A traffic light index for communicating forage balance of Atlantic Menhaden and resident Striped Bass in Maryland's portion of Chesapeake Bay

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Executive Summary

In February 2020, the Atlantic Menhaden Management Board of the Atlantic States Marine Fisheries Commission (ASMFC) accepted the results of a single species and an ecological reference point assessment for management use. These analyses considered the entirety of the coastal stock from Maine to Florida and indicated that fishing levels were below both single species and ecosystem targets (i.e., fulfilling Menhaden' forage role) and that the stock was not overfished. Despite the finding of a healthy coastal population, there has been broad concern among stakeholders that fishing levels were too high within the Chesapeake Bay (or Bay). Atlantic Menhaden (or Menhaden) in the Bay could be locally depleted, harming predators that rely on them. Because of the highly migratory nature of the Menhaden coastal stock and the lack of targeted surveys for adult Menhaden in Chesapeake Bay, it has not been possible to conduct a quantitative stock assessment for the Bay itself. In addition, an operational definition for the term 'localized depletion' does not exist. Localized depletion refers to a situation where concentrated local harvest, while not impactful to coastal stock health, has a deleterious effect on a local ecosystem.

Of particular interest to many of these concerned stakeholders is the impact of the Menhaden fishery on resident (nonmigratory) Striped Bass *Morone saxatilis* in the Bay. The coastal assessment determined that Striped Bass were most sensitive to Menhaden harvest of the array of species modeled. In order for the Bay's Menhaden fisheries to compete with resident Striped Bass in Maryland, ages 1+ Menhaden (ages harvested by the reduction fishery) need to make up a significant fraction of resident Striped Bass diet.

A traffic light index depicting Menhaden and Striped Bass balance in Maryland's portion of Chesapeake Bay based on indices can provide a cost-effective and timely alternative approach to communicate Menhaden's past and current forage status in Maryland's portion of the Bay. The traffic light approach was developed in the 1990s as part of the precautionary approach to fisheries management. It uses a three-color scheme patterned after familiar traffic lights to classify indicators as good or safe (green), intermediate or uncertain (yellow), and unacceptable or poor (red). The time-series of indicator colors are depicted in a table. The traffic light approach was originally proposed for situations where data and assessment capabilities were limited. Simplicity and communicability are of over-riding importance. Because of its flexibility, the traffic light approach can be adapted to ecosystem-based fisheries management.

Technically, the retrospective traffic light table for Bay Menhaden and Striped Bass balance would be a Traffic Light Index (TLI) because it will not be directly tied to management measures. Construction of a TLI requires selection of relevant indices, a reference period for traffic light categories, and objective criteria to delineate numerical boundaries for the traffic light categories that indicate good, poor, and between status.

We created two categories of indicators: core and supplemental. Core indices were used in the TLI. Supplemental indices were considered supporting information and were reported separately to corroborate status of core indices and provide an indication of uncertainty through their degree of agreement. There were three core indicators for Menhaden and they tracked trends in ages 1+ biomass, ages 1+ fishing mortality, and age 0 relative abundance. There were three core indicators for resident Striped Bass at sizes and ages capable of eating ages 1+ Menhaden. They tracked availability of age 0 and ages 1+ Menhaden (separately) and Striped Bass condition. Three supplemental indices were also examined for Menhaden: an additional ages 1+ biomass index, an ages 1+ relative abundance index, and an age 0 index. The supplemental ages 1+ indices had missing years. Separating indices into core and supplemental categories avoided over-weighting the TLI by multiple indicators that represented the same variable.

Potomac River Fisheries Commission (PRFC) estimates of Potomac River pound net catch per unit effort (CPUE; metric tons per net day; 1964-2021) served as the core index of ages 1+ Menhaden biomass (PRFC index). The geometric mean of catch per standard seine haul of age 0 Menhaden (GM JI) in Maryland's Juvenile Striped Bass Survey was the core index for young-of-year Menhaden (1959 -2021). Bay specific landings in year t were divided by the average of PRFC indices in year t and t+1 to estimate relative fishing mortality (Bay relative F, annual instantaneous biomass fishing mortality rate trend; 1964-2020). Bay-specific reduction landings were available through 2009 and for 2020. Bay reduction fishery caps were substituted as maximum estimates of Bay reduction landings when reduction estimates were not available. Ostensible total Bay landings were estimated by adding Bay-specific estimates of bait landings estimates to Bay reduction landings estimates when they were available or to the Bay reduction fishery cap when they were not.

Indices of ages 1+ or age 0 Menhaden availability to resident Striped Bass (AS indices; 1982-2021) were estimated as the ratios of the Menhaden PRFC index or GM JI to a Striped Bass biomass index. The latter index was based on ages likely to represent residents capable of consuming ages 1+ Menhaden (Striped Bass ages 3-6 biomass estimates from the most current ASMFC assessment). Striped Bass condition in October-November was estimated as the proportion of fish without visible body fat (P0; 1998-2021).

A 1995-2021 reference period was chosen to capture the prospect of providing adequate Menhaden forage under current prevailing management, environmental, and ecological conditions in Chesapeake Bay. Conditions considered were Striped Bass population status and management, major prey status, eutrophication, hypoxia, and the status of the Atlantic Multidecadal Oscillation (AMO), a major climate pattern that influences Menhaden year-class success in the Bay.

We used the median of the core indices during the reference period as their yellow / green boundaries (going from uncertain to good levels) and the 25th or 75th percentiles, depending on relevant direction, as the yellow / red boundaries for transition from uncertain to poor levels. Four of five metrics that relied on estimates of central tendency for status exhibited skewed distributions, supporting the use of the median and 25th percentile for boundaries of the TLI. The sixth core metric, proportion of Striped Bass without body fat (condition index), supplied its own boundaries.

We investigated important associations and relationships among core indices using correlation and regression. We used correlation to assess associations between Bay relative F and a similarly derived Atlantic coast biomass F from the ASMFC Menhaden assessment, and among core and supplemental indices. Linear and non-linear regression were used to examine relationships of core indices of age 0 and ages 1+ abundance or biomass, the core age 0 relative abundance index and the AMO, relative F and Striped Bass condition, Menhaden availability indices with condition, and availability of ages 1+ Menhaden with relative F on ages 1+ Menhaden.

The TLI exhibited a mix of core indicators that were poor (red) or uncertain (yellow) during 1995-2003. Core indicators were entirely or mostly red during 1995-1998 and became primarily yellow, with a few red core indicators during 1999-2000. The TLI turned back to predominately red during 2001-2004. A transition toward better indicator status was indicated by the TLI during 2005-2007. Red core indicators diminished, yellow core indicators became predominant, and green (good or safe) core indicators began to appear. After 2007, TLI core indicators were predominately green with some yellow indicators interspersed and a single red indicator. All indicators were green during 2018-2021. The TLI was not sensitive to removing three years from the beginning or end of the reference period to estimate boundaries; their removal did not alter the general pattern of the TLI based on the 1995-2021 reference period.

In the long term, there is potential to transform the TLI into Traffic Light Approach management triggers for the entire Chesapeake Bay. This would involve extensive work with Bay jurisdictional partners (Virginia Marine Resources Commission and PRFC), stakeholders, and the ASMFC.

There were three basic questions posed to be addressed by the TLI. These questions are paired with a brief answer about most recent conditions.

1) How are Menhaden doing in Maryland's part of the Bay? Biomass and recruitment are at a good level now given prevailing ecological and environmental conditions (green, above their reference period medians).
2) Are there enough Menhaden for resident Striped Bass? It appears that there are enough for resident Striped Bass now. Menhaden availability and Striped Bass condition indices are within their good boundaries (green).
3) Is the limitation of large-scale Menhaden fishing in the Bay having a positive effect on Menhaden and resident Striped Bass? It is plausible that there has been a positive effect since Bay relative F has declined and core indices improved concurrently with the decreases in cap on reduction harvest and other limitations to Menhaden harvest. Improvement also coincided with diminished harvest of age 0 Menhaden during fall-winter migration from the Bay that followed the closure of the Beaufort reduction plant in 2005, as well as the introduction of coastal quotas in 2013.

In general, supplemental indices and statistical analyses supported the cohesive trends of TLI metrics. Supplemental Menhaden indices indicated trends of core indices were more geographically widespread in the Bay.

Relative F in the Bay was compared to a biomass F estimated for the Atlantic coast using landings and ages 1+ biomass estimates from the coastal assessment. Time-series plots of Bay relative F and coastal F during the reference period had similar declining trends, but relative F in the Bay appeared higher on a relative basis than coastal F in 2001-2003 and lower in 2006-2008 and 2014-2019. Average coastal F for 2018-2020 was 49% lower than during 1995-1997, while average Bay relative F declined by 62%. Bay relative F and coastal F were well correlated ($r = 0.72$, $P < 0.0001$).

A supplemental index of ages 1+ Menhaden biomass for Maryland's portion of Chesapeake Bay was represented by an index of Menhaden pound net catch (MT) per net month during 1980-2020 with 1985-1989 and 1991 missing. A fishery-independent relative abundance index for ages 1+ Menhaden has been developed for the ASMFC coastal assessment from a Head-of-Bay multi-panel experimental drift gill survey for Striped Bass during spawning season (late March-May; 1985-2021). Annual Menhaden gill net CPUE for all ages pooled was adopted as a supplemental index of ages 1+ relative abundance for the TLI. The ASMFC Menhaden coastal stock assessment uses Maryland Juvenile Striped Bass Survey seine catch data to estimate a model-based index of age-0 Atlantic Menhaden abundance from Maryland's portion of the Bay (1959-2021); we considered this ASMFC index to be a supplement to the core GM JI index.

The MD and PRFC pound net CPUEs (ages 1+ Menhaden biomass indices) were well correlated during 1980-2020 ($r = 0.79$, $P < 0.0001$) and represented very similar trends over a large portion of the mesohaline Bay in Maryland during spring-fall. Head-of-Bay gill net survey CPUE (March-May in oligohaline waters) was modestly correlated with MD pound net CPUE ($r = 0.56$, $P < 0.0012$) and the PRFC index ($r = 0.45$, $P < 0.0053$). All three indices agreed that biomass and abundance were much higher during the 1980s, but the Head-of-Bay index did not support consistent elevation of Bay Menhaden relative abundance exhibited by both pound net biomass indices starting in the mid-2000s. Correlation analysis indicated that GM JI and ASMFC JI estimators of central tendency were strongly correlated during 1959-2021 ($r = 0.92$; $P < 0.0001$) and would indicate the same long-term trends. They were modestly correlated ($r = 0.51$, $P = 0.0065$) during the 1995-2021 reference period; the GM JI indicated an increase beginning after 2004 while the GLMM JI indicated a more random pattern.

We used linear regression to examine the relationship of the winter AMO and age 0 Atlantic Menhaden relative abundance (GM JI; 1959-2021) in Maryland's portion of the Bay since the AMO influenced the selection of our reference period. A regression with the $\log_e$-transformed GM JI indicated a moderate relationship with the AMO ($r^2 = 0.37$, $P < 0.0001$).

We estimated the relationship of the GM JI and the PRFC index two years later with linear regression. Inclusion of a categorical variable to represent the effect of regulatory changes after 2005 on the PRFC index improved the fit from $r^2 = 0.35$ for the simple linear regression to $R^2 = 0.59$ for the multiple linear regression ($P < 0.0001$ for both). The intercept during the more regulated period (2006-2021) was 78% higher than the less regulated period (1964-2005), indicating that recruitment from age 0 abundance to ages 1+ biomass two years later had increased concurrently with regulatory intensity.

Consumption of ages 1+Menhaden by resident Striped Bass has been described in the scientific literature as minimal from spring to fall; winter was left as the time when most age 1+ Menhaden might be consumed. Size and age composition of Menhaden consumed by resident Striped Bass in Maryland's portion of the Bay during winter has not been well described. Data for winter resident Striped Bass were collected during 2006-2015 by a citizen-science based Striped Bass diet monitoring program conducted by the Chesapeake Bay Ecological Foundation and we examined these data (N = 808) to judge whether consumption of ages 1+ Menhaden comprised a substantial fraction of winter consumption. Ages 2+ would be vulnerable to harvest the previous season as ages 1+. Menhaden categorized as ages 2+ in winter comprised 62% by weight and 16% by number of the 422 identifiable Menhaden in winter during 2006-2015. If there was a forage problem due to the fishery, it would be from depletion prior to winter.

Year-round collections during 2006-2015 provided an opportunity to examine seasonal and monthly dynamics of resident Striped Bass condition. Estimates of P0 (proportion without visible body fat) were 0.37 in summer (N = 2,921), 0.51 in fall (N = 1,866), and 0.08 in winter (N = 808). Comparisons of summer and fall estimates of P0 between the citizen science collections and MD DNR's Fish and Wildlife Program found that they were very close to one another. The ratios of Menhaden consumed per Striped Bass examined were 0.02 in summer, 0.64 in fall, and 0.48 in winter and on a Menhaden weight basis they were 21.0 gm in fall, 28.4 gm in winter, and 1.7 gm in summer. Sample sizes were sufficient for precise monthly estimates of P0 during June-February when pooled across years (CV range of 2-26%). Estimates of P0 were near 0.20 in June-July and then increased (i.e., condition worsened) to 0.51 in August and 0.68 (near the potential starvation threshold) during September-October. Estimated P0 dropped to 0.28 in November, 0.15 in December, and reached a nadir of 0.05-0.06 during January-February. Condition of Striped Bass in fall was strongly related to condition in the preceding summer of the same year (Weibull function, approximate $R^2 = 0.75$, $P < 0.0001$) and in the fall of the previous year (linear regression, $r^2 = 0.70$, $P < 0.0001$).

Consumption of ages 1+ Menhaden by resident (mostly male) Striped Bass is likely important because much of the latter's annual growth and gonadal development occurs in the fall-winter. Resident Striped Bass invested heavily in spring spawning that may be followed in late spring through early fall by low forage availability, depleted energy reserves, and increased risk of starvation, predation, and susceptibility to disease until successful feeding on Menhaden resumed.

We used regression analysis to examine the relationship of Bay relative F in year t and P0 in fall of year t+1 during 1998-2020. This lag was necessary since harvest had to impact P0 in the preceding winter. The scatter plot of P0 and Bay relative F suggested a sharp increase in P0 (worsening condition) as Bay relative F rose from 0.75 to 1.32 (scale is arbitrary) and an asymptote for P0 at Bay relative F higher than 1.32. A piecewise linear model with an ascending limb followed by an asymptote fit the estimates of P0 and Bay relative F well ($R^2 = 0.68$, $P < 0.0001$).

We used linear regression to investigate the strength of the relationships of the Menhaden availability indices (AS) for either ages 1+ or age 0 Menhaden with P0 or Bay relative F with AS of ages 1+ Menhaden. The AS for age 0 or ages 1+ Menhaden and P0 time-series trended in opposite directions, indicating that generally condition was improving as Menhaden availability went up. A linear regression indicated a modest relationship of availability of ages 1+ Menhaden and condition of resident Striped Bass in the following fall ($r^2 = 0.27$, $P = 0.019$). Relative availability of ages 1+ Menhaden declined with Bay relative F ($r^2 = 0.50$, $P < 0.00016$). A linear regression indicated availability of age 0 Menhaden exerted a modest influence on resident Striped Bass condition in fall ($r^2 = 0.32$, $P = 0.004$). The moderate amounts of variation accounted for by the two AS indices with P0 indicated potential for other prey (fish and invertebrates), competitors, and environmental conditions to influence P0 as well.

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Introduction

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

Purpose of this document - This document provides technical documentation for a communication tool describing Atlantic Menhaden *Brevoortia tyrannus* and Striped Bass *Morone saxatilis* forage balance in Maryland's portion of Chesapeake Bay developed by the Maryland Department of Natural Resources Fishing and Boating Services during 2022-2024. It covers status through 2021. Updates will be available separately.

Need - Atlantic Menhaden are important forage for many fish, bird and mammalian predators along the Atlantic coast and also support large fisheries (Munroe and Smith 2000; Buchheister et al. 2017; Chagaris et al. 2020; Drew et al. 2021; Anstead et al. 2021). Atlantic Menhaden (or Menhaden) have supported the largest commercial fishery by weight on the Atlantic coast for over a century - a reduction fishery that harvests Menhaden with purse seines for processing into fish meal and oil (Anstead et al. 2021). The reduction fishery is centered in Virginia's portion of Chesapeake Bay and its catches there surpassed all other areas of the Atlantic coast in the past (Smith 1999; Munroe and Smith 2000; Anstead et al. 2021).

Because Menhaden undergo extensive migrations and are mostly harvested from state waters, their management is coordinated through the Atlantic States Marine Fisheries Commission (ASMFC), a deliberative body of Atlantic coastal states' fisheries management agencies (Munroe and Smith 2000; SEDAR 2015; https://asmfc.org/species/atlantic-menhaden). The forage role of Atlantic Menhaden was recognized by ASMFC as early as 1981, but it was not raised as a multispecies management issue until the 1998 Atlantic Menhaden stock assessment (ASMFC 1999; Anstead et al. 2021). Single-species management prevailed until February 2020, when the Atlantic Menhaden Management Board of ASMFC accepted the results of both a single species stock assessment and an ecological (forage) reference point assessment for Atlantic coast management (ASMFC 2020a; Chagaris et al. 2020; Drew et al. 2021; Anstead et al. 2021). These analyses indicated that fishing levels of the entire coastal stock from Maine to Florida were below both single species and forage targets and that the stock was not overfished (ASMFC 2020a; Chagaris et al. 2020; Drew et al. 2021; Anstead et al. 2021). In August 2020, the ASMFC formally adopted an ecological modeling framework to set coastal reference points and harvest limits for Atlantic Menhaden that considers their role as a forage fish (Anstead et al. 2021).

Despite the finding of a healthy coastal stock, there are concerns among stakeholders that harvest from the reduction fishery is too high within the Chesapeake Bay (or Bay), depleting Menhaden there and harming predators that rely on them, particularly Striped Bass (Uphoff 2003a, Maryland Sea Grant 2009). The phrase "localized depletion" has been used to describe concerns about Menhaden depletion in the Bay, but the definition was not quantified (Maryland Sea Grant 2011). The issue of localized depletion in Chesapeake Bay first appeared in the 1998 Atlantic Menhaden stock assessment (ASMFC 1999). The issue led to an *ad hoc* 109,020 MT cap that was imposed on the reduction fishery in 2006 by ASMFC as a precautionary measure to address Bay ecosystem concerns (Maryland Sea Grant 2011; Anstead et al. 2021). The cap was reduced to 87,216 MT in 2013 and to 51,000 MT in 2020 (Anstead et al. 2021).

Striped Bass is the main predator of concern for the ASMFC's ecological (forage) reference points for Atlantic Menhaden along the Atlantic coast due to their high sensitivity to Menhaden population size in the coastal Ecopath with Ecosim model used to estimate forage reference points (ASMFC 2020a; Chagaris et al. 2020; Drew et al. 2021). The coastal assessment determined that Striped Bass were most sensitive to Menhaden harvest of the array of species modeled (ASMFC 2020a; Chagaris et al. 2020). A large contingent of Chesapeake Bay Striped Bass that do not participate in the Atlantic coast migration (hereafter, resident Striped Bass) constitute a year-round population of predators in the Bay that provide Chesapeake Bay's major saltwater recreational fishery and an important commercial fishery (Maryland Sea Grant 2009). An Ecopath with Ecosim model developed for Chesapeake Bay indicated that Striped Bass in the Bay were moderately sensitive to changes in Menhaden fishing pressure during 1950-2002 (Christensen et al. 2009).

Maryland DNR has an interest in using available data collected from ongoing surveys and fishery to address stakeholder concerns around localized depletion of Menhaden and their balance with Striped Bass due to large scale fishing in Chesapeake Bay. Determining Menhaden and resident Striped Bass stock status for the Bay or Maryland's portion of it through formal stock assessment is problematic because of their migratory nature and absence of long-term targeted surveys.

Indicators based on monitoring, such as forage indices, prey-predator ratios, condition indices, and prey abundance in diet samples have been suggested as a basis for ecosystem-based assessment and management of Striped Bass and Menhaden in Chesapeake Bay (Maryland Sea Grant 2009; 2011). Indicators are widely used for environmental reporting, research, and management support and they may be able to detect rapid shifts in ecosystem state that models cannot (Rice 2003; Jennings 2005; Dettmers et al. 2012; Fogarty 2014). The Maryland Department of Natural Resources (MD DNR) has used a modest index-based approach to communicate forage status for resident Striped Bass in Maryland's portion of Chesapeake Bay since 2014 (Uphoff et al. 2022). It provides a narrative based on a suite of indicators of major forage relative abundance (in surveys and fall diet sampling) and Striped Bass well-being developed from existing MD DNR sampling programs. Additional objectives are low cost and tractability for available staff. Given the high cost of implementing new programs, this approach uses information from existing sampling programs and indices. This index-based approach is not a comprehensive ecosystem assessment nor is it tied directly to management. Young-of-year (YOY or age 0) Menhaden are among the major Striped Bass forage items monitored, along with Bay Anchovy *Anchoa mitchelli*, Spot *Leiostomus xanthurus*, Blue Crab *Callinectes sapidus*, and benthic invertebrates (multiple taxa; Uphoff et al. 2022). However, it does not include ages 1+ Menhaden that are subject to fishing and the focus of concerns about localized depletion.

Maryland's fisheries managers and stakeholders want to know whether there are enough Menhaden to support resident Striped Bass in Maryland's portion of the Bay. There are three basic questions to be addressed:

1) How are Menhaden doing in Maryland's part of the Bay?
2) Are there enough Menhaden for Striped Bass there?
3) Is limiting large-scale fishing for Menhaden in the Bay having a positive effect on Menhaden and resident Striped Bass?

Striped Bass in Maryland's portion of Chesapeake Bay - Striped bass are anadromous, long-lived, late maturing, and highly fecund; they undergo complex migrations and support important Atlantic coast and Chesapeake Bay fisheries (Boreman and Lewis 1987; Rago and Goodyear 1987; Rago 1992; Dorazio et al. 1994; Richards and Rago 1999; Secor and Piccoli 2007; Maryland Sea Grant 2009; NEFSC 2019; Secor et al. 2020). Production from Chesapeake Bay spawning areas has been estimated to contribute up to 90% of landings along the entire Atlantic Coast and its fisheries are driven by strong, environmentally influenced year-classes (Ulanowicz and Polgar 1980; Richards and Rago 1999; Maryland Sea Grant 2009; Uphoff 2023). Striped Bass, fueled by a series of strong year-classes in Chesapeake Bay, were abundant in the 1960s and early 1970s, then declined as recruitment faltered and fishing mortality rates increased (Richards and Rago 1999; Maryland Sea Grant 2009; Uphoff 2023). Moratoria were imposed in several Mid-Atlantic States (including foremost Maryland) in the mid-to-late 1980s and conservative regulations were put in place elsewhere (Uphoff 1997; Richards and Rago 1999). Recovery of Atlantic coast Striped Bass was declared in 1995 after rapid Chesapeake Bay stock growth (Richards and Rago 1999; Maryland Sea Grant 2009; ASMFC 2021; Uphoff 2023). Management since recovery has been based on much lower fishing mortality and much higher size limits than were in place into the early 1980s (Hollis et al. 1967; Richards and Rago 1999; Maryland Sea Grant 2009; ASMFC 2021). These more conservative regulations increased demand for Menhaden by Striped Bass in the Bay (Uphoff 2003a).

The Bay has two main migratory contingents of Striped Bass: coastal migrants and residents (Maryland Sea Grant 2009). The migratory contingent consists mostly of females that leave the Bay for all (mostly immature females and some mature females that skip spawning) or most of the year (mature females; Secor 2008; Maryland Sea Grant 2009). Females that primarily use the Chesapeake Bay spawning areas migrate up the coast for most of the year are ages 4 and older (Setzler et al. 1980; Kohlenstein 1981; Dorazio et al. 1994; Secor and Piccoli 2007; Maryland Sea Grant 2009; Secor et al. 2020). Migratory Striped Bass ascend the Bay in winter – early spring, but their distribution in winter ranges between New Jersey and Cape Hatteras and does not center in Chesapeake Bay (Maryland Sea Grant 2009; Waldman et al 2012). Mature female Striped Bass spawn in Bay tributaries in April-May and quickly leave the Bay afterward (Maryland Sea Grant 2009).

Resident Striped Bass are younger (generally up to 8 years old), consisting of mostly males with some immature females and very few mature females (Kohlenstein 1981; Maryland Sea Grant 2009). Most commercially harvested resident Striped Bass (457 mm TL minimum) are males, ages 4-6 in summer (Maryland Sea Grant 2009; Horne 2021a). Commercially harvested fish from the Bay in winter were typically ages 4-6 until about 2015 when ages 7 and 8 began to contribute a larger share and ages 4 and 5 less (Horne 2021b).

After spawning, resident Striped Bass move downstream from the spawning areas into the Bay and its tributaries; many move to between Poole's Island and Tilghman Island (Figure 1; Maryland Sea Grant 2009). Schools are concentrated along shoal areas during July-October (Lippson 1973). With the approach of cold weather, these aggregations move downriver or down-Bay towards deeper water and in February-March they disperse toward their respective spawning rivers (Lippson 1973).

Ages 1+ Menhaden and resident Striped Bass – In order for the Bay's Menhaden fisheries to compete with resident Striped Bass both must seek the same sized prey (ages 1+ Menhaden). Uphoff and Sharov (2018) examined published fish prey length versus predator length plots of Striped Bass to select a minimum size-age class capable of eating age 1 Atlantic Menhaden. Three-year-old (~370-430 mm TL) Striped Bass were the first age-class capable of consuming age-1 Atlantic Menhaden (Uphoff and Sharov 2018), although this ability would start low and increase with Striped Bass length; it was not knife-edged.

Bioenergetics modeling indicated prey fish availability was low in the Bay during spring and early summer for ages 3+ Striped Bass during 1990-1992 and 1998-2000 (Hartman and Brandt 1995; Overton et al. 2009). Hartman and Brandt (1995) expected that yearling and older Menhaden would contribute more to diets in May-June, 1990-1992, if they were available. They attributed that shortfall to interception of age 0 Menhaden by the reduction fishery as they emigrated from the Bay in the previous late fall and winter. Striped Bass in the Bay fed heavily on Menhaden during fall and winter (Hartman and Brandt 1995; Griffin and Margraf 2003; Walter et al. 2003; Overton et al. 2009; Overton et al. 2015; Buchheister and Houde 2016; Uphoff et al. 2022). Age 0 Menhaden comprised nearly all Menhaden detected in resident Striped Bass diets in October-November during 2006-2020 (2,922 stomachs examined; Uphoff et al. 2022).

Chesapeake Bay Atlantic Menhaden Harvest - In the 1950s, 25 Atlantic Menhaden factories operated along the coast, but today only one reduction plant in Virginia remains in operation (Anstead et al. 2021). Atlantic Menhaden are also harvested coastwide as bait by mixed gear fisheries. While coastal reduction landings have decreased in recent years, bait landings have increased and comprise about 25% of total coastwide landings (Anstead et al. 2021).

Within the Bay, the reduction fishery operates in Virginia mainstem waters during May – November (Maryland Sea Grant 2011; Figure 1). Purse seines are not allowed in Maryland, Potomac River, or in Virginia's Bay tributaries (for the most part; Smith 1999; Maryland Sea Grant 2011). A cap was imposed on the Bay reduction fishery harvest in 2006 and has remained in place (see below).

The reduction fishery has harvested all ages of Menhaden, including age 0, but ages 1-3 account for most harvest by number (SEDAR 2020a). Age 2 Menhaden typically dominated the catch. Age 0 Menhaden (known as "peanuts") were sometimes targeted for harvest in fall and winter along the coasts of Virginia and North Carolina as they migrated southward from the Bay. Their harvest subsided greatly in the mid-2000s and now appears incidental (SEDAR 2020a). This directed fishery on age 0 Menhaden ceased concurrently with the closure of the plant in Beaufort, NC, in 2005 (Anstead et al. 2021).

