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C2.0 Terms of Reference for Weakfish

1. Evaluate biases, precision, uncertainty, and sampling methodology of the commercial and recreational catch (including landings and discards) and effort.

2. Evaluate precision, geographical coverage, representation of stock structure, and relative accuracy of the fisheries independent and dependent indices of abundance. Review preliminary work on standardization of abundance indices.

3. Evaluate the ADAPT VPA catch at age modeling methods and the estimates of F, Z, spawning stock biomass, and total abundance of weakfish produced, along with the uncertainty and potential bias of those estimates. Review the severity of retrospective pattern.

4. Evaluate the index based methods and the estimates of F, ages 1+ stock biomass, surplus production, and time-varying natural mortality of weakfish produced, along with the uncertainty of those estimates. Determine whether these techniques could complement or substitute for age-based modeling for management advice.

5. Evaluate testing of fishing and additional trophic and environmental covariates and modeling of hypotheses using biomass dynamic models featuring multiple indices blended into a single index with and without a Steele-Henderson (Type III) predator-prey extension. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.

6. Evaluate AIC-based hypotheses is testing of fishing and additional predation-competition effects using multi-index biomass dynamic models with and without prey-based, predator-based, or ratio dependent predator-prey extensions. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.

7. Review evidence for constant or recent systematic changes in natural mortality, productivity, and/or unreported removals.

8. Estimate biological reference points using equilibrium and non-equilibrium assumptions and evaluate stock status relative to these BRPs.

9. Review stock projections and impacts on the stock under different assumptions of fishing and natural mortality.

10. Make research recommendations for improving data collection and assessment.

C3.0 Executive Summary

C3.1 Major findings for TOR 1 –Evaluate biases, precision, uncertainty, and sampling methodology of the commercial and recreational catch (including landings and discards) and effort.

Weakfish fishery data were evaluated from four fishery sectors: commercial harvest, commercial discards, recreational harvest, and recreational discards. Commercial harvest data were obtained from state and federal harvest reporting systems. Commercial discards were estimated following the method of de Silva (2004) for key gear-species combinations found to be associated with relatively high rates of weakfish discards. A 100% discard mortality rate was assumed. Recreational harvest and discards were obtained from the Marine Recreational Fisheries Statistics Survey (MRFSS) conducted by the National Marine Fisheries Service. Harvest numbers and weight are directly available; discard numbers were estimated as the number of weakfish released alive times a discard mortality rate of 10% which is based on quantitative studies.

Harvest and discard estimates were stratified by region (north/south), year, and season (early = January to June and late = July to December). Commercial harvest was further stratified by state and gear. Where available, stratum specific biological data (length data and length-weight equations) were used to convert harvest and discard weights to number of weakfish removals at size. Where stratum specific data were not available (some commercial harvest strata), samples were substituted from the next most representative stratum. Numbers at size was then converted to numbers at age using region/year/season specific age-length keys. Numbers at age were summed across strata within a year to develop annual estimates of total weakfish catch at age.

Several sources of potential bias were identified that may result in uncertainty in annual catch at age estimates. These include inaccurate harvest/discard estimates as a result of under/over reporting or inappropriate survey methods; insufficient sample size to characterize length distributions; substitution of data from alternate cells in the catch at size characterization and age-length keys; errors in aging techniques or the scale-otolith age conversion; and others. Several of these sources are generic and not specific to weakfish. Attempts have been made to quantify some of these error sources; however, the extent of uncertainty associated with each of these sources, and their cumulative effect, remains largely unknown. Improvements in data collection from commercial landings have been instituted since 2000 that have greatly increased coverage and reliability of data.

C3.2 Major findings for TOR 2 – Evaluate precision, geographical coverage, representation of stock structure, and relative accuracy of the fisheries independent and dependent indices of abundance. Review preliminary work on standardization of abundance indices.

Five fishery independent age structured surveys were evaluated for use in the stock assessment. Surveys were evaluated relative to criteria such as geographic coverage, ability to accurately track weakfish abundance, and survey precision, among other factors. Catch per unit effort indices of abundance from three age-structured fishery independent surveys, including the Delaware Bay and SEAMAP trawl surveys and the North Carolina gillnet survey, were found to be suitable for use in the assessment. The North Carolina gillnet survey began in 2001, and this is the first time it has been included as a tuning index for weakfish. The NEFSC fall trawl survey, which has been used as a tuning index in previous weakfish stock assessments, was found to be unsuitable for use because of high interannual variability in catches, limited ability to capture weakfish greater

than 34 cm, and instances of negative mortality in year class catch curves. Similarly, a CPUE index based on two fall cruises of the New Jersey ocean trawl survey was found to provide little information on weakfish stock abundance; however, an alternate index based on the proportion of "positive" (*i.e.* non-zero) tows from the August cruise was found to be a suitable indicator of abundance. Ten young of year fishery independent surveys were also evaluated, one of which (Massachusetts trawl survey) was eliminated from further analysis due to exceptionally large coefficients of vavriation. Two fishery dependent indices of abundance were also included in the assessment. One is based on total catch per trip in the Mid-Atlantic private boat recreational fishery and encompasses an age aggregate index for ages 2+. The other is based on harvest per trip in the Mid-Atlantic private boat recreational fishery and is separated into age specific indices for ages 3, 4, 5, and 6+.

A team of researchers at Virginia Tech University has recently begun investigating the utility of standardizing weakfish abundance indices relative to spatial, temporal, and environmental factors using GLM and GAM methods. Although further evaluation of the methods and results is required by the Weakfish Technical Committee (WTC), preliminary results of the standardization analyses are presented in this report.

C3.3 Major findings for TOR 3 – Evaluate the ADAPT VPA catch at age modeling methods and the estimates of F, Z, spawning stock biomass, and total abundance of weakfish produced, along with the uncertainty and potential bias of those estimates. Review the severity of retrospective bias.

Age structured modeling was conducted using ADAPT VPA. Various runs were conducted using different sets of tuning indices. The different runs were evaluated with respect to model fit, residuals, and retrospective patterns. All runs produced consistent estimates of parameter values for the years 1982 to 2002. Trends in estimated parameters for the years 2003 to 2007 varied widely, and were confounded by a prominent retrospective pattern. Fishing mortality was generally underestimated, while biomass and abundance parameters were over estimated. Model runs that included fishery dependent indices had smaller retrospective bias and better model fit (lower mean squared residual (MSR)) than runs tuned solely with fishery independent indices. Runs that included young of year indices had extended retrospective patterns but inconsistent effect on MSR. The run tuned solely with fishery dependent indices produced the best fit and minimal retrospective pattern; but inclusion of fishery independent indices from New Jersey, Delaware, and North Carolina did not substantially increase the retrospective pattern and produced the second lowest MSR of all runs investigated. This run was therefore selected as the preferred run. These indices correspond well with changes in harvest, abundance, CPUE, and population age structure. Terminal year estimates were estimated as $F_{2007} = 0.51$ (unweighted, ages 4-5) and $SSB_{2007} = 7,236$ MT, although these were poorly estimated given the observed retrospective pattern. Attempts to correct for retrospective pattern were conducted but were not specifically endorsed by the WTC. Because ADAPT VPA calculates fishing mortality as $F = Z-M$, estimates of F are dependent on input values of natural mortality. The WTC has expressed concern regarding the assumption of constant natural mortality of $M = 0.25$ across all ages and years. To circumvent the concerns regarding input M, the WTC prefers to combine model estimated F rates and input M values to portray the trend in total mortality, Z. Following record low levels in the mid 1990s, total mortality increased dramatically and exceeded $Z = 2.6$ in 2003. Z has declined in recent years to $Z_{2007} = 0.76$, but values in recent years are likely underestimated given the observed retrospective pattern.

C3.4   Major findings of TOR 4 – Evaluate the index based methods and the estimates of F, ages 1+ stock biomass, surplus production, and time-varying natural mortality of weakfish produced, along with the uncertainty of those estimates. Determine whether these techniques could complement or substitute for age-based modeling for management advice.

Because of systematic retrospective bias exhibited in recent F and stock biomass (mt) estimates from the catch-at-age (ADAPT) model, the 2006 Weakfish Assessment (Kahn et al 2006, Uphoff 2006a; Crecco 2006) relied primarily on an index-based (ages 1+) approach to monitor temporal changes in weakfish biomass (mt) and fishing mortality (F) from 1981 to 2003. Given that the index-based approach produced F and weakfish stock biomass (mt) estimates that displayed a similar trend to that from the converged portion (1982-1999) of the 2006 VPA, F and biomass estimates were updated with this approach through 2008 using an annual blended index based on the recreational private boat cpue, as well as on the New Jersey and Delaware trawl indices. The index-based approach was also used to estimate the magnitude and trend in ages 1+ weakfish surplus production from 1981 to 2008.

Biomass weighted fishing mortality (FWt) estimates on ages 1+ weakfish from the index-based approach were high (FWt range: 0.69- 1.16) by most standards from 1981 to 1987. The magnitude of FWt estimates, however, rose even higher to beyond 1.0 from 1988 to 1991, and greatly exceeded our current overfishing threshold for weakfish (Fmsy = 0.53). The magnitude of FWt estimates declined steadily thereafter to below 0.60 in most years from 2000 to 2008. The ages 1+ fishing mortality rates weighted by number (FNt) were almost always lower in magnitude than the corresponding biomass weighted fishing rates. The ages 1+ numbers weighted (FNt) estimates from 1981 to 2008 followed a similar trend over time as the biomass weighted FWt estimates, but unlike the biomass weighted fishing rates (FWt), the FNt estimates fell abruptly after 1991 and remained below 0.30 from 1993 to 2008.

Weakfish ages 1+ biomass levels from the index-based approach exhibited wide contrast from 1981 to 2008. Weakfish ages 1+ biomass (mt) remained relatively high (14,200 and 41,500 mt) from 1981 to 1988 but biomass levels fell steadily to below 10,000 mt from 1989 to 1993. Weakfish coast-wide biomass rose again temporarily from 1994 to 1996, but biomass fell steadily thereafter to the lowest level in the time series in 2008 (1,333 mt). The time series of weakfish ages 1+ surplus production (SURPt) from 1981 to 2008 followed the same general trend as stock biomass. Weakfish surplus production remained relatively high from 1982 to 1986 and again in 1993 and 1994, but SURPt levels fell steadily after 1995 and remained very low in most years from 2001 to 2008 despite relatively low and stable fishing mortality. The unexpected drop in weakfish surplus production after 1999 coincided with a sharp rise in the coast-wide abundance of two potential predators: striped bass and spiny dogfish.

C3.5 Major findings of TOR 5 – Evaluate testing of fishing and additional trophic and environmental covariates and modeling of hypotheses using biomass dynamic models featuring multiple indices blended into a single index with and without a Steele-Henderson (Type III) predator-prey extension. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and

whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.

Since the index-based approach produced 1982-1998 ages 1+ F and weakfish stock biomass estimates that were similar to those over the converged portion (1982-1998) of the 2006 VPA, this approach was used to update ages 1+ F and stock biomass through 2008 using the recreational private boat cpue, as well as New Jersey and Delaware trawl indices. In addition, ages 1+ surplus production estimates were derived from 1981 to 2008 from which steady-state overfishing (Fmsy, Bmsy) thresholds were derived for Atlantic coast weakfish. Third, the age aggregated (ages 1+) Steele and Henderson (1984) (S-H) production model was updated through 2008 to further examine the joint effects of fishing and predation from striped bass (*Morone saxatilus*) and spiny dogfish (*Squalus acanthias*). The results from the S-H model were also used to estimate equilibrium and non-equilibrium *Fmsy* and *Bmsy* thresholds. Fourth, to provide a more thorough examination of the Predation Hypothesis, additional candidate predators such as bluefish (*Pomatomus saltatrix*) and summer flounder (*Paralichthys dentatus*) were also considered as candidate predators on weakfish, especially since both finfish predators have risen sharply inshore along the Atlantic coast after 1998. Finally, environmental disturbances have been proposed as a major process governing shifts in finfish production and recruitment (Hollowed et al 2000b), so environmental factors such as decadal shifts in mean sea surface water temperature and deviations in the winter North Atlantic Oscillation Index were also examined as potential explanatory variables.

The preponderance of statistical evidence given here supports the Predation Hypothesis involving enhanced predation by striped bass and spiny dogfish as the primary factor behind the recent and unexpected decline in weakfish productivity. Statistical evidence in support of the Predation Hypothesis consists of a significant (P <0.0001) inverse correlation between declining weakfish biomass and surplus production from 1999 to 2008 and striped bass and spiny dogfish abundance from 1982 to 2004. Striped bass abundance along the Atlantic coast rose 10 fold from 1994 to 2006 (Kahn 2005), although the 2008 striped bass abundance estimate fell by over 40% since 2006. Similarly, spiny dogfish abundance has increased 10 fold since 1999 and has remained high thereafter. During this recent period (1999-2008) of declining weakfish productivity, fishing mortality (FW) and discard mortality (Fdisc) rates remained low and relatively stable, indicating that the recent drop in weakfish productivity did not coincide with rising exploitation. The strong positive correlation (Pearson r = 0.91, P <0.0001) between the recent rise in weakfish juvenile mortality (Z0) and rising striped bass and spiny dogfish abundance further suggests that the recent emergence of a weakfish recruitment bottleneck at age 0 was largely due to enhanced predation by these two finfish predators. By contrast, discard mortality rates on small (< age 2) weakfish remained low and stable after 1999 during which juvenile mortality (Z0) rose steadily. Third, the residual patterns in all Logistics and Gompertz model runs that included only fishing effects (landings) produced inordinately low overfishing thresholds (Fmsy, Bmsy), poor precision around the estimates, and the residuals exhibited a pronounced serial correlation over time, clearly indicating model misspecification. However, when the predation term (Tpred), reflecting the joint predation by striped bass and spiny dogfish, was added to the models, the fit of the models to weakfish surplus production and biomass dramatically improved, the precision and magnitude of Fmsy and Bmsy rose to more plausible levels (Fmsy > 0.45), and, most importantly, the direction of the residuals over time shifted to a more random pattern and were therefore free of model misspecification.

When the equilibrium overfishing thresholds (Fmsy = 0.72, Bmsy = 17,009 mt) from the S-H model are considered, ages 1+ biomass weighted fishing mortality (FW) on weakfish exceeded the

estimated Fmsy threshold of 0.72 in most years from 1981 to 1992. Weakfish coast-wide biomass (mt) exceeded the biomass threshold (Bmsy = 17,009 mt) in 1981 and 1982, but biomass fell quickly below Bmsy thereafter. When more stringent management regulations were enacted after 1991, fishing mortality (FW) fell by 50 to 70% and biomass began to rise toward the Bmsy threshold. However, weakfish biomass fell unexpectedly after 1999 to the lowest level in the time series in 2007 despite the fact that fishing mortality rates remained below Fmsy in most years from 1998 to 2008 (exception: 2002).

Although most of the statistical and empirical evidence given here and elsewhere (see TOR #6 this assessment) supports the Predation Hypothesis, other factors such as unreported commercial and recreational landings, disease, toxins and parasitism cannot be ruled out at this time to explain the annual production loss of between 3,000 and 5,000 mt of weakfish. There is no evidence thus far that would link recent increases in disease, toxins and parasitism to the recent failure of weakfish. There has been a recent rise in sea surface water temperatures along the Atlantic coast (Oviatt 2004), but these analyses indicated that water temperature shifts were not significantly (P <0.05) linked to recent increases in weakfish juvenile mortality (Z0), nor in the decline in weakfish surplus production and stock biomass. It is possible that an enormous upsurge in unreported weakfish landings and commercial and recreational discards took place between 1996 and 2008 to account for the estimated 3,000 to 4,000 mt annual loss of weakfish surplus production, but a recent upsurge in unreported landings seems unlikely for several reasons. First, if the sources of this rapid upsurge in unreported weakfish landings and discards are thus far unknown, it would be nearly impossible to remove this source of mortality without closing virtually all inshore fishing activity between North Carolina and Rhode Island. Second, if a recent rise in unreported landings and discards resulted in the recent weakfish stock collapse, we would expect that other finfish stocks with a similar temporal and spatial distribution as weakfish (i. e. Atlantic croaker and summer flounder) to be likewise depleted. But Atlantic croaker and summer flounder stocks have either grown or have remained relatively stable from 1998 to 2008. Third, if a recent rise in weakfish unreported landings caused the recent weakfish stock collapse, all of the statistical and empirical evidence presented elsewhere in this assessment on enhanced predation would have to be regarded as a mere coincidence. Finally, unreported landings in the order of 3,000 to 4,000 mt annually are equivalent to about 5 times the current (2007-2008) known landings and estimated discards used in this weakfish assessment. The possibility that such an astronomical rise in unreported landings and discards took place after 1998 and then remain unnoticed by port agents, enforcement and management agencies seems remote. The management implications associated with a rise in predatory mortality on weakfish are also discussed.

C3.6 Major findings of TOR 6 – Evaluate AIC-based hypothesis testing of fishing and additional predation-competition effects using multi-index biomass dynamic models with and without prey-based, predator-based, or ratio dependent predator-prey extensions. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice.

The most reliable estimates of trends or values of F from the previous weakfish *Cynoscion regalis* assessment indicated it had been modest since at least 1995, while weakfish abundance and

surplus production declined to low levels, most likely due to increased natural mortality. Hypotheses featuring fishing, environmental conditions, forage abundance, competition, and predation were examined and two strong covariates emerged: Atlantic menhaden *Brevoortia tyrannus* (forage) abundance and predation- competition from striped bass *Morone saxatilis*.

In this 1981-2006 assessment of weakfish, biomass dynamics models were used to test multiple hypotheses about fishing alone or fishing plus interactions with striped bass (alone or influenced by Atlantic menhaden) as the cause of the recent failure of weakfish recovery. Spatial, temporal, and diet overlaps were sufficient for interactions. Logistic and Gompertz production functions were considered and six models were developed for each function. These models considered fishing alone and fishing in combination with five predator-competitor functions; three models considered prey-based Type I, II, and III functional responses, while two explicitly mimicked depensatory mortality by considering additional natural mortality solely as a function of striped bass biomass or as a function of striped bass biomass and the ratio of Atlantic menhaden to striped bass biomass.

We used three exploitable biomass indices (EBI; indices of weakfish 250 mm or greater) to evaluate biomass dynamics during 1981-2006: mid-Atlantic private/rental boat catch per trip (WRI; as biomass and estimated from MRFSS), DE (1990-2006), and NJ (1989-2006) trawl survey EBIs. Biomass dynamic models used total weight of aggregated harvest (NMFS estimates) and discards by both the commercial and recreational fisheries. The Weakfish Technical Committee (WTC) considered all estimated commercial discards to have died and updated ratio-based estimates of commercial discards using the general method of De Silva (2004), but used all years combined rather than annual estimates (latter were variable and imprecise). Discard sampling was not conducted until 1994 and market-related discard ratios estimated for 1994-2002 were used to estimate weakfish discards in prior years. Estimates after 1993 used total discard ratios (market + regulatory). Recreational discard losses were estimated as MRFSS number discarded*mean weight *mortality. The MRFSS does not estimate weight of released weakfish and discard mean weight (0.15 kg) estimated from MRFSS 2004-2007 headboat surveys was substituted for harvest mean weight used in the previous assessment. Weakfish hook-and-release experiments produced dichotomous mean mortality estimates ($\approx$3% or 15%) and 10% release mortality was adopted by the WTC.

We used Akaike information criteria adjusted for small sample size, AIC_c, to evaluate the 12 hypotheses. AIC_c indicated a 98% chance that the Gompertz production model with a depensatory function relating D_t (weakfish biomass lost to striped bass predation-competition) to the biomass of age 2+ striped bass and the ratio of menhaden to striped bass biomass was best (Gompertz Depensatory Ratio model or GDR) given the data. This model explained 90% of the variation in EBI. Fishing only models were poor choices for describing biomass dynamics of the data, ranking seventh and ninth out of twelve.

Annual (year t) estimates of F_t and M_{pt} (instantaneous natural mortality rate due to striped bass predation-competition) were combined to estimate Z_{pt} (total mortality excluding residual M). Loss of weakfish biomass per unit of striped bass biomass (D_t / P_t) was estimated. Biomass estimates (B_t) provided a basis for estimating surplus production and production that accounted for losses due to striped bass predation and competition.

Equilibrium biological reference points (EBRPs) were estimated (F_{msy} and B_{msy}) and two approaches were used to estimate predator-competitor reference points (NBRPs) for mortality when predation-competition losses were included: total mortality at maximum sustained yield (Z_{msy}) and non-equilibrium F_{msy} (or $F_{psyt} = F_{msy} - M_{pt}$). The former simply involved renaming equilibrium

estimates of F_{msy} as Z_{msy} in models with predator-prey terms and comparing it to Z_{pt}. Although biomass dynamic models do not provide SSB thresholds explicitly, the early maturity of weakfish allowed B_t / K to serve as a proxy for MSP to compare to the target and threshold. Amendment 4 to the Interstate Fishery Management Plan for Weakfish (ASMFC 2002) lists SSB that is 20% of an unfished stock as a maximum spawning potential (MSP) threshold and 30% as a target.

BRPs were F_{msy} or Z_{msy} = 0.48 and B_{msy} = 18,941 mt based on r = 0.48, K = 51,521 mt. Jackknifing and bootstrapping indicated all parameters were precisely estimated and parameter values of the base run were very close to their medians, indicating minimal bias. Several approaches were used to investigate sensitivity of model parameters and estimates of Z_t, F_t, and B_t. Estimates of B_t were standardized to K (B_t / K) and estimates of Z_t and F_t were standardized to Z_{msy} in sensitivity analyses. Sensitivity of model parameters to data from the beginning or ending of the time-series was tested by removing blocks of data and rerunning the model. Biomass estimated in 1981 for the 1981-2006 time-series was greater than K and we ran a version of this model with B_{1981} constrained to be less than K for comparison. Sensitivity to different assumed recreational discard mean weights was tested as well because of their importance in estimating WRI.

Overall, differences in B_t / K, Z_t / Z_{msy}, and F / Z_{msy} stabilized by 1983 among all initial time-block removal treatments. Biomass dynamics of weakfish were portrayed similarly. We chose to keep the results of the unconstrained GDR (all years), but removed 1981 and 1982 estimates from consideration. Removing up to three years from the end of the time-series or changing mean weight of recreational discards had little impact.

Biomass dynamic modeling indicated weakfish biomass in 2006-2007 was depleted well below its threshold, the stock was not overfished based on equilibrium F_{msy}, but was subject to high natural mortality that eroded the safe level of fishing. The proxy for MSP (2007 MSP = 6%) was far below the 20% threshold in Amendment 4, while high negative values of F_{psyt} (non-equilibrium F_{msy}) and surplus production indicated that complete (and unlikely) elimination of harvest and bycatch would not be sufficient to end the decline. Production persisted at a modest level in recent years, although it was falling gradually. Depensatory mortality, driven by high striped bass biomass and a low ratio of Atlantic menhaden to striped bass (an indicator of low feeding success on striped bass' main prey), appeared the most likely explanation for increasing natural mortality that undermined recovery given the data, hypotheses, and models developed. The GDR indicated that as menhaden have become less abundant and striped bass more-so, striped bass searching has lead to increasing encounters with vulnerable-sized weakfish (up to 400 mm) searching for smaller prey-fish (anchovies and age 0 menhaden) that are also found in diets of the largest striped bass. High natural mortality of weakfish in recent years was derived from very low loss per striped bass applied over a large striped bass biomass and was independent of weakfish biomass over the years modeled. Fishing played a secondary role in recent biomass dynamics. Striped bass predation-competition dominated weakfish biomass dynamics after overharvesting had been controlled in the early 1990s. Predation-competition from striped bass has increasingly eroded weakfish surplus production and F_{msy} reference points and there seems little chance of restoring weakfish by manipulating its fisheries alone because F has become a low fraction of total mortality (20% by 2006). At this time, leverage for manipulating weakfish may mostly reside in the menhaden to bass ratio; however, it can be difficult to predict the effects of fishing or culling policies from fairly simple representations of predation processes.

Additional regression analyses reinforced the high potential for striped bass, menhaden, and weakfish linkage. Predicted mean weight of weakfish at 340 mm had undergone a significant decline during 1992-2006, and was negatively related to striped bass biomass and positively related

to the ratio of menhaden to striped bass biomass. This would be consistent with the expected effects of intense competition. Estimates of D_t / P_t were strongly related to field-based estimates of feeding success of striped bass in coastal VA and NC during winter. During 1959-2006, weakfish commercial harvest and the DE PSD Q+ length quality index closely followed the ratio of Atlantic menhaden to striped bass and correlations were strong ($\rho \approx 0.82$). These associations indicated that this ratio was important in dynamics of weakfish beyond the period covered by the GDR.

Additional Gompertz biomass dynamic models mimicked various commercial bycatch scenarios (additional losses increasing as functions of time, a constant multiple of bycatch estimates, or constant additional weight) that imposed additional losses after 1995 to reflect regulatory discards. Best models of bycatch scenarios invoked about the same additional biomass loss as estimated by GDR. The failure of recovery since the late 1990s cannot be attributed to overfishing unless bycatch and under-reported catches were much greater than estimated, growing from about 3-4 times the estimates in 1996 to 15-20 times by 2006. If results of hypothetical bycatch scenarios are taken at face value, then weakfish regulations created this massive boost in discards and represent a colossal management failure. Implementation of further conservation measures short of a coast-wide moratorium on all Atlantic coast fisheries would not minimize this nominal discard problem. There is no evidence available thus far of an Atlantic coast fishery capable of generating additional unreported weakfish losses of this magnitude.

C3.7 Major findings of TOR 7 –Review evidence for constant or recent systematic changes in natural mortality, productivity, and/or unreported removals.

During development of the 2006 weakfish stock assessment, the WTC noticed an unexpected decline in stock size at low levels of fishing mortality that had previously resulted in stock growth. Further investigation indicated that weakfish had been experiencing increased predation/competition that could be a major driving force in stock dynamics. This section presents updates on analyses investigating these multispecies hypotheses and evaluates additional sources of data that provide support for the theory that decreased production has contributed to recent stock declines.

One of the major concerns the WTC has expressed regarding the age structured modeling is the assumption of constant natural mortality across all ages and years. As an alternative, the WTC has investigated trends in relative fishing mortality which is not influenced by assumptions regarding M. Relative F, calculated as the ratio between annual harvest and an annual index of abundance, was rescaled to instantaneous rates using a scalar vector based on a short time series of F rates from the converged portion of the ADAPT VPA. Rescaled relative F follows a similar pattern to VPA estimated F from 1982 to 1998. Following 1998, however, F_{VPA} follows a nearly exponential increase while relative F remains stable at moderate levels. This discrepancy suggests that estimates of natural mortality in the ADAPT input are inaccurate.

In addition, biomass dynamic modeling was conducted to investigate possible environmental and ecosystem covariates that might be influencing weakfish stock dynamics. Two similar but independent analyses were pursued using simple (fishing only) and extended (fishing plus ecological covariates) production models. Production models incorporating only fishing effects provided poorer model fits and greater parameter uncertainty relative to the extended models. Of the extended models, the "predation" model indicated that weakfish natural mortality had increased during the last decade as the biomass of two predators (striped bass and spiny dogfish) had increased, while the "forage" model concluded that increased striped bass abundance in conjunction with a decline in their primary forage (menhaden) had resulted in increased weakfish mortality. In

both cases, the increased losses from predation/competition eroded weakfish productivity while fishing mortality remained relatively low.

Finally, the WTC investigated additional data sources that could indicate changes in weakfish productivity. Commercial landings, as a proxy for weakfish abundance, were correlated with the Atlantic Multidecadal Oscillation, a time series of sea surface temperatures from the North Atlantic that exhibit a 65 to 70 year oscillation. Strong correlations between the two time series over more than 70 years suggest that weakfish abundance may be influenced by environmental parameters such as temperature. Qualitative analysis extends the association several more decades. Weakfish food habit data obtained from the NEFSC Food Habits Database showed a shift in prey items from forage fish and large invertebrates to smaller invertebrates and an increased incidence of empty stomachs during the 1990s. The incidence of empty stomachs is strongly correlated with total mortality estimated by ADAPT VPA. These data are consistent with shifts in weakfish diets observed during the same period in the Chesapeake Bay, and suggest that weakfish productivity may have been compromised during the 1990s as primary prey items were less available.

Taken as a whole, there are several pieces of evidence that suggest that weakfish productivity underwent a shift during the 1990s, either directly (*e.g.* predation) or indirectly (*e.g.* shift in environmental conditions).

C3.8 Major findings of TOR 8 – Estimate biological reference points using equilibrium and non-equilibrium assumptions and evaluate stock status relative to these BRPs.

Weakfish are currently managed relative to reference points developed under Amendment 4 of the Weakfish Fishery Management Plan. Reference points were updated for this assessment using a spreadsheet based model using age-specific input values and length-weight-age relationships. Fishing mortality reference points were found by solving for F rates that provided spawning stock biomass of 30% (target) and 20% (threshold) relative to unfished stock. Assuming constant natural mortality of M = 0.25 and partial recruitment equal to the average of the most recent three years estimated by ADAPT, new fishing mortality reference points were estimated as $F_{target} = F_{30\%} = 0.28$ and $F_{threshold} = F_{20\%} = 0.42$, a decrease of 10% and 16% respectively relative to Amendment 4 reference points. Similarly threshold biomass declined nearly 10% to the new estimate of $SSB_{20\%} = 10,179$ MT. Comparison of VPA based parameter estimates to these reference points indicates that weakfish are overfished and overfishing is occurring.

The WTC has expressed concern with a few aspects of the ADAPT VPA, including a prominent retrospective pattern in recent years and the use of a constant input natural mortality rate, both of which could affect reference point estimation. The retrospective pattern generally overestimates abundance for the last five years, resulting in underestimated partial recruitment. Using a partial recruitment vector from a more stable portion of the time series (1999 to 2001) decreased F reference points even further relative to Amendment 4. In addition, recent analyses indicate that natural mortality has increased substantially over the last decade to values approaching M = 1.0 or higher. Reference points calculated using an input of M = 0.8 and the more stable partial recruitment values provides estimates of a fishing mortality target of $F_{30\%} = 0.78$, and a fishing mortality threshold of $F_{20\%} = 1.20$. Under these assumptions, F_{2007} is below the target mortality rate, but is likely underestimated given the observed retrospective pattern.

In addition to the equilibrium reference points calculated based on ADAPT output, both of the biomass dynamic models investigated during this assessment produced estimates of equilibrium and

non-equilibrium MSY reference points. The full Steele-Henderson model with predation terms provided estimates of equilibrium $F_{MSY} = 0.72$ and $B_{MSY} = 17,009$ mt. The best fit model evaluated under the forage hypothesis resulted in equilibrium reference points of $F_{MSY} = 0.48$, and $B_{MSY} = 18,941$ MT. Nonequilibrium reference points calculated by both models indicate that F_{MSY} has declined in recent years as predation/competition has eroded the amount of weakfish productivity "available" to fishing. Total biomass (as proxy for SSB) has declined to less than 10% of MSY thresholds in 2007 for both models.

C3.9 Major findings of TOR 9 – Review stock projections and impacts on the stock under different assumptions of fishing and natural mortality.

Projections were conducted for each of the three main models investigated. Relative to output from the ADAPT VPA, biomass was projected for 25 years using the AgePro (version 3.1) module of the NFT Toolbox. Multiple runs were conducted assuming a range of fishing and natural mortality values assuming recruitment followed an empirical distribution of Age 1 numbers estimated through the full time series of the VPA. All projection runs resulted in increased biomass over time but reached different asymptotic values depending on assumptions regarding F and M. Under the assumption of constant $M = 0.25$ and a harvest moratorium, SSB is projected to increase to more than 275,000 MT by the year 2032; however, increasing mortality to $M = 0.75$ results in a biomass projection of 45,000 MT by 2013. Projections at given M values are lower if harvest is allowed.

For the predation hypothesis model, weakfish relative spawning stock biomass (TSSB) projections were made from 2010 to 2020 following the imposition of a simulated coast-wide moratorium ($F = 0$) to harvest beginning in 2009. Given the uncertainty surrounding the current and future trend in natural mortality (M) estimates, the following three scenarios that bracket a wide range of possibilities were examined with the weakfish Harvest Control Model (HCM) following a simulated 2009 moratorium to harvest: 1) M is fixed at 0.25 throughout the time series (1980-2020) as in ADAPT, there is no recent rise in trophic impacts on weakfish productivity, and recent (1999-2008) fishing mortality (F) has remained high ($F > 1.0$) as per ADAPT; 2) there is a moderate rise in M (from 0.25 to 0.65) from 1999 to 2020 due to predation but the magnitude of predatory mortality is less than predicted by the Steele-Henderson Model, and recent (1999-2008) F estimates have risen to moderate (0.7 to 1.0) levels; and 3) M on weakfish after 1997 has risen four-fold in magnitude (from 0.25 to 1.0) as per the Steele-Henderson Model, and fishing mortality (F) rates have remained relatively low ($F < 0.50$) from 1996 to 2008 as per the Index-based Analysis. In Scenario #1 under a relatively low and fixed natural mortality ($M = 0.25$) throughout the time series (1980-2020), the HCM predicted that a moratorium to all weakfish harvest ($F = 0$) enacted in 2009 and thereafter would lead to rapid TSSB recovery that would approached the Bmsy threshold by 2020. In scenario #2 under the assumption of a moderate rise in M from 0.25 to 0.65 after 1997, the HCM predicted that a moratorium ($F = 0$) enacted in 2009 would result in some measurable TSSB rebuilding by 2020, but the magnitude of stock growth would fall far short of the Bmsy threshold. In scenario #3 under a pronounced rise in M from 0.25 to 1.0 ostensibly due to enhanced predation, the HCM predicted that a moratorium to harvest in 2009 and thereafter would result in little if any TSSB rebuilding by 2020.

For the forage hypothesis model, jackknife and bootstrap estimates of parameter estimates from the best fit model and biomass in 2007 were projected to evaluate the effect of fishing restriction scenarios through 2015. Three fishery management scenarios were portrayed: an

approximation of the minimum F reduction in Amendment 4 ($F_{recover}$), F = 0.2 (a literal interpretation of Amendment 4), and a moratorium.

Stock recovery was not possible under $F_{recover}$ and F = 0.2, and there was about 1.4% chance of recovery under moratorium conditions for bootstrap runs and 0% chance for jackknife moratorium scenarios. It should be noted that estimates of F since 2003 have been at or below $F_{recover}$. Jackknifing and bootstrapping indicated greater than 90% chance that weakfish biomass would fall to zero by 2015 even under a moratorium if trophic conditions prevailing in 2006 continued. These projections are excessively grim and are considered a worst case scenario.

C3.10 Major findings of TOR 10 – Make research recommendations for improving data collection and assessment.

The list of prioritized research recommendations presented in the 2008 Weakfish Fishery Management Plan Review was updated by the WTC. Several recommendations were identified as completed or under investigation, while several new recommendations were identified and added to the list.

C4.0 Introduction

This is the first update to the weakfish stock assessment since 2006 when the assessment was peer reviewed through the Atlantic States Marine Fisheries Commission (ASMFC) External Peer Review process. The 2006 assessment updated the stock through the 2003 fishing season. The current assessment includes harvest data and survey indices through 2007.

C4.1 Management Unit Definition

Weakfish stocks on the U.S. Atlantic coast are managed through the ASMFC Interstate Fishery Management Plan (FMP) for Weakfish. Under this FMP, weakfish are managed as a single unit stock throughout their coastal range. Historically, all states from Massachusetts through Florida had a declared interest in the species. Currently, however, Massachusetts, Connecticut, South Carolina, Georgia, and Florida maintain *de minimus* status, and are therefore exempt from certain regulatory and monitoring requirements.

C4.2 Management History

The following is a brief review of the history of weakfish fishery management through the ASMFC. Additional details are provided in the various amendments and addenda to the original Weakfish Fishery Management Plan, which are available online at *www.asmfc.org*.

The first fishery management plan for weakfish was implemented by ASMFC in 1985 to address stock declines, bycatch concerns, the lack of sufficient data for management, and interstate user conflicts (Mercer 1985). The management measures under the FMP were voluntary, and no state implemented the full set of management provisions outlined in the FMP.

Amendment 1, adopted in 1991, established a target fishing mortality rate of $F_{20\%} = 0.34$ (Seagraves 1991). This target was to be achieved by a 52% reduction in directed harvest over the course of four years, as well as a 50% reduction in bycatch mortality in the penaeid shrimp fisheries by 1994. Although adoption of turtle excluder devices (TEDs) in the shrimp fishery led to bycatch reductions, none of the states with directed fisheries adopted the full complement of regulations recommended in the amendment.

Continued concern regarding the status of the weakfish stock was a major impetus for the development and passage of the Atlantic Coastal Fisheries Cooperative Management Act (1993),

which made compliance with ASMFC fishery management plans mandatory for member states. Following the Act's passage, the ASMFC approved Amendment 2 to the Weakfish FMP for implementation in April 1995 (ASMFC 1994). The provisions of Amendment 2 were mandatory and included harvest control strategies such as a 12" (305 mm) total length (TL) minimum size, maintenance of existing minimum mesh sizes, and a 50% shrimp trawl bycatch reduction requirement by 1996. Fishing mortality would be reduced in a stepwise fashion, with a 25% reduction in weakfish fishing mortality in 1995 followed by a 25% reduction in exploitation in 1996.

Following implementation of Amendment 2, below average fishery catch rates and spawning stock biomass continued, along with a lack of older fish. In response, Amendment 3 was developed to reduce fishing mortality to $F = 0.50$ by the year 2000, restore an expanded age structure, and restore fish to their full geographical range (ASMFC 1996). Commercial fisheries were regulated by a combination of season and area closures, mesh regulations to minimize harvest of fish less than 12" TL, and stricter requirements for bycatch reduction devices (BRDs). The minimum recreational requirements were a 12" TL minimum size limit and four fish possession limit. States were allowed to implement alternate size and bag limit regulations if they were conservationally equivalent to the minimum requirements. Bag limits were not required for minimum sizes of 16" TL or greater.

In 2000, a peer review of a stock assessment with data through 1998 indicated that weakfish biomass was high and fishing mortality rate was below the target of $F = 0.50$. Despite being ahead of schedule, it was recommended that low fishing mortality rates be continued to maintain an appropriate spawning biomass and promote expansion of stock size and age composition. Also as a result of the assessment, the WTC recognized several inconsistencies between management practices and stock dynamics. These could only be addressed through the development of a new FMP amendment. In the meantime, however, Addendum I to Amendment 3 was passed to maintain existing regulations until approval of the new amendment.

Weakfish stocks on the U.S. Atlantic coast are currently managed under Amendment 4 to the FMP (ASMFC 2002). Reference points established in Amendment 3 were too high to ensure sufficient spawning stock biomass, and the reference period used to develop recreational management measures represented an overexploited stock (insufficient abundance of older, larger individuals). In response to these concerns, Amendment 4, implemented in July 2003, established new fishing mortality and spawning stock biomass reference points, and adjusted the reference period to a period of greater stock health (1981 to 1985). Amendment 4 established new reference points: a fishing mortality target of $F_{target} = F_{30\%} = 0.31$; a fishing mortality threshold of $F_{threshold} = F_{20\%} = 0.5$; and a spawning stock biomass threshold of $SSB_{threshold} = SSB_{20\%} = 14,428$ metric tons (MT; 31.8 million pounds). A fishing mortality rate greater than $F = 0.5$ constitutes overfishing, and the stock is considered overfished if SSB is less than 14,428 MT. If it is determined that the weakfish stock is overfished, Amendment 4 requires ASMFC to implement measures to rebuild the population within six years (1½ generations).

Several addenda have been passed to improve management capabilities under Amendment 4. Addendum I was passed in December 2005 to modify biological sampling targets. Addendum III (May 2007) modified bycatch reduction requirements to maintain consistency with the South Atlantic Fishery Management Council. Of greater significance was passage of Addendum II in February 2007. A stock assessment conducted in 2006 showed a significant turn of events from previous assessment results (see full discussion in Section **C4.3**, **Assessment History**). Model results indicated that weakfish stocks were at historic low levels, and that fishing mortality was a relatively minor component of total mortality. Projection analyses indicated that even with a full moratorium on harvest, stock rebuilding would occur slowly at best without a significant decrease in

other sources of mortality. To minimize overall mortality without unduly penalizing fishermen, and to prevent expansion of the fishery in the event the stock begins to rebuild, Addendum II required that all states: 1) maintain current minimum sizes, 2) implement a recreational six fish bag limit (except South Carolina which was in the process of implementing a 10 fish limit), and 3) impose a 150 pound commercial bycatch trip limit (except *de minimus* states). Addendum II also established landings-based triggers to re-evaluate these criteria.

C4.3 Assessment History

Early stock assessment analyses for weakfish were conducted using a variety of virtual population models, such as the Murphy VPA (Vaughan et al 1991) and CAGEAN. The first peer reviewed assessment analyzed data through 1996 using Extended Survivor Analysis (XSA). The peer review was conducted in 1997 by the Stock Assessment Review Committee (SARC) at the 26th Northeast Regional Stock Assessment Workshop (SAW; NEFSC 1998a). The SARC had concerns with the XSA model runs and requested updated runs as well as exploratory CAGEAN and ADAPT model runs. These were conducted during the SAW, but there was insufficient time to fully review the results. As such, the SARC did not endorse the point estimates of F and SSB. Regardless, all models used indicated that SSB was increasing rapidly and fishing mortality rates were decreasing rapidly. SSB had increased an average of 22.5% per year since 1991, while F had decreased an average of 21.4% per year since 1990 (NEFSC 1998a). The SARC concluded that continuation of low fishing mortality rates and good recruitment would allow for age expansion to a point comparable to that observed in the early 1980s.

The subsequent assessment, including data through 1998, was peer reviewed at the 30th SAW/SARC in 1999 (NEFSC 2000). The stock was assessed using the ADAPT VPA as recommended by the 26th SARC. Ages in recent years were taken from otoliths, which required a conversion of scale-based ages from earlier years to otolith-based ages. The approved VPA run included only indices from the core abundance area (New York to North Carolina). The model indicated that fishing mortality rates had declined to 0.21 in 1998, well below both $F_{MAX} = 0.27$ and $F_{MSY} = 0.6$. In addition, SSB had increased to about 39,000 MT, approximately 55% of an unfished stock. The SARC did observe a noticeable retrospective pattern, which overestimated stock size and underestimated fishing mortality in the last few years. Regardless, the SARC concluded that results of the ADAPT VPA could be used to calculate biological reference points, and that figures illustrating the expanded size and age composition of weakfish would be useful for developing management advice.

A stock assessment update was conducted in 2002 (with data through 2000) using the SARC approved methodology (ADAPT VPA with tuning indices from the core area; Kahn 2002). The assessment showed that estimates of fishing mortality decreased further to $F = 0.12$, while SSB increased to over 50,000 MT. Although this assessment was not peer reviewed, the WTC expressed concern about a strong retrospective pattern that resulted in high levels of uncertainty in recent year estimates. The WTC recognized poor biological sampling of commercial catches, commercial discards, and recreational discards as a likely source of much of this error, especially when coupled with the assumption of error-free catch at age estimates used by ADAPT. Estimates of F and SSB were "corrected" by multiplying each parameter by the average amount each parameter changed in recent years with the addition of more data. Even so, the corrected estimate of $F = 0.23$ was substantially below $F_{Target} = 0.31$, and corrected SSB = 35,000 MT was more than double SSB$_{Threshold}$ = 14,428 MT.

In 2003, the Weakfish Stock Assessment Subcommittee (WSAS) began preparation for a

2004 peer review through the 40th SAW. Model results using the SARC approved methodology still exhibited a strong retrospective pattern, and results from both ADAPT VPA and biomass dynamic models indicated the stock was at very high levels (carrying capacity in the case of the biomass dynamic model; see Uphoff 2005c) with very low fishing mortality. The WTC was concerned that these results were not consistent with low catch rates and diminishing size structure being observed by commercial and recreational fishermen targeting weakfish.

For these reasons, the WSAS deemed the ADAPT VPA methodology as insufficient to characterize the weakfish resource and proceeded to investigate alternative assessment methods. Although the revised weakfish assessment was incomplete at the time of the 40th SAW, the SARC agreed to review the work and provide guidance on issues that were impeding the progress of the assessment (such as the inconsistency between survey indices and fishery-dependent indices of abundance and catch at age).

The SARC agreed with the WSAS that the results of the work in progress, although using the same approach as the SARC-approved assessment in 1999, were not suitable for management (*e.g.* Cook 2005). The SARC indicated that it felt the problem was conflicting data, and expressed skepticism about the reliability of some survey indices, especially the Northeast Fishery Science Center Fall Survey. Recommendations from the SARC proved to be useful, and some were incorporated into the stock assessment. The assessment was also expanded to include some alternative approaches previously explored by the WSAS in the 2002 update process (ASMFC 2006a, Part A).

The stock assessment was completed in February 2006 and submitted to ASMFC for evaluation through the ASMFC External Peer Review process. The Peer Review Panel consisted of four fisheries biologists with expertise in population dynamics and stock assessment methods. The Panel did not endorse the statements regarding weakfish stock status and identified several issues that required additional work or attention by the WTC before the report would be suitable for management purposes (ASMFC 2006a, Part B). In particular, the Panel had concerns regarding stock structure, age composition data, and fishery discards.

The Weakfish Management Board directed the WTC to address the issues identified by the Peer Review Panel. Specifically, the Management Board tasked the WTC to further investigate stock structure and discards; determine agreements and disagreements among the assessment report, the peer review panel report, and the 40th SARC report; and provide an account of the implementation of recommendations from the 40th SARC.

In August 2006, the WTC provided a response to these tasks (ASMFC 2006a, Part C). Based on these responses, the WTC's analyses, and significant evidence, the Management Board accepted the following five points for management use:

1. The stock is declining;
2. Total mortality is increasing;
3. There is little evidence of overfishing occurring;
4. Something other than fishing mortality is causing the stock decline, and;
5. There is a strong chance that regulating the fishery will not, in itself, reverse the stock decline.

In December 2008, the NEFSC held the 2008 Northeast Data Poor Stocks Workshop (DPSW) to evaluate reference points for stocks with limited data. Although weakfish is not considered a data poor stock, the current assessment was reviewed as a work in progress. The intent of the review was

not a formal evaluation of the work (*i.e.* not a "Pass/Fail" evaluation), but a cursory review of the general data, methods, and preliminary results to provide guidance on ways to improve the analysis. The review panel expressed several concerns with the analysis, particularly with input data and lack of empirical data to support the species interaction hypotheses (Miller et al 2009). The WTC has reviewed the report and made appropriate modifications to the analyses for this final product. The weakfish portion of Miller et al (2009), along with the WTC's responses is provided in Appendix C-1.

C4.4 Life History

Weakfish, *Cynoscion regalis*, are estuarine dependent members of the drum family (Sciaenidae). Commonly occurring from Massachusetts to Florida, weakfish are most common in the Mid-Atlantic region from North Carolina to New York (Wilk 1979). Common migration patterns for weakfish include spring spawning movement into estuaries and bays and reverse movements out of the estuaries in the fall either offshore and/or to more southern regions to overwinter (Bigelow and Schroeder 1953, Wilk 1979). The spawning season is protracted and begins in the spring taking place in coastal estuaries and bays. Weakfish mature early (age-1) and the maximum recorded age using otoliths is seventeen years.

C4.4.1 Reproduction

Weakfish spawn in the nearshore and estuarine areas of the coast. In North Carolina, the spawning season occurs from March to September and peaks from April to June (Merriner 1976). Spawning further north occurs later and is less protracted. In Chesapeake Bay, spawning has been documented to occur from May to August (Lowerre-Barbieri et al. 1996). From Delaware Bay to New York spawning occurs from May to mid-July (Shepherd and Grimes 1984).

