessment Division 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.

A new Nontidal Anadromous Fish Spawning Map tool was made available on our website at https://dnr.maryland.gov/fisheries/pages/fhep/anadromous.aspx. Marek Toposki worked with George Edmonds (Cooperative Oxford Laboratory) to finish development of this application. This geographical information system (GIS) application is the culmination of work conducted under F-63-R-10 and F-63-R-11. Maryland Department of Natural Resources (MD DNR) 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, U.S. Environmental Protection Agency, U.S. Fish and Wildlife Service, National Oceanic and Atmospheric Administration, Maryland Department of Environment, Maryland Department of Natural Resources, Maryland Historical Trust, and Maryland's Critical Areas Commission.

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 published in Marine and Coastal Fisheries: Dynamics, Management, and Ecosystem Science, Volume 15 Issue 5 in October 2023, as part of a Striped Bass themed issue that features papers recruited from the 2021 American Fisheries Society Meeting in Baltimore. This paper can be found at https://afspubs.onlinelibrary.wiley.com/doi/10.1002/mcf2.10248.

ResearchGate was updated with annual technical reports. Performance Report for Federal Aid Grant F-63-R, Segment 4 2013 Marine and Estuarine Finfish Ecological and Habitat

Investigations has reached over 50 readers on ResearchGate, while the Performance Report for Federal Aid Grant F-63-R, Segment 6 2015 report has reached over 100 readers. The most recently added Performance Report for Federal Aid Grant F-63-R, Segment 13 2022 already has 5 reads.

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.

A. Park reviewed comprehensive growth plans for Annapolis and the towns of Denton, Millington, Port Deposit, and Rock Hall. J. Uphoff reviewed the plan for Havre de Grace.

J. Uphoff, along with Chesapeake and Coastal Services and Environmental Review staff, briefed MD DNR leadership on recent and proposed changes in zoning in the Charles County portion of Mattawoman Creek watershed set aside for limited development. These changes potentially jeopardize stabilization of development and negatively impact its fisheries and anadromous fish spawning area. These were important natural resource objectives of an interagency workgroup that worked with Charles County in the 2010s (https://dnr.maryland.gov/fisheries/documents/Mattawoman_Ecosystem_Final_Report_March_2012.pdf) to develop a groundbreaking, lower impact comprehensive growth plan that was passed in 2017. Meetings during the current grant cycle between Maryland Department of Environment, Department of Planning, and MD DNR reviewed inconsistencies between zoning actions and commitments in the Plan. A letter from the MD DNR Secretary on zoning changes in Mattawoman Creek's Watershed Conservation District (WCD) was sent to the Charles County commissioners reminding them that the WCD is needed for water quality improvement and conservation of wildlife and fish habitat.

J. Uphoff and M. Toposki provided comments to the Environmental Review Program on proposed expansion of a small private airport located in the Watershed Conservation District of Mattawoman Creek. Among the issues brought forth were absences of stormwater management and descriptions of plane and runway de-icing measures and management. The airport is on septic but the proposal said it was on sanitary sewer. The airport is located in one of two adjacent watersheds with very low levels of impervious surface compared to the rest of the watershed.

FEAD met with MD DNR's Environmental Review Program and MDE's Municipal Surface Discharge Division to discuss the proposed upgrade of the Centreville WWTP from 500,000 gallons/day capacity to 1 million gallons per day of capacity. The upgrade is currently in the planning process but the possible relocation of the discharge point of the plant further downstream in the tidal portion raised concern for impacts on anadromous fish spawning in tidal Corsica River and its tributaries. MDE will require that additional modeling be performed by the Town. FEAD compiled water quality and Yellow Perch larval presence data collected during various surveys on the Corsica River for MDE. These data were requested to analyze the mixing model and assessment of the outfall of the new wastewater treatment plant for Centreville.

Cooperative Research and Monitoring - J. Uphoff peer-reviewed an article on Striped Bass spawning habitat in the Hudson River for the Marine Ecological Progress Series.

M. Topolski worked with Fish and Wildlife Health Program staff at the Cooperative Oxford Lab to map land use for portions of watersheds (Patapsco, Patuxent, Potomac, Chickahominy, Sassafras, Wicomico, Nanticoke, Choptank, Piscataway, and Severn Rivers) as part of a White Perch disease analysis.

J. Uphoff attended multiple meetings on the possible future shift of Chesapeake Bay Program emphasis to shallow water habitats. These meetings were held to recommend priorities and outline next steps for meeting the goals and outcomes of the Chesapeake Bay Watershed Agreement leading up to and beyond 2025.

J. Uphoff met with researchers from Chesapeake Biological Laboratory (CBL; (Hongsheng Bi, Ryan Woodland, and others) prior to CBL sampling zooplankton and larvae in the Patuxent and Choptank rivers' Striped Bass spawning areas during 2024 using PlanktonScope, a shadowgraph imaging that can sample organisms from turbid waters. S. Moorhead shared digital images of larval Striped Bass and White Perch with CBL; these species can be difficult to separate and these images captured characteristics for differentiating them.

J. Uphoff provided comments to NOAA Chesapeake Bay Office and Shore Rivers on how high salinity in fall of 2023 would affect distribution of fish and other aquatic organisms. The interconnections of the Bay ecosystem make it hard to isolate a factor like salinity that is driven by climate (precipitation) and resulting flow that, in turn, drive nutrient and organic matter inputs that affect other habitat conditions (DO, food web, salinity, extent, runoff, etc.). Salinity is one of the (or maybe the) determining factors of distribution of aquatic life in estuaries. Most of the abundant estuarine organisms are adapted to handle salinity swings as juveniles and adults. Eggs and larvae are sensitive. Freshwater oriented species will have less habitat when it is dry and marine oriented species will have more.

J. Uphoff met with Barnett Rattner (USGS) regarding an upcoming USGS Osprey Survey in Choptank River and other Bay tributaries. The survey is intended to evaluate various hypotheses about nesting success. Atlantic Menhaden availability is among the hypotheses and we had a conversation about collaboration.

A. Park provided information and historical summer seine and trawl data to the SeaQL Lab at Virginia Tech and the Invasive Species Program within MD DNR.

A. Park and M. Toposki provided data to The Nature Conservancy on fish surveys and anadromous spawning habitat in the Pocomoke Sound area for a restoration project.

Presentations and Outreach - J. Uphoff took part in multiple presentations with Resource Assessment Service staff on summer habitat for resident Striped Bass given to the Sportfisheries Advisory Commission, two Chesapeake Bay Program (CBP) workgroups, and the CBP Fisheries Goal Implementation Team. The Striped Bass Program, Resource Assessment Service (RAS), and FEAD had reviewed multiple studies to develop water temperature and DO criteria that reflected summer habitat for Striped Bass likely to be encountered by Maryland's sport fishery. These criteria were summarized in F-63-R-11. Summer temperature and DO data from RAS's long-term monitoring program were compared to these criteria. 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 contributed information for a legislative report on the consequences of a summer catch-and-release season for Striped Bass.

M. Toposki and S. Moorehead attended the Maryland Water Monitoring Council Conference.

J. Uphoff, M. Toposki, S. Moorehead, and M. Ponte attended the Chesapeake Community Research Symposium 2024. J. Uphoff presented *Temperature and flow associated with recent declining Maryland Striped Bass recruitment* and *Managing expectations for fishable* *urban Chesapeake Bay waters* and S. Moorhead presented a poster, *Exploring the role of zooplankton abundance in the failure of Striped Bass year-class: an investigation of 2023.*

We worked with MD DNR's communications group on an article describing our survey of Yellow Perch and Striped Bass early life stages that appeared on Maryland DNR's Fishing and Boating Services Website https://news.maryland.gov/dnr/2024/05/07/maryland-dnr-biologists-monitor-a-critical-year-of-striped-bass-eggs-larvae/.

Interjurisdictional Management - M. Toposki attended the summer ASMFC meeting of the Habitat Committee and Atlantic Coast Fish Habitat Partnership (ACFHP) Steering Committee to finalize revisions to the strategic plan and the two-year action plan. Project proposals were reviewed and submitted to ACFHP for National Fish Habitat Partnership (NFHP) funding.

M. Topolski attended the Atlantic Coastal Fish Habitat Partnership's Steering Committee meeting. Principal issues discussed included the need to assess funded project outcomes, selection of ecologically meaningful projects, how to reduce the 18 month lag from RFP to fund disbursement, revision to the MOU including partnership dues, and mechanisms for engagement with underserved communities.

J. Uphoff and M. Topolski drafted a revision of the Striped Bass Fish Habitat of Concern description to ASMFC and worked with the Habitat Committee and its coordinator to have it incorporated into the Fish Habitat of Concern Designations for Fish and Shellfish Species document (available: https://asmfc.org/files/Habitat/FHOC_Designations_January2024.pdf). The original document was out of date.

Atlantic Menhaden and Striped Bass Traffic Light Index (TLI) - J. Uphoff worked on a TLI for Atlantic Menhaden and resident Striped Bass balance in Maryland's portion of the Bay. The TLI is intended to communicate forage status to the general public in MD based on a series of indicators. The TLI was presented to Potomac River Fisheries Commission, Virginia Marine Resources Commission, Chesapeake Bay Foundation, and Chesapeake Biological Laboratory. It appeared that the various organizations looked on the TLI neutrally to positively. It is scheduled for peer-review on August 21, 2024.

Training - A. Park attended and completed NOAA vessel training at the Cooperative Oxford Laboratory.

Objective 3: Develop spatial data to assist in conserving priority fish habitat.

Marek Topolski

Nontidal Anadromous Fish Spawning Map tool - Marek Topolski worked with George Edmonds (Oxford Cooperative Laboratory) to finish the Nontidal Anadromous Fish Spawning Map tool (available at https://dnr.maryland.gov/fisheries/pages/fhep/anadromous.aspx). This GIS application is the culmination of work conducted under F-63-R-10 and F-63-R-11. 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 use these maps to limit habitat impacts, but also to identify potential locations to apply mitigation approaches.

Impervious surface estimates for catchments - Marek Topolski evaluated models to estimate percent impervious surface (%IS) for large scale catchments (mapped at 1:24,000) available from the USGS NHDPlus HR (https://www.usgs.gov/national-hydrography/nhdplus-high-resolution). The target users for these estimates were staff conducting environmental reviews. Two issues arise at the catchment scale: accuracy of tax data coordinates and lack of explicit road data.

The tributary catchments that intersect land (n = 104,203) are substantially smaller (mean = 29 ha) than the MD DNR 12-digit watersheds (n = 1,120; mean = 2,882 ha) used to develop the tax-based index of %IS (see General Spatial and Analytical Methods used in Project 1 Sections 1-3). Tax data coordinates (property parcel centroid) are not necessarily the primary structure's location which increases the risk of tax data being assigned to the wrong catchment. This misalignment between tax data and actual impervious surface location becomes increasingly relevant as catchment size decreases (minimum size = 0.01 ha; 27,686 <1 ha). The tax index implicitly incorporates a relationship between road density and structure density. At the large scale of catchments, exclusion of road data will increase %IS estimate error for catchments where roads exist absent of structures.

Both MD tax data and U.S. Census Bureau TIGER (Topologically Integrated Geographic Encoding and Referencing system) road centerline data have been considered as predictors for model development. Both data sources are readily available for years corresponding to the Chesapeake Conservancy's high resolution 2017/2018 land use/land cover data. Effort to date has focused on the generalized additive model (GAM) framework. Future work should consider alternate model frameworks, Bayesian, for example, and data processing steps to improve both accuracy of tax data coordinates and road area.

Shoreline hardening analysis – Chesapeake Bay Program has identified shoreline hardening as a driver of shallow water fish habitat distinct from watershed development.

Introduction

Substantial increase to projected sea level rise (IPCC 2007) will increase pressure for shoreline stabilization as a means to protect cities and residences from coastal flooding (Arkema et al. 2013). Shoreline hardening (i.e. bulkheads and riprap) is one mechanism to stabilize a shoreline and reduce erosion though it will have varying effects on living resources (Kornis et al. 2017; Batiuk et al. 2023). For example, abundance of some Chesapeake Bay forage species declined when shoreline hardening reached threshold values from 10-30% per kilometer (Vogt et al. 2023). Implementation of living shorelines as a substitute for hardened shorelines to benefit living resources, in addition to meeting total maximum daily load (TMDL) requirements, should be a consideration when identifying restoration actions (Scientific and Technical Advisory Committee [STAC] 2023).

Ecologically centered shoreline conservation and stabilization efforts may lag behind hardened impacts resulting in perpetual shoreline restoration unless appropriate predictors of shoreline hardening are identified. Vulnerability to physical damage is a predictor of shoreline hardening, although better predictors of shoreline hardening in sheltered (non-ocean facing) areas such as Chesapeake Bay are housing density and gross domestic product (Gittman et al. 2015). Shoreline hardening has been treated as an independent influence on fish habitat by the Chesapeake Bay Program but its influence may be conflated with watershed development. This analysis is to determine if shoreline hardening in the Maryland portion of Chesapeake Bay is independent of development intensity. I will consider multiple types of hardened shoreline and adjacent land uses across a development intensity gradient.

Methods

Catchments from the USGS NHDPlus HR dataset (https://www.usgs.gov/national-hydrography/nhdplus-high-resolution) were used to spatially define sample locations. Each catchment selected intersected the Chesapeake Bay Critical Area (1000 ft of tidal water). The Maryland Critical Area Boundary Map Viewer (https://webmaps.esrgc.org/cbca/) contained GIS polygon layers delineating the Critical Area Designations (approved and in-progress) which were used to select NHDPlus HR catchments (n = 38,052). Land area and percent impervious surface (%IS) derived from Chesapeake Conservancy 2017/2018 land cover data (see General Spatial and Analytical Methods used in Project 1 Sections 1-3) were calculated for each catchment as a whole (Catch) and the Critical Area (CA) portion of each catchment. Percent impervious surface was categorized into 20 bins of five %IS ranging from 0-5 to 95-100.

Maryland Chesapeake Bay shorelines were inventoried for each jurisdiction having tidal waters during the years 2020-2023 by the Center for Coastal Resources Management, at the Virginia Institute of Marine Science, William & Mary (https://www.vims.edu/ccrm/research/inventory/maryland/). Shorelines were inventoried by the riparian zone land use and bank condition (LUBC shapefile) and the presence of structure for shoreline defense (SSTRU shapefile). Each type of shoreline data for each jurisdiction were combined with the Merge tool to create a statewide LUBC shapefile and SSTRU shapefile. The geoprocessing tool Identity was used to overlay and combine the statewide LUBC (input) and SSTRU (identity feature) shapefiles into a single shoreline shapefile (herein CCRM2020). Structures that were adjacent to or extended from the shoreline were omitted: 510 of 512 breakwater; 1,985 of 1,986 groin; 507 jetty; and 1,592 of 1,625 marsh toe. Several shoreline types were combined into broader categories based on the nature of construction materials (Table 1); for example, breakwater, groin, and marsh toe are constructed of the same material as riprap. Wharfs were not grouped in with bulkheads because the structural component of the shoreline can vary. Shoreline segments lacking structure were labeled as natural. An additional shoreline attribute was coded as 1 = structure and 0 = no structure. The structure category debris used for shoreline segments where materials were "… haphazardly scattered and not providing shoreline protection" was not coded as structure. Catchment shorelines were evaluated by the probability of each shoreline type being present and the percent length of each shoreline type. Ninety-five percent confidence intervals (CI) were constructed as a measure of variability around each mean (x) with the equation

$$CI = x \pm z \cdot \left(\frac{\sigma}{\sqrt{n}}\right)$$

where z was the z-score, σ was the standard deviation, and n was the sample size (Sokal and Rohlf 2000).

Shorelines depicted by the CCRM2020 shapefile were spatially incongruous with the Critical Area Boundary shapefile, so whole catchments were used as boundaries to quantify shoreline parameters. Shoreline types and land use were quantified as presence/absence and percent of shoreline length within each catchment using the Summarize Within geoprocessing tool. Catchments that lacked CCRM2020 data did not have shoreline and were excluded from further analysis reducing the number of catchments to 20,139. Percent IS could not be calculated for catchments that lacked land area and were removed from further analysis leaving 20,113 whole catchments and 20,098 Critical Area only catchments.

