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Historical Recount of the Great South Bay Ecosystem, Long Island, New York

and

A Quantitative Assessment of the Ecosystem Structure of Great South Bay using Ecopath

A Thesis Presented

by

Matthew Andrew Nuttall

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Abstract of the Thesis

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The Great South Bay (GSB) ecosystem has provided marine resources to Long

Island residents for well over 300 years. However, various external stressors have

threatened this system, marked with declines in multiple stocks and ecosystem indices. A

historical review was conducted, indicating GSB has shifted to an dominated by lower

trophic groups. Of the twelve stocks with identifiable temporal abundance trends, eight

are currently declining. These stock declines have been met with drops in recent harvests

of GSB fisheries. In addition to monetary losses to local fishermen, the lack of a

commercially dominant shellfish stock may leave GSB without a dominant filter feeder,

facilitating the brown tide blooms that have affected stocks of plankton, shellfish, finfish,

and eelgrass since 1985. Ecosystem models were developed to elucidate the predominant

drivers of the ecosystem and determine the expected impact of external GSB stressors

over the last 120 years. Mass-balanced food web models indicated GSB has seen

concurrent drops in size and system maturity. Twenty two of the twenty four ecosystem

maturity indices measured an overall drop. GSB has experienced consistent reductions in

size and structure indicating the system is under stress. Indeed, trends consistent with

habitat degradation, alterations to physical conditions, phosphorus loading, and

overfishing were observed. Determination of cause and effect between multiple system

stressors and modeled ecosystem structure was no achieved but the results can help

enhance efforts aimed at restoration by providing an understanding of system changes

and historic baselines.   Future modeling attempts should address the feasibility of a

return to historic baselines and the management strategy that would be required to

achieve such a change.

Dedication

This work is dedicated to all who put up with me while compiling this study

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Chapter 1: Fluctuations in Great South Bay Stocks over the last 120 years with

emphasis on Ecosystem Stressors

Great South Bay (GSB) resides along the south shore of Long Island, New York.

Its border is defined by South Oyster Bay to the west, Fire Island National Seashore to

the south, and Moriches Bay to the east. This ecosystem provides approximately 290

km^2 of lagoonal habitat (Hanlon 1983) to various species of finfish (Gabriel 1921,

Neville et al. 1938, Schreiber 1973), shellfish (Gabriel 1921, Greene 1982), migrating

birds and resident waterfowl (Hanlon 1983). It has offered resources for commercial

income and recreational activities to Long Island residents for well over a century

(Gabriel 1921, Dickinson 1938, Westman 1938, Hanlon 1983, McHugh 1991, Gobler et

al. 2005). Despite this production, GSB has undergone multiple changes under various

external ecosystem pressures over the last 120 years.

During the 1880s, GSB supported a diverse ecosystem with an abundance of

finfish and shellfish (Gabriel 1921, Van Popering and Glancy 1947) including species

presently rare or extirpated from the bay such as Atlantic cod, Spanish mackerel,

sheepshead (Dickinson 1938), and large sharks (NY Times 1884, 1914, Murphy and

Nichols 1916, Thorne 1916, 1928). Prior to this period, the closure of Bellport and

Moriches Inlets left Fire Island Inlet as the only source of oceanic exchange. These

events diminished both the salinity and circulation in eastern GSB. One of the striking

characteristics in comparison to later decades is the dominance of eastern oyster under

the new salinity regime (Van Popering and Glancy 1947) and the abundance of apex

predators, prominently tiger and sandbar sharks (*Carcharinus milberti*) which utilized the

bay as a nursery area (Thorne 1928). Professor Tarleton Bean of the Smithsonian (1891)

and others published several articles in the New York Times documenting common fish

species in the bay. These articles documented the presence of large sharks (2.4 – 3.0 m)

in the bay that were abundant enough to tangle in commercial fishing nets and were

considered a persistent nuisance (NY Times 1914, Murphy and Nichols 1916). Atlantic

cod were captured on hand line by small dories near Fire Island Inlet in 1880, producing

2,000,000 pounds for New York City markets (Mather 1884) when the highest landings

of Atlantic menhaden (*Brevoortia tyrannus*; McHugh 1972), Spanish mackerel

(*Scomberomorus maculatus*; Dickinson 1938), sheepshead (*Achrosargus

probatocephalus*; Dickinson 1938), and blue crabs (*Callinectes sapidus*; Briggs 1998)

were also reported over the last 120 years. In addition, the bay supported an abundance

of bluefish (*Pomatomus saltatrix*; Gabriel 1921), weakfish (*Cynoscion regalis*; Bean

1891), American eels (*Anguilla rostrata*; Mather 1884, NY Times 1902), winter flounder

(*Pseudopleuronectes americanus*; Lobell 1938), summer flounder (*Paralichthys dentatus*;

Neville et al. 1938), and eastern oyster (*Crassostrea virginica*; Gabriel 1921, Van

Popering and Glancy 1947). However, in 1880, natural oyster beds of GSB were already

in a state of decline due presumably to increased harvest by dredges (Gabriel 1921).

As the 20th century began, various stressors altered the structure of the GSB

ecosystem. In 1902, a New York Times article declared the bay's fisheries had been

destroyed (NY Times 1902). Overfishing concentrated on various finfish stocks and

extirpated natural oyster harvest (Gabriel 1921, Van Popering and Glancy 1947, McHugh

1972). Despite the collapse of the natural oyster beds in 1893, local Long Islanders were able to sustain harvest with imported seed (Gabriel 1921, Van Popering and Glancy 1947) for another 60 years (McHugh 1972). During this time, inputs of inorganic nutrients from coastal farms initiated persistent blooms of green algae (Ryther 1954) with the potential to alter the planktonic community (Lonsdale et al. 1996, Gobler et al. 2005) and reduce eelgrass (*Zostera marina*) habitat (Cosper et al. 1987, Deegan and Buchsbaum 2005). The severe winter and summer temperatures recorded between 1930 and 1933 (Rasmussen 1977) and the release of sewage (Hinga 2005) led to an outbreak of wasting disease along the eastern US coast that devastated eelgrass beds (Short et al. 1987, Carpenter and Brinkhuis 1991). Arguably the most important impact occurred in 1931, when the reopening of Moriches Inlet by a Nor'easter caused an increase in GSB salinity and oceanic exchange, restructuring the ecosystem community starting with the bay's shellfish (Conley 1999; Van Popering and Glancy 1947, Greene 1982). A breach in the current system could increase salinity by almost 15% and reduce residence time by half (Hinga 2005), showing the potential impact from these events. In eastern GSB, salinity became almost full ocean content, causing an increase in predatory oyster drills (Van Popering and Glancy 1974, Greene 1982, Hinga 2005). Coupled with the sudden appearance of a small green alga (Ryther 1954), the harvest of oysters became sporadic and undependable after 1931 and the fishery began to decline, eventually collapsing in the 1950s (McHugh 1972). During the 1930s, the suspension feeding niche of GSB was in a state of transition between oysters and hard clams.

Although GSB supported historically high abundances of many finfish species in the 1880s; several stocks had declined by the 1930s. Little evidence exists for the

presence of large apex predators within GSB after the first few decades of the 1900s, suggesting their extirpation from the bay. Without apex predators, the highest levels of GSB's trophic pyramid were represented by bluefish, striped bass and summer flounder. Declines were also noted for menhaden (Gabriel 1921), Spanish mackerel, sheepshead (Dickinson 1938) and blue crab (Briggs 1998). It is difficult to determine causation of stock collapses that occurred 90 years ago, but evidence suggests fisheries played a significant role (Gabriel 1921, Lobell 1938, Neville et al. 1938, McHugh 1972).

In the early 1900s, GSB fisheries utilized hand lines, pound nets, fyke nets, and otter trawls (Dickinson 1938). Declining stocks led to the adoption of gear restrictions on fishermen. Fyke nets operating in Islip were subjected to a series of restrictions starting in 1937 whereas Babylon banned their usage (Poole 1969). Otter trawls were commonly used in the bay through the 1930s until the towns of Babylon and Islip banned trawling in 1937 and 1945 respectively (Poole 1969). However, these restrictions did not lower the overall fishing mortality because of corresponding increases in the recreational sector (Poole 1969). Trawling is still prohibited in Great South Bay under NY Code 13-0341 Stat. 8 (2010).

The 1980s GSB ecosystem was distinct from that which existed in the 1930s. Following decades of boom and bust in the hard clam fishery; related to the associated physical changes from the opening (1931), closing (1951), and reopening (1958) of Moriches Inlet and the pressures of overfishing; the stock finally collapsed in 1979 (Conrad 1982, Schubel 1991, Kraeuter et al. 2008). Hard clams filtered an estimated 40% of total water volume each day through the 1960s and 1970s (Kassner 1993), thus filtration capacity had drastically decreased. One repercussion of this collapse has been

the recurring brown algal blooms since 1985 (Cerrato et al. 2004). The hard clam

collapse allowed the planktonic grazing niche to become dominated by zooplankton

capable of selective feeding that limit the impacts of brown tide (Gobler et al. 2005).

Additionally, reduced offshore landings in the late 1900s caused commercial fishermen to

switch their focus inshore (Hanlon 1983) and augment an already substantial recreational

and commercial harvest in GSB. Naturally, declines were noted in stocks of bluefish,

blackfish (*Tautoga onitis*), winter flounder, and summer flounder (Hanlon 1983).

Furthermore, about 50% of the sport fishery for summer flounder in 1960 was comprised

of one year old fish (Poole 1961). By the end of the decade, the blue crab stock increased

from near absence in landings between the 1930s and 1970s (Briggs 1998).

Our data series represents an ecosystem that has undergone 120 years of

biological shifts that are largely the result of anthropogenic activities and storm triggered

barrier breaches. Commercial landings suggest the dominance of the suspension feeding

niche has switched between oysters, a shared state between oysters and hard clams, hard

clams, and more recently a near absence of both shellfish stocks. The apex predators of

the system have changed from large sharks in the 1880s to piscivorous finfish by the

middle of the 1900s. Further, the once abundant winter flounder has become scarce

throughout the bay (Frisk and Munch 2008). Overall, abundance and landings of

groundfish have drastically declined while abundances of migratory striped bass and

bluefish have likely increased predation on forage species. However, an increase in scup,

kingfish, and blue crab that began in the 1980s continued in the 2000s.

While the overcapacity of the GSB fisheries is commonly cited as the

predominant driver in abundance shifts (Gabriel 1921, Van Popering and Glancy 1947,

Gobler et al. 2005, Kraeuter et al. 2008), the underlying source of these changes has not

been conclusively shown. The thesis presented here attempts to model the GSB

ecosystem in hopes of elucidating the drivers of ecosystem fluctuations over time. We

describe the stock-specific variations that have occurred in GSB (Chapter 1) as well as

model the impacts of these variations on the ecosystem's structure (Chapter 2). These

trends were used to reconstruct the trophic structure of GSB with the potential to

elucidate the underlying processes that govern the ecosystem. A multispecies approach

will be used to model GSB and the interactions between various trophic levels using the

Ecopath software.

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Chapter 2: Historical Recount of the Great South Bay Ecosystem, Long Island, New York

Introduction

The largest saltwater bay in New York, Great South Bay (GSB) provides approximately 290 km^2 of lagoonal habitat (Hanlon 1983). This estuarine environment extends 40 km long with widths varying considerably, measuring 11 km across Bay Shore but only 300 m at Smith Point (Schubel 1991) (Figure 1.1). The mean low water depth of GSB ranges from 0.3 to 3.6 m (Hanlon 1983) and averages 2 m (Schubel 1991). Fire Island Inlet, where depths reach greater than 12 m (Hanlon 1983), provides the only direct influx of ocean water into GSB, although Shinnecock, Moriches, and Jones Inlets provide sources of indirect flow (Schubel 1991). Eelgrass beds (*Zostera* sp.), salt marshes (primarily cordgrass: *Spartina alterniflora* and *S. patens*), muddy sandflats, and sandflats comprise the majority of the benthic habitats (Dowhan et al. 1996).

During the last century, GSB can be characterized as an ecosystem in constant transition propelled by numerous drivers. Both natural and anthropogenic alterations to the system's physical and biological properties have resulted in dramatic shifts in the community structure. Natural modification of barrier islands through unpredictable storm surges and sediment deposition have changed the circulation patterns throughout the bay (Greene 1982, Hinga 2005); an estimated 28 inlets have existed along the Fire Island National Seashore over the last 300 years (Leatherman 1985). Species targeted for commercial and recreational harvest constantly varied and spread substantial fishing

mortality on species across all trophic levels. Harmful algal blooms (HABs), both past green algae and recent brown algae, have also affected composition of plankton within the bay, increasing the mortality on eelgrass (Cosper et al. 1987) and shellfish (Greenfield and Lonsdale 2002, Gobler et al. 2005). As GSB represents a valuable commercial, recreational, and ecological resource (McHugh 1972, Hanlon 1983), understanding drivers and the relative magnitude of their ecosystem impacts are of great importance towards managing the system effectively.

Methods

A literary review was compiled to provide an overall depiction of the GSB ecosystem and highlight changes that have occurred; landings and catch-per-unit-effort (CPUE) provide indices of species' abundances.

Commercial GSB landings were obtained for multiple years from Mather (1884), Dickinson (1938), Van Popering and Glancy (1947), annual commercial shellfish landings from Babylon, Brookhaven, and Islip, and annual commercial landings from the New York State Department of Environmental Conservation (NYSDEC). NYSDEC landings are organized by port where fish were landed, not necessarily where fishing took place. Therefore, stock-specific estimates from Philip Briggs were used to calculate the relative contribution of bayfish to the reported GSB landings. Recreational GSB landings were from the Marine Recreational Fisheries Survey Statistics (MRFSS) dataset (Table 1.1), Westman (1938), Briggs (1962), and Fox (1981).

Measures of finfish CPUE were gathered from studies using otter trawl. Multiple seine studies were also available from GSB, but the random distribution of sampling locations between surveys made time-based CPUE comparisons impossible. For this analysis, CPUE was defined as biomass (tonnes) per area sampled (km^2). Benthic trawling data from 1981 (Hanlon 1983) and 2007 (Frisk and Munch 2008) was analyzed to assess the ecological structure of GSB. A comparable CPUE was extracted for winter flounder (Lobell 1938) and summer flounder (Neville et al. 1938) from a commercial trawl survey. There were two additional trawl surveys available, but data were deemed incomparable. A 1940 survey (Neville and Bevelander 1941) was discarded as researchers attained their data from the catch of commercial shrimpers, utilizing gear with different dimensions and a much slower tow speed. A 1971 survey (Gaw 1972) was neglected due to use of a semi-balloon trawl (vs. the benthic variety) and differences in sampling locations. Details of the sampling protocol (Table 1.2) and area sampled (Figure 1.2) were noted for these trawl surveys.

Benthic species CPUEs were gathered from Greene (1982), Hanlon (1983), Wiggins (1986), MacKenzie (2003), and Kraeuter et al. (2008). Details on sampling protocol (Table 1.3) and location (Figure 1.3) were noted. Ponar grab data obtained by Hanlon (1983) was discarded as sampling effort was insufficient to accurately assess the GSB benthos. WAPORA Inc.'s study was also neglected as use of a suction dredge complicated comparison to other studies using ponar grabs (Greene 1982). An assessment of GSB blue crabs was also included (Briggs 1998).

Results

Historical Recount of the Fisheries of Great South Bay

Shellfish Fisheries: Shellfishing in GSB is perhaps the most revered pastime of Long Island (LI) fishermen. Shellfish fishermen, nicknamed baymen, alternated among three primary species: blue mussel (*Mytilus edulis*), eastern oyster (*Crassostrea virginica*), and hard clam (*Mercenaria mercenaria*). Bay scallops (*Argopecten irradians*), soft shell clams (*Mya arenaria*), razor clams (*Ensis directus*), and conchs (*Busycon* sp.) have also provided appreciable harvest (Van Popering and Glancy 1947, Greene 1982). In the 1700s, blue mussels were the easiest shellfish to harvest within GSB (Van Popering and Glancy 1947). The inlets at Fire Island, Bellport Bay, and Moriches (the latter two found west and east of Smith Pt respectively) (Figure 1.1) maintained a high salinity and circulation throughout GSB. The highest salinity areas, close to the GSB inlets, were more conducive to the growth of mussels (Brenko and Calabrese 1969) than to oysters and hard clams (Davis and Calabrese 1964). Once established, mussels may have hindered development of other shellfish: mussels have relatively fast growth rates (Bayne and Worrall 1980) which may lead to overgrowth of other shellfish (Van Popering and Glancy 1947) and establishment of mussel beds has been attributed to "muddying" the benthos (Smith and Shackley 2004) which impedes settlement of other shellfish. Additionally, predatory oyster drills (*Urosalpinx cinerea*) and seastars (*Asterias forbesi*) flourished in the high salinity waters (Van Popering and Glancy 1947). Blue mussels continued to be the only shellfish appreciably targeted by baymen until the early 1800s, when the discovery of a massive oyster bed left mussel harvest to only a few baymen (Ingersoll 1884, Gabriel 1921).

The first big oyster bed discovered in GSB was that off Blue Point at the end of the 18th century (Gabriel 1921). Harvest required a set of tongs and an open rowboat, appealing to both the rich and poor. Overcapacity plagued the developing fishery and uncontrolled harvest effectively closed the Blue Point fishery in 1824. Baymen searched for new fields, discovering a massive bed in eastern Patchogue (Gabriel 1921). The natural closure of the Bellport Bay Inlet (early 1800s) strengthened these beds, as reduced salinity (Greene 1982) posed problems for predatory seastars and oyster drills (Van Popering and Glancy 1947). These new conditions also brought an interesting option to the oyster fishermen. Oysters effectively devote energy to either growth or reproduction, not both (Gabriel 1921, Hofmann et al. 1994). The reduced circulation and salinity in eastern GSB favored the reproduction and proliferation of young oysters, while the western beds (still subjected to high salinity and circulation) were better suited for the growth and fattening of mature oysters (Van Popering and Glancy 1947). Thus, eastern beds became a reservoir for young oysters for baymen to transplant in the west to grow. This practice became common for baymen in the early 1800s, lasting until the collapse of the fishery. During this period, the LI towns of Blue Point, Sayville, and West Sayville were the largest suppliers of oysters in the United States (Hanlon 1983).

GSB oyster beds began to falter by the mid 19th century. By 1860, most baymen on GSB used power dredgers (Gabriel 1921). Cheap to manufacture, this new tool increased efficiency of harvest and opened up new fishing grounds previously too deep for exploitation. Like the collapsed Blue Point stock, Patchogue oysters started to decline. In 1865, GSB baymen began trying to cultivate seed oysters through planting (Gabriel 1921). The occupation proved effective as an increased harvest from the seeded

beds attracted other baymen and new planters (Van Popering and Glancy 1947). Oyster

planters began to monopolize the bay bottom, reducing available habitat for natural beds.

Coupled with high fishing rates, loss of habitat caused natural beds to fail and ultimately

collapse in 1893 (Gabriel 1921), leaving oyster fishermen to rely solely on seeded

oysters. From 1880 into the early 1900s, planting became the dominant profession

among baymen. Supported by the transplanted seed, a massive harvest ensued reaching

seventy thousand barrels per year in the early 1900s (Van Popering and Glancy 1947). In

1931, a powerful Nor'easter created a breach in the barrier islands, reopening Moriches

Inlet (Conley 1999) after its natural closure in 1886. This breach increased the salinity of

Moriches Bay and eastern GSB, causing an explosion in the predatory oyster drills,

devastating seeded and natural oyster beds (Van Popering and Glancy 1947, Greene

1982). Coupled with the sudden appearance of small green algae (Ryther 1954) and

deterioration in eelgrass habitat (Carpenter and Brinkhuis 1991), harvest of oysters

became sporadic and undependable after 1931. The fishery persisted through the 1930s,

limped through the 1940s, but eventually collapsed in the early 1950s (McHugh 1972).

The hard clam fishery dates back to Native Americans who used shells as

currency, referred to as wampum (McHugh 1985, 1991). Harvest started to grow in Cape

Cod at the start of the 1800s so, by 1860, a consistent harvest of 1500 metric tons was

extracted from NY waters until 1895 (McHugh 1991). When the natural oyster stock

collapsed in 1893 and the fishery turned to planting, baymen were displaced by oyster

planters. Hard clam landings plummeted as GSB benthic habitat became monopolized by

the oyster industry, remaining low until 1931. As the reopening of Moriches Inlet wiped

out the remaining oyster sets (McHugh 1991), the saline-tolerant hard clams flourished in

response to increased salinity (Van Popering and Glancy 1947, Greene 1982). As the last

50 years of oyster harvest had persisted from planted oysters, baymen were hesitant to

reclose Moriches Inlet and jeopardize the self-sustained hard clam stock (Van Popering

and Glancy 1947). Landings of hard clams steadily increased through the 1930s

(McHugh 1972). The green algae *Nannochloris atomus* and *Stichococcus* sp. had also

become persistent in GSB during this time. Although this HAB has been linked to a

reduction in eelgrass habitat and feeding in various shellfish (Van Popering and Glancy

1947, Tracey 1988, Bricelj and Kuenstner 1989, Gainey and Shumway 1991, Schubel

1991, Gobler et al. 2005), hard clam harvest remained appreciable through the 1940s,

increasing in 1947 as the NY harvest totaled almost 11 million pounds (McHugh 1972).

In 1951, Moriches Inlet again closed, producing a 65% reduction in the bay's circulation

(Glancy 1956 cited in Greene 1982). The reduced circulation and proliferation of green

algae caused NY hard clam landings to drop by almost two-thirds over the next few years

(McHugh 1972). To counteract the decline, Moriches Inlet was permanently reopened in

1958 (Greene 1982, McHugh 1991). The act seemed to work, as GSB hard clam

landings grew from 1.5 million pounds in 1959 (McHugh 1972) to over 6 million in

1969. The eastern grounds of GSB supplied over 70% of US hard clams in the late 1970s

(McHugh 1991, Kurlansky 2006). Unfortunately, a drastic decline in hard clam landings

began in 1976 (Schubel 1991), attributed to overharvest (Conrad 1982, Gobler et al.

2005, Kraeuter et al. 2008). By 1984, the hard clam stock provided less than half the

landings of the 1970s peak harvest (McHugh 1985, Schubel 1991). The next season,

brown algae (*Aureococcus anophageffrens*) started to bloom (Bricelj and Lonsdale 1997)

which imparted increased mortality on juvenile hard clams (Greenfield and Lonsdale

2002). The recovery of current hard clam stocks is complicated by the suppression of growth rates and feeding activity from brown algal blooms (Cosper et al. 1987, Schaffner 1999, Greenfield and Lonsdale 2002, Gobler et al. 2005).

Crustacean Fisheries: Various species of crustacean have also supported GSB fisheries. Blue crabs (*Callinectes sapidus*) have been harvested from GSB for over a century (Gabriel 1921) and were a valuable addition to the income of past NY fishermen (Briggs 1998). Crabbers would bait a heavy line of about 200 yards, typically with salted eel. Raising the line out of the water, fishermen could remove crabs with dip-nets (Gabriel 1921). However, since 1880, landings of blue crabs had been in a state of steady decline. The 1880 NY catch of over 1.6 million pounds declined to an average annual harvest of 17 thousand pounds from the 1940s–70s (Briggs 1998). Crab fishermen began using dredges and pots in the 1950s, completely replacing dip-nets by 1960 (Briggs 1998). However, the blue crab stock appeared to rebound through the late 1900s; NY landings increased from 53 thousand pounds in 1980 to over 1.2 million pounds in 1993, the majority of this harvest attributed from GSB (Briggs pers. comm.). Although not as prominent as the blue crab harvest, lady crabs (*Ovalipes ocellatus*) and rock crabs (*Cancer irroratus*) are seasonally important in GSB (Briggs 1998). The jonah crab (*Cancer borealis*) has also provided income for fishermen, although not targeted directly (Briggs 1998). There also exists a small fishery for grass (*Palaemonetes* sp.) and sand (*Crangon* sp.) shrimp within GSB, providing bait for weakfish fishermen (Westman 1938, Neville and Bevelander 1941, Hanlon 1983). Blood worms (*Glycera* sp.), sand worms (*Nereis virens*), and squid (*Loligo paelei*) are also popular baitfish targets (Westman 1938, Hanlon 1983).

Atlantic Menhaden Industry: Menhaden (*Brevoortia tyrannus*) were historically harvested as baitfish by LI fishermen (Gabriel 1921). At the end of the 1700s, an industry began to grow as farmers recognized the potential of menhaden as a prized fertilizer (Gabriel 1921, Dickinson 1938). As 8,000 fish were needed to dress an acre of farmland, hundreds of thousands of these fish were caught (Gabriel 1921). By 1825, landings grew to millions. In 1850, D.D. Wells established a factory in Greenport that allowed for the extraction of oil from the fish. The application of this oil in painting and tanning further pushed the development of the fishery (Gabriel 1921). More factories were founded so, by 1877, 23 menhaden processing factories were in existence across LI with the port of Sayville providing one of three focal points around LI (Gabriel 1921). As the fishery prospered, increased competition for a limited commodity became inevitable, pushing fishermen further offshore (Gabriel 1921). Increased operating costs pushed many independent fishermen from the business, consolidating the fishing effort of the fishery. Those that remained utilized purse seines, harvesting entire schools of menhaden with a single net. The increased efficiency made menhaden harder to find, driving fishing further offshore. Steamships became necessary, providing access to deeper waters. However, the cost drove most small factory owners from the business and further consolidated the fishery. By 1895, only 8 of the 23 LI menhaden factories remained. Competition remained high and continued until 1897, when the entire Atlantic coast industry was consolidated under the American Fisheries Company (AFC). At the beginning of the 20th century, the AFC moved the focal point of LI menhaden production to the east, away from the fishermen of GSB (Gabriel 1921). Harvest remained

consistent until 1958, when landings started to decline, culminating in 1966 (McHugh 1972) as catch proved inadequate to support the industry.

Commercial Finfish Fisheries: Primarily a destination for recreational anglers before the 1880s (Mather 1884), GSB has supported commercial fisheries targeting other finfish stocks. Although the stock was incomparable to systems north of LI, landings from the cod (*Gadus morhua*) fishery started as early as 1669 (Gabriel 1921). An important addition to the sustenance of local colonists, GSB fishermen pursued schools in dories and smacks. Both offshore and bay stocks were targeted by locals armed with handlines (Gabriel 1921). In 1840, the prosperous whaling industry drew half of the cod fishing vessels to eastern LI. However, fishermen returned to their handlines as the whaling industry began to struggle in the late 1800s. Bluefish (*Pomatomus saltatrix*) harvest increased in the 1890s to satisfy a growing demand for protein (Gabriel 1921). The combination of fishing bluefish in the summer and cod in the winter provided income year round. American eels (*Anguilla rostrata*) were also harvested during winter (Gabriel 1921). In the early 1900s, reduced offshore landings caused commercial fishermen to switch their focus to inshore GSB stocks (Hanlon 1983). This increased an already substantial recreational harvest. During this period, landings from the GSB commercial sector were dominated by trap fishermen (Dickinson 1938). In the late 1800s, these pound nets would typically catch stocks of Spanish mackerel (*Scomberomorus maculatus*), sheepshead (*Achrosargus probatocephalus*), scup (*Stenotomus chrysops*), weakfish (*Cynoscion regalis*), and bluefish. Although frequently landed, butterfish (*Poronotus triacanthus*) were considered a trash fish and discarded. By 1930, Spanish mackerel and sheepshead had disappeared from trap net catches and were

replaced by squid (*Loligo* sp.) and butterfish (Dickinson 1938). Menhaden and cod

continued to be harvested (Gabriel 1921, Dickinson 1938). Fyke nets were also

commonly set within GSB, but declining stocks caused multiple LI towns to adopt

restrictions on the gear throughout the 1930s, even being banned from Babylon in 1937

(Poole 1969).

      Recreational Finfish Fisheries: The recreational sector of the GSB fishery has

experienced transitions in targeted species as well. In 1938, a large recreational effort

was placed on flatfish stocks (Westman 1938, Briggs 1962). This effort increased

sequentially through the 1940s so by the 1950s, winter flounder (*Pseudopleuronectes*

americanus) and summer flounder (*Paralichthys dentatus*) made up about 90 percent of

the annual recreational catch (Briggs 1962). Bluefish and blackfish (*Tautoga onitis*) also

composed a significant portion of the recreational landings. Northern kingfish

(*Menticirrhus saxatilis*) and striped bass (*Morone saxatilis*) landings were also

significant, but were dominated by surf fishermen (Briggs 1962) implying offshore stocks

may govern the fishery. Commonly caught, the northern puffer (*Sphoeroides maculatus*)

was considered a trash fish by many fishermen during the 1960s and discarded.

Weakfish, common eels, and black sea bass (*Centropristis striatus*) were also targeted by

GSB anglers, but landings were relatively minor (Briggs 1962). In the 1980s, flatfish still

dominated the recreational catch across the southern bays of LI, including Shinnecock,

Moriches, and GSB (Hanlon 1983). Striped bass and northern puffer were still targeted

by recreational fishermen, as were white perch (*Morone americana*) and butterfish.

Multiple baitfish were also exploited within the bay including Atlantic silverside

(*Menidia menidia*), striped killifish (*Fundulus majalis*), and mummichug (*Fundulus heteroclitus*) (Hanlon 1983).

Abundance and Landing Trends

Of the numerous species that are included in GSB landings and abundance surveys, only fifteen provided sufficient data to compare temporal fluctuations. These stocks are analyzed below.

Hard Clam: In 1880, commercial hard clam landings measured 150,000 bushels. Little change was recorded in 1958, as the commercial sector extracted about 160,000 bushels. Landings consistently increased until 700,000 bushels were harvested from the system in 1976 (Figure 1.4A), 4 times the landings of the late 1950s. Landings then declined to 300,000 bushels in 1981 and 100,000 bushels from 1986–89. The decrease continued into 2007 when less than 6,000 bushels were processed. Kraeuter et al. (2008) showed a steady decrease in the Islip hard clam stock CPUE from 1978–2003 (Figure 1.4B). A decrease in unidentified clams, the classification made necessary from the broad groupings used in MacKenzie (2003), was also noted in CPUE from ponar grabs between 1986 and 2001 (Figure 1.4C).

Blue Crab: Commercial landings of blue crabs in 1880 have been unmatched by any subsequent year in the fishery (Figure 1.5). Consistent declines stretched into 1940 where harvest remained negligible until the 1980s, after which catch has consistently increased. Landings in 1993 reached almost 75 percent of the record 1880 yield.

Flatfish: The 1938 CPUE of winter and summer flounder (Figures 1.6B and 1.7B respectively) showed winter flounder were 4 times more abundant than summer flounder. A winter flounder population decline was captured in the commercial landings data from 1950–70 (Figure 1.6A). Summer flounder landings remained relatively consistent over this period (Figure 1.7A), although low relative to winter flounder. Winter flounder landings spiked in 1970, decreased into 1975 and increased into 1980. Conversely, summer flounder landings increased and decreased into 1975 and 1980, respectively. The CPUE data shows winter and summer flounder were at relatively high abundances in the 1980s. Since 1980, recreational landings have declined with a steeper decline observed for winter flounder. Commercial winter flounder landings in the early 2000s were about 10 fold lower than those of the late 1970s. Summer flounder landings seemed to show no change in magnitude over this time. The CPUE data from 1983–2007 shows a 70 and 5 fold decrease in abundance of winter and summer flounder, respectively. The trawl CPUE indicates a 2 fold increase in windowpane flounder since 1983 (Figure 1.8).

Bluefish: Commercial bluefish landings remained relatively low from 1938–50s (Figure 1.9A). Increased landings began in the 1960s, but were followed by declines in the 1970s. The 2000s commercial landings were 2 fold less than the late 1970s. The recreational sector experienced the same decline. Trawl CPUE data shows the opposite trend, recording a 2 fold increase in bluefish between 1980 and the 2000s (Figure 1.9B).

Northern Kingfish: The recreational landings of kingfish gradually increased through the late 1950s (Figure 1.10A). Although sporadic, recreational landings continued to grow in magnitude from the 1980s into the 2000s. The trawling data shows

a doubling in kingfish biomass from 1981–2007, supporting the landings data (Figure 1.10B).

Weakfish: Commercial weakfish landings in the 1950s were less than 25% of that reported in 1938 (Figure 1.11A), remaining relatively constant through the 1970s. Current trawl data shows a similar decline from the early 1980s (Figure 1.11B), but recent commercial landings show an appreciable increase.

Atlantic Menhaden: Largely ignored by the recreational sector, there are no recreational landings of menhaden to assess. Commercial menhaden landings increased from 1938–54 (Figure 1.12A) but started to decline by 1970. From 1970–73, commercial landings increased. Landings gradually declined from 1973–79 and again in the 2000s. The CPUE of menhaden in 2007 was 25% of that found in 1981 (Figure 1.12B).

Blackfish: Blackfish provide little to the commercial sector, so landings are neglected. Recreational landings were appreciable in the 1980s, producing a 1986 peak harvest (Figure 1.13A). Landings gradually tapered off into the 2000s. The trawl CPUE supports this with a 4 fold decrease recorded from 1981–2007 (Figure 1.13B).

Northern Puffer: The 1950 commercial landings of northern puffer were unmatched by any subsequent year in the fishery (Figure 1.14A). By 1956, commercial harvest dropped over 50%. Commercial puffer landings became trivial beyond 1972. A peak in the recreational harvest was recorded in 1981. A sequential decline by almost 20 fold ensued. The decline was captured in the trawl CPUE (Figure 1.14B).

Scup: There was a paucity of GSB commercial landings data (Figure 1.15A). Recreational landings decreased from 1981–2007 while trawl CPUE shows 2007 abundance 5 fold higher than 1981 (Figure 1.15B).

American Eel: The commercial landings of American eel peaked in 1880 (Figure 1.16A). Landings increased from 1952–61, but gradually declined through the 1960s. Although consistent, the 1970s harvest remained almost 4 fold lower than 1961 landings. Current landings show a decrease from the 1970s, being about half that of the 1970s. The decrease is supported by the trawl CPUE data, in which a 10 fold decline was recorded (Figure 1.16B).

Striped Bass: Commercial landings were relatively low for this species through the 1960s (Figure 1.17A). Peak landings were reported for 1970, but quickly decreased through the rest of the decade. The 1980s and '90s brought prosperity to the recreational sector, as landings steadily increased, reaching a maximum in 1996. However, landings quickly bottomed out by 2000. The commercial and recreational sector both show steady rises in harvest through the 2000s. As the 1981 trawl survey produced the only capture of striped bass, no assessment using CPUE data is possible (Figure 1.17B).

Black Sea Bass: Data was sparse for this species. From 1981–2007, trawl CPUE doubled (Figure 1.18B), agreeing with the increase in recreational harvest from the 1980s–2000s (Figure 1.18A).

Discussion

A review of abundance indices suggests various GSB finfish stocks are currently at historically low levels. Of the twelve stocks with identifiable temporal trends, eight are currently declining. Historically abundant, GSB hard clam stocks have collapsed, CPUE recording an 84% drop (Kraeuter et al. 2008) from peak harvest rates in the late 1970s. The reduction in hard clams has coincided with stresses from eutrophication, overcapacity in the fisheries, and alterations to barrier beaches (Gabriel 1921, Van Popering and Glancy 1947, Conrad 1982, McHugh 1991, Gobler et al. 2005). CPUE data shows winter flounder has retained less than 2% of the 1980s biomass from which record landings were extracted. Inshore habitat degradation (Manderson et al. 2000), overfishing (Vonderweidt 2006), and changing temperatures (Keller and Klein-MacPhee 2000) are cited as potential drivers suppressing the stock. Similarly, CPUE indicates summer flounder are 81% less abundant than in the 1980s. Conversely, blue crab, northern kingfish, windowpane flounder, and black seabass have recently increased in abundance. Blue crab have collapsed and rebounded through the bay's history, the 1943 yield dropping to less than 1% of 1880 peak landings and increasing 15 fold between 1980 and 1994. These fluctuations may be attributed to variations in fishing pressure, predation from other fished species, or overwintering mortality. The observed increase in abundance for black sea bass, however, may represent abundances of offshore stocks and inaccurately model bayfish: as most of the black seabass stock resides offshore (Neville et al. 1938, Briggs 1978), landings trends likely stem from outside GSB.

Fishing pressures have proven influential in the development of the current GSB ecosystem. Although multiple external pressures have existed over the last 120 years, the consistent mention of fishing mortality (Gabriel 1921, Van Popering and Glancy 1947,

McHugh 1991, Vonderweidt 2006, Kraeuter et al. 2008) indicates this pressure may be a dominant driver in stock fluctuations. As target stock abundances became low, fishermen simply switched to another stock, producing a consistent strain on the ecosystem that has manifested into its current state. Indeed, variations in stocks have been predictable and consistent in that observed stock fluctuations followed fishery trends which appear to consistently suppress the abundances of targeted stock.

The current degraded state of the GSB's resources is a cause of concern. In addition to the monetary losses to local fishermen, systems devoid of a species previously occupying an important ecological niche can become unstable and shift into a system dominated by different dynamics. In the Chesapeake Bay system, the loss of the predominant oyster stocks to overharvest has been met with outbreaks of eutrophication, hypoxia, and disease (Jackson et al. 2001). Throughout the 1960s–70s, hard clams filtered an estimated 40% of the total water volume of GSB each day (Kassner 1993). In 1993, the same shellfish stock was only able to filter 1-2% of the bay's volume daily (Kassner 1993). Reduction in filtration capacity by the hard clam stock is a possible driver for the relative frequency of brown tide events currently plaguing GSB (Cerrato et al. 2004, Gobler et al. 2005). Once established, these blooms have been shown to affect the composition of plankton, various shellfish and finfish stocks, and eelgrass habitat (Cosper et al. 1987, Bricelj and Kuenstner 1989, Bricelj and Lonsdale 1997, Greenfield and Lonsdale 2002, Cerrato et al. 2004). In particular, the reduction in eelgrass cover limits the habitat (Deegan and Buchsbaum 2005) and predator cover (Orth et al. 1984, Bell et al. 1987, Pohle et al. 1991) available for many species of juvenile finfish and shellfish, further suppressing these stocks. Unfortunately, these impacts also impede any

management aimed at the potential recovery of GSB stocks and must be remedied before the system is able to recover. The ability of some mollusks to feed in the presence of the brown tide algae, such as the slipper snail *Crepidula fornicata* (Harke 2009), may shift the GSB community to another dominant filter feeding stock. However, as eelgrass habitat declines (NY Seagrass Experts Meeting 2007, NYS Seagrass Task Force 2009, Pickerell 2010) and *A. anophageffrens* continues to bloom in GSB (Bricelj and Lonsdale 1997, Greenfield and Lonsdale 2002, Gobler et al. 2005), other stocks have so far been unsuccessful in occupying the empty filter feeding niche.

As with any study, uncertainty is present within this analysis. While pivotal to our study, CPUE and landings indices for stock abundance pose analytical problems. First, landings data is not a measure of stock biomass. Although tied to trends outlined in the literature, these trends only represent abundances if fishing effort remains constant. However, as the GSB fisheries have a history of overfishing (Gabriel 1921, Van Popering and Glancy 1947, McHugh 1991, Vonderweidt 2006, Kraeuter et al. 2008), I am confident that fluctuations in landings are driven by variations in abundance, in addition to demand. Second, CPUE comparisons were limited to periods where data was available. As the few scientific surveys conducted in GSB over the last century utilize different sampling protocols and fishing gear, there are multiple gaps in the data that could show important trends. Third, CPUE and landings data fail to capture variations in both the age and size structure of stocks, both of which may be important to the analysis; should growth rates increase yet abundance decrease, there may be no observable change in biomass. Despite these shortcomings, the conclusions drawn from this data can be valuable if used with caution.

Table 1.1. Site Codes for ports of Great South Bay anglers in the NOAA MRFSS dataset.

Sites included in GSB Recreational Landings

	State	36	New York		
	County	103	Suffolk		

Site	8	295	357	470	608	1303
	11	304	358	473	609	1304
	12	327	373	492	610	1305
	58	331	394	514	611	1306
	73	332	397	525	613	1613
	78	334	398	527	615	1780
	180	335	408	553	616	1851
	196	336	409	564	617	1896
	221	340	415	575	625	1999
	224	342	440	576	1300	3080
	226	345	460	606	1301	3229
	249	346	468	607	1302	

Table 1.2. Details of Sampling Protocol for included Trawl Surveys.

Parameter	Trawl Studies		
	Gaw 1972	Hanlon 1983	Frisk & Munch 2008
Gear	Semi-Balloon Otter Trawl	Benthic Otter Trawl	Benthic Otter Trawl
Sampling Method	3 Stations south of East Islip	Stratified Random Sampling	Random Sampling
Days Sampled	Twice each week from Oct 1970 - Oct 1971 - not in Jan & Feb	Twice each Month from Mar-Nov 1981	over 4 days in May, 3 days in July, and 4 days in Oct 2007
Number Trawls each Day	12	max of 11	average of 9
Total Number Trawls	1056	290	99
Groundline (m)	4.88	9.45	7.47
Average Tow Speed (m/sec)	1.543	assumed 1.286	1.224
Area Covered per Trawl (m^2)	283.33	3646.28	2640.89
Wing Mesh (cm)	3.81	3.81	3.81
Bag Mesh (cm)	3.175	2.54	NA
Cod End Mesh (cm)	1.27	1.905	0.635

Table 1.3. Details of Sampling Protocol for Benthic Grab Surveys.

