

# **Fisheries Ecosystem Model of the Chesapeake Bay: Methodology, Parameterization, and Model Exploration**

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## 1 Abstract

This report describes an ecosystem model of the Chesapeake Bay, the Chesapeake Bay Fisheries Ecosystem Model (CBFEM), prepared using the Ecopath with Ecosim approach and software. The CBFEM was created in response to a management need in the Chesapeake region for a quantified estimation of trophic pathways in the Bay. This information can be used to understand how one stock affects another within the food web and how the many Bay fisheries impact both target and nontarget species. Because the life histories and population dynamics of the thousands of organisms that live within the Bay are complicated, a model is necessary to provide an accurate estimation of the system.

Model construction has been carried out in close consultation with Chesapeake Bay researchers through a series of workshops. Currently, the model includes 45 functional groups of organisms, some of which represent life history stanzas of individual species, representing all trophic levels. The input data primarily includes assessment results from the Chesapeake Bay (including biomasses, mortality rates, catches, and effort) supplemented with research vessel survey data (fisheries and biological oceanography studies), ecological studies (as available from researchers and institutions in the region), and parameter estimates obtained from literature where necessary to supplement local data. Activities are under way to refine the temporal and spatial resolution of the CBFEM and to continue to incorporate hydrographic data.

This documentation is intended to facilitate use and further development of the CBFEM, so that it can serve as a ‘living’ model. Future revisions to the CBFEM and supporting documentation will be made available on the NOAA Chesapeake Bay Office web site (<http://chesapeakebay.noaa.gov>).

## 2 Introduction

### 2.1 *The Chesapeake Bay*

The Chesapeake Bay is the largest estuary in the continental United States, located midway along the Atlantic coast of the United States. The surface area of the tidal portion of the Chesapeake Bay system is approximately 10,000 km<sup>2</sup>, while the area including tributaries is estimated to 18,580 km<sup>2</sup>. More than 20 major tributaries drain into the Bay from a watershed that stretches across six states: New York, Pennsylvania, Maryland, Delaware, Virginia, West Virginia, and the District of Columbia. The largest of these tributaries, the Susquehanna River, provides more than half of the freshwater flow to the Bay. The waters of the Chesapeake Bay and tidal portions of its tributaries are governed by Maryland and Virginia (Figure 1).

The Bay is a partially mixed estuary, with an average tidal range of approximately 1 m at its mouth to less than 30 cm at its head (cited in 1989). Salinity within the Bay ranges from less than 0.5 ppt at its northern extreme to 32 ppt near its mouth. The Bay can be divided into three major salinity regions: oligohaline (0-5 ppt), mesohaline (6-18 ppt), and polyhaline (> 18 ppt). Water temperatures in the Bay vary greatly throughout the year, reaching 28-30°C in late summer and 1-4°C in late winter (Murdy *et al.*, 1997).

The estuarine circulation pattern of a flow of deeper, more saline water from the Atlantic Ocean into the Bay and its tributaries and surface fresher water out of the Bay serves to transport larval fishes and crabs from the ocean to their nursery habitats and juvenile fishes from tributaries to the coastal waters of the Atlantic. This transport mechanism is very important to the population dynamics of many Bay species.

The mixture of freshwater from the tributaries and seawater from the coastal ocean creates and maintains a variety of brackish habitats within the Bay. Tidally influenced habitat types in the Bay include: pelagic waters, nearshore littoral areas, and the benthic zone. Littoral habitats, such as marshes on intertidal lowlands, aquatic grass beds in the shallow flooded flatlands, and oyster reefs, are highly productive, serving as nursery areas to many fish and shellfish species, facilitating rapid growth under relatively protected conditions. The diversity of habitats within the Chesapeake Bay system enables it to support nearly 3,000 species of plants and animals within its waters and tidal margins.

Finfish species inhabiting the Chesapeake Bay have a wide variety of life history strategies. The American eel, *Anguilla rostrata*, is a catadromous species, spending most of its life in tributaries of the Chesapeake Bay, returning to the Atlantic Ocean to spawn. Some marine fishes, like the weakfish, *Cynoscion regalis*, enter the Bay to feed and spawn seasonally and then return to the coastal ocean. Anadromous species, like the American shad, *Alosa sapidissima*, and striped bass, *Morone saxatilis*, spend most of their adult lives migrating in the Atlantic Ocean, but return to Bay tributaries to spawn. Other species, like the white perch, *Morone americana*, spend their entire lives within the Chesapeake Bay system, undergoing 'semi-anadromous' seasonal migrations within the Bay. Due to the complexity of the Chesapeake Bay ecosystem, it is necessary to develop modeling tools like the CBFEM to simulate interactions between these many different species, to quantitatively estimate how they fit together within the larger food web and how human impacts are likely to affect this complex system.

The diversity of habitats within the Chesapeake Bay, combined with wide ranges of temperatures throughout the year, result in very dynamic seasonal changes in fish assemblages. During late summer and early autumn, fish diversity reaches its maximum due to a movement of tropical species into the lower portion of the Bay. When the cooler temperatures of autumn arrive, most marine fish within the Bay begin to migrate either south to Cape Hatteras, North Carolina, or offshore to the edge of the continental shelf. During winter, the abundance and diversity of fish in the Bay is relatively low. However, by early spring, abundance and diversity rebound significantly as anadromous species enter the Bay, followed soon after by the warm-temperate and subtropical summer residents.

Since the early 1800s, the Chesapeake Bay has supported a variety of large-scale commercial and recreational fisheries of both finfish and shellfish. The predominant invertebrate fisheries in the Chesapeake Bay have included the eastern oyster (*Crassostrea virginica*), blue crab (*Callinectes sapidus*), soft clam (*Mya arenaria*), and hard clam (*Mercenaria mercenaria*). The large-scale finfish fisheries have included striped bass, American shad, river herring (*Alosa aestivalis*), white perch, bluefish (*Pomatomus saltatrix*), Atlantic menhaden (*Brevoortia tyrannus*), summer flounder (*Paralichthys dentatus*), weakfish, Atlantic croaker (*Micropogonias undulates*), and spot (*Leiostomus xanthurus*). Several species, like white perch and Atlantic croaker, have sustained significant harvest levels, although trends in the commercial and recreational landings have varied over the last several decades. Striped bass landings may be the most dramatic in terms of variability from the 1960s to present. Many species, such as the eastern oyster, American shad, and striped bass, have suffered overexploitation in the Chesapeake Bay. Overfishing and the collapse of several Bay and coastal fish

stocks during the 1900s prompted the creation of fisheries management agencies both along the Atlantic coast and within the Chesapeake Bay.

In coastal areas, the Atlantic States Marine Fisheries Commission (ASMFC) serves as a deliberative body, coordinating the conservation and management of fisheries in near-shore state waters along the eastern seaboard from Maine to Florida (to 4.8 km or 3 miles off the coast). The Mid-Atlantic States Fishery Management Council (MASFC) is responsible for managing fisheries in federal waters, which occur predominantly off the mid-Atlantic coast (from 4.8 to 322 km or 3 to 200 miles offshore). Within the Bay, tidal fisheries are managed on a jurisdiction-specific basis, by the Virginia Marine Resources Commission (VMRC), the Maryland Department of Natural Resources (MD DNR), and the Potomac River Fisheries Commission (PRFC). The three jurisdictions have agreed upon management strategies, as outlined in Chesapeake Bay fisheries management plans, for commercially and recreationally targeted species within the tidal portion of the Chesapeake Bay.

## 2.2 *Multispecies management*

Traditionally, fisheries management plans have been targeted to manage a single species, (e.g., stock assessments designed to derive estimates of population size and fishing mortality rates, synthesis of life history characteristics to determine fishing seasons). However, single-species analyses have not traditionally considered the ecology of the species under management, (e.g., habitat requirements, response to environmental change), ecological interactions among species, (e.g., predation, competition), or technical interactions, (e.g., discards, bycatch) (NMFS, 1999; Link, 2002b; a). Basing fisheries management plans on both single-species characteristics and ecological processes is now mandated federally by the Magnuson-Stevens Act Reauthorization (NMFS, 1999; NRC, 1999; U.S. Commission on Ocean Policy, 2004) and regionally by the *Chesapeake 2000* agreement (Chesapeake Bay Program, 2000). Ecosystem-based fisheries management plans are currently under development for five target species/species groups in the Chesapeake Bay: striped bass, alosines (shad, alewife, and blueback herring), blue crab, menhaden, and the eastern oyster.

Ecosystem-based fisheries management is preferable in the Chesapeake Bay as traditional methods of management do not explicitly allow consideration of how fish populations and habitats are variably impacted by increasing human populations. To understand how one species is likely to respond to changing conditions, it is necessary to look at the response of all other species that affect it within its ecosystem. Not only do humans affect fish populations directly through fishing, but they also have major indirect impacts such as nutrient, sediment, and toxics pollution; disease; and physical destruction of habitat. All of these factors influence fish stock levels. Ecosystem-based fisheries management attempts to address all of these factors to lessen their impact on fish stocks. As part of this process, it is necessary to create tools to accurately portray how the ecosystem is currently functioning and how it is likely to function in the future given changes in habitat and stock management. One such tool is the CBFEM, which helps to understand how the food web is being impacted by these changes to the ecosystem and how fisheries management can be altered to address them.

The concept that ecological processes can strongly influence stock abundance is not new to fisheries science. During the 1970s and 1980s, several single-species population models were extended to include multiple species and the implied ecological interactions (Andersen and Ursin, 1977; May *et al.*, 1979; Mercer, 1982; Kerr and Ryder, 1989; Daan and Sissenwine, 1991). These models fostered awareness of the importance and role of ecological processes on yield performances of fish stocks, but were generally

viewed as underdeveloped. Recently, this belief has changed due to the sophistication and increased availability of multispecies assessment and ecosystem models (Hollowed *et al.*, 2000; Whipple *et al.*, 2000; Latour *et al.*, 2003). The analytical evolution of these modeling approaches has reached the point where it can now provide meaningful management advice. However, a significant deterrent to the use of multispecies modeling approaches is that more model parameters may need to be estimated compared with traditional single-species analyses, which in turn creates the need for additional types of data.

In the Chesapeake Bay region, single-species fisheries management plans have been developed for numerous species (Table 1). There has been a growing interest in ecosystem approaches to fisheries management in the Chesapeake Bay region since the mid-1990s. Efforts dedicated to ecosystem-based fisheries management in the region include: 1) creation of two ASMFC multispecies subcommittees, the first in 1999 and the second in 2004; 2) convening of multispecies technical workshops (Miller *et al.*, 1996; Houde *et al.*, 1998); 3) development of the multispecies and cross-jurisdictional sampling platform, the Chesapeake Multispecies Monitoring and Assessment Program (ChesMMAAP) in 2002; 4) continued development of the Fisheries Ecosystem Model for the Chesapeake Bay; 5) development of a multispecies virtual population analysis for the Chesapeake Bay; 6) release in 2006 of the *Fisheries Ecosystem Planning for the Chesapeake Bay* (NOAA CBFEAP, 2006) as the first fisheries ecosystem plan in the U.S.; and 7) ongoing development of ecosystem-based fisheries management plans for five target species/groups in the Chesapeake Bay (allosines, blue crab, eastern oysters, striped bass, and menhaden) in response to the regional goals for multispecies fisheries management set by the *Chesapeake 2000* agreement (Chesapeake Bay Program, 2000).

The language of the *Chesapeake 2000* agreement as it pertains to multispecies fisheries management, reads as follows:

- By 2004, assess the effects of different population levels of filter feeders such as menhaden, oysters, and clams on Bay water quality and habitat.
- By 2005, develop ecosystem-based multispecies management plans for targeted species.
- By 2007, revise and implement existing fisheries management plans to incorporate ecological, social, and economic considerations, multispecies fisheries management and ecosystem approaches.

In response to the *Chesapeake 2000* agreement (Chesapeake Bay Program, 2000), the NOAA Chesapeake Bay Office, working through the Chesapeake Bay Stock Assessment Committee, initiated a project in October 2001 to develop an Ecopath with Ecosim (EwE) model of the Chesapeake Bay. This Technical Report summarizes the results of the first years of development. Specifically, the 1950-2002 version of the Chesapeake Bay EwE model is presented, complete with detailed descriptions of the data used for model parameterization and calibration included in the appendices.

The current Chesapeake Bay EwE model should be considered a work in progress, and will evolve and improve in response to the collection of new information. For many components of the model, accurate and precise data were not available to define key input parameters. In an effort to fill these data gaps, research activities have been initiated to quantify missing parameters. However, several years of work will be necessary before the data will become available. Consequently, this Technical Report should be considered descriptive of the 1950-2002 version of the Chesapeake Bay Ecosystem Model. The

Technical Report will be under continuous development in the medium term, and electronic versions of the report will be released periodically and will be accessible on the NOAA Chesapeake Bay Office FTP site: <ftp://noaa.chesapeakebay.net/CB> Fisheries Ecosystem Model.

### **3 Methods**

#### **3.1 *EwE General Methodology***

##### **3.1.1 The EwE ecosystem modeling approach**

The EwE software is a modeling tool used to evaluate quantitative trophic interactions within an ecosystem in order to assess options for ecosystem-based management of fisheries. To run the Ecopath model, four groups of basic input parameters must be entered into the model for each of the species groups: diet composition, biomass accumulation, net migration, and catch. Three of the following four additional input parameters must also be input: biomass, production/biomass ( $Z$ ), consumption/biomass, and ecotrophic efficiency. Forcing functions have been developed for the system, including one for climate, primary production, and habitat area. The model uses the input data along with algorithms and a routine for matrix inversion to estimate any missing basic parameters so that mass balance is achieved. Basic input parameters for the Chesapeake Bay Fisheries Ecosystem Model are listed in Table 3.

Once all basic parameters have been input or estimated, the model balances the input and output of each group with two linear equations of production and consumption, using varying rates of respiration for adjustment. The model can come up with numerous balanced scenarios, given the input data and forcing functions. The balancing process results in predictions of biomass, production, and consumption values for each functional group and measures how closely these predictions match the input data. The modeler chooses which model run matches time-series data most closely based on the sum of least squares test performed by the model. The modeler then searches the selected model run for errors, makes necessary adjustments and documents these adjustments thoroughly. The modeler decides which parameters to include in the final estimation and then re-runs the model with the appropriate changes. This process (Figure 2) typically repeats several times until the model closely predicts existing data for that year.

The mass-balanced linear equations of Ecopath are then re-expressed as coupled differential equations so that they can be used by the Ecosim module to simulate what happens to the species groups over time. Model runs are compared with time-series data and the closest fit is chosen to represent the system. The Ecosim module can be used to simulate various management options for the system, by varying parameters over time to estimate potential ecosystem changes.

##### **3.1.2 Ecopath**

An Ecopath model uses trophically-linked biomass pools to create a mass-balanced snapshot of the resources and interactions in an ecosystem (Christensen and Pauly, 1992; Pauly *et al.*, 2000; Christensen and Walters, 2004). The biomass pools typically represent either a single species or a group of species that comprise an ecological guild. These pools may be split into ontogenetic age categories (juvenile, subadult, adult, *etc.*), commonly called ‘stanzas,’ and a detailed accounting of growth and survival for monthly cohorts is conducted for such groups. Biomass pools are created for all major components of the ecosystem, regardless of trophic level.

The parameterization of an Ecopath model is based on satisfying two ‘master’ equations. The first equation describes how the production term for each group can be divided for an arbitrary time period:

$$\begin{aligned} \text{production} &= \text{catch} + \text{predation} + \text{net migration} + \text{biomass accumulation} \\ &+ \text{other mortality.} \end{aligned} \quad (1)$$

More formally, equation (1) can be expressed as:

$$B_i(P/B)_i EE_i = Y_i + E_i + BA_i + \sum_{j=1}^n B_j(Q/B)_j DC_{ji} \quad (2)$$

where for biomass pool  $i = 1, \dots, n$ :

- $B_i$  is total biomass during the period of question
- $(P/B)_i$  is the production to biomass ratio
- $EE_i$  is the ecotrophic efficiency, defined as the fraction of the production that is consumed within or harvested from the system
- $Y_i$  is the yield or catch in weight (note that  $Y_i = F_i B_i$  where  $F$  is the fishing mortality rate)
- $E_i$  is the net migration rate (emigration – immigration)
- $BA_i$  is the biomass accumulation rate for ( $i$ )
- $B_j$  is the biomass of the consumers or predators of ( $i$ );  $(Q/B)_j$  is the food consumption per unit biomass for consumer  $j$
- $DC_{ji}$  is the average fraction of  $i$  in the diet of  $j$  (note that  $DC_{ji} = 0$  when  $j$  does not eat  $i$ )

At a minimum, Ecopath requires input of  $DC_{ji}$ ,  $Y_i$ , and three of the following four parameters for each species or biomass pool in the model:  $B_i$ ,  $(P/B)_i$ ,  $(Q/B)_i$ , and  $EE_i$  (mass balance principles are used to estimate the fourth parameter). If all four parameters are known, then Ecopath can be used to estimate either  $BA_i$  or  $E_i$ . Equation (2) implies that an ecosystem under study is described completely by an  $n$ -dimensional system of linear equations, the solutions of which can be easily calculated (Mackay, 1981); the resulting estimates of biomass, production, and consumption can be used to construct a quantitative network diagram of energy flow for the system (Ulanowicz, 1986).

The second ‘master’ equation is based on the principle of conservation of matter within a group and is designed to balance the energy flows of a biomass pool:

$$\text{consumption} = \text{production} + \text{respiration} + \text{unassimilated food} \quad (3)$$

Winberg (1956) defined consumption as the sum of somatic and gonadal growth, metabolic costs, and waste products. Equation (3) generally follows this definition, but differs in the sense that it is used to estimate losses rather than to measure growth. Balance of the energy equation is achieved by estimating respiration from the difference between the consumption, production, and unassimilated food terms. For more details on Ecopath, see Christensen and Pauly (1992) and Christensen and Walters (2004).

### 3.1.3 Ecosim

Ecopath is used to describe the interactions among resources within an ecosystem. Additional modules are created to simulate the dynamics of the ecosystem resources and the effects of different management

strategies on the structure and function of an ecosystem. The time-dynamic module, called Ecosim, provides a simulation capability that facilitates policy exploration at the ecosystem level, with initial parameters inherited from the base Ecopath model. To construct an Ecosim model, it is necessary to re-express the system of linear equations in (2) as a system of coupled differential equations. This transformation takes the following form (Walters *et al.*, 1997; Walters *et al.*, 2000; Christensen and Walters, 2004):

$$\frac{dB_i}{dt} = g_i \sum_{j=1}^n c_{ji} - \sum_{j=1}^n c_{ij} + I_i - (M_i + F_i + e_i)B_i \quad (4)$$

where:

- $g_i$  is growth efficiency
- $F_i$  is the instantaneous rate of fishing mortality
- $e_i$  is the rate of emigration
- $I_i$  is the rate of immigration
- $c_{ij}$  ( $c_{ji}$ ) is the consumption of biomass pool  $i$  ( $j$ ) by biomass pool  $j$  ( $i$ )

This system of equations is used to represent the spatially aggregated dynamics of entire ecosystems and is combined with explicit age/size-structured delay-difference equations to represent populations that have complex life histories and selective harvesting of older animals. An important aspect of Ecosim is the expression of the consumption or ‘flow’ rates among linked species or biomass pools. Consumption of prey  $i$  by predator  $j$  is modeled as:

$$Q_{ij}(B_i, B_j) = \frac{a_{ij} v_{ij} B_i B_j}{(2v_{ij} + a_{ij} B_j)}, \quad (5)$$

where  $a_{ij}$  is the rate of effective search for prey  $i$  by predator  $j$ , and  $v_{ij}$  is the behavioral exchange rate between vulnerable and invulnerable prey pools (Figure 3). Equation (5) is based on the notion that consumption is limited by ‘risk management’ behaviors of predators and prey at very small time scales. That is, predator-prey interactions are assumed to take place primarily in restricted ‘foraging arenas’ where prey only become vulnerable to predation through their own requirements for resource acquisition (Walters *et al.*, 1997; Walters *et al.*, 2000).

Relative to Ecopath, Ecosim introduces a number of new parameters, of which the simulations are especially sensitive to the vulnerability settings (Christensen and Walters, 2004). For this we use a vulnerability factor, (which we often just call ‘vulnerability’). The vulnerability factor expresses how much the predation mortality for a given prey can increase if the predator abundance is increased. When the predator is close to its carrying capacity with regard to the given prey, the predation mortality cannot be increased any further ( $v = 1$ ), and an increase in predator abundance, (e.g., due to good recruitment) will be compensated for by a decrease in predator consumption rates. This in turn will result in lower predator production, and the predator abundance will move back toward its carrying capacity. In an opposite response, a decline in predator population size when it is close to its carrying capacity will be compensated for by a comparative increase in average consumption rates, which will bring the predator back toward its carrying capacity. A population at its carrying capacity is a stable population.

On the other hand, if the predator is far from its carrying capacity for a given prey, the situation is very different. An increase in predator biomass will lead to an increase in prey mortality rate. In Ecosim terminology, the vulnerability factor for the prey will be high. The consumption rate of the predator will remain relatively constant, and the increase in its biomass will manifest itself in population growth. There will be only limited compensatory effects.

In general, it is not possible to estimate vulnerability factors from field or laboratory data. However, to assist with identifying appropriate settings, Ecosim includes several methods of estimation (see Christensen *et al.*, 2004 for details on these methods), and it is recommended that vulnerabilities be estimated based on time-series analyses, *i.e.*, by evaluating how groups in the ecosystem has reacted to changes in the past.

Time-series data for model calibration are thus essential for developing and validating an Ecosim model. Therefore, time-series data depicting trends in relative and absolute biomass, fishing effort by gear type, fishing and total mortality rates, and catches for as long a period as possible should be viewed as additional data requirements.

### ***Using Ecosim for stock reduction analysis***

Kimura's 'stock reduction analysis' (SRA, Kimura *et al.*, 1996) can be used to analyze long-term data in stock assessment. Historical catches are treated as fixed, known quantities, and are subtracted from simulated stock size over time to aid in estimating how large (and/or productive) the stock must have been in order to have sustained those catches and to have been reduced by some estimated fraction from its historical level.

A drawback of treating catches as fixed values, as is commonly done in stock reduction analysis (SRA), is that catches arise from the interaction of fishing effort and stock abundance. Ignoring this dynamic interaction amounts to treating the catches as purely dispensary impacts on stock size. Consequently, the fixed catches can cause progressively larger calculated fishing mortality rates (F) if simulated stock size declines. This can lead to a rapid collapse in the simulated stock, unreflective of what may have happened in the real system.

A modified version of Kimura's SRA is used in Ecosim. This allows catch series data to be treated as a forcing input (with simulated F calculated each year as  $[\text{input catch}] / [\text{simulated stock size}]$ ) or be used for evaluating model fit where F values are available from assessments. This SRA is used where estimates of initial biomasses and time trends in stock size are unavailable, and where no 'drivers' (such as effort or fishing mortality) is available to force the simulations.

### **3.1.4 Addressing uncertainty**

The EwE model presently incorporates several approaches for explicitly addressing uncertainty, but given the number of parameters involved in a complex model such as the CBFEM it is not possible to consider the uncertainty of all input parameters on all output parameters and all predictions. We rather recommend that focus is on what impact uncertainty with regard to key input parameters as well as of the model fitting procedure (evaluating alternative fitting procedures, see e.g., Walters *et al.*, in press) have on key model predictions.



- A ‘pedigree’ routine for characterizing the origin of input data and for developing an overall index of model quality
- The Ecoranger routine for explicit consideration, in a Bayesian context, of the uncertainty inherent in all input and its impact on estimated parameters
- A formal sensitivity analysis for documenting the effect of inputs on estimated parameters
- A Monte Carlo routine that can be used in the time-dynamic module to evaluate the effect of parameter uncertainty on policy questions

One should keep in mind that any given model represents only one possible synopsis of the trophic and other ecological interactions of species and functional groups of interest to fisheries and environmental managers. Developers should only attempt to adjust parameters in a way that makes sound ecological sense, rather than relying on the software’s automated tweaking, which potentially could develop a model parameterization that fits the data extremely well but may produce spurious results because of unsound assumptions fitting the noise rather than the signal (Walters and Martell, 2004).

Upon finalizing the CBFEM, Monte Carlo simulations (accounting for uncertainty in the parameterization) were run to determine if the patterns produced by the model were robust to parameter uncertainty. Further details on how EwE deals with parameter uncertainty are described in the software user guide (Christensen *et al.*, 2005).

### **3.2      *Development of an Ecopath Model of the Chesapeake Bay***

The construction of an Ecopath with Ecosim model for the Chesapeake Bay has been under way for several years. The effort has involved a large number of scientists from the Chesapeake Bay area, supported by modelers from the University of British Columbia. An initial workshop was held in October 2001 to introduce the Ecopath/Ecosim modeling approach to the Chesapeake Bay research community. A review of the Fisheries Ecosystem Model prototype was conducted to look for gaps in parameters, missing trophic linkages, and potential data sources to address concerns. A major focus of the workshop was to formulate research questions that can be addressed by ecosystem modeling (Table 2). This report addresses some of the questions in Table 2.

An introductory seminar/lab course on the use of the Ecopath portion of the EwE software was conducted in February 2002, with a follow-up seminar on dynamic simulation modeling in May 2002 at the Smithsonian Environmental Research Center, Edgewater, Maryland. A second workshop was held in May 2002 at the Virginia Institute of Marine Science, Gloucester Point, Virginia, to further develop the Fisheries Ecosystem Model and discuss its parameterization.

On April 28-29, 2003, a modeling workshop was held at the U.S. Fish and Wildlife Service Patuxent Research Refuge’s National Wildlife Visitor Center in Laurel, Maryland, to assess progress on the FEM to date. Workshop participants generally agreed:

- The FEM, as was implemented with 45 major species groups, reproduces many of the important time-series trends well;

- There remained a need to incorporate (or link to) water-quality parameters, abiotic processes, and lower trophic level dynamics; and
- There will be tradeoffs among many of the stated objectives of the *Chesapeake 2000* agreement. With finite ecosystem resources, it is unrealistic to believe that all fish species can be returned to their historic peak levels of abundance.

Other noteworthy results from the 2003 workshop include:

- Adjustments were made to the parameterization of species at lower trophic levels to secure both better data quality and better resolution;
- More reliable data were incorporated for the biomass of zooplankton and oyster groups; and
- Abundance indices and a new life history stage were introduced for oysters. The intention was to capture dynamics and mortality differences influenced by oyster population ontogenetics.

A draft of the CBFEM report was finalized in 2004, and has been subsequently subject to review and ensuing updating. Since the 2003 EwE modeling workshop, time-series data have been revised and added to the model, data sources have been verified, and data has been reviewed by a panel of Bay researchers, the Ecosystem Modeling Technical Advisory Panel 2. Ecopath with Ecosim 6 software version 6.0.4.2 was used to run this current version of the CBFEM, with a resulting goodness-of-fit criterion, the sum of squares, equaling 1098. The model currently uses more than 90 time series.

### 3.2.1 System boundaries

The FEM focuses on modeling fisheries stocks within the Chesapeake Bay and tidal portions of its tributaries. Some groups being modeled, such as black drum, reside in the Chesapeake Bay but are considered part of larger ‘stocks’ usually encompassed by the eastern or northeast United States. Further complicating matters, many of the groups spend only part of the year or different parts of their life histories within the Chesapeake Bay, such as weakfish or bluefish. Therefore, in order to derive time series for EwE time simulations, it was often necessary to develop assumptions and correction factors such that stock assessments for a larger population could be applied to the Chesapeake Bay EwE model.

### 3.2.2 Basic parameters and catches

The basic parameters for Ecopath models, with their units of measurement and commonly used abbreviations in parentheses, are:

- biomass ( $t \cdot km^{-2}$ , B)
- biomass accumulation ( $t \cdot km^{-2} \cdot year^{-1}$ , BA, default value 0)
- consumption per unit biomass ( $year^{-1}$ , Q/B)
- detritus import ( $t \cdot km^{-2} \cdot year^{-1}$ )
- diet composition of species ‘i’ from prey ‘j<sub>1</sub>, j<sub>2</sub>, j<sub>3</sub>, ..., j<sub>n</sub>’ (fraction, DC)

- ecotrophic efficiency (EE)
- fishery landings and discards by gear sector ‘j’ imparted upon each species ‘ $j_1, j_2, j_3, \dots, j_n$ ’ ( $t \cdot \text{km}^{-2} \cdot \text{year}^{-1}$ ).
- fraction unassimilated food (GS, default value 0.2)
- net migration ( $t \cdot \text{km}^{-2} \cdot \text{year}^{-1}$ , NM, default value 0)
- production divided by consumption (P/Q, only used if P/B or Q/B not given)
- production per unit biomass, *i.e.*, total mortality ( $\text{year}^{-1}$ , P/B, *i.e.*, Z)

When developing an Ecopath model, the diet compositions (DC) for all species must be entered. Gear sectors are designated by the modeler/user, who must enter catches and discards by gear type upon species or species groups.

Of the four basic input parameters B, P/B, Q/B, and EE, one may be left as unknown, because “the Ecopath model ‘links’ the production of each group with the consumption of all groups” (Christensen *et al.*, 2004) based on the trophic relations mapped out by the DC.

Typically, building an Ecopath model emphasizes collecting data for three of the four basic input parameters listed above: B, P/B, and Q/B. Other basic input parameters are usually not as well understood for most modeled species. In the case of fraction unassimilated food, 0.2 is set as a default value based on the experiments of Ivlev (1961). While this estimate may be appropriate for carnivorous fish, it is typically too low for many herbivorous species. This is especially true for species relying on low-energy food, notably zooplankton, where a value of 0.4 results in more appropriate respiration/biomass ratios (Christensen *et al.*, 2004). Because P/B and Q/B are usually entered, the P/Q rate will be set by the ratio of inputs set by the modeler for those values. However, if the modeler is incapable of providing an estimate of either P/B or Q/B, then the P/Q ratio may be entered instead. Because of the definition of P/Q, high-trophic-level predators with low production should have low P/Q values, (e.g.,  $\approx 0.05$ ), while low-trophic-level, highly productive organisms will tend to have high P/Q ratios, (e.g.,  $\approx 0.3$ ) (Christensen *et al.*, 2004).

Ecopath models are ‘snapshots’ that are intended to serve as the basis for time-dynamic Ecosim simulations. For this reason, the BA may be entered to represent the rate at which biomass is increasing or decreasing for the species group modeled: Ecopath models do not assume steady-state. This may especially be required in order to improve Ecosim simulations. ‘Detritus import’ is only of concern to the detritus group, and can therefore be omitted for ‘living’ groups.

Given these parameter characterizations, most Ecopath modelers prefer to leave the default values for ‘fraction unassimilated food’ and BA, adjusting these values only for species that have documented evidence suggesting different values. Because many species have not been studied in enough detail to yield published estimations of B, P/B, or Q/B, the modeler may let Ecopath estimate one as an unknown while estimating the others. In such cases, remember that P/B and Q/B values to some degree scale with allometric relationships, and therefore are conservative for similar species in similar ecosystems. This implies that even if a reliable P/B or Q/B estimate is unavailable for the species or species group modeled, then estimates for similar (or the same) species in similar (or the same) ecosystem may have to

suffice as proxies. Where possible, these may be modified up or down to reflect differences in exploitation pressure. Where biomass estimates are unavailable, they can be left for Ecopath to estimate given that the modeler can provide a value for EE, *i.e.*, the fraction of production used in the ecosystem (Christensen *et al.*, 2004).

One final aspect about general parameterization and grouping species in an Ecopath model – species may be modeled as one of three types: 1) an aggregation of trophically similar organisms, *i.e.*, a ‘functional’ group; 2) a single-species group; or 3) as a life history stage that is part of two or more groups representing life history stages of a ‘multi-stanza’ group. Generally, species to be examined in terms of policy questions are best dealt with as single-species or multi-stanza groups. Multi-stanza groups are preferred if there may be ontogenetic issues in the species’ ecosystem role that could play a part in the policy issues to be examined. To ensure that the biomasses for the different age-groups are consistent, Ecopath will estimate the stanza biomass and consumption rates for all stanza when the following parameters are supplied: the von Bertalanffy growth (curvature) parameter K (which is available for fish species through FishBase); B for one (‘leading’) stanza; estimates of Z (= P/B) for each stanza; Q/B for one stanza; and an estimate of the ratio of the weight-at-maturity to the asymptotic weight,  $W_{inf}$ . For a discussion of the calculations used in the Ecopath model, see Christensen and Walters (2004).

In most models, there will be a higher degree of aggregation in species that are trophically distant from the focal species. The desire to enrich the model with detail must be tempered by a realistic examination of the modeler’s ability to flesh out that detail and to obtain data or estimates for the required parameters. To examine specific policy issues for any particular species, detailed information must be available from surveys or assessments, and similar information should be available for species with which the focal species likely interacts in the environment. In particular, well-documented diet composition data and time-series data of biomass, natural and fishing mortality, fishing effort, and average weight are required to explore ecosystem relations when applying the time-dynamic Ecosim model.

Input parameters for the models are described in Appendix A. The four basic input parameters – biomass ( $t \cdot km^{-2}$ ); production / biomass (P/B,  $year^{-1}$ , corresponding to total mortality, Z); consumption / biomass (Q/B,  $year^{-1}$ ); and catches ( $t \cdot km^{-2} \cdot year^{-1}$ ) – are described for each functional group in the model, along with a description of how diet compositions were obtained. A summary of basic input parameters of the Chesapeake Bay Fisheries Ecosystem Model is presented in Table 3.

### **3.3      *Development of an Ecosim Model of the Chesapeake Bay***

#### **3.3.1      Time periods covered**

The strength of any model to be used for testing management action outcomes is measured by how well it is validated based on observed data for that system. This is true whether the model is a traditional single-species or a multispecies model. A 1950-Ecopath model was created to represent a snapshot of roughly what the Chesapeake Bay system may have looked like in the middle of the 20<sup>th</sup> century. This model was then run time-dynamically using Ecosim and tuned to observed data or to data estimated from other models for the time period 1950-2002 to estimate changes in biomass over 50 years. Descriptions and sources for the various time-series data used in driving, tuning, calibrating, and verifying the model are included in Appendix B of this report.

### 3.3.2 Primary production rate forcing: the Chesapeake Bay Regional Estuarine Ecology Model (CBREEM)

At the inception of the Chesapeake Bay Fisheries Ecosystem Model project, EwE did not have the ability to account for physical and chemical factors as a part of the ecosystem being modeled. The Ecospace component of the EwE modeling approach has been enhanced to alleviate this shortcoming. This is done by linking the Ecospace model to the Florida Bay Ecosystem Model (FBEM). The FBEM is a simple 2D-hydrographic model that was developed for Florida and Tampa Bays by Carl Walters (UBC) during a series of workshops sponsored by the South Florida Water Management District, The Nature Conservancy, and the Florida Keys Water Quality Joint Action Group. The FBEM was developed to provide a simple, calibrated model of salinity, nutrients and oxygen dynamics in Florida Bay (Walters and Gunderson, unpublished results).

While there are complicated hydrographic models for the Chesapeake Bay producing similar estimates with high spatial and temporal resolution, we have been unable to obtain information about nutrient conditions for the entire Bay over the long time period considered here (*i.e.* back to 1950). The existing models either only cover short, recent time periods or are geographically limited to a portion of the Bay. Hagy *et al.*, 2004 was used in the development of CBREEM. This study does cover the time period from 1950-2001, however it only describes nitrogen loading for the Susquehanna River and not the entire Bay. It was therefore of interest to emulate the Florida models to simulate long-term variability in nutrient conditions.

To this effect, we have developed the Chesapeake Regional Estuarine Ecology Model (CBREEM), which includes three sub-models, operating with monthly time steps: 1) a physical, or hydrodynamic, model computing current speed and direction for surface and deep layers; 2) a static chemistry model estimating concentration of nitrogen, salt, phytoplankton, and suspended particulates; and 3) a dynamic ecological model simulating growth and mortality of sea grass and epiphytic algae. CBREEM output provides a historical pattern in primary production, which has been used to drive the primary production rate in the CBFEM. CBREEM is a simple hydrographic model with two layers, deep and shallow. The horizontal spatial resolution is 1-2 km with monthly time steps covering 50+ years. The model solves for equilibrium velocity fields and calculates mass-balanced chemical concentrations. The model uses wind, rainfall, river inflow, and relative loading as major inputs. The outputs include spatial distributions of nutrient, salinity, and chlorophyll-a at monthly intervals.

Historical information on nutrient loading and physical mixing has been added to the CBREEM to calculate changes in primary production in the Chesapeake Bay. Monthly chlorophyll-a output from CBREEM was used to drive primary production rate in the Ecosim model. Both the CBREEM and supporting data are presented in a manuscript by Ma *et al.* (in prep).

The hydrodynamic model requires time-series data on wind vectors, river gauge data from major freshwater inputs, and bathymetry data, as well as some basic water chemistry information. Hydrodynamic model outputs include a time series of total primary production, nitrogen, oxygen, and salinity concentrations. These outputs have been compared with historical data to aid in model parameterization. The chemistry model calculates concentration of nutrients and estimates phytoplankton growth as a function of nitrogen concentrations. The sea grass submodel predicts total biomass and spatial distribution of sea grasses in response to changes in water chemistry and light penetration. Future efforts to improve the CBFEM will include a historical reconstruction of sea grass

communities in the Chesapeake Bay. This will serve to test hypotheses about how changes in sea grass beds have affected biota that are associated and/or dependent on sea grasses.

### **3.3.3 Catches**

For many species, catches are extracted from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the Chesapeake region, 1950-2002. The Chesapeake region includes Maryland and Virginia catches, including catches made on the Atlantic Ocean side of the Delmarva Peninsula. The CBFEM aims to characterize stocks within the Chesapeake Bay and tidal portions of its tributaries, not coastal stocks or freshwater stocks. In many cases, the catches have not been corrected for this discrepancy due to a lack of correction factors; however, this does not contribute any major bias to the analysis. Many of the species in the model for which this may potentially be of concern are migratory species that spend a major part of the year in the Chesapeake Bay. Typically, they spend only a limited part of the year in the coastal waters of Maryland and Virginia outside the Bay, and the catches there as a rule will be similarly limited.

While the NOAA marine catch database provides estimates for commercial catches from 1950 to the present, similar time series for recreational catches are not as readily available. The official recreational catch database, the Marine Recreational Fishery Statistics Survey (MRFSS), only includes information from 1982 to the most currently reported year; hence, recreational catches for the period 1950-1981 are derived from estimates. Commercial versus recreational catches were plotted for 1982 to the present, and the data was checked for trend. If a trend was clear, which rarely was the case, commercial versus recreational catches were regressed, and the pre-1982 recreational catches were estimated from the regression. Where no trend was detected, either the arithmetic mean (when there were few outliers) or the median value of the commercial/recreational catch rate was used to estimate the recreational catches for the earlier time period. For the time period from 1982 to the present, the commercial/recreational catch ratio, as estimated from the catches, was always used. The estimates of commercial and recreational catches are Atlantic coast-wide for many species and, for lack of better estimates, considered representative for the Chesapeake Bay as well. Catch series data are presented in Tables 4 and 5.

### **3.3.4 Time-series information**

For many groups in the Chesapeake Bay model, there is time-series information available from catch monitoring, surveys, and stock assessments that can be incorporated into EwE simulations. EwE builds on the more traditional stock assessment, using much of the information available from traditional assessments, while integrating to the ecosystem level.

The time-series fitting uses either fishing effort or fishing mortality data as driving factors for the Ecosim model runs. A statistical measure of goodness-of-fit to the time-series data outlined above is generated each time Ecosim is run. This goodness-of-fit measure is a weighted sum of squared deviations (SS) of log biomasses from log predicted biomasses, scaled in the case of relative abundance data by the maximum likelihood estimate of the relative abundance scaling factor  $q$  in the equation  $y = q \cdot B$  ( $y$  = relative abundance,  $B$  = absolute abundance). Each reference data series can be assigned a relative weight representing a prior assessment of relative data reliability.