Results

Whole catchment and Critical Area portion estimates of %IS were usually comparable, average difference = 0.1% (0.45% difference), but substantial differences did occur with a maximum difference of 98% (1,358% difference). Eighteen percent (n = 3,700) of catchments intersecting the Critical Area had shoreline structure present; 1% of the catchments had shorelines that were 100% structure. Positive trends between %IS and both probability of structure and percent of shoreline length as structure were comparable between the whole catchment and the Critical Area component of the catchment. The probability of shoreline structure being present was 0.086 (CI ± 0.004) when %IS was 0-5%, increased to 0.62 (CI_{Catch} ± 0.033, CI_{CA} ± 0.034) when the catchment had 5-10% IS, then tapered into a positive sigmoidal curve through to 90-95% IS (Figure 1 A and C). Confidence intervals began to increase when IS exceeded 20-25% and were large enough when IS was 45-50% to be uncertain about the extent of continued increase in probability of shoreline structure being present. Percent of catchment shoreline length that was structure increased from 2.3% (CI ± 0.17) at 0-5% IS to between $26.9\%_{Catch}$ and $27.1\%_{CA}$ (CI ± 2.1%) at 5-10% IS then increased and tapered into a positive sigmoidal curve through 90-95% IS (Figure 1 B and D). Confidence intervals for the percent of shoreline length began to increase substantially at 25% IS. Results for the whole catchment and Critical Area component of the catchment were comparable and so only whole catchment results are reported here on out; although, the Critical Area results are provided in part B of figures 3-4 and 7-9.

Catchment shorelines were often comprised of more than one type. Independent of development intensity (%IS), natural shoreline had a high probability (0.99 [CI ± 0.002]) of being present (Figure 2A) while the probability of structure being present was substantially lower: riprap = 0.15 (CI ± 0.005), bulkhead = 0.11 (CI ± 0.004), unconventional = 0.017 (CI ± 0.002), debris = 0.009 (CI ± 0.001), and wharf = 0.004 (CI ± 0.001). Natural shoreline occupied 94% (CI ± 0.24) of catchment shoreline length when present; however, when catchment shorelines were hardened, the two dominant types were riprap (29% [CI ± 0.96] of length) and bulkhead (25% [CI ± 1.1] of length; Figure 2B). Less common shoreline hardening structures were wharfs (15% [CI ± 4.7] of length) and unconventional (7.2% [CI ± 1.4] of length); the occurrence of debris (5.4% [CI ± 1.1] of length) was uncommon.

Percent IS was linked to the use of shoreline structure but was not an absolute determinant. The probability of natural shoreline generally remained above 0.9 as %IS increased until IS was greater than 50-55% at which time natural shoreline became less common with greater variability in presence (Figure 3A). The probability of bulkhead and riprap being present increased dramatically between 0-5% IS and 5-10% IS. Bulkhead presence went from 0.036 (CI ± 0.003) to 0.45 (CI ± 0.035) and riprap presence went from 0.069 (CI ± 0.004) to 0.51 (CI ± 0.035). Above 5-10% IS, bulkhead presence increased linearly to a value of 1 whereas riprap use remained fairly constant near 0.5 (Figure 3A); for both shoreline types, confidence intervals increased as %IS increased. Probabilities for shoreline types debris, unconventional structures, and wharf remained below 0.1 until IS reached approximately 55% at which point it increased but was highly variable (Figure 3A). Increased %IS was synonymous with a decrease in the percent length of natural shoreline from 91% (CI ± 0.3) at 0-5% IS to as low as 1.5% (CI ± 1.8%) at 80-85% IS and an increase in percent length of bulkhead from 0% (CI ± 0.1%) at 0-5 % IS to a high of 80% (CI ± 38%) at 90-95% IS (Figure 4A). Above 35% IS the percent length of shoreline, when present, became increasingly variable. The use of riprap to stabilize shoreline was typically <15% of shoreline length regardless of catchment %IS; exceptions were at 55-60% IS (20% [CI ± 15%]) and 95-100% (23% [CI ± 30%]; Figure 4A). Wharfs, unconventional structures, and debris were typically negligible components of catchment shorelines, but when present the extent of their use was variable.

Sixty one percent of the shoreline land use was not classified in the CCRM2020 data and was not randomly scattered throughout the MD Chesapeake Bay shoreline (Figure 5). Land use adjacent to the shoreline was best described for Anne Arundel, Calvert, and Talbot counties; Baltimore City; and the islands and wetlands in Tangier Sound (Wicomico, Somerset, and Dorchester counties). Land use adjacent to the shoreline, where described, did suggest some spatial patterns (Figure 6). Residential and grass land uses were common features for both western and eastern shores of the Chesapeake Bay occurring in tandem with forested shorelines on the western shore and agriculture on the eastern shore. Baltimore City shoreline was a commercial and industrial hub, and the Interstate-95 corridor was captured by the presence of pavement adjacent to shoreline.

The probability that the adjacent land use was specified was ≤0.1 (Figure 7A). When land use was specified, mean percent shoreline length per land use was 69% (CI ± 2%) agriculture, 10% (CI ± 3%) bare, 31% (CI ± 2%) commercial, 49% (CI ± 2%) forested, 24% (CI ± 2%) grass, 44% (CI ± 7%) industrial, 68% (CI ± 2%) marsh island, 27% (CI ± 2%) paved, 63% (CI ± 1%) residential, 34% (CI ± 4%) scrub/shrub, and 10% (CI ± 7%) timbered (Figure 7B). The presence and shoreline length of the three most common shoreline types (bulkhead, natural, and riprap) were compared to four land uses that exemplify alteration of the landscape (agriculture, commercial, industrial, and residential). The probability of natural shorelines adjacent to agriculture varied little as %IS increased (mean = 0.99), though when shoreline structure was used to stabilize agricultural shorelines, it was most often riprap and its use generally increased as %IS increased (Figure 8A). Commercial and industrial land uses had similar probability patterns of hardened and natural shorelines although there was more variability along industrial shorelines. In general, the probability of shoreline structure increased, and natural shoreline decreased as %IS increased except for the use of riprap along commercial shorelines which was relatively constant with a mean of 0.34 (Figure 8A). Natural shoreline along residential land use had a high probability of being present (mean = 0.95) and the probability of bulkhead and riprap being present increased from 0.15 (bulkhead) and 0.33 (riprap) at 0-5% IS to 0.65 (bulkhead and riprap) at 15-20% IS; above 20% IS the probability became relatively constant (bulkhead mean = 0.74 and riprap mean = 0.68; Figure 8A).

Percent length of natural shoreline along agricultural land was generally high (mean = 93%) with little variability except at 30-35% and 50-55% IS; bulkhead length was generally low and increased with high variability at >10% IS and riprap varied from 22-38% of shoreline length and variability generally increased as %IS increased (Figure 9A). Natural shoreline along commercial land use constituted 86% of length at 0-5% IS above which it dropped to approximately half of the shoreline length; when replaced with structure, bulkhead length was greater than riprap length which became highly variable at >55% IS (Figure 9A). Industrial shoreline composition was highly variable between the length of natural, bulkhead, and riprap present; although, natural shoreline was a dominant component averaging 89% of length when IS was <40% (Figure 9A). Residential land use had natural shoreline lengths of 84% (CI ± 2%) at 0-5% IS, averaged 61% of length through 60% IS, and then dropped to <9%. Bulkhead length generally increased from a low of 17% to as much as 59% of the shoreline as %IS increased, the exception being at 65-70% IS where the bulkhead length dropped to 16%; riprap use averaged 37% of shoreline length regardless of %IS except at 65-70% IS where the riprap constituted 82% of the shoreline (Figure 9A). The length of residential shoreline type increased in variability when IS exceeded 45%.

Discussion

Increased development, measured as %IS, is an indicator of increased use of shoreline structure (notably bulkhead and riprap) and its percentage of the shoreline length. These relationships were nonlinear. Five %IS was a transitional point where shoreline structure use became substantially more common and occupied longer segments of the shoreline. Variability in the shoreline composition (probability of occurrence and percent of length) increased when %IS was roughly 40-50%. Shoreline fragmentation increased and became less predictable as land was developed, but in general shoreline stabilization was a symptom of land development. Land use was not described for the majority of shoreline preventing definitive conclusions to patterns and trends; specifically, the dynamics between land use and shoreline structure at 5% IS and >40% IS. Alternate measures of land use, such as zoning, will be explored to describe land use for the entirety of Maryland's shoreline to improve understanding of how different land uses could be predictive of shoreline hardening. Additional predictors of shoreline hardening, i.e. economic and social metrics, should be examined considering shoreline hardening incurs a substantial economic expense for its installation. Finally, shoreline hardening and %IS will be compared to adjacent dissolved oxygen (DO), a measure of fish habitat suitability, to determine if they are colinear predictors of DO.

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Table 1. Aggregation of shoreline structure descriptions listed in CCRM2020 into broad categories of shoreline structure type.

Shoreline type	Shoreline description in CCRM2020
Bulkhead	Bulkhead
	Dilapidated Bulkhead
	Marina < 50 slips
	Marina > 50 slips
Debris	Debris
Natural	blank (no structure indicated)
Riprap	Groin
	Jetty
	Marsh Toe
	Riprap
Unconventional	Unconventional
Wharf	Wharf

Figure 1. Presence and extent of shoreline structure within a catchment as percent impervious surface (%IS) increases. All types of shoreline structure among all land use categories were pooled together to assess structure presence or absence and percent of shoreline length with structure within whole catchments (plots A and B) or Critical Area portions of catchments (plots C and D). Catchment %IS was binned in 5%IS increments. Means for proportion of structure presence (plots A and C) and percent of shoreline length with structure (plots B and D) were calculated for each bin. Ninety-five percent confidence intervals are shown with vertical lines.

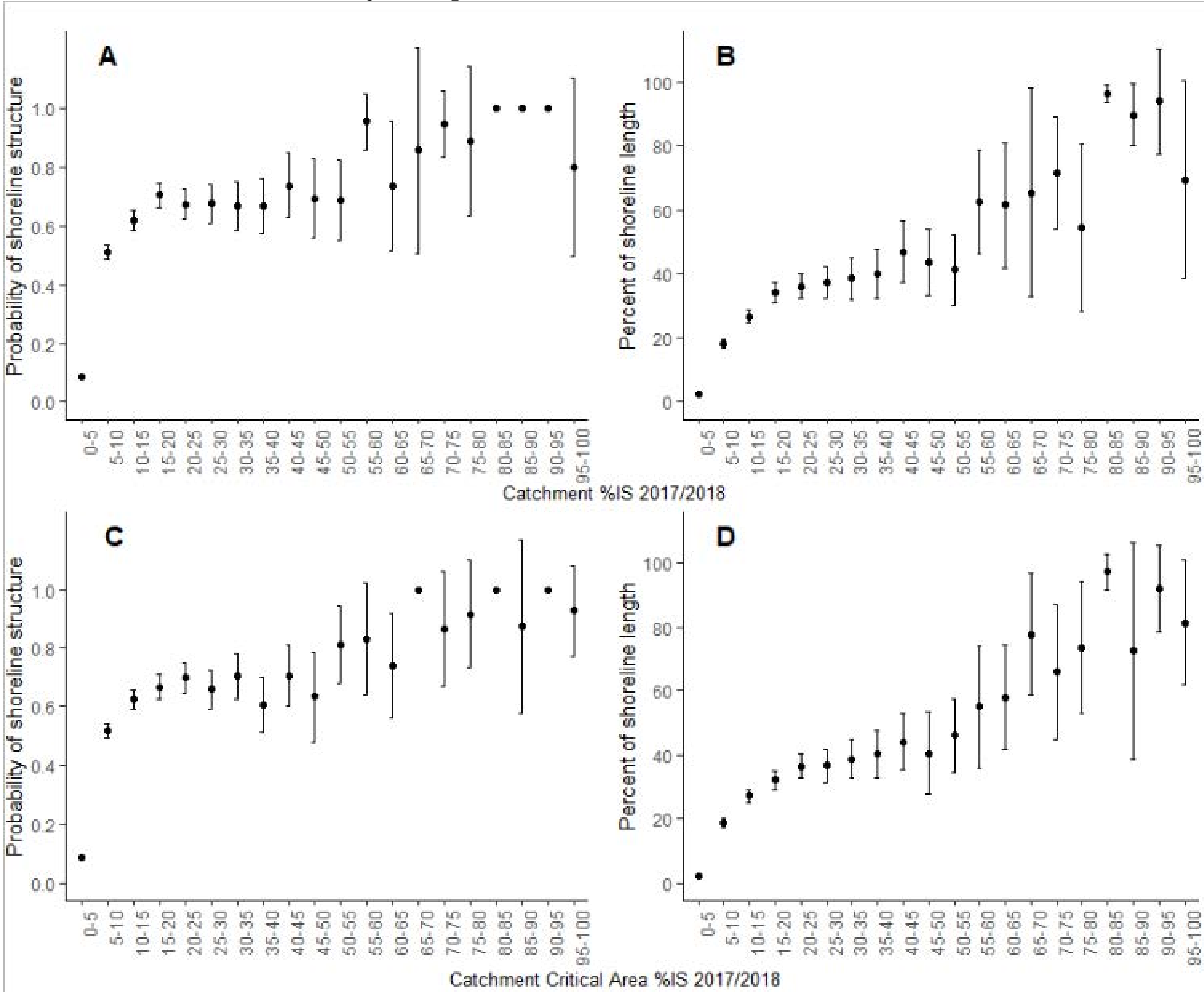


Figure 2. Shoreline composition among catchments; shorelines may be comprised of multiple types within a single catchment. Scatter plot A depicts the mean probability of a shoreline type being present in a catchment. Scatter plot B depicts the mean percent of shoreline length by shoreline type, when present, among the catchments. Ninety-five percent confidence intervals are shown with vertical lines.

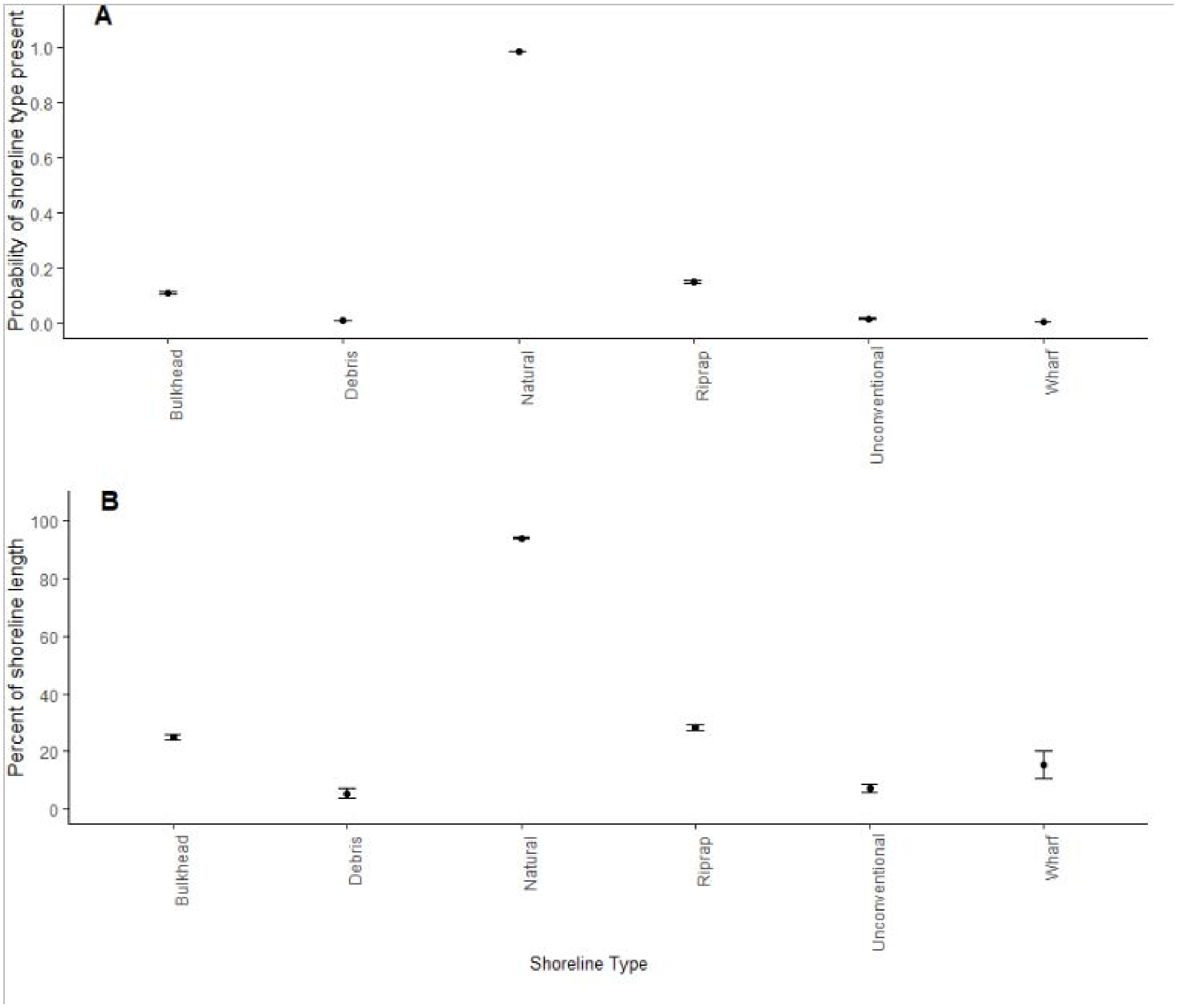


Figure 3. Scatterplots depict mean probability of each shoreline type being present in a catchment as %IS increases. Percent IS was binned in 5%IS increments for the whole catchment (plot A) and the Critical Area portion of the catchment (plot B). Ninety-five percent confidence intervals are shown with vertical lines and the y-axis is constrained for readability.

