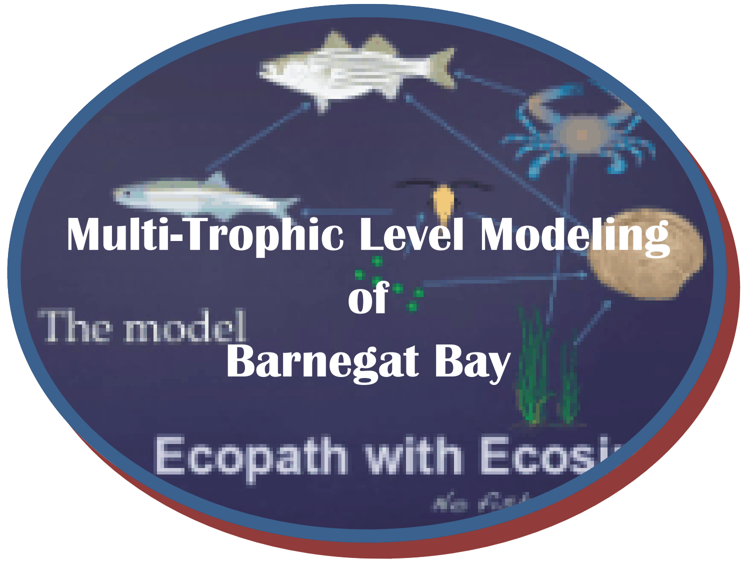
Barnegat Bay— Year 3



Dr. Olaf Jensen, Rutgers University, Principal Investigator

Heidi Fuchs and Jim Vasslides,
Rutgers University, Co-Investigators

Project Manager:
Tom Belton, Division of Science, Research and Environmental Health

Thomas Belton, Barnegat Bay Research Coordinator

Dr. Gary Buchanan, Director—Division of Science, Research & Environmental Health

Bob Martin, Commissioner, NJDEP

Chris Christie, Governor

February 2016

Ecosystem Modeling of Barnegat Bay
Year 3: Final Report

Jim Vasslides[1,2], Michael McCann[2], and Olaf Jensen[2]

[1]Barnegat Bay Partnership
PO Box 2001
Toms River, NJ 08754-2001

[2]Department of Marine & Coastal Sciences
Rutgers University
71 Dudley Rd.
New Brunswick, NJ 08901

Table of Contents

Table of Contents (*continued*)

Executive Summary

As part of the Governor's Barnegat Bay Initiative a bay-wide trophic model of the Barnegat Bay was constructed utilizing the Ecopath with Ecosim (EwE) software platform. The software consists of three distinct modules: Ecopath, a trophic mass balance model; Ecosim, for time-dynamic modeling; and Ecospace, which allows for the construction of a spatially-explicit model. Taken together, this software allows for the integration of disparate datasets into a cohesive description of the trophic interactions of an ecosystem, and allows for the investigation of the effects of various management scenarios on not only the group of interest, but on all of the components of the system contained within the model. These "what-if" scenarios can then be used to provide strategic advice to managers and stakeholders, and can form the basis for future detailed investigations.

We developed an Ecopath model for Barnegat Bay circa 1981 that contains 27 biomass groups, including 12 fish groups, 9 invertebrate groups, 3 primary producer groups, and 2 bird groups. The model was constructed utilizing a combination of Barnegat Bay-specific data (fish diets, selected species biomasses and vital rates, harvest), data from similar systems (vital rates, invertebrate diets), and software estimated values (biomasses). Piscivorous seabirds and phytoplankton are "keystone" groups in the Barnegat Bay system as determined by ecological network analysis.

Ecosim was used to model the changes in the biomass groups through time (1981-2013), using harvest data (commercial and recreational finfish and blue crabs) to drive the model. The model runs were then compared to fishery independent time series data (51 time series in total) and tuned as required to find the best model "fit". The overall fit of the model prediction to the available data is reasonable, and the model generally behaves as expected. While the biomasses of all of the groups vary through time, several groups display dramatic increases or decreases. The 17-fold increase in the biomass of croaker through time is reflective of the increase in its overwintering survivability and general population increase in the Mid-Atlantic as documented by Hare and Able (2007). Conversely, blue crab undergo a dramatic decrease, ending the period at 0.62 t/km^2, or approximately 9.9% of their starting biomass.

Once the Ecosim scenario was fit to the available time-series data, the model was extended beyond the current timeframe (1981-2013) to make predictions about the future state of the ecosystem under different management strategies. Scenarios run included the closure of Oyster Creek Nuclear Generating Station, changes in blue crab harvest, changes in hard clam harvest, and a reduction in nutrient loads to the bay. Reducing the "effort" of OCNGS by 96% in 2020 has a positive impact on the biomass of some groups, and a negative impact on others, by the end of the simulation in 2030. Of the groups directly impacted by OCNGS, Atlantic croaker has the greatest response to the reduction in mortality associated with the plant closure, decreasing in biomass by nearly 2.5% compared to the baseline simulation. This predicted decrease in biomass appears to be an indirect impact of the increases in biomass of Atlantic croaker's predators, primarily weakfish which were projected to increase by 2.1%. The other group with a greater than 1% change in biomass is blue crab, who realizes a 1.5% increase in biomass due to a decrease in mortality associated with OCNGS. The effects of changing blue crab harvest was more pronounced. Modifying the level of effort for each gear type in the blue crab fishery led to a variety of outcomes, from a near depletion of crab biomass by doubling the pot fishery to a near doubling of the baseline biomass by reducing both the winter dredge and pot fisheries. Evaluating the effects of changes to hard clam harvest proved to be problematic as the current model does not fit known hard clam population dynamics well. This is likely due to the importance of environmental factors, non-trophic related drivers, the relative paucity of survey data on hard clam biomasses through time, and the total lack of landings data.

Results from the nutrient reduction scenarios are problematic for use in management decisions because of difficulty capturing the expected changes in SAV. The primary interaction between nutrients and SAV is likely via increased shading of SAV as biomass of phytoplankton and epiphytic algae increase in response to greater nutrient loading (or decrease with decreased nutrients). This non-trophic interaction is difficult to adequately capture in a trophic model such as EwE. Because we forced changes to SAV in the nutrient reduction scenario (i.e., it is not an unconstrained model output), we recommend caution when interpreting results of the nutrient reduction scenarios with respect to SAV cover or changes in other model groups which are strongly influenced by SAV.

A three-zone model was created in Ecospace to evaluate the need to consider spatially-explicit processes in Barnegat Bay. The majority of the spatially-explicit data needed to parameterize the model came from recent years as part of the Barnegat Bay research initiative led by NJDEP, with older data used for hard clams and SAV. In general, initiating the model with present-day spatially-explicit data (i.e., 2012-2014) for a large number of taxa provided a better model fit than using historical data (i.e., early 1980s) for the relatively few taxa for which it is available (i.e., hard clams, phytoplankton, SAV). Now that this Ecospace model has been developed, it can be used to evaluate alternative, spatially-explicit management options in the future.

One way to maximize the utility of the Barnegat Bay Ecosystem Model is to use it as a tool to highlight where key data gaps still exist. In regard to model construction and fitting, the scarcity of Barnegat Bay specific data was particularly acute when it came to biomass and catch information. The initial biomass of most groups had to be estimated by the Ecopath module due to a lack of historic information, and data on entire fishery sectors were missing (recreational blue crabs, commercial and recreational hard clams). While the overall fit of the model was reasonable, additional data would not only improve the fit of the model (and our confidence in model predictions), but potentially allow us to increase the model resolution (*i.e.* age structured stanzas) to better investigate management actions. In particular, the lack of any hard clam landing information, and sporadic biomass surveys, makes it very hard to model population dynamics of this commercially important species, as evidenced in our inability to recreate known population changes in the model, and makes evaluating management strategies for this fishery highly problematic.

While the state, under the Governor's 10-point Plan, undertook a number of scientific studies that provided important details to this model, there are still a number of key data gaps that need to be addressed to improve the accuracy of this model, or future models for Barnegat Bay. Among the data most needed are:

- Regularly collected chlorophyll *a* data at relevant spatial and temporal scales. Recent efforts have generated data at the scales needed, and should be continued.
- Regularly collected hard clam biomass and landing data. Hard clam biomass data has been collected every10-15 years, and landings data (commercial and recreational) are not currently collected at all.
- Estimates of blue crab recreational landings. Our current assumption, that recreational landings are approximately 80% of total commercial landings, are based off of a study of a single year. This assumption makes the recreational fishery the largest component of blue crab harvest, and suggests that additional effort should be expended to understand how it effects blue crab biomass.
- A consistently scheduled fishery independent survey. An annual survey at the appropriate spatial scale would allow for the development of indices of relative abundances (and potentially biomasses), which would be a benefit to any future modeling exercises or fishery management efforts.
- Biomass and diet data on piscivorous seabirds, the highest rated keystone species in the model. All data for this group was modified from the Chesapeake Bay model.

An additional benefit of this type of holistic model is the ability to develop and evaluate a number of potential management scenarios from an ecosystem-wide perspective. For the Barnegat Bay the model suggests that management of a target fishery itself may not be sufficient to achieve management objectives, and that trophic or external factors may be an important contributor to population changes. Thus, management actions that address non-fishing activities (e.g., SAV restoration, stormwater runoff, pathogen loading, temperature/salinity changes, brown-tide blooms) may be necessary to restore or maintain populations of recreationally or commercially important species at their target abundance.

Chapter 1 – Introduction

1.1 Barnegat Bay

The Barnegat Bay is a lagoonal estuary located in central New Jersey. The surrounding 1,730 km^2 watershed is home to an estimated 580,000 year round residents (US Census Bureau 2012), with a summer population that swells to over 1 million with the influx of tourists. Land use is a mix of urban and suburban uses in the northeast and along the barrier islands, grading to less sparsely populated forested areas to the south and west (Kennish 2001a). Portions of the E.B. Forsythe National Wildlife Refuge and the Pinelands National Reserve are located along the eastern and western sides of the watershed, respectively. The blue crab fishery is the main commercial fishery within the bay, though there are still remnants of a historic wild-harvest hard clam fishery that was highly productive in the past (Bricelj *et al.* 2012). Commercial fishing, once an important source of income for local baymen, is now a minor component of the regional economy (Kennish 2001b), though there is a burgeoning shellfish aquaculture industry in the southern portion of the estuary (Bricelj *et al.* 2012). The Barnegat Bay is a popular destination for recreational fishing, crabbing, and clamming. The watershed is considered "highly eutrophic" (Bricker *et al.* 2007), mainly due to nutrient enrichment through non-point source pollution. The nation's oldest continuously operating nuclear power plant, Oyster Creek Nuclear Generating Station, is also located within the watershed.

1.2 Multispecies management

Historically, management of natural resources, particularly in marine systems, has occurred on a species or sector level. This single species approach has had mixed success, with recent analyses suggesting that 28% of the world's major fish resources are overexploited or depleted (FAO, 2009). In response to perceived shortcomings in the single species approach, management agencies began to utilize a multi-species approach in some circumstances, whereby the trophic interactions between a target stock and its prey were taken into account, with the assumption that a reduction in a predator's forage base would lead to reduced productivity of the predator, and thus reduced biomass available to the fishery.

While the multi-species approach accounted for single predator -prey dynamics, it was broadly recognized that fish stocks of interest were impacted by more than this simple interaction; that there was a need to consider the effects of the broader environment when managing fisheries (Ecosystem Principles Advisory Panel 1998, Pew Oceans Commission 2003, U.S. Commission on Ocean Policy 2004). Thus, was advanced the concept of ecosystem-based management (EBM), an integrated approach that considers the interaction between ecosystem components and the cumulative impacts of a full range of management activities (Rosenberg and McLeod 2005). This broad definition of EBM thus describes a gradient of interconnectivity, from a focus on multi-species interactions across a range of trophic levels, including some abiotic factors, to a comprehensive view which includes human impacts other than fishing (Hilborn 2011).

From the EBM perspective, a myriad of quantitative models have been developed to support the transition from single species/sector to ecosystem-based management, each with particular strengths and weaknesses (see Plagányi 2007 for a thorough overview). As with any other modeling exercise, the selection of an ecosystem model(s) is partially dictated by the questions that need to be answered and by the data available compared to the model's requirements (Plagányi 2007). For addressing questions on an ecosystem-wide scale the number of potential components that could be included in a model is very large, and care needs to be taken not to over-parameterize, as this can lead to the accumulation of errors and both process and parameter uncertainty (Fulton *et al.* 2003). On the opposite end of the spectrum, there may be few components for which there is robust data, and limited means to collect additional data. Therefore some *a priori* knowledge of which components are important to include to accurately portray dynamics within the ecosystem is paramount.

Ecopath with Ecosim (EwE) is a well-known program for addressing questions of aquatic ecosystem changes (Christensen and Walters 2004), and has been used within the mid-Atlantic region to construct a model of the lower Delaware Bay (Frisk *et al.* 2011) and one of Chesapeake Bay (Christensen et al 2009). Ecopath is a trophic mass balance analysis program that parameterizes an initial model using two master equations, one to describe the production term for each group and one for the energy balance for each group. This "base" model provides the foundation for the simulation component of EwE, Ecosim, where a series of coupled differential equations are used to simulate biomass dynamics through time, fitting the model to time-series reference data and forcing functions entered by the user.

The Barnegat Bay EwE model was developed to help researchers and managers approach a suite of questions related to the Barnegat Bay ecosystem from a whole system, rather than single species, perspective. Through model parameterization, simulation, and fitting, historic and current Barnegat Bay-specific data related to species biomasses, vital rates, and harvest levels were compiled into a single repository. The modeling process also helped identify gaps in our understanding of crucial ecosystem processes and relevant data needs. The model can also be used to develop and evaluate the effects of potential management scenarios from an ecosystem-wide perspective.

Chapter 2 – Methods

2.1 Barnegat Bay model boundaries

The Barnegat Bay EwE model encompasses the Barnegat Bay – Little Egg Harbor and tidal portions of its tributaries, for a total surface area of 279km^2 (Figure 1). Like most estuaries, many of the groups included in the model spend only a portion of the year or different parts of their life history within the Barnegat Bay (Able and Fahay 2010). While we recognize that migration and ontogenic habitat shifts are important in understanding community interactions, for the purposes of the Barnegat Bay model we have set biomass accumulation and net migration to zero for all of our species groups. This is equivalent to the assumption that biomass of all species groups was at equilibrium, and is a typical assumption in the absence of information to the contrary.

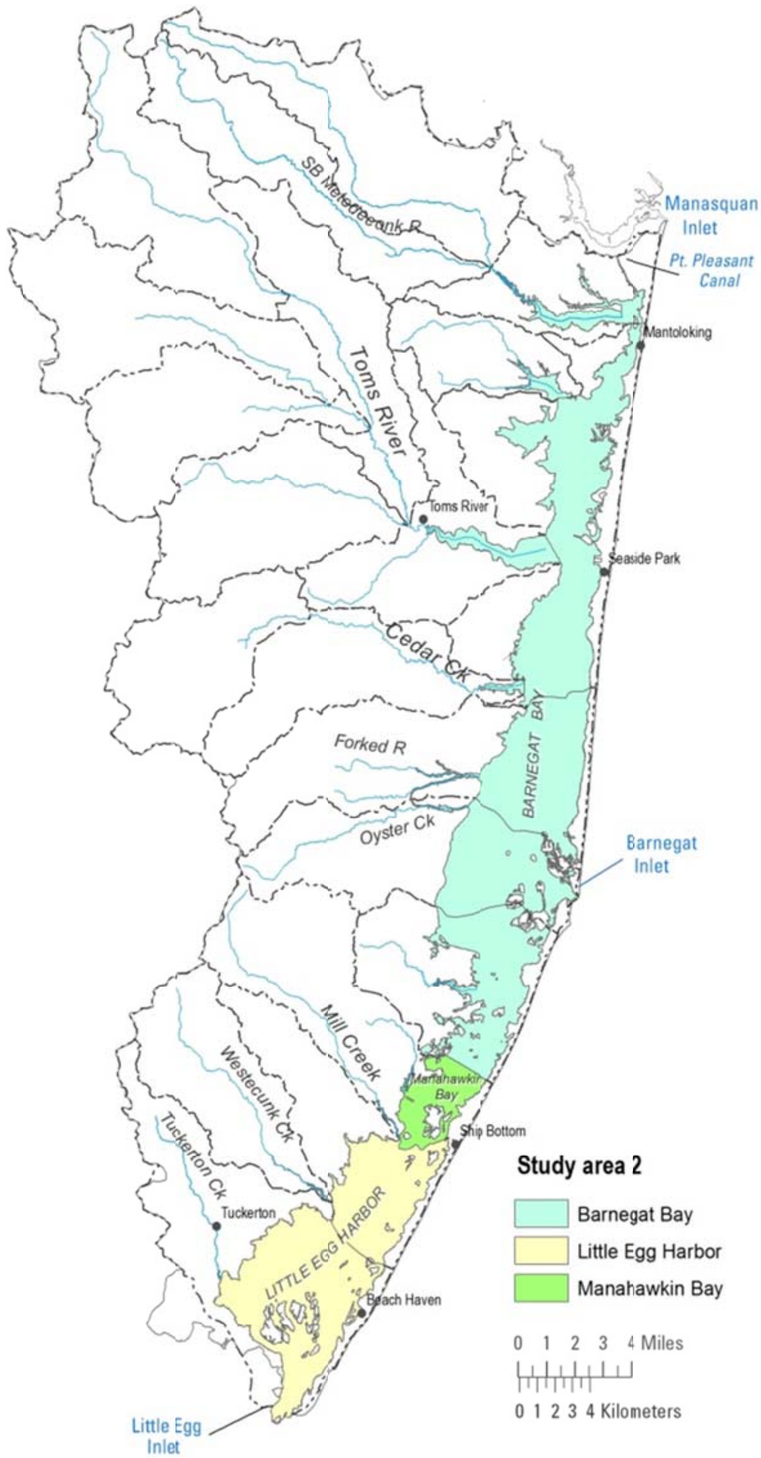
2.2 Ecopath model development

Ecopath requires four groups of basic input parameters to be entered into the model for each of the species (or groups) of interest: diet composition, biomass accumulation, net migration, and catch (for fished species). Three of the following four additional input parameters must also be input: biomass, production/biomass (P/B), consumption/biomass (Q/B), and ecotrophic efficiency. 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 (Christensen *et al.* 2008).

2.2.1 Input Parameters

The basic input parameters used in the model can be found in Table 1, and a description of the sources used to determine these values is in Appendix 1. Many of the parameter values utilized in the model at this time were not developed specifically for Barnegat Bay, as was anticipated. The use of literature derived values for P/B and Q/B is common in the development of EwE models as those parameter values are fairly consistent across systems or can be modified based on local conditions (Christensen *et al.* 2008). Barnegat Bay specific biomass estimates for the time around the beginning of the model run (1981) were available for SAV (Lathrop *et al.* 2001), hard clams (Celestino 2002), and bay anchovy (Vougliotis *et al.* 1987). We used the data collected as part of the first year of the NJDEP Barnegat Bay field research projects to estimate current biomass for ctenophores and assumed that to be constant through time (U. Howson, pers. comm.). We used the same study

Figure 1: Map of the Barnegat Bay study area.



to develop a present-day estimate of sea nettle biomass, though we set the initial biomass to 0.25% of current levels to reflect the apparent scarcity of sea nettles in the bay at that time. The biomass for Atlantic croaker, which was recorded only sporadically during field surveys at the starting time of the model (McClain *et al.* 1976), was estimated by the software to balance the requirements of its predators and fishery, as described below. Biomass of the remaining groups was estimated by the software or modified from Chesapeake Bay values (seabirds).

We attempted to utilize data from the first year of the NJDEP Barnegat Bay field research projects in combination with other Barnegat Bay specific studies for phytoplankton and amphipods (Haskin and Ray 1977), but the biomasses estimated by these studies was substantially less than that required to support the remainder of the model. We also attempted a catch-conditioned surplus production model (Walters and Martell 2004) to estimate biomass for blue crab given their importance to the recreational and commercial fishery sector. This model uses fishery landings in conjunction with a fishery independent survey to estimate the initial biomass and carrying capacity. Unfortunately, the model was non-informative, likely due to the relatively short timeframe over which landings data are available.

2.2.2 Diet Information

The diet data for most of the fish groups are based on a study conducted in the Barnegat Bay by Festa (1978), with the diets of the remaining groups taken from the primary literature or from other models (see Appendix 1 for details and Appendix 2 for the initial diet matrix). For predatory fishes, if stomach contents were listed as “unidentified fish” or as a species not included in the model that percentage of the diet was redistributed amongst the other diet categories. As described above, Atlantic croaker were scarce in the Barnegat Bay at the time of the diet study, and were not listed as a prey item for any of the piscivorus fish in the model. We know from studies in other nearby systems that when croaker are present they are a common food source for weakfish, striped bass, and bluefish (Nemerson and Able 1994, Frisk *et al.* 2006, Christensen *et al.* 2009). If they are not included in the initial diet matrix of the model there is no way to add them as a prey item during the simulation procedure. Therefore, we included croaker in the diet of the those species based on the consumption rates found in EwE models of the Delaware Bay (Frisk *et al.* 2006) and Chesapeake Bay (Christensen *et al.* 2009).

Table 1: Basic parameters for the Barnegat Bay Ecosystem Model. Values estimated by Ecopath are shown in *italics*. Estimated from a variety of sources as described in Appendix 1.						
Group #	Group name	Biomass (t/km^2)	P/B (year^{-1})	Q/B (year^{-1})	Ecotrophic Efficiency	Prod./Cons.
1	Piscivorous seabirds	0.250	0.163	120	*0.0*	*0.001*
2	Non-piscivorous seabirds	0.121	0.511	120	*0.0*	*0.004*
3	Weakfish	*3.969288*	0.26	3	0.95	0.08666667
4	Striped bass	*1.383361*	0.4	2.4	0.9	0.1666667
5	Summer flounder	*2.300436*	0.52	2.6	*0.95*	0.2
6	Bluefish	*2.732983*	0.52	3.1	0.95	0.1677419
7	Winter flounder	*4.66109*	0.52	3.4	0.95	0.1529412
8	Atlantic silversides	*4.461261*	0.8	*4*	0.95	0.2
9	Atlantic croaker	*0.1797899*	0.916	4.2	0.9	0.2180952
10	Spot	*0.5610008*	0.9	6.2	0.9	0.1451613
11	Atlantic menhaden	*12.42741*	0.5	31.42	0.95	0.01591343
12	River herring	*1.180491*	0.75	8.4	0.95	0.08928572
13	Mummichog	*3.464921*	1.2	3.65	0.95	0.3287671
14	Bay anchovy	4.86	3	9.7	*0.8390257*	0.3092784
15	Benthic infauna/epifauna	*77.42549*	2	*10*	0.9	0.2
16	Amphipods	*3.323004*	3.8	*19*	0.9	0.2
17	Blue crabs	*6.256972*	1.21	4	0.95	0.3025
18	Hard clams	26.18	0.5	5.1	*0.6292655*	0.09803922
19	Oyster	0.001	0.63	2	*0*	0.315
20	Copepods	*12.4849*	25	*83.33333*	0.95	0.3
21	Microzooplankton	*6.694112*	140	*350*	0.95	0.4
22	Sea nettles	0.345	13	20	*0*	0.65
23	Ctenophores	5.29	16.2	35	*0.04227053*	0.4628572
24	Benthic algae	*3.532623*	80	0	*0.8999999*	
25	Phytoplantkon	*21.27254*	160	0	0.95	
26	SAV	5.82	5.11	0	*0.3168661*	
27	Detritus	1			*0.1317724*	

2.2.3 Harvest Information

Harvest data for recreationally and commercially important species can also be incorporated into the EcoPath model as the landings (t/km^2/year) for the year in which the model is initiated. Landings can be entered as a fishery type (*i.e.* trawlers, recreational hook and line), as species specific landings across all fisheries, or a combination, depending upon the questions of interest. The Barnegat Bay model includes gear specific landings for the blue crab fishery provided by the NJ Bureau of Marine Fisheries and species specific landings for other fish and invertebrates, which combines recreational and commercial landings as recorded by the National Marine Fisheries Service. To assess the impacts of the Oyster Creek Nuclear Generating Station (OCNGS) on the biota of Barnegat Bay we also included a "fishery" that takes into account fish mortality due to the use of bay water for cooling the power plant (Amergen 2008). Because the mortality caused by OCNGS is not removed from the system, as fishery landings would be, we modeled it as discards that flow into the detrital pool. The landings values included in the model are provided in Table 2, and details on their derivations can be found in Appendix 3.

Table 2: Landings values used in the 1981 Barnegat Bay Ecopath model. All values are in tons/km^2/yr. Sources and calculations can be found in Appendix 3. * OCNGS mortality was modeled as a discard rather than landing; with biomass contributing to the detrital pool.

Group name	crab - recreational	crab pot and trap	crab winter dredge	commercial clam	OCNGS*	weakfish	striped bass
Piscivorous seabirds							
Non-piscivorous seabirds							
Weakfish					0.026182	0.01208	
Striped bass							0.0931
Summer flounder					0.001699		
Bluefish					2.15E-05		
Winter flounder					0.007052		
Atlantic silversides					0.024835		
Atlantic Croaker					0.013108		
Spot							
Atlantic Menhaden					0.057949		
River herring					0.000742		
Mummichog					7.00E-07		
Bay anchovy					0.011175		
Benthic infauna/epifauna							
Amphipods							
Blue crabs	0.634767	0.656989	0.136559		0.011571		
Hard clams				0.39			
Oyster							
Copepods							
Microzooplankton							
Sea nettles							
Ctenophores							
Benthic algae							
Phytoplantkon							
SAV							
Detritus							
Sum	0.634767	0.656989	0.136559	0.8129	0.154333	0.01208	0.0931

Table 2 cont'd: Landings values used in the 1981 Barnegat Bay Ecopath model. All values are in tons/km^2/yr. Sources and calculations can be found in Appendix 3.								
Group name	summer flounder	bluefish	winter flounder	croaker	spot	menhaden	river herring	Total
Piscivorous seabirds								
Non-piscivorous seabirds								
Weakfish								0.038262
Striped bass								0.0931
Summer flounder	0.804717							0.806416
Bluefish		0.750072						0.750094
Winter flounder			0.9253					0.932352
Atlantic silversides								0.024835
Atlantic Croaker				0.0001				0.013108
Spot					0.00398			0.00398
Atlantic Menhaden						0.000716		0.058665
River herring							0.000358	0.0011
Mummichog								7.00E-07
Bay anchovy								0.011175
Benthic infauna/epifauna								0
Amphipods								0
Blue crabs								1.439886
Hard clams								0.8129
Oyster								0
Copepods								0
Microzooplankton								0
Sea nettles								1.38
Ctenophores								0
Benthic algae								0
Phytoplantkon								0
SAV								0
Detritus								0
Sum	0.804717	0.750072	0.9253	0	0.00398	0.000716	0.000358	6.365871

2.2.4 Parameter uncertainty

The EwE software allows for the evaluation of the effects of parameter uncertainty through the use of Monte Carlo estimations in the time-dynamic module. In order to utilize this feature, estimates of uncertainty for each input parameter are required. This is accomplished through the use of a pedigree routine, whereby each input value is given a "confidence level" characterized by its origin. The highest pedigree (lowest uncertainty) is given to sampling based high precision values (+/- 10%), followed in decreasing order by sampling based low precision (+/- 30%), approximate or indirect method (+/- 50%), from other model (+/-

60%), best professional judgement (+/- 70%), and estimated internally by Ecopath (+/- 80%). Table 3 provides a graphical representation of the pedigrees for each of the input variables.

Table 3: Data pedigree representing “confidence” levels for the Ecopath input parameters. Red represents “sampling based high precision”, orange represents “sampling based low precision”, green represents “approximate or indirect method or empirical relationship”, dark blue represents “best professional judgement”, brown represents “from other model”, and black represents “estimated by Ecopath”.

Group name	Biomass (t/km^2)	P/B (year^{-1})	Q/B (year^{-1})	Diet
Piscivorous seabirds				
Non-piscivorous seabirds				
Weakfish				
Striped bass				
Summer flounder				
Bluefish				
Winter flounder				
Atlantic silversides				
Atlantic croaker				
Spot				
Atlantic menhaden				
River herring				
Mummichog				
Bay anchovy				
Benthic infauna/epifauna				
Amphipods				
Blue crabs				
Hard clams				
Oyster				
Copepods				
Microzooplankton				
Sea nettles				
Ctenophores				
Benthic algae				
Phytoplantkon				
SAV				

2.3 Ecosim time series data

Once the Ecopath model has been balanced, the mass-balanced linear equations 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 (Christensen and Walters, 2004). Model runs are compared with time-series data and the closest fit is chosen to represent the system. Time-series data for model calibration are thus essential for developing and validating an Ecosim model (Christensen *et al.* 2009). 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.

