PERFORMANCE REPORT FOR FEDERAL AID GRANT F-63-R, SEGMENT 11

2020

MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

Maryland Department of Natural Resources

Fishing and Boating Services

Tawes State Office Building B-2

Annapolis, Maryland 21401

January, 2022

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For today and tomorrow,

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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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This grant was funded by the State of Maryland Fisheries Management and Protection Fund

and

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

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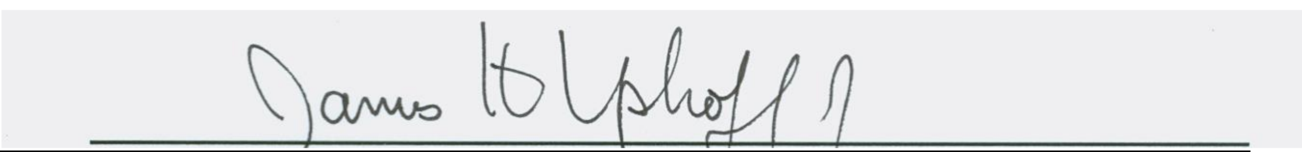

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Acknowledgements

The Maryland Department of Natural Resources and program staff would like to thank all volunteers and organizations who assisted us in 2018.

Sampling	Volunteer / organization
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Mapping	MD DNR

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This report was completed during January, 2022. It consists of summaries of activities for Jobs 1–4 under this grant cycle. All pages are numbered sequentially; there are no separate page numbering systems for each Job. Job 1 activities are reported in separate numbered sections. For example, Job 1, section 1 would cover development reference points (Job 1) for stream spawning habitat of anadromous fish (Section 1). Tables in Job 1 are numbered as section number – table number (1-1, 1-2, etc). Figures are numbered in the same fashion. Throughout the report, multiple references to past annual report analyses are referred to. The complete PDF versions of many past annual reports can be found under the Publications and Report link on the Fisheries Habitat and Ecosystem (FHEP) website page on the Maryland DNR website. The website address is http://dnr.maryland.gov/fisheries/Pages/FHEP/pubs.aspx . Table 1 provides the page number for each job and section.

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

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Maryland: 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

Executive Summary

Spatial Analyses - We used property tax map-based counts of structures in a watershed (C), standardized to hectares (C/ha), as our indicator of development. Recalculation of the equation previously used to convert annual estimates of C/ha to estimates of impervious surface (IS) was necessary in 2018 due to a new time-series provided by the Maryland Department of Planning, as well as inconsistencies found in the data for some watersheds up to 2002. New estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.37, 0.86, and 1.35 C/ha, respectively. Previous C/ha estimates corresponding to 5%, 10%, and 15% IS were 0.27, 0.83, and 1.59, respectively. Maryland Department of Planning estimates of percent of watershed in agriculture, forest, and wetlands in 2010 were used for other land use categories.

Section 1, Stream Ichthyoplankton - Spring sampling for stream ichthyoplankton was not conducted in 2020 due to restrictions imposed related to the ongoing COVID-19 pandemic, so there are no data to report.

Section 2: Investigation of estuarine spawning and larval habitat status of anadromous fish in Maryland - Sampling was not conducted for Section 2 in 2020 due to COVID-19. Since there were no data to enter or analyze for these sections, 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 estuarine reaches (primarily tidal-fresh) of Maryland's Chesapeake Bay tributaries. These data can be used to refine habitat maps and conduct investigations of changes in habitat use. Approximately 14,000 lines of data were entered, covering 67 years and 20 different Chesapeake Bay tributaries in Maryland. Most of these reports focused on Striped Bass, but information on other target species or species groups (i.e., Herring) were sometimes available in tables and appendices that allowed data to be traced to a per sample basis.

Section 3 - Estuarine Fish Community Sampling - The choice of subestuaries sampled during summer 2020 was influenced by State of Maryland travel policies imposed due to COVID-19; only one biologist could drive the state vehicle that towed the boat and remaining survey crew typically used their own vehicles. The subestuaries chosen had resource issues of interest and minimized driving.

In spite of similar land uses in their watersheds (primarily agriculture), habitat conditions varied widely in the three Talbot County subestuaries surveyed in 2020. In Tred Avon River, a watershed approaching the development threshold, numerous measurements of bottom dissolved oxygen (DO) were below target (5.0 mg/L) and threshold (3.0 mg/L) DO at station 01 (Easton), indicating development as its root cause. The two rural, primarily agricultural, watersheds sampled had strongly contrasting DO conditions. Broad Creek median bottom DO was within previous years' ranges and no threshold violations were recorded; however, all stations fell below the time-series median for the first time. Poor bottom DO was observed at all stations in Miles River during 2020. Other water quality metrics (pH, salinity, and Secchi depth) sampled during 2020 were within previous years' ranges for Broad Creek, Miles River, and Tred Avon River.

Finfish catches in trawls sampling bottom water habitat in the Talbot County subestuaries in 2020 were the lowest among all sampling years and particularly low in Miles River. Species composition changed slightly. Atlantic Croaker were present throughout most of the Talbot County subestuaries sampled; Spot and Hogchoker presence was widespread in 2020. Bay Anchovy remained in the top 90% of species in Broad Creek and Miles River, but were not prevalent enough in the Tred Avon River to avoid being grouped into the other species category. White Perch GMs in the Choptank River tributaries indicated a modest population in 2020; highly variable fluctuations in White Perch populations have been observed in previous years. Modified Proportional Stock Densities (PSDs) for trawl and seine samples located in the Choptank River tributaries sampled in 2020 had greater population densities of White Perch of interest to anglers. Inshore seine catches remained steady, although not at the highest levels previously observed.

We sampled Sassafras River, a fresh-tidal subestuary with a predominately agricultural watershed, in 2020. However, due to a harmful algal bloom (HAB) during sampling season, we were unable to collect finfish data from inshore by seining. Seine data acquired from Juvenile Index monitoring station just upstream of our sampling area indicated higher species richness inshore. Harmful algal blooms appear to be a major negative habitat feature of low salinity subestuaries in the Head-of-Bay region; HABS have occurred in Gunpowder (2004 and 2017), Middle River (2015), and Sassafras River (2018 and 2020), that have greatly contrasting dominant land uses (urban in Gunpowder and Middle, agricultural in Sassafras; MDE 2020). These HABs were not accompanied by depleted DO in any of our surveys. Finfish composition in Sassafras River was comparable to the other Head-of-Bay subestuaries with White Perch predominating. The Head-of-Bay subestuaries we have sampled were primarily habitat for smaller White Perch and modified PSDs were very low. Other species that are notable in the Sassafras River during 2020 were Spot and Channel Catfish; Spot relative abundance in our surveys was the strongest since 2010.

Common Background for Project 1, Sections 1-3

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

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

Forests and wetlands in the Chesapeake Bay watershed have been converted to agriculture and residential areas to accommodate increased human populations since colonial times (Brush 2009). These watershed alterations have affected major ecological processes and have been most visibly manifested in Chesapeake Bay eutrophication, hypoxia, and anoxia

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

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

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

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

Impact of development on estuarine systems has not been well documented, but measurable adverse changes in physical and chemical characteristics and living resources have occurred at IS of 10-30% (Mallin et al. 2000; Holland et al. 2004; Uphoff et al. 2011; Seitz et al. 2016). Habitat reference points based on IS have been developed (ISRPs) for Chesapeake Bay estuarine watersheds (Uphoff et al. 2011). They provide a quantitative basis for managing fisheries in increasingly urbanizing Chesapeake Bay watersheds and enhance communication of limits of fisheries resources to withstand development-related habitat changes to fishers, land-use planners, watershed-based advocacy groups, developers, and elected officials (Uphoff et al. 2011; Interagency Mattawoman Ecosystem Management Task Force 2012). These guidelines have held for Herring stream spawning, Yellow Perch larval habitat (they are incorporated into the current Maryland's tidal Yellow Perch management plan; MD DNR 2017), and summer habitat in tidal-fresh subestuaries (Uphoff et al. 2015). Conserving watersheds at or below 5% IS would be a viable fisheries management strategy. Increasingly stringent fishery regulation might compensate for habitat stress as IS increases from 5 to 10%. Above a 10% IS threshold, habitat stress mounts and successful management by harvest adjustments alone becomes unlikely (Uphoff et al. 2011; Interagency Mattawoman Ecosystem Management Task Force 2012; Uphoff et al. 2020). We have estimated that impervious surface in Maryland's portion of the Chesapeake Bay watershed will exceed 10% by 2020; a preliminary estimate of IS in 2018 equaled 9.3%. We expect adverse habitat conditions for important forage and gamefish to worsen with future growth. Managing this growth with an eye towards conserving fish habitat is important to the future of sportfishing in Maryland.

We now consider tax map derived development indices as the best source for standardized, readily updated, and accessible watershed development indicators in Maryland and have development targets and thresholds based on it that are the same as ISRPs (Uphoff et al. 2020; Topolski 2015). Counts of structures per hectare (C/ha) had strong relationships with IS in years when all were estimated (1999-2000; Uphoff et al. 2020). Tax map data can be used as the basis for estimating target and threshold levels of development in Maryland and these estimates can be converted to IS. Estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.37, 0.86, and 1.35 C/ha, respectively. Tax map data provide a development time-series that goes back to 1950, making retrospective analyses possible (Uphoff et al. 2020).

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

Agricultural pesticides and fertilizers were thought to be potential sources of toxic metals implicated in some episodic mortality of Striped Bass larvae in Bay spawning tributaries in the early 1980s (Uphoff 1989; 1992; Richards and Rago 1999; Uphoff 2008; Uphoff et al. 2020). A correlation analysis of Choptank River watershed agricultural best management practices (BMPs) and estimates of postlarval survival during 1980-1990 indicated that as many as four BMPs were positively associated with survival (Uphoff 2008). Two measures that accounted for the greatest acreage, conservation tillage and cover crops, were strongly associated with increased postlarval survival (r = 0.88 and r = 0.80, respectively). These correlations cannot explain whether toxicity was lowered by BMPs, but it is possible that reduced contaminant runoff was a positive byproduct of agricultural BMPs aimed at reducing nutrients (Uphoff 2008).

Agriculturally derived nutrients have been identified as the primary driver of hypoxia and anoxia in the mainstem Chesapeake Bay (or Bay; Hagy et al. 2004; Kemp et al. 2005; Fisher et al. 2006; Brush 2009). Hypoxia is also associated with transition from rural to suburban landscapes in brackish Chesapeake Bay subestuaries (Uphoff et al. 2011). Hypoxia's greatest impact on gamefish habitat occurs during summer when its extent is greatest, but hypoxic conditions are present at lesser levels during spring and fall (Hagy et al. 2004; Costantini et al. 2008). Episodic hypoxia may elevate catch rates in various types of fishing gears by concentrating fish at the edges of normoxic waters, masking associations of landings and hypoxia (Kraus et al. 2015).

Habitat loss due to hypoxia in coastal waters is often associated with fish avoiding DO that reduces growth and requires greater energy expenditures, as well as lethal conditions (Breitburg 2002; Eby and Crowder 2002; Bell and Eggleston 2005). There is evidence of cascading effects of low DO on demersal fish production in marine coastal systems through loss of invertebrate populations on the seafloor (Breitburg et al. 2002; Baird et al. 2004). A long-term decline in an important Chesapeake Bay pelagic forage fish, Bay Anchovy, may be linked to declining abundance of the common calanoid copepod *Acartia tonsa* in Maryland's portion of Chesapeake Bay that, in turn, may be linked to rising long-term water temperatures and eutrophication that drive hypoxia (Kimmel et al. 2012; 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 *M* and deteriorating condition in Chesapeake Bay through a temperature-oxygen squeeze (mismatch of water column regions of desirable temperature and dissolved oxygen in stratified Chesapeake Bay during summer; Coutant 1985; Price et al. 1985; Coutant 1990; Coutant 2013). Constantini et al. (2008); Kraus et al. 2015; Itakura et al. 2021) examined the impact of hypoxia on 2 year-old and older Striped Bass in Chesapeake Bay through bioenergetics modeling and acoustic tagging and concluded that a temperature-oxygen squeeze by itself was not limiting for Striped Bass. However, Groner et al (2018) suggested that Striped Bass are living at their maximum thermal tolerance and that this is driving increased mycobacteriosis and associated mortality. Adequate levels of Striped Bass prey can offset negative effects of warm temperatures and suboptimal dissolved oxygen in reservoirs (Thompson et al. 2010; Coutant 2013).

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General spatial and analytical methods used in Project 1

Spatial Methods - We used property tax map-based counts of structures in a watershed, standardized to hectares (C/ha), as our indicator of development (Uphoff et al. 2012; Topolski 2015). This indicator was estimated by M. Topolski (MD DNR). Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MD DOP 2019). All tax data were organized by county. Since watersheds straddle political boundaries, one statewide tax map was created for each year of available tax data, and then subdivided into watersheds. Maryland's tax maps are updated and maintained electronically as part of MD DOP's GIS database. Files were managed and geoprocessed in ArcGIS 10.3.1 from Environmental Systems Research Institute (ESRI 2015). All feature datasets, feature classes, and shapefiles were spatially referenced using the NAD_1983_StatePlane_Maryland_FIPS_1900 projection to ensure accurate feature overlays and data extraction. ArcGIS geoprocessing models were developed using Model Builder to automate assembly of statewide tax maps, query tax map data, and assemble summary data. MdProperty View tax data are annually updated by each Maryland jurisdiction to monitor the type of parcel development for tax assessment purposes, although there is typically a two-year lag in processing by MD DOP. Tax data through 2018 were available for the 2020 report. To create watershed land tax maps, each year's statewide tax map was clipped using the MD 8-digit watershed boundary file; estuarine waters were excluded. These watershed tax maps were queried for all parcels having a structure built from 1700 to the tax data year. A large portion of parcels did not have any record of year built for structures, but consistent undercounts should not have presented a problem since we were interested in the trend and not absolute magnitude.

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

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

A linear regression described the updated relationship well:

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

New estimates of C/ha that were equivalent to 5% IS (target level of development for fisheries; a rural watershed), 10% IS (development threshold for a suburban watershed), and 15% IS (highly developed suburban watershed) were estimated as 0.37, 0.86, and 1.35 C/ha, respectively. The previous C/ha estimates, based on a nonlinear power function, corresponding to 5%, 10%, and 15% IS were 0.27, 0.83, and 1.59, respectively (Uphoff et al. 2018).

Percent of watershed in agriculture, forest, and wetlands were estimated from MD DOP spatial data. The MD DOP forest cover estimates have a minimum mapping unit of 10 acres that mixes forest cover in residential areas (trees over lawns) with true forest cover, clouding interpretation of forest influence (R. Feldt, MD DNR Forest Service, personal communication). An urban category was available as well, but was not featured in many subsequent analyses since we have adopted C/ha as our preferred index of development. Urban land consisted of high and low density residential, commercial, and institutional acreages and was not a direct measure of IS.

Land use and land cover (LULC) shapefiles were available for 1973, 1994, 1997, 2002, and 2010 for each Maryland jurisdiction and as an aggregated statewide file. Metadata for the LULC categories is available for download from MD DOP. The statewide LULC shapefiles were clipped using boundary shapefiles for each watershed of interest. Once clipped, polygon geometry was recalculated. Polygons designated as water were omitted when calculating watershed area; that is only land was considered when calculating the ratio of LULC for each category. For each LULC category, polygons were queried and its land area in hectares was calculated. The land use total was divided by the watershed total to the nearest tenth of a hectare and multiplied by 100%.

Statistical Analyses – A combination of correlation analysis, plotting of data, and curve-fitting was commonly used to explore trends among land use types (land that was developed or in agriculture, forest, or wetland) and among fish habitat responses. Typical fish habitat responses were the proportion of stream samples with Herring eggs and-or larvae (P_{herr}; Section 1); proportion of subestuary samples with Yellow Perch larvae (L_p; Section 2); or subestuary bottom dissolved oxygen, fish presence-absence or relative abundance, and fish diversity in summer (Section 3).

Correlations among watershed estimates of C/ha and percent of watershed estimated in urban, agriculture, forest, and wetland based on MD DOP spatial data were used to describe associations among land cover types. These analyses explored (1) whether C/ha estimates were correlated with another indicator of development, percent urban and (2) general associations among major landscape features in our study watersheds. Scatter plots were inspected to examine whether nonlinear associations were possible. Land use was assigned from MD DOP estimates for 1973, 1994, 1997, 2002, or 2010 that fell closest to a sampling year. We were particularly interested in knowing whether these land uses might be closely correlated enough (r greater than 0.80; Ricker 1975) that only one should be considered in analyses of land use and L_p and P_{herr}. We further examined relationships using descriptive models as a standard of comparison (Pielou 1981). Once the initial associations and scatter plots were examined, linear or nonlinear regression analyses (power, logistic, or Weibull functions) were used to determine the general shape of trends among land use types. This same strategy was pursued for analyses of land use and L_p or P_{herr}. Level of significance was reported, but potential management and biological significance took precedence over significance at $P < 0.05$ (Anderson et al. 2000; Smith 2020). We classified correlations as strong, based on $r \geq 0.80$; weak correlations were indicated by $r < 0.50$; and moderate correlations fell in between. Relationships indicated by regressions were considered strong at $r^2 \geq 0.64$; weak relationships were indicated by $r^2 \leq 0.25$; and moderate relationships fell in between. Confidence intervals (95% CIs were standard output) of the model parameters for each indicator species were estimated to examine whether parameters were different from 0 (Freund and Littell 2006). If parameter estimates were often not different from 0, rejection of the model was considered. Residuals of regressions were inspected for trends, non-normality, and need for additional terms. A general description of equations used follows, while more specific applications will be described in later sections.

Linear regressions described continuous change in variable Y as X changed:

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

where m is the slope and b is the Y-intercept (Freund and Littel 2006). Multiple regression models accommodated an additional variable (Z):

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

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

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

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

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

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

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

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

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

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

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

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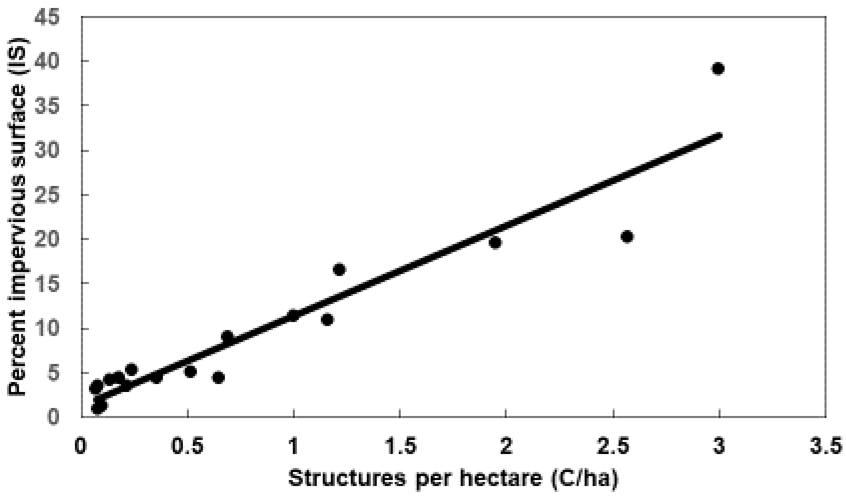
Table 1. Structures per hectare (C/ha) and percent impervious surface estimates (IS) used to estimate the relationship for predicting IS from C/ha. Old C/ha were estimates used previous to this report and New C/ha were revised estimates used to estimate the current relationship.

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

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



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

Carrie Hoover and Jim Uphoff

Section 1: Stream Ichthyoplankton Sampling

Spring sampling for stream ichthyoplankton was not conducted in 2020 due to restrictions imposed related to the ongoing COVID-19 pandemic, so there are no data to report.

Section 2: Investigation of estuarine spawning and larval habitat status of anadromous fish in Maryland

Sampling was also not conducted for Section 2 in 2020 due to COVID-19. Since there were no data to enter or analyze for these sections, 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 estuarine reaches (primarily tidal-fresh) of Maryland's Chesapeake Bay tributaries. These data can be used to refine habitat maps and conduct investigations of changes in habitat use. In total, approximately 14,000 lines of data were entered, covering 67 years and 20 different Chesapeake Bay tributaries in Maryland (Figure 1).

Most of these reports focused on Striped Bass, but information on other species were sometimes available in tables and appendices that allowed data to be traced to a per sample basis. Reports used were Boynton et al. 1977; Burton et al. 1996; Burton et al. 1985; Houde et al. 1988a; Houde et al. 1988b; Houde and Rutherford 1992; Houde et al. 1990; Houde et al. 1996; Mihursky et al. 1974; Mihursky et al. 1976; Otto and Peterson 1980; Portner and Kohlenstein 1979; Secor et al. 1994; Setzler-Hamilton et al. 1980a; Setzler-Hamilton et al. 1980b; Setzler et al. 1979; and Stroup et al. 1991, and PDFs are available for all reports referenced. Historical data was also obtained from physical copies or PDF files of scanned Maryland DNR field and lab data sheets (1953-2011; see Table 2.1.2 in Uphoff et al. 2020).

Water quality variables entered from historical reports and field data sheets included year, river, station, date, time, temperature, salinity, tidal stage, dissolved oxygen, pH, and conductivity (Table 1). Water quality parameters available varied, but were generally confined to temperature (˚C), salinity (‰), and tide stage until the early 1980s. During the 1980s and after, dissolved oxygen (DO; mg/L), pH, and conductivity (µS/cm) were monitored routinely (Table 1).

Counts or presence-absence of Striped Bass eggs and larvae, Yellow Perch eggs and larvae, Clupeidae eggs and larvae, and White Perch eggs and larvae were also entered (Table 2). In general, surveys prior to the early 1990s had count data for Striped Bass eggs and larvae, and presence-absence for other species. Count data are easily converted to presence-absence. If PDFs of scanned historical data sheets were blurry, or writing was too light for counts to be read clearly, presence or absence was recorded. After 1994, surveys were based on presence-absence only (Table 2).

Approximate locations of stations were developed from maps or descriptions of sites sampled (i.e., location names, statute or nautical miles, or kilometers from the mouth that were reported). Station locations and numbering changed over the 65 year span of sampling, and while georeferencing of these sites is ongoing, this work has not been completed. Additional reports are being obtained and these data will be added as they become available.

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Table 1. Summary of years, spawning areas, and variables entered from historical and present Striped Bass spawning studies. Temperature = ºC, salinity = ppt (‰), Tide = tidal stage, DO = dissolved oxygen (mg/L), and conductivity = µS/cm. "Historical data sheets" indicate actual field or lab records were available, while "Historical Excel spreadsheet" indicates this data had been entered previously and hard copies (or pdf scans) were not available for comparison or additional information. All historical data was collected by MD DNR. An "x" indicates that data was available for that system in that year.

Year	Spawning area	Temperature	Salinity	Tide	DO	pH	Conductivity	Reference
1953	Patuxent	x	x	x				Historical data sheets
1954	Blackwater	x	x	x				Historical data sheets
	Choptank	x	x	x				Historical data sheets
	Elk	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Patuxent	x	x	x				Historical data sheets
	Pocomoke	x	x	x				Historical data sheets
	Potomac	x	x	x				Historical data sheets
	Susquehanna	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Wicomico	x	x	x				Historical data sheets
1955	Blackwater	x	x	x				Historical data sheets
	Bohemia	x	x	x				Historical data sheets
	Chester	x	x	x				Historical data sheets
	Elk	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Northeast	x	x	x				Historical data sheets
	Patuxent	x	x	x				Historical data sheets
	Potomac	x	x	x				Historical data sheets
	Sassafras	x	x	x				Historical data sheets
	Susquehanna	x	x	x				Historical data sheets
1956	Annemessex	x	x	x				Historical data sheets
	Blackwater	x	x	x				Historical data sheets
	Bohemia	x	x	x				Historical data sheets
	Elk	x	x	x				Historical data sheets
	Manokin	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Northeast	x	x	x				Historical data sheets
	Pocomoke	x	x	x				Historical data sheets
	Potomac	x	x	x				Historical data sheets
	Sassafras	x	x	x				Historical data sheets
	Susquehanna	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Upper Bay		x	x				Historical data sheets

Table 1 cont.

Year	Spawning area	Temperature	Salinity	Tide	DO	pH	Conductivity	Reference
1957	Blackwater	x	x	x				Historical data sheets
	Chester	x	x	x				Historical data sheets
	Chicamacomico	x	x	x				Historical data sheets
	Choptank	x	x	x				Historical data sheets
	Elk			x				Historical data sheets
	Manokin	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Pocomoke	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Upper Bay	x	x	x				Historical data sheets
	Wicomico	x	x	x				Historical data sheets
1958	Blackwater	x	x	x				Historical data sheets
	Upper Bay	x	x	x				Historical data sheets
	Choptank	x	x	x				Historical data sheets
	Elk	x	x	x				Historical data sheets
	Manokin	x	x					Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Pocomoke	x	x	x				Historical data sheets
	Sassafras	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Wicomico	x	x	x				Historical data sheets
1959	Blackwater	x	x	x				Historical data sheets
	Chicamacomico	x	x	x				Historical data sheets
	Choptank	x	x	x				Historical data sheets
	Manokin	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Patuxent	x	x	x				Historical data sheets
	Pocomoke	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Wicomico	x	x	x				Historical data sheets
1960	Blackwater	x	x	x				Historical data sheets
	C&D Canal	x	x	x				Historical data sheets
	Upper Bay	x	x	x				Historical data sheets
	Choptank	x	x	x				Historical data sheets
	Manokin	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Pocomoke	x	x	x				Historical data sheets
	Sassafras	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Wicomico	x	x	x				Historical data sheets
1961	Blackwater	x	x	x				Historical data sheets
	Bohemia	x	x	x				Historical data sheets
	Upper Bay	x	x	x				Historical data sheets
	Choptank	x	x	x				Historical data sheets
	Elk	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Northeast	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets

Table 1 cont.

Year	Spawning area	Temperature	Salinity	Tide	DO	pH	Conductivity	Reference
1962	Blackwater	x	x	x				Historical data sheets
	Choptank	x	x	x				Historical data sheets
	Elk	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Wicomico	x	x	x				Historical data sheets
1963	Elk	x	x					Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Upper Bay	x	x					Historical data sheets
1964	Nanticoke	x	x	x				Historical data sheets
1965	Nanticoke	x	x	x				Historical data sheets
1966	Nanticoke	x	x	x				Historical data sheets
1967	Blackwater	x	x	x				Historical data sheets
	Chicamacomico	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
	Transquaking	x	x	x				Historical data sheets
	Tuckahoe	x	x	x				Historical data sheets
1968	Nanticoke	x	x	x				Historical data sheets
1969	Nanticoke	x						Historical data sheets
1970	Manokin	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
1971	Nanticoke	x	x	x				Historical data sheets
	Potomac	x	x	x				Historical data sheets
1972	Nanticoke	x	x	x				Historical data sheets
1973	Nanticoke	x	x	x				Historical data sheets
	Potomac	x	x	x				Historical data sheets
1974	Nanticoke	x	x	x				Historical data sheets
	Potomac	x	x		x			Mihursky et al. 1974
1975	Nanticoke	x	x	x				Historical data sheets
	Potomac	x	x		x			Mihursky et al. 1976
1976	Bohemia							Otto and Peterson 1980
	Elk							Otto and Peterson 1980
	Nanticoke	x	x					Historical data sheets
	Northeast							Otto and Peterson 1980
	Potomac	x	x		x			Boynton et al. 1977
	Sassafras							Otto and Peterson 1980
1977	Nanticoke	x	x	x				Historical data sheets
	Potomac	x	x		x			Setzler-Hamilton et al. 1980a
1978	Nanticoke	x	x	x				Historical data sheets
	Nanticoke (Vienna)	x	x					Portner and Kohlenstein 1979
	Patuxent	x	x					Setzler et al. 1979
1979	Nanticoke	x	x	x				Historical data sheets
	Patuxent	x	x					Setzler-Hamilton et al. 1980b
1980	Choptank	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets
1981	Choptank	x	x	x				Historical data sheets
	Nanticoke	x	x	x				Historical data sheets

Table 1 cont.

Year	Spawning area	Temperature	Salinity	Tide	DO	pH	Conductivity	Reference
1982	Choptank	x	x	x				Historical data sheets
1983	Choptank	x	x	x	x	x	x	Historical data sheets
1984	Choptank	x	x	x	x	x	x	Historical data sheets
1985	Choptank	x	x	x	x	x	x	Historical data sheets
	Nanticoke	x	x					Historical data sheets
1986	Choptank	x	x	x	x	x	x	Historical data sheets
1987	Choptank	x	x	x	x	x	x	Historical data sheets
	Potomac	x				x		Houde et al 1988a
1988	Choptank	x	x	x	x	x	x	Historical data sheets
	Potomac	x						Houde et al. 1990
	Upper Bay	x						Houde et al. 1990
1989	Choptank	x	x	x	x	x	x	Historical data sheets
	Elk							Historical data sheets
	Nanticoke (Vienna)	x	x	x	x	x	x	Stroup et al. 1991
	Potomac	x				x	x	Houde and Rutherford 1992
	Upper Bay	x			x	x	x	Houde and Rutherford 1992
1991	Patuxent	x	x		x	x	x	Secor et al. 1994
1992	Nanticoke	x	x		x	x	x	Houde et al. 1996
1993	Nanticoke	x	x		x	x	x	Houde et al. 1996
1994	Choptank	x	x					Historical data sheets
	Nanticoke	x	x					Historical data sheets
1995	Elk	x	x					Historical data sheets
1996	Chester	x	x		x	x	x	Burton et al. 1996
	Elk	x	x					Historical data sheets
1997	Choptank	x						Historical data sheets
1998	Choptank	x	x					Historical data sheets
1999	Choptank	x	x					Historical data sheets
2000	Choptank	x	x					Historical data sheets
2001	Choptank	x	x					Historical data sheets
2002	Choptank	x	x					Historical data sheets
2003	Choptank	x	x					Historical data sheets
2004	Choptank	x	x					Historical data sheets
	Nanticoke							Historical Excel spreadsheet
2005	Nanticoke							Historical Excel spreadsheet
2006	Nanticoke							Historical Excel spreadsheet
2007	Nanticoke							Historical Excel spreadsheet
2008	Nanticoke	x	x					Historical Excel spreadsheet
2009	Nanticoke	x	x					Historical Excel spreadsheet
2010	Nanticoke	x	x					Historical Excel spreadsheet
2011	Elk	x	x		x	x	x	Historical data sheets
	Nanticoke	x	x				x	Historical Excel spreadsheet
	Northeast	x	x		x	x	x	Historical data sheets
2012	Elk	x	x		x		x	Present studies
	Nanticoke	x	x		x		x	Present studies
	Northeast	x	x		x		x	Present studies
	Patuxent	x	x		x		x	Present studies

Table 1 cont.

Year	Spawning area	Temperature	Salinity	Tide	DO	pH	Conductivity	Reference
2013	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x		x	Present studies
	Northeast	x	x		x		x	Present studies
	Patuxent	x			x	x	x	Present studies
2014	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x		x	Present studies
	Northeast	x	x		x		x	Present studies
	Patuxent	x	x		x	x	x	Present studies
2015	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x		x	Present studies
	Patuxent	x	x		x	x	x	Present studies
2016	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x		x	Present studies
	Patuxent	x	x		x	x	x	Present studies
2017	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x		x	Present studies
	Wicomico	x	x		x	x	x	Present studies
2018	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x		x	Present studies
	Wicomico	x	x		x	x	x	Present studies
2019	Chester	x	x		x	x	x	Present studies
	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x		x	Present studies
2021	Choptank	x	x		x	x	x	Present studies
	Nanticoke	x	x		x	x	x	Present studies
	Sassafras	x	x		x	x	x	Present studies

Table 2. Summary of years, spawning areas, and species reported in historical and present Striped Bass spawning studies. SB = Striped Bass, YP = Yellow Perch, Clup = Clupeidae species, and WP = White Perch. C = actual counts of eggs and/or larvae were available, while P/A = presence or absence information only. From 1994 on, only presence or absence was recorded for all systems and species. "Historical data sheets" indicate actual field or lab records were available, while "Historical Excel spreadsheet" indicates this data had been entered previously and hard copies (or pdf scans) were not available for comparison or additional information. All historical data was collected by MD DNR.

Year	Spawning area	SB Eggs	SB Larvae	YP Eggs	YP Larvae	Clup Eggs	Clup Larvae	WP Eggs	WP Larvae	Reference
1953	Patuxent	C	C							Historical data sheets
1954	Blackwater	C	C							Historical data sheets
	Choptank	C	C							Historical data sheets
	Elk	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Patuxent	C	C							Historical data sheets
	Pocomoke	C	C							Historical data sheets
	Potomac	C	C							Historical data sheets
	Susquehanna	C	C							Historical data sheets
	Transquaking	C	C							Historical data sheets
	Wicomico	C	C							Historical data sheets
1955	Blackwater	C	C							Historical data sheets
	Bohemia	C	C							Historical data sheets
	Chester	C	C							Historical data sheets
	Elk	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Northeast	C	C							Historical data sheets
	Patuxent	C	C							Historical data sheets
	Potomac	C	C							Historical data sheets
	Sassafras	C	C							Historical data sheets
	Susquehanna	C	C							Historical data sheets
1956	Annemessex	C	C							Historical data sheets
	Blackwater	C	C							Historical data sheets
	Bohemia	C	C							Historical data sheets
	Elk	C	C							Historical data sheets
	Manokin	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Northeast	C	C							Historical data sheets

Table 2 cont.

Year	Spawning area	SB Eggs	SB Larvae	YP Eggs	YP Larvae	Clup Eggs	Clup Larvae	WP Eggs	WP Larvae	Reference
1956 cont.	Pocomoke	C	C							Historical data sheets
	Potomac	C	C							Historical data sheets
	Sassafras	C	C							Historical data sheets
	Susquehanna	C	C							Historical data sheets
	Transquaking	C	C							Historical data sheets
	Upper Bay	C	C							Historical data sheets
1957	Blackwater	C	C							Historical data sheets
	Chester	C	C							Historical data sheets
	Chicamacomico	C	C							Historical data sheets
	Choptank	C	C							Historical data sheets
	Elk	C	C							Historical data sheets
	Manokin	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Pocomoke	C	C							Historical data sheets
	Transquaking	C	C							Historical data sheets
	Upper Bay	C	C							Historical data sheets
	Wicomico	C	C							Historical data sheets
1958	Blackwater	C	C							Historical data sheets
	Choptank	C	C							Historical data sheets
	Elk	C	C							Historical data sheets
	Manokin	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Pocomoke	C	C							Historical data sheets
	Sassafras	C	C							Historical data sheets
	Transquaking	C	C							Historical data sheets
	Upper Bay	C	C							Historical data sheets
	Wicomico	C	C							Historical data sheets

Table 2 cont.

Year	Spawning area	SB Eggs	SB Larvae	YP Eggs	YP Larvae	Clup Eggs	Clup Larvae	WP Eggs	WP Larvae	Reference
1959	Blackwater	C	C							Historical data sheets
	Chicamacomico	C	C							Historical data sheets
	Choptank	C	C							Historical data sheets
	Manokin	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Patuxent	C	C							Historical data sheets
	Pocomoke	C	C							Historical data sheets
	Transquaking	C	C							Historical data sheets
	Wicomico	C	C							Historical data sheets
1960	Blackwater	C	C							Historical data sheets
	C&D Canal	C	C							Historical data sheets
	Choptank	C	C							Historical data sheets
	Manokin	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Pocomoke	C	C							Historical data sheets
	Sassafras	C	C							Historical data sheets
	Transquaking	C	C							Historical data sheets
	Upper Bay	C	C							Historical data sheets
	Wicomico	C	C							Historical data sheets
1961	Blackwater	C	C							Historical data sheets
	Bohemia	C	C							Historical data sheets
	Choptank	C	C							Historical data sheets
	Elk	C	C							Historical data sheets
	Nanticoke	C	C							Historical data sheets
	Northeast	C	C							Historical data sheets
	Transquaking	C	C							Historical data sheets
	Upper Bay	C	C							Historical data sheets

Table 2 cont.

Year	Spawning area	SB Eggs	SB Larvae	YP Eggs	YP Larvae	Clup Eggs	Clup Larvae	WP Eggs	WP Larvae	Reference
1962	Blackwater	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Choptank	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Elk	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Transquaking	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Wicomico	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1963	Elk	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Upper Bay	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1964	Nanticoke	C	C							Historical data sheets
1965	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1966	Nanticoke	C	C			P/A	P/A	P/A	P/A	Historical data sheets
1967	Blackwater	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Chicamacomico	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Transquaking	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Tuckahoe	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1968	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1969	Nanticoke	C	C							Historical data sheets
1970	Manokin	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Nanticoke	C	C							Historical data sheets
1971	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Potomac	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1972	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1973	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Potomac	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
1974	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Potomac	C	C							Mihursky et al. 1974
1975	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Potomac	C	C							Mihursky et al. 1976

Table 2 cont.

Year	Spawning area	SB Eggs	SB Larvae	YP Eggs	YP Larvae	Clup Eggs	Clup Larvae	WP Eggs	WP Larvae	Reference
1976	Bohemia	C	C		C		C	C	C	Otto and Peterson 1980
	Elk	C	C		C		C	C	C	Otto and Peterson 1980
	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Northeast	C	C		C		C	C	C	Otto and Peterson 1980
	Potomac	C	C							Boynton et al. 1977
	Sassafras	C	C		C		C	C	C	Otto and Peterson 1980
1977	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Potomac	C	C							Setzler-Hamilton et al. 1980a
1978	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Nanticoke (Vienna)	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Portner and Kohlenstein 1979
	Patuxent	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Setzler et al. 1979
1979	Nanticoke	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Historical data sheets
	Patuxent	C	C	P/A	P/A	P/A	P/A	P/A	P/A	Setzler-Hamilton et al. 1980b
1980	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
	Nanticoke	C	C							Historical data sheets
1981	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
	Nanticoke	C	C							Historical data sheets
1982	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
1983	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
1984	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
1985	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
	Nanticoke	C	C							Historical data sheets
1986	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
1987	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
	Potomac	C	C				C		C	Houde et al 1988a
1988	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
	Potomac	C	C							Houde et al. 1990
	Upper Bay	C	C							Houde et al. 1990

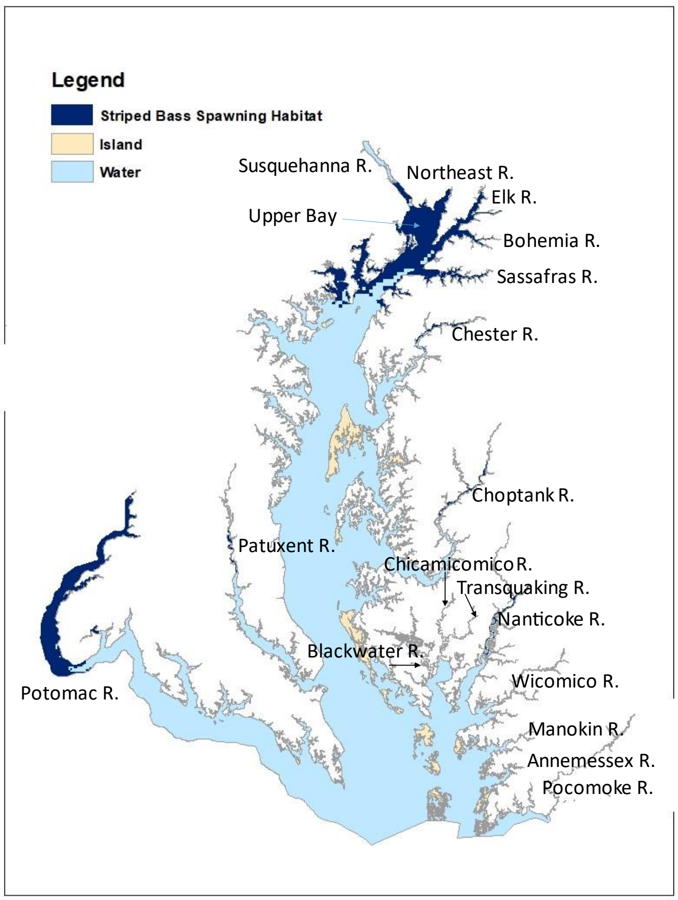
Table 2 cont.

Year	Spawning area	SB Eggs	SB Larvae	YP Eggs	YP Larvae	Clup Eggs	Clup Larvae	WP Eggs	WP Larvae	Reference
1989	Choptank	C	C	C	C	C	C	C	C	Historical data sheets
	Elk	C	C	C	C	C	C	C	C	Historical data sheets
	Nanticoke (Vienna)	C	C		C		C		C	Stroup et al. 1991
	Potomac	C	C				C		C	Houde and Rutherford 1992
	Upper Bay	C	C				C		C	Houde and Rutherford 1992
1991	Patuxent	C	C							Secor et al. 1994
1992	Nanticoke	C	C							Houde et al. 1996
1993	Nanticoke	C	C							Houde et al. 1996
1994	Choptank	P/A								Historical data sheets
	Nanticoke	P/A								Historical data sheets
1995	Elk	P/A								Historical data sheets
1996	Chester	C								Burton et al. 1996
	Elk	P/A								Historical data sheets
1997	Choptank	P/A								Historical data sheets
1998	Choptank	P/A			P/A					Historical data sheets
1999	Choptank	P/A			P/A					Historical data sheets
2000	Choptank	P/A			P/A					Historical data sheets
2001	Choptank	P/A			P/A					Historical data sheets
2002	Choptank	P/A			P/A					Historical data sheets
2003	Choptank	P/A			P/A					Historical data sheets
2004	Choptank	P/A			P/A					Historical data sheets
	Nanticoke	P/A			P/A					Historical Excel spreadsheet
2005	Nanticoke	P/A			P/A					Historical Excel spreadsheet
2006	Nanticoke	P/A			P/A					Historical Excel spreadsheet
2007	Nanticoke	P/A			P/A					Historical Excel spreadsheet
2008	Nanticoke	P/A			P/A					Historical Excel spreadsheet
2009	Nanticoke									Historical Excel spreadsheet
2010	Nanticoke									Historical Excel spreadsheet
2011	Elk	P/A			P/A					Historical data sheets
	Nanticoke	P/A			P/A					Historical Excel spreadsheet
	Northeast	P/A			P/A					Historical data sheets

Table 2 cont.

Year	Spawning area	SB Eggs	SB Larvae	YP Eggs	YP Larvae	Clup Eggs	Clup Larvae	WP Eggs	WP Larvae	Reference
2012	Elk	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Northeast	P/A			P/A					Present studies
	Patuxent	P/A			P/A					Present studies
2013	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Northeast	P/A			P/A					Present studies
	Patuxent	P/A			P/A					Present studies
2014	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Northeast	P/A			P/A					Present studies
	Patuxent	P/A			P/A					Present studies
2015	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Patuxent	P/A			P/A					Present studies
2016	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Patuxent	P/A			P/A					Present studies
2017	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Wicomico	P/A			P/A					Present studies
2018	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Wicomico	P/A			P/A					Present studies
2019	Chester	P/A			P/A					Present studies
	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
2021	Choptank	P/A			P/A					Present studies
	Nanticoke	P/A			P/A					Present studies
	Sassafras	P/A			P/A					Present studies

Figure 1. Location of areas with historic surveys (1953-2021) of Striped Bass and other anadromous fish egg and larval data. Dark blue shading indicates where salinity is 2 ppt that would indicate potential Striped Bass spawning and larval habitat.



Section 3 - Estuarine Fish Community Sampling

Alexis Park, Carrie Hoover, Margaret McGinty, Jim Uphoff

Changes to Project 4 Activities due to Coronavirus

The choice of subestuaries sampled during summer 2020 was influenced by State of Maryland travel policies imposed due to COVID-19; only one biologist could drive the state vehicle that towed the boat and remaining survey crew typically used their own vehicles. The subestuaries chosen had resource issues of interest and minimized driving.

Introduction

Human population growth since the 1950s added a suburban landscape layer to the Chesapeake Bay (or Bay) watershed (Brush 2009) that has been identified as a threat (Chesapeake Bay Program or CBP 1999). Development converts land use typical of rural areas (farms, wetlands, and forests) to residential and industrial uses (Wheeler et al. 2005; National Research Council or NRC 2009; Brush 2009; Meals et al. 2010; Sharpley et al. 2013; Zhang et al. 2016). These are the basic trade-off in land use facing Maryland as its population grows (Maryland Department of Planning; MD DOP 2020a) and they have ecological, economic, and societal consequences (Szaro et al. 1999).

Water quality and aquatic habitat are altered by agricultural activity and urbanization. Both land-uses include pesticide and fertilizer application. Agriculturally derived nutrients have been identified as the primary driver of hypoxia and anoxia in the mainstem of the Bay (Hagy et al. 2004; Kemp et al. 2005; Fisher et al. 2006; Brush 2009; Zhang et al. 2016). Land in agriculture has been relatively stable but farming itself has become much more intensive (fertilizer and pesticide use has increased) to support crop production and population growth (Fisher et al. 2006; Brush 2009).

Urbanization may introduce additional industrial wastes, contaminants, stormwater runoff, and road salt (Brown 2000; NRC 2009; Benejam et al. 2010; McBryan et al. 2013; Branco et al. 2016) that act as ecological stressors and alter fish production. Extended exposure to biological and environmental stressors affect fish condition and survival (Rice 2002; Barton et al. 2002; Benejam et al. 2008; Benejam et al. 2010; Branco et al. 2016). Reviews by Wheeler et al. (2005), the National Research Council (NRC 2009) and Hughes et al. (2014a; 2014b) documented deterioration of non-tidal stream habitat with urbanization. Todd et al. (2019) reviewed impacts of three interacting drivers of marine urbanization (resource exploitation, pollution, and proliferation of manmade marine structures) and described negative impacts that were symptomatic of urban marine ecosystems. Taylor and Suthers (2021) outlined how urban estuarine fisheries management was defined by unique ecological attributes of urbanized estuaries, the socio-economic objectives of anglers, and bottlenecks to productivity of exploited species.

Development of the Bay watershed brings with it ecologically stressful factors that conflict with demand for fish production and recreational fishing opportunities from its estuary (Uphoff et al. 2011; Uphoff et al. 2020). Uphoff et al. (2011) estimated target and limit impervious surface reference points (ISRPs) for productive juvenile and adult fish habitat in brackish (mesohaline; 5.0 – 18.0 ‰; Oertli, 1964) Chesapeake Bay subestuaries based on dissolved oxygen (DO) criteria, and associations and relationships of watershed impervious surface (IS), summer DO, and presence-absence of recreationally important finfish in bottom waters. Watersheds of mesohaline subestuaries at a target of 5.5 % IS (expressed as IS equivalent to that estimated by the methodology used by Towson University for 1999-2000) or less (rural watershed) maintained mean bottom DO above 3.0 mg/L (threshold DO), but mean bottom DO was only occasionally at or above 5.0 mg/L (target DO). Mean bottom DO seldom exceeded 3.0 mg/L above 10 % IS (suburban threshold; Uphoff et al. 2011). Although bottom DO concentrations were negatively influenced by development (indicated by IS) in mesohaline subestuaries, Uphoff et al. (2020) have found adequate concentrations of DO in bottom channel habitat of tidal-fresh (0 – 0.5 ‰) and oligohaline (0.5 – 5.0 ‰) and) 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 2020, we continued to evaluate summer nursery and adult habitat for recreationally important finfish in tidal-fresh, oligohaline, and mesohaline subestuaries of the Chesapeake Bay. In this section, we analyzed the associations of land use (i.e., agriculture, forest, urban, and wetlands) and C/ha (structures per hectare) on the annual median bottom DO among subestuaries sampled during 2003 – 2020. We evaluated the influence of watershed development on target species presence-absence and abundance, total abundance of finfish, and finfish species richness. We continued to examine Tred Avon River, a tributary of the Choptank River located in Talbot County (Table 3-1; Figure 3-1). We returned to two subestuaries located in Talbot County in 2020, Broad Creek, another tributary of the Choptank River previously sampled from 2012 to 2017, and Miles River previously sampled from 2003 to 2005 (Table 3-1; Figure 3-1). We visited the Sassafras River located at the Head-of-Bay for the first time in 2020 (Table 3-1; Figure 3-1). We examined associations among relative abundance of all finfish from Choptank River and the Head-of-Bay with Tred Avon River, Broad Creek, Miles River, and Sassafras River to evaluate potential contributions of the two large outside regions to the abundance in tributaries and subestuaries in our study. We added a more detailed evaluation of species composition and richness to our analysis in order to better understand the possible changes occurring throughout the subestuaries of the Chesapeake Bay.

Methods

Each subestuary sampled was classified into a salinity category based on the Venice System for Classification of Marine Waters (Oertli 1964). Tidal-fresh ranged from 0 – 0.5 ‰; oligohaline, 0.5 – 5.0 ‰; and mesohaline, 5.0 – 18.0 ‰ (Oertli 1964). Salinity influences distribution and abundance of fish (Allen 1982; Cyrus and Blaber, 1992; Hopkins and Cech 2003) and DO (Kemp et al. 2005). We calculated an arithmetic mean of all bottom salinity and measurements for all years available through 2020 to determine salinity class of each subestuary. We sampled three Chesapeake Bay mesohaline subestuaries located in Talbot County during 2020: Tred Avon River and Broad Creek (mesohaline tributaries of the Choptank River), and Miles River. We have sampled Tred Avon River since 2006; Broad Creek was previously sampled from 2012 to 2017; and Miles River was previously sampled from 2003 to 2005. In 2020 we sampled the Sassafras River, a tidal-fresh subestuary located between Kent and Cecil Counties, for the first time.

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

We used property tax map-based counts of structures in a watershed (C), standardized to hectares (C/ha), as our indicator of development (Uphoff et al. 2012; Topolski 2015). Estimates of C/ha and Maryland Department of Planning land use and water percentages were used for analyses of data from mesohaline subestuaries sampled during 2003 – 2020 (Table 3-2). Maryland DOP only has structure estimates available through 2018; 2018 estimates were used to represent 2018 – 2020 in analyses. Methods used to estimate development (C/ha) and land use indicators (percent of agriculture, forest, wetlands, urban land use, and water in the watershed) are explained in **General Spatial and Analytical Methods used in Project 1, Sections 1-3.** The C/ha to impervious surfaces (IS) conversion based on 1999 – 2000 property tax map estimates and subestuaries was revised in 2019, to reflect updates and led to revised C/ha levels for IS reference points (5 % IS = 0.38; 10 % IS = 0.86; and 15 % IS = 1.35). Impervious surface estimates were made by Towson University from Landsat, 30-meter pixel resolution satellite imagery (eastern shore of Chesapeake Bay in 1999 and western shore in 2001; Barnes et al. 2002). Development targets and limits, and general statistical methods (analytical strategy and equations) are described in **General Spatial and Analytical Methods used in Project 1, Sections 1 – 3** as well. Specific spatial and analytical methods for this section of the report are described below.

2020 Sampling - Ideally, four evenly spaced haul seine and bottom trawl sample sites were located in the upper two-thirds of each subestuary. Lower portions of a subestuary were not sampled in order to minimize the impact of mainstem water and maximize subestuary watershed influence. We used GPS to record latitude and longitude at the beginning and end of each trawl site, while latitude and longitude at seining sites were taken at the seine starting point on the beach. We focused on using previously sampled historical sites in 2020 at each of the previously sampled subestuaries unless they were no longer accessible. Miles River original seine site 01 lacked a shoreline. We tried to create a new seine site on the other side of the river where a small shoreline was found, but seining this upper section became impossible due to woody debris. All other seine and trawl sites remained the same. A seine site was not available in the past or present for site 02 in Miles River. Broad Creek's original seine site 04 was not sampled in 2020 due to large amounts of SAV and *Ulva* present; SAV has caused seining issues at this site in the past. A seine site was not available in the past or present for site 01 in Broad Creek. Sassafras River was sampled for the first time by this project in 2020. Seine sites in the Sassafras River were not sampled (but were established) during 2020 due to health risks from a harmful algal bloom (HAB) that occurred within the river throughout the sampling season; seine data was acquired from Juvenile Index (JI) monitoring station Sassafras River Natural Resource Management Area (NRMA) for 2020 to examine the inshore fish community. We partnered with Sassafras RiverKeeper and Maryland Department of Environment (MDE) to collect water samples for testing, an ELISA Microcystin analysis indicated that samples were above the state threshold of 8 ppb for water contact and recreation throughout our sampling schedule.
Sites were sampled once every two weeks during July – September, totaling six visits per system during 2020. The number of total samples collected from each system varied based on the number of sites available, SAV interference, weather/tidal influences, and equipment issues. All sites on one river were sampled on the same day, usually during morning through mid-afternoon.

Sites were numbered from upstream (station 01) to downstream (station 04). The crew determined whether to start upstream or downstream based on tidal direction; this helped randomized potential effects of location and time of day on catches and dissolved oxygen, and 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 then the crew leader deviated from the sample route to accommodate this need. Bottom trawl sites were generally in the channel, adjacent to haul seine sites. At some sites, haul seines could not be made because of permanent obstructions, dense SAV beds, or lack of beaches. Bottom trawl and haul seine sampling was conducted one right after the other at a site to minimize time of day or tidal influences between samples.

Water Quality Sampling - Water quality parameters were recorded at all stations for every individual sampling event in 2020. Temperature (ºC), DO (mg/L), conductivity (µS/cm), salinity (parts per thousand; ppt = ‰), and pH were recorded at the surface, middle, and bottom of the water column at the trawl sites depending on depth, and at the surface of each seine site. Mid-depth measurements were omitted at sites with less than 1.0 m difference between surface and bottom. Secchi depth was measured to the nearest 0.1 m at each trawl site. Weather, tide state (flood, ebb, high or low slack), date, and start time were recorded for all sites.

Dissolved oxygen concentrations were evaluated against a target of 5.0 mg/L and a threshold of 3.0 mg/L (Batiuk et al. 2009; Uphoff et al. 2011). 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. 2011). Target DO was considered sufficient to support aquatic life needs in Chesapeake Bay (Batiuk et al. 2009) and has been used in a regulatory framework to determine if a water body is meeting its designated aquatic life uses. Presence of target species in bottom channel trawls declined sharply when bottom DO fell below the 3.0 mg/L threshold (Uphoff et al. 2011). We estimated the percentages of DO samples in each subestuary that did not meet the target or threshold for all DO samples (surface, middle, and bottom DO) and for bottom DO alone. Percentages not meeting target or threshold conditions were termed “violations”, but the term did not have a regulatory meaning. The percentages of DO measurements that met or fell below the 5 mg/L target (V_{target}) or fell at or below the 3 mg/L threshold ($V_{threshold}$) were estimated as:

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

and

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

where N_{target} was the number of measurements meeting or falling below 5 mg/L, $N_{threshold}$ was the number of measurements falling at or below 3 mg/L, and N_{total} was total sample size.
Separate Pearson correlation analyses were conducted for surface or bottom temperature or C/ha with surface or bottom DO for all subestuaries sampled since 2003. This analysis explored multiple hypotheses related to DO conditions. Structures per hectare estimates were considered proxies for nutrient loading and processing due to development in the subestuaries in this analysis (Uphoff et. al 2011). Water temperature would influence system respiration and stratification (Kemp et al. 2005; Murphy et al. 2011; Harding et al. 2016). Conducting correlation analyses by salinity classification provided a means of isolating the increasing influence of salinity on stratification from the influence of temperature. Our primary interest was in associations of C/ha to DO in surface and bottom channel waters. Temperature and salinity

were potential influences on DO because of their relationships with DO saturation and stratification (Kemp et al. 2005; Murphy et al. 2011; Harding et al. 2016). We correlated mean surface temperature with mean surface DO, mean bottom temperature with mean bottom DO, and C/ha with surface and bottom DO for each salinity class. We chose annual survey means of surface or bottom DO and water temperature in summer at all sites within a subestuary for analyses to match the geographic scale of C/ha estimates (whole watershed) and characterize chronic conditions.

We obtained land use estimates for our watersheds from the Maryland Department of Planning (MD DOP) for 2002 and 2010 (MD DOP 2020b). The MD DOP provides agriculture, forest, urban, and wetlands estimates periodically rather than annually, but C/ha is estimated annually. Median summer bottom DO estimates made before 2010 were compared with 2002 MD DOP land use estimates and those made for 2010 – 2020 were matched with 2010 MD DOP estimates (the most current available). Four categories of land use (percent in agriculture, forest, urban, and wetlands) were estimated based on the land portion of the watershed (water area was excluded from these categories). A fifth category, percent in water, was estimated based on the water plus land area of the watershed. Newer land use estimates have not been released by MD DOP.

We analyzed the associations of land use (i.e., agriculture, forest, urban, and wetlands) and C/ha (structures per hectare) with annual median bottom DO among mesohaline systems sampled during 2003 – 2020 using correlation analysis. We further examined the influence of percent of land in agriculture on median bottom DO using linear, multiple linear, and quadratic regression models. We focused this analysis on mesohaline subestuaries because bottom DO does not exhibit a negative response to development in the other salinity categories.

Water Quality in Mesohaline Tributaries and Subestuaries in Talbot County - In 2020, we sampled three tributaries and subestuaries located within Talbot County, Broad Creek, Miles River, and Tred Avon River (Figure 3-1). The original four stations were sampled in Broad Creek, Miles River, and Tred Avon River (Figure 3-2). We contrasted Tred Avon River to Broad Creek (sampled during 2012 – 2017, 2020), Miles River (2003 – 2005, 2020), and previously sampled Harris Creek (2012 – 2016; Figure 3-3). Trajectories of C/ha since 1950 were plotted for the three Choptank tributaries, Broad Creek, Harris Creek, and Tred Avon River, and Miles River. Bottom DO measurements during 2006 – 2020 were plotted against C/ha and percent of target and threshold DO violations were estimated using all measurements combined (surface, middle, and bottom) and for bottom DO only. Annual mean bottom DO (depth most sensitive to violations) in Tred Avon River at each station for 2006 – 2020 was estimated and plotted by year. We examined correlations of Secchi depths, SAV coverage, DO, pH, and salinity within the three Choptank tributaries and Miles River.

