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MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

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Report Organization

This report was completed during April, 2015. It consists of summaries of 2014 activities for Jobs 1–4 under this grant. All pages are numbered sequentially; there are no separate page numbering systems for each Job. Job 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 a Job are numbered as section number – table number (1-1, 1-2, etc). Figures are numbered in the same fashion. This nomenclature applies to Jobs 1, 3, and 4.

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://www.dnr.maryland.gov/fisheries/fhep/.

<u>SURVEY TITLE:</u> MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS
<u>PROJECT 1</u>: FINFISH HABITAT AND MANAGEMENT

Job 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern: development targets and thresholds

Jim Uphoff, Margaret McGinty, Alexis Maple, Carrie Hoover, Shaun Miller, and Brian Redding

Executive Summary

Section 1: Stream Ichthyoplankton - Proportion of samples with Herring (Blueback Herring, Alewife, American Shad, and Hickory Shad) eggs and-or larvae (P_{herr}) provided reasonably precise annual estimates of relative abundance based on encounter rate. Regression analyses indicated a negative relationships of P_{herr} with development (indicated by structures per hectare or C / ha) and conductivity (a measure of dissolved salts), and a positive relationship of C / ha with conductivity consistent with the hypothesis that urbanization was detrimental to stream spawning. Magnitude of P_{herr} may indicate how much habitat is available or how attractive it is from year to year more-so than fluctuations in abundance of spawners. In developed watersheds, a combination of urban and natural stream processes may create varying amounts of ephemeral spawning habitat annually and dampen spawning migrations through changes in water chemistry (indicated by conductivity).

We pooled Mattawoman Creek data across years in order to estimate proportions of samples with White Perch eggs and larvae or Yellow Perch larvae to overcome the effect of their limited spatial distribution on annual sample size. This allowed us to compare for 1989-1991 collections (C / ha = 0.43–0.47) with 2008-2010 (C / ha = 0.87-0.90), and 2011-2014 (C / ha = 0.90-0.91) at the same combinations of downstream sites. These estimates did not detect a loss in stream spawning for Yellow Perch. Less White Perch stream spawning was detected during 2008-2010 than the other time periods. Proportions of stream samples with White Perch eggs or larvae were similar for 1989-1991 and 2011-2014.

Section 2: Estuarine Yellow Perch Larval Sampling - Estimates of proportion of plankton net tows in a subestuary with Yellow Perch larvae, L_p (an indicator of egg and larval viability), declined perceptibly once watershed development exceeded the suburban threshold (0.83 structures per hectare, C / ha, equivalent to 10% impervious surface, IS), most likely from endocrine disrupting contaminants. Interpretation of the influence of salinity class (tidal-fresh or brackish) or other types of land cover (agriculture and forest) on L_p is hindered because existing patterns of development do not represent all possible combinations.

There appears to be some potential for development to negatively influence flow of organic matter (OM) off the watershed. However, development's influence on OM may not matter much unless it prevents important, but intermittent, episodes of high watershed OM delivery that would have been followed by matches of high copepod abundance and successful feeding of Yellow Perch larvae.

We combined an egg per recruit (EPR) model for Chesapeake Bay Yellow Perch with estimated relative larval survival (L_p) at different levels of development to explore fishing mortality (F) reductions needed offset egg and larval viability declines from development and maintain egg production at a target level. At 10% IS (suburban watershed threshold), a 24-25% reduction in F was needed to maintain target EPR produced at the target level of development

(5% IS, a rural level of development); a 63-64% reduction was necessary at 15% IS; and at 20% IS it was not possible to compensate for diminished survival. Percentage reductions in F needed to maintain target EPR were independent of size limits imposed. We do not expect that managers are going to apply the development EPR model tactically, but it can provide a strategic sense of sacrifices needed to maintain target EPR as habitat deteriorates to judge whether it is worth doing them. Fishing reductions can buy time for effective growth management and habitat reconstruction measures to be put in place.

Section 3: Estuarine Fish Community Summer Sampling – Plots of species richness (number of species encountered) in 4.9 m trawl collections against C / ha did not suggest relationships for either tidal-fresh or oligohaline (low salinity) subestuaries. Plots did suggest that species richness declined when development went beyond the threshold in watersheds of mesohaline (mid-strength salinity) subestuaries. In general, these exploratory analyses of species richness and development supported trends found in analyses of development and dissolved oxygen (DO). Bottom DO was not negatively influenced by development in tidal-fresh or oligohaline subestuaries, but was in mesohaline subestuaries. Depletion of DO in bottom waters of mesohaline subestuaries to hypoxic or anoxic levels represented a direct loss of habitat.

We continued to track bottom DO, submerged aquatic vegetation (SAV), total ammonia nitrogen (TAN; NH3 plus NH4), development (C / ha), and number and diversity of finfish in 3.1 m and 4.9 m trawl samples from Mattawoman Creek. Development in Mattawoman Creek's watershed more than doubled between 1989 (0.43 C / ha) and 2011 (0.91 C / ha) and reached the suburban threshold in 2006. A downward shift of bottom DO after 2000 corresponded to changes in Mattawoman Creek's subestuary chlorophyll a from high to low and SAV acreage shifting from low (coverage of ~10% or less of water area) to high (coverage of > 30%). Median TAN was low and stable through 2000 and then began a rapid rise to a spike in 2002. Median TAN dropped after 2002, but was elevated beyond that seen prior to 2001; during 2007-2009, median TAN was consistently elevated beyond this period's baseline. Mattawoman Creek's finfish abundance appeared to be susceptible to boom and bust dynamics after 2001. "Busts" were concurrent with spikes (2002) or plateaus (2007-2009) of TAN. Collapses of the magnitude exhibited during 2002 and 2008-2009 were not detected previously. Recovery of fish abundance since 2011 has coincided with moderate values of median TAN. Dominant species (those comprising of 90% of catches) are now dominated by White Perch (YOY) and Spottail Shiners. Since 2003, four planktivores and adult White Perch have largely dropped out of the dominant category.

During 2014, we sampled TAN at four sites and three within-site locations (channel, edge of SAV bed, and in the SAV bed) in Mattawoman Creek over a three month period. Few samples exceeded the minimum detection limit. We conducted one 24-hour survey of TAN in a 1-m deep, dense SAV bed and found that detectable TAN and conditions of concern were present (high ammonia at the surface and low DO at the bottom) in the bed. High TAN occurred during late afternoon; our monitoring would normally have concluded by then. Channel-based monitoring may not reflect conditions within SAV beds.

<u>SURVEY TITLE:</u> MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS
<u>PROJECT 1</u>: FINFISH HABITAT AND MANAGEMENT

JOB 1: Development of habitat-based reference points for recreationally important Chesapeake Bay fishes of special concern: development targets and thresholds

Section 1: Stream Ichthyoplankton Sampling

Carrie Hoover, Alexis Park, Margaret McGinty, Jim Uphoff, Shaun Miller, and Brian Redding

Introduction

Surveys to identify spawning habitat of White Perch, Yellow Perch and "Herring" (Blueback Herring, Alewife, American Shad, and Hickory Shad) were conducted in Maryland during 1970-1986. These data were used to develop statewide maps depicting anadromous fish spawning habitat (O'Dell et al. 1970; 1975; 1980; Mowrer and McGinty 2002). Many of these watersheds have undergone considerable development and recreating these surveys provided an opportunity to explore whether spawning habitat declined in response to urbanization. Surveys based on the sites and methods of O'Dell et al. (1975) were used to sample Mattawoman Creek (2008-2014), Piscataway Creek (2008-2009 and 2012-2014), Bush River (2005-2008 and 2014) and Deer Creek (2012-2014; Figure 1-1).

Mattawoman and Piscataway Creeks are adjacent Coastal Plain watersheds along an urban gradient emanating from Washington, DC (Figure 1-1). Piscataway Creek's watershed is both smaller than Mattawoman Creek's and closer to Washington, DC. Bush River is located in the urban gradient originating from Baltimore, Maryland, and is located in both the Coastal Plain and Piedmont physiographic provinces. Deer Creek is entirely located in the Piedmont north of Baltimore, near the Pennsylvania border (Clearwater et al. 2000). Bush River and Deer Creek are adjacent to each other (Figure 1-1).

We developed two indicators of anadromous fish spawning in a watershed based on presence-absence of eggs and larvae: occurrence at a site and proportion of samples with eggs and larvae. Occurrence of eggs or larvae of an anadromous fish group (White Perch, Yellow Perch, and Herring) at a site, recreated the indicator developed by O'Dell et al. (1975; 1980). This indicator was compared to the extent of development in the watershed (counts of structures per hectare or C/ha) between the 1970s and the present. We also developed an indicator of relative abundance, proportion of samples with eggs and-or larvae of anadromous fish groups, from collections in the 2000s and compared it to C/ha and summarized conductivity data. Conductivity was monitored and examined to see whether urbanization had affected stream water quality. Increases in conductivity have been strongly associated with urbanization (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012).

Methods

Stream sites sampled for the anadromous fish eggs and larvae during 2005-2014 were typically at road crossings that O'Dell et al. (1975) determined were anadromous fish spawning sites during the 1970s. O'Dell et al. (1975) summarized spawning activity as the presence of any species group egg, larva, or adult at a site. O'Dell et al. (1975) sampled eggs and larvae with stream drift ichthyoplankton nets and adults were sampled by wire traps.

All collections during 2005-20114, with the exception of Deer Creek during 2012-2014, were made by citizen volunteers who were trained and monitored by program biologists. During March to May, between 2008 and 2014, ichthyoplankton samples were collected in Mattawoman Creek from three tributary sites (MUT3-MUT5) and four mainstem sites (MC1-MC4; Figure 1-2; Table 1-1). Tributary site (MUT4) was selected based on volunteer interest and added in 2010, while tributary site (MUTX) was added in 2014. Piscataway Creek stations were sampled during 2008-2009 and 2012-2014 (Figure 1-3; Uphoff et al. 2010). Bush River stations were sampled during 2005-2008 and 2014 (Figure 1-4; McGinty et al. 2009). Deer Creek sites SU01-SU04 were added to sampling in 2012 and sampling continued in 2013-2014 with the addition of site SU05 (Figure 1-5). Table 1-1 summarizes sites, dates, and sample sizes in Mattawoman, Piscataway and Deer Creeks, and Bush River during 2005-2014.

Ichthyoplankton samples were collected at each site using stream drift nets constructed of 360-micron mesh. Nets were attached to a square frame with a 300 • 460 mm opening. The stream drift net configuration and techniques were the same as those used by O'Dell et al. (1975). The frame was connected to a handle so that the net could be held stationary in the stream. A threaded collar on the end of the net connected a mason jar to the net. Nets were placed in the stream for five minutes with the opening facing upstream. Nets were retrieved and rinsed in the stream by repeatedly dipping the lower part of the net and splashing water through the outside of the net to avoid sample contamination. The jar was removed from the net and an identification label describing site, date, time, and collectors was placed in the jar. The jar was sealed and placed in a cooler with ice for transport when collections were made by volunteers. Preservative was not added by volunteers at a site because of safety and liability concerns. Formalin was added on site by DNR personnel. Water temperature (°C), conductivity (µS/cm), and dissolved oxygen (DO, mg/L) were recorded at each site using a hand-held YSI Model 85 meter. Meters were calibrated for DO each day prior to use. All data were recorded on standard field data forms and verified at the site by a volunteer. After a team finished sampling for the day, the samples were preserved with 10% buffered formalin. Approximately 2-ml of rose bengal dye was added in order to stain the organisms red to aid sorting.

Ichthyoplankton samples were sorted in the laboratory by project personnel. All samples were rinsed with water to remove formalin and placed into a white sorting pan. Samples were sorted systematically (from one end of the pan to another) under a 10x bench magnifier. All eggs and-or larvae were removed and were retained in a small vial with a label (site, date, and time) and stored with 20% ethanol for later identification under a microscope. Each sample was sorted systematically a second time for quality assurance (QA). Any additional eggs and-or larvae found were removed and placed in a vial with a label (site, date, time, and QA) and stored with 20% ethanol for identification under a microscope. All eggs and larvae found during sorting (both in original and QA vials) were identified as either Herring (Blueback Herring, Alewife, Hickory Shad, and American Shad), Yellow Perch, White Perch, unknown (eggs and-or larvae that were too damaged to identify) or other (indicating another fish species) and a total count (combining both original and QA vials) for each site was recorded, as well as the presence and absence of each of the above species. The four Herring species' eggs and larvae are very similar (Lippson and Moran 1974) and identification to species can be problematic. Quality assurance vials only contained additional eggs and-or larvae of target species already present in the original vials. No new target species were detected during the assessment of the QA vials.

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). This indicator has been provided to us by M. Topolski (MD DNR). Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MDP 2013). 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 MDP's Geographic Information System's (GIS) database. Files were managed and geoprocessed in ArcGIS 9.3.1 from Environmental Systems Research Institute (ESRI 2009). 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. 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 (Uphoff et al. 2012). Mattawoman Creek exhibited a one year downward trend in C/ha of -0.3% between 2011 and 2012, indicating some annual variability is possible that may be due to duplication or omission of records during annual database development. Determination of the exact cause of the trend shifts requires verification of database records and comparison of specific tax records with corresponding parcel maps within suspect sub-watersheds. The time frame for completion of this analysis exceeds that available for completion of this 2014 Federal Aid Report.

Uphoff et al. (2012) developed an equation to convert annual estimates of C/ha to estimates of impervious surface (IS) calculated by Towson University from 1999-2000 satellite imagery. 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 0.27, 0.83, and 1.59 C/ha, respectively (Uphoff et al. 2012).

Mattawoman Creek's watershed equaled 25,168 ha and estimated C/ha was 0.87-0.91 during 2008-2014; Piscataway Creek's watershed equaled 17,999 ha and estimated C/ha was 1.41-1.47 during 2008-2014; and Bush River's watershed equaled 39,644 ha and estimated C/ha was 1.37-1.49 during 2005-2014; (M. Topolski, MD DNR, personal communication). Deer Creek (Figure 1-1), a tributary of the Susquehanna River, was added in 2012 as a spawning stream with low watershed development (watershed area = 37,702 ha and development level = 0.24 C/ha; (M. Topolski, MD DNR, personal communication). It was sampled in 2012-2014 by DNR biologists from the Fishery Management Planning and Fish Passage Program at no charge to this grant.

Conductivity measurements collected for each date and stream site (mainstem and tributaries) during 2008-2014 from Mattawoman Creek were plotted and mainstem measurements were summarized for each year. Unnamed tributaries were excluded from calculation of summary statistics to capture conditions in the largest portion of habitat. Comparisons were made with conductivity minimum and maximum reported for Mattawoman

Creek during 1991 by Hall et al. (1992). Conductivity data were similarly summarized for Piscataway Creek mainstem stations during 2008-2009 and 2012-2014. A subset of Bush River stations that were sampled each year during 2005-2008 and 2014 (i.e., stations in common) were summarized; stations within largely undeveloped Aberdeen Proving Grounds were excluded because they were not sampled every year. Conductivity was measured with each sample in Deer Creek in 2012-2014.

A water quality database maintained by DNR's Tidewater Ecosystem Assessment (TEA) Division (S. Garrison, MD DNR, personal communication) provided conductivity measurements for Mattawoman Creek during 1970-1989. These historical measurements were compared with those collected in 2008-2014 to examine changes in conductivity over time. Monitoring was irregular for many of the historical stations. Table 1-2 summarizes site location, month sampled, total measurements at a site, and what years were sampled. Historical stations and those sampled in 2008-2014 were assigned river kilometers (RKM) using a GIS ruler tool that measured a transect approximating the center of the creek from the mouth of the subestuary to each station location. Stations were categorized as tidal or non-tidal. Conductivity measurements from eight non-tidal sites sampled during 1970-1989 were summarized as monthly medians. These sites bounded Mattawoman Creek from its junction with the estuary to the city of Waldorf (Route 301 crossing), the major urban influence on the watershed. Historical monthly median conductivities at each mainstem Mattawoman Creek non-tidal site were plotted with 2008-2014 spawning season median conductivities.

Presence of White Perch, Yellow Perch, and Herring eggs and-or larvae at each station in 2014 was compared to past surveys to determine which sites still supported spawning. We used the criterion of detection of eggs and-or larvae at a site (O'Dell et al. 1975) as evidence of spawning. Raw data from early 1970s collections were not available to formulate other metrics.

Four Mattawoman Creek mainstem stations sampled in 1971 by O'Dell et al. (1975) were sampled by Hall et al. (1992) during 1989-1991 for water quality and ichthyoplankton. Count data were available for 1991 in a tabular summary at the sample level and these data were converted to presence-absence. Hall et al. (1992) collected ichthyoplankton with 0.5 m diameter plankton nets (3:1 length to opening ratio and 363µ mesh set for 2-5 minutes, depending on flow) suspended in the stream channel between two posts instead of stream drift nets. Changes in spawning site occupation among the current study (2008-2014), 1971 (O'Dell et al. 1975) and 1991 (Hall et al. 1992) were compared to C/ha in Mattawoman Creek. Historical and recent C/ha were compared to site occupation for Piscataway Creek 1971 (O'Dell et al. 1975), 2008-2009, and 2012-2014; Bush River 1973 (O'Dell et al. 1975), 2005-2008 (McGinty et al. 2009; Uphoff et al. 2010) and 2014; and Deer Creek 1972 (O'Dell et al. 1975) and 2012-2014.

The proportion of samples where Herring eggs and-or larvae were present (P_{herr}) was estimated for Mattawoman Creek mainstem stations (MC1-MC4) during 1991 and 2008-2014. Volunteer sampling of ichthyoplankton in Piscataway Creek (2008-2009 and 2012-2014), Bush River (2005-2008 and 2014; McGinty et al. 2009), and Deer Creek (2012-2014) also provided sufficient sample sizes to estimate $P_{herr.}$ Herring was the only species group represented with adequate sample sizes for annual estimates with reasonable precision. Mainstem stations (PC1-PC3) and Tinkers Creek (PTC1) were used in Piscataway Creek (Figure 1-3). Streams that were sampled in all years in Bush River were analyzed (Figure 1-4; see Uphoff et al. 2014 for sites sampled in other years). Deer Creek stations SU01, SU04, and SU05 correspond to O'Dell et al. (1975) sites 1, 2, and 3 respectively (Figure 1-5). Two additional sites, SU02 and SU03 are sampled in this system as well.

For the rivers and stations described above, the proportion of samples with Herring eggs and-or larvae present was estimated as:

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

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

$$^{(2)}\ \mathrm{SD} = [(P_{herr} \bullet (1- P_{herr})) / \mathrm{N}_{\mathrm{total}}]^{0.5} \quad \text{(Ott 1977).}$$

The 90% confidence intervals were constructed as:

$$^{(3)}\ P_{herr} \pm (1.44 \bullet \mathrm{SD}).$$

White Perch and Yellow Perch have been present in samples at the downstream-most one or two stations during 1989-1991 (Hall et al. 1992) and 2008-2014 in Mattawoman Creek. We pooled three to four years (1989-1991, 2008-2010, and 2011-2014) to estimate the proportion of samples with White or Yellow Perch eggs and larvae in order to gain enough precision to separate these estimates from zero. Formulae for estimating proportions were the same as for estimating P_{herr} and its SD and 90% CI's (see above). White Perch spawning occurred at MC1 and MC2. Yellow Perch spawning was only detected at Station MC1.

Regression analyses examined relationships of development (C/ha) with standardized conductivity measurements (median conductivity adjusted for Coastal Plain or Piedmont background level; see below), C/ha and Herring spawning intensity (P_{herr}), and standardized conductivity with P_{herr}. Data were from Bush River and Mattawoman, Piscataway, and Deer Creeks. Twenty estimates of C/ha and P_{herr} were available (1991 estimates for Mattawoman Creek could be included), while nineteen estimates were available for standardized conductivity (Mattawoman Creek data were not available for 1991). Examination of scatter plots suggested that a linear relationship was the obvious choice for C/ha and P_{herr}, but that either linear or curvilinear relationships might be applicable to C/ha with standardized conductivity and standardized conductivity with P_{herr}. Power functions were used to fit curvilinear models:

$$^{(4)}\ \mathrm{Y} = \mathrm{a} \bullet \mathrm{X}^{\mathrm{b}};$$

where Y = dependent variable (standardized conductivity or P_{herr}), X = independent variable (standardized conductivity or C/ha), a is a scaling coefficient and b is a shape parameter. Linear regressions were analyzed in Excel, while the non-linear regression analysis used Proc NLIN (Freund and Littell 2006). A linear or nonlinear model was considered the best descriptor if it was significant at $\alpha < 0.05$ (both were two parameter models), it explained more variability than the other (r^2 for linear and approximate r^2 for nonlinear) and examination of residuals did not suggest a pattern. We expected negative relationships of P_{herr} with C/ha and standardized conductivity, while standardized conductivity and C/ha were expected to be positively related.

Conductivity was summarized as the median for the same stations that were used to estimate P_{herr} and was standardized by dividing by an estimate of the background expected from a stream absent anthropogenic influence (Morgan et al. 2012; see below). Piedmont and Coastal Plain streams in Maryland have different background levels of conductivity (Morgan et al. 2012). Morgan et al. (2012) provided two sets of methods of estimating spring base flow background conductivity for two different sets of Maryland ecoregions, for a total set of four potential background estimates. We chose the option featuring Maryland Biological Stream Survey (MBSS) Coastal Plain and Piedmont regions and the 25th percentile background level for conductivity. These regions had larger sample sizes than the other options and background conductivity in the Coastal Plain fell much closer to the observed range estimated for Mattawoman Creek in 1991 (61-114 μS/cm) when development was relatively low (Hall et al.

1992). Background conductivity used to adjust median conductivities was 109 µS/cm in Coastal Plain streams and 150 µS/cm in Piedmont streams.

Results

Development level of the watersheds of Piscataway, Mattawoman, and Deer Creeks and Bush River started at approximately 0.05 C/ha in 1950, (Figure 1-6). Surveys conducted by O'Dell et al. (1975, 1980) in the 1970s, sampled largely rural watersheds (C/ha < 0.27) except for Piscataway Creek (C/ha = 0.47). By 1991, C/ha in Mattawoman Creek was similar to that of Piscataway in 1971. By the mid-2000s Bush River and Piscataway Creek were at higher suburban levels of development (~1.30 C/ha) than Mattawoman Creek (~0.80 C/ha). Deer Creek, zoned for agriculture and preservation, remained rural through 2014 (0.24 C/ha; Figure 1-6).

In 2014, conductivity measurements in mainstem Mattawoman Creek were highly elevated in March (> 200 µS / cm) and declined for nearly two months before approaching the 1991 maximum (114 µS/cm; Figure 1-7). Two of 12 measurements at MC1 and one measurement each at MC2 and MC3 fell below the 1991 maximum. Conductivity in tributary MUT3 was elevated above the 1991 maximum for four of 12 measurements, which has not been observed since 2009. Conductivity values in tributaries MUT4 and MUT5 all fell within or below the range reported by Hall et al. (1992) for the mainstem. Conductivities in Mattawoman Creek's mainstem stations in 2009 were highly elevated in early March following application of road salt in response to a significant snowfall that occurred just prior to the start of the survey (Uphoff et al. 2010). Measurements during 2009 steadily declined for nearly a month before leveling off slightly above the 1989-1991 maximum. 2014 also had higher snowfall than the previous four years and a conductivity pattern similar to 2009 (Figure 1-7). Higher conductivity at the most upstream mainstem site (MC4) followed by declining conductivity downstream to the site on the tidal border is a general pattern in all years. This, along with low conductivities typically seen at the unnamed tributaries, indicates that development at and above MC4 is affecting water quality (Figure 1-7).

Conductivity levels in Piscataway Creek and Bush River were elevated when compared to Mattawoman Creek (Table 1-3). With the exception of Piscataway Creek in 2012 (median = 195 µS/cm), median conductivity estimates during spawning surveys were always greater than 200 µS/cm in Piscataway Creek and Bush River during the 2000s. Median conductivity in Mattawoman Creek was in excess of 200 µS/cm during 2009 and was less than 155 µS/cm during the next five years, with median conductivity in 2014 approaching 166 µS/cm (Table 1-3).

During 1970-1989, 73% of monthly median conductivity estimates in Mattawoman Creek were at or below the background level for Coastal Plain streams; C/ha in the watershed increased from 0.25 to 0.41. Higher monthly median conductivities in the non-tidal stream were more frequent nearest the confluence with Mattawoman Creek's estuary and in the vicinity of Waldorf (RKM 35) (Figure 1-8). Conductivity medians were highly variable at the upstream station nearest Waldorf during 1970-1989. During 2008-2014 (C/ha = 0.85-0.91), median spawning survey conductivities at mainstem stations MC2 to MC4, above the confluence of Mattawoman Creek's stream and estuary (MC1), were elevated beyond nearly all 1979-1989 monthly medians and increased with upstream distance toward Waldorf. Most measurements at MC1 fell within the upper half of the range observed during 1970-1989 (Figure 1-8). None of the non-tidal conductivity medians estimated at any site during 2008-2014 were at or below the Coastal Plain stream background criterion.

Anadromous fish spawning site occupation in fluvial Mattawoman Creek improved during 2008-2013 but was less consistent than during 1971 and 1989-1991 (historical spawning period), while 2014 had site occupations at historical levels (Table 1-4). Herring spawning was detected during 2008-2014 at historical mainstem stations, but was absent at stations MC2, MC4, and MUT3 during 2008-2009. Site occupation has increased since 2009 and all four mainstem stations had Herring eggs and-or larvae during 2010-2014. Herring spawning was detected at MUT3 in 2011-2014, at MUT4 in 2012, and at MUT5 in 2014. Herring spawning was detected at all mainstem stations in 1971 and 1991. Stream spawning of White Perch in Mattawoman Creek was not detected during 2009, 2011, and 2012, but spawning was detected at MC1 during 2008 and 2010, and at MC1 and MC2 during 2013 and 2014. During 1971 and 1989-1991, White Perch spawning occurred annually at MC1 and intermittently at MC2. Prior to 2008-2014, MC3 was sampled in 1971 and 1991 and White Perch were only present during 1971. Yellow Perch spawning occurred at station MC1 every year except 2009 and 2012. Station MC1 was the only stream station in Mattawoman Creek where Yellow Perch spawning has been detected in surveys conducted since 1971 (Table 1-4).

Herring spawning was detected at all mainstem sites in Piscataway Creek in 2012-2014. Stream spawning of anadromous fish had nearly ceased in Piscataway Creek between 1971 and 2008-2009 (Table 1-5). Herring spawning was not detected at any site in the Piscataway Creek drainage during 2008 and was only detected on one date and location (one Herring larvae on April 28 at PC2) in 2009. Stream spawning of White Perch was detected at PC1 and PC2 in 1971, was not detected during 2008-2009 and 2012-2013, but was detected at PC1 in 2014 (Table 1-5).

Changes in stream site spawning of Herring, White Perch, and Yellow Perch in the Bush River stations during 1973, 2005-2008, and 2014 were not obvious (Table 1-6). Occurrences of White and Yellow Perch eggs and larvae were far less frequent during 2005-2008 than 1973 and 2014. Herring eggs and larvae were present at three to five stations (not necessarily the same ones) in any given year sampled (Table 1-6).

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

The 90% confidence intervals of P_{herr} (Figure 1-9) provided sufficient precision for us to categorize four levels of stream spawning: very low levels at or indistinguishable from zero based on confidence interval overlap (level 0); a low level of spawning that could be distinguished from zero (level 1); a mid-level of spawning that could usually be separated from the low levels (level 2); and a high level (3) of spawning likely to be higher than the mid-level. Stream spawning in Mattawoman Creek was categorized at levels 1 (2008-2009), 2 (2010 and 2012), and level 3 (1991, 2011, and 2013-2014). Spawning in Piscataway Creek was at level 0 during 2008-2009, at level 2 during 2012, and at level 1 during 2013-2014. Bush River spawning was characterized by levels 0 (2006), 1 (2005 and 2007-2008), and level 2 (2014). Deer Creek, with the least developed watershed, was characterized by the highest level of spawning (level 3) during 2012-2014 (Figure 1-9).

The 90% CI's of proportions of samples with White Perch eggs and larvae at stations MC1 and MC2, pooled in 3-to-4-year intervals, indicated that less stream spawning occurred in Mattawoman Creek during 2008-2010 than during 1989-1991 (Figure 1-10). Status of spawning during 2011-2014 was not clear since 90% CI's of the proportion of samples with White Perch eggs and larvae during 2011-2014 overlapped both 1989-1991 and 2008-2010. The 90% CI's for stream spawning of Yellow Perch (at MC1 only) overlapped for 1989-1991, 2008-2010, and 2011-2014, indicating significant change in stream spawning had not been detected (Figure 1-10).

Standardized conductivity increased with development, while P_{herr} declined with both development and standardized conductivity. Regression analyses indicated significant and logical relationships among $P_{herr,}$ C/ha, and standardized median conductivity (Table 1-8). The relationship of C/ha with standardized median conductivity was linear, significant, and positive ($r^2 = 0.45$, $P = 0.001$, $N = 20$; Figure 1-11). Estimates of P_{herr} were linearly, significantly, and negatively related to C/ha ($r^2 = 0.54$, $P = 0.0002$, $N = 21$). A negative curvilinear regression best described the relationship of P_{herr} and standardized median conductivity (approximate $r^2 = 0.35$, $P < 0.0001$, $N = 20$; Figure 1-12). Low estimates of P_{herr} were much more frequent beyond the C/ha threshold (0.83 C/ha) or when standardized conductivity was 1.5-times or more than the baseline level (Figure 1-12).

Discussion

Proportion of samples with Herring eggs and-or larvae (P_{herr}) provided a reasonably precise estimate of relative abundance based on encounter rate. Regression analyses indicated significant and logical relationships among $P_{herr,}$ C/ha, and conductivity consistent with the hypothesis that urbanization was detrimental to stream spawning. Conductivity was positively related with C/ha in our analysis and with urbanization in other studies (Wang and Yin 1997; Paul and Meyer 2001; Wenner et al. 2003; Morgan et al. 2007; Carlisle et al. 2010; Morgan et al. 2012). Limburg and Schmidt (1990) found a highly nonlinear relationship of densities of anadromous fish (mostly Alewife) eggs and larvae to urbanization in Hudson River tributaries with a strong negative threshold at low levels of development.

An unavoidable assumption of regression analyses of P_{herr}, C/ha, and summarized conductivity was that watersheds at different levels of development were a substitute for time-series. Extended time-series of watershed specific data were not available. Mixing physiographic provinces in this analysis had the potential to increase scatter of points, but standardizing median conductivity to background conductivity moderated the province effect in analyses with that variable. Differential changes in physical stream habitat and flow due to differences in geographic provinces could also have affected fits of regressions. Estimates of C/ha would have indexed these physical changes as well as water chemistry changes, while standardized conductivity would only have represented changes in water chemistry. Estimates of C/ha explained more variation in P_{herr} (54%) than standardized conductivity (35%).

Elevated conductivity, related primarily to chloride from road salt (but including most inorganic acids and bases; APHA 1979), has emerged as an indicator of watershed development (Wenner et al. 2003; Kaushal 2005; Morgan et al. 2007; Morgan et al. 2012). Use of salt as a deicer may lead to both "shock loads" of salt that may be acutely toxic to freshwater biota and elevated baselines (increased average concentrations) of chloride that have been associated with decreased fish and benthic diversity (Kaushal et al. 2005; Wheeler et al. 2005; Morgan et al. 2007; 2012). Commonly used anti-clumping agents for road salt (ferro- and ferricyanide) that are not thought to be directly toxic are of concern because they can break down into toxic cyanide under exposure to ultraviolet light. Although the degree of breakdown into cyanide in nature is unclear (Pablo et al. 1996; Transportation Research Board 2007), these compounds have been implicated in fish kills (Burdick and Lipschuetz 1950; Pablo et al. 1996; Transportation Research Board 2007). Heavy metals and phosphorous may also be associated with road salt (Transportation Research Board 2007).

At least two hypotheses can be formed to relate decreased anadromous fish spawning to conductivity and road salt use. First, eggs and larvae may die in response to sudden changes in salinity and potentially toxic amounts of associated contaminants and additives. Second, changing stream chemistry may cause disorientation and disrupted upstream migration.

Levels of salinity associated with our conductivity measurements are very low (maximum 0.1 ppt) and anadromous fish spawn successfully in brackish water (Klauda et al. 1991; Piavis et al. 1991; Setzler-Hamilton 1991). A rapid increase might result in osmotic stress and lower survival since salinity represents osmotic cost for fish eggs and larvae (Research Council of Norway 2009).

Elevated stream conductivity may prevent anadromous fish from recognizing and ascending streams. Alewife and Herring are thought to home to natal rivers to spawn (ASMFC 2009a; ASMFC 2009b), while Yellow and White Perch populations are generally tributary-specific (Setzler-Hamilton 1991; Yellow Perch Workgroup 2002). Physiological details of spawning migration are not well described for our target species, but homing migrations in anadromous American Shad and Salmon have been connected with chemical composition, smell, and pH of spawning streams (Royce-Malmgren and Watson 1987; Dittman and Quinn 1996; Carruth et al. 2002; Leggett 2004). Conductivity is related to total dissolved solids in water (Cole 1975) which reflects chemical composition.

Processes such as flooding, riverbank erosion, and landslides vary by geographic province (Cleaves 2003) and influence physical characteristics of streams. Unconsolidated sediments (layers of sand, silt, and clay) underlie the Coastal Plain and broad plains of low relief and wetlands characterize the terrain (Cleaves 2003). Coastal Plain streams have low flows and sand or gravel bottoms (Boward et al. 1999). The Piedmont is underlain by metamorphic rocks and characterized by narrow valleys and steep slopes, with regions of higher land between streams in the same drainage. Most Piedmont streams are of moderate slope with rock or bedrock bottoms (Boward et al. 1999). The Piedmont is an area of higher gradient change and more diverse and larger substrates than the Coastal Plain (Harris and Hightower 2011) that may offer greater variety of Herring spawning habitats.

Urbanization and physiographic province both affect discharge and sediment supply of streams (Paul and Meyer 2001; Cleaves 2003) that, in turn, could affect location, substrate composition, extent and success of spawning. Alewife spawn in sluggish flows, while Blueback Herring spawn in sluggish to swift flows (Pardue 1983). American Shad select spawning habitat based on macrohabitat features (Harris and Hightower 2011) and spawn in moderate to swift flows (Hightower and Sparks 2003). Spawning substrates for Herring include gravel, sand, and detritus (Pardue 1983). Detritus loads in subestuaries are strongly associated with development (see Section 2) and urbanization affects the quality and quantity of organic matter in streams (Paul and Meyer 2001) that feed into subestuaries.

Herring spawning became more variable in streams as watersheds developed. The surveys from watersheds with C/ha of 0.46 or less had high P_{herr}. Estimates of P_{herr} from Mattawoman Creek during 2008-2014 (C/ha was 0.85-0.91) varied from barely different from zero to high. Eggs and larvae were nearly absent from fluvial Piscataway Creek during 2008-2009, but P_{herr} rebounded to 0.45 in 2012 and then dropped again to 0.2 in 2013-2014 (C/ha was 1.39-1.47). The rebound in Herring spawning in Piscataway Creek during 2012 was concurrent with the lowest mean and median conductivities encountered there in the four years sampled. Variability of Herring spawning in Bush River during 2005-2008 and 2014 involved "colonization" of new sites as well as absence from sites of historical spawning (Uphoff et al. 2014).

Magnitude of P_{herr} may indicate how much habitat is available or how attractive it is from year to year more-so than indicating abundance of spawners. In developed watersheds, a combination of urban and natural stream processes may create varying amounts of ephemeral spawning habitat annually and dampen spawning migrations through increased conductivity. Observed variation in P_{herr} would indicate wide annual and regional fluctuations in population size. However, stock assessments of Alewife and Blueback Herring along the Atlantic coast, including those in Maryland, indicate they are in decline or are at depressed stable levels (ASMFC 2009a; 2009b; Limburg and Waldman 2009; Maryland Fisheries Service 2012) rather than fluctuating.

Application of presence-absence data in management needs to consider whether absence reflects a disappearance from suitable habitat or whether habitat sampled is not really habitat for the species in question (MacKenzie 2005). Our site occupation comparisons were based on the assumption that spawning sites detected in the 1970s were indicative of the extent of habitat. O'Dell et al. (1975) summarized spawning activity as the presence of any species group's egg, larva, or adult (latter from wire fish trap sampling) at a site and we used this criterion (spawning detected at a site or not) for a set of comparisons. Raw data for the 1970s were not available to formulate other metrics. This presence-absence approach did not detect permanent site occupation changes or an absence of change since only a small number of sites could be sampled (limited by road crossings) and the positive statistical effect of repeated visits (Strayer 1999) was lost by summarizing all samples into a single record of occurrence in a sampling season. A single year's record was available for each of the watersheds in the 1970s and we were left assuming this distribution applied over multiple years of low development.

Proportion of positive samples (P_{herr} for example) provided an economical and precise alternative estimate of relative abundance based on encounter rate rather than counts. Encounter rate is readily related to the probability of detecting a population (Strayer 1999). Proportions of positive or zero catch indices were found to be robust indicators of abundance of Yellowtail Snapper *Ocyurus chrysurus* (Bannerot and Austin 1983), age-0 White Sturgeon *Acipenser transmontanus* (Counihan et al. 1999), Pacific Sardine *Sardinops sagax* eggs (Mangel and Smith 1990), Chesapeake Bay Striped Bass eggs (Uphoff 1997), and Longfin Inshore Squid *Loligo pealeii* fishery performance (Lange 1991).

Unfortunately, estimating reasonably precise proportions of stream samples with White or Yellow Perch eggs annually will not be logistically feasible without major changes in sampling priorities. Estimates for Yellow or White Perch stream spawning would require more frequent sampling to obtain precision similar to that attained by P_{herr} since spawning occurred at fewer sites. Given staff and volunteer time limitations, this would not be possible within our current scope of operations. In Mattawoman Creek, it was, however, possible to pool data across years to form estimates of proportions of samples with White Perch eggs and larvae (sites MC1 and MC2) or Yellow Perch larvae (MC1) for 1989-1991 collections to compare with 2008-2014 collections at the same combinations of sites. These estimates did not indicate a loss in stream spawning in these downstream sites.

Volunteer-based sampling of stream spawning during 2005-2014 used only stream drift nets, while O'Dell et al. (1975) and Hall et al. (1992) determined spawning activity with ichthyoplankton nets and wire traps for adults. Tabular summaries of egg, larval, and adult catches in Hall et al. (1992) allowed for a comparison of how site use in Mattawoman Creek might have varied in 1991 with and without adult wire trap sampling. Sites estimated when eggs and-or larvae were present in one or more samples were identical to those when adults present in wire traps were included with the ichthyoplankton data (Hall et al. 1992). Similar results were obtained from the Bush River during 2006 at sites where ichthyoplankton drift nets and wire traps were used; adults were captured by traps at one site and eggs and-or larvae at nine sites with ichthyoplankton nets (Uphoff et al. 2007). Wire traps set in the Bush River during 2007 did not indicate different results than ichthyoplankton sampling for Herring and Yellow Perch, but White Perch adults were observed in two trap samples and not in plankton drift nets (Uphoff et al. 2008). These comparisons of trap and ichthyoplankton sampling indicated it was unlikely that an absence of adult wire trap sampling would impact interpretation of spawning sites when multiple years of data were available.

The different method used to collect ichthyoplankton in Mattawoman Creek during 1991 could bias that estimate of P_{herr}, although presence-absence data tend to be robust to errors and biases in sampling (Green 1979). Removal of 1991 data lowered the fit between C/ha and P_{herr} (from $r^2 = 0.54$, P = 0.0002 to $r^2 = 0.50$, P = 0.0005), but did not alter the negative relationship (95% CI's of slopes and intercepts of both models overlapped).

Absence of detectable stream spawning does not necessarily indicate an absence of spawning in the estuarine portion of these systems. Estuarine Yellow Perch presence-absence surveys in Mattawoman and Piscataway Creeks, and Bush River did not indicate that lack of detectable stream spawning corresponded to their elimination from these subestuaries. Yellow Perch larvae were present in upper reaches of both subestuaries (see Section 2). Yellow Perch do not appear to be dependent on non-tidal stream spawning, but their use may confer benefit to the population through expanded spawning habitat diversity. Stream spawning is very important to Yellow Perch anglers since it provides access for shore fisherman and most recreational harvest probably occurs during spawning season (Yellow Perch Workgroup 2002).

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

Subestuary	Year	Number of Sites	1st Sampling Date	Last Sampling Date	Number of Dates	N
Bush	2005	13	18-Mar	15-May	16	99
Bush	2006	13	18-Mar	15-May	20	114
Bush	2007	14	21-Mar	13-May	17	83
Bush	2008	12	22-Mar	26-Apr	17	77
Bush	2014	6	22-Mar	1-Jun	10	60
Piscataway	2008	5	17-Mar	4-May	8	39
Piscataway	2009	6	9-Mar	14-May	11	60
Piscataway	2012	5	5-Mar	16-May	11	55
Piscataway	2013	5	11-Mar	28-May	11	55
Piscataway	2014	5	10-Mar	1-Jun	9	45
Mattawoman	2008	9	8-Mar	9-May	10	90
Mattawoman	2009	9	8-Mar	11-May	10	70
Mattawoman	2010	7	7-Mar	15-May	11	75
Mattawoman	2011	7	5-Mar	15-May	14	73
Mattawoman	2012	7	4-Mar	13-May	11	75
Mattawoman	2013	7	10-Mar	25-May	12	80
Mattawoman	2014	8	9-Mar	25-May	12	87
Deer	2012	4	20-Mar	7-May	11	44
Deer	2013	5	19-Mar	23-May	19	87
Deer	2014	5	2-Apr	28-May	12	60

Table 1-2. Summary of historical conductivity sampling in non-tidal Mattawoman Creek. RKM = site location in river kilometers from the mouth; Months = months when samples were drawn; Sum = sum of samples for all years.

RKM	Months	Sum	Years Sampled
12.4	1 to 12	218	1971, 1974-1989
18.1	4 to 9	8	1974
27	4 to 9	9	1970, 1974
30	8 and 9	2	1970
34.9	4 to 9	9	1970, 1974
38.8	8 and 9	2	1970

Table 1-3. Summary statistics of conductivity (μS/cm) for mainstem stations in Mattawoman, Piscataway, and Deer Creeks, and Bush River during 2005-2014. Unnamed tributaries were excluded from analysis. Tinkers Creek was included with mainstem stations in Piscataway Creek.

	Year									
Conductivity	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014
Mattawoman										
Mean				120.1	244.5	153.7	147.5	128.9	126.1	179.4
Standard Error				3.8	19.2	38	2.8	1.9	2.4	9.1
Median				124.6	211	152.3	147.3	130.9	126.5	165.8
Kurtosis				2.1	1.41	1.3	8.29	-0.26	5.01	0.33
Skewness				-1.41	1.37	0.03	1.72	-0.67	-1.70	1.00
Range				102	495	111	117	49	96	261
Minimum				47	115	99	109	102	63	88
Maximum				148	610	210	225	151	158	350
Count				39	40	43	44	44	48	48
Bush										
Mean	269	206	263	237						276.7
Standard Error	25	5	16	6						15
Median	230	208	219	234						253.4
Kurtosis	38	2	22	7						3.16
Skewness	6	-1	4	0						1.56
Range	1861	321	1083	425						606
Minimum	79	0	105	10						107
Maximum	1940	321	1187	435						713
Count	81	106	79	77						60
Piscataway										
Mean				218.4	305.4			211.4	245	249.4
Standard Error				7.4	19.4			5.9	6.9	11.1
Median				210.4	260.6			195.1	238.4	230
Kurtosis				-0.38	1.85			0.11	-0.29	2.56
Skewness				0.75	1.32			0.92	0.73	1.50
Range				138	641			163	173	274
Minimum				163	97			145	181	174
Maximum				301	737			308	354	449
Count				29	50			44	44	36
Deer										
Mean								174.9	175.6	170.3
Standard Error								1.02	1.5	1.4
Median								176.8	177.7	171.7
Kurtosis								17.22	13.88	9.21
Skewness								-3.78	-2.25	-2.42
Range								39.3	122	66
Minimum								140.2	93	116
Maximum								179.5	215	183
Count								44	87	60

Table 1-4. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Mattawoman Creek during 1971, 1989-1991, and 2008-2014. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-2.

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

Table 1-5. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife) and White Perch stream spawning in Piscataway Creek during 1971, 2008-2009, and 2012-2014. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-3.

	Year					
Station	1971	2008	2009	2012	2013	2014
Herring						
PC1	1	0	0	1	1	1
PC2	1	0	1	1	1	1
PC3	1	0	0	1	1	1
PTC1	1	0	0	1	1	0
PUT4	1		0	0	0	0
White Perch						
PC1	1	0	0	0	0	1
PC2	1	0	0	0	0	0

Table 1-6. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Bush River during 1973, 2005-2008, and 2014. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-4.

	Year					
Station	1973	2005	2006	2007	2008	2014
			Herring			
BBR1	0	1	1	1	1	1
BCR1	1	0	0	1	0	1
BHH1	0	0	1	1	1	1
BJR1	0	1	1	1	0	1
BOP1	1	1	1	1	1	1
BWR1	1	0	0	1	0	1
			White Perch			
BBR1	1	0	0	0	0	1
BCR1	1	0	0	0	0	1
BHH1	0	0	0	0	0	0
BJR1	0	0	0	0	0	0
BOP1	1	0	0	1	0	1
BWR1	1	0	0	0	0	0
			Yellow Perch			
BBR1	1	0	0	0	0	0
BCR1	0	0	0	0	0	1
BHH1	0	0	0	0	0	1
BJR1	1	0	0	0	0	1
BOP1	0	0	0	0	0	0
BWR1	1	0	1	0	0	0

Table 1-7. Presence-absence of Herring (Blueback Herring, Hickory and American Shad, and Alewife), White Perch, and Yellow Perch stream spawning in Deer Creek during 1972 and 2012-2014. 0 = site sampled, but spawning not detected; 1 = site sampled, spawning detected; and blank indicates no sample. Station locations are identified on Figure 1-5.

	Year			
Station	1972	2012	2013	2014
		Herring		
SU01	1	1	1	1
SU02		1	1	1
SU03		1	1	1
SU04	1	1	1	1
SU05	0		1	1
		White Perch		
SU01	1	0	1	1
SU02		0	1	0
SU03		0	0	1
SU04	0	0	1	1
SU05	0		0	0
		Yellow Perch		
SU01	1	1	0	1
SU02		1	0	1
SU03		0	0	1
SU04	0	0	0	0
SU05	0		0	0

Table 1-8. Summary of best regression models for standardized conductivity (annual median/province background) versus development level (C/ha), proportion of samples with herring eggs or larvae (P_{herr}) versus C/ha, and P_{herr} versus standardized conductivity.

Linear Model	Standardized conductivity = Structure density (C/ha)					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	1.23337	1.23337	14.47	0.0013	
Residual	18	1.53394	0.08522			
Total	19	2.7673				
r^2 = 0.4457						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.94282	0.17555	5.37	<.0001	0.57401	1.31164
C / ha	0.57973	0.15239	3.8	0.0013	0.25958	0.89989

Linear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Structure density (C/ha)					
ANOVA	df	SS	MS	F	Significance F	
Regression	1	0.76954	0.76954	21.91	0.0002	
Residual	19	0.66747	0.03513			
Total	20	1.43701				
r^2 = 0.5355						
	Estimate	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.81857	0.10547	7.76	<.0001	0.59782	1.03931
C / ha	-0.43734	0.09344	-4.68	0.0002	-0.63292	-0.24176

Nonlinear Model	Proportion of samples with herring eggs or larvae (P_{herr}) = Standardized Conductivity				
Source	df	SS	MS	F	P
Model	2	2.8761	1.4381	30.08	<.0001
Error	18	0.8607	0.0478		
Uncorrected Total	20	3.7368			
Approximate r^2 = 0.3533					
Parameter	Estimate	Approximate SE	Lower 95%	Upper 95%	
a	0.7515	0.1868	0.3591	1.1439	
b	-2.0778	0.8212	-3.8031	-0.3525	

Figure 1-1. Watersheds sampled for stream spawning anadromous fish eggs and larvae in 2005-2014. Coastal Plain and Piedmont Regions are indicated.