2.3.1 Fishery dependent time-series

Fishery dependent time-series data was used to “drive” changes in the Ecosim module. To use a time-series to force the model, data is needed for each year in the series, there can be no gaps. Because of this restriction there are a limited number of datasets available for Barnegat Bay. A time-series of commercial and recreational finfish landings collected by the NMFS for New Jersey was reduced through a series of gear and location filters to approximate landings for Barnegat Bay as those data are not collected at the estuary level (see Appendix 4 for the process). Many other ecosystem models glean data from formal stock assessments, which utilize similar time series data for single species management plans. Unfortunately, there are no stock assessments specific to the Barnegat Bay. While there is no formal stock assessment for blue crab, the NJDEP does collect commercial blue crab landings data by gear and location, and this was used to create gear specific time series which were converted to effort and used to force the model.

While hard clams have historically been an important resource within Barnegat Bay, New Jersey has not collected commercial or recreational landings data for the fishery. Therefore, no time-series for this group was developed. The final source of Barnegat Bay specific fishery dependent time series data comes from OCNGS. Because of the nature of OCNGS operations, the cooling and dilution intake structures function as an on/off type activity, with the only shutdowns associated with temporary, short term maintenance. As such the plant flow has been fairly consistent over the timeframe in question (1981-2013), and therefore the impacts of the plant have been modeled as a steady forced effort.

2.3.2 Fishery independent time-series

Fishery independent time-series were used to assess how well the model fit “real-world” patterns in the data. Again, there are limited repetitive assessments of biota specific to Barnegat Bay; however to assess model fit it is not necessary to have records for each year in the time series. Thus, we used a combination of surveys and assessments, spanning a variety of timeframes, to determine how well our model reflects changes in the ecosystem (Appendix 4). For the finfish, we used a long-term (1988-2013, except 1991-1995) otter trawl data set from the Rutgers Marine Field Station that includes five regularly sampled sites located in Little Egg Harbor (Vasslides *et al.* 2011). A shorter (2012-2013) time-series utilizing the same survey methodology was available estuary-wide and was also incorporated into the model. The CPUEs generated from this data were entered as an index of relative biomass for a number fish species, as well as blue crab. While data on mummichog are available through this dataset, we have chosen not use it in this analysis as they are a marsh species that are not well sampled by the gear.

An additional source of fish time-series data incorporated into the model is an index of biomass generated from the near-shore trawl surveys conducted each fall by the NJDEP. While sampling for this survey occurs along the entire New Jersey coast, it provides an estimate of relative biomass in each year for those species that leave the estuary each fall for offshore or southern waters. There are some species collected in the trawl, though, for which a coast-wide sample may not be appropriate to include in the model. For instance, in an initial run of the simulation the model fit for spot in the coast-wide dataset represented almost 30% of the total sum of squares value. Because spot abundance in the Barnegat Bay is typically very low, and the fit to the Barnegat Bay specific sampling was substantially more robust, we elected to discount the coast-wide data for this species.

In addition to the fish and crab data referenced above, hard clam surveys were conducted by the NJDEP in 1986/1987 in Barnegat Bay and Little Egg Harbor, 2001 in Little Egg Harbor, 2011 in Little Egg Harbor, and 2012 in Barnegat Bay (Celestino 2002, Celestino 2013, Dacanay 2015). The 1986/1987, 2001, and 2011 data for Little Egg Harbor have been incorporated into the model. While not offering estuary-wide coverage, these data were selected as 2001 was an apparent low in the biomass, with a rebound in population size in 2011. If the Little Egg Harbor and Barnegat Bay data are combined, necessitating the dropping of the 2001 Little Egg Harbor data, the time-series would consist of only 2 points showing a constant, but less steep, decline through time.

Invertebrates that are not commercially harvested within Barnegat Bay have not traditionally received as much attention as the higher trophic level taxa, and thus useable time-series data for these groups was primarily limited to recent studies funded by the NJDEP as part of the Barnegat Bay Initiative. Taghon et al. (2012, 2013) sampled at 100 locations throughout the Barnegat Bay in July of 2012 and 2013 for benthic infauna and epifauna. An index of the average number of individuals found in a 0.04 m^2 grab on or near the sediment surface was developed for the amphipod stanza (Order Amphipoda) and the benthic infauna and epifauna stanza (all other taxa excluding amphipods, blue claw crabs, and hard clams). A time-series for the copepod stanza was developed from samples collected from June to November of 2012 and 2013 at 3 locations (northern, central, and southern) within Barnegat Bay using paired 200 *u*m plankton nets (Howson et al. 2014). The data are the average yearly CPUE for the major copepod fauna found in Barnegat Bay (n=57). The same sampling schema was used to develop a time-series for the microzooplankton stanza, though this was limited to the average yearly CPUE for foraminifera, the only microzooplankton identified.

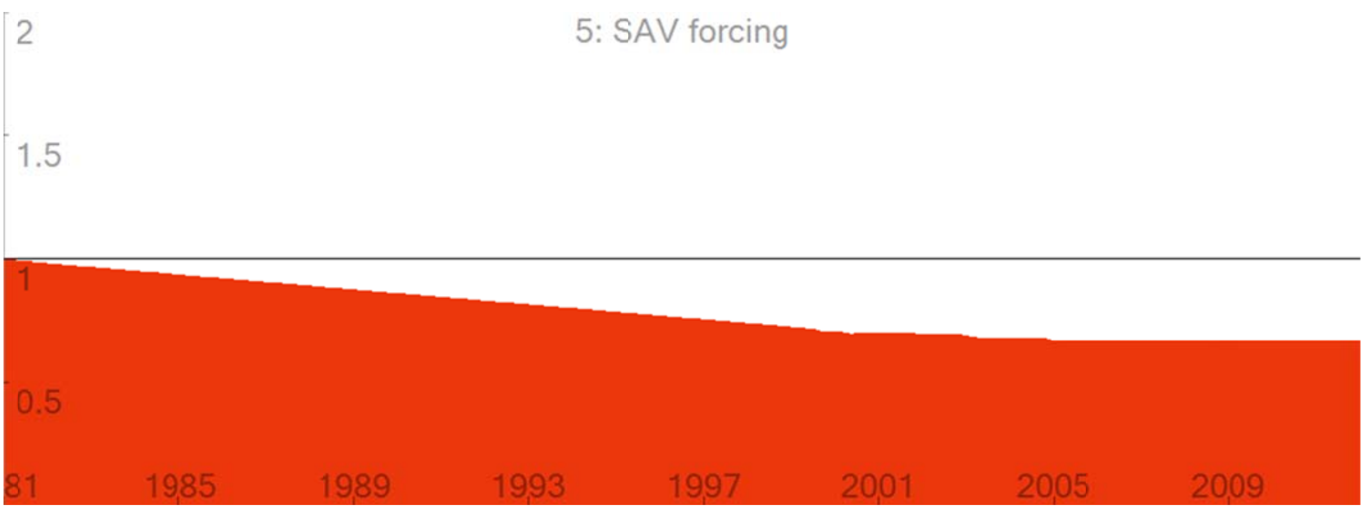
Developing a time series for sea nettles within Barnegat Bay proved to be particularly difficult. There is limited data available in the scientific literature, with only passing references to occasional collections. This lack of scientific reporting continued even during times of anecdotally high abundances up until the 2010’s. Recent work by Young et al. (in review) using local ecological knowledge suggests that sea nettle abundances were at very low levels (unrecorded in contemporary research) until the late 1990’s, with a large increase occurring around 2007. To simulate this bloom pattern, we forced a low biomass until the mid-1990s, and then a progressive increase leading to current (2013) population estimates beginning in 2007. The population estimate is based on spring/summer/fall sampling using plankton and lift nets conducted as part of the Barnegat Bay Initiative (Bologna *et al.* 2014, Howson *et al*. 2014).

The only consistent time series directly available for primary producers is for submerged aquatic vegetation. SAV coverage for the bay is available for 1980, 1987, 1999, 2003, and 2009 based on aerial photograph analysis in Lathrop *et al.* (2001) and Lathrop and Haag (2011). The acreage of seagrass in each year serves as a data point of relative abundance, though this method can mask declines in overall biomass due to changes in density or condition. A shorter time series (2008-2013) of relative abundance of phytoplankton bay-wide was estimated from chlorophyll *a* readings taken via aircraft remote sensing collected six days a week from March through October each year.

2.3.3 Forcing functions

Forcing functions represent physical or other environmental parameters that may influence trophic interactions, but are not directly included in the Ecopath parameters (Christensen and Walters 2004). These forcing functions can be used to directly influence primary production or modify the Q/B ratio of consumer groups. For the Barnegat Bay Ecosystem Model we created a forcing function to influence the biomass of SAV based on known changes in SAV coverage in the bay (Figure 2). No other forcing functions were applied.

Figure 2: The forcing function developed for SAV based on known relative abundance of SAV.



2.4 Scenario modeling

Once the Ecosim scenario is fit to the available time-series data, the model can be extended beyond the current timeframe (1981-2013) to make predictions about the future state of the ecosystem under different management strategies. We received input from scientists and managers within the Department regarding what change scenarios they would be most interested in. At this time we were able to create scenarios for the closure of Oyster Creek Nuclear Generating Station, changes in blue crab harvest, and changes in hard clam harvest. For all time-series forcing data (catches, sea nettle biomass), the 2013 values were extended to 2030, with the exception of the management actions being investigated.

2.4.1 Oyster Creek Nuclear Generating Station (OCNGS) closure

As America's oldest continuously operating nuclear plant, the facility uses a once-through cooling water system, where water is drawn from Forked River, used to cool the plant, and is then returned to Oyster Creek to flow into the bay. The impingement and entrainment of fish, crab, and hard clam larvae, as well as other zooplankters, is well documented. The effects of OCNGS were included in the EwE model as catch data and an effort time-series. As part of the Governor's 10-point Plan, the Oyster Creek Nuclear Generating Station (OCNGS) will cease power generation by 2020. To model this scenario we extended the time-series so that the effort remains constant from 1981-2019, and then reduces to 4% of the full operating capacity from 2020-2030.

2.4.2 Changes to blue crab harvest strategy

Blue crabs are Barnegat Bay's largest commercial fishery, and are currently managed based on a mix of sex and size limits and seasonal closures (NJAC7E:25 and 25A). We modeled the effects of increasing the commercial dredge harvest to 98 metric tons (twice the 1995-2013 average of 49 MT) and of decreasing the commercial dredge harvest to 25 MT (one-half the 1995-2013 average) from 2012 to 2030, while keeping the commercial pot fishery and recreation fisheries at their 1995-2011 averages and the other effort series at their 2013 values. We also modeled the effects of doubling the commercial pot fishery over the 1995-2013 average of 206 MT to 412 MT and of halving it to 103 MT.

2.4.3 Changes to hard clam management strategy

Hard clams were historically one of the most important commercial fisheries in the Bay, but anecdotal evidence suggests landings have declined dramatically over the past several decades. We will model the effects of limiting the commercial harvest effort to 10% of 1981 levels during the prediction period (2014-2030) and of closing the fishery entirely for a period of ten years (2014-2024) and then returning to the 10% level.

2.4.4 Nutrient reduction scenarios

Changes in nutrient loading can be incorporated in Ecosim through a nutrient loading forcing function. Like other forcing functions in Ecosim, this function specifies the change in the relative concentration of nutrients through time, and the shape of the function is specified by the user. The total nutrients are partitioned between primary producer biomass (in the case of this model: phytoplankton, benthic algae, and submerged aquatic vegetation (SAV)) and the pool of free nutrients in the environment. The primary production rates for each group are linked to the free nutrient concentrations via Michaelis-Menten uptake relationships, where the current P/B value depends on the maximum P/B value for that group (P/B_{max}) and the current available nutrient concentration. The P/B_{max} value sets the sensitivity of the primary producer group to nutrient levels, with a higher P/B_{max} value causing greater sensitivity to changes in nutrient concentration. The base proportion of free nutrients (N_f) can be used to increase the strength of nutrient limitation, with lower values causing greater competition among the primary producer groups.

P/B_{max} values for each of the primary producer groups were estimated from available Ecopath-Ecosim models of other coastal systems. Specifically, P/B_{max} values were estimated as the ratio of the highest P/B value for that group observed in any Ecopath-Ecosim model to the actual P/B value used in this model. P/B_{max} values were 1.5625, 1.76, and 2.0 for phytoplankton, SAV, and benthic algae, respectively.

The base proportion of free nutrients (N_f) was kept at its default value of 1.0, which assumes that all nutrients not bound in biomass are freely available for uptake by primary producers.

Two nutrient reduction scenarios were run: one with a 20% reduction and one with a 40% reduction occurring linearly from 2013 to 2018. A number of different approaches were attempted to assess the effects of changing nitrogen levels on primary production. In the first approach, nitrogen levels were assumed to be consistent from 1981 to 2013 and the SAV forcing function continued at its 2013 value through 2030. While the phytoplankton response was reasonable, SAV biomass decreased throughout the simulation as it was responding solely to the reduction in nutrients. The effects of improved light availability due to reductions in phytoplankton shading and decreased epiphyte loads, typically seen with reduced nutrients, were not present in the model. Further, nutrient levels fluctuated throughout the simulation period, and this was not taken into account.

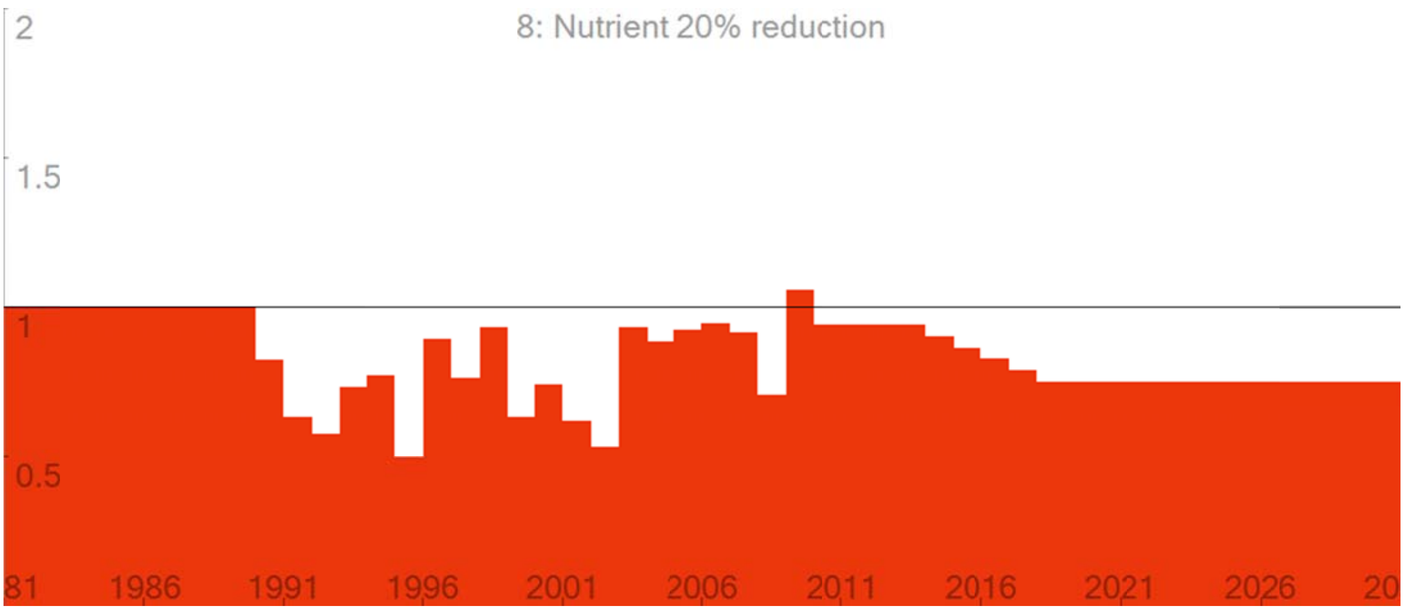
Upon further research, we were able to incorporate total nitrogen loads for Barnegat Bay for 1989-2011 as taken from Baker et al. (2014). Total nitrogen loads from 1981-1988 were set at the 1989 level and 2012-2013 set at the 2011 level for lack of other data. In both scenarios, the nutrient reduction occurred linearly from 2013 to 2018, at which point the target nutrient reduction was maintained for the remainder of the simulation (Figure 3a and b).

While the nutrient loads now more accurately reflected historic conditions, a forcing or mediation function was needed to relate the known response of SAV to changes in nutrients. A forcing function was developed for SAV based on the relationship between seagrass above-ground biomass and loading of total nitrogen found in Kennish et al. (2014). When applied throughout the simulation period, seagrass biomass fluctuated inversely to total nitrogen load, with the 2013 biomass greater than the initial (1981) biomass, a result we know to be inaccurate. In order to maintain the known decrease in SAV biomass during the 1981-2013 timeframe, yet capture the anticipated increase in SAV associated with a reduction in nutrients, we combined the two forcing functions into a single function. The empirical relationship was applied to 2014-2030, the time during which the nutrient reductions are to take place, while the forcing function developed for the 1981-2013 model (Figure 2) was used prior to 2014 (Figure 4a and b). Because the nutrient scenarios incorporated revised

P/B_{max} values for the primary producers throughout the entire time series, a new baseline scenario was also run, incorporating those same values, for comparison purposes. The 2013 value in the nutrient and SAV forcing functions was extended through 2030 for the baseline scenario.

Figure 3: The nutrient forcing functions showing a a) 20% and b) 40% reduction in nutrient loading after 2013. Data for 1989-2011 taken from Baker et al. 2014. The loading values for 1981-1988 are taken from 1989, and 2012-2013 were from 2012.

a)



b)

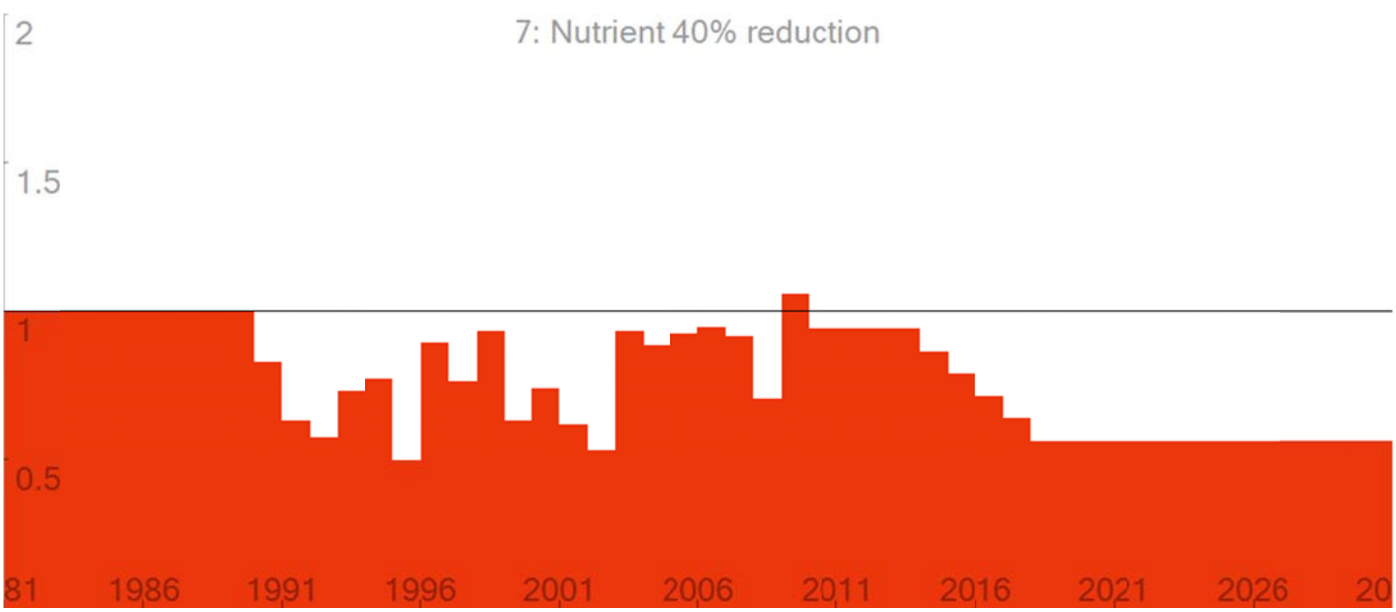
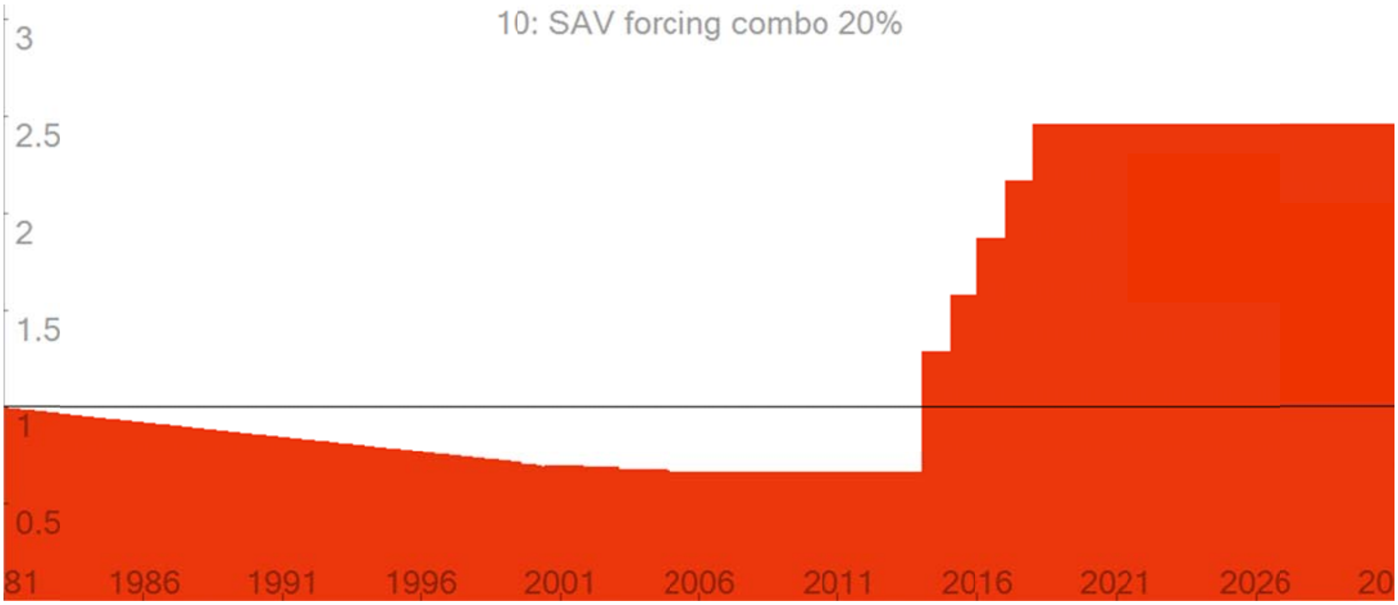
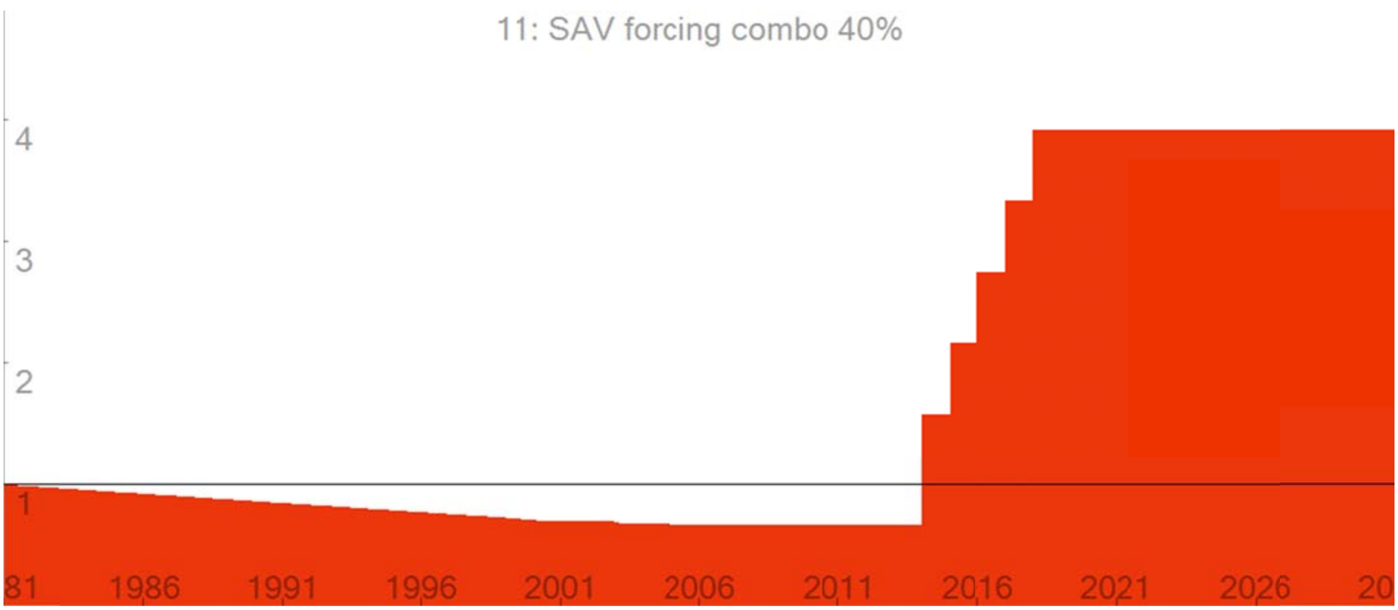


Figure 4: The SAV forcing functions developed for the a) 20% and b) 40% nutrient reduction scenarios to take place in 2014. The relationship between SAV above-ground biomass and Total Nitrogen loading are from Kennish et al 2014.

a)



b)



Chapter 3 – Results and discussion

3.1 Ecopath model

The Ecopath model for the Barnegat Bay contains 27 biomass groups, including 12 fish groups, 9 invertebrate groups, 3 primary producer groups, and 2 bird groups. The model shown in Figure 5 represents a possible configuration of Barnegat Bay for 1981, with the groups arranged by trophic level. The model is balanced, in that there is sufficient food for the consumers and enough production to meet consumptive demands. There are no surprises in the trophic level of any of the groups, though striped bass in our system do occupy a slightly higher level than those in the Chesapeake Bay. The fact that the structure of the model is consistent with other estuarine models from within this region (Chesapeake Bay, Delaware Bay, Narraganset Bay) lends additional support to its interpretation.

The Ecopath software includes packages for conducting ecological network analysis (ENA), based on the work of Ulanowicz (1986) and others. One of the indicators available is known as mixed trophic impacts, which is a routine that assess the effect that changes in the biomass of a group will have on the biomass of the other groups in a steady-state system through both direct and indirect means (Ulanowicz and Puccia 1990). As seen in Figure 6, most of the impacts appear to be directly related to feeding interactions, with few unexpected indirect impacts. There is also a routine to determine which group(s) occupies a keystone role in the ecosystem as described by Libralato *et al*. (2006). In this analysis, groups with an index value greater than 0 are considered keystone groups. For the Barnegat Bay Ecosystem Model, piscivorous seabirds (#1) and phytoplankton (#25) are keystone groups (Figure 7).

3.2 Ecosim simulations

When the time series data is incorporated into the model and the vulnerability values are adjusted to fit to the time series, the overall fit of the model prediction to the available data is reasonable, and the model generally behaves as expected (Figure 8, Sum of Squares = 497.2). Changes in an individual group's biomass over the course of the time period are shown in Figure 9. While the biomasses of all of the groups vary through time, several groups display dramatic increases or decreases. The 17-fold increase in the biomass of croaker through time is reflective of the increase in its overwintering survivability and general population increase in

Figure 5: Barnegat Bay Ecosystem Model for 1981. Numbered horizontal lines indicate trophic level. The size of the circle indicates relative biomass, while the thickness of the lines indicates the relative amount of energy flow from one group to another.