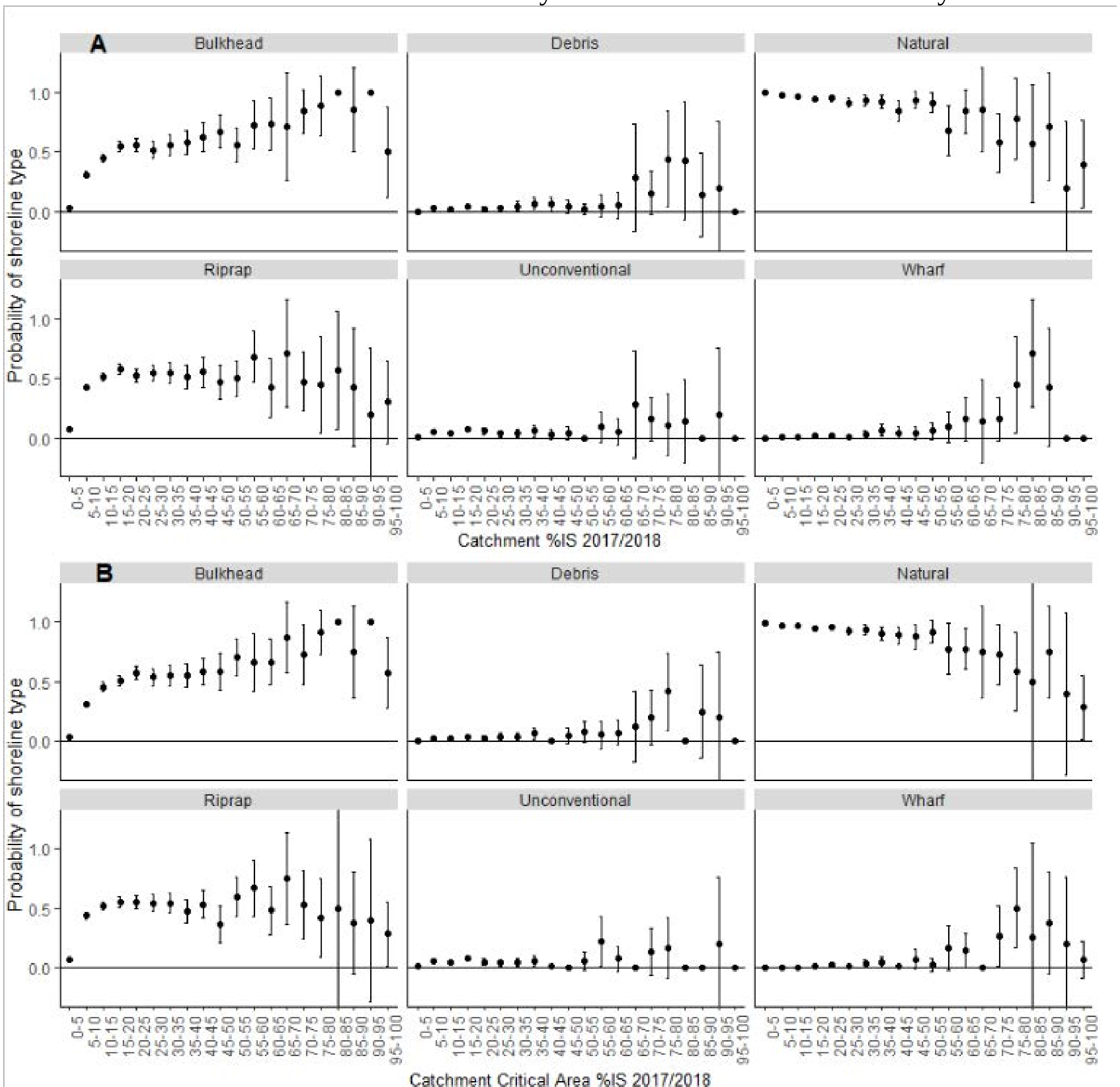


Figure 4. Scatterplots depict mean percent of shoreline length for each shoreline type, when present, as %IS increases within catchments. Percent IS was binned in 5%IS increments for the whole catchment (plot A) and the Critical Area portion of the catchment (plot B). Ninety-five percent confidence intervals are shown with vertical lines and the y-axis is constrained for readability.

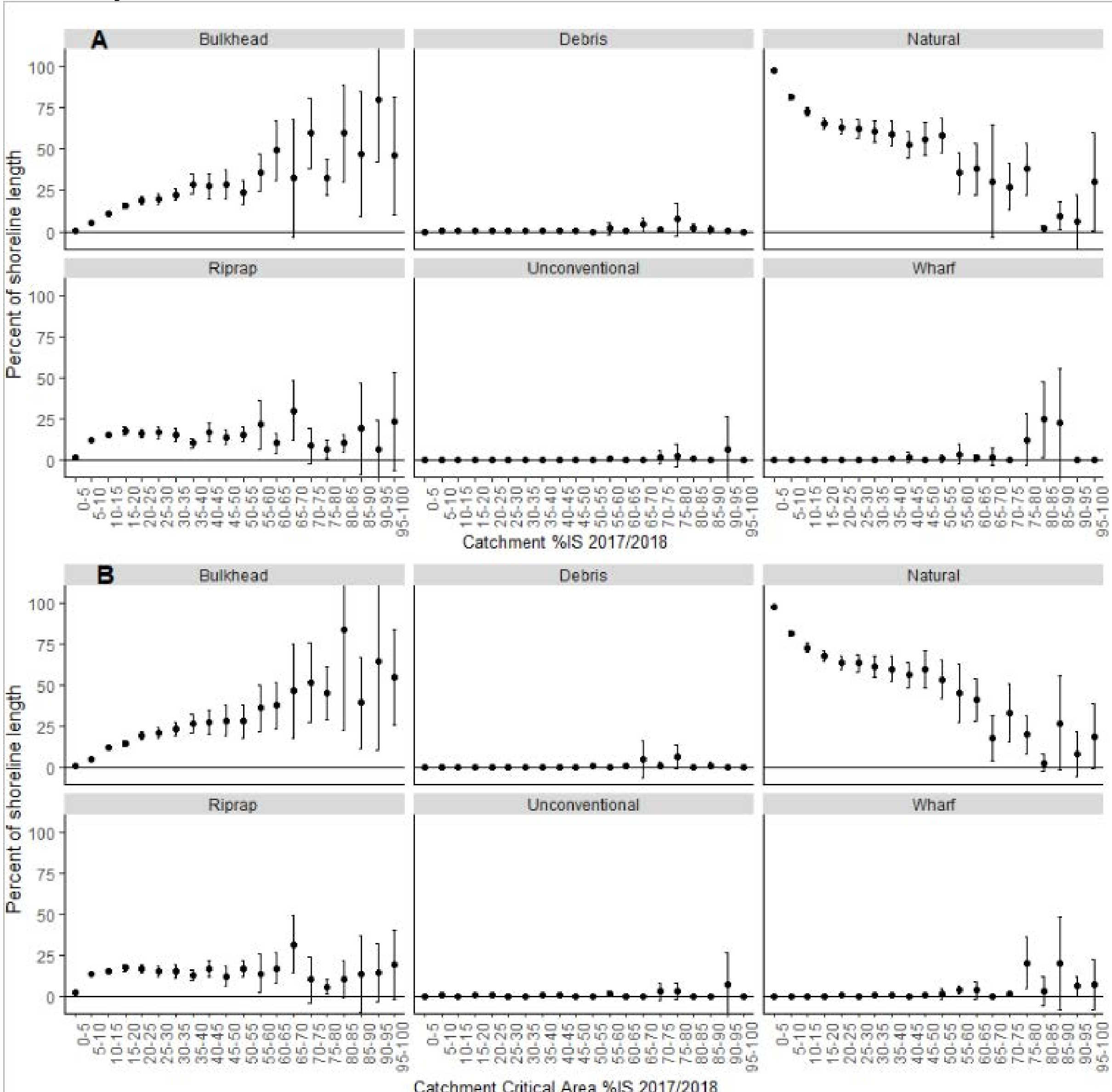


Figure 5. Map showing where shoreline land use (LUBC) data for Maryland was provided (orange) or missing (black) in the shoreline inventory shapefiles.

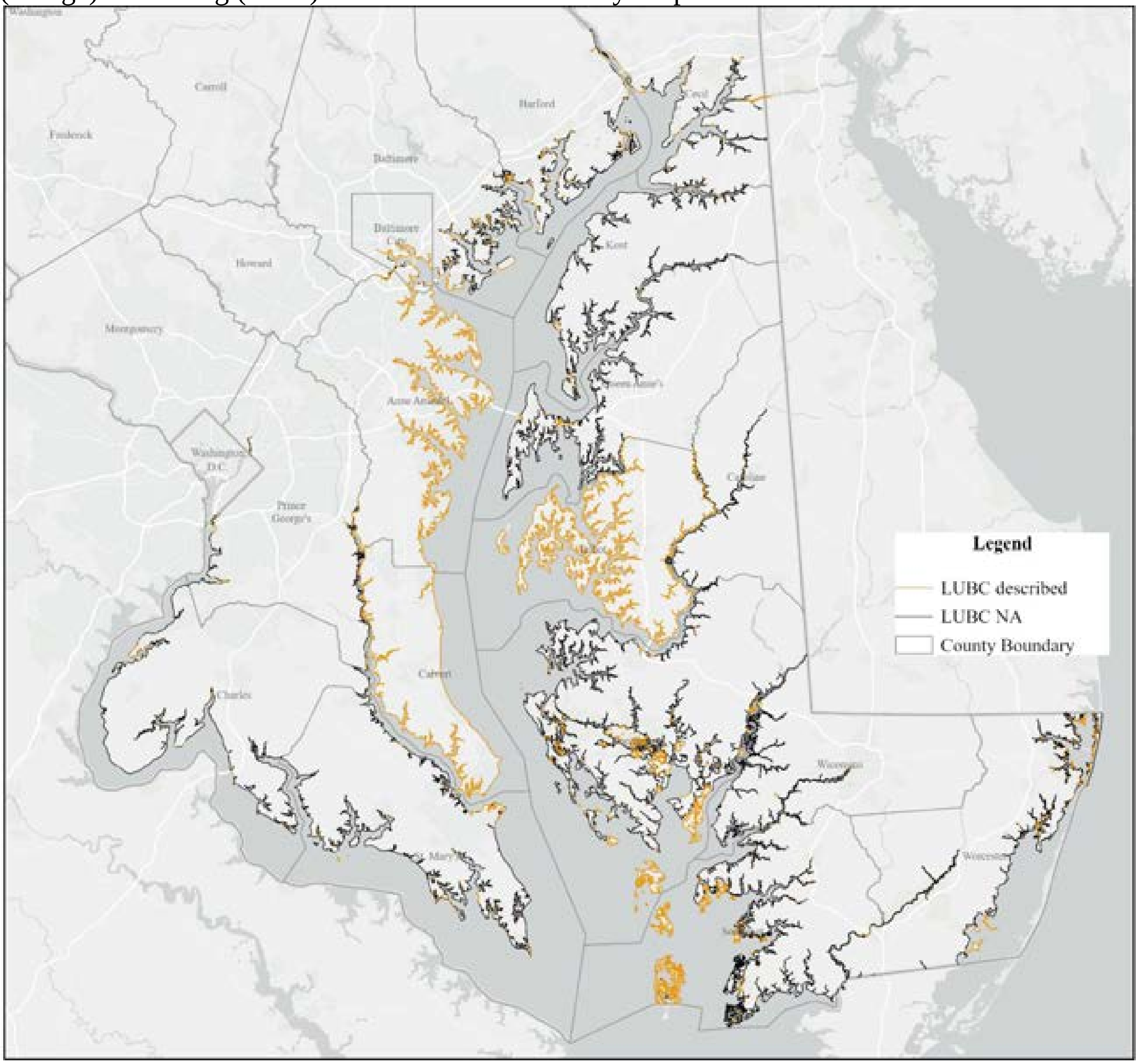


Figure 6. Maps of line segment density for each shoreline land use in Maryland, Chesapeake Bay. Patterns reflect the availability of land use data in shoreline inventory shapefiles.

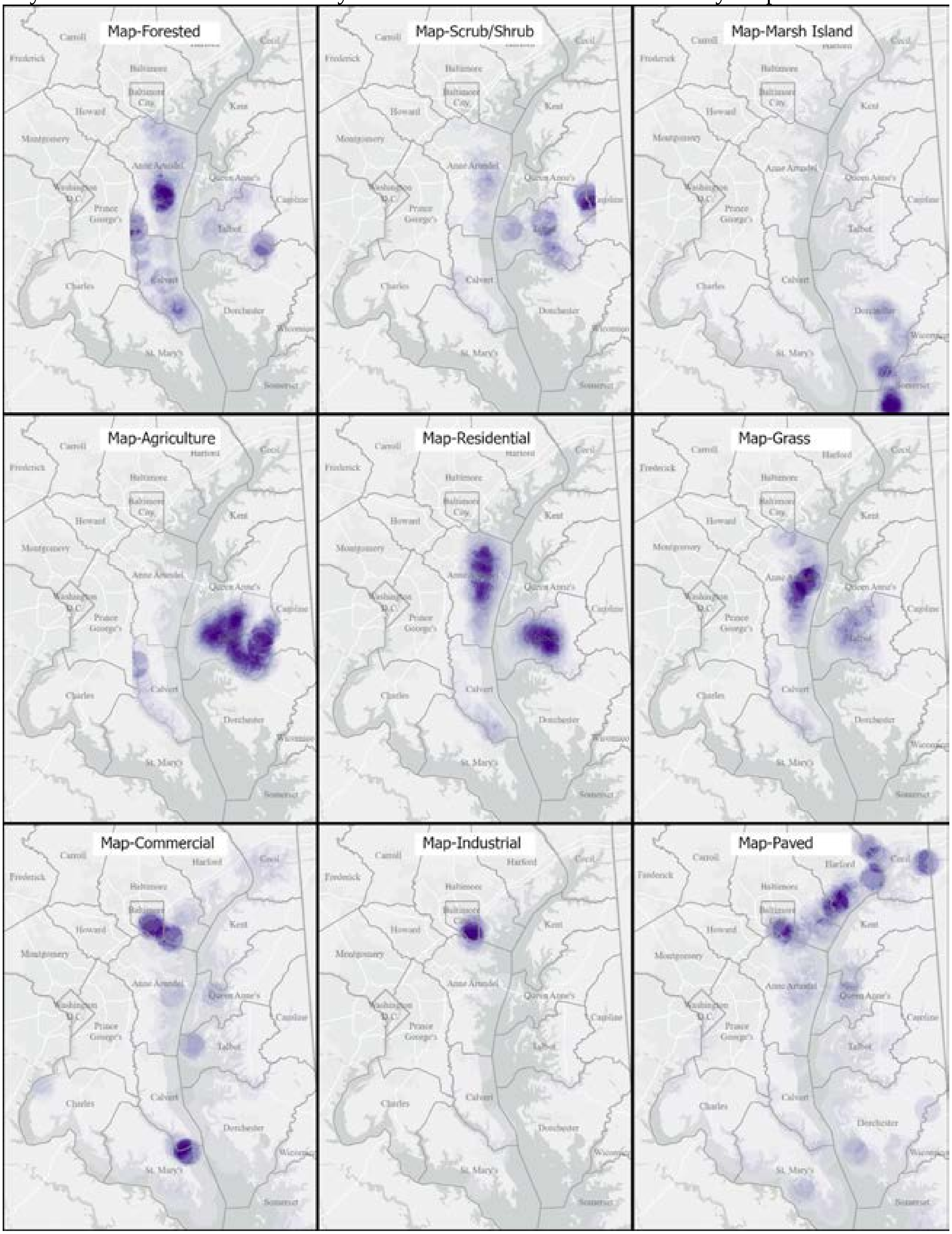


Figure 7. Mean probability of land use categories being adjacent to the shoreline among catchments (plot A); shorelines may be adjacent to multiple land use categories within a single catchment. Scatter plot B depicts the mean percent length of shoreline adjacent to each land use among the catchments. Ninety-five percent confidence intervals are shown with vertical lines.