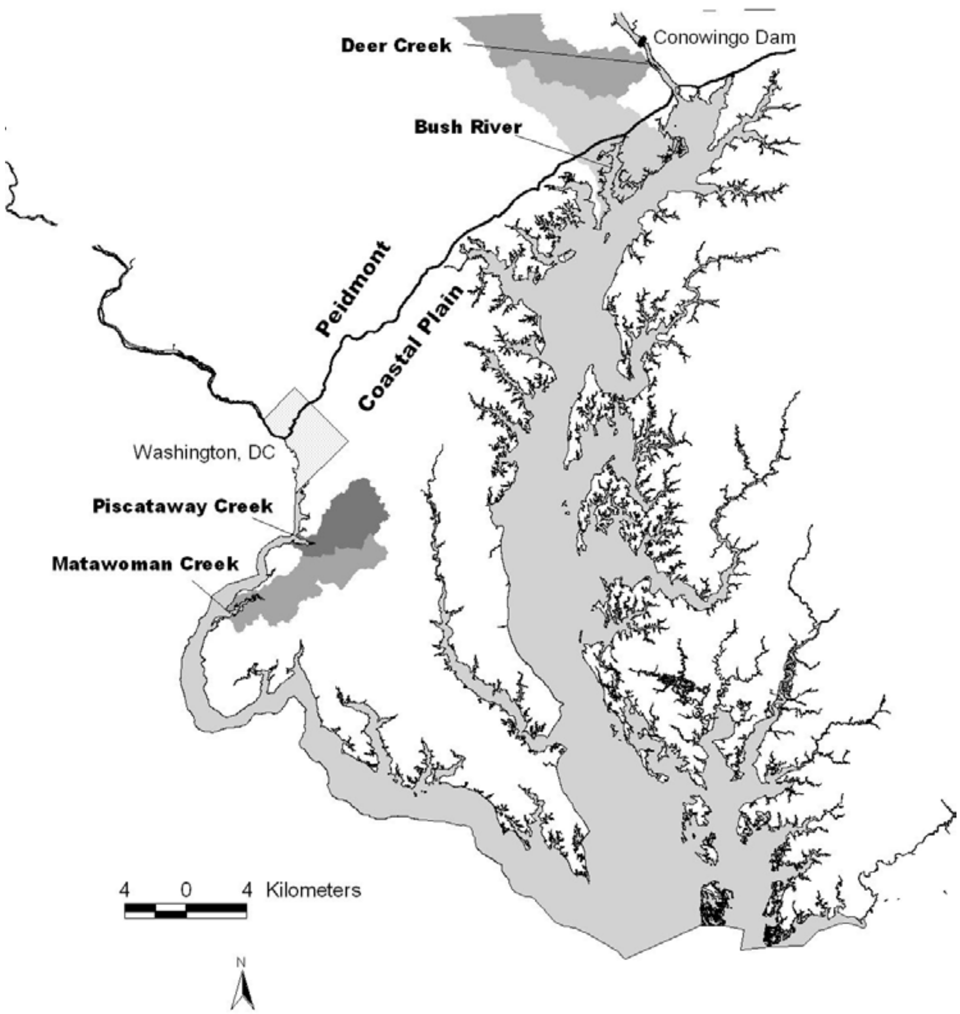


Figure 1-2. Mattawoman Creek's 1971 and 2008-2014 sampling stations.

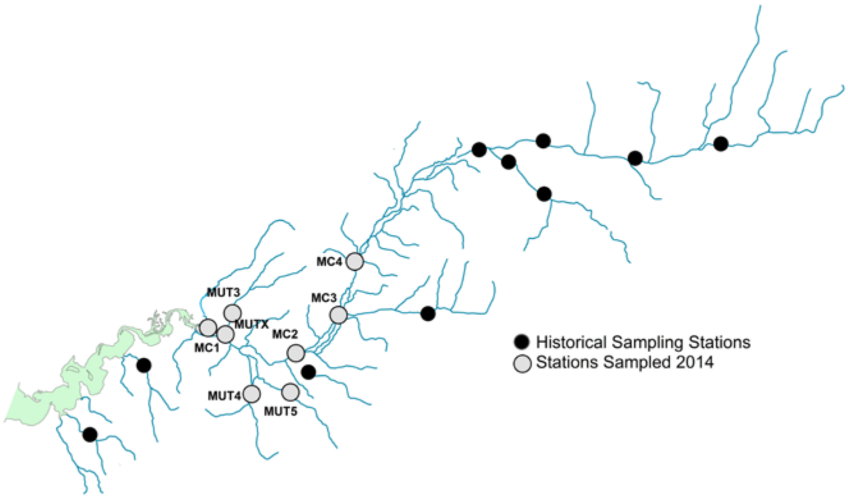


Figure 1-3. Piscataway Creek's 1971, 2008-2009, and 2012-2014 sampling stations.

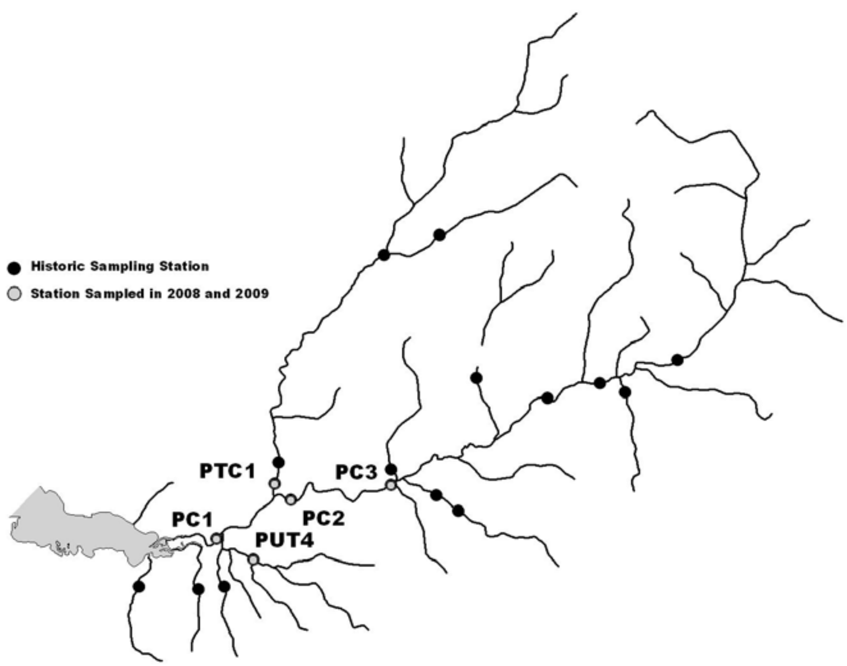


Figure 1-4. Bush River's 1973, 2005-2008, and 2014 sampling stations. Stations in Aberdeen Proving Grounds have been separated from other Bush River stations.

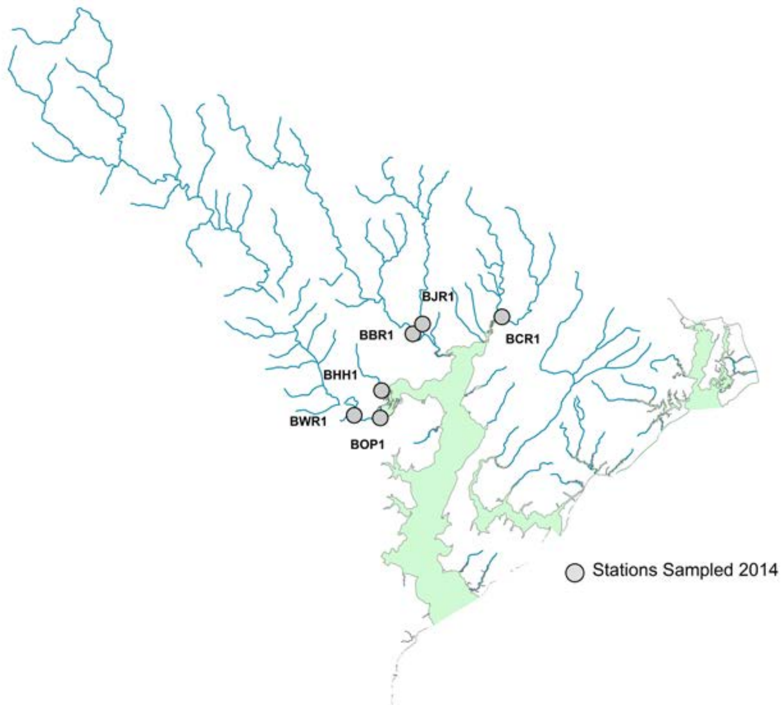


Figure 1-5. Deer Creek's 1972 and 2012-2014 sampling stations.

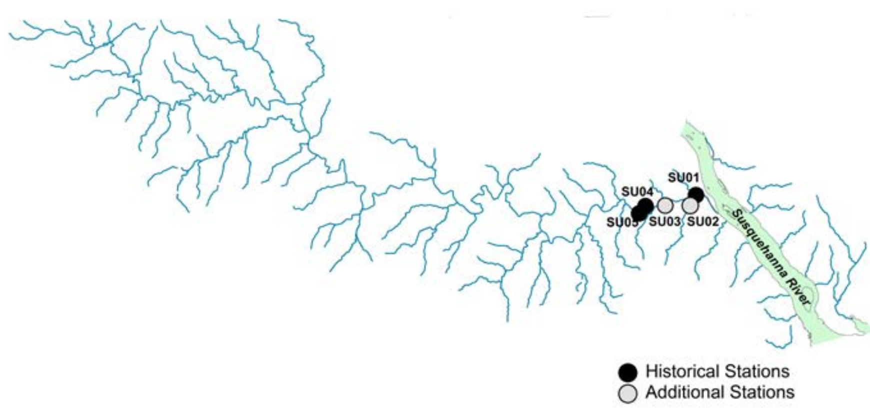


Figure 1-6. Trends in counts of structures per hectare (C / ha) during 1950-2014 in Piscataway Creek, Mattawoman Creek, Deer Creek, and Bush River watersheds. Updated estimates of C / ha were not available for 2013 or 2014. Large symbols indicate years when stream ichthyoplankton was sampled.

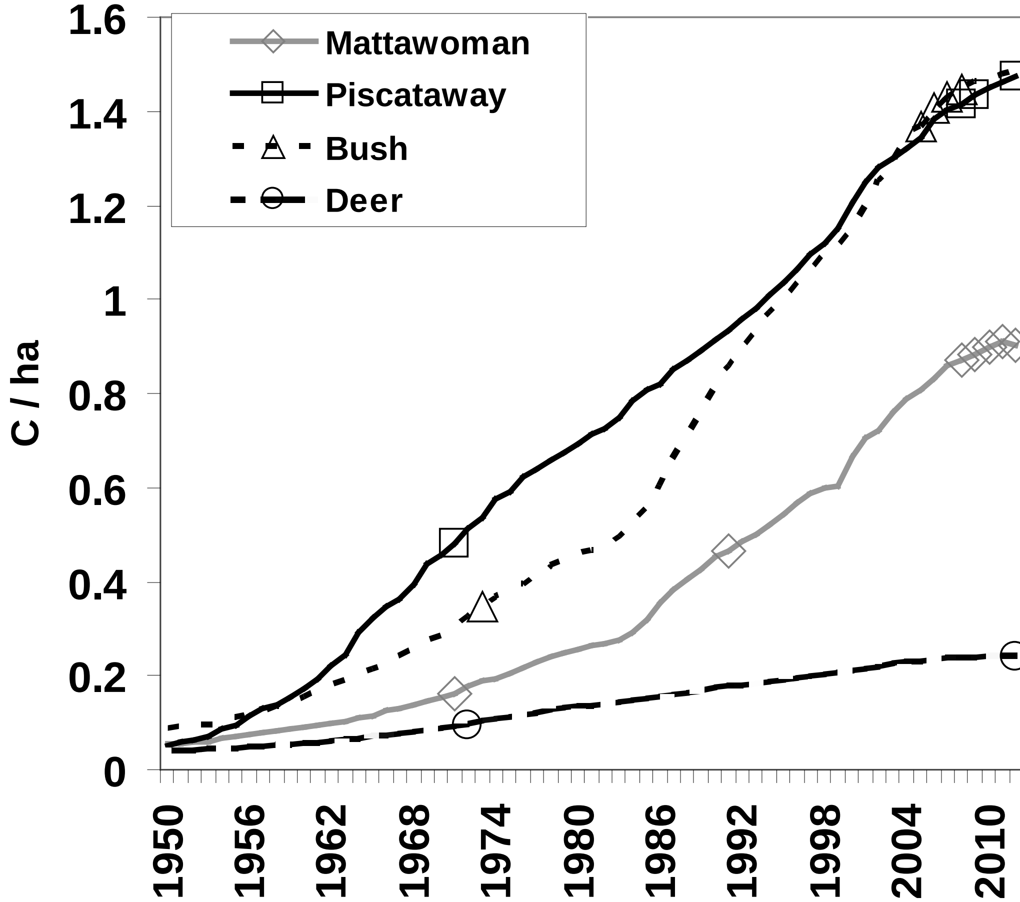


Figure 1-7. Stream conductivity measurements (µS / cm), by station and date, in Mattawoman Creek during (A) 2009, (B) 2010, (C) 2011, (D) 2012, (E) 2013, and (F) 2014. Lines indicate conductivity range measured at mainstem sites (MC1 – MC4) during 1991 by Hall et al. (1992).

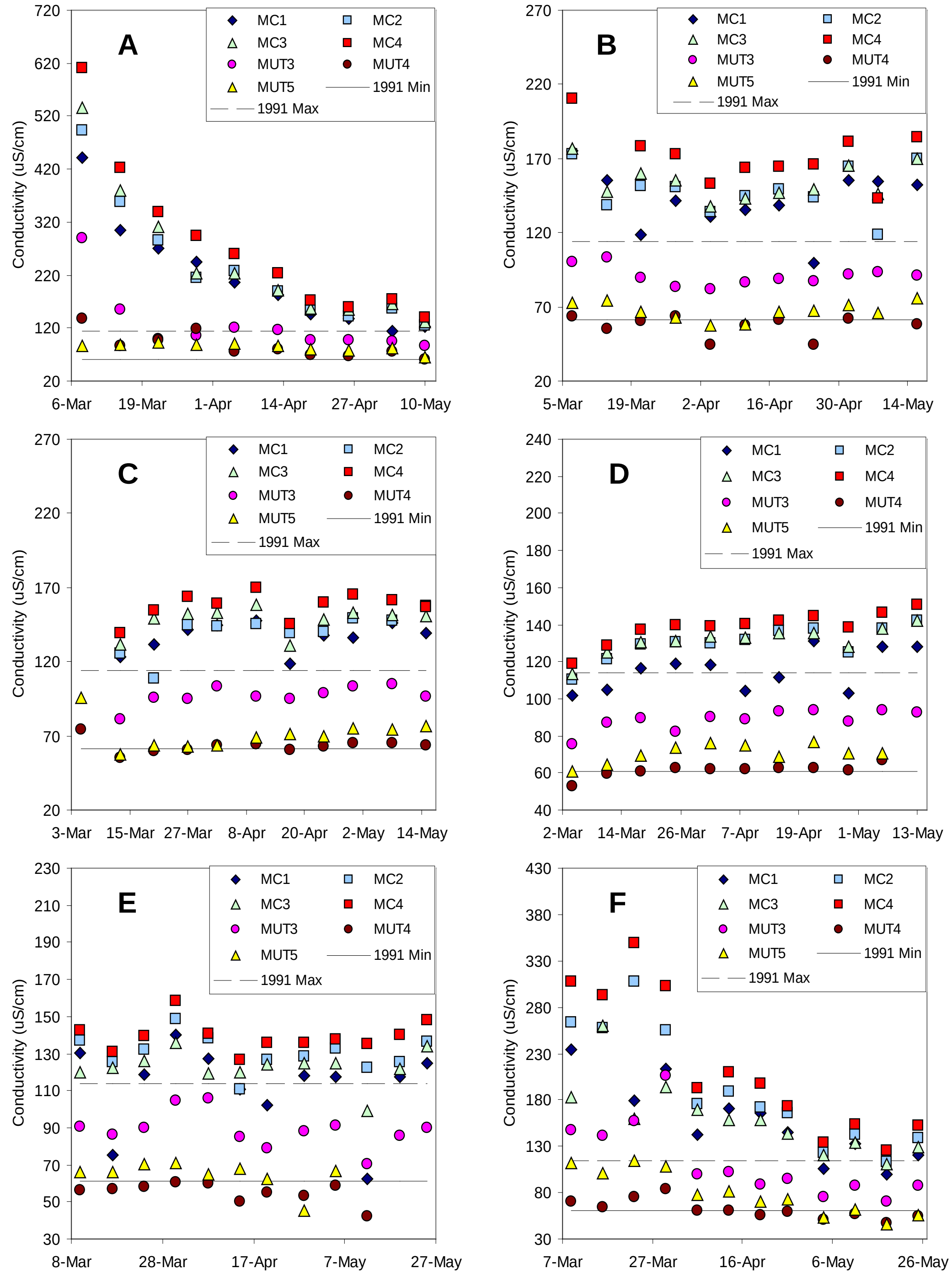


Figure 1-8. Historical (1970-1989) median conductivity measurements and current (2008-2014) anadromous spawning survey median conductivity in non-tidal Mattawoman Creek (between the junction with the subestuary and Waldorf) plotted against distance from the mouth. The two stations furthest upstream are nearest Waldorf. Median conductivity was measured during March-May, 2008-2014, and varying time periods (see Table 1-2) during 1970-1989.

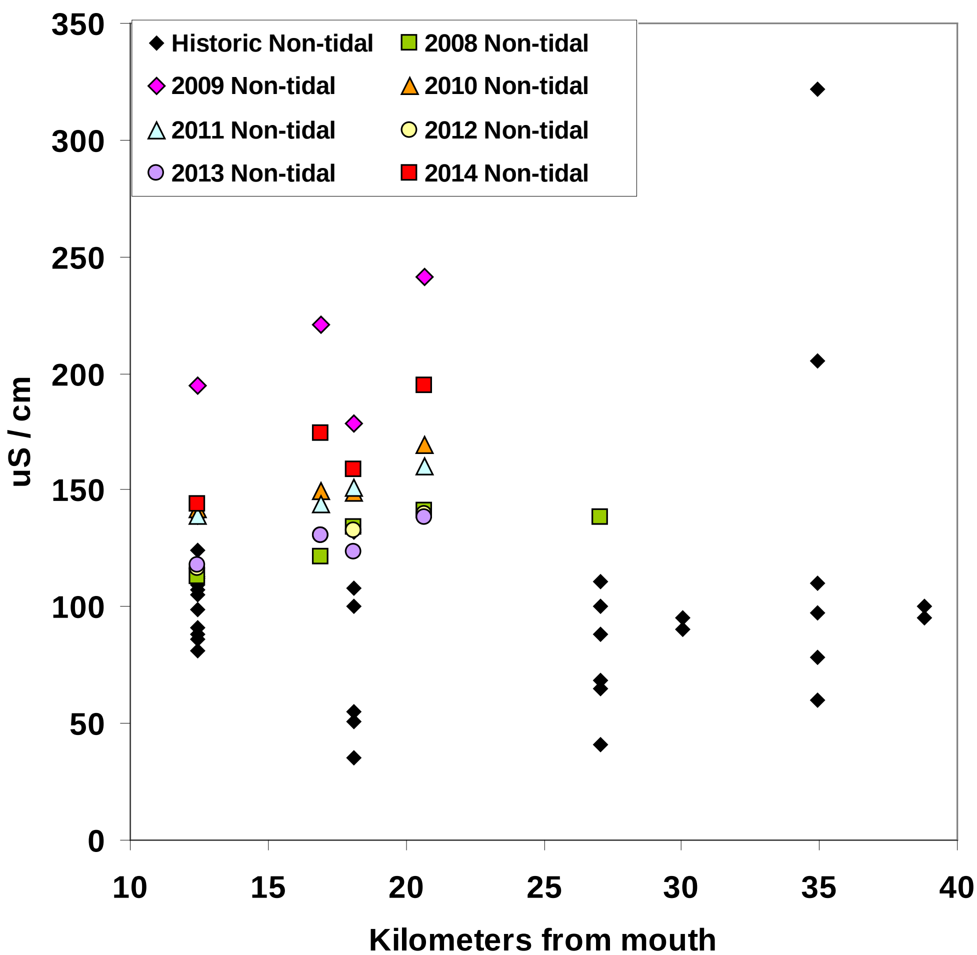


Figure 1-9. Proportion of samples (P_{herr}) with Herring and its 90% confidence interval for stream ichthyoplankton surveys in Mattawoman, Piscataway and Deer Creeks, and Bush River.

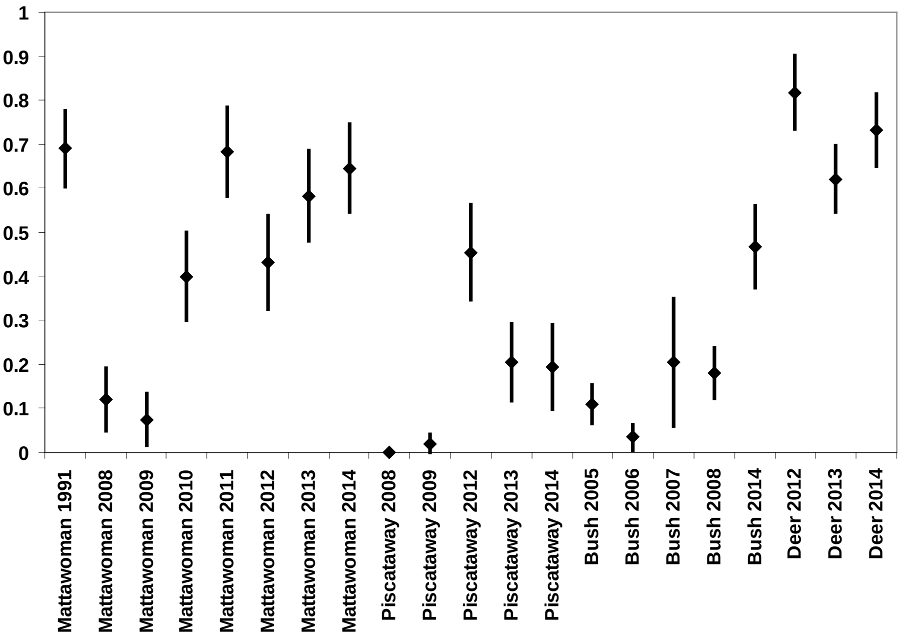


Figure 1-10. Mattawoman data pooled across years to form estimates of proportions of samples with White Perch (WP) eggs and-or larvae (sites MC1 and MC2) or Yellow Perch (YP) eggs and-or larvae (MC1) for 1989-1991 collections compared to 2008-2010 and 2011-2014 collections at the same combination of sites.

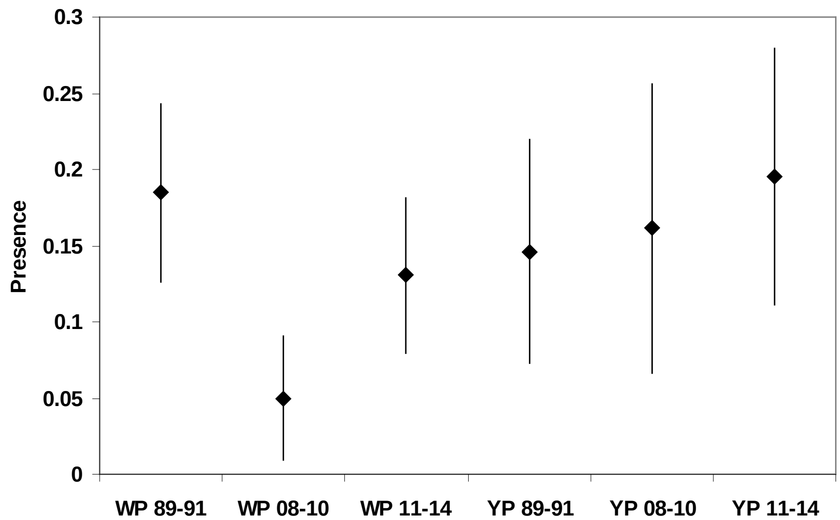


Figure 1-11. Standardized median conductivity during spring spawning surveys and level of development (C / ha). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).

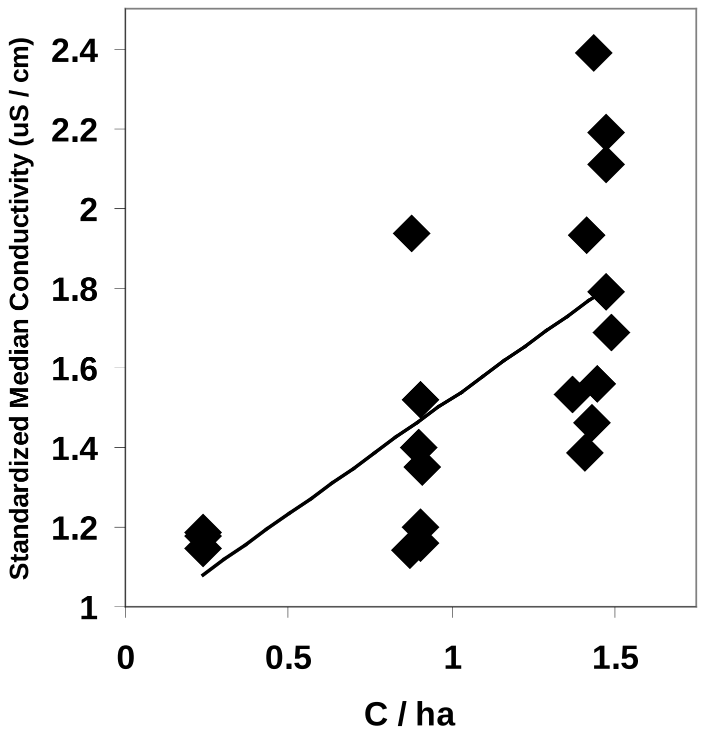
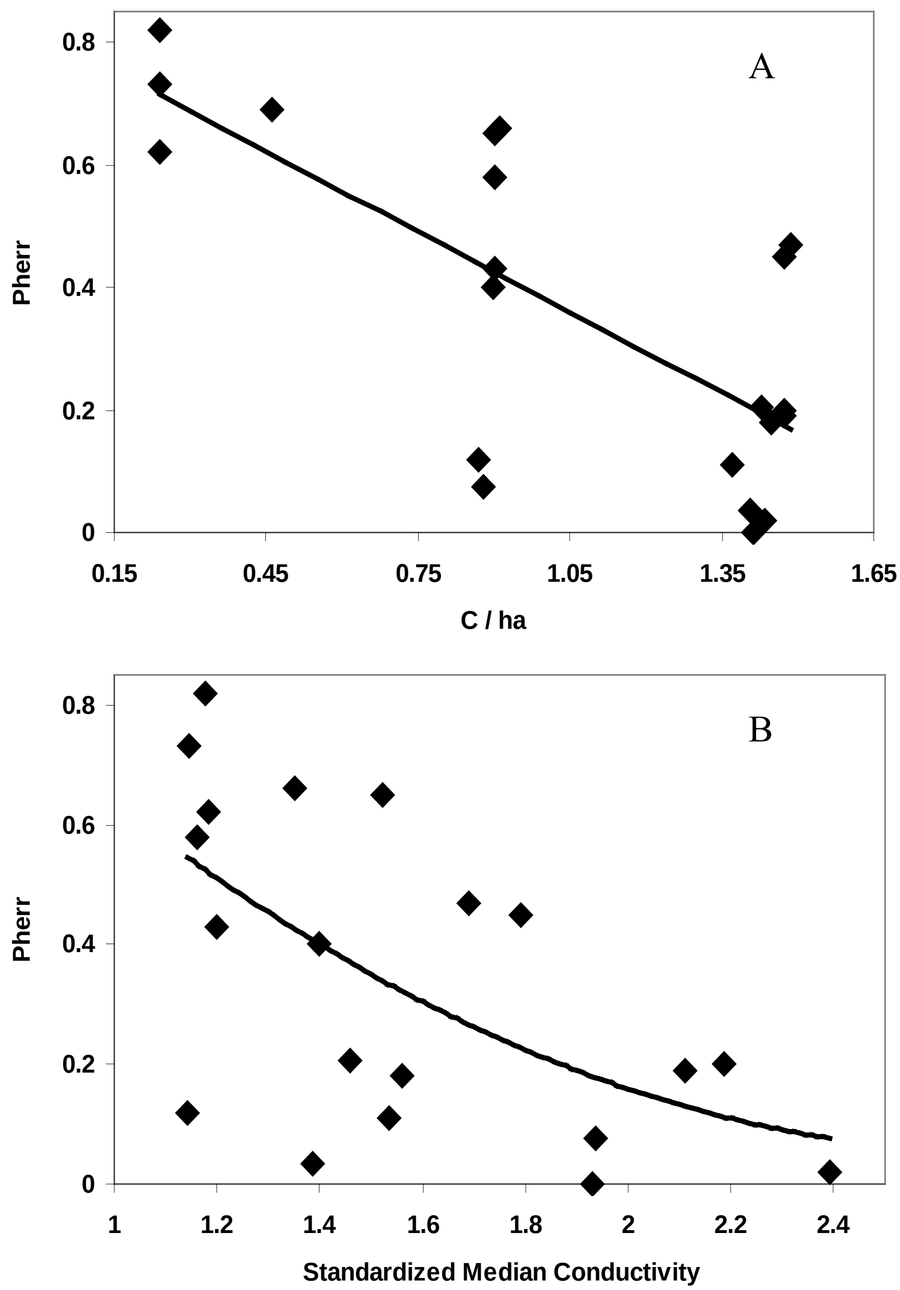


Figure 1-12. (A) Proportion of stream samples with Herring eggs or larvae (P_{herr}) and level of development (C / ha). (B) P_{herr} and standardized median spawning survey conductivity (uS / cm). Median conductivity was standardized to background estimates for Coastal Plain and Piedmont regions based on estimates in Morgan et al. (2012).



JOB 1: Development of habitat-based r eference p oints f or r ecreationally i mportant Chesapeake Bay fishes of special concern: development targets and thresholds

Section 2: Estuarine Yellow Perch Larval Presence-Absence Sampling

Carrie Hoover, Alexis Maple, Jim Uphoff, Margaret McGinty, Shaun Miller, and Brian Redding

Introduction

Presence-absence sampling for Yellow Perch larvae in 2014 was conducted in the upper tidal reaches of the Nanticoke, Choptank, Patuxent, and Bush Rivers, and Mattawoman and Nanjemoy Creeks during the month of April, and through the first week of May in the Northeast River (Figure 2-1). Annual L_p, the proportion of tows with Yellow Perch larvae during a standard time period and where larvae would be expected, provides a cost-effective measure of the product of egg production and survival through the early post-larval stage. In 2014 we continued examining relationships of L_p with estimates of development and other land uses.

We examined a hypothesis that development negatively influenced watershed organic matter (OM) dynamics, altering zooplankton production important for Yellow Perch larval feeding success and survival (the OM hypothesis) using the empirical-statistical approach recommended by Austin and Ingham (1978) and Crecco and Savoy (1984) for resolving the effects of environment on fish recruitment. This approach offers a working hypothesis that is tested for validity with empirical data and a thorough statistical analysis.

Years of high spring discharge favor anadromous fish recruitment in Chesapeake Bay (Hoffman et al. 2007; Martino and Houde 2010) and may represent episodes of hydrologic transport of accumulated OM from watersheds that fuel zooplankton production and feeding success (McClain et al. 2003). Under natural conditions, riparian marshes and forests would provide OM subsidies in high discharge years (Hoffman et al. 2007), while phytoplankton would be the primary source of OM in years of lesser flow. Shortage of appropriate food has been frequently hypothesized to cause high mortality of fish larvae (Martin et al. 1985; Miller et al. 1988; Heath 1992).

Urbanization was expected to negatively impact Yellow Perch larval feeding success because it affects the quality and quantity of OM in streams (Paul and Meyer 2001) and was negatively associated with extent of wetlands in many subestuary watersheds evaluated in this study (Uphoff et al. 2011). Riparian zones and floodplains that are sources of OM become disconnected from stream channels by stormwater management in suburban and urban watersheds (Craig et al. 2008; Kaushal et al. 2008; Elmore and Kaushal 2008; Brush 2009; NRC 2009), altering quantity and transport of OM (Paul and Meyer 2001; McClain et al. 2003; Stanley et al. 2012).

Correlation analyses examined associations of C/ha and 2010-2014 feeding success, L_p, larval TL, diet composition, and relative detritus levels collected during spring surveys. Larval fish size was included because it can be critical to larval feeding and starvation (Miller et al. 1988). Uphoff et al. (2012) included factors in addition to C/ha in analyses of 2010-2011 feeding success: relative amounts of OM, larval TL, mean water temperature, and mean conductivity in analyses of feeding success. Organic matter and larval length were found to be significant influences on feeding success, but water temperature and mean conductivity were not. Analyses of 2010-2014 feeding data in this report concentrated on variables that were significant in Uphoff et al. (2012).

During 2012-2014, Yellow Perch were also collected for analysis of the ratio of ribonucleic acid (RNA) concentration to deoxyribonucleic acid (DNA) concentration in body tissue in addition to estimating L_p and feeding success. The quantity of DNA within a cell is constant within a species while the quantity of RNA varies with protein synthesis (Tardiff et al. 2005). Since growth is a function of protein synthesis, RNA/DNA ratios provide a sensitive indicator of recent growth at any given time (Buckley 1984). This ratio is a useful indicator of nutritional status and somatic growth in larval fish (Buckley 1984) that provides a method for examining connections of feeding success and larval condition (Buckley 1984; Martin et al. 1985; Wright and Martin 1985; Clemmesen 1994; Blom et al. 1997) without requiring extensive sampling and sample processing needed to measure mortality directly. Tardif et al. (2005) used RNA/DNA ratios of Yellow Perch larvae and juveniles to determine differences in productivity of managed and natural wetlands of Lake St. Pierre, Canada.

Samples were gathered from two adjacent Potomac River subestuaries with watersheds exhibiting rural (C/ha = 0.09) and suburban levels of development (C/ha = 0.90) during 2014. We expected RNA/DNA ratios to decline with increased development.

Fishery managers may be interested in the impacts of habitat deterioration on the reproductive potential of a fish population (Boreman 1997). We combined the Thompson-Bell spawner biomass per recruit (SBR) model (Gabriel et al. 1989) used to calculate Yellow Perch biological reference points for Chesapeake Bay fisheries with estimated relative larval survival (L_p) at different levels of development to look at F reductions in a hypothetical fishery that offset egg and larval viability declines with development and maintain target SBR (35% of an unfished stock spawning biomass; spawner biomass per recruit that is equivalent to eggs per recruit; Boreman 1997; Yellow Perch Workgroup 2002). Management based on SBR links a harvest strategy to robustness of the stock to recruitment overfishing based on a measured or assumed stock-recruitment relationship. This allowed for exploration of the long-term effect of development (pollution) on the population's capability to be exploited at a sustainable level (Boreman 1997).

Methods

Conical plankton nets were towed from boats in upper portions of subestuaries to collect Yellow Perch larvae. Nets were 0.5-m in diameter, 1.0-m long, and constructed of 0.5 mm mesh. Nets were towed with the current for two minutes at a speed that maintained the net near the surface (approximately 2.8 km per hour). Temperature, dissolved oxygen, conductivity, and salinity were measured at each site on each sample date.

Ten sites were sampled in Nanjemoy and Mattawoman Creeks, and in Choptank, Northeast, and Nanticoke Rivers (Figure 2-1). Five to ten stations were sampled on the Bush and Patuxent Rivers. All subestuaries were sampled twice per week, although sampling in the Bush and Patuxent was not consistent and did not always follow this schedule. Larval sampling usually occurs during late March through mid-to-late April, but due to a late winter in 2014, sampling only occurred during the month of April in all subestuaries except Northeast River (it continued through the first week of May there). Boundaries of areas sampled were determined from Yellow Perch larval presence in estuarine surveys conducted during the 1970s and 1980s (O'Dell 1987). Sites in all subestuaries (except the Nanticoke and Choptank Rivers) were sampled with little spacing between tows because their larval nurseries were small.

Each sample was emptied into a glass jar and checked for larvae. Yellow Perch larvae can be readily identified in the field since they are larger and more developed than *Morone* larvae with which they could be confused (Lippson and Moran 1974).

Contents of the jar were allowed to settle and then the amount of settled OM was assigned a rank: 0 = a defined layer was absent; 1 = defined layer on bottom; 2 = more than defined layer and up to ¼ full; 3 = more than ¼ to ½ and; 4 = more than ½ full.

If a jar contained enough OM to obscure seeing larvae, it was emptied into a pan with a dark background and observed through a 5X magnifying lens. Organic matter was moved with a probe or forceps to free larvae for observation. If OM loads, wave action, or collector uncertainty prevented positive identification, samples were preserved and taken back to the lab for sorting.

Nanjemoy and Mattawoman Creeks, and Choptank River, were sampled by program personnel. Nanticoke and Northeast Rivers were voluntarily sampled by other Maryland Fisheries Service projects without charge to this grant. Patuxent and Bush Rivers were sampled by staff from the Chesapeake Bay National Estuarine Research Reserve Program and volunteers trained by our program biologists.

Composite samples of larvae were collected for feeding analyses from several sites in Mattawoman and Nanjemoy Creeks, and Choptank, Nanticoke, and Northeast Rivers during several sample trips. Subsamples of postlarvae 12 mm TL or less were examined for gut contents from each day's samples of each subestuary. These larvae represented first-feeding and early postlarvae, larvae that absorbed their yolk and began active feeding (Hardy 1978). Larvae were measured to the nearest millimeter. Gut fullness was judged visually and assigned a rank: 0 = empty; 1 = up to ¼ full; 2 = up to ½ full; 3 = up to ¾ full; and 4 = full. Major food items were classified as copepods, cladocerans, or other and the presence (coded 1) or absence (coded 0) of each group was noted.

The proportion of tows with Yellow Perch larvae (L_p) for each subestuary was determined annually for dates spanning the first catch through the last date that larvae were consistently present as:

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

where $N_{present}$ equaled the number of samples with Yellow Perch larvae present and N_{total} equaled the total number of samples. The SD of L_p was estimated as:

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

The 95% confidence intervals were constructed as:

$$^{(3)}\, L_p + 1.96 \text{ SD; (Ott 1977).}$$

In general, sampling to determine L_p began during the last days of March or first days of April and ended after larvae were absent (or nearly so) for two consecutive sampling rounds. In years where larvae disappeared quickly, sampling rounds into the third week of April were included in analysis even if larvae were not collected. This sampling schedule has been maintained for tributaries sampled by program personnel since 2006. Sampling by other Fisheries Service projects and volunteers sometimes did not adhere as strictly to this schedule.

Uphoff et al. (2012) developed L_p thresholds for brackish and tidal-fresh systems. Three brackish subestuaries with C/ha > 1.59 (10 estimates from Severn, South, and Magothy Rivers) exhibited chronically depressed L_p and their maximum L_p (0.33) was chosen as a threshold indicating serious deterioration of brackish subestuary larval nursery habitat. Similarly, tidal-fresh Piscataway Creek's four estimates of L_p (2008-2011) consistently ranked low when compared to other tidal-fresh subestuaries sampled (13th to 17th out of 17 estimates). The maximum for Piscataway Creek's four estimates, L_p = 0.65, was chosen as a threshold indicating serious deterioration of tidal-fresh larval habitat. Estimates of L_p would need to be consistently at or below this level to be considered "abnormal" as opposed to occasional depressions (Uphoff et al. 2012).

We estimated the risk that L_p was below a threshold for a tidal-fresh or brackish subestuary as one minus the cumulative proportion (expressed as a percentage) of the L_p binomial distribution function equaling or exceeding the restoration criterion. This calculation was used by Uphoff (1997) to estimate the risk that the proportion of plankton tows with Striped Bass eggs was not at a restored level.

Historical collections in the Choptank and Nanticoke Rivers targeted striped bass eggs and larvae (Uphoff 1997), but Yellow Perch larvae were also common (Uphoff 1991). Uphoff et al. (2005) reviewed presence-absence of Yellow Perch larvae in past Choptank and Nanticoke River collections and found that starting dates during the first week of April or early in the second week were typical and end dates occurred during the last week of April through the first week of May. Larval presence-absence was calculated from data sheets (reflecting lab sorting) for surveys through 1990. During 1998-2004, L_p in the Choptank River was determined directly in the field and recorded on data sheets (P. Piavis, MD DNR, personal communication). All tows were made for two minutes. Standard 0.5 m diameter nets were used in the Nanticoke River during 1965-1971 (1.0 • 0.5 mm mesh) and after 1998 in the Choptank River (0.5 mm mesh). Trawls with 0.5 m nets (0.5 mm mesh) mounted in the cod-end were used in the Choptank River during 1980-1990 (Uphoff 1997; Uphoff et al. 2005). Survey designs for the Choptank and Nanticoke Rivers were described in Uphoff (1997).

We used property tax map-based counts of structures per hectare (C/ha) in a watershed as our indicator of development (Uphoff et al. 2012). This indicator has been estimated for us by Marek Topolski of the Fishery Management Planning and Fish Passage Program. Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MDP 2010). 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 MDP's Geographic Information System's (GIS) database. Files were managed and geoprocessed in ArcGIS 9.3.1 from Environmental Systems Research Institute (ESRI 2009). 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. Each year's statewide tax map was clipped using the MD 8-digit watershed boundary file, and modified to exclude estuarine waters, to create watershed land tax maps. 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 present a problem since we are interested in the trend and not absolute magnitude (Uphoff et al. 2012).

Estimates of C/ha were used as a measure of watershed development intensity for analysis with L_p. Generally, whole watershed estimates were used with the following exceptions: Nanticoke and Choptank River watersheds were truncated at the lower boundaries of their striped bass spawning areas and at the Delaware border (latter due to lack of comparable data). Estimates of C/ha were available from 1950 through 2012 (M. Topolski, MD DNR, personal communication). While several watersheds had anomalous estimates exhibiting a one-year downward trend in C/ha (including Choptank and Nanticoke Rivers in 2000-2001, and Mattawoman Creek in 2011-2012; see Results section), estimates of C/ha for 2012 were used to represent 2013 and 2014 for all systems.

Estimates of C/ha for the IS target and limit were estimated from a power function that converts C/ha to IS based on Towson University satellite data interpretation (Uphoff et al. 2012). The target proposed in Uphoff et al. (2011), 5.5% IS, was reduced to 5% to meet IS guideline being developed by Maryland's Department of Natural Resources (MD DNR 2012). The IS threshold of 10% in Uphoff et al. (2011) remained unchanged. An estimate equivalent to 15% IS was also made to designate suburban watersheds that were developed well beyond the threshold. Estimates of C/ha that were equivalent to 5% IS, 10% IS, and 15% IS were estimated as 0.27, 0.83, and 1.59 C/ha, respectively by Uphoff et al. (2012).

Two regression approaches were used to examine the relationship between C/ha and L_p. First, separate linear regressions of C/ha against L_p were estimated for brackish and tidal-fresh subestuaries. If 95% CIs of slopes overlapped and 95% CIs of the intercepts did not overlap, we used the multiple regression of C/ha and salinity class against L_p. This latter approach assumed slopes were equal for two subestuary salinity categories, but intercepts were different (Freund and Littell 2006). Salinity was modeled as an indicator variable in the multiple regression with 0 indicating tidal-fresh conditions and 1 indicating brackish conditions. High salinity has been implicated in contributing to low L_p in Severn River (Uphoff et al. 2005). The association of mean salinity and IS can be significant and strong (Uphoff et al. 2010), and salinity is important to formation of stressful DO conditions in summer in mesohaline tributaries (see Section 3). Ricker (1975) warned against using well correlated variables in multiple regressions, so categorizing salinity for multiple or separate regressions of C/ha against L_p minimized confounding salinity with level of development. Level of significance was set at $\alpha < 0.05$. Residuals were inspected for trends, non-normality, and need for additional terms.

We used Akaike information criteria adjusted for small sample size, AIC_c, to evaluate the models that describe hypotheses that related changes in L_p to C/ha for each salinity category (separate slopes) or to C/ha and salinity category (common slopes, separate intercepts; Burnham and Anderson 2001):

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

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

An additional view of the relationship of L_p and C/ha was developed by considering dominant land use classification when interpreting salinity classification (brackish or tidal-fresh), C/ha, and L_p regressions. Primary land use (forest, agriculture, or urban) was determined from Maryland Department of Planning estimates for 1973, 1994, 1997, 2002, or 2010 that fell closest to a sampling year. These latter categories were not used in regression analyses, but were considered in the interpretation of results. Urban land consisted of high and low density residential, commercial, and institutional acreages (MD DNR 1999).

The mean of feeding success rank was calculated annually for each subestuary sampled in 2010-2014, as was mean total length (TL in mm) of larvae. The proportion of guts without food (P0) was estimated for each subestuary as was the proportion of larvae with copepods (P_{cope}), cladocerans (P_{clad}), or other (P_{othr}) food items. The latter three proportions were not additive.

Associations of C/ha with mean feeding rank, P0, mean TL, P_{cope}, P_{clad}, and P_{othr} (2010-2014 estimates) were tested with correlation analysis. Correlations of L_p with P0 and mean feeding rank were used to evaluate whether larval relative abundance was associated with feeding success. An additional set of correlation analyses examined associations among mean feeding success rank, mean TL, P_{cope}, P_{clad}, and P_{othr}.

We used OM0 (proportion of samples without OM, i.e., rank = 0) as our indicator of detritus availability and correlated OM0 against C/ha and feeding parameters that were significantly associated with C/ha. Proportions of samples without OM were estimated during 2011-2014, so fewer observations were available for analysis. The distribution of OM ranks assigned to samples in 2011-2014 was highly skewed towards zero and few ranks greater than one were reported.

We were specifically interested in the relationships of the amount of organic matter to development and larval feeding success. Examination of the plot of OM0 and C/ha suggested that the relationship could be nonlinear, with OM0 increasing at a decreasing rate with C/ha. We fit a power and logistic growth functions to these data using Proc NLIN in SAS (Freund and Littel 2006). The power function was described by the equation:

$$^{(5)}\ OM0 = a \bullet (C/ha)^b;$$

where a is a scaling coefficient and b is a shape parameter. The logistic growth function was described by the equation:

$$^{(6)}\ OM0 = b / ((1 + ((b - c) / c) \bullet (\exp(-a \bullet C/ha)));$$

where a is the growth rate of OM0 with C/ha, b is maximum OM0, and c is OM0 at C/ha = 0 (Prager et al. 1989).

We used linear and quadratic regressions to explore relationships of feeding success (mean of feeding ranks) with OM0 (Freund and Littel 2006). Linear and quadratic regressions explored this relationship for all data, with the linear regression describing a hypothesis about steady change, while the dome-shaped quadratic relationship would indicate an optimum value of OM0 for feeding success. A linear regression was also used on points representing only forested and urban watersheds, removing larger, agricultural (the only watersheds dominated by agriculture) Eastern Shore watersheds from consideration and confined remaining comparisons to western shore subestuaries.

During 2014, we collected Yellow Perch larvae for RNA/DNA analysis from a regional urban gradient represented by the watersheds of Mattawoman Creek (C/ha = 0.90) and Nanjemoy Creek (C/ha = 0.09; Figure 2-1). This design, based on several previous years' collections, anticipated that sampling from these two Rivers on three occasions would provide 30 larvae per system, per date for a total of 180.

Samples for RNA/DNA analysis were collected when larvae were gathered for analysis of gut contents. In the field, Yellow Perch larvae were composited from several stations (where possible) that bracketed where larvae are abundant. Once a candidate jar had been checked for larvae and OM, the sample was poured through a 500 µ screen and larvae were transferred to a large tube with special preservative (RNAlater®). The vial was labeled with the subestuary name and sample date. Larvae from other sites from one subestuary were composited into the vial on the same date.

In the lab, larvae for each date were processed for both gut contents and RNA/DNA ratios. Yellow Perch larvae 12 mm TL or less were examined for gut contents from each sample. These larvae represented first-feeding and early postlarvae, larvae that absorbed their yolk and began active feeding. Generally, 6 mm larvae were the smallest that contained food. Larvae were removed from the composite sample and placed in a Petri dish of water, examined for gut contents and then the guts were removed. The RNA/DNA ratio estimate did not contain food items. If a larva had not fed, the guts were teased away to be safe. Each processed larva was placed in a small individual vial of RNAlater® preservative. The vial was coded on the outside as follows: letter designating which creek, number designating which sample date, and number designating which individual larva was placed in the vial.

RNA/DNA ratios were estimated by science staff at the Cooperative Oxford Laboratory and partners from the University of Maryland Eastern Shore. Protocols for estimating RNA/DNA generally followed Kaplan et al. (2001). Larvae were stored at 4°C in RNAlater® for up to three weeks until processing. Whole body samples, minus gut contents, were digested with 1% sodium dodecylsulfate, proteinase K digestion buffer (66ug/ml), and 0.1M NaCl at 55^{o}C for several hours until completely digested. Samples were centrifuged at 11,000 rpm for 10 minutes, and the supernatant containing the nucleic acids was removed and stored at -80^{o}C until ready for processing.

A 400 µl portion of the supernatant was removed for digestion of DNA prior to analysis of RNA. Removal of DNA was accomplished by treating this portion of supernatant with DNase digestion buffer (0.2M Tris-HCl pH=7.5, 0.1M MgCl and 0.02M CaCl, and 10 U RNase-free DNase I). Samples incubated at 37 ^{o}C for 45 minutes in a dry bath. Samples were centrifuged for five minutes at 8,000 rpm. The supernatant was removed and stored at -80 °C until ready for processing.

Samples were analyzed for DNA and RNA using Quant-it™ PicoGreen® and Quant-it™ RiboGreen® (Molecular Probes, Oregon), respectively, according to the manufacturer's protocol. Samples were plated in triplicate on solid black 96-well microplates and fluorescence was measured at 480 nm excitation and 520 nm.