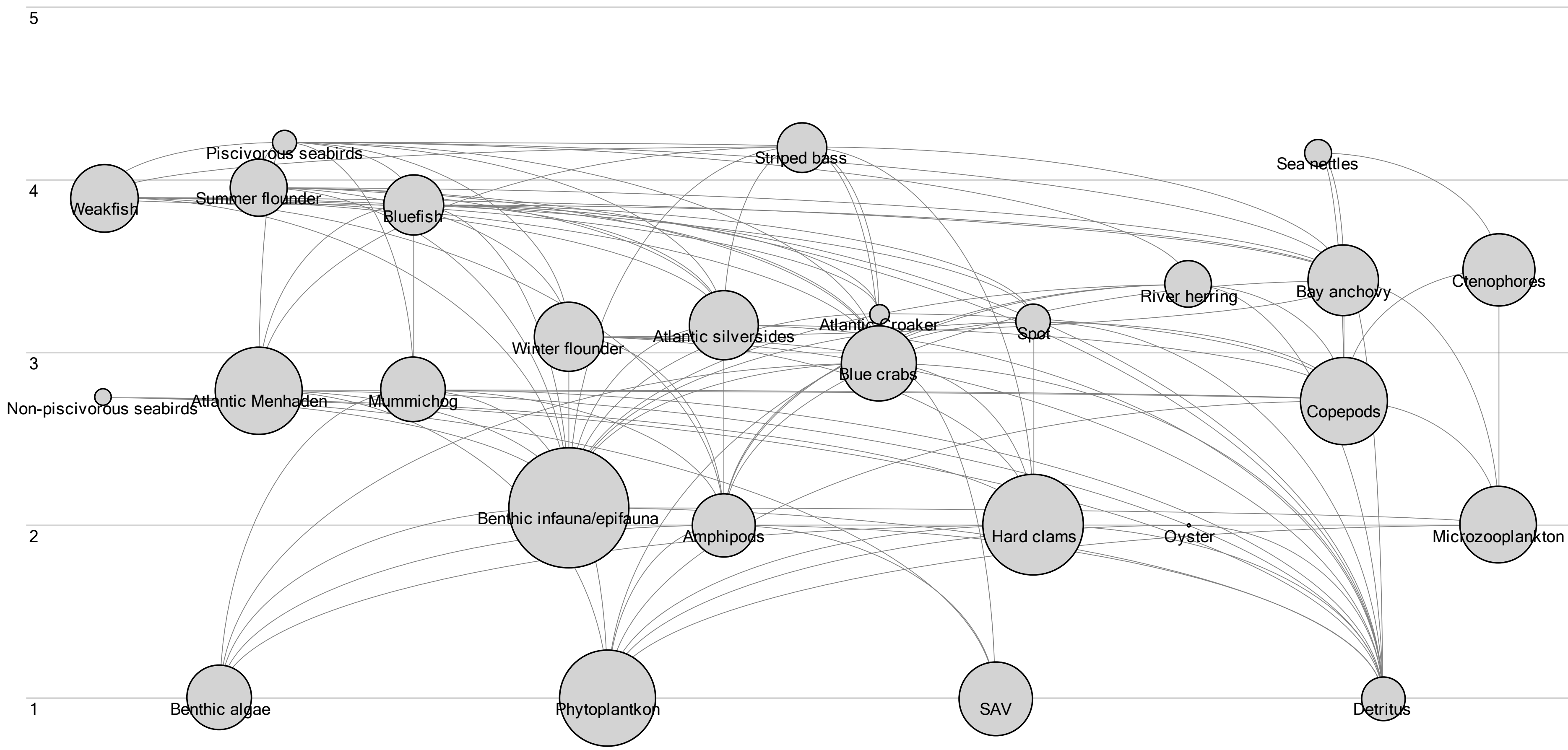


Figure 6: Mixed Trophic Impact analysis for the Barnegat Bay Ecosystem Model.

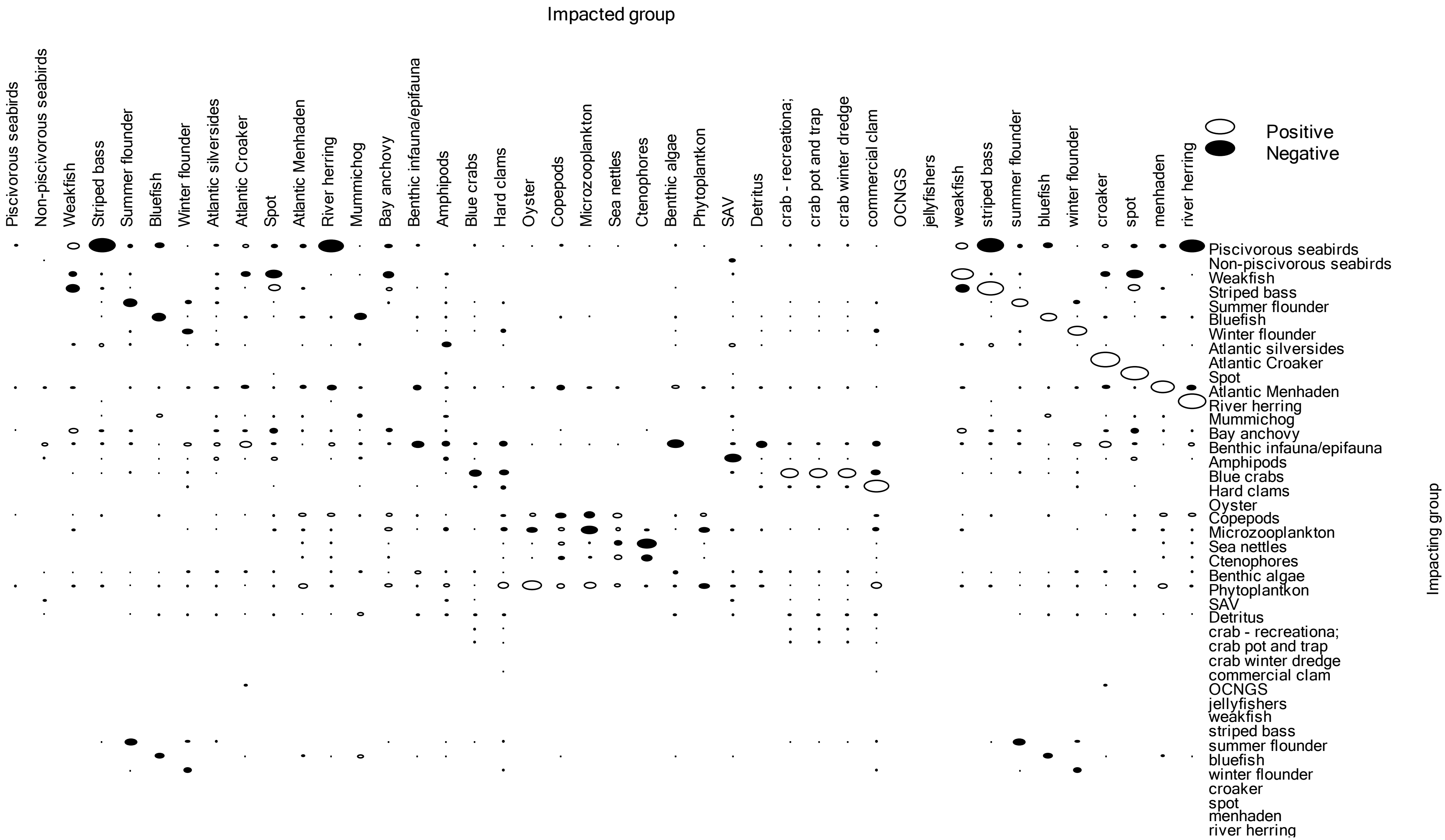
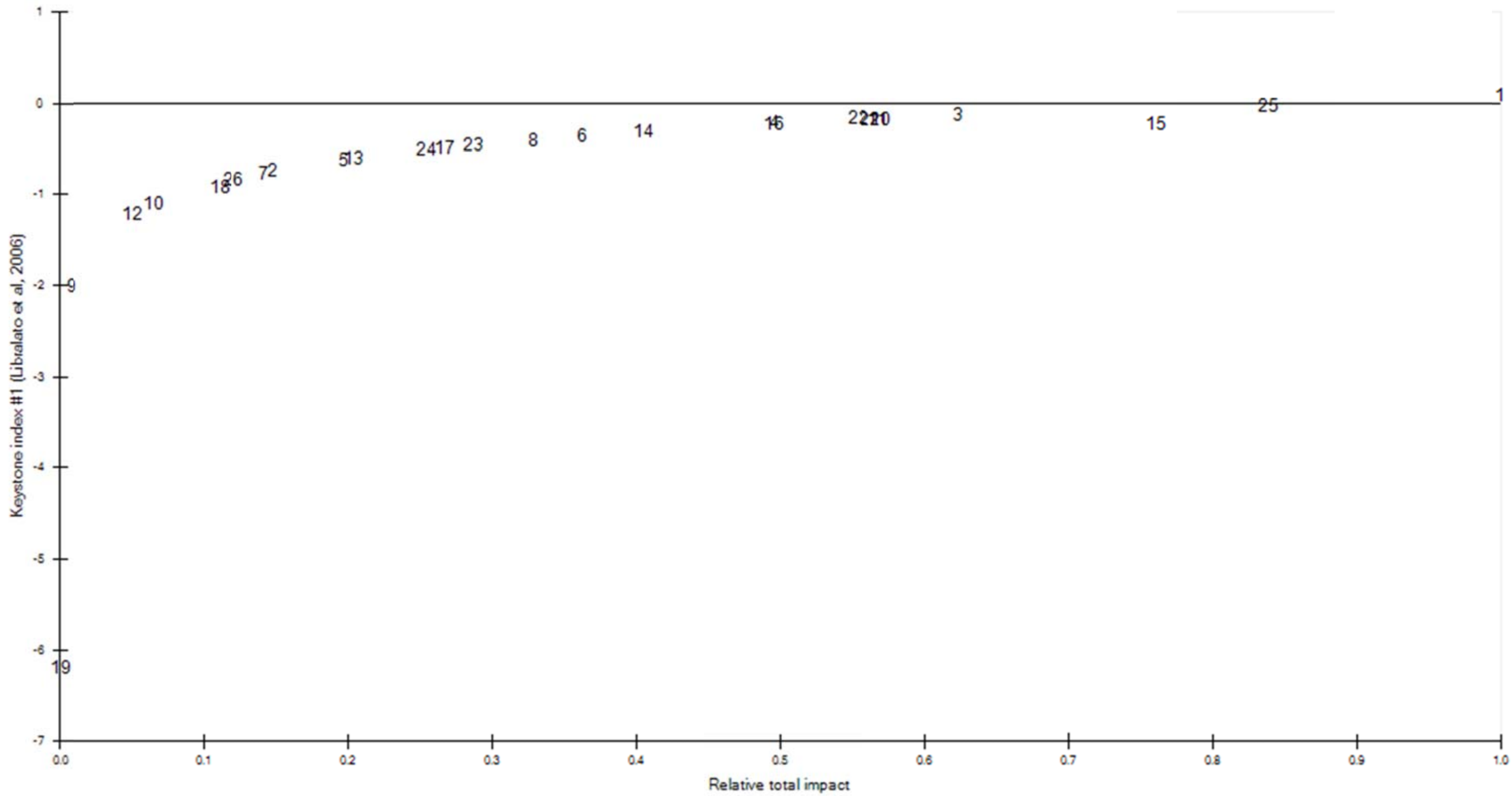


Figure 7: Keystone index for the Barnegat Bay Ecosystem Model. Numbers on the figure refer to the group number provided in Table 1. Groups with an index value greater than 0 (#1, piscivorous seabirds; #25, phytoplankton) are considered keystone groups.



the Mid-Atlantic as documented by Hare and Able (2007). Conversely, blue crab undergo a dramatic decrease, ending the period at 0.62 t/km^2, or approximately 9.9% of their starting biomass.

There is variability in how well the predicted biomass trends match the available time-series data among the groups (Figure 10). In general, the model fits the Barnegat Bay specific data better than the coastwide data, though some of that is explained by the fewer number of datapoints in the Barnegat Bay time-series. However, for winter flounder and summer flounder both time-series are well replicated. The increase in Atlantic croaker and decrease in blue crab populations seen in the time-series are also well captured by the model. In contrast, the decline in hard clams that occurred during the early part of the time period is not at all captured in the model. This poor fit for hard clam, as well as that of unfished or lightly fished groups (bay anchovy, Atlantic silverside, menhaden) suggests that there are ecosystem drivers other than fishing or consumption by predators that are influencing biomass and are not included in the model. One of the most important non-fishing drivers that can be represented in the EwE software is changes in primary productivity through time. This is generally represented in the model by a time series of phytoplankton biomass or some other measure of abundance. As seen in Figure 9, the biomass of phytoplankton does not vary much over the course of the model run, and this variation is due to consumption by predators rather than changes in environmental conditions. This is further reinforced by the poor fit to the limited time series information we have (Figure 10). We attempted to find a proxy for a phytoplankton time-series by comparing the phytoplankton and nutrient data available (sporadic NJDEP sampling during 1990-2011) to river discharge (USGS Toms River gauging station), which has been collected on a regular basis throughout the time period. Unfortunately we were unable to find a correlation in either same year or time-lagged comparisons.

When we ran 100 Monte-Carlo simulations utilizing the pedigree values set during the Ecopath model construction the current model was the best-fit, suggesting our model is robust to changes in the input parameters. For the remainder of the trials the biomass trends were similar, though the relative abundance varied between simulations.

Figure 8: Model predictions of relative biomass versus time series data for 1981 through 2011.

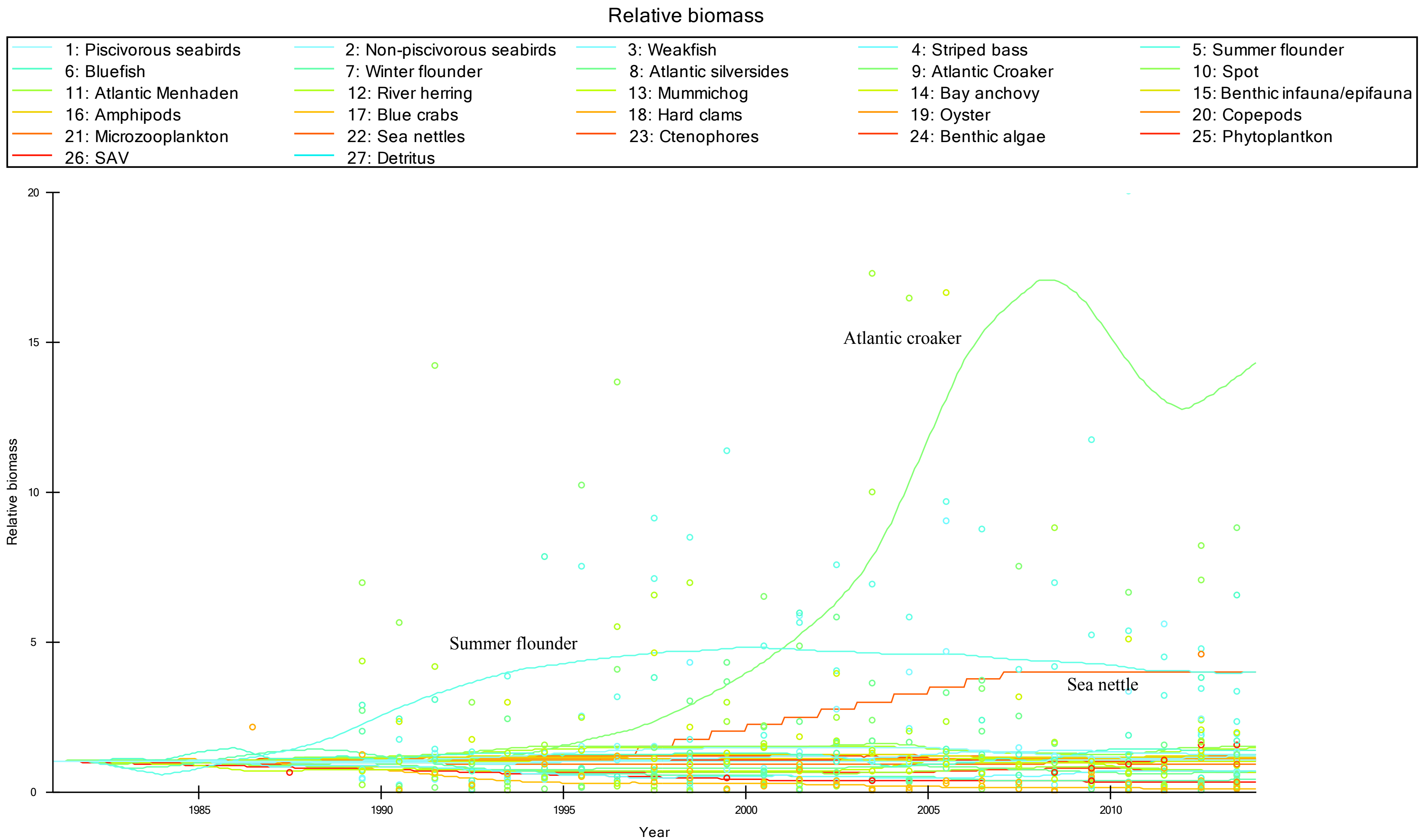
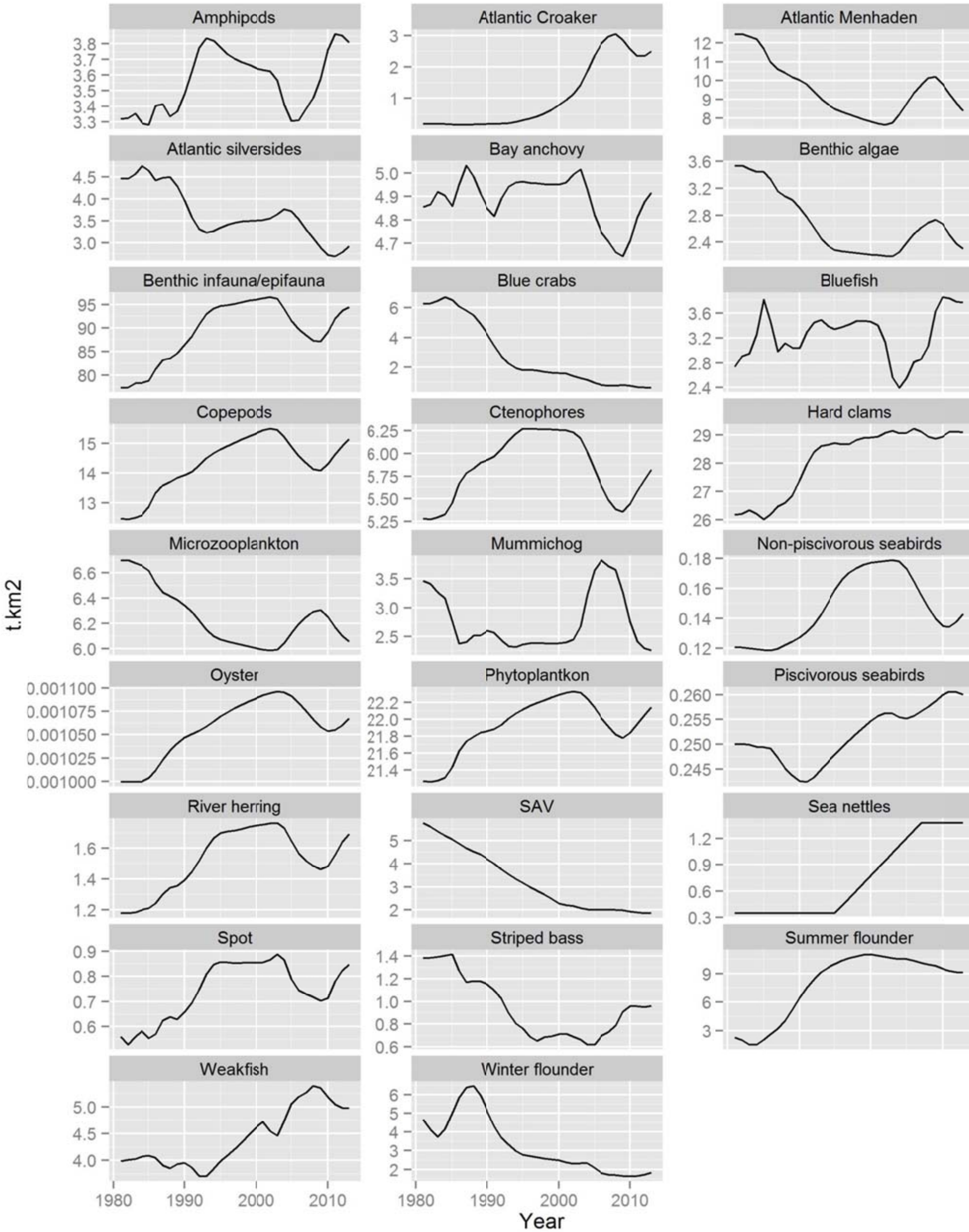
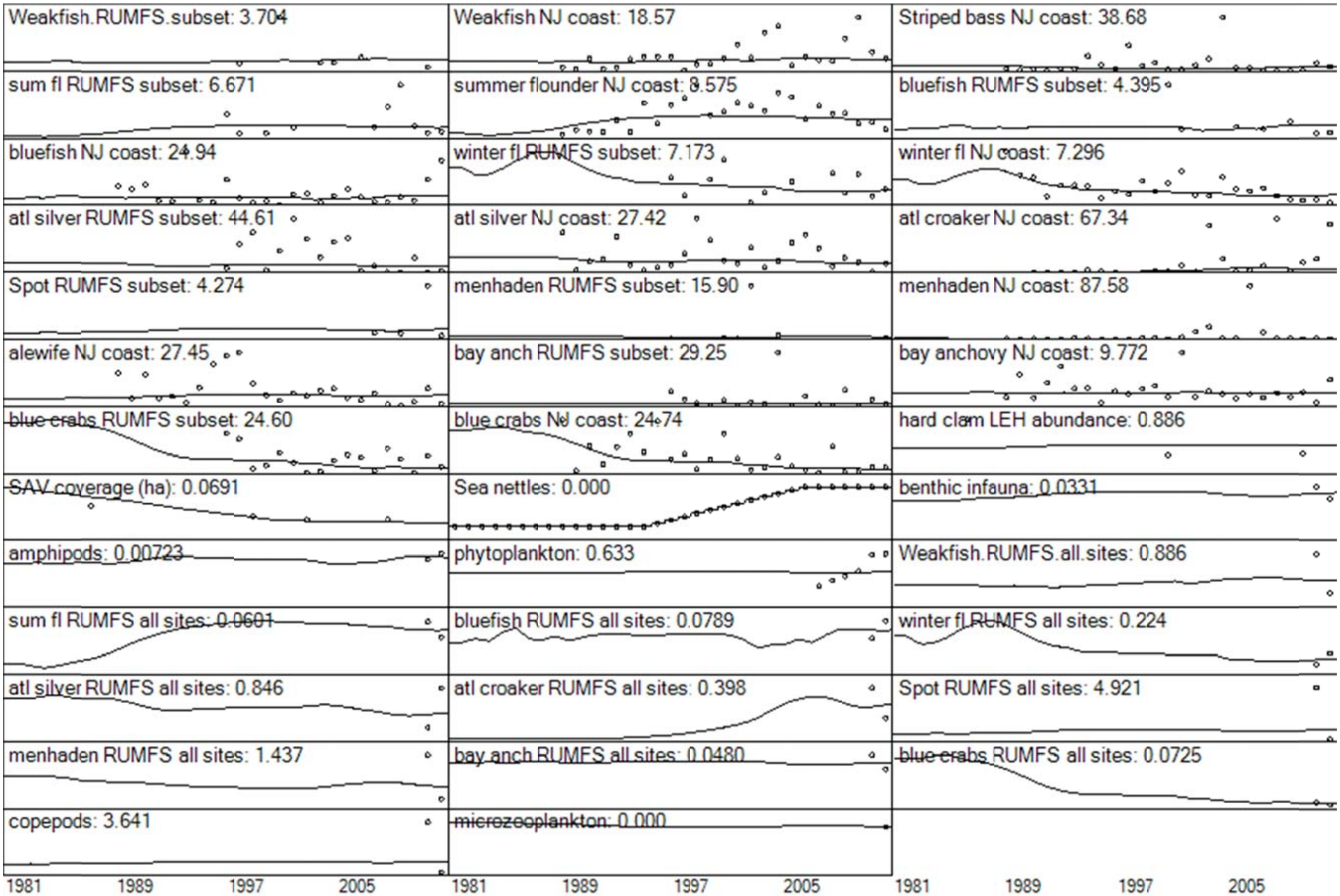


Figure 9: Changes in a group's biomass through time as predicted by the model. Note the different scales along the y-axis for each group.



This page intentionally blank

Figure 10: Graphs of the model fit to time series data. Numbers in each box represent the Sum of Squares contribution.



3.3 Management scenario modeling results

When the Ecosim scenario developed for 1981-2013 is continued to 2030 using the 2013 values for the time-series forcing data, most of the groups continue along at their 2013 biomass with slight increases or decreases (Figure 11). The major exception is Atlantic croaker, which continues to increase to 5.53 t/km^2. This is due to the high vulnerability setting needed to rapidly increase croaker biomass in the middle of the time series to meet the coast wide trawl data. A reduction in the vulnerability will lower the overall croaker abundance and flatten the curve, but increase the SS score as it does not fit the coast wide survey as well.

3.3.1 Closure of OCNGS

Reducing the "effort" of OCNGS by 96% in 2020 leads to small changes (<3%) in the biomass of the groups by the end of the simulation in 2030, with weakfish and blue crabs experiencing the largest increase and Atlantic croaker the largest decrease (Figure 12). Of the groups directly impacted by OCNGS, Atlantic croaker has the greatest response to the reduction in mortality associated with the plant closure, decreasing by nearly 2.5% compared to the baseline simulation. While this seems counterintuitive, it makes sense in light of the reduced mortality to croaker's main predator, weakfish, which has a positive response to the closure (2.1% increase). Thus, the increase in predation more than offsets the reduced mortality. The other group with a greater than 1% change in biomass is blue crab, which realizes a 1.5% increase in biomass due to a decrease in mortality associated with OCNGS.

3.3.2 Changes to blue crab harvest strategy

Modifying the level of effort for each gear type led to a variety of outcomes, from a near depletion of crab biomass by doubling the pot fishery to a near doubling of the baseline biomass by reducing both the winter dredge and pot fisheries (Figure 13). Under the baseline scenario the crab population remains nearly steady. Because of the historic differences in the sizes of the fisheries, doubling the pot fishery leads to a dramatic decline in biomass (84%), while doubling the dredge fishery leads to a much smaller (9%) decline. However, halving either fishery leads to a similar (43%) increase in biomass by the end of the simulation. Halving both fisheries leads to an 85% increase in biomass.

Figure 11: Model predictions of relative biomass through time for 1981 through 2030.