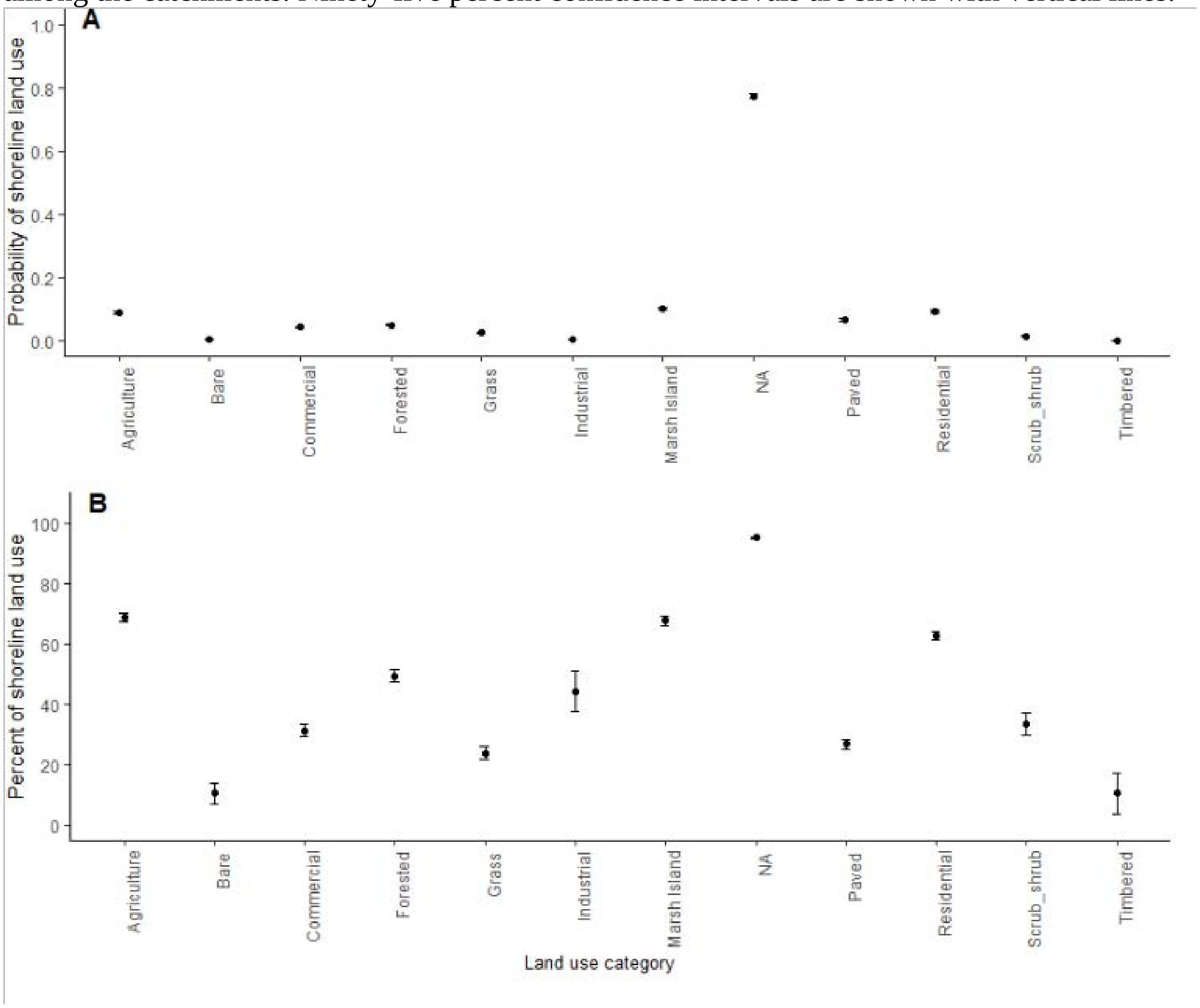


Figure 8. Scatterplots for the mean probability of a shoreline structure type being adjacent to a land use category as %IS increases. A subset of the structure types (three most common) and four land use categories (impacted lands) are shown. Percent IS was binned in 5% IS increments. Plot A represents the whole catchment and plot B depicts the Critical Area portion of the catchment. Ninety-five percent confidence intervals are shown with vertical lines and the y-axis is constrained for readability.

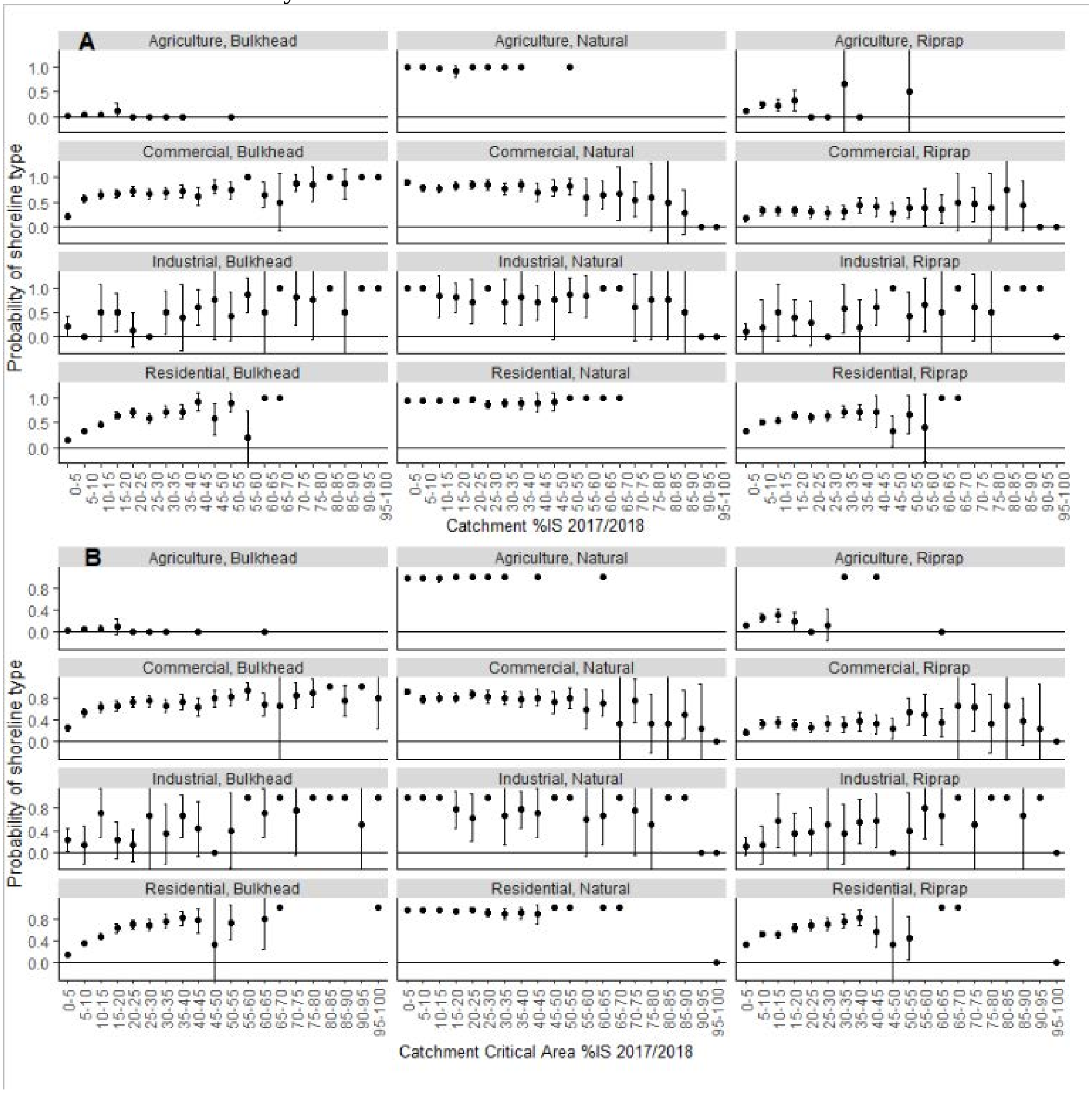
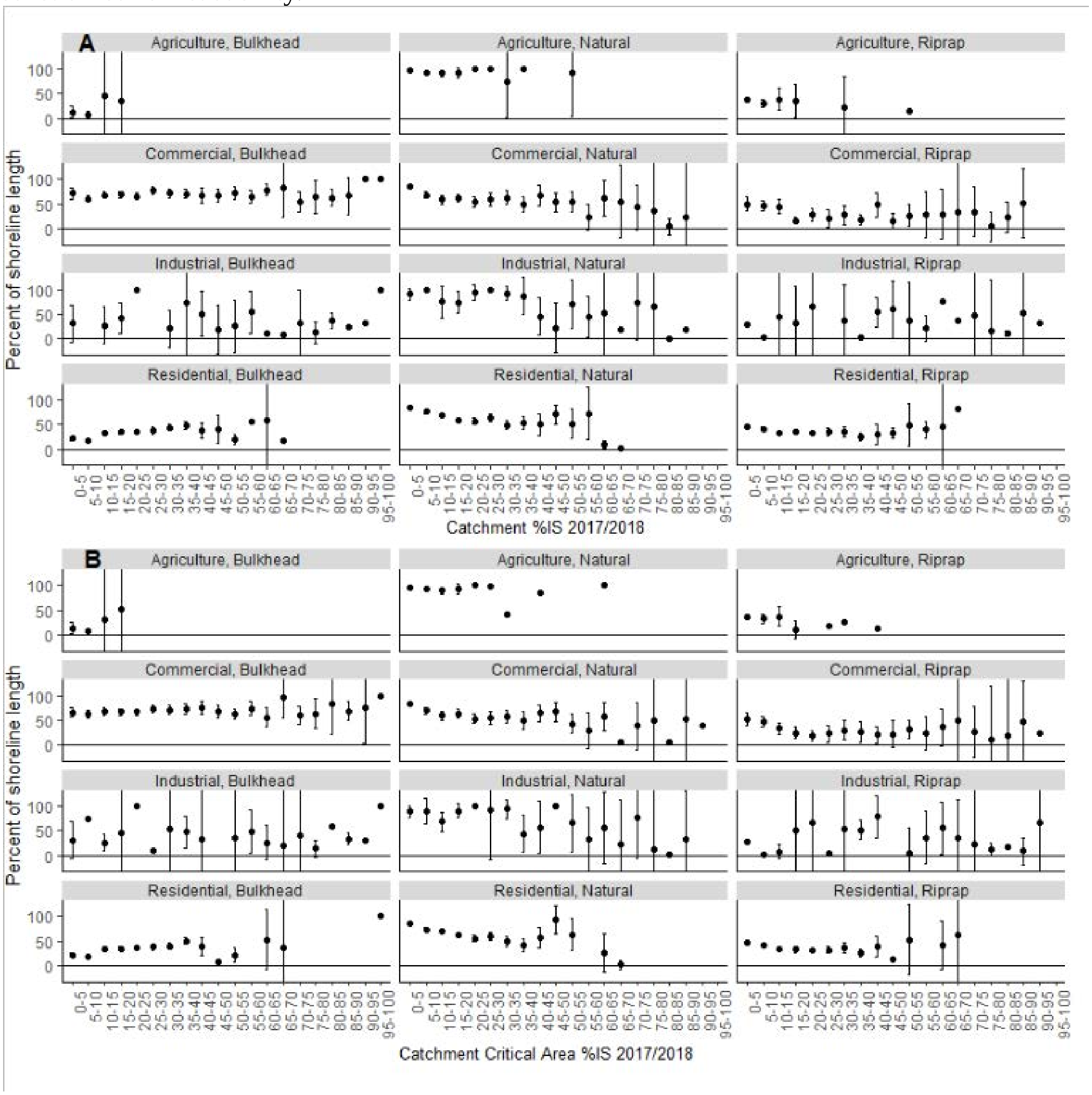


Figure 9. Mean percent of shoreline length of a given structure type, if present, adjacent to a land use category as %IS increases. A subset of the structure types (three most common) and four land use categories (impacted lands) are shown. Percent IS was binned in 5% IS increments. Plot A represents the whole catchment and plot B depicts the Critical Area portion of the catchment. Ninety-five percent confidence intervals are shown with vertical lines and the y-axis is constrained for readability.



Objective 4: Resident Striped Bass forage benchmarks

Project Staff

Jim Uphoff, Alexis Park, Carrie Hoover, and Tyler Fowler

Executive Summary

Indices of Striped Bass condition, relative abundance, natural mortality, and forage relative abundance from annual surveys and fall diets provided 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 for good and poor levels, respectively. An index of survival (SR) that reflected natural mortality (M) from age 0 to 3 was developed for small Striped Bass. 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 time-series of relative abundance indices of major prey (FRs; focal prey species are Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab).

In 2023, P0 for each size class was 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 a poor range) in 6 of 7 years during 1998-2004 and has improved to only slightly missing its target once since 2014.

Confidence intervals (90%) for PE of small Striped Bass overlapped the best values in this time-series in 2023. Estimated PE of small fish has been variable since 2012 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 2023. Estimates of PE for large Striped Bass have improved from threshold conditions prior to 2007 and 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 lower prey-predator length ratio 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 2023. 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. Some bias in small fish diet composition may have resulted 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.

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 2022-2023. These estimates were from poor Striped Bass year-classes. 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; Uphoff 2023). 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 approximately 2- to 5-times through changes in age-at-entry and fishing mortality (Uphoff 2003).

Concern emerged about the impact of high Striped Bass population size on its prey-base shortly after recovery from severe depletion was declared 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." Objective 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 2023. 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 (≥ 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. Uphoff et al. (2023) were concerned that small Striped Bass diets may be biased by the minimum sizes available in samples. They suggested dividing small Striped Bass into a smaller category not capable of feeding on Atlantic Menhaden and a "mid-sized" small category for Striped Bass transitioning to Atlantic Menhaden.

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 proportion of fish below a certain threshold of poor condition rather than mean condition is most likely related to starvation rates (Regular et al. 2022). 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).

A benthic invertebrate index (invertebrates other than Blue Crabs) is included in this report ("soft bottom" benthic index) 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) but could not be estimated for this report due to shortage of staff.

The ratio of age-3 relative abundance of male Striped Bass in spring spawning ground gill net surveys (Versak 2023) to their year-class-specific juvenile indices (Durell and Weedon 2023) 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. 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 or pattern 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 year-round 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 Cooperative Oxford Laboratory's 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 arose due to length limit changes.

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 with ice and either processed upon return to shore or held on ice for processing the next day. Collections of large Striped Bass were supplemented by CBEF sampling of 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 the FWHP have been processed since 2014 by the Fisheries Ecosystem Assessment Division (FEAD; formerly Fish Habitat and Ecosystem Program or FHEP) 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 the FWHP Striped Bass health survey. 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. 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 very close 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).

We 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 proportion by number was negatively biased to some extent.

Relative availability of prey biomass (biomass consumed or C) was estimated by dividing the sum of diet item weights by the sum of weight of all Striped Bass sampled (including those with empty stomachs; Pope et al. 2001). Estimates of C were subdivided by contribution of each major prey to overall diet mass (species-specific C).

Proportion of Striped Bass with empty stomachs (PE) was an indicator of total prey availability (Hyslop 1980). Standard deviations and 90% CIs 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. We used correlation analysis to determine the strength of the association of PE with C.

Level of significance was reported for correlations and regressions, 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 $r > 0.80$; weak correlations were indicated by $r < 0.50$; and moderate correlations fell in between. Relationships indicated by regressions were considered strong at $r^2 > 0.64$; weak relationships were indicated by $r^2 < 0.25$; and moderate relationships fell in between.

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 (0.34) was estimated for large fish from periods when PE corresponded with target estimates of P0; a target could not be estimated for small fish. Uphoff et al. (2022a) reviewed the plot of PE and P0 and indicated that P0 at the target level was more likely when PE was 0.34 or less (7 of 9 points) than above it (1 of 4). The PE target for large fish was set at 0.34

We used correlation analysis to examine associations of PE and P0 for each size group of Striped Bass. We examine the bivariate plots to see if threshold values might be suggested for PE associated with good or poor feeding conditions.

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).

