PERFORMANCE REPORT FOR FEDERAL AID GRANT F-63-R, SEGMENT 15

2024 Activities

MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

Maryland Department of Natural Resources
Fishing and Boating Services
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December 2025

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Department of Natural Resources Mission

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

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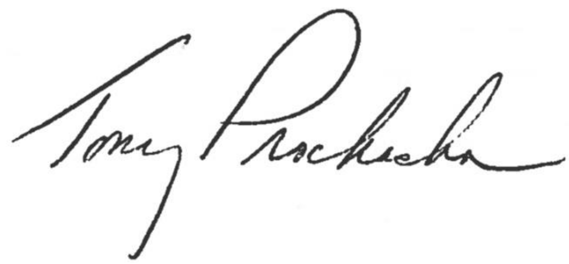
Approval



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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).

Report Organization

This report was completed during December 2025. 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 Ecosystem Assessment Division (FEAD) 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 production from this habitat. 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 (eggs) or growth (juvenile-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. Recently, 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 (rural; C/ha ≤ 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 suburban 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: Anadromous Fish Stream Spawning Ichthyoplankton – We surveyed four stations in Mattawoman Creek (1.04 C/ha or 11.5% IS) during February 28-May 15, 2024, adding to its 2008-2018 time-series. Estimates of C/ha progressed from 0.87 to 1.04 and %IS progressed from 10.2% to 11.5% IS during 2008-2024.

Proportion of samples with eggs and/or larvae of anadromous fish groups provided an indicator of habitat occupation in space and time. We emphasized the proportion of samples with Herring (Alewife, Blueback Herring, and Hickory Shad) eggs and-or larvae present (P_{herr}) because of its adequate sample size for precise annual estimates.

Mattawoman Creek is the sole forested watershed included in our surveys and much of this forest resides in 14,568 ha (out of 24,239 watershed ha) set aside in 2016 as the Watershed Conservation District – a low development resource conservation area. The county has been considering proposals that would rezone portions of the WCD for residential and industrial development. We returned to this watershed to update fish habitat indicators prior to the 10-year update of the county comprehensive growth plan.

Specific conductance (hereafter, conductance) measurements from Mattawoman Creek in 2024 ranged from 99-164 µS/cm (the lowest maximum value measured since 2013) reflecting low snowfall and low application of road salt. As in past years, conductance measurements during 2024 were generally highest at the most upstream site closest to Waldorf, declining as collections moved downstream through the WCD towards the site on the tidal border. The estimate of P_{herr} in 2024, 0.77, was tied for the second highest of the time-series. Herring spawning was detected at all four Mattawoman Creek sites during 2024.

We updated multiple regressions that regressed P_{herr} with standardized conductivity (specific conductance / region background level) or with C/ha from nine watersheds surveyed during 2005-2024. These regressions featured a categorical variable that accounted for the absence (0) or presence (1) of measures to reduce harvest and bycatch along the Atlantic coast that started in 2012. Conductance standardized to a coastal plain or Piedmont baseline increased with development, while P_{herr} declined with both development or standardized conductance. Predicted P_{herr} declined by nearly 50% over the observed range of C/ha (0.07-1.52). Both terms (harvest period and stressor) of these two models were significant and both models explained about 70% of variation in P_{herr}. Estimates of P_{herr} were consistently high in watersheds dominated by agriculture; forest cover is typically preserved along streams in rural watersheds.

Section 2: Estuarine Yellow Perch Larval Presence-Absence Sampling - Proportion of tows containing Yellow Perch larvae during a standard time period and where larvae would be expected (L_p), 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 in 2024 during late winter – early spring. Estimated L_p was determined annually from samples collected between the first day Yellow Perch larvae were caught until an 18°C water temperature cutoff was met.

Choptank River has a rural, agricultural watershed (C/ha = 0.13). Estimates of L_p from 2024 were compared to thresholds from brackish subestuaries based on a time-series of surveys from subestuaries with rural to urban watersheds stretching back to 1963.

In addition to examining the effects of development, we investigated the influence of winter temperature conditions on L_p during 1963-2024. Yellow Perch require a period of low temperature for reproductive success. We used summarized average winter air temperatures (December-February) at Baltimore as an indicator of regional winter intensity to investigate their relationship with L_p estimates for the Nanticoke River and Choptank River. These rivers were chosen because they have remained rural, had long time-series, and were lightly exploited.

The estimate for of mean L_p in Choptank River in 2024 (L_p = 0.61, SD = 0.06) was the tenth highest L_p out of 51 estimates for large brackish subestuaries. The chance that L_p fell below the brackish threshold in Choptank River during 2024 was 0%. The range of C/ha values available for analysis with L_p was 0.05-2.86 for brackish subestuaries. Estimates of L_p declined with development in brackish tributaries sampled. An extensive range of L_p estimates were present when C/ha was 0.22 and median L_p was 0.52. Beyond C/ha = 0.22, the range was similar but there were fewer high L_p values and median L_p decreased to 0.24.

Winter mean air temperatures in Baltimore (a regional winter temperature intensity indicator) increased during 1963-2024. Mean air temperature in Baltimore was modestly related to L_p in Nanticoke River and Choptank River and when 18°C was reached (end point for estimating L_p). This analysis provided some support for the hypothesis that reproductive success declines followed short, warm winters. Regression results revealed the presence of an underlying influence of winter intensity on L_p but did not explain annual variability that reflected other long and short-term influences. Correlation analysis indicated that winter temperature was positively and moderately associated with the North Atlantic Oscillation (NAO). The NAO is influenced by climate warming (higher CO_2 levels).

Section 2.1: Investigation of Striped Bass Spawning and Larval habitat Status in Maryland – This Choptank River survey did not cover general hypotheses about development. It investigated habitat conditions for Striped Bass eggs and larvae, reflecting concern about habitat during a current series of poor year-classes in Maryland spawning areas. During 2024, we collected basic water quality data (temperature, conductivity, dissolved oxygen or DO, pH, and alkalinity) from Choptank River and contrasted them with conditions during previous years. Our primary interest was in water temperature – an important driver of year-class success. Temperature data has been summarized from surveys with adequate sampling during 1954 to the present. 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 2024.

Estimated baywide *Ep* (1955-2024 time-series), based on Choptank River *Ep* in 2024 (0.58) was the lowest since 1989. There was a high chance it had crossed a threshold (0.60) to lower levels during 1982-1988 when depleted spawning stock negatively influenced year-class success. There was reason to believe that the 2024 estimate of *Ep* was negatively biased. Methodology required including samples from dates at temperatures or locations where spawning was rare. With these dates removed, estimated *Ep* would have been as high as 0.66-0.69 – a range with considerably lower risk of depletion. Estimated *Ep* in 2024 substantially overlapped *Ep* estimates in 2001 and 2011 that were concurrent with strong year-classes.

Estimated relative larval survival (RLS; baywide juvenile index / *Ep*) in 2024 indicated poor survival; most of the poor RLS estimates were concentrated in 1980-1991 and since 2019. Estimates of RLS near or below the poor survival criterion were absent during 1993-2001 but returned afterward and occurred intermittently through 2019.

We added temperature loggers at two stations within the spawning area to collect data at 30-minute intervals during 2024. Linear regressions did not indicate significant differences when water temperatures were measured at the same site by surveys or loggers. We added an egg volume index in addition to egg presence-absence to better define spawning intensity during 2024. In lieu of counting, we assigned a rank to egg volume in a sample jar. We compared the daily mean intensity indices to water temperature measurements from two continuous temperature loggers.

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, 16ºC, and 20ºC (D as days from April 1; April 1 = day 0) were consistently met in the Choptank and Nanticoke rivers. Temperature conditions in these two adjacent rivers should have been similar. We used summarized average winter air temperatures (December-February) at Baltimore during 1954-2024 as an indicator of regional winter intensity (T_w) and explored its relationship with temperature milestone dates (D). Inspection of the bivariate plot indicated the possibility of an asymptote through an initial portion of T_w (an asymptote) and a decrease afterward. To avoid applying a complex nonlinear equation to fit these data, we used T_w^2 as the independent term in a linear regression with D, transforming negative to positive values and approximating an asymptote for lower values.

The first egg was collected on March 25, 2024, the second earliest date that spawning has been detected in Choptank River and Nanticoke River ichthyoplankton surveys. The 12ºC milestone was reached on April 1, the third earliest date that milestone was reached. The mid-milestone, 16ºC, was reached on April 15 and was the tenth earliest of the time-series. The 20ºC milestone was reached on April 29 and was also the 10th earliest. These milestones have been reached earlier since 2017. The 2024 spawning season, based on the dates that 12°C and 21°C were reached, ran 28 days. The egg intensity index detected peak spawning on April 10.

Changes in average dates and plots of 12ºC and 16ºC milestone dates since 2010 suggested clustering at earlier dates below series medians for 1954-1999 and medians for years with strong year-classes. These shifts were noticeable for 2019 and later. Clustering was not readily apparent for the 20ºC milestone. Changes between 12ºC and 16ºC could be important to the formation of strong year-classes because most spawning occurs between these temperatures.

Spawning may start at low, lethal temperatures and a linear regression indicated that early spawning (first egg date) at temperatures below the 12ºC date was becoming more frequent. There was a positive trend over time for the difference between the 12ºC date and date that a first egg was collected during 1954-2024, indicating a shift of earliest spawning (first egg) after the 12ºC milestone date to the earliest spawning preceding 12ºC. Early spawning became less synchronous with the initial water temperature trigger.

Winter mean air temperatures in Baltimore (T_w) increased during 1954-2024. Linear regressions of D with T_w^2 did not account for a large amount of variation but they indicated that winter intensity could underlie long-term changes in temperature milestones relevant to spawning and prolarval survival. We found a moderate association of the North Atlantic Oscillation (NAO) with regional winter intensity that was a potential underlying influence on milestone temperatures. The NAO is influenced by climate warming.

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 near or above average baseline flow of 1957-2020 (1.0) during 2024 in Choptank River (1.83), Nanticoke River (1.71), Head-of-Bay (1.17), and Potomac River (0.99). Choptank River and Nanticoke River flows were among the highest since 1993.

Section 2.2 - Influence of feeding on zooplankton on Striped Bass postlarval mortality, growth, and year-class success - We examined first-feeding Striped Bass postlarvae in Choptank River during 2023-2024 to address whether feeding success on zooplankton could be a major factor behind a series of poor year-classes during 2019-2024. We estimated Choptank River Striped Bass postlarval feeding incidences on primary zooplankton prey and their associations with daily instantaneous mortality rates (Z) from seven 1980s surveys (low and high Z and poor to strong year-classes) to establish criteria to evaluate 2023-2024 collections. Feeding incidences of first-feeding Striped Bass postlarvae on copepods in Choptank River during 2023-2024 were high; feeding incidence on cladocerans was also high in 2024. Estimates of a proxy index for postlarval Z during 2023 and 2024 were low. However, year-class success was dismal during 2023 and low in 2024. This feeding investigation did not encompass the entire 2019-2024 drought in year-class success, but 2023-2024 did not indicate a consistent, prominent role for feeding success of postlarvae. High feeding incidence of first-feeding Striped Bass postlarvae on zooplankton and low mortality did not always translate to better year-class success during the 1980s and 2023-2024.

Section 3 - Estuarine Fish Community Sampling – Sampling of juvenile and adult target fish habitat in watersheds at various levels of salinity and development occurred during summer. Dissolved oxygen (DO) was the primary environmental response variable for development. Sampling during 2003-2024 resulted in 173 subestuary and year combinations: 98 combinations have been in mesohaline (5.0-18.0‰) subestuaries, 18 have been in oligohaline (0.5-5.0‰), and 57 have been tidal-fresh (< 0.5‰).

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 (structures per hectare, C/ha, and percent impervious surface, %IS) in mesohaline subestuaries, reaching average levels below 3.0 mg/L (threshold level) when development was beyond its threshold (0.84 C/ha or 10% IS); occupation of bottom channel habitat diminishes at or below threshold DO. The target level of development that provided best habitat was 0.31 C/ha (5% IS) or less. 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 chronically 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 become important.

Median bottom DO in mesohaline subestuaries increased as agricultural coverage of a watershed went from 3% to 50% and the DO trend appeared to be stable or slightly declining when agricultural coverage was 43-72%. A dome-shaped quadratic model of summer median bottom DO and agricultural coverage provided a moderate fit to the data. Below threshold median bottom DO was predicted when agricultural coverage fell below 14%. Median bottom DO was predicted to peak at about 50% agricultural coverage and modest declines in bottom DO would occur through 72% agricultural coverage. Predicted median bottom DO was 5 mg/L or more (target level) between 34% and 70% agricultural coverage. Agricultural coverage and C/ha were moderately 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 chronic 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 chronically low DO was absent.

In 2024, we evaluated summer nursery and adult habitat for recreationally important finfish in Mattawoman Creek (tidal-fresh; C/ha = 1.04), Piscataway Creek (tidal-fresh, C/ha = 1.61), Tred Avon River (mesohaline; C/ha = 0.79), Miles River (mesohaline, C/ha = 0.27), West-Rhode River (mesohaline, C/ha = 0.62), and Magothy River (mesohaline, C/ha = 2.95). Dissolved oxygen was most frequently below the 5 mg/L target or 3 mg/L threshold in bottom channel waters of mesohaline Magothy River that had over threshold watershed development. Frequency of below threshold DO in mesohaline subestuaries were 45.8% in Magothy River, 4.2% in Tred Avon River, 12.5% in West-Rhode River, and 0% in Miles Creek. 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). Piscataway Creek was too shallow to measure bottom DO.

A total of 35,219 finfish representing 40 species were captured by beach seine in 2024 and total of 55,884 finfish and 42 fish species were captured by bottom trawl. Anadromous fish target species encountered in 2024 were American Shad (3), Hickory Shad (0), Blueback Herring (74), Alewife (4), and Striped Bass (191 YOY and 5 age 1). Estuarine resident target species encountered were White Perch (6,322 juvenile, 701 adult), Yellow Perch (8), and Bay Anchovy (6,645). Marine target species encountered in 2024 were Atlantic Menhaden (25,141) and Spot

(40,225). Tidal-fresh target species encountered in 2024 were Spottail Shiner (566), Eastern Eastern Silvery Minnow (43), and Gizzard Shad (122). Atlantic Menhaden were the most abundant species captured in beach seines at 71% of the total catch and Spot were the most abundant species in bottom trawls (67%) during 2024.

Bottom trawl geometric means (GMs) for all finfish combined during 2003–2024 did not exhibit an obvious decline with C/ha for oligohaline or tidal-fresh subestuaries. They declined with C/ha in mesohaline subestuaries and a negative threshold response was suggested at C/ha between 0.8 and 1.2; this decline reflected a change to consistently low DO in mesohaline bottom channel waters with increasing development. There was wide variation in GMs prior to the development threshold and they declined to very low levels afterward. The median trawl GM during 2003-2024 calculated for mesohaline subestuaries with below target watershed development (C/ha=0.31) was 114 (N=50); it was 89 (N=33) with watershed development between the target and threshold (C/ha=0.84); and 10 (N=11) when development was greater than threshold. Geometric means specific to 2024 sampling conformed to the relationships between C/ha and GM catch for the three salinity types encountered.

A linear regression of proportion of positive tows (P-A) of finfish in bottom channel trawl samples with C/ha for watersheds with mesohaline subestuaries indicated a moderate negative influence of C/ha on the P-A of finfish in the bottom trawls. A plot of P-A of finfish in bottom channel trawl samples and median bottom DO in mesohaline subestuaries exhibited considerable variability (0-1.0) after passing the threshold of 3.0 mg/L DO; P-A of all finfish only exceeded 0.6 when DO was near or above 3.0 mg/L.

Relative abundance of juvenile Striped Bass in our surveys was important as supplemental information on distribution in areas not normally assessed for the Maryland juvenile index. Catches in our 2024 survey were low, reflecting poor juvenile indices in the four spawning areas routinely monitored.

White Perch is an important target species and the only popular gamefish that we can survey well as juveniles and adults. A negative threshold response of relative abundance and presence-absence was suggested at C/ha between 0.8 and 1.2 during 2003–2024 in mesohaline subestuaries. This decline reflected the change to consistent low DO conditions in mesohaline bottom channel waters that occurred with increasing development. Negative responses were not evident in oligohaline or tidal-fresh subestuaries.

Modified Proportional Stock Density (PSD) indices indicated that tidal-fresh subestuaries were primarily habitat for juvenile fish too small for anglers to harvest. Mesohaline subestuaries with chronic extensive low bottom channel DO measurements had highly variable modified 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 mesohaline subestuaries with rural or transition watersheds and least likely to be found in subestuaries with suburban-urban watersheds.

Common Background for Objective 1, Sections 1-3

Jim Uphoff

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

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

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

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 (eggs) or growth (juvenile-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 human 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). 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). Ecological stress from development of the Bay watershed conflicts with demand for fish production and recreational fishing opportunities (Uphoff et al. 2011).

Impervious surface is used as an indicator of development because of compelling scientific evidence of its effect in freshwater systems (Wheeler et al. 2005; NRC 2009) and because it is a critical input variable in many water quality and quantity models (Arnold and Gibbons 1996; Cappiella and Brown 2001). Impervious surface itself increases runoff volume and intensity in streams, leading to increased physical instability, erosion, sedimentation, thermal pollution, contaminant loads, and nutrients (Beach 2002; Wheeler et al. 2005; NRC 2009: Hughes et al. 2014a; 2014b). Urbanization may introduce additional industrial wastes, contaminants, stormwater runoff and road salt (Brown 2000; NRC 2009; Benejam et al. 2010; McBryan et al. 2013; Branco et al. 2016; Kaushal et al. 2018; Baker et al. 2019) that act as ecological stressors and are indexed by impervious surface. The NRC (2009) estimated that urban stormwater was 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 accounted 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). 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 fishery managers, 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 subestuaries (Uphoff et al. 2024). Conserving watersheds at or below 5% IS would be a viable fisheries management strategy. Increasingly stringent fishery regulation or hatchery supplementation might compensate for habitat related production losses as IS increases from 5 to 10%. Above a 10% IS threshold, habitat stress mounts and successful management by harvest adjustments and stocking alone becomes increasingly unlikely (Uphoff et al. 2011; Interagency Mattawoman Ecosystem Management Task Force 2012; Uphoff et al. 2024). 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. 2022; see **General Spatial and Analytical Methods used in Objective 1, Sections 1-3**). Counts of structures per hectare (C/ha) had strong relationships with IS (Topolski 2015; 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, stressed suburban watershed) were estimated as 0.31, 0.84, and 1.51 C/ha, respectively (Uphoff et al. 2022). Tax map data provide a development time-series that goes back to 1950, making retrospective analyses possible. Development in Maryland's portion of the Chesapeake Bay watershed, approximately 0.17 C/ha in 1950, reached 0.82 C/ha (9.9% IS) in 2023 (M. Topolski, MD DNR, personal communication).

The major spawning tributaries used by Striped Bass, White Perch, Yellow Perch, Alewife, Blueback Herring, Hickory Shad, and American Shad are typically on the receiving end of large amounts of agricultural drainage because of their location at the junction of large fluvial systems and brackish estuaries. Trends in juvenile indices of these species are similar, indicating similar influences on year-class success (Uphoff 2008; 2023).

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

A report, *Comprehensive Evaluation of System Response* (CESR), was released in 2023 to the CBP that advocated for more explicit consideration of living resources responses in planning and actions to improve water quality (STAC 2023). Four decades of efforts to manage nutrient and sediment pollutants have improved water quality conditions in some portions of the Chesapeake Bay, but results have been mixed. Additionally, changing conditions from population growth, land use, and climate will make future restoration more challenging. The CESR report recommended refocusing water quality management efforts on improving living resource response and shifting emphasis from slow to respond deep channel waters of the main Bay to shallow waters to accelerate and better understand attainment of water quality standards and benefits to living resources (STAC 2023). Activities under F-63 have focused on habitat, forage fish, and gamefish responses in subestuaries for decades that should qualify as shallow waters that fit the new CBP emphasis.

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MD – Marine and estuarine finfish ecological and habitat investigations

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

General Spatial and Analytical Methods

Marek Topolski and Jim Uphoff

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). 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. The equation that best described the relationship was

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

The C/ha equivalents for 5%, 10%, and 15% IS were 0.31, 0.84, and 1.51, respectively (Uphoff et al. 2022). These 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 (Uphoff et al. 2024). Current calibrations are based on two tax data years (2013 and 2018) which align with the Chesapeake Conservancy LC data years (Uphoff et al. 2024).

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) categories were estimated for each year of MD DOP spatial data to track broad patterns of LULC (Uphoff et al. 2024). 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

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 (Uphoff et al. 2024). 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. 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; 2024).

Watersheds used to model %IS underwent modest increases 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; Uphoff et al. 2024). 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. 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; Uphoff et al. 2024).

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 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 $r \geq \pm 0.80$; weak or poor correlations were indicated by $r < \pm 0.50$; and moderate or modest 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:

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

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

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

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

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

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

Shannon Moorhead, Marisa Ponte, Jeffrey Horne, Marek Topolski, Robin Minch, and Jim Uphoff

Introduction

Urbanization, spurred by increased population growth, has been a factor in the decline of diadromous fishes since the late 20^{th} century (Limburg and Waldman 2009). Associated increases in development have contributed to substantial diadromous fish habitat loss (Limburg and Waldman 2009). In some watersheds, such as the Hudson River, anadromous fish egg densities (Alewife and White Perch) have been shown to exhibit a strong negative threshold response to urbanization (Limburg and Schmidt 1990). We were interested in understanding how impervious surface reference points (ISRPs; Uphoff et al. 2011) or analogous structure per hectare (C/ha) reference points (Uphoff et al. 2024), developed for Chesapeake Bay subestuaries, were related to anadromous fish spawning in streams in Maryland's portion of the Chesapeake Bay watershed.

To address this question, we have conducted periodic surveys to identify spawning habitat of White Perch, Yellow Perch, and Herring (Blueback Herring, Alewife, American Shad, and Hickory Shad) in Maryland streams. These surveys were based on the sites and sampling methods of "historic" surveys conducted in Maryland during 1970-1986; data from these surveys were used to develop statewide maps depicting anadromous fish spawning habitat (O'Dell et al. 1970, 1975, 1980; O'Dell and Mowrer 1984; Mowrer and McGinty 2002; Uphoff et al. 2020; Maryland Department of Natural Resources [accessed Oct. 6, 2025]). In the time elapsed since these surveys occurred, many of these watersheds have undergone considerable development, and recreating the surveys provided an opportunity to explore whether spawning habitat has declined in response. Over the past 20 years, we have employed O'Dell et al. (1975, 1980) and O'Dell and Mower's (1984) methodology in nine Chesapeake subestuaries: Bush River (2005-2008, 2014), Piscataway Creek (2008-2009, 2012-2014), Deer Creek (2012-2015), Patapsco River (2013-2017), Choptank River (2016-2017), Tuckahoe Creek (2016-2017), Chester River (2019), Patuxent River (2021), and Mattawoman Creek (2008-2018, 2024; Figure 1-1).

We developed two indicators of anadromous fish spawning in a watershed based on presence-absence of eggs and/or larvae (i.e., ichthyoplankton): occurrence at a site (a spatial indicator) and proportion of samples with eggs and/or larvae (a spatiotemporal indicator). Occurrence of eggs or larvae of an anadromous fish group (White Perch, Yellow Perch, or Herring) at a site recreated the indicator developed by O'Dell et al. (1975, 1980) and O'Dell and Mowrer (1984). This spatial indicator was compared to the extent of development in the watershed (counts of structures per hectare or C/ha; Topolski 2015) between the 1970s and the present. Proportion of samples with eggs and/or larvae of anadromous fish groups, estimated from collections that began in the 2000s, provided an indicator of habitat occupation in space and time. This spatiotemporal indicator was compared to level of development (C/ha) and specific conductance (hereto after referred to as "conductance"), a freshwater quality metric strongly associated with development (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012; Moore et al. 2017; Bird et al. 2018).

Additionally, we attempted to discern how the proportion of samples with anadromous Herring eggs and/or larvae observed throughout the time-series may have been affected by increases in spawning stock abundance resulting from more restrictive regulatory measures implemented coast-wide over the past decade. It is a possibility that in-river fisheries closures along the Atlantic Coast, with most in place by 2012 (including Maryland in 2011; ASMFC 2019), and caps on River Herring bycatch in Atlantic Herring and Mackerel fisheries, beginning in 2014, (MAFMC 2019) boosted Herring spawning stocks. Subsequently, increases in the presence of Herring eggs and/or larvae due to regulatory measures (or other undetected large-scale factors, such as decreased predation or increased at-sea survival, due to improved feeding and/or environmental conditions) would theoretically be evident across the three watersheds studied before and after regulatory measures were put in place. Increases in spawning stock abundance over time could potentially bias estimated relationships of C/ha and conductance with indicators of anadromous Herring stream spawning intensity.

In 2024, our survey focused on Mattawoman Creek for the first time since 2018 (Figure 1-2). This new year of sampling further expanded what is already our longest data time-series (Table 1-1). Our renewed interest in surveying anadromous spawning in Mattawoman Creek was in response to an upcoming ten-year update to the Charles County Comprehensive Plan and signs of retreat on its natural resource conservation aspects. The only primarily forested watershed included in our surveys, this system historically supported anadromous fish spawning, as well as healthy, biodiverse stream and estuarine communities with productive fisheries, including one of the country's most prominent Largemouth Bass fisheries (Interagency Mattawoman Ecosystem Management Task Force 2012). Carmichael et al. (1992) described Mattawoman Creek as "near to the ideal conditions as can be found in the northern Chesapeake Bay, perhaps unattainable in other systems", going on to state that it "should be protected from overdevelopment".

Mattawoman Creek's drainage lies within Charles County, and land use within the watershed is governed by the county's Comprehensive Plan, last updated in 2016. Ahead of a scheduled update to this plan, an Interagency Mattawoman Ecosystem Management Task Force was formed, comprised of representatives from an array of MDNR services and units (e.g., Fishing and Boating Services, Maryland Biological Stream Survey, Forest Service, Environmental Review Unit), as well as other state and federal agencies (e.g., MD Department of Planning, MD Department of Environment, MD State Highway Administration, USDOT, US Fish and Wildlife Service, US EPA) and non-governmental organizations (i.e., University of Maryland Anthropology Department); this task force developed Land Use recommendations to best conserve the Mattawoman Creek watershed's resources, sharing them with Charles County in 2012 (Interagency Mattawoman Ecosystem Management Task Force 2012). Specifically, the task force voiced trepidations regarding development and acknowledged the potential for irreversible damage to resources within the Mattawoman Creek drainage. At the time, the county's existing plan designated the majority of the Mattawoman watershed land as some form of "deferred development district". The task force highlighted this as especially concerning; at the time, impervious surface cover within the Mattawoman watershed already reached the 10% threshold, at which the negative impacts of impervious surface begin to manifest as declines in biodiversity and fisheries production. Furthermore, if development continued unchecked, the watershed was projected to reach over 14% impervious surface in 2020, and over 22% if developed to the maximum extent allowed by the county's plan (Table 1-1; USACOE 2003; Interagency Mattawoman Ecosystem Management Task Force 2012).

Upon reviewing task force recommendations, the Charles County Board of Commissioners voted to amend their 2016 Comprehensive Plan, vowing to impose new development standards that promote conservation and rezoning approximately 36,000 acres of land for resource preservation via the establishment of a Watershed Conservation District (WCD; Charles County Government 2016; Chesapeake Bay Foundation 2017). Since its creation, the WCD has been fairly effective at maintaining forest cover in this watershed. However, in recent years the county Board of Commissioners has been considering proposals that would rezone portions of the WCD for residential and industrial development (J. Uphoff, personal communication). Reinstituting our survey here in 2024 illuminated whether Mattawoman Creek continues to provide valuable spawning habitat for anadromous fish species; in the event development plans move forward, subsequent annual sampling would allow us to directly monitor how spawning viability may change with increasing development in this watershed.

Methods

Study Area - The Bush River watershed, located along an urban gradient originating from Baltimore, Maryland, falls within both the Coastal Plain and Piedmont physiographic provinces. Adjacent to the north, Deer Creek's drainage lies within a conservation district located entirely in the Piedmont province, near the Pennsylvania border (Clearwater et al. 2000). Like the Bush, the Patapsco River watershed falls within both the Coastal Plain and Piedmont provinces, with characteristic Piedmont rolling hills covering much of its area while the southeast portion of the watershed, within the Coastal Plain, borders the western shore of the Chesapeake Bay (O'Dell et al. 1975). Fluvial Patapsco River meets the Chesapeake Bay and forms the Port of Baltimore. The Patuxent River, also located within both Piedmont and Coastal Plain provinces, is a major tributary of the Chesapeake Bay and the largest river located entirely within the state of Maryland. The upper portion of the drainage (north of MD Route 214, including the Little Patuxent River drainage) is located between Washington, D.C. and Baltimore, while the middle portion of the drainage (south from MD Route 214 to Hall Creek) extends through Anne Arundel, Prince George's, and Calvert counties (O'Dell and Mowrer 1984; Figure 1-1). The Patuxent River is urbanized, with extensive development that has negatively affected water quality, physical characteristics, and Herring spawning distribution and year-class success (O'Dell and Mowrer 1984; Uphoff et al. 2018, 2023; Table 1-1; Figure 1-1). Southwest of the Patuxent drainage, Piscataway and Mattawoman Creeks are adjacent Coastal Plain watersheds, lying along an urban gradient emanating from Washington, D.C.; Piscataway Creek's drainage, proximal to District of Columbia's development, is smaller than that of Mattawoman Creek (Table 1-1; Figure 1-1). On the eastern shore of the Chesapeake, which lies entirely within the Coastal Plain, the Choptank River is a major tributary of the Bay; this agriculturally dominated watershed also includes Tuckahoe Creek's drainage. Similarly, the Chester River, a fluvial-tidal system, is located on the eastern shore and agriculture is the primary land use type within its watershed (O'Dell et al. 1975; Table 1-1; Figure 1-1).

Within the Mattawoman Creek watershed, the focus of our 2024 ichthyoplankton sampling effort, our survey has sampled ten distinct sites throughout 2008-2018 (March-May) and 2024 (February-May) for anadromous fish spawning. In 2008, citizen volunteers collected ichthyoplankton samples from five mainstem sites (MC1-MC5) and four tributary sites (MUT3-MUT5, MOWR1; Table 1-2; Figure 1-2); MC5 and MOWR1 were removed in subsequent years, as spawning was not detected for any anadromous species group. Between 2009-2015, four mainstem sites (MC1-MC4) and three tributary sites (MUT3-MUT5; Table 1-2; Figure 1-2) were sampled consistently. Volunteer interest led to sporadic sampling of an additional tributary site (MUTX, n = 4) in 2014-2015, however the site was discontinued in 2015 due to restricted access and limited indication of spawning. In 2016-2018, citizen volunteers continued sampling at four mainstem sites and the MUT3 tributary site; MUT4 and MUT5 were excluded, as spawning access was blocked by beaver dams. In 2024, DNR Fisheries Ecosystem and Assessment Division (FEAD) biologists reinstituted sampling at four mainstem sites (MC1-MC4), which are those most likely to be impacted by threats of increased development within the watershed. Table 1-2 summarizes the number of sites sampled, sampling dates, and total sample sizes for Mattawoman Creek during 2008-2018 and 2024, as well as other watersheds sampled in prior years. More information on specific sites sampled in other subestuaries in previous years, as well as watershed-specific site maps, can be found in Uphoff et al. (2023).

Field Sampling Methods - Following the same protocols as 2005-2021, sampling for anadromous fish eggs and larvae in 2024 occurred at stream sites identified as anadromous fish spawning sites during the 1970s and 1980s (O'Dell et al. 1975, 1980; O'Dell and Mowrer 1984). O'Dell et al. (1975, 1980) and O'Dell and Mowrer (1984) summarized spawning activity as the presence of an anadromous fish species (White Perch, Yellow Perch, or Herring) group's egg, larva, or adult at a site sampled with stream drift ichthyoplankton nets and wire traps. In 2024, FEAD biologists recreated the stream drift net methodology at Mattawoman Creek sites, typically located near road crossings.

These 2024 collections expanded upon data collected from 2005-2021 across Chesapeake Bay subestuaries exposed to varying levels of development. Citizen volunteers, trained and monitored by program biologists, conducted sampling in the Bush River (2005-2008, 2014), Piscataway Creek (2008-2009, 2012-2014), and Mattawoman Creek (2008-2018). DNR biologists from the Fishery Management Planning and Fish Passage Programs sampled Deer Creek (2012-2015), Choptank River (2016-2017), Tuckahoe Creek (2016-2017), Chester River (2019), and most Patuxent River sites (2021); DNR biologists from the Fish Health and Hatcheries, Anadromous Species Division also made middle Patuxent River collections by boat, at no charge to this grant. Patapsco River (2013-2017) collections made by U.S. Fish and Wildlife Service, at no charge to this grant, were also included in this data set.

Ichthyoplankton samples were collected in all subestuaries and years using stream drift nets constructed of 360-micron mesh. Each net was attached to a square frame with a 300 x 460 mm opening, connecting to a handle so that it can be held stationary in the stream while wading. The stream drift net configuration and techniques mimicked those used by O'Dell et al. (1975). A threaded collar at the end of the net connected to a mason jar that retained the sample. Nets were held in the stream for five minutes, with the opening facing upstream. After five minutes, the contents of the net were rinsed down into the jar by repeatedly dipping the net's lower portion into the stream and splashing water through the outside, taking care to avoid any additional water entering through the net opening to prevent sample contamination. The mason jar was then removed from the net, and sample labels (including site, date, time, and collector initials) were added inside the jar and to its lid. During 2024 sampling, we immediately fixed the sample with 10% buffered formalin. While each sample was being collected, we used a YSI 556 Multi Probe System (MPS) water quality meter, calibrated weekly, to measure water temperature (°C), specific conductance (μS/cm), dissolved oxygen (DO, mg/L), and salinity (ppt). It should be noted that in prior reports, we referred to specific conductance as "conductivity"; we would like to clarify that conductivity and specific conductance are differing metrics, and we have been measuring specific conductance throughout the entire survey timeseries. Data was recorded on standardized field data sheets and verified at the site during collections. Upon return to the lab, approximately 2-ml of rose bengal dye was added to each sample, staining ichthyoplankton pink to aid sorting. In some previous years, minor adjustments to these protocols have been made as needed, depending on the site and year sampled, as well as personnel conducting the sampling; see Uphoff et al. (2023) for details on specific modifications to sample collection, fixative protocol, water quality meters and meter calibration, and data sheet verification.

An exception to the above methodology, the sample processing protocol for collections from Mattawoman Creek in 2018 was adjusted due to staffing limitations. Citizen volunteers received training on field identification of Herring eggs and larvae prior to the start of sampling season and, if they were able to determine presence in the field, the sample was not retained. Samples where Herring presence could not be confirmed were preserved for laboratory examination

Laboratory Methods - Ichthyoplankton samples were sorted in the laboratory by division biologists. After rinsing with water to remove formalin, each sample was transferred to a white sorting pan and sorted systematically (i.e., from one end of the pan to another) under a 10x bench magnifier. All ichthyoplankton were removed and retained in small vials (separate vials for eggs and larvae) labeled with site, date, and time and filled with 20% ethanol. For each sample, this process was repeated a second time for quality assurance (QA). Any additional eggs and/or larvae found were removed and added to the corresponding vials for that sample. Eggs and larvae found during sorting were later identified under a microscope as either White Perch, Yellow Perch, target Herring species (Blueback Herring, Alewife, and Hickory Shad), unknown (eggs and/or larvae that were too damaged to identify) or other (indicating another fish species); the presence or absence of each was recorded. Because the target Herring species' eggs and larvae are very similar, identification to species-level can be challenging (Lippson and Moran 1974). Though American Shad eggs and larvae would be larger at the same stages of development than those identified as the target Herring, that species has yet to be detected in our surveys (Lippson and Moran 1974).

As mentioned above, due to staffing limitations, Mattawoman Creek sample processing protocol in 2018 differed from the protocol in other years. Samples that were retained because the presence of Herring eggs and/or larvae could not be verified in the field were sorted only for presence of Herring eggs and/or larvae in the laboratory. Once a Herring egg or larvae was encountered, processing of the sample was considered complete, regardless of the quantity sorted.

Data Analysis - Methods used to estimate development (C/ha), impervious surface coverage (%IS), and land use indicators (percent of watershed in agriculture, forest, wetlands, and developed land use) are explained in **General Spatial and Analytical Methods used in Project 1, Sections 1-3.** Development targets and limits, as well as general statistical methods (analytical strategy and equations) are also described in this section. Specific spatial and analytical methods for this section (**Project 1, Section 1**) are as follows: watershed area draining into the anadromous spawning areas (hereafter, watershed), land use indicators, %IS, and C/ha in those anadromous spawning areas were estimated. C/ha and %IS were estimated for each watershed in each year it was sampled, to provide a frame of reference for changes in development levels over the time-series.

Mattawoman Creek Analyses - We plotted conductance for each Mattawoman Creek sampling date and stream site (mainstem and tributaries) during 2008-2018 and 2024, then compared them to the minimum and maximum conductance reported for Mattawoman Creek during anadromous fish spawning season in 1991 by Hall et al. (1992). Change in conductance over time was also demonstrated by box plots of each mainstem station's conductance observations. We generated summary statistics for conductance measurements during 2024 sampling for comparison to annual conductance summaries for Mattawoman Creek from previous years. Only measurements from mainstem sites were summarized, as these are more likely than tributary sites to be impacted by development in Waldorf, the major urban influence on the watershed, and Bryan's Road, a growing residential-commercial area that abuts the Mattawoman Creek WCD (Figure 1-2). Furthermore, tributaries were excluded to better represent conditions in the largest portion of habitat. Conductance measurements from other watersheds have been similarly summarized in prior years; past conductance summaries for other watersheds, along with descriptions of included sites, are available in Uphoff et al. (2023). Annual median conductance for each Mattawoman mainstem station was also plotted.

For further historic comparison, we utilized a water quality database maintained by DNR's Tidewater Ecosystem Assessment (TEA) Division that provided conductance measurements for Mattawoman Creek from 1970-1989. These historical measurements were compared with those collected in 2008-2018 and 2024 to explore changes in conductance over time. Monitoring was irregular for many of the historical stations, and Table 1-3 summarizes site locations, months sampled, total measurements at a site, and the years sampled. We assigned river kilometers (RKM) to historical stations and those sampled in 2008-2018 and 2024 using a GIS ruler tool that measures a transect approximating the center of the creek from the mouth of the subestuary to each station location. Stations were categorized as tidal or non-tidal. Conductance measurements from eight non-tidal sites sampled during 1970-1989 were summarized as monthly medians. These sites bound Mattawoman Creek from its junction with the estuary to the city of Waldorf (Route 301 crossing). Historical monthly median conductance at each mainstem Mattawoman Creek non-tidal site and 2008-2018 and 2024 spawning season median conductance values were plotted together.

We used ANOVA to assess how conductance varied among Mattawoman Creek mainstem stations across all sampling years combined. To ensure the data fit test assumptions, we visually examined the Q-Q plot of conductance residuals for normality and performed Bartlett's test for homogeneity of variance (Bartlett 1937; Kozak and Piepho 2017). If data violated the homogeneity of variance assumption, Welch's ANOVA for unequal variances was utilized (Tomarken and Serlin 1986). If significant differences were detected, post-hoc Tukey HSD analysis was performed to determine which sites were significantly different than others (Abdi and Williams 2010).

For use in regression analyses, we standardized Mattawoman Creek conductance annual medians by an estimate of the background conductance expected for a Coastal Plain stream devoid of anthropogenic influence (109 µS/cm, Morgan et al. 2012). Morgan et al. (2012) provided two methods of estimating spring base flow background conductance for two different sets of Maryland ecoregions, totaling four potential background estimates. For our purposes, the option featuring Maryland Biological Stream Survey (MBSS) Coastal Plain and Piedmont regions and the 25th percentile background level for conductance was chosen; these regions had larger sample sizes than other options, and background conductance in the Coastal Plain fell much closer to the observed range estimated for Mattawoman Creek in 1991 (61-114 µS/cm), when development was relatively low (Hall et al. 1992). For details on the standardization of median conductance values from other watersheds in previous years, see Uphoff et al. (2023).

To determine which sites continued to support anadromous spawning, we compared the presence of eggs and/or larvae of White Perch, Yellow Perch, and Herring at each Mattawoman Creek station to past surveys, with the exception of 2018 when only presence of Herring eggs and/or larvae was determined. Historical site occupation was available for Mattawoman Creek mainstem stations sampled in 1971 and 1989-1991, by O'Dell et al. (1975) and Hall et al. (1992), respectively. Instead of stream drift nets, Hall et al. (1992) collected ichthyoplankton with 0.5 m diameter plankton nets (3:1 length to opening ratio and 363µ mesh set for 2-5 minutes, depending on flow) suspended in the stream channel between two posts. The authors provided a tabular summary of egg and larvae counts at the sample level; we converted these data to presence-absence for comparison. We used the criterion of detection of eggs and/or larvae at a site (O'Dell et al. 1975, 1980; O'Dell and Mowrer 1984) as evidence of spawning. Raw data from early 1970s and 1980s collections were not available to formulate other metrics. Sites where Herring spawning was detected (site occupation) during the current and historical studies were compared to changes in C/ha. We also calculated site occupation persistence for each target species group by dividing the number of years a species was detected at a site by the total number of years the site was sampled. A description of historical and recent site occupation data for other watersheds in other years can be found in Uphoff et al. (2023). Additionally, we plotted the date that the first Herring egg, first Herring larvae, last Herring egg, and last Herring larvae were observed in Mattawoman Creek mainstem samples each sampling year to visually examine how the timing of Herring spawning in this system has changed over the timeseries.

We estimated the proportion of samples where Herring eggs and/or larvae were present (P_{herr}; described below) for Mattawoman Creek mainstem stations (MC1-MC4) for 2024, allowing for comparisons to 1991 and 2008-2018. We also calculated site-specific P_{herr} for these Mattawoman Creek mainstem stations in each sampling year. Previously, P_{herr} has been estimated for all other watersheds in the years they were sampled, with values used in regression analyses, described below. Herring was the only species group with adequate sample sizes for annual proportion of sample estimates with reasonable precision. A description of the sites used to estimate P_{herr} in each watershed each year is available in Uphoff et al. (2023).

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

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

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

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

The 90% confidence intervals were constructed as:

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

To explore whether increases in Herring spawning stock biomass over time existed that may have influenced Mattawoman Creek P_{herr}, we performed a t-test to compare Mattawoman P_{herr} estimates from two time periods: (1) 2005-2011, representing lower spawning stock before strict fisheries regulations were in place, and (2) 2012-2024, after the implementation of conservation measures such as river closures and bycatch reductions (ASMFC 2019; MAFMC 2019). Spawning stock was categorized in this manner because independent indicators of spawning stock size are not available for Mattawoman Creek, or any sampled watersheds. Prior to testing, we assessed data for violations of t-test assumptions by visually examining Q-Q plots of conductance residuals for normality and performing a Folded F test for equality of variances (Park 2009; Kozak and Piepho 2017).

Multiple Chesapeake Watersheds Analyses - With new estimates of Herring spawning intensity (P_{herr}), development (C/ha), and standardized median conductance from Mattawoman Creek in 2024, we updated two regression approaches that describe relationships among these variables: (1) simple linear regression, and (2) multiple regression with two independent variables, a categorical variable to indicate two levels of spawning stock (low and high) and C/ha or standardized conductance.

Background on the development of these relationships can be found in Uphoff et al. (2023) and earlier reports. Updates included data from all watersheds sampled from 2005-2024 and 1991 data from Mattawoman Creek (Hall et al. 1992). Thirty-eight paired estimates of C/ha and P_{herr} were available, while 37 estimates were available for standardized median conductance (median conductance estimates were not available for Mattawoman Creek in 1991).

The updated relationship of standardized conductance and C/ha was described by a simple linear regression (Uphoff et al. 2023). We also updated multiple regression analyses that described P_{herr} as a product of C/ha or standardized conductance and spawning stock class (Uphoff et al. 2023). We assumed equal slopes for two stock size categories, but different intercepts (Neter and Wasserman 1974; Rose et al. 1986; Freund and Littell 2006). This common slope would describe the relationship of C/ha or standardized conductance to P_{herr}, while the intercept would indicate the effect of high or low spawning stock size. This analysis was conducted for the 2005-2021 and 2024 time-series, excluding data from 1991 (Hall et al. 1992). These analyses were initially done in Excel and run again in SAS (Proc Reg) to confirm estimates. Spawning stock size was modeled as an independent variable in multiple regression analyses, with 0 indicating lower spawning stock prior to the full implementation of river closures and bycatch reductions (2005-2011) and 1 indicating higher spawning stock following these measures (2012-2021, 2024). Spawning stock was categorized in this manner because independent indicators of spawning stock size were not available for any sampled watersheds. P_{herr} may serve as an indicator of spawning stock size for each watershed, however P_{herr} is used as the continuous dependent variable in these analyses. The use of categorized variables and linear regression as an alternative to Box-Jenkins models and time-series regression was presented by Rose et al. (1986). In addition to standard regression output, we examined the type II sums of squared partial correlation coefficients to assess the amounts of variation in P_{herr} explained by each independent term in the regression models while holding the other constant (Ott 1977; Sokal and Rohlf 1981; Afifi and Clark 1984). This multiple regression was tested for multiple years and provided a good fit and serial patterning of residuals was minor compared to a linear model without a time category (Uphoff et al. 2023).

Results

Mattawoman Creek - Development in Mattawoman Creek's 24,329 ha watershed increased from approximately 0.05 C/ha in 1950 to 0.48 C/ha in 1991, when sampling was conducted by Hall et al. (1992; Table 1-1; Figure 1-3); during this same time frame, estimates of impervious surface (% IS) coverage in the watershed increased by more than five percent, from 1.5% in 1950 to 6.7% in 1991. By the time our sampling of Mattawoman Creek began in 2008, the watershed had been further developed to 0.87 C/ha and 10.2% IS. Development continued to grow throughout our sampling time-series to 0.97 C/ha and 11.0% IS in 2018. Our most recent estimates suggest that in 2024, when we reinstituted sampling here, development had reached at least 1.04 C/ha and 11.4% IS in the Mattawoman Creek watershed (Table 1-1; Figure 1-3). Figure 1-3 depicts the change in C/ha over time in all sampled watersheds since 1950.

Watershed sizes, levels of development, and primary land use types for each subestuary sampled are available in Table 1-1, and more detailed information on subestuaries other than Mattawoman Creek can be found in Uphoff et al. (2023).

Conductance measurements from Mattawoman Creek in 2024 ranged from 99-164 µS/cm, the lowest maximum value measured since 2013, with measurements from five samples (all occurring on April 3rd and 10th) falling below Coastal Plain stream background levels (109 µS/cm, Morgan et al. 2012; Table 1-4; Figure 1-4). Mainstem conductance was highest when sampling began in February (> 130 µS/cm), then declined throughout March to fall below the 1991 maximum (114 µS/cm; Hall et al. 1992) for two dates in early April (April 3rd and 10th; Figure 1-4). Conductance returned to comparatively elevated levels shortly thereafter and remained so throughout May, following a similar pattern to 2010-2013 and 2016 (Figure 1-4).

As in past F-63 surveys, 2024 conductance measurements were generally highest at the most upstream site closest to Waldorf (MC4), declining as collections moved downstream towards the tidal border (Figure 1-4). This was also reflected by 2024 median conductance values for stations MC2 to MC4, above the confluence of Mattawoman Creek's stream and estuary (MC1), which were elevated beyond nearly all historic monthly medians and increase further upstream, with proximity to Waldorf (Figure 1-5). Median conductance at MC1 in 2024 fell within the upper half of the range observed during 1970-1989 (Figure 1-5). Consistent with previous estimates from 2008-2018, non-tidal median conductance estimates for all mainstem sites did not meet or fall below the Coastal Plain stream background criterion in 2024 (109 µS/cm, Morgan et al. 2012; Figure 1-5).

Annual standardized median conductance estimates in Mattawoman Creek (all mainstem sites combined) ranged from 1.14- to 1.94-times background levels, exhibiting the highest inter-annual variation of any watershed sampled during 2005-2021 and 2024; 2024 is the third lowest estimate of standardized median conductance in Mattawoman Creek throughout the time-series (Table 1-4; see Uphoff et al. 2023 for summarized conductance estimates from other watersheds and years). Box plots of raw conductance values from each mainstem site in each year suggest spring Mattawoman conductance observations follow one of two general patterns: either lower, fairly stable conductance observations or higher, more variable conductance observations (Figure 1-6).

An ANOVA testing for differences in conductance among stations indicated that conductance was greater at MC4 (near Waldorf) than MC1 (just above the estuarine confluence); we would have expected the opposite in the absence of development. A one-way Welch's ANOVA revealed that, when all sampling years were grouped, conductance varied significantly among mainstem Mattawoman Creek stations ($F[3, 292.6] = 6.67$, $P = 0.0002$; Table 1-5). Post-hoc Tukey HSD analysis showed this was driven by a significant difference between conductance observations at MC1 (mean [M] = 144, SE = 3.941), the most downstream site, and MC4 (M = 173.9, SE = 5.545), the most upstream site (Figure 1-7).

In 2024, Herring spawning was detected at all Mattawoman Creek mainstem stations (Table 1-6). Spawning has been consistently detected at these stations during historic sampling (1971, 1991) and during the 2010-2018 portion of our sampling time-series; spawning was detected at only two of four mainstem sites in 2008-2009 (Table 1-6). Tributary sites were not sampled in 2024, however spawning was detected consistently at MUT3 in 2011-2016, and again in 2018, while indication of spawning at MUT4, MUT5, and MUTX was intermittent throughout the time-series (Table 1-6). Overall, Herring occupation persistence at each mainstem station varied from 0.86-1.00 across all sampling years (Table 1-6). A summary of Herring spawning detection within other watersheds in other years is available in Uphoff et al. (2023). In 2024, the first Herring eggs were observed on the first day of sampling, February 28th, and remained present throughout the duration of the sampling period, until May 15th (Figure 1-8). Herring eggs have been observed on the first day of sampling in five other years: 2008, 2012, and 2016-2018. The first Herring larvae appeared 14 days after the first egg, on March 13th, while the last larvae were observed a week before the conclusion of sampling, on May 8th (Figure 1-8).

White Perch spawning was detected at MC1, MC2, and MC4 in 2024 (Table 1-6). During historic sampling (1971, 1989-1991), White Perch spawning was detected annually at MC1, intermittently at MC2, and only once at MC3 (Table 1-6). Our survey has observed a similar pattern, with White Perch spawning detected consistently at MC1 from 2008-2017, excluding 2009 and 2012. White Perch spawning was not detected at any site in 2009 and 2012 (Table 1-6). We detected White Perch spawning at MC2 during 2013-2014 and 2016-2017, at MC3 in 2016, and at MC4 in 2015 (Table 1-6). White Perch spawning in tributaries has been detected in three years: at MUT3 in 2016, at MUT5 in 2013, and at MUTX in 2014 (Table 1-6). As mentioned previously, the presence of White Perch spawning in Mattawoman Creek was not assessed in 2018 due to time and staffing limitations. Overall, White Perch occupation persistence varied from 0.15-0.87 among mainstem stations across all sampling years; the highest detection persistence was at MC1, with mid-range persistence at MC2 (0.40) and persistence dropping off to 0.15 at MC3-MC4 (Table 1-6). A summary of White Perch spawning detection within other watersheds in other years is available in Uphoff et al. (2023).

In 2024, we detected Yellow Perch spawning at only one site, MC1 (Table 1-6). During historic sampling (1971, 1989-1991), MC1 was the only site known to support Yellow Perch spawning (Table 1-6). This was reflected in our own surveys, and Yellow Perch spawning was detected annually at MC1 from 2008-2017, with the exceptions of 2009 and 2012 when spawning was not detected at any site (Table 1-6). For a brief period of our survey, we did detect Yellow Perch spawning consistently at MC2, during 2013-2016; possible observations of Yellow Perch larvae were also recorded at MC3 and MC4 in 2016, however these are unverified due to later collection dates and an apparent isolated occurrence, respectively (Table 1-6). Yellow Perch spawning was also detected at two tributary sites, MUT5 and MUTX, in 2014 (Table 1-6). As stated above, the presence of Yellow Perch spawning in fluvial Mattawoman Creek was not assessed in 2018 due to time and staffing limitations. Overall, Yellow Perch occupation persistence varied from 0.08-0.87 among mainstem stations across all sampling years; the highest detection persistence was at MC1, with lower persistence at MC2 (0.27) and persistence dropping off to 0.08 at MC3-MC4 (Table 1-6). A summary of Yellow Perch spawning detection within other watersheds in other years is available in Uphoff et al. (2023).
In general, estimates of P_{herr} have increased in Mattawoman Creek throughout our survey time-series, both for the system as a whole and at each mainstem sampling station individually (Figure 1-9, 1-10). A two-sample t-test indicated that P_{herr} was significantly higher during the 2012-2024 time period (M = 0.658, SD = 0.120) than the 2005-2011 time period, before regulations took effect (M = 0.314, SD = 0.272; t[10] = -3.13, P = 0.012; Table 1-7; Figure 1-9. It should be noted that this observed increase in Mattawoman P_{herr}, occurred concurrently with a slight increase in development within the watershed between 2008 (0.87 C/ha, 0.9 %IS) and 2024 (1.04 C/ha, 11.5 %IS). Despite the increase of C/ha within the watershed, P_{herr} in Mattawoman Creek continues to approach levels exhibited in streams in rural watersheds; Mattawoman P_{herr} estimates reached 0.78 in 2018 and 0.77 in 2024 (Table 1-1; Figure 1-9).

Multiple Chesapeake Watersheds - Across all watersheds, standardized conductance increased with development, while P_{herr} declined with both development and standardized conductance. Watershed-specific trends in P_{herr} differed by system (Figure 1-11). Both simple and multiple regression analyses, updated to include 2024 estimates, continued to indicate significant and logical relationships among P_{herr}, C/ha, and standardized median conductance. The relationship of C/ha with standardized median conductance was linear, moderate, and positive ($r^2 = 0.34$, $P = 0.00016$, $N = 37$; Table 1-8; Figure 1-12).

The C/ha and spawning stock time category multiple regression explained 70% of variation in P_{herr} ($P < 0.0001$; Table 1-9). The intercept (mean = 0.49, SE = 0.08) and both coefficients (C/ha slope = -0.26, SE = 0.05; spawning stock coefficient = 0.33, SE = 0.06) were estimated with reasonable precision (CV < 30%). Predicted P_{herr} declined by 47% over the range of observed C/ha (0.07-1.52; Figure 1-13). Predicted P_{herr} increased by 66% between the two spawning stock categories (Table 1-9). Only the high spawning stock category contained estimates from the three land use types; the low stock size category consisted of estimates above the C/ha threshold (Mattawoman Creek, predominately forested, and Bush River tributaries, predominately urban; Figure 1-13).

The standardized conductance and spawning stock time category multiple regression explained 66% of variation in P_{herr} ($P < 0.0001$; Table 1-10). The intercept (mean = 0.65, SE = 0.12) and both coefficients (standardized conductance slope = -0.29, SE = 0.07; spawning stock coefficient = 0.42, SE = 0.06) were estimated with reasonable precision (CV < 32%). Predicted P_{herr} declined by 49% over the range of observed standardized conductance (1.14-2.42; Figure 1-13). Predicted P_{herr} increased by 66% between the two spawning stock categories (Table 1-10). Only the high spawning stock category contained estimates from all three land use types (Figure 1-13). Standardized median conductance observations exceeding 1.75 were exclusively from watersheds categorized as urban, with the exception of Mattawoman Creek in 2009 (a year of a late snowfall and high application of road salt). Higher standardized median conductance (up to about 1.70) in agricultural and forested watersheds did not appear to be associated with distinctly lower P_{herr}; declines appeared concurrent with higher conductance associated with urban development (Figure 1-13).

An increasing trend in residuals, evident in the linear regressions of P_{herr} or standardized median conductance with C/ha, was reduced to the point of near elimination for residuals of the multiple regressions that added a spawning stock size time category (Figure 1-14). Linear regressions of residuals from the multiple regressions and year indicated a slight increasing trend over time was possible for standardized conductance ($r^2 = 0.12$, $P = 0.03$; Figure 1-14) but, less likely for C/ha ($r^2 = 0.08$, $P = 0.09$; Figure 1-14). Cook's distance statistics identified 2011 as an outlier in both multiple regressions; the 2011 estimate of P_{herr} was more consistent with estimates from the high spawning stock (2012-2018) period than the low.

Discussion

Proportion of samples with Herring eggs and/or larvae (P_{herr}) provided a reasonably precise estimate of habitat occupation based on encounter rate. Regression analyses that ostensibly accounted for shifting spawner abundance between 2005-2011 and 2012-2024, indicated significant and logical relationships among P_{herr} and C/ha, consistent with the hypothesis that urbanization is detrimental to stream spawning. Predicted P_{herr} declined by nearly 50% over the observed range of C/ha (0.07-1.52). Estimates of P_{herr} were consistently high in rural watersheds dominated by agriculture; forest cover is typically preserved along streams in these watersheds. Limburg and Schmidt (1990) found a highly nonlinear relationship between densities of anadromous fish (mostly Alewife) eggs and larvae with urbanization in Hudson River tributaries, reflecting a strong negative threshold, even at low levels of development. In Mattawoman Creek, despite increases in C/ha throughout our time-series, P_{herr} estimates remain high and similar to those from less-developed, rural watersheds. Though development levels are increasing in the Mattawoman Creek watershed, it remains the least developed of surveyed watersheds not dominated by agriculture.

Uphoff et al. (2017, 2024) reported that there were strong negative correlations between agricultural watershed percentages with C/ha, that forest cover and agriculture were strongly and negatively correlated, and that forest cover was poorly correlated with C/ha. The Maryland Department of Planning (MDP) forest cover estimates mix forest cover in residential areas (trees over lawns) with true forest cover, clouding interpretation of forest influence. High resolution Chesapeake Conservancy estimates of land cover (2013-2014, 2017-2018, and 2021-2022) provide estimates of true forest cover; however, that resolution cannot be used for long-term trends based on earlier MDP estimates. Uphoff et al. (2017) determined that subsequent analyses with P_{herr} beyond comparisons with C/ha were likely to be confounded by the close negative correlations, therefore we did not pursue statistical analyses with land use metrics other than C/ha. Our preference for using C/ha in current analyses was two-fold: we had a history of using C/ha in prior analyses, and C/ha provided a continuous, rather than episodic, time-series. However, we did note when these other land uses were predominant for particular P_{herr} outcomes.

Mattawoman Creek, our study's only predominately forested watershed, is above the development threshold, yet estimates of P_{herr} have been near or above 0.60 there since 2013. Forest was identified as an important macroscale habitat feature for American Shad in the Mattaponi and Pamunkey Rivers, Virginia (Bilkovic et al. 2002). American Shad eggs were primarily collected in reaches that were more than 60% forested and less than 20% emergent marsh, indicative of upstream and midriver reaches. These rivers lacked intense urbanization (Bilkovic et al. 2002).

Mattawoman Creek's watershed has a diverse mix of upland and riparian forests (Interagency Mattawoman Ecosystem Management Task Force 2012). The two types of forests are characterized based on topographic influences, from the flat coastal plains to the wide stream valleys which comprise the watershed. Upland forests are found on the flat coastal plains while riparian forests, as well as a number of forested wetlands, are found in the stream valleys and throughout the non-tidal wetlands which surround Mattawoman Creek. The wide valleys function as a floodplain, allowing for nutrient cycling and filtering of many types of pollutants – though, unfortunately, not some of the more harmful substances (e.g., road salt). The floodplain also serves as a large habitat corridor, as the areas immediately surrounding the creek possess extensive forest cover (Interagency Mattawoman Ecosystem Management Task Force 2012).

Reforestation was mentioned as part of a portfolio of strategies that could be used to counter the negative effects of climate warming on Herring (Kritzer et al. 2022). Herring have been identified as highly vulnerable to climate change, due to their high biological sensitivity to climate stressors and very high exposure to a changing climate in the northeastern USA (Hare et al. 2021). High sensitivity for Herring was related to the complexity of their reproduction, their relatively narrow spawning season, and their exposure to a multitude of other stressors (Hare et al. 2021). Analysis of adult Herring presence-absence in spawning tributaries of Ablemarle Sound, North Carolina during 1973-2016 indicated that the spawning season started earlier and had become truncated, providing evidence of the impact of climate change on the group of fishes (Lombardo et al. 2020).

Reforestation projects are a commonly accepted means to reduce pollutant runoff, support healthy soils, sequester carbon dioxide, and, in certain jurisdictions within Maryland, potentially earn the stormwater credits necessary for permit compliance (USEPA 2023). This approach is a core strategy in the State of Maryland's Watershed Implementation Plan to reduce pollution in Chesapeake Bay (USEPA 2023).

Mattawoman Creek's watershed has lost forest cover, with estimates based on MDP methodology dropping from 70.5% in 1973 to 52.8% in 2018. Forest loss may have slowed with the adoption of the 2016 Charles County Comprehensive Growth Plan; loss averaged -0.45% per year during 1973-2010 and -0.21% per year during 2013-2018. Chesapeake Conservancy high resolution estimates of true tree cover indicate that the WCD accounts for the majority and an increasing percentage of tree cover in Mattawoman Creek's watershed: 62.9% in 2013-2014, 63.7% in 2017-2018, and 64.0% in 2021-2022. Increased forest percentage represented by the WCD likely reflects forest loss outside the WCD with development; the amount of forest in Mattawoman Creek's watershed (including the WCD) has fallen from 59.1% to 57.8% during the same period. Meanwhile, forest cover in the WCD has remained close to 75%.

Conductance measurements in Mattawoman Creek in 2024 were most comparable to 2008, 2011-2013, and 2017-2018 observations, likely because these years all experienced lower snowfall (i.e., less road salt use; NOAA 2025). When road salt enters a stream via runoff, it can directly impact stream conductance by increasing the concentration of chloride ions (Morgan et al. 2012; Hintz and Relyea 2019). This is reflected in our Mattawoman Creek dataset, where we observed elevated conductance levels in years with higher seasonal snowfall, such as 2014-2015. Elevated conductance was also especially apparent in 2009, when levels spiked in early March following heavy road salt application in response to significant snowfall just prior to the start of the survey (Uphoff et al. 2010); measurements during 2009 steadily declined for nearly a month before leveling off slightly above the 1991 maximum.

Elevated conductance, primarily due to chloride from road salt (although it includes most inorganic acids and bases; APHA 1979), has been found to be positively related with urbanization in other studies and, consequently, has emerged as an indicator of watershed development (Wang and Yin 1997; Paul and Meyer 2001;Wenner et al. 2003; Kaushal et al. 2005; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012; Moore et al. 2017; Bird et al. 2018; Kaushal et al. 2018; Baker et al. 2019). Our regression analysis supported this, showing that conductance is positively related with C/ha across all watersheds. A relationship between development and conductance within the Mattawoman Creek watershed was also suggested by our ANOVA examining differences in conductance among mainstem sites. Post hoc analysis demonstrated that, across all sampling years, mean conductance at the upstream-most station, MC4, was significantly higher than at the downstream-most station, MC1. Site MC4 is closest to Waldorf and Bryan's Road, and updated census data designates much of the land directly north and east of the site as "urban" (MDP 2020).

In 2015, two forest-dominated tributary sites in Mattawoman Creek subwatersheds within the WCD (MUT3 and MUT4) had much lower conductance than sites in developed areas with higher road densities, suggesting that forest watershed background levels were much lower (Uphoff et al. 2016). Citizen scientists evaluated conductance at nine sites along the longitudinal axis of Mattawoman Creek from May 2015 through 2016 (Uphoff et al. 2017). Samples were analyzed for conductance: anions, including chloride, bromide, nitrate and sulfate; cations, including sodium, potassium, magnesium, and calcium; total alkalinity; and closed pH. Baseflow conditions had higher conductance than a higher flow event, suggesting high flows diluted concentrations of ions and lowered specific conductance. The observed higher conductance at baseflow would have been driven by groundwater infiltrated by salt. Mean conductance slightly increased with upstream distance, until a reach upstream of Waldorf and adjacent to state-managed land at Cedarville State Forest. Sodium and chloride were the dominant ions in all samples collected, except for those from MUT4, where calcium and bicarbonate were prevalent (Figure 3-30, Uphoff et al. 2017). Sodium and chloride exhibited similar responses to road density as conductance (i.e., increasing rapidly as road density increased; Uphoff et al. 2017).

Regression analyses illustrated a negative effect of elevated conductance on Herring spawning. Different mixtures of salt ions (e.g., sodium, bicarbonate, magnesium, sulfate) produce differential toxicity to aquatic life (Kaushal et al. 2018). Use of salt as a deicer may lead to both "shock loads" of salt that may be acutely toxic to freshwater biota, as well as elevated baselines (i.e., increased average concentrations) of chloride that have been associated with decreased fish and benthic diversity (Kaushal et al. 2005; Wheeler et al. 2005; Morgan et al. 2007, 2012). Furthermore, commonly used anti-clumping agents for road salt (i.e., ferro- and ferricyanide) are also of concern; while not thought to be directly toxic, when exposed to ultraviolet light, these agents can break down into toxic cyanide. Although the degree of breakdown into cyanide in nature is unclear, these compounds have been implicated in fish kills (Burdick and Lipschuetz 1950; Pablo et al. 1996; Transportation Research Board 2007). Heavy metals and phosphorous may also be associated with road salt (Transportation Research Board 2007). Salt concentration, salt type, and, to a lesser extent, type of stormwater management practices (BMPs) determine the composition and concentrations of chemical cocktails released from stormwater BMPs (Galella et al. 2023). Salt ion amounts had significant effects on mobilization of a wide variety of contaminants, including nutrients, other salt ions, and metals. Though NaCl is often the most cost-effective deicer available, the use of brines (e.g., MgCl2 and CaCl2) should be considered as an alternative, especially in environmentally fragile areas or where fisheries are affected by Cu pollution (Galella et al. 2023).

At least two hypotheses can be formed to relate decreased anadromous fish spawning to conductance and road salt use. First, eggs and larvae may die in response to sudden changes in salinity and potentially toxic concentrations of associated contaminants and additives; a rapid increase might result in osmotic stress and lower survival, as salinity fluctuations result in osmotic cost for fish eggs and larvae (Research Council of Norway 2009). Second, changing stream chemistry may disorient spawning adults and disrupt upstream migration. Levels of salinity associated with our conductance measurements are very low (maximum 0.2 ppt), and anadromous fish spawn successfully in brackish water (Klauda et al. 1991; Piavis et al. 1991; Setzler-Hamilton 1991). Elevated stream conductance may prevent anadromous fish from recognizing and ascending streams. Alewife and Blueback Herring are thought to return to natal rivers to spawn, while Yellow and White Perch populations are generally tributary-specific (Setzler-Hamilton 1991; Yellow Perch Workgroup 2002; ASMFC 2009a, 2009b). Though the physiological details of spawning migrations are not well described for our target species, the homing migrations in anadromous American Shad and salmon species have been connected to chemical composition, smell, and pH of spawning streams (Royce-Malmgren and Watson 1987; Dittman and Quinn 1996; Carruth et al. 2002; Leggett 2004).

Road salt pollution, along with anthropogenically accelerated weathering, have altered the natural concentrations of major ions in freshwater streams across North America, increasing salinization and alkalinization (i.e., freshwater salinization syndrome, Kaushal et al. 2018, 2025). Conductance is related to total dissolved solids in water, a reflection of chemical composition (Cole 1975). Sodium chloride is the dominant salt pollutant responsible for freshwater salinization syndrome but increases in other mixtures of salt ions (e.g., bicarbonate, magnesium, sulfate) contribute as well (Kaushal et al. 2018). The resulting changes in conductance, chemical composition, and pH have impacted the water quality of most streams in the eastern United States since the early and middle 20th century. Densities of urban and agricultural land within a watershed can be strong predictors of base cations and pH in streams and rivers. While road salt is an important source of salinization in more urban areas with colder climates, agriculture can also contribute substantial base cations and bicarbonate via practices such as fertilizer application, liming, and potash (Kaushal et al. 2018).

Higher standardized conductance (up to about 1.7-times background levels) in agricultural and forested watersheds (i.e., Mattawoman Creek) did not appear to be associated with distinctly lower P_{herr}. However, declines in P_{herr} appeared with higher conductance in developing watersheds, suggesting that other urban stressors accompanied increasing conductance.

Including data from watersheds in both the Coastal Plain and Piedmont physiographic provinces in our regression analyses had the potential to increase scatter of points; however, standardizing median conductance to background conductance moderated province effects in analyses including that variable. Differential changes in physical stream habitat and flow with urbanization, due to differences in geographic provinces, could also have influenced fits of regressions. Estimates of C/ha may index these physical changes, as well as water chemistry changes, while standardized conductance would only represent changes in water chemistry. Squared type II partial correlation coefficients for regressions of C/ha with P_{herr} were higher (0.41) than for standardized conductance (0.34), possibly reflecting the wider coverage of stressors by C/ha.

Liess et al. (2016) developed a stress addition model for meta-analysis of toxicants that combined additional stressors of aquatic vertebrates and invertebrates, finding that the presence of multiple environmental stressors could amplify the effects of toxicants 100-fold. This general concept may explain the difference in regression fit of P_{herr} with C/ha and median standardized conductance, with conductance accounting for water quality and C/ha accounting for multiple stressors. This concept may also warn against expectations of overcoming Herring spawning stream habitat deterioration due to development through stringent management of directed fisheries and bycatch. An underlying negative relationship of P_{herr} with C/ha was present across all watersheds but only described how the spatial and temporal distribution of earliest life stages of Herring may be impacted. Increasingly frequent poor juvenile indices of Blueback Herring and Alewife in the urbanizing Patuxent River after the late 1990s did not indicate that increased spawning stock overcame deterioration of habitat (Uphoff et al. 2018).

In Uphoff et al. (2023), a simple plot and linear regression of C/ha and P_{herr} suggested that spawning across all watersheds both declined and became more variable as development increased. Increased variability, however, was likely an artifact of increasing spawning stock size with time. A multiple regression modelling P_{herr} as a product of both C/ha and a time category term, that we assume accounts for changing spawner abundance, considerably reduced the variability around the predicted slopes. Within the Mattawoman Creek watershed, both P_{herr}

and C/ha have increased during our time-series. Our t-test suggested the former could reflect increases in spawning stock size. Maryland closed its Herring fisheries in 2011, and most other in-river fisheries along the Atlantic Coast were closed by 2012 (AFMFC 2019; Hare et al. 2021; Kritzer et al. 2022). Caps on Herring bycatch in Atlantic Herring and Atlantic Mackerel fisheries were also implemented in 2014 (MAFMC 2019; Hare et al. 2021; Kritzer et al. 2022) and estimates of P_{herr} increased concurrently with these reductions. The 2017 ASMFC River Herring stock assessment update indicated that, at the time, 16 stocks had increased in abundance, two decreased in abundance, eight experienced stable abundance, and 10 exhibited no discernable trends in abundance over the final 10 years of the times series (2006-2015, ASMFC 2019). However, long-term monitoring of adult Blueback Herring and Alewife during spawning runs in the Nanticoke River have not indicated an increase in recent years, though Herring may have increased in the Head-of-Bay region (Bourdon and Jarzynski 2020; Bourdon 2022). The most current River Herring stock assessment update, published in 2024, stated that there is no discernible coast-wide trend in Alewife or Blueback Herring stocks in recent years (ASMFC 2024). However, a higher proportion of rivers in the Mid-Atlantic stock-region had a low probability of stock mortality surpassing a 40% mortality threshold, for both species (ASMFC 2024). It is also possible that improved survival to maturity in response to changes in undescribed sources of at-sea mortality unrelated to fishing (e.g., predation and feeding success) could have contributed to increased spawning stock or supplied an alternative hypothesis to harvest reductions for the increase.

Successful implementation of fisheries regulations may have also contributed to increased habitat use by spawning Herring farther upstream in Mattawoman Creek. P_{herr} increased over the time series at all Mattawoman mainstem sites, most notably MC3. It is possible that closing the fishery subsequently resulted in more fish available to move upstream. Positive relationships have been documented between spatial dispersion of sites occupied by a species and abundance for many taxa, including fishes, but they are not universal (Gaston et al. 2000; Miranda 2023). Furthermore, closing the fishery may have facilitated the progression of fish farther upstream by reducing fishing pressure at downstream sites; successful spawning of these fish would manifest as an increase in P_{herr} estimates at upstream sites, preceding any expected increase in spawning stock biomass.

Additionally, the establishment of the Mattawoman Creek WCD may have further amplified the positive effects of stringent fisheries management. We observed a dramatic increase in P_{herr} at MC3 over our time series, however the adjacent down- and upstream stations, MC2 and MC4, exhibited only moderate increases. Anthropogenic influence may be stronger at these stations due to their proximity to a regional airport and more densely developed land, respectively. In more recent years, development has encroached onto land originally designated as WCD and, in its current state, the protection provided by the WCD may not be enough to offset many negative impacts of nearby development.

Urbanization and physiographic province both affect discharge and sediment supply of streams (Paul and Meyer 2001; Cleaves 2003). These, in turn, can affect substrate composition, and, consequently, the location, extent, and success of spawning. Processes such as flooding, riverbank erosion, and landslides vary by geographic province and influence physical characteristics of anadromous fish spawning streams (Cleaves 2003). The Coastal Plain, with natural terrain characterized by wetlands and broad plains of low relief, is comprised of unconsolidated layers of sand, silt, and clay (Cleaves 2003); streams in this region are usually slow flowing, with sand or gravel bottoms (Boward et al. 1999). Contrastingly, the Piedmont region is an area of higher gradient change, with more diverse and larger substrates than the Coastal Plain (Harris and Hightower 2011). The region is underlain by metamorphic rocks and characterized by narrow valleys and steep slopes, with areas of higher elevation between streams within the same drainage. Reflecting this topography, most Piedmont streams are of moderate slope with rock or bedrock bottoms (Boward et al. 1999). The diversity of the Piedmont landscape, when compared to the Coastal Plain, may offer a greater variety of Herring spawning habitats.

Herring and Shad utilize a variety of stream types for spawning, depending on the species. Alewife spawn in sluggish flows, while Blueback Herring spawn in sluggish to swift flows (Pardue 1983). American Shad select spawning habitat based on macrohabitat features and spawn in moderate to swift flows (Hightower and Sparks 2003; Harris and Hightower 2011). Spawning substrates for Herring include gravel, sand, and detritus, all of which can be impacted by development (Pardue 1983). Detriments of urbanization on lithophilic spawners include loss of suitable substrate, increased embeddedness, lack of bed stability, and siltation of interstitial spaces (Kemp 2014). Broadcast spawning species, such as Herring, could be severely affected, as they do not clean substrate during spawning or provide protection to eggs and larvae in nests (Kemp 2014). Urbanization also affects the quality and quantity of organic matter, another source of spawning substrate (i.e., detritus) in streams that feed into subestuaries (Pardue 1983; Paul and Meyer 2001). In more rural areas, organic matter may be positively impacted by nutrients but, it can also be negatively impacted by fine sediment from agriculture (Piggot et al. 2015).

An unavoidable assumption of our regression analyses of P_{herr}, C/ha, and standardized conductance is that watersheds at different levels of development acted as a substitute for time-series; extended time-series of watershed-specific P_{herr} were not available (Mattawoman Creek is our longest, with 13 years of data). However, application of presence-absence data in management needs to consider whether absence reflects a disappearance from an area or whether habitat sampled is not truly suitable for the species in question (MacKenzie 2005). Our site occupation comparisons assumed that spawning sites detected in the 1970s and 1980s indicated the extent of spawning habitat. O'Dell et al. (1975, 1980) and O'Dell and Mowrer (1984) summarized spawning activity as the presence of any species group's egg, larva, or adult (latter from wire fish trap sampling) for all samples at a site, and we used this criterion (spawning detected at a site or not) for a set of comparisons. Raw data for the 1970s and early 1980s were not available to formulate other metrics. This site-specific presence-absence approach did not detect permanent site occupation changes or an absence of change. Only a small number of sites could be sampled (limited by road crossings), and the positive statistical effect of repeated visits was lost by summarizing all samples into a single record of occurrence in a sampling season (Strayer 1999). A single year's record was available for each of the watersheds in the 1970s and early 1980s, and we were left assuming this distribution applied over multiple years of low development.

Proportion of positive samples with Herring (P_{herr}) incorporated spatial and temporal presence-absence and provided an economical and precise alternative to the O'Dell et al. (1975, 1980) and O'Dell and Mowrer (1984) estimates of habitat occupation based on spatial presence alone. Encounter rate is readily related to the probability of detecting a population (Strayer 1999). Proportions of positive or zero catch indices were found to be robust indicators of abundance for Yellowtail Snapper *Ocyurus chrysurus* (Bannerot and Austin 1983), age-0 White Sturgeon *Acipenser transmontanus* (Counihan et al. 1999; Ward et al. 2017), Pacific Sardine

Sardinops sagax eggs (Mangel and Smith 1990), Chesapeake Bay Striped Bass eggs (Uphoff 1997), and Longfin Inshore Squid *Loligo pealeii* fishery performance (Lange 1991).

Unfortunately, estimating reasonably precise proportions of stream samples with White or Yellow Perch eggs and/or larvae annually would not be logistically feasible without major changes in sampling priorities. The likelihood of detecting White and Yellow Perch at more upstream sites was far less than that of Herring. Though comparable among Herring, White, and Yellow Perch at MC1, occupation persistence dropped off substantially for both Perch species at sites farther upstream, especially MC3-MC4. In addition, there is some doubt in the accuracy of Yellow Perch observations at MC3 and MC4 in 2016, the only year the species was detected at these stations. Yellow Perch were recorded at MC3 on two dates in May, however, this is outside the usual spawning time period for the species (March-April, Lippson and Moran 1974). Alternatively, the larvae recorded as Yellow Perch may have actually been a darter species (e.g., *Etheostoma* spp.), which are similar in appearance to Yellow Perch at that life stage and known to spawn during that time of year (Lippson and Moran 1974). Yellow Perch were recorded on one date in late March at MC4; however, the isolated nature of this observation calls it into question. Because spawning occurred at fewer sites for these species, estimates for Yellow or White Perch stream spawning would require more frequent sampling to obtain precision similar to that attained by P_{herr}. Given staff and volunteer time limitations, this would not be possible within our current scope of operations.

Sampling of stream spawning during our survey (2005-2021, 2024) used only stream drift nets, while O'Dell et al. (1975, 1980), O'Dell and Mowrer (1984), and Hall et al. (1992) determined spawning activity with both ichthyoplankton nets and wire traps to capture adults. Tabular summaries of egg, larval, and adult catches in Hall et al. (1992) allowed for a comparison of how site use in Mattawoman Creek might have varied in 1991 with and without adult wire trap sampling. Sites estimated when eggs and/or larvae were present in one or more samples were identical to those when adults present in wire traps were included with the ichthyoplankton data (Hall et al. 1992). Similar results were obtained from the Bush River during 2006 at sites where ichthyoplankton drift nets and wire traps were used; adults were captured by traps at one site and eggs and/or larvae at nine sites with ichthyoplankton nets (Uphoff et al. 2007). Wire traps set in the Bush River during 2007 did not indicate different results than ichthyoplankton sampling for Herring and Yellow Perch, but White Perch adults were observed in two trap samples while eggs and/or larva were not observed in plankton drift nets (Uphoff et al. 2008). These comparisons of trap and ichthyoplankton sampling indicated it was unlikely that an absence of adult wire trap sampling would impact interpretation of spawning sites when multiple years of data were available. The different method used to collect ichthyoplankton in Mattawoman Creek during 1991 could have biased that estimate of P_{herr}, although presence-absence data tend to be robust to errors and biases in sampling (Green 1979; Uphoff 1997).

Absence of detectable stream spawning does not necessarily indicate an absence of spawning in the estuarine portion of these systems. Estuarine Yellow Perch presence-absence surveys in Mattawoman and Piscataway Creeks, Bush River, and Patuxent River indicated that lack of detectable stream spawning does not correspond to their complete elimination from these subestuaries. Yellow Perch larvae were present in lower reaches of these subestuaries, (see Uphoff et al. 2023 Project 1, Section 2). Yellow Perch do not appear to be dependent on non-tidal stream spawning, but their use may confer benefit to the population through expanded spawning habitat diversity. Additionally, Yellow Perch stream spawning benefits anglers, providing access to the fishery from shore and likely facilitating most recreational harvest (Yellow Perch Workgroup 2002).

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Table 1-1. Summary of subestuaries and watershed size, Department of Planning (MDP) land use designation and estimates of land use types, and level of development (C/ha) and percent impervious surface coverage (%IS) during years sampled. Bush (w/o APG) refers to the portion of the Bush River watershed not including Aberdeen Proving Grounds.

Subestuary	Sample Year	C / ha	% IS	% Agriculture	% Forest	% Wetlands	% Urban	Watershed Size (ha)	Primary Land Use
Bush (w/o APG)	2005	1.37	14.0	25.4	35.0	3.2	36.2	35,956	Urban
Bush (w/o APG)	2006	1.41	14.3	25.4	35.0	3.2	36.2		
Bush (w/o APG)	2007	1.43	14.5	18.0	29.9	3.2	47.8		
Bush (w/o APG)	2008	1.45	14.6	18.0	29.9	3.2	47.8		
Bush (w/o APG)	2014	1.52	15.1	18.0	29.9	3.2	47.8		
Chester	2019	0.13	2.8	65.6	24.8	1.0	8.3	77,357	Agriculture
Choptank	2016	0.18	3.4	54.7	28.0	1.5	15.8	38,263	Agriculture
Choptank	2017	0.18	3.4	54.7	28.0	1.5	15.8		
Deer	2012	0.24	4.2	44.6	28.4	0.0	26.8	37,612	Agriculture
Deer	2013	0.24	4.2	44.6	28.4	0.0	26.8		
Deer	2014	0.24	4.2	44.6	28.4	0.0	26.8		
Deer	2015	0.24	4.2	44.6	28.4	0.0	26.8		
Mattawoman	1991	0.48	6.7	13.8	62.6	0.9	22.5	24,329	Forest
Mattawoman	2008	0.87	10.2	9.3	53.9	1.1	34.2		
Mattawoman	2009	0.88	10.3	9.3	53.9	1.1	34.2		
Mattawoman	2010	0.90	10.4	9.3	53.9	1.1	34.2		
Mattawoman	2011	0.91	10.6	9.3	53.9	1.1	34.2		
Mattawoman	2012	0.90	10.5	9.3	53.9	1.1	34.2		
Mattawoman	2013	0.92	10.6	9.3	53.9	1.1	34.2		
Mattawoman	2014	0.93	10.7	9.3	53.9	1.1	34.2		
Mattawoman	2015	0.94	10.8	9.3	53.9	1.1	34.2		
Mattawoman	2016	0.96	10.9	9.3	53.9	1.1	34.2		
Mattawoman	2017	0.97	11.0	9.3	53.9	1.1	34.2		
Mattawoman	2018	0.97	11.0	8.6	52.8	1.1	35.7		
Mattawoman	2024	1.04	11.5	8.6	52.8	1.1	35.7		

Table 1-1 cont.

Subestuary	Sample Year	C / ha	% IS	% Agriculture	% Forest	% Wetlands	% Urban	Watershed Size (ha)	Primary Land Use
Patapsco	2013	1.11	12.1	24.4	30.4	0.2	43.7		
Patapsco	2014	1.12	12.2	24.4	30.4	0.2	43.7		
Patapsco	2015	1.13	12.3	24.4	30.4	0.2	43.7	93,728	Urban
Patapsco	2016	1.14	12.3	24.4	30.4	0.2	43.7		
Patapsco	2017	1.15	12.4	24.4	30.4	0.2	43.7		
Patuxent	2021	1.41	14.3	20.5	32.1	0.1	45.8	99,960	Urban
Piscataway	2008	1.41	14.3	10.0	40.4	0.2	47.0		
Piscataway	2009	1.43	14.5	10.0	40.4	0.2	47.0		
Piscataway	2012	1.47	14.8	10.0	40.4	0.2	47.0	17,536	Urban
Piscataway	2013	1.50	14.9	10.0	40.4	0.2	47.0		
Piscataway	2014	1.51	15.0	10.0	40.4	0.2	47.0		
Tuckahoe	2016	0.07	1.8	66.6	25.4	0.7	7.3	39,272	Agriculture
Tuckahoe	2017	0.07	1.8	66.6	25.4	0.7	7.3		

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

Subestuary	Year	Number of Sites	1st Sampling Date	Last Sampling Date	Number of Dates	N
Bush	2005	13	18-Mar	15-May	16	99
Bush	2006	13	18-Mar	15-May	20	114
Bush	2007	14	21-Mar	13-May	17	83
Bush	2008	12	22-Mar	26-Apr	17	77
Bush	2014	6	22-Mar	1-Jun	10	60
Chester	2019	14	18-Mar	7-May	8	93
Choptank	2016	12	17-Mar	18-May	10	101
Choptank	2017	11	9-Mar	24-May	14	109
Deer	2012	4	20-Mar	7-May	11	44
Deer	2013	5	19-Mar	23-May	19	87
Deer	2014	5	2-Apr	28-May	12	60
Deer	2015	5	23-Mar	26-May	15	75
Mattawoman	2008	9	8-Mar	11-May	25	90
Mattawoman	2009	7	8-Mar	11-May	16	70
Mattawoman	2010	7	7-Mar	16-May	13	75
Mattawoman	2011	7	5-Mar	15-May	14	73
Mattawoman	2012	7	4-Mar	13-May	11	75
Mattawoman	2013	7	10-Mar	25-May	13	80
Mattawoman	2014	8	9-Mar	25-May	13	87
Mattawoman	2015	7	15-Mar	24-May	11	60
Mattawoman	2016	5	13-Mar	22-May	11	55
Mattawoman	2017	5	5-Mar	28-May	13	65
Mattawoman	2018	5	11-Mar	19-May	11	55
Mattawoman	2024	4	28-Feb	15-May	11	44
Patapsco	2013	4	19-Mar	30-May	22	40
Patapsco	2014	4	4-Apr	29-May	19	28
Patapsco	2015	4	25-Mar	28-May	18	32
Patapsco	2016	4	7-Mar	2-Jun	26	40
Patapsco	2017	4	9-Mar	6-Jun	21	40
Patuxent	2021	12	18-Mar	9-Jun	18	100
Piscataway	2008	5	17-Mar	4-May	8	39
Piscataway	2009	6	9-Mar	14-May	11	60
Piscataway	2012	5	5-Mar	16-May	11	55
Piscataway	2013	5	11-Mar	28-May	11	55
Piscataway	2014	5	10-Mar	1-Jun	9	45
Tuckahoe	2016	10	16-Mar	16-May	12	97
Tuckahoe	2017	10	8-Mar	23-May	11	102

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

RKM	Months	Sum	Years Sampled
12.4	Jan-Dec	218	1971, 1974-1989
18.1	Apr-Sep	8	1974
27	Apr-Sep	9	1970, 1974
30	Aug-Sep	2	1970
34.9	Apr-Sep	9	1970, 1974
38.8	Aug-Sep	2	1970

Table 1-4. Summary statistics of specific conductance (µS/cm) for mainstem stations in Mattawoman Creek during 2008-2018 and 2024. Unnamed tributaries were excluded from analysis.