Early to mature, weakfish spawn multiple times in a season and have indeterminate fecundity (Lowerre-Barbieri et al. 1996). Reproductive work in Chesapeake Bay during 1991 and 1992 found that 90% of age-1 weakfish were mature. Batch fecundity ranged from 75,289 to 517,845 eggs/female and significantly increased with both total length and somatic weight (Lowerre-Barbieri et al. 1996). During 1999 and 2000, a study conducted in Delaware and Chesapeake Bays noted no increase in the size at maturity (168 mm) from that previously estimated despite a marked increase in the overall population size (Nye and Targett 2008). Similarly, most (97%) age-1 fish were mature. Both studies indicated that spawning frequency and batch fecundity vary by year and that these two variables act jointly to determine total egg production (Nye and Targett 2008). Nye and Targett (2008) also noted that despite maturing early, age-1 weakfish spawned less frequently, arrived later to the estuary and had lower batch fecundity than did older fish, likely resulting in an overly optimistic assumption about the contribution of age-1 fish to the overall reproductive success of the stock. This is currently amplified by the fact that larger, older fish comprise a small proportion of the overall population.

C4.4.2 Feeding Habits

Spatial and temporal variation in juvenile weakfish diet has been observed in studies conducted in Delaware Bay and Chesapeake Bay (Hartman and Brandt 1995, Grecay and Targett 1996, R. Latour, Virginia Institute of Marine Science, pers.comm). In Delaware Bay, Grecay and Targett (1996) found mysid shrimp to dominate the diet of juvenile weakfish collected in 1986, while the bay anchovy dominated the diet of juvenile weakfish collected in the Chesapeake Bay in the early 1990s. Latour et al. (*in review*) examined the diet of weakfish from the Chesapeake Bay

from 2002 to 2004 and found that mysid shrimp were an important component of the diet not only in juvenile weakfish but also for adults in contrast to earlier diet studies of Chesapeake Bay weakfish (Hartman and Brandt 1995).

Older weakfish typically have been shown to become increasingly piscivorous with age, with Atlantic menhaden or other clupeids comprising a significant portion of the diet of older weakfish (Merriner 1975, Hartman and Brandt 1995). Recently, Latour et al (*in review*) found mysids and bay anchovy to comprise a significant portion of the diet of all age weakfish in Chesapeake Bay, with Atlantic menhaden comprising only a small portion of the diet of age-5+ weakfish. Differences in the two studies were attributed to different sampling methods or temporal changes in the abundance of prey items between the time periods of the two studies. The low prevalence of other sciaenids, spot and croaker, in light of high commercial landings of those species was also noted (Latour et al. *in review*). Section **C10.0** of this report examines changes in reported weakfish diet in detail.

C4.4.3 Age and Growth

Weakfish growth is rapid during the first year, and age-1 fish typically cover a wide range of sizes, a result of the protracted spawning season. After age-1, length becomes much less reliable as predictor of age due to an increasing overlap in lengths occurring over several age groups. Lowerre-Barbierri et al. (1995) found length at age to be similar between sexes with females attaining slightly greater length at age than males. Pooled across sexes, they reported observed TL's for weakfish collected in the spring (1989-1992) from Chesapeake Bay to be 176, 311,412, 510, 558, and 631 mm for ages 1-6 respectively. Growth was described using the Von Bertalanffy growth model ($r^2 = 0.98$; L_∞=919; K=0.19; t_0=-0.13). The L_∞ reported for other regions were similar: 893 mm TL for Delaware Bay (Villoso 1990) and 917 mm FL for North Carolina (Hawkins 1988) with the exception of Shephard and Grimes (1983) which reported lower L_∞ estimates for Chesapeake Bay (686 mm TL) and North Carolina (400 mm TL). The historical maximum age recorded using otoliths is 17 years. The fish was collected from Delaware Bay in 1985. The maximum age used in previous assessments considers Tmax to be 12 years (Kahn 2002). The world record weight for hook-and-line was recently captured on May 6, 2008 off of New York (8.67 kg). Weakfish have undergone large fluctuations in landings since the late 1800s, and there are reports from New England in the 1700s of decadal-scale abrupt shifts in abundance (Cushing 1982). Similar to landings, historic changes in the maximum size and age have been reported with weakfish typically obtaining their maximum size and age during periods of higher landings (Lowerre-Barbieri et al. 1995). More recent growth rates have slowed to the point that mean lengths at age of adults are several centimeters shorter than they were in the early 1990s (Lowerre-Barbieri et al. 1995, Kahn 2002). Weakfish weight at age plummeted in the 1990s by nearly half for 3-5 year-old weakfish.

C4.4.4 Natural Mortality

The 26th SARC recommended that M = 0.25 be used in modeling constant natural mortality and this value was used in prior assessments on weakfish (NEFSC 2000, Kahn 2002, ASMFC 2006a, Part A). A recent review of indirect, life history based methods to estimate natural mortality was conducted for weakfish along the Atlantic coast. This review found age-independent M rates ranging from 0.25 to 0.68 using various methods (Munyandorero 2008). The majority of the estimates ranged from 0.25 to 0.38. Where applicable, Tmax was set to age-12. Age-dependent estimates using the Lorenzen method resulted in M estimates ranging from 0.44 to 0.13*year^{-1} for age-1 to age-8 fish respectively.

The most recent age-structured VPA utilized data through 2003 and was run under the

assumption of constant M (0.25). Kahn et al. 2006). During the mid-1990's weakfish underwent a series of regulatory changes through Amendment 3 to the ASMFC weakfish FMP. After this period, the stock had an initial positive response including an increase in abundance, an expansion in the number of older fish in the population and an increase in the maximum observed age (up to age-12). Since that time, weakfish stocks have declined with landings currently at historic lows in both the recreational and commercial fisheries. These declines have occurred in spite of increased regulations to protect the fishery and no apparent increasing trend in either the effort of directed fisheries or the occurrence of discards. As a result, the WTC investigated possible causes for the recent decline in weakfish landings and age-structure. While the cause of the decline is not readily apparent, it does not appear that fishing mortality is not likely the primary culprit. Relative F estimates for the most recent years showed that fishing mortality had been low and stable from 1995 through 2003, in sharp contrast to the ADAPT estimates of a consistent increasing trend in F over that period. Consequently, the WTC does not support all results of the age-structured VPA under constant M assumptions, although the WTC does consider the VPA results with the reconstructed total mortality (based on adding the input M to the annual estimates of fishing mortality) to be reasonably accurate, along with the estimates of declining biomass through 2000. However, due to a retrospective bias extending back three years from the terminal estimate, the WTC regarded estimates for the last three years (2001-2003) as unstable and unreliable for manangement purposes. As a result of these analyses and conclusions, the WTC determined that the assumption of constant M was violated. Several investigations into possible systematic changes in M have been explored.

C4.4.5 Stock Definitions

The weakfish range extends along the Atlantic coast from Massachusetts to southern Florida, although strays are occasionally found as far as Nova Scotia, Canada and into the eastern Gulf of Mexico. Primary abundance occurs between New York and North Carolina. Within their range there is evidence of multiple stocks. Munyandorero (2006; see ASMFC 2006a, Part C) provides a concise but thorough overview of available information on weakfish stock structure. The following is an excerpt.

> Investigations of weakfish population structure along the US Atlantic coast have been undertaken through tagging, meristic, morphological, life history, genetic and otolith chemistry. The conclusions reached are conflicting. While Crawford et al. (1988), Graves et al. (1992) and Cordes and Graves (2003) did not detect genetic differentiation within the weakfish population, Chapman et al. (unpublished report) found that weakfish are made up of a series of overlapping stocks, without complete panmixia. Non-genetic studies found evidence of existence of multiple weakfish sub-populations (e.g., Nesbit 1954; Shepherd & Grimes 1983, 1984; Scoles 1990) or important spatial structure of the weakfish population (Thorrold et al. 1998, 2001). Mark-recapture, meristic, morphological and life-history studies (e.g., review by Crawford et al. 1988) indicated that weakfish could be partitioned into sub-stocks…

Crawford et al. (1988) recommended that weakfish be managed as separate northern and southern stocks, while Graves et al. (1992) recommended management of a single unit stock. The WTC reviewed the available information and reached the following conclusions.

- Evidence of stock structure exists
- Data is inadequate to define stock structure, and there is enough potential mixing that pinpointing the location of a north/south split is not possible at this time
- If a north to mid-Atlantic subpopulation is in serious decline, this does not warrant a north-south split based on conservation concerns (ASMFC 2006a, Part C).

Based on those recommendations, the ASMFC Weakfish FMP continues to manage Atlantic coast weakfish as a single unit stock throughout their coastal range.

C4.5 Habitat Description

Weakfish are found in shallow marine and estuarine waters along the Atlantic coast. They can be found in salinities as low as 6 ppt (Dahlberg 1972) and temperatures ranging from 17° to 26.5^{o} C (Merriner 1976).

Like many other North Atlantic species, weakfish exhibit a north-inshore/south-offshore migration pattern, although in the southern part of their range they are considered resident. Shepherd and Grimes (1983) observed that migrations occur in conjunction with movements of the $16\text{-}24^{o}$ isotherms. Warming of coastal waters during springtime triggers a northward and inshore migration of adults from their wintering grounds in the Mid-Atlantic. The spring migration brings fish to nearshore coastal waters, coastal bays, and estuaries where spawning occurs.

Weakfish spawn in estuarine and nearshore habitats throughout their range. Principal spawning areas are from North Carolina to Montauk, NY, although spawning and presence of juveniles has been observed in the bays and inlets of Georgia and South Carolina (Lunz and Schwartz 1969, Mahood 1974, and Powles and Stender 1978, all as cited in Mercer 1985). Larval and juvenile weakfish generally inhabit estuarine rivers, bays, and sounds, but have been taken in freshwater (Thomas 1971) and as far as 70 km offshore (Berrien et al 1978). Mercer (1983) found that juveniles are most prevalent in shallow bays and navigation channels and are commonly associated with sand or sand/grass bottoms.

Weakfish form aggregations and move southward and offshore as temperatures decline in the fall. Important wintering grounds for the stock are located on the continental shelf from Chesapeake Bay to Cape Lookout, North Carolina (Merriner 1973, as cited in Mercer 1985).

C4.6 Fishery description

C4.6.1 Overview of fisheries

C4.6.1.1 Commercial Fishery

Records of commercial weakfish landings are available back to 1950 through the National Marine Fisheries Service (NMFS) website. From 1950 through the 1960s commercial landings ranged from about 2,000 to 4,000 metric tons (MT) per year (Figure C4.6-1). Beginning in 1970, reported landings exhibited a dramatic increase to a record high of more than 16,000 MT in 1980. From 1982 to 1988, landings fluctuated between approximately 8,000 and 10,000 MT. Since 1989, landings have declined continuously, except for a brief increase to about 4,000 MT in the mid- to late-1990s. Estimated commercial harvest in 2007 is the lowest on record at approximately 388 MT.

The general pattern for the commercial fishery is that in the winter, most landings occur in North Carolina as that state's fishery targets the overwintering aggregation off the North Carolina

coast. This seasonal fishery has accounted for the largest proportion of the commercial landings on the Atlantic coast. In spring, weakfish migrate back to spawning ares, primarily estuaries in North Carolina and the Mid-Atlantic states. Fishing occurs on the migrating fish along the coast and then concentrates on estuaries for the remainder of spring and summer, from Pamlico Sound in North Carolina through Peconic bay on eastern Long Island, New York. In mid-summer, some larger fish arrive in southern New England, including Rhode Island and Connecticut. With fall, weakfish leave estuaries and begin their fall migration south to the overwintering grounds, and are targeted as they move down the coast.

Three states - New Jersey, Virginia, and North Carolina - have consistently accounted for 70 to 90% of the coastwide commercial harvest since 1950 (Table C4.6-1; Figure C4.6-2). North Carolina has predominated with nearly 37% of the coastwide harvest over the last ten years, while Virginia and New Jersey have averaged 25.6% and 17.0% respectively. During this same time period, New York has accounted for nearly 10% of coastwide harvest.

From the mid 1950s to the early 1980s landings from the trawl fishery generally accounted for 50 to 70% of commercial landings (Figure C4.6-3). Beginning in the early 1980s, harvest from trawlers began a gradual decline, and recently have accounted for approximately 20% of total harvest. Conversely, between 1979 and 1987, landings from gillnets increased from around 10% of annual harvest to 45% of annual harvest, and have remained relatively stable since that time. Over the entire time period, pound nets and haul seines have each averaged between 10 and 20% of total harvest annually, despite declining trends.

Discarding of weakfish by commercial fishermen is known to occur, and discard mortality is assumed to be 100%. The first quantitative analysis of weakfish discards is provided in de Silva (2004). Most discarding occurs in conjunction with two gears (trawls and gillnets) and a limited number of target species. Prior to 1994, discards are assumed to have occurred for non-regulatory reasons because few regulations were in place to limit the fishery. Since 1994, both regulatory and non-regulatory discarding has occurred. Regardless, population removals as a result of commercial discarding appear to be minor relative to harvest, even in recent years as harvest has decreased.

From 1982 to 1990, estimates of biomass of discarded weakfish generally declined from around 600 MT to 200 MT, where it remained stable for several years (Table C4.6-2). With the implementation of state specific regulations in 1993, and mandatory coastwide measures in 1995, estimated discards increased dramatically to near 1,000 MT in the mid 1990s. Except for the time series maximum of approximately 1,150 MT in 2001, discards decreased steadily from 1996 to 2004, and have remained stable around 200 MT since that time. From 1982 to 1999, age 1 fish generally dominated the discards, with a few exceptions in the mid 1990s when age 0 fish outnumbered age 1 fish. Since 2000, however, discards of age 2 and 3 fish have exceeded those of age 1.

Changes in catch per unit effort (CPUE) over time can be indicative of changes in abundance/availability or a shift in target species. Where available, commercial weakfish CPUE was examined to evaluate trends in fishery performance over time. Data were evaluated for all trips where weakfish were harvested, as well as only trips that harvested more than 150 pounds (68 kg) of weakfish, in an attempt to include directed trips during an open season.

North Carolina has historically been the largest commercial harvester of weakfish, and CPUE data are available back to 1994. When all positive weakfish trips (only trips where weakfish were caught) are considered, all but two (flounder trawl and haul seine) of the eight fisheries exhibited a strong decline in CPUE (Figure C4.6-4). When only trips harvesting 150 or more pounds are considered, CPUE typically declines, though generally less severely than when trips with bycatch

allowances were included (Figure C4.6-4). In Virginia, commercial weakfish CPUE has fallen since the late 1990s, particularly since 2002 (Figure 4.6-5). Delaware gillnet fishery CPUE (positive trips) declined in the early 1990s, rebounded in the late 1990s, and has declined steadily since 2000 (Figure C4.6-6). Data are available from the Potomac River pound net fishery for 1976 to 1980 and 1988 to 2007 (Figure C4.6-7). Between 1976 and 1980, CPUE was high. By 1988, CPUE had dropped to less than one-third of peak levels, dropping below 20% of the 1980 peak between 1988 and 1993. CPUE rebounded to between 20 and 40% of the time series peak during 1994-2002 and then fell sharply (Figure C4.6-7).

Although there is some regional and temporal variability, commercial CPUE generally present a consistent pattern of recovery during the late 1990s and then a severe decline in the early 2000s. Commercial CPUE since the mid to late 1990s corresponds well with model estimates of population trends, fishery independent and fishery dependent abundance indices, and observed size and age structure. The WTC are not aware of any changes in regulations or fisherman behavior that would explain the recent decline in CPUE.

C4.6.1.2 Recreational Fishery

Recreational harvest statistics for the weakfish fishery are available on the NMFS Marine Recreational Fishery Statistics Survey (MRFSS) website for the period 1981 to 2007 (*www.st.nmfs.gov/st1/*). From 1981 to 1988, the number of weakfish caught and the number harvested fluctuated without trend between 2 million and around 11 million fish; however, during this same time period, harvested weight generally declined from around 7,259 MT to 2,722 MT (Figure C4.6-8). Nearly 90% of all fish caught were retained during these years.

From 1989 to 1993, catch (numbers) and harvest (numbers and weight) remained relatively stable. Catch fluctuated between 1.6 and 2.2 million fish, while harvest ranged between 0.95 and 1.8 million fish and 499 to 998 MT. Percentage of total catch that was harvested during this period decreased from around 90% to less than 50%.

In 1994, weakfish catches increased and averaged around 6 million fish until 2000. Harvest numbers increased to a lesser extent and fluctuated between approximately 1.5 and 2.5 million fish. Harvest weight also increased to 1,814 MT during this period. By 2003, catches and harvest had declined to at or near time series minimum and have remained relatively stable. In 2007, total catch was 2.01 million fish, with a harvest of 0.58 million fish and 313 MT. Since 1994, harvest has accounted for approximately 20-40% of all fish caught.

Recreational harvest has been dominated by the five Mid-Atlantic states between New Jersey and North Carolina (Table C4.6-3; Figure C4.6-9). New Jersey dominated landings in most years, averaging 35% of coastwide harvest across the time series. Virginia consistently produced greater than 20% of coastwide landings from 1981 to 1992 but has since declined, averaging about 10% over the last five years. Since 1995, several states have each had periods of substantial landings, with Delaware contributing 20-30% of total harvest for 1995-1998, Maryland accounting for approximately 25% from 1999 to 2001, and North Carolina averaging 22.5% from 2003 to 2007.

Recreational discard mortality is assumed to be 10% of all discarded fish based on catch-and-release experiments with weakfish and the closely related spotted seatrout (*Cynoscion nebulosis*; (*e.g.* Murphy et al 1995, Malchoff and Heins 1997, Swihart et al 2000, Duffy 2002, Gearhart 2002). Weakfish hook-and-release experiments produced dichotomous mean mortality estimates, either near 3% or 15%, and 10% release mortality was adopted by the WTC. From 1981 to 1989, harvested weakfish averaged 89% of total catch (numbers). Even with high landings, discard losses during this period were lowest of the time series, with all but one year having fewer than 100,000

fish discarded coastwide (Figure C4.6-10). Between 1989 and 1995, harvest fell to 27% of catch, and discard losses increased to more than 400,000 in 1995. Harvest rebounded slightly to 41% of catch in 1997 and 1998, but dropped back to between 20-40% since 1999. Despite relatively stable release rates since 1995, discard losses have varied greatly due to large interannual fluctuations in catch. Discard losses peaked at approximately 500,000 fish in 1996 and 2000, but have since decreased along with catch. For the last five years, discard losses have ranged between 135,000 and 225,000 fish.

Throughout the time series, total removals have been dominated by commercial and recreational harvest (Figure C4.6-11). Removals were greatest during the early portion of the time series, averaging 13,500 MT between 1981 and 1988. Between 1989 and 1993, removals dropped off quickly to 4,000 MT. The next few years showed a slight rebound to a peak of 6,500 MT in 1998. Since then, removals have declined continuously to the time series minimum of only 852 MT in 2007. Combined commercial and recreational discard losses were generally less than 5% of total removals prior to 1993. Discarding increased rapidly following implementation of management measures. Regardless, discard losses have averaged less than 20% of total removals since 1994.

C4.7 Current status

Throughout the 1980s and early 1990s, weakfish stocks experienced unsustainably high fishing mortality rates, which led to a decline in abundance into the 1990s. Fishing mortality rates declined during the early 1990s, and an increase in biomass was evident during the mid to late 1990s. The 2006 stock assessment indicates that fishing mortality has remained low under Amendment 4, yet weakfish biomass had dropped back to near historic low levels by 2003. Available evidence indicates that factors other than fishing mortality were the primary cause for biomass decline (ASMFC 2006a, Part A). A peer review of the stock assessment did not endorse the statements regarding weakfish stock status and identified several issues that required additional work or attention by the WTC before they would support its use for management purposes (ASMFC 2006a, part B). In particular, the review panel had concerns regarding stock structure, age composition data, and fishery discards. In August 2006, the WTC responded to the peer review panel's concerns (ASMFC 2006a, Part C). Based on these responses, the Technical Committee's analyses, and significant evidence, the Weakfish Management Board accepted the following five points for management use:

1. The stock is declining;
2. Total mortality is increasing;
3. There is little evidence of overfishing occurring;
4. Something other than fishing mortality is causing the stock decline, and;
5. There is a strong chance that regulating the fishery will not, in itself, reverse the stock decline.

C5.0 Evaluate biases, precision, uncertainty, and sampling methodology of the commercial and recreational catch (including landings and discards) and effort. (TOR #1)

C5. 1 Commercial

C5.1.1 Landings

Commercial landings data were taken from two sources. Where available, state-specific

harvest records collected through a mandatory reporting system were considered the most reliable source for landings. Unfortunately, not all states require mandatory reporting of weakfish harvest. In such cases, landings estimates were obtained from the NMFS commercial landings database, available through the NMFS Office of Science and Technology, Fisheries Statistics Division website (*http://www.st.nmfs.gov/st1/*). Although estimates are available from NMFS, it is not mandatory to report weakfish harvest to NMFS, so these records (like those of most species) may be incomplete. Discrepancies between NMFS reported harvest and state reported harvest under mandatory reporting suggest that NMFS harvest estimates for weakfish are a potential source of uncertainty. In an attempt to quantify the uncertainty between the two reporting systems, state reported landings from Delaware and Virginia were compared to federally reported landings in these two states for the period 2004 to 2006. Combined across all gears, NMFS reported landings for a given year differed from state landings by less than 10% in all instances except Virginia in 2006, when the difference exceeded 23% (Figure C5.1-1). However, when evaluated at the gear level, more than one-third of all year/state/gear combinations differed by more than 20%, and in three cases exceeded 100% (Figure C5.1-2). Generally speaking, then, annual estimates of weakfish harvest reported by state and federal agencies are relatively consistent when combined across all gears, but the allocation of landings by gear are less certain. Lack of state landings data and discrepancies between state and federal estimates are not confined to just weakfish, but are observed in most state managed species.

C5.1.1.1 *Biological samples*

Commercial biological samples include lengths, weights, and ages from state-specific port sampling programs. Commercial samples were combined with similar data from recreational and fishery independent sources to develop length-weight relationships and age-length keys (ALK) for use in the estimation of commercial catch at age.

Lengths

Commercial length data were used for two primary purposes: the development of length-weight equations and characterizing the distribution of commercial catches by length and age. Because a combination of both total length and fork length data were available, lengths were standardized to fork length measurements. A conversion factor was developed using data pooled across all sources in 2004 to 2006 that reported both total length and fork length. Total length (mm) was converted to fork length (mm) using the equation

$$FL = (TL + 5.8106) / 1.0437$$

Length-weight equations were developed as in the 2006 assessment (ASMFC 2006a, Part A). Length and weight data from all sources were pooled, and relationships were developed by region/year/season. Sample sizes and parameter estimates are presented in Table C5.1-1.

Characterization of fishery catch at size was conducted using similar procedures as the 2006 assessment (ASMFC 2006a, Part A), following methods described by Quinn and Deriso (1999). Commercial harvest estimates and length samples were stratified by region/year/season/state/gear. Landings not identified to specific gear were pooled at the region/year/season level and classified as "Other." In addition, fisheries (state-gear combination) with minimal landings (< 1% of region/year/season total) generally had insufficient sample size (see below) to characterize that fishery. These cells were also classified as "Other."

For northern region cells with significant landings (≥ 1% of region/year/season total) and sufficient sample size, harvest weight was converted to harvest number at size using predicted weight at length (from region/year/season specific length-weight equations) and observed length frequency distributions. Mean weight at length was estimated using the appropriate length-weight equation. Sample weight at length (average weight at length multiplied by number of samples at length) was then divided by total sample weight (weight at length summed across lengths) to determine weight distribution by size. These proportions were then multiplied by total harvest weight for that state/gear/season to determine harvest weight at length. Harvest weight at length was then converted to number at length by dividing weight by average weight at length.

For cells with significant harvest and insufficient sample size, the same methods were employed using length data borrowed from an appropriate substitution cell with sufficient sample size. Finally, the "Other" category was split into harvest from states with a 12" minimum size and states with a 16" minimum size. For each region/year/season/minimum size category, all available samples greater than the minimum size from the respective states were used to convert "Other" harvest weight to harvest number at size.

For the southern region, characterization of the fisheries was done slightly differently. Commercial sampling in North Carolina includes collection of both lengths and weights, so it was possible to develop an average fish weight by gear and season for each fishery. The average weight was divided into the harvest weight to estimate number harvested. The number harvested was then partitioned to catch at size using the length frequency distribution of the samples. Florida, the only other southern region state with commercial landings, collects no biological samples. Biological sample data from North Carolina were used as proxy information for Florida landings.

Results of the fishery specific catch at size analyses were combined across states and gears within a region to develop estimates of commercial harvest number at size by region/year/season.

Uncertainty in estimated harvest number at size could be introduced from two primary sources: 1) sample size of fish used to characterize a fishery, and 2) substitution of data from alternate strata for fisheries with insufficient sample size. These are discussed in more detail below.

Sample size and ratio of samples per metric ton of landings were used to evaluate adequacy of sampling intensity (Table C5.1-3). It was determined that a minimum sample of 30 lengths per stratum (region/year/season/state/gear) was required to adequately characterize a fishery. The minimum of 30 samples is much lower than sample sizes suggested in the literature for characterizing a population (Miranda 2007; Vokoun et al. 2001). Insufficient sampling would tend to introduce uncertainty into the catch at size estimates; however, these studies recommend sample sizes necessary to meet an objective (characterizing entire population) much different than the current analysis (characterizing harvest of specific gear). However, Miranda (2007) notes that distributions with a smaller size range require a smaller sample size. Considering minimum size limits and gear selectivity, the sample size required to characterize a fishery is likely lower than those to characterize an entire population. For example, Burns et al (1983) indicate that 100 fish per 200 MT of landings appears adequate to characterize many of the northeast U.S. groundfish stocks. ASMFC (2005) requires states to collect a minimum of 6 weakfish lengths per MT of landings, although not all states have been able to comply with this due to staff/funding constraints and difficulty obtaining samples. For this assessment, there are 141 cells (defined as region/year/season/state/gear combination with significant landings) that require expansion. All have more than 0.5 fish per MT (*i.e.* 100 fish per 200 MT), and only 13 (9.2%) have fewer than 6 fish per MT of landings. Sample size in the cells with less than 6/MT ranges from 30 to 419 fish. Also, 41 of the 141 cells have fewer than 30 samples and require substitution from alternate strata.

A higher minimum sample size requirement would increase the number of cells that require substitution, which could increase uncertainty in length and age distributions for those cells. Therefore, a sample size of 30 fish is considered an appropriate compromise between adequate lengths to fully characterize a fishery and the need to substitute data.

For strata with insufficient (< 30) length samples, data were usually substituted from the same region/year/season, but were sometimes substituted from another state and/or gear (Table C5.1-2). Care was taken to minimize differences in gear selectivity, and when necessary substituted data were truncated to account for differences in minimum size requirements between the two states. Regardless, the WTC recognizes that substituted data are not always representative of the stratum to which they are applied, resulting in uncertainty in the length frequency distribution of the catch. Of greatest concern are the geographic differences in fish size, coupled with the general lack of samples north of Delaware. Minimum size limits and average size of harvested fish were much larger in the northern portions of the range. When commercial samples from these states were insufficient and data were substituted from more southern states, the effect was an underestimation of the proportion of large fish in the harvest. In 2006, New Jersey began collecting biological samples from their commercial fisheries. By estimating length and age distributions of New Jersey commercial harvest using New Jersey data and data substituted from other cells, Brust (2009) shows that data substitution resulted in fish ages 1 to 5 being overestimated in the coastwide CAA by less than 5%, but fish ages 6 and older were underestimated by up to 52%. The 6+ plus group as a whole was underestimated by 9.9% in 2006 and nearly 32% in 2007 (Brust 2009). Improved sampling in the northern region since 2005 will better characterize landings from this region, decreasing uncertainty in catch at size estimates.

Ages

The principal use of age data is in the development of ALKs. Sample sizes of ages by year, season, and source are provided for recent years in Table C5.1-3. During the 1980s, ages were based on scale samples. During the 1990s, otoliths became the principal method for aging weakfish. For the 1998 stock assessment, scale-based ages in previous years were converted to otolith-based ages using a scale-otolith conversion matrix (similar to an ALK) based on direct comparison of approximately 2,300 samples (Daniel and Vaughan 1997; NEFSC 1998b). Uncertainty in either aging method, as well as in the scale-otolith conversion matrix would be propagated through the catch-at-age matrix.

Age-length data from all available sources (commercial, recreational, fishery independent) were pooled by region/year/season to develop stratum specific age-length keys (four keys per year) as described by Vaughan (2000). Length intervals with missing information in the keys were filled by either averaging age distribution for lengths above and below, substitution from another stratum, or interpolation. These filling procedures could lead to uncertainty in catch at age if the substituted age distributions are not representative of the cells into which they are substituted.

Once the age-length keys were complete, catch at size estimates by region/year/season were converted to catch at age using the appropriate age-length key. Catch at age estimates were pooled across regions and seasons to develop annual estimates of commercial catch at age.

C5.1.2 Discards

Discard mortality of weakfish by commercial fisheries was assumed to be 100%. The first quantitative analysis of weakfish commercial discards was provided by de Silva (2004). Most discarding occurs in conjunction with two gears (trawls and gillnets) and a limited number of target

species. Several methods to estimate discards were investigated, including effort based estimates, regression analysis, and ratio extrapolation. Effort data were not available for all states and years to develop effort based estimates. Regression estimation was conducted, but the predictive models fit poorly ($r^2 \sim 0.08$) and were considered inappropriate for use in the assessment. Ratio estimates work best when there is evidence of a positive linear relationship between the response and explanatory variable. Although there was no evidence of such a relationship in several of the gear-species combinations evaluated, it was determined that ratio extrapolation provided the most reliable estimates of discards from the methods investigated. With this method, discards were calculated using seasonal, annual, and multi-year (all years combined) ratios. It was determined that the multi-year estimates provided the most reliable estimates, and this method was selected as the final estimation methodology. Discards in the southern region (North Carolina to Florida) were considered insignificant, so commercial discards were only evaluated for the northern region. A full description of the methods is provided in de Silva (2004) and summarized below.

Data from the NEFSC observer database (1994 to present) were queried to obtain haul level estimates of weakfish discard weight and target species harvest weight for each gear/target species combination. Discard and harvest weight estimates were pooled across all years by gear and target species to develop gear/species specific weakfish discard ratios (calculated as summed weakfish discard weight divided by summed target species harvest weight for each gear and target species). The gear/species specific ratios were then applied to annual harvest estimates of that gear/species combination (from the NMFS commercial harvest website) to estimate total weakfish discard weight in that fishery. Weakfish discard length frequency data by gear (all species combined) were used to convert discard weight to discard numbers at size. Annual discard estimates by gear were partitioned into seasonal estimates by using the proportion of annual landings by season and gear from the NMFS landings database. Gear-species-season discards at size were summed across gears and species to determine total seasonal discards at size.

Prior to 1994, discards are assumed to have occurred for non-regulatory reasons only because few regulations were in place to limit the fishery. Unfortunately, observer data are not available prior to 1994. As such, data from 1994 forward were subset by regulatory and non-regulatory discards, and the method described above was used to develop discard ratios for non-regulatory discards only for 1994 forward. These ratios were applied to annual estimates of gear/target species harvest for 1981 to 1993 to estimate non-regulatory discards during this time period.

For the current assessment, commercial discard estimates were updated for 1994 forward in order to include additional years of data and significant updates to data used in the 2006 assessment. Using the general method of de Silva (2004), haul level data for the same gear and target species combinations were used to evaluate annual, multiyear (5 and 7 year), and all-year (1994 to 2007) weakfish discard ratios. Because of concerns with high interannual variability and uncomfortably large standard errors with estimates based on short time groupings, the all-year ratio estimation method was selected. Ratio-based estimates of weakfish discard weight were made for butterfish, long-fin squid, summer flounder and weakfish for trawl gear, and Atlantic croaker, bluefish, spiny dogfish, and weakfish in the gillnet fishery. Discard weight was converted to numbers at size using observed and re-sampled length frequencies, then converted to discard number at age using the appropriate region/year/season age-length key described in section **C5.1.1 Landings**. Catch at age estimates were summed across regions and seasons to determine annual commercial discard catch at age.

There are several potential sources of uncertainty with the commercial discard estimates. Sample size of observed hauls with weakfish discards is low in many years for some gear-species

combinations (Table C5.1-4), and discard ratios vary greatly (see *e.g.* Table 13 in de Silva 2004). Combining data across years improves sample size but may mask temporal trends in discarding. In addition, in order to convert discard weight to number, biological data from discarded fish were pooled across species for a given gear type, and/or resampled from observed fish to attain a minimum sample size (30 fish). For both concerns, larger sample sizes would provide more reliable estimates of discard rates.

Another source of uncertainty is the gear-species combinations used. The initial analysis found 14 gear-species combinations with substantial weakfish discards (de Silva 2004). However, many of these species are often captured together. To minimize the potential for duplicate counting of discards when discard ratios were multiplied by total harvest of each of the gear-target species combinations, principal component analysis was conducted to identify species groupings. Discard ratios were then multiplied by harvest of only one of the species within a group. Selecting a different species from the group would result in different estimates of total discards.

During the NEFSC DPSW in December 2008, reviewers were concerned that the methods used to estimate discards could result in substantial uncertainty. However, the reviewers could not agree on the direction of the bias. At least one reviewer was concerned that discards were overestimated because multiplying a discard ratio for a given target species by total harvest of that species includes harvest when that species was not the target species (*i.e.* harvest estimates applied to ratios were overestimated). This was addressed to some degree by minimizing the number of gear-species combinations through PCA. On the other hand, a second reviewer was concerned that the number of gear-species combinations was too limited and may have missed historic fisheries with large weakfish discards. The WTC is aware of these potential sources of bias. Unfortunately, the methods used to estimate weakfish commercial discards are constrained in many ways by the amount of available data. The methodology has been used in other ASMFC assessments (i.e. Atlantic croaker) and is a more comprehensive analysis than most assessments attempt at quantifying discards.

C5.2 Recreational

C5.2.1. *Landings*

Recreational landings data were obtained from the NMFS Marine Recreational Fishery Statistics Survey (MRFSS) database, which is available through the NMFS Office of Science and Technology, Fisheries Statistics Division website (*http://www.st.nmfs.gov/st1/*). MRFSS provides estimates for three subcategories of catch, including observed harvest (Type A), unobserved harvest (*e.g.* filleted before observation, discarded dead; Type B1) and discarded alive (Type B2). Estimates of harvest were developed for each region/year/season combination as a sum of observed and unobserved harvest (Type A + B1). Because sand seatrout and weakfish are indiscernible except through genetic analysis, MRFSS estimates in Florida are for the *Cynoscion* complex of weakfish, sand seatrout, and their hybrids. Estimates for true weakfish in Florida were found by multiplying MRFSS estimates by the proportion of true weakfish observed by Tringali et al. (2006) before combining with other southern region states.

Precision in recreational catch and harvest estimates are calculated as a percent standard error (PSE). Lower values indicate better precision than higher values, and most commonly caught species generally have PSEs less than 20% (NMFS 1999). As such, estimates with PSE values less than 20% are generally considered "acceptable" (NEFSC 1998b). Estimates of weakfish harvest were relatively precise (Table C5.2-1), with PSEs less than 15% for most years since 1982

(numbers) or 1983 (weight). Estimates of the number of weakfish discarded were less precise in the beginning of the time series, but PSEs have been below 20% for all but one year since 1986. However, a recent review of the survey identified several potential biases and inadequacies of the sampling and estimation methodologies (NRC 2006; see http://www.nap.edu/catalog.php?record_id=11616). These include the inability to interview anglers at private access sites; the increasing use of household cell phones which are unavailable to the telephone sampling frame; reliance on unverified assumptions; and differences in statistical properties of data collected through different survey methods. The effects of these biases on estimates of recreational catch, harvest, and discards can not easily be quantified, leading to uncertainty in MRFSS recreational estimates. This uncertainty applies to all catch types over the entire time series, which has been collected using the same general methodology throughout.

C5.2.1.1 Biological samples

Biological samples collected by MRFSS include lengths and weights of a subsample of Type A fish. No ages are collected from the recreational fishery. Recreational length-weight data were combined with similar data from commercial and fishery independent sources to develop length-weight relationships (see section **C5.1 Commercial**). Length data were also used to partition harvest into harvest at size. Because of small sample sizes (Table C5.1-3), length observations were pooled by region/year/season to expand harvest estimates at the same level of stratification. Unlike commercial data, estimates of recreational harvest in numbers are directly available from the MRFSS website. Catch at size was estimated as the proportion measured at size by stratum multiplied by the estimated harvest (number A+B1 fish) for that stratum. Number at size was converted to number at age using the appropriate region/year/season age-length key described in section **C5.1.1 Landings**. The number of length samples collected by MRFSS is above the criterion of 100 lengths per 200 MT of landings (Burns et al 1983).

C5.2.2 Discards

Estimates of the number of recreational weakfish discards (Type B2 fish) were obtained from the MRFSS database. Estimates in Florida were corrected for weakfish-sand seatrout hybridization using ratios reported by Tringali et al. (2006). In previous assessments, discard mortality was assumed to equal 20% of all discards. However, based on a review of available data, the WTC has decreased the discard mortality to 10% (*e.g.* Murphy et al 1995, Malchoff and Heins 1997, Swihart et al 2000, Duffy 2002, Gearhart 2002).

Prior to 2004, discarded fish were not sampled. Since 2004, MRFSS has collected lengths of discarded fish from the for-hire sector (party and charter boats). As such, this is the first assessment for which recreational discard length frequencies are available. Observed length frequencies were applied to discard mortality estimates to estimate the number of dead discards at size. For the northern region, this was done by year and season. Due to low sample size, southern region samples were pooled across seasons to develop annual length frequencies. Number at size was converted to number at age using the appropriate region/year/season age-length key described in section C5.1.1 Landings.

C5.3 Catch at Age Matrix

The catch-at-age matrix for 2004-2007 was developed using the same general procedures as previous assessments. Catch at size from the four major sources of removals (commercial harvest, commercial discards, recreational harvest, recreational discards) were converted to catch at age

using the appropriate region/year/season age-length key. Results were pooled across regions, seasons and sectors to estimate total annual removals at age (Table C5.3-1).

As described in each of the pertinent sections, there are several potential sources of uncertainty in the overall catch at age estimates. These include inaccurate harvest/discard estimates as a result of under/over reporting or inappropriate survey methods; insufficient sample size to characterize length distributions; substitution of data from alternate cells in the catch at size characterization and ALKs; errors in aging techniques or the scale-otolith age conversion; and others. Attempts have been made to quantify some of these error sources; however, the extent of uncertainty associated with each of these sources, and their cumulative effect, remains largely unknown. A persistent cumulative trend in either direction would result in inaccurate catch at age estimates and may influence assessment results. However, the sources of potential error and the methodologies used to develop the catch-at-age matrix in this assessment are similar to those used for other ASMFC species assessments that have passed peer review.

C6.0 Evaluate precision, geographical coverage, representation of stock structure, and relative accuracy of the fisheries independent and dependent indices of abundance. Review preliminary work on standardization of abundance indices. (TOR#2)

C6.1 Aged fishery independent surveys

C6.1.1 NEFSC Bottom Trawl Survey

The National Marine Fisheries Service (NMFS) Northeast Fishery Science Center (NEFSC) conducts seasonal trawl surveys between Nova Scotia and Cape Hatteras. Stratified random sampling is conducted using a #36 Yankee otter trawl equipped with roller gear and a 1.25 cm mesh codend liner. The survey covers a large portion of the geographic range of weakfish, including their "core" distribution area (NEFSC 2000) of New Jersey to North Carolina. Despite the extended latitudinal range, the survey is not capable of sampling in shallow waters, and few sites are conducted in waters less than 9 m. In addition, the survey does not sample the South Atlantic portion of the range.

Weakfish are infrequent in the winter, spring, and summer surveys, but are commonly intercepted in the fall during their offshore migration. Because weakfish are rarely caught in this survey north of New Jersey, the 30th SAW/SARC recommended developing an index of weakfish abundance using only strata from the south end of Long Island to Cape Hatteras during the fall survey. Indices at age are developed by applying annual survey specific length frequency data to the annual mean catch per tow and then applying either survey specific ALKs (when available) or the pooled northern region late season ALK (see section **C5.0)**. Because this survey occurs in the fall, true ages are increased by one year to develop an index of abundance on January 1 of the year following the survey (*e.g.* fall 1997 age 0 fish are treated as January 1, 1998 age 1 fish).

The annual mean catch per tow appears nearly cyclical, with relative peaks in abundance generally every 4 to 6 years (Figure C6.1-1). From 1982 through the mid 1990s, mean catch per tow cycled without trend, generally ranging between 40 and 120 fish per tow. Beginning in the mid 1990s, abundance gradually increased to a time series maximum of approximately 500 fish per tow in 2004. During 2005 – 2007, abundance decreased to about 200 fish per tow, but increased in 2008 to over 300 fish per tow. Standard errors (SE) follow a similar pattern as CPUE, with an overall cyclical pattern and a gradual increase beginning in the mid 1990s. During the early portion of the time series, SE varied between approximately 10 and 50, increasing to a peak of 90 in 2004.

Coefficient of variation (CV; SE as a ratio of the mean) has varied without trend between approximately 10 and 40% since 1990.

The survey index is dominated by age 1 fish (age 0 fish progressed to age 1), although fish have been observed out to age 6 (Figure C6.1-1). Age distribution was greatest in the early 1980s, but was truncated to predominantly ages 1-3 by the early 1990s. Age distribution expanded somewhat during the late 1990s as the stock began rebuilding as a result of management measures, but has since declined to primarily ages 1-4.

The WTC evaluated many of the age aggregated survey indices and found the NEFSC weakfish index performed poorly compared to others (see section **C10.0** of this report for a summary of the analyses). The timing of the survey, along with the highly contagious distribution of weakfish, leads to high variability between years and between tows within a year. Proportional stock density analysis indicates that the survey's ability to capture large weakfish, even when they are present, is poor. Using correlation analysis, it was found that the index was not well correlated with the other indices or the converged portion of the VPA, and was negatively correlated with harvest trends. Finally, catch curve analysis determined abundance of several year class increased over time (*i.e.* negative mortality; see Table 3 of ASMFC 2006a, Part A). Efforts to develop a more representative index of weakfish abundance (*e.g.* geometric mean, percent positive tows, etc.) were unsuccessful. Although the survey has several advantages, such as being the longest running and widest ranging fishery independent survey, the WTC concluded that the NEFSC fall survey is not suitable for use as an index of relative abundance in the assessment.

C6.1.2 New Jersey Ocean Trawl Program

New Jersey has conducted a stratified random trawl survey in nearshore ocean waters (to 27 meters depth) from Ambrose Channel (entrance to New York Harbor) to Cape Henlopen Channel (entrance to Delaware Bay) since 1988. The survey originated as bi-monthly cruises, but since 1991 has consisted of five cruises per year (January, April, June, August, and October). Strata are nearly identical to those used by NEFSC in this region (New Jersey's northern- and southern-most strata are truncated at New Jersey state boundaries). The gear used is a two-seam trawl with a 25 m headrope and 6.4 mm bar mesh codend liner. Due to funding constraints, several different vessels have been used to conduct the survey.

The geographic range of the survey is limited to nearshore ocean waters of the species distribution within the northern and southern borders of New Jersey. The survey occurs within the region sampled by the NEFSC trawl survey. The use of a smaller vessel, however, allows the New Jersey survey to provide better coverage in shallow waters.

The majority of weakfish are observed during the June, August and October cruises, although catches in June are inconsistent. During previous assessments, an index of weakfish abundance was developed using the August and October cruises. However, recent work has shown that the August-October index is a poor indicator of weakfish abundance (see section **C10.0** of this report). As with the NEFSC index, tow-level and annual mean catch rates show great variability, abundance of some year classes was shown to increase over time based on year class catch curves, and CPUE is not well correlated with harvest, other weakfish indices, or the converged portion of the VPA. Unlike the NEFSC index, however, the New Jersey index is capable of capturing large weakfish when present. Efforts to develop a more suitable index found that the percent of tows during the August cruise that captured weakfish was coherent with other indicators of weakfish abundance. For the current assessment then, the New Jersey index is based on the proportion of positive tows (PPT) from the August cruise. The index was aged using a combination of survey specific proportion at age data for

1995 to 1997 and DNREC 30-foot trawl survey proportion at age data from August cruises (supplemented with July and September data as needed) in all other years.

PPT was greatest between 1994 and 1997, with two-thirds of all tows containing weakfish in three of those years (Figure C6.1-2). PPT dropped precipitously to 28% in 1998 and remained generally between 25 and 50% until 2002. In 2003, PPT again dropped precipitously to the time series low of just 10.3%. Since then, PPT has rebounded slightly, with 32 to 35% of tows containing weakfish in three of the last four years. Binomial confidence intervals (90%) indicate the upper and lower CI differ from the mean by approximately 7 to 16% in all years.

Indices at age document the occurrence of strong 1993 and 1996 year classes as they appear in 1994 and 1997 and move through the age structure (Figure C6.1-2). As these cohorts increase in age, the proportion of fish ages 4 to 6+ was increased and exceeded 40% in three years between 1997 and 2000. Since 2002, however, the survey has been dominated (>= 73%) by age 1 and 2, with 4+ fish making up less than 5% of the annual catch..

Detailed investigation into the New Jersey Ocean Trawl Survey provides evidence that the percent of positive tows from the August cruise provides a reliable index of weakfish abundance. The WTC has therefore determined that the New Jersey trawl survey index is acceptable for use in the stock assessment.

C6.1.3 Delaware DFW Delaware Bay Trawl Survey

The Delaware Division of Fish and Wildlife has conducted a trawl survey within the Delaware Bay intermittently since 1966 (1966-1971, 1979-1984, and 1990 – present). The survey collects monthly samples (March through December) at nine fixed stations throughout the Delaware portion of the Bay. The net used has a 30.5 foot headrope and 2" stretch mesh codend. For the current assessment, only the 1981-1984 and 1990-present time series are evaluated. Weakfish abundance is calculated as an average number of age 1+ fish per nautical mile for June to October cruises, and the index is treated as a mid-year abundance (*i.e.* not progressed forward to January 1). Since 1991, length frequencies have been aged using survey specific age-length keys.

The geographic range of this survey is limited to the Delaware Bay, a very small portion of the weakfish stock range; however, Delaware Bay is known to be a major spawning ground for weakfish on the Atlantic coast (Nye et al 2008). As the survey occurs monthly for a large portion of the year, fish from a wide size and age distribution are available to the survey, from young of year to large older spawners.

Weakfish abundance was moderate in the early 1980s and early 1990s, ranging between approximately 15-30 fish/nm (Figure C6.1-3). Beginning in 1992, abundance increased sharply to a time series high of over 230 fish in 1996. Abundance decreased by more than half in 1997, and has exhibited a generally declining trend since that time. CV of the composite index showed relatively high variability from 1991 to 1995, ranging from 20 to 60%. Interannual variability in CV stabilized in 1995 and generally ranged from 19 to 26% until 2001. Since 2001, CV has shown a slight increase, estimated at 33% in 2006.

Age structure advanced from primarily age 1 and 2 fish in the early 1990s to include ages 7 and 8 in 1998-2000 (Figure C6.1-3). Abundance of age 4+ fish accounted for 30 to 35% of the total index in 1997 and 1998 as the large 1993 year class moved through. Abundance of older ages has since declined to levels observed in the early 1990s, with 4+ fish accounting for less than 1% of the total.

The Delaware 30-foot trawl survey occurs in one of the major weakfish spawning areas and has been shown to capture a wide size and age range of weakfish throughout the year. Trends in

abundance correspond well with anecdotal and observed information from commercial and recreational fisheries and are coherent with other indicators of weakfish abundance. The WTC has determined that the Delaware 30-foot trawl survey provides a reliable index of weakfish abundance.