An ANOVA was used to examine differences in mean bottom DO among stations in Broad Creek, Harris Creek, Miles River, and Tred Avon River. Tukey Studentized Range and Tukey Honestly Significant Difference (HSD) tests examined whether stations within each tributary and subestuary were significantly different from one another. An overall median DO was calculated for all time-series data available for each system and used to detect how annual station DO compared with the time-series median. Correlation analysis of annual median DO measurements was used among the three Choptank tributaries.

Water Quality in Head-of-Bay Subestuaries - In 2020, we sampled the Sassafras River for the first time (Figure 3-2). Sassafras represented the only Maryland low salinity subestuary with a watershed dominated by agriculture. Sassafras River was associated with previously sampled Head-of-Bay subestuaries, Bohemia River (2006), Bush River (2006 – 2010), Gunpowder River (2009 – 2016), Middle River (2009 – 2017), and Northeast River (2007 – 2017; Figure 3-4). Trajectories of C/ha since 1950 were plotted for the Head-of-Bay subestuaries. Bottom DO measurements during 2006 – 2020 were plotted against C/ha and percent of target and threshold DO violations were estimated using all measurements combined (surface, middle, and bottom) and for bottom DO only. Annual mean bottom DO (depth most sensitive to violations) for each station within the Head-of-Bay subestuaries for 2006 – 2020 was estimated and plotted by year. We examined annual medians of Secchi depths, DO, pH, and salinity within the Head-of-Bay subestuaries, additional analyses involving Secchi depths, DO, pH, and salinity were limited due to the quantity of data. A subset of Bush River sampling years, 2006 – 2010, were summarized for this report; years sampled by citizen volunteers, 2011 – 2020, were excluded because not all stations were sampled consistently each year, therefore, data was incomplete.

An ANOVA was used to evaluate station differences in mean bottom DO; Tukey Studentized Range and Tukey Honestly Significant Difference (HSD) tests examined which station(s) within each subestuary were significantly different from others. An overall median bottom DO was calculated for all time-series data available for each system and compared with annual mean station DO.

Finfish Community Sampling - Surveys focused on twelve target species of finfish that fell within four broad life history groups: anadromous (American Shad, Alewife, Blueback Herring, Striped Bass), estuarine residents (semi-anadromous White Perch and Yellow Perch, and estuarine Bay Anchovy), marine migrants (Atlantic Menhaden and Spot), and tidal-fresh forage (Spottail Shiner, Silvery Minnow, and Gizzard Shad). With the exception of White Perch, adult sportfish of the target species were rare and juveniles were common. Use of target species is widespread in studies of pollution and environmental conditions (Rice 2003). These species are widespread and support important recreational fisheries in the Bay (directly or as forage); they are well represented in commonly applied seine and-or trawl techniques (Bonzek et al. 2007); and the Bay serves as an important nursery for them (Lippson 1973; Funderburk et al. 1991; Deegan et al. 1997). Gear specifications and techniques were selected to be compatible with past and present MD DNR Fishing and Boating Services' surveys (Carmichael et al. 1992; Bonzek et al. 2007; Durell and Weedon 2020).

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 30.5 m × 1.2 m bag-less beach seine, constructed of untreated knotted 6.4 mm stretch mesh nylon, was used to sample inshore habitat. The float-line was rigged with 38.1 mm by 66 mm floats spaced at 0.61 m intervals and the lead-line rigged with 57 gm lead weights spaced evenly at 0.55 m intervals. One end of the seine was held on shore, while the other was stretched perpendicular from shore as far as depth permitted and then pulled with the tide in a quarter-arc. The open end of the net was moved towards shore once the net was stretched to its maximum. When both ends of the net were on shore, the net was retrieved by hand in a diminishing arc until the net was entirely pursed. The section of the net containing the fish was then placed in a tub for processing. The distance the net was stretched from shore, maximum depth of the seine haul, primary and secondary bottom types (i.e., gravel, sand, mud, and shell), and percent of seine area containing submerged aquatic vegetation were recorded. All fish captured were identified to species and counted. Striped Bass and Yellow Perch were separated into two age categories, juveniles (Juv; young of year = YOY) and adults (ages 1+). White Perch were separated into three age categories based on size and life stage, juveniles, small adults (ages 1+ fish measuring < 200 mm), and harvestable size adults (fish measuring > 200 mm). Harvestable size adult White Perch were measured and the measurements were recorded for a modified proportional stock density analysis (PSD; Willis et al. 1993).

Three basic metrics of finfish community composition were estimated for tributaries and subestuaries sampled: geometric mean (GM) catch of all species, total number of species (species richness), and species comprising 90 % of the catch. The GM of seine and trawl catches were estimated as the back-transformed mean of $\log_e$-transformed catches (Ricker 1975; Hubert and Fabrizio 2007). The GM is a more precise estimate of central tendency of fish catches than the arithmetic mean but is on a different scale (Ricker 1975; Hubert and Fabrizio 2007). In addition, we noted which target species were within the group that comprised 90 % of fish collected, grouping the remaining 10 % of species into the "other species" category. We summarized these metrics by salinity type since some important ecological attributes (DO and high or low SAV densities) appeared to reflect salinity class.

We plotted species richness in seine and trawl collections against C/ha by salinity class. A greater range of years (1989 – 2020) was available for beach seine samples than the 4.9 m bottom trawl (2003 – 2020) due to a change from the 3.1 m trawl used during 1989 – 2002 (Carmichael et al. 1992). Gear comparisons between the 3.1 m and 4.9 m trawls can be reviewed in Uphoff et al. (2016). We set a minimum number of samples (15 for seine and trawl) for a subestuary in a year to include estimates of species richness based on species accumulation versus sample size analyses in Uphoff et al. (2014). This eliminated years where sampling in a subestuary ended early due to site losses (typically from SAV growth) or high tides. We separated all subestuaries sampled by salinity class, then ranked their 2003 – 2020 bottom trawl GMs by year for all species combined to find where the 2020 subestuaries sampled ranked when compared to other subestuaries in their respective salinity classes.

A modified Proportional Stock Density (PSD; Anderson 1980; Anderson and Neumann 1996; Neumann and Allen 2007) was calculated using trawl catch data for White Perch in the tributaries and subestuaries of Talbot County and the subestuaries of Head-of-Bay for each year available 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:

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

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 fish at the minimum length most anglers like to catch (≥ 200 mm TL; Piavis and Webb 2020). 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 2020). We substituted for stock length with the total number of small adults plus harvestable length White Perch to estimate a modified PSD since we did not measure small adults. White Perch greater than or equal to 130 mm TL is 20 – 26% of the world record length TL (Gablehouse et al 1984) is considered stock length category minimum; 125 mm TL is used as the length cut-off for White Perch in Chesapeake Bay recruitment and length-frequency assessments (Piavis and Webb 2020). Modified stock length category included small adults under 200 mm TL and could have fish as small as 90 mm TL. White Perch greater than or equal to 200 mm TL were measured to the nearest millimeter. White Perch greater than or equal to 200 mm TL corresponded to the quality length category minimum (36 – 41% of the world record TL) proposed by Gablehouse et al. (1984); 200 mm TL is used as the quality length category minimum length cut-off for White Perch in Chesapeake Bay (Piavis and Webb 2020). These data provided an opportunity to evaluate the influence of development on the availability of fish for anglers to harvest.

Fish Community Sampling in Mesohaline Tributaries and Subestuaries in Talbot County - In 2020, we examined correlations of 4.9 m bottom trawl geometric mean catches of all finfish or adult White Perch within the three Choptank tributaries and Miles River. We estimated GMs of trawl and seine catches, modified PSD of White Perch, and species composition.

We used a percent similarity index to evaluate variation in finfish species composition among the three Choptank tributaries trawl stations by year (Kwak and Peterson 2007). Finfish species abundances at a trawl station were standardized to percentages by dividing the abundance of each finfish species in a trawl station by the total number of fish collected at that trawl station, by year. The similarity among stations, P_{jklm} was calculated as:

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

where $p_{ji,}$ p_{ki}, p_{li}, and p_{mi} refers to the finfish species abundance of one particular finfish species *i* in trawl stations *j*, *k*, *l*, and *m*, by year, and the minimum indicates that the smallest of the four relative abundances was used in the summation (Kwak and Peterson 2007). The percent similarity index varies from 0% (no species in common) to 100 % (all species in common) and is considered a robust measure (Kwak and Peterson 2007).

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

Fish Community Sampling in Head-of-Bay Subestuaries - In 2020, we sampled the Sassafras River for the first time to collect information on fish habitat status. Sassafras represented the only low salinity subestuary with a watershed dominated by agriculture. Sassafras River metrics were compared with previously sampled Head-of-Bay subestuaries: Bohemia River (2006), Bush River (2006 – 2010), Gunpowder River (2009 – 2016), Middle River (2009 – 2017), and Northeast River (2007 – 2017).

Annual GMs of total fish relative abundance and their 95 % CIs were estimated for 4.9 m trawl and beach seine. The top 90 % of finfish species occurring in annual trawl and seine catches were estimated for each subestuary time-series. Due to increased HABs, seining was not conducted in Sassafras River during 2020 by FHEP staff; staff compiled seine data collected by the Juvenile Striped Bass Survey (Juvenile Index or JI) at the Sassafras River Natural Resource Management Area (NRMA monitoring station) in the Head-of-Bay for catch composition. The JI monitoring station was sampled monthly (July, August, and September), using replicate seine hauls, a minimum of thirty minutes apart, were taken at each site in each month. The NRMA station is located 1.61 km (1.0 miles) downriver of trawl site 04. The NRMA seine GM was calculated only using the first seine haul (comparative to FHEP sampling methods) and had only three samples.

Results and Discussion

2020 Water Quality Summary – Table 3-3 provides summary statistics for surface and bottom water quality for each tributary and subestuary sampled in 2020. Three of the four tributaries and subestuaries sampled had bottom DO reading less than the target level (5.0 mg/L) during 2020 (Table 3-4). Sassafras River did not have any DO readings below target level (N = 75). Six percent of all DO measurements (surface and bottom) from Broad Creek were below the target; in Miles River, 38% were below the target; and 27% in Tred Avon River. In 2020, two subestuaries did not have any bottom DO estimates below the 3 mg/L threshold, Broad Creek and Sassafras River. The remaining subestuaries had threshold bottom DO violations: Miles River, 48%; and Tred Avon River, 17% (Table 3-4).

Salinities in the Choptank tributaries and Miles River were within mesohaline bounds in 2020 (Table 3-4). Sassafras River was classified as a tidal-fresh subestuary in 2020.

Dissolved Oxygen Dynamics – Analyses of DO with temperature and C/ha in subestuaries sampled since 2003 (Table 3-5) indicated that DO responded to temperature and C/ha differently depending on salinity classification (Table 3-6). Mean bottom DO in summer surveys declined below the threshold level in mesohaline subestuaries, but did not in oligohaline or tidal-fresh (Figure 3-5). There were a few years in summer surveys where mean bottom DO fell below the target in oligohaline subestuaries, but remained above 4.0 mg/L; these below target conditions would not affect occupation of this habitat (Uphoff et al. 2011). Mean surface DO in summer surveys did not fall below the threshold, but two mesohaline subestuaries (Chester River, 2011 – 2012; Corsica River, 2012) fell below the target conditions (Figure 3-6).

A moderate negative association of surface dissolved oxygen (DO) and a strong negative association of bottom DO with corresponding mean water temperatures at depth were detected for oligohaline subestuaries by correlation analyses (Table 3-6), suggesting respiration was a factor in oligohaline subestuaries. Oligohaline subestuaries were shallower than most subestuaries of the other salinity categories, making them more likely to be warmer throughout. Associations of temperature and DO were weak in mesohaline and tidal-fresh subestuaries. A strong negative association between bottom DO and C/ha was found in mesohaline subestuaries; mesohaline subestuaries were where strongest stratification was expected. Oligohaline and tidal-fresh subestuaries were less likely to stratify because of low or absent salinity and the biological consequences of no or positive relationships would be similar (i.e., a negative impact on habitat would be absent). Remaining correlations were weak, although some were significant at $P <$ 0.012. Given that multiple comparisons were made, correlations that were significant at $P <$ 0.012 might be considered spurious if one rigorously adheres to significance testing (Nakagawa 2004; Anderson et al. 2000; Smith 2020). Sample sizes of mesohaline subestuaries (N = 87) were over twice as high as oligohaline (N = 33) or tidal-fresh subestuaries (N = 49), so ability to detect significant associations in mesohaline subestuaries was greater (Table 3-6).

Depletion of bottom DO to below target levels in mesohaline subestuaries with suburban-urban watersheds resulted in lost habitat. Uphoff et al. (2011) determined that the odds of adult and juvenile White Perch, juvenile Striped Bass, Spot, and Blue Crabs being present in shore zone seine samples from mesohaline subestuaries were not influenced by development, but odds of these target species being present in bottom channel trawl samples were negatively influenced by development through its negative influence on DO.

The extent of bottom channel habitat that can be occupied does not appear to diminish due to low DO with increasing watershed development in tidal-fresh and oligohaline subestuaries. However, more localized or episodic habitat issues appear to be important. Sampling of DO in dense SAV beds in tidal-fresh Mattawoman Creek in 2011 indicated that shallow water habitat could be negatively impacted by low DO within the beds (Uphoff et al. 2012; 2013; 2014; 2015; 2016). Unfortunately, it was not feasible for us to routinely monitor fish within the beds and the impact on target finfish could not be estimated. Ammonia toxicity that was potentially associated with high SAV coverage was suspected as a cause of boom and bust dynamics of trawl GMs in Mattawoman Creek during the 2000s (Uphoff et al. 2016). During 2015, the oligohaline Middle River subestuary experienced an extensive fish kill attributed to HABs (MDE 2016). During 2020, Sassafras River was subject to HABs throughout summer sampling, but fish kills were not detected.

Land Use Categories, C/ha, and Mesohaline Subestuary Bottom Dissolved Oxygen - Correlation of agriculture with C/ha was negative and considered moderate (r = -0.75; P < 0.0001); the correlation of urban land cover with C/ha was positive and considered strong (r = 0.89; P < 0.0001; Table 3-7). Correlation between forest cover with agriculture cover was negative and considered moderate (r = -0.57; P < 0.0001); urban cover with agriculture was negative and considered strong (r = -0.81; P < 0.0001). Wetland cover and C/ha were negative and considered weak (r = -0.27; P = 0.02). Remaining pairings of categories were not well correlated (Table 3-7).

After inspection of scatter plots, agricultural cover was further divided into regional categories (east and west of Chesapeake Bay) reflecting lower percentages of forest cover on the eastern shore, for analyses with DO in mesohaline subestuaries (Figure 3-7). Two western shore sub-regions reflected agricultural coverage: subestuaries located on the western shore of Chesapeake Bay (Magothy, Rhode, Severn, and South Rivers) fluctuated between 2.6% to 34.1% agricultural coverage, while lower Potomac River watersheds (Breton Bay, St. Clements, and Wicomico Rivers) ranged from 31.6% to 38.6% agricultural coverage. Eastern shore watersheds were divided into two divisions: Choptank River drainage (Broad and Harris Creeks, Miles River, and Tred Avon River) ranged from 42.6% to 53.7% agricultural coverage; mid-eastern shore watersheds (Chester, Corsica, Miles, Wye Rivers, and Langford Creek) ranged from 53.7% to 71.6% agricultural coverage.

Inspection of the scatter plot of percent of watershed in agriculture versus median bottom DO in mesohaline subestuaries indicated an ascending limb of median DO when agricultural coverage went from 2.6% to 40.9% comprised entirely of western shore subestuaries (Figure 3-7). Median DO measurements beyond this level of agricultural coverage (42.6% – 71.6% agriculture) were from eastern shore subestuaries and the DO trend appeared to be stable or declining. Development was predominant at low levels of agriculture (< 20% agricultural coverage). Agricultural coverage and C/ha were inversely correlated, so the positive trend of DO with agriculture when agricultural coverage was low was likely to reflect development's negative impact.

We split agricultural coverage and median bottom DO data into western and eastern regions and used a linear regression for each region to describe regional changes in annual median subestuary bottom DO with percent agriculture. The relationship was positive and considered strong for the western shore (slope = 0.13; SE = 0.02; r^2 = 0.73; P < 0.0001; N = 21;

Table 3-8) and negative and weak for the eastern shore (slope = -0.03; SE = 0.01; r^2 = 0.15; *P* = 0.0023; N = 59; Table 3-8). Predictions of median DO for mesohaline western shore subestuaries rose from 0.51 mg/L at 2.6% agricultural coverage to 5.18 mg/L at 38.6%. Predictions of median DO for mesohaline eastern shore subestuaries started at 5.35 mg/L at 42.6% agricultural coverage, increased to 5.48 mg/L at 50.1%, and then decreased to 4.46 mg/L at 71.6%. A quadratic regression of median bottom DO versus agricultural coverage described the relationship of median bottom DO with agricultural coverage well (R^2 = 0.57, P < 0.001; Table 3-9; Figure 3-7). Median bottom DO residuals were inspected and then plotted against agricultural coverage, residuals did not indicate substantial bias. However, residuals suggested that the predications at the highest coverage (≥ 65%) may have negatively biased. In addition, mesohaline subestuaries identified with heavy rainfall from 2018 to 2020 did not create noticeable changes in the relationship.

Water Quality Summary in Mesohaline Tributaries and Subestuaries in Talbot County – Percentages of land in agriculture (43% – 49%), forest (20% – 27%), and urban (23% – 34%) categories were similar among the three Choptank tributaries and Miles River (MD DOP 2020b; Table 3-10; Figure 3-1); however, wetlands varied among the three systems, comprising <1% of Broad Creek's watershed, 6% of Harris Creek's, and 1% of Miles and Tred Avon River's watersheds. Water comprised a larger fraction of the area in Broad and Harris Creeks (57% and 62%) than Tred Avon River (24%) and Miles River (22%; i.e., water to watershed ratios were higher in the former; Table 3-10; MD DOP 2020b).

Tax map estimates of C/ha indicated that the Tred Avon River watershed was subjected to more development than Broad Creek, Harris Creek, and Miles River watersheds (Figure 3-8) and more than indicated by the Maryland Department of Planning urban category (Table 3-10). Time-series for all watersheds started at a rural level of development (C/ha ranged from 0.05 to 0.2) in 1950. Harris Creek watershed passed the rural development target (C/ha = 0.38) in 2009, while Broad Creek (C/ha = 0.30) and Miles River (C/ha = 0.26) are still under the rural development target. More growth occurred in Tred Avon River's watershed and the rural development target was passed in 1982, reaching a C/ha of 0.77 in 2017 (Figure 3-8). Development accelerated noticeably in the Tred Avon River watershed during 1996-2011 and then slowed. Tred Avon River's watershed has been approaching the suburban threshold (C/ha = 0.86).

During 2020, 63% of Tred Avon River bottom DO samples were below the target and 17% were below the threshold. In Broad Creek, 21% were at or below the target and 0% were at or below the threshold. In Miles River, 81% were at or below the target and 48% were at or below the threshold (Table 3-11; Figure 3-9). During 2006 – 2020, 9% of bottom DO measurements from Tred Avon River were below the DO threshold and 38% were below the DO target. Less than 1% of Broad Creek bottom DO measurements during 2012 – 2017 and 2020 were below the threshold and 15% were below the target. Harris Creek did not have any bottom DO measurements that fell below the threshold, and 3% were below the target during 2012 – 2016. Miles River had 68% of bottom DO samples fall below the target in 2003 – 2005 and 2020; and 29% of bottom DO samples were below the threshold (Table 3-11; Figure 3-9).

There was more variation in annual summer median DO in Miles River (3.3 mg/L – 5.3 mg/L and in Tred Avon River (4.5 mg/L – 6.3 mg/L; Figure 3-10) than in Broad Creek (5.6 mg/L – 6.6 mg/L) and Harris Creek (5.7 mg/L – 6.4 mg/L; Figure 3-10). Correlations of median bottom DO between Tred Avon River and Broad Creek or Harris Creek were modest to low, while Broad and Harris Creek's correlations were very low. (Table 3-12). Years available for correlation analysis were low (5 – 7 years) and this pattern does not seem meaningful.

An ANOVA of Tred Avon River stations and bottom DO during 2006 – 2020 indicated significant differences among stations (F = 57.52; DF = 3; P < 0.0001; N = 359). Tukey Studentized Range and Tukey Honestly Significant Difference (HSD) tests indicated that bottom DO at station 01 (station at Easton, Maryland) was significantly lower than downstream stations 02, 03, and 04 (Figure 3-3). This decline in bottom DO with upstream distance was consistent with other mesohaline tributaries with high impervious surface (Uphoff et al. 2011). The mean and SE for bottom DO at all stations in Tred Avon River for all years were 5.17 mg/L and 0.08, respectively. Mean and SE for bottom DO at station 01 were 3.70 mg/L and 0.18; station 02 was 5.60 mg/L and 0.11; station 03 was 5.71 mg/L and 0.11; and station 04 was 5.70 mg/L and 0.10. Deterioration of DO at the uppermost station (station 01; Figure 3-11) since 2012 indicated that stormwater from increased watershed development around Easton was the source of poor water quality rather than runoff from the whole watershed or water intruding from downstream. Mean bottom DO at station 01 was the lowest of the time-series in 2020 (Figure 3-11).

An ANOVA of Broad and Harris Creeks and Miles River station bottom DO measurements did not indicate significant differences among stations in either of the subestuaries during sampling years. Annual station means in subestuaries varied without trend around the time-series median for all sites (Figure 3-11). The mean and SE for bottom DO at all stations for all years were 5.99 mg/L and 0.08 in Broad Creek; Harris Creek, 6.21 mg/L and 0.07; and Miles River, 4.08 mg/L and 0.21, respectively.

Miles River, an agricultural system and the least developed subestuary, had more widespread low bottom DO than the other Talbot County subestuaries. Tred Avon River, the subestuary with the most developed watershed exhibited low DO at Easton. Both of the Talbot County subestuaries exhibiting low DO (Miles River and Tred Avon River) had low percentages of water hectares per area of water and land (22% and 24% respectively). Broad and Harris Creeks had higher percentages, 57% and 62%, respectively. Low percentages may indicate that intrusion of "good" mainstem water into a subestuary is limited and internal nutrient loading and processing is important.

Median Secchi depths fluctuated slightly from year to year, while yearly ranges of Secchi depths reveal larger fluctuations within each system (Figure 3-12). Upper ranges were generally higher in Harris and Broad Creeks than in Tred Avon and Miles Rivers. Tred Avon River median Secchi depths ranged from 0.4 m to 0.75 m during 2006 – 2020; from 0.5 m to 0.9 m in Broad Creek during 2012 – 2017, 2020; and from 0.5 m to 1.1 m in Harris Creek during 2012 – 2016 (Figure 3-12). The three Choptank River tributaries Secchi depths were strongly correlated with each other (Table 3-13). Miles River median Secchi depth ranged from 0.5 m to 0.65 m during 2003 – 2005 and 2020 (Figure 3-12).

Tred Avon River, Broad Creek, and Harris Creek SAV coverage were combined in the VIMS (2021) mouth of the Choptank River region estimates. Coverage of SAV increased substantially from 1% in 2012 to 11.8% in 2017, and slightly declined to 7.2% in 2019 (Figure 3-13); since mapping started, the least SAV coverage was recorded in 1991 at 0.3%. The percentage of SAV coverage has remained above the time-series median of 4.5% since 2014 and displayed a similar trend present in the 1990s (Figure 3-13). The 2018 survey was only partially mapped. An SAV estimate for 2020 was not available at the time of this report.

Median pH in Tred Avon River from 2006 to 2020 ranged from 7.5 (2007) to 8.1 (2019; Figure 3-14). Broad Creek median pH during 2012 – 2017 and 2020 ranged from 7.8 (2014) to

8.1 (2015). Harris Creek median pH during 2012-2016 ranged from 7.7 (2013, 2014) to 7.9 (2012; Figure 3-14). Median pH estimates in Broad Creek and Harris Creek were strongly correlated, but remaining combinations were not (Table 3-14). Miles River only had one year (2020) of pH measurements and the median pH was 7.6 and ranged from 7.4 to 7.7, similar to the Choptank tributaries (Figure 3-14).

All salinity measurements remained in the mesohaline classification for the Choptank River tributaries and Miles River; salinity ranged varied the least in Tred Avon River and greatest in Miles River during 2020 (Figure 3-15). Overall, salinity range in 2020 for each of the mesohaline systems appeared normal compared to previous sampled years. Highest salinities for the Choptank subestuaries were observed in 2016, 12.8 ‰ in Tred Avon River and 13.6 ‰ in both Broad and Harris Creeks (Figure 3-15). Lowest salinity measurements differed by year in each subestuary, 2011 in Tred Avon River (7.5 ‰), 2013 in Broad Creek (10.2 ‰), and 2014 in Harris Creek (10.0 ‰; Figure 3-15). Median salinities of all three Choptank tributaries were positively and strongly correlated among each other; these strong correlations among these tributaries reflected their proximity to one another (Table 3-15). All the Choptank tributaries, indicated a positive, weak association between bottom salinity and bottom DO suggesting minimal influence of the former on the latter (Figure 3-16). Miles River (2003 – 2005, 2020) increased in median salinity from 9.1 ‰ (2003) to 11.6 ‰ (2005) and remained similar to 2005 in 2020 (11.5 ‰; Figure 3-15). Miles River had a negative, weak relationship between bottom salinity and bottom DO (Figure 3-16).

In 2020, there was little indication that low DO was more widespread than usual, nor did the other water quality measurements offer an obvious connection to changes in finfish abundance. Broad Creek bottom DO for all stations were below the time-series median for the first time, but the range of bottom DO measurements in 2020 was not the lowest for all years sampled. Tred Avon River's station 01 mean bottom DO has continuously declined since 2014 and has fallen below the target; mean bottom DO for stations 02 – 04 remained slightly under the time-series median. Miles River, an agricultural system, had the most extensive bottom DO violations in 2020. Miles River, along with Corsica River, appeared to represent agricultural, rural watersheds with poorer habitat than expected based on the quadratic relationship of bottom DO and agricultural land cover. We cannot offer a ready explanation for more extensive hypoxia in Miles River at this time. Dissolved oxygen conditions in Corsica River during 2003-2012 sampling were poor and may have been made worse by repeated sewage spills before the Centerville wastewater treatment plant was upgraded in 2010 (Uphoff et al. 2020). Corsica River bottom DO noticeably improved during 2018-2019 (Uphoff et al. 2020). This increase may also have been aided by additional resources provided from the State's designation of Corsica River as a targeted restoration watershed in 2005.

Water Quality Summary in Head-of-Bay Subestuaries – Sassafras River was added to our sampling during 2020 because it was the only non-mesohaline subestuary with an agriculturally dominated watershed. Sassafras River was contrasted with other non-mesohaline subestuaries sampled in the Head-of-Bay region. Estimated percentages of watershed in agriculture (3% – 68%), forest (23% – 39%), urban (8% – 71%), and wetlands (0.1% – 3%) varied throughout the Head-of-Bay subestuaries (MD DOP 2020b; Table 3-16; Figure 3-1). Water comprised a larger fraction of the Middle River drainage (28%) than in the Sassafras River (15%), Bohemia River (11%), Bush and Northeast Rivers (both 9%); Gunpowder River (5%; MD DOP 2020b) had the lowest fraction of water coverage (Table 3-16). Bohemia River was another subestuary with an agricultural watershed, but watershed effects on its fish community were difficult to detect because of the marine migrants that came through the nearby Chesapeake & Delaware Canal (J. Uphoff, personal communication).

Estimates of C/ha indicated that the Middle River has been subject to greatest development in the Head-of-Bay (Figure 3-17) and more than indicated by the Maryland Department of Planning urban category (Table 3-2). Bohemia and Sassafras Rivers were below the rural development target (IS 5% = 0.38). Time-series for Bohemia, Bush, Gunpowder, Northeast, and Sassafras Rivers started at a rural level of development in 1950 (C/ha ranged from 0.03 to 0.09; Figure 3-17). Middle River's level of development was already above the suburban level target (IS 10% = 0.86) in 1950 (C/ha = 0.97). Sassafras River's watershed has experienced the lowest growth (C/ha = 0.11 in 2018), while the most growth had occurred in Middle River's watershed (C/ha = 3.39 in 2018). Northeast River's C/ha progressed slowly, exceeding a rural level in 2003 (C/ha = 0.39). Gunpowder River progressed a little more quickly than Northeast River, exceeding a rural level in 1979 (C/ha = 0.39). Bush River developed above the C/ha target in 1976, exceeded the threshold (C/ha = 0.87) in 1991, and reached 1.37 (well-developed suburb) in 2005 (Figure 3-17).

In 2020, DO readings for the Sassafras River did not fall below the threshold (3.0 mg/L) or target (5.0 mg/L) levels even though HAB conditions were present throughout sampling (Table 3-17). In addition, the Chesapeake Bay program Sassafras River monitoring station (CBP ET3.1), located near the MD route 213 bridge, did not record DO readings below threshold or target levels in 2020. Since 1989, CBP ET3.1 has only recorded DO measurements below target level in 1991 (32%), 1992 (17%), and 1995 (25%; Figure 3-18); no threshold violations have been recorded. Other Head-of-Bay subestuaries sampled in previous years by FHEP all had target level breaches: 20% of Bohemia River readings; 2% of Bush River; 3% of Gunpowder River; 20% of Middle River; and 10% of Northeast River (Table 3-17). Only two subestuaries at the Head-of-Bay had bottom DO measurements that breached the threshold, Middle River (1%) and Northeast River (2%; Figure 3-19). The Chesapeake Bay Program Head-of-Bay monitoring station (CBP CB1.1), located at the mouth of the Susquehanna River, did not record DO readings below threshold or target levels since 1989 (Figure 3-18).

Sassafras River bottom DO measurements in 2020 ranged from 5.09 mg/L to 9.82 mg/L and median bottom DO was 6.83 mg/L (Figure 3-20). In 2020, Sassafras River bottom DO means were 7.40 mg/L at station 01, 6.73 mg/L at station 02, 7.15 mg/L at station 03, and 6.93 mg/L at station 04 (Figure 3-21). The CBP ET3.1 monitoring station bottom DO in 2020 ranged from 5.2 mg/L to 7.4 mg/L with a median bottom DO of 5.4 mg/L (Figure 3-18).
The other Head-of-Bay subestuaries (Bohemia, Bush, Gunpowder, Middle, and Northeast Rivers) station annual mean bottom DO readings fluctuated above and below the time-series median (Figure 3-21). Middle River was an exception; stations 01 and 02 sometimes diverged from stations 03 and 04. Bottom DO could not be collected at Gunpowder River stations 02 and 03 during 2016 because they were too shallow.

The overall mean and SE for bottom DO in Bohemia River for 2006 was 6.41 mg/L and 0.31, respectively; 7.21 mg/L and 0.24 for Bush River (2006 – 2010); 6.75 mg/L and 0.10 for Gunpowder River (2009 – 2016); 6.10 mg/L and 0.10 for Middle (2009 – 2017); 6.93 mg/L and 0.11 for Northeast River (2007 – 2017); and 7.06 mg/L and 0.24 for Sassafras River in 2020. The CBP CB1.1 monitoring station summer (July – September) overall mean and SE was 7.15 mg/ L and 0.06 from 1989 to 2020, respectively. Bottom DO measurements for CBP CB1.1 in 2020 were 5.6 mg/L in July; 7.0 mg/L and 7.1 mg/L in August; and 8.2 mg/L in September. During 2020, median summer bottom DO at CBP CB1.1 was 7.05 mg/L (Figure 3-18). Correlation analyses of annual survey median bottom DO among Head-of-Bay subestuaries suggested weak associations among Gunpowder River, Bush River, and Middle River, when they were sampled in adjacent years, and very weak associations that were either positive or negative among the remaining subestuaries, respectively. Bohemia and Sassafras Rivers were limited to only a single year of data and could not be used in this analysis.

Differences in mean bottom DO among stations in the each of the Head-of-Bay subestuaries for years sampled were not detected with ANOVA in Bohemia, Bush, Gunpowder, Northeast, and Sassafras Rivers (Table 3-18). An ANOVA of Middle River stations and bottom DO during 2009 – 2017 indicated significant differences among stations (F = 9.11; DF = 3; $P < 0.0001$; N = 201; Table 3-18). Tukey Studentized Range and Tukey Honestly Significant Difference (HSD) tests indicated that bottom DO at station 01 (located at the head of Dark Head Creek in front of Wilson Point Park landing) was significantly lower than downstream stations 03 and 04 (Figure 3-21). Station 02 (located at Clark Point where Middle River branch and Dark Head Creek converge) was significantly lower than downstream station 03 (Figure 3-21). Lower DO at the uppermost stations (station 01 and 02) indicated a negative influence of watershed development.

Median Secchi depths had variable annual ranges (Figure 3-22). Bohemia River median Secchi depths ranged from 0.2 m to 0.8 m during 2006; from 0.2 m to 0.6 m in Bush River during 2006 – 2010; from 0.2 m to 0.9 m in Gunpowder River during 2009 – 2016; from 0.0 m to 1.5 m in Middle River during 2009 – 2017; from 0.2 m to 1.0 m in Northeast River during 2007 – 2017; and from 0.2 m to 0.5 m in Sassafras River in 2020 (Figure 3-22). Gunpowder and Middle Rivers showed the greatest variation in Secchi depths during 2014 – 2016. In 2015, MD DNR biologists discovered and confirmed zebra mussel presence in the Middle River.

The pH in the low salinity subestuaries investigated between 2006 and 2020 ranged from 6.3 (Middle, 2014) to 9.45 (Northeast, 2017; Figure 3-23). Bohemia River median pH for 2006 was 7.33. Bush River median pH from 2006 to 2010 ranged from 6.95 (2007) to 7.90 (2009). Gunpowder River median pH from 2009 to 2016 ranged from 7.34 (2016) to 7.90 (2010). Middle River median pH from 2009 to 2017 ranged from 7.14 (2016) to 8.10 (2011). Northeast River median pH from 2007 to 2017 ranged from 7.80 (2008) to 8.70 (2010). Sassafras River median pH for 2020 was 8.27 (Figure 3-23). The yearly ranges of pH within Bush, Gunpowder, Middle, and Northeast Rivers varied slightly to considerably; Bohemia and Sassafras Rivers only had one year of pH measurements collected (Figure 3-23).

Head-of-Bay subestuaries were typically stable as tidal-fresh and oligohaline; however, some fluctuated between tidal-fresh and oligohaline. Oligohaline subestuaries consisted of Bohemia River, which was sampled in 2006; Bush River, 2006 – 2010; Gunpowder River, 2009 – 2016; and Middle River, 2009 – 2017. The tidal-fresh subestuaries sampled were Northeast River from 2007 to 2017 and Sassafras River in 2020. Salinity range varied the least in Northeast River and was greatest in Middle River for all years sampled (Figure 3-24). Highest salinity for the Head-of-Bay subestuaries differed by year in each subestuary, 3.5 ‰ in Bohemia River (2006), 3.6 ‰ in Bush River (2010), 6.74 ‰ in Gunpowder River (2016), 8.53 ‰ in Middle River (2016), 3.3 ‰ in Northeast River (2008), and 3.06 ‰ in Sassafras River (2020; Figure 3-24). Lowest salinity measurements differed by year in each subestuary, 2006 in Bohemia River (0.1 ‰), 2006 in Broad Creek (0.1 ‰), 2015 in Gunpowder (0.11 ‰), 2011 in Middle (0.5 ‰), 2009 and 2017 in Northeast River (0.06 ‰), and 2020 in Harris Creek (0.4 ‰; Figure 3-24).
In the fall of 2015, a fish kill occurred in Middle River. The Maryland Department of the Environment reported that the fish kill was caused by high amounts of toxic algae, *Karlodinium*

veneficum, whose toxin causes gill damage to fish when in high concentrations (MDE 2016; 2017). A toxic algae event occurred in the Sassafras in 2020, but a fish kill did not occur. A previous microcystin toxin event in the Sassafras River involving *Oscillatoria lemnosa* was noted in 2018; no fish kill was recorded (SR 2020).

2020 Finfish Community Summary - Geometric mean catch per seine haul ranged from 88 to 200 among the four subestuaries sampled during 2020 (Broad Creek, Miles River, Sassafras River, and Tred Avon River; Table 3-19). Geometric mean seine catches in 2020 ranked Broad Creek, 1st; Miles River, 2nd; Tred Avon River, 3rd; and Sassafras River, 4th. Between 20 and 31 species were encountered in seine samples (Table 3-19). Sassafras River seine catch was substantially smaller due to only three seine samples (first seine hauls only) examined in 2020, compared to the 12 samples in Miles River and Broad Creek, and the 24 samples in Tred Avon River.

A plot of species richness in seine samples against C/ha during 1989 – 2020 did not suggest a strong relationship in tidal-fresh, oligohaline, or mesohaline subestuaries (Figure 3-25). Tidal-fresh subestuary watersheds were represented by a limited range of C/ha (0.43 – 0.69). Oligohaline subestuary watersheds were represented by the widest range of C/ha (0.08 – 3.33) of the three salinity classes. Mesohaline subestuary watersheds were represented by a larger number of surveys (N = 72; C/ha range = 0.07 – 2.68) than tidal-fresh and oligohaline subestuaries (N = 22 and 36, respectively; Figure 3-25).

A total of 8,635 fish representing 34 species were captured by beach seines in 2020 (Table 3-19). Eleven species comprised 90% of the total fish caught in 2020, including (from greatest to least) Atlantic Silverside, Mummichog, White Perch (adults), Striped Killifish, Atlantic Menhaden, Bay Anchovy, Banded Killifish, and Spot. Atlantic Menhaden, White Perch, Bay Anchovy, and Spot represented target species. Four target species were present among species comprising 90% of the seine catch throughout all subestuaries: White Perch (adults) were present in this category in all three subestuaries; Atlantic Menhaden in two; and Bay Anchovy in two (Table 3-19).

Geometric mean catches per trawl were between 8 and 24 during 2020 (Table 3-20). All subestuaries had 24 samples (four stations) in 2020. Broad Creek had the greatest GM (24) and Miles River had the lowest (8); Sassafras River ranked, 2nd (GM = 22); and Tred Avon River (GM = 21) ranked, 3rd (Table 3-20). A plot of trawl GMs against C/ha (all subestuaries during 2003 – 2020) declined with development in mesohaline subestuaries and a possible negative threshold response at C/ha between 0.8 and 1.2 (Figure 3-26). Trawl GM catches did not exhibit an obvious decline with C/ha in tidal-fresh and oligohaline subestuaries (Figure 3-26).
Number of species captured by trawl in subestuaries sampled during 2020 ranged from 7 to 12 (Table 3-20). A plot of species richness in trawl samples against C/ha (all subestuaries during 2003 – 2020) did not indicate a relationship of development and number of species for tidal-fresh (species richness ranging from 14 to 25) or oligohaline subestuaries (species richness ranging from 12 to 26; Figure 3-27). Species richness (ranging from 3 to 23) declined in mesohaline subestuaries as C/ha advanced beyond the threshold (C/ha = 0.86 = 10% IS; Figure 3-27).
A total of 2,888 fish and 20 fish species were captured by bottom trawl during 2020 (Table 3-20). Five species comprised 90% of the total catch for 2020 (from greatest to least): White Perch (adult), Spot, Bay Anchovy, Hogchoker, White Perch (juvenile), and Atlantic Croaker; four of the five species were target species; Hogchoker was the exception. Target species comprising 90% of the catch in each of the four subestuaries sampled during 2020 were White Perch (adult)

and Spot in four subestuaries; Atlantic Croaker and Bay Anchovy each in two subestuaries; and White Perch (juvenile) in one subestuary (Table 3-20).

Subestuaries in 2020 had low GMs and species richness for their salinity classes and C/ha. Miles River had extensive system-wide DO issues and had a much lower trawl GM than the other subestuaries. Sassafras River had a HAB that caused water quality issues throughout most of the upper subestuary. Broad Creek had lower than normal bottom DOs. Tred Avon River has a localized upper-subestuary DO issue that has worsened since 2014. GMs appeared to bottom out following a decline of Bay Anchovies.

Finfish Community Summary in Mesohaline Tributaries and Subestuaries in Talbot County – Mesohaline subestuaries sampled with bottom trawl in 2020 had GMs ranked relatively low compared to previous years: Broad Creek ranked 73rd out of 86; Miles River, 84th; and Tred Avon River, 75th (Table 3-21). Annual GMs of catches of all species of finfish in 4.9 m bottom trawls in Broad Creek, Harris Creek, Miles River, and Tred Avon River for all sampling years and their 95% CIs were plotted (Figure 3-28). Geometric means during 2020 were the among the lowest for each subestuaries sampled (Figure 3-28). Each time-series sampled in 2020 had the potential for a GM that was at least an order of magnitude higher (Table 3-21).

Correlations of trawl GMs among the three Choptank tributaries did suggest coherence in annual relative abundance of finfish (Table 3-22). Strong positive correlations of GMs were present between Broad Creek and Harris Creek (r = 0.87, P = 0.05, N = 5); Broad Creek and Tred Avon River (r = 0.88, P = 0.009, N = 7); a positive, moderate correlation was present between Tred Avon River and Harris Creek (r = 0.69, P = 0.20, N = 5; Table 3-22). Correlations of beach seine GMs with bottom trawl GMs for Choptank tributaries were positive and strong to moderate, Broad Creek (r = 0.889, P = 0.006, N = 7); Harris Creek, (r = 0.988, P = 0.0001, N = 5); and Tred Avon River, (r = 0.890, P < 0.0001, N = 15; Table 3-23). Correlation between beach seine GM with bottom trawl GM for Miles River was positive and moderate (r = 0.790, P 0.21, but sample size (N = 4) was small (Table 3-23).

Bay Anchovy was the most abundant species found throughout the Choptank tributaries, making up greater than 50% of species present in systems when they were sampled during 2006 – 2020 (Figure 3-29). Two species in Broad Creek comprised the top 90% of finfish caught from 2012 to 2017 and in 2020, Bay Anchovy (87.8%) and Weakfish (2.8%); 27 species other species made up the remaining 10% of Harris Creek had three species in the top 90% during 2012 – 2016, Bay Anchovy (85.8%), White Perch (adults and juveniles; 2.6%), and Weakfish (2.1%), with 25 other species making up the remaining 10%. Five species were in the top 90% of finfish caught in the Tred Avon River from 2006 to 2020, Bay Anchovy (57.3%), Spot (16.7%), White Perch (adults and juveniles; 7.6%), Hogchoker (7.2%), and Striped Bass (adults and juveniles; 3.6%; Figure 3-29); all except Hogchoker, were target species; an additional 33 other species comprised the last 10% (Figure 3-29). Miles River species composition from 2003 to 2005 and in 2020 differed from the Choptank tributaries; the top 90% of species consisted of White Perch (adults and juveniles; 42.2%), Bay Anchovy (33.6%), and Striped Bass (adults and juveniles; 13.9%). Twenty-one species made up the remainder within Miles River. In these comparisons of samples with years combined, the number of other species appeared to be a function of how many years were sampled.

Species comprising the top 90% collected in Tred Avon River trawl samples was similar during 2019 – 2020 (4 species); 2011 and 2018 had the highest species richness (6 species; Figure 3-30). The usually common Bay Anchovy was not in the top 90% during 2020 and has not been since 2017. Atlantic Croaker reappeared in the top 90% of species in 2020; they were last noted in the top 90% in 2017. In Broad Creek, five species were present in the top 90% compared to one to three species in 2012 – 2017. In Miles River during 2020, four species were in the top 90%, compared to two and three during previously sampled years. The Choptank River tributaries and Miles River each had an increase in the species comprising the top 90% starting in 2018, but this appears to reflect reduced prevalence of Bay Anchovy (Figure 3-30). Bay anchovies transfer energy from zooplankton to higher levels of the food web and are a major prey for smaller piscivorous fishes in Chesapeake Bay, (Hartman and Brandt 1995; Christensen et al. 2009; Overton et al. 2015) and depletion could have ramifications for production of Striped Bass, Weakfish, and Bluefish in these subestuaries.

Percent similarity in trawl sample finfish species composition among stations 01 – 04 in the Tred Avon River was low but increased slightly between 2019 (7%) and 2020 (16%); percent similarity was above 50% during 2007 – 2017 (Figure 3-31). During 2006 and 2018 – 2020, the similarity index was below 25%, reflecting impacts of rainfall and low salinity on fish community composition (Figure 3-31). Percent similarity in Broad Creek fell, but did not show the same drastic drop that appeared in the Tred Avon River; Broad Creek remained above 50% and Harris Creek, above 40% for all years sampled (Figure 3-31). Previous analyses in 2018 (Uphoff et al. 2018), suggested wet years with lower salinity would have species composition dissimilar to dry years with higher salinity. Prevalent species in bottom trawl samples shifted during 2003 – 2020 (Figure 3-32). White Perch, Spot, and Bay Anchovy were predominant during 2003-2010; the latter two species predominated in 2012; Bay Anchovy predominated during 2013-2017; White Perch during 2018-2019; and Spot and White Perch during 2020. Low salinity in 2011 was not accompanied by loss of Bay Anchovy in all mesohaline tributaries as it was during 2018-2020 (Figure 3-32).

Tred Avon River adult White Perch trawl GMs in 2009 – 2011 and 2014 – 2016 fell below the median time-series GM (6; Figure 3-33). The greatest White Perch GM in Tred Avon River was in 2012 (14) and the least was in 2010 (2). During 2016, adult White Perch GMs in Broad and Harris Creeks and Tred Avon River were similar (4; Figure 3-33). In 2020, White Perch GMs in Broad Creek (8) and Tred Avon River (10; Figure 3-33) were greater than the time-series median; Broad Creek recorded the highest White Perch GM in 2020. The greatest White Perch GM in Miles River was in 2004 (26) and the least was in 2020 (5); only years, 2004 and 2005, were above the median time-series GM (12; Figure 3-33). Correlations of White Perch GMs among Choptank tributaries were weakly positive (Table 3-24). Miles River was not included in correlations because only one year of data available (2020).

Modified PSDs for White Perch in Choptank tributaries (Broad Creek, Harris Creek, and Tred Avon River) and in Miles River for 4.9 m trawl samples varied greatly among subestuaries and years, but were generally lower in Tred Avon River (Table 3-25; Figure 3-34). Between 2019 and 2020, modified PSD declined in Broad Creek but was still relatively high; both Miles and Tred Avon Rivers modified PSDs were greater than the previous year sampled. Tred Avon River modified PSD ranged from 4.7% (2012) to 51.1% (2018), and an increase after 2016 reflected the size progression of the strong 2011 year-class (juvenile index = 35.2, respectively; Durell and Weedon 2020) into harvestable size. The decline after 2018 may indicate recruitment of two top quartile year-classes (2014 and 2015 juvenile indices = 14.4 and 14.8, respectively) into the stock category. The 2011 year-class followed a stretch of lesser year-classes during the 2000s (Durell and Weedon 2020). The less developed Choptank River tributary, Harris Creek, had higher modified PSDs for trawl samples than Tred Avon River during corresponding sampling years (2012 – 2016). Modified PSDs for trawl samples in Broad Creek fluctuated above and below modified PSDs in Tred Avon River for trawl samples during 2012 – 2017 and 2020 (Table 3-25; Figure 3-34). Modified PSDs were more than often greater than 10%; about one-third of modified PSDs calculated were below 10% (Table 3-25). Miles River had the highest modified PSD in 2020 at 48.6%, a value similar to 2003 (Table 3-25). Miles River exhibited a substantial reduction in modified PSDs between 2003 and 2004-2005, decreasing from 58.4% (2003) to 0% (Table 3-25; Figure 3-34).

Seine GMs (relative abundance of all species combined) in 2020 for Choptank River tributaries, Broad Creek and Tred Avon River indicated similar status for years in common (Figure 3-35). Finfish seine GMs in the three Choptank River tributaries, Broad Creek, Harris Creek, and Tred Avon River, were highest during 2015; 2012 – 2016 represented years in common among these three tributaries. Seine GMs for all finfish in Tred Avon River samples were lowest in 2008 (77). Broad Creek and Harris Creek had their lowest GMs in 2012 (106 and 131, respectively). Tred Avon River seine GMs had a sharp, one year peak in 2015 and have generally been in a similar range over the rest of the time-series (Figure 3-35).

Seven species comprised the top 90% of finfish in beach seines when all years were combined in Broad Creek and Tred Avon River (Figure 3-36). Tred Avon River's (2006 – 2020) top species were Atlantic Silverside (37.3%), Atlantic Menhaden (18.0%), White Perch (15.2%), Striped Killifish (7.7%), Mummichog (7.4%), Bay Anchovy (3.4%), and Banded Killifish (2.8%); an additional 41 other species (8.3%) were collected in Tred Avon River. Broad Creek (2012 – 2017, 2020) also had 7 species in the top 90% of finfish collected, Atlantic Silverside (34.8%), Atlantic Menhaden (20.8%), Striped Killifish (10.9%), Mummichog (8.9%), Banded Killifish (8.7%), White Perch (4.7%), and Sheepshead Minnow (2.9%); an additional 32 other species (8.2%) were collected in Broad Creek. Harris Creek, not sampled in 2020, had only 6 species in the top 90% of finfish from 2012 to 2016, with an additional 32 other species collected. Miles River (2003 – 2005, 2020) had 6 species in the top 90% of finfish collected, Atlantic Silverside (38.3%), Atlantic Menhaden (20.9%), White Perch (17.2%), Striped Killifish (8.8 %), Mummichog (4.0%), and Striped Bass (3.9%); 6.8% were other species (Figure 3-36). All species in the top 90% in the subestuaries were target species, except Atlantic Silverside, Banded Killifish, Mummichog, Sheepshead Minnow, and Striped Killifish.

In 2020, finfish trawl catches in Broad Creek, Miles River, and Tred Avon River bottom channel fell to their lowest levels for all years sampled, while inshore seine catches were average and remained steady. Tred Avon River trawl catches were similar to 2018, both Broad Creek and Miles River trawl catches declined substantially since they were last sampled. Typically, low finfish catches in the bottom channel within mesohaline systems are associated with increased development and low DO measurements. However, Miles River, an agricultural system, had the greatest bottom DO violations in 2020 and a corresponding crash in trawl finfish catches; a similar crash in bottom channel finfish catches was observed in another agricultural subestuary, Wye River, in 2018 and 2019 (Uphoff et al. 2019). A change in the species present and richness in bottom trawl catches in 2020 was notable for Broad Creek, Miles River, and Tred Avon River (Figure 3-30); all mesohaline systems saw a noteworthy shift in species composition in bottom trawl catches from 2018 to 2020 as well (Figure 3-32). Bay Anchovy was completely absent in Tred Avon River, but were present in Broad Creek and Miles River. Spot increased noticeably in 2020. The changes in species composition could be due to the changes in salinity, as well as increased development and DO violations.

Finfish Community Summary in Head-of-Bay Subestuaries – Geometric means of catches of all species sampled in Head-of-Bay subestuaries since 2003 varied considerably among years. Sassafras River had a GM of 25 in 2020 and ranked 33rd out of 35 fresh-tidal subestuaries sampled with bottom trawls since 2003 (Table 3-21). Bohemia River, previously sample in 2006, ranked 31st out of 33 with a GM of 115. Bush River, sampled during 2006 – 2010, achieved its highest ranking in 2010 with a GM of 473 (3rd). Gunpowder River, previously sampled during 2009 – 2016, achieved its highest ranking in 2010 with a GM of 401 (5th). Middle River previously sampled from 2009 to 2017, ranked 2nd with a GM of 520 in 2011. Northeast River, sampled during 2007 – 2017, had its greatest GM (392) ranking 2nd in 2010. Annual GMs of catches of all species of finfish in 4.9 m bottom trawls in Head-of-Bay subestuaries for all sampling years and their 95% CIs were plotted (Figure 3-37). Bush River GMs ranged from 153 (2006) to 474 (2010); Gunpowder River, ranged from 147 (2013) to 402 (2010); Middle River, ranged from 75 (2017) to 520 (2011); and Northeast River, ranged from 96 (2016) to 392 (2010). Modified PSD data revealed White Perch primarily use Head-of-Bay subestuaries as nursery habitat. Modified PSDs fluctuated between 0% and 1.4% (Table 3-26). The earliest quality size White Perch appeared in the Head-of-Bay subestuaries was in 2011, prior years indicate that only stock size White Perch were caught while sampling. After 2011, quality size White Perch are regularly present in the Head-of-Bay subestuaries sampled although modified PSDs were extremely low.

Sassafras River bottom trawl catches for 2020 were composed of White Perch (adults and juveniles; 73%), Spot (14%), Channel Catfish (4%), and other species (7 species; 9%; Figure 3-38). White Perch was the dominant species for all years combined in all the Head-of-Bay subestuaries trawl samples and ranged from 57% to 83% of all finfish. Bay Anchovies were prevalent throughout the Head-of-Bay subestuaries, but they were absent in some years in some subestuaries. Gizzard Shad were present occasionally in trawl samples from the Northeast, Bush, and Gunpowder rivers (Figure 3-39).

Seine GM for 2020 at the NRMA station in the Sassafras River was 88 and was substantially greater than the trawl GM (25). Twenty species were observed in the three seine hauls conducted from July to September; twice as many species were observed in shallow water habitat compared to bottom water habitat. Fish abundance was not impacted by DO since it was above target level throughout shallow and bottom water habitat.

Nine species comprised the top 90% of finfish in Sassafras River NRMA beach seines in 2020 (Figure 3-40). Sassafras River's (2020) top species were White Perch (juveniles and adults; 34.5%), Atlantic Silverside (20.2%), Striped Bass (juveniles and adults; 13.9%), Atlantic Croaker (6.4%), Gizzard Shad (5.6%), Pumpkinseed (3.7%), Spottail Shiner (3.0%), Atlantic Needlefish (2.2%), and Golden Shiner (1.9%); an additional 11 other species (8.6%) were collected in Sassafras River. Five of the nine species in the top 90% in the subestuaries were target species; nontarget species were Atlantic Silverside, Pumpkinseed, Atlantic Needlefish, and Golden Shiner.

2020 Sampling Summary – In spite of having fairly similar land uses in their watersheds (primarily agriculture), habitat conditions varied in the three Talbot County subestuaries surveyed in 2020. In Tred Avon River, a watershed approaching the development threshold, bottom DO continued to have numerous target and threshold DO violations at station 01 (Easton) indicating development as its root cause. Broad Creek median bottom DO was within previous years' ranges and no threshold violations were recorded; however, all stations fell below the time-series median for the first time. Poor bottom DO was observed at all stations in Miles River during 2020; threshold DO violations were similar in 2005. Other water quality metrics (pH, salinity, and Secchi depth) sampled during 2020 were within previous years' ranges for Broad Creek, Miles River, and Tred Avon River.

Finfish catches in trawls sampling bottom water habitat in the Talbot County subestuaries in 2020 were the lowest among all sampling years and particularly low in Miles River. Species composition changed slightly. Atlantic Croaker were present throughout most of the Talbot County subestuaries sampled; Spot and Hogchoker presence was widespread in 2020. Bay Anchovy remained in the top 90% of species in Broad Creek and Miles River, but were not prevalent enough in the Tred Avon River to avoid being grouped into the other species category. White Perch GMs in the Choptank River tributaries indicated a modest population in 2020; highly variable fluctuations in White Perch populations have been observed in previous years. Modified PSDs for trawl and seine samples located in the Choptank tributaries sampled in 2020 had greater population densities of White Perch of interest to anglers. Inshore seine catches remained steady, although not at the highest levels previously observed.

We sampled Sassafras River, a fresh-tidal subestuary with a predominately agricultural watershed, in 2020. However, due to a microcystis event during sampling season, FHEP was unable to collect finfish data from inshore by seining. Seine data acquired from Juvenile Index (JI) monitoring station Sassafras River Natural Resource Management Area (NRMA) allowed limited insight into the finfish community for 2020, which indicated higher species richness inshore. Harmful algal blooms (HABs) appear to be a major negative habitat feature of low salinity subestuaries in the Head-of-Bay region; HABS have occurred in Gunpowder (2004 and 2017), Middle River (2015), and Sassafras River (2018 and 2020), that have greatly contrasting dominant land uses (urban in Gunpowder and Middle, agricultural in Sassafras; MDE 2020). These HABs did not result in depleted DO in any of our surveys. However, 22 fish kill cases from 1984 to 2019 were attributed to HABs (MDE 2020). Finfish composition in Sassafras River was comparable to the other Head-of-Bay subestuaries with White Perch predominating. The Head-of-Bay subestuaries we have sampled were primarily habitat for smaller White Perch and modified PSDs were very low. Other species that are notable in the Sassafras River during 2020 were Spot and Channel Catfish; Spot relative abundance in our surveys was the strongest since 2010.

During 2018, heavy rainfall and high freshwater discharge into the Chesapeake Bay and its subestuaries may have impacted the upper- and mid-Bay subestuaries with lower salinities, lower DO, and resulted in smaller finfish catches that lingered into 2019 and 2020. Overall, we saw minimal changes in water quality parameters in Broad Creek and Miles River, and moderate changes were observed in Tred Avon River. Our assessment of habitat, particularly the subestuaries sampled for Talbot County, provided additional insight into the subestuaries and what can be expected during dry and wet years and how long the effects linger.

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Tables

Table 3-1. Percent impervious cover (IS), structures per hectare (C / ha), watershed area (land hectares), area of tidal water (water hectares), and salinity class for the subestuaries sampled in 2020.