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 2023). 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 2024a; 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 2024b). 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 / m^2) of benthic invertebrates for Maryland tidal waters as our index (Figure 3-38 in 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) during 1983-2023 from the National Marine Fisheries Service's (NMFS) Marine Recreational Information Program (MRIP) website (https://www.fisheries.noaa.gov/data-tools/recreational-fisheries-statistics-queries) as an index. The query tool provided 2-month wave-based estimates of catch and trips with proportional standard errors (PSEs). Catch was comprised of harvest and releases. 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. Ages 2-5 abundance estimated for the Atlantic coast stock assessment (ASMFC 2022) was moderately related to RI from Maryland's portion of Chesapeake Bay during 1983-2022 (Uphoff et al. 2023). The trend in RI tracked the trend in estimated aggregate abundance of 2- to 5-year-old Striped Bass along the Atlantic Coast well through 2014 and less so after (Uphoff et al. 2023).

We used forage indices divided by RI (forage index-to-Striped Bass index ratios, i.e., forage ratio or FR) as 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 forage indices; RI estimates were available for every year since 1983 except 1987 (RI was not estimated).

We estimated relative survival as relative abundance at age-3 from a spawning season gill net index (Versak 2022) divided by age-0 relative abundance three years prior (juvenile index in year – 3; Durell and Weedon 2023) for 1985-2020 and 2022-2023. We did not estimate relative survival (SR) for 2021 due to concerns about the validity of the gill net index for that year due to an outbreak of Covid that shut down sampling during a key period (B. Versak, MD DNR, personal communication). 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 2022). Table 8 in Versak (2022) provided mean values for annual, pooled, weighted, age-specific CPUEs since 1985 and Table 11 provided coefficients of variation 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 (Versak 2022). 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 2023), 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 CPUE3 / relative q. After 1995, reported CPUEs were used (Uphoff et al. 2019). The hybrid index was compared to abundance of age 3 Striped Bass along the Atlantic Coast estimated by the ASMFC (2022) statistical catch-at-age model.

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 SEs in their respective sources (Versak 2023; MD DNR 2024b). 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 2023). 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 1998-2023, 2,257 small and 3,455 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 117. Annual sample sizes for large fish ranged from 49 to 327 with a median of 205. 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 2014-2023 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 2021-2023 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 2021-2023 (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, 2021-2023; 2021-2023 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, 2006-2023, 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 (0.54; Overton et al. 2009) was the threshold for small fish (Uphoff et al. 2016). Lowest estimates of PE for small fish (2009-2011, 2014, 2017, 2019, and 2023) could be separated from most higher estimates based on 90% confidence interval overlap. Estimates of PE during 2008-2011, 2014, 2016-2021, and 2023 were clearly lower than the 90% CIs of years that breached the threshold. Estimated PE in 2023 (0.145) was below the threshold and time-series median (0.31; Figure 3). Estimates of PE for small fish were not correlated with P0 ($r = 0.06$, $P = 0.82$); this may reflect that small fish were likely to have eaten small items that may not have supplied much nutrition.

In combination and by number, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab accounted for 96.1% of diet items encountered in small Striped Bass collected from upper Bay during fall, 2006-2023 (Figure 4). Bay Anchovy accounted for the highest percentage by number when all years were combined (61.4%, annual range = 9.3-87.9%); Atlantic Menhaden, 17.4% (annual range = 0-74.1%); Spot 5.4% (annual range = 0-70.7%); Blue Crab, 11.9% (annual range = 0.8-34.6%); and other items accounted for 3.9% (annual range = 0-12.9%; Figure 4). During 2023, Atlantic Menhaden accounted for 43.9% of the diet items; Bay Anchovy, 48.8%; Spot, 0%; Blue Crab, 7.3%; and other items, 0%. The vast majority of major prey in small Striped Bass diet samples during fall fell within young-of-year length cut-offs used by Virginia Institute of Marine Science for their seine and trawl surveys (https://www.vims.edu/research/units/programs/juvenile_surveys/data_products/indices/).

By weight, small Striped Bass diets in fall 2006-2023 (combined) were comprised of Atlantic Menhaden (73.8%), Bay Anchovy (12.8%), Spot (7.8%), Blue Crab (1.5%) and other items (3.8%; Figure 5). Estimates of C (total grams of prey consumed per gram of Striped Bass) for small Striped Bass varied as much as 8.8-times during 2006-2023. 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 2006-2023 median) either Spot (2010) or Atlantic Menhaden (2013-2014) dominated diet mass. The 2023 estimate of C of small fish (0.021) was above the median (0.0131) of the year time-series (Figure 5). Estimates of C for small fish were poorly correlated with PE ($r = -0.40$, $P = 0.10$).

Median PPLRs of large prey (Spot and Atlantic Menhaden combined) of small Striped Bass were 0.20-0.38 during 2006-2023 (Figure 6). 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. Median PPLRs have steadily fallen since 2019 and were 0.27 in 2022 and 2023; these were the third lowest of the time-series (Figure 6). These PPLRs would not be affected by unbalanced sampling of small sized Striped Bass 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 7). Large Striped Bass were at the target level of condition ($P0 \leq 0.30$) during 2008-2010, 2014-2015, and 2017-2023. Estimated P0 was 0.087 in 2023. Large fish during fall were usually in poorest condition ($P0 \geq 0.70$) during 1998-2004. 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 based on CI overlap (Figure 7).

Estimates of PE of large Striped Bass during fall were at threshold level in 2006, 2012, and 2017 based on 90% CI overlap (Figure 8). The PE target for large fish, 0.34, was met during 2014-2015, 2018-2021, and 2023. Estimated PE was 0.27 in 2023 (Figure 8). There was a moderate association of PE and P0 ($r = 0.66$, $P = 0.003$) during 2006-2023; the plot of these variables indicated that P0 at the target level was more likely when PE was 0.35 or less and a rapid ascent of most P0 points towards poorer condition beyond PE = 0.35 (Figure 9).

Major prey accounted for 92.8% of diet items, by number, encountered in large Striped Bass diet samples during fall 2006-2023 (Figure 10). Atlantic Menhaden accounted for 50.1% by number when all years were combined (annual range =12.4-97.0%); Bay Anchovy, 15.9% (annual range = 0-32.5%); Spot, 7.1% (annual range = 0-52.4%); Blue Crab, 19.7% (annual range = 0-59.4%); and other items, 7.2% (annual range = 0-40.0%). 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 2023, Atlantic Menhaden accounted for 54.3% of October-November diet items; Bay Anchovy, 36.8%; Spot, 0%; Blue Crab, 5.7%; and other items accounted for 3.2% (Figure 10). The vast majority of major prey fell within young-of-year length cut-offs.

By weight, Atlantic Menhaden predominated in large fish sampled (88.3% of diet weight during fall, 2006-2023, combined); Bay Anchovy accounted for 1.1%; Spot, 3.1%; Blue Crab, 3.2%; and other items, 4.3% (Figure 11). Estimates of C for large Striped Bass varied as much as 3.8-times among years sampled. The 2023 estimate of C of large fish (0.0134) was below the time-series median (0.0136; Figure 11). Estimates of C for large fish were moderately correlated with PE ($r = -0.56$, $P = 0.015$).

Median PPLRs of large prey (Spot and Atlantic Menhaden) for large Striped Bass were 0.19-0.30 during 2006-2023 (Figure 12). The median PPLR was 0.23 for 2023 (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) was lowest during 1983-1993 (mean RI < 0.7 fish per trip; Figure 13). Estimates of RI then rose abruptly to a high level and remained there during 1995-2007 (median = 2.6). Estimates of RI fell during 2008-2013 (mean = 1.3) then rose to 2.4-3.6 during 2014-2019 (2019 was the second highest of the time-series). The RI steadily fell from 1.8 in 2020 to 0.9 in 2023. 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 2020-2023 was lower than other years during 1995-2019 (Figure 13). Uphoff et al. (2022) determined that the RI was moderately related to ages 2-5 abundance estimated for the Atlantic coast by ASMFC (2022; linear regression, $r^2 = 0.52$, $P < 0.0001$).

Major pelagic prey were generally much more abundant during 1959-1994 than 1995-2022 (Figure 14). Seine indices for Bay Anchovy and Atlantic Menhaden improved considerably in 2023. Bay Anchovy seine indices following the early to mid-1990s were typically low during 1959-1993 but 2023 fell within the higher 1959-1994 range. Highest Bay Anchovy trawl indices (top quartile) occurred in 1989-1992, 1998-2000, 2013-2014, and 2020-2021, while lowest quartile indices occurred after 2006. 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). The 1990 Atlantic Menhaden seine index was the highest since 1990; 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 14). 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).

Spot seine indices have been above average during 2020-2023 (Figure 15). Major benthic forage indices were low after the 1990s, but years of higher relative abundance were interspersed during the 2000s. Seine (1959-2022) and trawl (1989-2022) indices for Spot generally indicated high abundance during 1971-1994 and low abundance during 1959-1970 and 1995-2019 (with 3 or 4 years of higher indices interspersed). The two Spot indices were strongly correlated ($r = 0.82$, $P < 0.0001$). Blue Crab densities (1989-2019) were generally at or above the time-series median during 1989-1998, and 2009-2015. Blue Crab densities in 2020-2023 were among the lowest of the time-series (Figure 15).

Most of the annual indices of biomass of soft bottom benthic invertebrates during 2000-2009 were well above the time-series median (Figure 16). Indices in the lowest quartile occurred during 1996, 1998, 2003, 2014, and 2021-2023 (Figure 16).

Species-specific standardized FRs exhibited similar general patterns during 1983-2023 (Figure 17). 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 abundance 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 of Blue Crab in 2023 and trawl survey Bay Anchovy were below their 1989-2023 medians, while remaining species were at or above their medians in 2023 (Figure 17). The Atlantic Menhaden FR in 2023 was the highest since 1991 (Figure 18). Prior to this year, it was generally elevated during 2005-2023 from its nadir during 1997-2004 but has been well below levels prior to the early 1990s (Figure 18). The Bay Anchovy seine FR in 2023 was the highest since 1991 and was above years of higher FRs since 1995 (2006-2009 and 2010-2013; Figure 19). The Bay Anchovy trawl FR for 2023 was in the top third of the time-series (Figure 20). The Spot seine FR during 2020-2021 was in the higher portion of the range exhibited since 1995 (Figure 21). The Spot seine (Figure 21) and trawl FRs (Figure 22) for 2020-2023 were similar in magnitude and indicated considerable improvement over lows exhibited during 2014-2019. The Blue Crab FR was above the time-series median in 2023 (Figure 23).

Relative survival of small Striped Bass – The unadjusted age 3 gill net index of male relative abundance on the spawning grounds indicated abundance during 1985-1995 was at least as high as any other period of the time-series (Figure 24). 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. The HI_3 for 2023 (2020 year-class) was very low and reflected the poor juvenile index for that year (Figure 24).

Ninety percent CIs of relative survival (SR; HI_3 / JI_{t-3}) allowed for separation of years of high and low survival, and some years in between (Figure 25). Estimated SR in 2022 was among the peak values of the time-series and SR in 2023 was the second highest (Figure 25).

Estimated SR was more often high during 1986-1998 with 9 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 26). Low SR during this rapid increase of the RI may have indicated a lagged response. After 1998, SR shifted 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 fifth highest of the time-series and 2023 was second. 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 26). 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 2023 and was among 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). Response of condition may be lagged since 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 anglers 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. Large fish fall diets were typically dominated by young-of-the-year (YOY) Atlantic Menhaden by weight and number while small fish diets were more variable and had a higher frequency of small items.

Small Striped Bass diet summaries may be biased by the minimum sizes available in samples (Uphoff et al. 2023). This year we were unable to consider an additional smaller category of small fish not capable of feeding on Atlantic Menhaden and a "mid-sized" small category for Striped Bass transitioning to Atlantic Menhaden as recommended in Uphoff et al. (2023). We hope to make this correction for the next annual report. We based estimates of proportion of food represented by diet items for small fish on a standard TL range; however, Striped Bass in the lower end of the size distribution were not always represented. The proportion of diet by number represented by Bay Anchovy and Atlantic Menhaden was affected by size of Striped Bass present during a sample year. Uphoff et al. (2023) found that 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 (Uphoff et al. 2023). The proportion of Atlantic Menhaden in small Striped Bass diets was moderately and positively correlated with annual small Striped Bass mean TL. The proportion of Bay Anchovy in small Striped Bass diets was moderately to strongly and negatively correlated with small Striped Bass mean TL. These associations indicated that small Striped Bass diet composition was influenced by the length of fish sampled.

Atlantic Menhaden YOY 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 diets 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 2023.

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 be subject to a prey bottleneck. 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 as 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.

Extensive research (laboratory, field studies, and stock assessment modeling) on the links between forage, condition, and M have been conducted for some stocks of Atlantic Cod that provide a narrative that seems to apply well to Striped Bass in the Bay. Similar to resident Striped Bass, these stocks experienced forage fish declines, followed by declining 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). Dutil and Lambert (2000) found that the response of M of Atlantic Cod could be delayed after unfavorable conditions. Fish condition has been used to estimate time-varying M in Baltic Sea and Canadian Atlantic Cod stock assessments (Casini et al. 2016; Regular et al. 2022). Recovery of the northern stock of Atlantic Cod has paralleled recovery of Capelin *Mallotus villosus*, its main prey; increases in size composition and fish condition and apparent declines in mortality followed increased Capelin abundance (Rose and Rowe 2015).

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 between 2005 and 2022 have been well below those that occurred in 1990, the year used to set target conditions for P0 in Jacobs et al. (2013) but reached the levels of the early 1990s in 2023 through a combination of higher forage indices and lower abundance of resident Striped Bass. 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 and fell 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 became a sizeable fraction (by number) of small and large fish diets (46% and 38%, respectively) in 2023, possibly reflecting the advent of a large year-class (indicated by the seine index but not the trawl index). 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).

The soft bottom benthic index time-series covered 1995-2023 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 three 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 (such as polychaete or "May worm" blooms or the high consumption of small clams, presumably *Macoma*, observed by J. Uphoff in spring 2015) 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 of 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 (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 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; $r = -0.37$, $P = 0.013$, 1959-2023; 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).

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 2022-2023. This estimate was from poor Striped Bass year-classes in 2019 and 2020 that were the first of a series of poor years through 2023 (Durell and Weedon 2023). If SR remains elevated through this series of poor year-classes, it may indicate lessening of density-dependent M up to age 3.

Higher frequency of SR below the 1985-2020 median 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.

Tagging studies described above did not describe mortality/survival of small Striped Bass. 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). The possibility of a prey bottleneck for small Striped Bass at the transition from small to large major prey was described previously. Fish reaching age 3 in spring (numerator of the SR index) would have been dependent on small prey and invertebrates.

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 (2014-2023) 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 P0 were similar. Fall diets were dominated by Atlantic Menhaden and Spot were absent in both cases. Differences arose in smaller major prey, particularly Bay Anchovy, and in the importance of “Other” prey (Uphoff et al. 2018). There was not a readily discernable shift in patterns of PE, C, and frequency of diet items by number detected 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 some 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) for large fish 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, 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. Atlantic Menhaden, Bay Anchovy, 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 believe 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 few 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) indicated 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 hierarchal Bayesian technique. Falcy et al. (2016) found that ratios of means provided a reasonable method for combining indices into a composite index to be calibrated with population estimates of Chinook Salmon *Oncorhynchus tshawytscha*, but there was no one optimal method among the four techniques applied.

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 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 target, threshold, or in-between status approach for P0 and PE were 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.

Recent discussions with MD 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 communicating the status of Atlantic Menhaden and Striped Bass balance in Maryland's portion of the Bay (J. Uphoff, MD DNR, personal communication). This Traffic Light Index (TLI) is scheduled for an outside peer-review on August 21, 2024. If successful, we will consider the TLI approach for metrics developed here.

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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.
FEAD	Fish Ecosystem Assessment Divison
FHEP	Fish Habitat and Ecosystem Program
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
2023	7	55	277	24-Oct	29-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
2023	407	420	440	284	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.