	Mattawoman											
Specific Conductance	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2024
Mean	120.1	244.5	153.7	147.5	128.9	126.1	179.4	181.8	180.3	151.2	160.7	127.2
Standard Error	3.8	19.2	3.0	2.8	1.9	2.4	9.0	6.4	4.0	3.7	4.3	2.1
Median	124.6	211	152.3	147.3	130.9	126.5	165.8	172.5	188.8	150.2	165.5	129.0
Stzd. Median	1.143	1.936	1.397	1.351	1.201	1.161	1.521	1.583	1.732	1.378	1.518	1.183
Kurtosis	2.1	1.41	1.33	8.29	-0.26	5.01	0.33	1.49	-0.80	-0.55	2.99	0.50
Skewness	-1.41	1.37	0.03	1.72	-0.67	-1.70	1.00	1.33	-0.68	-0.36	-1.70	0.08
Range	102	495	111	117	49	96	261	185	93	102	120	65
Minimum	47	115	99	109	102	63	88	130	121	91	79	99
Maximum	148	610	210	225	151	158	350	315	214	193	198	164
Count	39	40	43	44	44	48	48	44	44	52	44	44

Table 1-5. Summary statistics of ANOVA testing for differences in specific conductance (μS/cm) among mainstem Mattawoman Creek stations.

ANOVA					
Source	df	SS	MS	F	P
Model	3	59890.194	19963.398	6.99	0.0001
Error	530	1512888.406	2854.506		
Corrected Total	533	1572778.6			
$r^2 = 0.0381$					

CV	Root MSE	Conductance Mean
33.7295	53.4276	158.4

Bartlett's Test for Homogeneity of Variance			
Source	df	Chi-Square	P
Station	3	16.478	0.0009

Welch's ANOVA			
Source	df	F	P
Station	3	6.67	0.0002
Error	292.6		

Table 1-6. Site-specific presence-absence of Herring (Blueback Herring, Hickory Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Mattawoman Creek during 1971, 1989-1991, 2008-2018, and 2024. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Shaded box indicates identification is uncertain due to date of observation. Station locations are identified on Figure 1-2.

	Year															
Station	1971	1989	1990	1991	2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	2024
								Herring								
MC1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
MC2	1	1	1	1	0	0	1	1	1	1	1	1	1	1	1	1
MC3	1			1	1	1	1	1	1	1	1	1	1	1	1	1
MC4	1			1	0	0	1	1	1	1	1	1	1	1	1	1
MUT3	1				0	0	0	1	1	1	1	1	1	0	1	
MUT4					0	0	0	0	1	0	0	0				
MUT5	1				1	0	0	0	0	0	1	0				
MUTX											1	0				
								White Perch								
MC1	1	1	1	1	1	0	1	1	0	1	1	1	1	1		1
MC2	0	0	1	0	0	0	0	0	0	1	1	0	1	1		1
MC3	1			0	0	0	0	0	0	0	0	0	1	0		0
MC4	0			0	0	0	0	0	0	0	0	1	0	0		1
MUT3	0				0	0	0	0	0	0	0	0	1	0		
MUT5	0				0	0	0	0	0	1	0	0				
MUTX											1	0				
								Yellow Perch								
MC1	1	1	1	1	1	0	1	1	0	1	1	1	1	1		1
MC2	0	0	0	0	0	0	0	0	0	1	1	1	1	0		0
MC3	0			0	0	0	0	0	0	0	0	0	1	0		0
MC4	0			0	0	0	0	0	0	0	0	0	1	0		0
MUT5	0				0	0	0	0	0	0	1	0				
MUTX											1	0				

Table 1-7. Summary statistics of two-sample t test analyzing the difference in mean proportion of Mattawoman Creek samples with Herring eggs and-or larvae (P_{herr}) between spawning stock time categories (0 = 2008-2011, 1 = 2012-2024).

Two-Sample t Test						
Time Category	N	Mean	Std Dev	Std Err	Minimum	Maximum
0	4	0.3138	0.2719	0.136	0.075	0.66
1	8	0.6575	0.1199	0.0424	0.43	0.78
Diff (1-2)		-0.3437	0.1796	0.11		

Method	Variances	df	t	P
Pooled	Equal	10	-3.13	0.0108

Folded F Test for Equality of Variances				
Method	Num df	Den df	F	P
Folded F	3	7	5.15	0.0686

Table 1-8. Summary of a linear regression model for standardized specific conductance (annual median/province background, µS/cm) versus development level (C/ha).

Linear Model	Standardized conductance = Structure density (C/ha)					
ANOVA	df	SS	MS	F	P	
Regression	1	1.67364	1.67364	17.91	0.00016	
Residual	35	3.27004	0.09343			
Total	36	4.94368				
r^2 = 0.3385						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	1.18467	0.11043	10.72773	0.00000	0.96049	1.40886
C / ha	0.44841	0.10595	4.23242	0.00016	0.23333	0.66350

Table 1-9. Summary statistics of the multiple regression model for development level (C/ha) and spawning stock time category versus proportion of samples with Herring eggs and-or larvae (P_{herr}).

ANOVA			Multiple Regression			
Source	df	SS	MS	F	P	
Regression	2	1.83307	0.91653	42.65	5.40094E-10	
Residual	34	0.73068	0.02149			
Total	36	2.56375				
Adjusted $R^2 = 0.6982$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.49766	0.08032	6.19612	0.00000		
C / ha	-0.26299	0.05439	-4.83557	0.00003	0.44952	0.40749
Time category	0.32688	0.05808	5.62760	0.00000	0.48226	0.48226

Table 1-10. Summary statistics of the multiple regression model for standardized specific conductance (annual median/province background, µS/cm) and spawning stock time category versus proportion of samples with Herring eggs and-or larvae (P_{herr}).

ANOVA			Multiple Regression			
Source	df	SS	MS	F	P	
Regression	2	1.74663722	0.87331861	36.3388791	3.61315E-09	
Residual	34	0.81710921	0.02403			
Total	36	2.56374643				
Adjusted $R^2 = 0.6625$						
	Estimate	SE	t Stat	P-value	Squared Partial Corr Type I	Squared Partial Corr Type II
Intercept	0.64728667	0.12242802	5.28707939	7.2816E-06		
Standardized conductance	-0.29013461	0.06972907	-4.1608845	0.0002037	0.17008	0.33740
Time category	0.42383071	0.05739288	7.38	1.463E-08	0.61597	0.61597

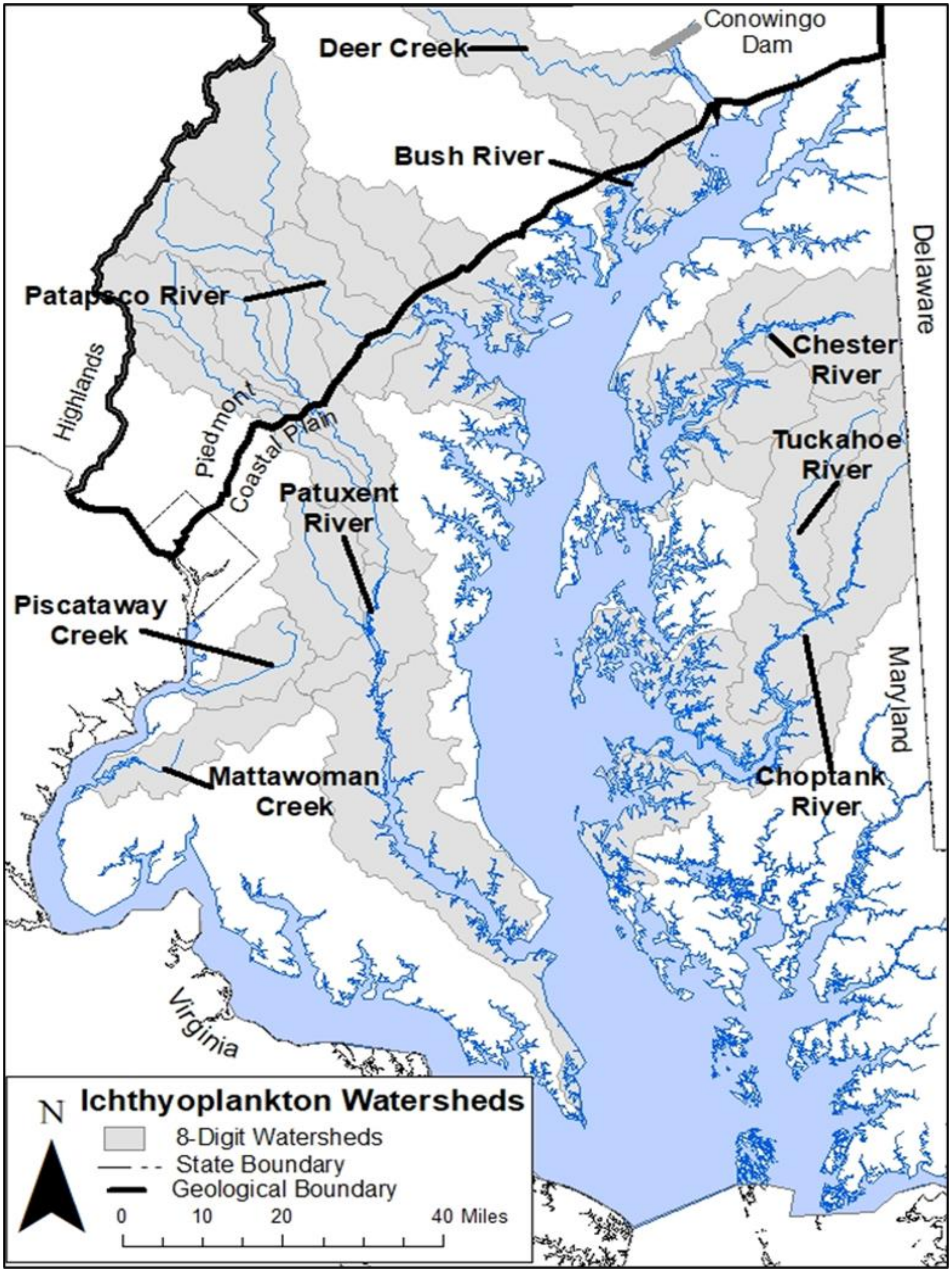

Figure 1-1. Watersheds sampled for stream spawning (i.e., anadromous fish eggs and larvae) during 2005-2021 and 2024. Coastal Plain and Piedmont regions are indicated.

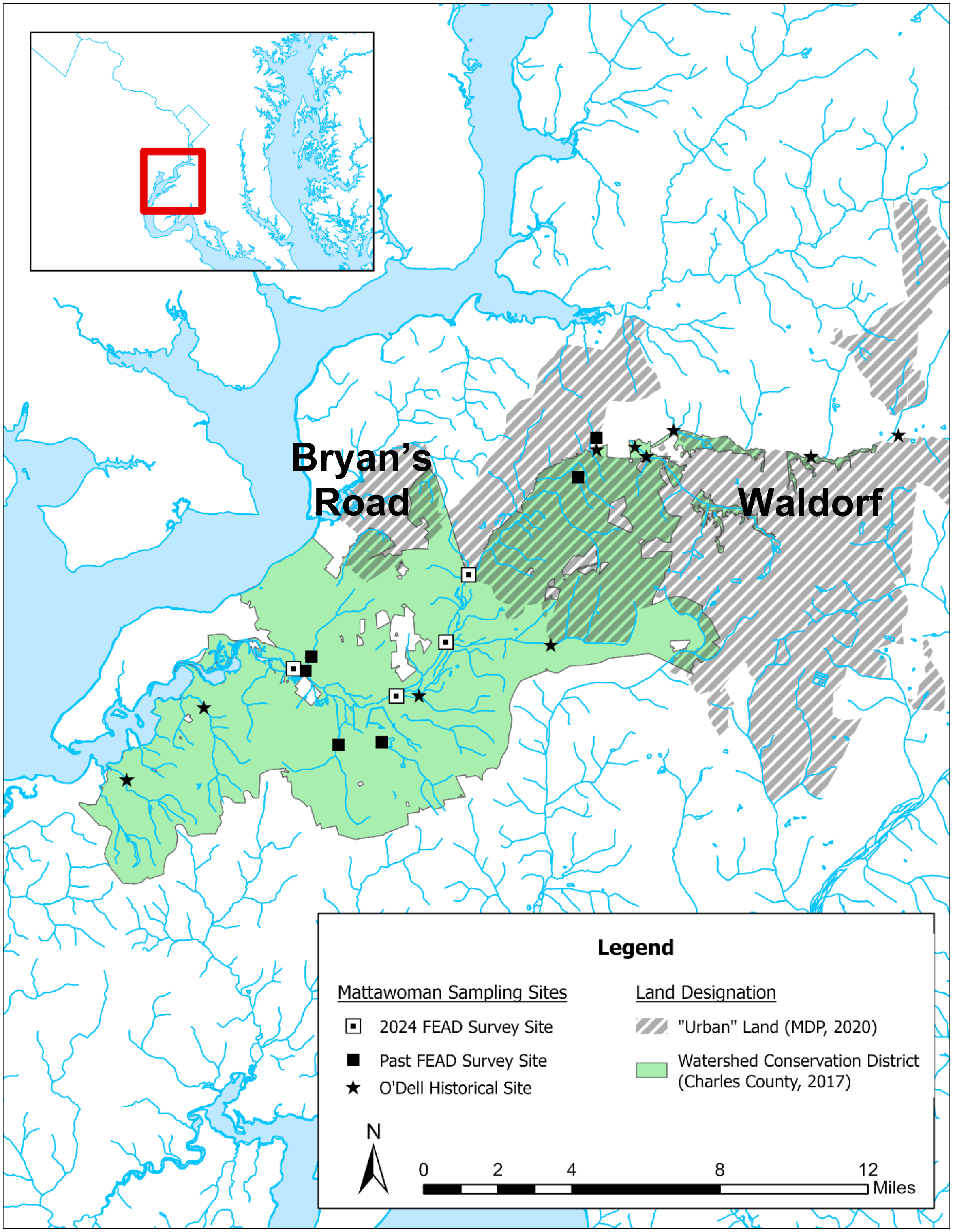

Figure 1-2. Mattawoman Creek's 1971 (O'Dell et al. 1975), 2008-2018, and 2024 sampling stations. Green area represents the area designated by Charles County as the Mattawoman Creek Watershed Conservation District in 2017. Gray area represents developed land, associated with the towns of Waldorf and Bryan's Road, designated as "urban" by the U.S. Census Bureau (Ratcliffe 2022; MDP 2020).

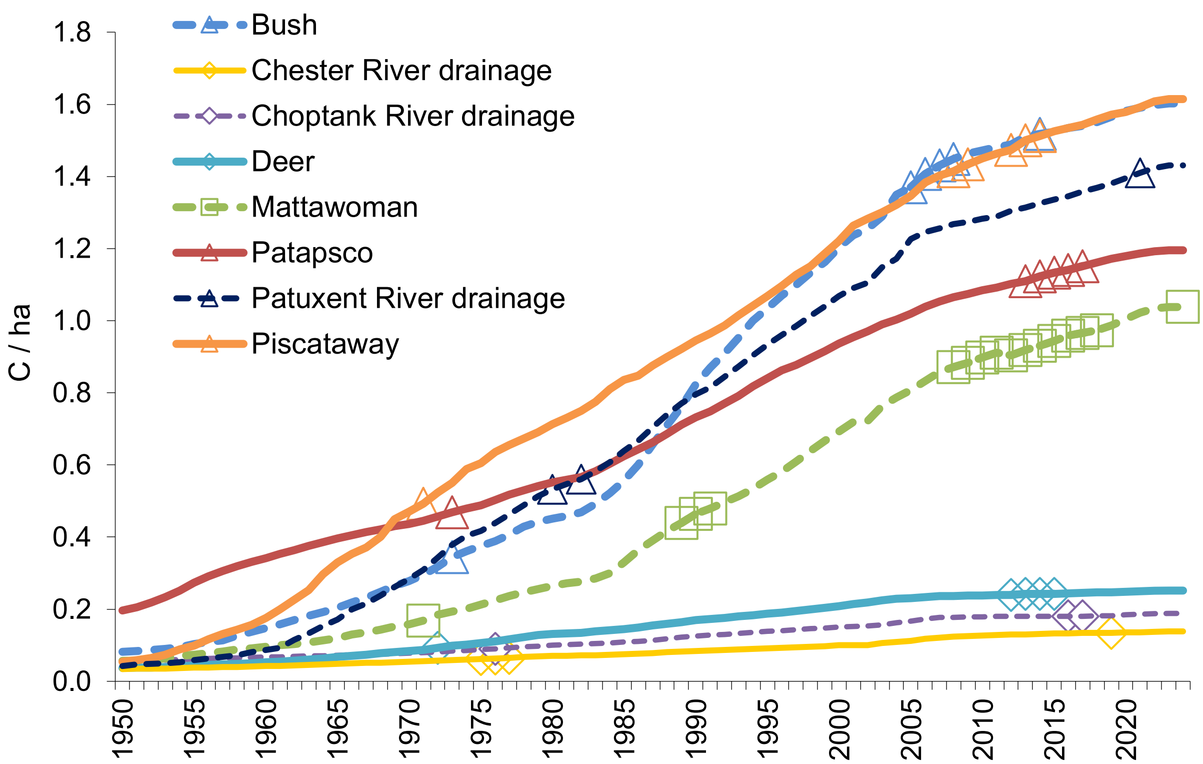

Figure 1-3. Trends in counts of structures per hectare (C/ha) during 1950-2024 in Deer, Mattawoman, and Piscataway creeks; Bush and Patapsco rivers; Chester, Choptank, and Patuxent River drainages. Estimates of C/ha were only available until 2023. Large symbols indicate years when stream ichthyoplankton was sampled.

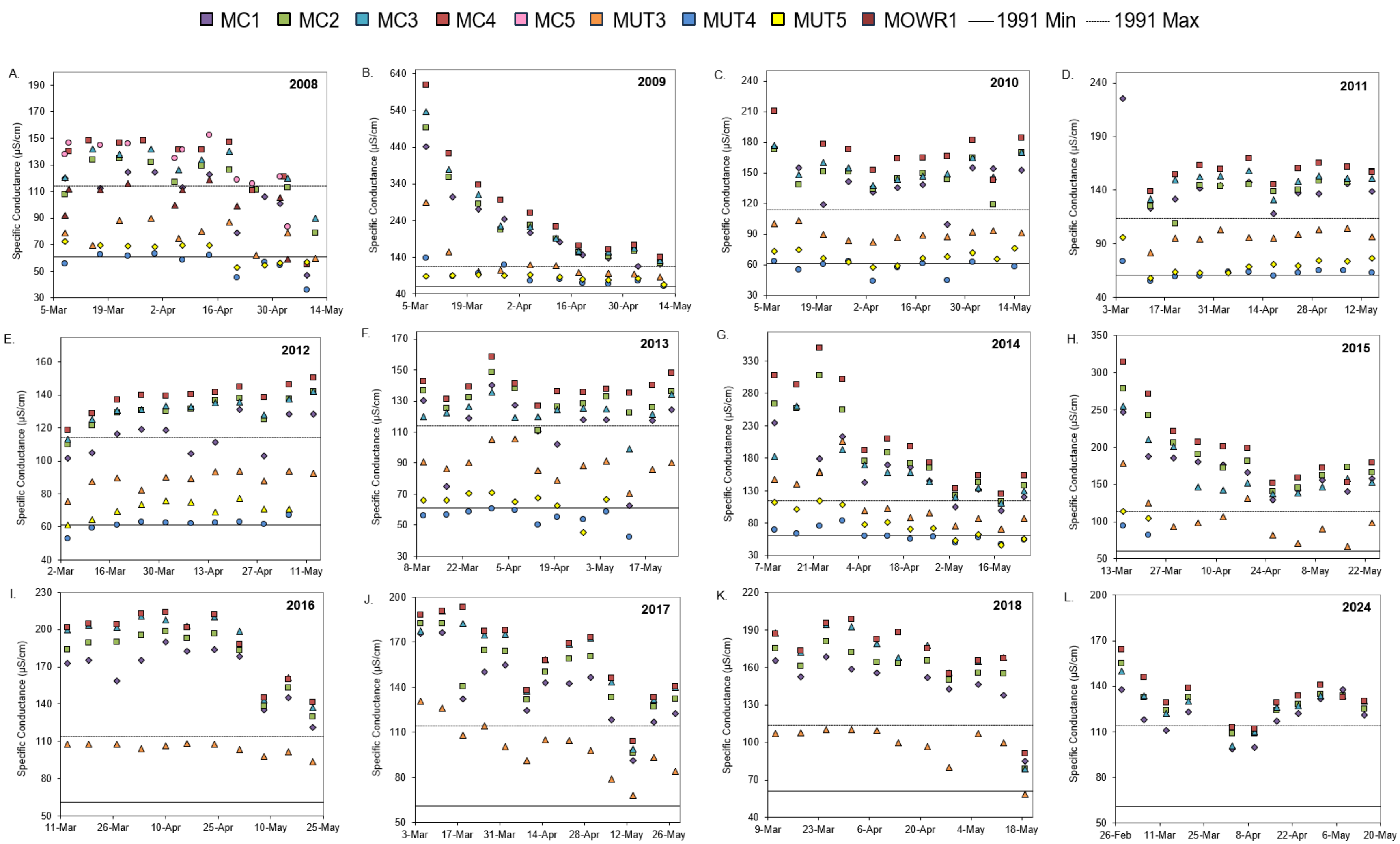

Figure 1-4. Stream conductance measurements (µS/cm), by station and date, in Mattawoman Creek during (A) 2008, (B) 2009, (C) 2010, (D) 2011, (E) 2012, (F) 2013, (G) 2014, (H) 2015, (I) 2016, (J) 2017, (K) 2018, and (L) 2024. Lines indicate conductance range measured at mainstem sites (MC1 - MC4) during 1991 by Hall et al. (1992). Note changes in axis scale among years.

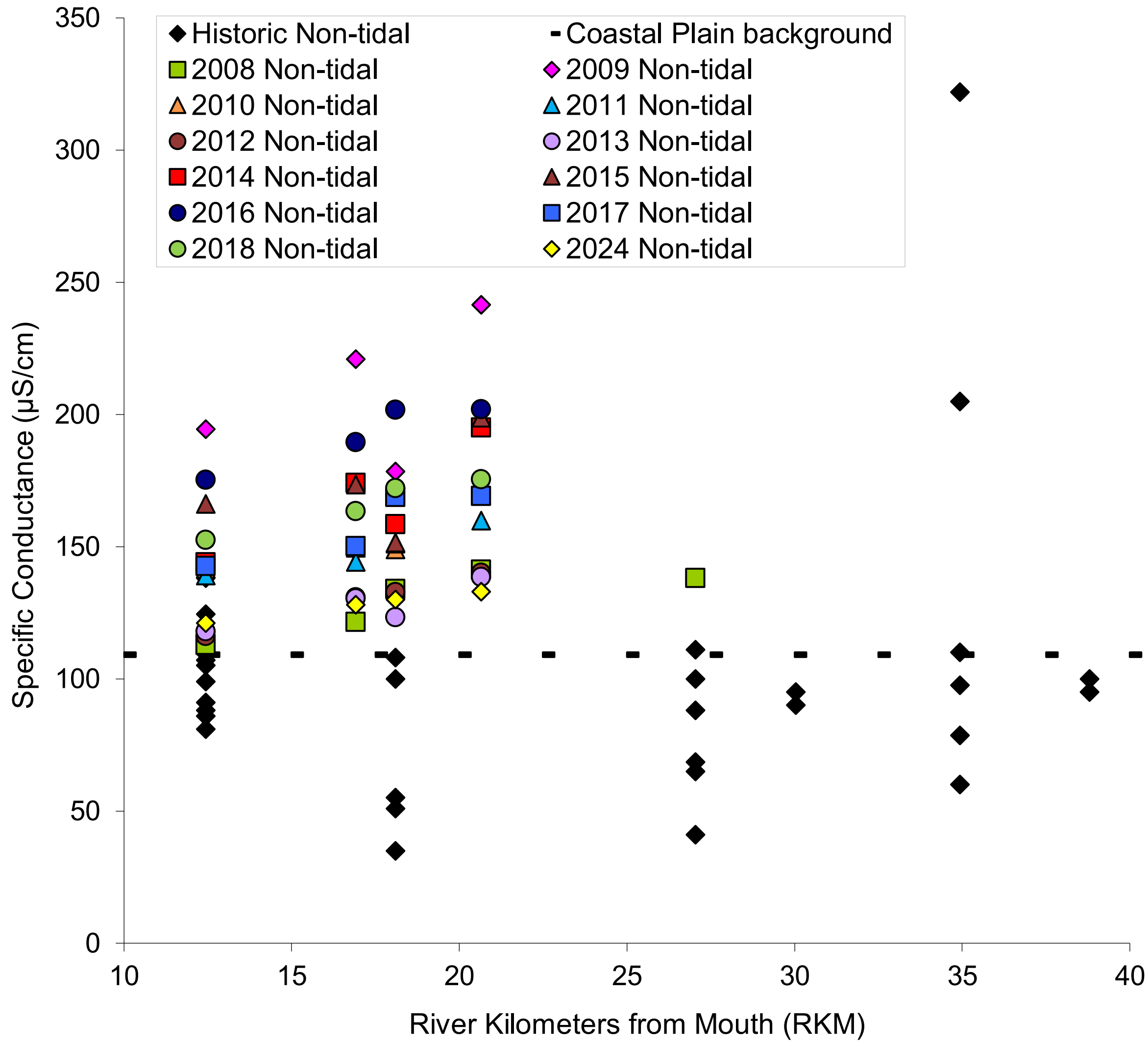

Figure 1-5. Historical (1970-1989) median specific conductance (µS/cm) measurements and current (2008-2018 and 2024) anadromous spawning survey median specific conductance in non-tidal Mattawoman Creek (between the junction with the subestuary and Waldorf) plotted against distance from the mouth (RKM). The two stations farthest upstream are nearest Waldorf. Median conductance was measured during March-May 2008-2018, February-May 2024, and varying time periods during 1970-1989 (see Table 1-3).

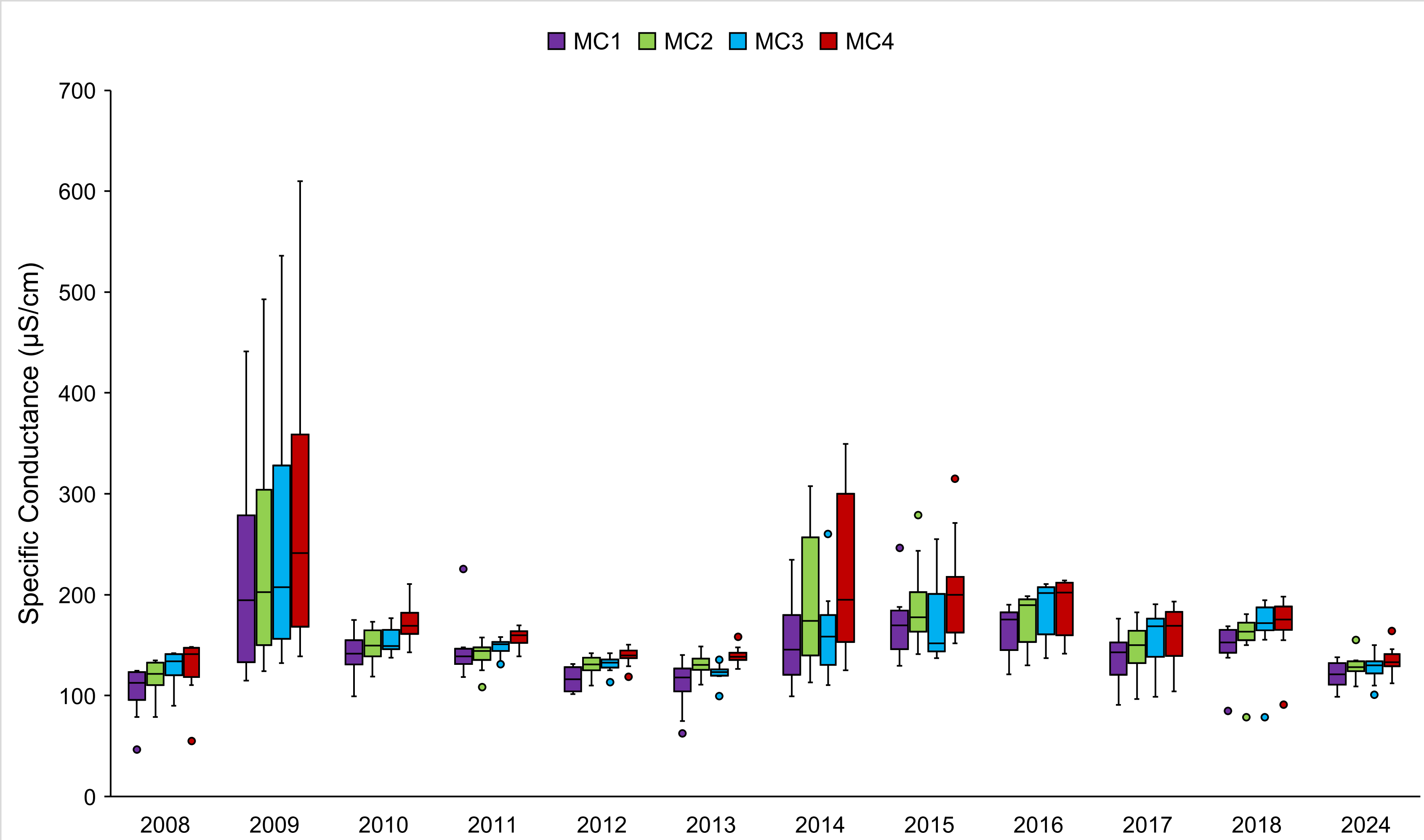

Figure 1-6. Box plots of specific conductance measurements (µS/cm) recorded at each mainstem Mattawoman Creek station in 2008-2018 and 2024. Lines inside each box represent median values and circles represent outlier values.

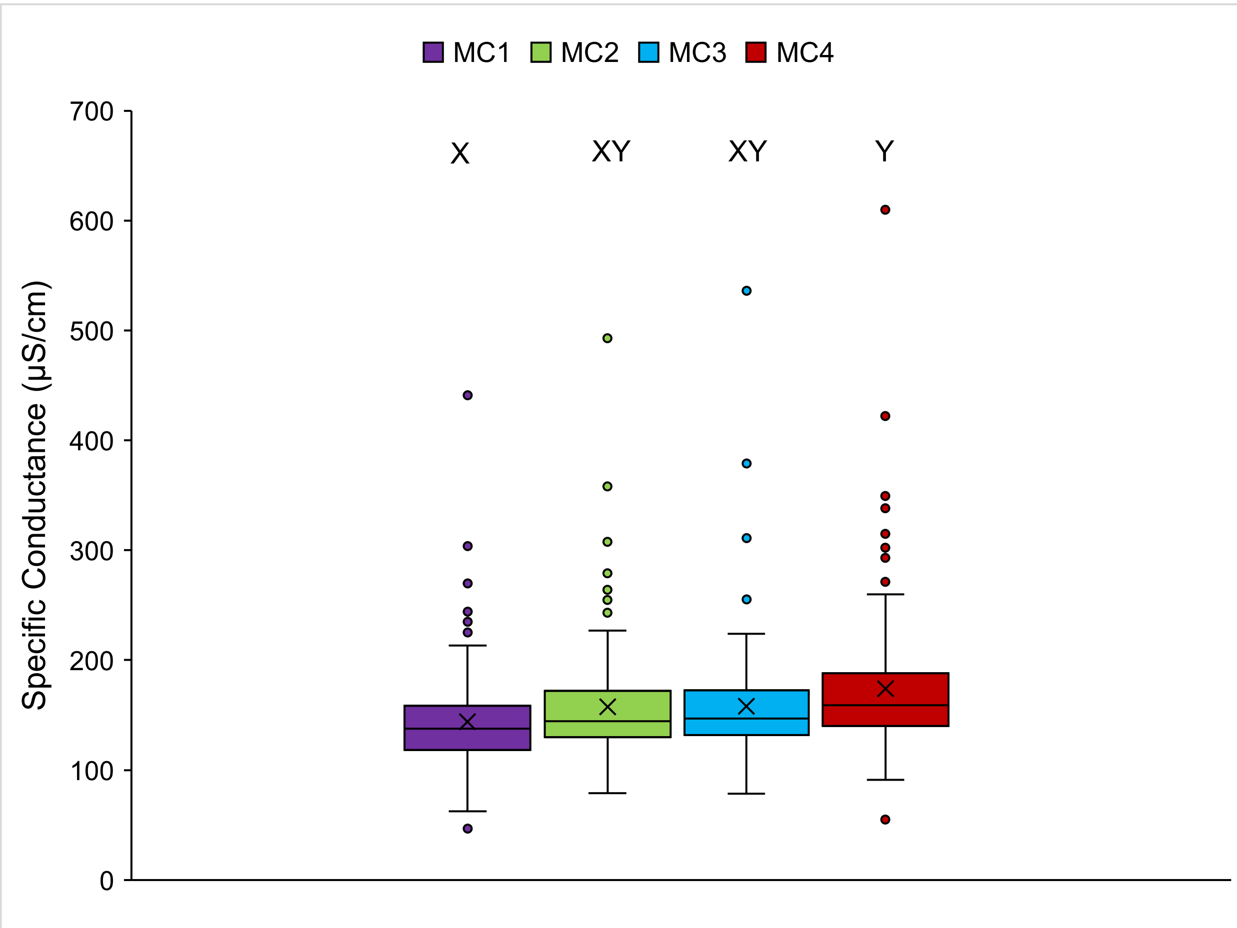

Figure 1-7. Box plots of all specific conductance measurements (µS/cm) recorded across all sampling years (2008-2018 and 2024) at each mainstem Mattawoman Creek station. Lines within each box represent median values, "X"'s within each box represent mean values, and circles represent outlier values. Letters (X, Y) represent significantly different stations.

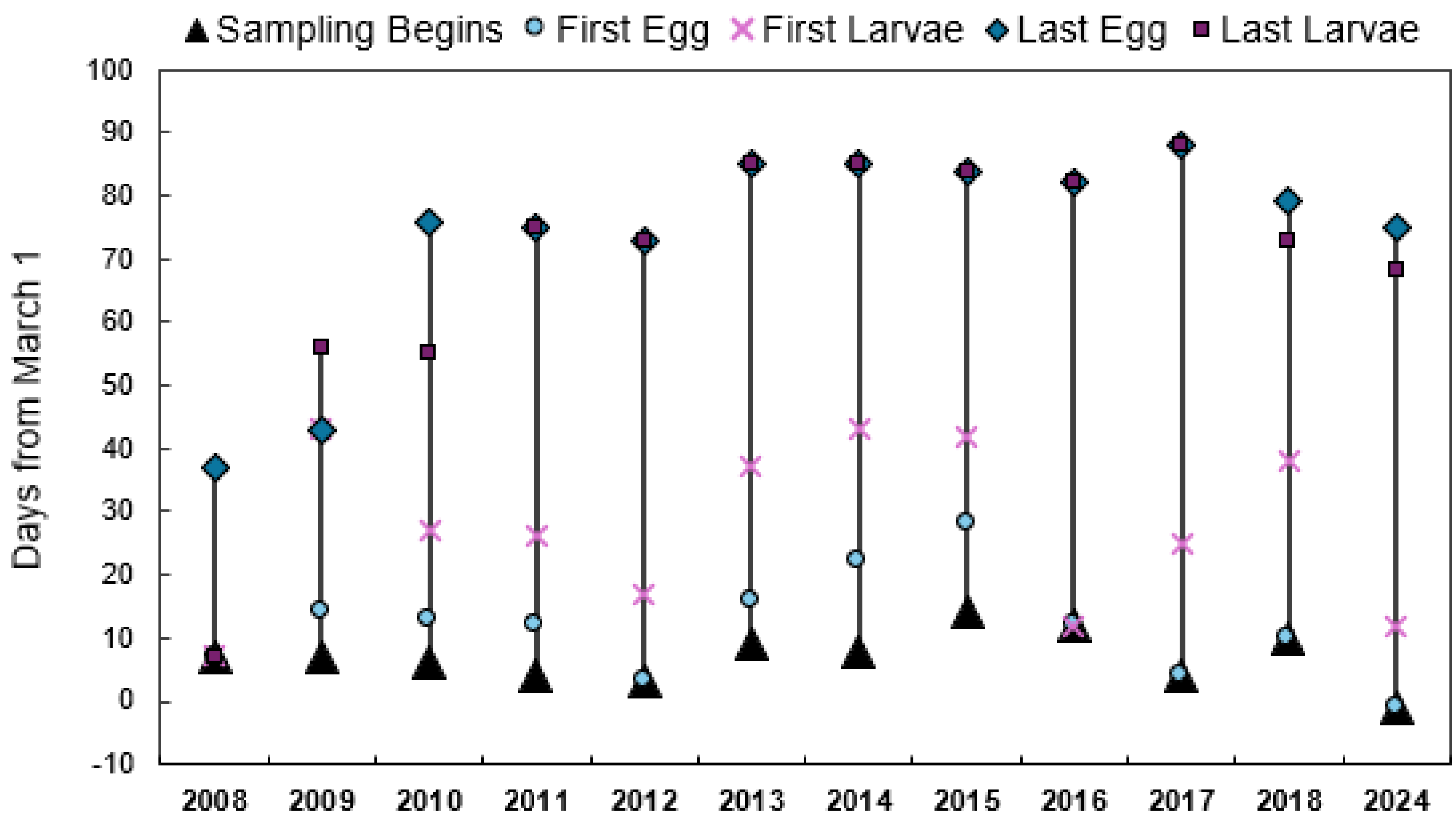

Figure 1-8. Date of first Herring egg, first Herring larvae, last Herring egg, and last Herring larvae observations in Mattawoman Creek samples from 2008-2018 and 2024, displayed as the number of days elapsed since March 1st each year.

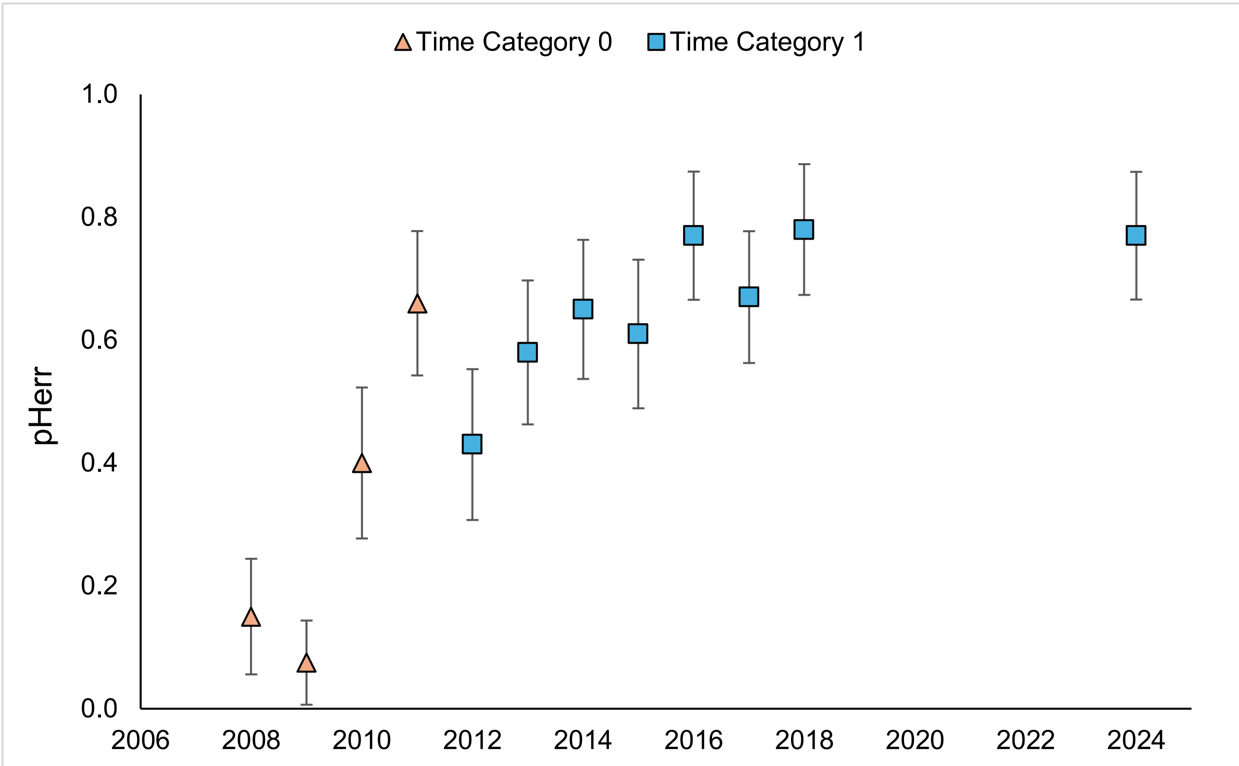

Figure 1-9. Mattawoman Creek P_{herr} (proportion of stream samples with Herring eggs and-or larvae) estimates plotted against year, with triangles representing spawning stock time category 0 (2008-2011, less regulated) and squares representing spawning stock time category 1 (2012-2024, more regulated).

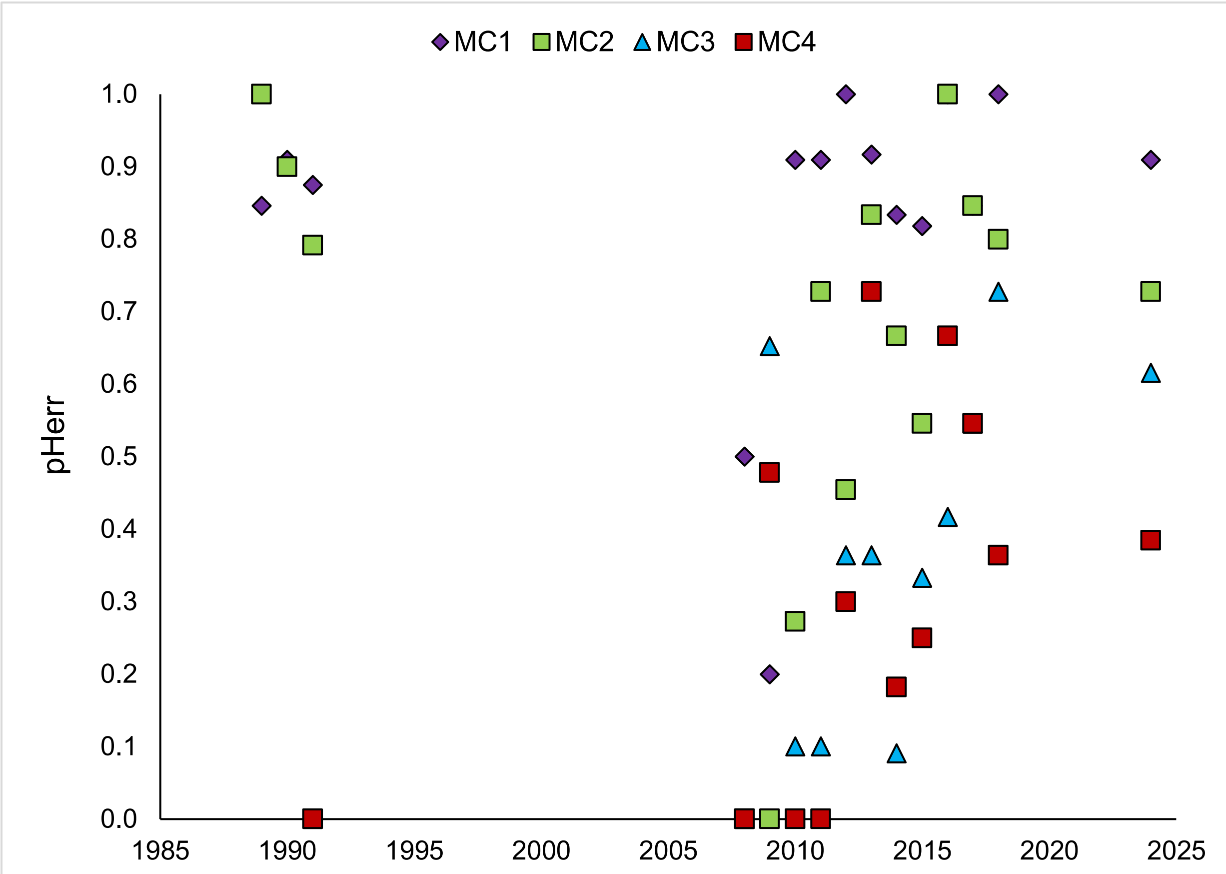

Figure 1-10. Annual P_{herr} (proportion of stream samples with Herring eggs and-or larvae) estimates for each Mattawoman Creek mainstem station plotted against year.

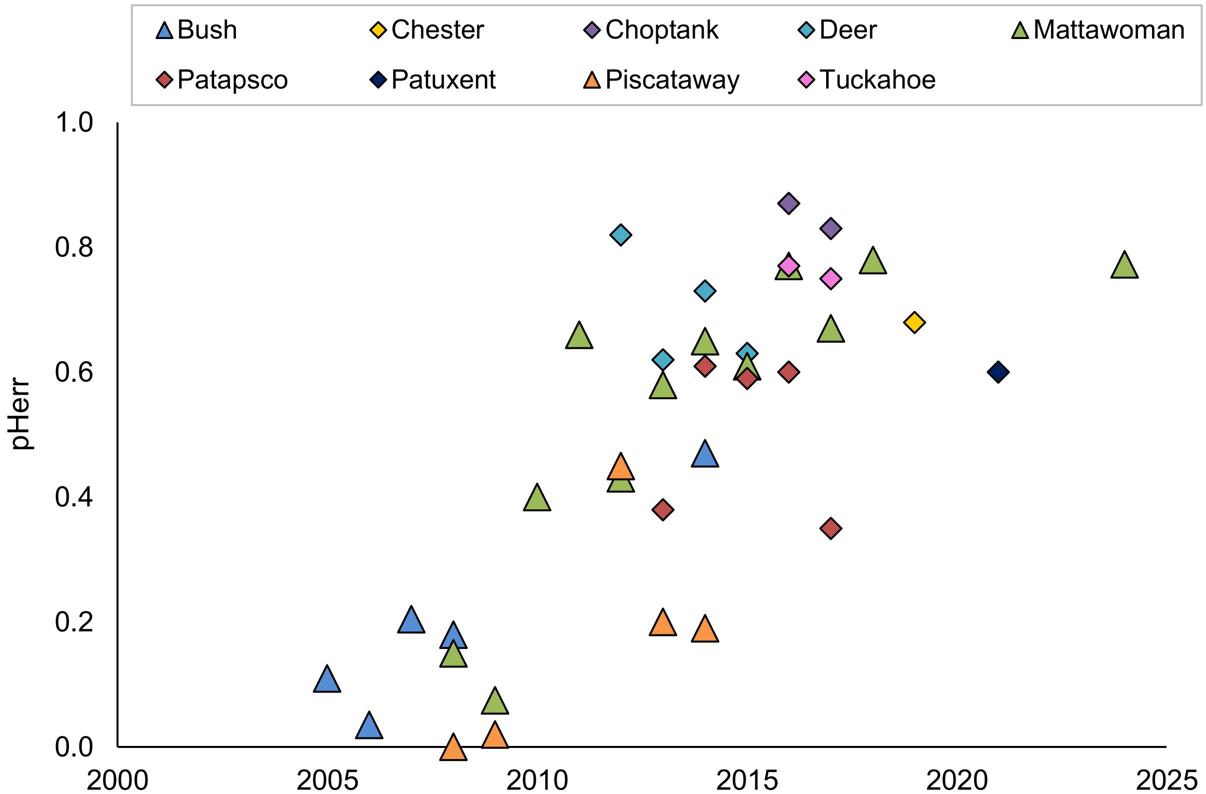

Figure 1-11. Trends in P_{herr} (proportion of stream samples with Herring eggs and-or larvae) by watershed. Watersheds sampled in both early (2005-2011) and late (2012-2024) spawning periods are indicated by large triangles.

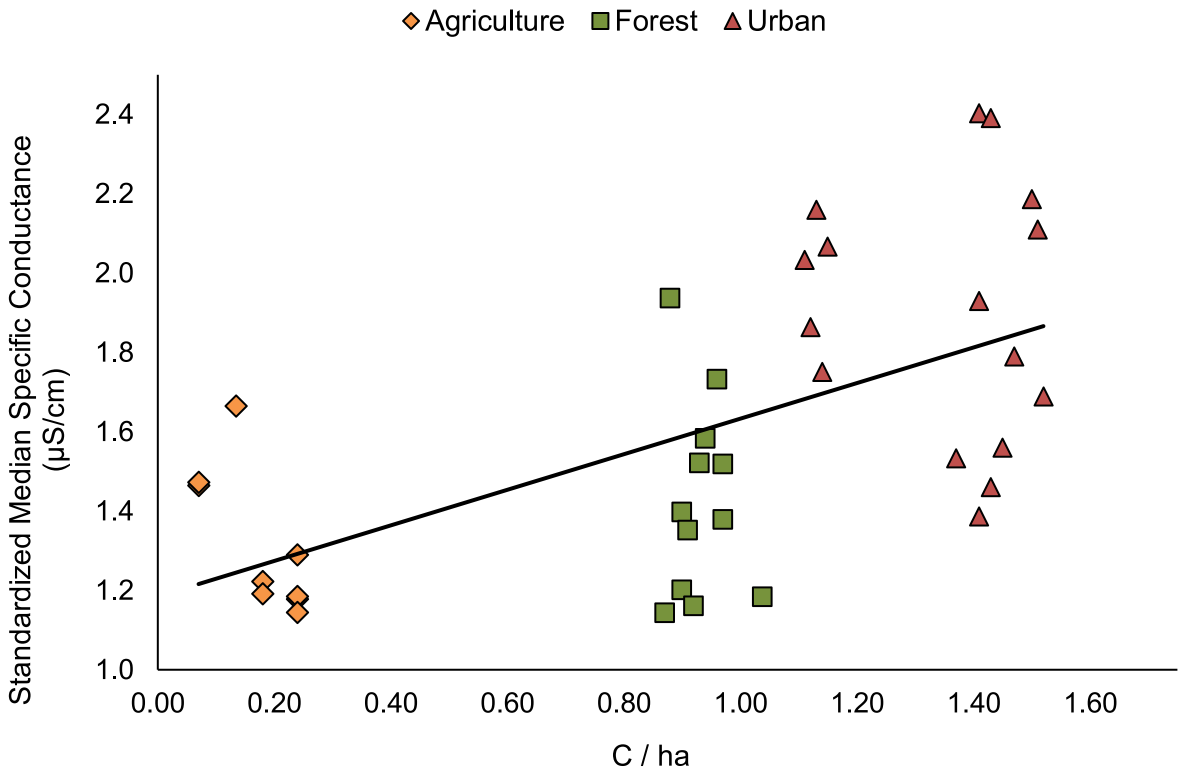

Figure 1-12. Standardized median specific conductance (µS/cm) during anadromous spawning surveys and level of development (C/ha) with dominant MDP land use designations. Median conductance was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates from Morgan et al. (2012).

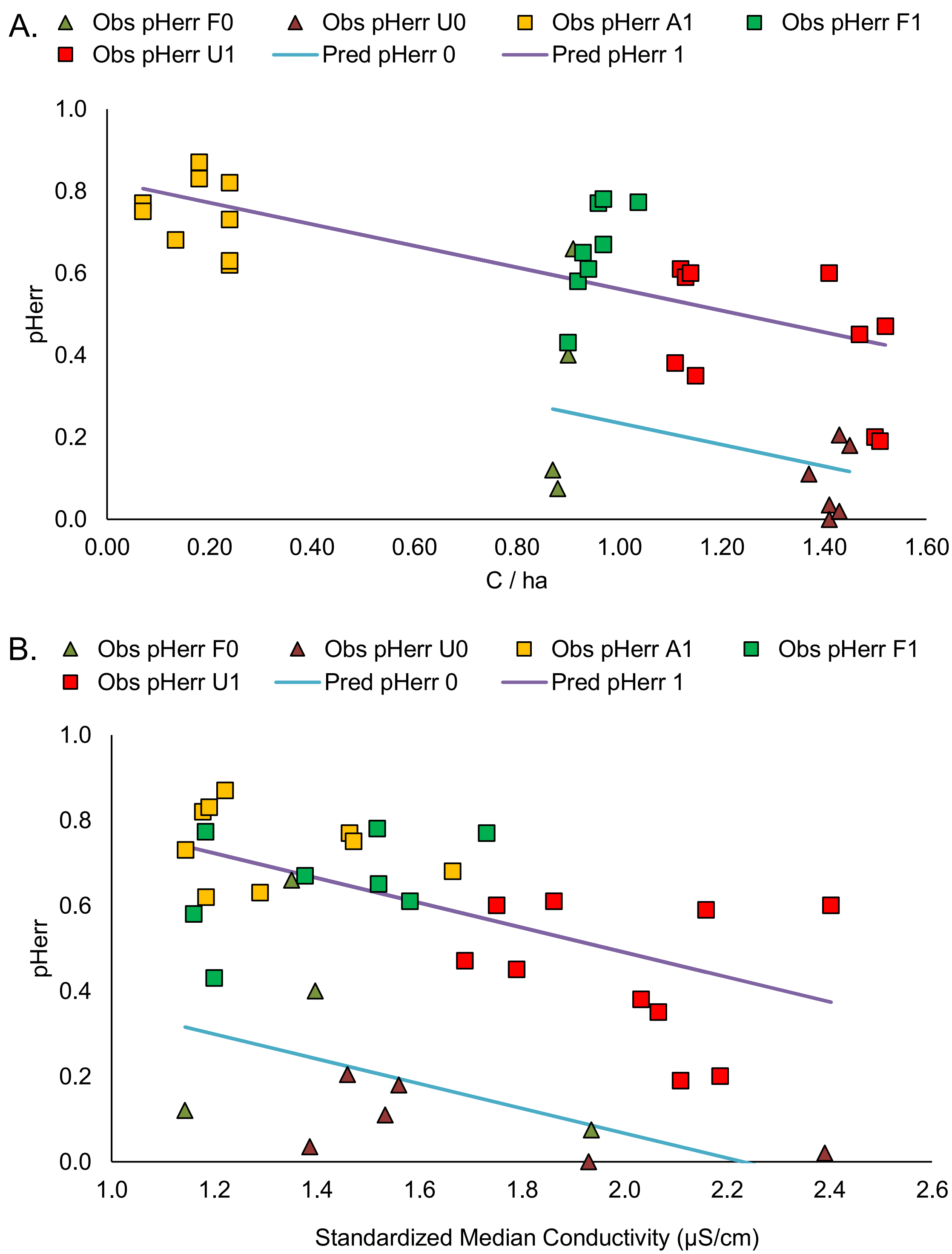

Figure 1-13. Plots of regressions of P_{herr} (proportion of stream samples with Herring eggs and-or larvae) against (A) level of development (C/ha) or (B) standardized median specific conductance (µS/cm) with spawning stock time categories (0 = 2005-2011; 1 = 2012-2024) included. Median specific conductance was standardized to background estimates for the Coastal Plain and Piedmont regions based on Morgan et al. (2012).

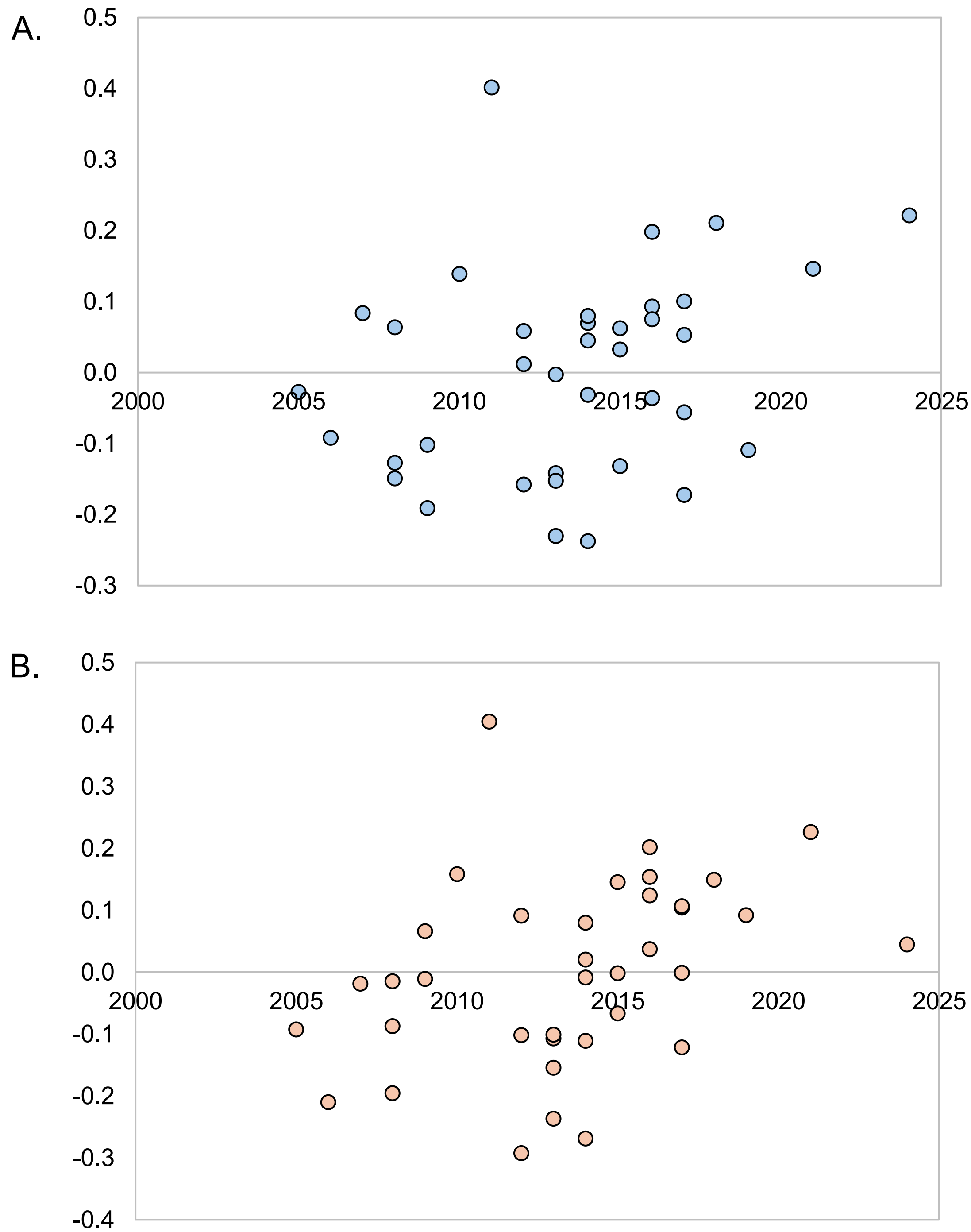

Figure 1-14. Residuals plotted against year for multiple regressions of spawning stock time category and (A) level of development (C/ha) or (B) standardized median spawning survey specific conductance (µS/cm) against proportion of stream samples with Herring eggs and-or larvae (P_{herr}). Median specific conductance was standardized to background estimates for the Coastal Plain and Piedmont regions based on Morgan et al. (2012).

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

Jim Uphoff, Alexis Park, Shannon Moorhead, Marisa Ponte, 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 in 2024. We use L_p as a response variable when examining the impact of major land uses (development or urban, agriculture, forest, and wetlands) and stressors that may be associated with these land uses.

Choptank River is a Coastal Plain watershed on the eastern side of Chesapeake Bay. 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 ‰) from the mouth and upriver to Cambridge (MD DNR 2024). The Striped Bass and Yellow Perch nursery that we sample is tidal-fresh to oligohaline and its extent varies annually due to river discharge.

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 about degradation of Striped Bass spawning and larval nursery habitat in Chesapeake Bay and 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 2024 investigation of Striped Bass egg and larval habitat.

In addition to examining the effects of development, we investigated the influence of winter temperature conditions on L_p during 1963-2024. We used summarized average winter air temperatures (December-February) at Baltimore as an indicator of winter intensity to investigate their relationship with L_p estimates for the Nanticoke River and Choptank River. Annual L_p provides a measure of the product of egg production and egg through early postlarval survival and detecting the effect of winter conditions on hatching success through L_p seemed reasonable. Both rivers have long-term estimates of L_p and have remained rural (Table 1). Changes in L_p due to development would not be expected. Widespread low L_p occurs sporadically in Chesapeake Bay subestuaries with rural watersheds, reflecting March temperatures (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. 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 (Farmer et al. 2015).

We did not repeat linear regressions of land use and L_p in this report and they are covered in Uphoff et al. (2024). A single point represented by Choptank River was very unlikely to change the analyses since the Choptank River has been well represented. We also did not repeat reporting results for tidal-fresh subestuaries since none were sampled in 2024. Those are summarized in Uphoff et al. (2024) as well.

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 eachsample date.

Ten sites were sampled twice weekly in all systems unless weather or salinity did not allow. Boundaries of areas sampled in watershed of small subestuaries (watershed area < 73,000 ha) were determined from Yellow Perch larval presence in estuarine surveys conducted during the 1970s and 1980s (O'Dell 1987). 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; 2023) and continuity with past surveys was maintained by sampling these Striped Bass spawning areas.

The Choptank River spawning area was divided into 19 1.61-km segments, starting at km 47.2 and proceeding upstream (Figure 2.1). We could not access two of the furthest upstream stations sampled in the past (stations 17 and 21) because of shallowing. 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.

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.** Development targets and limits, and general statistical methods (analytical strategy and equations) are described there as well.

Estimates of C/ha and Maryland Department of Planning (DOP) land cover (agriculture, forest, and wetland) percentages in watershed or portions of watershed located in Maryland were used as measures of watershed land use for analyses from 1973-2010, while estimates after 2012 were made from newer Chesapeake Conservancy data (Table 2-1; MD DOP 2015; Chesapeake Conservancy 2023). Updates to MD DOP land use estimates have not been released since 2010. Conversion factors for Chesapeake Conservancy high resolution land us datasets (2013/2014 and 2017/2018) were developed and used to generate agriculture, forest, wetland, and urban estimates comparable to those for MD DOP data (Table 2-1). 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 available west of the Susquehanna River, but some estimates for watersheds to the east were truncated at the Maryland Border due to lack of comparable land use data (Figure 2.2). The percentage of the watershed in Maryland as a percent of the total watershed (USGS estimates) was 35% for the Nanticoke River, 80% for Choptank and Chester rivers, 99% for Wicomico River (eastern shore region of Maryland or ES), and 61% of Elk River. Nanticoke River, Choptank River, Wicomico River ES, Chester River, and Patuxent River watersheds were truncated at the lower boundaries of their Striped Bass spawning areas (Figure 2.2). Estimates of C/ha were available from 1950 through 2024.

We classified Yellow Perch spawning subestuaries as brackish or tidal-fresh (salinity > 2.0‰ in the subestuary outside of the larval nursery or salinity always ≤ 2.0‰, respectively). Optimal salinity for Yellow Perch spawning was less than 2.0‰ (Piavis 1991) and area for spawning and larval nursery would be limited in brackish subestuaries. Choptank River was classified as brackish and plotted with the time-series of all brackish subestuaries sampled for L_p. We denoted whether watersheds were small (< 60,000 ha) or large on this plot. Choptank, Nanticoke, Chester, and Patuxent rivers were classified as large watersheds. The relationship between C/ha and L_p was strong in small, brackish systems while a relationship was not detected for large, brackish systems (Uphoff et al. 2024).

Correlation analysis was used to explore associations among temperature, DO, pH, and conductivity during the period L_p 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.

A view of the relationship of L_p and C/ha was developed by considering dominant land use (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. Urban land consisted of high and low density residential, commercial, and institutional acreages.

We used summarized average winter air temperatures (December-February) at Baltimore (https://marylandclimateandweather.weathertogether.net/maryland-climate-data/) during 1963-2024 as an indicator of regional winter intensity to investigate their relationship with L_p estimates for the Nanticoke River and Choptank River. These temperature data extend back to 1871 (Taylor 2024). They were tabulated as °F and we converted them to °C. Their sources are the National Weather Service and the National Center for Environmental Information. This data for Baltimore includes downtown locations (1871-1950) and Baltimore-Washington International Airport (1950-Present). The differences in the weather instrument sitings between the urban location rooftop and the suburban airport often resulted in large temperature differences but the National Weather Service/National Centers for Environmental Information's official long-term historical archive for Baltimore is maintained in this record (Taylor 2024).

We used linear regression to examine the relationship of winter temperature to L_p in the Choptank and Nanticoke rivers. These rivers were chosen because they have remained rural and had long time-series (Table 2-1). They were closed to harvest during 1989-1999 and open only for recreational harvest after 1999 and we would expect exploitation to be light under those conditions (MD DNR 2002). Commercial landings in both rivers prior to 1989 were small (P. Piavis, MD DNR, personal communication). Estimates of L_p were available for the Choptank River during 1980-1990, 1998-2004, and 2013-2024 (except 2020; Table 2.2). Nanticoke River surveys for 1963-1968, 1970-1978, 1979, 1981, and 2004-2019 were available (Table 2.2). Choptank River surveys during 1980-1990 used plankton trawls (Uphoff 1992; 1993) and these nets were more likely to detect larger larvae than the 0.5 m nets that were used in other surveys (Uphoff 1991; J. Uphoff, personal observation). This could induce a negative trend in the overall time-series due to positive bias since the initial years of the Choptank River time-series were all based on plankton trawl surveys. We excluded 1980-1990 Choptank surveys from analysis and confined analysis to surveys based on 0.50 m conical plankton net collections.

We examined L_p and winter temperature (°C) for Nanticoke River only and for combined Choptank River and Nanticoke River. The Nanticoke River analysis used the L_p estimates as the dependent variable. Review of L_p estimates strongly suggested that Choptank River L_p estimates based on 0.5 m plankton net collections were consistently higher than Nanticoke River L_p. To account for this difference in scale, we estimated the time-series median L_p for each river and divided observations by their respective medians to produce standardized estimates. These standardized estimates of L_p (denoted as SL_p) were regressed against mean winter temperature.

We examined the relationship of winter air temperature to days to reach the 18°C cutoff (as days from April 1; April 1 = day 0) for estimating L_p in Choptank River and Nanticoke River with linear regression to address whether this ending date was linked to overall winter conditions. We used both winter temperature (T) and T2 as independent variables in two different regression analyses. Inspection of the plot indicated the possibility of an asymptote through an initial portion of T (an asymptote) and a decrease in the latter portion (Figure 2.10). To avoid applying a complex nonlinear equation to fit these data, we used T^2 as the independent term in a linear regression with D. This transformed the small negative to small positive values and approximated an asymptote for the lower values. Choptank and Nanticoke rivers are adjacent to one another in Maryland's eastern coastal plain and temperature conditions should have been similar enough that they could be combined. We could not address the date of the start of the L_p interval since many of the early surveys in these rivers were directed at Striped Bass eggs and often did not provide survey visits prior to when Yellow Perch larvae were present.

Results

Watershed land use estimates and L_p for 1963-2024 by salinity type are summarized in Table 2-1. Estimates of L_p for all waterhseds and years, their SD, N, and starting and end dates for estimating Lp are summarized in Table 2-2.

Sampling in 2024 began on March 12 in Choptank River and lasted until May 6. Samples between March 21 and April 10 were used to estimate L_p in Choptank River. The estimate for of mean L_p in Choptank River in 2024 (L_p = 0.61, SD = 0.06) was the tenth highest L_p out of 51 estimates for large subestuaries (Table 2.1; Figure 2.3). The chance that L_p fell below the brackish threshold in Choptank River during 2024 was 0%.

The range of C/ha values available for analysis with L_p was 0.05-2.86 for brackish subestuaries (Table 2-1). Estimates of L_p declined with development in brackish tributaries sampled (Figure 2.4). An extensive range of L_p estimates were present when C/ha was 0.22 and median L_p was 0.52. Beyond C/ha = 0.22, the range was similar but there were fewer high L_p values and median L_p decreased to 0.24.

Although we have analyzed these data by distinguishing tidal-fresh and brackish subestuaries, an alternative interpretation based on primary land use was possible. 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 agriculture as its dominant land was available (Figure 2.4). 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; 65 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 forested Nanjemoy Creek. Increasing suburban land cover led to lower L_p regardless of rural land cover type (Figure 2.4).

Water temperature varied between 9 and 12˚C during March 21 and April 8; it increased to about 14˚C on April 10 and increased beyond the 18˚C cutoff for estimating L_p by April 15 (Figure 2.5). Dissolved oxygen was between 7.0 and 10.0 mg/L during March 21-April 10 and fell to between 5.4 and 7.5 mg/L on April 15 (Figure 2.5). Measurements of pH varied between 6.18 and 7.42 (Figure 2.6). Conductivity ranged from 75 to 1,200 µS/cm (Figure 2.6). Conductivity was low in Choptank River in 2024 and reflected high flows in late winter-early spring (see Striped Bass section 2.1).

Correlation analysis of these parameters in Choptank River during the L_p period (March 21 – April 7; N = 70) indicated a modest negative association of temperature and DO (r = -0.50, P < 0.0001), a poor correlation of temperature and pH (r = -0.37, P = 0.0013) and a moderate correlation of pH and DO ($r = 0.54$, $P < 0.0001$). This correlation analysis suggested that temperature and phytoplankton photosynthesis modestly influenced water quality dynamics.

Winter mean air temperatures in Baltimore increased during 1963-2024 (Figure 2.7). The lowest temperature was -1.61°C in 1963; mean temperatures remained below 1.67°C through 1970. They varied between -0.94 and 3.44°C through 2015 and between 2.44 and 5.56°C afterward (Figure 2.7). A linear regression indicated that temperature increased, on average, by 0.036°C per year since 1954 ($r^2 = 0.20$, $P < 0.0001$).

Estimates of L_p in the Nanticoke River during 1963-2019 varied from 0.04 to 0.77 (Table 2.2). The two lowest estimates of L_p occurred after 2004 and estimates greater than 0.60 occurred before 2008 (Table 2.2). A linear regression of winter mean air temperature (T) and L_p was negative and accounted for a modest amount of variation ($r^2 = 0.27$, $P < 0.002$; Figure 2.8). The relationship was described by the equation:

$$L_p = (-0.054 \bullet T) + 0.51;$$

the SE for the slope was 0.016 and 0.04 for the intercept. Estimates of L_p greater than 0.5 only occurred when T was 2.6°C or less (Figure 2.8). Serial patterning of residuals was not evident.

Median estimates of L_p were 0.39 for the Nanticoke River and 0.56 for Choptank River for their entire time-series (0.5 m diameter plankton nets only). Confining estimation of medians to a more recent period (1998-2024) to better match the two time-series resulted in the same estimated medians. These medians were used to standardize L_p between the two rivers. There were concurrent estimates of L_p in 2004 and 2013-2019 and Choptank River L_p was always greater than for Nanticoke River. Differences (Choptank L_p – Nanticoke L_p) ranged from +2% to +95% of Nanticoke River L_p and were +35% overall.

Estimates of SL_p (standardized L_p) in the Choptank and Nanticoke river during 1963-2024 varied from 0.10 to 1.77. A linear regression of T and SL_p was negative and accounted for a limited amount of variation ($r^2 = 0.13$, $P < 0.013$; Figure 2.9). The relationship was described by the equation:

$$SL_p = (-0.078 \bullet T) + 1.17;$$

the SE for the slope was 0.031 and 0.08 for the intercept. Proportion of tows with larvae was predicted to be 1.29-times the median at the lowest temperature in the time-series, -1.61°C, and 0.74-times the median at the maximum observed, 5.56°C; this represents a predicted decline of 43%. Highest values were present at 2.61°C or less but observed values above the median occurred throughout the range of observed temperatures (Figure 2.9). Serial patterning of residuals was not evident. This analysis would be sensitive to the accuracy of standardization to the median to match the proper scale of differences between Choptank River and Nanticoke River L_p.

The linear regression to evaluate the relationship of T with days (D) from April 1 (day 0) it took to reach the 18°C cut-off for calculating L_p (Choptank and Nanticoke rivers combined). was described by the equation:

$$D = (-1.87 \bullet T) + 29.15 \ (r^2 = 0.18, P = 0.0008; \text{Figure 2.10}).$$

The SE for the intercept and slope were 0.53 and 1.41, respectively.

This relationship asymptotic relationship approximate with T^2 fit the data better and was described by the equation:

$$D = (-0.21 \bullet T^2) – 28.59 \ (r^2 = 0.25, P = <0.0001; \text{Figure 2.10});$$

the SE was 0.11 for the slope and 1.13 for the intercept. The predicted line indicated little change in D (D ~ 27 or April 28) when T was less than 2°C and steadily declined afterward to D

~ 14 (April 15) by 5.6°C (Figure 2.10). Winter mean temperatures above 2°C became more frequent in the 1980s and have been entirely above 2°C since 2016.

Discussion

General patterns of land use and L_p have emerged: L_p was negatively related to development and positively associated with two rural features: forest and agriculture. 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 less than 0.60. 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.

Choptank River L_p in 2024 was above the highest level observed in the western shore subestuaries with heavily developed watersheds (Magothy, Severn, and South rivers; median $L_p = 0.13$ and maximum = 0.40). Similarly sized subestuaries with rural watersheds did not exhibit consistently depressed L_p. It became evident that Nanticoke River had consistently lower L_p at low development (below target) throughout its time-series and this low L_p was shared by Wicomico River in 2017-2018 (the only other lower eastern shore subestuary sampled). Salisbury and its suburbs are located on Wicomico River and development is between the target and threshold level; there could be some development influence there. These consistently lower L_p estimates suggest a lower baseline for lower eastern shore brackish subestuaries L_p that may reflect other habitat limitations.

Other factors can be identified that potentially contribute to variation in L_p: winter temperature intensity; salinity, summer hypoxia, and maternal influence. Some of these factors may not be independent and there is considerable potential for interactions among them and with development.

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).

Mean air temperature in Baltimore (a regional winter temperature intensity indicator) was modestly related to L_p in Nanticoke River and Choptank River and when 18°C was reached (end point for estimating L_p). This analysis provided some support for the hypothesis that reproductive success declines followed short, warm winters. Regression results likely revealed the presence of an underlying influence of winter intensity on L_p but did not explain annual variability that reflected other long and short-term influences.

Austin (2002) described low frequency patterns in lower Chesapeake Bay (Virginia's portion) water temperature, river discharge and surface winds and characterized them as dominant decadal regimes (warm-wet or cold-dry periods) of oscillatory waves with dramatic phase shifts (Austin 2002). Winter-spring climate variability was considered a prime candidate as an environmental driver of anadromous fish recruitment in the Bay that resulted in positive or negative shifts in anadromous fish recruitment success that lasted for a decade or more. (Wood and Austin 2009).

Szuwalski et al. (2015) offered synchronous shifts in long-term climate patterns as common environmental drivers of shifts in fish production. Knowledge of the prevailing background climatic regime can provide managers an estimate of the relative chance for high or low year-class success as reflected by recruitment patterns of the dominant production regime (Austin 2002). Austin (2002) suggested that positive correlations of Lowess smoothed water temperatures in the lower Bay (Virginia Institute of Marine Science pier, 1960-2000) and the North Atlantic Oscillation (NAO) indicated coherence of low frequency trends. The NAO indexes wind balances of the northwest Atlantic Ocean; a strong NAO in winter results in a strong westerly flow that drains off cold Canadian air (Austin 2002). We created a winter NAO index (tabulated monthly indices from the National Weather Service's Climate Prediction Center website: https://www.cpc.ncep.noaa.gov/products/precip/CWlink/pna/nao_index.html) as the mean of December (year t) through February (year t+1) monthly NAO indices that matched timing of our winter air temperature indicator of regional winter intensity (Figure 2.11). Correlation analysis indicated that these indices were positively and moderately associated ($r = 0.55$, $P < 0.0001$; Figure 2.11). The NAO is influenced by climate warming (higher CO_2 levels increase temperatures) and volcanic activity (cooler temperatures; Mitevski et al. 2025; Smith et al. 2025). It is possible that the NAO resembles an atmospheric oscillation driven by atmospheric and oceanic interactions but is not, rendering it unpredictable. Earth System Models generally project a more positive and less variable NAO under 21st century high-emission scenarios (Mitevski et al. 2025; Smith et al. 2025).

Widespread low L_p has occurred sporadically in Chesapeake Bay subestuaries with rural watersheds and appeared to be linked to high March 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).

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; National Research Council 2009). 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; National Research Council 2009). 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 (Uphoff et al. 2024). 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. Previous regression analyses indicated that brackish tributaries have lower L_p under rural conditions (as indicated by different intercepts) than fresh-tidal and separation into salinity classes was warranted. A multiple regression approach that categorized salinity into two classes and separate regressions for each salinity type explained moderate amounts of variation in L_p (Uphoff et al. 2024).

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 our summer habitat assessments begin in early July (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). 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). Significant annual differences were exhibited in amount of DNA per sample for Yellow Perch larvae Chesapeake Bay tributaries during 2014-2016, indicating that maternal influence on size of first-feeding larvae was not constant among years (Uphoff et al. 2017). Estimated RNA/DNA ratios for 6-9 mm (first-feeding) larvae did not indicate consistent differences in larval condition between two watersheds below the target level of development (Nanjemoy Creek, 0.09 C/ha and Choptank River, 0.13 C/ha) and two at or above the threshold (Mattawoman Creek, 0.93 C/ha and Patuxent River, 1.42 C/ha) and the latter two watersheds appeared to be holding their own on L_p and initial feeding success of larvae. Mattawoman Creek was considered a developed treatment but most of its watershed was classified by MD DOP as forested, while Patuxent River was classified as urban; regions adjacent to their larval nurseries were zoned for rural land use. Much of the development in both watersheds occurs upstream along the fluvial region above the larval estuarine nursery and water moves through a more rural region with floodplain swamps before reaching the tidal-fresh larval nursery. Both developed watersheds had 1% or greater of their watershed area in wetlands. While wetland coverage in these two developed watersheds was less than encountered in the rural ones, all four systems had fringing wetlands along the larval nursery region. Patuxent River had two drinking water reservoirs (Tridelphia and Rocky Gorge) upstream of the larval nursery (Uphoff et al. 2017). These features may have mitigated the impact of upstream urban influences.

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 ≥ 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 in the early 2000s, but egg viability declined drastically by then 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 were typically 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.

Watershed development was negatively, and often nonlinearly, linked to organic matter and feeding metrics of 6-9 mm (first-feeding) Yellow Perch larvae in Chesapeake Bay subestuaries (Uphoff et al. 2017). Correlation analyses did not suggest that processes covered in the feeding analysis of 6-9 mm larvae would influence L_p; L_p is not a measure of year-class success and the processes influencing feeding success could impact older larvae and not be detected by L_p surveys (Uphoff et al. 2017). Episodes of hydrologic transport of accumulated OM from watersheds may fuel zooplankton production and feeding success (McClain et al. 2003; Hoffman et al. 2007). Availability of zooplankton prey affects larval fish nutritional condition, growth, size, and survival (Houde 2008).

Annual L_p provides an economical measure of the product of egg production and egg through early postlarval survival. We used L_p as an index to detect “normal” and “abnormal” egg and early larvae dynamics. We considered L_p estimates from subestuaries with suburban to urban watersheds 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. Tighter budgets necessitate development of low-cost indicators of larval survival and relative abundance 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 required laboratory analysis (Uphoff et al. 2017). These latter two analyses represented separate studies rather than a requirement for estimating L_p.

We have relied on correlation and regression analyses to judge the effects of watershed development on Yellow Perch early larval dynamics. 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 influenced 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 (a 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 1963-2024 and data used for regressions with watershed hectares, 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 and 2018 that were closest to a sampling year.

River	Sample Year	LULC Year	Hectares	C / ha	% Ag	% Forest	% Wetland	% Urban	1° Land Use	Salinity	Lp
Bush (w/ APG)	2006	2002	44136	1.17	20.99	36.26	5.55	36.99	Urban	0	0.78
Bush (w/ APG)	2007	2010	44136	1.19	14.94	32.14	5.54	46.44	Urban	0	0.90
Bush (w/ APG)	2008	2010	44136	1.20	14.94	32.14	5.54	46.44	Urban	0	0.69
Bush (w/ APG)	2009	2010	44136	1.21	14.94	32.14	5.54	46.44	Urban	0	0.92
Bush (w/ APG)	2011	2010	44136	1.23	14.94	32.14	5.54	46.44	Urban	0	0.96
Bush (w/ APG)	2012	2010	44136	1.24	14.94	32.14	5.54	46.44	Urban	0	0.34
Bush (w/ APG)	2013	2013	44136	1.25	14.94	32.14	5.54	46.44	Urban	0	0.15
Chester	2019	2018	50276	0.13	66.39	24.46	0.84	8.11	Agriculture	1	0.82
Choptank	1980	1973	110017	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.71
Choptank	1981	1973	110017	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.86
Choptank	1982	1973	110017	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.89
Choptank	1983	1973	110017	0.07	65.24	30.61	1.99	2.11	Agriculture	1	0.32
Choptank	1984	1994	110017	0.07	64.03	29.16	2.3	4.15	Agriculture	1	0.71
Choptank	1985	1994	110017	0.07	64.03	29.16	2.3	4.15	Agriculture	1	1.00
Choptank	1986	1994	110017	0.07	64.03	29.16	2.3	4.15	Agriculture	1	0.73
Choptank	1987	1994	110017	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.75
Choptank	1988	1994	110017	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.70
Choptank	1989	1994	110017	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.64
Choptank	1990	1994	110017	0.08	64.03	29.16	2.3	4.15	Agriculture	1	0.62
Choptank	1998	1997	110017	0.10	63.6	27.72	2.2	6.44	Agriculture	1	0.57
Choptank	1999	1997	110017	0.11	63.6	27.72	2.2	6.44	Agriculture	1	0.60
Choptank	2000	2000	110017	0.11	63.55	27.5	2.12	6.79	Agriculture	1	0.19
Choptank	2001	2000	110017	0.11	63.55	27.5	2.12	6.79	Agriculture	1	0.25
Choptank	2002	2002	110017	0.11	63.85	27.14	2.02	6.94	Agriculture	1	0.32
Choptank	2003	2002	110017	0.11	63.85	27.14	2.02	6.94	Agriculture	1	0.54

Table 2.1 (continued)

River	Sample Year	LULC Year	Hectares	C / ha	% Ag	% Forest	% Wetland	% Urban	1° Land Use	Salinity	Lp
Choptank	2004	2002	110017	0.12	63.85	27.14	2.02	6.94	Agriculture	1	0.50
Choptank	2013	2013	110017	0.13	61.02	25.58	2.11	11.19	Agriculture	1	0.58
Choptank	2014	2013	110017	0.13	61.02	25.58	2.11	11.19	Agriculture	1	0.68
Choptank	2015	2013	110017	0.13	61.02	25.58	2.11	11.19	Agriculture	1	0.81
Choptank	2016	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.59
Choptank	2017	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.43
Choptank	2018	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.44
Choptank	2019	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.68
Choptank	2021	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.44
Choptank	2022	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.46
Choptank	2023	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.64
Choptank	2024	2018	110017	0.13	60.72	25.57	2.09	11.73	Agriculture	1	0.61
Corsica	2006	2002	9676	0.21	64.32	27.36	0.43	7.88	Agriculture	1	0.47
Corsica	2007	2010	9676	0.22	60.37	25.51	0.39	13.16	Agriculture	1	0.83
Elk	2010	2010	21040	0.59	28.02	38.7	1.13	31.15	Forest	0	0.75
Elk	2011	2010	21040	0.59	28.02	38.7	1.13	31.15	Forest	0	0.79
Elk	2012	2010	21040	0.60	28.02	38.7	1.13	31.15	Forest	0	0.66
Langford	2007	2010	9641	0.07	70.19	20.35	1.46	7.97	Agriculture	1	0.54
Magothy	2009	2010	9205	2.74	1.24	21.03	0.01	76.77	Urban	1	0.10
Magothy	2016	2018	9205	2.86	1.2	20.42	0.01	77.87	Urban	1	0.10
Mattawoman	2008	2010	24430	0.87	9.33	53.88	1.13	34.18	Forest	0	0.58
Mattawoman	2009	2010	24430	0.88	9.33	53.88	1.13	34.18	Forest	0	0.90
Mattawoman	2010	2010	24430	0.90	9.33	53.88	1.13	34.18	Forest	0	0.82
Mattawoman	2011	2010	24430	0.91	9.33	53.88	1.13	34.18	Forest	0	0.92
Mattawoman	2012	2010	24430	0.90	9.33	53.88	1.13	34.18	Forest	0	0.20
Mattawoman	2013	2013	24430	0.92	9.33	53.88	1.13	34.18	Forest	0	0.64
Mattawoman	2014	2013	24430	0.93	9.33	53.88	1.13	34.18	Forest	0	0.67
Mattawoman	2015	2013	24430	0.94	9.33	53.88	1.13	34.18	Forest	0	1.00

Table 2.1 (continued)

River	Sample Year	LULC Year	Hectares	C / ha	% Ag	% Forest	% Wetland	% Urban	1° Land Use	Salinity	Lp
Mattawoman	2016	2018	24430	0.96	8.63	52.83	1.14	35.65	Forest	0	0.90
Mattawoman	2023	2018	24430	1.03	8.63	52.83	1.14	35.65	Forest	0	0.68
Middle	2012	2010	2753	3.33	3.41	23.32	2.12	70.98	Urban	0	0.00
Nanjemoy	2009	2010	18891	0.09	12.38	68.7	4.09	14.74	Forest	1	0.74
Nanjemoy	2010	2010	18891	0.09	12.38	68.7	4.09	14.74	Forest	1	0.90
Nanjemoy	2011	2010	18891	0.09	12.38	68.7	4.09	14.74	Forest	1	0.92
Nanjemoy	2012	2010	18891	0.09	12.38	68.7	4.09	14.74	Forest	1	0.03
Nanjemoy	2013	2013	18891	0.09	12.38	68.7	4.09	14.74	Forest	1	0.52
Nanjemoy	2014	2013	18891	0.09	12.38	68.7	4.09	14.74	Forest	1	0.88
Nanticoke	1963	1973	71401	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.65
Nanticoke	1964	1973	71401	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.50
Nanticoke	1965	1973	71401	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.34
Nanticoke	1966	1973	71401	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.39
Nanticoke	1967	1973	71401	0.05	46.57	43.38	8.06	1.92	Agriculture	1	0.29
Nanticoke	1968	1973	71401	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.40
Nanticoke	1970	1973	71401	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.65
Nanticoke	1971	1973	71401	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.24
Nanticoke	1972	1973	71401	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.26
Nanticoke	1973	1973	71401	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.53
Nanticoke	1974	1973	71401	0.06	46.57	43.38	8.06	1.92	Agriculture	1	0.35
Nanticoke	1975	1973	71401	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.48
Nanticoke	1976	1973	71401	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.30
Nanticoke	1977	1973	71401	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.72
Nanticoke	1979	1973	71401	0.07	46.57	43.38	8.06	1.92	Agriculture	1	0.30
Nanticoke	1981	1973	71401	0.08	46.57	43.38	8.06	1.92	Agriculture	1	0.39
Nanticoke	2004	2002	71401	0.11	46.3	40.73	7.4	5.54	Agriculture	1	0.49
Nanticoke	2005	2002	71401	0.11	46.3	40.73	7.4	5.54	Agriculture	1	0.44
Nanticoke	2006	2002	71401	0.11	46.3	40.73	7.4	5.54	Agriculture	1	0.35
Nanticoke	2007	2010	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.69

Table 2.1 (continued).