The model allows four types of analysis with the SS measure:

1. Determine sensitivity of SS to the critical Ecosim vulnerability factors by changing each one slightly (1%) then rerunning the model to see how much SS is changed (*i.e.*, how sensitive the time-series predictions ‘supported’ by data are to the vulnerability settings);
2. Search for vulnerability factors that give better ‘fits’ of Ecosim to the time-series data (lower SS), with vulnerabilities ‘blocked’ by the user into sets that are expected to be similar (the search is typically conducted on the most sensitive interactions, as identified above);
3. Search for time-series values of annual relative primary productivity that may represent historical productivity shifts affecting biomasses throughout the ecosystem; and,
4. Estimate a probability distribution for the null hypothesis that all of the deviations between model and predicted abundances are due to chance alone, *i.e.*, under the hypothesis that there are no real productivity anomalies.

In addition to these nonlinear optimization routines, the fit to data can also be improved in a feedback process by examining some of the crucial ecological parameters in the EwE model (notably total mortality rates and the settings for top-down/bottom-up control). Such fitting does not include any ‘fiddling factors’ internal to the model. Instead, the type of question addressed after each run is, “Which species parameters or ecological settings are not set such that the model adequately captures the observed trends over time?”

The inclusion of time-series data in EwE facilitates the model’s use for exploring policy options for ecosystem-based management of fisheries. Analyses in this report illustrate how the model can be used to address some of the policy questions defined by workshop participants (Table 2). Further development and policy exploration activities will be carried out by NOAA Chesapeake Bay Office staff in cooperation with local experts and fisheries managers.

Time-series information for use with EwE can be of the following types:

- For functional groups
  - Biomass information (does not need to cover all years in the time series)
    - Relative biomass series: Can be from surveys, assessment, *etc.*
    - Absolute biomass: Rarely used, as it assumes that the absolute values (per unit area) are estimated in the same manner for the original data and for the ecosystem model. Absolute data, as a rule, are entered as relative data instead, using only the trend in the data for the fitting.
    - Biomass for forcing: Used to force the simulation at each time step; typically used for groups whose dynamics depend on processes that are not covered by the ecosystem model.
  - Fishing mortality: Used to ‘drive’ the Ecosim model and needs to be entered for all years of the time series.

- Total mortality: Used to compare how the simulation matches the observed data; data set need not cover all years.
- Catches.
  - Comparison of model simulation and observed data or for estimating fishing mortalities based on stock reduction analysis; data need not cover all years; time-series catch information is presented in Tables 4 and 5.
  - As part of a stock reduction analysis, where calculations in Ecosim are made for each time step of growth, mortality, and recruitment, and the catches subsequently are used to estimate a fishing mortality (catch/biomass), which is applied as well.
- Average weight: Used to compare observed and estimated weights for multi-stanza groups.
- For fleets
  - Effort data by gear type: Expressed as relative to the effort in the first year of the time series; used to ‘drive’ the Ecosim model; effort data need to be complete for the time series.
- Environmental data
  - Time-forcing data: Typically relative primary production (monthly or annual) over the time period, but can be any kind of environmental data as long as it can be related to the productivity for a group .

The actual procedure applied for fitting the model to time series can be summarized as follows:

- The primary production forcing time series from the spatial, hydrodynamic model described elsewhere in this document was used to force the system productivity in the fitted run.
- The ‘fit to time series’ interface was used to search for the most sensitive interactions in the model, *i.e.*, those interactions for which the vulnerability setting has most impact on the summed squared residuals between time series and the simulation.
- Catches and estimates from juvenile surveys were not included in the search, *i.e.*, their weighting factor was set to 0.
- The 25 most sensitive consumers were selected, and vulnerability factors were estimated for these groups. Vulnerability factors exceeding 100 were truncated at this value.
- These groups were then analyzed, and if comparison of time series and trend from time series (or the expected trend where there were no time series) warranted it, the vulnerabilities for the group in question were manually changed to improve time series fit.



The vulnerabilities were always changed by consumer group, *i.e.*, only one parameter was used per consumer (and for some groups vulnerabilities were not changed at all).

Information on time-series data is included in Appendix B of this report.

### **3.4 Stock Assessment Methods for Developing Time Series**

A requirement for the EwE reconstruction approach is to provide at least one (more is preferable) biomass input into Ecopath as well as historical information on the removals or fishing mortality rates. Biomass data can be obtained from single-species assessment models. In cases where model groups are partitioned into multiple life-history stanzas, an age-structured model is preferable so that fishing mortality rates for each stanza (if applicable, *e.g.*, certain fishing gears harvest a specific stanza) can be calculated from the estimated age-composition. Typical statistical catch-at-age models are notorious for having hundreds and sometimes thousands of parameters. However, this “over-parameterization” is not necessary (Walters and Martell, 2004).

For many Chesapeake Bay species, there is insufficient data to carry out detailed biomass assessments, and a simpler method is needed. Two stock assessment methods were used to develop time series for the Chesapeake Bay Fisheries Ecosystem Model, both based on production modeling: an age-structured single-species model and a multispecies production approach. These two methods are described in the following two sections and are applied to the trophic groups in the appendices of this report.

#### **3.4.1 Stock Reduction Analysis (SRA)**

In this section we briefly describe the derivation of an age-structured single-species assessment model, which requires few parameters and can be used to estimate biomass for Bay species. This model is parameterized with two leading (unknown) parameters that are equivalent to the maximum intrinsic rate of growth and the carrying capacity of a simple surplus production model. These two leading parameters represent the long-term unfished biomass ( $B_o$ ) and the maximum juvenile survival rate or recruitment compensation. For the estimation of the long-term unfished biomass, we rely heavily on meta-analytical results of Myers *et al.* (1999) to provide prior information for recruitment compensation at low spawning abundance. This is especially important in cases where relative abundance indices lack sufficient contrast to make it possible to estimate both parameters. A notable example is oysters in the current model. Without more diverse trajectories, notably decline and subsequent recovery, we cannot tell if a given harvest was taken from a large population with low productivity or a small population with high productivity.

For bluefish and several other species, an age-structured assessment model was used to reconstruct historical biomass and a time series of fishing mortality rates, which were used to force Ecosim simulations. Input data for the assessment model include: growth information (von Bertalanffy growth parameters), length-weight relationships (*i.e.*,  $w_a = aL^b$ ), parameters for a maturity cumulative frequency curve to calculate spawning stock biomass, natural mortality rate estimates, and parameters that describe size selectivity. Model parameters were estimated by fitting the model to abundance data and to catch rate information. Each of the abundance indices was assumed to be proportional to stock size, and observation errors were assumed to be lognormal. The age-structured population model includes a Beverton-Holt type stock recruitment function. The model was parameterized using a leading parameter setup, where the population scale (or capacity) was determined by  $R_o$  (the equilibrium unfished recruits),

and the maximum rate of population change was defined by a recruitment compensation parameter ( $k$ ). In most cases, only observation errors were assumed.

### 3.4.1.1 Equilibrium conditions

We start out by assuming that the age-structure is in equilibrium, *i.e.*, that the population structure has been stable over the years prior to the first model year. Beginning with the Beverton-Holt recruitment model:

$$R_e = \frac{\alpha E_o}{1 + \beta E_o} \quad (6)$$

the two parameters ( $\alpha$  and  $\beta$ ) can be derived given initial estimates of  $R_o$ ,  $M$ , and  $k$ . The maximum survival rate ( $\alpha$ ) is simply a multiple of number of recruits produced per unit of egg production, or:

$$\alpha = K \frac{R_o}{E_o} \quad (7)$$

and the asymptote of the recruitment function is defined by:

$$\beta = \left[ \frac{\alpha E_o}{R_o} - 1 \right] / E_o \quad (8)$$

The equilibrium egg production ( $E_o$ ) is the product of the equilibrium recruits and the number of eggs produced per recruit. The number of eggs per recruit ( $\phi_e$ ) is the product of survivorship to age  $a$  times mean fecundity of age  $a$  individuals. It is not necessary to know the exact fecundity of any specific age group, but rather the relative differences in fecundity between separate age classes. Here, it is assumed that egg production is proportional to body weight. The equilibrium egg production ( $E_o$ ) for a population at equilibrium is calculated as follows:

$$E_o = R_o \phi_e = R_o \sum_{a=0}^{\infty} (e^{-M})^a w_a m_a \quad (9)$$

where  $w_a$  is the weight-at-age and  $m_a$  is the proportion of that age class that is sexually mature. A simple logistic function is used to describe maturity-at-age:

$$m_a = \frac{1}{1 + e^{-g(l_a - l_h)}} \quad (10)$$

where  $g$  is a shape parameter that describes the variation in maturity-at-age,  $l_h$  is the length at 50% maturity, and  $l_a$  is the mean length-at-age.

### 3.4.1.2 Population dynamics

The numbers-at-age ( $N_a$ ) matrix is initialized assuming a stable age distribution, and the oldest age class ( $A$ ) is a plus group containing individuals ages  $A$  and older:

$$N_a = \delta R_o (e^{-M})^a \quad (11a)$$

$$N_A = \delta R_o \frac{(e^{-M})^A}{1 - e^{-M}} \quad (11b)$$

The  $\delta$  parameter is constrained to the interval [0-2] and represents the ratio of initial numbers to the unfished equilibrium numbers. Numbers-at-age are propagated over time using historical catch information and size selectivity to calculate age-specific fishing mortality rates. Since our interest was to develop a fishing mortality rate time series to force Ecosim, annual fishing mortality is conditioned on observed total catch:

$$F_t = \frac{C_t}{B_t} \quad (12)$$

where  $C_t$  is the observed total catch from all fisheries combined and biomass is simply the product of numbers-at-age times mean weight-at-age. Given predictions from equation (12), numbers-at-age are updated using:

$$E_t = \sum_{a=0}^A N_{t,a} m_a w_a \quad (13a)$$

$$N_{t+1,1} = \frac{\alpha E_t}{1 + \beta E_t} e^{\omega_t \sigma} \quad (13b)$$

$$N_{t+1,a+1} = N_{t,a} e^{(-M - F_t v_a)} \quad \text{for } a < A \quad (13c)$$

$$N_{t+1,A} = N_{t,A-1} e^{(-M - F_t v_A)} + N_{t,A} e^{(-M - F_t v_A)} \quad \text{for } a = A \quad (13d)$$

Equation (13a) represents the total egg production in year  $t$ , and equation (13b) is the Beverton-Holt recruitment function; note that process errors  $\omega_t$  may be included if  $\sigma > 0$ . The instantaneous natural mortality rate is represented by  $M$ , and the vulnerability-at-age ( $v_a$ ) is calculated using the same logistic function in equation (10). However, separate parameters ( $g$  and  $l_h$ ) are used and unless otherwise noted are fixed values (*i.e.*, not estimated).

### 3.4.1.3 Estimating model parameters

Model parameters were estimated by fitting the models to time-series data on relative abundance and composition information, if available. All abundance indices were assumed to be proportional to stock size or a specific component of the stock, such as age-0 recruitment indices. Observation errors were assumed to be log-normally distributed. In the case of relative abundance indices, the observation model is:

$$Y_t = qX_t e^{v_t} \quad (14)$$

where  $X_t$  is the predicted biomass or age group or population numbers (depending on what the observation  $Y_t$  represents), and  $q$  is simply a scaling parameter or the slope of the regression between  $Y$  and  $X$ . The scaling parameter,  $q$ , is a nuisance parameter (a parameter that is fundamental to the model, but of no particular interest in itself), and the model simply integrates over this parameter as well as the variance in the observation errors using the methods suggested by Walters and Ludwig (1994). The model uses the maximum likelihood estimates for  $q$  and the variance in the likelihood kernel; thus, each independent observation series is weighted by the relative standard deviation in the observation errors. The corresponding negative log-likelihood is:

$$\ln l_Y = -\frac{(n-1)}{2} \sum_{i=1}^n (Z_i - \bar{Z})^2 \quad (15)$$

where  $Z_t = \ln(Y_t/X_t)$  and

$$\bar{Z} = 1/n \sum Z_t = \ln(q) \quad (16)$$

In cases where catch-at-age information is available, a multinomial likelihood is added to the overall objective function. Here, it is assumed that no aging errors exist and that the catch-at-age composition is representative of the age-structure in the Chesapeake Bay region. The negative log-likelihood for the multinomial distribution is:

$$\ln(l_a) = -\sum_{t=1}^T \sum_{a=1}^A n_{ta} \ln(p_{ta}) \quad (17)$$

where  $n_{ta}$  is the observed numbers-at-age in the catch sampling programs and  $p_{ta}$  is the vulnerable proportion-at-age based on the numbers-at-age and vulnerability schedule in the population dynamics model.

For the majority of the assessments, only observation errors are assumed and the unknown parameter set is limited to ( $R_o$ ,  $k$ , and  $\delta$ ). In cases where catch-at-age data were available, parameters for the selectivity function ( $g$  and  $l_h$ ) are also estimated. Neither process errors nor recruitment anomalies are estimated in any of the assessment models.

### 3.4.2 Multispecies Production Method (MSP)

A series of Bay assessments (striped bass, weakfish, bluefish, white perch, spot, Atlantic croaker, Atlantic menhaden, bay anchovy, gizzard shad, and oyster) were developed to provide estimates of exploitable biomass to compare with EwE estimates. The boundary of these analyses, the mouth of the Chesapeake Bay, was not a physical boundary to many of the species modeled. This analysis was conducted in the context of a closed Chesapeake Bay population, which may not be realistic, but provides useful information nonetheless.

Gulland (1988) considered definition of a unit stock an essentially operational matter, being tied to the models used, the questions asked, and the information available. When the bounds of the unit stock

extend beyond the limits of the fishery being analyzed, then the pattern of exploitation beyond the limit of analysis will determine whether the analysis of portion of the stock will be misleading. If the fishery outside the boundary is similar to that inside, correct answers may be provided (Gulland, 1988).

#### **3.4.2.1 Calculation of indices of abundance**

Relative biomass indices for many Bay finfish were developed from the Marine Recreational Fisheries Statistics Survey (MRFSS) database. Prior to 1981, recreational harvests were not estimated, and only species that were exclusively harvested by commercial fisheries could be assessed. MRFSS data are available back to 1981. Time-series end points were variable (2000-2004) and depended on when the first version of the spreadsheet model was developed as much as data availability. There are no long-term alternative fishery-independent surveys of exploitable biomass for the Chesapeake Bay. Commercial effort data in Maryland begins in 1980, but is discontinuous (lost tapes). Virginia can likely generate recent effort from its trip ticket system. PRFC has a long continuous catch and effort database.

These indices were developed as a catch-effort ratio for private boat anglers in Maryland and Virginia in the MRFSS inland fishing area (inshore saltwater and brackish water bodies such as bays, estuaries, sounds, *etc.*, excluding inland freshwater areas). These indices were generally calculated as  $EB = Hp / Ep$ ; where EB is relative exploitable biomass,  $Hp$  is private boat harvest, and  $Ep$  is private boat trips. Coastal bays are included in these totals, but these fisheries are usually minor compared to those of the Chesapeake Bay. All private boat trips were included in the denominator, but not all private boat trips were directed toward the suite of species indexed. It was assumed that the composition of trips (bottom fishing, trolling, casting, *etc.*) have not changed a great deal over time, so that biases in effort were not great enough to influence trends for a particular species. Private boat recreational fishing occurs over the entire Bay, and this index would be as close to a global survey as could be obtained. There is a possibility of hyperstability (catch per effort remains high even though the stock is declining) in these estimates; the recent dependence of the recreational striped bass fishery on chumming may have resulted in an inverse catchability-biomass relationship. These data are fishery-dependent, and the harvest component is contained in the landings data as well. A general recommendation for data in stock assessment is that information only be used once (Cotter *et al.*, 2004). In the case of an MRFSS harvest-based index, information is contained in both the landings and the index. However, division by effort in the index reduces the direct dependency in the data, and there is little alternative for a Bay-wide index.

Changes in length limits can affect age/size classes represented by these indices. Length limits have been fairly stable or nonexistent for spot, Atlantic croaker, white perch, and bluefish. Moderate increases in size limits have been imposed on weakfish and major changes have been imposed on striped bass. The short time series for striped bass estimates (1991-2003) represents a period of somewhat stable length limits.

#### **3.4.2.2 Biomass dynamic models**

Biomass dynamic modeling (also known as surplus production models) is the simplest analytical method that provides a full stock assessment (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; minimum data needed are an index of relative abundance and landings (both in weight). The stock is considered as undifferentiated biomass and age, size, and sex structure are ignored

(Haddon, 2001). Variations in exploitation and biomass are important for fitting the model; length of the time series is not (Hilborn and Walters, 1992).

The time-series fitting method was used to estimate production model parameters (Hilborn and Walters, 1992; Haddon, 2001). A spreadsheet version of the discrete form of the biomass dynamic model based on the logistic function was used:

$$U_t = U_{t-1} + rU_{t-1} \left( 1 - \left( \frac{U_{t-1}}{sK} \right) \right) - sC_{t-1} + e \quad (18)$$

where:

- $U_t$  was the index of abundance in year  $t$
- $U_{t-1}$  = index of abundance the previous year,
- $r$  = intrinsic rate of population increase
- $s$  = scalar for the abundance index
- $K$  = maximum population biomass
- $C_{t-1}$  = harvest (commercial and recreational) in the previous year
- $\varepsilon$  is measurement error (Hilborn and Walters, 1992)

Combined recreational and commercial directed harvest was used; discard estimates were not available. A genetic algorithm super solver (Evolver, Palisade Corporation) was used to estimate parameters  $r$ ,  $K$ ,  $s$ , and  $U_0$  (an estimated abundance index in the initial year of the time series) that minimized observation error ( $observed \ln U_t - predicted \ln U_t$ )<sup>2</sup> (Prager, 1994). The previous year's estimated index was then used to predict the following year's estimate.

The spreadsheet version of the model combined with the genetic algorithm provided a great deal of flexibility for trying different model variations. Auxiliary data such as tagging estimates of  $F$ ,  $M$ , or disease mortality were added in some versions of the biomass dynamic model. In some cases, the scalar  $s$  was the only parameter used from the biomass dynamic model and each observed index was divided by  $s$  to obtain a biomass estimate.

An observation error model was used that assumed all residual errors were in the index observations and that the logistic equation used to describe the time series was deterministic and without error (Haddon, 2001). Residuals were examined to see if they were normally distributed with a mean of zero and to see if serial trends were present.

Biomass of the exploitable stock in year ( $B_t$ ) was estimated as predicted  $U_t / s$  (Hilborn and Walters, 1992). Instantaneous annual fishing mortality rate in year  $t$  was estimated, based on Ricker (1975) as:

$$F_t = \frac{C_t}{(B_t + B_{t-1})/2} \quad (19)$$

### 3.4.2.3 Predator-prey modeling

Abundances of many exploited fish stocks vary widely, and these variations may not be explained with single-species population models (Spencer, 1997). Often environmental factors are sought as underlying causes for changes in status, but sustained periods of high and low abundance are often exhibited by fish populations that are much more dramatic than shifts in climate or other environmental factors (Steele and Henderson 1984). These shifts can be regarded as jumps between alternative equilibrium states of ecological systems (Steele and Henderson, 1984). Continuous changes in intrinsic population growth rate, carrying capacity, F, or rate of predation can lead to ‘flips’ into periods of high or low abundance (Spencer and Collie, 1996). A classical logistic population growth formula combined with an S-shaped predation function reproduces these fluctuations when subjected to simulated directional environmental fluctuations or changes in fishing pressure (Steele and Henderson, 1984; Collie and Spencer, 1993).

A simple predator-prey model was applied to examine the relative effects of fishing and striped bass predation and competition on recent (1981-2003) population dynamics of selected fish species in the Bay. This model provided an alternative to attributing all changes in biomass to F under stable ecological conditions. In addition, the ‘effect’ of striped bass was considered to include collateral damage such as starvation, cannibalism, or inability to occupy habitat due to competition, as well as direct consumption by striped bass. Results of this predator-prey model were contrasted with fishing mortality and biomass estimates from ‘straight’ biomass dynamic models.

When applied generally, this predator-prey model has reproduced the types of rapid shifts in abundance that have been exhibited by marine populations, and it was useful in exploring the role of dogfish predation on Georges Bank haddock recovery and management (Spencer and Collie, 1996; 1997). In the current weakfish assessment process, use of this model has allowed for exploration of food web dysfunction as a hypothesis for their decline (Uphoff, 2006). This predator-prey model is essentially a Schaefer biomass dynamic model with a sigmoidal type III predation function added to estimate additional predation losses (Collie and Spencer, 1993). In this analysis, it provided a method for quantifying the extent that striped bass predation and competition or fishing mortality could be influencing another species biomass.

The spreadsheet version of a Schaefer biomass dynamic model formulated by Haddon (2001) was used, and the type III predation function was added (Steele and Henderson, 1984; Collie and Spencer, 1993; Spencer and Collie, 1996). The predator-prey model used the following discrete time-step equation:

$$B_t = B_{t-1} + rB_{t-1} \left( 1 - \left( \frac{B_{t-1}}{K} \right) \right) - H_{t-1} - \left[ \frac{(cP_{t-1}(B_{t-1})^2)}{A^2 + (B_{t-1})^2} \right] + \varepsilon \quad (20)$$

where:

- $B_t$  was biomass in year t
- $B_{t-1}$  = index of 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
- $c$  is per capita consumption by striped bass biomass ( $P_{t-1}$ ) in the previous year
- $A$  is weakfish biomass where predator satiation begins

- $\varepsilon$  is measurement error (Hilborn and Walters, 1992; Collie and Spencer, 1993; Spencer and Collie, 1996)

In fitting the model to an index of biomass ( $U$ ), an additional catchability parameter ( $q$ ) had to be estimated;  $U_t/q$  or  $U_{t-1}/q$  would be substituted for weakfish biomass ( $B_t$  or  $B_{t-1}$ , respectively) in the predator-prey model equation. The Haddon version of the spreadsheet model was easier to adapt to this predator-prey formulation because it estimated biomass as a first step and then estimated  $q$  as  $B_t/U_t$  (Haddon, 2001). Biomass was estimated directly for 1981 ( $B_{1981}$ ), and then the model was used to estimate subsequent years (Haddon, 2001). Estimating biomass first allowed striped bass biomass to be used directly and parameter  $A$  to be estimated directly rather than converting biomass to index equivalents. Two estimates of 1982-2003 striped bass biomass were used—current estimates of coastal biomass and a Bay estimate. The Bay estimate (1982-2003) equaled annual coastal biomass estimate multiplied by the median percentage of coastal age 2+ biomass represented by a Bay estimate (biomass dynamic model using 1991-2002 MRFSS catch per unit effort (CPUE) index with auxiliary information on lesions, and  $F$  and  $M$  from tagging). This median percentage equaled 23% of the coastal biomass. This estimate tracked the coastal estimate in trend, but would represent the exploitable fraction of bass in the Bay.

An observation error model was used that assumed all residual errors were in the index observations and the logistic equation used to describe the time series was deterministic and without error (Haddon, 2001). A genetic algorithm super solver (Evolver, Palisade Corporation) was used to estimate predator-prey model parameters that minimized observation error in the indices ( $observed \log_e U_t - predicted \log_e U_t$ )<sup>2</sup> and auxiliary data used (Haddon, 2001). The spreadsheet version of the model combined with the genetic algorithm provided a great deal of flexibility for trying different model variations. Residuals were examined to see if they were normally distributed with a mean of zero and to see if serial trends were present.

Instantaneous annual fishing mortality rate in year  $t$  was estimated from Ricker (1975) as:

$$F_t = \frac{H_{t-1}}{(B_t + B_{t-1})/2} \quad (21)$$

An equivalent instantaneous natural mortality rate associated with striped bass predation and competition was estimated. The type III predator-prey term,

$$\frac{(cP_{t-1}(B_{t-1})^2)}{A^2 + (B_{t-1})^2} \quad (22)$$

estimated loss of biomass,  $D_t$ , was equivalent to catch  $H_t$ . Instantaneous annual natural mortality associated with striped bass was estimated as

$$M_{bt} = \frac{D_{t-1}}{(B_t + B_{t-1})/2} \quad (23)$$



Total instantaneous annual mortality of ( $Z_t$ ) due to due to fishing and striped bass predation equaled  $F_t + M_{bt}$ . This estimate did not account for other sources of natural mortality not associated with food web effects represented by striped bass.

## 4 Results and discussion

### 4.1 *Ecopath models*

The Ecopath model represents a possible configuration of the Chesapeake Bay in 1950; its groups are shown arranged by trophic level in Figure 4. The model configuration is mass balanced to the degree of ensuring internal consistency, *i.e.*, there is enough food for the consumers in the model, and enough production to meet any demand. This, however, is not the only possible configuration, and the Monte Carlo routines of EwE can be used to explore other possibilities. Using the Ecoranger routine of EwE, balanced models can be obtained through a resampling method with values drawn from the confidence intervals dictated by the pedigree values.

We used Ecoranger to obtain 200 balanced model parameterizations based on the confidence intervals obtained from the model pedigree. The resulting values were assigned to the original Ecopath parameters. There is a clear tendency to estimate higher available production for lower trophic levels and lower production for higher trophic levels. Model construction should ensure that the model is constrained from both the bottom up and the top down (biomasses and catches of higher trophic level groups).

One of the few remarkable results indicated by the Ecoranger runs is a lower biomass for black drum. The tendency for lower black drum biomasses is likely caused by difficulties in balancing the hard clam group, which in the original model has an ecotrophic efficiency very close to 1.

Trophic impact in the model can be explored through the mixed trophic analysis, via a process originally developed by Leontif (1951) to describe the American economy and later modified for ecological use by Hannon and Joiris (1989) and Ulanowicz and Puccia (1990). Mixed trophic impacts are shown in Figure 6 for groups of focal impact to fisheries, and the analysis serves to illustrate, for example, that alewife and herring have strong effects on many other groups, yet very little is known quantitatively about their history in the Chesapeake.

### 4.2 *Ecosim simulations*

Ecosim simulations were performed for the period 1950-2002 using default Ecosim settings except as noted below. Feeding time adjustment was not included in the model runs; thus, feeding time was assumed constant for all groups over time. The ‘fraction of other mortality sensitive to changes in feeding time’ was set to zero for the two older stanzas of striped bass to reflect that they are unlikely to be impacted much by predation. Time-series data was input as described in Appendix B, and covers most of the important groups or species in the system.

### **4.2.1 Vulnerabilities**

Vulnerability factor settings were estimated as described in the Methods section of this document, and aimed at changing as few parameters as possible. In total, vulnerability factors were changed for 23 groups, using only one setting for each consumer (*i.e.*, where a group had several prey types, and hence several vulnerability settings, the same value was used for all consumer-prey interactions). The time-series fitting routine of EwE could have been used to estimate vulnerability settings for all 218 diet components in the model. However, doing this would have increased the model's ability to fit the time series to the detriment of its predictive capabilities (Walters and Martell, 2004). Vulnerabilities for species groups included in the CBFEM are presented in Table 6.

The key criterion for changing vulnerabilities is that there must be availability of time-series information for the group in question. There must be time-series data for prey where members of the group are important predators, or for predators, where members of the group are important prey. For many groups, these criteria are not met because the data is unavailable. This limits current possibilities for using the model's use for predictive purposes. A lack of historic information limits our capability to project what will happen in the future. We emphasize that this is a property of all modeling, not a specific feature of EwE.

The vulnerabilities can be interpreted as a measure of how far a consumer is from its carrying capacity in the 1950-start situation. Thus, the default setting of 2 indicates that the given group would at most be able to double the predation mortality it is causing on its prey were its abundance to increase to its carrying capacity. For groups at their carrying capacity, the vulnerability should be closer to 1, indicating that the groups cannot increase the predation mortality they are causing on their prey.

### **4.2.2 Time-series fitting**

We fitted the model to the available relative abundance data available for the various groups. Catch data were used either for fitting the Ecosim simulation or to drive Ecosim conditioning on catch. We generally found that where long time series of data on catches, fishing mortality, or relative abundance exists, the model fits well. In the absence of long-term data to drive the simulations, the ability of the model to explain short-term ecosystem changes is unclear. Long-term time-series information is of utmost importance for ecosystem-based management of fisheries.

#### **4.2.2.1 Commercial fish**

##### **4.2.2.1.1 Striped bass**

For striped bass, the assessments used in the model only cover the period from 1982 through 2002. The fit to the biomass trend for 1982-2002 is good for all stanzas along with the fit to catches for the same period. To obtain this fit, it was necessary to assume that for both resident and non-resident striped bass, the fishing mortality was relatively high for the period prior to 1982, which leads to a marked overestimation of catches in recent years for both groups. We could have obtained a better fit to catches by increasing the 1950 biomass of striped bass considerably. However, we would then have been unable to make the striped bass return to their 'historic level,' as conventional wisdom dictates has occurred.

A major problem for striped bass fitting is that we overestimate the catches for the last 20 years, with high biomass even though the currently estimated rather low F-values lead to quite high catches. If we

instead fit the time series to catches and hence ignore the uncertain, high estimates we have for fishing mortality in the pre-1982 period, we can obtain reasonable fit to the biomass time-series trend, but in that case we obtain quite low fishing mortality estimates for the pre-moratorium period (*i.e.*, F values around 0.2-0.3 year<sup>-1</sup>, with a peak for migratory in the late 1970s and early 1980s where F values reached values of 2-3 year<sup>-1</sup>).

The current striped bass assessment and simulation agree that the increased fishing mortality in recent years is likely to have caused the stock trend to level off or decrease. The striped bass assessments need to be carried back further than 1982 in order to maximize the accuracy of model simulation.

Menhaden was assumed to contribute 52.5% to the diet of resident striped bass, and 68.5% to the diet of migrant striped bass in the 1950s. Due to decline in menhaden abundance over time, menhaden in the diet was reduced (dynamically by Ecosim) in recent years to around 20% to 40% for the different stanzas of striped bass.

The main conclusion for striped bass is that the assessments should be carried back further than 1982, even if it means digging deep to obtain estimates for recreational catch prior to the establishment of the computerized NOAA recreational catch survey system.

#### **4.2.2.1.2 Bluefish**

The bluefish assessment indicates the following general trend: An increase in biomass in the 1950s and 1960s associated with a reduction in fishing mortality, a peak in the mid-1970s, a gradual decline in the mid-1990s back to the 1950 level, and a small increase again in recent years. This trend is repeated closely by the Ecosim simulation, though the simulations tend to produce higher catches since the peak. The ease with which this general trend was reproduced can be attributed to the fact that the assessment and Ecosim are being driven by the same factor, fishing mortality; the adult group does not have predation mortality in the model, and it is minimal for the juveniles. Further, trophic interactions do not provide any confounding issues, because any trend seen in prey consumption by bluefish is countered by an inverse trend demonstrated by the feeding scheme of their main competitor, striped bass.

#### **4.2.2.1.3 Weakfish**

For adult weakfish, the assessment we conducted indicated a peak in abundance around 1970. While this peak is not reproduced fully in Ecosim, the trend for the rest of the time period is matched more closely. The driving force for the simulation is the fishing mortality estimated in the assessment. The simulations tend to overestimate the catches throughout the modeling period.

#### **4.2.2.1.4 Atlantic croaker**

The only two time series for Atlantic croaker are a juvenile trawl series estimate from VIMS going back to 1979, and an ASMFC 2003 stock assessment. The VIMS series is highly variable, with indications of a decreased biomass in recent years (associated with increased catches), while the ASMFC series indicates an increase since the early 1980s with a leveling off in the later years. The simulation is not conclusive, and there is little information on which to base the simulations, notably a lack of drivers in form of fishing mortality estimates from assessments covering the simulation time period. Indeed, we had to use forced catches to drive the group. We ran the simulations with a low-vulnerability setting for Atlantic croaker (based on a search using the time-series-fitting module designed for this purpose, indicating that the croaker would have been close to its carrying capacity in 1950).

#### **4.2.2.1.5 Black drum**

For black drum, the catches can be used for estimating fishing mortalities only if the ecotrophic efficiency in 1950 is assumed to have been very low. This assumption increases the 1950-biomass, which was required to allow the estimated catches to be extracted from the population. The resulting decline is a gradual erosion of the black drum biomass over time, possibly associated with decline in soft clam, one of its major prey items. We have no time-series information to evaluate the legitimacy of the finding.

#### **4.2.2.1.6 Summer flounder**

The simulation for summer flounder is impacted by a burst in the late 1970s, and the juvenile trawl series indicates a sharp decline in the 1980s (where there were no catches, according to the catch information), followed by a marked increase in the 1990s (where the catches returned to a lower level). An NFSC assessment gives a trend very similar to the VIMS survey, and indicates that the summer flounder had high fishing mortality in the 1980s with a more recent declining trend. This, however, does not match the catch information at hand, and using the assessment-F series for 1983 on, we overestimate the catches for the period since then. Given that the mortality is fully dominated by fishing mortality, we cannot use a lower initial biomass to reduce the more recent catches, as this would make it impossible to balance the group. Our conclusion is that the data for this group is inconsistent.

#### **4.2.2.1.7 Menhaden**

Menhaden is the only group for which a long-term assessment was available from the regular stock assessments conducted in the Chesapeake Bay region (ASMFC, 2003c). The Ecosim simulation replicates the biomass trend well, even if there is some uncertainty about conclusions regarding the early 1950s.

While it was a welcome surprise to find an assessment going as far back as 1955, if ASMFC would take one more step and continue their data back to 1950, the accuracy of modeling results for menhaden would improve appreciably.

In the simulations we tend to underestimate the catches of the YOY menhaden, while the biomasses are in line with the assessment, *i.e.*, do not show any clear trend over time. For the older age group, the simulation replicates the multispecies production method assessment remarkably well, and is also in agreement with the ASMFC assessment from the mid-1960s onward. We were, however, not able to replicate the marked decline in the late 1950s and first half of 1960s indicated by the assessment, and ascribe this to a lack of information for the earlier part of the 1950s, or possibly to data inconsistency in form of underestimation of the catches for the early period.

We compared the catches to two catch series, one representing coast-wide catch trends, and one trends in Bay catches. Our catch series initially matches the coast-wide series best, but since 1970 matches the Bay series very well.

#### **4.2.2.1.8 Alewife and herring, eel, catfish, white perch, and spot**

The simulations for alewife and herring, American eel, and catfish are all characterized by very little available information on which to drive the simulations and evaluate the results. We chose to condition the simulations on catch for these groups, because there were no realistic time-series trends available. As

such, the simulations mainly demonstrate the lack of information about what has happened to these groups in historical time.

For white perch we had two (quite similar) multispecies production method assessments covering the period from 1981. We conditioned the simulations on catches up to 1970, and used an F series from one of the assessments for the period since then. Based on this, we overestimated the catches with a factor of three or so, while the biomass trends were more in line with the assessments (and perhaps the trawl-series though this is very variable). We cannot explain the catch-dilemma: Given the F-values, we should have lower biomasses, but with lower biomasses we would not have enough white perch for the striped bass.

For spot we also needed drivers for the modeling, and we chose to use the trend from the VIMS survey series to fit a stock reduction model to the catch data. The Ecosim simulation fitted the biomass trend quite well when conditioned on the catch series.

#### **4.2.2.2 Commercial invertebrates**

##### **4.2.2.2.1 Blue crab**

The adult biomass trend is associated with a marked increase in total mortality for adult blue crab, up above  $3 \text{ year}^{-1}$ , which may be excessive. The simulation indicates a marked decline in the early 1980s associated with a sharp increase in catches at the time. This biomass decline is not reflected in the assessments (where biomass and catches increase in parallel), leading us to think that an external productivity factor may have positively impacted crab abundance at the same time as catches were increasing.

Juvenile blue crab biomass does not show any time trend in the Ecosim simulations; similar results are obtained from the juvenile trawl survey indices since the 1960s and 1970s. Total mortality for juvenile blue crab seems to be declining over time, due to lower predation pressure from adult crabs.

The fit to the catch series is reasonably good, indicating that the F and biomass estimates are internally consistent.

##### **4.2.2.2.2 Oyster**

For oysters, we had some recent trend data, and estimated the population trend back to 1950 from a stock reduction analysis. The model does not have predation on adult oysters, so it is not surprising that we were able to replicate the biomass and catch trends throughout the time periods quite closely – even without invoking any oyster mortality due to diseases. It is generally recognized that diseases are a contributing factor to the current poor stock status for oysters in the Chesapeake Bay, but we currently lack quantitative information that will allow us to incorporate diseases in the model. We consider evaluation of historical biomass, catch, and other mortality trends important for understanding the role of oyster in the Chesapeake Bay, and encourage such studies.

The CBFEM allows for exploration of management scenarios. As an example of such a scenario, we present the potential outcome of a fishing moratorium on the native oyster since 1950 in the Chesapeake Bay in Table 8.

#### **4.2.2.2.3 *Soft and hard clam***

Our simulations for soft and hard clam for both show a marked decrease over the simulation period, much in line with observations. However, time-series information about abundance is lacking, as is information about mortality rates caused by diseases (which are considered important). The decline of clams in the model is caused by the combined impact of catches and sedimentation caused by hurricanes. The simulations for clams should be considered very tentative.

#### **4.2.2.3 *Other groups***

For the remaining fish species, birds, and most invertebrates, we had no time-series information that could be used to constrain and validate the simulations. Hence, our model is not very illustrative for these groups; they serve mostly as ‘place-keepers.’ They are in the ecosystem and require resources for their sustenance, but their dynamic over time is difficult to evaluate. In no case do they markedly influence the groups for which we have more information.

### **4.2.3 *Uncertainty/sensitivity***

We used the Monte Carlo option in Ecosim to search for a better fit to the time-series data, drawing parameters from defined ranges based on the model pedigree. For this we included only time series for the key groups, thus excluding juvenile trawl survey estimates and catches in estimating the summed squared residuals (SS). We let the search routine conduct 500 Ecosim simulations (each involving up to several thousand iterations to find a balanced model), but were unable to find any constellation with lower sum squared residuals than we obtained through the fitting procedure.

## **4.3 *Evaluating policy questions***

Emphasis in this report has been on model validation—on examining model fit to qualify performance and to ascertain whether the model could provide plausible hypotheses for the ecosystem changes that occurred from 1950 to the present. If the model can successfully mimic system function and recreate historical trends, then it lends some credence to its prospective as a predictive tool. We will use the model to explore some policy questions. We emphasize that the examples we present in this report are for demonstration purposes—the range of questions that the model can be used to address is much greater.

### **4.3.1 *Predatory and forage fish ecosystem dynamics***

In a recent study, Walters *et al.* (2005) concluded that analysis of single-species versus ecosystem harvest strategies underlined the need to provide explicit protection for species whose value derives in part from support of other species as well as from harvesting. Harvesting all species at their single-species maximum sustainable yield (MSY) may lead to ecosystem erosion. With this in mind, we examined the role forage fishes play in the Chesapeake Bay ecosystem based on model simulations.

#### **4.3.1.1 *Menhaden and striped bass***

There has been a menhaden fishery in the Bay for many decades, and there is still considerable interest in harvesting both menhaden and one of its major predators, striped bass, which relies heavily on menhaden for sustenance. Striped bass are said to be at their ‘historic level’ (Hartman and Margraf,

2003). Here we evaluate if their population growth may be impacted by the availability of menhaden as suggested in recent reports (Uphoff, 2003). Whether the menhaden fishery has any adverse impact is unclear, as “[no] studies have shown that the menhaden purse seine fishery has had any significant biological effect on any other species or fishery” (ASMFC, 2004).

#### **4.3.1.1.1 *Are the striped bass back at their historic level?***

Our simulations indicate that the resident striped bass have increased above their ‘historic’ (*i.e.*, 1950) level, while the migratory are back at it. These findings seem ‘semi-robust’ and it is indeed difficult to radically change the finding that striped bass will recover from the low fishing pressure it has experienced in recent decades.

These results are, however, inconclusive. We particularly question what may have happened in the pre-1982 period, where we had to estimate recreational catches based on post-1982 behavior. Again, this calls for a closer evaluation of historical exploitation and trends of striped bass biomass in the Chesapeake Bay.