During sample processing, it was discovered that a dilution had been missed in the instructions used to estimate ratios during 2013 and 2014. Fortunately, samples had been retained for both years and could be used to develop adjustments for the missed dilution. To quantify nucleic acids, sample fluorescent readings were compared to DNA and RNA standard curves. These curves were developed by creating eight separate solutions of tissue digestion buffer and nucleic acid standard solutions. Lamda phage DNA and E. coli ribosomal 16S and 23S RNA (Molecular Probes, Oregon) were used as DNA and RNA standards, respectively. Serial dilutions of the 16 standard solutions (eight solutions per nucleic acid) were plated on 96-well microplates followed by the addition of PicoGreen® for DNA and RiboGreen® for RNA. Fluorescence was read at 480 nm excitation and 520 nm. The natural log-transformed fluorescent measures from each standard solution (F) were plotted against their respective nucleic acid concentration (C). Polynomial linear regression was used to determine the coefficients (Table 2-1) for each curve. The regression model used was

$$^{(7)}\ \mathrm{Log_e}\ F = (a \cdot C) + (b \cdot C^2) + d;$$

Where F and C are as defined previously, a and b are coefficients and d is the intercept. These coefficients were used to determine sample concentrations of DNA and RNA (after back-transformation) during 2014. Curves will be developed for 2013 as well, but were not available for this report.

The RNA/DNA ratios for each subestuary were plotted against larval TL or date. Reference lines indicating starving (RNA/DNA < 2; Blom et al. 1997) and fed larvae (RNA/DNA > 3; Buckley 1984; Wright and Martin 1985) based on values from larvae of several marine species and Striped Bass were added to the plots. A tabular summary of C/ha, median RNA/DNA ratio, mean fullness rank, N, N < 2, and N > 3 was constructed. The proportions of larvae with RNA/DNA ratios less than 2 (proportion starved or *Ps*) and their 90% confidence intervals were estimated for each subestuary as

$$^{(8)}\ Ps = N_{<2} / N_{total};$$

where $N_{<2}$ equaled the number of samples with RNA/DNA ratios less than 2 and N_{total} equaled the total number of RNA/DNA samples. The SD of *Ps* was estimated as

$$^{(9)}\ SD = [(Ps \bullet (1- Ps)) / N_{total}]^{0.5} \text{ (Ott 1977).}$$

The 90% confidence intervals were constructed as

$$^{(10)}\ Ps + (1.44\ SD;\ \text{Ott 1977}).$$

Proportions of larvae with RNA/DNA ratios greater than 3 (proportion fed or *Pf*) were estimated as in equations 8-10, but *Pf* was estimated with the number of larvae with RNA/DNA ratios greater than 3 ($N_{>3}$) in the numerator of equation 8 and *Pf* was substituted for *Ps* in the remaining equations.

For each subestuary and sample date, RNA/DNA ratio means and the number of samples in the *Pf* and *Ps* categories were summarized along with mean fullness rank, mean TL, and total sample size. *Ps* and *Pf* and their 90% CI's were estimated (Johnson 1999). Confidence interval comparisons were limited to larvae with a common TL range among all subestuaries.

The Yellow Perch Thompson-Bell model and its inputs were from Piavis and Webb (2014a). We used pooled (by sex and for 1998-2013) allometry and von Bertalanffy equations for upper Bay Yellow Perch to parameterize it (Piavis and Webb 2014b). We modified the SBR model to calculate eggs per recruit (EPR) so that relative larval survival (an index of egg-postlarval survival) could be applied in relevant currency. We used a weight-to-fecundity relationship from the Patuxent River (Tsai and Gibson 1971) for EPR and estimated target EPR as 35% of an unfished stock. Other fecundity relationships were available as well, but all produced the same percent of unfished EPR under the scenarios examined as the Patuxent River relationship. We used the selectivity pattern of the 8.5 – 11.0 inch slot limit applied to Maryland's Chesapeake Bay commercial fisheries and the 9.0 inch minimum size limits applied to recreational fisheries in two sets of runs (Piavis and Webb 2014a). An initial population of 100,000 age-1 recruits was used.

The relative impact of development on relative larval survival (S_r) was indexed as the ratio of L_p at a given level of development (indicated by structures per hectare in a watershed or L_{px}) to L_p at our target level of development (rural = 0.27 C/ha or 5% impervious surface or IS or L_{pt}); $S_r = L_{px}$ / L_p target. Structures per hectare approximated the transition to a suburban landscape (0.82 C/ha or 10% IS; equivalent to Mattawoman Creek's watershed) and two higher levels of suburban development (1.59 C/ha or 15% IS, equivalent to Piscataway Creek; and 2.58 C/ha or 20% IS, equivalent to Severn River). We used separate linear regressions that described the relationship of L_p to C/ha for brackish and freshwater systems (described below, but from Uphoff et al. 2013).

The Thompson-Bell model was modified to accommodate S_r by multiplying fecundity by S_r at each age class. Reductions in larval survival were assumed to affect eggs of all age-classes equally. Fecundity · S_r at each age class was summed to estimate EPR. This sum was divided by an estimate of EPR at F = 0 and S_r = 1.0 to obtain % of unfished EPR in good habitat. Since 35% of unfished EPR was the target, the Thompson-Bell equations were solved for F35% at each level of S_r (development). Reductions in F35% at S_r different from the target condition (S_r = 1.0 at 5% IS) were estimated as S_r at X% IS / S_r at 5% (target) IS.

Results

During 2014, sampling began on April 2 in Mattawoman and Nanjemoy Creeks, and they were sampled through April 30; samples through April 24 and April 30 were used to estimate L_p in Mattawoman and Nanjemoy Creeks, respectively. Sampling began on April 3 in the Northeast River and ended on May 9; dates between April 9 and May 9 were used for estimating L_p. Nanticoke River was sampled between April 7 and 29 and samples taken during April 7-25 were used to estimate L_p. Choptank River was first sampled on April 3 and last sampled May 13; dates between April 8 and April 25 were used for estimating L_p. Bush and Patuxent Rivers were each sampled four times between April 1 and April 22. Sampling in both of these systems was inconsistent in 2014 (between five and 10 sites per date), however, and was not used to estimate L_p.

Based on 95% CIs, estimates of L_p during 2014 were judged sufficiently precise to detect significant differences among subestuaries (Figure 2-2). Estimates of L_p for brackish subestuaries (Nanjemoy Creek and Choptank River) were similar to estimates for tidal-fresh subestuaries (Mattawoman Creek and Northeast River) in 2014 (range 0.68 to 0.83), with the exception of brackish Nanticoke River (L_p = 0.35).

None of the brackish subestuaries sampled during 2014 fell persistently below the threshold L_p (Figure 2-3). Tidal-fresh Mattawoman Creek (C/ha = 0.90) had exhibited below threshold L_p during 2012-2103, while Northeast River (C/ha = 0.47) has been above the threshold during 2010-2014 (Figure 2-4).

The range of C/ha values available for analysis with L_p was lower in brackish subestuaries (C/ha range = 0.05-2.73) than tidal-fresh (0.46-3.32; Table 2-2). None of the tidal-fresh estuaries analyzed were at or below the target condition (Figure 2-2).

Separate linear regressions of C/ha and L_p by salinity category were significant at $P < 0.0006$; Table 2-3). These analyses indicated that C/ha was negatively related to L_p and L_p was, on average, higher in tidal-fresh subestuaries than in brackish subestuaries.

Estimates of C/ha accounted for 30% of variation of L_p in brackish subestuaries and 35% in tidal-fresh subestuaries. Based on 95% CI overlap, intercepts were significantly different between tidal-fresh (mean = 0.93, SE = 0.09) and brackish (mean = 0.57, SE = 0.04) subestuaries. Mean slope for C/ha estimated for tidal-fresh subestuaries (mean = -0.28, SE = 0.07) were steeper, but 95% CI's overlapped CI's estimated for the slope of brackish subestuaries (mean = -0.17, SE = 0.04; Table 2-3). Both regressions indicated that L_p would be extinguished between 3.0 and 3.5 C/ha (Figure 2-5).

Overall, the multiple regression approach offered a similar fit (R^2 = 0.36; Table 2-3) to separate regressions for each type of subestuary. Intercepts of tidal-fresh and brackish subestuaries equaled 0.93 and 0.57, respectively; the common slope was -0.20. Predicted L_p over the observed ranges of C/ha would decline from 0.56 to 0.10 in brackish subestuaries and from 0.80 to -0.02 in tidal-fresh subestuaries (Figure 2-5).

Akaike's Information Criteria values equaled 9.5 for the regression of C/ha and L_p for brackish subestuaries, 10.0 for tidal-fresh estuaries, and 11.5 for the multiple regression that included salinity category. Calculations of Δ_i for brackish or tidal-fresh versus multiple regressions were approximately 2.01 and 1.58 (respectively), indicating that either hypothesis (different intercepts for tidal-fresh and brackish subestuaries with different or common slopes describing the decline of L_p with C/ha) were plausible.

Although we have analyzed these data in terms of tidal-fresh and brackish subestuaries, inspection of Table 2-2 indicated an alternative interpretation based on primary land use estimated by MDP. Predominant land use at lowest levels of development may be influencing the intercept estimates. Rural watersheds were absent for tidal-fresh subestuaries analyzed and the lowest levels of development were dominated by forest (Figure 2-6). Nearly all rural land in brackish tributaries was dominated by agriculture. Dominant land cover estimated by MD DOP for watersheds of tidal-fresh subestuaries was equally split between forest (C/ha = 0.46-0.91; 16 observations) and urban (C/ha > 1.17; 14 observations). Brackish subestuary watershed rural lands were dominated by agriculture (C/ha < 0.22; 32 observations), while forest land cover (C/ha ~ 0.09) was represented by six observations. The range of L_p was similar in brackish subestuaries with forest and agricultural cover, but the distribution seemed shifted towards higher L_p in the limited sample from the forested watershed (Nanjemoy Creek). Urban land cover predominated in ten observations of brackish subestuaries (C/ha > 1.61; Table 2-2; Figure 2-6). Tidal-fresh subestuary intercepts may have represented the intercept for forest cover and brackish subestuary intercepts may have represented agricultural influence. If this is the case, then forest cover provides for higher L_p than agriculture. Increasing suburban land cover leads to a significant decline in L_p regardless of rural land cover type.

We examined 332 larval guts during 2010, 523 in 2011, 466 in 2012, 706 in 2013, and 1891 in 2014. Samples were drawn primarily from tidal-fresh subestuaries (21 of 27 subestuary and year combinations). A smaller sample size was available for correlations with OM0 (N = 22) than other variables (N = 27) because observations of OM did not start until 2011.

Larvae averaged 8.4-11.1 mm in 2010 (size range among days sampled), 8.3-9.3 mm in 2011, 7.5-8.8 mm in 2012, 7.3-8.8 mm in 2013, and 7.3-8.7 mm in 2014 (Table 2-4). Larval Yellow Perch guts contained food in all years and subestuaries except Piscataway Creek during 2011. Copepods were the most prevalent food item during 2010 and 2011, and were found in 51-100% of guts sampled (excluding Piscataway Creek). Copepods were not as prevalent in 2012 and only Piscataway and Mattawoman Creeks had P_{cope} estimates within the range observed in 2010-2011. In 2013, copepods were still not as prevalent and were found in 0-69% of guts sampled (Northeast River larvae did not contain copepods). In 2014, copepods were again found in higher numbers, being present in 48-86% of guts sampled. Cladocerans were found in a higher proportion of guts sampled in 2014 (49-96%), then guts in 2013 (20-84%), or during 2010-2012 (0-56%), with the exception of the Nanticoke River in 2011 (71%). The “other” food item category represented a high fraction of guts in Piscataway Creek (53%) in 2010 and 1-30% of guts in remaining subestuaries during 2010-2011. This category was predominant in larval gut samples from all five subestuaries during 2012, but it should be noted that most gut contents were already too digested to be identifiable and could not be categorized any other way during that year (70-100%; Table 2-4). Gut content identification was more straightforward in 2013 and the “other” food item category represented what was seen in previous years (44%). In 2014, the range of “other” food item presence was highly variable (6-100%), again due to the amount of digested material present that could not be identified.

During 2010-2014, percentage of guts without food ranged from 0 to 46% in all subestuary and year combinations except Piscataway Creek during 2011 (100%). Mean fullness rank ranged between 0.55 and 3.27 in all subestuary and year combinations except Piscataway Creek during 2011 (where it was 0; Table 2-4). In comparison with 2010 and 2011, feeding success was low in 2012-2014 (Table 2-4).

The type of food present in larval Yellow Perch guts was significantly associated OM, but not with development, with P_{cope} being negatively correlated with OM0 ($r = -0.52$, $P = 0.04$; Table 2-5). The amount of food present in larval guts was also correlated with the presence of copepods, with both mean fullness rank and P0 being significantly associated with P_{cope} ($r = 0.87$, $P = <0.0001$ and $r = -0.61$, $P = 0.0003$, respectively). Mean TL was negatively correlated with P_{clad} ($r = -0.48$, $P = 0.02$), indicating larger larvae had cladocerans present in their diets less often. Estimates of L_p were significantly and negatively correlated with P_{clad} ($r = -0.46$, $P = 0.03$) and P_{othr} ($r = -0.53$, $P = 0.01$; Table 2-5).

Estimates of C/ha and OM0 were significantly related. A non-linear power function fit the data (approximate $r^2 = 0.46$, $\alpha < 0.0001$; N = 16), depicting OM0 increasing towards 1.0 at a decreasing rate as C/ha approached 1.50 (Figure 2-7). The relationship was depicted by the equation:

$$^{(11)}\ OM0 = 0.87 \cdot ((C/ha)^{0.14});$$

Approximate standard errors were 0.04 and 0.05 for parameters a and b, respectively. The logistic growth function (equation 6) fit these data similarly, but term a was not significantly different from zero.

Regression analyses indicated that organic matter may have a limited influence on larval feeding success, at best. A linear regression of OM0 and mean fullness rank using all data (agricultural, forest, and urban watersheds) was not significant ($r^2 = 0.04$, $\alpha = 0.38$, N = 21; Figure 2-8) and did not indicate that OM0 influenced feeding success of Yellow Perch larvae. A linear regression of subset of watersheds (western shore subestuaries that were forested or urban, omitting Eastern Shore agricultural watersheds) explained about 40% of variation in feeding success ($\alpha = 0.009$, N = 16; Figure 2-8). A dome-shaped quadratic regression to all data fit about as well ($r^2 = 0.34$, $\alpha = 0.02$, N = 21; Figure 2-8) as the forest and urban subset fit to the linear regression. The descending portion of the quadratic model was consistent with the decline of forest and urban subset described above. The quadratic model suggested an optimum level of OM0 of about 0.56 that would produce a predicted mean fullness of 2.12 (Figure 2-8). The curve fitting OM0 and feeding success data may not have provided a means of understanding a phenomenon (high feeding success) that occurs episodically when first-feeding Yellow Perch larvae and abundant copepods match. We believe 2011 represents a year where timing of Yellow Perch larvae and copepods matched, enhancing feeding success over a broad geographic area (Nanticoke River, Nanjemoy Creek, Elk River and Northeast River encompassed the lower Eastern Shore, Potomac River, and upper Chesapeake Bay). In 2011, four of five mean fullness ranks greater than 2 were encountered during the span that OM was measured (2011-2014). The two suburban watersheds sampled in 2011, both tributaries of the Potomac River, clearly had higher OM0 and low feeding scores, while rural watersheds had high mean fullness ranks and lower OM0 (Figure 2-8). Remaining years with mostly lower feeding success did not exhibit a clear pattern of feeding success with OM0 and it was likely that timing of zooplankton did not match first-feeding larvae. Anecdotally, we observed that high copepod abundance in sample jars peaked earlier than Yellow Perch larvae during 2014.

Yellow Perch larvae were collected (as designed) for RNA/DNA analysis from Mattawoman (N = 236: C/ha = 0.90) and Nanjemoy Creeks (N = 352; C/ha = 0.09) on April 9, 14, 17, 22 and 24, 2014 (Table 2-6). Mattawoman and Nanjemoy Creek collections both had larvae exceeding 12 mm in length, but analysis was restricted to sizes in common (5.5-12 mm) and typical of that of first-feeding larvae (Figure 2-9). Nanjemoy Creek's watershed was below the threshold development level, while Mattawoman Creek has passed the suburban threshold. Estimates of OM0 were 0.72 and 0.53 in Mattawoman and Nanjemoy Creeks, respectively.

Ratios of RNA/DNA were highest for 6.5-8 mm TL postlarvae during 2014, several larvae in each system had a ratio greater than 3 (well fed larvae based on marine and Striped Bass larvae), and 33% of all larvae (193 out of a total of 588) had an RNA/DNA ratio not indicative of starvation (all above a ratio = 2.0; Figure 2-9). Ratios of RNA/DNA declined with TL in both Mattawoman and Nanjemoy Creeks to as low as

0.73 for larvae 9 mm or larger, while mean fullness, on average, increased over time (Table 2-6).

Estimates of Pf were 0.05 (90% CI = 0.03-0.07; C/ha = 0.09) in Nanjemoy Creek and 0.08 (90% CI = 0.06-0.11; C/ha = 0.90) in Mattawoman Creek in 2014. Estimates of Ps were 0.66 (90% CI = 0.62-0.70) in Nanjemoy Creek in 2014, and 0.57 in Mattawoman Creek (90% CI = 0.52-0.62). Construction of 90% CI's was confined to 6-9 mm TL larvae only for Pf and Ps, the sizes most representative of both systems. The great majority of larvae collected during 2014 would have been considered in starved condition (67%, or 395 of the 588 total samples collected, with RNA/DNA ratios $\leq$ 2) under the criterion developed from other fish larvae regardless of level of development or OM0.

Values of S_r for brackish or tidal-fresh subestuaries converged when standardized as $S_r = L_{px} / L_p$, so separate relationships were not used (Table 2-7). The survival index (S_r) at 5% IS equaled 1.0 and was 0.82 at 10% IS, 0.56 at 15% IS, and 0.25 at 20% IS. We originally intended to look at combinations of recreational and commercial fisheries, but it turned out that while estimates of F35% were different for the recreational and commercial selectivity schedules, the percentage reductions in F needed for F35% were nearly identical. At 10% IS, a 24-25% reduction in F was needed to maintain EPR at 35% of an unfished stock (target level); a 63-64% reduction was necessary at 15% IS; and at 20% IS it was not possible to compensate for diminished larval survival with reductions in F (Table 2-7). These percentage reductions were independent of the size limits imposed.

Discussion

Estimates of L_p declined perceptibly once development exceeded the threshold (0.83 C/ha or 10% IS). A forest cover classification in a watershed was associated with higher L_p (median L_p = 0.78) than agriculture (median L_p = 0.51) or development (median L_p = 0.30). Interpretation of the influence of salinity class or primary land cover on L_p needs to consider that our survey design was limited to existing patterns of development. All estimates of L_p at or below target levels of development (0.27 C/ha or 5% IS; forested and agricultural watersheds) or at and beyond high levels of development (1.59 C/ha or 15% IS; urban watersheds) were from brackish subestuaries; estimates of L_p for development between these levels were from tidal-fresh subestuaries with forested watersheds. Larval dynamics below the target level of development primarily reflected Eastern Shore agricultural watersheds. Two types of land use would be needed to balance analyses: (1) primarily agricultural, tidal-fresh watersheds with below target development and (2) forested, brackish watersheds with development between the target and threshold. We do not believe that these combinations exist where Yellow Perch spawning occurs in Maryland's portion of Chesapeake Bay.

Salinity may restrict L_p in brackish subestuaries by limiting the amount of available low salinity habitat over that in a tidal-fresh subestuaries. Uphoff (1991) found that 90% of larvae collected in Choptank River during 1980-1985 were from 1‰ or less. Mortality of Yellow Perch eggs and prolarvae in experiments generally increased with salinity and was complete by 12‰ (Sanderson 1950; Victoria et al. 1992). The range of suitable salinities for prolarvae was lower than that for eggs (Victoria et al. 1992).

Development was an important influence on Yellow Perch egg and larval dynamics and negative changes generally conformed to ISRP guidelines in Uphoff et al.

(2011). Hilborn and Stokes (2010) advocated setting reference points related to harvest for fisheries (stressor) based on historical stock performance (outcome) because they were based on experience, easily understood, and not based on modeling. We believe applying IS or C/ha watershed development reference points (stressor) based on L_p (outcome) conforms to the approach advocated by Hilborn and Stokes (2010).

There appears to be some potential for development to influence organic matter (OM) and larval Yellow Perch feeding dynamics. However, OM may not matter much if there is not a match in the timing of copepod abundance and early feeding stages of Yellow Perch larvae. Timing of larvae and zooplankton abundance was an important aspect for the formation of strong year-classes of Striped Bass and White Perch (Limburg et al. 1999; Martino and Houde 2010). This analysis suggests that an influence of OM delivery on larval feeding success in urban watersheds may be episodic, occurring during years of high OM transport coupled with favorable timing of zooplankton and Yellow Perch larvae in rural watersheds.

Uphoff et al. (2013) found March temperature conditions also influenced L_p, and multiple regression models provided evidence that widespread climate factors (March precipitation as a proxy for OM transport and air temperature) influenced survival of Yellow Perch egg and larvae in Chesapeake Bay subestuaries and also supported the OM hypothesis. Yellow Perch require a period of low temperature for reproductive success (Heidinger and Kayes 1986; Ciereszko et al. 1997) and warm temperatures may preclude that from occurring.

Years of high spring discharge favor anadromous fish recruitment in Chesapeake Bay (Hoffman et al. 2007; Martino and Houde 2010) and may represent episodes of hydrologic transport of accumulated OM from watersheds (McClain et al. 2003) that fuel zooplankton production and feeding success. Under natural conditions, riparian marshes and forests would provide OM subsidies in high discharge years (Hoffman et al. 2007), while phytoplankton would be the primary source of OM in years of lesser flow. Stable isotope signatures of York River, Virginia, American shad larvae and zooplankton indicated that terrestrial OM largely supported one of its most successful year-classes. Lesser year-classes of American shad on the York River were associated with low flows, OM largely based on phytoplankton, and lesser zooplankton production (Hoffman et al. 2007). The York River watershed, with large riparian marshes and forest, was largely intact relative to other Chesapeake Bay tributaries (Hoffman et al. 2007).

Zooplankton supply (cladocerans and copepods) for first-feeding Yellow Perch larvae has been identified as an influence on survival in Lake Michigan (Dettmers et al. 2003; Redman et al. 2011; Weber et al. 2011) and Canadian boreal lakes (Leclerc et al. 2011), and survival of European perch *Perca fluviatis* in the Baltic Sea (Ljunggren et al. 2003). In a two-year study in Lake Saint Pierre, Canada, Tardif et al. (2005) attributed larval Yellow Perch RNA/DNA response to wetland types, cumulative degree days, and feeding conditions. The importance of adequate zooplankton supply and factors influencing zooplankton dynamics have been established for survival of Chesapeake Bay Striped Bass, White Perch, and American Shad larvae (North and Houde 2001; 2003; Hoffman et al. 2007; Martino and Houde 2010). Yellow Perch larvae share habitat in Chesapeake Bay subestuaries with these species, but little has been published on larval Yellow Perch dynamics and feeding ecology in Chesapeake Bay (Uphoff 1991).

Urbanization reduces quantity and quality of OM in streams (Paul and Meyer 2001; Gücker et al. 2011; Stanley et al. 2012). Riparian zones and floodplains that are sources of OM become disconnected from stream channels by stormwater management in suburban and urban watersheds (Craig et al. 2008; Kaushal et al. 2008; Brush 2009; NRC 2009). Small headwater streams in the Gunpowder and Patapsco Rivers watersheds (tributaries of Chesapeake Bay) were sometimes buried in culverts and pipes, or were paved over (Elmore and Kaushal 2008). Decay of leaves occurred much faster in urban streams, apparently due to greater fragmentation from higher stormflow rather than biological activity (Paul and Meyer 2001). Altered flowpaths associated with urbanization affect the timing and delivery of OM to streams (McClain et al. 2003). Organic matter was transported further and retained less in urban streams (Paul and Meyer 2001). Uphoff et al. (2011) found that the percentage of Maryland's Chesapeake Bay subestuary watersheds in wetlands declined hyperbolically as IS increased, so this source of OM diminished with development.

Management for OM (organic carbon) is nearly non-existent despite its role as a great modifier of the influence and consequence of other chemicals and processes in aquatic systems (Stanley et al. 2012). It is unmentioned in the Chesapeake Bay region as reductions in nutrients (N and P) and sediment are pursued for ecological restoration (http://www.epa.gov/reg3wapd/pdf/pdf_chesbay/BayTMDLFactSheet8_6.pdf). However, most watershed management and restoration practices have the potential to increase OM delivery and processing, although it is unclear how ecologically meaningful these changes may be. Stanley et al. (2012) recommended beginning with riparian protection or re-establishment and expand outward as opportunities permit. Wetland management represents an expansion of effort beyond the riparian zone (Stanley et al. 2012).

Agriculture also has the potential to alter OM dynamics within a watershed (Stanley et al. 2012) and the effect of this major land use on fish habitat warrants further study. Agriculture has been associated with increased, decreased, and undetectable changes in OM that may reflect diversity of farming practices (Stanley et al. 2012). As indicated earlier, extensive forest cover in a watershed may be linked to higher L_p than agriculture. However, Uphoff et al (2011) noted that agricultural watersheds had more area in wetlands than urban watersheds and this could buffer loss of OM from decreased forest cover. Streams in agricultural watersheds were unlikely to become disconnected since urban stormwater controls would not be in use (Uphoff et al. 2011).

In addition to feeding success, Yellow Perch egg viability declined greatly in highly developed suburban watersheds of Chesapeake Bay (Blazer et al. 2013). Abnormalities in ovaries and testes of adult Yellow Perch during spawning season were found most frequently in subestuaries with suburban watersheds and these abnormalities were consistent with contaminant effects (Blazer et al. 2013). Blazer et al. (2013) explained the biology behind low egg viability observed by Uphoff et al. (2005) in Severn River during 2001-2003 and persistently low L_p detected in three western shore subestuaries with highly developed suburban watersheds (C/ha > 1.59; Severn, South, and Magothy Rivers). Endocrine disrupting chemicals were more likely to cause observed egg hatching failure in well developed tributaries than hypoxia and increased salinity (Blazer et al. 2013), factors identified as potential contributors to poor egg hatching success in Severn River (Uphoff et al. 2005). Low L_p occurs sporadically in subestuaries with rural watersheds and appears linked to high temperatures (Uphoff et al. 2013).

It is unlikely that low L_p has always existed in these suburban subestuaries since all supported productive and lightly regulated recreational fisheries into the 1970s (the C/ha threshold was met in Severn River during 1972). Severn River supported a state hatchery through the first half of the twentieth century and hatching rates of eggs in the hatchery were high into 1955, when records ended (Muncy 1962). Egg hatching success of Severn River Yellow Perch had declined drastically by the early 2000s when estimates of L_p were persistently low (Uphoff et al. 2005).

We used a general indicator of development (C/ha) in our analyses because negative effects of development involve multiple stressors difficult to isolate. Effects of multiple stressors are usually worse than the worst single stressor alone (Breitburg et al. 1998; Folt et al. 1999). Our results suggest a general sequence of stressors impacted Yellow Perch larvae as development increased. Feeding success declined as development proceeded past the target level of development and was followed by reduced egg hatching in highly developed subestuaries, implying initial stress related to disruption of OM dynamics followed by endocrine disrupting contaminants.

Estimates of C / ha in Mattawoman Creek's watershed during 2012-2014 (0.90) were less than during 2011 (0.91), indicating some annual variability is possible (~ 1% in this case) due duplication or omission of records during annual database development. Determination of the exact cause of the trend shifts requires verification of database records and comparison of specific tax records with corresponding parcel maps within suspect sub-watersheds. The time frame for completion of this analysis exceeds that available for completion of this 2014 Federal Aid Report.

Our RNA/DNA sampling of Mattawoman and Nanjemoy creeks during 2014 indicated that most Yellow Perch larvae collected were in the starved category (395 of 588 larvae). Differences in RNA/DNA ratios between these two systems were not readily apparent in plotted data. The response time of RNA/DNA ratios of larval fishes characterizes the feeding environment within a week of sampling (Tardif et al. 2005). Ratios of RNA/DNA of fed larvae were expected to increase with body size (Clemmensen 1994), but did not in 2014 samples. Surveys of larval Striped Bass RNA/DNA in 1981 in the Potomac River estuary exhibited a similar declining pattern that we detected for Yellow Perch larvae, but Striped Bass ratios stabilized above starvation values (Martin et al. 1985). Blom et al. (1997) detected a decline in RNA/DNA ratios of Atlantic herring *Clupea harengus*; but few herring larvae were observed with ratios indicating starvation. Laboratory studies of RNA/DNA ratios of fed and starved larval Yellow Perch have not been conducted and we have relied on general guidelines from other species (Blom et al. 1997). Tardif et al. (2005) determined that RNA/DNA ratios of Yellow Perch in Lake Saint Pierre, Canada, averaged below 2, but did not provide indication of nutritional state of these larvae.

Low RNA/DNA ratios exhibited by some Yellow Perch at 7-9 mm may have reflected problems as they changed to external nutrition. RNA/DNA ratios of Atlantic Herring larvae fed shortly after hatching were in the same range as those found for starved larvae and were thought to result from the problems in changing from internal to external nutrition (Clemmenson 1994). There was no difference in RNA/DNA ratios for starved and fed Atlantic Herring larvae up to an age of 10 days. After 10 days, deprivation of food lead to a significant decrease in RNA/DNA ratios in comparison to fed Atlantic Herring larvae (Clemmensen 1994). Low RNA/DNA ratios of larger and presumably older Yellow Perch larvae sampled from our subestuaries may have been more indicative of poor feeding conditions, although it was possible that bias may have resulted from starving, weaker, poorly growing larvae being more vulnerable to our plankton nets than fed larvae.

In our analyses, we assumed that mainstem Potomac or Susquehanna River water was not a major influence on subestuary water quantity, water quality, and zooplankton supply. Sampling for Yellow Perch larvae occurred in the upper portions of subestuaries and this should have minimized the influence of mainstem waters, although some intrusion would have been possible at the most downstream sites in the smallest systems closest to the major Rivers (i.e., Piscataway Creek for the Potomac). Strong correlations of C/ha, L_p, and OM0 indicated that local conditions prevailed.

High estimates of L_p that were equal to or approaching 1.0 have been routinely encountered in the past, and it is likely that counts would be needed to measure relative abundance if greater resolution was desired. Mangel and Smith (1990) indicated that presence-absence sampling of eggs would be more useful for indicating the status of depleted stocks and count-based indices would be more accurate for recovered stocks. Larval indices based on counts have been used as a measure of year-class strength of fishes generally (Sammons and Bettoli 1998) and specifically for Yellow Perch (Anderson et al. 1998). Tighter budgets necessitate development of less costly indicators of larval survival and relative abundance in order to pursue ecosystem-based fisheries management. Characterizations of larval survival and relative abundance normally are derived from counts requiring labor-intensive sorting and processing. Estimates of L_p were largely derived in the field and only gut contents and RNA/DNA required laboratory analysis. These latter two analyses represented separate studies rather than a requirement for estimating L_p.

We do not expect that managers are going to apply the development EPR model tactically, but it can provide a strategic sense what sacrifices are needed to maintain target EPR to judge whether it is worth doing them. Fishing reductions can buy time, but effective growth management and habitat reconstruction need to follow to make sacrifices of fishing opportunity lead to sustainable fishing. The reductions in F needed to maintain F35% start at a manageable level at the onset of suburbanization (25% reduction at 10% IS) and quickly progress to a daunting level (> 60% at 15% IS). At intense suburban development, it is no longer possible to maintain target EPR (or SBR) because of relentless habitat decline. In the case of subestuaries with recreational fisheries only, fishing mortality rates may be low enough that reductions have negligible impact.

It is interesting to note that the timeline of concern and management activity in Severn River (Uphoff et al. 2005) would have roughly corresponded to the progression of reduced EPR implied by the results above. Articles about declining catches began when this watershed reached about 10% IS. Management action was taken in 1989 as part of widespread partial moratoria strategy to duplicate the Striped Bass recovery with Yellow Perch (Jensen 1993). This fishery closure started in 1989 when IS reached about 13-14% IS. Over the following years, IS grew in excess of 20%. Egg viability, estimated from historical hatchery records (1920-1960; a rural waterhsed) to be in excess of 80% fell to far less than 10% during 2001-2003 (well developed suburban watershed; Uphoff et al.

2005). The ratio of current to past egg viability was the equivalent of percent EPR at F = 0 (J. Uphoff, unpublished analysis) and indicated that recovery of Yellow Perch EPR in Severn River (and other developed tributaries) by managing the fishery alone would not be possible (an optimistic estimate of percent of unfished EPR was 12%). The fishery was reopened in 2009 to allow for some recreational benefit of fish that migrated in but did not reproduce successfully.

As noted by Boreman (1997), fishing is probably the only anthropogenic source of mortality that is density-dependent. Other sources, such as pollution from development modeled here, are insensitive to the abundance of the animals they are killing (fishery-independent; Boreman 1997). The fishery-independent aspects of mortality overwhelm density-dependent compensation from reducing F with the progression of suburban development.

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Table 2-1. Regression coefficients for DNA and RNA standard curves used for quantification of sample concentrations of nucleic acids.

Model	Intercept	Slope ([DNA or RNA])	Slope ([DNA or RNA]^2	p-value	R^2
DNA	2.111	4.15E-03	-2.71E-06	<0.0001	0.90
RNA	5.802	6.85E-03	-4.16E-06	<0.0001	0.95

Table 2-2. Estimates of proportions of ichthyoplankton net tows with Yellow Perch larvae (L_p) during 1965-2014 and data used for regression with counts of structures per hectare (C/ha). Salinity class 0 = tidal-fresh (≤ 2.0 ‰) and 1 = brackish (> 2.0 ‰). Year is the year a subestuary was sampled. Primary land use was determined for Maryland Department of Planning estimates for 1973, 1994, 1997, 2002, or 2010 that were closest to a sampling year. These latter categories were not used in regression analysis.

River	Year	C / ha	Primary Land Use	Salinity	Lp
Bush	2006	1.17	Urban	0	0.79
Bush	2007	1.19	Urban	0	0.92
Bush	2008	1.20	Urban	0	0.55
Bush	2009	1.21	Urban	0	0.86
Bush	2011	1.23	Urban	0	0.96
Bush	2012	1.24	Urban	0	0.28
Bush	2013	1.24	Urban	0	0.15
Choptank	1986	0.09	Agriculture	1	0.53
Choptank	1987	0.09	Agriculture	1	0.73
Choptank	1988	0.10	Agriculture	1	0.80
Choptank	1989	0.10	Agriculture	1	0.71
Choptank	1990	0.10	Agriculture	1	0.66
Choptank	1998	0.13	Agriculture	1	0.60
Choptank	1999	0.13	Agriculture	1	0.76
Choptank	2000	0.13	Agriculture	1	0.25
Choptank	2001	0.13	Agriculture	1	0.21
Choptank	2002	0.14	Agriculture	1	0.38
Choptank	2003	0.14	Agriculture	1	0.52
Choptank	2004	0.15	Agriculture	1	0.41
Choptank	2013	0.16	Agriculture	1	0.47
Choptank	2014	0.16	Agriculture	1	0.68
Corsica	2006	0.21	Agriculture	1	0.47
Corsica	2007	0.22	Agriculture	1	0.83
Elk	2010	0.59	Forest	0	0.75
Elk	2011	0.59	Forest	0	0.79
Elk	2012	0.59	Forest	0	0.55
Langford	2007	0.07	Agriculture	1	0.83
Magothy	2009	2.73	Urban	1	0.17
Mattawoman	1990	0.46	Forest	0	0.81
Mattawoman	2008	0.87	Forest	0	0.66
Mattawoman	2009	0.88	Forest	0	0.92
Mattawoman	2010	0.90	Forest	0	0.82
Mattawoman	2011	0.91	Forest	0	0.99
Mattawoman	2012	0.90	Forest	0	0.20
Mattawoman	2013	0.90	Forest	0	0.47
Mattawoman	2014	0.90	Forest	0	0.78
Middle	2012	3.33	Urban	0	0.00
Nanjemoy	2009	0.09	Forest	1	0.83
Nanjemoy	2010	0.09	Forest	1	0.96
Nanjemoy	2011	0.09	Forest	1	0.99
Nanjemoy	2012	0.09	Forest	1	0.03
Nanjemoy	2013	0.09	Forest	1	0.46
Nanjemoy	2014	0.09	Forest	1	0.82
Nanticoke	1965	0.05	Agriculture	1	0.50
Nanticoke	1967	0.05	Agriculture	1	0.43
Nanticoke	1968	0.05	Agriculture	1	1.00
Nanticoke	1970	0.06	Agriculture	1	0.81
Nanticoke	1971	0.06	Agriculture	1	0.33
Nanticoke	2004	0.11	Agriculture	1	0.49
Nanticoke	2005	0.11	Agriculture	1	0.67
Nanticoke	2006	0.11	Agriculture	1	0.35
Nanticoke	2007	0.11	Agriculture	1	0.55
Nanticoke	2008	0.11	Agriculture	1	0.19
Nanticoke	2009	0.11	Agriculture	1	0.41
Nanticoke	2011	0.11	Agriculture	1	0.55
Nanticoke	2012	0.11	Agriculture	1	0.04
Nanticoke	2013	0.11	Agriculture	1	0.43
Nanticoke	2014	0.11	Agriculture	1	0.35
Northeast	2010	0.46	Forest	0	0.68
Northeast	2011	0.46	Forest	0	1.00
Northeast	2012	0.47	Forest	0	0.77
Northeast	2013	0.47	Forest	0	0.72
Northeast	2014	0.47	Forest	0	0.77
Piscataway	2008	1.41	Urban	0	0.47
Piscataway	2009	1.43	Urban	0	0.39
Piscataway	2010	1.45	Urban	0	0.54
Piscataway	2011	1.46	Urban	0	0.65
Piscataway	2012	1.47	Urban	0	0.16
Piscataway	2013	1.47	Urban	0	0.50
Severn	2002	2.02	Urban	1	0.16
Severn	2004	2.09	Urban	1	0.29
Severn	2005	2.15	Urban	1	0.33
Severn	2006	2.18	Urban	1	0.27
Severn	2007	2.21	Urban	1	0.30
Severn	2008	2.24	Urban	1	0.08
Severn	2009	2.25	Urban	1	0.15
Severn	2010	2.26	Urban	1	0.03
South	2008	1.61	Urban	1	0.14

Table 2-3. Summary of results of regressions of proportions of tows with Yellow Perch larvae (L_p) and counts of structures per hectare (C/ha). Separate regressions by salinity (tidal-fresh ≤ 2.0 ‰ and brackish > 2.0 ‰) and a multiple regression using salinity as a class variable (tidal-fresh = 0 and brackish = 1) are presented.

ANOVA			Brackish			
Source	df	SS	MS	F	P	
Model	1	1.03302	1.03302	20.13	<.0001	
Error	46	2.36019	0.05131			
Total	47	3.3932				
r^2 = 0.3044						
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.57171	0.03876	14.75	<.0001	0.4937	0.64973
C / ha	-0.17311	0.03858	-4.49	<.0001	-0.25076	-0.09545

ANOVA			Tidal-Fresh			
Source	df	SS	MS	F	P	
Model	1	0.73656	0.73656	15.2	0.0006	
Error	28	1.35693	0.04846			
Total	29	2.09349				
r^2 = 0.3518						
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.92532	0.08578	10.79	<.0001	0.74961	1.10102
C / ha	-0.28298	0.07259	-3.9	0.0006	-0.43166	-0.13429

ANOVA			Multiple Regression			
Source	df	SS	MS	F	P	
Model	2	2.10608	1.05304	20.76	<.0001	
Error	75	3.80477	0.05073			
Total	77	5.91085				
r^2 = 0.3563						
	Coefficients	SE	t Stat	P-value	Lower 95%	Upper 95%
Intercept	0.83477	0.05438	15.35	<.0001	0.72644	0.9431
C / ha	-0.19625	0.03408	-5.76	<.0001	-0.26414	-0.12835
Salinity	-0.25057	0.05517	-4.54	<.0001	-0.36047	-0.14067

Table 2-4. Summary of estimates used in correlation analysis of Yellow Perch larval feeding success. C/ha = counts of structures per hectare. Mean full = mean of fullness ranks assigned to larval guts. OM0 = proportion of samples without organic matter (detritus). P0 = proportion of guts without food. Pclad = proportion of guts with cladocerans. Pcope = proportion of guts with copepods. Pother = proportion of guts with "other" food items. Mean TL = mean TL of larvae in mm. N = number of Yellow Perch larvae examined.

River	Year	C / ha	Mean full	OM0	P0	Pclad	Pcope	Pother	Mean TL	N
Elk	2010	0.59	2.75		0.05	0.02	0.95	0.13	11.1	110
Mattawoman	2010	0.90	2.00		0.09	0.15	0.78	0.09	9.2	55
Nanjemoy	2010	0.09	2.88		0.00	0.10	1.00	0.15	9.1	48
Northeast	2010	0.46	2.34		0.19	0.22	0.72	0.30	8.4	64
Piscataway	2010	1.45	1.85		0.13	0.00	0.55	0.53	9.4	55
Elk	2011	0.59	2.81	0.76	0.08	0.00	0.96	0.01	8.9	90
Mattawoman	2011	0.91	0.90	0.78	0.42	0.02	0.51	0.07	9.3	110
Nanjemoy	2011	0.09	2.18	0.56	0.07	0.03	0.83	0.20	9.0	150
Nanticoke	2011	0.11	3.27	0.55	0.08	0.71	0.92	0.16	8.6	51
Northeast	2011	0.46	2.44	0.58	0.08	0.00	0.91	0.09	8.3	90
Piscataway	2011	1.46	0.00	1.00	1.00	0.00	0.00	0.00	8.4	32
Bush	2012	1.23	2.48		0.00	0.55	0.53	1.00	8.6	40
Elk	2012	0.59	0.77	0.77	0.24	0.02	0.00	0.70	7.7	198
Mattawoman	2012	0.90	1.81	1.00	0.00	0.44	0.88	1.00	8.8	16
Northeast	2012	0.47	1.17	0.99	0.01	0.04	0.08	0.99	7.5	203
Piscataway	2012	1.46	1.67	0.98	0.00	0.56	0.67	1.00	8.7	9
Choptank	2013	0.16	1.04	0.33	0.21	0.37	0.34	0.33	7.6	319
Mattawoman	2013	0.90	1.69	0.79	0.00	0.84	0.69	0.04	7.6	98
Nanjemoy	2013	0.09	1.59	0.65	0.00	0.59	0.42	0.23	7.3	64
Nanticoke	2013	0.11	1.08	0.13	0.33	0.40	0.25	0.23	8.3	132
Northeast	2013	0.47	0.55	1.00	0.46	0.20	0.00	0.44	8.8	80
Piscataway	2013	1.46	2.31	0.74	0.00	0.38	0.69	0.23	7.9	13
Choptank	2014	0.16	1.59	0.60	0.005	0.85	0.56	0.85	8.1	610
Mattawoman	2014	0.90	1.88	0.72	0.00	0.96	0.86	1.00	7.3	271
Nanjemoy	2014	0.09	2.57	0.53	0.00	0.49	0.80	0.54	8.6	403
Nanticoke	2014	0.11	1.45	0.11	0.00	0.58	0.48	0.06	8.3	31
Northeast	2014	0.47	1.57	0.86	0.04	0.63	0.56	0.56	8.7	576

Table 2-5. Correlation matrix for Yellow Perch larval feeding success. C/ha = counts of structures per hectare. Mean fullness = average feeding rank of larvae. OM0 = proportion of samples without organic matter (detritus). P0 = proportion of guts without food. Pclad = proportion of guts with cladocerans. Pcope = proportion of guts with copepods. Pother = proportion of guts with "other" food items. Mean TL = mean TL of larvae in mm. L_p = proportion of plankton tows with larvae. Statistic r = Pearson correlation coefficient, P = level of significance, and N = number of observations. Gray shading indicates correlation of interest at P ≤ 0.05.

Parameter	Statistic	C / ha	Mean Fullness	OM0	P0	P_{clad}	P_{cope}	P_{other}	Mean TL
Mean Fullness	r	-0.26							
	P	0.24							
	N	22							
OM0	r	0.54	-0.63						
	P	0.03	0.009						
	N	16	16						
P0	r	0.27	-0.73	0.38					
	P	0.23	0.0001	0.14					
	N	22	22	16					
P_{clad}	r	0.11	0.08	-0.04	-0.42				
	P	0.64	0.72	0.88	0.05				
	N	22	22	16	22				
P_{cope}	r	-0.16	0.87	-0.52	-0.61	0.17			
	P	0.47	<.0001	0.04	0.003	0.45			
	N	22	22	16	22	22			
P_{other}	r	0.19	-0.10	0.38	-0.34	0.42	-0.18		
	P	0.39	0.67	0.15	0.13	0.05	0.41		
	N	22	22	16	22	22	22		
Mean TL	r	0.02	0.31	0.09	0.07	-0.48	0.36	-0.28	
	P	0.92	0.17	0.73	0.74	0.02	0.10	0.20	
	N	22	22	16	22	22	22	22	
L_p	r	-0.55	0.13	-0.50	0.15	-0.46	0.17	-0.53	0.21
	P	0.008	0.57	0.05	0.49	0.03	0.44	0.01	0.34
	N	22	22	16	22	22	22	22	22

Table 2-6. Summary of feeding success, larval length, sample size, and RNA/DNA characteristics, by subestuary and sample date. Data only for dates with feeding information and with RNA/DNA analysis are summarized. Mean fullness = mean feeding rank. Mean TL is in mm. N = the sample size of larvae processed for gut contents and with RNA/DNA ratios available for the date. Mean RNA/DNA is the average for the date. SE RNA/DNA is the standard error for the date. N RNA/DNA > 3 is the number of ratios above the fed criterion. N RNA/DNA < 2 is the number of ratios below the starvation criterion.

Subestuary	Variable	9-Apr	14-Apr	17-Apr	22-Apr	24-Apr
Mattawoman	Mean Fullness	1.7	2	2.3		2.2
	Mean TL	6.3	7.5	9		10.8
	N	102	122	6		6
	Mean RNA/DNA	2.37	1.69	1.22		1.06
	SE RNA/DNA	0.06	0.06	0.07		0.08
	N RNA/DNA > 3	13	6	0		0
	N RNA/DNA < 2	35	90	6		6
Nanjemoy	Mean Fullness	1.8	2.8	2.8	2.9	3
	Mean TL	6.8	8.1	8.5	11	11.1
	N	89	102	78	67	16
	Mean RNA/DNA	2.27	1.54	1.52	1.27	1.09
	SE RNA/DNA	0.07	0.05	0.07	0.08	0.04
	N RNA/DNA > 3	10	1	2	2	0
	N RNA/DNA < 2	31	87	62	62	1

Table 2-7. Results of Thompson-Bell modeling of eggs per recruit for effects of reduced larval survival on target fishing mortality rate (F35%). Commercial selectivity refers to an 8.5-11.0 inch slot limit and recreational refers to a 9.0 inch minimum length limit. C / ha = structures per hectare; IS = impervious surface; Sr = relative larval survival; F35 is the F needed for 35% of the egg production of an unfished stock, and F reduce is the reduction in F needed to maintain 35% of unfished egg per recruit. NP = not possible.

Selectivity	C/ha	IS	Sr	F35	F reduce
Commercial	0.27	5%	1.00	0.52	0%
Commercial	0.82	10%	0.82	0.39	25%
Commercial	1.59	15%	0.57	0.19	63%
Commercial	2.58	20%	0.25	NP	NP
Recreational	0.27	5%	1.00	0.33	0%
Recreational	0.82	10%	0.82	0.25	24%
Recreational	1.59	15%	0.57	0.12	64%
Recreational	2.58	20%	0.25	NP	NP

Figure 2-1. Sampling areas for the 2014 Yellow Perch larval presence-absence study. Nanticoke River watershed delineation was unavailable for Delaware and Northeast and was unavailable for Pennsylvania.