River	Sample Year	LULC Year	Hectares	C / ha	% Ag	% Forest	% Wetland	% Urban	1° Land Use	Salinity	Lp
Nanticoke	2008	2010	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.11
Nanticoke	2009	2010	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.32
Nanticoke	2010	2010	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.39
Nanticoke	2011	2010	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.55
Nanticoke	2012	2010	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.04
Nanticoke	2013	2013	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.48
Nanticoke	2014	2013	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.35
Nanticoke	2015	2013	71401	0.11	45.02	39.4	7.36	8.08	Agriculture	1	0.59
Nanticoke	2016	2018	71401	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.38
Nanticoke	2017	2018	71401	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.22
Nanticoke	2018	2018	71401	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.28
Nanticoke	2019	2018	71401	0.11	44.56	39.6	7.29	8.37	Agriculture	1	0.41
Northeast	2010	2010	16342	0.46	31.08	38.65	0.11	28.86	Forest	0	0.68
Northeast	2011	2010	16342	0.46	31.08	38.65	0.11	28.86	Forest	0	1.00
Northeast	2012	2010	16342	0.47	31.08	38.65	0.11	28.86	Forest	0	0.66
Northeast	2013	2013	16342	0.48	31.08	38.65	0.11	28.86	Forest	0	0.72
Northeast	2014	2013	16342	0.48	31.08	38.65	0.11	28.86	Forest	0	0.77
Patuxent	2015	2013	170644	1.24	20.51	35.07	1.02	41.67	Urban	1	0.74
Patuxent	2016	2018	170644	1.25	20.21	33.98	1.07	43.17	Urban	1	0.72
Piscataway	2008	2010	17634	1.41	9.98	40.37	0.24	47.01	Urban	0	0.41
Piscataway	2009	2010	17634	1.43	9.98	40.37	0.24	47.01	Urban	0	0.39
Piscataway	2010	2010	17634	1.45	9.98	40.37	0.24	47.01	Urban	0	0.54
Piscataway	2011	2010	17634	1.46	9.98	40.37	0.24	47.01	Urban	0	0.59
Piscataway	2012	2010	17634	1.47	9.98	40.37	0.24	47.01	Urban	0	0.18
Piscataway	2013	2013	17634	1.50	9.98	40.37	0.24	47.01	Urban	0	0.59
Sassafras	2021	2018	19580	0.11	63.98	25.8	1.28	8.55	Agriculture	0	0.60
Sassafras	2022	2018	19580	0.11	63.98	25.8	1.28	8.55	Agriculture	0	0.82
Severn	2002	2002	17937	2.02	8.57	35.18	0.18	55.84	Urban	1	0.16
Severn	2004	2002	17937	2.09	8.57	35.18	0.18	55.84	Urban	1	0.35

Table 2.1 (continued).

River	Sample Year	LULC Year	Hectares	C / ha	% Ag	% Forest	% Wetland	% Urban	1° Land Use	Salinity	Lp
Severn	2005	2002	17937	2.15	8.57	35.18	0.18	55.84	Urban	1	0.40
Severn	2006	2002	17937	2.18	8.57	35.18	0.18	55.84	Urban	1	0.24
Severn	2007	2010	17937	2.21	4.97	27.97	0.2	65.07	Urban	1	0.35
Severn	2008	2010	17937	2.24	4.97	27.97	0.2	65.07	Urban	1	0.08
Severn	2009	2010	17937	2.25	4.97	27.97	0.2	65.07	Urban	1	0.13
Severn	2010	2010	17937	2.26	4.97	27.97	0.2	65.07	Urban	1	0.03
South	2008	2010	14773	1.32	10.24	39.15	0.47	48.82	Urban	1	0.12
Wicomico (ES)	2017	2018	41352	0.69	29.07	37.68	2.03	30.68	Forest	1	0.46
Wicomico (ES)	2018	2018	41352	0.69	29.07	37.68	2.03	30.68	Forest	1	0.34

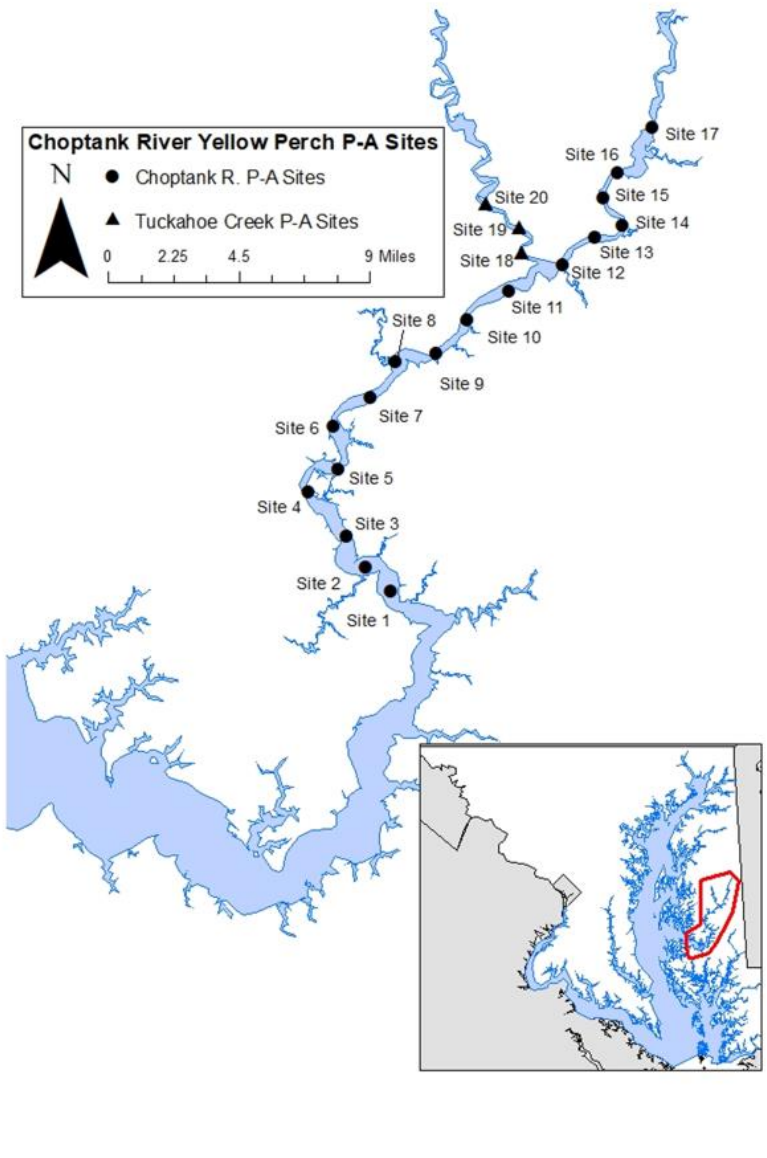
Table 2.2. Sampling summary for L_p surveys. APG = Aberdeen Proving Ground. ES = eastern shore of Chesapeake Bay. Unshaded entries indicate Yellow Perch larvae were sampled by 0.5 m diameter cone shaped plankton nets; shading indicates plankton trawls were used. Start and End indicate span of dates used to estimate L_p.

River	Sample Year	Lp	SD	N_total	Start	End
Bush (w/ APG)	2006	0.78	0.05	70	March 29	April 19
Bush (w/ APG)	2007	0.90	0.04	50	April 2	April 24
Bush (w/ APG)	2008	0.69	0.06	68	March 27	April 24
Bush (w/ APG)	2009	0.92	0.04	40	April 2	April 24
Bush (w/ APG)	2011	0.96	0.02	79	March 28	April 26
Bush (w/ APG)	2012	0.34	0.07	50	April 3	April 17
Bush (w/ APG)	2013	0.15	0.06	39	April 4	April 25
Chester	2019	0.82	0.06	44	April 3	April 17
Choptank	1980	0.71	0.08	35	April 10	April 24
Choptank	1981	0.86	0.05	52	April 2	April 28
Choptank	1982	0.89	0.05	44	April 12	May 3
Choptank	1983	0.32	0.07	44	April 5	May 2
Choptank	1984	0.71	0.06	56	April 9	May 1
Choptank	1985	1.00	.	28	April 9	April 22
Choptank	1986	0.73	0.05	66	April 7	May 1
Choptank	1987	0.75	0.05	68	April 13	May 11
Choptank	1988	0.70	0.07	44	April 11	May 9
Choptank	1989	0.64	0.06	74	April 4	April 27
Choptank	1990	0.62	0.06	64	April 9	April 25
Choptank	1998	0.57	0.06	70	April 8	April 29
Choptank	1999	0.60	0.05	100	April 7	May 7
Choptank	2000	0.19	0.04	100	April 3	May 5
Choptank	2001	0.25	0.07	40	March 30	April 23
Choptank	2002	0.32	0.08	38	April 8	April 15
Choptank	2003	0.54	0.05	90	April 7	April 29
Choptank	2004	0.50	0.08	40	April 8	April 21
Choptank	2013	0.58	0.07	50	April 2	April 18
Choptank	2014	0.68	0.06	56	April 8	April 25
Choptank	2015	0.81	0.06	48	April 7	April 22
Choptank	2016	0.59	0.05	90	March 28	April 25
Choptank	2017	0.43	0.05	90	March 21	April 17
Choptank	2018	0.44	0.05	99	March 27	May 1
Choptank	2019	0.68	0.06	60	April 2	April 18
Choptank	2021	0.44	0.06	70	March 25	April 20
Choptank	2022	0.46	0.06	74	March 22	April 18
Choptank	2023	0.67	0.08	32	March 21	April 7
Choptank	2024	0.61	0.06	70	March 21	April 10
Corsica	2006	0.47	0.06	60	March 28	April 13
Corsica	2007	0.83	0.05	59	April 3	April 25
Elk	2010	0.75	0.07	36	April 6	April 29
Elk	2011	0.79	0.06	39	April 8	April 22
Elk	2012	0.66	0.07	50	March 22	April 17
Langford	2007	0.54	0.07	56	April 3	April 25

Magothy	2009	0.10	0.05	40	March 30	April 17
Magothy	2016	0.10	0.05	41	March 23	April 13
Mattawoman	1990				March 21	April 18
Mattawoman	2008	0.58	0.06	80	March 25	April 17
Mattawoman	2009	0.90	0.03	80	March 31	April 23
Mattawoman	2010	0.82	0.05	50	March 31	April 15
Mattawoman	2011	0.92	0.03	80	March 29	April 21
Mattawoman	2012	0.20	0.05	60	March 27	April 17
Mattawoman	2013	0.64	0.07	50	April 1	April 15
Mattawoman	2014	0.67	0.06	70	April 7	April 28
Mattawoman	2015	1.00	.	40	April 14	April 29
Mattawoman	2016	0.90	0.06	29	April 4	April 17
Mattawoman	2023	0.68	0.08	34	March 27	April 11
Middle	2012	0.00	.	30	March 28	April 9
Nanjemoy	2009	0.74	0.05	70	March 31	April 22
Nanjemoy	2010	0.9	0.04	60	March 31	April 22
Nanjemoy	2011	0.92	0.03	80	March 29	April 21
Nanjemoy	2012	0.03	0.02	60	March 27	April 17
Nanjemoy	2013	0.52	0.08	40	April 3	April 15
Nanjemoy	2014	0.88	0.04	70	April 7	April 28
Nanticoke	1963	0.65	0.05	81	April 1	May 6
Nanticoke	1964	0.50	0.05	92	March 26	May 9
Nanticoke	1965	0.34	0.06	64	April 5	May 5
Nanticoke	1966	0.36	0.07	44	April 6	April 26
Nanticoke	1967	0.29	0.08	31	April 7	May 5
Nanticoke	1968	0.40	0.07	47	April 2	May 4
Nanticoke	1970	0.65	0.10	23	April 16	April 29
Nanticoke	1971	0.24	0.07	34	April 9	May 1
Nanticoke	1972	0.26	0.08	31	April 8	May 2
Nanticoke	1973	0.53	0.09	30	March 30	April 23
Nanticoke	1974	0.35	0.08	31	April 3	April 29
Nanticoke	1975	0.48	0.09	33	April 7	May 5
Nanticoke	1976	0.30	0.09	27	March 30	April 23
Nanticoke	1977	0.72	0.11	18	April 4	April 19
Nanticoke	1979	0.30	0.08	33	April 5	May 3
Nanticoke	1981	0.39	0.08	33	April 8	May 4
Nanticoke	2004	0.49	0.07	49	April 6	April 20
Nanticoke	2005	0.44	0.07	54	April 8	April 25
Nanticoke	2006	0.35	0.09	31	April 3	April 17
Nanticoke	2007	0.69	0.06	68	April 2	April 27
Nanticoke	2008	0.11	0.03	99	March 24	April 25
Nanticoke	2009	0.32	0.06	65	March 31	April 23
Nanticoke	2010	0.39	0.06	59	April 1	April 23
Nanticoke	2011	0.55	0.07	56	April 6	April 25
Nanticoke	2012	0.04	0.02	56	April 2	April 20
Nanticoke	2013	0.48	0.08	40	April 8	April 19
Nanticoke	2014	0.35	0.07	52	April 7	April 25
Nanticoke	2015	0.59	0.06	56	April 7	April 23
Nanticoke	2016	0.38	0.07	50	April 4	April 26

Nanticoke	2017	0.22	0.07	32	April 3	April 18
Nanticoke	2018	0.28	0.08	32	April 10	April 30
Nanticoke	2019	0.41	0.08	39	April 5	April 16
Northeast	2010	0.68	0.09	25	April 6	April 29
Northeast	2011	1.00	.	49	March 29	April 22
Northeast	2012	0.66	0.06	70	March 22	April 17
Northeast	2013	0.72	0.07	40	April 9	April 30
Northeast	2014	0.77	0.05	70	April 9	May 9
Patuxent	2015	0.74	0.07	38	April 8	April 21
Patuxent	2016	0.72	0.05	71	March 30	April 19
Piscataway	2008	0.41	0.07	54	March 25	April 17
Piscataway	2009	0.39	0.08	33	March 31	April 16
Piscataway	2010	0.54	0.08	35	March 31	April 22
Piscataway	2011	0.59	0.06	56	March 29	April 21
Piscataway	2012	0.18	0.06	39	March 27	April 17
Piscataway	2013	0.59	0.10	22	April 3	April 15
Sassafras	2021	0.60	0.07	50	April 5	April 19
Sassafras	2022	0.82	0.05	60	March 30	April 22
Severn	2002	0.16				
Severn	2004	0.35				
Severn	2005	0.40	0.06	60	April 6	April 20
Severn	2006	0.24	0.05	80	March 28	April 21
Severn	2007	0.35	0.06	60	March 30	April 20
Severn	2008	0.08	0.04	50	April 3	April 18
Severn	2009	0.13	0.04	70	March 31	April 21
Severn	2010	0.03	0.03	30	April 8	April 18
South	2008	0.12	0.04	60	March 31	April 16
Wicomico (ES)	2017	0.46	0.06	80	March 22	April 13
Wicomico (ES)	2018	0.34	0.04	110	March 26	April 30

Figure 2.1. Location of Choptank River stations sampled for larval Yellow Perch presence-absence. Inset shows the location of the Choptank River in Chesapeake Bay.



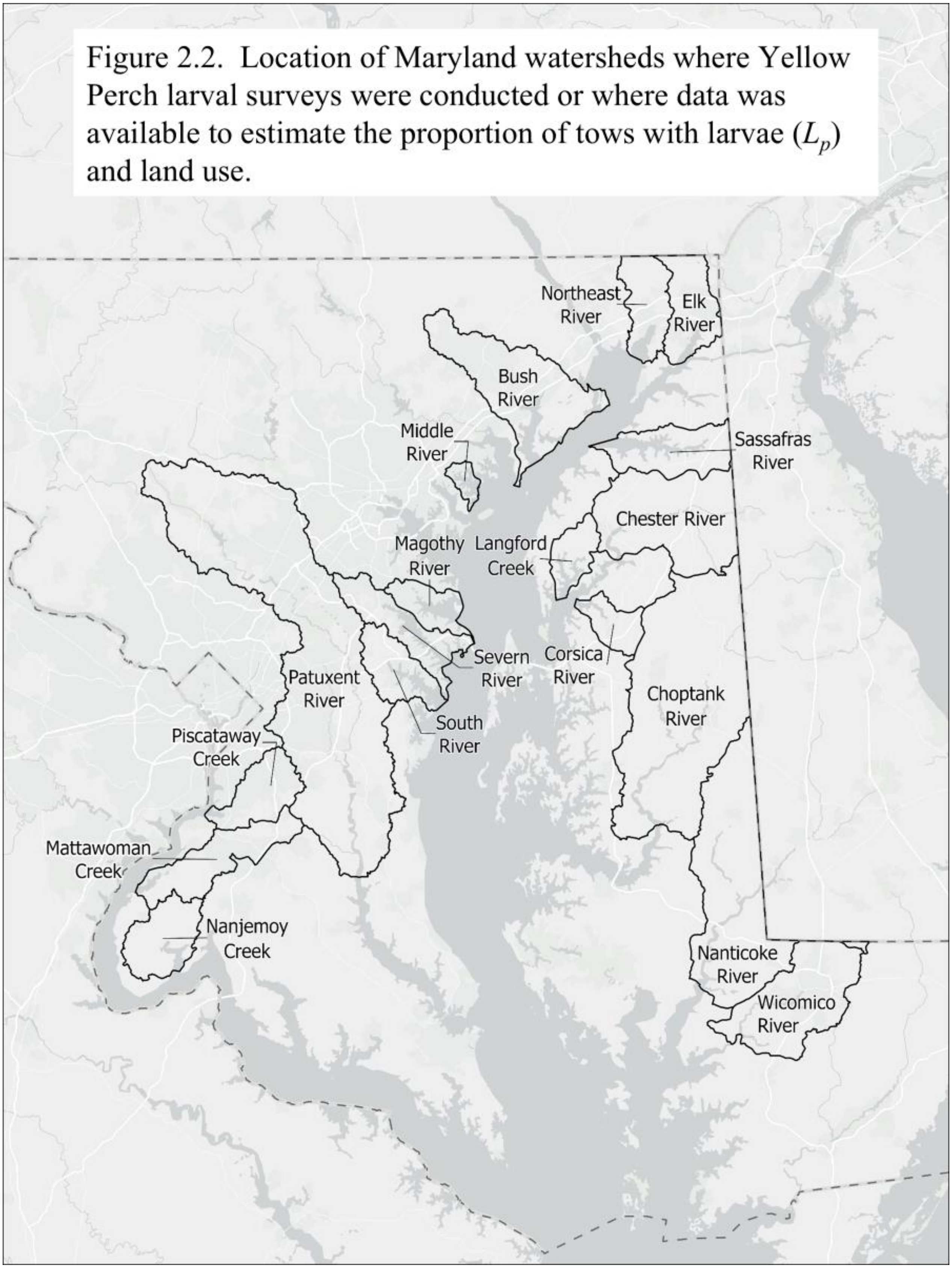


Figure 2.2. Location of Maryland watersheds where Yellow Perch larval surveys were conducted or where data was available to estimate the proportion of tows with larvae (L_p) and land use.

Figure 2.3. Proportion of tows with Yellow Perch larvae (Lp) by year sampled with 0.5 m plankton nets for brackish subestuaries, 1963-2024. Symbols indicate primary watershed land use: gold = agriculture; green = forest; and red = urban. Thick borders indicate watersheds > 73,000 ha (large watersheds).

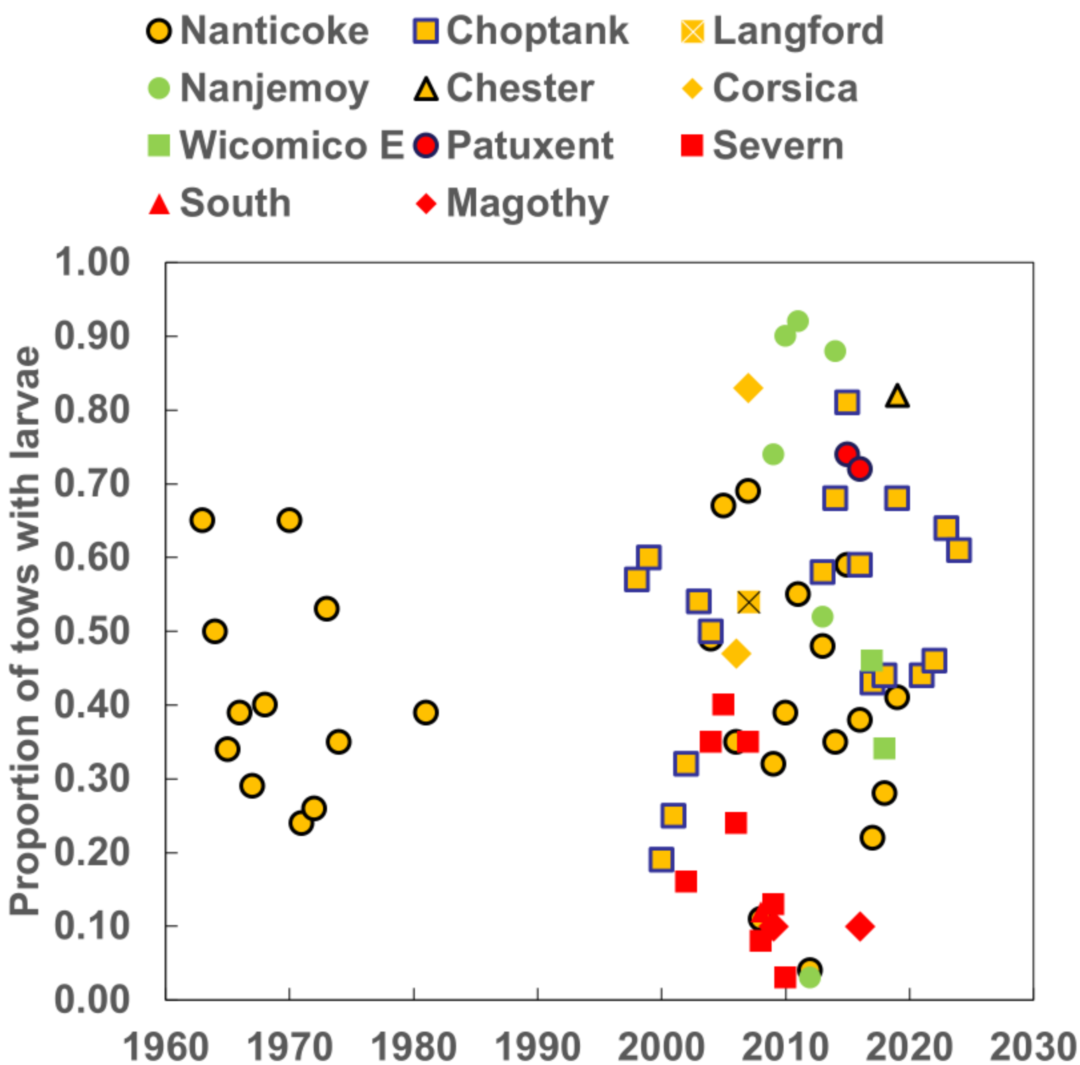


Figure 2.4. Proportion of tows with Yellow Perch larvae plotted against structures per hectare (development intensity) with primary land use and salinity type (0 = tidal-fresh, ≤ 2 ‰; and 1 = brackish, > 2 ‰) indicated for all surveys since 1963.

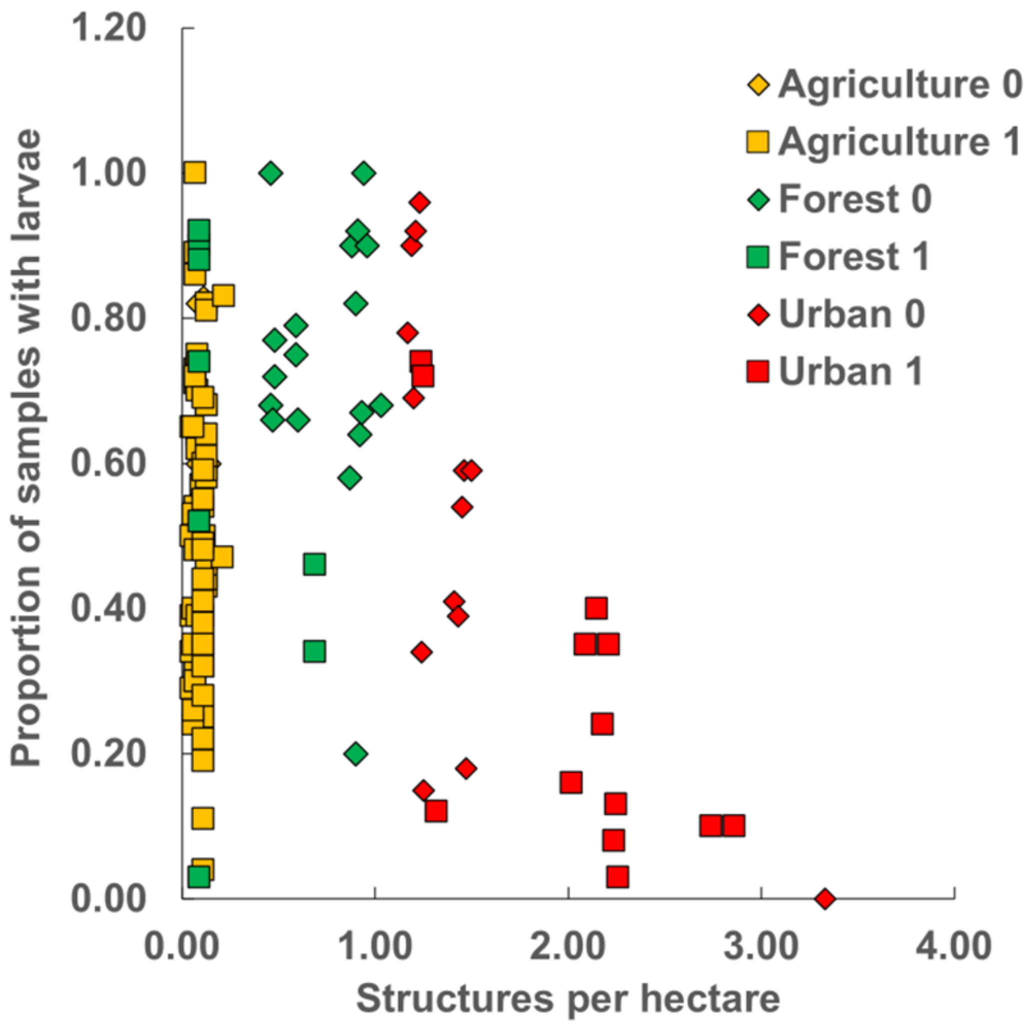


Figure 2.5. Water temperature and dissolved oxygen measurements during the period that proportion of tows with Yellow Perch larvae was estimated during 2024 in Choptank River.

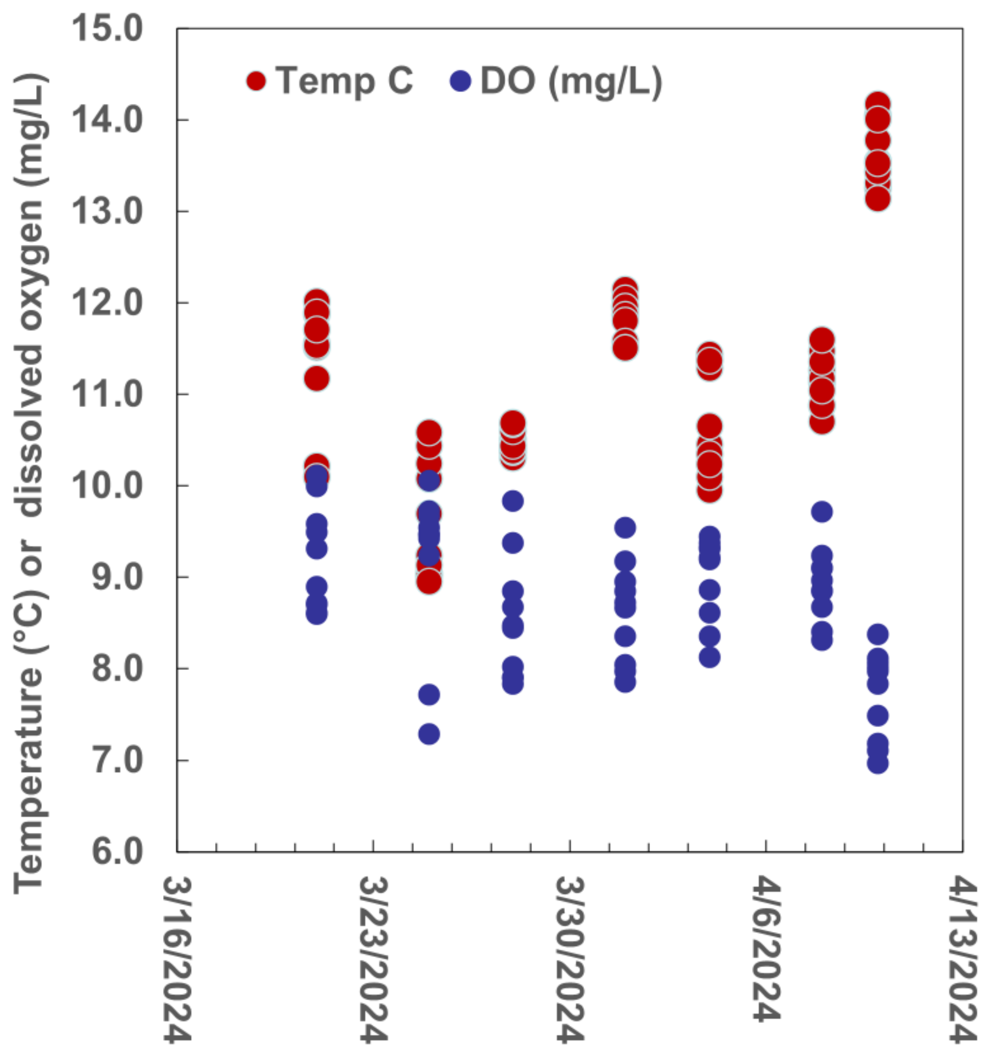


Figure 2.6. Conductivity and pH measurements during the period that proportion of tows with Yellow Perch larvae was estimated during 2024 in Choptank River.

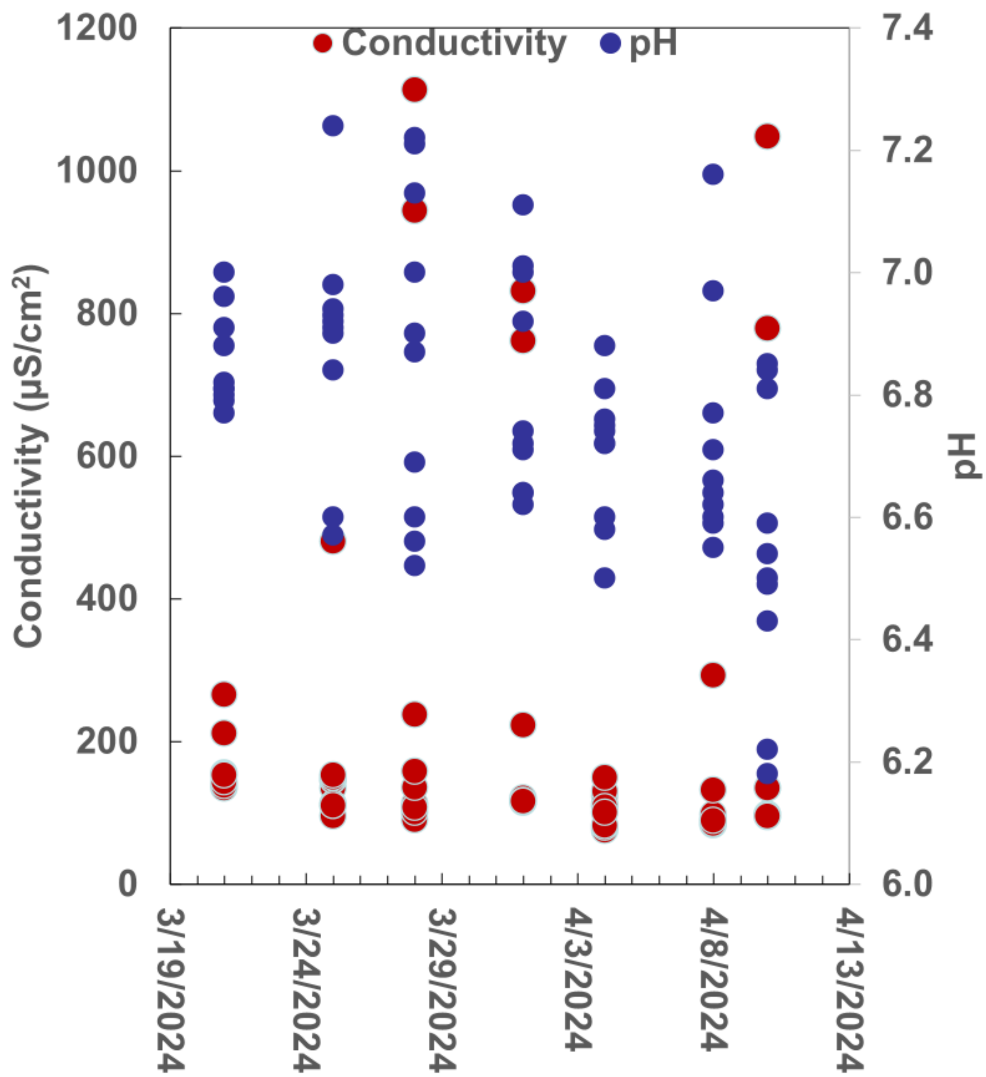


Figure 2.7. Winter (December-February) mean air temperature (˚C) trend at Baltimore, Maryland, during 1963-2024.

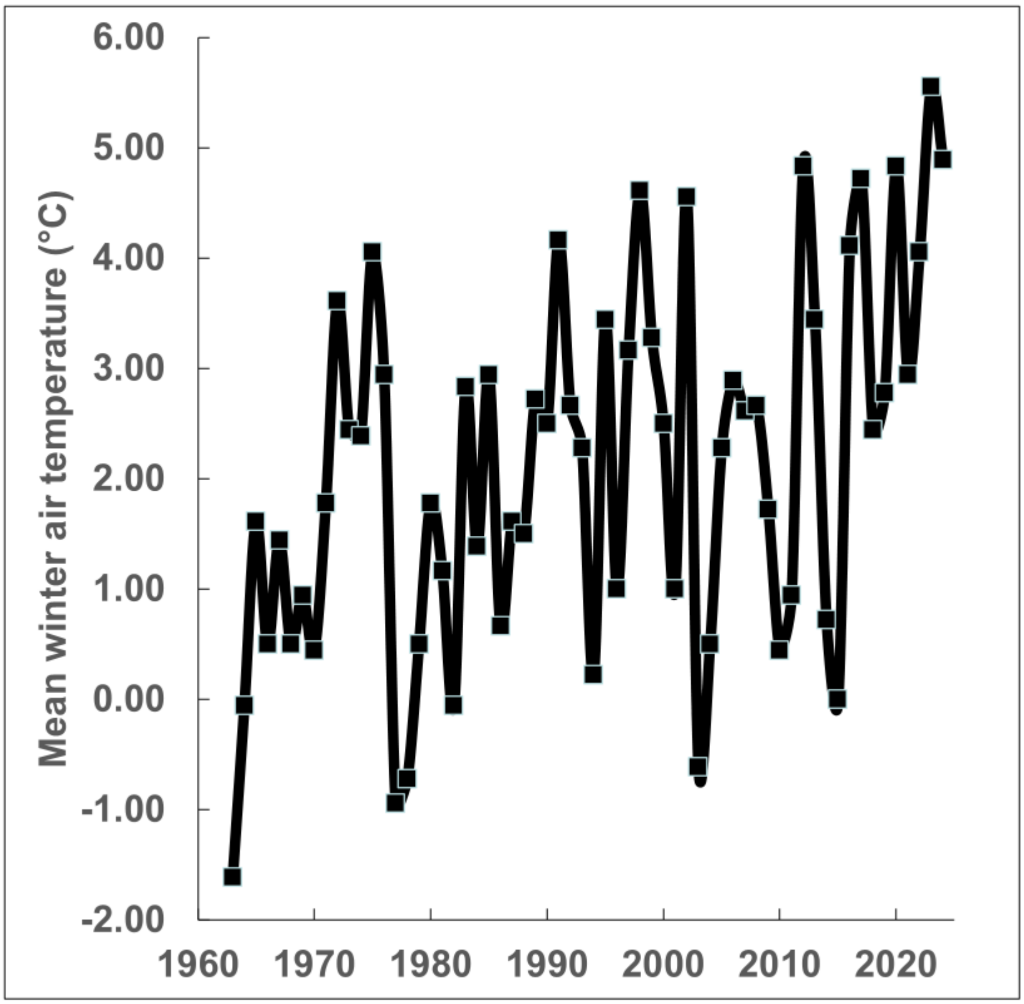


Figure 2.8. Linear regression of proportion of tows with Yellow Perch larvae in Nanticoke River versus mean winter temperature (December-February) at Baltimore, MD.

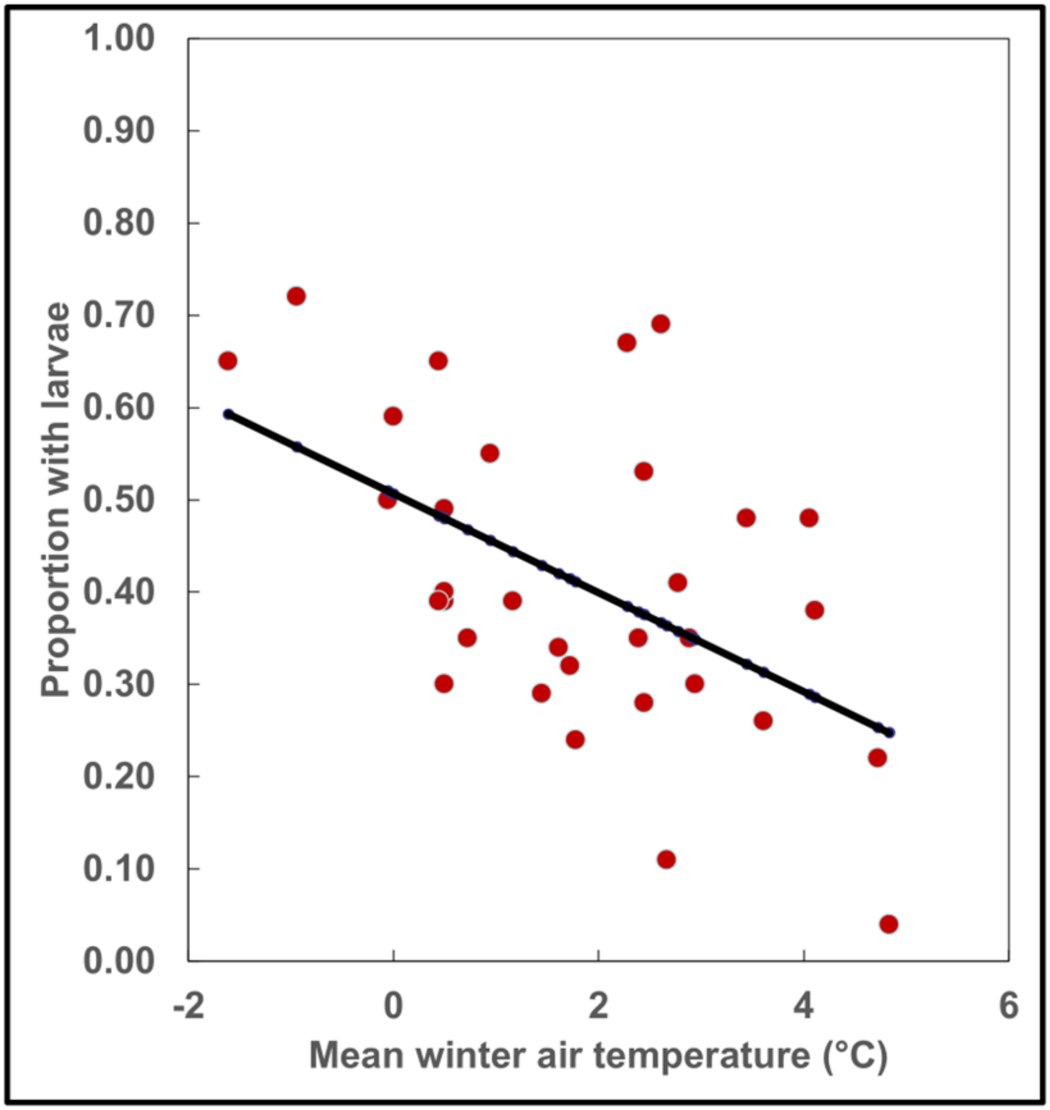


Figure 2.9. Linear regression of proportion of tows with Yellow Perch larvae in Nanticoke River and Choptank River standardized to their time-series medians versus mean winter temperature (December-February) at Baltimore, MD.

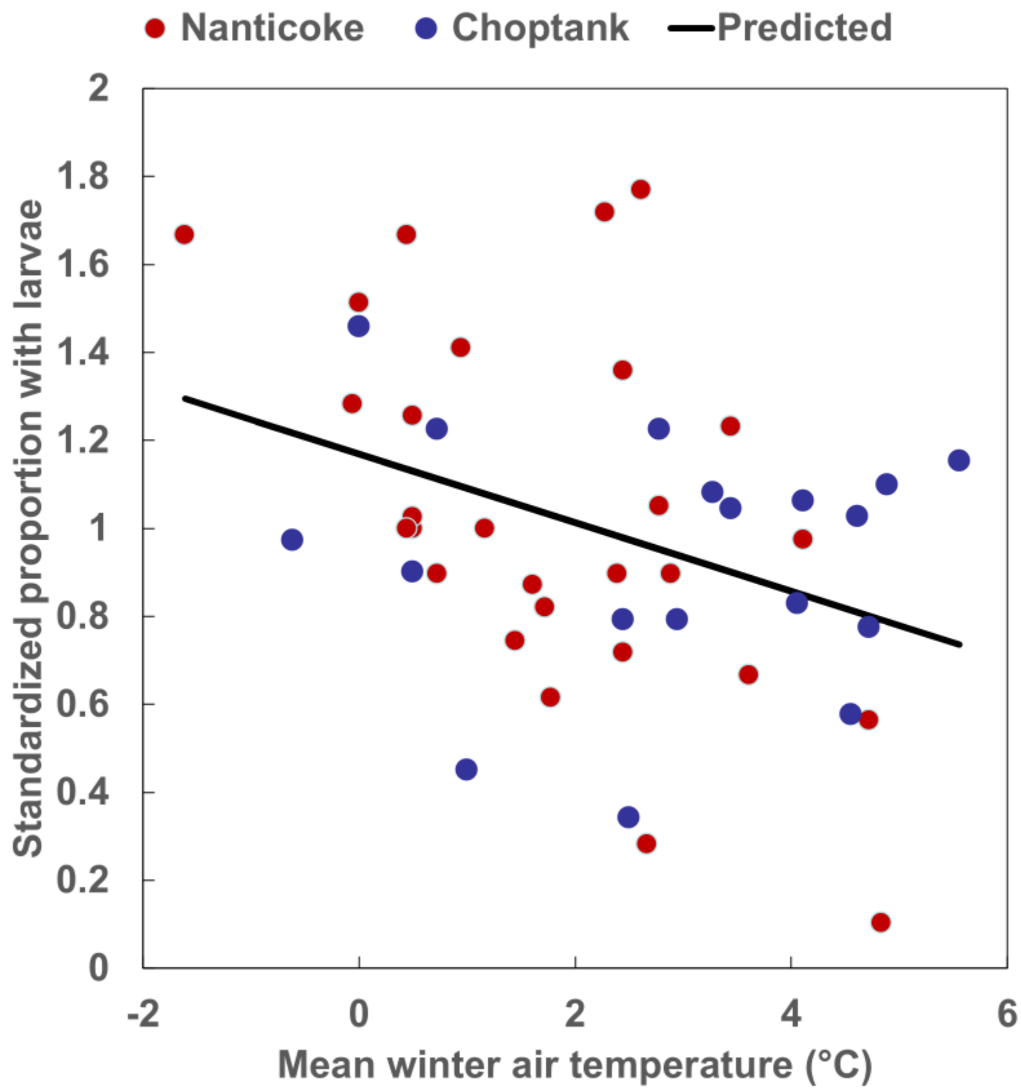


Figure 2.10. Mean winter temperature and date to reach 18°C (April 1 = 0) in Choptank River and Nanticoke River estuarine Yellow Perch nursery, 1963-2024. Linear model prediction = black line and temperature squared model prediction = dark red dots.

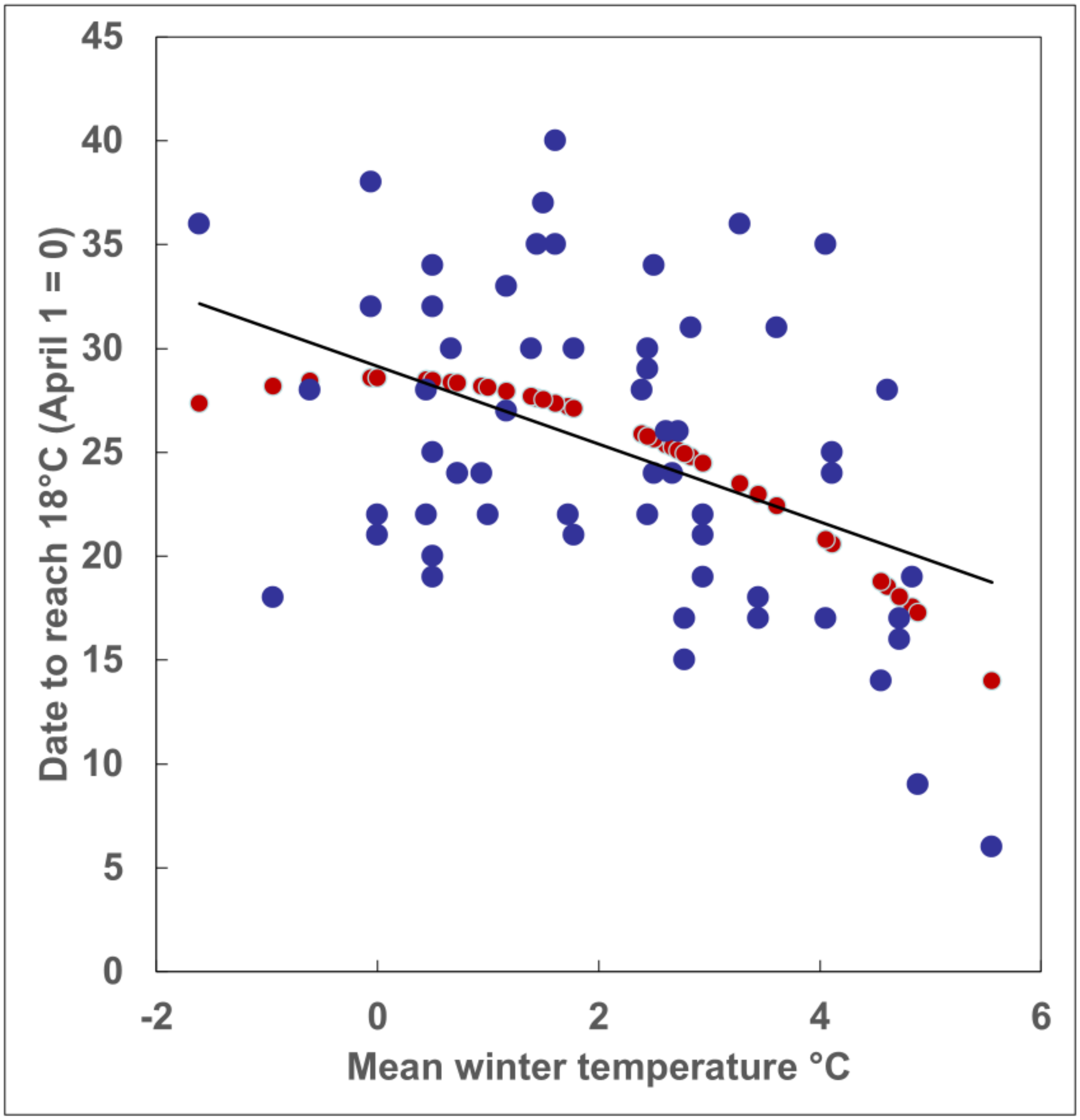
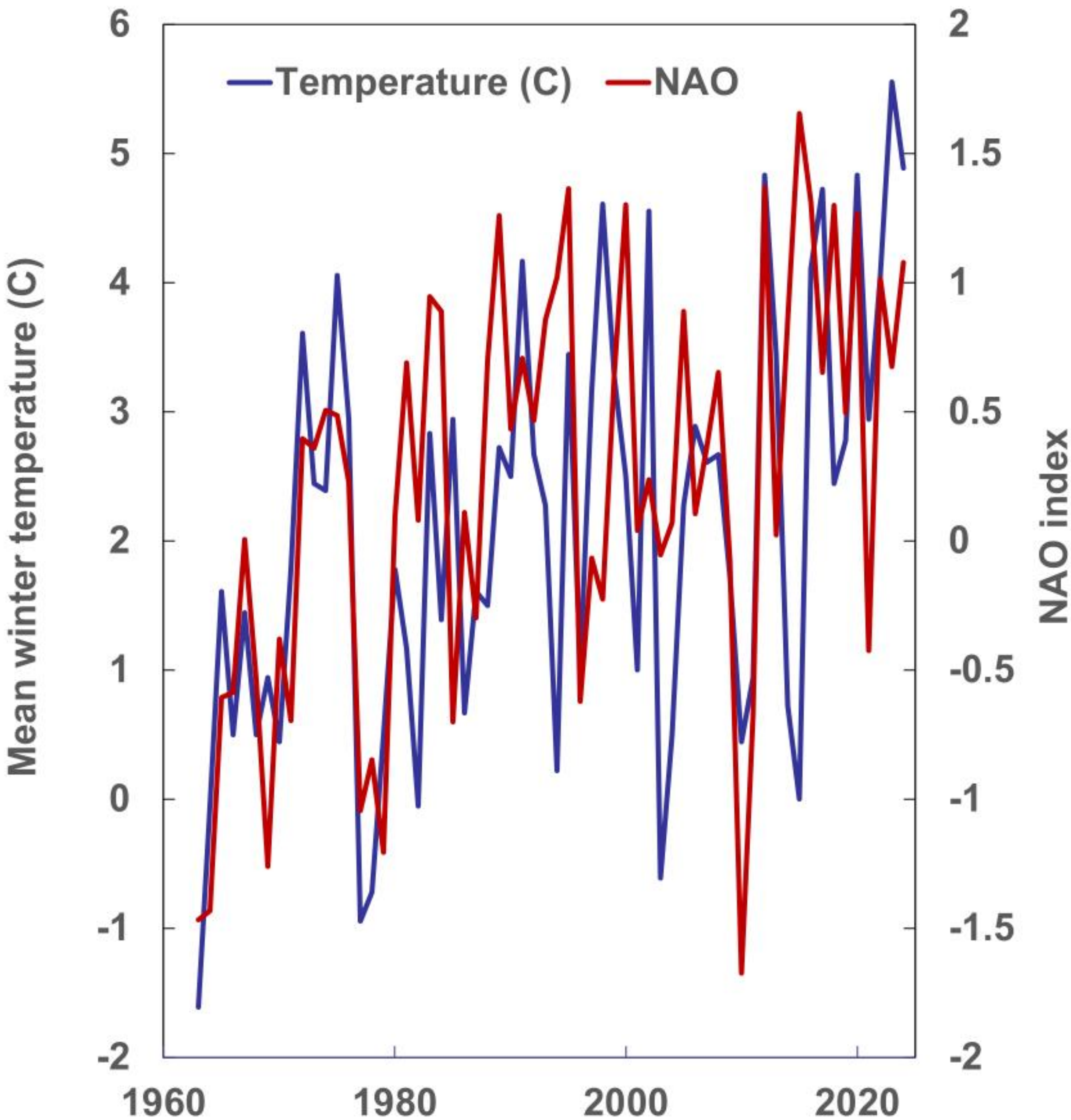


Figure 2.11. Trends in winter temperature in Baltimore (regional indicator of winter conditions) and the North Atlantic Oscillation index (NAO) during 1963-2024.



MD – Marine and estuarine finfish ecological and habitat investigations
Project 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern
Section 2.1: Investigation of Striped Bass spawning and larval habitat status in Maryland

Jim Uphoff, Alexis Park, Shannon Moorhead, and Marisa Ponte

Introduction

An overfishing declaration and successive poor year-classes of Striped Bass in Maryland spawning areas during 2019-2024 have generated concern in the fisheries management and angling community. Unease has been expressed about degradation of Striped Bass spawning and larval nursery habitat in Chesapeake Bay. We have assembled historical data and oriented some of our spring monitoring to respond to these concerns. This report updates efforts begun in the last five annual reports (Uphoff et al. 2020; 2022a; 2022b; 2023; 2024) to assess spawning and larval habitat. These reports provide extensive background for this report and Uphoff et al. (2022a) provide detail on the data set assembled for those analyses and this report.

Although much management effort has focused on the abundance of spawning stock, 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 (Ulanowicz and Polgar 1980; Uphoff 1989; 1993; Houde 1996; Maryland Sea Grant 2009; Shideler and Houde 2014; Martino and Houde 2010; Secor et al. 2017; Uphoff 2023). Other spawning stock attributes such as age structure, spatial and temporal dispersion, behavior, body size, condition, and maternal age may be more influential for a broad array of stocks, including Striped Bass (Secor 2000a; 2000b; Berkeley et al. 2004; Petitgas et al. 2010; Marshall 2016; Barneche et al. 2018; Uphoff 2023). Spawning and larval nursery habitat (both are basically the same) are 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) with cooler and wetter winters and springs considered 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). 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; 2024).

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 its four largest 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 2024) 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 it is a strong indicator of the future 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 2024). 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 2019-2024 (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 (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 during 2021-2023. 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 Striped Bass egg and larval 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).

We 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 update temperature patterns through 2024 for Choptank River.

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. High 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 (<14ºC) and late (>21ºC) 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).

We investigated the influence of winter temperature conditions on temperature milestones. We used summarized average winter air temperatures (December-February) at Baltimore as an indicator of winter intensity to investigate their relationship with milestone date estimates for the Nanticoke River and Choptank River. Recent and ongoing work on environmental influences by the Striped Bass Program have indicated an influence of winter intensity on the MD JI using the same air temperature data (S. Brown, MD DNR, personal communication) and we wanted to examine whether a connection was possible between spawning temperature milestones and winter intensity changes. Austin (2002) suggested that positive correlations of Lowess smoothed water temperatures in the lower Bay (Virginia Institute of Marine Science pier, 1960-2000) and the North Atlantic Oscillation (NAO) indicated coherence of low frequency trends. The NAO indexes wind balances of the northwest Atlantic Ocean; a strong NAO in winter results in a strong westerly flow that drains off cold Canadian air (Austin 2002). We created a winter NAO index (tabulated monthly indices from the National Weather Service's Climate Prediction Center website: https://www.cpc.ncep.noaa.gov/products/precip/CWlink/pna/nao_index.html) as the mean of December (year t) through February (year t+1) monthly NAO indices that matched timing of our winter air temperature indicator of regional winter intensity. We used correlation analysis to determine the strength of the association of regional winter intensity and the NAO during 1954-2024.

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). Gross et al. (2022) proposed a "poor recruitment paradigm" for Chesapeake Bay Striped Bass based on the consistency of poor juvenile indices with below average flow in seven spawning areas.

Under F-63, 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; 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 have been minimized during 1993-2020 (Uphoff et al. 2022b).

We updated the following metrics developed in Uphoff et al. (2020; 2022b) through 2024 in this report: *Ep*, JI, RLS, temperature, DO, pH, salinity, and conductivity. We updated the occurrence of spawning temperature milestones and flow patterns in the four major spawning areas. We implemented an egg volume index to better define spawning intensity during 2024. We deployed loggers at two stations that recorded water temperature every 30 minutes to fill in gaps between surveys.

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) 2024 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. 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 added to 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 2024, 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) in the last several years that were sampled during 1987-1990 because of shallow depths. 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 (dead or alive) were readily seen in a sample during or after processing, the sample was discarded, and presence of eggs was recorded. Dead eggs could consist of chorions (clear or with yolk or oil), partially intact (not spherical), or intact eggs that were cloudy. 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 1994-2024 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 2.0‰ (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).

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).

Spawning intensity - In lieu of counting, we assigned a rank to egg volume in a sample jar to better discriminate spawning intensity: 0 = eggs not detected; 1 = one to a few scattered eggs; 2 = a layer up to 25%; 3 = 25% to 50%; and 4 = 50% or more of the jar. Ranks for each sample collected on a date were averaged as an index of intensity for that day. We compared the daily intensity indices to water temperature measurements from two continuous temperature loggers (described below).

Juvenile index 2024 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 2024). 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 2024). 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 2024).

Relative Larval Survival (RLS) update - We used the JI and baywide *Ep* to estimate annual relative larval survival (RLS) during 1957-2024 as:

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

Estimates of the JI concurrent with Ep were available. The baywide *Ep* time-series started in 1955 and continued through 2024; 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 –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 in Choptank River during 2014-2024 (Uphoff 2023). The meter was calibrated frequently. The Choptank River is turbulent and did not show signs of stratification during 1983-1991 surveys when surface, mid-depth, and bottom measurements or measurements at 2-m increments were taken (Uphoff 1992), so surface measurements should have been comparable to those at multiple depths (Uphoff 2023).

During 2021-2023, total alkalinity (mg/L $CaCO_3$), was measured in Choptank River using a YSI 9500 Photometer (Uphoff et al. 2024). Alkalinity was not sampled in 2024.

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 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 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). These estimates were made from survey data to maintain continuity with past surveys. Correlation analysis was used to explore associations among temperature, DO and pH during this period. 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.

We examined four spawning milestones from survey data 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 of or all 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 years available varied among milestones.

We determined the median dates for each water temperature milestone during 1957-1999 and compared them to milestone dates during 2000-2024. We determined milestone dates for years with strong year-classes for the Choptank and Nanticoke rivers. Strong year-classes were defined as the 75th percentile for their respective 1957-2009 time-series.

Surveys from the Nanticoke River during 1954-1981, 1985, 1989, 1992-1994, 2004-2019, and 2021 were used (Uphoff et al. 2022a). Choptank River surveys consisted of 1954, 1957-1962, 1980-1989, 1994, 1997-2004, 2013-2019, and 2021-2024. 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.

Examination of the plot of milestone dates for the first egg appearance and 12ºC indicated a shift over time. First egg appearance date early in the time-series often appeared to occur after the 12ºC date was met and shifted to the opposite later in the time-series. We tested this with a linear regression of (12ºC date – first egg date) against year. The first egg date indicated early presence of females on the spawning ground (an indicator of migratory behavior) while 12ºC was a physical marker of when spawning was typically initiated. A negative value would indicate that earliest spawning (first egg) as initiated after the 12ºC milestone date and a positive value would indicate that earliest spawning preceded it.

Two HOBO model Pro v2 temperature loggers were deployed on buoys (one on each buoy) marking the outside of shoal water at sites 10 and 11 during 2024. They were set approximately 0.7-1.0 m below the surface. Temperature was recorded every 30 minutes between January 31 and May 13. The recorders were set and retrieved with major assistance by Boating Services personnel from the Cambridge office. Temperatures from the loggers during the time span for estimating *Ep* (described below) were compared to those collected at the same site by our YSI model 556 at the closest time available from the HOBO logger using linear regression. Survey temperatures were overlaid onto the plot of logger data by date and time to examine how temperature varied by location (survey data) and time. Survey times were not synched exactly to logger temperatures but the starting time of survey temperatures on a given day was used as an initial match. Logger temperatures were considered for subsequent analyses if there was more than a one-day gap with the survey date if there was a reasonable match in conditions on concurrent dates.

We used summarized average winter air temperatures (December-February) at Baltimore (https://marylandclimateandweather.weathertogether.net/maryland-climate-data/) during 1954-2024 as an indicator of regional winter intensity (T_w). These temperature data extend back to 1871 (Taylor 2024). They were tabulated as °F and we converted them to °C. Their sources are the National Weather Service and the National Center for Environmental Information. This data for Baltimore includes downtown locations (1871-1950) and Baltimore-Washington International Airport (1950-Present). The differences in the weather instrument sitings between the urban location rooftop and the suburban airport often resulted in large temperature differences but the National Weather Service/National Centers for Environmental Information's official long-term historical archive for Baltimore is maintained in this record (Taylor 2024).

Linear regression was used to examine the relationship of winter intensity (T_w) to days to reach the 12°C, 16°C, or 20°C milestones (D as days from April 1; April 1 = day 0) in the Choptank and Nanticoke rivers. These two rivers are adjacent to one another in Maryland's eastern coastal plain and temperature conditions should have been similar enough that they could be combined. Inspection of the bivariate plot indicated the possibility of an asymptote through an initial portion years of T_w and a decrease in the latter portion. To avoid applying a complex nonlinear equation to fit these data, we used T_w^2 as the independent term in a linear regression with D. This transformed the small negative to small positive values and approximated an asymptote for lower values.

Flow – We updated the standardized flows developed in Uphoff et al. (2022b) through 2024. 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) 2024 update – Sample size was sufficient for estimating *Ep* in the Choptank River (N = 78) during 2024. Samples used to estimate *Ep* began on March 25 and ended on April 25. The temperature cutoff was reached on April 29.

The estimate of *Ep* in Choptank River during 2024 was 0.58 (N = 78 and 45 tows with eggs) with a 90% CI of 0.49-0.67 (SD = 0.048; Figure 2.1.3) and this estimate served as the baywide *Ep* estimate (Appendix; Figure 2.1.4) as well. Estimated *Ep* in 2024 overlapped lower 90% CIs of of baywide values indicating adequate levels for a full range of recruitment (*Ep* > 0.60) and upper CIs of baywide *Ep* estimates during 1982-1988 ($\leq$ 0.60) that were reflected by JIs lower than expected given their estimates of relative survival (Uphoff et al. 2023b). There was a 62% chance that estimates of *Ep* were 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 0% chance that *Ep* was above the 1989-2019 baseline median (0.77).

There is reason to believe that the 2024 estimate of *Ep* was biased low. Two sampling dates at the beginning were included because in each case one egg was found at two sites at temperatures where spawning is rare (March 25 and 28; water temperature was 9-10˚C). Cumulative distributions of egg counts for the Nanticoke and Choptank rivers during 1954-1993 (105,336 and 113,503 eggs, respectively) indicated that 0.1% of eggs were collected by 10 ˚C

(Uphoff et al. 2022b). This introduced 16 instances of absence into the *Ep* estimate. With these two early dates removed, estimated *Ep* would have been 0.66 (SD = 0.060). Seven additional zeros were introduced because eggs were detected in the Tuckahoe Creek on the last eligible date (April 25) but not before. Spawning in the upper mainstem (sites 15 and 16) was detected on April 29; this spawning was not included because the temperature cutoff had been reached. Excluding Tuckahoe Creek from the estimate and starting the *Ep* estimate on March 25, *Ep* was 0.69 (SD = 0.056). Both treatments resulted in a high chance (> 90%) of not meeting the 1989-2019 *Ep* median and a lower chance (< 15%) of falling below a threshold where year-class success would be lower due to low *Ep*.

The spawning area was located between sites 1 and 13 in the mainstem Choptank River and in Tuckahoe Creek in 2024 based on egg presence. Spawning intensity was greatest in the mainstem (volume indices ranged from 0.75 to 1.33) and light in Tuckahoe Creek (0.20-0.25). Highest intensity indices (1.33) were found at stations 7 and 8. This downstream distribution likely reflecting high winter-spring precipitation and flow due to an El Niño climate pattern.

Juvenile index 2024 update – The Baywide JI was 1.06 in 2024 (90% CI = 0.84-1.34) and the year-class was poor (Figure 2.1.5; Durell and Weedon 2024).

Relative larval survival 2024 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 1.83 in 2024 (Figure 2.1.6). The simulated mean was 1.86 and the SD was 0.33. The probability of falling below the poor RLS criterion in 2024 was approximately 0.75.

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 2024, -0.32, was below the normal range but above the range during 1982-1988; this negative deviation may have reflected negative bias in the *Ep* estimate (explained above; Figure 2.1.7). If the lower of the bias adjustments for *Ep* were used (*Ep* = 0.66) then RLS would equal 1.61 and the deviation would equal -0.14; this deviation would be within the normal range.

Water quality update - During 2024, median pH during April 1-May 8 (a standard period across years) in Choptank River was 6.90 and measurements ranged between 6.18 and 7.61 (Table 2.2.1; Figure 2.1.8). Dissolved oxygen and pH measurements were poorly correlated (r = 0.18, P = 0,06), indicating that pH was unlikely to have been influenced by phytoplankton photosynthesis. The 2024 median pH was the first since 2014 to fall below 7.0. A pattern of above neutral median pH measurements was routine in 2014-2023. Medians during 2014-2023 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 than 2014-2024. 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).

Minimum and median conductivity in 2024 were the lowest of the 2013-2024 time-series, reflecting high flows during March-April (see flow section below). Conductivity measurements were at or less than 1,200 $\mu S/cm^2$ through April 18; they were very low (<292 $\mu S/cm^2$) during surveys conducted on April 4 and April 8. Conductivities at lower stations (stations 1-7) began to climb past 1,200 $\mu S/cm^2$ after April 18. 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 2024 were 149, 75, and 3,350 µS/cm^2, respectively. Disparity between the mean (379 µS/cm^2) and median in 2024 indicated values were skewed towards low conductivity and were not normally distributed (Table 2.1.1). Conductivity measurements above 1,300 µS/cm^2 were not detected until April 25.

Water temperature – There were 16 instances for station 10 and 15 for station 11 when water temperatures were measured at the same site and approximate time by loggers and the boat survey. Linear regressions did not indicate that measurements were significantly different. Slopes were not different from 1.0 and intercepts were not different from 0 based on 95% CI overlap for both sites. The equation for predicting survey water temperature (T_S; ˚C) from the site 10 logger temperature (T_L) was

$$T_S = (1.02 * T_L) - 0.38 \ (r^2 = 0.99, P < 0.0001);$$

the SE for T_L was 0.022 and 0.33 for the intercept. The equation for predicting survey water temperature (T_S) from the site 11 logger temperature (T_L) was

$$T_S = (1.02 * T_L) - 0.52 \ (r^2 = 0.99, P < 0.0001);$$

the SE for T_L was 0.031 and 0.47 for the intercept. An overlay of all survey water temperatures on logger temperatures indicated they generally tracked each other but there was more variability in survey temperatures during periods of warming (Figure 2.1.9).

Lowest spawning intensity (egg volume index = average of ranks) in 2024 occurred during March 25 - April 8 when water temperatures were approximately 10-12˚C and indices were between 0.25 and 0.56 (Figure 2.1.10). There was a slight rise in intensity between March 28 and April 1 that was concurrent with a slight temperature rise from 10.7˚C to 12.0˚C but intensity diminished on April 4 and April 8 as water temperatures fell back to 10.7-11.6 ˚C. Intensity surged to 1.0 on April 10 following a substantial rise in water temperature to 13.8-15.0˚C; surface spawning was observed on this date. Intensity remained between 0.71 and 0.88 during April 15 – April 25, then fell to 0.63 on April 29; temperatures were approximately 17.0-18.5˚C. Temperatures passed the 21.0˚C sampling cutoff for *Ep* and intensity on April 29 (Figure 2.1.10).

The first egg was collected on March 25, 2024. This was the second earliest date that spawning has been detected in Choptank River and Nanticoke River ichthyoplankton surveys (Figure 2.1.12). There was one first egg date earlier than April 1 out of 23 dates prior to 2000 and six out of 16 dates after (Figure 2.1.12). This earlier date for detection of the first egg may indicate that females have been arriving earlier on the spawning grounds since 2000.

Strong year-classes for the Choptank River or Nanticoke River were GMs at or greater than 7.9 and 4.6, respectively (Figure 2.1.11). Years with temperature milestone dates matching strong year-classes are summarized in Table 2.1.2.

Water temperatures reached 12ºC on April 1 on the survey and March 31 with the loggers. The survey date was used for this milestone. This was the third earliest date for this milestone based on survey data (Figure 2.1.13). Eighteen of 23 dates since 1999 have fallen below the 1954-1999 median. These early spawning milestones seem to be occurring sooner more often since 2017. Strong year-classes during 1957-2024 (in the 75th percentile of Choptank or Nanticoke River juvenile indices) occurred within a narrower band of 12ºC dates between April 9 and 16 along the 1954-1999 median (April 12) than lesser year-classes (March 25-April 21). The median 12ºC date for the strong year-classes was April 13 (N = 9; Figure 2.1.13).

The mid-milestone, 16ºC, was reached in 2024 on April 15 on the survey and on April 12 with the temperature loggers (Figure 2.1.14). Logger temperatures were used to determine this milestone date. Twenty-two of 29 dates since 1999 have been sooner than the 1954-1999 median (April 23). This milestone has been reached earlier and more consistently since 2017. Strong year-classes were likely within a narrower band of 16ºC dates between April 9 and May 2, with 6 above the 1954-1999 median and 9 below; all but two of these dates were between April 14 and April 27. Lesser year-classes had a wider range of dates (April 5 – May 13). The median 16ºC date for the strong year-classes was April 19. There has been overlap of dates when 16ºC has been reached between strong and lesser year-classes; the band of dates for strong year-classes has not been unique (Figure 2.1.14).

The 20ºC milestone was detected in 2024 on the April 29 survey and on April 30 with the loggers (Figure 2.1.15). The survey date was used for this milestone. Thirteen of 16 dates since 1999 have fallen below the 1954-1999 median date (May 13). Nine out of 10 years with strong year-classes were located below the 1954-1999 median milestone date and the median date for strong year-classes was May 5. The span of days to the 20ºC milestone for weak year-classes was April 15 to June 4 while strong year-classes had a span between April 21 and May 22 (Figure 2.1.15).

Choptank River offered a series of years where all three temperature milestones were available within spawning seasons up through 2024: 1954, 1981-1985, 1987, 2001-2003, 2013, 2014, 2017-2019, 2021, 2023, and 2024 (Figure 2.1.16). Several changes were evident. The span of dates of the milestones shortened in 2001-2003 but remained within the bounds of earlier years. During 2013-2024 there was a shift to an earlier spawn for all but two years (2014 and 2018). These two years resulted in the strongest year-classes of the available years since 2013; 2018 met the criterion for the upper quartile in Choptank River, GM = 7.87, and the GM for 2014 was 6.28. Very rapid warming was evident as 1-2 day intervals between milestones in 2004, 2013, 2017, and 2019 (4 of 10 years available after 1999). After 2000, the 16ºC was frequently hit on dates when 12ºC was met during the 1980s (Figure 2.1.16).

First egg appearance date (indicating migration of females onto the spawning grounds) early in the time-series often appeared to occur after the 12ºC date (physical marker of when spawning was typically initiated) was met and shifted to the opposite later in the time-series. There was a positive trend over time for the difference between the 12ºC date and date that a first egg was collected during 1954-2024, indicating a shift of earliest spawning (first egg) after the 12ºC milestone date to the earliest spawning preceding 12ºC ($r^2 = 0.14$, $P = 0.0006$; Figure 2.1.17). Early spawning became less synchronous with the initial water temperature trigger. There was a break in the data between 1986 and 2001 and it is possible that this change was not continuous as indicated by a linear fit.

Winter mean air temperatures in Baltimore (T_w) increased during 1954-2024 (Figure 2.1.18). The lowest temperature was -1.61°C in 1963; T_w remained below 1.67°C through 1970. T_w varied between -0.94 and 3.44°C through 2015 and between 2.44 and 5.56°C afterward. A linear regression indicated that T_w increased, on average, by 0.044°C per year since 1954 ($r^2 = 0.23$, $P < 0.0001$). Decadal medians (with partial decades at the beginning and end of the time-series lumped with the adjacent decade) did not indicate a steady progression: 0.89ºC for 1954-1969, 2.08ºC for 1970-1979, 1.56ºC for 1980-1989, 2.92ºC for 1990-1999, 2.39ºC for 2000-2009, and 3.44ºC for 2010-2024. Large changes in median T_w were present at the beginning and end decades of the time-series with some variation in between without much indication of change. Correlation analysis indicated that T_w and the NAO were positively and moderately associated ($r = 0.53$, $P < 0.0001$; Figure 2.1.18). This climate feature may be an influence on regional winter intensity.

Regression analyses featuring T_w^2 as the independent variable with D provided a better fit than T_w for all three milestones, indicating an asymptote at lower temperatures and an accelerating negative effect at higher temperatures (Table 2.1.3; Figures 2.1.19–2.1.21). All slopes were negative. These regressions with T_w^2 did not account for a large amount of variation (r^2 ranged from 0.09 to 0.20 and P ranged from 0.02 to <0.0001) but indicated an underlying influence of winter intensity could be reflected by temperature milestones relevant to spawning and prolarval survival (Table 2.1.3). The relationship at the 12°C milestone had the shallowest trajectory (Figure 2.1.18) and the difference in predicted D when T_w was 2.3°C (D = 8.1; median T_w for 1970-2009) and the maximum T_w observed, 5.6°C (D = 1.2) was 6.9 days earlier than that median. At 16°C, the difference at the same T_w range was 12.5 days earlier (20.7 - 8.2; Figure 2.1.20) and the difference at 20°C was 14.2 days earlier (40.0 – 25.8; Figure 2.1.21). Observed days to reach all three temperature milestones well above the predicted line of the D and T_w relationship were frequent out to 3-4°C became close to or below the predicted relationship afterwards (T_w extended as far as 5.6°C).

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 since 1993. Standardized flows were near or above average baseline flow of 1957-2020 (1.0) during 2024 in Choptank River (1.83), Nanticoke River (1.71), Head-of-Bay (1.17), and Potomac River (0.99). Choptank River and Nanticoke River flows were among the highest since 1993 (Table 2.1.4; Figure 2.1.22).

Discussion

Proportion of ichthyoplankton tows with Striped Bass eggs (Ep) 2024 update – The estimate of baywide *Ep* (0.58) for 2024 was the lowest since 1989. There was a high chance it had crossed a threshold (0.60) to levels during 1982-1988 when spawning stock was depleted enough to affect year-class success. However, there is good reason to believe that the 2024 estimate of *Ep* was negatively biased because of unusual egg presence at very low temperatures and late appearance of spawning in Tuckahoe Creek that introduced "extra" zeros into *Ep*. Removal of these events singly or in combination resulted in *Ep* (0.66-0.69) with a high chance (> 90%) of not meeting the 1989-2019 *Ep* median and a lower chance (< 15%) of falling below a threshold where year-class success would be lower due to low *Ep*. Uphoff (1997) described how concern from focusing on poor results from a single system and year could be alleviated by sampling more than one spawning area, including at least one adjacent year to diagnose whether *Ep* was critically low, or specifying a tolerable frequency of high risk within a time period. In this case, including one or more years of additional *Ep* is a reasonable response given that our capacity to sample more areas is limited, the level of tolerable risk is not defined, and *Ep* is not used as a management trigger. The previous two years' *Ep* estimates, 0.69 and 0.74, both had low risk of falling below the threshold criterion (Uphoff et al. 2023; 2024).

Juvenile index 2024 update – The 2024 Choptank River JI was poor. Use of JI quartiles to designate poor and strong year-classes was convenient and use of the lower quartile 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 transitions. For Striped Bass in Maryland's portion of the Chesapeake Bay, RLS and *Ep* can be used to identify periods of productivity (Uphoff 2023). However, quartiles may not align with the fishery needs. 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).

Relative larval survival 2024 update – Estimated *Ep* in 2024 substantially overlapped *Ep* estimates in 2001 and 2011 that were associated with strong year-classes. Estimated RLS in 2024 (and 2023) breached 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. Consecutive years of low baywide JIs have occurred since 2019 and presumably low RLS; *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). Six consecutive years of low year-class success is worrisome and will impact the fishery.

Water quality update - Comparisons of pH and alkalinity (the latter was measured in 2021-2023) in Choptank River between 1986-1991 and 2013-2024 indicated improvement (higher averages) that would have lowered toxicity of metals implicated in elevated larval mortality and 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 2014-2024 in Choptank River. Median and mean pH during 2024 surveys was lower than estimated for remaining surveys since 2014. Average alkalinity was at least 3-times higher in 2021-2023 than 1986-1991. Low survival of Striped Bass postlarvae 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), decreased erosion (sediment is a vector for contaminants), and a general increase in alkalinity of freshwater across the U.S. (Uphoff 2023). While recent measurements of metals are unavailable, it seems unlikely that poor survival of larvae during 2019-2024 could be attributed to a return of toxic water quality conditions implicated in high postlarval mortality and poor recruitment during the 1980s. Estimates of a proxy index for postlarval Z during 2023 and 2024 were low (see Section 2.2, zooplankton investigation).

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 10 surveys during 2014-2024. 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 did not measure alkalinity in 2024 and will not in subsequent years since three years of monitoring indicated it was stable. A correlation analysis indicated that minimum pH tracked all of the alkalinity summary statistics (correlation range = 0.72-0.80) during the 1980s and 2021-2023. 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 for several years could warrant resumption of alkalinity monitoring.

Water temperature – Survey water temperatures on logger temperatures generally tracked each other but there was more variability in survey temperatures during periods of warming.

Changes were not uniform among temperature milestones and early milestones appeared to have changed the least. 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 progressed 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) was updated through 2023 and the conclusions did not change (J. Uphoff, MD DNR, unpublished).

First egg appearance date (indicating migration of females onto the spawning grounds) early in the time-series often appeared to occur after the 12ºC date (physical marker of when spawning was typically initiated) was met and shifted to the opposite later in the time-series. There was a positive trend over time for the difference between the 12ºC date and date that a first egg was collected during 1954-2024, indicating a shift of earliest spawning (first egg) after the 12ºC milestone date to the earliest spawning preceding 12ºC. Early spawning became less synchronous with the initial water temperature trigger.

In addition to changes in average dates, the plots of 12ºC and 16ºC milestone dates since 2010 suggested clustering at earlier dates that were below the medians for 1954-1999 and medians for years with strong year-classes. These shifts were noticeable for 2019 and later. The recent cluster was below the range where strong year-classes had developed for the 12ºC milestone. Six of seven points in the cluster for 2019 and afterwards for 16ºC were out of the range for 14 of 15 strong year-classes. Clustering was not readily apparent for the 20ºC milestone.

Changes between 12ºC and 16ºC could be important to the formation of strong year-classes because most spawning occurs between these temperatures. Cumulative distributions of egg counts for the Nanticoke and Choptank rivers by temperature during 1954-1993 (105,336 and 113,503 eggs, respectively; surveys with egg count data were not made after 1993) indicated that most egg deposition would occur between 12°C and 16˚C (83.2% for Nanticoke River and 89.2% for Choptank River; Uphoff et al. 2022b). A large portion of spawning may occur over a few days early in the season in a gamble that subsequent conditions will favor offspring survival, but this can often be a bad bet (Maryland Sea Grant 2009). Spawning may start at low, more lethal temperatures and linear regression indicated that early spawning (first egg date) at temperatures below the 12ºC date was becoming more frequent. Losses from much of mistimed early spawning in 2024, as indicated by the egg volume index, were steady but not large. However, at least some of the initial peak spawning was at lower, less optimal temperatures. Minimum water temperatures during peak spawning (time needed to collect 85% of eggs) were linearly related to survival from egg to 6 mm larvae in the Choptank River during 1980-1988; these minimum temperatures ranged from approximately 11ºC to 15ºC and predicted survival increased from 0 to approximately 0.30 in this range (Uphoff 1992). Cold fronts can quickly drop suitable temperatures to lethal levels and rapid warming can hasten lethally high temperatures. Occasional strong year-classes have been produced and longevity (in the absence of heavy fishing) ensured that these strong year-classes reproduced over many years and dampened the effects of environmental variation (Florence 1980; Rago and Goodyear 1987; Rago 1992; Richards and Rago 1999; Secor 2000a; 2007; Maryland Sea Grant 2009).

Knowledge of the prevailing background climatic regime may provide managers an estimate of the relative chance for high or low year-class success as reflected by recruitment patterns of the dominant production regime (Austin 2002). Winter-spring climate variability was considered a prime candidate as an environmental driver of anadromous fish recruitment in the Bay that resulted in shifts in anadromous fish recruitment that were positive (cooler, wetter springs) or negative (warmer, drier springs) that lasted for a decade or more. (Austin 2002; Wood and Austin 2009). Szuwalski et al. (2015) offered synchronous shifts in long-term climate patterns as common environmental drivers of shifts in fish production, more-so than spawning stock.

Low frequency patterns in lower Chesapeake Bay (Virginia's portion) water temperature, river discharge and surface winds have been characterized as dominant decadal oscillatory waves with dramatic phase shifts (Austin 2002). Austin (2002) suggested that positive correlations of Lowess smoothed water temperatures in the lower Bay (Virginia Institute of Marine Science pier, 1960-2000) and the North Atlantic Oscillation (NAO) indicated coherence of low frequency trends. The NAO indexes wind balances of the northwest Atlantic Ocean; a strong NAO in winter results in a strong westerly flow that drains off cold Canadian air (Austin 2002). The NAO is influenced by climate warming (higher CO_2 levels increase temperatures) and volcanic activity (cooler temperatures; Mitevski et al. 2025; Smith et al. 2025). It is possible that the NAO resembles an atmospheric oscillation driven by atmospheric and oceanic interactions but is not, rendering it unpredictable. Earth System Models generally project a more positive and less variable NAO under 21st century high-emission scenarios (Mitevski et al. 2025; Smith et al. 2025).

We found a moderate association of the NAO with regional winter intensity in the Maryland portion of the Bay and this regional intensity was a potential underlying influence on milestone temperatures indicating initiation, intensity, and duration of Striped Bass spawning. These regressions with T_w^2 did not account for a large amount of variation but they indicated that winter intensity could underlie long-term changes in temperature milestones relevant to spawning and prolarval survival. The low amount of variation explained is not particularly surprising. Chesapeake Bay is located among a number of climatologically important features (Appalachian Mountains, the Gulf Stream, position of Arctic fronts to the north, and the Westerlies) that also influence its climate (Austin 2002). Local weather conditions (described previously) operating on a scale of days can have a large influence on year-class strength. Uphoff

(2023) described changes in major long-term large-scale anthropogenic factors (acidic precipitation and agriculture) that provided little indication of stable conditions within the spawning and larval nursery areas of Chesapeake Bay since the 1950s.

Water temperature and Striped Bass spawning changes described here were similar to expectations described by MD Sea Grant (2009) for Chesapeake Bay and Nack et al. (2019) for the Hudson River. 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). 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). 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.

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 as water temperatures increased 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 10-14 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 days 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 intervals could have an impact as well. None of the surveys were conducted daily and most were conducted several days a week. There was 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. The use of continuous temperature recorders in 2024 filled the gap between surveys and readings between the recorders and survey were comparable.

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 of the available data. None of these surveys were specifically designed to monitor for long-term temperature changes and they represented "targets of opportunity". 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 on 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.

Flow - Spawning area standardized flows appear to have shifted downward after 2011. Spawning season flows in Choptank and Nanticoke rivers were well above average in 2024, while Head-of-Bay and Potomac River flows were slightly above average and average, respectively. With the exception of 2024, 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). However, there was still a substantial chance that a poor year-class could be formed; 13 of 56 year and spawning area combinations for above average flows had poor year-classes during 1993-2020. Plots of area-specific JIs and standardized flow during 1993-2020 suggested that high standardized flows above ~1.70 (1.0 would be average for this period) would not be followed by a strong year-class in Choptank and Nanticoke rivers (juvenile indices were poor in both during 2024). 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 (Uphoff et al. 2022b). 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. 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.

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Table 2.1.1. Summary of pH, conductivity ($\mu S/cm^2$), and total alkalinity (mg/L) during a standard period (April 1 – May 8), 1986-1991 and 2014-2024. 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
2024	6.84	6.90	7.38	6.52	6.18	7.61	160

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
2024	379	149	457	300	75	3350	160

Table 2.1.2. Days from April 1 (= day 0) that spawning water temperature milestones were met for strong year-classes (≥ 75th percentile) in Choptank and Nanticoke rivers for data available from 1955-2024.

Year	River	12˚C days	16˚C days	20˚C days
1964	Nanticoke	10	19	41
1965	Nanticoke	15	32	40
1970	Nanticoke	8	26	30
1972	Nanticoke	13	18	
1989	Choptank		23	
1998	Choptank		14	36
1999	Choptank		8	
2000	Choptank			38
2001	Choptank	12	22	32
2003	Choptank	15	24	31
2004	Choptank		20	22
2011	Nanticoke	10	17	
2014	Nanticoke		14	
2015	Choptank	6	21	35
2016	Nanticoke		17	
2018	Choptank	12	25	

Table 2.1.3. Summary of linear regressions of winter intensity (Tw, average air temperature in Baltimore during December-February) and spawning temperature milestones.

Variable	Milestone	Slope	Intercept	r^2	P	Slope SE	Intercept SE
T_w	12°C	-0.64	8.96	0.03	0.22	0.52	1.33
T_w^2	12°C	-2.68	9.59	0.09	0.02	0.11	1.12
T_w	16°C	-1.75	23.46	0.14	0.0013	0.52	1.34
T_w^2	16°C	-0.49	23.22	0.20	<0.0001	0.11	1.05
T_w	20°C	-1.80	42.96	0.07	0.04	0.88	2.17
T_w^2	20°C	-0.57	42.94	0.12	0.007	0.20	1.85

Table 2.1.4. Average annual flow during two-month periods for the four major spawning areas 1957-2024. 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
2024	73,336	22,006	444	258
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.