C6.1.4 SEAMAP Fall Survey

The Southeast Area Monitoring and Assessment Program (SEAMAP) has conducted three seasonal trawl surveys since 1989 between Cape Hatteras, NC and Cape Canaveral, FL. A stratified random design is employed to sample inner (4.6 to 9.1 m) and outer (9.1 to 18.2 m) depth strata using twin 75-foot (22.9 m) highrise mongoose trawls towed behind a double rigged St. Augustine shrimp trawler. The geographic range of the survey encompasses nearshore ocean waters south of Cape Hatteras, and SEAMAP is the only fishery independent survey conducted in the southern portion of the weakfish range. Unfortunately, catches of weakfish south of North Carolina are extremely small and of little value as an index of abundance. An index of abundance is therefore generated using only strata off North Carolina during fall cruises. Catch is aged using survey specific length and age data (where available) or southern region late season ALKs, which are primarily North Carolina data. Fall aged fish are progressed one age to estimate January 1 abundance in the following year.

From 1990 to 2001, the survey ranged from approximately 5 to 30 fish per tow, with the exceptions of 1992 with an index of less than 1 fish per tow, and 1993 and 1994 with indices of approximately 44 and 52 fish per tow (Figure C6.1-4). From 2002 to 2004, the index increased to between 35 and 60 fish per tow, before jumping drastically to nearly 500 fish per tow in 2005. In 2006, the index dropped back down to 45 fish per tow. Removing 2005 as an outlier, the index has increased gradually throughout the time series. CV for the survey is relatively high, ranging from 24% in 1996 to 76% in 1997, with an average of nearly 48% over the time series.

Age structure is truncated in the survey catch-at-age matrix, and the survey is driven primarily by age 1 and age 2 fish (Figure C6.1-4). Barring the 2006 (lagged year) index value, strongest recruitment (age 1) events occurred in 1995 and 2003. The 2006 index is anomalously high, with an age 3 index greater than the age 1 index in most years. Age 4+ fish generally constitute less than 1% of the total catch, with a maximum of 11.2% in 1998 and 7.7% in 1999 as the strong 1995 recruits moved through.

The SEAMAP survey index is highly variable, with CV's greater than most other available weakfish indices. Regardless, this is the only offshore survey in the southern region of the weakfish range, and the WTC determined that it should be considered for inclusion in stock assessment analyses.

C6.1.5 Pamlico Sound Independent Gillnet Study (PSIGNS)

In May 2001, the NCDMF began a gillnet survey in Pamlico Sound to provide fishery independent relative abundance indices for key estuarine species. This is the first weakfish stock assessment for which a sufficient time series has been available from this survey.

The survey is conducted throughout the year, providing a mean index of abundance in the survey year (*i.e.* indices at age are not progressed to January 1). Sampling uses a stratified random design based on area and water depth. Twice per month a deep-water and shallow-water sample are collected from each of 8 areas using a gillnet consisting of eight 27.4 m segments of 7.6, 8.9, 10.2, 11.4, 12.7, 14.0, 15.2, and 16.5 cm (3.0" to 6.5" by half inch) stretched mesh gill net. Nets are typically deployed within an hour of sunset and retrieved the next morning, for approximate soak

times of 12 h. This sampling design results in a total of approximately 32 gill net samples (16 deep and 16 shallow samples) being collected per month across both the Rivers and Sound. Catch rates of target species are calculated annually and expressed as an overall CPUE along with corresponding length class distributions. The overall CPUE provides a relative index of abundance showing availability of each species to the study, while the length distribution and aged CPUE estimates show the size structure of each species for a given year. The overall CPUE was defined as the mean number captured per sample and was further expressed as the number at length per sample, with a sample being one array of nets fished for 12 hours. Due to disproportionate sizes among stratum and region, the final CPUE estimate was weighted. For weakfish, the CPUE at age was calculated for 6-month periods (Jan-Jun and Jul-Dec) in the same manner as was done for the harvest catch at age (see section **C5.1**).

Total CPUE showed a decline for the first few years of the survey, stabilized for a few years, and declined again in the last years (Figure C6.1-5). CPUE in 2007 was approximately one-third of the value in the first year of the survey. No estimates of precision were provided for this survey.

Age 1 fish are not fully recruited to the gear, and in most years it appears age 2 are also under represented as the index for age 3 fish is generally equal to or greater than the index for age 2 in the previous year (Figure C6.1-5). One prominent feature of this survey is the tracking of an apparent strong 1999 year class as it progresses through the first years of the survey.

Although the time series is short and no estimates of survey variability are available, the survey appears to adequately track abundance of weakfish across years. The WTC accepted the survey for use in the stock assessment.

C6.2 Young of year fishery independent surveys

C6.2.1 Massachusetts DMF Trawl Survey

The Massachusetts Division of Marine Fisheries conducts a stratified random trawl survey in six depth zones (0-9.1, 9.1-18.3, 18.3-27.4, 27.4-36.6, 36.6-54.9, and >54.9 m) and five geographic regions within the state. Sampling has been conducted twice per year (May and September) since 1978. Survey gear consists of a two-seam whiting trawl with a 11.9 m headrope and a 12.7 mm stretch mesh codend liner. Weakfish, primarily young of year, are most commonly observed during the fall survey in the three regions south of Cape Cod. Arithmetic mean catch per tow is used as an index of young of year abundance in the survey year.

The MA DMF trawl survey area encompasses nearshore ocean and estuarine areas within Massachusetts state boundaries. Like the New Jersey trawl survey, the survey area overlaps a portion of the NEFSC trawl survey area, but a smaller vessel allows more comprehensive sampling of shallow waters. Although large numbers of weakfish have been observed in Cape Cod Bay and Massachusetts Bay (Collette and Klein-MacPhee 2002), these waters are generally considered the northern extent of the weakfish range.

Mean annual catch per tow is consistently under 2 fish, with only three exceptions since 1981 (Table C6.2-1, Figure C6.2-1). Abundance generally declined from 1981 to 1984. In 1985, abundance increased more than 100-fold to the time series high of more than 15 fish per tow. Recruitment was again relatively high in 1986 (2.7 fish per tow), before dropping back to near zero levels for 1987 to 1994. Since 1994, abundance has shown a general upward trend, while at the same time exhibiting greater interannual variability. The second highest index value of 2.9 fish per tow occurred in 2006, before dropping back to just 0.2 fish per tow in 2007.

Standard errors were high and exhibited a similar trend as mean abundance. The CV was

generally greater than 60%, and exceeded 90% in eight years. Because of the low catch rates and high variability, the WTC has determined that this index provides little information on the abundance of weakfish. The survey was not used in the assessment.

C6.2.2 Rhode Island Trawl Survey

The Rhode Island Division of Fish and Wildlife seasonal trawl survey was initiated in 1979 to monitor recreationally important finfish stocks in Narragansett Bay, Rhode Island Sound, and Block Island Sound. The survey employs a stratified random design and records aggregate weight by species, frequency, individual length measurements, and various physical data. In 1990, a monthly component was added to the survey, which includes 13 fixed stations in Narragansett Bay. Sampling is conducted using a two seam trawl net with a 12.2 m headrope and 6.4 mm codend liner. Calibration studies have been conducted to ensure continuity of the catch series when changes in vessel and gear were required. An index of weakfish abundance is calculated as the geometric mean number per tow.

CPUE was relatively high in 1981 and 1982, but dropped off in 1983 and remained below 2.5 fish per tow until 1995 (Table C6.2-1, Figure C6.2-1). In 1996 and 1997, YOY abundance increased to more than 6 fish per tow, before dropping off to previous levels for 1998 to 2002. Since 2003, recruitment has shown great interannual variability, ranging from the time series high of 16.5 in 2003 to the second lowest value of 0.17 in 2006. CV of the log mean values are generally between 10 and 30%, with only two values (1995, 2006) exceeding 40%.

The RI survey occurs in a very small portion of the weakfish range and is outside the core area described by NEFSC (2000). However, several episodes of strong recruitment have been observed, and CV is moderate. For these reasons, the WTC has determined that the Rhode Island YOY survey is suitable for use in the assessment.

C6.2.3 Connecticut DEP Long Island Sound Trawl Survey

Since 1984, the Connecticut DEP has conducted spring and fall trawl surveys in the Connecticut portion of Long Island Sound between the New York/Connecticut border in the west and New London, CT in the east. Survey effort consists of three spring cruises conducted during April, May and June, and three fall cruises during September/October. Stratified random sampling is employed based on four depth zones and three bottom types. Survey gear consists of a 14 x 9.1 m high-rise otter trawl with 5 mm codend mesh. The survey catches mostly YOY and age 1 weakfish as defined by examination of length frequencies. Indices of abundance for age 0 and age 1+ are developed as geometric mean catch per tow, but only the YOY index was considered for this assessment.

Sampling is limited to Long Island Sound. The Sound encompasses a very small portion of the weakfish range, but may serve as a primary nursery habitat in this region.

From 1984 to 1998, the YOY index varied without trend, and generally ranged from approximately 3 to 10 fish per tow, with relatively strong year classes (10-15 fish per tow) occurring in five years (Table C6.2-1; Figure C6.2-2). In 1999, recruitment increased sharply and has remained above 30 fish per tow in all years except 2005 and 2006. Time series highs of more than 63 fish per tow occurred in 2000 and 2007, while minimum catches of approximately 1 fish or less occurred in 1984, 1986, and 2006. CV of the YOY index has exhibited a generally negative trend over the time series.

NEFSC (2000) recommended that this survey not be used as an index of abundance because it occurs outside the core area of weakfish abundance. However, large recruitment events have been

observed in this area over the last ten years, suggesting it may provide prime nursery habitat. In addition, precision of the YOY catches is strong. For these reasons, the WTC concluded that the Long Island Sound YOY index was suitable for use in the assessment.

C6.2.4 NYDEC Peconic Bay Juvenile Trawl Survey

The New York Division of Fish, Wildlife and Marine Resources has conducted a juvenile trawl survey in the Peconic Bay estuary of Long Island since 1985. Weakfish was the primary target species when the survey was initiated, and Peconic Bay was selected for the survey area because of its importance as a weakfish spawning ground. Random sampling occurs weekly between May and October using a semi-balloon shrimp trawl with a 4.9 m headrope and 12.7 mm stretch mesh codend liner. The survey samples mainly young of year weakfish, and a YOY index has historically been calculated as a geometric mean catch per tow over all sampling months. In 2005 and 2006, technical difficulties constrained sampling to May – July (2005) and July – October (2006), so a revised index using only July and August has been calculated. The two indices (all months and July-August) show a similar increasing trend and are well correlated ($r = 0.96$). The July-August index provides higher estimates of abundance and appears to be more variable between years, although CV is lower for the July-August index than for all months combined.

The July-August index ranges from less than one to more than 30 fish per tow (Table C6.2-1; Figure C6.2-3). Despite large interannual variations, there appears to be a gradual increase in recruitment over the time series. Strong year classes occurred in 1991, 1996, and 2005 (time series high). Standard error of the catch has increased over the time series as well; however, 95% confidence limits around the mean are moderate. The lower bound averages approximately 35% less than the mean value, while the average upper bound is approximately 50% larger than the mean.

Because this survey is conducted outside the apparent core area, NEFSC (2000) recommended that this survey not be used as an index of abundance. However, the survey was developed specifically to monitor trends in weakfish populations on an important spawning ground, and some strong year classes have been observed. Precision of the survey is acceptable. For these reasons, the WTC has used the Peconic Bay YOY survey in the assessment.

C6.2.5 Delaware DFW Delaware Bay Juvenile Trawl Survey

In addition to their 30-foot trawl survey, the Delaware DFW conducts a fixed station survey in Delaware Bay targeting juvenile finfish. Sampling is conducted monthly from April through October using a semi-balloon otter trawl. The net has a 5.2 m headrope and a 12.7 mm stretch mesh codend liner. Weakfish are a significant component of the catch, with the greatest majority of these weakfish (more than 99% in some years) being young of the year. A YOY index is calculated as the geometric mean number per tow during the June to October cruises.

As with the Delaware 30-foot trawl index, the survey is restricted to Delaware Bay. Although this encompasses only a small portion of the geographic range of weakfish, Delaware Bay is known to provide significant spawning and nursery habitat for the species.

Throughout this time series, recruitment indices have generally fallen between 5 and 15 fish per tow, with only 2 values below and three values above this range (Table C6.2-1, Figure C6.2-3). Weak recruitment occurred in 1983 and 1988, with less than 5 fish per tow, while the two strongest recruitment events of 20.1 and 16.8 fish per tow occurred in 1991 and 2005, respectively. Average recruitment over the time series has been approximately 10.8 fish per tow. The index indicates three general stanzas in recruitment since 1981. From 1981 to 1990, recruitment was generally below the long term average. In 1991, recruitment increased to the time series high, beginning a decade of

above-average recruitment. In 2001, recruitment dropped below average and has remained there for five of the last seven years. Estimated means appear precise, with 95% confidence limits typically only 25% to 35% larger or smaller than the mean value.

The Delaware young of year survey occurs within the core area of weakfish abundance and encompasses a major spawning/nursery area for the species during months when weakfish are present. The survey has captured the occurrence of several strong year classes with good precision. The WTC has used this survey in the stock assessment.

C6.2.6 *Maryland DNR Chesapeake Bay and Coastal Bays Juvenile Trawl Surveys*

The Maryland Department of Natural Resources conducts two juvenile trawl surveys: one in the lower eastern region of Maryland's portion of Chesapeake Bay from 1980 to the present, and one in the Atlantic coastal bays from 1972 to the present. Both surveys sample fixed stations using a 4.9 m semi-balloon otter trawl with a 12.7 mm stretch mesh codend liner. The coastal bays project samples monthly from April through October, while the Chesapeake survey runs monthly from May through October. Due to non-standardized survey methods during the early portions of both surveys, only data from 1989 onward are used to calculate YOY abundance indices. Indices are calculated as geometric mean catch per tow.

Both surveys are confined to Maryland state waters which constitute only a small portion of the weakfish range. Regardless, both survey areas are sheltered estuarine environments and may provide suitable spawning and nursery habitat for the species.

The Chesapeake index steadily increased from 0.4 fish per tow in 1989 to the time series high of 8.1 fish per tow in 2001 (Table C6.2-1, Figure C6.2-4). The index has steadily declined to less than 2 fish per tow in 2007. The coastal bays index appears stable between 0.9 and 1.9 fish per tow during1989-1994. In 1995, recruitment increased to 4.4 fish per tow and then decreased gradually to 2.6 in 2001. During the period 1989 to 2001, interannual variability was minor with few exceptions. Beginning in 2001, interannual variability increased dramatically. The weakest recruitment of the time series occurred in 2002, followed in 2003 by the time series high of 5.6 fish per tow. Coefficients of variation have ranged from 3-22% for the Chesapeake Bay survey and 4-15% for the coastal bays survey (except 30% in 2008).

Both surveys occur within the core region of weakfish abundance during months when weakfish would be present. Precision is uncertain, but the WTC has determined both are suitable for use in the assessment.

C6.2.7 *Virginia Institute of Marine Science Chesapeake Bay Trawl Survey*

The Virginia Institute of Marine Science (VIMS) has conducted a trawl survey in lower Chesapeake Bay since 1955. Over time there have been several changes to sampling strategy and survey area. Currently, sampling is conducted using a 9.1 m semi-balloon otter trawl with a 6.4 mm codend liner. Sampling occurs monthly throughout the year using stratified random sampling in the mainstem bay and fixed stations in tributaries. Young of year are identified through examination of length frequencies (monthly ranges), and an index of recruitment is computed as the geometric mean catch per tow during August to October from the three major tributaries.

The geographic region covered by the survey includes the Virginia portion of the Chesapeake Bay and lower portions of its three main tributaries (James, York, and Rappahannock Rivers). Although sampling does occur in the main stem, catches of weakfish are generally minimal in the Bay, so the index is limited to the three tributaries. Few large weakfish are present year round, but the estuaries provide suitable nursery grounds for juveniles.

Recruitment varies widely over the time series, ranging from less than 5 fish per tow to more than 35 fish per tow (Table C6.2-1, Figure C6.2-4). Interannual variability is often large, particularly in the early portion of the time series, with the maximum and minimum indices occurring in consecutive years (1985, 1986). From 1986 to 1990, the survey shows a rapid increase from 4.7 to 30.0 fish per tow, followed by a sharp drop back to 7.0 fish per tow by 1994. Recruitment rebounded slightly through 1999, but generally has been declining since.

No estimates of survey variability are available for the current index; however, 95% CIs for an index that includes Bay and River stations (data not shown) indicate good precision which has improved as the survey progressed. Sine 1989, CIs have generally been within 25 to 40% of the observed mean value. It could be expected that precision of the "river only" index would be greater, as catches of weakfish are less variable in the rivers than the Chesapeake Bay.

The VIMS trawl survey occurs within the core region of weakfish abundance during months when weakfish would be present. Precision is uncertain, but proxy data indicate low to moderate variability. The WTC has determined that this survey is suitable for use in the assessment.

C6.2.8 North Carolina DMF Pamlico Sound Juvenile Trawl Survey

The North Carolina Division of Marine Fisheries conducts a juvenile trawl survey in Pamlico Sound. Sampling is conducted in June and September using a stratified random design. Survey gear consists of twin 9.1 m mongoose trawl nets with 19.1 mm codend mesh. Data from these surveys are used to develop an ages 1+ index (June) and a YOY index (September), both based on length frequency analysis. Indices are calculated as geometric mean catch per tow.

Between 1987 and 1999, the YOY index ranged from approximately 1 to 60 fish per tow. Catch was characterized by large interannual fluctuations, but shows a consistent increase over that period from the time series low in 1987 to the time series high in 1999 (Table C6.2-1, Figure C6.2-5). From 1999 to 2002, recruitment dropped rapidly from 60 to 4 fish per tow. Since 2002, the index indicates a modest rebound to approximately 13 fish per tow in 2007. Since 2000, interannual variability is much less pronounce than in the early portion of the time series.

From 1987 to 1999, the ages 1+ index varies without trend between approximately 5 and 20 fish per tow. The index does capture some of the larger year classes observed in the YOY index, and peaks in 2000 (one year after the YOY peak) at 51.5 fish per tow. The index drops rapidly back to less than 5 fish per tow and remains stable for the remainder of the time series except for the 2006 index of nearly 30 fish per tow.

Variability for both indices was moderately large, with 95% CIs in many years deviating by more than 40% from the mean value. For the YOY index, variability decreased as abundance increased. For both surveys, variability has increased in recent years.

The survey area encompasses only a small portion of the weakfish range and survey variability for both indices is moderate. However, the survey occurs within a prime weakfish spawning/nursery ground and provides the only recruitment index in the southern portion of the range. The WTC has included both the YOY and 1+ indices in the assessment.

C6.3 Other fishery independent surveys

In addition to the fisheries independent surveys listed above for consideration as tuning indices, weakfish biological data are also obtained from two other trawl surveys in the Mid-Atlantic region. The Chesapeake Bay Multispecies Monitoring and Assessment Program (ChesMMAP) is a large mesh trawl survey that began in 2002 (Bonzek et al 2007). Sampling is conducted bi-monthly from March through November, targeting juvenile to adult finfish throughout the main stem of the

Chesapeake Bay. The Northeast Area Monitoring and Assessment Program (NEAMAP) Trawl Survey is a nearshore ocean trawl survey operating between Cape Hatteras, NC and Montauk, NY (Bonzek et al 2008). The intent of this survey is to sample nearshore ocean waters which have typically been under represented in NEFSC trawl surveys. A pilot survey was conducted in 2006, and bi-annual (spring and fall) cruises have occurred since.

Although the time series from these surveys are not yet sufficient to provide trends, weakfish biological data collected during survey cruises have been made available for use in the stock assessment. When a sufficient time series has been attained, the WTC will evaluate these surveys as candidates for tuning indices.

Recently, information was obtained on weakfish juvenile abundance collected through the Georgia Coastal Resources Division Ecological Monitoring Trawl Survey (EMS). Sampling occurs at 43 fixed stations throughout six coastal sounds using a 12.2 m flat otter trawl equipped with 4.8 cm stretch mesh in the codend. The weakfish index is calculated from sound and creek strata during monthly cruises between July and October. Juveniles are identified by monthly length cutoffs. The survey began in 2003, so the time series is currently too short. When additional years of data are available, the WTC will re-evaluate this survey.

C6.4 Fishery dependent surveys

Historically, a fishery dependent index of weakfish abundance was developed using recreational catch per "directed trip" (trips where weakfish was identified as a target species; *cf.* NEFSC 1998b, 2000). During the review of the 2000 assessment, the SARC expressed concerns about fishery dependent indices, and they were removed from the assessment. Potential sources of bias in fishery dependent indices include non-random distribution of effort, and hyperstability of the index (Hilborn and Walters 1992).

In 2006, a revised recreational index was developed that the WTC feels largely addresses the concerns expressed by the 30th SARC. Estimates of catch include all weakfish (A+B1+B2) captured by the recreational private/rental boat mode in state waters of the mid-Atlantic region (New York to Virginia). As described in Crecco (2005a), the private/rental mode is highly mobile and capable of catching weakfish over a large range of sizes. Catches were constrained to the mid-Atlantic region because private/rental boat catches from this region have accounted for greater than 60% of annual recreational catch. Two estimates of effort were used to convert catch to CPUE. The first uses all private/rental boat trips in state waters of the mid-Atlantic region (Crecco 2005a). The second, detailed by Brust (2004), estimates effort as the number of private/rental boat trips in mid-Atlantic state waters that captured any of a suite of species typically associated with catches of weakfish. The two indices were highly correlated (r > 0.98). Although the 2006 assessment used the index based on the suite of associated species, the current assessment uses the index based on all mid-Atlantic private/rental boat trips because of its ease of calculation without loss of information, and because it provides consistency between assessment methodologies.

The methods described above provide an index of total catch per trip, including both harvested and discarded fish. Based on assumptions regarding availability to the fishery (gear and area) at size, this index is considered to represent ages 1+. Age specific indices for harvested fish were developed by dividing total harvest by the effort index (Mid-Atlantic private/rental boat trips) and applying the age structure of the recreational harvest of 2+ fish. When the index is lagged forward, this method provides indices at age for 3 through 6+.

Both indices (age aggregated 1+ and age specific 3 to 6+) are developed using data from mid-Atlantic state waters. This region encompasses the primary distribution of weakfish within its

range. Although all sizes and ages of weakfish are present in this region, younger fish are not considered to be captured or harvested by the fishery, so the indices are only representative of mature fish.

CPUE of all ages varied greatly from 0.2 to 0.8 during 1981 to 1988 (Figure C6.3-1). From 1989 to 1993, CPUE was stable between 0.15 and 0.2, following which it increased rapidly to a peak of 0.61 in 1996. Since that time, CPUE has exhibited a consistent decline to the time series minimum of 0.09 in 2007.

During the early portion of the time series, age structure of harvested fish was dominated by fish aged 2 and 3 (Figure C6.4-1). By the mid 1990s, minimum size limits and expanding age structure increased the influence of older fish, and 4 year old fish were the most prevalent age group between 1996 and 1999, with 5 year olds dominant in 2000. Since 2001, older ages have fallen off to near zero levels, and harvest consists mostly of ages 2 and 3.

Estimates of precision are available for the different components of the indices (catch, harvest, and effort); however, there are no direct estimates of survey precision. Discussions on uncertainty in catch and effort estimates are presented in section **C5.2**.

Fishery dependent indices were employed in some of the early stock assessments, but were dropped in subsequent years due to concerns over their reliability. During the 2006 assessment, the WTC developed a new recreational index that covers the entire core area of the population range based on a highly mobile fleet whose effort is not specific to weakfish. The index covers the entire core area of the population range, is highly mobile, but effort is not specific to weakfish. Results are coherent with other indicators of weakfish population size, as well as harvest and the converged portion of the VPA (see Section **C10.0** of this report). The WTC has therefore determined that the fishery dependent indices are suitable for use in the stock assessment.

C6.5 Discussion

Of all surveys reviewed, only NEFSC and MA YOY were excluded from further analysis due to concerns over information content. The remaining indices include age specific indices from four fishery independent surveys (NJ, DE, NC gillnet, SEAMAP), nine fishery independent young of year surveys, and age specific indices from the recreational fishery. Although previous peer reviews recommend limiting indices to the core population area, the WTC has determined that three YOY surveys from outside the core area provide reliable information on weakfish recruitment, and these surveys have been included in modeling investigations.

The informative value of different tuning indices have received a lot of consideration during recent reviews of the weakfish assessment, either as a work in progress (SAW 40, 2008 DPSW) or the completed work (ASMFC 2006a). Each review panel had varying opinions of the indices, but a common recommendation from all three is that the indices need further evaluation. The WTC has attempted to address these concerns as well as possible (*e.g.* ASMFC 2006a, Parts A and C; Technical Committee response to 2008 DPSW panel report (Appendix C-1)), and a summary of recent work is presented in Section **C10.0** of this report. Most recently, the DPSW panel recognized and supported much of the work done by the WTC to address previous concerns, but still identified several issues with the indices (Miller et al 2009, bullets d through f). These concerns are summarized below, as are the WTC's responses to the panel report.

In bullet d) of the panel report, the panel recognizes that decisions to exclude indices were based on "several valid reasons." They continue, however, by stating that the remaining indices may not be indicative of true population dynamics because they "were restricted to limited spatial areas within the overall weakfish stock area." Further, they were concerned that similar trends in

remaining age-aggregate indices could be due to these indices containing little information. The WTC recognizes that the remaining fishery independent surveys are localized; however, the review panel supported the exclusion of the only coastwide index (NEFSC) for "valid reasons." In addition, the three remaining "aged" fishery independent surveys occur in the state with the most commercial landings (NC gillnet), the largest recreational harvest (NJ), and a principal spawning area (DE 30-foot). When considered on an age-aggregated basis, all three surveys show the same general trend, as does the recreational index which encompasses the entire core region. These findings suggest that the core area is adequately covered. In response to the concern over limited data content, the WTC points out that the trends in indices reflect changes in commercial CPUE, population trends from previous assessments, and observed age structure from fishery dependent and independent sampling.

Bullet e) of the panel report identifies concerns regarding the recreational index. In particular, the Panel expressed concerns that the MRFSS index was one of few that exhibited any clear pattern, which could be due to anglers switching methods as abundance of a more favorable species (striped bass) increased. The WTC responds that the pattern observed in the MRFSS index is coherent with three other aged surveys. Only two aged indices – NEFSC and SEAMAP - showed trends different from MRFSS and the others. Further, NEFSC was found to be uninformative, and the SEAMAP index exhibits moderate to large CVs. Although no alternative estimators were provided in the panel report, one suggestion made during the face to face meeting was found to produce the same general trend as the MRFSS index presented.

In bullet f), the review panel recognizes that although four surveys (NJ, DE, NC gillnet, MRFSS) show coherence when age aggregated, VPA results vary greatly when using only fishery dependent or fishery independent tuning indices. Preliminary evaluation of this issue was presented in the WTC response; however, further work identified incorrect age data in the MRFSS index used for preliminary runs. Additional information on VPA runs is presented in section **C7.0**.

In summary, the WTC recognizes that many of the concerns expressed over the last few years regarding the tuning indices are valid concerns. Significant time has been spent investigating and evaluating each of these concerns. The result is a set of tuning indices that the WTC considers is representative of weakfish population dynamics throughout the core region.

Another topic that has been gaining popularity in fisheries stock assessment is the process of standardizing abundance indices relative to temporal, spatial, and environmental factors. The three most recent weakfish peer reviews have all recommended that available weakfish indices of abundance be scrutinized to better understand their information content. During sampling, factors external to population dynamics may contribute "noise" that, if not accounted for, is generally attributed to fluctuations in population size. Standardizing indices using methods such as generalized linear models (GLM) and generalized additive models (GAM) seeks to identify factors that contribute to survey variability and minimize their influence, in order to better observe the true population signal.

Recently, a team of researchers from Virginia Polytechnic Institute and State University (Virginia Tech) has begun investigating the use of CPUE standardization for weakfish (Winter et al 2009). Raw data from nearly all abundance indices (fishery independent and fishery dependent) considered by the WTC were analyzed using GLM and (where appropriate) GAM. Preliminary results were presented to the WTC in April 2009. The WTC was encouraged by the results, but many questions remained. For example, indices investigated by Winter et al (2009) were not all directly comparable to indices considered by the WTC, in particular the New Jersey (CPUE all cruises vs percent positive tows during August cruise) and fishery dependent indices (Mid-Atlantic private boat CPUE vs CPUE for all sub-regions and modes). In the case of the fishery dependent

index, factors were included to account for sub-region and mode, but the discrepancies made comparisons difficult. Also, preliminary results did not include summary statistics of the various factors investigated for each index. It was not possible, therefore, for the WTC to evaluate which factors were significant or the proportion of total variability explained by each factor.

The researchers from Virginia Tech were receptive to comments and suggestions from the WTC, and the full report, updated with responses to some of the WTC's concerns (*e.g.* p-values for different factors), is included with the materials for SAW 48 as a supplementary report for weakfish (entitled "Winter et al 2009 – Weakfish CPUE standardization.doc"). However, given the WTC's concerns regarding the preliminary results, and the short timeline available to complete this assessment for peer review in June 2009, the WTC has determined that there was insufficient time to incorporate standardized indices into the current assessment. The WTC supports continuing this work to address the research recommendation and for consideration in future stock assessments.

C7.0 Evaluate the ADAPT VPA catch at age modeling methods and the estimates of F, Z, spawning stock biomass, and total abundance of weakfish produced, along with the uncertainty and potential bias of those estimates. Review the severity of retrospective pattern. (TOR #3)

C7.1 Introduction

Age structured modeling was conducted using ADAPT VPA (version 2.8, available for download at *http://nft.nefsc.noaa.gov*) as recommended by the 30th SARC (NEFSC 2000). Development of catch at age inputs from four fishery sectors (commercial harvest, commercial discards, recreational harvest, and recreational discards) is discussed in detail in section **C5.0** (see Table C5.3-1). A total of 38 abundance indices were considered suitable for use in the assessment (section **C6.0** and Figures C6.1-1 through C6.4-1), including 24 age specific fishery independent indices (four surveys age 1 to 6+), nine fishery independent young of year surveys, four age specific fishery dependent indices (MRFSS 3 to 6+), and one fishery dependent age aggregated index (MRFSS 2+). Specific configuration of the model, such as biological specifications and options selected, is presented in the supplementary report entitled "Weakfish ADAPT final run output.txt."

Multiple runs were conducted using various combinations of tuning indices and/or modified input values (sensitivity runs). All model runs were reviewed and the preferred model run was selected based on evaluation of model fit, residuals, retrospective patterns, and other considerations.

C7.2 General findings

Regardless of tuning indices selected, all runs using baseline data (calculated CAA, constant $M = 0.25$) provided consistent, virtually identical results through 2002. Trends in estimated parameters for the years 2003 to 2007 varied widely, as did the magnitude and duration of retrospective patterns. Model runs that included fishery dependent indices produced more conservative results (lower abundance, higher F), lower mean squared residual (MSR), and less severe retrospective patterns than those tuned solely with fishery independent indices (Table C7.2-1; Figure 7.2-1 through 7.2-3). When YOY indices were used to tune the model, the magnitude and duration of retrospective patterns were greater than when YOY indices were excluded, but exclusion of YOY indices did not necessarily result in lower MSR.

Significant consideration was given to selection of tuning indices. Section **C10.0** provides a detailed summary of work conducted to evaluate indices based on criteria such as size structure of the catch, and coherence with other indices, landings and the converged portion of the VPA. This

method resulted in the NEFSC fall trawl survey being dropped from consideration, and the New Jersey ocean trawl index being changed from average CPUE to proportion positive tows (PPT). Further work was conducted to investigate combinations of indices that produced consistent results while minimizing the retrospective pattern. Some committee members cautioned that this method of index selection might appear subjective, and recommended an alternate method of index selection. The proposed method developed a list of criteria against which each index could be scored, including survey design, spatial and temporal coverage, and catch characteristics. Indices receiving the highest total scores would be selected as tuning indices. An exploratory implementation of this method selected NEFSC ages 1-2, DE 30 foot trawl age 1, NC gillnet age 1, SEAMAP ages 1-6+, and the RI, CT, NY, DE, MD, and NC YOY indices for use in tuning the model. During the converged portion of the time series (1982 to 2002), ADAPT results using these tuning indices were consistent with results from other runs with other tuning indices (Figure C7.2-4); however, the WTC was concerned that the magnitude and duration of the retrospective pattern in recent years were greater using this method than observed in other runs (Figure C7.2-5), and MSR for this run was among the highest observed (Table C7.2-1). In addition, the WTC was concerned that this method selected two ages from the NEFSC survey, which the WTC had excluded based on poor performance relative to established criteria, and all 6 ages of the SEAMAP survey, which is confounded by large interannual variability and an extreme outlier in 2005 that likely influences estimates in recent years. Based on these concerns, the WTC did not pursue this method any further for this assessment. It is recognized that alternative criteria and/or a different cut-off score for "acceptable" indices might influence the results, but additional work was postponed until completion of the ongoing assessment.

The WTC also investigated the use of tuning indices that were standardized relative to spatial, temporal, and environmental factors using methods such as GLM and GAM (Winter et al 2009; see section **C6.5**). Exploratory model runs using standardized indices were conducted in ADAPT VPA and compared to results using non-standardized indices. Winter et al (2009) indicate that GAM methods were superior to GLM for modeling weakfish catch rates. Model runs were therefore conducted using GAM standardized indices where possible, and GLM only for those indices that could not be standardized using GAM. Because the MRFSS harvest index was not standardized, comparisons were restricted to runs using only fishery independent indices. Also, indices developed by Winter et al (2009) were not all directly comparable to indices considered by the WTC, in particular the New Jersey (CPUE all cruises vs. proportion positive tows during August cruise). Regardless, ADAPT results using standardized indices were very similar to comparable runs using non-standardized indices and produced somewhat smaller retrospective patterns (Figures C7.2-6 and C7.2-7). As noted in section **C6.5**, although the WTC is encouraged by these findings, initial results of the index standardization analysis were presented too late in the development of this assessment report to be fully evaluated. The above discussion is based on examples of preliminary work that have not been fully endorsed by the WTC. The WTC will address their concerns regarding the index standardization analyses and continue to investigate the use of standardized tuning indices for future assessments.

The 2008 DPSW panel expressed concern over the inconsistency of model results between runs using only fishery dependent tuning indices and runs using only fishery independent indices. In the preliminary results presented during the 2008 DPSW, the New Jersey, Delaware, North Carolina gillnet, and MRFSS CPUE indices all showed a similar pattern in abundance when aggregated across ages; however, when disaggregated and used to tune the VPA, the fishery dependent results were inconsistent with the results using fishery independent indices. These results suggested that

one or more of the indices performed poorly at tracking weakfish age structure. The DPSW reviewers suggested that the WTC investigate this discrepancy. Preliminary results of these investigations were presented in the WTC response to the review panel's comments (see Appendix C-1). Additional evaluation shows, however, that the results presented at the DPSW were inaccurate due to incorrect age allocation of the fishery dependent indices. Updated results show that runs using only NJ, DE, or MRFSS indices produce consistent results (Figure C7.2-8).

C7.3 Preferred model run

C7.3.1 Model output

The previous (2006) assessment selected a model run tuned with only the fishery dependent indices, mainly because it produced a greatly reduced retrospective pattern relative to other runs. For this current assessment, the fishery dependent only run still produced the smallest retrospective (Figures C7.2-2 and C7.2-3) and best model fit (Table C7.2-1); however, including indices from the New Jersey ocean trawl, Delaware 30-foot trawl, and North Carolina gillnet surveys produces nearly identical trends in recent years with only marginal increases in the retrospective pattern. MSR nearly doubled relative to the fishery dependent only run, but is still lower than all other runs investigated. Minor improvements were made to the output by expressing index values to four decimal places and including preliminary estimates of abundance in 2008 for all selected tuning indices where available (only NC gillnet data were not). This run is hereafter referred to as the "final" or "preferred" run. Index values are shown in Table C7.3-1. Selected parameter estimates are presented in Figures C7.3-1 and C7.3-2, and the full model output is provided in the supplementary report entitled "Weakfish ADAPT final run output.txt."

Following the 2006 assessment, the WTC received criticism for not including any fishery independent tuning indices. The preferred run for the current (2009) assessment addresses this concern by including fishery independent indices from New Jersey, Delaware, and North Carolina along with the fishery dependent tuning indices. However, another criticism received by the WTC following the 2006 assessment was the selection of indices that produced desired results. As all the tuning indices used in the preferred run of the current assessment exhibit the same trend, this criticism may reappear. The 2008 DPSW suggested that these indices were coherent because they contained little information about weakfish abundance (Miller et al 2009). The WTC argues that these surveys were selected, not because they produce the desired result, but because they most accurately reflect other indicators of stock dynamics. These indices correspond well with changes in harvest, abundance, CPUE, and population age structure. It is doubtful that indices that lack useful information would track these parameters so closely. In addition, parameter estimates derived using tuning indices selected based on scored criteria were nearly identical to results of the preferred run for the years 1982 to 2002. The retrospective pattern based on scored criteria was longer and of much greater magnitude than the preferred run.

Estimates of fishing mortality (unweighted average, ages 4-5) from the preferred run approached or exceeded F = 1.5 in most years between 1982 and 1988 (Table C7.3-2, Figure C7.3-1). Between 1989 and 1994, F values generally varied around F = 1.0 before dropping rapidly to the time series low of F = 0.32 in 1995. A nearly exponential increase ensued during most of the following decade, peaking in 2003 at the time series high of F = 2.39. Fishing mortality rates decline after 2003, and the terminal year estimate of F_{2007} = 0.51 (80% CI range 0.38 to 0.82) is the second lowest value in the time series. A prominent retrospective pattern (Figure C7.3-2) indicates estimates since 2003 may be inaccurate (see section **C7.3.2**).

Between 1982 and 1985, spawning stock biomass varied generally between 8,000 and 13,000 MT, increased rapidly to approximately 20,000 MT in 1986-1987, then returned to its previous range during 1988 to 1993 (Figure C7.3-1). A gradual increase followed until SSB reached a relative peak around 19,000 MT in 1997. Since that time, SSB has undergone a gradual decline to the time series low of 1,987 MT in 2005 before rebounding slightly in the last two years. SSB in the terminal year is estimated as $SSB_{2007} = 7,236$ MT. Bootstrapping indicates an 80% CI range for SSB in 2007 from 5,300 to 10,733 MT. A prominent retrospective pattern (Figure C7.3-2) makes these estimates unreliable.

From 1982 to 1984, total abundance was estimated to range between 80 and 100 million individuals per year (Figure C7.3-1). Abundance increased rapidly to the time series high of 120 million in 1986, then declined by more than 70% in just 4 years to less than 50 million individuals in 1989. A steady increase was observed between 1989 and 1994, peaking at around 90 million. A nearly continuous decline has been observed since that time, with the time series low of just 10.6 million individuals estimated in 2005. Recent years' estimates are slightly higher, and the terminal year is estimated as $N_{2007} = 32.8$ million with an 80% CI range of 11.95 to 90.05 million. A prominent retrospective pattern (Figure C7.3-2) makes these estimates unreliable.

Recruitment to age 1 followed a similar trend as total abundance (Figure C7.3-1). The time series high of 70.9 million in 1986 was followed by a sharp decrease to just 20.8 million four years later. Recruitment increased steadily to around 49.3 million in 1994 and has since declined, with the time series low of just 4.5 million recruits estimated in 2005. Recent years' estimates are slightly higher, and the terminal year is estimated as $R_{2007} = 16.4$ million with an 80% CI range of 3.8 to 59.1 million. A prominent retrospective pattern (Figure C7.3-2) makes these estimates unreliable.

Survey residuals tend to show a strong serial pattern, particularly for the younger ages (Figure C7.3-3). Abundance at age of the New Jersey and Delaware trawl surveys is generally overestimated by the model during the early years of the surveys, shifting to an underestimation during the mid to late 1990s. The same pattern is observed for the recreational harvest index of abundance (aggregated ages 2-6+). In contrast, the recreational harvest indices are underestimated by the model during the first decade of the time series, but the early 1990s, residuals have shifted to negative values before increasing back to positive values since 2002.

C7.3.2 Retrospective pattern

As in previous stock assessments (NEFSC 2000, Kahn 2002, ASMFC 2006a, Part A) reliability of estimates in recent years is diminished by the presence of a prominent retrospective pattern. For the current assessment, fishing mortality is underestimated for the most recent five to seven years, while SSB, total abundance, and recruitment are all overestimated (Figure C7.3-2). The degree of error is substantial, with fishing mortality in 2003 increasing 130% from $F_{2003} = 1.04$ when 2003 is the terminal year to $F_{2003} = 2.39$ when data through 2007 are included. Similarly, SSB_{2003} decreases by nearly 75% from 8,282 MT to 2,190 MT between terminal years 2003 and 2007. Improvements to the ADAPT VPA model implemented in 2007 (version 2.6) allow users to combine retrospective analysis with bootstrapping (ADAPT version history, available at *http://nft.nefsc.noaa.gov*). Results of such an analysis further elaborate the degree of uncertainty in recent year estimates. Frequency distributions of bootstrapped estimates of F_{2005} from terminal year 2005 and terminal year 2007 do not overlap, and those of SSB overlap only slightly (Figure C7.3-4). Estimated 80% confidence intervals do not overlap for either parameter (Table C7.3-3).

Despite a significant retrospective pattern, the 30th SARC determined that results from the ADAPT model could be used to establish reference points and provide guidance to management

(NEFSC 2000). Quantitative analysis of the retrospective pattern observed in this assessment shows that estimates differ by more than 100% between the time they are made in the terminal year and five years later when estimates converge (Figure C7.3-2). In addition, bootstrap confidence intervals of terminal year estimates are not sufficient to capture the uncertainty (Figure C7.3-4, Table C7.3-3). The WTC has concluded that ADAPT parameter estimates in recent years are unreliable and should not be used for management at face value. An attempt to correct parameter estimates for the retrospective pattern is presented below. The WTC does not necessarily endorse the corrected estimates, but is presenting them here for review and discussion by the 48th SAW/SARC.

An attempt was made to correct parameter estimates in recent years using the observed retrospective pattern. For this analysis, it is assumed that parameter estimates in 2003 using data through terminal year 2007 have "converged" and are robust to additional years of data being added in the future, and that the retrospective pattern observed since 2003 continues into the future. Violation of these assumptions will invalidate the following retrospective correction analysis.

The estimate of F_{2003} calculated for terminal year 2007 was divided by estimates of F_{2003} from each of the other terminal years 2003 to 2006 to calculate a ratio of change for each terminal year relative to the year it would converge. Assuming four years of additional data are required for a terminal year estimate to converge, estimates of F_{2004} to F_{2007} from terminal year 2007 were multiplied by the appropriate ratio of change before convergence. For example, the ratio of F_{2003} calculated in 2007 relative to 2005 (2.39/2.09 = 1.1460) indicates how much F_{2003} changed in the last two years before stabilizing. F_{2005} is expected to converge in terminal year 2009, or two years after the current estimate with data through 2007 ($F_{2005} = 2.07$). A corrected estimate of F_{2005} in 2009 was calculated by multiplying the estimate of F_{2005} from 2007 by the ratio of change in the last two years before convergence. Specifically, $convergedF_{2005} = 2.07 * 1.146 = 2.37$.

Similar methods were used to correct estimates of SSB, January 1 abundance, and recruitment.

Results of the retrospective correction analysis are presented in Figure C7.3-5. Correcting for the retrospective pattern produces higher estimates of F and lower estimates of SSB, total abundance, and R during the period 2003 to 2007. As noted above, these results are dependent on the assumption that estimates from fishing year 2003 have stabilized with data through 2007 and that the observed retrospective pattern continues into the future. There is no way to validate these assumptions without additional years of data.

C7.3.3 Total mortality

Within ADAPT VPA, calculation of fishing mortality is constructed around the Baranov catch equation (NEFSC 2003). Given abundance of age a+1 in year t+1 and catch of age a in year t, total mortality on age a in year t is calculated iteratively using the equation

$$\frac{N_{a+1,t+1}}{C_{a,t}} = \frac{Z_{a,t}}{F_{a,t}} * \frac{e^{-Z_{a,t}}}{1 - e^{-Z_{a,t}}}.$$

Fishing mortality on age a in year t is then estimated as

$$F_{a,t} = Z_{a,t} - M.$$

Following this procedure, estimates of fishing mortality are dependent on assumptions regarding natural mortality. Incorrect assumptions (*i.e.* inaccurate input values) for M would produce inaccurate estimates of fishing mortality. No direct estimates of natural mortality are available for weakfish, so the WTC has historically operated on the assumption of constant $M = 0.25$ across all years and ages. Alternative assumptions would produce different fishing mortality trends than presented above. If M is indeed constant but of a different magnitude, F estimates would follow a similar trajectory but be scaled up or down from the current estimates. Of greater concern is natural mortality that is not constant across time. Assumptions of constant natural mortality when M is actually increasing over time would lead to overestimates of fishing mortality. Alternatively, a negative trend in M would result in F being underestimated.

To circumvent these issues regarding assumptions about M, it is possible to increase the ADAPT estimated fishing mortality rates by the value of input natural mortality to evaluate trends in total mortality. Since natural mortality was assumed constant at all ages for all years ($M = 0.25$), the trend in total mortality is the same as the trend in F scaled up by 0.25 (Figure C7.3-1). Z declines from values between 1.5 and 2.0 in the early 1980s to the time series low of 0.57 in 1995. In the years following, Z increased nearly exponentially to a peak of more than 2.6 in 2003. Correcting for the retrospective pattern seen in F, total mortality appears to level off at approximately 2.5 to 2.6 before dropping by about 45% in 2007 (Figure C7.3-5).

C7.3.4 Other age-based models

ADAPT VPA operates under the assumption that catch at age is known without error. If catch at age is mis-specified in the input data, the errors are carried through to the results of abundance and fishing mortality at age. If errors in the CAA matrix are known or assumed to be large, other age structured models that are more robust to CAA uncertainty should be explored.

For weakfish, there are several known sources of uncertainty in the CAA. These include conversion of scale-based ages from the 1980s and early 1990s to otolith-based ages used since around 1994, and the substitution of commercial length frequency data to characterize fisheries in states with insufficient data (see section **C5.0**). During the development of the 2002 stock assessment update, the WTC began investigating the use of Integrated Catch at Age Analysis (ICA), which incorporates a statistical error model and is not dependent on error-free CAA (de Silva 2002). Due to loss of staff, this methodology was not pursued beyond the initial investigation.

The most recent peer reviews of the weakfish stock assessment, either as a work in progress (SAW 40, 2008 DPSW) or the completed work (ASMFC 2006a), have focused on concerns with input data. The 2008 DPSW review panel recommended the WTC investigate the use of a forward projecting statistical catch at age model which is robust to errors in the CAA (Miller et al 2009). On that recommendation, the WTC has begun investigating the use of the Age Structured Assessment Program (ASAP) available through the NEFSC toolbox. Trends in fishing mortality and SSB generally follow similar patterns as those estimated in ADAPT (Figure C7.3-6). To date, only preliminary runs have been completed, and the results presented should be considered as examples. The WTC has not had sufficient time to fine tune the model inputs, nor evaluate the reasons for, or the implications of, the discrepancies between the two models. The WTC will continue investigating the use of ASAP and/or other age-structured models for use in future stock assessments.

C7.4 Discussion

Fishing mortality estimated using ADAPT VPA exhibited a generally declining trend between 1982 and 1995, followed a nearly exponential increase during the next decade, and then dropping off in recent years. SSB generally declined through the 1980s, exhibited some rebuilding during the mid to late 1990s, then declined to record lows by 2005. A number of exploratory modeling runs using available data indicate good precision of estimates between 1982 and about 2002. From 2003 to 2007, different sets of tuning indices produced different parameter estimates, and all were compromised by a prominent retrospective pattern. Fishing mortality is generally underestimated, while SSB, January 1 stock abundance, and recruitment are overestimated. Results after 2002 are therefore considered unreliable. An attempt was made to correct for the observed retrospective pattern, but the results have not been endorsed by the WTC.