2020 Sampled Subestuaries						
Area	**Subestuary**	**IS**	**C/ha**	**Land Hectares**	**Water Hectares**	**Salinity Class**
Upper-Bay	Sassafras River	2.4	0.11	19,595	3,461	Tidal-Fresh
Mid-Bay	Broad Creek	4.3	0.30	4,730	6,344	Mesohaline
Mid-Bay	Miles River	3.9	0.26	11,072	3,035	Mesohaline
Mid-Bay	Tred Avon River	9.1	0.77	9,561	3,086	Mesohaline

Table 3-2. Estimates of structures per hectare (C / ha) and land use percentages from Maryland Department of Planning (2002 and 2010) for subestuaries sampled 2003 – 2020.

River	Year	C/ha	Agriculture	Wetland	Forest	Urban
Breton Bay	2003	0.27	23.8	0.8	56.1	18.7
Breton Bay	2004	0.28	23.8	0.8	56.1	18.7
Breton Bay	2005	0.30	23.8	0.8	56.1	18.7
Broad Creek	2012	0.29	42.6	0.4	25.4	31.5
Broad Creek	2013	0.30	42.6	0.4	25.4	31.5
Broad Creek	2014	0.30	42.6	0.4	25.4	31.5
Broad Creek	2015	0.30	42.6	0.4	25.4	31.5
Broad Creek	2016	0.30	42.6	0.4	25.4	31.5
Broad Creek	2017	0.30	42.6	0.4	25.4	31.5
Broad Creek	2020	0.30	42.6	0.4	25.4	31.5
Bush River	2006	1.41	25.4	3.2	35.0	36.2
Bush River	2007	1.43	25.4	3.2	35.0	36.2
Bush River	2008	1.45	25.4	3.2	35.0	36.2
Bush River	2009	1.46	25.4	3.2	35.0	36.2
Bush River	2010	1.47	18.0	3.2	29.9	47.8
Chester River	2007	0.14	66.5	2.0	25.8	5.8
Chester River	2008	0.14	66.5	2.0	25.8	5.8
Chester River	2009	0.15	66.5	2.0	25.8	5.8
Chester River	2010	0.15	64.2	2.0	24.7	8.9
Chester River	2011	0.15	64.2	2.0	24.7	8.9
Chester River	2012	0.15	64.2	2.0	24.7	8.9
Chester River	2018	0.15	64.2	2.0	24.7	8.9
Chester River	2019	0.15	64.2	2.0	24.7	8.9
Corsica River	2003	0.17	64.3	0.4	27.4	7.9
Corsica River	2004	0.18	64.3	0.4	27.4	7.9
Corsica River	2005	0.19	64.3	0.4	27.4	7.9
Corsica River	2006	0.21	64.3	0.4	27.4	7.9
Corsica River	2007	0.22	64.3	0.4	27.4	7.9
Corsica River	2008	0.24	64.3	0.4	27.4	7.9
Corsica River	2010	0.24	60.4	0.1	25.5	13.2
Corsica River	2011	0.25	60.4	0.1	25.5	13.2
Corsica River	2012	0.25	60.4	0.1	25.5	13.2
Corsica River	2018	0.27	60.4	0.1	25.5	13.2
Corsica River	2019	0.27	60.4	0.1	25.5	13.2
Gunpowder River	2009	0.72	30.6	1.0	32.1	35.6
Gunpowder River	2010	0.72	30.6	1.0	32.1	35.6
Gunpowder River	2011	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2012	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2013	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2014	0.73	30.6	1.0	32.1	35.6
Gunpowder River	2015	0.74	30.6	1.0	32.1	35.6
Gunpowder River	2016	0.74	30.6	1.0	32.1	35.6
Harris Creek	2012	0.39	44.9	5.6	19.7	29.8
Harris Creek	2013	0.39	44.9	5.6	19.7	29.8
Harris Creek	2014	0.39	44.9	5.6	19.7	29.8
Harris Creek	2015	0.39	44.9	5.6	19.7	29.8
Harris Creek	2016	0.39	44.9	5.6	19.7	29.8

Table 3-2 (Cont).

Langford Creek	2006	0.07	71.6	1.5	23.0	3.9
Langford Creek	2007	0.07	71.6	1.5	23.0	3.9
Langford Creek	2008	0.07	71.6	1.5	23.0	3.9
Langford Creek	2018	0.07	70.2	1.5	20.4	8.0
Langford Creek	2019	0.07	70.2	1.5	20.4	8.0
Magothy River	2003	2.68	2.6	0.0	27.8	69.5
Mattawoman Creek	2003	0.76	11.9	1.2	59.4	27.4
Mattawoman Creek	2004	0.79	11.9	1.2	59.4	27.4
Mattawoman Creek	2005	0.81	11.9	1.2	59.4	27.4
Mattawoman Creek	2006	0.83	11.9	1.2	59.4	27.4
Mattawoman Creek	2007	0.86	11.9	1.2	59.4	27.4
Mattawoman Creek	2008	0.87	11.9	1.2	59.4	27.4
Mattawoman Creek	2009	0.88	11.9	1.2	59.4	27.4
Mattawoman Creek	2010	0.90	9.3	2.8	53.9	34.2
Mattawoman Creek	2011	0.91	9.3	2.8	53.9	34.2
Mattawoman Creek	2012	0.90	9.3	2.8	53.9	34.2
Mattawoman Creek	2013	0.91	9.3	2.8	53.9	34.2
Mattawoman Creek	2014	0.93	9.3	2.8	53.9	34.2
Mattawoman Creek	2015	0.93	9.3	2.8	53.9	34.2
Mattawoman Creek	2016	0.93	9.3	2.8	53.9	34.2
Middle River	2009	3.30	4.5	2.2	27.9	63.9
Middle River	2010	3.32	3.4	2.1	23.3	71.0
Middle River	2011	3.33	3.4	2.1	23.3	71.0
Middle River	2012	3.33	3.4	2.1	23.3	71.0
Middle River	2013	3.34	3.4	2.1	23.3	71.0
Middle River	2014	3.35	3.4	2.1	23.3	71.0
Middle River	2015	3.36	3.4	2.1	23.3	71.0
Middle River	2016	3.38	3.4	2.1	23.3	71.0
Middle River	2017	3.38	3.4	2.1	23.3	71.0
Miles River	2003	0.24	53.7	0.9	27.2	18.1
Miles River	2004	0.24	53.7	0.9	27.2	18.1
Miles River	2005	0.24	53.7	0.9	27.2	18.1
Miles River	2020	0.26	49.0	0.8	26.7	23.1
Nanjemoy Creek	2003	0.08	15.1	4.1	73.1	7.6
Nanjemoy Creek	2008	0.09	15.1	4.1	73.1	7.6
Nanjemoy Creek	2009	0.09	15.1	4.1	73.1	7.6
Nanjemoy Creek	2010	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2011	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2012	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2013	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2014	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2015	0.09	12.4	4.1	68.7	14.7
Nanjemoy Creek	2016	0.09	12.4	4.1	68.7	14.7
Northeast River	2007	0.44	36.7	0.1	42.7	20.1
Northeast River	2008	0.44	36.7	0.1	42.7	20.1
Northeast River	2009	0.45	36.7	0.1	42.7	20.1
Northeast River	2010	0.46	31.1	0.1	38.6	28.9
Northeast River	2011	0.46	31.1	0.1	38.6	28.9

Table 3-2 (Cont.)

Northeast River	2012	0.47	31.1	0.1	38.6	28.9
Northeast River	2013	0.47	31.1	0.1	38.6	28.9
Northeast River	2014	0.48	31.1	0.1	38.6	28.9
Northeast River	2015	0.48	31.1	0.1	38.6	28.9
Northeast River	2016	0.49	31.1	0.1	38.6	28.9
Northeast River	2017	0.49	31.1	0.1	38.6	28.9
Piscataway Creek	2003	1.30	12.8	0.3	45.8	40.6
Piscataway Creek	2006	1.38	12.8	0.3	45.8	40.6
Piscataway Creek	2007	1.40	12.8	0.3	45.8	40.6
Piscataway Creek	2009	1.43	12.8	0.3	45.8	40.6
Piscataway Creek	2010	1.45	10.0	0.2	40.4	47.0
Piscataway Creek	2011	1.46	10.0	0.2	40.4	47.0
Piscataway Creek	2012	1.47	10.0	0.2	40.4	47.0
Piscataway Creek	2013	1.49	10.0	0.2	40.4	47.0
Piscataway Creek	2014	1.50	10.0	0.2	40.4	47.0
Rhode/West Rivers	2003	0.55	34.1	0.8	45.3	19.8
Rhode/West Rivers	2004	0.56	34.1	0.8	45.3	19.8
Rhode/West Rivers	2005	0.56	34.1	0.8	45.3	19.8
Sassafras River	2020	0.11	64.1	1.3	25.9	8.3
Severn River	2003	2.06	8.6	0.2	35.2	55.8
Severn River	2004	2.09	8.6	0.2	35.2	55.8
Severn River	2005	2.15	8.6	0.2	35.2	55.8
Severn River	2017	2.36	5.0	0.2	28.0	65.1
South River	2003	1.24	15.2	0.4	45.6	38.8
South River	2004	1.25	15.2	0.4	45.6	38.8
South River	2005	1.27	15.2	0.4	45.6	38.8
St. Clements River	2003	0.19	38.6	0.9	48.6	11.8
St. Clements River	2004	0.20	38.6	0.9	48.6	11.8
St. Clements River	2005	0.20	38.6	0.9	48.6	11.8
Tred Avon River	2006	0.69	50.1	1.0	21.6	27.2
Tred Avon River	2007	0.71	50.1	1.0	21.6	27.2
Tred Avon River	2008	0.73	50.1	1.0	21.6	27.2
Tred Avon River	2009	0.74	50.1	1.0	21.6	27.2
Tred Avon River	2010	0.75	43.2	0.8	21.6	33.6
Tred Avon River	2011	0.75	43.2	0.8	21.6	33.6
Tred Avon River	2012	0.75	43.2	0.8	21.6	33.6
Tred Avon River	2013	0.76	43.2	0.8	21.6	33.6
Tred Avon River	2014	0.76	43.2	0.8	21.6	33.6
Tred Avon River	2015	0.76	43.2	0.8	21.6	33.6
Tred Avon River	2016	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2017	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2018	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2019	0.77	43.2	0.8	21.6	33.6
Tred Avon River	2020	0.77	43.2	0.8	21.6	33.6
Wicomico River	2003	0.19	34.7	4.6	48.5	12.0
Wicomico River	2010	0.21	31.6	4.6	44.9	18.7
Wicomico River	2011	0.21	31.6	4.6	44.9	18.7
Wicomico River	2012	0.21	31.6	4.6	44.9	18.7
Wicomico River	2017	0.22	31.6	4.6	44.9	18.7
Wye River	2007	0.10	67.7	0.7	23.5	8.1
Wye River	2008	0.10	67.7	0.7	23.5	8.1
Wye River	2018	0.10	64.9	0.6	23.0	10.9
Wye River	2019	0.10	64.9	0.6	23.0	10.9

Table 3-3. Summary of water quality parameter statistics collected during both seine and trawl samples for subestuaries in 2020. Measurements for pH were calculated from H+ concentrations and converted back to pH.

		Surface Measurements					Bottom Measurements					
System	Statistics	Temp ©	DO (mg / L)	Cond (umhols)	Salinity	pH	Temp ©	DO (mg / L)	Cond (umhols)	Salinity	pH	Secchi
Broad Creek	Mean	27.94	7.55	18375.76	10.95	8.02	27.57	5.57	18832.29	11.13	7.83	0.54
	Standard Error	0.43	0.12	274.86	0.11	9.33	0.52	0.14	220.72	0.14	9.07	0.03
	Median	28.14	7.50	18725.00	11.10	8.03	28.07	5.69	18906.00	11.17	7.86	0.50
	Mode	.	7.22	19327.00	11.41	7.95	.	.	18906.00	10.77	7.97	0.50
	Kurtosis	-0.42	-0.35	12.99	-0.64	0.11	-0.12	0.24	0.20	0.62	2.69	-0.28
	Skewness	-0.68	0.15	-2.98	-0.44	0.05	-0.71	-0.70	0.14	0.25	0.03	0.98
	Minimum	22.48	5.95	10533.00	9.45	8.24	22.34	4.01	16745.00	9.81	7.61	0.40
	Maximum	31.46	9.14	20174.00	11.94	7.75	31.25	6.75	21440.00	12.88	8.03	0.90
	Count	36	37	37	37	37	24	24	24	24	24	24
Miles River	Mean	27.88	6.50	17776.32	10.45	7.82	26.90	3.42	18936.14	11.20	7.55	0.49
	Standard Error	0.64	0.27	344.52	0.22	9.00	0.83	0.52	430.64	0.28	8.59	0.05
	Median	29.35	6.45	18107.00	10.64	7.88	28.29	3.27	19399.00	11.47	7.58	0.50
	Mode	30.43	4.24	.	10.41	7.60	.	.	.	.	7.70	0.50
	Kurtosis	0.63	4.23	1.43	1.44	.	0.25	-0.58	5.15	4.90	.	0.13
	Skewness	-1.35	-1.22	-1.28	-1.27	0.23	-1.19	0.68	-2.01	-1.94	0.65	0.75
	Minimum	18.99	0.28	11781.00	6.67	8.22	19.19	0.40	12396.00	7.04	7.32	0.20
	Maximum	32.60	8.94	20361.00	12.20	7.56	31.33	7.74	20990.00	12.61	7.91	1.00
	Count	37	37	37	37	37	21	21	21	21	21	23
Sassafras River	Mean	28.14	9.83	2271.82	1.16	8.38	27.27	7.06	2412.75	1.24	8.09	0.33
	Standard Error	0.72	0.37	187.42	0.10	8.95	0.78	0.24	230.07	0.13	8.84	0.02
	Median	29.93	9.30	2206.00	1.12	8.79	28.55	6.83	2201.00	1.12	8.26	0.30
	Mode	30.19	.	.	0.92	9.33	.	6.56	.	.	8.69	0.40
	Kurtosis	0.94	-0.43	1.04	1.37	-0.39	0.49	0.00	1.61	2.22	0.91	-1.18
	Skewness	-1.42	0.52	0.77	0.89	-0.29	-1.29	0.64	1.01	1.19	-0.08	0.16
	Minimum	19.69	7.08	591.00	0.28	9.50	19.31	5.09	699.00	0.34	7.63	0.20
	Maximum	31.83	14.05	5085.00	2.74	7.71	31.18	9.82	5635.00	3.06	8.82	0.50
	Count	28	28	28	28	28	24	24	24	24	24	24
Tred Avon River	Mean	28.29	6.91	16253.40	9.47	7.78	28.11	4.35	17148.58	10.04	7.60	0.44
	Standard Error	0.40	0.19	303.24	0.19	8.83	0.46	0.33	278.80	0.18	8.64	0.02
	Median	28.55	6.80	16678.00	9.74	7.88	28.45	4.69	17577.50	10.35	7.67	0.48
	Mode	26.96	6.27	16936.00	11.04	7.66	.	4.63	.	10.48	7.82	0.50
	Kurtosis	-0.81	0.00	-0.78	-0.82	0.23	-0.79	1.44	-0.85	-0.84	0.67	-0.79
	Skewness	-0.43	-0.09	-0.61	-0.60	-0.05	-0.44	-1.37	-0.63	-0.65	0.04	-0.40
	Minimum	21.78	3.70	11796.00	6.71	8.30	23.74	0.27	14508.00	8.39	7.27	0.25
	Maximum	32.52	9.97	18764.00	11.05	7.36	31.35	6.49	18784.00	11.07	7.98	0.60
	Count	48	48	48	48	48	24	24	24	24	24	24

Table 3-4. Percentages of all dissolved oxygen (DO) measurements and all bottom DO measurements that did not meet target (5.0 mg / L) or threshold (3.0 mg / L) conditions for each subestuary sampled in 2020. C / ha = structures per hectare. N = number of samples.

				All DO		Bottom DO	
Subestuary	**Salinity Class**	**C/ha**	**N**	**% < 5.0 mg/L**	**N**	**% < 5.0 mg/L**	**% < 3.0 mg/L**
Broad Creek	Mesohaline	0.30	79	6%	24	21%	0%
Miles River	Mesohaline	0.26	76	38%	21	81%	48%
Sassafras River	Tidal-Fresh	0.11	75	0%	24	0%	0%
Tred Avon River	Mesohaline	0.77	96	27%	24	63%	17%

Table 3-5. Subestuaries sampled during 2003 – 2020, by salinity class, with C / ha (watershed structures per hectare), mean annual surface and bottom temperatures, and mean annual surface and bottom dissolved oxygen (mg / L).

			Temperature (ºC)		Dissolved Oxygen (mg / L)	
River	Year	C / ha	Surface	Bottom	Surface	Bottom
			Mesohaline			
Blackwater River	2006	0.04	28.14	27.98	5.27	4.12
Breton Bay	2003	0.27	26.40	25.69	8.10	3.75
	2004	0.28	27.01	25.95	7.36	3.73
	2005	0.30	28.62	27.51	6.98	3.99
Broad Creek	2012	0.29	27.50	26.60	8.30	5.97
	2013	0.30	27.30	26.49	7.26	5.76
	2014	0.30	27.62	26.64	7.65	5.78
	2015	0.30	28.05	27.05	7.93	6.63
	2016	0.30	29.16	28.33	7.30	6.16
	2017	0.30	27.01	26.29	7.50	6.11
	2020	0.30	27.94	27.57	7.55	5.57
Chester River	1995	0.11	26.70	26.32	7.58	6.28
	1996	0.11	23.86	23.63	7.62	6.71
	1997	0.11	27.02	26.32	7.66	6.37
	1998	0.11	28.00	27.49	7.40	6.26
	2007	0.14	25.59	24.18	5.38	4.53
	2008	0.14	25.09	25.35	5.24	4.20
	2009	0.15	25.79	25.77	5.74	5.21
	2010	0.15	26.12	24.97	5.84	5.71
	2011	0.15	25.31	25.41	4.90	4.28
	2012	0.15	27.12	27.12	4.67	4.39
	2018	0.15	27.54	26.90	6.83	6.00
	2019	0.15	27.45	27.05	6.75	5.77
Corsica River	2003	0.17	25.90	26.13	6.50	4.67
	2004	0.18	27.18	26.88	5.57	4.57
	2005	0.19	28.54	28.14	6.48	3.08
	2006	0.21	27.39	26.84	7.55	4.05
	2007	0.22	25.94	25.82	6.24	4.22
	2008	0.24	26.20	25.22	7.32	4.21
	2010	0.24	34.36	26.62	5.69	5.01
	2011	0.25	27.00	27.01	5.30	3.28
	2012	0.25	27.79	27.47	4.71	3.40
	2018	0.27	27.23	26.71	7.02	5.12
	2019	0.27	27.24	27.04	6.82	4.39
Fishing Bay	2006	0.04	26.23	25.28	7.24	6.79
Harris Creek	2012	0.39	26.55	26.42	7.44	6.35
	2013	0.39	26.39	26.05	7.02	6.01
	2014	0.39	27.61	26.68	6.84	4.84
	2015	0.39	26.62	26.62	7.19	6.56
	2016	0.39	27.82	27.75	6.65	6.02
Langford Creek	2006	0.07	27.05	26.52	6.95	5.68
	2007	0.07	26.23	25.48	6.69	5.68
	2008	0.07	27.47	26.65	6.85	5.05
	2018	0.07	27.08	31.78	6.40	5.10
	2019	0.07	27.77	27.51	6.69	5.07
Magothy River	2003	2.68	25.70	25.31	7.30	2.04
Miles River	2003	0.24	25.50	25.60	6.50	4.09
	2004	0.24	25.75	25.64	6.08	5.47
	2005	0.24	28.03	27.44	5.96	3.31
	2020	0.26	27.88	26.90	6.50	3.42

Table 3-5 (Cont.)

Rhode River	2003	0.47	25.00	24.69	7.10	4.80
	2004	0.47	27.00	26.95	6.58	5.39
	2005	0.48	27.78	27.16	6.50	4.03
Severn River	2003	2.06	26.30	24.75	7.60	1.57
	2004	2.09	27.42	26.18	7.05	2.64
	2005	2.15	28.01	26.23	7.07	0.96
	2017	2.36	26.93	26.07	6.86	1.78
South River	2003	1.24	25.40	24.56	7.60	2.61
	2004	1.25	25.79	25.48	6.46	3.77
	2005	1.27	27.57	26.67	6.02	2.49
St. Clements River	2003	0.19	26.00	25.29	8.20	3.48
	2004	0.20	26.08	25.78	6.84	4.61
	2005	0.20	27.12	26.36	6.85	4.42
Transquaking River	2006	0.03	26.68	22.75	5.75	5.85
Tred Avon River	2006	0.69	27.12	26.72	6.18	5.34
	2007	0.71	26.85	26.59	6.49	5.39
	2008	0.73	26.28	25.61	6.90	4.83
	2009	0.74	26.15	26.03	7.37	6.31
	2010	0.75	27.47	26.93	7.08	5.26
	2011	0.75	28.48	28.18	6.82	5.11
	2012	0.75	27.27	27.16	7.02	5.47
	2013	0.76	26.79	26.39	7.15	5.00
	2014	0.76	26.66	26.51	6.12	5.90
	2015	0.76	28.00	27.60	6.92	5.54
	2016	0.77	28.89	28.44	7.27	5.15
	2017	0.77	26.49	26.13	7.01	5.04
	2018	0.77	27.79	27.34	7.34	4.81
	2019	0.77	28.62	28.22	6.79	4.49
	2020	0.77	28.29	28.11	6.91	4.35
West River	2003	0.64	24.90	24.31	7.40	4.84
	2004	0.65	26.83	26.59	7.37	5.58
	2005	0.66	27.96	27.15	6.72	3.99
Wicomico River	2003	0.19	25.40	23.83	7.00	5.85
	2010	0.21	25.43	25.30	6.06	5.21
	2011	0.21	27.08	26.89	5.57	4.30
	2012	0.22	27.57	27.38	6.59	5.44
	2017	0.22	26.70	25.73	7.55	4.62
Wye River	2007	0.10	26.75	26.45	7.08	5.70
	2008	0.10	26.98	26.22	5.70	5.11
	2018	0.10	28.36	27.78	8.07	4.67
	2019	0.10	27.68	27.67	6.33	4.68
			Oligohaline			
Bohemia River	2006	0.11	26.79	26.02	7.01	6.41
Bush River	2006	1.41	25.48	24.28	7.96	7.47
	2007	1.43	27.02	26.42	7.68	6.54
	2008	1.45	26.59	24.20	9.00	5.43
	2009	1.46	25.88	24.34	9.41	8.54
	2010	1.47	27.72	23.80	7.79	7.04

Table 3-5 (Cont.)

Gunpowder River	2009	0.72	25.71	26.05	7.39	6.79
	2010	0.72	25.17	25.91	7.89	7.13
	2011	0.73	25.09	25.56	8.28	7.14
	2012	0.73	26.48	25.93	8.19	6.71
	2013	0.73	25.85	27.46	8.05	6.10
	2014	0.73	26.65	26.15	7.28	5.76
	2015	0.74	27.51	27.65	8.02	6.63
	2016	0.74	27.70	26.46	7.43	6.18
Middle River	2009	3.30	26.50	25.78	7.27	6.07
	2010	3.32	24.65	24.20	8.44	7.11
	2011	3.33	27.13	26.42	8.35	7.33
	2012	3.33	28.05	26.60	8.82	5.21
	2013	3.34	27.12	26.46	7.58	5.79
	2014	3.35	26.56	26.01	7.55	6.04
	2015	3.36	28.47	27.20	8.20	6.23
	2016	3.38	28.87	27.82	7.56	5.69
	2017	3.38	25.54	25.17	7.80	5.36
Nanjemoy Creek	2003	0.08	25.90	28.80	7.30	4.96
	2008	0.09	27.53	26.58	7.85	6.65
	2009	0.09	26.31	24.64	7.05	7.49
	2010	0.09	26.50	24.80	7.66	7.02
	2011	0.09	29.34	28.55	6.13	5.30
	2012	0.09	26.18	25.92	6.73	5.98
	2013	0.09	26.88	26.30	6.76	5.86
	2014	0.09	26.78	26.36	7.66	6.25
	2015	0.09	27.40	27.10	7.16	6.32
	2016	0.09	28.49	28.21	6.86	5.16
			Tidal Fresh			
Mattawoman Creek	2003	0.76	26.00	25.75	9.00	8.81
	2004	0.79	27.33	27.14	8.34	7.95
	2005	0.81	28.77	28.09	7.74	7.27
	2006	0.83	27.05	26.44	7.10	6.50
	2007	0.86	26.89	26.85	6.70	6.48
	2008	0.87	26.40	24.52	7.97	6.33
	2009	0.88	26.20	26.64	7.92	7.86
	2010	0.90	26.21	26.10	6.95	6.62
	2011	0.91	27.08	27.46	6.33	6.51
	2012	0.90	26.70	26.82	7.40	7.00
	2013	0.91	26.35	25.94	9.22	8.40
	2014	0.93	26.73	26.24	7.48	6.17
	2015	0.93	27.91	26.84	8.66	7.74
	2016	0.93	28.47	28.03	6.96	6.54
Northeast River	2007	0.44	26.83	26.43	9.73	7.75
	2008	0.44	25.35	24.98	8.43	7.70
	2009	0.45	26.33	25.55	9.35	7.36
	2010	0.46	25.90	26.21	7.76	6.78
	2011	0.46	25.97	25.71	6.87	5.79

Table 3-5 (Cont.)

	2012	0.47	27.78	27.59	7.88	6.03
	2013	0.47	26.61	26.11	9.33	7.06
	2014	0.48	26.94	26.52	7.72	6.81
	2015	0.48	26.66	26.23	7.84	6.17
	2016	0.49	27.95	26.86	8.81	7.10
	2017	0.49	26.38	25.68	9.38	7.80
Piscataway Creek	2003	1.30	25.60	24.63	10.20	8.33
	2006	1.38	28.16	24.97	8.70	6.85
	2007	1.40	27.47	26.00	8.57	7.60
	2009	1.43	26.72	27.07	8.56	6.62
	2010	1.45	27.07	25.08	9.36	7.63
	2011	1.46	28.25	30.07	9.05	9.47
	2012	1.47	27.92	25.51	9.53	9.34
	2013	1.49	27.19	26.22	9.87	7.65
	2014	1.50	26.98	26.28	8.66	7.33
Sassafras River	2020	0.11	28.14	27.27	9.83	7.06

Table 3-6. Pearson correlations (r) of mean survey surface and bottom dissolved oxygen (DO; mg / L) with water temperatures at depth (surface and bottom) and with watershed development (C / ha = structures per hectare) from subestuaries sampled during 2003 – 2020, by salinity class. Level of significance = *P*. N = sample size.

DO Depth	Statistics	Temperature	C / ha
		Mesohaline	
Surface	r	-0.003	0.234
	P	0.979	0.029
	N	87	87
Bottom	r	0.074	-0.589
	P	0.493	<.0001
	N	87	87
		Oligohaline	
Surface	r	-0.308	0.432
	P	0.081	0.012
	N	33	33
Bottom	r	-0.601	-0.061
	P	0.0002	0.737
	N	33	33
		Tidal Fresh	
Surface	r	0.041	-0.162
	P	0.779	0.265
	N	49	49
Bottom	r	0.060	-0.098
	P	0.684	0.501
	N	49	49

Table 3-7. Pearson correlations (r) of C/ha for mesohaline subestuaries sampled during 2003 – 2020 with Maryland Department of Planning (DOP) land use categories. Pearson correlations (r) between land use categories estimated by MD DOP for 2002 and 2010. *P* = level of significance. N = sample size. Duplicate entries of C/ha for mesohaline subestuaries from 2003 to 2020 were not included in analysis.

			Land Use Categories			
	Statistics	**C/ha**	**Agriculture**	**Forest**	**Wetland**	**Urban**
C/ha	r					
	P	1				
	N					
Agriculture	r	-0.75				
	P	<.0001	1			
	N	73				
Forest	r	0.07	-0.57			
	P	0.57	<.0001	1		
	N	73	73			
Wetland	r	-0.27	0.02	-0.01		
	P	0.02	0.86	0.94	1	
	N	73	73	73		
Urban	r	0.89	-0.81	-0.12	-0.01	
	P	<.0001	<.0001	0.32	0.92	1
	N	73	73	73	73	

Table 3-8. Statistics and parameter estimates for regional (western and eastern shores) linear regressions of median bottom dissolved oxygen (DO) versus percent agricultural coverage.

Linear Model	**Western Shore: Median Bottom DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	52.61	52.61	53.84	<.0001	
Residual	20	19.54	0.98			
Total	21	72.15				
$r^2 = 0.7292$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.50	0.48	1.04	0.31	-0.50	1.50
Agriculture (%)	0.13	0.02	7.34	<.0001	0.09	0.17
Linear Model	**Eastern Shore: Median Bottom DO = Agriculture (%)**					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	7.03	7.03	10.20	0.0023	
Residual	58	39.98	0.69			
Total	59	47.01				
$r^2 = 0.1495$						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	6.92	0.57	12.12	<.0001	5.78	8.06
Agriculture (%)	-0.03	0.01	-3.19	0.0023	-0.05	-0.01

Table 3-9. Statistics and parameter estimates for a quadratic regression of median bottom dissolved oxygen (DO) versus percent agricultural coverage (western and eastern shore combined).

Linear Model	**Median Bottom DO = Agriculture (%) Coverage**					
ANOVA	df	SS	MS	F	Significance F	
Regression	2	87.93	43.97	52.58	<0.0001	
Residual	79	66.06	0.84			
Total	81	153.99				
r^2 = 0.571						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	-0.05	0.48	-0.11	0.91	-1.00	0.90
Agriculture (%)	0.22	0.02	9.36	<0.0001	0.18	0.27
Agriculture (%)^2	-0.002	0.0003	-8.07	<0.0001	-0.003	-0.002

Table 3-10. Percent of watershed in major land use categories estimated by Maryland Department of Planning (DOP) for each of the Choptank River subestuaries. Land use estimates are determined from MD DOP 2010 data. The first four land use categories contain only land area (hectares) of the watershed; water area (hectares) is removed from each of these categories. Water is the percent of water hectares per area of water and land.

	Talbot County Subestuaries			
		Choptank River Subestuaries		
Land Use Category	**Miles River**	**Broad Creek**	**Harris Creek**	**Tred Avon River**
Agriculture	49	43	45	43
Forest	27	25	20	22
Urban	23	31	30	34
Wetlands	1	<1	6	1
Water	22	57	62	24

Table 3-11. Percentages of all dissolved oxygen (DO) measurements (surface, middle, and bottom) and all bottom DO measurements that did not meet target (5.0 mg / L) or threshold (3.0 mg / L) conditions during July-September for years sampled. N = sample size.

				All DO		Bottom DO	
Subestuary	**Year**	**C / ha**	**N**	**% < 5.0 mg/L**	**N**	**% < 5.0 mg/L**	**% < 3.0 mg/L**
Broad Creek	2012	0.29	83	7%	24	17%	4%
	2013	0.30	78	10%	23	30%	0%
	2014	0.30	81	6%	24	21%	0%
	2015	0.30	82	1%	23	0%	0%
	2016	0.30	76	4%	22	9%	0%
	2017	0.30	72	3%	22	9%	0%
	2020	0.30	79	6%	24	21%	0%
Harris Creek	2012	0.39	82	0%	23	0%	0%
	2013	0.39	83	0%	24	0%	0%
	2014	0.39	84	1%	23	4%	0%
	2015	0.39	85	0%	24	0%	0%
	2016	0.39	79	3%	23	9%	0%
Miles River	2003	0.24	96	45%	25	7%	24%
	2004	0.24	81	26%	23	0%	0%
	2005	0.24	86	44%	24	17%	46%
	2020	0.26	76	37%	21	81%	48%
Tred Avon River	2006	0.69	91	19%	24	38%	0%
	2007	0.71	93	11%	23	26%	4%
	2008	0.73	89	24%	21	48%	14%
	2009	0.74	95	6%	24	13%	0%
	2010	0.75	89	20%	24	38%	13%
	2011	0.75	82	22%	21	48%	10%
	2012	0.75	94	10%	24	29%	0%
	2013	0.76	103	15%	26	31%	15%
	2014	0.76	96	11%	24	21%	0%
	2015	0.76	96	8%	24	21%	13%
	2016	0.77	96	13%	24	38%	13%
	2017	0.77	89	17%	24	42%	13%
	2018	0.77	110	17%	28	50%	14%
	2019	0.77	96	30%	24	71%	17%
	2020	0.77	96	27%	24	63%	17%

Table 3-12. Pearson correlations (r) of annual median bottom dissolved oxygen (DO; mg / L) for Broad Creek, Harris Creek, and Tred Avon River with year and among subestuaries. *P* = level of significance. N = number of annual median DO measurements for each subestuary sampled.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	-0.07		
	P	0.91	1	
	N	5		
Tred Avon River	r	0.54	0.73	
	P	0.21	0.16	1
	N	7	5	

Table 3-13. Pearson correlations (r) of annual survey median Secchi depths for Broad Creek, Harris Creek, and Tred Avon River among subestuaries. *P* = level of significance. N = number of annual survey median Secchi depths.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.972		
	P	0.006	1	
	N	5		
Tred Avon River	r	0.841	0.928	
	P	0.018	0.023	1
	N	7	5	

Table 3-14. Pearson correlations (r) of annual median pH for Broad Creek, Harris Creek, and Tred Avon River among subestuaries. *P* = level of significance. N = number of annual survey median pH estimates.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.937		
	P	0.019	1	
	N	5		
Tred Avon River	r	0.490	0.174	
	P	0.264	0.779	1
	N	7	5	

Table 3-15. Pearson correlations (r) of annual survey median salinity (‰) for Broad Creek, Harris Creek, and Tred Avon River among subestuaries. *P* = level of significance. N = number of annual survey median salinity estimates.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.990		
	P	0.001	1	
	N	5		
Tred Avon River	r	0.995	0.979	
	P	<.0001	0.004	1
	N	7	5	

Table 3-16. Percent of watershed in major land use categories estimated by Maryland Department of Planning (DOP 2010) for each of Head-of-Bay subestuary. The first four land use categories contain only land area (hectares) of the watershed; water area (hectares) is removed from each of these categories. Water is the percent of water hectares per area of water and land.

	Head-of-Bay Subestuaries					
Land Use Category	Bohemia River	Bush River	Gunpowder River	Middle River	Northeast River	Sassafras River
Agriculture	68.0	18.0	30.6	3.4	31.1	64.1
Forest	22.8	29.9	32.1	23.3	38.6	25.9
Urban	7.7	47.8	35.6	71.0	28.9	8.3
Wetlands	1.5	3.2	1.0	2.1	0.1	1.3
Water	10.7	9.2	5.3	27.9	9.2	15.0

Table 3-17. Percent of all dissolved oxygen (DO) measurements (surface, middle, and bottom) and all bottom DO measurements that did not meet target (5.0 mg / L) or threshold (3.0 mg / L) conditions during July – September, by year sampled, for Head-of-Bay subestuaries. N = number of DO measurements.

				All DO	Bottom DO		
Subestuary	Year	C / ha	N	% < 5.0 mg/L	N	% < 5.0 mg/L	% < 3.0 mg/L
Bohemia River	2006	0.11	72	17%	20	20%	0%
Bush River	2006	1.41	49	0%	11	0%	0%
Bush River	2007	1.43	48	2%	11	0%	0%
Bush River	2008	1.45	45	2%	3	33%	0%
Bush River	2009	1.46	50	2%	8	13%	0%
Bush River	2010	1.47	41	7%	5	0%	0%
Gunpowder River	2009	0.72	65	2%	18	0%	0%
Gunpowder River	2010	0.72	58	0%	13	0%	0%
Gunpowder River	2011	0.73	41	0%	9	0%	0%
Gunpowder River	2012	0.73	60	3%	15	7%	0%
Gunpowder River	2013	0.73	51	4%	12	17%	0%
Gunpowder River	2014	0.73	49	0%	15	0%	0%
Gunpowder River	2015	0.74	54	0%	13	0%	0%
Gunpowder River	2016	0.74	50	2%	8	0%	0%
Middle River	2009	3.30	70	7%	22	18%	5%
Middle River	2010	3.32	78	4%	22	14%	0%
Middle River	2011	3.33	60	2%	18	0%	0%
Middle River	2012	3.33	77	13%	24	42%	0%
Middle River	2013	3.34	71	7%	23	17%	0%
Middle River	2014	3.35	68	1%	23	4%	0%
Middle River	2015	3.36	67	7%	24	17%	0%
Middle River	2016	3.38	59	8%	22	23%	0%
Middle River	2017	3.39	72	11%	24	33%	8%
Northeast River	2007	0.44	86	3%	23	9%	0%
Northeast River	2008	0.44	74	7%	19	11%	0%
Northeast River	2009	0.45	78	1%	23	4%	0%
Northeast River	2010	0.46	71	1%	17	0%	0%
Northeast River	2011	0.46	88	13%	24	33%	13%
Northeast River	2012	0.47	82	7%	24	21%	0%
Northeast River	2013	0.47	85	2%	24	8%	0%
Northeast River	2014	0.48	80	1%	24	4%	0%
Northeast River	2015	0.48	85	5%	24	13%	4%
Northeast River	2016	0.49	84	0%	24	0%	0%
Northeast River	2017	0.49	93	1%	24	4%	0%
Sassafras River	2020	0.11	75	0%	24	0%	0%

Table 3-18. Basic summary statistics (F, DF, *P*, and N) for Head-of-Bay subestuaries linear regressions of bottom dissolved oxygen (mg/L) versus stations.

Linear Model	**Bottom DO (mg/L) = Stations**			
ANOVA	**F**	**DF**	***P***	**N**
Bohemia River	0.08	3	0.97	19
Bush River	1.00	2	0.38	37
Gunpowder River	0.42	3	0.74	102
Middle River	9.11	3	< 0.0001	201
Northeast River	1.69	3	0.17	249
Sassafras River	0.28	3	0.84	23

Table 3-19. Beach seine catch summary, 2020. C / ha = structures per hectare. GM CPUE = geometric mean catch per seine sample. Sassafras River data acquired from Juvenile Index monitoring station Sassafras River Natural Resource Management Area (NRMA). Italics designate target species. Young of the year or juveniles = JUV.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Broad Creek	2	12	21	Mummichog *Atlantic Menhaden* Altantic Silverside Banded Killifish *White Perch (Adults)* Striped Killifish Rainwater Killifish	0.3	2,668	200
Miles River	3	12	24	Striped Killifish Atlantic Silverside *Atlantic Menhaden* *Bay Anchovy* *White Perch (Adults)* Mummichog	0.26	2,198	161
Sassafras River (JI NRMA)	1	3	20	Atlantic Silverside *White Perch (Juv)* *White Perch (Adults)* *Striped Bass (Juv)* *Atlantic Croaker* *Gizzard Shad* Pumpkinseed *Spottail Shiner* Atlantic Needlefish Golden Shiner *Striped Bass (Adults)*	0.11	267	88
Tred Avon River	4	24	31	Atlantic Silverside Mummichog *White Perch (Adults)* *Bay Anchovy* Striped Killifish Sheepshead Minnow Inland Silverside	0.77	3,769	142
Grand Total	6	27	39	Atlantic Silverside Mummichog *White Perch (Adults)* Striped Killifish *Atlantic Menhaden* *Bay Anchovy* Banded Killifish *Spot*		8,902	

Table 3-20. Bottom trawl catch summary, 2020. C / ha = structures per hectare. GM CPUE = geometric mean catch per trawl sample. Italics designate target species. Young-of-the-year or juveniles = JUV.

River	Stations Sampled	Number of Samples	Number of Species	Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Broad Creek	4	24	12	*Spot* *Bay Anchovy* *White Perch (Adult)* *Atlantic Croaker* Hogchoker	0.3	749	24
Miles River	4	24	7	*Spot* *Bay Anchovy* *White Perch (Adult)* Hogchoker	0.26	386	8
Sassafras River	4	24	10	*White Perch (Adult)* *Spot* *White Perch (Juv)* Channel Catfish	0.11	1,188	22
Tred Avon River	4	24	12	*Spot* *White Perch (Adult)* Hogchoker *Atlantic Croaker*	0.77	565	21
Grand Total	16	96	20	*White Perch (Adult)* *Spot* *Bay Anchovy* Hogchoker *White Perch (Juv)* *Atlantic Croaker*		2,888	

Table 3-21. Subestuaries sampled during 2003 – 2020, grouped by salinity class and ranked by annual 4.9 m trawl catch geometric mean (GM) of all species combined.

River	Year	GM	Rank
Mesohaline			
Broad Creek	2014	401	1
Corsica River	2003	378	2
Miles River	2003	313	3
Broad Creek	2012	294	4
Langford Creek	2007	273	5
West River	2003	272	6
Tred Avon River	2010	264	7
Rhode River	2003	262	8
Chester River	2011	259	9
Langford Creek	2006	258	10
Corsica River	2004	251	11
Corsica River	2011	238	12
Corsica River	2009	218	13
Tred Avon River	2014	192	14
Harris Creek	2014	174	15
Corsica River	2006	174	16
Chester River	2010	172	17
Wye River	2007	170	18
Rhode River	2005	163	19
Corsica River	2012	162	20
Langford Creek	2008	161	21
Corsica River	2010	161	22
Tred Avon River	2008	155	23
Tred Avon River	2012	155	24
Harris Creek	2012	155	25
Chester River	2007	152	26
Broad Creek	2017	148	27
Broad Creek	2016	147	28
Broad Creek	2013	142	29
Tred Avon River	2007	137	30
Corsica River	2007	131	31
Fishing Bay River	2006	131	32
Transquaking River	2006	131	33
Chester River	2012	130	34
West River	2005	125	35
Tred Avon River	2016	121	36
Chester River	2008	120	37
Wicomico River	2010	120	38
Wye River	2008	114	39
Wicomico River	2012	110	40
Corsica River	2005	109	41
Tred Avon River	2009	104	42
Broad Creek	2015	103	43
Tred Avon River	2017	98	44
Tred Avon River	2011	92	45

Table 3-21 (Cont.)

Harris Creek	2013	89	46
Corsica River	2008	86	47
Miles River	2004	82	48
Wicomico River	2017	81	49
Tred Avon River	2015	80	50
Chester River	2019	78	51
Tred Avon River	2013	77	52
Chester River	2009	76	53
Tred Avon River	2006	76	54
Miles River	2005	72	55
Wicomico River	2011	65	56
Wicomico River	2003	59	57
St. Clements River	2005	56	58
South River	2003	55	59
Harris Creek	2016	51	60
Langford Creek	2019	42	61
Harris Creek	2015	40	62
Rhode River	2004	38	63
Tred Avon River	2019	37	64
South River	2005	35	65
Blackwater River	2006	35	66
Breton Bay	2005	34	67
West River	2004	34	68
Broad Creek	2020	34	69
Magothy River	2003	33	70
Corsica River	2019	32	71
St. Clements River	2003	31	72
Tred Avon River	2020	29	73
Langford Creek	2018	27	74
South River	2004	21	75
Tred Avon River	2018	20	76
Wye River	2019	19	77
Breton Bay	2003	18	78
St. Clements River	2004	17	79
Breton Bay	2004	16	80
Severn River	2017	16	81
Corsica River	2018	16	82
Wye River	2018	12	83
Miles River	2020	9	84
Severn River	2004	5	85
Severn River	2003	5	86
Severn River	2005	3	87
Oligohaline			
Nanjemoy Creek	2013	576	1
Middle River	2011	520	2
Bush River	2010	473	3
Nanjemoy Creek	2015	416	4
Gunpowder River	2010	401	5
Nanjemoy Creek	2014	396	6

Table 3-21 (Cont.)

Gunpowder River	2011	394	7
Nanjemoy Creek	2011	385	8
Bush River	2007	324	9
Bush River	2009	319	10
Middle River	2010	315	11
Nanjemoy Creek	2010	309	12
Nanjemoy Creek	2016	297	13
Middle River	2009	292	14
Gunpowder River	2009	289	15
Middle River	2015	286	16
Nanjemoy Creek	2009	284	17
Middle River	2016	261	18
Middle River	2014	251	19
Nanjemoy Creek	2012	224	20
Gunpowder River	2012	224	21
Gunpowder River	2014	219	22
Gunpowder River	2015	218	23
Bush River	2008	210	24
Nanjemoy Creek	2008	209	25
Gunpowder River	2016	206	26
Middle River	2013	181	27
Bush River	2006	152	28
Middle River	2012	148	29
Gunpowder River	2013	147	30
Bohemia River	2006	115	31
Nanjemoy Creek	2003	93	32
Middle River	2017	74	33
Tidal-Fresh			
Mattawoman Creek	2014	580	1
Northeast River	2010	392	2
Piscataway Creek	2011	320	3
Northeast River	2014	291	4
Northeast River	2011	290	5
Piscataway Creek	2010	290	6
Mattawoman Creek	2013	283	7
Mattawoman Creek	2004	252	8
Piscataway Creek	2014	221	9
Mattawoman Creek	2015	217	10
Mattawoman Creek	2011	208	11
Northeast River	2009	198	12
Northeast River	2012	191	13
Mattawoman Creek	2005	187	14
Northeast River	2013	186	15
Piscataway Creek	2013	184	16
Northeast River	2008	152	17
Northeast River	2015	150	18
Northeast River	2007	149	19
Mattawoman Creek	2016	149	20
Mattawoman Creek	2003	144	21

Table 3-21 (Cont.)

Piscataway Creek	2012	119	22
Northeast River	2017	105	23
Piscataway Creek	2009	105	24
Northeast River	2016	96	25
Mattawoman Creek	2010	84	26
Mattawoman Creek	2006	75	27
Mattawoman Creek	2012	72	28
Mattawoman Creek	2007	56	29
Piscataway Creek	2003	42	30
Piscataway Creek	2006	28	31
Mattawoman Creek	2008	27	32
Sassafras River	2020	25	33
Piscataway Creek	2007	8	34
Mattawoman Creek	2009	6	35

Table 3-22. Pearson correlations (r) of annual 4.9 m trawl finfish catch geometric mean (GM; all species combined) for Broad Creek, Harris Creek, and Tred Avon River, with year and among subestuaries. *P* = level of significance. N = number of annual GMs for each subestuary.

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	0.87		
	P	0.05	1	
	N	5		
Tred Avon River	r	0.88	0.69	
	P	0.009	0.20	1
	N	7	5	

Table 3-23. Pearson correlations (r) of annual beach seine GM (all finfish species) against annual 4.9 m trawl catch GM for Choptank subestuaries and Miles River. Level of significance of Pearson correlation = P. Sample size (N) for the number of GM measurements for each subestuary sampled.

		Seine Geometric Mean			
Trawl Geometric Mean	**Statistics**	**Broad Creek**	**Harris Creek**	**Tred Avon River**	**Miles River**
Broad Creek	r	0.899			
	P	0.006			
	N	7			
Harris Creek	r		0.998		
	P		0.0001		
	N		5		
Tred Avon River	r			0.890	
	P			<0.0001	
	N			15	
Miles River	r				0.790
	P				0.210
	N				4

Table 3-24. Pearson correlations (r) of annual 4.9 m trawl catch of adult White Perch geometric mean (GM) for Choptank subestuaries, Broad Creek, Harris Creek, and Tred Avon River, with year and among each subestuary. Level of significance of Pearson correlation = P. Sample size (N) for the number of adult White Perch GM measurements for each subestuary sampled. Bold numbers indicate a significant associations (α = 0.05).

	Statistics	Broad Creek	Harris Creek	Tred Avon River
Broad Creek	r			
	P	1		
	N			
Harris Creek	r	-0.459		
	P	0.437	1	
	N	5		
Tred Avon River	r	0.402	0.377	
	P	0.372	0.532	1
	N	7	5	

Table 3-25. Modified proportional stock density (PSD) of White Perch in Choptank River subestuaries and Miles River are the proportion of 4.9m trawl samples with quality length or greater White Perch. N_{TOTAL} is the total number of White Perch (all juveniles and adults) captured in trawl catches. Number of L_{STOCK} is the number of all adult White Perch (adults age +1). Number of $L_{QUALITY}$ is the number of harvestable adults (≥ 200 mm).

Subestuary	Years	N_{TOTAL}	N L_{STOCK}	N $L_{QUALITY}$	Modified PSD
Broad Creek	2012	86	86	4	4.7%
	2013	42	42	3	7.1%
	2014	38	38	14	36.8%
	2015	214	21	1	4.8%
	2016	60	51	15	29.4%
	2017	16	16	5	31.3%
	2020	40	40	6	15.0%
Harris Creek	2012	106	106	45	42.5%
	2013	244	237	26	11.0%
	2014	52	51	11	21.6%
	2015	39	39	27	69.2%
	2016	96	96	41	42.7%
Miles River	2003	6704	185	108	58.4%
	2004	941	798	0	0%
	2005	1081	537	0	0%
	2020	74	74	36	48.6%
Tred Avon River	2006	323	321	45	14.0%
	2007	404	375	22	5.9%
	2008	234	234	29	12.4%
	2009	120	120	30	25.0%
	2010	21	15	6	40.0%
	2011	809	76	19	25.0%
	2012	570	570	27	4.7%
	2013	225	225	11	4.9%
	2014	62	60	4	6.7%
	2015	282	80	18	22.5%
	2016	102	102	6	5.9%
	2017	126	118	39	33.1%
	2018	105	88	45	51.1%
	2019	554	553	147	26.6%
	2020	165	165	56	33.9%

Table 3-26. Modified proportional stock density (PSD) of White Perch in Head-of-Bay subestuaries, Bohemia, Bush, Gunpowder, Middle, Northeast, and Sassafras Rivers, are the proportion of 4.9m trawl samples with quality length or greater White Perch. N_{TOTAL} is the total number of White Perch (all juveniles and adults) captured in trawl catches. Number of L_{STOCK} is the number of all adult White Perch (adults age +1). Number of $L_{QUALITY}$ is the number of harvestable adults (≥ 200 mm).

Subestuary	Years	N_{TOTAL}	N L_{STOCK}	N $L_{QUALITY}$	Modified PSD
Bohemia River	2006	1725	745	0	0%
Bush River	2006	1543	1133	0	0%
	2007	5651	477	0	0%
	2008	3263	1929	0	0%
	2009	4695	1130	0	0%
	2010	7392	1905	0	0%
Gunpowder River	2009	6076	743	0	0%
	2010	3913	917	0	0%
	2011	6164	254	0	0%
	2012	4013	1776	6	0.3%
	2013	1645	789	5	0.6%
	2014	2946	182	0	0%
	2015	4859	290	0	0%
	2016	1254	890	7	0.8%
Middle River	2009	5851	1664	0	0%
	2010	4586	866	0	0%
	2011	11978	801	0	0%
	2012	2830	2267	2	0.1%
	2013	1679	1137	2	0.2%
	2014	3098	336	4	1.2%
	2015	7108	283	4	1.4%
	2016	1248	744	2	0.3%
	2017	1439	374	3	0.8%
Northeast River	2007	2961	1222	0	0%
	2008	2967	1468	0	0%
	2009	4681	2841	0	0%
	2010	7929	2104	0	0%
	2011	6692	1485	20	1.3%
	2012	6699	4962	63	1.3%
	2013	4781	3875	13	0.3%
	2014	6929	2254	5	0.2%
	2015	3828	2247	4	0.2%
	2016	2073	968	2	0.2%
	2017	3123	557	4	0.7%
Sassafras River	2020	873	722	4	0.6%

Figures

Figure 3-1. Map illustrating subestuaries sampled in summer 2020: Sassafras River (1), Miles River (2), Tred Avon River (3), and Broad Creek (4), and their land use categories. Land use data is based on Maryland Department of Planning (DOP) 2010 land use land cover data. Figure includes previously sampled subestuaries referenced throughout this report (black stars; Harris Creek, Bohemia River, Northeast River, Gunpowder River, Middle River, and Bush River).

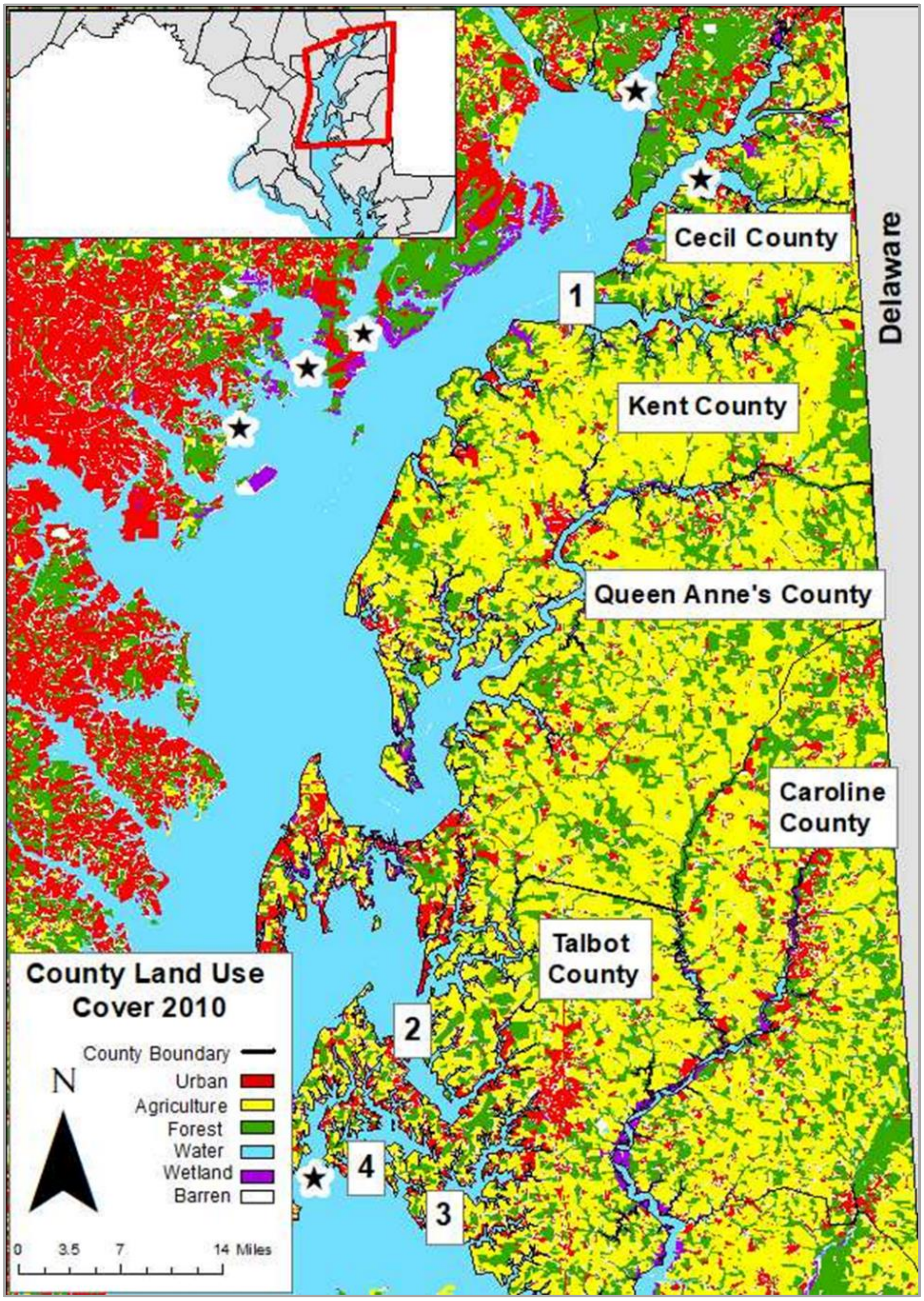


Figure 3-2. Map indicating 2020 locations of sampling sites for subestuaries, Broad Creek, Miles River, Sassafras River, and Tred Avon River.

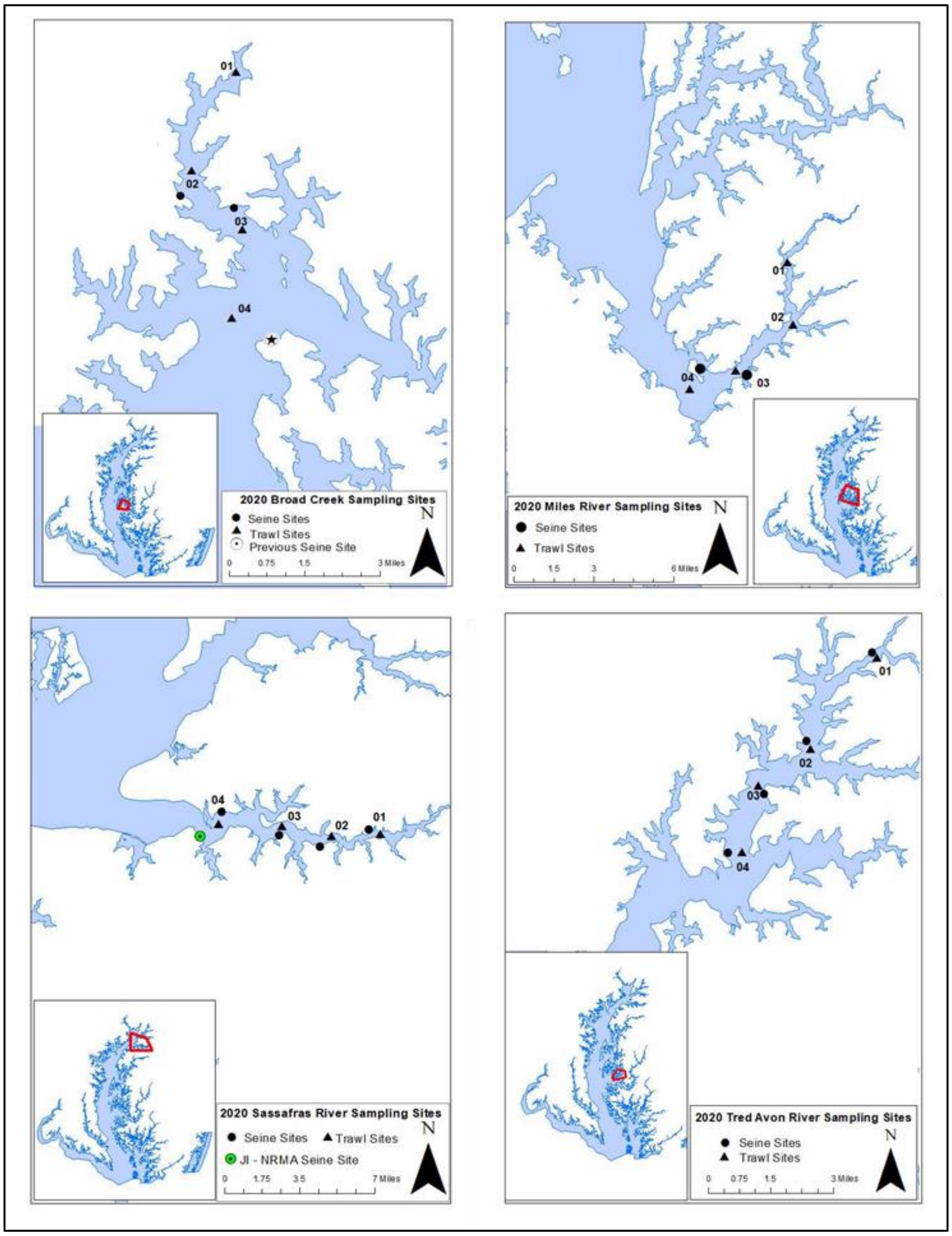


Figure 3-3. Map indicating the locations of seine and bottom trawl sites for the lower Choptank River tributaries, Broad Creek (2012 – 2017, 2020), Harris Creek (2012 – 2016), and Tred Avon River (2006 – 2020), and the subestuary, Miles River (2003 – 2005, 2020).