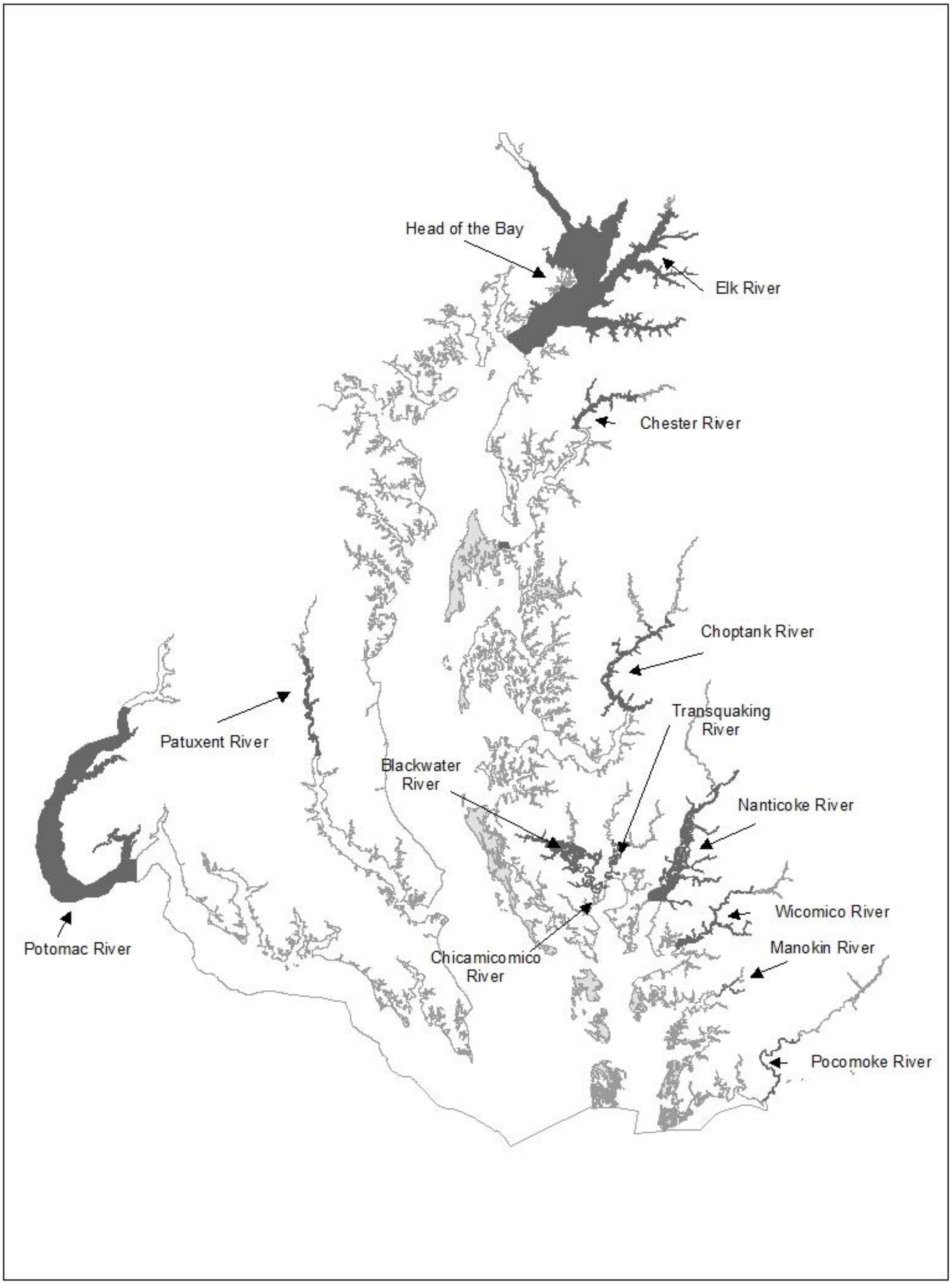


Figure 2.1.2. Location of historic and sites sampled in Choptank River during 2024 and mainstem sites or within Tuckahoe Creek (triangles). Stations 17 and 21 were not sampled in 2024 because access was too shallow. Inset shows location of Choptank River within Chesapeake Bay.

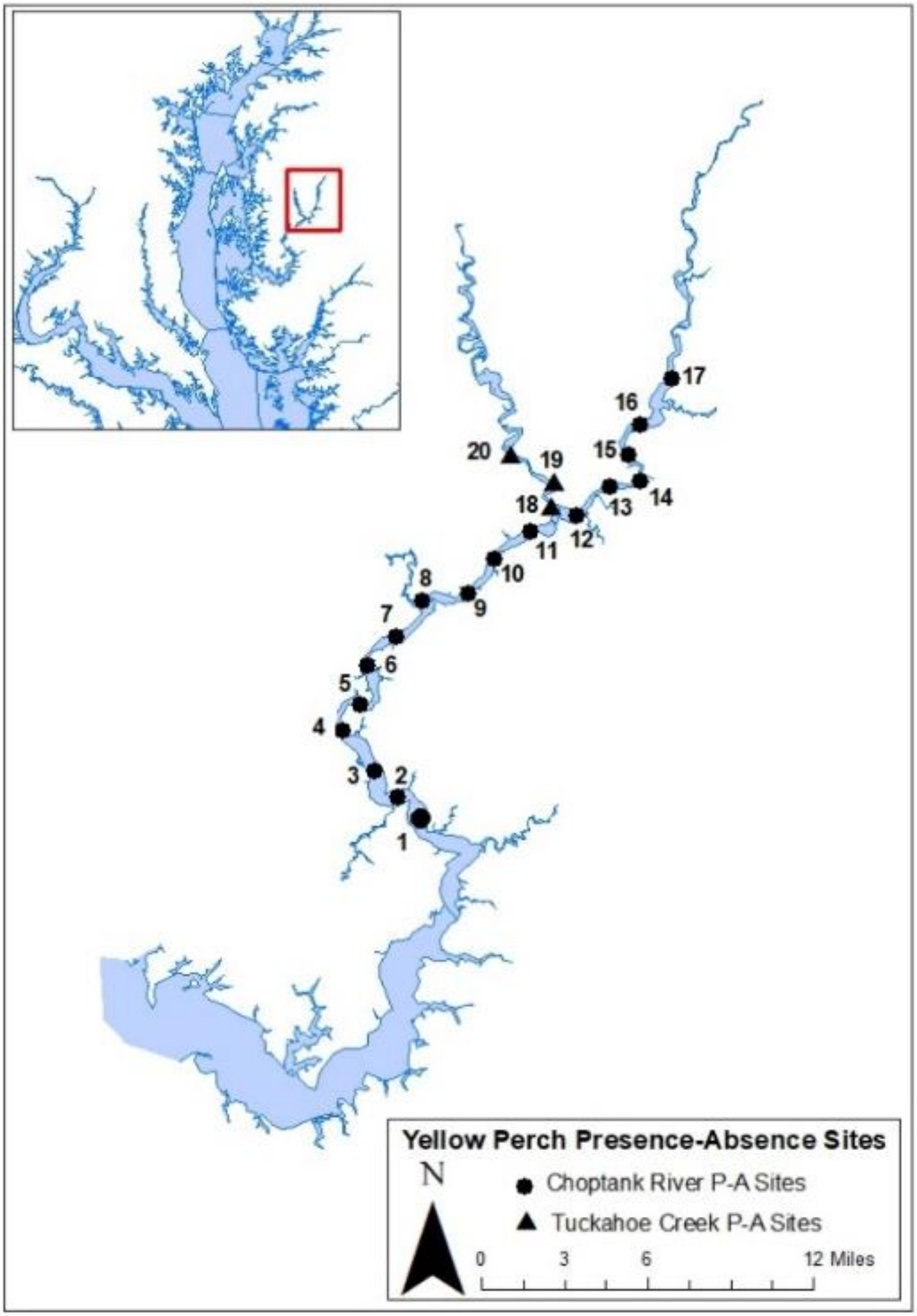


Figure 2.1.3. Spawning area specific proportion of tows with Striped Bass eggs (*Ep*) estimated from surveys in juvenile index rivers conducted during 1955-2024. Elk River represents a portion of the Head-of-Bay.

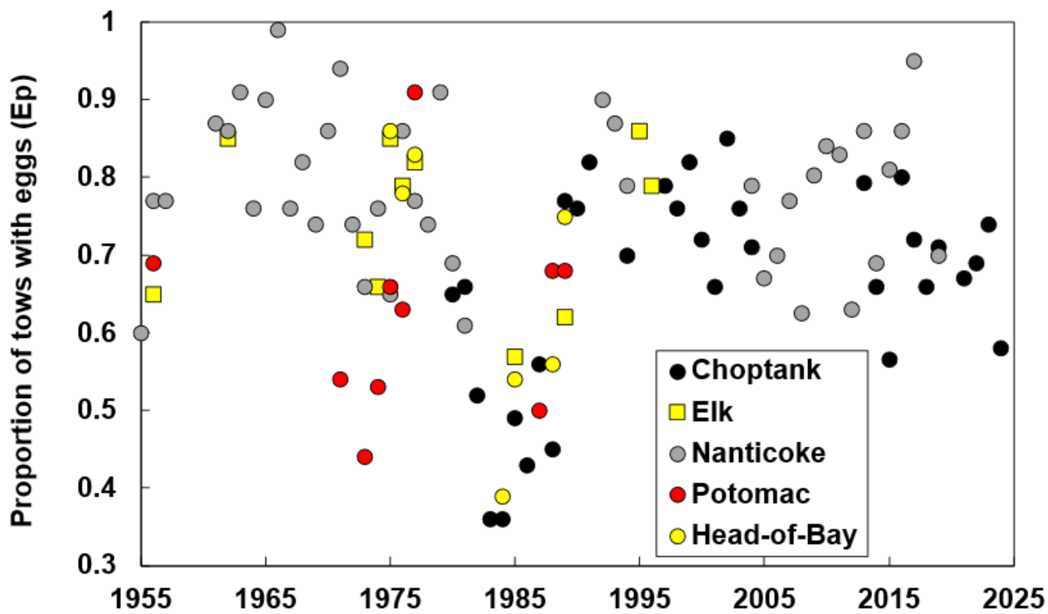


Figure 2.1.4. Baywide (Maryland's spawning areas) proportion of tows with Striped Bass eggs (*Ep;* diamond) and its 90% CI (vertical line) estimated from surveys in juvenile index rivers conducted during 1955-2024. Baywide estimate pools available data from spawning surveys conducted in four areas surveyed for the juvenile index: Head-of-Bay, Potomac River, Nanticoke River, and Choptank River.

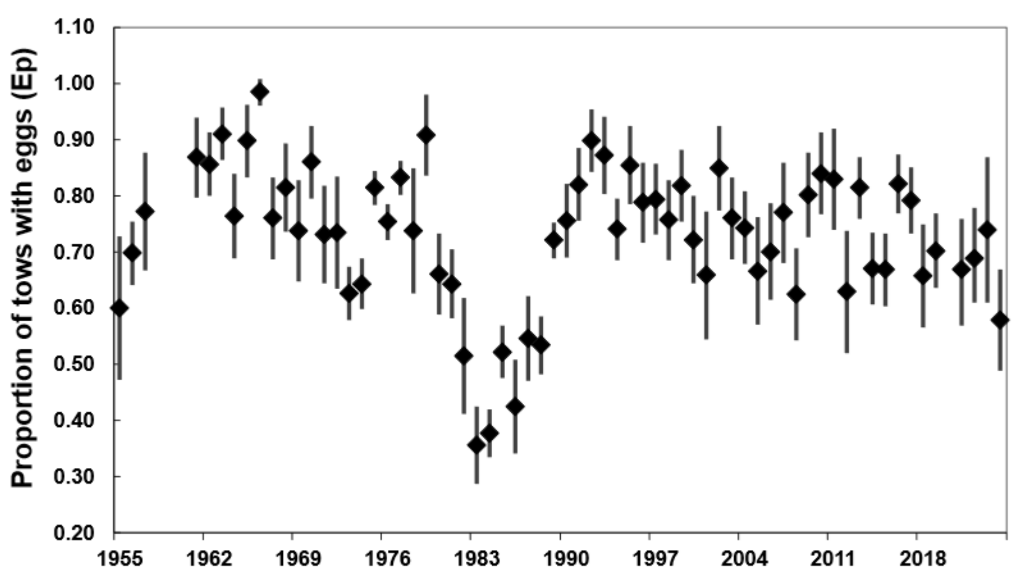


Figure 2.1.5. Baywide Striped Bass juvenile indices (geometric mean catch per standard seine haul; diamonds) and their 90% confidence interval (vertical line) estimated for Maryland's major spawning areas during 1957-2024 (Durrell and Weedon 2024).

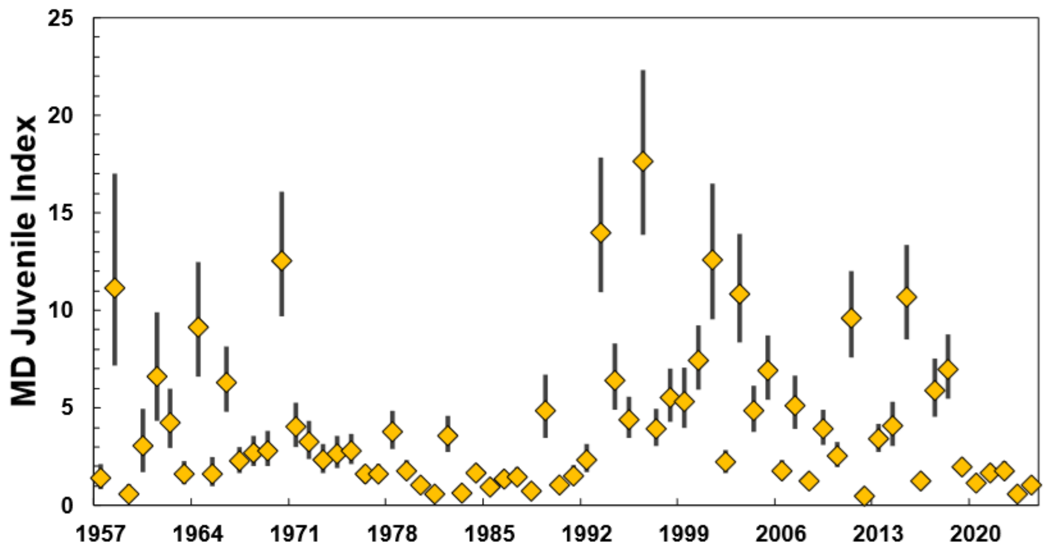


Figure 2.1.6. Relative larval survival (baywide JI / baywide *Ep*) mean and 90% CIs, 1957-2024. Quartiles (green and grey dashed lines) based on 1957-2009 (ASMFC JI base years); upper quartile = green dashed line and lowest quartile = grey dashed line.

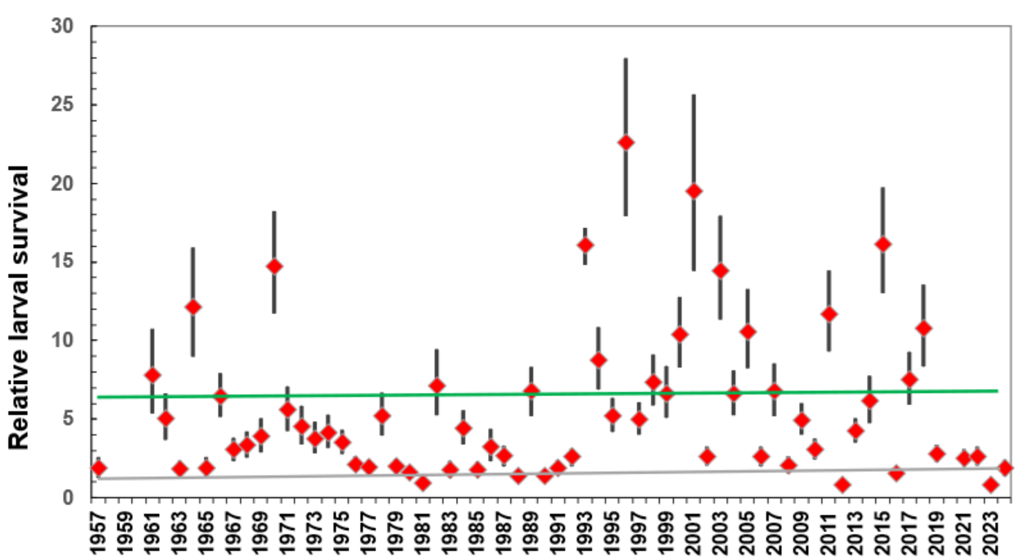


Figure 2.1.7. Difference of standardized juvenile index (Std JI) and standardized relative larval survival (Std RLS) as proportion of standardized JI during 1957-2024. Large negative deviations indicate overfishing in 1982-1988. Indices standardized to mean of common years (same scale).

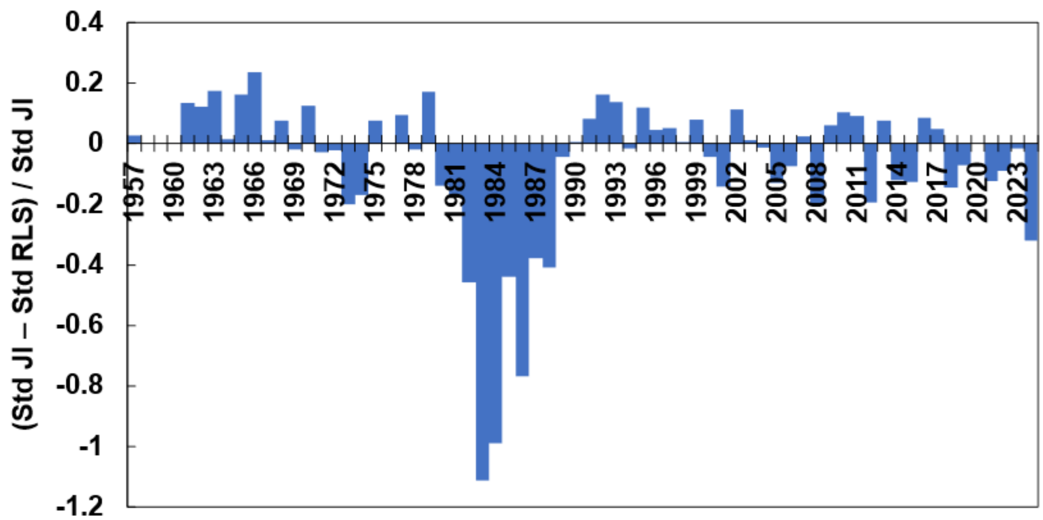


Figure 2.1.8. Choptank River pH median and range during April 1 – May 7, 1986-1991 and 2014-2024.

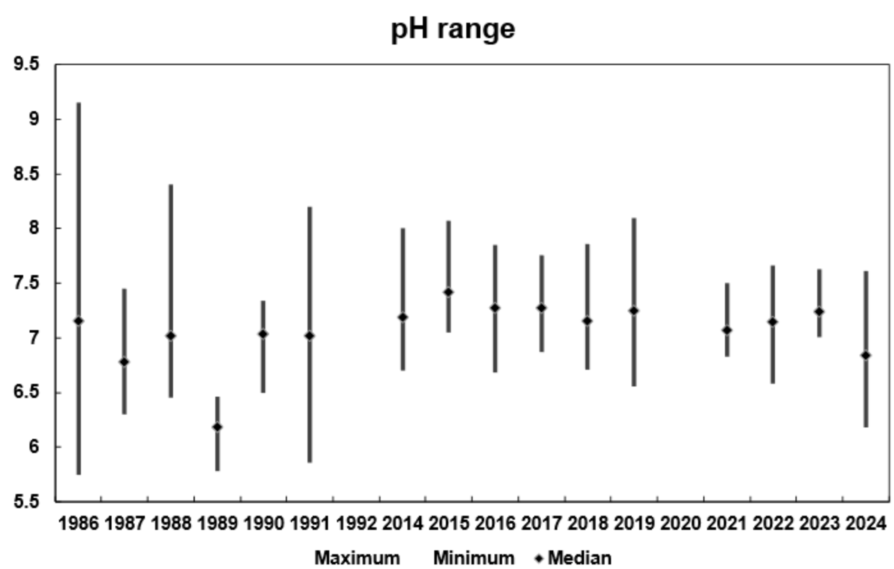


Figure 2.1.9. Water temperature from continuous recorders and egg presence - absence surveys between when eggs were present and the temperature cut-off was reached during 2024.

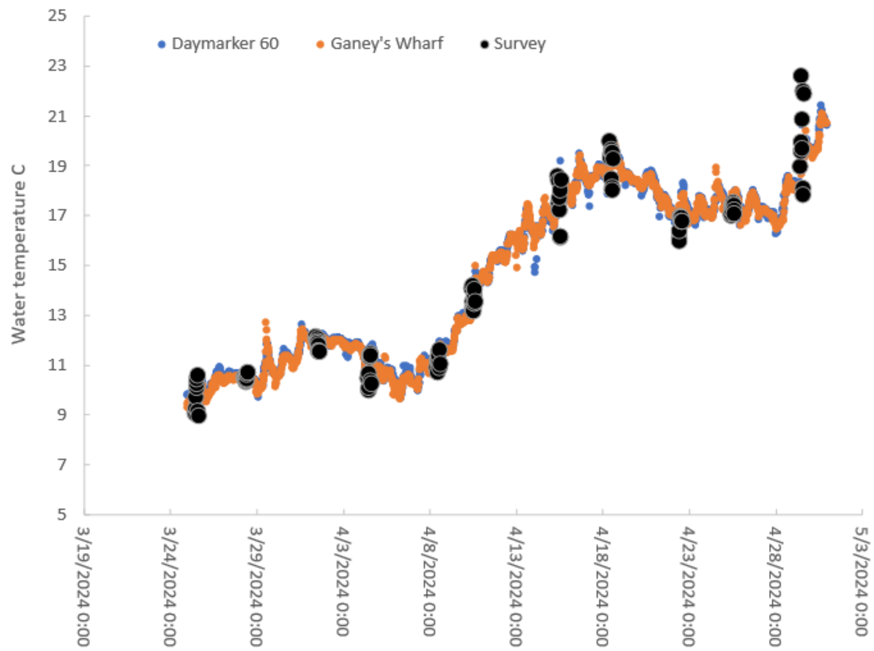


Figure 2.1.10. Water temperature from continuous recorders and the Striped Bass egg volume intensity index during 2024.

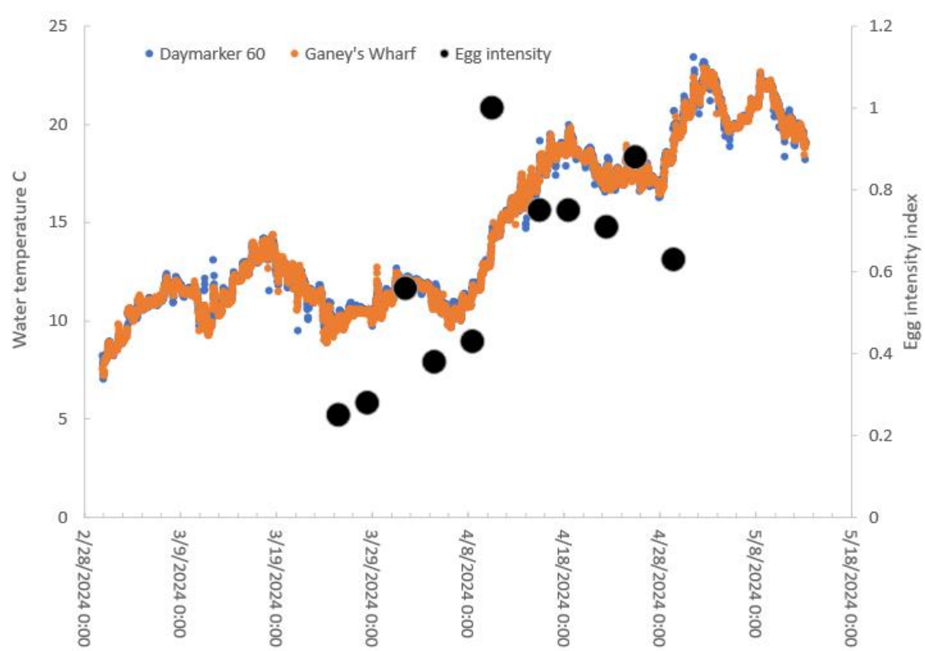


Figure 2.1.11. Choptank River and Nanticoke River juvenile index time -series with strong year-class (upper quartile) boundary indicated by a red line.

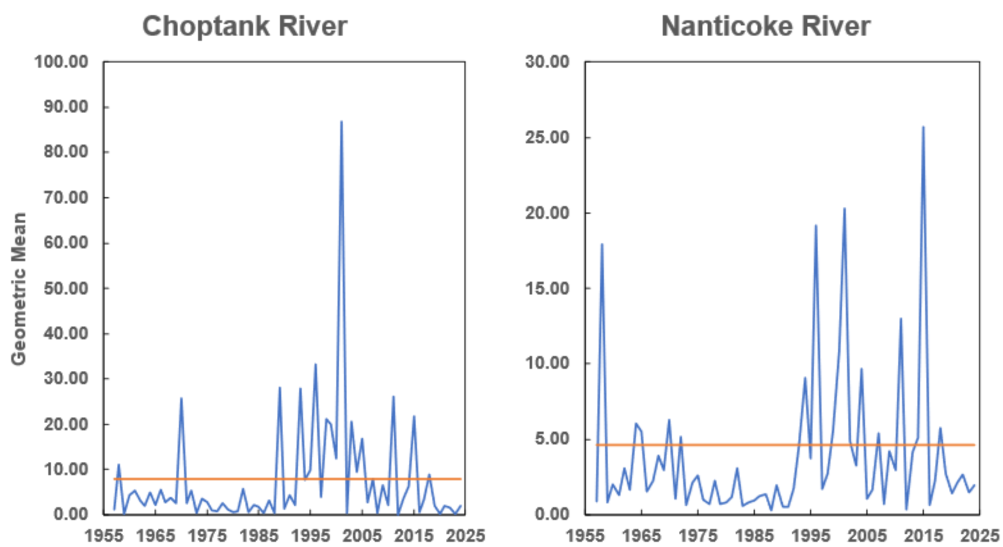


Figure 2.1.12. Days from April 1 (day = 0) that the first egg was collected in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2024. Median = median day for both rivers combined (day 7) during 1954-1999.

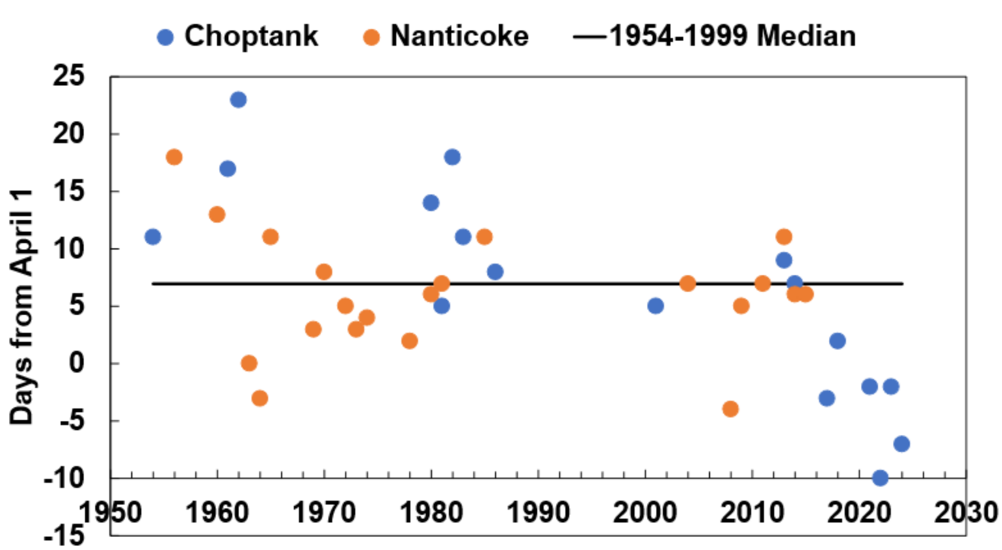


Figure 2.1.13. Days from April 1 (day = 0) that 12ºC was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2024. Median = median day for both rivers combined (day 11) during 1954-1999. Points with thick borders indicate strong year-classes and the dashed line is the median milestone for strong year-classes.

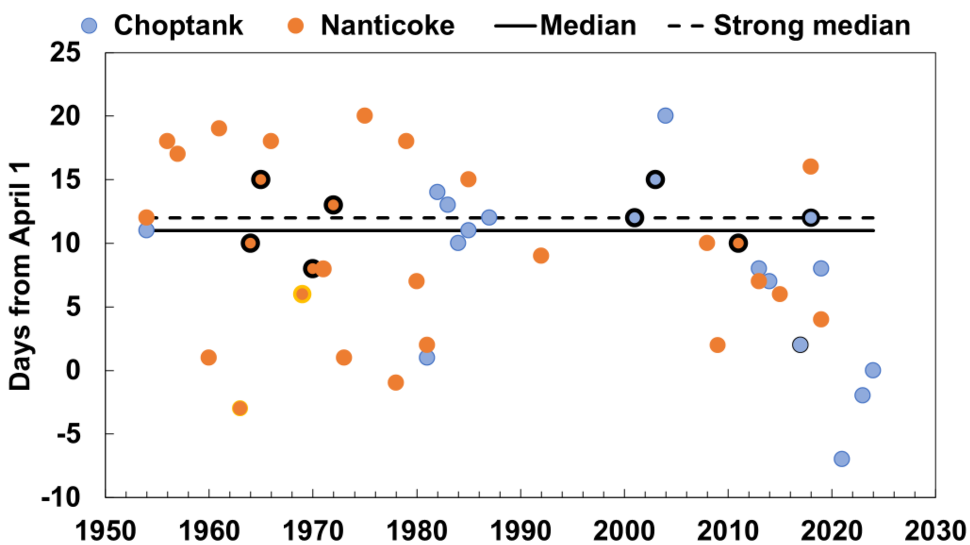


Figure 2.1.14. Days from April 1 (day = 0) that 16ºC was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2024. Median = median day for both rivers combined during 1954-1999. Thick point borders indicate strong year-classes and the dashed line is the median milestone date for strong year-classes.

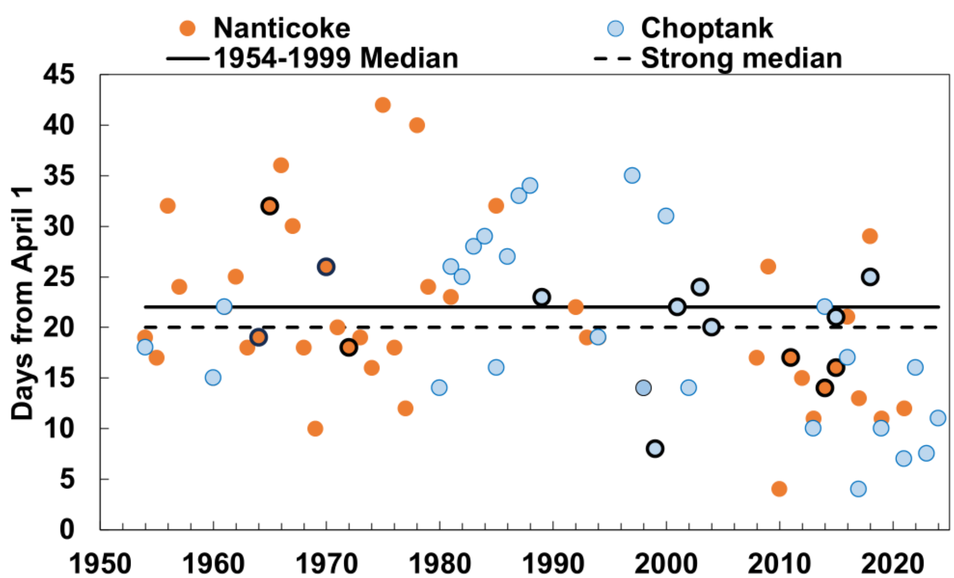


Figure 2.1.15. Days from April 1 (day = 0) that 20ºC was reached in Choptank River and Nanticoke River Striped Bass ichthyoplankton surveys during 1954-2024. Median = median day for both rivers combined (day 41) during 1954-1999. Thick point borders indicate strong year-classes and the dashed line is the median milestone date for strong year-classes.

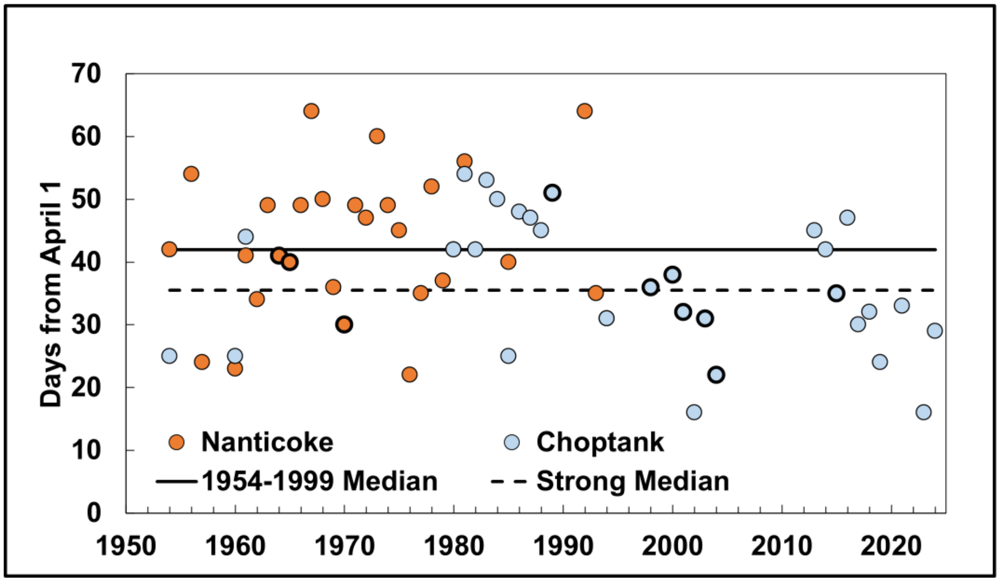


Figure 2.1.16. Days from April 1 (day = 0) that 12ºC, 16ºC, and 20ºC were reached in the Choptank River during 1954-2024. Star indicates a strong year-class.

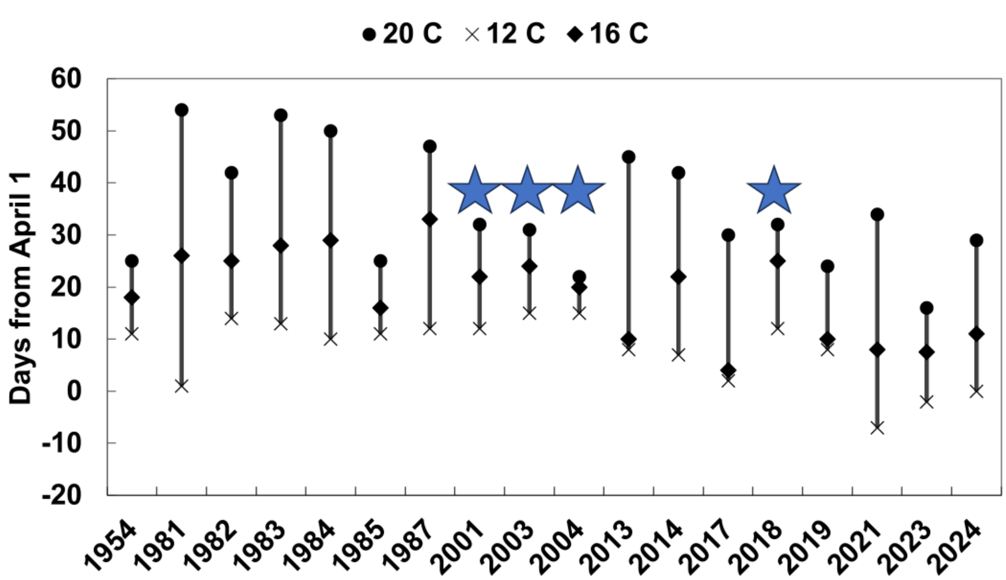


Figure 2.1.17. Trend (dotted line) in day 12°C was reached – day first egg was collected in Nanticoke and Choptank rivers, 1954-2024. April 1 = day 0.

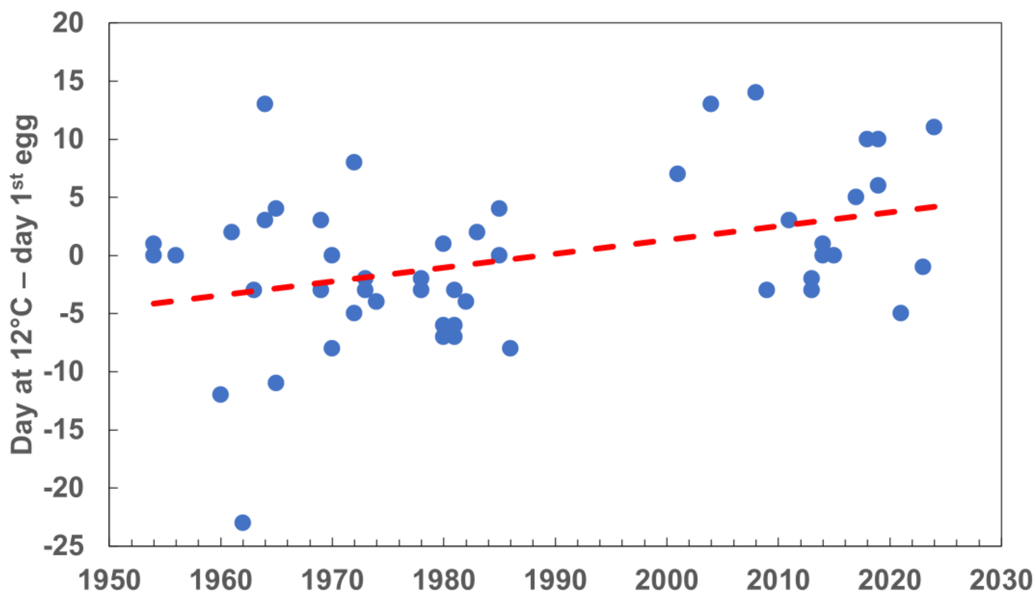


Figure 2.1.18. Trends in winter air temperature in Baltimore (regional indicator of winter conditions) and the winter North Atlantic Oscillation index (NAO) during 1954-2024.

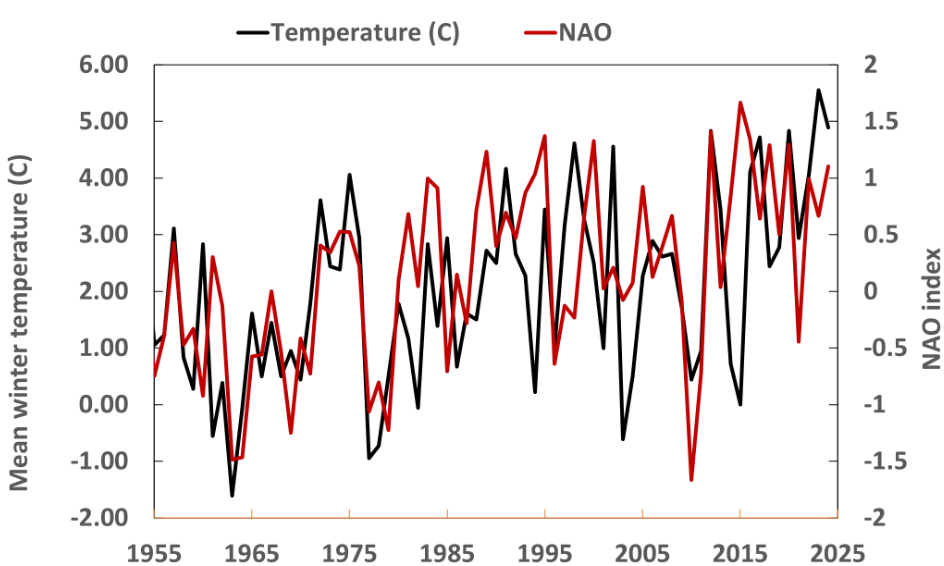


Figure 2.1.19. Observed and predicted days when 12°C spawning temperature milestone was reached on the Choptank and Nanticoke rivers versus winter intensity.

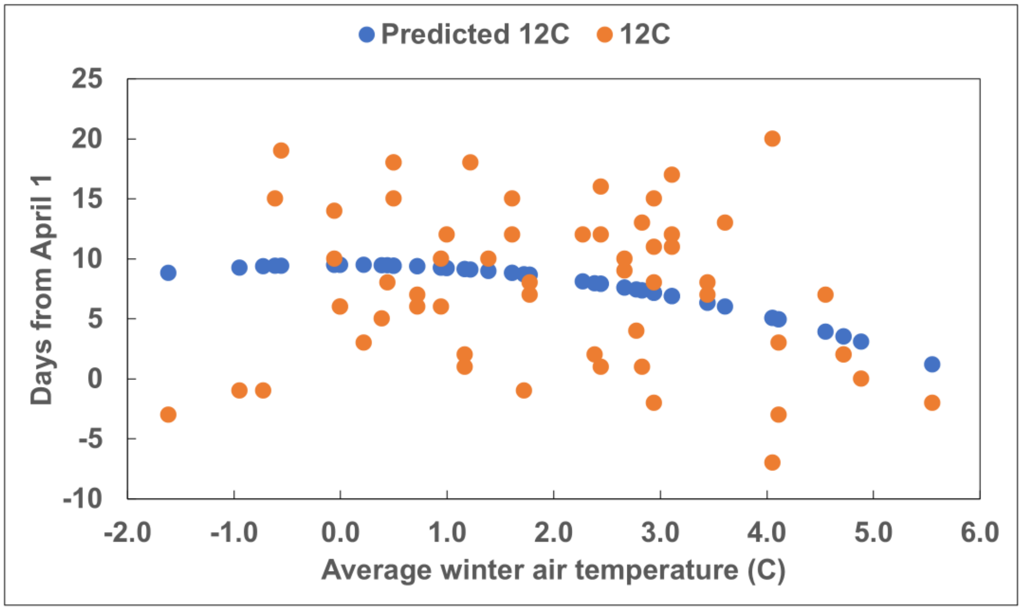


Figure 2.1.20. Observed and predicted days when 16°C spawning temperature milestone was reached on the Choptank and Nanticoke rivers versus winter intensity.

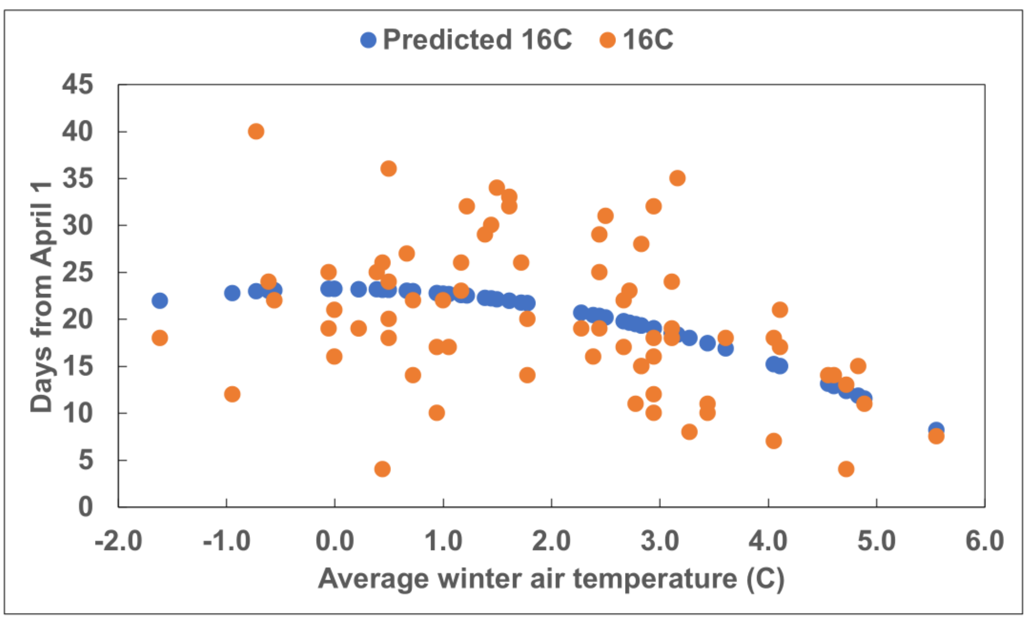


Figure 2.1.21. Observed and predicted days when 20°C spawning temperature milestone was reached on the Choptank and Nanticoke rivers versus winter intensity.

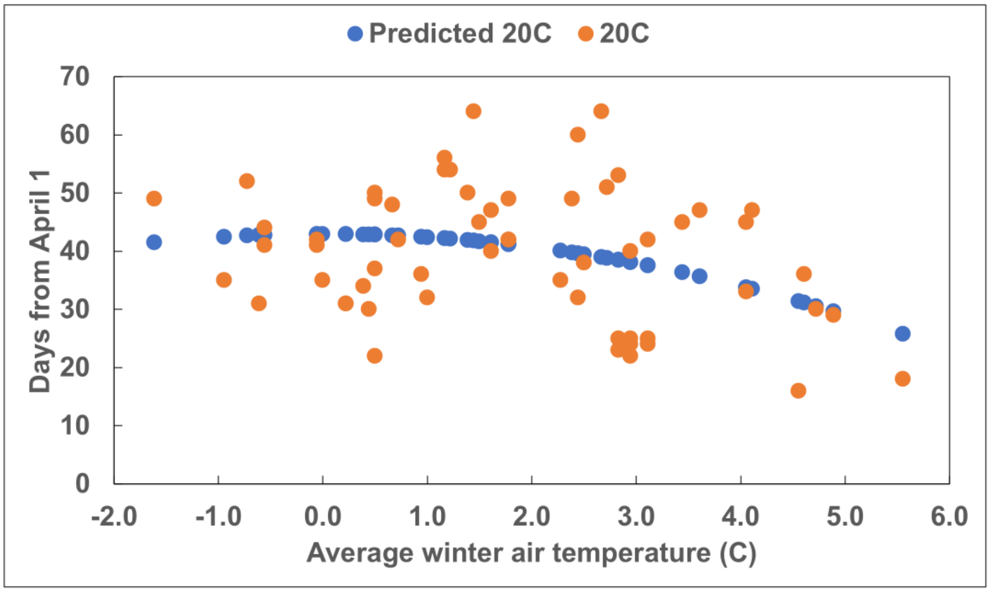
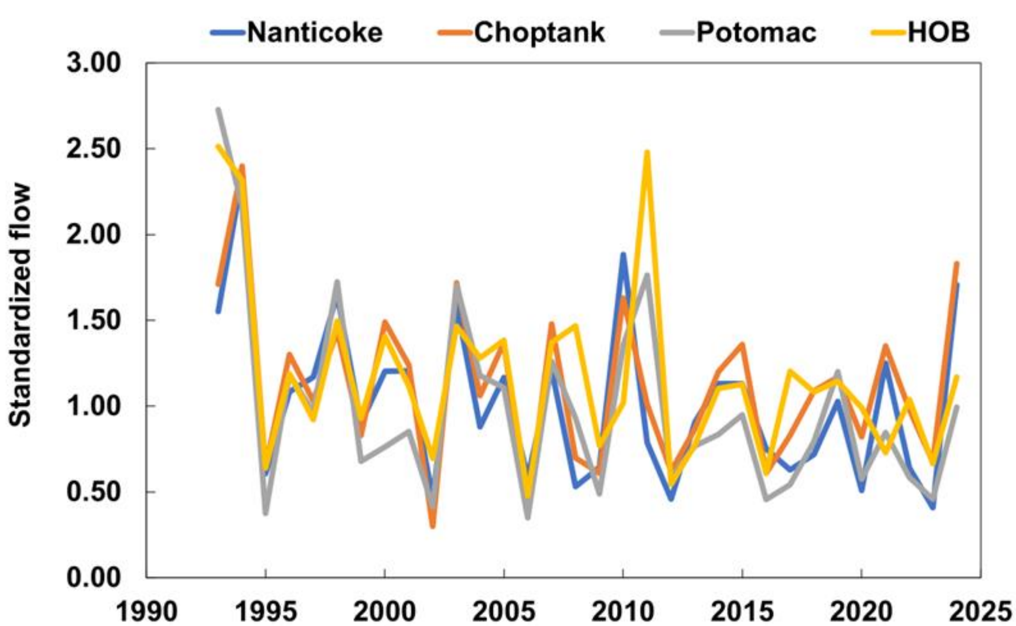


Figure 2.1.22. Two-month average flows for months prior to and including spawning season during 1993-2024 standardized to their averages for years in common during 1957-2020. HOB = Head-of-Bay.



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
2024	45	78	0.58	0.06	0.10	0.67	0.48

MD – Marine and estuarine finfish ecological and habitat investigations

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

Section 2.2: Investigation of Striped Bass spawning and larval habitat status in Maryland

Influence of feeding on zooplankton on Striped Bass postlarval mortality, growth, and year-class success in Choptank River, Maryland, during the 1980s and 2023-2024

James H. Uphoff, Jr., Shannon Moorhead, Marisa Ponte, Jeffrey Horne, Alexis Park, and Robin Minch

Our investigation of the influence of zooplankton feeding on Striped Bass postlarvae during 2023-2024 was accepted for publication in the American Fisheries Society's Marine and Coastal Fisheries Striped Bass themed issue. We have provided link to the paper and its abstract in lieu of a section that featured an earlier draft.

Standard link:
https://academic.oup.com/mcf/article-abstract/doi/10.1093/mcfafs/vtaf047/8392953?utm_source=authortollfreelink&utm_campaign=mcf&utm_medium=email

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https://academic.oup.com/mcf/article-abstract/doi/10.1093/mcfafs/vtaf047/8392953?utm_source=authortollfreelink&utm_campaign=mcf&utm_medium=email&guestAccessKey=0c3d6d5f-9ac9-4e3e-ba0e-3a0f9e93d1e3

Abstract

Objective: We examined first-feeding Striped Bass postlarvae (larvae that had absorbed their yolk-sac) in Choptank River during 2023-2024 to address whether their feeding success on zooplankton could be a major factor behind a series of poor year-classes during 2019-2024.

Methods: We estimated Choptank River Striped Bass postlarval feeding incidences on primary zooplankton prey and their associations with daily instantaneous growth (G) and mortality rates (Z) of postlarvae from seven 1980s surveys (low and high mortality and poor to strong year-classes) to establish criteria to evaluate feeding during 2023-2024. Distributions of larvae by water temperature and conductivity during 1980s surveys were used to direct 2023-2024 postlarval collections for feeding analysis.

Result: Feeding incidences of first-feeding Striped Bass postlarvae on copepods in Choptank River during 2023-2024 were high; feeding incidence on cladocerans was also high in 2024. Estimates of a proxy index for postlarval Z during 2023 and 2024 were low. However, year-class success was dismal during 2023 and low in 2024

Conclusion: This feeding investigation did not encompass the entire 2019-2024 drought in year-class success, but 2023-2024 surveys did not indicate a consistent, prominent role for feeding success of postlarvae.

Impact Statement: High feeding incidence of first-feeding Striped Bass postlarvae on zooplankton and low mortality did not always translate to better year-class success during the 1980s and 2023-2024. A prominent role of poor larval feeding success on zooplankton was not suggested for continuous poor year-class success during 2019-2024.

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

Jeffrey Horne, Marisa Ponte, Shannon Moorhead, Robin Minch, Marek Topolski, 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 uses 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 used 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). Soil compaction has 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 Chesapeake 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). Using impervious surface (IS) as a proxy for development, 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. ISRPs were based on dissolved oxygen (DO) criteria and associations and relationships between watershed impervious surface, 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 exceeds 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. (2024) 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 2024, we continued to evaluate summer nursery and adult habitat for recreationally important finfish in tidal-fresh 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, a development proxy equivalent to IS) on the annual median bottom DO among subestuaries sampled during 2003–2024. We evaluated target species presence-absence and abundance, total abundance of finfish, and finfish species richness. We continued to examine Tred Avon River, a tributary of the Choptank River located in Talbot County, which has been sampled consistently since 2006 (Table 3-1; Figure 3-1). We also returned to five previously sampled systems. Two were tidal-fresh subestuaries of the Potomac River: Mattawoman Creek, previously sampled from 1989 to 2016 and 2022 to 2023, and Piscataway Creek, previously sampled from 2003, 2006 to 2007, and 2009 to 2014. The remaining subestuaries were mesohaline: Magothy River on the Western Shore, previously sampled in 2003; Miles River on the Eastern Shore, previously sampled from 2003 to 2005 and in 2020, 2023; and West-Rhode River, mesohaline subestuaries located on the Western Shore, previously sampled from 2003 to 2005 (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 2024.

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. 2024; Topolski 2015). Estimates of C/ha were used for analyses of data from subestuaries sampled during 2024 (Table 3-2). Maryland Department of Planning (MD DOP) only has structure estimates available through 2020; 2021–2024 estimates are extracted from MD DOP property sales data. 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 3-2) 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 (Uphoff et al. 2024). 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 (Uphoff et al. 2024).

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 survey median bottom DO among mesohaline systems sampled during 2003–2024 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 has not exhibited a negative response to development in the other salinity categories (Uphoff et al. 2024).

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 revisited previously sampled historical sites at each of the subestuaries unless they were no longer accessible. Sites were sampled once every two weeks during July–September, totaling six visits per system during 2024.

The number of trawl and seine samples collected from each system varied based on 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. Some historic seine sites could not be sampled due to permanent obstructions, dense submerged aquatic vegetation (SAV), or lack of shore/beach availability; these were replaced with nearby beaches if possible. 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 each sampling event at each station using a YSI 556 MPS water quality meter, barring any equipment issues. 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 site. Weather, tide state (flood, ebb, high or low slack), and time were recorded for all sites.

Water Quality Analysis - Surface water temperatures were evaluated for differences between subestuaries using an Analysis of Covariance test to account for temperature changes due to date. Post hoc analysis was performed with a Tukey test and a Duncan multiple range test to determine which subestuaries were driving significant results. With a new database compiled in the last year, the opportunity existed to do deeper analysis of the longest continuous dataset, 2006-2024 in the Tred Avon River. Analysis of Variance was used to investigate changes in mean surface water temperature in Tred Avon by year, as well as linear regression to observe the overall trend and relationship. When looking at the effect of station on individual system temperatures, an ANOVA test was conducted and followed up with a post hoc Duncan's multiple range test. Lastly, 2024 surface temperatures were checked for impacts of salinity class, development (using both C/ha and % IS as metrics), and gear type (seine or trawl). This was done using a regression or student's t test, depending on data type.

Dissolved oxygen concentrations were evaluated against the biologically significant target of 5.0 mg/L and threshold of 3.0 mg/L established by Batiuk et al. (2009) for Chesapeake Bay living resources. 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 by MDE 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 does 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:

$$Vtarget = (Ntarget / Ntotal) \bullet 100; \text{ and,}$$

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

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.

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 waters. Trajectories of C/ha since 1950 were described for watersheds of subestuaries sampled in 2024. Annual median bottom DO (depth most sensitive to violations in mesohaline subestuaries) at each station was calculated for tributaries sampled in 2024 and plotted by year sampled. We explored the association between subestuary median annual bottom DO for all systems sampled by FEAD 2003-2024 and annual system C/ha with Pearson's correlations. Quadratic regression was used to assess relative impact of agricultural land use on bottom DO.

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 used Pearson's correlation analysis to investigate mean surface temperature with median surface DO, median bottom temperature with median bottom DO, and C/ha with surface and bottom DO for each salinity class. We chose annual survey medians 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. We first examined the data for 2024 systems alone, then expanded to all 19 mesohaline subestuaries sampled by FEAD since 2003 to include a wider range of development levels. Quadratic regression was also performed on the full data set, separated by region (Eastern and Western shore) to determine differences in the relationship between these two factors between the two sides of the Chesapeake Bay. At the smaller scale, an ANOVA examined variations in mean bottom DO among stations in 2024 subestuaries to determine whether stations within each subestuary were significantly different from one another. The overall median DO was calculated for all time-series data available for each 2024 subestuary and used to detect how annual station DO compared with the time-series median.

Measurements of pH were not made prior to 2006; some years of data had pH probe issues and measurements were not available. Due to the logarithmic nature of pH, readings were log transformed into H+ activity before being analyzed, then converted back to pH for reporting, following Kuna-Broniowska and Smal (2017). Relationships between surface and bottom pH values and DO at relative depth were assessed using Pearson's correlation and linear regression, however, due to non-normality of the pH/H+ data, analysis of variance was inappropriate. We also examined plots of salinity, pH, and Secchi depths within subestuaries by year. Impacts of development on Secchi depth were investigated by linear regression, using both C/ha and percent agricultural land use as a representative metric.

Finfish Community Sampling – Surveys focused on thirteen target species of finfish that fell within four broad life history groups: anadromous (American Shad, Hickory 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 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 2024).

Striped Bass and Yellow Perch were separated into two age categories, juveniles (JUV, <120 mm) and adults (ages 1+, >120 mm). White Perch were separated into three age categories based on size and life stage: juveniles (<120 mm), small adults (ages 1+ fish measuring >120 mm to < 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 the 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.

Finfish Community and Target Species Metrics - Basic metrics of finfish community composition and target species were estimated for subestuaries sampled: catch of all species and by gear, catch of target species and by gear, geometric mean (GM) catch of all species, geometric mean (GM) catch of target species, and total number of species (species richness) and target species. The GM of seine and trawl catches was the back-transformed mean of loge-transformed (+1) catches (Ricker 1975; Hubert and Fabrizio 2007; Durell and Weedon 2024). The GM is a more precise estimate of central tendency of finfish catches than the arithmetic mean but is on a different scale (Ricker 1975; Hubert and Fabrizio 2007; Durell and Weedon 2024).

Catch distributions of all finfish and target species were also analyzed by using Ward's Minimum Cluster Analysis to look at clustering of the data by subestuary and gear to describe similarity of assemblages among subestuaries. Since finfish data tends to have high variances, variable standardization for each species of finfish was performed using a Euclid distance matrix. This matrix was used to calculate the Ward's Minimum Cluster Analysis for each subestuary. This procedure creates a dendrogram for the specific gear and calculates the R^2 among subestuaries. The higher the R^2, the closer finfish assemblages in these systems are to one another (Singh et al. 2011; Bellwood et al. 2006).

We plotted geometric means of bottom trawl collections against C/ha by salinity class for all years in our database (2003-2024) to determine the effects of development on finfish abundance in the subestuaries. Median GM by development category (below target, target to threshold, and above threshold) were calculated to determine the relationship of development and bottom trawl GM. Finally, the proportion of positive tows with finfish was calculated to determine the relationship of development and the presence-absence of finfish in the bottom trawls; linear regression was used to analyze the relationship. Proportion of positive tows was also compared to median bottom dissolved oxygen with linear regression. Median presence-absence (P-A) of all finfish was also analyzed by dissolved oxygen groups (below threshold, threshold to target, and above target). Calculations of the presence-absence of target species will be included in future reports.

Tidal-Fresh Subestuary Metrics – Mattawoman and Piscataway Creek were directly compared to determine impacts of development on the finfish communities of each system. Catch distributions were examined to determine if differences were evident in species composition. Species richness and a Sorensen Similarity Index (S) were used to evaluate the two systems (Sorensen 1948) using the following equation:

$$S = 2a / (2a + b + c);$$

where S is the Sorensen similarity index, a is the number of species shared between the two samples, b is the number of species unique to the first sample, and c is the number of species unique to the second sample.

Geometric means were also calculated from the bottom trawls in these systems. We noted which target species were consistently sampled and summarized the GM by salinity type since some important ecological attributes (DO, SAV, etc) appeared to reflect salinity class. A two-sample t-test was used to examine any differences between the GM of juvenile or adult White Perch and Spottail Shiners in the two systems over time.

Mesohaline Subestuary Metrics – Bottom trawl and beach seine samples from the Magothy River, Miles River, Tred Avon River, and West-Rhode River were directly compared to determine impacts of development on the finfish communities. Species Richness and a Sorensen Similarity Index were used to evaluate the different systems. Catch distributions were analyzed to determine differences in species composition.

Geometric means were calculated from the bottom trawls and beach seines in these systems. We noted which target species were consistently present in a sample and summarized the GM by salinity type since some important ecological attributes (DO, SAV, etc) appeared to reflect salinity class.

Proc GLM in Statistical Analysis Systems (SAS) was used with a Tukey's test to evaluate differences in log-transformed catches (+1) of target species that are consistently encountered in sampling. A Duncan's grouping test was used to evaluate the differences for Bay Anchovy.

Forage Target Species Metrics – Basic fish metrics were calculated for the forage target species: Spot, Atlantic Menhaden, Eastern Silvery Minnow, Spottail Shiner, and Gizzard Shad. Geometric means were calculated for each of the species over time. These estimates were used to determine if there were changes in overall target species abundance in all systems combined. A Pearson's correlation analysis was used to evaluate GM of Spot (bottom trawls) and Atlantic Menhaden (beach seines) over the time series. The annual proportion of positive hauls for Atlantic Menhaden over time was calculated for each year of data.

Annual GM of tidal-fresh target species, Eastern Silvery Minnow, Spottail Shiner, and Gizzard Shad were calculated over the time series. Future reports will examine the changes of observed GM and presence/absence for these species with other studies in Chesapeake Bay. Bay Anchovy is the only estuarine resident forage species with abundance estimates (GM) calculated in this report.

Population Dynamics of Select Target Species - There has been increasing interest from anglers and the legislature in the distribution of juvenile Striped Bass in subestuaries of the mid-Bay region which have not been covered by the existing Striped Bass Juvenile Abundance Index (JAI) survey (Durell and Weedon 2024). Mid-Bay subestuaries of concern have been intermittently surveyed by Fisheries Ecosystem Assessment Division (FEAD) and its predecessors since 2003. The Striped Bass Program and FEAD use the same beach seine and sampling methodology; therefore, direct comparisons could be made between our data and the Striped Bass Juvenile Abundance Index (JAI) survey (Durell and Weedon 2024). Annual subestuary geometric means of young of year (YOY) Striped Bass were calculated for historical and current data from our surveys. Pearson's correlation was used to compare trends in GMs among mid-Bay subestuaries with Baywide, Upper Bay and Choptank River JAIs.

White Perch presence-absence was analyzed based on data from the bottom trawls. Juvenile and adult data from all years and all systems were analyzed, and a linear regression analysis was developed to compare juvenile and adult White Perch P-A to development and median bottom dissolved oxygen. Tred Avon River was also analyzed independently since there was a long-term data set available for this system (2006-2024).

A modified Proportional Stock Density (PSD; Anderson 1980; Anderson and Neumann 1996; Neumann and Allen 2007) was calculated using bottom trawl catch data for White Perch in subestuaries sampled each year (and in 2024) 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 to estimate a modified PSD since we did not measure small adults. The stock length category minimum for White Perch is 130 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 120 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.

Results and Discussion

2024 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 in this tidal-fresh Potomac tributary more than doubled from 0.44 C/ha to 1.00 C/ha. 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 3-2). We returned to sample Mattawoman Creek from 2022 to 2024 after a six-year hiatus, and C/ha was estimated at 1.04. All historical trawl stations were sampled in 2024, 4 trawl sites per sample period for a total of 24 trawls (Figure 3-2, Table 3-1). This monitoring was an important part of Maryland DNR's 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 from 2022 to 2024 was a response to 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.

Piscataway Creek is a tidal-fresh tributary of the Potomac River located in the suburbs emanating from Washington D.C. This system was previously sampled 2003, 2006-2007, and 2009-2014 (Table 3-1). This system was once again sampled in 2024 to use as a comparison with Mattawoman Creek, and to indicate where Mattawoman Creek could be heading with continued development. Development in the Piscataway Creek watershed is currently 1.61 C/ha (Table 3-2), whereas Mattawoman Creek is 1.04 C/ha. Due to shallow depth (< 2m) at all 3 trawl sites in Piscataway Creek, only surface water quality measurements were collected in 2024. In the decade since our last sampling, all 3 previous seine sites were rendered inaccessible by shallowing and heavy SAV growth, so sampling was limited to the 3 historical trawl sites (18 total trawls; Figure 3-2, Table 3-1). An MDE Biological Stressor Identification Analysis concluded that increased sedimentation as a result of urban land use was the main source of biological impairment in Piscataway Creek (MDE 2015). Continued sediment deposition and erosion has likely resulted in shallowing of the upper portion of this subestuary. Trawl station 01 (Figure 3-2) may have to be relocated in the coming years due to inaccessibility by boat.

Miles River is a mesohaline tributary located Mid-Bay on the Eastern shore. It was previously sampled in 2003–2005 when C/ha ranged from 0.23 to 0.24 (Table 3-2). In 2020 we returned to sample the Miles River - C/ha had slightly increased to 0.27. We once again returned to sample Miles River in 2023 and 2024 due to concerns within the local fishing community about decreasing observations of adult White Perch in 2022, to evaluate Striped Bass spawning success in 2024, and to further explore its idiosyncratic DO dynamics that switch from hypoxic to non-hypoxic from year to year. Development remained rural at 0.27 C/ha. Historical trawl stations were sampled, and seine station 01 was replaced with a nearby beach, therefore, three seine stations were once again sampled in 2024 (24 total trawl samples and 18 total seine samples; Figure 3-2, Table 3-1; see Uphoff et al. 2023 for additional analyses of Mid-Bay subestuaries).

Magothy River is a mesohaline tributary located on the western shore of the Mid-Bay area and was previously sampled in 2003 when C/ha was 2.68 (Table 3-2). In 2024, we returned to sample the Magothy River and C/ha was 2.95. We returned to this subestuary to assess changes during the past 20 years, to compare a high-development system directly with other less developed mesohaline systems, and to evaluate juvenile Striped Bass habitat use in the Mid-Bay region. In the years since last sampling, seine station 01 was replaced with a seawall, and a new seine station 01 was selected at a nearby beach (Figure 3-2). All other historic seine and trawl sites were sampled, maintaining 4 seines and 4 trawls per sampling period (24 samples of each total; Table 3-1).

West-Rhode River is a mesohaline tributary located on the western shore of the Mid-Bay area and was previously sampled from 2003 to 2005, when C/ha ranged from 0.55 to 0.56 (Table 3-2). In 2024, we returned to sample the West-Rhode River and C/ha was 0.62. The sampling of this subestuary occurred to evaluate changes in the system, to directly compare with other mesohaline systems, and to evaluate juvenile Striped Bass habitat use in the Mid-Bay region. Due to their close proximity (Figure 3-1), our analysis treats the West and Rhode rivers as one system. Rhode River seine station 02 became overgrown by marsh and was relocated to an adjacent beach. All other historical seine and trawl sites were sampled, for a total of 2 trawls and 2 seines in Rhode River, and 2 trawls and 1 seine in West River (24 total bottom trawls and 18 total beach seines; Table 3-1).

Tred Avon River (0.80 C/ha in 2024; Figure 3-1) reached the target for rural development (C/ha = 0.31) in 1972 and remains just under the 10% IS (C/ha = 0.84) threshold for suburban watersheds (Table 3-2). Our sampling in this mesohaline Eastern shore subestuary began in 2006 when development was at 0.69 structures per hectare, one year ahead of a substantial development project. We have monitored Tred Avon River continuously for the last 18 years in anticipation of DO and fish community changes as its watershed continues to develop. 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. We returned to the same historical sampling sites this year – all have been stable since 2006 (4 sites each, 24 total trawls and seines each; Table 3-1).

2024 Land Use and Development Summary - The six subestuaries sampled in 2024 ranged in land development from rural (0.26 C/ha, Miles River) to urban (2.95 C/ha, Magothy River; Table 3-2; Figure 3-3). Magothy River exceeded rural levels of development in 1955, suburban in 1968, and highly developed suburban in 1976. It is one of the most developed Chesapeake Bay subestuaries we have surveyed with 2.95 structures per hectare in 2024. Piscataway Creek is Maryland's fourth most developed Chesapeake Bay subestuary, currently at 1.61 structures per hectare. It passed a rural level of development by 1967, suburban in 1987, and highly developed suburban in 2006. Though Mattawoman Creek was still rural until 1987, it had grown to a suburban level in 2008, continuing to 1.04 C/ha in 2024. Tred Avon surpassed the rural threshold in 1972 but remains just below suburban development (0.80 C/ha in 2024). West-Rhode River surpassed the rural threshold in 1989 but remains comfortably below suburban development (0.62 C/ha). Miles River is the only subestuary sampled in 2024 that remains rural (0.26 C/ha). Overall, development in Maryland's portion of the Chesapeake Bay watershed has grown from 0.17 C/ha in 1950 to 0.83 C/ha in 2024 and now verges on suburban (Table 3-2; Figure 3-3).

Land use composition for subestuaries sampled in 2024 covered a wide range reflective of their relative development - Miles River has the lowest urban land use of all systems sampled and the highest agriculture, and Magothy has the highest urban percentage and lowest agriculture. In the six subestuaries, agricultural land use varied from 1.2 - 48.87%; forest, 20.42 - 52.83%; wetlands, 0.01- 1.14%; and urban, 23.39 - 77.87% (Table 3-2). Looking at the associations between development and the various land use types estimated by DOP, the strongest correlation was between structures per hectare (C/ha) and urban land cover (r = 0.96; $P < 0.0001$; Table 3-3). Correlations of C/ha with agriculture (r = -0.79; $P < 0.0001$) and wetland (r = -0.69, $P = 0.0001$) were negative and moderate. Negative correlations were found between urban land use and both agriculture (r = -0.55, $P < 0.0001$) and wetland (r = -0.71, $P = 0.0001$) categories, as well as between agriculture and forest (r = -0.79, $P = 0.0001$); the correlation of urban and agriculture was considered moderate and the remaining correlations were weak. The remaining pairings of land use categories were poorly correlated. Correlations among land uses and year were weak, although annual forest land use estimates and year bordered on moderate (r = -0.34; $P = 0.046$), indicating a decline in percentage of forest land in all systems over time.

2024 Water Quality Summary - Table 3-4 provides summary statistics for surface and bottom water quality for each tributary and subestuary sampled in 2024, including temperature (°C), dissolved oxygen (mg/L), specific conductance (µS/cm), salinity (‰), and pH. Each metric is described by mean, standard error (SE), median, maximum, minimum, and total count.

Water Temperature - Surface water temperatures ranged from 21.3°C to 32.33°C during the 2024 survey. Bottom water temperatures ranged from 21.8°C to 30.8°C in 2024 for all subestuaries. Surface water temperatures from all six systems show a gradual decline in temperature from June to September. There was an outlying heat wave the week of 7/29 – 8/7 (Figure 3-4); all systems sampled during this week show a similar jump in temperature regardless of region or development. An ANCOVA investigating the impact of river system, accounting for changes due to date, found significant results for both factors ($P < .0001$, $R^2 = 0.887$; Table 3-5a). Even after compensating for the expected change in temperature over the course of the summer, water temperatures were not the same in all systems. Post hoc Tukey's HSD testing found that the mean temperature in Tred Avon River was higher than some subestuaries, and West-Rhode River was lower than others, though neither was significantly different from *all* other subestuaries. A follow up Duncan's MRT, which doesn't account for secondary factors, displays similar results in Table 3-5b. The surface river temperatures were as follows in descending mean temperature order: Tred Avon River, Piscataway Creek, Magothy River, Miles River, Mattawoman Creek, and West-Rhode River.

Subestuaries with the most developed watersheds tended to have higher temperatures, but Tred Avon River and Mattawoman Creek were exceptions. This may be influenced less by the overall watershed development than the landscape immediately surrounding each river. Much of Mattawoman Creek's watershed adjacent to the subestuary is in forest and wetlands and the adjacent portion of the watershed with fluvial waters lies within the Watershed Conservation District (a large tract of low development) before suburban development in Waldorf. The headwaters of Tred Avon River, on the other hand, are surrounded by localized development (shopping centers, housing, a marina, and light industry), so precipitation is warmed on the hot pavement before entering the river. Though the uppermost 2 sites in Tred Avon did have higher average temperatures in 2024 than those downstream, the surface temperatures between sites were not statistically different ($P = 0.67$). Overall, there were no statistically significant differences in surface temperatures between subestuaries sampled in 2024, based on development (using either C/ha or % IS as a metric), salinity class, or gear type.

With the highest average temperature and the longest continuous sampling effort, Tred Avon River surface water temperatures were examined for changes between 2006 and 2024 (Figure 3-5). Despite inter-annual variability, a linear regression fitted to the continuous Tred Avon temperature data shows a weak upward trend in surface water temperature over the last 20 years (slope = 0.08, $P < .0001$, $r^2 = 0.20$). An ANOVA of among year differences showed a significant difference between yearly average temperatures for the full Tred Avon time series ($F=5.65$, $P < .0001$, $r^2 = 0.10$). However, 2024 was only significantly warmer than 6 years: 2007-2009, 2013-2014, and 2017, all of which were below the overall mean temperature of 27.6°C. The maximum median temperature over the nearly 20-year time series was 30°C in 2016, followed immediately by the minimum mean temperature in 2017 at 26.4°C. Subsequent years have less variation around the trendline than the rest of the time series.

Dissolved Oxygen - In 2024 surface dissolved oxygen ranged between 4.60mg/L (Magothy River) and 13.24 mg/L (Piscataway Creek; Table 3-4). Only 4 surface measurements were below the target DO of 5 mg/L this year (Table 3-6), and only in the mesohaline systems – two in Magothy River, and one each in West-Rhode River and Tred Avon River - all at up-river sites. However, bottom DO, a primary fish habitat concern, fared less well. Bottom DO in 2024 ranged from 0.00 mg/L (Magothy River), up to 8.71 mg/L (Mattawoman Creek; Table 3-6; Figure 3-6). Magothy River had the highest percentage of DO violations: 25% of measurements from all depths were below the 5mg/L target, 71% of bottom DO readings were below target, and 45% were below the 3mg/L threshold. This is a small improvement over the last sampling in Magothy River in 2003 (89% and 67% violations respectively). West-Rhode River had over 45% of bottom DO readings below target, and 13% below threshold. Tred Avon River, which has previously had up to 71% target and 21% threshold violations, dropped to 37.5% and 4.2% in 2024. Target violations for the Tred Avon and West-Rhode Rivers were similar to previous years but showed a small decline in threshold violations compared to each system's average over the time series. While all four mesohaline rivers sampled in 2024 had readings under the target, Miles River showed the greatest improvement in DO violations, with only 5 readings out of 57 below 5 mg/L, and none below 3 mg/L (Table 3-6). This may be in part due to the lack of development in Miles River over the years – it remains rural at 0.27 C/ha and may have avoided much of the biological stress due to development that other systems have been subject to.

Tidal-fresh Mattawoman and Piscataway Creeks were the only subestuaries sampled that did not have any DO target or threshold violations this year (Table 3-6). Piscataway has not had any DO violations since sampling began in 2003, and Mattawoman has only had 2 readings below target since 2011 - in 2012 and 2022. This is likely due to heavy SAV growth in both systems. There were no threshold violations in median surface DO regardless of salinity class or development level in 2024 (Figure 3-8).

Over the full 2003-2024 time series, Magothy River has had the lowest median bottom DO (Table 3-7) of the systems visited in 2024. Annual median bottom DO has declined in the Tred Avon River since 2009, however, in 2024 the median bottom DO was above target for the first time since 2017. Mattawoman Creek annual median bottom DO has fluctuated between 6.45 and 9.15 mg/L without an apparent trend during 2003–2016 and 2022–2024. In 2024 all trawl sites at Piscataway Creek were below 2 m in depth, therefore bottom measures were not taken, but median surface DO was the highest of any subestuary at 10.15 mg/L. Since 2003, all sites at Piscataway Creek have been above the DO target at both surface and bottom depth. Despite more than adequate oxygenation, shallowing due to sediment accumulation may still be reducing available water column habitat for fish communities in this system (MDE 2015).

Median bottom DO measurements for the time-series of each subestuary sampled during 2024 are depicted in Figure 3-7, organized by salinity class and subestuary. Median bottom DO levels were all above the DO target in the tidal-fresh subestuaries, but 38% were below the target in mesohaline subestuaries. Magothy River was the only system with a median bottom DO below the threshold (1.1 mg/L in 2003), but all four mesohaline subestuaries sampled in 2024 have had past median DO readings below the target. Although Tred Avon was just above target this year (5.25 mg/L), since 2006 the median DO has ranged between 4.5 and 6.3 mg/L, only exceeding 6 mg/L once (2009). All median bottom DOs below 5 mg/L in Tred Avon River have occurred since 2017, which coincides with the increase in average temperature and decrease in variability shown in Figure 3-5. West-Rhode Rivers also fluctuate around the target, with median bottom DOs between 4.9 and 5.8 mg/L. In 2024 Miles River had the highest median DO in its time series at 5.94 mg/L – past measurements ranged between 3.3 and 5.3 mg/L in 2003-2023.

Dissolved Oxygen and Land Use - Examining associations between DO, temperature and development (C/ha) in 2024 subestuaries indicated that DO responded to temperature and C/ha differently depending on salinity classification (Table 3-8). In mesohaline systems, a decline in bottom DO correlated with increased C/ha ($r = -0.495$, $P = 0.005$), while in tidal-fresh systems there was no significant association of bottom DO with development. Conversely, surface DO showed a moderate increase with development in tidal-fresh systems ($r = 0.61$, $P = 0.001$), likely due to the heavy SAV coverage which occurred only in the tidal-fresh subestuaries included in this analysis. In mesohaline systems, however, there was a weak but not statistically significant association found between these two factors ($r=0.32$, $P=0.08$). Correlations were poor between temperature and DO at any depth or salinity class. Mesohaline subestuaries were where strongest stratification was expected, which exacerbates oxygen depletion at depth (Kemp et al. 2005; Murphy et al. 2011; Harding et al. 2016).

As development significantly impacted bottom DO in mesohaline systems only, further analysis focused only on this salinity class. Pooling together all four mesohaline systems sampled in 2024, agricultural land use had a significant positive impact on annual median bottom DO ($F=15.83$, $P = 0.0004$; Table 3-9), although agriculture only accounted for about 35% of the variability in DO readings. Expanding to all 19 mesohaline subestuaries sampled by FEAD since 2003 supports this conclusion from the 2024 subestuary data ($n = 99$; see Table 3-1 for subestuaries). A quadratic regression showed a moderate dome shaped relationship between annual median bottom DO and agricultural land use ($R^2 = 0.49$, $P < 0.001$; Table 3-10a; Figure 3-9). Agricultural coverage accounted for nearly 50% of the variation in bottom DO in mesohaline systems.

Evaluating the full 20-year data set of mesohaline DO readings by region of the Chesapeake Bay (Eastern vs Western shore) with quadratic regressions paints a different picture (Table 3-10b). Subestuaries on the Eastern shore showed no significant influence of agricultural land use on bottom DO ($F=1.37$, $P = 0.26$, $R^2 = 0.038$). However, there was a moderate relationship between these two factors on the Western shore ($F=14.63$, $P < .0001$, $R^2 = 0.56$). This was also the case when looking at 2024 data alone, though there were insufficient samples on the Western shore in 2024 ($n=6$) to draw any substantial conclusions. In the subestuaries sampled in 2024, on the Eastern shore there was little evidence of impact of agricultural land use on median bottom DO ($P=0.20$, $R^2=0.07$), while Western shore bottom DO was positively influenced by agriculture ($P=0.04$, $R^2=0.694$).

In both 2024 annual analysis and over the full time-series, development was predominant at low levels of agriculture (< 20% agricultural coverage). Agricultural coverage and C/ha are inversely correlated, so the positive trend of DO with agriculture when agricultural coverage was low was likely to reflect development's negative impact. Median bottom DO residuals were inspected and 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.

Salinity - Salinities in Magothy, Miles, Tred Avon and West-Rhode Rivers were within mesohaline bounds (5.0‰-18‰) in 2024 (Table 3-4). Salinity in Mattawoman and Piscataway Creeks was tidal-fresh (<0.5 ‰). Despite deviations into oligohaline salinities (0.5–5.0 ‰) in Mattawoman Creek in 2007 and 2023 due to below average rainfall (Ruiz-Barradas 2023; NOAA NCBO 2024), the median annual bottom salinity for the current year was nearly identical to the median bottom salinity over the full 2003-2024 time series (0.17‰ in 2024, 0.18‰ over all years). Surface salinity in Piscataway Creek has been stable within the tidal-fresh class over the 2003-2024 time-series, with all measured salinities falling at or below 0.21‰.
Annual median salinity measurements in Magothy River ranged from 5.4 – 6.46‰ for all years sampled; Mattawoman Creek, 0.1 to 1.0‰; Miles River, 9.1–14.3‰; Piscataway Creek, 0.1–0.18‰; Tred Avon River, 7.5–13.2‰; and West-Rhode River, 7.1–10.9‰.

pH - Measurements of pH were not available prior to 2006. Annual mean surface pH measurements in Mattawoman Creek ranged from 7.14 to 8.54 for all years sampled: Miles River, 7.78-7.89; Piscataway Creek 7.78-8.27; and Tred Avon River, 7.3-8.15. Average pH measurements for 2024 for systems not sampled between 2006 and 2023 were as follows: Magothy River 7.9, and West-Rhode River 8.05. There have not been notable trends observed in surface or bottom pH over the last 20 years in the systems sampled in 2024.

Pearson's correlations of 2024 surface and bottom pH measurements with DO at the same depths, showed strong negative associations between the two water quality measures at both depths (surface: r=-0.67, P<.0001; bottom: r=-0.86, P<.0001). These correlations remained strong within each system at each depth (all P<0.001). Variations in pH at surface and bottom depth are driven by changes in DO and carbon dioxide caused by respiration by SAV and phytoplankton (Su et al. 2021). This relationship can be more intense in bottom waters due to stratification – decreased water circulation causing hypoxic and acidification events. There did not appear to be any difference in this biochemical relationship among systems sampled.

Clarity - There was little improvement or deterioration of water clarity over time in the subestuaries sampled during 2024 (Figure 3-10). Median annual Secchi depths in Mattawoman Creek ranged from 0.50 - 1.3 m; Tred Avon River, 0.40 – 0.75 m; Miles River, 0.45 – 0.65 m; West-Rhode River, 0.59 – 0.70 m; and Piscataway Creek, 0.50 – 0.75 m (with one outlier in 2007 at 1.05 m). Both years that Magothy River was sampled had a median Secchi depth of 0.90 m.
A linear regression analysis of 2024 systems (data from sampling years 2003-2024) indicated a moderate negative relationship between median Secchi depth and % Agriculture (F=22.4, P<.0001, r^2 = 0.29), indicating that higher agricultural land use contributed to declining water clarity. The same analysis between median Secchi depth and C/ha found a weak positive relationship (P=0.01, F=6.8), but the low coefficient of determination (r^2 =0.12) indicated that development was not a major driver of variation in Secchi depth. Examining both salinity classes separately showed a second order polynomial trend between both development metrics and water clarity in mesohaline systems (R^2=0.49), with Secchi depth increasing with increased C/ha.

Conversely, in tidal-fresh systems, a second order polynomial line also best described the relationship between development and water clarity, but with a weak decline in Secchi depth as development increased (R^2=0.17). Overall, median Secchi depth was highly variable within systems and years, ranging from 0.4m up to 1.3m.

Although both C/ha and agricultural land use had a moderate impact on Secchi depth in all 2024 systems (r^2=0.49 and 0.46, respectively), in both cases the statistical relationship was driven largely by the 2 high-development outliers in Magothy River. Further sampling in suburban systems with development between 1-2.5 C/ha is needed to strengthen conclusions on the effect of development on water clarity.

2024 Finfish Community Summary - A total of 91,103 finfish representing 58 species were captured in beach seines and bottom trawls in 2024. Four target species were present in the top 5 species encountered for both gears combined: Spot, Atlantic Menhaden, Bay Anchovy and White Perch.

A total of 35,219 finfish representing 40 species were captured by beach seines in 2024 (Table 3-11). Target species collected from beach seines in 2024 were Alewife, Atlantic Menhaden, Bay Anchovy, Blueback Herring, Gizzard Shad, Spot, Striped Bass and White Perch. Atlantic Menhaden were the most abundant species captured in beach seines at 71% of the total catch (25,057 fish; Table 3-11).

A total of 55,884 finfish and 42 fish species were captured by bottom trawl during 2024 (Table 3-12). Twelve of 13 target species collected from bottom trawls in 2024 - Alewife, American Shad, Atlantic Menhaden, Bay Anchovy, Blueback Herring, Gizzard Shad, Eastern Silvery Minnow, Spot, Spottail Shiner, Striped Bass, White Perch, and Yellow Perch. Spot were the most abundant species captured in bottom trawls at 67% of the total catch (37,667 fish; Table 3-12).

Beach seine species richness for the mesohaline systems was greater in Miles River (N=29) than in Tred Avon River (27), Magothy River (23), and West-Rhode River (18). Species richness from bottom trawls for the tidal-fresh systems was higher in Piscataway Creek (N=23) than in Mattawoman Creek (N=21). In mesohaline subestuaries, bottom trawl species richness was greatest in Tred Avon River (N=23), followed by Miles River (18), West-Rhode (13), and Magothy River (8).

Ward's Minimum Variance Cluster Analysis was used to examine relationships of finfish community similarities within gears and among systems (Figure 3-11, Figure 3-12). In beach seines, Miles and Magothy Rivers clustered the closest together, indicating the fish communities were similar in beach seines (R^2 = 0.706). Tred Avon River and West-Rhode River clustered together, indicating some degree of similarity in the finfish communities (R^2 = 0.362). The finfish communities of West-Rhode River/Tred Avon River did not cluster with Miles and Magothy and were not similar to these systems (R^2 = 0.000; Figure 3-11). The similarity of the beach seine communities in West-Rhode River and Tred Avon River were likely attributable to the similar variety of species in these systems. The similarity of the beach seine communities in Miles and Magothy is likely attributable to the abundance of Striped Killifish and Mummichog. For the Ward's Cluster Analysis of bottom trawl catch, the tidal-fresh systems of Mattawoman and Piscataway Creek were similar to each other (R^2 = 0.361) and not clustered with the mesohaline systems (R^2 = 0.000). The abundance of freshwater species in the catch likely resulted in the differences between systems by salinity class. The bottom trawls in Magothy, Miles, West-Rhode Rivers were all clustered together (R^2 ranged from 0.799 to 0.921). Tred Avon River was less similar to these systems but still clustered together with these systems (R^2 =

0.601; Figure 3-12). The finfish community structures of these systems were mostly influenced by the same species in each system (Spot, Bay Anchovy, Atlantic Croaker, etc.); however, the difference in Tred Avon River is likely attributed to higher catches of Oyster Toadfish, Summer Flounder, Weakfish and Spotted Seatrout.

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, 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 in 2018 and 2019 (Uphoff et al. 2018).

Geometric mean (GM) of total catch per seine haul ranged from 109 to 234 among the four subestuaries sampled with a beach seine during 2024 (Table 3-11). Miles River had 18 seine samples at 3 stations; Magothy River, 24 seine samples at 4 stations; West-Rhode River, 18 seine samples at 3 stations and Tred Avon River, 24 seine samples at 4 stations (Table 3-11; Figure 3-2). The number of samples varied among subestuaries due to the number of stations established and availability; extensive SAV has precluded seining in Mattawoman Creek and Piscataway Creek. A seine site was not available adjacent to trawl site 01 on the West River due to extensive bulkheading.

Subestuary GMs of all species caught in bottom trawls of all finfish were between 14 and 486 during 2024 and were heavily influenced by Spot (Table 3-12). Tred Avon River had the greatest GM (486), and Magothy River had the lowest GM (14; Table 3-12).

Bottom trawl GMs for all species combined did not exhibit an obvious decline with C/ha for oligohaline or tidal-fresh systems (Figure 3-13a). However, in mesohaline subestuaries GMs declined with C/ha during 2003–2024, and a negative threshold response was suggested at C/ha between 0.8 and 1.2 (Figure 3-13b; r^2=0.30, P<0.001 as a linear decline); this decline reflected the change to consistent low DO conditions in mesohaline bottom channel waters with increasing development. There was wide variation in GMs prior to the development threshold and they declined to very low levels afterward. Median trawl GMs calculated for mesohaline subestuaries with below target watershed development (C/ha=0.31) was 114 (N=50); it was 89 (N=33) with watershed development between the target and threshold (C/ha=0.84); and 10 (N=11) when development was greater than threshold (Figure 13b).

Presence-Absence (P-A) of finfish (all species combined) in the bottom trawls was examined to evaluate the impacts of development and to account for high variability in fish data. Proportions of positive tows were calculated for each mesohaline subestuary by year and compared to C/ha (Figure 3-14) and median bottom dissolved oxygen (mg/L; Figure 3-15). A linear regression of proportion of positive tows and C/ha had an r^2 of 0.47 (P<0.001) that indicated a moderate negative influence of C/ha on the P-A of finfish in the bottom trawls. A linear regression of proportion of positive tows and median bottom dissolved oxygen for mesohaline subestuaries had a r^2 of 0.06 (P=0.02) that indicated the influence of median bottom dissolved oxygen on P-A of finfish in bottom trawls was described poorly by a linear function. There was considerable variability above the threshold of 3.0 mg/L of median bottom dissolved oxygen; P-A of all finfish only exceeded 0.6 when DO was near or above 3.0 mg/L (Figure 3-15). Once C/ha crossed the thresholds of development or DO, bottom trawls no longer had finfish in all samples (Figures 3-14 and 3-15). Presence-absence of a finfish combined indicated persistence of habitat occupation, provided less data variability, and have been linked to finfish abundance (Gaston et al. 2000, Miranda 2022).