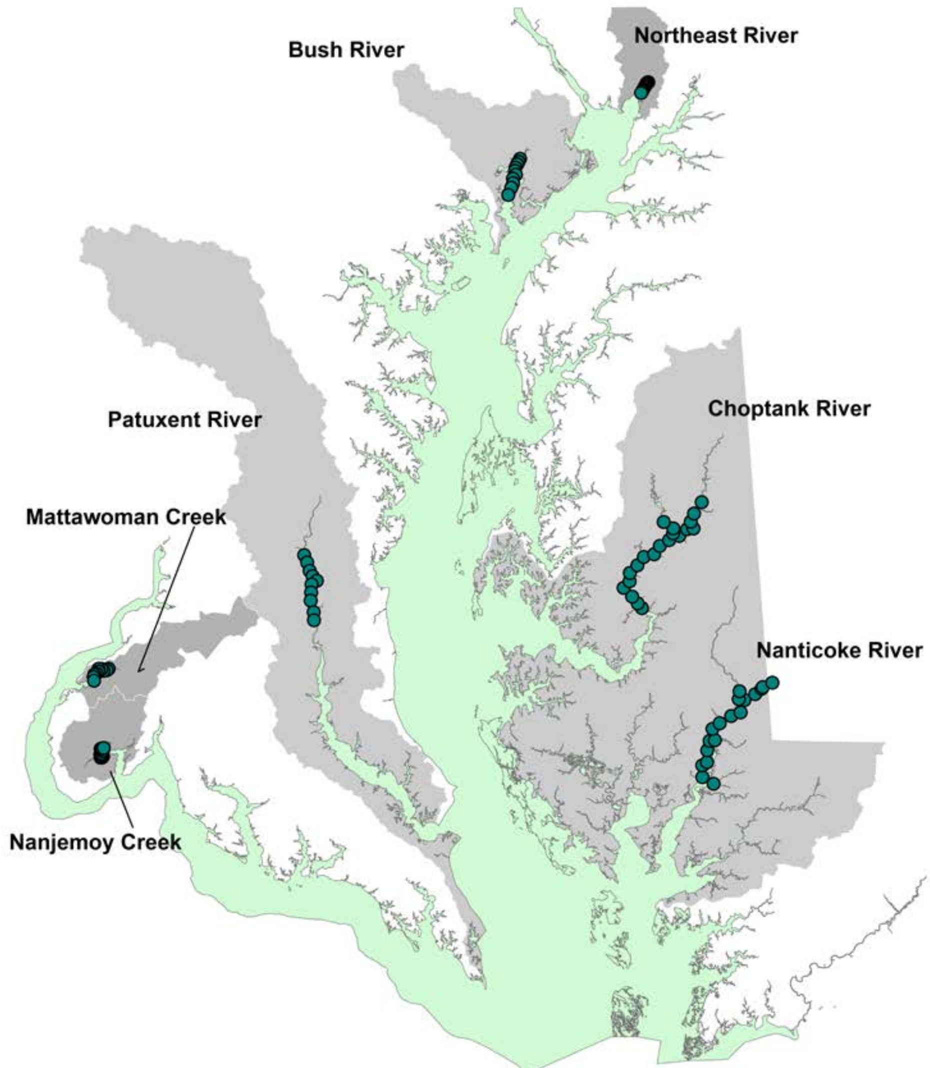


Figure 2-2. Proportion of tows with larval Yellow Perch (*L*p) and its 95% confidence interval in systems studied during 2014. Mean *L*p of brackish tributaries indicated by diamond and tidal-fresh mean indicated by dash.

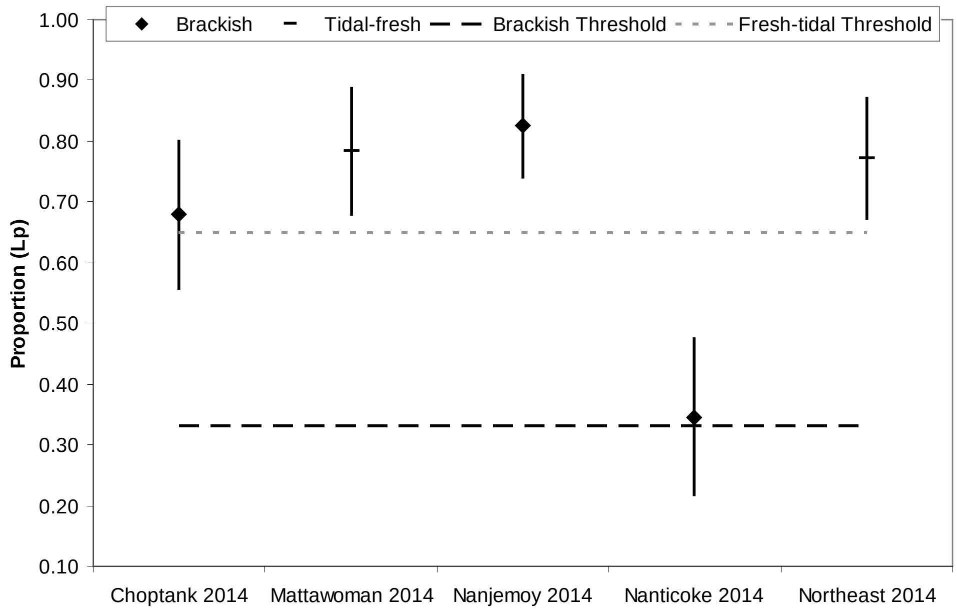


Figure 2-3. Proportion of tows with Yellow Perch larvae (*Lp*) for brackish subestuaries, during 1965-2014. Dotted line provides reference for persistent poor *Lp* exhibited in developed brackish subestuaries.

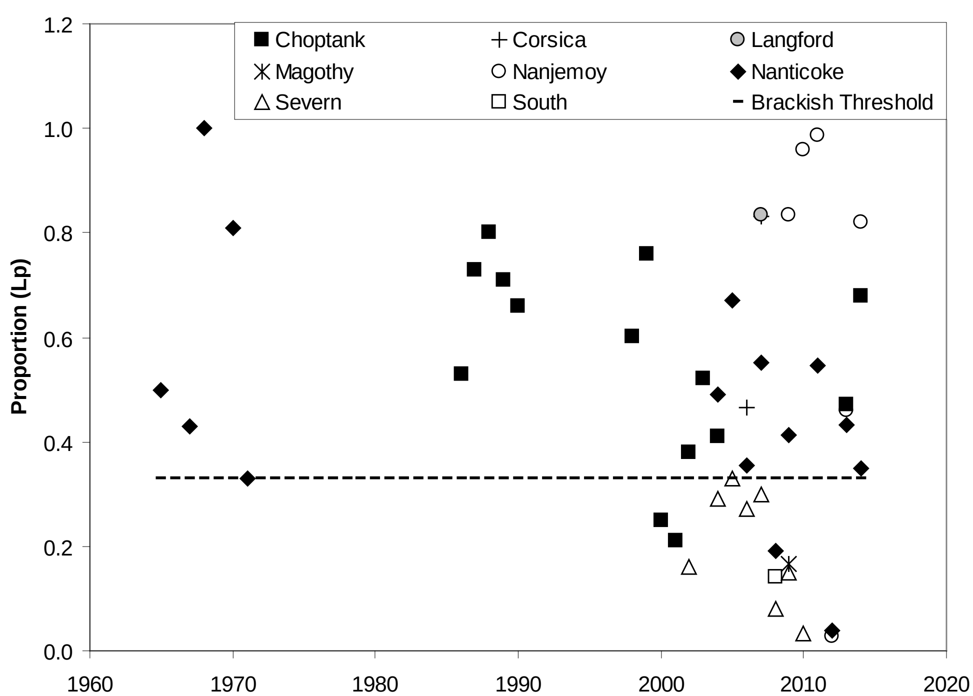


Figure 2-4. Proportion of tows with Yellow Perch larvae (*Lp*) for fresh-tidal subestuaries, during 1990-2014. Dotted line provides reference for consistent poor *Lp* exhibited in a more developed fresh-tidal subestuary (Piscataway Creek).

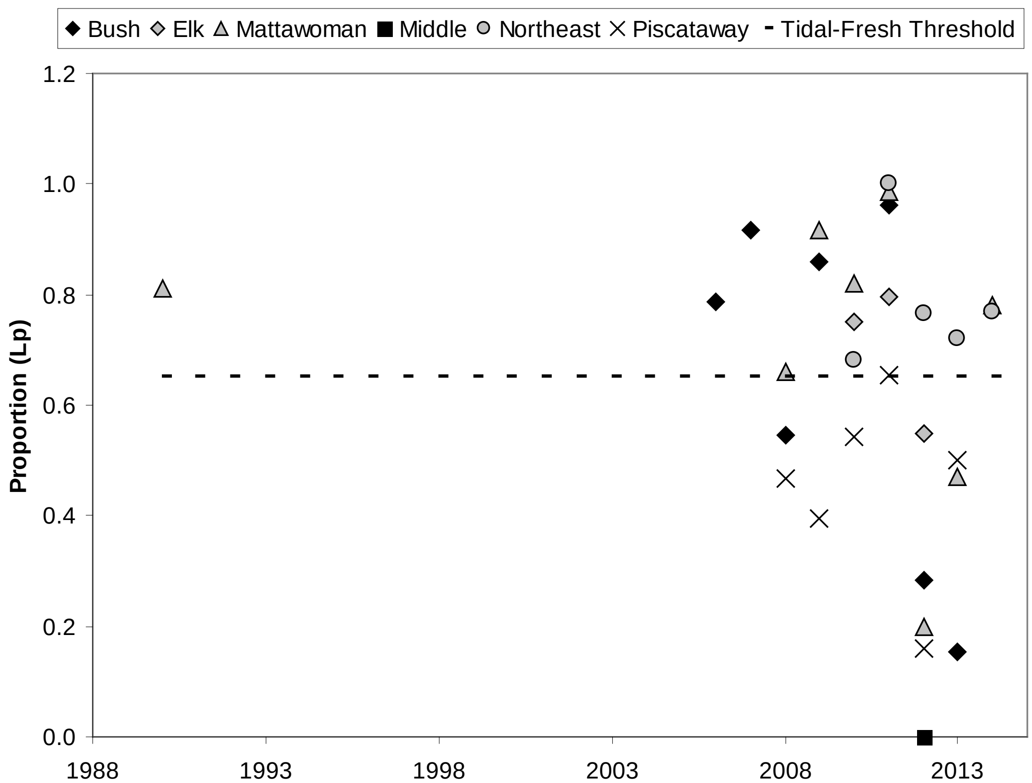


Figure 2-5. Relationship of proportion of plankton tows with Yellow Perch larvae and development (structures per hectare or C/ha) indicated by multiple regression of fresh and brackish subestuaries combined (prediction = MR) and separate linear regressions for both (prediction = LR).

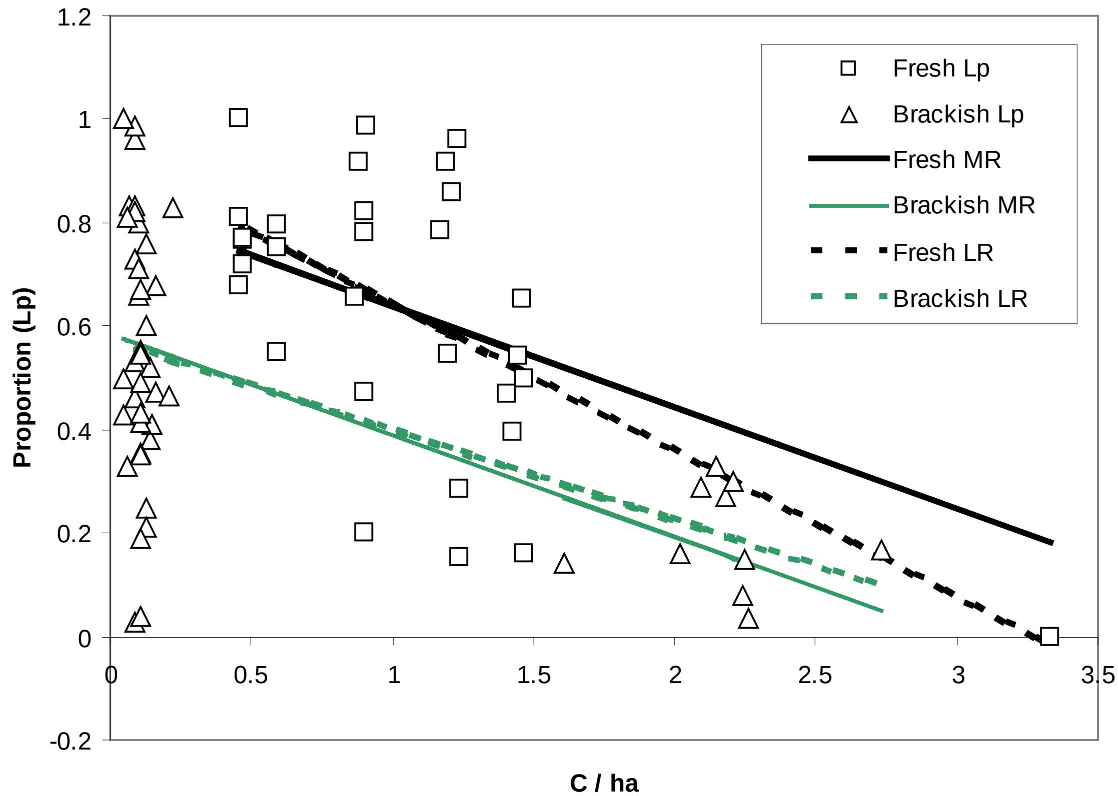


Figure 2-6. Proportion of plankton tows with Yellow Perch larvae plotted against development (C/ha) with Department of Planning land use designations and salinity class indicated by symbols. Diamonds and a “1” behind land use in the key indicate brackish subestuaries, while squares and a “0” indicate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

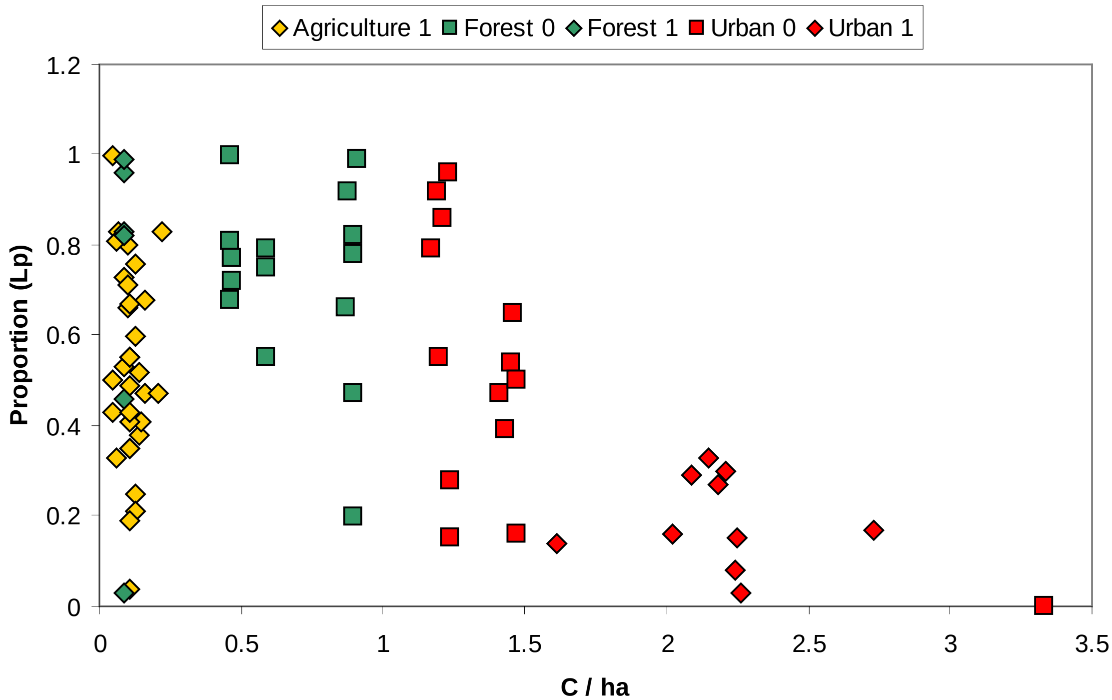
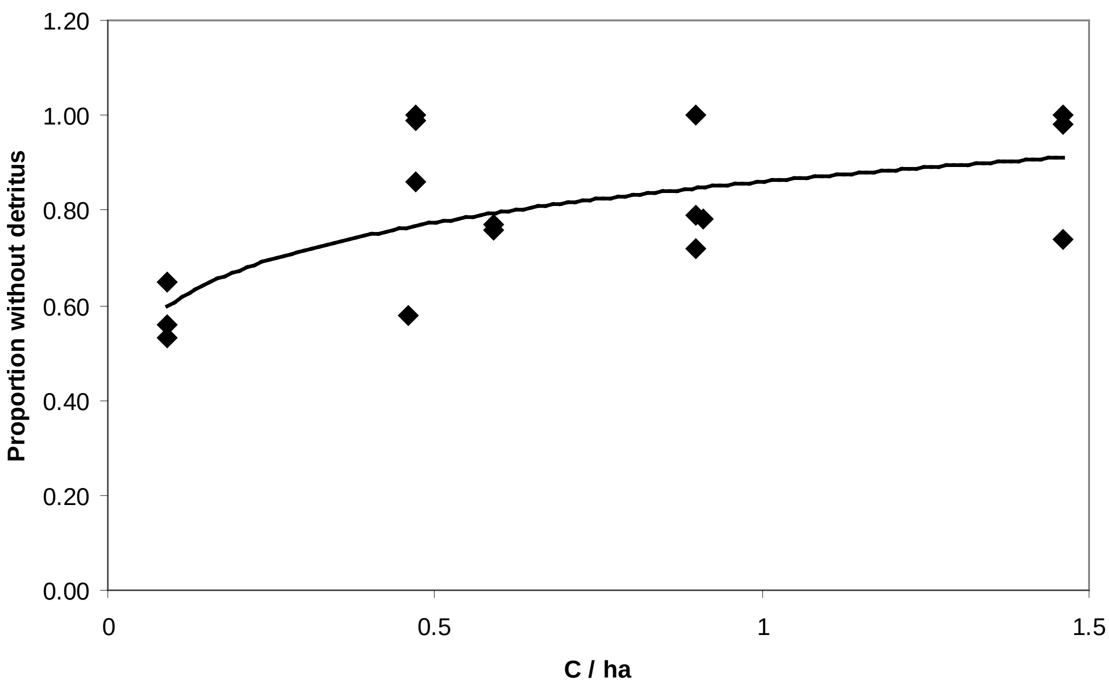


Figure 2-7. Relationship of proportion of plankton tows without detritus (OM0) and development (structures per hectare or C/ha).



JOB 1: Development of habitat-based r eference p oints f or r ecreationally i mportant Chesapeake Bay fishes of special concern: development targets and thresholds

Section 3 - Estuarine Fish Community Sampling

Alexis Park, Carrie Hoover, Shaun Miller, Margaret McGinty, Jim Uphoff, and Brian Redding

Introduction

Water quality and aquatic habitat within watersheds is altered by agricultural activity and urbanization; both include use of pesticides and fertilizers, while the latter may have additional industrial wastes, contaminants, stormwater runoff and road salt (Brown 2000; NRC 2009; Benejam et al. 2010) that act as ecological stressors. Extended exposure to biological and environmental stressors affect fish condition and survival (Rice 2002; Barton et al. 2002; Benejam et al. 2008; Benejam et al. 2010). 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.

Uphoff et al. (2011a) estimated target and limit impervious surface reference points (ISRPs) for productive fish habitat in brackish (mesohaline) Chesapeake Bay subestuaries based on dissolved oxygen (DO) criteria, and associations and relationships of watershed impervious surface (IS), summer DO, and presence-absence of recreationally important finfish in bottom waters. Watersheds at a target of 5.5% IS (based on Towson University IS estimates 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 mg / L (target DO). Mean bottom DO seldom exceeded 3.0 mg / L above 10% IS (suburban threshold; Uphoff et al. 2011a). Although bottom DO concentrations were influenced by development (indicated by IS) in brackish subestuaries, Uphoff et al. (2011b; 2012; 2013; 2014) have found adequate concentrations of DO in bottom channel habitat of tidal-fresh and oligohaline 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, allowing them to become well mixed. However, the summer fish community of tidal-fresh Mattawoman Creek underwent drastic changes in abundance and species richness as development threshold was approached that were unrelated to adequacy of DO in channel waters, indicating other stressors were important (Uphoff et al. 2009; 2012; 2013; 2014).

In 2014, we continued to evaluate summer nursery and adult habitat for recreationally important finfish in tidal-fresh, oligohaline, and mesohaline subestuaries of Chesapeake Bay. In this report, we evaluated the influence of watershed development on target species presence-absence and abundance, total abundance of finfish, finfish species richness, and tested our hypothesis that we formulated during 2013 that the water quality dynamics in Mattawoman Creek's extensive submerged aquatic vegetation (SAV) beds (low DO, high pH, and high organic matter) may be creating episodes of ammonia toxicity for fish. We continued to emphasize Mattawoman Creek in this report as part of Maryland DNRs' efforts to influence Charles County into modifying its comprehensive growth plan to conserve natural resources of its watershed (MDDNR 2013).

Methods

We sampled nine subestaries in Chesapeake Bay during 2014: Broad Creek, Harris Creek, and Tred Avon River, tributaries of the Choptank River; Mattawoman Creek, Piscataway Creek, and Nanjemoy Creek, tributaries of the Potomac River; Northeast River, Middle River, and Gunpowder River located in the upper Chesapeake Bay (Table 3-1; Figure 3-1). This is the third year of sampling of Broad Creek and Harris Creek. These watersheds, downstream of Tred Avon River (sampled since 2006), represented a gradient of development from 0.29 C / ha (Broad Creek) to 0.75 C / ha (Tred Avon) within a single watershed (Table 3-1); Harris Creek is undergoing an extensive Oyster restoration effort (MD DNR 2014). Three Potomac River tributaries were sampled in 2014; Mattawoman Creek has been sampled since 1989, Piscataway Creek since 2006 (except in 2008), and Nanjemoy Creek since 2008. Three subestuaries were sampled in upper Chesapeake Bay in 2014: Northeast River (sampled since 2007), Middle River (since 2009), and Gunpowder River (since 2009; Table 3-1).

We obtained compatible data from Bush River monitoring by citizen volunteers and staff from the Anita C. Leight Estuary Center (Bush River; Table 3-1; Figure 3-1). The Bush River has been sampled since 2006; the Estuary Center and its citizen volunteers, trained in 2011 by the Fisheries Service staff, have taken over sampling.

Housing density (C / ha) and impervious surface (IS) were estimated for each watershed (Table 3-1). 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). This indicator has been provided to us by M. Topolski (MD DNR).

Tax maps are graphic representations of individual property boundaries and existing structures that help State tax assessors locate properties (Maryland Department of Planning or MDP 2013). 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 MDP's Geographic Information System's (GIS) database. Files were managed and geoprocessed in ArcGIS from Environmental Systems Research Institute (ESRI 2009). 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 ArcGIS Model Builder to automate assembly of statewide tax maps, query tax map data, and assemble summary data. Each year's statewide tax map was clipped using the Maryland 8-digit watershed boundary file to create watershed land tax maps. Watershed area estimates excluded estuarine waters. 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 (Uphoff et al. 2012).

Uphoff et al. (2012) developed a nonlinear regression equation to convert annual estimates of C / ha to IS calculated by Towson University based on 1999-2000 (years in common) satellite imagery. The relationship of C / ha and IS was well described by the equation:

$$IS = 10.98\ (C / ha)^{0.63}, (r^2 = 0.96; P < 0.0001).$$

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 0.27, 0.83, and 1.59 C/ha, respectively (Uphoff et al. 2012).

MdProperty View tax data are annually updated by each Maryland jurisdiction to monitor the type of parcel development for tax assessment purposes. Detailed records of each structures composition, including the foundation's square footage, are included. Therefore, the tax data can be used to estimate increasing development within a given area: total number of structures (C / ha) and total structure square feet (SQFT / ha). Several watersheds have exhibited a one year downward trend in C / ha: Broad Creek and Mattawoman Creek (2011-2012; shifts of -0.3% and -0.9%, respectively), and Harris Creek (2000-2001; a shift of -2.19%), indicating some annual variability is possible that may be due to duplication or omission of records during annual database development. Determination of the exact cause of the trend shifts requires verification of database records and comparison of specific tax records with corresponding parcel maps within suspect sub-watersheds. The time frame for completion of this analysis exceeds that available for completion of this 2014 Federal Aid Report.

Tidal water surface area of each subestuary was estimated using the planimeter function on MDMerlin satellite photographs and maps (www.mdmerlin.net; Table 3-1). Shorelines were traced five-times for each system, and an average area was calculated. The lower limit of each water body was arbitrarily determined by drawing a straight line between the lowest downriver points on opposite shores (the mouth of each system) and the upper limits were to include all waters influenced by tides.

Surveys focused on eleven target species of finfish that fell within four broad life history groups: anadromous (American Shad, Alewife, Blueback Herring, Striped Bass), estuarine residents (White Perch, Yellow Perch), marine migrants (Atlantic Menhaden and Spot), and tidal-fresh forage (Spottail Shiner, Silvery Minnow, Gizzard Shad). With the exception of White Perch, adults 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 Chesapeake Bay (directly or as forage); they are sampled well by 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). Gear specifications and techniques were selected to be compatible with other Fisheries Service surveys.

Ideally, four evenly spaced haul seine and bottom trawl sample sites were located in the upper two-thirds of each subestuary. Nanjemoy and Piscataway creeks were covered sufficiently by three sites. However, during 2011 and 2012, NOAA, who was assisting with sampling, added an additional site in Nanjemoy Creek upstream of our three sites; the data collected during those years were added into all analyses for 2011 and 2012. Sites were not located near a subestuary's mouth to reduce influence of mainstem waters on fish habitat. We used GPS to record latitude and longitude at the middle of the trawl site, while latitude and longitude at seining sites were taken at the seine starting point on the beach.

Sites were sampled once every two weeks during July-September. All sites on one river were sampled on the same day, usually during morning through mid-afternoon. Sites were numbered from upstream (site 1) to downstream (site 4). The crew leader flipped a coin each day to determine whether to start upstream or downstream. This coin-flip somewhat randomized potential effects of location and time of day on catches and DO. However, sites located in the middle would not be as influenced by the random start location as much as sites on the extremes because of the bus-route nature of the sampling design. If certain sites needed to be sampled on a given tide then the crew leader deviated from the sample route to accommodate this need. Trawl sites were generally in the channel, adjacent to seine sites. At some sites, seine hauls could not be made because of permanent obstructions, SAV beds, or lack of beaches.

Water quality parameters were recorded at all sites. Temperature (ºC), DO (mg / L), conductivity (mS / cm), salinity (‰), and pH were recorded at the surface, middle, and bottom of the water column at the trawl sites and at the surface of the seine site. Mid-depth measurements were omitted at sites with less than 1.0 m difference between surface and bottom. Secchi depth was measured to the nearest 0.1 m at each trawl site. Weather, tide state (flood, ebb, high or low slack), date, and start time were recorded for all sites.

Dissolved oxygen concentrations were evaluated by watershed against a target of 5.0 mg / L and a threshold of 3.0 mg / L (Batiuk et al. 2009; Uphoff et al. 2011a). This target DO is considered sufficient to support aquatic life needs in Chesapeake Bay (Batiuk et al. 2009) and has been used in a regulatory framework to determine if a water body is meeting its designated aquatic life uses. The target criterion was associated with asymptotically high presence of target species in bottom channel habitat in brackish subestuaries (Uphoff et al. 2011a). Presence of target species declined sharply when bottom DO fell below the 3.0 mg / L threshold (Uphoff et al. 2011a). In each subestuary, we estimated the percentages of DO samples that did not meet the target or threshold for all samples (surface to bottom) and for bottom waters alone. 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.

Associations of dissolved oxygen (DO) depth (surface and bottom) estimates from 2003–2014 were tested with correlation analysis. Correlations of DO (surface and bottom) with temperature depth (surface and bottom) and watershed development (C / ha) by salinity class were used to evaluate whether DO stratification would occur and to observe if DO stratification was associated with salinity classification.

Conductivity measurements were collected at each site in every system from July to September. Conductivity measurements recorded in 2012-2013 were recorded incorrectly. The raw conductivity was recorded instead of the specific conductivity, which compensates for temperature. An equation was used to correct the error and convert the raw conductivity measurements that were recorded to specific conductivity (Fofonoff and Millard 1983):

$$\text{Specific Conductivity} = \text{Conductivity} / (1 + ((0.02 \cdot T) - 25));$$

for each ºC change in water temperature (T) there was a 2% change in conductivity.

Each subestuary was classified into a salinity category based on the Venice System for Classification of Marine Waters (Oertli, 1964). Salinity influences distribution and abundance of fish (Hopkins and Cech, 2003; Cyrus and Blaber, 1992; Allen, 1982) and DO (Kemp et al. 2005). Uphoff et al. (2012) calculated an arithmetic mean of all bottom salinity measurements over all years available to determine salinity class for each subestuary. Tidal-fresh ranged from 0-0.5 ‰; oligohaline, 0.5-5.0 ‰; and meshohaline, 5.0-18.0 ‰ (Oertli, 1964). Mattawoman Creek, Piscataway Creek, and Northeast River were classified as tidal-fresh subestuaries (Table 3-1). Gunpowder River, Bush River, Middle River, and Nanjemoy Creek were considered oligohaline. Broad Creek, Harris Creek, and Tred Avon River, were mesohaline subestuaries (Table 3-1). We grouped data by these classifications when examining effects of development.

An additional water quality parameter, total ammonia nitrogen (TAN; mg / L), was measured during July, August, and September 2014 in Mattawoman Creek. Total ammonia nitrogen sampling was conducted using an YSI 9500 Photometer at each of the four stations in Mattawoman Creek in three different locations, (1) channel, (2) edge of SAV bed, and (3) in the middle of the SAV bed. A total of twelve samples were taken and tested for each sampling date, a thirteenth sample was taken in the channel (to minimize debris-particulates) to use as the “blank” sample. Each water sample was taken just below the surface (0.5 m) and poured into a test tube. Reagents were then crushed into each sample (except for the blank) and allowed to process for exactly ten minutes before inserting the sample into the Photometer for a reading. The “blank” was tested in between each sample. The YSI 9500 Photometer has a "minimum detection limit" of 0.05 mg / L. A reading greater than or equal to 0.05 mg / L would be a “true detection reading”, indicating that TAN was detected in the water at some known level. Sampling for TAN occurred on Mattawoman trawl sample dates (twice a month for three months) and were conducted after sampling with 3.1 m trawls (see below). Therefore, TAN samples were randomly collected at different times during morning through mid-afternoon and during different tidal stages.

In addition to monthly sampling, 24-hour sampling was conducted on July 31, 2014, in the fullest SAV bed present at Mattawoman Creek (near Station 1) to look at diel response of ammonia in a dense SAV bed. A water sample was collected every hour starting at 12:00am on July 31st and ending at 12:00am on August 1st. The water sample was poured into four test tubes, one test tube was identified as the blank and three additional test tubes were processed as samples with the same protocol described previously. After ten minutes, each sample was read and recorded. In addition to TAN sampling, surface and bottom temperature, DO, pH, conductivity, and salinity were recorded.

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 codend, with an untreated 12 mm stretch-mesh knotless mesh liner. The headrope was equipped with floats and the footrope was equipped with a 3.2 mm chain. The net used 0.61 m long by

0.30 m high trawl doors attached to a 6.1 m bridle leading to a 24.4 m towrope. Trawls were towed in the same direction as the tide. The trawl was set up tide to pass the site halfway through the tow, allowing the same general area to be sampled regardless of tide direction. A single tow was made for six minutes at 3.2 km / hr (2.0 miles / hr) per site on each visit. The contents of the trawl were emptied into a tub for processing.

During 2009-2014, a 3.1 m box trawl made of 12.7 mm stretch-mesh nylon towed for five minutes was used on the same day sampling was conducted with a 4.9 m trawl in Mattawoman Creek to create a catch-effort time-series directly comparable to monitoring conducted during 1989-2002 (Carmichael et al. 1992). The initial choice of net to start with on each day in Mattawoman Creek was decided by a coin flip.

An untreated 30.5 m 1.2 m bagless knotted 6.4 mm stretch mesh beach seine, the standard gear for Bay inshore fish surveys (Carmichael et al. 1992; Durell 2007), 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 to 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 washtub for processing. The distance the net was stretched from shore, maximum depth of the seine haul, primary and secondary bottom type, and percent of seine area containing aquatic vegetation were recorded.

All fish captured were identified to species and counted. Striped Bass and Yellow Perch were separated into juveniles and adults. White Perch were separated into three categories (juvenile, small adults and harvestable size) based on size and life stage. The small adult White Perch category consisted of ages-1+ White Perch smaller than 200 mm. White Perch greater than or equal to 200 mm were considered to be of harvestable size and all captured were measured to the nearest millimeter. White Perch of this size or larger corresponded to the quality length category minimum (36-41% of the world record TL) proposed by Anderson (1980) for proportional stock density (PSD) indices; 200 mm TL is used as the length cut-off for White Perch in Chesapeake Bay assessments of White Perch (Piavis and Webb 2013). Small adult and harvestable White Perch were combined for adult counts. Catch data were summarized and catch statistics were reported for both gears combined and each gear separately.

Three basic metrics of community composition were estimated for subestuaries sampled: geometric mean catch of all species, total number of species (species richness), and species comprising 90% of the catch. The geometric mean (GM) was estimated as the back-transformed mean of $\log_e$-transformed catches (Ricker 1975; Hubert and Fabrizio 2007). Prior to this report, only the arithmetic mean of catches (AM) was reported. The GM is a more precise estimate of central tendency of fish catches than the AM, but is on a different scale than the AM (Ricker 1975; Hubert and Fabrizio 2007). Both values are given in this report for comparison to prior reports and for future reports. We noted which target species were within the group that comprised 90% of fish collected. We summarized these metrics by salinity type since some important ecological attributes (DO and high or low SAV densities) appeared to reflect salinity class (Uphoff et al. 2012).

We plotted number of species collected by seine or 4.9 m trawl against C / ha and denoted salinity class on these two plots. A greater range of years (1989-2014) was available for seine samples than the 4.9 m trawl (2003-2014) due to a change from the 3.1 m trawl used during 1989-2002 (Carmichael et al. 1992). This was an exploratory analysis because not all subestuaries and years had C / ha estimates. The same plot was constructed for GMs of total catch. These plots would provide insight on how salinity class and C / ha influenced species richness and total abundance. We set a minimum number of samples (15) for a subestuary in a year to include estimates of species richness. This eliminated years where sampling in a subestuary had to be ended due to site losses, typically from SAV growth, that did not permit sampling throughout a season. We plotted the total number of species by their respective number of trawl or seine samples collected to see if we could detect an influence of sample size on accumulation of species (Kwak and Peterson 2007). If a linear or non-linear relationship of richness was suggested, a suitable regression was run. If significant at $\alpha \leq 0.05$, the residuals were used as an effort-corrected time-series of relative richness (above or below average, with the average indicated by 0) plotted against C / ha to examine whether a trend in species richness might be suggested.

We discovered an error in Nanjemoy Creek data during 2011-2012 due to station identifications being switched; stations were correctly identified using the latitudes-longitudes recorded at the time of sampling and matched to the correct sites. Errors were also discovered in Mattawoman 2009-2011 trawl data, the 3.1 m box trawls and 4.9 m headrope semi-balloon otter trawls were not correctly identified; trawls were correctly identified from field data sheets.

We continued to track bottom DO, SAV area, finfish abundance and finfish species richness in 3.1 m and 4.9 m trawl samples from Mattawoman Creek and compared them to changes in C / ha.

We obtained measurements of total ammonia nitrogen (TAN; NH_3 plus NH_4; US EPA 2013) in Mattawoman Creek during the SAV growing season (April-October) from Chesapeake Bay Program (CBP; 2015) monitoring site MAT0016, located in the channel between our stations 3 and 4 (W. Romano, MD DNR, personal communication). Estimates were available for 1986-2012, but we eliminated 1986-1990 from analysis because of methodology differences. During 1991-2009, TAN samples were collected twice a month, only the first TAN sample of each month was selected for analysis (except in 1991, the first sample during May was not available so the second sample was selected) to correspond equally with the number of samples in the following years 2010-2014 (N=7). In 2014, only 6 TAN samples were used in analysis because samples were not collected in July. Measurements of growing season TAN were annually summarized as minimum, median, and maximum and compared to US EPA ambient water quality criteria for TAN (US EPA 2013) to capture the potential for acute and chronic toxicity.

In addition to TAN, we obtained pH and Chlorophyll a (μg / L) readings from the MD DNR Continuous Monitoring Program at their Mattawoman Creek monitoring site located at Sweden Point Marina (MD DNR 2015). Chlorophyll a and pH readings were collected at fifteen minute intervals during late March-early April through October. Estimates were available from 2004 – 2014. Chlorophyll a sample sizes varied from year to year depending on when sampling began; 2004 had the lowest amount of recorded readings (17,416) and 2007 had the highest (21,501). All measurements were used to estimate an annual minimum, median, and maximum. Samples sizes for pH varied: 2004 had the lowest (15,975) and 2006 had the highest (21,497). All measurements were used in the annual analysis of minimum, median, and maximum.

Sampling with 3.1 m trawls was conducted during 1989-2002 and 2009-2014 and 4.9 m trawls have been used since 2003. Geometric means of total fish abundance and their 95% CI's were estimated for the 3.1 m and 4.9 m trawls for samples from Mattawoman Creek. We compared trends of GMs of total fish abundance in the years in common for the 3.1 m and 4.9 m trawls in Mattawoman Creek using linear regression.

Estimates of species richness in Mattawoman Creek (number of species encountered) were made for 3.1 m trawl samples during 1989-2002 and 2009-2014. Sampling during 1989-2002 was based on monthly sampling of five stations (Carmichael et al. 1992). Station 5, sampled during 1989-2002, was dropped because it was outside the range of stations 1-4 sampled during 2009-2014. Remaining stations were the same throughout the time-series, but were sampled monthly during 1989-2002 (annual N = 12) and bi-monthly during 2009-2014 (annual N = 24). In order to match the annual sample sizes of 1989-2002, we made two sets of estimates for each sample year during 2009-2014: one for the first round of the month and one for the second. As a result, all comparisons of species richness in Mattawoman Creek were based on the same annual sample size.

Results and Discussion

Gunpowder, Nanjemoy, Mattawoman, Bush, and Piscataway did not have DO readings less than the target level (5.0 mg / L) during 2014 (Table 3-2). Remaining subestuaries had non-zero estimates of V_{target} in surface and bottom waters. Thirteen percent of DO measurements from Tred Avon River were below the target (V_{target} = 13%); Broad Creek, 8%; Northeast, 2%; Middle River, 2%; and Harris Creek, 2%. When we evaluated V_{target} in bottom channel waters, Tred Avon River and Broad Creek both had the highest estimate at 21%; followed by Middle River, Harris Creek, and Northeast at 4%, all other subestuaries had V_{target} estimates of zero. In 2014, no subestuaries had measurements of bottom DO below the 3 mg / L threshold (Table 3-2).

Correlation analyses of 2003-2014 data suggested that the sign and significance ($\alpha \leq 0.05$) of associations of mean surface or bottom DO with C / ha were influenced by salinity classification in a manner consistent with potential for stratification (Table 3-3). In mesohaline subestuaries, where strongest stratification was expected, the association between bottom DO with C / ha was negative and significant (r = -0.54, $\alpha < 0.0001$), while remaining comparisons at the mesohaline and other salinity classifications were not. Given that multiple comparisons were made, the positive correlation of bottom DO with C / ha for tidal-fresh subestuaries ($\alpha = 0.03$) was considered spurious (Nakagawa 2004). Sample sizes of mesohaline subestuaries (N = 55) were over twice as high as oligohaline (N = 31) or tidal-fresh subestuaries (N = 29), so ability to detect significant associations in mesohaline subestuaries was greater (Table 3-3).

During 2014, dense SAV prevented seining in Mattawoman and Piscataway Creeks. Seining in Middle River was sporadic because of high tides that limited beach availability and dense SAV in seine sites; only two seine sites were available when tide and SAV allowed. Additional seine sites sampled in Middle River and Nanjemoy Creek for NOAA's Integrated Assessment were dropped since NOAA terminated field collections. In Gunpowder River, one seine site (Site 2) was not sampled at all after it was roped off for swimming.

Geometric mean of seine hauls ranged from 41 to 356 fish during 2014, with little indication that salinity class or development level exerted an influence (Table 3-4). Interestingly, two adjacent oligohaline subestuaries (Middle and Bush rivers) had the highest and lowest GMs. Number of species estimated for Middle River (5 seine hauls; 2 of the seine hauls caught no fish) was excluded from analysis. Remaining subestuaries had 10-24 samples. Bush River, an oligohaline tributary, had the greatest number of species (27) during 2014. The three mesohaline subestuaries had 25, 22, and 20 species (Table 3-4).

A total of 33,378 fish representing 49 species were captured by beach seine in 2014 (Table 3-4). Eight species comprised 90% of the total fish caught in 2014, including Atlantic Silverside (31%), Atlantic Menhaden (20%), Gizzard Shad (15%), YOY White Perch (13%), Spottail Shiner (4%), Striped Killifish (3%), Banded Killifish (2%), and YOY Striped Bass (2%). White Perch (juveniles), Gizzard Shad, Striped Bass, and Atlantic Menhaden represented four target species that were among species comprising 90% of the total catch (Table 3-4). Nine target species were present among species comprising 90% of the seine catch (dominant species) when viewed by subestuary; White Perch (juvenile) were present in six of the eight subestuaries seined; Atlantic Menhaden in four; Gizzard Shad and Spottail Shiner in three; Yellow Perch (juvenile) and Striped Bass (juvenile) in two; Silvery Minnow, Alewife and Blueback Herring in one (Table 3-4). Three of five subestuaries with White Perch (juvenile) comprising 90% of the catch were oligohaline. Three out of four subestuaries where Atlantic Menhaden were observed were mesohaline. Remaining target species were not among dominant species collected by seine (Table 3-4).

Bottom trawling with a 4.9 m headrope trawl was conducted in all ten subestuaries in 2014. A total of 89,140 fish and 43 fish species were captured (Table 3-5). Four species comprised 90% of the total catch for 2014, Bay Anchovy (36%), White Perch (juvenile) (36%), Spottail Shiner (13%) and White Perch (adults) (5%). White Perch (juveniles and adults) and Spottail Shiner were target species (Table 3-5).

Geometric mean trawl catches during 2014 were between 159 and 580 (Table 3-5). Subestuaries had 18-24 samples; except Bush River, which had 15. Number of species captured by trawl in subestuaries sampled during 2014 (17-28) overlapped for all three salinity classifications (Table 3-5).

White Perch (juveniles) were among species comprising 90% of 4.9 m trawl catches in 7 of the 10 subestuaries (Table 3-5). Bay Anchovy were the most frequent species comprising 90% in 8 subestuaries. Mattawoman Creek had the highest total catch at 18,135 (580 GM CPUE, respectively) and Tred Avon had the lowest total catch at 5,633 (181 fish per trawl, respectively). Mattawoman Creek had the highest GM (580) and the Harris Creek had the lowest GM (159 collections made; Table 3-5).

Species richness in seine mesohaline subestuaries appeared to be influenced by effort, while bivariate plots did not suggest a relationship for tidal-fresh or oligohaline subestuaries (Figure 3-2). Plots of species richness and C / ha did not suggest a relationship in tidal-fresh or oligohaline subestuaries (Figure 3-3). Tidal-fresh subestuary watersheds were represented by a limited range of C / ha (0.43 - 0.72) that fell between the rural watershed target and suburban threshold. Oligohaline subestuary watersheds were represented by the widest range of C / ha (0.09 - 3.33, rural to urban) of the three salinity classes (Figure 3-3).

Similar to what was found with seine samples, species richness in 4.9 m trawl collections from mesohaline subestuaries that met the effort criterion appeared to be influenced by effort (Figure 3-4). Bivariate plots did not suggest a relationship for tidal-fresh or oligohaline subestuaries (Figure 3-5). Again, high scatter of the relationship of species richness and 4.9 m trawl effort in mesohaline subestuaries made selection of a suitable function difficult and we selected a linear regression. This linear regression was significant ($P = 0.003$, $N = 40$), but explained only 22% of variation.

In general these exploratory analyses of species richness and development supported trends found in analyses of development and DO. Levels of bottom DO were not negatively influenced by development in tidal-fresh or oligohaline subestuaries, but were in mesohaline subestuaries. Depletion of bottom DO in mesohaline subestuaries to hypoxic or anoxic levels represented a direct loss of habitat to be occupied. Uphoff et al. (2011b) determined that the odds of target species (same as in this report, less the tidal-fresh forage component) being present in seine samples from mesohaline subestuaries were not influenced by development, but odds of target species being present in bottom channel trawl samples were negatively influenced. The extent of bottom channel habitat that can be occupied does not appear to diminish with development in tidal-fresh and oligohaline subestuaries due to low DO. However, 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); it was not feasible for us to sample fish within the beds so the impact on target finfish could not be estimated.

The level of development in Mattawoman Creek's watershed more than doubled between 1989 (0.43 C / ha) and 2011 (0.91 C / ha; Figure 3-6). This watershed reached the threshold for suburban development (C / ha = 0.83) in 2006 (Figure 3-6).

There appeared to be two periods of bottom DO in the Mattawoman Creek time-series (Figure 3-7). Mean bottom DO was near or above the median for the time-series (8.5 mg / L) during 1989-2000 (C / ha $\leq$ 0.67) and then fell below the median afterward (with the exceptions of 2003, 2013 and 2014). Mean bottom DO in 2014, 8.4 mg / L, was very near the median for the time-series. Annual mean bottom DO has never fallen below the target of 5.0 mg / L and excursions below this level were rare (Figure 3-7). These shifts in bottom DO corresponded to changes in Mattawoman Creek's subestuary chlorophyll a from high (16-40 µg / L) to low (< 15 µg / L) and shift in SAV acreage from low (coverage of ~10% or less of water area) to high (coverage of > 30%; Figure 3-8; Uphoff et al. 2011b; 2012; 2013; 2014).

Monthly TAN samples in Mattawoman Creek only exceeded the "minimum detection limit" of ≥ 0.05 mg / L four times out of 60 samples, while the TAN measurements collected by Chesapeake Bay Program (2015) ranged from 0.003 mg / L to 0.064 mg / L and had a median of 0.022 mg / L during 2014 (Figure 3-9). We expected to see a higher number of readings exceeding the minimum detection limit in the SAV bed, but only saw one reading ≥ 0.05 mg / L; the edge of the SAV bed had two readings ≥ 0.05 mg / L; and the channel had one reading ≥ 0.05 mg / L.

The 24-hour ammonia sampling conducted in a Mattawoman Creek SAV bed (depth < 1 m), indicated that TAN at the surface and DO at the bottom were at levels indicating poor habitat. Estimates of TAN were above the minimum detection limit from 07:40 till 18:58 (Figure 3-10) and followed a diurnal trend. A number of the TAN estimates observed were higher than the maximum TAN measurements collected by CBP (2015) during 2014 (Figure 3-9); however, no TAN measurements were recorded by CBP during the time we conducted our sampling in July 2014. Surface temperatures and pH reached levels where median estimates of TAN met the EPA chronic criterion (US EPA 2013) in late afternoon (at 1510 hours and again at 1758 hours) (Figure 3-10). Water temperatures and pH were lower in bottom sample and the EPA chronic criterion for ammoniawas not met there. However, bottom DO estimates were below the threshold level (3.0mg / L) for 9 of 12 daylight samples (Figure 3-10). Surface DO only fell below target level (5.0 mg / L) once and bottom DO only went above target level twice during the 24 hours sampled (Figure 3-10). Measurements of TAN during April-October, 1991-2014 exhibited two time periods (Figure 3-9) corresponding to those observed for bottom DO (Figure 3-7) and SAV (Figure 3-8). Median TAN was low and stable at 0.01 mg / L or lower through 2000 and then began a rapid rise to a spike of 0.08 mg / L in 2002 (Figure 3-9). Median TAN dropped after 2002, but was elevated beyond that seen prior to 2001; during 2007-2009, median TAN was consistently elevated at 0.03 mg / L. Estimates of median TAN were generally much closer to minimum than maximum estimates. Maximum estimates of TAN were 2-6 times higher than their respective medians, while differences between the minimum and median were much less (Figure 3-9).

Measurements of pH collected from the MD DNR continuous monitoring site (MD DNR 2015) during April – October, 2004 – 2014 exhibited a slight increase in median pH levels and range since 2012 (Figure 3-11). Median pH levels have remained relatively stable since 2005, fluctuating between 7.70 and 8.24. The most dramatic episodes occurred from 2004 to 2005 when median pH dropped from 8.27 to 7.70 and during 2007 and 2010 when pH reached the highest levels observed (9.88 and 9.84, respectively; Figure 3-11). High pH, primarily greater than 9, promotes ammonium (NH_4^+) to change to toxic ammonia nitrogen (NH_3), causing fish kills (US EPA 2013). During our 24-hour ammonia sampling, surface pH observed in Mattawoman Creek was often near maximum pH estimates observed at the MD DNR continuous monitoring site at Sweden Point Marina during 2004 to 2014 (Figures 3-10 and 3-11). Nineteen surface pH values observed were above 9, with the maximum pH level of 10.14 occurring at 17:58 (Figure 3-10).

Geometric mean catches and their 95% CIs for 3.1 m and 4.9 m trawls in Mattawoman Creek are presented in Table 3-5. The linear regression of GM catches of 4.9 m and 3.1 m trawls during 2009-2014 indicated that their trends were closely and linearly related ($r^2 = 0.99$, $\alpha = 0.001$, N = 6). The slope was significant ($\alpha = <0.0001$), but the intercept was not ($\alpha = 0.32$) and we predicted missing portion of the 3.1 m trawl GM time-series from the slope alone (Figure 3-12). The span of GMs in the regression was similar to those that were predicted, so values did not have to be extrapolated beyond limits of data. The full 3.1 m GM time-series (observations and predictions) suggested total abundance became much more variable after 2001. During 1989-2001, minimum, maximum, and median GM catches of all species were 30.3, 111.7, and 48.7, respectively; during 2002-2014, minimum, maximum, and median GM catches of all species (predictions for missing years included) were 2.3, 196.1, and 32.3, respectively (Figure 3-12).

Species richness in 3.1 m trawl samples declined between 1989-2002 and 2009-2012 (Figure 3-13). During 1989-2002, minimum, maximum, and median number of species collected annually were 8, 19, and 14 respectively; during 2009-2014, minimum, maximum, and median annual number of species collected annually were 5, 20, and 11, respectively (Figure 3-13). Between 1989-2002 and 2009-2014, YOY White Perch were largely unchanged, but presence of adult White Perch declined noticeably. Planktivorous Blueback Herring, Alewife, Gizzard Shad, and Bay Anchovy declined drastically and were replaced by Spottail Shiners. Pumpkinseeds and Bluegills were among the dominant species during 2001 and 2009-2010 (Figure 3-15).