Atlantic coast bait landings of Atlantic Menhaden have been dominated by catches in Chesapeake Bay and New Jersey (Maryland Sea Grant 2011; SEDAR 2020a; Anstead et al. 2021). The bait fishery harvests ages 1+ Menhaden; ages 2 and 3 are usually predominant by number (SEDAR 2020a). Bait landings of Menhaden in Chesapeake Bay are primarily drawn from pound nets in Maryland and Potomac River, while Virginia bait landings are dominated by small purse seines (snapper rigs; Maryland Sea Grant 2011; SEDAR 2020a). Pound net monitoring in Maryland's portion of the Bay since 2005 has detected ages 0-7 with ages 1-3 predominant (Rickabaugh and Messer 2021).

The traffic light assessment method – A traffic light index (TLI) for Menhaden and Striped Bass balance in Maryland's portion of the Bay (including Potomac River) may be a reasonable, timely alternative for communicating status in the absence of model-based stock assessments and aerial surveys. A traffic light index uses a three-color scheme, patterned after familiar traffic lights, to classify multiple indicators as good or safe (green), intermediate or uncertain (yellow), and unacceptable or poor (red; Caddy and McGarvey 1996; Caddy 1998; 2002; 2015;

Halliday et al. 2001). This retrospective table of traffic lights would communicate a historical perspective on Menhaden's forage status in Maryland's portion of the Bay and would not be directly tied to management measures. This TLI can be developed at low cost using available, ongoing catch and survey indices to depict Menhaden status in Maryland's portion of the Bay. The TLI can provide stakeholders with a view of relative status based on indicators and criteria that Maryland DNR managers consult and hopefully provide an understandable framework for communication (Halliday et al. 2001).

A strength of the TLI is its ability to account for a broad spectrum of information, qualitative as well as quantitative, which might be relevant to an issue (Halliday et al. 2001). Simplicity and communicability are of over-riding importance (Halliday et al. 2001). The approach allows a system of indicators to be developed that are not constrained by narrower model-driven approaches to sampling and data collecting (Caddy 2015). This method provides a way of looking at high-dimensional data sets without premature modeling or resorting to advanced statistical techniques (Caddy 2015).

Construction of an Atlantic Menhaden TLI for Maryland's portion of Chesapeake Bay requires selection of relevant indicators. These can be time-series from stock assessments, fishery independent surveys, fisheries statistics, and socio-economic surveys (Caddy 2002; 2015). The next step is to choose a reference period to develop the traffic light categories. Finally, objective criteria are applied to each indicator for the reference period to delineate numerical boundaries that indicate good, poor, and between status (Caddy 2002; 2015).

We chose the basic or "strict" three light stoplight as opposed to the fuzzy (blended color) approach for the TLI (Halliday et al. 2001). We created two categories of indicators: core and supplemental. Core indices were used in the TLI, while supplemental indices were reported separately as supporting information that could corroborate status of core indices and provide an indication of their uncertainty through their degree of agreement. There are six core indicators for Maryland's portion of the Bay, three for Menhaden that track trends in ages 1+ biomass, ages 1+ fishing mortality, and age 0 relative abundance, and three for resident Striped Bass that track relative availability of age 0 and ages 1+ Menhaden (separately) and Striped Bass condition. Three supplemental indices were also examined for Menhaden: an additional ages 1+ biomass index, an ages 1+ relative abundance index, and an age 0 index. The supplemental ages 1+ indices had missing years in their time-series. Separating indices into core and supplemental categories avoided over-weighting the TLI by multiple indicators that represented the same variable.

Data series of different lengths may bias a TLI (Halliday 2001) and we used a 1995-2021 reference period shorter than what was available for most metrics. Red and green boundaries were determined from this reference period for five core indicators; the condition indicator had its own boundaries. If a time-series for a core metric was longer than the reference period, the entire time-series was included as supplemental information. The reference period will be updated to include additional years as necessary rather than annually. We conducted a sensitivity analysis of the TLI to changes in reference period time-series.

The 1995-2021 reference period attempted to capture the prospect of providing adequate Menhaden forage under current prevailing environmental and ecological conditions rather than relying on an assumption implicit with the longer time-series that conditions might be reversed to an earlier preferred state. This latter expectation is not well supported by scientific literature. Periods of high and low productivity that drive stock dynamics due to environmental forcing are far more common for fish stocks than stable environmental conditions (Gilbert 1997; Jiao 2009; Vert-pre et al. 2013; Szuwalski et al. 2015). Shifts in ecosystem status often represent shifts to different persistent states rather than steady, reversable change (Steele and Henderson 1984; Duarte et al. 2009; Vert-pre et al. 2013; Szuwalski et al. 2015; Cloern et al. 2016). The terms "regime shift" or "recruitment states" have been used to suggest jumps between alternative population states that are nonlinear, causally connected, and linked to other changes in an ecosystem (Steele and Henderson 1984; Steele 1996; Gilbert 1997; Jiao 2009; Duarte et al. 2009; Kemp et al. 2009). The concept implies that different regimes have inherent stability, so that significant forcing is required to flip the system into alternative states (Steele 1996). Fishing, predation, climate, and eutrophication are among forcing mechanisms (Walters 1987; Jiao 2009; Duarte et al. 2009; Kemp et al. 2009; Szuwalski et al. 2015). Dynamics of predation, competition, environmental regime shifts, and habitat alteration or deterioration may take over once overharvesting has been controlled (Link 2002).

Concern emerged about an imbalance of high Striped Bass population size and its prey-base shortly after recovery from severe depletion was declared in 1995 (Hartman 2003; Hartman and Margraf 2003; Uphoff 2003a; Savoy and Crecco 2004; Heimbuch 2008; Davis et al. 2012; Overton et al. 2009; 2015; Uphoff and Sharov 2018). An equilibrium Atlantic Menhaden consumption per Striped Bass recruit model indicated that conservative regulation changes designed for increased age-at-entry and lower fishing mortality imposed after stock recovery could have increased demand for Menhaden by 2- to 5-times (Uphoff 2003a). Major declines in abundance of important prey (Bay Anchovy, Atlantic Menhaden, and Spot in Maryland's portion of Chesapeake Bay coincided with Striped Bass recovery (Uphoff 2003a; Overton et al. 2015; Uphoff et al. 2022). Reports of Striped Bass in poor condition and with ulcerative lesions increased in Chesapeake Bay shortly after recovery was declared in 1995; linkage of these phenomena and poor feeding success on Atlantic Menhaden and other prey was considered plausible (Overton et al. 2003; Uphoff 2003a; Gauthier et al. 2008; Overton et al. 2015; Uphoff and Sharov 2018). Mycobacteriosis, a chronic wasting disease, became widespread in Chesapeake Bay in the late 1990s and was concurrent with lesions and poor condition (Overton et al. 2003; Jiang et al. 2007; Gauthier et al. 2008; Jacobs et al. 2009b). Challenge experiments with Striped Bass linked nutrition with disease progression and severity, and reduced survival (Jacobs et al. 2009a). Tagging models indicated that annual instantaneous natural mortality rates (M) of 457-711 mm TL sized (i.e., resident) Striped Bass in Chesapeake Bay increased substantially during the mid-1990s while annual instantaneous fishing mortality rates (F) remained low (Kahn and Crecco 2006; Jiang et al. 2007; NEFSC 2013; NEFSC 2019). Prevalence of mycobacteriosis and magnitude of M appeared to be lower outside Chesapeake Bay (Matsche et al. 2010; NEFSC 2019; Secor et al. 2020), but abundance, condition, and M of the coastal migratory contingent appears linked to ages 1+ Atlantic Menhaden (Buchheister et al. 2017; Uphoff and Sharov 2018; SEDAR 2020b; Chagaris et al. 2020).

While top-down control of forage in the Bay is suggested by opposing trends of major forage and Striped Bass, bottom-up processes may also be in play. Widespread hypoxia in summer has been a major habitat feature in Chesapeake Bay since the mid-1980s (Hagy et al. 2004; Kemp et al. 2005; Maryland Sea Grant 2009; Ni and Li 2023). Hypoxia has increased in extent and duration in bottom waters as a result of eutrophication and it is the target of substantial nutrient management efforts (Breitburg 2002; Hagy et al. 2004; Kemp et al. 2005; Maryland Sea Grant 2009; Ni and Li 2023). Hypoxic bottom waters expanded southward from a small area at the upstream limit of the mesohaline Bay in the late 1950s to encompass the entire mesohaline Bay and a portion of the polyhaline Bay in Virginia by the early 1990s (Kemp et al. 2005). Water quality monitoring indicates that Chesapeake Bay has been slow to respond to nutrient and sediment reductions (Scientific and Technical Advisory Committee 2023). The Bay faces permanent and ongoing changes in land use, climate change, population growth, and economic development that challenge notions of restoration to historical conditions. Additional funding of existing implementation efforts is unlikely to produce the intended nutrient reduction outcomes that will likely require development and adoption of new implementation programs and tools (Scientific and Technical Advisory Committee 2023).

Houde et al. (2016) found Chlorophyll a and variables associated with freshwater flow (Secchi disk depth and zooplankton assemblages) were correlated with age 0 Menhaden abundance in the Bay. Variations in river flows to the Chesapeake Bay set up stratification, drive estuarine circulation, and cause fluctuations in inputs of freshwater, sediments, and nutrients that greatly influence hypoxia (Hagy et al. 2004; Kemp et al. 2005; Maryland Sea Grant 2009). Chlorophyll a concentrations in the Bay increased between the 1950s and 1980s and remained high and unchanged afterward (Kemp et al. 2005). The phytoplankton community shifted from larger to smaller cells; diatoms decreased and dinoflagellates, cyanobacteria, and small flagellates increased (Kemp et al. 2005). Some of these smaller phytoplankton (cyanobacteria) were too small to be ingested by Menhaden and passed through their guts intact (Friedland et al. 2005).

Late-stage Menhaden larvae that enter the Bay in winter feed primarily on zooplankton, mostly crustaceans such as copepods (Maryland Sea Grant 2011). After metamorphosis, young-of-year Menhaden feed primarily on phytoplankton. Zooplankton become increasingly important for ages 1+ (Maryland Sea Grant 2011).

A long-term decline of Bay Anchovy in Maryland's portion of Chesapeake Bay was linked to declining abundance of its common calanoid copepod *Acartia tonsa* that, in turn, was linked to rising long-term water temperatures, eutrophication, and hypoxia (Kimmel et al. 2012; Roman et al. 2019; Slater et al. 2020). Copepod mortality was higher under hypoxic conditions and implied a direct linkage between low dissolved oxygen and reduced copepod abundances (Slater et al. 2020).

ASMFC (1999) noted that Chesapeake Bay Atlantic Menhaden recruitment indices were autocorrelated, indicating recruitment may be affected by non-independent decadal scale changes (Walters and Martell 2004). Buchheister et al. (2016) suggested that broad-scale climate forcing was an important controller of recruitment dynamics. The Atlantic Multidecadal Oscillation (AMO) climate pattern was one of the best predictors of relative abundance of age 0 Menhaden in the Chesapeake Bay and southern New England regions during 1959-2013. The AMO exhibited a minimum in the mid-1970s, transitioned into a positive phase in the 1990s, and reached a maximum in the 2000s (Buchheister et al. 2016). Our reference period corresponded to the recent positive phase. Three AMO-related hypotheses for potential mechanisms affecting year-class success included changes in oceanographic processes and larval transport caused by AMO-associated changes in wind and water circulation patterns; temperature-mediated changes in distribution, timing, and location of spawning that influence larval transport to estuaries; and effects of temperature and other climate-related factors on habitat quality or system production that influenced survival of larval or juvenile menhaden (Buchheister et al. 2016).

Testing for associations and relationships among metrics - Statistical analyses can provide insight into important ecological processes such as predation, competition, environmental forcing, and forage availability and act as a bridge between assessing a stock in isolation from its environment in a single species assessment and more complex process-based multispecies and ecological models (Bax 1998; Sainsbury 1998; Whipple et al. 2000). We investigated important associations and relationships among core indices using correlation and regression. We assessed associations among core and supplemental indices, and relationships of core indices of age 0 and ages 1+ abundance or biomass of Menhaden, the core age 0 Menhaden relative abundance index and the AMO, an index of fishing mortality of Menhaden in the Bay and Striped Bass condition, indices of Menhaden availability to Striped Bass (availability indices) with Striped Bass condition, and availability of ages 1+ Menhaden with the index of Bay fishing mortality on Menhaden.

Methods

Winter consumption of Ages 1+ Menhaden by resident Striped Bass – Since consumption of ages 1+Menhaden appeared minimal from spring to fall (Hartman and Brandt 1995; Overton et al. 2009; Uphoff et al. 2022), winter seemed to be when most age 1+ Menhaden would be consumed. Size and age composition of Menhaden consumed by resident Striped Bass in Maryland's portion of the Bay during winter has not been described in enough detail to address the plausibility that Menhaden large enough to be harvested were important for resident Striped Bass well-being. Data for winter resident Striped Bass were collected during 2006-2015 for DNR by a citizen-science based Striped Bass diet monitoring program conducted by the Chesapeake Bay Ecological Foundation (CBEF) under a MD DNR scientific collector's permit (Uphoff et al. 2022). We examined these data to judge whether consumption of ages 1+ Menhaden was important (i.e., they comprised a substantial fraction of winter consumption).

Chesapeake Bay Ecological Foundation's Winter diet samples came from Striped Bass caught by the gill net fishery. Legal minimum size was 457 mm (18 inches) TL and there was no maximum. Striped Bass 457 - 711 mm TL fall within TL boundaries used for the ASMFC tag-based assessment of F and M of resident Striped Bass in the Bay (NEFSC 2019) and we used 711 mm (28 inches) TL as our upper boundary for resident fish in winter. Walter and Austin (2003) used a size range of 458-710 mm TL to categorize large resident Striped Bass in a 1997-1998 Bay diet study. Striped Bass larger than the 711 mm TL upper bound were fairly abundant in CBEF samples but were considered migratory. We felt that migratory Striped Bass were handled in the coastal forage reference points (Chagaris et al. 2020) and focused on resident Striped Bass.

Striped Bass in winter were mostly sampled at a fish cleaning business that drew fish from mid-Bay. Locations reported for these fish were predominately from the mainstem Chesapeake Bay between Kent Island and Patuxent River, Eastern Bay, and Choptank River (Figure 1). More detail on the CBEF sampling program can be found in Uphoff et al. (2014; 2015).

Lengths of Menhaden collected by CBEF during winter 2006-2015 were used to determine the prevalence of age 0, ages 1+, and ages 2+ Menhaden in the resident Striped Bass diet based on total lengths of intact Menhaden in Striped Bass guts. Ages 2+ in winter would have been vulnerable to harvest during the previous fishing season while younger Menhaden would not. We used a minimum TL of 159 mm as an age 0 cut-off for Menhaden; this was estimated by adjusting the 150 mm FL cut-off for August 16-November 30 used in ASMFC stock assessments (ASMFC Atlantic Menhaden Technical Committee 2010). We examined a scatter plot of lengths of Menhaden consumed (TL) to resident Striped Bass (TL) that consumed them and classified Menhaden as age 0 or 1+ based on the 159 mm TL cut-off. We constructed a length-frequency histogram of Menhaden consumed and used it to determine a cut-off for ages 2+. We examined this histogram of the total lengths of Menhaden consumed (25 mm increments), concentrating on the formation of a nadir and smaller peak at larger sizes beyond the most abundant portion of the distribution. The nadir prior to the smaller peak was interpreted as a break between Menhaden less than 2 years old (not subject to fishing the previous season as age 0s) and ages 2+ (vulnerable to harvest the previous season as ages 1+).

Based on the age 0, age 1+, and age 2+ cut-offs, we estimated the fraction of total weight of Menhaden consumed by resident Striped Bass that each age category represented.

Estimating Chesapeake Bay Menhaden Harvest - We constructed ostensible landings from Bay specific estimates in past stock assessments, reports on the Bay reduction fishery cap, the reduction fishery Bay caps, and estimates of Bay bait landings. Estimates of Atlantic Menhaden harvest from the Bay proper by the reduction fishery during 1955-2001 were taken from Vaughan et al. (2002; their Appendix B, Table B1). Bay reduction fishery landings estimates continued to be reported routinely through 2004. With the closure of Beaufort Fisheries in 2005, the fishery contracted to a single company; reduction landings became protected by confidentiality rules and Chesapeake Bay specific estimates were no longer reported routinely. Bay reduction landings estimates for 2005 and 2006 were available from Addendum 2 reporting on the Bay reduction fishery cap during management board meetings. Bay reduction landings for 2007 and 2008 were reported for the Chesapeake Bay ecosystem-based fisheries management plan. Bay-specific reduction fishery landings were no longer reported after 2009, with the exception of a cap overage reported in newspapers during 2020. Bay reduction fishery caps were substituted as maximum estimates of Bay reduction landings after 2009, with the exception of 2020.

Bait landings for 1964-2002 estimated for a Chesapeake Bay Atlantic Menhaden biomass dynamic model (Uphoff 2003b) were available. They were derived from gear specific landings available for Chesapeake Bay through the NMFS website at that time. Total landings for Chesapeake Bay were reported and purse seine harvest was subtracted from the total. Snapper rigs may have been included in purse seine landings, but the vast majority of purse seine landings would have been from the reduction fishery (Uphoff 2003b). Bait landings for 1985-2020 (Maryland + Virginia + PRFC) were those provided to the ASMFC Plan Development Team (PDT). We assumed that these landings were from Chesapeake Bay, although a small portion would be coastal catch. A 1964-2020 time-series of bait landings was formed by combining the two time-series: 1964-1984 bait landings from Uphoff (2003b) and the PDT bait landings for the remainder of the time-series. We did not include recreational harvest or estimates of unreported harvest; these were considered very minor losses that would not change the findings of the TLI. Ostensible Chesapeake Bay total landings were estimated by adding bait landings estimates to Bay reduction landings estimates when they were available or to the Bay reduction fishery cap when they were not.

Statistical analyses of indices - Correlation and linear regression were the primary means of analyzing associations and relationships among indices, but the potential for nonlinear relationships was not ignored. Cause and effect were not implied with associations (correlation analysis) but were for relationships (regression analyses).

For all analyses, scatter plots were examined for the need for data transformations and to identify candidate models. A general description of relationships we considered and equations used to describe them follows, while more specific applications are described as needed.

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

$$(1) \qquad Y = (m \cdot X) + b;$$

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

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

where n is the coefficient for variable Z and other parameters are as described previously (Freund and Littel 2006). We did not consider multiple regression models with more than two variables.

Potential symmetric curvilinear, U-shaped, or dome-shaped relationships were examined with quadratic models (Freund and Littell 2006):

$$(3) \qquad Y = (m \cdot X) + (n \cdot X^2) + b;$$

where m and n are coefficients and b is the Y-intercept (Freund and Littell 2006).

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

$$(4) \qquad Y = a \bullet (X)^b;$$

where a is a scaling coefficient and b is a shape parameter (Prager et al. 1989). The symmetric logistic growth function described growth to an asymptote through the equation:

$$(5) \qquad Y = b / ((1 + ((b - c) / c) \bullet (\exp(-a \bullet X)));$$

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

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

$$(6) \qquad Y = K \bullet \{1 - \exp[-(Y / S)^b]\};$$

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

Level of significance was reported, but potential management and biological significance took precedence over significance at $P < 0.05$ (Anderson et al. 2000; Smith 2020). We classified correlations as strong, based on $r > 0.80$ (Ricker 1975); weak correlations were indicated by $r < 0.50$; and moderate correlations fell in between. Relationships indicated by regressions were considered strong at $r^2 > 0.64$; weak relationships were indicated by $r^2 < 0.25$; and moderate relationships fell in between. Moderate to strong correlations and relationships were considered of interest to management. Confidence intervals (95% CIs were standard output) of the model parameters for each indicator species were estimated to examine whether parameters were different from 0 (Freund and Littell 2006). Residuals of regressions were inspected for outliers, serial trends, and non-normality. In more complex cases, diagnostic statistics such as AIC (Akaike Information Criteria) and Mallow's C(P) were used (Burnham and Anderson 2001; Freund and Littell 2006).

Hilborn (2016) reviewed the use of correlation in fisheries and ecosystem management and we adopted this advice. It should apply to regression analyses as well since the underlying math of the two techniques is very similar. Correlative evidence is strongest when (1) correlation is high, (2) it is found consistently across multiple situations, (3) there are not competing explanations, and (4) the correlation is consistent with mechanistic explanations that can be supported by experimental evidence. Ideally, manipulative experiments and formal adaptive management should be employed. In large-scale aquatic ecosystems these opportunities for experiments are limited; correlations may not be causal, but they often represent all the evidence available (Hilborn 2016).

Core index of ages 1+ biomass: Potomac River Fisheries Commission pound net catch per effort - Fishery-dependent Potomac River Fisheries Commission (PRFC) pound net catch per unit effort (CPUE as metric tons caught per net day) was used as a biomass index for the TLI (hereafter, PRFC index). Net days fished may be the number of times nets were visited, rather than how long they soaked and fished. Pound nets are stationary, nonselective fishing gear. The time-series extends back to 1964. It is based on annual aggregated catch and effort. A more detailed description of the PRFC index is provided in Appendix 1.

Landings, license counts, and net day estimates were provided by PRFC (Ellen Cosby, PRFC retired, personal communication). This time-series was originally developed by A.C. Carpenter (PRFC retired). During 1964-1993, numbers of Potomac River pound net licenses were not restricted. After 1993, the number of licenses was capped at 100 (A. C. Carpenter, PRFC retired, personal communication; Uphoff 2003b). Catch (MT) per net day fished estimates were available, but discontinuous (1976-1980 and 1988-2021). Predictions of missing net day effort from a linear regression with licenses during the period prior to imposition of the license cap, i.e., data from years where both estimates were available (1976-1980 and 1988-1994) were used to fill in missing catch per net estimates back to 1964 when only uncapped license counts were available. Net days estimates from fishing reports peaked between 4,627 and 5,899 during 1976-1991 and then steadily fell; estimates have been close to 1,000 net days since 2017.

Pound nets were located in Potomac River from Maryland Point to the mouth of the river during 2021 (C. Friend, PRFC, personal communication); this region represents about 54% of the length of the tidal region of the Potomac River (Figure 1). Salinities transition from mesohaline (5-18‰) to oligohaline (0.5-5‰) in this region (MD DNR 2023). A few pound nets are set above St. Clement's Island, but most of the Potomac River commercial menhaden quota is caught by pound nets downstream from St. Clement's Island in the mesohaline region (M. Gary, PRFC, personal communication; MD DNR 2023). As a linear measurement, the distance from St Clement's Island to the Mouth is about 25% of the length of the tidal region of the Potomac River (Figure 1).

The PRFC index has been used as a tuning index of biomass of Menhaden (ages-0 through 5, primarily 1 through 3) in some past Atlantic coast age-structured assessments (ASMFC 2004; 2011; see Appendix 1). More recently, it was used as part of a multiple model approach that incorporated predation on Menhaden and changes in productivity over time to evaluate trade-offs between harvest and ecosystem management objectives (SEDAR 2022b; Drew 2021). The PRFC index was used as a substitute index in sensitivity analyses of a surplus production model that included predation that were based on fishery-dependent indices (SEDAR 2020b).

Core index of ages 1+ Menhaden relative fishing mortality – We estimated relative Bay fishing mortality (Bay relative F; Sinclair 1998) based on the ratio of ostensible landings and the PRFC index to depict the trend in biomass-based F. The ratio of landings to an index can be useful for stocks where estimation of F is not possible (Sinclair 1998; NEFSC 2009). Relative F is not constrained by assumptions about natural mortality (Sinclair 1998; NEFSC 2009).

The theoretical foundation of the Bay relative F approach we used was based on re-arrangement of the Baranov catch equation in Ricker (1975). Instantaneous fishing mortality rate (as weight) in year t can be estimated by rearranging equation 1.40 in Ricker (1975) to

$$(6)\ F_t = C_t / [(B_t + B_{t+1}) / 2];$$

where C_t is total catch as harvested biomass in year *t*, and B are estimated biomass in year *t* and *t*+1 (NEFSC 2009). We substituted ostensible Bay landings as an estimate of harvest and PRFC indices for estimates of biomass to estimate Bay relative F as

$$(7)\ H_t / [(I_t + I_{t+1}) / 2];$$

where H_t is total ostensible Bay harvested biomass (reduction and bait estimates) in year *t*, and I_t and I_{t+1} are PRFC indices of ages 1+ biomass in year *t* and *t*+1 (NEFSC 2009). The numerator was multiplied by a scalar to control leading zeros of the estimate of relative F. The scale of Bay relative F is arbitrary.

Bay relative F was compared to a biomass-based F (Fb) estimated for the Atlantic coast from equation 6 using landings (in weight) and ages 1+ biomass estimates from the Beaufort Assessment Model (BAM; SEDAR 2020a) during 1995-2020. Correlation was used to measure the association between Bay relative F and Fb.

Core index of age 0 relative abundance - We used annual geometric mean (GM) of catches of Atlantic Menhaden per standard seine haul in Maryland's survey of Striped Bass juveniles conducted in Head-of-Bay (roughly from Still Pond to the C and D Canal), and Potomac, Choptank, and Nanticoke rivers (combined) as the juvenile index (GM JI; Durell and Weedon 2022; Figure 1). This time-series runs from 1959 to 2021. The juvenile index was derived annually from sampling at 22 fixed stations within Maryland's portion of Chesapeake Bay. There were seven stations each in the Potomac River and Head-of-Bay and four each in the

Nanticoke and Choptank Rivers. Two seine hauls, a minimum of thirty minutes apart, were taken at each site on each sample round. Sampling occurred during July prior to 1962 (44 samples per year), during July and August during 1962-1965 (88 samples), and during July - September after 1965 (132 samples; Durell and Weedon 2022). The GM JI or data used to derive it have been used in ASMFC Menhaden coastal stock assessments since 2004.

Core indices of availability of Menhaden to Striped Bass – Availability of Menhaden to resident Striped Bass during 1982-2021 was indexed as ratios of relative abundance of age 0 or relative biomass of ages 1+ Atlantic Menhaden to the estimated biomass of Striped Bass capable of eating them and likely to be Bay residents:

$$(8)\ AS = (I_t / (P_t / c));$$

where, AS is an availability ratio for either age 0 or ages 1+ Menhaden, I_t is the PRFC index of Menhaden ages 1+ biomass or the GM JI for age 0 in year t, P_t is the aggregated biomass of Atlantic coast 3- to 6-year-old Striped Bass (as an index of Bay resident biomass able to consume ages 1+ Menhaden) from the ASMFC (2022) stock assessment update, and c is a constant scalar used to reduce leading zeros of the decimal. Striped Bass biomass was chosen because consumption by Striped Bass is a function of weight (Hartman & Brandt 1995). These estimates of Striped Bass biomass contain Delaware River and Hudson River stocks but are dominated by the Chesapeake Bay stock (NEFSC 2019). We viewed trends of the numerators and denominators to determine how the components were contributing to trends in the prey-predator ratios.

Core index of Striped Bass condition – Condition based on visible body fat has been assessed in fall in Maryland's portion of Chesapeake Bay since 1998 and in summer during 1999-2012 as part of a Striped Bass health survey conducted by the Fish and Wildlife Health Program (FWHP). Fish were collected by hook-and-line during multiple trips. They were iced thoroughly on each trip and examined in the laboratory, usually the next day. A relative score was assigned to each fish based on the visual prevalence of mesenteric storage lipids (body fat) and scaled as follows: 0 = no detectable storage lipids, 1 = lipids present, but less than 25% of viscera covered, 2 = approximately 25% to 50% of viscera covered, and 3 = approximately 75% or greater of viscera covered (Jacobs et al. 2013). This body fat index (BFI) has been found to be well-related to measured lipid concentration of fed and unfed Striped Bass in the laboratory. It provided a simple and rapid means of evaluating lipid reserves in sacrificed fish without expensive laboratory equipment or technician time, although classification error increased with the number of classes employed. Conventional weight-at-length indices are coarse indicators of condition of Striped Bass since starving fish replace lipids with water to conserve weight loss (Jacobs et al. 2013).

The proportion of Striped Bass 457-711 mm TL (size capable of consuming ages 1+ Menhaden) without visible lipid reserves (P0) was used as our condition indicator (Jacobs et al. 2013; Uphoff et al. 2022). Uphoff et al. (2022) have used the proportion of Striped Bass without visible body fat (P0) based on BFI data during October-November as their index of condition; P0 was chosen because it could be derived from data collected by CBEF with the same interpretation as FWHP estimates (Uphoff et al. 2018). Simple presence-absence of lipid reserves offered exceptional discriminatory capability in classification, but at the cost of reduced information (Jacobs et al. 2013). Presence or absence of visceral lipids was subject to less error than categories of relative quantity (Jacobs et al. 2013). The proportion of fish below a certain threshold of poor condition rather than mean condition is most likely related to starvation rates (Regular et al. 2022).