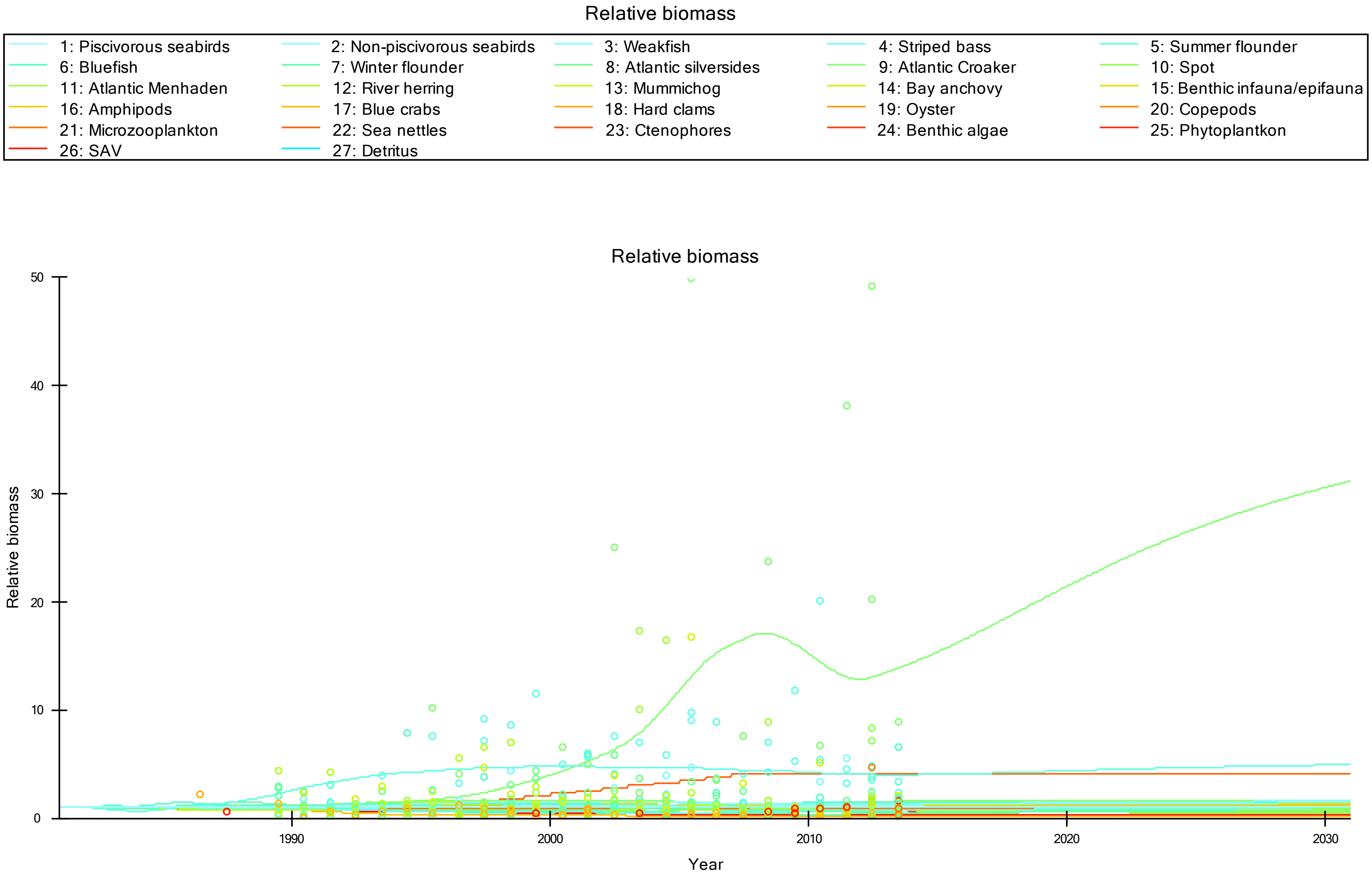


Figure 12: Percent change in biomass between the OCNGS closure simulation and the baseline simulation for 2030.

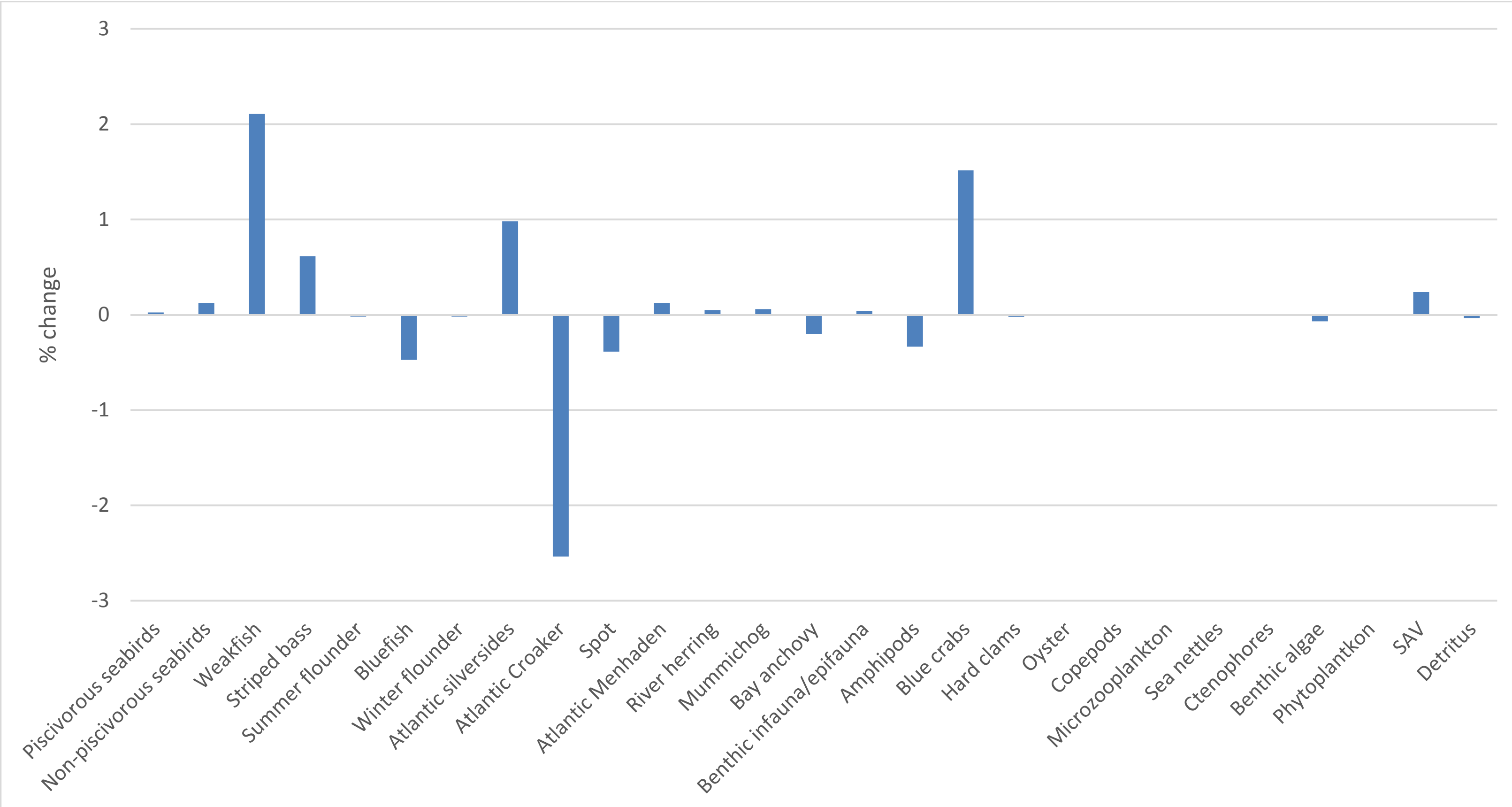
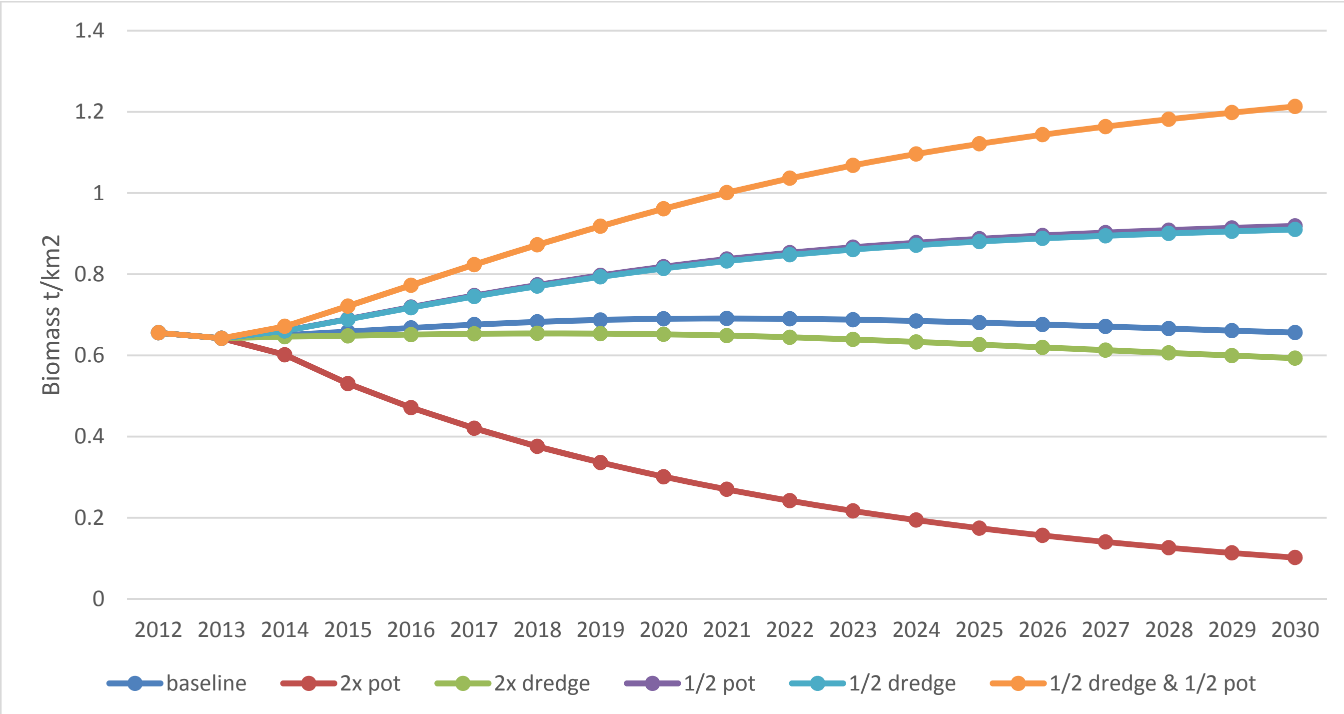


Figure 13: Change in the biomass (t/km^2) of blue crab under various harvest strategies beginning in 2014.



3.3.3 Changes to the hard clam management strategy

Under the baseline scenario the commercial harvest effort falls to below 5% of the 1981 level in 1999 and remains relatively steady at that rate for the remainder of the simulation. This equates to a catch of approximately 6 metric tons per year. A ten-year moratorium (2014-2024) on commercial harvest followed by six years of harvest at 5% effort resulted in a negligible decrease in biomass (0.00109 MT) compared to the baseline. The lack of biomass response to changes in harvest is not surprising given the low rate of fishing mortality post-2000 (F~ 0.001) compared to overall mortality (Z~ 0.45). As discussed in Section 3.2, the current model does not fit known hard clam population dynamics well, likely due to the misspecification of harvest (or lack of harvest information), the importance of environmental factors, and non-trophic related drivers. Furthermore, the relative paucity of survey data on hard clam biomasses through time, as well as the total lack of landings data, makes evaluating management strategies for this fishery problematic at this time.

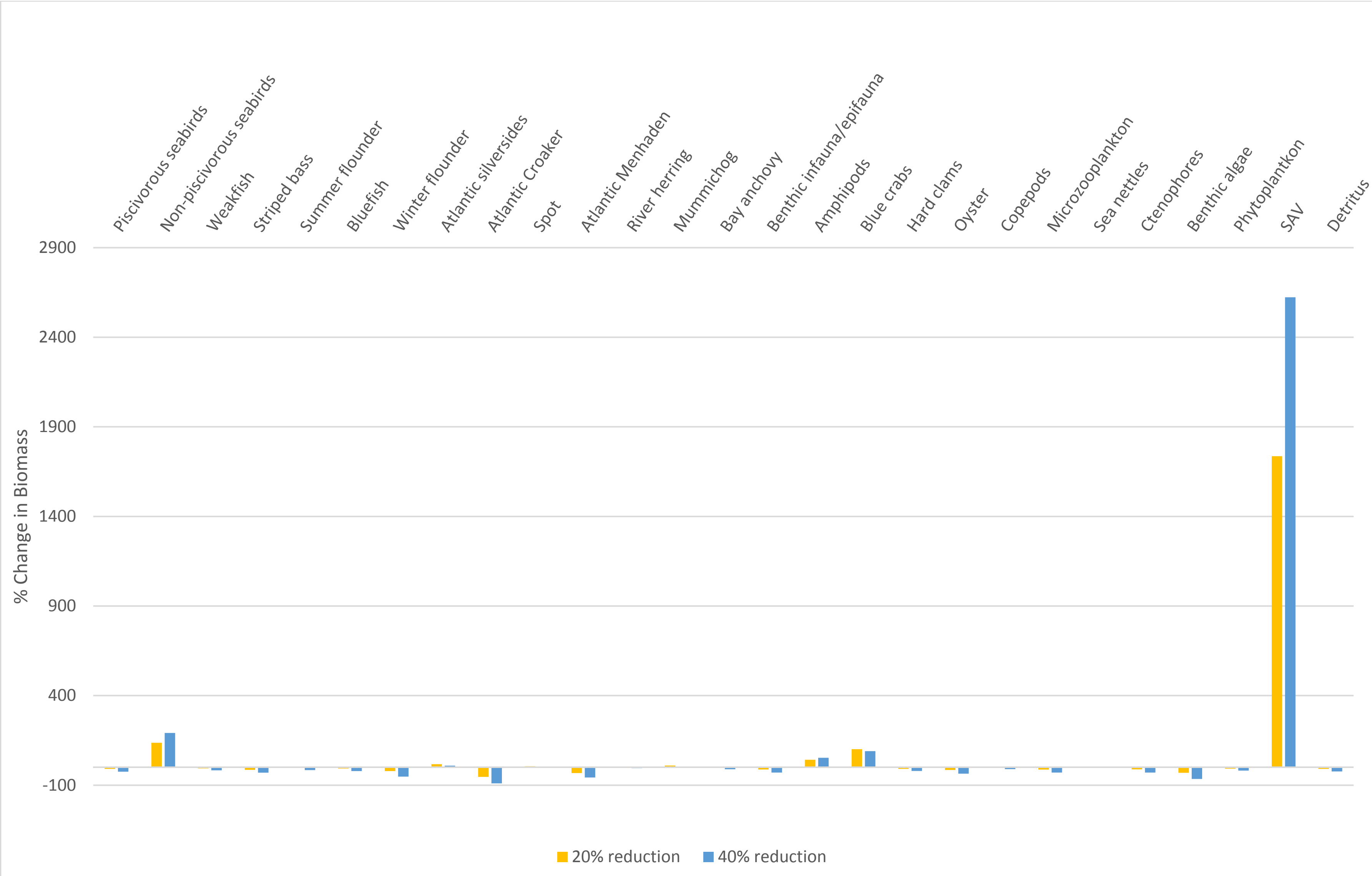
3.3.4 *Nutrient Reduction Scenarios*

Both nutrient reduction scenarios lead to lower total system biomass at the end of the simulation period (20% reduction = 228.08 t/km^2, 40% reduction = 204.21 t/km^2) compared to the baseline scenario (230.82 t/km^2). Among the primary producers, submerged aquatic vegetation was positively affected by the reduced nutrient load (>1000%), while phytoplankton (7%-18%) and benthic algae (31%-65%) both decreased in biomass (Figure 14). Non-piscivorous seabirds (>100%), blue crabs (100%), and amphipods (>40%) all saw increases in biomass. Most other groups had modest declines in biomass of less than 20%, though Atlantic croaker (53% - 90%) and Atlantic menhaden (32%-57%) had slightly larger reductions.

The rapid recovery of SAV under these scenarios is based on data from a study that investigated historic levels of seagrass in Barnegat Bay relative to the Total Nitrogen load at the same time (Kennish *et al.* 2014, Figure 3-17). Thus it is an estimate of what the seagrass aboveground biomass *could be* if other factors that limit seagrass (temperature, TSS, etc.) were favorable. While no shading effects due to phytoplankton, epiphytes, or microalgae were explicitly defined in this model, these impacts are implicit in the SAV-TN relationship as it is based on field measurements.

The reductions to finfish and other consumers observed in the nutrient reduction scenarios are the result of bottom-up trophic effects that start with lower phytoplankton productivity in response to lower nutrient availability. This is most clearly demonstrated by the reduction in Atlantic menhaden, a filter feeding species that depends heavily on phytoplankton. Detritivores, such as mummichog and blue crabs, benefited from an increased flow to the detrital pool associated with the increase in SAV.

Figure 14: Percent change in biomass for each group compared to the baseline scenario for the 20% (yellow) and 40% (blue) nutrient reduction scenarios.



Chapter 4 – Ecopath with Ecosim Conclusions

4.1 Model structure and data availability

The Barnegat Bay Ecosystem Model developed here includes 27 of the most important species and functional groups found within this complex estuarine ecosystem. While all attempts were made to make the model as comprehensive as possible, it is not without its limitations, both structurally and functionally. Most of these limitations are driven by a shortage of system-specific data. This was recognized at the beginning of model development, and the timeframe selected for model construction and fitting was a balance between when Barnegat Bay specific data was available and the desire to have as long a time-series as possible to fit the model to.

The limited availability of Barnegat Bay specific data led to compromises in the overall structure of the model. For instances, all of the biomass groups within the model are represented by a single age stanza. For many of the species/groups this does not represent an issue as their importance as a trophic link does not depend on life history, *i.e.* phytoplankton, zooplankton, benthic invertebrates, SAV. However, for species for which we wish to investigate management actions (blue crabs) or where there may be ontogenic shifts in diet preferences (weakfish, striped bass, flounders) age-structured stanzas provide increased resolution into the interactions in question. Of course, this increased level of resolution requires ever increasing amounts of data to populate the input parameters. However, the current single stanzas model appears to capture the overall trends in biomass (where available) reasonably well, and is thus useful for investigating questions of ecosystem functioning and exploring scenario development.

The scarcity of Barnegat Bay specific data was particularly acute when it came to biomass and catch information. The initial biomass of most groups had to be estimated by the Ecopath module due to a lack of historic information. While the biomasses generated appear reasonable, they are underpinned by only five field researched values. While alternate methods of developing biomass estimates were attempted (*e.g.* a surplus production model for blue crab), the general lack of bay specific data often precluded these as well. We were more successful in developing catch estimates for many finfish fisheries, as most of the species within the bay are managed on a regional basis, and the data could be broken down by gear type and fishing location. For state managed fisheries there tended to be more data for recent landings, though data on entire sectors were often missing (recreational blue crabs, commercial and recreational hard clams).

The Barnegat Bay Ecosystem Model can be used as a tool to highlight where key data gaps still exist. The state, under the Governor's 10-point Plan, has undertaken a number of scientific studies that provided important details to this model. Even with this commitment, there are still a number of key data gaps that need to be addressed to improve the accuracy of this model, or future models for Barnegat Bay or other estuarine systems. Among the data needs are:

- Biomass and diet data on piscivorous seabirds, the highest rated keystone species in the model. All data for this group was modified from the Chesapeake Bay model.
- Regularly collected chlorophyll *a* data at relevant spatial and temporal scales. The lack of a primary production forcing function affected the transmission of dominant bottom-up effects through the model. While the state has been collecting data since the late 1990's, it was not always regularly collected estuary wide. Recent efforts have generated data at the scales needed, and should be continued.
- Regularly collected hard clam biomass and landing data. Hard clam biomass data has been collected every10-15 years, and landings data (commercial and recreational) are not currently collected at all. This lack of data makes it very hard to model population dynamics of this commercially important species, as evidenced in our inability to recreate known population changes in the model.
- Estimates of blue crab recreational landings. Our current assumption, that recreational landings are approximately 80% of total commercial landings, are based off of a study of a single year. This assumption makes the recreational fishery the largest component of blue crab harvest, and suggests that additional effort should be expended to understand how it effects blue crab biomass.
- A consistently scheduled fishery independent survey. The only regularly conducted fishery independent survey in Barnegat Bay is done by Rutgers University and has historically focused on the southern end of the bay (Little Egg Harbor). While this sampling program has expanded over the last three years as part of the Governor's 10-point Plan, its future is uncertain. An annual survey at the appropriate temporal and spatial scale would allow for the development of indices of relative abundances (and potentially biomasses), which would be a benefit to any future modeling exercises or fishery management efforts.

4.2 Management Scenarios

One of the main benefits of this type of holistic model is the ability to develop and evaluate a number of potential management scenarios from an ecosystem-wide perspective. This approach can lead to some surprising findings, as was seen in the OCNGS closure scenario. While a reduction in mortality associated with impingement and entrainment was expected to lead to increased biomass for many of the directly impacted species, the opposite was often the case. The decreased mortality on upper trophic level species such as weakfish led to increases in their biomass, which then translated into increased mortality on their prey items, usually in excess of that caused by the power plant.

The use of whole ecosystem models also points out that management of the target fishery itself is sometimes not sufficient to affect biomass trajectories. This was true for both hard clams and blue crab to a lesser degree. Placing a moratorium on hard clam harvest had almost no effect on overall biomass due to the low fishing mortality expressed in the model. Predation mortality in the model is also low, suggesting that some other aspect of natural mortality (disease, habitat changes, recruitment failure, etc.) is the primary control on hard clam biomass in Barnegat Bay. Therefore, management actions that address non-fishing activities (e.g., stormwater runoff, pathogen loading, temperature/salinity changes, brown-tide blooms) may be necessary to restore hard clam populations.

It should be noted that the outcomes of these management scenarios are only as reliable as the data used to construct them. The issues associated with the hard clam and blue crab data have been described above. The OCNGS "fishery" data had to be extensively manipulated as detailed in Appendix 3 to expand the reported mortality from numbers of individuals to weights, particularly for entrainment losses. The methodology used to determine impingement and entrainment losses and mortality in the Amergen (2008) report were slightly modified from those of earlier studies at OCNGS (EA Engineering 1981), which were the subject of a critical external peer review (Summers *et al.* 1989). In addition, there was a change in intake protection structures, and thus mortality rates, between the start of our model and the 2008 mortality study. Thus, there is a likelihood that the OCNGS removals used here are a conservative estimate.

Chapter 5 - Developing a spatially-explicit model

5.1 Introduction to Ecospace

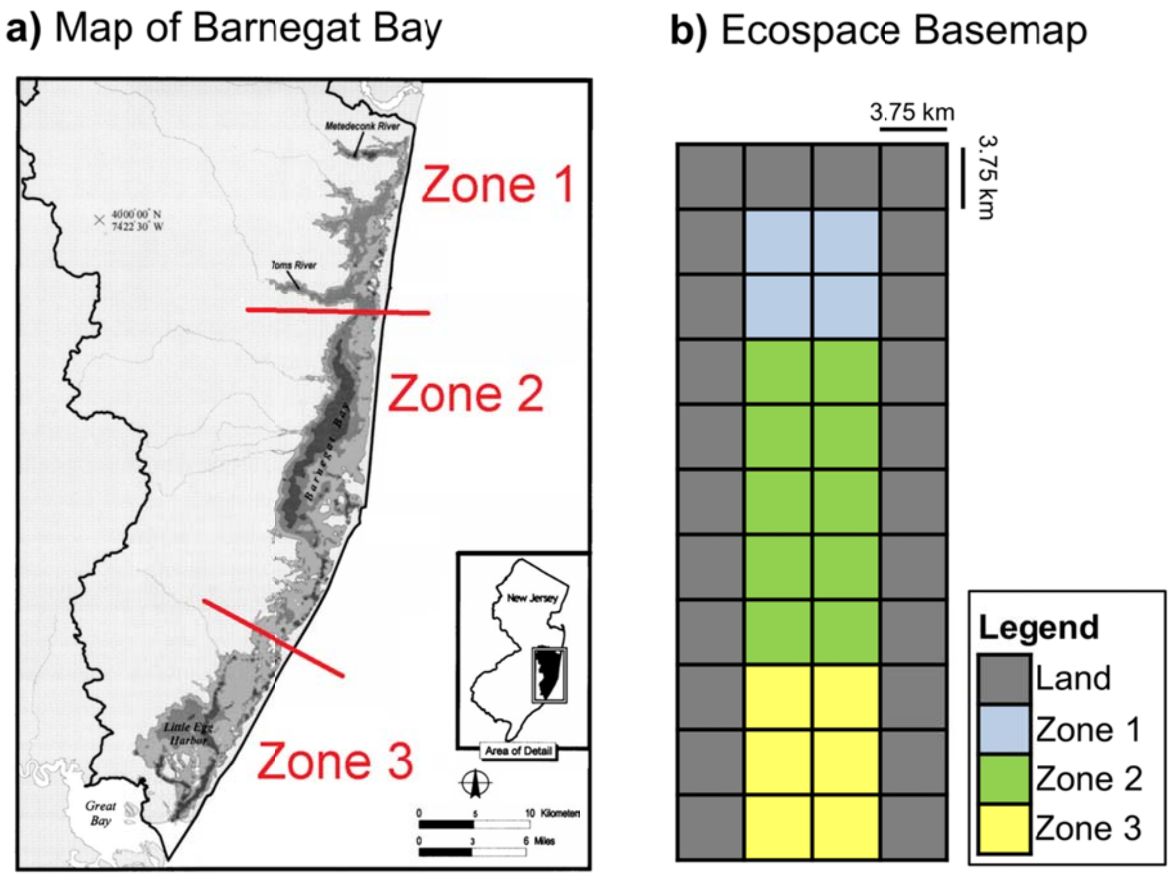
Ecospace is a spatially-explicit extension of the Ecopath with Ecosim modeling approach (Walters et al. 1999, Pauly 2000). In Ecospace, biomass dynamics are simulated via the Ecosim equations over a two-dimensional grid of cells that are linked via movement of organisms from each focal cell to its four neighboring cells. Ecospace requires a calibrated and balanced Ecopath-Ecosim model and can incorporate a variety of spatial processes. Ecospace models typically define habitat types, with each cell in the grid specified as a single habitat. Each group can have a preferred habitat and responses to its non-preferred habitats (e.g., habitat-dependent movement, vulnerability to predation, and feeding rates).Other possible spatially-explicit processes can include spatial distributions of physical conditions and group-specific responses; directed, seasonal migrations; and physical, oceanographic processes such as advection and upwelling (Christensen and Walters 2004). Although Ecospace incorporates a number of realistic, spatial processes, the intended use of Ecospace is not for specific, quantitative predictions (Walters et al. 1999). Rather, Ecospace is intended for the screening of potential management and policy alternatives that may warrant further investigation (Walters et al. 1999). A common use of Ecospace is to provide preliminary evaluation of the effect of marine protected areas on fisheries (e.g., Walters et al. 1999, Walters et al. 2010). Although the Ecospace scenarios described below are not applied to particular spatially-explicit management scenarios, the development of an Ecospace model is important for comparing results from non-spatial versions of the model (i.e., Ecopath and Ecosim). Once developed, this Ecospace model can be used to evaluate spatially-explicit management options in the future.

5.2 Ecospace Parameters and Basemap

For this Ecospace model, Barnegat Bay was divided into three zones according to Lathrop et al. (2001) (Figure 15a). Lathrop et al. (2001) chose these three zones "to represent distinct regions within Barnegat Bay-Little Egg Harbor as well as the gradient of watershed development" with the most development in Zone 1 and less development in Zones 2 and 3. Zone 1 (~55 km^2) extends from the northern portions of Barnegat Bay to just south of Toms River. Zone 2 (~138 km^2) is the central portion of Barnegat Bay to the Route 72 Bridge, with Zone 3 (~83 km^2), to the south, including Little Egg Harbor. These divisions also satisfy the criteria that at least one sampling station is located in each zone and they allow for the direct use of the submerged aquatic vegetation (SAV) estimates of Lathrop et al. (2001) (see *Spatially-explicit data* below). The Ecospace basemap was specified as an eleven row by four column grid, with each cell measuring 3.75 x 3.75 km (~14 km^2) (Figure 15b). A border of land cells was constructed on the eastern, northern, and western edges of the map (Figure 15b). Based on the approximate area of each zone, four, ten, and six cells were allocated to Zones 1, 2, and 3 respectively (Figure 15b).

In addition to a basemap, Ecospace requires the specification of movement rates. These movement rates are meant to represent the random, local movements of organisms and do not consider directed, seasonal migrations (Walters et al. 1999). A rate of 300 km/year is the default movement rate in Ecospace. Previous studies have used three relative magnitudes of movement (3, 30 and 300 km/year) for sessile or non-dispersing, demersal or small-bodied, and pelagic or large-bodied organisms, respectively (Chen et al. 2009, Fouzai et al. 2012). The scenarios below will explore the use of both the default movement rate (Scenarios 1 and 3) and biologically-based movement rates (Scenario 2).

Figure 15: a) Map of Barnegat Bay with geographic zones of Lathrop et al. (2001) indicated (Figure modified from Lathrop et al. 2001), and b) the Ecospace basemap of Barnegat Bay.