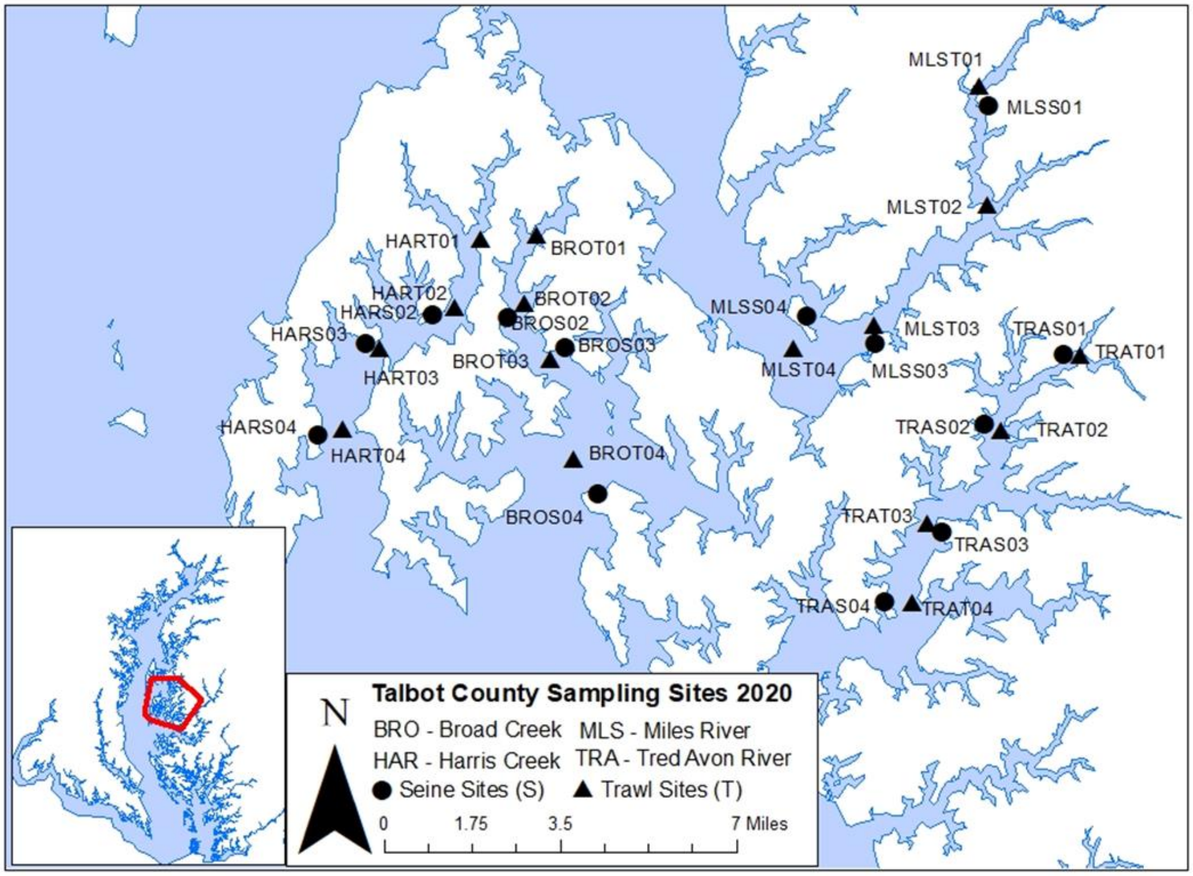


Figure 3-4. Map indicating the locations of seine and bottom trawl sites for Head-of-Bay subestuaries , Bohemia River (2006), Bush River (2006 – 2010), Gunpowder River (2009 – 2016), Middle River (2009 – 2017), Northeast River (2007 – 2017), and Sassafras River (2020); including juvenile index (JI) seine site at Sassafras Natural Resource Management Area (NRMA).

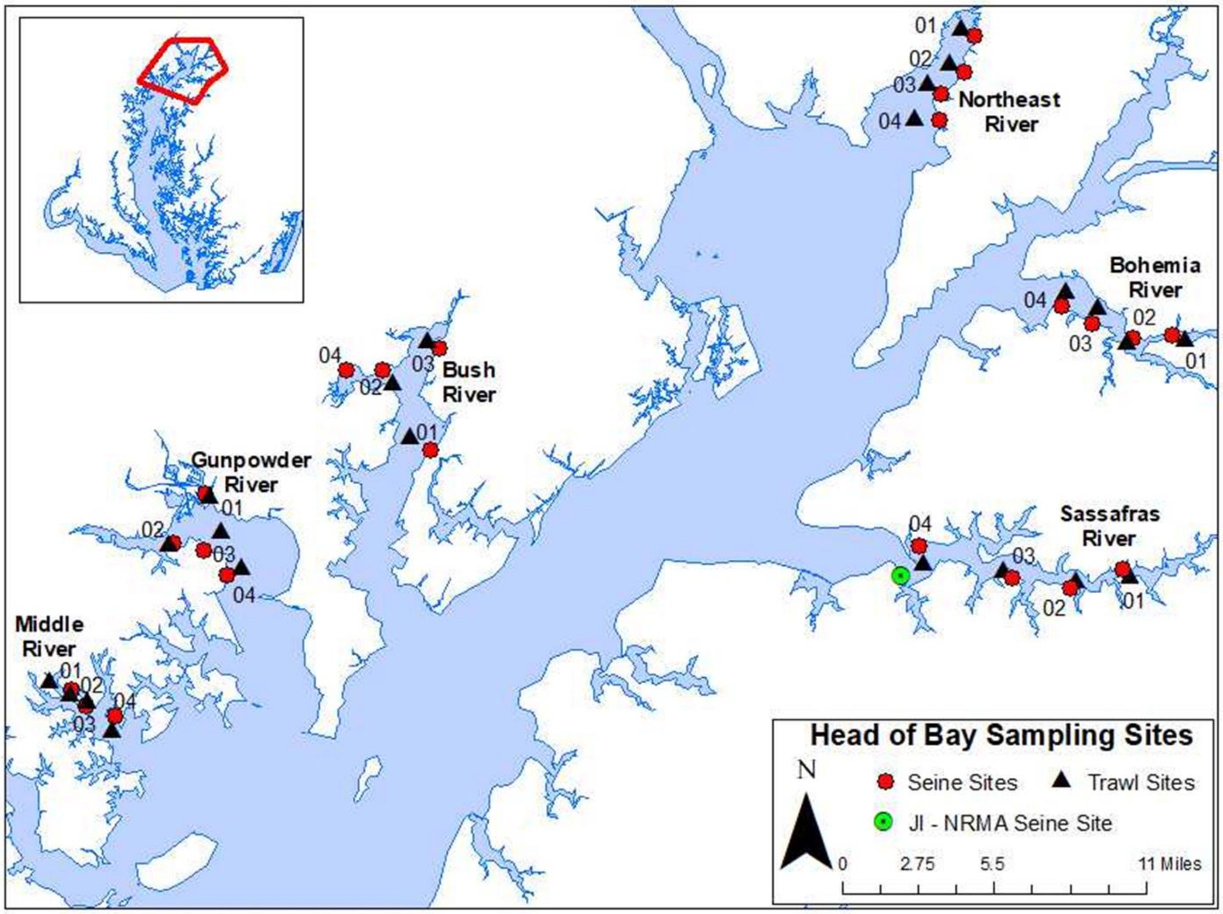


Figure 3-5. Mean subestuary bottom dissolved oxygen during summer sampling, 2003 – 2020, plotted against level of development (C / ha or structures per hectare).

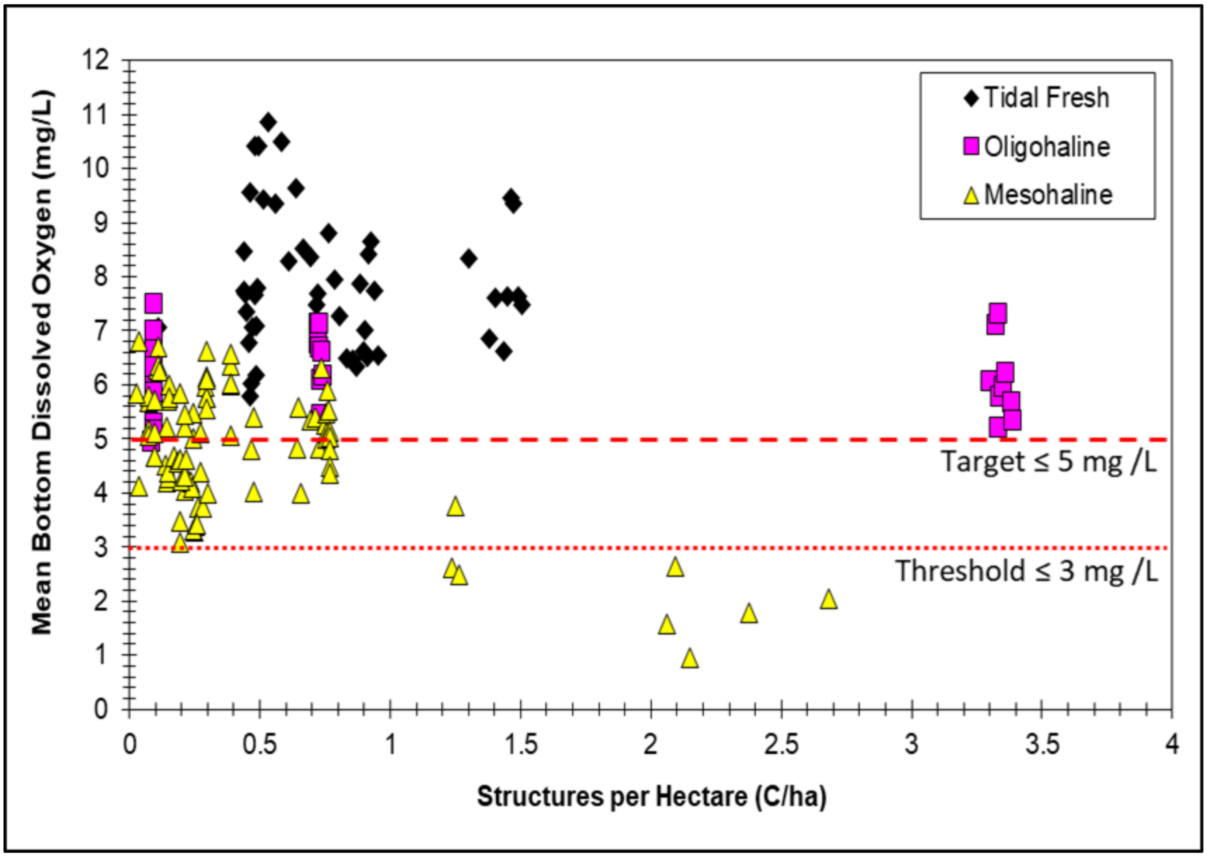


Figure 3-6. Mean subestuary surface dissolved oxygen during summer (July – October) sampling, 2003 – 2020, plotted against level of development (C / ha or structures per hectare).

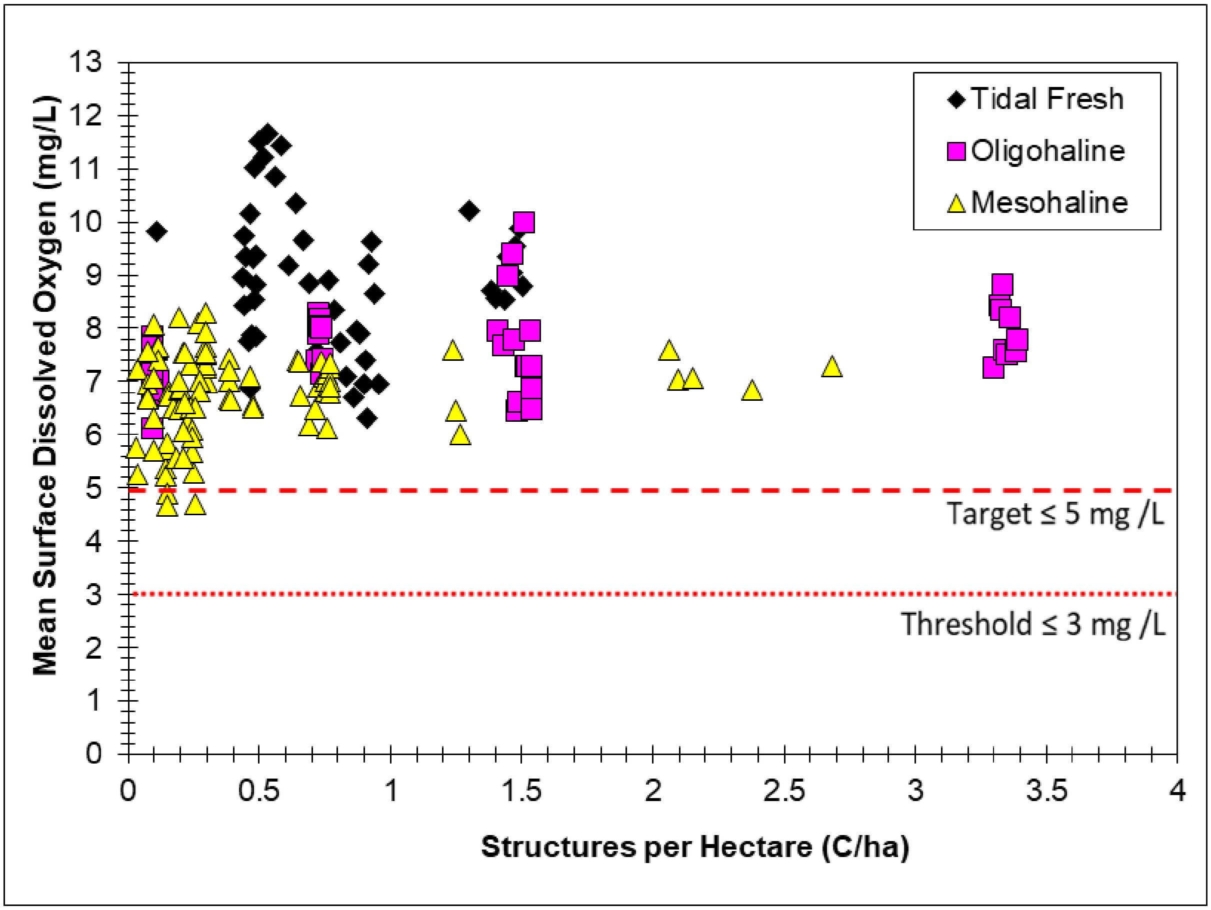


Figure 3-7. Maryland Department of Planning (DOP) estimates agricultural land coverage (% watershed land area) by region (western or eastern shore) versus median bottom dissolved oxygen (DO) in mesohaline subestuaries (2003 – 2020). Quadratic model predicts median bottom DO and agricultural coverage (%) using data from both regions.

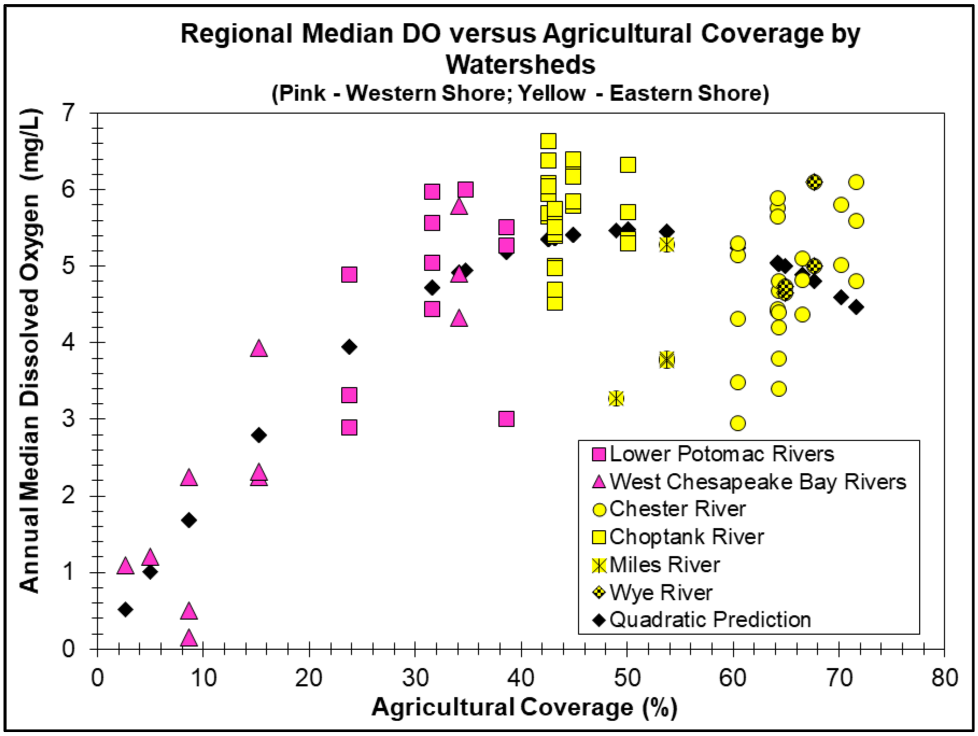


Figure 3-8. Trends in development (structures per hectare = C / ha) from 1950 to 2018 of watersheds of three subestuaries surveyed in the Choptank River, Broad Creek, Harris Creek, and Tred Avon River. Black diamond markers indicate the years that subestuaries were sampled. Development data was not available for 2019 and 2020 and 2018 was used for these years.

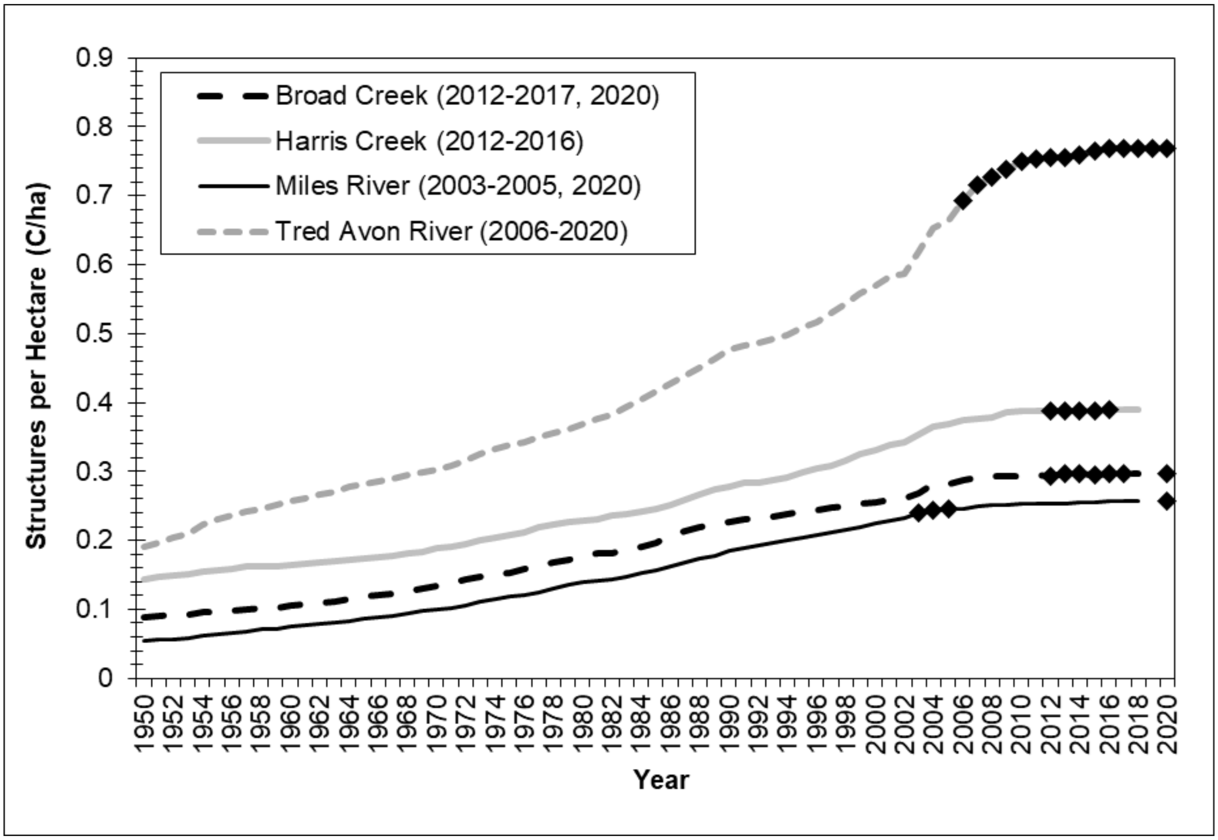


Figure 3-9. Bottom dissolved oxygen (DO; mg / L) readings (2003 – 2020) in Choptank River subestuaries, Broad Creek, Harris Creek, and Tred Avon River, and Miles River versus intensity of development (C / ha = structures per hectare) in Talbot County. Target (5 mg / L) and threshold (3 mg / L) boundaries are indicated by red dashed lines. See legend for years subestuaries were sampled.

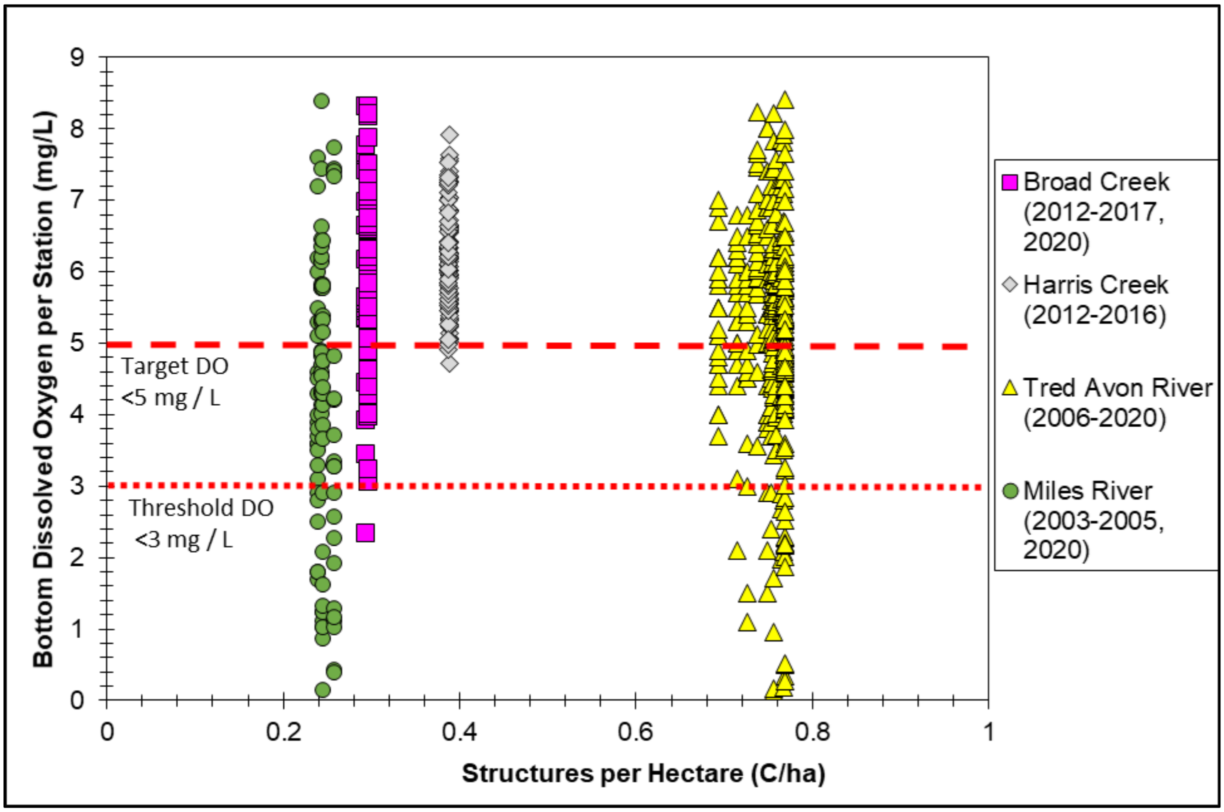


Figure 3-10. Median bottom dissolved oxygen (DO; red squares; mg / L) by year sampled for Broad Creek, Harris Creek, Miles River, and Tred Avon River. Solid black bars indicate range of all bottom DO measurements for that year. The y-axes range from 0 to 9 mg / L; x-axes vary based on years sampled.

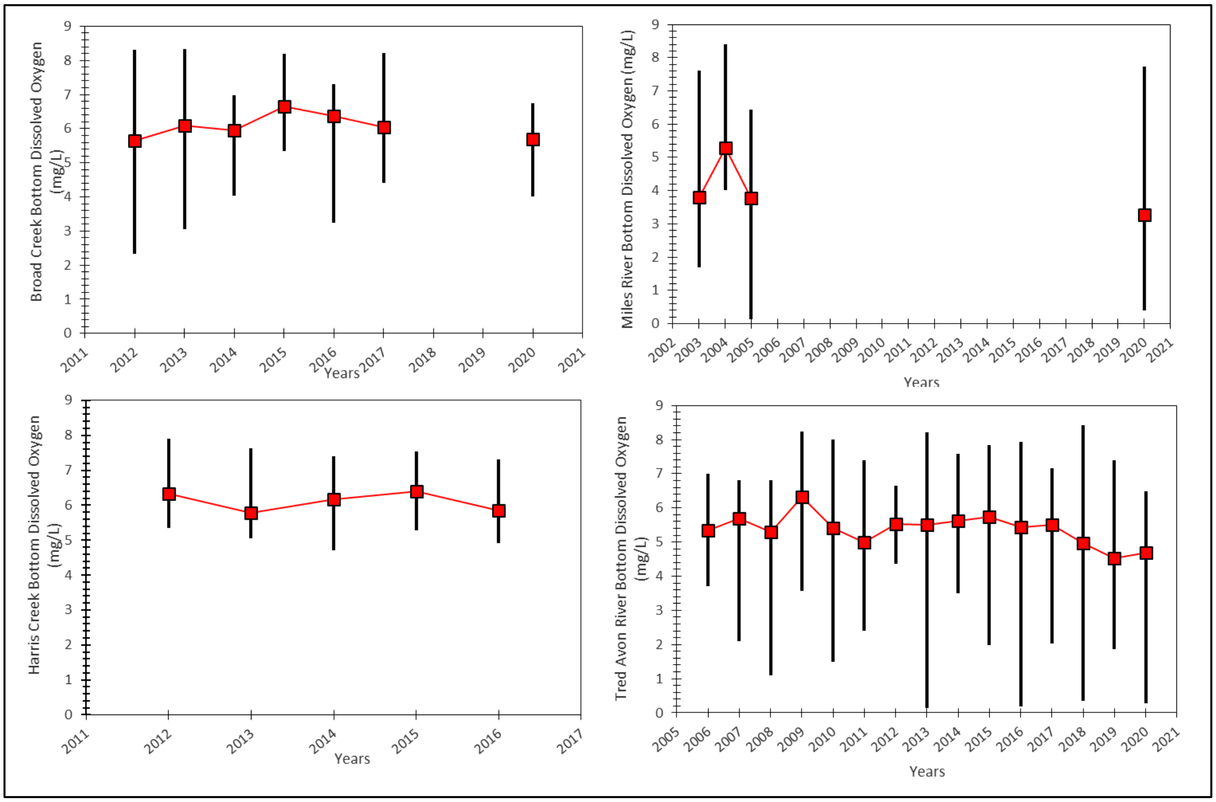


Figure 3-11. Mean bottom dissolved oxygen (DO; mg / L) for all years surveyed for Broad Creek, Harris Creek, Miles River, and Tred Avon River, by sampling station. Dotted line indicates the median of all DO measurement data for the time-series. The y-axes range from 0 to 8 mg / L; x-axes vary based on years sampled.

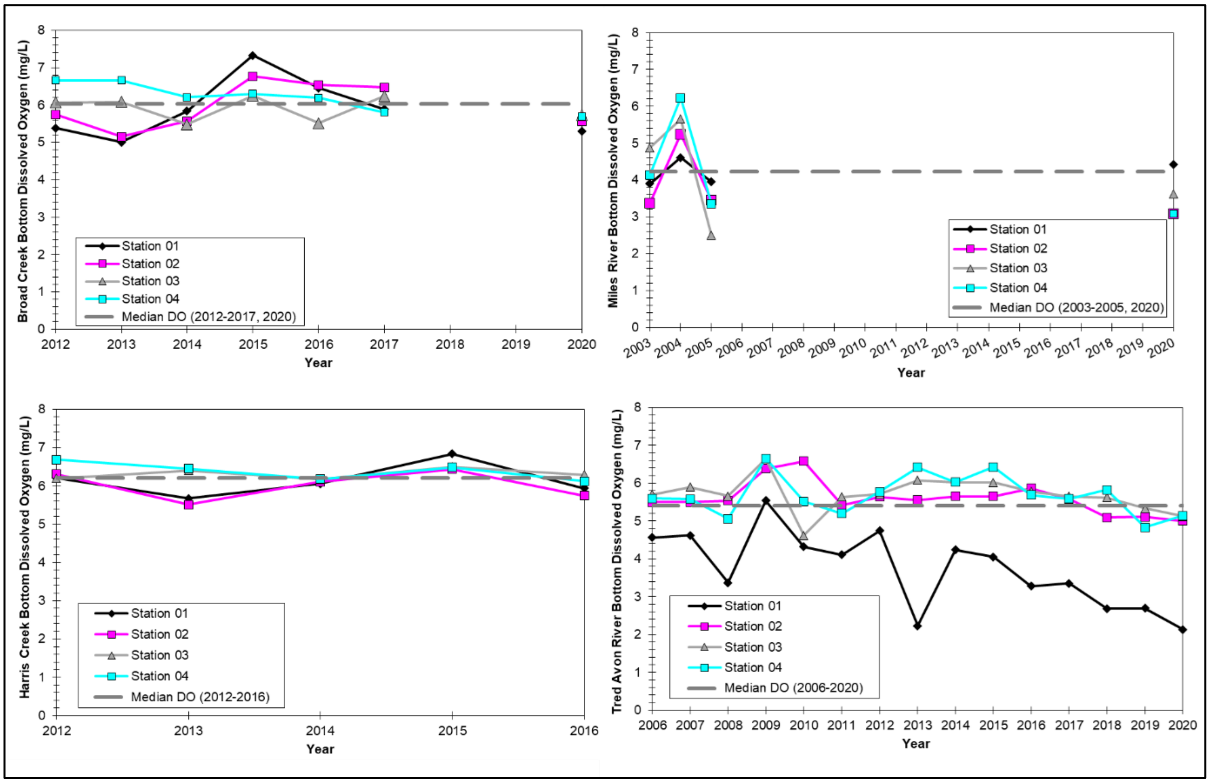


Figure 3-12. Median Secchi depth (m) for Broad Creek, Harris Creek, Miles River, and Tred Avon River (red squares), by year. Solid black bars indicate the range of Secchi depth (m) measurements by year. The y-axes range from 0 to 2.5 m; x-axes vary based on years sampled.

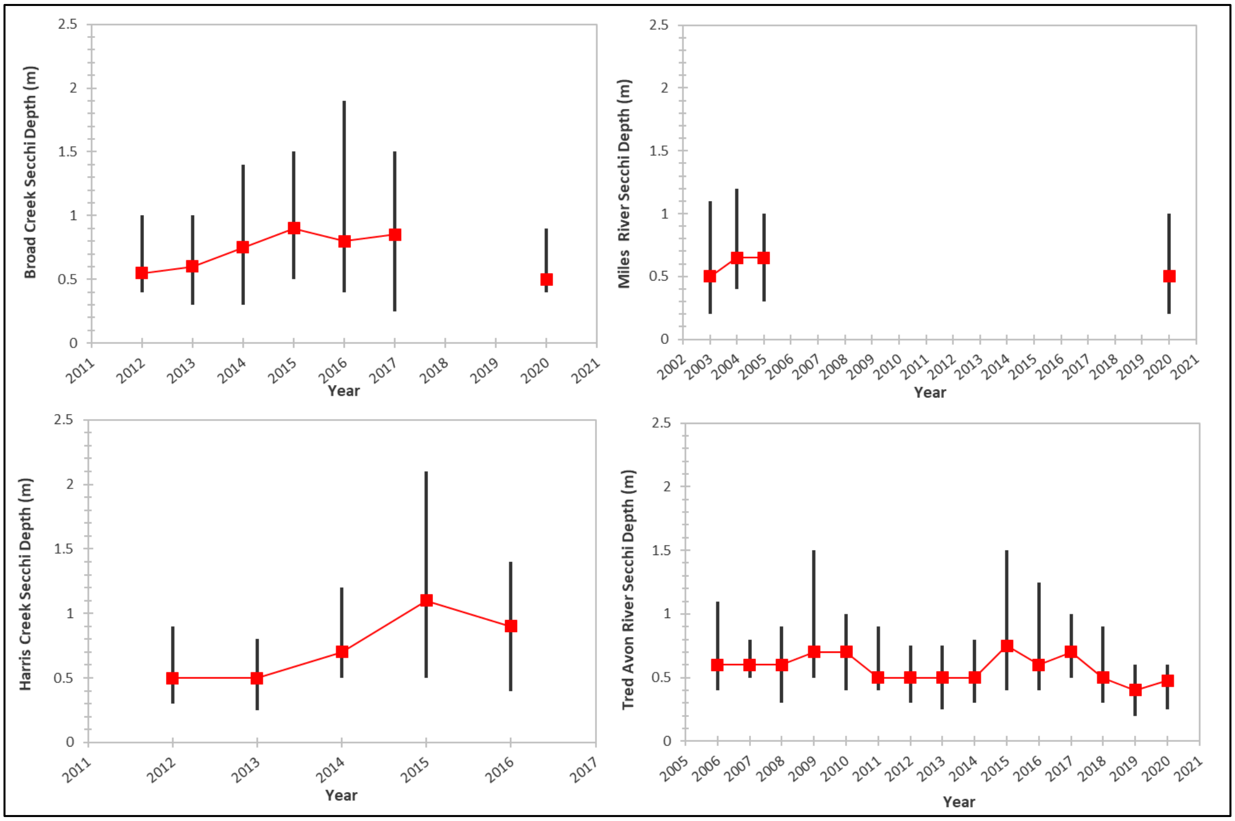


Figure 3-13. Coverage of SAV (percent of coverage in water area) for the mouth of the Choptank River (containing Broad Creek, Harris Creek, and Tred Avon) during 1989 – 2019. Median of only fully mapped years (1989 – 2017, 2019) for the time-series is indicated by the dashed line. Data for 2020 was not available at the time of this report.

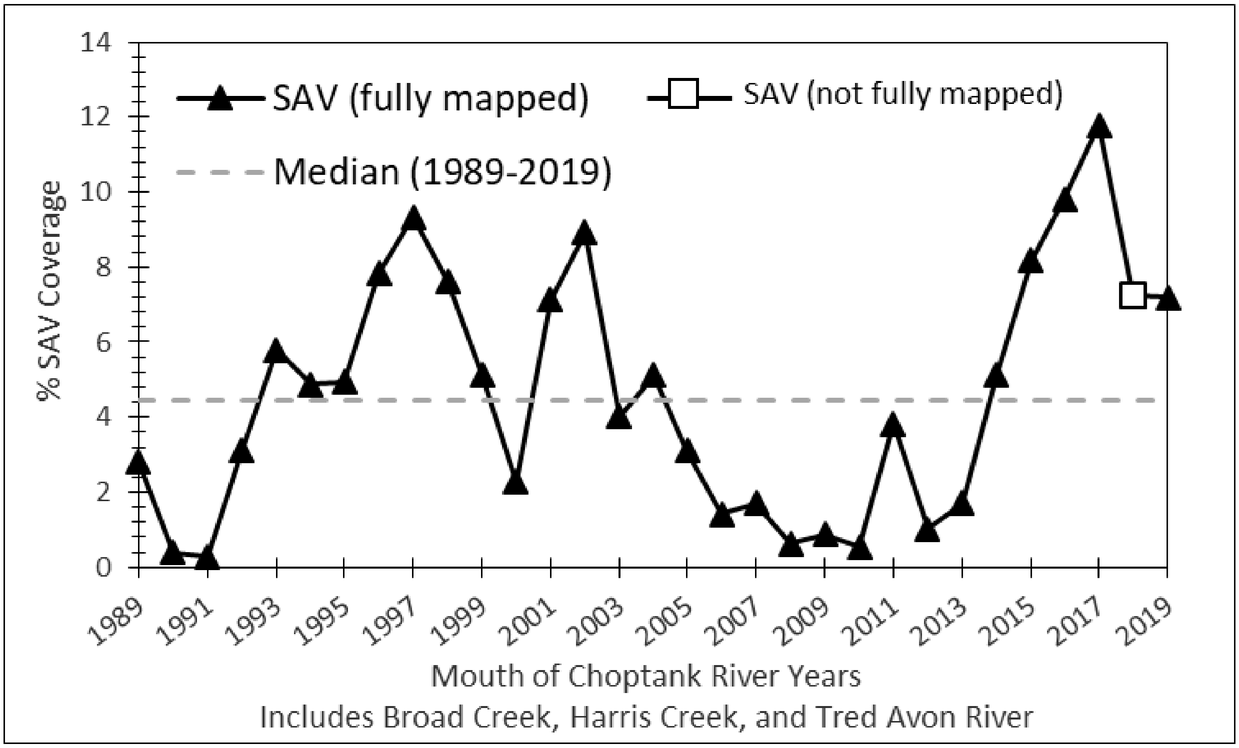


Figure 3-14. Median bottom pH (red squares) for Broad Creek, Harris Creek, Miles River, and Tred Avon River, by sampling year. Solid black bars indicate the range of pH measurements by year. The y-axes range from 5.5 to 9.5; x-axes vary based on years sampled.

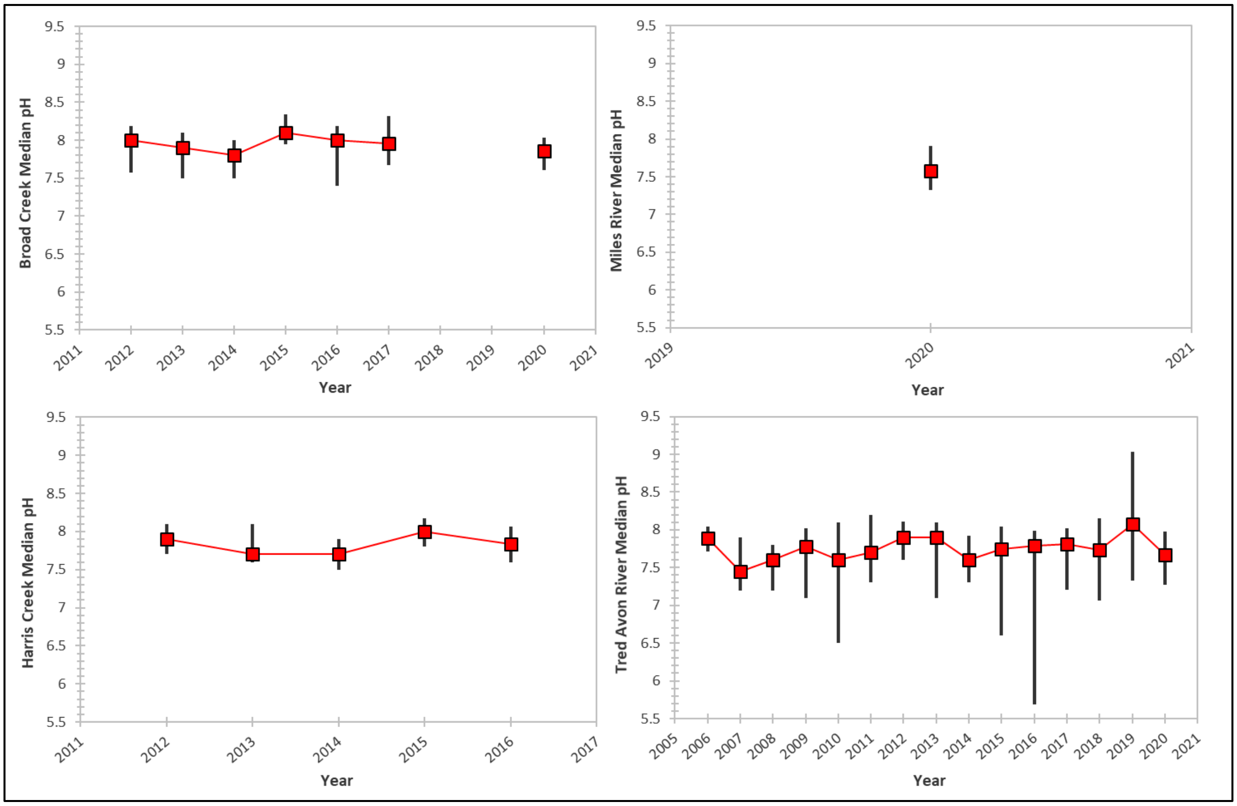


Figure 3-15. Median bottom salinity (red squares; ppt = ‰) for Broad Creek, Harris Creek, Miles River, and Tred Avon River, by sampling year. Solid black bars indicate the range of salinity measurements by year. The y-axes range from 0 to 18 ppt; x-axes vary based on years sampled.

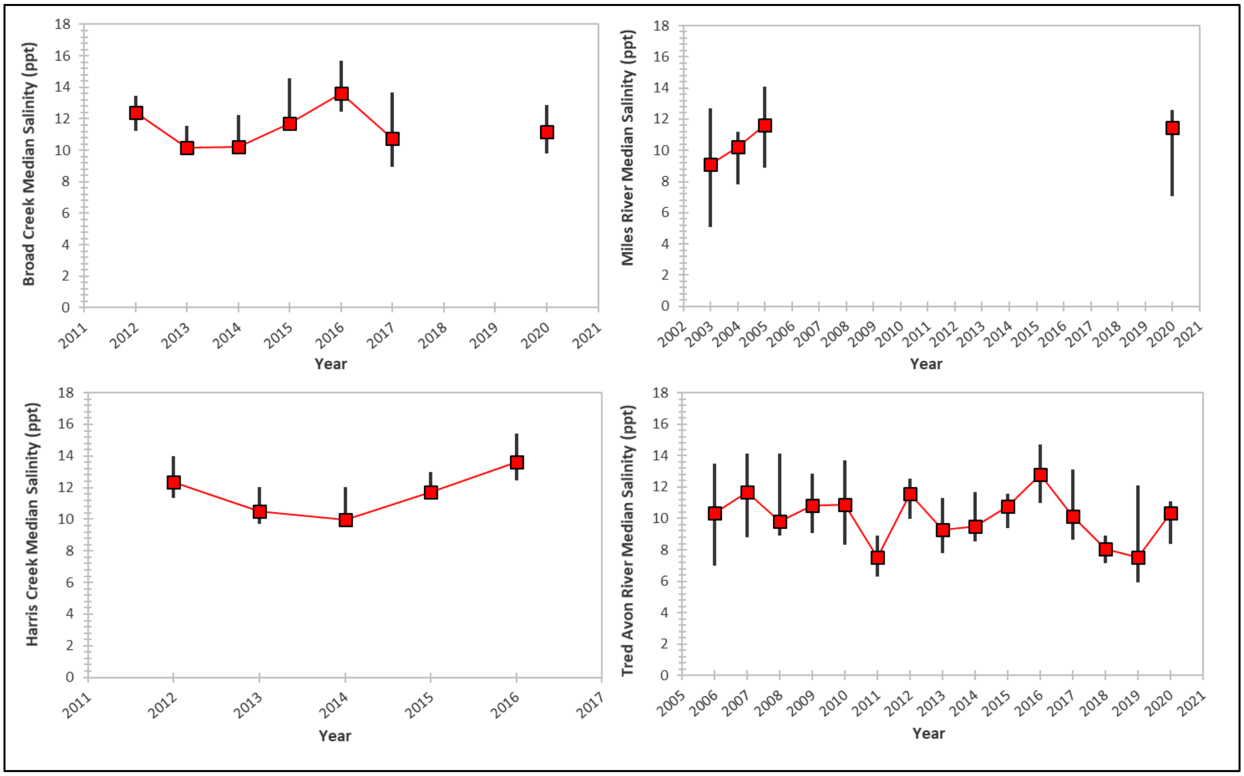


Figure 3-16. Linear trends for Choptank River tributaries and Miles River bottom DO (mg / L) versus bottom salinity (‰) by year. Dashed lines indicate DO target and threshold values. The y-axes range from 0 to 9 mg / L; x-axes range from 0 to 16 ‰.

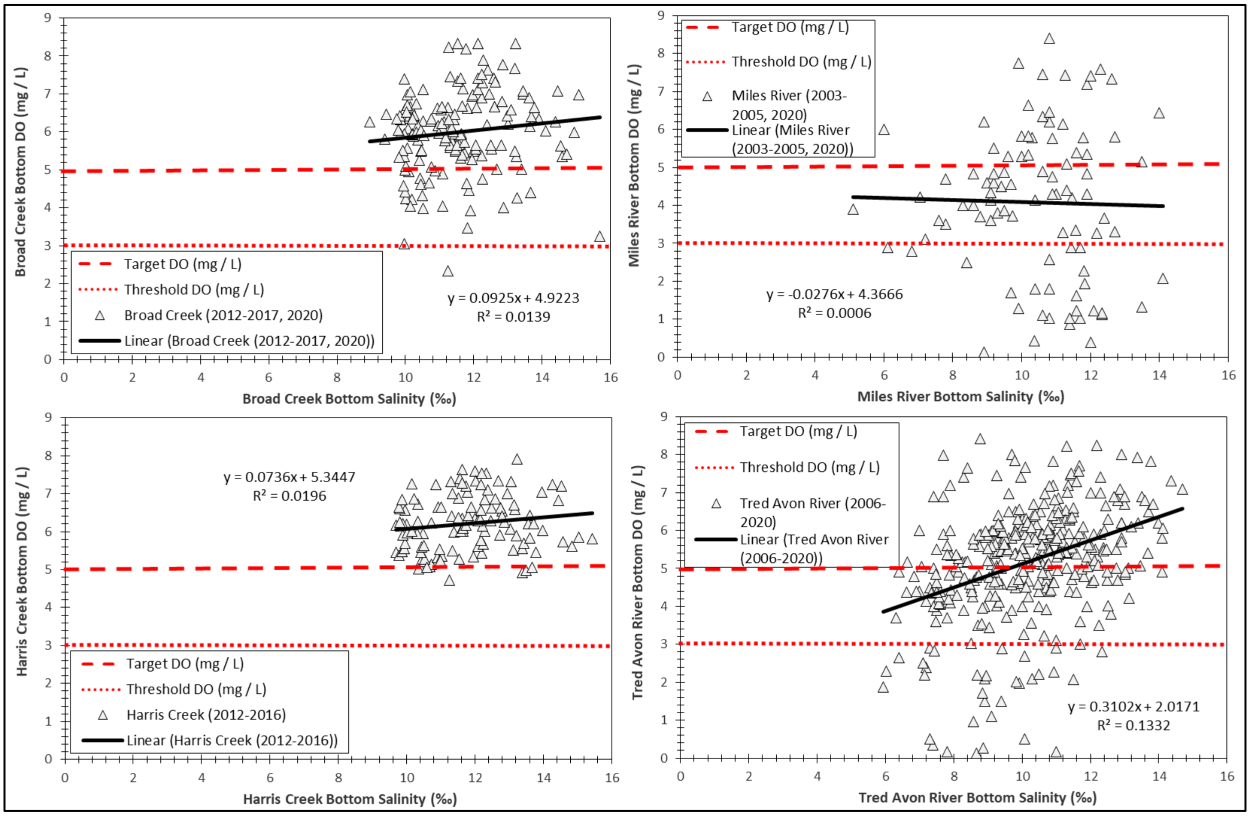


Figure 3-17. Trends in levels of development (structures per hectare = C / ha) during 1950 – 2018 in the Head-of-Bay subestuaries, Bohemia River, Bush River, Gunpowder River, Middle River, Northeast River, and Sassafras River. Black diamond markers indicate the years that subestuaries were sampled. Tax map data were not available for 2019 and 2020.

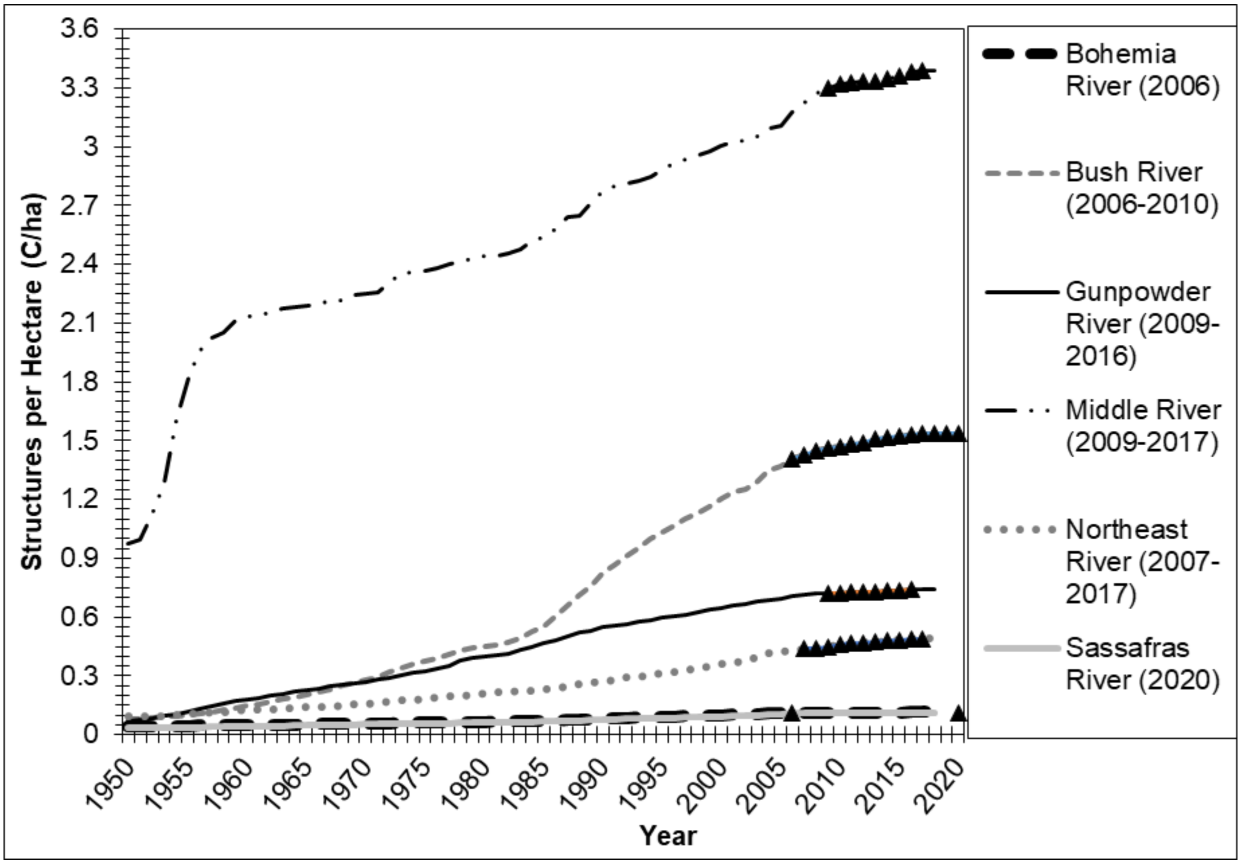


Figure 3-18. Summer (July – September) median bottom dissolved oxygen (DO; red squares; mg/L) for Chesapeake Bay Program Sassafras River (CBP ET3.1) and Head-of-Bay (CBP CB1.1) monitoring stations from 1989 to 2020. Solid black bars indicate range of bottom DO measurements for each year.

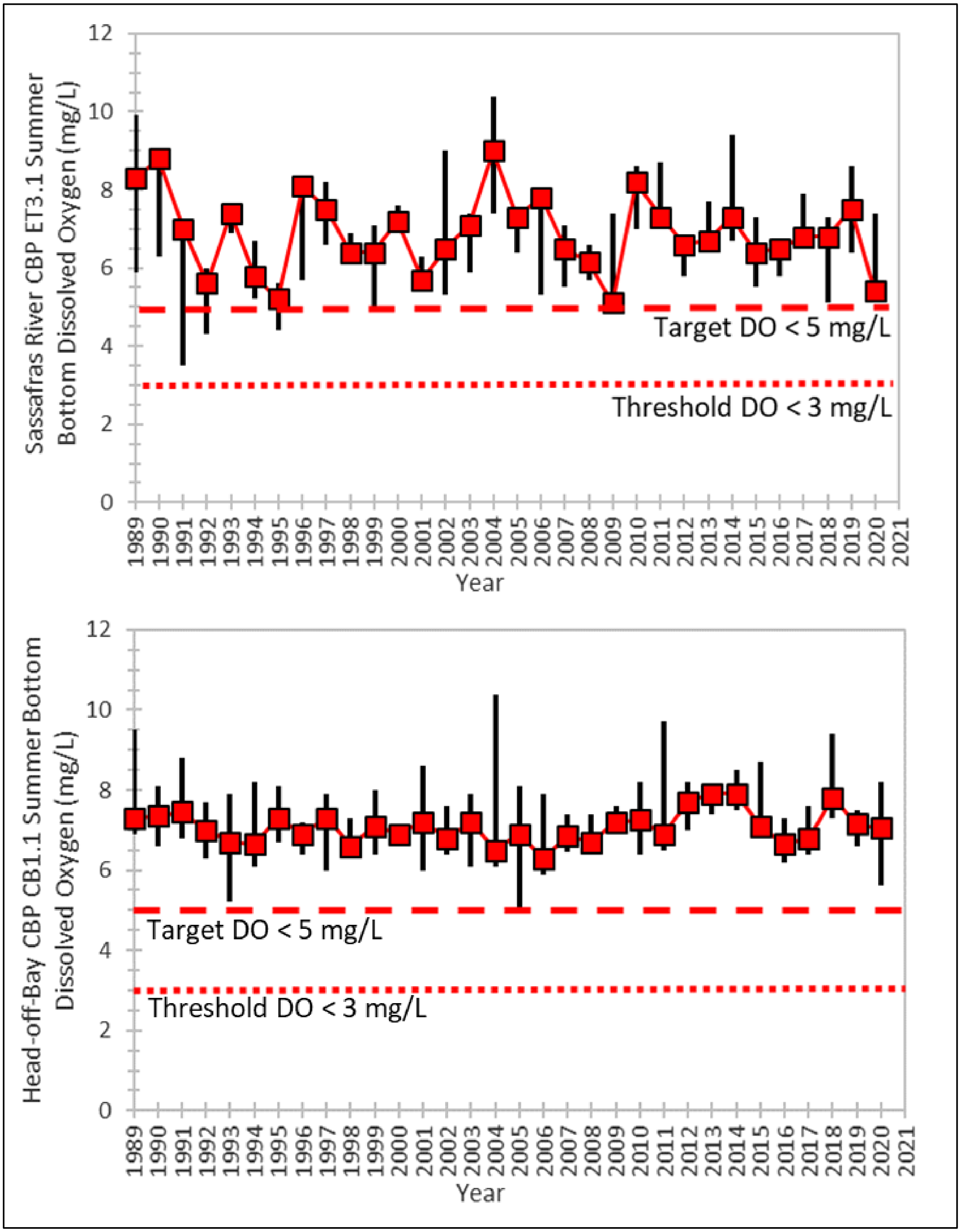


Figure 3-19. Bottom dissolved oxygen (DO; mg / L) versus intensity of development (C / ha = structures per hectare) in the Head-of-Bay subestuaries. Target (= 5 mg / L) and threshold (= 3 mg / L) boundaries are indicated (red dashed lines).