Parameter	Hanlon 1983	Greene 1982	Benthic Studies Wiggins 1986	MacKenzie 2003	Kraeuter et al. 2008
Gear	Petite Ponar Grab	Suction Dredge	Ponar Grab	Petite Ponar Grab	Commercial Clam Shell Bucket
Sampling Method	Stratified Random Sampling (same stations as trawl)	Stratified Random Sampling along 1.295 km^2 grids	2 sites found at 4 stations in Patchogue Bay	2 stations in Patchogue Bay	Stratified Random Sampling along 400 m^2 grids
Days Sampled	Mar-Nov 1981	June-Nov 1978		2001	Not Stated
# Samples / Day	Sporadic sampling	Not stated	average of 18	4-6	332 samples/year
Total # Samples	14	392	54	Not Stated (maybe just the one day sampling)	8956 (27 years)
Area per Sample (m^2)	0.0231	0.16	0.04	0.0225	0.995 average
Penetration Depth (cm)	6.99	7.6	about 7	same gear as Hanlon, assume 6.99	Not Stated
Mesh Size for Sieve (mm)	1	1	1	0.00025	6.4 (1978-84) and 3.2 (1985-2004)

Figure 1.1. The Great South Bay System represented using ArcGIS software.
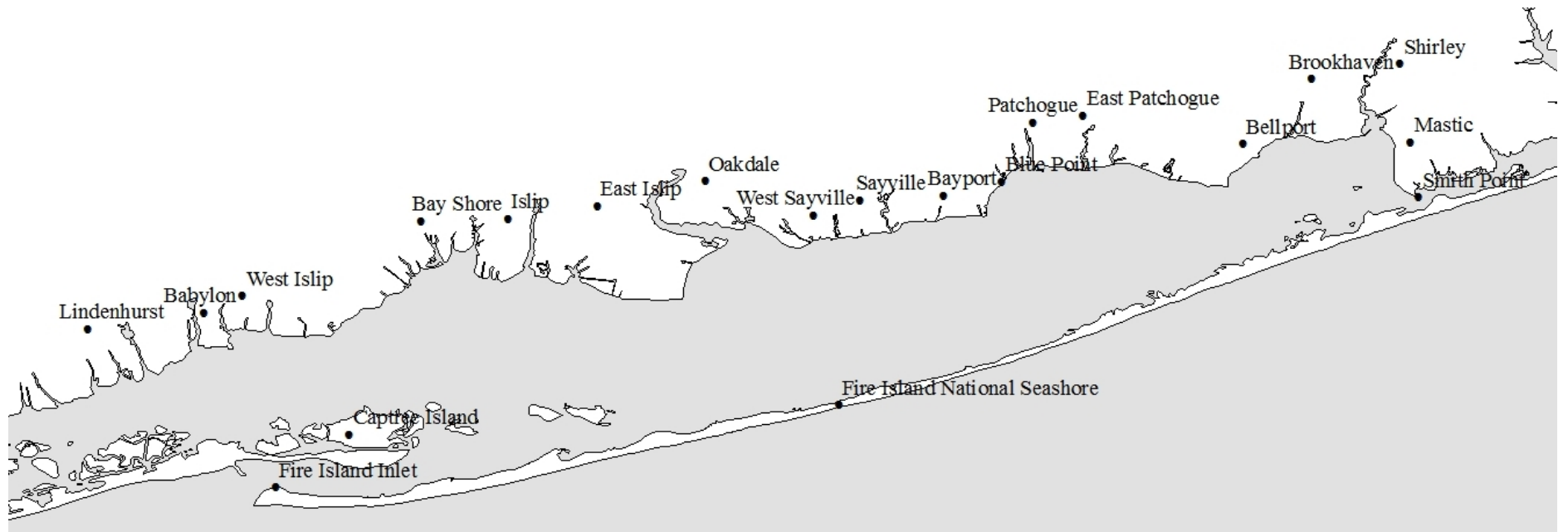
Figure 1.2. Map of the Sampling Stations from GSB Trawling Surveys. Stations G1, G12, and G13 were those sampled repeatedly by Gaw 1972. Hanlon (1983) divided GSB into six areas in which a stratified random sampling procedure was adopted. Stations were as follows: 1 – Fire Island Inlet, 2 – Babylon, 3 – Islip, 4 – Sayville, 5 – Patchogue, and 6 – Bellport Bay. Note that station 1 was never sampled due to the water depth and fast current speed. Frisk and Munch (2008) sampled randomly throughout the entirety of GSB, so no site key was needed.
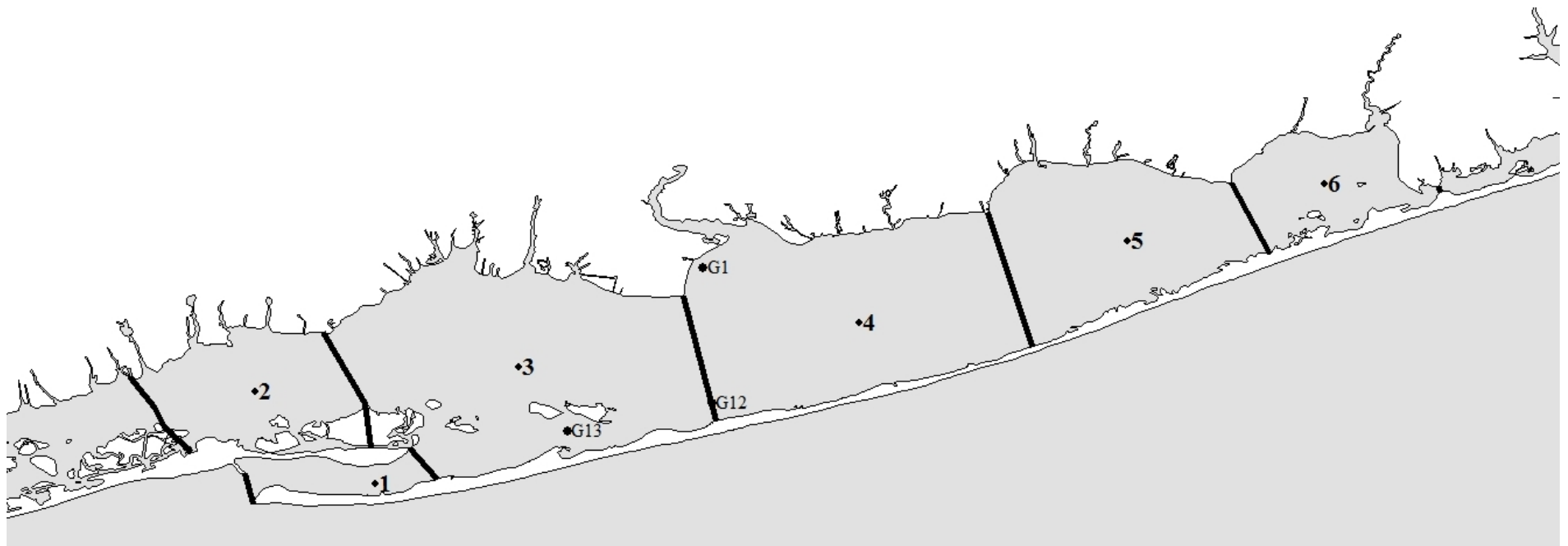
Figure 1.3. Map of the Sampling Stations from GSB Benthic Grab Surveys. Wiggins (1986) sampled four stations outside various creeks running from Patchogue: Swan (WA and WB), Mud (WC), and Hedges Creek (WD). MacKenzie (2003) also sampled within Patchogue Bay, whereas one station was close to shore (M1) and the other further offshore (M2). Hanlon (1983) and Greene (1982) sampled the entire bay.
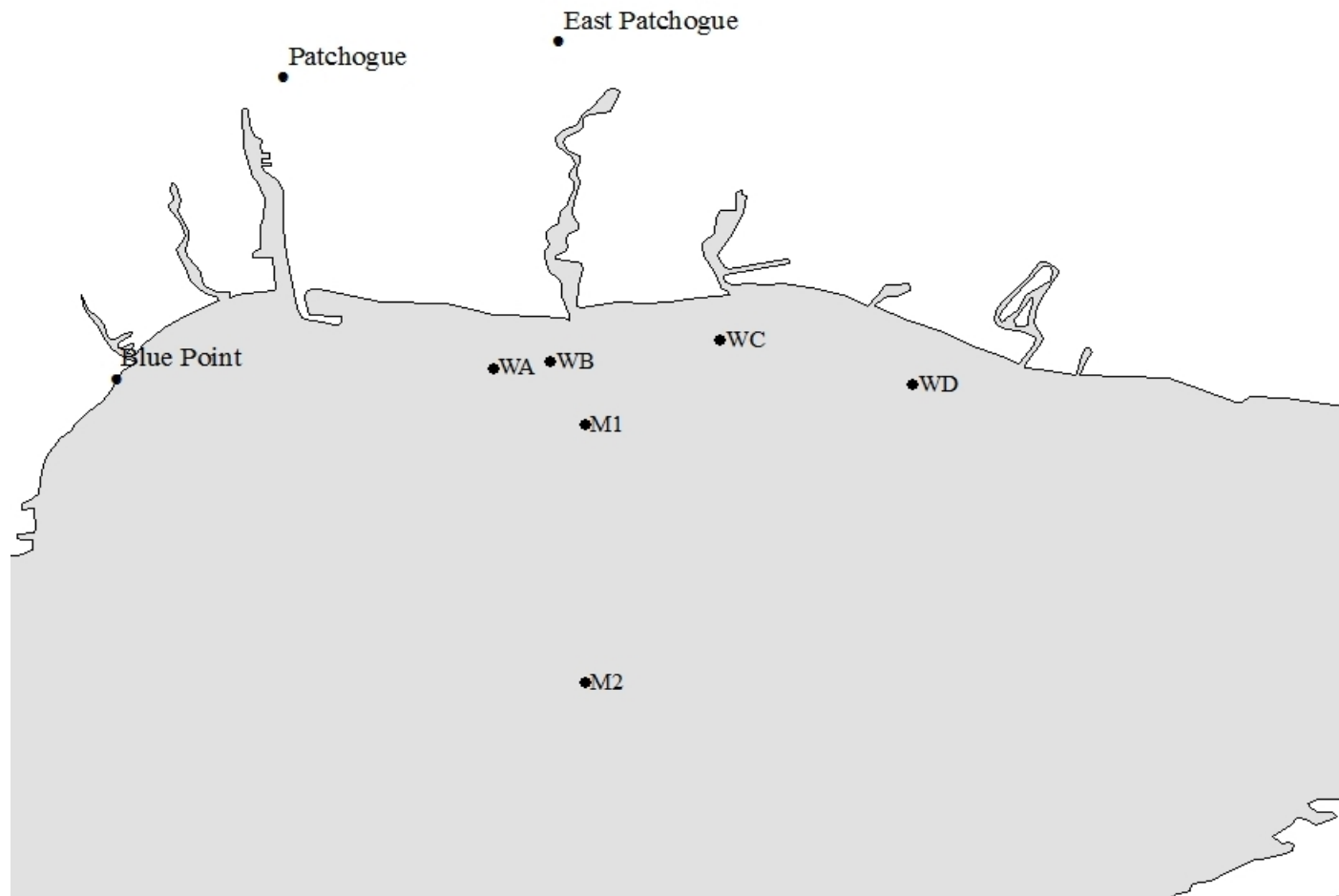
Figure 1.4. Compiled Commercial Landings for Hard Clams (A) and benthic CPUEs for Hard Clams (B) and Unidentified Clams (C).
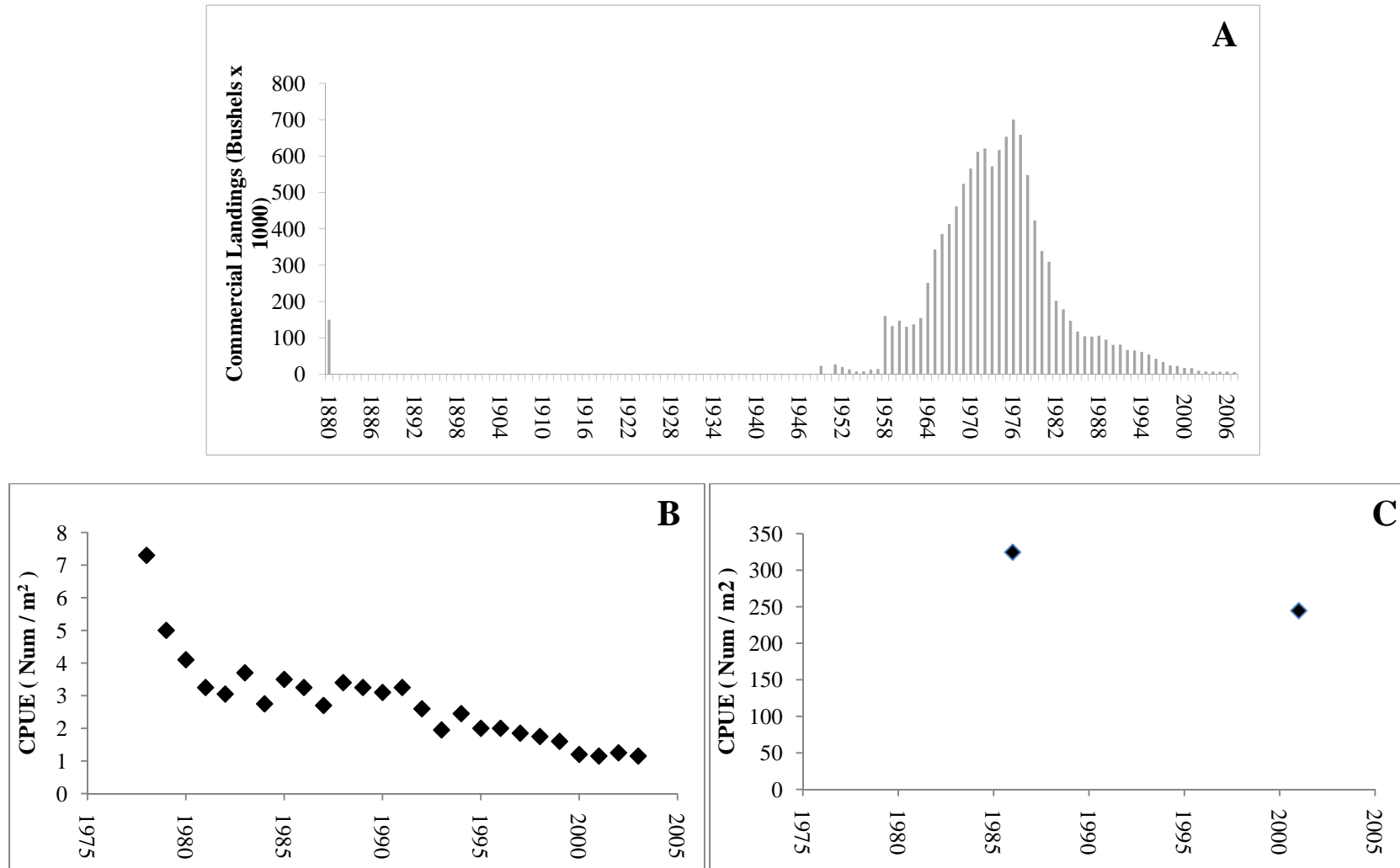
Figure 1.5. Compiled Commercial Landings for Blue Crabs.
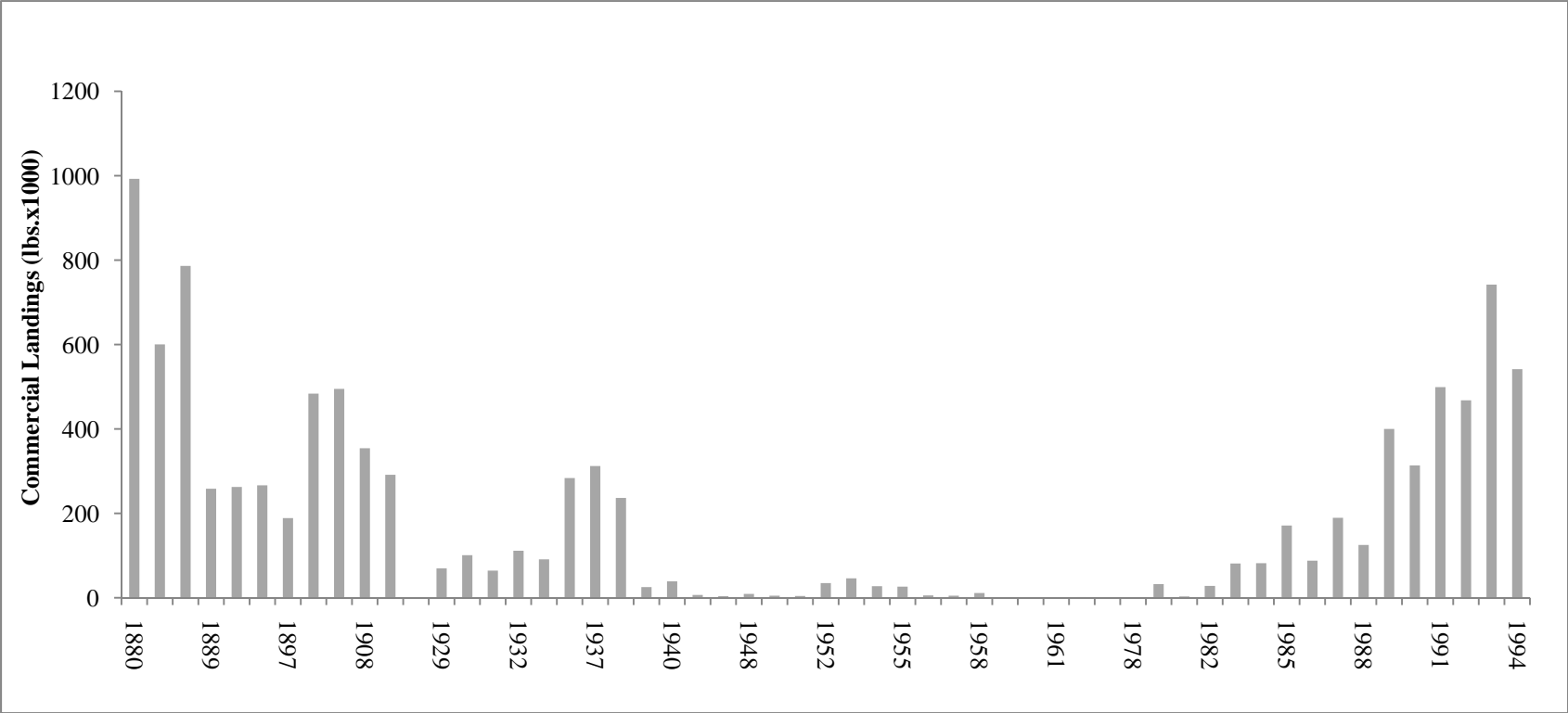
Figure 1.6. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Winter Flounder.
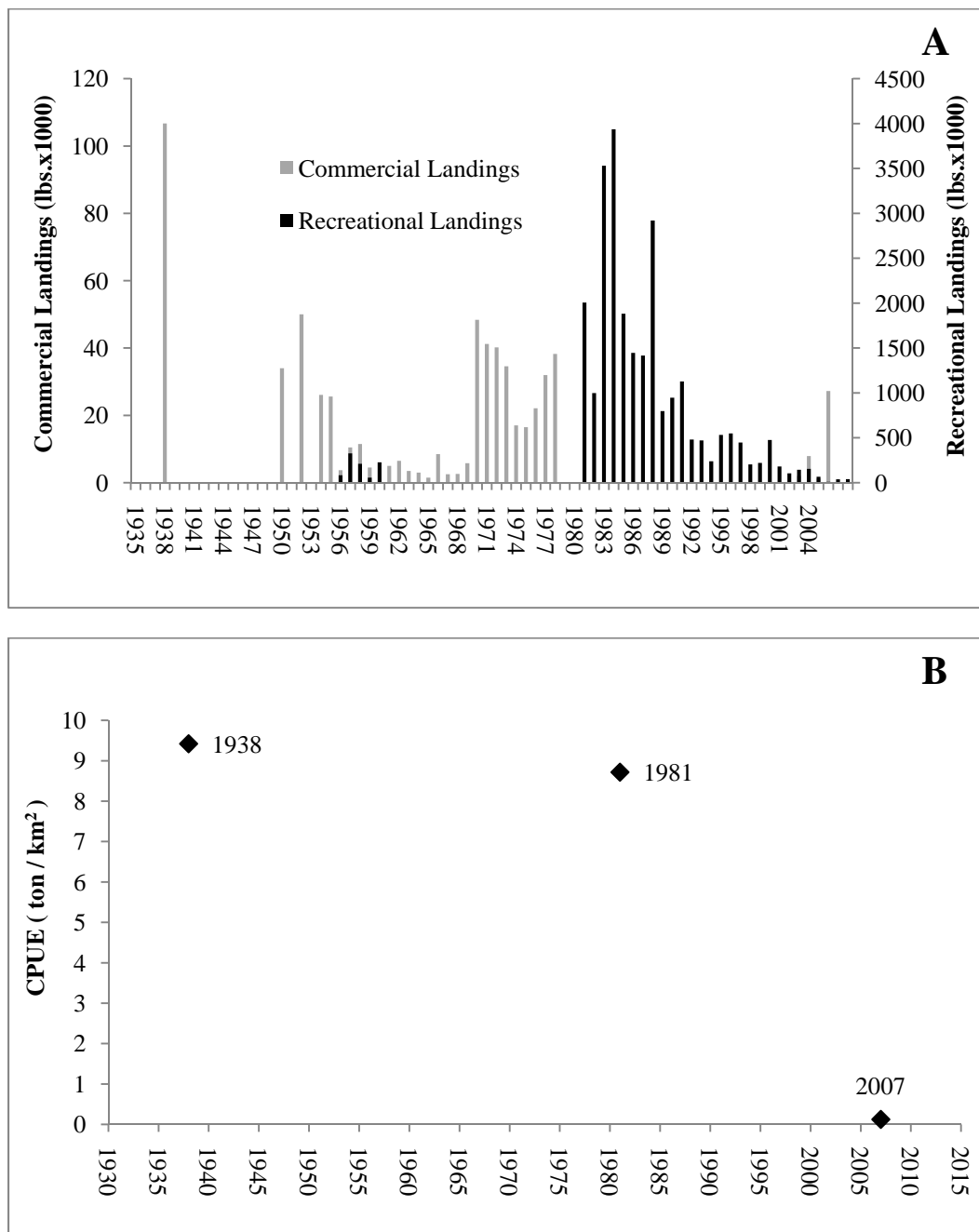
Figure 1.7. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Summer Flounder.
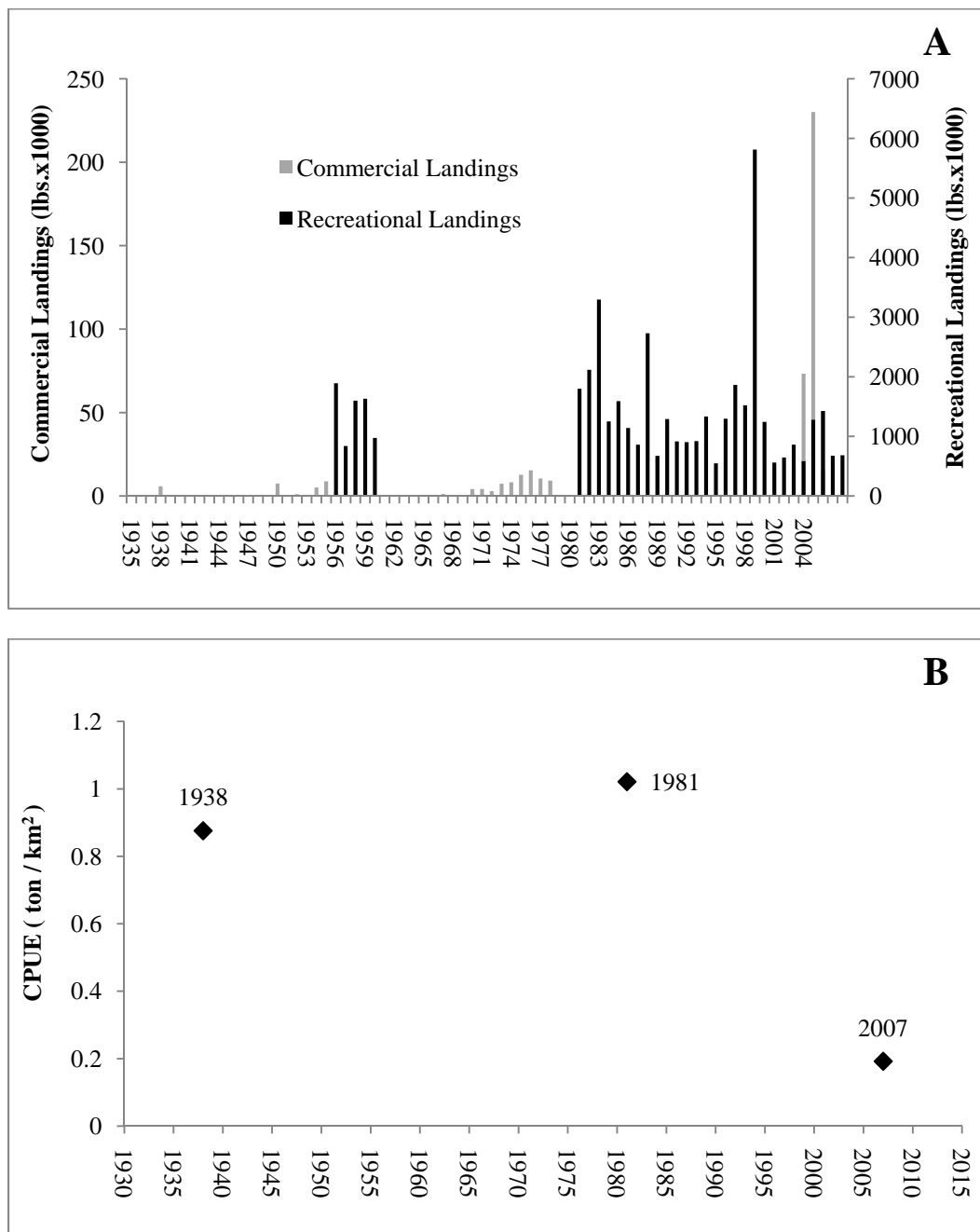
Figure 1.8. Compiled trawl CPUEs for Windowpane Flounder.
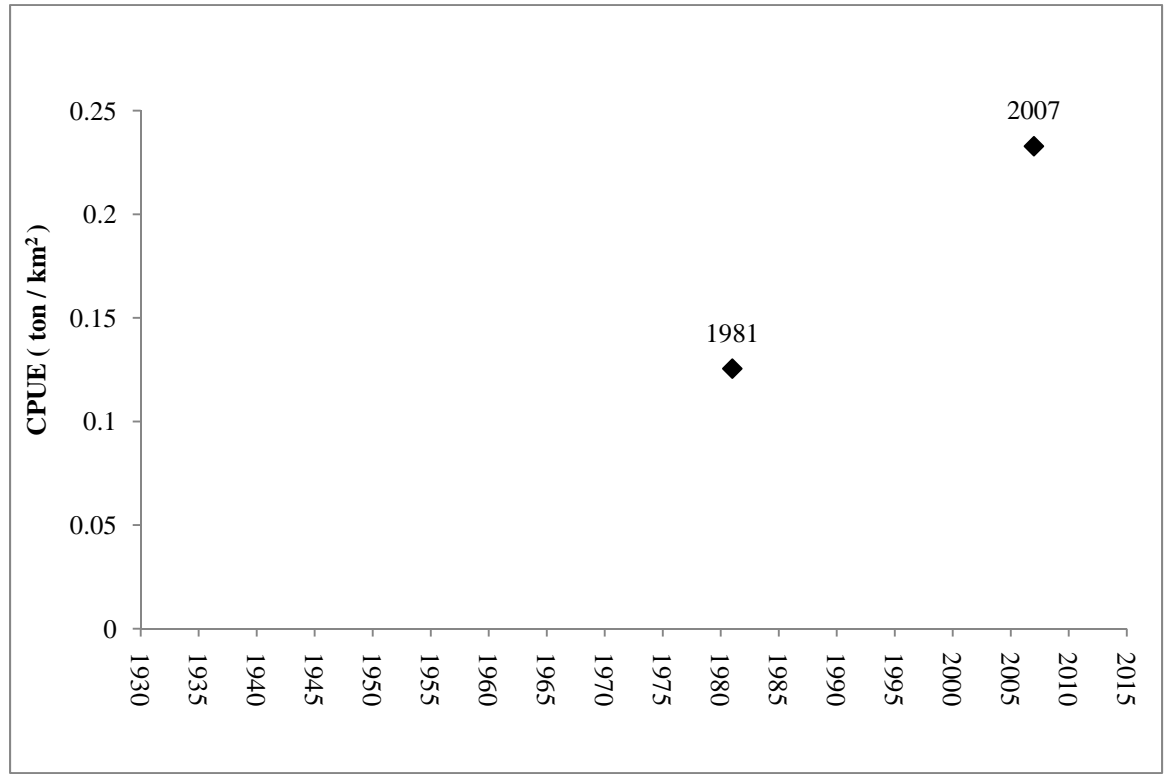
Figure 1.9. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Bluefish.
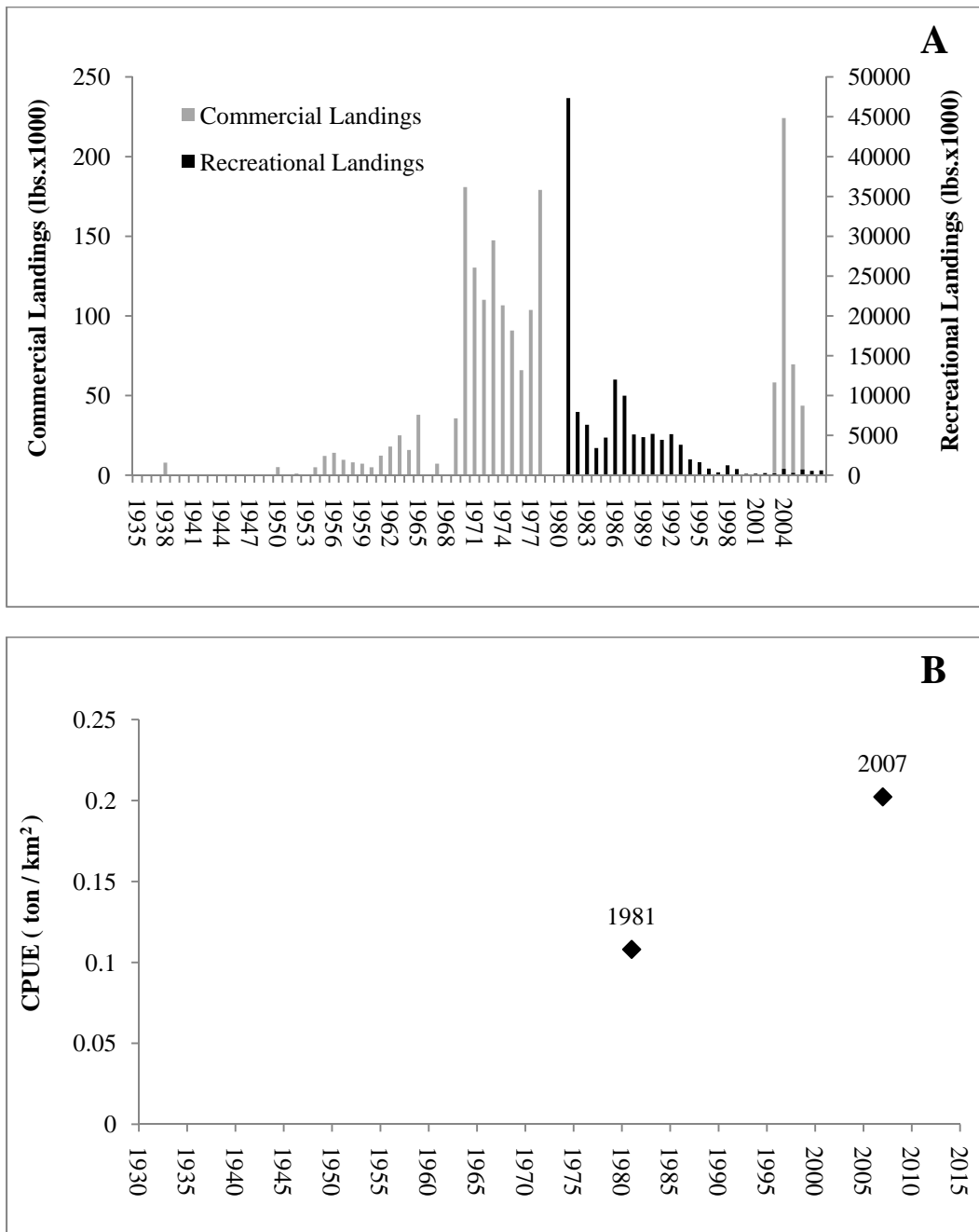
Figure 1.10. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Northern Kingfish.
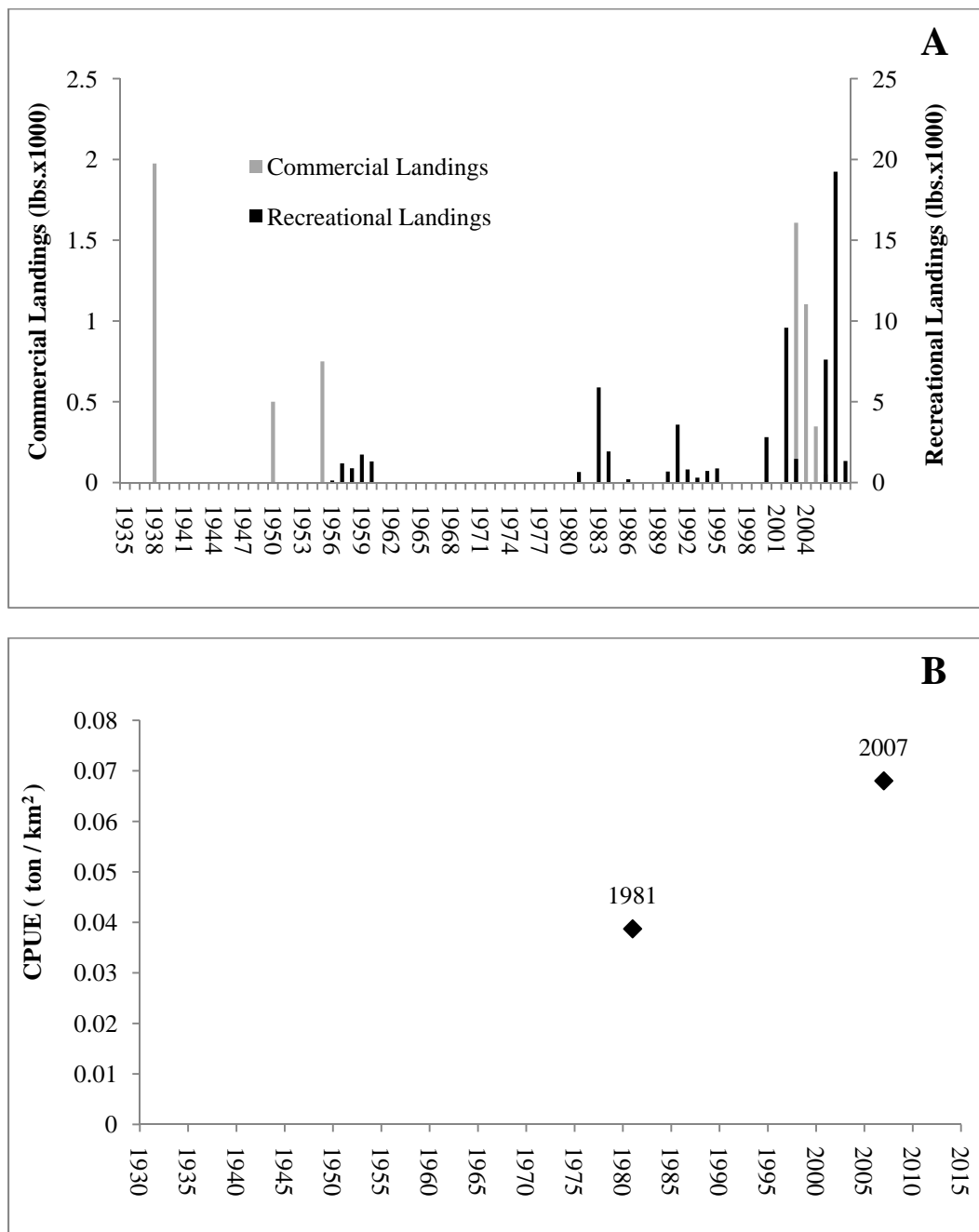
Figure 1.11. Compiled Commercial Landings (A) and trawl CPUEs (B) for Weakfish.
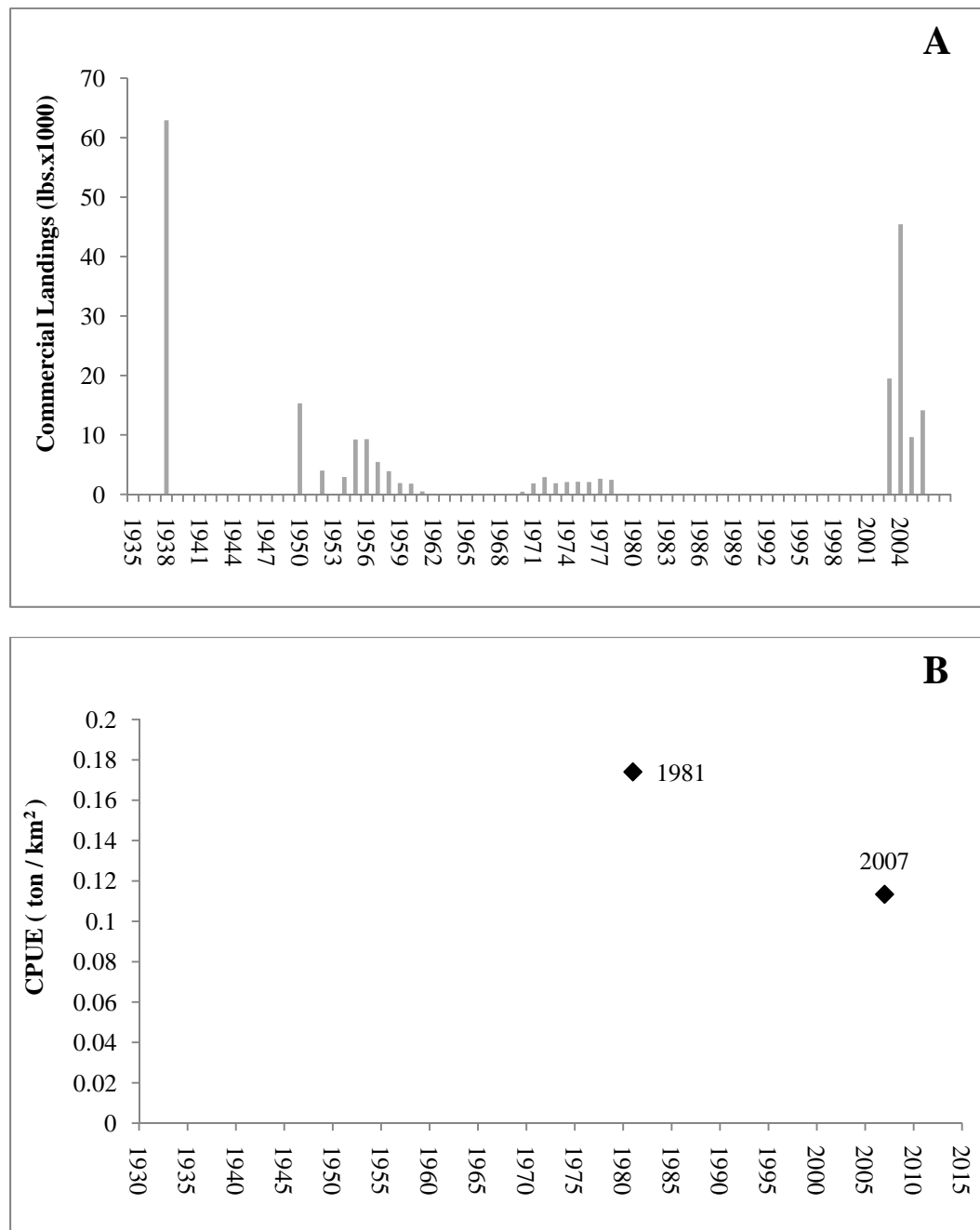
Figure 1.12. Compiled Commercial Landings (A) and trawl CPUEs (B) for Atlantic Menhaden.
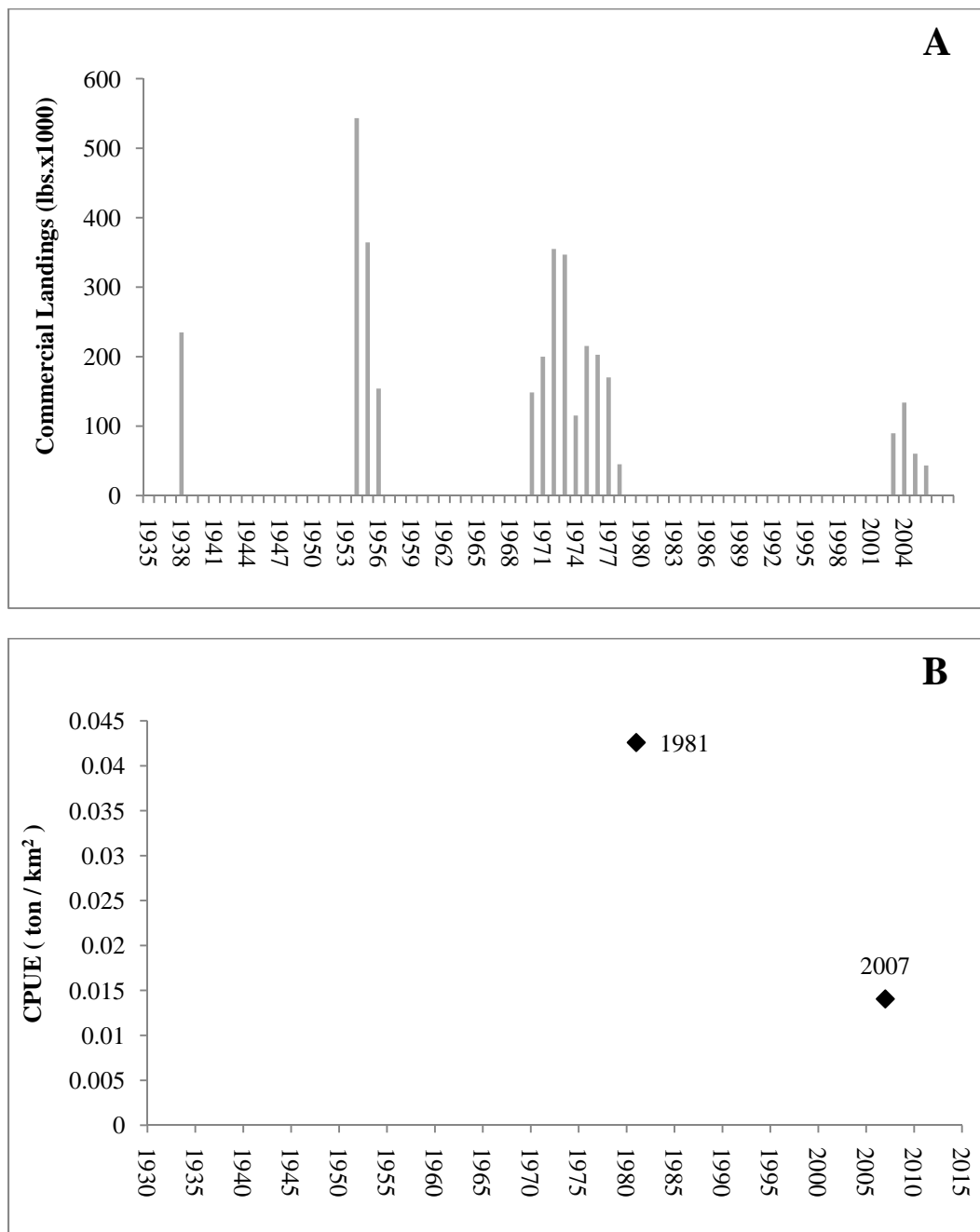
Figure 1.13. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Blackfish.
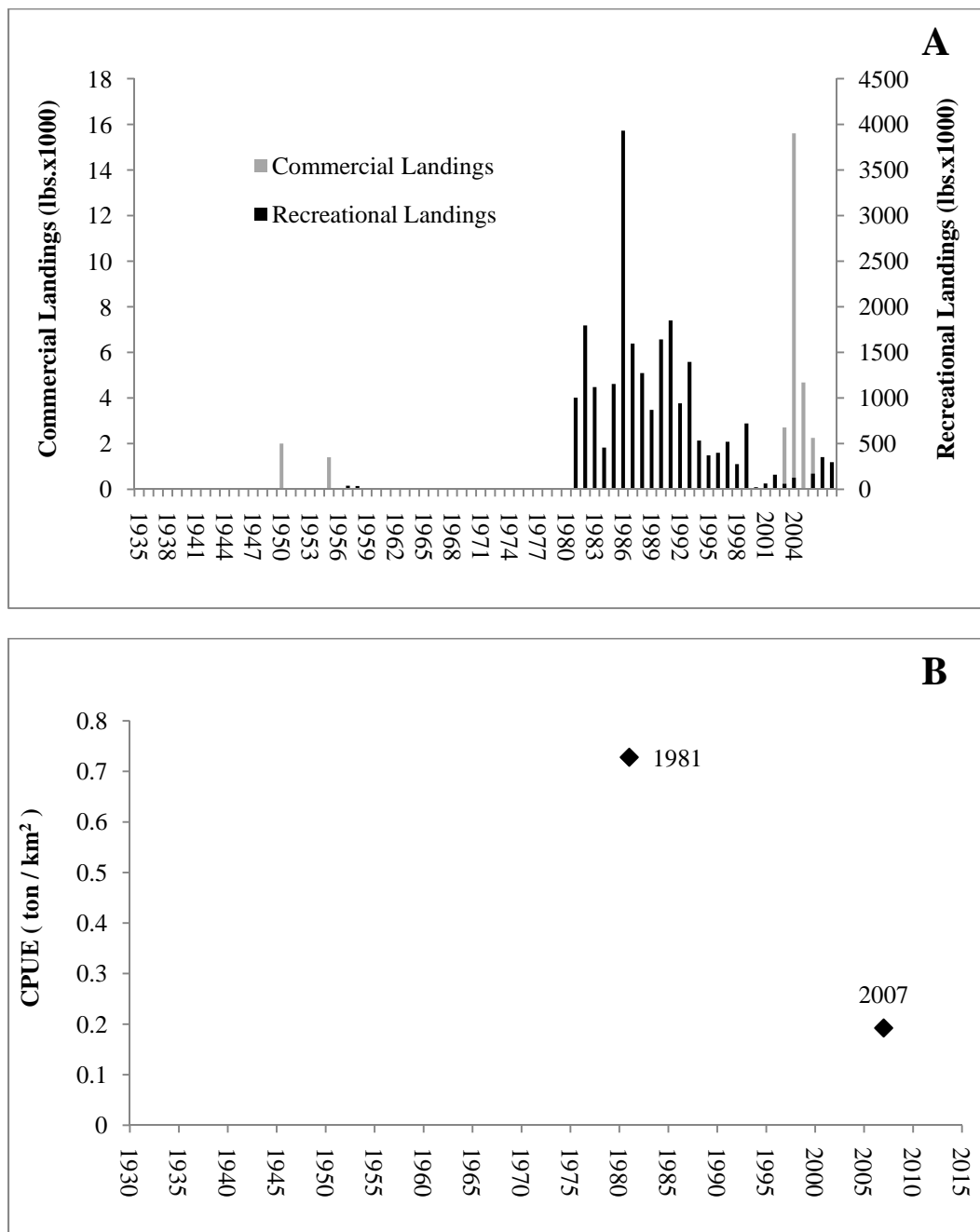
Figure 1.14. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Northern Puffer.
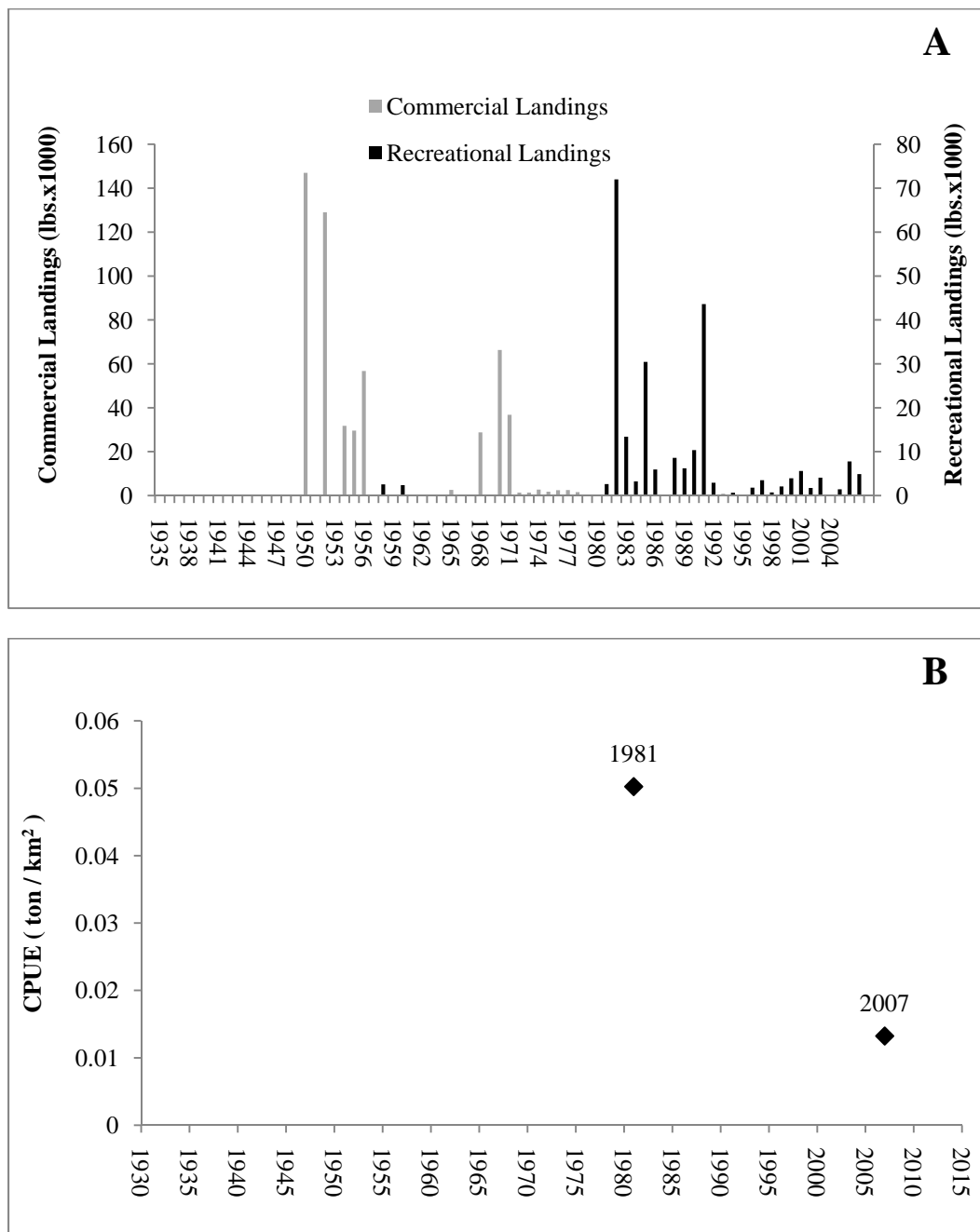
Figure 1.15. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Scup.
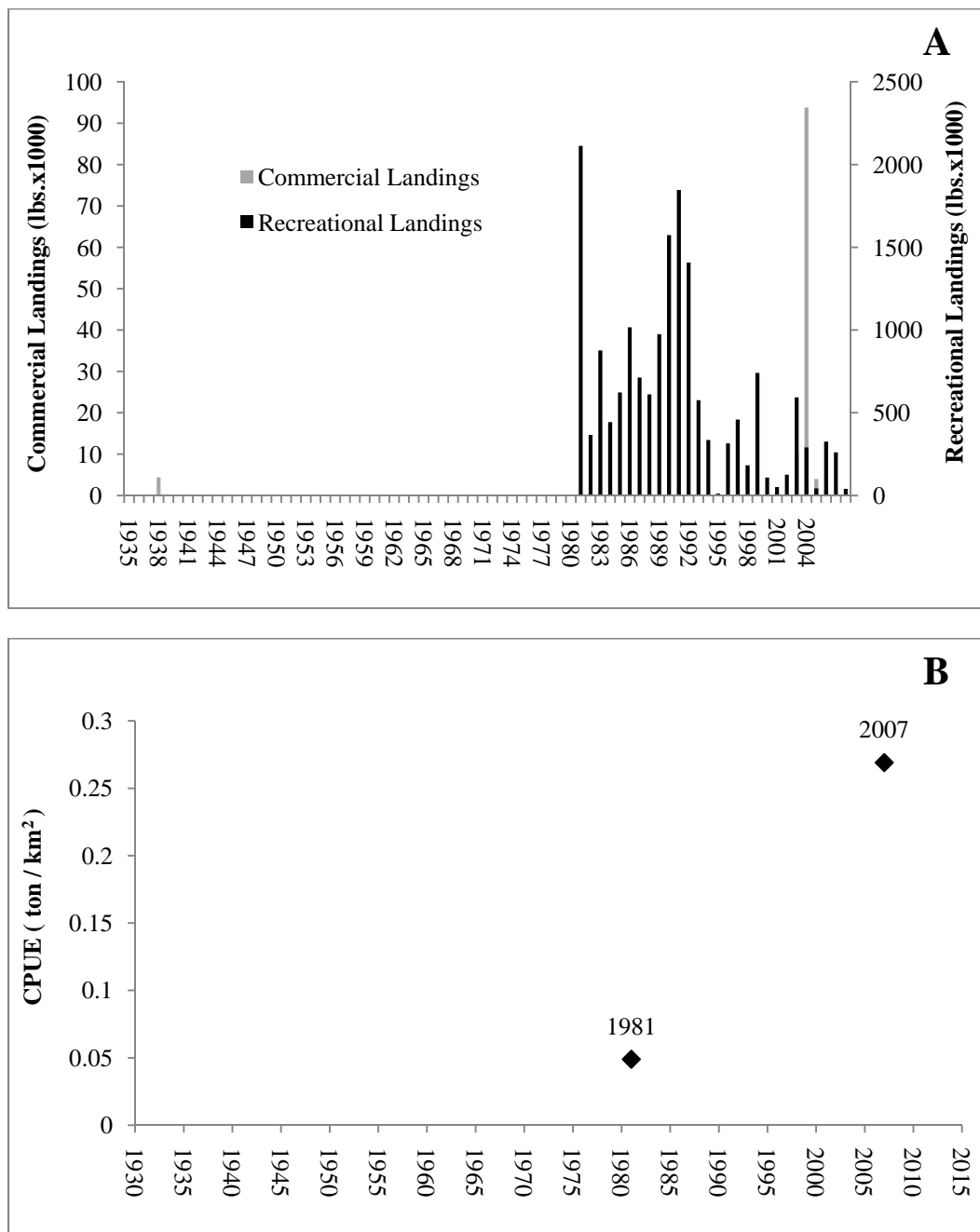
Figure 1.16. Compiled Commercial Landings (A) and trawl CPUEs (B) for American Eel.
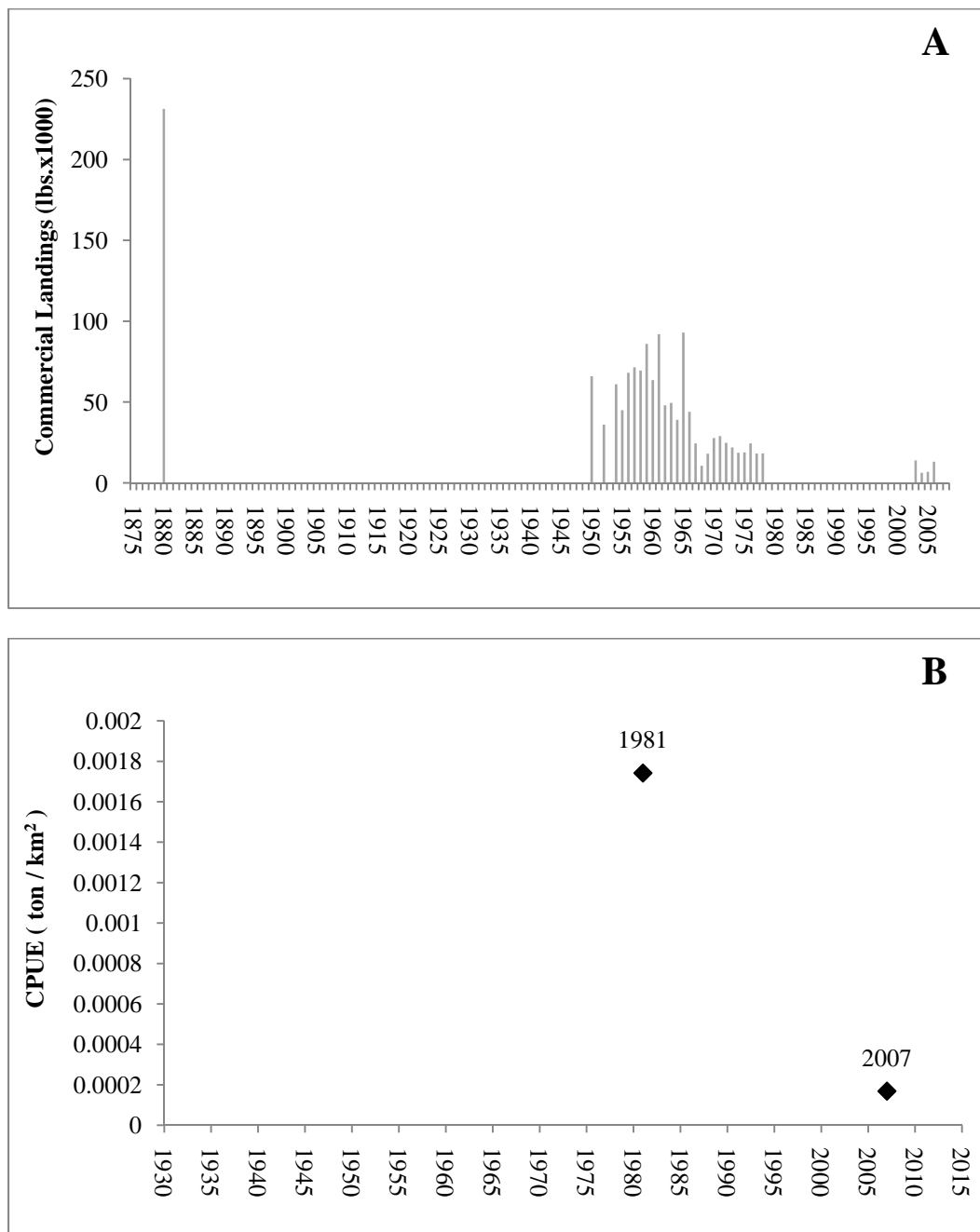
Figure 1.17. Compiled Commercial & Recreational Landings (A) and trawl CPUEs (B) for Striped Bass.
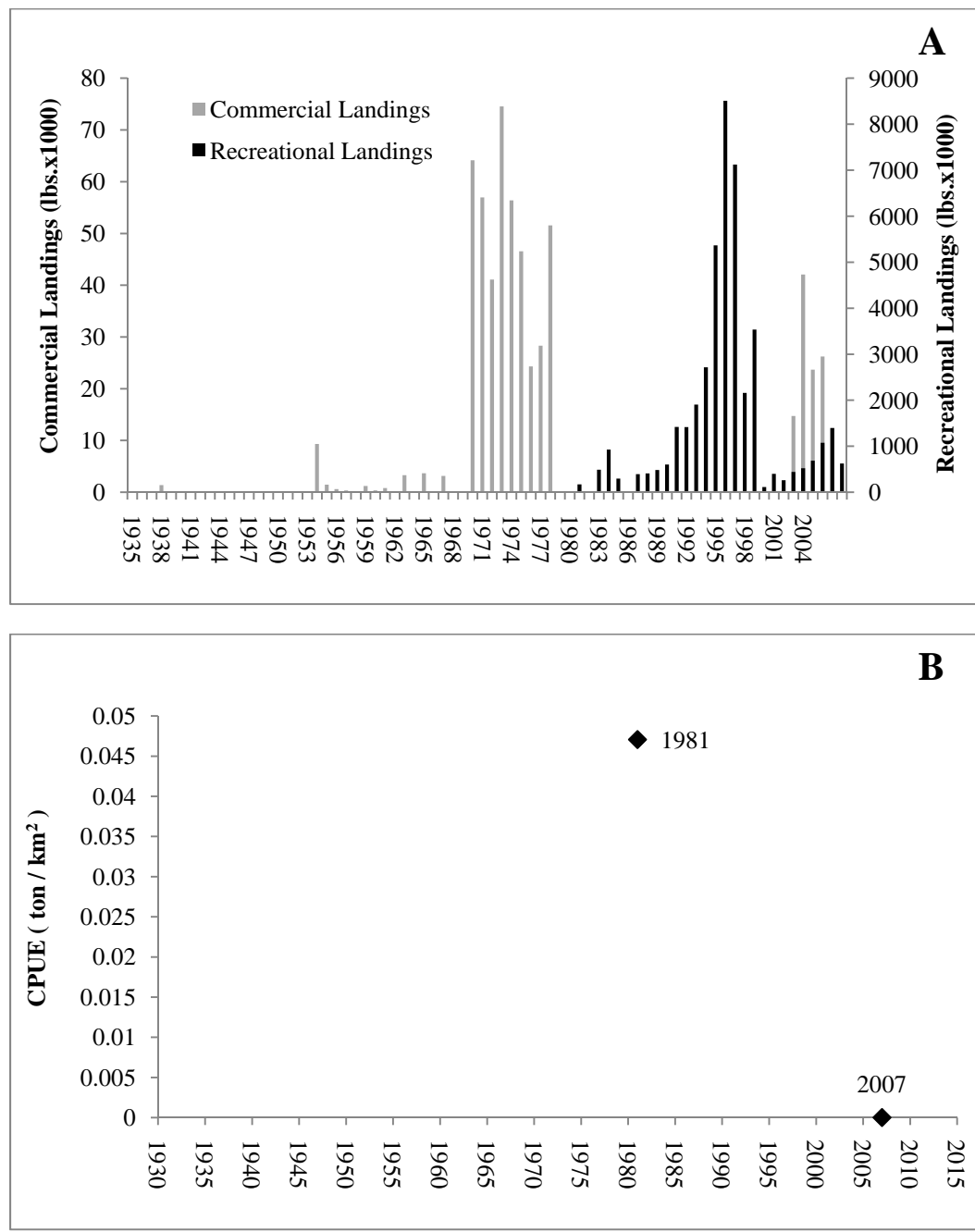
Figure 1.18. Compiled Recreational Landings (A) and trawl CPUEs (B) for Black Sea Bass.
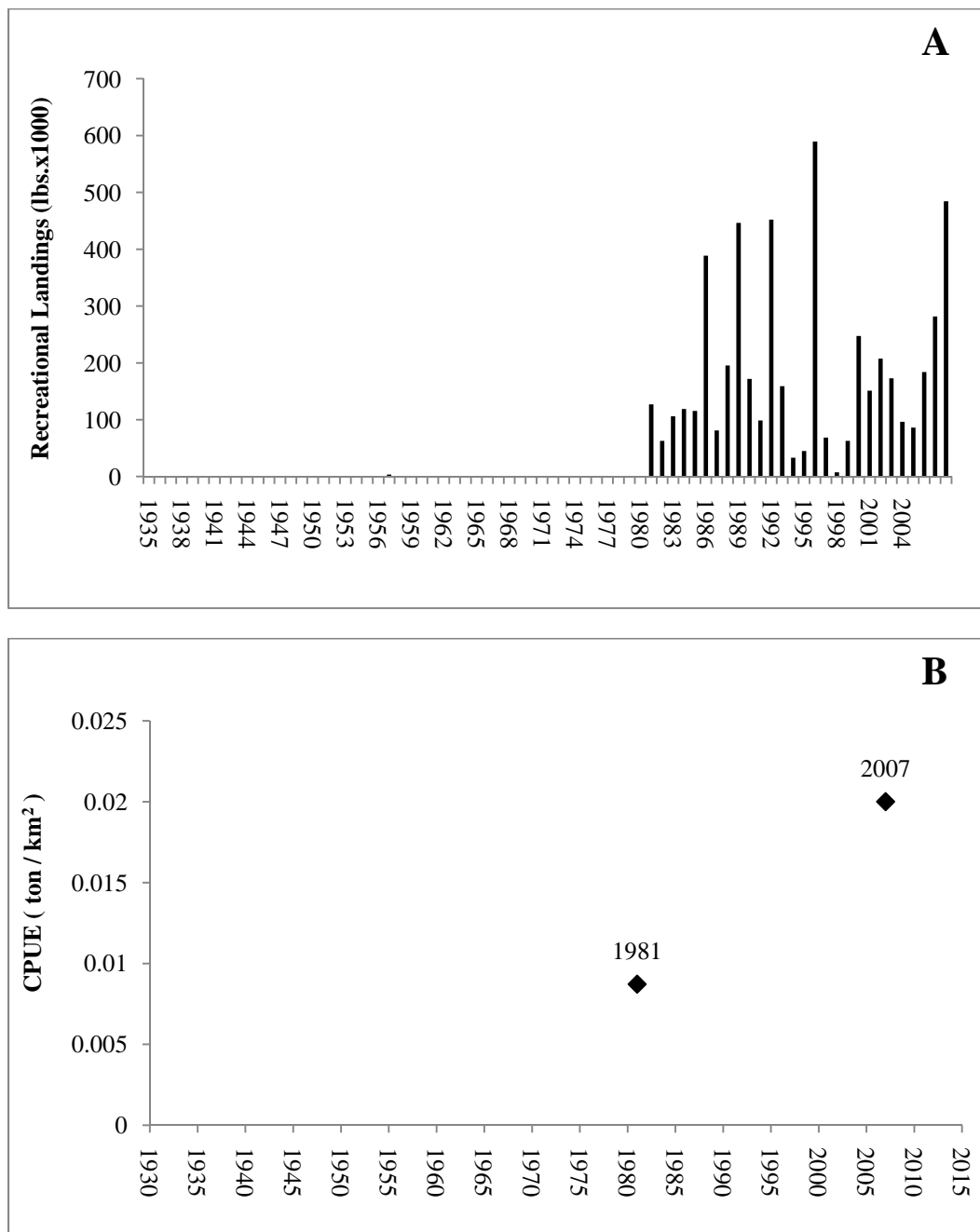
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Chapter 3: A Quantitative Assessment of the Ecosystem Structure of Great South Bay
using Ecopath