#### **4.3.1.1.2 *Impact of menhaden fishery on striped bass***

Using the model as fitted to the time series, we let the model run for an additional 25 years, and evaluated three alternative menhaden-harvest scenarios: *status quo*, half the fishing on menhaden, and no fishing on menhaden. We found at the end of the simulation that the striped bass biomass will decrease under *status quo*; that they would be stable under the reduced fishing scenario; and that they would increase with no fishing for menhaden. The striped bass are thus moderately sensitive to changes in menhaden fishing pressure in this model.

The predictions for impact of menhaden fishery on striped bass are sensitive to the assumptions about carrying capacity for both menhaden and striped bass. If we use the default assumption for vulnerability ( $v=2$ ) for menhaden, we obtain a very good fit to menhaden (and others groups’) time series, and we see that the reduction in menhaden fishery is having a fair impact on the menhaden population (the menhaden population roughly doubles if fishing is stopped altogether). On the other hand, if we assume that menhaden were much closer to their carrying capacity in 1950, we are not able to get as clear a decline for menhaden in the early part of the time series as indicated by the assessments, and a stop to the menhaden fishery at present is indicated to have very little impact on the striped bass.

Likewise, the calculations are sensitive to the assumptions about carrying capacity for striped bass. We ended up using low vulnerabilities for the species, indicating that it would have been close to carrying capacity in 1950. We doubt that this is the case, but higher vulnerabilities (*i.e.*, assuming it to be further from carrying capacity) would let the fishing mortality crash the stock much earlier than what likely happened.

The problem we face with regard to evaluating the impact of the menhaden fishery is clearly linked to our lack of knowledge (*i.e.*, lack of assessments) about what happened to striped bass prior to 1982, as well as to the exploitation and population history of menhaden pre-1958.

#### **4.3.1.2 Bay anchovy**

Bay anchovy is considered an important forage species in the Chesapeake Bay, and a large number of predators feed on the group in the present ecosystem model. The fitted model was used to evaluate the impact of bay anchovy on other ecosystem groupings. The simulation period was increased to 100 years, and two simulations were run with all biomasses recorded at the end of the simulation: one *status quo* simulation maintaining the 2002 fishing pressure for an additional 50 years and another simulation in which a very high fishing pressure was entered for bay anchovy.

Comparing the ratios of the end-states in Table 7, a few groups are predicted to benefit from the decline in bay anchovy. These are mainly competitors (juvenile menhaden, and through them the adult menhaden) of the bay anchovy, not its prey. A noteworthy result is that striped bass are predicted to benefit, which is assumed to be linked to improved feeding conditions for menhaden. The groups that decline with the bay anchovy are mainly the predators for which bay anchovy are an important prey. The predictions in Table 7 are in general agreement with the mixed trophic impact analysis of EwE, confirming the finding of Libralato *et al.* (2006), who found a strong correlation between Ecosim simulation results and those of mixed-trophic-impact analysis. However, there are differences. For instance, mixed-trophic-impact analysis did predict the impact on striped bass, juvenile menhaden, and spot, but also showed that other flounders would be impacted by bay anchovy, and we did not see that in the Ecosim simulation.

The perhaps most important finding from this simulation is that one cannot simply assume a direct relation between what happens to a group and what happens to its prey or predators; the food web is more intricate than that.

#### **4.3.2 Invertebrates**

The model may lack sufficient detail and be limited by its design regarding certain lower-level processes. Policy questions that concern alterations in planktonic community structure and resulting ramifications on trophically dependent higher trophic level species, or *vice versa*, cannot be addressed confidently due to a lack of detail and partitioning at lower levels. This lack of detail at planktonic levels prohibits shifts in community composition due to nutrient enrichment and/or differential responses to predation that may be essential to the reproduction of historical changes in the Chesapeake Bay ecosystem. The model can easily be modified to provide more detail to accommodate such questions; the main issue is whether there is sufficient empirical background to do so.

##### **4.3.2.1 Blue crab**

When participants in a Chesapeake Bay Ecosystem Modeling Workshop were asked to formulate policy questions (Table 2), several focused on blue crab. One question dealt with ecosystem manipulation: Can the crab stock be increased by control of its predators? The current model will not be good at answering this question, as no evidence could be found of blue crab being important prey for other groups in the system. The model says that predation pressure on adult blue crab is negligible, and the only important predator on young-of-year blue crab is older blue crab. Hence, no predator control mechanism can be identified for enhancing blue crab abundance short of providing refuge for small crabs to hide from bigger ones. This may well be a shortcoming in examining diet composition, and any additional



information about predator-prey interactions involving blue crab that could be added to the model inputs would be helpful.

The model was used to evaluate whether the crab stock can be restored through fishery reductions. The fitted scenario was run for an additional 47 years, using productivity and exploitation patterns from 2002 for all groups but blue crab, while bringing the blue crab fishing pressure back to the 1950 level. The simulation predicts that this would result in a blue crab biomass of 77% over the 1950 level, while catches would settle at 75% over the 1950 level. Thus, indications are that it is possible to restore blue crab abundance, and that it can be done through effort restriction with limited impact on overall catches. We assume the major reason for the blue crab settling at a level higher than that of 1950 is the increased productivity in the Bay as based on the nutrient loading patterns.

It is worth noting that we see two- to three-year cycles in blue crab abundance within the model runs, somewhat like what is observed in the Bay.

#### **4.3.2.2 Oyster**

The Ecosim simulation shows good agreement with the biomass time-series trend available for oysters. The model indicates that the oyster biomass over the time period has decreased to 4% of its 1950 value.

Noting that the *Chesapeake 2000* agreement (Chesapeake Bay Program, 2000) targets a ten-fold increase in oyster biomass by 2010, a ‘what-if’ scenario was run. We ran a simulation for 40 years into the future and forced the biomass of oyster to ten times the 2002 level (*i.e.*, from  $.75$  to  $7.5 \text{ t} \cdot \text{km}^{-2}$ , illustrating that modeling is much easier than doing it empirically), and compared to a similar simulation also continuing for 40 years with current fishing patterns, but without forcing the biomass of oysters.

The increased oyster biomass (to around half of the 1950 level) is not predicted to have any noticeable impact on phytoplankton in the Bay. We attribute this to spatial factors; it will have impact around oyster beds, but based on the numbers, it will not have any Bay-wide influence, much in line with what other models have shown (R. Fulford, The University of Southern Mississippi, pers. comm.)

For the other groups, the increased oyster biomass is predicted to have only negligible effect (<4%), and the impact is negative for nearly all groups, due largely to there being ‘one more competitor on the block’; predation pressure on adult oysters is nonexistent in the model, hence, they do not contribute to funneling energy up the food web. Also, we do not include in the model any mediation effects related to the protection oyster reefs may offer as refuge for juvenile fishes and invertebrates. This could be done straightforwardly, given information to that effect.

#### **4.3.3 Ecosystem drivers: climate variation and fishing pressure**

The primary production rate for the Chesapeake Bay was estimated based on the hydrographic/climatic model. The resulting monthly time series is shown in Figure 9. The summed squared residuals between simulated values and ‘observed’ time-series estimates decreased substantially when primary production rate was forced.

Estimating “the relative importance of climate variation on fish populations versus that of harvesting pressure” (see Table 2) is a more complicated matter – or rather, it is something that cannot be estimated. There are systems where it is possible to evaluate population trends based on fishing pressure

alone (Christensen, 1998), but it is almost always necessary to consider fisheries as well as environmental factors to explain what has happened in an ecosystem over time (Christensen and Walters, 2005; Walters *et al.*, 2005). This is clearly indicated to be the case for the Chesapeake Bay, based on the impact of nutrient loading on the residuals as described above.

The simulations here raise a question about the effect of decreasing nutrient loading to the Bay. This model will likely predict that higher-trophic level production will decrease even more than the amount to which phytoplankton is reduced, based on experience from other models, and to some degree supported by empirical studies (Nielsen and Richardson, 1996). It is important to keep in mind that primary production in estuaries may not be linearly related to the concentration of one nutrient, but that the combined effect of several factors (*e.g.*, nutrients, light, salinity, and temperature) appears to be paramount.

## 5 Conclusions

### 5.1 *Data availability*

This model attempts to reconstruct the recent history of exploitation and trophic interactions in the Chesapeake Bay. The simulations rely very heavily on data availability to reproduce the past. It is a major hindrance to the Fisheries Ecosystem Model that little systematic effort has been allocated in the Bay region to collecting and making available fisheries data from before 1982.

From a data perspective, the biggest problem is the lack of pre-1982 recreational catch data. There is no doubt that older data is more difficult to obtain; the NOAA Recreational Catch database only starts in 1982. Nevertheless, finding this data is a priority. It may call for more assumptions than were necessary for the post-1982 data, but its inclusion is of utter importance in order to develop an understanding of how life in the Bay has developed and reacted to exploitation.

The ecosystem model is strongly influenced by nutrient loading trends. Excellent hydrographic/climatic modeling is available for more recent years, but do not cover the full time period required to understand what has happened in the Bay and why. A simple two-layer hydrographic model has been applied to the model, forced by river gauge data, nutrient loading, wind, and rainfall, and based on detailed bathymetry. The model runs are validated based on observed data. The hydrographic model is simple, but it provides the time series of estimates required for the Ecosim simulations. It would be best if in the future, more detailed habitat models were applied to the full time period, so that the trophic model can be linked to those as well.

The striped bass/menhaden simulation discussed above illustrates how assumptions about trophic interactions can be important for evaluating impact of fisheries. This warrants a closer look when setting target and limit reference points as part of the stock assessment process. It also calls for digging into the archives to extract historical diet information (as done by Griffin and Margraf, 2003), for continued sampling of diet information, and for the creation of community-accessible databases with available diet information.

## 5.2 *Stock assessment*

Current fisheries management practices are tactical. They deal with how to best use available resources in the short term. The data analysis to determine use of Bay fisheries tends to be quite limited in scope, often concentrating on recent catch data and single-species assessments. In the Chesapeake Bay, hardly any assessment goes back to before 1982. The lack of recreational catch data makes this a difficult task, but any effort toward increasing the body of historical data available for inclusion in the model would be highly valuable. If possible, it would be best to input data for all assessments back to 1950.

When taking an ecosystem perspective, more species than just those that are exploited matter. In order to understand quantitative trophic interactions in the Bay, we recommend that assessments include all species of ecological importance in the Bay. When evaluating a time-series trend for a given species or group, the two most important data points are a biomass from the early part of the modeled time period and a biomass from the late part of the time period. Of course, it is also valuable to get data points in between or for all years. Are there old surveys that may be of use, even for non-target species? Or egg surveys?

Perhaps there should be two standards for stock assessments: one for use in fisheries management, where decisions with short-term economical consequences must be made; and another for use in long-term management of the ecosystem, where emphasis is on increasing or understanding of ecological processes and strategically evaluating management scenarios.

## 5.3 *Spatial modeling*

EwE models, as well as their implementation, are very open-ended; their expansion is straightforward. The number of functional groups is not limited in any way—groups can be removed, aggregated, or added as new information is received. Fishing fleets can be added or removed, and new fleets can be included (or invented) to test suggested management interventions. Time-series data can be easily updated in the time steps required by the modeler/user. This requires only that the user create a new spreadsheet with comma-separated values (\*.csv file) that contains the new data and load it into the EwE database.

EwE can also model trophic and fishing-effort dynamics in an explicit spatial setting. The spatial component of EwE, Ecospace, is essentially a grid or two-dimensional matrix of ‘cells,’ each cell incorporating an Ecosim model (initially identical, as Ecosim inherits its parameters from Ecopath) and expressed at the user-interface level as a map (Walters *et al.*, 1999). Each cell in the map, excluding land cells, is linked through two processes: dispersal of organisms and the redistribution of fishing effort due to changing profit patterns and/or the creation of areas closed to fishing. The user/modeler defines the base map (land/water areas) by sketching it on the interface with the computer mouse. Over the top of the base map the following attributes can be sketched: 1) patterns of relative fishing cost (effort ‘avoids’ high-cost cells; for example, cells far from their home port that require high fuel costs to reach); 2) patterns of relative primary production; 3) patterns of habitats to which biomass pools and fishing fleets can be assigned; and 4) areas closed to fishing (fleet and season specific). This allows for the exploration of policies that include spatial components, including the evaluation of the size and placement of marine protected areas (MPAs). A new sub-module of Ecospace has been developed to evaluate ecological and economic aspects of the placement of MPAs (Beattie *et al.*, 2002).

The geographical scale of a model needs to be defined appropriately to consider the questions that are to be addressed to it. An example of this could be a study of how coastal eutrophication, or nutrient enrichment, influences marine ecosystems. It is known that enrichment (and sedimentation) generally decreases the transparency of coastal water bodies, but the broader indirect effects of such changes in light regimes on continental shelf ecosystems are not well explored. Shelf systems may be particularly susceptible to pollution-related decreases in water transparency, as they may cause shifts in energy flows from benthos-based to plankton-based food webs. Nutrient loading can be incorporated as a forcing function in Ecosim that directly influences phytoplankton production (as done here) and indirectly (causing shading effects on benthic producers and the food webs they support). This model is very capable of modeling ecosystem effects through food webs.

Much effort has been devoted to showing the effects of climate changes on fish populations. Most of this work presupposes that there is some connection between climate and recruitment and correlates some climate index with an aspect of the life history of a particular stock of fish. For example, see Hollowed *et al.* (2000) for production and recruitment of various fish stocks in the North Pacific. The linkages between climate change and fish populations have also been studied in the Atlantic using the North Atlantic Oscillation as a climate index (Attrill and Power, 2002). Ecosim can be used to examine how primary production changes might be driven by climate and evaluate how 'bottom-up' cascade of production might differentially affect mortality and stock size of commercially important species in the Chesapeake Bay.

Chlorophyll-a time series from CBREEM can be incorporated as a forcing function in Ecosim to influence phytoplankton production directly and benthic primary producers indirectly (by simulating interference in Ecosim's 'mediation' tools). Simulated shifts in the system's food webs supported by each of these primary production realms can be scrutinized and compared with related empirical and theoretical information.

#### **5.4 *Ecosystem boundaries and model structure***

Few ecosystems are clearly demarcated; the Chesapeake Bay certainly is not. Many species spend a good part of their life cycle or year outside the Bay, and it is an open question how best to model the population dynamics of such species. We believe there is no way but to include the full extent of such populations in the model. For example, for menhaden, this may mean incorporating the entire Atlantic population and its production, consumption, and catches. Lesser nuisances, such as ensuring that consumption taken up outside the Bay does not lead to exaggerated estimates for the consumption in the Bay, can be handled by lowering the consumption ratio.

It may be necessary to make several versions of the fitted model developed here. Different policy questions may call for different model structures. This is simply a question of focusing the modeling appropriately. The biggest problem is, rather, that it is necessary to have good data on which to build our models and simulations. Models cannot be designed to answer all questions – at least not without having a very good understanding of how an ecosystem and its components function and what has happened to the resources over time. A model cannot predict what will happen to parts of a system for which we have no information; the model is a formulation built on our knowledge.

Based on solid information and with an appropriate structure, a model can be used to evaluate a range of scenarios and guide us to what are likely consequences and causes. The ecosystem model presented in this report has this potential.

This report has sought to clarify what Chesapeake Bay ecosystem resource information has been included in the Fisheries Ecosystem Model to date. Hopefully, by assembling this material in one place, data gaps have been made more evident. Any feedback that will enable the next generation of this modeling effort to go beyond what has been presented here is welcomed and encouraged.

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## 7 Tables

### 7.1 *Relating species and management plan entities*

Table 1. Chesapeake Bay fish and shellfish species with both common and scientific names. From Bonzek (2004) with updates.

Species Common Name	Species Latin Name(s)	Management Plan Entity*
American eel	<i>Anguilla rostrata</i>	ASMFC, CBP
Atlantic croaker	<i>Micropogonias undulatus</i>	ASMFC, CBP
Atlantic menhaden	<i>Brevoortia tyrannus</i>	ASMFC
Sturgeon, Atlantic and shortnose	<i>Acipenser oxyrhynchus</i> , <i>A. brevirostrum</i>	ASMFC
Bay anchovy	<i>Anchoa mitchilli</i>	No management plan
Black drum	<i>Pogonius cromis</i>	CBP
Black seabass	<i>Centropristis striata</i>	ASMFC, CBP, MAFMC
Blue crab	<i>Callinectes sapidus</i>	CBP; MDNR (Coastal Bays)
Bluefish	<i>Pomatomus saltatrix</i>	ASMFC, MAFMC, CBP
Butterfish	<i>Peprilus triacanthus</i>	MAFMC
Catfish (several closely related species)	<i>Ictaluridae</i>	No management plan
Dogfish and coastal sharks	<i>Elasmobranchii</i>	ASMFC, MAFMC
Eastern oyster	<i>Crassostrea virginica</i>	CBP
Killifishes (several)	<i>Fundulus</i>	No management plan
Gizzard shad	<i>Dorosoma cepedianum</i>	No management plan
Hard clam	<i>Mercenaria mercenaria</i>	MDNR (Coastal Bays)
Mackerel, king and Spanish (both)	<i>Scomberomorus cavallas</i> , <i>S. maculatus</i>	ASMFC (Spanish), CBP (Spanish and king); MAFMC (Spanish)
Red drum	<i>Sciaenops ocellatus</i>	ASMFC, CBP
River herring (alewife and blueback herring)	<i>Alosa pseudoharengus</i> , <i>A. aestivalis</i>	ASMFC, CBP
Scup	<i>Stenotomus chrysops</i>	ASMFC, MAFMC
Spot	<i>Leiostomus xanthurus</i>	ASMFC, CBP
Shad (American and hickory)	<i>Alosa sapidissima</i> , <i>A. mediocris</i>	ASMFC (American), CBP (both)
Spotted seatrout	<i>Cynoscion nebulosus</i>	ASMFC
Striped bass	<i>Morone saxatilis</i>	ASMFC, CBP
Summer flounder	<i>Paralichthys dentatus</i>	ASMFC, CBP, MAFMC
Tautog	<i>Tautoga onitis</i>	ASMFC, CBP
Threadfin shad	<i>Dorosoma petenense</i>	No management plan
Weakfish	<i>Cynoscion regalis</i>	ASMFC, CBP
White perch	<i>Morone Americana</i>	No management plan
Yellow perch	<i>Perca flavescens</i>	MDNR (for Maryland section)
<i>Other species of possible interest:</i>		
Cobia	<i>Rachycentron canadum</i>	SAFMC; however, states in the region are not on this council
Horseshoe crab	<i>Limulus poytphemus</i>	ASMFC, CBP
Squid (long-finned & short-finned)	<i>Illex illecebrosus</i> <i>Loligo pealei</i>	MAFMC

\* ASMFC = Atlantic States Marine Fisheries Commission; CBP = Chesapeake Bay Program; MAFMC = Mid Atlantic Fishery Management Council; and MDNR = Maryland Department of Natural Resources; SAFMC = South Atlantic Fishery Management Council

## 7.2 *Questions*

Table 2. Topics that the Chesapeake Bay Fisheries Ecosystem Model (or a submodel derived from it) has been or can be used to address.

At the October 2001 Chesapeake Bay EwE workshop, the following subsample of questions and issues to be addressed by the Chesapeake Bay EwE model were formulated:

1. Can water quality (e.g., dissolved oxygen) be managed by top-down actions such as fishery regulations?
2. What is the role of forage fish in Chesapeake Bay ecosystem dynamics?
3. Are there too many striped bass in the Chesapeake Bay?
4. What is the relative importance of the effects of climate variation on fish populations versus that of harvesting pressure?
5. Can the crab stock be restored through fishery reductions and the use of protected areas?
6. Can the crab stock be increased by the 'control' of other mortality agents, particularly predators?
7. What are the consequences of a tenfold increase in the oyster population in the Chesapeake Bay?
8. Can protected areas for oysters enhance abundance and aid in their restoration?

### 7.3 Basic parameters

Table 3. Basic parameters for the Chesapeake Bay Fisheries Ecosystem Model. Values estimated by Ecopath are shown in *italics*. Estimated from a variety of sources as described in the text.

EwE Group #	Group name	Trophic level	Biomass (t · km <sup>-2</sup> )	Prod. / biomass (year <sup>-1</sup> )	Cons. / biomass (year <sup>-1</sup> )	Ecotrophic efficiency	Prod. / cons.
1	Striped bass YOY	3.56	0.0125	1.800	23.266	<i>0.401</i>	<i>0.077</i>
2	Striped bass resident	3.52	2.100	0.400	4.441	<i>0.554</i>	<i>0.090</i>
3	Striped bass migratory	3.36	2.946	0.300	2.300	<i>0.483</i>	<i>0.130</i>
4	Bluefish YOY	4.17	0.0161	5.650	18.111	<i>0.014</i>	<i>0.312</i>
5	Bluefish adult	4.05	0.240	0.589	3.300	<i>0.630</i>	<i>0.178</i>
6	Weakfish YOY	4.26	0.0257	4.000	13.525	<i>0.304</i>	<i>0.296</i>
7	Weakfish adult	4.15	0.489	0.685	3.100	<i>0.906</i>	<i>0.221</i>
8	Atlantic croaker	3.25	1.670	0.916	5.400	<i>0.801</i>	<i>0.170</i>
9	Black drum	3.03	<i>1.263</i>	0.190	2.100	0.100	<i>0.090</i>
10	Summer flounder	3.66	<i>0.454</i>	0.520	2.900	0.950	<i>0.179</i>
11	Menhaden YOY	2.99	18.089	1.500	15.860	<i>0.686</i>	<i>0.095</i>
12	Menhaden adult	2.13	33.000	0.800	7.800	<i>0.941</i>	<i>0.103</i>
13	Alewife and herring	3.13	5.986	0.750	9.400	0.950	<i>0.080</i>
14	American eel	3.38	<i>3.220</i>	0.250	<i>2.500</i>	0.500	0.100
15	Catfish	3.09	<i>1.155</i>	0.280	2.500	0.950	<i>0.112</i>
16	White perch YOY	3.55	0.00305	2.000	19.921	<i>0.576</i>	<i>0.100</i>
17	White perch adult	3.55	0.300	0.500	4.200	<i>0.886</i>	<i>0.119</i>
18	Spot	2.86	<i>1.674</i>	1.000	5.800	0.900	<i>0.172</i>
19	American shad	3.04	0.400	0.700	3.500	<i>0.725</i>	<i>0.200</i>
20	Bay anchovy	3.41	3.400	3.000	10.900	<i>0.494</i>	<i>0.275</i>
21	Other flatfish	2.99	<i>0.169</i>	0.460	4.900	0.950	<i>0.094</i>
22	Gizzard shad	2.43	<i>2.086</i>	0.530	14.500	0.950	<i>0.037</i>
23	Reef-associated fish	3.40	<i>0.232</i>	0.510	3.100	0.900	<i>0.165</i>
24	Non-reef-associated fish	3.05	<i>1.228</i>	1.000	<i>5.000</i>	0.900	0.200
25	Littoral forage fish	2.85	<i>5.210</i>	0.800	<i>4.000</i>	0.950	0.200
26	Sandbar shark	4.05	0.0240	0.230	1.400	<i>0.217</i>	<i>0.164</i>

27	Other elasmobranchs	3.33	0.500	0.150	0.938	0.112	0.160
28	Piscivorous birds	3.98	0.300	0.163	120.000	0.000	0.001
29	Non-piscivorous seabirds	2.73	0.121	0.511	120.000	0.000	0.004
30	Blue crab YOY	2.80	1.580	5.000	12.057	0.879	0.415
31	Blue crab adult	3.09	4.000	1.000	4.000	0.881	0.250
32	Oyster YOY	2.00	3.280	6.000	8.965	0.096	0.669
33	Oyster 1+	2.09	20.400	0.150	2.000	0.414	0.075
34	Soft clam	2.09	6.923	0.450	2.250	0.950	0.200
35	Hard clam	2.00	2.626	1.020	5.100	0.950	0.200
36	Ctenophores	3.48	3.400	8.800	35.200	0.205	0.250
37	Sea nettles	4.13	0.583	5.000	20.000	0.000	0.250
38	Microzooplankton	2.00	6.239	140.000	350.000	0.950	0.400
39	Mesozooplankton	2.72	10.300	25.000	83.333	0.956	0.300
40	Other suspension feeders	2.00	6.000	2.000	8.000	0.823	0.250
41	Other in/epi fauna	2.10	66.675	1.000	5.000	0.900	0.200
42	Benthic algae	1.00	1.717	80.000	-	0.900	-
43	SAV	1.00	419.000	5.110	-	0.084	-
44	Phytoplankton	1.00	27.000	160.000	-	0.684	-
45	Detritus	1.00	1.000	-	-	0.031	-

## 7.4 Catches, species groups 1-12

Table 4. Estimated catches of groups 1-12 ( $t \cdot km^2 \cdot year^{-1}$ ) for the Chesapeake Bay since 1950 used in the Chesapeake Bay Fisheries Ecosystem Model. Estimated from a variety of sources as described in the text.

Group	SB resident	SB resident	SB Migratory	SB migratory	Blue-fish	Blue-fish	Weak-fish	Atl. croaker	Atl. croaker	Black drum	Summer flounder	Men-haden Age 0-1	Men-haden Age 2+	Menhaden ASMFC	Men-haden CB
Group no.	2	2	3	3	5	5	7	8	8	9	10	11	12	12	12
1950	0.463	0.792	0.427	0.825	0.116	0.067	0.286	0.656	0.542	0.005	0.209		12.706	12.71	7.753
1951	0.329	0.549	0.303	0.572	0.073	0.042	0.137	0.434	0.358	0.006	0.212		13.954	13.95	5.78
1952	0.271	0.442	0.25	0.46	0.071	0.041	0.111	0.321	0.265	0.004	0.194		16.758	16.76	4.19
1953	0.247	0.393	0.228	0.409	0.062	0.035	0.142	0.323	0.267	0.005	0.273		22.844	22.84	7.359
1954	0.243	0.377	0.224	0.392	0.076	0.044	0.148	0.431	0.356	0.034	0.304		24.25	24.25	13.101
1955	0.275	0.416	0.254	0.433	0.079	0.045	0.264	0.818	0.676	0.01	0.255	3.836	21.82	24.71	14.305
1956	0.25	0.368	0.23	0.383	0.09	0.052	0.232	0.815	0.673	0.014	0.292	8.514	19.97	28.46	8.638
1957	0.221	0.317	0.204	0.33	0.079	0.046	0.147	1.114	0.92	0.011	0.26	6.951	17.161	23.75	12.15
1958	0.351	0.489	0.324	0.509	0.052	0.03	0.11	0.894	0.738	0.005	0.332	3.086	17.314	20.33	14.641
1959	0.512	0.693	0.472	0.721	0.059	0.034	0.049	0.607	0.501	0.02	0.436	14.648	11.716	26.35	18.802
1960	0.531	0.697	0.49	0.726	0.039	0.022	0.067	0.323	0.266	0.014	0.357	0.966	20.226	21.13	11.29
1961	0.576	0.734	0.532	0.764	0.087	0.05	0.091	0.224	0.185	0.02	0.26	3.14	19.896	23.04	13.548
1962	0.47	0.58	0.434	0.604	0.163	0.094	0.105	0.093	0.077	0.029	0.25	2.534	18.974	21.4	14.874
1963	0.516	0.616	0.476	0.641	0.187	0.108	0.074	0.009	0.007	0.028	0.216	3.435	10.441	13.7	11.749
1964	0.412	0.476	0.38	0.495	0.112	0.064	0.11	0.028	0.023	0.006	0.195	3.529	7.239	10.28	15.26
1965	0.41	0.457	0.378	0.476	0.059	0.034	0.14	0.109	0.09	0.009	0.258	4.488	6.448	10.42	16.327

1966	0.488	0.525	0.451	0.546	0.072	0.041	0.074	0.105	0.086	0.038	0.282	3.964	4.82	7.89	12.605
1967	0.462	0.479	0.427	0.499	0.038	0.022	0.043	0.023	0.019	0.017	0.222	3.701	4.039	7.72	10.127
1968	0.488	0.486	0.45	0.506	0.106	0.061	0.079	0	0	0.028	0.239	2.579	6.813	8.98	12.432
1969	0.616	0.589	0.568	0.613	0.077	0.044	0.065	0.005	0.004	0.009	0.163	2.53	3.934	6.04	8.24
1970	0.457	0.419	0.422	0.436	0.199	0.114	0.153	0.009	0.008	0.006	0.239	4.61	5.766	10.33	20.403
1971	0.315	0.276	0.29	0.287	0.209	0.12	0.17	0.019	0.016	0.008	0.191	1.77	8.242	9.85	18.149
1972	0.467	0.391	0.431	0.407	0.354	0.203	0.182	0.035	0.029	0.002	0.203	4.748	9.888	14.59	25.243
1973	0.624	0.498	0.576	0.518	0.884	0.508	0.35	0.184	0.082	0.001	0.354	2.544	11.332	13.79	22.935
1974	0.481	0.365	0.444	0.38	1.027	0.59	0.216	0.223	0.096	0.003	0.363	2.552	9.136	11.32	17.453
1975	0.336	0.241	0.31	0.251	0.99	0.569	0.309	0.544	0.316	0.003	0.411	2.169	7.839	9.77	14.601
1976	0.226	0.153	0.209	0.16	1.301	0.747	0.274	0.734	0.411	0.002	0.38	4.007	9.613	13.42	20.237
1977	0.21	0.134	0.194	0.14	1.027	0.59	0.283	0.894	0.548	0.001	0.502	2.302	11.342	13.34	23.118
1978	0.136	0.081	0.125	0.084	0.852	0.489	0.274	0.826	0.513	0.003	0.629	1.972	11.792	13.33	19.411
1979	0.112	0.063	0.104	0.065	0.941	0.54	0.428	0.333	0.132	0.003	1.115	2.957	12.071	13.74	20.909
1980	0.207	0.107	0.191	0.112	0.904	0.52	0.423	0.144	0.042	0	0.934	3.784	12.276	15.99	24.764
1981	0.162	0.065	0.149	0.068	0.623	0.26	0.175	0.033	0.022	0.007	0.275	2.45	12.802	14.35	18.661
1982	0.031	0.02	0.029	0.02	0.571	0.318	0.155	0.007	0.007	0.002	0.287	2.565	12.731	15.15	27.56
1983	0.034	0.021	0.032	0.022	0.824	0.379	0.226	0.023	0.008	0.063	0.684	2.348	14.396	15.74	29.544
1984	0.057	0.049	0.053	0.051	0.466	0.226	0.15	0.061	0.051	0.004	0.616	3.873	9.179	11.92	22.281
1985	0.016	0.009	0.015	0.009	0.776	0.529	0.144	0.153	0.101	0.017	0.309	3.415	8.853	11.74	29.252
1986	0.007	0.001	0.007	0.001	0.796	0.344	0.156	0.233	0.163	0.037	0.232	0.625	8.895	9.43	20.463

1987	0.011	0.003	0.011	0.003	0.805	0.311	0.147	0.21	0.164	0.026	0.388	1.515	11.565	13.03	28.183
1988	0.023	0.006	0.021	0.006	0.649	0.334	0.137	0.213	0.121	0.009	0.511	1.102	11.27	12.04	25.265
1989	0.028	0.01	0.026	0.01	0.348	0.306	0.093	0.109	0.061	0.012	0.21	3.779	9.101	12.67	28.047
1990	0.033	0.014	0.031	0.014	0.404	0.278	0.097	0.051	0.012	0.005	0.131	1.051	14.997	15.56	32.078
1991	0.033	0.022	0.03	0.022	0.471	0.222	0.078	0.093	0.013	0.011	0.256	5.342	9.914	13.78	27.43
1992	0.061	0.038	0.056	0.04	0.235	0.072	0.05	0.153	0.065	0.017	0.317	3.413	8.491	11.41	26.178
1993	0.085	0.052	0.079	0.054	0.24	0.052	0.067	0.42	0.329	0.004	0.216	2.007	10.817	12.67	29.14
1994	0.112	0.062	0.104	0.064	0.208	0.07	0.084	0.518	0.394	0.007	0.206	0.931	9.469	10.33	23.441
1995	0.188	0.104	0.174	0.108	0.19	0.049	0.089	0.575	0.44	0.011	0.212	2.212	11.384	13.54	31.954
1996	0.255	0.129	0.235	0.135	0.148	0.054	0.109	0.725	0.556	0.008	0.172	0.953	10.763	11.69	26.503
1997	0.31	0.164	0.286	0.17	0.142	0.065	0.12	1.09	0.832	0.009	0.191	1.095	9.269	10.33	22.773
1998	0.315	0.18	0.29	0.187	0.175	0.068	0.142	1.08	0.797	0.008	0.22	1.042	8.794	9.68	23.278
1999	0.295	0.152	0.273	0.158	0.114	0.047	0.125	1.068	0.834	0.003	0.151	1.433	5.415	6.59	17.413
2000	0.436	0.192	0.403	0.2	0.142	0.046	0.126	1.174	0.89	0.004	0.183	0.631	6.057	6.61	16.874
2001	0.403	0.166	0.372	0.173	0.181	0.076	0.092	1.277	0.954	0.003	0.262	0.348	9	9.29	22.306
2002	0.467	0.148	0.431	0.154	0.147	0.049	0.083	1.166	0.858	0.003	0.227	1.396	5.564	6.71	16.774

## 7.5 Catches, species groups 13-35

Table 5. Estimated catches for species groups 13-35 ( $t \cdot km^2 \cdot year^{-1}$ ) for the Chesapeake Bay since 1950 used in the Chesapeake Bay Fisheries Ecosystem Model. Estimated from a variety of sources as described in the text.

Group	Alewife	Eel	Catfish	White perch	Spot	Spot	Shad	Gizzard shad	Blue crab	Oyster	Oyster	Soft clam	Hard clam
Group no.	13	14	15	17	18	18	19	22	31	33	33	34	35
1950	1.577	0.065	0.091	0.073		0.456	0.203	128.1	4.329	1.36		0.073	0.071
1951	1.789	0.056	0.079	0.064		0.508	0.22	140.4	4.005	1.34		0.011	0.067
1952	1.513	0.055	0.091	0.058		0.619	0.256	165.2	3.796	1.56		0.126	0.059
1953	1.299	0.045	0.11	0.053		0.407	0.204	225.3	3.732	1.68		0.011	0.045
1954	1.448	0.036	0.135	0.056		0.452	0.212	240.2	3.332	1.89		0.059	0.037
1955	1.225	0.04	0.156	0.056		0.416	0.225	1389.8	2.968	1.78		0.128	0.043
1956	1.234	0.041	0.154	0.06		0.333	0.24	1021	3.176	1.68		0.126	0.042
1957	1.006	0.035	0.148	0.051		0.382	0.239	579.6	3.529	1.58		0.178	0.05
1958	1.032	0.037	0.14	0.056		0.547	0.188	422.7	3.222	1.7		0.205	0.045
1959	0.995	0.037	0.168	0.073		0.356	0.148	150.1	2.942	1.51		0.253	0.088
1960	0.862	0.018	0.161	0.056		0.405	0.122	113.6	3.847	1.23		0.213	0.083
1961	0.816	0.017	0.132	0.068		0.109	0.143	51	3.937	1.25		0.307	0.105
1962	1.218	0.015	0.124	0.095		0.215	0.172	43.8	4.337	0.9		0.311	0.094
1963	1.251	0.026	0.092	0.067		0.134	0.142	11.9	3.52	0.83		0.37	0.117
1964	1.269	0.023	0.084	0.035		0.288	0.161	39.1	3.952	1		0.357	0.126
1965	1.746	0.043	0.06	0.068		0.155	0.195	65.5	4.266	0.96		0.336	0.124
1966	1.363	0.031	0.069	0.091		0.102	0.162	20.7	4.772	0.96		0.238	0.092
1967	1.431	0.044	0.06	0.064		0.392	0.136	13.1	4.278	1.17		0.253	0.098
1968	1.648	0.044	0.064	0.085		0.1	0.159	16.1	3.155	1.03		0.359	0.109
1969	1.54	0.049	0.072	0.104		0.092	0.161	8.2	3.471	1.01		0.282	0.122
1970	0.958	0.068	0.061	0.075		0.547	0.234	6.5	3.705	1.12		0.272	0.086
1971	0.705	0.066	0.081	0.077		0.044	0.112	30.1	3.945	1.16		0.088	0.098
1972	0.582	0.033	0.089	0.054		0.252	0.137	26.4	3.791	1.09		0.03	0.069
1973	0.516	0.02	0.069	0.039		0.215	0.138	56.1	3.154	1.15		0.095	0.065
1974	0.686	0.073	0.08	0.026		0.188	0.081	71.8	3.435	1.13		0.057	0.068
1975	0.552	0.068	0.081	0.027		0.164	0.06	69.5	3.2	1.03	0.744	0.079	0.053
1976	0.2	0.033	0.066	0.023		0.097	0.046	36.8	2.696	0.95	0.675	0.075	0.042



1977	0.068	0.025	0.089	0.032		0.151	0.07	7.4	3.127	0.82	0.591	0.157	0.048
1978	0.106	0.062	0.068	0.048		0.256	0.06	2.7	2.839	1.02	0.652	0.131	0.024
1979	0.083	0.067	0.069	0.033		0.2	0.046	4.8	3.313	0.98	0.612	0.087	0.029
1980	0.062	0.034	0.103	0.043		0.14	0.045	15.9	3.182	1.03	0.678	0.071	0.036
1981	0.028	0.067	0.08	0.042	0.118	0.091	0.023	17.6	4.242	0.98	0.713	0.072	0.053
1982	0.064	0.036	0.08	0.045	0.079	0.064	0.027	5.8	3.619	0.79	0.557	0.089	0.032
1983	0.091	0.036	0.088	0.037	0.15	0.131	0.028	23.1	3.941	0.53	0.338	0.043	0.053
1984	0.063	0.041	0.079	0.052	0.049	0.046	0.061	240.5	3.94	0.56	0.365	0.06	0.034
1985	0.028	0.04	0.088	0.041	0.136	0.13	0.037	193.7	3.968	0.6	0.39	0.048	0.033
1986	0.052	0.038	0.115	0.049	0.187	0.131	0.032	301.1	3.402	0.62	0.361	0.159	0.042
1987	0.104	0.039	0.1	0.051	0.299	0.266	0.037	242.5	3.086	0.39	0.174	0.198	0.046
1988	0.062	0.036	0.097	0.064	0.126	0.127	0.041	341.7	3.169	0.24	0.109	0.183	0.059
1989	0.038	0.039	0.129	0.044	0.193	0.172	0.05	333.4	3.431	0.2	0.111	0.097	0.069
1990	0.036	0.032	0.119	0.065	0.2	0.123	0.039	263.3	3.909	0.2	0.129	0.077	0.071
1991	0.044	0.04	0.101	0.066	0.274	0.204	0.034	293.5	3.726	0.15	0.106	0.016	0.048
1992	0.085	0.04	0.099	0.061	0.258	0.205	0.034	347.7	2.885	0.1	0.057	0.036	0.05
1993	0.08	0.045	0.107	0.09	0.229	0.234	0.03	709.6	4.012	0.03	0.025	0.02	0.072
1994	0.058	0.043	0.14	0.089	0.274	0.3	0.018	948.9	3.62	0.05	0.038	0.015	0.053
1995	0.019	0.032	0.105	0.08	0.208	0.233	0.007	976.7	3.367	0.07	0.06	0.014	0.043
1996	0.006	0.028	0.155	0.104	0.174	0.2	0.011	1314.3	3.211	0.05	0.04	0.011	0.036
1997	0.016	0.028	0.135	0.15	0.211	0.23	0.024	694.1	3.555	0.08	0.065	0.01	0.031
1998	0.009	0.03	0.182	0.097	0.261	0.29	0.021	1009.3	2.609	0.12	0.083	0.007	0.025
1999	0.009	0.029	0.169	0.09	0.165	0.187	0.013	1068.7	2.843	0.13	0.123	0.008	0.03
2000	0.009	0.025	0.138	0.124	0.215	0.231	0.009	166.9	2.24	0.11	0.108	0.003	0.023
2001	0.013	0.025	0.162			0.222	0.014	471.7	2.154	0.07		0.01	0.028
2002	0.015	0.018	0.147			0.199	0.008	615.7	2.264	0.03		0.002	0.031

## 7.6 Vulnerabilities for species groups

Table 6. Vulnerability settings for species groups in the Chesapeake Bay Fisheries Ecosystem Model. Only groups for which the vulnerabilities were changed from the default value of 2 are displayed.