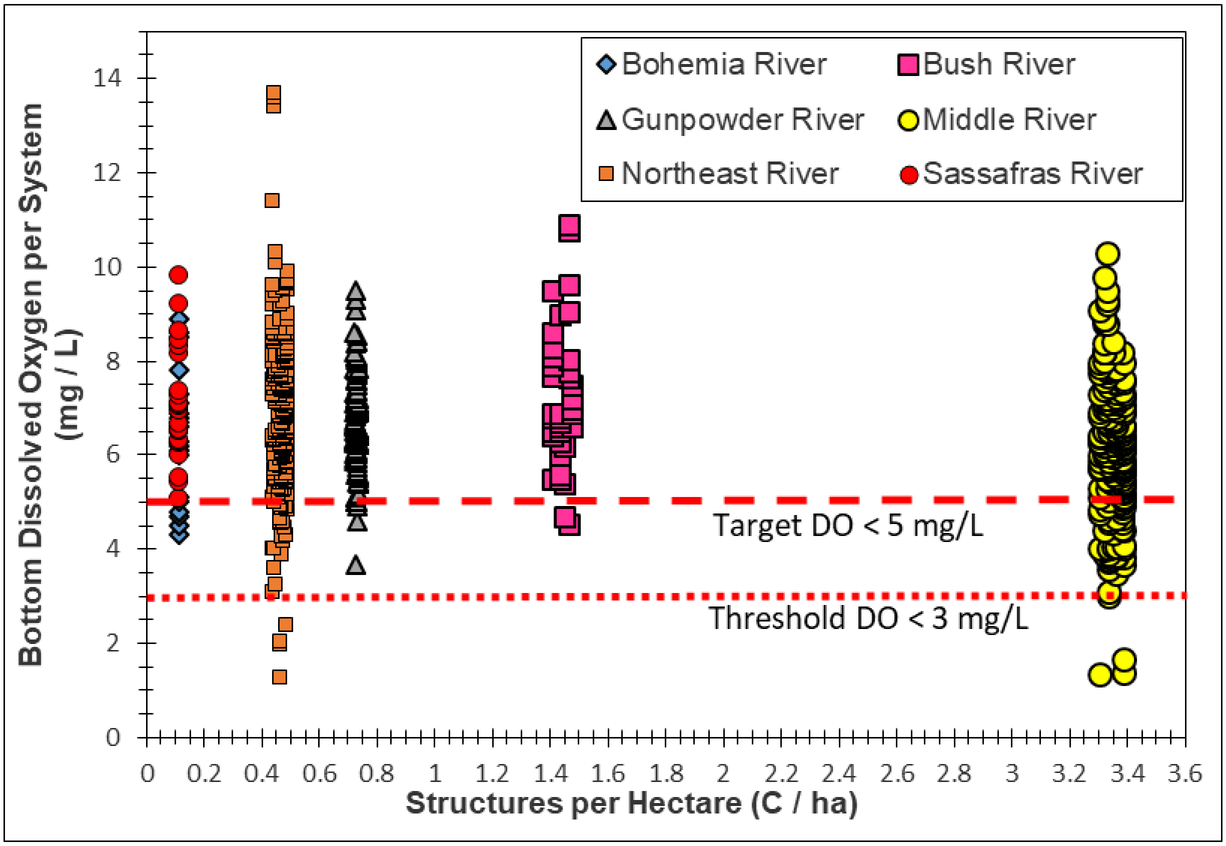


Figure 3-20. Median bottom dissolved oxygen (DO; red squares; mg / L) for Head-of-Bay subestuaries for each year sampled. Solid black bars indicate range of bottom DO measurements for that year. The y-axes range from 0 to 16 mg / L; x-axes vary based on years systems were sampled.

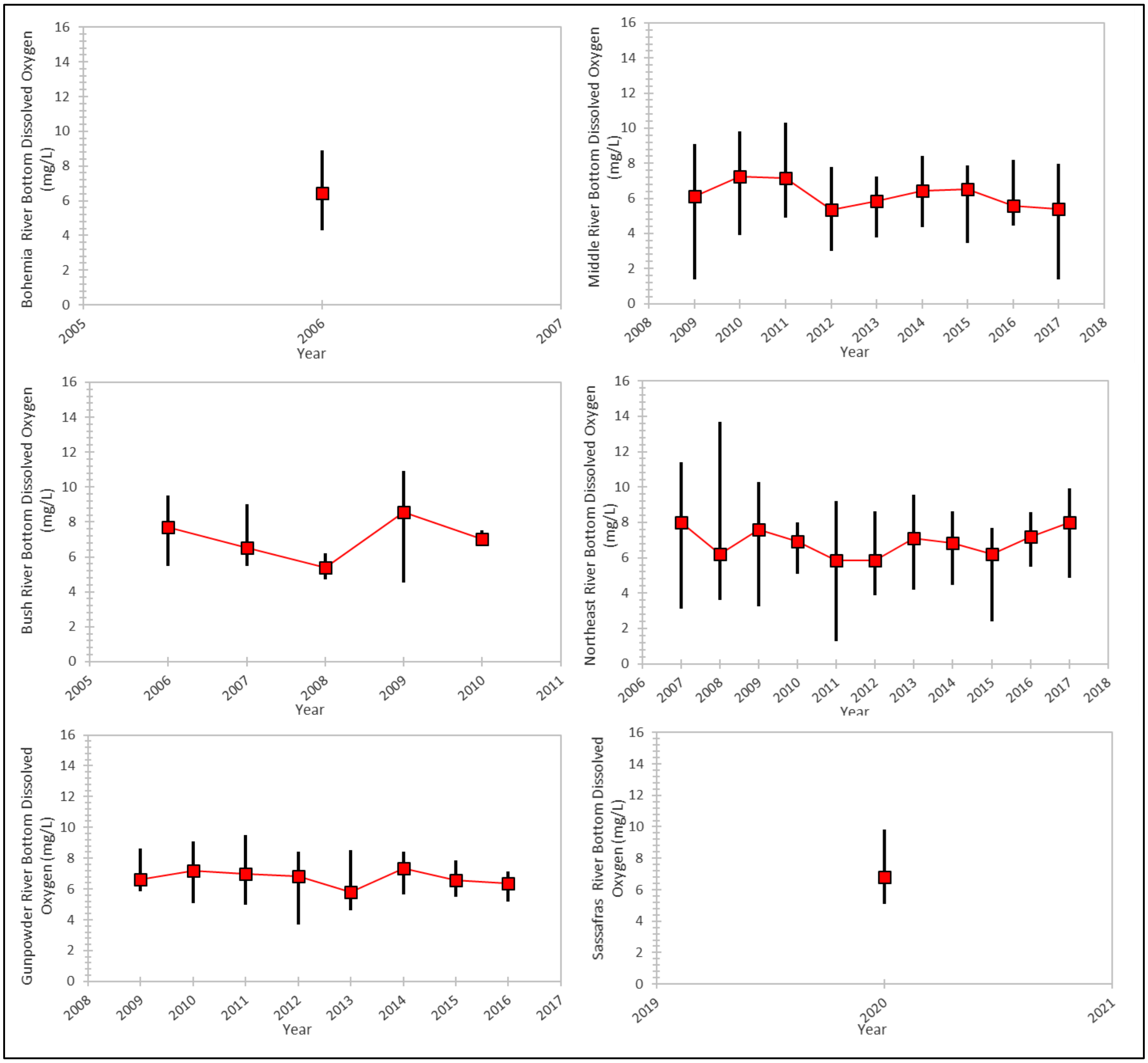


Figure 3-21. Mean bottom dissolved oxygen (DO; mg / L) for all years surveyed for Head-of-Bay subestuaries, by sampling station. Dotted line indicates the median of all DO measurement data for the time-series available. The y-axes start at 3 mg / L; x-axes vary based on years systems were sampled.

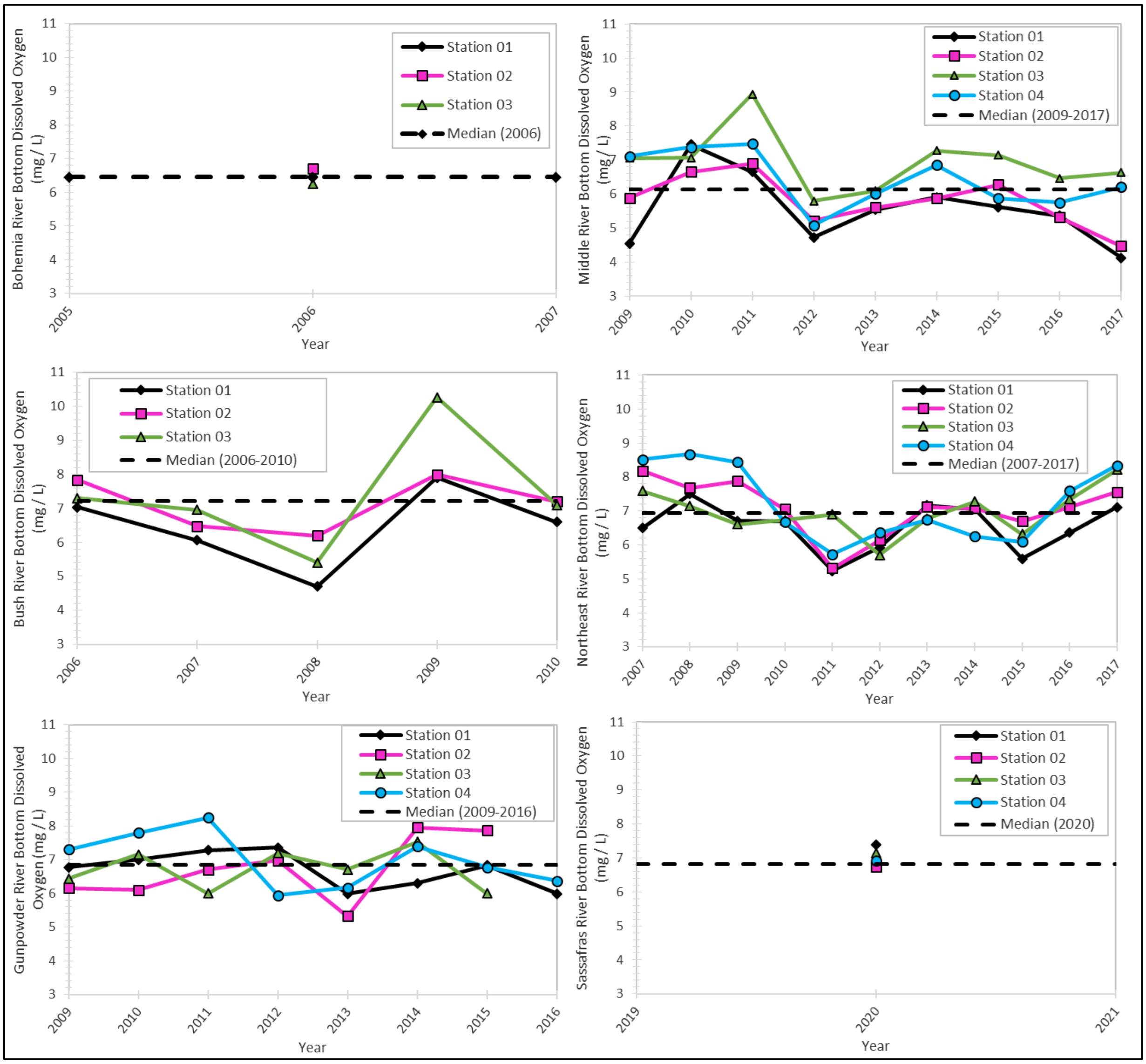


Figure 3-22. Median Secchi depth (m) for Bohemia River, Bush River, Gunpowder River, Middle River, Northeast River, and Sassafras River (red squares), by year. Solid black bars indicate the range of Secchi depth (m) measurements by year. The y-axes range from 0 to 2.5 m; x-axes vary based on years sampled.

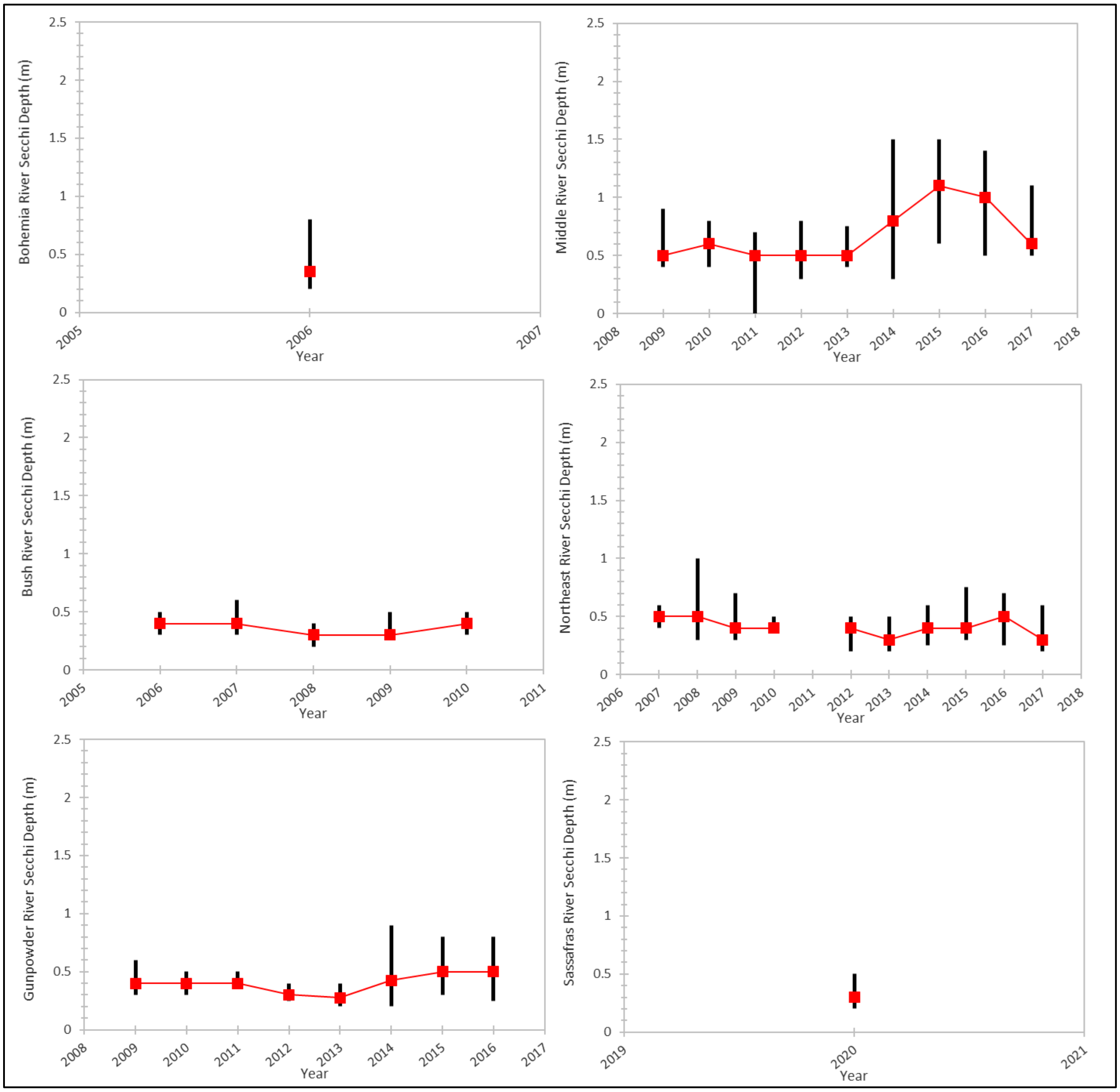


Figure 3-23. Median bottom pH (red squares) for Bohemia River, Bush River, Gunpowder River, Middle River, Middle River, Northeast River, and Sassafras River, by sampling year. Solid black bars indicate the range of pH measurements by year. The y-axes range from 6 to 10; x-axes vary based on years sampled.

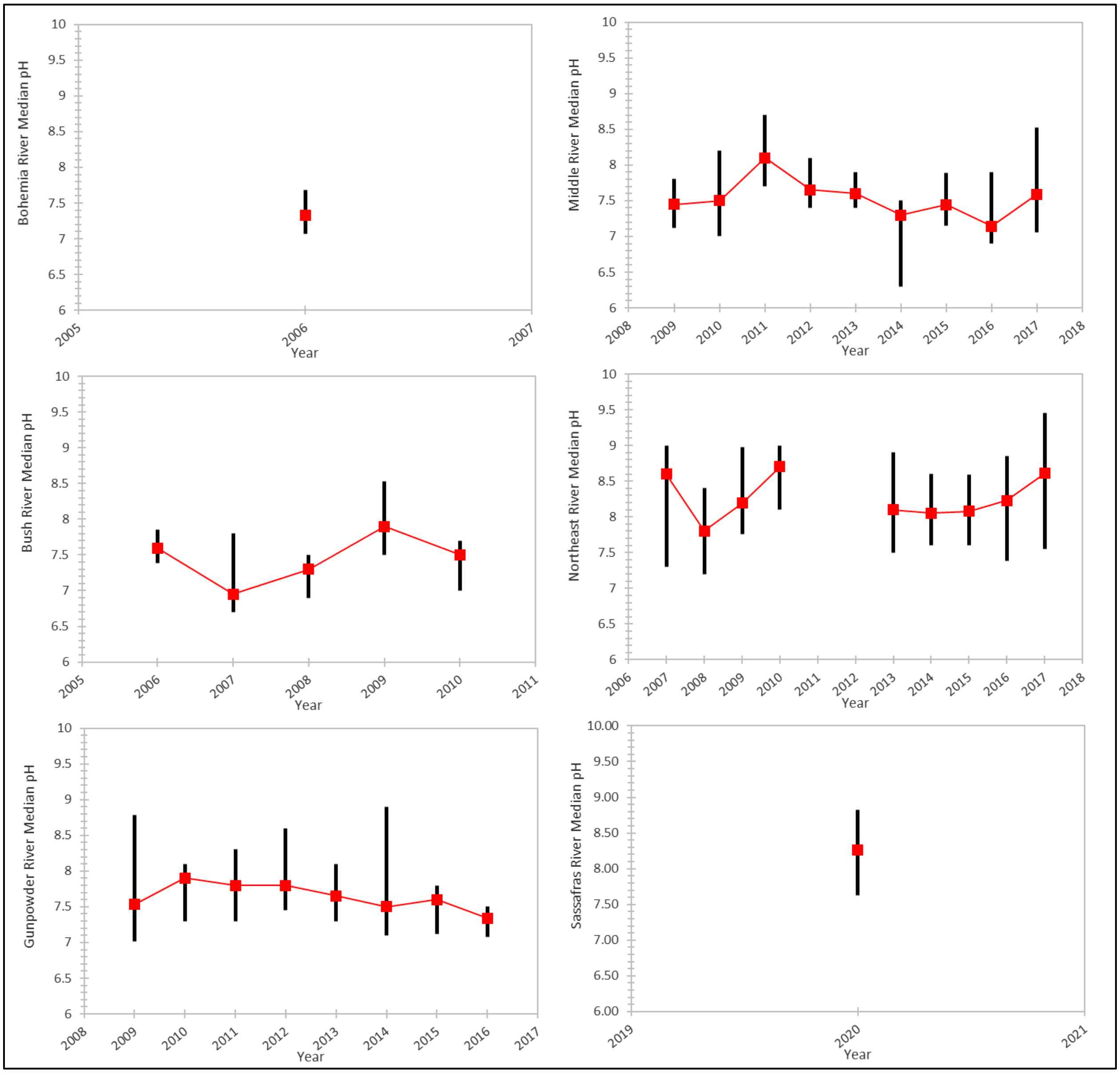


Figure 3-24. Median bottom salinity (red squares; ppt = ‰) for Bohemia River, Bush River, Gunpowder River, Middle River, Northeast River, and Sassafras River, by sampling year. Solid black bars indicate the range of salinity measurements by year. The y-axes range from 0 to 9 ppt; x-axes vary based on years sampled.

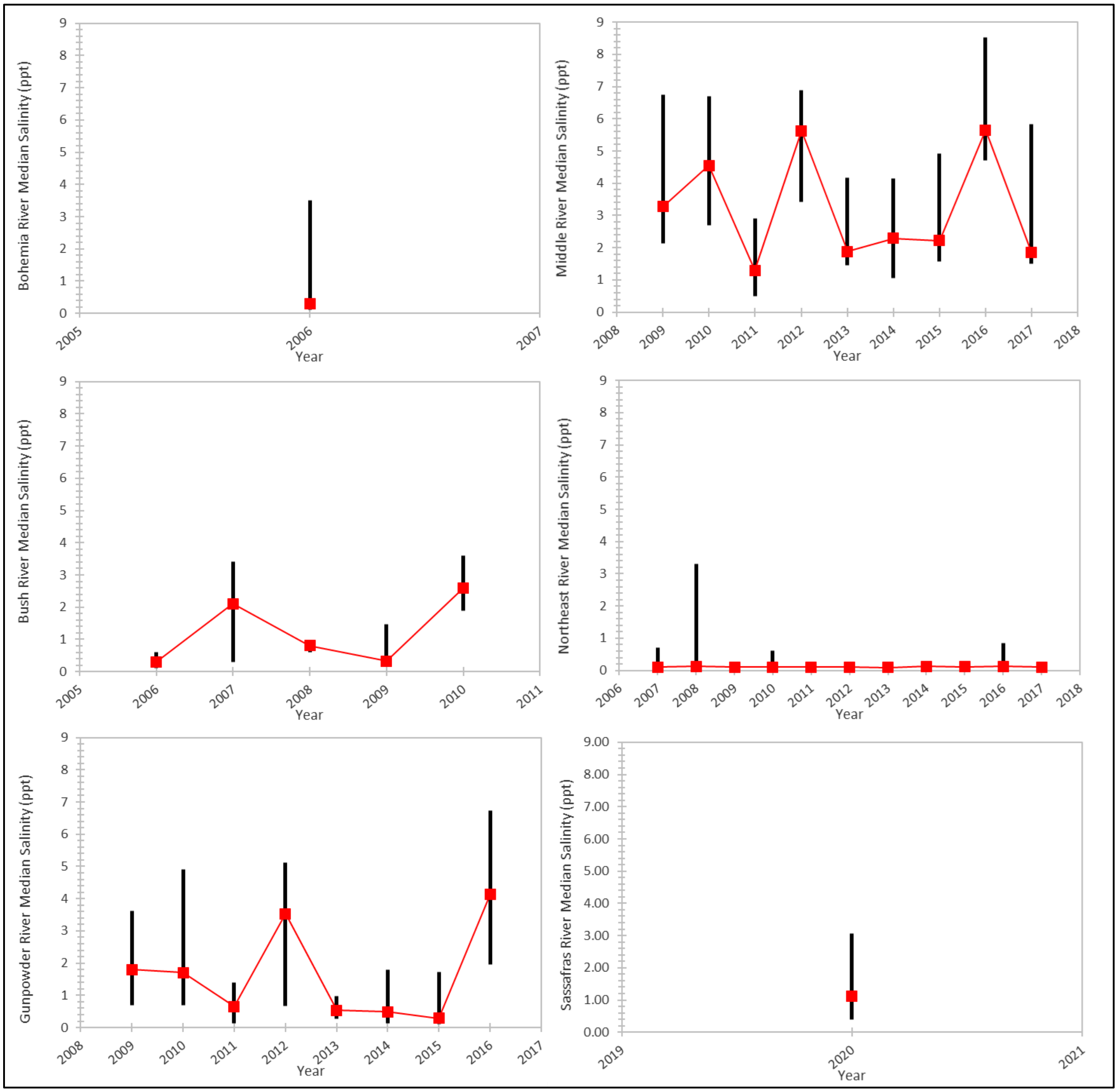


Figure 3-25. Number of finfish species (richness) collected by beach seines in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of watershed development (C / ha = structures per hectare). Points were omitted if beach seine effort (number of samples) < 15 samples.

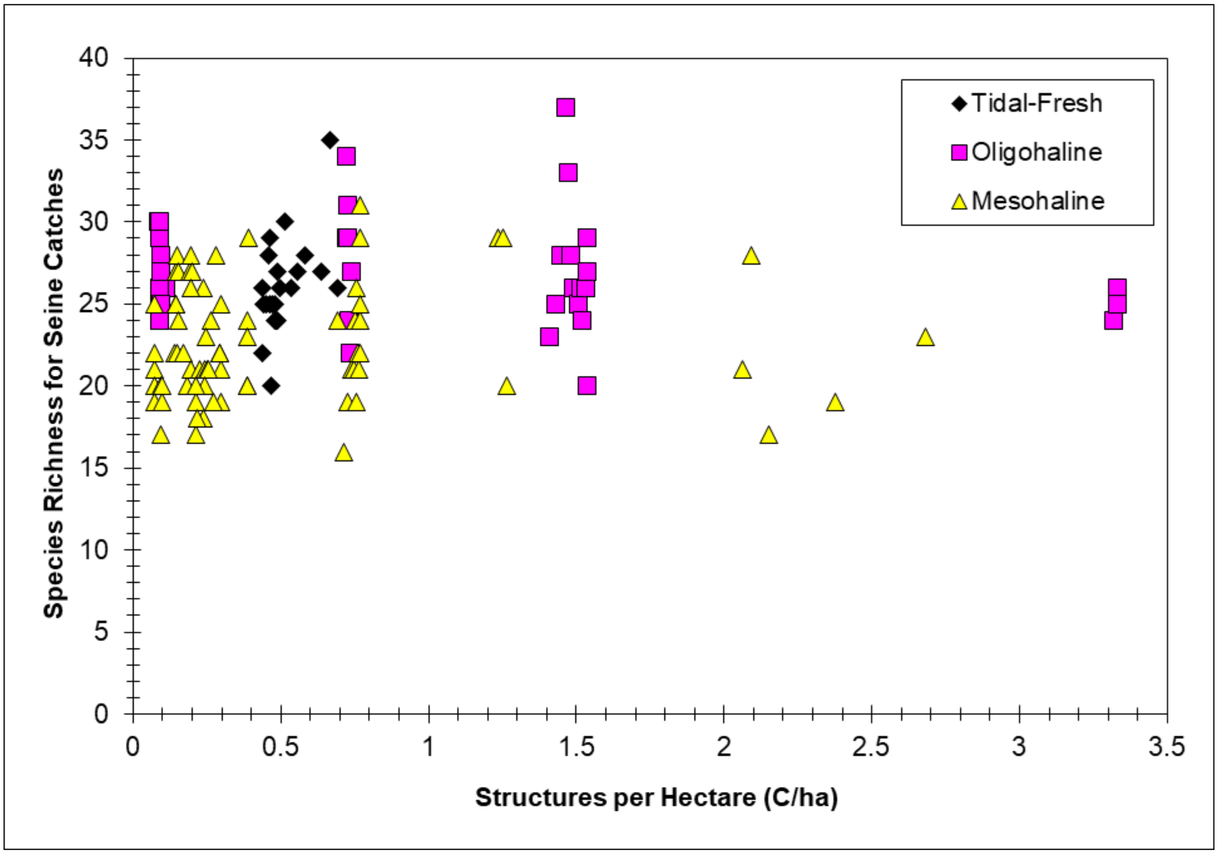


Figure 3-26. Annual 4.9 m trawl geometric mean (GM) catches plotted against C / ha subestuaries sampled during 2003 – 2020, separated by salinity class.

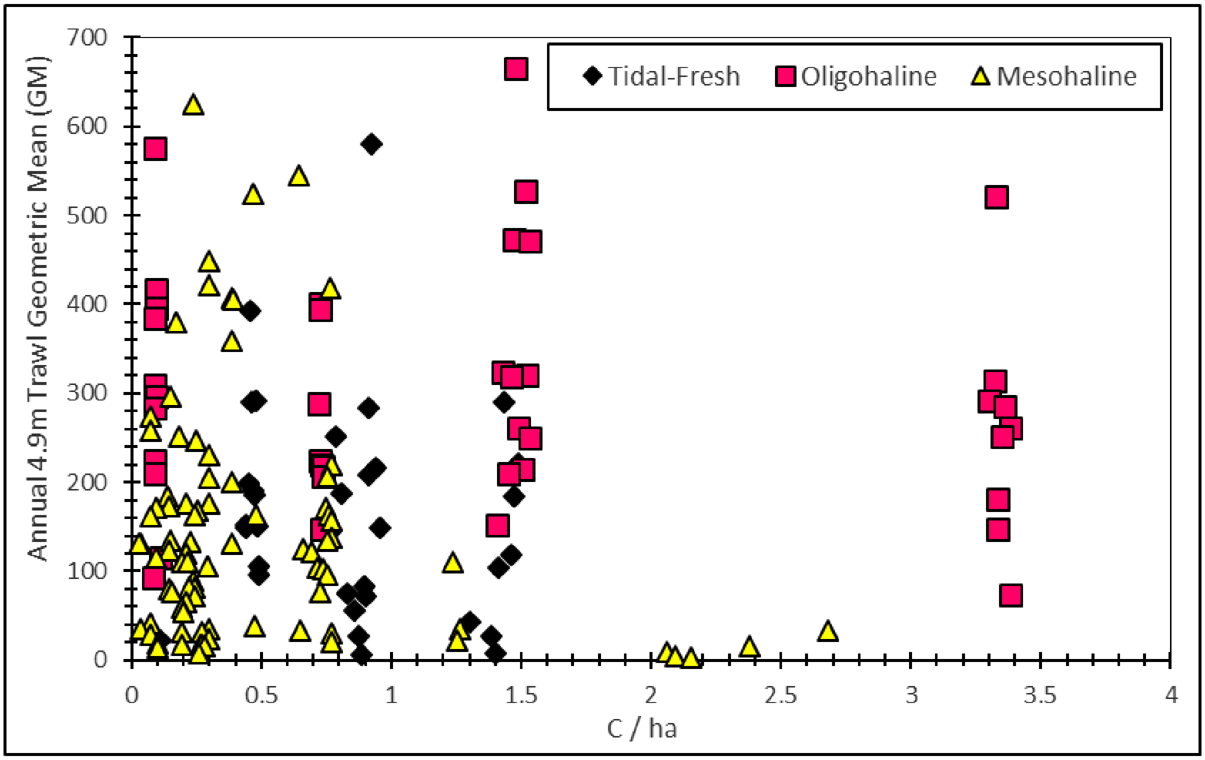


Figure 3-27. Number of finfish species (richness) collected by 4.9 m bottom trawl in tidal-fresh, oligohaline, and mesohaline subestuaries versus intensity of development (C / ha = structures per hectare). Points were omitted if number of samples was less than 15.

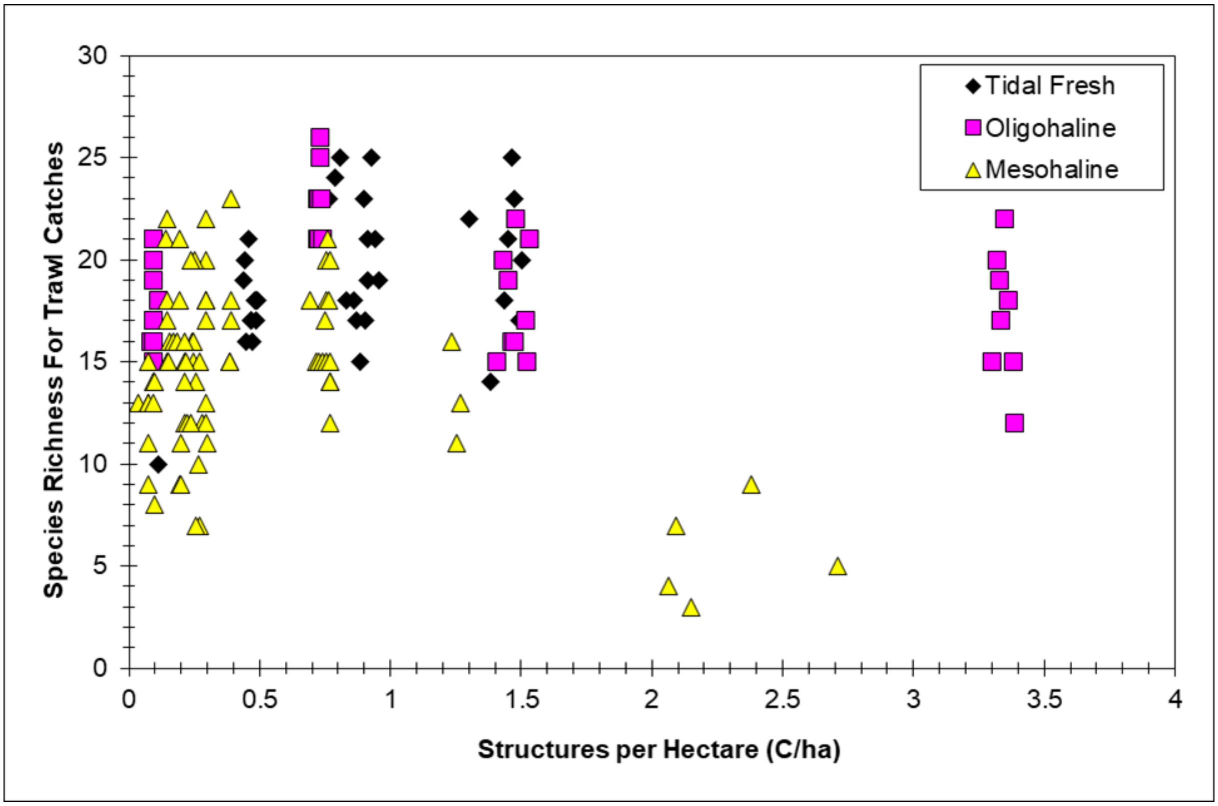


Figure 3-28. Annual 4.9 m bottom trawl catch geometric mean (GM) of all finfish species (red squares) for Broad Creek, Harris Creek, Miles River, and Tred Avon River, by sampling year. Black bars indicate the 95 % confidence intervals. The y-axes range from 0 to 600; x-axes vary based on years sampled.

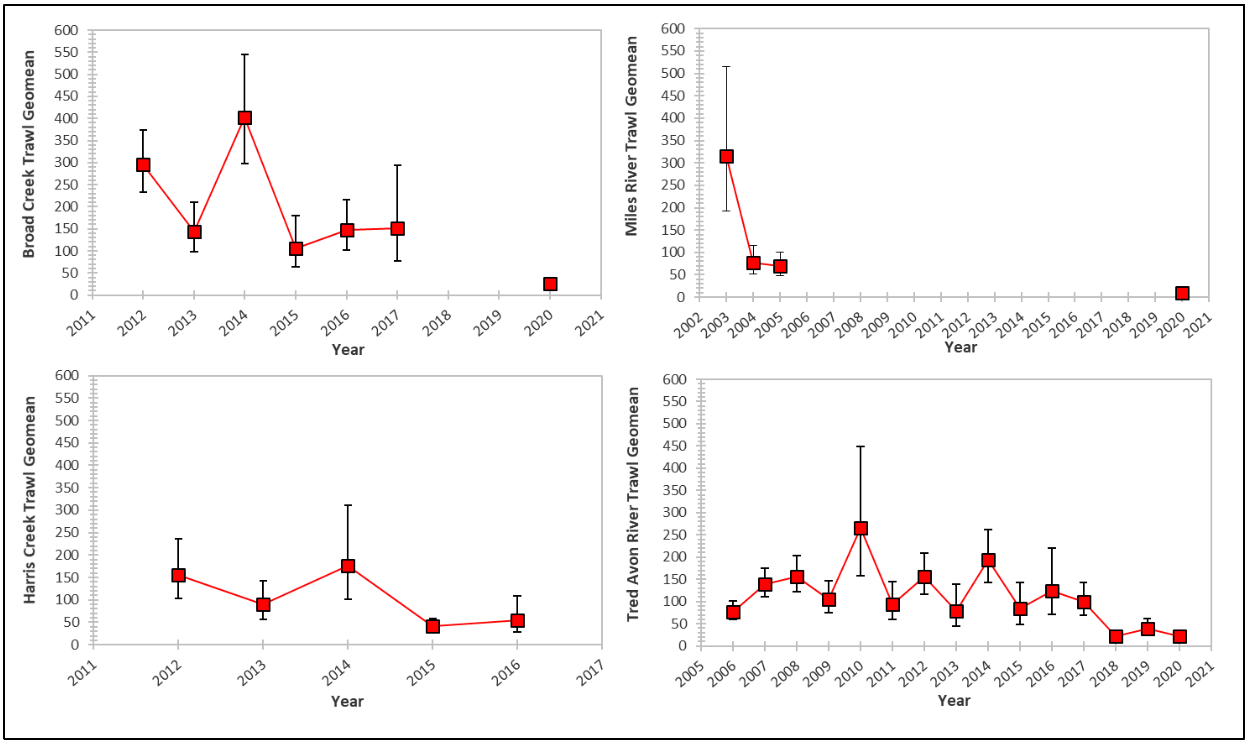


Figure 3-29. Finfish species composition for 4.9 m bottom trawl catch in Broad Creek, Harris Creek, Miles River, and Tred Avon River for all sampling years combined. Species that define the top 90 % are identified, and the remainder of species are grouped and labeled as “other species”.

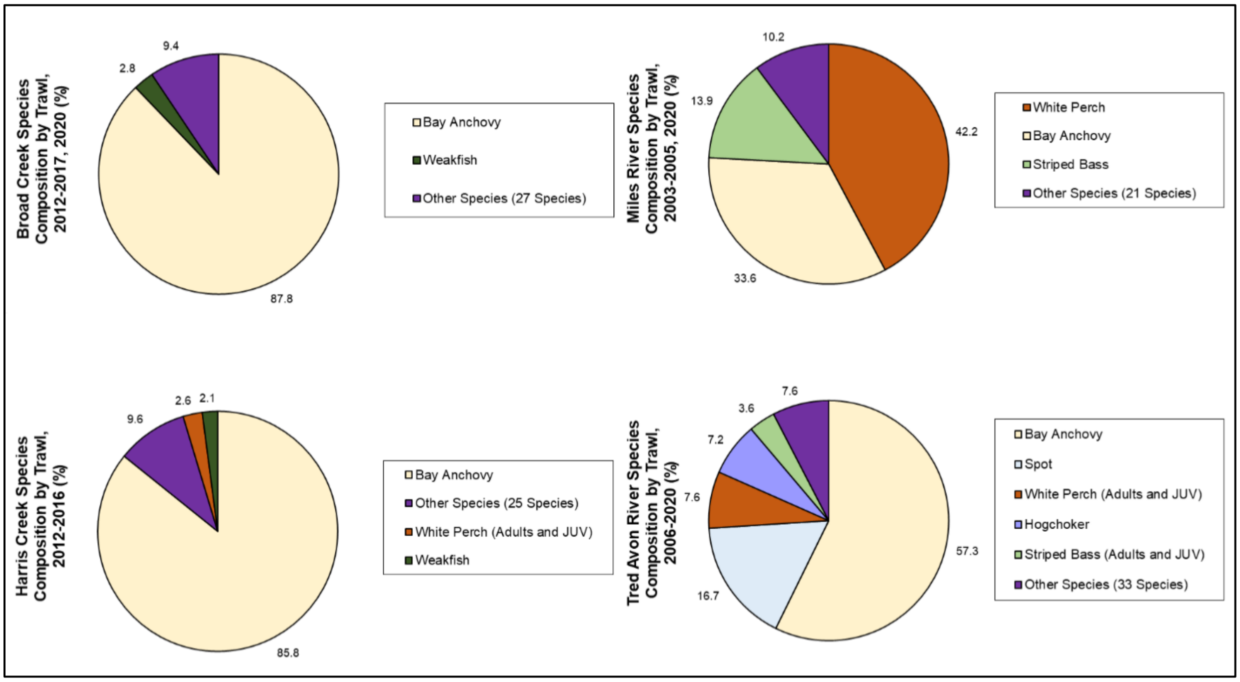


Figure 3-30. Finfish species composition for 4.9 m bottom trawl catch in Broad Creek, Harris Creek, Miles River, and Tred Avon River for each year sampled. Species that define the top 90 % are identified, and the remainder of species are grouped and labeled as “other species”. The y-axes range from 0 to 100 %; x-axes vary based on years sampled.

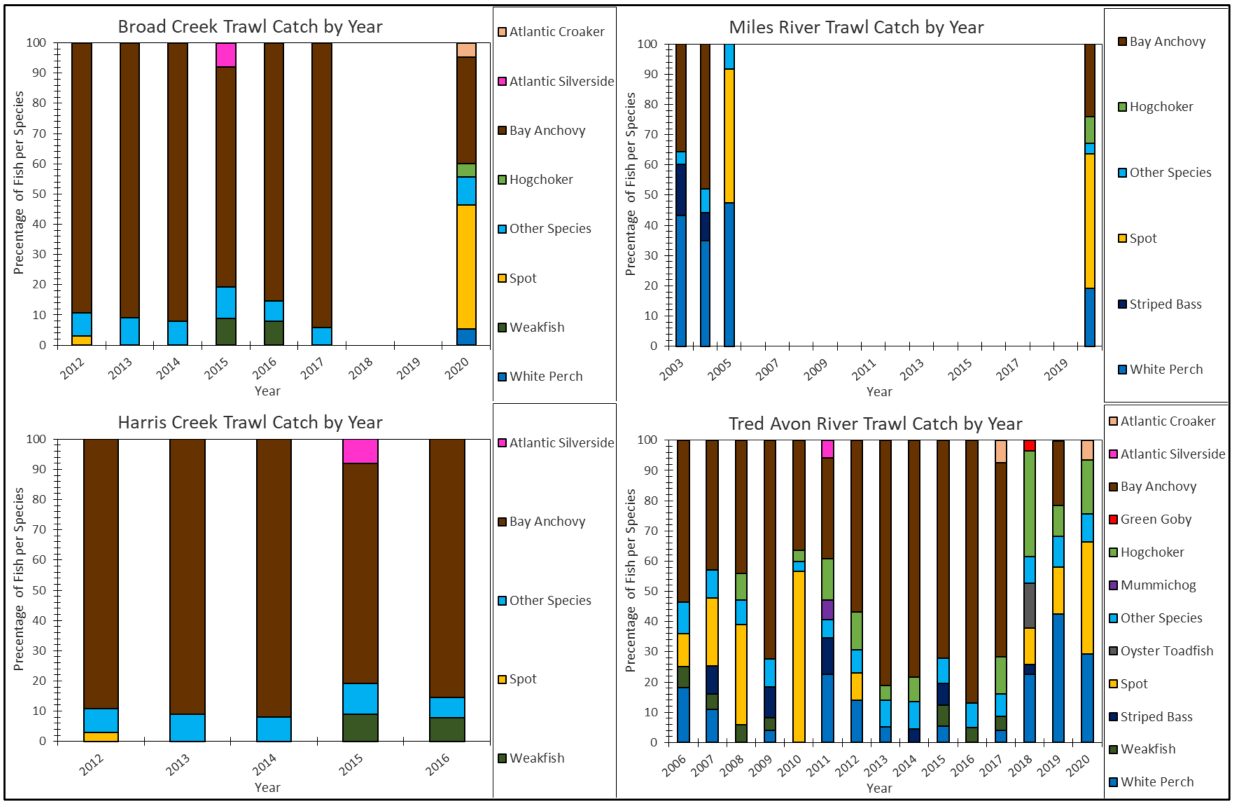


Figure 3-31. Percent similarity index (%) for 4.9 m bottom trawl stations 01 – 04 in Choptank River tributaries, Broad Creek, Harris Creek, and Tred Avon River, by year. The greater the similarity value, the more finfish species there are in common throughout all four bottom trawl stations. The y-axes range from 0 to 100 %; x-axes vary based on years sampled.

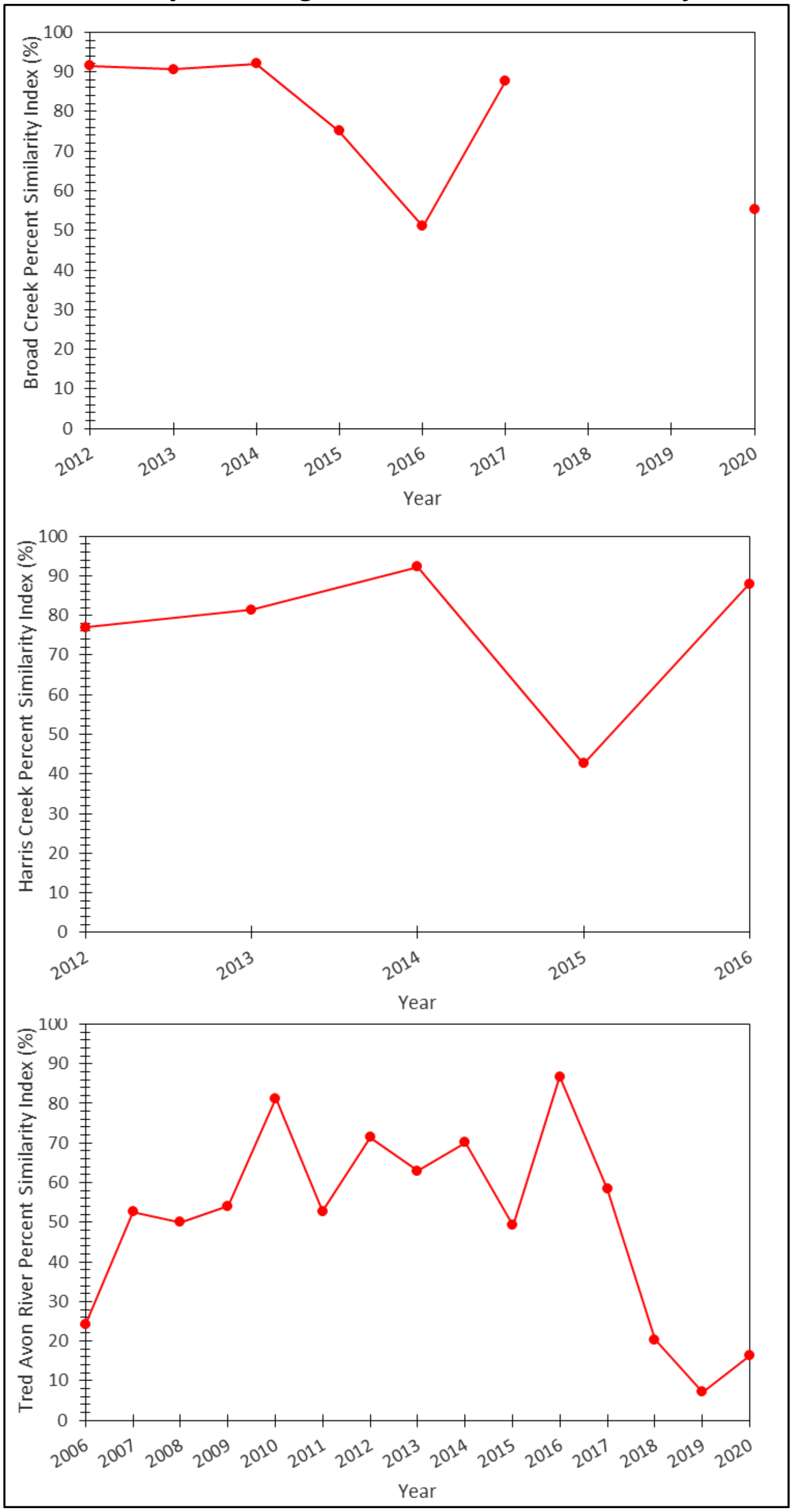


Figure 3-32. Finfish species composition for 4.9 m bottom trawl catch in all mesohaline subestuaries sampled during 2003 – 2020, by year. Finfish species that define the top 90 % are identified, and the remainder of species are grouped and labeled as “other species”.

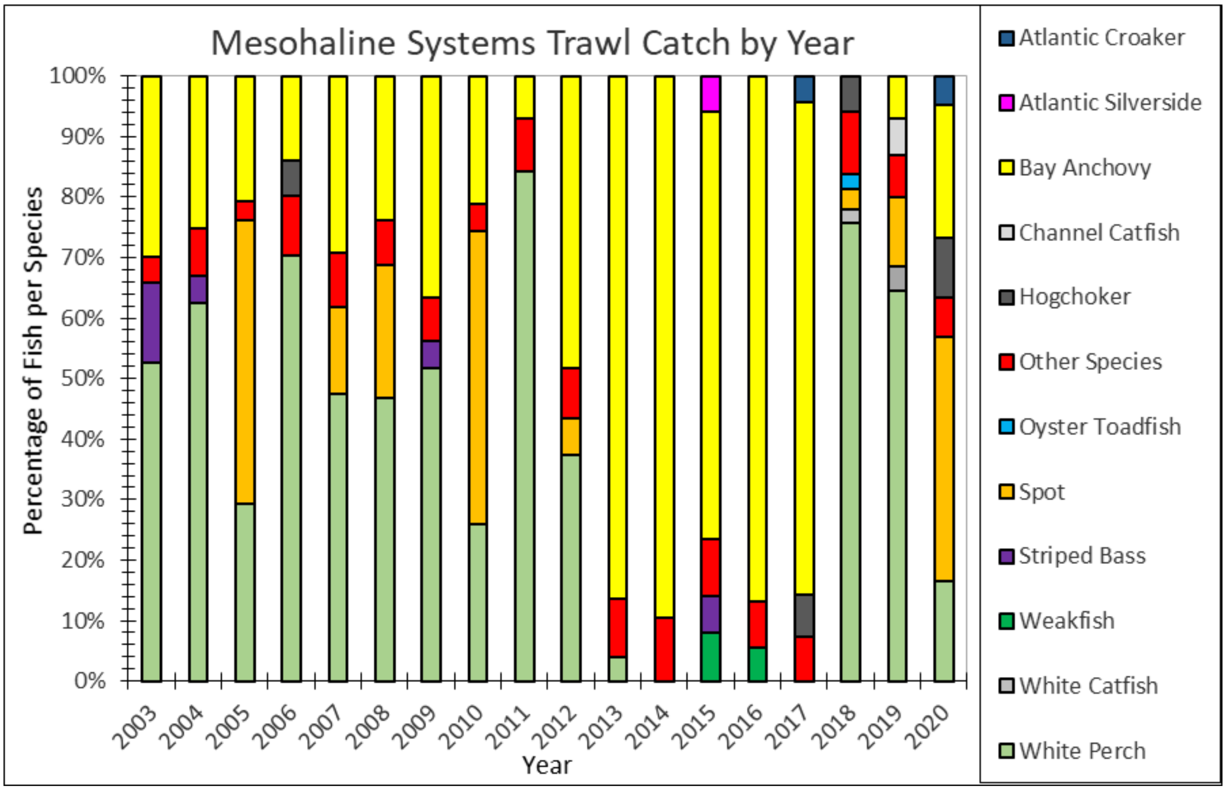


Figure 3-33. Geometric mean (GM) per 4.9 m bottom trawl catch for adult White Perch in Broad Creek, Harris Creek, Miles River, and Tred Avon River, by sampling year. Black bars indicate the 95% confidence intervals. The y-axes range from 0 to 50; x-axes vary based on years sampled.

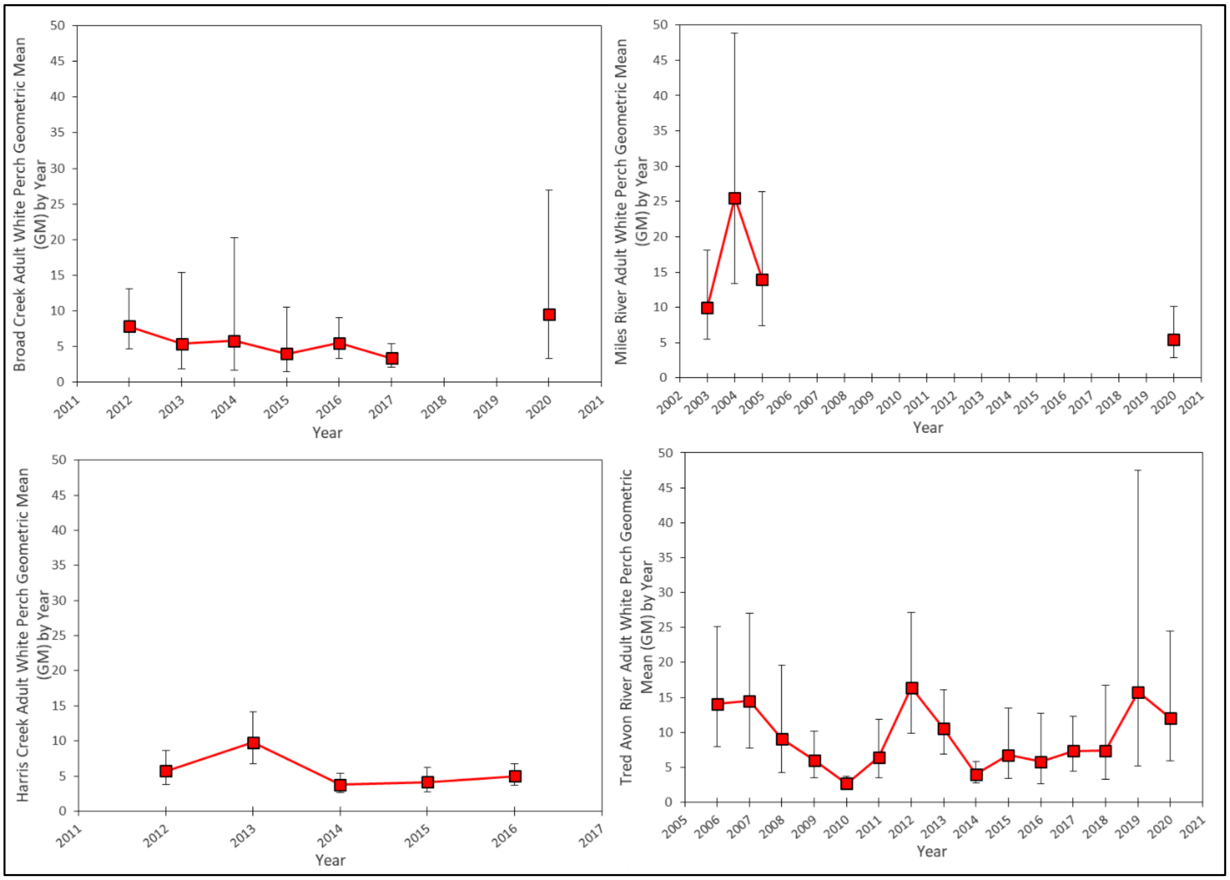


Figure 3-34. Modified proportional stock density (PSD) of White Perch in Choptank River tributaries, Broad Creek, Harris Creek, and Tred Avon River, and the subestuary, Miles River, is the proportion of 4.9 m trawl samples with quality length or greater White Perch.

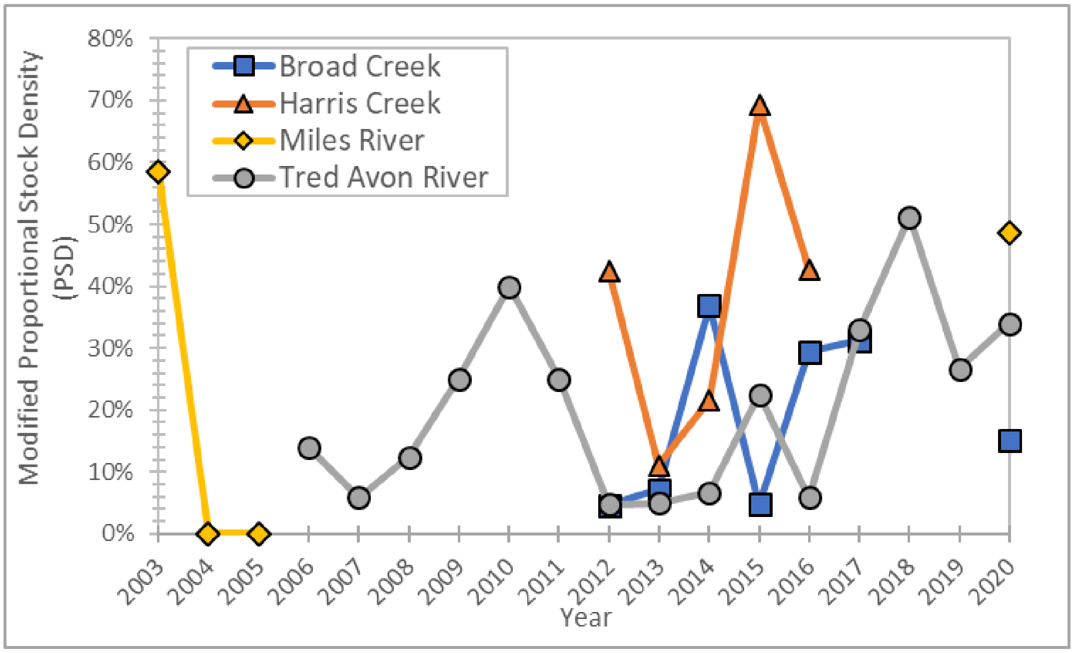


Figure 3-35. Annual beach seine catch geometric mean (GM) per of all finfish species (red squares) for Broad Creek, Harris Creek, Miles River, and Tred Avon River, by sampling year. Black bars indicate the 95 % confidence intervals. The y-axes range from 0 to 700; x-axes vary based on years sampled.