Introduction

It has been established that ecosystems undergo periodic "state" changes that, in some cases, constitute regime shifts (Scheffer et al. 2001, Folke et al. 2004, McManus and Polsenberg 2004, Steneck et al. 2004, Peterson et al. 2007). These ecosystem shifts have been linked to various pressures, such as anthropogenic impacts of fishing and pollution (Folke et al. 2004, Troell et al. 2005) or natural alterations to physical processes (Jeppesen et al. 2007). Ecosystem alterations large enough to be considered regime shifts are considered low frequency events (Carpenter 2003), occurring at a scale of decades to centuries.

The Great South Bay (GSB) resides along the southern shore of Long Island, New York and has undergone a number of these large scale shifts throughout its existence. This lagoonal system is a shallow, well mixed, highly productive ecosystem (Carpenter and Brinkhuis 1991) supporting a diverse array of finfish (Gabriel 1921, Neville et al. 1938, Schreiber 1973) and shellfish (Gabriel 1921, Greene 1982) and serves as an important nursery area for many species of waterfowl and migratory birds (Hanlon 1983). It has provided commercial, recreational, and ecological benefits to New York and the eastern United States seaboard for over a century (Gabriel 1921, Dickinson 1938, Westman 1938, McHugh 1991, Gobler et al. 2005).

Lagoonal systems are unique in that their ecosystem structure is generally retained despite the potential impact of stochastic events initiating departures from typically stable conditions (Day et al. 1989). However, even innately resilient ecosystems, such as lagoons, can become vulnerable to regime shifts (Scheffer et al. 2001, Folke et al. 2004). Indeed, GSB has undergone significant changes under various external pressures (Introduction). Most notable of these changes was the 1931 Nor'easter that reopened Moriches Inlet (Conley 1999) and initiated a transition from an oyster to hard clam dominated benthic community. Fishing effort, which had been overexploiting stocks for decades, transitioned from oysters to hard clams (Gabriel 1921) that would soon become overfished (Conrad 1982, McHugh 1991, Kraeuter et al. 2008). Phosphorus loading from duck farms (Ryther et al. 1957) and deterioration of eelgrass habitat (Short et al. 1987, Carpenter and Brinkhuis 1991) has also caused variations in GSB stocks.

The prominent role GSB has played in Long Island has not resulted in the level of research afforded to the Chesapeake Bay or Delaware Bay and, as a result, consistent biological sampling in the bay is lacking. An extensive literature review (Chapter 2) identified four time periods for which commercial and scientific data exist, allowing for the exploration of the ecosystem structure and species community of GSB. Each of these periods represented unique ecosystem "states" and included the following periods: 1880s, 1930s, 1980s and 2000s. Here we will present four ecosystem models that represent these four "states" to elucidate the interaction of biological, physical and anthropogenic (barrier island alterations, fisheries, pollution) influences that have shaped GSB over the last 120 years. Using ecopath, we will estimate trends in ecosystem structure, maturity, keystone species, fishery pressures, and ecosystem efficiency.

Methods

Ecopath model structure

The ecopath modeling approach utilizes a mass-balanced framework composed of trophically-linked biomass pools representing all major ecosystem functional groups (stocks) (Polovina 1984a, Polovina 1984b, Christensen and Pauly 1992, Walter et al. 1997, Pauly et al. 2000). Ecopath is based on two governing equations representing the total production (Equation 1) and consumption (Equation 2) of each stock within a system. In a mass balanced model, production is expressed as a function of all loss and gain processes to a stocks biomass and is estimated with the following equation:

Equation 1: $P_i = Y_i + (B_i \times M2_i) + E_i + BA_i + P_i \times (1 - EE_i)$

where P_i is the total production rate of stock (i), Y_i is the yield of the stock resulting from fishing mortality, $(B_i \times M2_i)$ is the total predation rate $(M2_i)$ acting on the biomass of the stock (B_i), E_i is the net migration rate (emigration – immigration), and BA_i is the biomass accumulation rate. $P_i \times (1 - EE_i)$ is the 'other mortality' rate (also denoted $M0_i$) that represents mortality from sources outside the system; contrary to production used by the system which is represented by $P_i \times EE_i$ (ecotrophic efficiency).

The consumption equation expands the removal component of system production; specifically, predation mortality $(B_i \times M2_i)$ for each stock modeled. Consumption represents the energetic needs of a particular predator and how much prey biomass is consumed. It is estimated by:

Equation 2: $(B_i \times M2_i) = (B_1 \times Q/B_1 \times DC_{1i}) + (B_2 \times Q/B_2 \times DC_{2i}) + \ldots + (B_n \times Q/B_n \times DC_{ni})$

where the proportion of the biomass of prey item (i) that passes to upper trophic levels via predation mortality ($M2_i$) is equal to the summation of the mortality components representing the various predators of prey (i) in the ecosystem. The mortality component of each predator on prey (i) is quantified as the product of the biomass of that predator (B_n), the ratio of the consumption (Q) to the biomass (B) of that predator (Q/B_n), and the percentage that prey (i) composes predator (1)'s total diet (DC_{ni}).

The Ecopath approach is similar to that of Winberg (1956) who assumed consumption equals the summation of gonadal and somatic growth, metabolic costs, and associated wastes. However, instead of measuring growth, the ecopath approach focuses on losses of a stock from both predation and fishery removals (Christensen et al. 2008). Thus, respiration (which is rarely measured directly) can be estimated as the difference between consumption, production and unassimilated biomass (Christensen et al. 2008). By accounting for production, consumption, and the resultant predator prey interactions, a mass-balanced Ecopath model assesses all the potential flows of biomass in a particular stock. Through this flow of biomass, namely predation, each stock can be tied to other trophic guilds. The mass-balanced approach relaxes the requirement for initial input parameters as only three of the four ecopath parameters (P/B, Q/B, EE, and P/Q) are needed to be estimated by the modeler while the fourth can be calculated by the software.

Historic Ecosystem Structure

The ecosystem structure developed in the GSB models incorporated information from scientific literature, state and federal government reports, and fishing periodicals. State and federal institutions included the New York State Department of Environmental

Conservation (NYSDEC) and from NOAA's Marine Recreational Fisheries Statistics Survey (MRFSS) and National Oceanographic Data Center (NODC). Historical records at the Long Island Maritime Museum were also accessed. Finally, newspaper articles were searched for the conditions of GSB waters and its fisheries for the 1880s and 1930s.

Data Sources

An extensive search was conducted using multiple resources to satisfy the demanding requirements of the ecopath parameterization routine to obtain initial ecopath parameters (P/B, Q/B, EE, or P/Q), biomass, dietary habits, and landings. Primary and secondary scientific sources provided the majority of our data. As these models were developed for the GSB ecosystem, studies that sampled within this bay were favored. When parameters were unavailable within GSB, other New York bays were used (Moriches Bay, Shinnecock Bay, Peconic-Gardiners Bay, Long Island Sound). For some parameters, we were required to obtain data from ecosystems in the vicinity of New York (Delaware Bay, Chesapeake Bay, Mid-Atlantic Bight, Gulf of Maine).

Biomass: Biomass estimates were obtained primarily from catch-per-unit effort (CPUE) measures in trawling studies conducted in GSB over the last century. While seining data was also available, seine surveys were conducted in different locations and did not incorporate the same range of years (1962 to 1981) as the trawling studies (1938 to 2007). However, seine surveys were used to estimate other needed parameters: length-weight relationships, stock-specific average weights, and measures of P/B. Productivity studies were also compiled to estimate the primary production of GSB.

Ecopath Parameters: Estimates of ecopath parameters were provided by multiple methods. Based on the work of Allen (1971), the P/B ratio was calculated to be equivalent with a stock's total mortality (Z). Using this, P/B was estimated from mark-recapture studies and length (or age) frequency data in catch surveys. The Hoenig (1983) approximation of mortality was also computed and compared to other P/B estimates. The Q/B ratio was estimated primarily by the method outlined by Fishbase (Froese and Pauly 2005). As a method to calculate EE was unavailable from the literature, this parameter was estimated solely from other ecopath models compiled from systems in the vicinity of GSB.

Diet: As with other parameters, preference was given to studies that were in relative proximity to GSB and came from stocks with similar dietary habits. A recent study on the diet of finfish in the Long Island region provided a wealth of information for many finfish stocks (Sagarese 2009). In an attempt to mitigate some of the uncertainty in the diet input, multiple studies were included for a particular stock when possible.

Compiling Initial Inputs

Data was collected and pooled into four distinct periods: the 1880s, 1930s, 1980s, and 2000s. These periods were chosen for two reasons: 1) they are representative of a unique state in the bay's history, and 2) available data allows for the construction of a model.

Once necessary parameters were obtained and compiled, estimates were compared within individual groups. If multiple estimates were available for a given stock parameter, this provided a general range of accepted values possible for that group and

allowed for an assessment of the accuracy in individual parameter estimates. As ecopath

only requires three of the four parameters to be initially input for a grouping and many

stocks had multiple estimates available for various parameters, certain parameters were

held in higher regard when choosing initial inputs. Estimates of stock abundance (CPUE)

were used whenever possible. Area swept, used as the units for fishing effort, was used

to compare trawl-based CPUE measures (used as the biomass index) between time

periods. P/B estimates were also highly regarded, although seldom found. Q/B values

were considered fairly consistent within an ecopath grouping and were altered only if the

program changed initial inputs. EE was only input when other parameters were lacking.

This decision related to the difficulties in estimating this parameter independent of the

ecopath routine (Christensen et al. 2005). However, if inaccuracy was suspected in the

dynamics of a modeled stock and measures of Q/B and EE were available, ecopath was

used to estimate either the stock biomass or P/B as allowed under the assumption of

mass-balance in the ecopath routine (Equation 1).

Balancing Ecopath

The first run of all four models required the initial parameter inputs to be altered.

Parameters with the most uncertainty were altered before parameters with greater

accuracy. Estimates derived from GSB were deemed the most accurate followed by

estimates from nearby systems. Furthermore, as our models represented the same system

across four distinct periods, decade-dependent inputs were required. However, a paucity

of data necessitated a recycling of some parameter inputs. Initial inputs were only

changed if dictated by the balancing process and adjustment could be justified by the

dynamics of the stock. While this eliminated some of the modeler's inherent bias, any

inter-decadal variation present between the system parameters may have been missed, introducing error into the models.

Comparing System States

Once constructed, various summary statistics were analyzed to evaluate the ecosystem structure of GSB and make comparisons between periods or system states. These indices were chosen based on their ability to measure one of four aspects of ecosystem structure: 1) the basic structural properties of the ecosystem, 2) maturity of the system, 3) relative impact of particular stocks within each model as well as their importance, defined here as "keystoneness", and 4) overall impact of the fishery. Ecosystem structure was represented by measures of net primary production (PP), system respiration (RESP), total system throughput (TST), and total system biomass in the present study. Appropriate measures of ecosystem maturity were obtained from various sources. The findings of Odum (1969), Ulanowicz (1986), Herendeen (1989), and Christensen (1995) were considered to assess maturity. Measures of stock-specific ecosystem impacts included mixed trophic impact (ε) and keystoneness (KS) while the impact of the fishery was modeled with fishing mortality and landings. A brief explanation of these calculations is provided below.

Basic Ecosystem Structural Properties

PP: Total primary production (PP) provides an activity index of the ecosystem across its lower trophic levels. Calculated as the summation of production across all primary producers, the PP estimate requires the biomass and P/B outputs from the ecopath program.

RESP: Conversely, system respiration (RESP) provides an activity measure of the top levels of the trophic web. Represented as the proportion of consumption not put into production, it can be calculated using:

$$\text{Equation 3:} \quad \text{RESP} = (1 - GS_i)\, Q_i - (1 - TM_i)\, P_i$$

where GS_i is the unassimilated consumption of group (i) and the TM_i parameter is reserved for mixed producer/consumer groupings and represents the production of group (i) that is attributed to primary production (PP). By estimating the total production and consumption from biomass and P/B or Q/B ratios respectively and assuming the proportion of unassimilated consumption is approximately 0.2 (Winberg 1956), the total system respiration can be calculated.

TST: Total system throughput (TST) represents all of the biomass flows within an ecosystem. Estimated as the summation of four energy flow quantities; total system consumption, system respiration, flows out of the system (export), and flows to detritus; TST signifies the 'size of the entire system in terms of flow' (Ulanowicz 1986) and is an important parameter for comparisons of energy flow between systems. While total biomass is also an index for ecosystem size, the measure does not account for variations in species-specific parameters (production, respiration, consumption, EE) that are important to energy flow. Furthermore, as the ecopath routine connects different trophic groups via the flow of biomass, TST was believed a more appropriate measure of ecosystem size.

System Maturity

The first attempt to characterize ecosystem maturity was made by Odum (1969) in which he presented 24 potential system attributes. Ulanowicz (1986) augmented this list with his work on ascendency. Herendeen (1989) evaluated four of these indices based on their sensitivity to fluctuations in the system: energy intensity, residence time, ascendency, and exergy. While Herendeen's study assumed the more sensitive indicators deserved the most attention, the systems analyzed in this study are separated by forty years on average. Therefore, we were equally concerned with sensitive indicators as measures that may change slowly over time. The assessment of Christensen (1995) was, therefore, thought more applicable to the present analysis. Combined with various measures of exergy (Mejer and Jorgensen 1979) and ascendency (Ulanowicz 1986), Christensen (1995) investigated the Odum indices based on their explanatory power. Four maturity measures were arbitrarily selected from the Christensen study and assessed in the GSB system: the ratio of primary production to respiration (PP/RESP), system biomass to total system throughput (B/TST), ascendency to capacity (A/C), and the system omnivory index (SOI).

PP/RESP: The ratio of system primary production to respiration (PP/RESP) is considered an important index of ecosystem maturity. As an ecosystem begins development, the majority of its production is expected to go unutilized (Christensen et al. 2005). In these early systems, even if primary consumers exploit 100% of system primary production, higher level consumer stocks (secondary, tertiary, quaternary) may not have equilibrated with an abundant prey, leaving a large proportion of production unused and producing a PP/RESP > 1. However, as the system develops into a mature

stable state and consumers become more efficient at utilizing the production at the bottom of the trophic web, the PP/RESP ratio approaches one (Odum 1971).

B/TST: The ratio of total system biomass to total system throughput (B/TST) provides a measure of the amount of energy that is utilized by an ecosystem (B) relative to the amount of energy available (TST). This ratio provides another index of maturity in that large values of B/TST are observed in mature systems (Odum 1971).

A/C: Ascendancy (A) provides a measure of an ecosystem's size (flow of biomass) and development (organization) of network links. Specifically, ascendancy measures the relative uncertainty in which path a particle of biomass will travel within a trophic web (Christensen et al. 2005, Cruz-Escalona et al. 2007). High measures of ascendency are correlated with systems in which biomass can flow with similar likelihood through numerous trophic paths such that the uncertainty in flow of an energy particle is relatively large. As the number of potential trophic linkages between groupings is finite, ecosystems possess an upper limit in ascendancy, termed the development capacity (C). Unlike ascendancy, the ratio of A/C is inversely related to system maturity (Christensen 1995) as it measures the fraction of potential organization that the system has attained (Cruz-Escalona et al. 2007). The difference between ascendancy and capacity is defined as the system overhead, serving as a reserve source of energy should the ecosystem become stressed (Ulanowicz 1986) where stocks consume resources not typically utilized. In this study, energy-based ascendancy was measured.