5.3 Spatially-explicit data

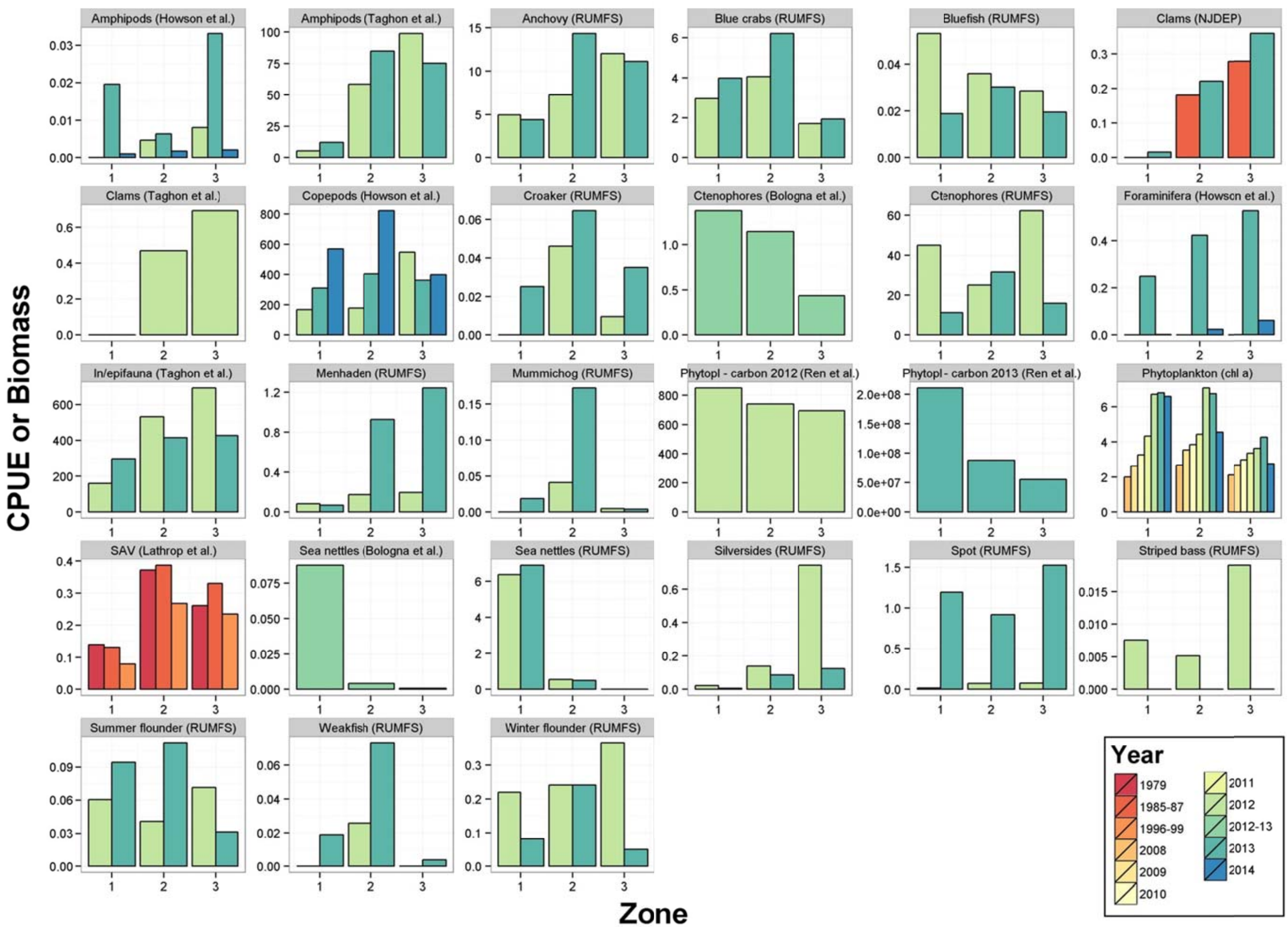
Spatially-explicit biomass data can be used for two purposes with an Ecospace model: for model initiation or for model testing and validation. In the first case, the observed distributions of biomass among regions can be used to allocate initial biomasses in an Ecospace model (see Scenarios 1 and 2 below). In the second case, the Ecospace model can be initiated with relatively simple assumptions about habitat, physical conditions, or fishing effort and the modeled spatial patterns can be compared to observed distributions of biomass among the regions (see Scenario 3 below). Spatially-explicit biomass data were gathered from a variety of sources (Table 4). All groups except piscivorous seabirds, non-piscivorous seabirds, river herring, oysters, and benthic algae had sufficient spatially-explicit data to partition the biomass among the three geographic zones. The majority of spatially-explicit data came from recent years as part of the Barnegat Bay research initiative lead by NJDEP (i.e., mostly 2012 and later), but older data were available for both hard clams and SAV (1979 and 1985-1987, respectively).

The distribution of biomass or catch per unit effort (CPUE) for each group is shown in Figure 16. Multiple data sources or years are plotted for most groups. For all data sources each sample (e.g., a plankton tow, trawl, or benthic grab) was attributed to a zone based on its geographic coordinates (i.e., latitude and longitude) and then the average response (e.g., CPUE, density, concentration) of all samples was determined for that zone. To compare model predictions to observed data, the biomass distributions from multiple sources or years were combined for a particular group. In order to account for differences in methods, measurement units, and annual variability, the values were scaled so the geographic zone with the highest value equaled 1.0 and all other zones were proportional. Then, the scaled values were averaged across years or sources.

Table 4: Spatially-explicit relative abundance/biomass data for each group in the Ecospace model. Groups without information did not have spatially-explicit data available.

Group name	Source(s)	Year(s)	Gear / Method	Units	Notes
Piscivorous seabirds					
Non-Piscivorous seabirds					
Striped bass	RUMFS	2012, 2013	otter trawl	CPUE	
River herring					insufficient catch (RUMFS)
Bay Anchovy	RUMFS	2012, 2013	otter trawl	CPUE	
Atlantic Menhaden	RUMFS	2012, 2013	otter trawl	CPUE	
Weakfish	RUMFS	2012, 2013	otter trawl	CPUE	
Mummichog	RUMFS	2012, 2013	otter trawl	CPUE	
Spot	RUMFS	2012, 2013	otter trawl	CPUE	
Atlantic Silverside	RUMFS	2012, 2013	otter trawl	CPUE	
Atlantic Croaker	RUMFS	2012, 2013	otter trawl	CPUE	
Summer Flounder	RUMFS	2012, 2013	otter trawl	CPUE	
Bluefish	RUMFS	2012, 2013	otter trawl	CPUE	
Winter Flounder	RUMFS	2012, 2013	otter trawl	CPUE	
Microzooplankton	NJDEP	2012-2013	tow	CPUE	Foraminifera only
Oysters					
Blue crabs	RUMFS	2012, 2013	otter trawl	CPUE	
Hard Clams	Joseph 1986, 1987, Celestino et al. Taghon et al. 2013 NJDEP	1985-87 2012 2013	dredge Van Veen grab dredge	# / sq. ft. # / 0.04 m^2 # / sq. ft.	adj. for dredge efficiency adj. for dredge efficiency
Sea Nettles	RUMFS Bologna et al.	2012, 2013 2012-2013	otter trawl lift net	CPUE CPUE	
Ctenophores	RUMFS Bologna et al.	2012, 2013 2012-2013	otter trawl lift net	CPUE CPUE	
Benthic infauna/epifauna	Taghon et al. 2013	2012, 2013	Van Veen grab	# / 0.04 m^2	
Amphipods	Taghon et al. 2013 Howson et al.	2012, 2013 2012-2014	Van Veen grab tow	# / 0.04 m^2 CPUE	
Copepods	Howson et al.	2012-2014	tow	CPUE	
Benthic algae					
SAV	Lathrop et al. 2001	1979, 1985-1987, 1996-1999	aerial / satellite	% cover	
Phytoplankton	Ren et al. NJDEP	2012-13 2008-14	tow overflight	µg / L mg / L	Total carbon Chlorophyll a
Detritus					

Figure 16: Spatial distribution of relative abundance of groups in Barnegat Bay. See Table 4 for units of vertical axis.



5.4 Scenario 1: Initiate with current biomass divisions – default movement rates

This scenario simulated the spatial patterning of biomass under conditions where initial biomass was divided according to current geographic divisions in relative abundance. The current distribution of relative abundance was determined for each group by averaging multiple sources and years 2008 and later. Prior to averaging, abundances were scaled so the highest value equaled 1.0 as described in *5.3 Spatially-explicit data.*

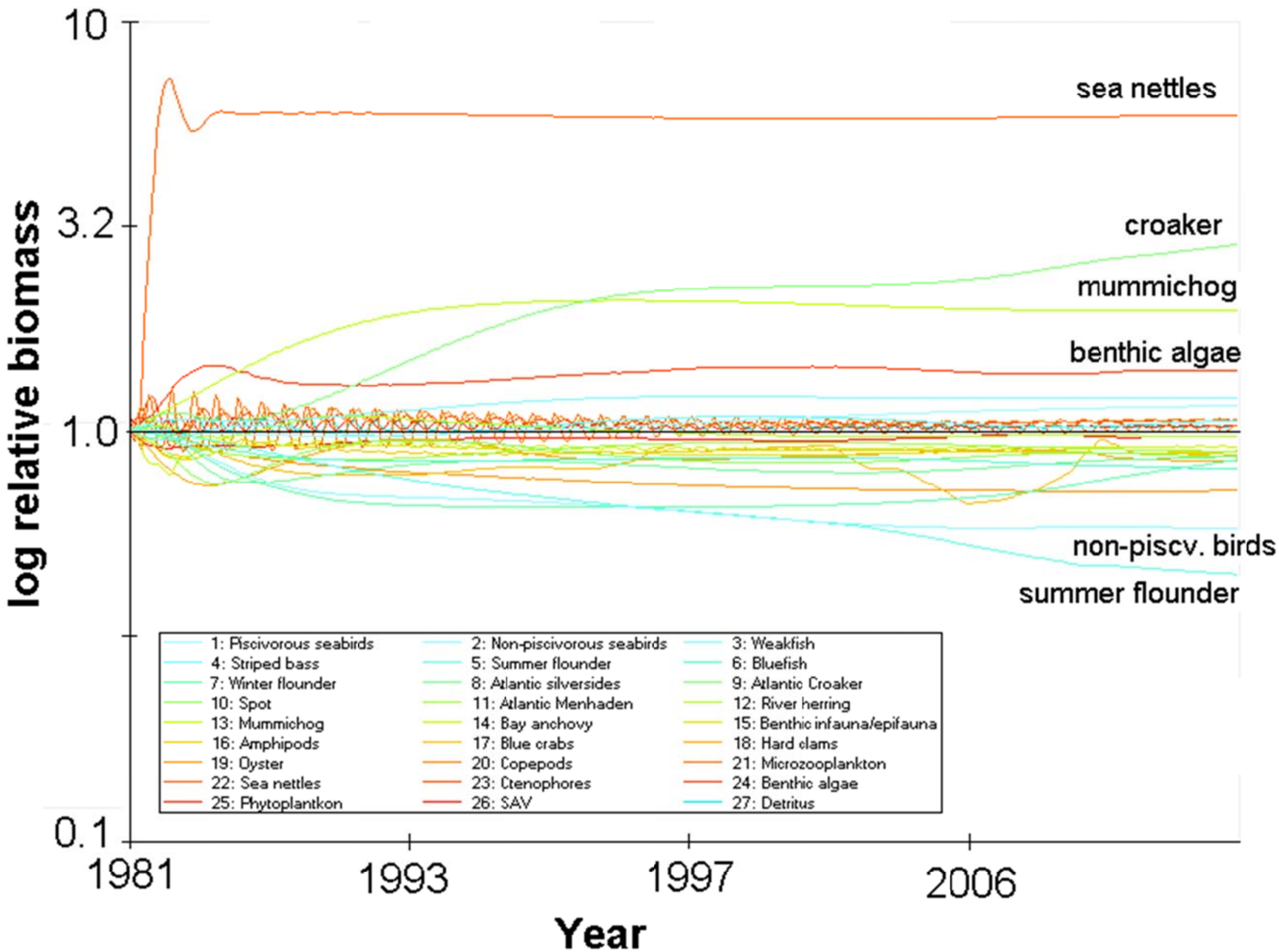
Ecospace does not have a direct means to specify an initial spatial distribution of biomass. Instead, the "Habitat-adjusted biomasses" initialization was used in conjunction with specifying a "Habitat foraging usage" for each group (Table 5). Since these habitats designations were only intended to initiate the model, conditions in non-preferred habitat were set equal to the preferred habitat in order to make the habitat-types functionally equivalent (i.e., the relative dispersal, relative vulnerability to predation, and relative feeding rate in non-preferred habitat was set to 1.0). Table 5 presents the values used for the "Habitat foraging usage" input. Groups that did not have spatially explicit data (Table 4) received a value of 1 for all habitats (i.e., zones). The default movement rate for all non-detrital groups (300 km/year) was used. Detritus has a default movement rate of 10 km/year in Ecospace.

The Ecospace model was run from 1981 to 2013. At the end of the simulation, sea nettles had the largest increase in relative biomass, followed by Atlantic croaker and mummichog (Figure 17). Summer flounder and non-piscivorous seabirds had the largest decreases in relative biomass (Figure 17). Many of the groups showed substantial spatial patterning at the end of the simulation (Figure 18). Groups including sea nettles, benthic algae, phytoplankton, and copepods had their highest relative biomass in Zone 1 in the north of Barnegat Bay (Figure 18). Other groups including weakfish, Atlantic croaker, blue crabs, and mummichog had their highest relative biomass in the central region (Zone 2) (Figure 18). Figure 19 shows the predicted (i.e., modelled) and observed relative abundance among the three geographic regions. Many of the fish groups (e.g., bluefish, Altantic croaker, mummichog, Atlantic silversides, spot, stripped bass, and weakfish) appear to show good agreement between predictions and observed values (Figure 19). Whereas, many of the lower trophic level consumers (e.g., amphipods, benthic in/epifauna, and microzooplankton/Foraminifera) showed greater discrepancies (Figure 19).

Table 5: Habitat foraging usage used to initiate relative biomass distributions in Scenarios 1 and 2.

Group	**Zone 1**	**Zone 2**	**Zone 3**
Piscivorous seabirds	1	1	1
Non-piscivorous seabirds	1	1	1
Weakfish	0.129	1	0.027
Striped bass	0.398	0.269	1
Summer flounder	1	0.931	0.757
Bluefish	0.969	1	0.707
Winter flounder	0.566	1	0.73
Atlantic silversides	0.041	0.438	1
Atlantic Croaker	0.195	1	0.375
Spot	0.491	0.772	1
Atlantic Menhaden	0.242	0.82	1
River herring	1	1	1
Mummichog	0.055	1	0.07
Bay anchovy	0.405	0.904	1
Benthic infauna/epifauna	0.463	0.871	1
Amphipods	0.266	0.66	1
Blue crabs	0.685	1	0.366
Hard clams	0.023	0.647	1
Oyster	1	1	1
Copepods	0.742	0.977	1
Microzooplankton	0.257	0.599	1
Sea nettles	1	0.068	0.004
Ctenophores	0.929	1	0.814
Benthic algae	1	1	1
Phytoplantkon	1	0.965	0.691
SAV	0.295	1	0.876
Detritus	1	1	1

Figure 17: Model predictions of relative biomass from 1981 to 2013 (Scenario 1 – default dispersal)



+

Figure 18: Model predictions of spatial distribution of relative biomass in 2013 (Scenario 1 – default dispersal)

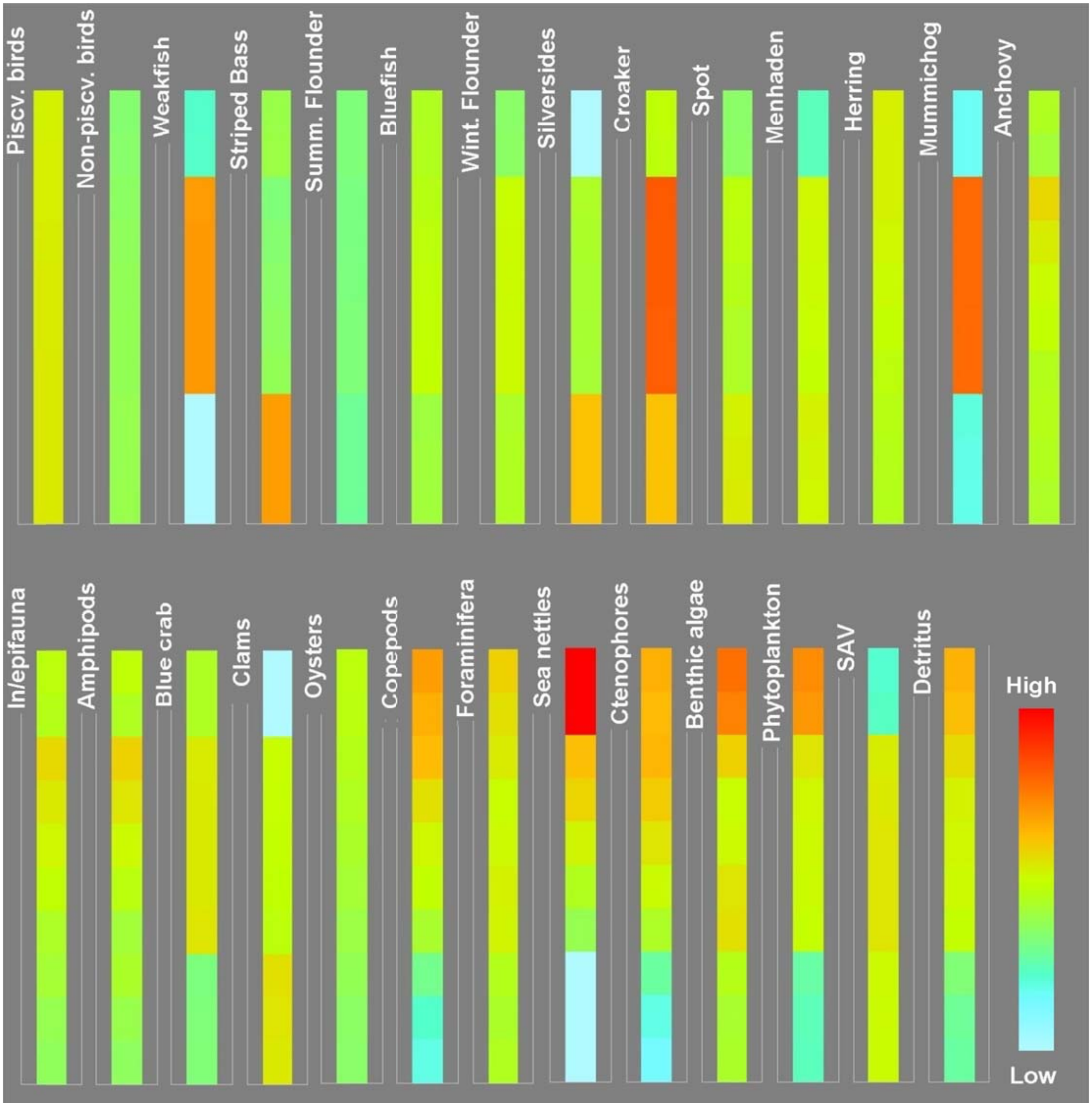
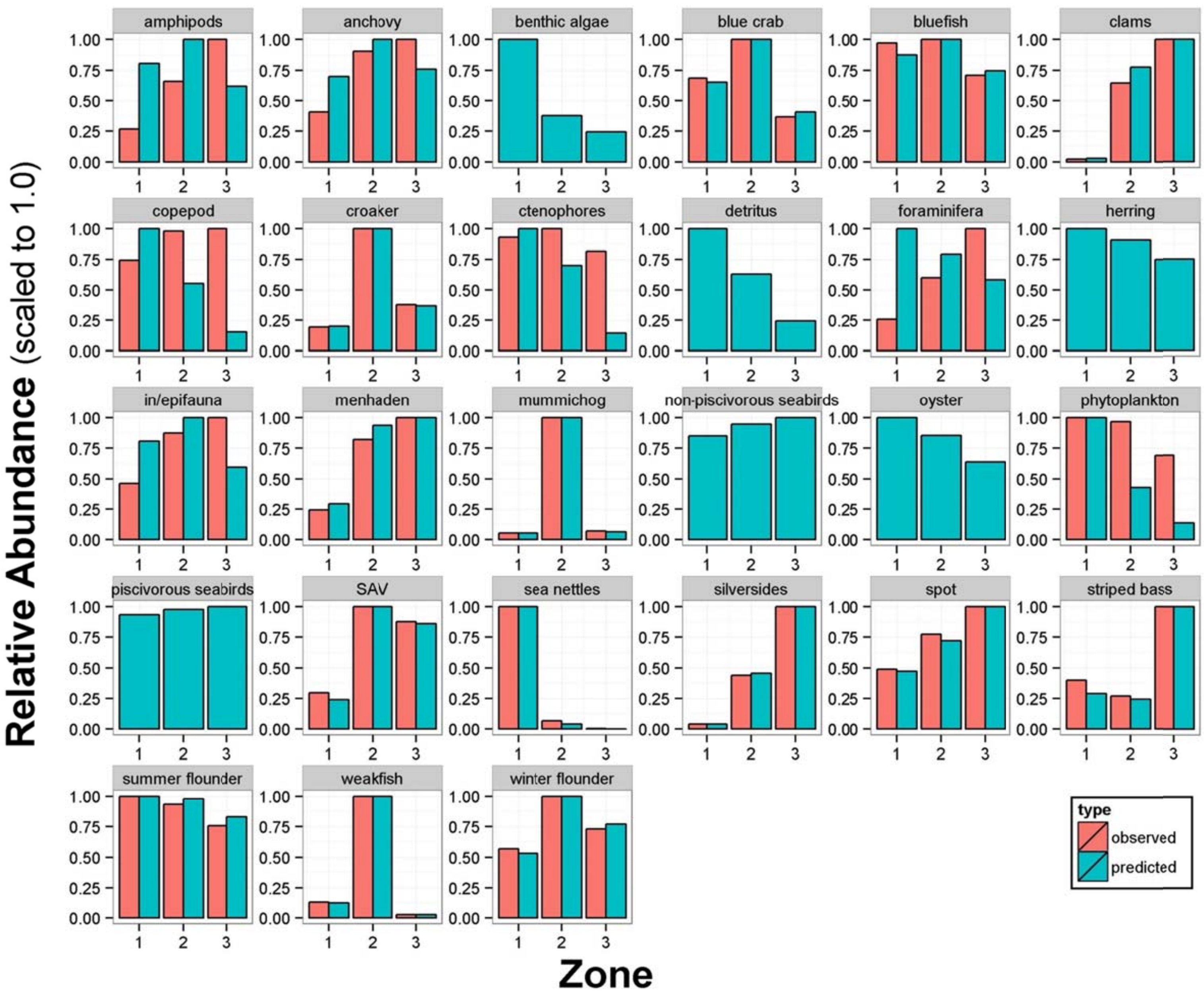


Figure 19: Predicted vs. observed relative abundance among the three geographic zones (Scenario 1 – default dispersal)



5.5 Scenario 2: Initiate with current biomass divisions – biologically-based movement rates

Scenario 2 was identical to Scenario 1 except movement rates were biologically-based, rather than the default rate of 300 km/year. Table 6 displays the movement rates used in this scenario. Like previous studies (Chen et al. 2009, Fouzai et al. 2012), three levels of movement were used (3, 30, and 300 km/year) for organisms that are sessile or non-dispersing, demersal or small-bodied, and pelagic or large-bodied organisms, respectively.

The Ecospace model was run from 1981 to 2013. At the end of the simulation, sea nettles had the largest increase in relative biomass (Figure 20). Atlantic croaker had a large decrease in relative biomass (Figure 20). This scenario also showed larger annual cycles in many groups, especially those in lower trophic levels (e.g., microzooplankton, copepods) (Figure 20). Many of the groups showed substantial spatial patterning at the end of the simulation (Figure 21). Groups including sea nettles and copepods still had their highest relative biomass in Zone 1, as in Scenario 1, but now groups including benthic epi/infauna and amphipods also showed this pattern of higher biomass in the north (Figure 21). More groups in Scenario 2 displayed a pattern of highest relative biomass in the southern Zone 3 (e.g., striped bass, Bay anchovy, Atlantic silversides) than in Scenario 1 (Figure 21). Many of the fish groups that had a good match between predicted and observed spatial distributions in Scenario 1 had slight decreases in fit (e.g., summer and winter flounder, striped bass, Atlantic silversides) (Figure 22). As in Scenario 1, many of the lower trophic level consumers (e.g., amphipods, benthic in/epifauna, and microzooplankton/Foraminifera) showed large discrepancies between predicted and observed distributions (Figure 22).

Table 6: Movement rates for each group in Scenario 2.