White Perch (YOY) and Spottail Shiners became he only target species to qualify as dominant (part of the species that comprise 90% of catch) in 4.9 m trawls after 2011 (Figure 3-16). Since 2003, planktivores have been uncommon and adult White Perch have dropped out of the dominant category (Figure 3-16).

Mattawoman Creek's finfish abundance appeared to be susceptible to boom and bust dynamics after 2001. "Busts" were concurrent with spikes (2002) or plateaus (2007-2009) of TAN (Figure 3-10). Collapses of the magnitude exhibited during 2002 and 2008-2009 were not detected previously (Figure 3-12). Uphoff et al. (2010) determined that the collapse of abundance in 2008-2009 was local to Mattawoman Creek and not widespread in the Potomac River. Recovery of fish abundance since 2011 has coincided with moderate values of median TAN (Figures 3-10 and 3-12).

Shifts in ecosystem status such as that observed in Mattawoman Creek may represent shifts to different unstable or stable states (shifting baselines or regime shifts, respectively) of ecological systems rather than steady declines (Steele and Henderson 1984; Duarte et al. 2009). The term "regime shift" has been used to suggest jumps between alternative equilibrium states are nonlinear, causally connected, and linked to other changes in an ecosystem (Steele 1996; Duarte et al. 2009). The regime shift concept implies that different regimes have inherent stability, so that significant forcing is required to flip the system into alternative states (Steele 1996). Eutrophication is one of these forcing mechanisms (Duarte et al. 2009), while urbanization creates a set of stream conditions (urban stream syndrome; Hughes et al. 2014a; 2014b) that qualifies as a shift as well. Both of these processes (eutrophication and urban stream syndrome) are inter-related products of development in Mattawoman Creek's watershed. Sediment loads in Mattawoman Creek from construction and stream bank erosion were high (Gellis et al. 2009) and increased nutrient loading there was strongly associated with sediment level increases that occurred after 2003 (J. Uphoff, MDDNR, unpublished analysis of USGS data obtained by W. Romano, MDDNR). Approaching and breaching the development threshold in Mattawoman Creek's watershed has been concurrent with changes in stream hydrology and water quality, increased sediment and nutrient loading from stream erosion and construction, decreased chlorophyll a (a powerful indicator of ecosystem response to nutrients; Duarte et al. 2009) and DO, increased water clarity, TAN and SAV, and more variable and less diverse finfish abundance (particularly planktivores) in the subestuary (Gellis et al. 2009; Uphoff et al 2009; 2010; 2011b; 2012; 2013). When evaluated in the context of Chesapeake Bay Program's habitat goals, Mattawoman Creek superficially resembles a restored system with reduced nutrient loads, i.e., increased clarity, reduced chlorophyll a, and increased SAV. Together, these factors were expected to increase habitat for fish (Chesapeake Bay Program 2014). However, Chanat et al, (2102) reported that nutrient and sediment loads in Mattawoman Creek were nearly twice those of the Choptank River, an agriculturally dominated watershed twice the size of Mattawoman Creek. Boyton et al (2012) modeled nutrient inputs and outputs in Mattawoman Creek and found that nutrients were not exported out of the subestuary, suggesting that wetlands, emergent vegetation, and SAV in Mattawoman Creek were efficiently metabolizing and sequestering nutrients. The fish community has become highly variable and less diverse under these conditions. Duarte et al. (2009) analyzed responses of phytoplankton of four coastal ecosystems to eutrophication and oligotrophication and found diverse and idiosyncratic responses. An expectation that ecosystems would revert to an expected reference condition was unsupported (Duarte et al. 2009). During 2014, we further explored a hypothesis that water quality dynamics in Mattawoman Creek's extensive SAV beds (low DO, high pH, and high organic matter) may be creating episodes of ammonia toxicity for fish. Our 24-hour study suggested that fish could be caught in a habitat squeeze in SAV from high ammonia at the surface and low DO at the bottom.

Ammonia is considered one of the most important pollutants in the aquatic environment because it is both common and highly toxic (US EPA 2013). Ammonia toxicity in fish is heavily influenced by pH; temperature and salinity are considered minor influences (Randall and Tsui 2002). Low DO may lead to positive feedback of nutrient cycling that enhances NH_4 levels (Testa and Kemp 2012). The toxic substance profile for ammonia developed by the United Kingdom's Marine Special Areas of Conservation Project (2001) determined that toxicity of ammonia increased with low DO.

Breakdown of organic matter is a source of ammonia (US EPA 2013). Macrophyte beds have high primary productivity and are an important source of organic matter (Caraco and Cole 2002). The microorganisms of decay assimilate some of the organic material in the dead remains to build their cells (Cole 1975). Other organic material is converted to ammonia. This, in turn, is oxidized to nitrite and then to nitrate. Both aerobic and anaerobic bacteria function in ammonification, while only aerobic forms participate in nitrification (Cole 1975). Low DO in SAV beds can impact this biogeochemical cycle within the bed (Caraco and Cole 2002).

Some species of SAV create low DO conditions and introduced species, in particular, may induce hypoxia (Caraco and Cole 2002). Uphoff et al. (2011b) found low DO patches were not uncommon within an extensive SAV bed in Mattawoman Creek and DO conditions were generally worse within the SAV bed than in bottom channel waters. Introduced Hydrilla and Eurasian Milfoil are commonly encountered in Mattawoman Creek and often form dense beds (J. Uphoff, MD DNR, personal observation). In general, SAV are two orders of magnitude less sensitive to ammonia than aquatic invertebrates and vertebrates (US EPA 2013).

If toxic ammonia contributed to episodic "disappearances" of Mattawoman Creek's estuarine fish community, it either did so at levels well below EPA's acute criteria for aquatic life (US EPA 2013) or at levels much greater than indicated by TAN monitoring in the channel at MAT0016. Our sampling during 2014 supports the latter hypothesis. While we rarely encountered TAN above detection levels in our channel-based monitoring, detectable TAN and conditions of concern were present in a 24-hour study of a dense SAV bed. Under the temperature and pH conditions used by US EPA (2013) for chronic ammonia conditions (longer term reductions in survival, growth, or reproduction), the range of TAN maximum measurements at MAT0016 (0.08-0.015 mg / L) and the Sweden Point continuous monitor measurements (in SAV) of pH and temperature indicate a potential match for pH from 8.6 to 9.0 at water temperatures from 21 to 30 ºC during 2004-2011. Anecdotally, we have observed multiple fish kills in Mattawoman Creek since the early 2000s. Some have followed tournament releases of Largemouth Bass at Sweden Point Marina; at least one was fairly widespread and involved multiple species.

Randall and Tsui (2002) criticized ammonia criteria for fish because they were based on starved, resting, non-stressed fish. These criteria did not protect swimming and stressed fish, nor did criteria reflect that feeding reduced the toxicity response. Fish may employ strategies, such as reduced ammonia excretion, that ameliorate ammonia toxicity (Randall and Tsui 2002).

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

Area	Watershed	Subestuary	IS	C/ha	Total Hectares	Water Hectares	Salinity Class
Mid-Bay	Lower Choptank	Broad Creek	5.1	0.29	4,730	3,148	Mesohaline
Mid-Bay	Lower Choptank	Harris Creek	6.0	0.39	3696	2,919	Mesohaline
Mid-Bay	Gunpowder River	Middle River	23.4	3.33	2,753	982	Oligohaline
Mid-Bay	Lower Choptank	Tred Avon River	9.2	0.75	9,563	2,429	Mesohaline
Potomac	Lower Potomac	Mattawoman Creek	10.3	0.90	24,441	729	Tidal Fresh
Potomac	Lower Potomac	Nanjemoy Creek	2.4	0.09	18,893	1,131	Oligohaline
Potomac	Upper Potomac	Piscataway Creek	14.0	1.47	17,642	361	Tidal Fresh
Upper-Bay	Bush River	Bush River	14.1	1.49	36,038	2,962	Oligohaline
Upper-Bay	Gunpowder River	Gunpowder River	9.0	0.73	113,760	4,108	Oligohaline
Upper-Bay	Elk River	Northeast River	6.8	0.47	16,342	1,579	Tidal Fresh

Table 3-2. Percentages of all DO measurements and bottom DO measurements that did not meet target (= 5.0 mg/L) and threshold (= 3.0 mg/L) conditions during July-September, 2014, for each subeastuary. C/ha = structures per hectare.

			All DO	Bottom DO	
Subestuary	Salinity Class	C/ha	% < 5.0 mg/L	% < 5.0 mg/L	% < 3.0 mg/L
Broad Creek	Mesohaline	0.29	8	21	0
Harris Creek	Mesohaline	0.39	2	4	0
Middle River	Mesohaline	3.33	2	4	0
Tred Avon River	Mesohaline	0.75	13	21	0
Gunpowder River	Oligohaline	0.73	0	0	0
Nanjemoy Creek	Oligohaline	0.09	0	0	0
Bush River	Oligohaline	1.49	0	0	0
Mattawoman Creek	Tidal Fresh	0.90	0	0	0
Northeast River	Tidal Fresh	0.47	2	4	0
Piscataway Creek	Tidal Fresh	1.47	0	0	0

Table 3-3. Correlations of 2003-2014 arithmetic mean annual surface and bottom DO (mg/L) with like water temperatures at depth (surface and bottom) or watershed development (C/ha = structures per hectare), by salinity class.

DO Depth	Statistics	Temperature Depth	C/ha
Mesohaline			
Surface	r	-0.1383	0.03071
	α	0.314	0.8238
	N	55	55
Bottom	r	-0.07187	**-0.54952**
	α	0.6021	**<.0001**
	N	55	**55**
Oligohaline			
Surface	r	**-0.38375**	**0.3698**
	α	**0.0331**	**0.0406**
	N	**31**	**31**
Bottom	r	**-0.37525**	-0.13612
	α	**0.0375**	0.4653
	N	**31**	31
Tidal-fresh			
Surface	r	0.02168	0.34369
	α	0.9111	0.0679
	N	29	29
Bottom	r	0.10554	**0.39768**
	α	0.5858	**0.0327**
	N	29	**29**

Table 3-4. Beach seine catch summary, 2014. C/ha = structures per hectare. GM is the geometric mean catch of all fish per seine.

River	Stations Sampled	Number of Samples	Number of Species	Species Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Broad Creek	3	15	25	Atltantic Menhaden Atlantic Silverside Striped Killifish Striped Bass (YOY) White Perch (YOY)	0.29	4594	199
Bush River	4	20	27	Gizzard Shad White Perch (YOY) Altantic Menhaden Spottail Shiner White Perch (Adult)	1.49	5386	204
Gunpowder River	3	10	24	Spottail Shiner White Perch (YOY) Yellow Perch (YOY) Gizzard Shad Tesselated Darter Banded Killifish Inland Silverside Pumpkinseed	0.73	1501	74
Harris Creek	3	18	20	Atlantic Silverside Atlantic Menhaden	0.39	8396	356
Middle River	2	5	12	White Perch (YOY) Banded Killifish Bluegill Largemouth Bass (YOY) Pumpkinseed Yellow Perch (YOY)	3.33	406	41
Nanjemoy Creek	3	18	25	White Perch (YOY) Atlantic Silverside Inland Silverside Mummichug Silvery Minnow Bay Anchovy White Perch Spottail Shiner	0.09	2682	105
Northeast River	4	24	25	Gizzard Shad White Perch (YOY) Bay Anchovy Blueback Herring White Perch Alewife	0.47	4628	151
Tred Avon River	4	24	22	Atlantic Silverside Atlantic Menhaden Striped Killifish Banded Killifish Striped Bass (YOY)	0.75	5785	162
Grand Total	26	134	49	Atlantic Silverside Atlantic Menhaden Gizzard Shad White Perch (YOY) Spottail Shiner Striped Killifish Banded Killifish Striped Bass (YOY)		33378	

Table 3-5. Bottom trawl (4.9m) catch summary, 2014. C/ha = structures per hectare. GM is the geometric mean catch of all fish per seine.

River	Stations Sampled	Number of Samples	Number of Species	Species Comprising 90% of Catch	C / ha	Total Catch	GM CPUE
Broad Creek	4	24	19	Bay Anchovy	0.29	11746	384
Bush River	3	15	17	White Perch (YOY) White Perch (Adult) Gizzard Shad Bay Anchovy	1.49	8402	528
Gunpowder River	4	24	28	White Perch (YOY) Bay Anchovy Spottail Shiner White Perch Gizzard Shad Pumpkinseed	0.73	6411	218
Harris Creek	4	24	18	Bay Anchovy	0.39	7546	159
Mattawoman Creek	4	24	24	White Perch (YOY) Spottail Shiner	0.9	18135	580
Middle River	4	24	23	Bay Anchovy White Perch (YOY) White Perch	3.33	7856	251.3
Nanjemoy Creek	3	18	22	White Perch (YOY) Bay Anchovy	0.09	7974	395
Northeast River	4	24	20	White Perch (YOY) White Perch Bay Anchovy	0.47	8258	291.1
Piscataway Creek	3	18	21	White Perch (YOY) Spottail Shiner Blueback Herring	1.47	7179	220.6
Tred Avon River	4	24	22	Bay Anchovy Hogchoker Striped Bass (YOY)	0.75	5633	181
Grand Total	37	219	43	Bay Anchovy White Perch (YOY) Spottail Shiner White Perch (Adult)		89140	

Figure 3-1. Subestuaries sampled in 2014, estuarine fish summer sampling.

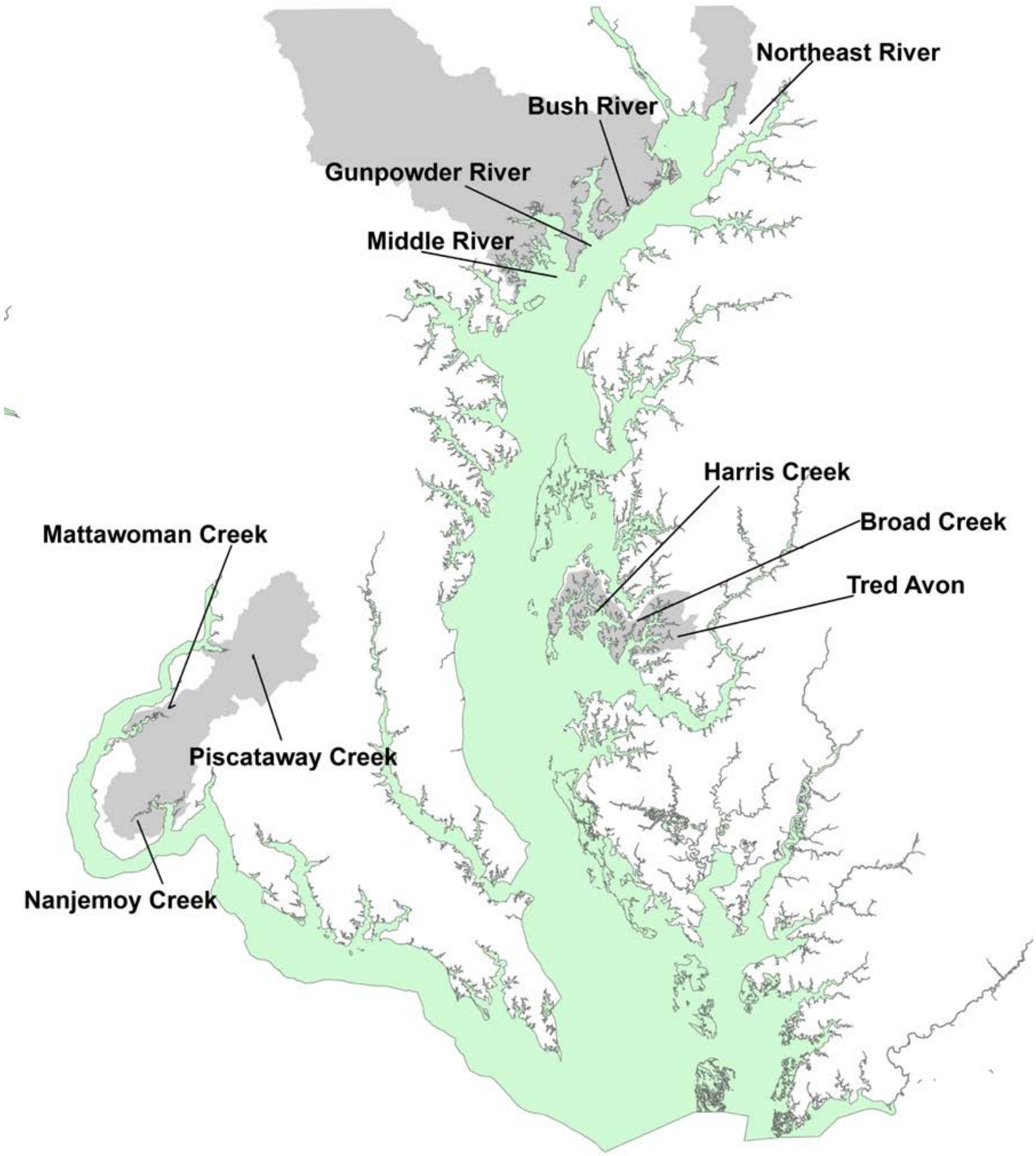


Figure 3-2. Number of species captured annually during 2003-2014 in subestuaries by seining plotted against number of seine samples taken, by salinity class.

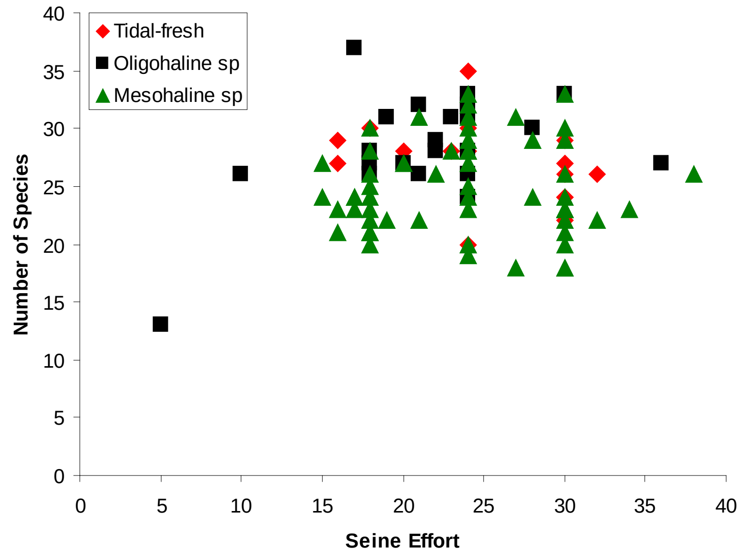


Figure 3-3. Number of finfish species collected by seining in fresh-tidal or oligohaline subestuaries versus intensity of watershed development (C/ha = structures per hectare).

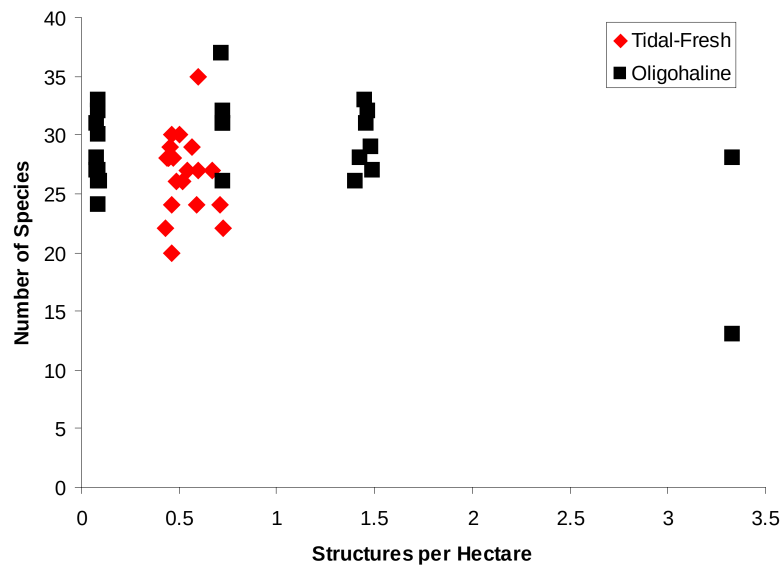


Figure 3-4. Number of species collected by 4.9 m trawl and sample size for tidal-fresh, oligohaline, and mesohaline subestuaries during 2003-2014.

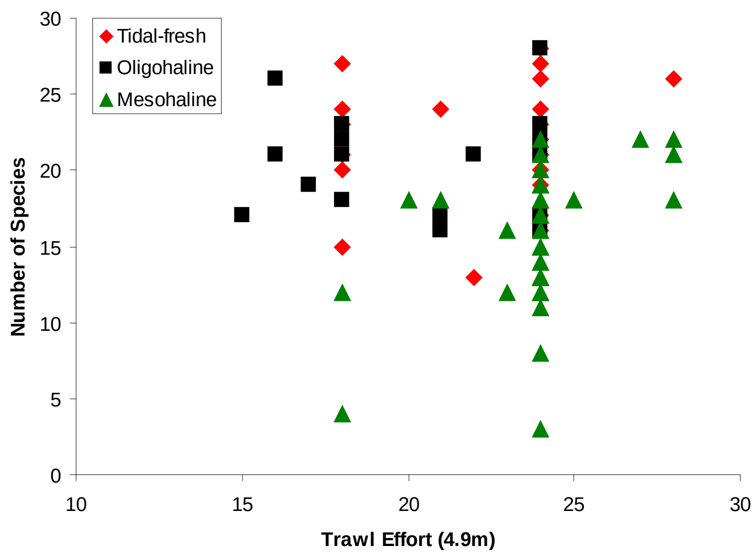


Figure 3-5. Number of finfish species collected by 4.9 m trawl in fresh-tidal or oligohaline subestuaries versus intensity of development (C/ ha = structures per hectare).

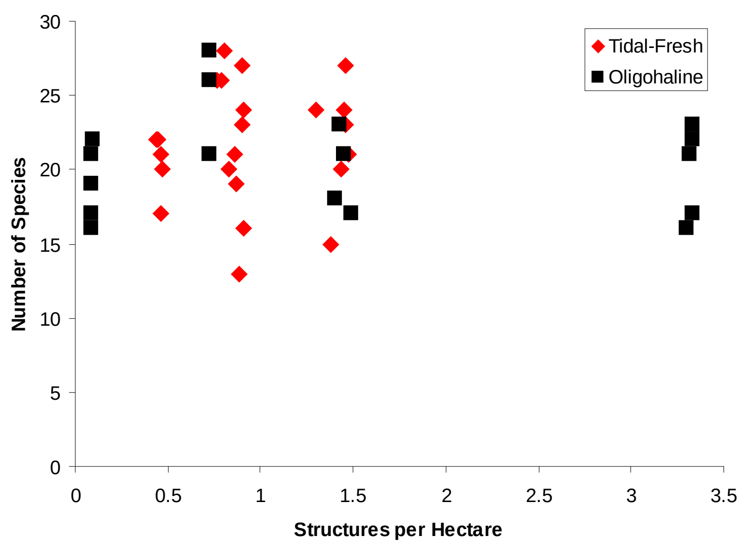


Figure 3-6. Trend in development (structures per hectare or C / ha) of Mattawoman Creek's watershed during 1989-2014. Black square indicates values that are at or beyond the threshold for a suburban watershed.

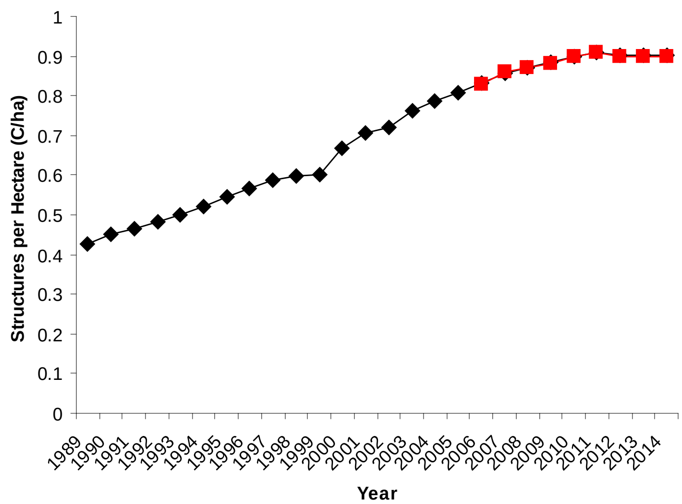


Figure 3-7. Mean bottom dissolved oxygen (DO) during July-September in Mattawoman Creek's subestuary, 1989-2014. Dotted line indicates median for the time-series of annual means.

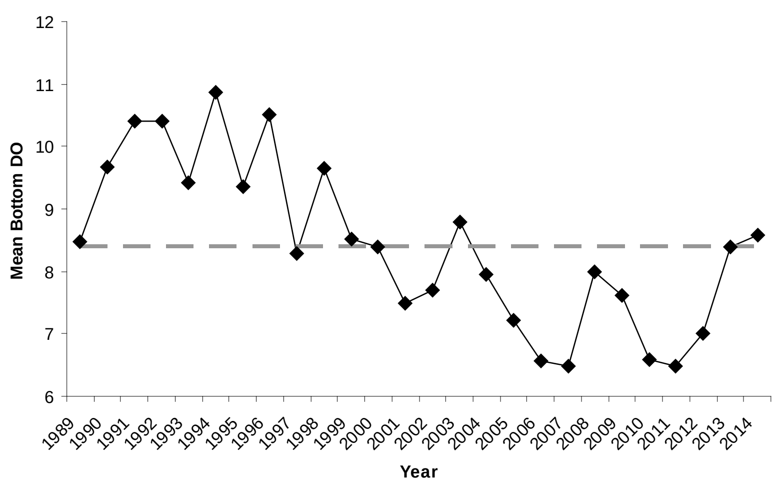


Figure 3-8. Percent of Mattawoman Creek's subestuary covered by SAV during 1989-2014.

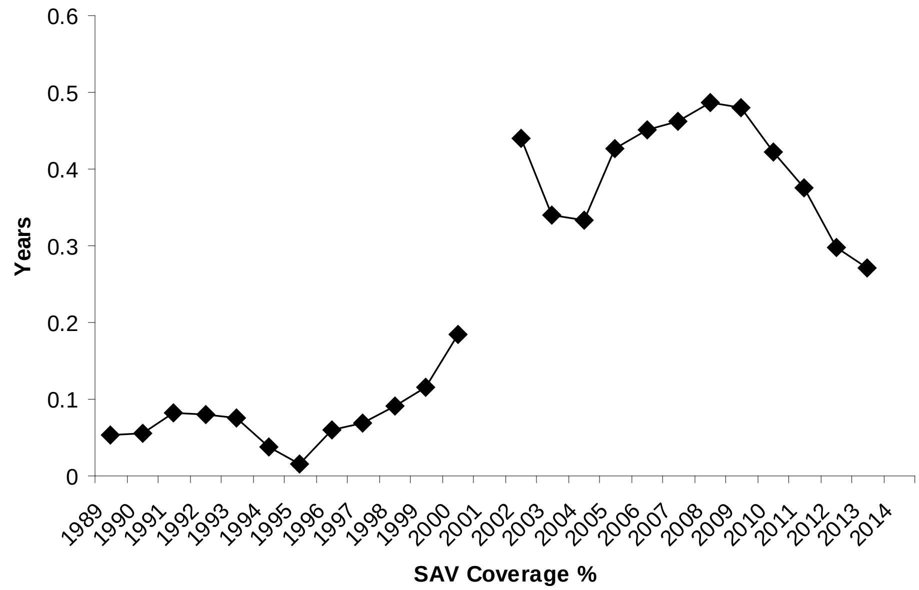


Figure 3-9. Range (solid gray line) and the median total ammonia nitrogen (TAN; mg/L) (solid black line) at a Chesapeake Bay Program monitoring station in Mattawoman Creek (MAT0016) during SAV growing season (April-October).

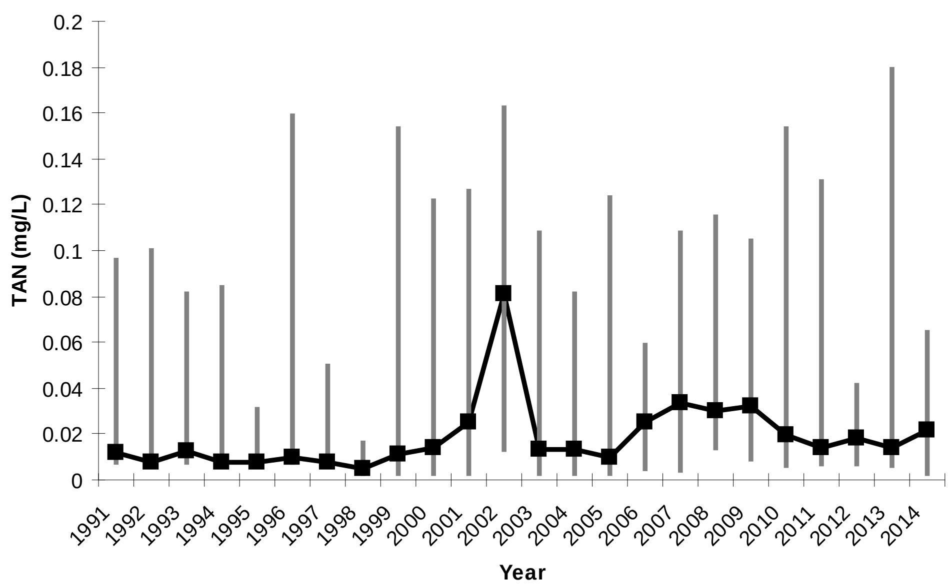


Figure 3-10. Total ammonia nitrogen (TAN; mg/L) estimates in a SAV bed during 24-hour sampling period in Mattawoman Creek. Surface and bottom water quality measurements taken during the ammonia sampling. Total ammonia nitrgoen and the minimum detection limit are located on the primary axis. The minimum detection limit line indicates the level in which a true TAN level is detected by the equipment used. Additional water quality measurements are located on the secondary axis.

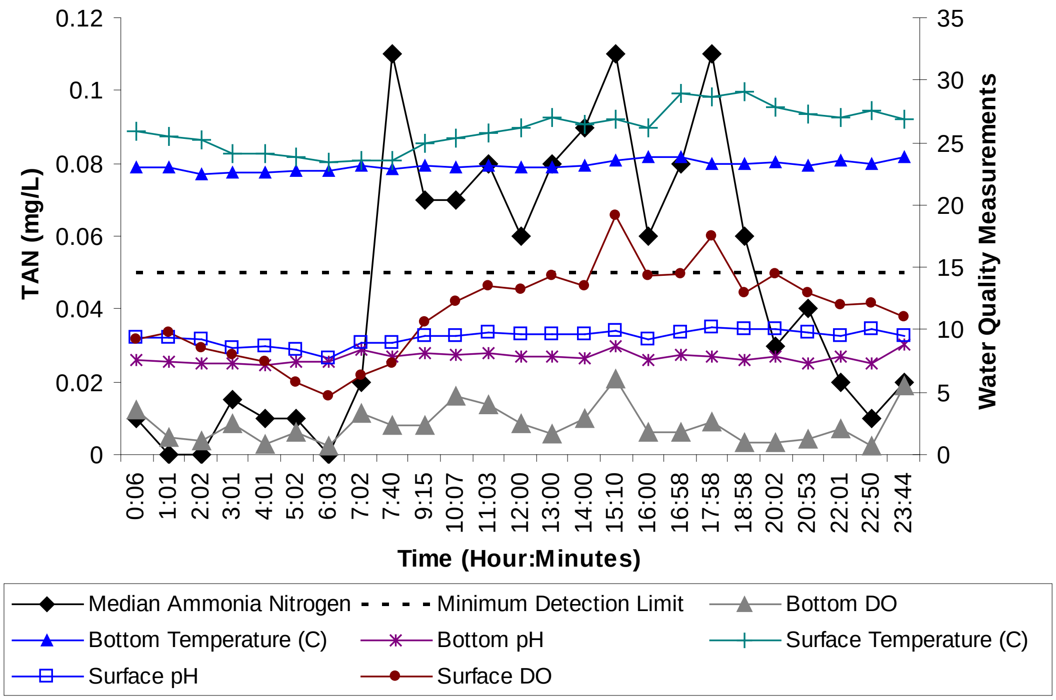


Figure 3-11. Range of pH (solid gray line) and median pH (solid black line) at a MD DNR Continuous Monitoring station (MD DNR 2015) located in Mattawoman Creek during SAV growing season (April-October).

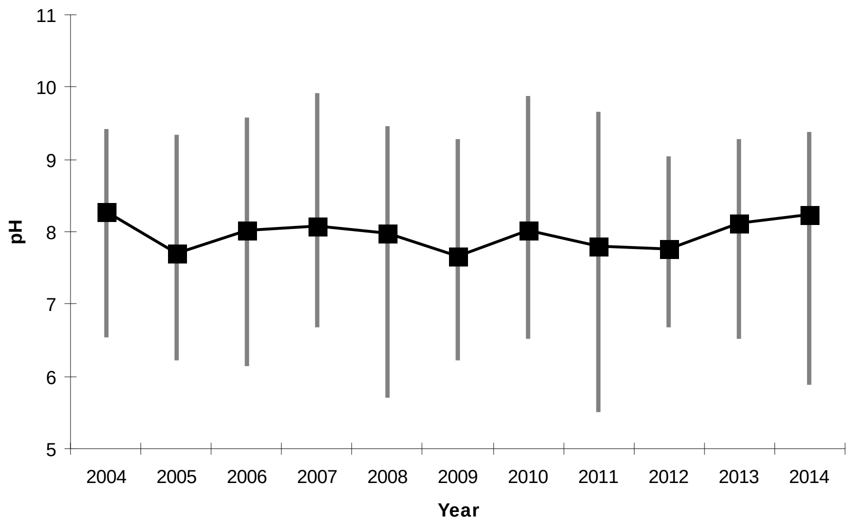


Figure 3-12. Geometric mean (GM) catches per trawl of all species of finfish in Mattawoman Creek during 1989-2014. Note dual axes for 3.1 m and 4.9 m trawls. Predicted 3.1 m GM is based on a linear regression of 3.1 m and 4.9 m trawl GMs during 2009-2014. Dotted horizontal lines indicate median GM of 3.1 m trawl samples for 1989- 2001 (red dotted line) and 2002-2014 (blue dotted line).

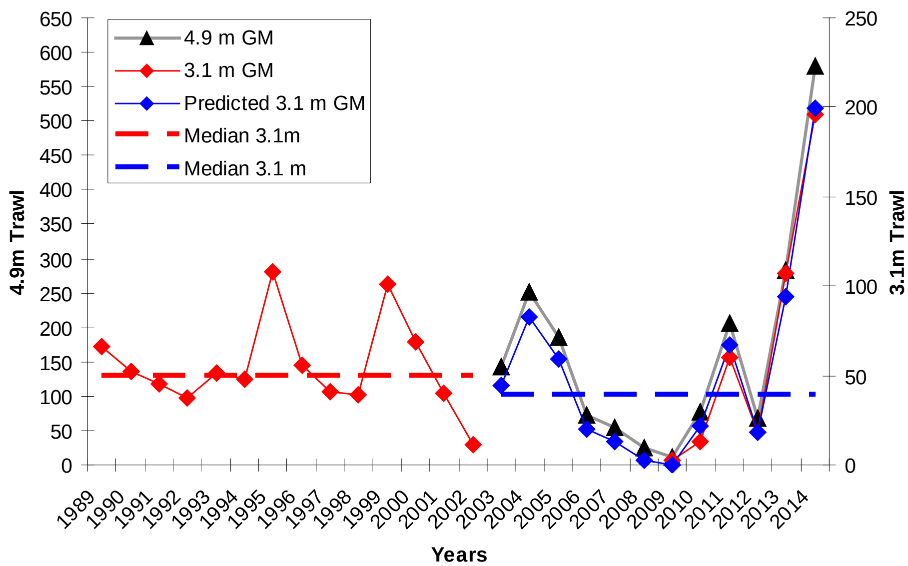


Figure 3-13. Species richness (number of species) in 3.1 m trawl samples during summer sampling. N = 12 for all points. Bimonthly sampling during 2009-2014 allowed for two estimates of N = 12 per year. Median number of species during 1989-2002 is indicated by the green line; median number of species during 2009-2014 is indicated by the red line.

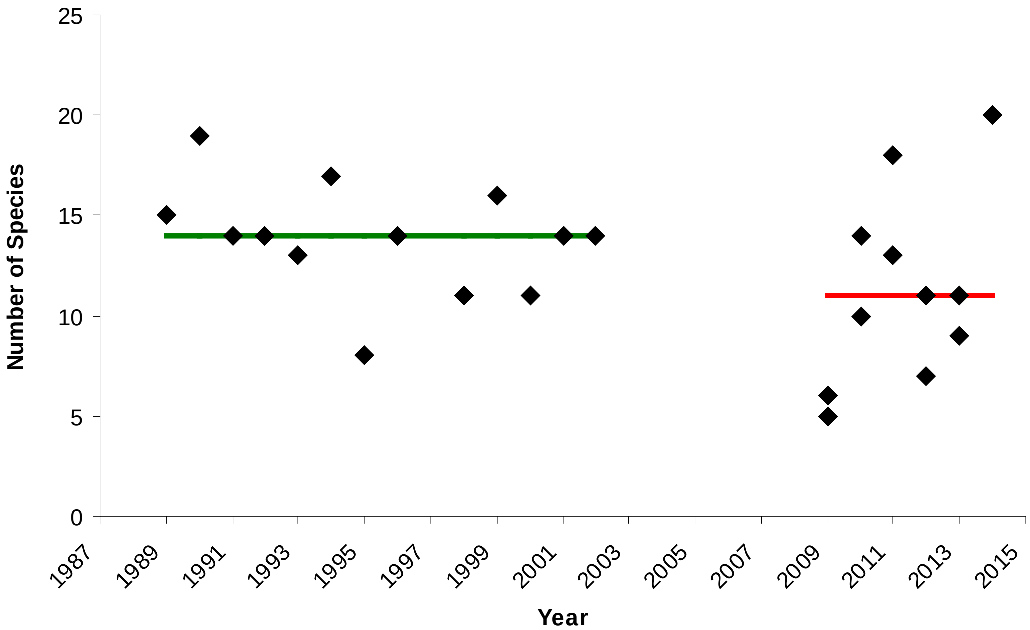


Figure 3-14. Fish species comprising of 90% of 3.1m annual trawl catch for Mattawoman Creek.

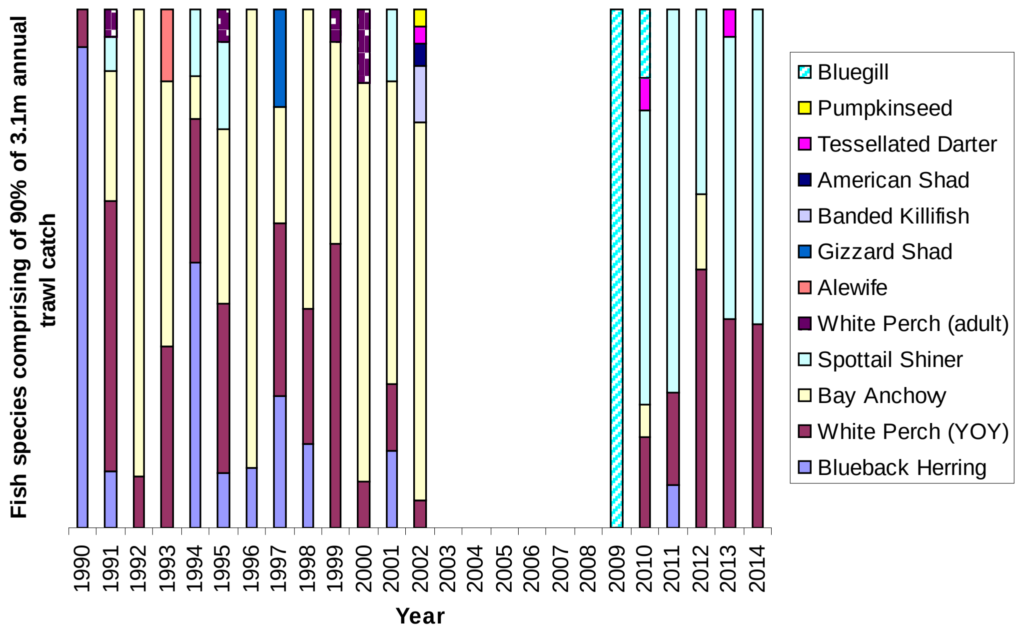
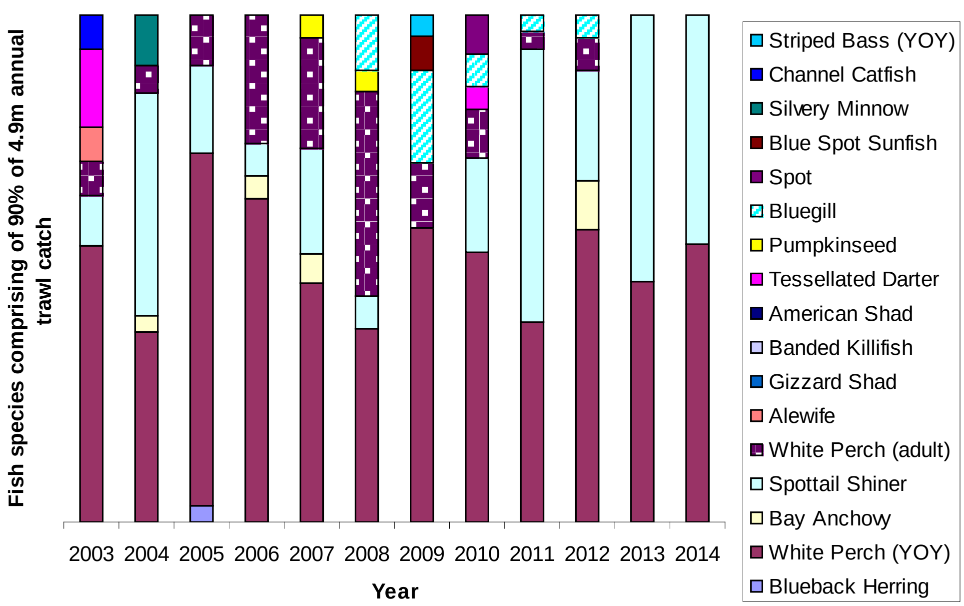


Figure 3-15. Fish species comprising of 90% of 4.9m annual trawl catch for Mattawoman Creek.



<u>SURVEY TITLE:</u> MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

<u>PROJECT 2</u>: HABITAT AND ECOLOGICAL ASSESSMENT FOR RECREATIONALLY IMPORTANT FINFISH

JOB 1: Support multi-agency efforts to assess finfish habitat and ecosystem-based fisheries management.

Jim Uphoff, Margaret McGinty, Alexis Park, Carrie Hoover, Shaun Miller, Brian Redding, and Paul Parzynski

Introduction

The objective of Project 2 was to document participation of the Fisheries Habitat and Ecosystem Program (FHEP) in habitat, multispecies, and ecosystem-based management approaches important to recreationally important finfish in Maryland's Chesapeake Bay and Atlantic coast. Contributions to various research and management forums by Program staff through data collection and compilation, analysis, and expertise are vital if Maryland is to successfully develop an ecosystem approach to fisheries management.

Maryland Fisheries Service – Fisheries Habitat and Ecosystem Program Website

We continued to populate the website with new reports and information to keep it up to date with project developments. We are working on a new website design that will launch in 2015, which will be easier to navigate and contain additional valuable information.

Publications

Uphoff, J. H., Jr. and Co-Authors. 2014. Marine and estuarine finfish ecological and habitat investigations. Performance Report for Federal Aid Grant F- 63-R, Segment 4. Maryland Department of Natural Resources, Annapolis, Maryland.

Environmental Review Unit Bibliography Database

FHEP staff continues to compile an Environmental Review Unit database, adding recent literature and additional topics including effectiveness of Best Management Practices.

Review of County Comprehensive Growth Plans

M. McGinty reviewed plans for Talbot and Charles County and supplied comments to MD DNR.

MD DNR Interagency Effort on Mattawoman Creek

M. McGinty supported a collaborative effort by MD DNR to draft a fact sheet to describe the challenges in the present Charles County Tier map. The adopted map allowed for 9,000 acres to be designated as Tier II, which are areas to be served by sewer. This designation allows for these acres to be rezoned from a 1:10 density to high density subdivision. If this development is permitted, it will cause irreparable harm to Mattawoman Creek and losses in fisheries. Staff also drafted a form letter and provided talking points for those who want to give a verbal testimony at the upcoming public hearing. Our comments and testimony are consistent with recommendations from Maryland Department of Planning The Charles County Board of County Commissioners voted to designate the 9000 acres as Tier IV (low density).

M. McGinty presented to the Charles County Board of County Commissioners information on the impacts of impervious surface to ecosystems. Commissioners asked M. McGinty to describe the expected fisheries impacts if Mattawoman Creek is over-developed. M. McGinty outlined various losses in recreational and commercial habitat that can have local quality of life impacts and regional economic impacts.

J. Uphoff supplied comments on requested expansion of Charles County's sewage treatment plant to handle infiltration of groundwater. The expansion was inconsistent with projected growth and would result in a great deal of excess capacity that could accommodate hundreds of years of growth. It appeared that the county may reconsider the proposal.

Database Development

Scientific Collection Permits (SCP) are issued by the State to groups (agencies, organizations, individuals) who wish to legally try to collect finfish, shellfish, other target species, or data in the State of Maryland waters. They in return submit a report on their findings providing the location, date, species collected, number count, and gear used, or any other parameters collected.

Through the auspices of the State of Maryland, the permit coordinator authorizes the validity of the requests based on standard parameters consisting of location, time of year, gear used, type of species targeted and number collected, and use of such data. Other restrictions may be applied based on newly updated regulations.

These findings from the collectors are then scrutinized to identify which data would be appropriate to include in a database. Data from 2003 to present has been archived and the invaluable information is now being prioritized and recorded.

After ascertaining whether the data is from tidal or nontidal waters, the tidal waters finfish data was entered in 2013 on an Excel worksheet. It consisted of permit number, location (coordinates if available), scientific (genus/species) as well as common name of fish, number of fish, collecting agency and any pertinent comments.

The SCP data from year 2003 to present will be used in conjunction with MBSS's nontidal data for future map plotting and verifications.

P. Parzynski used ArcGIS 10.1 to build a GIS database using water quality and fish presence/absence data from the SCP data, VIMS's ChesMMap and NEAMAP programs, and the Chesapeake Bay Program's data library. The creation of these geospatial relationships will help FHEP better indentify critical fish habitat. These maps are intended to shed light on main bay juvenile habitat to compliment FHEP's data on spawning habitat.

Cooperative Research

M. McGinty, A. Park, C. Hoover, S. Miller, B. Redding, and P. Parzynski supported field sampling efforts of various state and federal projects including: the DNR's Coastal Bays Program, Resident Fish Species, Fish Passage, the Alosid Project, and the Fish Health Program.

J. Uphoff collaborated with NOAA and the University of Maryland Eastern Shore on a project for a graduate student that would process and analyze RNA-DNA samples from Yellow Perch larvae collected by FHEP staff. This arrangement results in FHEP getting these data at no cost and provides a graduate student data for a thesis.

A. Park and C. Hoover worked with the Resource Assessment Service program on developing housing units for conductivity data loggers that would be deployed in four Chesapeake tributaries to continuous record conductivity measurements over a year.

M. McGinty, A. Park, and C. Hoover worked with staff from MD DNR Inland Fisheries and RAS to coordinate summer sampling in Mattawoman Creek. The purpose of this work is to further investigate the impacts of development in Mattawoman Creek. We will be focusing on understanding the benefits and impacts of ammonia and SAV on water quality and fish habitat.

A. Park, C. Hoover, S. Miller, and B. Redding worked with the Fish Health Program at the Oxford Lab assessing Striped bass stomach contents collected from the middle and lower Chesapeake Bay.

P. Parzynski worked with Regulations with entering citation data into an access database.

M. McGinty worked with the Tidal Bass Program and RAS about coordinating monitoring to aid an investigation of changes in the Potomac River's tidal Largemouth Bass fishery. In conjunction w/ the LMB work in Mattawoman, staff is working w/ TEA to support the two continuous monitors in Mattawoman Creek.

J. Uphoff worked with MD DNR Environmental Review to provide comments on a water withdrawal permit on Tuckahoe Creek.

M. McGinty assisted MD DNR RAS program in the required 10 year review for the State Wildlife Grant, Species of Greatest Conservation Need (SGCN). No tidal species were listed in the previous assessment. M. McGinty reviewed the regional list to determine if the status of these regional species warrants listing them in need of conservation.

We, along with additional MD DNR programs, worked to identify key metrics to be included in the Bay Report Card. Staff provided indicators of development with a brief description of land-use impacts on fish habitat. This will include development thresholds and what they mean in a fisheries management context.

Presentations and Outreach

The following technical presentations were given during the project year.

Tracking and Understanding Changes: 25 Years of Monitoring Mattawoman Creek (and more) at the 20th Annual Maryland Water Monitoring Annual Conference.

Managing Land Use, Fish Habitat and Fisheries in the Chesapeake Bay at MD DNR staff professional development day held by the Conservation Education Matrix Team.