SOI: The system omnivory index (SOI), defined as the average OI_i for all consumers weighted by their respective food intake, gauges the degree of connectance between trophic levels. The omnivory index of each consumer (OI_i) is estimated by:

$$\text{Equation 4.} \quad OI_i = \sum_{j=1}^{n} (TL_j - (TL_i - 1))^2 \; x \; DC_{ij}$$

When $OI_i = 0$, the predator is considered specialized in that it feeds completely at one trophic level whereas a large OI suggests an indiscriminate consumer. While the connectance index (CI) offers a similar measure, it is too dependent on the taxonomic detail arbitrarily chosen in the separation of stocks into ecopath groupings (Christensen et al. 2005) and was not assessed in this section. Mature ecosystems are expected to possess a more weblike trophic structure than developing systems (Christensen 1995), so SOI was expected to increase with maturity.

Unfortunately, small changes were observed between the above maturity indices, warranting the inclusion of additional indices from Christensen's analysis (1995) of the Odum study (1969). While some values were included, others were found more applicable as indicators of ecosystem structure: PP, RESP, TST, and system biomass (B). Additional maturity measures included the residence time of energy in the system (B/[RESP+EXP]), connectance (C), net community yield (PP – RESP), dominance of detritus (Dom.Det), Finn's cycling index (FCI), predatory cycling index (PCI), nutrient regeneration (FCI – PCI), path length (PL), straight-through path length (SPL), the Schrodinger ratio (RESP/B), system ascendency (A), system overhead (O), internal redundancy or overhead on internal flow (Redund.), nutrient conservation or overhead on external flow (O_{ex}), information content of flows (I), exergy (EX), structural exergy

(EX$_{st}$), inverse of system production to biomass (P/B^{-1}), the ratio of PP/B, and another measure of biomass supported per unit energy flow (B/[PP+RESP]) in which B/TST provided a similar index. Of the given maturity indices, all were assumed to increase with system maturity except PP/B, PP – RESP, and A/C based on Christensen (1995). For further details on the computation and explanation of these measures, refer to Christensen (1995).

Species Impact and Keystoneness

The Ecopath software estimates the interaction between stocks with the calculation of the mixed trophic impact (MTI). Introduced by Ulanowicz and Puccia (1990), the net impact of stock (i) on (j) (q_{ij}) is calculated with:

$$\text{Equation 5.} \quad q_{ij} = d_{ij} - f_{ji}$$

where d_{ij} represents the proportion of prey (j) in the diet of predator (i) (positive effect) and f_{ji} represents the fraction of total predation on (j) that is from predator (i) (negative effect). In this, the software assumes predator-prey relationships, not competition, form the foundation of inter-stock relationships. Once the various q_{ij}'s are calculated, the mixed trophic impact of (i) on (j) (MTI$_{ij}$) is estimated as the product of all the net impacts (q_{ij}) across all potential trophic pathways connecting the two stocks.

In addition to the comparisons drawn between two distinct stocks, the impact that a particular stock has on the entire ecosystem (ε) can be assessed. This ecosystem effect is measured as the MTI with:

$$\text{Equation 6.} \quad \varepsilon_i = \sqrt{\sum_{j \neq i}^{n} MTI_{ij}^2}$$

and estimates the net effect of group (i) on all other stocks in the system. However, as abundant stocks are more likely to impart a large impact on their ecosystem, Libralato et al. (2006) introduced the keystoneness (KS) index. This measure scales ε with biomass, "penalizing" a stock with high abundance. With this, the KS index attributes high values to stocks which impart large impacts while maintaining a low biomass. The biomass component (p_i) is represented as the contribution of a particular stock to the total biomass of the system:

$$\text{Equation 7.} \quad p_i = B_i / \Sigma_k B_k$$

Using ε_i and p_i, the keystoneness of species (i) (KS_i) can be calculated with:

$$\text{Equation 8.} \quad KS_i = \log [\varepsilon_i (1 - p_i)]$$

Fishery impacts

Two indices of fishing pressure were available to estimate the impact of GSB fisheries on stock abundances: fishing mortality (F) and total landings. As these two values are correlated through stock biomass, only one measure was used. The periodic fluctuations in fishing mortality, therefore, were used to assess the relationship of biomass with fishing pressure.

Ecosystem Efficiency

In addition to these indices, the efficiency of GSB was assessed by analyzing the proportion of available production utilized by system consumers. Preliminary models were compiled by allowing the software to estimate the primary production required by the ecosystem (PPR) based on the energetic demands of consumers. Once balanced,

models were re-run with inputs of primary production from independent productivity surveys. The direct measures of GSB productivity were then compared to the consumer-based estimates computed by ecopath to assess the efficiency of system consumption. PP/RESP and PP/B ratios were also thought indicative of ecosystem efficiency as these maturity indices provided measures of system production that were converted into useable biomass.

Shifts in Trophic Assemblages

The relative abundance of migratory stocks and groundfish was assessed a priori after an assessment of model outputs. Migratory stocks are defined as groups that consistently move in and out of the GSB system; black seabass, bluefish, gadids, menhaden, scup, sharks, squid, striped bass, summer flounder, weakfish, winter flounder, and tropical fish comprise this group. Groundfish are described as stocks that live on, in, or near the benthos and include blackfish, drums and croakers, gadids, flatfish-other, oyster toadfish, scup, skates, sea robins, striped bass, summer flounder, weakfish, and winter flounder.

Results

Sources of Input Data

  Biomass: Few comparable surveys were done within GSB, complicating the acquisition of abundance data. Sources included: Thorne (1928); Dickinson (1938); Lobell (1938); Nesbit (1938); Neville et al. (1938); Townes (1938); Westman (1938); Hanlon (1983); and Frisk and Munch (2008). CPUE estimates were obtained from Frisk and Munch (2008) for the 2000s stocks in terms of length, not biomass as required by ecopath. Therefore, length:weight conversion factors (Table 2.1) were used to estimate stock abundances. Similarly, the Hanlon survey (1983) measured 1980s abundance in terms of numbers caught. This index was converted into biomass using both length:weight conversion factors (Table 2.1) and estimates of average individual lengths. Species-specific average lengths of the 1980s were provided by multiple sources: Alperin and Schaefer (1964), Schaefer (1967), Briggs and O'Connor (1971), Gaw (1972), Schreiber (1973), Briggs (1977), and Briggs (1978). While direct measures of abundance were found for the 2000s and 1980s models, estimates of abundances in the 1930s and 1880s were obtained from various literature sources. CPUE measures from the 1930s were estimated from landings provided by commercial trawlers operating within GSB (Dickinson 1938). However, effort units in this study (days fished) needed to be converted into a measure of area covered to allow for comparison to the 1980s and 2000s abundance indices. This calculation assumed: 1) the net's mouth spread 12.2 meters when dragged (Dickinson 1938), 2) the "slow" speed at which trawls were pulled was assumed 1 m/sec, and 3) one day fished was equivalent to the standard used by the National Marine Fisheries Service (24 hours of fishing). The biomasses of stocks not

landed by the commercial trawls were predicted from landings supplied by pound nets, gill nets, haul seines, and benthic grabs. As no quantitative abundance estimates were discovered for the 1880s, catch records, qualitative data sources, and comparisons to the 1930s model were used to estimate values for the 1880s. Productivity estimates were supplied by phytoplankton assessments from Lively et al. (1983) and Lonsdale et al. (1996).

Ecopath Parameters and Diet: Initial estimates of parameters were obtained from various studies conducted within and outside GSB including the following: Delaware Bay, Chesapeake Bay, Middle Atlantic Bight, and Gulf of Maine. Decade dependent estimates were used when possible, but these were rarely available from the literature. For brevity, the sources used to estimate these parameters and an explanation of the alterations required to these inputs are not listed here, but are explained in detail in Appendix 2.1.

Fishery Landings: Landings data reported by the New York State Department of Environmental Conservation (NYSDEC) and the Marine Recreational Fisheries Statistics Survey (MRFSS) provided estimates of the fishery yields for the 2000s and 1980s. Thorne (1928), Dickinson (1938), Westman (1938), and Briggs (1998) provided landings for the 1930s; while 1880s landings were found in Mather (1884), Bean (1891), Thorne (1928), and Briggs (1998).

The Balanced Models

The final 2000s and 1980s ecopath models contained 42 functional groups. The 1930s and 1880s models were composed of 44 groupings, the additional categories the

result of the separation of oysters from the suspension feeder grouping and the inclusion of sharks. The final parameters for each ecopath grouping are listed in: Table 2.2 for the 1880s, Table 2.3 for the 1930s, Table 2.4 for the 1980s, and Table 2.5 for the 2000s.

Basic Ecosystem Structural Properties

Various summary statistics exhibited significant declines between models (Table 2.6). Despite differing fluctuations in stock abundances, the total system biomass of GSB (detritus excluded) has consistently dropped since 1880 (Figure 2.1A). To mimic these trends, the calculated total system throughput (TST) also showed consecutive declines (Figure 2.1A). Despite the agreement in direction, the magnitude of decline varied amongst these indices. While the drop in system biomass followed a linear reduction (r^2=0.997), the periodic decline in TST showed more variation. Similar reductions were calculated in the total respiratory flows of the system (Figure 2.1B). These indices all suggest the ecosystem size of GSB has been drastically reduced over the last 120 years.

While no fluctuations were recorded in net primary production across the compiled system models (Figure 2.1B), the lack of contrast was by design (Appendix 2.1). Therefore, PP was not used as an indicator of ecosystem size for the compiled models. However, the increase in net system production (Figure 2.1C) indicates a reduction in the utilization of system production, supporting the measured declines in system consumers.

System Maturity

The four original indices used to gauge the maturity of the GSB ecosystem showed an overall decline in system maturity over the last century. The large deviations

from unity in the PP/RESP index (Figure 2.2A) indicate a large reduction in maturity

occurred. Although the decline in B/TST (Figure 2.2B) and A/C (Figure 2.2C) support

this reduction in system maturity, only small drops were recorded. The calculated

measures of SOI (Figure 2.2D) predicted an overall decline in maturity. However, a

relatively large increase (+8.4%) in SOI was observed between the 1980s and 2000s

models, signifying an increase in system maturity.

Although the contradicting increase in the 2000s SOI index was the only apparent

inconsistency, the inclusion of only four indices raised suspicion on even a single

conflicting measure. Therefore, additional maturity indices were included to assess the

accuracy of the predicted trends described above (Table 2.7). Including the four indices

above, 22 of the 24 included maturity measures estimated an overall drop in the

ecosystem maturity of GSB. Of the two remaining indices, one predicted an increase

(PCI) while the other showed no variation (Dom.Det.). These trends agree with the

overall maturity decline predicted by the four original indices. The computed trends

predict a significant reduction in the maturity of the GSB ecosystem since the 1880s.

Species Impact and Keystoneness

As relative stock abundances fluctuated across periods, variations in the net

impact of each stock (Figures 2.3-2.6) were also observed. Starting with ecopath

groupings with the largest overall impact on the system (ε), the 1880s system was heavily

influenced by menhaden ($\varepsilon = 2.402$, $KS = 0.332$), phytoplankton ($\varepsilon = 2.161$, $KS = 0.324$),

bluefish ($\varepsilon = 2.044$, $KS = 0.303$), forage fish ($\varepsilon = 2.041$, $KS = 0.299$), and striped bass (ε

$= 2.029$, $KS = 0.307$). Fluctuations to the 1930s system caused variations in stock

impacts, where weakfish (ε = 2.139, KS = 0.325), crabs (ε = 1.612, KS = 0.179),

phytoplankton (ε = 1.593, KS = 0.196), forage fish (ε = 1.498, KS = 0.173), and benthic

fauna (ε = 1.479, KS = 0.143) had the largest measureable effects. In the 1980s system,

striped bass (ε = 2.625, KS = 0.416), bluefish (ε = 2.452, KS = 0.385), forage fish (ε =

2.055, KS = 0.307), benthic fauna (ε = 1.632, KS = 0.180), and phytoplankton (ε = 1.421,

KS = 0.146) had the greatest influence on GSB. Currently, summer flounder (ε = 2.564,

KS = 0.401), striped bass (ε = 2.162, KS = 0.325), detritus (ε = 1.497, KS = 0.171), crabs

(ε = 1.425, KS = 0.121), and bluefish (ε = 1.379, KS = 0.136) impart the largest impact

on the 2000s GSB ecosystem. Note that sorting the stocks by either ε or KS would result

in a different order, owing to the inclusion of a biomass parameter in the KS calculation.

Fishery Impacts

Intense fishing pressures were found for a few GSB stocks within each modeled

period (Figure 2.7). The 1880s system exerted intense fishing pressures on, in

descending order, gadids (F=1.29), menhaden (F=0.90), blue crabs (F=0.47), striped bass

(F=0.42), and sharks (F=0.24). The impact of fishermen on the 1930s ecosystem varied,

in which the largest fishing mortalities were imparted on sharks (F=0.55), menhaden

(F=0.43), blue crab (F=0.29), northern kingfish (F=0.12), and weakfish (0.09). In the

1980s, fishing mortality was high upon the GSB stocks of blackfish (F=1.55), bluefish

(F=1.05), summer flounder (F=0.68), weakfish (F=0.63), and striped bass (F=0.44).

Interestingly, the exploited stocks of fishermen in the 2000s was similar to that of the

1980s, in which blackfish (F=1.21), bluefish (F=1.04), winter flounder (F=0.97), summer

flounder (F=0.86), and weakfish (F=0.69) experienced high levels of fishing pressure.

These fluctuations represent the consistent transition in fishing effort across GSB stocks in response to variations in relative stock abundances.

Ecosystem Efficiency

The ecopath models revealed the low efficiency of the GSB system in utilizing available primary production. The primary production required by GSB consumers measured 2016, 877, 826, and 488 t/km^2/yr for the 1880s, 1930s, 1980s, and 2000s models respectively (Table 2.6), as computed by the ecopath software. Lively et al. (1983) provided an independent production estimate, calculating a phytoplankton productivity measure of 4050 t/km^2/yr (Appendix 2.1) for the 1980s. Considering a phytoplankton production estimate provided by Lonsdale et al. (1996) proved higher than the Lively measure, even less of the available production in GSB is being utilized by consumers in the current system. Measures of PP/B and PP/RESP (Table 2.7) further support the reduction in GSB's efficiency to incorporate production into useable biomass that flows throughout the ecosystem.

Shifts in Trophic Assemblages

The relative abundance of migratory stocks (Figure 2.13) and groundfish (Figure 2.14) both declined, indicating each grouping has comprised a smaller portion of the total ecosystem over the modeled periods.

Discussion

The concurrent reduction in ecosystem size and system maturity suggests GSB is highly impacted. Despite the innate resilience of lagoonal systems to perturbations (Day et al. 1989), GSB has consistently changed under a multitude of stressors. Over the last 100 years, the bay has experienced intense fishing, large scale physical alterations, phosphorus loading, massive eelgrass die offs, and changes in keystone species. The food web has shifted from a system dominated by oysters, menhaden, and a diverse array of piscivorous fish to one dominated by lower trophic species with reduced production from migratory and groundfish stocks. Further, following boom-and-bust fisheries and large-scale physical changes, GSB is presently without a dominant filter feeder for the first time in at least 200 years.

As a system matures, its complexity and stability are thought to increase, leading to a more resilient ecosystem (Odum 1971, many others, but see Pimm 1984). The one hundred year history of GSB indicates the system has responded to multiple stressors by decreased maturity and, as a result, less stability and resilience. The drop in overall size also coincides with declines in the proportion of system biomass that is exploitable to consumers (B/TST, B / PP+RESP). Reductions in the various measures of exergy (EX and EX_{st}) suggest GSB is capable of supporting a greater diversity of energy flows and food web complexity. However, the GSB ecosystem has become increasingly stressed over the last 120 years, seen in various overhead measures (O, Oex, and Redund.). Decades of phosphorus loading, exploitation, and habitat destruction or alteration or some combination of these and unknown factors may be hindering the full potential of the ecosystem and maintaining an undeveloped state. Determining which stressors had

the most influence on the structure of GSB is difficult. However, intense fishing and large scale physical changes appear to have played important roles in structuring the GSB ecosystem. Here I will discuss various model outputs in relation to changes in GSB over the last 120 years including the following: eutrophication, physical stresses, eelgrass deterioration, phosphorus loading, and fishing.

The GSB ecosystem utilized a smaller portion of available resources with each new model, seen in PP/RESP, PP/B, and PPR/PP. Reduced capabilities of system consumers suggest GSB is becoming more eutrophic. Measures of ecosystem efficiency support eutrophication as the models show a reduction in the utilization of production with time and current consumption the lowest in the time series. A drop in consumer abundance coinciding with consistently high inputs of nutrients (Adamson 1982, Nixon et al. 1994) would explain the observed reduction in ecosystem efficiency. In this, the eutrophication of GSB may be a corollary to overfishing (Jackson et al. 2001). Similarly, a growing percentage of system production tied up in inedible phytoplankton stocks also helps explain this trend.

Numerous alterations have occurred to the chain of barrier islands along the south shore of Long Island (Van Popering and Glancy 1947, Greene 1982, Conley 1999), changing the input of nitrogenous nutrients (Nixon et al. 1994), salinity, and circulation patterns in GSB. Similarly, the derived models predict food webs with unique community compositions, likely in response to the new physical regimes. Moriches Inlet has closed and reopened twice over the last 120 years, influencing the composition of the bay's shellfish stocks. The hard clam stock increased by 8.92 t/km^2/yr between the 1880s and 1930s models as higher salinities were observed following the inlet's reopening in

1931. Conversely, the biomass of the oyster decreased under the new conditions, dropping to 21.87 t/km^2/yr. The relatively low fishing mortality on GSB oysters calculated for the 1930s model suggests their decrease, and eventual extirpation, was likely influenced by the change in GSB's salinity regime.

In addition to the alterations in GSB's physical conditions, fluctuations in eelgrass coverage have been observed to influence the abundance of various finfish (Phillips and McRoy 1980, Orth et al. 1984, and Deegan and Buchsbaum 2005), shellfish (Nelson 1924, Marshall 1947, and Pohle et al. 1991), and developing larvae (Bell et al. 1987). Attributed to extreme winter and summer temperatures, an outbreak of wasting disease reduced benthic eelgrass coverage along the entire eastern US coast in the 1930s (Short et al. 1987, Carpenter and Brinkhuis 1991). Extensive commercial trawling conducted in GSB during this period (Dickinson 1938) further degraded the coverage of benthic seagrass (Dorsey and Pederson 1998, Auster and Langton 1999). Therefore, epifaunal stocks that depend on the habitat provided by eelgrass, such as shrimp, (Orth et al. 1984) declined in the 1930s model. Despite a rebound in eelgrass by the 1980s (Carpenter and Brinkhuis 1991), susceptibility to eutrophication and light limitation (Cosper et al. 1987, Deegan and Buchsbaum 2005) has likely reduced the current eelgrass standing stock. Largely dependent on crustacean prey (Steimle et al. 2000, Buckel and McKown 2002, and Nemerson and Able 2003), the recent increase in striped bass coupled with a potential reduction in predation cover (Orth et al. 1984) may be instigating the recent decline in shrimp predicted by the models.

Anthropogenic inputs from duck and agricultural farms have lead to a reduction in system nitrogen to phosphorus ratio (N:P) and a corresponding increase in small-form

algal stocks (Ryther 1954, Ryther et al. 1957, Pederson and Borum 1996) that may experience lower grazing pressures than other phytoplankton (Gobler et al. 2002, Caron et al. 2004). Blooms of *Rhodomonas* sp., *Phaeocystis pouchetti*, and various HABs (*Nannochloris* sp. and *Aureococcus anophagefferens*) have dominated the standing stock of GSB producers in various studies (Ryther et al. 1957, Weaver and Hirshfield 1976, Cassin 1978, Lively et al. 1983). Larger producers, such as diatoms *Thallasiosira pseudonana*, *Rhizoselenia deliculata*, and *Skeletonema costatum* (Ryther et al. 1957, Lively et al. 1983), require a larger N:P than small-form phytoplankton. A negative correlation between green HABs and the 1930s oyster stock observed in Carpenter and Brinkhuis (1991) was reflected in the disappearance of these shellfish by the 1980s. Further, the drastic reduction in fishing mortality between the 1980s and 2000s hard clam stock has not been met with a similar increase in biomass, suggesting brown HABs are suppressing a potential rebound in GSB hard clams (Greenfield and Lonsdale 2002, Gobler et al. 2005). Similar suppression has been noted in bay scallops and blue mussels (Cosper et al. 1987), explaining the modeled decline in suspension feeders. A final indication of these blooms may be observed in the reduced utilization of system productivity predicted by the models. Driven by an influx of excess nutrients into GSB, blooms of small-form inedible algae may account for a growing percentage of unconsumed system production.

Recent stock assessments for sharks (Cortés et al. 2002), weakfish (NEFSC 2009), winter flounder (Terceiro 2008), striped bass (NEFSC 2008), and summer flounder (Terceiro 2003) indicate the fishing mortalities predicted by the models are unsustainable. The models, however, show declines in only some of these stocks. Sand

tiger sharks, which once used the bay as a nursery (Thorne 1916, 1928), are now

extirpated. Similarly, little evidence exists for the presence of inshore winter flounder

(Sagarese 2009), termed "bay fish" (Lobell 1938), which supported a fishery in the 1930s

(Dickinson 1938).

Constant transition in targeted GSB fishery stocks resulted in variations to the

internal pressures that shape trophic structure. Recorded in the measures of

keystoneness, the extirpation of GSB sharks has essentially removed an apex pressure

that proved influential in the 1880s and 1930s models. Intense fishing pressure on winter

flounder and weakfish (Dickinson 1938, Lobell 1938, Nesbit 1938) reduced much of their

influence in the 1930s. Further, disappearance of inshore menhaden under the industrial

fishery (McHugh 1972) reduced the importance of this once abundant prey on dynamics

of the GSB ecosystem. Striped bass and summer flounder, conversely, have had a

growing impact on system dynamics as their biomass has grown.

The fluctuations in various GSB predators and their influence on the system have

altered ecosystem structure, with trends showing a steady decline in food-web

complexity. The highest trophic levels in current models are occupied by finfish that are

seasonal visitors to the bay, such as recently abundant summer flounder and striped bass,

compared to large sharks which provided a higher trophic level consumer to the GSB

ecosystem. Consecutive drops in R/B and $(P/B)^{-1}$ indicate top-level consumers contribute

less to recent than earlier model systems. As predators decline, fewer omnivorous

consumers are found within the system; as such, the degree of connectivity between

trophic groupings and the complexity of trophic flow has dropped, reflected in system

connectivity (C and SOI) and path length (PL and SPL), respectively. Declines in A and

A/C reflect a further decline in system complexity as fewer potential pathways exist for biomass to flow when strength of relationships between predators and their various prey is reduced.

Conclusion

Although the synchrony of various stock fluctuations with multiple system stressors makes the separation of cause-and-effect difficult, alterations to the system are clear. GSB has experienced consistent reductions in ecosystem size, seen in measures of RESP and two indices of biomass (TST and total system biomass). While measures of ecosystem productivity independent of this study reveal a highly productive ecosystem, the low calculations of PPR/PP, PP/RESP, and PP/B indicate system consumers are able to utilize less available system production. Consistent declines in various indices of trophic complexity further indicate a trend towards simplifying the trophic structure of the ecosystem. As a reduction in any of these indices is sufficient to cause major alterations to an ecosystem, the concurrent decline in these measures may represent regime shifts that have occurred in GSB over the last 120 years. Historically, the GSB ecosystem supported an immensely productive exploitation of abundant marine resources. The current system exhibits different dynamics as the application of numerous stressors has consistently reset GSB's ecosystem structure, impacting both the system's maturity and stability. With such radical alterations, attempts to return the system to

previous states may be difficult as the current system structure may be too far removed to return to historical baselines.

Table 2.1. Sources of length:weight conversion parameters for Ecopath Functional Groups.

Ecopath Grouping	L:W Conversion
Benthic Fauna	
Macrobenthos	
Sand Shrimp	Pihl and Rosenberg 1984
Shrimp	
Mantis Shrimp	
Squid	Belcari 1996
Forage Fish	Dawson 1965; Wilk et al. 1978; Jessop 1983; Froese & Pauly 2005
Suspension Feeders	
Hard Clams	
Tropical Fish	Wilk et al. 1978; Duarte et al. 1999; Bohnsack & Harper 1988; Froese & Pauly 2005
Crabs	
Blue Crab	Lipcius & Stockhausen 2002
Horseshoe Crabs	Graham 2007
Spider Crabs	Guerao & Rotllant 2009
Skate	Waring 1984; Froese & Pauly 2005
American Eel	Wilk et al. 1978
Atlantic Menhaden	Cooper 1965
Black Sea Bass	Bohnsack & Harper 1988
Blackfish	Hostetter & Munroe 1993
Bluefish	Barger 1990
Butterfish	Wilk et al. 1978
Cunner	
Dogfish	van der Elst 1981
Drums & Croakers	Beckman et al. 1990
Flatfish-Other	Koski 1978
Gadids	Froese & Pauly 2005
Northern Kingfish	Schaefer 1965
Northern Pipefish	
Oyster Toadfish	Schwartz & Dutcher 1963
Scup	Crawford 1993
Striped Bass	
Searobins	Crawford 1993
Summer Flounder	Henderson 1979
Weakfish	Shepherd & Grimes 1983
Windowpane Flounder	
Winter Flounder	Crawford 1993
Zooplankton	
Phytoplankton	
Ctenophores	
Detritus	

Table 2.2. Final Parameters (Biomass, P/B, Q/B, EE, P/Q) for the 1880s Model.

Group Name	1880 Model					
	Trophic Lvl	Biomass	P/B	Q/B	EE	P/Q
Benthic Fauna	2.00	19.78	4.58	24.36	0.859	0.188
Macrobenthos	3.14	1.71	1.00	1.67	0.804	0.600
Sand Shrimp	2.08	5.83	3.75	24.90	0.999	0.151
Shrimp	2.34	10.06	3.30	16.10	0.860	0.205
Mantis Shrimp	3.01	0.14	1.34	7.43	0.950	0.180
Squid	2.25	0.22	0.80	7.00	0.990	0.115
Forage Fish	3.08	7.00	1.29	15.00	0.999	0.086
Suspension Feeders	2.09	109.90	0.30	2.17	0.931	0.138
Hard Clams	2.09	4.46	0.57	5.10	0.999	0.111
Oysters	2.09	38.38	0.20	2.02	0.999	0.099
Tropical Fish	3.11	0.14	1.37	7.97	0.999	0.172
Crabs	2.92	8.79	1.38	8.50	0.963	0.162
Blue Crab	3.19	0.67	1.40	4.00	0.950	0.350
Horseshoe Crabs	2.97	0.00	0.60	3.00	0.500	0.200
Spider Crabs	2.55	0.09	2.30	15.18	0.950	0.150
Skate	3.54	0.16	0.15	4.10	0.983	0.037
Sharks	4.07	0.21	0.56	4.00	0.431	0.141
American Eel	2.01	3.45	0.22	3.70	0.500	0.059
Menhaden	2.06	31.40	1.10	28.00	0.990	0.039
Black Sea Bass	3.17	0.03	0.60	6.90	0.614	0.087
Blackfish	2.97	0.00	0.03	3.10	0.000	0.009
Bluefish	3.61	4.88	0.10	15.40	0.821	0.007
Butterfish	2.58	0.11	0.80	5.50	0.040	0.145
Cunner	3.12	0.52	0.00	7.50	0.990	0.000
Dogfish	3.50	0.10	0.05	4.77	0.000	0.010
Drums & Croakers	3.04	2.05	0.40	3.90	0.497	0.103
Flatfish-Other	3.25	0.05	0.43	7.03	0.950	0.061
Gadids	3.48	0.25	1.35	3.66	0.999	0.370
Northern Kingfish	3.76	0.00	0.30	5.90	0.414	0.051
Northern Pipefish	3.27	0.21	1.30	6.90	0.999	0.188
Oyster Toadfish	3.65	0.60	0.36	6.20	0.000	0.058
Scup	3.08	0.09	0.61	5.50	0.999	0.110
Striped Bass	3.69	0.11	0.48	2.45	0.900	0.194
Sea Robins	3.29	0.12	0.43	7.15	0.986	0.060
Summer Flounder	3.63	2.87	0.07	4.00	0.161	0.018
Weakfish	3.37	5.76	0.09	4.10	0.999	0.022
Windowpane Flounder	3.22	0.01	0.47	6.20	0.383	0.076
Winter Flounder	2.84	18.18	0.52	6.30	0.113	0.082
Zooplankton	2.00	7.20	25.00	90.91	0.900	0.275
Phytoplankton	1.00	4050.00	60.00		0.006	
Ctenophores	2.50	0.81	8.80	35.20	0.900	0.250
Detritus	1.00	1.00			0.003	

Table 2.3. Final Parameters (Biomass, P/B, Q/B, EE, P/Q) for the 1930s Model.

| Group Name | 1930 Model | | | | |
	Trophic Lvl	Biomass	P/B	Q/B	EE	P/Q
Benthic Fauna	2.00	13.74	4.58	24.36	0.859	0.188
Macrobenthos	3.14	1.44	1.00	1.67	0.804	0.600
Sand Shrimp	2.08	1.57	3.75	24.90	0.999	0.151
Shrimp	2.34	1.87	3.30	16.10	0.860	0.205
Mantis Shrimp	3.01	0.04	1.34	7.43	0.950	0.180
Squid	2.24	0.04	0.81	7.00	0.990	0.115
Forage Fish	3.08	1.09	1.29	15.00	0.999	0.086
Suspension Feeders	2.09	136.53	0.30	2.17	0.931	0.138
Hard Clams	2.09	13.38	0.73	5.10	0.999	0.144
Oysters	2.09	16.51	0.20	2.02	0.999	0.099
Tropical Fish	3.11	0.06	1.37	7.97	0.999	0.172
Crabs	2.94	14.45	1.38	8.50	0.963	0.162
Blue Crab	3.19	0.25	1.40	4.00	0.950	0.350
Horseshoe Crabs	2.97	0.00	0.60	3.00	0.500	0.200
Spider Crabs	2.55	0.13	2.30	15.18	0.950	0.150
Skate	3.54	0.02	0.15	4.10	0.983	0.037
Sharks	4.07	0.03	0.56	4.00	0.968	0.141
American Eel	2.01	0.24	0.22	3.70	0.500	0.059
Menhaden	2.06	0.87	1.10	28.00	0.990	0.039
Black Sea Bass	3.17	0.01	0.60	6.90	0.614	0.087
Blackfish	2.97	0.00	0.03	3.10	1.000	0.009
Bluefish	3.61	0.46	0.05	15.40	0.999	0.003
Butterfish	2.58	0.26	0.80	5.50	0.035	0.145
Cunner	3.12	0.01	0.00	7.50	0.990	0.000
Dogfish	3.51	0.10	0.05	4.77	0.000	0.010
Drums & Croakers	3.04	0.97	0.40	3.90	0.497	0.103
Flatfish-Other	3.25	0.01	0.43	7.03	0.950	0.061
Gadids	3.16	0.00	1.35	3.66	0.999	0.370
Northern Kingfish	3.77	0.32	0.30	5.90	0.414	0.051
Northern Pipefish	3.27	0.05	1.30	6.90	0.999	0.188
Oyster Toadfish	3.80	0.60	0.36	6.20	0.000	0.058
Scup	3.08	0.01	0.61	5.50	0.999	0.110
Striped Bass	3.69	0.07	0.04	2.45	0.900	0.015
Sea Robins	3.30	0.02	0.43	7.15	0.986	0.060
Summer Flounder	3.63	0.88	0.40	4.00	0.161	0.100
Weakfish	3.37	2.88	0.09	4.10	0.999	0.023
Windowpane Flounder	3.22	0.00	0.47	6.20	0.369	0.076
Winter Flounder	2.85	9.42	0.78	6.30	0.113	0.124
Zooplankton	2.00	2.49	25.00	90.91	0.900	0.275
Phytoplankton	1.00	4050.00	60.00		0.003	
Ctenophores	2.50	0.13	8.80	35.20	0.900	0.250
Detritus	1.00	1.00			0.002	

Table 2.4. Final Parameters (Biomass, P/B, Q/B, EE, P/Q) for the 1980s Model.

Group Name	Trophic Lvl	Biomass	P/B	Q/B	EE	P/Q
Benthic Fauna	2.00	12.68	4.58	24.36	0.859	0.188
Macrobenthos	3.14	1.27	1.00	1.67	0.804	0.600
Sand Shrimp	2.08	2.35	3.75	24.90	0.999	0.151
Shrimp	2.34	3.59	3.30	16.10	0.860	0.205
Mantis Shrimp	3.01	0.04	1.34	7.43	0.950	0.180
Squid	2.24	0.01	1.74	7.00	0.990	0.249
Forage Fish	3.08	2.46	1.29	15.00	0.999	0.086
Suspension Feeders	2.09	109.02	0.30	2.17	0.931	0.138
Hard Clams	2.09	4.95	2.04	5.10	0.999	0.400
Tropical Fish	3.11	0.64	1.37	7.97	0.999	0.172
Crabs	2.94	10.87	1.38	8.50	0.963	0.162
Blue Crab	3.19	0.20	1.40	4.00	1.000	0.350
Horseshoe Crabs	2.97	0.04	0.60	3.00	0.500	0.200
Spider Crabs	2.55	0.11	2.30	15.18	0.950	0.150
Skate	3.54	0.43	0.15	4.10	0.983	0.037
American Eel	2.01	0.86	0.22	3.70	0.500	0.059
Menhaden	2.06	2.25	1.10	28.00	0.990	0.039
Black Sea Bass	3.17	0.01	0.60	6.90	0.614	0.087
Blackfish	2.97	1.50	1.55	3.10	1.000	0.500
Bluefish	3.61	1.90	1.07	15.40	0.999	0.070
Butterfish	2.58	0.01	0.80	5.50	0.142	0.145
Cunner	3.12	0.02	0.00	7.50	0.990	0.000
Dogfish	3.51	0.01	0.05	4.77	0.000	0.010
Drums & Croakers	3.04	0.21	0.40	3.90	0.497	0.103
Flatfish-Other	3.25	0.03	0.43	7.03	0.950	0.061
Gadids	3.16	0.00	2.29	3.66	0.999	0.626
Northern Kingfish	3.77	0.04	0.30	5.90	0.248	0.051
Northern Pipefish	3.27	0.06	1.30	6.90	0.999	0.188
Oyster Toadfish	3.80	0.44	0.36	6.20	0.039	0.058
Scup	3.08	0.05	0.61	5.50	0.999	0.110
Striped Bass	3.68	1.45	0.49	2.45	0.900	0.200
Sea Robins	3.30	0.12	0.43	7.15	0.986	0.060
Summer Flounder	3.53	1.02	0.88	4.00	0.879	0.220
Weakfish	3.38	0.17	0.73	4.10	0.999	0.179
Windowpane Flounder	3.22	0.13	0.47	6.20	0.369	0.076
Winter Flounder	2.85	8.72	0.72	6.30	0.593	0.114
Zooplankton	2.00	2.82	25.00	90.91	0.900	0.275
Phytoplankton	1.00	4050.00	60.00		0.002	
Ctenophores	2.50	0.28	8.80	35.20	0.900	0.250
Detritus	1.00	1.00			0.002	

Table 2.5. Final Parameters (Biomass, P/B, Q/B, EE, P/Q) for the 2000s Model.

Group Name	Trophic Lvl	Biomass	P/B	Q/B	EE	P/Q
		2000 Model				
Benthic Fauna	2.00	7.54	4.58	24.36	0.859	0.188
Macrobenthos	3.14	0.57	1.00	1.67	0.804	0.600
Sand Shrimp	2.08	1.58	3.75	24.90	0.999	0.151
Shrimp	2.34	2.19	3.30	16.10	0.860	0.205
Mantis Shrimp	3.01	0.22	1.34	7.43	0.950	0.180
Squid	2.29	0.05	2.74	7.00	0.990	0.392
Forage Fish	3.08	1.30	1.29	15.00	0.999	0.086
Suspension Feeders	2.09	68.25	0.30	2.18	0.931	0.138
Hard Clams	2.09	1.78	3.73	5.10	0.999	0.731
Tropical Fish	3.11	0.38	1.27	7.97	0.999	0.159
Crabs	2.95	7.61	1.38	8.50	0.963	0.162
Blue Crab	3.20	0.43	1.40	4.00	0.950	0.350
Horseshoe Crabs	2.97	0.06	0.60	3.00	0.500	0.200
Spider Crabs	2.55	0.10	2.30	15.30	0.950	0.150
Skate	3.55	0.82	0.15	4.10	0.983	0.037
American Eel	2.01	0.27	0.22	3.70	0.500	0.059
Menhaden	2.06	1.02	1.10	28.00	0.990	0.039
Black Sea Bass	3.17	0.02	0.30	6.90	0.614	0.043
Blackfish	3.78	0.19	1.21	3.10	1.000	0.391
Bluefish	3.62	0.80	1.06	15.40	0.999	0.069
Butterfish	2.58	0.03	0.80	5.50	0.142	0.145
Cunner	3.12	0.01	0.48	7.50	0.990	0.064
Dogfish	3.52	0.10	0.05	4.77	0.950	0.010
Drums & Croakers	3.04	0.27	0.40	3.90	0.497	0.103
Flatfish-Other	3.28	0.01	0.74	7.03	0.950	0.105
Gadids	3.16	0.01	1.94	3.66	0.999	0.530
Northern Kingfish	3.77	0.07	0.30	5.90	0.538	0.051
Northern Pipefish	3.27	0.01	1.30	6.90	0.999	0.188
Oyster Toadfish	3.49	0.00	0.36	6.20	0.058	0.058
Scup	3.08	0.27	0.60	5.50	0.999	0.110
Striped Bass	3.73	2.30	0.49	2.45	0.900	0.200
Sea Robins	3.30	0.20	0.43	7.15	0.980	0.060
Summer Flounder	3.80	1.79	1.00	4.00	0.950	0.250
Weakfish	3.40	0.11	1.33	4.10	0.999	0.325
Windowpane Flounder	3.22	0.23	0.47	6.20	0.051	0.076
Winter Flounder	2.94	0.24	1.50	6.30	0.999	0.238
Zooplankton	2.00	1.64	25.00	90.91	0.900	0.275
Phytoplankton	1.00	4050.00	60.00		0.001	
Ctenophores	2.50	0.15	8.80	35.20	0.900	0.250
Detritus	1.00	1.00			0.001	

Table 2.6. Summary Statistics for the 4 composed Ecopath Models.

Statistic	1880 Model	1930 Model	1980 Model	2000 Model
Total System Production (ΣP)	243448.40	243226.10	243231.10	243138.00
Net Primary Production (PP)	243000.00	243000.00	243000.00	243000.00
Primary Production Required	2015.86	877.08	825.64	488.23
Total Respiration (RESP)	1927.41	766.06	727.72	416.44
Net System Production	241072.60	242233.90	242272.30	242583.60
Total System Throughput (TST)	489083.00	487172.30	487208.50	486690.70
Total Biomass w/out Detritus (B)	4336.33	4270.93	4220.77	4152.61
Connectance Index	0.20	0.20	0.19	0.19
System Omnivory Index (SOI)	0.33	0.31	0.29	0.32
Total Consumption (ΣQ)	3140.40	1297.94	1265.08	731.75
Total Flows to Detritus	242324.90	242756.80	242799.80	242888.20
Total Exports (EXP)	241690.30	242351.40	242415.80	242654.40
Mean Trophic Level of Catch	2.13	2.82	3.13	3.59
Total Catch	30.61	1.48	10.30	4.46
Gross efficiency (Catch/PP)	0.02	0.00	0.02	0.01

Table 2.7. Maturity Indices used for the 4 composed Ecopath Models.