Group name	Movement rate (km/year)
Piscivorous seabirds	300
Non-Piscivorous seabirds	300
Striped bass	300
River herring	300
Bay Anchovy	30
Atlantic Menhaden	300
Weakfish	300
Mummichog	3
Spot	300
Atlantic Silverside	30
Atlantic Croaker	300
Summer Flounder	300
Bluefish	300
Winter Flounder	30
Microzooplankton	3
Oyster	3
Blue crabs	30
Hard Clams	3
Sea Nettles	3
Ctenophores	3
Benthic infauna/epifauna	3
Amphipods	3
Copepods (Mesozooplankton)	3
Benthic algae	3
SAV	3
Phytoplankton	3
Detritus	3

Figure 20: Model predictions of relative biomass 1981 through 2013 (Scenario 2 – biologically-based dispersal)

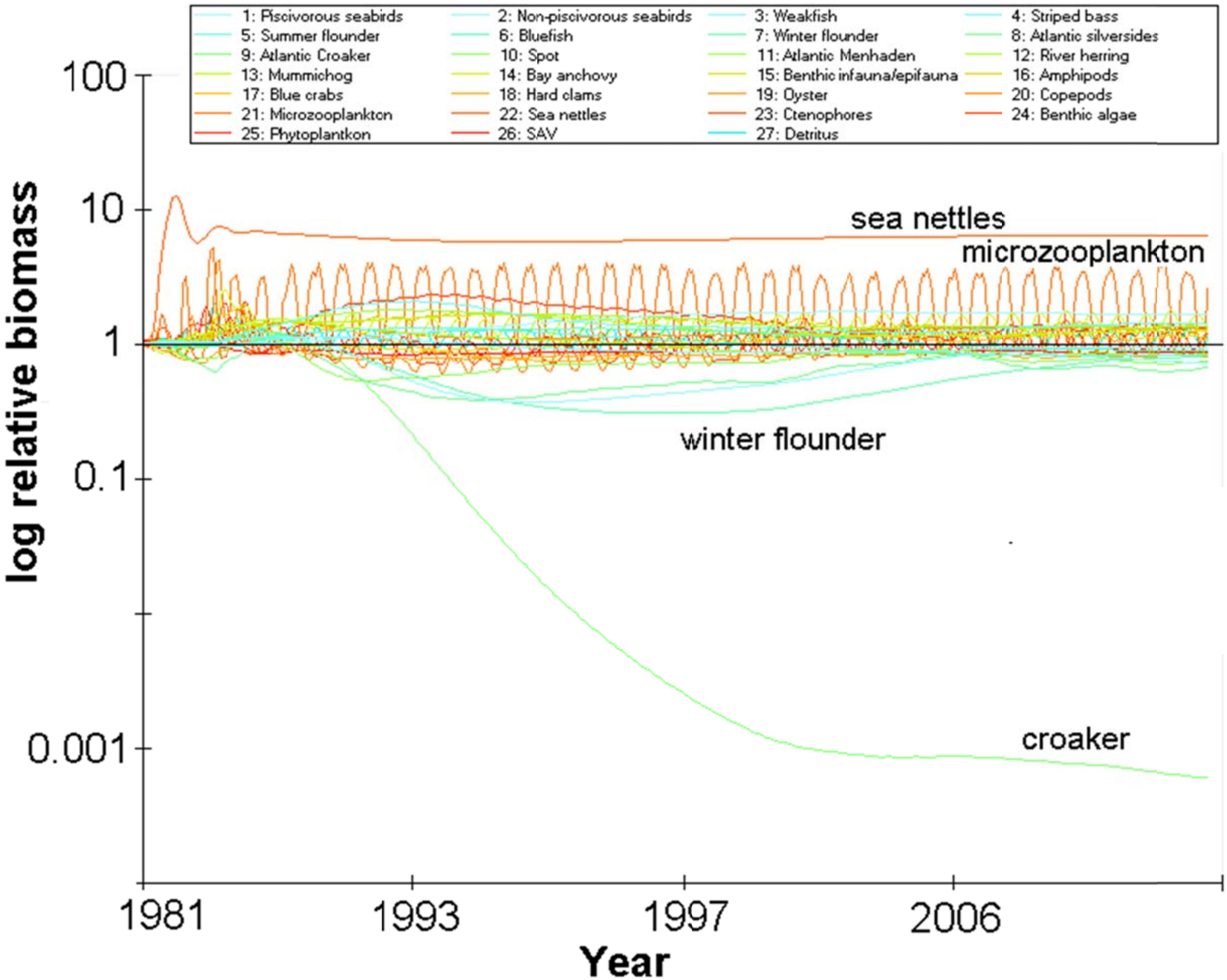


Figure 21: Model predictions of spatial distribution of relative biomass in 2013 (Scenario 2 – biologically-based dispersal)

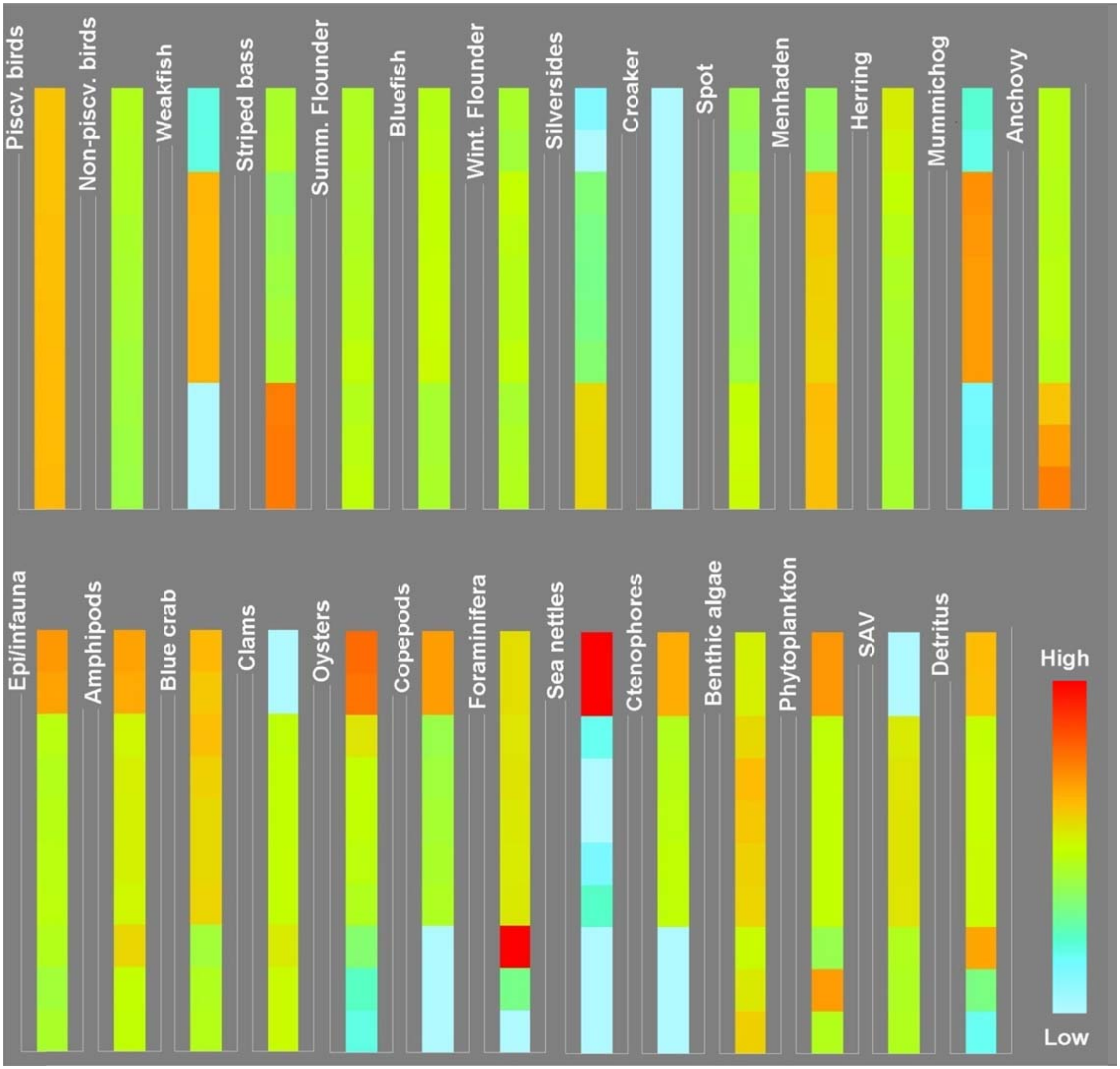
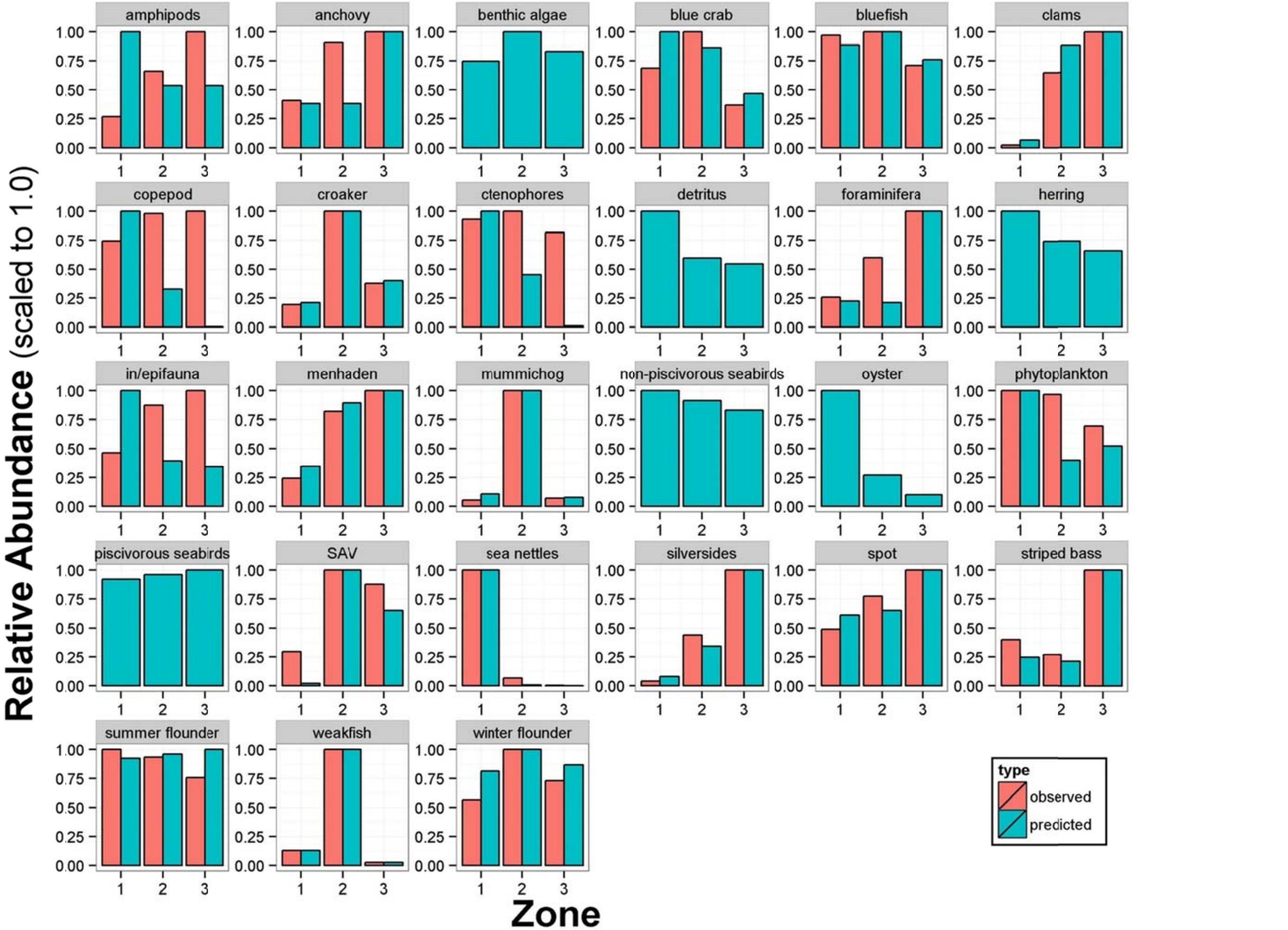


Figure 22: Predicted vs. observed relative abundance among the three geographic zones (Scenario 2 – biologically-based dispersal)



5.6 Scenario 3: Initiate with 1980s data and minimal assumptions – default dispersal rates

Rather than use the current biomass distributions to initiate the model as in Scenarios 1 and 2, Ecospace could be initiated with minimal assumptions about past conditions and then compared to current geographic divisions in relative abundance. In particular, this scenario used the observed biomass divisions of hard clams from 1985-1987 (Joseph 1986, 1987, Celestino 2003) and of SAV from 1979 (Lathrop et al. 2001) to initiate the model. It also made the assumption that the current geographic distribution of phytoplankton was present in 1981. The initiation of biomass for these three groups was done in the same way as Scenarios 1 and 2 – by using the "Habitat foraging usage" input (Table 7). All groups that did not have spatial distribution data near the start of the Ecospace model (~1981) received a value of 1.0 for all habitats (i.e., zones). As in previous scenarios, the relative dispersal, relative vulnerability to predation, and relative feeding rate in non-preferred habitat was set to 1.0. This makes the habitat designations a means to initiate the model and makes all habitats functionally equivalent as the model is run.

In this scenario, Atlantic croaker, followed by sea nettles, had the largest increases in relative biomass from 1981 to 2013 (Figure 23). No groups displayed major decreases in biomass, although blue crabs displayed a decrease and a rebound (Figure 23). Little spatial patterning of biomass was apparent at the end of the simulation for most groups (Figure 24). Some of the groups that had initial spatial data (i.e., hard clams and SAV) displayed a spatial pattern at the end of the simulation (Figure 24). Despite not having any spatial data inputs, a spatial pattern of higher biomass in the north, especially Zone 1, was apparent for ctenophores, copepods, amphipods, and especially the sea nettles (Figure 24). Since the model did not predict substantial spatial patterns for biomass of most groups, there was a large discrepancy between predicted and observed relative abundances for many groups (Figure 25).

Table 7: Habitat foraging usage used to initiate relative biomass distributions in Scenario 3.

Group	**Zone 1**	**Zone 2**	**Zone 3**
Piscivorous seabirds	1	1	1
Non-piscivorous seabirds	1	1	1
Weakfish	1	1	1
Striped bass	1	1	1
Summer flounder	1	1	1
Bluefish	1	1	1
Winter flounder	1	1	1
Atlantic silversides	1	1	1
Atlantic Croaker	1	1	1
Spot	1	1	1
Atlantic Menhaden	1	1	1
River herring	1	1	1
Mummichog	1	1	1
Bay anchovy	1	1	1
Benthic infauna/epifauna	1	1	1
Amphipods	1	1	1
Blue crabs	1	1	1
Hard clams	0	0.652	1
Oyster	1	1	1
Copepods	1	1	1
Microzooplankton	1	1	1
Sea nettles	1	1	1
Ctenophores	1	1	1
Benthic algae	1	1	1
Phytoplantkon	1	0.965	0.691
SAV	0.374	1	0.702
Detritus	1	1	1

Figure 23: Model predictions of relative biomass 1981 through 2013 (Scenario 3)

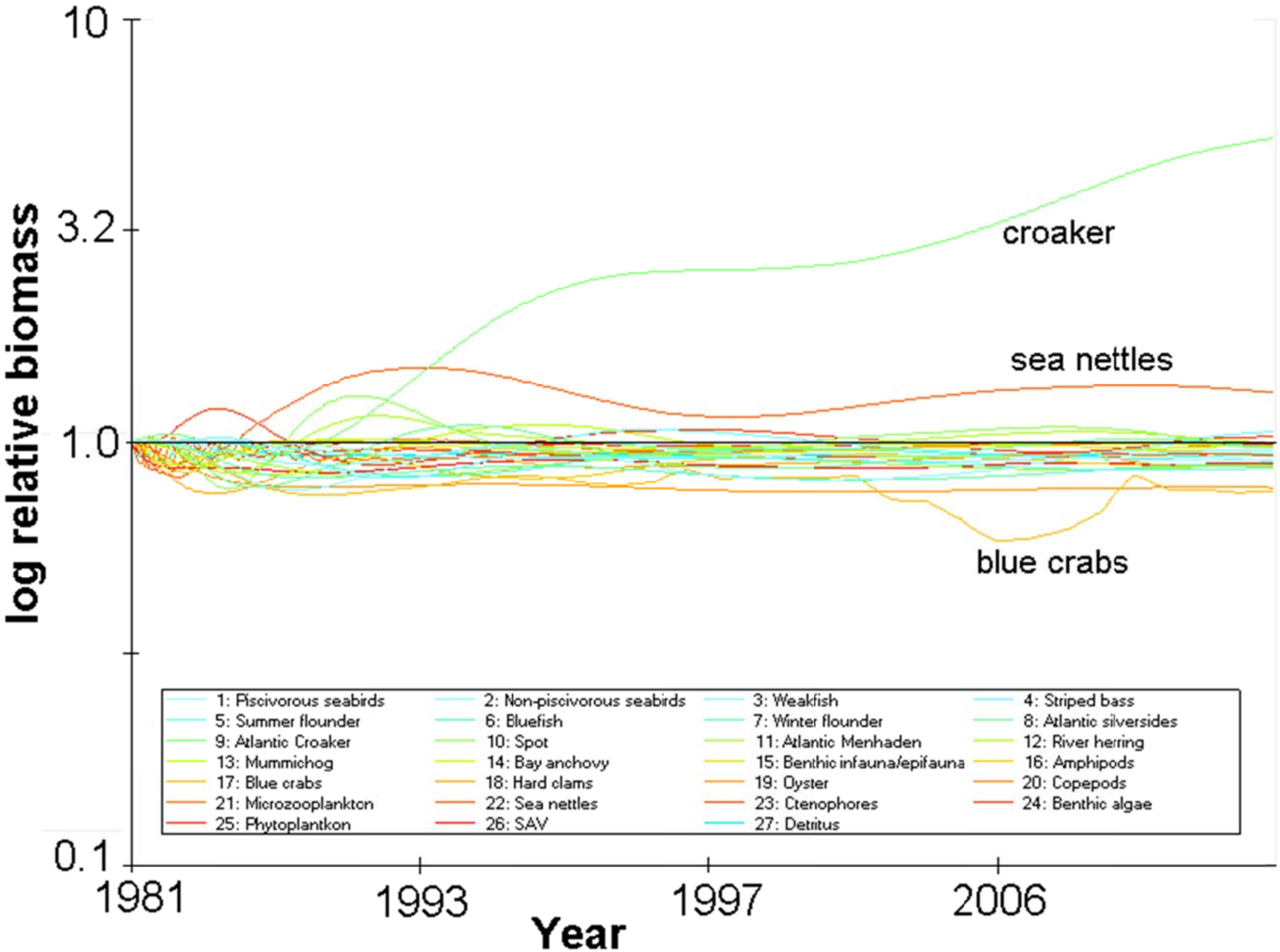


Figure 24: Model predictions of spatial distribution of relative biomass in 2013 (Scenario 3)

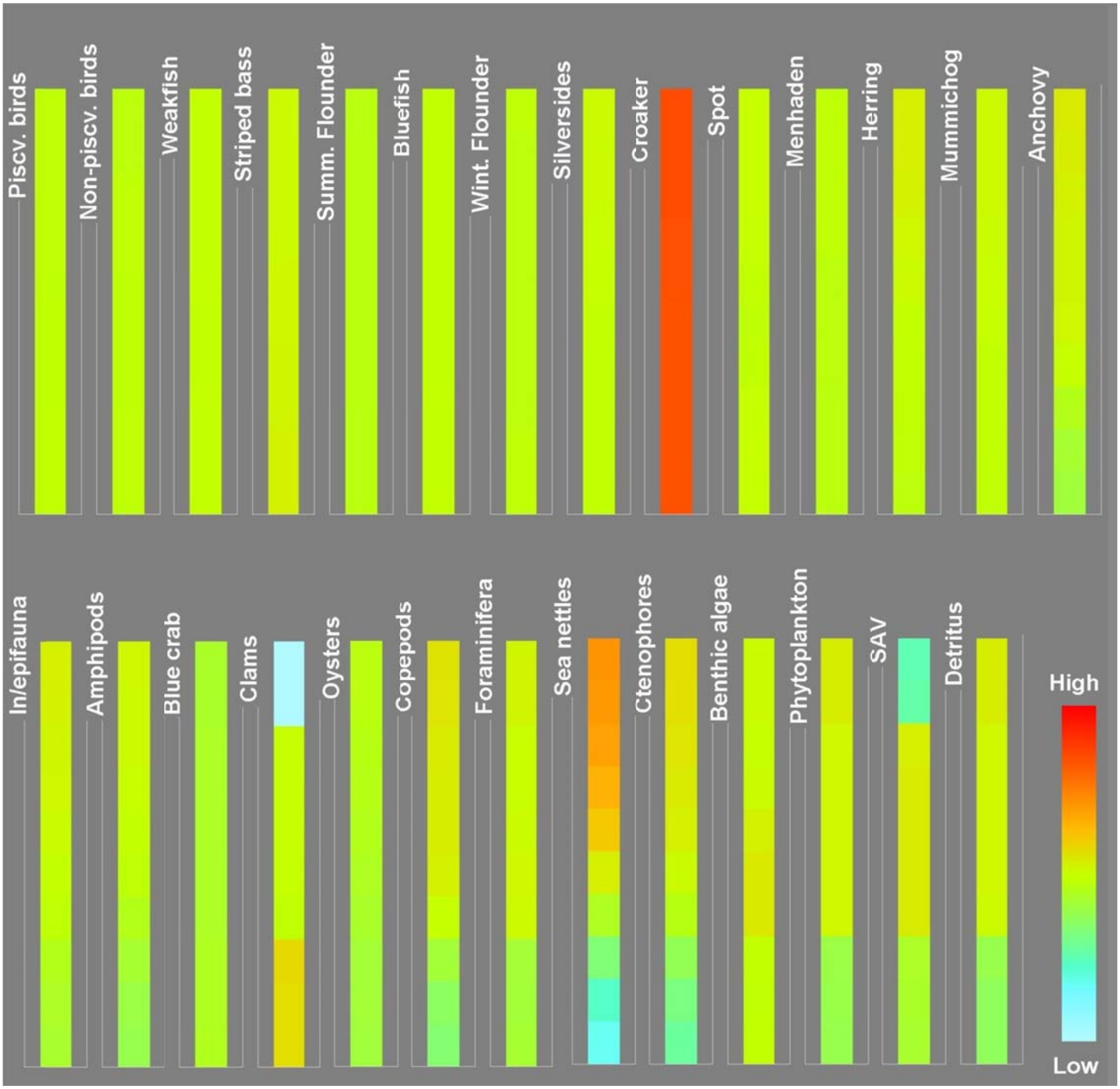
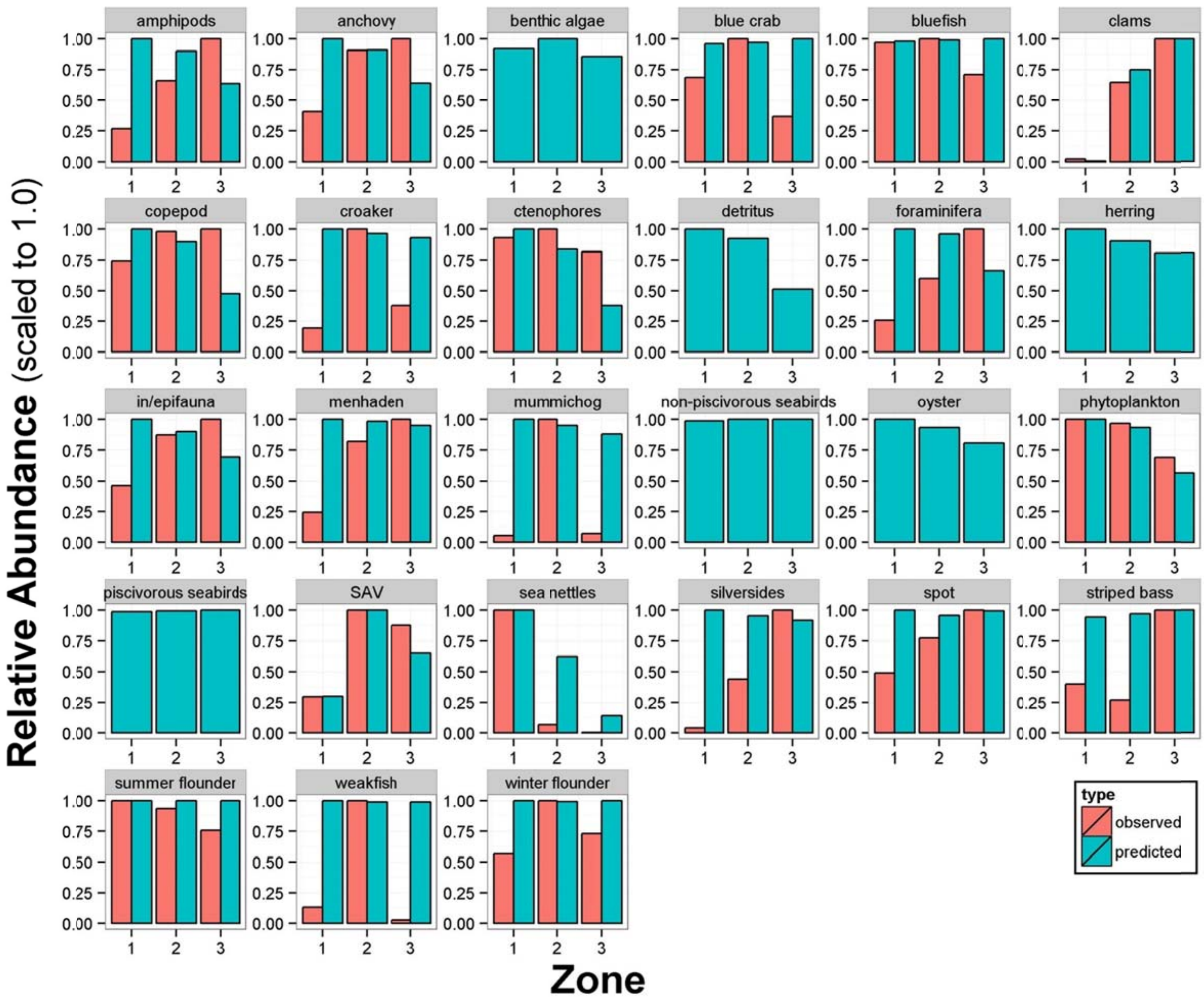


Figure 25: Predicted vs. observed relative abundance among the three geographic zones (Scenario 3)



5.7 Ecospace Conclusions

Although the Ecospace scenarios described above are not addressing particular spatially-explicit management scenarios, the development of an Ecospace model is important for assessing the performance of non-spatial versions of the model (i.e., Ecopath and Ecosim). For some questions, a non-spatial model may be adequate, but for others capturing spatially-explicit processes maybe essential. Now that it has been developed, this Ecospace model can be used to evaluate alternative, spatially-explicit management options in the future.

Depending on the methods used to establish the Ecospace model (Scenarios 1-3), some of the model outcomes differed. Using current, spatial data to initialize the model (Scenarios 1 and 2) resulted in greater spatial patterning at the end of the simulation, compared to using historical, spatial data for initiation (Scenario 3). Using the historical spatial data to initialize the model (Scenario 3) also resulted in greater discrepancies between predicted and observed spatial patterns for many fish species (e.g., striped bass, menhaden, mummichog) (Figure 21) compared to the other scenarios (Figures 15 and 18). Although, all scenarios produced mismatches between predicted and observed spatial patterns for lower trophic level groups such as copepods, amphipods, phytoplankton, and microzooplankton (i.e., foraminifera). The choice of using default or realistic dispersal rates (Scenario 1 vs. 2) did not have a large effect on the outcome of the simulations, although using realistic dispersal rates produced a substantial decrease in Croaker biomass (Scenario 2) that was not observed in the other scenarios.

6.0 Environmental Management Charge Questions

1. **How can the BBay ecosystem-based model (EWE-NPZ-WASP) assist NJDEP in developing an ecosystem-based management (EBM) plan for decision-making?**

   Through model parameterization, simulation, and fitting process, historic and current Barnegat Bay-specific data related to species biomasses, vital rates, and harvest levels were compiled into a single repository. The modeling process identified gaps in our understanding of crucial ecosystem processes and relevant data needs. The model was also used to develop and evaluate the effects of potential management scenarios from an ecosystem-wide perspective. We recommend that these ecosystem model outputs be used for strategic rather than tactical decision-making. For example, traditional single-species stock assessment models are likely to be a better tool for estimating the impact of specific harvest levels on the target species (a tactical decision). However, these ecosystem model outputs are valuable for understanding likely indirect impacts through trophic relationships between target species and their predators and prey. Thus ecosystem models such as this one can provide advance warning of possible ecological surprises arising from indirect food web impacts of management decisions. Within an EBM framework, ecosystem models are often useful for identifying threshold biomass levels of key prey species below which there are likely to be unacceptably large negative consequences on their predators.

2. **What are the EBM identified critical drivers and specific pressures on the ecosystem of BBay and their feedback relationships?**

   Based on the results of the modeling scenarios, from a trophic standpoint the Barnegat Bay appears to be dominated by "bottom-up" processes, where primary producers, both phytoplankton and SAV, exert a large influence on the functioning of the system. Piscivorous seabirds appear to be important in controlling the middle and upper trophic levels, but there is no Barnegat Bay specific information available for them, making our understanding of their role less precise. While commercial harvest for finfish is limited in Barnegat Bay, the commercial harvest for blue crabs has a large influence on the size of the population as evidenced by the various harvest scenarios. This effect is almost certainly compounded by what is assumed to be a robust recreational fishery, but with limited recreational data it is difficult to assess its impact. While eutrophication is known to be an important anthropogenic driver of conditions in Barnegat Bay, the EwE modeling approach has limited capacity to predict the impacts of changing nutrient levels. Once the WASP model is built and validated, its outputs can be used to understand the impacts of changing nutrient loads on higher trophic level components of the Bay.

3. **How can the EBM model be used to test hypothesis about 1) how predicted changes in climate will affect changes to the fishery resource, 2) how closure of Oyster Creek Nuclear Generating Station will affect the fish and shellfish of the bay, and 3) how management actions related to the hard clam and blue crab fisheries would impact the populations of those and other species in the bay, both directly through harvest and indirectly through cascading food web effects?**

   Under the current model formulation the effects of climate change (increasing water temperatures, changes in precipitation, sea level rise) cannot be explicitly incorporated. However, changes to primary production due to climate change can be included in the WASP model, and the resultant effects can be incorporated into EwE. Changes to fish and shellfish populations as a result of a) the closure of OCNGS have been discussed in Section 3.3.1, b) changes to blue crab harvest strategies in Section 3.3.2, and c) changes to hard clam harvest in Section 3.3.3.

4. **Will the EBM model require spatial separation to account for strong gradients of salinity and biotic variability in Barnegat Bay? How will this limit the accuracy of the non-spatial aspects of the ecosystem model?**

   For overall ecosystem assessment and many management scenario evaluations the non-spatial model is robust to gradients in biotic variability and salinity. However, if management actions will be undertaken on a spatial basis (i.e. area closures, MPAs) then a spatially explicit model (i.e., Ecospace) would be more appropriate. Our Ecospace model results suggest that the non-spatial EwE model is generally robust to spatial patterns in different model groups (taxa) within the Bay.

   The Barnegat Bay EwE model, by design, only models dynamics of the food web within Barnegat Bay. Several species of interest, e.g., Striped Bass, live much of their life cycle outside of the Bay. Predictions for these species will have a higher degree of uncertainty because the model does not capture processes outside the Bay, e.g., coastal fishing, that may influence their abundance.

5. **How will the hydrodynamic model of the Bay being developed by the USGS be integrated with the EBM model to offers insight into the spatial dynamics of nutrients and phytoplankton?**

Ideally, the USGS ROMS model will feed into the WASP water quality model, and the outputs from the WASP model, particularly biomass and vital rates for primary producers, can be utilized as inputs for the EwE model. A programmer at the USGS, Southeast Ecological Science Center, Mark McKelvy, developed code for us to convert WASP output files into a form useable by EwE. However, the WASP model is not yet complete.