Implementation of ADAPT for this and previous stock assessments has been conducted under the assumption of constant natural mortality. Violation of this assumption results in inaccurate fishing mortality estimates. The WTC has strong reservations about this assumption and has expressed concern regarding the trend in fishing mortality calculated through ADAPT. As an alternative, the WTC prefers to combine calculated F estimates with input M rates to portray a trend in total mortality, Z. The trend in Z is identical to the trend in F scaled upward by 0.25, and is also influenced by the retrospective pattern in recent years. Regardless, estimates of Z prior to 2002 are not limited by assumptions regarding M and are therefore considered more accurate.

ADAPT VPA has long been the accepted modeling approach for weakfish based on the recommendation of the 26th SARC (NEFSC 1998b). The subsequent peer review (NEFSC 2000) determined that estimates based on ADAPT VPA were suitable for management. In recent years, however, the WTC has doubted the utility of ADAPT for a number of reasons. First, the severe retrospective pattern indicates that parameter estimates in the last five years or more are unreliable. A combined bootstrap and retrospective analysis provides evidence that bootstrapping is insufficient to capture the uncertainty in terminal year estimates. As such, the WTC considers 2002 or 2003 as the most recent year of "acceptable" parameter estimates. Estimates that are now more than five years old are of limited utility for determining future management strategies. Secondly, calculations within ADAPT are conditional upon certain assumptions, either hard programmed (*e.g.* error-free catch at age) or user defined (*e.g.* constant M). The WTC is aware of several sources of uncertainty in the weakfish catch at age matrix, including the scale to otolith conversion, substitution of commercial samples, and commercial discards estimates. The extent of error is unknown, but could be substantial, making ADAPT estimates unreliable. In addition, without direct evidence suggesting otherwise, the WTC has historically assumed natural mortality was constant over all ages and years. This assumption was called into question during development of the 2004 stock assessment when it was observed that stock size began declining under low fishing pressure. Additional work (presented in later sections) provides evidence that suggests weakfish productivity has changed over the available time series, which may be affecting our evaluation of the stock and appropriate management strategies.

The WTC has been tasked to evaluate weakfish stock status in order to support sustainable management of the stock. Without an approved alternative analytical method, results of ADAPT VPA modeling are presented for review and discussion by the 48th SARC. However, given the concerns presented above, the WTC does not believe the terminal year estimates are suitable for management purposes. Beginning with the 2004 assessment, alternative analytical methods have been (and will continue to be) investigated in an attempt to find a more suitable model that is not constrained by (*i.e.* not dependent on, or at least more robust to violations of) assumptions

underlying the current model. The WTC also intends to continue evaluating available data sources to improve historical and future input data. The WTC welcomes specific comments from the SARC on how to improve the input data and model parameterization, and/or comments on appropriate alternative analytical methods. The goal is to find a suitable method that is capable of accurately depicting recent and historical trends in abundance and mortality rates to allow ASMFC the ability to manage for the long term sustainability of the stock.

C8.0 Evaluate the index based methods and the estimates of F, ages 1+ stock biomass, surplus production, and time-varying natural mortality of weakfish produced, along with the uncertainty of those estimates. Determine whether these techniques could complement or substitute for age-based modeling for management advice. (TOR #4)

C8.1 INTRODUCTION

Our ability to assess the current status of Atlantic coast weakfish has been continually plagued by a pronounced retrospective bias associated with all previous ADAPT model runs. Previous ADAPT runs made with trawl-based tuning indices (Kahn et al 2006) exhibited severe systematic retrospective bias that underestimated F and overestimated stock size in the most recent years by as much as 80%. The degree of retrospective bias from ADAPT was so severe that the results gave the false impression that weakfish stock size had actually risen to record high levels by 2002 despite the presence of record low coast-wide landings after 1999. Such a large systematic bias greatly confounds our ability to determine whether or not weakfish abundance has fallen and whether or not the stock is overfished. The degree of retrospective bias from ADAPT was reduced to some extent when the model was tuned specifically to recreational catch-effort data from the MRFSS (Kahn et al 2006).

Because of the aforementioned limitations thus far in the catch-at-age approach, the 2006 Assessment (Kahn et al 2006, Uphoff 2006a; Crecco 2006) relied primarily on an index-based (ages 1+) method to monitor temporal changes in weakfish biomass (mt) and fishing mortality (F) from 1981 to 2003. In the 2006 assessment, relative weakfish abundance was represented solely by the MRFSS recreational cpue from the Mid-Atlantic private boat sector of the recreational fishery. More recently, Uphoff (2008) reported that trends in the MRFSS recreational indices as well as trawl based indices from the States of New Jersey (August data only) and Delaware were highly correlated (P < 0.01) with trends in coast-wide landings and with ages 1+ weakfish abundance from the converged portion (1981-2001) of the most recent VPA model run. Since the index-based approach produced a trend in ages 1+F that closely followed the F trend from the converged (1982-2000) portion of ADAPT without having to rely on the often untested assumption of a fixed M (i. e. F = Z-fixed M), a blended index was derived here based on the recreational cpue, New Jersey and Delaware trawl indices and used to update ages 1+ F and stock biomass estimates (mt) from 1981 to 2008.

Results from the last assessment (Kahn et al 2006) revealed that overfishing (F > Fmsy) had occurred on weakfish from about 1981 to 1991. More stringent fishery regulations were imposed on the weakfish recreational and commercial fisheries in 1992 and 1993 after which fishing mortality (F) rates fell to well below Fmsy from 1996 to 2003. Following a drop in F, coast-wide weakfish abundance initially rose from 1992 until about 1998, but stock size dropped unexpectedly thereafter to the lowest levels in the time series by 2007 despite low and stable fishing mortality (F) rates during this period. Because F levels remained relatively low and stable after 1999, additional analyses have begun to find evidence for trophic interactions involving enhanced striped bass

(*Morone saxatilus*) and spiny dogfish (*Squalus acanthias*) predation or a more complex trophic triangle among weakfish, striped bass and Atlantic menhaden (*Brevoortia tyrannus)* (see sections **C9.0** through **C11.0**).

C8.2 METHODS

C8.2.1 Recreational Abundance Index

A weakfish relative abundance index in numbers (RelNt) was developed annually from 1981 to 2008 (Table C8.2-1) as a recreational catch-effort ratio:

$$\text{RelNt} = \text{MIDN} / \text{Effort}. \tag{1}$$

The recreational catches (MIDN, numbers) (type A, B1 and B2) and fishing effort estimates (Effort in millions of trips) in equation (1) were taken from the MRFSS annual surveys from the private boat sector within the Mid-Atlantic subregion (Virginia to New York). Weakfish catch and fishing effort data were confined to the Mid-Atlantic subregion because private boats catches from this subregion represent the major (> 60%) component of the total coast-wide recreational weakfish catches with acceptably high relative precision about the mean catch estimates after 1984 (annual CV values < 0.13). Moreover, the private boat fishery is highly mobile and capable of catching weakfish of all sizes throughout their range.

A second time series (1981-2008) of weakfish relative abundance in weight (RelWt) was also derived as a ratio of recreational catches (A, B1, B2) in weight (MIDW, mt) to fishing effort (Effort) from the Mid-Atlantic private boat fishery (Table C8.2-1). Since the MRFSS has obtained weight (kg) data from only the harvest (A, B1), the average weight of released weakfish (the B2 component) was assumed to be constant (0.15 kg) each year based on length frequencies (cm) of several thousand released weakfish in the coast-wide head-boat fishery from 2004 to 2007. The weakfish lengths (cm) from this fishery were converted to weight (kg) using the coast-wide length-weight equation for weakfish.

The proposed recreational abundance indices for weakfish (RelNt and RelWt) are fishery dependent and thus partially included in the total (sport, commercial and discards) coast-wide landings. However, the problem of colinearity between the recreational indices and total coast-wide landings should be relatively minor for several reasons. First, auto-correlation between the relative abundance indices (RelNt and RelWt) and total recreational and commercial weakfish landings is minimized by the fact that Mid-Atlantic private boat recreational catches (type A, B1 and B2) rather than harvest (type A and B1) was used to derive the RelNt. The recreational catches are usually three to four times higher each year than the harvest after 1990. Second, the private boat catches (A, B1, B2) in the MRFSS were further divided by private boat fishing effort (Et) in which the effort trend from 1981 to 2003 is inversely correlated (r = -0.39, P <0.08) with the trend in total coast-wide harvest. Finally, the time series (1982 to 2002) of recreational abundance indices (RelNt, RelWt) was shown to be highly correlated (Pearson r = 0.84, P < 0.009) with ages 1+ weakfish stock size from the converged portion (1982 to 2003) of the most recent VPA model run (Uphoff 2008).

Additional age 1+ weakfish abundance indices are available in weight (mean kg/tow) and number (mean N/tow) from the New Jersey (NJ DFW) and Delaware (DE FW) inshore trawl surveys (Table C8.2-2). Uphoff (2008) noted that the time series (1990-2007) of Delaware weakfish trawl indices were highly correlated (P <0.01) to the recreational indices (RelWt), to coast-wide weakfish landings and to age1+ abundance from the converged portion (1990-2000) of the most recent

ADAPT VPA model run. By contrast, the New Jersey age 1+ indices from 1989 to 2007 were poorly correlated (P <0.54) to landings and the converged portion of the VPA. Uphoff (2008) found that the New Jersey trawl indices could be made more informative about trends in coast-wide abundance if the August proportion of positive weakfish (converted to a weight index by multiplying by an estimate of mean weight of exploitable sized weakfish) was used instead of the geometric mean number/tow index from August and October. For this reason, the August indices of positive tows from the New Jersey survey were used to index weakfish relative abundance. Given that these trawl surveys spatially overlap the recreational private boat RelNt indices, the most representative coast-wide weakfish abundance index was chosen as a blended (scaled and averaged) index based on the recreational private boat indices (RelNt, RelWt) (Table C8.2-1), the Delaware trawl indices and the revised August New Jersey trawl indices (Table C8.2-2). The rationale for blending the three indices together seemed justified since trends in the weakfish indices from the three surveys were highly correlated from 1981 to 2007 (Figures C8.2-1 and C8.2-2). Before the three data sets indices could be combined into a coast-wide index **(WkNt, WkWt)**, the time series from each survey had to be standardized to equivalent abundance units. Equivalent units were established in a three-step process. First, the long-term mean abundance index was derived separately for the recreational, Delaware and New Jersey abundance indices. Second, a scalar was derived as a ratio of the long-term average Delaware and New Jersey indices to the long-term average recreational private boat index (catch/trip). Finally, each annual index from the Delaware and New Jersey time series was then multiplied times the respective scalar, thereby transforming the magnitude of the Delaware and New Jersey trawl indices to relative units of the recreational private boat indices (Table C8.2-3). Note that the Delaware and New Jersey indices began in 1990 and 1989, respectively and are not yet available for 2008. For this reason, the 1981 – 1988 and 2008 recreational private boat indices were used to reflect coast-wide weakfish abundance during those years.

C8.2.2 Relative Fishing Mortality (RelFt) and Scaled F Estimates

In this analysis, relative fishing mortality estimates (RelFt) were derived on ages 1+ weakfish from 1981 to 2008. The theoretical foundation of the relative F approach is based on a simple re-arrangement of the Baranov catch equation (Ricker 1975, page 13, equation 1.17) with respect to F:

$$\textbf{F = Harvest / Mean Stock Size,} \qquad \textbf{(2)}$$

where: mean relative stock size in equation 2 is typically expressed as the average of relative abundance indices in years t and t+1. In this analysis, RelFt estimates were based on the ratio of coast-wide annual (commercial and recreational landings plus discards) landings (numbers) of ages 1+ weakfish in year t (Catchnt) to the corresponding blended weakfish relative abundance index (WkNt, WkNt+1) in year t and t+1:

$$\textbf{RelFnt = Catchnt / [(WkNt + WkNt+1)/2].} \qquad \textbf{(3)}$$

Equation (3) is very similar to the equation introduced earlier by Sinclair (1998) except that he used relative exploitation:

$$\textbf{Relu = Catchnt/ RelNt} \qquad \textbf{(4)}$$

In this analysis, average (ages 1+) relative fishing mortality rates in weight and number

(RelFnt and RelFwt) (Table C8.2-4) were estimated from 1981 to 2008 via equation (3). RelFwt values were expressed by the ratio of annual total coast-wide (commercial and recreational plus discards) weakfish landings in weight (Catchw, mt) to the blended relative abundance indices in weight (WkWt) (Table C8.2-4). The 1981-2007 coast-wide weakfish landings (Catchn) in numbers (N*1000) and weight (Catchw) of age 1+ fish were taken directly from age aggregate landings data. The 2008 recreational landings and discard estimates are available from the MRFSS, but the 2008 commercial landings and discard estimates are not yet available. To estimate total commercial and recreational landings and discards (Catchn, Catchw) indirectly in 2008, the 2008 recreational landings were divided by the average ratio of recreational landings to total landings in 2006 and 2007. The same ratio approach was also used to estimate total weakfish discards in 2008.

The relative F estimates from equation 3 do not consider temporal and spatial shifts in the age structure, so this approach is designed only to address average annual shifts in ages 1+ F across time (1981-2008). Thus, the RelFt values are uninformative about year-class and age-specific changes in F over the time series. The strength of the relative F method, however, is in its simplicity and intuitive appeal, allowing scientists to evaluate the relative accuracy of tuning indices and how they might affect the magnitude and trend in F estimates. Most importantly, since RelFt estimates are expressed as a ratio of annual harvest to mean relative abundance, the trends in relative F are not confounded by the often untested assumption of constant natural mortality ($M = 0.25$) used explicitly to derive F estimates ($F = Z - 0.25$) in the ADAPT and Forward Projection models.

The next step in this analysis was to transform the ages 1+ relative fishing mortality rates (RelFnt and RelFwt) from 1981 to 2008 into units of instantaneous fishing mortality (F). This transformation was based on two scalars consisting of the average ratios of F to relative F (RelFnt, RelFwt) across some known time period. The instantaneous fishing mortality (F) rates used for scaling were taken directly from the 2009 ADAPT runs that was configured with tuning indices from the Delaware and New Jersey trawl surveys and North Carolina gillnet survey, as well as from the recreational cpue indices. Since a severe retrospective bias in F was present for all ADAPT runs during recent years (2004-2007), a block of annual F (ages 1+) from 1982 to 1985 was chosen for scaling because the magnitude and trend in F estimates from the converged portion of the model were robust to changes to the tuning indices in ADAPT model runs. Finally, since ages 1+ F estimates for weakfish based on ADAPT were derived assuming a constant natural mortality ($F = Z - 0.25$) across all ages and years, it was assumed that the chosen M of 0.25 on ages 1+ weakfish was a reasonable approximation of average natural mortality over a narrow period (from 1982 to 1985), without having to make the more problematic assumption that M was fixed over the entire (1982-2007) time series.

The first scalar used to transform relative weighted F in numbers (RelFnt) to units of instantaneous F (FNt) consisted of the ratio between the long-term (1982-1985) average numbers weighted F from the most recent ADAPT VPA (Fvpa = 0.58, Se = 0.07) for ages 1+ and the corresponding (1982-1985) RelFn values (Table C8.2-5). The resulting scalar for converting RelFn in numbers to units of FN was 0.0000067. The second scalar from the same ADAPT run was used to convert relative biomass weighted F (RelFw) estimates into units of instantaneous biomass weighted fishing mortality (FWt) (Table C8.2-5). This second scalar consisted of the average biomass weighted F estimates (ages 1+) from 1982 to 1985 (Fvpa = 0.89, SE = 0.12). The resulting scalar used to convert RelFw to units of FW was 0.000024. The time series (1981-2008) of fishing mortality (F) rates in weight (FW) and number (FN) (Table C8.2-4) were derived by multiplying the annual relative fishing rates (RelFn, RelFw) by the corresponding fixed scalar.

C8.2.3 Stock Biomass and Surplus Production

Average stock biomass (Biowt, mt) and average stock numbers (Biont*1000) of ages 1+ weakfish were estimated from 1981 to 2008 (Table C8.2-6). The biomass series (Biowt) was derived by dividing the annual coast-wide weakfish harvest and discards in mt (Catchw) (Table C8.2-4) by the average biomass weighted F on age 1+ weakfish (FWt). The coast-wide number of weakfish (Biont) was determined by dividing coast-wide harvest in numbers and discards (Catchn) by the corresponding ages 1+ F in numbers (FNt) (Table C8.2-6).

A time series (1981-2008) of surplus production estimates in year t (SURPt) was also derived (Table C8.2-6). As in Jacobson et al (2002) and more recently in Walters et al (2008), the SURPt values were expressed each year by subtracting weakfish biomass in year t (Biowt) from the biomass in year t+1 (Biowt+1), and then adding the coast-wide harvest (mt) (Catchw):

$$\text{SURPt} = \text{Biowt+1} - \text{Biowt} + \text{Catchw}. \qquad (5)$$

To examine whether or not overfishing has adversely affected weakfish surplus production (SURPt), the average (ages 1+) biomass weighted F estimates (FWt) lagged one year (t-1) (Table C8.2-4) were linearly regressed against surplus production (SURPt) from 1981 to 2008. If overfishing has recently occurred, the slope of the regression should be negative and statistically significant ($P < 0.05$).

C8.3 RESULTS

C8.3.1 Scaled Fishing Mortality (F)

Biomass weighted fishing mortality (FWt) estimates on ages 1+ weakfish were high (FWt range: 0.69- 1.16) by most standards from 1981 to 1987 (Table C8.2-4). The magnitude of FWt estimates, however, rose even higher to beyond 1.0 from 1988 to 1991 (Table C8.2-4, Figure C8.3-1), and greatly exceeded our current overfishing threshold for weakfish (Fmsy = 0.53). The magnitude of FWt estimates declined steadily thereafter to below 0.60 in most years from 2000 to 2008. The ages 1+ fishing mortality rates weighted by number (FNt) were almost always lower in magnitude than the corresponding biomass weighted fishing rates (FWt) (Table C8.2-4, Figure C8.3-1). The ages 1+ FNt estimates from 1981 to 2008 followed a similar trend over time as the biomass weighted FWt estimates (Table C8.2-4), but unlike the biomass weighted fishing rates (FWt), the FNt estimates fell abruptly after 1991 and remained below 0.30 from 1993 to 2008.

Weakfish biomass levels (Biow) have exhibited wide contrast from 1981 to 2008 (Table C8.2-6, Figure C8.3-2). Weakfish ages 1+ biomass (Biow, mt) remained relatively high (14,200 and 41,500 mt) from 1981 to 1988 but biomass levels fell steadily to below 10,000 mt from 1989 to 1993 (Table C8.2-6). Weakfish coast-wide biomass rose again temporarily from 1994 to 1996, but weakfish biomass fell steadily thereafter to the lowest level in the time series in 2008 (1,333 mt).

The time series of weakfish surplus production (SURPt) from 1981 to 2008 followed the same general trend as stock biomass (Biow) (Table C8.2-6, Figure C8.3-2). Weakfish surplus production remained relatively high from 1982 to 1986 and again in 1993 and 1994 (Table C8.2-6), but SURPt levels fell steadily after 1995 and remained very low in most years from 2001 to 2008 despite relatively low and stable fishing mortality (Figure C8.3-1). The unexpected drop in weakfish surplus production after 1999 coincided with, and may be attributed to, a sharp rise in the abundance of two potential predators: striped bass and spiny dogfish.(Figures C8.3-3 and C8.3-4). Since the

time series of striped bass and spiny dogfish abundance is highly correlated over time (Pearson r = 0.87, P < 0.0001), there is no way to statistically separate their potential predatory effects on weakfish. As a result, a scaled and blended index of striped bass and dogfish abundance (Tpred) (Figure C8.3-4) was developed and used instead of individual striped bass and spiny dogfish indices in the investigation of trophic linkages to weakfish stock dynamics (section **C9.0**)

C8.4 Management and Scientific Implications

Like virtually all single-species stock assessments conducted along the Atlantic coast, natural mortality (M) of age 1+ weakfish was initially assumed to be constant (M = 0.25) in previous Yield-per-Recruit and VPA model runs. Results from the index-based analysis indicated that ages 1+ F estimates remained low and steady from 1999 to 2008 despite a systematic rise in ages 1+ total mortality (Z) after 1998 as indicated by ADAPT. These conflicting trends between Z and F strongly suggest that natural mortality (M) has recently risen systematically (Figure C8.4-1), and was therefore the primary cause for the recent weakfish stock failure along the Atlantic coast. The management consequences of assuming a fixed M when the annual M values actually rises systematically over time can be serious (Swain and Chouinard 2008). As noted in this weakfish assessment, by holding M constant, the resulting ages 1+ fishing mortality rates (F) on weakfish would have exceeded 1.40 in recent years based on the 2009 VPA run despite the recent precipitous drop in landings and tuning indices. If the constant M assumption and ensuing VPA results were accepted without qualification, we would have concluded falsely that the recent failure in weakfish productivity was due to overfishing. In future assessments here and elsewhere, the assumption of constant M for ages 1+ fish needs to be critically examined. In addition, the impacts of trophic and environmental effects on exploited finfish stocks should be integrated into fisheries models and rigorously tested as a potential alternative hypothesis to the overfishing hypothesis.

C9.0 Evaluate testing of fishing and additional trophic and environmental covariates and modeling of hypotheses using biomass dynamic models featuring multiple indices blended into a single index with and without a Steele-Henderson (Type III) predator-prey extension. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice. (TOR #5)

C9.1 INTRODUCTION

Our ability to assess the current status of Atlantic coast weakfish has been continually plagued by a pronounced retrospective bias associated with all previous ADAPT model runs. Previous ADAPT runs made with trawl-based tuning indices (Kahn et al 2006) exhibited severe retrospective bias that underestimated F and overestimated stock size in the most recent years by as much as 80%. The degree of retrospective bias from ADAPT was so severe that the results gave the false impression that weakfish stock size had actually risen to record high levels by 2002 despite the presence of record low coast-wide landings after 1999. Such a large systematic bias greatly confounds our ability to determine whether or not weakfish abundance has fallen and whether or not the stock is overfished. The degree of retrospective bias from ADAPT was reduced to some extent when the model was tuned specifically to recreational catch-effort data from the MRFSS (Kahn et al

2006).

Because of the aforementioned limitations to the catch-at-age approach, the 2006 Assessment (Kahn et al 2006, Uphoff 2006a; Crecco 2006) relied primarily on an index-based (ages 1+) method to monitor temporal changes in weakfish biomass (mt) and fishing mortality (F) from 1981 to 2003. This index-based approach was used to update ages 1+ F and stock biomass estimates through 2008. The index-based analysis for weakfish is described in detail in section **C8.0** of this assessment and was used as the foundation for the following biomass dynamic analyses. Many of the tables and figures presented in section **C8.0** are referred to within this section.

Results from the last assessment (Kahn et al 2006) revealed that overfishing (F > Fmsy) had occurred on weakfish from about 1981 to 1991. As a result, more stringent regulations were imposed on the weakfish recreational and commercial fisheries in 1992 and 1993 after which fishing mortality (F) rates fell to well below Fmsy from 1996 to 2003. Following a drop in F, coast-wide weakfish abundance initially rose from 1992 until about 1998, but stock size dropped unexpectedly thereafter to the lowest levels in the time series by 2007 despite low and stable fishing mortality (F) rates during this period. Because F levels were low after 1999, other studies (Crecco 2006; Uphoff 2006a) have begun to find evidence that linked enhanced striped bass (*Morone saxatilis*) predation to the decline in weakfish productivity after 1998. Striped bass abundance along the Atlantic coast has risen to record high levels after 1998 (Nelson 2007) coincident with the recent failure in weakfish productivity. Larger (> 50 cm) striped bass are known to consume a wide variety of finfish prey including weakfish (Overton et al 2008; Hartman and Brandt 1995; Walter and Austin 2003), and striped bass are known to overlap the temporal and spatial distribution of weakfish north of Cape Hatteras NC (Rudershausen et al 2005). Since predation is generally regarded as a major force structuring marine fish communities (Bax 1991), the age aggregated Steele and Henderson (1984) (S-H) production model was updated through 2008 to further examine the joint effects of fishing and predation on Atlantic coast weakfish. Also, to provide a more thorough examination of the the hypothesis of increased predation on weakfish, additional candidate predators such as spiny dogfish (*Squalus acanthias*), bluefish (*Pomatomus saltatrix*) and summer flounder (*Paralichthys dentatus)* were also considered, especially since all three finfish predators have risen sharply inshore along the Atlantic coast after 1998. Finally, environmental disturbances have been proposed as a major process governing finfish production and recruitment (Hollowed et al 2000b), so environmental factors such as decadal shifts in mean sea surface water temperature and deviations in the winter North Atlantic Oscillation Index were also examined as potential explanatory variables.

The following analysis focuses on the direct interaction between potential predators and weakfish and is hereafter referred to as the "predation hypothesis" to distinguish it from the more indirect "forage hypothesis" presented in section **C10.0**.

C9.2 METHODS

C9.2.1 Recreational Abundance Index

As described in section **C8.2**, a weakfish relative abundance index in numbers (RelNt) was developed annually from 1981 to 2008 (Table C8.2-1) as a recreational catch-effort ratio:

$$\text{RelNt} = \text{MIDN} / \text{Effort.} \qquad\qquad (1)$$

C9.2.2 Relative Fishing Mortality (RelFt) and Scaled F Estimates

In this analysis, relative fishing mortality estimates (RelFt) were derived on ages 1+ weakfish from 1981 to 2008. The theoretical foundation of the relative F approach is based on a simple re-arrangement of the Baranov catch equation (Ricker 1975, page 13, equation 1.17) with respect to F:

$$\textbf{F = Harvest / Mean Stock Size,} \qquad\qquad (2)$$

where: mean relative stock size in equation 2 is typically expressed as the average of relative abundance indices in years t and t+1. In this analysis, RelFt estimates were based on the ratio of coast-wide annual (commercial and recreational landings plus discards) landings (numbers) of ages 1+ weakfish in year t (Catchnt) to the corresponding blended weakfish relative abundance index (WkNt, WkNt+1) in year t and t+1:

$$\textbf{RelFnt = Catchnt / [(WkNt + WkNt+1)/2].} \qquad\qquad (3)$$

Equation (3) is very similar to the equation introduced earlier by Sinclair (1998) except that he used relative exploitation:

$$\textbf{Relu = Catchnt/ RelNt} \qquad\qquad (4)$$

Additional description of these methods and equations is presented in section **C8.2** of this report.

C9.2.3 Stock Biomass and Surplus Production

Average stock biomass (Biowt, mt) and average stock numbers (Biont*1000) of ages 1+ weakfish were estimated from 1981 to 2008 (Table C8.2-6). The biomass series (Biowt) was derived by dividing the annual coast-wide weakfish harvest and discards in mt (Catchw) (Table C8.2-4) by the average biomass weighted F on age 1+ weakfish (FWt). The coast-wide number of weakfish (Biont) was determined by dividing coast-wide harvest in numbers and discards (Catchn) by the corresponding ages 1+ F in numbers (FNt) (Table C8.2-6).

A time series (1981-2008) of surplus production estimates in year t (SURPt) was also derived (Table C8.2-6). As in Jacobson et al (2002) and more recently in Walters et al (2008), the SURPt values were expressed each year by subtracting weakfish biomass in year t (Biowt) from the biomass in year t+1 (Biowt+1), and then adding the coast-wide harvest (mt) (Catchw):

$$\textbf{SURPt = Biowt+1 – Biowt + Catchw.} \qquad\qquad (5)$$

Additional description of these methods and equations is presented in section **C8.2** of this report.

C9.2.4 Overfishing Thresholds (Fmsy, Bmsy)

Surplus production estimates have been used to monitor trends in per capita stock productivity for many exploited finfish populations (Jacobson et al 2002).). Walters et al (2008) noted that few stock assessments conducted thus far have examined the temporal trends in surplus production against biomass and how these trends may relate to the degree of density dependence and to the presence of enhanced trophic and environmental effects. Having a time series (1981-2008) of

weakfish surplus production (SURPt) and ages 1+ stock biomass in year t (Biowt) (Table C8.2-6), updated F_{msy} and B_{msy} thresholds were estimated for weakfish using the dynamic Gompertz external surplus production model (Quinn and Deriso 1999; Jacobson et al 2002). The Gompertz form was selected over the more widely used logistics equation because Yoshimoto and Clarke (1993) reported that, under simulation conditions, the Gompertz model produced more realistic (positive) and stable overfishing thresholds than the logistics model. In the asymmetrical Gompertz model, surplus production estimates (SURPt) from 1981-2008 were regressed against weakfish biomass (Biowt) and the product of the log weakfish biomass and biomass (LogBiowt*Biowt) in a two variable linear regression model without a y-axis intercept:

$$\text{SURPt} = a*\text{Biowt} + b * [(\text{LogBiowt})*\text{Biowt}], \tag{6}$$

where: K – theoretical carrying capacity (mt) = exp (a / b);
        MSY- maximum sustainable yield (mt) = (-b * K)/2.72;
        B_{msy} – stock size (mt) at MSY = K / 2.72;
        F_{msy} – instantaneous fishing mortality at MSY= MSY / B_{msy};
        Fcoll – instantaneous fishing mortality at stock collapse = F_{msy} *2.72.

    Surplus production and stock biomass estimates are often plagued by moderate to high measurement errors (Quinn and Deriso 1999). For this reason, the Gompertz model (equation 6) was fitted as a linear robust regression model using the least trimmed squares regression (LTS) objective function as recommended by Rousseeuw and Van Driessen (2000). The parameter estimates (a, b) and resulting reference points (F_{msy}, N_{msy}, Fcoll) from the production model (equation 6) were derived from the ROBUSTREG procedure contained in the Statistical Analysis System (SAS 2002). The parameter estimates (a, b) and their standard errors based on least squares (LS) are highly prone to the presence of outliers. With robust linear regression like LTS, outlying observations are identified and automatically down-weighted, resulting in higher precision and more robust parameter estimates (a, b) over those derived from ordinary least squares.

    In all model runs with equation 6, residual plots were examined against time (year) to check for the presence of serial correlations. A serial correlation in the residuals would suggest model misspecification, implying that additional biotic and abiotic factors other than fishing may be affecting weakfish surplus production (equation 6). To test for potential joint effects of fishing (F), environmental and trophic variables on weakfish surplus production, additional explanatory variable were added to equation 6 such as striped bass, bluefish, summer flounder and spiny dogfish abundance in a stepwise multiple regression (see section below for details). In these multiple regression models, residual plots across time were also examined to detect for the presence of serial correlations in the residuals. If enhanced predation is adversely affecting weakfish surplus production, the extended production models with predation effects should generate more precise parameter estimates (a, b) in equation (6) and more importantly, the residual plots should exhibit little if any serial correlation over time.

C9.2.5 Screening for Potential Trophic and Environmental Effects

    There is a vast array of potential finfish predators that could prey on weakfish and thus undermine weakfish surplus production and stock size in recent years. Due to the recent and unexpected drop in weakfish stock size since 1999 under relatively low (F < 0.32) fishing mortality (Kahn et al 2006), other factors such as enhanced predation or temporal shifts in environmental

factors may have both played a measurable role in the recent decline in weakfish productivity. In this report, a predation hypothesis was tested for weakfish based on the potential trophic interactions among weakfish, and one or more candidate finfish predators. To be included as a candidate predator in this analysis, each potential finfish predator must overlap the temporal and spatial distribution of weakfish, have a documented history of consuming weakfish and, most importantly, the relative abundance of the candidate predators must have been at high abundance during some portion of the time series (1981-2007) in which weakfish abundance is being evaluated. Four finfish predators including striped bass (*Morone saxatilus*), spiny dogfish (*Squalus acanthias*), bluefish (*Pomatomus saltatrix*) and summer flounder (*Paralichthys dentatus)* were considered as candidate predators on weakfish, especially since all four have risen sharply inshore along the Atlantic coast after 1998. Bluefish, striped bass, summer flounder and spiny dogfish are major inshore finfish predators that have recently risen sharply in abundance along the Atlantic coast coincident with the drop in weakfish productivity. Moreover, these finfish predators overlap the spatial and temporal distribution of weakfish, and all are considered, to some extent, as potential candidate predators on weakfish (Overton et al 2008; Latour et al 2007; Bowman et al 2000). Uphoff (2003) has hypothesized that the predator-prey interaction between striped bass and Atlantic menhaden (*Brevoortia tyrannus*) may play an important role on the degree to which striped bass may have recently switched prey preference from menhaden to weakfish. For this reason, a time series (1981-2007) of menhaden abundance was also included as an additional explanatory variable in the stepwise regressions. Striped bass is regarded as a voracious predator from the Mid and North Atlantic on menhaden, gizzard shad and herring (Hartman 1993). Larger (> 70 cm) striped bass, however, have been reported to eat spot, flounder and weakfish in Chesapeake Bay (Overton et al 2008; Hartman and Brandt 1995; Walter and Austin 2003). A recent study in Chesapeake Bay (Latour et al 2007) reported that medium to large (> 38 cm) summer flounder fed extensively on ages 0 and 1 weakfish. Bluefish and spiny dogfish also prey upon a variety of finfishes including weakfish throughout the Atlantic coast (Bowman et al 2000; Stehlik 2007).

Annual coast-wide abundance of striped bass (ages 8+) in numbers (Table C9.2-1) has been estimated from 1981 to 2006 by the Statistical Catch-at-Age Model (SCAM) (Nelson 2007). Since the striped bass VPA underestimates recent (2004-2006) stock size estimates, a second time series (1981-2008) of coast-wide striped bass abundance (mean catch/trip) was derived as a ratio of striped bass recreational catch from the MRFSS private boat sector to private boat fishing effort (trips) within the Mid and North Atlantic sub-regions (Kahn 2007). This recreational time series is believed to be very informative about trends in coast-wide striped bass abundance since the MRFSS indices were highly correlated to ages 8+ abundance from the converged portion (1982-2001) of SCAM. For this reason, the MRFSS relative abundance indices were scaled to units of ages 8+ striped bass abundance from SCAM and used to reflect striped bass abundance in all subsequent analyses. Annual abundance changes in spiny dogfish, summer flounder and bluefish from 1981 to 2007 were indexed here as catch/trip based on the coast-wide recreational catches in number (A, B1, B2) and coast-wide effort (trips) from the private boat fishery in the MRFSS surveys (Table C9.2-1). These trends in coast-wide recreational cpue of summer flounder, dogfish and bluefish were assumed to be informative about coast-wide trends in these stocks from 1981 to 2008. A time series (1981-2005) of ages 1+ menhaden abundance (numbers) (Table C9.2-1) was taken from the most recent menhaden stock assessment (ASMFC 2006b).

Pearson correlation (r) and stepwise multiple regression models were used to screen for several key biotic and abiotic factors that may be linked statistically ($P < 0.05$) to changes in weakfish surplus production (SURPt) as well as to changes in the blended coast-wide biomass

indices (**WkWt**) 1981 to 2008 (Table C8.2-3). To examine whether surplus production and weakfish relative abundance may be linked to trophic and environmental factors, each of the response variables was related to the time series of the four candidate finfish predators (striped bass, spiny dogfish, summer flounder and bluefish) on weakfish (Table C9.2-1) and two environmental variables (deviations in the mean sea surface water temperature and deviations in the winter North Atlantic Oscillation index (NAO)) in thestepwise regression models. Annual deviations in the winter NAO indices from 1981 to 2008 (Table C9.2-1) were taken from the NOAA web site reported in Collie et al (2008). A time series (1976-2008) of average summer (July-September) sea surface water temperatures (C) was taken from a continuous temperature recorder in Long Island Sound located at the Millstone Nuclear Power Station, Waterford CT (Table C9.2-1). Temperature effects in the stepwise model (equation 1) were expressed as annual deviations from the long-term (1981-1998) mean temperature (devtemp) recorded prior to 1999.

Since enhanced predation should negatively impact weakfish productivity, the stepwise model selected a candidate predator when the slope (b) for predation effects is negative and statistically significant ($P < 0.05$). The stepwise regression models were run in SAS (SAS 2002) using the PROC REG procedure. Temperature effects in the stepwise model (equation 1) were expressed as annual deviations from the long-term (1976-1998) mean temperature (devtemp) recorded prior to 1999. In addition, deviations in the winter North Atlantic Oscillation (NAO) index (**Environj**) from 1981 to 2008 (Table C9.2-1) was also included as a potential explanatory variable in a stepwise linear regression model:

$$\text{WkNt} = a + b * \text{FWt} + c * (\text{Pred, Environj}).\tag{7}$$

Unlike enhanced predatory effects that are assumed to be negative on weakfish surplus production, climatic effects such as rising sea surface temperature could have a positive or negative impact on weakfish productivity. Climatic disturbances (**Environj**) in the form of a decadal rise in water temperature have been proposed as a major process restructuring ecological systems in southern New England (Oviatt 2004). A rise in temperature may cause stress and direct mortality or perhaps alter the competitive advantage of weakfish with other finfishes, thereby restructuring the inshore finfish community. Temporal changes in the NAO are thought to influence wind fields and levels of precipitation over the North Atlantic, thus affecting finfish larval drift, their subsequent recruitment and ecosystem productivity (Collie et al 2008).

There has been a coast-wide commercial and recreational fishery on weakfish for at least a hundred years (Kahn et al 2006). Thus, the time series (1981-2008) of ages 1+ biomass weighted (FWt) fishing mortality rates (Table C8.2-4) was also included as an explanatory variable in the weakfish stepwise (equation 7) regression models. Since an assessment of fishing effects on weakfish is an important priority, fishing mortality was always included (INCLUDE statement in PROC REG procedure) in the stepwise regression models regardless of its level of statistical significance. If fishing mortality effects on weakfish is excessive, the slope for fishing mortality effect will be negative and statistically significant ($P < 0.05$).

Statistical support for the predation and environmental hypotheses would be evident if the slope (c) for predation and environmental effects in equation (7) was negative and statistically significant ($P < 0.05$). This would imply that enhanced trophic and environmental factors have eroded weakfish abundance and surplus production independently of fishery effects. Moreover, if the slope for predation effects is significant, the inclusion of an extra predation term in the Gompertz model (equation 6) can greatly enhance the precision around the (a) and (b) parameters, thus

allowing more precise estimates of Fmsy and Bmsy thresholds. To test for potential joint effects of fishing (F) and trophic interactions on weakfish productivity, residual plots against time were examined for the presence of serial correlations. As a result, a negative and statistically significant slope estimate in equation 8 would imply enhanced predation effects on weakfish surplus production. To test for potential joint effects of fishing (F) and trophic interactions on weakfish productivity, residual plots across time were derived for each production model with and without predation. If predation is an important variable, the plot of residuals from the Gompertz models without predation terms (equation 6) should exhibit a pronounced serial correlation over time, indicating model misspecification. By contrast, the residuals from production models that include predation effects (equation 6) should exhibit no serial correlation over time.

C9.2.6 Age 0 Mortality

One major problem in quantifying predation or other environmental effects on weakfish is pinpointing the life history period where the highest predation risk takes place. A temporal shift in predation mortality can occur across many weakfish ages (ages 0+) or may be confined mainly to a single age group (i. e. age 0 mortality). Since age 0 weakfish rarely exceed 18 cm TL, juvenile weakfish are particularly at risk to an array of potential finfish predators. Several recent predation studies (Beck 1997; Wahle 2003) have shown that a systematic rise in mortality during the juvenile stage may produce a demographic bottleneck that can constrict the flow of recruitment to older ages. If this bottleneck is severe and persists over time, prey abundance will eventually cascade downward, resulting in a stock collapse emanating from the youngest to the oldest ages (i. e. bottom-up effect). To examine whether or not a demographic bottleneck has occurred for age 0 weakfish, a time series of relative weakfish juvenile mortality (Z0) was derived for the 1981 to 2006 year-classes. The Z0 estimate by year-class was expressed by a log ratio between coast-wide age 1 abundance (N1t+1) in year t+1 to the mean coast-wide juvenile abundance index (N0t) in year t:

$$Z0 = - \log (N1t+1 / N0t). \tag{8}$$

Weakfish juvenile abundance surveys (N0) have been conducted along the Atlantic coast from Rhode Island to North Carolina from 1981-2007 (Table C9.2-2). Eight juvenile surveys (Table C9.2-28) were used to construct average coast-wide juvenile indices (N0) from 1981 to 2007. Details of the various surveys are presented in section **C6.0**. Note that juvenile weakfish indices are not yet available in 2008. The abundance indices were expressed as the geometric mean catch per tow, resulting in indices of varying magnitude. As a result, before the indices from the eight surveys were combined into a coast-wide average index (N0), the relative abundance values for each survey had to be standardized to equivalent abundance units. Equivalent units were established in a three-step process. First, the long-term (1982-2007) geometric mean abundance index was derived annually for the recruitment time series of each of the nine surveys. Second, a scalar for each data set was derived as a ratio of the long-term average index to the long-term average New Jersey index. Each annual index from each data set was then multiplied times the respective scalar, thereby transforming the magnitude of the eight indices into units of the New Jersey indices (Table C9.2-2). Finally, the coast-wide geometric mean index (N0) was derived as the grand mean across the scaled indices from 1982-2004 (Table C9.2-3).

A time series (1982-2007) of weakfish age 1 abundance (in millions of fish) used in the numerator of equation (8) was taken directly from the preferred 2009 ADAPT VPA model run (Table C9.2-3). The N1 estimates were derived independently of the relative juvenile abundance

estimates (Table C9.2-2) since none of the juvenile indices were used to tune the VPA. Note that the juvenile abundance indices in the denominator of equation (8) are expressed in relative units so that the juvenile total mortality rates (Z0) are expressed as relative mortality estimates. Also, note that the current VPA generated age 1 abundance estimates (t+1) for the 1981 to 2006 year-classes during the years 1982 to 2007. The age 1 abundance (N1) values from the VPA were larger than the coast-wide juvenile index (Table C9.2-2) resulting in some negative values of mortality (Z0) via equation (8a). To generate positive juvenile mortality rates (Z0), all age 1 abundance data (N1) were reduced in magnitude by dividing N1 by an arbitrary value of 100.0.

To examine the hypothesis that the recent emergence of a recruitment bottleneck between age 0 and age 1 was linked to shifts in trophic and environmental factors, a time series of candidate predators such as striped bass, bluefish and spiny dogfish and summer flounder abundance (Table C9.2-1) were included as explanatory variables in a stepwise regression where the time series of Z0 estimates was the response variable (Table C9.2-3). In addition, environmental variables such as mean summer sea surface water temperature and deviations in the winter North Atlantic Oscillation (NAO) index (environ term) from 1981 to 2007 (Table C9.2-1) were also considered as additional explanatory variables in the stepwise model. Finally, a recent rise in juvenile mortality (Z0) could be due to a rise in weakfish discards from the commercial and recreational fisheries. To test this hypothesis, the annual contribution of fishing mortality due to discards number (Fdisn) was derived as the fraction of annual discards in number to total weakfish landings in number times the current numbers weighted F (FN) (Table C8.2-4). The time series (1981-2007) of Fdisn estimates were included in the stepwise model as a final explanatory variable.

C9.2.7 Stock-Recruitment Effects

In this report, the shape and residual pattern of the weakfish stock-recruitment (S-R) relationship was explored with the Ricker (1975) model:

$$\text{Rec0} = A * Biow * \exp(B * Biow), \tag{9}$$

where: A = the magnitude of compensatory reserve;
   B = the coefficient of compensatory density-dependent mortality;
   Rec0 = blended coast-wide index of weakfish age 0 recruits (Table C9.2-3);
   Biow = average weakfish biomass (mt) (Table C8.2-6).

The parameter estimates (A, B) from the S-R model (equation 9) were derived from the NLIN procedure (Marquardt Algorithm) contained in the Statistical Analysis System (SAS 2002).

Given the likely presence of outliers in the S-R data, the Ricker S-R model was fitted as a nonlinear robust regression using the iterative reweighted least squares method outlined by Holland and Welsch (1978). The algorithm and rationale for this approach is described in SAS (2002). This re-weighting scheme is designed to detect outliers, thereby allowing the down weighting of S-R data from certain years in the model where model residuals, regardless of direction, exceeded a previously defined threshold level. As indicated by Holland and Welsch (1978), the choice of a threshold is subjective and always represents a trade-off between minimizing the variances around the parameters (A, Kp) and at the same time generating globally converged parameter estimates. As suggested by Holland and Welsch (1978), a range of threshold estimates was used initially and the final threshold value was selected that satisfied the trade-off between global convergence of all parameter estimates and parameter estimates with maximum precision and minimum variance. The

two-step re-weighting approach always produced converged estimates (global estimates) that were within 10% of the parameter estimates (A, Kp) derived by the nonlinear least squares approach. However, the standard errors about the estimates based on iterative re-weighting were always 30 to 45% lower than the standard errors from the least squares method.

To examine for potential predatory and environmental effects on the weakfish S-R model, the nonlinear Ricker S-R model (equation 9) was linearized and included in the stepwise model that included an additional exponent (c) reflecting potential predation (pred) and environmental (environ) effects:

$$\text{Log (Rec0/Biow)} = A * Biow * \exp(B * Biow) * \exp(c*pred, environ). \qquad (10)$$

As in the surplus production analyses, potential explanatory variables included striped bass, summer flounder, bluefish and spiny dogfish abundance (Table C9.2-1) as well as environmental variables (environ) including annual sea surface water temperature and deviations in the winter North Atlantic Oscillation (NAO) index (Table C9.2-1). Statistical evidence consistent with predation would exist if additional exponents (c) for predation effects in equation (10) were negative and statistically significant ($P < 0.05$). Further statistical support for the predation hypotheses would be evident, if the serial correlation in residuals evident in the basic Ricker S-R model (equation 9) was minimized following the addition of predation effects to the linearized S-R model (equation 10).

C9.2.8 Steele-Henderson (S-H) Production Model

The last weakfish assessment (Kahn et al 2006) reported statistical evidence of a strong linkage between enhanced striped bass predation and the recent unexpected decline in coast-wide weakfish. As a result, the age aggregated Steele-Henderson (S-H) production model (Steele and Henderson 1984) was updated to link fishing and predation effects to the recent drop in weakfish productivity. The S-H model has extensive theoretical appeal since it incorporates the compensatory stock dynamics of the prey (weakfish) with fishing effects, plus a sigmoid foraging response by the predatory finfish that may lead to critical depensation at low prey abundance (Spencer and Collie 1997b). The Steele-Henderson (S-H) model incorporates compensatory stock dynamics of the prey with fishing effects plus a sigmoid type III functional response by the predator. The Type III response adds a degree of realism to the model since it may lead to either prey stability at low to intermediate predator abundance, or to critical depensation of the prey at low prey abundance (Spencer and Collie 1997b; Collie and DeLong 1999). The age aggregated Steele-Henderson (S-H) production model was used to estimate equilibrium and time varying overfishing thresholds (F_{msy}, N_{msy}) for weakfish in the presence of a significant ($P < 0.05$) predatory response. The S-H model assumes the existence of compensatory density-dependent mortality for finfish populations, a position widely held by most fish population ecologists (Wahle 2003). All of the weakfish population dynamics processes (somatic growth, natural mortality and recruitment) in the S-H model are subsumed in the intrinsic rate of population increase (r) and to a lesser extent in the carrying capacity (K) parameters. Like all production models, successful fitting (precise and robust parameter estimates) of the S-H model requires a high degree of contrast in the time series (1981-2007) of stock sizes. The S-H model was originally configured as a logistics production model with an added sigmoid function that reflected the foraging response by the predator. Previous simulation studies (Yoshimoto and Clarke 1993) have indicated that the Gompertz asymmetrical model produced more realistic (positive values of F_{msy}) and robust parameter estimates than the logistics model. As a result, the surplus production portion of the S-H model was converted from the logistics to the

Gompertz form:

Biowt+1= Biowt+log(K)*r*Biowt*(1-(log(Biowt)/log(K)))- Catchw-

(c*Pred*(Biowt)**2)/(A**2+(Biowt)**2)] (11)

where: Biowt+1 =weakfish stock biomass (mt) in year t+1 (Table C8.2-6);
Biowt = weakfish stock biomass (mt) in year t (Table C8.2-6);
Pred = abundance of the selected predator during year t (Table C9.2-1);
Catchw = harvest and discards (mt) of weakfish in year t (Table C8.2-4);
K = estimated carrying capacity of weakfish biomass;
r = intrinsic rate of weakfish population increase;
c = per capita consumption rate of the predator;
A = weakfish biomass (mt) at which predator satiation takes place.