Managing Land Use, Fish Habitat and Fisheries in the Chesapeake Bay at the 2014 MET Land Conservation Conference.

Evaluating Land Use, Fish Habitat and Fisheries at the Quarterly Critical Area Commission Meeting.

Linking Fall Diets of Striped Bass in Maryland's Portion of Chesapeake Bay to Proposed Nutrition Reference Points and Other Ecological Indicators at the Chesapeake Bay Program's Fisheries Goal Implementation Team meeting in Gloucester, VA.

Linking Fall Diets of Striped Bass in Chesapeake Bay to Nutrition Reference Points at the Annual American Fisheries Society Conference in Quebec City.

Managing Chesapeake Bay's Land Use, Fish Habitat, and Fisheries at the Restore America's Estuaries conference.

Quantifying effects of habitat change for management. Presentation to the ASMFC Habitat Committee

Anadroumous herring and development. Presentation for a regional ASMFC and The Nature Conservancy habitat scoping workshop.

What lurks in the SAV, too much a good thing? at the 20th Annual Maryland Water Monitoring Annual Conference. FHEP staff provided the data and were co-authors of the poster.

A. Park organized and led sampling and fish identification training at the 16th Annual Bush River Wade in. M. Margaret, A. Park, and C. Hoover led a volunteer training for spring ichthyoplankton sampling for the Anita C. Leight Estuary Center staff and volunteers to resurrect the Larval Sampling program that occurred between 2005 and 2008 in the Bush River by holding an informational meeting and training session. This program was re-initiated to determine the quality of migratory fish spawning habitat in the Bush River. A. Park and C. Hoover presented sampling results and led volunteer summer juvenile fish sampling training for the Anita C. Leight Estuary Center staff and volunteers. The Bush River is one of our sampling areas. This volunteer group samples the Bush River and provides data to the project.

C. Hoover participated in TEAM training, through which staff will be certified to assist with outreach activities. TEAM is dedicated to educating elementary and middle school students about the Chesapeake Bay and other natural resource issues in Maryland by helping students understand and care for their natural environment.

C. Hoover attended the National Aquarium Aquatic Teacher workshop and was trained in various programs and techniques to improve effectiveness in conducting both fisheries and wildlife outreach/education. C. Hoover worked with Aquarium staff and local teachers on issues relevant to fisheries and wildlife health and management and how human activities can have a great impact on these and other natural resources.

M. McGinty, A. Park, C. Hoover, and P. Parzynski attended the 20th Annual Maryland Water Monitoring Annual Conference and presented *Tracking and Understanding Changes: 25 Years of Monitoring Mattawoman Creek (and more)* at the *Maryland Streams: Are They Getting Much Better, or Not So Much?* seminar. The presentation communicated results from the last 25 years of monitoring, the work of the citizen scientists, and the Charles County Comp Plan. The presentation included data provided by FHEP staff.

J. Uphoff, M. McGinty, A. Park, and C. Hoover attended the 2014 Chesapeake Bay River Herring Workshop at the Smithsonian Environmental Research Center in Edgewater, MD and presented on FHEP's spring sampling procedures and the data collected. The meeting addressed updates on Chesapeake Bay river herring research and monitoring conducted in Maryland, Delaware, and Virginia during 2013, discussed plans and explored possible collaborations for river herring research and monitoring in 2014, and discussed the formation of a Chesapeake Bay River Herring Monitoring Network that would work towards bay-wide monitoring of river herring spawning runs.

M. McGinty attended the Potomac River Fisheries Commission's Water Quality Exchange and presented findings on landuse impacts on fisheries and fish habitat.

M. McGinty presented *Managing Land Use, Fish Habitat and Fisheries in the Chespeake Bay* at the 2014 Maryland Environmental Trust Land Conservation Conference. The presentation focused on impacts of land development and emphasizing the need to conserve rural landscapes. The presentation was part of the Healthy Watersheds Session and complimented a presentation by Scott Stranko (MD DNR) highlighting MBSS results.

M. McGinty presented *Evaluating Land Use, Fish Habitat and Fisheries* at the Quarterly Critical Area Commission Meeting. There were many local planning representatives present. M. McGinty mentioned the Habitat Workgroup's interest in collaborating with localities to promote sound planning for people and fish and invited input from the localities in this effort.

B. Redding assisted MD DNR Educational staff with the *Raising Horseshoe Crabs in the Classroom* program field day.

J. Uphoff attended the ASMFC and The Nature Conservancy river herring workshop. The objective of the workshop was to identify and prioritize river herring habitat restoration needs in key watersheds of Chesapeake Bay. J. Uphoff presented on FHEP's work on the effect of development on stream spawning of herring and on mapping priority areas. As a result of this presentation, USFW has contacted M. McGinty to provide spatial data to develop a regional assessment to identify key stressors and target priority habitat.

J. Uphoff presented *Linking Fall Diets of Striped Bass in Maryland's Portion of Chesapeake Bay to Proposed Nutrition Reference Points and Other Ecological Indicators* at the Fisheries Goal Implementation Team meeting in Gloucester, VA. Work to date has linked nutritional status of sublegal striped bass in fall to prey consumption, with juvenile menhaden being particularly important. When Striped Bass fed well during 2006-2012, far fewer were vulnerable to starvation, disease, and predation.

J. Uphoff presented *Linking Fall Diets of Striped Bass in Chesapeake Bay to Nutrition Reference Points* at the Annual American Fisheries Society meeting in the Trophic Interaction of Fishes session. Work to date has linked nutritional status of sublegal striped bass in fall to prey consumption, with juvenile menhaden being particularly important.

J. Uphoff presented *Managing Chesapeake Bay's Land Use, Fish Habitat, and Fisheries* at the Restore America's Estuaries conference. J. Uphoff represented a state perspective in a panel discussion on fish habitat at the conference.

Program and Staff Development

M. McGinty participated in the Chesapeake Bay Program workshop titled *Designing Sustainable Stream Restoration Projects within the Chesapeake Bay Watershed.*

J. Uphoff, A. Park, C. Hoover, and B. Redding attended the Bioelectrical Impedance Analysis (BIA) training workshop taught by Kyle Hartmann at the Cooperative Oxford Lab. The training included background of BIA technology and protocol, hands-on training, and discussion of field implementation strategies.

A. Park, C. Hoover, and B. Redding were certified and trained in CPR and First Aid.

The FHEP program received an additional permanent position on July, 2014. The position was filled by A. Park, a long-term contractual with the program since 2011. At the end of 2014, the FHEP program consists of two senior biologists and one mid-level biologist, one mid-level contractual biologist, one contractual entry-level biologist, and one seasonal technician.

ASMFC

J. Uphoff presented to the ASMFC Habitat Committee on how to quantify habitat changes and include them in stock assessments. The primary method used measured reductions in spawning habitat or egg-larval survival to discount egg or spawner biomass per recruit based on normal conditions. This analysis put habitat loss in terms of F reductions needed to compensate for lost habitat.

J. Uphoff was appointed to the ASMFC Biological and Ecological Reference Point Committee and left the Weakfish Technical Committee. J. Uphoff attended and presented on modeling predator-prey dynamics using biomass dynamic models with predation terms (Striped Bass alone and Striped Bass plus Spiny Dogfish) and nutrition reference points for the ASMFC Biological and Ecological Reference Point workgroup. A strategy of using a continuum of indicators, from simple indices of forage, nutrition reference points, simple models and more complex ones emerged as a likely approach. The indicator concepts were reviewed as part of the 2015 menhaden peer-review.

Fisheries Habitat Workgroup

M. McGinty led the first meeting of the Fisheries Habitat Workgroup as MD DNR support staff. The following vision was adopted by the workgroup members: Protect and restore fisheries habitat, using ecosystem-based management and practical understanding of watershed ecology, to educate and engage people and influence decisions and policies, respecting all voices.

<u>SURVEY TITLE:</u> MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

<u>PROJECT 3 TITLE</u>: DEVELOPING PRIORITY FISHERIES HABITAT SPATIAL TOOLS

JOB 1: *Develop spatial data to assist in developing management priorities for protecting priority fish habitat.*

Margaret McGinty

Abstract

This analysis compared how different GIS spatial scales affected interpretation of watersheds with anadromous fish spawning habitat quality and quantity. Our analyses have been at an 8-digit watershed scale, but there have been frequent requests for 12-digit resolution. Percent coverage of high priority habitat (high priority because of high quality) was similar for maps at the 8-digit and 12-digit scale, but mid- and low priority habitat (low priority because of low quality) coverage was quite different. At both scales, the general patterns of high priority watersheds with anadromous spawning habitat in eastern and western Maryland were similar, but the approximate coverage of mid-priority habitat watersheds on the western shore was much greater at the 8-digit scale. Estimated total watershed area of anadromous spawning areas was ~440,000 acres in eastern Maryland (divided at the Susquehanna River and Chesapeake Bay) and ~390,000 acres in western Maryland. Estimated total watershed area of anadromous spawning areas was less for eastern (~414,000 acres) and western Maryland (~232,000 acres) at 12-digit resolution.

Introduction

Maryland's population is projected to increase by 1.1 million people by 2030 (Maryland Department of Planning 2011), with an attendant increase in urbanization to accommodate this growth. Increased development associated with urbanization has been identified as a threat to the health and recovery of the Chesapeake Bay (Chesapeake Bay Program 2014). Uphoff et al. (2011a; 2011b; 2012; 2013; 2014) have identified development (indicated by impervious surface coverage) as a stressor of Chesapeake Bay fish habitat.

Uphoff et al. (2011a) proposed development related (impervious surface or IS) targets and limits for communication of watershed and fisheries management connections. Under watershed development reference points for estuarine fisheries, conservation of natural streams, wetlands, and forests, and agricultural land is paramount. Fisheries management emphasis shifts from harvest control and stocking in rural watersheds to watershed renovation, reengineering and reconstruction as development proceeds from exurb to city. In the Chesapeake Bay region, nearly all watershed land use (zoning) and water quality responsibilities lie with local, state, and federal agencies not involved in fisheries management. Fisheries managers need to effectively communicate potential losses of fish habitat, fishing opportunities, and ecological services in hope that stakeholders, responsible agencies, and governing bodies will favor conservation of rural landscapes needed for fisheries (Uphoff 2011a).

This job describes the approach we are applying to develop spatial tools to delineate key fish habitats of target species using anadromous fish spawning habitat as an example. We updated GIS maps at a smaller watershed scale in response to requests from land-use managers. Our analyses have been at an 8-digit watershed scale, but there have been frequent requests for 12-digit resolution. This analysis compares how different scales affect interpretation of anadromous fish spawning habitat quality and quantity. These spatial tools will support an effort to communicate fisheries management priorities to local land managers.

Methods

Uphoff et al. (2014) described the general approach applied to delineate natural limits to distribution and develop habitat criteria for life stages of target species occurring in Maryland's tidal water. This approach involves 1) identifying the natural range of a species by critical life stages; 2) rating the habitat based on prevalence of a life stage; 3) identifying anthropogenic stressors that limit habitat use; and 4) rating these stressors based on degree of impact on use of habitat. In step 1), we classified habitat of a species' life stage into four categories: preferred (area of high occurrence of a species' life stage), acceptable (area of moderate occurrence), marginal (area of low occurrence) and not suitable (life stage absent). In step 2), we scored watershed habitat by category, assigning 5 to preferred habitat, 3 to acceptable habitat, 1 to marginal, and 0 to habitat where a life stage would be absent. In step 4), we scored stressor categories by assigning a score of 5 to the area with lowest impacts, a 3 to areas with moderate impacts, a 1 to areas with intense impact, and 0 where habitat impacts would render an area unsuitable as fish habitat. We then summed the habitat and stressor scores to derive a total score, then ranked these combined scores into terciles (thirds) that assigned watersheds it to a high, mid-, or low habitat management priority category. See Uphoff et al. (2014), Job 3: Developing Priority Fisheries Habitat Spatial Tools, for a detailed description of this approach.

The following summary for anadromous fish spawning habitat builds on the aforementioned approach and describes changes applied to evaluate habitat on a smaller spatial scale. Previous maps were developed at Maryland's 8-digit watershed scale (Figure 1). The average watershed area at this scale was 4,840 acres. The metadata (MD DNR 1998a) for this 8-digit spatial layer describe derivation of the 8-digit data as follows:

> "This file (SWSUB8) is a statewide digital watershed file. It depicts the State with 138 separate watersheds each with an 8-digit numeric code. The file was created primarily for State and Federal agency use. The creation of this file goes back many years and involved several State and Federal agencies. This file was derived from a more detailed watershed file (Maryland's Third-Order Watershed). The U.S. Natural Resources Conservation Service (NRCS) redefined the third order watersheds creating the HUA14 file. The SUB1998 file contains all of the HUA14 Watersheds and some added Watersheds to maintain water quality sampling sites."

To develop the tool on a finer scale, we obtained Maryland 12-digit watershed data and re-evaluated habitat at this smaller scale (Figure 2). The average watershed area at this scale was 651 acres. This layer is described as follows (MD DNR 1998b):

> "This file (SWSHED12) is a statewide digital watershed file. It was created

primarily for state and federal agency use. The watersheds define Strahler (Strahler 1952 p. 1120) third order stream drainage by contours on U.S. Geological Survey (USGS) 7.5 minute quadrangle map sheets. Some watersheds drainage areas were defined for streams less than third order and some large area watershed were split to maintain a maximum size of 15,000 acres. The watershed boundaries in this file were developed in a joint state and federal effort to create a consistent watershed file for use by all government agencies with an interest in Maryland's watersheds. The U.S. Natural Resources Conservation Service (NRCS) redefined the third order watersheds creating the HUA14 file. This file contains all of the HUA14 watersheds and some added watersheds to maintain water quality sampling sites. It was also used to create the Maryland Sub-Watershed file."

We identified salinity as a natural limiting factor for all early life stages of the target anadromous fishes and the approach for defining anadromous and semi-anadromous spawning salinity preferences remained the same as described by Uphoff et al. 2014. We obtained historical data from a longitudinal survey of eggs and larvae in Chesapeake Bay with a wide range of salinities (Dovel, 1971) to develop cumulative percentage distributions of abundance of eggs by salinity for estuarine spawning. We then fit lines to 3 segments of the cumulative percentage distributions to determine salinity categories by species. Lines with highest slopes (starting at 0 on the x- and y-axes) indicated preferred habitat; the next, more moderate slope indicated acceptable habitat; and marginal habitat was represented by slope with the least amount of change (starting at the end of the acceptable habitat segment) until 100% of the cumulative distribution was reached (Uphoff et al. 2014).

We used interpolated average seasonal bottom salinity data obtained from the Chesapeake Bay Program data to map salinity in the Bay (Tom Parham, MD DNR Resource Assessment Service, personal communication). This data layer contained over 39,000 cells with salinity concentrations defined. We developed an associated data set that ranked each cell for each species' salinity category. For example, cells with salinity between 0 and 1 received a rank of 5 for Alewife, American Shad, Blueback Herring and White Perch spawning habitat. Cells with salinity between 1 and 2, received a rank of 3 for Alewife, Blueback Herring, and White Perch, and so on according to the salinity categories described in Table 1. Once each cell was scored by species, we intersected this layer with the watershed data layer. This delineated salinity scores by watershed for each species. We averaged these scores by watershed to derive an average score for each species and all species combined. This is referred as the habitat score.

To compensate for omission of watersheds that do not intersect tidal waters, we used presence data from historical anadromous fish spawning habitat surveys (Alewife, Blueback Herring, Hickory Shad, American Shad, Yellow Perch, White Perch, and Striped Bass; O'Dell et al. 1975; 1980) to identify nontidal watersheds where spawning was observed in the 1970s. O'Dell et al. (1975; 1980) combined Alosids into a single category (Herring) because identification of eggs and larvae to species can be difficult. Presence of a species or group was based on a single occurrence of an egg, larva, or adult at a site during a survey. Striped Bass were not observed in any freshwater streams because their spawning areas in Maryland are limited to tidal waters (Hollis 1967). Given these data limitations, we identified the number of species present within a non-tidal portion of a watershed and scored them by assigning a 5 when 3 species groups (Herring, White Perch, and Yellow Perch) were present, a 3 to watersheds with 2 groups, and a 1 to watersheds with just one group present. This represented the habitat score for nontidal watersheds (anadromous spawning streams). This ranking approach allowed us to combine tidal and nontidal watersheds into one data set based on a common scoring approach.

Using the approach introduced in Uphoff et al. (2014), we updated anadromous fish spawning habitat maps using IS estimates derived from the most recent (2012) Maryland Property Tax Map data. These tax map data are regularly updated and readily available for estimating levels of development in a watershed. Localities use these data in tax assessments and local and state planners are familiar with the reliability of the data. We use structures per hectare (C / ha) as our development indicator derived from tax map data. Uphoff et al. (2012) developed a nonlinear regression equation to convert annual estimates of C / ha to IS as calculated by Towson University based on 1999-2000 (years in common) satellite imagery. The relationship of C / ha and IS was well described by the equation:

$IS = 10.98\ (C / ha)^{0.63}$, ($r^2 = 0.96$; $P < 0.0001$).

We used IS reference points to assign stressor scores to watersheds: watersheds with less than 5% IS received a stressor score of 5; watersheds with 5% to 10% IS, 3; 10% to 15% IS, 1; and greater than 15% IS, 0.

We combined habitat and stressor scores to form a spawning score for each watershed. The watersheds were ranked by terciles (thirds) based on the spawning score and assigned into a management priority category for habitat quality. Watersheds falling into the upper third of the distribution received a score of 5 (high priority because of high quality), the middle third of the distribution received a score of 3 (mid-priority) and the lower a 1 (low priority because of low quality). Habitat management priorities assigned in this analysis represented our recommendations to the Department of Natural Resources, but did not necessarily constitute a final anadromous fish spawning habitat and watershed policy.

Total acreage of the three priority categories combined was determined for both levels of resolution for eastern and western Maryland, with the split occurring along the Susquehanna River and Chesapeake Bay. Percentages in the four IS categories described above were estimated for each region's watersheds supporting anadromous fish spawning to judge how much influence it may have in each region.

Results and Discussion

At the 8-digit scale, most of the high priority anadromous fish spawning watersheds were located in eastern Maryland (Figure 3), usually along the fresh-tidal to oligohaline regions of major tributaries. High priority watersheds of western Maryland were limited to a region close to the Susquehanna River, the upper tidal Patuxent River, and some watersheds along the tidal mid- Potomac River. Most of the spawning areas of tidal Potomac and Patuxent rivers fell in the mid-priority category (Figure 3). Estimated total watershed area of anadromous spawning areas was greater in eastern Maryland (~440,000 acres) than western Maryland (~390,000 acres). Approximately 32% of anadromous fish stream spawning habitat in eastern Maryland was high priority, 44% was mid-priority, and 24% was low priority at 8-digit resolution. In western Maryland, 21% of stream spawning habitat was in the high priority category, 32% was mid-priority, and 47% was identified as low priority (Table 2).

At the 12-digit scale, most of the high priority anadromous fish spawning watersheds were also located in eastern Maryland (Figure 4, Table 2). However, blank gaps appeared and high priority habitats were less continuous. There was much less mid-priority habitat in western Maryland than identified on the 8-digit maps (Figure 4). Estimated total watershed area of anadromous spawning areas was greater in eastern Maryland (~414,000 acres) than western (~232,000 acres) at 12-digit resolution. Approximately 38% of anadromous fish stream spawning habitat in eastern Maryland was high priority, 43% was mid-priority, and 19% was low priority at 12-digit resolution. In western Maryland, 21% of stream spawning habitat was in the high priority category, 21% was mid-priority, and 58% was identified as low priority.

Percent coverage of high priority habitat was similar for maps at the 8-digit and 12-digit scale, but mid- and low priority habitat coverage was quite different. At both scales, the general patterns of high priority anadromous spawning habitat in eastern and western Maryland were similar, but the approximate coverage of mid-priority habitat on the western shore was much greater at the 8-digit scale.

In the mid- and low priority areas, spawning habitat may be have been limited due to high salinity gradients or impaired by development. In western Maryland, much of the low and mid-priority anadromous fish spawning habitat (59% at 8-digit and 37% at 12-digit) coincided with IS of 10% or more (Table 3; Figure 5). This was not the case in eastern Maryland (3% at 8-digit and 6% at 12-digit with > 10% IS) and we infer that high salinity gradients would be the main spawning habitat limitation in this region's mid- and low priority watersheds (Table 3; Figure 5).

Impervious surface targets and limits for anadromous spawning habitat watersheds were the same as targets and limits outlined in Uphoff et al. (2011a) for summer habitat of target species juveniles and adults in brackish subestuaries (based on a negative relationship of IS and bottom dissolved oxygen). Surveys of anadromous fish stream and estuarine spawning and larval nursery habitat under F-63-R (Uphoff et al. 2011b; 2012; 2013; 2014) have confirmed the applicability of Uphoff et al. (2011a) IS targets and limits to these habitats. Stressors impacting anadromous fish spawning habitat would be different than Uphoff et al. 2011a and would become more numerous as development proceeds (Uphoff et al. 2011b; 2012; 2013; 2014).

Watersheds with IS between 0 and 5% IS, represented watersheds with target levels of development (rural watersheds) where typical fisheries management strategies (harvest management and stocking) should be effective and watershed conservation and renovation are vital for maintaining productive habitat. Watersheds with 5-10% IS (exurban to early suburban) were designated as areas where increased management may compensate for habitat loss through harvest management and stocking; conservation and watershed renovation, reengineering and reconstruction are needed to uphold habitat conditions. Watersheds with 10-15% IS represented areas impacted by development. Fisheries management options (harvest management and stocking) are unlikely to result in sustainable outcomes and habitat should be reengineered and reconstructed to stabilize streams and minimize estuarine water quality impacts. Watersheds beyond 15% IS were considered severely degraded with very few options for fisheries management, although extensive reengineering and reconstruction may confer water quality benefit to estuaries. Project 1, Job1, Section 2 provides an illustration of how development impacts harvest management and attainment of yellow perch fishery targets and limits.

We originally used the 8-digit scale four our mapping needs, reasoning that watershed conditions upstream of spawning habitat contributed to the quality of downstream spawning habitat. This assumption was supported by the river continuum theory (Minshall et al. 1985). While 8-digit maps have been useful in visualizing regions of the state to focus management actions, they lack the smaller scale resolution needed for local conservation and planning. We envision using both scales in the future. The 8-digit scale is useful for state managers to develop regional priorities and the 12-digit scale allows us to target key watersheds within the region to promote specific management actions.

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Table 1. Spawning habitat for anadromous and semi-anadromous target species based on salinity. *Yellow Perch salinity limits were derived from the literature.

Species	Preferred	Acceptable	Marginal	No Occurrence
Alewife	0-1	1-2	2-3	>3
American Shad	0-1			>1
Blueback Herring	0-1	1-2		>2
Striped Bass	0-3	3-9		>9
White Perch	0-1	1-2	2-10	>10
Yellow Perch*	0-2			
Anadromous and Semi-Anadromous Combined	0-1	1-3	3-11	>11

Table 2. Percent of watershed area supporting anadromous fish spawning in management priority categories, by region, for 8- and 12-digit watersheds.

Fisheries Management Priority	Eastern Shore 8-Digit Scale	Eastern Shore 12-Digit Scate	Western Shore 8-Digit Scale	Western Shore 12-Digit Scale
5 (high)	32%	38%	21%	21%
3 (mid)	44%	43%	32%	21%
1 (low	24%	19%	47%	58%

Table 3. Percent of watershed area supporting anadromous fish spawning in impervious surface categories, by region, for 8- and 12-digit watersheds.

Impervious surface	Eastern Shore 8-Digit Scale	Eastern Shore 12-Digit Scate	Western Shore 8-Digit Scale	Western Shore 12-Digit Scale
> 15%	0%	42%	1%	20%
10-15%	3%	17%	5%	17%
5-10%	32%	21%	17%	25%
<5%	65%	20%	77%	38%

Figure 1. Eight-digit watersheds in Maryland.

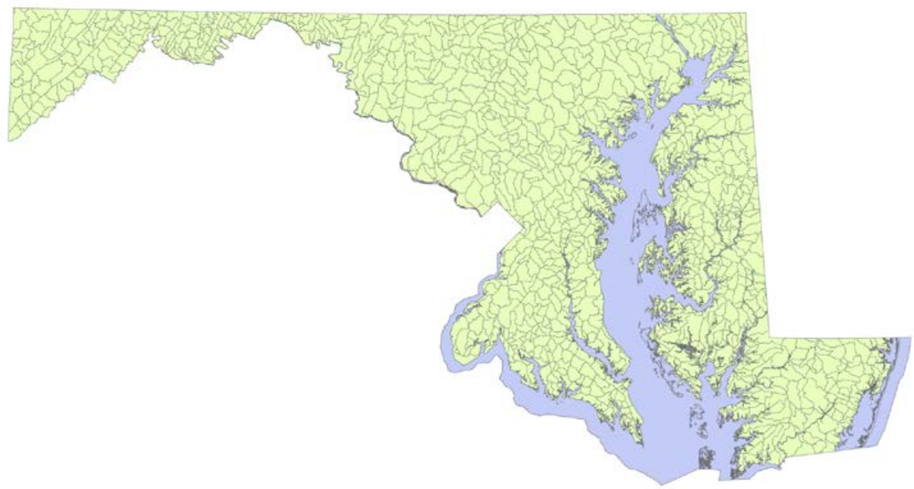

Figure 2. Twelve-digit watersheds in Maryland.

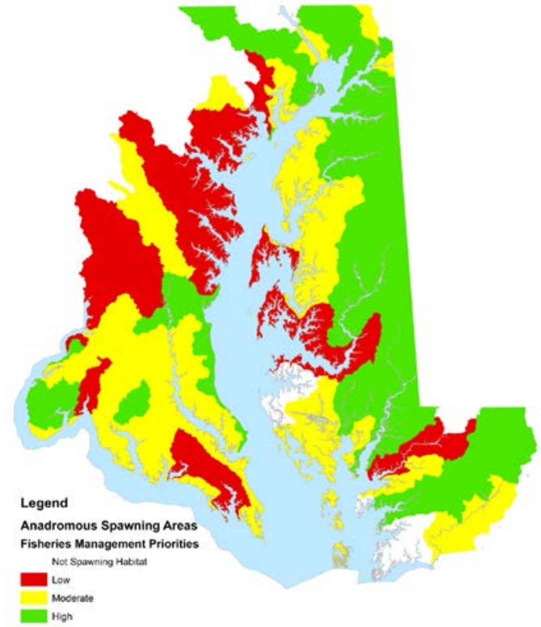

Figure 3. Anadromous spawning watersheds depicted by management priority at the 8-digit watershed scale.

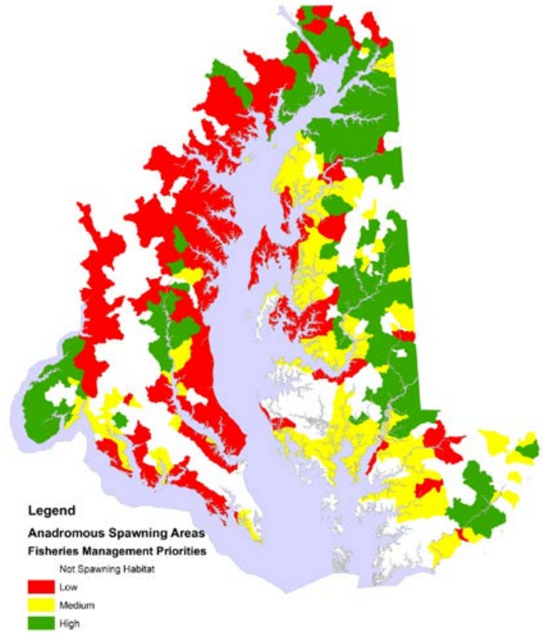

Figure 4. Anadromous spawning watersheds depicted by management priority at the 12-Digit watershed scale.

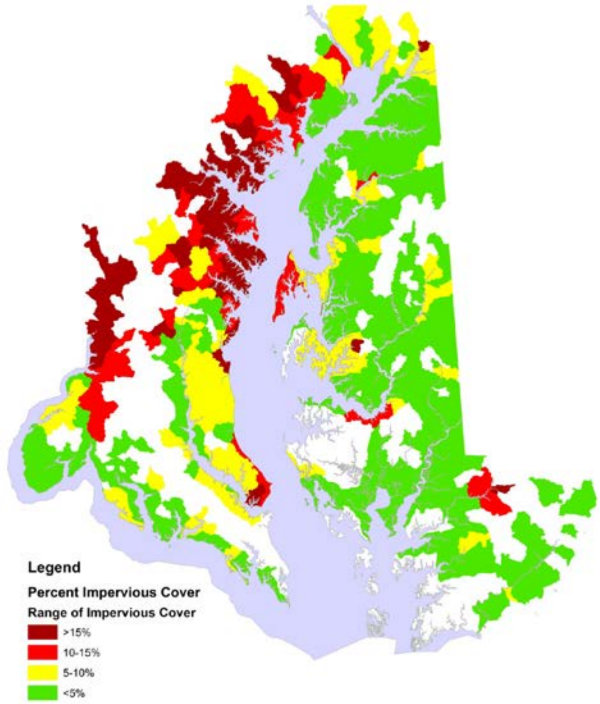

Figure 5. Percentage of Impervious surface, by category, at the 12-digit scale.

SURVEY TITLE: MARINE AND ESTUARINE FINFISH ECOLOGICAL AND HABITAT INVESTIGATIONS

PROJECT 4: FINFISH ECOSYSTEM-BASED MANAGEMENT

JOB 1: Development of ecosystem-based reference points for recreationally important Chesapeake Bay fishes of special concern: striped bass nutrition and forage availability benchmarks

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Executive Summary

Two major activities related to an ecosystem approach to fisheries management were conducted during this report cycle: development of Striped Bass diet database from multi-year, year-round citizen-science based diet monitoring and development of an index-based indicator approach to evaluate forage and resident Striped Bass status in Maryland's portion of Chesapeake Bay.

Data sets were completed for summer (June 21 – September 30) 2007 (N = 631) and 2008 (N = 517). Striped Bass ranged from 209 mm to 914 mm, TL. For any given year and Striped Bass size class combination, three diet items would comprise about 95% of food consumed during summer (Table 4-6). Pelagic fish (Atlantic Menhaden and Bay Anchovy) contributed 59-87% of weight in diet samples of both size-classes. Sublegal Striped Bass diet weight in 2007 was dominated by Bay Anchovy (86%), Blue Crab (8%), and Spot (2%); 2008 diets were dominated by YOY Atlantic Menhaden (66%), Spot (21%), and Bay Anchovy (10%). By weight, large Atlantic Menhaden (ages 1+) predominated in legal-sized Striped Bass diets in 2007 (59%) and 2008 (82%), but were found in a few Striped Bass greater than 700 mm (4% of all fish examined) in June-July. In 2007, Spot (22%) and White Perch (ages 1+; 13%) were also important to legal-sized Striped bass, as were Spot (10%) and YOY Atlantic Menhaden (5%) during 2008. Condition of Striped Bass, indicated by the proportion without body fat (Pf0), deteriorated over the course of the summer. For both size-classes of Striped Bass, Pf0 was lowest during July or July-August and highest in September. During 2007, 91% of legal-sized Striped Bass had empty guts, while 73% were empty in 2008. Legal-size Striped Bass consumed less food during summer 2007 (0.0026 grams of food per gram of Striped Bass) than during summer 2008 (0.0074 grams of food per gram of Striped Bass). These estimates were 15% and 37% of respective estimates for October-November.

Chesapeake Bay Program and Atlantic States Marine Fisheries Commission efforts provided convergent needs for ecological indicators that assess adequacy of forage for resident Striped Bass in Maryland's portion of Chesapeake Bay. Information from ongoing monitoring was used to devise five annual forage indicators for resident Striped Bass in Maryland's portion of Chesapeake Bay. A Striped Bass recreational catch per trip index provided an index of relative demand. A forage-to-Striped Bass ratio (focal species combined) and grams of all forage consumed per gram of Striped Bass in fall provided trends in supply relative to demand based on relative abundance indices and diet sampling, respectively. Proportion of resident Striped Bass without visible body fat and

trends in survival due to natural mortality indices provided indicators of Striped Bass well-being. Statistical analyses provided evidence that forage and Striped Bass abundance and well-being were inter-related. Targets and thresholds were then developed for each of these indicators to assign them scores. A score of 1 indicated threshold conditions; a score of 5 indicated target conditions; and scores of 2-4 indicate grades between.

Introduction

Reports of Striped Bass in poor condition and exhibiting ulcerative lesions increased in Chesapeake Bay during the mid-to-late 1990s (Overton et al. 2003; Gauthier et al. 2008), spurring concerns about the effect of low Atlantic Menhaden abundance on the health of the contingent of Striped Bass that remains in the Chesapeake Bay after spawning ((i.e., residents; Uphoff 2003; Maryland Sea Grant 2009). Uphoff (2003) determined that these phenomena could be linked with poor feeding success on Atlantic Menhaden *Brevortia tyrannus*. Mycobacteriosis emerged in Chesapeake Bay in the late 1990s and an epizootic has affected Striped Bass in Chesapeake Bay (Jiang et al. 2007; Gauthier et al. 2008; Jacobs et al. 2009); challenge studies have linked nutrition and mycobacteriosis (Jacobs et al. 2009). Tagging and epidemiological models have provided evidence of increased M (total annual instantaneous natural mortality rate) of Striped Bass in Chesapeake Bay that is concurrent with the mycobacteriosis outbreak (Jiang et al. 2007; Gauthier et al. 2008; Sadler 2010). High M of Chesapeake Bay Striped Bass may have serious implications for management since this stock is the main contributor to Atlantic coast fisheries (Richards and Rago 1999; Sadler 2010).

Maryland Sea Grant (2009) identified a need for diet sampling and condition or nutritional health indicators for Striped Bass to address concerns about the effect of low forage abundance on Striped Bass *Morone saxatilis* well-being in Chesapeake Bay. Job 4 provided support for two major activities related to an ecosystem approach to management of Striped Bass during this cycle: development of diet database from multi-year, year-round citizen-science based diet monitoring and development of an index-based indicator approach (Rice and Rochet 2005; Jennings 2005) to evaluate forage and resident Striped Bass status in Maryland's portion of Chesapeake Bay. Because these two activities are fundamentally different, we have broken Job 4 into two sections. The first reports on our effort to build the diet database and summarizes information of summer feeding of Striped Bass in Maryland's portion of Chesapeake Bay. The second section describes the development of a prototype indicator-based approach for assessing forage of resident Striped Bass. Each section has separate Introductions, Methods, Result, and Discussion.

Diet Monitoring Introduction

In this federal aid report, we document steps needed to transform raw data from CBEF's paper ledgers into an Excel spreadsheet data base for June – September, 2007-2012. Data entry protocols were developed for annual summer Striped Bass diets; diets were summarized by numerical and weight composition. Numbers of prey ingested provide insight into feeding behavior, while weight and caloric content of prey consumed reflect nutritional value (MacDonald and Green 1983; Pope et al. 2001).

Diet Monitoring Methods

Field Collections – Field collections continued for 2013-2014 as described in Uphoff et al. (2014). During late September through November, 2014, we also obtained diet information from Striped Bass sampled by the Fish and Wildlife Health Program (FWHP). Samples from this program have been use to estimate Pf0 during 1998-2013 (Uphoff et al. 2014). These samples were obtained by hook-and-line in Chesapeake Bay from the lower and mid-Bay, placed on ice, and then processed back at Oxford Lab the next day. Our staff identified, measured, and weighed diet items as FWHP staff processed Striped Bass in the lab. Diet data will be merged with the other information collected by FWHP.

Fish examination – There was no change from what was described in F-63-R-4 (Uphoff et al. 2014).

Data entry – The CBEF ledger for 2013 was obtained and initial data entry begun as described in Uphoff et al. (2014). Each year's data will be recorded in a worksheet for sublegal-sized Striped Bass (< 457 mm, TL) and legal-sized (≥ 457 mm, TL). Variable names and labels were standardized and additional variables will be created (Table 4-1). In general, the data entry and editing process attempted to standardize variable names and labels, convert English units to metric (lengths in mm and weights in grams), and provide estimated weights of food items and Striped Bass without weights. See Uphoff et al. (2014) for greater detail. Table 4-1 provides an updated description of variables entered from CBEF ledgers and variables created through the editing process.

Data editing – Creation of a data set for summer (June 21 through September 30, 2006-2012) began. The process used for editing was largely the same as described in Uphoff (2014); additions or changes will only be described here.

Table 4-2 provides an update of labels assigned under "Contents edit name", the standard common name or taxonomic category, genus and species epitaph, and whether an item is considered "wild" food or not (described below). The two parameters for a non-linear allometry equation for converting diet item length to weight (grams; Hartman and Brandt 1995b) are provided in Table 4-2. Most of these diet item allometry equations were used to reconstruct diets for Overton (2003), Overton et al. (2009), Griffin (2001), and Griffin and Margraf 2003), and were originally developed by Hartman and Brandt (1995b). Allometry equations described changes in diet item weight (W) with length as

$$^{(1)}\ W = a \cdot (L^b);$$

where lengths (L) were TL for fish and shrimp, carapace width (CW) for crabs, or shell length (BL) for bivalves; a is a constant and b is an exponent (usually between 2.5 and 4.0 for fish; Pope and Kruse 2007; Table 4-2).

Bait, either live fish (predominantly Spot and infrequently White Perch) or fish pieces occurred frequently enough during summer 2006-2012 that the editing and analytical strategy of omitting bait in Uphoff et al. (2014) had to be reconsidered. Bait alone was no longer categorized as "none" (Uphoff et al. 2014), but was labeled as "live-line Spot", "live-line White Perch", or bait pieces. Weights of whole fish used for bait were recreated from their lengths as described for prey in Uphoff et al. (2014). This would allow for an estimate of the weight of bait present in the guts for comparison with recreated weights of natural food. An additional column was added that designated whether wild (natural) food was present (coded 1) or absent (coded 0). This coding allowed wild food to be easily separated from bait. This data element will need to be added to previously created data sets (October-November and winter 2006-2012). It also aided the creation of diet weight estimates for invertebrates represented by frequency of occurrence and not counts or measured weight (described below).

Invertebrates were much more common in summer diets than what we encountered in October-November or winter, 2006-2012 (Uphoff et al. 2014). Some invertebrate groups (polychaetes, amphipods, soft invertebrates, and Sand Shrimp) could not be counted or had inconsistent counts; they were represented by single entry per fish indicating presence. For these species, frequency of occurrence and empirical relationships developed by Stobberup et al. (2009; based mainly on diet data from a variety of fish in the North Atlantic) were used to estimate relative weight (weight of a prey item divided by total weight of prey items observed in a predator diet) represented by the particular food items. Relative weight of each of the invertebrate items were estimated from frequency of occurrence (number of prey items divided by the total number of non-empty stomachs expressed as a percentage; Stobberup et al. 2009) and we only used items in non-empty guts in the wild food category. Relative weight (RWi) was the percentage of total diet weight represented by the four items (i) representing uncountable invertebrates. The estimated reconstructed weights of other items (Wi; as described in Uphoff et al. 2014) were summed and the total weight (TW) of diet items was estimated as

$$^{(2)}\ TW = (\sum Wi) / [(100 - (\sum RWi)) / 100].$$

We used the equation Stobberup et al. (2009) developed for RWi of benthic crustaceans for shrimp, amphipods and soft invertebrate residue estimates of RWi were developed from the equation for zooplankton (plankton), and the equation for worms was used for polychaetes (Table 4-3).

Weights had to be assigned to some unmeasured mollusks (Soft Clams and Ribbed Mussels = 3 grams) or mollusk parts (clam snouts, 0.5 grams). We found that the TL versus weight relationship developed by Hartman and Brandt 1995a) for White Perch was the same as for Atlantic Silverside and we substituted a relationship for Choptank River White Perch for the one used by Hartman and Brandt (1995a). The TL-weight (mm to grams) equation for White Perch was $0.0000032*(TL^{3.29})$ and was based on 2000-2013 data for Choptank River for both sexes combined (P. Piavis, MD DNR, personal communication). Estimates for October-November and winter 2006-2012 need to be corrected, but these species did not occur frequently enough in these diets for serious errors in Uphoff et al. (2014).

Once all data have been edited and corrected, we anticipate combining annual data into a single database.

Length-weight regressions ($\log_e$-transformed lengths and weights) from Striped Bass sampled from the commercial hook-and-line fishery were used to estimate weights of legal-sized Striped Bass during summer 2006-2012 (E. Durell, MD DNR, personal communication; Table 4-4). The equation for these estimates was

$$^{(3)}\ \log_e(W) = a + b\cdot(\log_e L);$$

where W was weight in grams, L was total length in mm, a was the intercept, and b was the slope (Pope and Kruse 2007). The estimate of $\log_e(W)$ was exponentiated to estimate weight. We did not have sufficient data to develop annual summer length-weight relationships for sublegal Striped Bass.

Data Analysis - Two groups of Striped Bass were formed for analysis of summer diet: sublegal (< 457 mm TL) and legal ($\geq$ 457mm TL; hereafter, all lengths are TL unless otherwise noted). These categories accounted for ontogenic changes in Striped Bass diet, but also reflected sampling differences. Sublegal sized fish could only be collected by fishing for them directly, whereas, legal sized fish were collected by fishing and cleaning station visits.

We confined analysis of food items to those we believed were recently consumed in an attempt to keep odds of detection as even as possible. Items with "flesh" (whole or partial fish and invertebrates) were considered recently consumed, while hard, indigestible parts such as gizzards, mollusk shells, and backbones were excluded (Table 4-2). "Unknown fish parts" were excluded as well (Table 4-2). Partially intact items with flesh that were identified to species or other taxonomic group were assigned the mean weight estimated for intact items in the same group. Guts classified as "Regurgitated, empty", or with "Unknown residue" were also classified as "none" under "Contents edit name".

Bait was excluded from natural diet analyses, but we did estimate the total weight of bait present in the guts for comparison with the natural diet. Whole Spot or White Perch were checked carefully for hook wounds around the head, mid-dorsal, or tail when encountered in samples since they were often used as live bait by charter boats in mid-Bay. Similarly, chunks or pieces of Atlantic Menhaden, Spot, and soft or peeler Blue Crabs might have represented bait or chum (dispersed in water as attractant) and were identified from hook marks or straight knife cuts.

Feeding metrics were calculated for both subgroups of Striped Bass for each year: proportion of food represented by an item in weight (PWi), proportion of Striped Bass without food (P_{none}), and mean grams of an item per gram Striped Bass (MWi). We did not estimate mean calories per gram of Striped Bass (MCi). Uphoff et al. (2014) found that trends in MWi and MCi in October diets were nearly identical. Estimates of PWi were based on Striped Bass with stomach contents only, while remaining estimators were derived from all fish sampled including those without food (Pope et al. 2001). Weight of a Striped Bass was represented by measured weights when available or from weight predicted from the relevant length-weight regression when measured weights were absent. Estimates of MWi could be summed for legal Striped Bass (weights could be estimated) to calculate combined summer averages for all items ($\sum$MWi). Weights were not available for sublegal Striped Bass and we compared summer consumption as weight of items consumed divided by number of sublegal Striped Bass sampled. Once these metrics were available, a subset of items that accounted for 95% or more of diet by weight were identified as major items. In addition to metrics described above, we also estimated the proportion of Striped Bass guts containing live bait (live-lined Spot or White Perch).

We estimated Pf0 (described below) for each subclass of Striped Bass for July (includes the remainder of June), August, and September to view its progression over time. The 95% confidence intervals indicated which months were different from one another.

Diet Monitoring Results and Discussion

Data sets were completed for summer 2007 and 2008. Striped Bass ranged from 209 mm to 914 mm, TL. During 2007, 59% of sublegal Striped Bass examined were collected from the Bay region and 41% were collected from the Choptank River (N = 198). Thirty-seven percent of sublegal Striped Bass examined during 2008 were from the Bay region and 63% were from the Choptank River. Ninety-three percent of legal-sized Striped Bass examined during 2007 and 2008 were from the Bay region (N = 430 and N = 389, respectively). Differences in distribution of sublegal and legal-sized Striped Bass samples reflected sampling differences. Sublegal fish were obtained by us fishing for them directly and they were readily available in Choptank River. Legal-sized fish were obtained mostly from charter boats that had fished in the Bay because they were not available in the Choptank River in summer.

For any given year and Striped Bass size class combination, three diet items would comprise about 95% of food consumed during summer (Table 4-6). Pelagic fish (Atlantic Menhaden and Bay Anchovy) contributed 59-87% of weight in diet samples of both size-classes. Sublegal Striped Bass diet weight in 2007 was dominated by Bay Anchovy (86%), Blue Crab (8%), and Spot (2%); 2008 diets were dominated by YOY Atlantic Menhaden (66%), Spot (21%), and Bay Anchovy (10%). By weight, large Atlantic Menhaden (ages 1+) predominated in legal-sized Striped Bass diets in 2007 (59%) and 2008 (82%). In 2007, Spot (22%) and White Perch (ages 1+; 13%) were also important to legal-sized Striped Bass, as were Spot (10%) and YOY Atlantic Menhaden (5%) during 2008. Large Atlantic Menhaden were only found in Striped Bass greater than 700 mm, TL (4.4% of sampled legal-sized Striped Bass in 2007 and 10.1% in 2008), and the great majority of these were caught in June-July. Additional diet items groups during summer 2007-2008 for both size classes of Striped Bass were other fish (Naked Goby and Atlantic Silverside), Mud Crabs, polychaetes, shrimp (Sand and Grass Shrimp), soft invertebrates, and bivalves. None of these items individually contributed more than 2% of summer diet weight (Table 4-6).

Sixty-three and 65% of sublegal Striped Bass guts sampled during summer 2007 and 2008 (respectively) did not contain food (Table 4-6). Sublegal Striped Bass consumed less food during 2007 (1.4 grams per fish) than during 2008 (2.7 grams per fish; Table 4-6). Comparable estimates for October-November were 1.8 and 10.6 grams per fish, respectively. These estimates could be affected by the availability of forage and the size of fish within the sublegal category.

During 2007, 91% of legal-sized Striped Bass had empty guts, while 73% were empty in 2008 (Table 4-6). Legal-size Striped Bass consumed less food during summer 2007 (0.0026 grams of food per gram of Striped Bass) than during summer 2008 (0.0074 grams of food per gram of Striped Bass; Table 4-6). These estimates were 15% and 37% of respective estimates for October-November (Uphoff et al. 2014).

Live-lined bait fish were less common in sublegal than legal-sized Striped Bass during summer (Table 4-6). One percent of sublegal Striped Bass guts examined in 2007 contained live bait, while 3% did so in 2008; about 35% of sublegal fish examined had food. The combined weight of live-lined bait represented 41% of wild food consumed in summer by sublegal Striped Bass (bait excluded) sampled during 2007 and 17% in 2008 (Table 4-6). These fish were primarily caught on artificial lures.

Live-lined bait fish were present more often in legal-sized Striped Bass than sublegal ones during summer (Table 4-6). Six and 10% of legal-sized Striped Bass guts contained live bait in 2007 and 2008, respectively, compared to 8% and 27% of guts containing food. Since the great majority of these fish were sampled from the Bay by the charter boat fleet at Tilghman, it was very likely that live-lining bait fish was used to catch them (Table 4-6).

Condition, indicated by Pf0, deteriorated over the course of the summer (Table 4-7). For both size-classes of Striped Bass, Pf0 was lowest during July or July-August (if these months were statistically indistinguishable) and highest in September. Confidence intervals of monthly Pf0 estimated overlapped for sublegal fish during 2007 and 2008 and legal fish during 2007, i.e., each month's estimate for these classes overlapped one another. Monthly estimates for legal-sized fish during 2008 indicated Pf0 was lower in any given month than the other month, year, and size-class combinations (Table 4-7).

Indicator Introduction

Indicators can be defined as indices of a phenomenon, (in our case, forage fish abundance and availability) and are widely used for environmental reporting (several Chesapeake Bay "report" cards are issued each year), research, and management support (Jennings 2005). In the late 2000s, Maryland Sea Grant facilitated a panel of experts that developed potential indicators of ecological stress, including forage and predation, for Striped Bass in Chesapeake Bay; monitoring based indicators listed were prey-predator ratios, condition indices, and prey abundance in diet samples (Maryland Sea Grant 2009).

In 2014, a forage fish outcome was included in the Chesapeake Bay Agreement (Chesapeake Bay Program or CBP 2014): "By 2016, develop a strategy for assessing the forage fish base available as food for predatory species in the Chesapeake Bay." During November 12-13, 2014, a CBP Scientific and Technical Advisory Committee workshop, comprised of Bay fisheries managers and scientists representing diverse geographical and ecological interests, was held in Solomons, Maryland, to explore how to assess the Chesapeake Bay forage fish base. Managers wanted practical guidance on what forage fish are important in the Bay, how abundant forage fish are, and what monitoring could be used that is already in place or could be inexpensively implemented.

Recently, the 2015 ASMFC Atlantic Menhaden stock assessment identified potential monitoring-and model-based ecological reference points that account for Atlantic Menhaden's (and other species) role as forage fish (SEDAR 2015). Monitoring-based forage indices, predator-prey ratios, and condition reference points, and prey abundance in diet samples received favorable consideration (SEDAR 2015).