6. **How effective and predictive is the integration of the planktonic ecosystem and higher trophic levels models for exploring past and future anthropogenic and environmental perturbations?**

The EwE model is specifically designed to account for the trophic interactions between biomass groups, starting with primary producers up through top consumers. While there is the ability to incorporate the effects of nutrients on the production rate of primary producers, the relatively simplistic manner in which it is handled may not be of sufficient resolution to satisfy management needs. We recommend "linking" the EwE to the WASP model (as discussed above) for best results.

It should be noted that given the importance of primary production in driving this system (#2 above), the model would benefit greatly from a phytoplankton biomass time series, which was not available. For the Chesapeake Bay EwE model (Christensen et al 2009) a 50+ year phytoplankton time series was developed using a separate "simple" hydrodynamic model (major inputs included wind, rainfall, gage inflow, and relative loading), which substantially improved the overall fit of the EwE model.

7. **Can the model separate and integrate simultaneous perturbations in the model system (e.g., nutrient flows and fish harvest)?**

Yes. In the current report we have run separate scenarios for each perturbation, but they can be combined to model the effect of several management changes at once.

8. **Will the EBM model allow observations of not only changes in individual species parameters over time but also changes to key indicators of ecosystem condition such as maturity, throughput, connectance, and trophic transfer efficiency?**

Yes, the Ecopath module includes a routine for “Ecological Network Analysis”, which provides information on all of those indicators. An Ecopath model can be automatically created for each time step in the Ecosim simulation, and the indicators can be produced.

9. **Will the model identify key species to be included in future monitoring plans? What species?**

Yes, there are two ways in which the model can identify key species for future monitoring. The keystone species routine (Section 3.1, Figure 7) identifies those biomass groups that have the largest cumulative effect on the system per biomass amount. In this model it was piscivorous sea birds and phytoplankton. The second method is through the Mixed Trophic Impact analysis (Section 3.1, Figure 6), where a small change is made to the biomass of a particular group and its relative effect on all other groups is calculated.

Literature Cited

Able, K.W., and Fahay, M.P. 2010. Ecology of estuarine fishes: temperate waters of the Western North Atlantic. Johns Hopkins University Press, Baltimore, MD.

Amergen, 2008. Characterization of the Aquatic Resources and Impingement and Entrainment at Oyster Creek Nuclear Generating Station, pp. 1-33.

Anderson, D.R. 1975. Population ecology of the mallard, V: Temporal and geographic estimates of survival, recovery, and harvest rates. U.S. Fish and Wildlife Service Resource Publication 25: 110.

Atlantic States Marine Fisheries Commission. 2003 Atlantic striped bass advisory report. ASMFC Striped Bass Technical Committee Report 2003-03, Atlantic States Marine Fisheries Commission, Washington, D.C.

Bologna, P., and Gaynor, J. 2013. Assessment of the distribution and abundance of sting sea nettles (jellyfish) in Barnegat Bay. New Jersey Department of Environmental Protection, pp. 1-92.

Brey, T. 1999. A collection of empirical relations for use in ecological modelling. The ICLARM Quarterly 22: 24-28.

Brey, T., Muller-Wiegmann, C., Zittier, Z., and Hagen, W. 2010. Body composition in aquatic organisms – a global data bank of relationships between mass, element composition, and energy content. Journal of Sea Research 64: 334-340.

Bricelj, V. M., Kraeuter, J.N., and Flimin, G. 2012. Status and Trends of Hard Clam, *Mercenaria mercenaria*, Shellfish Populations in Barnegat Bay, New Jersey. Barnegat Bay Partnership Technical Report. Toms River, Barnegat Bay Partnership**:** 143.

Bricker, S., Longstaff, B., Dennison, W., Jones, A., Boicourt, K., Wicks, C., and Woerner, J. 2007. Effects of Nutrient Enrichment In the Nation's Estuaries: A Decade of Change. NOAA Coastal Ocean Program Decision Analysis Series No. 26. National Centers for Coastal Ocean Science, Silver Spring, MD. 328 pp.

Buckley, L.J., Smigielski, A.S., Halavik, T.A., Caldarone, E.M., Burns, B.R., and Laurence, G.C. 1991. Winter flounder *Pseudopleuronectes americanus* reproductive success. I. Among-location variability in size and survival of larvae reared in the laboratory. Marine Ecology Progress Series 74: 117-124.

Calvo, G. 2010. 2008-2009 Oyster Restoration Activities at Good Luck Point, Barnegat Bay. New Jersey Bureau of Shellfisheries, pp. 1-12.

Celestino, M. P. 2002. Shellfish stock assessment of Little Egg Harbor bay. New Jersey Department of Environmental Protection, pp. 1-47.

Celestino, M.P. 2013. Shellfish stock assessment of Little Egg Harbor bay (2011). New Jersey Department of Environmental Protection, pp. 1-38.

Chen, Z., Xu, S., Qiu, Y., Lin, Z., and Jia, X. 2009. Modeling the effects of fishery management and marine protected areas on the Beibu Gulf using spatial ecosystem simulation. Fisheries Research 100: 222-229.

Christensen, V. and Walters, C.J. 2004. Ecopath with Ecosim: methods, capabilities and limitations. Ecological Modelling 172: 109-139.

Christensen, V., Beattie, A., Buchanan, C., Ma, H., Martell, S.J.D., Latour, R.J., Preikshot, D., Sigrist, M.B., Uphoff, J.H., Walters, C.J., Wood, R.J. and Townsend, H., 2009. Fisheries Ecosystem Model of the Chesapeake Bay: Methodology, Parameterization, and Model Explanation. U.S. Dep. Commerce, NOAA Tech. Memo. NMFS-F/SPO-106, 146 pp.

Christensen, V., Walters, C.J., Pauly, D, and Forrest, R. 2008. Ecopath with Ecosim version 6: a User's Guide, November 2008 Edition. Fisheries Centre, University of British Columbia, Vancouver, Canada.

Dalsgaard, J., and Pauly, D. 1997. Preliminary mass-balance model of Prince William Sound, Alaska, for the pre-spill period, 1980-1989. Fisheries Centre Research Report 5(2): 34 pp.

Duffy, J.T., and Epifanio, C.E. 1994. Effects of larval density on the growth and survival of weakfish *Cynoscion regalis* in large-volume enclosures. Marine Ecology Progress Series 104: 227-233.

EA Engineering, Science, and Technology Inc. 1986. Entrainment and Impingement Studies at Oyster Creek Nuclear Station 1984-1985. GPU Nuclear Corporation, pp. 1-208.

Ecosystem Principals Advisory Panel. 1998. Ecosystem based fishery management: a report to the Congress. Available from: www.nmfs.noaa.gov/sfa/EPAPrpt.pdf

Food and Agricultural Organization (FAO). 2009. State of the World's Fisheries and Aquaculture 2008. Food and Agriculture Organization of the United Nations. FAO, Rome, Italy.

Festa, P.J. 1978. Analysis of the fish forage base in the Little Egg Harbor estuary. New Jersey Division of Fish, Game, and Shellfisheries. Trenton, New Jersey. 324 pp.

Fouzai, N., Coll, M., Palomera, I., Santojanni, A., Arneri, E., and Christensen, V. 2012. Journal of Marine Systems 102-104: 39-51.

Franz, D.R., and Tanacredi, J.T. 1992. Secondary production of the amphipod *Ampelisca abdita* and its importance in the diet of juvenile winter flounder (*Pleuronectes americanus*) in Jamaica Bay, New York. Estuaries 15: 193-203.

Frisk, M.G., Miller, T.J., Latour, R.J., and Martell, S. 2006. An ecosystem model of Delaware Bay.

Frisk, M.G., Miller, T.J., Latour, R.J., and Martell, S, 2011. Assessing biomass gains from marsh restoration in Delaware Bay using Ecopath with Ecosim. Ecological Modelling 222: 190-200

Froese, R., and Pauly, D. 2004. FishBase World Wide Web electronic publication, www.fishbase.org,version Accessed: March 2013.

Hare, J.A., and Able, K.W. 2007. Mechanistic links between climate and fisheries along the east coast of the United States: explaining population outbursts of Atlantic croaker (*Micropogonias undulatus*). Fisheries Oceanography. 16: 31-45.

Haskin, H.H. and Ray, G. 1977. Estuarine evaluation study: Benthic invertebrates. Four year report, 1973-1977. A report submitted to the New Jersey Department of Environmental Protection. Center for Coastal and Environmental Studies, Rutgers University. New Brunswick. 300 pp.

Hettler, W.F. 1976. Influence of temperature and salinity on routine metabolic rate and growth of young Atlantic menhaden. Journal of Fish Biology 8: 55-65.

Hilborn, R. 2011. Future directions in ecosystem based fisheries management: A personal perspective. Fisheries Research. 108: 235-239.

Hoenig, J. M. 1983. Empirical Use of Longevity Data to Estimate Mortality-Rates. Fishery Bulletin 81: 898-903.

Howson, U. 2013. Baseline survey of zooplankton of Barnegat Bay. New Jersey Department of Environmental Protection, pp. 1-92.

ICES. 2000. Report of the working group on seabird ecology, ICES CM 2000/C:04

Kahn, D. M. 2003. Stock assessment of Delaware Bay blue crab (*Callinectes sapidus*) for 2003. Division of Fish and Wildlife, Dover, DE.

Joseph, J.W. 1986. Inventory of New Jersey's estuarine shellfish resources. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Rockville, MD.

Joseph, J.W. 1987. Inventory of New Jersey's estuarine shellfish resources. US Department of Commerce, National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Rockville, MD.

Kennish, M.J. 2001a. Physical description of the Barnegat Bay—Little Egg Harbor estuarine system. Journal of Coastal Research SI(32): 13-27.

Kennish, M.J. 2001b. The Scientific Characterization of the Barnegat Bay – Little Egg Harbor Estuary and Watershed. Jacques Cousteau National Estuarine Research Reserve Contribution #100-5-01.

Kennish, M.J., Ferting, B.M.. and Sakowicz, G.P. 2013. In situ Surveys of Seagrass Habitat in the Northern Segment of the Barnegat Bay - Little Egg Harbor Estuary: Eutrophication Assessment. Barnegat Bay Partnership Technical Report. 43 pp.

Lathrop, R.G., and Haag, S.M. 2011. Assessment of Seagrass Status in the Barnegat Bay - Little Egg Harbor Estuary System, pp. 1-58.

Lathrop, R.G., Styles, R.M., Seitzinger, S.P., and Bognar, J.A. 2001. Use of GIS mapping and modeling approaches to examine the spatial distribution of seagrasses in Barnegat Bay, New Jersey. Estuaries 24: 904-916.

Laurence, G.C. 1975. Laboratory growth and metabolism of the winter flounder *Pseudopleuronectes americanus* from hatching through metamorphosis at three temperatures. Marine Biology 32: 223-229.

Leak, J.C., and Houde, E.D. 1987. Cohort growth and survival of bay anchovy *Anchoa mitchilli* larvae in Biscayne Bay, Florida. Marine Ecology Progress Series 37: 109-122.

Libralato, S., Christensen, V. and Pauly, D. 2006. A method for identifying keystone species in food web models. Ecological Modelling 195: 153-171.

Lowerre-barbieri, S. K., Chittenden M. E., and Barbieri L. R. 1995. Age and Growth of Weakfish, Cynoscion Regalis, in the Chesapeake Bay-Region with a Discussion of Historical Changes in Maximum Size. Fishery Bulletin 93: 643-656.

Luo, J. and Brandt, S.B. 1993. Bay anchovy, *Anchoa mitchilli*, production and consumption in mid-Chesapeake Bay based on a bioenergetics model and acoustic measurement of fish abundance. Marine Ecology Progress Series 98: 223-236.

Macro International Inc. 2008. New Jersey Blue Crab Recreational Fishery Survey 2007 Final Report.

Matishov, G.G. and Denisov, V.V. 1999. Ecosystems and biological resources of Russian European seas at the turn of the 21st century, Murmansk Marine Biological Institute, Murmansk

McClain, J., Makai, J., and Himchak, P. 1976. Studies of the Manahawkin Bay – Little Egg Harbor system: finfish study, physical-chemical study, and use study. New Jersey Division of Fish, Game, and Shellfisheries. 463 pp.

Nemerson, D. M., and Able K. W. 2004. Spatial patterns in diet and distribution of juveniles of four fish species in Delaware Bay marsh creeks: factors influencing fish abundance. Marine Ecology Progress Series 276: 249-262.

Oshima, Y., Kishi, M.J. and Sugimoto, T. 1999. Evaluation of the nutrient budget in a seagrass bed. Ecological Modelling 115: 19-33.

Palomares, M. L. D. 1991. La consummation de nourriture chez les poissons: étude comparative, mise au point d'un modèle prédictif et application à l'étude des réseaux trophiques. Thèse de Doctorat, Institut National Polytechnique de Toulouse: 211.

Palomares, M.L.D. and Pauly, D. 1998. Predicting food consumption of fish populations as functions of mortality, food type, morphometrics, temperature and salinity. Marine and Freshwater Research 49: 447-453.

Pauly, D. 1989. Food consumption by tropical and temperate fish populations: some generalizations. Journal of Fisheries Biology 35: 11-20

Pauly, D., Christensen, V., and Walters, C. 2000. Ecopath, Ecosim, and Ecospace as tools for evaluating ecosystem impact of fisheries. ICES Journal of Marine Science 57: 697-706.

Pew Oceans Commission. 2003. America's Living Oceans: Charting a course for sea change. Washington, DC. 166 pp.

Piner, K. R., and Jones C. M. 2004. Age, growth and the potential for growth overfishing of spot (*Leiostomus xanthurus*) from the Chesapeake Bay, eastern USA. Marine and Freshwater Research 55: 553-560.

Plagányi, É.E. 2007. Models for an ecosystem approach to fisheries. FAO Fisheries Technical Paper. No. 477. Rome, FAO. 108 pp.

Preikshot, D. 2007. The influence of geographic scale, climate and trophic dynamics upon North Pacific oceanic ecosystem models. Ph.D. Dissertation. University of British Columbia, Vancouver

Randall, R.G. and Minns, C.K. 2000. Use of fish production per unit biomass ratios for measuring the productive capacity of fish habitats. Canadian Journal of Fisheries and Aquatic Sciences 57:1657-1667.

Ren, L. 2013. Baseline characterization of phytoplankton and harmful algal blooms in Barnegat Bay-Little Egg Harbor, New Jersey. New Jersey Department of Environmental Protection, pp. 1-100.

Rosenberg, A.A., and McLeod, K.L. 2005. Implementing ecosystem-based approaches to management for the conservation of ecosystem services. Marine Ecology Progress Series 300: 270-274.

Sellner, K.G., Fisher, N., Hager, C.H., Walter , J.F. and Latour, R.J. 2001. Ecopath with Ecosim Workshop, Patuxent Wildlife Center, October 22-24, 2001, Chesapeake Research Consortium, Edgewater, MD

Shushkina, E.A., Musaeva, E.I., Anokhina, L.L. and Lukasheva, T.A. 2000. The role of gelatinous macroplankton, jellyfish *Aurelia*, and Ctenophores *Mnemiopsis* and *Beroe* in the planktonic communities of the Black Sea. Russian Academy of Sciences. Oceanology 40: 809-816.

Smith, D.R., Burnham, K.P., Kahn, D.M., He, X. and Goshorn, C.J. 2000. Bias in survival estimates from tag-recovery models where catch-and-release is common, with an example from Atlantic striped bass. Canadian Journal of Fisheries and Aquatic Sciences 57: 886-997

Summers, J.K., Holland, A.F., Weisburg, S.B., Wendling, L.C., Stroup, C.F., Dwyer, R.L., Turner, M.A., Burton, W. 1989. Technical Review and Evaluation of Thermal Effects Studies and Cooling Water Intake Structure Demonstration of Impact For the Oyster Creek Nuclear Generating Station. Versar, Inc., pp. 1-500.

Taghon, G.L., Grassle, J.P., Fuller, C.M., Petrecca, R.F., Ramey, P. 2013. Benthic Invertebrate Community Monitoring and Indicator Development of Barnegat Bay-Little Egg Harbor Estuary. Rutgers University, pp. 1-92.

Ulanowicz, R.E. 1986. Growth and Development: Ecosystem Phenomenology. Springer Verlag (reprinted by iUniverse, 2000), New York, 203 pp.

Ulanowicz, R.E. and Puccia, C.J. 1990. Mixed trophic impacts in ecosystems. Coenoses 5: 7-16.

United States Census Bureau. 2012. State and County QuickFacts. http://quickfacts.census.gov/qfd/states/34/34029.html. Accessed: 09 January 2014.

U.S. Commission on Ocean Policy. 2004. An Ocean Blueprint for the 21st Century. Final Report. Washington, DC.

Vasslides, J.M., Rackovan, J.L., Toth, J.L., Hagan, R., Able, K.W. 2011. Metadata Manual for Fish and Environmental Records at the Rutgers University Marine Field Station. Institute of Marine and Coastal Sciences, Rutgers University, 173 pp.

Vouglitois, J.J., Able, K.W., Kurtz, R.J., and Tighe, K.A. 1987. Life history and population dynamics of the bay anchovy in New Jersey. Transactions of the American Fisheries Society 116: 141-153.

Walters, C., Christensen, V., Walters, W., and Rose, K. 2010. Representation of multistanza life histories in Ecospace models for spatial organization of ecosystem trophic interaction patterns. Bulletin of Marine Science 86: 439-459.

Walters, C.J. and Martell, S.J. 2004. An Overview of Single-Species Assessment Models, in Fisheries Ecology and Management. Princeton University Press. Princeton

Walters, C., Pauly, D., and Christensen, V. 1999. Ecospace: Prediction of mesoscale spatial patterns in trophic relationships of exploited ecosystems, with emphasis on the impacts of marine protected areas. Ecosystems 2: 539-554.

Appendix 1 – Ecopath Parameter Derivations

FISH

Atlantic Croaker

Q/B - Estimates of consumption to biomass ratio was calculated in FishBase (Froese and Pauly, 2004) as 4.2 year^{-1}, assuming an annual temperature of the Barnegat Bay of T = 15 °C, aspect ratio = 1.32, W*inf* = 815.3, and carnivorous feeding.

P/B - An annual total mortality for the Chesapeake Bay Atlantic croaker stock was estimated to be 55 to 60% per year (Christensen *et al*. 2009). Using the higher end as a conservative mortality estimate yields a P/B = 0.916 year^{-1}.

Biomass – An EE value of 0.90 was used and EwE estimated the biomass. Croaker were rarely identified in the McClain *et al*. (1976) study and thus the Delaware Bay and Chesapeake models likely overestimate the biomass present here.

Diet – The diet data is based on the general diet found in the Delaware Bay model, which is a composite of the Nemerson and Able (1994) study.

Atlantic Menhaden

Q/B – A value of 31.42 year^{-1} taken from Palomares and Pauly (1998).

P/B – As there was no commercial fishery for menhaden in Barnegat Bay and only a limited bait fishery, total mortality was set equal to natural mortality, which is estimated at 0.50 year^{-1} (MSVPA-X averaged across all ages and 1982-2008; in 2010 Stock Assessment Table 2.13).

Biomass – Biomass was calculated by EwE setting the EE to 0.95.

Diet – Diet data is from Festa 1978.

Atlantic Silverside

Q/B – The consumption ratio for silversides of 4.0 year^{-1} was determined by setting a production/consumption ratio of 0.2 (Christensen *et al.* 2009).

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 year^{-1}.

Biomass - The biomass for the group was estimated by setting ecotrophic efficiency to 0.95. While baywide biomass was not determined by Vougliotis *et al.* (1987), they suggested it should be comparable, if not great than what they determined for bay anchovy, given Atlantic silverside was numerically dominant.

Diet – Diet data is from Festa 1978.

Bay Anchovy

Q/B - Assuming habitat temperature of 15 °C, $W\infty$ = 20 (g), an aspect ratio of 1.32, and carnivorous diet, the consumption to biomass ratio is calculated by Fishbase to be 9.7 year^{-1}.

P/B – Christensen *et al.* used an initial P/B of 3.0 year^{-1} for the Chesapeake Bay model based on a 95% annual mortality rate reported by Luo and Brandt (1993), while Frisk *et al.* (2006) estimated a P/B of 2.19 year^{-1} from catch curve analysis on adults in Delaware Bay. We elected to use the higher rate.

Biomass – Vougliotis *et al.* (1987) estimated biomass for 1976 to range from 0.83 to 4.83 g/m^2. In the same study the catch per unit effort for 1981 was comparable to that for 1976, and thus the biomass range should be similar. Given the ubiquity of the species within the Barnegat Bay, we chose to use 4.83g/m^2 for an initial biomass.

Diet - Diet data is from Festa 1978.

Bluefish

Q/B - Assuming habitat temperature of 15 °C, Wmax = 16,962.1 (g), carnivorous feeding, and an aspect ratio of 2.55, the resulting consumption to biomass ratio is 3.1 $year^{-1}$.

P/B – Production/biomass was determined as 0.52 $year^{-1}$ based on an M = 0.25 $year^{-1}$ (Christensen *et al.*2009) and an estimate of F = 0.27 $year^{-1}$ for 1982 from the 41st Stock Assessment Workshop (2005) for Bluefish (Figure B2).

Biomass – Biomass was calculated by EwE setting the EE to 0.95.

Diet – Diet data is from Festa 1978 averaged for all size classes.

Mummichog

Q/B – A Q/B of 3.65 $year^{-1}$ was used (Pauly 1989).

P/B – We opted to utilize a P/B of 1.2 $year^{-1}$ as given in Frisk *et al.* (2006) from "best professional judgement" compared to Valiela 0.287 $year^{-1}$ (1977 mortality tables) or Christensen *et al.*'s 0.8 $year^{-1}$.

Biomass- The biomass for the group was estimated by setting ecotrophic efficiency to 0.95.

Diet – Diet data is from Festa 1978.

River herring

Q/B – We used a Q/B = 8.4 $year^{-1}$, which is the average of Pauly (1989; 8.63 at temperature = 10°C) and Palomares (1991; 8.23 at temperature= 20°C).

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

Biomass – Biomass was estimated by EcoPath assuming that the ecotrophic efficiency of these species in the Bay was 0.95.

Diet – Diet data is from Festa 1978.

Spot

Q/B – The consumption biomass ratio was estimated as 6.2 $year^{-1}$ using the model in Fishbase.org and a habitat temperature of 15 °C, $W\infty$ = 190g (Piner and Jones, 2004) and an aspect ratio of 1.39 (Christensen *et al.*).

P/B - Hoenig's method estimated an M = 0.9 $year^{-1}$ given a maximum age of 5 (Piner and Jones, 2004). This is consistent with the Z used in the Delaware Bay model.

Biomass – Biomass was estimated by the software using an EE value of 0.90, which was taken from the Chesapeake Bay model.

Diet – Diet data is from Festa 1978.

Striped bass

Q/B - Based on empirical relationship provided by Fishbase.org and assuming an aspect ratio of 2.31 (Chesapeake Bay Ecopath Model), temperature T = 15 °C, and $W\infty$ = 46.6 kg, the estimated consumption ratio was 2.4 $year^{-1}$.

P/B – The 1981 ASMFC (2003) FMP suggest an M=.15 and an F=.3 for the coastwide stock. Given the reduced fishing mortality in the Barnegat Bay, an F=.25 is appropriate leading to a P/B of 0.4 $year^{-1}$. This is equal to the Chesapeake model for resident bass (1-7 years old), though their YOY P/B = 1.8 $year^{-1}$.

Biomass – The biomass was estimated by EcoPath based on an EE of 0.90.

Diet – Diet data is from Festa 1978 and was averaged across all size classes.

Summer Flounder

Q/B- Assuming an aspect ratio of 1.32, Wmax = 12kg (Frisk *et al.* 2006), carnivorous feeding, and habitat temperature of 15 °C, the consumption to biomass ratio is = 2.6 year^{-1}.

P/B- The Chesapeake Bay and Delaware Bay models utilized P/B=0.52 year^{-1} based on the 2002 NEFSC determination of M=0.2 and F ranging between 0.24 and 0.32.

Biomass – Biomass was estimated by the software using an EE value of 0.95.

Diet – Diet data is from Festa 1978.

Weakfish

Q/B - Using Fishbase, consumption to biomass was estimated = 3.0 year^{-1}, assuming average habitat temperature of 15 °C, aspect ratio of 1.32, maximum weight W∞ = 6,190g (Lowerre-Barbieri *et al.*., 1995) and carnivorous feeding habitats.

P/B –Total mortality of Z = 0.26 year^{-1} was estimated using Hoenig's method (1983) assuming a longevity of 17 years (Lowerre-Barbieri *et al.*., 1995). This is in-line with an estimated M of .25 year^{-1} as used for stock assessment purposes (Smith *et al.*, 2000). Given the low rate of fishing in Barnegat Bay, Hoenig's estimation of Z seems reasonable.

Biomass – Biomass was estimated by the software using an EE value of 0.90.

Diet – Diet data is from Festa 1978 and averaged across all size classes.

Winter Flounder

Q/B - The estimated consumption ratio of 3.4 year^{-1} was derived using the empirical equation in FishBase and was calculated assuming that T = 15 °C, Winf = 3,600 g (Fishbase), an aspect ratio of 1.32, and a carnivorous diet.

P/B – The 2011 Southern New England/Mid-Atlantic stock assessment updated natural mortality (M) to 0.30 year^{-1} for all ages and all years. Fishing mortality for ages 4-6 was determined as 0.61 year^{-1} for 1981. If one assumes only natural mortality for ages 0-3 and then F+M for ages 4-6, total mortality (Z) is 0.52 averaged across all ages.

Biomass – Biomass was estimated by the software using an EE value of 0.95.

Diet – Diet data is from Festa 1978.

Piscivorous seabirds

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

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

Biomass - The biomass estimate for piscivorous seabirds of 0.25 t · km-2 is a reduction of the Chesapeake Bay model estimate (Sellner *et al.* 2001).

Diet compositions - The diet composition for piscivorous seabirds was taken from the Chesapeake Bay model (Christensen *et al.* 2009) and was modified by reducing predation on menhaden and increasing imports based on the large number of migratory seabirds.

Non-Piscivorous seabirds

Q/B - The consumption ratio estimate of 120 year^{-1} was from data for the non-piscivorous seabirds group in Preikshot (2007).

P/B - A total mortality estimate for non-piscivorous seabirds of 0.51 year^{-1} was taken from the Chesapeake model and was based on annual mortality rate of 37% for mallard males and 44% females (Anderson, 1975).

Biomass - The biomass estimate for non-piscivorous seabirds of 0.121 t · km-2 was taken from the Chesapeake Bay model and was based on advice provided in a Chesapeake Ecopath Workshop (Sellner *et al.* 2001).

Diet compositions - The diet composition for non-piscivorous seabirds was taken from the Chesapeake Bay model (Christensen *et al.* 2009).

INVERTEBRATES

Blue crabs

Q/B- The consumption ratio of 4.0 $year^{-1}$was taken from the Chesapeake Bay model (Christensen *et al.* 2009).

P/B – The Delaware Bay model utilized a P/B= 1.21 $year^{-1}$. This was based on a stock assessment for Delaware Bay that used a natural morality of M = 0.8 $year^{-1}$ assuming a lifespan of 4 years (Kahn, 2003) and fishing mortality on total stock (recruits and post recruits) was F = 0.41 $year^{-1}$ (2000-2002).

Biomass – Biomass was estimated using an ecotrophic efficiency of 0.95.

Diet – Diet taken from Chesapeake Bay model (Christensen *et al.* 2009), averaged across stanzas.

Hard Clams

Q/B - The consumption ratio was estimated to be 5.1 $year^{-1}$ assuming a P/Q = 0.20 (Christensen *et al.* 2009)

P/B - A production/biomass ratio of 0.5 $year^{-1}$ was used based on the empirical studies of Hibbert (1976).

Biomass – A value of 26.18 t/km2 was used based on a density of 1,309,233 clams per km^2 (adjusted values for the 1985-1987 surveys, Celestino 2002) and an average mass of 20 g (mean length of 7.46cm, Celestino 2013).

Diet – Diet taken from Christensen *et al.* (2009).

Oyster

Q/B - The Q/B ratio of 2.0 $year^{-1}$ was taken from the adult stanza of the Chesapeake Bay Model (Christensen *et al.* 2009).

P/B – A 2009 survey of the restored oyster reef at Good Luck Point determined a mean annual mortality of 47%, or an M=0.63 $year^{-1}$ (Calvo 2010). As oysters in Barnegat Bay are an unfished resource, Z=M=.63 $year^{-1}$.