Maturity Index	Symbol	Trend	1880	1930	1980	2000
Total primary production / total respiration	PP / R	→ 1	126.08	317.21	333.92	583.51
Total primary production / total biomass	PP / B	-	56.04	56.90	57.57	58.52
Biomass supported per unit energy flow	B / TST	+	0.01	0.01	0.01	0.01
Biomass supported per unit energy flow	$\dfrac{B}{(PP + R)}$	+	0.02	0.02	0.02	0.02
Net Community Yield	PP - R	-	241072.59	242233.94	242272.28	242583.56
Connectance	C	+	0.20	0.20	0.19	0.19
System Omnivory Index	SOI	+	0.33	0.31	0.29	0.32
Dominance of Detritus	Dom.Det.	+	0.50	0.50	0.50	0.50
Average Organism Size - $(P/B)^{-1}$	B / P	+	0.02	0.02	0.02	0.02
Finn's Cycling Index	FCI	+	0.04	0.03	0.02	0.01
Predatory Cycling Index	PCI	+	0.18	0.90	0.70	0.86
Nutrient Regeneration	FCI - PCI	+	-0.14	-0.87	-0.68	-0.85
Path Length	PL	+	2.01	2.00	2.00	2.00
Straight-through Path Length	SPL	+	2.01	2.00	2.00	2.00
Residence Time	B / (R+EXP)	+	0.02	0.02	0.02	0.02
Nutrient Conservation	O_{ex}	+	1570.90	626.90	791.90	445.90
System Overhead	O	+	48209.56	22806.56	22199.91	13567.80
Schrodinger Ratio	R / B	+	0.44	0.18	0.17	0.10
Information Content of Flows	I	+	1.04	1.02	1.02	1.01
Energy-based Ascendency	A	+	510408.52	495902.49	496032.22	492066.89
Relative Ascendency	A / C	-	0.91	0.96	0.96	0.97
Internal Redundancy	Redund.	+	37916.00	18424.90	17652.10	10918.90
Exergy	EX	+	11275.23	11105.22	10974.82	10797.65
Structural Exergy	EX_{st}	+	0.04	0.04	0.04	0.03

Figure 2.1. Indices of Ecosystem Size and Structural Properties for GSB: A) System Biomass and TST , B) Net Primary Production and Total Respiratory Flow , and C) Net System Production.
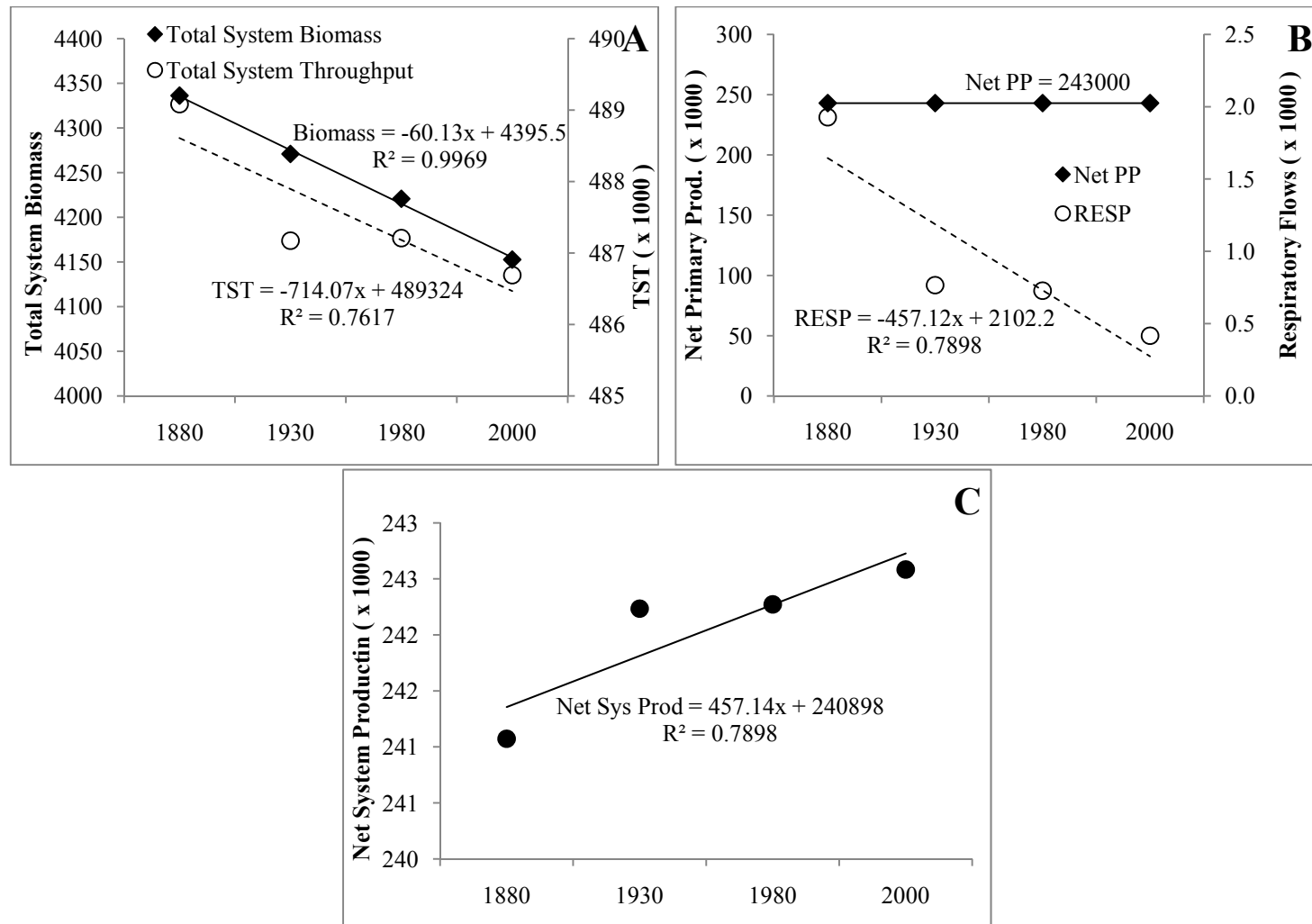
Figure 2.2. Select Indices of Ecosystem Maturity for GSB: A) PP/Respiration , B) Biomass/TST , C) Ascendency/Capacity , and D) System Omnivory Index. Symbols along lines indicate an increase (+) or decrease (-) in ecosystem maturity.
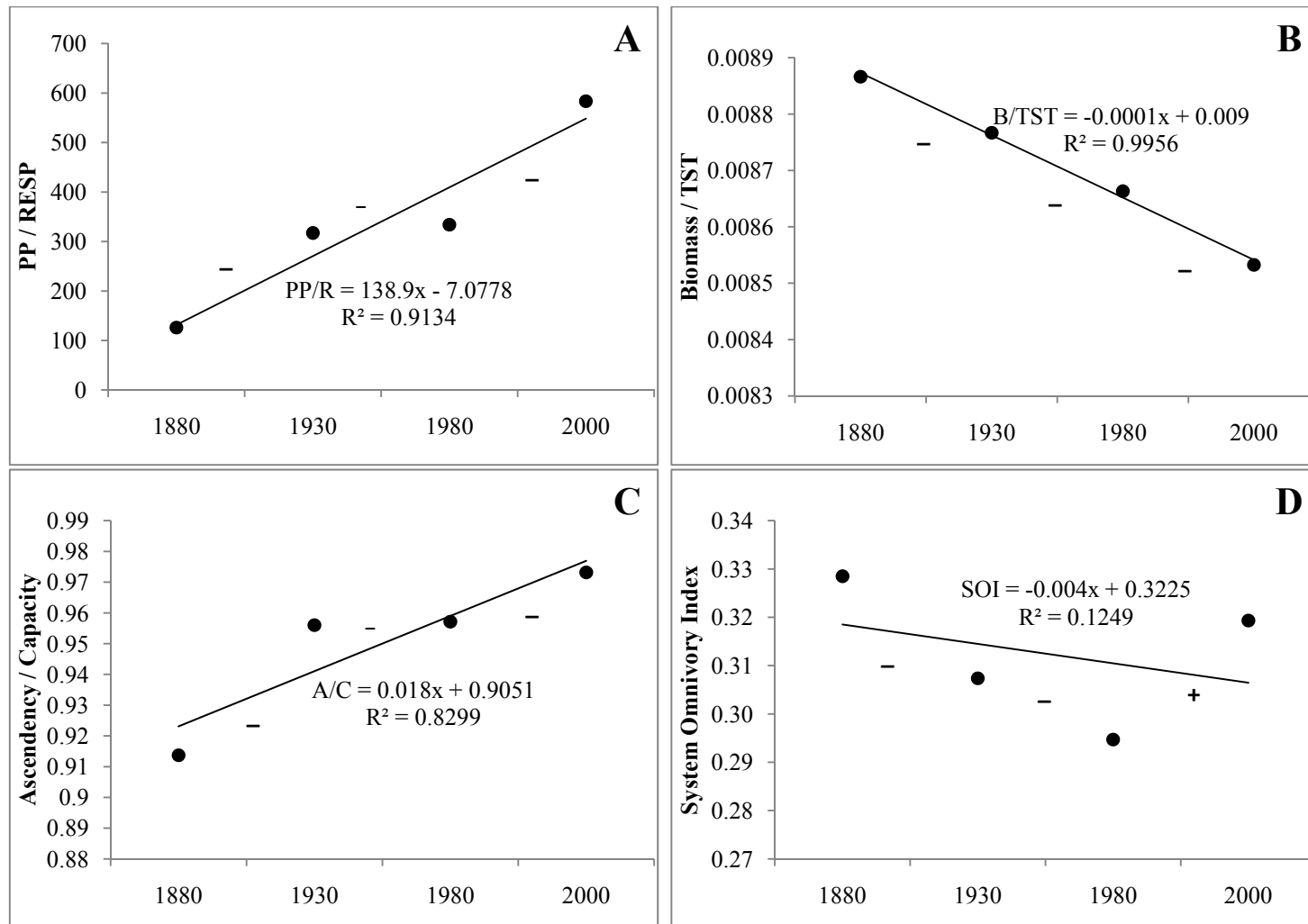
Figure 2.3. Total Ecosystem Impact (ε) vs. Keystoneness for the 1880s GSB Ecosystem. Stocks with high values of ε and KS represent groups that impose large influences on the ecosystem.
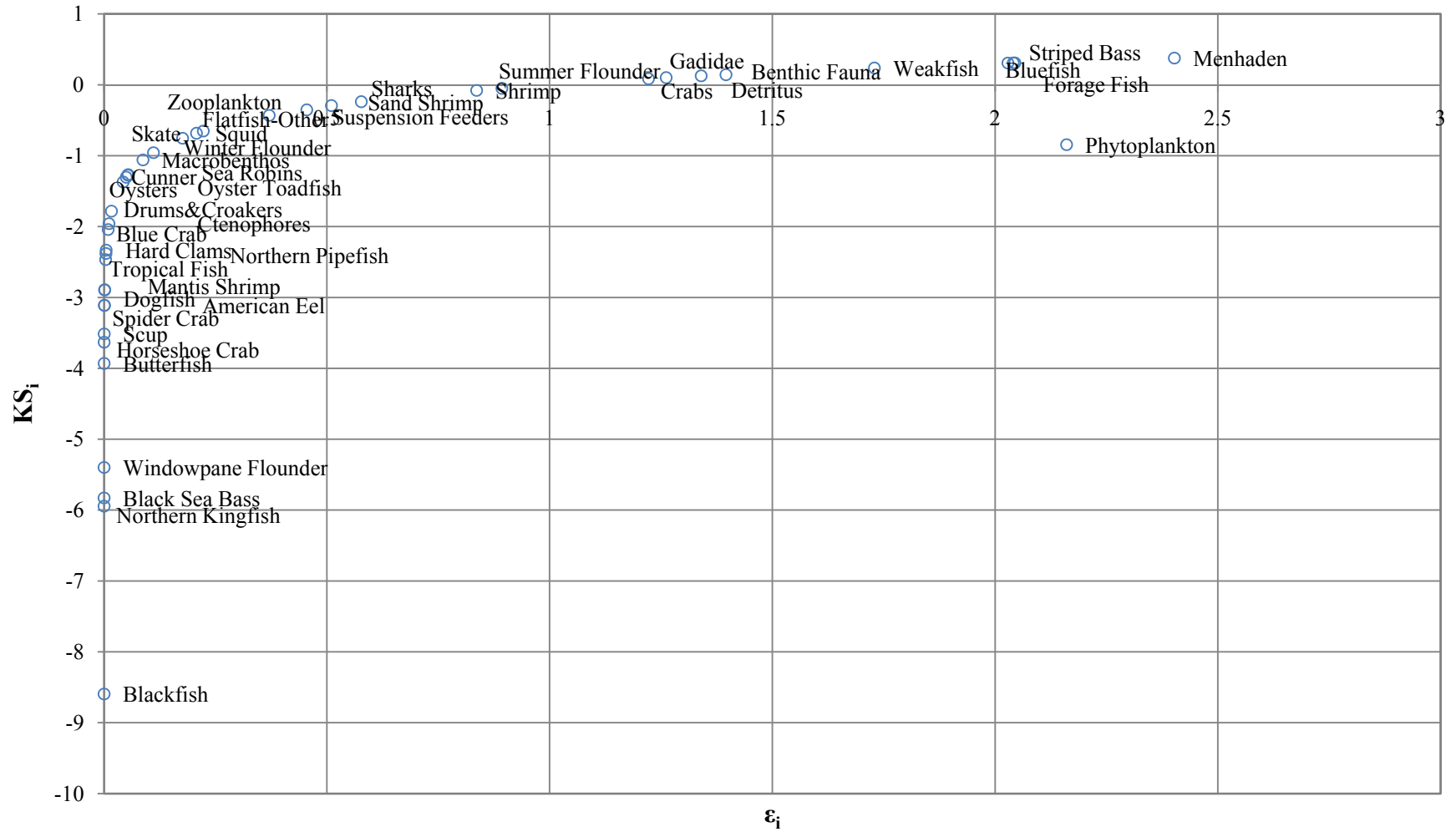
Figure 2.4. Total Ecosystem Impact (ε) vs. Keystoneness for the 1930s GSB Ecosystem. Stocks with high values of ε and KS represent groups that impose large influences on the ecosystem.
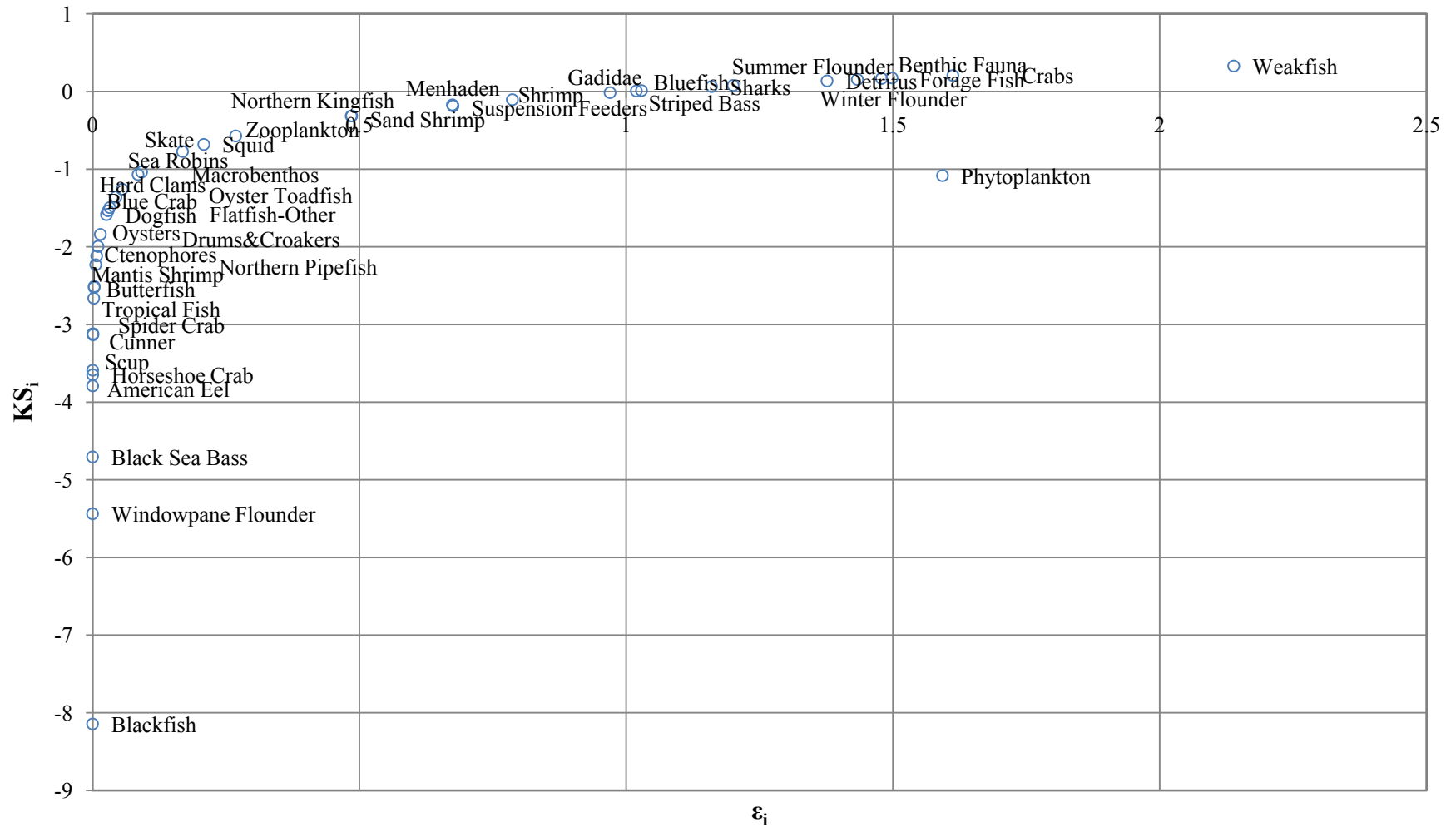
Figure 2.5. Total Ecosystem Impact (ε) vs. Keystoneness for the 1980s GSB Ecosystem. Stocks with high values of ε and KS represent groups that impose large influences on the ecosystem.
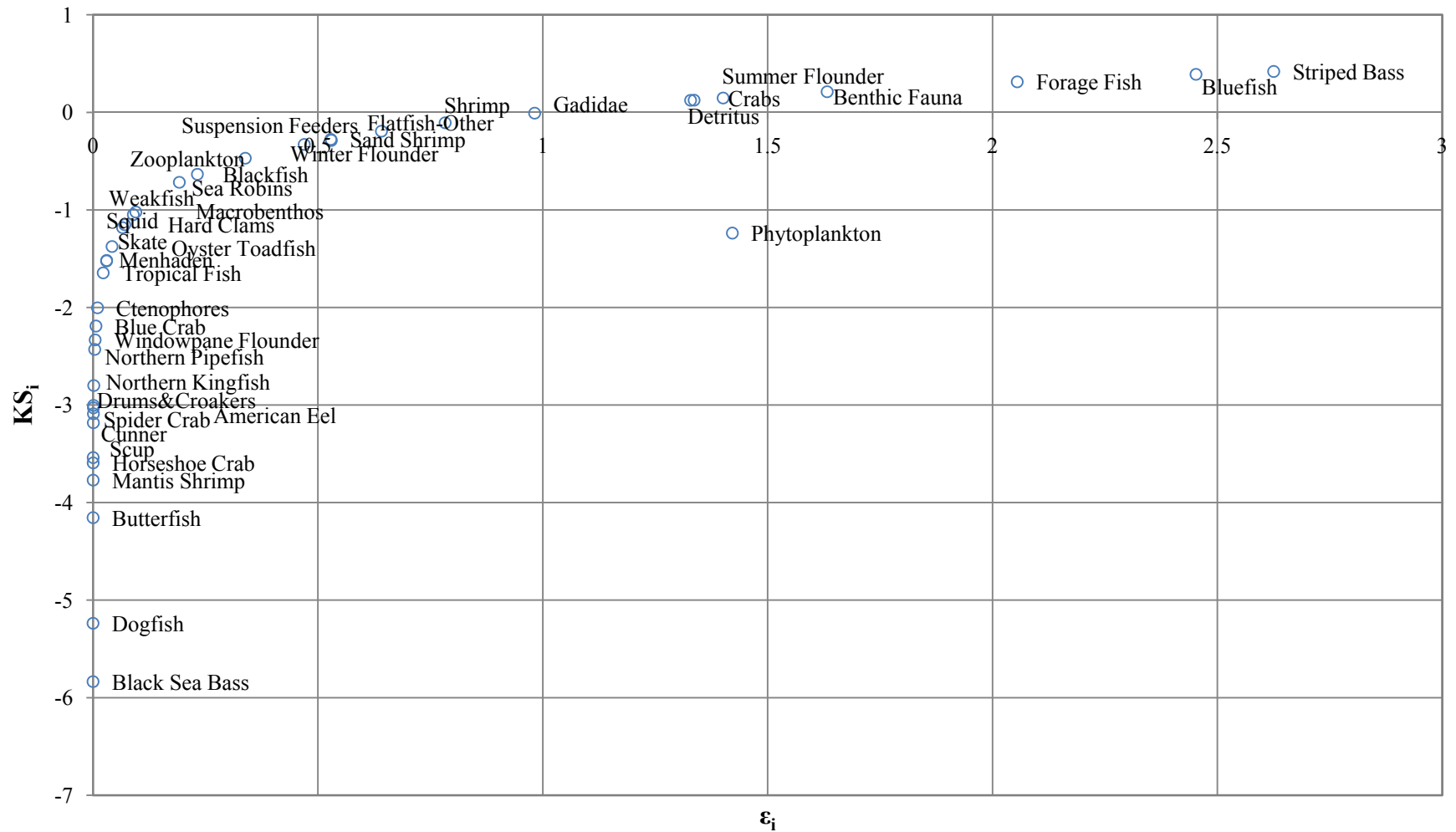
Figure 2.6. Total Ecosystem Impact (ε) vs. Keystoneness for the 2000s GSB Ecosystem. Stocks with high values of ε and KS represent groups that impose large influences on the ecosystem.
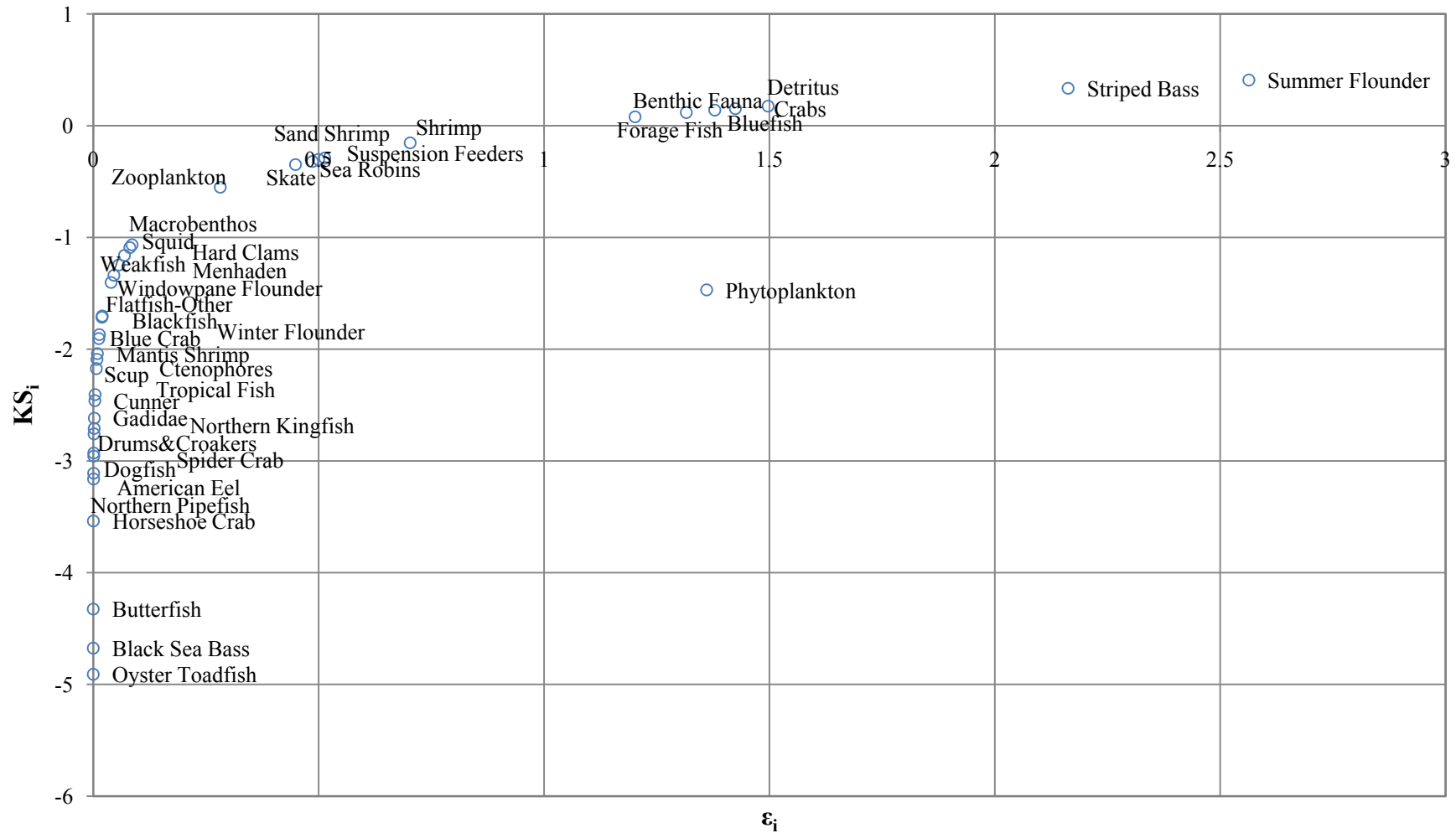
Figure 2.7. Temporal Variations in Fishing Mortality on specific Stocks.
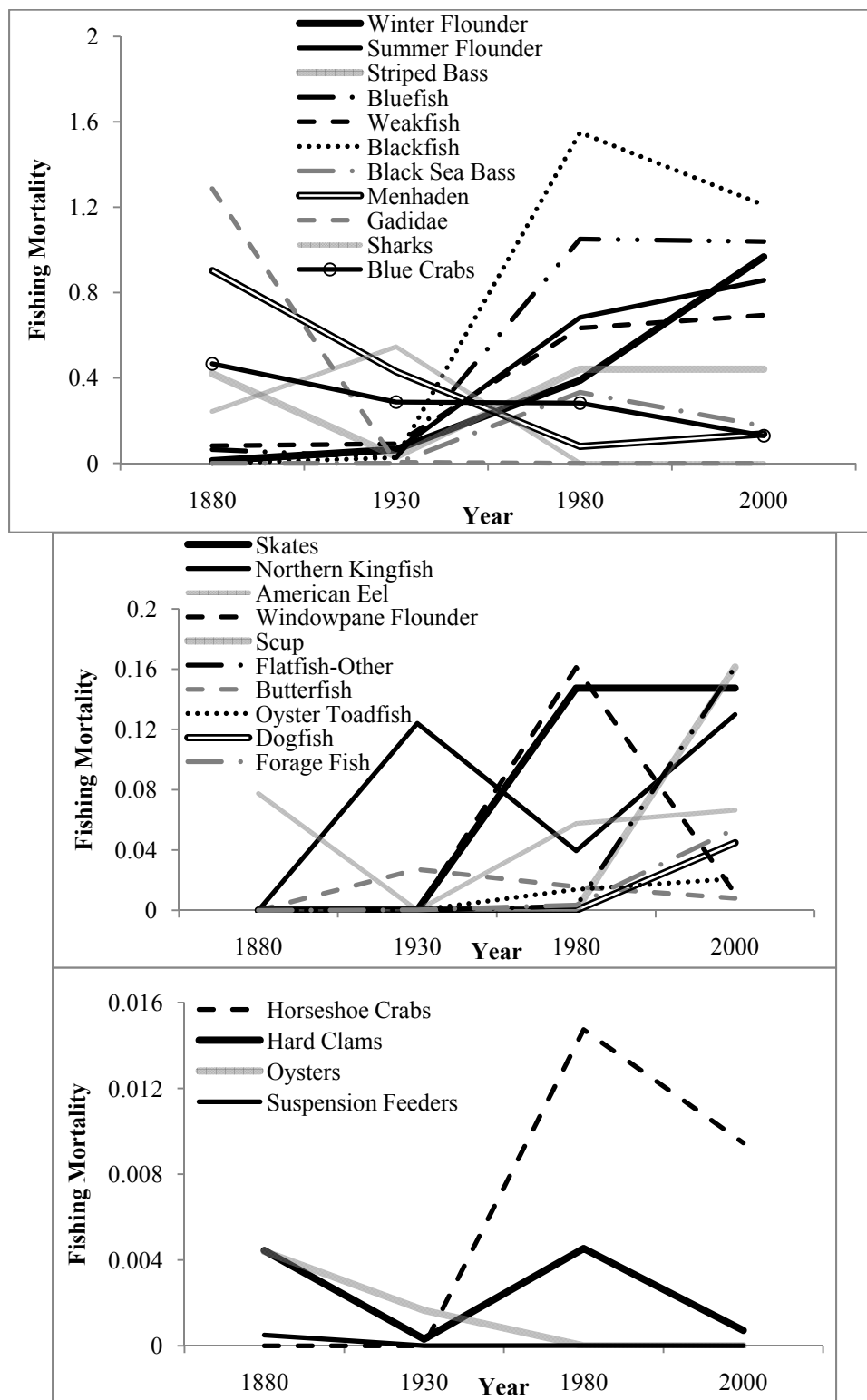
Figure 2.8. Flow Diagram of the 1880s GSB Ecosystem.
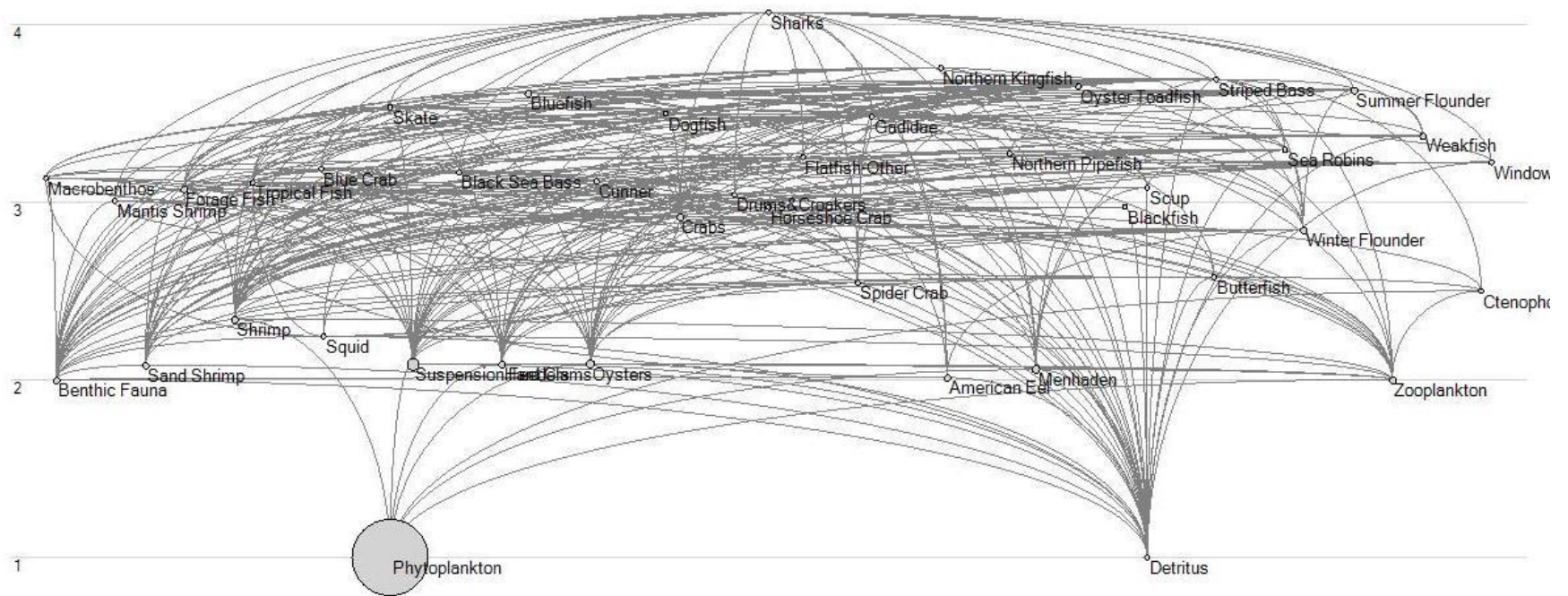
Figure 2.9. Flow Diagram of the 1930s GSB Ecosystem.
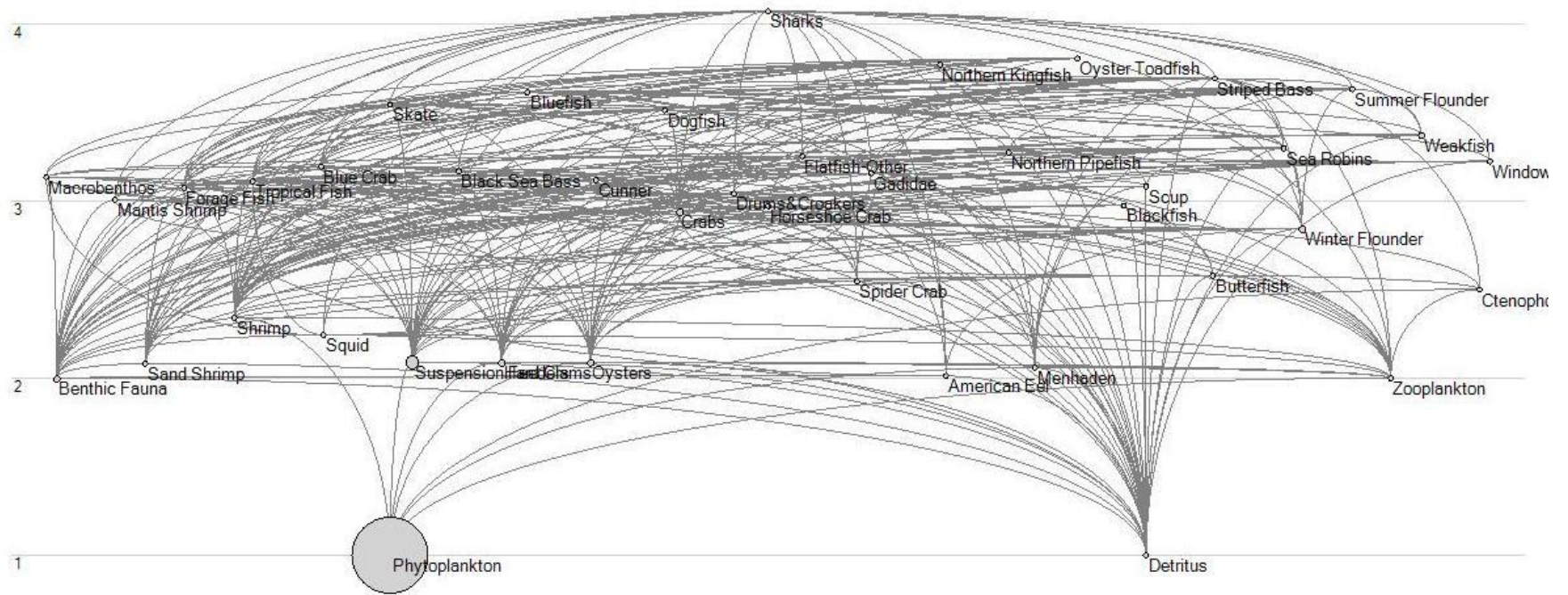
Figure 2.10. Flow Diagram of the 1980s GSB Ecosystem.
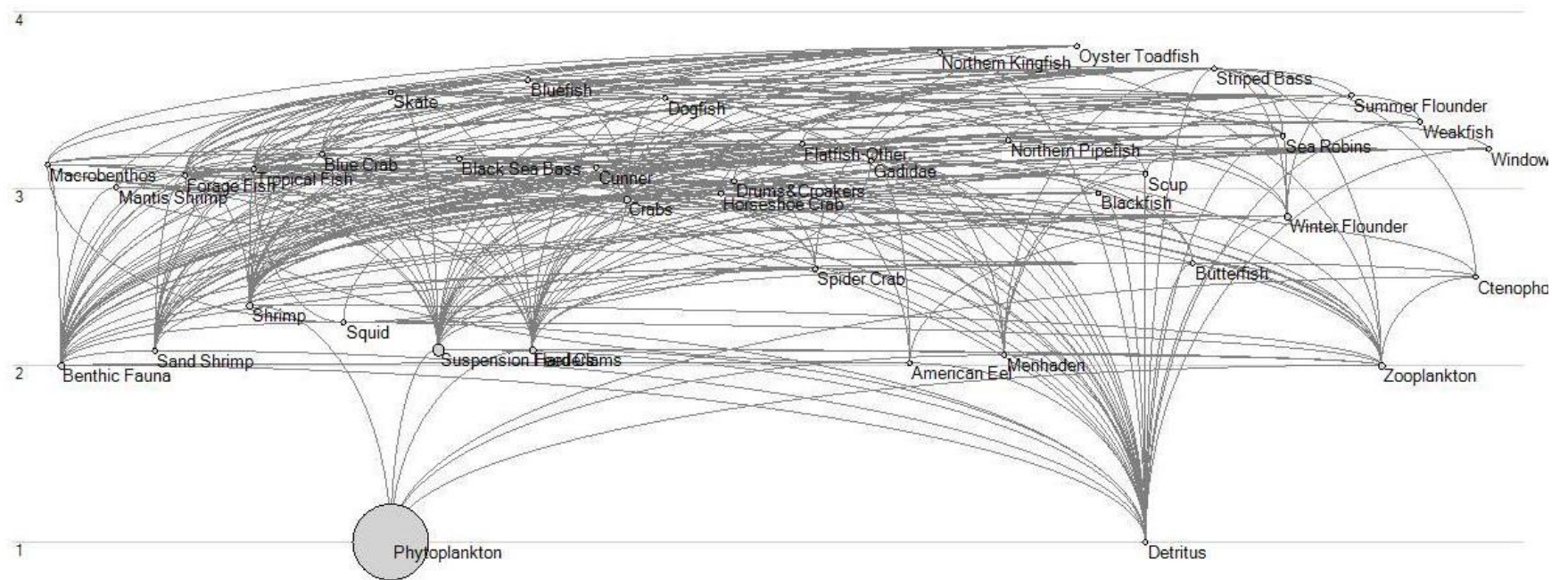
Figure 2.11. Flow Diagram of the 2000s GSB Ecosystem.
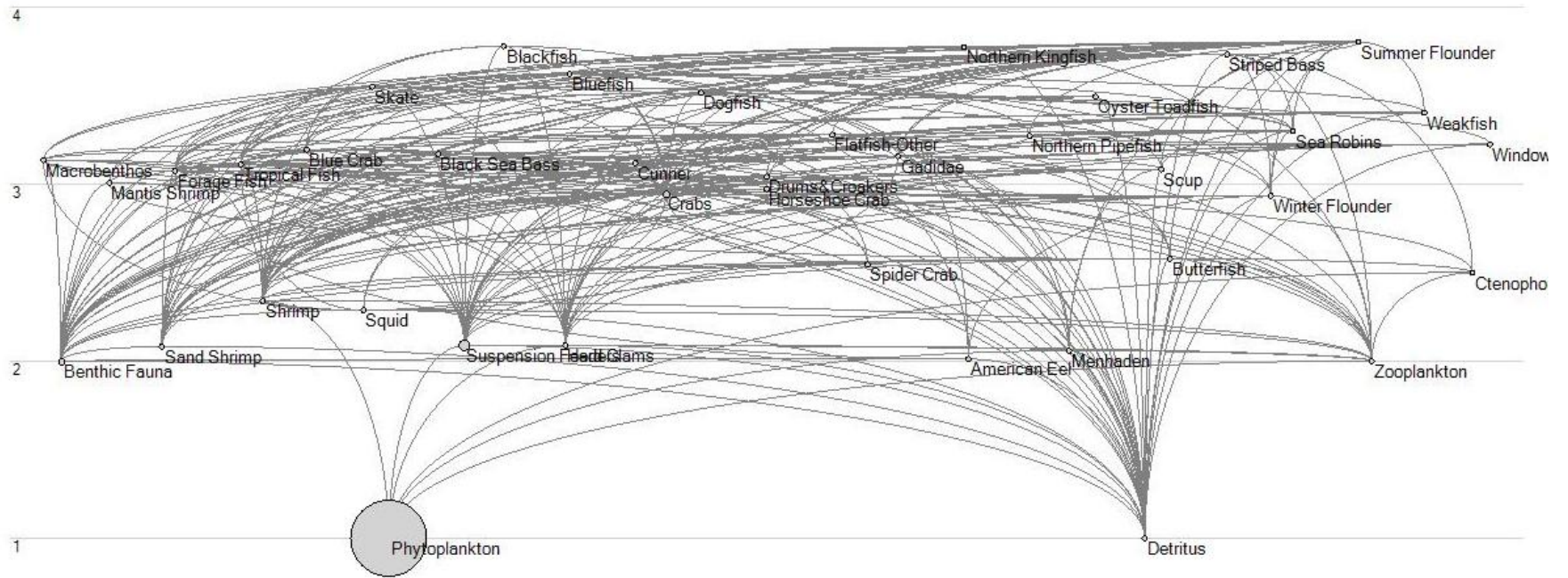
Figure 2.12. Temporal Variations in the relationship between F (fishing mortality) and B (stock biomass) for Hard Clams and Oysters.
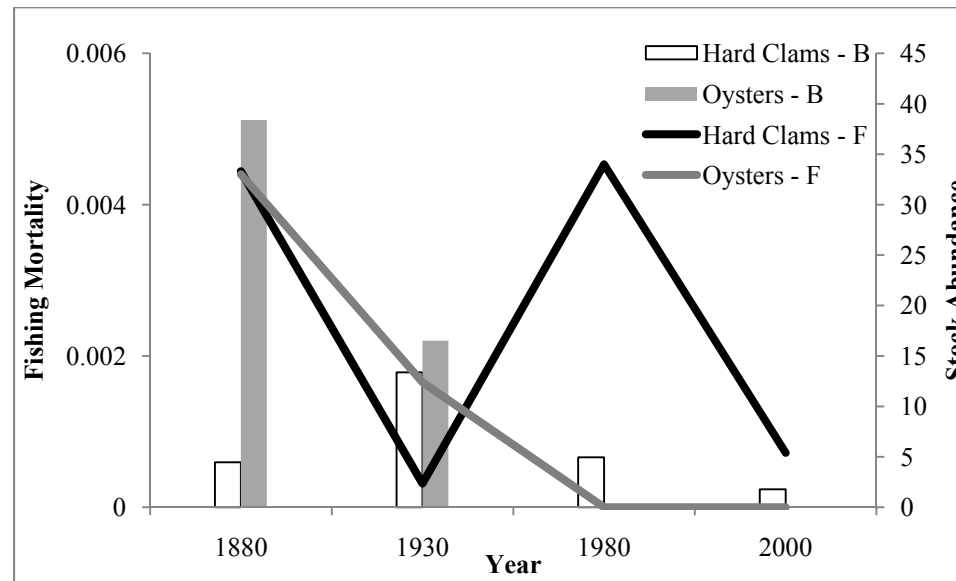
Figure 2.13. Ratio of Biomass of "Migratory" Stocks compared to Total System Biomass. Figure includes stocks that consistently move in and out of the GSB system: black seabass, bluefish, gadids, menhaden, scup, sharks, skates, squid, striped bass, summer flounder, weakfish, winter flounder, and tropical fish.