Estimates of P0 for 1998–2013 were provided by FWHP and remaining years were estimated from FWHP data (Uphoff et al. 2022). Standard deviations and confidence intervals (90%) of P0 were estimated using the normal distribution approximation of the binomial distribution (Ott 1977). This approximation of the binomial distribution can be used when the sample size is greater than or equal to 5 divided by the smaller of the proportion of positive or zero tows (Ott 1977). The proportion of Striped Bass without body fat (P0) was estimated for October-November of each year as

$$(9)\ P0 = N0 / Ntotal;$$

where N0 equalled the number of qualifying samples without body fat present and Ntotal equalled the total number of qualifying samples. The SD of P0 was estimated as

$$(10)\ SD = [(P0 \cdot (1 - P0)) / Ntotal]^{0.5} \text{ (Ott 1977).}$$

Ninety percent confidence intervals were constructed as:

$$(11)\ P0 \pm (1.645 \cdot SD); \text{ (Ott 1977).}$$

Jacobs et al. (2013) stressed that comparisons of Striped Bass body fat to a nutritional target or threshold in Chesapeake Bay should be based on October-November data since they were developed from samples during that time span. We used October-November samples to estimate P0. These nutritional reference points would indicate vulnerability to starvation and processes associated with inadequate nutrition such as disease (mycobacteriosis) and predation (Jacobs et al. 2013). In the context of the TLI, "target" refers to the boundary of good (green) and uncertain (yellow) status and threshold is the boundary between uncertain and poor (red) status.

A level of P0 of 0.30 or less was used by Uphoff et al. (2022) as a boundary to judge whether resident Striped Bass were in good condition. A target for P0 was not presented in Jacobs et al. (2013). However, Jacobs et al. (2013) presented a condition target for Chesapeake Bay ecosystem-based fisheries management based on body moisture (25% or less of fish with starved status) as a surrogate for lipid content estimated from proximate composition of well-fed Striped Bass. This target was derived from fall 1990 field collections by Karahadian et al. (1995) - the only field samples available from favorable feeding conditions (relatively high forage and low Striped Bass). Jacobs et al. (2013) reported mean tissue lipid of Striped Bass without visible body fat was identical to that estimated from percent moisture, meaning that P0 related strongly to the proportion exceeding the moisture criteria. Variation of tissue lipids estimated from body fat indices was greater than for moisture and the higher P0 target accounted for this with an additional buffer for misjudging status (J. Jacobs, NOAA, personal communication). Our boundary for poor condition (red / yellow) for resident Striped Bass at or greater than 457 mm TL during fall was derived from a period of asymptotically high P0.

Traffic Light Index – We used the median of the PRFC, relative F, GM JI, and both AS indices during the reference period as their yellow / green boundaries (going from uncertain to good conditions) and the 25th or 75th percentiles, depending on relevant direction, as the yellow / red boundaries for poor conditions. This differs from the guidelines in Halliday et al. (2001)

where the yellow / red boundary is set at 60% of the reference period mean, which would indicate a 40% decline from the reference period mean. It also differs from how they are applied to Spot and Atlantic Croaker by ASMFC, which use the fuzzy blend approach based on the reference period mean and the associated upper and lower 95% confidence limits (ASMFC 2014; 2020b). The median and mean would have the same meaning if an indicator was normally distributed within the time-series, while the median is a more robust indicator of central tendency if an indicator has a skewed distribution (Manikandan 2011a; 2011b). The 25th percentile represents a 50% decline from the series median (and mean if the indicators are normally distributed).

We used the Fit Comparison in @Risk (default settings, AIC option) to judge what distribution or distributions might describe these data within the reference period best. @Risk is an add-on to Excel that conducts Monte Carlo simulations for risk analysis (Palisade 2022). Our primary interest was whether data could be considered normally distributed or not. The Fit Comparison fits multiple distributions to data and AIC is used to estimate the relative quality of the fitted distribution (Burnham and Anderson 2001; Palisade 2022). If most of the core metrics were normally distributed, then the criteria of Halliday et al. (2001) might be a reasonable choice rather than the median and 25th percentile. @Risk estimated Δi, (AICi – minimum AICi), where i is an individual modeled distribution (Burnham and Anderson 2001). The Δi values provided a quick "strength of evidence" comparison and ranking of models and hypotheses. Values of Δi ≤ 2 have substantial support, while those > 10 have essentially no support (Burnham and Anderson 2001). We considered distributions with Δi ≤ 2 as alternatives to the best distribution indicated and inspected graphs of the distributions to examine how strongly the candidates corresponded to a normal distribution.

We conducted a sensitivity analysis of the TLI to changes in the 1995-2021 reference period time-series. by removing three years from the beginning or end. These reference period changes were applied to core metrics that depended on the reference period for their boundaries. Additional TLI tables were constructed for 1995-2021 using each of these two changes. The body fat index supplied its own boundaries and was not subject to the sensitivity analysis but was kept in all versions of the TLI. We determined how many changes of each core indicator (except P0) were "better" (advanced into a more positive category) or worse (declined into a more negative category).

Supplemental Menhaden ages 1+ biomass index: Maryland pound net CPUE – Maryland's portion of Chesapeake Bay was represented by an index of Menhaden pound net catch (MT) per net month. This time-series was discontinuous (1980-1984, 1990, and 1992-2020). This index attempts to account for soak time since nets are assumed to fish all month. It is the annual sum of each month's Menhaden pound net catch divided by the annual sum of each month's reported pound nets fished. Indices are adjusted to reflect differences in number of days in the month in the denominator. Fishermen reported monthly pound net catches and number of pound nets fished from 1980-2006. Later estimates from the daily trip records instituted in 2006 were manipulated to be consistent with previous estimates calculated from MD record system monthly data.

Within a series of years readily available to look at catch distribution (2006, 2012-2013, and 2020), 77-93% of Menhaden harvested by pound nets in Maryland were from the region between the Bay Bridge (US Route 50) and Virginia line. This region was mesohaline (MD

DNR 2023). The remainder came from a wide number of reporting areas. The distance from the Maryland – Virginia border to the Bay Bridge represents approximately 60% of the linear distance from the mouth of the Susquehanna River to the Maryland-Virginia border (Figure 1).

Supplemental ages 1+ Menhaden abundance index: Menhaden CPUE from the Striped Bass spawning experimental gill net survey - The Maryland Department of Natural Resources has employed multi-panel experimental drift gill nets to monitor the Chesapeake Bay component of the Atlantic coast Striped Bass population during spawning season since 1985 (SEDAR 2020a). An age-structured fishery-independent relative abundance index for ages 1+ Menhaden has been developed for the ASMFC coastal assessment from these data (SEDAR 2020a).

Multi-panel drift gill nets are deployed in the Potomac River and in the Upper Chesapeake Bay and fished 6 days per week from late March through May, totaling 30-40 sample days (SEDAR 2020a). Individual net panels are 150 feet long. The panels are constructed of multifilament nylon webbing in 3.0, 3.75, 4.5, 5.25, 6.0, 6.5, 7.0, 8.0, 9.0, and 10.0-inch stretch mesh. Sampling locations are assigned using a stratified random design. The Potomac River and Upper Bay spawning areas are each considered a stratum. In both systems, all 10 panels are fished twice daily at randomly selected sites within the strata. Atlantic Menhaden caught in the gill nets are counted and measured for total length (mm TL), when possible, and released. Based on the survey's random sampling design and catch frequency distribution, a General Linear Mixed Model (GLMM) with a negative binomial link function was chosen to develop an index of Menhaden abundance. All Menhaden caught in the survey were above the cutoff size that separates Age 0 and Ages 1+ Menhaden, so all sampled Menhaden were used to calculate the ages 1+ index of abundance. March samples and all samples for the Potomac River were excluded due to low occurrence of Menhaden. In addition, records with missing observations of environmental covariates were excluded, as were records for the 3.13-inch mesh panel because it was not consistently used throughout the time series. Years when Menhaden were not caught (1996, 1997, 2003, and 2004) were removed from the analysis. Significant explanatory variables were determined based on a run of full GLMM negative binomial model. Based on AIC criteria and tests of significance, the final version of negative binomial GLMM model predicted catch as a linear function of year, mesh size, depth, salinity, water temperature, and an offset for logarithm of effort. The calculated index is in units of 1,000 square yards per hour (SEDAR 2020a).

The resulting index is for the Head-of-Bay mainstem between the mouths of the Elk River and Susquehanna Flats downstream to Stillpond Creek, just north of Worton Point, during April and May (Figure 1). This region ranges from tidal-freshwater to oligohaline and represents about 9% of the distance from the mouth of the Susquehanna River downstream to the Maryland – Virginia border.

Supplemental index of age 0 relative abundance - The coastal stock assessment uses Maryland seine data (Durell and Weedon 2022) to estimate an index of age-0 Atlantic Menhaden abundance from Maryland's portion of the Bay (SEDAR 2020a). Prior to index development, site numbers with zero catch and sites that were sampled in less than 50% of time-series were eliminated. A zero-inflation GLMM (generalized linear mixed model) catch per effort estimator was used (SEDAR 2020a). We used correlation analysis to measure the association of the GM JI and GLMM JI for the whole 1959-2021 time-series and for the 1995-2021 reference period.

Table 2 provides a summary of the core and supplemental indices: their location, metric represented, parameter represented, gear type, data type (fishery independent or dependent), and agency source.

Core index of age 0 relative abundance and the Atlantic Multidecadal Oscillation - We used linear regression to examine the relationship of the AMO to age 0 Atlantic Menhaden relative abundance (GM JI) through 2021 in Maryland's portion of the Bay since it influenced the selection of our reference period. An average of AMO monthly indices in the winter – early spring preceding the GM JI (January-April) were used as the independent variable. Menhaden larvae entered Chesapeake Bay from November to April, but very few juveniles recruited from larvae that were spawned prior to January (Lozano et al. 2012; Atkinson and Secor 2017). Monthly AMO indices were obtained from the NOAA Physical Sciences Laboratory http://www.cdc.noaa.gov/data/timeseries/AMO/.

Relationship of core indices of age 0 relative abundance and ages 1+ relative biomass - We estimated the relationship of the GM JI in year t-2 to the PRFC index in year t during 1964-2021 with linear regression. We used the two-year lag suggested for the GMJI in ASMFC (2004).

A multiple linear regression approach was also used to test for possible time-series effects on PRFC CPUE as the fishery changed from less to more regulated in 2006. It was possible that closure of the fishery in 2005 that harvested age 0 Menhaden in fall-winter, the Bay cap on the reduction fishery imposed in 2006, and trip limits for the bait fishery added in mid-2013 altered the relationship of the GM JI to the core adult index. Rose et al. (1986) used categorical variables with linear regression as an alternative to Box-Jenkins models and time-series regression and we used that approach. The multiple linear regression of the PRFC index in year t as a function of the GM JI in year t-2 included an additional categorical variable representing when the Bay cap was in place: "0" indicated the less regulated period (1964-2005 and "1" indicated more regulated (2006-2021). This multiple regression assumed slopes were equal for the two Bay cap categories, but intercepts were different (Neter and Wasserman 1974; Rose et al. 1986; Freund and Littell 2006). This common slope would describe the relationship of GM JI in year t-2 and an adult index in year t while the intercept would indicate the effect of the Bay cap and other restrictions.

We used stepwise selection (SAS PROC REG) to determine whether the linear or multiple regression model was the best choice for describing the relationship of the GM JI with the PRFC index (Freund and Littell 2006). We used improvement in the regression coefficient, Mallow's C(P), and AIC to judge the best model. Inclusion of the cap category coefficient in the regression would indicate that change occurred between the two periods.

Striped Bass condition dynamics – Year-round collections by CBEF during 2006-2015 provided an opportunity to examine seasonal dynamics of Striped Bass condition, particularly in winter. Changes in fish condition are often seasonal and age specific (Tocher 2003; Jacobs et al. 2013; Regular et al. 2022; Cadigan et al. 2022).

Seasonal proportions of resident Striped Bass 457- 711 mm TL (capable of consuming ages 1+ Menhaden) without visible body fat (P0) could be estimated from CBEF collections during summer (June-September), fall (October-November), and winter (December-March) during 2006-2015. Winter collections were from samples of commercial gill net catch and remaining seasons were primarily hook-and-line samples. All years were pooled for seasonal estimates. Summer CBEF collections were compared to estimates made from FWHP summer collections during 2006-2012. The seasonal ratio of the sum of intact Menhaden examined in Striped Bass guts to the sum of Striped Bass examined was estimated as an index of relative prey availability. This numeric ratio was multiplied by the mean weight of intact Menhaden to calculate a ratio of weight of consumed Menhaden per Striped Bass. Estimates of P0 were further refined to monthly intervals (all years pooled). Monthly estimates of P0 by year were not possible because monthly sample sizes were not always adequate throughout the time-series. Seasonal and monthly estimates of condition give a general pattern of changes in P0, but absence of annual changes at the monthly scale prevented determination of annual variability within and among seasons.

Menhaden Bay Relative F and Striped Bass condition - We used regression analysis to determine if there was a relationship of Bay relative F in year t and P0 in fall of year t+1 during 1998-2020. This lag was necessary since harvest had to impact P0 in the preceding winter. After examination of the scatter plot, we used piecewise regression (Neter and Wasserman 1974; Freund and Littel 2006) to test for a "hockey stick" shaped relationship where there were two distinct slopes over the range of the data rather than one continuous relationship. We were interested in estimating what levels of Bay relative F were when P0 was predicted to be at its good or poor boundaries in the TLI. Piecewise regression uses indicator variables in a multiple regression to estimate a slope shift coefficient (V) that allows the relationship to shift for the two or more different classes of data (Neter and Wasserman 1974; Freund and Littel 2006). Analysis was based a general model where V = 0 when relative F was associated with changes in P0 from low to high and V = 1 when relative F was associated with high P0 (near or at the boundary for classifying it as poor). The terms produced for this equation were an intercept (b), an indicator variable coefficient (v), a maximum slope (m_1), and the slope at consistently poor condition (m_2). These terms were then used to estimate relationships of P0 and relative F below the boundary for poor classification (v • 0) and when relative F was at or above the boundary (v • 1). When relative F was low, the relationship was described by

$$\text{(15)} \qquad P0 = (\text{relative F} \bullet (m_1 + m_2)) + (b + v).$$

When relative F was high, relative F was described by the equation:

$$\text{(16)} \qquad P0 = (\text{relative F} \bullet m_2) + b.$$

Dynamics of Menhaden availability (AS) indices – We used linear regression to investigate the strength of the relationships of AS of either ages 1+ or age 0 Menhaden with P0 of resident Striped Bass or AS of ages 1+ Menhaden with relative F.

Results and Discussion

Ages 1+ Menhaden and resident Striped Bass in winter - During December-March, 2006-2015, there were 792 Striped Bass 457-711 mm TL with individual consumed fish identified: 422 Menhaden (59 gm average) were consumed, 633 Atlantic Croaker (3.5 gm average), 177 Bay Anchovy (0.4 gm average), and 107 White Perch (45 gm average). By weight, Menhaden represented 76% of identifiable fish in Striped Bass guts.

Intact Menhaden that could be measured were present during December-March. There were 280 less than or equal to 159 mm TL (i.e., age 0 sized) and they accounted for 20% of total Menhaden weight (Figure 2). An additional 142 Menhaden were greater than 159 mm TL (ages 1+) and they accounted for 80% of total Menhaden weight (Figure 2); some to many of these Menhaden would have been age 0 in fall that were promoted to age 1 in winter.

Examination of the histogram of the total lengths of Menhaden consumed (25 mm increments) indicated a peak at the 175 mm TL increment, after which lengths quickly tapered off (Figure 3). A nadir was reached at the 225 mm TL increment, followed by a secondary peak at the 250 mm TL increment (Figure 3). The nadir was interpreted as a break between Menhaden less than 2 years old for fish below 225 mm TL (not subject to fishing the previous season as age 0s) and ages 2+ above (vulnerable to harvest the previous season as ages 1+). Menhaden categorized as ages 2+ in winter comprised 62% by weight and 16% by number of the 422 identifiable Menhaden in in Striped Bass guts during winter, 2006-2015.

We used static length cutoffs based on data pooled across years of diet collections. However, size-at-age of Menhaden consumed may vary annually. Mean lengths and weights at age of Menhaden are linearly and negatively related to recruitment, indicating density-dependent growth (Schueller and Williams 2017). In addition to the number of prey available via recruitment, associated variations in Menhaden growth would influence feeding through length available and its effect on handling time and swimming speed. Poor Striped Bass feeding success associated with small Menhaden year-classes could be compounded by greater size at age of Menhaden. The optimum prey-predator length ratio for Striped Bass was predicted to be 0.21 and low abundance and greater Menhaden size could force a switch to smaller prey (Overton et al. 2009). Overton et al. (2015) noted that Striped Bass diets in Chesapeake Bay had shifted over time to smaller prey, Bay Anchovy and Blue Crab, in Bay diet studies conducted during the 1950s, 1990-1992, and 1998-2001, possibly in response to declines in Menhaden abundance.

Review of bivariate plots of total lengths of Menhaden consumed and of Striped Bass that consumed them during 1997-1998 (Walter and Austin 2003) and 1998-2001 (Overton et al. 2009) indicated that few Menhaden over 200 mm were present in guts of 457-711 mm TL Striped Bass. These plots included Striped Bass sampled during winter. Both studies were conducted before the directed fishery on age 0 Menhaden ceased and the Bay harvest reduction fishery cap was in place.

Ages 1+ and 2+ Menhaden comprised an important portion of winter diet biomass of large resident Striped Bass during 2006-2015. If there was a problem due to the fishery, it would be from depletion of ages 1+ Menhaden prior to winter. Changes in Menhaden migration would be another possibility.

Estimated Chesapeake Bay Menhaden harvest – Estimated harvest by the reduction fishery from the Bay ranged from 5,733 MT to 178,200 MT during 1964-2008 (Table 3; Figure 4). Estimated reduction harvest was between 5,733 and 69,800 MT during 1964-1969. It rapidly rose to 178,200 MT by 1972 and fluctuated between 31,440 and 175,300 MT through 1987. Reduction harvest estimates remained high through 1995, fluctuating between 155,700 MT and 171,000 MT. Estimates dropped to between 80,900 and 135,300 MT through 2008. A 109,020 MT cap was imposed on the reduction fishery in 2006. The cap was reduced to 87,216 MT in

2013 and to 51,000 MT in 2020. Reduction catch was reported as 65,000 MT in 2020 (Table 2; Figure 4).

There were two estimates (5,733 MT in 1964 and 31,400 MT in 1981) that appeared anomalously low (Table 2; Figure 4). Reduction landings prior to 1964 (1955-1963) ranged from 25,646 MT to 66,408 MT and remained in that range during 1965-1969. Landings for 1981 were sandwiched between estimates of 177,157 MT and 145,576 MT and all other landings during 1979-1999 were above 100,000 MT (Table 2; Figure 4).

Bait landings estimates ranged from 4,200 MT to 30,700 MT during 1964-2020 (Table 2; Figure 5). Estimates from the old NMFS website fluctuated from 9,500 to 30,700 MT during 1964-1989 and were consistently above 19,000 MT during 1975-1983. Bait landings were near 10,000 MT during 1990-1993 and fell below 10,000 MT through 2002. Bait landings from the PDT were similar to those estimated from the old NMFS website during 1987-1999 with the exception of much a much higher PDT estimate in 1988. After 1999, PDT bait landings shifted upward and remained between 15,000 and 28,300 MT (Table 2; Figure 5). We do not know the reason for this upward shift but speculate that snapper rig landings may have been separated out from reduction fishery purse seine landings and then added to the bait category.

Estimates of ostensible total landings from the Bay were below 100,000 MT during 1964-1969 (Table 2; Figure 6). They rose afterward and were usually between 150,000 to 200,000 MT during 1972-2003. They fell below 150,000 MT through 2012, remained near 100,000 MT through 2019, and were approximately 82,000 MT in 2020 (Table 2; Figure 6).

Percent of reduction harvest drawn from the Bay after 1984 was mostly above the median percent (40.1%) for 1964-2005 (i.e., prior to the Bay cap). Even with the advent of a cap of 109,000 MT on reduction landings, the percent potentially drawn from Chesapeake Bay did not drop below the 1964-2005 median percent during 2006-2009 (based on reported reduction landings from the Bay). Movement toward the 1964-2005 median percentage in 2018-2019 corresponded with the Bay cap lowering to 87,000 MT and a rise in the coastal quota. Lowering the Bay cap further to 51,000 MT dropped the expected percentage extracted by the reduction fishery from the Bay to 26% of the coastal quota; percentages this consistently low were last seen in the 1960s.

Bay caps on reduction fishery harvest may be interpreted as an experiment in harvest reduction, fulfilling the manipulative experiment ideal that aids in interpreting correlation (or regression) results (Hilborn 2016). The 51,000 MT cap has not been in place long enough to judge its effect.

Core index of ages 1+ biomass: Potomac River Fisheries Commission pound net catch per effort – Estimated PRFC CPUE was at its lowest level, 0.13-0.56 MT / net day, during 1964-1971 (Figure 7). There was a rapid rise in CPUE between 1971 and 1973 from 0.40 to 1.53 MT / net day. Catch per effort remained high (0.94-1.95 MT / net day) through 1988 and then fell by about 60% between 1988 and 1990. The index varied at a low level (0.45-0.83 MT / net day) during 1990-2003 and then rose. It has varied from 0.75 to 1.50 MT / net day since 2004 (Figure 7).

Median PRFC CPUE during the 1995-2021 reference period was 0.98 MT / net day and the 25th percentile was 0.72 MT / net day (Figure 8). Indices below the 25th percentile (red) occurred during 1995-2003 and above median indices (green) were intermittent after 2004. The last four years of the time-series were above the median (Figure 8). PRFC indices during 1995-2021 ranged from 0.48 to 1.51 MT / net day with an approximate mode of 1.14 (Figure 9). The mean (0.92) was similar to the median (Figure 9). The Fit Comparison in @Risk indicated that a normal distribution fit the time-series of indices best (mean = 0.92 and SD = 0.26). Five other distributions (not shown) had Δ_i within 2. Visually, four of these distributions were very similar to the normal distribution, while one (triangular distribution) was highly skewed to low indices.

We have assumed that pound net catchability has been relatively stable, particularly during the reference period. Pound nets are not suitable for a mobile search fishery (unlike purse seines), where an inverse catchability relationship with biomass and abundance (hyperstability) would be expected (Schaaf 1975; Walters and Martell 2004). Pound nets require a considerable investment of money and physical effort to set. However, reduced density of pound nets could allow for increased catchability if nets remain in productive locations and removed nets were in less optimal locations. Positive relationships have been documented between spatial dispersion of sites occupied by a species and abundance for many taxa, including fishes, but they are not universal (Gaston et al. 2000; Miranda 2023). Declines of marine fishes often involve loss of spatial stock structure and spatial spread of the stock may increase as a population increases (Mangel and Smith 1990; Walters and Martell 2004).

Anecdotal information indicated possible change in catchability early in the time-series. Prior to the 1970s, pound nets in the Potomac River were primarily set for Blueback Herring *Alosa aestivalis* and American Shad *Alosa sapidissima* and ran from shallow to deep water (A. C. Carpenter, PRFC retired, personal communication). The pound net fishery shifted to a Menhaden bait fishery as the Alosid fisheries collapsed; nets set for Menhaden were scattered and set mostly in shallow water (A. C. Carpenter, PRFC retired, personal communication). Pound nets may have been set differently as Striped Bass allocations increased in the 1990s, but many to most pound nets were set to catch Menhaden as bait for the Blue Crab pot fishery and recreational fisheries, and more recently as bait for Lobster *Homarus americanus* in New England (Maryland Sea Grant 2011).

Dean (2014) reviewed the PRFC index as a fishery-dependent index of abundance for the Beaufort Assessment Model (BAM) using a GLM approach. Data with daily information (1989-2012) were summarized to monthly resolution and used to develop an index. Zero catches made a small component of catches when daily records were available and were included in the index. A model with port and month as predictors was selected as the preferred model and it achieved an average CV of 13% (Dean 2014). The PRFC index was not used in coastal stock assessments after fishery-independent indices (FI) were developed for the 2015 assessment (SEDAR 2015). Fishery-dependent (FD) indices were considered for the BAM used in SEDAR (2015) but were rejected because of the requirement for individual trip records resulted in shorter FD time-series than available for FI indices, FD indices lacked length and age data needed for age-based modeling, and they were correlated with FI indices (SEDAR 2015). The PRFC biomass index was included in sensitivity analyses of surplus production models used as part of the assessment that developed ecological reference points for Atlantic Menhaden along the Atlantic coast (SEDAR 2020b; Drew et al. 2021).

Core Index of ages 1+ Menhaden Bay relative fishing mortality – Bay relative F underwent fairly rapid transitions from low to high and back during 1964-2020. It was at its highest level, roughly above 2.00 (the scale is arbitrary), during 1966-1972 and 1989-2003 (Figure 10). The index varied at a lower level (below 1.30) during 1974-1986 and after 2005 (Figure 10).

Median Bay relative F during the 1995-2021 reference period was 1.16 and the 75th percentile was 1.90 (Figure 11). Bay relative F above the 75th percentile (red) occurred during 1995-1998 and 2001-2003. Bay relative F was at or below the median (green) after 2006 with the exception of 2009 (Figure 11). Bay relative F during 1995-2020 ranged from 0.76 to 2.73 and was highly skewed toward low values (Figure 12). The mean (1.46) was dissimilar to the median (1.14) and there was an approximate mode at 1.03 (Figure 12). The Fit Comparison in @Risk indicated that a triangular distribution (triangle with the ascending side at the lowest values of Bay relative F) fit the time-series of indices best and two others had similarly skewed distributions (not shown) with Δ_i within 2.

Time-series plots of Bay relative F and Fb (Atlantic coast biomass F) during the reference period had similar declining trends (Figure 13), but relative F in the Bay appeared higher on a relative basis than Fb in 2001-2003 and lower in 2006-2008 and 2014-2019. Average Fb for 2018-2020 was 49% lower than during 1995-1997, while average relative F declined by 62% (Figure 13). Bay relative F and Fb were well correlated ($r = 0.72$, $P < 0.0001$).

Sinclair (1998) developed relative F and presented a case study with Atlantic Cod *Gadus morhua* where relative F at age and length were compared with F estimates obtained with sequential population analysis. These two sets of estimates were found to be of similar magnitude and trend. The form of relative F for Bay Menhaden used here does not consider age structure, so our aggregated biomass approach addresses average annual shifts in F as ages 1+ biomass across time. An aggregated relative F was a key component of the 2009 ASMFC Weakfish *Cynoscion regalis* stock assessment (NEFSC 2009) that determined a rise in M rather than F was responsible for the decline of the stock. Kahn (2019) used aggregated relative F based on recreational CPUE in the denominator to depict trends in American Eel fishing mortality independent of those estimated by stock assessment models. Petrie et al. (2022) used relative F in an investigation of changes in Flemish Cap Atlantic Cod spawning stock biomass, maturity-at-age, weight-at-age, and recruitment and their relationship to stock collapse and recovery.

Sinclair (1998) developed relative F as the ratio of commercial catch divided by a research vessel survey index of relative population abundance. If the survey was conducted near the middle of the fishing year, its catchability was constant, and the catch reporting rate remained constant, relative F would be proportional to the actual F and trends in relative F would reflect trends in F. Relative F is insensitive to changes in M provided the research survey occurs close to the middle of the fishing year. Relative F offers useful diagnostics for interpreting stock assessment results (Sinclair 1998).

We have substituted fishery-dependent pound net CPUE biomass indices in the absence of a research vessel survey index with age-aggregated estimates of ostensible landings (both in biomass) to estimate Bay relative F in the absence of long-term age structure information. Pound nets are mostly fished during spring-fall (peak months are May-September), approximating the mid-year requirement. Possible pound net catchability changes were described previously.