Biomass – Based on NJDEP experience there does not appear to be a viable oyster set in Barnegat Bay; the known oyster reef is seeded by the NJDEP and others. In order to keep oysters in the model for future management considerations the biomass was set to 0.001t/km^2 to simulate a very small population.

Diet – Data taken from the Chesapeake model (Christensen *et al.* 2009).

Sea Nettles

Q/B – A Q/B of 20 $year^{-1}$ was taken from the Chesapeake Bay model (Christensen *et al.* 2009). This value is based on an assumed P/Q of 0.25.

P/B – As reported in the Christensen *et al.* (2006), Matishov and Denisov (1999) estimated a daily growth rate for *Aurelia aurita* of 0.053 at 5 ºC to 0.15 at 16.5 ºC. Sea nettle medusa are present in the Barnegat Bay during the summer months, when waters are typically warmer than 16.5 ºC. As such the P/B for Barnegat Bay was calculated as (0.15*365)/4 ~ 13 $year^{-1}$.

Biomass – In 2012, baywide survey data from Monmouth University led to an estimate of 24,711 individuals per km^2. Assuming an average wet weight of 56g per individual, this translates to a biomass of 1.38t/km^2. Recent work by Young et al. (in prep) suggests that sea nettle abundances were at very low levels (unrecorded in contemporary research) until the late 1990's, and did not reach current levels until the mid-2000s. Therefore we estimate that sea nettle biomass was about 1/4th of the current biomass, or 0.345t/km^2.

Diet – The sea nettle diet data was taken from the Chesapeake Bay model (Christensen *et al.* 2009).

Ctenophores

Q/B - Shushkina *et al.* (1989) found that ctenophores in their study had growth rates 1.5 to 2 times greater than true jellyfish. Therefore, the Q/B value for ctenophores was the value for sea nettles multiplied by 1.75, *i.e.*Q/B was 35 year^{-1}.

P/B – Shushkina *et al.* (1989) found that ctenophores in their study had growth rates 1.5 to 2 times greater than true jellyfish. Ctenophores tend to be present in Barnegat Bay at cooler temperatures than those of sea nettles, therefore the P/B was calculated as 1.75 times the average estimated daily growth rate of *Aurelia aurita* over the course of 3 months $((((0.053+0.15)/2)*365)/4)*1.75 \sim 16.2$ year^{-1}.

Biomass – A biomass of 7.86 t/km^2 was calculated using bay-wide survey data collected by Monmouth University during 2012 and an average weight of 3.42g per individual.

Diet - The ctenophore diet data was taken from the Chesapeake Bay model (Christensen *et al.* 2009).

Benthic infauna/epifauna (shrimp, worms, non-blue claw crabs)

Q/B – A consumption ration of 5.0 year^{-1} was estimated by Ecopath after designating a P/Q ratio of 0.2, as taken from the Chesapeake Bay Model (Christensen *et al.* 2009).

P/B – A P/B of 2.0 year^{-1} was taken from the Chesapeake Bay model (Christensen *et al.* 2009).

Biomass – Estimated by Ecopath, based on a group ecotrophic efficiency of 0.9 as taken from the Chesapeake Bay model (Christensen *et al.* 2009).

Diet – Diet data taken from Chesapeake Bay model (Christensen *et al.* 2009).

Amphipods

This category consists mainly of the genus *Ampelisca* (*A. abdita* and *A.verrilli*) and *Elasmopus levis* based on the work conducted by Haskin and Ray (1979) and Taghon et al (2012).

Q/B – Ecopath estimated a Q/B = 5.0 year^{-1} using a P/Q ratio of 0.2, following the Chesapeake Bay model (Christensen *et al.* 2009).

P/B – A P/B of 3.8 year^{-1} was used based on the average P/B of *Ampelisca abdita* at 3 locations within Jamaica Bay (Franz and Tanacredi 1992). *A. abdita* was the most common amphipod found in Barnegat Bay sampling in 2012.

Biomass – The biomass of amphipods was estimated by Ecopath using an EE=0.900. We attempted to utilize the first year of NJDEP Barnegat Bay research program data (Taghon *et al*. 2012), which is the only study of amphipod density bay-wide, though it is restricted to summer sampling only. A 1974/1975 study (Haskin and Ray 1979) documented amphipod density throughout the year, but on a limited spatial scale. In the 1974/75 study the average yearly density across all sites was approximately 2.5 times larger than the summer density during the same time period. To estimate amphipod biomass, the average density of the 2012 study was multiplied by 2.5, and the resulting density multiplied by the weight of an average amphipod (0.003g) to reach an estimate of 1.53g/m^2. This empirically determined biomass is approximately one-half of the biomass required to balance the model as found by Ecopath.

Diet – The diet data for this group was taken from the benthic infauna group.

Copepods (Mesozooplankton)

Q/B – A consumption ration of 83.333 year $^{-1}$ was estimated by Ecopath after designating a P/Q ratio of 0.3, as taken from the Chesapeake Bay Model (Christensen *et al.* 2009).

P/B – A mortality rate of 25 year $^{-1}$ was taken from the Chesapeake Model, as estimated during the Chesapeake Bay Ecopath Workshop.

Biomass – Copepod biomass was estimated using an ecotrophic efficiency of 0.95.

Diet – The diet ratio, 72% microzooplankton, and 28% phytoplankton is from the Chesapeake Bay model (Christensen *et al.* 2009).

Microzooplankton

Q/B – A consumption ration of 350 year $^{-1}$ was estimated by Ecopath after designating a P/Q ration of 0.4, as taken from the Chesapeake Bay Model (Christensen *et al.* 2009).

P/B – A total mortality rate for microzooplankton of 140 year^{-1} was taken from the Chesapeake Bay model (Christensen *et al.* 2009).

Biomass – Biomass was estimated based on an assumed EE of 0.95.

Diet – The 100% phytoplankton diet follows the Chesapeake Bay model (Christensen *et al.* 2009).

Phytoplankton

P/B – We elected to use the Chesapeake value of 160 year^{-1} over the Delaware Bay value of 60 year^{-1} as the Chesapeake is a highly eutrophic system more similar to the conditions found in Barnegat Bay (Christensen *et al.* 2009).

Biomass – Biomass was estimated by the software assuming an ecotrophic efficiency of 0.95. We originally attempted to calculate a biomass estimate using the August 2011 to September 2012 data (ugC/L) collected as part of the Governor's Barnegat Bay Initiative and a conversion ratio of 10 mg wet weight:mg C (Emax report, Dalsgaard and Pauly 1997). This lead to an estimated wet weight of 7.705 t/km2. However, this biomass is far too small to support the grazing pressure calculated.

Benthic algae

P/B – The Chesapeake model assumed a value of 80 year^{-1} (Christensen *et al.* 2009).

Biomass – Biomass of benthic algae was estimated based on an assumed EE of 0.9 (Christensen *et al.* 2009).

SAV

P/B – Mortality for *Z. marina* was estimated in the Chesapeake as Z = P/B =5.11 year^{-1}, which was taken from a similar system in Japan (Oshima *et al.*., 1999).

Biomass – In 1979 there was approximately 8,053 ha of mapped submerged aquatic vegetation (Northern segment: 767, Central segment: 5,126, Southern segment: 2,160) out of the 27,900 hectares of Barnegat Bay (Lathrop *et al.* 2001). The highest recorded annual eelgrass maximum biomass in the southern and central portions of the bay occurred in 2004 and was 219.7 g dry wt /m^2, while the highest Ruppia biomass recorded in the northern segment occurred in 2011 and was 32.8 g dry wt/ m^2 (Kennish *et al.* 2013). Expanding the biomass estimates over the 1979 SAV acreage yields a baywide total biomass of 1,625.891t, or 5.82t/km^2

Appendix 2 – Ecopath Initial Diet Composition

	Piscivorous seabirds	Non-piscivorous	weakfish	striped bass	summer flounder	bluefish	winter flounder	Atlantic silversides	Atlantic croaker	spot	Atlantic menhaden
Piscivorous seabirds											
Non-piscivorous											
weakfish	0.0056			0.2		0.013					
striped bass	0.0166										
summer flounder	0.011										
bluefish	0.02										
winter flounder	0.0058				0.2						
Atlantic silversides	0.017		0.05	0.221	0.132	0.087					
Atlantic croaker			0.005	0.01		0.005					
spot			0.03			0.011					
Atlantic menhaden	0.1			0.206		0.255					
river herring	0.028										
mummichog	0.03					0.36					
bay anchovy	0.07		0.535	0.2	0.273	0.094	0.018				
benthic infauna/epifauna		0.276	0.352	0.06	0.186	0.066	0.742	0.59	0.8	0.509	0.18
amphipods			0.022				0.07	0.244		0.25	
blue crabs	0.004		0.006	0.1	0.2	0.103	0.002				
hard clams		0.01		0.003			0.157			0.057	
oysters											
copepods								0.154	0.2	0.18	0.338
Microzooplankton											
sea nettles											
ctenophores											
benthic algae											
phytoplankton											0.421
SAV		0.128									
detritus		0.011			0.009	0.006	0.011	0.012		0.004	0.061
import	0.692	0.575									

	river herring	mummichog	bay anchovy	benthic infauna	amphipods	blue crabs	hard clams	oysters	copepods	Micro zoo	sea nettles	ctenophores
Piscivorous seabirds												
Non-piscivorous												
weakfish												
striped bass												
summer flounder												
bluefish												
winter flounder												
Atlantic silversides												
Atlantic croaker												
spot												
Atlantic menhaden												
river herring												
mummichog												
bay anchovy											0.054	
benthic infauna/epifauna	0.435	0.260		0.02	0.02	0.5						
amphipods	0.055	0.170	0.044									
blue crabs						0.125						
hard clams						0.175						
oysters												
copepods	0.5	0.19	0.582								0.421	0.666
Microzooplankton			.370	0.08	0.08				0.72			0.334
sea nettles												
ctenophores											0.525	
benthic algae		0.12		0.3	0.3	0.05	0.5					
phytoplankton	0.005			0.4	0.4		0.25	0.99	0.28	1		
SAV						0.05						
detritus	0.005	0.26	0.004	0.2	0.2	0.1	0.25	0.01				
import												

Appendix 3 - Landing Calculations for the Barnegat Bay Ecopath Model

Directed Fisheries

The National Marine Fisheries Service (NMFS) commercial landings database is the most comprehensive record of commercial landings available for the time period of interest (1950-2011). However, these data represent landings for all of New Jersey, and are not Barnegat Bay specific. The NMFS landings data used below are a subset of the statewide landings based on gear that could be used within an estuary. Gear types considered usable in the bay include the following: by hand; cast nets; dip nets, common; fyke and hoop nets, fish; hand lines, other; pots and traps, blue crab; and weirs. Because these gear types have been used in the Barnegat Bay as well as other larger estuaries throughout the state (Raritan Bay, Delaware Bay, *etc.*), this subset likely overestimates commercial removals from Barnegat Bay. Where Barnegat Bay specific landings data are available they were used to the maximum extent possible.

Recreational landings for finfish were taken from the NMFS Marine Recreational Fisheries Statistics Survey (MRFSS) for Ocean County, inland waters only. The landings for 1981 were used to initialize the model as that is the earliest year for which data is available.

The source and calculations for each species are described below.

Atlantic croaker – Based on the subset of NMFS commercial landing data, there was no harvest of Atlantic croaker reported in the 1980s. There were no recreational landings of croaker reported for Ocean County.

Atlantic Menhaden – There was no commercial harvest of menhaden recorded in the NMFS landing data for the gear types used in Barnegat Bay in 1980. There were no recreational landings of menhaden reported for Ocean County in the MRFSS database. Menhaden are commonly used as bait in the recreational fishery in Barnegat Bay, therefore an estimated landing of 0.2MT was attributed to the recreational fishery, though this likely underestimates landings.

Blue Crab – In Barnegat Bay the commercial blue crab fishery can be divided into a winter dredge fishery and a pot/trap line fishery in the remainder of the year. Landings data specific to Barnegat Bay were available from the NJDEP for 1995-2013, while statewide landings were available from NMFSS for 1980-2011. The NJDEP data was regressed on the NMFS data and the results used to calculate bay specific total landings for 1981-1994. The winter dredge fishery represented approximately 17% of the baywide total (NJDEP data); this ratio was used to estimate the gear specific landings from the total baywide landings of 221 metric tons for 1981. Therefore the winter dredge fishery in 1981 landed an estimated 38.1 metric tons while the pots and trot lines accounted for an estimated 183.3 metric tons. In 2007 the recreational harvest of blue crabs in Barnegat Bay was estimated to be 80% of the total commercial harvest (Macro International Inc. 2008), leading to an estimated recreational harvest of 177.1 metric tons in 1981.

Bluefish – Barnegat Bay specific commercial landings were available for bluefish for 1997 only (Kennish 2001). The bay specific landings represented 21% of the subset landings for that year (NMFS). That ratio was utilized to calculate an estimated Barnegat Bay specific commercial landing of 0.02 metric tons for 1980. In 1981approximately 209.1 metric tons of bluefish were landed in Ocean County inland waters (MRFSS).

Hard Clam – Hard clams are historically one of the most important commercial fishery resources in Barnegat Bay. However there are no records for commercial or recreational landings of hard clams specific to Barnegat Bay. Based on recommendations of Bureau of Shellfisheries staff and long-time commercial clammers an estimate of 0.39t/km^2 was developed assuming 40 individuals harvesting 3 bushels per day working 200 days per year, with a bushel weighing approximately 10lbs. There are no estimates of hard clam recreational landings available.

River herring – Alewife and blueback herring have been combined into this single category given the similarities in their life history strategies and propensity to co-migrate. In 1981 there were no commercial landings of either species in the subset landings, and no landings reported for Ocean County's recreational inland fishery. However, there were known fisheries for river herring within the bay associated with bait collection. As such a total landing of 0.1MT was assumed based on the landings in subsequent years and split evenly between the recreational and commercial sectors.

Spot –There were no commercial landings of spot recorded in the subset landing data for the late 1970s through mid-1980s. There were 1.1 metric tons of spot landed in the Ocean County inland recreational fishery in 1981.

Striped Bass – In 1981 there were no commercial landings of striped bass recorded in the subset landing data. There were no landings reported for Ocean County's recreational inland fishery. However, there was a well-documented recreational fishery present at the time, therefore 26 MT was used, which is the average of reported landings from 1981-201.

Summer flounder – Commercial landings of summer flounder approached 0.2 metric tons in 1981 according to the subset NMFS database. There were 224.4 metric tons of summer flounder landed in the Ocean County inland recreational fishery in 1981.

Weakfish – Barnegat Bay specific commercial landings were available for weakfish for 1993 only (Kennish 2001). The bay specific landings represented approximately 5.2% of the gear specific statewide landings for that year (NMFS landing data). That ratio was utilized to calculate an estimated Barnegat Bay specific commercial landing of 0.078 metric tons for 1981. There were 3.29 metric tons of weakfish landings reported for Ocean County's recreational inland fishery in 1981.

Winter flounder – The NJDEP Bureau of Marine Fisheries estimates a commercial harvest of approximately 10.68 metric tons of winter flounder from Barnegat Bay in 1981. In 1981 there were 247 metric tons of winter flounder landed in the Ocean County inland recreational fishery.

OCNGS

The Oyster Creek Nuclear Generating Station "landings" info can be divided into two categories, impingement/impingeable size losses and entrainment losses. Impingement losses describe those animals that become trapped on the traveling Ristroph screens (9mm mesh) associated with the Circulating Water Intake Structure (CWIS) and are subsequently deposited into a fish return system and into the discharge canal. Impingeable size losses are biota that are large enough to be impinged on the Ristroph screens if they were present at the Dillution Water Intake Structure (DWIS). Entrainment losses are the biota that pass through the CWIS and DWIS structures and pass through the plant and dilution pumps, respectively. The data used to estimate these values were collected as part of periodic relicensing of the facility, and were most recently collected during 2005-2007 and include in the "Characterization of the aquatic resources and impingement and entrainment at Oyster Creek Nuclear Generating Station" September 2008 (Amergen 2008).

Impingement/Impingeable size losses

During 2006-2007 the estimated annual biomass of the young of year (YOY) and older ages of selected fish and crustaceans impinged on the traveling screens at the CWIS was calculated (Appendix A: Detailed Characterization of the aquatic resources and impingement and entrainment at Oyster Creek Nuclear Generating Station, Tables A-7 and A-8). The biomass of each species was then multiplied by the empirically determined impingement mortality rate (Appendix H, Tables H-2 and H-4) to derive a CWIS impingement mortality (kg/yr). The estimated annual biomass of impingeable sized fish and shellfish that were entrained through the DWIS was calculated (Tables A-15 and A-18) and multiplied by the empirically determined mortality rates (Tables H-5 and H-6) to derive a DWIS impingeable size mortality (kg/yr). It should be pointed out that the mortality rates were instantaneous, that is injured individuals were considered "live" at the time of counting, and thus the mortality rates are likely low.

Entrainment losses

Entrainment losses occur when biota are able to avoid or slip through the traveling screens at the CWIS and are carried through the cooling water system or are taken up by the DWIS. The number of individual fish in each species entrained into either the CWIS (Table A-10) or DWIS (A-20) are broken into 5 size categories; eggs, yolk sac larvae, post-yolk sac larvae, YOY, and YOY+. Blue crabs were divided into adult, juvenile, and megalops (tables A-12 and A-22). For this model the entrainment analysis was limited to post-yolk sac larvae, YOY, and YOY+ fish and megalops stage of blue crab. Biomass for each species/size class was calculated by taking the median or mode length from the CWIS entrainment sampling length frequency histograms (Appendix C: Impingement and entrainment studies at Oyster Creek Generating Station 2005-2007) and searching the literature for the corresponding weight. This weight was multiplied by the annual estimated number of individuals to derive an estimate of annual biomass. The biomass estimate was then multiplied by the appropriate empirically determined mortality rate to derive an estimate of entrainment losses for both the CWIS and DWIS. The latent mortality was calculated as the number of live, healthy entrainable-size specimens collected from the discharges who survived for 24 hours (Appendix F, Sections 2 and 3). The mortality was applied equally across all size classes. Given that this methodology does not take into account individuals that do not survive passage through the system it likely underestimates mortality. The specific values selected for the length, weight, and mortality rate for each species are detailed below.

Adult and juvenile blue crabs were not included in the entrainment analysis as there are a number of discrepancies in the crab data. The CWIS impingement sampling collected crabs in the 8-166mm size range; these specimens should not be able to pass through the Ristroph screen, thus nearly eliminating any entrainment at the CWIS. Further, any crabs of this size should be considered part of the "entrainment of impingeable sizes" DWIS calculations, and to include them in DWIS entrainment would be double counting.

Atlantic croaker –

Post-yolk sac – Lengths ranged from 4-16mm, with a rather uniform distribution between 7-15mm. The ASMFC 2005 stock assessment for larval croaker suggests a mode of 11mm and a weight range of 0.02 – 0.04g. An average weight of 0.03g was used in the analysis.

YOY – The lengths of YOY croaker ranged from 15-72mm, with the distribution skewed heavily to the left. The modal length was 21mm. An average weight of 0.06 grams at 21mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for croaker. The empirically determined weakfish mortality rate (CWIS 0.8, DWIS 0.75) was used as they are both Sciaenids and share similar characteristics at the larval stage.

Atlantic Menhaden –

Post-yolk sac – Lengths were bimodally distributed from 6 – 33 mm, with the larger mode at 24 mm. Hettler (1976) found an average weight of 0.195 grams at 28mm.

YOY – Lengths were evenly distributed between 27-42mm , with a mean length of 34. Hettler (1976) found an average weight of 0.494 grams at 34mm.

Mortality – A 24 hour mortality rate of 1 was used for the CWIS and 0.72 for the DWIS.

Atlantic silverside –

Post-yolk sac – Lengths were unimodally distributed from 4 – 8 mm, with the mode at 5mm.

YOY – Lengths were evenly distributed between 71-85mm. The silverside should be fully recruited to the Ristroph screen at 72mm, so 71mm was selected. An average weight of 0.2.25 grams at 71mm was calculated using the length-weight regression from FishBase.

YOY+ - Lengths were evenly distributed between 74-102mm, with a mean at 87mm. An average weight of 4.71 grams at 87mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for silverside. The empirically determined bay anchovy mortality rate (CWIS 0.97, DWIS 0.94) was used as they have similar body shapes and tolerances at the larval stage.

Bay anchovy –

Post-yolk sac – Lengths were unimodally distributed from 3 – 37 mm, with the mode at 8mm. Using the length-weight relationship in Table 5 of Leak and Houde (1987), an 8mm individual is approximately 11 days old, and would have a dry weight of 0.000114g. If larvae are assumed to be 95% water, this would lead to a wet weight of 0.0023

YOY – Lengths were unimodally distributed between 26-69mm , with a modal length of 34. An average weight of 0.32 grams at 34mm was calculated using the length-weight regression from FishBase.

Mortality - A 24 hour mortality rate of 0.97 was used for the CWIS and 0.94 for the DWIS.

Summer flounder –

Post-yolk sac – Lengths were unimodally distributed from 10 – 17 mm, with the mode at 14mm. An average weight of 0.04 grams at 14mm was calculated using the length-weight regression from FishBase.

YOY – Lengths were unimodally distributed between 12-17mm , with a modal length of 14. Given the overlap in lengths with post-yolk sac, it appears the demarcation between classes is based on eye migration. An average weight of 0.04 grams at 14mm was calculated using the length-weight regression from FishBase.

Mortality – A mortality rate was not determined for summer flounder. The empirically determined winter flounder mortality rate (CWIS 0.88, DWIS 0.90) was used as they have similar body shapes and tolerances at the larval stage.

Weakfish –

Post-yolk sac – Lengths were unimodally distributed from 2 – 14 mm, with the mode at 5mm. Using the empirically measured mean dry weight of 0.000171g for 5mm larvae from Duffy and Epifanio (1994) leads to a wet weight of 0.0034 grams assuming 95% water.

YOY – Lengths were evenly distributed between 11-123mm, with a mean length of 36. An average weight of 0.41 grams at 36mm was calculated using the length-weight regression from FishBase.

YOY+ - The only size captured in sampling was 172mm. An average weight of 0.44 grams at 172mm was calculated using the length-weight regression from FishBase.

Mortality - A 24 hour mortality rate of 0.80 was used for the CWIS and 0.75 for the DWIS.

Winter flounder –

Post-yolk sac – Lengths ranged from 2-11mm, with a relatively uniform distribution between 3-6mm. The average length was 5mm. Based on mean larval lengths in Buckley *et al.*. (1991), a 6mm winter flounder is approximately 4 weeks old. Laurence (1975) determined the mean dry weight of a 4 week old winter flounder kept at a similar temperature to be 0.000206g. This leads to a wet weight of 0.00412 grams assuming 95% water.

YOY – Lengths ranged between 6-7mm, with 6mm fish dominating the catch. Given the overlap in lengths with post-yolk sac, it appears the demarcation between classes is based on metamorphosis. Laurence (1975) determined the mean dry weight of a metamorphosed winter flounder to be 0.001243g. This leads to a wet weight of 0.02486 grams assuming 95% water.

Mortality - A 24 hour mortality rate of 0.88 was used for the CWIS and .90 for the DWIS.

Blue Crab –

Megalops – There was no information provided in the OCNGS reports on the length, weight, or mortality of blue crab megalopae with regard to entrainment sampling. Blue crab instar #1 have an average carapace width of 2.5mm, which is sufficiently small enough to pass through the Ristroph screen, and have an estimated average of weight of 0.0033 grams (Newcombe *et al.*., 1949). Mortality was assumed to be similar to that found empirically for *Mysidopsis bigelowi* during the study period of 0.66 and 0.17 for the CWIS and DWIS respectively.

Appendix 4 - Time Series data for the 1981-2013 Ecopath Model

NMFS finfish, fishery-dependent (Forced catches; -6)
This time series (1981-2013) is a combination of the NMFS commercial landings data subset (as described in the catch calculations) and recreational landings from the MRFSS (1981-2003) and MRIP (2004-2013) surveys.

For summer flounder the 1983 recreational landing of 932MT was 3 times that of the next highest value (1982) and nearly 40 times that of 1984. It was replaced with the average of the 1981-1982 landings.

While there is no large-scale commercial fishery for menhaden in Barnegat Bay, "bunker" are a popular bait fish among recreational fisherman and crabbers. Therefore a steady low harvest rate (0.2mt/yr) was assigned.

If no landings were recorded, or recorded as 0kg, it was entered as 0kg.

NJDEP Blue crab gear specific, fishery dependent (Forced effort by gear type, 3)
Blue crab landing estimates for each fishery in 1981 are described in the catch calculation and were set as a relative effort of 1. The landing values for each fishery from 1995-2013 were then scaled compared to the 1981 estimate. For 1982 to 1994 the total commercial landings were calculated based on a linear regression of the known Barnegat Bay landings against the NMFS statewide landings for 1980 and 1995-2011. The commercial gear specific and recreational landings were then calculated and scaled as previously described.

OCNGS, fishery dependent (Forced effort by gear type; 3)
Because of the nature of OCNGS operations, the cooling and dilution intake structures function as an on/off type activity, with the only shutdowns associated with temporary, short term maintenance. As such the plant flow is fairly consistent, and therefore forced effort is steady.

RUMFS subset, fishery independent (Relative biomass; 0)
These values are relative abundance (CPUE) found through otter trawling at 5 locations in the southern portion of Barnegat Bay from 1997-2011. Data is from the Tuckfile program at the Rutgers University Marine Field Station (Vasslides *et al.* 2011).

RUMFS all sites, fishery independent (Relative biomass; 0)
These values are relative abundance (CPUE) found through bay-wide otter trawling (47 sites) conducted by RUMFS in 2012 and 2013. The data is from the NJDEP's WQDE database.

NJ Coast, fishery independent (Relative Biomass; 0)
Since 1989 the NJDEP has conducted a coastal trawl survey five times a year. The data included here are yearly CPUE averaged across all sampling efforts in Stratum 15 and 18, which cover from Belmar to Sea Isle City to a depth of 10m.

Hard clam LEH abundance, fishery independent (Relative biomass; 0)
These values are density (#/ft^2) of hard clams in the southern portion (LEH) of the estuary based on stock surveys conducted in 1986 (adjusted values in Celestino 2002), 2001 (Celestino 2002), and 2011 (Celestino 2013).

SAV coverage, fishery independent (Relative biomass; 0)
SAV coverage for the bay is available for 1980, 1987, 1999, 2003, and 2009 based on aerial photograph analysis in Lathrop *et al.* (2001) and Lathrop and Haag (2011).

Sea nettles, fishery independent (Forced biomass, -1)
Recent work by Young *et al.* (in prep) using local ecological knowledge suggests that sea nettle abundances were at very low levels (unrecorded in contemporary research) until the late 1990's, with a large increase occurring around 2007. To simulate this bloom pattern we are forcing a low biomass until the mid-1990s, and then a progressive increase leading to current population estimates in 2007. The population estimate is based on spring/summer/fall sampling using plankton and lift nets conducted as part of the Barnegat Bay Initiative.

Benthic infauna and epifauna, fishery independent (Relative biomass; 0)
Benthic infauna and epifauna abundances were taken from Taghon et al. (2012 and 2013) and are from samples collected at 100 locations throughout the Barnegat Bay in July of each year. This index represents the average number of individuals found on or near the sediment surface (per $0.04m^2$) excluding amphipods, blue claw crabs, and hard clams.

Amphipods, fishery independent (Relative biomass; 0)
Benthic infauna and epifauna abundances were taken from Taghon et al. (2012 and 2013) and are from samples collected at 100 locations throughout the Barnegat Bay in July of each year. This index represents the average number of individuals per $0.04m^2$ belonging to Order Amphipoda.

Copepods, fishery independent (Relative biomass; 0)
This timeseries is based on samples collected from June to November of 2012 and 2013 at 3 locations (northern, central, and southern) within Barnegat Bay using paired 200*u*m plankton nets. The data are the average yearly CPUE for the major copepod fauna found in Barnegat Bay (n=57).

Microzooplankton, fishery independent (Relative biomass; 0)
This timeseries is based on samples collected from June to November of 2012 and 2013 at 3 locations (northern, central, and southern) within Barnegat Bay using paired 200*u*m plankton nets. The data are the average yearly CPUE for foraminifera, the only microzooplankton identified.

Phytoplankton, fishery independent (Relative biomass; 0)
The relative abundance of phytoplankton from 2008 to 2013 was estimated from chlorophyll *a* readings taken via aircraft remote sensing collected six days a week from March through October each year.