Target Species Summary - Total numbers of anadromous target species encountered in 2024 were American Shad (3), Hickory Shad (0), Blueback Herring (74), Alewife (4), and Striped Bass (191 YOY and 5 age 1+; Table 3-13). Total numbers of estuarine resident target species encountered in 2024 were White Perch (6,322 Juvenile, 701 Adult), Yellow Perch (8), and Bay Anchovy (6,645; Table 3-13). The total numbers of marine target species encountered in 2024 were Atlantic Menhaden (25,141) and Spot (40,225; Table 3-13). Total numbers of tidal-fresh target species encountered in 2024 were Spottail Shiner (566), Eastern Silvery Minnow (43), and Gizzard Shad (122; Table 3-13).

Distribution of Target Species in 2024. - Target species total 2024 catch for all systems combined was 80,050 finfish and comprised 87.9% of the total sample in both gears. Atlantic Menhaden comprised the highest catch of target species in beach seines at 25,057 fish (Table 3-14). Spot comprised the highest catch in bottom trawls at 37,667 fish (Table 3-15). The tidal-fresh system distributions were heavily influenced by Juvenile White Perch abundance.

Ward's Minimum Variance Cluster Analysis was used to examine relationships of target species finfish community similarities within gears and among systems (Figures 3-16 and 3-17). In beach seines (all mesohaline subestuaries), West-Rhode and Magothy Rivers clustered very closely together, indicating a high similarity in the target species communities (R^2 = 0.855). Miles, West-Rhode and Magothy Rivers also clustered well together (R^2 = 0.534), but the similarity was not as high as West-Rhode and Magothy River. Tred Avon River did not cluster with the other systems (R^2 = 0.000; Figure 3-16). In Tred Avon River, Blueback Herring were caught in the beach seine, which likely led to the difference in the target species compositions. Also to note, the Tred Avon River had the highest catch of adult White Perch in beach seines.

The Ward's Cluster Analysis was also used to examine the relationships of target finfish community similarities in the bottom trawl samples. The mesohaline systems of West-Rhode and Miles River were the most similar for target species catch (R^2 = 0.972). West-Rhode, Miles, and Tred Avon also clustered closely (R^2 = 0.933). These two cluster groupings of systems were also similar to the Magothy River (R^2 = 0.843). The tidal-fresh system of Mattawoman Creek clustered with the mesohaline systems, but the similarity was a lot lower than among those systems (R^2 = 0.578). This was likely influenced by catches of Spot in Mattawoman Creek - only two Spot were captured in Piscataway Creek. Piscataway Creek did not cluster with any of the other systems and was different from these systems (R^2 = 0.000). The difference in Piscataway Creek was the result of catching 11 of the 13 target species in this system. The next closest system was Mattawoman Creek where 8 of the target species were caught. The mesohaline systems clustered well together due to the presence of only three to five target species in these systems (Figure 3-17).

Comparison of Tidal-Fresh Systems Mattawoman Creek (1.04 C/ha) and Piscataway Creek (1.61 C/ha) - Bottom trawl samples from Mattawoman Creek and Piscataway Creek were directly compared to determine the impacts of development on the finfish communities of each system. Mattawoman Creek had a species richness of N=19 for all finfish species collected. For Piscataway Creek, species richness was N=22 for all finfish species collected. A total of 8 target species were collected in Mattawoman Creek, and 11 target species were collected in Piscataway Creek (Table 3-15). A Sorensen Similarity Index evaluation found S = 0.73, which indicates high similarity between the two systems with substantial species overlap. Distribution was heavily influenced by juvenile White Perch in both systems; catch distributions were created with juvenile White Perch removed (Figure 3-18). In these target species catch distributions, Mattawoman Creek was dominated by adult White Perch (35.6%) and Spottail Shiners (27.6%). Piscataway Creek was dominated by Spottail Shiners (55.1%) and adult White Perch (21.0%). Geometric means (GM) were calculated from bottom trawl data for select target species that were consistently captured in sampling to determine any differences in finfish abundance between the two systems. No significant differences were found in the GM of juvenile or adult White Perch in the two systems for 2024. No significant difference in juvenile White Perch GM was found between the two systems from 2003 to 2024; however, the adult White Perch GM was significantly higher in Mattawoman Creek during this time period ($P < 0.001$; Figure 3-19). Also, no significant differences were found in the GM of Spottail Shiners in the two systems for 2024.

Comparison of Mesohaline systems Magothy River (2.95 C/ha), Miles River (0.27 C/ha), Tred Avon River (0.80 C/ha), and West-Rhode River (0.62 C/ha). - Beach seine and bottom trawl samples from Magothy River, Miles River, Tred Avon River, and West-Rhode River were directly compared to determine the impacts of development on the finfish communities of each system. Magothy River had a species richness of N=23 for all species collected in beach seines and N=8 for bottom trawls. Miles River had a species richness of N=29 in beach seines and N=18 for bottom trawls. Tred Avon River had a species richness of N=27 in beach seines and N=20 for bottom trawls. West-Rhode River had a species richness of N=22 in beach seines and N=13 for bottom trawls. Target species catches are ranked by abundance in Table 3-11 for beach seines and Table 3-12 for bottom trawls. A total of six target species were collected in Magothy River, seven in Miles River, seven in Tred Avon River, and six in West-Rhode River (Table 3-13; Table 3-15). A Sorensen Similarity Index evaluation for both gears combined found the highest similarity between Miles and Tred Avon Rivers (S = 0.79). The lowest similarity was between Magothy and West-Rhode Rivers (S = 0.67). Most of the systems had a similarity of S = 0.71 to 0.76. Catch distribution was heavily influenced by Atlantic Menhaden in beach seine samples and Spot in bottom trawl samples (Tables 3-14 and 3-15). Spot and non-target species made up a large portion of beach seine catches after Atlantic Menhaden (1.5% to 12.7% and 9.2% to 53.5%, respectively). Bay Anchovy were the most commonly caught target species behind Spot in the bottom trawls (7.4% to 15.4% of total catch).

Geometric means were calculated from the beach seine or bottom trawl data for target species that were most consistently encountered in a system in 2024 to determine if any difference in target finfish abundance was present between the systems and their levels of development. The target species that were the most consistently sampled were Spot (bottom trawl and beach seine), Atlantic Menhaden (beach seine), and Bay Anchovy (bottom trawl). Atlantic Menhaden beach seine GM ranged from 4.57 to 39.14. Due to high variability of Atlantic Menhaden catches, no significant differences were found between any of the systems. Bottom trawl GM for Spot ranged from 11.68 (Magothy River) to 387.33 (Tred Avon River). A Tukey's HSD test was used to evaluate differences in Spot log-transformed catches (+1) between different systems. (P=0.05). Spot bottom trawl catch was similar for three of the four systems. Magothy River Spot catch was significantly lower than the other three systems. Bay Anchovy bottom trawl GM ranged from 1.77 to 40.19. A significant difference was found between the systems for log-transformed catches (+1) (F=4.13, P=0.009). According to the Duncan's MRT grouping, West-Rhode River and Miles River had the most similar bottom trawl catches for Bay Anchovy. Miles, Tred Avon and Magothy Rivers were not significantly different in catches of

Bay Anchovy. Due to high variability of finfish abundance, the differences in a specific year would be hard to determine.

Historical Trends of Forage Target Species - Abundance indices of forage target species were calculated over the time series (2003-2024) for Spot, Atlantic Menhaden, Eastern Silvery Minnow, Spottail Shiner, and Gizzard Shad. The marine migrant species (Spot and Atlantic Menhaden) data correlated very well over the time series (r = 0.91, $P<0.001$). The GM for Spot abundance was calculated from bottom trawl data, Spot catches in beach seines were highly variable. The GM for Atlantic Menhaden abundance was calculated from beach seine data; Atlantic Menhaden were rarely encountered in bottom trawls. The proportion of positive hauls with Atlantic Menhaden was 57% for 2024, the most prevalent in the time series (Figure 3-20). Even though Atlantic Menhaden catches are highly variable in the beach seine catches, the high proportion of positive hauls suggests that abundance is high for this species in the systems sampled with a beach seine.

Eastern Silvery Minnows were encountered in Mattawoman and Piscataway Creeks. The GM of abundance of Eastern Silvery Minnows from bottom trawls was low after a peak in 2004 (Figure 3-21). This decrease in Eastern Silvery Minnow abundance coincides with increased SAV (Miller et al. 2018).

Spottail Shiner abundance from bottom trawls gradually increased from 2003 to 2015 in the time series. In 2016, abundance fell to almost zero. The 2024 GM was higher than the previous 7 years (Figure 3-22).

Gizzard Shad abundance was estimated from beach seine data. Gizzard Shad abundance increased from 2003 to 2011; however, a decrease was noted from 2012 to 2024 (Figure 3-23). Future reports will examine this decrease in relation to other studies in the Chesapeake Bay. Gizzard Shad abundance could be impacted by lower recruitment in recent years due to dry conditions (Miranda et al. 2020). Gizzard Shad are also a preferred food item of invasive Blue Catfish. A Study in Virginia has shown they make up a large portion of the diet in the James River (52% by weight) and the Rappahannock River (48.8% by weight; Orth et al 2017).

Bay Anchovy was the only estuarine resident forage species with an abundance index. Bay Anchovy abundance was calculated from bottom trawl data. Bay Anchovy abundance tends to be highly variable with boom-and-bust cycles (Newberger and Houde 1995; Wang and Houde 1995). In the time series, low Bay Anchovy abundance was present during 2004-2006 and 2018-2022. Bay Anchovy abundance was highest in 2012-2015. The 2024 Bay Anchovy abundance was higher than the 6 previous years (Figure 3-24)

Population Dynamics of Select Target Species in Magothy River, Mattawoman Creek, Miles River, Piscataway Creek, Tred Avon River, and West-Rhode River - Geometric means for beach seine samples were calculated for young of year (YOY) Striped Bass in the subestuaries sampled and compared to previous years. For 2024, the GM for the subestuaries sampled were: Magothy (0.22), Tred Avon (0.73), Miles (0.70), and West-Rhode (0.17). Geometric means from the Striped Bass Program's Juvenile Abundance Index Survey were Bay-wide (1.06), Upper Bay (0.35) and Choptank River (1.90; Durell and Weedon 2024).

Historical GMs were calculated for each subestuary sampled in 2024 and correlated with Bay-wide and Upper Bay YOY Striped Bass GM from the Juvenile Abundance Index Survey (Figure 3-25); Magothy River was excluded due to low sample size (n=2). The West-Rhode Pearson correlations were 0.83 for Bay-wide (P=0.17, N=4) and 0.76 for Upper Bay (P=0.24, N=4). The Tred Avon River and Miles River had strong positive correlations with the Bay-wide GM, 0.82 and 0.86 respectively (P<0.001 and 0.03, N=19 and 6, respectively; Table 3-16). The

Tred Avon River and Miles River correlations with the Upper Bay GM were 0.52 and 0.83, respectively (P=0.02 and 0.04, N=19 and 6, respectively; Table 3-16). The Tred Avon River, a subestuary of the Choptank River, had a strong positive correlation with the Choptank River JAI ($r = 0.80$, $P<0.001$, N=19). Overall, the YOY Striped Bass GM's for the subestuaries sampled were positively correlated with the Striped Bass Program's estimates. Uphoff et al. (2011) found that the regional JAI was a significant and positive influence on the odds that Striped Bass juveniles would be present in seine samples in mesohaline subestuaries of the central Bay and Potomac River during 2003-2005; distance from the spawning area and level of development were not significant influences.

Juvenile White Perch P-A in mesohaline systems declined with increased development (r^2=0.10, P=0.002; Figure 3.26a). There was a large drop in P-A when C/ha was past the threshold and a linear function may not have been a good descriptor of a threshold response (Figure 3-26a). Linear regressions of tidal-fresh and oligohaline system White Perch P-A showed no clear relationships that were dependent on development (r^2=0.00, P=0.77; Figure 3.26b). A long-term data set for the Tred Avon River was analyzed to determine any impacts from development. Juvenile White Perch P-A over time was strongly correlated with year class strength from the Bay-wide Striped Bass JAI ($r = 0.88$; P=<0.001; Durell and Weedon 2024; Figure 3-27) but was not related to development there(Figure 3-28).

There was no clear impact of development indicated for P-A of adult White Perch in tidal-fresh or oligohaline systems. In mesohaline systems, a linear decline in the P-A of adult White Perch with increased development was a reasonable description (r^2=0.30, P<0.001; Figure 3-29). In these types of analyses, the regression line captures the decline from target to between conditions and drives the predictions. The regression did a poorer job of describing a possible low P-A asymptote past the threshold for development (between 0 and 0.30). Mesohaline systems also showed a threshold response in P-A of adult White Perch with median bottom DO; P-A was low (0 - 0.3) when DO was below 3.0 mg/L and highly variable after that (0 – 1.0; Figure 3-30). A threshold response was not well described by linear regression (r^2=0.07, P=0.01; Figure 3-30). A linear relationship was not suggested for P-A of adult White Perch in tidal-fresh or oligohaline systems with increased development (r^2=0.03, P=0.120).

In the Tred Avon River data set, a linear decline in P-A of adult White Perch was evident over the 18-year time series (r^2=0.43, P=0.003; Figure 3-31). When juvenile recruitment was considered (2-year lagged GM from JAI for the Choptank River), adult White Perch P-A followed the recruitment pattern (Figure 3-31). Adult White Perch P-A declined with increased development (C/ha) in the Tred Avon River (r^2=0.37, P=0.006; Figure 3-32). White Perch P-A linearly increased with median bottom DO (generally, 4.5 to 6.5 mg/L) in the Tred Avon River (r^2=0.23, P=0.04; Figure 3-33).

Modified proportional stock densities (PSDs) based on trawl samples revealed White Perch primarily use Mattawoman Creek and Piscataway Creek as nursery habitat. Modified PSDs for Mattawoman Creek fluctuated between 0% and 1.4%. Mattawoman Creek's modified PSD in 2024 was 0%, which was also the long-term median. Modified PSDs for Piscataway Creek fluctuated between 0% and 3.8%. Piscataway Creek's modified PSD in 2024 was 0%, which was also the long- term median (Table 3-17; Figure 3-34). Figure 3-35 shows the length frequency distribution of juvenile White Perch sampled in Mattawoman and Piscataway Creeks for 2024 was made up of Young of Year (YOY) and small age-1 fish.

A total of 20 Adult White Perch (9 Quality) were present in Miles River in 2024, so a modified PSD was estimated at 45%. Modified PSDs for White Perch on the Miles River ranged from 0% to 58.4% (Table 3-17; Figure 3-34). Modified PSDs were either high (>45%) or low (<1%) and did not align with mean bottom DO.

Modified PSD for Magothy River in 2003 was 29.8%. In 2024, only 1 adult white perch was collected, and this fish was also quality size (Table 3-17).

Adult White Perch were not present in West-Rhode River in 2024 and a modified PSD could not be estimated. Historical sampling of adult White Perch showed modified PSDs of 0.0 to 50% (Table 3-17).

Tred Avon River modified PSDs have ranged from 4.7% to 52.1%. In 2024, the modified PSD was 31.8%, which is above the time-series median (22.4%). Modified PSDS have been relatively high since 2017, suggesting a relatively stable adult population of White Perch in the Tred Avon River. Since 2017, six of the eight years have been above the time-series median (Table 3-17; Figure 3-34).

Modified PSD time-series indicated that bottom channel waters of tidal-fresh subestuaries sampled in 2024 (Mattawoman Creek and Piscataway) were primarily habitat for juvenile White Perch that consisted of YOY and Age 1 fish too small to be of interest to anglers (Figure 3-35). Mesohaline subestuaries with extensive low bottom channel DO measurements (Magothy and Miles River) had highly variable PSDs in bottom channel habitat 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).

Mattawoman Creek Ecosystem Shift and Restoration - 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 in the early 2000s (Gellis et al. 2009) and they increased nutrient loading.

In 2024, there was little indication that low DO was more widespread in Mattawoman Creek channel habitat than usual. Salinity was noticeably greater in 2024, and more marine species were present in trawl catches. Bottom DO at all stations remained above the target level (5.0 mg/L). 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 from the watershed (the main sewage outfall releases wastewater to the mainstem Potomac; however, the sewer line runs along Mattawoman Creek and leaks occur) 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 the channel's fish community recovery associated with recovery of this subestuary's SAV has not revealed itself.

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Table 3-1. Summary of all subestuaries and their regional area, year sampled, number of stations and sampling gear used.

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

Table 3-1. Cont.

Subestuary	Area	Year	# of 30.5m Seine Stations	# of 4.9m Trawl Stations	# of 3.1m Trawl Stations
Corsica River	Mid Bay (Chester)	2006	3	4	
		2007	3	4	
		2008	3	4	
		2009	3	4	
		2010	3	4	
		2011	3	4	
		2012	3	4	
		2018	3	4	
		2019		4	
Fishing Bay	Lower Bay	2006	4	4	
Gunpowder River	Upper Bay	2009	4	4	
		2010	4	4	
		2011	4	4	
		2012	4	4	
		2013	4	4	
		2014	3	4	
		2015	3	4	
		2016	3	4	
Harris Creek	Mid Bay (Choptank)	2012	3	4	
		2013	3	4	
		2014	3	4	
		2015	3	4	
		2016	3	4	
Langford Creek	Mid Bay (Chester)	2006	4	4	
		2007	4	4	
		2008	4	4	
		2018	3	4	
		2019	3	4	
Magothy River	Mid Bay	2003	4	4	
		2024	4	4	
Mattawoman Creek	Potomac	1989	5		5
		1990	5		5
		1991	5		5
		1992	5		5
		1993	5		5
		1994	5		5
		1995	5		5

Table 3-1. Cont.

Subestuary	Area	Year	# of 30.5m Seine Stations	# of 4.9m Trawl Stations	# of 3.1m Trawl Stations
Mattawoman Creek	Potomac	1996	5		5
		1997	5		5
		1998	5		5
		1999	5		5
		2000	5		5
		2001	5		5
		2002	4		5
		2003	4	4	
		2004	4	4	
		2005	4	4	
		2006		4	
		2007		4	
		2008		4	
		2009	1	4	4
		2010		4	4
		2011		4	4
		2012		4	4
		2013		4	4
		2014		4	4
		2015		4	4
		2016		4	4
		2022		4	
		2023		4	
		2024		4	
Middle River	Upper Bay	2009	2	4	
		2010	3	4	
		2011	3	4	
		2012	3	4	
		2013	2	4	
		2014	2	4	
		2015	1	4	
		2016		4	
		2017		4	
Miles River	Mid Bay	2003	3	4	
		2004	3	4	
		2005	3	4	
		2020	3	4	

Table 3-1. Cont.

Subestuary	Area	Year	# of 30.5m Seine Stations	# of 4.9m Trawl Stations	# of 3.1m Trawl Stations
Miles River	Mid Bay	2023	2	4	
		2024	3	4	
Nanjemoy Creek	Potomac	2003	3	3	
		2008	3	3	
		2009	3	3	
		2010	3	3	
		2011	4	4	
		2012	4	4	
		2013	3	3	
		2014	3	3	
		2015	3	3	
		2016	3	3	
Northeast River	Upper Bay	2007	4	4	
		2008	4	4	
		2009	4	4	
		2010	4	4	
		2011	4	4	
		2012	4	4	
		2013	4	4	
		2014	4	4	
		2015	4	4	
		2016	4	4	
		2017	4	4	
		2022	3	4	
		2023	4	4	
Piscataway Creek	Potomac	2003	3	3	
		2006	2	3	
		2007		3	
		2009		3	
		2010		3	
		2011		3	
		2012		3	
		2013		3	
		2014		3	
		2024		3	
Sassafras River	Upper Bay	2020		4	
		2021	4	4	

Table 3-1. Cont.

Subestuary	Area	Year	# of 30.5m Seine Stations	# of 4.9m Trawl Stations	# of 3.1m Trawl Stations
Severn River	Mid Bay	2003	5	4	
		2004	5	4	
		2005	5	4	
		2017	3	4	
South River	Mid Bay	2003	4	4	
		2004	4	4	
		2005	4	4	
		2022	4	4	
		2023	3	4	
St Clements River	Potomac	2003	4	4	
		2004	4	4	
		2005	4	4	
Transquaking River	Lower Bay	2006	4	1	
Tred Avon River	Mid Bay	2006	4	4	
	(Choptank)	2007	4	4	
		2008	4	4	
		2009	4	4	
		2010	4	4	
		2011	4	4	
		2012	4	4	
		2013	4	4	
		2014	4	4	
		2015	4	4	
		2016	4	4	
		2017	4	4	
		2018	4	4	
		2019	4	4	
		2020	4	4	
		2021	4	4	
		2022	4	4	
		2023	4	4	
		2024	4	4	
West-Rhode River	Mid Bay	2003	3	4	
		2004	3	4	
		2005	3	4	
		2024	3	4	

Table 3-1. Cont.

Subestuary	Area	Year	# of 30.5m Seine Stations	# of 4.9m Trawl Stations	# of 3.1m Trawl Stations
Wicomico River	Potomac	2003	4	4	
		2010	4	4	
		2011	4	4	
		2012	4	4	
		2017	4	4	
Wye River	Mid Bay	2007	4	4	
		2008	4	4	
		2018	3	4	
		2019	3	4	

Table 3-2. Estimates of impervious surface (% IS), structures per hectare (C/ha) and land use percentages (agriculture, forest, wetland, and urban) for subestuaries sampled in 2024.

Subestuary	Year	% IS	C/ha	% Land Use				% Water	Land (ha)	Water (ha)
				Ag	Forest	Wetland	Urban			
Magothy River	2003	22.40	2.68	2.57	27.82	0.00	69.51	24.75	9,227	2,283
	2024	23.86	2.95	1.20	20.42	0.01	77.87	25.85	9,147	2,364
Mattawoman Creek	2003	9.32	0.76	11.88	59.37	1.18	27.38	3.14	24,401	767
	2004	9.52	0.79	11.88	59.37	1.18	27.38	3.14	24,401	767
	2005	9.69	0.81	11.88	59.37	1.18	27.38	3.14	24,401	767
	2006	9.90	0.83	11.88	59.37	1.18	27.38	3.14	24,401	767
	2007	10.12	0.86	11.88	59.37	1.18	27.38	3.14	24,401	767
	2008	10.22	0.87	11.88	59.37	1.18	27.38	3.14	24,401	767
	2009	10.32	0.88	11.88	59.37	1.18	27.38	3.14	24,401	767
	2010	10.44	0.90	9.33	53.88	1.13	34.18	3.14	24,403	766
	2011	10.55	0.91	9.33	53.88	1.13	34.18	3.14	24,403	766
	2012	10.48	0.90	9.33	53.88	1.13	34.18	3.14	24,403	766
	2013	10.60	0.92	9.33	53.88	1.13	34.18	3.44	24,334	836
	2014	10.69	0.93	9.33	53.88	1.13	34.18	3.44	24,334	836
	2015	10.81	0.94	9.33	53.88	1.13	34.18	3.44	24,334	836
	2016	10.93	0.96	9.33	53.88	1.13	34.18	3.44	24,334	836
	2022	11.47	1.03	8.63	52.83	1.14	35.65	3.45	24,331	840
	2023	11.48	1.03	8.63	52.83	1.14	35.65	3.45	24,331	840
	2024	11.48	1.04	8.63	52.83	1.14	35.65	3.45	24,331	840
Miles River	2003	4.14	0.24	53.71	27.21	0.89	18.14	27.43	11,071	3,036
	2004	4.20	0.24	53.71	27.21	0.89	18.14	27.43	11,071	3,036
	2005	4.21	0.24	53.71	27.21	0.89	18.14	27.43	11,071	3,036
	2020	4.42	0.26	48.70	26.52	0.86	23.39	28.64	10,968	3,141
	2023	4.43	0.26	48.70	26.52	0.86	23.39	28.64	10,968	3,141
	2024	4.43	0.27	48.70	26.52	0.86	23.39	28.64	10,968	3,141
Piscataway Creek	2003	13.52	1.30	12.76	45.76	0.25	40.57	2.23	17,607	392
	2006	14.11	1.38	12.76	45.76	0.25	40.57	2.23	17,607	392
	2007	14.25	1.40	12.76	45.76	0.25	40.57	2.23	17,607	392
	2009	14.47	1.43	12.76	45.76	0.25	40.57	2.23	17,607	392
	2010	14.58	1.45	9.98	40.37	0.24	47.01	2.23	17,607	392
	2011	14.68	1.46	9.98	40.37	0.24	47.01	2.23	17,607	392
	2012	14.75	1.47	9.98	40.37	0.24	47.01	2.23	17,607	392
	2013	14.93	1.50	9.98	40.37	0.24	47.01	2.64	17,537	464
	2014	15.02	1.51	9.98	40.37	0.24	47.01	2.64	17,537	464
	2024	16.57	1.61	9.60	38.03	0.32	49.04	2.65	17,536	465

Table 3-2. Cont.

Subestuary	Year	% IS	C/ha	% Land Use				% Water	Land (ha)	Water (ha)
				Ag	Forest	Wetland	Urban			
Tred Avon River	2006	8.71	0.69	50.08	21.58	1.00	27.23	32.35	9,556	3,092
	2007	8.90	0.71	50.08	21.58	1.00	27.23	32.35	9,556	3,092
	2008	9.00	0.73	50.08	21.58	1.00	27.23	32.35	9,556	3,092
	2009	9.10	0.74	50.08	21.58	1.00	27.23	32.35	9,556	3,092
	2010	9.19	0.75	43.20	21.63	0.85	33.57	32.28	9,561	3,087
	2011	9.23	0.75	43.20	21.63	0.85	33.57	32.28	9,561	3,087
	2012	9.25	0.75	43.20	21.63	0.85	33.57	32.28	9,561	3,087
	2013	9.33	0.76	43.20	21.63	0.85	33.57	34.11	9,432	3,217
	2014	9.36	0.77	43.20	21.63	0.85	33.57	34.11	9,432	3,217
	2015	9.40	0.77	43.20	21.63	0.85	33.57	34.11	9,432	3,217
	2016	9.43	0.78	43.20	21.63	0.85	33.57	34.11	9,432	3,217
	2017	9.42	0.77	43.20	21.63	0.85	33.57	34.11	9,432	3,217
	2018	9.46	0.78	42.63	21.67	0.86	33.96	34.10	9,433	3,216
	2019	9.51	0.79	42.63	21.67	0.86	33.96	34.10	9,433	3,216
	2020	9.53	0.79	42.63	21.67	0.86	33.96	34.10	9,433	3,216
	2021	9.53	0.79	42.63	21.67	0.86	33.96	34.10	9,433	3,216
	2022	9.55	0.79	42.63	21.67	0.86	33.96	34.10	9,433	3,216
	2023	9.57	0.79	42.63	21.67	0.86	33.96	34.10	9,433	3,216
	2024	9.57	0.79	42.63	21.67	0.86	33.96	34.10	9,433	3,216
West-Rhode River	2003	7.40	0.55	34.07	45.30	0.79	19.84	21.73	6,604	1,435
	2004	7.47	0.56	34.07	45.30	0.79	19.84	21.73	6,604	1,435
	2005	7.52	0.56	34.07	45.30	0.79	19.84	21.73	6,604	1,435
	2024	7.96	0.62	27.08	44.12	0.86	27.96	22.86	6,544	1,496

Table 3-3. Pearson correlations (r) of structures per hectare (C/ha) and land use categories for subestuaries sampled during 2024. Land use data from MD Department of Planning (DOP; 2002 and 2010) and Chesapeake Conservancy (2013 and 2018). *P* = level of significance. Sample size (N) is the number of surveys conducted in the 2024 subestuaries between 2003 and 2024.

	Statistic	C/ha	Agriculture	Forest	Wetland	Urban
Agriculture	r	-0.67				
	P	<.0001	1			
	N	58				
Forest	r	-0.103	–0.79			
	P	0.44	<.0001	1		
	N	58	58			
Wetland	r	-0.69	0.20	0.27		
	P	<.0001	0.13	0.043	1	
	N	58	58	58		
Urban	r	0.96	-0.55	-0.071	-0.71	
	P	<.0001	<.0001	0.59	<.0001	1
	N	58	58	58	58	

Table 3-4. Summary of water quality parameter statistics collected during beach seine and bottom trawl samples for subestuaries in 2024. Depths at Piscataway Creek sites were insufficient (<2 m) to collect bottom water quality data.

		Surface Measurements					Bottom Measurements					
Subestuary	Statistics	Temp (°C)	DO (mg/L)	Cond (µS/cm)	Salinity	pH	Temp (°C)	DO (mg/L)	Cond (µS/cm)	Salinity	pH	Secchi
Magothy River	Mean	27.65	7.01	10976	6.21	7.98	27.01	3.24	11514	6.52	7.58	0.94
	SE	0.39	0.16	99	0.07	0.5	0.48	0.46	191	0.11	0.09	0.02
	Median	27.91	7.03	10823	6.08	7.98	27.46	3.16	11428	6.46	7.61	0.90
	Minimum	23.50	4.60	10093	5.64	7.27	23.32	0.00	10010	5.63	7.13	0.60
	Maximum	31.59	9.31	12734	7.59	8.32	30.12	6.56	13665	7.63	8.16	1.30
	Count	48	48	48	48	32	24	24	24	24	16	48
Mattawoman Creek	Mean	26.82	7.49	373	0.18	8.15	26.78	6.99	376	0.18	8.06	0.91
	SE	0.65	0.21	11	0.01	0.06	0.63	0.18	11	0.01	.05	0.05
	Median	26.77	7.53	363	0.17	8.07	27.84	7.03	364	0.17	8.02	0.90
	Minimum	23.04	5.77	300	0.14	7.73	23.01	5.36	304	0.14	7.69	0.77
	Maximum	30.99	9.63	464	0.22	8.56	30.68	8.71	484	0.23	8.42	1.35
	Count	24	24	24	24	20	24	24	24	24	20	24
Miles River	Mean	27.00	7.31	16604	9.73	7.95	26.25	5.82	16871	9.91	7.78	0.64
	SE	0.43	0.13	444	0.29	0.04	0.54	0.21	623	0.41	0.05	0.03
	Median	27.77	7.28	15612	9.07	7.99	26.14	5.94	15757	9.16	7.90	0.62
	Minimum	22.78	6.13	13499	7.71	7.50	22.22	4.22	13672	7.85	7.39	0.36
	Maximum	32.33	8.92	23704	14.36	8.24	30.11	7.05	23707	14.37	8.05	1.38
	Count	42	28	42	42	35	23	15	23	23	19	42
Piscataway Creek	Mean	27.69	10.15	362	0.17	8.38						0.66
	SE	0.81	0.41	14	0.01	0.13						0.04
	Median	27.44	9.92	374	0.18	8.35						0.63
	Minimum	23.65	7.35	266	0.13	7.69						0.45
	Maximum	32.04	13.54	434	0.21	9.29						0.94
	Count	18	18	18	18	14						18
Tred Avon River	Mean	28.28	7.12	14390	8.31	7.87	27.68	5.23	14652	8.48	7.59	0.57
	SE	0.40	0.14	203	0.13	0.03	0.50	0.24	283	0.18	0.03	0.02
	Median	29.07	6.99	14120	8.11	7.89	28.53	5.25	14329	8.24	7.57	0.58
	Minimum	23.67	4.78	12180	6.90	7.41	23.55	2.53	12891	7.35	7.35	0.30
	Maximum	32.21	9.95	17821	10.52	8.30	30.63	7.94	18227	10.79	7.96	0.96
	Count	48	47	48	48	40	24	24	24	24	20	47
West-Rhode River	Mean	26.37	7.56	14640	8.41	8.15	26.19	4.96	15668	9.13	7.78	0.60
	SE	0.48	0.20	321	0.19	0.05	0.59	0.29	338	0.22	0.05	0.02
	Median	26.96	7.64	14923	8.62	8.15	26.76	5.20	15244	8.84	7.80	0.59
	Minimum	21.34	4.62	10959	6.18	7.46	21.77	2.02	13007	7.33	7.24	0.35
	Maximum	31.70	10.62	18933	11.22	8.75	30.78	7.62	19416	11.56	8.16	0.92
	Count	42	42	42	42	40	24	24	24	24	24	42

Table 3-5. Statistics and parameter estimates for analysis of covariance and post hoc Duncan's multiple range test examining the effect of date and river system on 2024 surface water temperatures.

a. Statistics and parameter estimates for ANCOVA.

	df	SS	MS	F	*P* value
Model	6	1759.67	293.28	280.26	<0.001
Error	215	224.98	1.05		
Total	221	1984.66			
$R^2 = 0.887$					
	df	Type III SS	MS	F	*P* value
River	5	33.47	6.69	6.4	<0.001
Date	1	1658.93	1658.93	1585.31	<0.001

b. Duncan's multiple range test results.

Duncan Grouping	Mean Temperature	N	Subestuary
A	28.28	48	Tred Avon River
B	27.69	18	Piscataway Creek
B	27.65	48	Magothy River
C	27.00	42	Miles River
C	26.82	24	Mattawoman Creek
C	26.36	42	West-Rhode River

*Means with the same letter are not significantly different.

Table 3-6. Percentages of all DO measurements that did not meet the target (5.0 mg/L) and bottom DO that did not meet the target and threshold (3.0 mg/L). C/ha = structures per hectare. There was insufficient depth (<2 m) at Piscataway Creek sites to collect bottom water quality measures in 2024.

			All DO		Bottom DO		
Subestuary	Salinity Class	C/ha	N	% <5.0 mg/L	N	% <5.0 mg/L	% <3.0 mg/L
Magothy River	Mesohaline	2.95	96	25.0	24	70.8	45.8
Mattawoman Creek	Tidal-Fresh	1.04	57	0.0	24	0.0	0.0
Miles River	Mesohaline	0.27	57	8.6	15	20.0	0.0
Piscataway Creek	Tidal-Fresh	1.61	18	0.0			
Tred Avon River	Mesohaline	0.80	95	11.6	24	37.5	4.2
West-Rhode River	Mesohaline	0.62	41	2.4	24	45.8	12.5

Table 3-7. Median annual surface and bottom temperatures, median annual surface and bottom dissolved oxygen, and structures per hectare (C/ha) for subestuaries sampled during 2024, by salinity class. Piscataway Creek sites were too shallow (<2 m) to collect bottom water quality measures in 2024.

			Temperature (◦C)		Dissolved Oxygen (mg/L)	
Subestuary	Year	C/ha	Surface	Bottom	Surface	Bottom
			Mesohaline			
Magothy River	2003	2.68	26.35	25.60	7.20	1.10
	2024	2.95	27.91	27.46	7.03	3.16
Miles River	2003	0.24	27.80	27.20	6.40	3.80
	2004	0.24	26.50	26.20	5.95	5.29
	2005	0.24	28.40	27.80	5.80	3.77
	2020	0.26	29.35	28.29	6.45	3.27
	2023	0.26	28.58	28.17	6.22	4.33
	2024	0.27	27.77	26.14	7.28	5.94
Tred Avon River	2006	0.69	28.50	28.05	5.95	5.35
	2007	0.71	26.55	26.25	6.60	5.70
	2008	0.73	26.70	26.10	6.60	5.30
	2009	0.74	28.25	27.65	7.35	6.33
	2010	0.75	28.15	27.70	7.10	5.40
	2011	0.75	29.15	29.10	7.00	5.00
	2012	0.75	27.79	27.66	7.06	5.52
	2013	0.76	26.70	26.37	7.12	5.50
	2014	0.77	27.38	27.29	6.44	5.63
	2015	0.77	28.24	27.58	6.93	5.75
	2016	0.78	30.05	29.26	7.16	5.43
	2017	0.77	26.39	26.24	6.98	5.51
	2018	0.78	28.11	27.48	7.36	4.98
	2019	0.79	28.74	28.55	6.75	4.52
	2020	0.79	28.55	28.45	6.80	4.69
	2021	0.79	29.20	28.39	6.56	5.11
	2022	0.79	28.58	28.51	6.54	4.87
	2023	0.79	28.96	28.44	6.43	5.09
	2024	0.80	29.07	28.53	6.99	5.25
West-Rhode River	2003	0.55	27.10	25.25	7.30	4.90
	2004	0.56	27.50	27.25	6.64	5.80
	2005	0.56	28.00	27.50	6.65	4.33
	2024	0.62	26.96	26.76	7.64	5.20

Table 3-7. Cont.

Subestuary	Year	C/ha	Temperature (◦C) Surface	Temperature (◦C) Bottom	Dissolved Oxygen (mg/L) Surface	Dissolved Oxygen (mg/L) Bottom
Tidal-Fresh						
Mattawoman Creek	2003	0.76	28.30	28.05	9.20	9.15
	2004	0.79	27.90	27.70	8.10	7.83
	2005	0.81	28.90	28.25	7.60	7.33
	2006	0.83	27.90	27.80	7.80	6.50
	2007	0.86	26.95	26.90	6.90	6.60
	2008	0.87	27.05	25.60	7.20	6.45
	2009	0.88	27.85	27.55	7.60	7.63
	2010	0.90	28.15	28.15	6.95	6.70
	2011	0.91	27.60	28.35	6.40	6.75
	2012	0.90	27.96	28.13	7.17	6.80
	2013	0.92	26.95	26.94	8.82	8.23
	2014	0.93	27.94	27.24	9.60	8.65
	2015	0.94	28.10	27.73	8.63	7.79
	2016	0.96	29.93	29.57	6.91	6.68
	2022	1.03	27.03	26.88	7.73	6.63
	2023	1.03	27.63	27.48	7.50	6.82
	2024	1.04	26.77	27.84	7.53	7.03
Piscataway Creek	2003	1.30	27.45	24.60	10.60	8.55
	2006	1.38	29.60	22.70	8.70	6.95
	2007	1.40	27.40	26.00	8.60	7.60
	2009	1.43	28.75	28.65	7.70	6.46
	2010	1.45	28.85	26.25	9.30	7.60
	2011	1.46	28.15	29.70	9.00	9.50
	2012	1.47	28.91	25.95	9.27	8.35
	2013	1.50	27.37	26.68	9.22	7.72
	2014	1.51	26.46	25.90	8.52	7.59
	2024	1.61	27.44		9.92	

Table 3-8. Pearson correlations (r) of arithmetic median surface and bottom dissolved oxygen (DO; mg/L) with median water temperatures at depth (surface or bottom) and watershed development (C/ha = structures per hectare) for subestuaries sampled during 2024, by salinity class. *P* = level of significance. Sample size (N) = number of samples collected in 2024 systems between 2003 to 2024.

DO Depth	Statistic	Temperature	C/ha
	Mesohaline		
Surface	r	-0.211	0.316
	P	0.255	0.083
	N	31	31
Bottom	r	-0.004	-0.495
	P	0.982	0.005
	N	31	31
	Tidal-Fresh		
Surface	r	-0.028	0.610
	P	0.892	0.001
	N	27	27
Bottom	r	0.053	0.270
	P	0.796	0.183
	N	26	26

Table 3-9. Statistics and parameter estimates of Pearson correlation for median bottom dissolved oxygen (DO; mg/L) against agricultural land cover in mesohaline subestuaries sampled in 2024 (data from 2003-2024).

	df	SS	MS	F	*P* value
Model	1	11.19	11.19	15.83	0.0004
Error	29	20.50	0.707		
Total	30	31.69			
R^2 = 0.353					
	Estimate	SE	t value	*P* value	
Intercept	2.79	0.544	65.12	<0.001	
% Agriculture	0.05	0.013	3.98	0.0004	

Table 3-10. Statistics and parameter estimates for quadratic regression of median bottom dissolved oxygen (DO; mg/L) versus percent agricultural coverage in all mesohaline systems sampled by FEAD during 2003-2024 (see Table 3-1 for list of systems).

a. All systems combined.

	df	SS	MS	F	*P* value
Model	2	95.51	47.76	46.30	<0.001
Error	96	99.02	1.03		
Total	98	194.53			
$R^2 = 0.49$					
	Estimate	SE	t value	*P* value	
Intercept	0.79	0.432	1.82	0.072	
% Agriculture	0.19	0.022	8.15	<.0001	
% Ag * % Ag	-0.002	0.0003	-6.66	<.0001	

b. By region (East and West shores)

East

	df	SS	MS	F	*P* value
Model	2	2.24	1.12	1.37	0.26
Error	70	57.25	0.82		
Total	72	59.49			
$R^2 = 0.038$					
	Estimate	SE	t value	*P* value	
Intercept	5.62	1.84	3.06	0.003	
% Agriculture	0.001	0.07	0.01	0.989	
% Ag * % Ag	-.0001	0.0007	-0.23	0.821	

West

	df	SS	MS	F	*P* value
Model	2	44.75	22.38	14.63	<.0001
Error	23	35.18	1.53		
Total	25	79.93			
$R^2 = 0.56$					
	Estimate	SE	t value	*P* value	
Intercept	0.29	0.77	0.38	0.71	
% Agriculture	0.24	0.09	2.79	0.01	
% Ag * % Ag	-.003	0.002	-1.56	0.13	

Table 3-11. Beach seine catch summary for 2024. Total GM = total geometric mean catch per seine sample. Species captured list includes total catch and target species in order of abundance for each system.

Subestuary	Stations Sampled	Number of Samples	Number of Species	Total Catch	Total GM	Species Captured
Magothy River	4	24	23	4,387	119	All species Atlantic Menhaden Spot White Perch Gizzard Shad Striped Bass Bay Anchovy
Miles River	3	18	29	13,373	234	All species Atlantic Menhaden Spot Bay Anchovy White Perch Striped Bass Gizzard Shad Alewife
Tred Avon River	4	24	27	14,061	220	All species Atlantic Menhaden Spot Bay Anchovy White Perch Blueback Herring Striped Bass Gizzard Shad
West-Rhode River	3	18	22	3,398	109	All species Atlantic Menhaden Spot Bay Anchovy Gizzard Shad White Perch Striped Bass

Table 3-12. Bottom trawl catch summary for 2024. Total GM = total geometric mean catch per trawl sample. Species captured list includes total catch and target species in order of abundance for each system.

Subestuary	Stations Sampled	Number of Samples	Number of Species	Total Catch	Total GM	Species Captured
Magothy River	4	24	8	1,762	14	All species Spot Bay Anchovy White Perch
Mattawoman Creek	4	24	21	3,640	88	All species White Perch Spottail Shiner Spot Striped Bass Bay Anchovy Yellow Perch Atlantic Menhaden Blueback Herring
Miles River	4	24	18	16,094	462	All species Spot Bay Anchovy White Perch Atlantic Menhaden Striped Bass
Piscataway Creek	3	18	23	4,999	115	All species White Perch Spottail Shiner Bay Anchovy Eastern Silvery Minnow Blueback Herring Striped Bass Alewife American Shad Spot Atlantic Menhaden Gizzard Shad
Tred Avon River	4	24	20	14,767	486	All species Spot Bay Anchovy White Perch Atlantic Menhaden Striped Bass
West-Rhode River	4	24	13	14,623	452	All species Spot

Table 3-13. Total count of all target species collected in 2024 for bottom trawl and beach seine combined, by subestuary and salinity class.

	Mesohaline				Tidal-Fresh		
Species	Magothy	Miles	Tred Avon	West-Rhode	Mattawoman	Piscataway	Total
Alewife	0	1	0	0	0	3	4
American Shad	0	0	0	0	0	3	3
Atlantic Menhaden	1,769	11,739	9,637	1,994	1	1	25,141
Bay Anchovy	272	1,710	1,388	3,162	9	104	6,645
Blueback Herring	0	0	42	0	1	31	74
Gizzard Shad	9	25	11	76	0	1	122
Hickory Shad	0	0	0	0	0	0	0
Eastern Silvery Minnow	0	0	0	0	0	43	43
Spot	1,697	14,044	13,976	10,305	201	2	40,225
Spottail Shiner	0	0	0	0	222	344	566
Striped Bass (1+)	0	3	1	1	0	0	5
Striped Bass (YOY)	9	35	61	3	77	6	191
White Perch (Adult)	17	71	183	13	286	131	701
White Perch (Juv)	1	0	1	0	2,278	4,042	6,322
Yellow Perch	0	0	0	0	8	0	8

Table 3-14. Total count of all target species collected by beach seine in subestuaries sampled in 2024. Tidal-fresh subestuaries were not sampled by beach seine in 2024.

	Mesohaline				
Species	Magothy	Miles	Tred Avon	West-Rhode	Total
Alewife	0	1	0	0	1
American Shad	0	0	0	0	0
Atlantic Menhaden	1,769	11,730	9,595	1,963	25,057
Bay Anchovy	1	98	295	188	582
Blueback Herring	0	0	42	0	42
Gizzard Shad	9	25	11	76	121
Hickory Shad	0	0	0	0	0
Eastern Silvery Minnow	0	0	0	0	0
Spot	234	201	1,782	341	2,558
Spottail Shiner	0	0	0	0	0
Striped Bass (1+)	0	3	1	1	5
Striped Bass (YOY)	9	27	27	3	66
White Perch (Adult)	16	51	139	13	219
White Perch (Juv)	1	0	1	0	2
Yellow Perch	0	0	0	0	0

Table 3-15. Total count of all target species collected by bottom trawl in subestuaries sampled in 2024.

	Mesohaline				Tidal-Fresh		
Species	Magothy	Miles	Tred Avon	West-Rhode	Mattawoman	Piscataway	Total
Alewife	0	0	0	0	0	3	3
American Shad	0	0	0	0	0	3	3
Atlantic Menhaden	0	9	42	31	1	1	84
Bay Anchovy	271	1,612	1,093	2,974	9	104	6,063
Blueback Herring	0	0	0	0	1	31	32
Gizzard Shad	0	0	0	0	0	1	1
Hickory Shad	0	0	0	0	0	0	0
Eastern Silvery Minnow	0	0	0	0	0	43	43
Spot	1,463	13,843	12,194	9,964	201	2	37,667
Spottail Shiner	0	0	0	0	222	344	566
Striped Bass (1+)	0	0	0	0	0	0	0
Striped Bass (YOY)	0	8	34	0	77	6	125
White Perch (Adult)	1	20	44	0	286	131	482
White Perch (Juv)	0	0	0	0	2,278	4,042	6,320
Yellow Perch	0	0	0	0	8	0	8

Table 3-16. Pearson correlation coefficients for geometric means of young of year (YOY) Striped Bass from beach seines in the Tred Avon River and Miles River (Magothy River dropped due to low sample size). Striped Bass Program Juvenile Abundance Index Survey geometric means were used for the *Bay-wide, Upper Bay,* and *Choptank River* for comparisons. (P value in parentheses). Samples sizes range from 4 to 19, depending on length of time series.

	Bay-wide	Tred Avon	Miles	West-Rhode	*Upper Bay*	*Choptank*
Bay-wide	1.00					
Tred Avon	0.82 (<0.001)	1.00				
Miles	0.86 (0.029)	0.67 (0.144)	1.00			
West-Rhode	0.83 (0.167)			1.00		
Upper Bay	0.70 (<0.001)	0.52 (0.024)	0.83 (0.043)	0.76 (0.238)	1.00	
Choptank	0.94 (<0.001)	0.80 (<0.001)	0.76 (0.079)		0.46 (0.047)	1.00

Table 3-17. Annual modified proportional stock density (PSD) of White Perch for subestuaries sampled by bottom trawl in 2024 with historical estimates. N Total is the total number of White Perch (all juvenile and adult) in bottom trawl catches. N TLstock is the number of all adult White Perch (adults age 1+). N TLquality is number of harvestable adult White Perch (>200 mm).

Subestuary	Year	N Total	N TLstock	N TLquality	Modified PSD (%)
Magothy River	2003	448	429	128	29.8
	2024	1	1	1	100.0
Mattawoman Creek	2003	3,661	382	0	0.0
	2004	2,791	355	2	0.6
	2005	3,916	471	1	0.2
	2006	1,978	567	0	0.0
	2007	1,365	442	1	0.2
	2008	716	366	1	0.3
	2009	346	61	0	0.0
	2010	2,555	430	0	0.0
	2011	3,460	282	4	1.4
	2012	2,512	241	0	0.0
	2013	7,026	223	1	0.4
	2014	12,138	101	1	1.0
	2015	5,774	466	0	0.0
	2016	4,490	754	0	0.0
	2022	2,223	109	0	0.0
	2023	2,568	744	0	0.0
	2024	2,564	286	0	0.0
Miles River	2003	6,704	185	108	58.4
	2004	941	798	4	0.5
	2005	1,061	537	4	0.7
	2020	74	74	36	48.6
	2023	0	0	0	0.0
	2024	20	20	9	45.0
Piscataway Creek	2003	495	100	3	3.0
	2006	741	29	0	0.0
	2007	34	4	0	0.0
	2009	2,496	233	0	0.0
	2010	7,428	51	0	0.0
	2011	8,376	114	0	0.0
	2012	1,640	98	0	0.0
	2013	1,841	47	0	0.0
	2014	3,645	26	1	3.8
	2024	4,173	131	0	0.0

Table 3-17. Cont.

Subestuary	Year	N Total	N TLstock	N TLquality	Modified PSD (%)
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
	2024	44	44	14	31.8
West-Rhode River	2003	7,530	2	1	50.0
	2004	4	2	0	0.0
	2005	4	4	0	0.0
	2024	0	0	0	0.0

Figure 3-1. Map illustrating subestuaries sampled in summer 2024: Miles River (1), Tred Avon River (2), Magothy River (3), Rhode River (4), West River (5), Piscataway Creek (6), Mattawoman Creek (7), and their land use categories. Land use data are based on Maryland Department of Planning (DOP) 2018 land use and land cover data.

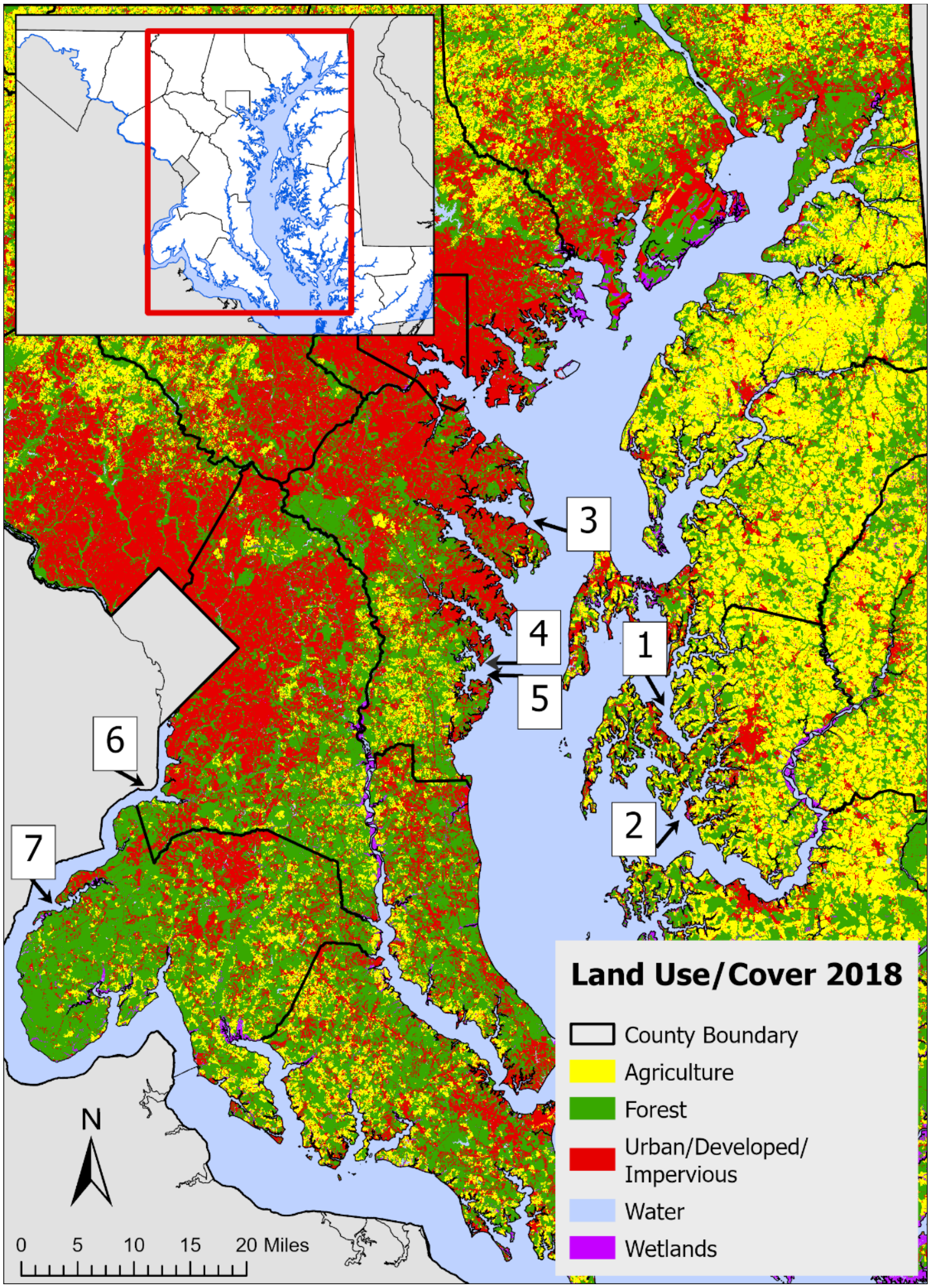


Figure 3-2. Maps indicating locations of stations sampled in 2024 located within Miles River, Tred Avon River, Magothy River, Rhode and West River, Piscataway Creek and Mattawoman Creek.

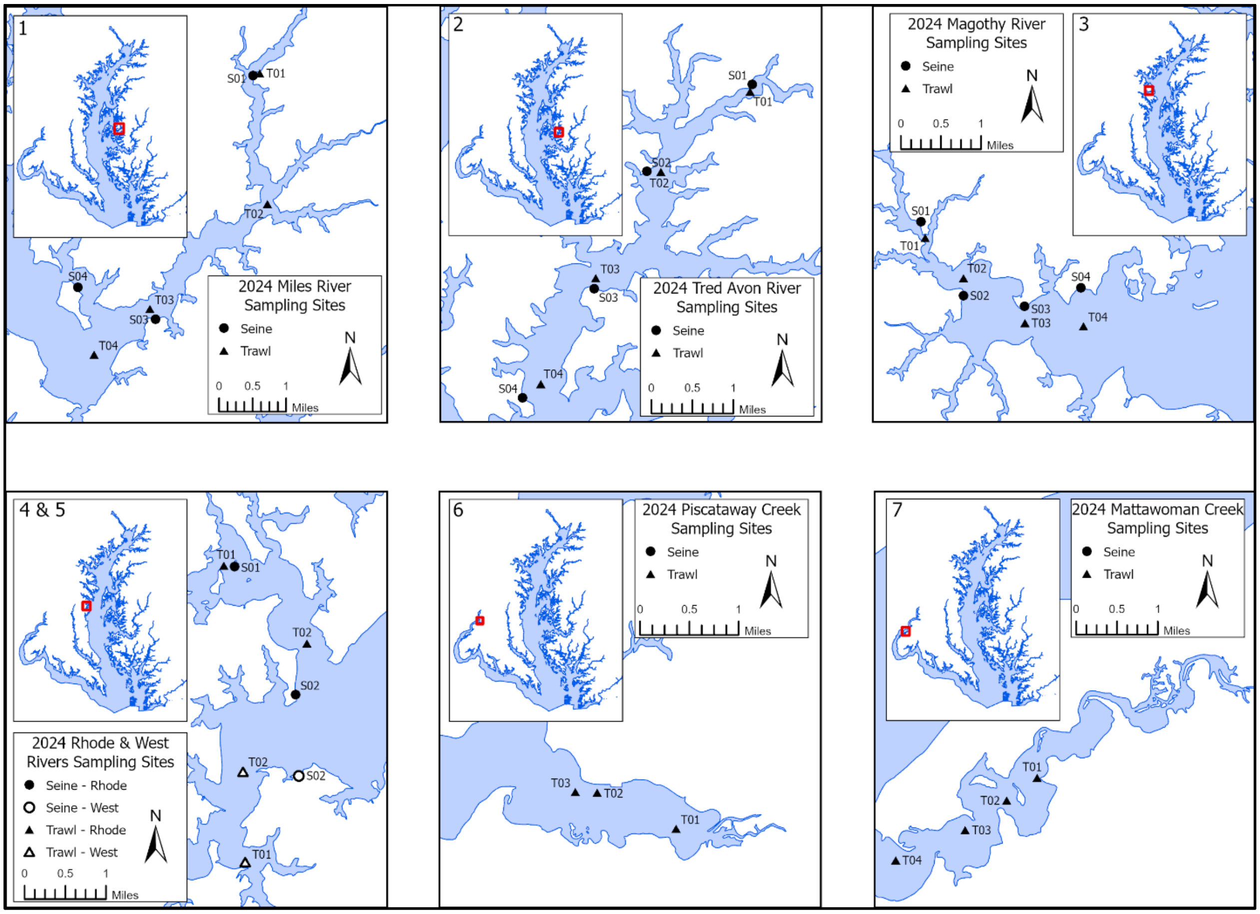


Figure 3-3. Trends in development (C/ha or structures per hectare) from 1950 to 2024 of the watersheds of subestuaries sampled in 2024. Black markers indicate the years that subestuaries were sampled. Dashed lines indicate thresholds for rural (0.37 C/ha), suburban (0.86 C/ha) and urban (1.35 C/ha) development.

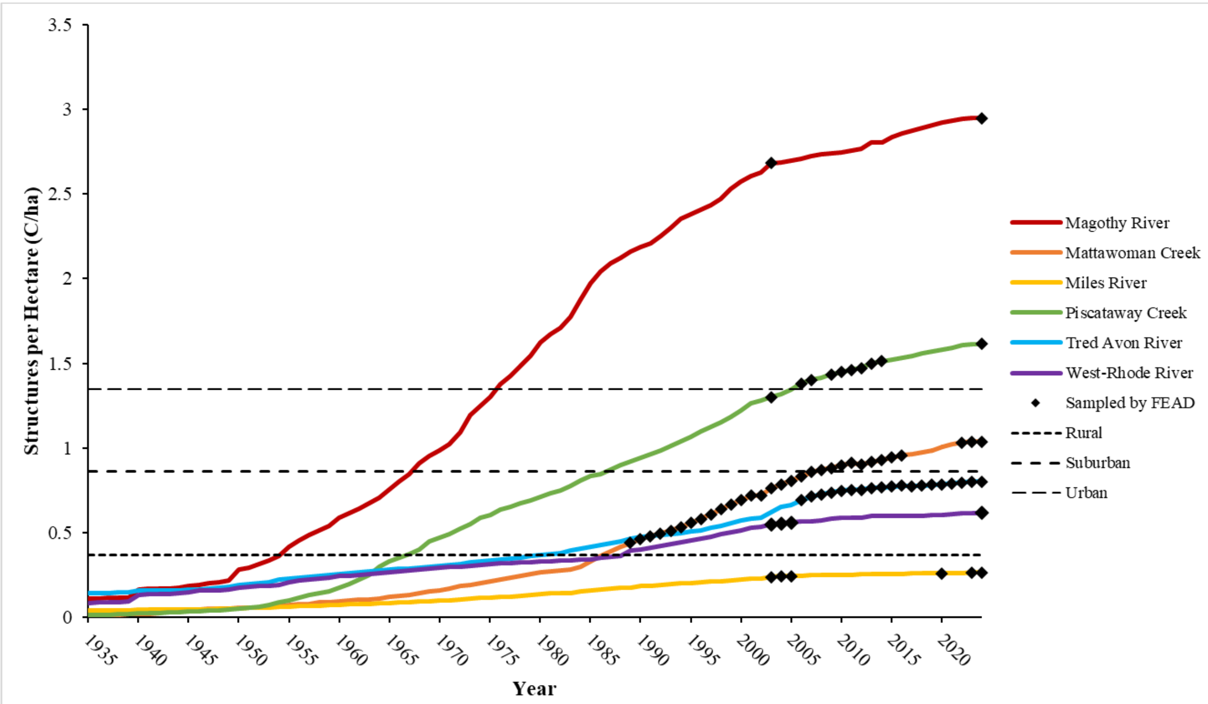


Figure 3-4. All surface water temperatures recorded in the 2024 summer season.

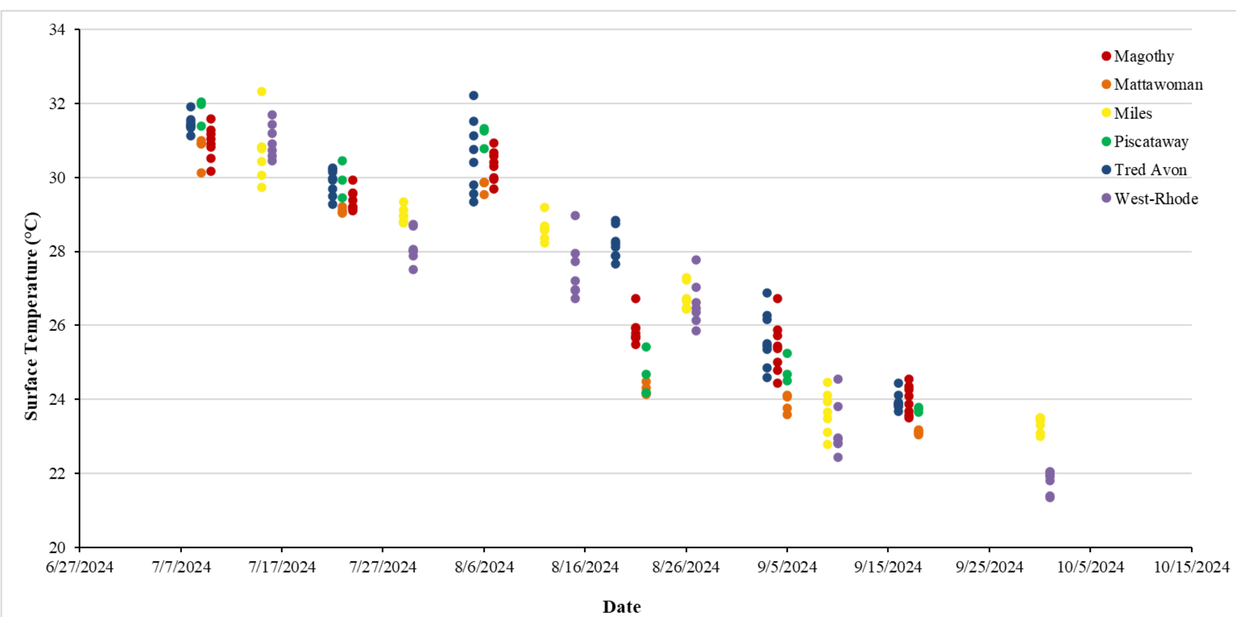


Figure 3-5. Median annual surface water temperature for Tred Avon River (2006-2024) with linear trendline.

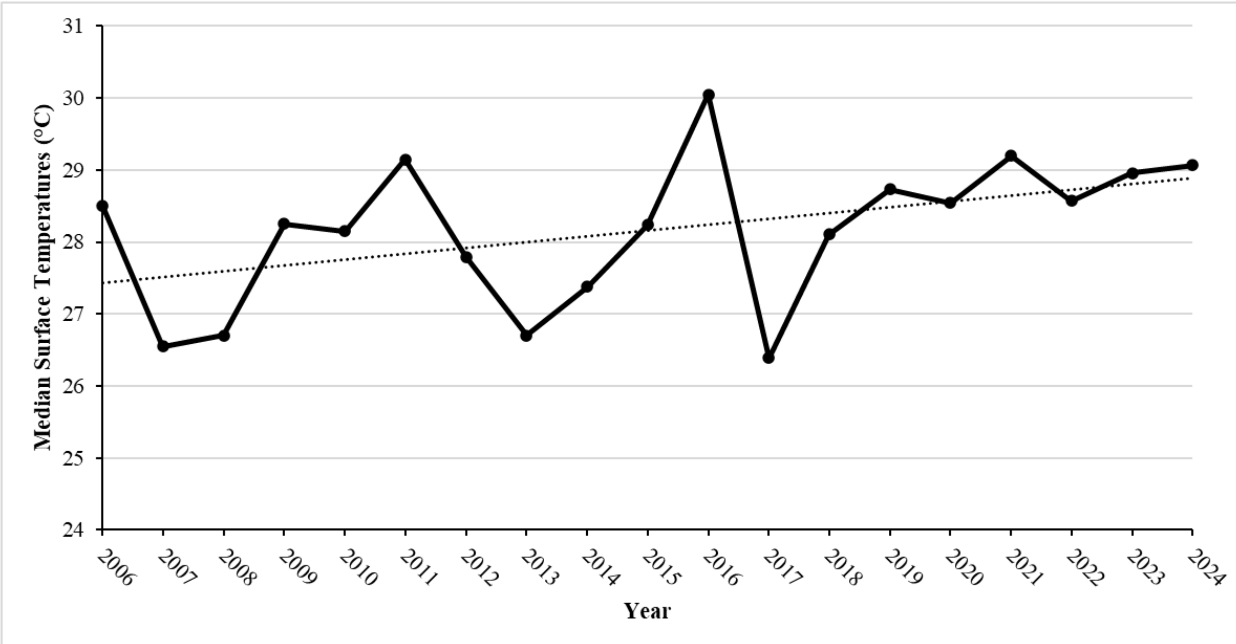


Figure 3-6. Bottom dissolved oxygen (DO; mg/L) readings in 2024 versus intensity of development (C/ha or structures per hectare). Target (5 mg/L) and threshold (3 mg/L) DO boundaries are indicated by the red dashed lines. Piscataway Creek was omitted due to insufficient depth (<2 m) at sites for bottom measurements in 2024.

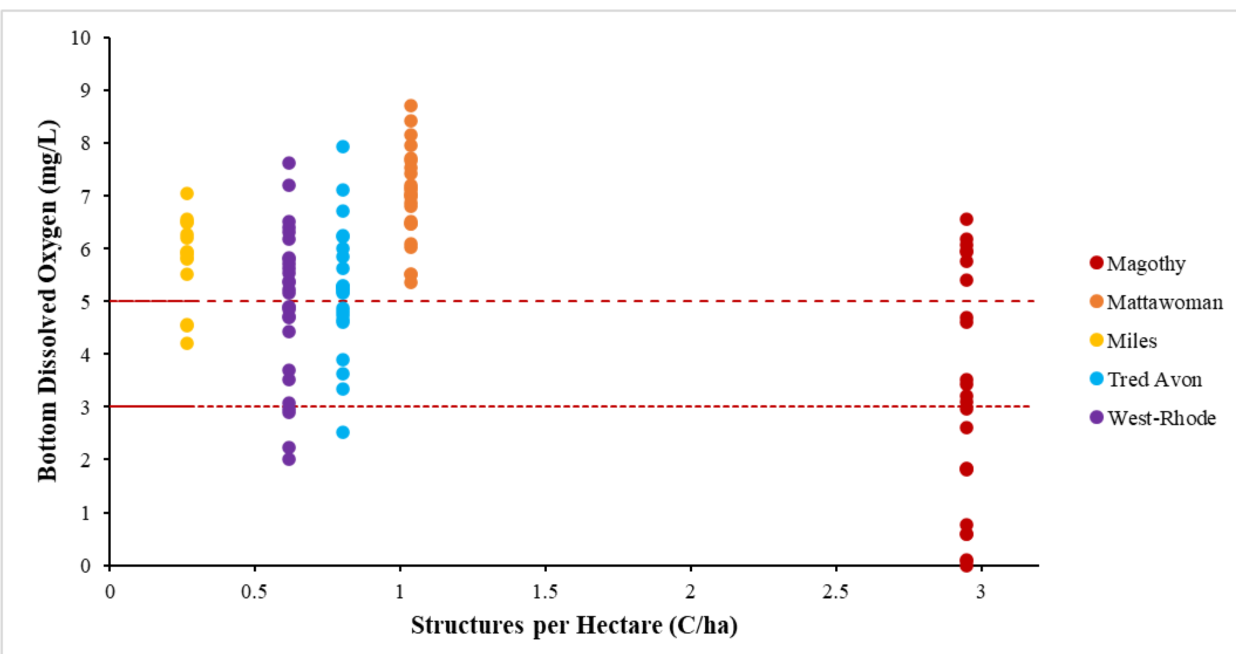


Figure 3-7. Median bottom dissolved oxygen (DO; mg/L) time-series for subestuaries sampled in 2024 by salinity class. Dashed red lines represent target DO (5 mg/L) and threshold DO (3 mg/L). Blue symbols represent mesohaline systems, orange symbols represent tidal-fresh.

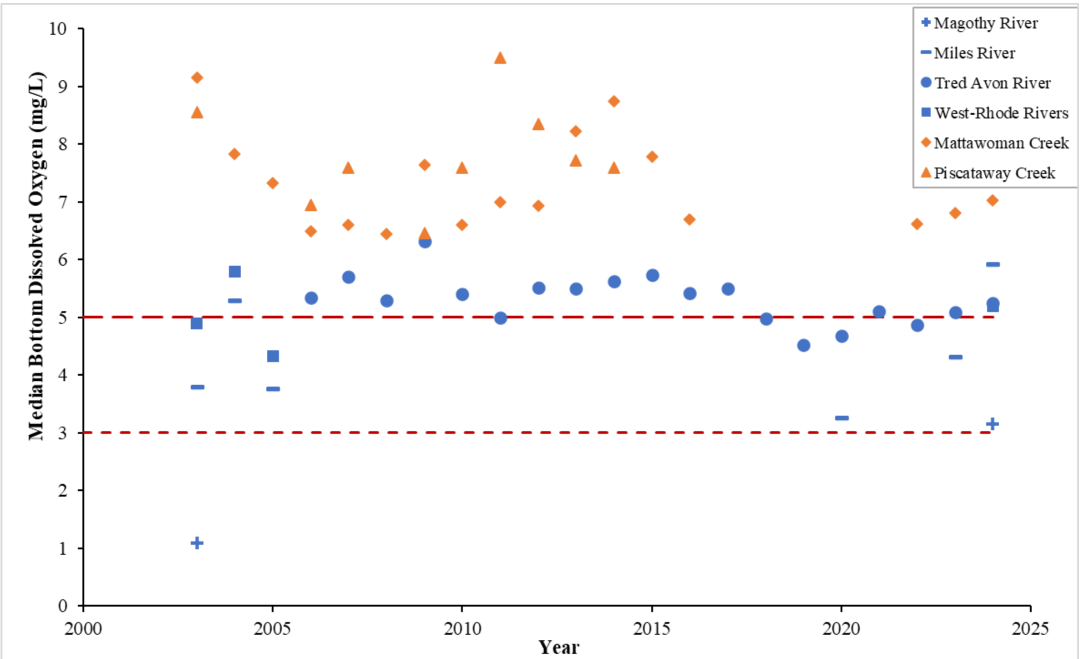


Figure 3-8. Median subestuary surface dissolved oxygen in 2024 (data from 2003-2024) and level of development (C/ha or structures per hectare). Dashed red lines represent target DO (5 mg/L) and threshold DO (3 mg/L). Blue symbols represent mesohaline systems, orange symbols represent tidal-fresh.

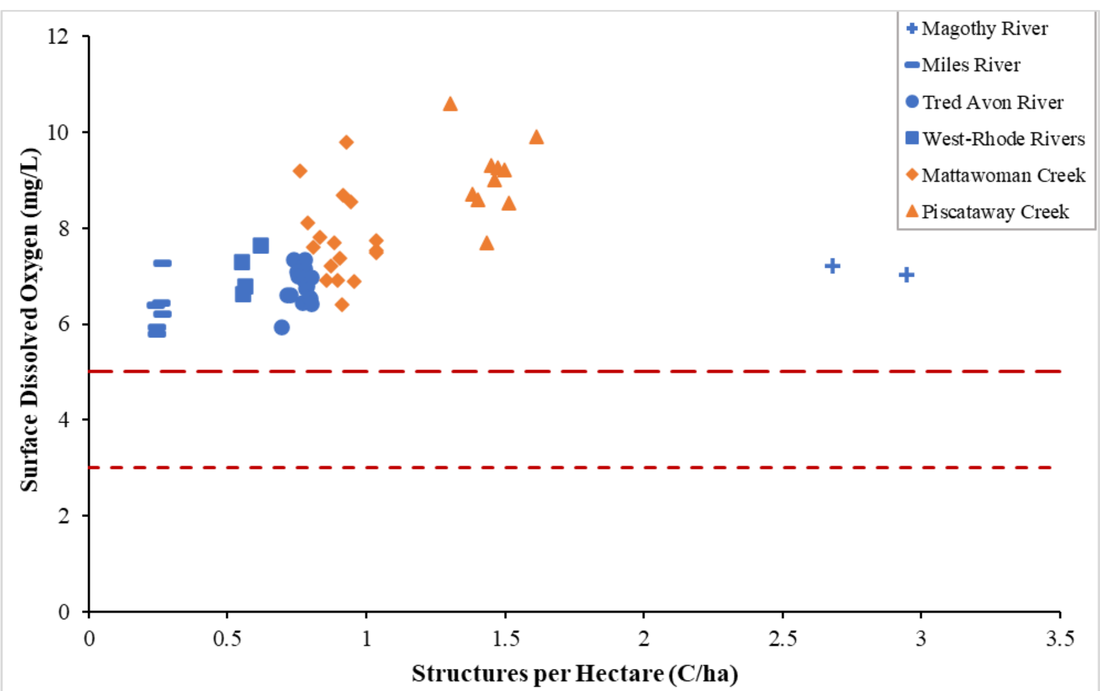


Figure 3-9. Estimates of agricultural land cover (% watershed land area) versus median bottom dissolved oxygen (DO; mg/L) in mesohaline subestuaries (2003-2024). The quadratic model predicts median bottom DO and agricultural coverage (%).

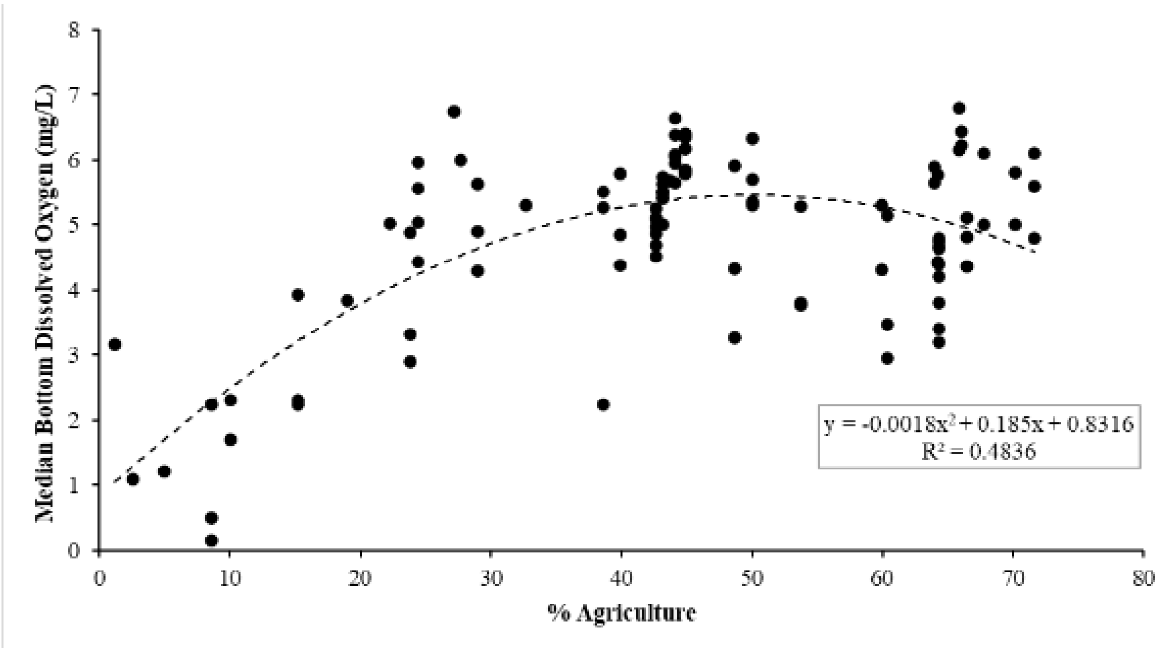


Figure 3-10. Median Secchi depth (m) time series for subestuaries sampled in 2024 (data from 2003-2024). Blue symbols represent mesohaline systems, orange symbols represent tidal-fresh.

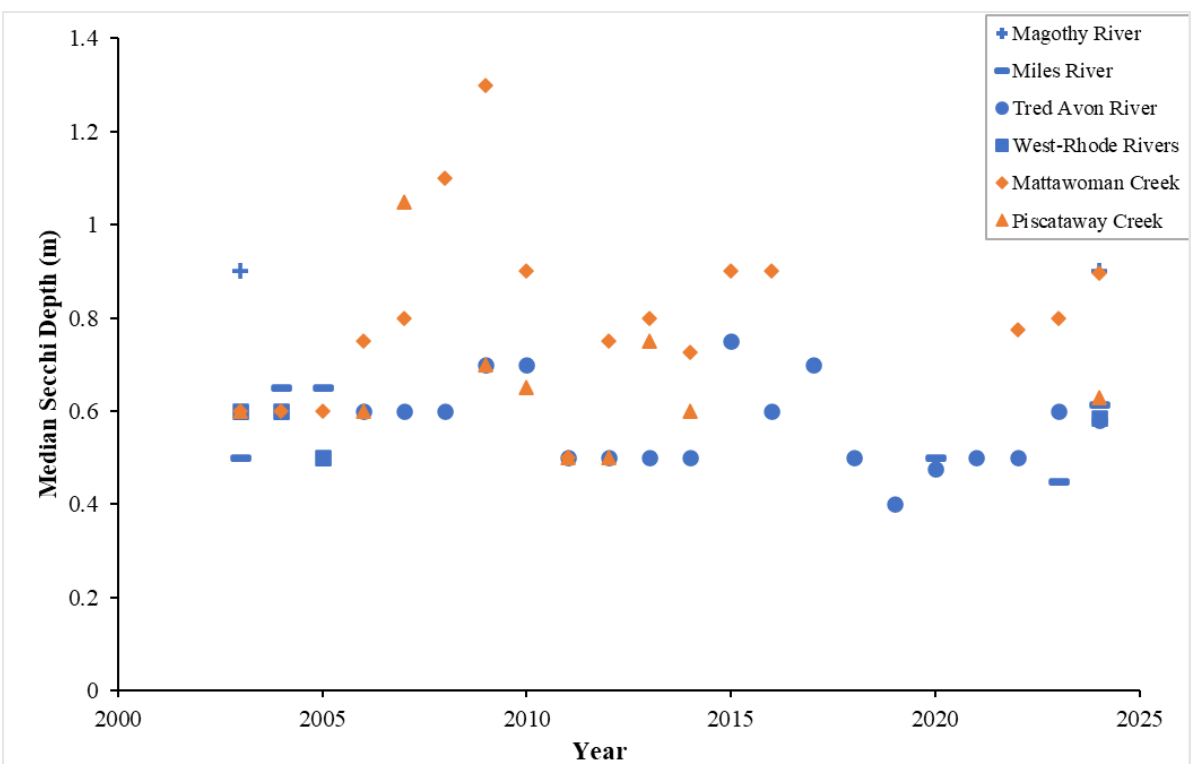


Figure 3-11. Ward's Minimum Variance Cluster Analysis for all fish captured in beach seines in the subestuaries sampled in 2024.

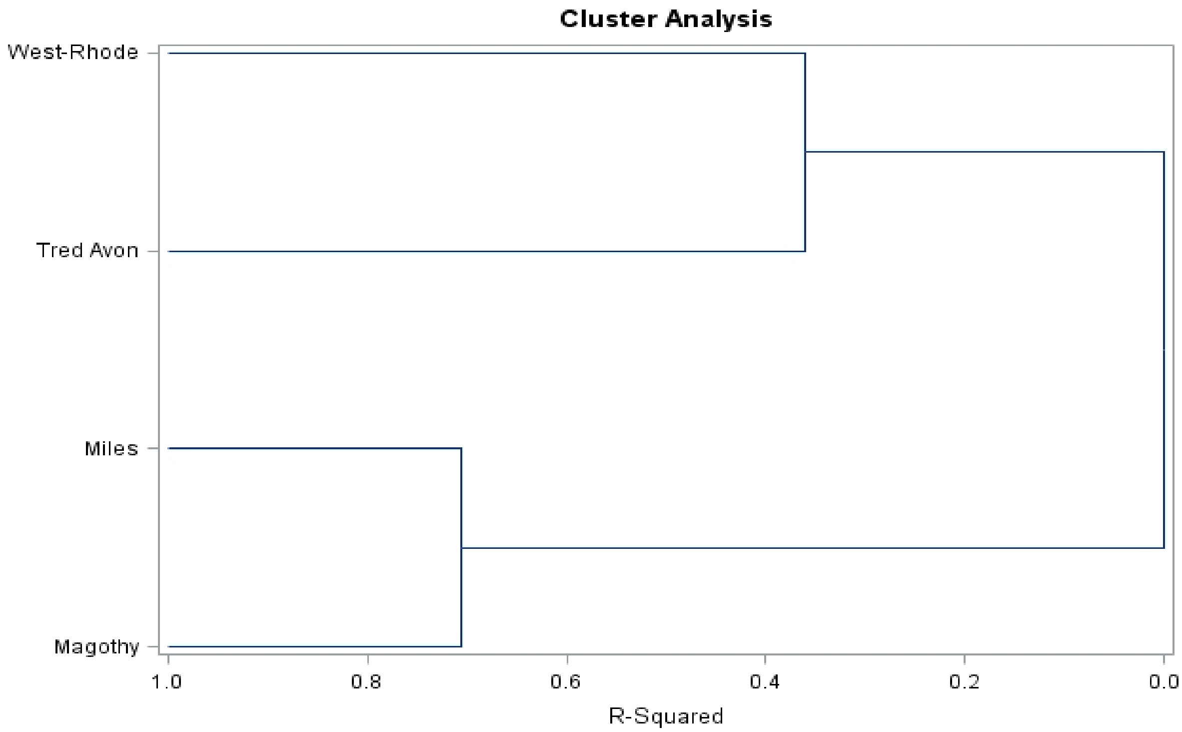


Figure 3-12. Ward's Minimum Variance Cluster Analysis for all fish captured in bottom trawls in the subestuaries sampled in 2024.

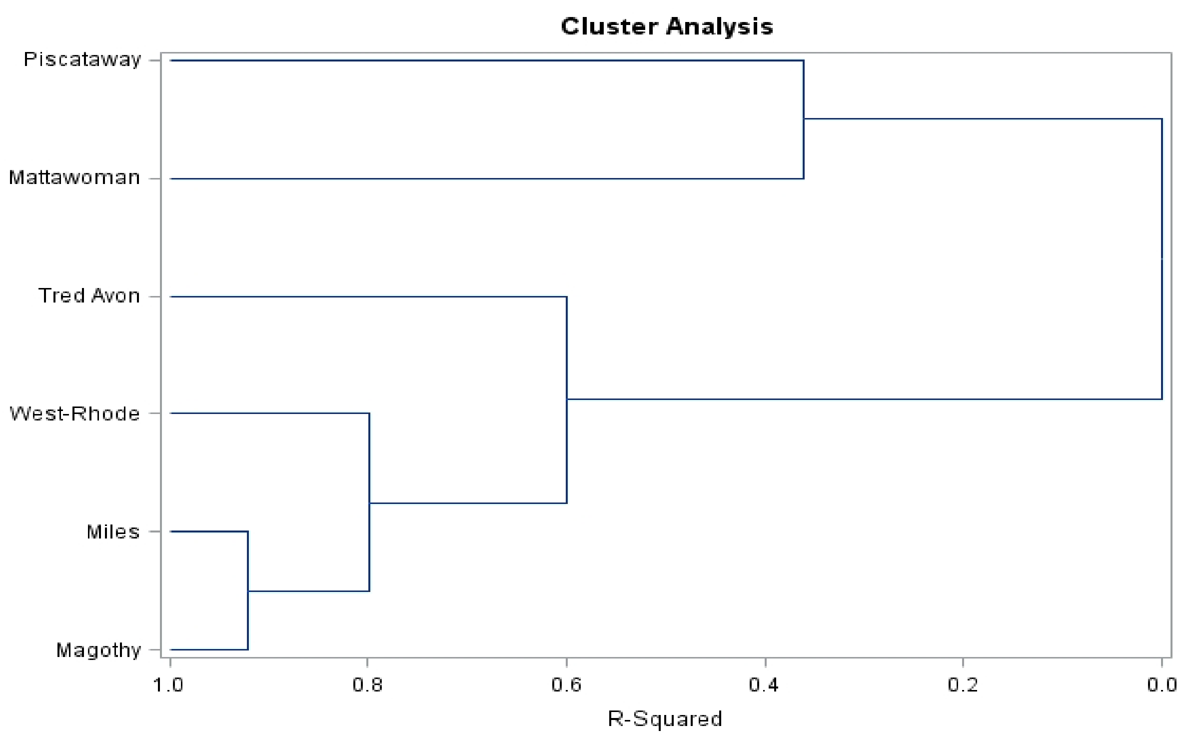


Figure 3-13a. Annual bottom trawl catch geometric mean (GM) for oligohaline and tidal-fresh systems from 2003-2024 for all species of finfish in relation to development (C/ha or structures per hectare).

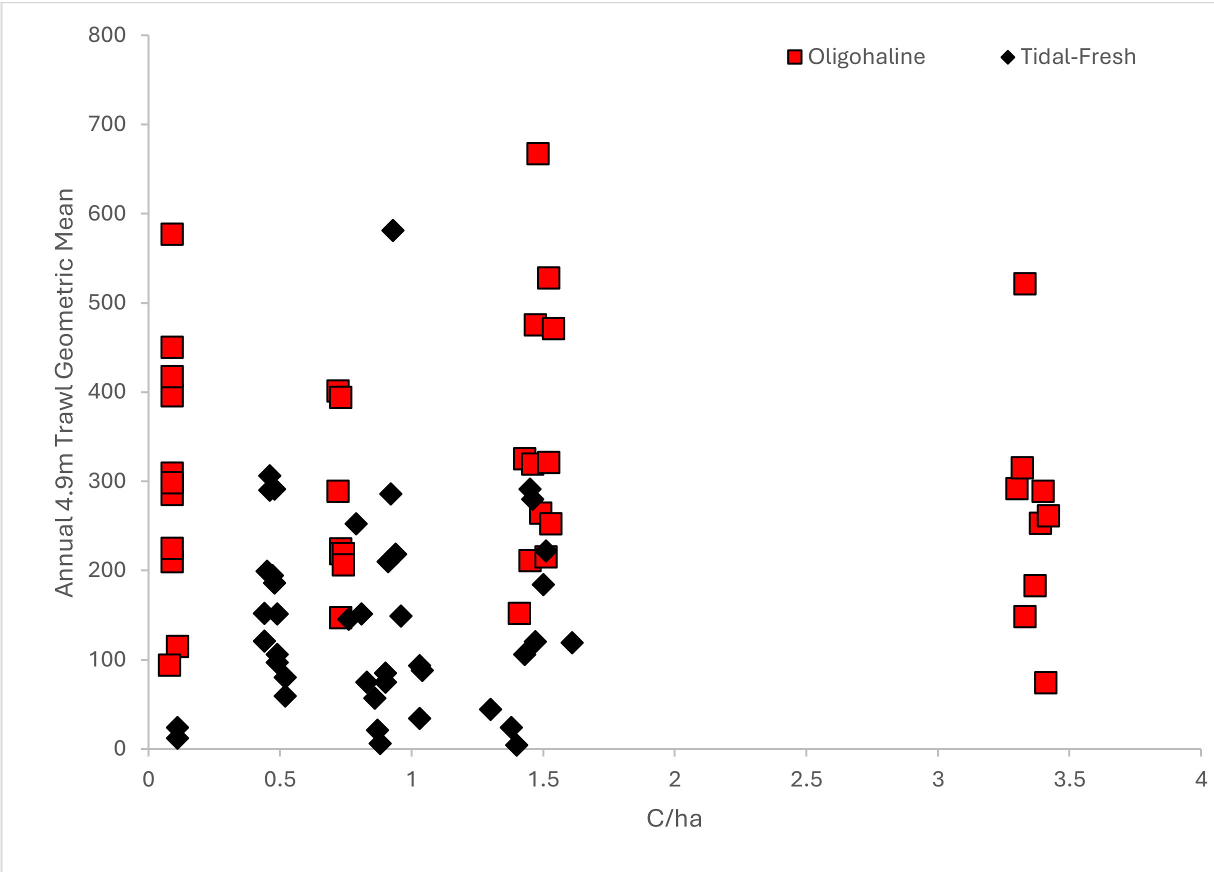


Figure 3-13b. Annual bottom trawl log of geometric mean (GM, all species) of mesohaline systems from 2003 to 2024 in relation to development (C/ha or structures per hectare). Geometric means were grouped by development class (below development target-green, target to threshold-yellow, above threshold-red).