EwE Group #	Group	Vulnerability
2	Striped bass resident	1.20
3	Striped bass migratory	1.01
4	Bluefish YOY	10.00
5	Bluefish adult	15.0
6	Weakfish YOY	1.02
7	Weakfish adult	1.10
8	Atlantic croaker	1.00
9	Black drum	1.50
10	Summer flounder	1.00
13	Alewife and herring	1.00
14	American eel	1.10
17	White perch adult	1.10
19	American shad	1.20
20	Bay anchovy	1.66
22	Gizzard shad	5.00
30	Blue crab YOY	1.20
33	Oyster 1+	1.50
34	Soft clam	12.0
35	Hard clam	7.00
36	Ctenophores	1.16
38	Microzooplankton	2
39	Mesozooplankton	1.08
41	Other in/epi fauna	91.7

## 7.7 Bay anchovy simulation – change in biomass

Table 7. Estimated change in biomass of ecosystem groups resulting from applying a strong fishing pressure on bay anchovy. Groups for which the predicted change was less than  $\pm 2\%$  are omitted.

Group	Change (%)
Menhaden adult	16%
Menhaden 0-1	10%
Spot	3%
Striped bass migratory	3%
Striped bass resident	2%
Mesozooplankton	-2%
Sea nettles	-2%
Striped bass YOY	-3%
Atl. Croaker	-4%

Sandbar shark	-5%
Piscivorous birds	-7%
Summer flounder	-7%
Other elasmobranchs	-8%
Reef assoc. fish	-8%
White perch adult	-11%
Bluefish adult	-14%
Weakfish Adult	-27%
Bluefish YOY	-29%
Weakfish YOY	-38%
Bay anchovy	-46%

### 7.8 *Oyster fishing moratorium simulation*

Table 8. Predicted effect on group biomasses if fishing for oyster had been stopped 1950. The biomass ratios are expressed as current biomass (assuming no oyster fishing)/current biomass (with historic oyster fishing). Only groups for which the absolute difference exceeds 5% are included in the table.

<b>Group</b>	<b>Biomass ratio</b>
Oyster 1+	4.28
Oyster YOY	3.14
Weakfish adult	0.95
Summer flounder	0.94
Phytoplankton	0.94
Other elasmobranchs	0.94
Striped bass YOY	0.93
Non-piscivorous seabirds	0.93
Menhaden adult	0.92
Piscivorous birds	0.92
Menhaden YOY	0.91
Other suspension feeders	0.90
Striped bass migratory	0.89
Striped bass resident	0.88
Black drum	0.74
Ctenophores	0.73
Sea nettles	0.69
Catfish	0.67
Bluefish adult	0.67
Soft clam	0.63
Bluefish YOY	0.60
Hard clam	0.41

## 8 List of figures

### 8.1 *The Chesapeake Bay*

Figure 1. Bathymetry map of the Chesapeake Bay. The salinity provinces are defined as oligohaline ( $S < 10$  psu), mesohaline ( $10 \text{ psu} < S < 20 \text{ psu}$ ), and polyhaline ( $S > 20 \text{ psu}$ ) according to Harding and Perry (1997). The bathymetry data are from NOAA/NOS Data Explorer Service and the bathymetric elevation is referenced to the local tidal datum (Mean Lowest Low Water) averaged over a 19 year tidal epoch.

### 8.2 *Model development*

Figure 2. Ecopath model development process.

### 8.3 *Foraging arena*

Figure 3. Flow between available and unavailable biomass in Ecosim. The assumption of fast equilibrium between the two prey states implies that  $V_i = vB_i / (2v + a_{ij}B_j)$ .

### 8.4 *Food web components*

Figure 4. Overview of the groups in the Chesapeake Bay Fisheries Ecosystem Model. Groups are placed according to their trophic level; the size of the boxes is a function of the group biomasses.

### 8.5 *Ecoranger*

Figure 5. Estimated mean biomasses from 200 Ecoranger runs compared to the original Ecopath biomasses. Note tendency to estimate higher available production for lower trophic level and lower production for higher. The slope of the regression line is -0.06. Accepted Ecoranger runs tend to produce a very low biomass for black drum compared with original Ecopath biomass.

### 8.6 *Mixed trophic impacts*

Figure 6. The mixed-trophic-impact analysis for the Chesapeake Bay Fisheries Ecosystem Model shows direct and indirect impact through the food web. Impacting groups are shown in rows, impacted in columns. Positive impacts are shown above the baselines, negative below. Impacts are relative but comparable between groups. Only selected groups are shown.

### 8.7 *Time-series fit, biomass*

Figure 7. Time-series fit for biomasses in the Chesapeake Bay Fisheries Ecosystem Model. Time series from assessments or surveys are shown as dots, while Ecosim simulation results are indicated with lines. The time period (X-axis) is 1950-2002 for all plots.

## **8.8**      *Time-series fit, catches*

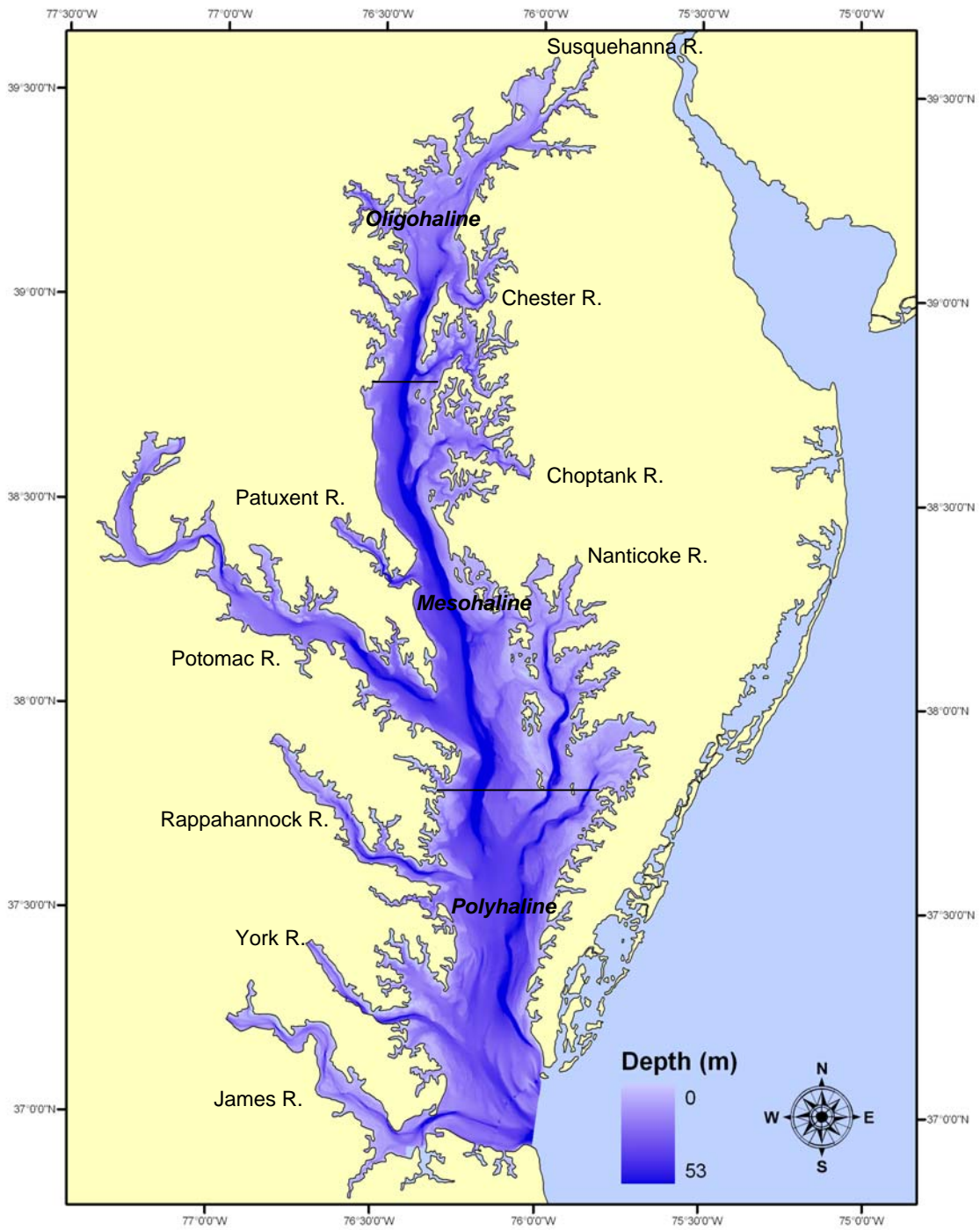
Figure 8. Time-series fit for catches in the Chesapeake Bay Fisheries Ecosystem Model. Catch time series are shown as dots, while the catches predicted by Ecosim (from biomasses and fishing mortalities) are shown as lines. The time period (X-axis) is 1950-2002 for all plots. Where a simulation matches the catches for all years, it indicates that the catches were used to estimate fishing mortalities for the Ecosim run.

## **8.9**      *Impact of nutrient loading on primary production*

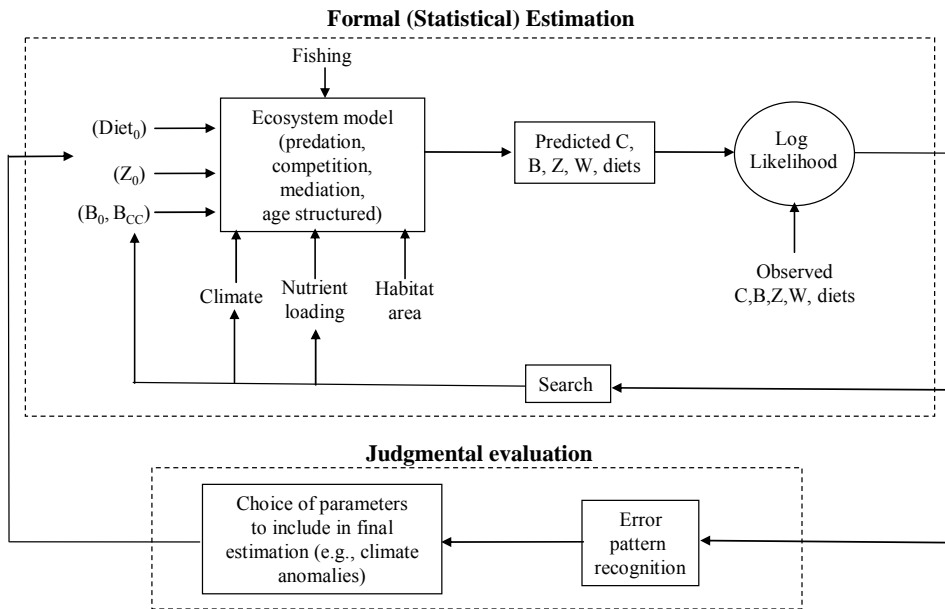
Figure 9. Chlorophyll-a ration from CBREEM (1950-2001) used to drive primary production rate in Ecosim.

# 9 Figures

9.1 Figure 1.

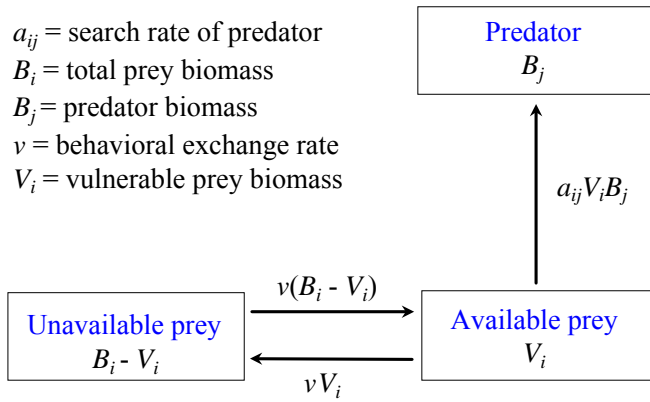


9.2 *Figure 2.*



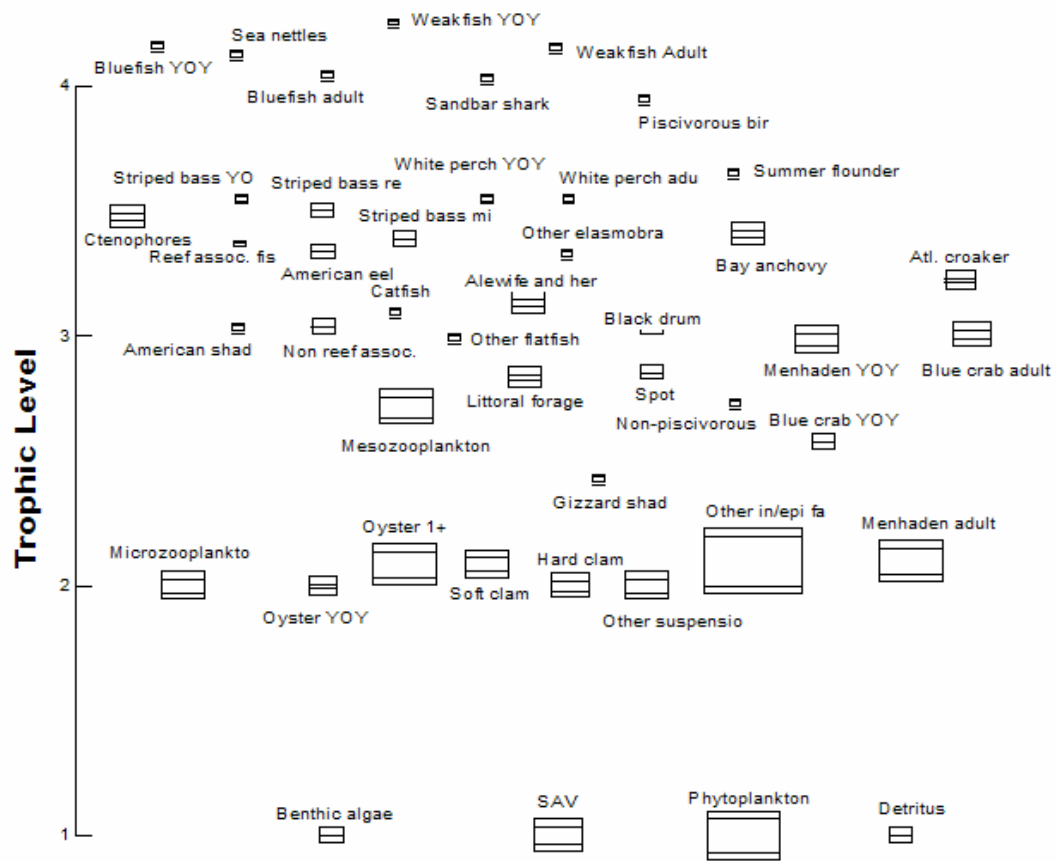
9.3 *Figure 3.*

$a_{ij}$  = search rate of predator  
 $B_i$  = total prey biomass  
 $B_j$  = predator biomass  
 $v$  = behavioral exchange rate  
 $V_i$  = vulnerable prey biomass

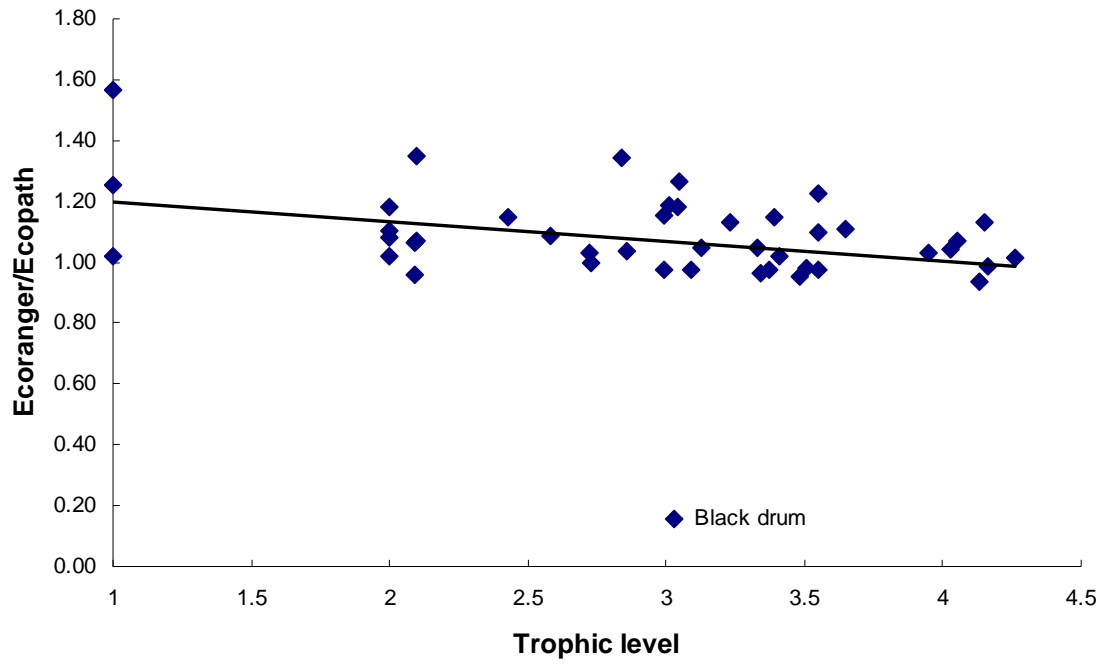




9.4 Figure 4.

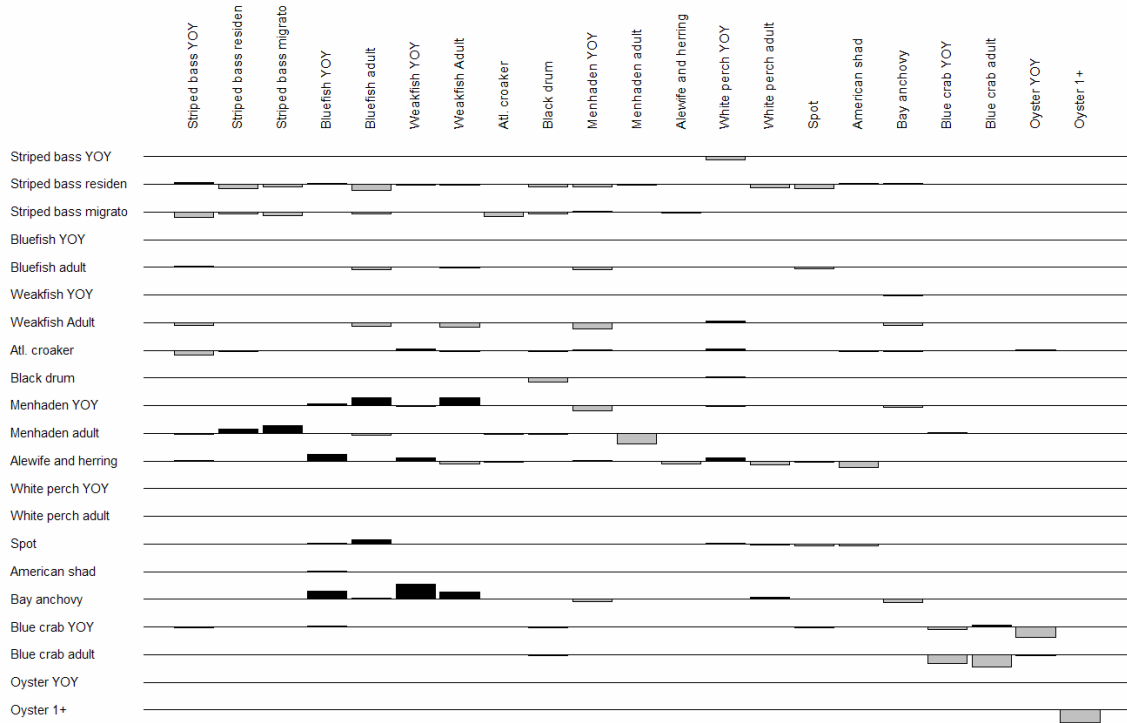


9.5 *Figure 5.*

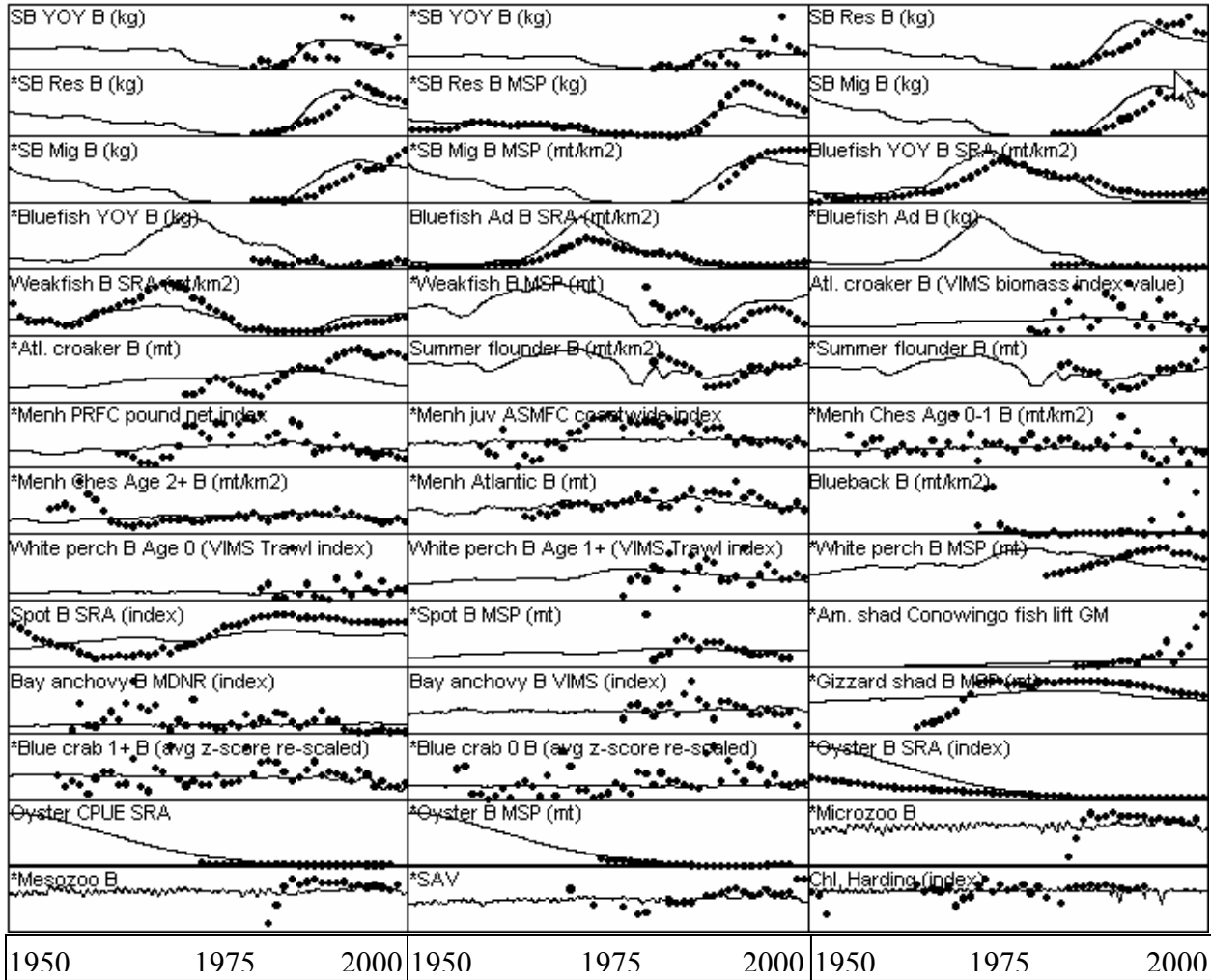


9.6

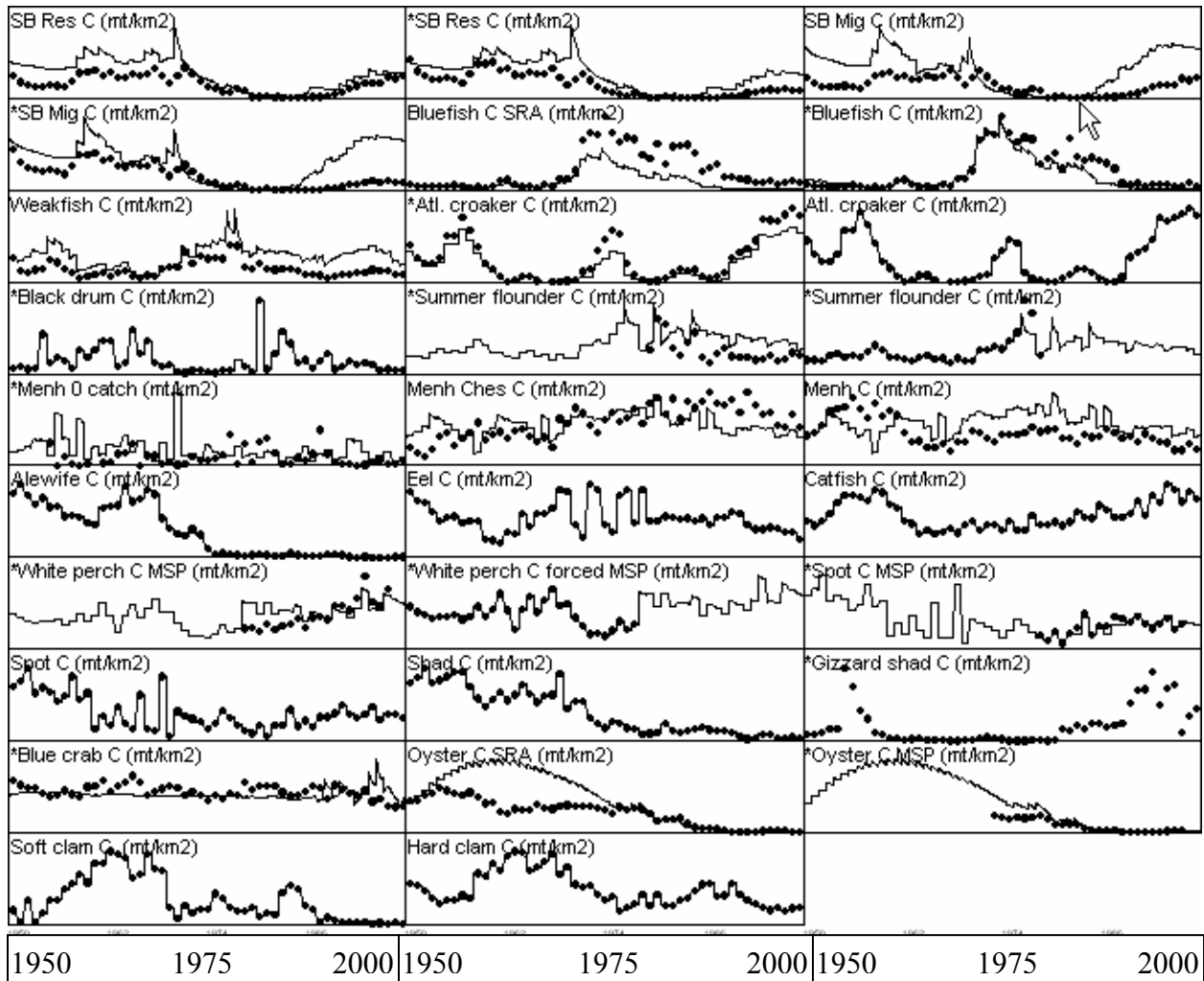
Figure 6.



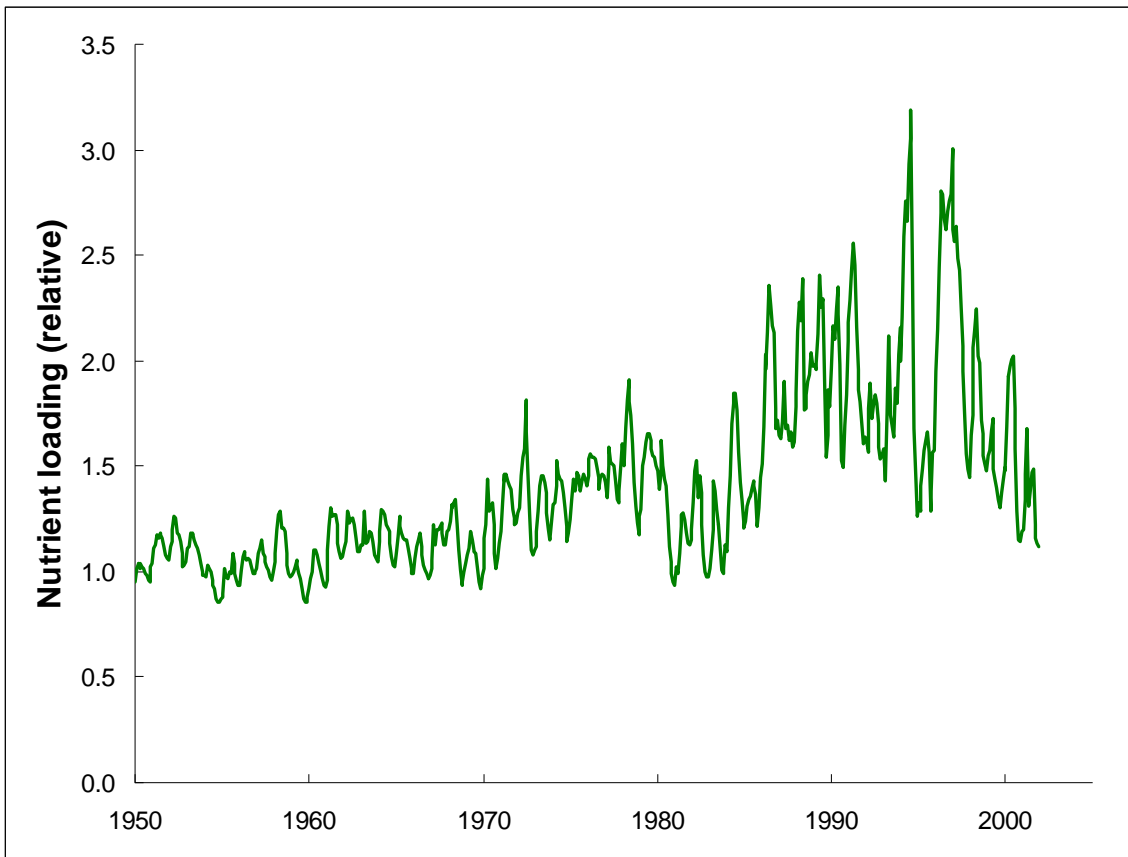
9.7 *Figure 7.*



9.8 *Figure 8.*



9.9 *Figure 9.*



## 10 Appendices – Description of methods for estimating basic input parameters and determining time series for CBFEM

In the appendices, the fish groups have been separated into “Commercial” and “Other fish.” The split is not used in the EwE model; it is only introduced as a matter of convenience in this report. Likewise, the invertebrates have been grouped into “Commercial” and “Other invertebrates.”

Where sources for diet compositions are omitted in the following data sections, they were based upon advice from local experts at the Chesapeake Bay Ecopath Workshop (Sellner *et al.*, 2001) and general knowledge of these species’ trophic behavior as reported in Hagy (2002) and Baird and Ulanowicz (1989). Commercial fish diet compositions are listed in Tables 20 and 21. Diets for other fish, commercial invertebrates, other invertebrates, and birds are listed in Tables 22, 23, 24, and 25, respectively.

### 10.1 Appendix A: Basic Input Parameters

#### 10.1.1 Commercial fishes

##### 10.1.1.1 Striped bass: young of the year (YOY), resident and migratory (*Morone saxatilis*) (FEM Groups 1, 2, and 3)

Striped bass is one of the higher trophic level predators in the Chesapeake Bay. It is a prized sport fish and of great value for both commercial and recreational fisheries in the Bay (Hartman, 2003). The present fisheries are the result of a successful recovery effort that began in the early 1980s, when catch levels were heavily curtailed after the stock had collapsed. By 1995, the stock was deemed to have recovered, and biomass is now often described as being at or near ‘historic levels’ (Hartman and Margraf, 2003).

Three stanzas (life stages) were created to represent this species: young-of-the-year (YOY), resident, and migratory. These age divisions mirror behavioral changes exhibited by the species on the Atlantic coast (Walter and Austin, 2003) and were based on discussions with local striped bass experts at workshops sponsored by this project. YOY are aged 0-11 months. The resident component is defined as fish less than 711 mm, a length representing the age at which the ASMFC considers striped bass to be migratory. This corresponds to ages 12–83 months (1–6.9 years). The migratory component includes ages 84+ months ( $\geq 7$  years).

The leading stanza for entry of biomass for this group is the resident component, as biomass estimates for YOY as well as the migratory component of the stock utilizing the Bay are poorly understood. For Q/B, the migrant population is the leading stanza. Ecopath estimates YOY biomass and consumption rates, resident Q/B, and migrant B, based on the lead parameters, the von Bertalanffy growth parameter (annual  $K = 0.11$ , average of FishBase values, Froese and Pauly, 2004), and an estimate of the ratio of the weight-at-maturity to the  $W_{inf}$  of 0.1. For a presentation of the life stage calculations used in the Ecopath model, see Christensen and Walters (2004).

#### **10.1.1.1.1 1950 biomass**

We used an estimate of  $2.1 \text{ t} \cdot \text{km}^{-2}$  as leading biomass for the resident part of the population based on the assumption that the population is back at its ‘historic level’. Migratory biomass is estimated at  $2.949 \text{ t} \cdot \text{km}^{-2}$  and young-of-the-year biomass at  $0.0125 \text{ t} \cdot \text{km}^{-2}$ .

#### **10.1.1.1.2 1950 P/B**

Estimates of  $Z (= P/B)$  for resident and migratory fish were obtained from virtual population analysis (VPA) (ASMFC, 2003b) and tagging results (Latour, unpublished results). The ASMFC assumes natural mortality  $M$  to be  $0.15 \text{ year}^{-1}$  (Smith *et al.*, 2000). The average  $F$  for the reference years (ages 4-13) used by the ASMFC was  $0.32 \text{ year}^{-1}$ ; thus  $Z = P/B = 0.47 \text{ year}^{-1}$ . Based on this, we used an assumed  $P/B$ -estimate of  $0.4 \text{ year}^{-1}$  for resident striped bass, a lower  $P/B$ -estimate of  $0.3 \text{ year}^{-1}$  for the migrant portion of the population, and  $1.8 \text{ year}^{-1}$  for the YOY.

#### **10.1.1.1.3 1950 Q/B**

For striped bass and most of the fish groups, consumption ( $Q/B$ ) values were determined by the empirical equation available in FishBase (Froese and Pauly, 2004), which requires that estimates be provided for  $W_{\text{inf}}$ ; average environmental temperature; fin aspect ratio (ratio of the ratio of the square of the height of the caudal fin and its surface area); and food type (detritivore, herbivore, omnivore, or carnivore, Palomares and Pauly, 1998). For the migratory striped bass,  $Q/B$  was estimated to  $2.3 \text{ year}^{-1}$ , given the parameter estimates of  $T = 17 \text{ }^{\circ}\text{C}$ ,  $W_{\text{inf}} = 115,760 \text{ g}$ , aspect ratio = 2.31, and carnivore diet.  $Q/B$  for resident striped bass was estimated to  $4.41 \text{ year}^{-1}$ , and YOY at  $23.266 \text{ year}^{-1}$ .

#### **10.1.1.1.4 Diet compositions**

For all striped bass stanzas, a diet item contributing less than 1% to total diet in a referenced study was not considered for determination of the modeled diet composition. Striped bass YOY diets were derived from three sources: Hartman and Brandt (1995), Rudershausen (1994), and Markle and Grant (1970). Grass shrimp, mysids, stone crabs, and benthic invertebrates were combined as ‘other in/epi fauna.’ Killifish, naked gobies, silversides, and other small fishes were combined as littoral zone forage fish. For striped bass residents, diet data were found in Hartman and Brandt (1995) and Walter (1999). Grass shrimp, mysids, stone crabs, polychaetes, and other benthic invertebrates were combined as ‘other in/epi fauna’. Weighted averages were used to determine resident diet using three age-classes. Diet data were weighted 1.0 for ages 1 and 2; 4.0 for ages 3+. For the migratory stanza diets, composition data were based on Hartman and Brandt (1995) and Walter (1999). Grass shrimp, mysids, stone crabs, mantis shrimp, and other benthic invertebrates were combined as ‘other in/epi fauna.’

### **10.1.1.2 Bluefish: YOY and adult (*Pomatomus saltatrix*) (FEM groups 4 and 5)**

#### **10.1.1.2.1 1950 biomass**

Bluefish are represented in the model by two stanzas: YOY (aged 0–11 months) and adults (aged 12+ months). The adult stanza is the lead stanza for this group. Adult biomass was based on the  $F$  derived from a coast-wide biomass dynamic model (Lee, 2003b) and catches in the Chesapeake Bay (Piavis, personal communication, 2003).  $M$  was assumed to be  $0.26 \text{ year}^{-1}$ . The YOY biomass was derived by Ecopath based on estimates of  $K = 0.26 \text{ year}^{-1}$  from FishBase (Froese and Pauly, 2004) and  $W_m/W_{\text{inf}} = 0.20$  (Table 7). The YOY biomass was estimated to be  $0.0161 \text{ t} \cdot \text{km}^{-2}$ . No changes were made to the  $Z$  of YOY. The adult biomass estimate was based upon a delay difference model of the group, tuned to the coast-wide VPA data available from the ASMFC. The adult biomass estimate is  $0.24 \text{ t} \cdot \text{km}^{-2}$ .



#### **10.1.1.2.2 1950 P/B**

A biomass dynamic model (Lee, 2003b) was used to derive coast-wide estimates of  $F$  (0.257-0.718 year<sup>-1</sup>) for adult bluefish age-classes. Note that coast-wide values for  $F$  are likely to be higher than Bay-specific  $F$ s (Gartland, 2006). For 1950, the  $F$  for the older stanza was estimated to be 0.483 year<sup>-1</sup>, and an overall, assumed  $Z$  of 0.589 year<sup>-1</sup> was used for this group.  $Z$  for the younger stanza was assumed, at 5.65 year<sup>-1</sup>.

#### **10.1.1.2.3 1950 Q/B**

The adult bluefish leading parameter  $Q/B$  was estimated using the empirical relationship in FishBase as 3.3 year<sup>-1</sup> assuming  $T = 17$  °C,  $W_{inf} = 16,962$  g, a fin aspect ratio of 2.55, and carnivorous diet. YOY  $Q/B$  was estimated to be 18.111 year<sup>-1</sup>.

#### **10.1.1.2.4 Diet compositions**

Adult bluefish diet was based on information contained in Hartman and Brandt (1995). Any component less than 1% in the diet was ignored in the model. Butterfish and harvestfish were combined as non-reef demersal fish. The diet data were averaged over six months (summer, fall, and part of winter), representing the time that they are resident in the Bay. Diet composition for the YOY stanza was derived from Hartman and Brandt (1995) and Gartland (2006), ignoring those elements that compose less than 1% of the diet. Bay anchovy and striped anchovy in Gartland (2006) were combined as bay anchovy; 'unknown fish' in Gartland (2006) was included with littoral forage fish; and 'shrimp' was placed into 'other in/epi fauna.' Diet data were averaged over the six months of residency (summer, fall, and part of winter) and over the two studies.