Confounding F with migration is a concern with Bay relative F, particularly for years that the Bay cap was substituted for reduction fishery landings and for as long as that substitution is necessary. Major changes in overall migration should be reflected in the PRFC ages 1+ biomass index while the reduction cap would not change. The PRFC and MD pound net indices have exhibited similar variations at relatively high reference period levels since 2007, two years prior to the adoption of caps as a substitute for Bay landings estimates, so an overall migration shift was not supported. Both reduction and bait landings are now subject to confidentiality rules that necessitated substituting the cap for reduction landings. As it stands, future landings based on caps will be largely frozen in place and a sudden, large upward shift in relative F should prompt an effort to detect migration change.

Movement of tagged adult Menhaden in the 1960s to and from a region consisting of Chesapeake Bay and adjacent coastal waters from regions north or south occurred primarily during October-May (Liljestrand et al. 2019). Exchange among regions was low during June-September (Liljestrand et al. 2019). Based on Maryland pound net landings and assuming seasonality is the same in Potomac River, June-September accounted for about 60% of their annual catch during 2012-2020. April, May, and October accounted for about 33% of the remainder. Most harvest occurs when exchange among regions should be low, but a sizeable fraction does occur during months when Menhaden migrate into and out of the Bay.

Catch limits that affected pound nets were imposed in 2014 and percent caught in June-September 2012-2013 (53%), when limits were not in place, was less than 2014-2020 (63%) when they were. January-May percentages were fairly similar for catches when limits were not in place and when they were (31% and 28%, respectively), but relative differences were larger during October-December (16% and 9%, respectively).

The boundary of this analysis, the mouth of Chesapeake Bay, was not a physical boundary to Atlantic Menhaden. We estimated Bay relative F in the context of a (mostly) closed Chesapeake Bay population. Gulland (1983) considered definition of a unit stock an essentially operational matter, being tied to the models used, the questions asked, and information available. When the bounds of the unit stock extend beyond the limits of the fishery being analyzed, then the pattern of exploitation beyond the limit of analysis will determine whether the analysis of portion of the stock will be misleading. If the fishery outside the boundary is similar to that inside, correct answers may be provided (Gulland 1983). Coastal Menhaden reduction landings have been largely drawn from Chesapeake Bay and mid-Atlantic waters adjacent to reduction factories (Smith 1999; Anstead et al. 2021). The reduction fishery has operated largely in the same manner outside the Bay as inside. Changes in Bay relative F were similar to those estimated for the Atlantic coast using biomass and landings. These characteristics supported the idea that Bay relative F captured essential information on the status of Menhaden in the Bay.

Core index of age 0 Menhaden relative abundance - The GM JI was at its lowest level, 0.16-0.89, during 1959-1970 (1962 was an exception at 2.34; Figure 14). The GM JI became elevated between 1971 and 1973 (from 2.61 to 4.42). It increased sharply in 1974 and generally remained 2- to 4-times higher than in 1973 through 1981. The GM JI fell to an intermediate level between high and low during 1982-1991. It returned to a low level in 1993 and has remained there through 2021; the GM JI has ranged from 0.28-1.84 since 1993 (Figure 14).

Median GM JI during the 1995-2021 reference period was 0.87 and the 25th percentile was 0.57 (Figure 15). Indices below the 25th percentile (red) occurred mostly before 2004 (2017 was an exception). Above median indices (green) occurred in 1997, 1999, 2005 and fairly consistently after 2008. Overlap of 90% CIs of the GM JI indicated that the lowest estimates (2002-2004 and 2017) were likely to be less than most others after 2006. The GM JI during 1995-2021 ranged from 0.28 to 1.84 with an approximate mode at 0.94 (Figure 16). The mean (0.91) was near the median (0.87; Figure 16). The Fit Comparison in @Risk indicated that a triangular distribution skewed towards low values fit the reference period time-series best and one other similar distribution (not shown) had Δ_i within 2.

Core indices of Menhaden availability to Striped Bass – The AS for Striped Bass on ages 1+ Menhaden ranged from 0.04 to 1.54 over the entire 1982-2021 time-series (Figure 17). The index peaked at 1.54 in 1983 when Striped Bass biomass was extremely low and Menhaden biomass was high. The ratio fell rapidly to 0.09 by 1990 and remained close to that level through 2003. The AS for ages 1+ Menhaden began to climb after 2003 and reached a peak at 0.30 in 2012. It remained between 0.14 and 0.25 during 2013-2021 (Figure 17).

The AS for age 0 Menhaden ranged from 0.03 to 4.00 over the entire 1982-2021 time-series (Figure 17). The index peaked in the early to mid-1980s when Striped Bass biomass was extremely low and Menhaden year-class success was high. The ratio fell continuously from 3.89 in 1985 but remained above 0.47 through 1991. It then fell to 0.09 in 1993 and, with two exceptions, remained between 0.04 and 0.10 through 2007. It was often between 0.10 and 0.30 after 2007 (Figure 17).

Median AS for ages 1+ Menhaden during the 1995-2021 reference period was 0.14 and the 25th percentile was 0.07 (Figure 18). Indices below the 25th percentile (red) occurred before 2004. At or above median indices (green) occurred after 2006 (Figure 18). The AS for ages 1+ ranged from 0.04 to 0.30 during the reference period with an approximate mode at 0.061 (Figure 19). The mean and median were the same (0.14). The Fit Comparison in @Risk indicated that a triangular distribution (skewed to lower values; not shown) fit the time-series of indices best.

Median AS for age 0 Menhaden during the 1995-2021 reference period was 0.11 and the 25th percentile was 0.07 (Figure 20). Indices below the 25th percentile (red) occurred before 2007 and in 2017. At or above median indices (green) occurred in 2005 and during 2008-2021 (but not 2017; Figure 20). The AS for age 0 Menhaden ranged from 0.03 to 0.30 during the reference period with an approximate mode at 0.07 (Figure 21). The mean was 0.14. The Fit Comparison in @Risk (not shown) indicated that a triangular distribution skewed toward lower values fit the time-series of indices best (Figure 21).

Increased AS for ages 1+ and age 0 Menhaden during 1995-2012 reflected increased Menhaden indices and a steady decline in ages 3-6 Striped Bass biomass estimates (Figure 22). After 2012, Striped Bass biomass began to increase and ages 1+ AS declined but remained at or above the good boundary. The PRFC index largely remained above or near the good boundary during this period. The age 0 AS appeared to vary without trend above the good boundary during 2013-2021, although the poor 2017 Menhaden year-class resulted in the age 0 Menhaden AS dropping to uncertain (yellow) status. The GM JI was somewhat higher during 2019-2021 and this increase offset the increase in Striped Bass biomass (Figure 22).

Uphoff (2003a) indexed changes in potential attack success along the Atlantic coast during 1982-1998 as Menhaden per Striped Bass in numbers or biomass; we have labelled Menhaden to Striped Bass ratios as availability instead of potential attack success to avoid concerns that AS was not determined from diet data. Decreased potential attack success in Uphoff (2003a) was inferred from a 97% decline in ratios of forage-sized Atlantic Menhaden to Striped Bass. These estimates came from coastal stock assessments and were used as a proxy for trends in Chesapeake Bay (Uphoff 2003a).

Current monitoring of resident Striped Bass forage and well-being in Maryland's portion of the Bay in fall (Uphoff et al. 2015; 2022) uses the ratio of the Menhaden GM JI to a resident Striped Bass recreational CPUE to index relative attack success in Maryland's portion of the Bay in fall when age 0 Menhaden dominate their diet and migratory Striped Bass are very rare. Striped Bass too small to consume ages 1+ Menhaden were included in the recreational CPUE index and were likely abundant enough to positively biased the recreational CPUE index if it was used as the denominator for an AS index for ages 1+ Menhaden.

A predator's functional response (number of prey consumed per unit area and time by an individual predator) is both a function of attack success and prey handling time. Handling time usually varies little for a given predator, but search time and interference influence consumption by changing the attack rate (prey consumed per unit of search time; Yodzis 1994). Predators in natural systems may be closer to prey-predator ratio dependence than dependence on prey abundance alone (Ginzburg and Akçakaya 1992). Attack success indexed from the ratio of prey-to-predator allows for the effect of predator interference to be included (Ginzburg and Akçakaya 1992; Yodzis 1994; Ulltang 1996; Uphoff 2003a; Walters and Martell 2004). Some predators are less efficient when there are fewer prey or there is interference from other predators (including their own species) and these factors may change attack rates in a nonlinear fashion (Yodzis 1994).

Core index of Striped Bass condition – Condition of resident Striped Bass in fall transitioned from mostly poor during 1998-2004 to mostly good condition after 2013 (Figure 23). Striped Bass were in good condition (P0 $\leq$ 0.30) during fall 2008-2010, 2014-2015, and 2017-2021. Estimated P0 (0.01) in 2021 was the lowest (fish were in best condition) of the time-series. The poor boundary (red) was set at P0 = 0.70 based on asymptotically high estimates during 1998-2004. This level of body fat allowed for separation of most of the years when condition was considered poor from intermediate estimates (yellow). Resident Striped Bass during fall were in poor condition (red) during 1998-2004 (except 2002). The 90% confidence intervals of P0 allowed for separation of years within the good boundary from remaining estimates and estimates within the poor boundary from those within the good boundary (Table 4). Confidence intervals of P0 that indicated poor condition during fall 1998-2001 and 2004 could be separated from most (7 of 8) P0 estimates after 2004 that fell between the poor and good (yellow region; Table 4, Figure 23).

Jacobs et al. (2013) developed their body fat target (our good boundary) from field collections by Karahadian et al. (1995) during 1990-1992 for comparisons with their 1998-2001 field collections; threshold condition used here was derived from the peak in P0 estimates (1998-2004). While AS for age 0 was high during 1990-1992, AS for ages 1+ (based on the PRFC index) was low. Estimates of P0 have improved beyond the proposed target of Jacobs et al. (2013) since 2014 concurrently with improvement in AS for ages 1+ beyond 1990-1992 levels while AS for age 0 was above its lowest point (1995-2004) but did not approach 1990-1992.

Traffic Light Index (TLI) - The TLI exhibited a mix of core indicators that were poor (red) or uncertain (yellow) during 1995-2004 (Figure 24). Core indicators were entirely or mostly red during 1995-1998 and became primarily yellow, with two or three red core indicators during 1999-2000. The TLI returned to predominately red during 2001-2004. A transition toward better metrics was indicated by the TLI during 2005-2007. Red core indicators diminished, yellow core indicators became predominant, and green (good or safe) core indicators began to appear. After 2007, TLI core indicators were predominately green with some yellow indicators interspersed and a single red indicator. All indicators have been green since 2018 (Figure 24).

Removing three years from the beginning or end of the reference period to estimate boundaries did not alter the general pattern of the TLI. Indicators were predominately red through 2004 with both treatments and a transition to mostly green occurred during 2008-2010 in both the unaltered TLI (Figure 24) and the TLI based on a 1998-2021 reference period (Figure 25). The transition to mostly green occurred by 2007 when the reference period boundaries were based on 1995-2019 (Figure 26).

Ten of the 135 metric-year combinations during 1995-2021 worsened (green to yellow or yellow to red) when the first three years were removed from the reference period to calculated TLI boundaries and none improved. The juvenile index accounted for three years; the PRFC index, Relative F, and AS for ages 1+ Menhaden accounted for two years each; and AS for age 0 Menhaden accounted for one year.

When the latest three years of the time-series were removed to estimate TLI boundaries, 16 of 135 TLI elements improved and one worsened. The PRFC index, Relative F, and AS for age 0 Menhaden each improved in 4 years; the juvenile index improved in three years and worsened in one; and AS for ages 1+ improved in one year.

All but six of these changes occurred during 1995-2009 for both treatments. The predominant worsening or improvement exhibited for reference period changes reflected increased improvement in metrics over time. The first three years were among the worst for the five metrics in the full reference point time-series and the last three years were among the best.

There were three basic questions to be addressed by the TLI. These questions are paired with a brief answer about conditions in 2021.

1) How are Menhaden doing in Maryland's part of the Bay? Biomass and recruitment are at a good level (green, above their reference period medians) given prevailing ecological conditions.
2) Are there enough Menhaden for resident Striped Bass? It appears that there are enough for Striped Bass. Menhaden per Striped Bass and condition indices are within their good boundaries (green).
3) Is the limitation of Menhaden harvest with the Bay through a reduction fishery cap and other measures having a positive effect on Menhaden and resident Striped Bass? It is plausible that there has been a positive effect since relative F has declined and core indices improved concurrently with the cap. Improvement also coincided with diminished harvest of age 0 Menhaden during fall-winter migration from the Bay that followed the closure of the Beaufort reduction plant in 2005, as well as management imposing quotas along the Atlantic coast.

Four of the five metrics that relied on estimates of central tendency for status exhibited skewed distributions, supporting the use of the median and 25th percentile for boundaries of the TLI. Only one metric, the PRFC index, was classified with the normal distribution as the best model of its distribution. The condition indicator, P0, supplied its own boundaries (it was the sixth metric).

Not all indicators were available for the entire 1995-2021 time-series. The condition indicator was not available until 1998. Relative F will always be blank for the terminal year of the TLI since it requires the average of the PRFC index in years t and t+1 for the denominator unless it is approximated by H_t / I_t in the terminal year.

In the long term, there is potential to expand the TLI into a Baywide index by incorporating metrics from Virginia and transform the TLI into Traffic Light Approach (TLA) management triggers for the entire Chesapeake Bay. The TLA would involve extensive work with Bay jurisdictional partners (Virginia Marine Resources Commission or VMRC, and PRFC), stakeholders, and the ASMFC.

Application of a full TLA needs a decision rule structure for management (Halliday et al. 2001). The TLA was originally proposed for applying precautionary fisheries target and limit reference points in situations where data and assessment capabilities were limited (Caddy and McGarvey 1996; Caddy 1998; 2002; 2015; Halliday et al. 2001). Multiple indicators with limit reference points would be used to judge a fishery and the fraction of traffic lights that were red would indicate severity of the needed management response (Caddy 2015).

The TLA is a method by which diverse sets of qualitative and quantitative indicators can be used to monitor and manage fish populations (i.e., a Traffic Light Precautionary Management Framework). Because of its flexibility, the TLA can be adapted to ecosystem-based fisheries management (Fogarty 2014; Caddy 2015). It is usually applied in fisheries management when data are too limited to use a quantitative stock assessment model, but the approach is not limited to data poor situations (Halliday et al. 2001). Current examples of the TLA can be found in the ASMFC Interstate Fisheries Management Plans for Spot and Atlantic Croaker (ASMFC 2014; 2020b).

Supplemental Atlantic Menhaden Age 1+ Biomass: Maryland Pound Net Catch per Effort – Maryland pound net CPUE was highest at the beginning of the time-series, 1980-1982 (13.9-22.5 MT/net month; Figure 25). A continuous decline started after the peak in 1981 and CPUE fell to 2.8-3.9 MT/net month during 1992-1996, and then fell further to between 1.8 and 2.4 MT/net month during1997-2003. MD CPUE became elevated after 2006 and ranged from 4.4 to 12.0 MT/net month through 2020. Estimates were absent for 1985-1991 (Figure 25).

On a relative basis, MD CPUE was a good bit higher than PRFC CPUE in 1980-1981, but remaining years were similar in relative abundance trends (Figure 25). Correlation analysis indicated that MD and PRFC CPUE were well correlated during 1980-2020 ($r = 0.79$, $P < 0.0001$) and represented very similar trends over a large portion of the Bay in Maryland.

Dean (2014) reviewed the MD CPUE index as a fishery-dependent index of abundance for the Beaufort Assessment Model (BAM) using a GLM approach. Data (1992-2012) were summarized to monthly resolution and used to develop an index. Some permits, areas, and months with low catches were removed from analysis. Zero catches made up 33% of effort included in the index. A delta-lognormal GLM Index of adult menhaden abundance from the Maryland pound net fishery model with year, month, and permit as predictors was selected as the preferred model and it achieved an average CV of 15% (Dean 2014). The MD CPUE index was not used after fishery-independent indices were developed for the 2015 assessment for the same reasons outlined earlier for the PRFC index (SEDAR 2015).

Supplemental age 1+ Menhaden abundance index: Menhaden CPUE from the Striped Bass spawning experimental gill net survey - Head-of-Bay relative abundance was highest at the start of the time-series, 1985-1988 (1.15-1.88 per 1,000 square yards per hour; Figure 25). An abrupt decline occurred in 1989 when CPUE fell to 0.13 and it was no higher than 0.33 during 1990-2011. Head-of-Bay CPUE elevated to 0.42 to 0.77 during 2012-2016. It fell afterwards and was no higher than 0.20 after 2016 (Figure 25). Estimates were not available for 1993, 1996-1998, and 2003-2004, years when TLI indicators were often red.

Correlation analysis was used to measure the strength of the association of MD pound net CPUE or PRFC CPUE with Head-of-Bay CPUE. Head-of-Bay CPUE was modestly correlated with MD pound net CPUE ($r = 0.56$, $P < 0.0012$, $N = 30$) and the PRFC index ($r = 0.45$, $P < 0.0053$, $N = 37$). All three indices seemed to agree that biomass and abundance of ages 1+ Menhaden were much higher during the 1980s, but the Head-of-Bay index did not support consistent elevation of Bay Menhaden relative abundance exhibited by both pound net biomass indices starting in the mid-2000s (Figure 25). Some of this inconsistency may be explained by the different currencies of the indices (abundance or biomass) and their temporal and geographic separation (the gill net survey occurs as Menhaden migrate into the Bay and is the greatest distance from the mouth of Chesapeake Bay). In addition, the Head-of-Bay gill net survey is located in fresh-tidal to oligohaline salinities, while most pound net catches for MD and PRFC pound net biomass indices are drawn from downstream mesohaline regions. It is also possible that Menhaden enter or leave the Head-of-Bay through the Chesapeake and Delaware Canal that connects Head-of-Bay and Delaware Bay.

Supplemental index of age 0 relative abundance - The GLMM JI (Figure 26) appeared to have a similar pattern of year-class success as the GM JI during 1959-2021. Correlation analysis indicated that GM JI and GLMM JI estimators of central tendency were strongly correlated during 1959-2021 ($r = 0.92$; $P < 0.0001$) and would indicate the same long-term trends. They were not as coherent during the 1995-2021 reference period; the GM JI indicated an increase beginning after 2004 while the GLMM JI indicated a more random pattern (Figure 27). The two indices of age 0 relative abundance were modestly correlated during the 1995-2021 reference period ($r = 0.51$, $P \leq 0.0065$). The differences in fit between the long-term and reference period may represent the difference between an index derived from observations (GM JI) and a model-based index developed from a period of high contrast in relative abundance being applied to a period of much less data contrast.

Core index of recruitment and the Atlantic Multidecadal Oscillation - Based on visual inspection, juvenile indices during 1959-2021 greater than the maximum detected in the reference period were present when the AMO was negative (Figure 28). The relationship of the winter AMO and GM JI was described by the equation:

$$\text{(17)} \qquad \text{GM JI} = (-13.89 \bullet \text{AMO}) + 2.77; \ (r^2 = 0.39, P < 0.0001).$$

Standard errors for the slope and intercept were 2.25 and 0.40, respectively. There was patterning of residuals with the winter AMO (not shown). There was wide random scatter of residuals when the AMO was negative and a tight, increasing linear pattern (residuals increasing from approximately -2.2 to 1.85) as AMO indices increased from 0 to 0.30. This relationship was rejected based on the residuals.

The regression was rerun after $\log_e$-transforming the GM JI and the fit was similar ($r^2 = 0.37$, $P < 0.0001$), but the residuals (not shown) were substantially improved and appeared random throughout the range of values. The relationship was described by the equation:

$$(18) \qquad \log_e \text{GM JI} = (-4.22 \bullet \text{AMO}) + 0.28;$$

standard errors for the slope and intercept were 0.70 and 0.12, respectively. Environmental conditions can have a multiplicative effect on year-class success and the $\log_e$-transformation linearized the index of age 0 relative abundance. Lognormal recruitment distributions are common in fish datasets (Hilborn and Walters 1992; Walters and Martell 2004).

Relationship of core indices of age 0 relative abundance and ages 1+ relative biomass - The relationship of the GM JI in year t-2 to the PRFC index in year t (Figure 29) was described by the equation:

$$(19) \qquad \text{PRFC}_t = (0.056 \bullet \text{GM JI}_{t-2}) + 0.73 \ (r^2 = 0.35,\ P < 0.0001).$$

The SEs for the slope and intercept were 0.010 and 0.051, respectively; all terms were significant at $P < 0.0001$. A plot of residuals against time (not shown) produced periods of consistently positive residuals (1980-1984 and 2004-2021) and negative residuals (1964-1972 and 1990-2003).

Inclusion of a categorical variable intended to account for the effect of regulatory changes on the PRFC index improved the fit from the single variable regression to $R^2 = 0.59$, Mallow's C(P) (from 33.9 to 3.0), and AIC (from -134 to -158). The relationship of the PRFC index in year t to the GM JI in year t-2 was described by the equation:

$$(21) \qquad \text{PRFC}_t = (0.072 \bullet \text{GM JI}_{t-2}) + (0.438 \bullet \text{category}) + 0.564 \ (P < 0.0001;\ \text{Figure 30}).$$

The SEs for the slope, regulatory category coefficient, and intercept equaled 0.008, 0.076, and 0.050, respectively. The range in GM JIs for the regulated category (0.32-1.80) was considerably smaller than for the unregulated period (0.14-16.67; Figure 30). Serial patterning of residuals (not shown) was reduced from the simple linear regression; early periods of consistently positive residuals during 1980-1984 and negative residuals 1964-1972 were still present, but residuals appeared random after that. It is possible that the remaining, early pattern of residuals was related to the effect of the AMO on age 0 relative abundance. The intercept during the more regulated period (2006-2021) was 78% higher than the less regulated period (1964-2005), suggesting that recruitment from age 0 abundance to ages 1+ biomass two years later had increased concurrently with regulatory intensity.

Striped Bass condition dynamics – Estimates of P0 from CBEF data were 0.37 in summer (N = 2,921; the FWHP summer estimate was 0.38, N = 721), 0.51 in fall (N = 1,866), and 0.08 in winter (N = 808; Table 4). The ratios of Menhaden consumed per Striped Bass examined were 0.02 in summer, 0.64 in fall, and 0.48 in winter; these ratios were not synchronous with seasonal condition. When multiplied by mean weight of intact Menhaden (32.8 gm in fall, 59.2 gm in winter, and 86.9 gm in summer), the ratio as weight of Menhaden consumed per Striped Bass was 21.0 gm per Striped Bass in fall, 28.4 in winter, and 1.7 in summer; these ratios are relative and not absolute estimates (Table 4). The fall ratio would have consisted of nearly all age 0 Menhaden, while the winter and summer ratios consisted of both age 0 and ages 1+ (i.e., larger and heavier Menhaden comprised a greater percentage of weight of consumed Menhaden).

Uphoff et al. (2018) determined that P0 estimates for resident Striped Bass from CBEF collections in fall were strongly and linearly related to those estimated from FWHP collections during 2006-2014 (r^2 = 0.79, P = 0.001; Uphoff et al. 2018). The intercept of this relationship was not different from zero and the slope was not different than one, indicating 1:1 correspondence between the estimates (Uphoff et al. 2018). Interpretation of whether body fat was present or not was the same between FWHP and the CBEF citizen science surveys in fall and summer. Presumably, this would not change in winter.

Sample sizes were sufficient for precise monthly estimates during June-February, 2006-2015, when pooled across years (CV range of 2-26% and N range of 127-1,083; Table 5). Estimates of P0 were near 0.20 in June-July and then increased (i.e., condition worsened) to 0.51 in August and 0.68 during September-October. Estimated P0 dropped to 0.28 in November, 0.15 in December, and reached a nadir at 0.05-0.06 during January-February (Table 5).

Uphoff et al. (2017) found that condition of Striped Bass in summer was strongly related to condition in the following fall (1999-2012, Weibull function, approximate R^2 = 0.75, P < 0.0001; Figure 31). Condition in fall of the previous year was linearly related to condition in the next fall during 1998-2021 (r^2 = 0.70, P < 0.0001; Figure 32). These analyses indicated that previous condition and feeding at least a year earlier may have influenced condition in fall.

Summer preceding fall may be particularly stressful and potentially lethal for resident Striped Bass. Poor condition contributes to this stress. Summer represented a period with limited forage leading to no or negative growth in weight for ages 3-6 (Hartman and Brandt 1995), higher mortality of diseased and healthy Striped Bass (Groner et al. 2018), hypoxia and temperature stress (Constantini et a. 2008; Maryland Sea Grant 2009; Coutant 2013; LaPointe et al. 2014; Kraus et al. 2015; Itakura et al. 2021), and high catch-and-release mortality (Lukacovic and Uphoff 2007).

Seasonal and monthly estimates of P0 did not directly follow prey availability described in Chesapeake Bay diet and bioenergetic studies but seemed to lag behind. Prey fish availability is low during spring and early summer for ages 3+ Striped Bass (Hartman and Brandt 1995; Overton et al. 2009). Based on bioenergetics modeling, prey demand during 1990-1992 by Striped Bass ages 3-6 was equal to supply during winter (days 330-365 and 1-60, approximately December-February), but demand was greater than supply for the rest of the year (Hartman and Brandt 1995). Major prey levels were moderate and Striped Bass abundance was low during 1990-1992; Uphoff et al. 2022). Striped Bass in the Bay fed heavily on Menhaden during fall and winter (Hartman and Brandt 1995; Griffin and Margraf 2003; Walter et al. 2003; Overton et al. 2009; Overton et al. 2015; Buchheister and Houde 2016; Uphoff et al. 2022).

Lipids are the source of metabolic energy for growth, reproduction, and swimming for fish and these energy reserves relate strongly to foraging success, reproductive success, potential prey density, habitat conditions, environmental stressors, and subsequent fish health and survival (Tocher 2003; Jacobs et al. 2013). Poor condition is a common problem for Striped Bass in lakes when prey supply is inadequate (Axon and Whitehurst 1985; Matthews et al. 1988; Cyterski and Ney 2005; Raborn et al. 2007; Sutton et al. 2013; Wilson et al. 2013).

Menhaden consumption in fall and winter is likely important to resident Striped Bass because much of the latter's annual growth and gonadal development occur then (Hartman and Brandt 1995; Walter and Austin 2003; Jacobs et al. 2013). Mature Striped Bass have to trade-off energetic costs of reproduction with growth and survival after the spring spawning period. Resident Striped Bass may be seen as capital breeders investing heavily in spring spawning that is followed later by depleted energy reserves, increased risk of starvation, predation, and susceptibility to disease during mid-summer through early fall to the potential detriment of survival (Secor 2007; Regular et al. 2022). Population die-offs would be expected when a stock is limited by resources (Regular et al. 2022). Because energy limitations are first experienced by individuals, these die-offs may be signaled by the systemic deterioration of body condition of large portions of the individuals in a population (Dutil and Lambert, 2000; Casini et al. 2016; Regular et al. 2022).

Multiple lines of evidence suggest that survival of resident Striped Bass decreased in Chesapeake Bay due to higher natural mortality since the late 1990s. Conventional tag-based estimates of survival of 457-711 mm of Striped Bass in Chesapeake Bay decreased from 77% during 1987-1996 to 44% during 1997-2017, a 43% reduction (based on Table B8.25 in NEFSC 2019); tag-based estimates of F in Chesapeake Bay have been low and estimates of M have been high. Reporting rate changes could have been behind the declining estimates (NEFSC 2019). Secor et al. (2020) implanted a size-stratified sample of Potomac River Striped Bass with acoustic transmitters and recorded their migration and mortality during 2014-2018 using telemetry receivers throughout the Mid-Atlantic Bight and Southern New England. These transmitters would not be susceptible to reporting rate assumptions. Analysis of the last day of transmission indicated that Chesapeake Bay resident Striped Bass experienced lower survival (30% per year) than coastal emigrants (63% per year; Secor et al. 2020).

Long-term analyses of M based on conventional tags indicated survival of large Striped Bass decreased after stock recovery (NEFSC 2019). However, the time blocks analyzed were large and only differentiated pre- and post-1997 periods, the former of low M and latter of high M. A finer temporal resolution of M estimates is needed to relate forage, condition, or other factors to survival of large fish.