All parameter estimates (r, K, c and A) from the S-H model (equation 11) were derived from the NLIN procedure (marquardt algorithm) contained in the Statistical Analysis System (SAS 2002).

The S-H model was fitted to weakfish biomass (Biowt, Biowt+1) (Table C8.2-6) and the abundance of one or more predators (Pred) by nonlinear least squares regression methods. The choice of one or more predators in equation (9) depends on the outcome of the stepwise regression procedure. Given the likely presence of measurement errors in the input data, the S-H model was fitted as a nonlinear robust regression using the iterative reweighted least squares method outlined by Holland and Welsch (1978).

As indicated by Spencer and Collie (1997b), the S-H model (equation 11) represents the merger of two models; one is a conventional Gompertz discrete time dynamic production model with only fishing effects (Catchw) present:

Biowt+1 = Biowt+log(K)*r*Biowt*(1-(log(Biowt)/log(K)))- Catchw (12)

whereas the other model expresses additional predatory effects via the Type III functional response:

[(c*Pred*(Biow$_t$) ^{2})/(A^2+(Biowt) ^{2})] (13)

To more fully examine whether or not fishing mortality (Ft) alone might account for the systematic decline in weakfish abundance, the discrete time Gompertz portion (equation 12) of the full S-H model (equation 11) was fitted separately to weakfish stock sizes (Biowt, Biowt+1) from 1981 to 2007 using nonlinear robust regression methods. If fishing effects are largely responsible for the current decline in weakfish, the predicted stocks sizes should closely follow the observed abundances (high coefficient of determination), the resulting parameter estimates (r, K) from the discrete Gompertz model (equation 12) should differ significantly (P <0.05) from zero, and the resulting equilibrium F_{msy} threshold estimate (r) should at least approximate previous F_{msy} estimates for weakfish based on previous stock assessment (Kahn et al 2006). If the discrete time model fails in this regard, then the full S-H model with predation effects (equation 11) was then applied to

determine whether or not the extended model could fulfill all three of the aforementioned requirements.

Uphoff (2006) noted that if the predation parameter estimates (c, A) from the S-H model are sufficiently robust and precise, then a time series of weakfish biomass consumed (Dt in mt) annually by predation (Pred) can be derived in the form:

$$Dt = [(c*Pred*(Biowt)**2) / (A^2 + (Biowt)^2] \qquad (14)$$

Once (Dt) is estimated via equation (14), the weakfish instantaneous consumption rate associated with predation (Mpt) can be derived annually for an annual predator:

$$Mpt = Dt /[(Biowt+Biowt+1)/2)] \ . \qquad (15)$$

In the dynamic Gompertz production model without predation (equation 12), the equilibrium F_{msy} threshold is solely expressed by the intrinsic rate (r) parameter, whereas B_{msy} is expressed by the carrying capacity (K) divided by 2.72 (Quinn and Deriso 1999). Since temporal shifts in striped bass predation are absent from the traditional dynamic models, the overfishing definitions (F_{msy}, B_{msy}) in these models are fixed in time. However, in the non-equilibrium S-H model (equation 11) the ability to identify steady-state conditions is far more difficult because predation effects are transient over time. In the non-equilibrium S-H model, weakfish surplus production and predation-induced mortality (Mpt) can vary greatly across years, resulting in time varying F_{msy} and B_{msy} thresholds. The degree of temporal variation in F_{msy} and B_{msy} depends on the magnitude and trend in predator abundance, the predator consumption exponent (c) and on the prey stock size (A) at which the consumption threshold of predation takes place in equation (11). Thus, the annual $F_{msy\ t}$ values from the S-H model are not fixed in time but rather are a function of the fixed intrinsic rate (r) minus the time varying predator consumption rate (Mpt):

$$F_{msy\ t} = r * exp(- Mpt) \ . \qquad (16)$$

Similarly, the weakfish biomass threshold (B_{msy}) can vary over time depending on the amount of weakfish biomass consumed annually by the predator (Dt):

$$B_{msy} = [K- Dt] / 2.72 \ . \qquad (17)$$

Although weakfish overfishing thresholds ($F_{msy\ t}$, $B_{msy\ t}$) derived from the S- H model are time varying, equilibrium reference points can be approximated as the long-term (1981-2007) mean F_{msy} t and $B_{msy\ t}$.

C9.3 RESULTS

C9.3.1 Scaled Ages 1+ Fishing Mortality (F), Biomass and Surplus Production

Estimates of rescaled relative F, weakfish stock biomass, and weakfish surplus production

are presented in section **C8.3** of this report.

Juvenile and Age 1 Abundance Including Juvenile Mortality (Z0)

The relative magnitude of weakfish juvenile indices varied greatly across the eight surveys for the 1981 to 2007 year-classes (Table C9.2-2). When the eight juvenile data sets were scaled and blended into a mean coast-wide index (Rec0) (Table C9.2-3), the coast-wide index (Rec0) showed a fairly persistent rise after 1995 (Figure C9.3-1). Although coast-wide average recruitment (Rec0) remained high in most years from 1998 to 2007, a pronounced drop in weakfish ages 1+ abundance (Nvpa) from the 2009 VPA was clearly evident from 1998 to 2006 (Table C9.2-3). Abundance of age 1 weakfish (Nvpa) from the most recent VPA run varied without trend from 1982 to about 1998, then age 1 abundance fell steadily thereafter in sharp contrast to the general rise in juvenile recruitment after 1995 (Figure C9.3-1). The resulting weakfish juvenile mortality (Z0) estimates increased in magnitude for the 1999 to 2006 year-classes (Table C9.2-3), and were positively correlated (Pearson $r = 0.88$, $P < 0.0001$) to the blended predation index (Tpred) consisting of striped bass and spiny dogfish (Figure C9.3-2). These findings strongly suggest that recent rise in juvenile natural mortality since 1998 is consistent with the recent emergence of a trophic bottleneck that may have interfered with the flow of recruitment from age 0 to ages 1+.

C9.3.2 Correlation Matrix and Stepwise Regression

The time series of weakfish surplus production (SURPt) (Table C8.2-6), age aggregated biomass index (WkW) (Table C8.2-3) and weakfish juvenile mortality (Z0) from 1981 to 2008 (Table C9.2-3) was examined in a correlation matrix against each of the ten potential explanatory variables (Table C9.2-1). The correlation matrix revealed that striped bass and spiny dogfish abundance were always inversely related ($P < 0.001$) to weakfish surplus production and weakfish biomass and positively related to juvenile mortality (Z0) (Table C9.3-1, Figure C9.3-2). The time series of menhaden abundance, a major finfish prey of both striped bass and spiny dogfish, was positively related ($P < 0.001$) to weakfish surplus production and weakfish biomass and inversely related to juvenile weakfish mortality (Table C9.3-1), suggesting that a recent drop in menhaden abundance may have enhanced the predation risk of weakfish by spiny dogfish and striped bass. None of the other potential explanatory variables, including fishing mortality (FWt) and discard effects, were correlated across all three (SURPt, WkW, Z0) response variables. These findings strongly suggest that the recent failure in weakfish productivity after 1998 was linked mainly to enhanced predation from striped bass and spiny dogfish, the magnitude of which appeared to be enhanced by the recent coast-wide decline in Atlantic menhaden.

The time series of weakfish juvenile mortality rates (Z0) for the 1981 to 2006 year-classes (Table C9.2-3, Figure C9.3-2) were used as a response variable in the stepwise regression model against the blended striped bass and dogfish abundance index (Tpred) as well as with the five additional explanatory variables (Table C9.2-1). The stepwise model selected only the blended predation index (Tpred) as the most significant ($P < 0.0001$) explanatory variable that was closely tied (Pearson $r = 0.84$) to the rise in weakfish juvenile mortality (Z0) (Table C9.3-2, Figure C9.3-2). No other explanatory variable was selected at the $P < 0.05$ level in the stepwise model. It is important to note that estimated weakfish coast-wide discards (Disn) and discard-related fishing mortality (FDisn) (Table C8.2-4) were included in the models but were not positively correlated ($P < 0.05$) to juvenile mortality (Z0). These findings suggest that fisheries-related activities were not tied directly to the recent rise in weakfish juvenile mortality. The emergence of an age 0 recruitment bottleneck after 1996 appeared to be linked mainly to enhanced predation from striped bass and

spiny dogfish and rather than to enhanced fisheries discards.

Weakfish surplus production (SURP) and coast-wide relative biomass (WKW) (Tables C8.2-3 and C8.2-6) from 2001 to 2008 were also related to six explanatory variables in stepwise regression models where biomass weighted fishing mortality (FWt) effects were always included (Table C9.3-2). Once again, the stepwise model always selected the blended predation index (Tpred), comprised of striped bass and spiny dogfish, as the most significant ($P < 0.0001$) explanatory variable (Table C9.3-2). The Tpred variable was inversely related (Pearson r = -0.80) to the post 1999 decline in weakfish surplus production (Figure C8.3-3) and biomass. Although biomass weighted fishing mortality effects (FW) were independent ($P < 0.45$) of weakfish surplus production (SURP), FWt were negative and statistically significant ($P < 0.007$) on weakfish relative biomass (WKW) but only in the presence of predation (Tpred) (Table C9.3-2). The biotic mechanism(s) behind the apparent negative interaction of predation and fishing mortality on weakfish biomass are not clear at this time. Most of the statistical evidence given thus far suggests that the recent (1999-2008) failure in weakfish surplus production and stock biomass was due primarily to enhanced predation from striped bass and spiny dogfish.

C9.3.4 *Gompertz Production Models With and Without Predation Effects*

The external Gompertz surplus production model (equation 6) was fitted by linear robust regression to weakfish stock biomass in year t and t+1 (Biot, Biot1) (Table C8.2-6). This model with only fishing effects (Catchw) did provide a reasonably good fit ($r**2 = 0.54$) to the biomass time series with statistically significant ($P < 0.05$) r and K parameter estimates (Table C9.3-3). The resulting overfishing thresholds (Fmsy, Bmsy) of 0.26 and 57,388 mt, respectively, were much different than earlier estimates (Fmsy = 0.41, Bmsy = 25,400 mt) based on the Steele-Henderson model in 2005. Moreover, the resulting production model consistently over predicted weakfish surplus production from 1998 to 2008 by 20% to 350% (Figure C9.3-3), indicating the presence of a distinct serial pattern in the residuals. This systematic trend in the direction of the residuals usually indicates that the model is lacking an informative parameter(s).

When the blended predation parameter (Tpred) was added to the basic Gompertz model, the fit to surplus production greatly improved ($r**2 = 0.84$) and the parameters (r, K, C) were estimated with much higher precision than the production model without predation (Table C9.3-3). The slope representing predation effects (C) was negative and statistically significant ($P < 0.0001$) which is consistent with the significant inverse relationship (Pearson r = -0.87, $P < 0.0001$) between weakfish surplus production and blended predation from 1981 to 2008 (Figure C8.3-4). The resulting equilibrium overfishing thresholds (Fmsy, Bmsy) from the extended Gompertz model were 0.54 and 19,289 mt, respectively (Table C9.3-3). Moreover, unlike the strong serial correlation in the residuals noted from the basic Gompertz model (Figure C9.3.3), there is little if any systematic residual pattern from the extended Gompertz model that includes predation effects of striped bass and spiny dogfish (Figure C9.3-4).

C9.3.5 *Stock –Recruitment Effects*

The Ricker S-R (equation 9) model was applied to relative coast-wide juvenile recruitment (Rec 0) (Table C9.2-3) and weakfish biomass (Biow, mt) (Table C8.2-6) from 1981-2007 using iterative re-weighted nonlinear least squares regression. The dome-shaped Ricker stock-recruitment (S-R) model was a good fit ($r^2 = 0.52$) to the S-R data from 1981 to 2007 (Figure C9.3-5). The steep ascending limb and pronounced dome indicated the strong presence of density-dependent mortality and the ability of weakfish to compensate for relatively high levels ($F < 0.8$) of fishing mortality.

The resulting parameter estimates were precise (A = 0.016, SE = 0.004, B = -0.00015, SE = 0.000027), both of which differed significantly (P <0.05) from zero (Table C9.3-4). However, the residual plot over time indicated the presence of a distinct serial correlation (Figure C9.3-6). The residuals were mostly negative from 1981 to 1994 but the direction of the residuals shifted abruptly in a positive direction thereafter (Figure C9.3-6), indicating model misspecification.

When the linearized version of the Ricker S-R model (equation 10) was fitted to several candidate predators (blended predators, summer flounder and bluefish) and environmental factors (deviations in summer sea surface temperatures and deviations in the NAO index) the stepwise model chose the blended predators (tpred) as the only significant explanatory variable to the S-R model. When tpred was added as a second explanatory variable to the S-R model, the extended model explained 81% of the recruitment variation, all three parameter estimates (A, B, c) became highly significant (P <0.0001) (Table C9.3-4) and, most importantly, the anomalous residual pattern seen in the basic Ricker S-R models virtually disappeared when Tpred was added to the model (Figure C9.3-7). These finding are consistent with the Predation Hypothesis, indicating that the transmission of age 0 weakfish recruits to the adult stock has been recently impeded due to enhanced predation from striped bass and spiny dogfish.

C9.3.6 Steele-Henderson Model

The discrete version of the Gompertz surplus production model with only fishing effects (equation 12) was fitted by nonlinear least squares and iterative re-weighted least squares regression to weakfish stock biomass in year t and t+1 (Biowt, Biowt+1) (Table C8.2-6). Like the results from the Gompertz external model (Table C9.3-3), this model provided a good fit ($r^{**}2$ = 0.68) to the biomass data with statistically significant (P < 0.05) r and K parameter estimates (Table C9.3-6). The resulting overfishing thresholds (Fmsy, Bmsy) for weakfish of 0.32 and 25,259 mt, respectively, were similar in magnitude to previous estimates. However, this production model consistently over predicted weakfish biomass from 1998 to 2008 by 10 to 250% (Figure C9.3-8), indicating that the basic Gompertz model was plagued by substantial process error. The model (equation 12) applied by iterative reweighting least squares substantially improved the fit ($r^{**}2$ = 0.82) and enhanced precision around the parameter estimates (r, K) and overfishing definitions (Fmsy, Bmsy) (Table C9.3-6), but severe process error nevertheless persisted in the residuals from 1998 to 2008. Even when the Logistics form of the surplus production model was used instead of the Gompertz, the same serial residual pattern persisted over time, indicating that the residual problem was not due to the configuration of the production model. Given the clear residual problem associated with the basic Gompertz and Logistics models, the surplus production models without predation were not used to estimate overfishing thresholds for weakfish.

The full Steele-Henderson (S-H) production model (equation 11) was applied to weakfish stock biomass in year t and t+1 (Biot, Biot1) and to the blended predators (Tpred) by iterative re-weighting (Tables C9.3-5 and C9.3-6). The S-H models provided a very good fit ($r^{**}2$ = 0.84 for unweighted and 0.94 with iterative reweighting) to the biomass and predation data (Figure C9.3-10) with statistically significant (P < 0.05) r, K, c and A parameter estimates (Table C9.3-6). Overall, the S-H models was not only a better fit to weakfish biomass than the Gompertz model with only fishing effects (Table C9.3-6), but the r and K parameters were estimated with much higher precision. The resulting weakfish equilibrium overfishing thresholds (Fmsy, Bmsy) from the nonlinear least squares S-H models for weakfish was 0.72 and 17,009 mt, respectively (Table C9.3-6). Moreover, unlike the severe residual pattern evident from 1998 to 2008 based on the basic Gompertz models (Figure C9.3-3), there is little if any systematic residual pattern from the S-H model fitted by iterative re-

weighting (Figure C9.3-9). Finally, the time series (1981-2008) of predation-based natural mortality (Mp) from striped bass and spiny dogfish (Tpred) was derived via equations (14) and (15) (Table C9.3-5). The predatory mortality rates (Mp) rose two to four fold in magnitude after 1997 coincident with a steady drop in weakfish biomass and surplus production. This inverse relationship between Mp and weakfish biomass is consistent with the presence of depensatory density-dependent predation mortality and, if persistent over time, could be highly destabilizing to future weakfish stock rebuilding. These findings strongly suggest that the recent rise in weakfish ages 1+ natural mortality (MWt) coupled with the failure in weakfish productivity (SURPt) are tied directly to the increase in striped bass and spiny dogfish predation mortality (Mp).

   The estimated instantaneous consumption rates (Mp) of weakfish by striped bass and spiny dogfish rose steadily in magnitude from 0.22 in 1986 to 1.64 in 2002 and then remained relatively high thereafter (Table C9.3-5, Figure C9.3-11). The estimated biomass (mt) of weakfish consumed (Dt) annually exceeded 4000 mt in most years from 1995 to 2008 during which weakfish biomass fell by 80% after 2000 (Table C9.3-5). Despite the systematic decline in weakfish biomass after 1999, the magnitude of weakfish biomass (Dt, mt) consumed by striped bass and spiny dogfish remained relatively high, equal or exceeding the weakfish coast-wide biomass after 2001 (Table C9.3-5, Figure C9.3-11). Moreover, weakfish biomass consumed (Dt) annually and the instantaneous consumption rates (Mp) from 1997 to 2008 (Table C9.3-5) were 30% to 300% higher than the coast-wide landings and fishing mortality (F) rates, respectively, on weakfish during those years (Table C9.3-7, Figure C9.3-12). The effects of enhanced predatory consumption (Mp) coupled with moderate fishing mortality (FW) on weakfish from 2000 to 2008 easily exceeded the Fmsy threshold, resulting in the systematic decline in weakfish surplus production and biomass from 2001 to 2008 (Figure C8.3-2).

   When the equilibrium overfishing thresholds (Fmsy = 0.72, Bmsy = 17,009 mt) from the S-H model (Table C9.3-6) are considered, ages 1+ biomass weighted fishing mortality (FW) on weakfish exceeded the estimated Fmsy threshold of 0.72 in most years from 1981 to 1992 (Figure C9.3-13). Weakfish coast-wide biomass (mt) exceeded the biomass threshold (Bmsy = 17,009 mt) in 1981 and 1982, but biomass fell quickly below Bmsy thereafter (Figure C9.3-14). When more stringent management regulations were enacted after 1991, fishing mortality (FW) fell by 50 to 70% (Figure C9.3-13) and biomass began to rise toward the Bmsy threshold (Figure C9.3-14). However, weakfish biomass fell unexpectedly after 1999 to the lowest level in the time series in 2007 (Figure C9.3-14) despite the fact that fishing mortality rates remained below Fmsy in most years (Figure C9.3-13) from 1998 to 2008 (exception: 2002).

   Because predatory consumption rates (Mp) on weakfish increased about five fold after 2000 (Figure C9.3-10), the overfishing definitions (F_{msy}, B_{msy}) did not remain fixed over time (Table C9.3-5). The non-equilibrium F_{msy} levels via equation (16) remained relatively stable around 0.60 to 0.78 from 1981 to 1989 during which weakfish biomass and surplus production remained high and stable (Figure C8.3-2). However, when predatory consumption rates (Mp) rose after 1997 (Figure C9.3-11) and surplus production dropped, annual $F_{msy\ t}$ thresholds dropped sharply from around 0.60 to 0.70 in the 1980's to below 0.30 after 2000 (Table C9.3-5) in concert with a steep rise in striped bass and spiny dogfish abundance (Figure C8.3-3) and predation related mortality rates (Mp) (Figure C9.3-11). In contrast, non-equilibrium biomass thresholds (B_{msy}) were more robust to rising Mp (Table C9.3-5). The B_{msy} thresholds remained relatively stable at between 14,000 and 15,000 mt from 1995 to 2008 despite rising Mp and Dt (Figures C9.3-11 and C9.3-12).

C9.4 Scientific and Management Implications

The preponderance of statistical evidence given here supports the Predation Hypothesis involving enhanced predation by striped bass and spiny dogfish as the primary factor behind the recent and unexpected decline in weakfish productivity. Statistical evidence in support of the Predation Hypothesis consists of a significant (P <0.0001) inverse correlation between declining weakfish biomass and surplus production from 1999 to 2008 and striped bass and spiny dogfish abundance from 1982 to 2004. Striped bass abundance along the Atlantic coast rose 10 fold from 1994 to 2006 (Kahn 2005), although the 2008 striped bass abundance estimate fell by over 40% since 2006. Similarly, spiny dogfish abundance has increased 10 fold since 1999 and has remained high thereafter. During this recent period (1999-2008) of declining weakfish productivity, fishing mortality (FW) and discard mortality (Fdisc) rates remained low and relatively stable, indicating that the recent drop in weakfish productivity did not coincide with rising exploitation. The strong positive correlation (Pearson r = 0.91, P <0.0001) between the recent rise in weakfish juvenile mortality (Z0) and rising striped bass and spiny dogfish abundance further suggests that the recent emergence of a weakfish recruitment bottleneck at age 0 was largely due to enhanced predation by these two finfish predators. By contrast, discard mortality estimates of small (< age 2) weakfish remained low and stable after 1999 during which juvenile mortality (Z0) rose steadily. Third, the residual patterns in all logistics and Gompertz model runs that included only fishing effects (landings) produced inordinately low overfishing thresholds (Fmsy, Bmsy), poor precision around the estimates, and the residuals exhibited a pronounced serially correlation over time, clearly indicating model misspecification. However, when the predation term (Tpred), reflecting the joint predation by striped bass and spiny dogfish, was added to the models, the fit of the models to weakfish surplus production and biomass dramatically improved, the precision and magnitude of Fmsy and Bmsy rose to more plausible levels (Fmsy > 0.45), and, most importantly, the direction of the residuals over time shifted to a more random pattern and were therefore free of model misspecification. Finally the instantaneous consumption rates (Mp) on ages 1+ weakfish derived empirically from the Steele-Henderson model from 1999 to 2008 were closely correlated (Pearson r = 0.81, P < 0.0002) to the rise in ages 1+ total mortality (Z) derived independently from the most recent ADAPT VPA run. This strongly suggests that the recent increase in ages 1+ total mortality (Z) in the VPA was not due to a rise in fishing mortality (F) but rather to enhanced predation by striped bass and spiny dogfish..

Although results from regression and production models alone do not demonstrate causality, recent empirical evidence is consistent with the Predation Hypothesis involving striped bass and spiny dogfish. Striped bass are known to consume finfish prey up to 60% of their own body length (Manooch 1973). Unlike fluke and bluefish that prey mainly on small (< 30 cm) fish, large (> 76 cm) striped bass can easily prey on larger (> 40 cm) weakfish, whereas smaller stripers (< 65 cm) can effectively prey on juvenile and age 1 weakfish. Recent food habits studies of spiny dogfish show that dogfish prey on a wide variety of finfishes including weakfish (Stenlik 2007; unpublished NEAMAP data). Striped bass and spiny dogfish seem to prey more commonly on menhaden and river herring (*Alosa* spp) (Uphoff 2003: Bowman et al 2000). But coast-wide menhaden and river herring abundance has fallen dramatically after 1995 (citation), thereby enhancing the prospects that striped bass and spiny dogfish would recently switch to alternative prey such as weakfish and summer flounder. Within the forage hypothesis model (section **C10.0**) it was shown that the decline in menhaden was likely responsible for the recent rise in striped bass predation on weakfish after 1996. Moreover, the recent drop in weakfish productivity after 1997 was shown to have a spatial component that matched the distribution of striped bass. Weakfish landings and surplus production

fell rapidly after 1998 from the Mid and North Atlantic subregions, but surplus production from the South Atlantic subregion had remained steady (Crecco 2005b). It so happens that anadromous striped bass are very abundant along the Mid and North Atlantic coast throughout the year, but are seldom encountered in any numbers south of Cape Lookout, NC. Spiny dogfish are found coast-wide from Canada to Florida (Stehnik 2007), but their post 1998 rise inshore has occurred mostly along the Mid and North Atlantic subregions, thus allowing spiny dogfish to spatially overlap weakfish. The predator assemblage south of Cape Lookout, NC is largely a different field of predators such as channel bass, spotted seatrout, king mackerel, and cobia. This well -defined spatial match between prey (weakfish) and predators (striped bass and spiny dogfish) clearly supports the validity of the Predation Hypothesis. Further empirical support of the hypothesis includes recent food habit studies of striped bass (Walter and Austin 2003; Rudershausen et al 2005), indicating that weakfish and other sciaenids (spot) were primary food items of larger (> 60 cm) striped bass in Chesapeake Bay and Albemarle Sound. Moreover, a recent rise in striped bass abundance has been linked empirically to the steady decline of blueback herring in the Connecticut River and Albemarle Sound (Savoy and Crecco 2004; Rudershausen et al 2005), as well as with the menhaden decline from Chesapeake Bay (Uphoff 2003; Walter and Austin 2003).

Although most of the statistical and empirical evidence given here and elsewhere (Uphoff 2009, this assessment) supports the Predation Hypothesis, other factors such as unreported commercial and recreational landings and discards, disease, toxins and parasitism cannot be ruled out at this time to explain the annual production loss of between 3,000 and 5,000 mt of weakfish. At this time, there is no evidence that would link recent increases in disease, toxins and parasitism to the recent failure of weakfish. There has been a recent rise in sea surface water temperatures along the Atlantic coast (Oviatt 2004), but this analyses indicated that water temperature shifts were not significantly (P <0.05) linked to recent increases in weakfish juvenile mortality (Z_0), nor in the decline in weakfish surplus production and stock biomass. It is possible that an enormous upsurge in unreported weakfish landings and commercial and recreational discards took place between 1996 and 2008 to account for the estimated 3,000 to 4,000 mt annual loss of weakfish surplus production, but a recent upsurge in unreported landings seems unlikely for several reasons. First, if the sources of this rapid upsurge in unreported weakfish landings and discards are thus far unknown, it would be nearly impossible to remove this source of mortality without closing virtually all inshore fishing activity between North Carolina and Rhode Island. Second, if a recent rise in unreported landings and discards resulted in the recent weakfish stock collapse, we would expect that other finfish stocks with a similar temporal and spatial distribution as weakfish (i. e. Atlantic croaker and summer flounder) to be likewise depleted. But Atlantic croaker and summer flounder stocks have either grown or have remained relatively stable from 1998 to 2008. Third, if a recent rise in weakfish unreported landings caused the recent weakfish stock collapse, all of the statistical and empirical evidence presented elsewhere in this assessment on enhanced predation would have to be regarded as a mere coincidence. Finally, unreported landings in the order of 3,000 to 4,000 mt annually are equivalent to about 5 times the current (2007-2008) known landings and estimated discards used in this weakfish assessment. The possibility that such an astronomical rise in unreported landings and discards took place after 1998 and then remain hidden from port agents, enforcement and management agencies seems remote.

The strong inverse relationship between predatory consumption rates (M_p) rates and weakfish biomass since 1997 is consistent with the presence of depensatory density-dependent mortality. This phenomenon plus the apparent emergence of a weakfish recruitment bottleneck between ages 0 and 1 makes stock rebuilding via the implementation of fisheries management

measures an exceedingly difficult task. As indicated by Spencer and Collie 1997b), fish stocks that are subject to moderate to severe depensatory predatory mortality, often undergo a sudden and persistent drop in surplus production over time even when fishing mortality rates have remained low for several years. Note that the biomass weighted fishing mortality (FWt) on ages 1+ weakfish have been below the Steele-Henderson Fmsy of 0.72 in all but one year since 1996. Under depensatory predation, the weakfish stock would be expected to remain low and unresponsive to favorable climatic events and to further fishery management restrictions. Note that weakfish stock biomass has fallen steadily since 1999 to a low 2008 stock biomass level of 1,333 mt which is about 92% below the new steady-state Bmsy level of 17,100 mt. The phenomenon of depensatory mortality, if driven largely by striped bass and spiny dogfish predation, could lead to a persistent and perhaps irreversible failure in weakfish productivity unless striped bass and spiny dogfish productivity in the next few years reverts back to pre 1998 levels.

The pessimistic outlook regarding the future of the Atlantic coast weakfish due to depensatory predation may be tempered somewhat by findings of compensatory density-dependent survival during or before the juvenile stage. Compensatory mortality was manifested here by the steep ascending limb and pronounced dome on the Ricker stock-recruitment curve, as well as by the recent and persistent rise in the coast-wide juvenile recruitment (N0) after 1995. This apparent recruitment surge took place despite the pronounced rise in juvenile mortality (Z0) and drop in weakfish biomass after 1998. It is notoriously difficult to sort out the ramifications on future weakfish stock growth when both depensatory and compensatory mechanisms are operating simultaneously. However, the added stock resiliency due to compensatory processes at the juvenile stage may persist over several more years. If so, it may in fact overcome or at least balance out the adverse effects of depensatory predation, allowing the weakfish stock to achieve equilibrium at the current low levels for the foreseeable future.

The most restrictive management measures, such as a coast-wide moratorium on weakfish harvest, would reduce the 2008 numbers weighted FN of 0.19 and landings to near zero. A coast-wide moratorium would also reduce the high total mortality (Z) levels on ages 1+ weakfish by about 25% based on the average 2004-2008 ratio of fishing (F) to total mortality (F / Z) (refer to Table C9.3-5). In a sense, the F/Z ratio is a relative measure of leverage that fishery managers can exert in order to enhance the chances of rebuilding depleted stocks. From 1981 to 1989, the F/Z ratios for weakfish were, in most years, well above 0.75, indicating the presence of relatively high leverage and thus a high probability, that management measures if implemented then, would have lead to stock rebuilding. As natural mortality (M) on ages 1+ weakfish increased after 1999, however, the F/Z ratios fell quickly to below 0.30 in most years, thereby greatly reducing the likelihood that management measures imposed after 2003 would eventually lead to a weakfish stock recovery. This relationship between the probability of stock rebuilding via management action and the F/Z ratio was recently addressed by (Uphoff 2005d) by forecasting weakfish stock abundance over the next 20 years from the logistic production model. If M was allowed to remain at the 2003 level, he reported little if any future weakfish stock rebuilding following a 50% reduction in fishing mortality (F) if the current weakfish fishing mortality rates (F) on weakfish comprised less than 40% of total mortality (Z). The problem of weakfish stock rebuilding is made even worse by the emergence of a recruitment bottleneck at age 0. Even a coast-wide moratorium to weakfish harvest would have little if any impact on the recently emergent recruitment bottleneck, where age 0 weakfish are not susceptible to direct harvest.

Like virtually all single-species stock assessments conducted along the Atlantic coast, natural mortality (M) of age 1+ weakfish was initially assumed to be constant (M = 0.25) in previous Yield-

per-Recruit and VPA model runs. Results from the index-based analysis (section **C8.0**) indicated that ages 1+ F estimates remained low and steady from 1999 to 2008 despite a systematic rise in ages 1+ total mortality (Z) after 1998 as indicated by ADAPT. These conflicting trends between Z and F strongly suggest that natural mortality (M) has recently tripled in magnitude, and was therefore the primary cause for the recent weakfish stock failure along the Atlantic coast. The management consequences of assuming a constant M when the annual M values actually rises systematically over time can be serious (Swain and Chouinard 2008). As noted in this weakfish assessment, by holding M constant, the resulting ages 1+ fishing mortality rates (F) on weakfish would have risen steadily to around 1.40 in 2007 based on the 2009 VPA run despite the recent precipitous drop in landings and tuning indices. If the constant M assumption and ensuing VPA results were accepted without qualification, we would have concluded falsely that the recent failure in weakfish productivity was due mainly to overfishing. In future assessments here and elsewhere, the assumption of constant M for ages 1+ fish needs to be critically examined. In addition, the impacts of trophic and environmental effects on exploited finfish stocks should be integrated into fisheries models and rigorously tested as a potential alternative hypothesis to the overfishing hypothesis.

C10.0'Evaluate AIC-based hypothesis testing of fishing and additional predation-competition effects using multi-index biomass dynamic models with and without prey-based, predator-based, or ratio dependent predator-prey extensions. Evaluate biomass dynamic model estimates of F, ages 1+ stock biomass, surplus production, time-varying natural mortality, and biological reference points along with uncertainty of those estimates. Advise on burden of proof necessary for acceptance of alternatives to constant M and whether these biomass dynamic techniques could complement or substitute for age-based modeling for management advice. (TOR #6)

C10.1 Introduction

Weakfish fisheries were subject to increasingly restrictive management during the early and mid-1990s (ASMFC 2002). In the late 1980s, states south of VA did not have length limits; commercial minimum length limits were 229 TL in VA and NJ, 250 mm in MD and DE, and 305 mm in NY, CT, and RI (Vaughan et al. 1991; see Figure 1 for map). Recreational fisheries were generally subject to the same length limits, although NJ and CT did not have any. States required a general permit for harvest and there were some gear and area restrictions at that time (Vaughan et al. 1991). After imposition of Amendment 3 in 1996 (a coastwide 305 mm length limit or its equivalent allowing at least 1-2 years of spawning, trawl and gill net mesh restrictions, recreational bag limits, reductions in directed commercial fishery effort, and fish excluder devices to minimize bycatch in southern shrimp fisheries; ASMFC 1996), weakfish were thought to be on the path to recovery, but by 2000-2003 recreational and commercial landings of weakfish along the Atlantic Coast approached all-time lows as population estimates derived from age structured or biomass dynamic models (De Silva 2002; Kahn 2002; Uphoff 2005b) reached all-time highs. This disconnection between weakfish stock assessment results, management actions, and fishery performance lead the WTC to consider external factors that lie outside of the purview of traditional single-species assessments.

The most reliable estimates of trends or values of F during 1981-2003 indicated it had been modest since at least 1995, while weakfish abundance and surplus production declined to low levels (Crecco 2006; Kahn et al. 2006; Uphoff 2006a). This lead to formulation of a hypothesis that the

most likely cause of declining weakfish fortune was increased natural mortality (Kahn et al. 2006). Decline of weakfish in response to increasing M was a default postulate reflecting the difficulty in finding evidence of increasing F, constant M, or recurring year-class failure. Fishing is often blamed for declines in fish stocks, but there is a preponderance of evidence that many other factors cause stock fluctuations (Deriso et al. 2008). During 2004-2006, a series of analyses (summarized below; Crecco 2006; Kahn et al. 2006; Uphoff 2006a) investigated hypotheses that weakfish population dynamics, growth, and survival were negatively affected by fishing, environmental conditions, forage abundance, competition and predation. As a result of these analyses, two strong covariates emerged: Atlantic menhaden (forage) abundance, and predation-competition from striped bass. Other candidate predator-competitors in particular were suggested by these analyses, but were less consistently identified as factors and often co-varied with striped bass (Crecco 2006; Uphoff 2006a).

Crecco (2006) and Uphoff (2006a) applied the predator-prey model of Steele and Henderson (1984) to examine relative effects of fishing and striped bass predation and competition on recent (1981-2003) weakfish dynamics. This predator-prey model was a biomass dynamic model with a sigmoidal type III predation function added to estimate additional predation losses (Collie and Spencer 1993). When applied generally, this predator-prey model reproduced rapid shifts in abundance exhibited by marine fish populations (Steele and Henderson 1984) and was useful in exploring the role of spiny dogfish *Squalus acanthias* predation on Georges Bank haddock *Melanogrammus aeglefinus* recovery and management (Spencer and Collie 1995; 1997b). Collie and DeLong (1999) applied delay-difference biomass dynamics models with additional species interaction terms (competition and Type I, II, or III predator functional responses) to evaluate multispecies interactions in the Georges Bank fish community.

Biomass dynamic models are the simplest full stock assessment method (Haddon 2001). They are relatively simple to apply because they pool the overall effects of growth, mortality, and recruitment into a single production function. Their data needs are small - an index of relative abundance, landings (both in weight), and, in the case of species interactions, biomass or indices of predator-competitors. The stock is considered as undifferentiated biomass and age, size, and sex structure are ignored (Haddon 2001). When species interactions are not considered, variation in exploitation and biomass is important for fitting the model - length of the time-series is not (Hilborn and Walters 1992). In some situations, biomass dynamic models do not perform well (National Research Council 1998) and fisheries scientists usually prefer age-structured models (Hilborn and Walters 1992). However, biomass dynamic models may provide as good or better estimates of management parameters at a fraction of the cost (Hilborn and Walters 1992; Walters and Kitchell 2001).

The WTC has pursued biomass dynamic modeling since the late 1990s (Uphoff 2005a) as a contrasting approach to age-structured models of weakfish dynamics. Three potential problems impacting age structured assessment approaches for weakfish have been identified by the WTC: (1) estimates of F and stock were based on scale ages up through the mid-1990s and otolith ages afterwards (these aging methods may not assign the same ages), (2) age structure was not sampled well in all regions over many years (states from NJ and north had ages assigned from states south even though larger weakfish were more common and higher size limits were in place), and (3) retrospective bias of most recent year estimates from ADAPT and ASAP has been severe (NEFSC 1998b; NEFSC 2000; Kahn et al. 2006; section **C7.0** of this report), leading to questions about the ability of these techniques to supply current management advice (Walters and Martell 2004). Hilborn and Walters (1992) suggested a pragmatic approach of using both age structured and surplus production modeling when data are available because they are fundamentally different approaches to

answer many of the same stock assessment questions. If both methods are applied and give different answers, then assessment scientists should try to understand why the answers are different and analyze their management implications (Hilborn and Walters 1992).

Biomass dynamic models could also be easily modified with predator-prey terms for hypothesis testing. In Crecco (2006) and Uphoff (2006a), estimates of M and F from the Steele-Henderson model were used for investigating whether F alone or additional non-equilibrium changes in M associated with predation-competition could be influencing weakfish. Crecco (2006) largely explored biomass dynamics using a Gompertz (asymmetric) production function, while Uphoff (2006a) employed a logistic-based (symmetric) version.

In this approach to assessing 1981-2006 weakfish biomass dynamics, we have evaluated the relative merits of multiple hypotheses examining whether fishing alone or fishing and interactions with striped bass influenced biomass dynamics of weakfish.   Included in this analysis is an evaluation of the two types of production functions used by Crecco (2006) and Uphoff (2006a). In addition to fishing only models, Type I, II, and III prey-dependent predation functions (Ginzburg and Akçakaya 1992) were considered as were two additional depensatory mortality functions. Detection of moderate to severe depensatory mortality is important because stocks that are subject to it often undergo sudden and persistent drops in surplus production and shifts into domains of population behavior that are unresponsive to management (Spencer 1997; Walters and Kitchell 2001). A decline toward extinction may even be possible (Hilborn and Walters 1992).

Platt (1964) offered that strong inference and rapid progress in science can be derived from (1) devising alternative hypotheses; (2) devising crucial experiments with alternative, exclusive outcomes (as much as possible); (3) carrying out the experiments to get as clean a result as possible; and (4) making sequential hypotheses from possibilities that remain. Burnham and Anderson (2001) described the use of Kullback-Leibler information as a basis for making valid inference from analyses of empirical ecological data and to objectively answer "What model should be used to approximate reality given the data at hand?" Three general principles guide model-based inference of ecological data: simplicity and parsimony, multiple working hypotheses, and strength of evidence (Burnham and Anderson 2001). As part of a new paradigm for stock assessment in the 21st century, Sainsbury (1998) advocated formulating multiple hypotheses about stock status and evaluating them with empirical data. Deriso et al. (2008) presented a framework for evaluating the cause of fishery declines by incorporating covariates into assessment models to evaluate fishing effects versus other natural or human impacts. This framework has several components: identifying factors potentially affecting the stock, developing a stock assessment model that incorporates them, fitting the model, testing hypotheses, and evaluating impact of the factors (Deriso et al. 2008). We have applied this outline to structure this investigation of weakfish population dynamics during 1981-2006. Identification of factors was largely carried out in the previous assessment (Crecco 2006; Kahn et al. 2006; Uphoff 2006a) and the remaining components related to model development and hypothesis testing are the focus of this assessment of 1981-2006 weakfish dynamics.

The following analysis investigates both direct and indirect trophic interactions between weakfish and potential predators and/or competitors. Hereafter it will be referred to as the "forage hypothesis" to distinguish it from the more direct "predation hypothesis" presented in section **C9.0**.

C10.2 Summary of Previous Analyses of External Factors Affecting Weakfish

C10.2.1 Predator-competitors and climatic variables

Crecco (2006) and Uphoff (2006a) used diet literature review, statistical analyses, and empirical models to examine major processes (predation, competition, environmental forcing, and forage availability) affecting weakfish biomass dynamics. This empirical approach provided a bridge between the common assumption that predation and competition impose a constant rate of mortality that allows the stock to be assessed in isolation from its environment (single species assessment; Bax 1998; Sainsbury 1998) and more complex process-based multispecies models (Whipple et al. 2000).

Crecco (2006) and Uphoff (2006a) explored whether shifts in weakfish landings, growth, natural mortality, and distribution or their proxies coincided with climatic factors, forage or predator-competitor abundance or biomass using correlation and regression analyses, inferring competition for forage if test results were logical and statistically significant. Potential predator-competitors (striped bass, summer flounder *Paralichthys dentatus* , bluefish *Pomatomus saltatrix*, spiny dogfish *Squalus acanthias*, and Atlantic croaker *Micropogonias undulatus*), major forage species for piscivores along the mid-Atlantic (bay anchovy *Anchoa mitchilli*, Atlantic menhaden, and spot *Leiostomus xanthurus*), and major climatic features (North Atlantic Oscillation or NAO, and water temperature) were evaluated as candidates for inclusion in assessment models. Analytical strategies of Crecco (2006) and Uphoff (2006a) consistently and strongly indicated striped bass was a major predator – competitor of weakfish.

Crecco (2006) used three stepwise multiple regression analyses ($P < 0.05$) to test linkages of biotic and abiotic factors to changes in weakfish surplus production (SP), M on ages 1+, and juvenile M. Explanatory variables in these analyses included F, abundance or relative abundance of candidate finfish predator-competitors, average annual surface water temperature, and deviations in the winter NAO (both lagged 1 or 2 years to coincide with weakfish recruitment to the adult stock). Coast-wide biomass of spiny dogfish, and coast-wide striped bass, summer flounder, and bluefish stock abundance estimates from 1982 to 2004 were taken from stock assessments. In the case of the Atlantic croaker, assessment estimates were available from 1982 to 2002, so 1982-2004 total recreational catch of croaker (MRFSS A, B1, B2 estimates) was used as a coast-wide index (1982-2002 recreational catches were highly correlated with abundance estimates; Crecco 2006).

Stepwise regressions selected striped bass abundance as the only ($P < 0.001$) explanatory variable inversely related to SP and juvenile M, while summer flounder biomass was inversely related to M of ages 1+ (Crecco 2006). Correlation analyses indicated that SP, M on ages 1+, and juvenile M were also inversely associated ($P < 0.05$) with striped bass abundance estimated from tagging (SP, M on ages 1+, and juvenile M) and to Atlantic croaker relative abundance (SP and juvenile M). Due to substantial co-linearity among striped bass, summer flounder, and Atlantic croaker indicators, stepwise models were unable to select second or third significant variables (Crecco 2006).

Uphoff (2006a) applied three approaches to evaluate the potential of striped bass, Atlantic croaker, bluefish, summer flounder, and spiny dogfish to influence weakfish biomass dynamics. First, long-term trends (1950-2003) in Atlantic Coast commercial landings (excluding spiny dogfish) were correlated with weakfish landings to determine if significant ($P \leq 0.10$) negative associations existed. Second, biomass estimates from single species assessments (generally 1981-2003) were used in a species-specific predator-weakfish model (logistic biomass dynamic model with a Type III predation function). Deterministic runs were made and model fit, its ability to produce a decline in weakfish biomass after the late 1990s, and how closely its levels of weakfish biomass were to those

estimated using the rescaled relative F technique (landings divided by average indices scaled into VPA numbers or biomass weighted F; Kahn et al. 2006) were used to evaluate the candidates. Finally, literature on candidate species' diets was reviewed to see if predation on weakfish and competition for bay anchovy, Atlantic menhaden, and spot were possible (Uphoff 2006a). Uphoff (2006a) developed a "scorecard" for each candidate predator-competitor based on these three components. Ten criteria were considered - four based on statistical or model results, and six on surveyed diet literature. The scorecard indicated that striped bass was by far the most likely predator-competitor candidate (9.5 out of 10 possible), followed by summer flounder (6.2), bluefish (6.1), spiny dogfish (5.2), and Atlantic croaker (2.6).

Uphoff (2006a) conducted additional explorations of the role of forage and striped bass on long-term commercial landings and the DE PSD Q+ index (a length-frequency index; discontinuous 1966-2003 time-series; Appendix C-2) with correlation, categorical or stepwise regression analyses (Neter and Wasserman 1974; Rose et al. 1986; Freund and Littel 2000). The DE PSD Q+ index was considered an indicator of long-term weakfish dynamics because of its significant and positive associations with recreational trophy citations, commercial and recreational yield, MRFSS catch per trip, and inshore-offshore recreational harvest distribution (Uphoff 2004; Kahn et al. 2006; Appendix C-2). Forage abundance indices and a long-term indicator of striped bass biomass (LBI, a Chesapeake Bay egg presence-absence index of spawning biomass; Uphoff 1997) were independent variables. Indices of Atlantic menhaden, spot, and bay anchovy relative abundance from NC, VA, MD, DE, and NJ, age 0 abundance estimates of Atlantic menhaden along the Atlantic Coast, and LBI were used in correlation and stepwise regression analyses with DE PSD Q+ (Uphoff 2006a).

A negative relationship of striped bass and weakfish landings that reflected high or low periods of underlying weakfish productivity was described by categorical regression of period and striped bass landings ($P < 0.001$; see Figures 18 and 19 in Uphoff 2006a). This approach predicted high (1973-1989) and low (1950-1972 and 1989+) periods of weakfish landings as abrupt time-blocks. Stepwise regressions described a negative relationship of weakfish landings with LBI and a positive relationship with Atlantic menhaden abundance or $\log_e$-transformed juvenile menhaden indices during 1955-2002 ($P < 0.0001$). In general, menhaden accounted for most variation in weakfish landings and striped bass accounted for a lesser amount in these stepwise regressions and reproduced the time-blocks of weakfish landings described by the categorical regression of striped bass landings and productivity period (Uphoff 2006a).

Significant ($P < 0.10$) positive associations of weakfish size quality (1966-2003 DE PSD Q+ index) and forage abundance occurred consistently (but not among all surveys) in MD and VA for all three forage species and sporadically in NJ and NC (Uphoff 2006a). Stepwise regressions of striped bass biomass and juvenile menhaden abundance against DE PSD Q+ indicated a significant positive influence of forage and negative influence of LBI (Uphoff 2006a).

Changes in weakfish landings and size distribution were logically consistent with trends in menhaden relative abundance and striped bass biomass over a broad geographic area and time-span. These changes, coupled with little evidence of a rise in F, supported the development of hypotheses linking the aborted weakfish recovery to striped bass and menhaden.

C10.3 Background for Weakfish, Striped Bass, and Atlantic Menhaden Interactions

C10.3.1 Spatial and Temporal Distributions

Weakfish occur along the Atlantic coast from FL to MA, straying as far north as Nova Scotia (Mercer 1985; Figure C10.3-1). They are most abundant from NC to NY. Adult weakfish disperse

from continental shelf wintering grounds located from lower Chesapeake Bay to Cape Lookout, NC, as waters warm in spring and initiate an inshore, northerly migration (Mercer 1985; ASMFC 2002). They enter sounds, bays, and estuaries in early spring and may stay through summer, although most in northern states return to the ocean (Mercer 1985). In fall, a general southerly movement back to overwintering grounds occurs (Mercer 1985).