y = -0.0038x + 0.0156
R² = 0.6962
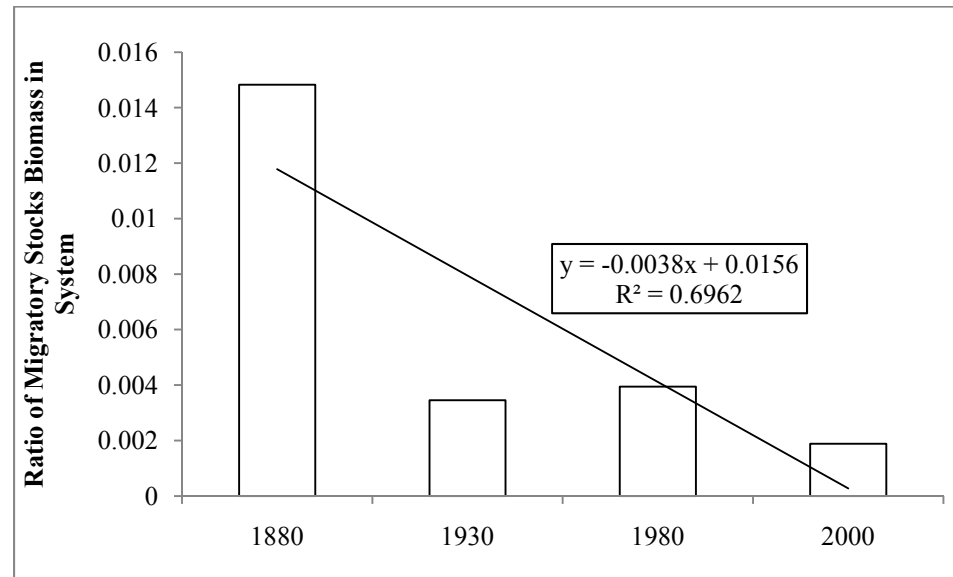
Figure 2.14. Ratio of Biomass of "Groundfish" Stocks compared to Total System Biomass. Figure includes stocks that live on, in, or near the benthos: blackfish, drums and croakers, gadids, flatfish-other, oyster toadfish, scup, skates, sea robins, striped bass, summer flounder, weakfish, and winter flounder.
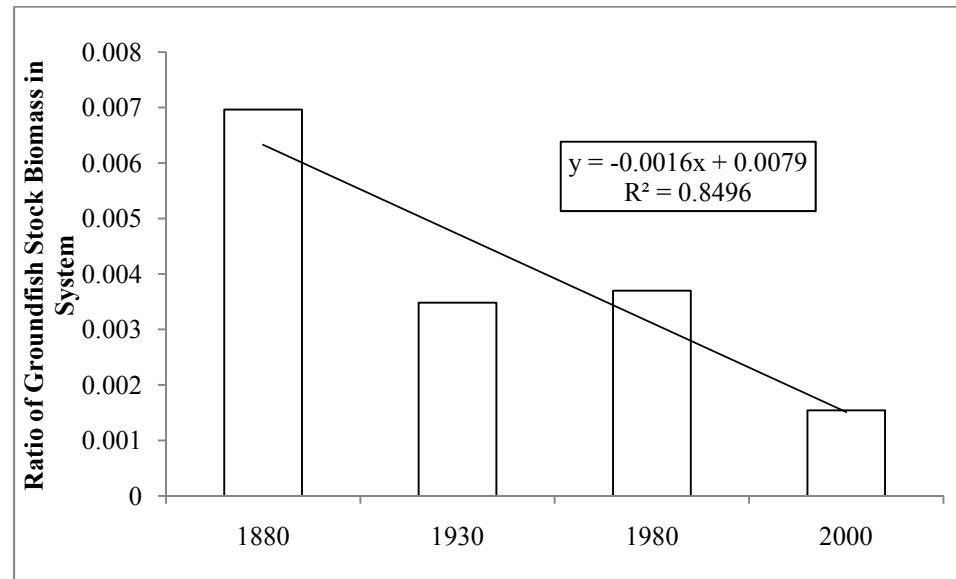
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Chapter 4: Summary

The Great South Bay (GSB) ecosystem encompasses 290 km^2, supplying marine resources from a multitude of shellfish, finfish, and crustaceans to Long Island, New York (Gabriel 1921, Dickinson 1938, Westman 1938, Hanlon 1983). The history of GSB shellfishing is marked by periods of discovery and overexploitation. Commercial shellfish fishermen, nicknamed baymen, have subsided between three predominant species: blue mussel, eastern oyster, and hard clam. Baymen targeted mussels through the 1700s as these shellfish were the easiest to harvest (Van Popering and Glancy 1947). Discovery of the Blue Point oyster beds in the early 1800s marked the beginning of the GSB oyster industry. Overcapacity of the fishery collapsed the Blue Point beds in 1824, only to be replaced by new beds found off Patchogue (Gabriel 1921). Oystering continued into the early 1900s, unnaturally high yields fueled by imported seed oyster (Gabriel 1921,Van Popering and Glancy 1947). In 1931, the reopening of Moriches Inlet altered the salinity and circulation patterns of GSB. As oystering declined, harvest of hard clams increased under the new conditions (McHugh 1972). Harvest remained appreciable through the 1940s, steadily increasing until 1976 when fishery pressures caused a collapse in the stock (Conrad 1982, Gobler et al. 2005, Kraeuter et al. 2008).

Declines in various finfish stocks are also attributed to the GSB fisheries. Primarily a destination for recreational anglers before the 1880s (Mather 1884), GSB

fishermen historically targeted large sharks (Thorne 1916, 1928). As these elasmobranches are essentially extirpated, stocks of winter and summer flounder, bluefish, blackfish, and striped bass (Westman 1938, Briggs 1962, Hanlon 1983) have become popular targets of anglers. Commercially, the usage of Atlantic menhaden as a fertilizer in the late 1700s prompted the creation of a fishery (Gabriel 1921, Dickinson 1938). This fishery grew into an industry in 1850, when a process to extract menhaden oil, used in painting and tanning, was discovered. Unsustainable fishing caused stocks to decline, pushing fishermen further and further offshore (Gabriel 1921). In 1897, the entire Atlantic coast menhaden industry was consolidated under the American Fisheries Company (AFC). The AFC closed many of the existing menhaden factories, including those in GSB, effectively ending the bay's involvement in the industry (Gabriel 1921). Winter and summer flounder, bluefish, and weakfish have also supported commercial harvest (Dickinson 1938). Blue crabs maintained immense yields in the late 1800s. Collapsing in the early 1900s, this stock is currently experiencing a rebound (Briggs 1998).

Under various ecosystem stressors (Van Popering and Glancy 1947, Ryther et al. 1957, Leatherman 1985, Short et al. 1987, Gobler et al. 2005), the resident GSB stocks that support fisheries have experienced drastic variations over the last century. CPUE data was analyzed as an index of stock abundance. Of the twelve stocks with identifiable temporal abundance trends, eight are currently declining. Commercially important flatfish stocks of winter and summer flounder currently exist at less than 20% of abundances recorded during peak landings in the 1980s. American eel, blackfish, and northern puffer are recently found at 25% of their estimated 1980s stock abundances.

Weakfish have also experienced a recent decline (-35%). Landings statistics show these abundance trends are represented by fishery data. Recreational yield from winter flounder and blackfish are less than 10% of that recorded in the 1980s while summer flounder and northern puffer has dropped about 50%. The commercial harvest of American eels is less than 50% of that exploited in the 1970s. The commercial importance of winter flounder, menhaden, striped bass, and northern puffer has also declined. Conversely, CPUE shows an increase in northern kingfish and windowpane flounder, 76% and 86% respectively.

The reduced abundance of these stocks is a cause of concern for many Long Island locals. In addition to monetary losses to local fishermen, the current system is lacking a commercially dominant shellfish stock for the first time in over 200 years. These filter feeders were responsible for filtering an estimated 40% of the total GSB water volume each day in the 1960s–70s (Kassner 1993). In 1993, the same shellfish stock filtered 1-2% of the total volume (Kassner 1993). This reduction has been cited as a possible driver for the relative frequency of brown tide events currently plaguing GSB (Cerrato et al. 2004, Gobler et al. 2005). Once established, these blooms have been shown to affect the composition of plankton, various shellfish and finfish stocks, and eelgrass habitat (Cosper et al. 1987, Bricelj and Kuenstner 1989, Bricelj and Lonsdale 1997, Greenfield and Lonsdale2002, Cerrato et al. 2004). Unfortunately, these impacts may also impede any management aimed at the potential recovery of GSB stocks.

To benefit efforts aimed at restoration, the cause of these shifts should be analyzed. The consistent mention of fishing pressures as a driver of stock abundances (Gabriel 1921, Van Popering and Glancy 1947, Gobler et al. 2005, Kraeuter et al. 2008)

implies it to be a predominant factor. However, anthropogenic eutrophication (Ryther 1954) and alterations to physical conditions (Van Popering and Glancy 1947, Leatherman 1985, Conley 1999) and structural habitat (Short et al. 1987, Carpenter and Brinkhuis 1991) have also been influential. While the historical data proved insufficient to assess causation, ecosystem models were developed to delve further into the system to determine the predominant drivers of the GSB ecosystem.

Using Ecopath software, ecosystem models were constructed for four periods: 1880s, 1930s, 1980s, and 2000s. These models were assessed for various indices of ecosystem structure, maturity, keystoneness, fishery pressures, and ecosystem efficiency. Twenty two of the twenty four included maturity indices estimated an overall drop in the ecosystem maturity of GSB. Seen in various measures of overhead (O, O_{ex}, and Redund.), the GSB ecosystem has become increasingly stressed over the last 120 years. Concurrent reductions in ecosystem size and maturity further suggest these pressures have drastically altered the dynamics of GSB. Declines in ecosystem size (system biomass and TST) coincide with drops in the proportion of biomass that is exploitable to consumers (B/TST, B / PP+RESP). Reductions in various measures of exergy (EX and EX_{st}) suggest GSB is capable of supporting a greater diversity of energy flows and food web complexity than currently supported. The food web has shifted from a system dominated by oysters, menhaden, and a diverse array of piscivorous fish to one dominated by lower trophic species with reduced production from migratory and groundfish stocks.

Sequentially modeled ecosystems also utilized a smaller portion of available production, seen in PP/RESP, PP/B, and PPR/PP, suggesting GSB is becoming more

eutrophic. This trend can be explained by pressures from phosphorus loading (Adamson 1982, Nixon et al. 1994) or overfishing (Jackson et al. 2001). High inorganic nutrient inputs (Ryther 1954, Gobler et al. 2005) have instigated harmful algal blooms since the 1930s. These stocks may experience lower grazing pressures than other phytoplankton (Gobler et al. 2002, Caron et al. 2004), so a growing percentage of system production may be tied up in inedible plankton. Similarly, a drop in consumer abundance, symptomatic of overfishing, coinciding with consistent nutrient inputs would also create eutrophic conditions.

In addition to eutrophication, the models were analyzed for any expected trends from ecosystem stressors identified in GSB over the last 120 years. Variations to the southern barrier islands have impacted resident shellfish stocks (Van Popering and Glancy 1947, Greene 1982). Indeed, hard clams increased by 8.92 t/km^2/yr while oysters dropped 21.87 t/km^2/yr between the 1880s and 1930s models, corresponded to the reopening of Moriches Inlet. Fluctuations in eelgrass habitat (Carpenter and Brinkhuis 1991) co-occurred with expected variations in eelgrass epifauna, such as GSB shrimp. While the impact of phosphorus loading was not assessed through the standing stock of phytoplankton, the positive relationship between this stressor and the presence of harmful algal blooms was considered. Reductions in the bay's suspension feeders under these blooms (Cosper et al. 1987) were observed in the models. The impact of the GSB fisheries was also seen in the models. Fishing mortalities computed by Ecopath were found unsustainable for some GSB stocks (Cortés et al. 2002, Terceiro 2003, NEFSC 2009, Terceiro 2008), each showing modeled abundance declines. The removal of the influential shark grouping ($KS_{1930} = 0.064$) from GSB allowed summer flounder to seize

the apex predator trophic position, dropping the upper trophic level from 4.07 (TL_{shark}) to 3.80 (TL_{fluke}). Summer flounder have grown more influential to the GSB ecosystem $KS_{1880} = -0.049$ and $KS_{2000} = 0.409$). Consecutive drops in R/B and $(P/B)^{-1}$ indicate top-level consumers contribute less to recent than earlier modeled systems while drops in C and SOI reflect a reduction in omnivorous consumers, further showing the impact of the GSB fisheries.

Historically diverse and highly productive, GSB has shifted to an ecosystem with reductions across various species and total ecosystem size and maturity. As lagoonal systems are typically resilient to change (Day et al. 1989), the frequency of observed variations within stocks and the ecosystem from which they reside is alarming. Given such alterations, many conservation or management strategies may prove futile as the new system structure may be too far removed to return to historic baselines. Cause-and-effect information may enhance the efficiency of current management attempts aimed at restoration. Unfortunately, the synchrony of various stock fluctuations and modeled structural differences with multiple system stressors makes the separation of cause-and-effect difficult. However, trends consistent with phosphorus loading, overfishing, and habitat alteration and degradation were observed. Future modeling attempts should address causation.

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Appendix

Appendix 2.1. Species included in Ecopath Groupings and Estimation of Parameters

Benthic Fauna

Includes:
- Various Marine Worms
  - Platyhelminthes
  - Nematodes
  - Nemertean
  - Sipunculids
  - Annelids (excluding *Tomopteris* sp.)
- Pycnogonids
- Various Mollusks
  - Aplacophora (Shell-less Mollusks)
  - Various Gastropods
    - Cerithiidae (Ceriths)
    - Marine Snails (excluding *Crepidula* sp.)
      - *Nassarius* sp. (Mud Snails)
    - *Acteon punctostriatus* (Adam's Baby Bubble)
    - Unidentified Gastropods
- Various Crustaceans
  - Tanaids
  - Amphipods
  - Isopods
  - Cumaceans (Hooded Shrimps)
  - *Lernaea* sp. (Anchor Worms)
  - Horseshoe Crab Eggs
  - Unidentified Crustaceans
- Fish Eggs

Parameter Estimates: The acquisition of an accurate biomass assessment was thought unlikely for this grouping based on the number of families included, small body sizes, and the nature of these individuals to burrow into the benthos. Instead, Ecopath parameters were used. Initial estimates came from a Middle Atlantic Bight study (Okey 2001). The polychaete and small crustacean groupings from Okey (2001) were averaged to estimate P/B, Q/B, and EE. The initial 2000s model used P/B = 4.58, Q/B = 24.36, and EE = 0.847. The balancing process of the 2000s model adjusted the EE to 0.859, which was used for the 1980s, 1930s, and 1880s models.

Diet: The dietary trend for this grouping was obtained from benthic macrofauna of Heymans (2001) Gulf of Maine model, using information from benthic invertebrates in Bundy et al. (2000). Bundy et al. used detritus as the sole component of diet for various mollusks, polychaetes, and other benthic invertebrates like amphipods and isopods. Assuming a similar trend in GSB, 100% of the benthic fauna diet was attributed to detritus for all four models.

Macrobenthos

Includes:
- Cnidarians (excluding Ctenophores)
- Echinoderms (excluding *Chelophyes appendiculata*)
  - *Asterias forbesi* (Forbe's Starfish)
- Various Mollusks
  - Marine Octopi
  - Various Gastropods
    - *Busycon canaliculatum* (Channeled Whelk)
    - *Busycon carica* (Knobbed Whelk)
    - Naticidae (Moon Snails)
    - *Urosalpinx cinera* (Eastern Oyster Drill)
  - Various Bivalves
    - *Macoma* sp.
    - *Nucula proxima* (Atlantic Nutclam)
    - *Yoldia* sp.
- Crustaceans
  - Lobsters – although essentially absent in GSB, these crustaceans needed to be classified to include diet studies from other systems in which lobsters were more abundant (when GSB-specific diet studies were unavailable)

Parameter Estimates: The number of benthic taxa classified under this grouping necessitated Ecopath to provide a biomass measure for this grouping. As such, Ecopath parameter estimates were used for each model. These estimates were obtained from studies conducted in the Gulf of Maine (Zhang and Chen 2007) and Mid-Atlantic Bight (Okey 2001). Values for the initial model were averages from the mollusca (Zhang and Chen 2007), echinoderm, and sessile epibenthos (Okey 2001) groupings and included $P/B = 1.0$, $EE = 0.804$, and $P/Q = 0.6$. These values remained unchanged across our models.

Diet: Dietary trends were combined from two separate species in Himmelman and Hamel (1993). Diet data was provided for common whelks, *Buccinum undatum*, and common seastars, *Asterias vulgaris*. These trends were used for the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Macrobenthos
2000s & 1980s

- Benthic Fauna
- Macrobenthos
- Suspension Feeders
- Hard Clams
- Detritus

Macrobenthos
1930s

- Benthic Fauna
- Macrobenthos
- Suspension Feeders
- Hard Clams
- Oysters
- Detritus

Macrobenthos
1880s
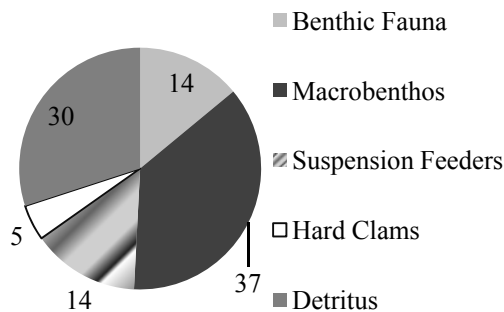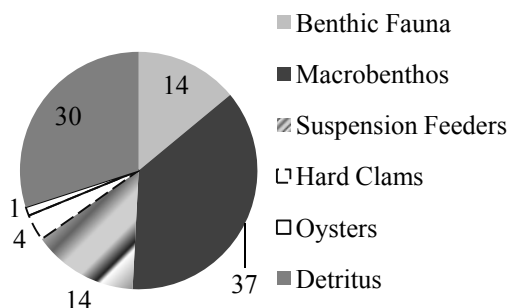
- Benthic Fauna
- Macrobenthos
- Suspension Feeders
- Hard Clams
- Oysters
- Detritus

Ctenophores

Includes:
- Ctenophores
- Scyphozoa (Jellyfish)

Parameter Estimates: A biomass estimate for ctenophores was unavailable from the literature. Initial Ecopath estimates came from Delaware Bay (Frisk et al. in review) in which P/B = 8.8, EE = 0.9, and P/Q = 0.25. These values were applied to all four models. The unassimilated consumption of this grouping was raised to 0.4, thought to be more accurate for planktonic feeders (Christensen et al. 2008).

Diet: Initial dietary estimates were obtained from an Ecopath model of Delaware Bay (Frisk et al. in review). This model asserted zooplankton and phytoplankton comprise the entirety of the ctenophore diet and assumed a 50:50 ratio. These trends were assumed representative across all four models.

Sand Shrimp

Includes:
- *Crangon septemspinosa*
- *Crangon* sp.

Parameter Estimates: The reclusive nature and small size of these crustaceans made an accurate biomass estimation difficult to obtain across our modeled periods. Therefore, three Ecopath parameter estimates were required to balance this grouping. The initial model inputs for this grouping came from two locations within the northeast Atlantic Ocean. A study conducted in the Irish Sea (Oh et al. 1999) provided an estimate of P/B = 3.75. Multiple estimates of Q/B were obtained and averaged from western Sweden (Pihl and Rosenberg 1984); the initial Q/B input was 24.9. EE was estimated at 0.999 based on the frequency of these crustaceans in the diets of predators within Sagarese (2009). No changes were deemed necessary to these parameters throughout the balancing process.

Diet: Dietary trends were obtained from a stomach analysis conducted from the Pettaquamscutt River, Rhode Island (Wilcox and Jeffries 1974). This same analysis was used as the estimate for all four models.

Sand Shrimp

Benthic Fauna

Detritus
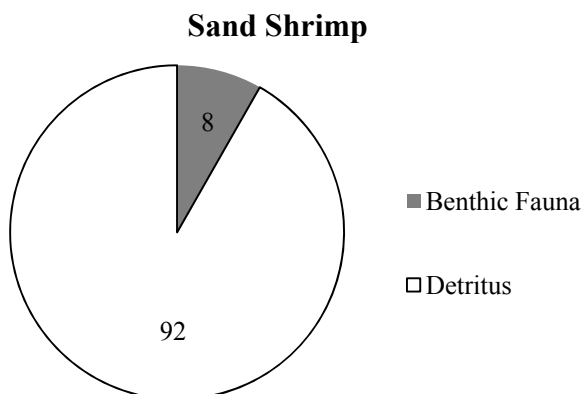
Mantis Shrimp

Includes:
- Stomatopods
- *Squilla empusa*

Parameter Estimates: Few mantis shrimp were obtained in the trawling surveys conducted in GSB, necessitating a biomass estimate from the Ecopath software. The initial estimates for the mantis shrimp grouping came from the Middle Atlantic Bight (Okey 2001) in which P/B = 1.34, Q/B = 7.43, and EE = 0.95. These values remained unchanged across the four periods.

Diet: Dietary estimates for mantis shrimp were provided from samples obtained from the lower York River, Virginia in Pihl et al. (1992). These samples were used as direct estimates of mantis shrimp diet in the 2000s and 1980s. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.
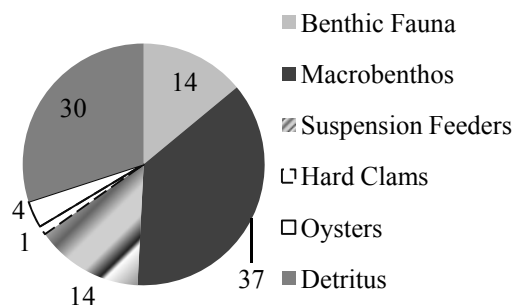

Mantis Shrimp
2000s & 1980s

■ Benthic Fauna
■ Suspension Feeders
□ Hard Clams

Mantis Shrimp
1930s

■ Benthic Fauna
■ Suspension Feeders
□ Hard Clams
▫ Oysters

Mantis Shrimp
1880s

■ Benthic Fauna
■ Suspension Feeders
□ Hard Clams
▫ Oysters
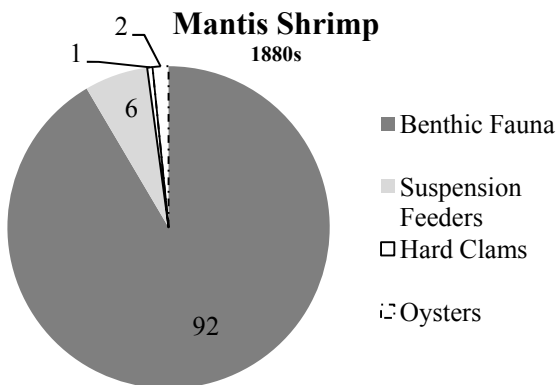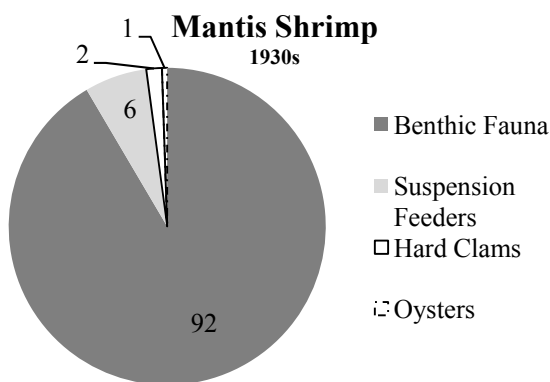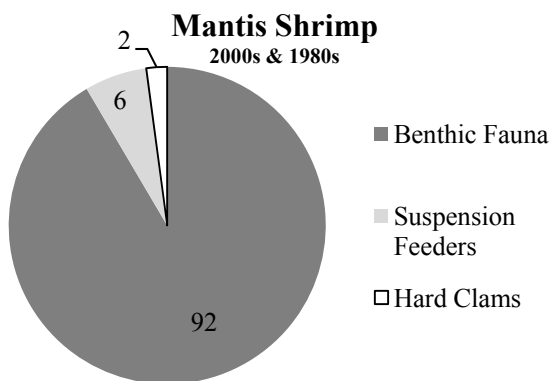
Shrimp

Includes:
- Caridean Shrimp (True Shrimp)
  - Pandalidae
- Penaeoideia (Penaeid Shrimp/Prawns)
- Mysidae (Mysid Shrimp)
- Unidentified Shrimp

Parameter Estimates: Although the 2000s GSB survey frequently observed shrimp within the catch (Frisk and Munch, 2008), the selectivity of these small crustaceans to the otter trawl likely hampered the comparability of the shrimp catch to other stocks. With doubt in the 2000s biomass estimate and no estimates available for other periods, initial inputs included three Ecopath parameters. These initial values came from averages of specimens obtained from the Gulf of Maine (Zhang and Chen 2007) and the Middle Atlantic Bight (Okey 2001) systems. Initial estimates included $P/B = 3.3$, $Q/B = 16.1$, and $EE = 0.86$ and were consistent throughout our modeled periods.

Diet: Dietary estimates for all four models were obtained from the pelagic mysids *Mysis mixta* and *Mysis relicta* from the Gulf of Finland in the northern Baltic Sea (Viherluoto et al. 2000). This study was assumed representative of all four GSB systems.

Shrimp
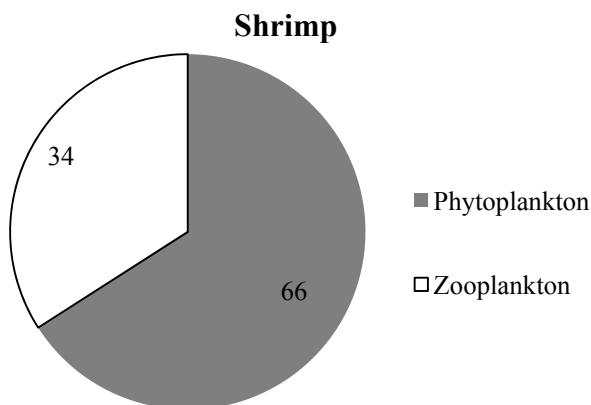
Squid

Includes:
- *Loligo pealeii* (Longfin Squid)
- *Illex illecebrosus* (Shortfin Squid)

Parameter Estimates: A stock biomass of 0.0527 tonnes/km^2 was estimated for the 2000s model by a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial parameter estimates for this grouping came from the Gulf of Maine (Zhang and Chen 2007) in which Q/B (7.0) and EE (0.99) were calculated. These values were used in all four models. As a biomass estimate was unavailable for the 1980s, the

balanced P/B from the final 2000s model (2.744) was input in the 1980s model. Input with the Q/B and EE from the 2000s, a biomass estimate was provided by Ecopath. However, after balancing all four models, the difference in the calculated mortalities between the 2000s and 1930s models (see next paragraph) raised concerns that the 2000s P/B may have overestimated the mortality on the 1980s stock. Therefore, the 1980s mortality was lowered to an intermediate value of 1.744. A stock biomass of 0.0383 t/km^2 was measured for the 1930s model from data supplied by Westman (1938) and Dickinson (1938), leaving Ecopath to calculate a P/B. Since a biomass estimate was also unavailable for the 1880s model, the balanced P/B ratio from the 1930s model (0.803) was input with Q/B and EE estimates.

Diet: Dietary trends were combined from two separate species. Data for northern shortfin squid, *Illex illecebrosus*, and longfin inshore squid, *Loligo pealeii*, from Cape Hatteras was combined to yield the initial diet of the squid grouping (Bowman et al. 2000). The balancing process necessitated a reduction in the gadidae component in the 2000s model. A further reduction was needed when balancing the 1980s model. The 1930s and 1880s models used the final diet from the 1980s model.
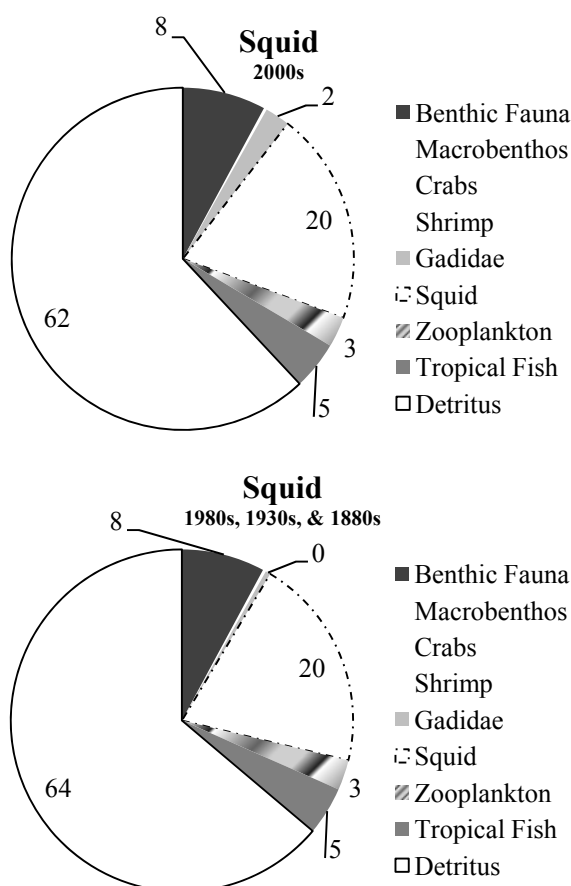
Hard Clams

    Includes:
- *Mercenaria mercenaria*

    Parameter Estimates: Since the 2000s hard clam biomass estimate (16.1692 t/km^2) from Kraeuter et al. (2008) was provided by clam beds solely from Islip waters, this estimate was believed inaccurate and discarded (inflated from the relatively large stock in Islip compared to other areas in GSB). However, the time series of hard clam abundance trends in the Kraeuter et al. (2008) study was still thought representative of the GSB stock. Using the 1980s hard clam abundance from (Greene 1982) and assuming the Kraeuter et al. (2008) time series was indeed accurate, the theoretical hard clam abundance in the 2000s was calculated (1.7837 t/km^2) and input into the model. The initial Q/B estimate for this grouping (5.1) came from the Chesapeake Bay ecosystem model (Christensen et al. 2005). We estimated EE = 0.999 as shellfish have a long harvest history within GSB and are believed to be consumed by multiple predators. As biomass estimates were available for all four periods, these two Ecopath parameters were applied to all four models, allowing the program to estimate the stock mortality (P/B). A stock biomass of 4.9493 t/km^2, obtained from the Wapora survey (Greene 1982), was used for the 1980s model. The biomass of the 1930s, 13.375 t/km^2, was estimated from data presented by Townes (1938). As the selective harvest of baymen was focused on oysters in the early 1900s, the hard clam stock was assumed under-utilized by the fishery through the 1880s. However, the hard clam stock of the 1880s was assumed less than that found in the 1930s based on three factors: the physical conditions of GSB during this period were optimal for oysters, not hard clams; the monopolization of available benthos by oyster planters likely suppressed clam growth; and the reopening of Moriches Inlet in 1931 may have increased hard clam abundance, suggesting the 1930s model (representing a transition state between oysters and hard clams) supported more clams than the 1880s. Since the predation on hard clams in the 1880s was estimated as 33% of that found in the 1930s (dietary input), this ratio was used to roughly estimate hard clam biomass as well. Therefore, we estimated the 1880s hard clam biomass at 4.458 t/km^2.

    Diet: The dietary trends used in all four models were obtained from the Delaware Bay model of Frisk et al (in review).

Hard Clam

- Phytoplankton
- Zooplankton
- Detritus
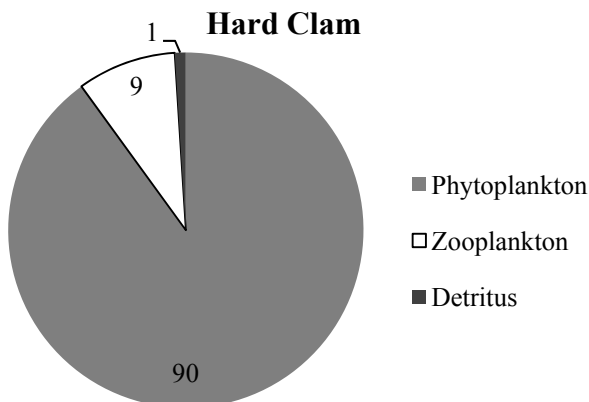
Oysters

Includes:

- *Crassostrea virginica*

This abundant species provided a large harvest up until the early 1900s, but was considered relatively unimportant to the system during the latter part of the 1900s. Therefore, this grouping was only separated for the 1930s and 1880s models. In the 1980s and 2000s models, oysters were fed into the suspension feeder category (see below).

Parameter Estimates: Biomass estimates for the oyster stock were unavailable from the literature. Furthermore, the usage of oyster planting in GSB through the early 1900s supported an unnaturally large harvest of oysters, suggesting an estimation of actual biomass may be difficult to obtain through the estimation of natural processes. Using Ecopath estimates, the dynamics of the oyster stock were modeled from the Delaware Bay (Frisk et al. in review). Values of $P/B = 0.2$ and $Q/B = 2.02$ were used. We estimated $EE = 0.999$ as these shellfish have a long harvest history within GSB and are believed to be consumed by multiple predators. These values remained unchanged across our models.

Diet: Dietary trends were obtained from the Chesapeake Bay model of Christensen et al (2005). These results were input into both the 1930s and 1880s models.

Oysters

- Phytoplankton
- Zooplankton
- Detritus

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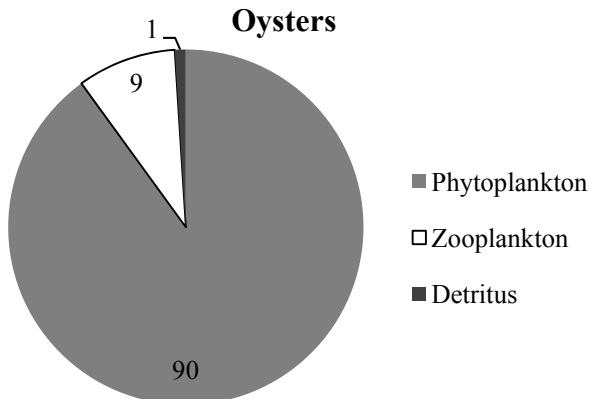
Suspension Feeders

Includes:
- Porifera (Sponges)
- Urochordata (Tunicates)
    - Salpidae (Salps)
    - Ascidiacea (Sea Squirts)
    - Unidentified Tunicates
- Various Mollusks
    - *Crepidula* sp. (Slipper Snails)
    - Various Bivalves
        - *Mytilus edulis* (Blue Mussels)
        - *Crassostrea virginica* (Eastern Oyster)
        - *Argopecten irradians* (Bay Scallop)
        - *Mya arenaria* (Softshell Clam)
        - *Ensis directus* (Razor Clam)
        - *Gemma gemma* (Amethyst Gem Clam)
        - *Mulinia lateralis* (Dwarf Surf Clam)
        - Unidentified Pelecypoda (Freshwater Clams)
        - Unidentified Bivalves
- Cirripedia
    - *Balanus improvisus* (Bay Barnacles)

Note: the eastern oyster was a large fishery in Great South Bay up until the mid 1900s. Therefore, this species was only included in the suspension feeder grouping for the 1980s and 2000s models.

Parameter Estimates: Based on the number of families included in this grouping, the acquisition of an accurate biomass assessment was unlikely. Instead, Ecopath parameters were used. The initial estimates for this grouping came from the Chesapeake Bay ecosystem model (Christensen et al. 2005). Average values of P/B = 0.3 and P/Q = 0.138 were used. We estimated EE = 0.9 as numerous shellfish in this grouping have history of harvest within GSB. However, these bivalves were never under

the same fishing pressures as hard clams or oysters and so a lower EE was thought more accurate. The balancing process of the 2000s model adjusted the EE to 0.9308, which was used for the 1980s, 1930s, and 1880s models.

Diet: Dietary trends were obtained from the Chesapeake Bay model of Christensen et al (2005). These trends were applied to all four models.

Suspension Feeders
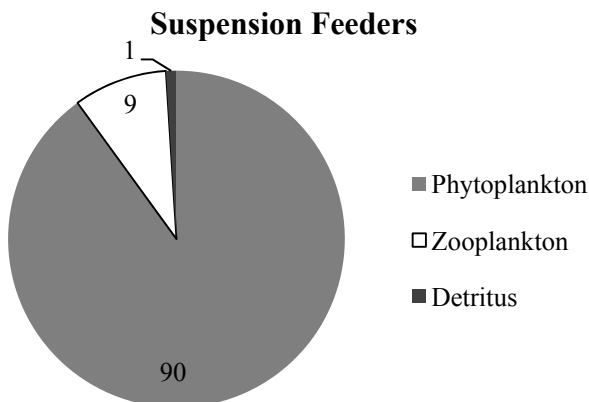
Blue Crabs

Includes:
- *Callinectes sapidus*
- *Callinectes* sp.

Parameter Estimates: The initial 2000s model was based on Ecopath parameter estimates from the Delaware Bay (Frisk et al. in review) and Chesapeake Bay (Christensen et al. 2005). The Delaware model provided estimates of P/B = 1.4 and EE = 0.95 for blue crabs. The Q/B estimate (4.0) was taken from the Chesapeake model. Using the estimated biomass from the final 2000s model (0.4232 t/km^2) and stock abundance trends shown in the landings data of Briggs (1998), the biomass of the 1980s blue crab stock was set at 0.2 t/km^2. Inputting the P/B and Q/B values from the 2000s model, EE was left to be estimated by the 1980s model. As the calculated EE was greater than one, a slight increase in the biomass (0.202 t/km^2) was required to produce a possible EE. The agreement of the two biomass measures suggests the method for estimating the 1980s biomass was relatively accurate. The 1930s and 1880s models lacked a biomass estimate. Therefore, the initial estimates of P/B (1.4), Q/B (4.0), and EE (0.95) were used.

Diet: The initial dietary trends were combined from multiple studies. Data was compiled from the Hudson-Raritan estuary (Stehlik et al. 2004), Navesink, New Jersey (Meise and Stehlik 2003), and Rhode Island (Ropes 1988). The unidentified shrimp category in Meise and Stehlik (2003) was classified as sand shrimp based on presence in Stehlik et al. (2004). These inputs were used for the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as

these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Blue Crabs
2000s & 1980s

Benthic Fauna
Macrobenthos
Sand Shrimp
Suspension Feeders
Hard Clams
Crabs
Blue Crabs
Spider Crabs
Zooplankton
Detritus

Blue Crabs
1930s

Benthic Fauna
Macrobenthos
Sand Shrimp
Suspension Feeders
Hard Clams
Oysters
Crabs
Blue Crabs
Spider Crabs
Zooplankton
Detritus

Blue Crabs
1880s

Benthic Fauna
Macrobenthos
Sand Shrimp
Suspension Feeders
Hard Clams
Oysters
Crabs
Blue Crabs
Spider Crabs
Zooplankton
Detritus
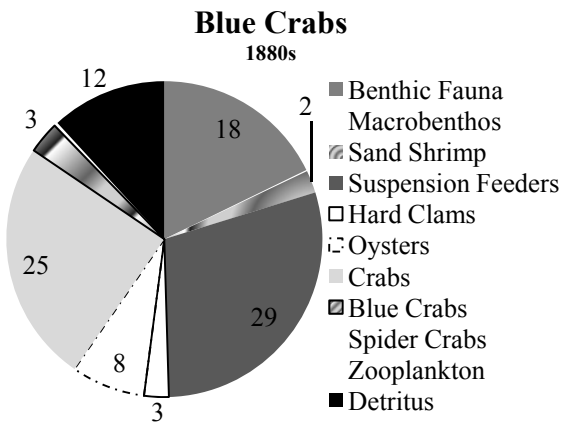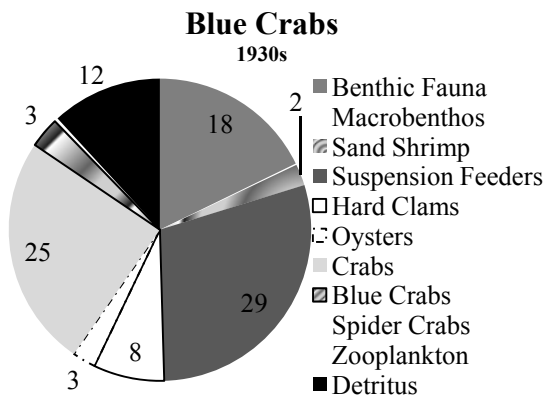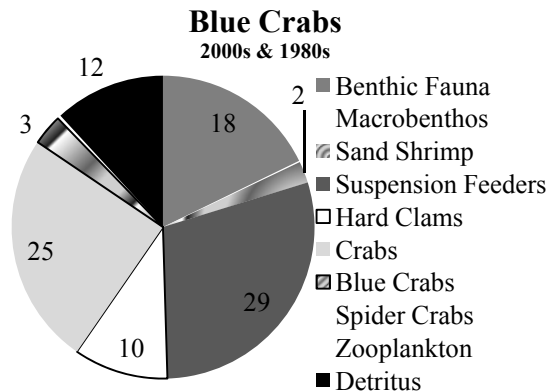
Crabs

Includes:
- *Ovalipes ocellatus* (Lady Crab)
- *Cancer irroratus* (Atlantic Rock Crab)
- *Pagurus* sp. (Hermit Crabs)
- *Carcinus maenas* (European Green Crab)
- Unidentified Xanthids (Mud Crabs)
  - *Panopeus herbstii* (Atlantic Mud Crab)
  - *Neopanopes texana*
- Lithodidae (Stone and King Crabs)
- Unidentified Crabs (Brachyurans)

Parameter Estimates: The variety of families incorporated in this grouping suggested using the Ecopath software may provide the most accurate estimate of biomass. As such, Ecopath parameter estimates were used to balance the crabs grouping. The initial estimates for this grouping came from the Gulf of Maine (Zhang and Chen 2007) which calculated a P/B = 1.38, Q/B = 8.5, and EE = 0.99. The balancing process of the 2000s model adjusted the EE to 0.9633, which was used for the 1980s, 1930s, and 1880s models.