Mortality of fish due to starvation 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). Decreased survival of resident Striped Bass estimated from conventional tags during 1987-1996 and 1997-2017 in NEFSC (2019) was attributed to mycobacteriosis. Mycobacteriosis in isolation would not necessarily be the only source of increased M of Chesapeake Bay Striped Bass. Jacobs et al. (2009b) were able to experimentally link the progression of mycobacterial disease in Striped Bass to their diet; inadequate diet led to more severe disease progression and rapid mortality compared with a higher ration. Abundant individuals competing for limited prey may hinder one another's feeding activities, leading directly to starvation (Yodzis 1994). Shifts from high survival during 1987-1996 to lower survival afterwards (Kahn and Crecco 2006; Jiang et al. 2007; NEFSC 2013; NEFSC 2019) lagged two years behind downward shifts in major forage-to-Striped Bass ratios in Chesapeake Bay (major forage are Ages 0 and 1+Atlantic Menhaden, Bay Anchovy, Spot, and Blue Crab; Uphoff et al. 2022).

With high size limits and low fishing mortality in place since restoration, intraspecific competition for limited forage should be greater for smaller Striped Bass since they compete with one another and larger fish. All things being equal, larger Striped Bass should forage more efficiently and outcompete smaller fish through greater vision, swimming speed, and experience

(Ward et al. 2006). After the mycobacteriosis epizootic became established in Striped Bass in Chesapeake Bay in the late 1990s, prevalence increased with age and growth, potentially dampening competitive advantages for larger, older fish if nutritional challenges were severe enough (Gauthier et al. 2008; Jacobs et al. 2009b; Latour et al. 2012). Condition of Striped Bass smaller than 457 mm TL (small Striped Bass) in fall has transitioned from consistently poor during 1998-2007 to a mix of at or near target P0 interspersed with scattered years of poor P0 afterward (Uphoff et al. 2022). Small Striped Bass were at the target level of condition (P0 < 0.30) during 2008, 2015, 2017, and 2021 and were in poorest condition during 1998-2007, 2011-2012, 2016, and 2019. Age 0 Menhaden may be too large in fall for small Striped Bass to feed on them effectively in some years (Uphoff et al. 2022).

A hypoxia-based hypothesis has linked increased M and deteriorating condition of resident Striped Bass in Chesapeake Bay through a temperature-oxygen squeeze (Coutant 1985; Price et al. 1985; Coutant 1990; Coutant 2013). A temperature-oxygen squeeze is a mismatch of water column regions of desirable temperature and dissolved oxygen in stratified waters during summer. This hypothesis was originally formed by Coutant (1985; 1990; 2013) to explain die-offs of large (10-20 kg) trophy sized adult Striped Bass in southeastern reservoirs and this particular hypothesis has often been advanced for Chesapeake Bay. These trophy fish were considerably larger than fish comprising Maryland's resident Striped Bass fishery. Constantini et al. (2008), Kraus et al. (2015), and Itakura et al. (2021) examined the impact of hypoxia on 2-year-old and older Striped Bass in Chesapeake Bay through bioenergetics modeling and acoustic tagging and concluded that a temperature-oxygen squeeze by itself was not limiting for Striped Bass. Coutant (2013) modified his original temperature-oxygen squeeze hypothesis to reflect additional studies and experience, but this has not been widely recognized by Bay researchers. Tolerance of warm water was influenced by Striped Bass size and-or age (2-4 kg fish, more aligned with Maryland's resident fish, were more tolerant than trophy sized), duration of exposure, quantity of food available, and stress from catch-and-release (Coutant 2013). Groner et al (2018) suggested that Striped Bass were living at their maximum thermal tolerance and that this has driven increased mycobacteriosis and associated mortality. Adequate levels of Striped Bass prey can offset negative effects of warm temperatures and suboptimal dissolved oxygen in reservoirs (Thompson et al. 2010; Coutant 2013). Condition of resident Striped Bass in fall has improved since the mid-2000s in concert with improvement in Atlantic Menhaden to Striped Bass ratios (age 0 and ages 1+) and consumption (Uphoff et al. 2022); temperature and oxygen conditions have not improved (T. Parham, Resource Assessment Service, MD DNR, personal communication). Management to balance abundance of prey with Striped Bass may represent a strategy to offset the effect of increased warming of the Bay in the future.

Extensive research (laboratory, field studies, and stock assessment modeling) on the links between forage, condition, and M have been conducted for some stocks of Atlantic Cod that provide a narrative that seems to apply well to Striped Bass in the Bay. Similar to resident Striped Bass, these stocks experienced forage fish declines, followed by declining condition and increased M; starvation caused declines in energy reserves, physiological condition, and enzyme activity (Lilly 1994; Lambert and Dutil 1997; Dutil and Lambert 2000; Shelton and Lilly 2000; Rose and O'Driscoll 2002). Dutil and Lambert (2000) found that the response of M of Atlantic Cod could be delayed after unfavorable conditions. Condition has been used to estimate time-varying M in Baltic Sea and Canadian Atlantic Cod stock assessments (Casini et al. 2016; Regular et al. 2022). Recovery of the northern stock of Atlantic Cod has paralleled recovery of

Capelin *Mallotus villosus*, its main prey; increases in size composition and fish condition and apparent declines in mortality followed increased Capelin abundance (Rose and Rowe 2015).

Relative F and condition - The scatter plot of P0 and relative F suggested a sharp increase in P0 (worsening condition) as relative F rose from 0.75 to 1.32 (Figure 33). An asymptote formed at relative F higher than 1.32. We chose the point where relative F equaled 1.59 and P0 = 0.78 as the location of the slope shift coefficient. At this point, values of P0 indicating poor conditions (P0 ≥ 0.70) were established (one point was to the left of the remaining five points at that level). The piecewise model fit the estimates of P0 and relative F well ($R^2 = 0.68$, $P < 0.0001$; Figure 37). Means and standard errors for the model terms were the following: intercept (b) = 0.944, SE = 0.425; indicator variable coefficient (v) = -1.422, SE = 0.473; maximum slope (m_1) = 0.814, SE = 0.264; and the slope at consistently poor condition (m_2) = -0.097, SE = 0.185. These terms were then used to estimate relationships of P0 and relative F for F ≥1.59 (v • 1) when relative F was < 1.59 (v • 0). When relative F was < 1.59, the relationship was described by the equation:

(20) P0 = (0.717 • relative F) – 0.478 (Figure 33).

When F was ≥ 1.59,

(21) P0 = (-0.097• relative F) + 0.944 (Figure 33).

Since the slope in equation 21 was not different than 0, an alternative would be to use equation 20 up to where it predicts that P0 is at the boundary for poor conditions (P0 = 0.70 at relative F = 1.64) and then assume that the slope is 0 after that. Equation 20 predicted that P0 would meet the good boundary condition (P0 = 0.30) when relative F was 1.08. Boundaries of relative F estimated from the 25th percentile and median were 1.90 and 1.16, respectively.

The relationship of Relative F with P0 may also reflect that the PRFC index is inversely correlated with Relative F (r = -0.73, P < 0.0001); the PRFC index is the denominator for relative F, so it is present in both variables. Ricker (1975) cautioned against using strongly correlated variables as independent terms and suggest that r ≥ 0.80 or ≤ -0.80 be considered strong enough that only one variable should be chosen. The correlation found between relative F and the PRFC index does not meet the strong correlation criterion of Ricker (1975) but is close enough to it that lack of independence between the two variables should be considered. However, it is logical that high F on forage fish would eventually result in low forage fish biomass that could ultimately result in poor predator condition and vice-versa.

Landings, the numerator of relative F, were weakly correlated with the PRFC index (r = 0.25, P = 0.063) and relative F (r = 0.26, P = 0.048). These low correlations with landings reflect high harvest during 1970-2005 regardless of whether the PRFC index and relative F were high or low. Hyperstability of landings has been found when catchability increased while the stock declined (Hilborn and Walters 1992). Schaaf (1975) determined that catchability in the Atlantic Menhaden purse seine fishery varied inversely with population size (high catchability at low abundance that decreased nonlinearly to an asymptote as abundance increased). Catchability inversely related to stock size has been a nearly universal feature of clupeid and purse seine fisheries (Saetersdal 1980; Saville and Bailey 1980; Crecco and Savoy 1985; Gulland 1983; Hilborn and Walters 1992).

The 75th percentile of relative F (1.90) is a good bit higher than the location of the slope shift coefficient used in equations 20 and 21 (1.59) or the alternative estimate when equation 20 is used up to where it predicts that P0 is at the boundary for poor conditions (P0 = 0.70 at relative F = 1.64) and then we assume that the slope is 0. The use of relative F = 1.59 would pull one more year (1999) into the red category and another (2005) would be just below the red boundary. An additional year would not be over the red boundary if relative F = 1.64 was chosen. There was little practical difference in relative F between the median and the prediction for P0 = 0.30.

Exact predictions from the equations describing the relationship of P0 and relative F may be subject to biases due to lagged responses (P0 may be responding to the level in previous seasons up to at least the previous year), consumption of other food items, including age 0 Menhaden, and unaccounted for environmental phenomena that may be exerting an impact that is interpreted as the effect of fishing. None-the-less, this analysis supports the hypothesis that Menhaden harvest intensity impacts Striped Bass condition in Maryland's portion of Chesapeake Bay.

Index of Menhaden availability and Striped Bass dynamics – The AS for age 0 or ages 1+ Menhaden and P0 time-series trended in opposite directions, indicating that generally condition was improving as availability went up (Figure 34). A linear regression indicated a modest relationship of availability of resident Striped Bass on ages 1+ Menhaden and condition in the following fall ($r^2 = 0.27$, $P = 0.019$; Figure 35). The relationship was described by the equation:

$$(22) \qquad P0 = (-2.14 \bullet AS) + 0.52.$$

Standard errors for the slope and intercept were 0.83 and 0.12, respectively. The modest amount of variation accounted for indicated potential for other prey, predators, and environmental conditions to influence P0 as well.

Availability of ages 1+ Menhaden declined with relative F (Figure 36). The relationship of AS on ages 1+ Menhaden to relative F was described by the equation:

$$(23) \qquad AS = (-0.067 \bullet \text{relative F}) + 0.22, \ (r^2 = 0.50, P < 0.00016).$$

Standard errors for the slope and intercept were 0.015 and 0.024, respectively.

A linear regression indicated availability age 0 Menhaden exerted a modest influence on condition in fall ($r^2 = 0.32$, $P = 0.004$; Figure 37). The relationship was described by the equation:

$$(24) \qquad P0 = (-1.88 \bullet AS) + 0.67.$$

Standard errors for the slope and intercept were 0.58 and 0.10, respectively. The modest amount of variation accounted for indicated potential for previous feeding history and condition, other prey, predators, and environmental conditions to influence P0 as well.

A general recommendation for stock assessments is that information only be used once (Cotter et al. 2004). The PRFC index is contained in the numerator of AS on ages 1+ Menhaden and in the denominator of relative F. However, direct dependency of these indices on the PRFC index was altered by the denominator in each ratio. Exact predictions from the equations describing the relationship of AS with P0 and relative F with AS are subject to the same caveats as listed for P0 and relative F. The equations may be subject to biases due to lagged responses (P0 may be responding to the level in previous seasons up to at least the previous year), consumption of other food items, interference from other predators, and poorly understood environmental phenomena.

In general, supplemental indices and statistical analyses supported the cohesive trends of TLI metrics. Supplemental Menhaden indices indicated trends of core indices were widespread (Maryland and Potomac River). Trends in Virginia's portion of the Bay were not examined in this initial stage of the TLI; however, Buccheister et al. (2016) found that Chesapeake Bay juvenile indices for Maryland and Virginia were strongly correlated.

Other Bay Menhaden information not included in the TLI - The Maryland Department of Natural Resources has conducted summer pound net sampling of Atlantic Menhaden since 2005 and has conducted a fishery-independent gill net survey in the lower Choptank River since 2013 (Rickabaugh and Messer 2021). These time-series are too short to be included in the TLI but length and age frequency, and survey CPUE summaries are available. The data collected from these efforts provide information for stock assessments and fishery management plans by MD DNR and ASMFC. Federal Aid to Sportfishing reports for Project F-61 contain this information and are available through the Reports section of the Striped Bass Program tab of the Fishing and Boating Services webpage. The latest report on Menhaden monitoring (Rickabaugh and Messer 2021) can be accessed at https://dnr.maryland.gov/fisheries/Documents/Final%20Report%20F-61-R-16%202020-2021.pdf. Atlantic Menhaden monitoring summaries are found in Project 2, Job 2: *Stock assessment of selected recreationally important adult migratory finfish in Maryland's Chesapeake Bay*.

Appendices – A memo describing the PRFC index is Appendix 1. Values of core and supplemental indices are presented in three appendices. Appendix 2 presents values of Atlantic Menhaden core indices. Appendix 3 presents values of Striped Bass core indices. Appendix 4 presents values of supplemental indices. All of these data are depicted in report figures as well.

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Table 1. Important abbreviations.

Abbreviation	Definition
AIC	Akaike Information Criteria
AS	Index of Menhaden availability; ratio of a Menhaden index to biomass of age 3-6 Striped Bass
@Risk	Software for fitting and simulating distributions
ASMFC	Atlantic States Marine Fisheries Commission
B	Biomass (population weight)
BAM	Beaufort Assessment Model
CBEF	Chesapeake Bay Ecological Foundation
C	Reported catch (weight)
CI	Confidence interval
CPUE	Catch per unit of effort
CV	Coefficient of variation
F	Instantaneous annual fishing mortality rate
Fb	Biomass based F derived from BAM estimates of biomass and landings
FWHP	Fish and Wildlife Health Program (DNR)
GLM	Generalized linear model
GLLM	Generalized linear mixed model
GM	Geometric mean
H	Ostensible Menhaden landings (weight) estimated for the Bay
I	Index
JI	Juvenile index of relative abundance of a species
M	Instantaneous annual natural mortality rate
MD DNR	Maryland Department of Natural Resources
MRIP	Marine Recreational Information Program
MT	Metric tons
N	Abundance (assessment) or sample size (statistics)
NB	Negative binomial distribution
NMFS	National Marine Fisheries Service
P	Level of significance
P0	Proportion of Striped Bass without visible body fat
PDT	ASMFC Atlantic Menhaden Plan Development Team
PRFC	Potomac River Fisheries Commission
r	Correlation coefficient
r^2	Regression coefficient
SD	Standard deviation
SE	Standard error
t	Time, year
t-1	Year - 1; previous year
YOY	Young-of-year; age 0 fish

Table 2. Summary of the core and supplemental indices: their location, metric represented, parameter represented, gear type, data type (fishery independent or dependent), and agency source. Shading is used to provide contrast for type of parameter. SA = stock assessment. See Table 1 for other abbreviations.

Species	Parameter	Metric	Type	Location	Gear	Type	Source
Menhaden	Age 1+ B	PRFC MT / net day (PRFC index)	Core	Potomac River	Pound net	FD	PRFC
Menhaden	Age 1+ B	MD MT/net month	Supplement	MD Chesapeake	Pound net	FD	MD DNR
Menhaden	Age 1+ N	GLMM N/1,000 square yard hour	Supplement	Head-of Bay	Experimental gill net	FI	MD DNR
Menhaden	Age 0 N	JI, geometric mean (GM JI)	Core	MD Chesapeake	Seine	FI	MD DNR
Menhaden	Age 0 N	JI, GLM neg binomial	Supplement	MD Chesapeake	Seine	FI	ASMFC
Menhaden	Relative F	Ostensible harvest /PRFC index	Core	MD Chesapeake	All Bay harvest	FD	ASMFC, PRFC, VMRC, MD DNR
Striped Bass	Age 0 prey per predator	Menhaden GM JI per bass biomass	Core	MD Chesapeake	JI, SA	FI, SA	MD DNR, ASFMC
Striped Bass	Ages 1+ prey per predator	PRFC index per bass biomass	Core	MD Chesapeake	Pound net, SA	FD, SA	PRFC, ASMFC
Striped Bass	Condition	Proportion without visible body fat	Core	MD Chesapeake	Fish Health survey	FI	MD DNR

Table 3. Estimated landings (MT) from Chesapeake Bay, by fishery and source. Old NMFS bait MT = the online version of federal commercial landings available on the NMFS website in the early 2000s.

Year	Reduction landings MT	Bay cap MT	Old NMFS bait MT	PDT bait MT	Total MT
1964	5,733		20,976		26,729
1965	69,679		25,871		95,550
1966	62,907		15,572		78,479
1967	44,934		11,118		56,052
1968	62,235		9,497		71,732
1969	30,958		10,002		40,960
1970	89,799		21,850		111,649
1971	121,750		12,924		134,674
1972	178,185		10,742		188,927
1973	154,227		17,098		171,325
1974	108,321		16,539		124,860
1975	85,181		24,174		109,355
1976	117,806		22,639		140,445
1977	131,090		30,732		161,822
1978	121,021		26,920		147,941
1979	104,542		19,088		123,630
1980	177,157		28,470		205,627
1981	31,393		28,214		59,607
1982	145,576		22,156		167,732
1983	150,532		23,487		174,019
1984	98,287		14,835		113,122
1985	142,166		17,873	17,916	160,082
1986	124,307		11,892	11,943	136,250
1987	175,530		15,025	15,088	190,618
1988	141,910		14,272	29,346	171,256
1989	155,707		17,589	17,620	173,327
1990	151,184		11,082	11,082	162,266
1991	160,015		10,625	10,625	170,640
1992	154,443		10,475	10,475	164,918
1993	168,046		7,888	4,884	172,930
1994	128,479		6,681	4,211	132,690
1995	171,092		7,910	5,610	176,702
1996	152,554		6,545	4,906	157,460
1997	136,028		6,581	4,750	140,778
1998	135,276		4,856	21,313	156,589

1999	104,976		6,451	20,597	125,573
2000	80,919		6,881	17,859	98,778
2001	127,300		5,253	21,587	148,887
2002	99,300		4,971	22,937	122,237
2003	124,100			24,409	148,509
2004	98,000			23,816	121,816
2005	98,000			28,291	126,291
2006	65,000			15,070	80,070
2007	85,344			21,831	107,175
2008	84,562			21,175	105,737
2009		109,000		19,641	128,641
2010		109,000		16,598	125,598
2011		109,000		17,085	126,085
2012		109,000		23,628	132,628
2013		87,000		17,020	104,020
2014		87,000		20,003	107,003
2015		87,000		20,239	107,239
2016		87,000		17,877	104,877
2017		87,000		16,979	103,979
2018		87,000		16,356	103,356
2019		87,000		18,606	105,606
2020	65,000			17,000	82,000
2021		51,000			

Table 4. Seasonal body fat summary, 2006-2015 pooled. Menhaden/Bass is the ratio of the number of identifiable Menhaden in resident Striped Bass guts to the number of resident Striped Bass examined. Gm Menhaden/Bass is the ratio of grams of identifiable Menhaden per Striped Bass examined. Summer = June-September; fall = October-November; and winter = December-March. Striped Bass were 457-711 mm, TL.

Collector	FWHP	CBEF	CBEF	CBEF
Season	Summer	Summer	Fall	Winter
P0	0.377	0.374	0.513	0.075
SD	0.018	0.009	0.012	0.009
Up 90% CI	0.407	0.388	0.532	0.091
Low 90% CI	0.347	0.359	0.494	0.060
N	721	2921	1866	808
Years	2007-2012	2007-2015	2007-2014	2007-2015
Proportion male		0.84	0.80	0.88
Menhaden/Bass		0.022	0.641	0.481
Grams Menhaden/Bass		1.7	21.0	28.4

Table 5. Monthly body summary of proportion Striped Bass 457-711 mm TL, without body fat (P0), 2006-2015 pooled. N is the sample size; SD is the standard deviation; Up 90% is the upper 90% confidence interval; Low 90% is the lower 90% confidence interval; and CV is the coefficient of variation.

Month	N	P0	SD	Up 90%	Low 90%	CV
June	852	0.173	0.013	0.194	0.151	0.075
July	765	0.222	0.015	0.247	0.198	0.067
August	645	0.507	0.020	0.539	0.475	0.039
September	659	0.678	0.018	0.708	0.649	0.027
October	1083	0.683	0.014	0.707	0.660	0.021
November	782	0.276	0.016	0.303	0.250	0.058
December	127	0.150	0.032	0.202	0.098	0.212
January	256	0.055	0.014	0.078	0.031	0.260
February	374	0.061	0.012	0.082	0.041	0.202
March	13	0.154	0.100	0.318	-0.011	0.650

Appendix 1. Potomac River Fisheries Commission Pound Net Catch per Effort Estimates Report to the ASMFC Ecological Reference Point Workgroup

Jim Uphoff

January 24, 2019

The fishery-dependent Potomac River Fisheries Commission (PRFC) pound net catch per effort (CPUE) time-series extends back to 1964. It is based on annual aggregated catch and effort, and it was used as a biomass index in the 2004 and 2011 Atlantic Menhaden stock assessments (appendices 1A and 1B contain the methods descriptions copied and pasted from the assessments). This was the only index for age 1+ Menhaden used in those assessments. It was not used after fishery-independent indices (FI) were developed for the 2015 assessment (SEDAR 2015). Fishery-dependent (FD) indices were considered for the Beaufort Assessment Model used in SEDAR (2015) but were rejected because of the requirement for individual trip records that resulted in shorter FD time-series than available for FI indices, FD indices lacked length and age data needed for age-based modeling, and they were correlated with FI indices (SEDAR 2015).

Two approaches being considered for developing Ecological Reference Points (ERPs), Time-Varying r and Steele-Henderson models, are biomass dynamic model based and require a long-term biomass index. The need for a long-term index has revived of interest in PRFC CPUE.

Annual Potomac River pound net catches with effort indicators were available for 1964-2018 from the PRFC. During 1964-1993, PRFC required a license for each pound net and did not restrict number of pound net licenses sold. Since pound nets were expensive and labor intensive to fish, it seemed reasonable to assume that each licensee would maintain stable fishing practices, so that number of licenses approximated effort during this period. Potomac River pound net catch per license (PRFC FD1) was estimated for each year during 1964-1993 as annual catch divided by number of licenses. After 1993, licenses were capped at 100 and this estimator may have stopped representing effort in the same manner as before the cap (fishermen may have bought more licenses than needed to keep from being excluded from fishing). Prior to the imposition of the cap, licenses had steadily fallen by half (to 72 between 1985 and 1993).

After the cap was imposed, 100 licenses were issued every year, but that did not mean 100 were fished.

Catch per net day fished estimates (PRFC FD2) were also available, but discontinuous (1976-1980 and 1988-2018). Previous single-species stock assessment (ASMFC 2004; 2011; see appendices) used a regression approach to combine these two PRFC FD indices into a single catch per net day time-series stretching back to 1964. Predictions of missing net day effort from a linear regression with licenses during the period prior to imposition of the license cap, i.e., data from years where both estimates were available (1976-1980 and 1988-1994) were used to fill in missing catch per net estimates back to 1964 when only license data were available. Even though the cap on licenses was imposed after 1993, 1994 had to be included in the original regression for sample size to equal 11. "Observed" estimates are used for the time-series and only missing values are filled in by predictions.

Two sources of indices exist, the time-series used for the 2010 assessment (A. Shueller, NMFS, personal communication) and spreadsheets with landings, license counts, and net day estimates from PRFC (A. C. Carpenter or Ellen Cosby, PRFC, personal communication). These spreadsheets have been provided to the ERP workgoup. **Unfortunately, the estimates of catch per net day (FD2) prior to 1995 provided from the 2011 assessment do not match estimates made by the same method based on information in the PRFC spreadsheets**. The regression equations and reported fit are different (Table 1). A third option for backfilling CPUE estimates is available, estimating FD2 (C/net d in Table 1) from FD1 (C/L) for years where both are available, also exists and the linear regression has a much stronger fit than for licenses alone (Table 1).

Table 1. Comparison of regressions used to estimate missing catch per net day effort estimates to create a continuous set of estimates starting in 1964. Weight units for catch per license vs Catch per net day (fourth column) are metric tons. N = 11 for all regressions.

Analysis	Lic vs net days	Lic vs net days	C/L vs C/net d
Source	2011 assessment	PRFC spreadsheets	PRFC spreadsheets
Predictor	Licenses	Licenses	MT / License
Predicted	Net days	Net days	MT / Net day
Slope	17.944	19.214	0.0245
Slope SE		6.340	0.0026
Intercept	3094.2	2794.5	-0.1349
Int SE		675.8	0.1236
r^2	0.485	0.505	0.909
P	0.104*	0.014	<0.0001

* I'm wondering if these numbers get superimposed since the fit is fairly close to the recreation based on the PRFC spreadsheets, but the P values are quite different.

Overall, differences among MT / net day for the observed and predicted time-series are not great, although there are a few years in the of where predictions of FD2 from FD1 (fourth column) are noticeably higher (1973-1975 and 1987-1988; Figure 1; Table 2). The PRFC spreadsheet-based recreation of the method used in the 2003 and 2011 stock assessments (third column) comes very close to recreating the estimates from the 2011 assessment and the underlying data they are based on is readily available (Figure 1; Table 2). Unless someone can find the underlying data used to make estimates for the 2011 assessment, the PRFC spreadsheet-based estimates may be a more transparent proxy for the "old" PRFC time-series.

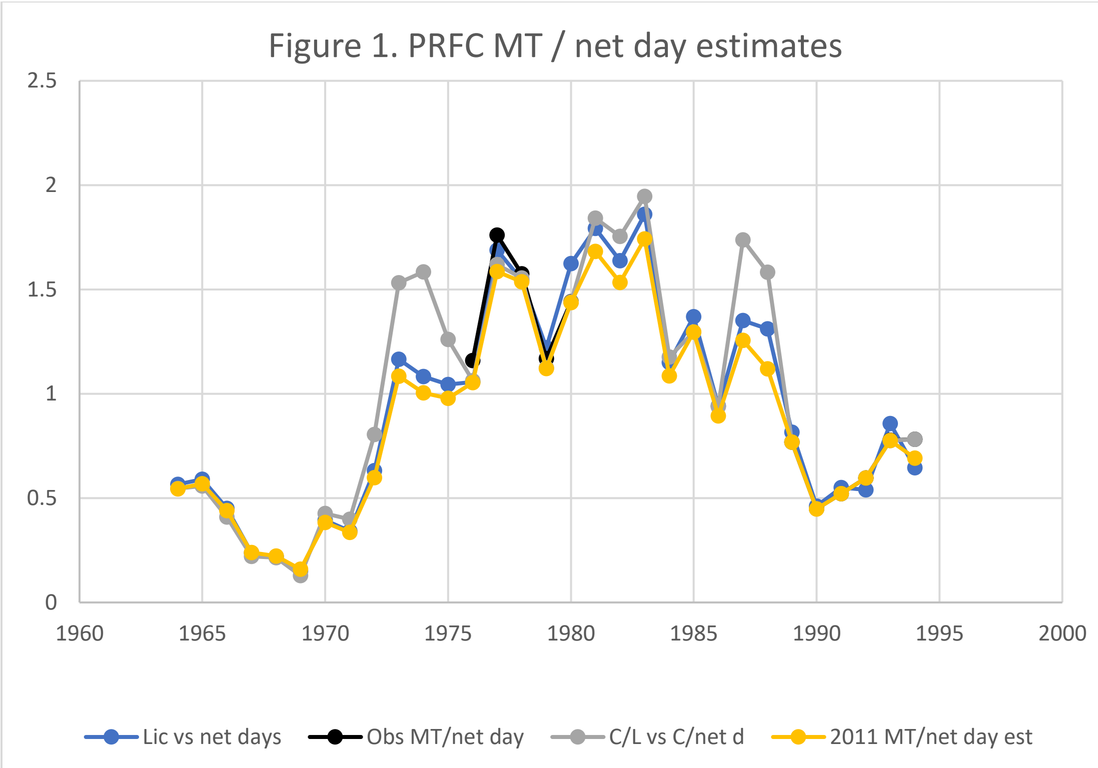
Figure 1. PRFC MT / net day estimates
2.5
2
1.5
1
0.5
0
1960
1965
1970
1975
1980
1985
1990
1995
2000
Lic vs net days
Obs MT/net day
C/L vs C/net d
2011 MT/net day est

Table 2. Time-series of PRFC pound net catch per net day. Old PRFC MT / net day = estimates from the 2011 stock assessment plus updated estimates from PRFC. New PRFC MT / net day1 = time-series based on PRFC spreadsheets that recreate the regression used for "old" estimates. New PRFC MT / net day2 is a time-series based on a MT / license versus MT / net day regression.