### **10.1.1.3 Weakfish: YOY and adult (*Cynoscion regalis*) (FEM groups 6 and 7)**

#### **10.1.1.3.1 1950 biomass**

Weakfish are represented by two stanzas, YOY (0-11 months) and adults (12+ months), with adults as the leading stanza. Adult biomass was derived from the coast-wide VPA (Kahn, 2002) adjusted to reflect the Chesapeake Bay. Specifically, catch data from the Chesapeake Bay was compared to that of the entire coast. That fraction was then applied to the overall coast-wide population estimate to derive a population biomass estimate for the Bay. YOY biomass was estimated by Ecopath assuming  $K = 0.26$  year<sup>-1</sup> from FishBase (Froese and Pauly 2004) and  $W_m/W_{inf} = 0.1$ . As leading biomass we used an estimate for the adult stanza derived from an age-structured model and estimated to 0.489 t · km<sup>-2</sup>. For the YOY, biomass was estimated at 0.0257 t · km<sup>-2</sup>.

#### **10.1.1.3.2 1950 P/B**

$F$  was estimated to be approximately 0.2 year<sup>-1</sup> in the late 1990s for the coast-wide weakfish stock (Spear *et al.*, 2003), and  $M$  was estimated to be 0.25 year<sup>-1</sup> for all stock assessment purposes (Smith *et al.*, 2000). Thus, the current  $Z = P/B \approx 0.45$  year<sup>-1</sup>. For 1950,  $F$  was estimated at 0.585 year<sup>-1</sup>, and a  $Z$  value of 0.685 year<sup>-1</sup> was used, *i.e.*, a lower natural mortality than used in the current stock assessments was assumed as we cannot quantify as high predation mortality as assumed in the assessments. YOY  $Z$  was estimated to be 4.0 year<sup>-1</sup>.

### **10.1.1.3.3 1950 Q/B**

The adult weakfish Q/B value was the leading parameter, and was estimated at 3.1 year<sup>-1</sup> using the empirical formula from FishBase (Froese and Pauly, 2004) with T = 17 °C, W<sub>inf</sub> = 8,850 g, aspect ratio = 1.32, and carnivorous diet. YOY Q/B was estimated to be 13.525 year<sup>-1</sup>.

### **10.1.1.3.4 Diet compositions**

Both stanzas of weakfish had diet compositions derived from Hartman and Brandt (1995). For both stanzas, grass shrimp and mysids were added to ‘other in/epi fauna.’ Diet data were averaged over six months (summer, fall, and part of winter), representing residency time in the Bay.

### **10.1.1.4 Atlantic croaker (*Micropogonias undulatus*) (FEM group 8)**

#### **10.1.1.4.1 1950 biomass**

Little stock assessment data are available for the Atlantic croaker (Desfosse *et al.*, 1999; Austin *et al.*, 2003), although they are one of the most abundant bottom fish in the Chesapeake Bay. Good year-classes appear to have sustained relatively high catches for the commercial fishery from 1997 to 2002, and the stock appears resistant to growth overfishing, depending on F assumptions. Abundance estimates were calculated based on sampling area of trawl (5,402 km<sup>2</sup>) and on the assumption that the trawl net efficiency is 0.4, based on hydroacoustic data (Hoffman, Personal communication). Abundance estimates were converted to biomass assuming an average weight value from trawl catches. Densities were calculated using an area of 5,402 km<sup>2</sup>, and were entered into the model under the assumption that those densities apply to the entire Bay. These data were based on unpublished information provided by staff of the Virginia Institute of Marine Science (VIMS) Chesapeake Bay Multispecies Monitoring (ChesMMAP) survey ([www.fisheries.vims.edu/chesmmap/](http://www.fisheries.vims.edu/chesmmap/)). The resulting biomass was 1.67 t · km<sup>-2</sup>, and this biomass was used for the model in the absence of time trend information for croaker.

#### **10.1.1.4.2 1950 P/B**

An annual total mortality for the Chesapeake Bay Atlantic croaker stock was estimated to be 55 to 60% per year (Austin *et al.*, 2003). Using the higher end as a conservative mortality estimate yields a P/B = 0.916 year<sup>-1</sup>.

#### **10.1.1.4.3 1950 Q/B**

Q/B for Atlantic croaker was estimated from empirical relationship in FishBase to be 5.4 year<sup>-1</sup>, assuming that T = 17 °C, W<sub>inf</sub> = 2580 g, aspect ratio = 1.32, and carnivorous diet, (Desfosse *et al.*, 1999)

#### **10.1.1.4.4 Diet compositions**

About half of the diet for Atlantic croaker was designated as ‘imported,’ representative of their six-month residency period in the Bay. FishBase (Froese and Pauly, 2004) suggests that their diet is made up mostly of demersal invertebrates and some larval fish.

### **10.1.1.5 Black drum (*Pogonias cromis*) (FEM group 9)**

#### **10.1.1.5.1 1950 biomass**

Black drum are managed as a single stock along the continental East Coast (Jones and Wells, 2001). No estimate of stock size was available for the Chesapeake Bay. Ecopath estimated biomass by using an

assumed ecotrophic efficiency for 1950 of 0.1. This low EE resulted in an initial biomass sufficient to balance the values of F estimated from catches. The biomass estimate is  $1.263 \text{ t} \cdot \text{km}^{-2}$ .

#### **10.1.1.5.2 1950 P/B**

Total annual mortality for black drum is estimated to range from 0.08 to 0.11 year<sup>-1</sup> (Jones and Wells, 2001). In the absence of other evidence, the median value of  $M = 0.095 \text{ year}^{-1}$  was used. No reliable estimate of F was available, so it was assumed, conservatively, to equal M. Thus,  $Z = P/B \approx 0.190 \text{ year}^{-1}$ . The model as outlined does not include any predation on black drum.

#### **10.1.1.5.3 1950 Q/B**

Q/B for black drum was estimated using the empirical relationship in FishBase (Froese and Pauly, 2004) as  $2.100 \text{ year}^{-1}$ , assuming that  $T = 17 \text{ }^\circ\text{C}$ ,  $W_{\text{inf}} = 57612 \text{ g}$ , aspect ratio = 1.32, and carnivore diet.

#### **10.1.1.5.4 Diet compositions**

The black drum diet composition was based on information made available by VIMS ChesMMAAP.

#### **10.1.1.6 Summer flounder (*Paralichthys dentatus*) (FEM group 10)**

##### **10.1.1.6.1 1950 biomass**

The 1950 biomass of summer flounder was estimated to be  $0.454 \text{ t} \cdot \text{km}^{-2}$ , based on an assumed EE of 0.95.

##### **10.1.1.6.2 1950 P/B**

The summer flounder 2002 advisory report noted that this species is overfished in the Northeast, and that there was an 80% chance that F in 2001 was between 0.24 and 0.32 year<sup>-1</sup>, having declined from about 1.32 year<sup>-1</sup> in 1994 (NFSC, 2002a). The more detailed analysis of the stock (NFSC, 2002b) suggests that natural mortality is about 0.2 year<sup>-1</sup>. An estimate of total mortality would therefore be  $Z = P/B \approx 0.520 \text{ year}^{-1}$ . This estimate was used for the model for lack of any other information.

##### **10.1.1.6.3 1950 Q/B**

Summer flounder Q/B was calculated to be  $2.900 \text{ year}^{-1}$  using the empirical equation available in FishBase (Froese and Pauly, 2004) assuming  $T = 17 \text{ }^\circ\text{C}$ ,  $W_{\text{inf}} = 12,000 \text{ g}$ , an aspect ratio of 1.32, and carnivorous diet.

##### **10.1.1.6.4 Diet compositions**

The diet composition of summer flounder was derived from information provided by the ChesMMAAP 2002 Bay-wide trawl survey ([www.fisheries.vims.edu/chesmmap/](http://www.fisheries.vims.edu/chesmmap/)) using samples from the main stem of the Bay. Anything contribution less than 1.0% of the diet was ignored. Bay anchovy and striped anchovy were combined as one group. The diet category ‘non-reef-associated fish’ included spotted hake, silver perch, and northern sea robin. ‘Other in/epi fauna’ included mantis shrimp and mysids.

### **10.1.1.7 Atlantic menhaden (*Brevoortia tyrannus*): YOY and adult (FEM groups 11 and 12)**

#### **10.1.1.7.1 1950 biomass**

This species was represented in the model by two stanzas, juveniles (age 0-1) and adults (age 2+), with adults as the leading stanza. The 1950-biomass was assumed to be  $33.000 \text{ t} \cdot \text{km}^{-2}$  for the adult group to balance the demand for the stanza, while the juvenile biomass was estimated by Ecopath to  $18.089 \text{ t} \cdot \text{km}^{-2}$ , assuming  $K = 0.424 \text{ year}^{-1}$  based on estimated  $K$  from 1992 to 2002 (ASMFC, 2004) and  $W_m/W_{inf} = 0.24$ . These weights were calculated from length data (ASMFC, 2004), which showed that menhaden mature when their length is 180–230 mm, (average 205 mm), and that from 1992-2002,  $L_{\infty}$  averaged 328 mm. The weight ratio for the multi-stanza group was approximated by cubing these values.

#### **10.1.1.7.2 1950 P/B**

P/B of the two stanzas was derived from  $M$  and  $F$  values in the Atlantic coast menhaden stock assessment (ASMFC, 2004).  $M$  was estimated to be  $\sim 1.5 \text{ year}^{-1}$  for juveniles, and assumed to be  $0.3 \text{ year}^{-1}$  for age 2+ fish. Since fishery on age 0-1 is negligible, the juvenile P/B is approximately  $1.5 \text{ year}^{-1}$ . For the age 2+ group, a P/B of  $0.8 \text{ year}^{-1}$  was used.

#### **10.1.1.7.3 1950 Q/B**

For the adult group, a Q/B of  $7.8 \text{ year}^{-1}$  was used, which for the juveniles leads to a Q/B of  $15.86 \text{ year}^{-1}$  based on stanza-calculations.

#### **10.1.1.7.4 Diet compositions**

The diet composition of menhaden is poorly understood, and only qualitative knowledge of feeding characteristics was available. Much of this knowledge was synthesized by the Menhaden Working Group of the Chesapeake Research Consortium Scientific and Technical Advisory Committee (2002). This report suggests menhaden shifts diet from primarily zooplanktivorous as YOY to almost entirely phytoplanktivorous for age 1+. This qualitative ontogenetic shift was mirrored in the modeled diet, with one third of adult diet designated as ‘imported material’ to represent the time they spend out of the Bay ecosystem.

### **10.1.1.8 Alewife/herring (*Alosa pseudoharengus/Clupea harengus*) (FEM group 13)**

#### **10.1.1.8.1 Biomass for 1950**

This group includes alewife and blueback herring (*Alosa aestivalis*). Based on an annual average of the four seasonal models in Baird and Ulanowicz (1989), a biomass was estimated and converted to wet weight. The conversion factor ( $0.16 \text{ g DW} / \text{g WW}$ ) was determined by taking an average of weight carbon to dry weight and dry to wet weight in Jørgensen *et al.* (2000). The resulting biomass seemed rather low to local experts interviewed for this report. Therefore, the biomass was estimated by Ecopath instead at  $5.986 \text{ t} \cdot \text{km}^{-2}$ , assuming that the ecotrophic efficiency of these species in the Bay was 0.95.

#### **10.1.1.8.2 1950 P/B**

Total mortality for this group was based on the P/B of  $0.75 \text{ year}^{-1}$  for alewife in Randall and Minns (2000).

#### **10.1.1.8.3 1950 Q/B**

The consumption ratio ( $9.4 \text{ year}^{-1}$ ) for this group was the average of Q/B values listed for herring ( $10.1 \text{ year}^{-1}$ ) and alewife ( $8.62 \text{ year}^{-1}$ ) in FishBase (Froese and Pauly, 2004).

#### **10.1.1.8.4 Diet compositions**

Alewife and herring spend a large portion of their life in the open ocean, but make annual spawning runs to rivers that feed the Bay and spend about half the year in the Bay. Based on qualitative information available from VIMS (2004), the diet of alewife and herring consists of a mix of mostly zooplankton and some phytoplankton.

#### **10.1.1.9 American eel (*Anguilla rostrata*) (FEM group 14)**

##### **10.1.1.9.1 1950 biomass and EE**

Stock assessments for this group were not available, so Ecopath was made to estimate biomass at  $3.22 \text{ t} \cdot \text{km}^{-2}$ , by setting EE to 0.5. This EE estimate was based on well-known aspects of the life history of American eel: They can live for 25 years and leave the Bay to spawn and die in the Sargasso Sea.

##### **10.1.1.9.2 1950 P/B**

The total mortality,  $0.25 \text{ year}^{-1}$ , was based on P/B for American eel in Randall and Minns (2000).

##### **10.1.1.9.3 1950 Q/B**

The consumption ratio,  $2.5 \text{ year}^{-1}$ , was obtained from the empirical equation in FishBase (Froese and Pauly, 2004) assuming,  $T = 17 \text{ }^{\circ}\text{C}$ ,  $W_{\text{inf}} = 9,065 \text{ g}$ , an aspect ratio of 1.32, and carnivore diet.

##### **10.1.1.9.4 Diet compositions**

The diet composition for American eel was based upon qualitative information found in the American Eel Plan Development Report (ASMFC, 2000).

#### **10.1.1.10 Catfishes (*Ameiurus catus*, *A. nebulous*, *A. natalis*, *Ictalurus punctatus*, *I. furcatus*, *Pylodictis olivaris*) (FEM group 15)**

Catfish are predominantly freshwater species, but also occur in estuarine areas. There are three native species in the Chesapeake Bay: white catfish (*Ameiurus catus*), brown bullhead (*A. nebulous*), and yellow bullhead (*A. natalis*). The introduced channel catfish (*Ictalurus punctatus*) and blue catfish (*I. furcatus*), both of which have economic importance in the Bay, and the rarer flathead catfish (*Pylodictis olivaris*), also reside in the Bay ([www.chesapeakebay.net](http://www.chesapeakebay.net)).

##### **10.1.1.10.1 1950 biomass**

Catfish biomass was estimated by Ecopath to be  $1.155 \text{ t} \cdot \text{km}^{-2}$ , assuming that ecotrophic efficiency was 0.95, *i.e.*, that the model explains 95% of the mortality of the catfish.

##### **10.1.1.10.2 1950 P/B**

Total mortality for catfish of  $0.28 \text{ year}^{-1}$  was based upon the P/B value for channel catfish in Randall and Minns (2000).

#### **10.1.1.10.3 1950 Q/B**

Consumption/biomass ratio was estimated at  $2.5 \text{ year}^{-1}$  using empirical relationship in FishBase (Froese and Pauly, 2004) and parameters for channel catfish,  $T = 17 \text{ }^{\circ}\text{C}$ ,  $W_{\text{inf}} = 26 \text{ kg}$ , an aspect ratio of 1.32, and carnivorous diet.

#### **10.1.1.10.4 Diet compositions**

The diet composition of the catfish group was based on knowledge of these fishes provided by local experts as part of the Chesapeake Bay-area modeling workshops (Sellner *et al.*, 2001).

#### **10.1.1.11 White perch: YOY and adult (*Morone americana*) (FEM groups 16 and 17)**

##### **10.1.1.11.1 1950 biomass**

White perch was represented in the model by two stanzas: YOY age 0–11 months, and adults aged 12+ months. The adult stanza was the lead stanza for this group. The biomass for the adult group for 1950 of  $0.300 \text{ t} \cdot \text{km}^{-2}$  is a guessed value. YOY biomass of  $0.00305 \text{ t} \cdot \text{km}^{-2}$  was estimated by Ecopath assuming  $K = 0.10 \text{ year}^{-1}$ , an average of values from FishBase (Froese and Pauly, 2004), and  $W_{\text{m}}/W_{\text{inf}} = 0.1$ .

##### **10.1.1.11.2 1950 P/B**

Otolith aging from the Choptank River indicated that  $M = 0.15 \text{ year}^{-1}$  for adults (Casey *et al.*, 1988). A biomass dynamic model of white perch suggested that fishing mortality from 1996 to 2000 averaged  $F = 0.54 \text{ year}^{-1}$ . For 1950, an estimate of  $0.500 \text{ year}^{-1}$  was used. The YOY P/B was assumed to be  $2.0 \text{ year}^{-1}$  for 1950.

##### **10.1.1.11.3 1950 Q/B**

The consumption ratio of white perch adults,  $4.2 \text{ year}^{-1}$ , was estimated with the empirical equation in FishBase (Froese and Pauly, 2004) assuming  $T = 17^{\circ}\text{C}$ ,  $W_{\text{inf}} = 2178\text{g}$ , an aspect ratio of 1.32, and carnivorous diet. YOY Q/B was estimated to be  $19.921 \text{ year}^{-1}$ .

##### **10.1.1.11.4 Diet compositions**

The diet composition of YOY white perch in the model was obtained from Rudershausen (1994), who used beach seine and trawl sampling to collect juveniles in the James River. Anything comprising less than 1% of the diet was ignored, and decapods, mysids, polychaetes, amphipods, *etc.*, were combined into ‘other in/epi fauna’. Fish as prey items were assumed to be littoral zone forage fish. A small portion of ‘imported food’ was used to account for insects and insect larvae. The diet of white perch adults consists almost entirely of benthic invertebrates (Luo *et al.*, 1994). Because this preference appears to become greater as the fish age (St-Hilaire *et al.*, 2002), the adult white perch diet is almost entirely ‘other in/epi fauna,’ with some small fishes also included.

#### **10.1.1.12 Spot (*Leiostomus xanthurus*) (FEM group 18)**

##### **10.1.1.12.1 1950 biomass**

The biomass of spot was estimated by Ecopath to be  $1.674 \text{ t} \cdot \text{km}^{-2}$ , by setting the ecotrophic efficiency to an assumed value of 0.90.

#### **10.1.1.12.2 1950 P/B**

The annual mortality rate for adult spot has been estimated to be 80%, *i.e.*,  $Z = P/B \approx 1.6 \text{ year}^{-1}$ , with a maximum life span of five years; few fish over three years old are found (Pacheco, 1962; cited by Homer and Mihursky, 1991). This mortality estimate was considered to be on the high side, and instead a lower guessed value of  $Z = 1.0 \text{ year}^{-1}$  was used.

#### **10.1.1.12.3 1950 Q/B**

The consumption ratio of spot was estimated at  $5.8 \text{ year}^{-1}$  using the empirical equation in FishBase (Froese and Pauly 2004), assuming  $T = 17 \text{ }^\circ\text{C}$ ,  $W_{\text{inf}} = 466 \text{ g}$ , an aspect ratio of 1.39, and carnivorous diet.

#### **10.1.1.12.4 Diet compositions**

The diet composition of spot was adapted from Homer and Mihursky (1991) and adjusted to reflect migration, although juveniles are present nearly all year round.

### **10.1.1.13 American shad (*Alosa sapidissima*) (FEM group 19)**

#### **10.1.1.13.1 1950 biomass**

Biomass estimates for American shad in 1950 are unavailable, and a biomass of  $0.4 \text{ t} \cdot \text{km}^{-2}$  was assumed.

#### **10.1.1.13.2 1950 P/B**

The total mortality estimate for American shad of  $0.7 \text{ year}^{-1}$  was based upon P/B for alewife (*Alosa pseudoharengus*) in Randall and Minns (2000).

#### **10.1.1.13.3 1950 Q/B**

The consumption ratio for American shad,  $3.500 \text{ year}^{-1}$ , was estimated with FishBase, assuming  $T = 17^\circ\text{C}$ ,  $W_{\text{inf}} = 5,500 \text{ g}$ , an aspect ratio of 1.32, and carnivorous diet.

#### **10.1.1.13.4 Diet compositions**

American shad diet composition was derived from Walter and Olney (2003), which used percentage by weight from diet analysis of adult American shad during their spawning run in the York River, Virginia. Diet items contributing less than 1% of the diet were ignored. YOY American shad were assumed to eat 100% mesozooplankton (Hoffman, pers. comm.), and to reside in the Bay from April to November. Therefore, YOY diet composition is one third 'imported' matter. For adults, calanoid copepod food items were included as 'mesozooplankton' and mysids as 'other in/epi fauna.' About one-third of the adult diet was assumed to be 'imported' to account for migratory behavior. To generate final input values, weighted averages were calculated for the diet items based on eight age-groups (juveniles plus seven 'adult' age-classes; adults ranged from age 3-9).

## 10.1.2 Other fishes

### 10.1.2.1 Bay anchovy (*Anchoa mitchilli*) (FEM group 20)

#### 10.1.2.1.1 1950 biomass

A recent biomass estimate for bay anchovy was based on data from Jung (2002), which showed that the standing stock biomass from 1995 to 2000 averaged 34,000 t. With 10,000 km<sup>2</sup> as the area of the Chesapeake Bay, the biomass in the late 1990s would have been around 3.4 t · km<sup>-2</sup>. The growth model of Luo and Brandt (1993) suggested a higher biomass of around 16 t · km<sup>-2</sup>. Here, the biomass of 3.4 t · km<sup>-2</sup> was used for the bay anchovy group for 1950 as well (while we estimate a similar biomass for the late 1990s).

#### 10.1.2.1.2 1950 P/B

Houde and Zastrow (1991) reported bay anchovy adult mortality rates ranging between 89% and 95% annually. Luo and Brandt (1993) suggested that a 95% mortality rate was appropriate for the species, although Jung (2002) found that mortality rates can be higher, and are in fact much higher for larvae and juveniles. Because the population being modeled will be dominated by the biomass of adult anchovy, the P/B ratio was calculated based on the 95% mortality rate, *i.e.*,  $P/B \approx 3.0 \text{ year}^{-1}$ .

#### 10.1.2.1.3 1950 Q/B

The consumption ratio for bay anchovy, 10.900 year<sup>-1</sup>, was estimated with the empirical equation in FishBase (Froese and Pauly, 2004), assuming  $T = 17^\circ\text{C}$ ,  $W_{\text{inf}} = 20\text{g}$  (based on length weight relationships reported by Jung, 2002), an aspect ratio of 1.32, and carnivorous diet.

#### 10.1.2.1.4 Diet compositions

Bay anchovy diet was based on Houde and Zastrow (1991), which contains a general description of anchovy diet, and on Hagy (2002), which reported diet composition as 67% mesozooplankton, 28% microzooplankton, 4% meroplankton (fish larvae, *etc.*), and 1% suspension feeders. Juveniles were assumed to eat copepodites and copepod nauplii, which were included as ‘mesozooplankton.’

### 10.1.2.2 Other flatfishes (FEM group 21)

#### 10.1.2.2.1 1950 biomass

The biomass of this diverse group, which includes hogchoker, tonguefish, window pane flounder, and winter flounder, was estimated by Ecopath to be 0.169 t · km<sup>-2</sup>, by assuming that ecotrophic efficiency was 0.95.

#### 10.1.2.2.2 1950 P/B

The P/B estimate for this group of 0.460 year<sup>-1</sup> is based on a value given for flatfish off the Atlantic seaboard in Sissenwine (1987).

#### 10.1.2.2.3 1950 Q/B

The estimated consumption ratio of 4.9 year<sup>-1</sup> was derived using the empirical equation in FishBase (Froese and Pauly, 2004), and is the average of Q/Bs calculated for winter flounder, windowpane, and hogchoker as representative species for the group, assuming that  $T = 17^\circ\text{C}$ ;  $W_{\text{inf}} = 3,600 \text{ g}$  (for winter



flounder);  $W_{inf} = 689$  g (for window pane);  $W_{inf} = 188$ g (for hog choker); an aspect ratio of 1.32; and carnivorous diet.

#### **10.1.2.2.4 Diet compositions**

The diet composition for ‘other flatfish’ was based on a synthesis of diet information for windowpane, winter flounder, and hogchoker in FishBase (Froese and Pauly, 2004) and diet information for hogchoker in Baird and Ulanowicz (1989).

#### **10.1.2.3 Gizzard shad (*Dorosoma cepedianum*) (FEM group 22)**

##### **10.1.2.3.1 1950 biomass**

The biomass of gizzard shad of  $2.086 \text{ t} \cdot \text{km}^{-2}$  was estimated by Ecopath assuming an ecotrophic efficiency of 0.95.

##### **10.1.2.3.2 1950 P/B**

Gizzard shad P/B was estimated to  $0.530 \text{ year}^{-1}$ , based on an estimate of M from empirical equations in FishBase assuming  $T = 17 \text{ }^\circ\text{C}$ ,  $K = 0.18 \text{ year}^{-1}$ ,  $L_{inf} = 43.6 \text{ cm}$ , and no fishing mortality. We tried to use a lower P/B for gizzard shad from Randall and Minns (2000), but this production rate was not sufficient to meet predation mortality requirements without resulting in demand for a huge biomass.

##### **10.1.2.3.3 1950 Q/B**

The consumption ratio for gizzard shad,  $14.5 \text{ year}^{-1}$ , was estimated using the empirical equation in FishBase (Froese and Pauly, 2004) assuming  $T = 17 \text{ }^\circ\text{C}$ ,  $W_{inf} = 1,980 \text{ g}$ , an aspect ratio of 1.32, and herbivorous diet.

##### **10.1.2.3.4 1950 EE**

The ecotrophic efficiency was assumed to be high (0.95) for gizzard shad, as it is a common forage fish, with predation accounting for much of its mortality. The species is exploited commercially only as bait and is not important to recreational fishers.

##### **10.1.2.3.5 Diet compositions**

Publications cited in FishBase (Froese and Pauly, 2004) suggested that the majority of gizzard shad diet is phytoplankton, with some zooplankton included for very large individuals. Local experts at the Chesapeake Bay Workshop (Sellner *et al.*, 2001) suggested that phytoplankton should be considered the vast majority of the diet.

#### **10.1.2.4 Reef-associated fishes (FEM group 23)**

This is a diverse group that includes several species, e.g., spadefish (*Chaetodipterus faber*), tautog (*Tautoga onitis*), toadfish (*Opsanus tau*), blennies (Blenniidae), adult gobies (Gobiidae), and black seabass (*Centropristis striata*). Tautog is considered a prized recreational species along the eastern seaboard; it is slow-growing and susceptible to overfishing, and is therefore subject to management as part of SAW/SARC and the ASMFC.

##### **10.1.2.4.1 1950 biomass**

The biomass was estimated by Ecopath to be  $0.232 \text{ t} \cdot \text{km}^{-2}$  by assuming an ecotrophic efficiency of 0.9.

#### **10.1.2.4.2 1950 P/B**

According to a recent coastwide assessment report (Stirratt *et al.*, 2002a) during the period from 1995-2000, F for tautog averaged  $0.4 \text{ year}^{-1}$ . Assessments assumed a natural mortality rate, M, of  $0.15 \text{ year}^{-1}$  (Stirratt *et al.*, 2002a; Stirratt *et al.*, 2002b). Therefore P/B for tautog  $\approx 0.55 \text{ year}^{-1}$ , and this value is assumed to be representative for the rest of the group. We used a lower estimate of  $0.51 \text{ year}^{-1}$  for the 1950 model.

#### **10.1.2.4.3 1950 Q/B**

The Q/B for this group,  $3.1 \text{ year}^{-1}$ , was estimated using the empirical equation in FishBase (Froese and Pauly, 2004), based on data for tautog and assuming that  $T = 17 \text{ }^{\circ}\text{C}$ ,  $W_{\text{inf}} = 8688 \text{ g}$ , an aspect ratio of 1.32, and a carnivorous diet.

#### **10.1.2.4.4 Diet compositions**

The diet composition of this group was estimated from the diet and food item entries cited in FishBase (Froese and Pauly, 2004) for tautog, toadfish, and black seabass. Tautog diet was described as benthic organisms, including mussels, gastropods, and crustaceans. Toadfish diet was described as 33% fish, 25% gastropods, 25% bivalves, 5% crustaceans, and the remainder as plant matter. Black seabass diet was described as mostly benthic crustaceans with some clupeids and zooplankton.

#### **10.1.2.5 Non-reef-associated fishes (FEM group 24)**

##### **10.1.2.5.1 1950 biomass**

This group was represented by species such as spotted hake (*Urophycis regia*), sea robins (*Prionotus carolinus*, *P. evolans*, and *P. tribulus*), lizard fish (*Synodus foetens*), butterfish (*Peprilus burti* and *P. triacanthus*), and harvest fish (*Peprilus alepudotus* and *P. paru*). Biomass for this group was estimated by Ecopath to be  $1.228 \text{ t} \cdot \text{km}^{-2}$ , assuming an ecotrophic efficiency of 0.9.

##### **10.1.2.5.2 1950 P/B**

Total mortality was estimated from mortality values empirically derived in FishBase (Froese and Pauly, 2004): for spotted hake,  $0.49 \text{ year}^{-1}$ ; for sea robins, 0.53, 0.5, and  $0.56 \text{ year}^{-1}$ ; for butterfish, 1.85 and  $1.19 \text{ year}^{-1}$ ; for harvest fish, 1.33 and  $1.26 \text{ year}^{-1}$ . Values were derived by assuming  $T = 17 \text{ }^{\circ}\text{C}$  and that total length  $L_{\infty} \approx L_{\text{max}}$ , when no estimate of  $L_{\infty}$  was available. These values suggest a group P/B of about  $1.0 \text{ year}^{-1}$ .

##### **10.1.2.5.3 1950 Q/B**

The consumption ratio of the group was estimated by Ecopath to be  $5.000 \text{ year}^{-1}$  by assuming that the production/consumption ratio for this group was 0.2. The P/Q ratio for most species will vary from  $\approx 0.05$  for long-lived, slow-growing creatures to  $\approx 0.3$  for small, fast-growing organisms (Christensen *et al.*, 2004). Given that many of the species in this group tended to be small and fast-growing, e.g., butterfish and harvest fish, while others were slower to mature, e.g., sea robins, the P/Q estimate of 0.2 should be reasonable.

##### **10.1.2.5.4 Diet compositions**

The diet composition for this group was synthesized from information for each species cited in FishBase (Froese and Pauly, 2004). Spotted hake adults were said to eat a mixture of fish and squid, whereas the

juveniles fed upon a mixture of benthos including filter feeders, crustaceans, and molluscs. Sea robin adults were described as eating mostly fish, while the juveniles targeted a variety of crustaceans. Lizard fish juveniles and adults were reported as eating mostly fishes. Butterfish and harvest fish were described as feeding on benthic invertebrates and detritus.

#### **10.1.2.6 Littoral forage fishes (FEM group 25)**

##### **10.1.2.6.1 1950 biomass**

Species that made up this group included striped and rainwater killifish (*Fundulus majalis* and *Lucania parva*), mummichogs (*Fundulus heteroclitus*), silversides (*Membras spp.* and *Menidia spp.*), silverperch (*Bairdiella chrysoura*), tonguefish (*Symphurus plagiusa*), and gobies (Gobiidae). The biomass for the group was estimated to be  $5.21 \text{ t} \cdot \text{km}^{-2}$ , by setting ecotrophic efficiency to 0.95.

##### **10.1.2.6.2 1950 P/B**

Total mortality for littoral forage fish was estimated by local experts at a Chesapeake Bay Ecopath Workshop (Sellner *et al.*, 2001) to be  $0.8 \text{ year}^{-1}$  and was assumed to be similar to other forage fish groups.

##### **10.1.2.6.3 1950 Q/B**

The consumption ratio for littoral forage fish of  $4.0 \text{ year}^{-1}$  was determined by setting a production/consumption ratio of 0.2.

##### **10.1.2.6.4 Diet compositions**

The diet composition for littoral forage fish was derived from data in Cicchetti (1998), which was reported in percent by volume. The study was conducted at Goodwin Island, at the mouth of the York River. In order to apply the data to the model group, anything less than 1% in the diet study was ignored. Grass shrimp, mysids, polychaetes, *etc.*, were added into 'other in/epi fauna.' Diet data were averaged over the habitats (five) and time period (June to October 1995) covered by the study and were also averaged over species: striped and rainwater killifish, mummichogs, silversides, silver perch, tonguefish, and several species of gobies.

#### **10.1.2.7 Sandbar shark (*Carcharhinus plumbeus*) (FEM group 26)**

##### **10.1.2.7.1 1950 biomass**

Using a model based upon fishing effort, Cortes *et al.* (2002) suggested that an  $F_{\text{msy}}$  of about 0.05 was appropriate for sandbar shark and was representative of the likely, present-day fishing mortality. If it is assumed that this was the fishing rate in the Chesapeake Bay and if it is further assumed that this fishing mortality rate can be used to back calculate survival, then approximately 95% of sandbar sharks survive fishing each year. Given average recent catches in the Chesapeake Bay, from the VIMS Fisheries Ecosystem Modeling and Assessment Project web site (<http://www.fisheries.vims.edu/multispecies/femap/femap.htm>), the catch from 1995 to 2000 was approximately 12 t per year. If this corresponds to a fishing mortality of 0.05, the shark biomass computes to approximately  $0.0240 \text{ t} \cdot \text{km}^{-2}$ .

#### **10.1.2.7.2 1950 P/B**

Cortes *et al.* (2002) estimated a natural mortality rate of  $0.18 \text{ year}^{-1}$  for sandbar sharks ( $>$  age 1). We thus assume a  $Z = P/B = 0.23 \text{ year}^{-1}$  for the model.

#### **10.1.2.7.3 1950 Q/B**

The consumption/biomass ratio for sandbar sharks of  $1.4 \text{ year}^{-1}$  was estimated with FishBase (Froese and Pauly, 2004) assuming  $T = 17^\circ$ ,  $W_\infty = 616,292 \text{ g}$ , an aspect ratio of 1.63, and a carnivorous diet.

#### **10.1.2.7.4 Diet compositions**

The diet composition of the sandbar shark was based upon Ellis (2003), who sampled sharks in four size classes:  $\leq 60 \text{ cm}$  precaudal length (PCL), 61-80 cm PCL, 81-100 cm PCL, and  $> 100 \text{ cm}$  PCL. Giving all size classes equal weight from summed diet data resulted in an average wet weight diet of: teleosts (47.9), crustaceans (27.075), elasmobranchs (22.1), cephalopods (1), unknown (1.275), and other (0.65). We used these values as guidelines for the diet (Table 22).

#### **10.1.2.8 Other elasmobranchs (FEM group 27)**

##### **10.1.2.8.1 1950 biomass**

This group includes skates and rays, e.g., the cownose ray (*Rhinoptera bonasus*) and other sharks, e.g., the spiny dogfish (*Squalus acanthias*), which are common in the Bay. The biomass was assumed to be similar to that of the benthic rays and skates group of the southeast United States continental shelf model as reported by Okey and Pugliese (2001). The biomass estimate for that earlier model was, in turn, derived from the Southeast Area Monitoring and Assessment Program- South Atlantic, ([http://www.gsmfc.org/sm\\_ov.html/](http://www.gsmfc.org/sm_ov.html/)). The biomass estimate was  $0.5 \text{ t} \cdot \text{km}^{-2}$ .

##### **10.1.2.8.2 1950 P/B**

The P/B estimate for the ‘other elasmobranchs’ group of  $0.15 \text{ year}^{-1}$  was based on values given for similar groups in other EwE models, e.g., skates in Beattie (2001) and benthic rays and skates in Okey and Pugliese (2001).

##### **10.1.2.8.3 1950 Q/B**

The Q/B value for the ‘other elasmobranchs’ was calculated by Ecopath to be  $0.938 \text{ year}^{-1}$  by estimating the P/Q ratio for the group as 0.16.

##### **10.1.2.8.4 Diet compositions**

Diet composition for the ‘other elasmobranchs’ was derived from the ChesMMA 2002 Bay-wide trawl survey ([www.fisheries.vims.edu/chesmmap/](http://www.fisheries.vims.edu/chesmmap/)), using samples from the main stem of the Chesapeake Bay. Data for cownose rays, clearnose skate, bluntnose ray, bullnose ray, spiny butterfly ray, southern stingray, smooth dogfish, and spiny dogfish were used. Anything  $< 1.0\%$  of the diet for any of the noted species was ignored for determination of the group diet.

### **10.1.3 Birds and other vertebrates**

The EwE model includes two groups of seabirds—piscivorous and non-piscivorous. In addition, the odd marine mammal occurs in the Chesapeake Bay, as do turtles. However, these groups were excluded from the 1950-2002 version of the Chesapeake Ecosystem Model due to their perceived minimal trophic

and economic impacts. Should ecological considerations warrant explicit inclusion of such groups, this can be easily achieved.

#### **10.1.3.1 Piscivorous seabirds (FEM group 28)**

The birds included in biomass for this group were based on advice from local experts, and listed by D. Forsell (personal communication, workshop October 2001) (Table 27).

##### **10.1.3.1.1 1950 Biomass**

The biomass estimate for piscivorous seabirds of  $0.3 \text{ t} \cdot \text{km}^{-2}$  was based on advice provided in a Chesapeake Ecopath Workshop (Sellner *et al.*, 2001).

##### **10.1.3.1.2 1950 P/B**

A total mortality estimate for piscivorous seabirds of  $0.163 \text{ year}^{-1}$  was based on survival rate values of 85-90% for cormorants and 80-93% for alcids in the northeast Atlantic (ICES, 2000).

##### **10.1.3.1.3 1950 Q/B**

The consumption ratio estimate of  $120 \text{ year}^{-1}$  was from data for the piscivorous seabirds group in Preikshot (2007).

##### **10.1.3.1.4 Diet compositions**

The diet composition for piscivorous seabirds was based upon advice from D. Forsell (Personal communication), modified to include a substantial predation on juvenile menhaden (E. Houde, personal communication).

#### **10.1.3.2 Non-piscivorous seabirds (FEM group 29)**

##### **10.1.3.2.1 1950 Biomass**

Bird species included in this group are presented in Table 26. The biomass estimate for this group of  $0.121 \text{ t} \cdot \text{km}^{-2}$  was based on advice from local experts in a Chesapeake Ecopath Workshop (Sellner *et al.*, 2001).

##### **10.1.3.2.2 1950 P/B**

The total mortality estimate for non-piscivorous seabirds of  $0.511 \text{ year}^{-1}$  was based on an annual mortality rate of 37% for mallard males and 44% females (Anderson, 1975).

##### **10.1.3.2.3 1950 Q/B**

The consumption ratio for non-piscivorous seabirds of  $120 \text{ year}^{-1}$  was based on the estimated Q/B for the group in Preikshot (Preikshot, 2007)

##### **10.1.3.2.4 Diet compositions**

The diet composition for non-piscivorous seabirds was based upon advice from D. Forsell (pers. comm.)

## 10.1.4 Commercial invertebrates

### 10.1.4.1 Blue crab: YOY and adult (*Callinectes sapidus*) (FEM groups 30 and 31)

#### 10.1.4.1.1 1950 biomass

The blue crab is represented by two stanzas, YOY (0-11 months) and adults (12+ months), with adults as the leading stanza. A recent VPA (A. Sharov, pers. comm.) suggests a current biomass of  $1.580 \text{ t} \cdot \text{km}^{-2}$  for juveniles. Juvenile biomass was estimated by Ecopath, assuming that  $K = 0.59 \text{ year}^{-1}$  (1998) and  $W_m/W_{inf} = 0.4$ . For 1950, we used a higher biomass of  $4.0 \text{ t} \cdot \text{km}^{-2}$  for the adult blue crab.

#### 10.1.4.1.2 1950 P/B

Total mortality for 1950 was assumed at 1.0 for adults and  $5.0 \text{ year}^{-1}$  for juveniles.

#### 10.1.4.1.3 1950 Q/B

The consumption ratio was assumed to be  $4.0 \text{ year}^{-1}$  for the adult group and was estimated to be  $12.057 \text{ year}^{-1}$  from multi-stanza calculations for the juveniles.

#### 10.1.4.1.4 Diet compositions

Blue crab diet compositions were provided by R. Lipcius (pers. comm.)

### 10.1.4.2 Oyster (*Crassostrea virginica*) (FEM groups 32 and 33)

Eastern oysters are separated into young-of-year (YOY) and age 1+ stanzas in the FEM.

#### 10.1.4.2.1 1950 Biomass

An oyster biomass of  $20.4 \text{ t} \cdot \text{km}^{-2}$  for age 1+ was used as a leading biomass for 1950 based on a stock reduction analysis. YOY oysters were estimated to have a biomass of  $3.28 \text{ t} \cdot \text{km}^{-2}$ .