These efforts provided convergent needs for ecological indicators that assess adequacy of forage for resident Striped Bass in Maryland's portion of Chesapeake Bay and we have developed a prototype indicator-approach to address them. Two extensive reviews of the development of indicator-based approaches in support of an ecosystem approach to fisheries (EAF; Jennings 2005; Rice and Rochet 2005) were consulted to develop this approach and the reader is directed to them. Resident Striped Bass were chosen by Maryland's fisheries managers as an indicator predator for the upper Chesapeake Bay (Maryland waters). Use of indicator species is widespread in environmental studies (Rice 2003).

Resident Striped Bass offered an immediate opportunity to develop an indicator-based assessment approach because they and their prey are monitored in Maryland's portion of Chesapeake Bay. Resident Striped Bass constitute a year-round population (predominantly male) of predators that provides Maryland's major recreational fishery and an important commercial fishery (Maryland Sea Grant 2009). The impact of Striped Bass on its forage base and the impact of forage on Striped Bass are of major concern in Chesapeake Bay and interstate management (see Introduction). Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab were selected as focal forage species because they were consistently identified as items accounting for most of diet biomass of Striped Bass in Maryland's portion of Chesapeake Bay (Hartman and Brandt 1995a; Griffin and Margraf 2003; Walter and Austin 2003; Overton et al. 2009; Uphoff et al. 2014) and because managers identified clear concerns about them. Targets (a "safe" level) and thresholds (a level with an unacceptable risk of a negative outcome) were to be developed for each indicator (Jennings 2005) and indicators would be judged relative to targets and limits.

Recent development of Striped Bass nutritional indicators that reflect lipid content by Jacobs et al. (2013) anchored our approach. Lipids serve as the energy currency in marine fish (Rose and O'Driscoll 2002) and are the source of metabolic energy for growth, reproduction, and swimming (Tocher 2003). Lipid allocation among storage, maintenance, and growth changes between fish that are feeding successfully or starving (Jacobs et al. 2013). Starvation caused declines in energy reserves, physiological condition, enzyme activity, increased natural mortality in Atlantic Cod *Gadus morhua*, and caused degeneration of swimming muscle in Winter Flounder *Pleuronectes americanus* (Dutil and Lambert 2000).

In the well-studied Atlantic Cod *Gadus morhua*, forage fish, nutritional state, and natural mortality have been linked to form a viable alternative hypothesis to overfishing (Lambert and Dutil 1997; Dutil and Lambert 2000; Shelton and Lilly 2000; Rose and O'Driscoll 2002). We felt we had an opportunity to form a set of indicators that could provide similar linkages, albeit with less elegant information. We had a viable indicator of nutritional state of resident Striped Bass (proportion without body fat; Jacobs et al. 2013). We had multiple indicators of forage status to address supply, but we needed to develop an indicator of demand (a resident Striped Bass index of abundance) an index of relative natural mortality.

Indicator Methods

Maryland DNR's Fish and Wildlife Health Program (or FWHP) has monitored Striped Bass health in Chesapeake Bay during fall (late September – November) since 1998 (M. Matsche, MD DNR, personal communication). A categorical body fat index was used by FWHP to evaluate visible reserves of visceral body fat: 0 = no detectable fat; 1 = fat present, but coverage was less than 25%; 2 = 25-75% of viscera covered; and 3 = 75% or greater coverage of viscera. Jacobs et al. (2013) analyzed an identical classification to develop nutritional reference points for Chesapeake Bay Striped Bass. These body fat index data, collected by FWHP, were provided to us for analysis with our data by M. Matsche and K. Rosemary.

For both CBEF and FWHP body fat data, the nutrition threshold for individual Striped Bass was indicated by a body fat index of 0 (no visible fat) and the proportion of Striped Bass with that score (Pf0) in the size class sample indicated what fraction met the threshold condition and were vulnerable to starvation (Jacobs et al. 2013). Standard deviations and confidence intervals (95% CI) of Pf0 were estimated from the normal distribution approximation of the binomial distribution (Ott 1977). The probability of meeting a body fat target criterion (see below) equaled the cumulative proportion (expressed as a percentage) of the Pf0 distribution function equaling or falling below the target.

A target level of Pf0 of 30% (John Jacobs, NOAA, personal communication) was used to judge whether mid-Bay Striped Bass had fed successfully during October-November. A target for body fat was not presented in Jacobs et al. (2013), but mean tissue lipid of Striped Bass with a body fat index of 0 was identical to that estimated from percent moisture. Jacobs et al. (2013) presented a target for body moisture (25% or less of fish with starved status) that was derived from mean moisture in fall 1990 field collections and variation in moisture from experiments conducted during 1996-2005 (an estimate of variability of 1990 samples was not available). Feeding conditions were considered favorable in 1990 and these samples offered the only opportunity for a target condition (Jacobs et al. 2013). Variation of tissue lipids estimated from body fat indices was greater than for moisture and the Pf0 target of 30% for body fat accounted for this additional variation plus a buffer for misjudging status (John Jacobs, NOAA, personal communication).

Previously, estimates of Pf0 were made for sublegal and legal sized Striped Bass in FWHP surveys in order to compare results with CBEF diet data (Uphoff et al. 2014). However, estimation of the Pf0 threshold did not necessarily require the length categories. We used correlation analysis to examine how closely associated Pf0 estimates were for the two size classes. If estimates were closely correlated ($r > 0.80$; Ricker 1975), we then considered combining the two size class estimates into a single estimate with greater precision due to larger sample size. We needed to balance the ability of being able to separate more years from one another with the possibility of missing a size-specific divergence of trends. We estimated the 95% confidence intervals for each categorization (all fish, sublegal fish, legal fish) and compared the potential interpretation of being at the threshold condition (described below), at the target condition, or in between.

A threshold for nutritional condition was developed from the selected distribution of Pf0 estimates made by the FWHP during 1998-2013. We examined the 1998-2013 time-series of Pf0 and their 95% CI's to identify persistent high Pf0 estimates that could be separated from estimates at or near the target level. We used the average of the lower 95% CI of these high Pf0 estimates as our threshold.

We used geometric mean indices of Atlantic Menhaden, Bay Anchovy, and Spot, from MD DNR's long-term (1959-2013) seine survey (Durell and Weedon 2014), and Spot and Bay Anchovy indices from the Blue Crab trawl survey (1989-2013; MD DNR 2014a; estimates provided by H. Rickabaugh, MD DNR) were used as indicators of relative abundance of important fish prey. We used seine or trawl indices summarized for all systems sampled. Sampling occurred during summer – early fall. The bay-wide seine index consisted of samples collected from the Head-of-Bay region, and Potomac, Nanticoke, and Choptank rivers (permanent stations sampled since 1954; Durell and Weedon 2014; Figure 4-1). Trawl indices were estimated from samples collected from Chester River, Eastern Bay, Choptank River, Patuxent River, Tangier Sound and

Pocomoke Sound (MD DNR 2014a; Figure 4-1). Density of juvenile Blue Crabs in a winter dredge survey (1989-2013; MD DNR 2014b) was our indicator of Blue Crab relative abundance. The winter dredge survey sampled Blue Crabs bay-wide (MD DNR 2014b).

We assumed these indices would reflect relative abundance of major prey species in Maryland's portion of Chesapeake Bay. Correlation analysis was used to judge whether Bay Anchovy or Spot indices from seine and trawl indices were similar enough that one could be chosen to represent a common trend. We used the criterion that a correlation coefficient needed to exceed 0.8 in order to give preference for one indicator (Ricker 1975); preference would be given to seine indices because of their longer time series. Each forage index was divided by its time-series mean in order to place them on the same scale for graphs.

While indices of prey abundance exist for Maryland's portion of Chesapeake Bay (Uphoff et al. 2014), a long-term index or estimate of relative abundance of resident Striped Bass (other than young-of-year) has not been developed. We developed a catch-per-private boat trip index (released and harvested fish) for 1981-2013 from the National Marine Fisheries Service's (NMFS) Marine Recreational Information Program (MRIP) database as a measure of resident Striped Bass relative abundance. MRIP indices are used as abundance indicators in ASMFC stock assessments of major pelagic finfish predators along the Atlantic coast (Striped Bass, Bluefish, and Weakfish; ASMFC 2013; NEFSC 2012; ASMFC 2009).

This index was developed as a catch-effort ratio for private 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 2014). Coastal bays are included in these totals, but these fisheries are usually minor compared to Chesapeake Bay. This fishery-dependent Striped Bass index (RI) was estimated as the wave 5 (September-October) recreational private/rental catch (harvest + releases, in numbers) divided by MRIP estimates of trips for the private/rental boat sector during wave 5. Private boat recreational fishing occurs over the entire portion of the Bay in MD and this index would be as close to a global survey as could be obtained. The September-October wave of MRIP represents the period when seasons were always open following Maryland's 1985-1990 harvest moratorium, a 457 mm TL (18 inch) size limit was consistently applied, and migratory fish had left the Bay.

We used two linear regression approaches to examine if trends in the RI were related to other measures of Striped Bass abundance used as inputs or available as outputs in the most recent ASMFC Striped Bass stock assessment (ASMFC 2013). The first approach developed a multiple regression of lagged Maryland Striped Bass juvenile indices (JIs; based on geometric means; Durell and Weedon 2014) that would reproduce the RI. This analytical strategy was similar to an approach used by Goodyear (1985) with MD juvenile indices (as arithmetic means) to determine age-classes that predominated in Maryland's commercial fishery. We followed the analytical strategy of Goodyear (1985) and used stepwise selection (Freund and Littel 2006) to select a set of lagged juvenile indices (geometric mean Striped Bass JIs) that could be modeled to estimate the RI. Juvenile indices were entered at $P < 0.15$ and were retained at $P < 0.05$. Mallow's Cp was used to judge the regression that best fit the data and minimized parameters (Freund and Littel 2006). We examined partial R^2 coefficients to judge how much variation significant (at $P \leq 0.05$) lagged JIs accounted for. If a partial R^2 coefficient was judged to be low enough, this lagged JI could be omitted to minimize parameterization. We used the convention employed by Goodyear (1985) of considering the JI as an indicator of age-1 abundance, so that a one-year lag corresponded to a juvenile index two years earlier and so on. One- to five-year lags were considered reasonable since the RI would consist of both released fish (mostly sublegal) and harvested resident fish. Maryland's Chesapeake Bay recreational fisheries harvest Striped Bass four years-old and older, but ages 4 to 6 predominate. Maryland's Striped Bass juvenile indices (as geometric means) are used in the ASMFC stock assessment as an indicator of yearling abundance (ASMFC 2013).

The second approach to evaluate the RI summed abundance estimates from the ASMFC (2013) stock assessment's statistical catch-at-age model (or SCAM) for ages identified as important in the multiple linear regression of JIs and RI.

The relationship of RI to Pf0 was examined using linear regression. If Pf0 was not reflecting density-dependent effects, a negative relationship with abundance should not be detected. Examination of the plot of Pf0 and RI suggested that a curvilinear relationship might be possible, so an inverse transformation (1 / RI) was considered for linearizing data. Therefore, two models were used: (1) Pf0 = RI and (2) Pf0 = 1 / RI.

Forage-to-Striped Bass ratios were calculated annually for each focal forage species (Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab) as Fi / RI; where Fi was the index of abundance of forage species i and RI was the resident Striped Bass index described above. The latter two species were classified as benthic forage, while the remaining two species were classified as pelagic. Each forage-to-Striped Bass ratio was divided by their respective means in order to place them on the same scale for graphs. Correlation analysis was used to explore associations of forage-to-RI ratios and Pf0 during 1998-2013.

The ratios of age 3 or age 4 annual relative abundance of male Striped Bass in spring gill net surveys to their year-class-specific juvenile indices (as geometric means) were used as indicators of change in survival due to natural mortality (SR). Maryland estimates age-specific mean indices of Striped Bass relative abundance from spawning season gill net surveys on the Potomac River and Upper Bay (~39% and 47% of Maryland's spawning area; Hollis 1967) and these estimates are used in the ASMFC stock assessment (Giuliano and Versak 2014; ASMFC 2013). We estimated a combined (Potomac and Upper Bay) relative survival to each age. To combine the estimates, we first standardized each area's time series of gill net catch per unit effort to its time-series mean (1985-2013). These standardized estimates were then averaged for each year (Potomac River was not sampled in 1994) and this average of standardized CPUE was divided by its respective JI three or four years prior to estimate SR.

Confining the gill net relative abundance indices to 3 or 4 year-old males makes it likely that trends in SR will reflect resident Striped Bass before harvest. Age 3 male Striped Bass in the spring gill net survey were well below legal-size (Giuliano and Versak 2014), but they could be subject to catch-and-release mortality. Mean lengths of age 4 male Striped Bass were close to, but less than legal-size, in the gill net survey. Some had reached legal-size by the spring gill net survey, but estimates of F on Chesapeake Bay Striped Bass by SCAM or tagging have been low and fairly steady (ASMFC 2013), so changes in age 4 SR should primarily reflect natural mortality. We expected SR for both ages to vary without trend if natural mortality (M) remained constant.

Trends in SR were compared to tag-based estimates of survival for legal-size (457-711 mm) Striped Bass from Chesapeake Bay in the ASMFC benchmark stock assessment (ASMFC 2013). Tag-based estimates of M were determined for two time periods in the recent stock assessment (1987-1996 and 1997-2011; ASMFC 2013). We converted the two estimates of M to survival (S) using the equation $S = e^{-M}$ (Ricker 1975); S during 1987-1996 equaled 0.77 and S equaled 0.44 during 1997-2011.

Linear regressions were used to examine the relationship of SR at age 3 (SR3) with SR at age 4 (SR4), Pf0 with SR3 or SR4, and RI with SR3 or SR4. Because relative survival was estimated in spring and Pf0 was estimated in the fall, we used SR3 and SR4 estimates in the following year (year + 1) in these regressions. If 95% CI's of regression slopes and intercepts overlapped for separate regressions of Pf0 and RI with SR3 or SR4, we combined SR3 and SR4 estimates into a single regression. We used correlation analysis to explore associations of SR3 or SR4 with forage ratios.

Level of significance for correlation and regression analyses was $P \leq 0.05$. Scatter plots were examined for the need for transformations. Residuals were inspected for outliers, trends, non-normality, and need for additional terms. If a large outlier was identified, the data from that year was removed and the regression 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 (Nakagowa 2004). Close correlations among forage-to-RI indices ($r \sim 0.8$ or greater; Ricker 1975) would cloud whether species-specific ratios could be interpreted as unique contributors to associations with Pf0 or SR, so correlations among forage-to-RI indices were examined. If the ability to cleanly interpret these analyses was in doubt, a correlation analysis with the forage indices alone was conducted; correlations among forage indices were examined for close correlations (as defined by Ricker 1975).

A composite forage-to-Striped Bass ratio was constructed from individual forage-to-Striped Bass ratios that were significantly related to Pf0, SR3, and SR4; these forage species are referred to as focal species. Standardized ratios of forage fish-to-Striped Bass were on the same scale, but this did not reflect their importance to Striped Bass. Contribution of diet items to nutritional state can be estimated from their weight (Bowen 1989) and a forage fish-to-Striped Bass ratio for all focal species combined was calculated by multiplying each ratio times the percentage each forage fish represented in sublegal Striped Bass diets during October-November, 2006-2012 (years combined; Uphoff et al. 2014) and then summing these four products. Sublegal Striped Bass were chosen because their estimated consumption was well-related to their nutritional state; a relationship could not be detected for legal-size Striped Bass (Uphoff et al. 2014). This combined index was regressed against sublegal Striped Bass consumption during 2006-2012 to determine if this index was related to consumption. Correlation analyses estimated the strength of associations of each focal species-to-Striped Bass ratio to the composite ratio. Linear regression was used to determine the relationships of the composite forage-to-Striped Bass ratio to Pf0, grams of forage consumed per gram of sublegal Striped Bass, SR3, and SR4. If the associations and relationships with the composite forage-to-Striped Bass ratio were significant and at least as strong as those for the individual focal species ratios, the composite ratio could be used in place of the individual ratios.

Statistical analyses can provide insight into important processes related to predation (Whipple et al. 2000). The combination of statistical analyses and empirical information was intended to confirm or falsify whether population dynamics of resident Striped Bass in Maryland's portion of Chesapeake Bay were meeting underlying assumptions of the single species assessment model for Atlantic coast Striped Bass (constant M across years and little density-dependence) that hypothesizes that fishing is the major driver of dynamics (Hare 2014). In this case, few, if any, associations or relationships would be apparent among indicators. A viable alternative hypothesis relating to density-dependent processes would be supported if a preponderance of analyses linked Striped Bass density, forage abundance and consumption, nutritional state, and changing annual natural mortality, while evidence of overfishing was lacking.

The prototype of the indicator-based approach was based on a suite of statistically linked indicators. Interpretation of status would be judged by whether target or threshold reference points were met for each indicator. Time-periods where nutrition indicators (1998-2013) or October-November diet information (2006-2012) were available provided a time-frame for developing targets and thresholds for other indicators. Targets and limits based on historical performance are desirable because they are based on experience and easily understood (Hilborn and Stokes 2010). A combination of scoring and stoplight colors communicated status from indicators judged both from their most recent value and from their recent 3-year trend. Each indicator in the most recent year is assigned a score from 1 to 5 (bad to good); a score of 1 indicates bad or threshold conditions; a score of 5 indicates good or target conditions; and scores of 2-4 indicate grades of status in between. Indicators with a score of 1 are depicted in red, scores of 5 are indicated by green, and 2-4 are shaded yellow. The trend in the most recent three years is provided in the next column. Improving trends would be indicated by green shading if the target was met in the terminal year (score = 5 in the terminal year) and yellow if improvement was in a positive direction, but the target was not met. Deteriorating trends would be shaded red if scores moved in a negative direction and the threshold (score = 1 in the terminal year) was met in the terminal year. A deteriorating trend would be coded yellow if the threshold was not met in the terminal year. "No trend" could be shaded green if target conditions were maintained (in two or three years), yellow if neither target nor threshold conditions were met, and red if threshold conditions were maintained.

Indicator Results

Estimates of Pf0 for sublegal and legal Striped Bass during October-November, 1998-2013, by the FWHP were strongly correlated ($r = 0.95$, $P < 0.0001$), suggesting the categories were indicating the same nutritional conditions. Estimates of Pf0 for sublegal, legal, and categories combined (Figure 4-2) did not suggest that interpretation of nutritional status would be different if a single pooled estimate of Pf0 was estimated for each year. Nutritional conditions were poorest in all three categories during 1998-2004, near or at the target condition during 2008-2010, and intermediate but nearer the poorest conditions in the remaining years. We used estimates of Pf0 for all sizes of Striped Bass sampled by FWHP for subsequent analyses (Table 4-8).

We used the average of the lower 95% CI of high Pf0 estimates during 1998-2004, 0.68, as our threshold nutritional condition (Table 4-8; Figure 4-2). The chance that this threshold was exceeded ranged from 77-100% during 1998-2004, with the exception of 2002 when it was 0%. A non-zero chance of meeting the threshold, 21%, existed in 2012 as well. The probability of meeting the target nutritional condition was 100% in 2008, 19% in 2009, and 100% in 2010; the chance of meeting the target was zero in remaining years (Table 4-8; Figure 4-3).

Pelagic forage fish (Atlantic Menhaden and Bay Anchovy) have generally been at a lower level in Maryland's portion of Chesapeake Bay since Pf0 has been estimated (1998-2013; Figure 4-4). Atlantic Menhaden indices indicated a period of high productivity between 1971 and 1994. Indices during 1959-1970 and 1995-2013 were much lower (Figure 4-4).

The two sets of Bay Anchovy indices were not significantly correlated (1989-2012, 1990 missing; r =0.27, P = 0.20), indicating that they were not depicting the same trends over this period. Seine indices were at low levels during the early to mid-1990s, typically at or below the bottom quartile of indices during 1959-1993 (Figure 4-4). Highest trawl indices occurred in brief spurts in 1989-1992 and 2001-2002, while lowest indices occurred during 2006-2011 (Figure 4-4).

Seine and trawl indices for Spot were significantly correlated (1989-2012, 1990 missing; r = 0.86, P < 0.0001). Based on the strength of this correlation, we chose to use the longer time-series of seine survey indices as the indicator of Spot relative abundance in Maryland's portion of Chesapeake Bay. The general pattern for Spot in the long-term (1959-2013) was very similar to Atlantic Menhaden; indices indicated a period of high productivity between 1971 and 1994 and were much lower during 1959-1970 and 1995-2013 (Figure 4-5). Blue Crab densities were highest during 1989-1996, 2009, and 2011 (Figure 4-5).

Recreational private and rental boat catch and effort estimates for Maryland during wave 5 (September-October) were not available for 1982 and 1987 (Table 4-9; NMFS Fisheries Statistics Division 2014). Proportional standard errors (PSE; SE as a percent of the mean) were large enough to be of concern during the early years of the survey (36-92% during 1981-1989) and during 2007 (47%). In general, RI was lowest prior to 1994 (1990-1993 mean = 0.54 fish per trip; these were years of low abundance and adequate PSE's; Figure 4-6). It then rose very rapidly to a high level by 1995 and then remained there until 2006 (mean = 2.65). The RI fell rapidly after 2006 to about half the high level (mean = 1.21) during 2008-2013 (Figure 4-6).

Stepwise regression selected Striped Bass JI's for ages 2-5 for the multiple regression with RI (Table 4-10). The regression accounted for 87% of variation and was highly significant (P < 0.0001). Partial correlation coefficients were highest for ages 4 and 3, followed by ages 2 and 5 (Table 4-10). This multiple regression model predicted the RI well during 1983-1997 and then tended to underpredict somewhat afterwards.

The regression of RI against summed SCAM abundance estimates for ages 2-5 was significant ($r^2 = 0.67$, P < 0.0001; Table 4-11; Figure 4-6). Inspection of residuals indicated that 2004 represented a potential outlier. The regression was rerun without 2004 and the fit was improved ($r^2 = 0.80$, P < 0.0001; Table 4-11; Figure 4-6). The SCAM-based regression models tended to overestimate RI in the early 1990s and underestimate it during the early 2000s (Figure 4-6). The results of both regression analyses supported using RI as an indictor of basic trends in the Striped Bass population in Maryland's portion of the Bay.

Standardized forage-to-Striped Bass ratios exhibited similar patterns since 1990 (Figure 4-7). Ratios were high in the early 1990s when Striped Bass were not abundant. Lowest ratios started in 1995 and continued for a decade, followed by a modest rise. We did not use the earliest years with consistently large PSE's (1981-1989) to form forage-to-Striped Bass ratios. An isolated year with a high PSE (2007) was not omitted.

Significant relationships were detected for Pf0 versus RI ($r^2 = 0.32$, $P < 0.03$) and Pf0 versus inverse RI ($r^2 = 0.51$, $P < 0.002$; Figure 4-8); the latter relationship was considered a better description of dynamics due to better fit.

Correlation analyses indicated that FWHP estimates of Pf0 were significantly associated with the ratio of Atlantic Menhaden to RI ($r = -0.63$, $P = 0.009$), and the ratio of Spot to RI ($r = -0.51$, $P = 0.04$) during 1998-2013 (Table 4-12). The correlation of the Atlantic Menhaden ratios with Spot ratios was significant ($r = 0.59$, $P = 0.017$), but would not be considered close based on the criterion of Ricker (1975). These significant correlations were consistent with a hypothesis that attack success on forage and density of resident Striped Bass would influence condition.

Trends in relative survival of age 3 and age 4 (SR3 and SR4, respectively) sublegal resident Striped Bass indicated high (1985-1996) and low (1997-2013) periods of survival (Figure 4-9). These periods of SR3 and SR4 aligned well with periods of natural mortality-based high survival (1987-1995) and low survival (1996-2012) estimated for legal-sized (457-711 mm, TL) Striped Bass in Chesapeake Bay from tagging models (ASMFC 2013; Figure 4-9).

Examination of the plot of SR3 against SR4 indicated that a $\log_e$-transformation could reduce high variation at higher values of SR and linearize these data. The regression of $\log_e$-transformed SR3 against $\log_e$-transformed SR4 was significant ($r^2 = 0.48$, $P < 0.0001$; Figure 4-10) and the relationship was described by the equation:

$$^{(4)}\ \log_e SR4 = (0.63 * \log_e SR3) - 0.41;$$

where SR3 and SR4 are survival indices for ages 3 and 4 respectively. Standard errors equaled 0.12 for the slope and 0.23 for the intercept.

The regression of SR3 with Pf0 was not significant ($r^2 = 0.23$, $P = 0.07$, $N = 15$), but the regression of SR4 with Pf0 was significant ($r^2 = 0.44$, $P < 0.007$, $N = 15$). The equations describing the relationships were similar:

$$^{(5)}\ SR3 = (-0.263 \cdot Pf0) + 0.295 \text{ and } SR4 = (-0.246 \cdot Pf0) + 0.299.$$

Standard errors for the slope and intercept of the equation for SR3 were 0.133 and 0.081 and were 0.076 and 0.046 for the slopes and intercepts of the equation for SR4. The 95% confidence intervals of the slopes and intercepts of both equations overlapped, so we combined the data from the separate SR3 versus Pf0 and SR4 versus Pf0 regressions and estimated a single relationship, i.e., SR versus Pf0. This relationship ($r^2 = 0.30$, $P = 0.0018$; $N = 30$) was described by the equation:

$$^{(6)}\ SR = (-0.255 \cdot Pf0) + 0.297;$$

standard errors of the slope and intercept were 0.074 and 0.045, respectively.

$\log_e$-transformed estimates of SR3 and SR4 were significantly, positively, and similarly related to RI ($r^2 = 0.43$, $P < 0.0002$, and $N = 28$ for both comparisons; Figure 4-10). The equations describing the relationships were

$$^{(7)}\ \log_e SR3 = (-0.701 \cdot RI) - 0.322 \text{ and } \log_e SR4 = (-0.624 \cdot RI) - 0.365.$$

Standard errors of the slope and intercept of the $\log_e$ SR3 relationship were 0.160 and 0.312, respectively; standard errors of the slope and intercept of the $\log_e$ SR4 relationship were 0.141 and 0.276, respectively. These estimated relationships were similar, so a relationship was estimated for $\log_e$ SR3 and $\log_e$ SR4 combined (abbreviated as $\log_e$ SR). The combined relationship ($r^2 = 0.42$, $P < 0.0001$, $N = 56$) was described by the equation:

$$^{(8)}\ \log_e SR = (-0.663 \cdot RI) - 0.344;$$

standard errors of the slope and intercept were 0.105 and 0.205. Examination of residuals suggested a decline over time (possible autocorrelation; Figure 4-11). The Durbin-Watson (DW) statistic equaled 1.79 and the critical DW statistic tabular value was 1.39, indicating autocorrelation with a one-year lag was not likely (Freund and Littel 2006). Further analyses of autocorrelation were not possible on the SAS package available. However, this patterning may suggest that additional terms are needed or that a linear relationship may not be the only possibility. Time-series of SR3 and SR4 suggest a threshold response in survival (rapid shift from high to low) between 1994 and 1995 when RI abruptly shifted from low to high; residuals averaged 0.43 during 1985-1994 and -0.20 during 1995-2013. The best model for tag-based estimates of M of Chesapeake Bay Striped Bass (legal-sized) indicated similar periods of high and low survival as well (ASMFC 2013).

Correlations of SR and forage-to-Striped Bass ratios were significant for Atlantic Menhaden ($r = 0.58$ and 0.44 for SR3 and SR4, respectively), Bay Anchovy (seine-based, 0.59 and 0.47; trawl-based, $r = 0.48$ and 0.51) and Blue Crab ($r = 0.40$ and 0.43), but not for Spot (Table 4-13). However, correlations among ratios were so strong and positive (9 of 15 comparisons with $r < 0.79$ and $P < 0.0001$; Table 4-14), that it would be difficult to interpret which indices were truly important. Both SR3 and SR4 were significantly correlated with indices (instead of ratios) for Atlantic Menhaden ($r = 0.79$ and 0.52 for SR3 and SR4, respectively) and seine-based Bay Anchovy indices ($r = 0.73$ and 0.52). SR4 estimates were significantly correlated with trawl-based Bay Anchovy ($r = 0.54$) and Blue Crab ($r = 0.56$), but SR3 estimates were not ($r = 0.17$ for both species; Table 4-15). Relative abundance indices for Spot were not significantly associated with SR3 or SR4. Forage indices did not exhibit the pattern of strong correlations among species found with forage ratios. While many correlations among species indices were significant, only Atlantic Menhaden and Bay Anchovy seine indices were positively and significantly correlated ($r = 0.60$) with each other (Table 4-16), but would not be considered closely correlated based on the criterion of Ricker (1975).

Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab met the criteria for inclusion in the combined forage-to-Striped Bass ratio (FR). Based on the summed grams eaten per gram of Striped Bass (October-November, 2006-2012), Atlantic Menhaden received the highest weighting (0.56), followed by Spot (0.23), Bay Anchovy (0.11), and Blue Crab (0.03). FR was highest during 1990 and declined rapidly through 1995 (Figure 4-14). It was lowest during 1995-2004 and then rose and varied at somewhat higher levels (Figure 4-14). FR was well correlated with the species-specific forage-to-Striped Bass ratios: Atlantic Menhaden, $r = 0.97$, $P < 0.0001$; Bay Anchovy, $r = 0.90$, $P < 0.0001$; Spot, $r = 0.69$, $P = 0.0003$; and Blue Crab, $r = 0.70$, $P = 0.002$.

Estimates of FR were related to Pf0 during October-November, 1998-2013 ($r^2 = 0.42$, $P < 0.007$, $N = 16$; Figure 4-15). The equation describing the relationship was

$$^{(9)}\ Pf0 = (-0.134*FR) + 0.71.$$

Standard errors of the slope and intercept of the relationship were 0.04 and 0.06, respectively.

FR was related to grams of focal forage species consumed per gram of sublegal Striped Bass (C) during October-November, 2006-2012 ($r^2 = 0.58$, $P < 0.05$, $N = 7$; Figure 4-16) . The equation describing the relationship was

$$^{(10)}\ C = (0.0046) + 0.006.$$

Standard errors of the slope and intercept of the relationship were 0.0017 and 0.003, respectively.

The regression of SR3 with FR was significant ($r^2 = 0.64$, $P < 0.0001$, $N = 24$; Figure 4-17). The equation describing the relationship was

$$^{(11)}\ SR3 = (0.12 \cdot FR) + 0.069.$$

Standard errors for the slope and intercept of the equation for SR3 were 0.019 and 0.063, respectively.

The regression of SR4 with FR was significant ($r^2 = 0.23$, $P = 0.016$, $N = 24$); however, examination of residuals indicated that a point was a potential outlier (Figure 4-17). This point was removed and the regression was rerun; fit was improved ($r^2 = 0.38$, $P = 0.0017$, $N = 23$; Figure 4-17). The equation describing the relationship (outlier removed) was

$$^{(12)}\ SR4 = (0.060 \cdot FR) + 0.158.$$

Standard errors for the slope and intercept of the equation for SR4 were 0.017and 0.053.

Targets and limits for Pf0 were described previously. Descriptions of development of reference points for FR, RI, consumption, and SR follow.

Reference levels for FR were judged using 1998-2013 estimates – a period when body fat information was consistently available and Striped Bass abundance was not depleted from overfishing (Figure 4-14). FR was lowest when Pf0 was below the threshold (1998-2004) and highest during the two years (2008 and 2010) that target conditions were met (Figure 4-14). FR was lowest when Pf0 was above the threshold (1998-2004 FR range = 0.23-0.65) and highest during the two years (2008 and 2010, FR = 1.99 and 3.94, respectively) that target Pf0 was met. Lowest FR occurred during 1998-2004, when Pf0 was at threshold levels. We chose 0.65, the maximum FR during this period, as the threshold. The lowest ratio where target Pf0 was met (2008, rounded to 2.0) was selected for a target. Threshold-level FR was also present during 1995-1997. Target level FR was present during 1990-1993, when RI was low, and 1994, when RI was moderate (Figure 4-6).

The equation relating Pf0 to the inverse of RI,

$$^{(13)}\ Pf0 = (-0.542 \cdot (1 / RI)) + 0.880,$$

was back-transformed into RI (Figure 4-18) and predicted RI was used as an aid for determining reference points. Threshold Pf0 was predicted to have been reached when predicted RI was 2.7 or more (RI was sometimes at this level during 1995-2007), but observed Pf0 was at threshold values when RI was as low as 1.9. Target Pf0 would be reached when predicted RI was 0.9 or less (met during 1983-1993 and during 2008 and 2010; there was a significant chance (19%) that target Pf0 was reached in 2009 even though RI jumped to 1.6 (Figure 4-18).

Our choice of RI reference points took into consideration trade-offs between Striped Bass well-being and fishery needs. Unless forage levels can be increased greatly, high levels of Striped Bass in Maryland's portion of the Bay indicated by RI during 1995-2007 (RI range = 1.9 to 3.8), will lead to threshold Pf0. Minimum RI during 1995-2007, 1.9, was considered the threshold. Moderate levels of Striped Bass, indicated by RI during 1994 and 2008-2013 (RI range = 0.8 to 1.6) supported recreational and commercial fisheries, avoided body fat threshold conditions, and the Pf0 target was met during two years. An RI of 1.3 (observed in 2013) clearly avoided threshold conditions during 2008-2013 and was chosen as a target that may balance abundance needed for a fishery with concerns about condition and forage. Two very similar RI estimates (1.5 and 1.6) during 2008-2013 were associated with Pf0 near its threshold or target. While maximum RI observed during 2009 (1.6) was associated with near-target Pf0, it was also sandwiched between the two lowest RI's (0.8 each). The second highest RI during 2008-2013 (1.5 in 2012) was associated with a 21% chance of meeting the Pf0 threshold (Table 4-8).

The period 1998-2013 was used to develop reference points for SR3 and SR4 (Figure 4-19). This time-period provided Pf0 estimates to compare to SR estimates and Striped Bass abundance was not depleted from overfishing. We chose the maximum SR estimate during 1998-2004 (SR3 = 0.25) as the threshold. There was not enough distinction in ranges of SR3 or SR4 between 1998-2004, when Pf0 was at threshold levels, and the remaining years for a clear choice of target SR.

Indicator tables were developed for 2013 (most recent year), 2004 (a year of threshold Pf0), and 2010 (a year of target Pf0). Grams forage eaten per gram Striped Bass indices for 2012 were substituted for 2013 since 2013 consumption estimates are not yet available.

Overall, forage conditions were poor for Striped Bass in Maryland's portion of Chesapeake Bay in 2013 based on five indicators, but threshold conditions were usually avoided (Table 4-17). One indicator was at threshold levels (SR3 and SR4; score = 1), three were just above threshold conditions (FR, C, and Pf0; score = 2), and one was at its target (RI; score = 5). Three-year trends for C and SR had deteriorated to threshold conditions (status of C would have been based on 2012 in this example). No trend was assigned to FR and Pf0 and these avoided their thresholds. RI maintained target conditions in two of three years, so no trend at target level was assigned (Table 4-17).

In 2004, all four forage indicators available (C was not available) were at their thresholds (scores assigned were all 1; Table 4-17). Trends away from threshold conditions were not apparent in any of the elements (Table 4-17).

Forage indicators in 2010 were at target conditions (assigned scores = 5), with the exception of SR (Table 4-17). Estimates of SR3 in 2010 were at the threshold, while SR4 was slightly above it; a score of 1 was assigned because the threshold was not avoided by both. RI was moderate and at target (score = 5). This target condition was maintained during 2008-2010 (no trend detected). Estimates of FR, Pf0, and C had improved to target conditions. SR was improving, but both indices had not exceeded the threshold.

Indicator Discussion

Information from ongoing monitoring was used to devise five annual forage indicators for resident Striped Bass in Maryland's portion of Chesapeake Bay. A Striped Bass recreational catch per trip index (RI) provided an index of relative demand. A forage-to-Striped Bass ratio (focal species combined; FR) and grams of all forage consumed per gram of Striped Bass (C) provided trends in supply relative to demand based on relative abundance indices and diet sampling, respectively. Proportion of resident Striped Bass without body fat (Pf0; an index of condition) and trends in survival due to natural mortality indices (SR) provided indicators of Striped Bass well-being. Statistical analyses provided evidence that forage and Striped Bass abundance and well-being were inter-related. Targets and thresholds were then developed for each of these indicators to assign them scores. A score of 1 indicated threshold conditions; a score of 5 indicated target conditions; and scores of 2-4 indicate grades between.

Jacobs et al. (2013) compared Fulton's condition factor, relative weight, percent moisture, and an index of coverage of viscera by visible body fat (body fat index) as indicators of Striped Bass lipid content (nutritional status) with proximate composition. Proximate Analysis partitions compounds in a Striped Bass into moisture, ash, crude protein, crude lipid, crude fiber, and digestible carbohydrates and is the standard for judging nutritional condition. Statistical models developed for both moisture content and the body fat index (including presence or absence of body fat) adequately predicted tissue lipids, offered clear indication of lipid depletion, and cost far less than proximate composition for routine monitoring of nutritional status (Jacobs et al. 2013).

Jacobs et al. (2013) reported a target for body moisture, but a visible body fat target (< 30% of Striped Bass without fat) was also developed (J. Jacobs, NOAA, personal communication). Attainment of "safe" or target nutritional status (low vulnerability to starvation) during October-November was judged by comparing the proportion of Striped Bass without observable visceral fat (Pf0) to a target of 30% or less of Striped Bass with Pf0 (John Jacobs, NOAA, personal communication). In this report, we developed a Pf0 threshold (Pf0 = 68%) that indicated poor condition and high vulnerability to starvation. Jacobs et al. (2013) stressed that comparisons of body fat to nutritional criteria (the body fat target or moisture threshold) should be based on October-November data since the criteria for Chesapeake Bay Striped Bass were developed from samples during that time span.

A Maryland-only RI was defensible based on migratory behavior of resident Striped Bass. Contingent behaviors (fish that share migration patterns) have been identified for Chesapeake Bay Striped Bass based on tagging (Mansueti 1961; Hollis 1967; Setzler et al. 1980; Kohlenstein 1981) and otolith microchemistry (Secor and Piccoli 2007). A small fraction were freshwater residents (remained within the Bay), while most exhibited periods of estuarine or marine residency (former remained within the Bay) after spawning (Secor and Piccoli 2007). Kohlenstein (1981) determined that few young males leave the Chesapeake Bay. Studies of within Bay movements appear to be confined to tagging of 280-430 mm fish in Maryland during 1954-1961 (Mansueti 1961; Hollis 1967). Most tagged Striped Bass remained within Maryland's portion of Chesapeake Bay and very few were recaptured in Virginia. Generally, fish spawning in lower Bay rivers moved out of these systems during summer and then moved northwards in the Bay, while fish that spawned in the upper Bay shifted south. Most fish tagged within the Potomac River were recaptured there (Mansueti 1961; Hollis 1967). Migration studies conducted during the 1930s-1970s found that most Striped Bass (85%-90%) along the coast were females (Setzler et al. 1980). More recent migration studies of Chesapeake Bay Striped Bass based on otolith microchemistry have generally confirmed oceanic movements of females, but have indicated more participation of males in oceanic migrations (Secor and Piccoli 2007). Tag-based estimates of natural mortality for resident Striped Bass in Maryland based on tagging are confined to males (ASMFC 2013).

We used ratios of forage-to-Striped Bass, estimated as a forage index divided by RI, as our indicator of forage supply relative to demand rather than forage indices alone. Rate of consumption of prey by a predator is often based on an assumption that only prey density is important and in this case forage indices would be used. However, this approach does not consider that predators may experience interference from other predators (including their own species) that restricts their feeding success (Ginzburg and Akçakaya 1992; Yodzis 1994; Walters and Juanes 1993; Walters and Martell 2004). A predator's functional response (number of prey consumed per unit area, per unit time by an individual predator; Yodzis 1994) is both a function of attack success and prey handling time. Handling time varies little for a given predator (Yodzis 1994) so attack success can be indexed from the ratio of prey-to-predator (in relative or absolute abundance or biomass; Ginzburg and Akçakaya 1992; Ulltang 1996; Uphoff 2003; Walters and Martell 2004; Uphoff et al. 2009), allowing for the effect of predator interference to be included in the indicator.

A general recommendation for data in stock assessment is that information only be used once (Cotter et al. 2004). In the case of the RI and FR, the same information (RI) is contained in both. However, dividing indices of forage relative abundance by RI to create forage-to-Striped Bass ratios reduced the direct dependency in the data and weighting species-specific indices by the proportion of diet weight each represented further reduced dependency.

Simple consumption indices (grams of each forage fish eaten per gram of sublegal Striped Bass) in fall provided information on forage relative abundance (Maryland Sea Grant 2009). Fall is a period of active feeding and growth for resident Striped Bass and forage fish biomass is at its peak (Hartman and Brandt 1995a; Walter and Austin 2003; Overton et al. 2009; Uphoff et al. 2014). Sublegal Striped Bass diet samples at this time provided an additional index of focal species relative abundance. Focal species accounted for 94% of food consumed, by weight, during October-November 2006-2012. Consumption indices for Maryland's portion of the Bay were available for October-November 2006-2012; data for 2013 and 2014 have been collected. Relative consumption of Spot and Blue Crab was similar in trend to relative abundance indices, but consumption of Atlantic Menhaden and Bay Anchovy was not (Uphoff et al. 2014). Dominance of pelagic prey in Striped Bass diets in years of low pelagic forage abundance indices suggests either larger variations in pelagic prey abundance existed than were measured by surveys or availability to Striped Bass varied beyond abundance due to prey or predator distribution and behavior (Uphoff et al. 2014).

Shifts of SR3 and SR4 during 1990-2013 lagged behind those of other indices, but shifted from states of higher to lower survival very rapidly (Figure 4-19). Estimates of SR3 and SR4 rose to or above their threshold in 2011, even though target conditions were met for other forage indicators during 2008 and 2010 (Figure 4-19). Below target forage conditions returned in 2011, but SR3 and SR4 did not fall below its threshold until 2012. Estimates of FR first reached their threshold in 1995; SR3 fell below its threshold the following year, while SR4 fell below it two years later. High SR during 1990-1996 reflected stable and above target FR. Low SR afterwards has reflected FR that has mostly been at its threshold or below target (Figure 4-20). Dutil and Lambert (2000) found that natural mortality of Atlantic Cod could be delayed after unfavorable conditions. This lagged response in SR and a possible need for persistent target forage conditions make it insensitive to immediate changes that could create "false-positives" or "false-negatives" for decision-makers.

The best tagging model estimating natural mortality (M) of legal-size male Striped Bass in Chesapeake Bay (residents) in the recent stock assessment determined M for two time periods (ASMFC 2013). Based on our conversion of M to survival, estimates of survival during 1987-1996 (77%) were considerably better than 1997-2011 (44%), duplicating the high-to-low patterns observed in SR3 and SR4.

Adopting a target for FR based on early time-series conditions (1990-1994) would lead to managing for hard to duplicate forage conditions. High FR during 1990-1994 reflected high forage status and depleted Striped Bass. Using Atlantic Menhaden-to-Striped Bass ratios as an example, the average of ratios during 1990-1994 equaled the period's average Atlantic Menhaden GM (2.28) divided by the average RI (0.66). Twenty-one (62%) of Atlantic Menhaden GMs during 1959-2013 were at or above this index and a value this large or large was last reached in 1991. If average RI during 2008-2013 (1.21, representing moderate resident Striped Bass abundance) is used as the Striped Bass management target (denominator of the ratio), an Atlantic Menhaden GM of 8.5 is required. This Atlantic Menhaden GM would be met or exceeded by only seven other GMs during 1959-2013 (11% of all GMs) and a GM this high has not occurred since 1981. If high Striped Bass abundance is desired (indicated by the 1995-2007 average RI, 2.65), then Atlantic Menhaden GM's need to be 18.6 or higher. An Atlantic Menhaden GM this high has never been measured in Maryland's portion of Chesapeake Bay.

Piscivorous fish such as Striped Bass depend on high densities of proper forage and safe foraging opportunities to grow and survive (Persson and Brönmark 2002). In Atlantic Cod and Striped Bass, body condition declined and natural mortality increased dramatically concurrent with declines of important forage fish (Lilly 1994; Lambert and Dutil 1997; Dutil and Lambert 2000; Shelton and Lilly 2000; Uphoff 2003; Maryland Sea Grant 2009; Jacobs et al. 2013). It has been suggested that rebuilding of some depleted Atlantic Cod stocks and Atlantic coast Weakfish is dependent on forage fish production (Rose and O'Driscoll 2002; ASMFC 2009). Energy reserves of individual fish and populations relate strongly to foraging success, and subsequent fish health and survival (Jacobs et al. 2013). Low forage fish relative to piscivore abundance leading to poor nutritional state and high natural mortality form a viable alternative hypothesis to overfishing that has direct implications for managing fisheries (Hare 2014).

Sublegal Striped Bass initiate feeding on fish, primarily Bay Anchovy, as yearlings, (Hartman and Brandt 1995b). Within an additional two years, sublegal Striped Bass grow enough to switch to juvenile Atlantic Menhaden and Spot (Hartman and Brandt 1995b). Bay Anchovy, juvenile Atlantic Menhaden, and Spot have generally been at reduced levels in Maryland's portion of Chesapeake Bay since the mid-1990s, while abundance of resident Striped Bass has been high, falling to moderate, in recent years. Early switching to a fish diet requires high growth rate and high densities of proper forage (Persson and Brönmark 2002). Abundant individuals competing for limited prey may hinder one another's feeding activities, leading to starvation (Yodzis 1994). 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). Poor foraging conditions for resident Striped Bass during 1995-2013 were also concurrent with a Mycobacteriosis epizootic in Chesapeake Bay and this disease's progression was linked to nutrition through a series of challenge studies (Jacobs et al. 2009).

Using a forage fish indicator approach for EAF in Chesapeake Bay requires a broader management perspective than is currently encompassed in single-species management (Maryland Sea Grant 2009). Using management of Atlantic coast Striped Bass as an example, fishing mortality is controlled to maintain high spawning stock biomass through interstate management coordinated by Atlantic States Marine Fisheries Commission (Maryland Sea Grant 2009). This strategy does not address factors such as forage supply, predation, competition, and disease or regional ecological problems for resident Striped Bass in Chesapeake Bay. Confounding of migration and mortality complicates assessment of resident Striped Bass using a technique such as the statistical catch-at-age model used for the Atlantic Coast.

A framework of indicator targets and thresholds depicted issues in an ecological context (Maryland Sea Grant 2009). Analyses of indictors supported the overall hypothesis that forage-related, density-dependent processes were influencing resident Striped Bass in Maryland's portion of Chesapeake Bay. These analyses did not confirm that underlying assumptions of the single-species model for the Atlantic coast (constant M and little density-dependence) provided an adequate description for resident Striped Bass in Chesapeake Bay. Evidence of overfishing of Chesapeake Bay Striped Bass was lacking in the benchmark assessment after the 1980s, (ASMFC 2013).

Inadequate prey resources for Striped Bass are a common problem in lakes (Axon and Whitehurst 1985; Brown and Murphy 1991; Raborn et al. 2007; Cyterski and Ney 2005; Thompson et al. 2010). Managers of Striped Bass populations in lakes with inadequate forage may attempt to manipulate prey supply through introductions but is more feasible to reduce demand through reduced stocking and increased harvest of Striped Bass (Raborn et al. 2007; Cyterski and Ney 2005). However, controlling prey-predator imbalances in lakes by reducing demand can be limited by natural variability in prey recruitment (Raborn et al. 2007). Management of prey may seem more viable for Chesapeake Bay Striped Bass than for lakes since a large fishery exists for an important prey (Atlantic Menhaden). However, young-of-year Atlantic Menhaden are most important to resident Striped Bass, particularly sublegal fish, and increasing their production by manipulating the Atlantic Menhaden fishery to benefit resident Striped Bass depends on how directly Atlantic Menhaden recruitment responds to spawning stock. Unfortunately, this relationship is not direct for Atlantic Menhaden (SEDAR 2015). The recent Atlantic Menhaden stock assessment indicates that fishing mortality has fallen substantially, and biomass and population fecundity has risen. Atlantic Menhaden juvenile indices in Maryland's portion of Chesapeake Bay have not responded strongly, even though they increased north of the Bay (SEDAR 2015). Larger legal-sized resident Striped Bass would benefit from more abundant age 1+ Atlantic Menhaden, but they are too large to be consumed by sublegal Striped Bass.

Competition for forage should have its greatest effect on sublegal resident Striped Bass because they compete with one another and legal-sized Striped Bass for much of the same forage. Legal-sized Striped Bass should forage more efficiently and outcompete sublegal fish through greater visual acuity, swimming speed, and experience with the competitive arena (Ward et al. 2006).

Management of resident Chesapeake Bay Striped Bass shares with lakes the possibility of increasing harvest to reduce demand. Uphoff (2003) estimated that demand for Atlantic Menhaden per Chesapeake Bay Striped Bass recruit increased from 11-16% to 31-55% of an unfished stock due to regulatory changes enacted after the 1980s. Decreasing age-at-entry to the fishery decreased demand by 8-10% at any given F. Demand fell as F increased, but the proportionally largest declines in demand occurred when F was increasing between 0 and 0.4 (Uphoff 2003). Adjustments to demand through increased harvest of Striped Bass would need to be balanced with need for escapement of fish from the Chesapeake Bay to the Atlantic Coast (Richards and Rago 1999). Strong year-classes are a predominant feature of Striped Bass population dynamics (Richards and Rago 1999) that may swamp harvest-based actions to reduce forage demand for periods of time.