Diet: The dietary trends used in the 2000s model were combined from four separate species. The Atlantic rock crab, *Cancer irroratus*, diet was estimated from the Hudson-Raritan estuary (Stehlik et al. 2004). The prey preferences of lady crabs, *Ovalipes ocellatus*, were provided by samples from the Hudson-Raritan estuary (Stehlik et al. 2004) and Pettaquamscutt River, Rhode Island (Ropes 1988). The dietary habits of the mud crab, *Neopanopeus texana*, were taken from the Pettaquamscutt River (Ropes 1988). The prey of the invasive European green crab, *Carcinus maenas*, was also assumed to be accurately represented in Ropes (1988). The unidentified fish category in Ropes (1988) was classified as detritus based on the discussion of Stehlik et al (2004). For the 1980s model, a reduction in the blue crab component was required during the balancing process; as only a 0.37 percent reduction in total weight was required, no visible difference was evident between the two periods and only one pie chart was given below (for the 2000s and 1980s). The 1980s results were also applied to the 1930s model. The final model of the 1880s required a reduction in the crab category (cannibalism). Furthermore, the 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Crabs
2000s & 1980s

- ■ Benthic Fauna
  Macrobenthos
  Sand Shrimp
  Shrimp
- ▨ Suspension Feeders
- ▢ Hard Clams
- ■ Crabs
  Blue Crabs
  Spider Crabs
  Zooplankton
- ▨ Detritus

Crabs
1930s

- ■ Benthic Fauna
  Macrobenthos
  Sand Shrimp
  Shrimp
- ▨ Suspension Feeders
- ▢ Hard Clams
- ▢ Oysters
- ■ Crabs
  Blue Crabs
  Spider Crabs
  Zooplankton
- ▨ Detritus

Crabs
1880s

- ■ Benthic Fauna
  Macrobenthos
  Sand Shrimp
  Shrimp
- ▨ Suspension Feeders
- ▢ Hard Clams
- ▢ Oysters
- ■ Crabs
  Blue Crabs
  Spider Crabs
  Zooplankton
- ▨ Detritus

Horseshoe Crabs

Includes:
- *Limulus polyphemus*

Parameter Estimates: The low biomass estimate calculated from the Frisk and Munch trawling survey (0.0003 t/km^2) was believed low based on the importance of this species to waterfowl that utilize GSB during annual migrations. Therefore, the estimated

2000s biomass was discarded, leaving the dynamics of horseshoe crabs to be based on parameter estimates from Delaware Bay (Frisk et al. in review). From Frisk et al, P/B was set to 0.6 and Q/B at 3.0. As few studies reported horseshoe crabs in the diets of our Ecopath stock groupings, these crabs were assumed underutilized by the system and EE was estimated at 0.5.

Diet: The diet of horseshoe crabs was provided from a study conducted in Delaware Bay, New Jersey (Botton 1984). Considering the benthic nature and lethargic movements of this arthropod, we assumed horseshoe crabs were inefficient at capturing finfish; as such, the unidentified fish category within the diet was assumed the result of chance encounters with decaying material along the benthos and was added to detritus. This study was used for the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.
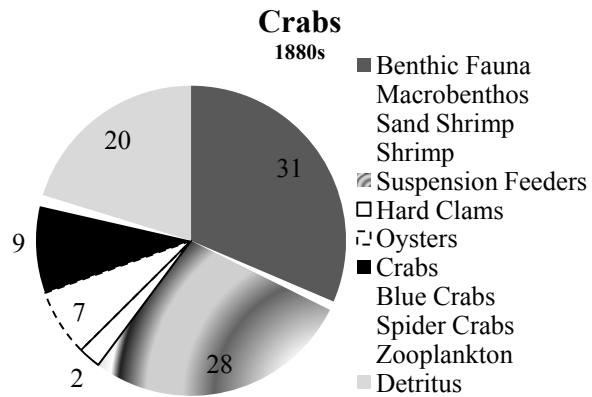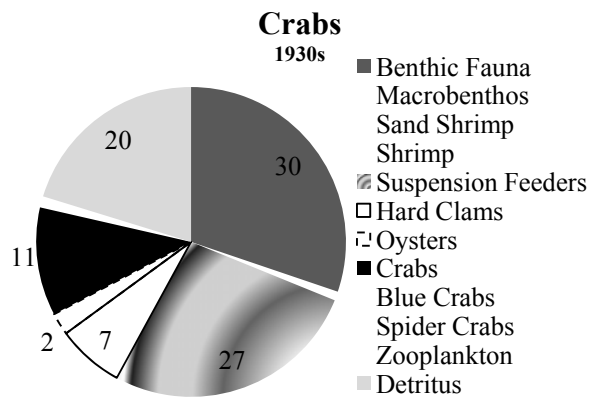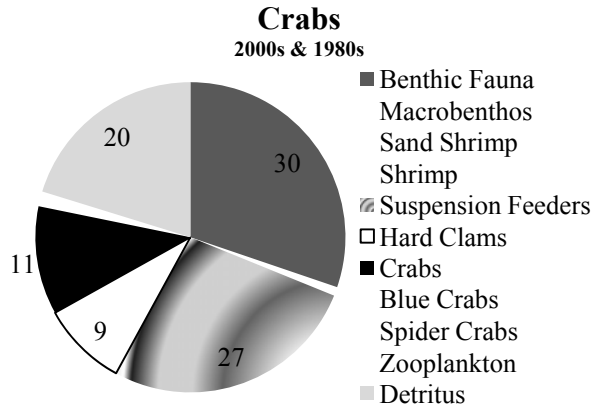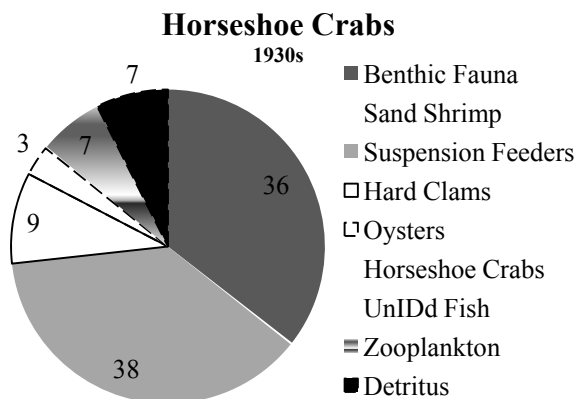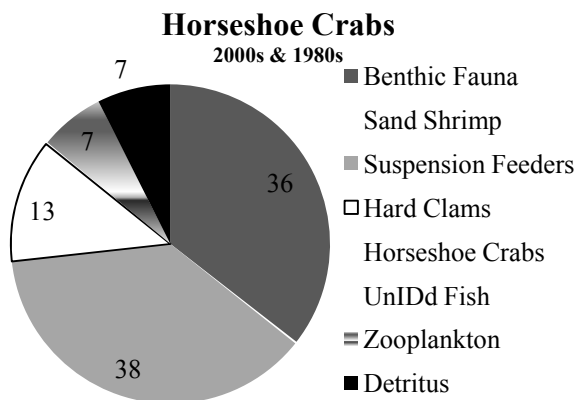

Horseshoe Crabs
2000s & 1980s

Horseshoe Crabs
1930s

Horseshoe Crabs

1880s

Legend:
- Benthic Fauna
- Sand Shrimp
- Suspension Feeders
- Hard Clams
- Oysters
- Horseshoe Crabs
- UnIDd Fish
- Zooplankton
- Detritus

Values shown on pie: 36, 38, 3, 9, 7, 7

Spider Crabs

Includes:
- *Libinia dubia* (Longnose Spider Crab)
- *Libinia emarginata* (Portly Spider Crab)
- *Libinia* sp.

Parameter Estimates: An initial stock biomass of 0.0987 t/km^2 was estimated by a GSB benthic trawling survey conducted in 2007 (Frisk and Munch 2008). Since studies specific to the US coastline were lacking for spider crabs, the P/Q (0.15) provided by Stanford and Pitcher (2004) from the English Channel was used. Based on the high ecotrophic efficiencies of blue crabs (0.95) and the generic crab grouping (0.99), an EE = 0.95 was assumed adequate and used as an initial estimate. Estimates of stock biomass were missing for the other three models, so the estimated P/B (2.296) and initial EE and P/Q values from the 2000s model were used in the 1980s, 1930s, and 1880s models.

Diet: The dietary habits of the spider crab, *Libinia emarginata*, were estimated within the Pettaquamscutt River, Rhode Island (Ropes 1988). This study was used for the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Spider Crabs
2000s & 1980s
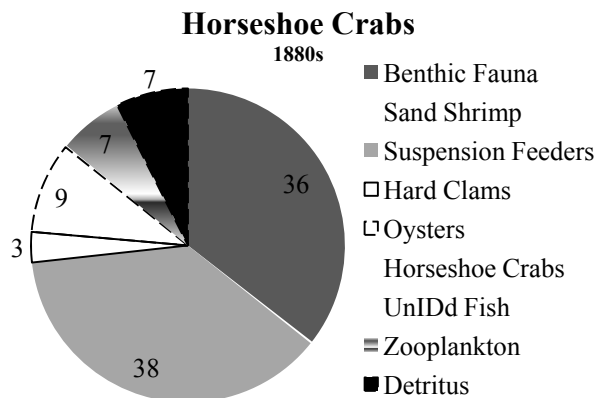

- Benthic Fauna
- Suspension Feeders
- Hard Clams
- Crabs
- Detritus

Spider Crabs
1930s

- Benthic Fauna
- Suspension Feeders
- Hard Clams
- Oysters
- Crabs
- Detritus

Spider Crabs
1880s
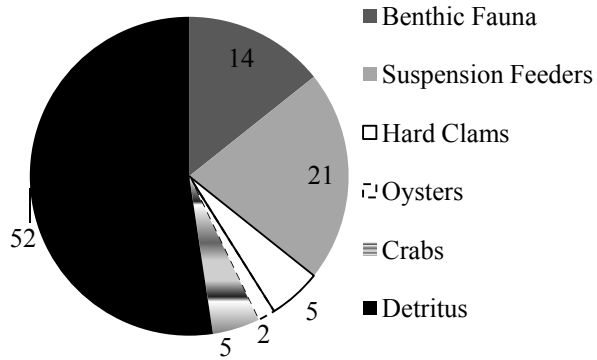
- Benthic Fauna
- Suspension Feeders
- Hard Clams
- Oysters
- Crabs
- Detritus

Skates

Includes:
- *Leucoraja erinacea* (Little Skate)
- *Leucoraja ocellata* (Winter Skate)
- *Raja eglanteria* (Clearnose Skate)
- *Dipturus laevis* (Barndoor Skate)
- *Dasyatis centroura* (Roughtail Stingray)

Parameter Estimates: A skate stock biomass of 0.8219 t/km^2 was estimated by the GSB trawl survey of Frisk and Munch (2008). The initial parameter estimates for this grouping came from Delaware Bay (Frisk et al. in review), supplying values of P/B = 0.46 and Q/B = 4.1. The balancing process of the 2000s model adjusted the P/B to 0.15, which was used for the 1980s, 1930s, and 1880s models. As no stock biomass were available for the subsequent periods, the estimated EE = 0.9827 and Q/B value (4.1) from the final 2000s model was used for the other three periods.

Diet: Data combined over three skate species supplied the dietary trends for all four models. Sagarese (2009) provided diet data for the clearnose skate, *Raja eglanteria*, in Shinnecock Bay of Long Island, New York. Winter skate, *Leucoraja ocellata*, and little skate, *Leucoraja erinacea*, samples from the Hudson-Raritan estuary (Steimle et al. 2000) also contributed to the overall diet. However, the high percentage of winter flounder in the diet samples from the Hudson-Raritan estuary were considered inaccurate owing to the absence of these flatfish in the Sagarese (2009) study; this component was therefore reduced for the initial dietary input. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to 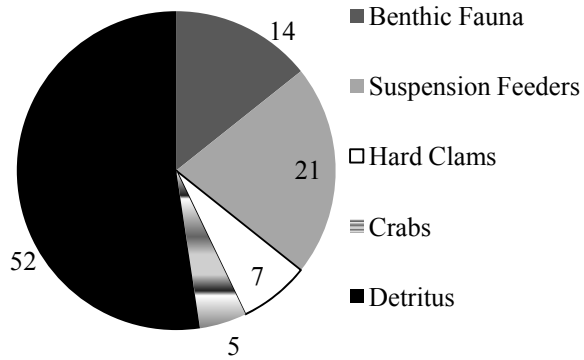h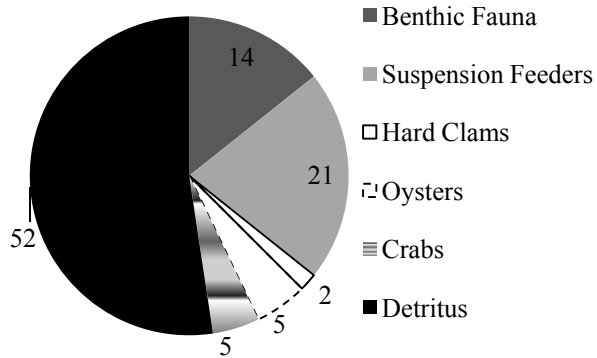ard clams while the remaining 75% were fed into oysters. As only a 0.57% of the overall diet was changed, no visible difference was evident between the pie charts of the two periods and only one figure was given below.

Skates

Legend:
- Benthic Fauna
- Crabs
- Sand Shrimp
- Shrimp
- Blue Crab
- Spider Crabs
- Detritus
- Suspension Feeders
- Hard Clams
- Winter Flounder
- Windowpane Flounder
- Tropical Fish
- Sea Robins
- Forage Fish
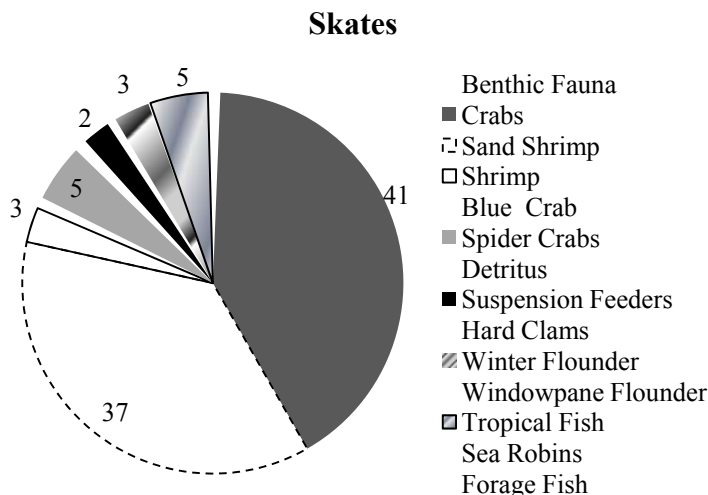
Sharks

> **Includes:**
> - *Carcharinus plumbeus* (Sandbar Shark)
> - *Isurus oxyrinchus* (Shortfin Mako)
> - *Prionace glauca* (Blue Shark)
> - *Negaprion brevirostris* (Lemon Shark)

Sandbar sharks (*Carcharhinus milberti*) utilized GSB as a nursery in the early 1900s, but this group was considered relatively unimportant to the system after the early 1900s. Therefore, this grouping was only used in the 1930s and 1880s model.

Parameter Estimates: As the inclusion of this group was necessitated by the importance of the *Carcharinus plumbeus* stock before the 1930s, the initial estimates for this grouping were specific to the sandbar shark species. A stock biomass of 0.0306 t/km^2 was used for the 1930s and 0.2125 t/km^2 for the 1880s, estimated from trends in Thorne (1928). Initial Ecopath parameters came from conclusions drawn by Wetherbee et al. (1990) in data gathered by Medved et al. (1985). From Wetherbee et al. (1990), P/B = 0.564 and Q/B = 4.0.

Diet: As sandbar sharks, *Carcharhinus plumbeus*, used Great South Bay as a nursery during these years and composed the majority of shark landings (Thorne 1928), the dietary trends of this species were assumed representative of the grouping. The diet was obtained from samples originating from Chesapeake Bay (Ellis 2003).

Sharks

Legend:
- Menhaden
- American Eel
- Winter Flounder
- Summer Flounder
- Windowpane Flounder
- Flatfish-Other
- Drums & Croakers
- Gadidae
- Striped Bass
- Black Seabass
- Weakfish
- Searobins
- Skates
- Forage Fish
- Tropical Fish
- Benthic Fauna
- Suspension Feeders
- Squid
- Mantis Shrimp
- Shrimp
- Blue Crab
- Spider Crab
- Crab
- Detritus
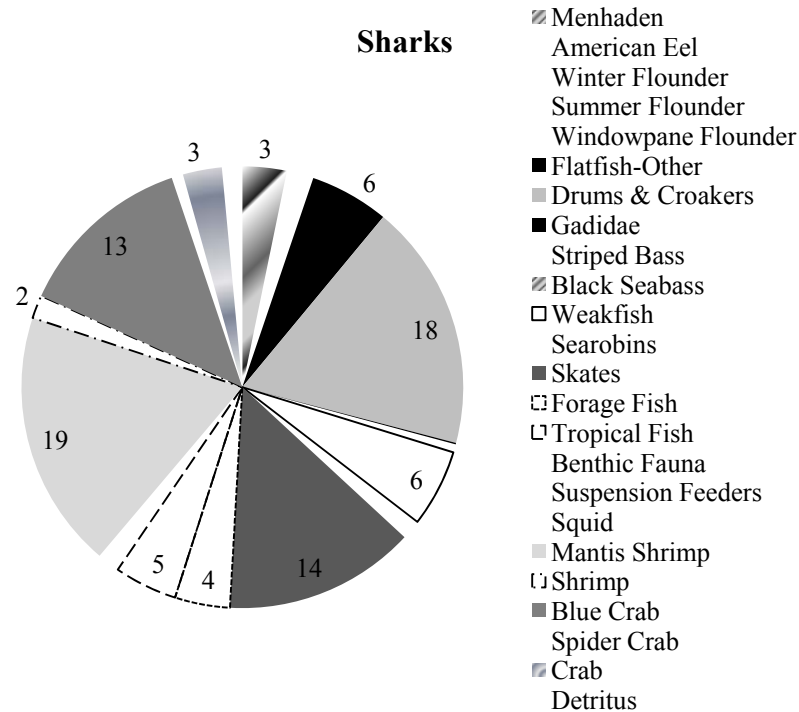
Forage Fish

Includes:
- Shads and Herrings
  - *Alosa aestivalis* (Blueback Herring)
  - *Alosa pseudoharengus* (Alewife)
  - *Alosa sapidissima* (American Shad)
  - *Alosa* sp.
  - *Clupea harengus* (Atlantic Herring)
  - Unidentified Clupeidae
  - "Whitebait" – generally refers to young herring
- Anchovies
  - *Anchoa mitchilli* (Bay Anchovy)
  - *Anchoa hepsetus* (Striped Anchovy)
  - *Anchoa* sp.
  - Engraulidae
- Killifish
  - *Fundulus heteroclitus* (Mummichug)
  - *Fundulus majalis* (Striped Killifish)
  - *Fundulus* sp.
- Atherinidae (Silversides)
  - *Menidia menidia* (Atlantic silverside)
- *Leuresthes tenuis* (California Grunion)
- *Leuresthes sardine* (Gulf Grunion)
- Gasterosteidae (Sticklebacks)

- Stichaeidae (Shannies or Pricklebacks)

Parameter Estimates: Over 97% of forage fish landed in 2007 (Frisk and Munch 2008) were bay anchovies, *Anchoa mitchilli*. As such, the initial estimates of this grouping focused on parameters specific to bay anchovies. Assuming an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C, functions found on the Fishbase website provided estimates of P/B = 1.29 and Q/B = 15.0 (Froese and Pauly 2005). Catch data from Frisk and Munch (2008) of bay anchovies supported this P/B, estimating mortality at 1.167. EE was estimated at 0.95 owing to the consistent presence of small fish in the piscivorous stomachs of Sagarese (2009). The combination of these three parameters (P/B, Q/B, and EE) was used to calculate forage fish biomass for all four models. Although biomass estimates were available for the 2000s model (0.1795 t/km^2) and 1980s models (0.0059 t/km^2), this grouping contained a disproportionately large number of stocks, making it difficult to assess the efficiency of the surveys ability to capture this grouping. The large variation between the two values supports this suspicion, where small differences in the spatial and temporal sampling protocol may have been an issue. Furthermore, the mesh used in the cod end of the Hanlon trawl measured 1.905 cm^2 whereas Frisk and Munch used 0.635 cm^2. For a grouping composed of small fish, this difference complicates the comparability of the two studies.

Diet: Dietary trends were combined for multiple species, all from Cape Hatteras, North Carolina (Bowman et al. 2000). Data was obtained for blueback herring, *Alosa aestivalis*, Atlantic herring, *Clupea harengus*, round herring, *Etrumeus teres*, hickory shad, *Alosa mediocris*, American shad, *Alosa sapidissima*, striped anchovy, *Anchoa hepsetus*, and unidentified silversides (Atherinidae). These trends were used for all four models.

Forage Fish

- Benthic Fauna
- Sand Shrimp
- Mantis Shrimp
- Shrimp
- Suspension Feeders
- Ctenophores
- Zooplankton
- Detritus
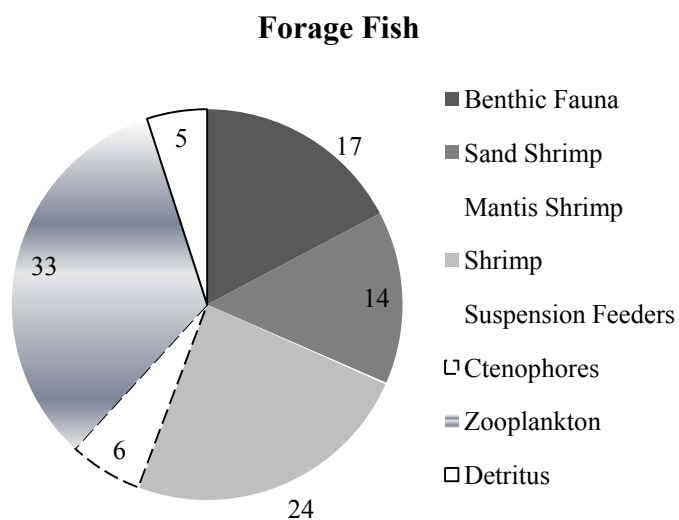
Tropical Fish

Includes:
- Albulidae (Bonefish)
- Ammodytidae
  - *Ammodytes dubius* (Sand Lance)
- Anarhichadidae (Wolffish)
- Apogonidae (Cardinalfish)
- Ariidae (Marine Catfish)
- Blennidae (Blennies)
- Carangidae (Jacks)
- *Chromis* sp.
- Cyprinidae (Carps)
- Gobiidae (Gobies)
- Hemiramphidae (Halfbeaks)
- Labridae (Wrasses)
- Mugilidae (Mullets)
- Mullidae (Goatfish)
- Myctophidae (Lanternfish)
- *Pareques* sp. (Highhats)
- Pholidae (Gunnels)
- Pomacentridae (Damselfish/Clownfish)
- Pomadasyidae (Grunts)
- Salmonidae (Salmons)
  - *Salmo trutta trutta* (Spotted Sea Trout)
- *Sebastes* sp. (Rockfish)
- *Selene setapinnis* (Atlantic Moonfish)
- *Sphoeroides* sp. (Pufferfish)
  - *Sphoeroides maculatus* (Northern Puffer)
- *Sphyraena barracuda* (Great Barracuda)
- *Stelifer* sp. (Stardrums)
- *Strongylura* sp. (Needlefish)
- *Symphurus* sp. (Tonguefish)
- Synodontidae (Lizardfish)
- Unidentified Eels (excluding Anguilla rostrata)
  - *Ophichthus cruentifer* (Marginated Snake Eel)
  - Ophidiidae (Cusk Eels)
  - *Ophidion marginatum* (Striped Cusk Eel)
- Unidentified Mackerel
  - Scombriadae
  - *Scomber scombrus* (Atlantic Mackerel)
  - *Scomberomorus maculates* (Spanish Mackerel)
  - *Scomberomorus cavalla* (King Mackerel)
- Unidentified Perches
  - Embiotocidae (Surfperches)

Parameter Estimates: The stock biomass of tropical fish was estimated in the 2000s by Frisk and Munch (2008) at 0.0765 t/km^2 and in the 1980s by Hanlon (1983) at 0.0556 t/km^2. However, the stochastic nature in the appearance of tropical fish created doubt in any accurate estimation of abundance for this grouping. Therefore, three Ecopath parameter estimates were used in each of the four GSB models. The Fishbase website (Froese and Pauly 2005) provided estimates of P/B and Q/B for northern puffer, American sand lance, and striped burrfish. These assumed an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. Calculated input values included a Q/B$_{average}$ = 7.967 and P/B$_{average}$ = 0.833. However, this mortality measure was thought low due to the high mortality on exotic species when in unnatural territory. Instead, catch data from Frisk and Munch (2008) of moonfish established a mortality measure of 1.2676. An EE of 0.999 was used. Since no biomass estimates were used from the literature, P/B, Q/B, and EE estimates were required to balance the 1980s, 1930s, and 1880s models. The Q/B (7.967) and EE (0.999) from the final 2000s model were both used as inputs for the remaining three models. Mortality was measured from two cohorts of bonefish in Alperin and Schaefer (1964) and amounted to 1.7006 and 1.0397. An average value of 1.3702 was used.

Diet: Dietary trends for this grouping came from samples across three species. Data for the American sand lance, *Ammodytes dubius*, was obtained from the Gulf of Maine (Meyer et al. 1979). The northern puffer, *Sphoeroides maculatus*, diet was represented by data from Great South Bay (Schreiber 1973). The diet of the striped burrfish, *Chilomycterus schoepfii*, was estimated from Tampa Bay, Florida (Motta et al. 1995). These trends were applied to the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Tropical Fish

2000s & 1980s

Tropical Fish
1930s

- ■ Benthic Fauna
-   Shrimp
- ■ Suspension Feeders
- □ Hard Clams
- ⁚⁚ Oysters
- ▨ Crabs
- ☰ Zooplankton
- ■ Detritus

Tropical Fish
1880s

- ■ Benthic Fauna
-   Shrimp
- ■ Suspension Feeders
- □ Hard Clams
- ⁚⁚ Oysters
- ▨ Crabs
- ☰ Zooplankton
- ■ Detritus
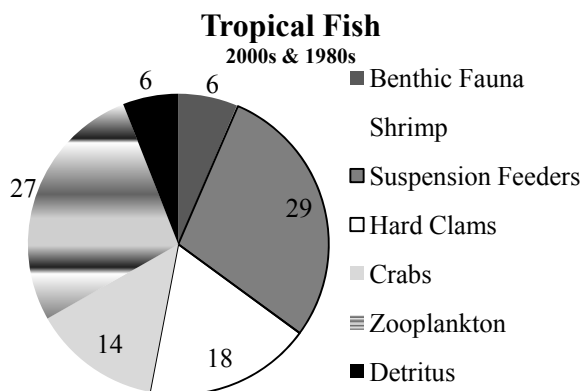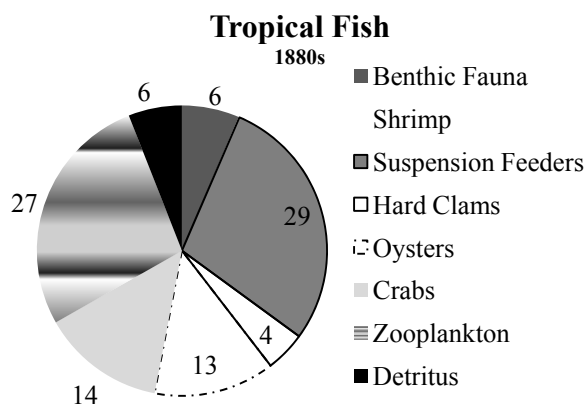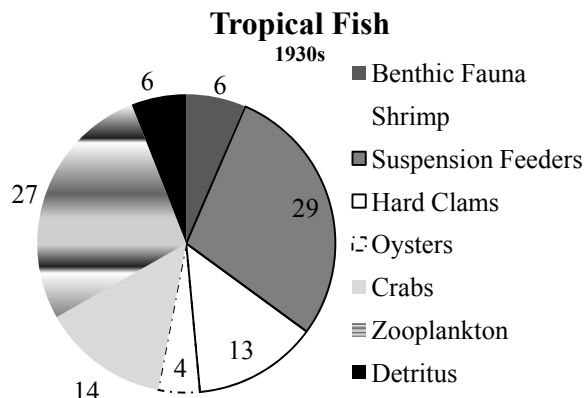
American Eel

Includes:
- *Anguilla rostrata*

Parameter Estimates: As no biomass estimates were available for this grouping, all four models required three initial estimates of Ecopath parameters. These estimates came from two sources: outputs from Fishbase (Froese and Pauly 2005) and Delaware Bay (Frisk et al. in review). Froese and Pauly estimated P/B = 0.22 and Q/B = 3.7 while assuming an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. Frisk et al. (in review) provided an EE of 0.5.

Diet: Dietary trends were provided by data from the Delaware River (Denoncourt and Stauffer 1993). Although finfish made up only a small proportion of the diet, Denoncourt and Stauffer (1993) did not distinguish between different species of fish. A study conducted in the Chesapeake Bay (Wenner and Musick 1975) found American Eel fed predominantly on the alewife, *Alosa pseudoharengus*. Therefore, the "fish" category from Denoncourt and Stauffer (1993) was attributed to Forage Fish in the model. The diet input was the same across all four models.

American Eel

0   1

99

- ■ Benthic Fauna
- □ Detritus
- ■ Forage Fish
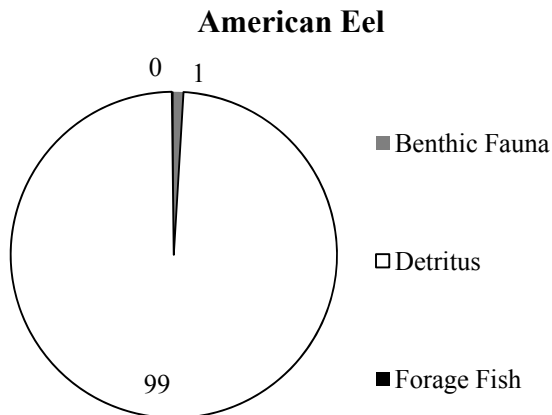
Atlantic Menhaden

Includes:
- *Brevoortia tyrannus* (Atlantic Menhaden)
- *Brevoortia* sp.

Parameter Estimates: Two biomass estimates were available for Atlantic menhaden: 0.0140 t/km^2 for 2000s (Frisk & Munch 2008) and 0.0426 t/km^2 for 1980s (Hanlon 1983). However, the prevalence of menhaden in the diets of GSB predators raised doubts as to whether these relatively low abundances were accurate. Therefore, biomass was left to be estimated by the models, necessitating the input of three Ecopath parameters. The initial estimates for menhaden came from both the Delaware Bay (Frisk et al. in review) and Middle Atlantic Bight (Zhang and Chen 2007). Input values included P/B = 1.1 and Q/B = 28.0 from Frisk et al. (in review) and EE = 0.99 from Zhang and Chen.

Diet: Dietary trends for menhaden obtained along the Atlantic coast of the United States (Lewis and Peters 1994) provided initial estimates for all four models.

Atlantic Menhaden

1   6

93

- ■ Zooplankton
- □ Phytoplankton
- ■ Detritus
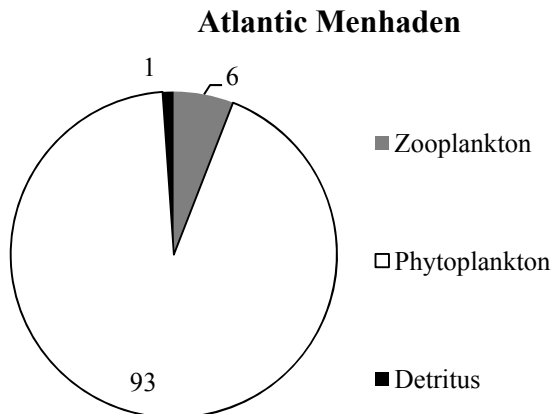
Black Seabass

Includes:
- *Centropristis striata*
- Unidentified Serranidae (Seabasses and Groupers)

Parameter Estimates: The black seabass stock biomass was set to 0.0200 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The limited number of studies available for black seabass necessitated the usage of the Fishbase website (Froese and Pauly 2005) to estimate the initial Ecopath parameters. Input values were P/B = 0.8 and Q/B = 6.9, assuming an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. As this stock was poorly utilized in the 2000s model (seen in the low EE when P/B = 0.8), the balancing process dictated a lowering of mortality was necessary. The final model set P/B = 0.3 as this value was thought on the border of possibility for the species. Since this stock was not fully utilized in the 2000s model, the mortality was calculated based on what was thought possible and likely did not provide an accurate estimate. Therefore, P/B was again left for Ecopath to estimate in the 1980s model. The stock biomass was estimated at 0.0087 t/km^2 from another GSB trawl survey carried out by Hanlon (1983). The Q/B (6.9) and EE (0.614) from the 2000s model were also input. Since biomass estimates were not available for the 1930s and 1880s models, the P/B estimated by the 1980s model (0.9192) was used. Coupled with the final 1980s Q/B and EE, biomass was left to be calculated by the program in both periods.

Diet: Dietary trends were estimated from the Hudson-Raritan estuary (Steimle et al. 2000). This survey was used for all four models.

Black Seabass
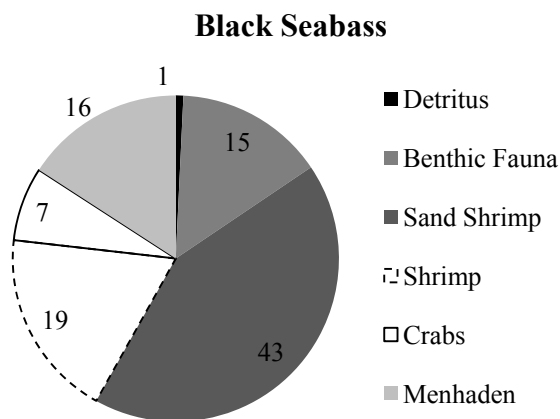
Blackfish/Tautog

Includes:
- *Tautoga onitis*

Parameter Estimates: Stock biomass for the 2000s model was set at 0.1923 t/km^2 based on data from a 2007 trawl survey (Frisk and Munch 2008). The initial Ecopath inputs for blackfish came from a Chesapeake Bay study (Christensen et al. 2005). Although the grouping from this study (reef associated fish) represented multiple species of finfish, the authors assumed blackfish were representative of the entire group so the estimated Ecopath parameters were derived solely from blackfish data, justifying their usage in our model. These estimates placed P/B = 0.55 and Q/B = 3.1. However, the high yields from this stock adjusted the 2000s P/B of blackfish to be 1.211. As an adjustment was required for the P/B measure in the 2000s model, the accuracy of our initial mortality measure was uncertain and left out of the initial 1980s inputs. Instead, the 2000s EE estimate (0.9996) was input, allowing the model to estimate the 1980s P/B (1.5506). Although a 1980s biomass of 0.7282 t/km^2 was estimated by Hanlon (1983), the high fishery yield of blackfish in the 1980s required the biomass to be raised (1.5 t/km^2) until a reasonable P/B was estimated (1.551). The stock biomass of blackfish for the 1930s was estimated (0.0012 t/km^2) from Dickinson (1938) and Westman (1938). Using the 1980s Q/B (3.1) and EE (0.9996) inputs, P/B was left to be calculated. As no data pertaining to the fishery landings, migrational patterns, and biomass accumulation of blackfish were found in the literature, both biomass and P/B estimates were necessary inputs in the 1880s model. As little data accounted for the abundance of the blackfish stock before the 1930s, we assumed this stock was relatively not targeted before this period and estimated the 1880s abundance was similar to that in the 1930s (0.0012 t/km^2). P/B (0.0267) and Q/B (3.1) values from the 1930s model were also used, allowing the program to estimate EE.

Diet: Estimates of the dietary habits of blackfish for the 2000s model were supplied by Steimle et al. (2000) in the Hudson-Raritan system. The 1980s model used trends provided across Long Island, New York (Olla et al. 1975). The unidentified fish category was attributed to detritus due to the benthic nature of other prey items. The 1980s trends were also used for the 1930s and 1880s.

Blackfish

2000s

- Sand Shrimp
- Crabs
- Hard Clams
- Suspension Feeders
- Detritus
- Blue Crabs

Blackfish

1980s, 1930s, & 1880s

- Suspension Feeders
- Benthic Fauna
- Crabs
- Detritus
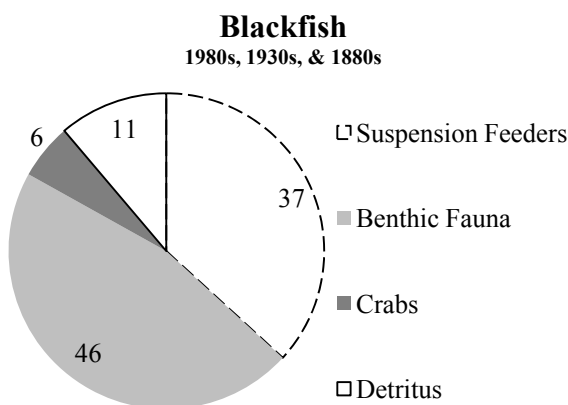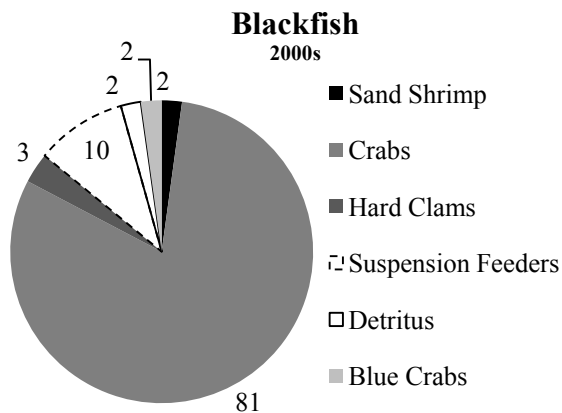
Young-of-the-Year (YOY) Bluefish (Snappers) and Adult Bluefish

Includes:

- *Pomatomus saltatrix*

YOY Bluefish are defined as Bluefish that are no more than one year old while Adult Bluefish are representative of Bluefish aged one year or more.

Parameter Estimates: Frisk and Munch (2008) provided data that calculated bluefish biomass for the 2000s model (0.2023 t/km^2). However, this value was believed inaccurate based on the mortality that would be necessary to balance out this biomass with the yield from the fishery. Therefore, the 2000s biomass was increased until a reasonable P/B ratio was obtained; biomass was set to 0.8 t/km^2, yielding a P/B of 1.061. The mortality estimated from age-frequency data in Frisk and Munch (1.073) supported this value. This procedure required Q/B and EE to be input for the 2000s model. These values were obtained from both the Delaware Bay (Frisk et al. in review) and Chesapeake Bay (Christensen et al. 2005). The Q/B of YOY bluefish from Frisk et al. (in review) (15.4) was thought more accurate than adult bluefish (3.1) so as to incorporate the large consumption of these stocks during the spawning season. Indeed, 88% of the bluefish landed in Frisk and Munch (2008) were determined to be YOY. The EE from Christensen et al.(2005) (0.821) was readjusted to 0.999 owing to the magnitude of

landings extracted from this stock. The 1980s biomass estimate (0.1080 t/km^2) was provided by data from Hanlon (1983). However, similar to the 2000s model, the yield from the fishery suggested an increase in bluefish biomass was needed. The 1980s biomass was increased to 1.9 t/km^2, yielding a P/B of 1.071. Although data used from Nyman and Conover (1988) yielded a P/B estimate of 0.518 for bluefish from 1985 and 1986, this study focused solely on YOY bluefish and may have missed a large portion of the mortality on older individuals. The Q/B (15.4) and EE (0.999) values from the 2000s model were also used in the 1980s model. Biomass estimates were available for the remaining two periods, allowing an estimate of P/B to be calculated from inputs of Q/B (15.4) and EE (0.999). A 1930s biomass of 0.4578 t/km^2 was computed from catch data by Dickinson (1938) and Westman (1938). Using landings trends given by McHugh (1972), the 1930s biomass was scaled to give a 1880s abundance estimate (4.8837 t/km^2).