Year	Old PRFC MT / net day	New PRFC MT / net day1	New PRFC MT / net day2
1964	0.54	0.57	0.55
1965	0.57	0.59	0.56
1966	0.44	0.45	0.41
1967	0.24	0.24	0.22
1968	0.22	0.22	0.21
1969	0.16	0.15	0.13
1970	0.38	0.39	0.43
1971	0.34	0.34	0.40
1972	0.60	0.63	0.81
1973	1.08	1.16	1.53
1974	1.01	1.08	1.58
1975	0.98	1.04	1.26
1976	1.05	1.06	1.06
1977	1.59	1.69	1.62
1978	1.54	1.55	1.55
1979	1.12	1.22	1.12
1980	1.44	1.62	1.44
1981	1.68	1.79	1.84
1982	1.53	1.64	1.75
1983	1.74	1.86	1.95
1984	1.09	1.15	1.18
1985	1.30	1.37	1.30
1986	0.89	0.94	0.94
1987	1.26	1.35	1.74
1988	1.12	1.31	1.58
1989	0.77	0.82	0.77
1990	0.45	0.46	0.45
1991	0.52	0.55	0.52
1992	0.60	0.54	0.60
1993	0.78	0.86	0.78
1994	0.69	0.65	0.78
1995	0.70	0.70	0.70
1996	0.67	0.67	0.67
1997	0.66	0.66	0.66
1998	0.52	0.52	0.52

Year	Old PRFC MT / net / day	New PRFC MT / net day1	New PRFC MT / net day2
1999	0.74	0.74	0.74
2000	0.84	0.84	0.84
2001	0.58	0.58	0.58
2002	0.51	0.51	0.51
2003	0.48	0.48	0.48
2004	1.00	1.00	1.00
2005	0.85	0.85	0.85
2006	0.75	0.75	0.75
2007	1.15	1.15	1.15
2008	1.20	1.20	1.20
2009	0.91	0.91	0.91
2010	1.04	1.04	1.04
2011	1.13	1.13	1.13
2012	1.34	1.34	1.34
2013	0.99	0.99	0.99
2014	0.98	0.98	0.98
2015	0.98	0.98	0.98
2016	1.10	1.10	1.10
2017	0.92	0.92	0.92

Appendix 1A

4.2.4 Commercial Bait Catch Rates (CPUE) – 2011 Assessment

Pound net landings collected by the Potomac River Fisheries Commission (PRFC) were used to develop two fishery-dependent indices of relative abundance for adult menhaden. The pound net is a stationary presumably nonselective fishing gear that is used to harvest fishes in the Potomac River of Chesapeake Bay, including menhaden primarily aged-1 through 3 years. Other than the reduction landings, these data represent the only other available information that can be used to infer changes in relative abundance of adult menhaden along the east coast of the U.S. The first catch-per-unit-effort (CPUE) index was calculated as annual ratios of total pounds landed to total pound net days fished. Raw catch and effort data were available for the years 1976-1980 and 1988-2008. Recently, the PRFC was able to obtain and computerize more detailed data on pound net landings and effort, which allowed index values to be calculated for 1964-1975 and 1981-1987 (Carpenter 2005). To generate estimates of pound net landings (PN) for the missing years, a linear regression was fitted to annual PN and published landings (PB): $PN = 219035.8 + 0.953 \cdot PB$, which had an r^2 value of 0.996 and was highly significant ($p < 0.001$, $n = 26$). During 1964–1993, there were no restrictions on the number of licenses sold to fishers operating in the Potomac River, however after 1993, the number of licenses was capped at 100 (A. C. Carpenter, PRFC, personal communication). Therefore, to generate estimates of pound net days fished (DF) for the missing years, a second linear regression was fitted to DF as a function of the number of licenses (L): DF = 3094.2 +17.944 ·L , which had an r^2 value of 0.485 and was significant at an α-level of 0.104 (n = 11). The shorter period of overlap among DF and L and greater variability associated with the regression increases the uncertainty of the index for the reconstructed years, but not for the most recent years (1988–2008). This index was constructed in the same manner as those used for the 2003 and 2006 menhaden assessments, and it shows a variable trend over time with low values in the 1960s-1970s, peak values in the early 1980s, and intermediate values in recent years (Figure 4.13)

Appendix 1B

5.3.4 Biomass Indices Potomac River Fisheries Commission Dependent CPUE index- 2004 Assessment

Annual Potomac River pound net catches of menhaden and number of pound net licenses issued during 1964-2000 were available from the Potomac River Fisheries Commission (PRFC; A. C. Carpenter, PRFC, personal communication). Catch-per-unit-effort for each year was calculated as annual catch reported by all license holders divided by number of licenses. During 1964-1993, there were no restrictions on the number of licenses sold. After 1993, the number of licenses was capped at 100 (A. C. Carpenter, PRFC, personal communication). Pound net is a stationary nonselective fishing gear and it was believed to produce an index of relative abundance of menhaden (ages-0 through -5, primarily 1 through 3) in Potomac River and Chesapeake Bay (Fig. 5.13). The pound net CPUE, lagged 2-years, was highly positively correlated with the juvenile abundance seine indices from North Carolina, Virginia and Maryland, but negatively, although not significantly, with the seine indices from Connecticut and Rhode Island. This pattern is similar to the correlations among the seine indices between New England and the regions to the south.

Appendix 2. Core Indices for Atlantic Menhaden. Relative F scale is arbitrary and is based on biomass.

Species	Menhaden	Menhaden	Menhaden
Index	PRFC	Relative F	GM JI
Gear	Pound net		Haul seine
Ages	1+	1+	0
Type	Core	Core	Core
Unit	MT / net day	Instantaneous	GM catch per haul
1959			0.67
1960			0.60
1961			0.30
1962			2.34
1963			0.65
1964	0.54	0.48	0.25
1965	0.57	1.89	0.34
1966	0.44	2.31	0.32
1967	0.24	2.43	0.14
1968	0.22	3.76	0.31
1969	0.16	1.51	0.89
1970	0.38	3.11	0.16
1971	0.34	2.88	2.61
1972	0.60	2.25	2.76
1973	1.08	1.64	4.42
1974	1.01	1.26	11.34
1975	0.98	1.07	12.11
1976	1.06	1.05	16.67
1977	1.62	1.02	15.09
1978	1.55	1.10	4.81
1979	1.12	0.96	12.01
1980	1.44	1.32	8.64
1981	1.68	0.37	11.75
1982	1.53	1.02	2.83
1983	1.74	1.23	4.34
1984	1.09	0.95	4.64
1985	1.30	1.46	8.24
1986	0.89	1.27	7.61
1987	1.26	1.61	3.55
1988	1.12	1.81	5.90
1989	0.77	2.85	2.23
1990	0.45	3.35	4.68
1991	0.52	3.05	3.12
1992	0.60	2.40	1.78

1993	0.78	2.22	0.62
1994	0.78	1.79	1.21
1995	0.70	2.59	0.51
1996	0.67	2.38	0.53
1997	0.66	2.39	0.87
1998	0.52	2.49	0.43
1999	0.74	1.59	0.87
2000	0.84	1.39	0.67
2001	0.58	2.73	0.69
2002	0.51	2.47	0.28
2003	0.48	2.01	0.38
2004	1.00	1.32	0.32
2005	0.85	1.58	1.40
2006	0.75	0.84	0.62
2007	1.15	0.91	0.86
2008	1.20	1.00	0.93
2009	0.91	1.32	0.93
2010	1.04	1.16	0.96
2011	1.13	1.02	0.85
2012	1.34	1.14	1.11
2013	0.99	1.05	1.45
2014	0.98	1.09	1.20
2015	0.98	1.03	0.91
2016	1.10	1.04	1.30
2017	0.92	0.85	0.39
2018	1.51	0.81	1.24
2019	1.04	1.03	1.84
2020	1.02	0.76	1.28
2021	1.15		1.75

Appendix 3. Core Indices for Striped Bass.

Species	Menhaden / Bass	Menhaden / Bass	Striped Bass
Index	AS	AS	P0
Gear			
Ages	1+ / 3-6	0 / 3-6	4+
Type	Core	Core	Core
Unit	Biomass	JI / Biomass	Proportion without visible fat
1982	1.42	2.81	
1983	1.54	3.97	
1984	0.71	3.20	
1985	0.57	3.89	
1986	0.30	2.71	
1987	0.31	0.96	
1988	0.23	1.32	
1989	0.15	0.47	
1990	0.08	0.94	
1991	0.09	0.57	
1992	0.09	0.26	
1993	0.11	0.09	
1994	0.10	0.16	
1995	0.08	0.06	
1996	0.06	0.05	
1997	0.05	0.07	
1998	0.04	0.04	0.775
1999	0.06	0.07	0.819
2000	0.08	0.07	0.776
2001	0.06	0.07	0.770
2002	0.06	0.03	0.600
2003	0.07	0.05	0.697
2004	0.12	0.04	0.736
2005	0.11	0.19	0.556
2006	0.08	0.07	0.563
2007	0.13	0.10	0.466
2008	0.14	0.11	0.042
2009	0.11	0.11	0.243
2010	0.17	0.16	0.247
2011	0.22	0.17	0.465
2012	0.28	0.25	0.561
2013	0.18	0.30	0.480
2014	0.17	0.23	0.138
2015	0.16	0.17	0.081

2016	0.19	0.26	0.315
2017	0.15	0.07	0.171
2018		0.21	0.046
2019		0.28	0.067
2020		0.18	0.058
2021		0.24	0.011

Appendix 4. Supplemental indices.

Species	Menhaden	Menhaden	Menhaden
Index	MD	MD	Age 0 N
Gear	Pound net	Gill net	Seining survey
Ages	1+	1+	FI (optional)
Type	Supplement	Supplement	Supplement
Unit	MT / net month	N / 1,000 sq yd/hour	GLM negative binomial index
1959			59.4
1960			0.8
1961			0.4
1962			63.6
1963			22.5
1964			3.9
1965			0.9
1966			2.2
1967			0.7
1968			1.5
1969			15.6
1970			1.0
1971			92.0
1972			40.6
1973			230.5
1974			263.7
1975			420.7
1976			381.6
1977			359.0
1978			45.0
1979			152.9
1980	1.13		269.4
1981	0.86		235.3
1982	0.17		107.6
1983	1.20		82.6
1984	1.44		44.7
1985	1.93	1.24	139.0
1986		1.15	116.6
1987		1.88	53.0
1988		1.46	90.7
1989		0.13	34.7
1990	5.10	0.03	52.6
1991	4.95	0.01	38.8
1992	9.11	0.21	27.9

1993	4.15		1.9
1994	5.02	0.33	9.6
1995	4.04	0.26	2.9
1996	5.02		1.5
1997	5.66		5.2
1998	6.91		1.6
1999	6.63	0.08	14.3
2000	4.43	0.04	2.1
2001	3.71	0.07	0.4
2002	6.29	0.16	2.0
2003	4.74		0.4
2004	4.99		1.7
2005	1.81	0.04	8.3
2006	1.59	0.07	0.7
2007	0.96	0.04	2.3
2008	1.13	0.09	10.2
2009	1.35	0.28	1.7
2010	1.79	0.15	11.6
2011	1.86	0.01	2.6
2012	1.34	0.63	2.1
2013	1.22	0.42	2.9
2014	1.11	0.77	7.0
2015	0.86	0.50	0.6
2016	0.76	0.48	7.6
2017	0.98	0.06	0.6
2018	1.26	0.16	1.5
2019	1.12	0.01	5.2
2020	1.12	0.19	1.8
2021		0.20	18.4

Figure 1. Location of important geographic features of Maryland's portion of Chesapeake Bay mentioned in this report.

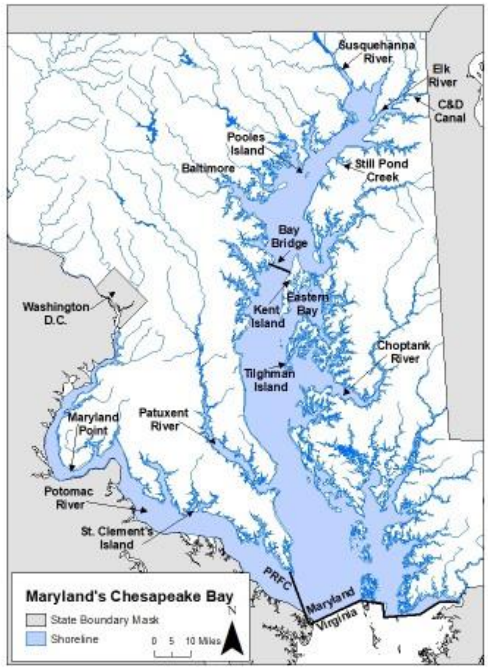

Figure 2. Plot of total lengths (TL) of Atlantic Menhaden consumed and resident Striped Bass (457-711 TL) that consumed them during winter, 2006-2015.

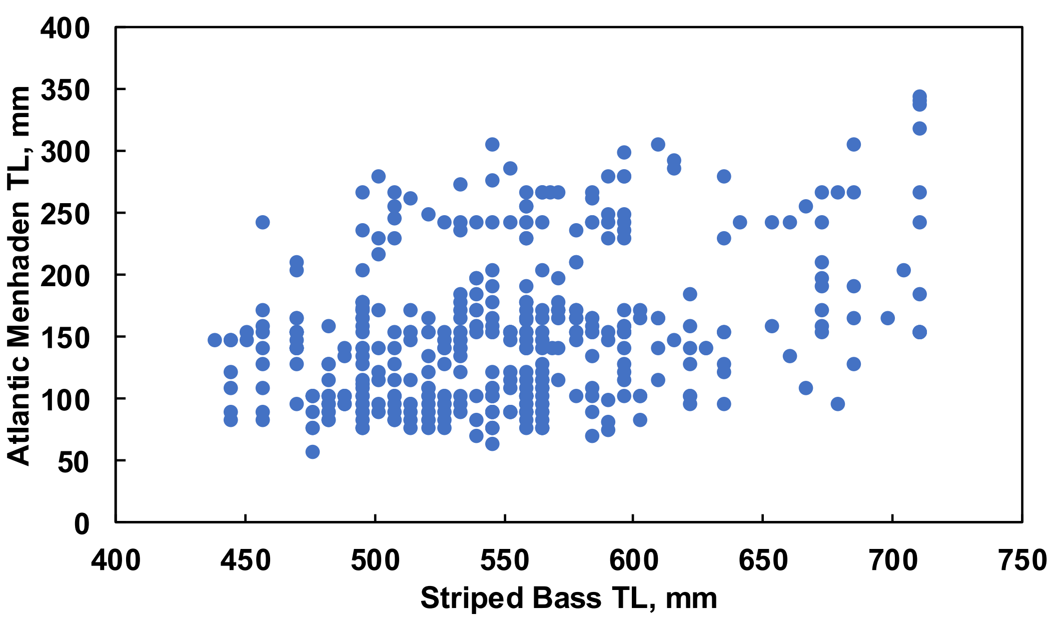

Figure 3. Frequency distribution of total lengths of Atlantic Menhaden (mm) consumed by 457-711 mm TL Striped Bass in winter samples, 2006-2015. This length range was chosen to represent resident Striped Bass.

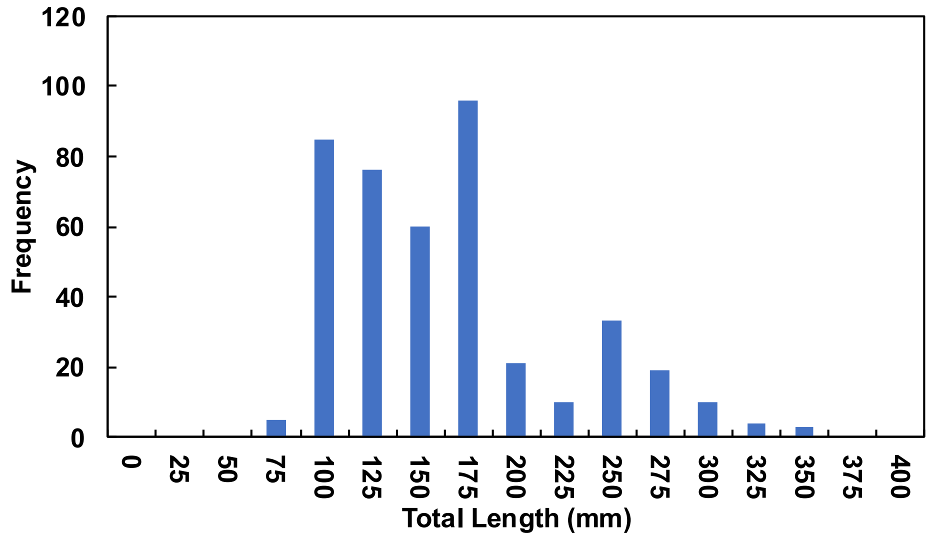

Figure 4. Chesapeake Bay reduction landing estimates and harvest caps for Atlantic Menhaden, 1964-2021.

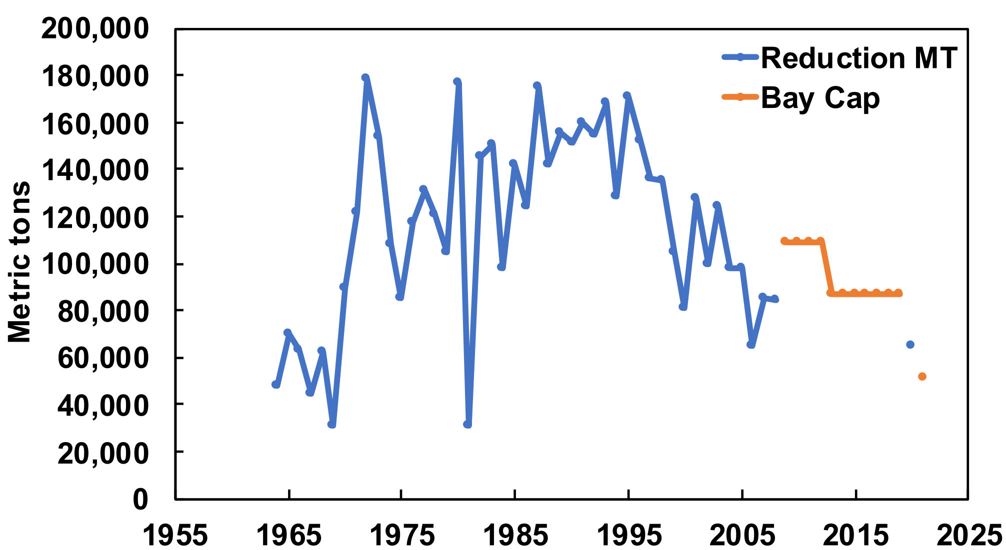

Figure 5. Estimates of Chesapeake Bay bait landings for Atlantic Menhaden. NMFS = landings estimated for Uphoff (2003b). PDT = landings provided by ASMFC for the Atlantic Menhaden plan development team.

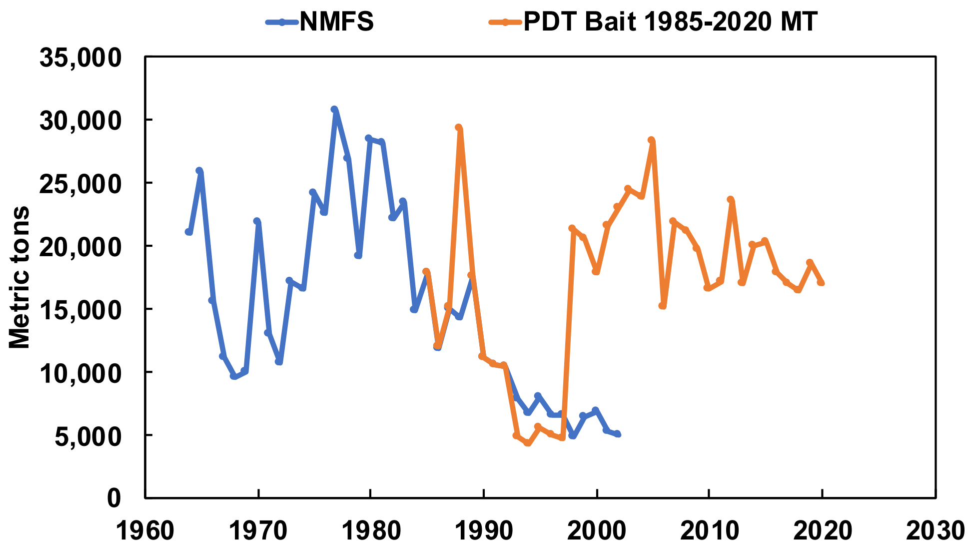

Figure 6. Total ostensible harvest of Atlantic Menhaden from Chesapeake Bay (Bait estimates + reduction landings estimates or cap), 1964-2020.

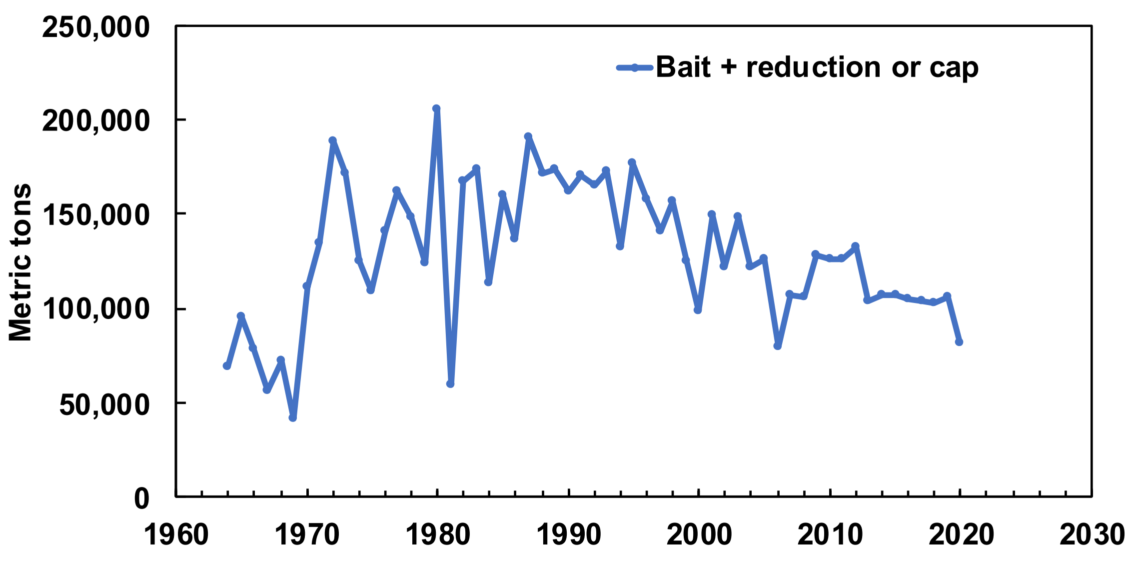

Figure 7. Long -term Potomac River Fisheries Commission(PRFC) ages 1+Atlantic Menhaden index, 1964-2021. MT = metric tons.

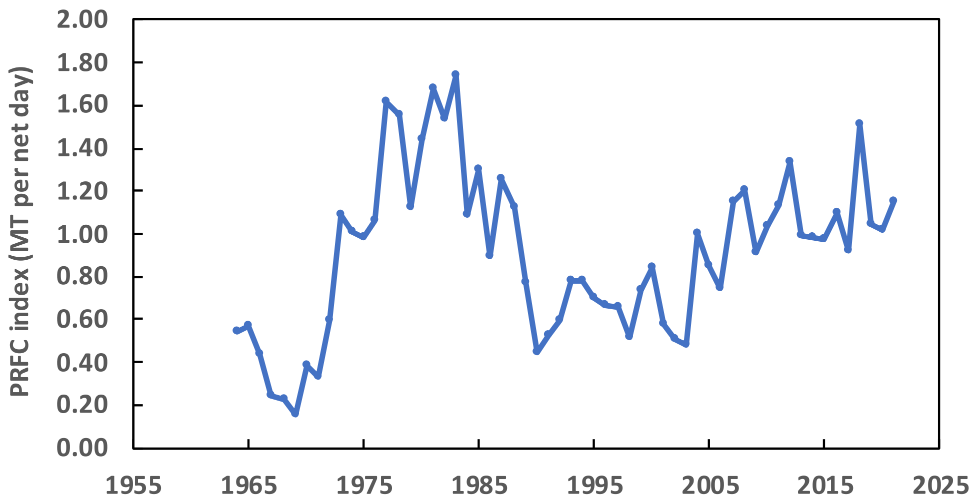

Figure 8. PRFC ages 1+ Atlantic Menhaden index traffic light boundaries during the 1995-2021 reference period.

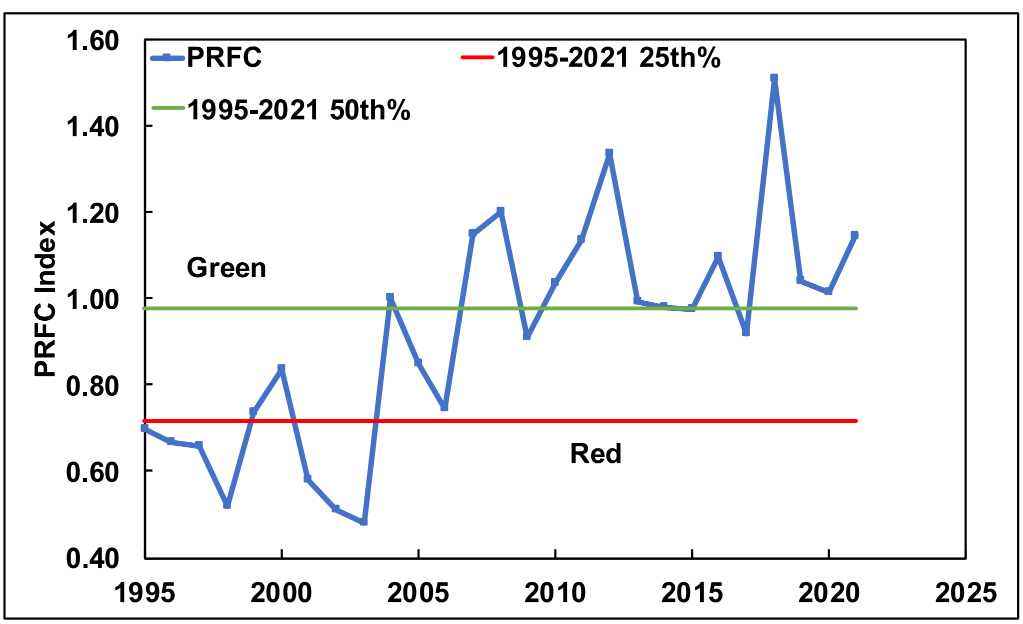

Figure 9. PRFC ages 1+ Atlantic Menhaden index frequency distribution, 1995 -2021.

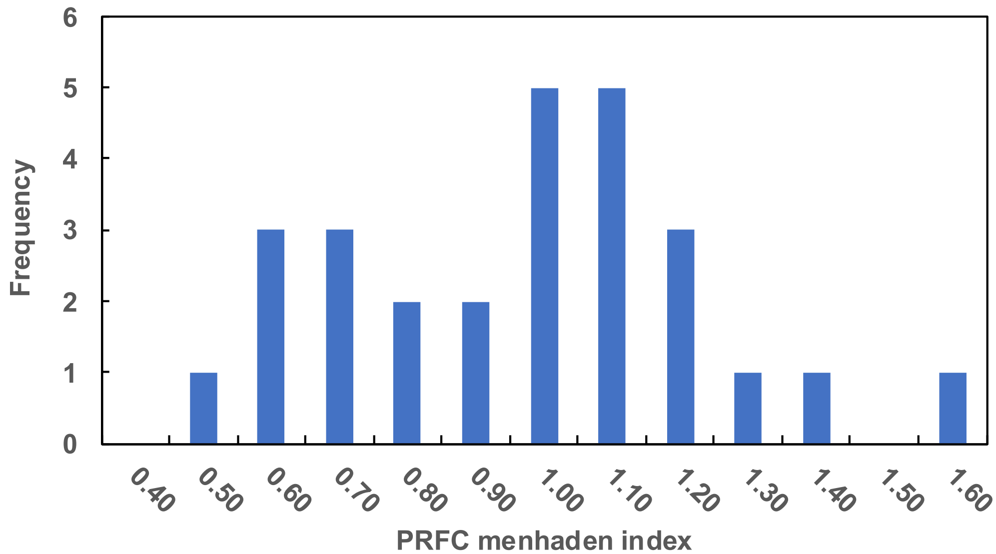

Figure 10. Relative F based on ostensible total Chesapeake Bay landings and PRFC ages 1+ Atlantic Menhaden during 1964-2020. Scale for relative F is arbitrary.