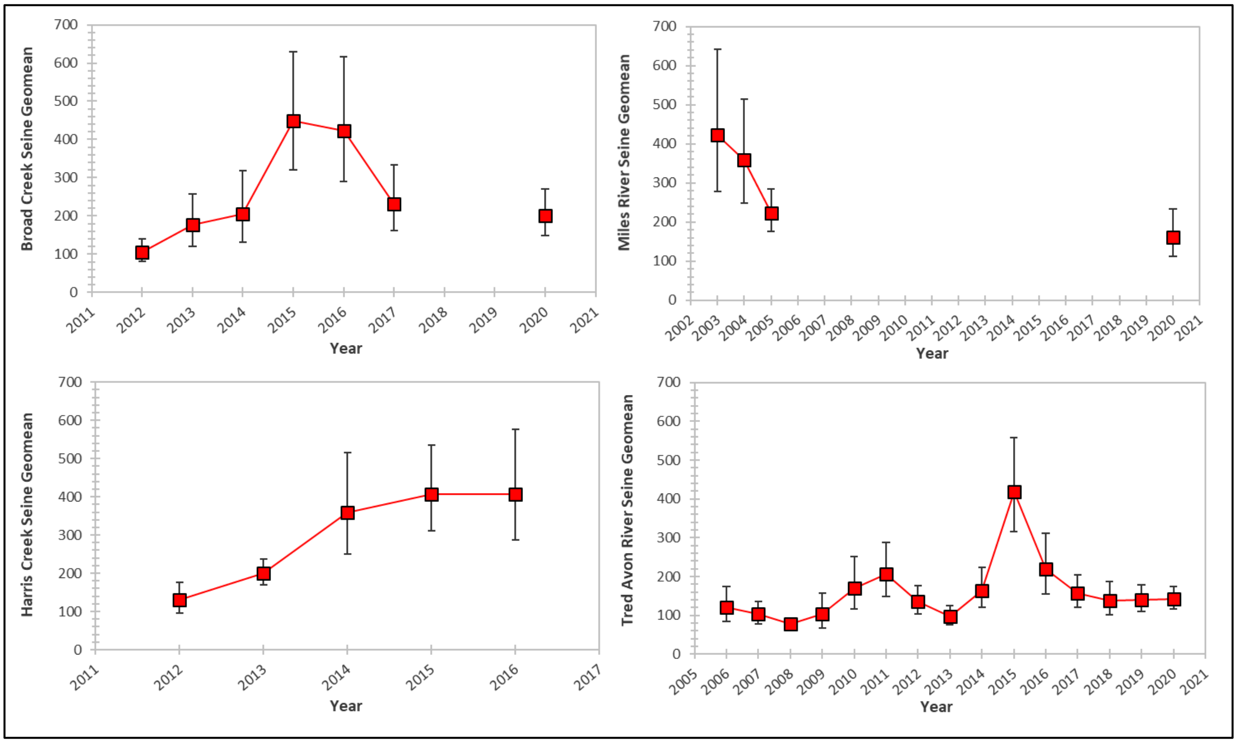


Figure 3-36. Finfish species composition for beach seine catch in Broad Creek, Harris Creek, Miles River, and Tred Avon River for all years combined. Species that define the top 90 % are identified, and the remainder of species are grouped and labeled as “other species”.

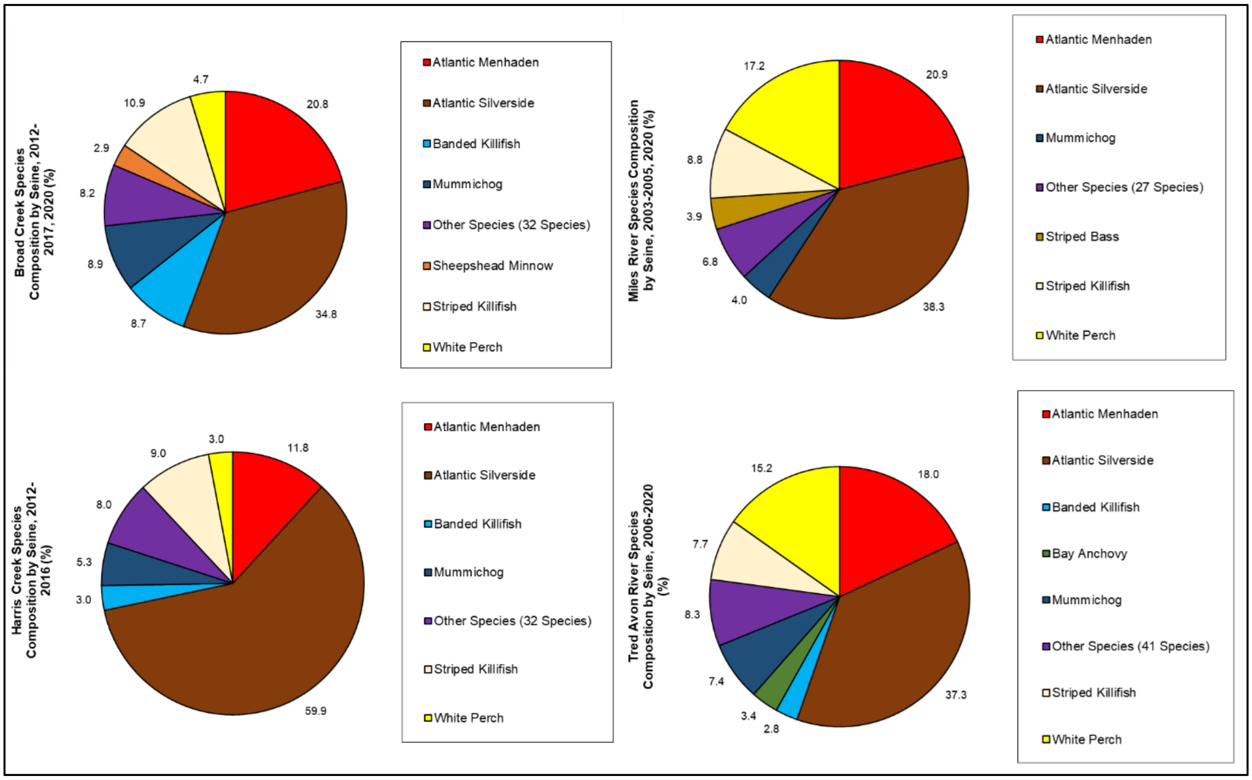


Figure 3-37. Annual 4.9 m bottom trawl catch geometric mean (GM) per of all finfish species (red squares) for Bohemia, Bush, Gunpowder, Middle, Northeast, and Sassafras Rivers, by sampling year. Black bars indicate the 95 % confidence intervals. The y-axes maximums vary for each system; x-axes vary based on years systems were sampled.

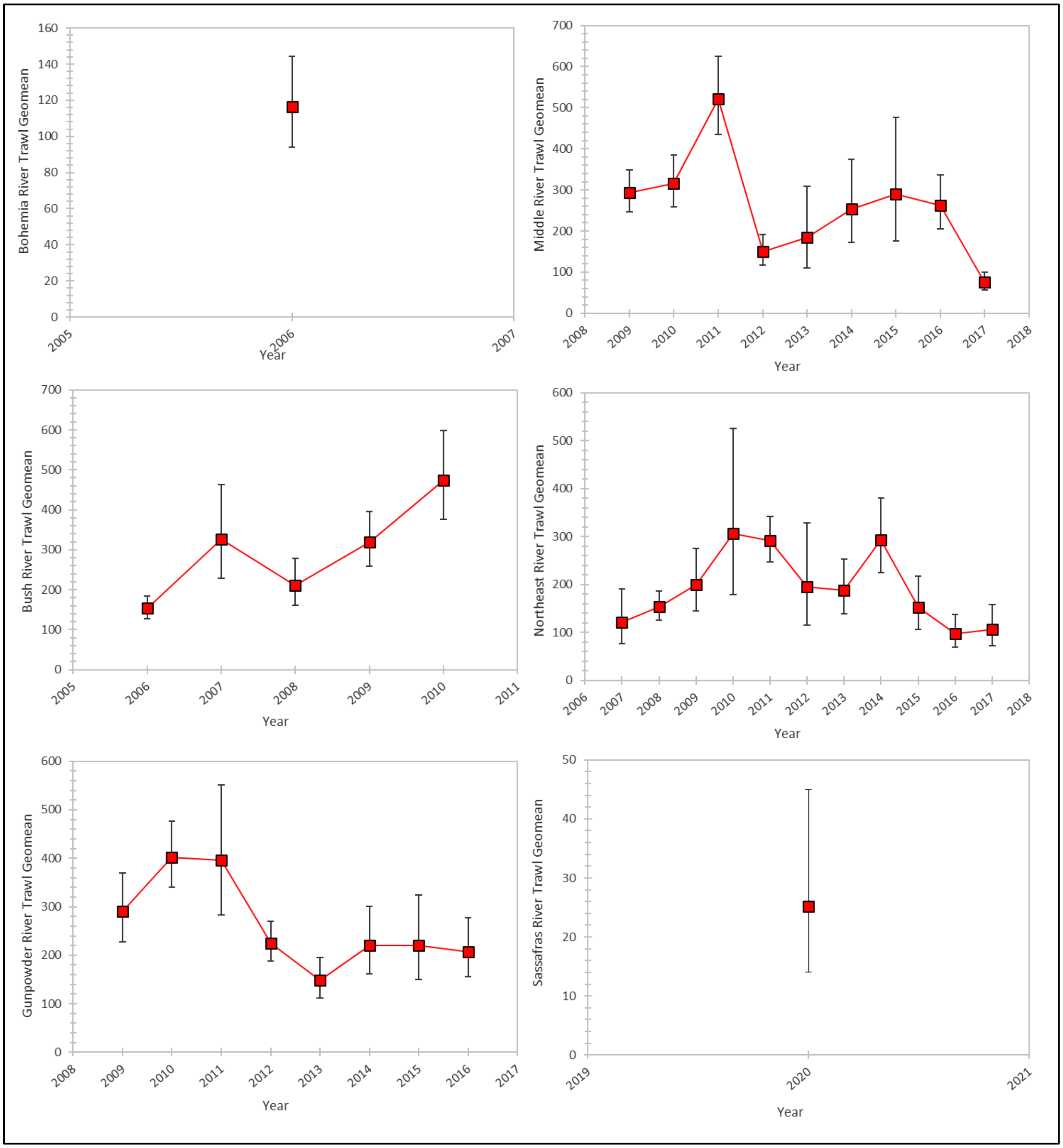


Figure 3-38. Finfish species composition for 4.9 m bottom trawl catch in Head-of-Bay subesestuaries for all sampling years combined. Species that define the top 90 % are identified, and the remainder of species are grouped and labeled as “other species”.

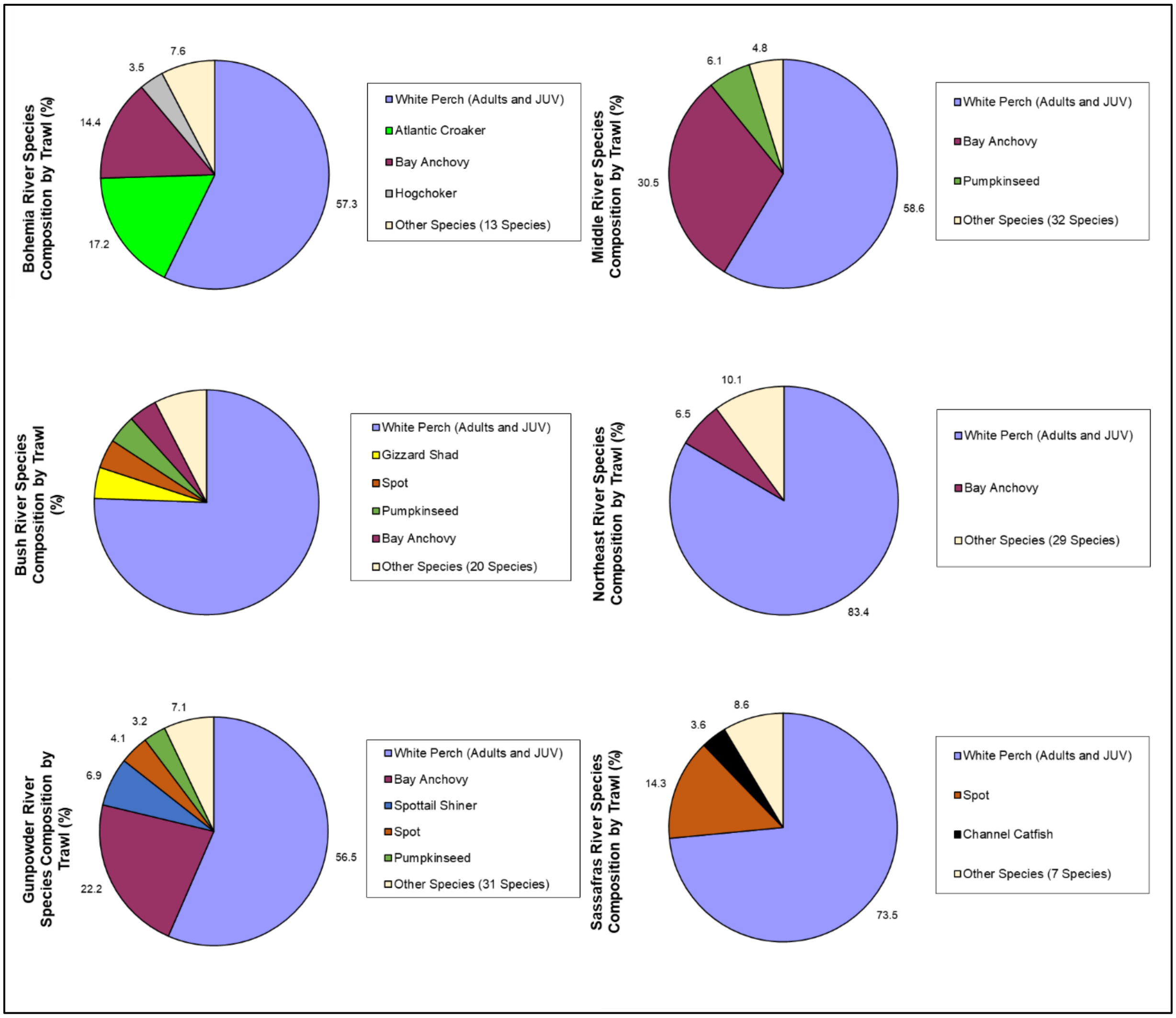


Figure 3-39. Finfish species composition for 4.9 m bottom trawl catch in Head-of-Bay subestuaries, by year. Species that define the top 90 % are identified, and the remainder of species are grouped and labeled as "other species". The y-axes range from 0 % to 100 %; x-axes vary based on years systems were sampled.

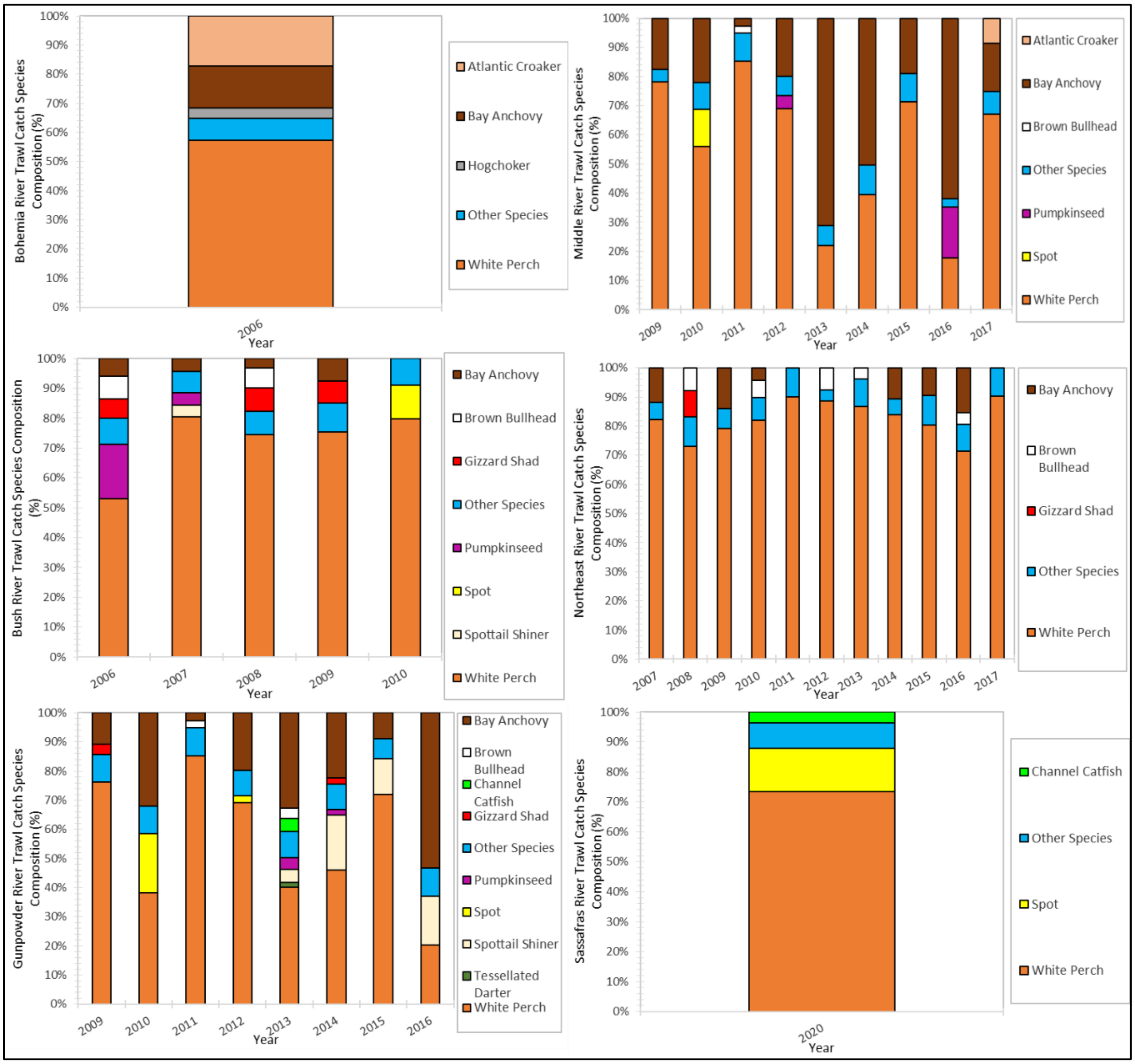
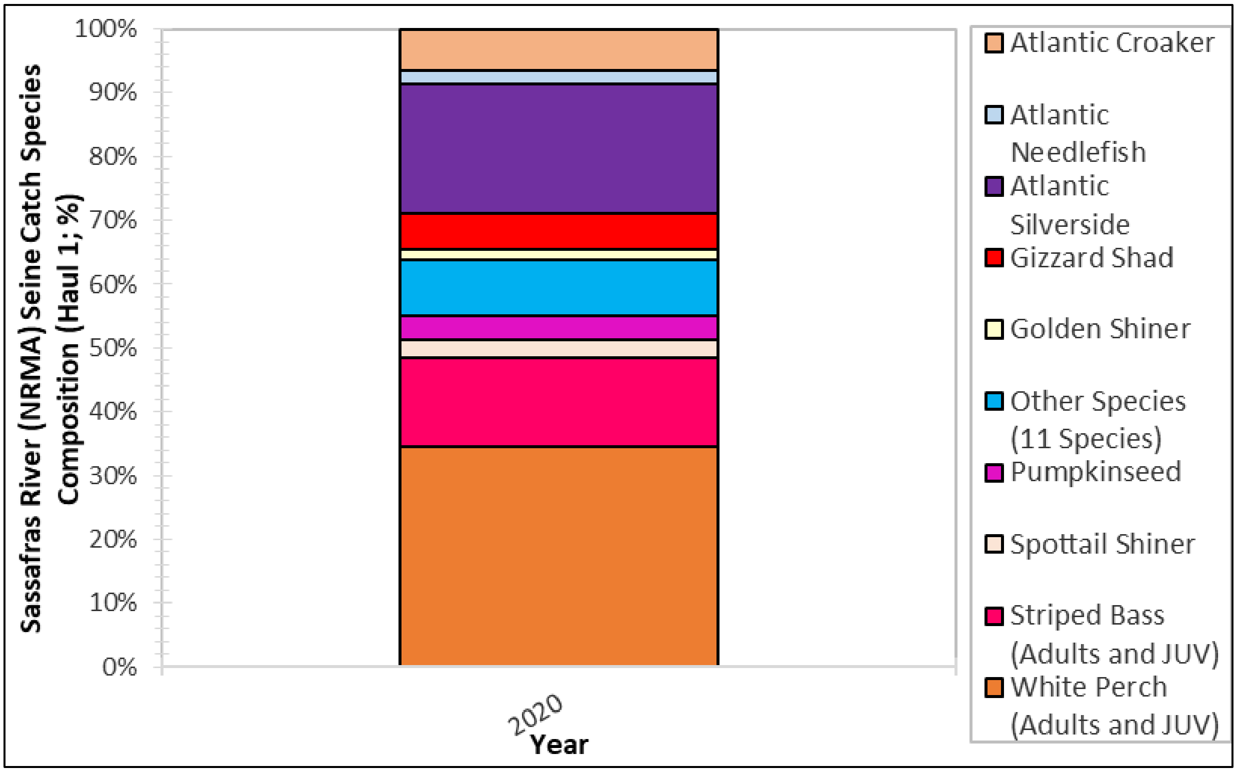


Figure 3-40. Finfish species composition for beach seine in Sassafras River at Juvenile Index (JI) – NRMA seine site during 2020. Species that define the top 90 % are identified, and the remainder of species are grouped and labeled as "other species".



Maryland: Marine and estuarine finfish ecological and habitat investigations

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

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

Introduction

Project 2 documents participation by the Fisheries Habitat and Ecosystem Program (FHEP) in habitat, multispecies, and ecosystem-based management forums that relate to recreationally important finfish in Maryland's Chesapeake Bay and Atlantic coast during July 1, 2020 - June 30, 2021. Activities used information generated by F-63 or were consistent with the goals of F-63.

Changes to Project 2 Planned Activities due to Coronavirus - Activities under Project 2 were altered due to the Pandemic, but virtual meetings and email provided opportunities.

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

Publications – J. Uphoff was a coauthor of two articles published in Frontiers of Marine Science: *Balancing Model Complexity, Data Requirements, and Management Objectives in Developing Ecological Reference Points for Atlantic Menhaden*, (https://www.frontiersin.org/articles/10.3389/fmars.2021.608059/full) and *The Path to an Ecosystem Approach for Forage Fish Management: A Case Study of Atlantic Menhaden (*https://www.frontiersin.org/articles/10.3389/fmars.2021.607657/full).

J. Uphoff reviewed a manuscript on mycobacteriosis in Chesapeake Bay Striped Bass.

Environmental Review Unit Bibliography Database – We maintain an Environmental Review Unit database, adding additional literature as it becomes available. Older reports that are not in electronic format are scanned and saved. Program staff continue to track research and literature regarding restoration effectiveness.

Review of County Comprehensive Growth Plans – We reviewed a Charles County comprehensive growth plan amendment for the Maryland Airport near Indianhead and for two town growth plans, Ellicott City and the Town of Berlin, providing recommendations consistent with maintaining viable fish habitat. These efforts included an assessment of local fisheries resources that represent recreational opportunities and the importance of fish habitat protection in planning. We corresponded with Queen Anne's County planning staff and Corsica River Association on fish habitat and planning as the County works on its comprehensive growth plan.

Cooperative Activities – We worked with Resource Assessment Service (RAS) on criteria for Striped Bass dissolved oxygen and water temperature in summer. J. Uphoff was the lead for FHEP and Tom Parham was the lead for RAS. M. McGinty and C. Hoover participated. A brief summary of the issue and criteria developed through FHEP and RAS exchanges is provided in Appendix 1.

M. McGinty and J. Uphoff worked with R. Bourdon (Fishing and Boating Services) on *Maryland's American Shad Habitat Plan* for the ASMFC. They provided assistance on habitat issues facing American shad and with habitat delineation. The plan can be found at http://www.asmfc.org/files/ShadHabitatPlans/MD_ShadHabitatPlan_2020.pdf .

J. Uphoff participated in a meeting with NOAA's Chesapeake Bay Office (NCBO) climate program staff to review an intern project investigating temperature trends on the Potomac River Striped Bass spawning grounds. Intensity of collections on the spawning ground was too low (few sites and dates) for meaningful interpretation.

A. Park provided data and report links on fish species collected within the Severn River by FHEP and other DNR units (past and present) to a consulting firm.

M. McGinty and J. Uphoff provided advice to Caroline County planners on a stream restoration project for a new park at Red Bridges on the Choptank River.

M. McGinty discussed with Maryland Department of the Environment (MDE) staff on how fish monitoring could be used to evaluate benefit of living shorelines.

M. McGinty met with staff from MD DNR Shellfish program to obtain and enter historical oyster fouling data to continue to refine the hard bottom benthic forage index.

J. Uphoff, M. McGinty, and A. Park provided information on anadromous fish spawning in the Gunpowder and Bush Rivers to the Gunpowder Riverkeeper.

M. McGinty and J. Uphoff provided comments to the Anne Arundel County Parks Department on whether an improved fish ladder is needed for Yellow Perch to access Lake Waterford in the urbanized Magothy River watershed. The fish ladder was highly unlikely to lead to any improvement in reproductive success due to poor viability of eggs and larvae.

J. Uphoff and/or M. McGinty responded to the Magothy River Association inquires about protecting Yellow Perch eggs from raccoon predation. M. McGinty assisted the Magothy River Association in interpretation of some of their habitat data.

J. Uphoff answered an inquiry from Canada's Department of Fisheries and Oceans about habitat conditions associated with Striped Bass spawning in Chesapeake Bay, the inquiry was related to the re-establishment of a spawning population in the Saint Lawrence River.

J. Uphoff, M. McGinty, A. Park, and C. Hoover met with M. Topolski to discuss how tax map estimates of impervious surface (based converting structure per hectare estimates to percent impervious surface based on their relationship with 1999-2000 Towson University impervious surface estimates) compared to impervious surface estimates from the high resolution Chesapeake Conservancy Land Use data. Despite having been developed from lower resolution data, tax index estimates of impervious surface from Towson University data compared well to these high resolution estimates. An analysis is being finalized.

M. McGinty participated in a virtual meeting with NOAA regarding the President's Executive Order on Climate Change. She worked with Fishing and Boating Services' staff to develop a list of talking points regarding climate impacts to Maryland's fisheries.

J. Uphoff, M. McGinty, and A. Park answered questions and provided comments on habitat in the Middle and Gunpowder Rivers with the MD DNR Tidal Bass Program. The issue was a disease outbreak in Largemouth Bass. Summer water quality and fish community data collected by FHEP was shared.

J. Uphoff and A. Park shared summer seine catch data of Four-Spine Stickleback with a professor at East Carolina University, a student is interested in a possible study involving color variation and distribution of sticklebacks.

J. Uphoff and M. McGinty reviewed a NOAA habitat assessment of the Choptank River.

J. Uphoff and M. McGinty, as members of the steering committee of a Virginia Institute of Marine Science doctoral student, attended several meetings on developing habitat suitability models for ages 0-4 Striped Bass in Chesapeake Bay.

Presentations and Outreach – J. Uphoff attended the 2020 annual national American Fisheries Society virtual meeting. He presented *Spawning Success of Anadromous Herring in Patuxent River (Maryland, USA)* as part of a symposium: *Confronting Present and Emerging Stressors in Rivers for Global Fisheries Conservation*, participated in a symposium panel discussion, and was interviewed for a podcast. M. McGinty, A. Park, and C. Hoover attended the annual national American Fisheries Society virtual meeting held on September 14th – 25th, several virtual webinars were attended on various topics.

J. Uphoff and M. McGinty attended the 2021 virtual meeting of the Maryland Water Monitoring Council.

J. Uphoff attended a webinar on the USFWS's new federal aid tracking system (TRACS).

J. Uphoff, M. McGinty, A. Park, and C. Hoover participated in various webinars, including seminars on effects of road salting, eDNA, stormwater design, submerged aquatic vegetation, environmental contaminants, restoration effectiveness, water quality and bacteria, microplastics, and ocean acidification.

J. Uphoff worked on proposals for two symposia that were accepted for the annual national American Fisheries Society meeting that will be held in Baltimore during November 2021.

A. Park presented on the Bush River estuarine fish community sampling conducted by volunteers during 2019 and 2020 for Anita C. Leight Estuary Center (ACLEC) via a webinar. Data was collected by FHEP during 2006-2010) and sampling was turned over to ACLEC volunteers in 2011. A presentation on their data is made annually during ACLEC's volunteer training workshop.

ASMFC – J. Uphoff continued to work with the Atlantic Menhaden Ecological Reference Point workgroup on forage reference points. Targets and limits for Menhaden that allow them to maintain their forage role were adopted by the Atlantic Menhaden Board at the August, 2020, meeting.

J. Uphoff participated in an ASMFC conference call on menhaden quota estimates.

J. Uphoff advised the Chair of the ASMFC Habitat Committee on habitat related sections of ASMFC's Striped Bass Amendment 7.

Chesapeake Bay Program – M. McGinty participated in Fish Habitat Action Team meetings and J. Uphoff participated in Forage Action Team meetings. Both participated in the Sustainable Fisheries Goal Implementation winter and summer meetings.

Envision the Choptank – Envision the Choptank (https://www.envisionthechoptank.org/) is a collaboration of conservation organizations, government agencies (town, county, state, and federal), and local citizens that work to maintain and improve the viability of the Choptank River's water quality and natural resources. J. Uphoff and M. McGinty participated in virtual meetings of the Envision the Choptank Working with Local Government workgroup. Engaging in county and town comprehensive plan updates to strengthen natural resource components is a priority of this workgroup and we hope the Envision effort will lead to more consideration of fish habitat in county planning. J. Uphoff and M. McGinty helped with the development of a National Fish and Wildlife Fund proposal for water quality projects and a letter of support from MD DNR.

Literature Searches - Literature searches were also conducted on the following topics: carbon cycling and its relationship to supply and restoration; impacts of road salt on denitrification and carbon processing; macroinvertebrates and restoration; the potential for restoration to have adverse consequences (particularly in regards to release of concentrated chemicals/contaminants previously sequestered); remediation approaches to reduce nutrients and organics (i.e. wetlands and wastewater treatment plants); impacts of road salt interfering with natural processes (elevated soil salinities and reduction of plant species); impacts of road salt on denitrification and carbon processing; and salinization and mobilization.

Appendix 1

Cooperative Research with Resource Assessment Service on development of criteria for depicting the effect of temperature and dissolved oxygen on summer habitat for Striped Bass in Maryland's portion of Chesapeake Bay

Introduction

Hypoxia may negatively impact summer habitat conditions for Striped Bass in Chesapeake Bay (Maryland Sea Grant 2009). The interplay of dissolved oxygen (DO) and water temperature and its impact on Striped Bass in Chesapeake Bay has become an increasing focus of modeling and forecasting because of their importance as a game and commercial fish in the Bay (J. Uphoff, personal observation). Early versions of the temperature-DO squeeze (TOS) hypothesis (mismatch of water column regions of desirable temperature and DO) developed from reservoir fisheries (Coutant 1985) and extended to estuarine and marine habitat (Price et al. 1985; Coutant 1990) are being used to interpret the results of models depicting the effect of hypoxia on Striped Bass habitat in Chesapeake Bay (J. Uphoff, personal observation).

The null hypothesis of the Coutant (1985) TOS hypothesis was that Striped Bass mortalities were due to limited availability of cool (<25˚C), oxygenated water (>2 mg/L; Coutant 1985). Summer mortalities of large Striped Bass (generally >5 kg) in some reservoirs and lakes in the southeastern US, attributed to TOS, have been a recurring management issue (Coutant 1985; Coutant 2013).

The TOS hypothesis has not been supported for Striped Bass in Chesapeake Bay, reflecting higher temperature tolerance indicated by bioenergetics models, conventional tagging, and acoustic telemetry (Hartman and Brandt 1995a; Constantini et al. 2008; Kraus et al. 2015; Groner et al. 2018; Itakura et al. 2021). In addition, Coutant (2013) modified his original TOS hypothesis to reflect additional studies and experience. Tolerance of warm water was influenced by Striped Bass size and-or age (smaller fish, 2-4 kg, were more tolerant), duration of exposure, quantity of food available, and stress from catch-and-release (Coutant 2013).

A draft summer Striped Bass water temperature and DO assessment by the Maryland Department of Natural Resources (MD DNR) Resource Assessment Service (RAS) was sent to FHEP and the MD DNR Striped Bass programs for review and comment in summer 2020. Lengthy discussions through email and phone followed and a collaborative project to develop summer criteria started.

Multiple studies of Striped Bass temperature and DO tolerance have been conducted in Chesapeake Bay from 1995 to 2021 (Table 1). Results of these studies did not appear to have been incorporated into recent Chesapeake Bay Program evaluations of the impact of summer water temperature and hypoxia on availability of Striped Bass Habitat (J. Uphoff, personal observation). The Striped Bass Program, RAS, and FHEP reviewed these studies to develop water temperature and DO criteria that reflected these more current evaluations (Table 1). These criteria would be applied by RAS to analyses of summer Striped Bass temperature and DO conditions for MD DNR. These analyses will look at condition during 2010 to 2019 (and possibly later; T. Parham, MD DNR RAS, personal communication).

Methods

Criteria were developed from a literature review of Chesapeake Bay Striped Bass studies that evaluated temperature and-or DO, and the Coutant (2013) update of the TOS hypothesis in southeastern US reservoirs (Table 1). The Coutant (2013) paper was chosen to represent multiple lake and reservoir studies because it was a summary of TOS information in a region known to experience TOS related issues and because it updated Coutant (1985; 1990) that are heavily relied on for Chesapeake Bay assessments of hypoxia. Three studies (Hartman and Brandt 1995a; Constantini et al. 2008; Kraus et al. 2015) used “scope for growth” (the potential for growth at a given temperature at maximum consumption) or “growth rate potential” (illustrates the potential for growth at a given temperature at less than maximum consumption) estimated from bioenergetics models to evaluate temperature and-or DO. Growth potential was used as the label for both, but reported ration size was noted. Growth potential is based on the difference between energy gained from consumption and used through metabolism. It can be positive (growth) or negative (depletion). Itakura et al. (2021) evaluated habitat occupation of Striped Bass during summer using acoustic tags. Groner et al. (2018) evaluated survival of conventionally tagged Striped Bass (healthy or with different degrees of mycobacteriosis) in Rappahanock River, Virginia. Coutant (2013) reviewed multiple studies conducted in 24 southeastern US reservoirs that used condition, growth, habitat occupation, and mortality as responses (Table 1).

We confined the size of Striped Bass to be evaluated to those likely to be residents (Table 1). Residents are a large contingent of Chesapeake Bay Striped Bass that do not participate in the Atlantic coast migration (mostly males along with some young females; Maryland Sea Grant 2009). They constitute a year-round population that provides Maryland's major saltwater recreational fishery and an important commercial fishery. About 90% of Striped Bass harvested in Chesapeake Bay during 2005-2007 were 3 to 6 years-old and 457-635 mm TL (457 mm TL minimum size; Maryland Sea Grant 2009). Based on a plot of 2,047 weights and lengths of Striped Bass collected in summer, 1999-2011, by the MD DNR Fish and Wildlife Program, the approximate range in average weight for Striped Bass 457-635 mm TL would have been from 1.8 to 2.5 kg (J. Uphoff, unpublished analysis). Ages 2-6 made up the vast majority of Striped Bass sampled from pound nets in Maryland's portion of Chesapeake Bay during June-November, 2017 (Horne 2020).

We noted whether salinity, temperature, DO, and food ration were considered by a study Table 1). A general description of the study's response variable (growth potential, habitat occupation, mortality, etc.) was provided and whether we considered the study was a test of the TOS hypothesis was indicated (Table 1).

We defined four categories for summer temperature and DO conditions for Striped Bass in Chesapeake Bay (suitable, tolerable, marginal, and unsuitable). Results from each study were interpreted into these categories.

Suitable habitat supported "normal" long-term occupancy and this category provided the upper bound of the best water temperature and DO combination. This category supported occupancy at all times, growth potential was positive, and did not cause excess mortality. Note that suitable is not necessarily the same as preferred, but preferred habitat would fall somewhere in the suitable category.

Tolerable habitat would support occupancy for a modest period, approximately 1 month. Growth potential would be limited or negative. Mortality would be minimal to modest within the time period and higher if conditions persisted.

Marginal habitat supported very brief incursions with little impact on growth potential because of the short duration of exposure that could be tolerated. Less severe conditions would be nearby. Marginal habitat had potential for high mortality beyond brief exposure.

Unsuitable habitat did not support incursions or occupancy. If it could be avoided, mortality would be minimal. It would be lethal if it could not be avoided.

Results and Discussion

Not all studies provided water temperature and DO values for all categories. Three of 6 papers reviewed had values for each water temperature category and 2 of 6 had DO values for each category (Table 2). Among the reviewed studies, water temperatures between 27 and 28˚C described the upper bound of the suitable category (6 papers); the lower bounds for DO was 4.0 mg/L (4 papers). The tolerable water temperature category upper bound was 29˚C (4 papers), while the lower bounds of tolerable DO values were from 3.0 mg/l to <5.0 mg/L (3 papers). Marginal temperature upper bounds were 30-31˚C (4 papers) and marginal DO lower bounds fell between 1.0 and 2.0 mg/L (2 papers). Unsuitable habitat descriptions ranged from >30 ˚C to >31

˚C (upper bound; 4 papers) for water temperature and <1.0 to <3.0 mg/L (lower bound; 5 papers; Table 2).

The following habitat criteria were reached by consensus. Suitable habitat boundaries (supports occupancy at all times, growth potential was positive, and did not cause mortality) were water temperature < 28˚C and DO ≥ 4.0 mg/L. Tolerable habitat (supports occupancy for a modest period of time, with limited or negative growth potential, and with little or no mortality) was bounded by water temperature >28 ˚C to 29˚C and DO > 3.0 to 4.0 mg/L. Marginal (supports very brief occupancy with potential for high mortality beyond brief exposure) was bounded by water temperature > 29˚C to 30˚C and DO ≥ 2.0 to 3.0 mg/L. Water temperature > 30 ˚C and DO < 2.0 mg/L (conforming to the Chesapeake Bay Program hypoxia definition) defined unsuitable habitat that would be avoided.

The DO criterion for suitable habitat we arrived at differed from DO concentrations of ≥ 5 mg/L or greater that are considered desirable for many Chesapeake Bay living resources and have been adopted into the Chesapeake Bay states water quality standards regulations (Batiuk et al. 2009). The lower bound of our tolerable category was the same as the DO criterion for deepwater fishes and shellfish that calls for maintaining a 30-d mean of 3 mg/L during June 1–September 30 in bottom waters (Batiuk et al. 2009). However, the 30 day mean (by definition of the mean) could include substantial periods within the marginal and even unsuitable categories.

Only water temperature and DO were considered in developing these criteria, but they may not sufficiently portray the stress experienced by Striped Bass in summer. Summer may be particularly stressful since it can also be a period of limited feeding success and poor condition (Uphoff et al. 2017), no to negative growth in weight for ages 3-6 (Hartman and Brandt 1995b), higher mortality of diseased and healthy Striped Bass (Groner et al. 2018), 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 DO in reservoirs (Coutant 2013). Condition of Striped Bass within Maryland’s portion of Chesapeake Bay has improved between 1998 and 2020, especially for fish >457 mm TL (Uphoff et al. 2020; see Project 4). Size is an influence and smaller fish are more tolerant of temperature. Two 2 to 4 kg Striped Bass in reservoirs occupied temperatures up to 30 ˚C without mortality (Coutant 2013). Most of the studies in Table 1 featured Striped Bass that would be considered larger residents (near the legal length limit and above). Multiple catch-and-release events limit growth and condition (Diodoti and Richards 1996; Stockwell et al. 2002), suggesting they could have an influence on reaction to temperature and hypoxia. The presence of salinity was associated with much lower catch-and-release mortality in experiments conducted in Maryland’s mesohaline portion of Chesapeake Bay than those estimated for freshwater fisheries (tidal and nontidal; Lukacovic and Uphoff 2007). Water temperature did not have a detectable influence survival of released Striped Bass in experiments, but air temperature did. Air temperature may have served as a surrogate for the difference in temperature between water and air in experiments, although air temperature may have also been a proxy for seasonal factors (condition and aggressiveness; Lukacovic and Uphoff 2007).

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Table 1. Characteristics of studies used to derive temperature and DO criteria for summer habitat of striped bass in Maryland's portion of Chesapeake Bay. Y (yes) indicates that a factor was present in a study and N (no) indicated it was not. TOS – temperature-oxygen squeeze.

Study	Hartman and Brandt 1995 (H & B 1995)	Kraus et al. 2015	Constanti et al. 2008	Coutant 2013
Type	Bioenergetics (model)	Observation (acoustic tag) and bioenergetics	Observation, Lab (DO), and bioenergetics	Review of 20 years, multiple studies, 24 locations
Habitat	Chesapeake Bay	Chesapeake Bay	Chesapeake Bay	SE US inland
Salinity	Y	Y	Y	N
Fish Size	1 – 3 kg	2 – 4 kg (ages 3-6)	Ages 2 & 4	2 – 9 kg
Temperature	Y	Y	Y	Y
DO	N	Y (% saturation)	Y (mg/L)	Y (mg/L)
Food ration	Y (maximum feed)	Y (assume ½ ration)	Y (field estimates)	Y (some studies)
Response variable	Growth potential	Growth potential Uses (H & B 1995)	Growth potential (uses H & B 1995)	Habitat occupation, die-offs, condition
Bay TOS test	N	Y	Y	N

Table 1. (continued)

Study	Itakura et al. 2021	Groner et al. 2018
Type	Observation (acoustic tag) TSO hypothesis test	Observation (conventional tag)
Habitat	Chesapeake Bay	Chesapeake Bay
Salinity	Y	Y
Fish Size	451-1060 mm, 65% > 600 mm 1.2-16.2 kg, mean 3.5 kg	457-610 mm (95%)
Temperature	Y	Y
DO	Y (% saturation)	Y (mg/L)
Food ration	N	N
Response variables	Habitat occupation	Survival
Bay TOS test	Y	Y

Table 2. Temperature and dissolved oxygen boundaries assigned to habitat categories, by study. Synthesis = consensus limits assigned to a variable category.

	Reference						
Category	Hartman and Brandt 1995	Kraus et al. 2015	Constantini et al. 2008	Coutant 2013	Groner et al. 2018	Itakura et al. 2021	Synthesis
			Temperature ˚C				
Suitable	28	Mean 28.4-29.2	28	28	27	27-28	28
Tolerable	29	for both		29		29	29
Marginal	30	31		30		31	30
Unsuitable	>30	>31		>30		>31	>30
			Dissolved Oxygen mg/L				
Suitable	N/A	Mean 5.6-6.6	4-4.5	4		≥5	4
Tolerable	N/A	for both	3	3		<5	3
Marginal	N/A		1	2			2
Unsuitable	N/A	<2	< 1	<2	<3	<2	<2

Table 2. Factors (stressors) potentially influencing temperature tolerance not included in Chesapeake Bay water temperature and DO criteria.

Factor	Influence	Conditions associated with influence	References
Feeding	Negative	If poorly fed or in poor condition	Coutant 2013
			Contstantini 2008
Mycobacteriosis	Negative	Severe status	Lapointe et al. 2014
			Groner et al. 2018
Catch and Release	Negative	Multiple releases lower condition	Diodoti and Richards 1996
		Multiple releases limit growth	Stockwell et al. 2002
Fish Size	Negative	Sensitivity increases with size	Coutant 2013
Salinity	Positive	Reduces osmotic stress	Hatchery experience
		Reduces catch and release mortality	Lukacovic and Uphoff 2007

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

Refining Anadromous spawning Maps for Management Applications

Margaret McGinty

Abstract

As part of an effort to provide mapping tools for environmental disturbance permit decisions (i.e., environmental review), a robust spatial dataset was developed that represented historical locations of anadromous fish spawning (non-tidal and upper tidal reaches of streams where American Shad, Alewife and Blueback Herring, White Perch and Yellow Perch spawn). These maps are used by reviewers from state and federal agencies. However, these maps did not account for changes due to development that depreciate anadromous spawning habitat. Nor did they differentiate habitat where anadromous spawning was absent from where it was undocumented because they only included stations where anadromous spawning was present. More recently, these maps were updated to include sites sampled where anadromous spawning was absent and the level development in a watershed. Afterwards, environmental reviewers requested additional information to identify potential upstream and adjacent reaches that could contribute to habitat quality of an anadromous spawning reach. I developed a methodology to identify anadromous spawning stream segments and habitat that contributed to anadromous spawning to expand the detail of maps for reviewers.

Project 3 Narrative

Changes to Project 3 Planned Activities due to Coronavirus – Face-to-face activities under Project 3 were cancelled due to the pandemic, but virtual meetings and email provided opportunities to continue developing spatial data and tools.

Spatial Data Development - With the advent and availability of desktop mapping applications, digital maps have become common in natural resource management. Maps allow visualization of the spatial dimensions of specific habitats or species of interest (Patterson et al. 2003; Ridgely et al. 2003), as well as stressors that could impair habitat suitability and species occurrence (Thornbrugh et al 2018; Oliver et al. 2018). Various map layers can be overlayed to assist managers in identifying key ecological features and prioritizing management based on specific objectives. For example, Torbick et al. (2013) used LANDSAT data to characterize lakes based on water quality metrics in Michigan and then overlayed land use to determine if land use could predict a lake's water quality. Others such as Halpern et al. (2008) and Paukert et al. (2011) have used maps to assess the spatial extent of anthropogenic stressors and assess ecosystem risk from stressors

Mowrer and McGinty (2002) produced the first digital map of anadromous spawning habitat in Maryland. Focal species included American Shad, Alewife, Blueback Herring, White Perch, and Yellow Perch. These maps were developed from historical survey data and represented presence of anadromous fish species based on observation of eggs, larvae or adults (and in some cases juveniles) present on spawning grounds during spawning season (Uphoff et al. 2020, Job 3). These maps have been used in print and digital form to identify anadromous spawning areas and promote protection against adverse impacts from building and landscape alterations that require permits. However, these maps represent anadromous spawning habitat occupation during the 1970s and 1980s and do not account for more contemporary changes in habitat due to land use change. Thus, all habitat is viewed equally regardless of level of development, although recent studies indicate anadromous fish spawning declines as development increases (Limburg and Schmidt 1990; Limburg and Waldman 2009; Uphoff et al. 2020; Hare et al. 2021). The level of development in a watershed measured by housing density or impervious cover is a good predictor of habitat quality as indicated by presence of early life stages (Uphoff et al. 2013; 2020).

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

To provide a more complete tool for IRT reviewers, we updated the database (Uphoff et al. 2020, Job 3). The initial database included species presence with locational information. The most recent update included presence and absence, along with information on collection date, reports cited, location and sample type. The addition of stations sampled that indicated absence of anadromous spawning was incorporated to give the reviewers confidence that a stream was sampled when it was categorized as non-anadromous spawning habitat (Uphoff et al. 2020, Job 3).

Additionally, we incorporated information about the level of development in a watershed, so reviewers could consider the present state of habitat (Uphoff et al. 2020). We offered guidance suggesting they focus detailed reviews that called for applying compensatory measures in watersheds with less than 10% impervious cover (Uphoff et al. 2020).

After review, the workgroup requested an expansion of the maps to identify upstream watershed segments that could potentially impact a specific site. After several meetings and review of historical reports, I developed an approach to expand utility of the maps for requested applications.

To keep the task tractable, I focused on 'inland stations" (Figure 1). I present Anne Arundel County maps in this report because the scale is smaller and allows for better visualization of features. Maps are available for all relevant counties statewide where anadromous spawning was observed. These were historical stations on streams that could be passively sampled by wading to the station and holding a net in place to capture suspended eggs and larvae (Uphoff et al. 2020). Most of these stations were on nontidal streams with a few on small tidal streams. I then overlayed a file documenting stream blockages developed by Jim Thompson of MD DNR's Fish Passage Program (MD DNR 2021).

I identified anadromous spawning segments using the following criteria: If anadromous spawning was observed at a station, I defined the segment by tracing the stream (1) up to a blockage or (2) the point at which the stream order changed (Figure 2). For example, if anadromous spawning was observed on a third order stream and the stream was unimpeded by a blockage above the station, the segment represented the length of unblocked stream up to the point at which the stream order changed from third to second order. I did not consider smaller order streams as part of the segment, because in the original survey, O'Dell et al (1970) limited sampling to third order streams and larger. I assumed they considered smaller streams unlikely to support anadromous spawning. By this approach, I proposed that an entire stream segment represents potential anadromous spawning habitat (i.e., habitat accessible to migrating adults for anadromous spawning). I consider this a liberal approach for defining anadromous spawning segments, as it does not account for potential water quality impairments from toxins or road salts. (Note, it is possible to incorporate contaminant data overlays when they are available if reviewers are interested in examining additional stressors; see Uphoff et al. 2018, Job 3).

Stream segments and contributing habitat were designated by editing a stream file (MBSS100c.shp) developed by Maryland Biological Stream Survey (MBSS 2021). Catchment data were developed from USGS National Hydrologic data (USGS 2012) with impervious cover estimates developed from housing density date estimated from Maryland Department of Planning Maryland (MDP) Property Tax Database (MDP 2020; See Project 1, **General spatial and analytical methods used in Project 1**).

After defining segments and presenting these to the IRT mapping workgroup, they indicated a need to consider additional upstream disturbances, including potential sediment impairments from construction as well as water quality impacts. In response I developed two additional layers to apply in the review process.

The first layer included unblocked contributing habitat. This layer was developed by identifying streams that contributed to an anadromous spawning segment through a connected network of streams that flowed to a direct confluence with the anadromous spawning segment (Figure 3). In other words, any unblocked streams that would eventually flow into an anadromous spawning segment were identified as unimpeded contributing habitat. These unblocked streams were considered habitat that could potentially contribute sediment to an anadromous spawning segment during a disturbance event. This assumed that all blockages entrain sediment under all flow conditions. I recognize this is not the case, but applied it as a standard approach to help reviewers focus on key issues and areas.

The second layer developed included blocked streams that would eventually flow either directly or indirectly (through unblocked contributing habitat) to an anadromous spawning segment (Figure 4). These streams were those above blockages on streams identified as contributing habitat, or above blockages on anadromous spawning segments. These were considered habitat that could contribute pollutants to the anadromous spawning segment.

To account for changes in habitat related to land use change, I also developed a catchment file that identifies the percentage of impervious cover in the catchment (Figure 5). Planners can include this information to determine if they want to pursue strict management and impose restrictions on a project. For example, if an anadromous spawning segment lies in a catchment or catchments with low impervious cover, they may want to encourage precautionary measures to limit impacts to anadromous spawning. Conversely if the segment lies in highly developed watersheds, reviewers may be more lenient or use mitigation to compensate.

These catchment files provide reviewers a holistic view of the area around a proposed site and may help them identify local mitigation banks. Depending on programmatic restrictions, practitioners may decide to go outside a disturbed watershed to identify a mitigation area that is less disturbed and would benefit from local restoration actions to increase habitat resilience.

These additional data layers provided maps that project reviewers requested to identify potential free flowing anadromous spawning habitat (unimpeded stream segments where anadromous spawning presence was indicated at a site or sites on a segment) and habitat that could potentially impact anadromous spawning segments if events cause a water quality disturbance (sediment movement and mobilization of contaminants) or increased concentrations of salt due to increased road salting and urban weathering (Kaushal et al. 2017). This will allow reviewers to map a project location and visualize the potential for it to impact an anadromous spawning segment. This satisfied concerns raised by the IRT workgroup regarding limitations of point data and their concerns that they under-represent anadromous spawning habitat.

I recognize that a better approach would involve identifying a defined radius or distance of influence related to a sampling station. It is unlikely we will get to this point, because the variety of stream and watershed characteristics at play would require much more refined data than presently available, along with development of predictive models that account for changing climate patterns and other anthropogenic influences.

I recommend exploring the application of the recently developed RAD (Resist, Accept, Direct) framework for evaluating projects (Shuurman et al. 2020). This approach offers managers a decision framework they can apply in the face of irreversible ecosystem change instigated by global climate change and other anthropogenic stressors such as development (Williams 2021). Managers are encouraged to consider resisting ecological changes in habitats that can maintain natural resilience in the face of stressors. In habitats where ecological changes cannot be overcome, managers accept the changes and consider alternative management strategies. In habitats where ecological stressors are reshaping the ecosystem, managers may manage for an ecological regime that provides new services or enhances services that remain viable.

I recommend applying the RAD approach in context of already developed impervious thresholds (Uphoff et al. 2011; 2020). Watersheds with low impervious cover (<10%) continue to support productive anadromous spawning and are areas where managers should continue to work to resist change. Actions to resist change could include working with localities to apply conservative zoning that maintains the rural character of the watershed and restoring streams within the watershed that promote habitat connectivity (restoring hydrological connections) and quality, and building resilience (reinforcing riparian areas, limiting water withdrawals). In watersheds with impervious cover between 10 and 15%, streams may no longer fully support anadromous spawning functions. They may function inconsistently, providing suitable habitat under certain conditions, but do not provide consistent anadromous spawning habitat because processes are vulnerable to annual variations in natural plus additional anthropogenic stressors. I would recommend accepting changes in ecological services while managing to maintain present anadromous spawning habitat conditions that do allow for successful production in some years. Management options could include capping growth and continuing to develop sound management policies like the innovative designation of the Watershed Conservation District applied to Mattawoman Creek by Charles County (Charles County Government 2016). In watersheds where impervious cover exceeds 15%, anadromous spawning habitat potential is severely degraded and ineffective to support internal production. To date, there are no proven management strategies to restore internal production, however, stream restoration to reduce nutrients and sediments can enhance downstream estuarine areas that continue to support nursery functions for species that migrate from elsewhere. In these areas, I recommend focusing management on maintaining the downstream ecological functions that have remain in the face of changing habitat. For example, offshore habitats in suburban mesohaline tributaries experience frequent hypoxic events in the summer, where nearshore zones remain suitable for juvenile fishes (Uphoff et al. 2011). In these watersheds, I would recommend focusing management attention to downstream subestuary nearshore habitats.

I recommend permit reviewers and managers consider the RAD approach as an alternate framework that can be used to develop decisions trees based on the level of development (impervious cover) in a watershed. This could help streamline the review process to focus more intensive reviews on rural watersheds where we want to resist habitat changes that could impair production. The new data layers developed could be useful in developing a decision tree. I look forward to opportunities to collaborate to develop and refine tools to streamline management and assure we are committing resources to conserve productive habitats.

The next step includes packaging the spatial files into a single tool that can be made available online. We will work with our GIS staff to complete this. We also will continue to promote application of the RAD approach to encourage adoption of the framework and increase the likelihood of meeting management goals.

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Figure 1. Inland stations where anadromous fish stream spawning was indicated from surveys in Anne Arundel County, MD. Anadromous species include American Shad, Alewife, and Blueback Herring, White Perch and Yellow Perch.

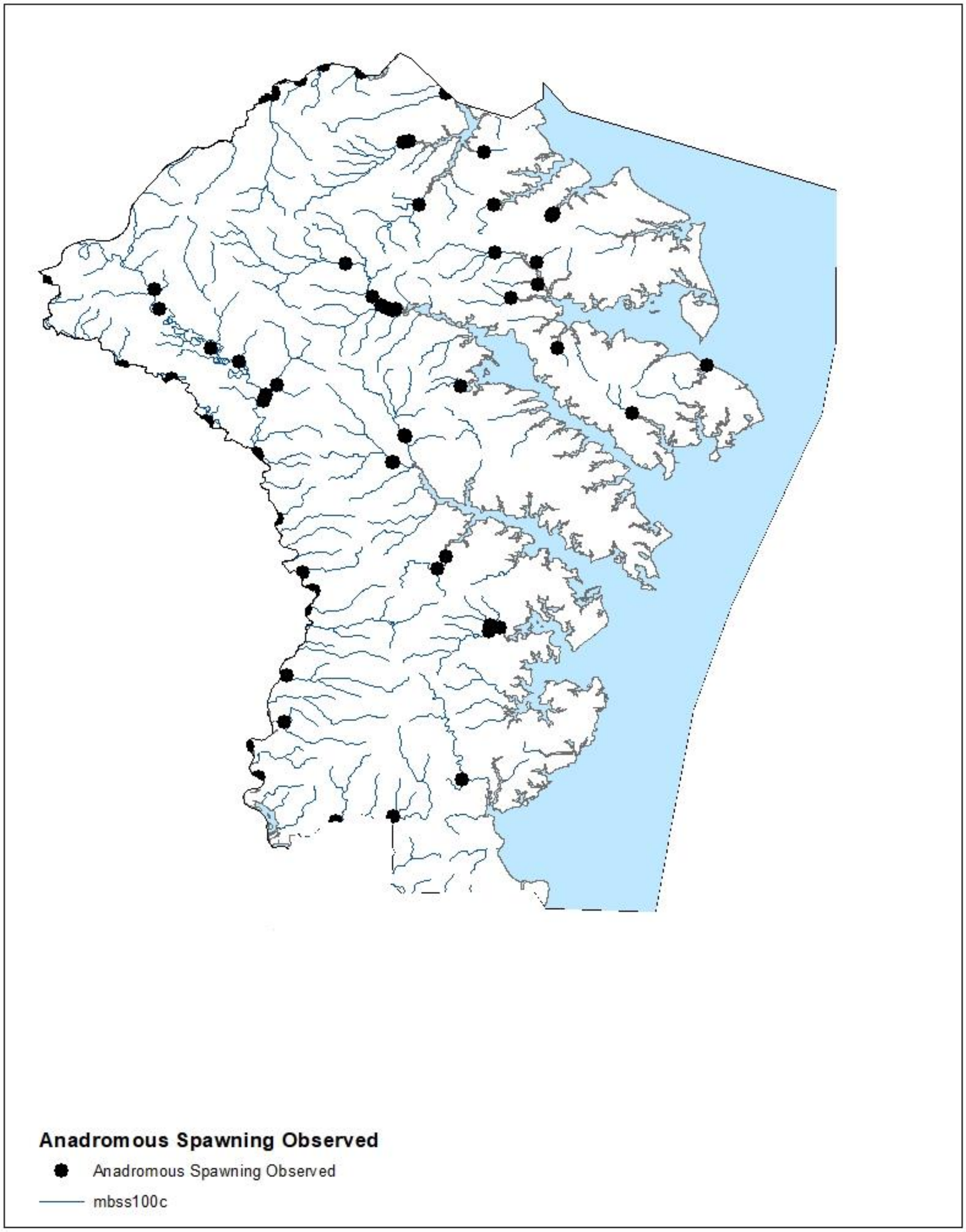


Figure 2. Inland stations where anadromous fish stream spawning was indicated from surveys in Anne Arundel County, MD, with spawning segments delineated.