Striped bass are found along the entire Atlantic coast from Canada to FL, but are most abundant between ME and NC. Migratory stocks of striped bass are mainly composed of fish spawned in Hudson River (NY) or Chesapeake Bay, but Chesapeake Bay fish predominate and are associated with most of the production along the Atlantic coast (Boreman and Austin 1985; Dorazio et al. 1994; Richards and Rago 1999). After spawning in tidal-fresh Chesapeake Bay rivers during April-May, most immature and mature striped bass females and some males migrate to spend summer in coastal waters of the mid-Atlantic (VA-NY) and New England (CT-ME; Dorazio et al. 1994). Immature females largely leave the Chesapeake Bay after ages 2-3 and reside in coastal waters until they mature. Remaining, mostly male, striped bass reside within estuarine waters of Chesapeake Bay throughout summer-fall (Dorazio et al. 1994; Secor and Piccoli 2007). In fall, southward movement of coastal fish begins (Dorazio et al. 1994). During November-March, some striped bass from Chesapeake Bay and the coastal contingent overwinter in the inshore zone between Cape Henry, VA, and Cape Lookout, NC (Setzler et al. 1980).

Atlantic menhaden are distributed from FL to ME during summer, with older, larger individuals found further northward and smaller, younger fish in the southern half of the range (Ahrenholz 1991). In September, the northerly portion of the population begins to migrate southwards. By December, these fish are in waters off the NC coast. These fish are followed by large numbers of young-of-year menhaden (Ahrenholz 1991). Although described as disappearing and dispersing from inshore waters by late January (Ahrenholz 1991), menhaden were dominant in striped bass diets in nearshore waters from the VA Capes to the Outer Banks of NC during December-March, 2000-2007 (Overton et al. 2008). During March-early April, schools move rapidly northward and have redistributed by June (Ahrenholz 1991). This general pattern of movement is closely approximated by weakfish (ASMFC 2004) and striped bass (Walter et al. 2003).

C10.3.2 Trophic Ecology of Weakfish and Striped Bass

Weakfish feed throughout the water column (Wilk 1979; Mercer 1985). Young weakfish feed primarily on mysid shrimp and anchovies, while older weakfish feed on the clupeid species that are abundant in a given area. Cannibalism has been reported. Functional morphology indicates that weakfish are upper midwater feeders (Wilk 1979; Mercer 1985).

Striped bass evolved as a schooling species to take advantage of the great energy resource of clupeids along the Atlantic Coast of North America (Stevens 1979). Striped bass actively select for Atlantic menhaden, but will feed on other species when menhaden are not sufficiently abundant (Overton 2003; ASMFC 2004; Rudershausen et al. 2005).

In the mid-Atlantic region, bay anchovy represented the prey most consumed by age 0 weakfish and age 1 striped bass as they initiated piscivory, but both switched mostly to similar, larger prey (clupeids, primarily Atlantic menhaden) within a year (Mercer 1985; Taylor 1987; Hartman and Brandt 1995). Juvenile menhaden and spot would be most relevant to weakfish and younger, smaller striped bass making the transition through piscivory because of size selectivity and gape limitations (Stein et al. 1988; Juanes 1994).

Weakfish expanded the size of items in their diet least of 18 species examined by Scharf et

al. (2000) as they grew. They largely remain dependent on items in the size range of bay anchovy, and juvenile menhaden or spot; maximum fish prey size of an 800 mm weakfish was about 180 mm (Scharf et al. 2000). Walter and Austin (2003) and Overton et al. (2008) found that large striped bass fed on small pelagic prey (bay anchovy and juvenile clupeids) as well as large prey items. The upper 99% quantile of prey fish total length that can be eaten by 800 mm striped bass is approximately 280 mm and a 1,200 mm striped bass can eat a 415 mm fish (F. Juanes, University of Massachusetts, personal communication). Minimum prey size is below 50 mm for the entire size ranges of weakfish and striped bass.

Weakfish have been specifically identified as a minor food item in some, but not all, striped bass diet studies, particularly in the mid-Atlantic region (Manooch 1973; Wilk 1979; Mercer 1985; Walter and Austin 2003; Overton et al. 2008). Weakfish were encountered in striped bass diets during fall or winter (a period of heavy pelagic feeding) in the mid-Atlantic region and accounted for no more than 1-3% of striped bass diet by weight or volume (Manooch 1973: Walter and Austin 2003; Overton et al. 2008). These overwintering striped bass feed mostly on Atlantic menhaden and bay anchovy (Manooch 1973; Austin and Walter 2003; Overton et al. 2008).

Early switching of weakfish and striped bass to a fish diet indicates that both are specialized piscivores (Persson and Brönmark 2002). Early switching requires high growth rate, which implies high densities of proper forage and safe foraging opportunities. If unfavorable conditions prevail, growth is reduced, and vulnerability to competitors and predators is increased. Population densities of specialists are generally a result of their success at a feeding stage where they experience intense competition (Persson and Brönmark 2002). Species undergoing ontogenetic diet shifts face a risk of delayed transitions among feeding stages if food resources are limited and competition is intense (Persson and Brönmark 2002). Competing individuals and predators may hinder one another's feeding activities, leading to starvation or they may eat one another (including cannibalism; Yodzis 1994). Ontogenetic bottlenecks are common in piscivorous fishes because competition may retard growth and prevent size advantage necessary to feed on the larger prey in the next stage (Persson and Brönmark 2002). Individuals not reaching size advantage over prey may become stunted at size where consumption balances metabolic requirements (Bax 1998; Persson and Brönmark 2002).

C10.4 Current Stock Assessment

C10.4.1 Harvest and Discard Estimates

Biomass dynamic models employed in this analysis used total weight of aggregated harvest and discards by both the commercial and recreational fisheries. Commercial landings and MRFSS recreational harvest estimates were obtained from the National Marine Fisheries Service (www.st.nmfs.gov/st1/).

De Silva (2004) provided the first quantitative analysis of weakfish discards based on ratios of discards to harvest in 1994-2003 NMFS at sea observer data. The NMFS database contained information on weakfish discards, target species catches, estimates of landings of species kept on the haul, and length data of both discards and fish kept. Only hauls that were completely sampled for discards were evaluated. During 1994-2003, discard data were primarily available for gillnets (287 trips and 615 hauls; 89% landed in VA, NC or NJ) and otter trawls (196 trips and 519 hauls; 41% landed in MD and 49% in NJ, NY, or RI.). The annual number of gillnet trips where weakfish discards were observed ranged from 5 to 68. Annual number of otter trawl trips where weakfish discards were observed ranged from 4 to 45 (De Silva 2004).

The WTC adopted De Silva's (2004) technique for estimates of weakfish discards using

annual (1994-2003) discard ratios (weakfish to target species) and NMFS commercial landings of a suite of target species for the previous assessment (Kahn et al. 2006). The ratio of at sea observations of weight of discarded weakfish to weight of harvested target species by gear (trawl and gill net) was scaled up to total discard estimates based on total landings of the target species (De Silva 2004). Discards were subdivided into regulatory and market discards. Identification of target species was only based on complete trips and hauls where weakfish discards were observed (De Silva 2004). The WTC assumed that gear-specific relationships among the target species in the data set reflected the entire fishery.

For the current assessment, the WTC considered all estimated commercial discards to have died and chose to update de Silva's (2004) method for estimating of commercial discards (section **C5.0** of this report). Haul level data for the same gear and target species combinations were used to evaluate annual, multiyear (5 or 7 year blocks), and all-year (1994 to 2007) weakfish discard ratios. Because of concerns with high interannual variability and uncomfortably large standard errors of estimates based on short time groupings, the WTC elected to use the all-year ratio estimates. Ratio-based estimates of weakfish discard weight were made for butterfish, long-fin squid, summer flounder and weakfish fisheries for trawl harvests, and Atlantic croaker, bluefish, spiny dogfish, and weakfish gillnet harvests.

Discard sampling was not conducted until 1994 and market-related discard ratios estimated for 1994-2002 were used to estimate weakfish discards in prior years (De Silva 2004). In 1993, New Jersey and Delaware enacted weakfish regulations that may have lead to non-market discarding and the total discard ratio was used for these states. For all other states, discard estimates for 1993 were based on the non-regulatory discard ratios. Discard estimates after 1993 used total discard ratios to estimate weakfish discards for all states.

Recreational discard losses were calculated as the product of estimates of MRFSS number discarded, mean weight, and discard mortality rate. The MRFSS does not estimate weight of released weakfish and we used mean weight of weakfish recreational discards (0.15 kg) estimated from MRFSS 2004-2007 headboat survey length-frequencies (D. Kahn, DE Fish and Wildlife, personal communication). This constant mean weight approximated the mode of six annual regional (VA and north or NC and south) mean discard weight estimates for headboat catches (range = 0.14-0.17 kg, three ≈ 0.15 kg). Weakfish hook-and-release experiments produced dichotomous mean mortality estimates either near 3% or 15% (Table C10.4-1) and 10% release mortality was adopted by the WTC.

C10.4.2 Biomass Indices

Exploitable biomass was defined as biomass of weakfish 250 mm and larger. Weakfish of this size and age (1+) and larger were heavily exploited in the past (305 mm length limit or its equivalent was not mandatory until 1994; ASMFC 2002) and represented spawning biomass (Nye et al. 2008). Size limits and gear restrictions imposed since Amendment 2 (1994) have greatly lessened vulnerability of weakfish less than 305 mm to directed harvest.

C10.4.2.1 1981-2003 Assessment's Exploitable Biomass Indices

Crecco (2006) and Uphoff (2006a) evaluated fishery and predator-weakfish biomass dynamics primarily with recreational catch (harvested and released) per effort index of relative biomass (WRI). DE and NJ trawl survey exploitable biomass indices (Kahn et al. 2006) provided additional indicators of weakfish exploitable stock biomass. These trawl surveys and WRI were positively correlated ($r \geq 0.71$; $P \leq 0.003$), but trawl survey indices were not used extensively in the

assessment (Crecco 2006; Kahn et al. 2006; Uphoff 2006a). A third trawl survey index (Northeast Fisheries Science Center fall survey) was rejected because exploitable-sized weakfish were poorly represented, it did not correlate consistently with other indices, cohort catch curves produced positive "mortality" rates, and the 40th SARC expressed concerns about high inter-annual variability (WTC 2006; Kahn et al. 2006; Uphoff 2006a).

C10.4.2.2 1981-2006 Indices of Exploitable Biomass

Three exploitable biomass indices (EBI) were used: one fishery-dependent and two fishery-independent. The fishery-dependent index (WRI) was estimated as weakfish recreational private/rental harvest and release weight per trip in all areas of the mid-Atlantic (VA-NY; Crecco 2009). A MRFSS directed trip index of weakfish catch (harvest and releases) per trip (Brust 2004) was very closely correlated (Pearson r = 0.96, P < 0.0001) with the more general WRI (Crecco 2005a), so it was not considered further in analyses.

The WRI represented the only EBI available for the entire 1981-2006 time series. Harvest weight estimates were obtained directly from the MRFSS. Weight of recreational releases was estimated from numbers released (MRFSS type B2) multiplied by average weight of released weakfish from the coastwide head-boat fishery (assumed constant at 0.15 kg; see section **C10.4**). Effort equaled MRFSS annual estimates of all trips for the private/rental boat sector. The WRI used previously by Crecco (2006) and Uphoff (2006a) assigned harvest mean weights to released weakfish and these weights were much higher than indicated by headboat releases (3-15 times higher; median ≈ 5-times).

The private boat fishery is highly mobile and capable of catching weakfish of all sizes. Weakfish 250 mm TL and larger have constituted the majority of weakfish measured by the MRFSS. Weakfish less than 250 mm and as large as 890 mm were consistently present in historic MRFSS length-frequencies (1979-1998; M. Gibson, RI Fish and Wildlife personal communication; these years are not currently available at the MRFSS website); smaller fish are still present in 2004-2008 length-frequencies available online (www.st.nmfs.gov/st1/), but larger weakfish were consistently present out to about 580 mm in 2008.

Summed mid-Atlantic catches (1981-2006) indicated that nearly all weakfish (numbers) were caught during three MRFSS waves: May-June (24%), July-August (42%), and September-October (33%). Summed mid-Atlantic private boat trips were distributed similarly: May-June (27%), July-August (40%), and September-October (22%). The Mid-Atlantic subregion of the MRFSS represented the major (38-76%, median = 62%) component of annual coastwide recreational weakfish catches.

A general recommendation for data in stock assessment is that information only be used once (Cotter et al. 2004). In the case of the WRI and this assessment, the same information is contained in both the landings and the index. However, division by effort in the index reduces the direct dependency in the data (V. Crecco, CT DEP, personal communication).

Delaware and New Jersey trawl survey EBI were used as additional indicators of weakfish stock biomass. Essential data for calculating EBI for all years included annual biomass (kg) or an estimate of central tendency of abundance of all weakfish caught per tow, annual length-frequencies (cm length groups), and an equation that predicted weight (kg) at length (cm). Each agency provided catch-per-effort data and length-frequencies for the available time-series. We assumed all length-frequencies represented random samples of lengths. Annual length-frequency was converted to weight-frequency by multiplying each cm-group catch by predicted mean weight (kg) of that cm-group.

Predicted weight-at-length was derived from annual regressions of $\log_e$-transformed weight (kg) on $\log_e$-transformed length (cm). Individual lengths and weights were available from the NEFSC fall trawl survey annually for 1992-2006 (Uphoff 2005a; G. Shepherd, NMFS, personal communication). We used a weight-length regression (converted from pounds and inches to kg and cm) to estimate weight-at-length during 1989-1991 based on combined data from NC during 1982-1983 and 1988-1989, NY during 1988-1989, and coastwide data collected by VIMS (Virginia Institute of Marine Science) during 1988 (D. Vaughan, NMFS, personal communication).

A DE EBI was derived for each year as $B_{DE} * (B_E / B_A)$; where B_{DE} is the kg per nautical mile estimated for all weakfish by DE; B_E equaled the summed weight of weakfish ≥ 25 cm created by multiplying cm length-group specific catch per tow by predicted weight; and B_A equaled the summed weight for all weakfish derived by the same procedure described for B_E.

DE data was available for 1966-1971, 1979-1984, and 1990-2006. We excluded years prior to 1990 because (1) recreational catches were only available since 1981, (2) the time series was discontinuous, (3) survey vessels had changed between 1984 and 1990, and (4) 1981-2003 DE EBI were poorly correlated with WRI, but 1990-2003 were significantly and positively correlated with WRI and NJ EBI (Uphoff 2006a).

Mean kg per nautical mile estimates were provided by DE for all weakfish captured. Twenty minute tows with a 9.1-m headrope trawl were made at nine fixed stations located along the eastern side of Delaware Bay each month during March-December (Michels and Greco 2004). May-October catches during 1990-2007 comprised 98% of weakfish biomass captured. May accounted for approximately 55% of biomass; June and July, 10-12%' and August-October 5-8%; CVs of annual estimates of biomass per nautical mile ranged from 16-54% with a median of 22%. A single CV (1994) was in excess of 40%, while 19 of 24 were below 25% (S. Michels, DEDFW, personal communication).

NJ has conducted a stratified random survey along its coast during 1988-2006 (catch data only were available for 1988) and provided a complete file of catch and length-frequency data. February, April, and June data were pared from estimates because of very high frequencies of zero catches (Uphoff 2005a). Indices and weight-at-length distributions were calculated from August and October rounds, the convention used for NJ age-structured indices in early versions of ADAPT (NEFSC 2000).

Uphoff (2005a) calculated three estimators of central tendency and their standard deviations for 1989-2003 NJ data: arithmetic mean (NJAM), the proportion of positive tows (NJPT), and mean $\log_e$-transformed catches + 1 (NJLN); NJLN was converted to a geometric mean (NJGM) to express central tendency. These estimates of central tendency were based on catches of all sizes of weakfish.

Precision of NJPT and NJLN during 1989-2003 was comparable (CV = 7.7-29.1%) and trends in survey time-series were similar when NJPT and NJLN were used as indicators of central tendency (Uphoff 2005a). NJAM was less precise (CV = 22.9-60.9%) and indicated different trends in abundance. Uphoff (2005a) chose NJGM as the estimator of central tendency for the NJ trawl survey because its precision was comparable or better than NJPT and it indicated the same trend in relative abundance as NJPT.

An annual NJ length-frequency was converted to weight-at-length distribution (described above). A mean weight of all weakfish sampled (MWT) was estimated by dividing the summed weight frequency by the summed numeric frequency. The exploitable fraction (B_E / B_A) of the sampled biomass was estimated as the summed weight of all weakfish ≥ 25 cm divided by the summed weight of all weakfish. An index of exploitable biomass (NJ EB) was then derived for each year as NJ EB = NJGM * MWT* (B_E / B_A).

The three EBI were different in scale and we used z-transformations to place them on the same scale to view relative trends and added two to each value to positively rescale them. Z-transformations were based upon means and standard deviations during 1990-2006 - years in common among all three surveys. This approach removed the need for multiple index scalars since all were in common scale.

Once these Z+2 EB indices time-series were plotted together, two general trends emerged (Figure C10.4-1). There was (1) general agreement between WRI Z+2 and DE Z+2 indices, and (2) NJ Z+2 (GM based) exhibited considerably more inter-annual variation and disagreed substantially with the others during 1998-1999 (much lower) and 2004-2005 (much higher; Figure C10.4-1). Substitution of NJ PT for NJ GM in the NJ EBI was a slight improvement at best.

This disagreement among indices led us to re-examine whether all three EBI were suitable indicators of stock status and, if not, could they be calculated from a different estimator of central tendency to conform to accepted indicators. This search for conformity implied a single stock of weakfish, even though evidence exists for multiple stocks (Munyandorero 2006). This assumption of a single stock was consistent with how assessments of weakfish have been conducted in the past and how the species has been managed (NEFSC 1998a; NEFSC 2000; ASMFC 2002; Kahn et al. 2006).

This re-examination considered four concepts for suitability: precision, accuracy, consistency among indices, and coherency (inter-annual variation of an index). Coefficient of variation (CV) was used to evaluate precision of trawl surveys and PSEs (proportional standard error; standard error of an estimate as a percentage of the estimate; Personal communication from the National Marine Fisheries Service, Fisheries Statistics Division) were used for MRFSS components. A CV (or PSE) of 40% was considered a threshold for rejection and CVs of 25% or less was desired. Estimates of CV were based on weight or count-based estimators or proportions of samples with weakfish of all sizes. Precision of WRI could not be determined directly, but PSE of harvest weight, catch, and effort were available from the MRFSS survey. Accuracy criteria were to reflect the true status of the population which, of course, was not readily answered. We used trends in 1981-2006 fishery losses (harvest and discard weight in commercial and recreational fisheries) as one indicator and biomass estimated from the converged portion (years exhibiting minimal or no retrospective bias - 982-2000) of the "best" VPA run (20) from Kahn et al. (2006) as another. Consistency was indicated by how well indices correlated among themselves. Coherency was indicated by correlation with a grand mean of Z+2 transformed WRI, DE EBI and NJ EBI. We considered the grand mean a form of smoothing. Correlation coefficients and P (< 0.05) were considered in evaluations of accuracy and consistency. Evaluation of coherency involved correlations with a grand mean of the indicators which rendered P meaningless. Coherency was indicated by similar, high correlations among indices. As a first step, the four available EBI (WRI, DE, NJGM, and NJPT) were evaluated with the four criteria.

Precision of the components of the WRI were within the desirable range. Proportional standard errors (PSE) of trips ranged from 2.6-6.7%, harvest weight PSEs ranged from 8.8-21.8%, and catch PSEs fell between 5.2% and 19.8%. Precision of NJ estimators of central tendency was largely described previously. The NJAM exceeded the 40% CV threshold and was generally above the desirable criterion (CV > 25%). Neither NJLN nor NJPT estimators had CVs above the threshold, but both had values outside the desired range. All but one DE trawl survey estimate of mean kg per nautical mile during 1990-2006 had a CV < 40% and 5 of the 17 remaining estimates had CVs between 25% and 40% (S Michels, DEDFW, personal communication).

In general, WRI and DE met the remaining criteria, while NJ EB indices performed poorly. None of the four EBI correlated at P < 0.05 with the converged portion of the VPA, but DE EBI and

NJ GM EBI correlated at P < 0.1, and WRI correlated at P < 0.2 (Table 2). WRI and DE EBI were significantly correlated with Atlantic coast fishery removals, but neither of the NJ EBIs were. The WRI and DE EBI were significantly correlated with each other and each was significantly correlated with one of the NJ EBI; the NJ indices were significantly correlated with one another. WRI and DE EBI ($\rho = 0.82$-0.95) were more strongly associated with both sets of grand means than either NJ EBI (NJGM, $\rho = 0.70$ and NJPT, $\rho = 0.49$; Table C10.4-2). We concluded that WRI and DE EBI were suitable indicators of weakfish biomass, but that other estimators should be considered for NJ.

We began by looking at trends in each month (August and October) used in the NJ EBI. We concentrated on the proportion of positive tows (PT) as an indicator of central tendency because its precision should not have been as sensitive to reductions in sample size (from N = 71-79 for two months to N = 34-41). I calculated the proportion of positive tows and its 95% confidence interval by using the normal distribution to approximate the binomial probability distribution (Ott 1977). Proportion of zero catches and its inverse, NJPT in this case, tend to be robust to errors and biases in sampling when zeros are frequently encountered (Green 1979; Mangel and Smith 1990). Presence-absence indices have been more robust indicators of yellowtail snapper and age 0 white sturgeon abundance (Bannerot and Austin 1983; Counihan et al. 1999), California sardine (Mangel and Smith 1990) and upper Chesapeake Bay striped bass egg abundance (a spawning stock biomass indicator; Uphoff 1997), and longfin squid fishery performance (Lange 1991) than catch per effort. Marine surveys often contain substantial proportions of zeros and log-normal distributions of positive catches that cause high variability and low precision of sample means (Pennington 1983; 1996). The use of more efficient estimators is one way of increasing survey precision (Pennington 1983; 1996; McConnaughey and Conquest 1993).

CVs of NJ PT for either month were generally within the threshold (CV < 40%), with one exception (August 2003). All October NJ PT CVs were less than 25%, while two August NJ PT CVs were greater than 25%, but less than 40%. CVs were usually higher in August (14 of 18 were greater than October), but confidence intervals of August or October NJPT were significantly different from 0 at P = 0.05 in all years.

We explored two options for calculating EBI from NJ August or October PT. The first option was EBI = PT * EWT and the second was EBI = PT * E * EWT, where E is the exploitable fraction (number of weakfish $\leq$ 250 mm / total number) for the August or October sampling round and EWT is mean weight of exploitable weakfish in either round. Criteria for evaluating EBI indices described previously were used to evaluate the four potential indices.

August round PT * EWT (Figure C10.4-2) met all criteria. It was significantly correlated with WRI, DE EBI, total fishery losses, and the converged portion of the VPA at P < 0.05 (Table C10.4-3). Correlations of this index with the grand mean of WRI, DE, and this NJ EBI (August round PT * EWT; all indices z-transformed +2; z-transformation based on 1990-2006) were similar ($\rho = 0.89$, 0.90, and 0.86, respectively). Other indices only met a single criterion each and were not as strongly correlated with their respective grand means (Table C10.4-3). We selected August round PT * EWT as NJ EBI and it is denoted as NJA EBI.

C10.4.3 *Striped Bass and Atlantic Menhaden Biomass*

Versions of predator-prey models required estimates of striped bass and Atlantic menhaden biomass. The annual sum of SCAM (Statistical Catch at Age Model) biomass estimates for 2 year-old and older striped bass were used as predator-competitor biomass during 1982-2006 (NEFSC 2008). Striped bass of this size would also be important predators of menhaden and would be capable of consuming weakfish (Hartman and Brandt 1995; Walter et al. 2003; Uphoff 2003; Uphoff

2006a; Overton et al. 2008). Atlantic menhaden biomass estimates (ages 1+ during 1981-2005) from a forward projection model were used (ASMFC 2006b).

Neither the striped bass nor menhaden assessments provided time-series that matched the weakfish time-series; 1981 was missing for striped bass and 2006 was missing for Atlantic menhaden. We applied two techniques to fill in these missing estimates. The approaches used to make these estimates will be described briefly.

We developed a known-biomass production model (MacCall 2002) for ages 2+ striped bass using SCAM estimates of biomass, landings, and discard biomass (assumed to be 10% of landings) to estimate biomass. We used SCAM biomass estimates for 1982-2006 (G. Nelson, MA DMF, personal communication; NEFSC 2008), and recreational and commercial landings for 1981-2006 to generate an estimate of biomass in 1981. Parameters r, K, and the model scalar (see modeling section) based on 1982-2006, combined with estimates of fishery losses would allow for an estimate of biomass in 1981. Biomass estimates from the known-biomass production model had a modest bias (10%) and we adjusted output downwards by this amount. Biomass of striped bass in 1981 was estimated to be 8,789 mt (Figure C10.4-3).

We used categorical regression (Neter and Wasserman 1974; Rose et al. 1986) to estimate 2006 biomass of ages 1+ menhaden (from the forward projection model; ASMFC 2006b) from landings. We constructed annual age 1+ biomass estimates by multiplying abundance at age by mean weight derived from length-at-age and weight-at-length equations in ASMFC (2006b). A categorical variable was used to split the 1955-2005 time-series into periods where bait landings were not estimated (reduction landings only during 1955-1984; coded 0; ASMFC 2006b) and the remaining period where bait estimates were added to reduction landings (coded 1). This model provided a significant fit ($R^2 = 0.70$, $P < 0.0001$; Figure C10.4-4) and serial patterning of residuals was not indicated. All parameters were significant at $P < 0.04$ and the relationship was described by the equation: $B_t = (L_t * 2.484) + (217,798 * C) - 221,922$; where B_t = biomass in year t in MT, L_t = landings in MT, and C = bait landings category. Biomass of ages 1+ Atlantic menhaden in 2006 was predicted to be 449,784 MT (Figure C10.4-4).

C10.4.4 Weakfish Biomass Dynamic Models

Biomass dynamic models provided a stock assessment framework that allowed the incorporation of covariates for hypothesis testing. Both standard and predator-prey production models evaluated the effect of fishing, while the latter allowed for estimation of predation-competition impacts as well. The small size (250 mm) that defined exploitable biomass of weakfish allowed for use of biomass dynamic models without lags necessary in some delay-difference predator-prey models of other species (Collie and DeLong 1999; Overholtz et al. 2008).

Harvest estimates are summarized in Table C10.4-4; indices, striped bass biomass, Atlantic menhaden biomass, and the ratio of menhaden to striped bass biomass are summarized in Table C10.4-5.

WRI (1981-2006), DE EBI (1990-2006), and NJA EBI (1989-2006) were used as indices of relative abundance. Each set of indices were Z-transformed based on means and standard deviations from a common time period (1990-2006) and 2 was added to remove negative values (designated as ZEBI). This transformation allowed for a single index scalar (a nuisance parameter). Landings during 1981-2006 included both directed harvest and discards by the recreational and commercial fisheries. Biomass estimates for 2 year-old and older striped bass and ages 1+ Atlantic menhaden were used.

Two types of production functions were considered: the symmetric Schaefer (logistic) model and the asymmetric Gompertz model (Haddon 2001). Previously, Uphoff (2006a) used the Schaefer model as the basis for exploration fishing and predation-competition effects on weakfish, while Crecco (2006) used the Gompertz version.

Biomass dynamics based on the Schaefer model were described by the following discrete time-step equation:

$$B_t = B_{t-1} + rB_{t-1} (1-(B_{t-1} / K)) - H_{t-1} - D_{t-1} + \varepsilon; \qquad (1)$$

where B_t was weakfish biomass in year t; B_{t-1} = biomass the previous year, r = intrinsic rate of population increase; K = maximum population biomass; H_{t-1} = harvest (commercial and recreational, including discard estimates) in the previous year; D_{t-1} was predation-competition losses from striped bass and ε observation error (described below; Hilborn and Walters 1992: Collie and Spencer 1993; Spencer and Collie 1995; Haddon 2001).

The Gompertz predator-prey model was expressed as

$$B_t = B_{t-1} + \log_e(K) \; r \, B_{t-1} [1-((\log_e (B_{t-1})) / (\log_e(K))] - H_{t-1} - D_{t-1} + \varepsilon \qquad (2)$$

Quinn and Deriso 1999; Haddon 2001; Crecco 2006). Parameter definitions are the same as equation 1.

In all, a dozen models were formulated, a half-dozen for each production function. Fishing only hypotheses were modeled by omitting functions that estimated D_{t-1}. Predator-prey functions described Type I, II, and III predator functional responses, as well as two functions designed to mimic depensation. The Type I, II, and III responses assumed that rates of consumption by striped bass were dependent on weakfish density (prey-dependent trophic function; Ginzburg and Akçakaya 1992).

The Type I functional response estimated D_{t-1} as

$$c \, P_{t-1} \, B_{t-1}; \qquad (3)$$

where c = a positive constant; P_{t-1} = striped bass biomass in the previous year; and B_{t-1} = biomass of weakfish in the previous year (Yodzis 1994; Collie and DeLong 1999). Estimates of D_{t-1} from the Type II function were estimated as

$$(cP_{t-1}B_{t-1}) / (1 + a \, B_{t-1}); \qquad (4)$$

where c is a positive constant and a is a coefficient that combines search time and probability of capture (Yodzis 1994; Collie and DeLong 1999). The Type III function estimated D_{t-1} as

$$[(dP_{t-1}(B_{t-1})^2)/) / (A^2+(B_{t-1})^2)] ; \qquad (5)$$

where d is maximum per capita consumption by striped bass biomass (P_{t-1}); and A is weakfish biomass where predator satiation begins (Collie and Spencer 1993; Spencer and Collie 1995).

Depensatory models did not include B_{t-1} to estimate D_{t-1}. The simplest depensatory model describes D_{t-1} solely as a function of striped bass (predator-competitor) biomass:

$$c\mathbf{P_{t-1}}. \tag{6}$$

A final depensatory model expressed D_{t-1} as a function of the ratio of striped bass biomass to the biomass of its main prey, Atlantic menhaden (R_{t-1}) and striped bass (predator-competitor) biomass:

$$\mathbf{P_{t-1}}(c/\ \mathbf{R_{t-1}}). \tag{7}$$

This equation expresses D_{t-1} solely as a function of striped bass attack success on its main prey, Atlantic menhaden and weakfish were alternative prey. Ratio-dependence is another approach to functional response theory, and the ratio summarizes interference of predators with one another and spatial restriction in foraging (Ginzburg and Akçakaya 1992; Walters and Juanes 1993; Yodzis 1994; Walters and Martell 2004). Natural systems may be closer to ratio-dependence than prey-dependence (Ginzburg and Akçakaya 1992).

The Haddon (2001) spreadsheet version of a biomass dynamic model was easy to adapt to predator-prey formulations because it estimated weakfish biomass as a first step and then estimated an annual scalar (q_t) as $B_t / ZEBI_t$ (Haddon 2001). Estimating weakfish biomass first allowed striped bass biomass to be used directly rather than converting them to weakfish index equivalents. The geometric mean of annual estimates of q_t (GM q_t) was used to predict $ZEBI_t$ as geometric mean GM $q_t * B_t$. Biomass was estimated directly for 1981 (B_{1981}) and then the model estimated subsequent years (Haddon 2001).

An observation error model was used that assumed all residual errors were in the index observations, and the equation used to describe the time-series was deterministic and without error (Haddon 2001). A genetic algorithm super solver (Evolver; Palisade Corporation 2001) was used to estimate predator-prey model parameters that minimized the sum of observation errors (observed $\log_e ZEBI_t$ - predicted $\log_e ZEBI_t)^2$ (Haddon 2001). The spreadsheet version of the model combined with the genetic algorithm provided a great deal of flexibility for trying different model variations.

C10.4.5 Hypothesis Testing

We used Akaike information criteria adjusted for small sample size, AIC_c, to evaluate the 12 models describing hypotheses that related changes in weakfish biomass to fishing alone or to fishing + predation-competition losses due to striped bass (Burnham and Anderson 2001):

$$\mathbf{AIC_c = -2(log\text{-}likelihood) + 2K + [(2K(K+1)) / (n\text{-}K\text{-}1)]}; \tag{8}$$

where n is sample size and K is the number of model parameters. We calculated lognormal likelihoods as described by equations 3.29 and 3.30 in Haddon (2001). We rescaled AIC_c values to Δ_i, ($AIC_{c\,i}$ – minimum AIC_c) where i is an individual model (Burnham and Anderson 2001). The Δ_i values provide a quick "strength of evidence" comparison and ranking of models and hypotheses. Values of $\Delta_i \leq 2$ have substantial support, while those > 10 have essentially no support. We used Δ_i to calculate Akaike weights that approximated the probability that model i constituted the best model of the hypothesis given the data. The best model or subset of models were used to generate parameters of interest, estimate variability, and explore model sensitivity. An R^2 was calculated for

the best model or models to describe goodness of fit (Burnham and Anderson 2001).

Once a best model (or models) was chosen, residuals were examined to see if they were normally distributed with a mean of zero and to see if serial trends were present. We used "BestFit" software (Palisade Corporation 1997) to examine whether residuals were normally distributed. Bestfit identifies a distribution that most likely produced the data using Chi-square, Kolmogorov-Smirnov, and Anderson-Darling statistics and provides graphical output for viewing the selected distributions (Palisade Corporation 1997). A regression of residuals against year explored whether a significant linear trend (autocorrelation) was present; other types of serial patterns were evaluated visually. If residual patterns were acceptable, precision of parameters (see below) was estimated. If parameters were significantly different from zero, the model was accepted.

All 12 models had the same number of observations (n = 60), while parameters varied from 4-6. Trial versions of biomass dynamic models indicated one data point (NJA ZEBI in 2003) would account for a large portion (nearly half in some cases) of the sums of squares (SSQ) and this point was eliminated from SSQ calculations in all models.

C10.4.6 Model Output and Biological Reference Points

Instantaneous annual fishing mortality rate in year t was estimated as:

$$\mathbf{F_t = H_{t-1} / [(B_t + B_{t-1}) / 2]} \tag{9}$$

(Ricker 1975). An equivalent instantaneous natural mortality rate associated with striped bass predation and competition was estimated as:

$$\mathbf{M_{pt} = D_{t-1} / [(B_t + B_{t-1}) / 2]}. \tag{10}$$

Total instantaneous annual mortality of weakfish (Z_{pt}) due to due to fishing and striped bass predation equaled $F_t + M_{pt}$. This estimate did not account for other sources of natural mortality not associated with food web effects represented by striped bass. The ratio of F_t to Z_{pt} provided an indicator of "leverage" for recovery by managing the weakfish fishery and bycatch (V. Crecco, CT DEP, personal communication).

Loss of weakfish biomass per unit of striped bass biomass was estimated as

$$\mathbf{D_t / P_t}. \tag{11}$$

The functional response of striped bass to weakfish was displayed visually by plotting D_t / P_t against B_t.

Surplus production estimated from the logistic predator-prey analysis (SP_t) was estimated with two equations. The first estimate was from the standard equation and applied to fishing only models:

$$\mathbf{B_t - B_{t-1} + H_{t-1}} \tag{12}$$

(Prager 1994; Jacobson et al. 2002). The second estimate of SP (SP_{Dt}) explicitly accounted for losses due to striped bass predation and competition:

$$B_t - B_{t-1} - D_{t-1} + H_{t-1}. \tag{13}$$

This approach defined surplus production similarly to Overholtz et al. (2008). Correlation analysis was used to estimate associations of SP or SP_{Dt} with F_t and/or M_{pt} to explore the possible linkage of these parameters to changes in surplus production.

Production (recruitment to exploitable biomass + individual weight gain) was estimated by adding estimates of D_{t-1} (if necessary) to the basic production equation described by Walters and Hilborn (2005):

$$P_{t-1} = B_t - B_{t-1} + H_{t-1} + D_{t-1}. \tag{14}$$

Estimates of F were compared to relative F estimates rescaled into F (Sinclair 1998; Crecco 2006) using:

$$F_{Rt} = (H_{t-1} * q_{mean}) / ((mean\ ZEBI_t + mean\ ZEBI_{t-1}) / 2); \tag{15}$$

where F_R is an F estimated by rescaling harvest in year t into ZEBI units using the geometric mean of q_t estimated from the biomass dynamic model and then dividing this product into the average of Z+2 transformed WRI, DE EBI, and NJA ZEBI in years t and t+1. This comparison allowed me to view the impact of the observation error assumption on trends in F based on indices.

Equilibrium biological reference points (EBRPs) could be estimated for relevant Schaefer (F_{msy} = r / 2 and B_{msy} = K / 2) and Gompertz models (F_{msy} = r $F_{collapse}$ = r * 2.72, B_{msy} = K / 2.72; Quinn and Deriso 1999; Crecco 2006). Nonequilibrium reference points were also developed for predator-prey models that accounted for additional losses from striped bass that were assumed to be responsible for all but F and residual natural mortality (denoted as M_1 by Overholtz et al. 2008).

Two approaches were used to estimate reference points (NBRPs) for mortality when predation-competition losses were included: total mortality at maximum sustained yield (Z_{msy}) and annual non-equilibrium F_{msy} (F_{psyt}). The former simply involved renaming equilibrium estimates of F_{msy} as Z_{msy} in models with predator-prey terms and comparing it to Z_{pt}. This estimate of total mortality at MSY would be constant across time and would serve as a benchmark for total mortality changes.

Annual F that provided MSY after accounting for predation was estimated as

$$F_{psyt} = F_{msy} - M_{pt}. \tag{16}$$

In this case, the predator or predator-prey conditions related to total mortality is considered "off-limits" for management and yield can only be influenced by managing the fishery.

Amendment 4 to the Interstate Fishery Management Plan for Weakfish (ASMFC 2002) lists SSB that is 20% of an unfished stock as a maximum spawning potential (MSP) threshold and 30% as a target. Although biomass dynamic models do not provide SSB thresholds explicitly, the biology of weakfish allows for MSP proxies. We used B_t / K as a proxy for MSP to compare to the target and threshold. Minimum size criterion for inclusion as exploitable biomass ($\geq$ 250 mm) was greater than minimum size at 50% maturity (168 mm; all age 1; Nye et al. 2008). Maturity was complete by

230 mm and all age 2 fish were mature; however, age 1 weakfish produced far fewer spawns per season than ages 2 and older (weakfish are indeterminate batch spawners; Lowerre-Barbieri et al. 1998; Nye et al. 2008). Examination of 2001-2003 length at age data (J. Brust NJDEP, personal communication) indicated that age 1 weakfish mostly ranged between 130-300 mm and age 2 ranged from 200-400 mm. Inclusion of age 1 weakfish in B_t as well as annual variation in batch fecundity could introduce some bias in trying to directly interpret it as a proxy for egg deposition, but these would be problems for equilibrium-based calculations from spawner biomass per recruit or equilibrium yield models as well.

C10.4.7 *Variability and Model Sensitivity*

Jackknifing and bootstrapping (Efron and Tibshirani 1993) were used to describe variability of parameters estimated in the final model. The bootstrapping technique recommended by Prager (1994) that preserved the original order of the time-series was used. For each set of bootstrap trials, a synthetic data set was constructed by combining a random sample of the original residuals with the ordered predictions from the original fit; the model was then rerun with each set of synthetic observations (Efron and Gong 1983; Efron and Tibshirani 1993; Prager 1994; Haddon 2001). Percentile confidence intervals of model parameters and the time-series of B_t, F_t, M_{pt} and Z_t / Z_{msy} were estimated.

First, jackknifing was conducted manually using Evolver. Then bootstrapping followed using Excel's Solver. In order for Solver to run, parameter constraints were imposed based on, but broader than, jackknife minimums and maximums.

Several approaches were used to investigate sensitivity of model parameters and estimates of Z_t, F_t, and B_t. All sensitivity analyses were deterministic. Estimates of B_t were standardized to K (B_t / K) and estimates of Z_t and F_t were standardized to Z_{msy}. These standardizations were used in an attempt to minimize likely correlations between r, K, and B_0 that would affect comparisons based on absolute values of Z_t, F_t, and B_t.

Sensitivity of model parameters to data from the beginning or ending of the time-series was tested by removing blocks of data and rerunning the model. Time blocks of years up through 1989 were removed from the beginning of the model. These time blocks at the beginning encompassed data where WRI was the only time-series until the first fishery-independent data were available. Up to three of the most recent years were removed from the time-series that began in 1981. Biomass estimated in 1981 for the 1981-2006 time-series was greater than K and we ran a version of this model with B_{1981} constrained to be less than K for comparison (Prager 1994). B_t / K estimates were compared in addition to model parameters in this last exercise.

C10.5 Results

C10.5.1 *Hypothesis Testing*

AIC_c scores indicated a 98% chance that the Gompertz production model with a depensatory function relating M_{pt} to the biomass of age 2+ striped bass and the ratio of menhaden to striped bass biomass was best (hereafter, Gompertz depensatory ratio model or GDR; equations 2 and 7) for describing the hypothesis given the data (Table C10.5-1). This model fit the data well and explained 90% of the variation in ZEBI (Figure C10.5-1). Remaining models will not be considered, but it is interesting to note that the second and third ranked models were also Gompertz functions with predator-prey functions. Fishing only models were poor choices for describing biomass dynamics of the data, ranking seventh and ninth out of twelve (Table C10.5-1).

The mean of residuals of the GDR model was close to zero (0.0045) and the normal distribution was either the first or second choice of the three procedures in BestFit (a triangular distribution was first in two). A linear serial trend in residuals was not indicated by regression ($r^2 = 0.014$, P = 0.37). There were periods of larger positive (1986-1989) and negative (1993-1995) residuals, but residuals of remaining years were mixed (Figure C10.5-2).

C10.5.2 Model Parameters

Model parameters were estimated as r = 0.48, K = 51,521 mt, c = 0.29, and B_{1981} = 82,472 mt. The EBRPs were F_{msy} or Z_{msy} = 0.48, $F_{collapse}$ = 1.31, and B_{msy} = 18,941 mt. Jackknifing and bootstrapping indicated all parameters were precisely estimated (Tables C10.5-2 and C10.5-3). Parameter values of the base run were very close to their medians, indicating minimal bias. Bootstrapping required constraints and changes to base settings of Excel after preliminary trials indicated repeated failure of Solver without them. Constraints were r, 0.40 – 0.55; K, 45,000 - 55,000 MT: B_{1981}, 65,000 - 90,000 MT; and c, 0.2 - 0.4. Iterations were set at 1,000, precision to 0.1, tolerance to 0.5%, and convergence to 0.1.

The initial biomass estimate (1981) exceeded the estimate of K. We applied the penalty function for large values of B_0 described by Prager (1994), but this did not change results substantially. We proceeded through a series of sensitivity analyses to determine a course of action. Estimates of other model parameters were influenced by the estimate of B_0. Estimates of r (and dependent reference points) declined steadily as time-blocks were removed and B_0 declined (Table C10.5-4). Estimates of K and c displayed a down and then up trajectory as years were removed. When models were initiated progressively later from 1981-1984, K declined from 51,000 mt to 46,000 mt and then rose at a quicker rate to 78,000 mt as 1985-1989 became the initial years in the time-series. Coefficient c declined steadily from 0.29 to 0.20 as 1981-1987 time-blocks were removed and then rose rapidly to 0.35 when the GDR model was initiated in 1988 or 1989. Constraining B_{1981} to be less than K lowered r and raised K substantially and resulted in a slight drop in c. The mean of the squared residuals was lowest with the unconstrained 1981-2006 data set (Table C10.5-4). The net impact of these parameter changes was judged by comparing time-series of B_t / K, Z_t / Z_{msy}, and F_t / Z_{msy}.

Sensitivity analyses indicated that B_0 / K exceeded one when the time-series was initiated in 1981 or 1982 and was close to one when initiated in 1983 or 1984 (Figure C10.5-3). Remaining treatments did not exhibit B_0 / K estimates exceeding or approaching one. Estimates of B_t / K converged for the remainder of the time-series by the late-1980s. Constraining B_{1981} to less than K resulted in convergence of B_t / K with the unrestrained GDR by 1983 (Figure C10.5-3).

Estimates of Z_t / Z_{msy} were similar in trend among time-block treatments (initial years removed) until the most recent three years of the time-series, when a bifurcation between a steady increase and a plateau occurred (Figure C10.5-4). The unconstrained, constrained, 1982+ time-series, and 1989+ time-series (additional indices present) treatments produced similar increasing trends to Z_{2006} / Z_{msy} ≈ 3.0-3.5,while remaining treatments indicated a leveling of Z_{2006} / Z_{msy} ≈ 2.0-2.5. There were mixed rankings of values of Z_t / Z_{msy} among treatments, but similar trends for 1985-2003. There was more variability among treatments of years prior to 1985 (Figure C10.5-4).

Trends in F_t / Z_{msy} were similar among time-block removal treatments (Figure C10.5-5). Ranking of treatments was mixed across the time-series; there was not any one treatment that produced a consistent higher or lower value. Time-series based on the unconstrained B_{1981} and B_{1981} constrained to be less than K converged by1983. Treatments using all years of data generally produced highest values of F_t / Z_{msy} in 2006 (0.66 for no constraint on B_{1981} / K and 0.63 for B_{1981} <

K). Over all treatments, F_t / Z_{msy} during 2006 varied between 0.45 and 0.66, with a median of 0.53 (Figure C10.5-5).

Overall, differences in B_t / K, Z_t / Z_{msy}, and F_t / Z_{msy} stabilized by 1983 when the entire time-series was used and B_{1981} was constrained. Biomass dynamics of weakfish were portrayed similarly among all initial time-block removal treatments. We chose to keep the results of the unconstrained GDR, but did not use 1981 and 1982 estimates. Prager (1994) recommended not using the first 2-4 years of production model estimates to draw inferences about biomass.

Removing up to three of the most recent years from the time-series had little impact on parameter estimates, except for B_{1981}. Estimates of r increased from 0.48 when all data were used, to 0.50 when 2006 or 2005-2006 were removed; estimates of K did not change; B_{1981} declined from 82,472 to 68,286 MT, and c fell from 0.292 to 0.291. Collectively, these changes did little to alter depiction of biomass dynamics after 1982 (illustrated by B_t / K; Figure C10.5-6).

Assigning larger mean weights to recreational discards influenced parameter estimates (Table C10.5-5), but did not discernibly alter estimates of B_t unless weights above 0.3 kg or harvest mean weights were assigned (Figure C10.5-7). We accepted the use of 0.15 kg for recreational discard weights in the WRI and fishery loss estimates. Only the run with 0.4 kg mean weight had B_t estimates that were consistently higher than other runs. Using harvest mean weight produced similar estimates of B_t as the remaining treatments during 1982-1991, but estimates were consistently higher afterwards (Figure C10.5-7). The timing of the separation of estimates based on harvest weights was similar to the timing of increasing minimum length limits beginning with Amendment 1 (ASMFC 2002); size limits would have increased recreational discarding and the impact of assumed discard weights.

Biomass was about 33,600 mt in 1983 and fell to 6,300 mt by 1990; B_t rose to 18,000 mt in 1996-1997 and then began to decline again, reaching a brief plateau at 5,000-6,000 mt during 2003-2005 (Figure C10.5-8). The pace of loss accelerated in 2006 (4,000 mt) and 2007 (3,000 mt). Landings exhibited the same general trend as B_t. Initial losses to striped bass (D_t) were very low and rose slowly to less than 1,000 mt during 1983-1993 and then began to accelerate to about 5,100 mt by 1997. Estimates of D_t fluctuated between 4,100 and 5,900 mt during 1998-2006 while B_t was dropping. Fishery-related loss estimates fell below D_t in 1999 and were less than 25% of D_t since 2004 (Figure C10.5-8). Jackknife and bootstrap estimates of B_t (Figures C10.5-9 and C10.5-10) and D_t (Figures C10.5-11 and C10.5-12) indicated little bias and estimated precision of B_t and D_t from these techniques was high.

Estimates of F_t were near Z_{msy} (0.48) during 1983-1985, then peaked quickly at 1.3 during 1988, then fell steadily to below Z_{msy} in 1992 (Figure C10.5-13). F_t remained below Z_{msy} during 1992-1997 and then rose slowly to a level slightly above Z_{msy} during 1998-2002. After 2002, F_t dropped sharply and remained at about 50-70% of Z_{msy} through 2006 (Figure C10.5-13).