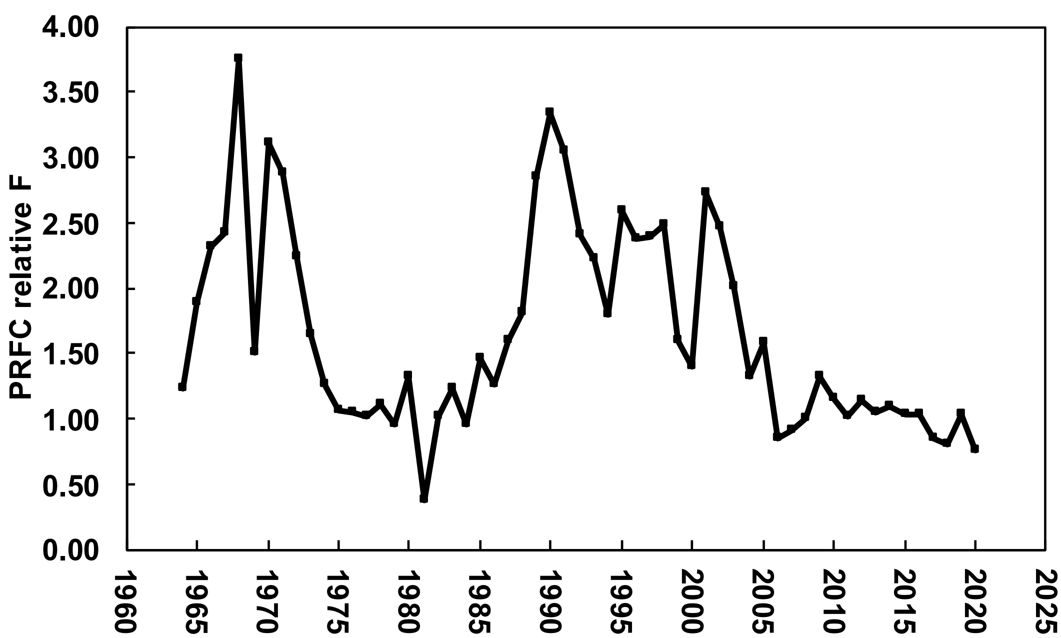

Figure 11. Relative F during the 1995-2021 reference period and its traffic light boundaries,1995-2020. Scale for relative F is arbitrary. It was not estimated for 2021 because the 2022 PRFC index was required in the denominator and it was unavailable.

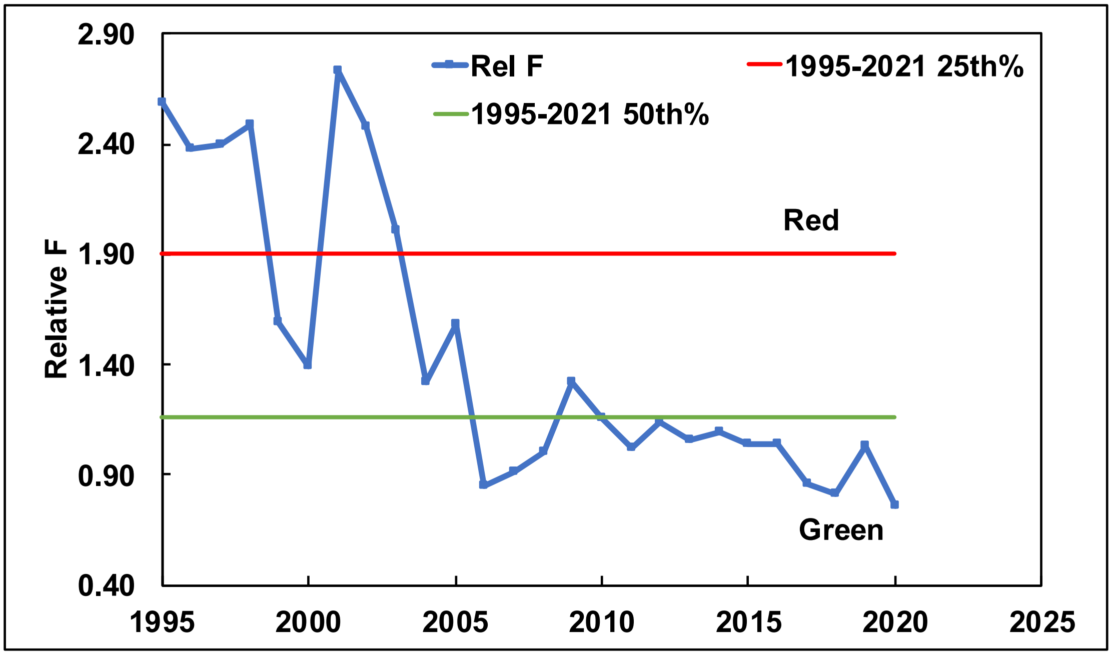

Figure 12. Relative F frequency distribution during the 1995 -2021 reference period. Scale for relative F is arbitrary.

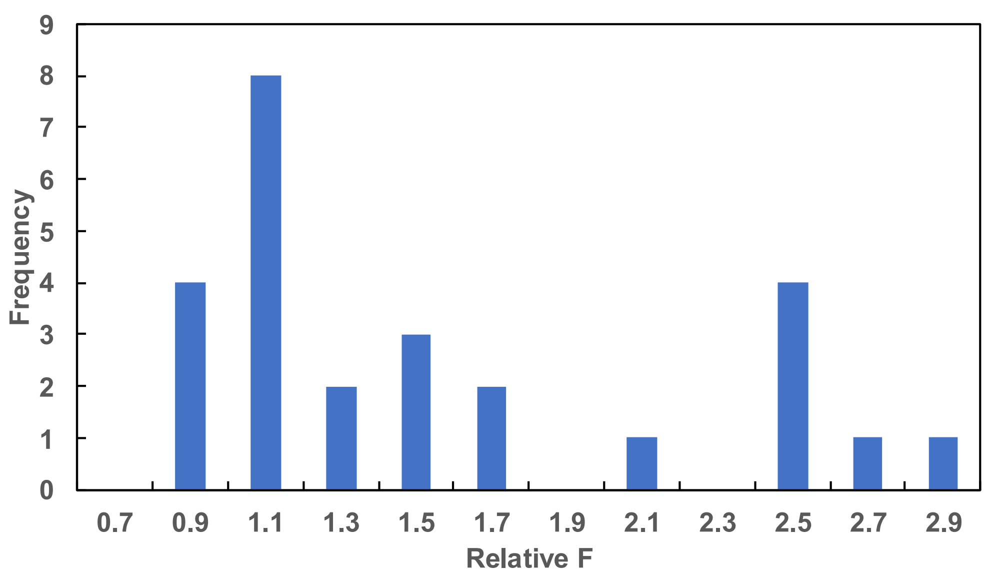

Figure 13. Estimated Atlantic coast fishing mortality (Fb) based on landings and biomass estimates from the Beaufort Assessment Model (BAM)and the trend in Chesapeake Bay F based on PRFC relative F during the 1995-2021 reference period. Both estimates are based on total landings weight estimates divided by ages 1+ biomass (BAM = biomass estimate; PRFC = biomass index. Scale for relative F is arbitrary.

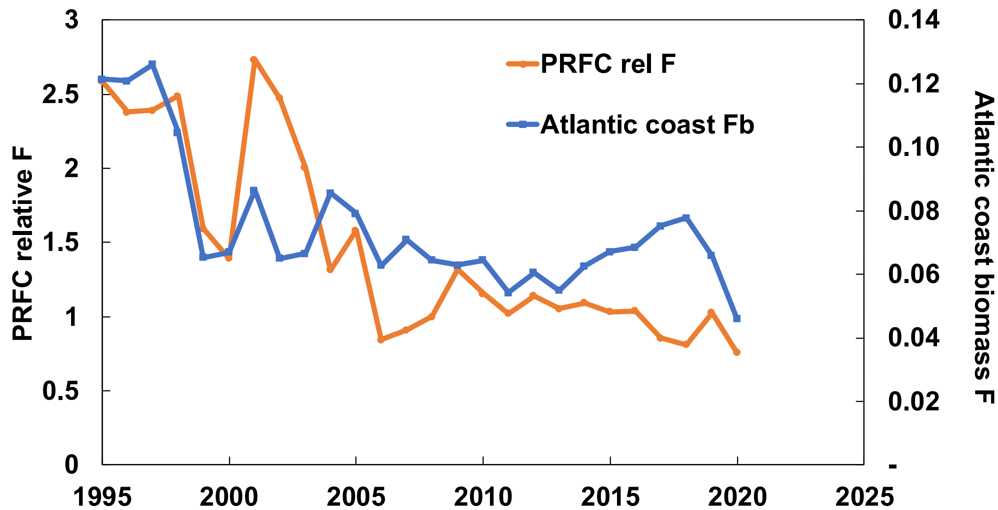

Figure 14. Geometric mean Atlantic Menhaden catch per standard seine haul (juvenile index or JI) long time -series 90% CI, 1959-2021. Survey is conducted in four regions of Maryland's portion of Chesapeake Bay (Head-of-Bay, Potomac River, Choptank River, and Nanticoke River.

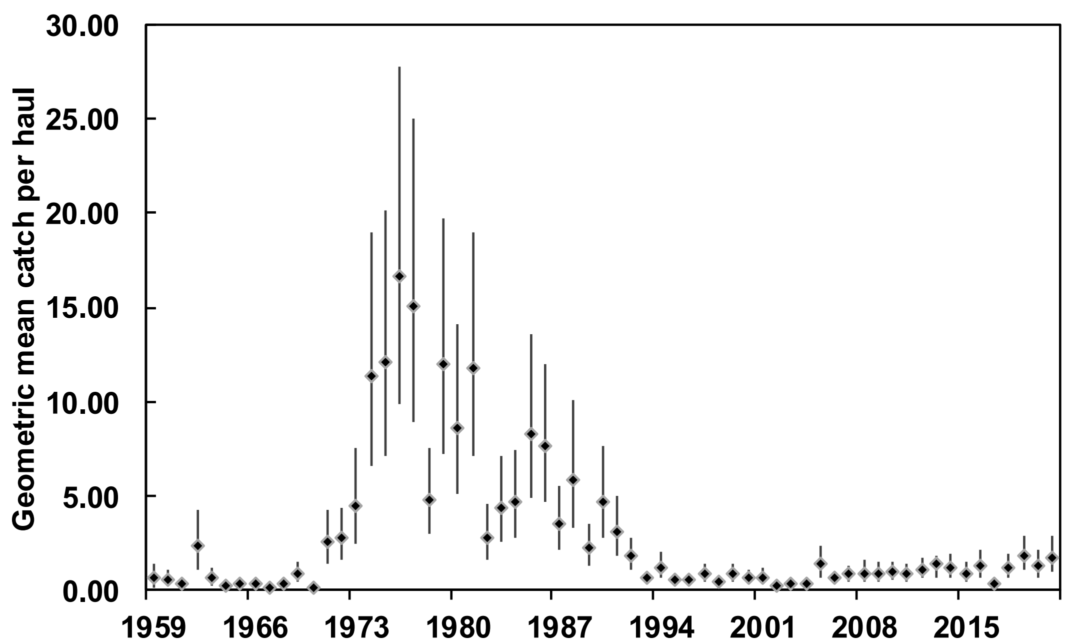

Figure 15. Maryland Atlantic Menhaden juvenile index (JI) and its traffic light boundaries for 1995-2021. GM = geometric mean.

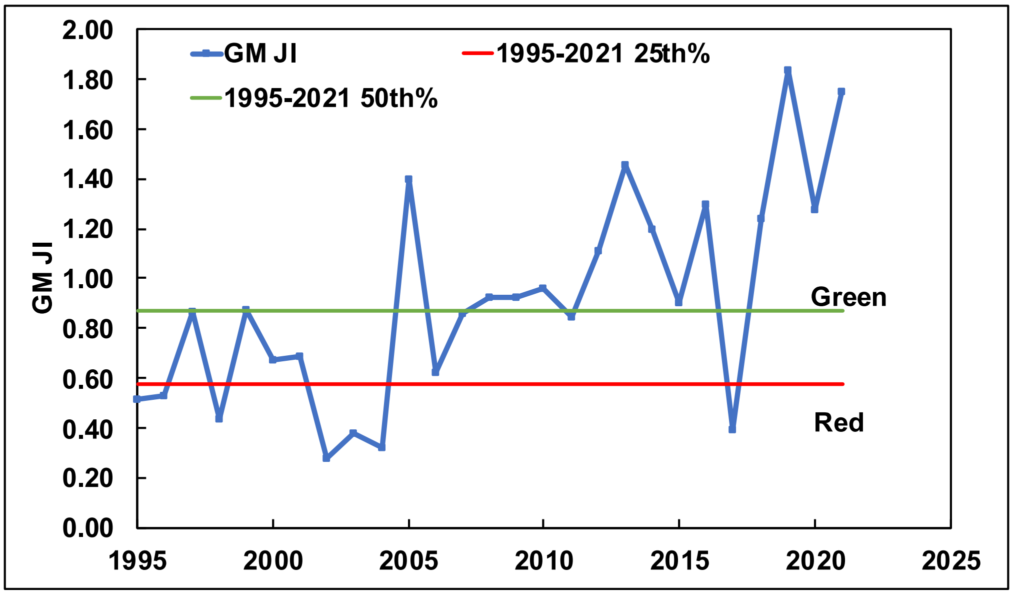

Figure 16. Atlantic Menhaden juvenile index (JI) for Maryland's portion of Chesapeake Bay frequency distribution, 1995-2021. GM = geometric mean.

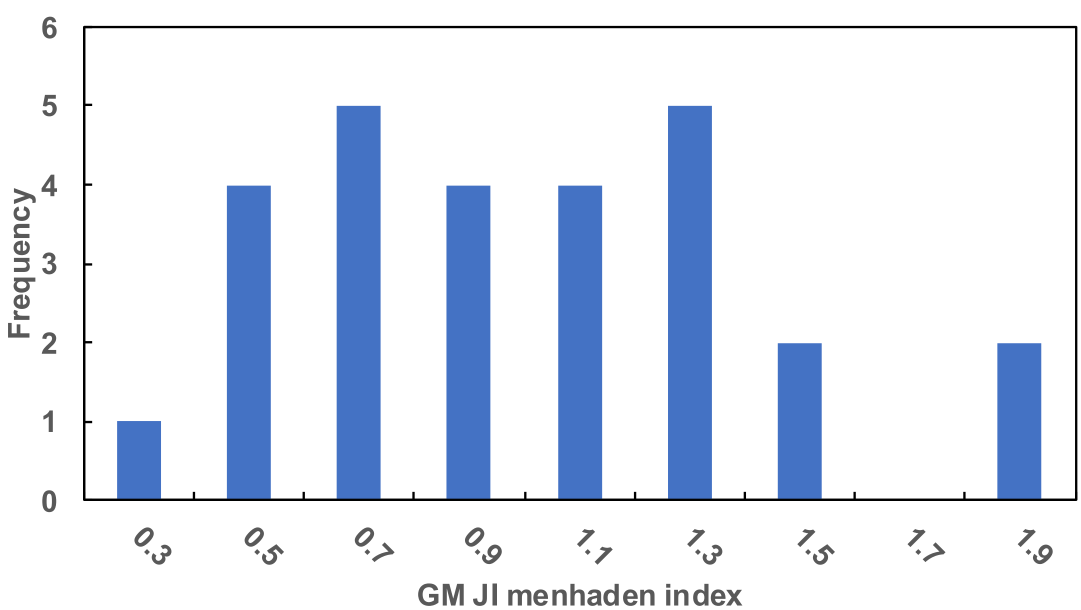

Figure 17. Indices of availability of Menhaden to resident Striped Bass (ages 3 -6) for age 0 (GM JI) or ages 1+ (PRFC) Atlantic Menhaden in Maryland's portion of Chespaeake Bay, 1982-2021. Availability index = Atlantic Menhaden index / Striped Bass index.

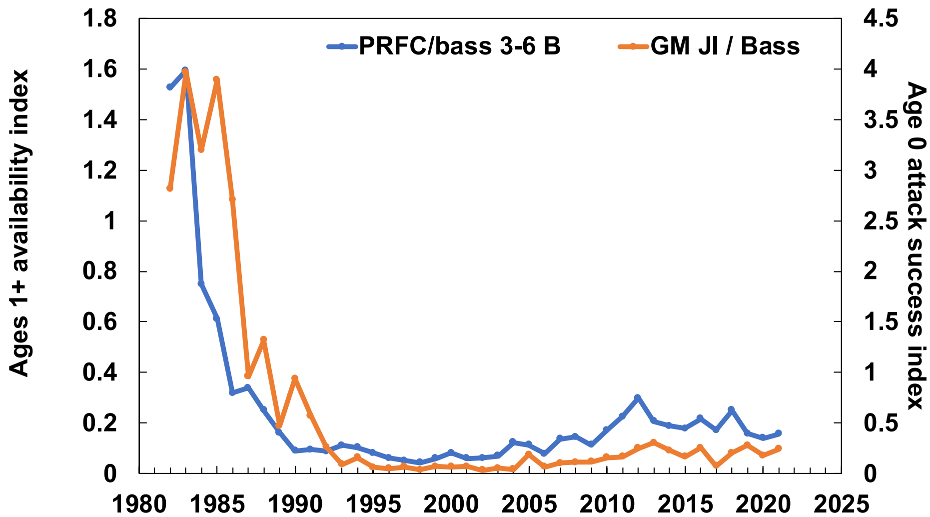

Figure 18. Ages 1+ Atlantic Menhaden availability index: PRFC index per age 3-6 Striped Bass biomass index (biomass ÷ 10,000 for scale) and its traffic light boundaries, 1995-2021

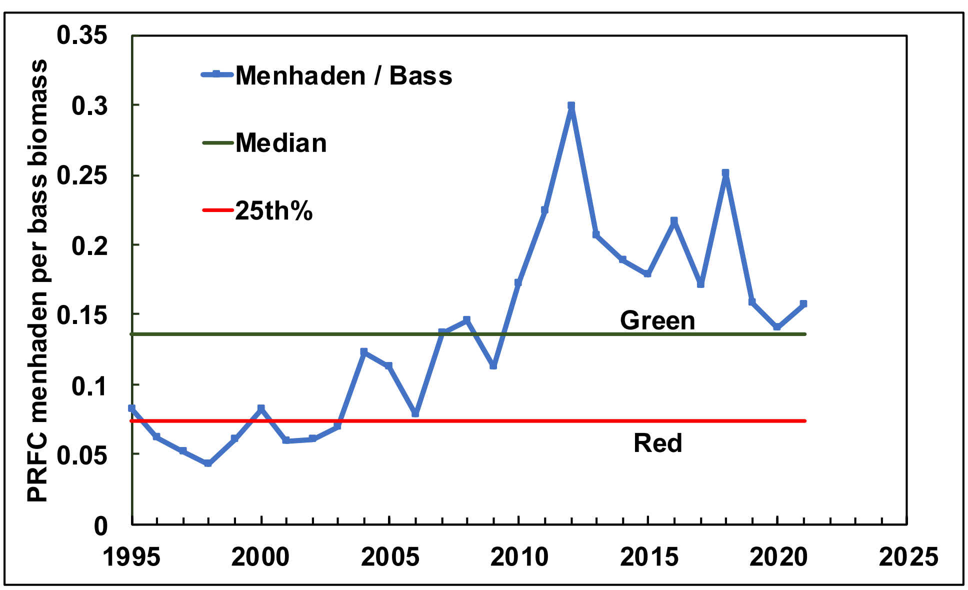

Figure 19. Ages 1+ Atlantic Menhaden availability index (PRFC index per Striped Bass index frequency distribution during the 1995-2021 reference period.

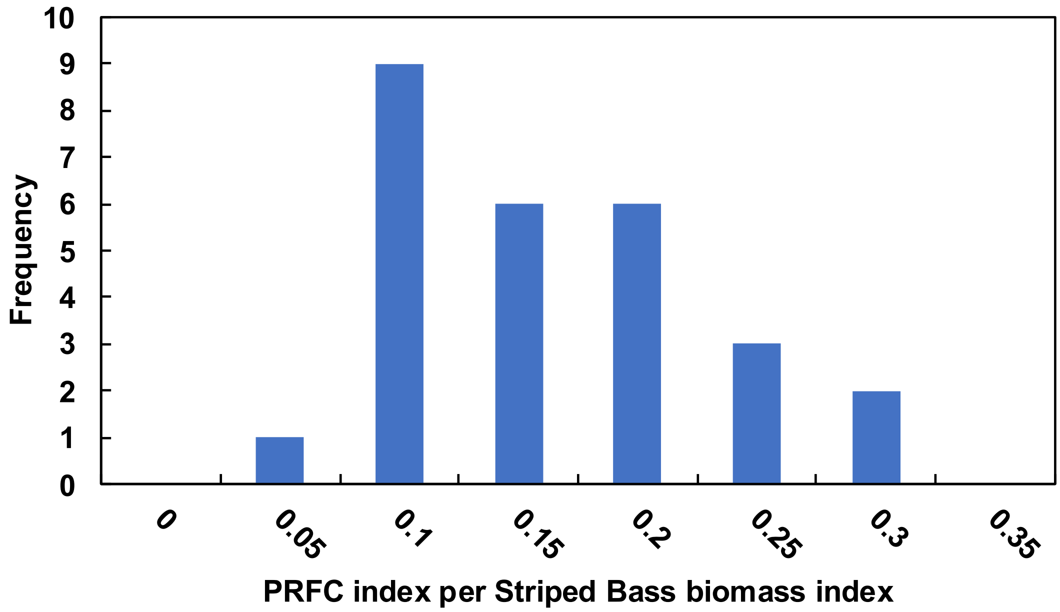

Figure 20. Availability index: Menhaden juvenile index (GM JI) per age 3-6 Striped Bass biomass (÷ 10,000 for scale) and its traffic light quartiles for 1995-2021. GM = geometric mean.

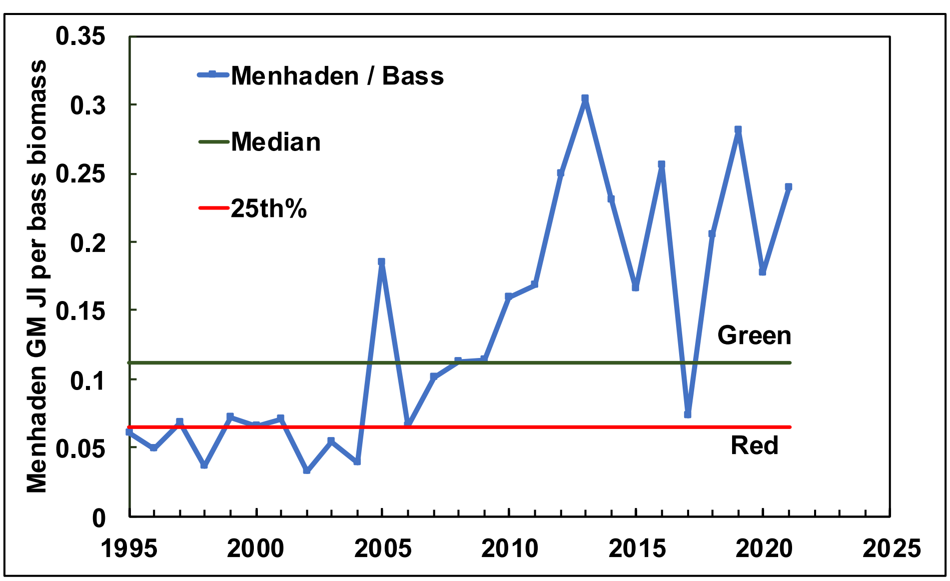

Figure 21. Age 0 availability index frequency distribution during the 1995-2021 traffic light reference period.

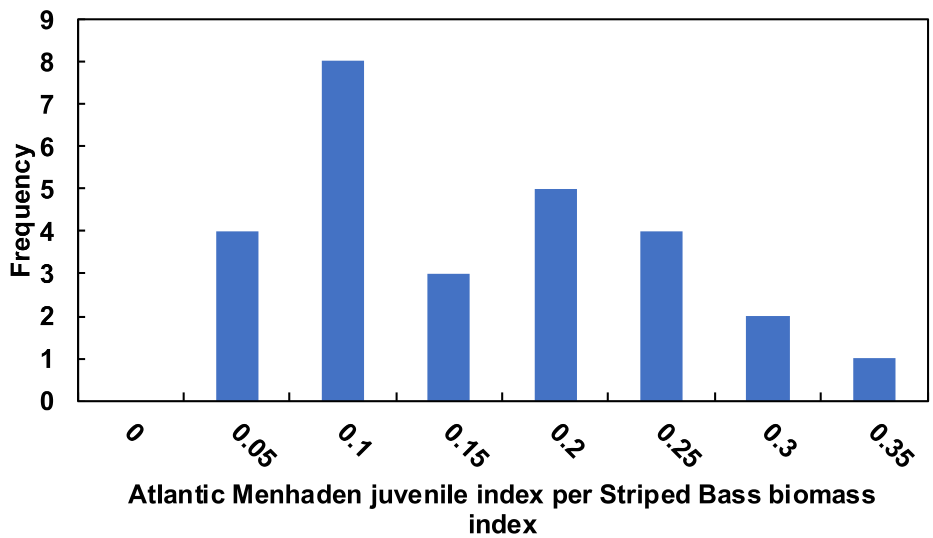

Figure 22. Trends in components of Atlantic Menhaden per Striped Bass (availability) indices during the 1995-2021 reference period. Striped Bass Biomass is for ages 3-6; PRFC index is for ages 1+ Menhaden; GM JI is the geometric mean juvenile (age 0) index.

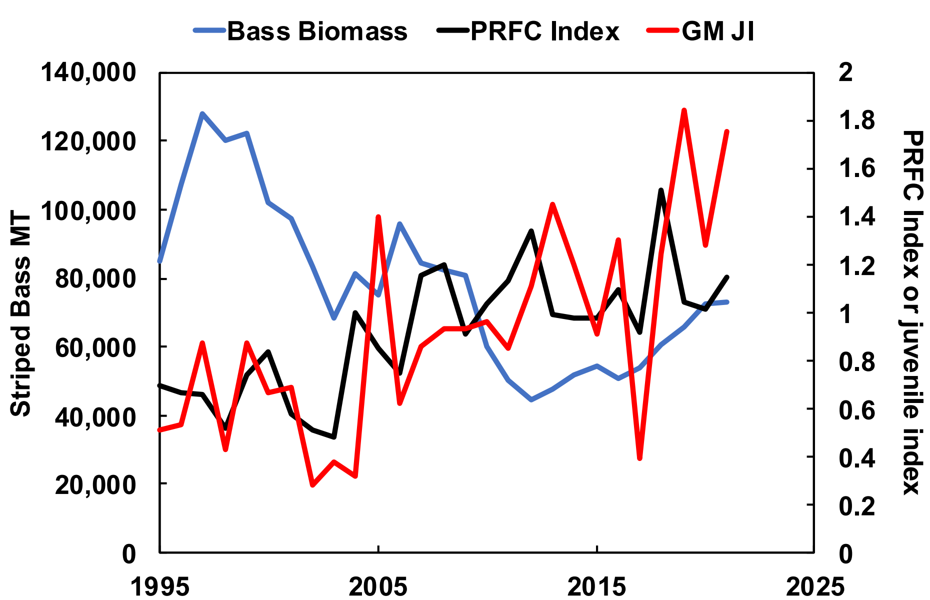

Figure 23. Core index: Proportion of resident Striped Bass ≥ 457 mm, TL without body fat (P0) during October-November, 1998-2021, (MD DNR Fish and Wildlife Health Program monitoring) and its 90% confidence interval, with body fat boundaries for good (green) and poor (red) condition.