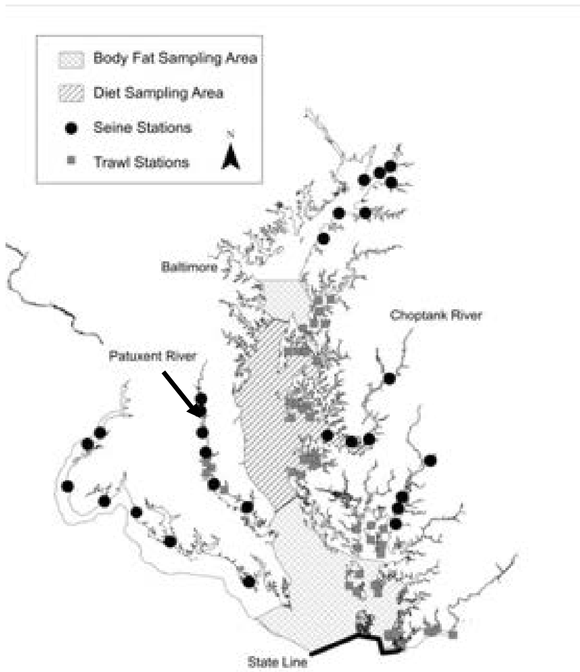

Figure 2. Proportion of small Striped Bass without body fat (P0) during October -November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

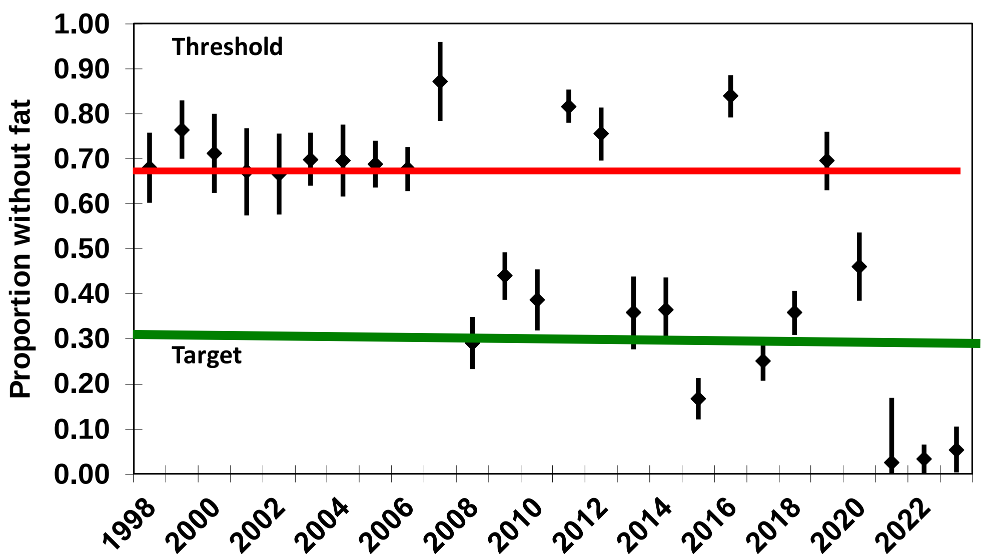

Figure 3. Proportion of small Striped Bass guts without food (PE) in fall and its 90% confidence interval. Red diamond represents threshold PE.

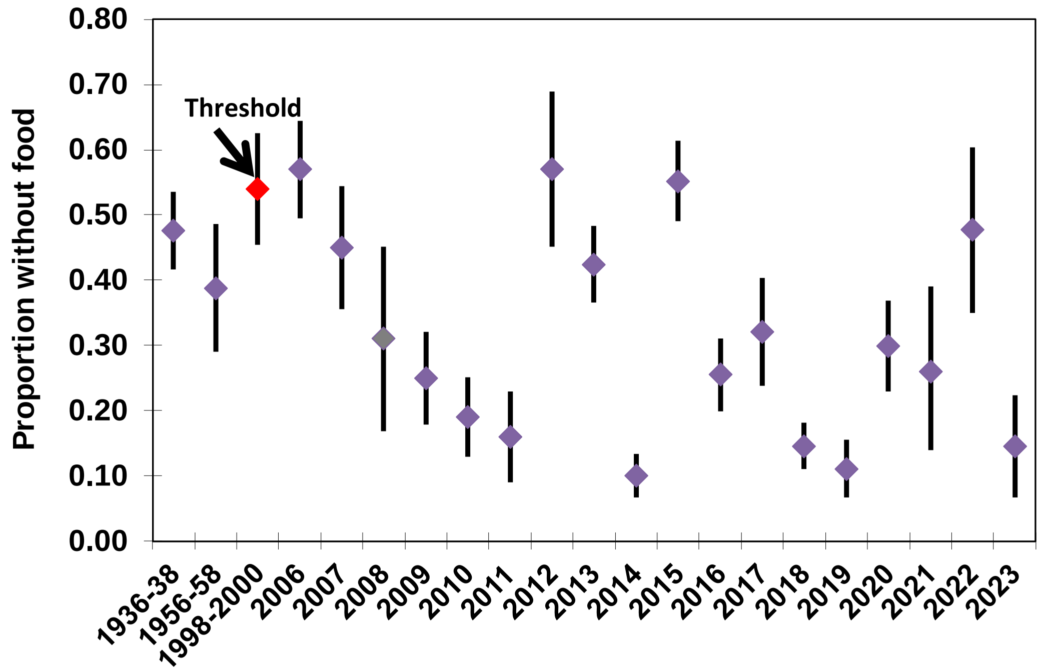

Figure 4. Percent, by number (counts of individuals plus presence of parts), of identifiable (excludes unknown) major forage groups in small Striped Bass (< 457 mm TL) guts, in fall.

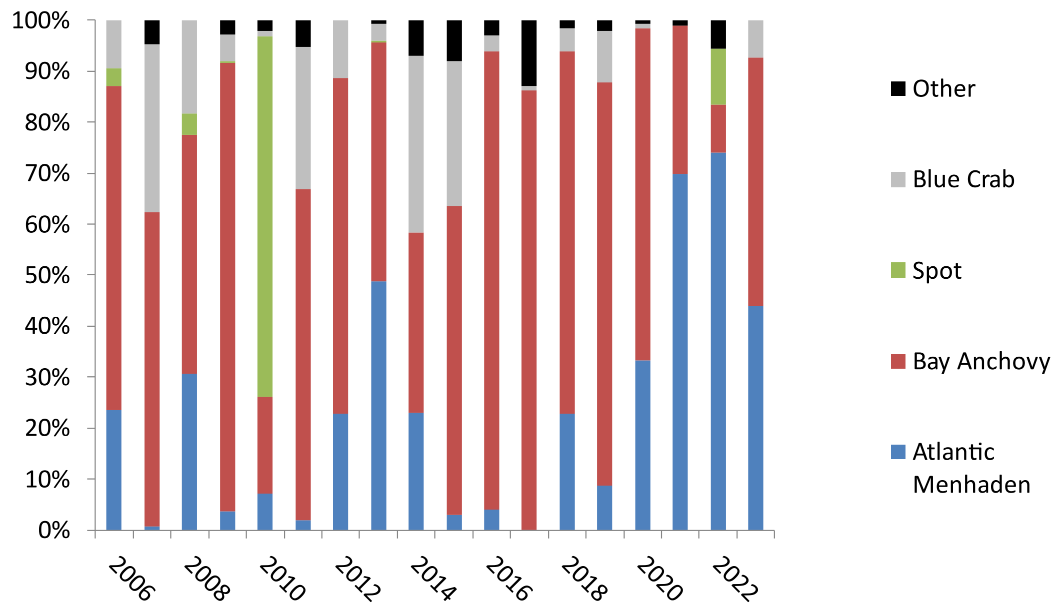

Figure 5. Gram prey consumed per gram (C) of small (< 457 mm TL) Striped Bass in fall hook-and-line samples. Age-0 forage dominate the diet. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

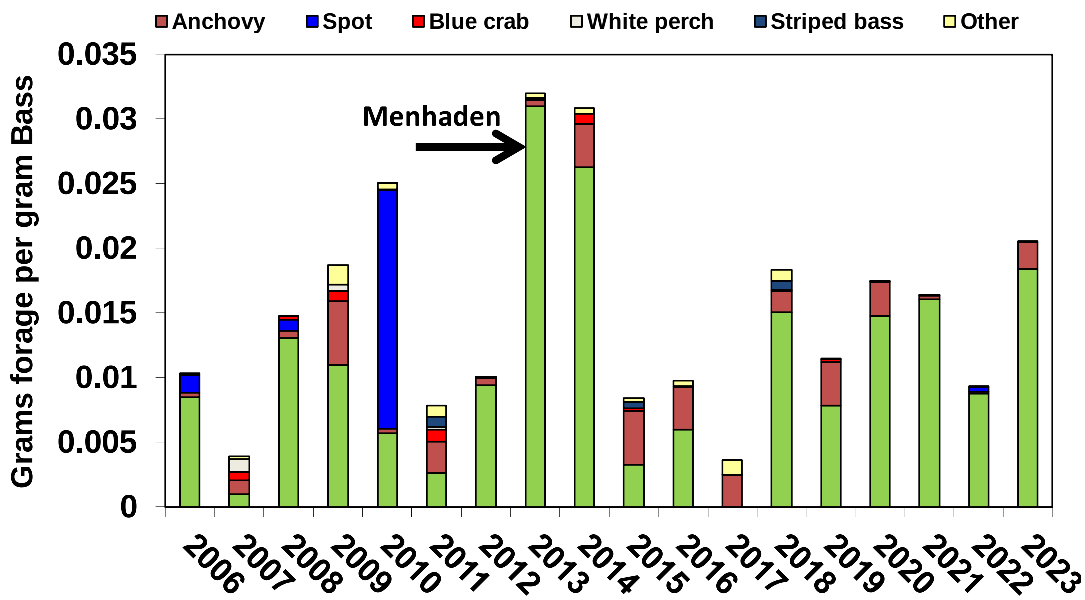

Figure 6. Median prey-predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for small (< 457 mm) Striped Bass. Optimum ratio was estimated by Overton et al. (2009).

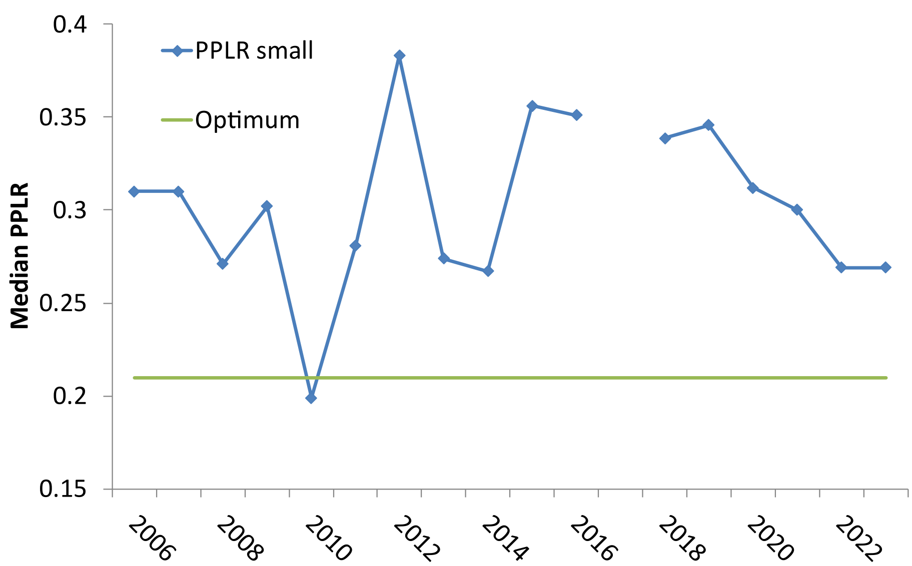

Figure 7. Proportion of large Striped Bass without body fat (P0) during October -November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

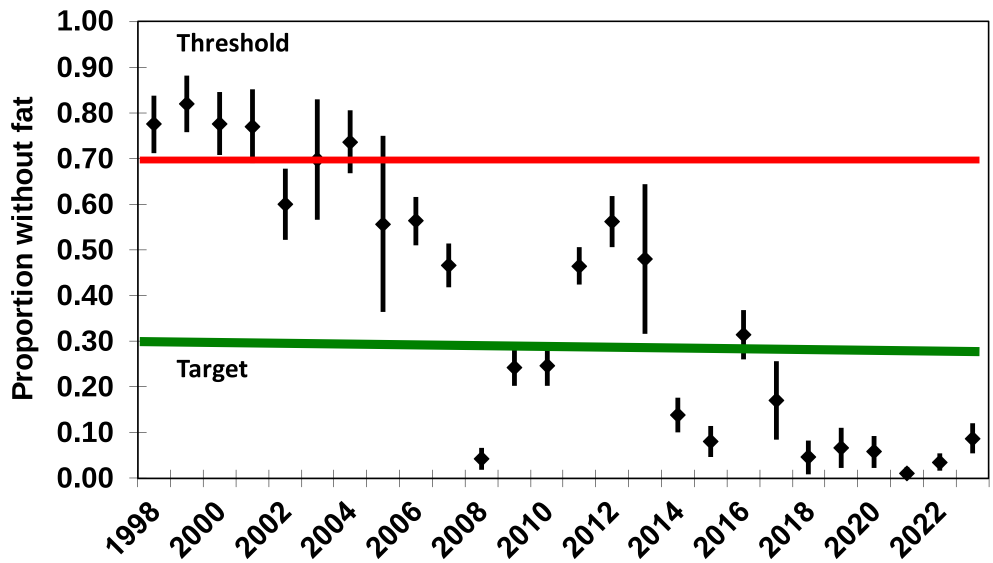

Figure 8. Proportion of large Striped Bass (≥ 457 mm or 18 in, TL) guts without food (PE) in fall and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

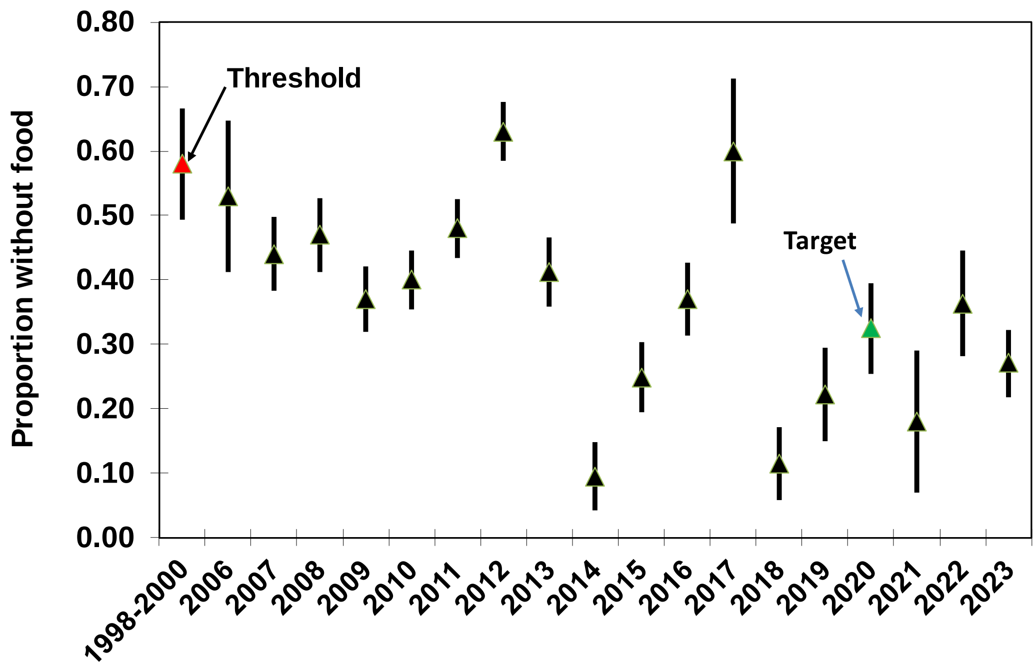

Figure 9. Plot of proportion of large (≤ 457 mm, TL) Striped Bass with empty guts in October-November against proportion without body fat, 2006 -2023.

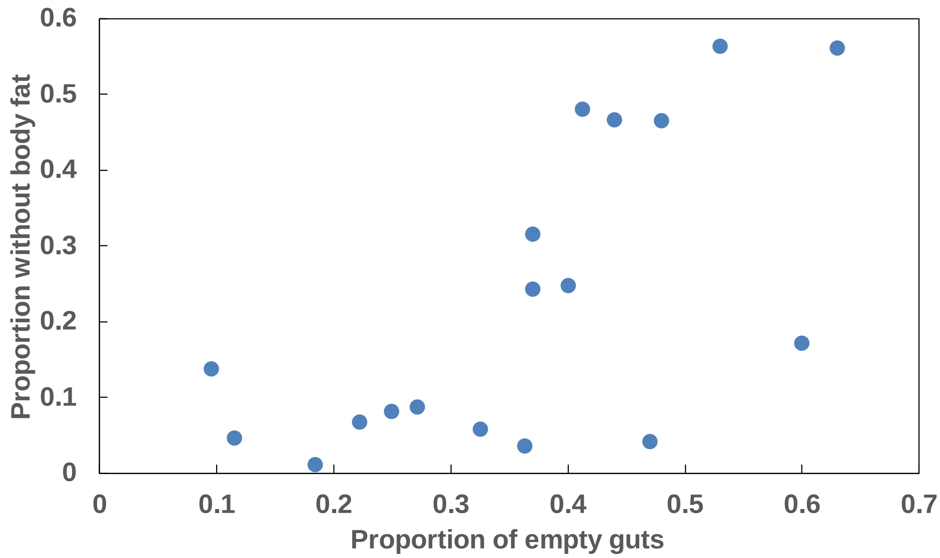

Figure 10. Percent of large Striped Bass (≥ 457 mm TL) identifiable diet represented by major forage groups, by number, in fall.