Estimates of M_{pt} rose from a very low level in the mid-1980s to estimates equivalent to F_t and Z_{msy} by 1999 (Figure C10.5-13). After 1999, M_{pt} accelerated rapidly and reached 1.4 in 2006 (Figure C10.5-13).

Comparing F_t to equilibrium F_{msy} and non-equilibrium F_{psyt} may provide the best depiction of how D_t influenced biomass dynamics (Figure C10.5-14). After 1992, F_t remained below or near what would be equilibrium F_{msy} (or Z_{msy}); however, increasing losses to striped bass were continuously eroding F_{psyt} (Figure C10.5-14). This erosion was reflected in continuous estimates of negative SP_{Dt} after 1996 (Figure C10.5-15). Negative surplus production (in absolute terms) exceeded landings since 2002 and biomass since 2006. Surplus production was generally in excess of 6,000 mt during 1983-1994 (Figure C10.5-15). Surplus production was significantly associated

($P < 0.01$) with F_t and M_{pt}, but the modest positive correlation ($\rho = +0.54$) with F_t was somewhat counter-intuitive (although F should increase surplus production initially in an unfished or lightly fished stock) while the association with M_{pt} was logical, negative, and strong ($\rho = -0.90$).

Production varied between 5,000 – 9,000 mt during 1984-2006 and was highest during periods when biomass was greater than production (1984-1988 and 1992-2002; Figure 20). High biomass during the mid-1980s was not reflective of concurrent production and its deterioration was not particularly reflective of F_t, or M_{pt}. Production was close to biomass when total mortality rates were high during 1989-1991 (due to fishing) and 2003-2006 (due to predation-competition; Figure C10.5-15). Production appeared to decline when proxy MSP (B_t / K) fell below 20% (Figure C10.5-16).

Estimates of proxy MSP (B_t / K) fell continuously after 1983 from 65% of K to reach about 12% by 1990 (Figure C10.5-16). F_t / Z_{pt} indicated that this early decline would have been attributed to F (ratio was near 1). Fishing mortality reductions lead to rebuilding to approximately 35% of MSP by 1996-1997 (target level in Amendment 4 = 30%; ASMFC 2002), while F_t / Z_{pt} was falling from near 1.0 to approximately 0.5. Spawning potential continuously fell after 1997, reaching 8% MSP in 2006 and 6% MSP in 2007 as F_t / Z_{pt} declined from about 0.5 in 1997-2002 to less than 0.2 in 2006 (Figure C10.5-16).

Estimates of F_t from the GDR were usually close to estimates of relative F_t, except during 1986-1989 (Figure C10.5-17). A plot of catchability over time ($q_t = ZEBI_t / B_t$) suggests this was a period of elevated catchability in the WRI (Figure C10.5-18). However, catchability was higher for the 1989 NJA EBI as well, indicating that there may have been regional aggregation of weakfish as biomass was declining as opposed to nonrandom behavior by recreational anglers alone (Figure C10.5-18).

Jackknife and bootstrap estimates of F_t were precise, with slightly more variability in estimates during 2005-2006 (Figures C10.5-19 and C10.5-20, respectively). Estimates of F_{2006} were likely to be between 0.23 and 0.32 based on jackknife 90% percentile confidence intervals, while bootstrapping indicated F_t was likely between 0.31 and 0.40. Median and original values of F_{2006} (0.31) were very close, indicating minimal bias.

As with F_t, jackknife and bootstrap estimates of M_{pt} were precise, with slightly more variability in estimates during 2005-2006 (Figures C10.5-21 and C10.5-22). Estimates of M_{p2006} were likely to be between 1.3 and 1.7 based on jackknife 90% percentile confidence intervals, while bootstrapping indicated M_{p2006} was likely between 1.0 and 1.4. Median estimates of M_{p2006} from both techniques and original values (1.41) were the same.

Estimated loss of weakfish biomass per biomass of striped bass (D_t / P_t) was very low (0.004 – 0.005 mt / mt) during 1983-1987 when the menhaden to striped bass biomass ratio (R_t) was at its zenith (70-80 mt / mt; Figure C10.5-23). A rapid drop in R_t to 20-30 during 1988-1993 produced a rise in estimated D_t / P_t to 0.01-0.02. By 1997, R_t became asymptotically low at less than 10, while modeled estimates of D_t / P_t became asymptotically high between 0.04 and 0.06 (Figure 28). These changes in R_t produced two striped bass functional response regimes (Figure C10.5-24). During 1983-1993, while weakfish biomass (B_t) fell from 34,000 to 6,300 mt, D_t / P_t was low – between 0.004 and 0.01. A rapid transition followed and during 1997-2006, D_t / P_t increased to 0.04-0.06 as B_t fell from 18,000 to 4,000 mt (Figure C10.5-24).

C10.6 Verification of Predation-Competition Loss Estimation

C10.6.1 Long-term Evidence of the Influence of the Menhaden to Striped Bass Ratio on Weakfish

A long-term view (1959-2006) of the influence of the menhaden to striped bass ratio on weakfish was developed from an index-based menhaden to bass ratio, Atlantic Coast weakfish commercial harvest, and a Delaware Bay trawl survey length-frequency index (DE PSD Q+; Appendix C-2). Chesapeake Bay menhaden to bass index ratios (CR_t) for 1959-2006 were developed from presence-absence of age 0 menhaden in the MD seine survey (E. Durell, MD DNR, personal communication; see http://www.dnr.state.md.us/fisheries/juvindex/index.html for survey description) and striped bass egg presence-absence indices in MD spawning areas (an index of mature striped bass female biomass; Uphoff 1997). Presence-absence of menhaden in the MD survey was significantly related to estimates of coastal menhaden abundance (all ages) two years later from the ASMFC assessment (linear regression, 1959-2005, $r^2 = 0.58$, P < 0.0001; J. Uphoff, MD DNR, unpublished analysis). As with Skud (1982), my basic premise was that a truly competitive relationship between striped bass and weakfish would transcend deficiencies of weakfish landings or indices. A strong correlation existed between weakfish landings and the fishery-independent DE PSD Q+ index ($\rho = 0.94$), indicating both reflected similar processes.

Trends in both sets of menhaden to striped bass biomass ratios (R_t and CR_t) tracked each other closely, falling from their highest levels in the early 1980s to asymptotic lows in the mid-1990s (Figure C10.6-1). Chesapeake Bay menhaden to bass index ratios (CR_t) indicated low attack success in the 1960s and a sudden rise in 1971. Higher ratios were maintained until the early 1990s (Figure C10.6-1). A linear regression of 1982-2005 estimates (strictly based on assessment results as opposed to 1981 and 2006 predictions) was significant ($r^2 = 0.80$, P < 0.001) and CR_t could be converted to R_t with the equation:

$$R_t = (67.3 * CR_t) - 5.8.$$

During 1959-2006, weakfish commercial harvest and the DE PSD Q+ index closely followed CR_t (Figure C10.6-2) and correlations with the CR_t were strong ($\rho \approx 0.82$ in both cases). These associations indicated that this ratio has been important in dynamics of weakfish beyond the period covered by the GDR.

C10.6.2 Short-term Evidence of the Influence of Striped Bass on Weakfish Condition

Competition between striped bass and weakfish would be expected to influence condition of weakfish. An estimate of mean weight at a standard length provides a measure of body condition (Deriso et al. 2008).

Individual lengths and weights were available from the NEFSC fall survey annually for 1992-2006 (Uphoff 2005; G. Shepherd, NMFS, personal communication). These samples are collected by standardized trawl in a stratified random design during a restricted period of time (NEFSC 2007) and should minimize gear, location, and seasonal variation of weight. Predicted weight-at-length was derived from annual regressions of log_e-transformed weight (kg) on log_e-transformed length (cm) that were applied to NJ trawl survey length-frequencies to derive exploitable biomass (*described previously in Trawl Survey Indices of Exploitable Biomass*). Generally, weakfish < 400 mm were well represented in the length-weight samples, although larger weakfish were present in the length-weight samples. Fits of the log_e-transformed length and weight data were very good ($r^2 \approx 0.99$, except 2003 $r^2 = 0.97$). The length used to designate quality

weakfish in PSD analysis (Appendix C-2), 340 mm, was used as a standard length and predicted weight at this length was used as a body condition indicator (hereafter, Q_{wt}; Table C10.6-1). Regression analysis tested whether Q_{wt} had undergone a significant linear decline since 1992, and whether it was negatively related to striped bass biomass or the ratio of menhaden to striped bass biomass (R_t). Inspection of residuals of these regressions indicated that 2003 Q_{wt} was a potential outlier and a separate set of analyses was run with this year removed.

Regression analysis (all years and with 2003 removed; r^2 range = 0.41-0.61 and P range = 0.0009-0.01) indicated that Q_{wt} had undergone a significant decline during 1992-2006, and that the decline in Q_{wt} was significantly related with both striped bass biomass and the ratio of menhaden to striped bass biomass (Table C10.6-2). Predicted body mass at 340 mm was negatively related to striped bass biomass and positively related to the ratio of menhaden to striped bass biomass. These relationships were consistent with expected effects of intense competition between weakfish and striped bass and with parameterization of the GDR model.

C10.6.3 Comparison of Model and Field Estimates of Weakfish Consumption by Striped Bass

Unlike harvest, we do not have a natural mortality "body count" to unequivocally estimate D_t or M_{pt}. However, estimates of D_t / P_t can be judged by comparing them to estimates of maximum consumption (C_{max}) and annual consumption of food (C_t; weight of all prey, fish and invertebrates, consumed) per weight of striped bass on an annual basis from bioenergetic models of striped bass. These estimates have been made for Chesapeake Bay during 1955-1959 (Griffin 2001), 1990-1992 (Hartman and Brandt 1995a; 1995b), and 1998-2001 (Overton 2003). Bioenergetics models of Griffin (2001) and Overton (2003) were the same as Hartman and Brandt (1995a; 1995b), but used diet and temperature data for their respective time periods. We confined comparisons of C_t to striped bass 2 years-old and older on day 365 (full year's consumption); these studies estimated C_t for ages 2-6. These comparisons allow for an estimate of the scale of individual consumption of weakfish by striped bass and may answer the questions "Are weakfish a major or minor item?" or "Are the estimates of D_t / P_t excessively high (in excess of C_{max} or C_t)?"

Overton et al. (2008) provided a time-series of striped bass diet trends to compare with D_t / P_t. This study examined 1,154 striped bass during winter in the Atlantic Ocean off the VA and NC coasts. These striped bass were collected during 1994-1996, 2000, 2002-2003, and 2005-2007 and their diets were dominated by Atlantic menhaden and bay anchovy (Overton et al. 2008). Percent of striped bass stomachs with food (SS_t) was relevant to GDR estimates of D_t / P_t. In the GDR, decreased feeding success by striped bass (indicated independently by SS_t) should lead to increased search times and more encounters with weakfish since the diet is dominated by other species and includes a large contribution by menhaden, i.e., D_t / P_t would be negatively related to SS_t and influenced by R_t. Linear regression with categorical variables and a slope shift coefficient was used to analyze these relationships (Freund and Littel 2000). The annual ratio of menhaden biomass to striped bass biomass (R_t) was converted to a categorical variable (MR_t), 0 for ratios 10 and above (1994-1996) and 1 for ratios below 10 (remaining years). Categories were used to minimize confounding with D_t / P_t estimates derived from the model that used these ratios to estimate D_t. A slope shift coefficient was estimated because the time trend in the functional response plot indicate two periods with different slopes that would have to be accounted for within the span of Overton et al. (2008) data (see Figure C10.5-24). The slope shift coefficient was estimated by including a variable equal to the product of SS_t, multiplied by MR_t (Freund and Littel 2000). These variables were tested for inclusion by multiple regression and those significant at $P < 0.05$ were retained. An inverse transformation of the dependent variable, D_t / P_t, was used to place the small ratios on a

larger scale.

Estimates of D_t / P_t were quite low when compared to estimates of weight of annual food consumption (fish and invertebrates) per weight of striped bass (C_t) in three Chesapeake Bay bioenergetics studies (Table C10.6-3; Hartman and Brandt 1995; Griffin 2001; Griffin and Margraf 2003; Overton 2003). Estimates of C_t ranged from 4.1 to 7.9, with a median of 5.5, while D_t / P_t fell between 0.004 and 0.06. Estimates of C_t were about half to a third of C_{max} (Hartman and Brandt 1995). Weakfish were a minor diet item subject to a very low level of consumption by striped bass (if all losses were strictly defined as consumption). This low level of consumption was applied over an increasingly large striped bass biomass that was engaging in higher search times (reflected by low R_t) for their main prey after 1996. Encounters with weakfish may have been facilitated by diet, spatial, and temporal overlap with striped bass, resulting in disproportionately high D_t as weakfish biomass declined. There was no feedback between D_t and P_t since weakfish were such a minor diet item.

Estimates of D_t / P_t were strongly related to estimates of feeding success of striped bass in coastal VA and NC during winter, indicated by SS_t (as a percentage). The relationship was described by linear regression with a slope shift coefficient:

$$D_t / P_t = 1 / [(0.204*(SS_t)) - (0.16*(MR_t* SS_t)) + 15.35]; (R^2 = 0.89, P = 0.0013);$$

where MR_t is the category for menhaden to striped bass biomass ratio (Tables C10.6-4 and C10.6-5).

C10.7 Could Bycatch be Responsible for the Recent Decline?

Weakfish Peer Review and Data Poor Workshop panels in 2006 and 2008, respectively, suggested that unreported commercial discards of weakfish after 1995 could have created the recent weakfish stock collapse in lieu of our hypothesis of a rise in natural mortality. We fit Gompertz biomass dynamic models to the three ZEBI's, harvest, estimated discards, and an additional loss term (L_F) mimicking various commercial bycatch scenarios (L_F increasing as linear, quadratic, or exponential functions of time, a constant multiple of bycatch estimates, or as a constant additional weight). Additional losses were imposed after 1995 to reflect regulatory discards. Results were contrasted with the GDR and a Gompertz model using estimated fishery losses without additions. We used AIC$_c$ to calculate Akaike weights that approximated the probability that model i constituted the best model of the hypothesis given the data.

Akaike weights (Table C10.7-1) indicated that there was approximately a 91% chance that the GDR provided the best model of the dynamics of the data, while there was a 6% chance when L_F was constant (an additional 3,495 MT since 1996), and a 3% chance when L_F increased as a quadratic function of time (increasing from an 2,273 MT in 1996 to 5,054 MT in 2006). These two bycatch scenarios invoke about the same biomass of L_F as D_t estimated by GDR (Figure C10.7-1). In general, bycatch estimates started at about 2-3 times the estimates of 1996 and ended up 15 to 20 times higher by 2006 (Figure C10.7-1). Each of these three scenarios provided an excellent fit to the EBI ($R^2 \approx 0.90$). Remaining models had near zero odds of explaining the data compared to these three.

C10.8 Discussion

Biomass dynamic modeling indicated the weakfish stock was not overfished in 2006 based on equilibrium F_{msy}, but was subject to high natural mortality that eroded the safe level of fishing. The

B_t / K proxy for MSP in 2007 (6%) was far below the 20% MSP threshold in Amendment 4, while high negative values of F_{psyt} (non-equilibrium F_{msy}) and surplus production indicated that complete (and unlikely) elimination of harvest and bycatch would not be sufficient to end the decline. Production persisted at a modest level in recent years, although it was falling gradually. Depensatory mortality, driven by high striped bass biomass and a low ratio of Atlantic menhaden to striped bass (an indicator of low feeding success on striped bass' main prey), appeared the most likely explanation for increasing natural mortality that undermined recovery given the data, hypotheses, and models developed. Fishing played a secondary role in recent biomass dynamics. Striped bass predation-competition dominated weakfish biomass dynamics after overharvesting had been controlled in the early 1990s.

Regression analyses and biomass dynamic models indicated high potential for striped bass, menhaden, and weakfish linkage. Spatial, temporal, and diet overlaps were sufficient for interactions to occur. Modeling indicated high mortality of weakfish from striped bass was derived from very low loss of weakfish per striped bass applied over a large striped bass biomass. Predation-competition from striped bass has increasingly eroded weakfish surplus production and F_{msy} reference points and there seems little chance of restoring weakfish by manipulating its fisheries alone because F has become a low fraction of total mortality (≈20% by 2006). At this time, leverage for manipulating weakfish may mostly reside in the menhaden to bass ratio.

Projections of future weakfish biomass under these conditions indicated little ability to generate recovery by manipulating the fishery and suggest extirpation in a time frame shorter than a moratorium could be implemented. These predictions should be greatly qualified. We believe these predictions indicate (1) a need for broader multispecies or ecosystem-based management perspective on weakfish and (2) managing the weakfish fishery alone is unlikely to have much impact unless conditions contributing to high M_{pt} (very high striped bass biomass and low menhaden-bass ratio) lessen. The possibility of extirpation is far more uncertain than indicated by projections because of high uncertainty of model specifications. Forecasts, starting with B_{2007}, have gone beyond weakfish biomass limits represented by data (up through 2006), which the model has fit well, and continued to represent predation-competition as unchanged. This representation of a constant predation-competition function in GDR projections becomes highly uncertain as weakfish densities drop and implies lack of refuge for weakfish, as well as uniform (and suicidal) linkage of weakfish migratory and feeding behavior with striped bass. Munyandorero's (2006) summarization of weakfish population structure literature would suggest the South Atlantic region could serve as a refuge since sub-populations below NC are not subject to commercial fisheries or predation from the migratory striped bass that roam from ME to NC. The Atlantic Coast weakfish stock appears to be a mosaic of regional sub-populations with enough exchange for genetic uniformity (Munyandorero 2006).

It can be difficult to predict the effects of fishing or culling policies from fairly simple representations of predation processes (Bax 1998; Yodzis 2001; Hollowed et al. 2000a). Hollowed et al. (2000b) cited Beverton's argument that only first order symmetric interactions in which a predator and fishery exploit the same prey species are predictable. First order asymmetric interactions, in which the predator (striped bass) and fishery (weakfish) exploit different interacting prey species (striped bass mainly exploit menhaden in this case and their exploitation of weakfish is a function of their failure to do this successfully) are unpredictable (Hollowed et al. 2000a; Yodzis 2001). Further complicating projections of weakfish biomass dynamics was the reliance on single species, constant M-based biomass estimates for striped bass and Atlantic menhaden biomass as the principle drivers of M_{pt}. There is evidence for both species that their natural mortality has changed systematically (tag-based estimates for striped bass and multispecies modeling for menhaden; Jiang

et al. 2007; ASMFC Multispecies Technical Committee 2008; Gauthier et al. 2008; NEFSC 2008) and mortality could be linked between these species (Uphoff 2003). Systematic and linked changes in M would alter estimates of striped bass and menhaden biomass which drive D_t and M_{pt} in the GDR.

Atlantic menhaden is a buffer species that absorbs predatory pressure from other species when they are sufficiently abundant (ASMFC 2004). Striped bass actively select for Atlantic menhaden, but will feed on other species when menhaden are not sufficiently abundant (Overton 2003; ASMFC 2004; Ruderhausen et al. 2005). Striped bass appear capable of limiting their prey populations along the Atlantic Coast at high biomass that existed during the mid-1990s into 2006 (Hartman 2003; Uphoff 2003; Heimbuch 2008). The GDR indicated that as menhaden have become less abundant and striped bass more-so, striped bass searching has lead to increasing encounters with vulnerable-sized weakfish searching for smaller prey-fish (anchovies and age 0 menhaden) that are also found in diets of the largest striped bass (Walter and Austin 2003; Overton et al. 2008).

Hypothesis testing indicated that depensatory mortality associated with high striped bass and low menhaden biomass has reduced productivity of weakfish along the Atlantic coast. Depensation may not be common, but it is not so rare a possibility that it should be considered implausible (Walters and Kitchell 2001). Movement from high to low equilibrium states can be induced by depensatory mortality (Collie and Spencer 1993). These shifts can be regarded as jumps between alternative equilibrium states of ecological systems (Steele and Henderson 1984). The term "regime shift" has been used to suggest these changes are causally connected and can be linked to other changes in an ecosystem (Steele 1996). The regime shift concept implies that different regimes have inherent stability, so that significant forcing is required to flip the system into alternative states (Steele 1996).

The effect of striped bass on weakfish in the GDR was minimal when R_t was above 20, but it intensified (more weakfish died) as R_t fell lower and their biomass dynamics were dominated by striped bass. Switching behavior by large predators may cause unexpected, sequential depletion in prey (Walters et al. 2005). In the early to mid-1990s, important forage species (Atlantic menhaden, spot, and bay anchovy) dropped to low levels (Uphoff 2006a; ASMFC Multispecies Technical Committee 2008) and depletion of weakfish followed in the late 1990s.

Hartman and Brandt (1995) found that striped bass in Chesapeake Bay during 1990-1992 increased their use of the pelagic food web (primarily menhaden) as they aged, while weakfish increased use of benthic resources (spot). Stable isotope analysis of striped bass scales collected during 1982-1997 from Chesapeake Bay indicated striped bass increased their use of the benthic food web as menhaden abundance decreased (Pruell et al. 2003). This shift would place them in more direct competition with weakfish and increase the chance striped bass would be occupying the same habitat as weakfish while feeding.

Estimates of D_t in the GDR represented combined effects of direct striped bass predation, as well as cannibalism and starvation induced by intense competition (Yodzis 1994). Estimates of D_t may have included covarying species effects as well, but striped bass predation-competition should predominate.

Weakfish are cannibalistic; weakfish juveniles were specifically mentioned in weakfish diet studies reviewed by Mercer (1985). They comprised about 20% of weakfish diet by weight in Delaware Bay during 1985 (Taylor 1987). Atlantic coast biomass estimates indicated striped bass were at low abundance and menhaden were at high abundance during 1985 ($R_t \approx 80$), but menhaden were not abundant in DE and NJ surveys (Uphoff 2006a). Weakfish were not specifically mentioned in diets of weakfish sampled in Chesapeake Bay during 1990-1992 ($R_t \approx 20$; Hartman and Brandt

1995), but other weakfish comprised 8% of ages 0-5 weakfish diets in Chesapeake Bay during 2002-2003 ($R_t \approx 5$; Bonzek et al. 2004).

Diminished size quality (indicated by PSD Q+: Appendix C-2) and condition (indicated by Q_{wt}) were both related to the biomass of striped bass and Atlantic menhaden (Uphoff 2006a; see section **C10.6-2**). Substantial decreases in mean weight at age during 1981-2003 in the VPA catch-at-age matrix also occurred (Uphoff 2006a); however, interpretation is confounded by the potential biases as aging shifted from scales to otoliths at the same time. Natural selection balances predation risk and the need to obtain food for growth and fear of being eaten may reduce foraging and growth (Walters and Juanes 1993; Walters and Martell 2004; Trussell et al. 2006) and even lead to starvation (Pine et al 2009). Mortality due to starvation is a size-dependent process and represents an alternative (albeit final) response to reduced growth and stunting during food shortages (Ney 1990; Persson and Brönmark 2002). Starvation may be more common than generally perceived (Ney 1990) and it might be expected once the bioenergetic limits of reduced growth were breached.

Comparisons of trends in biomass and production from biomass dynamic modeling and length quality (Appendix C-2) suggested large scale shifts in biomass and production relationships from the early-to-mid 1980s to current conditions. High biomass during the mid-1980s was not reflective of concurrent production, and its deterioration during 1983-1985 was not reflective of F_t ($\approx F_{msy}$) or M_{pt} (near 0). The DE PSD Quality+ index (Appendix C-2, Figure 1) indicates that larger weakfish were more common during this earlier period. The grand mean of standardized YOY indices has increased since the 1980s (see Figure 32 in Kahn et al. 2006) as size quality has diminished. This suggests a trade-off between high growth–low recruitment and low growth –high recruitment that coincided with a drop in R_t after 1987 (see Figure 30).

Models used in this investigation were of modest complexity because of the basic limitations of the observational data. Simplified procedures allowed for the testing of overfishing and predation-competition hypotheses. The strategy of including striped bass predation-competition was chosen in light of realization that dynamics of predation, competition, environmental regime shifts, and habitat alteration or deterioration could take over once overharvesting had been controlled (Link 2002). Stationarity of ecological conditions or constancy of M cannot always be an acceptable default assumption (Sainsbury 1998) and simply presuming that ceasing exploitation on an overfished stock will result in recovery ignores the uncertainty imposed by ecological systems (Link 2002). Evaluating effects of fishing on ecosystems often has to be pursued by testing hypothetical conceptual models and multispecies models are essential (Hollowed et al. 2000a). An important element of these models is that they must include predators having the greatest impact on commercial and recreational species (Hollowed et al. 2000a).

The GDR was a step in an inferential process described by Platt (1964), Burnham and Anderson (2001); and Deriso et al. (2008). Hypothesis testing in subsequent years may yield other or additional environmental or ecological factors or functional descriptions. The "right" functional form of predator-prey models can be difficult to define (Yodzis 1994), but typical parameters in age structured single species models (constant M, stock-recruitment steepness, selectivity) are sometimes confounded and difficult to estimate (Magnusson and Hilborn 2007). The exact form of the predation-competition term may not crucial. Management benchmarks from the "top" three models (GDR, bass only, and Type II; all Gompertz functions) were very close (F_{msy} range = 0.48-0.51 and B_{msy} range = 19,000-20,000 mt), as were estimates of current status (B_t / K in 2006-2007 range = 6-9%). Management advice would not vary appreciably since Z_{2006} / Z_{msy} ranged from 2.4 to 3.6 while F_{2006} / Z_{p2006} was between 0.18 and 0.20, i.e, total mortality was very high and mostly due to striped bass predation-competition.

Overfishing was the cause of the decline of weakfish in the mid-to-late 1980s, and management was successful in preventing it from reoccurring, at least by standard equilibrium benchmarks. The failure of recovery since the late 1990s cannot be attributed to overfishing unless bycatch and under-reported catches were much greater than those estimated, growing from about 3-4 times the estimates in 1996 to 15-20 times by 2006. If results of hypothetical bycatch scenarios are taken at face value, then weakfish regulations created a massive boost in discards and represent a colossal management failure. Implementation of further conservation measures short of a coast-wide moratorium on many Atlantic coast fisheries would not minimize this nominal discard problem. There is no evidence available thus far of an Atlantic coast fishery capable of generating additional unreported weakfish discards of this magnitude (Uphoff 2006b).

Equilibrium MSY limit reference points, consistent with the precautionary approach (Restrepo et al. 1998) and estimated from the GDR, indicated that F had been near or below F_{msy} for the last 15 years, but B_t fell well below 20% of K that approximated the MSP limit in Amendment 4 (ASMFC 2002). Biomass continued falling even as F_t fell to about 50-60% of F_{msy} after 2003. Fishing mortality rates in this range should have been associated with stock expansion or stability under constant M conditions and these equilibrium-based biological reference points had little utility when M was rising rapidly. Biological reference points can be sensitive with species subject to predation and cannibalism and may represent moving targets (Collie and Gislason 2001). Multispecies simulation models indicated it was impossible to derive a single fixed value for F_{msy}, but stock-recruitment and yield curves may be distorted if multispecies effects are disregarded and predictions of recovery can be much too optimistic (Hollowed et al. 2000a). Moustahfid et al. (2009) added predation loss estimates as a "fleet" in an age structured assessment of Atlantic mackerel and noted that their inclusion altered model outputs and biological reference points. Non-equilibrium F_{psyt} was informative; however, it was not developed until predation-competition effects were well established and is likely to be retrospective in practice. Once understood, non-equilibrium F_{psyt} does provide short-term understanding of underlying productivity and its relationship to fishing. In the case of the GDR, by 2000 rising M_{pt} ate away all F that represented a safe level of fishing.

The biomass dynamic approach applied here and by others (Collie and Spencer 1993; Spencer and Collie 1995; Spencer and Collie 1997a; Collie and DeLong 1999, Crecco 2006; Uphoff 2006a) is fundamentally different from biomass or age structured multispecies models employed by Hollowed et al. (2000b), Overholtz et al. (2008), and Moustahfid et al. (2009), and ASMFC (NEFSC 2006). The biomass dynamic approach applied here generates estimates of predation-competition losses through their predation-competition terms, predator-competitor biomass or trends (Crecco (2006) used relative abundance), and fits to observed indices or estimates of "victim" biomass. Estimates of individual or stock level consumption were not applied; however, estimates individual of consumption by striped bass were needed for verifying results or they could define constraints on generated estimates. Other methods require predator abundance and annual consumption rates or estimates of consumption by age (Hollowed et al. 2000b; ASMFC 2003; Overholtz et al. 2008). Size, and prey type preference parameters, evacuation rates, biomass of "other food" not explicitly modeled, and annual or semi-annual diets may also be needed (ASMFC 2003; Moustahfid et al. 2009).

It is not possible to look at all predation processes in an ecosystem (Bax 1998; Sainsbury 1998; Walters and Martell 2004) and we concentrated on a limited representation of the food web. The GDR is a minimal realistic model that represents first-order indirect (triangular) interactions (Bax 1998). Triangular food webs have been proposed as a mechanism promoting stability in marine ecosystems (Bax 1998). The virtue of a minimum-realistic approach is tractability in analyzing and

parameterizing (Yodzis 2001). Its shortcoming is that there is no *a priori* way to know how much complexity is the minimum and that can only be judged by comparing results to more complex ecosystem models. A minimum-realistic approach was used to develop an understanding of how culling seals might affect the hake (*Merluccius paradoxis* and *Merluccius capensis*) fishery in the Benguila ecosystem (Punt and Butterworth 1995; Yodzis 2001).

The GDR supports the existence of a dominance hierarchy, as defined by Skud (1982), between striped bass (dominant) and weakfish (subordinate). A dominant species is defined as the more abundant of two species that interact and whose densities are maintained at distinctly different levels (Skud 1982). Ignoring dominance hierarchy can lead to erroneous interpretation of factors contributing to changes in subordinate species abundance. Weakfish landings (1929-2006; Figure C10.8-1), GDR biomass (Figure C10.8-1), and DE PSD Q+ (see Figure 1 in Appendix C-2) maintained distinct high or low levels related to striped bass and R_t. Changes in abundance of dominant species are positively correlated with environmental factors that improve survival, while abundance of subordinate species depends on density of the dominant species (Skud 1982). Most variation in year-class strength of striped bass in Chesapeake Bay (the largest spawning area on the Atlantic Coast) can be explained by freshwater flow and temperature during March through May (Rutherford and Houde 1995; North et al. 2005) although recent high biomass and trophic demand also reflect imposition of conservative management measures (Richards and Rago 1999; Uphoff 2003). Variation in weakfish abundance is strongly related to striped bass and its success in feeding on menhaden (indexed through R_t). An assumption of constant M ignores dominance hierarchy and attributes all changes in weakfish biomass to fishing, which would grossly overestimate the potential for recovery by managing the weakfish fishery alone.

More complex fisheries ecosystem models of other species complexes in other geographic regions (Ecopath with Ecosim or EWE) have highlighted processes similar to those displayed by the GDR. Large dominant fishes may be successful due partly to "cultivation effects", where adults crop down forage species that are potential competitors of their own juveniles (striped bass cropping weakfish in this case; Walters and Kitchell 2001). Dominance is a result of not only being able to acquire resources but also by insuring the best possible trophic conditions for the dominant species' young. This hierarchy leads to the risk of persistent depensatory effects (low juvenile survival) that develop with some time lag following periods of adult stock depletion (Walters and Kitchell 2001). EWE modeling of multiple systems has indicated that trophic interactions in combination with widespread fishing at F_{msy} may cause considerable change in community structure and MSY than those predicted from single-species assessments (Walters et al. 2005). The most critical predictions of change in juvenile mortality rates from EWE simulations of the Gulf of Mexico ecosystem arose when abundant predators caused high mortality of prey without those rates being represented as high proportions of predator diets (Walters et al. 2008).

Experience with Northwest Atlantic Cod *Gadus morhua* provides a case history to consider when contemplating weakfish biomass dynamics presented here. Excessive fishing mortality was the predominant factor in depletion of these stocks of cod and led to their collapse by the early 1990s (Shelton et al. 2006). Despite imposition of severe catch restrictions for over a decade, stocks largely failed to recover at predicted rates due to declines in productivity. Factors contributing to this decline in productivity (in rank order) were increased natural mortality, decreased body growth, reduced recruitment was implicated in a few cases, and continued fishing and bycatch were important (Shelton et al. 2006). An increase in M is now routinely incorporated into several cod stock assessments (Chouinard et al. 2005). Changes in estimated M in southern Gulf of St. Lawrence cod matched fluctuations of in grey seal abundance, although some inconsistencies in seal diet data

conflicted with this hypothesis (Chouinard et al. 2005). Productivity of this stock of cod has declined to the point where the population is no longer viable even in the absence of fishing (Swain and Chouinard 2008).

C11.0 Review evidence for constant or recent systematic changes in natural mortality, productivity, and/or unreported removals. (TOR #7)

C11.1 Introduction

A period of stock rebuilding appears to have occurred during the early 1990s, as evidenced by expansion of the population age structure, increases in commercial and recreational harvest and CPUE, and positive trends in abundance and biomass in subsequent stock assessments. However, during development of the 2006 weakfish stock assessment (ASMFC 2006a, Part A), the WTC was confronted with anomalous patterns in weakfish abundance estimates and modeling results. Preliminary runs of age structured and simple biomass models indicated that the stock had rebuilt to record levels. In contrast, observed size and age structure had diminished, CPUE and harvest were down, and fishermen were complaining about lack of fish. Management measures had not changed, and there was no evidence of increased harvest or discard rates. Discrepancies in trends of fishing mortality estimated by VPA and relative F analysis provided further evidence that the modeling results were suspect. The WTC searched for possible causes of model uncertainty and the unexpected reversal in stock growth. Additional modeling exercises, including biomass modeling with environmental and ecological covariates, suggested that the anomalous pattern was due to an incorrect assumption of constant natural mortality, and that stock declines were a result of increased interaction with principal predator/competitor species (ASMFC 2006a, Parts A and C).

Results of the 2006 stock assessment were vetted through the ASMFC external peer review process (ASMFC 2006a, Part B). The peer review panel did not accept the WTC's evaluation of the stock, suggesting that the age structured modeling could be improved by addressing concerns with the input data. These issues were addressed as well as possible by the WTC prior to and during the development of the 2009 stock assessment. Regardless, preliminary age structured and simple biomass modeling attempts still produce unrealistic results. As such, biomass modeling with ecological and environmental covariates was again pursued, with updates and improvements made over the previous (2006) methods.

An interim review of the current stock assessment as a work in progress was conducted during the 2008 DPSW in Woods Hole, MA. Modeling results presented during the DPSW indicate that fishing mortality is a minor component of total mortality, suggesting trophic or environmental forces as major influences on weakfish stock dynamics. The reviewers had similar comments as the 2006 panel regarding the advanced modeling efforts, citing concerns with input data and lack of empirical evidence to support the predation/competition hypotheses (Miller et al 2009). Although the panel appears to agree that weakfish populations have decreased, they contend that any number of sources could have been the driving force of the decline, such as environmental factors or unreported/underestimated removals. The following is a review of a suite of analyses searching for evidence of declines in productivity due to non-fishing sources.

C11.2 Review of previous findings

C11.2.1 Relative F

By the early 1990s, harvest (total removals) had fallen to less than half the levels seen during most of the previous decade (Figure C11.2-1). As a result (presumably), biomass began a period of rebuilding around 1993. During the mid to late 1990s, harvest was increasing slightly, but with the increase in stock size, F rates had fallen to all time low levels between 1995 and 1997. Regardless, a dramatic drop in biomass was observed between 1997 and 1998, and the decline has continued to recent years. Harvest levels remained relatively stable and were similar to those that had presumably led to stock increases earlier in the decade, but now were causing a rapid stock decline. As biomass declined, predicted trends in fishing mortality from age structured modeling (ADAPT VPA) indicated a continuous, almost exponential increase in fishing mortality between 1995 and 2002, regardless of tuning indices used. (Trends after 2002 are unreliable due to retrospective patterns.) The WTC expressed concern that trends in F from the VPA might be influenced by the assumption of constant natural mortality, M. Relative F analysis was considered as an alternative method to calculate trends in fishing mortality. Relative F evaluates the stock as a whole, and is not influenced by changes in age structure. Some of the benefits of relative F are that it is simple and intuitive, easy to calculate, and not constrained by assumptions about natural mortality. The following is a summary of the relative F analysis presented in section **C8.0** of this report.

Relative F is calculated as

$$F_{rel} = \frac{H_t}{0.5 * (I_t + I_{t+1})}$$

where H_t is total harvest in year t, and I are index values of abundance in year t and $t+1$. Total harvest biomass was estimated as the sum of commercial harvest from state and NMFS reporting programs, commercial discards using the methods of de Silva (2004), recreational harvest (A+B1) from MRFSS, and 10% of total recreational discards (from MRFSS) multiplied by the MRFSS estimated average weight of discarded fish of 0.15 kg. Three surveys – MRFSS 1+ CPUE, New Jersey Ocean Trawl Survey positive tows, and Delaware DFW 30-foot Delaware Bay Trawl Survey – were standardized to a common scale and averaged to develop a single index of weakfish abundance. Additional information on these data and their sources are presented in sections **C5.0** (harvest) and **C6.0** (indices).

In order to directly compare relative F estimates with estimates derived from the VPA, the relative F time series was rescaled using a span of years during the converged portion of the VPA which is robust to the number and source of tuning indices used. This method assumes relative stability of M, which is one of the main concerns of the WTC regarding ADAPT; however, the assumption is only applied for a short time period (5 years) which is much more realistic than M remaining constant over decades.

To rescale the relative F vector, the ratio between F_{VPA} (biomass weighted 1+) and F_{rel} was found for the years 1982-1985. The average ratio for these years was computed and applied to F_{rel} for all years in the time series.

Comparison of trends in rescaled F_{rel} and F_{VPA} (biomass weighted ages 1+) show similar patterns between 1982 and 1998 (Figure C11.2-2). Fishing mortality was relatively high during the

1980s, generally ranging from 0.7 to 1.3. In the early 1990s, both trends begin a gradual decline to time series lows of 0.23 (F_{VPA}) and 0.35 (F_{rel}) in 1995 before gradually increasing to between 0.5 and 0.6 by 1999. Beginning in 1999, the trends differ greatly. F_{rel} exhibits relative stability, generally ranging between 0.4 and 0.6 until the end of the time series. In contrast, F_{VPA} continues to increase rapidly, reaching a time series high of 1.57 in 2003. F_{VPA} declines after 2003; however a strong retrospective pattern (Figure C7.3-2) suggests F_{VPA} estimates in recent years are greatly underestimated.

Harvest and abundance data used for the VPA and relative F analyses are essentially identical. Trajectories of biomass harvested and biomass indices used for relative F are strongly correlated with the corresponding age aggregated numbers based inputs for the VPA. Identical input data and divergent results suggest violation of model assumptions or mis-specification of one of the models. An underlying assumption of the ADAPT model is that catch at age is known without error. Section **C5.0** of this document identifies several sources of uncertainty with the overall catch at age, which likely lead to this assumption being violated through inaccurate harvest information and/or poorly specified age distribution. Error in the age structure would not be expected to produce such divergent trends in fishing mortality from the two methods. Inaccurate harvest information could produce the observed discrepancy, but only if there was a systematic increase in the amount of unrecorded landings. Several recent peer reviews (ASMFC 2006a, Part B; Miller et al 2009) suggest unreported landings could be contributing to the anomalous patterns. In an attempt to quantify the amount of unrecorded removals required to mimic trends in total mortality observed in the VPA, two alternative applications of the "forage model" were run using estimates of unrecorded removals that were constant or increasing over time (see section **C10.0** of this report). A constant level of approximately 3,500 MT of additional removals since 1995, or an increase from 2,300 to 5,000 MT additional removals between 1995 and 2007 both resulted in strong model fits ($R^2 \sim 0.90$; see Figure C10.7-1); however, neither performed as well as the "best" model fit which incorporated menhaden and striped bass abundance. In addition, the implications of these unrecorded removal scenarios are drastic. Total recorded removals in 2007 were less than 1,000 MT while estimated unrecorded removals were three to five times greater, suggesting our understanding of fleet dynamics is minimal and reporting requirements are ineffective. Also, the 2006 and DPSW review panels suggested the increases could be due to additional discarding. Such large increases in discarding rates following implementation of management measures imply a complete failure of management to constrain removals.

Another possible cause for the discrepancy in results between the VPA and relative F could be misspecification of input data. Without direct evidence to the contrary, natural mortality is often considered constant throughout time. This simplistic assumption may hold over short time periods, but is likely unrealistic over decades. Regardless, the baseline VPA model runs investigated for this and all previous stock assessments assumed constant natural mortality in weakfish of M = 0.25. ADAPT estimation procedures incorporate the input natural mortality rate, and model results are therefore influenced by the rate selected. Incorrect assumptions in the input M matrix would lead to inaccurate estimates of population parameters by the ADAPT model. In contrast, relative F calculations are independent of any assumptions regarding natural mortality rate. Discrepancies in F trends calculated using the two methods may indicate invalid assumptions regarding M.

C11.2.2 Biomass modeling

Due to concerns over the uncertainty in recent years' estimates from age structured modeling (*i.e.* retrospective patterns), as well as concerns regarding the assumptions upon which the ADAPT

runs were constructed (in particular, constant M and error free CAA), the WTC investigated biomass dynamic modeling. Methods are described in detail in sections **C9.0** and **C10.0**, and are summarized below.

Two similar but alternative models were developed to investigate both simple and extended biomass models. Three indices of weakfish abundance (two fishery independent and one fishery dependent) were standardized and combined into a single composite index to depict trends in biomass. Simple models used basic (Schaefer and/or Gompertz) formulations of surplus production models with fishing as the only source of mortality, while the extended models incorporated ecological and environmental covariates as well. One model (the "predation hypothesis") investigated trends in potential weakfish predators as well as several environmental variables. In contrast, the "forage model" investigated links between weakfish abundance and trends in potential predators/competitors and forage species. Suitable covariates for inclusion in the extended models were selected using correlation analysis and stepwise regression. For a particular covariate to be retained for further analysis, statistical results had to be significant and logical (*e.g.* negative correlation for predator, positive correlation for forage). The predation model implemented a Type III predator-prey functional response (Steele-Henderson), while the forage model investigated a suite of response functions. "Best" models were selected using model fits, parameter estimation, trends in residuals, and/or Akaike Information Criterion (AIC).

Detailed results of the predation hypothesis are presented in section **C9.0**, while the forage hypothesis is discussed in detail in section **C10.0**. The following is a summary of findings for the two models.

Predation model

Simple biomass dynamic modeling using the Gompertz formulation produced significant parameter estimates and strong model fit ($r^2 = 0.68$). However, uncertainty around parameter estimates was moderate, and a prominent systematic trend in residuals suggests process error (missing terms) in the population model (Figure C9.3-8).

Correlation analysis and stepwise regression were used to investigate potential links between key environmental and ecological covariates and trends in weakfish abundance. Results of the statistical analysis indicate that indices of abundance based on recreational CPUE for two key predator species (striped bass and spiny dogfish) are negatively correlated with weakfish biomass (Table C9.3-1). Including these indices in a biomass dynamic model with a Type III functional response increased model fit ($r^2 = 0.94$) and greatly reduced uncertainty in parameter estimates (Table C9.3-6). Additionally, residuals from the run including predation appear much more random (Figure C9.3-9).

Forage model

Statistical analyses identified significant correlations between weakfish and both striped bass abundance and menhaden abundance that might influence weakfish stock dynamics. These terms were included in biomass dynamic modeling that evaluated six different mortality hypotheses (fishing only, 3 functional responses, 2 depensatory responses) in both Schaefer and Gompertz formulations, for a total of 12 competing models. AIC showed that the model including fishing mortality, striped bass abundance, and a depensatory response to the ratio of menhaden to striped bass performed best, while the two models with fishing mortality only ranked 7th and 9th out of 12.

Results of these two models (predation hypothesis and forage hypothesis) suggest that a simple biomass dynamic model based on trends in weakfish abundance and harvest is insufficient for

characterizing population dynamics. Inclusion of predation terms greatly improves model results, providing trends that more accurately reflect field observations, and strongly support a systematic increase in M in the past decade or more. Comparison of Mp from the predation model and M calculated from $Z_{VPA} - F_{rel}$ show a very similar pattern (Figure C11.2-3). In addition, the fishing only models produce population parameters that are unrealistic for a species with a life history such as weakfish. Weakfish grow quickly and mature at a young age, suggesting that the population growth parameter, r, would be relatively high. The Gompertz model when only fishing mortality is assumed estimates r = 0.26 and a carrying capacity (K) over 150,000 MT (Table C9.3-3). This relatively slow growth rate and extreme carrying capacity seem unlikely for a species that exhibits rapid growth and early maturation. Including ecological covariates in the biomass models results in r and K estimates which are much more realistic for a species like weakfish.

It has been argued that spatial and temporal overlap of striped bass and weakfish are limited to the fall during southward migrations, and therefore opportunity for predation events is uncommon. The WTC recognizes that weakfish are not a principal prey item of striped bass. A comparison of estimated predation mortality and striped bass bioenergetic data indicates weakfish are a minor component of striped bass diet (see section **C10.6.3**). Regardless, with the dramatic increase in striped bass abundance over the last decade, even low consumption rates would lead to substantial increases in weakfish predation mortality.

C11.3 Additional analyses to investigate changes in productivity

C11.3.1 Environmental

North Atlantic sea surface temperatures have been found to exhibit a 65 to 70 year oscillation (Schlesinger and Ramankutty 1994; Enfield et al 2001). Kerr (2000) termed this oscillation the Atlantic Multidecadal Oscillation (AMO) to distinguish it from the atmospheric North Atlantic Oscillation (NAO). Since its discovery, the AMO has been linked to the occurrence and severity of climatic events throughout the northern hemisphere such as rainfall/drought, severity of Atlantic hurricanes, and North American and European summer climate (Enfield et al 2001). A concise, albeit "unofficial," review of the AMO is available from Wikipedia, the open access online encyclopedia (*http://en.wikipedia.org/wiki/*).

Data for the AMO are available from the National Oceanic and Atmospheric Administration (NOAA) Earth System Research Laboratory at *http://www.cdc.noaa.gov/data/timeseries/AMO* as both an annual index (1948 to 2007) and a 10-year moving average index (1861 to 2002). The longest official time series available for weakfish consists of reported commercial harvest from 1950 to 2007. Although many factors affect reported harvest (*e.g.* abundance, effort, gear selectivity, data collection methods), the harvest time series was considered proxy data for weakfish abundance. Trends in the smoothed AMO index were compared to commercial weakfish harvest estimates from 1950 to 2002. Pearson correlation was conducted using SAS Proc Corr (SAS 1990). To evaluate potential temporal delays in population response, correlations were conducted with the harvest time series lagged backward 0 to 10 years.

Using reported weakfish commercial harvest as proxy data for weakfish abundance, graphical analysis suggests that weakfish abundance increases when the AMO is in a negative phase (Figure C11.3-1). In 1950, the AMO was in a positive phase, but began a gradual decline in the mid 1950s, crossing into a negative phase in the mid 1960s, and continuing to a relative minimum a decade later. The AMO index then increased steadily until the early 1980s, leveled off for a number of years, before resuming an increasing trend in 1990. The index entered a positive phase around

1994, and continued to increase through 2002 (the last year available for the 10 year average). In contrast, weakfish landings, and presumably abundance, were low from 1950 to the 1960s. Rapid increases in harvest began in 1970 while the AMO was in a negative phase. Shortly after the AMO reached a minimum, landings peaked and rapidly dropped off. As the AMO entered the recent positive phase, landings have dropped to record low levels.

Correlations between the 10 year moving average AMO index and commercial weakfish landings were negative and highly significant ($P < 0.001$) for all comparisons (Table C11.3-1). The strongest association ($r = -0.83$) was found when a five year time lag was incorporated. Peaks in commercial landings and record numbers of citation size fish (Uphoff 2004) are reported during the late 1970s and early 1980s when the AMO was at its lowest point. Despite commencing an upward trend in the mid 1970s, a slight stabilization of the index coincided (when a five year lag is incorporated) with management measures, perhaps boosting their effectiveness. The index resumed a positive trend, and the transition from a negative to a positive phase occurred around the time of the unexpected reversal in weakfish stock growth.