#### 10.1.4.2.2 1950 P/B

A total mortality rate of  $1.5 \text{ year}^{-1}$  was available from the model of Dew *et al.* (2003), where the potential population dynamics of an introduced oyster species (*Crassostrea ariakensis*) in the Chesapeake Bay were described. Mortality rates of  $6.0 \text{ year}^{-1}$  and  $0.15 \text{ year}^{-1}$  were used for YOY and age 1+, respectively.

#### 10.1.4.3 1950 Q/B

A Q/B of  $2.0 \text{ year}^{-1}$  was assumed as a leading parameter for age 1+ oysters, while the Q/B for juveniles were estimated to be  $8.965 \text{ year}^{-1}$  based on the stanza calculations.

### 10.1.4.4 Soft clam (*Mya arenaria*) (FEM group 34)

#### 10.1.4.4.1 1950 biomass

Present-day soft clam biomass was estimated from data in Homer *et al.* (2004). The assumption was made that the population is limited to mesohaline waters. The derivation was based on the average of densities from survey sites, assumed body mass of 20 g, and that soft clam inhabits 10% of Bay waters. The resulting biomass was  $1.66 \text{ t} \cdot \text{km}^{-2} \cdot \text{year}^{-1}$ . It was assumed that the biomass of soft clams in the

Chesapeake Bay has decreased considerably over the last 50 years, but time-series data are not available. We estimated the 1950 biomass of soft clam from an assumed ecotrophic efficiency of 0.95 and obtained an estimate of  $6.923 \text{ t} \cdot \text{km}^{-2}$ .

#### **10.1.4.4.2 1950 P/B**

(Abraham and Dillon, 1986) estimate a soft clam survival rate for individuals  $> 30\text{mm}$  of 90%, *i.e.*,  $M = 0.105 \text{ year}^{-1}$ , whereas in exploited areas the exploitation rate alone has been estimated at 50-60% (2003), *i.e.*,  $F = 0.69\text{--}0.91 \text{ year}^{-1}$ . Assuming for 1950 a low exploitation rate, a  $Z$  value of  $0.45 \text{ year}^{-1}$  was used.

#### **10.1.4.4.3 1950 Q/B**

The consumption ratio for soft clam was estimated to be  $2.25 \text{ year}^{-1}$  by assuming a P/Q ratio of 0.20.

#### **10.1.4.5 Hard clam (*Mercenaria mercenaria*) (FEM group 35)**

##### **10.1.4.5.1 1950 biomass**

A present-day hard clam biomass was extrapolated from Mann *et al.* (2003), based on abundance and area surveyed divided by total Bay area, assuming that a one- to two-inch clam weighs 18-20 g and that 25% of Bay waters were exploited. The current biomass was thus estimated as  $2.24 \text{ t} \cdot \text{km}^{-2} \cdot \text{year}^{-1}$ . For 1950, a higher assumed biomass of  $2.626 \text{ t} \cdot \text{km}^{-2}$  was used to reflect that hard clams have decreased in the Bay over the last decades.

##### **10.1.4.5.2 1950 P/B**

A total production/biomass ratio of  $1.02 \text{ year}^{-1}$  was estimated from an empirical equation of Thomas Brey, AWI, included in the Ecopath software (see Christensen *et al.* (2000)] for a description of the algorithm), assuming an average mass of 20 g, water  $T = 17 \text{ }^\circ\text{C}$ , non-motile behavior, and an average water depth of 6.5 m.

##### **10.1.4.5.3 1950 Q/B**

The consumption ratio was estimated to be  $5.1 \text{ year}^{-1}$  assuming a  $P/Q = 0.20$ , the same as for soft clam.

#### **10.1.5 Other invertebrates**

##### **10.1.5.1 Ctenophores (*Mnemiopsis spp.*) (FEM group 36)**

###### **10.1.5.1.1 1950 biomass**

Present-day biomass of  $3.4 \text{ t} \cdot \text{km}^{-2}$  was estimated from data obtained from the VIMS ChesMMAAP survey (Sellner *et al.*, 2001). This estimate was used 1950 in lack of earlier abundance estimates.

###### **10.1.5.1.2 1950 P/B and Q/B**

Shushkina *et al.* (1989) observed that ctenophores in their study had growth rates 1.5 to 2 times greater than jellies. Therefore, the P/B and Q/B values for ctenophores were the values for sea nettles multiplied by 1.75: P/B was  $8.800 \text{ year}^{-1}$  and Q/B was  $35.2 \text{ year}^{-1}$ .

### **10.1.5.2 Sea nettles (*Chrysaora quinquecirrha*) (FEM group 37)**

#### **10.1.5.2.1 1950 biomass**

Present-day biomass for sea nettles of  $0.583 \text{ t} \cdot \text{km}^{-2}$  was based upon an average of the four seasonal models in Baird and Ulanowicz (1989) multiplied by a conversion factor of carbon to wet weight of 0.3% for jellies (Shushkina *et al.*, 2000). This biomass was used for 1950.

#### **10.1.5.2.2 1950 P/B**

(1999) estimated a daily growth rate for *Aurelia aurita* of 0.053 at 5 °C to 0.15 at 16.5 °C. The average conservative estimate was the basis for P/B in the Chesapeake Bay, *i.e.*,  $0.053 \cdot 365 = 19.3 \text{ year}^{-1}$ . Because they are only in the Bay for three to four months each year, an apparent P/B  $\approx 5.0 \text{ year}^{-1}$  was used in the model to lower the available production.

#### **10.1.5.2.3 1950 Q/B**

Matishov and Denisov (1999) found a diurnal consumption rate of 7% of biomass for the medusa in the Black Sea. This would translate to an annual consumption per unit biomass of  $365 \cdot 0.07 = 25.55 \text{ year}^{-1}$ . Thus, a three- to four-month residency period in the Bay implies a Q/B  $\approx 8 \text{ year}^{-1}$ . As this value implies a rather high P/Q ratio, we instead used an estimated Q/B of  $20 \text{ year}^{-1}$  based on an assumed P/Q of 0.25.

### **10.1.5.3 Microzooplankton (FEM group 38)**

#### **10.1.5.3.1 1950 biomass**

This group chiefly consisted of rotifers, copepod nauplii, and ciliates. A present-day estimate of  $0.13 \text{ t} \cdot \text{km}^{-2}$  was available for the Maryland portion of the Chesapeake (www.chesapeakebay.net; C. Buchanan, pers. comm.). Biomass was estimated based on an assumed EE of 0.95, which leads to an estimated value for 1950 of  $6.239 \text{ t} \cdot \text{km}^{-2}$ , much higher than that estimated for Maryland.

#### **10.1.5.3.2 1950 P/B**

Total mortality rate for microzooplankton was estimated to be  $140 \text{ year}^{-1}$  by local experts at a Chesapeake Bay Ecopath workshops (Park and Marshall, 2000). The value was used for as the 1950 estimate.

#### **10.1.5.3.3 1950 Q/B**

The consumption ratio for microzooplankton was estimated to be  $350 \text{ year}^{-1}$  by assigning a P/Q ratio of 0.4 to the group.

### **10.1.5.4 Mesozooplankton (FEM group 39)**

#### **10.1.5.4.1 1950 biomass**

This group was largely made up of copepods, which have been noted to reach densities greater than 1000 nauplii per liter at estuarine turbidity maximum zones (Sellner *et al.*, 2001). A present-day biomass for the Chesapeake Bay of  $10.3 \text{ t} \cdot \text{km}^{-2}$  was estimated from Maryland data provided by C. Buchanan (pers. comm.) from the Chesapeake Bay Program data set. This estimate was used as the 1950 estimate.



#### **10.1.5.4.2 1950 P/B**

A total mortality rate of 25 year<sup>-1</sup> for mesozooplankton was estimated by local experts at the Chesapeake Bay Ecopath Workshop (1989).

#### **10.1.5.4.3 1950 Q/B**

The consumption ratio for mesozooplankton of 83.333 year<sup>-1</sup> was estimated by assigning a P/Q ratio of 0.3 to the group.

#### **10.1.5.5 Other suspension feeders (FEM group 40)**

##### **10.1.5.5.1 1950 biomass**

Biomass was based on the ‘other suspension feeders’ group in Baird and Ulanowicz (2000), converted to dry weight, then wet weight using ratios listed for annelids and zooplankton in Jørgensen *et al.* (2000). The 1950 estimate was 6.0 t · km<sup>-2</sup>.

##### **10.1.5.5.2 1950 P/B**

A P/B for ‘other suspension feeders’ of 2.0 year<sup>-1</sup> was taken from a value for annelids in Jørgensen *et al.* (2000).

##### **10.1.5.5.3 1950 Q/B**

The consumption ratio for ‘other suspension feeders’ was estimated to be 8.0 year<sup>-1</sup> by assigning a P/Q ratio of 0.25 to the group.

#### **10.1.5.6 Other infauna/epifauna (FEM group 41)**

##### **10.1.5.6.1 1950 biomass**

The biomass for ‘other infauna/epifauna’ was estimated by Ecopath to be 66.675 t · km<sup>-2</sup>, assuming a group ecotrophic efficiency of 0.9.

##### **10.1.5.6.2 1950 P/B**

The P/B for ‘other infauna/epifauna’ of 1.0 year<sup>-1</sup> was taken from the value for annelids given in Jørgensen *et al.* (2000)

##### **10.1.5.6.3 1950 Q/B**

Q/B for ‘other infauna/epifauna’ was estimated to be 5.0 year<sup>-1</sup> by assigning a P/Q ratio of 0.2.

#### **10.1.5.7 Primary producers**

##### **10.1.5.7.1 Phytoplankton (FEM group 44)**

##### **10.1.5.7.2 1950 biomass**

Phytoplankton biomass for 1950 was assumed at 27 t · km<sup>-2</sup>.

##### **10.1.5.7.3 1950 P/B**

Phytoplankton P/B was assumed to be 160 year<sup>-1</sup>.

### **10.1.5.8 Benthic algae (FEM group 42)**

#### **10.1.5.8.1 1950 biomass**

Biomass of benthic algae was estimated to be  $1.717 \text{ t} \cdot \text{km}^{-2}$  based on an assumed EE of 0.9.

#### **10.1.5.8.2 1950 P/B**

P/B for benthic algae was assumed at  $80 \text{ year}^{-1}$ .

### **10.1.5.9 Submerged aquatic vegetation (SAV) (FEM group 43)**

#### **10.1.5.9.1 1950 biomass**

Four main groups dominate the macrophyte communities within the Chesapeake Bay; one group, eelgrass (*Zostera marina*), dominates these. Biomass ranges from a high in late summer of more than 20,000 t to a low of 5,000 t in December (Sellner *et al.*, 2001). Total SAV biomass, averaged annually for 1996, was 22,300 t in approximately 25,000 ha, or 2.1% of the total Chesapeake Bay area. Biomass is entered as  $419 \text{ t} \cdot \text{km}^{-2}$ , in a proportion of the total area corresponding to 0.021, and is also used for the 1950 model due to the lack of time trend information (Sellner *et al.*, 2001).

#### **10.1.5.9.2 1950 P/B**

Mortality for *Z. marina* was estimated in a similar system in Japan (Oshima *et al.*, 1999) as  $Z = P/B = 5.11 \text{ year}^{-1}$ .

## **10.1.6 Model pedigree**

Pedigree was defined for all input parameters as described in the *EwE User's Guide* (Christensen *et al.*, 2004). The pedigree indices were used to obtain confidence interval estimates for the input parameters (Table 31). The pedigree index was estimated as 0.45.

## **10.1.7 Prices**

Prices for the major species exploited in the Chesapeake Bay were obtained from the Sea Around Us global price database ([www.seaaroundus.org](http://www.seaaroundus.org)). The price database includes year-specific as well as consumer price index-corrected values for the various commodities. Prices for 2000 were used; the intention is to use prices for evaluating future policy options (Table 32).

For the U.S. recreational fisheries overall, Sumaila (pers. comm., UBC Fisheries Centre) estimated that recreational catches were worth approximately 32 times as much per unit weight of the catch (in US dollars). In this model, a much more conservative factor of two was used for the value of recreational/commercial fisheries.

## **10.2 Appendix B: Time Series**

### **10.2.1 Commercial fishes**

#### **10.2.1.1 Striped bass**

##### **10.2.1.1.1 1950-2002 catches**

For the most recent striped bass catch time series, Marine Recreational Fisheries Statistics Survey (MRFSS) estimates of recreational catch of striped bass for Maryland and Virginia combined from 1981 to 2002 were used. For the period from 1950 to 1980, the multispecies production method was used to estimate a recreational to commercial ratio to determine the recreational catch. For commercial catch of striped bass, NMFS estimates for the Chesapeake Bay from 1950 to 2002 were added to a discards estimate, determined as 33% of the commercial catches (NFSC, 2003). The catch distribution between resident (12-83 months) and migrant (84+ months) striped bass was estimated from the coastwide catches reported by ASMFC (2003b) Tables 1b, 12, and 13, estimating total weight in catch from the numbers in the catch, 1982-2002, and average weight. This resulted in an average distribution by weight in the catch of 49% for resident and 51% for migrant striped bass (standard deviation 13%).

An earlier version of the striped bass catch time series is also included in the model. This version uses a recreational:commercial catch ratio for the years 1982-2002 based on Table 1 of the 2003 ASMFC Striped Bass Advisory Report (ASMFC, 2003a). This ratio is then applied to the NMFS commercial data for the Chesapeake Bay to estimate the recreational catch for these years. Prior to 1982, the median of the recreational to commercial catch ratio for the years 1982-2002 was used to estimate the recreational catch. The resident and migrant portions of the catch were determined for 1982-2002 based on an average percent resident striped bass calculated from values in Tables 10 and 11 of the 2003 ASMFC Striped Bass Advisory Report (ASMFC, 2003a).

##### **10.2.1.1.2 Other 1950-2002 time series**

A number of time series were available for striped bass in the Chesapeake Bay. There are two striped bass young-of-the-year biomass estimates included in the model. The most recent version was calculated using values from Tables 13 and 18a of the 2005 Stock Assessment Report for Atlantic Striped Bass (ASMFC, 2005b). The earlier version was calculated with data from Tables 11 and 16 of the 2003 Stock Assessment Report for Atlantic Striped Bass (ASMFC, 2003a). Resident and migratory biomass are represented by data from the ASMFC striped bass stock assessment for 2003, the ASMFC striped bass stock assessment for 2005 (ASMFC, 2005b), and multispecies production method estimates. Biomass data for striped bass are presented in Table 4. Growth parameters are given in Table 3.

The combined fishing mortality series for both stocks were estimated for 1982-2002 with data from the ASMFC 2005 stock assessment (ASMFC, 2005b). The data from the ASMFC are estimated from a virtual population analysis model (VPA), *i.e.*, it recreates the population history by receding back in time and adding catches back into the population, based on an assumed natural mortality. Values of striped bass resident combined F for 1959–1984 are from Gibson (1993) Table 3, Run 2, and migratory combined F for 1959-1981 from Gibson (1993) Appendix Table 3, Run 3. The fishing mortalities for the period before 1959 are values estimated by Ecopath. A second migratory fishing mortality for 1988-2002 is included for the Maryland spring spawning stock in the Chesapeake Bay. This estimate is

provided by Table 24 of the 2005 Stock Assessment Report for Atlantic Striped Bass (ASMFC, 2005b). Fishing mortality data for striped bass are presented in Table 5.

#### ***10.2.1.1.3 Calculation of abundance indices***

Most of the recreational species indexed did not have fishing seasons. Striped bass was the exception, and the blanket application of the technique to make annual estimates probably led to some bias, because closed seasons were imposed until 1996 and the 18-inch minimum did not apply to resident stock for the entire season (a 28-inch minimum would apply early) until about 1999. Generally, it was only possible to model a stable or very slightly declining striped bass biomass in the Bay with the annual MRFSS index. A very different trend for resident striped bass was found if the September-October wave of the MRFSS data was used. This is the period when fishing seasons were always opened during the cautious period following the moratorium; an 18-inch size limit was consistently applied; and migratory fish had left the Bay. On a kg harvested per trip basis there has been a drop in relative abundance during this wave. If kg harvest per open day is used as an estimate of relative abundance, a dramatic, steady decline is displayed. Over the same time period, migratory biomass has increased in the Bay (May-June wave data). These would be much larger fish than residents. A general problem with hindcasting striped bass landings for the Bay has been the unknown relationship of recreational landings to commercial landings prior to the MRFSS survey. Eight estimates of Maryland recreational landings were compiled from creel surveys conducted between 1962 and 1984. These estimates were used in regression analysis with commercial landings to estimate the trend in recreational harvest back to 1950. The regression indicated that recreational landings became a decreasing fraction of total landings between 1962 and 1984 ( $r^2 = 0.77$ ,  $P < 0.004$ ). This relationship could be used to develop the long-term striped bass model described briefly below.

A striped bass kg harvest per open day estimate of relative abundance was developed along with a biomass dynamic model incorporating an additional loss function (a function of lesions presence-absence which was used as an indicator of health and subsequent mortality). It was tuned to bay estimates of  $F$  and  $M$  from tagging. Landings consisted of MRFSS and commercial reports for 1950-2004, recreational landings (Maryland and Virginia) predicted from the analysis described previously (1962-1980), and recreational landings back to 1950 predicted by using the 1962 ratio as a constant. Lesions were invoked as an additional increasing loss term from 1994 to 2004 and held constant at a very low level for all years previous. The analysis was conducted in three stages. An initial estimate of production parameters ( $r$  and  $K$ ) and biomass was made for 1991-2004. These estimates of biomass were then substituted as “observed” values and a model was run for the MD moratorium through present (1985-2004). A final run used these biomass values as “observations” and the predicted landings back to 1950 to derive a history of resident biomass by running the biomass dynamic model in reverse. Bear in mind that regulations prior to the moratorium were minimal (12- to 14-inch size limit, creel limits were not in place until the 1980s, and commercial harvests were restricted minimally) and were restrictive afterwards (18-inch size limit, two fish creel, strict seasons, mesh restrictions). It was assumed that production parameters would be the same regardless of size changes, but exploitable biomass prior to 1985 would be 12-inch fish and larger, while post-moratorium exploitable biomass would be 18-inch and larger striped bass.

### **10.2.1.2 Bluefish: YOY and adult (*Pomatomus saltatrix*) (FEM groups 4 and 5)**

#### **10.2.1.2.1 1950–2002 catches**

Total catch of bluefish for the Chesapeake Bay was determined by adding the recreational catch for Maryland and Virginia as reported by MRFSS to commercial catch of bluefish in the tidal waters of the Chesapeake Bay, as reported by NMFS for the years 1982-2002. The recreational to commercial ratio for years prior to 1982 was estimated using the median value for years 1982-2002 from Table 1 of the bluefish assessment in the 41<sup>st</sup> SAW Assessment Report (NFSC, 2005), 2.52. The recreational to commercial ratio was then applied to the commercial catch values for the Chesapeake reported by the National Marine Fisheries Service (Fisheries Statistics and Economics Division, Silver Spring, Maryland) for 1950 through 2002 to determine recreational portions of the Chesapeake catch. These two values were added together to determine total catch for the Chesapeake in the time period. It was assumed that there were no commercial discards, based on the 41<sup>st</sup> SAW Assessment Report (NFSC, 2005).

A second estimate of bluefish catch was assessed using stock reduction analysis.

#### **10.2.1.2.2 Other 1950–2002 time series**

Bluefish time series data are presented in Table 7. A relative abundance index for bluefish in the Chesapeake Bay region specifically was unobtainable. Consequently, it was assumed that coast-wide abundance indices for bluefish are representative. Both Ecosim and single-species assessment models were fitted to the same information. Two different sources of time-series information were available: Bluefish YOY and adult biomass were estimated using data from Tables 10 and 20 of the NFSC 41<sup>st</sup> SAW (NFSC, 2005) and YOY and adult bluefish biomass was estimated using stock reduction analysis.

Coast-wide trawl survey information (in numbers landed per tow and kg landed per tow) were taken from Lee (2003a) for the period 1972-2002. Total fishing mortality rates were estimated using a single-species assessment model, where the combined recreational and commercial catch data was used to drive the assessment model. Furthermore, an index of 0- to 12-month bluefish biomass and adult biomass for the entire time period was constructed from the age-structured model. Growth, size selectivity, and maturity parameters used in the assessment model were taken from Salerno *et al.* (2001), and length-weight relationships from Wigley *et al.* (2003) (provided in Table 6). Prior to 1972, there is no survey information on relative abundance, and the uncertainty associated with the bluefish abundance during 1950 to 1972 is high. Reported landings prior to 1970 suggest that bluefish abundance was relatively low and as such, the initial biomass ratio to the unfished equilibrium was estimated to be very low.

### **10.2.1.3 Weakfish: YOY and adult (*Cynoscion regalis*) (FEM groups 6 and 7)**

#### **10.2.1.3.1 1950–2002 catches**

Information about commercial catches of weakfish in the Chesapeake region was obtained from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the years 1950-2002. A ratio between recreational and total catches for weakfish on the Atlantic coast was obtained from Tables 1 and 2 of the ASMFC 2003 fishery management plan review for weakfish (Spear *et al.*, 2003), and estimated to average 27% (median 28%) for the period 1982-2002. This ratio was used for all years prior to 1982, while the actual ratio was used for 1982-2002.

#### ***10.2.1.3.2 Other 1950–2002 time series***

The multispecies production method was used to provide a biomass estimate for weakfish. Adult fishing mortality and biomass estimates were also estimated using stock reduction analysis. Weakfish biomass and fishing mortality data are presented in Table 9. Growth parameters are given in Table 8.

#### ***10.2.1.4 Atlantic croaker (*Micropogonias undulatus*) (FEM group 8)***

##### ***10.2.1.4.1 1950–2002 catches***

Commercial catches of Atlantic croaker for the Chesapeake region were extracted from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the years 1950–2002. A ratio between recreational and commercial catches for Atlantic croaker was calculated from data in Table G1 of the 2003 Atlantic Croaker Stock Assessment report (ASMFC, 2005a), with an average value of 0.22, a standard deviation of 0.99, and a median value of 0.23 for the period 1973–2002. This recreational to commercial catch ratio was used to determine the total catch based on NMFS data for the period 1950 through 1980. For 1981–2002, MRFSS estimates of recreational catch of Atlantic croaker for Maryland and Virginia combined from 1981–2002 were used.

Catch data was also estimated using data from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the years 1950–2002 with a recreational:commercial ratio from the ASMFC 2004 Atlantic Croaker Stock Assessment (ASMFC, 2005a) for the years 1981–2002. For the time period 1950–1980, the median recreational:commercial ratio for 1981–2002 was used to estimate recreational catch.

The catch series was used to obtain fishing mortality rates over time using the Ecosim stock reduction analysis.

##### ***10.2.1.4.2 Other 1950–2002 time series***

Time-series abundance data for Atlantic croaker were available from the VIMS Trawl Surveys ([www.fisheries.vims.edu/rawlseine](http://www.fisheries.vims.edu/rawlseine)), and are presented in Table 10. Atlantic croaker biomass and fishing mortality estimates for the mid-Atlantic region for the years 1973 through 2002 were available from the ASMFC 2003 Atlantic Croaker Stock Assessment (ASMFC, 2005a).

#### ***10.2.1.5 Black drum (*Pogonias cromis*) (FEM group 9)***

##### ***10.2.1.5.1 1950–2002 Catches***

To obtain time series for the black drum, commercial catches were extracted from the NMFS statistics for the Chesapeake region and recreational catches since 1981 for Maryland and Virginia state waters were extracted from the Marine Recreational Fishery Statistics Survey (MRFSS). A median recreational:commercial ratio value for the years 1981–2002 of 0.75 was used for the years 1950–1980.

The catches were used to calculate F estimates using the stock reduction analysis in Ecosim.

##### ***10.2.1.5.2 Other 1950–2002 time series***

No time-series data were available for black drum (apart from the catches discussed above).

### **10.2.1.6 Summer flounder (*Paralichthys dentatus*) (FEM group 10)**

#### **10.2.1.6.1 1950–2002 catches**

Commercial catches for ‘flatfish’ in the Chesapeake region were extracted from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the years 1950-1957. The flatfish category was assumed to be dominated by summer flounder, the main commercial flatfish. Recreational catches since 1981 for Maryland and Virginia state waters were extracted from the Marine Recreational Fishery Statistics Survey (MRFSS). Prior to 1981, a recreational:commercial ratio was used to estimate the recreational portion of the catch. This ratio is the median value of recreational:commercial ratios determined for 1998-2004 based on data from the 41<sup>st</sup> SAW (NFSC, 2005).

#### **10.2.1.6.2 Other 1950–2002 time series**

The summer flounder stock assessed by NFSC (2002a) was considered to extend from Cape Hatteras to New England. The summer flounder in the Chesapeake Bay represent a subset of that stock and not a separate population. Therefore, the stock assessment for the larger population was used for indicating the changes in the summer flounder group in the model.

The first biomass series (1982 and after) used in the model were taken from the results of a VPA (Figures A10 and A11), which used ADAPT as the calibration method (NFSC, 2002a). An additional biomass series was added, including data from the 41<sup>st</sup> Northeast Regional Stock Assessment Workshop Assessment Report (NFSC, 2005).

Fishing mortality data from the 35<sup>th</sup> Northeast Regional Stock Assessment Workshop Assessment Report (NFSC, 2002b) was used for summer flounder for the period from 1982 through 1997 and from the 41<sup>st</sup> Northeast Regional Stock Assessment Workshop Assessment Report (NFSC, 2005) for the period from 1998 through 2002. Summer flounder biomass data are presented in Table 11.

### **10.2.1.7 Atlantic menhaden (*Brevoortia tyrannus*): YOY and adult (FEM groups 11 and 12)**

#### **10.2.1.7.1 1950–2002 catches**

There are two main menhaden fisheries, a reduction fishery (major component) and a minor bait fishery (pound net gear type). The proportion of the Atlantic catches taken in the Bay has increased from 20% before the mid 1960s, to 50% in the 1970s, to more than 80% since 1980, based on the NMFS catch data. There are two catch series based on the NMFS catch database: 1) assume a constant proportion (0.4) of the menhaden catches being taken in the Chesapeake Bay, and 2) use the Chesapeake Bay catches as reported. There is a catch series calculated using the values from Table 2.2 of the Analyses of the Status of the Atlantic Menhaden Stock (Vaughan *et al.*, 2002). Specifically, the total weight of the catch on the Atlantic coast for a given year (in 1000 mt) is multiplied by the sum product of the weight at age in the fishery of age 1-8 menhaden in that year. There are also two catch series for the Chesapeake split into stanzas, one for the 0-1 age group and one for the 2+ age group, based on the ASMFC 2003 Menhaden Stock Assessment (ASMFC, 2004) for the Atlantic Coast, NMFS commercial data for the Chesapeake, and estimates of the proportion of the catch taken in the Chesapeake.

#### **10.2.1.7.2 Other 1950–2002 time series**

Relative biomass and F series were extracted from figures in the ASMFC stock assessment report (ASMFC, 2004). The estimates are representative for the total Atlantic menhaden population, not just the fraction occurring in the Chesapeake Bay, as this fraction is difficult to estimate. This is a minor problem, as more than 80% of the catches have been taken in the Bay in recent decades. Since several of the major predators on menhaden also move in and out of the Bay, it may well be best to include the total populations.

Relative biomass and fishing mortality for menhaden was estimated in several time series. From the ASMFC 2004 stock assessment, the Potomac River Fisheries Commission menhaden pound net index was used as an estimate of age 2+ abundance, and the ASMFC juvenile coast-wide index as an estimate of age 0-1 abundance. An estimate of both the age 0-1 and 2+ relative biomass and fishing mortality was made based on estimates from the ASMFC 2003 stock assessment (ASMFC, 2004). Another fishing mortality time series was used from Figure 7.8 of the ASMFC 2003 menhaden stock assessment (ASMFC, 2004). The multispecies production method was used to estimate Chesapeake Bay and Atlantic biomass as well as fishing mortality values for Atlantic menhaden. Menhaden biomass and fishing mortality data are presented in Table 12.

#### **10.2.1.8 Alewife/herring (*Alosa pseudoharengus/Clupea harengus*) (FEM group 13)**

##### **10.2.1.8.1 1950–2002 catches**

Commercial catches of alewife for the Chesapeake region were extracted from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov/st1](http://www.st.nmfs.gov/st1)) for the years 1950-2002. It was assumed that there were no recreational fisheries for these species, because neither species is included in the NOAA recreational fisheries database available through the same link as the commercial catches. The catch series was used to estimate F values based on the stock reduction analysis of Ecosim.

##### **10.2.1.8.2 Other 1950–2002 time series**

We obtained a relative measure of blueback herring abundance from fish lifts at Conowingo Dam (St. Pierre, pers. comm.). American shad and blueback herring biomass estimates are presented in Table 13. The effectiveness of the fish lifts is less in wet years, (e.g., 2000, 2002-2004) than in years with drier spring months, (e.g., 1997-1999, 2001).

#### **10.2.1.9 American eel (*Anguilla rostrata*) (FEM group 14)**

##### **10.2.1.9.1 1950–2002 catches**

Commercial catches of American eel for the Chesapeake region were extracted from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the years 1950-2002. The recreational catch of American eel is very limited (Munger *et al.*, 2002); the species is not included in the NOAA recreational catch database, and it is therefore ignored here. It may be noted that from 1987 to 1996, Maryland, Virginia, and the Potomac River accounted for approximately 60% of the American eel catch in the United States (ASMFC, 2000).

Catches were used to estimate F values using the stock reduction analysis of Ecosim.



**10.2.1.10 Catfishes (*Ameiurus catus*, *A. nebulous*, *A. natalis*, *Ictalurus punctatus*, *I. furcatus*, *Pylodictis olivaris*) (FEM group 15)**

**10.2.1.10.1 1950–2002 catches**

Commercial catch for the combined catfish group was based on catfishes and bullheads in the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the Chesapeake region for the years 1950-2002. Since catfish are limited to freshwater and estuarine areas, the Chesapeake region as defined in this database should be representative for the Chesapeake Bay. Information about recreational catches in the Bay was not available, and the assumption was made that the catches were miniscule and could be ignored. Hence, the commercial catches were assumed to be representative for the group (Tables 1 and 2).

Fishing mortalities were estimated from the catches using the stock reduction analysis in Ecosim.

**10.2.1.11 White perch: YOY and adult (*Morone americana*) (FEM groups 16 and 17)**

**10.2.1.11.1 1950–2002 catches**

The catch series for white perch was estimated using the multispecies production method and was based on commercial and recreational harvest data for Maryland, Virginia, and the Potomac River.

**10.2.1.11.2 Other 1950–2002 time series**

Biomass series for white perch age groups 0 and 1+ were available for ‘upper rivers’ from the Virginia Institute of Marine Science juvenile trawl survey. These were used here for comparison with the Ecosim simulations. The multispecies production method was used to estimate catch, biomass, and fishing mortality. White perch biomass and fishing mortality estimates are presented in Table 14.

**10.2.1.12 Spot (*Leiostomus xanthurus*) (FEM group 18)**

**10.2.1.12.1 1950–2002 catches**

A time series of spot catches in the Chesapeake Bay was estimated from the commercial catches in the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)), and the ratio between recreational and commercial catches reported by Desfosse *et al.* (2001). A second catch series was estimated using the multispecies production method, with combined estimates of recreational and commercial catch from NMFS, MRFSS, and the VIMS trawl survey.

**10.2.1.12.2 Other 1950–2002 time series**

Estimates of spot relative biomass for the period 1950-2002 were obtained from a stock reduction analysis based on catches and tuned to the mean of the VIMS survey estimates for age groups 0 and 1+ (Table 15). The multispecies production method was used to estimate catch, fishing mortality, and biomass estimates for spot from 1981 through 2000. We were, however, not able to use the F-series as this would lead to a major overestimation of catches.

### **10.2.1.13 American shad (*Alosa sapidissima*) (FEM group 19)**

#### **10.2.1.13.1 1950–2002 catches**

Commercial catches of American shad for the Chesapeake region were extracted from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the years 1950–2002. The recreational catch was assumed to be minimal in comparison and therefore was ignored. The catches were used to obtain F values based on the stock reduction analysis in Ecosim.

#### **10.2.1.13.2 Other 1950–2002 time series**

An estimate of relative biomass was provided by the geometric mean of fish caught at the Conowingo Fish Lift on the Susquehanna River in the upper Chesapeake Bay (Bob Sadzinski, MDNR, pers. comm.). This estimate does not adequately represent trends Bay-wide; however, these are not available at this time.

## **10.2.2 Other fishes**

### **10.2.2.1 Bay anchovy (*Anchoa mitchilli*) (FEM group 20)**

#### **10.2.2.1.1 1950–2002 time series**

Two time series of relative abundance for bay anchovy were used for comparison with the Ecosim simulations, both from surveys. One is from the Maryland DNR juvenile seine survey going back to 1958, the other is from the VIMS trawl surveys from 1978. These time series are presented in Table 16.

#### **10.2.2.2 Other flatfishes (FEM group 21)**

No time-series data were included in the CBFEM for other flatfish.

### **10.2.2.3 Gizzard shad (*Dorosoma cepedianum*) (FEM group 22)**

#### **10.2.2.3.1 1950–2002 time series**

The multispecies production method was used to estimate fishing mortality and biomass for gizzard shad from 1958 through 2000. F-values were impressively low and we chose to ignore them as they were below detection range. Commercial catches of gizzard shad for the Chesapeake region were extracted from the NOAA Fisheries Statistics and Economics Division online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) for the years 1950–2002.

Gizzard shad provided an additional exception to the general assessment framework. Five count-based, fishery-independent indices were considered for gizzard shad: the Maryland gizzard shad juvenile index (1958–2004; E. Durell, Maryland Department of Natural Resources, pers. comm.), two estimates of catch per effort at the Conowingo fish lifts (east or west catch per lift hour; 1991–2000 or 1972–2000, respectively; R. Sadzinski, MD DNR, pers. comm.), and Head-of-Bay or Potomac River catch per gill net hour (1985–2005; P. Piavis, MD DNR, pers. comm.) from the spring striped bass spawning stock survey (Warner *et al.*, 2006). They likely represented different sizes on average. Various trial versions combining all or some of the indices were tried, but eventually a long-term model (1958–2004) was developed based on the Maryland juvenile index time-series (GSJI; geometric mean for Choptank, Nanticoke, and Potomac rivers and Head-of-Bay). This model was acceptable because it was the only

one that came close to supplying an estimate of intrinsic rate of increase (0.66) that made sense for a fish with a short life-span. Most model runs were suited to much more long-lived species. The GSJI was comprised of young-of-year through large adults; some extra variability was probably caused by the inclusion of YOY that would not have been present in the harvest. Exploratory analyses indicated that NO<sub>3</sub> loading estimates of Hagy *et al.* (2004) was significantly related to the GSJI (1958-2000,  $r^2 = 0.38$ ,  $P < 0.001$ ). This relationship should be biologically significant because gizzard shad are well suited to taking advantage of eutrophic conditions because of their ability to feed on detritus, algae, and zooplankton.

Hagy *et al.* (2004) provided estimates of NO<sub>3</sub> loading through 2000. Loading was estimated for 2001-2004 from the relationship of mean annual Susquehanna River flow at Conowingo Dam and NO<sub>3</sub> loading estimates during 1968-2000. NO<sub>3</sub> loading was first plotted versus river flow for 1945-2000. Residuals from the regression of these data ( $r^2 = 0.47$ ,  $P < 0.001$ ) increased steadily from positive to negative over the course of the time-series and intercepted the x-axis at about 1968. The NO<sub>3</sub> loading data were then plotted as 1945-1968 and 1969-2000 time series. These data strongly suggested two distinct relationships of flow and NO<sub>3</sub> loading (different slopes). The 1969-2000 time-series was used and the fit of the regression of flow and NO<sub>3</sub> loading improved ( $r^2 = 0.80$ ,  $P < 0.001$ ); however, a serious serial trend was evident in the residuals. Autoregression analysis (Proc Autoreg in SAS) was used with a one-year lag, and this removed the patterning in the residuals and improved fit ( $r^2 = 0.94$ ,  $P < 0.001$ ). This adjusted relationship was used to estimate NO<sub>3</sub> loading for 2001-2005 from annual mean flow.

Three types of biomass dynamic models were assessed. The first was the standard Schaefer logistic model, the second was a logistic model with a predator function (striped bass predation as a linear function of bass biomass), and the third used standardized NO<sub>3</sub> loading (standardized to the time-series median) as a multiplier for the intrinsic rate parameter. Rose (2004) used a similar approach to model changes in cod productivity. A run with both predation and changing production due to shifting NO<sub>3</sub> loading was made as well. Ultimately, the version with changing production and without predation was selected for gizzard shad. This model is described by equation (18) above.

A combined bay-wide (Maryland and Virginia) commercial harvest was used; discard estimates were not available. A genetic algorithm super solver (Evolver, Palisade Corporation) was used to estimate parameters. The previous year's estimated index was then used to predict the following year's estimate. The spreadsheet version of the model combined with the genetic algorithm provided a great deal of flexibility for trying different model variations. The time-series for this model was shortened to 1964-2004, partially because a menhaden model with this time-span, was used as the basis for this model and partially to minimize patterning in residuals that occurs in the earliest years of the time-series. This patterning could reflect shifts from varying to fixed stations and from a single sampling round to two rounds in the seine survey.

The GSJI model index was numeric, while landings were in weight. Landings were converted to numbers landed by converting the mean length of gizzard shad sampled from Potomac River pound nets during summer 1995 (Austin *et al.* 1996) to mean weight (0.26 kg) based on the length-weight relationship described in Fishbase. Once abundance was estimated, it was converted back to biomass by using the 1996 mean weight.

Maximum biomass of gizzard shad estimated by this method was not large and was about 4,000 mt. This is considerably less than that estimated for Atlantic menhaden and bay anchovy.

#### **10.2.2.4 Reef-associated fishes (FEM group 23)**

##### **10.2.2.4.1 1950–2002 catches**

Catches for these species are very limited. For example, for tautog, the total catch reported in the NOAA commercial catch data is 316 t for the years 1950-2002 combined. Consequently, we did not include catches in the model.

##### **10.2.2.5 Non-reef-associated fishes (FEM group 24)**

No time-series data were included in the CBFEM for non-reef-associated fish.

##### **10.2.2.6 Littoral forage fishes (FEM group 25)**

No time-series data were included in the CBFEM for littoral forage fish.

##### **10.2.2.7 Sandbar shark (*Carcharhinus plumbeus*) (FEM group 26)**

No time-series data were included in the CBFEM for sandbar shark.

##### **10.2.2.8 Other elasmobranchs (F.E.M. group 27)**

No time-series data were included in the CBFEM for other elasmobranchs.

### **10.2.3 Birds and other vertebrates**

#### **10.2.3.1 Piscivorous seabirds (FEM group 28)**

No time-series data were included in the CBFEM for piscivorous seabirds.

#### **10.2.3.2 Non-piscivorous seabirds (FEM group 29)**

No time-series data were included in the CBFEM for non-piscivorous seabirds.

### **10.2.4 Commercial invertebrates**

#### **10.2.4.1 Blue crab: YOY and adult (*Callinectes sapidus*) (FEM groups 30 and 31)**

##### **10.2.4.1.1 1950–2002 catch**

Catch data were included from the 2005 Stock Assessment of the Blue Crab in Chesapeake Bay (Miller *et al.*, 2005). The total Chesapeake Bay catch time series were from the assessment data presented in the Chesapeake Bay Stock Assessment Committee report (2003a).

##### **10.2.4.1.2 Other 1950–2002 time series**

Effort data for 1945-1994 were available from CBSAC assessments (Rugolo *et al.*, 1997).

The age 0 (YOY) and age 1+ (adult biomass) time series were from the assessment data presented in the Chesapeake Bay Stock Assessment Committee report (2003a). Note that the original abundance indices

range around an average value of zero and may, therefore, be negative in some years, but have been re-scaled for this study. Blue crab biomass, effort, and fishing mortality estimates are presented in Table 17.