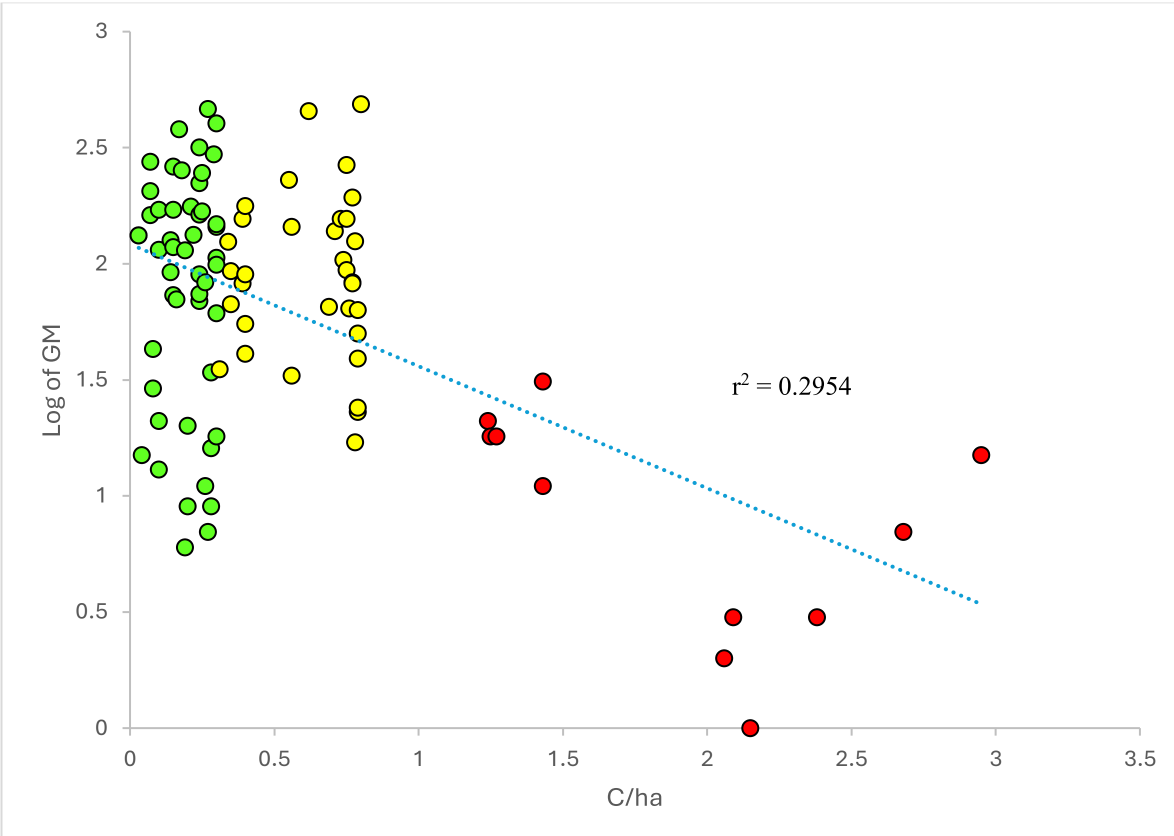


Figure 3-14. Proportion of positive bottom trawl tows and development (C/ha or structures per hectare) for mesohaline subestuaries sampled during 2003-2024. Proportion of positive tows were grouped by development class (below development target-green, target to threshold-yellow, above threshold-red).

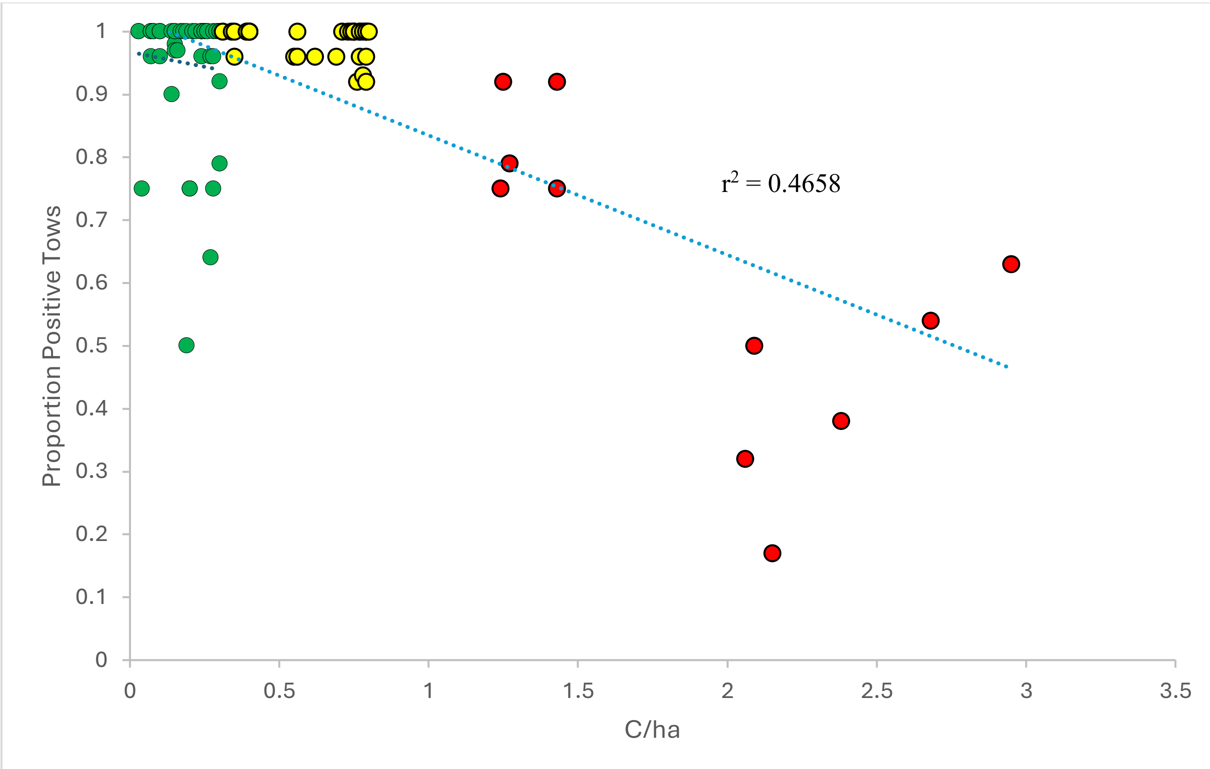


Figure 3-15. Annual proportion of positive bottom trawl tows of mesohaline systems in relation to annual median bottom dissolved oxygen (mg/L). Median bottom dissolved oxygen was grouped by dissolved oxygen levels (above target (>5.0 mg/L) - green, target to threshold (3.0 to 5.0 mg/L) - yellow, below threshold (<3.0 mg/L) - red).

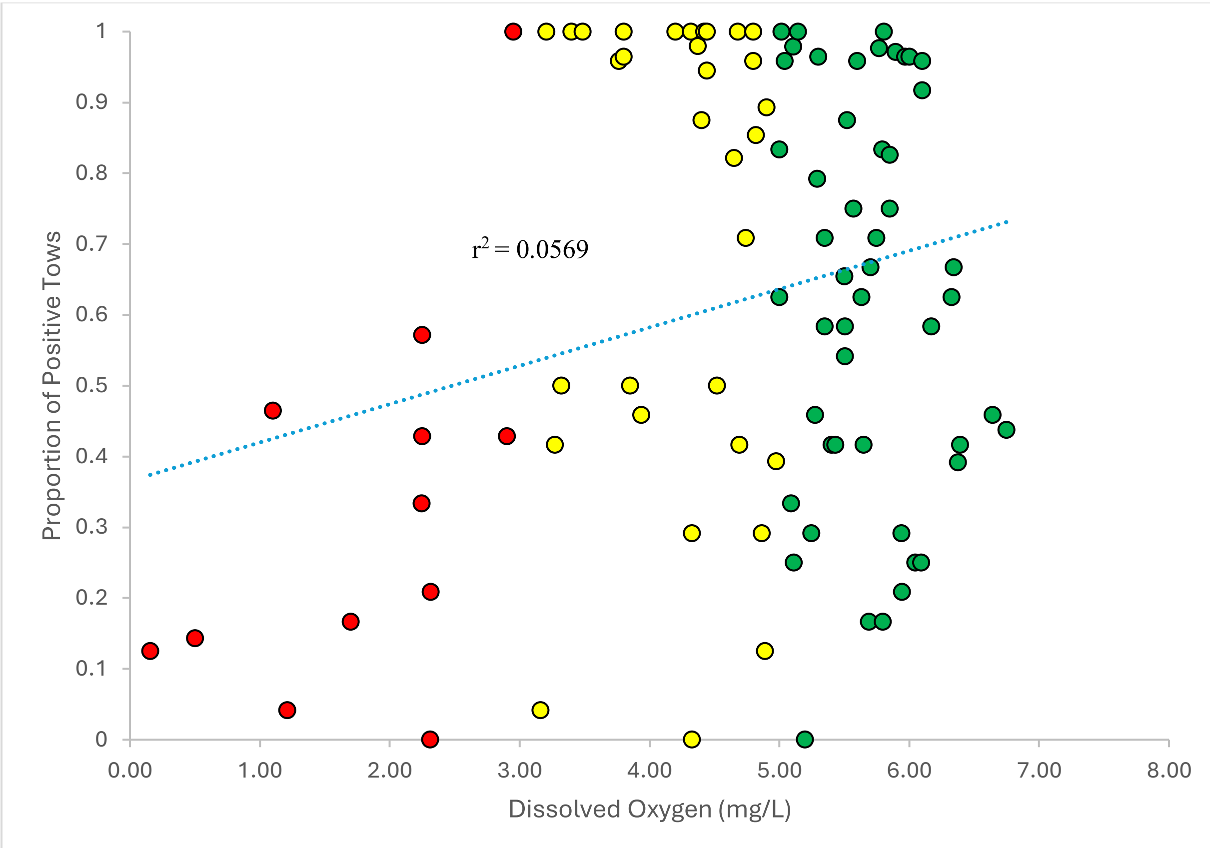


Figure 3-16. Ward's Minimum Variance Cluster Analysis for geometric means of target species captured in beach seines in subestuaries sampled in 2024.

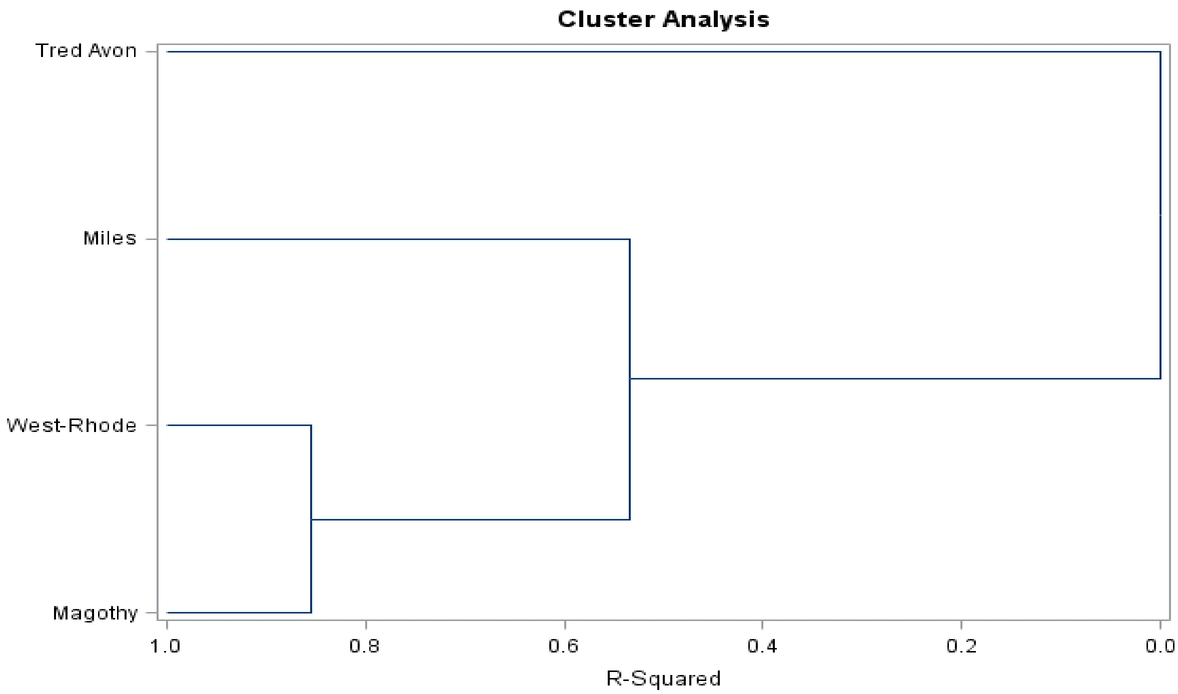


Figure 3-17. Ward's Minimum Variance Cluster Analysis for geometric means of target species captured in bottom trawls in subestuaries sampled in 2024.

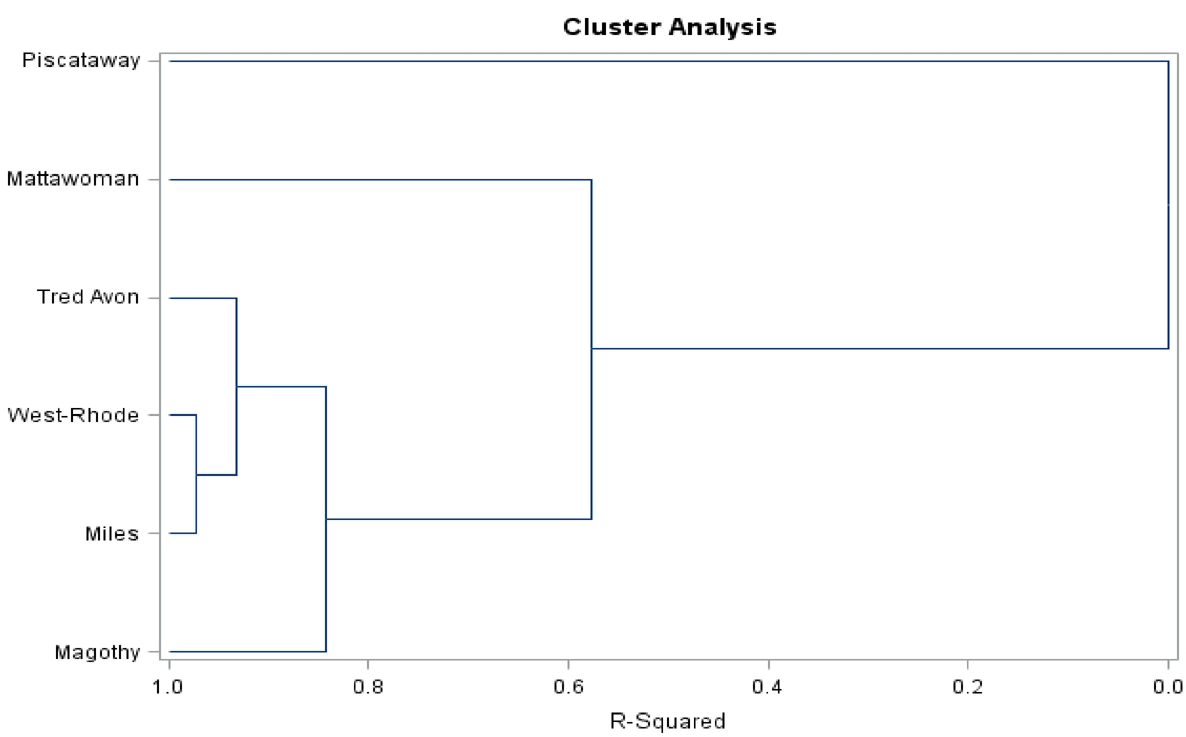
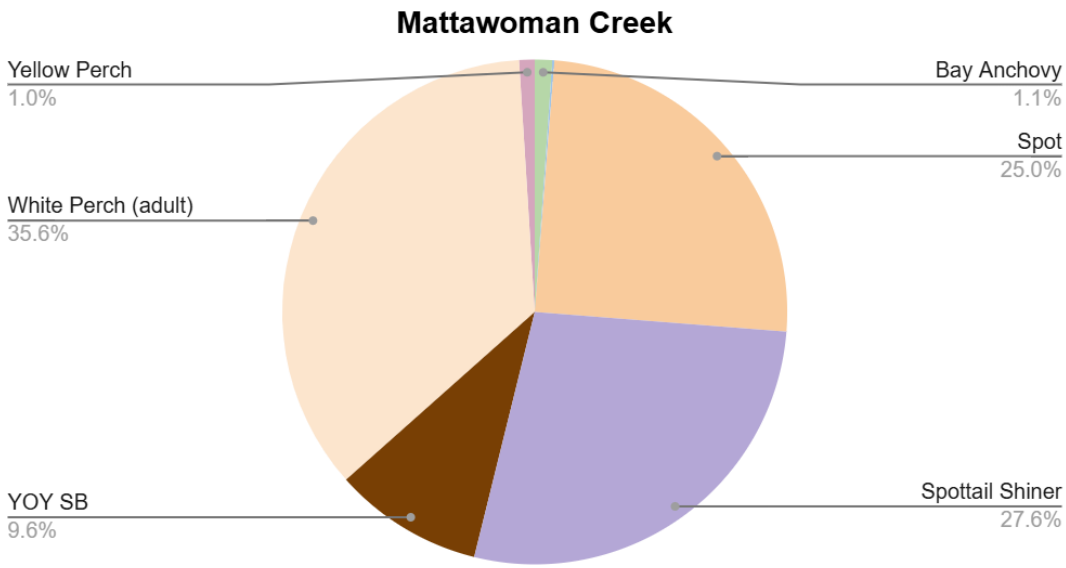


Figure 3-18. Target species catch distributions for 2024 for Mattawoman Creek and Piscataway Creek, with juvenile White Perch catch excluded.



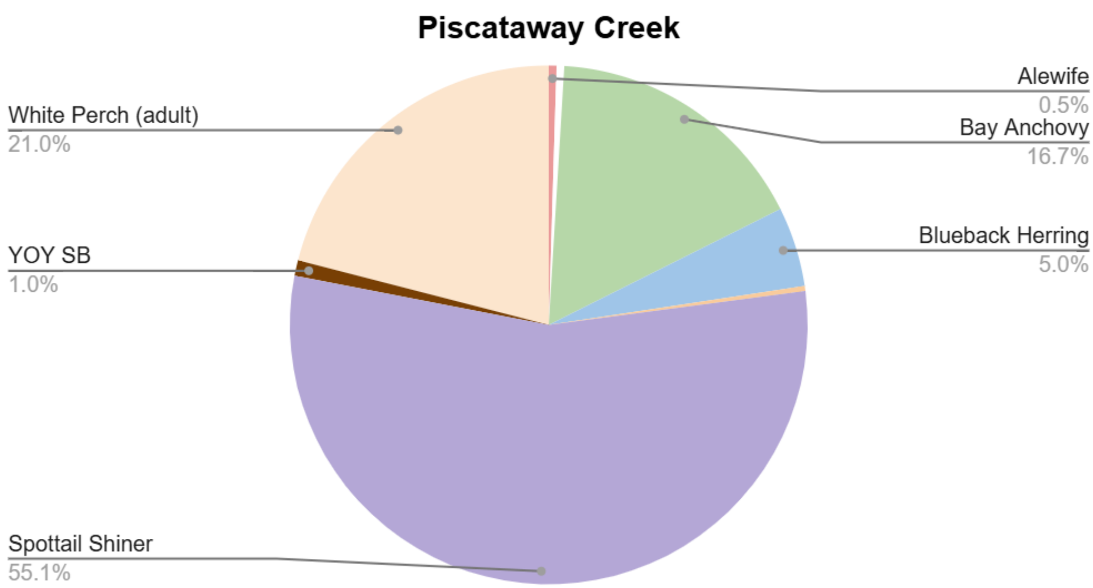
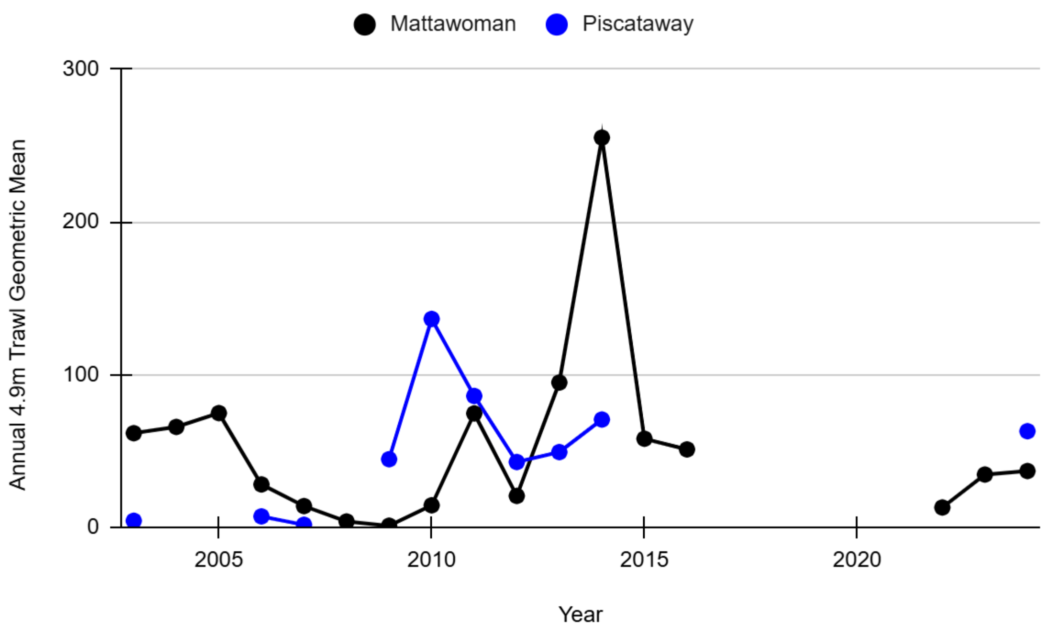


Figure 3-19. Times series of geometric mean for juvenile and adult White Perch in Mattawoman Creek and Piscataway Creek for years sampled from 2003-2024.

a. Juvenile White Perch



b. Adult White Perch

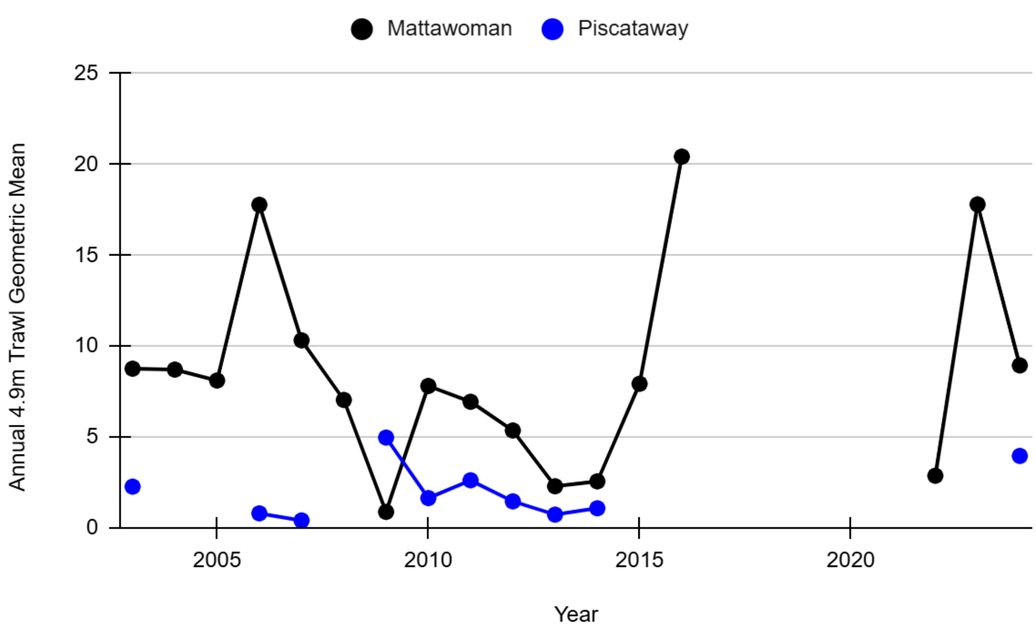


Figure 3-20. Atlantic Menhaden proportion of positive beach seine hauls (P-A) from mesohaline systems sampled 2003-2024.

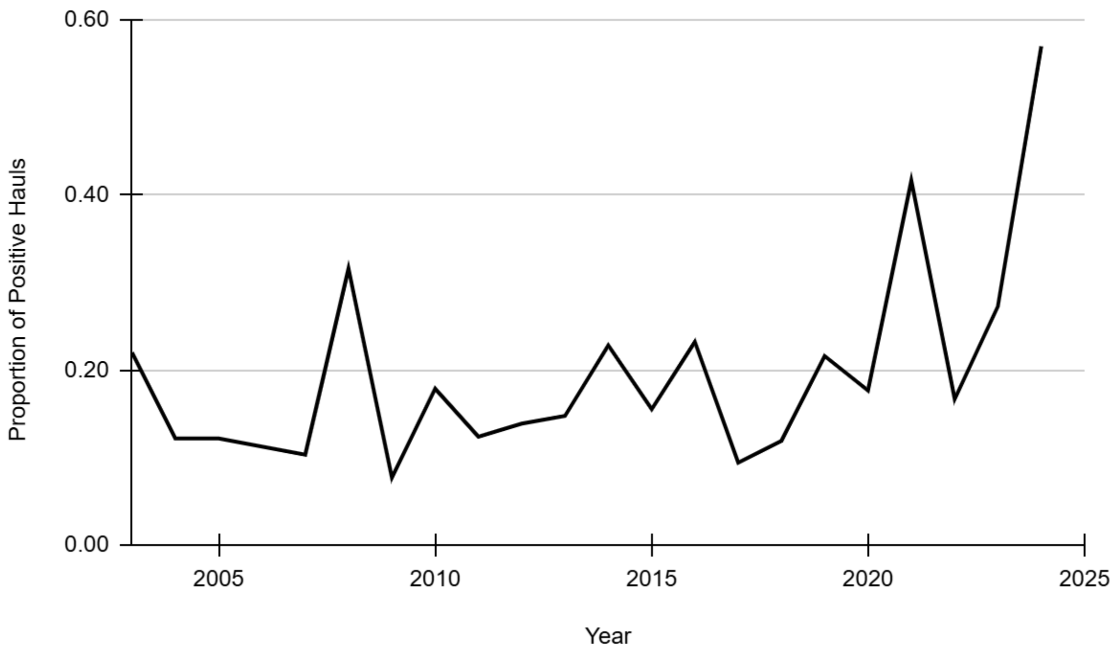


Figure 3-21. Eastern Silvery Minnow bottom trawl geometric mean (GM) from tidal-fresh Mattawoman and Piscataway Creek for years sampled from 2003-2024.

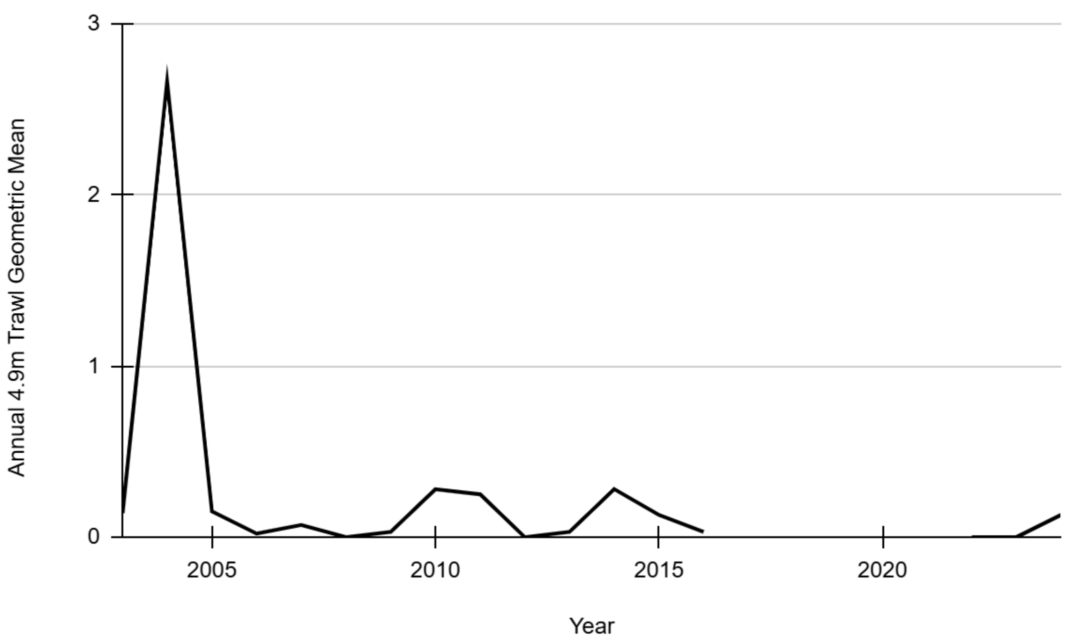


Figure 3-22. Spottail Shiner bottom trawl geometric mean (GM) from tidal-fresh Mattawoman and Piscataway Creeks for years sampled from 2003-2024.

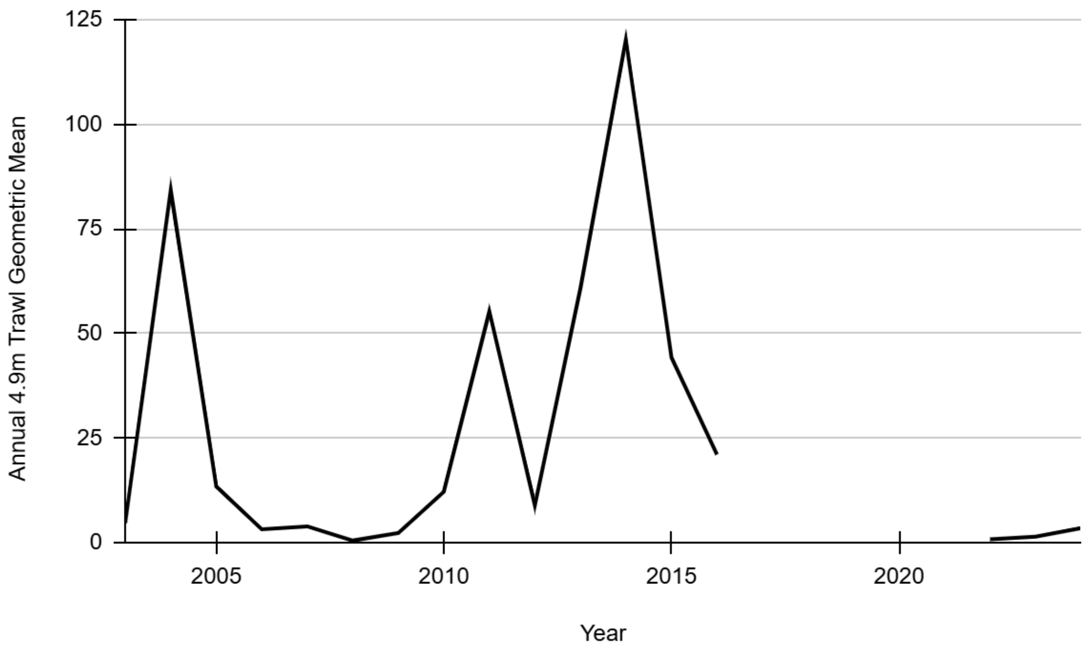


Figure 3-23. Gizzard Shad beach seine geometric mean (GM) from all systems sampled 2003-2024.

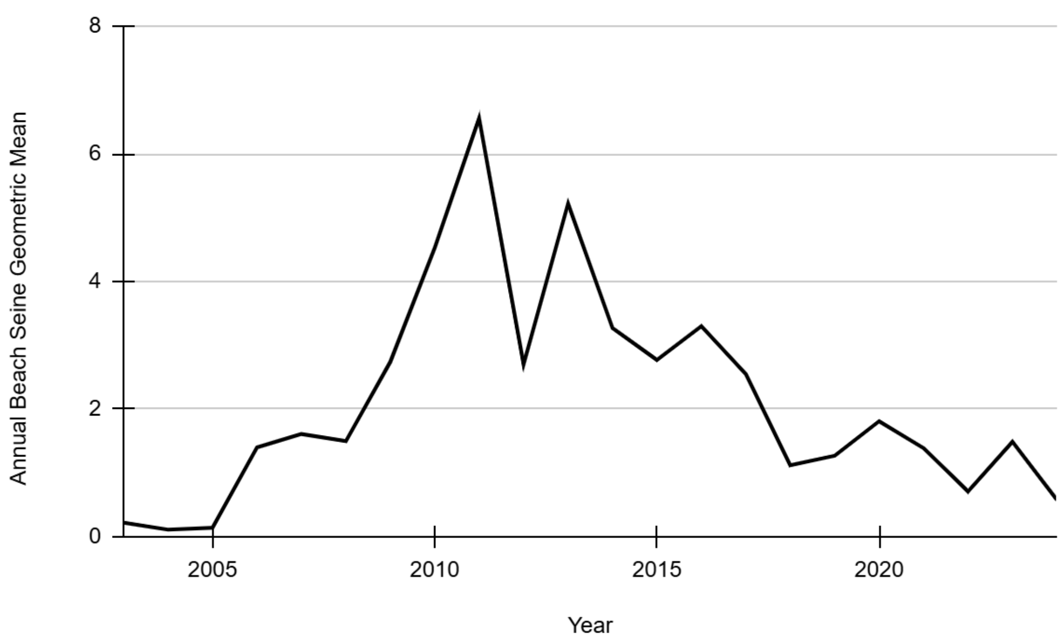


Figure 3-24. Bay Anchovy bottom trawl geometric mean (GM) from all systems sampled 2003-2024.

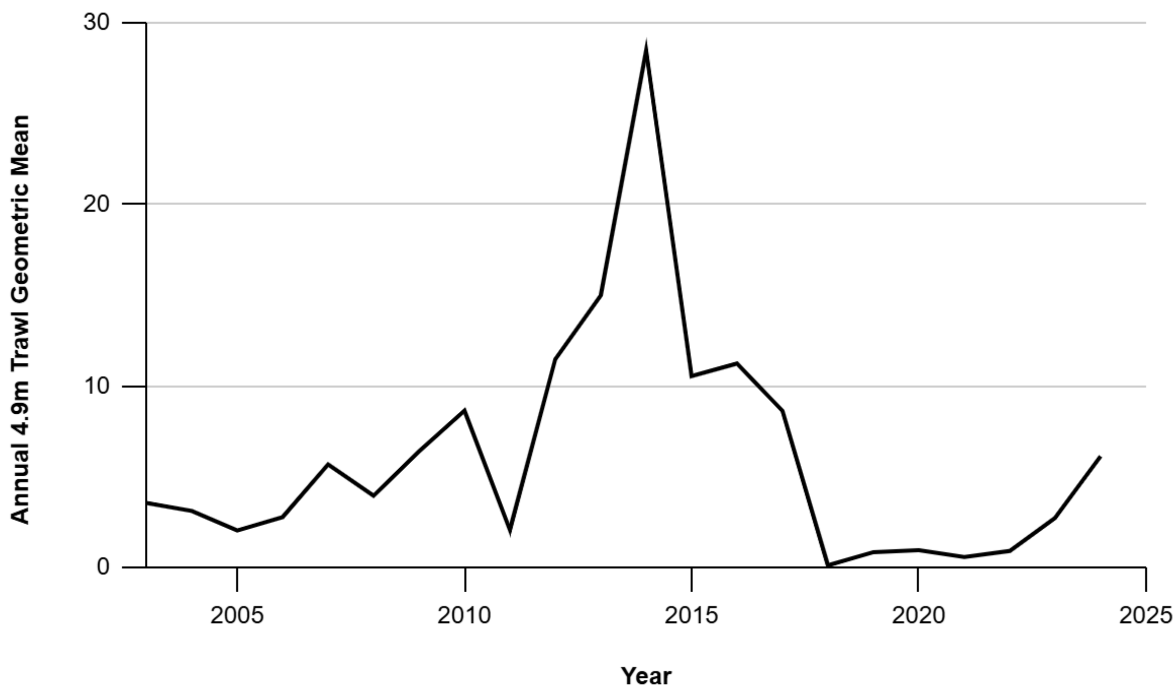


Figure 3-25. Young of year (YOY) Striped Bass beach seine geometric mean (GM) from Magothy, Miles, Tred Avon, and West/Rhode River. Also includes data from the Striped Bass Program Juvenile Abundance Index (JAI) for Bay-wide, Upper Bay, and Choptank River. West-Rhode River in 2003 (GM of 69.4) was excluded from the figure to provide better scale in the remaining data.

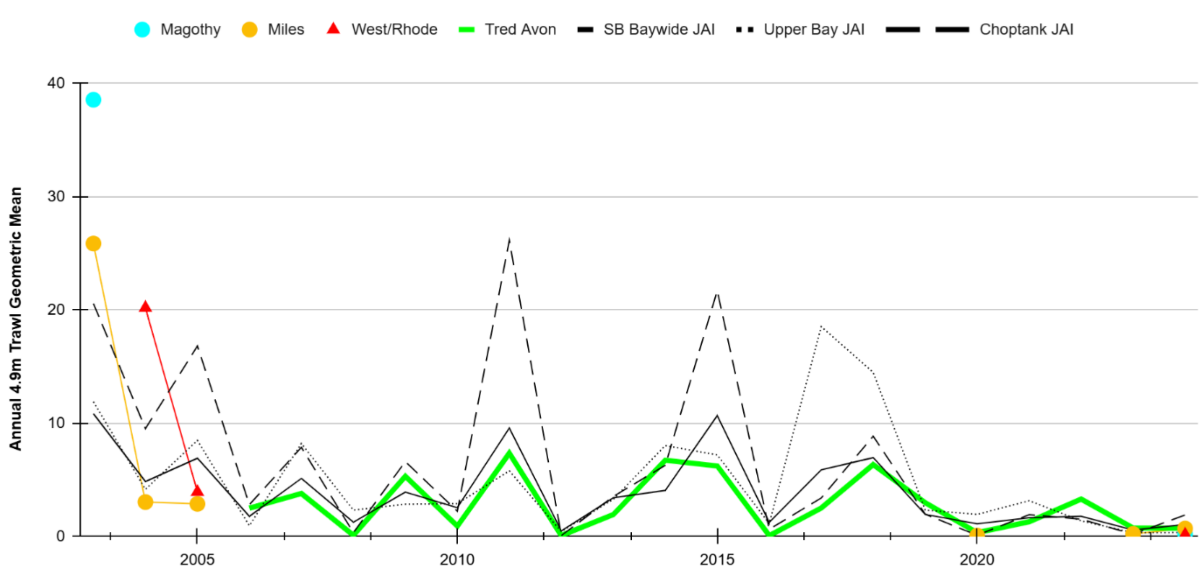


Figure 3-26a. Juvenile White Perch proportion of positive bottom trawl tows (P-A) and development (C/ha) in mesohaline systems sampled 2003 to 2024. Proportion of positive tows were grouped by development class (below development target-green, target to threshold-yellow, above threshold-red). Larger points represent the median P-A for that development class.

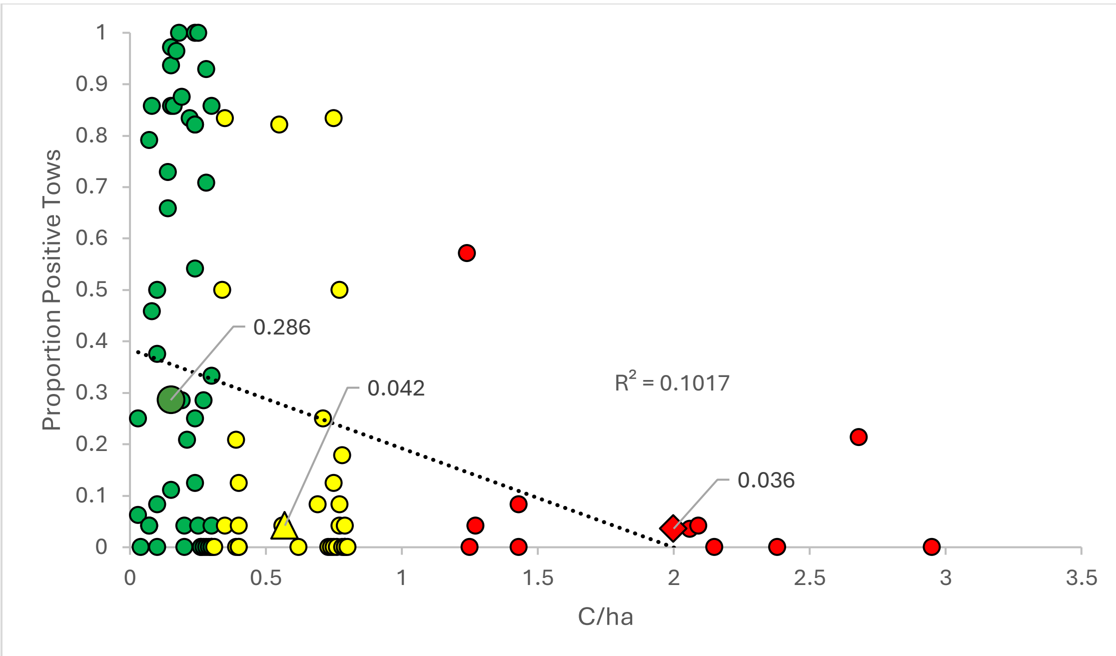


Figure 3-26b. Juvenile White Perch positive bottom trawl tows (P-A) and development (C/ha) in tidal-fresh and oligohaline systems sampled 2003 to 2024. Red squares are oligohaline systems and black diamonds are tidal-fresh systems.

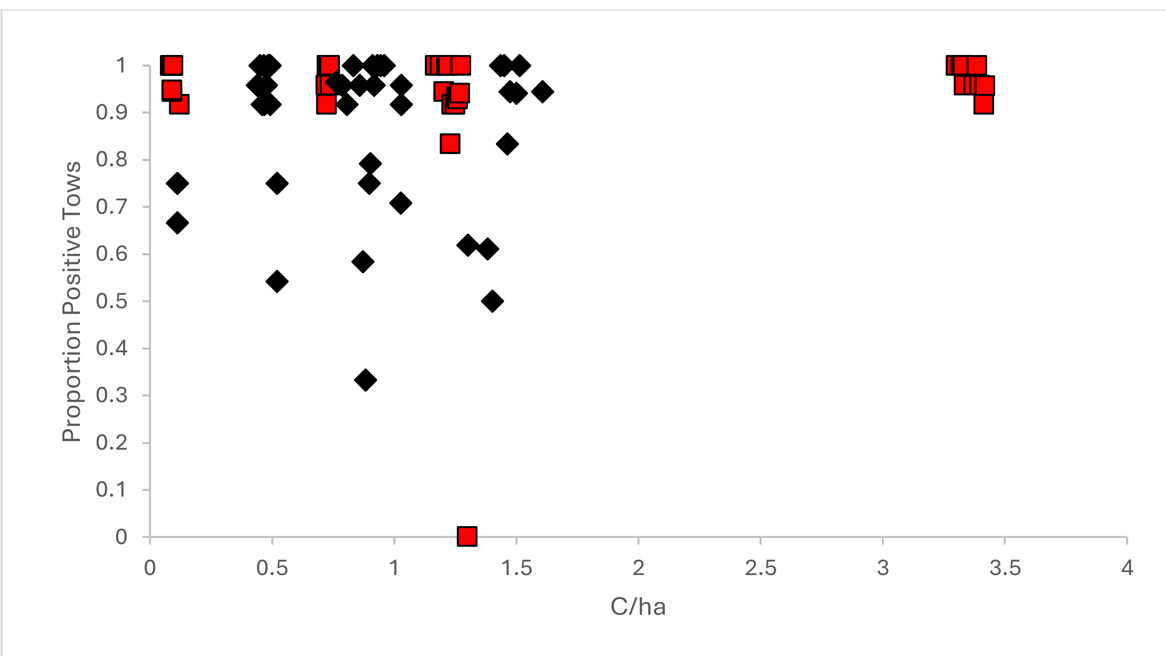


Figure 3-27. Juvenile White Perch positive bottom trawl tows (P-A) over time for the Tred Avon River (2006-2024). Juvenile White Perch juvenile abundance index data from the Striped Bass Bay-wide JAI survey is included for comparison.

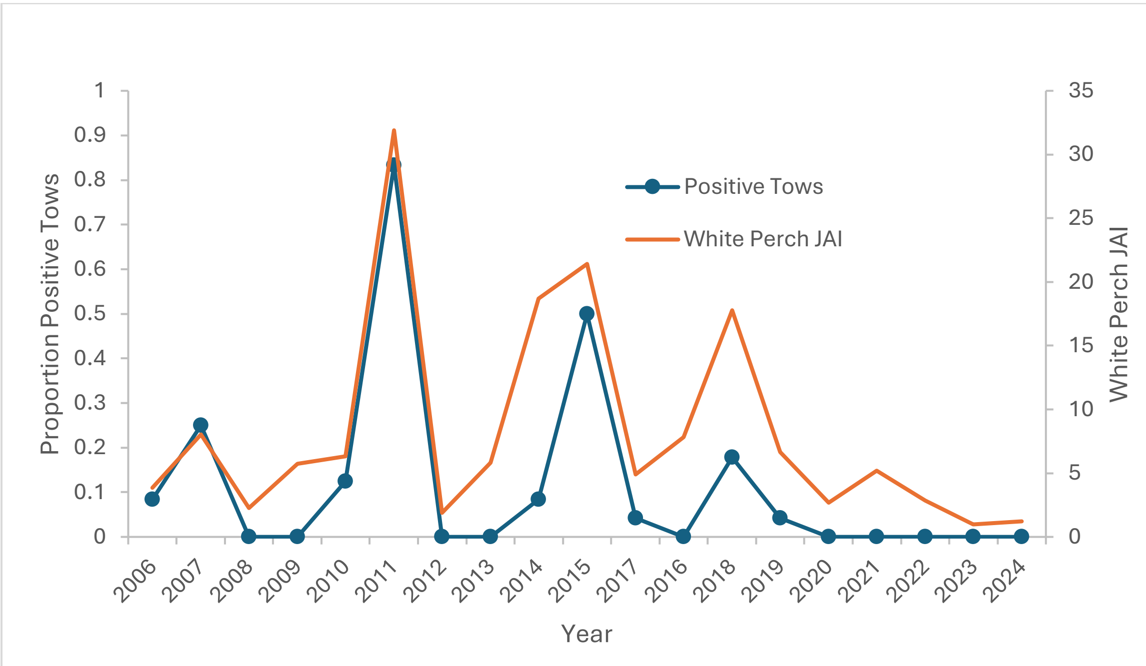


Figure 3-28. Juvenile White Perch positive bottom trawl tows (P-A) at all sites with development (C/ha) for the Tred Avon River (2006-2024).

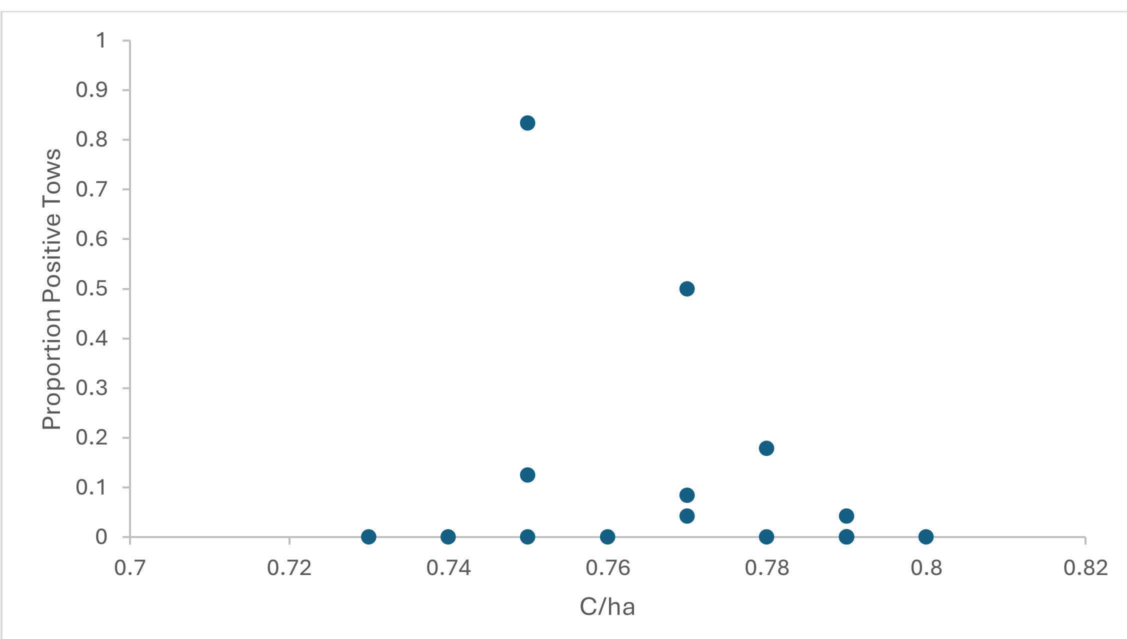


Figure 3-29. Adult White Perch positive bottom trawl tows (P-A) and development (C/ha) in mesohaline systems sampled 2003 to 2024. Proportion of positive tows were grouped by development class (below development target-green, target to threshold-yellow, above threshold-red). Larger points represent the median P-A for that development class.

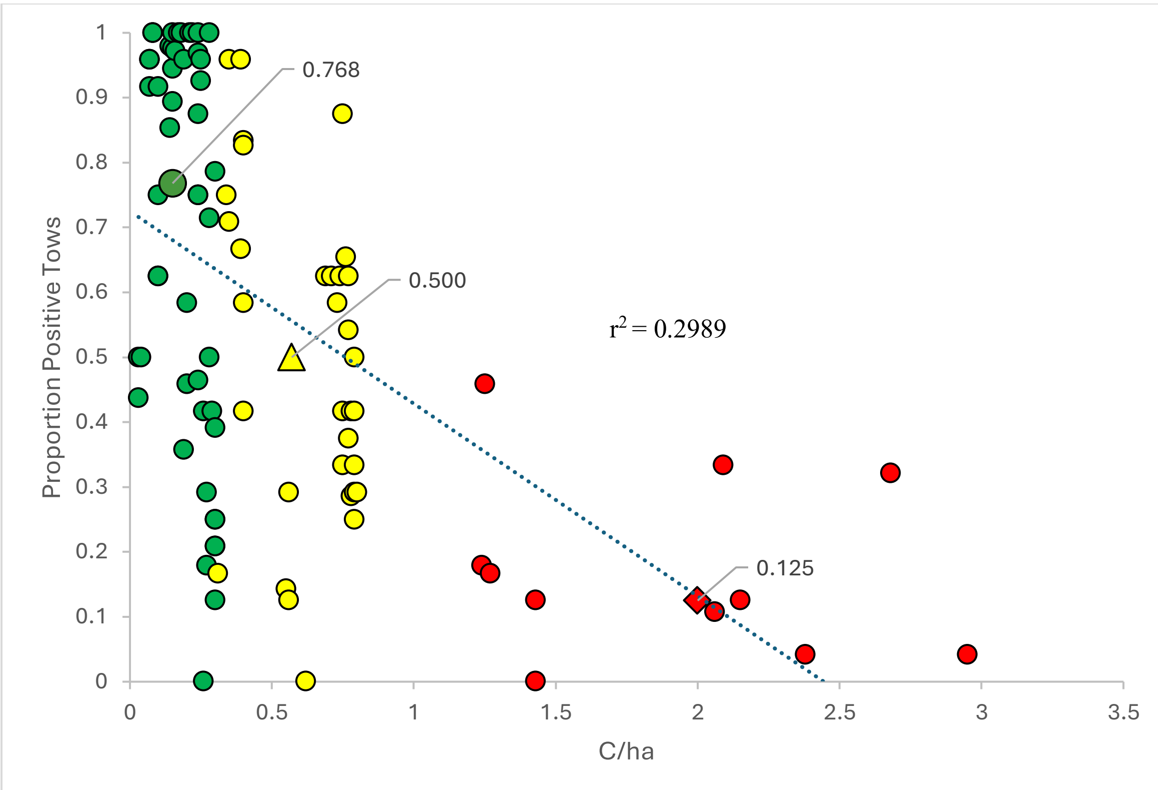


Figure 3-30. Adult White Perch positive bottom trawl tows (P-A) and median bottom dissolved oxygen (mg/L). Median bottom dissolved oxygen was grouped by dissolved oxygen levels (above target (>5.0 mg/L) - green, target to threshold (3.0 to 5.0 mg/L) - yellow, below threshold (<3.0 mg/L) - red).

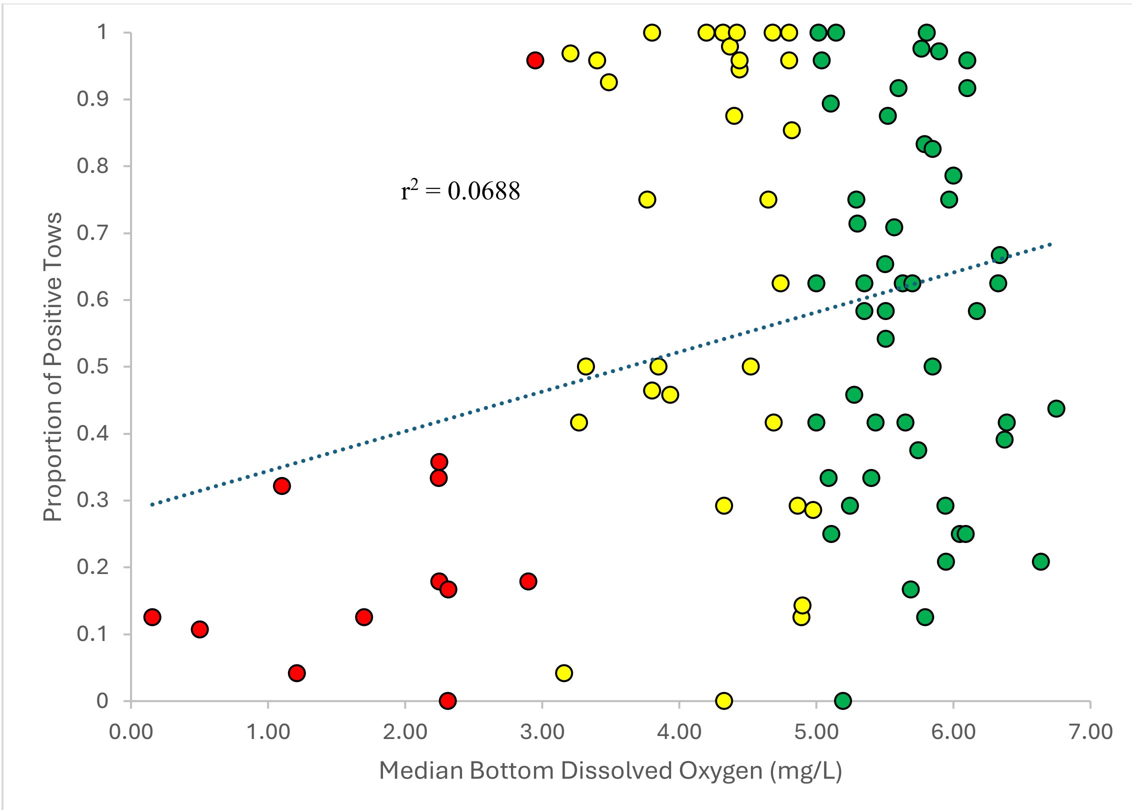


Figure 3-31. Adult White Perch positive bottom trawl tows (P-A) over time for Tred Avon River (blue line; 2006-2024) and 2-year lagged geometric mean from JAI for the Choptank River (orange line).

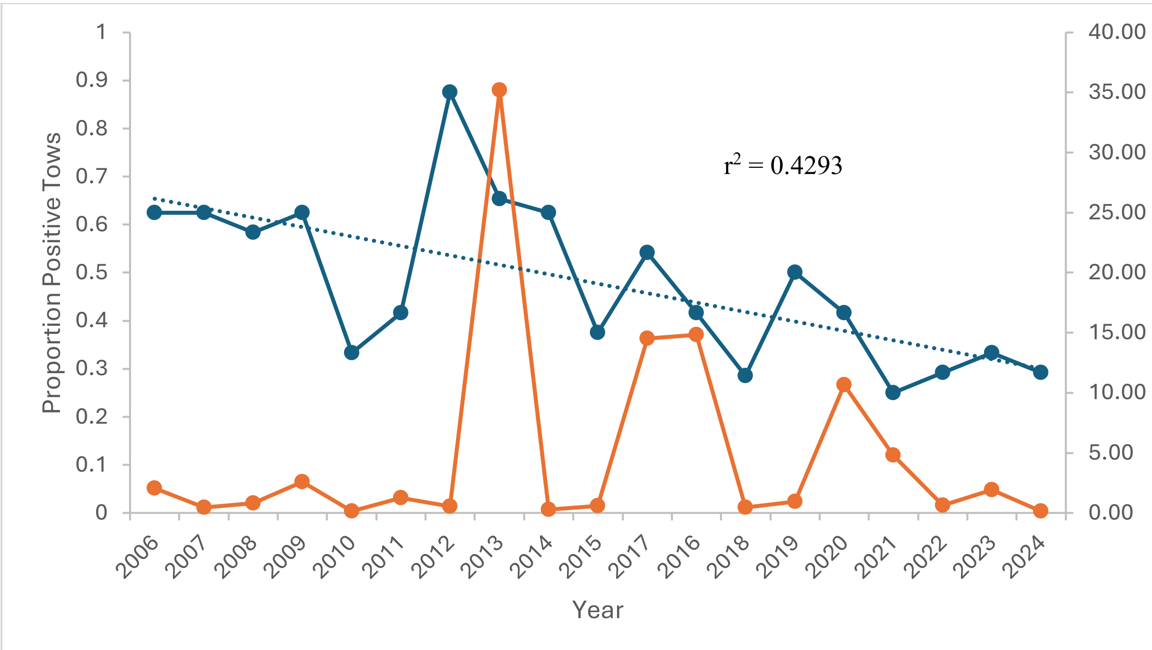


Figure 3-32. Adult White Perch positive bottom trawl tows (P-A) with development (C/ha) for Tred Avon River (2006-2024).

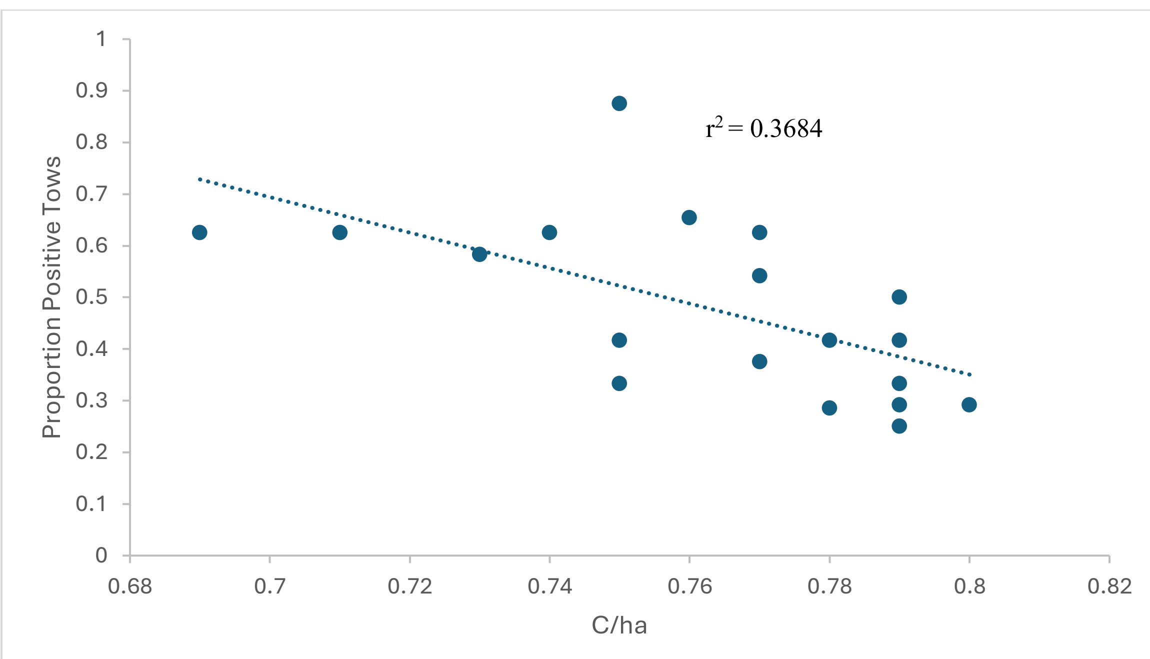


Figure 3-33. Linear relationship of adult White Perch annual positive bottom trawl tows (P-A) and median bottom dissolved oxygen for Tred Avon River (2006-2024).

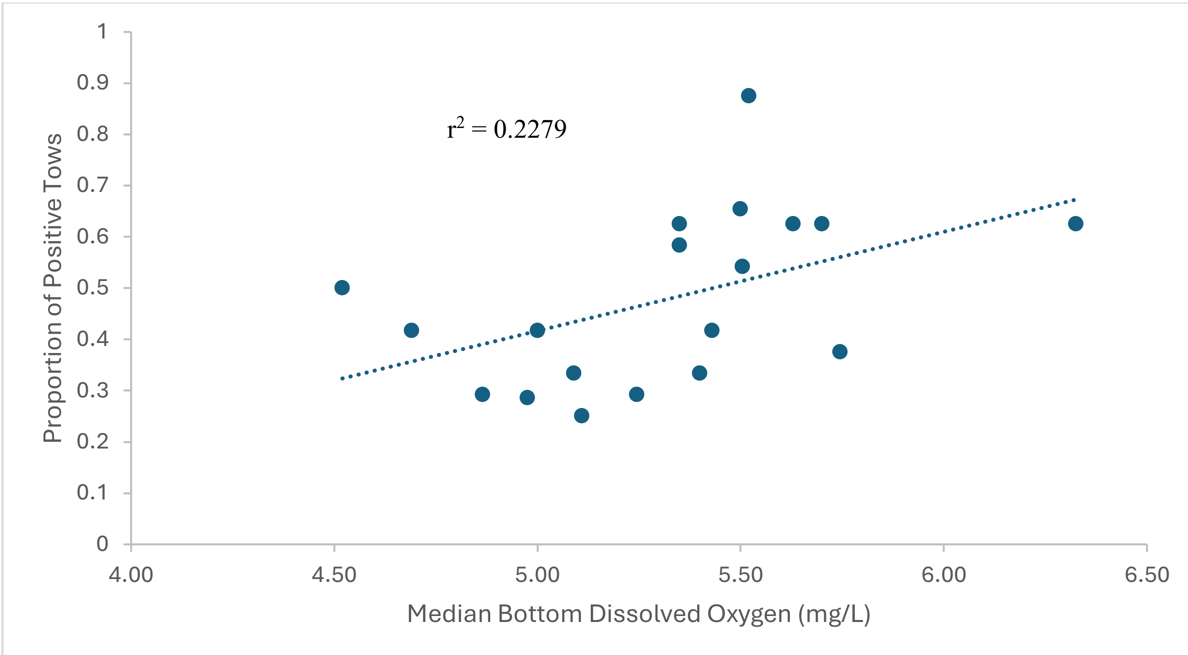


Figure 3-34. White Perch proportional stock density (PSD) from bottom trawl data over time for Mattawoman Creek, Miles River, Piscataway Creek and Tred Avon River (2003 to 2024).

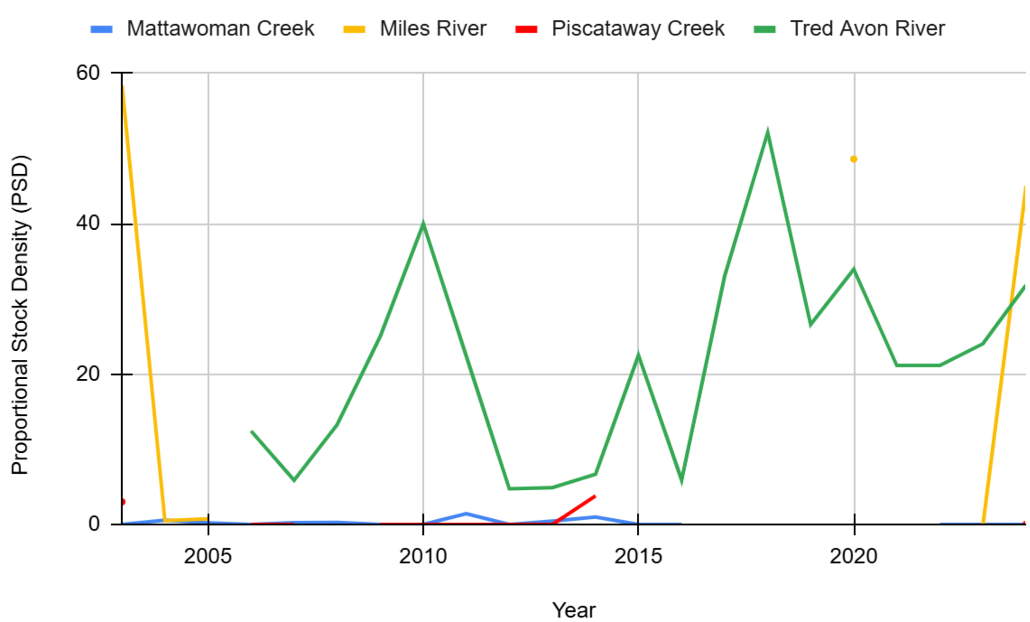
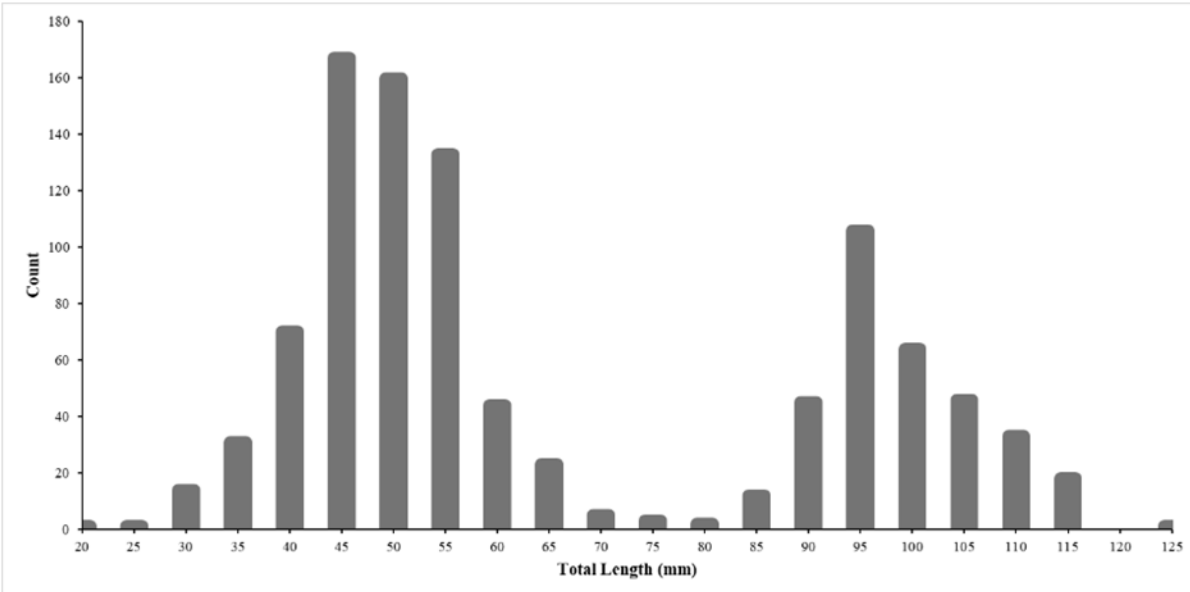


Figure 3-35. Length frequency distribution of juvenile White Perch sampled in Mattawoman and Piscataway Creek in 2024.



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, Jeffrey Horne, Marek Topolski, Shannon Moorhead, Marisa Ponte, Robin Minch, Zophia Galvan

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, communication, 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, 2024 – June 30, 2025. 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 sampling studies were refreshed with current projects and pictures. We also updated the Goal and Objectives of FEAD and updated the Biologist page with current personnel. The webpage can be found at: https://dnr.maryland.gov/fisheries/pages/fhep/index.aspx

Publications - We submitted a manuscript on our zooplankton and larval Striped Bass feeding study during 2023-2024 (see Objective 1, Section 2.2) to Marine and Coastal Fisheries.

Fish Habitat Conservation – Staff reviewed a draft Prince Frederick Town Center Master Plan, which is an amendment to its comprehensive plan adopted in 2019 and updates the current Prince Frederick Master Plan which was adopted in 1989. The plan contained relevant sections for review on land use, environment and natural resources, and water resources. Staff also reviewed comprehensive plans and submitted comments for St. Michael's, Leonardtown, Fruitland, Charlestown, and Somerset County. Comprehensive plan comments are available in the Objective 2 Appendix.

The rebuild of the Non-tidal Anadromous Fish Spawning Map on the ESRI Experience Builder platform was completed. This web map has been published to the existing three links on DNR webpages thereby replacing the previous web map.

Staff attended the Corsica River Implementers meeting and a meeting with Maryland Department of Environment and consultants for the town of Centreville. For the former, FEAD watershed impervious surface concerns and the effect of the outfall for the proposed wastewater treatment plant on anadromous fish movement and spawning were of interest. For the latter, the location of the proposed outlet for the plant and its potential to form a thermal barrier for Yellow Perch spawning movement during February to early March was discussed. The consultants will run a two-dimensional mixing model to see what portion of the subestuary will be impacted by increased temperatures.

Staff attended the Chesapeake and Coastal Services' (CCS) meeting to review and update the state's identified Targeted Ecological Areas (https://dnr.maryland.gov/land/documents/greenprint-lands-are-important.pdf). These areas prioritize DNR Program Open Space funds to purchase the "best of the best" ecological areas in Maryland. Criteria considered are green infrastructure (large blocks of forest and wetland hubs and the habitat pathways or corridors that connect them), water quality, aquatic life, and rare, threatened, and endangered species (terrestrial and aquatic). We reviewed the existing GIS layers for priority anadromous fish spawning watersheds and high priority Blue Infrastructure shorelines and watersheds (blue infrastructure is a spatial evaluation of coastal habitat, critical natural resources and associated human uses in the tidal waters and near-shore area of Maryland's coastal zone; https://dnr.maryland.gov/ccs/pages/bi.aspx). Additionally, CCS has interest in including existing statewide 12-digit watershed percent impervious surface estimates and impervious surface reference points produced by FEAD under F-63. A second meeting was held about inclusion of anadromous fish spawning areas into the Targeted Ecological Areas. Their inclusion seems likely as does the use of impervious surface reference points to determine high priority watersheds. There will not be complete overlap with the other Targeted Ecological Area criteria but a significant portion of priority anadromous fish spawning areas would be eligible.

We met with Chesapeake Bay Critical Areas Commission (CAC) to discuss additional protection of anadromous fish spawning streams. The CAC is charged with devising criteria to minimize adverse effects of human activities on water quality and natural habitats (https://dnr.maryland.gov/criticalarea/Pages/default.aspx). They look to foster consistent, uniform and more sensitive development activity in a ribbon of land within 1,000 feet of the tidal influence of the Bay (the "critical area") that has direct and immediate effects on the health of the Bay. In cooperation with the CAC, local critical area management programs are administered by the 61 local governments whose jurisdictions are partially or entirely within the Critical Area. The CAC has regulatory authority that can protect anadromous fish spawning streams and is interested in enhancing protection for these streams and their watersheds through existing and additional regulations. A question (CAC) and answer (FEAD) spreadsheet was circulated to meeting participants to foster a dialog on how CAC could help prioritize anadromous fish spawning areas for conservation. Impervious surface reference points developed by FEAD under F-63 will aid prioritization. Staff from FEAD will present information to CAC commissioners and local planners in upcoming meetings.

Cooperative Research and Monitoring – Jim Uphoff met with the federal director at the Cooperative Oxford Laboratory about federal research ideas that could be connected to Striped Bass recruitment. The idea primarily had to do with the impact of marsh migration and the importance of small tributaries that feed through the marshes into the spawning ground. The Nature Conservancy was interested in this as well. This proposed research had little bearing on the current recruitment shortfall but was of some long-term interest due to expected climate related sea level and salinity increases in the current spawning areas that could affect the location, extent, and upstream migration of spawning areas.

Staff participated in the Chesapeake Bay Program's Scientific and Technical Advisory Committee workshop for Striped Bass at the Smithsonian Environmental Research Center. We gave a presentation on surveys of spawning and larval nurseries under F-63. Current surveys, habitat and early life history, mortality and movement studies were evaluated for Striped Bass management. Roundtable discussions also occurred to determine priority areas of research for Striped Bass.

FEAD and Freshwater Fisheries and Hatcheries Division staff met to discuss eDNA sampling in Mattawoman Creek. The eDNA sampling was incorporated into the spring 2025 presence-absence monitoring of anadromous fish non-tidal stream spawning in Mattawoman Creek. Adding eDNA should resolve which herring species are using the stream for spawning and their spatial and temporal distribution. Currently, we cannot reasonably differentiate among herring species based on egg and larval characteristics without substantial cost in staff time. We will also evaluate the possibility of using eDNA as a rapid assessment tool to determine the presence of anadromous fish in Mattawoman Creek and elsewhere.

We aided a Chesapeake Biological Laboratory research project led by Hongsheng Bi and Ryan Woodland that tested a plankton scope for monitoring zooplankton and Striped Bass eggs and larvae. We aided egg and larvae identifications, supplied Choptank River sampling station coordinates, and provided regular updates on what we have seen in our surveys to aid their sampling.

We were contacted by the North Carolina Division of Marine Fisheries about how to identify live and dead Striped Bass eggs. North Carolina has been undergoing an extended period of poor recruitment similar to MD and are investigating early life history dynamics to understand the underlying cause or causes. We supplied them with some pictures of live and dead eggs, exchanged other information, and helped with their live and dead egg identification via the web.

Staff provided percent cover of watershed features including zoning categories, land use/land cover, and impervious surface along with Inverse Distance Weighted (IDW) adjusted values that account for feature proximity to the mainstem shoreline and tributaries for research by staff at the Cooperative Oxford Laboratory into the linkage of land use, contaminants, and liver tumors in White Perch. Watershed areas included portions of the Chickahominy River, Choptank River, Nanticoke River, Patapsco River, Patuxent River, Piscataway Creek, Potomac River, Sassafras River, Severn River, and Wicomico River (eastern shore). Albemarle River in North Carolina has been partially completed.

Presentations and Outreach –The presentation *Influences of Impervious Surfaces on Aquatic Resources* was given by Marek Topolski during the Friends of Mill Creek (chapter of American Chestnut Land Trust) meeting held at the Calvert Marine Museum in Solomons. The presentation highlighted various effects of impervious surface on fisheries production, an assessment of the Mill Creek watershed's impervious surface status, and the importance of land use planning in fisheries management. Following the presentation, a similar watershed assessment was requested by The Friends of Hunting Creek. This work was completed and used in a brochure by this organization with an acknowledgement of the assistance provided by Marek Topolski.

The presentation *Interactive Anadromous Fish Spawning Map* was given at the Towson University TUgis conference in Towson by Marek Topolski. The web map displays the documented historical extent of non-tidal anadromous fish spawning in tributaries and is used in the screening process for environmental review. The presentation highlighted the watershed impervious surface charting tool and proposed alterations for the map rebuild in the Experience Builder platform being implemented by ESRI (Web Map Builder platform is being retired).

Jim Uphoff and Shannon Moorhead participated in Science Week at the Cooperative Oxford Laboratory on September 25th. Jim Uphoff and Mark Matsche (Fish Health Program) demonstrated a Striped Bass dissection for DNR Secretary Josh Kurtz and members of the DNR communications team. Jim's portion of the demonstration focused on Striped Bass condition, showing how we assess fish for presence of body fat, an indicator of health and energy storage, and examine their gut contents to identify prey items and estimate feeding success. Jim emphasized the importance of Atlantic Menhaden in the Striped Bass diet, especially for larger fish.

Jim Uphoff gave a PowerPoint presentation to the Midshore Anglers Club, *What are we doing to understand what's going on with Striped Bass year-class success? Applying lessons of the past to understand now.* It described the work that FEAD is doing under F-63 to help understand how six years of poor year-class success have come about.

Jim Uphoff participated in a virtual meeting along with Dr. Michael Wilberg (CBL stock assessment scientist) of the Wicomico Environmental Trust (WET) on the status of Atlantic Menhaden in Chesapeake Bay. The meeting was organized by Tom Horton (Salisbury University) at the request of the WET. The WET was being pressed to support closure of the Bay reduction fishery by an outside group and wanted to make an informed decision. There were four questions that the WET wanted answered and all were predicated with the notion that Atlantic Menhaden declined severely in the Bay. This was not supported by indices and assessments currently available at that time. Both Wilberg and Uphoff gave similar answers, "There isn't a convincing indication of decline. Most evidence support that Atlantic Menhaden are at a safe level."

The proposed traffic light index (TLI) communication tool for Atlantic Menhaden and Striped Bass forage balance in Maryland's portion of Chesapeake Bay underwent a peer review in a virtual format by two outside reviewers who were familiar with Atlantic Menhaden and ecological issues. The review was favorable but there were recommendations for improvement. that were largely addressed in a report revision. We have worked with FABS communications staff to put this on the internet.

Staff attended the 30th annual conference for Maryland Water Monitoring Council. J. Uphoff presented at the conference on *Agriculture, Development, and Local Fish Habitat Conditions in Chesapeake Bay*.

Interjurisdictional Management – Marek Topolski reviewed the final draft of the ASMFC Habitat Committee's Habitat Management Series document titled *Anthropogenic Noise Impacts on Atlantic Fish and Fisheries: Implications for Managers and Long-Term Productivity.* The document provides an overview of the natural aquatic soundscape and its importance to fishes, sources of anthropogenic noise in the oceans, impacts of anthropogenic noise on fishes, and mitigation techniques.

Chesapeake Bay Program - Staff participated in the fish habitat and forage action teams meetings. The meetings provided an opportunity to submit feedback and comments on the 2025 revision of the Chesapeake Bay Program (i.e., Beyond 2025).

Staff attended the joint Forage and Fish Habitat Meeting and subsequent follow-up meetings to discuss the forage fish and fish habitat outcomes in the Beyond 2025 Plan. We offered an outline of a monitoring plan that was received well. Since the interest for the TMDL is nutrients, sediment, and clarity, the mesohaline subestuaries in summer would be where and when to work. These areas when stressed by watershed development have extensive areas of low DO regardless of depth. Less developed areas don't typically exhibit low DO in the upper reaches. The upper reaches would be a good place to look for changes in DO and site occupation by focal species. If there isn't much DO, there's not much site occupation. White Perch would be a good focal species since they are well sampled by standard techniques and spend much of their lives in the Bay. The NAJFM paper we published in 2011 indicated they responded to impervious surface which was the driver for DO.

Staff attended the spring 2025 Fisheries Goal Implementation Team meeting. The primary task was to finalize the fish habitat outcome draft language and align/add outputs (actions) to the revised outcome. Time was insufficient to complete a review of the indicators (metrics) for the outputs.

State Wildlife Action Plan, 2025 Update - Staff attended a meeting with other DNR staff to discuss the list of estuarine and marine species in the State Wildlife Action Plan. They reviewed the key habitat descriptions from the 2015 SWAP update and provided recommendations to update the Pelagic - Open Water habitat description to Chesapeake Bay Estuarine Connecting Waters habitat.

Training – Marek Topolski completed the ESRI GIS Massive Open Online Course (MOOC) "Spatial Data Science: The New Frontier in Analytics".

Staff completed CPR and First Aid training.

Appendix for Objective 2

Fisheries Ecosystem Assessment Division Comments on County and Municipal Comprehensive Plans, July 1, 2024 to June 30, 2025

Calvert County/Prince Frederick, submitted 7/26/2024 - There are three anadromous fish spawning streams that could be impacted: Parker Creek (drains into the Bay), Hunting Creek and Battle Creek (both of which drain into the Patuxent River). The non-tidal anadromous fish spawning map estimates impervious surface for these watersheds. Parker and Hunting Creeks are at 6% and Battle Creek is at 5%. This is a low level of development for a southern Maryland watershed and at or very close to what we consider a target level indicating good habitat. Parker Creek has White Perch spawning; Hunting Creek has Yellow Perch, White Perch, and Herring spawning; and Battle Creek has Herring and White Perch spawning. Conservation of these watersheds should be the priority.

There is a water quality station adjacent to Prince Frederick just below Benedict on the Patuxent River (Eyes on the Bay RET 1.1). Bottom dissolved oxygen is poor there in the summer. Development of the Patuxent River watershed above there is likely the main source, but Prince Frederick contributes to this.

The County adopted an updated Patuxent River Policy Plan in 2014. The original plan started in 1984 (impervious surface was around 7%) to deal with point and nonpoint nutrient and sediment pollution associated with development of the watershed of its largest native river. Impervious surface is now around 14% and the river has a variety of major fish and shellfish habitat issues. The policy plan by itself hasn't effectively dealt with these issues.

Culture Resources and Sustainable Fisheries – Commercial and recreational fishing are part of the heritage, but not much is mentioned about them.

The plan acknowledges urban management practices that impact stream biology and that recognition is good.

Stream restoration projects should be resisted and focused on restoration of heavily developed urban streams.

Cluster Development (Chapter 2, Vision 3) – The idea of having all the development in one area is good, but the concern would be pulses of water during high rain events. Proper stormwater management practices should be used, and this might be better than spread out development. The ideas for stormwater management practices seem adequate.

Prioritizing Trail Access to Open Spaces (Chapter 6) - The idea of providing more trail access to open spaces is great. An issue with a lot of trails is improper construction or lack of maintenance. Trails aren't a significant contributor to sediment and nutrients entering a waterway, but a trail through a steep, sandy/loamy area will lead to erosion and sediment entering the body of water if not built properly. The recommendation would be to use the best trail building practices and keep up with maintenance to limit erosion.

Wastewater Treatment (Chapter 10) - In Table 9-4, it looks like the planned capacity and 2040 demand will be close. The expected EDUs by 2040 are 4,120 compared to 3,062 in 2014, an increase of 1,058 EDUs or 35%. This seems rather high over a 26-year period. Hopefully this is mostly related to adding failing septic systems to the public wastewater treatment systems.

The plan will not permit public sewer service in rural and agricultural areas – more septic systems will be added with rural growth which will impact aquatic resources.

Toxic contaminants are mentioned in the plan numerous times, but we are unsure of the definition. Is this primarily road salts/contaminants or something else?

Centreville Land Use Map Amendments, submitted 10/9/2024 - There are 1,032 housing units available under the water supply limitations. We project that if all these units are built out, impervious surface coverage for the whole watershed will be about 5%. Growth outside of town (including areas to be annexed into town and Queen Anne's County) should be limited as much as possible to keep the watershed near 5% impervious surface. This is considered a safe target for fish habitat. Ten percent impervious coverage should not be considered as a growth target if aquatic resources are a concern; this is the tipping point for increasingly intractable aquatic habitat degradation that should be avoided.

There will be a greater impact on the streams in the annexed area than in the whole watershed. It appears a large portion of additional housing units proposed by Centreville would be located on parcel 0060 (Growth Area 4), which is adjacent to the existing municipal boundary. Mill Stream Branch is adjacent to the parcel boundary, so intense stormwater management and stream buffers will be critically important. Anadromous fish spawning (herring species and Yellow Perch) has been documented in Mill Stream Branch.

The use of cluster development to limit environmental impacts is good, but the concern would be pulses of water during high rain events. The use of proper stormwater management practices would be necessary. The plan to preserve open space around the development would be important.

St. Michael's, submitted 1/3/2025 - Overall, the plan does a good job of limiting development to in town locations. There is some concern of future development in outlying areas that are currently agriculture land.

Additional housing units – not likely to be more than 100. There may be some impact on fish habitat with the additional impervious surfaces, but not likely to be significant since all units would be within town limits.

Current wastewater system can handle any planned additional development in town.

Strausburg Farm Property – included in the agricultural and conservation estimate for town but is approved for a 10 lot subdivision. This area would also require well and septic systems which could impact fish habitat. Efforts should be made to conserve this property to protect fish habitat.

San Domingo Creek Park – This project looks like a great opportunity to reduce the amount of impervious surface both in town and along the creek (removal of multiple buildings, roads, and parking lots).

Trail Access to Open Spaces - The idea of providing more trail access to open spaces is great. An issue with a lot of trails is improper construction or lack of maintenance. Trails aren't a significant contributor to sediment and nutrients entering a waterway, but a trail not built properly can contribute to erosion and sediment entering a body of water. The recommendation would be to use the best trail building practices and keep up with maintenance to limit erosion.

Living shoreline - replacement of a bulkhead with a living shoreline is adequate in addressing some fish habitat issues. This would be better than an established bulkhead for the shoreline.

Leonardtown, submitted 2/3/2025 - Waterfront Development. The plans to develop a hotel and conference center on the Tudor Hall Farm property and residential development on the same property will have an impact on the amount of impervious surface (IS) in the watershed. In order to maintain healthy fish communities, the goal for the watershed should be a target IS of 5% and structures per acre (C/ac) of 0.13 C/ac for development. The Breton Bay watershed is at 6.4% IS and 0.18 C/ac which is above the safe target and below the threshold (IS of 10% and 0.34 C/ac) for increasingly intractable aquatic habitat issues. Development levels between target and threshold may show a negative response in the fish community. Once the development exceeds the threshold, significant negative impacts on the fish communities are shown to occur.

The additional proposed development of this area is projected to increase IS to 8.2% and 0.26 C/ac by 2037. The recommendation would be to preserve as much land as possible to minimize the increase in IS.