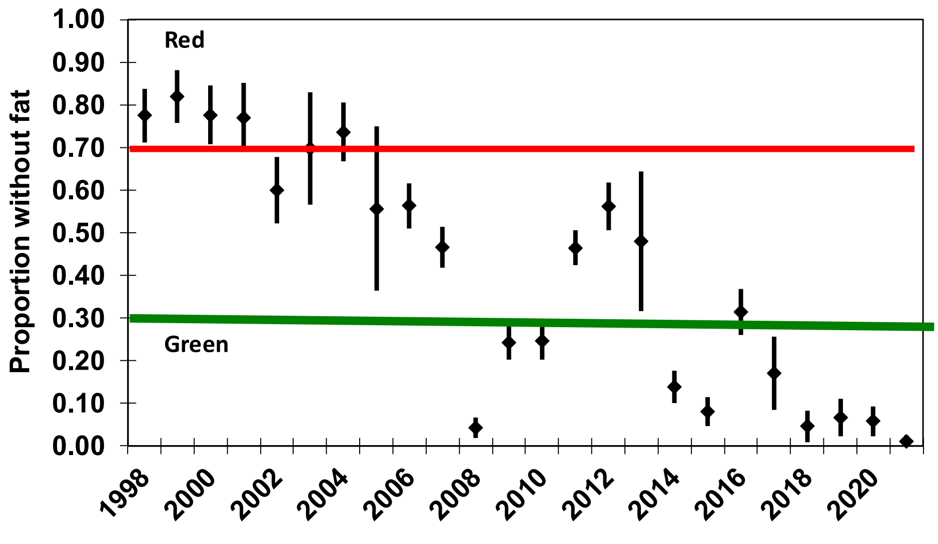

Figure 24. The traffic light index for Atlantic Menhaden in Maryland's portion of Chesapeake Bay, 1995-2021.

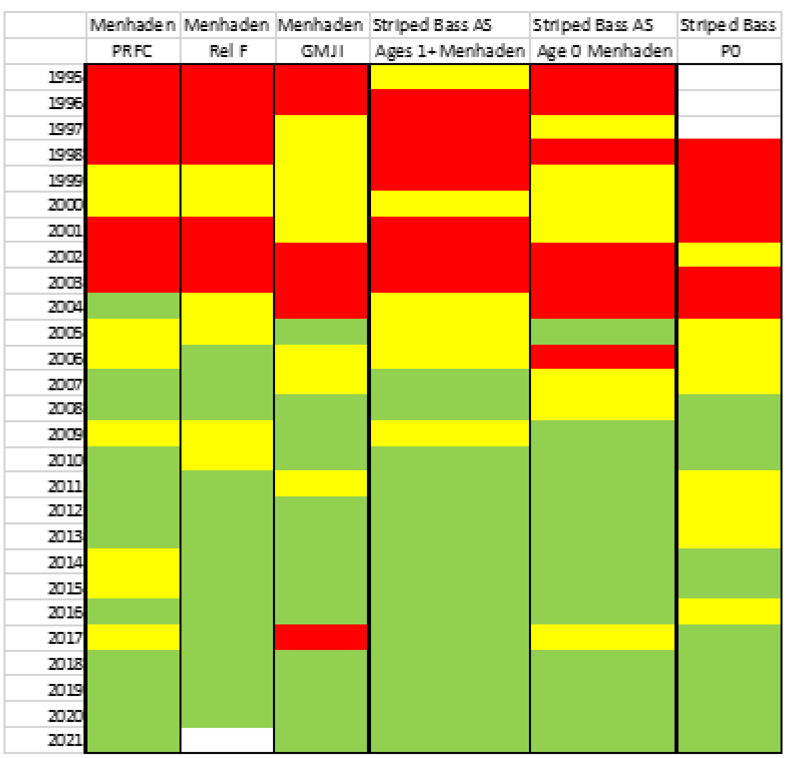

Figure 25. The traffic light index for Atlantic Menhaden in Maryland's portion of Chesapeake Bay when the first three years are removed(1995-1997) to form the traffic light boundaries for the first five indicators.

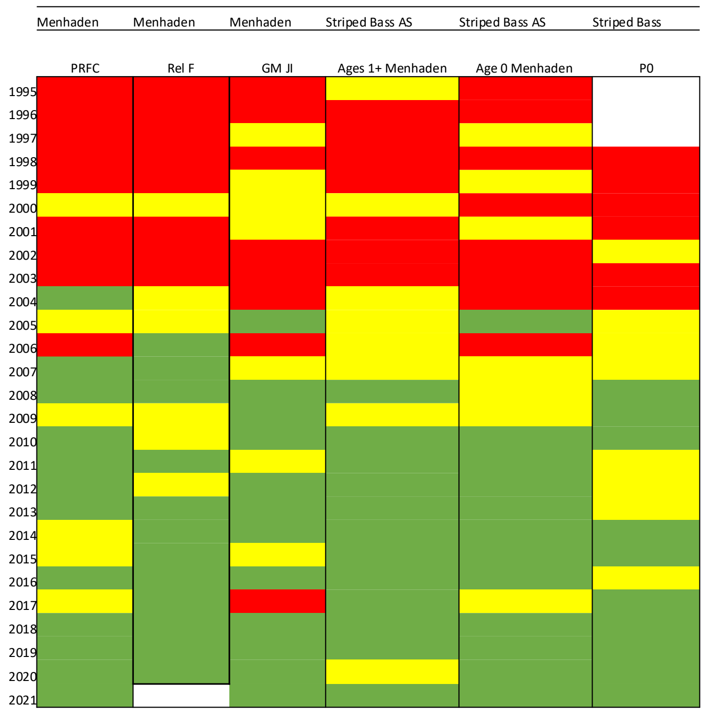

Figure 26. The traffic light index for Atlantic Menhaden in Maryland's portion of Chesapeake Bay when the last three years are removed(2019-2021) to form the traffic light boundaries for the first five indicators.

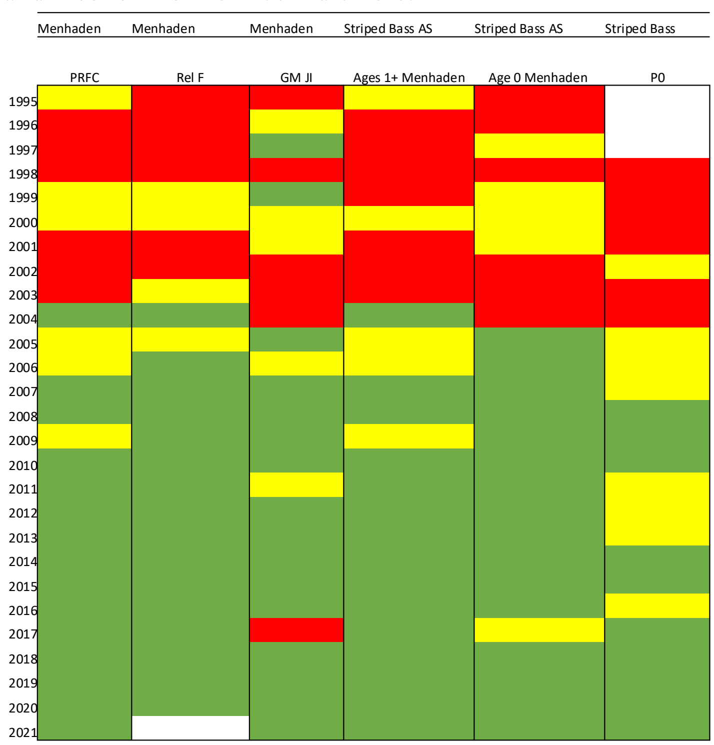

Figure 27. Long-term age 1+ menhaden indices of relative biomass (MD and PRFC pound net) or abundance (MD Head-of-Bay spring gill net survey).

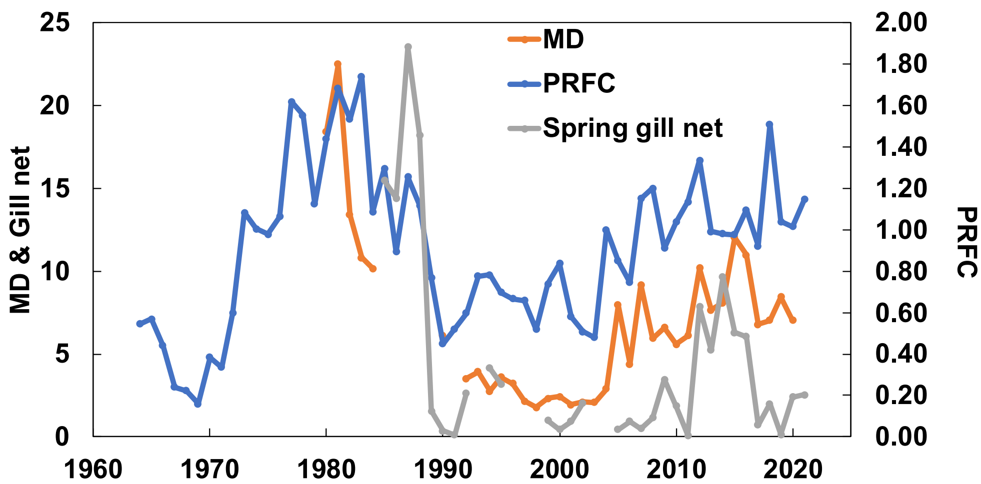

Figure 28. The Maryland juvenile index used in the ASMFC coastal stock assessment and its 90% confidence interval. The index is the result of a negative binomial generalized linear mixed model.

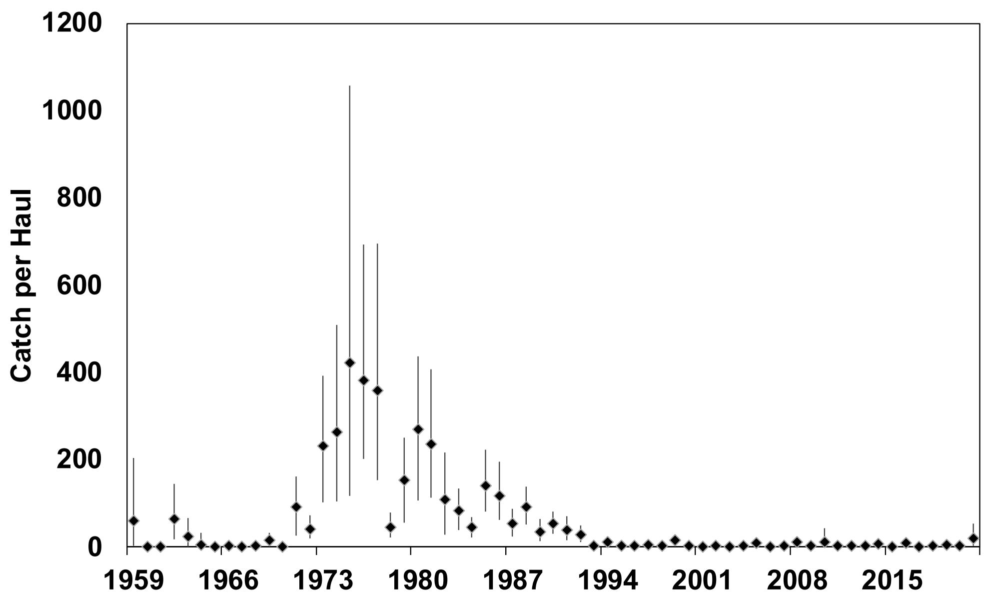

Figure 29. Comparison of the geometric mean (GM) and ASMFC (GLMM JI) Atlantic Menhaden juvenile indices for Maryland's portion of Chesapeake Bay during the reference period.

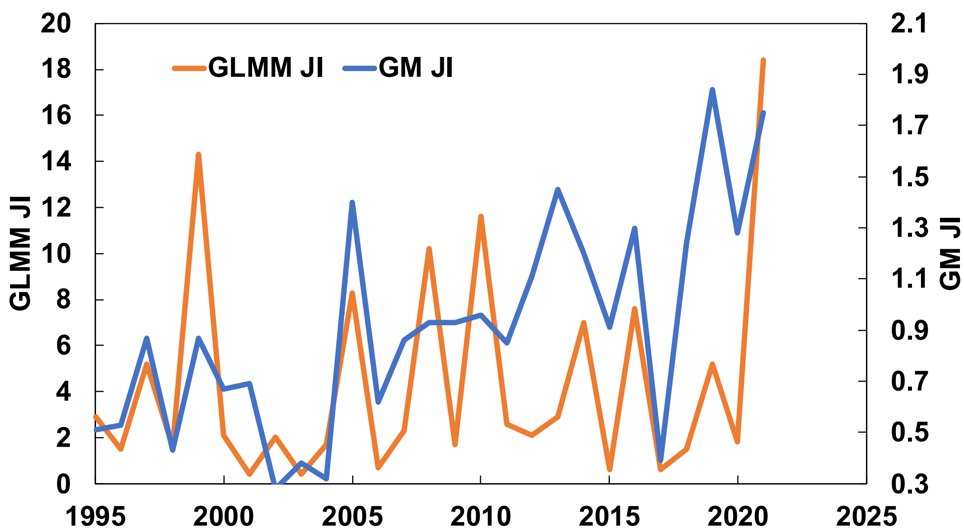

Figure 30. Winter Atlantic Multidecadal Oscillation (AMO) and the Maryland Atlantic Menhaden juvenile index (JI). GM = geometric mean.

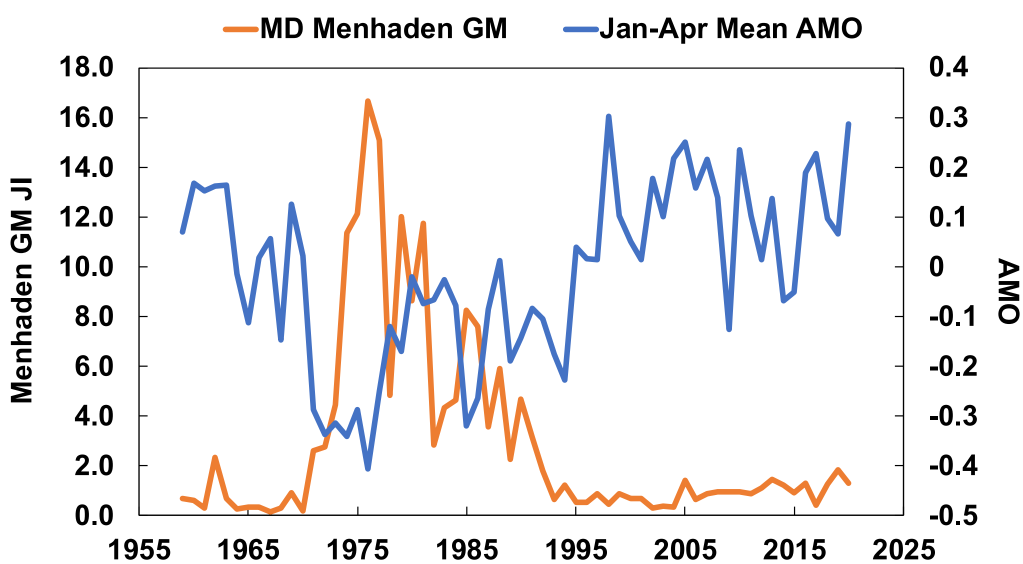

Figure 31. Atlantic Menhaden juvenile index (JI in year-2) and ages 1+ PRFC index (year t) trends. GM = geometric mean.

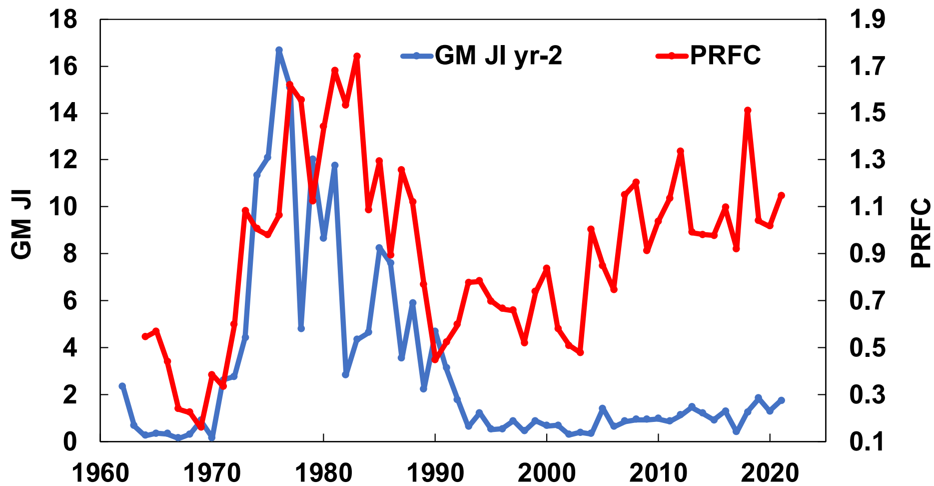

Figure 32. Plot of results of a multiple regression of PRFC biomass index as a function of the Maryland juvenile relative abundance index (GM JI) and time categories representing less regulated (1964-2005) and more regulated periods (2006-2021). PRFC Cat 0 = observed values during 1964-2005 and Pred Cat 0 = predictions. PRFC Cat 1 = observed values during 2006-2021 and Pred Cat 1 = predictions.

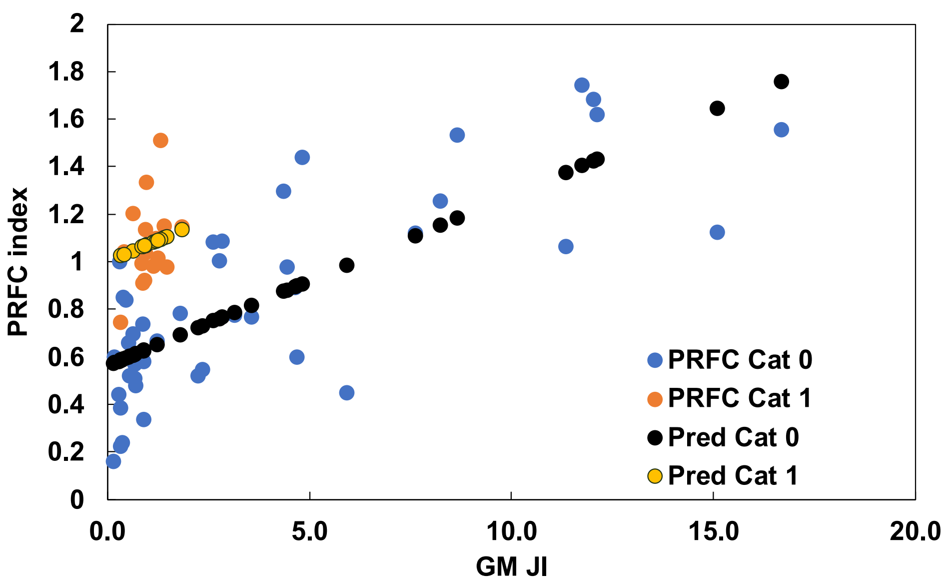

Figure 33. Proportion of striped bass (all sizes) without body fat in summer versus fall, 1999-2012. Summer sampling was discontinued after 2012.

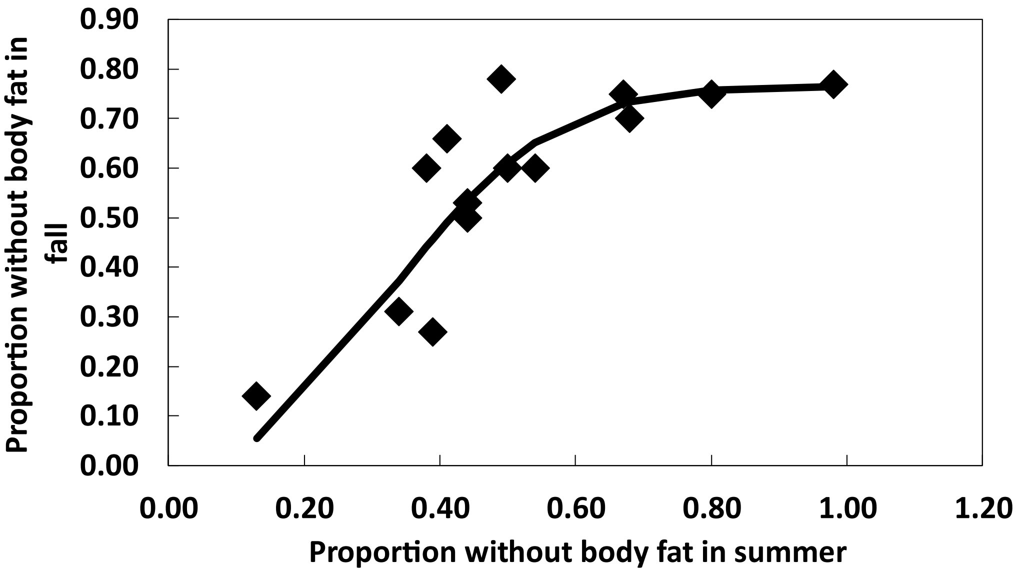

Figure 34. Relationship of body fat indices in fall of year t to fall of year t+1

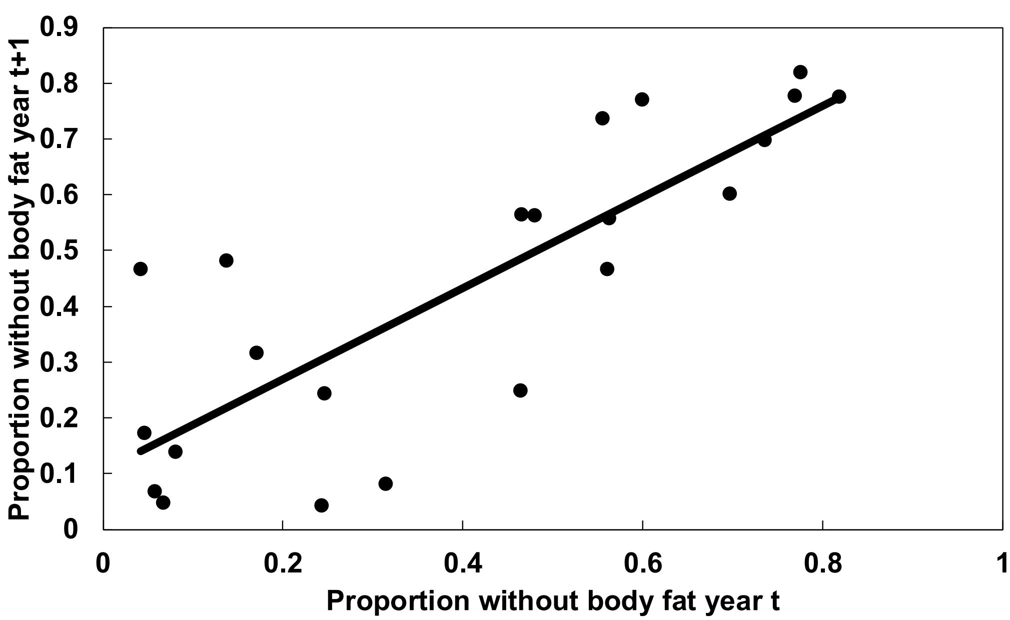

Figure 35. Relationship of proportion of resident striped bass ≥ 457 mm, TL, without body fat and PRFC relative F from piecewise regression. Slightly declining predicted slope at higher relative F is not different from 0.

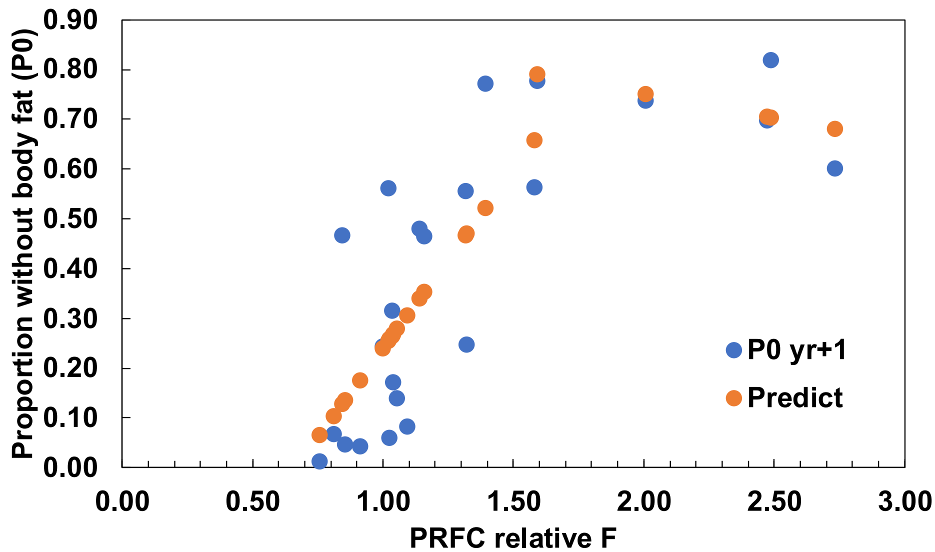

Figure 36. Trends in availability (AS) indices for ages 1+ or age 0 Atlantic Menhaden and proportion of Striped Bass without body fat (P0).

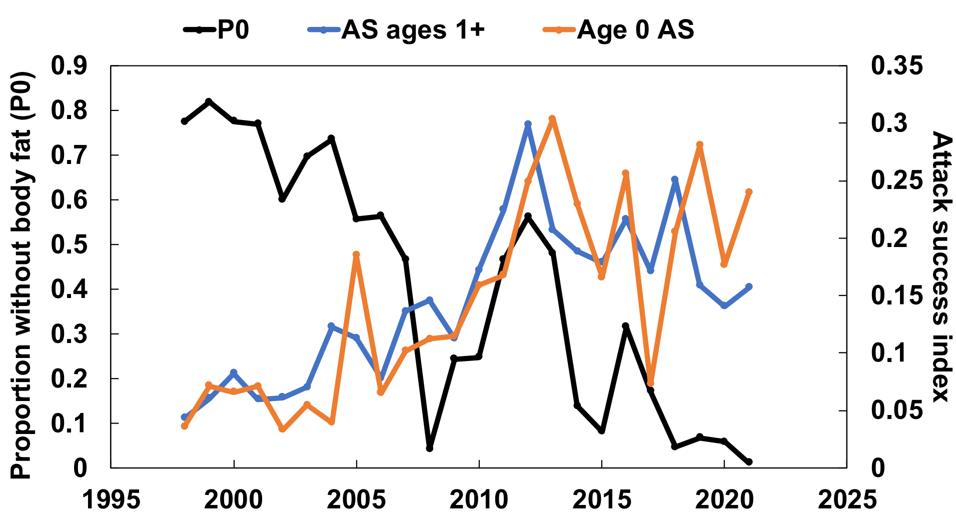

Figure 37. Relationship of proportion of resident striped bass ≥ 457 mm, TL, without body fat and the ages 1+ Menhaden availability index.

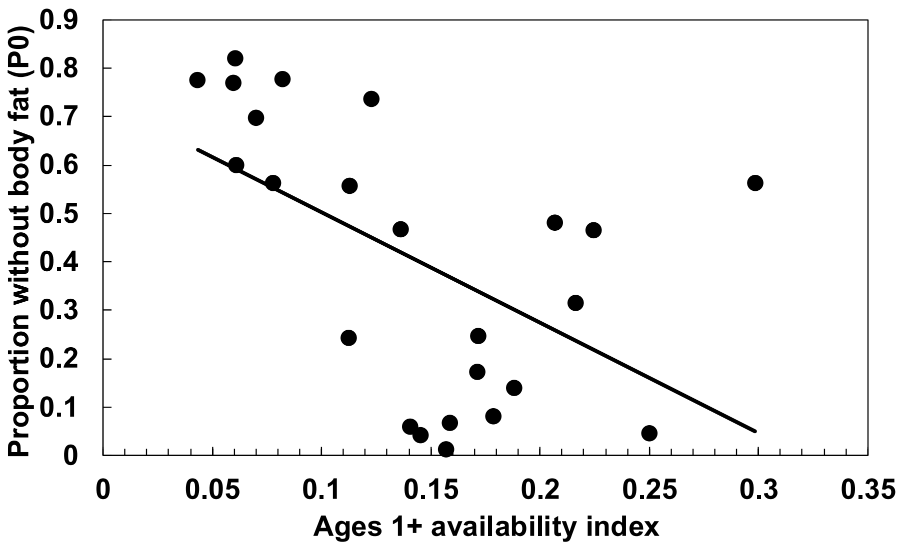

Figure 38. Relationship of ages 1+ Menhaden availability index (PRFC per Striped Bass) and relative F during the reference period (1995-2021).