#### **10.2.4.2 Eastern oyster (*Crassostrea virginica*) (FEM groups 32 and 33)**

##### **10.2.4.2.1 1950–2002 catch**

Oyster catches were extracted from the NOAA online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)) including all oyster catches for Maryland and Virginia. Recreational catch estimates were not available, and were omitted from the analysis. These estimates are provided in the stock reduction analysis. A catch estimate for oysters was also estimated using the multispecies production method.

##### **10.2.4.2.2 Other 1950–2002 time series**

Estimates of oyster catch per unit effort (CPUE) for the Maryland harvest were made available by the Maryland DNR Shellfish Division and included in the oyster stock reduction analysis (Table 18). Oyster abundance and fishing mortalities were estimated from a stock reduction analysis tuned to a CPUE series and are shown in the same table. The oyster fishing mortality combined time series includes data from the stock reduction analysis for the years 1950 through 1974 and data from the multispecies production method assessment for the years 1975 through 1999. A multispecies production method estimate was also made of catch and biomass for the years 1975 through 2000.

#### **10.2.4.3 Soft clam (*Mya arenaria*) (FEM group 34)**

##### **10.2.4.3.1 1950–2002 catch**

Commercial catch data were extracted from the NOAA Fisheries Commercial catch database ([www.st.nmfs.gov](http://www.st.nmfs.gov)), soft clam for the Chesapeake region, and used to force the simulations.

##### **10.2.4.3.2 Other 1950 - 2002 time series**

No actual time-series data were available for soft clam. There are indications – perhaps evidence – that flooding caused by hurricanes may severely affect soft and hard clams in the Chesapeake Bay. A forcing function was constructed to impact the P/B of the two groups based on the occurrence and severity of flooding caused by hurricanes in the Maryland/Virginia region. An overview of the hurricanes is presented (Table 28), while the time series with assumed relative P/B values for forcing the simulations is given in Table 29.

#### **10.2.4.4 Hard clam (*Mercenaria mercenaria*) (FEM group 35)**

##### **10.2.4.4.1 1950–2002 catch**

Commercial catches were extracted from the NOAA Fisheries Statistics online database ([www.st.nmfs.gov](http://www.st.nmfs.gov)), based on quahog, or hard clam, *Mercenaria mercenaria*, for the Chesapeake region, and used to force the simulations.

## **10.2.5 Other invertebrates**

### ***10.2.5.1 Ctenophores (Mnemiopsis spp.) (FEM group 36)***

No time-series data were included in the CBFEM for ctenophores.

### ***10.2.5.2 Sea nettles (Chrysaora quinquecirrha) (FEM group 37)***

No time-series data were included in the CBFEM for sea nettles.

### ***10.2.5.3 Microzooplankton (FEM group 38)***

#### ***10.2.5.3.1 1950–2002 time series***

Estimated of relative abundance of microzooplankton were based on the estimate from CRC workshop (Sellner, 2001) that the median rotifer and copepod nauplii biomass value is roughly 50% of the mesozooplankton biomass value over all station-dates available in the Chesapeake Bay Program database. Zooplankton biomass estimates are presented in Table 19.

### ***10.2.5.4 Mesozooplankton (FEM group 39)***

#### ***10.2.5.4.1 1950–2002 time series***

Estimates of relative abundance for mesozooplankton are based on data from the Chesapeake Bay Program database, weighed by station. Zooplankton biomass estimates are presented in Table 19.

### ***10.2.5.5 Other suspension feeders (FEM group 40)***

No time-series data were included in the CBFEM for other suspension feeders.

### ***10.2.5.6 Other infauna/epifauna (FEM group 41)***

No time-series data were included in the CBFEM for other infauna/epifauna.

## **10.2.6 Primary producers**

### ***10.2.6.1 Phytoplankton (FEM group 44)***

#### ***10.2.6.1.1 1950–2002 time series***

A relative chlorophyll abundance measure, 1950-1994, was estimated based on Harding and Perry (1997), and was used for comparison with the Ecosim simulations. Phytoplankton chlorophyll content is presented in Table 30.

### ***10.2.6.2 Benthic algae (FEM group 42)***

No time-series data were included in the CBFEM for benthic algae.

**10.2.6.3 Submerged aquatic vegetation (SAV) (FEM group 43)**

**10.2.6.3.1 1950-2002 time series**

SAV monitoring data is included from the VIMS SAV Monitoring Program by Chesapeake Bay Program segment from 1971-2002. Only those segments that were fully surveyed were included in the SAV index used in the CBFEM. For each year, SAV acreage for those segments that were fully surveyed was totaled and divided by the total surface area of the segments. This resulted in a single value for each year of the survey from 1971-2002, representing the ratio of SAV identified/SAV surveyed (Table 34). The year 1979 was not included in the index, as values for this year were based on only 2 segments and created outlier values.

**10.2.7 Nutrient loading**

An estimate of monthly primary production rate affected by nutrient loading was obtained based on a spatial hydrodynamic model, CBREEM (Ma *et al.*, in prep; Ma *et al.*, MS). The model runs are still of a preliminary nature, but are used in the absence of any other time-series information describing environmental loading factors through the modeled period.

**11 Tables related to Appendices A and B**

**11.1 Catches, species groups 1-12**

Table 1. Estimated catches Groups 1-12 ( $t \cdot km^2 \cdot year^{-1}$ ) for the Chesapeake Bay since 1950 used in the Chesapeake Bay Fisheries Ecosystem Model. Estimated from a variety of sources as described in the text.

Group	SB resident	SB resident	SB Migratory	SB migratory	Blue-fish	Blue-fish	Weak-fish	Atl. croaker	Atl. croaker	Black drum	Summer flounder	Menhaden Age 0-1	Menhaden Age 2+	Menhaden ASMFC	Menhaden CB
Group no.	2	2	3	3	5	5	7	8	8	9	10	11	12	12	12
1950	0.463	0.792	0.427	0.825	0.116	0.067	0.286	0.656	0.542	0.005	0.209		12.706	12.71	7.753
1951	0.329	0.549	0.303	0.572	0.073	0.042	0.137	0.434	0.358	0.006	0.212		13.954	13.95	5.78
1952	0.271	0.442	0.25	0.46	0.071	0.041	0.111	0.321	0.265	0.004	0.194		16.758	16.76	4.19
1953	0.247	0.393	0.228	0.409	0.062	0.035	0.142	0.323	0.267	0.005	0.273		22.844	22.84	7.359
1954	0.243	0.377	0.224	0.392	0.076	0.044	0.148	0.431	0.356	0.034	0.304		24.25	24.25	13.101
1955	0.275	0.416	0.254	0.433	0.079	0.045	0.264	0.818	0.676	0.01	0.255	3.836	21.82	24.71	14.305
1956	0.25	0.368	0.23	0.383	0.09	0.052	0.232	0.815	0.673	0.014	0.292	8.514	19.97	28.46	8.638
1957	0.221	0.317	0.204	0.33	0.079	0.046	0.147	1.114	0.92	0.011	0.26	6.951	17.161	23.75	12.15
1958	0.351	0.489	0.324	0.509	0.052	0.03	0.11	0.894	0.738	0.005	0.332	3.086	17.314	20.33	14.641

1959	0.512	0.693	0.472	0.721	0.059	0.034	0.049	0.607	0.501	0.02	0.436	14.648	11.716	26.35	18.802
1960	0.531	0.697	0.49	0.726	0.039	0.022	0.067	0.323	0.266	0.014	0.357	0.966	20.226	21.13	11.29
1961	0.576	0.734	0.532	0.764	0.087	0.05	0.091	0.224	0.185	0.02	0.26	3.14	19.896	23.04	13.548
1962	0.47	0.58	0.434	0.604	0.163	0.094	0.105	0.093	0.077	0.029	0.25	2.534	18.974	21.4	14.874
1963	0.516	0.616	0.476	0.641	0.187	0.108	0.074	0.009	0.007	0.028	0.216	3.435	10.441	13.7	11.749
1964	0.412	0.476	0.38	0.495	0.112	0.064	0.11	0.028	0.023	0.006	0.195	3.529	7.239	10.28	15.26
1965	0.41	0.457	0.378	0.476	0.059	0.034	0.14	0.109	0.09	0.009	0.258	4.488	6.448	10.42	16.327
1966	0.488	0.525	0.451	0.546	0.072	0.041	0.074	0.105	0.086	0.038	0.282	3.964	4.82	7.89	12.605
1967	0.462	0.479	0.427	0.499	0.038	0.022	0.043	0.023	0.019	0.017	0.222	3.701	4.039	7.72	10.127
1968	0.488	0.486	0.45	0.506	0.106	0.061	0.079	0	0	0.028	0.239	2.579	6.813	8.98	12.432
1969	0.616	0.589	0.568	0.613	0.077	0.044	0.065	0.005	0.004	0.009	0.163	2.53	3.934	6.04	8.24
1970	0.457	0.419	0.422	0.436	0.199	0.114	0.153	0.009	0.008	0.006	0.239	4.61	5.766	10.33	20.403
1971	0.315	0.276	0.29	0.287	0.209	0.12	0.17	0.019	0.016	0.008	0.191	1.77	8.242	9.85	18.149
1972	0.467	0.391	0.431	0.407	0.354	0.203	0.182	0.035	0.029	0.002	0.203	4.748	9.888	14.59	25.243
1973	0.624	0.498	0.576	0.518	0.884	0.508	0.35	0.184	0.082	0.001	0.354	2.544	11.332	13.79	22.935
1974	0.481	0.365	0.444	0.38	1.027	0.59	0.216	0.223	0.096	0.003	0.363	2.552	9.136	11.32	17.453
1975	0.336	0.241	0.31	0.251	0.99	0.569	0.309	0.544	0.316	0.003	0.411	2.169	7.839	9.77	14.601
1976	0.226	0.153	0.209	0.16	1.301	0.747	0.274	0.734	0.411	0.002	0.38	4.007	9.613	13.42	20.237
1977	0.21	0.134	0.194	0.14	1.027	0.59	0.283	0.894	0.548	0.001	0.502	2.302	11.342	13.34	23.118
1978	0.136	0.081	0.125	0.084	0.852	0.489	0.274	0.826	0.513	0.003	0.629	1.972	11.792	13.33	19.411
1979	0.112	0.063	0.104	0.065	0.941	0.54	0.428	0.333	0.132	0.003	1.115	2.957	12.071	13.74	20.909
1980	0.207	0.107	0.191	0.112	0.904	0.52	0.423	0.144	0.042	0	0.934	3.784	12.276	15.99	24.764
1981	0.162	0.065	0.149	0.068	0.623	0.26	0.175	0.033	0.022	0.007	0.275	2.45	12.802	14.35	18.661
1982	0.031	0.02	0.029	0.02	0.571	0.318	0.155	0.007	0.007	0.002	0.287	2.565	12.731	15.15	27.56
1983	0.034	0.021	0.032	0.022	0.824	0.379	0.226	0.023	0.008	0.063	0.684	2.348	14.396	15.74	29.544
1984	0.057	0.049	0.053	0.051	0.466	0.226	0.15	0.061	0.051	0.004	0.616	3.873	9.179	11.92	22.281
1985	0.016	0.009	0.015	0.009	0.776	0.529	0.144	0.153	0.101	0.017	0.309	3.415	8.853	11.74	29.252
1986	0.007	0.001	0.007	0.001	0.796	0.344	0.156	0.233	0.163	0.037	0.232	0.625	8.895	9.43	20.463
1987	0.011	0.003	0.011	0.003	0.805	0.311	0.147	0.21	0.164	0.026	0.388	1.515	11.565	13.03	28.183



1988	0.023	0.006	0.021	0.006	0.649	0.334	0.137	0.213	0.121	0.009	0.511	1.102	11.27	12.04	25.265
1989	0.028	0.01	0.026	0.01	0.348	0.306	0.093	0.109	0.061	0.012	0.21	3.779	9.101	12.67	28.047
1990	0.033	0.014	0.031	0.014	0.404	0.278	0.097	0.051	0.012	0.005	0.131	1.051	14.997	15.56	32.078
1991	0.033	0.022	0.03	0.022	0.471	0.222	0.078	0.093	0.013	0.011	0.256	5.342	9.914	13.78	27.43
1992	0.061	0.038	0.056	0.04	0.235	0.072	0.05	0.153	0.065	0.017	0.317	3.413	8.491	11.41	26.178
1993	0.085	0.052	0.079	0.054	0.24	0.052	0.067	0.42	0.329	0.004	0.216	2.007	10.817	12.67	29.14
1994	0.112	0.062	0.104	0.064	0.208	0.07	0.084	0.518	0.394	0.007	0.206	0.931	9.469	10.33	23.441
1995	0.188	0.104	0.174	0.108	0.19	0.049	0.089	0.575	0.44	0.011	0.212	2.212	11.384	13.54	31.954
1996	0.255	0.129	0.235	0.135	0.148	0.054	0.109	0.725	0.556	0.008	0.172	0.953	10.763	11.69	26.503
1997	0.31	0.164	0.286	0.17	0.142	0.065	0.12	1.09	0.832	0.009	0.191	1.095	9.269	10.33	22.773
1998	0.315	0.18	0.29	0.187	0.175	0.068	0.142	1.08	0.797	0.008	0.22	1.042	8.794	9.68	23.278
1999	0.295	0.152	0.273	0.158	0.114	0.047	0.125	1.068	0.834	0.003	0.151	1.433	5.415	6.59	17.413
2000	0.436	0.192	0.403	0.2	0.142	0.046	0.126	1.174	0.89	0.004	0.183	0.631	6.057	6.61	16.874
2001	0.403	0.166	0.372	0.173	0.181	0.076	0.092	1.277	0.954	0.003	0.262	0.348	9	9.29	22.306
2002	0.467	0.148	0.431	0.154	0.147	0.049	0.083	1.166	0.858	0.003	0.227	1.396	5.564	6.71	16.774

## 11.2 Catches, species groups 13-35

Table 2. Estimated catches for species groups 13 - 35 ( $t \cdot km^2 \cdot year^{-1}$ ) for the Chesapeake Bay since 1950 used in the Chesapeake Bay Fisheries Ecosystem Model. Estimated from a variety of sources as described in the text.

Group	Alewife	Eel	Catfish	White perch	Spot	Spot	Shad	Gizzard shad	Blue crab	Oyster	Oyster	Soft clam	Hard clam
Group no.	13	14	15	17	18	18	19	22	31	33	33	34	35
1950	1.577	0.065	0.091	0.073		0.456	0.203	128.1	4.329	1.36		0.073	0.071
1951	1.789	0.056	0.079	0.064		0.508	0.22	140.4	4.005	1.34		0.011	0.067
1952	1.513	0.055	0.091	0.058		0.619	0.256	165.2	3.796	1.56		0.126	0.059
1953	1.299	0.045	0.11	0.053		0.407	0.204	225.3	3.732	1.68		0.011	0.045
1954	1.448	0.036	0.135	0.056		0.452	0.212	240.2	3.332	1.89		0.059	0.037
1955	1.225	0.04	0.156	0.056		0.416	0.225	1389.8	2.968	1.78		0.128	0.043
1956	1.234	0.041	0.154	0.06		0.333	0.24	1021	3.176	1.68		0.126	0.042

1957	1.006	0.035	0.148	0.051		0.382	0.239	579.6	3.529	1.58		0.178	0.05
1958	1.032	0.037	0.14	0.056		0.547	0.188	422.7	3.222	1.7		0.205	0.045
1959	0.995	0.037	0.168	0.073		0.356	0.148	150.1	2.942	1.51		0.253	0.088
1960	0.862	0.018	0.161	0.056		0.405	0.122	113.6	3.847	1.23		0.213	0.083
1961	0.816	0.017	0.132	0.068		0.109	0.143	51	3.937	1.25		0.307	0.105
1962	1.218	0.015	0.124	0.095		0.215	0.172	43.8	4.337	0.9		0.311	0.094
1963	1.251	0.026	0.092	0.067		0.134	0.142	11.9	3.52	0.83		0.37	0.117
1964	1.269	0.023	0.084	0.035		0.288	0.161	39.1	3.952	1		0.357	0.126
1965	1.746	0.043	0.06	0.068		0.155	0.195	65.5	4.266	0.96		0.336	0.124
1966	1.363	0.031	0.069	0.091		0.102	0.162	20.7	4.772	0.96		0.238	0.092
1967	1.431	0.044	0.06	0.064		0.392	0.136	13.1	4.278	1.17		0.253	0.098
1968	1.648	0.044	0.064	0.085		0.1	0.159	16.1	3.155	1.03		0.359	0.109
1969	1.54	0.049	0.072	0.104		0.092	0.161	8.2	3.471	1.01		0.282	0.122
1970	0.958	0.068	0.061	0.075		0.547	0.234	6.5	3.705	1.12		0.272	0.086
1971	0.705	0.066	0.081	0.077		0.044	0.112	30.1	3.945	1.16		0.088	0.098
1972	0.582	0.033	0.089	0.054		0.252	0.137	26.4	3.791	1.09		0.03	0.069
1973	0.516	0.02	0.069	0.039		0.215	0.138	56.1	3.154	1.15		0.095	0.065
1974	0.686	0.073	0.08	0.026		0.188	0.081	71.8	3.435	1.13		0.057	0.068
1975	0.552	0.068	0.081	0.027		0.164	0.06	69.5	3.2	1.03	0.744	0.079	0.053
1976	0.2	0.033	0.066	0.023		0.097	0.046	36.8	2.696	0.95	0.675	0.075	0.042
1977	0.068	0.025	0.089	0.032		0.151	0.07	7.4	3.127	0.82	0.591	0.157	0.048
1978	0.106	0.062	0.068	0.048		0.256	0.06	2.7	2.839	1.02	0.652	0.131	0.024
1979	0.083	0.067	0.069	0.033		0.2	0.046	4.8	3.313	0.98	0.612	0.087	0.029
1980	0.062	0.034	0.103	0.043		0.14	0.045	15.9	3.182	1.03	0.678	0.071	0.036
1981	0.028	0.067	0.08	0.042	0.118	0.091	0.023	17.6	4.242	0.98	0.713	0.072	0.053
1982	0.064	0.036	0.08	0.045	0.079	0.064	0.027	5.8	3.619	0.79	0.557	0.089	0.032
1983	0.091	0.036	0.088	0.037	0.15	0.131	0.028	23.1	3.941	0.53	0.338	0.043	0.053
1984	0.063	0.041	0.079	0.052	0.049	0.046	0.061	240.5	3.94	0.56	0.365	0.06	0.034
1985	0.028	0.04	0.088	0.041	0.136	0.13	0.037	193.7	3.968	0.6	0.39	0.048	0.033
1986	0.052	0.038	0.115	0.049	0.187	0.131	0.032	301.1	3.402	0.62	0.361	0.159	0.042
1987	0.104	0.039	0.1	0.051	0.299	0.266	0.037	242.5	3.086	0.39	0.174	0.198	0.046
1988	0.062	0.036	0.097	0.064	0.126	0.127	0.041	341.7	3.169	0.24	0.109	0.183	0.059
1989	0.038	0.039	0.129	0.044	0.193	0.172	0.05	333.4	3.431	0.2	0.111	0.097	0.069
1990	0.036	0.032	0.119	0.065	0.2	0.123	0.039	263.3	3.909	0.2	0.129	0.077	0.071
1991	0.044	0.04	0.101	0.066	0.274	0.204	0.034	293.5	3.726	0.15	0.106	0.016	0.048

1992	0.085	0.04	0.099	0.061	0.258	0.205	0.034	347.7	2.885	0.1	0.057	0.036	0.05
1993	0.08	0.045	0.107	0.09	0.229	0.234	0.03	709.6	4.012	0.03	0.025	0.02	0.072
1994	0.058	0.043	0.14	0.089	0.274	0.3	0.018	948.9	3.62	0.05	0.038	0.015	0.053
1995	0.019	0.032	0.105	0.08	0.208	0.233	0.007	976.7	3.367	0.07	0.06	0.014	0.043
1996	0.006	0.028	0.155	0.104	0.174	0.2	0.011	1314.3	3.211	0.05	0.04	0.011	0.036
1997	0.016	0.028	0.135	0.15	0.211	0.23	0.024	694.1	3.555	0.08	0.065	0.01	0.031
1998	0.009	0.03	0.182	0.097	0.261	0.29	0.021	1009.3	2.609	0.12	0.083	0.007	0.025
1999	0.009	0.029	0.169	0.09	0.165	0.187	0.013	1068.7	2.843	0.13	0.123	0.008	0.03
2000	0.009	0.025	0.138	0.124	0.215	0.231	0.009	166.9	2.24	0.11	0.108	0.003	0.023
2001	0.013	0.025	0.162			0.222	0.014	471.7	2.154	0.07		0.01	0.028
2002	0.015	0.018	0.147			0.199	0.008	615.7	2.264	0.03		0.002	0.031

### 11.3 *Striped bass growth parameters*

Table 3. Striped bass growth parameters.

<i>Model group: Striped bass resident</i>			
<i>Growth parameters</i>		<i>Stock assessment parameters</i>	
		<b>Parameter</b>	<b>Value Estimated</b>
L <sub>∞</sub> (maximum length)	95.5	Log R <sub>0</sub> (unfished recruits)	6.095 1
K (growth constant)	0.110	Compensation (recruitment)	14.24 2
T <sub>0</sub> (time at length 0)	-1.126	natural mortality (M)	0.15 -1
A (length-weight coefficient)	9.99E-06	length @ 50% vulnerability	34.53 -1
B (length-weight exponent)	3.0851167	Shape	0.5 -1
W <sub>m</sub> /W <sub>∞</sub>	0.1	Delta	0.309 3
<i>Maturity ogive parameters</i>			
length @ 50% mature (lh)	71.1		
shape (g)	0.25		

## 11.4 Striped bass biomass

Table 4. Biomass (relative) of striped bass in the Chesapeake Bay Fisheries Ecosystem Model. Resident striped bass includes year classes 1 to 6 (12 to 83 months of age). Estimated from a variety of sources as described in the text.

Group	SB YOY B (kg)	*SB YOY B (kg)	SB Res B (kg)	*SB Res B (kg)	*SB Res B MSP(kg)	SB Mig B (kg)	*SB Mig B (kg)	*SB Mig B MSP (mt/km2)
Data Source	Calculated with data from tables 11 and 16 of ASMFC 2003 SB SA	Calculated with data from tables 13 and 18a of the 2005 ASMFC SB SA	Calculated with data from tables 11 and 16 of ASMFC 2003 SB SA	Calculated with data from tables 13 and 18a of the 2005 ASMFC SB SA	MSP estimate	Calculated with data from tables 11 and 16 of ASMFC 2003 SB SA	Calculated with the data from tables 13 and 18a of the 2005 ASMFC SB SA	MSP estimate
1950					5054056.086			
1951					4404263.816			
1952					4242201.800			
1953					4414565.898			
1954					4855072.339			
1955					5097882.750			
1956					6080353.035			
1957					8361671.314			
1958					9978378.041			
1959					10005523.960			
1960					9747647.087			
1961					8584217.538			
1962					8406517.611			
1963					7454031.873			
1964					7683335.424			
1965					8229713.024			
1966					8025512.177			
1967					8212299.446			
1968					8268757.809			
1969					6482731.512			
1970					5891137.404			
1971					7193377.369			
1972					7329748.884			
1973					5289128.147			
1974					3605957.530			
1975					2546979.554			
1976					2153180.469			
1977					1555873.213			
1978					1525929.522			
1979					1937919.104			
1980					1319880.278			
1981					577713.883			
1982	185	199	3690	3830	632694.641	2293	2563	
1983	586	636	3570	3755	906548.780	1925	2630	
1984	529	576	5306	5715	231288.372	2973	3183	

1985	195	215	5091	5611	250802.393	2909	2794	
1986	366	387	8551	9470	672590.965	2116	2327	
1987	679	789	11447	12514	1723760.466	3048	3813	
1988	1432	1618	15562	17163	4052464.804	3064	3693	
1989	807	897	19014	20855	8278235.282	6280	7868	
1990	636	674	20537	22679	14758505.323	8548	7644	
1991	1518	1815	24407	27194	21566483.107	12860	14625	0.561
1992	731	871	28354	32404	27712441.373	14393	17579	0.777
1993	622	775	34141	39108	32400576.860	17929	21303	1.032
1994	3114	3975	43587	51573	35593992.442	20859	25426	1.305
1995	3024	3735	45656	55683	35656745.204	24614	29366	1.546
1996	1489	1810	57632	72594	33612075.364	33920	40582	1.727
1997	1366	2026	52344	66915	29747954.775	28690	36612	1.823
1998	1054	4144	55402	61640	28034679.814	30038	38254	1.856
1999	1228	6809	56180	56721	25790968.200	29295	37498	1.896
2000	866	3057	63586	51959	22614237.572	40178	46929	1.911
2001	1958	2478	46825	51771	21146081.449	34950	52135	1.901
2002		2163	44382	47699	17671866.443	31939	58406	1.918

### 11.5 *Striped bass fishing mortality*

Table 5. Fishing mortality estimates, 1950-2000, for striped bass in the Chesapeake Bay used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	*SB Res Combined F	*SB Mig Combined F	*SB Mig F
Data Source	For 1950-1958 an Ecopath estimation was used; for 1959-1981 Gibson 1993, Appendix Table 3, Run 2 was used; 1982-2002 from file entitled, "2005 striped bass processing ASMFC_VPA 2005 ; MSP estimate	For 1950-1958 an Ecopath estimation was used; 1959-1981 from Gibson 1993, Appendix Table 3, Run 3; 1982-2002 from file entitled, "2005 striped bass processing ASMFC_VPA 2005 ; MSP estimate	From Table 24 ASMFC 2005 SB SA for MD CB Age 7+
1950	0.360	0.370	
1951	0.360	0.370	
1952	0.360	0.370	
1953	0.360	0.370	
1954	0.360	0.370	
1955	0.360	0.370	
1956	0.360	0.370	

1957	0.360	0.370	
1958	0.360	0.370	
1959	0.480	0.560	
1960	0.630	0.920	
1961	0.610	0.930	
1962	0.720	0.990	
1963	0.730	0.880	
1964	0.660	0.800	
1965	0.620	0.510	
1966	0.660	0.520	
1967	0.640	0.520	
1968	0.800	0.600	
1969	0.910	0.630	
1970	0.730	0.500	
1971	0.750	0.810	
1972	1.420	1.560	
1973	1.220	1.330	
1974	1.150	1.170	
1975	1.080	1.070	
1976	1.140	1.390	
1977	1.200	1.520	
1978	0.850	0.860	
1979	1.210	1.250	
1980	1.640	1.890	
1981	1.330	1.920	
1982	0.354	0.426	
1983	0.354	0.092	
1984	0.354	0.121	
1985	0.273	0.113	

1986	0.078	0.070	
1987	0.035	0.140	
1988	0.043	0.092	0.070
1989	0.040	0.174	0.040
1990	0.073	0.196	0.110
1991	0.063	0.168	0.150
1992	0.050	0.239	0.140
1993	0.043	0.208	0.130
1994	0.070	0.227	0.120
1995	0.098	0.256	0.240
1996	0.130	0.290	0.200
1997	0.143	0.320	0.280
1998	0.200	0.330	0.240
1999	0.178	0.353	0.390
2000	0.208	0.353	0.210
2001	0.208	0.353	0.130
2002	0.208	0.353	0.120

## 11.6 *Bluefish growth parameters*

Table 6. Bluefish growth and maturity parameters used in the single-species stock assessment, as well as ratio of weight at maturity and asymptotic weight used in multi-stanza calculations. Estimated parameter values from the stock assessment model are listed on the right (positive phase values indicate parameter was estimated by fitting the model to time-series data; negative values indicate parameter was fixed).

<i>Model Group: Bluefish</i>				
<i>Growth Parameters</i>		<i>Stock assessment parameters</i>		
		<b>Parameter</b>	<b>Value</b>	<b>Estimated</b>
$L_{\infty}$ (cm) (maximum length)	87.2	Log $R_0$ (unfished recruits)	7.729	1
K (year <sup>-1</sup> ) (growth constant)	0.26	Compensation (recruitment)	5.84	2
$T_0$ (year) (time at length 0)	-0.93	natural mortality (M)	0.26	-1
A (length-weight coefficient)	1.09E-05	length @ 50% vulnerability	34	-1
B (length-weight exponent)	3.0548	shape	0.5	-1
$W_m/W_{\infty}$	0.2007328	delta	0.01	3
<i>Maturity ogive Parameters</i>				
length @ 50% mature (lh)	35.433408			
shape (g)	0.3544103			

## 11.7 *Bluefish biomass and fishing mortality*

Table 7. Bluefish biomass (B, relative) and fishing mortality (F, year<sup>-1</sup>) used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	Bluefish YOY B SRA (t·km <sup>-2</sup> )	*Bluefish YOY B (kg)	Bluefish Ad B SRA (t·km <sup>-2</sup> )	*Bluefish Ad B (kg)	Bluefish Ad F SRA
Data Source	SRA estimate	Calculated from tables 10 and 20 of the NFSC 41st SAW	SRA estimate	Calculated from tables 10 and 20 of the NFSC 41st SAW	SRA estimate
1950	0.010		0.240		0.483
1951	0.010		0.153		0.481
1952	0.054		0.129		0.551
1953	0.035		0.150		0.409
1954	0.028		0.171		0.446
1955	0.033		0.174		0.451
1956	0.038		0.181		0.499
1957	0.039		0.185		0.429
1958	0.040		0.200		0.261
1959	0.041		0.244		0.242
1960	0.044		0.286		0.136
1961	0.053		0.358		0.242
1962	0.062		0.403		0.405
1963	0.077		0.403		0.465
1964	0.086		0.407		0.274
1965	0.086		0.491		0.120
1966	0.086		0.639		0.112
1967	0.103		0.791		0.048
1968	0.131		1.012		0.105
1969	0.158		1.217		0.063
1970	0.195		1.515		0.131



1971	0.226		1.765		0.118
1972	0.269		2.081		0.170
1973	0.301		2.343		0.377
1974	0.339		2.225		0.462
1975	0.368		2.065		0.479
1976	0.354		1.980		0.657
1977	0.335		1.674		0.613
1978	0.325		1.535		0.555
1979	0.287		1.492		0.630
1980	0.268		1.340		0.675
1981	0.263		1.183		0.526
1982	0.242	3250	1.204	85488	0.474
1983	0.219	2255	1.234	78837	0.667
1984	0.222	2764	1.056	70434	0.442
1985	0.227	1659	1.131	67245	0.686
1986	0.199	1259	0.978	100391	0.815
1987	0.211	1269	0.795	68569	1.013
1988	0.187	2542	0.616	50927	1.054
1989	0.156	2704	0.497	48886	0.702
1990	0.124	3320	0.515	57459	0.784
1991	0.102	2104	0.453	49542	1.041
1992	0.106	733	0.332	51727	0.707
1993	0.095	1019	0.347	58201	0.692
1994	0.070	851	0.338	43040	0.616
1995	0.074	1424	0.326	49101	0.584
1996	0.072	1740	0.324	40323	0.458
1997	0.070	971	0.352	35916	0.404
1998	0.070	1546	0.383	27546	0.457
1999	0.075	1744	0.390	28951	0.291

2000	0.082	1623	0.458	27784	0.310
2001	0.083	2997	0.514	28273	0.353
2002	0.096	2254	0.546	35576	0.270

### 11.8 Weakfish growth parameters

Table 8. Weakfish growth parameters.

<i>Model Group: Weakfish</i>				
<i>Growth Parameters</i>		<i>Stock assessment parameters</i>		
		<b>Parameter</b>	<b>Value</b>	<b>Estimated</b>
$L_{\infty}$	68.6	Log_ $R_0$	6.095	1
K	0.35	Compensation	14.24	2
$T_0$	-0.051	natural mortality (M)	0.25	-1
A	1.10E-05	length @ 50%		
B	2.9575	vulnerability	34.53	-1
$W_m/W_{\infty}$	0.1382508	Shape	0.5	-1
<i>Maturity Ogive Parameters</i>		Delta	0.309	3
length @ 50% mature				
(lh)	25			
shape (g)	1			

### 11.9 Weakfish biomass and fishing mortality

Table 9. Weakfish relative biomass (B) and fishing mortality (F) estimates used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	*Weakfish Ad F MSP	Weakfish B SRA (t·km <sup>-2</sup> )	*Weakfish B MSP(t)	Weakfish F SRA (t·km <sup>-2</sup> )
Data Source	MSP estimate	SRA estimate	MSP estimate	SRA estimate
1950		0.489		0.585
1951		0.290		0.473
1952		0.212		0.524
1953		0.201		0.707
1954		0.220		0.671

1955		0.232		1.140
1956		0.174		1.331
1957		0.142		1.035
1958		0.156		0.707
1959		0.190		0.258
1960		0.270		0.248
1961		0.332		0.274
1962		0.372		0.282
1963		0.402		0.184
1964		0.465		0.237
1965		0.501		0.279
1966		0.513		0.144
1967		0.582		0.074
1968		0.681		0.116
1969		0.739		0.088
1970		0.803		0.191
1971		0.783		0.217
1972		0.751		0.243
1973		0.714		0.490
1974		0.565		0.383
1975		0.523		0.591
1976		0.428		0.640
1977		0.365		0.776
1978		0.302		0.907
1979		0.250		1.709
1980		0.147		2.878
1981	0.410	0.084	50961	2.080
1982	0.400	0.095	34440	1.625
1983	0.500	0.105	29047	2.153

1984	0.630	0.076	21060	1.973
1985	0.450	0.073	24840	1.983
1986	0.560	0.073	27562	2.126
1987	0.500	0.066	23432	2.239
1988	0.850	0.059	15408	2.337
1989	1.060	0.056	7656	1.659
1990	0.800	0.067	6801	1.440
1991	0.650	0.073	8602	1.068
1992	0.610	0.086	7917	0.580
1993	0.460	0.120	9770	0.558
1994	0.320	0.146	16875	0.575
1995	0.250	0.162	23805	0.549
1996	0.270	0.182	25841	0.600
1997	0.260	0.196	28037	0.611
1998	0.270	0.209	29229	0.678
1999	0.220	0.212	28436	0.590
2000	0.260	0.230	23436	0.549
2001	0.310	0.249	17118	0.369
2002	0.350	0.297	12765	0.279

**11.10 Atlantic croaker biomass and fishing mortality**

Table 10. Atlantic croaker relative biomass (B) and fishing mortality (F) estimates used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	Atl. croaker B (VIMS biomass index value)	*Atl. Croaker B (t)	*Atl. Croaker F
Data Source	Estimated in: Croaker Atlantic.xls	Calculated with data from table G9 of the ASMFC 2003 Atlantic Croaker SA (Nov 05)	From Table G8 of ASMFC 2003 Atlantic Croaker SA (Nov 05)
1973		13196	0.170
1974		13196	0.280

1975		21261	0.270
1976		38338	0.220
1977		45165	0.270
1978		39883	0.330
1979	0.310	30057	0.500
1980	0.080	22389	0.410
1981	0.200	18566	0.280
1982	1.100	15387	0.350
1983	0.310	11746	0.230
1984	1.590	27920	0.150
1985	2.220	37804	0.120
1986	0.910	49070	0.090
1987	0.400	57373	0.080
1988	0.720	63800	0.090
1989	1.950	62221	0.060
1990	1.720	60099	0.050
1991	1.350	57085	0.040
1992	2.460	73274	0.030
1993	0.460	82924	0.040
1994	0.930	90023	0.050
1995	0.340	94131	0.060
1996	1.620	96686	0.060
1997	0.930	89624	0.100
1998	0.980	81590	0.100
1999	0.510	84412	0.090
2000	0.280	91040	0.090
2001	0.690	88773	0.110
2002	0.290	80328	0.110

## 11.11 *Summer flounder biomass and fishing mortality*

Table 11. Summer flounder relative biomass (B) and fishing mortality (F) estimates used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	Summer flounder B (t·km <sup>-2</sup> )	*Summer flounder B (t)	*Summer flounder F
Data Source	Estimated by Ecopath based on an EE of 0.95	From Table 3-5 of the NFSC 41st SAW report	For 1982-1997, values are from the Stock Assessment of Summer Flounder for 2003, NFSC Doc. 03-09; For 1998-2002, values are from the NFSC 41st SAW (2005) table entitled, "Catch & Status Table: Summer Flounder" on pg. 11
1982	41939		0.940
1983	48802	17501	2.150
1984	44553	18837	1.240
1985	40196	16087	1.360
1986	38453	14972	1.590
1987	35403	13934	1.060
1988	29412	14424	2.070
1989	16122	8130	1.900
1990	16449	5217	1.650
1991	17102	7453	1.580
1992	17647	6007	1.410
1993	21351	7303	0.980
1994	28214	9249	1.320
1995	35948	11960	1.240
1996	36928	15611	1.150
1997	32244	15886	1.220
1998	37800	15669	0.910
1999	36275	17794	0.970
2000	36819	16497	0.990
2001	43137	19381	0.860
2002		25544	0.650

## 11.12 Menhaden biomass and fishing mortality

Table 12. Menhaden biomass (B, relative values) and fishing mortality (F) estimates used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	*Menh PRFC pound net index	*Menhjuv ASMFC coastwide index	*Menh Chesapeake Age 0-1 B (t·km <sup>-2</sup> )	*MenhChesAge 2+ B (t·km <sup>-2</sup> )	*Menh Atlantic B MSP (t)	*Menh Ages 0-1 F	*Menh Ages 2+ F	Menh. F2	*Menhad en F MSP
Data Source	Fig. 7.6 ASMFC Menhaden Stock Assessment 2004	Fig. 7.5 of ASMFC Menhaden Stock Assessment 2004	Estimated in Menhaden ASMFC assessment 1993 extracts 07 revis - see green highlighted worksheets	Estimated in Menhaden ASMFC assessment 1993 extracts 07 revis - see green highlighted worksheets	MSP estimate	Estimated in Menhaden ASMFC assessment 1993 extracts 07 revis - see green highlighted worksheets	Estimated in Menhaden ASMFC assessment 1993 extracts 07 revis - see green highlighted worksheets	From Fig. 7.8 in ASMFC 2004 Menhaden Stock Assessment	MSP estimate
1950						0.030	0.379	0.510	
1951						0.033	0.416	0.560	
1952						0.040	0.500	0.680	
1953						0.054	0.682	0.920	
1954						0.058	0.724	0.980	
1955			274.253	55.841		0.023	0.651	0.996	
1956			119.871	59.941		0.118	0.555	1.597	
1957			189.130	67.519		0.061	0.424	1.117	
1958			232.391	57.330		0.022	0.503	0.821	
1959		2.455	240.063	118.991		0.102	0.164	0.992	
1960		1.681	113.551	88.831		0.014	0.379	0.500	
1961		1.453	143.217	76.415		0.037	0.434	0.905	
1962		4.133	192.628	53.086		0.022	0.596	1.273	
1963		2.359	134.977	30.621		0.042	0.568	1.653	
1964	50285.390	0.131	143.998	22.553		0.041	0.535	1.739	
1965	50285.390	1.357	123.087	19.910	367789.300	0.061	0.540	2.463	2.300
1966	38856.900	0.720	228	17.601	326579.775	0.029	0.456	2.180	2.262
1967	20571.300	0.946	123.205	23.780	445263.858	0.050	0.283	1.392	1.546
1968	19428.450	2.672	215.266	21.093	383623.587	0.020	0.538	1.637	1.636
1969	12571.350	2.762	432.059	24.751	440292.429	0.010	0.265	0.972	0.576
1970	38856.900	2.125	162.579	31.780	593658.464	0.047	0.302	1.242	1.013
1971	36571.200	3.760	279.069	32.125	579451.496	0.011	0.428	0.860	0.983
1972	73142.390	3.895	55.111	33.484	611307.975	0.144	0.492	1.388	1.417
1973	139427.700	4.576	212.670	34.902	635443.007	0.020	0.541	1.215	1.555
1974	143999.100	5.348	190.825	29.136	531375.140	0.022	0.523	1.227	1.103
1975	115427.800	5.301	212.988	27.389	500866.402	0.017	0.477	1.017	0.739
1976	100570.800	5.709	170.534	34.677	636569.546	0.039	0.462	1.188	0.793
1977	150856.200	5.435	143.594	35.740	585195.680	0.027	0.529	1.052	0.930
1978	147427.600	4.344	220.396	35.458	644519.508	0.015	0.554	1.175	0.878
1979	109713.600	5.070	277.266	37.563	680746.024	0.018	0.536	1.162	0.667

1980	149713.30 0	4.796	166.659	43.095	792437.012	0.038	0.475	1.284	1.185
1981	167998.90 0	5.067	223.388	38.133	686248.524	0.018	0.560	1.012	0.269
1982	158856.10 0	5.067	185.835	47.511	877809.128	0.023	0.447	1.196	0.796
1983	177141.70 0	4.611	233.383	36.565	551590.460	0.017	0.656	1.318	1.063
1984	105142.20 0	5.292	236.089	30.344	555868.213	0.027	0.504	1.502	0.607
1985	117713.50 0	4.973	200.706	40.007	704552	0.028	0.369	1.034	0.802
1986	84570.890	4.971	169.876	42.689	778336.459	0.006	0.347	0.590	0.703
1987	157713.30 0	3.834	170.380	49.914	898323.903	0.015	0.386	0.651	1.084
1988	145141.90 0	4.833	217.910	37.059	507110.261	0.008	0.507	0.860	1
1989	87999.450	4.196	173.684	45.654	844022.896	0.036	0.332	1.092	1.141
1990	52571.100	4.376	268.177	48.116	862412.886	0.007	0.519	0.919	1.127
1991	65142.440	4.375	416.227	43.657	805783.953	0.021	0.378	1.410	1.227
1992	69713.850	3.329	184.199	43.740	806222.868	0.031	0.324	0.893	1.244
1993	111999.30 0	2.691	296.487	57.952	1062836.038	0.011	0.311	0.708	1.757
1994	68570.990	3.236	113.533	41.616	743953.921	0.014	0.379	0.597	1.109
1995	71999.550	2.871	72.026	47.377	854719.103	0.051	0.400	0.780	1.652
1996	52571.100	2.597	34.788	41.640	739099.464	0.046	0.431	0.558	1.760
1997	58285.350	2.914	67.222	35.626	636683.833	0.027	0.434	0.730	1.789
1998	39999.750	2.959	219.277	31.866	575809.970	0.008	0.460	0.963	2.070
1999	50285.390	2.776	135.181	26.429	482852.894	0.018	0.341	0.900	1.881
2000	50285.390	2.411	37.519	28.900	522824.527	0.028	0.349	0.617	1.316
2001	34285.500	3.092	141.848	35.859	643162.852	0.004	0.418	0.739	1.961
2002	31999.800	2.546	128.452	27.273	491494.918	0.018	0.340	0.738	1.605

### 11.13 *American shad and blueback herring biomass*

Table 13. American shad and blueback herring relative biomass in the Conowingo Dam fish lifts, Susquehanna River, Maryland, 1972-2002, used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	Blueback B (t·km <sup>-2</sup> )	Am. Shad Conowingo fish lift GM
Data Source	Dick St. Pierre, pers. comm.	Conowingo fish lift Geometric Mean 1984- 2002 (From Bob Sadzinski, MDNR)
1972	58198	
1973	330341	
1974	340084	
1975	69916	
1976	35519	
1977	24395	
1978	13098	



1979	2282	
1980	502	
1981	618	
1982	25249	
1983	517	
1984	311	0
1985	6763	0.510
1986	6327	1.740
1987	5861	3.400
1988	14570	2.490
1989	3598	5.040
1990	9658	6.620
1991	15616	9.310
1992	27533	8.760
1993	8626	5.550
1994	2851	15.610
1995	97863	14.470
1996	1132	20.090
1997	376072	38.780
1998	6211	7.810
1999	138625	21.230
2000	29289	31.440
2001	301240	59.620
2002	2465	79.210

### ***11.14 Bay anchovy biomass***

Table 14. Bay Anchovy biomass indices, provided by MD DNR and VIMS.