The current impervious surface for the town limits of Leonardtown is 21.9% and structures per acre is 0.42. The addition of the proposed development by 2037 is projected to increase the IS to 27.8% and structures per acre to 0.59. The elevated IS can lead to negative impacts on the fish communities of the upper part of the Breton Bay subestuary. Bottom DO threshold of 3.0 mg/L is the minimum value for aquatic organisms, the target is 5.0 mg/L. Division sampling in the summer from 2003 to 2005 indicated bottom dissolved oxygen for the site near Leonardtown (Site 1) was below 3.0 mg/L for 22% of the samples. Site 2 (next downstream site) was below 3.0 mg/L for 42% of the samples. The two sites that were further down the subestuary (Sites 3 and 4) were below 3.0 mg/L for 5% and 0%, respectively, of the samples. The elevated impervious surfaces from the town appear to have been impacting bottom DO values from 2003-2005 and increased development will not improve the situation.

Development of the Tudor Hall Farm property may have impacts on sensitive species located in the adjacent stream and wetlands.

McIntosh Run that flows into Breton Bay is an important spawning location for Yellow Perch. Additional development could impact their spawning success.

Waste water treatment plant expansion in 2025 will increase capacity and will be able to accommodate expanded build-out growth. This will help prevent the expansion of the use of septic systems which will help with some of the issues with fish habitat. However, it also leads to increased development that will put further stress on stormwater issues because the growth can be accommodated. There should be careful though given to the trade-off in growth and ecological health of Breton Bay.

Additional development beyond the planned areas will be required to pay for upgrades to the wastewater treatment plant. Coordination with the county should be implemented to control growth in the watershed outside of town limits.

Municipal growth will use a smart growth strategy to concentrate development adjacent to existing developed areas. This will help keep important forest and agricultural lands intact. The use of cluster development to limit environmental impacts is good, but the concern would be pulses of water during high rain events. The use of proper stormwater management practices would be necessary. The plan to preserve open space around the development would be important.

Charlestown, submitted 4/28/2025 - *Fisheries in the Northeast River.* There are several recreational fisheries on the Northeast River that are important. The Northeast River is a premier destination for Largemouth Bass in Maryland. It also has very good Yellow Perch fishing in the spring and Striped Bass fishing on the Susquehanna Flats during the catch and release season. Important commercial fisheries include tidal fresh species such as Catfish, Yellow Perch, and Gizzard Shad.

Two streams that run through town were found to be anadromous fish spawning streams. Conductivity readings from the Chesapeake Monitoring Cooperative (https://cmc.vims.edu/dataexplorer#/home) indicated elevated conductivity at Red Rum Creek, an indication of urban impact, and more normal readings on Peddler's Run.

The Fisheries Ecosystem Assessment Division previously sampled the Northeast River for Yellow Perch larvae in the spring and found a high presence in the river. Summer sampling has also found a diverse assemblage of fish species in the river. However, there have been incidences of low bottom channel DO in the river.

Although a Fisheries component is not required in the town plan, consideration should be given for these important resources in the Northeast River.

On Page 17, Goal #1 indicates that there will be "meaningful impact" of extending sewer service to two unincorporated communities they want to bring into town boundaries. What does that mean? There's the potential for expanding sewer service inducing expanded growth.

Goal #2 – use of cluster development is a good idea to limit clear cutting of forest and reducing the impacts of runoff from impervious surfaces if sufficient stormwater features are created.

The plan doesn't suggest a large increase in impervious surface, but it is unclear of any potential impacts in the plan. The Northeast River watershed (7.14% impervious surface (IS) and 0.22 structures per acre (C/ac) has surpassed the development target (5% IS, 0.13 C/ac), but hasn't breached the threshold (10% IS, 0.34 C/ac).

The improvement of waterfront connectivity with trails and bridges is great. It would also be a great opportunity to provide shore fishing effort as part of this effort.

Fruitland, submitted 4/18/2025 - Additional development in the Wicomico drainage is a concern. The watershed is currently at 8.8% impervious surface (IS) and 0.28 structures per acre (C/ac). The target development for a watershed is 5% IS and 0.13 C/ac. Once development reaches a threshold of 10% IS and 0.34 C/ac, there are increasingly intractable aquatic habitat issues. The goal would be to conserve as much land as possible to stay under the threshold of development.

FEAD staff sampled the Wicomico River most recently in 2018. Striped Bass eggs, Yellow Perch larvae, White Perch larvae, and Herring larvae were all detected at the site adjacent to Fruitland.

Analysis of Yellow Perch larvae between the Wicomico River and Choptank River (rural control) indicated that primary productivity may be lower in the Wicomico River and depicting an impact from development.

Runoff from town drains into the nearby spawning and nursery areas, so stormwater runoff is a concern. Strong stormwater management should be implemented.

Allowing mining in the critical area should be reconsidered. In the critical area, this would be very near the anadromous fish spawning and larval fish nursery. Water quality concerns could impact the productivity of these areas. Additional considerations of other mining activities should be given to protect and maintain important fish habitat in the Wicomico River.

Additional protection of greater than a 25ft buffer along non-tidal streams should be given to protect fish habitat.

Annexing areas with failing septic systems is a good. This will help protect fish habitat from high nutrient runoff.

Infilling areas with development already present is a good idea. It will reduce the impact of increasing impervious surface in less developed areas.

Somerset County, submitted 5/9/2025 - Overall, the plan looks good and promotes the continued rural nature of the county.

Infill to already developed areas should minimize increase in impervious surfaces

Multiple towns are proposing extending sewer service to replace failing septic systems. This will help reduce nonpoint source discharge but could increase development.

The plan notes the desire to update ordinances that determine when work can be done in streams/ditches. The ordinance should be updated to include public and privately owned streams/ditches and minimize work during the fish spawning season (March to May) but allow for emergency repairs if needed.

The plan uses 10% impervious surface (IS) for declines in water quality and 25% IS for sharp declines. The target IS should be 5% to minimize the impacts from development. The threshold IS should be 10% at which point negative impacts on fish habitat are more likely.

Living shorelines should help protect shorelines from erosion and will also provide some shallow water habitat for fish.

Project 1: Marine and estuarine finfish ecological and habitat investigations
Objective 3: Develop spatial data to assist in conserving priority fish habitat.

Marek Topolski

Introduction

The Chesapeake Bay Program (CBP) has identified shoreline hardening, or armoring, as a driver of shallow water fish habitat distinct from watershed development. Shoreline armoring alters physical habitat and influences species composition in littoral waters (Chhor et al. 2020) in disparate ways through shoreline composition change including reductions in availability of shallow water and wetland habitats (Munsch et al. 2015; Kornis et al. 2017, 2018). Fish and crustacean assemblages in Chesapeake Bay within 3 m of the shoreline are structured according to the shoreline present with small bodied demersal species common to the littoral zone showing a negative response to hardened shoreline (Kornis et al. 2017); whereas, planktivore and larger bodied benthivore/piscivore individuals are commonly observed along hardened shorelines, bulkheads in particular, where water tends to be deeper (Kornis et al. 2018). Presence of larger bodied species along armored shorelines may result from a preference for deeper water rather than shoreline armoring (Toft et al. 2007). In the Salish Sea of Puget Sound, abundance of transient species (salmon and herring) were not associated with the percent of shoreline armoring at a local scale – radius of 100-500 m although a weak relationship was detected at larger spatial scales that could not be attributed to the presence of shoreline armor (Bishop et al. 2024). Causal relationship between shoreline armoring and abundance of migratory fishes is dubious since highly mobile anadromous species were unable to avoid travel along shorelines with variable amounts of armoring (Bishop et al. 2024).

Percent hardened shoreline is a correlate and consequence of land development (Gittman et al. 2015; Kornis et al. 2017). Analysis of Maryland's Critical Area shoreline (within 1,000 feet of tidal water) detected a high probability (0.62) of shoreline structure being present when land development in both the adjacent Critical Area and the whole catchment had 5-10% impervious surface (%IS, Uphoff et al. 2024) and the probability increased nonlinearly as percent impervious surface increased. Using housing density as a proxy of land development, Gittman et al. (2015) identified housing density as a significant predictor of shoreline armoring in the United States. Structure density (housing, commercial, and institutional) can be used to estimate a watershed's percent impervious surface (Topolski 2015; Uphoff et al. 2022), a measure of land development that is negatively linked to bottom dissolved oxygen (DO_b) and fish distribution in brackish waters (Uphoff et al. 2011a). Dissolved oxygen is a key habitat parameter for fishes (Breitburg 1990; Craig and Crowder 2005; Tyler and Targett 2007; Zhang et al. 2009; Buchheister et al. 2013), which during summer months can frequently drop to hypoxic levels. Hypoxia (<2.0 mg/L) is particularly prevalent in summer stratified mesohaline subestuaries in Chesapeake Bay (Hagy et al. 2004; Kemp et al. 2005; Batiuk et al. 2009). Low DO conditions in mainstem Chesapeake Bay are largely driven by agriculture derived nutrient enrichment, although urban areas are also sources of high nutrient enrichment (Kemp et al. 2005; National Research Council 2009; Brush 2009). Kornis et al. (2017) detected a significant negative DO_b response to the proportion of cropland and a non-significant ($P = 0.18$) negative relationship to developed land although in mesohaline Chesapeake Bay subestuaries a strong negative correlation existed between mean DO_b and both %IS and percent urban land cover (Uphoff et al. 2011a, 2024).

Balouskus and Targett (2018) observed that shoreline armoring was associated with lower fish abundance when DO was suboptimal (≤4.8 mg/L).

Armored shorelines alter the physical configuration of the littoral zone which has direct effects on the fish community composition. What is less clear is if shoreline armoring is associated with alteration of non-structural habitat required by fish? This analysis assessed if armored shoreline was a factor associated with suboptimal levels of DO for living resources.

Methods

Bottom DO data were assembled from F-63 estuarine fish summer habitat and community surveys of Maryland subestuaries from 2003-2023 (See Objective 1, Section 3 for a description of field methods; Figure 1). Surveys began between the last week of June and second week of July and concluded the final week of September, although in some years data collection extended into the first two weeks of October to ensure consistent time series sample size. Data were imported into ArcGIS Pro v3.x for all geoprocessing. A DO_b point feature class was created using latitude and longitude coordinates provided except when the coordinates were incorrect or missing. In these instances, default median station coordinates were created from available F-63 surveys using the Median Center geoprocessing tool. Summer measurements of DO_b are of particular interest since there was a greater likelihood of below target (5 mg/L) and threshold (3 mg/L) concentrations (Uphoff et al. 2011b). Bottom DO data were collected during the months June-October and compared using monthly DO_b boxplots to determine if a seasonal signal was present prior to calculating the mean summer DO_b. Of particular interest was if October data should be included as a summer measurement. Subestuaries were designated as mesohaline (5-18 ‰), oligohaline (0.5-5 ‰), or tidal-fresh [limnetic] (<0.5 ‰) (Figure 1) in accordance with prior watershed salinity classifications (Uphoff et al. 2024) based on the Venice System ("The Venice System for the classification of marine waters according to salinity" 1958). The Venice System categories are not incremental since they were developed in accordance with general biological zonation.

Shoreline structure for Maryland's Chesapeake Bay was surveyed twice, first during 2003-2006 and again from 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/). Each survey assessed the type of shoreline structure if present (SSTRU shapefile) and the land use and bank condition (LUBC shapefile) for each county and Baltimore City. Using ArcGIS Pro 3.x, the Intersect geoprocessing tool was used to combine the SSTRU and LUBC shapefiles for each jurisdiction and then combined into a single shoreline feature class for each set of survey years herein CCRM2003 and CCRM2020. Shoreline segments having structure were coded as present except when the shoreline structure was categorized as "debris" ("… haphazardly scattered and not providing shoreline protection"). Structures adjacent to (breakwater and marsh toe) or extending from (groin and jetty) the shoreline were not included. Shoreline segments were dissolved into structure and non-structure polyline features, which were then exploded into their individual composite segments.

The 12digit subwatershed was chosen as the spatial scale for the study since it represented the smallest watershed area delineated for tributaries by Maryland's Department of Environment and Department of Natural Resources and is referenced by various state regulations, monitoring and assessment programs, and restoration actions. The DO_b feature class was spatially joined with a DNR 12digit subwatershed feature class to link each DO_b observation with its corresponding subwatershed. These data were then assigned the annual estimate of subwatershed %IS for the year DO_b was measured (see General Spatial and Analytical Methods used in Objective 1 Sections 1-3 for %IS calculation). This feature class was saved as two separate feature classes categorized into the timeframes, 2003-2012 and 2013-2023, to correspond with the two surveys of shoreline structure. The trawl stations where DO_b was measured were in or near the subestuary channel rather than adjacent to the shoreline. Distance to shoreline (D_{shore}) was calculated using the Near geoprocessing tool and the percent of shoreline as structure (%L_{str}) adjacent to each station was calculated using the Summarize Nearby geoprocessing tool within a series of five distance buffers from each station: 75 m, 250 m, 500 m, 1,000 m, and 1,500 m. The mean %L_{str} (excluding buffers that did not intersect the shoreline) was used for the analyses. Monthly values of DO_b, %L_{str}, D_{shore}, and %IS were averaged to produce annual summer estimates by subestuary.

Uphoff et al. (2011b) reported that the 5 mg/L target and 3 mg/L threshold DO_b criteria could be used to assess watershed development induced habitat stress during summer months in mesohaline subestuaries. These criteria are less effective for identifying habitat stress in oligohaline and tidal-fresh subestuaries since they are less prone to water density stratification and depleted DO_b. Across a range of %IS levels, water depth (1.5-6.1 m) accounted for little variation in DO_b concentration (Uphoff et al. 2011a) and was not included as an explanatory variable in this analysis. Bottom DO was assessed with analysis of variance to determine if it was independent of salinity category. Multiple linear regression was used to determine if a significant relationship existed between the DO_b concentration and %L_{str}, D_{shore}, %IS, and a %L_{str}:D_{shore} interaction for each salinity classification. Percent L_{str} will typically increase with an increase in %IS within the Chesapeake Bay critical area (≤1000 ft from tidal water; Uphoff et al. 2024). However, %IS in this study was for the larger, adjacent 12digit subwatershed and any effect on DO_b was considered mechanistically independent of %L_{str} (index of many landscape perturbations versus discrete segments of shoreline alteration, respectively). A significant interaction was assessed using added-variable plots, which depicted each multiple regression term's partial residuals while controlling for the other terms. The benefit of added-variable plots is that a fitted linear regression to these plots has the same slope as the independent variable in the full regression model (Gallup 2019). A significant %L_{str}:D_{shore} interaction was further scrutinized with a Johnson-Neyman test using D_{shore} as the moderator. The Johnson-Neyman test estimates moderator values, where the predictor, %L_{str}, has a significant conditional slope and therefore an effect on the dependent variable DO_b. Significance tests were evaluated at an α = 0.05. All statistical analyses were done using R v4.4.2 and RStudio v2024.12.0.

Results

A total of 3,147 DO_b records were assembled for the months June through October, 2003-2023 and mapped to verify coordinates (Figure 1). Forty-eight percent (1,526) of the records had site-specific coordinates. Records having either no coordinates (1,520) or incorrect coordinates were assigned the median center coordinates for the respective sample station which was derived using the Median Center geoprocessing tool on records with accurate site-specific coordinates and a database of sample station coordinates (885). The spatial reference of 55 samples among four subestuaries could not be reconciled; however, they were clustered such that the cluster could be confidently centered on the respective median station coordinate. Median coordinates could not be assigned to 13 records. In all 3,134 DO_b records were georeferenced and included in the analysis.

Dissolved oxygen records spanned June through October, although not every station in each year was sampled in each month. Visual inspection of these data indicated that DO_b was inversely related to salinity category and increased over summer months except October when DO_b was higher in oligohaline systems than in tidal-fresh systems (Figure 2). Mesohaline mean DO_b was at or below the target of 5 mg/L in all months except October although threshold 3 mg/L DO_b violations did occur within the first two weeks of October in some years. Mean DO_b remained above the 5 mg/L target in oligohaline and tidal-fresh systems. Bottom DO below 3 mg/L did occur in all salinity regimes. Mean DO_b concentrations (Table 1) were ≥6.5 mg/L from June through October in tidal-fresh subestuaries. Oligohaline subestuaries had gradually increasing mean DO_b concentrations from June (5.21 mg/L) through September (6.85 mg/L) after which it increased to 8.59 mg/L within the first week of October. Monthly violations of target DO_b and threshold DO_b (Table 1) were uncommon in tidal-fresh subestuaries: 0-9% and 0-1% of measurements respectively. Oligohaline subestuaries had an increased occurrence of target DO_b violations (0-31%) although threshold DO_b violations remained uncommon (0-6%). Bottom DO in mesohaline subestuaries exhibited chronic violations of the 5 mg/L target concentration (11-57%) and to a lesser extent violation of the 3 mg/L threshold concentration (5-27%). Excluding October, mesohaline target and threshold violations were no less than 39% and 17% respectively of measurements. October DO_b data represented an upward summer to autumn seasonal shift in DO_b concentrations and was excluded from the remainder of analyses leaving a total of 3,075 DO_b observations.

Twenty-five subestuaries (16 mesohaline, five oligohaline, and four tidal-fresh) comprised the summer DO_b data which was combined with estimates of $\%L_{str}$, D_{Shore}, and %IS (Table 2). Sampling effort was not equally distributed among subestuaries. The number of mesohaline records (N = 1,820) was more than oligohaline (N = 550) and tidal-fresh systems (N = 705) combined (Table 1). Each subestuary had four sampling stations except for Bush River, Nanjemoy Creek (seven of 10 of years), and Piscataway Creek which had three stations and two stations each in Rhode and West Rivers. The range of DO_b was comparable among the subestuary categories (Table 1, Table 2): mesohaline = 0-14.3 mg/L, oligohaline = 1.4-10.9 mg/L, and tidal-fresh = 1.27-14 mg/L. Mean DO_b concentration was greater as subestuary salinity category went from mesohaline to tidal-fresh (Table 1, Table 2): 4.56 mg/L in mesohaline systems, 6.29 mg/L in oligohaline systems, and tidal-fresh systems were at 7.05 mg/L. Fifty percent of DO_b measurements in mesohaline subestuaries were <5 mg/L, compared to 17% in oligohaline and 7% in tidal-fresh subestuaries. Twenty-one percent of DO_b measurements were below the 3 mg/L threshold in mesohaline subestuaries compared to 1% in both oligohaline and tidal-fresh subestuaries. Of the mesohaline subestuaries, 63% had mean DO_b below the 5 mg/L DO target and 19% were below the 3 mg/L threshold (Table 2). In comparison, neither oligohaline nor tidal-fresh subestuaries had a mean DO_b below the 5 mg/L target or 3 mg/L threshold.

Estimates of $\%L_{str}$ ranged from 0-100% for each subestuary category and averaged 43%, 45%, and 35% in mesohaline, oligohaline, and tidal-fresh subestuaries respectively (Table 2). While the range of D_{shore} was greater as salinity category increased (mesohaline = 3-1,169 m, oligohaline = 6-344 m, and tidal-fresh = 61-307 m), mean D_{shore} was greater in tidal-fresh subestuaries: mesohaline = 244 m, oligohaline = 243 m, and tidal-fresh = 315 m (Table 2). Five of 16 mesohaline subestuaries had at least one 12digit subwatershed >10% IS (Table 2) although the sample size weighted (N-weighted) mean was 6% among all 12digit subwatersheds. A greater proportion (three of five) of oligohaline subestuaries examined had one or more 12digit subwatersheds >10% IS and the N-weighted mean of 15% IS. One tidal-fresh subestuary contained at least one 12digit subwatershed >10% IS (Table 2) while the N-weighted mean subestuary IS was 8%.

Bottom DO concentration measurements were taken for one to 18 summers depending on subestuary and were averaged by year for each subestuary (Table 3). Mesohaline mean summer DO_b was <5 mg/L during 56% of summers and <3 mg/L during 11% of summers. The three mesohaline subestuaries that averaged summer DO_b <3 mg/L threshold were urban (≥15% IS) and the six that averaged summer DO_b ≥5 mg/L target were rural (≤ 5%IS) (Table 3). Three percent of mean summer oligohaline DO_b were <5 mg/L and all values were >3 mg/L. No tidal-fresh mean summer DO_b values were <5 mg/L. Mean subestuary $\%L_{str}$ was generally greater where there was higher %IS. The sample years weighted (N_{yr}-weighted) mean summer $\%L_{str}$ was 42% for mesohaline subestuaries which was similar to the N-weighted mean although N_{yr}-weighted mean summer values decreased for oligohaline (40%) and tidal-fresh (29%) subestuaries. Mean summer D_{shore} range was 96-812 m for mesohaline, 153-482 m for oligohaline, and 191-497 m for tidal-fresh subestuaries which were sufficiently far from shorelines that DO_b measurements should not be considered adjacent (Table 3). Subwatersheds having >10% mean annual %IS were observed in four of 16 mesohaline, two of five oligohaline, and one of four tidal-fresh subestuaries (Table 3). Overall, N_{yr}-weighted average %IS was 6% in mesohaline, 12% in oligohaline, and 9% in tidal-fresh.

Mean summer DO_b was significantly lower as subestuary salinity category increased ($P < 0.0001$, Table 4, Figure 3A); suboptimal DO_b and violation of the three mg/L DO threshold were characteristic of mesohaline subestuaries (Figure 3A). Mean $\%L_{str}$ was highly variable within subestuaries but was not significantly different among the salinity categories (Table 4, Figure 3B). Distance to shoreline within the subestuaries generally increased the further downstream the station was located. Mean D_{shore} did not differ significantly among the subestuary salinity categories (Table 4, Figure 3C). Mean %IS was lower within mesohaline subwatersheds compared to oligohaline ($P_{adj} < 0.0001$) and tidal-fresh ($P_{adj} = 0.0626$) subwatersheds (Table 4, Figure 3D). High %IS estimates, well above the third quartile, occurred in mesohaline and oligohaline subestuaries; deviation in the tidal-fresh subwatersheds was due to a single watershed having low %IS (Figure 3D).

The multiple regression models indicated that DO_b had different relationships with $\%L_{str}$, D_{shore}, %IS, and the $\%L_{str}$:D_{shore} interaction among the salinity categories. In mesohaline subestuaries, all variables and the interaction term were significantly related to DO_b accounting for modest variation in DO_b ($P = < 0.0001$, adj $R^2 = 0.487$, Table 5). While both $\%L_{str}$ ($P = 0.0003$) and D_{shore} ($P = 0.0245$) were positively related to DO_b, their interaction was negatively related to DO_b ($P = 0.0147$) as was %IS ($P < 0.0001$, Table 5, Figure 4). Mesohaline DO_b had a strong negative relationship and narrow 95% confidence intervals with the adjacent subwatershed's %IS (Figure 4) while the negative slope of the $\%L_{str}$:D_{shore} interaction was influenced by extreme partial residual values at each end of the trend line (Figure 4). None of the explanatory terms ($\%L_{str}$, D_{shore}, %IS, and $\%L_{str}$:D_{shore} interaction) were significant predictors of DO_b in the oligohaline ($P \geq 0.363$, adj $R^2 = 0.0121$) and tidal-fresh ($P \geq 0.0766$, adj $R^2 = 0.119$) multiple regression models (Table 5).

Percent L_{str} was not a significant factor in DO_b concentration at all distances from the shoreline. The $\%L_{str}$ conditional slope was significant up to 266 m from the shoreline (Table 6) and had a negative effect on DO_b (Figure 5). Simple slopes analysis and Johnson-Neyman plots confirmed that for oligohaline and tidal-fresh subestuaries there were no significant D_{shore} ranges where $\%L_{str}$ influenced DO_b (Table 6).

Discussion

Shoreline armoring was not a sole explanatory variable for DO_b in this analysis. A positive association between $\%L_{str}$ and DO_b was detected when %IS was included in the analysis. The $\%L_{str}$:D_{shore} interaction term, while negative and significant, explained little variation in DO_b without inclusion of %IS as a main effect. Percent L_{str} was negatively associated with DO_b only when incorporated as an interaction effect suggesting that it was not a sole driver of suboptimal DO_b. The analysis assumed $\%L_{str}$ was static during the ~18 years between shoreline surveys which could confound detection of an effect on DO_b; however, during this same time period Kornis et al. (2018) did not detect a significant summer DO effect related to shoreline type (wetland, beach, bulkhead, and riprap) in the mid- to upper Chesapeake Bay. Analysis of the $\%L_{str}$:D_{shore} interaction revealed that remediating shoreline armoring should not be relied upon to ameliorate low DO_b concentrations, particularly in tributary mainstem waters. However, remediation of armored shorelines in mesohaline subestuaries may incur some nearshore (≤ 266 m from shoreline) DO_b improvement with the effect (simple slope) increasing closer to shore. Nearshore mesohaline DO_b measurements were typically (92%) taken at ≤6 m in water depth; approximately half (48%) of DO_b data were from ≤3 m depth. Water depth did not correlate with DO_b nor D_{shore} (M. Topolski, personal observation), but DO_b in shallow nearshore waters would respond to wind driven mixing and stratification (Scully 2013; Wang et al. 2025) that could obfuscate the effect of armored shorelines on DO_b concentration. The importance of %IS inclusion for the model to explain variation of DO_b indicated that without addressing watershed impacts the remediation of shoreline armoring will provide little improvement of DO_b.

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 (Uphoff et al. 2024). Increased development, measured as %IS, was 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 (rural watershed) was a transitional point where shoreline structure use became substantially more common and occupied longer segments of the shoreline. Variability in shoreline composition (probability of occurrence and percent of length) increased when %IS was roughly 40-50% (city watershed). Shoreline fragmentation increased and became less predictable as land was developed, but in general shoreline stabilization was a symptom of land development (Uphoff et al. 2024). Vulnerability to physical damage was an important predictor of shoreline hardening, although housing density and gross domestic product were better predictors in sheltered (non-ocean facing) areas such as Chesapeake Bay (Gittman et al. 2015).

Bottom DO differed among salinity categories but was considerably lower in mesohaline subestuaries for each month. October DO_b data was elevated compared to the months June-September with few violations of the 3 mg/L threshold. Classification of DO_b records as summer should avoid inclusion of October data when possible. Studies that aggregate DO_b data annually by season, such as this one, would underestimate summer DO_b threshold violations and the influence of land use on fish habitat.

Among the salinity categories in this study, mesohaline systems were vulnerable to water column stratification and deep water hypoxia (< 2 mg/L) during the summer (Kemp et al. 2005). Hypoxia is not limited to the Chesapeake Bay mainstem, but rather extends into bottom waters of smaller subestuaries (Lake et al. 2013). Unlike oligohaline and tidal-fresh subestuaries, threshold DO_b violations were anticipated and frequently observed among the mesohaline subestuaries. Significantly lower annual summer DO_b in mesohaline subestuaries suggests that this type of subestuary should be targeted by efforts aimed at ameliorating low DO_b in tidal waters.

Bottom DO measurements adjacent to the shoreline (<50 m) would clarify if shoreline armoring was a factor affecting DO_b independent of watershed development. A prior analysis used the mean of surface and bottom DO concentration and shoreline disposition, and it was unclear if suboptimal DO_b existed (Kornis et al. 2018). Unlike Kornis et al. (2018) who sampled ~16 m from shore, the current analysis had 0.4% of DO_b measurements within 16 m. DO_b measurements along the 1.25 m depth contour in close proximity (where possible ≤ 30.5 m, length of a Fisheries Ecosystem and Assessment Division seine net) to the shoreline were collected during the 2025 F-63 summer sampling. Identification of factors that affect the nearshore DO profile are relevant to refinement of fishery management tools such as habitat suitability index (HSI) models. For example, development of an age-0 striped bass HSI for Chesapeake Bay that relies on surface DO in shallow (< 2 m) nearshore (within 30.5 m) waters (Dixon et al. 2024) may need to factor in DO_b and proximity to armored shorelines.

The Comprehensive Evaluation of System Response (CESR) identified gaps and uncertainties in the Chesapeake Bay restoration efforts that impacted efforts designed to attain water quality standards and responses of living resources (Scientific and Technical Advisory Committee [STAC] 2023). Shallow water habitat was considered low hanging fruit for restoration that benefits living resources. Improvements in DO in shallow water habitats that support both nursery habitat and forage fish may generate larger living resource responses than similar levels of water quality improvement in deeper water habitats. The State of Maryland's Whole Watershed Act, based on CESR, has also targeted shallow water improvements for living resources (Whole Watershed Act 2024). The CBP has recommended that living shorelines and total maximum daily load (TMDL) requirements should be considered when identifying restoration actions that benefit living resources in shallow water (STAC 2023).

Shallow water depth has not been defined by the CBP or for the Whole Watershed Act even though depth will define the amount of habitat affected and relative importance of development, TMDLs, and living shorelines in mesohaline subestuaries. Uphoff et al. (2011a) did not detect a negative effect of %IS on the presence-absence of four Chesapeake Bay target species (White Perch juveniles or adults, Striped Bass juveniles, Spot juveniles, and Blue Crabs) in shore zone (seine) samples. If shallow water depth is 1-2 m, then nearshore areas along the shoreline are more likely to predominate as a percentage of habitat and living shorelines would potentially have more impact than TMDLs. There was a strong negative effect on presence in bottom channel habitat (trawl samples at depths from 1.5-6.0 m) that reflected a negative influence of development on DO_b (Uphoff et al. 2011a). If depth for shallow water is increased, DO_b is likely to be a bigger factor and nutrient TMDLs will be more important and living shorelines will be less so.

Response of DO_b to TMDLs may be difficult to detect or interpret in tidal-fresh and oligohaline subestuaries. Extent of bottom channel habitat that can be occupied by living resources does not diminish due to low DO_b with increasing watershed development in tidal-fresh and oligohaline subestuaries (Uphoff et al. 2024). More localized or episodic habitat issues such as low DO within dense submerged aquatic vegetation (SAV) beds, ammonia toxicity, and harmful algal blooms may be important (Uphoff et al. 2024). These events may not be detected through routine monitoring. Dense SAV, prevalent in some tidal-fresh subestuaries, prevents seine sampling of fish communities and techniques portable enough for routine monitoring do not appear to be available. Ammonia chemistry and toxicity is complex (Uphoff et al. 2017), making routine monitoring challenging. Harmful algal blooms are not necessarily toxic to fish, so presence is not indicative of fatal habitat stress.

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Table 1. Descriptive statistics (sample size [N], minimum [Min], maximum [Max], mean, standard error [SE], percent below the 5 mg/L target [% <5mg/L], and percent below the 3 mg/L threshold [% <3mg/L]) for monthly and seasonal bottom dissolved oxygen concentration (mg/L) for each category of subestuary salinity: mesohaline, oligohaline, and tidal-fresh. Stations were sampled multiple times per month during July, August, and September for the years 2003-2023. The summer category does not include October.

Estuary	Month	N	Min	Max	Mean	SE	% <5mg/L	% <3mg/L
Mesohaline	June	16	0.42	8.40	4.77	0.57	50	25
Mesohaline	July	557	0	14.30	4.21	0.09	57	27
Mesohaline	August	648	0	9.35	4.40	0.07	54	21
Mesohaline	September	599	0	9.07	5.03	0.08	39	17
Mesohaline	October	37	0.13	9.60	6.50	0.33	11	5
Oligohaline	June	16	2.76	6.77	5.21	0.29	31	6
Oligohaline	July	181	1.80	9.63	6.01	0.11	22	2
Oligohaline	August	185	2.13	9.54	6.14	0.08	14	1
Oligohaline	September	168	1.36	10.9	6.85	0.12	13	2
Oligohaline	October	9	6.85	9.80	8.59	0.27	0	0
Tidal-Fresh	June	1	6.50	6.50	6.50	---	0	0
Tidal-Fresh	July	237	2.39	13.70	7.24	0.11	6	0
Tidal-Fresh	August	239	1.27	14.00	6.91	0.09	6	1
Tidal-Fresh	September	228	2.70	13.30	7.01	0.10	9	1
Tidal-Fresh	October	13	6.20	11.40	7.84	0.49	0	0
Mesohaline	Summer	1820	0	14.3	4.56	0.05	50	21
Oligohaline	Summer	550	1.36	10.90	6.28	0.06	17	1
Tidal-Fresh	Summer	705	1.27	14.00	7.05	0.06	7	1

Table 2. Descriptive statistics (sample size [N], minimum [Min], maximum [Max], mean, and standard error [SE]) for all measurements of summer bottom dissolved oxygen (DO_b) concentration in mg/L, percent of shoreline as structure ($\%L_{str}$), distance in meters from station to shoreline (D_{shore}), and adjacent 12-digit subwatershed percent impervious surface (%IS) estimates within each subestuary. The S column designates the salinity classification for each subestuary: M = mesohaline, O = oligohaline, and T = tidal-fresh.

					DO			**$\%L_{str}$**			**D_{shore}**			**%IS**
Subestuary	**S**	**N**	**Min**	**Max**	**Mean (SE)**	**Min**	**Max**	**Mean (SE)**	**Min**	**Max**	**Mean (SE)**	**Min**	**Max**	**Mean (SE)**
Bohemia River	O	20	4.3	8.9	6.41 (0.31)	0	34.7	23.4 (2.8)	107	370	267 (23)	2.74	3.35	3.22 (0.05)
Breton Bay	M	75	0	9.07	3.74 (0.27)	0	100	37.8 (2.3)	68.3	517	323 (11)	7.27	7.78	7.52 (0.02)
Broad Creek	M	162	2.34	8.33	5.99 (0.08)	0	100	71.7 (1.9)	65.9	742	246 (12)	4.61	4.72	4.68 (0.00)
Bush River	O	129	2.13	10.9	6.53 (0.15)	0	100	37.2 (3.0)	188	344	285 (6)	1.92	12.4	7.52 (0.38)
Corsica River	M	231	0.02	9.35	4.12 (0.11)	0	100	19.3 (0.9)	59.2	246	146 (2)	2.3	2.55	2.43 (0.00)
Fishing Bay	M	14	6	7.5	6.79 (0.14)	0	56.8	17.9 (6.9)	405	1169	810 (82)	0.461	1.21	0.67 (0.09)
Gunpowder River	O	101	3.68	9.5	6.72 (0.10)	0	100	27.7 (3.2)	18.6	841	334 (26)	7.77	12.9	9.90 (0.21)
Harris Creek	M	117	4.72	7.91	6.21 (0.07)	0	100	48.1 (2.5)	67.4	527	238 (10)	5.01	5.09	5.07 (0.00)
Langford Creek	M	107	1.7	14.3	5.66 (0.20)	0	43.8	8.5 (1.4)	50.6	262	129 (5)	1.96	2.1	2.03 (0.01)
Magothy River	M	27	0.1	5.7	2.00 (0.40)	51.4	100	79.3 (3.5)	83.2	525	258 (31)	13	20.5	16.90 (0.74)
Mattawoman Creek	T	314	2.7	10.4	7.24 (0.08)	0	64.9	5.7 (0.7)	68.7	472	239 (6)	7.49	8.42	7.94 (0.02)
Middle River	O	198	1.36	10.3	6.04 (0.10)	0	100	75.3 (2.2)	4.49	486	171 (9)	30.4	31.2	30.80 (0.02)
Miles River	M	114	0.14	8.4	4.06 (0.17)	0	100	41.4 (3.1)	39.3	677	190 (10)	2.14	10.1	3.45 (0.27)
Nanjemoy Creek	O	102	1.8	8.83	6.00 (0.13)	0	100	15.4 (1.9)	28	537	236 (15)	1.35	1.51	1.45 (0.00)
Northeast River	T	297	1.27	13.7	6.77 (0.10)	0	100	73.0 (1.3)	179	715	424 (8)	8.44	9.93	9.03 (0.03)
Piscataway Creek	T	46	5.1	14	7.96 (0.27)	0	100	9.2 (3.3)	98.1	307	210 (9)	10.3	11.5	11.10 (0.06)
Rhode River	M	36	0.7	7.41	4.68 (0.26)	11.3	54.4	32.8 (3.7)	170	263	216 (8)	9.43	9.58	9.52 (0.01)
Sassafras River	T	48	4.56	9.82	6.68 (0.17)	0	86.2	20.9 (3.1)	61.1	467	242 (16)	2.02	3.47	2.50 (0.06)
Severn River	M	99	0	7.9	1.73 (0.20)	21.6	100	80.5 (2.3)	54.8	859	321 (28)	11.5	25	14.50 (0.34)
South River	M	120	0.01	6.7	2.49 (0.16)	20.8	100	70.1 (2.2)	101	564	251 (11)	9.67	29.1	19.70 (0.56)
St. Clements Bay	M	76	0	8.4	4.09 (0.27)	0	31.7	4.4 (0.7)	114	549	312 (12)	3.53	3.61	3.58 (0.00)
Tred Avon River	M	423	0.13	8.22	5.02 (0.07)	0	100	48.3 (1.4)	32.4	351	202 (4)	4.49	5.31	4.94 (0.01)
West River	M	36	1.1	6.89	4.67 (0.26)	64.6	95.6	80.1 (2.6)	131	267	199 (12)	11.3	11.5	11.40 (0.02)
Wicomico River: western shore	M	94	0.71	6.76	4.85 (0.14)	0	100	27.1 (2.7)	191	1103	720 (29)	1.54	3.38	1.88 (0.04)
Wye River	M	89	0.76	8.45	5.02 (0.16)	0	100	23.7 (2.7)	2.99	294	124 (10)	1.58	4.45	3.08 (0.14)

Table 3. Descriptive statistics (number of years [N_{yr}], minimum [Min], maximum [Max], mean, and standard error [SE]) for each subestuary's annual mean summer bottom dissolved oxygen (DO_b) concentration in mg/L, percent of shoreline as structure ($\%L_{str}$), distance in meters from station to shoreline (D_{shore}), and adjacent 12-digit subwatershed percent impervious surface (%IS). The S column designates the salinity classification for each subestuary: M = mesohaline, O = oligohaline, and T = tidal-fresh.

				DO			$\%L_{str}$			D_{shore}			%IS	
Subestuary	**S**	**N_{yr}**	**Min**	**Max**	**Mean (SE)**	**Min**	**Max**	**Mean (SE)**	**Min**	**Max**	**Mean (SE)**	**Min**	**Max**	**Mean (SE)**
Bohemia River	O	1	6.41	6.41	6.41 (---)	23.4	23.4	23.4 (---)	267	267	267 (---)	3.22	3.22	3.22 (---)
Breton Bay	M	3	3.52	3.99	3.74 (0.14)	37.7	37.8	37.8 (0.0)	301	336	324 (12)	7.27	7.78	7.53 (0.15)
Broad Creek	M	7	5.57	6.63	6.00 (0.13)	68.4	76.0	71.7 (1.2)	224	266	246 (6)	4.61	4.72	4.68 (0.01)
Bush River	O	12	4.66	8.54	6.47 (0.32)	8.0	66.8	41.5 (8.4)	280	299	286 (2)	6.27	7.96	7.43 (0.18)
Corsica River	M	12	3.08	5.09	4.13 (0.19)	14.2	22.0	19.1 (0.7)	116	161	146 (3)	2.30	2.55	2.43 (0.02)
Fishing Bay	M	1	6.79	6.79	6.79 (---)	17.9	17.9	17.9 (---)	810	810	810 (---)	0.68	0.68	0.68 (---)
Gunpowder River	O	8	6.10	7.14	6.69 (0.14)	13.1	46.3	27.2 (3.7)	228	482	328 (34)	9.19	10.7	9.89 (0.16)
Harris Creek	M	5	6.01	6.56	6.21 (0.11)	39.5	57.0	48.1 (3.6)	220	247	238 (5)	5.01	5.09	5.07 (0.02)
Langford Creek	M	5	5.07	6.46	5.68 (0.23)	6.1	11.6	8.4 (1.2)	97	177	126 (17)	2.01	2.05	2.03 (0.01)
Magothy River	M	1	2.00	2.00	2.00 (---)	79.3	79.3	79.3 (---)	258	258	258 (---)	16.90	16.90	16.90 (---)
Mattawoman Creek	T	16	5.95	8.66	7.22 (0.21)	1.3	17.9	5.4 (1.3)	194	263	240 (5)	7.49	8.42	7.99 (0.11)
Middle River	O	9	5.21	7.33	6.09 (0.22)	58.4	86.7	75.7 (3.1)	153	195	171 (5)	30.4	31.2	30.8 (0.09)
Miles River	M	5	3.31	5.47	4.05 (0.39)	30.3	45.8	41.6 (2.9)	161	292	192 (25)	2.17	6.69	3.50 (0.89)
Nanjemoy Creek	O	10	5.16	7.45	6.21 (0.24)	3.4	42.5	18.0 (3.2)	167	411	240 (23)	1.35	1.51	1.45 (0.02)
Northeast River	T	13	5.30	7.80	6.79 (0.22)	67.9	77.9	72.9 (0.8)	337	497	427 (16)	8.44	9.93	9.01 (0.14)
Piscataway Creek	T	9	6.61	9.47	7.88 (0.33)	0	17.3	8.4 (2.8)	191	232	213 (5)	10.3	11.5	11.1 (0.12)
Rhode River	M	3	4.03	5.49	4.68 (0.43)	32.8	32.8	32.8 (0)	216	216	216 (0)	9.43	9.58	9.52 (0.05)
Sassafras River	T	2	6.30	7.06	6.68 (0.38)	19.3	22.5	20.9 (1.6)	230	254	242 (12)	2.45	2.56	2.50 (0.06)
Severn River	M	4	0.96	2.64	1.74 (0.35)	46.6	91.5	80.2 (11.2)	292	337	321 (10)	13.7	16.5	14.5 (0.67)
South River	M	5	1.76	3.77	2.49 (0.34)	67.1	72.4	70.1 (0.8)	241	256	251 (3)	17.6	22.8	19.7 (1.18)
St. Clements Bay	M	3	3.39	4.61	4.13 (0.37)	3.4	6.0	4.3 (0.8)	296	322	313 (9)	3.53	3.61	3.58 (0.02)
Tred Avon River	M	18	4.35	6.11	5.03 (0.12)	35.1	60.9	48.6 (2.4)	170	246	203 (7)	4.49	5.28	4.93 (0.05)
West River	M	3	3.99	5.58	4.67 (0.48)	80.1	80.1	80.1 (0)	199	199	199 (0)	11.30	11.50	11.40 (0.06)
Wicomico River: western shore	M	5	4.30	5.44	4.91 (0.21)	17.0	41.6	26.8 (4.7)	593	812	722 (38)	1.78	2.18	1.89 (0.08)
Wye River	M	4	4.67	5.70	5.04 (0.24)	11.1	34.6	22.3 (6.3)	96	146	126 (11)	3.01	3.15	3.07 (0.03)

Table 4. Analysis of variance (ANOVA) tests to determine if the response and explanatory variables were independent of the salinity categories. Response variable was mean summer bottom dissolved oxygen (DO_b, mg/L) and response variables were percent of shoreline as structure ($\%L_{str}$), distance from station to shoreline (D_{shore}, meters), and subwatershed percent impervious surfaces (%IS). Tukey's Honest Significant Difference (HSD) test was used to determine if and which salinity categories (M = mesohaline, O = oligohaline, and T = tidal-fresh) were different for the variable being tested.

ANOVA Response: DO_b	Df	Sum Sq	Mean Sq	F value	P
Salinity	2	209	104.7	87.8	< 0.0001
Residuals	161	192	1.2		
Tukey's HSD	diff	Lwr	upr	P_{adj}	
O-M	1.763	1.267	2.259	< 0.0001	
T-M	2.603	2.107	3.099	< 0.0001	
T-O	0.840	0.262	1.417	0.0021	

Explanatory: $\%L_{str}$	Df	Sum Sq	Mean Sq	F value	P
Salinity	2	2450	1225	2.45	0.089
Residuals	161	80343	499		
Tukey's HSD	diff	lwr	upr	P_{adj}	
O-M	-0.900	-11.05	9.252	0.9760	
T-M	-9.251	-19.40	0.901	0.0821	
T-O	-8.351	-20.17	3.466	0.2192	

Explanatory: D_{shore}	Df	Sum Sq	Mean Sq	F value	P
Salinity	2	62286	31143	2	0.14
Residuals	161	2505990	15565		
Tukey's HSD	diff	lwr	upr	P_{adj}	
O-M	8.893	-47.804	65.59	0.9270	
T-M	47.462	-9.234	104.16	0.1204	
T-O	38.570	-27.424	104.56	0.3524	

Explanatory: %IS	Df	Sum Sq	Mean Sq	F value	P
Salinity	2	907	453	10.7	< 0.0001
Residuals	161	6804	42		
Tukey's HSD	diff	lwr	upr	P_{adj}	
O-M	5.686	2.732	8.641	< 0.0001	
T-M	2.838	-0.116	5.793	0.0626	
T-O	-2.848	-6.287	0.5907	0.1258	

Table 5. Multiple regression results for each of the three salinity categories: mesohaline, oligohaline, and tidal-fresh. The coefficients percent of shoreline as structure ($\%L_{str}$), distance from station to shoreline (D_{shore}, meters), the $\%L_{str}$:D_{shore} interaction, and percent impervious surface (%IS) were included in the model to predict bottom dissolved oxygen (DO_b, mg/L).

Mesohaline Coefficients	Estimate	SE	t value	P
Intercept	4.0086	0.4782	8.38	< 0.0001
$\%L_{str}$	0.0646	0.0172	3.76	0.0003
D_{shore}	0.0042	0.0018	2.29	0.0245
% IS	-0.2105	0.0264	-7.97	< 0.0001
$\%L_{str}$:D_{shore}	-0.0002	0.0001	-2.49	0.0147
adj R^2	0.487			
F-statistic	20.7	4 & 79 DF		
P	< 0.0001			

Oligohaline Coefficients	Estimate	SE	t value	P
Intercept	4.6007	1.2008	3.83	0.0005
$\%L_{str}$	0.0451	0.0397	1.14	0.2628
D_{shore}	0.0068	0.0048	1.42	0.1644
% IS	-0.0390	0.0343	-1.14	0.2635
$\%L_{str}$:D_{shore}	-0.0001	0.0001	-0.98	0.3335
adj R^2	0.012			
F-statistic	1.12	4 & 35 DF		
P	0.363			

Tidal-Fresh Coefficients	Estimate	SE	t value	P
Intercept	10.2840	2.2796	4.51	< 0.0001
$\%L_{str}$	-0.0748	0.0409	-1.83	0.076
D_{shore}	-0.0150	0.0085	-1.76	0.088
% IS	0.0839	0.0763	1.10	0.279
$\%L_{str}$:D_{shore}	0.0003	0.0001	1.79	0.083
adj R^2	0.119			
F-statistic	2.32	4 & 35 DF		
P	0.0766			

Table 6. Simple slopes table and Johnson-Neyman interval for each multiple regression to determine if there was a range of distances from shoreline (D_{shore}, meters) where the percent of shoreline as structure ($\%L_{str}$) was a significant predictor of bottom dissolved oxygen (DO_b, mg/L). Simple slopes of the $\%L_{str}$ relationship to DO_b were tested at the mean D_{shore} and at ±1 standard deviation (SD) of mean D_{shore}. Significant ($\alpha = 0.05$) slope indicates that $\%L_{str}$ influences DO_b at the specified D_{shore}. Percent impervious surface was mean centered to isolate the interaction between $\%L_{str}$ and D_{shore}. Range is the minimum and maximum distance from shore. Adjusted false discovery rate (Adj FD rate) indicated the proportion of Type I error. Interval indicates the range of D_{shore} where the $\%L_{str}$ slope (i.e., relationship to DO_b) was significant.

Mesohaline								
D_{shore}	Conditional intercept	SE	t value	P	Slope $\%L_{str}$	SE	t value	P
-1 SD: 97	3.17	0.35	9.01	< 0.0001	0.05	0.01	4.21	< 0.0001
Mean: 248	3.80	0.28	13.66	< 0.0001	0.02	0.01	3.14	< 0.0001
+1 SD: 398	4.42	0.42	10.42	< 0.0001	-0.00	0.01	-0.27	0.79
Range: 96-812 m								
Adj FD rate: t = 2.58								
Interval: inside [-4326; 266]								

Oligohaline								
Distance	Conditional intercept	SE	t value	P	Slope $\%L_{str}$	SE	t value	P
-1 SD: 179	5.37	0.67	7.98	< 0.0001	0.02	0.02	1.21	0.23
Mean: 256	5.89	0.46	12.92	< 0.0001	0.01	0.01	0.98	0.34
+1 SD: 334	6.42	0.49	13.15	< 0.0001	0.00	0.01	0.08	0.94
Range: 153-482 m								
Adj FD rate: t = 4.62								
Interval: NA								

Tidal-Fresh								
Distance	Conditional intercept	SE	t value	P	Slope $\%L_{str}$	SE	t value	P
-1 SD: 195	8.09	0.39	20.50	< 0.0001	-0.03	0.02	-1.50	0.14
Mean: 295	6.59	0.57	11.55	< 0.0001	-0.00	0.01	-0.07	0.94
+1 SD: 395	5.10	1.39	3.66	< 0.0001	0.02	0.02	1.15	0.26
Range: 191-497 m								
Adj FD rate: t = 4.62								
Interval: NA								

Figure 1. Sample location and subestuary map where bottom dissolved oxygen was measured. Subestuary polygons are identified by salinity classification: mesohaline (clear), oligohaline (stippled), or tidal-fresh (cross hatched). Sample locations are symbolized by coordinate correction: correct (black circle), incorrect assigned median station coordinates (MSC, blue square), missing assigned MSC (red circle), and cluster repositioned to MSC (orange diamond).

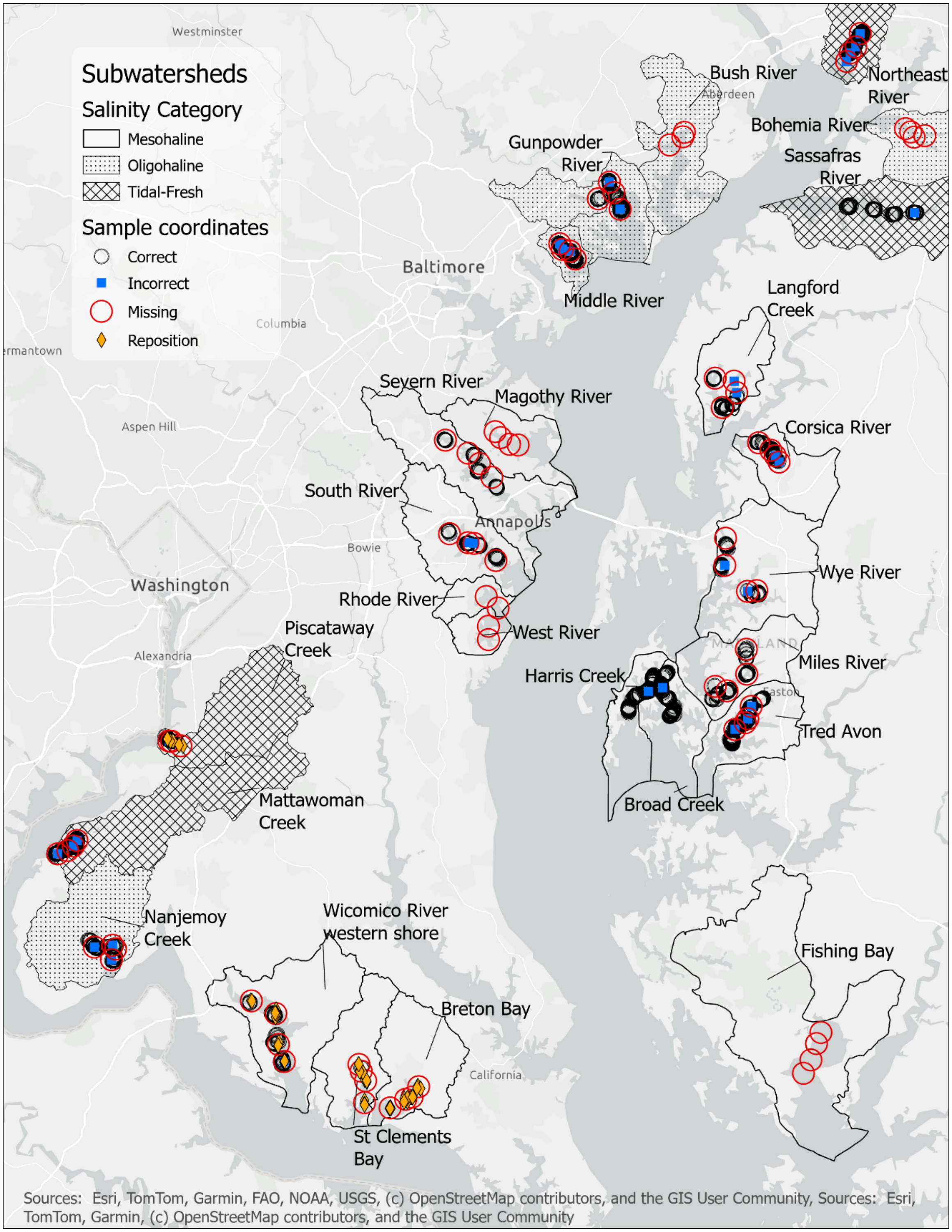


Figure 2. Boxplots of subestuary bottom dissolved oxygen (DO_b) concentration by month for each salinity category: mesohaline, oligohaline, and tidal-fresh. The bold line is the median, the box is the interquartile range (middle 50%), vertical lines are 1.5 times the interquartile range, and closed circles are outliers. Mean DO_b concentration is indicated by an open circle. Target 5 mg/L DO_b is indicated with a dotted line and the threshold 3 mg/L DO_b is indicated with a dashed line.

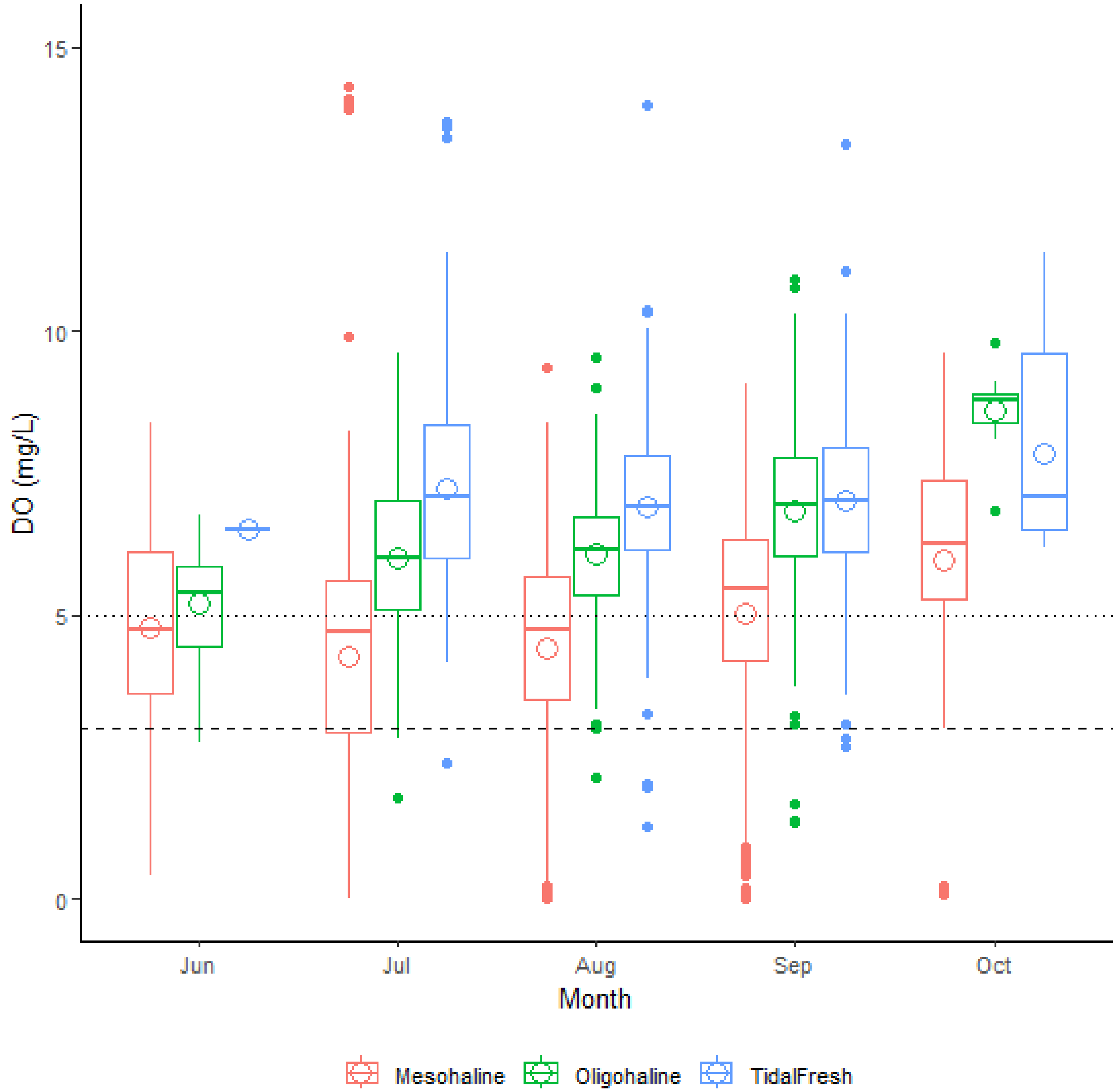


Figure 3. Boxplots depicting the distribution of (A) summer bottom dissolved oxygen (DO), (B) percent of shoreline as structure, (C) distance from station to shoreline, and (D) subwatershed percent impervious surfaces among the three salinity categories: mesohaline, oligohaline, and tidal-fresh. The bold line is the median, the box is the interquartile range (middle 50%), vertical lines are 1.5 times the interquartile range, and closed circles are outliers. In plot A, target 5 mg/L DO_b is indicated with a dotted line and the threshold 3 mg/L DO_b is indicated with a dashed line.

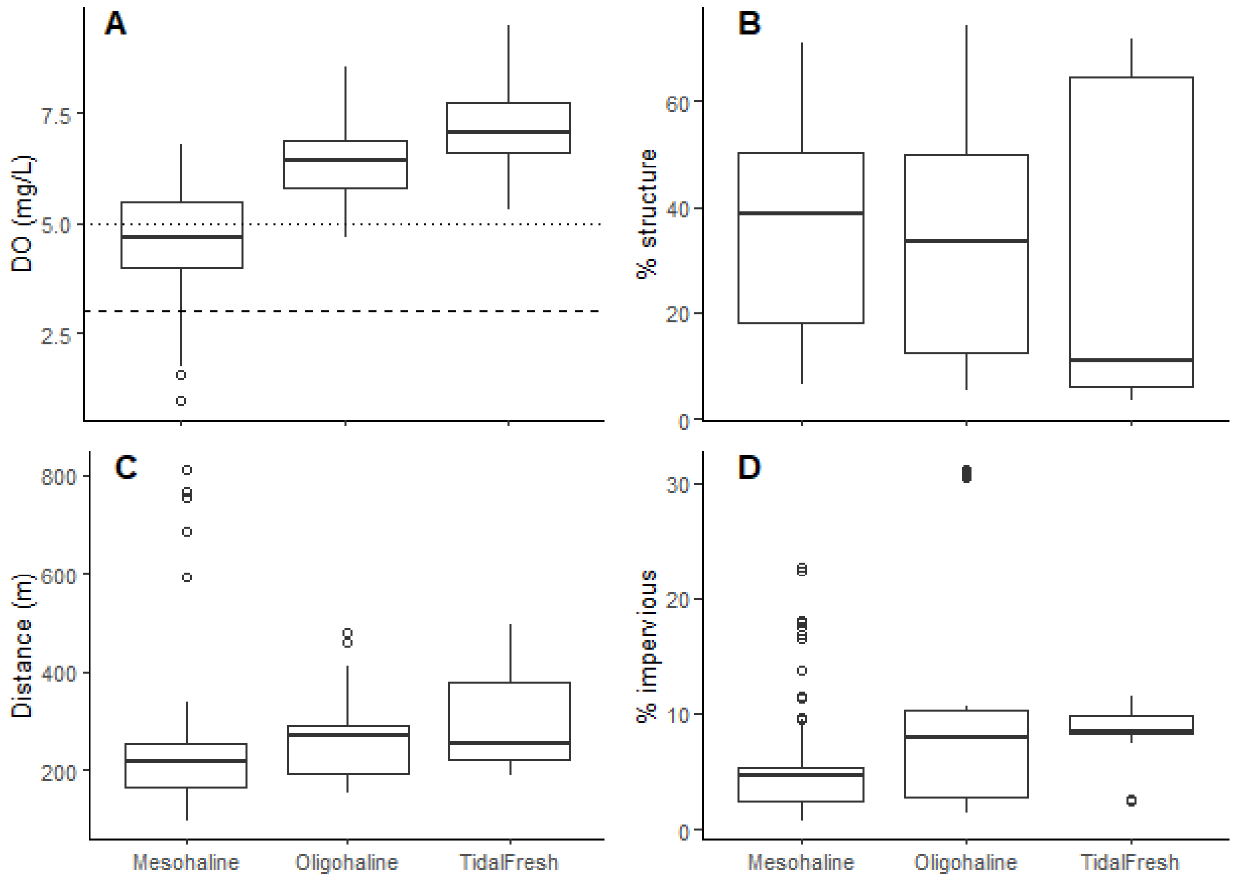


Figure 4. Added variable plots depicting the partial residuals for (left) bottom dissolved oxygen (DO_b) and percent impervious surfaces (%IS) and (right) DO_b and the interaction between percent of shoreline as structure (L_{str}) and distance between the sample station and shoreline (D_{shore}) while controlling the influence of other terms in the multiple regression model. Mesohaline, oligohaline, and tidal-fresh systems were modeled separately. Dashed lines are 95% confidence intervals.

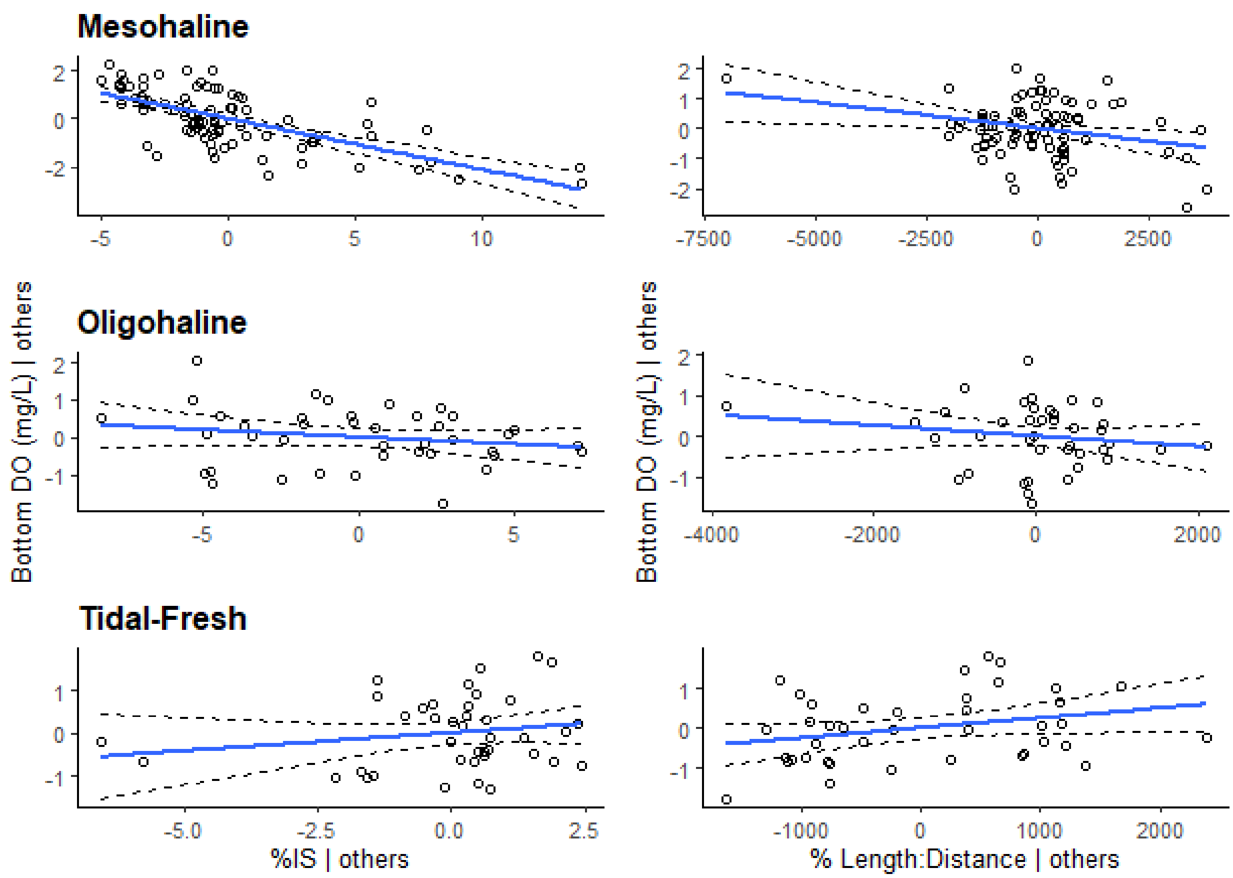
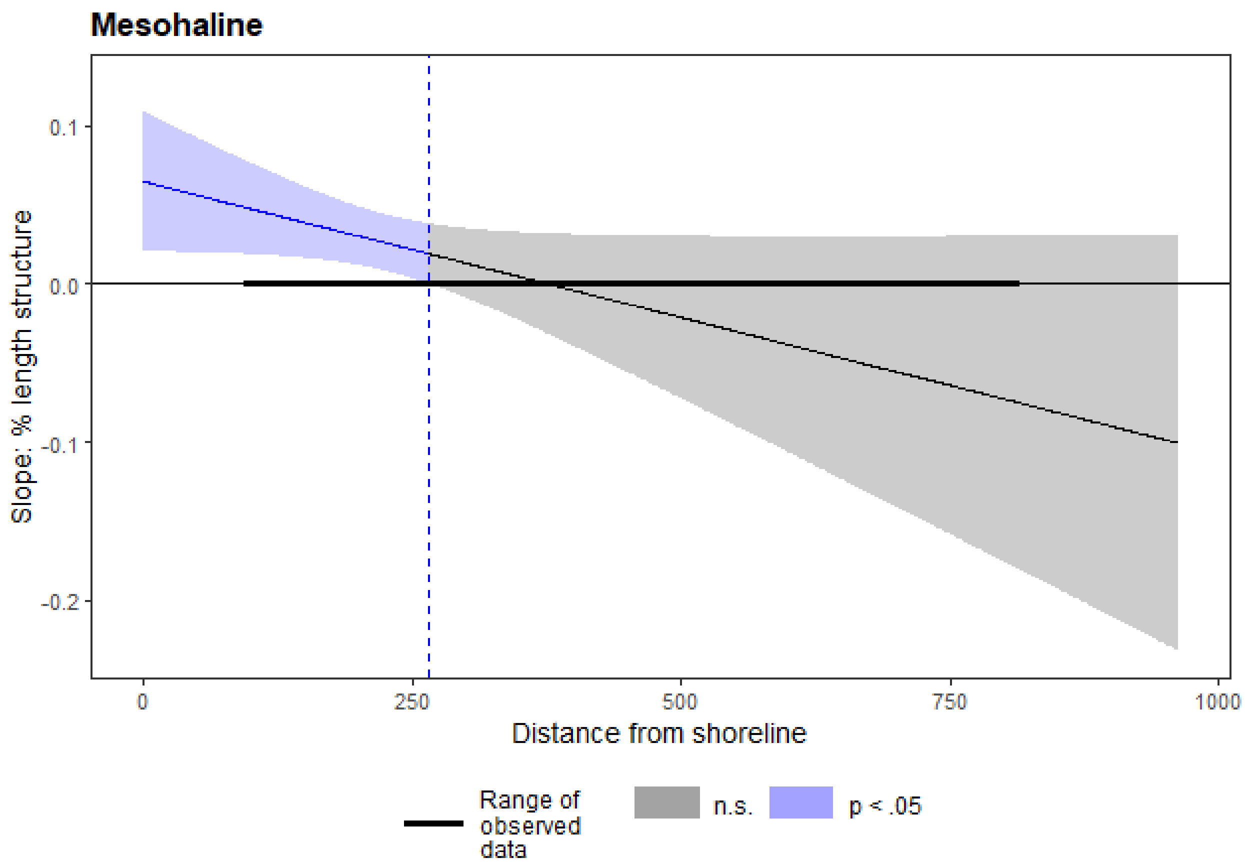


Figure 5. Johnson-Neyman plot for mesohaline subwatersheds that parses the multiple regression interaction term to identify the range of distances from the shoreline (D_{shore}) where the percent of shoreline as structure ($\%L_{str}$) is significantly related to the bottom dissolved oxygen (DO_b). Distances from shoreline with a significant ($\alpha = 0.05$) $\%L_{str}$ slope are highlighted (blue). The shaded area along the regression line represents the 95% confidence interval. The vertical dashed line indicates the boundary for significant slope values. The horizontal bold line represents the range of D_{shore} (96-812 m) where DO_b was measured and the plot extends one standard deviation beyond these distances (truncated at zero meter from shoreline). Plots are not shown for the oligohaline and tidal-fresh category due to non-significant $\%L_{str}$:D_{shore} interactions and the Johnson-Neyman test not being able to resolve significant D_{shore} intervals.



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

Project Staff
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Executive Summary

We developed a retrospective table of traffic lights (or TLI; red = poor, green = good, and yellow = uncertain) to communicate a historical perspective on forage status for resident Striped Bass in Maryland's portion of Chesapeake Bay (or Bay). This major fall forage TLI (FF TLI) can provide anglers and other stakeholders with a view of balance of resident Striped Bass and their major prey based on indicators and criteria that Maryland DNR managers consult and hopefully provide an understandable framework for communication. The FF TLI was organized into sections that describe well-being metrics for small (< 457 mm TL) or large Striped Bass, and metrics of availability of prey for both size classes (FRs). We followed the methods developed for a recent, peer-reviewed MD DNR TLI developed for resident Striped Bass and Atlantic Menhaden (ages 0 and older) in Maryland's portion of Chesapeake Bay (https://dnr.maryland.gov/fisheries/Pages/menhaden/traffic_light.aspx).

Red and green boundaries were determined from a1995-2021 reference period for our indicators (described below) except for the condition indicator, which had its own boundaries. The 1995-2021 reference period attempted to capture the prospect of providing adequate forage under current prevailing Striped Bass management, environmental, and ecological conditions. Indices of Striped Bass condition, relative abundance, natural mortality, and forage relative abundance from annual surveys and fall diets provided core indicators to assess forage status and Striped Bass well-being in Maryland's portion of Chesapeake Bay. We used the median of the core indices during the reference period as their yellow / green boundaries (going from uncertain to good levels) and the 25th or 75th percentiles, depending on relevant direction, as the yellow / red boundaries for transition from uncertain to poor levels. 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; core indicator), 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; core indicator) 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; supplemental indicator) augmented 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. An index of survival (SR; core indicator) 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; core indicators; focal prey species were Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab), but an index and FR for benthic invertebrate biomass was included. Other indices based on consumption were used as supplemental information rather than metrics evaluated in the FF TLI.

We identified four periods within the FF TLI time-series with similar color patterns of major forage indicators: 1995-2007, 2008-2013, 2014-2020, and 2021-2024. These time periods aligned with RI above its median (1995-2007 and 2014-2020) or below it. Red and yellow indicators were common during 1995-2007 for both size classes of Striped Bass and FRs. During 2008-2013, green indicators were predominant for FRs, while yellow and red indicators were common for well-being indicators of both Striped Bass size classes. During 2014-2020, FRs were often yellow or red. The FRs for Atlantic Menhaden and Bay Anchovy (seine only) were the exception and were predominately green. Well-being indicators for large Striped Bass were mostly good, while small fish indicators were more mixed among the color combinations. During 2021-2024, all classes of indicators were generally good with scattered uncertain indicators. Most of the changes of indicators from 1995-2020 were driven by variation in resident Striped Bass abundance; there was limited contrast in the forage indicators. After 2020, Atlantic Menhaden, Spot, and Bay Anchovy FRs reflected declining Striped Bass abundance and increased relative abundance of these forage fish. Blue Crab and benthic invertebrate FRs improved as well but reflected a more rapid drop in RI than the decline of relative abundance of these two forage groups.

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 ratios of large major prey and achievement of target P0 and PE for small fish in 2010, have been largely absent or minor in fall diets of both size classes between 2014 and 2024. Bay Anchovy were consistently present in fall diets of both size classes of Striped Bass during 2006-2014 but have fallen substantially as a numerical percentage of large fish diet since 2018 as Atlantic Menhaden became more frequent. Bay anchovy represented a variable percentage of small fish diets. Some bias towards larger prey 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 Maryland's portion of the Bay during the mid-1990s, followed by a dozen more years at high abundance after recovery was declared in 1995, coincided with declines in relative abundance of Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (i.e., major pelagic and benthic prey) to low levels. Changes in FRs largely reflected decreasing prey during 1983-1994 since RI was low. After 1995, prey indices stayed relatively low and 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 2022-2024. 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

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

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; Uphoff 2023).

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 2024). 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 2024). 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 under equilibrium conditions (Uphoff 2003). Increased year-class success was not accounted for in this analysis.

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 concurrently 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 457-711 mm TL 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).

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). 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).

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 (or upper Bay). Formal assessments of abundance and biomass of Striped Bass and most forage species in upper Bay are lacking due to cost and difficulty in mathematically separating migration from mortality. 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).

A traffic light index (TLI) for Atlantic Menhaden and Striped Bass balance in Maryland's portion of the Bay (including Potomac River) was developed during 2022-2024 as a reasonable, timely alternative for communicating forage status in the absence of model-based stock assessments and aerial surveys (Uphoff et al. 2024a). Participation by Jim Uphoff was funded by F-63.

The TLI for Menhaden and Striped Bass balance in Maryland's portion of the Bay was focused on age 0 and ages 1+ Menhaden and large (457-711 mm TL) resident Striped Bass capable of eating them (https://dnr.maryland.gov/fisheries/Pages/menhaden/traffic_light.aspx). It was intended to address the impact of the Menhaden fishery in the Bay that is of concern to some stakeholders. This TLI has been adopted by MD DNR and we used it as a basis for development of a TLI for major prey of resident Striped Bass under Objective 4. Objective 4's TLI is based on our broader suite of forage and well-being indicators (metrics) for small and large resident Striped Bass. This report provides indicators through 2024. The reader is referred to Uphoff (2024a) for more detailed explanation of reasoning and methodology beyond what follows here. In addition to providing information to judge whether the forage base is adequate to support resident Striped Bass in Maryland's portion of Chesapeake Bay, two additional objectives were low cost and tractability for staff.

A TLI uses a three-color scheme, patterned after familiar traffic lights, to classify multiple forage and Striped Bass well-being indicators as good or safe (green), intermediate or uncertain (yellow), and unacceptable or poor (red; Caddy and McGarvey 1996; Caddy 1998; 2002; 2015; Halliday et al. 2001). This retrospective table of traffic lights communicates a historical perspective on forage status in Maryland's portion of the Bay and would not be directly tied to management. This major fall forage TLI (hereafter, FF TLI) for Maryland's portion of the Bay can be developed at low cost using indices and TLI methods we have already developed through F-63. The FF TLI can provide anglers and other stakeholders with a view of relative status based on indicators and criteria that Maryland DNR managers consult and hopefully provide an understandable framework for public and management communication. There is potential to transform the FF TLI into Traffic Light Approach management triggers for the entire Chesapeake Bay. This would involve extensive future work with Bay jurisdictional partners (Virginia Marine Resources Commission and Potomac River Fisheries Commission), stakeholders, and the ASMFC (Uphoff et al. 2024a).

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 to Striped Bass ratios indexed potential attack success on major prey (Uphoff 2003; MD Sea Grant 2009) and 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. 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). Predators in natural systems may be closer to prey-predator ratio dependence than dependence on prey abundance alone (Ginzburg and Akçakaya 1992).

A benthic invertebrate index (invertebrates other than Blue Crabs) to Striped Bass ratio is included in this report (“soft bottom” benthic index) even though benthic invertebrates have not contributed much to fall diets. 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 Margaret McGinty (Uphoff et al. 2018) but could not be continued due to staff workload.

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 2024) 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 or varied without trend. 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

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 fall 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 by FEAD. 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 size classes 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 larger invertebrates that may have been 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 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 or weight 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). In addition, this indicator could be derived from published diet information from the 1930s (Hollis 1952), the 1950s (Griffin and Margraf 2003), and 1998-2000 (Overton et al. 2009) for comparisons within our small fish category. Overton et al. (2009) provided estimates of percent of Striped Bass stomachs with food during fall 1998-2000 (years combined) from a mid-Bay region that corresponded to our study area; 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). We examined the correlation of PE and PO 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.

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.

We determined predator-prey length ratios (PPLRs) for the two largest major prey in fall diets: Spot and Atlantic Menhaden. This analysis was based on ratios for whole prey only 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 2024). 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 2025a; 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 2025b). 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 2025). The BIBI has been employed to monitor water quality since 1995. 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 incorporated into a forage ratio for use in the TLI.

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 1983-2024 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 Bay in Maryland and this index would be as close to a global survey as could be obtained. Migratory Striped Bass 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. 2022). 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. 2022).

We used forage indices divided by RI (forage index-to-Striped Bass index ratios, i.e., forage ratio or FR) as indices 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 2022-2024. We did not estimate relative survival (SR) for 2021 due to concerns about the validity of the gill net index for that year because of an outbreak of Covid that shut down sampling during a key period (B. Versak, MDDNR, 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 2024). Table 8 in Versak (2024) provided mean values of 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 2024). 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) suggested 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; i.e., 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 most current ASMFC 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 FRs 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 2024; MD DNR 2025b). 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 2024). Geometric mean indices were back-transformed into their 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).

Fall Forage Traffic Light Index – The FF TLI was organized into sections that describe well-being metrics for small or large Striped Bass, or metrics for availability of prey (FRs). We created two categories of indicators: core and supplemental. Core indices were depicted in the FF TLI. Supplemental indices were considered supporting information and were reported separately to corroborate status of core indices, add additional insight, and provide an indication of uncertainty through their degree of agreement. Exclusion of supplemental indices from the FF TLI display averted over-weighting of a category by conceptually redundant information.

We used the 1995-2021 reference period adopted by Uphoff et al. (2024a). Red and green boundaries were determined from this reference period for our indicators (described below) except for the condition indicator which had its own boundaries. The 1995-2021 reference period attempted to capture the prospect of providing adequate forage under current prevailing Striped Bass management, environmental, and ecological conditions. Conditions considered for setting the reference period were Striped Bass population status and management, major prey status, eutrophication, hypoxia, and the status of the Atlantic Multidecadal Oscillation, a major climate pattern that influences Menhaden year-class success and other forage in the Bay (Woodland et al. 2021; Uphoff et al. 2024a). If a time-series for a core metric was longer than the reference period, the entire time-series was included as supplemental information. The reference period will be updated to include additional years as necessary rather than annually.

We used the median of the core indices during the reference period as their yellow / green boundaries (going from uncertain to good levels) and the 25th or 75th percentiles, depending on relevant direction, as the yellow / red boundaries for transition from uncertain to poor levels. The median and percentiles were used because skewed distributions were likely (Uphoff et al. 2025a). The median and mean would have the same meaning if an indicator was normally distributed within the time-series, while the median is a more robust indicator of central tendency if an indicator has a skewed distribution (Manikandan 2011a; 2011b). The 25th percentile represents a 50% decline from the series median (and mean if the indicators are normally distributed).

One metric, proportion of Striped Bass without body fat (condition index, P0), supplied its own boundaries. Justification and methodology for P0 was explained earlier.

Results

Sample Size Summary - During 1998-2024, 2,336 small and 3,624 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; 74 small Striped Bass were examined in 2024. Annual sample sizes for large fish ranged from 49 to 327 with a median of 205 and 179 large were examined in 2024. 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 (Table 2). The late start dates for 2021-2024 (October 14 – 29) reflected a dearth of fish available until before then (J. Uphoff, MD DNR, personal observation).

Small Striped Bass Condition, feeding success, and diet 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, 2021-2024 (2021-2024 have been the best of the time-series). Small fish in the upper Bay during fall were in poorest condition (threshold; $\geq$ 0.67) during 1998-2007, 2011-2012, 2016, and 2019. Estimates of P0 (0.36-0.46) were between poor and good during 2009-2010, 2013-2014, 2018, and 2020. The 90% confidence intervals of P0 allowed for separation of years at or near poor from remaining estimates (Figure 2).

Estimates of PE of small Striped Bass during fall, 2006-2024, ranged between 0.10 and 0.57 (Figure 3). Lowest estimates of PE for small fish (2009-2011, 2014, 2018, 2019, and 2023-2024) could be separated from most higher estimates (2006-2007, 2012, 2015, and 2022) based on 90% confidence interval overlap (Figure 3). Estimates of PE for small fish were poorly correlated with P0 (r = 0.08, P = 0.22); 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 96.2% of diet items encountered in small Striped Bass collected from upper Bay during fall, 2006-2024 (Figure 4). Bay Anchovy accounted for the highest percentage by number when all years were combined (60.9%, annual range = 9.3-87.9%); Atlantic Menhaden, 19.1% (annual range = 0-74.1%); Spot 5.4% (annual range = 0-70.7%); Blue Crab, 11.8% (annual range = 0.8-34.6%); and other items accounted for 3.8% (annual range = 0-12.9%; Figure 4). During 2024, Atlantic Menhaden accounted for 89.3% of the diet items; Bay Anchovy, 3.6%; Spot, 3.6%; Blue Crab, 3.6%; 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 2006-2024 (combined) were comprised of Atlantic Menhaden (75.1%), Bay Anchovy (12.1%), Spot (7.6%), Blue Crab (1.6%) and other items (3.6%; 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 2006-2024. 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 2006-2024 median) either Spot (2010) or Atlantic Menhaden (2013-2014) dominated diet mass. The 2024 estimate of C of small fish (0.016) was above the median (0.015) of the year time-series (Figure 5).

Median annual PPLRs of large prey (Spot and Atlantic Menhaden combined) of small Striped Bass were 0.20-0.38 during 2006-2024 (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 = 0.20) when Spot constituted a large fraction of their diet, and 2024 (PPLR = 0.23, second lowest of the time-series) when Menhaden were predominant. 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 intact 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 2017-2024. Estimated P0 was 0.033 in 2024. 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 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 high in 1998-2000, 2006, 2012, and 2017 based on 90% CI overlap and low during 2014-2015, 2018-2021, and 2023-2024 (Figure 8). Estimated PE was 0.27 in 2024 (Figure 8). There was a moderate association of PE and P0 (r = 0.64, P = 0.0015) during 2006-2024; the plot of these variables indicated that P0 at the target level was more likely when PE was 0.36 or less and a rapid ascent of most P0 points towards poorer condition beyond PE = 0.36 (Figure 9).

Major prey accounted for 94.9% of diet items, by number, encountered in large Striped

Bass diet samples during fall 2006-2023 (Figure 10). Atlantic Menhaden accounted for 50.8% by number when all years were combined (annual range =12.4-97.0%); Bay Anchovy, 15.6% (annual range = 0-32.5%); Spot, 7.0% (annual range = 0-52.4%); Blue Crab, 19.5% (annual range = 0-59.4%); and other items, 7.1% (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 2024, Atlantic Menhaden accounted for 82.5% of October-November diet items; Bay Anchovy, 0%; Spot, 2.5%; Blue Crab, 10.0%; and other items accounted for 5.0% (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 (88.7% of combined diet weight during fall, 2006-2024); Bay Anchovy accounted for 1.0%; Spot, 3.1%; Blue Crab, 3.1%; and other items, 4.1% (Figure 11). Estimates of C for large Striped Bass varied as much as 3.8-times among years sampled. During 2024, Atlantic Menhaden accounted for 97.1% of October-November diet items; Bay Anchovy, 0%; Spot, 1.3%; Blue Crab, 0%; and other items accounted for 1.0%. The 2024 estimate of C of large fish (0.0125) was below the time-series median (0.0135; Figure 11).

Median PPLRs of large prey (Spot and Atlantic Menhaden) for large Striped Bass were 0.19-0.30 during 2006-2024 (Figure 12). The median PPLR was 0.22 for 2024 (Figure 12). Median PPLRs for large prey of large Striped Bass were much closer to the optimum (0.21) estimated by Overton et al. (2009) than those of small fish.

Relative abundance indices of Striped Bass and major prey – Relative abundance of Striped Bass (RI) was lowest during 1983-1993 (median RI = 0.4 fish per trip; Figure 13). Estimates of RI then rose abruptly to a high level and remained there during 1995-2007 (median = 2.7). Estimates of RI fell during 2008-2013 (median = 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 and 0.6 in 2024. 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 2020-2024 was lower than 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 2023-2024. Bay Anchovy seine indices following the early to mid-1990s were typically low during 1959-1993 but 2023-2024 fell within the higher 1959-1994 range. Highest Bay Anchovy trawl indices (top quartile) occurred in 1989-1992, 1998-2000, 2013-2014, 2020-2021, and 2024, while lowest quartile indices occurred after 2006. There was little agreement between seine and trawl 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 2023 and 2024 Atlantic Menhaden seine indices were 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 is a need to take the influence of the Atlantic Multidecadal Oscillation (AMO) into account when judging Atlantic Menhaden seine indices (Buccheister et al. 2016; Uphoff et al. 2024a). The negative phase of the AMO had a strong concurrence with high seine indices during 1971-1994 (Uphoff et al. 2024a).

Spot seine indices have been above average during 2020-2024 and 2024 constituted one of the highest seine indices and the highest trawl index of their respective time-series (Figure 15). Major benthic forage indices were low after the 1990s, but years of higher relative abundance were interspersed during the 2000s. Seine (1959-2024) and trawl (1989-2024) 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 (1989-2024) were generally at or above the time-series median during 1989-1998, and 2009-2015. Blue Crab densities in 2020-2024 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 2021-2024 (Figure 16). This time-series covers 1995-2024 (Versar 2025).

Species-specific standardized FRs exhibited similar patterns during 1983-2024 (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, followed by an acceleration of FRs starting in 2020 (Figure 17).

The Atlantic Menhaden FRs in 2023-2024 were the highest since 1991 (Figure 18). Prior to 2023-2024, it was generally elevated during 2005-2022 from its nadir during 1997-2004 but well below levels prior to the early 1990s (Figure 18).

The Bay Anchovy seine FRs in 2023-2024 were the highest since 1991 and were above years of higher FRs since 1995 (2006-2009 and 2010-2013; Figure 19). The Bay Anchovy trawl FRs for 2023-2024 were in the top third of the time-series (Figure 20).

The Spot seine FR during 2020-2021 was in the higher portion of the 1959-2024 range exhibited since 1995 (Figure 21). The Spot seine (Figure 21) and trawl FRs (Figure 22) for 2024 were among the highest in their time-series and indicated considerable improvement over lows exhibited during 2014-2019.

The Blue Crab FR was above the time-series median in 2024 (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 mostly 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 stock assessment update (1982-2023). 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 2024 (2021 year-class, 106.6) was above the 1995-2024 post-recovery median (86.0; 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, SR in 2023 was the third highest, and 2024 was the 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 of M. 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). Estimates of SR in 2022-2024 were above the median. 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-dependent 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).