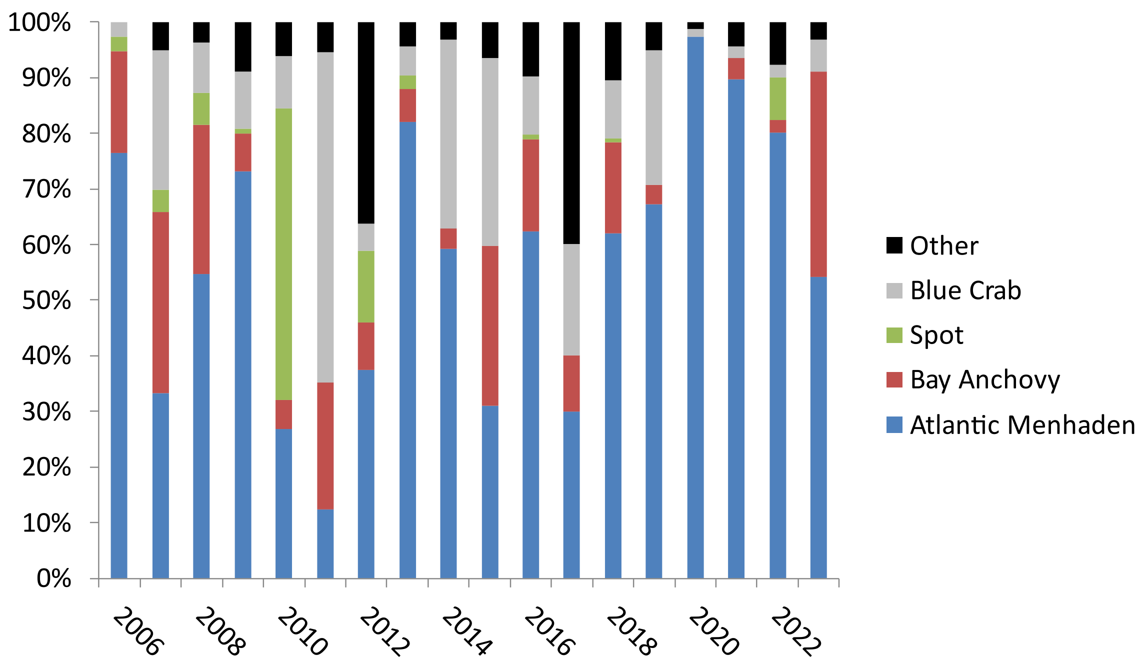

Figure 11. Grams of prey consumed per gram (C) of large (≥ 457 mm TL) Striped Bass during October-November. Fall consumption dominated by age 0 forage. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

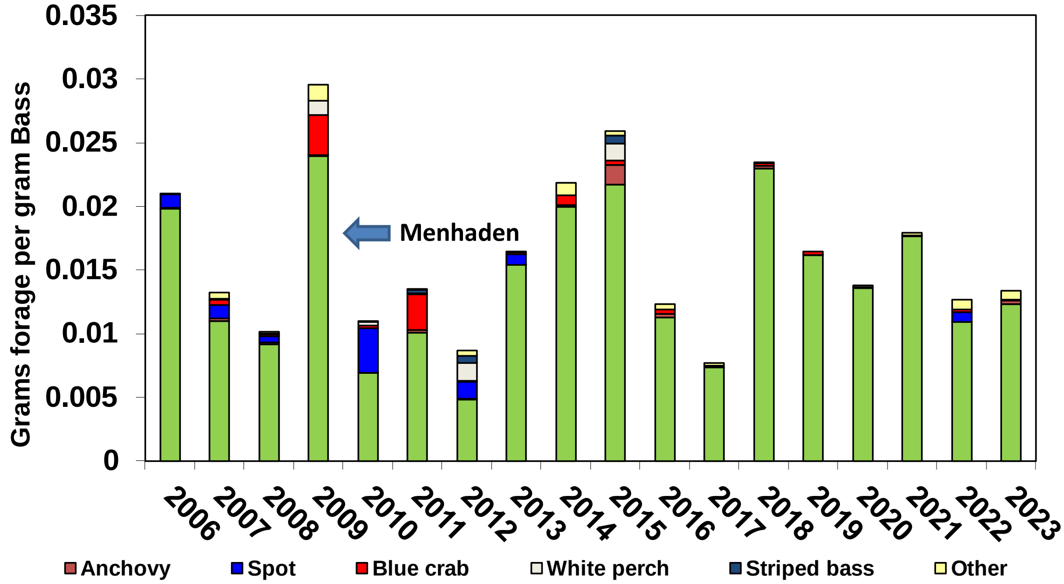

Figure 12. Median prey -predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for large Striped Bass (≥ 457 mm). Optimum ratio was estimated by Overton et al. (2009).

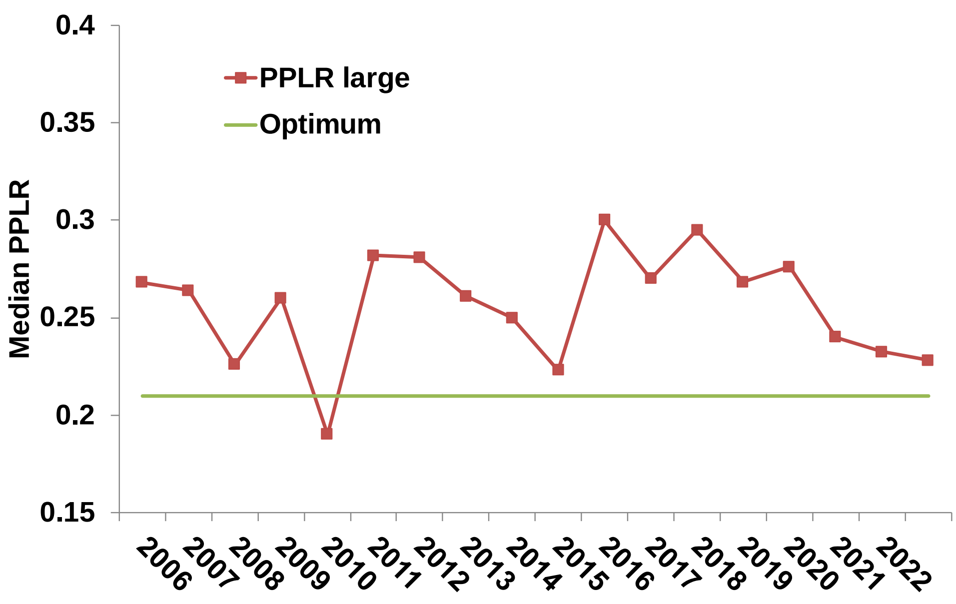

Figure 13. Maryland resident Bay Striped Bass annual abundance index (RI; MD MRIP inshore recreational catch per private boat trip during September -October; mean = black line) since 1983 and its 90% confidence intervals based on @Risk simulations of catch and effort distributions. Catch = number harvested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

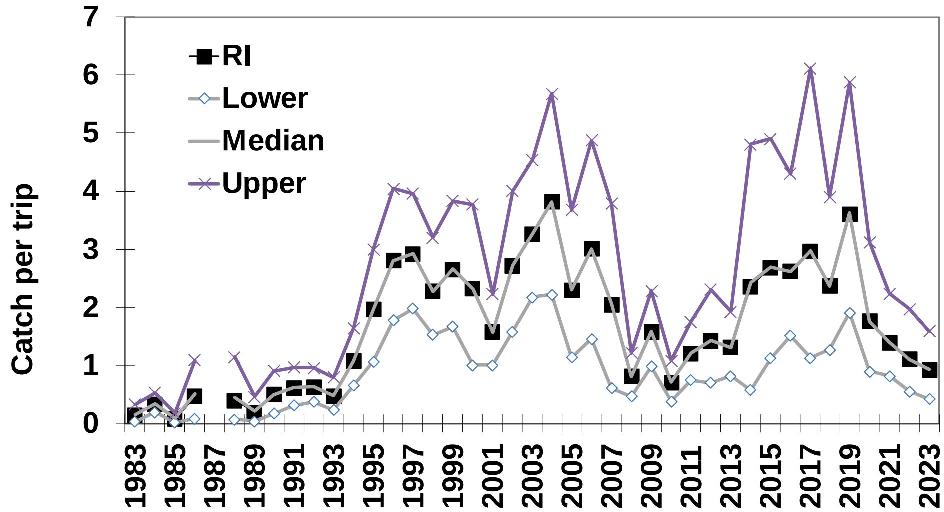

Figure 14. Trends in major pelagic prey of Striped Bass in Maryland Chesapeake Bay surveys since 1959. Indices were standardized to their means since 1989 (years in common). Menhaden = Atlantic Menhaden and Anchovy = Bay Anchovy.

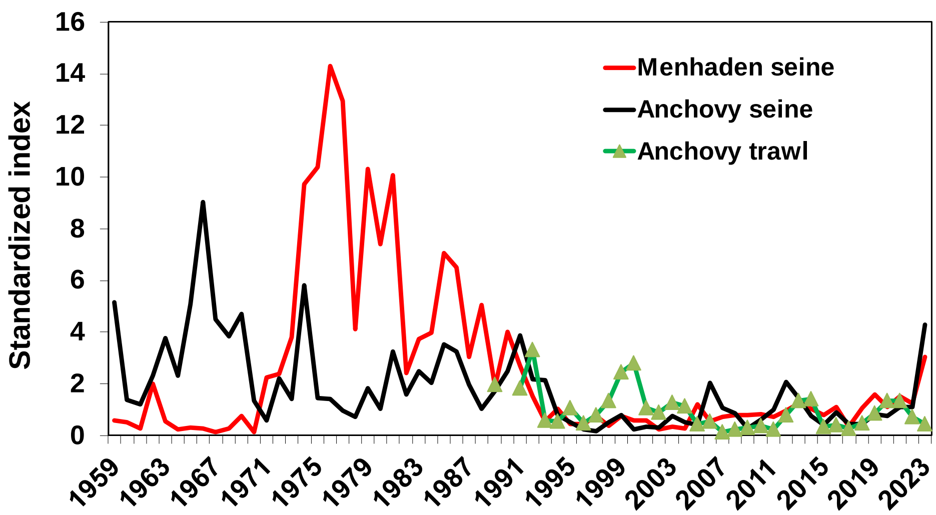

Figure 15. Trends in major benthic prey of Striped Bass in Maryland Chesapeake Bay surveys, since 1959. Indices were standardized to their means since 1989 (years in common).

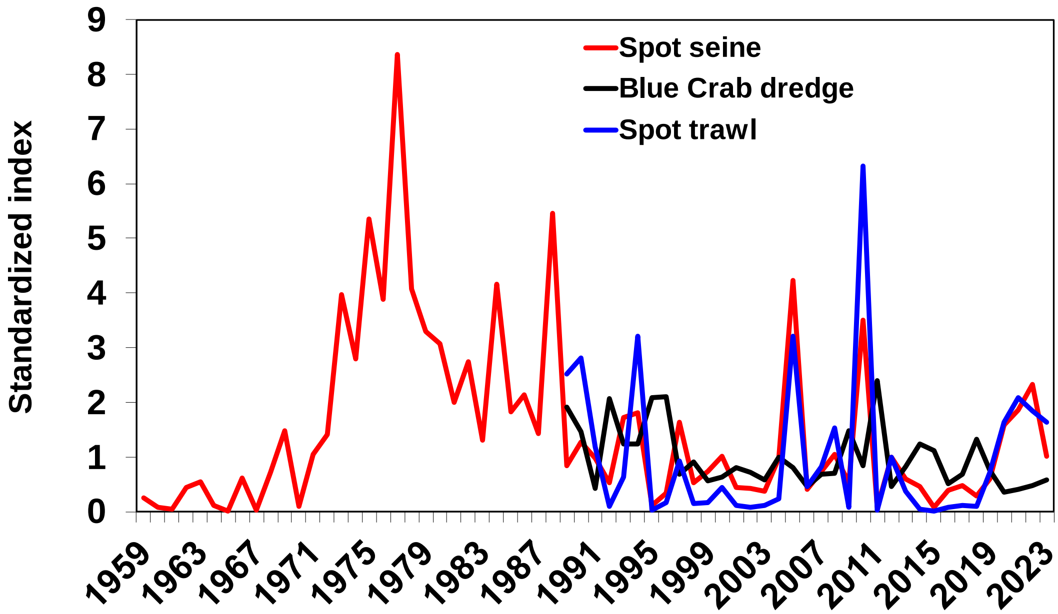

Figure 16. Trends in soft bottom benthic invertebrate biomass in Maryland waters (grams / m^2) and its median during 1995-2022 (based on Figure 3-38 in Versar 2024).

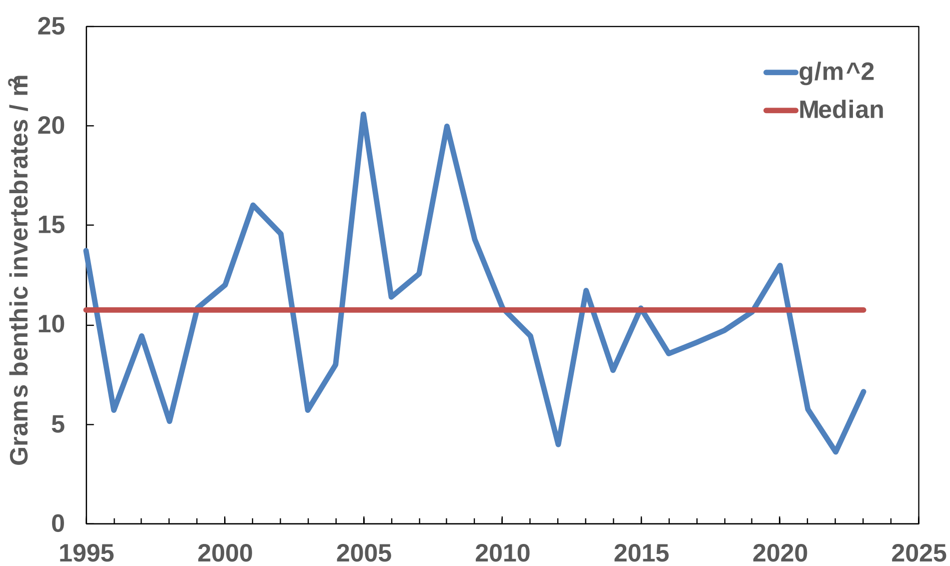

Figure 17. Trends of standardized ratios of major upper Bay forage species indices to Striped Bass relative abundance (RI). Forage ratios have been standardized to their means since 1989 to place them on the same scale. S indicates a seine survey index; T indicates a trawl survey index; and D indicates a dredge index. Note the log $_{10}$ scale on Y-axis.

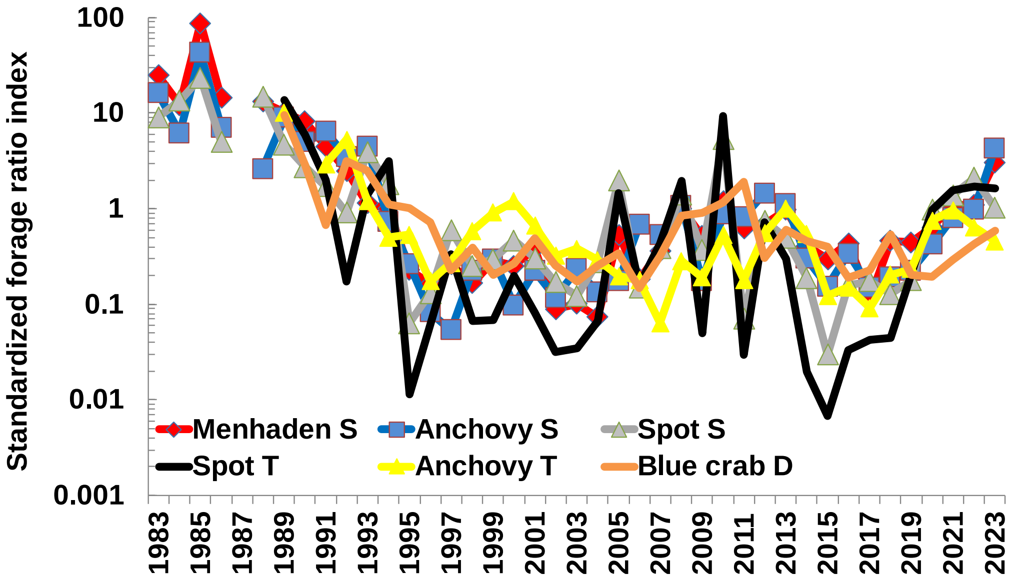

Figure 18. Atlantic Menhaden index to Striped Bass index (RI) ratios (Atlantic Menhaden FR) since 1983 and their 90% confidence intervals based on @Risk simulations of Atlantic Menhaden seine indices and RI distributions. Note log $_{10}$ scale on the Y-axis.