Year	Bay Anchovy B MDNR (index)	Bay anchovy B VIMS (index)
1958	0.460	
1959	4.400	
1960	1.190	
1961	0.990	
1962	1.920	
1963	3.140	
1964	1.980	
1965	4.300	
1966	7.670	
1967	3.870	
1968	3.210	
1969	4	
1970	1.090	
1971	0.460	
1972	1.880	

1973	1.160	
1974	4.960	
1975	1.160	
1976	1.190	
1977	0.730	
1978	0.660	0.570
1979	1.520	1.090
1980	0.860	1.140
1981	2.780	0.890
1982	1.290	0.690
1983	2.080	1.130
1984	1.650	1.210
1985	2.940	0.770
1986	2.740	1.530
1987	1.650	2.080
1988	0.830	1.350
1989	1.420	1.120
1990	2.020	0.570
1991	3.310	1.270
1992	1.820	1.240
1993	1.820	1.090
1994	0.630	1.030
1995	0.400	0.740
1996	0.200	0.720
1997	0.130	0.890
1998	0.400	0.790
1999	0.660	0.740
2000	0.200	0.860
2001	0.260	0.310
2002	0.230	

**11.15 White perch biomass and fishing mortality**

Table 15. White perch abundance (B) and fishing mortality (F, year<sup>-1</sup>) estimates used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	White Perch B Age 0 (VIMS Trawl index)	White Perch B Age 1+ (VIMS Trawl index)	White Perch B MSP(t)	White Perch F MSP
Data Source	From the VIMS Trawl Survey	From the VIMS Trawl Survey	MSP estimate	MSP estimate
1978		3.300		
1979		15.800		
1980		18.900		
1981		15.900	2467.087	0.167
1982		26.600	2578.735	0.174

1983	10	23.800	2677.583	0.134
1984	13.300	36.800	2880.024	0.176
1985	1.900	9.500	2979.551	0.133
1986	1.800	21.900	3202.804	0.147
1987	42.100	35.100	3395.362	0.146
1988	5.300	25.900	3600.671	0.175
1989	13.300	32	3714.726	0.114
1990	3.300	29.500	4048.144	0.156
1991	2.300	15.800	4236.807	0.151
1992	1.200	15	4449.518	0.132
1993	17.900	18.800	4749.236	0.189
1994	8.400	40.800	4805.407	0.185
1995	4.600	12.500	4881.603	0.160
1996	20.600	20.200	5069.711	0.207
1997	10	27.400	5041.325	0.314
1998	7.100	22.200	4548.173	0.214
1999	16.100	16.800	4503.868	0.200
2000	6	17.100	4519.615	0.284
2001	9.480	20.600	4201.165	0.265
2002	9.160	18.500	3982.242	0.256

### 11.16 *Spot biomass*

Table 16. Estimates of spot abundance (B) used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	Spot B SRA (index)	Spot B MSP (t)
Data Source	SRA estimate	MSP estimate
1950	1.400	
1951	1.260	
1952	1.120	
1953	0.920	
1954	0.870	
1955	0.780	
1956	0.710	
1957	0.700	
1958	0.640	
1959	0.480	
1960	0.410	
1961	0.290	
1962	0.340	
1963	0.330	
1964	0.380	
1965	0.330	
1966	0.360	

1967	0.460	
1968	0.360	
1969	0.460	
1970	0.610	
1971	0.440	
1972	0.630	
1973	0.670	
1974	0.750	
1975	0.860	
1976	1	
1977	1.190	
1978	1.320	
1979	1.340	
1980	1.390	
1981	1.480	7468.157
1982	1.580	1045.165
1983	1.670	1669.975
1984	1.670	2096.694
1985	1.730	3736.686
1986	1.710	4221.186
1987	1.700	3772.859
1988	1.600	2596.898
1989	1.630	3566.087
1990	1.620	3591.417
1991	1.650	3530.030
1992	1.610	2764.064
1993	1.580	2408.359
1994	1.550	2329.427
1995	1.480	1788.350
1996	1.470	1708.007
1997	1.490	1918.693
1998	1.490	1868.895
1999	1.440	1295.811
2000	1.480	1297.523
2001	1.480	
2002	1.480	

**11.17 Blue crab biomass, effort, and fishing mortality**

Table 17. Blue crab abundance (B), effort, and fishing mortality (F) estimates used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	*Blue Crab F	*Blue crab 1+ B (avg z-score re-scaled)	*Blue crab 0 B (avg z-score re-scaled)
Data Source	Estimated in: Table 23 Blue Crab Stock Assessment for Chesapeake	values are z-scores averaged from up to 4 surveys (Fig. 13 of 2005)	values are z-scores averaged from up to 4 surveys (Fig. 13 of 2005)

	Bay 2005	Blue Crab Stock Assessment) and re-scaled to get rid of negative numbers by dividing by the standard deviation	Blue Crab Stock Assessment) and re-scaled to get rid of negative numbers by dividing by the standard deviation
1950			
1951			
1952			
1953			
1954			
1955			
1956		1.793	2.129
1957		0.991	2.281
1958		1.348	0.400
1959		1.091	0.374
1960		0.444	0.054
1961		1.611	0.257
1962		2.041	0.784
1963		2.094	0.125
1964		1.090	1.087
1965		1.187	1.250
1966		3.053	0.951
1967		0.990	0.220
1968		0.649	1.632
1969		2.413	0.317
1970		2.126	3.313
1971		3.948	2.296
1972		1.542	0.610
1973		1.038	0.655
1974		1.350	0.135
1975		1.116	0.273
1976		1.075	0.468
1977		1.810	1.125
1978		1.199	0.846
1979		1.241	0.376
1980		1.392	1.908
1981		3.858	1.827
1982		1.866	1.016
1983		2.755	1.873
1984		2.934	1.276
1985		2.792	1.269
1986		2.101	1.046
1987		1.381	0.928
1988		1.679	0.756
1989		2.243	3.190
1990	0.775	2.851	3.654

1991	0.667	1.803	1.253
1992	1.120	2.051	2.728
1993	0.877	1.990	1.940
1994	1.241	1.742	1.281
1995	1.214	1.373	2.370
1996	0.772	2.243	2.111
1997	0.915	1.868	1.092
1998	1.483	0.947	1.738
1999	2.487	1.441	1.082
2000	1.726	1.028	1.048
2001	1.738	1.022	1.139
2002	1.315	1.273	1.143

**11.18 Eastern oyster biomass, effort, and fishing mortality**

Table 18. Eastern oyster effort (relative), fishing mortality (F), and abundance estimates (B) used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	Oyster F Comb	Oyster SRA F	*Oyster F MSP	*Oyster F ASPIC	*Oyster B SRA (index)	Oyster CPUE SRA	*Oyster B MSP (mt)
Data Source	MSP estimate	SRA estimate	MSP estimate	MSP estimate	SRA estimate	SRA estimate	MSP estimate
1950	0.060	0.070			20.415		
1951	0.074	0.070			19.561		
1952	0.088	0.080			18.756		
1953	0.102	0.090			17.855		
1954	0.116	0.110			16.922		
1955	0.130	0.110			15.901		
1956	0.144	0.110			14.977		
1957	0.158	0.110			14.138		
1958	0.172	0.120			13.380		
1959	0.186	0.110			12.568		
1960	0.200	0.100			11.882		
1961	0.214	0.100			11.370		
1962	0.228	0.080			10.853		
1963	0.242	0.070			10.544		
1964	0.256	0.090			10.283		
1965	0.270	0.090			9.919		
1966	0.284	0.100			9.581		
1967	0.298	0.130			9.243		
1968	0.312	0.110			8.785		
1969	0.326	0.120			8.409		
1970	0.340	0.140			8.045		
1971	0.354	0.150			7.616		
1972	0.368	0.150			7.162		
1973	0.382	0.170			6.743		

1974	0.391	0.180			6.288		
1975	0.391	0.180	0.417	0.391	5.838	16.100	19643.780
1976	0.380	0.170	0.446	0.380	5.443	15.500	16061.606
1977	0.345	0.160	0.429	0.345	5.084	12.100	14183.753
1978	0.397	0.210	0.514	0.397	4.793	14.700	13371.478
1979	0.397	0.220	0.533	0.397	4.389	14.700	11981.562
1980	0.485	0.260	0.674	0.485	3.997	13.700	10956.488
1981	0.622	0.280	0.898	0.622	3.574	14.300	9170.802
1982	0.624	0.250	0.947	0.624	3.173	13.100	6707.325
1983	0.439	0.180	0.682	0.439	2.858	9.400	5046.854
1984	0.519	0.210	0.796	0.519	2.675	8.100	4859.913
1985	0.677	0.250	1.034	0.677	2.470	7.800	4311.051
1986	0.920	0.290	1.426	0.920	2.242	9.800	3230.150
1987	0.643	0.200	1.064	0.643	1.995	6.700	1834.414
1988	0.462	0.130	0.773	0.462	1.860	5.200	1430.695
1989	0.497	0.110	0.820	0.497	1.803	5.500	1396.055
1990	0.678	0.110	1.113	0.678	1.770	5.200	1317.199
1991	0.732	0.070	1.240	0.732	1.731	5.200	1004.117
1992	0.473	0.060	0.840	0.473	1.721	5.300	701.602
1993	0.183	0.010	0.310	0.183	1.742	4.600	662.355
1994	0.227	0.020	0.353	0.227	1.811	6.200	921.641
1995	0.304	0.040	0.429	0.304	1.867	5.700	1236.115
1996	0.168	0.030	0.207	0.168	1.912	6.900	1558.448
1997	0.219	0.040	0.233	0.219	1.974	5.800	2304.557
1998	0.228	0.060	0.206	0.228	2.019	6.500	3296.258
1999	0.292	0.060	0.219	0.292	2.040	5.800	4725.387
2000	0.292	0.060			2.058	7.900	6525.113
2001	0.292	0.057			2.085		
2002	0.292	0.057			2.142		

### 11.19 Zooplankton biomass

Table 19. Estimate of relative abundance (B) for mesozooplankton and microzooplankton used in the Chesapeake Bay Fisheries Ecosystem Model.

Group	Microzoo B (ug C / liter)	Mesozoo B (ug C / liter)
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Data Source	Based on an estimate that "median rotifer and cop naup biomass value is roughly 50% of the mesozooplankton biomass value over all station-dates available in the CBP database."	Based on CBP database weighted by station
1985	18.086	36.171
1986	7.171	14.341
1987	5.381	10.762
1988	4.348	8.695
1989	10.878	21.755
1990	7.430	14.860
1991	5.781	11.563
1992	7.913	15.827
1993	6.442	12.883
1994	6.133	12.266
1995	6.179	12.358
1996	5.332	10.664
1997	4.580	9.160
1998	3.241	6.482
1999	7.214	14.428
2000	5.336	10.672
2001	4.509	9.018
2002		

### 11.20 *Diet compositions for high-trophic-level, multi-stanza, commercial fish*

Table 20. Diet compositions for high-trophic-level, multi-stanza, commercial fish in the Chesapeake Bay Fisheries Ecosystem Model. Diet compositions are expressed as proportions, and are expressed on a volume or wet weight basis. See text for sources.

EwE Group #	Prey \ Predator	Striped bass YOY	Striped bass resident	Striped bass migratory	Blue-fish YOY	Blue-fish adult	Weak-fish YOY	Weak-fish adult	White perch YOY	White perch adult
1	Striped bass YOY			0.000498				0.050		
6	Weakfish YOY		0.000856							
7	Weakfish adult		0.000994	0.000995						
8	Atlantic croaker		0.00994	0.0498		0.077	0.131	0.035		
10	Summer flounder							0.010		
11	Menhaden 0-1		0.0855		0.140	0.480		0.464		



12	Menhaden adult		0.439	0.682						
13	Alewife and blueback herring		0.0944	0.123						
14	American eel		0.0129							
16	White perch YOY	0.001								
17	White perch adult		0.00497	0.00199						
18	Spot		0.0676	0.0109	0.014	0.224		0.031		
19	American shad	0	0.00498	0.00498	0	0	0	0	0	0
20	Bay anchovy	0.105	0.00895	0.0110	0.546	0.168	0.768	0.429		0.167
22	Gizzard shad			0.0279						
23	Reef associated fish		0.00795							0.016
24	Non reef-associated fish		0.0855			0.022				
25	Littoral forage fish	0.321	0.0597		0.286	0.028			0.022	0.146
30	Blue crab YOY		0.00298	0.0229						
39	Mesozooplankton	0.124							0.549	0.167
40	Other suspension feeders		0.0587							
41	Other in/epi fauna	0.449	0.0547	0.0637	0.014		0.101	0.031	0.221	0.504
46	Import								0.208	

### 11.21 *Diet composition for other commercial fish species*

Table 21. Diets for other commercial fish species. Diets are expressed as proportions. (Numbers in the first column refer to EwE group numbers.)

EwE Group #	Prey \ Predator	Atlantic croaker	Black drum	Summer flounder	Menhaden YOY	Menhaden adult	Alewife and herring	American eel	American eel	Cat-fish	Spot
1	Striped bass YOY	0.0001000901	0	0	0	0	0	0	0	0	0.0001000901
4	Bluefish YOY	0	0	0	0	0	0	0	0	0	0
6	Weakfish YOY	0	0	0	0	0	0	0	0	0	0
8	Atlantic croaker	0	0	0.0009990011	0	0	0	0	0	0	0
13	Alewife and blueback herring	0	0	0	0	0	0	0	0	0	0
16	White perch YOY	0	0	0.009990009	0	0	0	0.000998004	0	0	0
18	Spot	0	0	0	0	0	0	0	0	0	0

19	American shad	0	0	0.05694306	0	0	0	0	0	0	0
20	Bay anchovy	0	0	0	0	0	0	0	0	0	0
22	Gizzard shad	0	0	0	0	0	0	0	0	0	0
24	Non reef-associated fish	0	0	0	0	0	0	0	0	0	0
25	Littoral forage fish	0	0	0	0	0	0	0	0	0	0
30	Blue crab YOY	0	0	0	0	0	0	0.0998004	0	0	0
32	Oyster YOY	0	0	0	0	0	0	0	0	0	0
34	Soft clam	0	0	0	0	0	0	0	0	0	0
35	Hard clam	0	0	0	0	0	0	0	0.001107663	0	0
38	Microzooplankton	0	0	0	0	0	0	0	0	0	0
39	Mesozooplankton	0	0	0.07392608	0	0	0	0	0	0	0
40	Other suspension feeders	0	0	0	0	0	0	0.001996008	0.001996008	0	0
41	Other in/epi fauna	0.05004504	0	0.1578422	0	0	0	0	0	0	0.05004504
42	Benthic algae	0	0	0	0	0	0	0	0	0	0
43	SAV	0	0	0	0	0	0	0	0.1107663	0	0
44	phytoplankton	0	0	0	0	0	0	0	0	0	0
45	Detritus	0	0	0.2137862	0	0	0	0	0	0	0
46	Import	0.5224702	0	0	0	0.2	0.4	0.038	0	0.208	0.5224702

**11.22 Diet composition for other fishes**

Table 22. Diet compositions for other fishes, expressed as proportions (sum = 1) and evaluated on a weight or volume basis. For sources, see text.

EwE Group #	Prey \ Predator	American shad	Bay anchovy	Other flatfish	Gizzard shad	Reef-associated fish	Non-reef associated. fish	Littoral forage fish	Sandbar shark	Other elasmobranchs
7	Weakfish adult								0.04	
8	Atlantic croaker								0.35	0.011
12	Menhaden adult								0.03	
13	Alewife and herring					0.05	0.05		0.01	
14	American eel								0.01	
18	Spot									0.034
20	Bay anchovy					0.101				0.073
21	Other flatfish								0.06	
24	Non-reef-associated fish									0.011
25	Littoral forage fish					0.07	0.05	0.015		

27	Other elasmobranchs								0.25	
30	Blue crab YOY					0.1		0.04	0.05	
31	Blue crab adult					0.1			0.05	
38	Microzooplankton	0.130	0.365							
39	Mesozooplankton	0.130	0.562		0.25		0.05	0.041		
40	Other suspension feeders			0.05		0.202	0.1			0.13
41	Other in/epi fauna	0.310	0.073	0.85		0.296	0.6	0.598	0.15	0.324
42	Benthic algae					0.031				
43	SAV					0.05				
44	Phytoplankton				0.75					
45	Detritus	0.100		0.1			0.15	0.283		
46	Import	0.330						0.023		0.417

### 11.23 Diet composition for commercial invertebrates

Table 23. Diet compositions for commercial invertebrates. Diet compositions are expressed as proportions and are on a volume or weight basis. For sources, see text.

EwE Group #	Prey \ Predator	Blue crab YOY	Blue crab adult	Oyster YOY	Oyster 1+	Soft clam	Hard clam
30	Blue crab YOY		0.25				
32	Oyster YOY	0.050	0.05				
34	Soft clam	0.050	0.075				
35	Hard clam	0.050	0.075				
38	Microzooplankton				0.09	0.09	
40	Other suspension feeders	0.150	0.05				
41	Other in/epi fauna	0.450	0.35				
42	Benthic algae	0.075	0.025				0.5
43	SAV	0.075	0.025				
44	Phytoplankton			1	0.9	0.9	0.25
45	Detritus	0.100	0.100		0.01	0.01	0.25

### 11.24 Diet composition for other invertebrates

Table 24. Diet compositions for other invertebrates expressed as proportions on a weight or volume basis. The diet compositions are based on general knowledge about the groups.

EwE Group #	Prey \ Predator	Ctenophores	Sea nettles	Micro-zooplankton	Meso-zooplankton	Other suspension feeders	Other in/epi fauna
23	Reef-associated fish		0.001				
25	Littoral forage fish		0.053				
36	Ctenophores		0.525				
38	Microzooplankton	0.334			0.72		0.08
39	Mesozooplankton	0.666	0.421				
41	Other in/epi fauna						0.02
42	Benthic algae					0.25	0.3

44	Phytoplankton			1	0.28	0.5	0.4
45	Detritus					0.25	0.2

## 11.25 *Diet composition for birds*

Table 25. Diets for piscivorous and non-piscivorous birds. Both diet compositions are based on general knowledge for the groups, and are expressed as proportions (wet weight or volume). Import includes food taken outside the Chesapeake Bay model area.

EwE Group #	Prey \ Predator	Piscivorous birds	Non-piscivorous seabirds
1	Striped bass YOY	011	
2	Striped bass resident	0056	
6	Weakfish YOY	0056	
11	Menhaden YOY	0.457	
12	Menhaden adult	0.022	
13	Alewife and herring	0.028	
14	American eel	0.006	
15	Catfish	0.006	
20	Bay anchovy	0.074	
21	Other flatfish	0.002	
22	Gizzard shad	0.015	
25	Littoral forage fish	0.017	
30	Blue crab YOY	0.006	
35	Hard clam		0.01
38	Microzooplankton	056	
39	Mesozooplankton	0.053	
40	Other suspension feeders		0.041
41	Other in/epi fauna		0.235
42	Benthic algae		
43	SAV		0.128
45	Detritus		0.011
46	Import	0.313	0.575

## 11.26 *Non-piscivorous bird species included in the Chesapeake Bay Fisheries Ecosystem Model*

Table 26. List of bird species (common name and scientific names given) included in the non-piscivorous group of the Chesapeake Bay Fisheries Ecosystem Model.

Common Name	Scientific Name
Brant	<i>Branta bernicla</i>
Canada Goose	<i>Branta Canadensis</i>
Snow Goose	<i>Chen caerulescens</i>
Mute Swan	<i>Cygnus olor</i>
Tundra Swan	<i>Cygnus columbianus</i>
Wood Duck	<i>Aix sponsa</i>
Gadwall	<i>Anas stepera</i>
American Wigeon	<i>Anas americana</i>
American Black Duck	<i>Anas rubripes</i>
Mallard	<i>Anas platyrhynchos</i>
Blue-winged Teal	<i>Anas discors</i>
Northern Shoveler	<i>A. Clypeata</i>
Northern Pintail	<i>Anas acuta</i>
Green-winged Teal	<i>Anas crecca</i>
Canvasback	<i>Aythya valisineria</i>

Redhead	<i>Aythya americana</i>
Ring-necked Duck	<i>Aythya collaris</i>
Greater and Lesser Scaup	<i>Aythya marila A. affinis</i>
Surf Scoter	<i>Melanitta perspicillata</i>
White-winged Scoter	<i>Melanitta fusca</i>
Black Scoter	<i>Melanitta nigra</i>
Long-tailed Duck	<i>Clangula hyemalis</i>
Bufflehead	<i>Bucephala albeola</i>
Common Goldeneye	<i>Bucephala clangula</i>
Hooded Merganser	<i>Lophodytes cucullatus</i>
Common Merganser	<i>Mergus merganser</i>
Red-Breasted Merganser	<i>Mergus serrator</i>
Ruddy Duck	<i>Oxyura jamaicensis</i>
Bald Eagle	<i>Haliaeetus leucocephalus</i>
Osprey	<i>Pandion haliaetus</i>
American Coot	<i>Fulica americana</i>
American Oystercatcher	<i>Haematopus palliatus</i>

### 11.27 *Piscivorous bird species included in the Chesapeake Bay Fisheries Ecosystem Model*

Table 27. List of piscivorous birds included in the Chesapeake Bay Fisheries Ecosystem Model. Common names and scientific names are presented. Based on input from mesotrophic levels group in Chesapeake Bay Workshop 1 (October 22-24, 2001). (D. Forsell, pers. comm.)

Common Name	Scientific Name
Common Loon	<i>Gavia immer</i>
Red-throated Loon	<i>Gavia stellata</i>
Horned Grebe	<i>Podiceps auritus</i>
Red-necked Grebe	<i>Podiceps grisegena</i>
Northern Gannet	<i>Sula bassanus</i>
Brown Pelican	<i>Pelecanus occidentalis</i>
Double-crested Cormorant	<i>Phalacrocorax auritus</i>
Great Blue Heron	<i>Ardea herodias</i>
Great Egret	<i>Casmerodius albus</i>
Snowy Egret	<i>Egretta thula</i>
Little Blue Heron	<i>Egretta caerulea</i>
Tricolored Heron	<i>Egretta tricolor</i>
Yellow-crowned Night Heron	<i>Nyctanassa violaea</i>
Black-crowned Night Heron	<i>Nycticorax nycticorax</i>
Green Heron	<i>Butorides virescens</i>
Brant	<i>Branta bernicla</i>
Canada Goose	<i>Branta canadensis</i>
Snow Goose	<i>Chen caerulescens</i>
Mute Swan	<i>Cygnus olor</i>
Tundra Swan	<i>Cygnus columbianus</i>
Wood Duck	<i>Aix sponsa</i>
Gadwall	<i>Anas stepera</i>
American Wigeon	<i>Anas americana</i>
American Black Duck	<i>Anas rubripes</i>
Mallard	<i>Anas platyrhynchos</i>

Blue-winged Teal	<i>Anas discors</i>
Northern Shoveler	<i>A. Clypeata</i>
Northern Pintail	<i>Anas acuta</i>
Green-winged Teal	<i>Anas crecca</i>
Canvasback	<i>Aythya valisineria</i>
Redhead	<i>Aythya americana</i>
Ring-necked Duck	<i>Aythya collaris</i>
Greater and Lesser Scaup	<i>Aythya marila A. affinis</i>
Surf Scoter	<i>Melanitta perspicillata</i>
White-winged Scoter	<i>Melanitta fusca</i>
Black Scoter	<i>Melanitta nigra</i>
Long-tailed Duck	<i>Clangula hyemalis</i>
Bufflehead	<i>Bucephala albeola</i>
Common Goldeneye	<i>Bucephala clangula</i>
Hooded Merganser	<i>Lophodytes cucullatus</i>
Common Merganser	<i>Mergus merganser</i>
Red-Breasted Merganser	<i>Mergus serrator</i>
Ruddy Duck	<i>Oxyura jamaicensis</i>
Bald Eagle	<i>Haliaeetus leucocephalus</i>
Osprey	<i>Pandion haliaetus</i>
American Coot	<i>Fulica americana</i>
American Oystercatcher	<i>Heamatopus palliatus</i>
Laughing Gull	<i>Larus atricilla</i>
Bonaparte's Gull	<i>Larus philadelphia</i>
Ring-billed Gull	<i>Larus delawarensis</i>
Herring Gull	<i>Larus argentatus</i>
Great Black-backed Gull	<i>Larus marinus</i>
Black-legged Kittiwake	<i>Rissa tridactyla</i>
Royal Tern	<i>Sterna maxima</i>
Common Tern	<i>Sterna hirundo</i>
Least Tern	<i>Sterna antillarum</i>
Sandwich Tern	<i>Sterna sandvicensis</i>
Black Skimmer	<i>Rynchops niger</i>

### 11.28 *Hurricanes and their assumed impact on the relative P/B for clams*

Table 28. Hurricanes in the Maryland/Virginia region, and assumed impact on the relative P/B for soft and hard clam. Data sources: [www.vdem.state.va.us/library/vahurr/va-hurr.htm](http://www.vdem.state.va.us/library/vahurr/va-hurr.htm) and [www.nhc.noaa.gov/HAW2/english/history.shtml#iris](http://www.nhc.noaa.gov/HAW2/english/history.shtml#iris).

Year	Date	Name	Hurricane category	VA/MD flooding	Assumed P/B (relative)
1952	31-Aug	Able	Weak	Minor	0.7
1953	14-Aug	Barbara	Weak		0.9
1954	15-Oct	Hazel	4		0.9
1955	12-Aug	Connie	Weak	Minor	0.7
1955	17-Aug	Diane	1	Minor	-

1955	19-Sep	Ione	Weak		-
1956	27-Sep	Flossy	Weak		0.9
1959	10-Jul	Cindy	Weak		0.9
1959	30-Sep	Gracie	3		0.9
1960	12-Sep	Donna	4	Major	0.1
1964	01-Sep	Cleo	weak	heavy rains	0.7
1969	20-Aug	Camille	5	historic record for Virginia	0.1
1971	27-Aug	Doria	TS	Minor	0.7
1972	21-Jun	Agnes	1	Major	0.1
1979	05-Sep	David	2		0.9
1985	25-Jul	Bob	2		-
1985	27-Sep	Gloria	3	Moderate	0.4
1986	17-Aug	Charley	weak	Minor	0.7
1996	12-Jul	Bertha	weak		-
1996	05-Sep	Fran	3	Major	0.1
1997	24-Jul	Danny		Minor	0.7
1998	27-Aug	Bonnie	2	Moderate	0.4
1999	04-Sep	Dennis		Minor	-
1999	15-Sep	Floyd	2	Major	0.1
2001	14-Jun	Allison	TS		0.9
2003	18-Sep	Isabel	3	historic record for Maryland	0.1

**11.29**      *Estimated relative P/B for clams, carried over to years subsequent to hurricane events*

Table 29. Estimated relative P/B for soft and hard clams assuming that the impacts of hurricanes are carried over to subsequent years, although with less severity (last column).

Year	P/B (relative)	P/B (relative), used to force simulations
1950	1	1
1951	1	1
1952	0.7	0.7
1953	0.9	0.8
1954	0.9	0.9
1955	0.7	0.7
1956	0.9	0.8
1957	1	0.9
1958	1	1
1959	0.9	0.9
1960	0.1	0.1
1961	1	0.2
1962	1	0.3
1963	1	0.4
1964	0.7	0.5
1965	1	0.6



1966	1	0.7
1967	1	0.8
1968	1	0.9
1969	0.1	0.1
1970	1	0.2
1971	0.7	0.3
1972	0.1	0.1
1973	1	0.2
1974	1	0.3
1975	1	0.4
1976	1	0.5
1977	1	0.6
1978	1	0.7
1979	0.9	0.8
1980	1	0.9
1981	1	1
1982	1	1
1983	1	1
1984	1	1
1985	0.4	0.4
1986	0.7	0.5
1987	1	0.6
1988	1	0.7
1989	1	0.8
1990	1	0.9
1991	1	1
1992	1	1
1993	1	1
1994	1	1
1995	1	1
1996	0.1	0.1
1997	0.7	0.2
1998	1	0.3
1999	0.1	0.1
2000	1	0.2
2001	0.9	0.3
2002	1	0.4
2003	0.1	0.1
2004	1	0.2

### 11.30 *Phytoplankton chlorophyll content*

Table 30. Estimate of relative chlorophyll content for the Chesapeake Bay, 1950-1994 (Harding and Perry, 1997). The series is used for comparison with Ecosim simulation results, not for driving the model. (Data not available for 1953-1963.)

Year	Chlorophyll-a
1950	5.068869
1951	4.770248
1952	1.9
1964	8.551069
1965	15.1605
1966	13.72456
1967	4.316667
1968	32.7125
1969	16.43147
1970	13.70561
1971	27.84041
1972	37.13371
1973	17.15287
1974	26.38086
1975	14.55696
1976	17.46835
1977	19.6949
1978	9.791241
1979	10.01648
1980	15.0723
1981	8.022222
1982	-----
1983	4.856979
1984	9.241438
1985	9.210595
1986	8.347284
1987	12.11841
1988	9.780312
1989	8.656951
1990	9.735246
1991	7.954487
1992	6.10325
1993	7.92401
1994	7.7015

### 11.31 Confidence intervals for basic input parameters

Table 31. Estimate of confidence intervals for basic input parameters of the Chesapeake Bay Fisheries Ecosystem Model. The confidence intervals are derived from the model pedigree. See the *EwE User's Guide* for information (Christensen *et al.*, 2004).

EwE Group #	Group	B	P/B	Q/B	Diet	Catch
1	Striped bass YOY	50	50	50	10	---
2	Striped bass resident	50	10	50	10	50
3	Striped bass migratory	50	10	50	10	50
4	Bluefish YOY	50	50	50	10	---
5	Bluefish adult	50	50	50	10	50
6	Weakfish YOY	50	50	50	10	---
7	Weakfish adult	50	50	50	10	50
8	Atlantic croaker	50	10	50	60	50
9	Black drum	80	10	50	30	50
10	Summer flounder	80	40	50	30	50
11	Menhaden YOY	50	50	50	50	50
12	Menhaden adult	50	50	50	50	50
13	Alewife and herring	80	20	50	50	50
14	American eel	80	20	50	60	50
15	Catfish	80	20	50	60	50
16	White perch YOY	50	50	50	10	---
17	White perch adult	50	50	50	60	50
18	Spot	80	10	50	30	50
19	American shad	50	30	50	10	50
20	Bay anchovy	50	30	50	50	---
21	Other flatfish	80	50	50	60	---
22	Gizzard shad	80	50	50	60	---
23	Reef-associated fish	80	10	50	60	---
24	Non-reef-associated fish	80	70	50	60	---
25	Littoral forage fish	80	70	50	10	---
26	Sandbar shark	50	70	50	60	50
27	Other elasmobranchs	50	70	50	30	---
28	Piscivorous birds	50	40	50	60	---
29	Non-piscivorous seabirds	50	40	50	60	---
30	Blue crab YOY	50	70	50	10	---
31	Blue crab adult	50	10	50	30	50
32	Oyster YOY	50	70	50	60	---
33	Oyster 1+	50	50	50	60	50
34	Soft clam	80	50	50	60	50
35	Hard clam	50	50	50	30	50
36	Ctenophores	80	40	50	60	---
37	Sea nettles	80	40	50	60	---
38	Microzooplankton	80	70	50	60	---
39	Mesozooplankton	50	40	50	60	---
40	Other suspension feeders	80	50	50	60	---

41	Other in/epi fauna	80	50	50	60	---
42	Benthic algae	80	70	---	---	---
43	SAV	80	20	---	---	---
44	Phytoplankton	80	40	---	---	---

### 11.32 *Prices for commercial catches*

Table 32. Year 2000 prices (\$US) for commercial catches of exploited groups in the Chesapeake Bay. For group 13, alewife/herring, price for alewife was used as these dominated the landings (98.6%). For group 15, catfish, average price for the two most common species (blue and channel catfish) was used.

Source: [www.searoundus.org](http://www.searoundus.org).

EWE Group #	Name	\$/kg
2, 3	Striped bass	3.91
5	Bluefish	0.77
7	Weakfish, grey	1.51
8	Atlantic croaker	0.83
9	Black drum	1.56
10	Summer flounder	3.94
12	Atlantic menhaden	0.14
13	Alewife	0.44
13	Atlantic herring	0.14
14	American eel	1.80
15	Catfish, blue	1.06
15	Catfish, channel	1.13
15	Catfish, flathead	1.03
17	White perch	1.14
18	Spot	1.14
19	American shad	0.63
26	Sandbar shark	0.71
31	Blue crab	1.79
33	Oyster, American cupped	0.74
34	Soft clam	5.48
35	Hard clam (quahog)	9.70

### 11.33 *Multispecies Production stock assessment model parameter values*

Table 33. MSP models and their extensions. B = biomass; r = intrinsic rate of increase; B<sub>0</sub> = biomass estimate to initiate time-series; t = time in years; H = harvest; c is a constant defined in “Additional Mortality Source” column; and A = prey biomass at satiation in Type III predator-prey function. Note that units of measure (Unit) vary among species. If “Additional Mortality Source” is blank, fishing mortality is the main source of loss.

Species	Equation	Unit	Bo	r	K	c	A	Additional Mortality Source	Time-series
Atlantic croaker	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1}$	Mt	4.42 10 <sup>3</sup>	1.96	1.19 10 <sup>5</sup>				1981-2003
Spot	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1} - [(cP_{t-1}(B_{t-1})^2)/(A^2 + (B_{t-1})^2)]$	Lbs	8.85 10 <sup>8</sup>	1.09	4.80 10 <sup>8</sup>	3.72	1.48 10 <sup>8</sup>	P = striped bass bay biomass estimated as below; predation losses; c=maximum per capita consumption.	1981-2004
White perch	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1}$	Lbs	4.05 10 <sup>6</sup>	0.48	1.64 10 <sup>7</sup>				1981-2003
Striped bass	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1} - (B_{t-1} L c)$	Kg	5.78 10 <sup>6</sup>	0.64	6.62 10 <sup>7</sup>	3.94		L = lesions; disease / starvation losses; c = scalar for lesions to loss	1950-2004
Weakfish	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1} - [(cP_{t-1}(B_{t-1})^2)/(A^2 + (B_{t-1})^2)]$	Mt	1.62 10 <sup>4</sup>	2.83	5.54 10 <sup>3</sup>	12.92	4.99 10 <sup>5</sup>	P = striped bass coast biomass; predation, competition, migration losses; c=maximum per capita consumption	1981-2003
Bluefish	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1} - [(cP_{t-1}(B_{t-1})^2)/(A^2 + (B_{t-1})^2)]$	Mt	1.17 10 <sup>5</sup>	1.54	3.05 10 <sup>4</sup>	7.42	4.32 10 <sup>4</sup>	P = striped bass coast biomass; predation, competition, migration losses; c=maximum per capita consumption	1981-2003

Atlantic menhaden	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1}$	Mt	2.29 $10^5$	2.27	$4.54 \cdot 10^4$				1965-2004
Gizzard shad	$B_t = B_{t-1} + rCB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1}$	N	$1.58 \cdot 10^7$	0.66	$1.51 \cdot 10^6$			C = NO3 load standardized to median	1964-2004
Oyster	$B_t = B_{t-1} + rB_{t-1} (1 - (B_{t-1} / K)) - H_{t-1} - D_{t-1}B_{t-1}$	Lbs meat	$5.85 \cdot 10^7$	0.57	$7.23 \cdot 10^7$			D = estimate percent of oysters killed by disease	1975-2006

### 11.34 Submerged Aquatic Vegetation biomass index

Table 34. Submerged Aquatic Vegetation Index based on the VIMS SAV Monitoring Program data

Year	SAV Biomass Index (ID/surveyed)
1971	0.025337
1972	
1973	
1974	0.015884
1975	
1976	
1977	
1978	0.014696
1979	
1980	0.010436
1981	0.011951
1982	
1983	
1984	0.017739
1985	0.017519
1986	0.017449

1987	0.017709
1988	
1989	0.021291
1990	0.021414
1991	0.02259
1992	0.025183
1993	0.026083
1994	0.023348
1995	0.021379
1996	0.022652
1997	0.024711
1998	0.022051
1999	0.022423
2000	0.024009
2001	0.030924
2002	0.031126