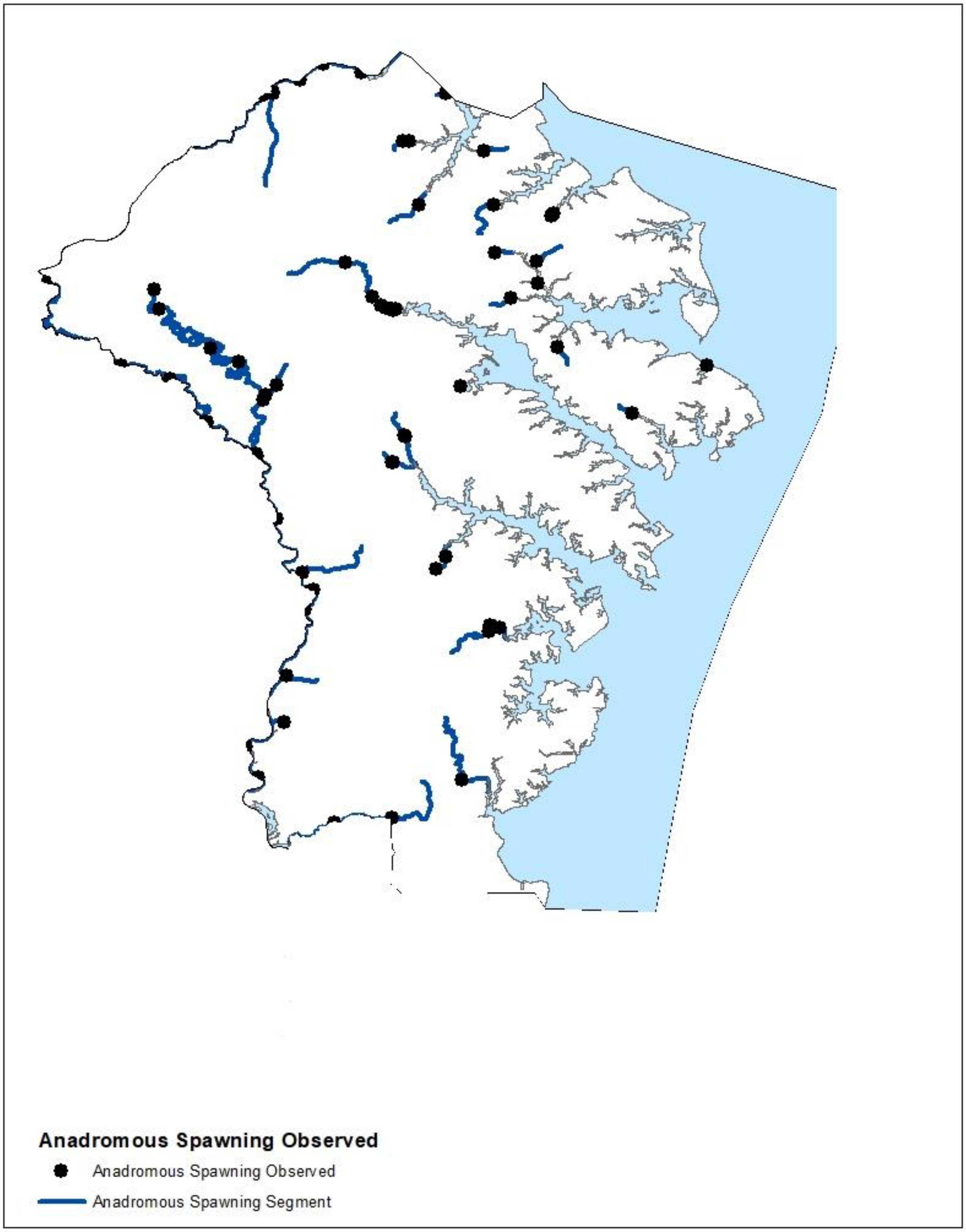


Figure 3. Inland stations where anadromous fish stream spawning was indicated from surveys in Anne Arundel County, MD, with anadromous spawning segments and unimpeded contributing anadromous spawning habitat delineated.

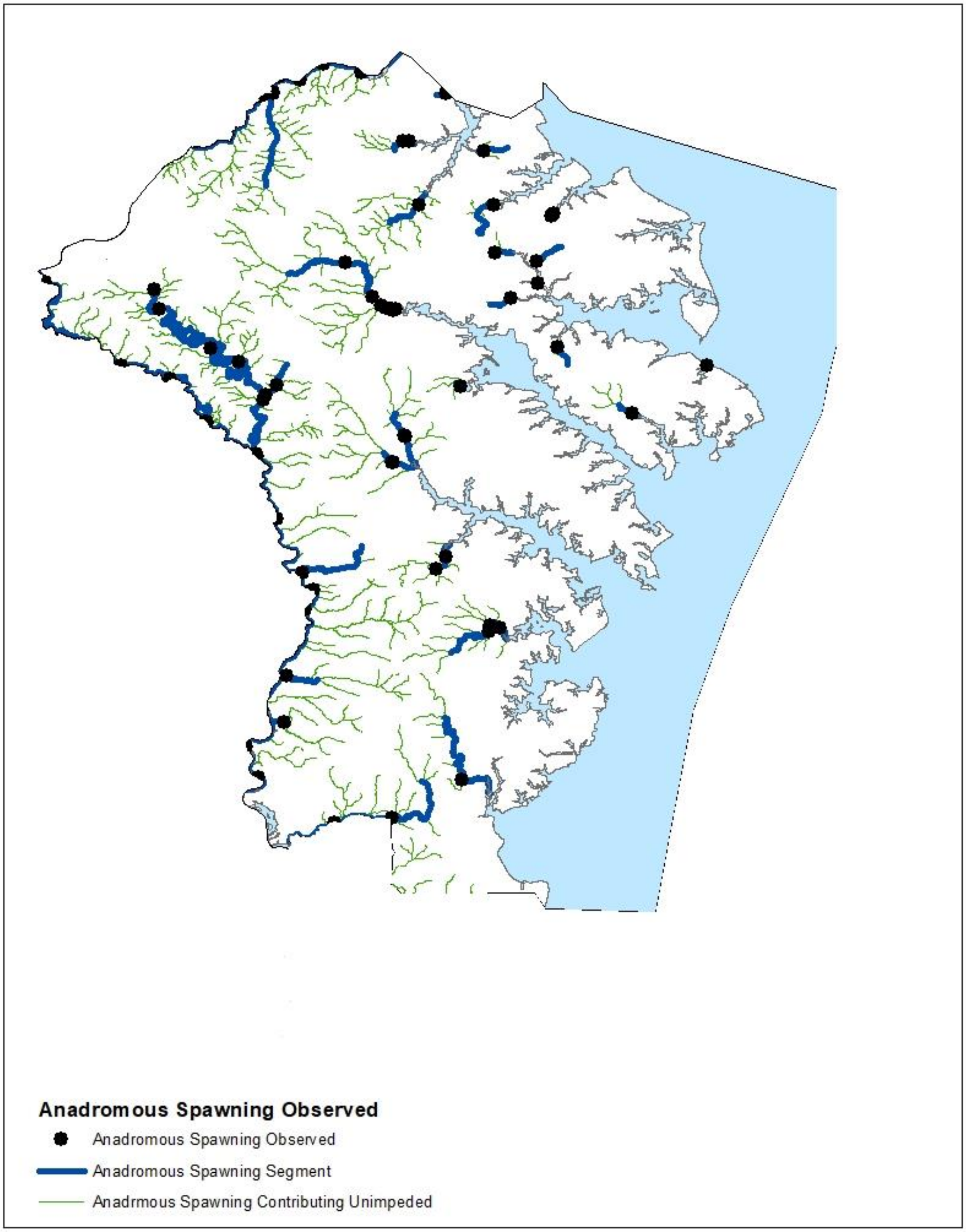


Figure 4. Inland stations where anadromous fish stream spawning was indicated from surveys in Anne Arundel County, MD, with anadromous spawning segments and unimpeded and impeded contributing anadromous spawning habitat delineated.

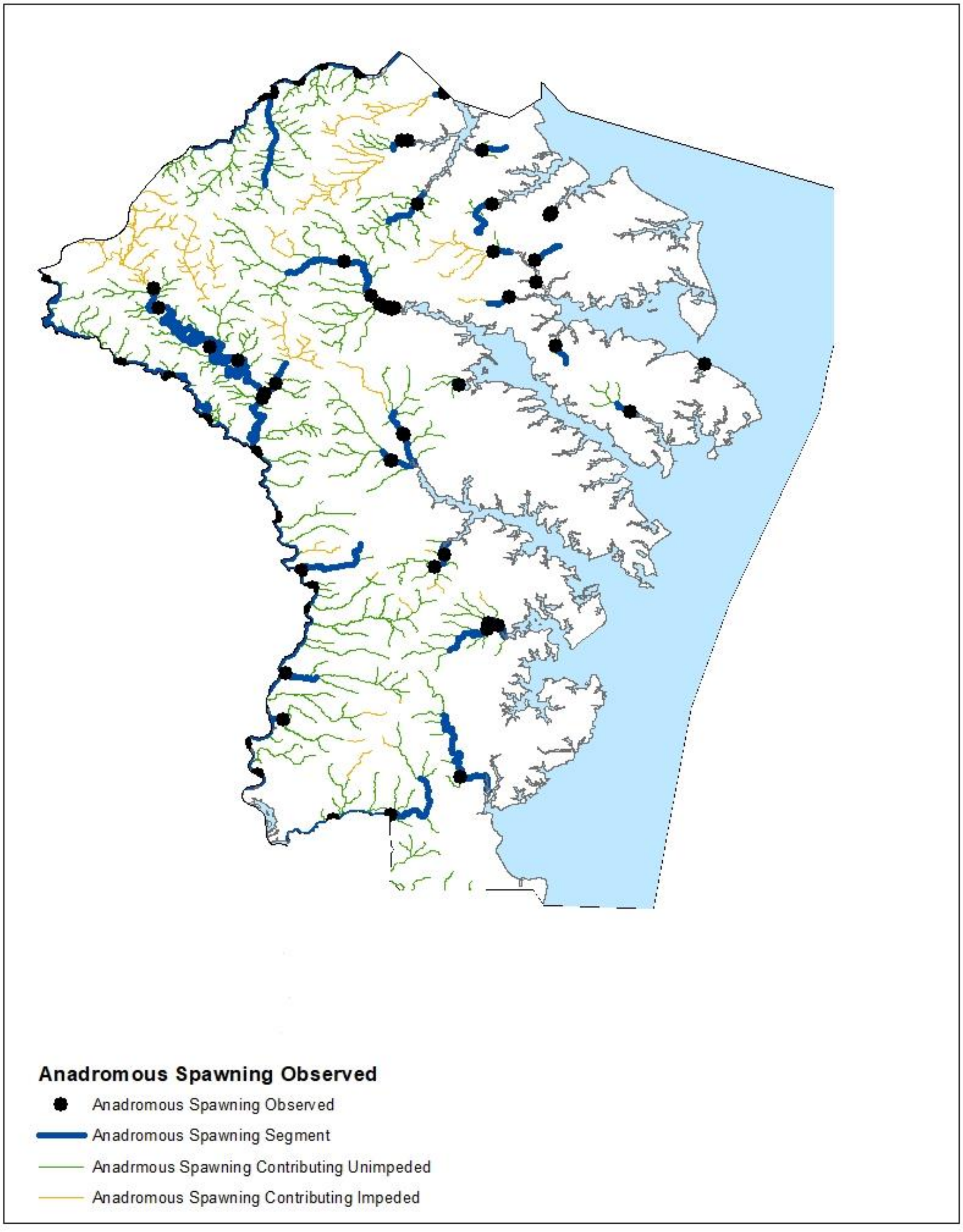
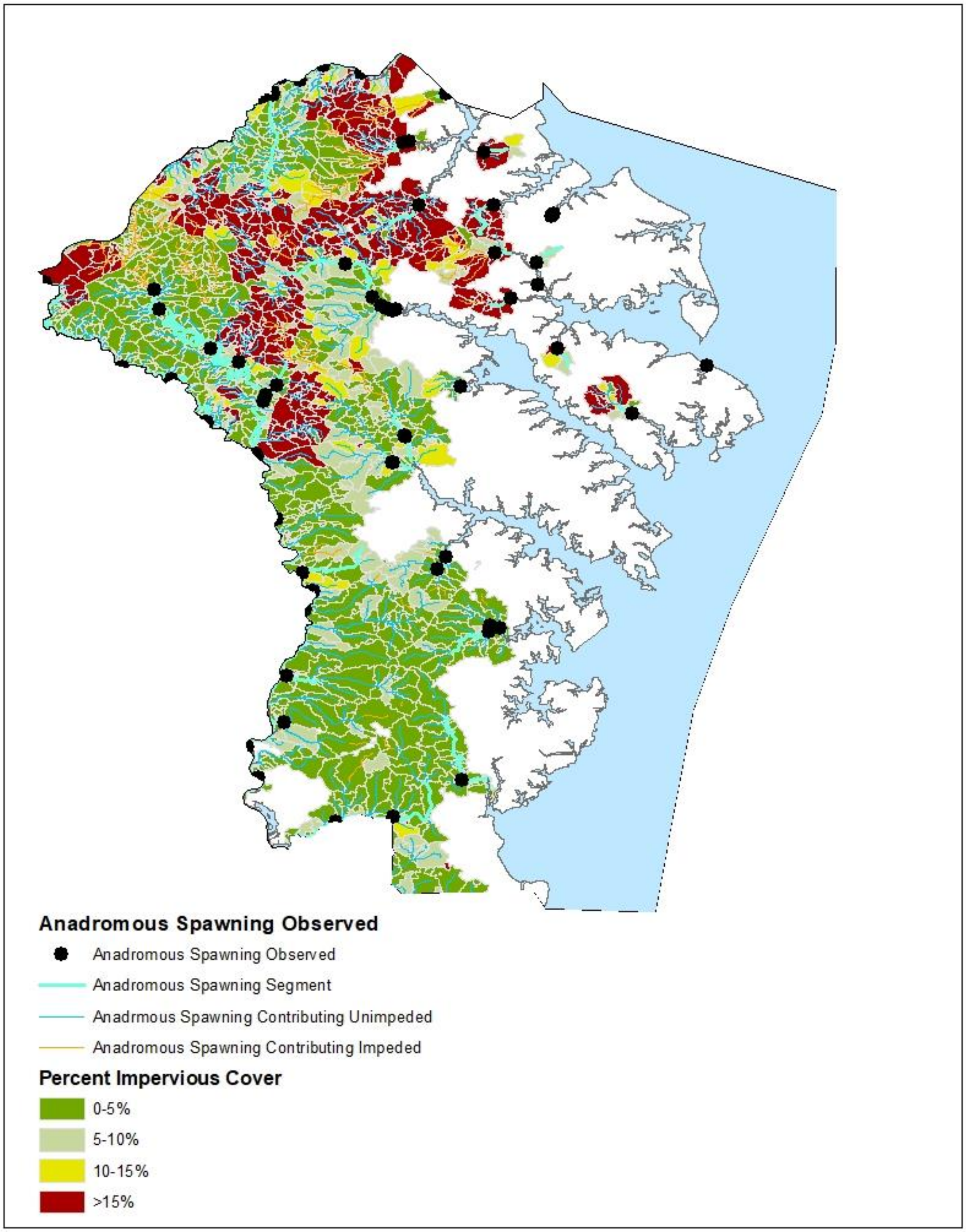


Figure 5. Inland stations where anadromous fish stream spawning was indicated from surveys in Anne Arundel County, MD, with anadromous spawning segments, unimpeded and impeded contributing anadromous spawning habitat delineated and catchments with impervious surface included.



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

Jim Uphoff, Alexis Park, and Carrie Hoover

Changes to Project 4 Activities due to Coronavirus

Sampling of Striped Bass condition and diets for Job 4 was not affected by the Pandemic.

Executive Summary

Indices of health (1998-2020), relative abundance (1983-2020), natural mortality (1986-2020), and forage relative abundance in surveys (1959-2020) and fall diets (1998-2000 and 2006-2020) provided metrics (indicators) to assess forage status and Striped Bass well-being in Maryland's portion of Chesapeake Bay. In addition to providing insight on forage status, these indicators had to be inexpensive and tractable for staff. The proportion of Striped Bass without body fat (P0), anchored our approach, providing a measure of condition and potential for starvation that was well-related to feeding. Proportion of Striped Bass in fall with empty guts (PE) provided trends in prey supply relative to predator demand based on relative abundance and diet sampling, respectively. The proportion of diet items by number and weight of prey per weight of Striped Bass (C) supplemented PE. Metrics based on examination of individual fish (P0, PE, and C) were split into two size classes (small, <457 mm TL and large, $\geq$ 457 mm TL) due to sampling considerations and recent divergence in trends in P0 between the size classes. The P0 and PE metrics had targets and thresholds and remaining metrics were considered supplemental. An index of survival (SR) that reflected natural mortality (M) was developed for small Striped Bass and trends could be compared with published estimates for large fish. Remaining metrics could not be split for size classes. A Striped Bass recreational catch per trip index (RI) provided an index of relative abundance. Forage-to-Striped Bass ratios were developed from relative abundance indices of major prey (FRs; focal prey species are Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab).

Condition of small Striped Bass was between the target and threshold level during 2020, while large fish were at the target. Small Striped Bass condition was consistently poor (breaching the threshold) during 1998-2007 and shifted to a mix afterward; there were 3 years where the P0 met the target, 4 that the threshold was breached, and 6 in between. Condition of large Striped Bass was at its threshold in 6 of 7 years during 1998-2004 and has improved to only slightly missing its target once since 2014.

Large Striped Bass have been mostly at target PE since 2014. A target was not readily suggested for PE of small fish, but PE was clearly below their threshold during 2008-2010, 2014, and 2016-2020. Estimates of PE for large and small Striped Bass have improved from threshold conditions prior to 2007. Atlantic Menhaden dominated small and large Striped Bass diets by weight during fall; C has been higher since 2013, more frequently ranking in the top half of

estimates. Bay Anchovy were dominant by number in small Striped Bass diets, but made up a low fraction of fall diet weight in all but the worst years. Small Blue Crabs were a minor component by weight as well, but were numerically abundant in some years. Spot, a major prey that had contributed to achievement of target P0 and PE for small fish in 2010, have been largely absent in fall diets of both size classes since 2015. Bay Anchovy were consistently present in fall diets of both size classes of Striped Bass during 2006-2014, but have fallen substantially as a percent of large fish diet since 2015 as Atlantic Menhaden became frequent. Bay anchovy represented a variable percentage of small fish diets during 2006-2015 and had a steadier, higher frequency afterwards. Diet changes since 2015 suggest the pelagic pathway is making a larger contribution to fall diets in recent years.

A rapid rise in Striped Bass abundance in upper Bay during the mid-1990s, followed by a dozen more years at high abundance after recovery was declared in 1995, coincided with declines in relative abundance of Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (i.e., major pelagic and benthic prey) to low levels. Changes in FRs largely reflected decreasing prey during 1983-1994 since RI was low. After 1995, prey indices stayed relatively low and RI increased; FR changes usually reflected fluctuations in RI. It appears that slightly higher (but not statistically significant) 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 (this study) and large (published estimates) Striped Bass decreased due to increased M in Chesapeake Bay since the late 1990s. Higher frequency of below time-series median SR after 1996 was concurrent with declines in conventional tag-based estimates of survival of large Striped Bass in Chesapeake Bay (based on time varying estimates of M). Natural mortality estimates for large fish based on acoustic tagging produced similar differences in mortality of coastal migrants (low M) and Chesapeake Bay residents (high M) as found with conventional tags. The fall in survival in the mid-to-late 1990s was consistent with a compensatory response to high Striped Bass abundance, low forage, and poor condition.

Introduction

The Chesapeake Bay stock of Striped Bass *Morone saxatilis* supports major commercial and recreational fisheries within Chesapeake Bay and along the Atlantic coast of the United States (Richards and Rago 1999; Maryland Sea Grant 2009). Striped Bass, fueled by a series of strong year-classes in Chesapeake Bay, were abundant in the 1960s and early 1970s, then declined as recruitment faltered and fishing mortality rates increased (Richards and Rago 1999). Moratoria were imposed in several Mid-Atlantic States in the mid-to-late 1980s and conservative regulations were put in place elsewhere (Uphoff 1997; Richards and Rago 1999). Recovery of Atlantic coast Striped Bass was declared in 1995 after rapid stock growth (Richards and Rago 1999; ASMFC 2021). Management since recovery has been based on much lower fishing mortality and much higher size limits than were in place into the early 1980s (Richards and Rago 1999; ASMFC 2021).

Concern emerged about the impact of high Striped Bass population size on its prey-base shortly after recovery (Hartman 2003; Hartman and Margraf 2003; Uphoff 2003; Savoy and Crecco 2004; Heimbuch 2008; Davis et al. 2012; Overton et al. 2015; Uphoff and Sharov 2018). Major declines in abundance of important prey (Bay Anchovy *Anchoa mitchilli*, Atlantic

Menhaden *Brevoortia tyrannus*, and Spot *Leiostomus xanthurus*) in Maryland's portion of Chesapeake Bay (hereafter upper Bay) coincided with recovery (Uphoff 2003; Overton et al. 2015). Maintaining a stable predator-prey base is a challenge for managing Striped Bass in lakes (Axon and Whitehurst 1985; Matthews et al. 1988; Cyterski and Ney 2005; Raborn et al. 2007; Sutton et al. 2013; Wilson et al. 2013).

A large contingent of Chesapeake Bay Striped Bass that do not participate in the Atlantic coast migration (mostly males along with some young, immature females; Setzler et al. 1980; Kohlenstein 1981; Dorazio et al. 1994; Secor and Piccoli 2007) constitute a year-round population of predators that provides Maryland's major saltwater recreational fishery and an important commercial fishery (Maryland Sea Grant 2009). Reports of Striped Bass in poor condition and with ulcerative lesions increased in Chesapeake Bay shortly after recovery; linkage of these phenomena and poor feeding success on Atlantic Menhaden and other prey was considered plausible (Overton et al. 2003; Uphoff 2003; Gauthier et al. 2008; Overton et al. 2015; Uphoff and Sharov 2018). Mycobacteriosis, a chronic wasting disease, became widespread in Chesapeake Bay in the late 1990s and was concurrent with lesions and poor condition (Overton et al. 2003; Jiang et al. 2007; Gauthier et al. 2008; Jacobs et al. 2009b). Challenge experiments with Striped Bass linked nutrition with progression and severity of the disease, and reduced survival (Jacobs et al. 2009a). Tagging models indicated that annual instantaneous natural mortality rates (M) of large 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; ASMFC 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 appears linked to ages 1+ Atlantic Menhaden (Buccheister et al. 2017; Uphoff and Sharov 2018; ASMFC 2020a; Chagaris et al. 2020)

Maryland's fisheries managers and stakeholders want to know whether there is enough forage to support Striped Bass in upper Bay. Formal assessments of abundance and biomass of Striped Bass and most forage species in upper Bay are lacking due to cost and difficulty in mathematically separating migration from mortality. The Atlantic States Marine Fisheries Commission (ASMFC) has adopted ecological (forage) reference points for Atlantic Menhaden's forage role along the Atlantic coast and Striped Bass is a predator of concern because of its sensitivity to Atlantic Menhaden population size (ASMFC 2020a; Chagaris et al. 2020; Drew et al. 2021; Anstead et al. 2021). In 2014, a forage fish outcome was included in the Chesapeake Bay Agreement (Chesapeake Bay Program): "By 2016, develop a strategy for assessing the forage fish base available as food for predatory species in the Chesapeake Bay." Project 4 is a direct response by MD DNR to this outcome.

Indicators based on monitoring, such as forage indices, prey-predator ratios, condition indices, and prey abundance in diet samples have been suggested as a basis for forage assessment for Striped Bass in Chesapeake Bay (Maryland Sea Grant 2009; SEDAR 2015) and formed the foundation of our approach. Indicators are widely used for environmental reporting, research, and management support (Rice 2003; Jennings 2005; Dettmers et al. 2012; Fogarty 2014).

The approach used here is based on a suite of indicators (metrics) that are inexpensively and easily developed from existing MD DNR sampling programs. This report provides indicators through 2020. In addition to providing information for judging whether the forage base is adequate to support Striped Bass in Maryland's portion of Chesapeake Bay, two additional objectives were low cost and tractability for available staff.

During 2014-2019, we developed an integrated index of forage or IF that was comprised of five metrics covering all sizes of Striped Bass within a defined size range (286-864 mm TL or 11.3-34.0 inches). Forage status was judged by whether target (indicating good forage conditions) or threshold (indicating poor forage conditions) reference points were met for each metric. Time periods where body fat indicators were at target or threshold levels provided a time frame for developing targets and thresholds for other metrics.

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

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

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

While upper Bay Striped Bass feed on a wide range of prey, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab *Callinectes sapidus* have consistently accounted for most annual diet biomass in Chesapeake Bay studies (Hartman and Brandt 1995c; Griffin and Margraf 2003; Walter et al. 2003; Overton et al. 2009; Overton et al. 2015; Buccheister and Houde 2016). We selected these species as focal prey (major prey) for forage indices. Forage ratios of species-specific indices of major prey relative abundance from fishery-independent surveys to an indicator of resident Striped Bass relative abundance were examined for each focal prey as an indicator of potential attach success. These forage ratios could not be split into size categories. Forage species indices alone would not consider the possibility of predator interference or the vulnerability exchange process of foraging arena theory (Ginzburg and Akçakaya 1992; Yodzis 1994; Ulltang 1996; Uphoff 2003; Walters and Martell 2004; Walters et al. 2016).

Benthic invertebrate indices (invertebrates other than Blue Crabs) are included in this report even though benthic invertebrates have not contributed much to fall diets. Uphoff et al. (2018) found that P0 the previous summer and the previous fall could influence P0; condition of Striped Bass in summer may be influenced by benthic invertebrates since they can be a significant component of their spring diet (Overton et al. 2015). The utility of estimates of biomass of invertebrates comprising a benthic IBI in Maryland's portion of the Bay used for water quality monitoring was explored in Uphoff (2018). A complementary index for hard (oyster) bottom was developed by M. McGinty (Uphoff et al. 2018).

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

Correlation and regression analyses and-or determination of how often target or threshold P0 were or were not met for a given metric were used to provide insight on potential influences on nutritional status. Targets and limits based on historical performance are desirable because they are based on experience and easily understood (Hilborn and Stokes 2010). We have used correlation and regression analyses in the past to explore to what degree indicators of upper Bay Striped Bass abundance, forage abundance, consumption, and relative survival estimates were linked to the body fat condition indicator (Uphoff et al. 2020). Statistical analyses can provide insight into important processes related to predation (Whipple et al. 2000), but relationships may change over time if they do not reflect underlying ecological processes or the processes themselves shift over time (Skern-Mauritzen et al. 2016). Some metrics were statistically linked to one another, but making strong connections can be difficult due to sampling issues, nonlinear, asymptotic relationships among variables, lagged responses, potential insensitivity of some indices, behavioral changes that increase feeding efficiency, episodes of good foraging conditions outside of those monitored in fall, larger major prey relative to size of Striped Bass and combinations of the above (Uphoff et al. 2020). Many of these issues were discussed in Uphoff et al. (2016; 2017; 2018) and the reader is referred to them.

We have corrected past errors that were discovered in two metrics: PE and SR. Errors in PE arose from inconsistency in the period used for estimation and inclusion of Striped Bass that were smaller than the common minimum size. Estimates in SR resulted from inadvertent use of the lower confidence interval of the juvenile index rather than the mean in some estimates.

Methods

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

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

Conditions of the collector's permit issued to CBEF allowed for samples of up to 15 Striped Bass less than 457 mm total length (or TL; small Striped Bass or fish; the minimum length limit for Striped Bass was 457 mm or 18 inches when the permit was issued) and 15 fish 457 mm TL or larger (large Striped Bass or fish) per trip during 2006-2014. The small and large designations replace sublegal and legal sized designations used in previous reports; this change was made to prevent confusion that may arise due to length limit changes (the length limit was 457 mm TL during 1998-2014; it was raised to 508 mm TL in 2015, lowered to 483 mm TL in 2018 and has remained there through 2020).

Striped Bass diet collections by CBEF were made in a portion of upper Bay bounded by the William Preston Lane Bridge to the north, the mouth of Patuxent River to the south, and into the lower Choptank River (Figure 1). Most active trips by CBEF occurred in Choptank River, but some occurred in the mainstem Chesapeake Bay. Active trips were our source of small sized fish, but large sized fish were caught as well. Striped Bass kept as samples during active trips were placed in a cooler and either processed upon return to shore or held on ice for processing the next day. Collections of large sized Striped Bass were supplemented by sampling charter boat hook-and-line catches at a fish cleaning business. These fish were predominately from the mainstem Chesapeake Bay; 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.

Since 2014, Striped Bass collected for health samples by Fish and Wildlife Health Program (FWHP) have been processed by Fish Habitat and Ecosystem Program (FHEP) biologists for diet information. Collections by FWHP were not constrained by collector's permit conditions like CBEF collections. Fish have been collected by hook-and-line from varying locations during fall since 1998 between Baltimore, Maryland (northern boundary) and the Maryland-Virginia state line (southern boundary; Figure 1). Sampling by FWHP was designed to fill size class categories corresponding to age-classes in an age-length key to assess Striped Bass health. Some trips occurred where fish in filled out length classes were discarded (typically small fish). Samples were usually obtained by fishing on a charter boat using the techniques considered most effective by the captain (bait or artificial lures). Bait was excluded from diet data.

Condition indices were estimated from an existing FWHP Striped Bass health survey that began in 1998. Nutritional status (condition) for upper Bay Striped Bass was estimated as the proportion of fish without visible body fat during October-November in FWHP samples (P0;

Jacobs et al. 2013). Estimates of P0 were made for the two size classes of Striped Bass. Estimates of P0 for 1998–2013 were provided by FWHP and remaining years were estimated from FWHP data by FHEP. Standard deviations and confidence intervals (90%) of P0 were estimated using the normal distribution approximation of the binomial distribution (Ott 1977).

As Striped Bass experience starvation, lipids are replaced by water, conserving weight loss and hampering the interpretation of weight-at-length condition indices (Jacobs et al. 2013). Jacobs et al. (2013) presented a condition target based on body moisture (25% or less of fish with starved status) as a surrogate for lipid content estimated from proximate composition of well-fed Striped Bass. This target was derived from fall 1990 field collections by Karahadian et al. (1995) - the only field samples available from favorable feeding conditions (high FRs). A target for visible body fat was not presented in Jacobs et al. (2013) because the index was not applied in the 1990 collection. However, mean tissue lipid of Striped Bass without visible body fat was reported to be identical to that estimated from percent moisture in the remainder of the data set, meaning that P0 related strongly to the proportion exceeding the moisture criteria (Jacobs et al. 2013). A level of P0 of 0.30 or less was used to judge whether Striped Bass were in good condition. Variation of tissue lipids estimated from body fat indices was greater than for moisture and the higher P0 target accounted for this additional variation plus a buffer for misjudging status (J. Jacobs, NOAA, personal communication). Jacobs et al. (2013) stressed that comparisons of Striped Bass body fat to a nutritional target or threshold in Chesapeake Bay should be based on October-November data since they were developed from samples during that time span. Uphoff et al. (2014) estimated the P0 threshold as 0.68 (average of the lower 95% CI of high P0 estimates during 1998-2004, a period of consistently poor condition). Other indicators of condition were described in Jacobs et al. (2013), but P0 was chosen because it could be applied to data collected by CBEF; P0 estimates from CBEF collections were similar to those estimated for FWHP collections for years in common (Uphoff et al. 2018).

Total length of each Striped Bass was recorded and whole fish were weighed on a calibrated scale for CBEF and FWHP samples. Striped Bass length-weight regressions based on that year’s October-November samples were used to estimate missing weights from filleted fish in CBEF collections.

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

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

Diets were analyzed separately for small and large Striped Bass for both CBEF and FWHP collections. These categories accounted for ontogenic changes in Striped Bass diet, but also reflected unbalanced sample availability to CBEF (small fish could only be collected by fishing for them directly, while large sized fish were supplemented by cleaning station samples). The lower limit of fish analyzed in the small category, 286 mm, was the minimum length in common among years during 2006-2013. An upper limit of 864 mm avoided inclusion of very large, migratory Striped Bass that reentered upper Bay in late fall.

We confined analysis of food items to those considered recently consumed in an attempt to keep odds of detection as even as possible. Items with “flesh”, including whole or partial fish and invertebrates, and intact crab carapaces were considered recently consumed. Hard, indigestible parts such as gizzards, mollusk shells, and backbones without flesh were excluded. Partially intact items with flesh were identified to lowest taxonomic group and assigned the mean weight estimated for intact items in the same group. Bait was excluded.

Percentage of food represented by an item in numbers in each year since 2006 was estimated for each Striped Bass size class based on fish with stomach contents (Pope et al. 2001). Estimates included both counts of whole items and presence of partially intact prey (portions that were intact enough to identify a prey, but not intact enough to measure and weigh as individuals). The latter could include multiple individuals, so percent by number was negatively biased to some extent.

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

Proportion of Striped Bass with empty stomachs (PE) was an indicator of total prey availability (Hyslop 1980). Standard deviations and 90% CI’s of PE were estimated using the normal distribution approximation of the binomial distribution (Ott 1977). Estimates of PE from Overton et al. (2009) were available to estimate threshold conditions during 1998-2000 (Uphoff et al. 2017). In addition, this indicator could be derived from published diet information from the 1930s (Hollis 1952) and the 1950s (Griffin and Margraf 2003) for comparisons with 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 that we converted to PE. Proportion of empty stomachs was 0.54 for fish between 301 and 500 mm TL (approximating our small class) and 0.57 for Striped Bass between 501 and 700 mm (approximating our large class; Overton et al. 2009). These 1998-2000 estimates were comparable to our highest estimates of PE and reflected high P0 and a nadir in major prey indices (except the Bay Anchovy trawl index) during that period. Target PE was estimated for small or large fish from periods when PE corresponded with target estimates of P0.

To aid interpretation of PE, we examined the influence of prey-predator length ratios (PPLR) of the two size classes of Striped Bass. For this analysis we determined PPLRs for the two largest major prey in fall diets: Spot and Atlantic Menhaden. This analysis was based on ratios for whole prey and was split for small and large Striped Bass. We determined median PPLR for each year and size class of Striped Bass. Optimum PPLR of Striped Bass was 0.21 (described in Overton et al. 2009) and we compared median PPLR of large major prey for each size class to this estimate of optimum PPLR.

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 2020). 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 2021a; 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 2021b). Spot and Blue Crabs were classified as benthic forage, while Atlantic Menhaden and Bay Anchovy were pelagic (Hartman and Brandt 1995c; Overton et al. 2009). Each forage index was divided by its mean for years in common among all surveys (1989-2019) to place their time-series on the same scale for graphical comparisons of trends among surveys.

A 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 component for Maryland tidal waters as our index (Figure 3-37 in Llansó and Zaveta 2019). The BIBI has been employed to monitor water quality since 1995 and the latest indices are for 2018. The benthic biomass component consists of 7 polychaetes, 10 mollusks, 1 isopod, 2 amphipods, and 2 ribbon worms (see Table 2-5 in Llansó and Zaveta 2019). Uphoff et al. (2018) explored the relationship of this benthic biomass index on resident Striped Bass condition. This index was not incorporated into a forage ratio (described below for major prey).

A fishery-independent index of relative abundance of upper Bay resident Striped Bass was not available and we used estimates of Maryland Striped Bass catch-per-private boat trip (released and harvested fish; RI) from the National Marine Fisheries Service's (NMFS) Marine Recreational Information Program (MRIP; NMFS Fisheries Statistics Division 2021) database as an index. Online estimates of catch and effort are available for 1981 and onwards. Similar recreational catch per trip indices have been used as abundance indicators in Atlantic coast stock assessments of major pelagic finfish predators: Striped Bass, Bluefish *Pomatomus saltatrix*, and Weakfish *Cynoscion regalis* (NEFSC 2019; NEFSC 2012; ASMFC 2013). Our RI estimates were based on revised MRIP estimates.

The RI was estimated as a catch-effort ratio for private and rental boat anglers in Maryland in the MRIP inland fishing area (inshore saltwater and brackish water bodies such as bays, estuaries, sounds, etc, excluding inland freshwater areas; NMFS Fisheries Statistics Division 2021). The RI equaled September-October recreational private and rental boat catch of Striped Bass divided by estimates of trips for all species for the private and rental boat sector. Recreational survey estimates are made in two-month waves and September-October constituted the fifth wave (NMFS Fisheries Statistics Division 2021). This wave was chosen because portions or the whole wave were continuously open for harvest of Striped Bass following the 1985-1990 moratorium, making it less impacted by regulatory measures than other waves that opened later. Recreational fishing by boat occurs over the entire portion of the upper Bay and this index would be as close to a global survey as could be obtained. Migratory fish were unlikely to have been present during this wave. The RI was related to juvenile indices 2-5 years earlier (determined by multiple regression) and to Atlantic coast abundance estimates (Uphoff et al. 2014). We compared the RI to the abundance estimates for 2-5 year-old Striped Bass estimated by the statistical catch at age model used in the recent stock assessment (NEFSC 2019).

We used forage indices divided by RI (forage index-to-Striped Bass index ratios, i.e., forage ratio or FR) as indicators of forage supply of major prey relative to Striped Bass demand (index of potential attack success). Ratios were standardized by dividing each year's FR estimate by the mean of FR during 1989 to the present, a time-period in common among all data; FR estimates were available for every year since 1983 except 1987 (RI was not estimated).

We estimated relative survival as relative abundance at age-3 divided by age-0 relative abundance three years prior (juvenile index in year - 3). Striped Bass spawning season experimental gill net surveys have been conducted since 1985 in Potomac River and the Head-of-Bay (~39% and 47%, respectively, of Maryland's total spawning area; Hollis 1967) that provide age-specific indices of relative abundance (Versak 2021). Table 8 in Versak (2021) provided mean values of for annual, pooled, weighted, age-specific CPUEs since 1985 for the Maryland Chesapeake Bay Striped Bass spawning stock and we used the age-3 index (CPUE3) as the basis for an adjusted index. Typically, the most recent year's CPUE3 was unavailable on this table and was provided by B. Versak, (MD DNR, personal communication). Even though males and females were included, females were extremely rare on the spawning grounds at age 3; nearly all of these fish would be resident males (Versak 2021). This CPUE3 index had the advantage of combining both spawning areas, a coefficient of variation (CV) estimate was provided, and it was regularly updated in an annual report.

Gill net indices used in the numerator of SR in Uphoff et al. (2015) were suggesting either no change in abundance since 1985 or a decrease; this was implausible when viewed against stock assessment estimates, juvenile indices, and harvest trends. Uphoff et al. (2016; 2017; 2018) determined that gill net survey catchability (q; estimated by dividing the catch per effort index by the stock assessment abundance estimate; rearrangement of equation 6.1 in Ricker 1975) of 3-year-old male Striped Bass changed as an inverse nonlinear function of population size.

We created a "hybrid" gill net time-series that used indices adjusted for rapid changes in catchability during 1985-1995 (stock went from severely depleted to recovered) and the unaltered estimates afterwards. First, we estimated a catchability coefficient (q) for each year during 1985-2017 for age 3 Striped Bass by dividing CPUE3 by the estimated abundance at age 3 from NEFSC (2019); 2017 was the last year in the assessment. We averaged q estimates for 1985-1995 (mean q) and used them to form a relative q as (annual q / mean q). An adjusted CPUE for each year from 1985-1995 was estimated as CPUE3 / relative q. After 1995, reported CPUEs were used (Uphoff et al. 2019).

Relative survival (SR) in year t was estimated as the hybrid gill net index for age-3 in year t (HI_t) divided by its respective juvenile index three years earlier (JI_{t-3});

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

The frequency of SR estimates above, below, and near the full time-series median was determined and trends in SR were compared to RI to examine whether density-dependent mortality was suggested.

Confidence intervals (90%) were developed for ratio-based metrics using an Excel add-in, @Risk, to simulate distributions reported for numerators and denominators. Each annual set of estimates was simulated 5,000-times. Ratio metrics simulated were RI, SR, and FR for Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab. Annual means and standard errors reported for these indices were used to generate simulations. Numerators and denominators of the RI, HI, and the Blue Crab index were considered normally distributed since their distributions were characterized by means and SE's in their respective sources (NMFS Fisheries Statistics Division 2021; Versak 2021; MD DNR 2020b). Remaining indices for Atlantic Menhaden (seine), Bay Anchovy (seine and trawl), and Spot (seine and trawl) and the JI for Striped Bass were based on geometric means (Durell and Weedon 2020). Geometric mean indices were back-transformed into the mean of $\log_e$-transformed catches (+1) and its standard error was derived from the 95% CI. The $\log_e$-transformation normalized the original catch data. Geometric means were recreated by exponentiating the simulated mean of $\log_e$-transformed catches (+1).

@Risk used Latin Hypercube sampling to recreate input distributions by stratifying their cumulative curves into equal intervals and then sampled each interval without replacement (Palisade Corporation 2016). Sampling was forced to represent values in each interval and recreated the original input distribution. Latin Hypercube sampling uses fewer iterations compared to random sampling employed by Monte Carlo simulations and is more effective when low probability outcomes are present (Palisade Corporation 2016).

Analyses – Statistical analyses provided evidence that forage and Striped Bass abundance and well-being were or were not inter-related for each size class. We were particularly interested in how our primary index of condition, P0, reflected other indices of feeding success (PE and C), forage size, forage relative abundance, intraspecific competition (RI), and water temperature. Water temperature was a new variable and we were concerned that higher temperatures might influence fat accumulation through delayed timing of aggressive feeding (temperature may be a trigger) and its effect on fat accumulation through metabolism. Median above pycnocline water temperature at eight Maryland mesohaline mainstem Bay water quality monitoring stations (T. Parham, Resource Assessment Service, MD DNR, personal communication) for October (1998-2020) or November (1998-2019) were used as independent variables.

Correlation and regression were the primary means of analyzing data. In some cases, regression and correlation analyses indicated a general linear association or relationship was not strong enough to be useful. In these cases, a probabilistic approach was used. The number of data points of a particular variable were determined that did or did not fall below the P0 target or above the P0 threshold to determine those odds. Comparisons with P0 fell into two time-series categories. Analyses that were not based on feeding metrics (comparisons with forage indices, water temperature, or RI) encompassed the full P0 time-series, 1998-2020. The time span for comparisons of feeding metrics (PE and C) with P0 was 2006-2020.

For all analyses, scatter plots were examined for the need for data transformations and to identify candidate models. Residuals of regressions were inspected for outliers, trends, and non-normality. If a large outlier was identified, the data from that year was removed and the analysis was rerun. Levels of significance of correlations were not adjusted for multiple comparisons as there is no formal consensus as to when these adjustment procedures should be applied (Nakagawa 2004). A general description of equations used follows, while more specific applications were described as needed.

Linear regressions described continuous change in variable Y as X changed:

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

where m is the slope and b is the Y-intercept (Freund and Littell 2006). When linear regression analyses exhibited serial patterning of residuals, a time category variable (T) that split the time-series into two time periods (T indicating time categories 0 and 1) was used to remove time-series bias (Rose et al. 1986):

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

Where m is the slope, n is a coefficient for the time-series, and b is the intercept.

Potential dome-shaped relationships were examined with quadratic models (Freund and Littell 2006):

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

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

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

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

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

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

where a is the growth rate of Y with X, b is maximum Y, and c is Y at X = 0 (Prager et al. 1989).

The Weibull function is a sigmoid curve that provides a depiction of asymmetric ecological relationships (Pielou 1981). A Weibull curve described the increase in Y as an asymmetric, ascending, asymptotic function of X:

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

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

Level of significance was reported, but potential management and biological significance took precedence over significance at $P < 0.05$ (Anderson et al. 2000; Smith 2020). We classified correlations as strong, based on $r \geq 0.80$; weak correlations were indicated by $r < 0.50$; and moderate correlations fell in between. Relationships indicated by regressions were considered strong at $r^2 \geq 0.64$; weak relationships were indicated by $r^2 \leq 0.25$; and moderate relationships fell in between. Moderate to strong correlations and relationships were considered biologically significant and of interest to management. Confidence intervals (95% CIs were standard output) of the model parameters for each indicator species were estimated to examine whether parameters were different from 0 (Freund and Littell 2006). If parameter estimates were often not different from 0, rejection of the model was considered.

Results

Sample Size Summary - During 1998-2020, 1,988 small and 2,737 large Striped Bass were sampled during October-November (Table 2). Annual sample sizes for small fish in October-November ranged from 24 to 271 with a median of 118. Annual sample sizes for large fish ranged from 49 to 327 with a median of 205. Fewer dates were sampled within similar time spans after the FWHP became the platform for sampling in 2014 because numbers collected per trip were not confined by the terms of the CBEF collector's permit (6-12 per trips in fall by FWHP during 2014-2019 versus 11-22 trips by CBEF during 2006-2013). Starting dates for surveys analyzed were similar between those conducted by CBEF and FWHP (October 1-9), but samples taken on September 24, 2015, were included in that year's analysis because the earliest date sampled in October would have been October 21, 2015. End dates tended to be earlier in November for FWHP surveys, reflecting when size categories were filled out (Table 2).

Small Striped Bass Condition, feeding success, and diet composition indices - Condition of small Striped Bass has transitioned from continuously poor during 1998-2007 to a mix of at or near target P0 interspersed with a year or two of poor P0 afterward (Figure 2). Small Striped Bass were at the target level of condition ($P0 \leq 0.30$) during 2008, 2015, and 2017. Small fish in the upper Bay during fall were usually in poorest condition ($P0 \geq 0.60$) during 1998-2007, 2011-2012, 2016, and 2019 and we adopted P0 = 0.60 as this size group's threshold. Estimates of P0 (0.36-0.46) were between the target and threshold during 2009-2010, 2013-2014, 2018, and 2020. The 90% confidence intervals of P0 allowed for separation of years at or near the threshold from remaining estimates (Figure 2). Correlation analysis suggested some potential for October and November above pycnocline water temperatures (Table 3) to positively influence P0 of small fish (October 1998-2020 median temperature, $r = 0.43$, $P = 0.040$; November 1998-2019 median temperature, $r = 0.37$, $P = 0.093$), i.e., there was a weak tendency for P0 in fall to be poor as median temperatures increased.

Estimates of PE (proportion of empty stomachs) of small Striped Bass during fall, 2006-2020, ranged between 0.10 and 0.57 (Figure 3). Estimates of PE during 2006-2007, 2012, and 2015 could not be clearly separated from the threshold based on 90% CI overlap; PE during 1998-2000 based on Overton et al. (2009) was the threshold for small fish (PE = 0.54; Uphoff et al. 2016). Lowest estimates of PE for small fish (2009-2011, 2014, 2017, and 2019) could be separated from remaining higher estimates (except 2008) based on 90% confidence interval overlap. Estimates of PE during 2008-2011, 2014, and 2016-2020 were clearly lower than the 90% CI's of years that breached the threshold. Estimated PE in 2020 (0.30) was below the threshold and ranked 8th out of 15 years (Figure 3). Neither the correlation of P0 with PE (r = 0.05, P = 0.85) nor the bivariate plot (not shown) offered an indication of a target level of PE for small fish.

In combination and by number, Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab accounted for 96.0% of diet items encountered in small Striped Bass collected from upper Bay during fall, 2006-2020 (Figure 4). Bay Anchovy accounted for the highest percentage by number when all years were combined (63.5%, annual range = 19.1-87.9%); Atlantic Menhaden, 14.4% (annual range = 0-48.8%); Spot 5.6% (annual range = 0-70.7%); Blue Crab, 12.5% (annual range = 0.8-34.6%); and other items accounted for 4.0% (annual range = 0-24.5%; Figure 4). The vast majority of major prey in small Striped Bass diet samples during fall were young-of-year (Uphoff et al. 2016).

By weight, small Striped Bass diets in fall 2006-2020 (combined) were comprised of Atlantic Menhaden (69.8%), Bay Anchovy (14.8%), Spot (9.3%), Blue Crab (2.0%) and other items (4.6%; Figure 5). Estimates of C (total grams of prey consumed per gram of Striped Bass) for small Striped Bass varied as much as 8.7-times during 2006-2020. 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 2006-2020 median) either Spot (2010) or Atlantic Menhaden (2013-2014) dominated diet mass. The 2020 estimate of C of small fish was sixth highest of the 15 year time-series (Figure 5). The correlation of P0 and C was moderate (r = -0.58, P = 0.028).

Median PPLRs of large prey of small Striped Bass (Spot and Atlantic Menhaden combined) were 0.20-0.38 during 2006-2020 (Figure 6). Median PPLRs for small fish were particularly high (0.34-0.38) during 2012 and 2015-2020. They were close to the optimum described by Overton et al. (2009; 0.21) in 2010 when Spot constituted a large fraction of their diet. High estimates of C (defined previously) coincided with three of the four lowest PPLRs. The correlation of small Striped Bass P0 and PPLR of large prey was weak (r = 0.31, P = 0.28).

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. Large Striped Bass were at the target level of condition (P0 $\leq$ 0.30) during 2008-2010, 2014-2015, and 2017-2020 (Figure 7). Large fish in the upper Bay during fall were usually in poorest condition (P0 $\geq$ 0.70) during 1998-2004 (except 2002) and we adopted P0 = 0.70 as this size group's threshold. The 90% confidence intervals of P0 allowed for separation of years at the target from remaining estimates and estimates at the threshold from those at the target. Five of six estimates were above the threshold during 1998-2001 and 2004 could be separated from most (7 of 8) P0 estimates that fell between the target and threshold (Figure 7). Correlations of October or November above pycnocline water temperatures (Table 3) were weak

(October 1998-2020 median temperature, r = 0.17, P = 0.44; November 1998-2019 median temperature, r = 0.39, P = 0.10).

Estimates of PE of large Striped Bass during fall were at the threshold level in 2006, 2012, and 2017 based on 90% CI overlap (Figure 8). Overton et al. (2009) provided an estimate of the percent of Striped Bass in their large size class (501-700 mm, TL) with food during 1998-2000 (within the period of threshold P0) and we used this estimate (0.58) as a threshold PE for large sized fish. There was a modest association of PE and P0 (r = 0.52, P = 0.047); review of the plot of these variables (not shown) indicated that P0 at the target level was more likely when PE was 0.34 or less (7 of 9 points) than above it (1 of 4). The PE target for large fish was set at 0.34 and was met during 2014-2015 and 2018-2020 (Figure 8). Estimates of PE for large and small Striped Bass were modestly correlated (r = 0.56, P = 0.029).

Major prey accounted for 93.0% of diet items, by number, encountered in large Striped Bass diet samples during fall 2006-2020 (Figure 9). Atlantic Menhaden accounted for 45.1% when all years were combined (annual range =12.4-97.0%); Bay Anchovy, 15.9% (annual range = 0-32.5%); Spot, 8.3% (annual range = 0-52.4%); Blue Crab, 23.0% (annual range = 0-59.4%); and other items, 7.0% (annual range = 0-40.0%). Spot have represented a noticeably lower percentage of fall diet items since 2014. The "Other" category accounted for a noticeably higher fraction of large Striped Bass diets by number in 2012 and 2017 (36.2% and 40.0%, respectively; Figure 9) than remaining years (< 9.7%). The vast majority of major prey were young-of-year (Uphoff et al. 2016).

By weight, Atlantic Menhaden predominated in large fish sampled (87.5% of diet weight during fall, 2006-2020, combined); Bay Anchovy accounted for 1.1%; Spot, 3.4%; Blue Crab, 3.7%; and other items, 4.3% (Figure 10). Estimates of C for large Striped Bass varied as much as 3.8-times among years sampled. The 2020 estimate of C of large fish was eighth highest of the 15 year time-series and represented its median (Figure 10).

Median PPLRs of large prey (Spot and Atlantic Menhaden) for large Striped Bass were 0.19-0.30 during 2006-2020 (Figure 11). Median PPLRs for large Striped Bass were much closer to the optimum (0.21 based on Overton et al. 2009) than for small fish.

Relative abundance indices of Striped Bass and major prey – Relative abundance of Striped Bass (RI) during 1981-2020 was lowest prior to 1994 (mean RI < 0.4 fish per trip; Figure 12). Estimates of RI then rose abruptly to a high level and remained there during 1995-2006 (mean = 2.6). Estimates of RI fell by about a third of the 1995-2006 mean during 2008-2013 (mean = 1.8), rose to 2.4-3.0 during 2014-2018, was 3.6 in 2019 (second highest of the time-series), then fell to 1.8 in 2020. The 90% confidence intervals indicated that RI was much lower during 1981-1993 than afterward and that there was some chance that RI during 2008-2013 was lower than other years during 1995-2019. Ninety percent CIs of the 2020 estimate were broad and overlapped both the lowest and highest estimates of RI during 1995-2019 (Figure 12). The trend in RI compared favorably to the trend in estimated aggregate abundance of 2- to 5-year old Striped Bass along the Atlantic Coast, particularly in the years after recovery was declared (1995; Uphoff et al. 2020). These estimates were well correlated (r = 0.79, P < 0.001).

Major pelagic prey were generally much more abundant during 1959-1994 than afterward (Figure 13). Bay Anchovy seine indices following the early to mid-1990s were typically at or below the bottom quartile of indices during 1959-1993. Highest Bay Anchovy trawl indices occurred in 1989-1992, 2001-2002, and 2020, while lowest indices occurred during 2006-2011 and 2015-2019. There was little agreement between the two sets of Bay Anchovy indices; however, there were few data points representing years of higher abundance in the years in common and contrast may have been an issue (comparisons are of mostly low abundance points). Atlantic Menhaden seine indices were high during 1971-1994 and much lower during 1959-1970 and 1995-2020. There has been a slight upward shift in Atlantic Menhaden seine indices from mostly below average during 1995-2012 to mostly just above average afterward (Figure 13)

Major benthic forage indices were low after the 1990s, but years of higher relative abundance were interspersed during the 2000s (Figure 14). Seine (1959-2019) and trawl (1989-2019) indices for Spot had similar trends and indicated high abundance during 1971-1994 and low abundance during 1959-1970 and after 1995 (with 3 or 4 years of higher indices interspersed). Spot indices in 2020 were much better than the previous nine years. Blue Crab densities (1989-2019) were generally at or above the time-series median during 1989-1998, and 2009-2015. Blue Crab density in 2020 was the lowest of the time-series (Figure 14).

Most of the annual indices of biomass of soft bottom benthic invertebrates during 1995-2018 were well above the time-series median during 2000-2009 (Figure 15). Indices well below the median indices occurred during 1996, 1998, 2003-2004, and 2011. Biomass indices have been below to near the median since 2010 (Figure 15).

Species-specific standardized FRs exhibited similar patterns during 1983-2019 (Figure16). Indices were at their highest in the early1980s and fell steadily. A nadir in the ratios appeared during 1995-2004, followed by occasional “spikes” of Spot and Blue Crab ratios and a slight elevation in Atlantic Menhaden ratios after 2004. With the exception of Blue Crab, FRs in 2020 were roughly as high as they were in 2005, 2008, and 2010 (Figure 16). The 90% CIs for prey to Striped Bass ratios indicated these ratios were high prior to 1994 and lower afterward (Atlantic Menhaden, Figure 17; Bay Anchovy, Figures 18 and 19; Spot, Figures 20 and 21; Blue Crab, Figure 22).

Relative survival of small Striped Bass - The unadjusted age 3 gill net index of male relative abundance on the spawning grounds did not indicate the same trend as age 3 abundance in the assessment (NEFSC 2019) during 1985-1995; abundance during 1985-1995 was at least as high as any other period of the time-series through 2020 (Figure 23). The hybrid approach resulted in much better agreement with age 3 abundance trends in the NEFSC (2019) stock assessment. The hybrid age 3 gill net index of male relative abundance (HI_3) on the spawning grounds indicated a dearth of high indices during 1985-1995. These low HI_3 year-classes were followed by appearances of large year-classes at age 3 in 1996, 1998, 1999, 2004, 2006, 2010, 2014, and 2018. The HI_3 indicated sharper changes in relative abundance of age 3 Striped Bass from year-to-year than the ASMFC (2019) assessment. Peaks generally aligned, but years of low abundance in the NEFSC (2019) assessment tended to be higher than would have been indicated by the hybrid gill net index (Figure 23).

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 24). Estimated SR was consistently high during 1986-1994 with 7 years above the median, 3 below, and 1 at the median; this time span coincided with consistently low RI estimates (Figure 25). After 1994, SR shifted to consistently below the median during 1999-2004 and varied during 2005-2020 (8 years were above the median, 7 were below. Large oscillations in SR above and below the median were evident during 2005-2011 and they dampened after 2011. There was very general support for a density-dependence survival hypothesis. Estimates of RI were usually much higher after 1994, although there was a period (2009-2009) where relative abundance was between its lows and highs (Figure 25). 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 2021).

Discussion

Average condition of small Striped Bass was between the target and threshold level during 2020 and large fish were at the target. Small Striped Bass condition was consistently poor (breaching the threshold) during 1998-2007 and shifted to a mix afterward. During 2008-2020, there were three 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 to 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, environmental conditions, mycobacteriosis, and catch-and-release and harvest mortality that reduce abundance and intraspecific competition among Striped Bass. The summer preceding our fall monitoring may be particularly stressful and potentially lethal. Summer has represented a period of no to negative growth in weight for ages 3-6 (Hartman and Brandt 1995b), higher mortality of diseased and healthy Striped Bass (Groner et al. 2018), hypoxia and temperature stress (Constantini et a. 2008; Maryland Sea Grant 2009; Coutant 2013; LaPointe et al. 2014; Kraus et al. 2015; Itakura et al. 2021), and high catch-and-release mortality (Lukacovic and Uphoff 2007). Condition of Striped Bass in summer was a good predictor of fall condition, and condition in fall of the previous year appeared related to condition in the next fall (Uphoff et al. 2017). If fewer fish make it through these hurdles, the survivors may benefit from reduced intraspecific competition for forage. The RI is a rather blunt indicator of resident abundance since it aggregates both large and small size groups and is likely to be dominated by the small size class. Improvement in condition due to greatly reduced abundance is not likely to be comforting to fishermen or managers.