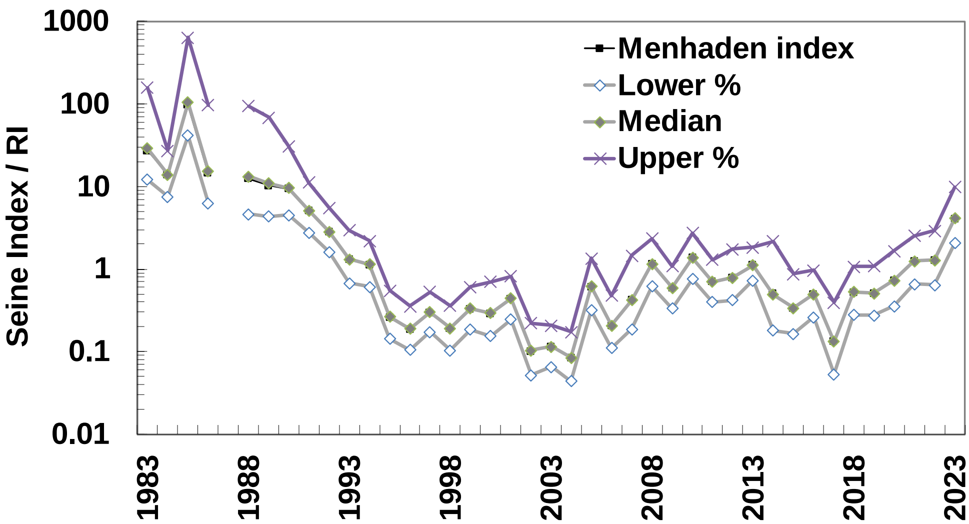

Figure 19. Bay Anchovy seine index to Striped Bass index (RI) ratios (Bay Anchovy seine FR) since 1983 and their 90% confidence intervals based on @Risk simulations of Bay Anchovy seine indices and RI distributions. Note $\log_{10}$ scale on the Y-axis.

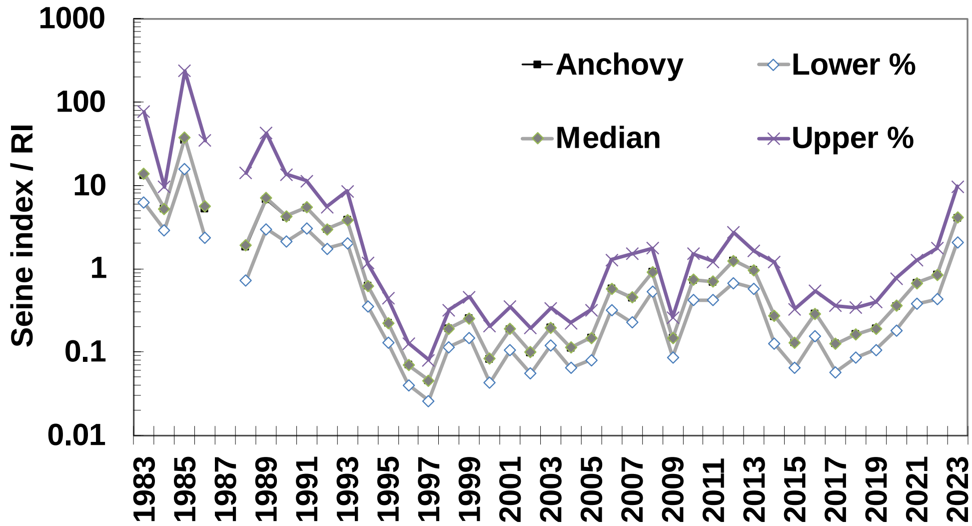

Figure 20. Bay Anchovy trawl index to Striped Bass index (RI) ratios (Bay Anchovy trawl FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale on the Y-axis.

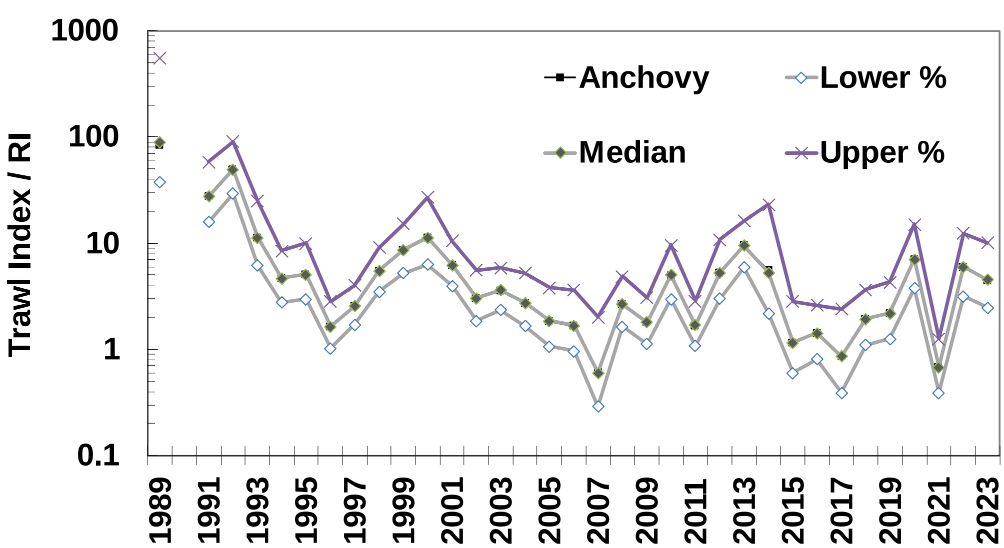

Figure 21. Spot seine index to Striped Bass index (RI) ratios (Spot seine FR) since 1983 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Spot seine indices and RI. Note log $_{10}$ scale on Y-axis.

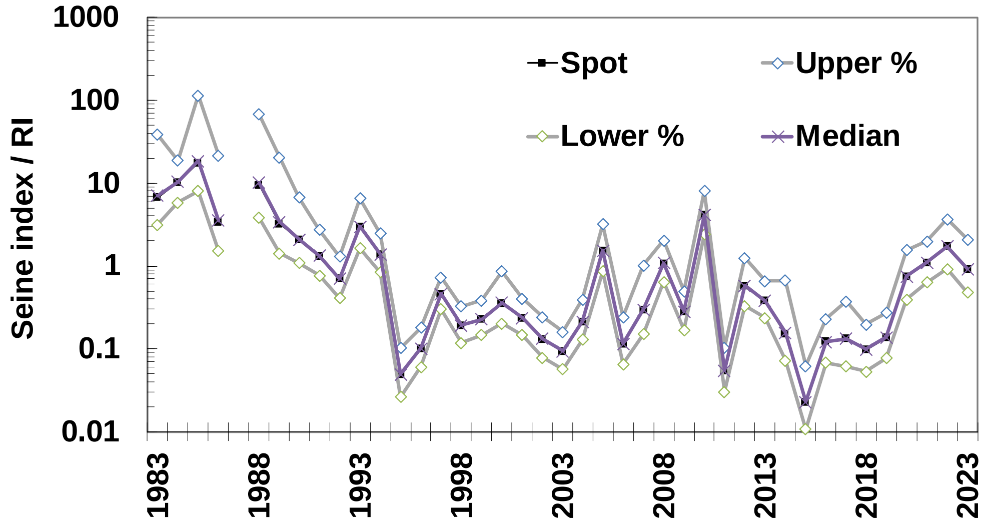

Figure 22. Spot trawl index to Striped Bass index (RI) ratios (Spot trawl FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note log $_{10}$ scale on Y-axis.

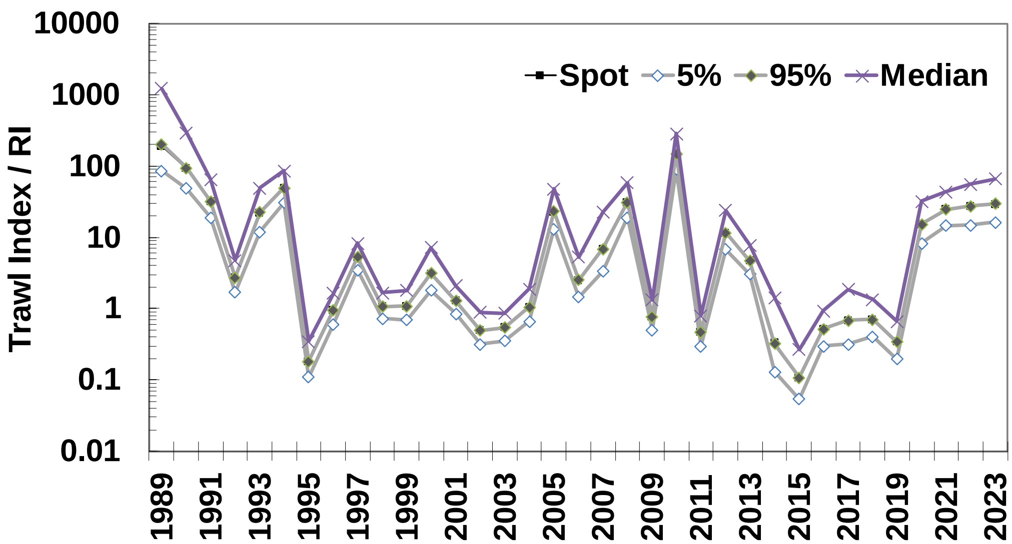

Figure 23. Blue Crab index to Striped Bass index (RI) ratios (Blue Crab FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Blue Crab (age 0) winter dredge densities and RI. Note the $\log_{10}$ scale on Y-axis.

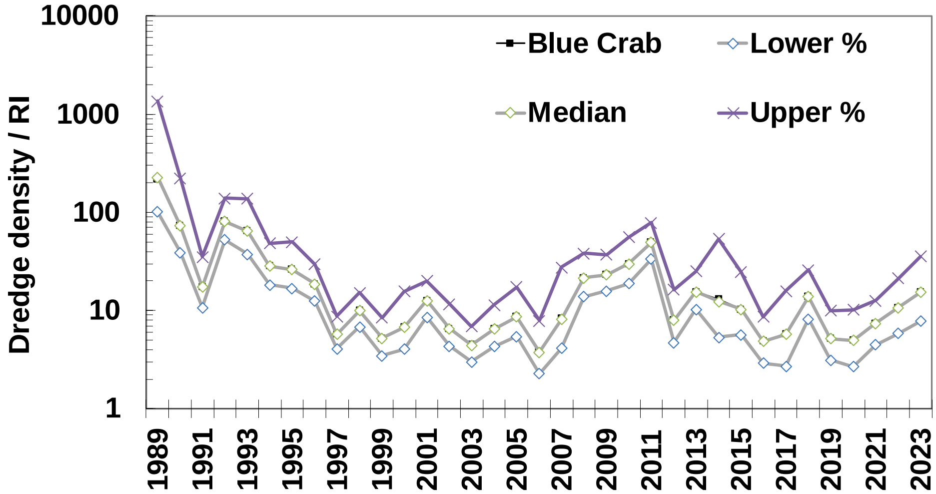

Figure 24. Time -series of age 3 Striped Bass relative abundance on two major Maryland spawning areas (hybrid index = gill net index adjusted for changing catchability during 1985-1995; units = number of fish captured in 1000 square yards of net per hour) and abundance of age 3 Striped Bass along the 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

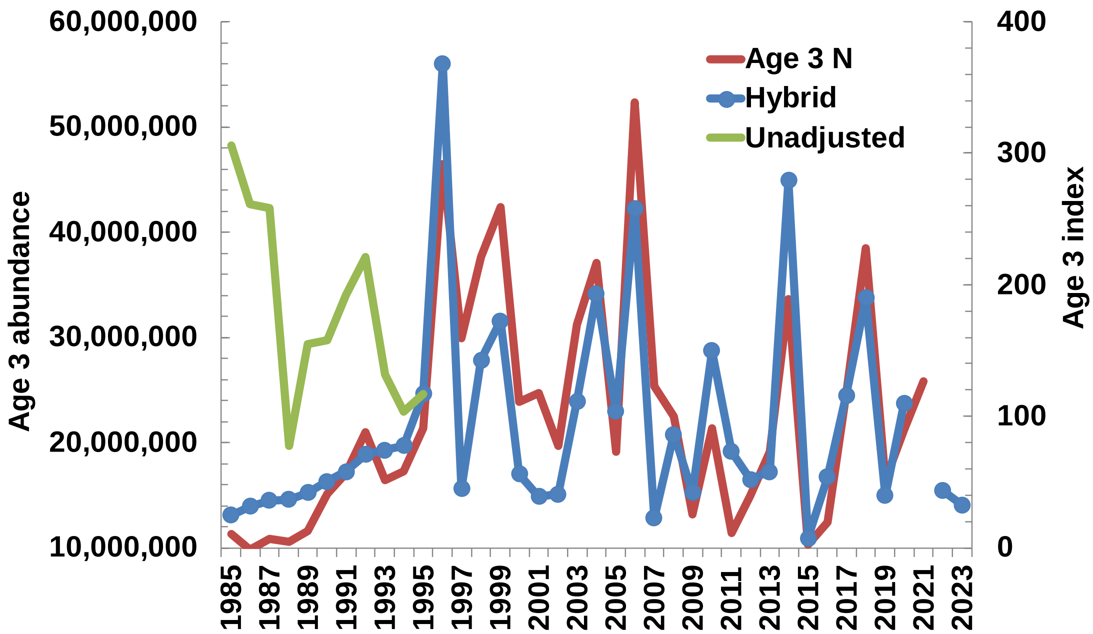

Figure 25. Relative survival (SR) of a Striped Bass year -class to approximately its third birthday during 1985-2022 and 90% confidence intervals based on @Risk simulations of age 3 hybrid gill net indices divided by juvenile index distributions. Year of estimate = year - class + 3. An estimate was not available for 2021 (2018 year -class).

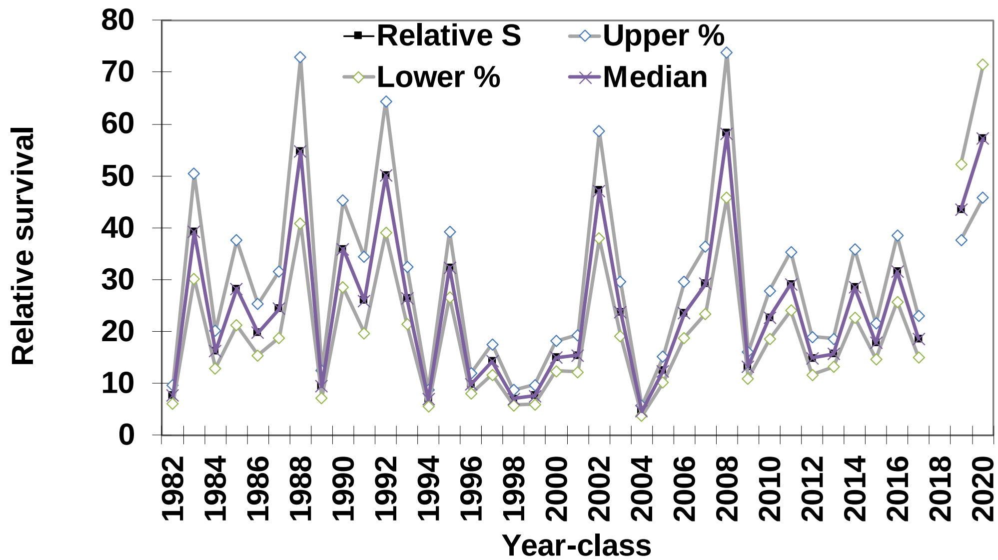

Figure 26. Relative survival (SR) of Striped Bass between ages 0 and 3, its median, and the relative abundance of resident Striped Bass (RI) in the previous year during 1985 -2023 (year-class = year – 3). An estimate was not available for 2021.