Fall Forage Traffic Light Index (FF TLI) – We identified four periods with similar color patterns of major forage indicators: 1995-2007, 2008-2013, 2014-2020, and 2021-2024 (Figure 27). These time periods aligned with RI above its median (1995-2007 and 2014-2020) or below it. Red and yellow indicators were common during 1995-2007 for both size classes of Striped Bass and FRs. During 2008-2013, green indicators were predominant for FRs, while yellow and red indicators were common for well-being indicators of both Striped Bass size classes. During 2014-2020, FRs were often yellow or red. The FRs for Atlantic Menhaden and Bay Anchovy (seine only) were the exception and were predominately green. Well-being indicators for large Striped Bass were mostly good, while small fish indicators were more mixed among the color combinations. During 2021-2024, all classes of indicators were generally good with scattered uncertain indicators (Figure 27). Figures 28-39 depict the numeric time-series of each indicator in the FF TLI since the beginning of the reference period (1995) and its estimated boundaries.

Discussion

Nearly all fall forage indicators have been at a good level since 2021. Most of the changes of indicators from 1995-2020 were driven by variation in resident Striped Bass abundance; forage indicators were generally low. After 2020, Atlantic Menhaden, Spot, and Bay Anchovy FRs reflected declining Striped Bass abundance and increased relative abundance of these forage fish. Blue Crab and benthic invertebrate FRs improved as well but reflected a more rapid drop in RI than the decline of relative abundance of these two forage groups.

The FF TLI represented 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).

Traffic light style representations can be used for the precautionary approach to fisheries management (Caddy 1998; Halliday et al. 2001) and can be adapted to ecosystem-based fisheries management (Fogarty 2014). The strength of the traffic light method is its ability to account for a broad spectrum of information, qualitative as well as quantitative, which might be relevant to an issue (Halliday et al. 2001). Simplicity and communicability are issues of over-riding importance (Halliday et al. 2001).

Discussions with DNR fisheries managers and stock assessment scientists indicated acceptance of a stoplight index for forage assessment in Maryland's portion of Chesapeake Bay. This approach has been developed for communicating the status of Atlantic Menhaden and Striped Bass balance in Maryland's portion of the Bay (Uphoff et al. 2024a). This Traffic Light Index (TLI) passed an outside peer-review as a communication tool on August 21, 2024, leading to the FF TLI developed here. There has been interest in a TLI approach for assessing forage in Chesapeake Bay from the Chesapeake Bay Program (J. Uphoff, personal observation).

The FF TLI was able to depict periods of forage related stress for both size classes of Striped Bass. While the FF TLI is a simple depiction of complex information, we chose to combine that with layers of supporting information. The full time-series of index values for each metric are provided as are (separately) the indices and their red and green boundaries for the reference time period. Hopefully, this helps stakeholders using the FF TLI.

Estimated P0 of small and large Striped Bass during 2023-2024 were among the best body fat indices for both size classes for the whole time-series (since 1998). 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, stressful environmental conditions (hypoxia, excessive heat), mycobacteriosis, and catch-and-release, natural, and harvest mortality that reduce abundance and intraspecific competition.

Summer 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). 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).

Response of condition may be lagged since condition of Striped Bass in summer was a good predictor of fall condition during 1999-2012 and condition in fall of the previous year appeared related to condition in the next fall during 1998-2021 (Uphoff et al. 2017; 2024a). If fewer fish make it through these hurdles, the survivors may benefit from reduced intraspecific competition for forage. Improvement in condition due to greatly reduced abundance of Striped Bass is not likely to be comforting to anglers or managers.

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 YOY Atlantic Menhaden by weight and number while small fish diets were more variable and had a higher frequency of small items, particularly Bay Anchovy.

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 fully 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 may make this correction in the future if staff time is sufficient. We based estimates of proportion of food represented by diet items for small fish on a standard TL range; however, Striped Bass in the smaller end of the size distribution were not always represented. Proportions of diet by number represented by Bay Anchovy and Atlantic Menhaden were affected by size of small 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). Proportions of Atlantic Menhaden in small Striped Bass diets were moderately and positively correlated with annual small Striped Bass mean, median, and minimum TL. The proportion of Bay Anchovy in small Striped Bass diets was moderately to strongly negatively correlated with small Striped Bass mean, median, and minimum TL. These associations indicated that small Striped Bass diet composition was influenced by the length of Striped Bass sampled (Uphoff et al. 2023).

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 in 2010 had contributed to lower PPLR of large major prey and achievement of target P0 and PE for small fish, have been largely absent or minor in fall diets of both size classes between 2014 and 2024. Bay Anchovy were consistently present in fall diets of both size classes of Striped Bass during 2006-2018 and fell afterward (except during 2023) as a percent of large fish diet as Atlantic Menhaden became frequent in their fall diet. 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 that coincided with declines in Atlantic Menhaden and Bay Anchovy (Pruell et al. 2003).

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; length-frequencies from pound net sampling 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 during 2008-2020 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, had biological significance based on improvement in recent body fat and fall diet metrics. Improvements in pelagic and benthic fish FRs since 2020 have reflected both increases in forage indices and declining resident Striped Bass relative abundance as poor year-classes now comprise most of the latter.

The soft bottom benthic index time-series covered 1995-2024 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 four years. The FR for benthic biomass (Figure 39) has been good during the three of the last four years as RI steadily declined. The benthic invertebrate FR may be optimistic since it does not take into account other benthic forage competitors that can be abundant (White Perch, Spot, Atlantic Croaker *Micropogonias undulatus*, Blue Crab, and Cownose Ray *Rhinoptera bonasus*).

Changes in benthic invertebrate populations have the potential to affect Striped Bass directly or through reductions in other benthic major prey that feed on invertebrates. 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). Non-predation copepod mortality 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 some 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 sizeable increase in SR was evident in 2022-2024. This estimate was from poor Striped Bass year-classes in 2019 and 2020 that were the first of a series of poor year-classes through 2024 (Durell and Weedon 2024). Elevation of SR of this series of poor year-classes 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). Striped Bass accounted for in SR are younger than those in the 457-711 mm interval. 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.

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 *Gadus morhua* 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).

Striped Bass tagging studies described above did not describe mortality/survival of small fish. 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 entering this transition and dependent on small prey and invertebrates beforehand.

Decreased survival in the mid-to-late 1990s was consistent with a density-dependent 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, organizer, and implementer 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. 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 (Uphoff et al. 2018).

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 (now FEAD) staff due to other duties. Piggybacking diet sampling onto the existing fall FWHP Striped Bass health survey provided a low-cost compromise that would provide some information on annual 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.

Year-round collections by CBEF during 2006-2015 provided an opportunity to examine seasonal and monthly dynamics of large resident Striped Bass condition (Uphoff et al. 2024a). Estimates of P0 (proportion without visible body fat) were 0.37 in summer, 0.51 in fall, and 0.08 in winter; consumption of Menhaden in summer was much less than fall and winter. Sample sizes were sufficient for precise monthly estimates of P0 during June-February when pooled across years (CV range of 2-26%). Estimates of P0 were near 0.20 in June-July and then increased (condition worsened) to 0.51 in August and 0.68 (near the potential starvation threshold) during September-October. Estimated P0 dropped to 0.28 in November, 0.15 in December, and reached a nadir of 0.05-0.06 during January-February. Condition of Striped Bass in fall was strongly related to condition in the preceding summer of the same year and in the fall of the previous year during 1998-2021 (Uphoff et al. 2024a).

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 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 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 likely changing. Schools of Striped Bass and their prey no longer have a fixed location, and become mixed (J. Uphoff, MD DNR, personal observation), making 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 worked in fall 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 numerous 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.

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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
FF TLI	Fall Forage Traffic Light Index – A traffic light display of relative status of Objective 4 forage indicators.
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_3	Hybrid gill net index of relative abundance of age-3 male Striped Bass that has been adjusted for catchability change with population size.
IF	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
2024	5	74	179	29-Oct	20-Nov	FWHP

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.

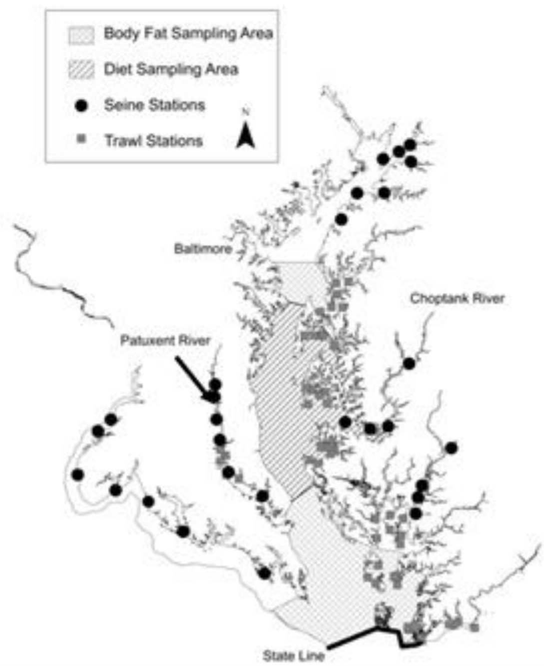

Figure 2. Proportion of small Striped Bass without body fat (P0) during October -November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

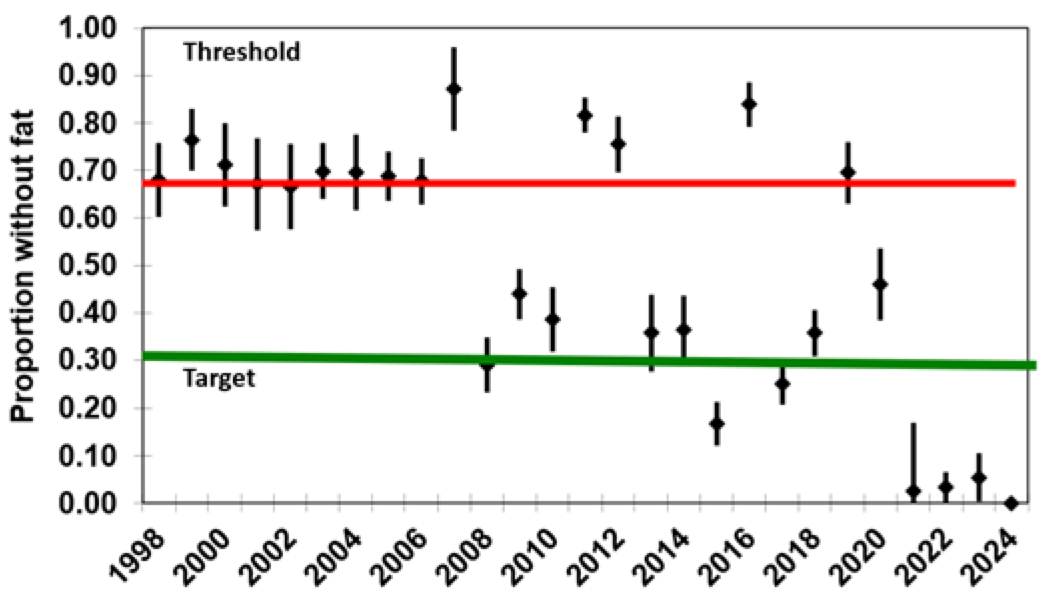

Figure 3. Proportion of small Striped Bass guts without food (PE) in fall and its 90% confidence interval.

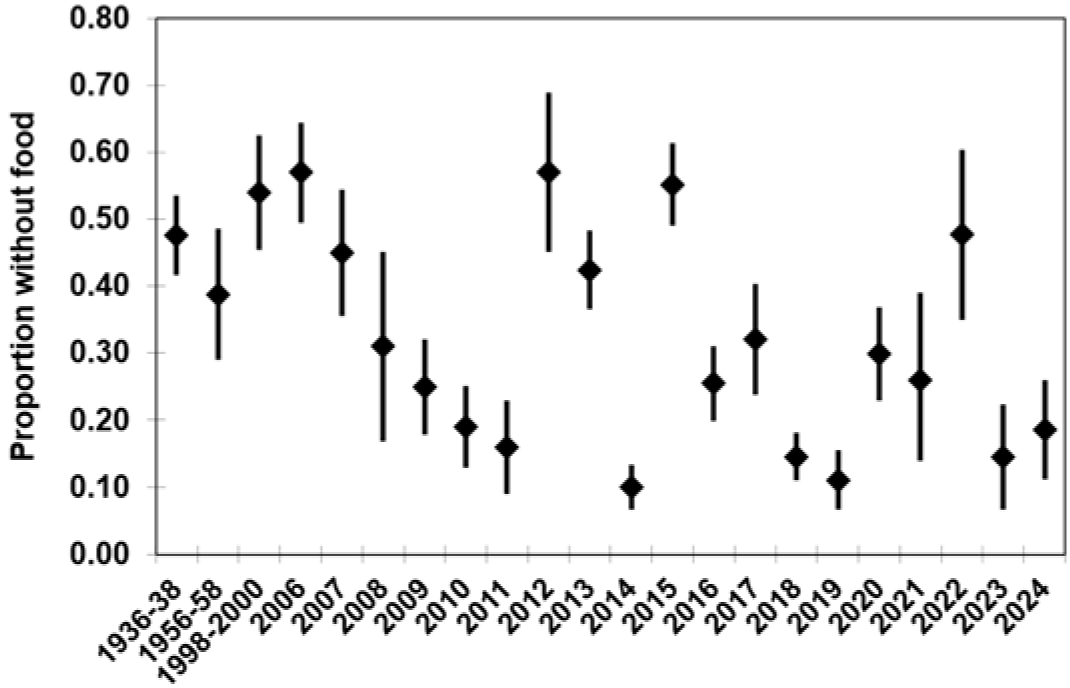

Figure 4. Percent, by number (counts of individuals plus presence of parts), of identifiable (excludes unknown) major forage groups in small Striped Bass (< 457 mm TL) guts, in fall.

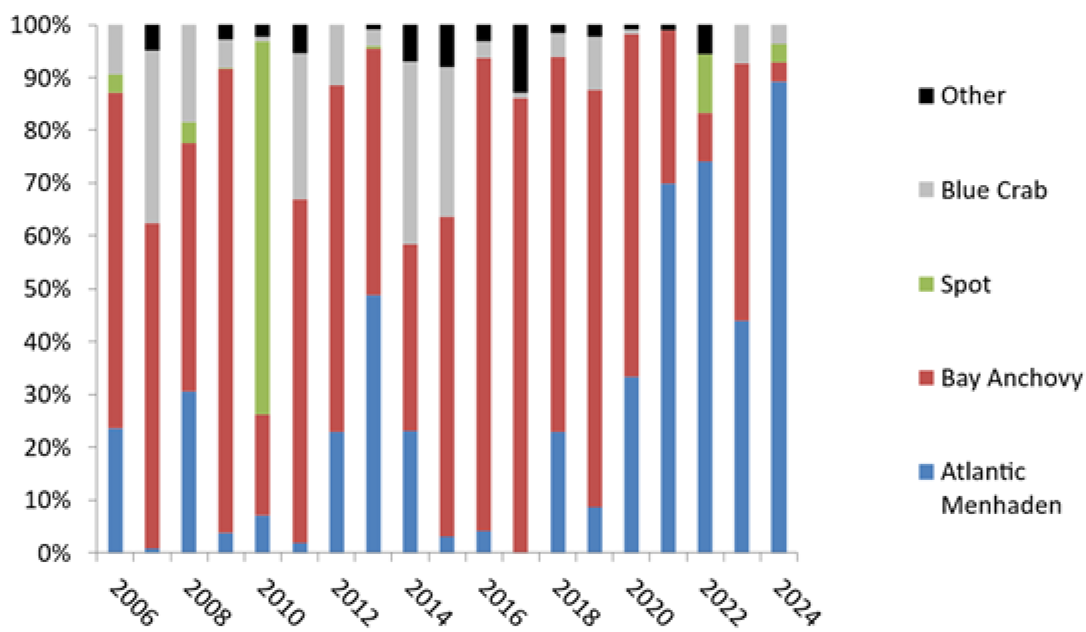

Figure 5. Gram prey consumed per gram (C) of small (< 457 mm TL) Striped Bass in fall hook -and-line samples. Age-0 forage dominate the diet. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

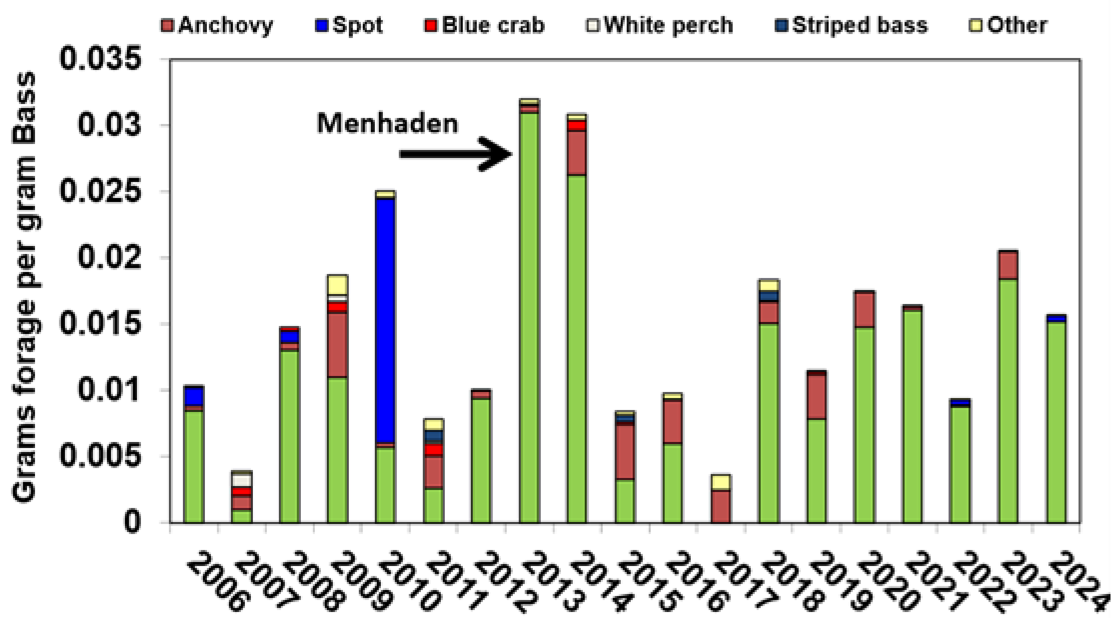

Figure 6. Median prey -predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for small (< 457 mm) Striped Bass. Optimum ratio was estimated by Overton et al. (2009).

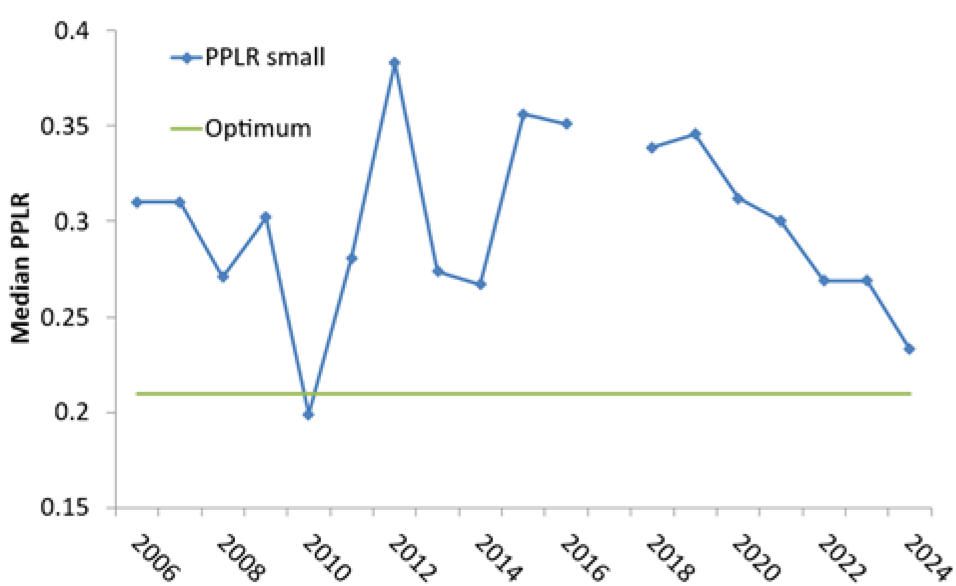

Figure 7. Proportion of large Striped Bass without body fat (P0) during October -November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

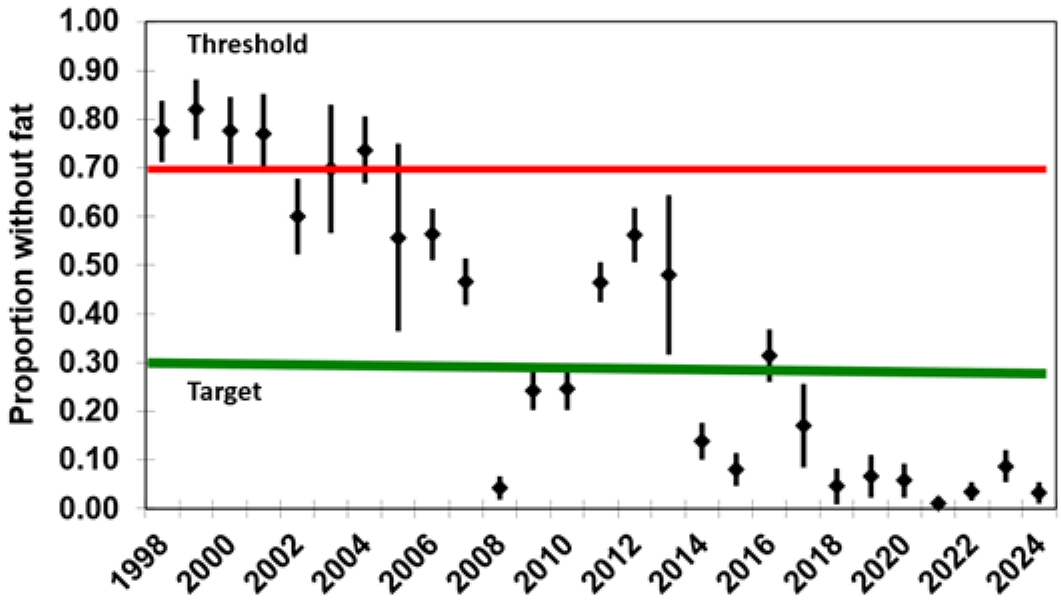

Figure 8. Proportion of large Striped Bass (≥ 457 mm or 18 in, TL) guts without food (PE) in fall and its 90% confidence interval.

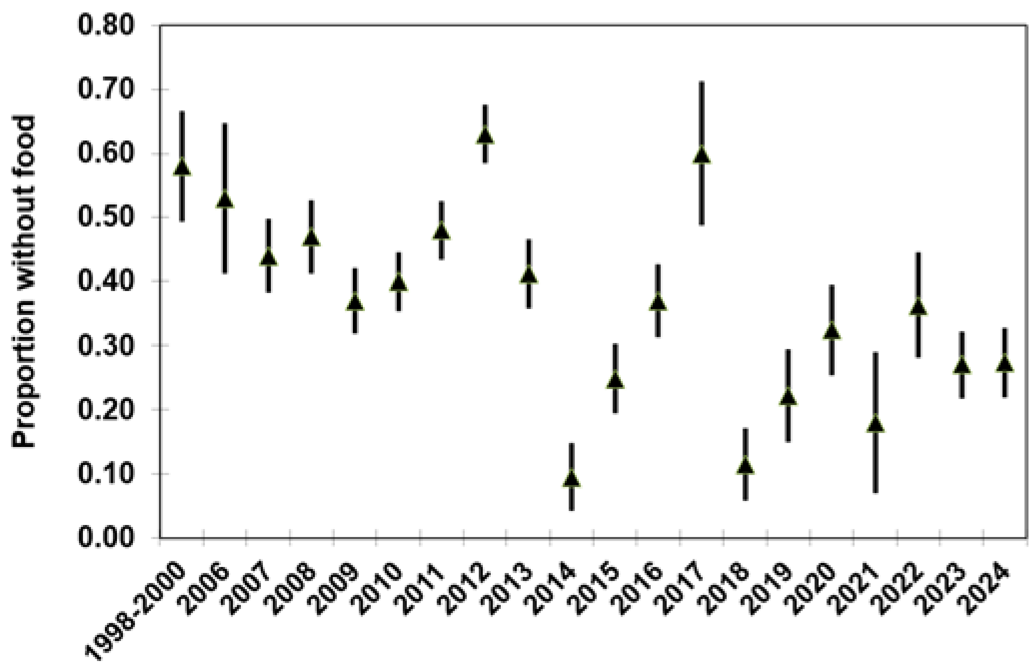

Figure 9. Plot of proportion of large (≤ 457 mm, TL) Striped Bass with empty guts in October-November against proportion without body fat, 2006 -2024.

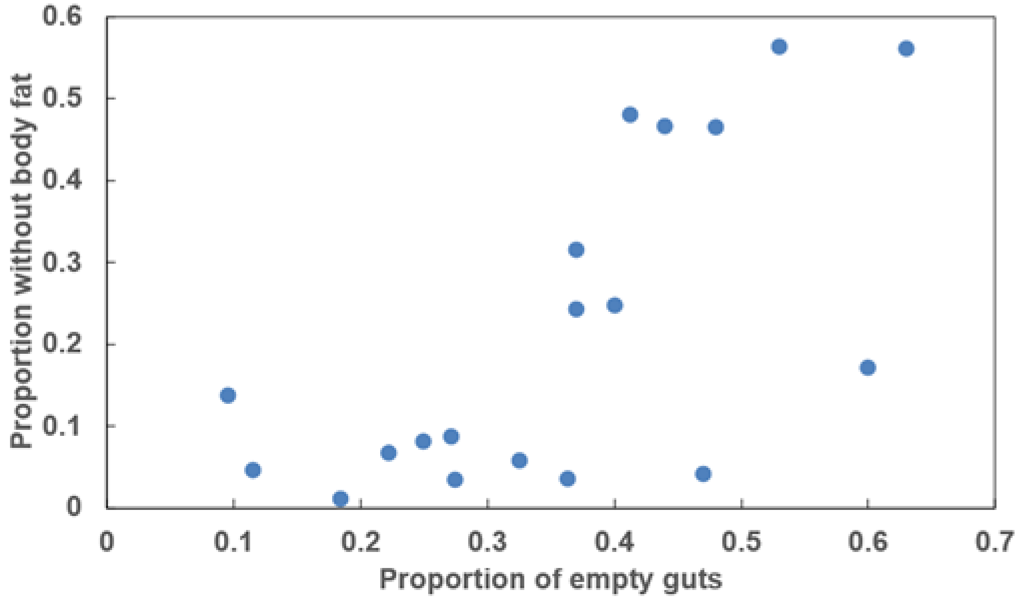

Figure 10. Percent of large Striped Bass (≥ 457 mm TL) identifiable diet represented by major forage groups, by number, in fall.

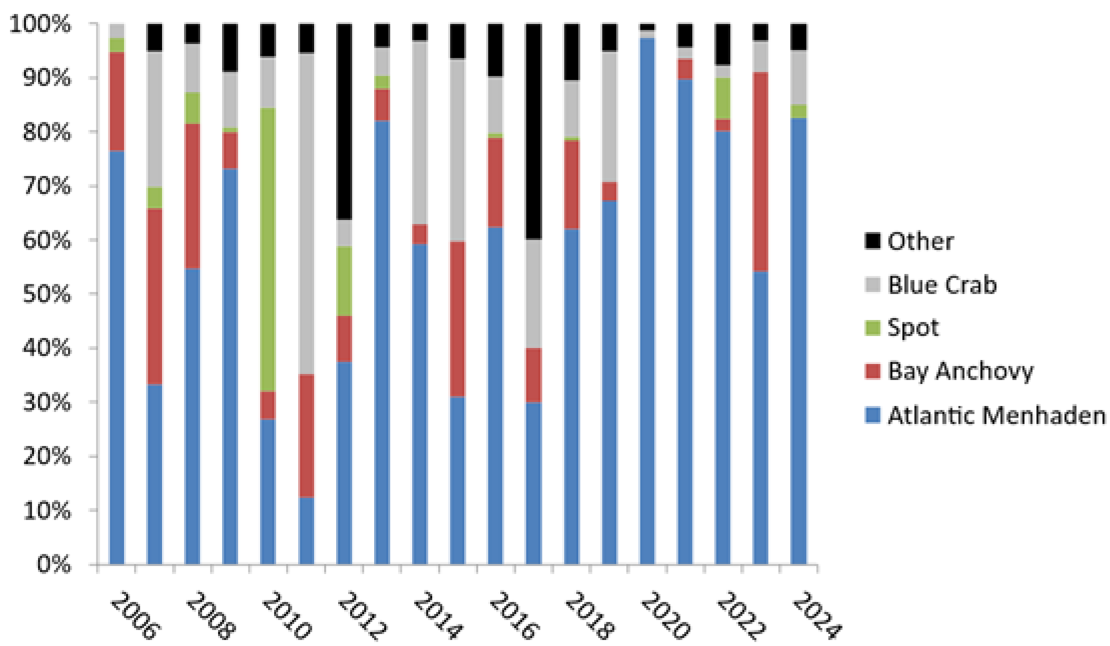

Figure 11. Grams of prey consumed per gram (C) of large (≥ 457 mm TL) Striped Bass during October-November. Fall consumption dominated by age 0 forage. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

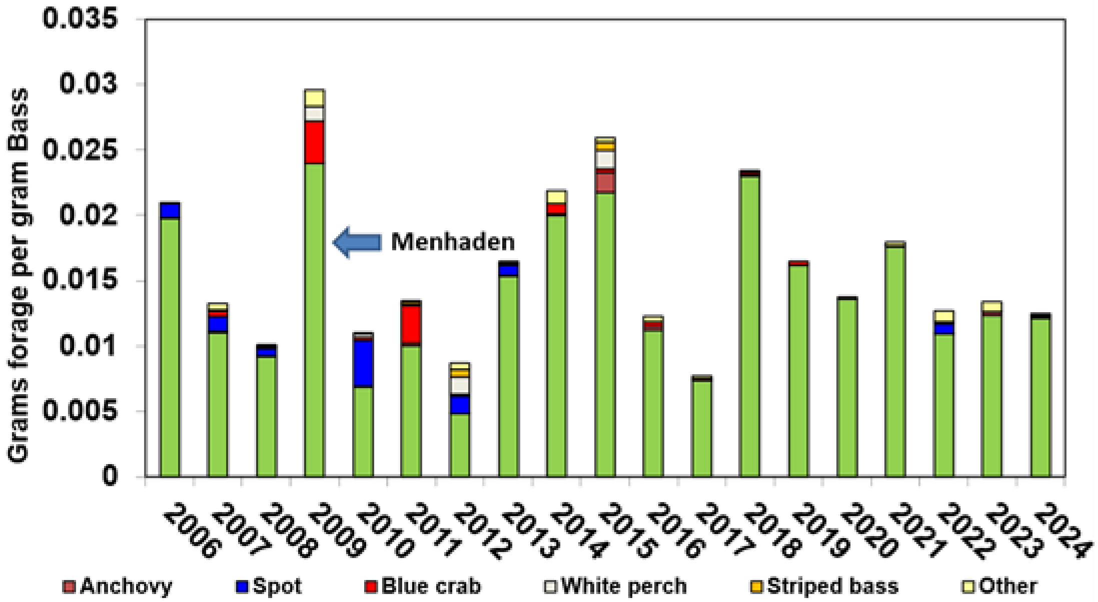

Figure 12. Median prey -predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for large Striped Bass (≥ 457 mm). Optimum ratio was estimated by Overton et al. (2009).

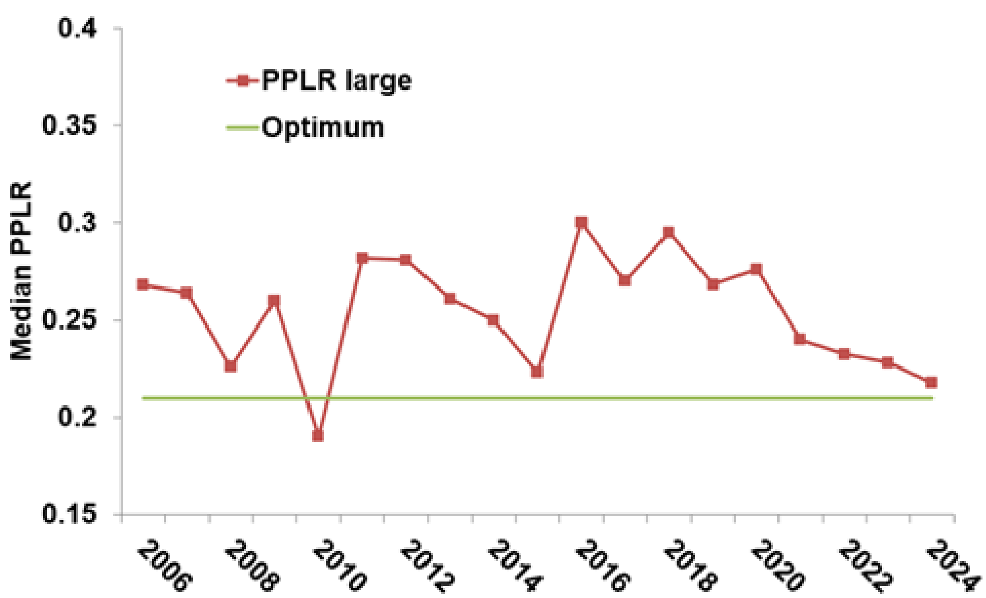

Figure 13. Maryland resident Bay Striped Bass annual abundance index (RI; MD MRIP inshore recreational catch per private boat trip during September-October; mean = black line) since 1983 and its 90% confidence intervals based on @Risk simulations of catch and effort distributions. Catch = number harvested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

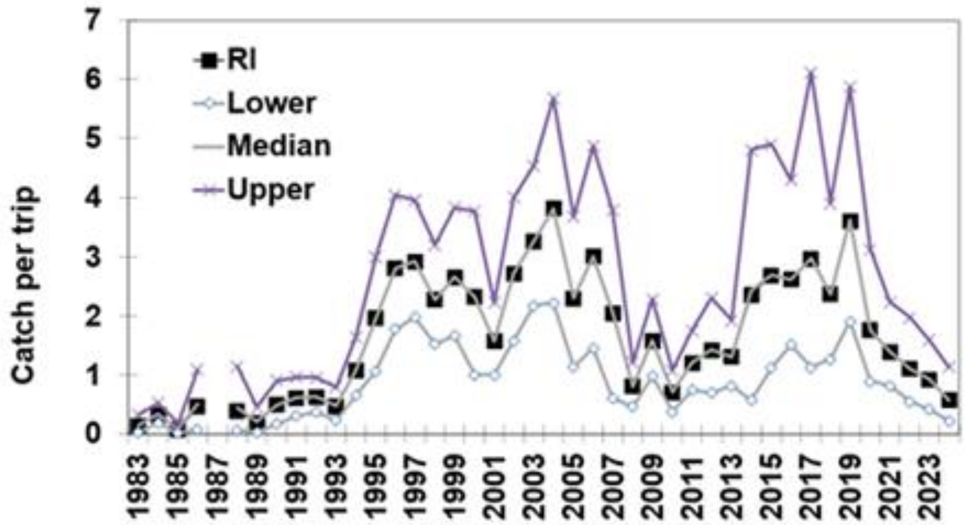

Figure 14. Trends in major pelagic prey of Striped Bass in Maryland Chesapeake Bay surveys since 1959. Indices were standardized to their means since 1989 (years in common). Menhaden = Atlantic Menhaden and Anchovy = Bay Anchovy.

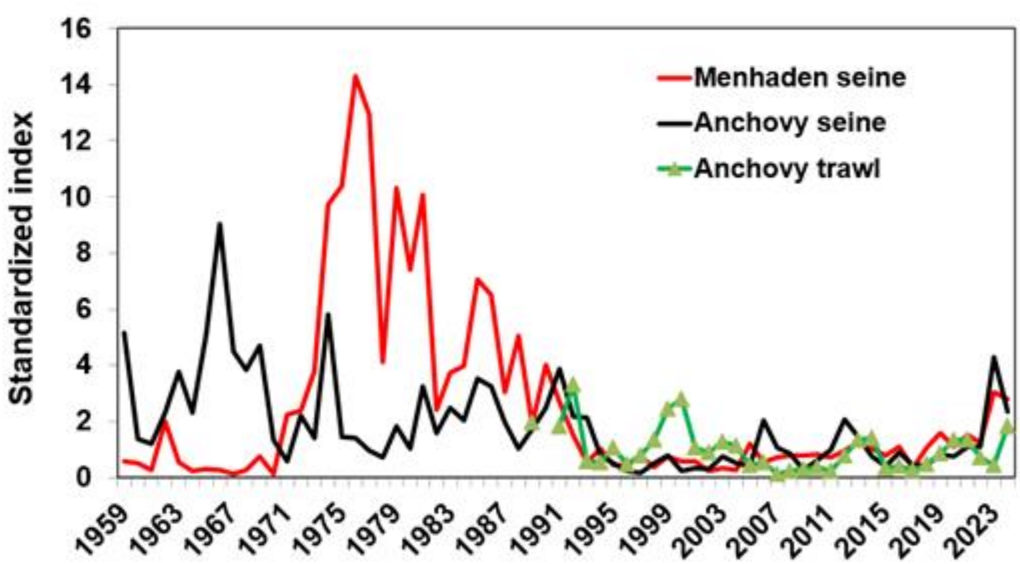

Figure 15. Trends in major benthic prey of Striped Bass in Maryland Chesapeake Bay surveys, since 1959. Indices were standardized to their means since 1989 (years in common).

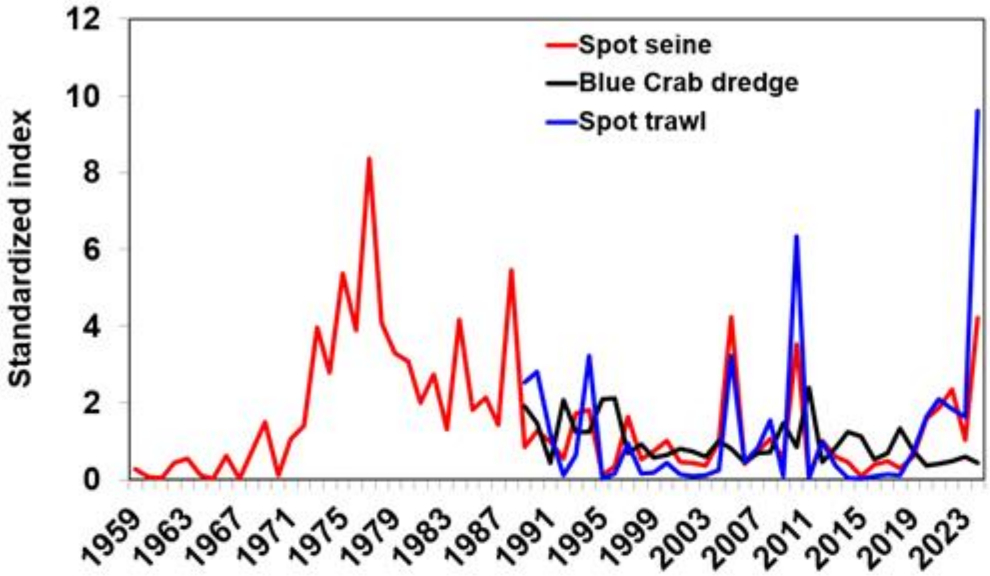

Figure 16. Trends in soft bottom benthic invertebrate biomass index (grams / m ^{2} based on Figure 3-40 in Versar 2025) in Maryland waters.

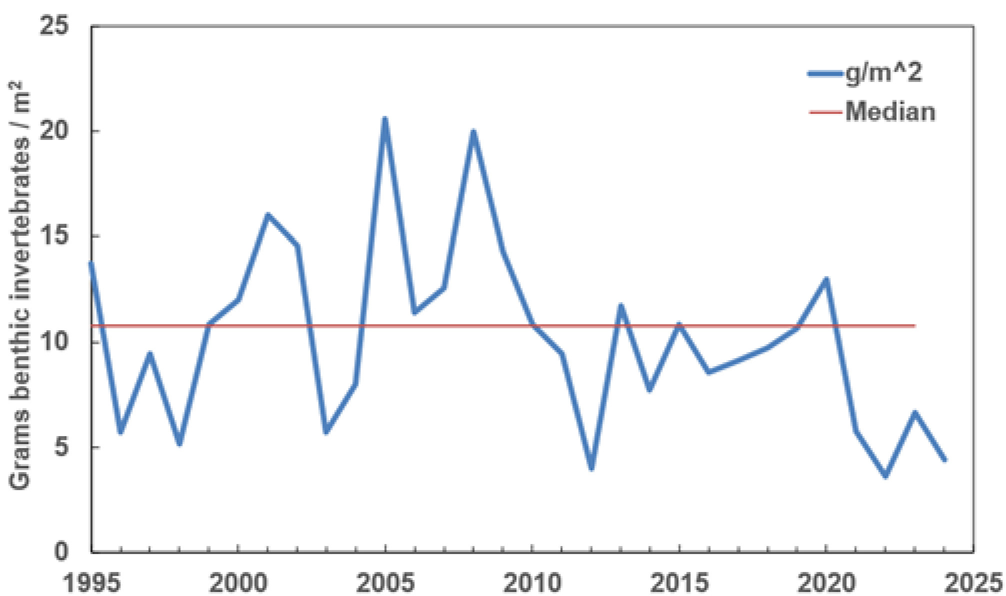

Figure 17. Trends of standardized ratios of major upper Bay forage species indices to Striped Bass relative abundance (RI). Forage ratios have been standardized to their means since 1989 to place them on the same scale. S indicates a seine survey index; T indicates a trawl survey index; and D indicates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

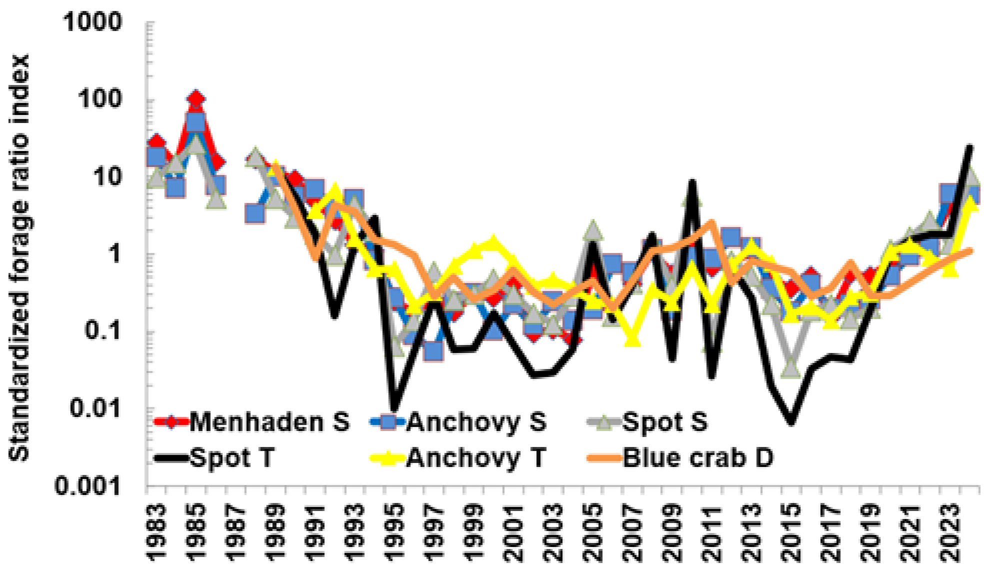

Figure 18. Atlantic Menhaden index to Striped Bass index (RI) ratios (Atlantic Menhaden FR) since 1983 and their 90% confidence intervals based on @Risk simulations of Atlantic Menhaden seine indices and RI distributions. Note log $_{10}$ scale on the Y-axis.

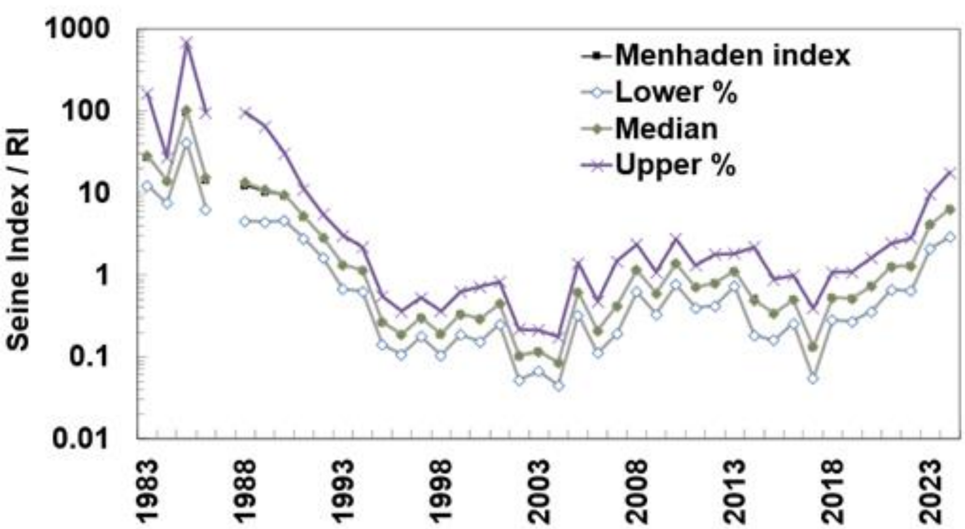

Figure 19. Bay Anchovy seine index to Striped Bass index (RI) ratios (Bay Anchovy seine FR) since 1983 and their 90% confidence intervals based on @Risk simulations of Bay Anchovy seine indices and RI distributions. Note log $_{10}$ scale on the Y-axis.

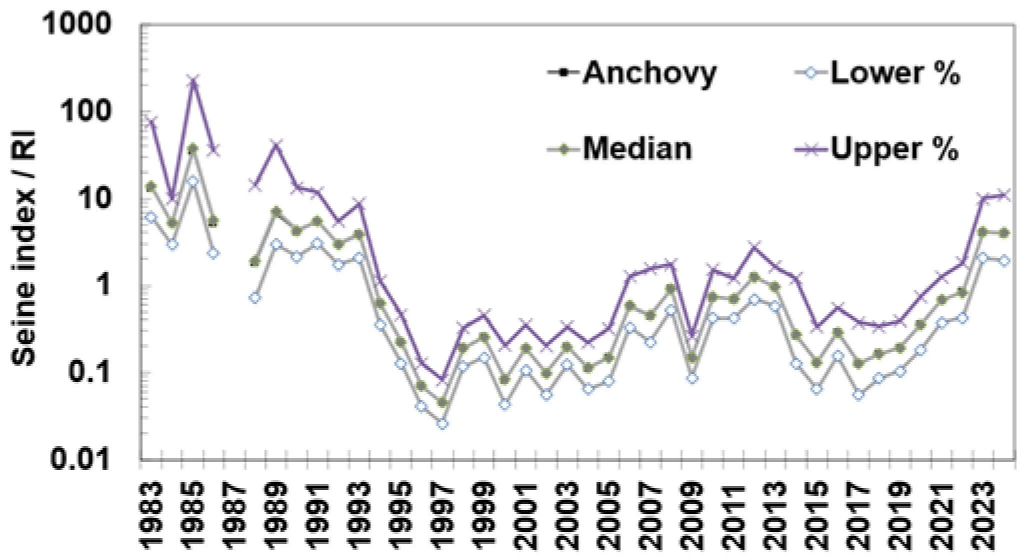

Figure 20. Bay Anchovy trawl index to Striped Bass index (RI) ratios (Bay Anchovy trawl FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note log $_{10}$ scale on the Y-axis.

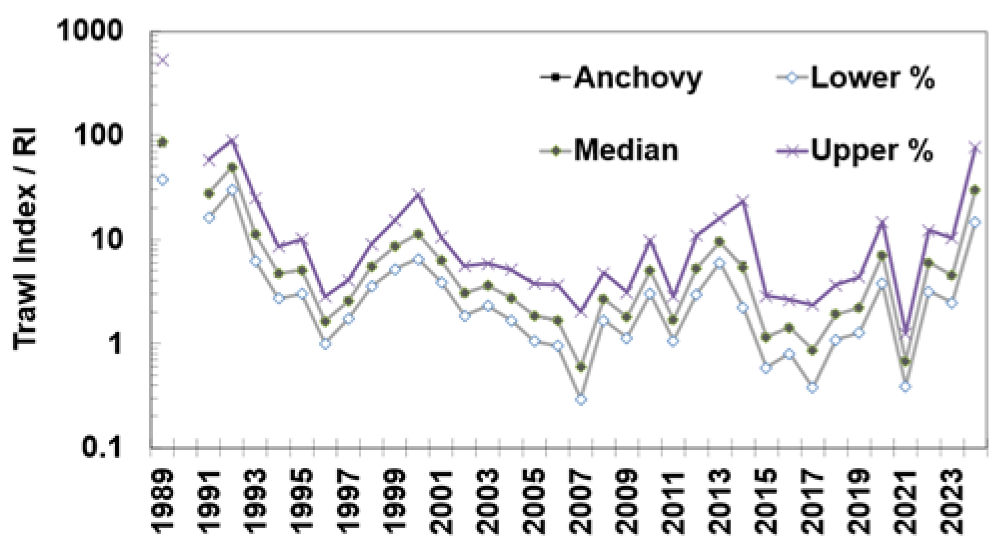

Figure 21. Spot seine index to Striped Bass index (RI) ratios (Spot seine FR) since 1983 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Spot seine indices and RI. Note log $_{10}$ scale on Y-axis.

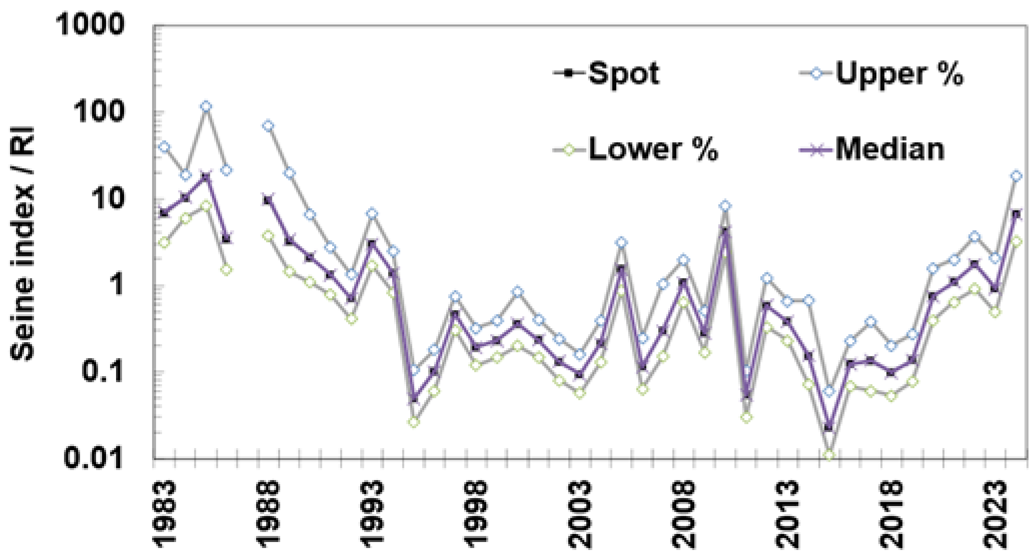

Figure 22. Spot trawl index to Striped Bass index (RI) ratios (Spot trawl FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note log $_{10}$ scale on Y-axis.

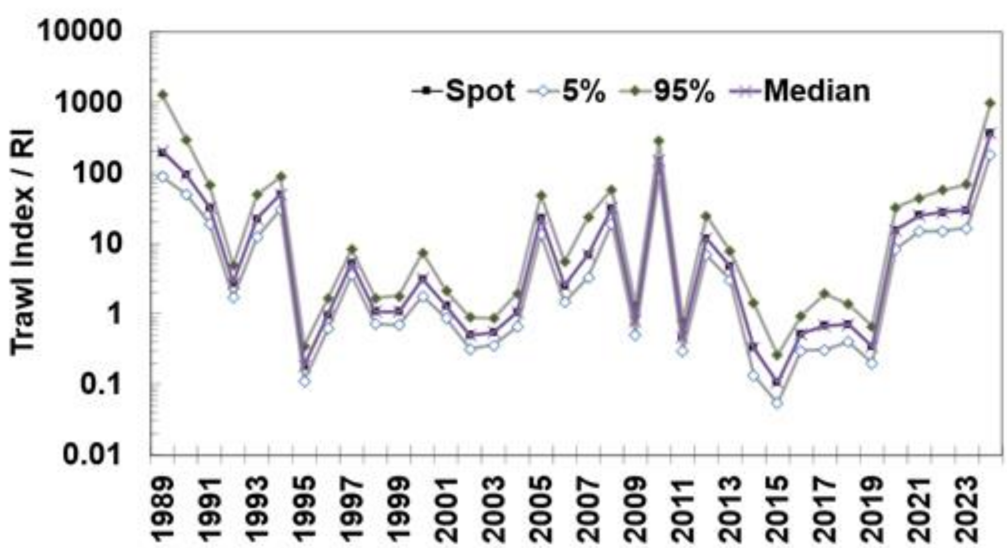

Figure 23. Blue Crab index to Striped Bass index (RI) ratios (Blue Crab FR) since 1989 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Blue Crab (age 0) winter dredge densities and RI. Note the $\log_{10}$ scale on Y-axis.

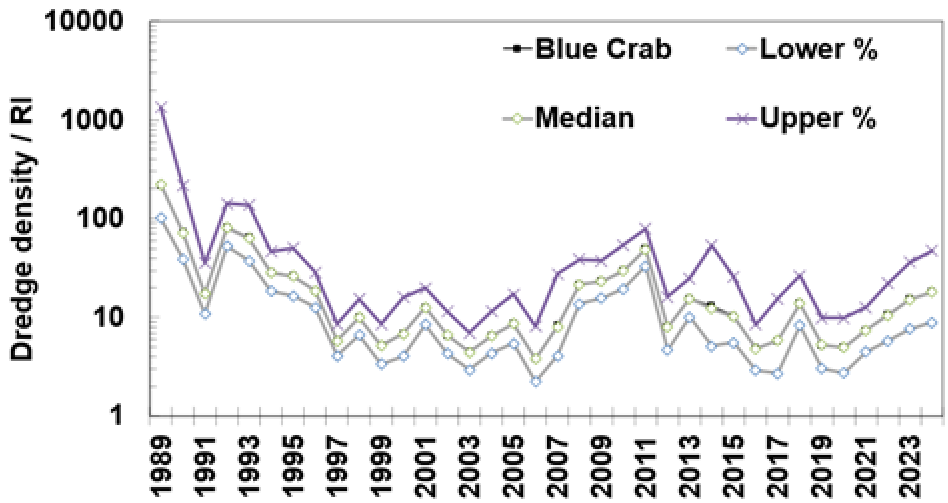

Figure 24. Time -series of age 3 Striped Bass relative abundance on two major Maryland spawning areas (hybrid index = gill net index adjusted for changing catchability during 1985-1995; units = number of fish captured in 1000 square yards of net per hour) and abundance of age 3 Striped Bass along the Atlantic Coast estimated by the ASMFC statistical catch-at-age model (SCAM). Hybrid index time series =1985-2024; SCAM time-series = 1985-2023. Unadjusted = gill net index not adjusted for catchability during 1985-1995. A hybrid index estimate was not made for 2021.

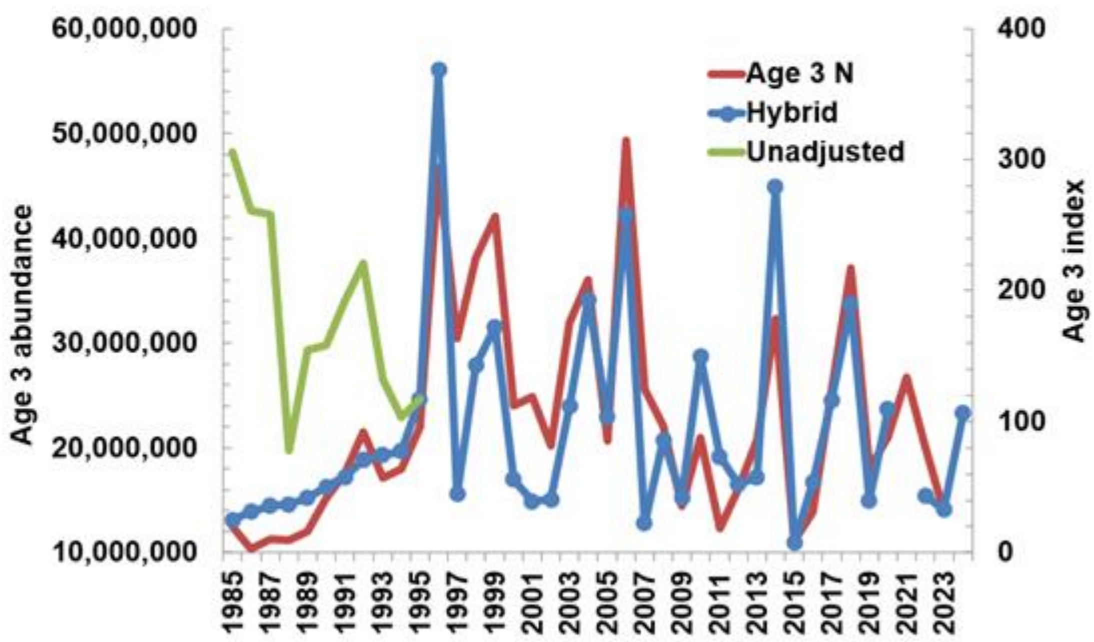

Figure 25. Relative survival (SR) of a Striped Bass year -class to approximately its third birthday during 1985-2024 and 90% confidence intervals based on @Risk simulations of age 3 hybrid gill net indices divided by juvenile index distributions. Year of estimate = year - class + 3. An estimate was not available for 2021 (2018 year -class).

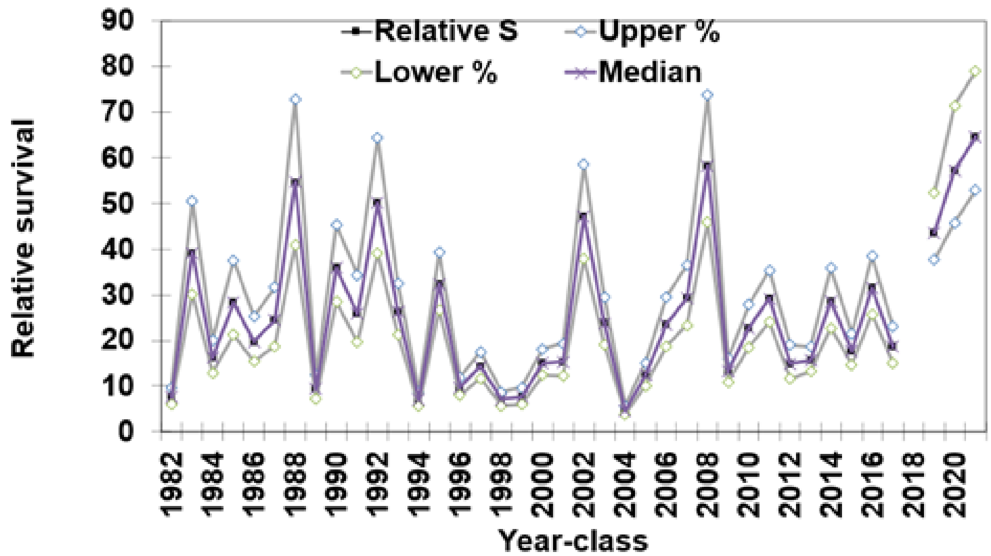

Figure 26. Relative survival (SR) of Striped Bass between ages 0 and 3, its median, and the relative abundance of resident Striped Bass (RI) in the previous year during 1985-2024 (year-class = year – 3). An estimate was not available for 2021.

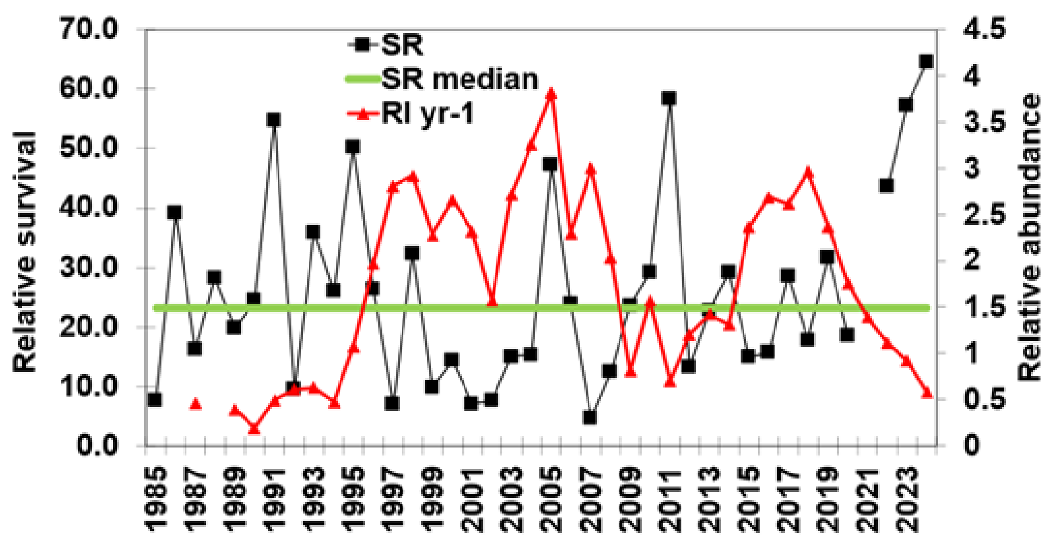

Figure 27. Traffic light index for small and large Striped Bass and forage to Striped Bass ratios during 1995-2024. Forage ratios = prey index / Striped Bass index (RI). White block (blank) = index was not available. Years with above median RI are within boxes on the year axis.

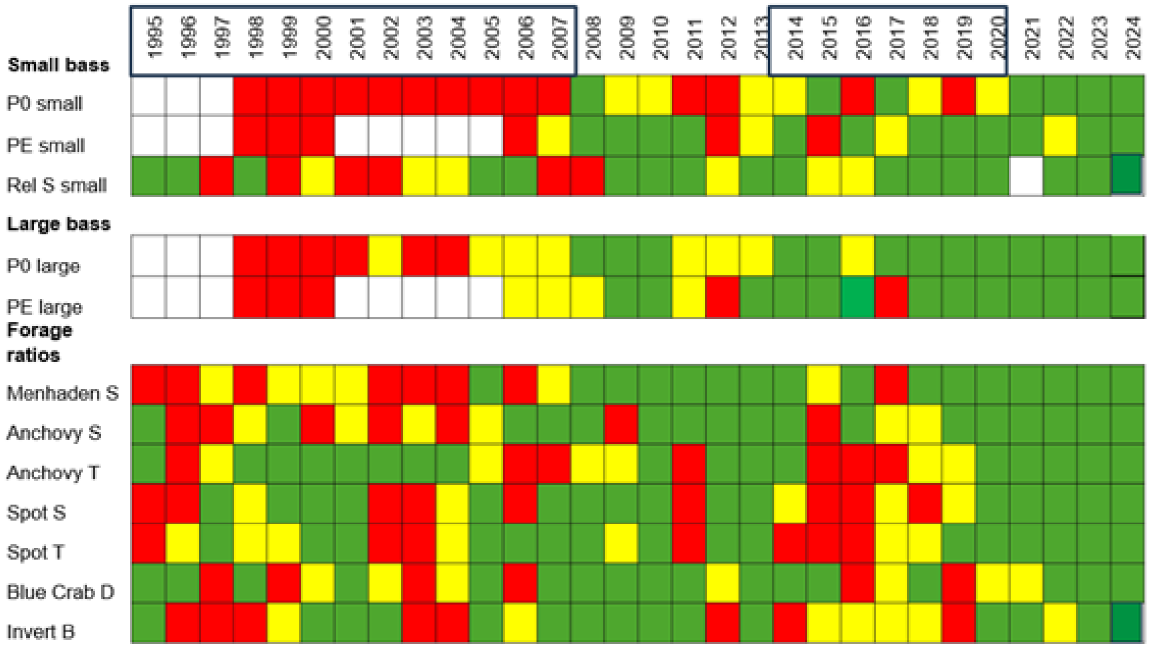

Figure 28. Proportion of small Striped Bass (< 457 mm, TL) guts without food (PE) and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions.

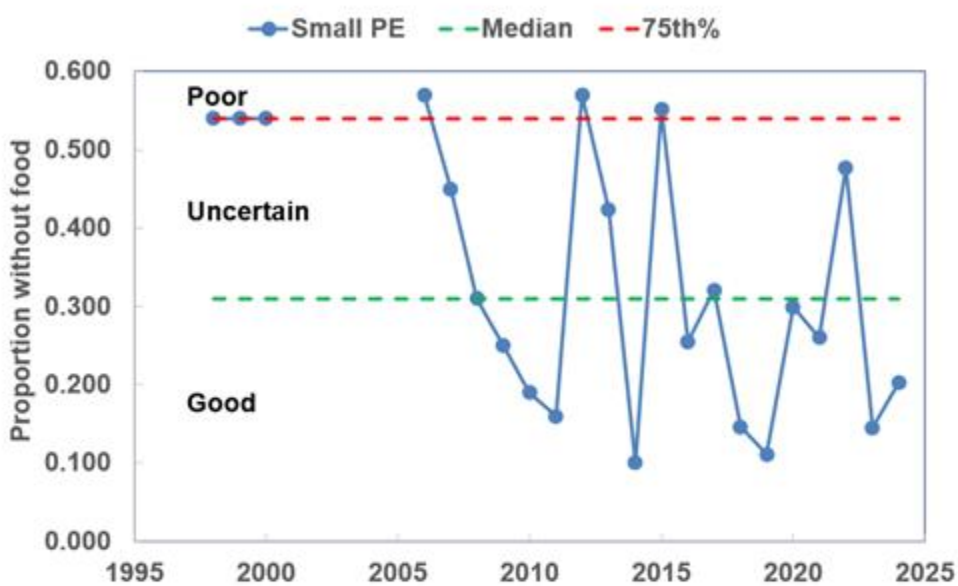

Figure 29. Proportion of large Striped Bass (≥ 457 mm, TL) guts without food and its traffic light boundaries for good, uncertain, and poor conditions; 1995-2021 is the reference period for judging conditions.

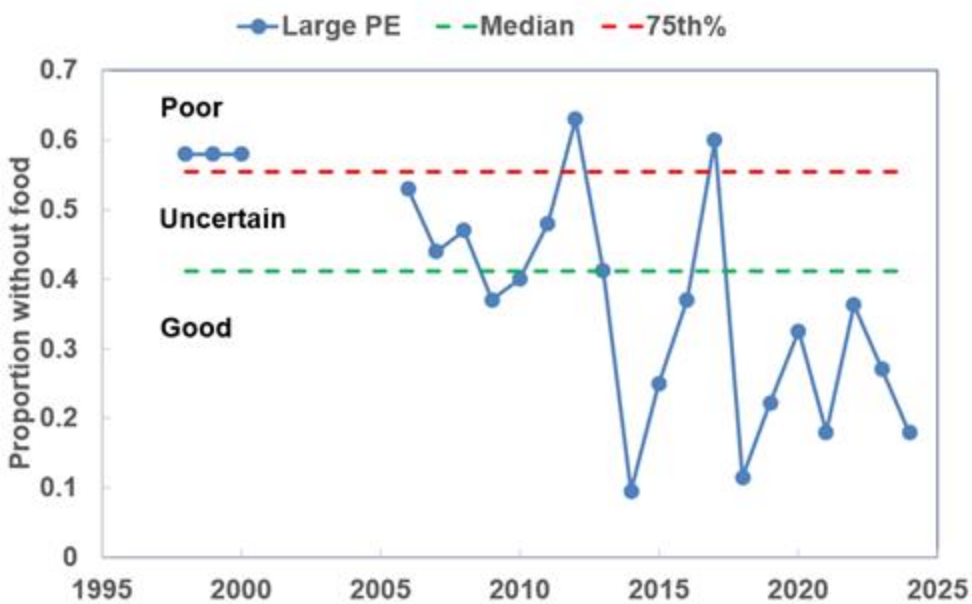

Figure 30. Proportion of small Striped Bass (< 457 mm, TL) without visible body fat (P0) since 1995 and its traffic light boundaries for good, uncertain, and poor conditions.

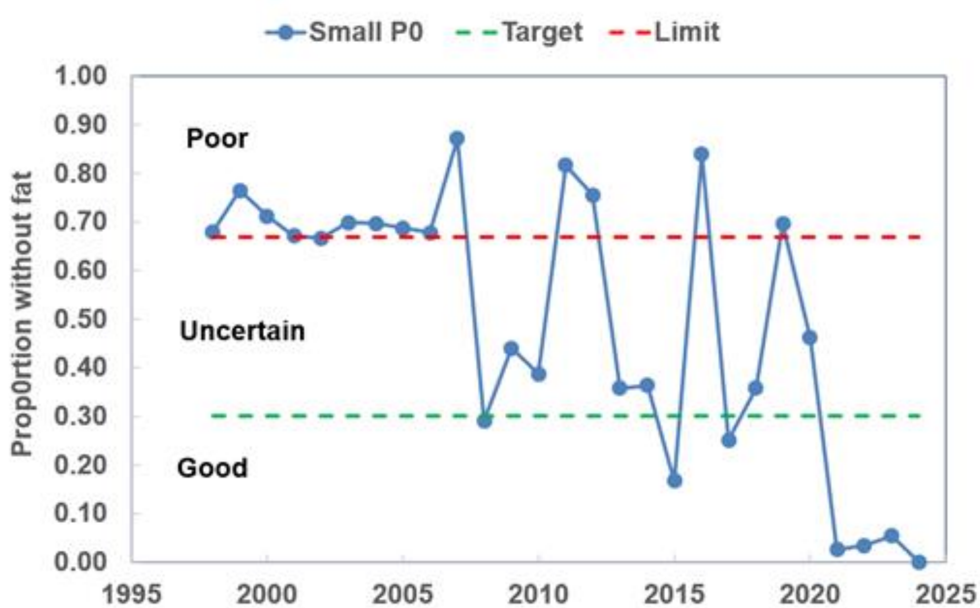

Figure 31. Proportion of large Striped Bass (≥ 457 mm, TL) without visible body fat (P0) since 1995 and its traffic light boundaries for good, uncertain, and poor conditions.

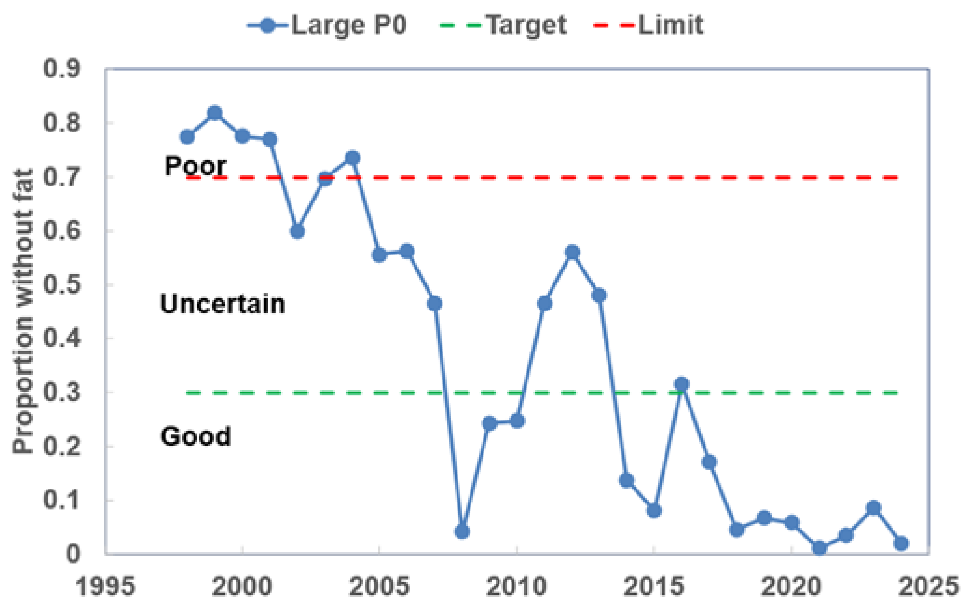

Figure 32. Relative survival Striped Bass between age 0 and age 3 since 1995 and its traffic light boundaries for good, uncertain, and poor conditions. Year -class = year – 3.

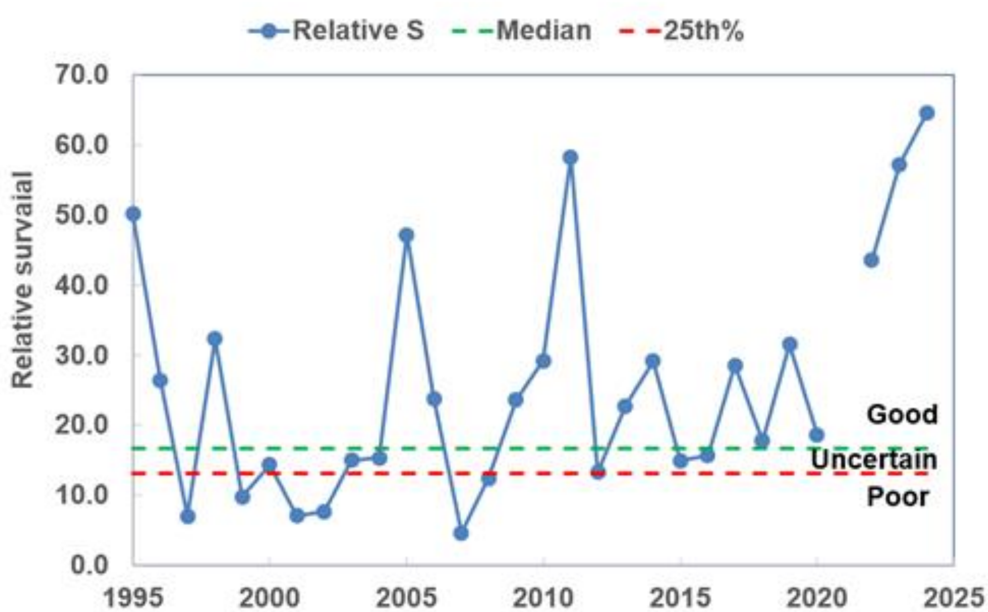

Figure 33. Atlantic Menhaden seine index to Striped Bass index (RI) ratios (Atlantic Menhaden FR) since and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions. The Atlantic Menhaden seine index (numerator) has been standardized to its 1989-2024 mean for comparability with other forage ratios. Note the log10 scale for the y-axis.

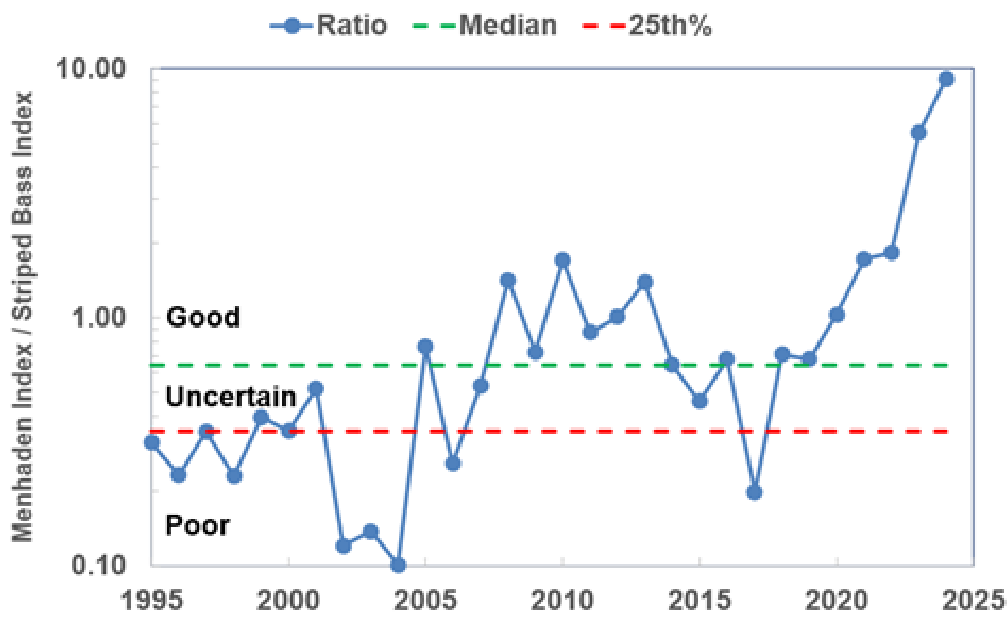

Figure 34. Bay Anchovy seine index to Striped Bass index (RI) ratios (Bay Anchovy seine FR) and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions. The Bay Anchovy seine index (numerator) has been standardized to its 1989-2024 mean for comparability with other forage ratios. Note the log10 scale for the y -axis.

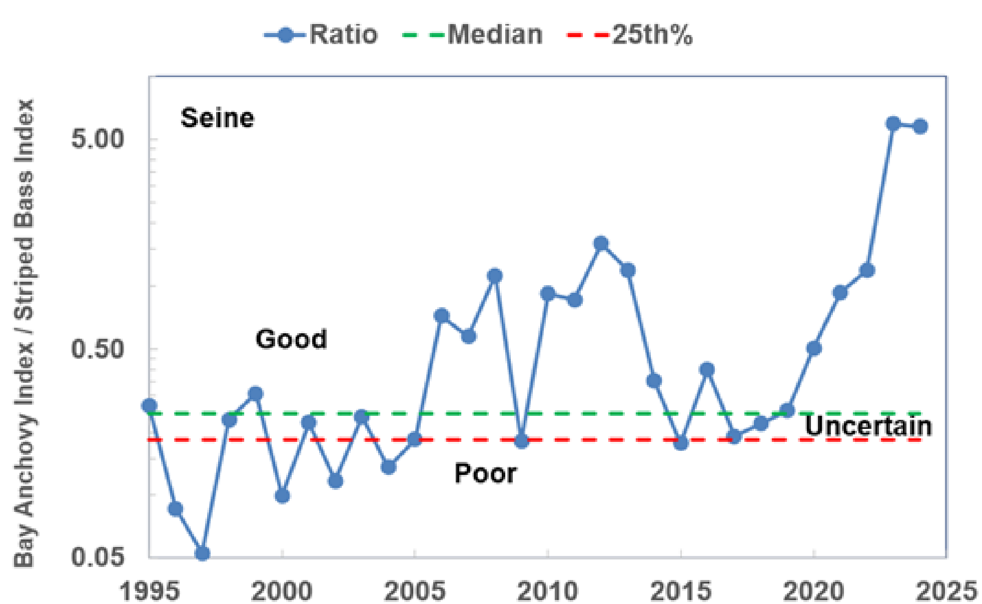

Figure 35. Bay Anchovy trawl index to Striped Bass index (RI) ratios (Bay Anchovy trawl FR) and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions. The Bay Anchovy trawl index (numerator) has been standardized to its 1989-2024 mean for comparability with other forage ratios. Note the log10 scale for the y -axis.

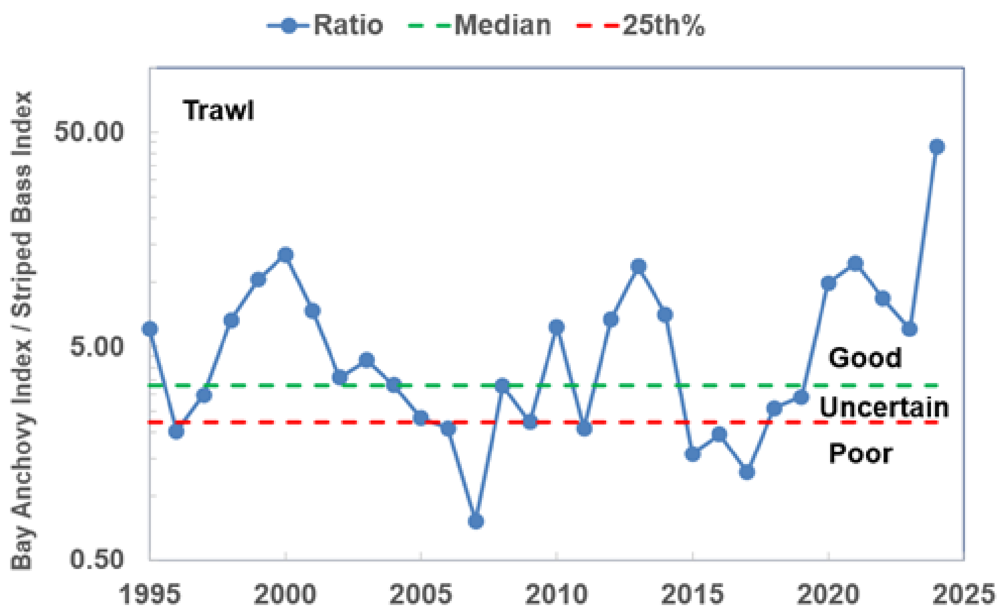

Figure 36. Spot seine index to Striped Bass index (RI) ratios (Spot seine FR) and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions. The Spot seine index (numerator) has been standardized to its 1989 -2024 mean for comparability with other forage ratios. Note the log10 scale for the y -axis.

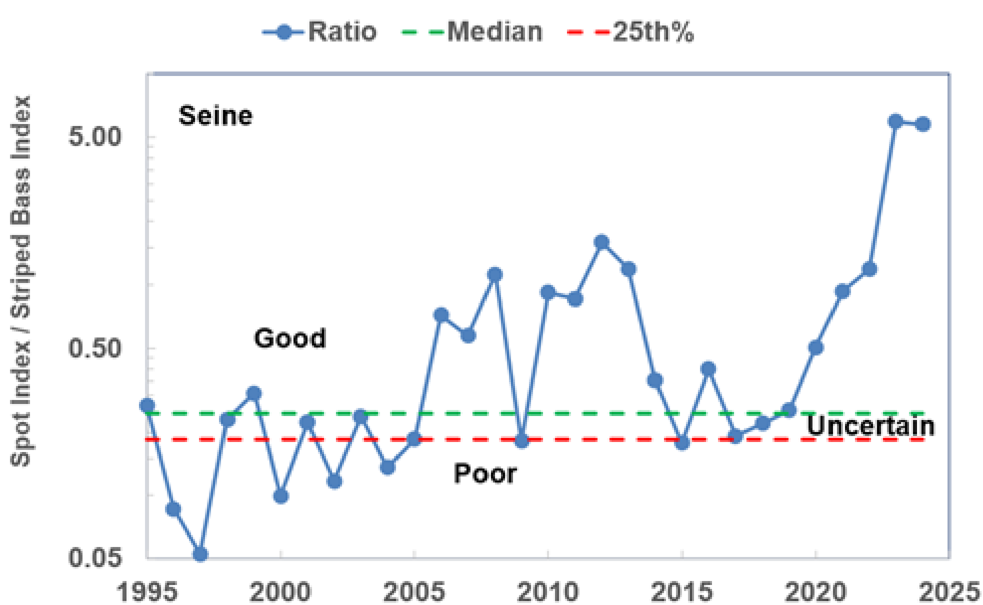

Figure 37. Spot trawl index to Striped Bass index (RI) ratios (Spot trawl FR) and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions. The Spot trawl index (numerator) has been standardized to its 1989 -2024 mean for comparability with other forage ratios. Note the log10 scale for the y -axis.

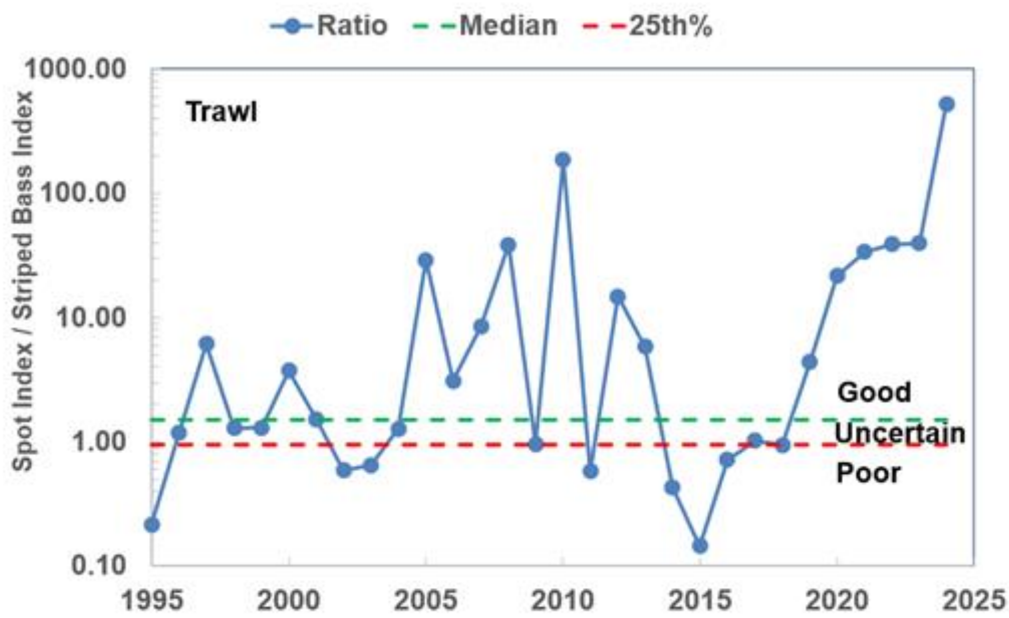

Figure 38. Blue Crab dredge index to Striped Bass index (RI) ratios (Blue Crab FR) and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions. The Blue Crab index (numerator) has been standardized to its 1989 -2024 mean for comparability with other forage ratios.

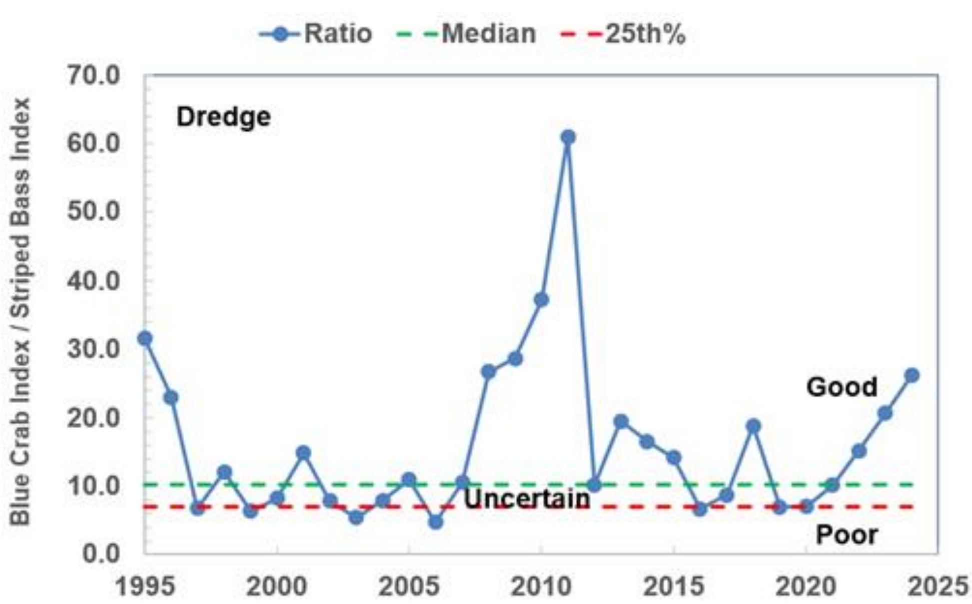

Figure 39. Benthic invertebrate biomass index to Striped Bass index (RI) ratios (Benthic FR) and its traffic light boundaries for good, uncertain, and poor conditions since 1995; 1995-2021 is the reference period for judging conditions.