Correlations of median above pyncnocline water temperature in October or November were weak, but did not preclude potential for warmer fall temperatures to negatively influence condition in fall. Water temperature should be considered if more comprehensive analyses (such as logistic regression) of factors influencing condition in fall are developed.

Large Striped Bass have been mostly at target PE associated with target P0 since 2014. A target was not readily suggested for PE of small fish, but PE was clearly below the threshold during 2008-2010, 2014, and 2016-2020. Estimates of PE for large and small Striped Bass were modestly correlated and both have improved from threshold conditions during 1998-2000 and 2006.

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

Atlantic Menhaden dominated small and large Striped Bass diets by weight during fall; 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 PPLR of large major prey and achievement of target P0 and PE for small fish in 2010, have been largely absent in fall diets of both size classes. The seine and trawl FRs for Spot during 2010 were much higher than other years in either the body fat or consumption time-series and were within the range estimated for 1990 (year used as a target for P0; Jacobs et al. 2013).

Small Striped Bass condition has improved since the mid-2000s, but not to the extent of large fish. The transition from small to large major prey may represent a bottleneck for small Striped Bass. Small Striped Bass would have more difficulty in catching and handling the same sized large major prey than large Striped Bass in any given year. Animal feeding in nature is composed of two distinct activities: searching for prey and handling prey (Yodzis 1994). Both can be influenced by prey size, with larger prey obtaining higher swimming speeds (typically a function of body length) that enable them to evade a smaller predator and larger size makes prey more difficult to retain if caught (Lundvall et al. 1999). With high size limits and low fishing mortality in place 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; historic pound net length-frequencies (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 reached the large size category (2011 year-class in 2016 and 2015 year-class in 2019).

Our concentration on fall diets did not directly consider some prey items in the "other" category that could be important in other seasons. White Perch (*Morone americana*) and benthic invertebrates other than Blue Crab are important diet items during winter and spring, respectively (Walter et al. 2003; Hartman and Brandt 1995c; Overton et al. 2009; 2015). These 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 seems anomalous.

A rapid rise in Striped Bass abundance in upper Bay during the mid-1990s, followed by a dozen more years at high abundance after recovery was declared in 1995, coincided with declines in relative abundance of Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab (i.e., major pelagic and benthic prey) to low levels. Changes in FRs largely reflected decreasing prey during 1983-1994 since RI was low. After 1995, prey indices stayed relatively low and RI increased; FR changes usually reflected fluctuations in RI. Striped Bass were often in poor condition during fall, 1998-2004, and vulnerable to starvation. Improvements in condition after 2007 coincided with lower Striped Bass abundance, spikes or slight increases in some major forage indices, and higher consumption of larger major prey (Spot and Atlantic Menhaden) in fall diets. A return of Striped Bass to high abundance after 2014 was not accompanied by increased major forage, but it appears that slightly higher Atlantic Menhaden seine indices since 2007, while not always statistically distinguishable from indices during the 1998-2004 when threshold P0 was predominant, may have biological significance based on improvement in recent body fat and fall diet metrics.

Forage to Striped Bass ratios indexed potential attack success on major prey. Atlantic Menhaden FR reached its nadir during 1995-2004 and has risen just above it since. The FRs for Atlantic Menhaden, Spot, and Bay Anchovy since 2005 have been well below those that occurred in 1990, the year used to set target conditions for P0 (Jacobs et al. 2013). Condition of both size classes improved after 2004, but improvement was steadier and more pronounced for large Striped Bass. Bay Anchovy were consistently present in fall diets of both size classes of Striped Bass during 2006-2014, but have fallen substantially as a percent of large fish diet since 2015 (from 28.8% to 0% in 2020) as Atlantic Menhaden became frequent in their fall diet (from 31.0% to 97.4%). Bay anchovy represented a variable percentage (22.7-87.9%) of small fish diets during 2006-2015 and had a steadier, higher frequency (65-90%) afterwards. 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 in recent years. Overton et al. (2015) described shifting prey dependence over time in Chesapeake Bay based on bioenergetics analyses of annual Striped Bass diets in the late 1950s, early 1990s, and early 2000s. By the early 2000s, there was a greater dependence on Bay Anchovy by all ages of Striped Bass and older fish had a greater dependence on the benthic component as Atlantic Menhaden declined in the diet (Overton et al. 2015). Stable isotope analyses of archived Striped Bass scales from Maryland's portion of Chesapeake Bay indicated an increasing shift from pelagic to benthic food sources during 1982-1997 (Pruell et al. 2003)

The soft bottom benthic index time-series covered 1995-2018 and changes prior to Striped Bass recovery and in the two most recent years could not be addressed. Benthic biomass has generally been lower since 2010. Changes in benthic invertebrate populations have the potential to affect Striped Bass directly or through reductions in benthic major prey. There was little indication of correspondence of the soft bottom benthic index to P0 of either size class of Striped Bass, but an exploratory analysis indicated a weak positive correlation of the two standardized Spot indices (combined into a single analysis) with the soft bottom index ($r = 0.31$, $P = 0.033$).

While top-down control of forage is suggested by opposing trends of major forage and Striped Bass, bottom-up processes may also be in play. A long-term decline Bay Anchovy in Maryland's portion of Chesapeake Bay may be 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-predatory 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, e.g. Secchi disk depth and zooplankton assemblages, were correlated with age-0 Menhaden abundance in the upper Bay. Variations in river flows to the Chesapeake Bay set up stratification, drive estuarine circulation, and cause fluctuations in inputs of freshwater, sediments, and nutrients, processes that greatly influence hypoxia (Hagy et al. 2004; Kemp et al. 2005; Maryland Sea Grant 2009). Woodland et al. (2021) demonstrated that bottom–up processes influenced fish and invertebrate forage (including our major forage species and benthic invertebrates included the BIBI based index; Blue Crabs were not examined) in Chesapeake Bay. Annual abundance indices of many forage taxa were higher in years when spring water temperatures warmed slowly. Forage indices also were related (in taxon-specific ways) to winter–spring chlorophyll concentration and freshwater discharge, and to three summer water quality variables: dissolved oxygen, salinity, and water temperature, in addition to a broad-scale climate indicator (Atlantic Multidecadal Oscillation or AMO; Woodland et al. 2021). The AMO was the best single predictor of recruitment patterns of Atlantic Menhaden in Chesapeake Bay and along the Atlantic coast, suggesting that broad-scale climate forcing was an important controller of recruitment dynamics, although the specific mechanisms were not identified (Buccheister et al. 2016). The MD Spot seine index was negatively and weakly correlated with the AMO (January-April mean; ($r = -0.41$, $P = 0.0012$, 1957-2017; J. Uphoff, unpublished).

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

Multiple lines of evidence suggest that survival of both small and large Striped Bass decreased in Chesapeake Bay since the late 1990s. Higher frequency of below time-series (1985-2020) median SR between ages 0 and 3 after 1996 was concurrent with declines in conventional tag-based estimates of survival of 457-711 mm of Striped Bass in Chesapeake Bay (based on time varying estimates of M). 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. 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 lagged two years behind downward shifts in forage-to-Striped Bass ratios. Dutil and Lambert (2000) found that the response of M of Atlantic Cod (*Gadus morhua*) could be delayed after unfavorable conditions. Similar to Striped Bass, some stocks of Atlantic Cod experienced forage fish declines, followed by declining body condition and increased M; starvation caused declines in energy reserves, physiological condition, and enzyme activity (Lilly 1994; Lambert and Dutil 1997; Dutil and Lambert 2000; Shelton and Lilly 2000; Rose and O'Driscoll 2002). Recovery of the northern stock of Atlantic Cod has paralleled recovery of Capelin (*Mallotus villosus*), its main prey (Rose and Rowe 2015); increases in size composition and fish condition and apparent declines in mortality followed. Condition of both size classes of resident Striped Bass has improved since the mid-2000s in concert with slight improvement in Atlantic Menhaden FR and consumption. No other major prey FR (or benthic invertebrate biomass) matches this timing. 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 fall in survival in the mid-to-late 1990s was consistent with a compensatory response to high Striped Bass abundance, low forage, and poor condition. The degree that M compensates with F may reduce effectiveness of management measures since total mortality, Z, may not be reduced by harvest restrictions when M increases as F decreases (Hilborn and Walters 1992; Hansen et al. 2011; Johnson et al. 2014). Single species stock assessments typically assume that M is constant and additive with F to keep calculations tractable (Hilborn and Walters 1992). Animal populations may exhibit additive mortality at low abundance and compensatory mortality at high abundance or compensatory mortality that changes continuously with density (Hansen et al. 2011). Increased M over time 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 fishing mortality, and compensatory mortality 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).

We developed the hybrid methodology for estimating SR over several years. It became apparent that SR estimates in Uphoff et al. (2015) were biased because age-3 gill net indices were not reflecting expected trends in abundance indicated by the stock assessment, juvenile indices, landings, and a long-term egg production index (Uphoff 1993; 1997; Richards and Rago 1999; NEFSC 2017; Durell and Weedon 2020). Uphoff et al. (2016) developed gill net indices adjusted for changes in catchability that reflected expected stock changes and used these as the numerator in the SR estimates. We revised the approach in Uphoff et al. (2018) and used it to estimate a SR time-series that reflected changes in catchability based on the most recent ASMFC Striped Bass stock assessment (NEFSC 2019).

Confining the spring gill net relative abundance index to 3-year-old males makes it likely that trends in SR will reflect survival of resident Striped Bass before harvest (i.e., due to M). Males are completely mature at age-3 (nearly all females mature at older ages), so they would be fully recruited to the gill net survey (Maryland Sea Grant 2009). Age-3 males in the spring gill net survey were nearly always well below minimum length limits for harvest (Versak 2021), but they could be subject to catch-and-release mortality. Observation error or changes in catchability of the spring gill net and juvenile surveys can also produce changes in SR. Uphoff et al. (2016) determined that gill net survey catchability of 3-year-old male Striped Bass changed as an inverse nonlinear function of population size. While there was some year-to-year variation in age 3 catchability, major changes that would lead to bias would require a sustained drop in total abundance. The SR index has an added complication in that it is a measure of survival over about 2.5 years, while other indices were annual or had potential lags less than 2.5 years.

An underlying assumption of the SR is a fairly constant migration schedule for male Striped Bass between when they are sampled as young-of-year and appear on the spawning grounds at age 3. Shifts in migration can produce similar changes as M. Migration estimates based on 1988-1991 spawning survey tagging (40-100 cm TL) indicated that larger Striped Bass were more likely to migrate from spawning areas of the Chesapeake Bay to coastal areas north of Cape May, NJ, than were smaller fish (Dorazio et al. 1994). Fewer males participate in the northward migration, but this difference appeared to reflect differences in size of mature males and females (Dorazio et al. 1994). Secor et al. (2020) confirmed this general migration schedule with acoustic tags. Kohlenstein (1981) determined that few young males leave the Chesapeake Bay.

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

“Other” prey (Uphoff et al. 2018). There has not been a readily discernable shift in patterns of PE, C, and frequency of diet items by number that would be readily attributed to changes from CBEF to FWHP sampling programs.

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

We treated hook-and-line samples in fall as random samples (Chipps and Garvey 2007) rather than as cluster samples (Rudershausen et al. 2005; Hansen et al. 2007; Overton 2009; Nelson 2014), i.e., individual fish rather than a school were considered the sampling unit. This choice reflected changing feeding behavior of Striped Bass in fall and the nature of hook-and-line fishing for them. Fall is a period of active feeding and growth for resident Striped Bass and forage fish biomass is at its peak (Hartman and Brandt 1995c; Walter and Austin 2003; Overton et al. 2009). Striped Bass leave the structures they occupied during summer-early fall and begin mobile, aggressive, open water feeding. Forage begins to migrate out of the Bay and its tributaries (and refuges therein) or to deeper water at this time and are much more vulnerable to predation. Both major forage and Striped Bass schools are constantly moving and changing. Schools of Striped Bass and their prey no longer have a fixed location, presenting well mixed populations (J. Uphoff, MD DNR, personal observation) that made a random sampling assumption reasonable. Treating hook-and-line samples as a cluster required a broad definition of a cluster in Overton et al. (2009), i.e., an entire day’s effort that assumed fish caught that day represented a non-independent sample. Neither assumption (random or cluster) provided a complete description of how hook-and-line sampling works and we believed that random sampling was a better fit.

Two additional objectives of this forage assessment are low cost and tractability for available staff. Ecosystem based fisheries management has been criticized for poor tractability, high cost, and difficulty in integrating ecosystem considerations into tactical fisheries management (Fogarty 2014). It has been the principal investigator’s unfortunate experience that complex and comprehensive ecosystem-based approaches to fisheries management for the entire Chesapeake Bay i.e., Chesapeake Bay Ecopath with Ecosim and MD Sea Grant’s Ecosystem Based Fisheries Management for Chesapeake Bay (Christensen et al. 2009; MD Sea Grant 2009) have not gained a foothold in Chesapeake Bay’s fisheries management. This is not surprising. While policy documents welcome ecosystem-based approaches to fisheries management and a large number of studies that have pointed out the deficiencies of single-species management, a review of 1,250 marine fish stocks worldwide found that only 2% had included ecosystem drivers in tactical management (Skern-Mauritzen et al. 2016).

The index-based forage assessment approach represents a less complex, low-cost attempt to integrate forage into Maryland’s Striped Bass management. Given the high cost of implementing new programs, we have used information from existing sampling programs and indices (i.e., convenience sampling and proxies for population level estimates, respectively;

Falcy et al. 2016). This trade-off is very common in fisheries and wildlife management (Falcy et al. 2016).

We used available estimates of central tendency and variability for ratio simulations. We did not attempt to standardize indices to account for influences such as latitude, date, and temperature. Use of standardizing techniques that "account" for other influences have increased, but they require additional staff time and often barely have a detectable effect on trends. Maunder and Punt (2004) described that their effect "can be disappointingly low" and they do not guarantee removal of biases.

Forage indices and forage to Striped Bass ratios were placed on the same scale by dividing them by arithmetic means over a common time period (ratio of means). Conn (2009) noted in several scenarios that arithmetic mean of scaled indices performed as well as the single index estimated by a 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.

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

By splitting into small and large fish size classes, the P0 and PE metrics represent four pieces of information. The science of decision making has shown that too much information can lead to objectively poorer choices (Begley 2011). The brain's working memory can hold roughly seven items and any more causes the brain to struggle with retention. Proliferation of choices can create paralysis when the stakes are high and information is complex (Begley 2011).

The P0 and PE targets and thresholds represent a framework for condensing complex ecological information so that it can be communicated simply to decision makers and stakeholders. The target, threshold, or in-between status approach for P0 and PE was similar to traffic light style representations (but without the colors) for applying the precautionary approach to fisheries management (Caddy 1998; Halliday et al. 2001). Traffic light representations can be adapted to ecosystem-based fisheries management (Fogarty 2014). The strength of the traffic light method is its ability to take into account a broad spectrum of information, qualitative as well as quantitative, which might be relevant to an issue (Halliday et al. 2001). It has three elements – a reference point system for categorization of indicators, an integration algorithm, and a decision rule structure based on the integrated score (Halliday et al. 2001). In the case of P0 and PE, it contains the first two elements, but not the last. Decision rules would need input and acceptance from managers and stakeholders.

Some form of integration of indicator values is required in the traffic light method to support decision making and simplicity and communicability are issues of over-riding importance (Halliday et al. 2001). Integration has two aspects, scaling the indicators to make them comparable (target, threshold, or in-between status in our case) and applying an operation to summarize the results from many indicators. The reduced number of metrics with our size-based approach (two metrics) does not require the latter operation. Caddy (1998) presented the simplest case for single-species management where indicators were scaled by converting their values to traffic lights, and decisions were made based on the proportion of the indicators that were red. In 2020, the P0 and PE indicators for both size classes would not have been red.

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Table 1. Important abbreviations and definitions.

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

Table 2. Number of dates sampled and number of small (<457 mm, TL) and large sized Striped Bass collected for October-November diet information in each size category, by year. Diet collections were made by Chesapeake Bay Ecological Foundation (CBEF) during 2006-2013 and MD DNR Fish and Wildlife Health Program (FWHP) during 2014-2020. 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

Table 3. Estimates of proportion of Striped Bass without body fat (P0) for small (< 457 mm, TL) and large (≥ 457 mm) Striped Bass and above pycnocline median water temperature (ºC) at eight Maryland mesohaline mainstem Bay water quality monitoring stations (T. Parham, Resource Assessment Service, MD DNR, personal communication) for October (1998-2020) or November (1998-2019).

Year	P0 small	P0 large	October Temperature	November Temperature
1998	0.680	0.775	20.2	13.0
1999	0.765	0.819	20.5	15.5
2000	0.712	0.776	17.6	13.7
2001	0.672	0.770	18.5	13.3
2002	0.667	0.600	20.8	12.8
2003	0.699	0.697	19.1	12.8
2004	0.697	0.736	19.4	14.2
2005	0.688	0.556	16.9	14.4
2006	0.678	0.563	18.3	13.4
2007	0.872	0.466	21.4	15.6
2008	0.291	0.042	18.4	12.9
2009	0.440	0.243	15.9	13.1
2010	0.387	0.247	18.1	12.0
2011	0.817	0.465	19.6	12.8
2012	0.755	0.561	18.6	11.1
2013	0.358	0.480	19.5	11.4
2014	0.364	0.138	19.0	11.1
2015	0.167	0.081	18.2	14.1
2016	0.840	0.315	19.6	14.4
2017	0.251	0.171	14.1	14.1
2018	0.358	0.046	20.6	10.2
2019	0.696	0.067	19.8	12.2
2020	0.462	0.058	19.9	

Figure 1. Upper Bay (Maryland's portion of Chesapeake Bay) with locations of forage index sites (black dots = seine site and grey squares = trawl site), and regions sampled for Striped Bass body fat and diet data. Patuxent River seine stations are not included in analyses.

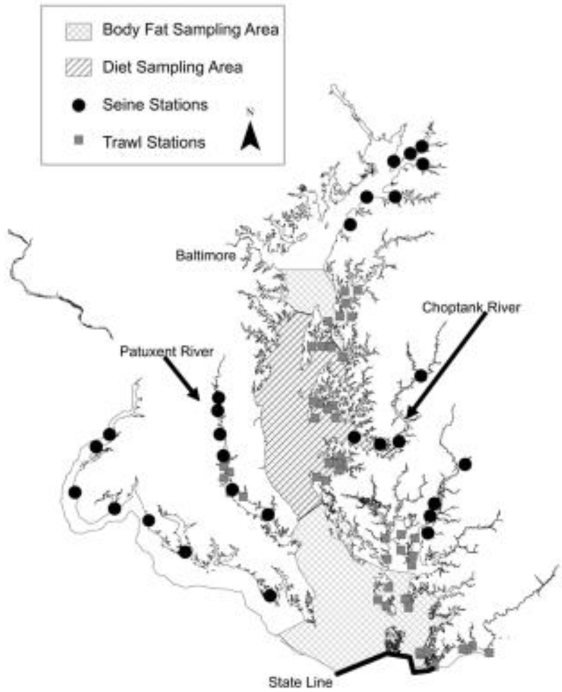

Figure 2. Proportion of small Striped Bass without body fat (P0) during October-November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

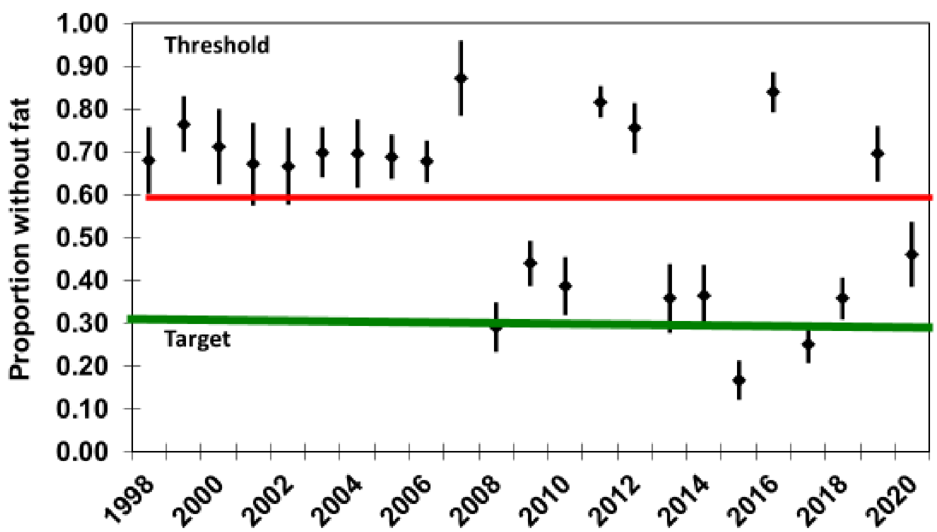

Figure 3. Proportion of small Striped Bass guts without food (PE) in fall and its 90% confidence interval. Red diamond represents threshold PE.

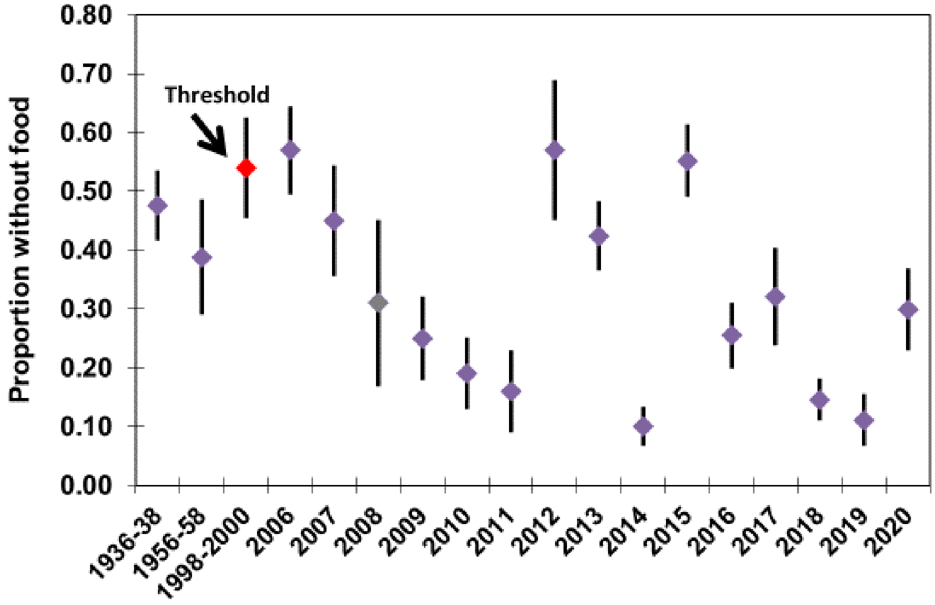

Figure 4. Percent, by number (counts of individuals plus presence of parts), of identifiable (excludes unknown) major forage groups in small Striped Bass (< 457 mm TL) guts, in fall.

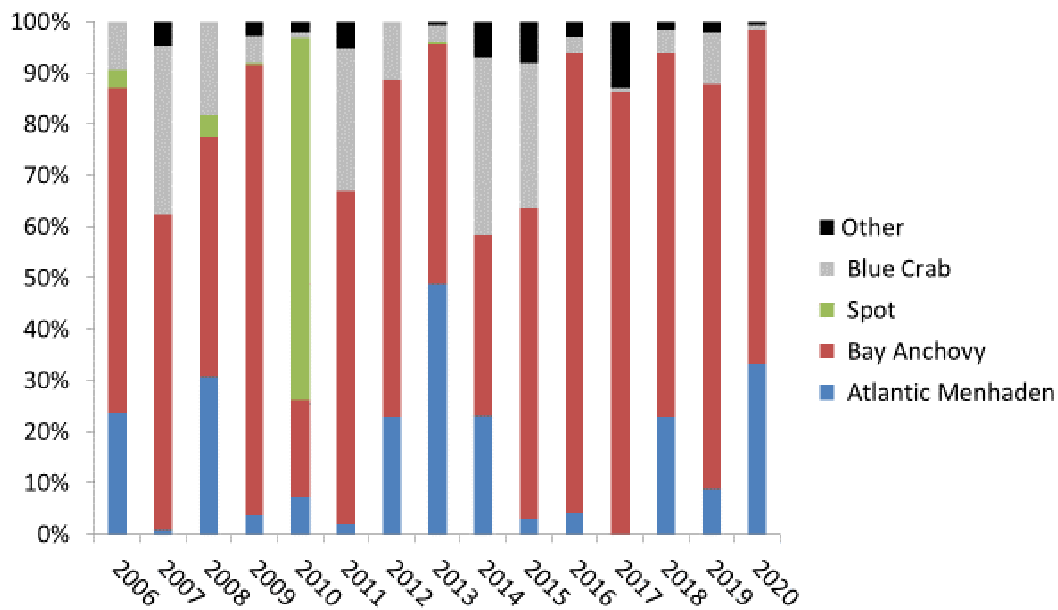

Figure 5. Gram prey consumed per gram (C) of small (< 457 mm TL) Striped Bass in fall hook-and-line samples. Age-0 forage dominate the diet. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

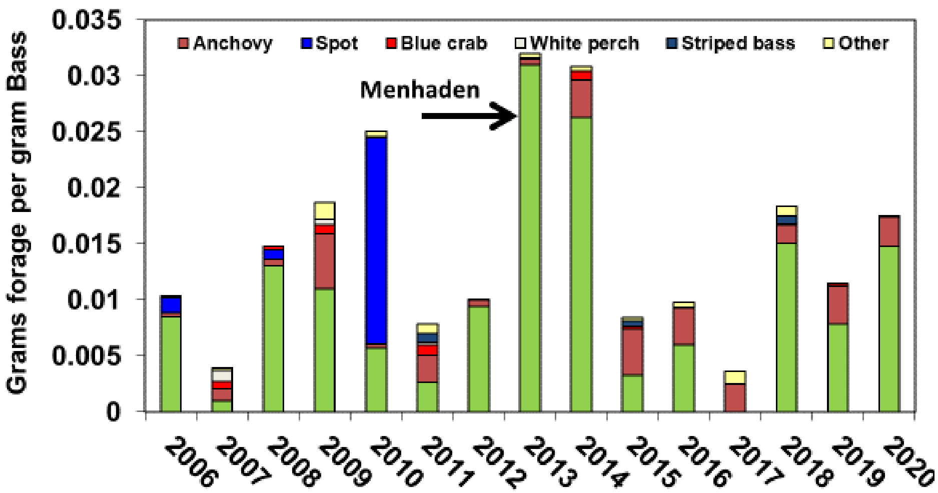

Figure 6. Median prey-predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for small (< 457 mm) Striped Bass. Optimum ratio was estimated by Overton et al. (2009).

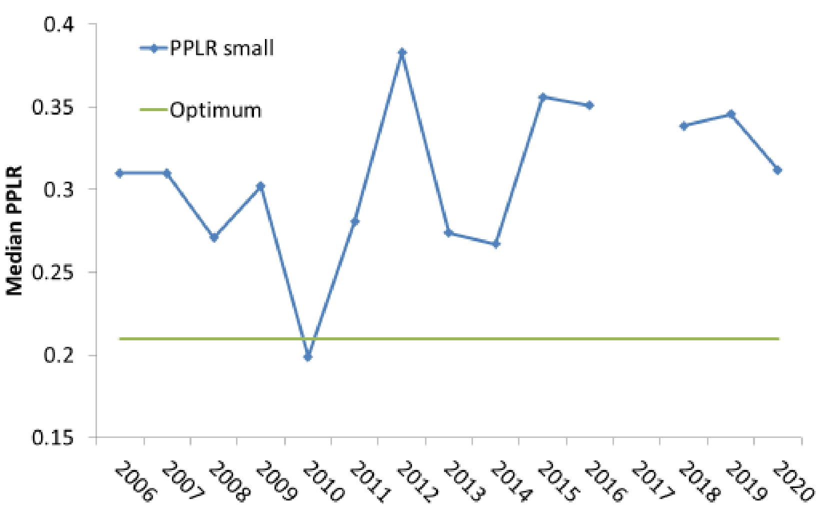

Figure 7. Proportion of large Striped Bass without body fat (P0) during October-November (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

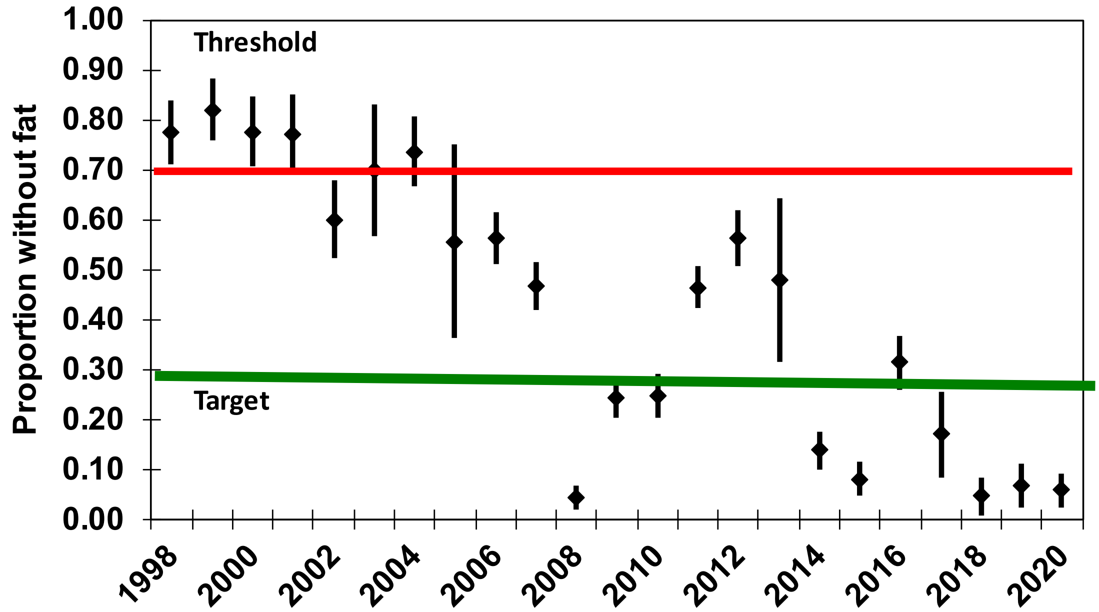

Figure 8. Proportion of large Striped Bass (≥ 457 mm or 18 in, TL) guts without food (PE) in fall and its 90% confidence interval, with body fat targets (best condition) and thresholds (poorest condition).

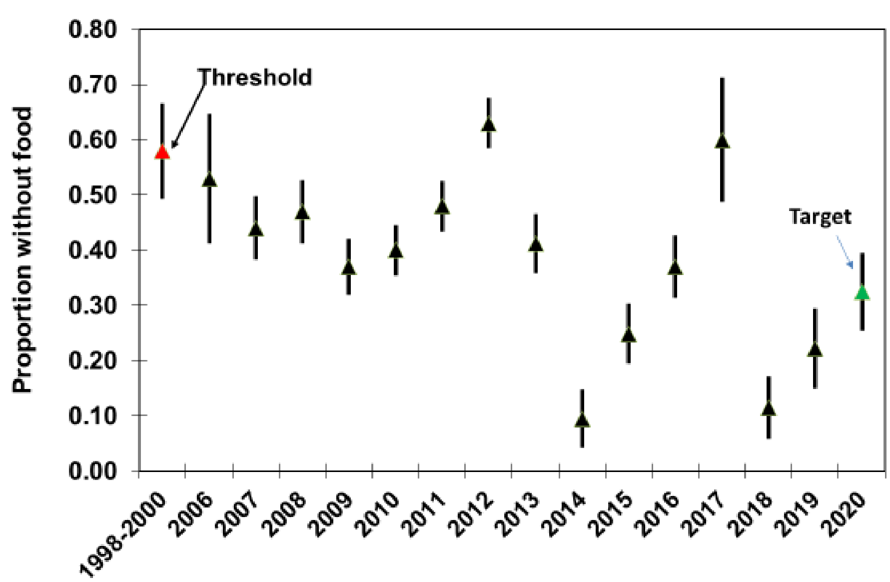

Figure 9. Percent of large Striped Bass (≥ 457 mm TL) identifiable diet represented by major forage groups, by number, in fall.

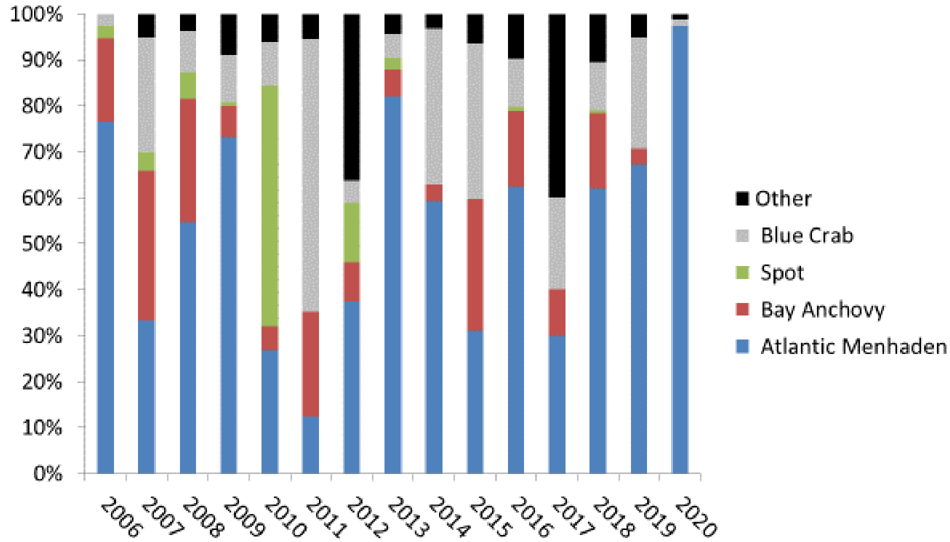

Figure 10. Grams of prey consumed per gram (C) of large (≥ 457 mm TL) Striped Bass during October-November. Fall consumption dominated by age 0 forage. Arrow indicates color representing Atlantic Menhaden which disappeared on the figure legend.

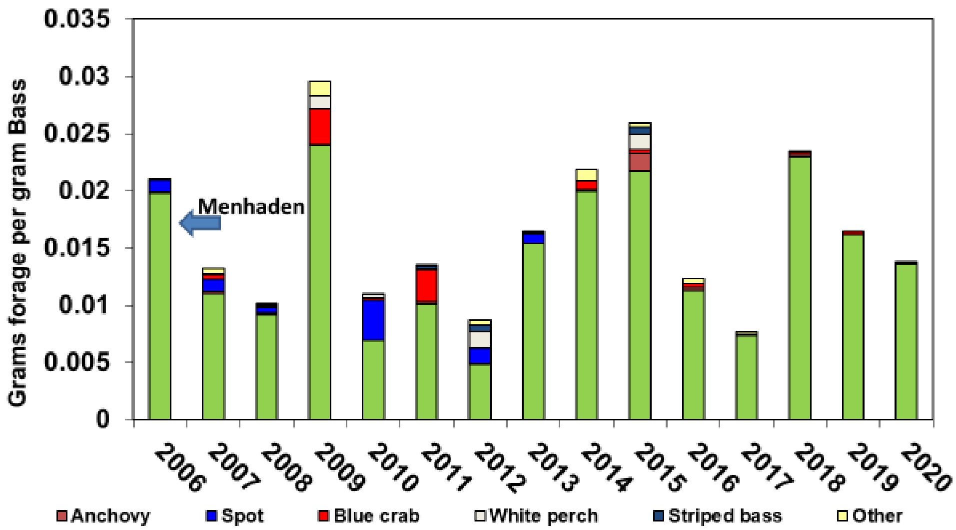

Figure 11. Median prey-predator length ratios (PPLR) for large major prey (Spot and Atlantic Menhaden) for large Striped Bass (≥ 457 mm). Optimum ratio was estimated by Overton et al. (2009).

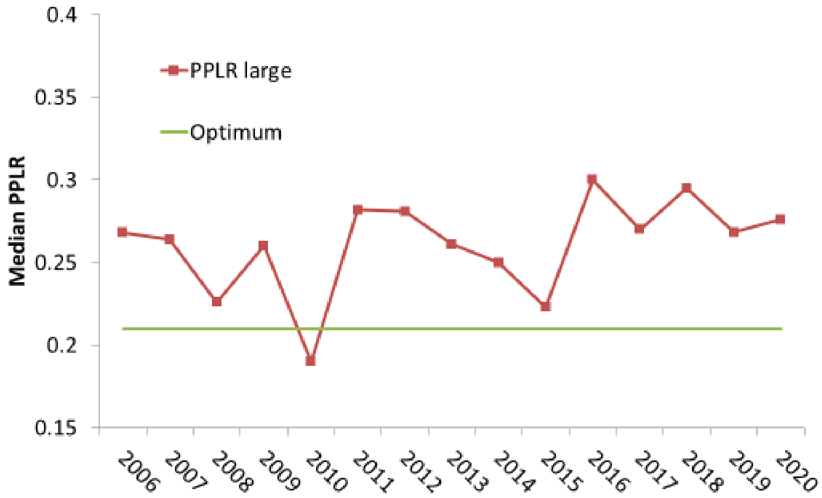

Figure 12. Maryland resident Bay Striped Bass annual abundance index (RI; MD MRIP inshore recreational catch per private boat trip during September-October; mean = black line) during 1981-2020 and its 90% confidence intervals based on @Risk simulations of catch and effort distributions. Catch = number harvested and released.

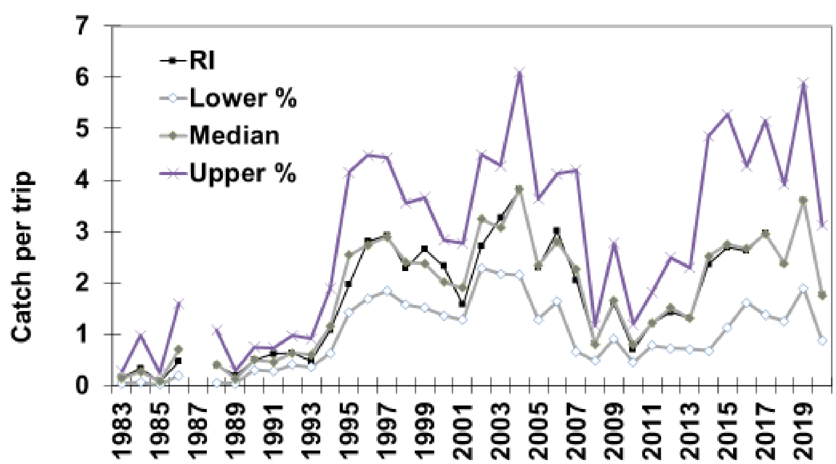

Figure 13. Trends in major pelagic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2020. Indices were standardized to their 1989-2020 means (years in common). Menhaden = Atlantic Menhaden and Anchovy = Bay Anchovy.

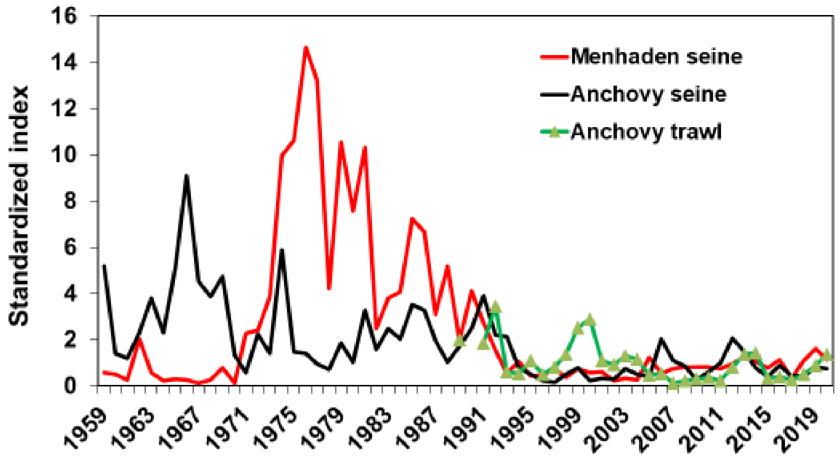

Figure 14. Trends in major benthic prey of Striped Bass in Maryland Chesapeake Bay surveys, 1959-2020. Indices were standardized to their 1989-2020 means (years in common).

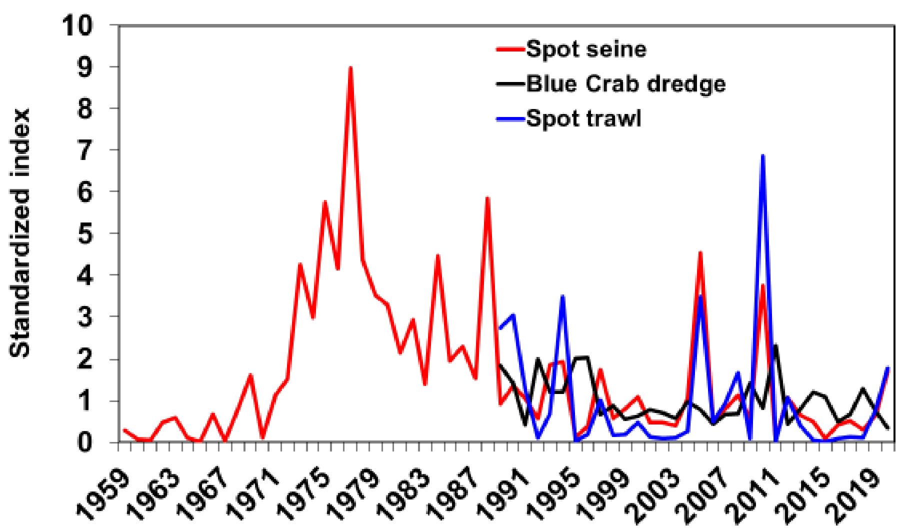

Figure 15. Trends in soft bottom benthic invertebrate biomass in Maryland waters (grams / m^2) and its median during 1995-2018 (based on Figure 3-37 in Llansó and Zaveta 2019).

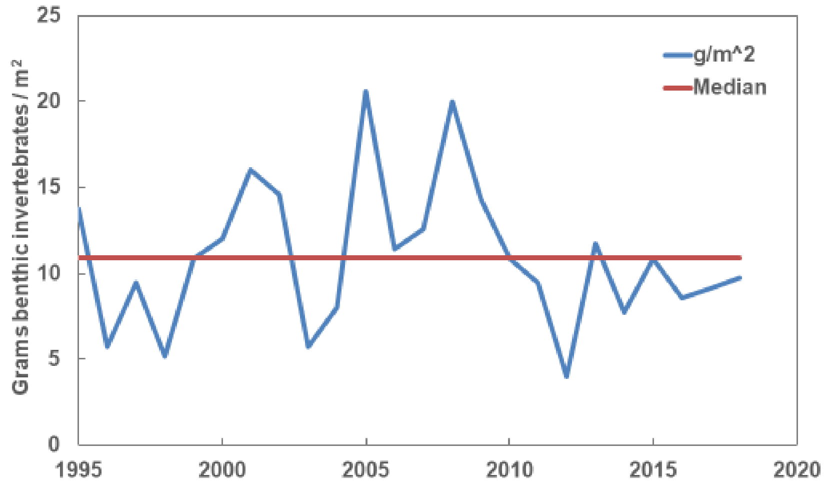

Figure 16. Trends of standardized ratios of major upper Bay forage species indices to Striped Bass relative abundance (RI). Forage ratios have been standardized to their 1989-2020 mean to place them on the same scale. S indicates a seine survey index; T indicates a trawl survey index; and D indicates a dredge index. Note the $\log_{10}$ scale on Y-axis.

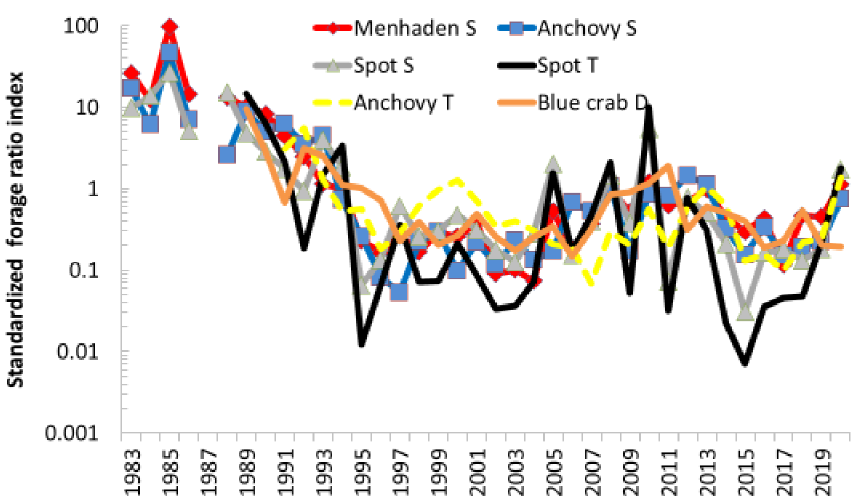

Figure 17. Atlantic Menhaden index to Striped Bass index (RI) ratios (Atlantic Menhaden FR) during 1983-2020 and their 90% confidence intervals based on @Risk simulations of Atlantic Menhaden seine indices and RI distributions. Note $\log_{10}$ scale on the Y-axis.

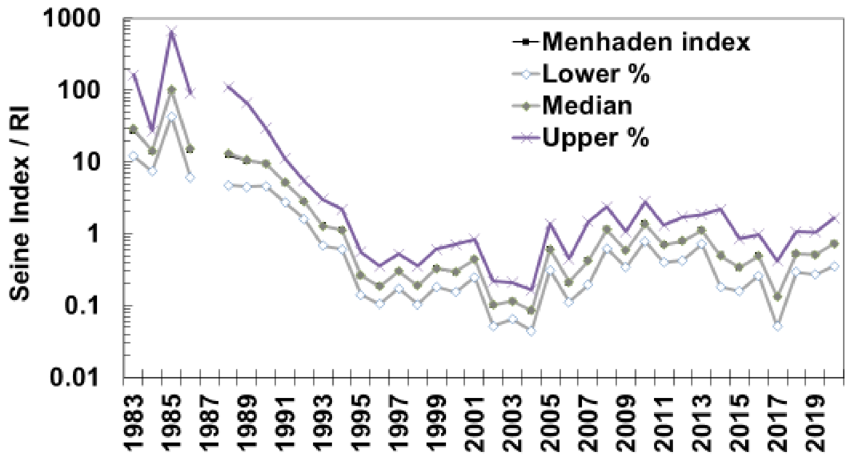

Figure 18. Bay Anchovy seine index to Striped Bass index (RI) ratios (Bay Anchovy seine FR) during 1983-2020 and their 90% confidence intervals based on @Risk simulations of Bay Anchovy seine indices and RI distributions. Note $\log_{10}$ scale on the Y-axis.

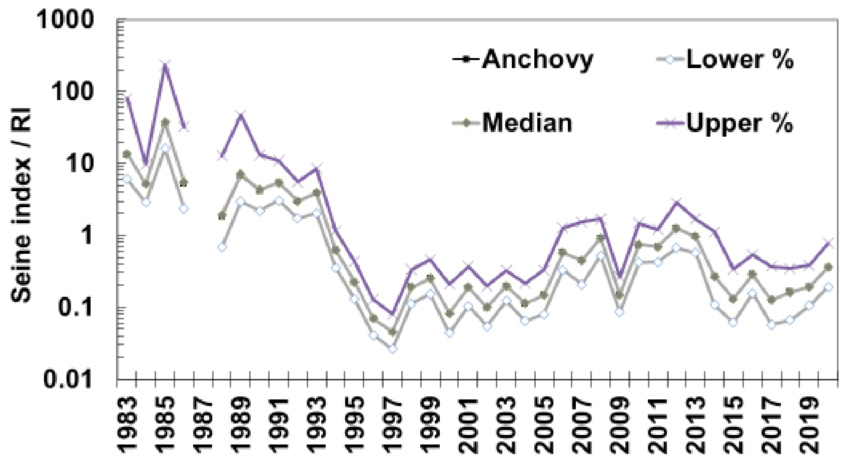

Figure 19. Bay Anchovy trawl index to Striped Bass index (RI) ratios (Bay Anchovy trawl FR) during 1989-2020 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale on the Y-axis.

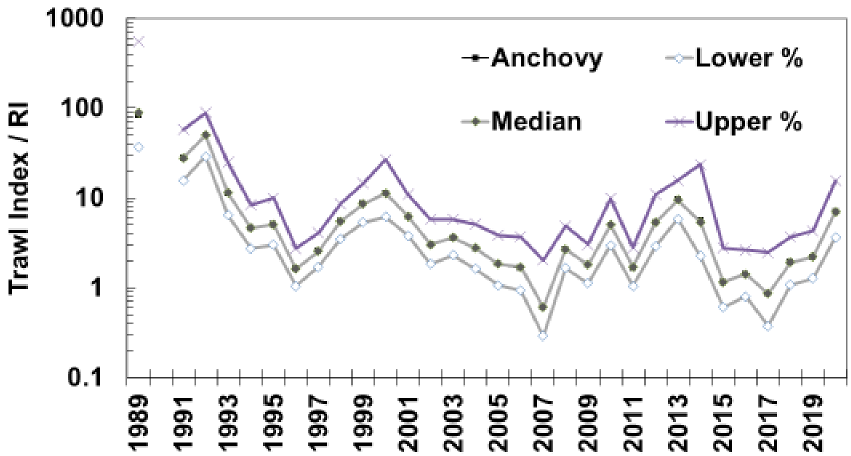

Figure 20. Spot seine index to Striped Bass index (RI) ratios (Spot seine FR) during 1983-2020 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Spot seine indices and RI. Note $\log_{10}$ scale on Y-axis

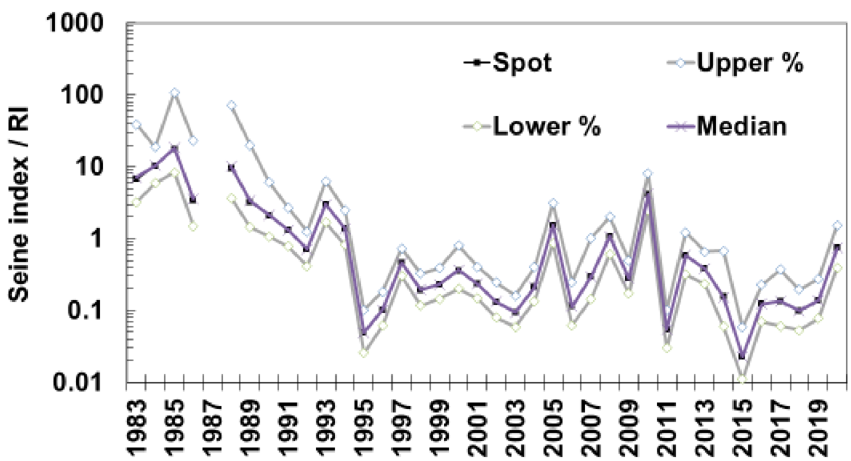

Figure 21. Spot trawl index to Striped Bass index (RI) ratios (Spot trawl FR) during 1989-2020 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of trawl indices and RI. Note $\log_{10}$ scale on Y-axis.

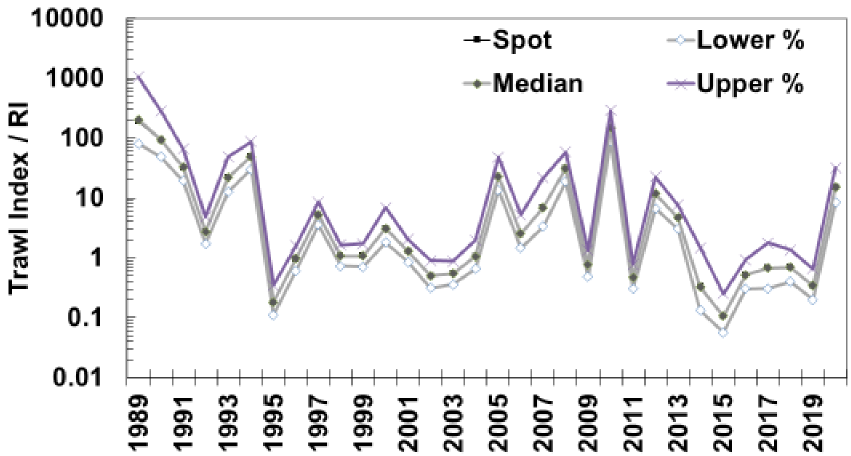

Figure 22. Blue Crab index to Striped Bass index (RI) ratios (Blue Crab FR) during 1989-2020 and their 90% confidence intervals based on @Risk simulations of central tendency and estimated dispersion of data of Blue Crab (age 0) winter dredge densities and RI. Note the $\log_{10}$ scale on Y-axis.

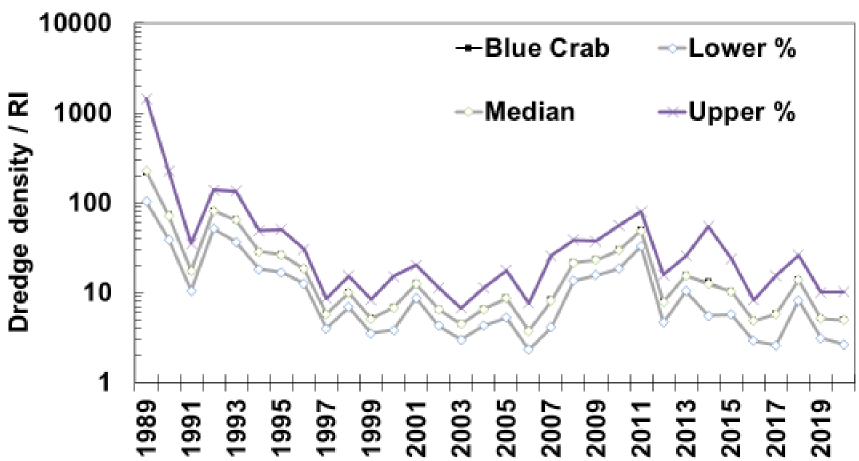

Figure 23. Time-series of age 3 Striped Bass relative abundance on two major Maryland spawning areas (hybrid index = gill net index adjusted for changing catchability during 1985-1995; units = number of fish captured in 1000 square yards of net per hour) and abundance of age 3 Striped Bass along the Atlantic Coast estimated by the NEFSC(2019) statistical catch-at-age model. Hybrid index time series =1985-2020; Statistical catch-at-age model time-series = 1985-2017. Unadjusted = gill net index not adjusted for catchability during 1985-1995.

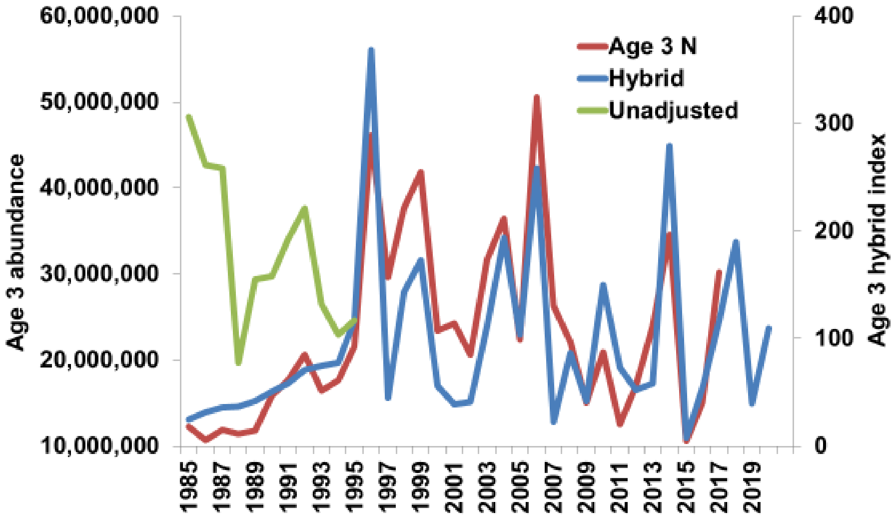

Figure 24. Relative survival (SR) of a Striped Bass year-class to approximately its third birthday during 1985-2020 and 90% confidence intervals based on @Risk simulations of age 3 hybrid gill net indices divided by juvenile index distributions. Year of estimate = year-class + 3.

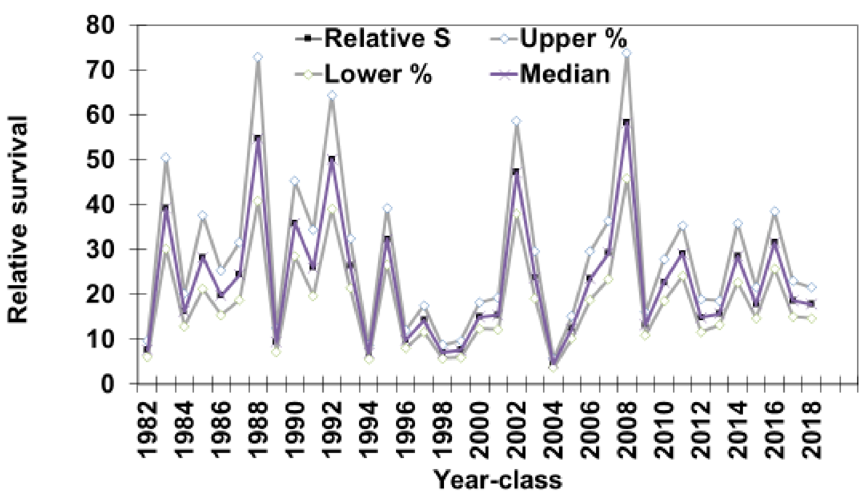

Figure 25. Relative survival (SR) of Striped Bass between ages 0 and 3, its median, and the relative abundance of resident Striped Bass (RI) in the previous year during 1985-2020 (year-class = year – 3).