Adopting an ecosystem approach to fisheries management is inherently evolutionary (Cowan et al. 2012). The primary use of the forage fish indicator at this time would be for reporting. Use for management support is possible, but a process for incorporating this information into the current single-species management framework is lacking. The suite of indicators developed herein to assess forage status was not viewed as permanent. In coming years, new information may need to be adopted and outdated information discarded. We have provided a starting point that seems acceptable to managers so far.

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Table 4-1. Comparison of variables entered from Chesapeake Bay Ecological Foundation ledgers (Entered variable) and variables created through entry and editing by the Fish Habitat and Ecosystems Program (Edited variable). Descriptions are not provided for variables that were self-evident.

Entered variable	Description	Edited variable	Description
location	Specific or general	Sequence	Number assigned to each line for year
year		Date	Derived from month day for each year
month		Area	Choptank R., Bay, or Ocean
day		location	Specific when available
gear	Hook and line unless noted	year	
fish #	Sequence of fish examined on a day	month	
TL(in)	Total length of Striped Bass in inches	day	
Sex	Male, female, unknown	fish_num	Sequence of fish examined on a day
Stomach Contents	Diet item label; may not be standardized	TL_in	Total length of Striped Bass in inches
Partial (1)	Blank = intact; 1 = part identifiable to species	TL_mm fill	TL in mm assigned to each line of data
inches	Length of diet item, inches	TL_mm	TL in mm assigned once to Striped Bass
mm	Length of diet item, mm	Sex	Male, female, unknown
Spleen	Nodule classification; 0-3; 0 = no nodules	Stomach Contents	Diet item label; may not be standardized
B. Fat	Classes 0-3; subclasses < 1	Contents edit	Standardized diet item label
Lb.	Striped Bass weight measured, grams	Wild food present	1 = present, 0 = absent, blank = bait of not food
Oz.	Striped Bass weight measured, grams	Part_1	Blank = intact; 1 = part identifiable to species
gms	Striped Bass weight measured, grams	Food_L_in	Length of diet item, inches
Gonad	Abbreviation indicating status	Food_L_mm	Length of diet item, mm
Comments	Miscellaneous	Food wt	Estimated weight of diet item, grams
		Gizzard_mm	Size of fish gizzard
		Spleen	Nodule classification; 0-3; 0 = none
		B. Fat	Classes 0-3; subclasses < 1
		Lb.	Striped Bass weight measured, grams
		Oz.	Striped Bass weight measured, grams
		gms	Striped Bass weight measured, grams
		Estimated gms	Striped Bass weight (grams) estimated from TL

Table 4-2. Summary of information on Striped Bass diet items. Wild = 1 indicates wild diet item and 0 indicates exclusion. Parameters a and b are for the allometric length-weight (mm and grams) equations ($Wt = a \cdot L^b$). A "Y" under Hartman indicates allometric equation in Hartman and Brandt (1995b); a "N" indicates an alternative source was used.

Contents edit name	Common name	Genus species	Wild	a	b	Comment	Hartman
Amphipod	Gammarus sp.	Gammarus sp.	1			See Table 3.	
Anchovy	Bay Anchovy	*Anchoa mitchilli*	1	0.0000005	3.57		Y
Bait Spot	Spot	*Leistomus xanthurus*	0	0.0000074	3.13		Y
Bait White Perch	White Perch	*Morone americana*	0	3.20E-06	3.29	Choptank MD	N
Bait pieces	Fish		0			Size unknown	
Blue Crab	Blue Crab	*Callinectes sapidus*	1	0.0000959	2.86		Y
Blueback Herring	Blueback Herring	*Alosa aestivalis*	1	0.0000046	3.52		Y
Butterfish	Butterfish	*Peprilus triachanthus*	1	0.000016	3.08		N
Clam	Use Soft Clam	*Mya arenaria*	1	0.0002341	2.899	3 gm if not measured	N
Clam shell			0				
Clam snout			1			0.5 gm	
Croaker	Atlantic Croaker	*Micropogonius undulatus*	1	0.0000022	3.33		Y
Flounder	Flounder sp.		1	0.0000056	3.1	Summer Flounder	N
Gizzard Shad	Gizzard Shad	*Dorosoma cepedianum*	1	0.0000007	3.6		Y
Goby	Naked Goby	*Gobiosoma bosc*	1	0.0002088	2.24		Y
Grass shrimp	Grass shrimp	*Palaemonetes pugio*	1	0.0000047	3.2		Y
Grasshopper			0				
Herring	Clupeid		1	0.0000007	3.6		Y
Mantis Shrimp	Mantis Shrimp	*Squilla empusa*	1	0.0000047	2.86		Y
Menhaden	Atlantic Menhaden	*Brevoortia tyrannus*	1	0.0000022	3.35		Y
Mud Crab		*Panopeous*	1	0.0000959	2.86	Blue Crab	
Mussel	Ribbed Mussel		1			Use soft clam or 3 gm	
None	None		0				
Oyster shell			0				
Parasitic arthropod	Isopod		0				
Pipefish	Northern Pipefish	*Sygnathus fucus*	1	0.0000007	3.6		Y
Polychaete	Polychaete		1			See Table 3.	
Razor Clam	Razor Clam	*Perkinsus chesapeaki*	1			3 gm	
Regurgitated empty			0				
Sand shrimp	Grey Sand Shrimp	*Crangon septimspinosa*	1	0.0000047	3.2	See Table 3.	Y
Shrimp	Grass or Sand		1	0.0000047	3.2	See Table 3.	Y
Silverside	Silverside	*Menidia sp*	1	0.0000074	2.95		
Skilletfish	Skilletfish	*Gobiesox strumosus*	1	0.0000046	3.52	Oyster Toadfish	Y
Soft Clam	Soft Clam	*Mya arenaria*	0	0.0002341	2.899	3 m if not measured	N
Soft invertebrate residue			1			See Table 3.	
Spine (mspine, etc)			0				
Spot	Spot	*Leistomus xanthurus*	1	0.0000074	3.13		Y
Tunicate			1			Mean weight 0.5 gm	
Unknown crabs			1			Blue Crab	Y
Unknown fish			1	0.0000007	3.6		Y
Unknown fish parts			0				
Unknown residue			0				
White Perch	White Perch	*Morone americana*	1	0.0000032	3.29	MD Choptank R.	N

Table 4-3. Equations for converting item frequency into relative weight (Stobberup et al. 2009). Relative weight (RWi) was the percentage of total diet weight represented by items representing uncountable invertebrates. Prey type refers to the conversion equation used. X = frequency of occurrence.

Contents Edit	Prey Type	Conversion equation
Shrimp, Sand Shrimp	Benthic Crustacean	1.051X - 0.654
Amphipod	Zooplankton	((1.051 + 0.89)X) - (0.654-0.023)
Soft Invertebrate residue	Zooplankton	((1.051 + 0.89)X) - (0.654-0.023)
Polychaete	Worm	((1.051 + 0.049)X) - (0.654-0.55)

Table 4-4. Summer length-weight regression parameters developed from hook and line harvested Striped Bass measured at checkstations (B. Versak, MD DNR, personal communication). Regressions are for $\log_e$ weight (grams) versus $\log_e$ length (mm, TL). These relationships were applied to legal-sized Striped Bass.

YEAR	r^2	n	intercept	slope
2006	0.91	2,100	-20.06	3.25
2007	0.89	1,675	-21.09	3.41
2008	0.88	1,624	-19.81	3.21
2009	0.93	2,259	-20.34	3.29
2010	0.91	1,789	-19.13	3.10
2011	0.94	1,328	-19.94	3.22
2012	0.93	1,988	-19.95	3.23
2013	0.94	1,952	-19.74	3.20

Table 4-5. Feeding metrics, their abbreviations, and formulas that were used to summarize annual Striped Bass diets during summer, 2007-2008.

Metric	Abbreviation	Formula
Proportion without food	P_{none}	Count "None" / count all Striped Bass
Proportion of all grams consumed represented by item i	Pwi	∑ Grams of item i / ∑ Grams of all items; fish with food only
Number of item i consumed per Striped Bass	MN_i	Count of item i / count of all Striped Bass
Grams of item i consumed per gram Striped Bass	MW_i	∑ Grams of item i / ∑ grams of all Striped Bass

Table 4-6. Summary of resident Striped Bass size class diet (by weight) during summer 2007-2008. Letter P before a diet item indicates proportion of the item represented by weight. P without food and P with bait indicate proportions based on counts.

Item	2007 Sublegal	2007 Legal	2008 Sublegal	2008 Legal
P Menhaden large		0.594		0.820
P Menhaden YOY		0.019	0.656	0.050
P Bay Anchovy	0.859	0.016	0.095	0.024
P Spot	0.020	0.225	0.208	0.098
P White Perch	0.000	0.134		
P Other fish	0.012			0.001
P Blue Crab	0.082	0.001		0.005
P Mud crab			0.018	0.001
P Polychaete	0.005	0.005	0.000	
P Shrimp	0.006	0.005	0.006	
P Soft Invert	0.005		0.007	
P Bivalves	0.011		0.010	0.001
gm / gm		0.0026		0.0074
gm / fish	1.5		2.7	
N with food	73	41	39	108
N examined	198	433	111	407
P without food	0.63	0.91	0.65	0.73
N with bait	2	28	3	40
P with bait	0.01	0.06	0.03	0.10

Table 4-7. Summary of monthly estimates of proportion of Striped Bass without body fat (Pf0) during summer 2007-2008. Shading indicates months within a size class that were not considered different based on confidence interval overlap.

Month	Size class	Pf0	N	SD	Lower 95%	Upper 95%
June-July	Sublegal	0.24	54	0.06	0.13	0.35
August	Sublegal	0.45	65	0.06	0.33	0.57
September	Sublegal	0.88	25	0.06	0.75	1.01
June-July	Legal	0.33	141	0.04	0.26	0.41
August	Legal	0.38	128	0.04	0.30	0.47
September	Legal	0.76	106	0.04	0.68	0.84
June-July	Sublegal	0.40	42	0.08	0.26	0.55
August	Sublegal	0.36	50	0.07	0.23	0.49
September	Sublegal	0.72	18	0.11	0.52	0.93
June-July	Legal	0.09	197	0.02	0.05	0.13
August	Legal	0.34	104	0.05	0.25	0.43
September	Legal	0.40	84	0.05	0.30	0.51

Table 4-8. Summary of Striped Bass body fat indicators for all sizes sampled by FWHP. N no fat = number without body fat; N = number examined; Pf0 = proportion without body fat; SD = standard deviation of the proportion; Up 95% = upper 95% confidence interval; Low 95% = lower 95% confidence interval; P < target = chance of being at the target body fat criterion (< 30% of sample without body fat); P > threshold = chance of being at the body fat threshold (> 68% of sample without body fat). Shading indicates non-zero chances.

Year	N	N no fat	Pfo	SD	Low 95%	Up 95%	P < target	P > threshold
1998	338	253	0.75	0.024	0.70	0.79	0.00	1.00
1999	344	268	0.78	0.022	0.74	0.82	0.00	1.00
2000	290	224	0.77	0.025	0.72	0.82	0.00	1.00
2001	224	167	0.75	0.029	0.69	0.80	0.00	0.98
2002	316	191	0.60	0.028	0.55	0.66	0.00	0.00
2003	237	166	0.70	0.030	0.64	0.76	0.00	0.77
2004	414	309	0.75	0.021	0.70	0.79	0.00	1.00
2005	524	312	0.60	0.021	0.55	0.64	0.00	0.00
2006	863	518	0.60	0.017	0.57	0.63	0.00	0.00
2007	662	331	0.50	0.019	0.46	0.54	0.00	0.00
2008	629	86	0.14	0.014	0.11	0.16	1.00	0.00
2009	1107	345	0.31	0.014	0.28	0.34	0.19	0.00
2010	693	187	0.27	0.017	0.24	0.30	0.96	0.00
2011	1202	638	0.53	0.014	0.50	0.56	0.00	0.00
2012	333	219	0.66	0.026	0.61	0.71	0.00	0.21
2013	441	254	0.58	0.024	0.53	0.62	0.00	0.00

Table 4-9. Information used to calculate the recreational catch per trip index for resident Striped Bass in Maryland's portion of Chesapeake Bay (RI; NMFS Fisheries Statistics Division 2014). Catch (harvest + releases) and effort are for private / rental boat categories. Estimates are for September-October (wave 5). PSE is the standard error of the estimate as a percentage of the mean.

Year	Catch	Catch PSE	Trips	Trips PSE	RI
1981	38,607	36.1	102,020	26	0.38
1982			200,118	43.3	
1983	49,078	48.8	366,459	19.3	0.13
1984	59,371	52.3	222,710	42.6	0.27
1985	28,410	79	396,670	32.7	0.07
1986	156,997	46	227,042	28.1	0.69
1987			150,496	24.8	
1988	82,152	92.9	233,818	25.4	0.35
1989	26,932	45.4	204,486	26.1	0.13
1990	152,258	22.7	305,262	13.7	0.50
1991	189,196	20.1	413,068	18.7	0.46
1992	205,337	19	329,043	17.9	0.62
1993	358,180	22	601,701	15.7	0.60
1994	496,777	23.6	432,623	20	1.15
1995	1,424,111	23.6	562,280	19.1	2.53
1996	1,090,206	20.9	395,827	18.3	2.75
1997	1,174,092	18.5	406,853	18.4	2.89
1998	1,013,374	17.4	422,189	16.3	2.40
1999	937,929	18.7	395,590	16.5	2.37
2000	985,773	17.6	488,566	14.3	2.02
2001	1,159,641	17.9	609,871	13.9	1.90
2002	1,375,625	17.1	423,692	11.6	3.25
2003	1,431,169	16.8	462,756	11.3	3.09
2004	1,341,184	25.3	350,431	16.3	3.83
2005	1,219,890	27.4	519,304	12.8	2.35
2006	1,125,725	24.4	403,967	9.9	2.79
2007	1,275,934	47.4	571,237	13.2	2.23
2008	352,624	23.4	440,312	12.2	0.80
2009	314,071	24.1	190,622	20.2	1.65
2010	369,037	24.8	463,230	12.4	0.80
2011	465,719	19.8	381,143	14.9	1.22
2012	466,941	30	310,632	15.2	1.50
2013	514,632	24.1	392,868	21.4	1.31

Table 4-10. Summary statistics for best regression model of the resident Striped Bass recreational catch per trip index (RI) and Maryland's juvenile indices (geometric means).

Analysis of Variance					
Source	df	SS	MS	F Value	Pr > F
Model	4	30.06263	7.51566	41.63	<.0001
Error	25	4.51305	0.18052		
Corrected Total	29	34.57568			
Root MSE	0.42488				
Dependent Mean	1.55502	R-Square	0.8695		
Coeff Var	27.3231	Adj R-Sq	0.8486		

	df	Parameter Estimate	Standard Error	t Value	Pr > |t|	Squared Partial Corr Type I	Squared Partial Corr Type II	Cumulative r^2
Intercept	1	-0.23232	0.15975	-1.45	0.1583	.	.	
JI Yr+2	1	0.08473	0.02005	4.23	0.0003	0.30143	0.41658	0.41
JI Yr+3	1	0.11993	0.01947	6.16	<.0001	0.47397	0.60292	0.69
JI Yr+4	1	0.12038	0.01952	6.17	<.0001	0.54493	0.60339	0.83
JI Yr+5	1	0.05402	0.02038	2.65	0.0137	0.21944	0.21944	0.87

Table 4-11. Summary statistics for regression models of the resident Striped Bass recreational catch per trip index (RI) and biomass estimates for age 2-5 resident and migratory Striped Bass from the ASMFC (2013) stock assessment with all data and a possible outlier removed.

Summary Output - all data

Regression Statistics	
R-Square	0.671066308
Adjusted R-Sq	0.66
Standard Error	0.67
Observations	29

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	24.71	24.71	55.08	5.53025E-08
Residual	27	12.11	0.45		
Total	28	36.82			

	Parameter	*Standard Error*	*t value*	*P-value*	*Lower 95%*	*Upper 95%*
Intercept	-0.7	0.3	-2.1	0.047	-1.38	-0.01
Age 2-5 biomass	0.0	0.0	7.4	5.53E-08	3.00497E-08	5.30133E-08

Summary Output – 2004 outlier removed

Regression Statistics	
R-Square	0.80
Adjusted R-Sq	0.80
Standard Error	0.46
Observations	28

ANOVA

	df	*SS*	*MS*	*F*	*Significance F*
Regression	1	22.79	22.79	106.20	1.12871E-10
Residual	26	5.58	0.21		
Total	27	28.37			

	Parameter	*Standard Error*	*t value*	*P-value*	*Lower 95%*	*Upper 95%*
Intercept	-0.70	0.23	-3.04	0.005389	-1.17	-0.23
Age 2-5 biomass	0.00	0.00	10.31	1.13E-10	3.2015E-08	4.79684E-08

Table 4-12. Correlations of forage indices to Striped Bass relative abundance (RI) ratios and proportion of Striped Bass without body fat (Pf0) during 1998-2013.

Forage Ratio	Statistics	Pf0
Atlantic Menhaden ratio	r	-0.63
	P	0.009
Anchovy ratio (seine)	r	-0.32
	P	0.220
Anchovy ratio (trawl)	r	0.45
	P	0.060
Spot ratio	r	-0.51
	P	0.042
Blue Crab ratio	r	-0.40
	P	0.126

Table 4-13. Correlations of forage indices to Striped Bass relative abundance (RI) ratios and relative survival of 3- and 4-year-old Striped Bass (SR3 and SR4, respectively), during 1990-2013.

Forage ratio	Statistic	SR3	SR4
Atlantic Menhaden	r	0.58	0.44
	P	0.0014	0.0217
	N	27	27
Bay Anchovy (seine)	r	0.60	0.47
	P	0.0011	0.0136
	N	27	27
Bay Anchovy (trawl)	r	0.48	0.51
	P	0.0177	0.0117
	N	24	24
Spot	r	0.27	0.17
	P	0.1689	0.399
	N	27	27
Blue Crab	r	0.40	0.43
	P	0.0492	0.0325
	N	25	25

Table 4-14. Correlations among ratios of forage indices to Striped Bass relative abundance during 1990-2013.

Ratio	Statistic	Bay Anchovy (seine)	Spot	Bay Anchovy (trawl)	Blue Crab
Atlantic Menhaden	r	0.79	0.81	0.96	0.89
	P	<0.0001	<0.0001	<0.0001	<0.0001
	N	27	27	24	25
Bay Anchovy (seine)	r		0.44	0.91	0.84
	P		0.0222	<0.0001	<0.0001
	N		27	24	25
Spot	r			0.72	0.80
	P			<0.0001	<0.0001
	N			24	25
Bay Anchovy (trawl)	r				0.96
	P				<0.0001
	N				24

Table 4-15. Correlations of forage indices and relative survival of 3- and 4-year-old Striped Bass (SR3 and SR4, respectively) during 1990-2013.

Ratio	Statistic	SR3	SR4
Atlantic Menhaden	r	0.79	0.53
	P	<.0001	0.0049
	N	27	27
Bay Anchovy (seine)	r	0.74	0.52
	P	<.0001	0.0056
	N	27	27
Bay Anchovy (trawl)	r	0.17	0.55
	P	0.4307	0.0056
	N	24	24
Spot	r	0.19	0.02
	P	0.3392	0.9214
	N	27	27
Blue Crab	r	0.17	0.56
	P	0.419	0.0038
	N	25	25

Table 4-16. Correlations among forage indices during 1990-2013.

Ratio	Statistic	Bay Anchovy (seine)	Spot	Bay Anchovy (trawl)	Blue Crab
Atlantic Menhaden	r	0.60	0.50	0.35	0.12
	P	0.0009	0.0083	0.0951	0.5686
	N	27	27	24	25
Bay Anchovy (seine)	r		0.0029	0.27	-0.05
	P		0.9885	0.2009	0.8096
	N		27	24	25
Spot	r			-0.24	-0.25
	P			0.2626	0.2306
	N			24	25
Bay Anchovy (trawl)	r				0.07
	P				0.7561
	N				24

Table 4-17. Examples of prototype indicator table for assessing forage fish status in Maryland's portion of Chesapeake Bay during 2013 (latest year), 2004 (poor year), and 2010 (good year). Estimates of grams of forage eaten per gram of sublegal (< 457 mm, TL) Striped Bass for 2012 were used for 2013 in this example.

Indicator	2013 Status	2011-2013 Trend
Forage fish-to-Striped Bass ratio (FR)	2	No trend
Grams forage eaten per gram sublegalStriped Bass (C)	2	Deteriorate
Resident Striped Bass relative abundance (RI)	5	No trend
Percent Striped Bass without body fat (Pf0)	2	No trend
Relative survival (SR3 and SR4)	1	Deteriorate

Indicator	2004 Status	2002-2004 Trend
Forage fish-to-Striped Bass ratio (FR)	1	No trend
Grams forage eaten per gram sublegal Striped Bass (C)	Not available	Not available
Resident Striped Bass relative abundance (RI)	1	No trend
Percent Striped Bass without body fat (Pf0)	1	No trend
Relative survival (SR3 and SR4)	1	No trend

Indicator	2010 Status	2008-2010 Trend
Forage fish-to-Striped Bass ratio (FR)	5	No trend
Grams forage eaten per gram sublegal Striped Bass (C)	5	Improve
Resident Striped Bass relative abundance (RI)	5	Improve
Percent Striped Bass without body fat (Pf0)	5	Improve
Relative survival (SR3 and SR4)	1	Improve

Figure 4-1. Maryland's portion of Chesapeake Bay and approximate regions with forage indices, diet data, and body fat information collected by MD DNR Fish and Wildlife Health Program.

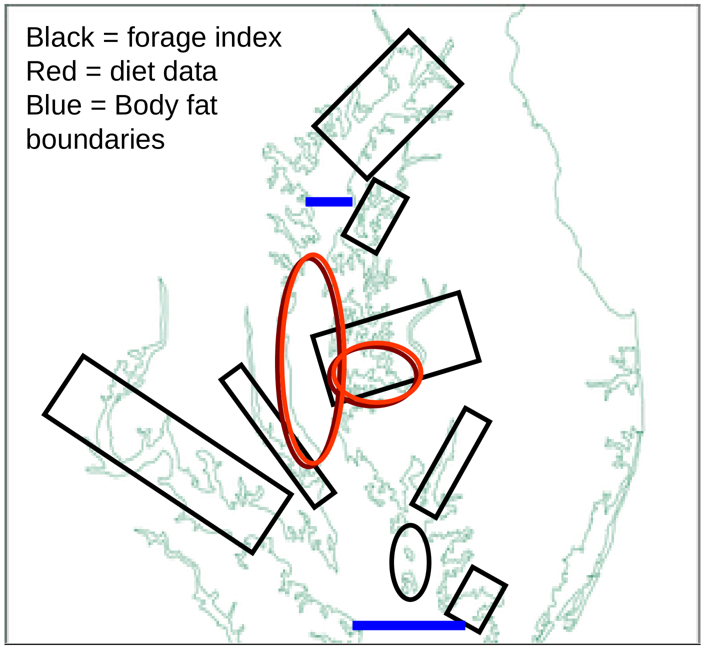

Figure 4-2. Proportions of Striped Bass without body fat by length category or categories combined. Data collected by MD DNR Fish and Wildlife Health Program.

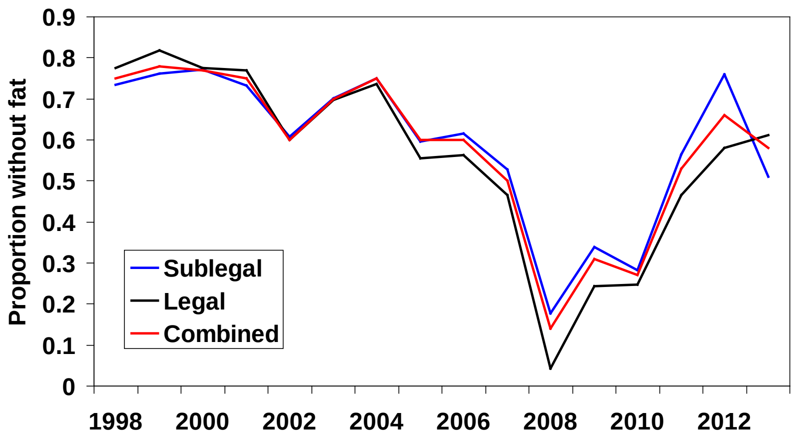

Figure 4-3. Estimates of the proportion of Striped Bass without body fat and their 95% confidence intervals during October-November monitoring by the Fish and Wildlife Health Program. Estimates at or below the green line meet the target for nutritional condition. Estimates above the threshold red line indicate a large fraction are vulnerable to starvation.

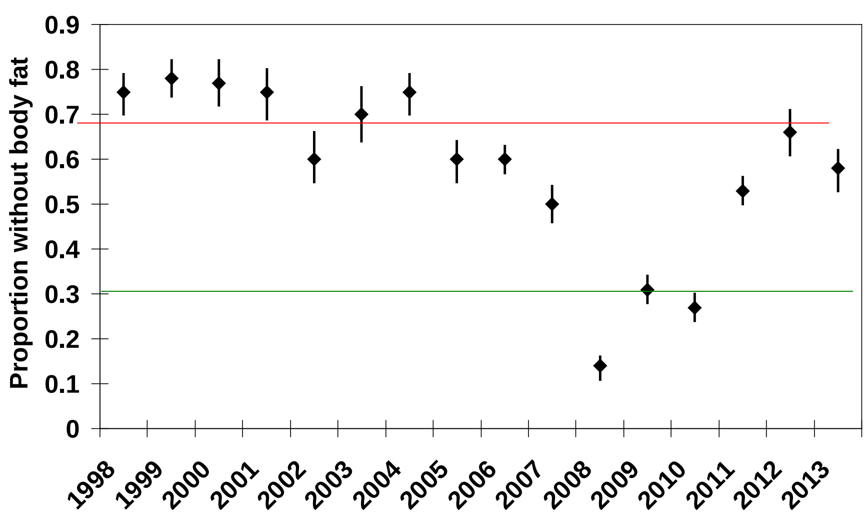

Figure 4-4. Trends in major pelagic prey of Striped Bass in Maryland's juvenile seine survey. Indices were standardized to their 1989-2013 means. For Bay Anchovy, S indicates seine survey and T indicates summer Blue Crab Trawl Survey.

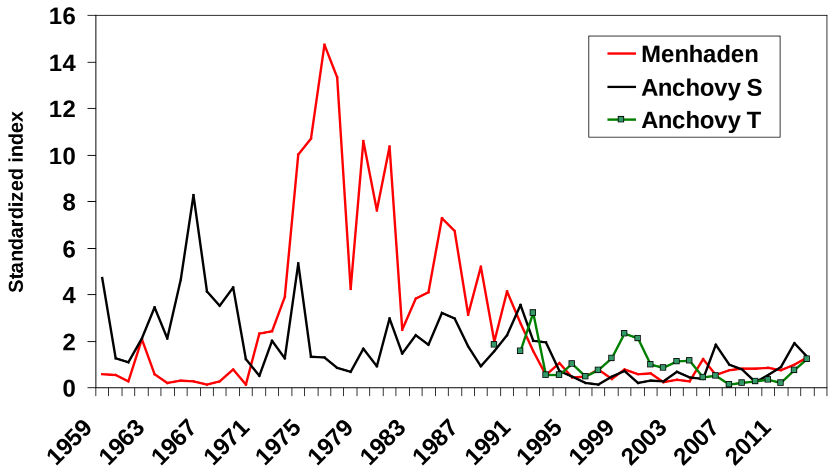

Figure 4-5. Trends in major benthic prey of Striped Bass in Maryland surveys. Indices were standardized to their 1989-2012 means.

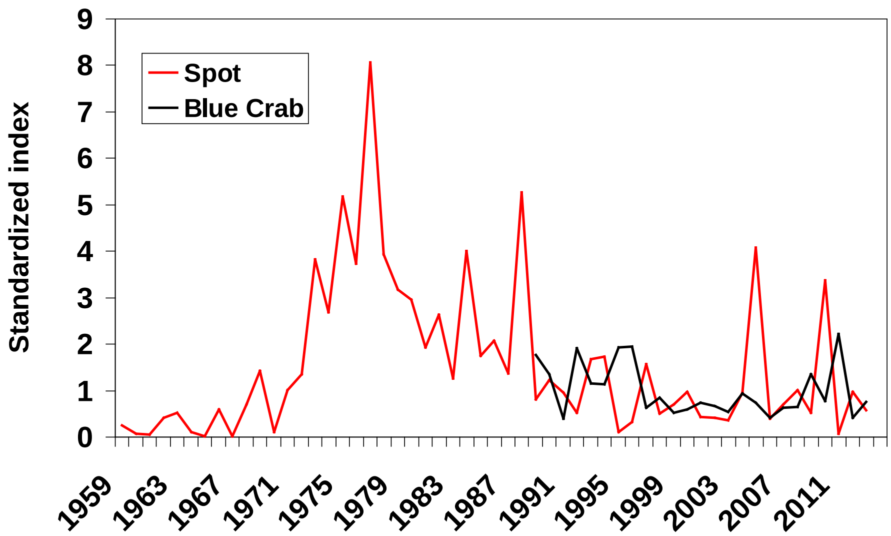

Figure 4-6. Recreational catch per trip index (RI) for resident Striped Bass in Maryland's portion of Chesapeake Bay and predictions from regression equations using Maryland juvenile indices (JI) or coastal assessment (SCAM) abundance estimates. A negative prediction of RI in 1983 was omitted from SCAM-based plots

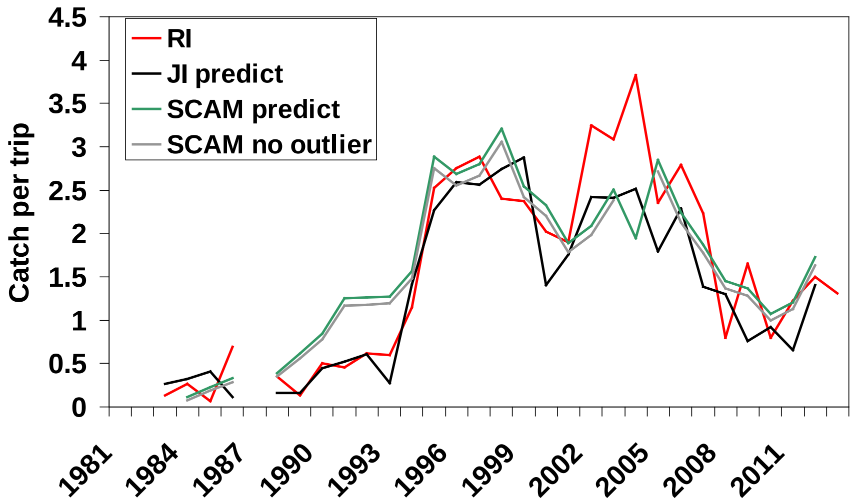

Figure 4-7. Trends in standardized forage-to-Striped Bass ratios, 1990-2013.

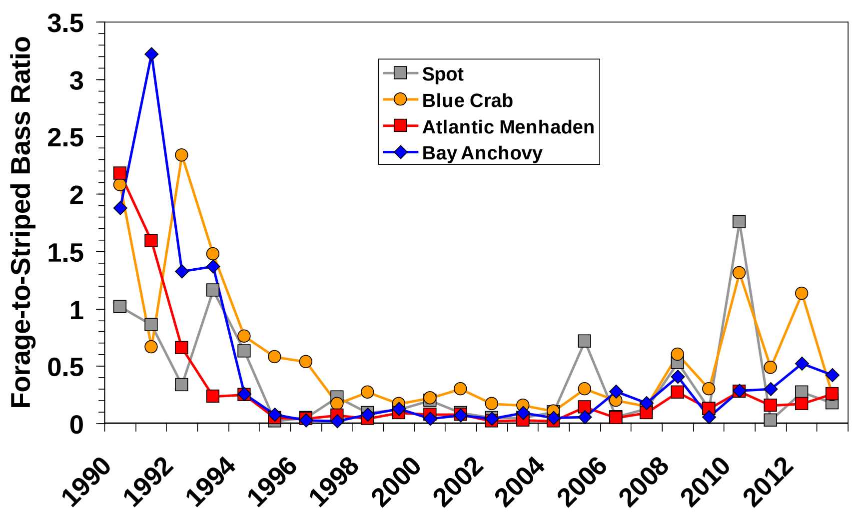

Figure 4-8. Relationship of proportion of Striped Bass without body fat and inverse of relative abundance of Striped Bass (catch per private / rental boat trip in Maryland's portion of Chesapeake Bay) during 1998-2013.

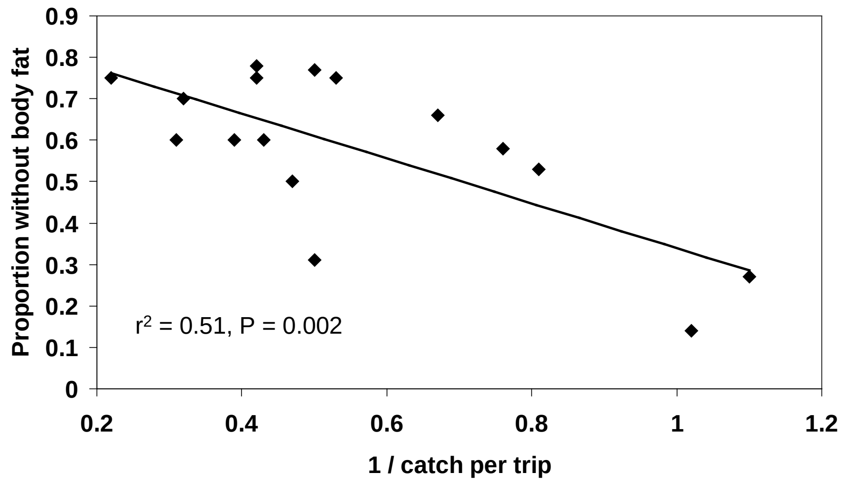

Figure 4-9. Trends in relative survival of sublegal male Striped Bass (1985-2013) and survival from tag M estimate (1987-2012; ASMFC 2013) of legal-size Striped Bass in MD's portion of Chesapeake Bay

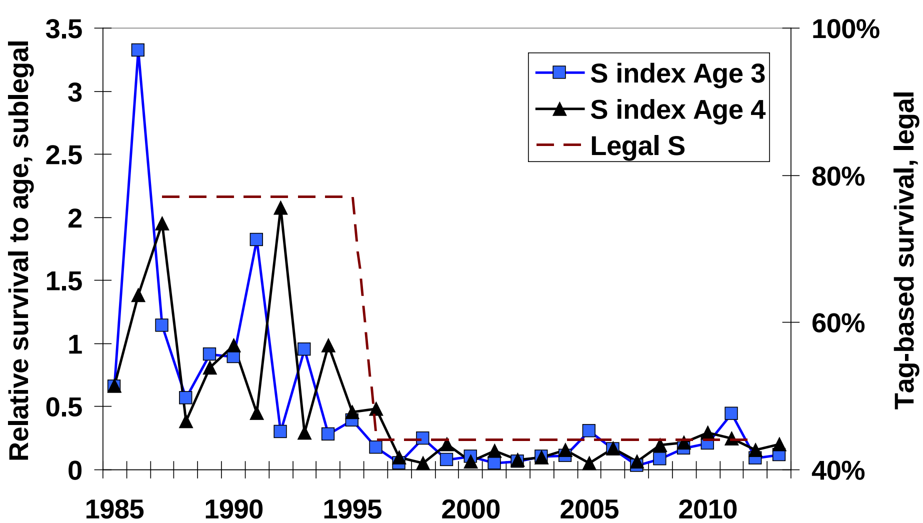

Figure 4-10. Log_e-transformed Striped Bass survival indices for ages 3 (SR3) and 4 (SR4) plotted against resident Striped Bass relative abundance. Predicted line depicts combined data.

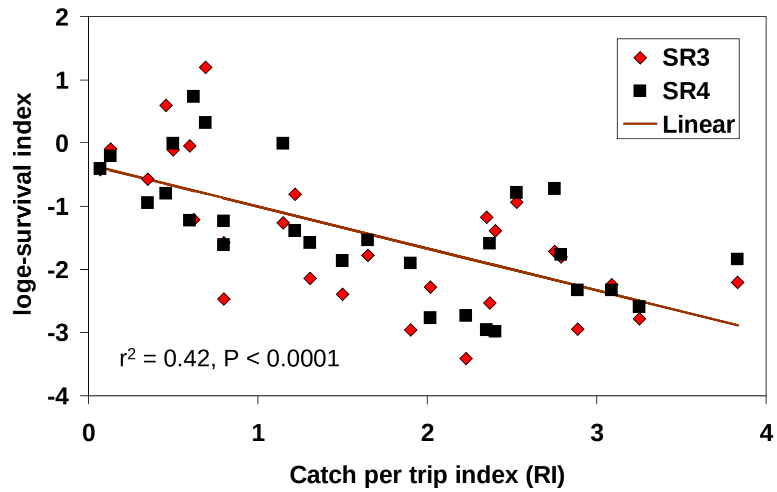

Figure 4-11. Residuals of the linear regression of log_e-transformed Striped Bass survival indices for ages 3 (SR3) and 4 (SR4) combined versus relative abundance (RI) against (A) RI and (B) year.

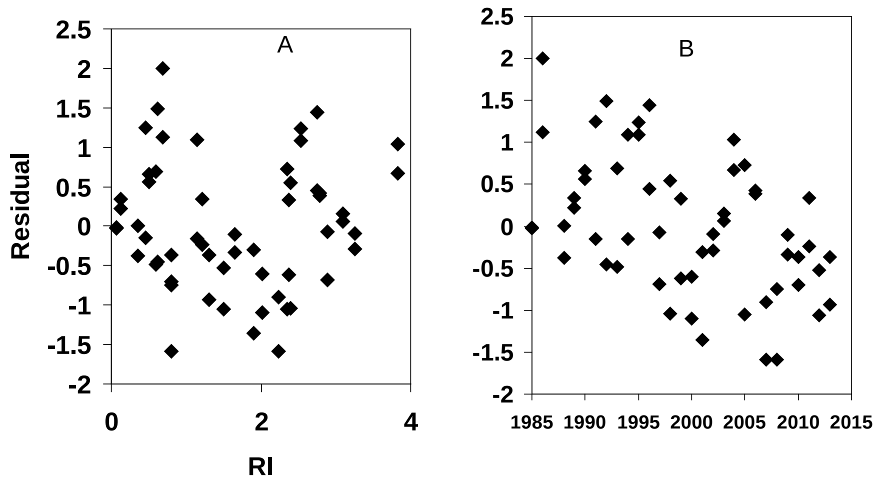

Figure 4-12. Trends in proportion of resident Striped Bass without body fat (Pf0) in October-November and survival indices for age 3 (SR3) and 4 (SR4) fish.

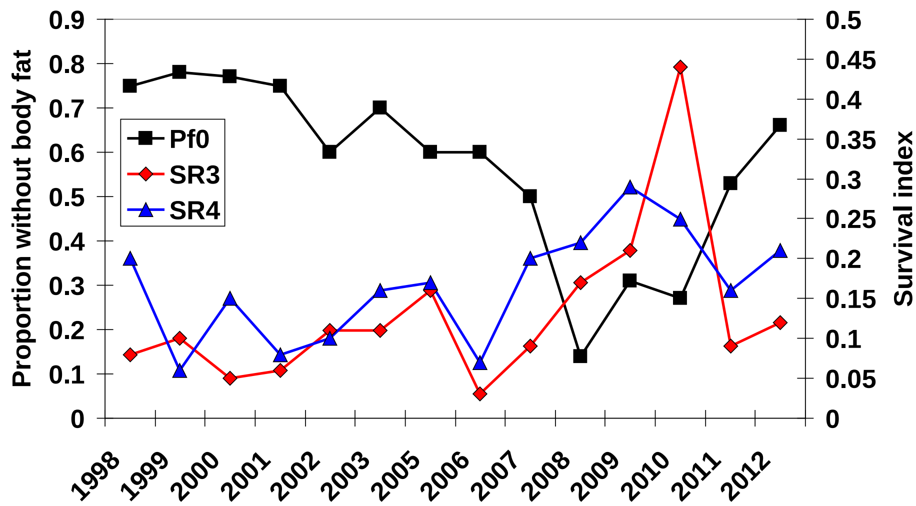

Figure 4-13. Relationship of Striped Bass survival indices for ages 3 (SR3) and 4 (SR4) and proportion of resident Striped Bass without body fat during 1998-2013. Predicted line is from the combined relationship.

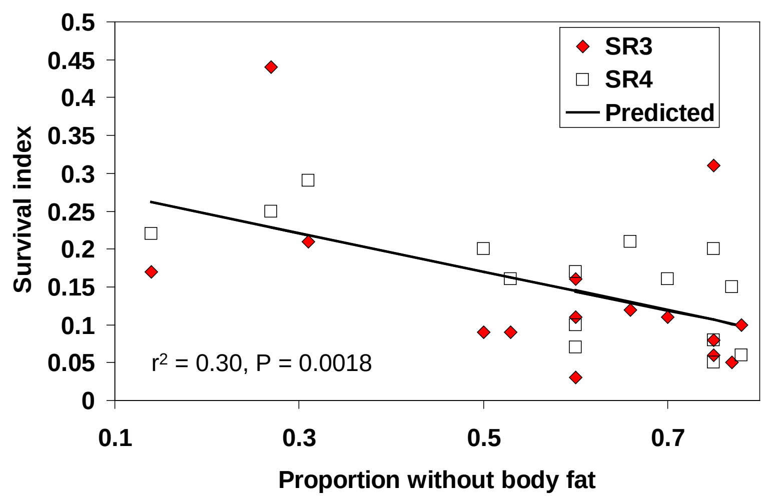

Figure 4-14. Trends in the combined forage-to-Striped Bass ratio, 1990-2013. Green squares denote years at target body fat levels and red diamonds indicate years with body fat at threshold levels.

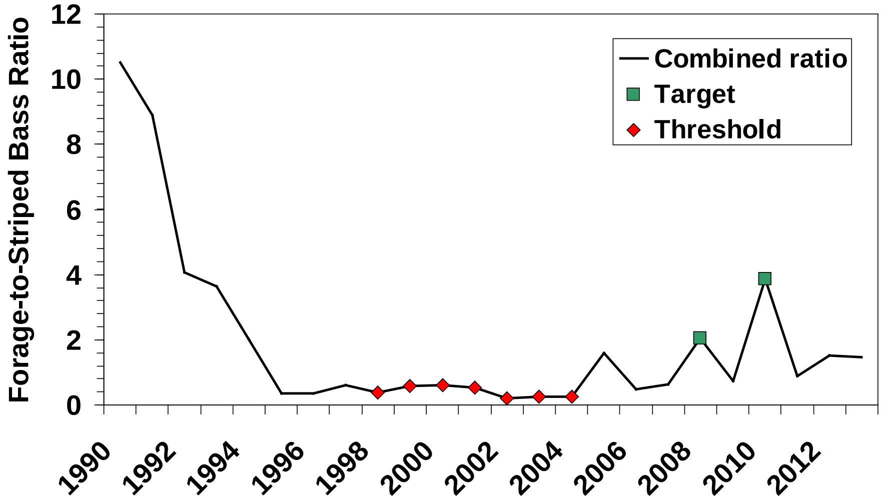

Figure 4-15. Relationship of the combined forage-to-Striped Bass ratio (FR) and the proportion of Striped Bass without visible visceral body fat during October-November, 1998-2013.

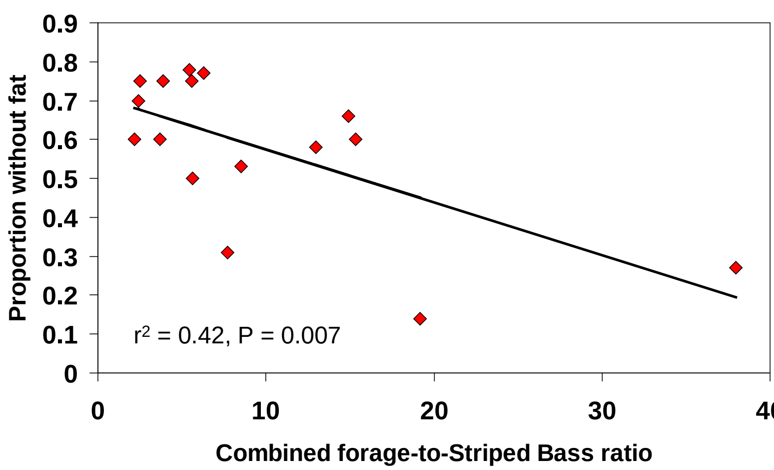

Figure 4-16. Relationship of the combined forage-to-Striped Bass ratio (FR) and grams of forage consumed per gram of sublegal Striped Bass during October-November, 2006-2012.

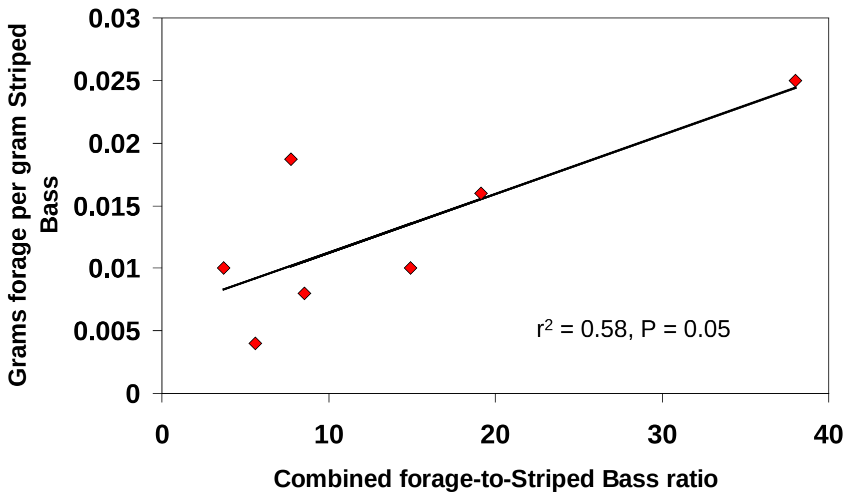

Figure 4-18. Relationship of proportion of Striped Bass without body fat and relative abundance of Striped Bass (catch per private / rental boat trip in Maryland's portion of Chesapeake Bay) during 1998-2013 with target (green dashed lines) and threshold (red dashed lines) body fat proportions.

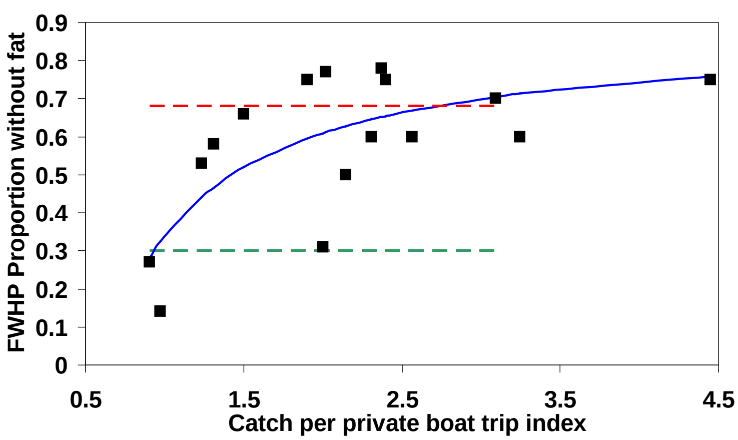

Figure 4-19. Trends in relative survival of sublegal male Striped Bass during 1998-2013 in MD's portion of Chesapeake Bay with proposed threshold.

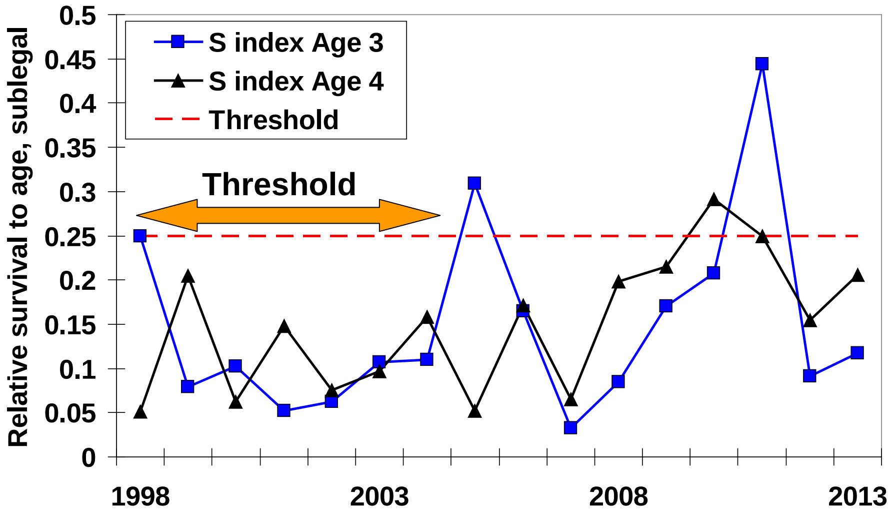

Figure 4-20. Trends in the combined forage-to-Striped Bass ratio (FR) and relative survival indices for ages 3 and 4 (SR3 and SR4), 1990-2013. Squares and diamonds indicate when nutrition reference points (Pf0; available 1998-2013) were at their targets or threshold. Dotted line indicates relative survival (SR) threshold.