The longest official index available for weakfish is the NMFS commercial harvest record extending from 1950 to present. However, Joseph (1972) provides a landings history back to 1929. Correlation analysis with the longer time series provides weaker, but still significant negative relationships between AMO and weakfish landings. The strongest correlations ($r \sim -0.38$, $P < 0.01$) occur with a 7 to 9 year time lag (Table C11.3-1). Between 1929 and 1935, AMO was increasing in a positive phase after which it stabilized. Landings during this period were decreasing, and reached relative low levels a short time after the AMO stabilized. Landings began to rebound shortly after the AMO began a declining trend around 1959.

Quantitative analysis is not possible prior to 1929, but a qualitative analysis suggests the correlations hold for earlier periods as well. Collette and Klein-MacPhee (2002) indicate periods of high landings in southern New England and into Cape Cod Bay during the early part of the 20th century. This harvest boom occurred as the AMO switched from a positive to negative phase, and during portions of the next 3 decades while the AMO remained negative. Landings from this region were not consistently high throughout this period, but were alternately abundant and scarce. This would not be unexpected for a species fished heavily near the extent of its range. High fishing pressure would diminish the stock, but during a period of high productivity, populations could rebuild quickly.

Crecco (2009) investigated localized temperature variation but found no correlation with weakfish abundance. Water temperature data from other New England stations (*e.g.* Oviatt 2004, Nixon et al 2004) show similar patterns to that observed by Crecco, but with localized differences in magnitude and duration of anomalies. The AMO incorporates data from the entire North Atlantic which may be more representative for a stock distributed over a wide range. In addition, Crecco evaluated correlations in temperature and abundance over a shorter time scale and without a smoothing function. Interannual variability in temperature, particularly when confined to a single site, may partially mask long term trends, making correlations harder to detect.

Decadal scale influence of climate and other variables on marine populations has been gaining recognition in the last two decades (*cf* Hare and Able 2007, Introduction pp 31-32). In some recent examples, Oviatt (2004) found that increased winter water temperatures led to a decline in boreal demersal species such and winter flounder (*Pseudopleuronectes americanus*) and windowpane flounder (*Scophthalmus aquosus*), followed closely by an increase in demersal decapod crustaceans. Increased winter temperatures were also strongly correlated with increases in abundance of grey snapper (*Lutjanus griseus*) in Texas (Tolan and Fisher 2007). Perhaps most

relevant to weakfish, Hare and Able (2007) identify a link between winter water temperature and juvenile abundance of Atlantic croaker (*Micropogonias undulatus*), another member of the scianid family. The authors propose a conceptual model whereby above average winter water temperatures result in high juvenile survival which leads to "outbursts" in Atlantic croaker populations. These studies provide evidence that productivity and stock dynamics can be significantly influenced by environmental factors such as water temperature. It is recognized that factors other than abundance influence harvest levels, but strong correlations and qualitative associations between weakfish harvest and the AMO over the last century provide support that oceanic conditions may have a significant influence on weakfish stock dynamics and may help explain the unexpected decline in productivity over the last decade. It is not possible from the available data to determine whether oceanic conditions are acting on weakfish directly (*e.g.* recruitment strength), indirectly (*e.g.* changes in predator and/or prey abundance), or a combination of both.

C11.3.2 Food habits

The NEFSC Food Habits Database contains information on type and prevalence of prey items for key species, obtained by analysis of stomach contents collected during seasonal trawl surveys. Preliminary review of the database showed that fall surveys from 1991 to present have used consistent methodology and have collected sufficient sample size of weakfish. Prey types were grouped into broad diet categories (amphipod/isopod, crab/shrimp, forage fish, empty, other), and the annual percent composition in all observed stomachs was calculated for each prey category. Three year moving averages were calculated to smooth out interannual variation in prevalence, and SAS Proc Corr (SAS 1990) was used to evaluate correlations between prey prevalence and trends in total mortality (biomass weighted, ages 1-5) estimated by ADAPT VPA. These correlations were conducted for 1-inch size intervals from 5" to 12", and for all sizes combined, for the years 1991 (first year of food habits data) to 2002 (most recent year of VPA results without retrospective pattern).

Negative correlations with Z_{VPA} were generally observed with percent composition of crab/shrimp and forage fish, while the percent composition of amphipod/isopod and the prevalence of empty stomachs were positively correlated (Table C11.3-2; Figure C11.3-2). More than half of these correlations were strong ($P < 0.10$), and only the 12" category was not significantly correlated. The "empty" category produced the greatest number of significant correlations (6 out of 9 size groups) and often produced the strongest correlations with Z. One interpretation of these data is that primary prey items (forage fish and large invertebrates) became scare, so weakfish shifted to other, less optimal prey (small invertebrates) or were unable to find suitable forage (empty stomachs).

The 2008 DPSW review panel expressed concern over the lack of empirical data to support the hypothesis that weakfish productivity has been compromised since the late 1990s. The data presented above provide direct empirical evidence that weakfish forage opportunities may have diminished during the mid to late 1990s. Decreased forage abundance and/or sub-optimal prey types could affect weakfish productivity through increased natural mortality or changes to other parameters (*e.g.* growth rates). Empirical data are also provided through a comparison of two studies on weakfish diet composition in Chesapeake Bay (Hartman and Brandt 1995; R. Latour, Virginia Institute of Marine Science, pers. comm.). The studies occurred approximately a decade apart on either side of the time period during which natural mortality was suspected to have increased. These studies indicate shifts in diet composition (Figure C11.3-3) that are consistent with those observed in the NEFSC food habits database. Weakfish initiate piscivory at an early age, beginning with bay anchovy at age 0 and moving to larger prey such as menhaden and spot by age 1

to 2. In Chesapeake Bay, Hartman and Brandt (1995) indicated that greater than 80% of weakfish diet from age 0 to 2 consisted of these principal prey items, with the majority of the remainder (5 to 20%) made up of invertebrates (Figure C11.3-3). In contrast, by 2002, 30% or more of weakfish diets age 0 to 5 consisted of invertebrate species, with forage fish accounting for less than 40% of the total (R. Latour, Virginia Institute of Marine Science, pers. comm.). Bay anchovy and Atlantic menhaden have become much less frequent in weakfish diets between the early 1990s and early 2000s, spot have disappeared, and invertebrates have comprised a greater part of the diet. During both periods, weakfish made a rapid transition to piscivory - bay anchovy dominated their diets as young of year. However, transition from bay anchovy to dominance of spot and Atlantic menhaden at age 2 that occurred during the early 1990s was not evident in recent years. Older weakfish (ages 2+) are subsisting on the same invertebrate and bay anchovy diet as ages 0-1, with a much smaller supplement of menhaden (Figure C11.3-3). Striped bass diets in Chesapeake Bay have undergone similar changes as those described for weakfish; small striped bass are now relying more on invertebrate prey and large striped bass on small pelagic prey such as bay anchovy and age 0 clupeids, and benthic invertebrates (Griffin and Margraf 2003; Overton 2003). This suggests that larger prey (juvenile menhaden and spot) were insufficiently available for both species to complete the transition to piscivory.

Early switching to a fish diet indicates that weakfish is a specialized piscivore (Persson and Brönmark 2002). Early switching requires high growth rate, which implies high densities of proper forage and safe foraging opportunities. Species undergoing ontogenetic diet shifts face a risk of delayed transitions among feeding stages if food resources are limited and competition is intense (Persson and Brönmark 2002). If unfavorable conditions prevail, then growth is reduced, often leading to ontogenetic bottlenecks where individuals fail to gain size advantage on larger prey items in the next stage. Unfavorable conditions can also result in increased vulnerability to competitors and predators. Competing individuals and predators may hinder one another's feeding activities, leading to starvation or they may eat one another (including cannibalism; Yodzis 1994). The two Chesapeake Bay studies and the NEFSC food habits database provide evidence that weakfish forage opportunities diminished during the late 1990s. Shifts in diet composition may have resulted in decreased productivity of the stock through decreased growth rates and/or increased competitive or predatory mortality.

C11.3.3 Size at Age

Another possible indicator of changing productivity would be changes in size at age over time. Preliminary investigations were conducted using separate time series of scale- and otolith-based age data, but are not presented here. Although some data produce significant trends (both positive and negative) in size at age during the time series, the WTC was concerned that the availability of samples could be affecting the results. Due to geographic differences in size (*e.g.* between southern and northern Mid-Atlantic states), annual differences in the source and sample size by source can affect estimated average size when evaluated on a regional or stockwide basis. Additionally, no single data source had a consistent time series of sufficient sample size at age to evaluate on an individual basis. The WTC intends to continue investigating potential changes in size at age by identifying alternative data sources and analytical methods.

In addition, trends in size (weight) at age were used to evaluate trends in natural morality (Lorenzen 1996). Preliminary investigations indicate that trends in natural mortality are well correlated with VPA-based estimates of total mortality. However, as mentioned above, the WTC is concerned that trends in weight at age may be affected by interannual variability in the location and

intensity of sampling, so formal results are not presented in this report.

C11.4 Discussion

Following the 2006 stock assessment, the WTC concluded that fishing mortality for weakfish had remained stable at low levels in recent years, but that the population was near all time low levels due to increased predation/competition with striped bass. These conclusions received criticism from the 2006 peer review panel and the 2008 DPSW review panel. The panels identified concerns with input data and lack of empirical data to support the conclusions. The DPSW panel expressed the opinion that any number of factors could have contributed to the trends in fishing mortality observed from the VPA, such as unreported harvest, changing fishing practices, or environmental/ecological influences other than predation. Responses to concerns regarding input data are discussed elsewhere in this report (*e.g.* sections **C5.0** and **C6.0**). This section of the report investigates a variety of data types and sources that might support or discount the hypothesis that weakfish productivity has declined.

The WTC recognizes that population dynamics of any species are influenced by a wide variety of anthropogenic, environmental, and ecological forces, both direct and indirect. As these forces work in concert, enhancing or diminishing the effects of each other, it is not possible to predict the magnitude or direction of their cumulative impact. This section is not meant to be a comprehensive review of all the possible influences on the weakfish stock, but attempts to investigate some of the more direct factors. The overwhelming majority of data types investigated lend support to the WTC's evaluation that factors other than fishing mortality were influencing stock dynamics.

- During the period 1995 to 2003, total mortality (biomass weighted Z_{VPA} ages 1 to 5) increased by 375% from 0.48 to 1.82 (Figure C11.3-2)
- During the period 1995 to 2002, F_{rel} estimates, which are independent of natural mortality, remained relatively stable and were much lower than F_{VPA} estimates based on assumptions of known constant M. (Figure C11.2-2)
- Biomass models that incorporated ecological interactions with key predators/competitors and forage species resulted in improved model fits and more realistic population parameters than models based on fishing mortality alone. (Tables C9.3-3; compare Figures C9.3-8 and C9.3-9).
- Estimates of unrecorded removals necessary to replicate F_{VPA} trends were unrealistically high and imply complete management failure. (Figure C10.7-1)
- Trends in north Atlantic sea surface temperature tracked through the AMO are strongly correlated with weakfish harvest trends over the last century. (Figure C11.3-1)
- Two independent studies identify shifts in prey types during the 1990s. (Table C11.3-1; Figures C11.3-3)
- The incidence of empty weakfish stomachs observed in the NEFSC food habits database correlates strongly with the increase in total mortality (Z_{VPA}) as fishing mortality (F_{rel}) remains stable (Figure C11.3-2)

The WTC admits that the conclusion in the 2006 stock assessment that weakfish declined solely as a result of striped bass predation may have been overly myopic; however, given the additional evidence presented in this section, the WTC still contends that productivity of the weakfish stock was compromised beginning in the mid to late 1990s due to one or more of the factors discussed

above. Potential changes in productivity include decreased growth rates, starvation, competition, or increased vulnerability to predation, among others. Regardless of the source, the overall effect was a decline in weakfish stock biomass which analyses suggest was not attributable to fishing pressure.

C12.0 Estimate biological reference points using equilibrium and non-equilibrium assumptions and evaluate stock status relative to these BRPs. (TOR #8)

C12.1 Thompson-Bell reference points

Weakfish are currently managed relative to reference points developed under Amendment 4 of the Weakfish Fishery Management Plan. Reference points were updated during the 2002 assessment (Kahn 2002) using a spreadsheet based model, but the updated reference points were not adopted because it was not a benchmark stock assessment. For the current benchmark assessment, reference points have been calculated using the spreadsheet model. The model requires input values for partial recruitment at age, natural mortality, and age specific maturity. Weight at age is calculated using relationship described by Vaughan 1999. SSB is calculated as the summed product of number and weight at age. Fishing mortality reference points were found by solving for F rates that provided spawning stock biomass of 30% (target) and 20% (threshold) relative to unfished stock (J. Uphoff, MD DNR, pers comm.).

Reference points established under Amendment 4 assumed equilibrium conditions with a constant natural mortality rate of $M = 0.25$. Implementing the spreadsheet method described above and updating partial recruitment values to the average of the most recent three years (2005-2007) allows direct comparison of reference points during the two periods. Amendment IV established new reference points for fishing mortality target of $F_{target} = F_{30\%} = 0.31$, and a fishing mortality threshold of $F_{threshold} = F_{20\%} = 0.5$. Updating partial recruitment to the average of 2005-2007 decreases reference points estimates to $F_{30\%} = 0.28$ and $F_{20\%} = 0.42$. The Shepherd spawner–recruit function was also updated using recent data, providing a new SSB threshold estimate of $SSB_{20\%} = 13,108$ MT. Comparison of VPA based parameter estimates indicates that fishing mortality on weakfish has been above the equilibrium target of $F_{30\%}$ in all years since 1981, and above the $F_{20\%}$ threshold in every year except 1995 (Figure C12.1-1). SSB has exceeded the $SSB_{20\%}$ threshold during only two time periods: 1986-1987 and 1994-1997. SSB_{2007} is approximately 55% of the threshold, but is likely overestimated based on the observed retrospective pattern.

Reference point estimates are conditional on the input values, such as partial recruitment at age. Given the pronounced retrospective pattern that overestimates abundance at age (see section **C7.0**), estimates of partial recruitment in recent years are likely to be underestimated. Consequently, fishing mortality reference points are likely to be overestimated. The retrospective pattern diminishes to near zero in approximately five years, so using partial recruitment vectors from an earlier time period may provide more stable estimates of reference points. Using average partial recruitment at age from 1999 to 2001 provides estimates of a fishing mortality target of $F_{30\%} = 0.25$, and a fishing mortality threshold of $F_{20\%} = 0.36$. The associated SSB threshold is estimated at $SSB_{20\%} = 10,179$ MT.

Another factor that could affect the reference point estimates is the assumption of constant natural mortality. The WTC has expressed concern regarding the assumption that M has remained constant throughout the time period analyzed. Recent analyses indicate that natural mortality may have increased dramatically over the last decade, to levels as high as $M = 1.0$ or higher. Reference points were also calculated using a natural mortality rate of $M = 0.8$. Using average partial

recruitment at age from 1999 to 2001 provides estimates of a fishing mortality target of $F_{30\%} = 0.78$, and a fishing mortality threshold of $F_{20\%} = 1.20$. Under these assumptions, fishing mortality on weakfish declined below the threshold for most years between 1989 and 2001, and was below the $F_{30\%}$ target between 1995 and 1999. F_{2007} is below the target mortality rate, but is likely underestimated given the observed retrospective pattern.

As noted above, reference points assume natural mortality is constant over time. Sensitivity of reference points can be investigated under different assumptions of constant natural mortality, but more advanced techniques are required to investigate the influence of systematic changes in M. Both of the biomass dynamic models investigated during this assessment produced estimates of both equilibrium and non-equilibrium reference points, and are discussed below.

C12.2 Reference points calculated by the "Predation Hypothesis"

The following is a summary of the reference point and stock status analyses conducted under the predation hypothesis. Additional details are presented in the full discussion of the predation hypothesis in section **C8.0** of this report.

C12.2.1 Equilibrium conditions

Updated F_{msy} and B_{msy} thresholds were estimated for weakfish using the dynamic Gompertz external surplus production model (Quinn and Deriso 1999; Jacobson et al 2002). Surplus production estimates from 1981-2008 were regressed against weakfish biomass (Biowt) and the product of the log weakfish biomass and biomass (LogBiowt*Biowt) in a two variable linear regression model without a y-axis intercept (see equation (6) in section **C8.0**) using the ROBUSTREG procedure contained in the Statistical Analysis System (SAS 2002). Equilibrium F_{MSY} threshold is expressed by the intrinsic growth rate (r) parameter, whereas B_{MSY} is expressed by the carrying capacity (K) divided by 2.72 (Quinn and Deriso 1999). Since temporal shifts in striped bass predation are absent from the traditional dynamic models, the overfishing definitions (F_{msy}, B_{msy}) in these models are fixed in time.

The analysis was conducted for the two scenarios of fishing mortality only and fishing mortality plus predation. The "fishing only" model provided a reasonably good fit (r**2 = 0.54) to the biomass time series with statistically significant (P < 0.05) r and K parameter estimates (Table C9.3-3). The resulting overfishing thresholds were $F_{MSY} = 0.26$ and $B_{MSY} = 57,388$ MT. However, a distinct serial pattern was present in the residuals, and the model consistently over predicted weakfish surplus production from 1998 to 2008 by 20% to 350% (Figure C9.3-3). This systematic trend in the direction of the residuals usually indicates that the model is lacking an informative parameter(s).

Addition of a predation term resulted in greatly improved model fit (r**2 = 0.84), and the parameters (r, K, C) were estimated with much higher precision than the production model without predation (Table C9.3-3). Resulting equilibrium overfishing thresholds (Fmsy, Bmsy) from the extended Gompertz model were 0.54 and 19,289 mt, respectively (Table C9.3-3). Moreover, the strong serial correlation in the residuals noted from the basic Gompertz model was minimized when predation effects of striped bass and spiny dogfish were included (Figure C9.3-4).

Within the predation hypothesis, reference points were also calculated using the discrete Gompertz model with and without predation (see equations 12 and 11 of section **C9.0**) using iterative reweighting. The discrete model with predation is referred to in this report as the Steele-Henderson (S-H) model. As above, the equilibrium F_{MSY} threshold is expressed by the intrinsic

growth rate (r) parameter, and B_{MSY} is expressed by the carrying capacity (K) divided by 2.72 (Quinn and Deriso 1999).

When only fishing effects are evaluated, this model provided a good fit ($r^{**}2 = 0.82$) to the biomass data with statistically significant ($P < 0.05$) r and K parameter estimates. Overfishing thresholds were calculated as $F_{MSY} = 0.32$ and $B_{MSY} = 25,259$ MT. As with the external model, the discrete model with fishing effects only consistently over predicted weakfish biomass from 1998 to 2008 by 10 to 250% (Figure C9.3-8), indicating that the basic Gompertz model was plagued by substantial process error.

The full Steele-Henderson (S-H) production model fit with iterative reweighting provided a very good fit ($r^{**}2 = 0.94$) to the biomass and predation data with statistically significant ($P < 0.05$) r, K, c and A parameter estimates (Table C9.3-6). Overall, the S-H models was not only a better fit to weakfish biomass than the Gompertz model with only fishing effects, but the r and K parameters were estimated with much higher precision. The resulting weakfish equilibrium overfishing thresholds were estimated at $F_{MSY} = 0.72$ and $B_{MSY} = 17,009$ mt (Table C9.3-6). Moreover, unlike the severe residual pattern evident from 1998 to 2008 based on the basic Gompertz models, there is little if any systematic residual pattern from the S-H model fitted by iterative re-weighting (Figure C9.3-9).

Given the clear residual problem associated with the Gompertz models estimated without predation, these models were not used to evaluate weakfish stock status. Relationships between parameter estimates and reference points evaluated through the Steele-Henderson model are shown in Figures C9.3-13 (fishing mortality) and C9.3-14 (biomass). Fishing mortality fell below the F_{MSY} = 0.72 1993 and has remained below the threshold in all years except 2002. SSB fell below B_{MSY} = 17,009 MT threshold in 1983, falling to less that one-third the threshold in 1990. Biomass subsequently rebounded to more than 80% of the threshold in 1995 and 1996, but has since declined.

C12.2.2 Non-equilibrium conditions

The non-equilibrium S-H model allows the direct calculation of the amount of mortality attributable to predation, M_p. Because predation effects are transient over time, weakfish surplus production and predation-induced mortality can vary greatly across years, resulting in time varying F_{msy} and B_{msy} thresholds. The degree of temporal variation in F_{msy} and B_{msy} depends on the magnitude and trend in predator abundance, the predator consumption exponent (c) and on the prey stock size (A) at which the consumption threshold of predation takes place in equation (11). Thus, the annual F_{MSY} values from the S-H model are not fixed in time but rather are a function of the fixed intrinsic rate (r) and the time varying predator consumption rate. Similarly, weakfish biomass threshold (B_{MSY}) can vary over time depending on the amount of weakfish biomass consumed annually (see equations 16 and 17 in section **C9.0**).

Calculation of non-equilibrium reference points indicates that F_{MSY} has decreased substantially from above 0.75 on the early 1980s to less than 0.25 in most years since 2001 (Table C9.3-5). Estimated fishing mortality rates dropped below the threshold in 1995 and 1996; however, as excessive predation has eroded stock size and associated surplus production, F has exceeded the non-equilibrium F_{MSY} estimates in all years since 1997 (Figure C12.2-1).

B_{MSY} has shown greater stability than F_{MSY}, but still declined from over 20,000 MT in 1981 to less than 15,000 MT between 1997 and 2002, before rebounding in recent years (Table C9.3-5). Stock size as a proportion of maximum spawning potential exceeded 25% MSP between 1981 and 1986, and again between 1994-1996, but has since decreased drastically to less than 5% of an

unfished stock since 2005 (Table C9.3-5).

C12.3 Reference points calculated by the "Forage Hypothesis"

The following is a summary of the reference point and stock status analyses conducted under the forage hypothesis. Additional details are presented in the full discussion of the forage hypothesis in section **C10.0** of this report.

C12.3.1 Equilibrium conditions

The forage hypothesis investigated both Schaefer and Gompertz forms of the surplus production function. For both methods, equilibrium reference points are estimated from the fit population parameter values. However, the Gompertz production function with depensatory predation mortality as a function of striped bass biomass and the striped bass:menhaden ratio provided the best model fit, so reference points are evaluated for that model only. For the Gompertz model, equilibrium reference points are calculated as $F_{MSY} = r$ and $B_{MSY} = K / 2.72$.

The best model fit resulted in parameter estimates of $r = 0.48$, $K = 51,521$ MT, $c = 0.29$, and $B_{1981} = 82,472$ MT, providing equilibrium reference points of $F_{MSY} = 0.48$, $F_{collapse} = 1.31$, and $B_{MSY} = 18,941$ MT. Jackknife and bootstrap procedures indicated that parameters were estimated with good precision (Tables C10.5-2 and C10.5-3). Estimated fishing mortality rates exceeded F_{MSY} for the first decade of the time series, but dropped below the threshold in 1992 (Figure C10.5-13). By 1995, F had fallen to approximately 60% of the threshold, but subsequently began a steady increase to a level slightly above F_{MSY} during 1998-2002. After 2002, F_t dropped sharply and remained at about 50-70% of the threshold through 2006 (Figure C10.5-13).

Biomass has been below the SSB_{MSY} threshold in all years except 1996. Estimates of proxy MSP, calculated as B_t / K, fell continuously after 1983 from 65% of K to reach about 12% by 1990 (Figure C10.5-16).

C12.3.2 Non-equilibrium conditions

Two methods were implemented to calculate nonequilibrium reference points from the forage hypothesis model. The first simply involved renaming F_{MSY} to $Z_{MSY} = 0.48$ and comparing it to annual estimates of total (fishing plus predation) mortality (Z_{pt}). This estimate of total mortality at MSY is constant across time and serves as a benchmark for total mortality changes. Estimates of M_{pt} rose from a very low level in the mid-1980s to estimates equivalent to F_t and Z_{msy} by 1999 (Figure C10.5-13). After 1999, M_{pt} accelerated rapidly and reached 1.4 in 2006 (Figure C10.5-13). When combined with estimates of fishing mortality, it is evident that total mortality ($Z_{pt} = F + M_{pt}$) has exceeded the equilibrium reference point throughout the time series, except for a brief period in the mid 1990s.

The second method used to calculate non-equilibrium reference points was to directly calculate F_{MSY} under non-equilibrium conditions as equilibrium F_{MSY} minus predation mortality (see equation 16 of section **C10.0**). Mortality associated with the predator or predator-prey conditions is considered "off-limits" for management and yield can only be influenced by managing the fishery. Low level of predation mortality occurring in the 1980s and early 1990s resulted in F_{PSYt} levels close to Z_{MSY} (Figure C10.5-14). As fishing mortality rate declined, estimates of F_t fell below the non-equilibrium threshold between 1993 and 1996. However, increasing losses due to striped bass began to erode F_{PSYt}. Consequently, the amount of production available for harvest has declined, and fishing mortality has remained above the threshold in all years since 1997.

C12.4 Discussion

Reference point estimates and stock status determinations for the different models are compared in Table C12.4-1. Under equilibrium conditions, the ADAPT VPA estimate of F_{2007} (0.51) exceeds the overfishing threshold ($F_{20\%}$ = 0.42), indicating that overfishing is occurring. However, the two production models provide estimates of F_{2007} that are lower than their respective F_{MSY} thresholds. Under these scenarios, overfishing is not occurring on weakfish. For all three models under equilibrium condition, SSB_{2007} is below the respective biomass threshold, indicating that weakfish are overfished.

It is not possible to calculate non-equilibrium reference points from the VPA; however assuming non-equilibrium conditions, both production models indicate that weakfish are overfished and overfishing is occurring. The transition from overfishing not occurring under equilibrium to overfishing occurring in a non-equilibrium system can be attributed to the erosion of "available" production due to increased weakfish removals through predation and/or competition over the last decade. Although fishing mortality is above the threshold level, both models indicate that the F/Z ratio has been below 50% since the mid to late 1990s.

The 2006 peer review panel did not accept the WTC's determination that stock size had declined due to increased natural mortality as a result of predation (ASMFC 2006a, Part B). In fact, little attention was given to weakfish stock status at all during that review. In the current analysis, all models investigated indicate that weakfish stocks are depressed. Regardless of the many factors influencing the stock that may have led to this condition, it is imperative that the 48th SARC evaluate the determination of stock status so that the Weakfish Management Board can be apprised of the situation. Within this document the WTC has presented a suite of hypotheses that attempt to explain the mechanisms that have resulted in the current low biomass, the implications of which will help determine future management actions. The WTC requests guidance from the SARC on the utility of the different models and how they may be improved in order to ensure sustainable management of the stock. However, before management measures can be considered, the true status of the stock must be determined. With the models investigated in this stock assessment, it is evident that the weakfish stock declined substantially and is at or near all time low levels.

C13.0 Review stock projections and impacts on the stock under different assumptions of fishing mortality. (TOR #9)

C13.1 ADAPT projections

Projections were conducted relative to output from the ADAPT VPA using the AgePro (version 3.1) module of the NFT Toolbox. Biomass was projected for 25 years starting from the terminal year estimates of the preferred ADAPT VPA run (see section **C7.0** of this report). Values for biological parameters (*e.g.* weights at age, partial recruitment, etc) were the same as those used in the terminal year of ADAPT input. Multiple runs were conducted assuming a range of fishing and natural mortality values. Recruitment followed an empirical distribution of Age 1 numbers estimated through the full time series of the VPA.

All projection runs resulted in increased biomass over time and appeared to reach asymptotic values (Figure C13.1-1). However, different assumptions regarding F and M resulted in a wide range of projected biomass. Under the assumption of constant M = 0.25 and a harvest moratorium, SSB is projected to increase to more than 275,000 MT by the year 2032. Allowing harvest to occur at a level F = 0.25 (slightly below the fishing mortality target of $F_{30\%}$ = 0.28) the stock is expected to

peak at approximately 135,000 MT by 2019.

The WTC has expressed concern that natural mortality has increased in recent years. Under this scenario, projections based on M = 0.25 would be overestimated. Recent analyses have indicated that natural mortality may be as high as M = 1.0 or more. To investigate the potential for stock rebuilding under similar conditions, projections were conducted assuming a constant M = 0.75 (Figure C13.1-1). Under a harvest moratorium, SSB is expected to increase to approximately 45,000 MT by 2013. Allowing fishing mortality to occur at a level of F = 0.25, rebuilding would occur to 36,000 MT by 2013.

C13.2 Projections based on the "Predation Hypothesis"

C13.2.1 Introduction

Results from previous weakfish assessments (Kahn et al 2002, 2006) revealed that overfishing (F > Fmsy) had occurred on weakfish from about 1981 to 1991. As a result, more stringent fishery regulations were imposed on the weakfish recreational and commercial fisheries in 1992 and 1993 and fishing mortality (F) rates from 1996 to 2003 fell to well below Fmsy. Following a drop in F, coast-wide weakfish abundance initially rose about five fold from 1992 until about 1998, but stock size dropped unexpectedly thereafter to the lowest levels in the time series by 2007. Because F levels were relatively low and stable after 1999, follow-up studies (see TOR #5-7) have begun to find evidence for a Trophic Hypothesis involving enhanced striped bass (*Morone saxatilis*) and spiny dogfish (*Squalus acanthias*) predation or a more complex trophic triangle among weakfish, striped bass and Atlantic menhaden (*Brevoortia tyrannus)*. Under non-steady-state conditions that might arise from a recent increase in natural mortality (M), the projected pace of weakfish stock rebuilding following a moratorium to harvest (F = 0) might occur very slowly if at all as was reported recently for Atlantic cod (*Gadus morhua*) in the Gulf of St. Lawrence (Swain and Chouinard 2008). As shown recently by Walters et al (2008), the rate of stock rebuilding of depleted finfish stocks can be highly unpredictable especially under a shifting environment. By contrast, if M for weakfish is assumed to be fixed over time at 0.25 as was assumed in the ADAPT model, and, if future recruitment remains average, a moratorium to harvest (F = 0) should result in rapid stock rebuilding to Bmsy over a relative short time horizon. Clearly, the degree to which additional management promotes rapid (< 15 years) weakfish stock rebuilding in the future depends, among other things, on whether or not future M remains low and fixed over time.

In this section, weakfish relative spawning stock biomass (TSSB, mt) projections were made from 2010 to 2020 following the imposition of a simulated coast-wide moratorium (F = 0) to harvest beginning in 2009. The projection model was an extension of the Harvest Control Model (HCM) (Rugolo and Crecco 1993) used to forecast decadal shifts in striped bass (*Morone saxatilis*) biomass under several management scenarios. Given the uncertainty surrounding the current and future trend in M estimates, the following three scenarios that bracket a wide range of possibilities were examined with the weakfish HCM following a simulated 2009 moratorium to harvest: 1) M is fixed at 0.25 throughout the time series (1980-2020) as in ADAPT, there is no recent rise in trophic impacts on weakfish productivity, and recent (1999-2008) fishing mortality (F) have remained high (F > 1.0) as per ADAPT; 2) there is a moderate rise in M (from 0.25 to 0.65) from 1999 to 2020 due to predation but the magnitude of predatory mortality is less than predicted by the Steele-Henderson Model, and recent (1999-2008) F estimates have risen to moderate (0.7 to 1.0) levels; and 3) M on weakfish after 1997 has risen four-fold in magnitude (from 0.25 to 1.0) as per the Steele-Henderson Model, and fishing mortality (F) rates have remained relatively low (F < 0.50) from 1996 to 2008 as

per the Index-based Analysis.

C13.2.2 Methods

A modification to the Harvest Control Model (HCM) was used to project relative changes in weakfish total spawning stock biomass (TSSB) from 2010 to 2020 following the imposition of a simulated moratorium to harvest ($F = 0$) beginning in 2009. The theory and application of the HCM are fully described for striped bass catch quota projections (Rugolo and Crecco 1993). The entire model is programmed in SAS (SAS 2002) (Appendix C-3). The model output (ages 1+ TSSB) in the HCM was expressed as year (1980-2020) and age (ages 1-10) effects. A small percentage of weakfish do survive beyond age 10, but the average contribution of ages 11+ weakfish to annual SSB was relatively modest from 1982 to 2007. The HCM is configured to operate off several age-related (ages 1 to 10) vectors that reflect somatic growth, maturity, natural mortality and partial recruitment (PR).

In this analysis, three scenarios are offered to examine how potential shifts in future natural mortality (M) levels might affect rebuilding of weakfish TSSB following the imposition of a simulated 2009 moratorium ($F = 0$) to harvest (Table 1). The scenarios are: 1) M is fixed at 0.25 throughout the time series (1980-2020) as in ADAPT, there is no recent rise in M due enhanced trophic impacts on weakfish productivity, unreported weakfish landings and discards have recently (1999-2008) increased steadily to where they now comprise over 80% of the total fishery losses and recent (1999-2008) fishing mortality (F) has remained high ($F > 1.0$) as per ADAPT; 2) there is a moderate rise in M (from 0.25 to 0.65) from 1999 to 2020 due to enhanced predation but the magnitude of predatory mortality is less than predicted by the Steele-Henderson Model, there is a more modest rise in recent (1998-2008) unreported landings and recent (1999-2008) F estimates have risen and remain moderately high (0.7 to 1.0) from 1999 to 2008; and 3) M levels on weakfish from 1999 to 2020 have exhibited a four- fold rise (0.25 to 1.0) in magnitude as per the Steele-Henderson Model, unreported landings have remained low and steady from 1999 to 2008 and fishing mortality (F) rates have remained relatively low ($F < 0.50$) from 1996 to 2008 as per the Index-based Analysis. In all of the above scenarios, input parameters in the weakfish HCM include fully recruited fishing mortality (F), somatic growth in average length (mm) and weight (kg) by age, percentage maturity by age, the partial recruitment vectors (PR) by age to the fisheries, either a fixed M of 0.25 (scenario #1) or a systematic rise in natural mortality (Scenarios #2 and 3) from 1997 to 2020. The fully recruited (ages 4+) F estimates, PR vectors, age at maturity and age-specific somatic growths in weight were taken directly from the preferred ADAPT run. Since retrospective bias in ADAPT severely underestimated recent (> 2004) F estimates, the fully recruited F in 2004 was repeated for the years 2005 to 2008. Discard mortality among sublegal weakfish was expressed in the HCM as 0.10 *F. Annual estimates of relative TSSB were summed in the HCM as the product of relative abundance by age, the average age-specific weight (kg) and the average age-specific maturity vector. The pace of future TSSB rebuilding among the three scenarios was measured arbitrarily against a relative Bmsy value that was scaled directly from the estimated equilibrium Bmsy threshold for weakfish from surplus production modeling.

In the HCM, a time series vector (1970-2020) of recruitment relative abundance is the primary variable that dictates temporal changes in future weakfish TSSB abundance by age. Thus relative TSSB in numbers from the HCM across the time series is expressed in recruitment units. There is a time series (1982-2007) of coast-wide average age 0 indices (Rec 0) that are expressed as mean catch/effort from nine State surveys from 1982 to 2007 (see section **C9.0**). However, the relatively high and stable age 0 indices occurring from 1997 to 2006 have thus far not translated into

high subsequent abundance of ages 1 and older fish, This mismatch in year-class abundance between subsequent age groups suggests the emergence of a recruitment bottleneck in recent years that has constricted the flow of recruitment via a rise in natural mortality (see sections **C7.0** to **C9.0**). Thus, instead of using average age 0 recruitment to express coast-wide recruitment in the HCM, a time series (1970-2008) of trial values of age 1 weakfish relative indices were imputed to the HCM so that the resulting trend in model-based TSSB from 1981 to 2008 closely approximated (Pearson correlation r> 0.90) the observed trend in ages 1+ relative biomass from the Index-based Analysis (refer to section **C8.0**). All available trawl data have indicated that coast-wide weakfish abundance was at peak levels from the mid 1970's until the early 1980's. To simulate these high TSSB levels from 1980 to 1985, a series (1970-1980) of dominant year-classes and relatively low fishing mortalities (F < 0.4) were imputed in the HCM. Note that projected TSSB each year is composed of 10 age groups. For this reason, it was necessary to begin recruitment to the HCM in 1970 so that the first full year of simulated TSSB composed of 10 age groups began in 1980. The long-term (1982-2008) average recruitment (long-term mean = 16.5) level was used to express age 1 recruitment from 2009 to 2020 (Table 1). We are primarily interested in comparing how a simulated moratorium (F = 0) might affect future stock rebuilding among three scenarios with a fixed M versus a rising future M. Thus, the use of constant recruitment from 2009 to 2020 would not alter the relative trajectory in projected TSSB rebuilding from 2010 to 2020. Random or log normal variation (stochastic) of future (2009-2020) recruitment using the observed recruitment time series (1982-2007) could have been used in TSSB projections from the HCM, but stochastic variation in future recruitment would have complicated the interpretation of TSSB trends during the rebuilding phase (2010-2020) of each scenario.

C13.2.3 Results and Discussion

In Scenario #1 under a relatively low and fixed natural mortality (M = 0.25) throughout the time series (1980-2020), the HCM predicted that a moratorium to all weakfish harvest (F = 0) enacted in 2009 and thereafter would lead to rapid TSSB recovery that would approached the Bmsy threshold by 2020 (Figure C13.2-1). Since M was assumed to have remained fixed at 0.25, reducing a high F (> 2.0) to zero from 2009 to 2020 exerts an enormous leverage on enhanced survival (high F/Z ratio) with which to rapidly rebuild TSSB from 2010 to 2020. However, it is important to note that in Scenario #1, we have assumed that the primary cause of the post 1999 weakfish stock collapse is overfishing mainly resulting in a systematic rise in unreported landings and discards from some yet unknown recreational and commercial fisheries. Since reported weakfish landings and estimated discards have fallen from 5500 mt in 1996 to about 800 mt by 2008, these unreported landings would need to have increased from about 1,000 mt to 5,000 mt. from 1998 to 2008 to have caused the post 1999 weakfish stock decline. Thus, in order to enact a complete moratorium on all weakfish landings in 2009, nearly all inshore fisheries activities from North Carolina to Rhode Island would have to cease from 2009 to 2020.

In scenario #2 under the assumption of a moderate rise in M from 0.25 to 0.65 after 1997, the HCM predicted that a moratorium (F = 0) enacted in 2009 would result in some measureable TSSB rebuilding by 2020, but the magnitude of stock growth would fall far short of the Bmsy threshold (Figure C13.2-2). Since M rose more moderately in this Scenario from 0.25 to 0.65, a moratorium to harvest exerts somewhat less leverage on survival (lower F/Z ratio) than Scenario #1 in order to rebuild TSSB over a 10 year horizon. Note that in Scenario #2, we have assumed that both a moderate rise in M (from 0.25 to 0.65) after 1996 coupled with a more modest systematic increase in unreported landings resulted in the post 1999 stock collapse. Thus, like Scenario #1, to enact an

effective moratorium to harvest of both reported and unreported weakfish landings, we would have to close nearly all inshore fisheries from North Carolina to Rhode Island from 2009 to 2020.

In scenario #3 under a pronounced rise in M from 0.25 to 1.0 ostensibly due to enhanced predation, the HCM predicted that a moratorium to harvest in 2009 and thereafter would result in little if any TSSB rebuilding by 2020 (Figure C13.2-3). Given the pronounced rise in M coupled with low and steady F from 1996 to 2008 in Scenario #3, a moratorium to harvest after 2008 provides very little leverage to enhanced survival (lowest F/Z ratio) on which measurable stock rebuilding depends. In Scenario #3, we note that fishing mortality (F) was largely driven by reported weakfish landings which have fallen by 80% from 1996 to 2008. By contrast, unreported weakfish landings and discards are assumed to have remained relatively low and steady from 1996 to 2008 during which M rose four-fold (0.25 to 1.0). In this Scenario, we have assumed that high M would persist throughout the rebuilding time frame (2010-2020). It is important to note that if M of 1.0 associated largely with enhanced predation during the recovery period (2010-2020) should drop back to the pre-1997 level of 0.25, measureable weakfish stock rebuilding would likely occur after 2010 without further management restrictions, although the imposition of a moratorium during this period would accelerate the pace of stock rebuilding.

C13.3 Projections based on the "Forage Hypothesis"

Jackknife and bootstrap estimates of GDR parameters and biomass in 2007 were projected with equations 2 and 7 to evaluate the effect of fishing restriction scenarios through 2015. Projections explored whether conditions set forward in Amendment 4 (ASMFC 2002) for stock rebuilding could be achieved by managing the weakfish fishery alone. Under overfished conditions (two consecutive years above the F threshold), F was to drop below 0.2 and SSB was to be rebuilt to 30% of an unfished stock (ASMFC 2002). While these conditions were based on VPA-based estimates and reference points, approximations could be developed from the GDR. The probability of biomass growing to 15,000 mt (approximating 30% of K) by 2015 under reduced fishing could be determined. Three fishery management scenarios were portrayed: an approximation of the minimum F reduction in Amendment 4 ($F_{recover}$), F = 0.2 (a literal interpretation of Amendment 4), and a moratorium.

Projections of biomass used each jackknife or bootstrap trial estimate of r, K, c, R_t, and B_{2007}. Estimates of H_t equaled $U_t * B_t$ during 2008-2015, where U_t was an assumed exploitation rate derived from F_{2006} for 2007-2008, a transition F for 2009, and a 2010-2015 target F (specified above) and biomass. Striped bass biomass and R_t were constant at the 2006 estimate (88,000 mt and 5, respectively). In trials where cuts in F were imposed, it was assumed that cuts in F would not occur until fall 2009, so biomass trajectory during 2008 was based on a continuation of 2007 conditions. Cuts were instituted in 2009 (F drops by a maximum of half to mimic regulations imposed by fall) and maintained until 2015. $F_{recover}$ was estimated as

$$(0.2 / 0.31) * Fmsy; \qquad (17)$$

where 0.2 equaled the maximum F allowed for rebuilding under Amendment 4 and 0.31 equaled target. F_{msy} was represented by the median of the bootstrap and jackknife trials (0.48), therefore $F_{recover}$ was approximated as 0.31. Losses under moratorium conditions (U_m) during 2010-2015 were approximated as

$$(1-(H_w / H_t)) * U_{2006}; \qquad (18)$$

where H_w = losses of weakfish from all harvest and weakfish commercial fishery discards in 2006; H_t total losses of weakfish in all fisheries, and U_{2006} was estimated exploitation rate in 2006. U_m was approximated as 0.056. The probability of B_{2015} reaching 15,000 mt was estimated separately for jackknife or bootstrap-based projections.

Stock recovery was not possible under $F_{recover}$ and F = 0.2, and there was about 1.4% chance of recovery under moratorium conditions for bootstrap runs and 0% chance for jackknife moratorium scenarios. It should be noted that estimates of F since 2003 have been below or at $F_{recover}$. Prospects for weakfish are grim under these projections; jackknifing and bootstrapping indicated 98% and 91% chances (respectively) that weakfish biomass would fall to zero by 2010 if trophic conditions prevailing in 2006 continued. Instances where extirpation did not occur were associated with estimates of B_{2007} in excess of 4,600 mt. If high predation-competition losses estimated for 2006 prevail, a moratorium cannot be initiated in time according to the great majority of trials. By 2015, extirpation becomes a virtual certainty even under a moratorium

C14.0 Make research recommendations for improving data collection and assessment. (TOR #10)

The following list of prioritized research needs updates the list provided in the 2008 Weakfish Fishery Management Plan Review. New research recommendations identified by the WTC are presented in ALLCAPS. Comments regarding existing recommendations are shown in *italics*.

High Priority
INVESTIGATE ALTERNATIVE AGE BASED MODELS WHICH ALLOW ERROR IN CATCH AT AGE ESTIMATES (*E.G.* STATISTICAL CATCH AT AGE) AND/OR ARE LESS PRONE TO RETROSPECTIVE PATTERNS (*E.G.* EXTENDED SURVIVORS ANALYSIS).

EVALUATE CONSUMPTION OF WEAKFISH BY PREDATORS USING A MORE ADVANCED MULTISPECIES MODEL, SUCH AS THE ASMFC MSVPA MODEL OR ECOPATH WITH ECOSIM, TO VALIDATE ESTIMATES CALCULATED BY PRODUCTION MODELS WITH PREDATION-COMPETITION EXTENSIONS.

DEVELOP A BIOENERGETICS MODEL FOR WEAKFISH THAT ENCOMPASSES A BROADER RANGE OF AGES THAN HARTMAN AND BRANDT (1995). USE IT TO EVALUATE DIET AND GROWTH DATA.

INITIATE MONITORING OF WEAKFISH, STRIPED BASS, AND SPINY DOGFISH DIETS OVER A BROAD REGIONAL AND SPATIAL SCALE.

Collect catch and effort data including size and age composition of the catch, determine stock mortality throughout the range, and define gear characteristics. In particular, increase length-frequency sampling, particularly in fisheries from Maryland and further north.

Derive alternative estimates of discard mortality rates and the magnitude of discards for all commercial gear types from both directed and non-directed fisheries. In particular, quantify trawl bycatch, refine estimates of mortality for below minimum size fish, and focus on factors such as

distance from shore and geographical differences. Improved estimates of discard mortality would best be obtained through increased observer coverage.

Conduct an age validation study. *This work has been completed in Lowerre-Barbieri et al (1995)*.

Identify stocks and determine coastal movements and the extent of stock mixing, including characterization of stocks in over-wintering grounds. Most direct method would be to develop a coastwide tagging program. *Otolith samples have been obtained by Old Dominion University, but funding has not been available for processing.*

Conduct spatial and temporal analysis of the fishery independent survey data. The analysis should assess the impact of the variability of the surveys in regards to gear, time of year and geographic coverage on their (survey) use as stock indicators. *Research is ongoing by Dr. Yan Jiao of Virginia Tech. University. See Winter et al 2009 in the supplementary material for this peer review.*

Analyze the spawner-recruit relationship and examine the relationships between parental stock size and environmental factors on year-class strength. *Work is currently underway by Weakfish Stock Assessment Subcommittee.*

Develop latitudinal/seasonal/gear specific age length keys for the Atlantic coast. Increase sample sizes to consider gear specific keys.

Medium Priority
Examine geographic and temporal differences in growth rate (length and weight at age).

Define reproductive biology of weakfish, including size at sexual maturity, maturity schedules, fecundity, and spawning periodicity. Continue research on female spawning patterns: what is the seasonal and geographical extent of "batch" spawning; do females exhibit spawning site fidelity? *This work has been completed by Nye et al 2008 and Lowerre-Barbieri et al 1996.*

Compile data on larval and juvenile distribution from existing databases in order to obtain preliminary indications of spawning and nursery habitat location and extent.

Conduct hydrophonic studies to delineate weakfish spawning habitat locations and environmental preferences (temperature, depth, substrate, etc.) and enable quantification of spawning habitat.

Continue studies on mesh-size selectivity; up-to-date (1995) information is available only for North Carolina's gill net fishery. Mesh-size selectivity studies for trawl fisheries are particularly sparse. *Gillnet selectivity has been investigated by Swihart et al (2000). Can also be obtained from the NC PSIGNS survey.*

Assemble socio-demographic-economic data as it becomes available from ACCSP.

Continue studies on recreational hook-and-release mortality rates, including factors such as depth, warmer water temperatures, and fish size in the analysis. Studies are needed in deep and warm water conditions. Further consideration of release mortality in both the recreational and

commercial fisheries is needed, and methods investigated to improve survival among released fish.

Low Priority
       Define restrictions necessary for implementation of projects in spawning and over-wintering areas and develop policies on limiting development projects seasonally or spatially.
       Document the impact of power plants and other water intakes on larval, post larval and juvenile weakfish mortality in spawning and nursery areas, and calculate the resultant impact to adult stock size. *Data are available for power plants in the Delaware Bay area and North Carolina. Data should be compiled and evaluated.*
Determine the onshore versus offshore components of the weakfish fishery.

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