Diet: Dietary trends were provided by two studies. YOY Bluefish was provided by a dietary study in two Long Island bays: Shinnecock Bay and Port Jefferson Harbor (Sagarese 2009). Adult Bluefish were estimated from Navesink, New Jersey (Scharf et al. 2004). The balancing procedure required a reduction in the weakfish and blue crab components for the 1980s. This same reduction was applied to the 1930s and 1880s inputs.

Bluefish
1980s, 1930s & 1880s

Legend:
- ■ Menhaden
- ■ Forage Fish
- Weakfish
- Sea Robins
- ⊡ Winter Flounder
- Bluefish
- Northern Pipefish
- American Eel
- Scup
- Tropical Fish
- Benthic Fauna
- ≡ Sand Shrimp
- Shrimp
- □ Blue Crab
- ■ Crabs
- Detritus

Pie values: 1, 4, 1, 3, 38, 50
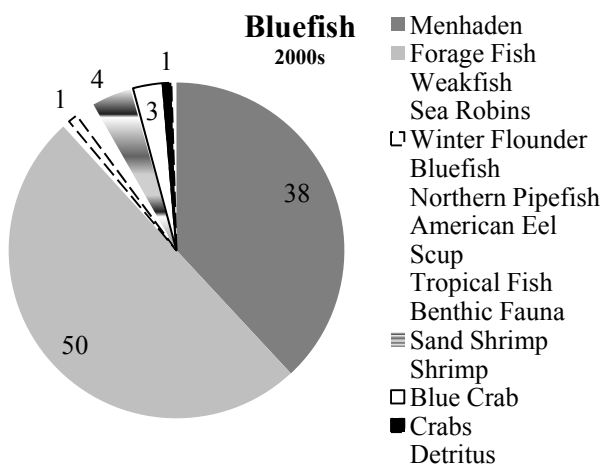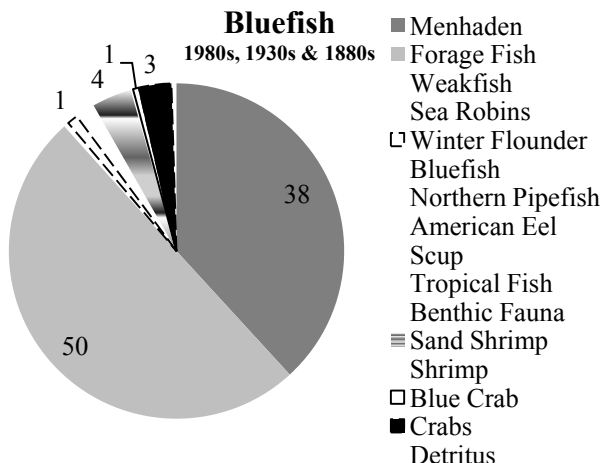
Butterfish

Includes:

- *Peprilus triacanthus* (American Butterfish)
- *Peprilus* sp.

Parameter Estimates: Butterfish biomass was set to 0.0268 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). Initial Ecopath estimates were provided by Okey (2001) in the Middle Atlantic Bight and included a P/B = 2.2 and Q/B = 5.5. This Q/B was used as an initial input for all four models. The balancing process of the 2000s model adjusted the P/B to 0.8, which was also used in each model. With the biomass estimate available from Hanlon 1983 (0.0003 t/km^2) thought too low, the 2000s EE (0.1420) was used in the 1980s model allowing the abundance measure to be calculated by Ecopath. The 1930s biomass of 0.2597 t/km^2 was computed from catch data by Dickinson (1938) and Westman (1938). Using qualitative trends given in Neville et al. (1938), the 1930s biomass was scaled to give a 1880s abundance estimate (0.1113 t/km^2). The 1930s and 1880s EE were left to be estimated by the program.

Diet: The dietary trends of butterfish for all four models were sampled from Cape Hatteras, North Carolina (Bowman et al. 2000).

Butterfish

- Zooplankton
- Suspension Feeders
- Benthic Fauna
- Mantis Shrimp
- Shrimp
- Ctenophores
- Detritus

21
46
29
2 2
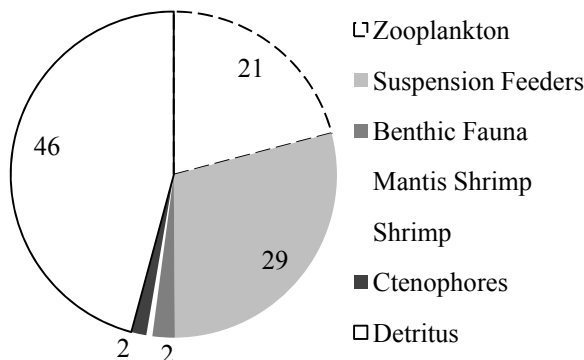
Cunner

Includes:

- *Tautogolabrus adspersus*

Parameter Estimates: A biomass estimate for the 2000s model was unavailable for cunner, owing to the absence of their capture throughout the Frisk and Munch (2008) study. The initial estimates for this grouping came from outputs of the Fishbase function (Froese and Pauly 2005) and the Gulf of Maine (Zhang and Chen 2007). Initial values included P/B = 0.48 and Q/B = 7.5 from Froese and Pauly (2005) and EE = 0.99 from Zhang and Chen (2007). The Fishbase estimates assumed an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. Hanlon (1983) provided a biomass estimate (0.0167 t/km^2) of the stock for the 1980s model. The P/B was left for Ecopath to estimate as trends in mortality fluctuations of this stock were not found in the literature. Instead, the Q/B and EE from the 2000s model were used. The cunner group lacked biomass estimates for the 1930s and 1880s models, requiring the input of P/B, Q/B, and EE. The Q/B (7.5) and EE (0.99) inputs were taken from the values used in both the 2000s and 1980s models. However, there was a large variation in the P/B values of the 2000s (0.48) and 1980s (0.0032), complicating the decision of a reasonable mortality estimate for the 1930s and 1880s. As cunner have no fishery associated with them and little predation mortality was evident in the dietary inputs, the 1980s P/B estimate was used in both models.

Diet: Dietary estimates for all four models were combined from two separate studies. The predominant trends were taken from cunner collected throughout Long Island, New York (Olla et al. 1975) and Great South Bay, New York (Schreiber 1973). Unfortunately, Olla et al. (1975) did not distinguish between the different species of fish they sampled. Schreiber (1973) noted that cunner fed on multiple species of finfish: namely silversides, mummichugs, sticklebacks, and pipefish. Therefore, the unidentified fish category from Olla et al. was divided amongst forage fish (75%) and northern pipefish (25%) in the model. This study was used for all four models.

Cunner
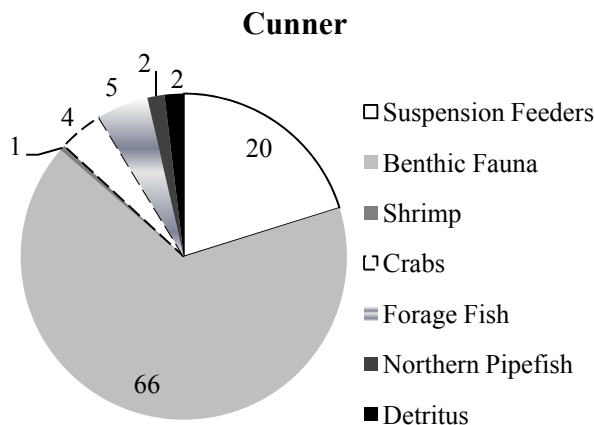
Dogfish

Includes:
- *Mustelus canis* (Smooth Dogfish)
- *Squalus acanthias* (Spiny Dogfish)

Parameter Estimates: The dogfish biomass for the 2000s model was set to 0.0953 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial Ecopath estimates for this grouping came from the Middle Atlantic Bight (Okey 2001). Initial values used were Q/B = 4.77 and EE = 0.95, leaving P/B to be estimated by the software. A stock biomass of 0.0076 t/km^2 was input based on trawling data from Hanlon (1983). However, as little statistics were available regarding the fishing and natural mortality of this group, a P/B was required to be input by the program. Therefore, the 2000s P/B estimate (0.04723) was input, leaving EE blank. Since landings data was unavailable for the 1930s and 1880s systems, no direct measure of abundance was able to be calculated for this grouping. However, biomass was set to 0.1 t/km^2 based on three abundance patterns: the absence of fishing effort and predators on dogfish in both the 2000s and 1930s suggested similar biomass trends may have been present; the decline in crab abundance (prey) estimated between the 1930s and 1980s models indicated dogfish were at a higher abundance in the 1930s; and large cartilaginous predators (sandbar sharks) were present in GSB during the early 1900s, suggesting a "healthy" system capable of supporting stocks at high trophic levels. Similar to the 1980s model inputs, a P/B input was required to balance the dogfish grouping in the 1930s and 1880s models. Therefore, the 2000s estimate of P/B (0.0472) was used, leaving EE to be calculated by the models.

Diet: Dietary trends were obtained using data from two separate studies. The predominant trends were collected in the Hudson-Raritan estuary (Steimle et al. 2000). Unfortunately, Steimle et al. (2008) did not distinguish between the different species of fish they sampled. Schreiber (1973) noted that the finfish diet of smooth dogfish was composed of silversides (70%), sea horses (5%), rainbow smelt (15%), and sea robins (10%). Therefore, the "unidentified fish" category from Steimle et al. was divided

amongst forage fish (70%), tropical fish (20%), and sea robins (10%) from these respective ratios. These trends were used in the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Dogfish
2000s & 1980s

10 9 2 Shrimp
7 Forage Fish
Suspension Feeders
15 Hard Clams
Tropical Fish
Crabs
58 Sea Robins
Detritus

Dogfish
1930s

10 9 2 Shrimp
7 Forage Fish
Suspension Feeders
11 Hard Clams
4 Oysters
Tropical Fish
Crabs
58 Sea Robins
Detritus

Dogfish
1880s

10 9 2 Shrimp
7 Forage Fish
Suspension Feeders
4 Hard Clams
11 Oysters
Tropical Fish
Crabs
58 Sea Robins
Detritus
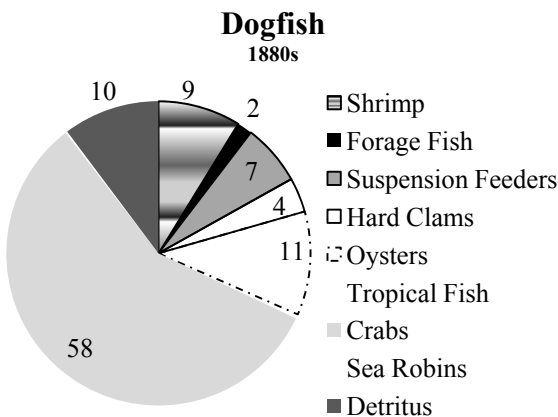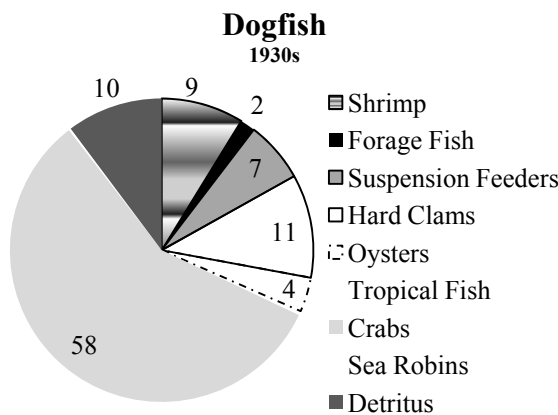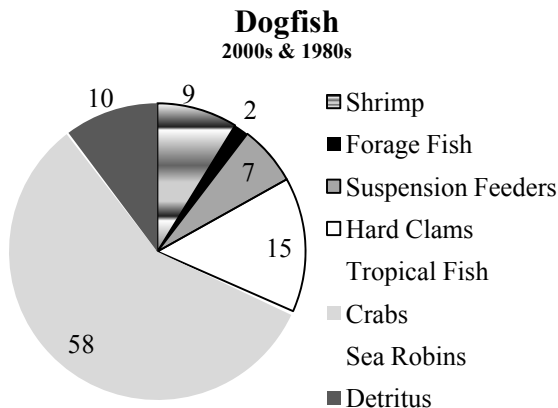
Drums and Croakers

Includes:
- *Pogonias cromis* (Black Drum)
- *Leiostomus xanthurus* (Spot)
- *Larimus* sp.
- *Umbrina* sp.
- *Micropogonias undulatus* (Atlantic Croaker)
- *Micropogonias* sp.
- Unidentified Sciaenidae

Parameter Estimates: Stock biomass was set to 0.2735 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial estimates for P/B and Q/B came from averages of three Sciaenids in Delaware Bay (Frisk et al. in review). The initial values of P/B = 0.93 and Q/B = 3.9 were used. The balancing process of the 2000s recalculated the P/B at 0.4. As no biomass estimates were available for subsequent years, the P/B (0.4), Q/B (3.9), and EE (0.4971) estimates from the final 2000s model were used in the 1980s, 1930s, and 1880s models.

Diet: Dietary trends were combined from two separate studies. Data for Atlantic croaker, *Micropogonias undulatus*, and spot, *Leiostomus xanthurus*, was obtained from Delaware Bay (Nemerson and Able 2004) while the diet of the Black Drum *Pogonias cromis* was estimated from southern Texas (Street et al. 1997). These trends were used for the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.
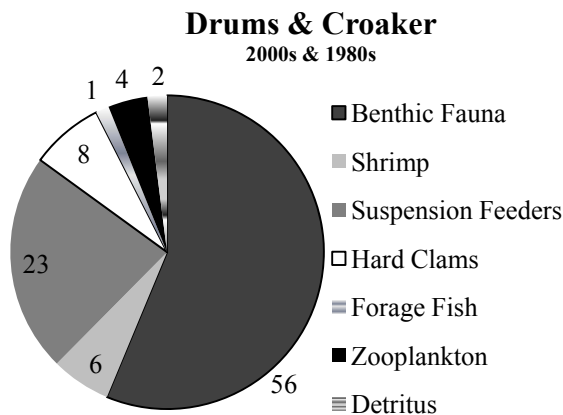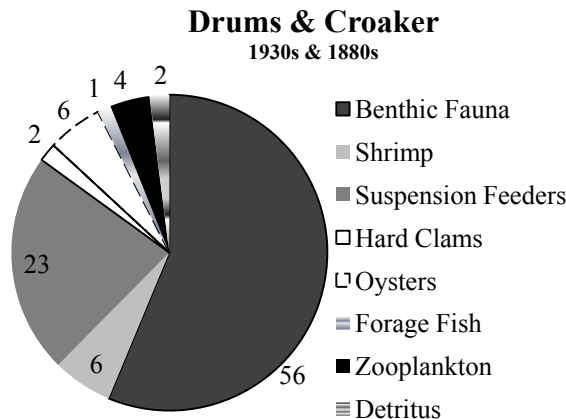
Drums & Croaker
2000s & 1980s

Benthic Fauna, Shrimp, Suspension Feeders, Hard Clams, Forage Fish, Zooplankton, Detritus

56, 6, 23, 8, 1, 4, 2

Drums & Croaker
1930s & 1880s

Legend:
- ■ Benthic Fauna
- ▨ Shrimp
- ▦ Suspension Feeders
- ☐ Hard Clams
- ⊔ Oysters
- ▤ Forage Fish
- ■ Zooplankton
- ▤ Detritus

Pie values: 1, 4, 2, 6, 2, 23, 6, 56

Flatfish – Other

Includes:
- *Citharichthys* sp. (Sanddabs)
- *Etropus crossotus* (Fringed Flounder)
- *Etropus microstomus* (Smallmouth Flounder)
- *Glyptocephalus cynogiossus* (Witch Flounder or Torbay Sole)
- *Pleuronectes ferruginea* (Yellowtail Flounder)
- *Syacium papillosum* (Dusky Flounder)
- Soleidae (Soles)
    - *Trinectes maculates* (Hogchoker)
- Unidentified Bothidae (Left-eyed Flounders)
- Unidentified Pleuronectidae (Right-eyed Flounders)
    - Excluding Paralichthyidae

Parameter Estimates: The stock biomass for the flatfish-other grouping was set to 0.0073 t/km^2 based on data from the Frisk and Munch (2008) benthic trawl in 2007. Ecopath estimates were available from two sources: Fishbase (Froese and Pauly 2005) and the Chesapeake Bay (Christensen et al. 2005). The Fishbase function provided a Q/B of 7.033 based on averages over hogchoker, yellowtail flounder, and dusky flounder. This assumed an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. Christensen et al. estimated an initial EE at 0.95, leaving P/B to be calculated. The 1980s biomass (0.0308 t/km^2) was estimated by Hanlon (1983) in a trawl conducted in GSB. Coupled with the Q/B and EE above, P/B was again left to be estimated by Ecopath. Initial P/B inputs were required for the 1930s and 1880s models as abundances were unavailable. The balancing process of the 1980s model set the P/B to 0.4297, which was used for the 1930s and 1880s models.

Diet: Dietary estimates were combined from a single study in which data was collected for three separate species. Data for the yellowtail flounder, *Pleuronectes ferruginea*, witch flounder, *Glyptocephalus cynogiossus*, and dusky flounder, *Syacium*

papillosum, were obtained from Cape Hatteras, North Carolina (Bowman et al. 2000). While the 2000s model used the original data, the balancing process of the 1980s model dictated a reduction in the gadidae category. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to ha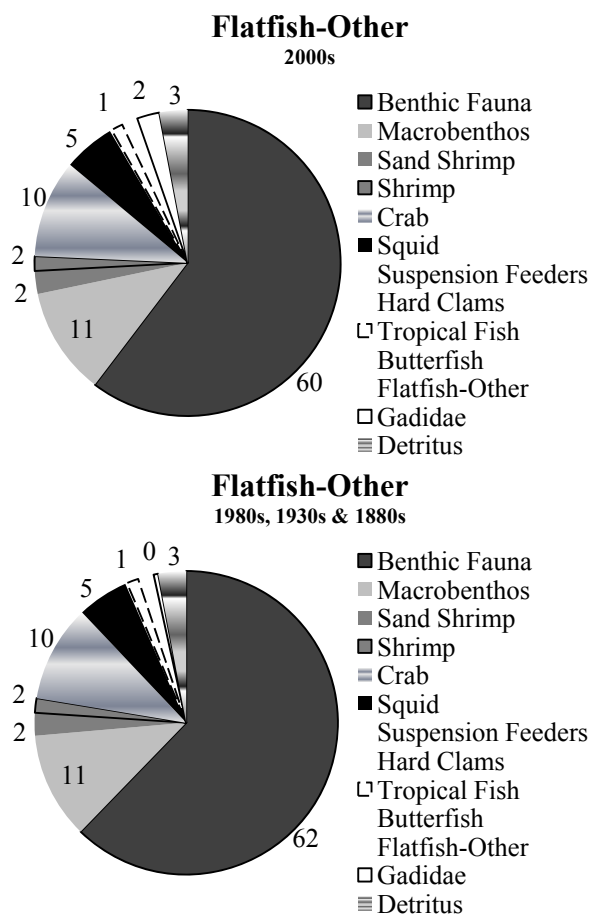rd clams while the remaining 75% were fed into oysters. However, as only 0.0141% of the overall diet was switched, there was no visible change in the flatfish diet and no additional charts were provided below.

Flatfish-Other

2000s

Legend
■ Benthic Fauna
■ Macrobenthos
■ Sand Shrimp
■ Shrimp
≡ Crab
■ Squid
Suspension Feeders
Hard Clams
⊔ Tropical Fish
Butterfish
Flatfish-Other
□ Gadidae
≡ Detritus

Flatfish-Other

1980s, 1930s & 1880s

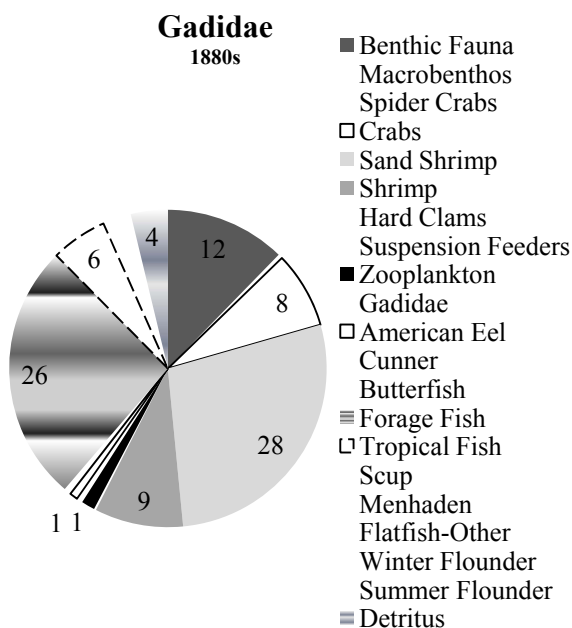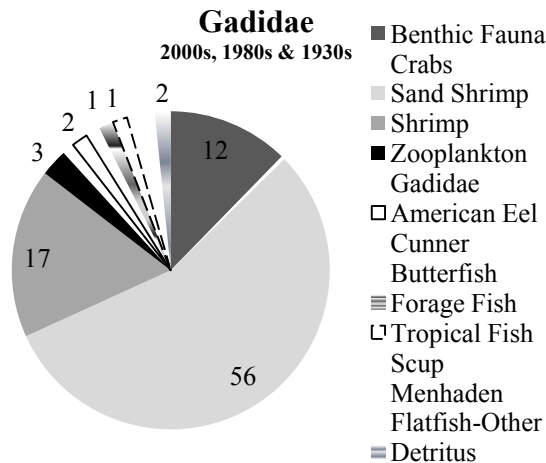| Legend |
| --- |
| ■ Benthic Fauna |
| ■ Macrobenthos |
| ■ Sand Shrimp |
| ■ Shrimp |
| ≡ Crab |
| ■ Squid |
| Suspension Feeders |
| Hard Clams |
| ⊔ Tropical Fish |
| Butterfish |
| Flatfish-Other |
| □ Gadidae |
| ≡ Detritus |

Gadidae

Includes:
- *Microgadus tomcod* (Atlantic Cod)
- *Pollachius virens* (Pollock)
- *Merluccius bilinearis* (Silver Hake or Whiting)
- *Urophycis chuss* (Red Hake or Ling)
- *Urophycis regia* (Spotted Hake)
- *Urophycis* sp.

Parameter Estimates: The stock biomass for the 2000s model was set to 0.0080 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial Ecopath estimates were provided by three gadids from the Gulf of Maine (Zhang and Chen 2007) and one gadid from Delaware Bay (Frisk et al. in review). Average input values included Q/B = 3.66, and EE = 0.68. However, the frequency with which gadids appeared in the diets of other groupings suggested that an EE of 0.68 was too low; we increased the initial estimate to 0.999. These values were used as initial estimates for all four models. The 1980s system was believed to hold 0.0040 t/km^2 of gadids based on the GSB survey of Hanlon (1983). The 2000s Q/B and EE estimates were put into the initial 1980s model, allowing the program to calculate P/B. A 1930s stock biomass of 0.0033 t/km2 was based on commercial trawl catch of silver hake supplied by Dickinson (1938). With no biomass estimate computed for the 1880s model, the P/B = 1.3546 from the 1930s model was used for the 1880s.

Diet: Dietary trends for Gadidae were averaged from three different species found in the Hudson-Raritan estuary. Data was obtained for red hake, *Urophycis chuss*, silver hake *Merluccius bilinearis*, and spotted hake, *Urophycis regia*, from Steimle et al. (2000). However, all three gadids from Steimle et al. (2000) had an unidentified fish category, accounting for over 8% of the averaged Gadidae diet. A silver hake diet in Schreiber (1973) and the identified finfish in Steimle et al. (2000) determined what ratio of the unidentified fish component should be attributed to each Ecopath group. Forage fish and tropical fish each received 18.2% while cunner, butterfish, menhaden, gadidae, other flatfish, and American eel received 9.1%. Note that some of the final dietary contributions to the Gadidae diet were higher than those expected from the above percentages (greater than 9.1 or 18.2 percent). As some groups (gadidae, American eel, unidentified flatfish) were identified in Steimle et al. (2000), the contribution from the unidentified category simply augmented the dietary contribution of the group that was already present. These trends were used for the 2000s, 1980s, and 1930s model. The 1880s model used a combination of these trends and the addition of diet data from Atlantic cod, *Gadus morhua*, from southern New England (Langton and Bowman 1980) as fishing records indicated this stock was important during this period.

Gadidae
2000s, 1980s & 1930s

■ Benthic Fauna
  Crabs
□ Sand Shrimp
■ Shrimp
■ Zooplankton
  Gadidae
□ American Eel
  Cunner
  Butterfish
≡ Forage Fish
⌐ Tropical Fish
  Scup
  Menhaden
  Flatfish-Other
≡ Detritus

Gadidae
1880s

■ Benthic Fauna
  Macrobenthos
  Spider Crabs
□ Crabs
□ Sand Shrimp
■ Shrimp
  Hard Clams
  Suspension Feeders
■ Zooplankton
  Gadidae
□ American Eel
  Cunner
  Butterfish
≡ Forage Fish
⌐ Tropical Fish
  Scup
  Menhaden
  Flatfish-Other
  Winter Flounder
  Summer Flounder
≡ Detritus

Northern Kingfish

Includes:
- *Menticirrhus saltatrix*
- *Menticirrhus* sp. (Kingcroakers)

Parameter Estimates: The 2000s biomass of northern kingfish was set to 0.0680 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial estimates of P/B = 0.87 and Q/B = 5.9 were obtained from Fishbase (Froese and Pauly 2005) assuming an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. The balancing process of the 2000s model set the P/B to 0.3, which was used for the 1980s, 1930s, and 1880s models. Hanlon estimated the biomass of this stock at 0.0387 t/km^2 in 1983 while Dickinson (1938) and Westman (1938) supplied the data used for the biomass measure of

the 1930s (0.3229 t/km^2). Coupled with the P/B and Q/B values from the final 2000s model, EE was estimated for northern kingfish in the 1980s and 1930s. As an estimate of stock biomass was unavailable in the 1880s, all three Ecopath parameter estimates were required. P/B (0.3), Q/B (5.9), and EE (0.414) were input into the 1880s based on values measured in the final 1930s model.

Diet: An estimate of the dietary habits of Northern Kingfish was obtained from samples in Cape Hatteras, North Carolina (Bowman et al. 2000). These were used for all four models.

Northern Kingfish

Pie chart legend:
- Benthic Fauna
- Sand Shrimp
- Mantis Shrimp
- Shrimp
- Crabs
- Zooplankton
- Tropical Fish
- Forage Fish
- Detritus

Values shown: 3, 6, 2, 5, 6, 1, 7, 69
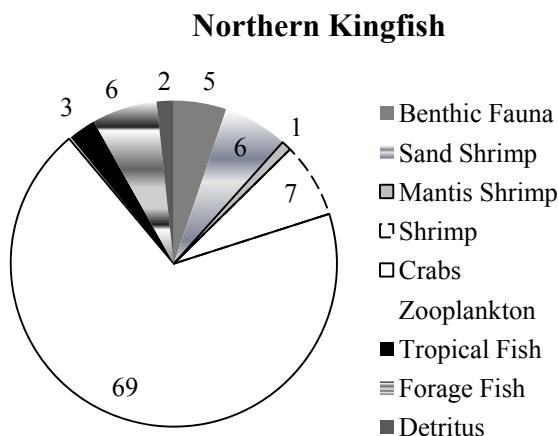
Northern Pipefish

Includes:

- *Syngnathus fuscus*
- *Syngnathus* sp.

Parameter Estimates: The CPUE indices estimated for pipefish (0.0034 t/km^2 in the 2000s and 0.0045 t/km^2 in the 1980s) were thought inaccurate due to the ability of thin pipefish to slide through a trawl net's mesh. Therefore, abundance was left for the program to calculate, necessitating the acquisition of three Ecopath parameter estimates. The initial estimates for this grouping came from the Fishbase website (Froese and Pauly 2005). However, an estimate of time at maturity was needed to use the Fishbase function. Data from Ripley and Foran (2006) was used to estimate $t_{maturity} = 0.312$ years. Assuming an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C, P/B (2.75) and Q/B (6.9) were calculated. The balancing process of the 2000s model set the P/B to 1.3, which was used for the 1980s, 1930s, and 1880s models. An initial estimate of EE (0.999) was used.

Diet: The dietary trends used in all four models were obtained from a study done in Cape Hatteras, North Carolina (Bowman et al. 2000). This data was input for all four periods.
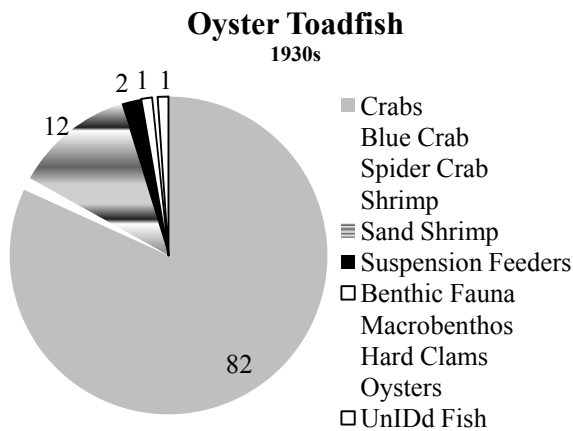
Northern Pipefish
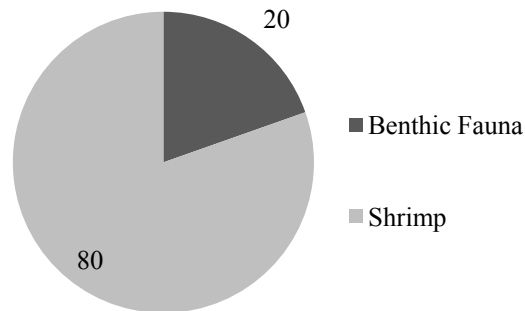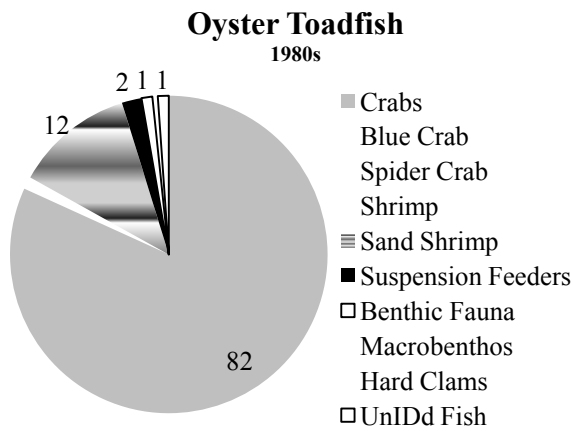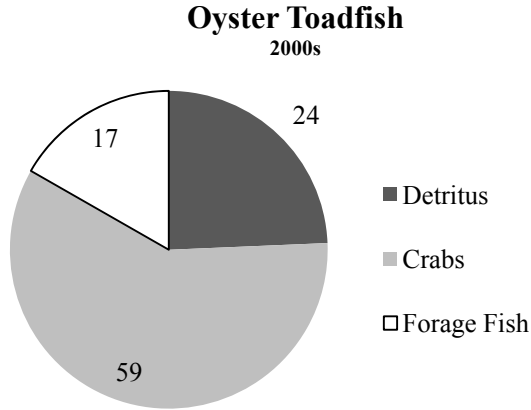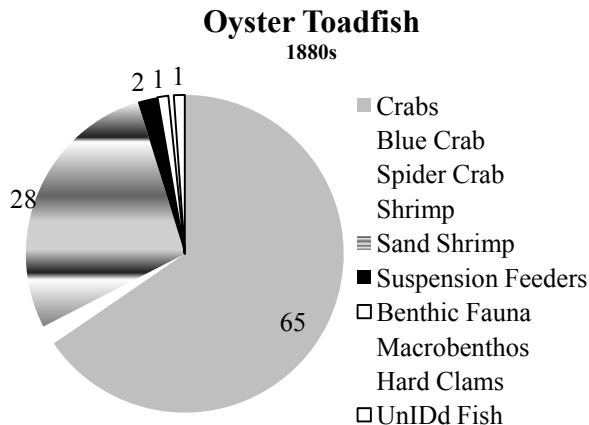

Oyster Toadfish

Includes:

- *Opsanus tau*
- *Porichthys* sp.
- Unidentified Batrachoididae (Toadfish)

Parameter Estimates: The 2000s oyster toadfish biomass was set to 0.0038 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial Ecopath parameter estimates came from the Fishbase equation (Froese and Pauly 2005). As we assumed an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C, a P/B = 0.36 and Q/B = 6.2 was calculated. In the 1980s, Hanlon (1983) estimated this same stock at a biomass of 0.4431 t/km^2. The P/B and Q/B estimates from the final 2000s model were also input, leaving EE to be calculated by Ecopath. As literature on the fishery, migration, or accumulation of biomass of oyster toadfish was lacking, the Ecopath program required an abundance estimate to be directly input. However, abundance data was unavailable for toadfish in the 1930s and 1880s systems. While prey was more abundant (namely crabs) in GSB during the 1930s and 1880s, toadfish biomass in the 1930s and 1880s was expected to be higher than that found in the 1980s. The biomass was arbitrarily raised to 0.6 t/km^2 (26.15%) and was considered reasonable.

Diet: Dietary habits for the 2000s were estimated from samples obtained from Shinnecock Bay and Port Jefferson Harbor (Sagarese 2009). However, almost 17% of the diet was classified as unidentified fish. An overview of oyster toadfish in Collette (2002) documented predation on various species of fish fry. As such, the unidentified fish category from Sagarese (2009) will be classified as forage fish. The trends used in the 1980s and 1930s were supplied by samples from Delaware Bay (McDermott 1965). The unidentified fish was attributed to forage fish based on data from Wilson et al. (1982). The 1880s model required a reduction in forage fish. Furthermore, the 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were

fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Oyster Toadfish
2000s

24

17

59

■ Detritus

■ Crabs

□ Forage Fish

Oyster Toadfish
1980s

2 1 1

12

82

■ Crabs
  Blue Crab
  Spider Crab
  Shrimp
≡ Sand Shrimp
■ Suspension Feeders
□ Benthic Fauna
  Macrobenthos
  Hard Clams
□ UnIDd Fish

Oyster Toadfish
1930s

2 1 1

12

82

■ Crabs
  Blue Crab
  Spider Crab
  Shrimp
≡ Sand Shrimp
■ Suspension Feeders
□ Benthic Fauna
  Macrobenthos
  Hard Clams
  Oysters
□ UnIDd Fish

Oyster Toadfish
1880s

Crabs
  Blue Crab
  Spider Crab
  Shrimp
Sand Shrimp
Suspension Feeders
Benthic Fauna
  Macrobenthos
  Hard Clams
UnIDd Fish

Scup

Includes:
- *Stenotomus chrysops*
- Unidentified Sparidae (Porgies)

Parameter Estimates: Scup biomass was set to 0.2690 t/km^2 in the 2000s based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial estimates for this grouping came from the Middle Atlantic Bight (Okey 2001) in which a Q/B = 5.5 and EE = 0.95 was used. While the estimated P/B for the 2000s (0.636) was lower than that calculated from the catch data of Frisk and Munch (0.538), EE was raised to 0.999 to reduce the variation. The 1980s biomass of 0.0489 t/km^2 was estimated by Hanlon (1983). The Q/B and EE from the final 2000s model was also input, allowing a mortality estimate to be calculated. As no biomass estimate was found for the 1930s and 1880s models, the P/B, Q/B, and EE from the 1980s model were used. The balancing process of the 1980s model set the P/B to 0.606.

Diet: The estimate of the dietary habits of scup for all four models was provided by Sagarese (2009) from Shinnecock Bay and Port Jefferson Harbor. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.
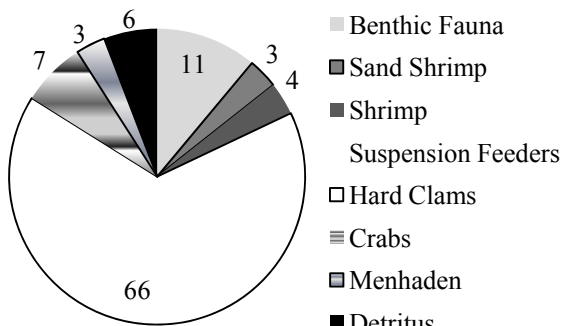
Scup
2000s & 1980s

- Benthic Fauna
- Sand Shrimp
- Shrimp
  Suspension Feeders
- Hard Clams
- Crabs
- Menhaden
- Detritus

Scup
1930s

- Benthic Fauna
- Sand Shrimp
- Shrimp
  Suspension Feeders
- Hard Clams
- Oysters
- Crabs
- Menhaden
- Detritus

Scup
1880s

- Benthic Fauna
- Sand Shrimp
- Shrimp
  Suspension Feeders
- Hard Clams
- Oysters
- Crabs
- Menhaden
- Detritus
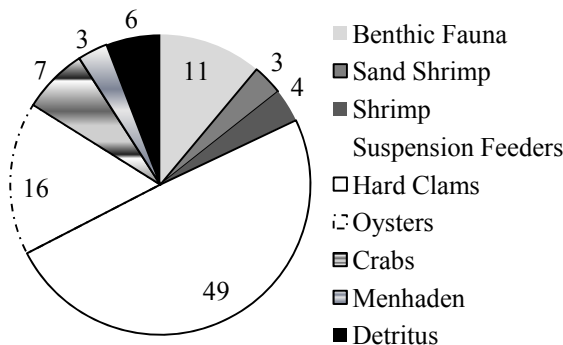
Young-of-the-Year (YOY) and Adult Striped Bass

Includes:
- *Morone saxatilis*

YOY Striped Bass are defined as bass that are no more than one year old while Adult Striped Bass are representative of bass aged one year or more.

Parameter Estimates: The initial estimates for this grouping came from Delaware Bay (Frisk et al. in review) in which Ecopath inputs included P/B = 0.49 and Q/B = 2.86. EE was estimated at 0.9. The balancing process of the 2000s model adjusted the Q/B to 2.45, which was used for the 1980s, 1930s, and 1880s models. The same set of inputs (P/B, Q/B, and EE) was used to model the striped bass stock in the 1980s. The 1930s stock biomass estimate of 0.0678 t/km^2 was computed from the data provided by Dickinson (1938) and Westman (1938). Using qualitative trends given in Neville et al. (1938), the 1930s biomass was scaled to give a 1880s abundance estimate (0.1130 t/km^2). Using these biomasses and the 1980s values of Q/B and EE, the P/B ratio was left to be estimated by the Ecopath program in the 1930s and 1880s models.

Diet: Dietary trends for striped bass were estimated from multiple studies. The predominant trends were estimated from Shinnecock Bay and Port Jefferson Harbor (Sagarese 2009), the Hudson-Raritan estuary (Steimle et al. 2000), the western bays of Long Island (Buckel and McKown 2002), and Delaware Bay (Nemerson and Able 2003). However, as limited predation had been observed on bluefish throughout the literature research, data from Walter and Austin in the Chesapeake (2003) was also included so as to incorporate some predation on YOY and adult Bluefish. Only the contribution of Bluefish to the diet was used from Walter and Austin. Using these studies as a framework, a reduction in the gadidae component was required during the balancing process in the 2000s model. A further reduction in gadidae was deemed necessary when finalizing the 1980s model. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the 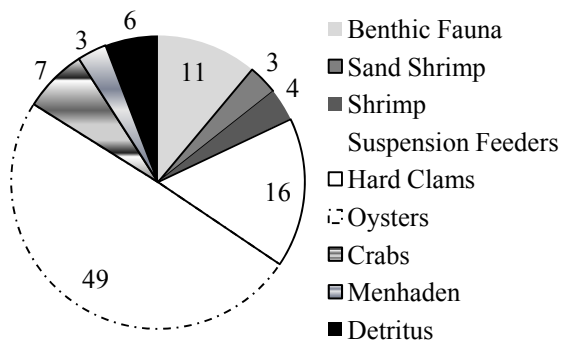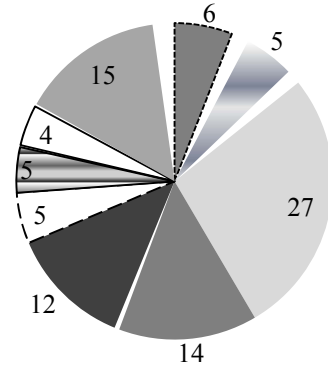remaining 75% were fed into oysters. However, as only 0.166% of the overall striped bass diet was effected by the inclusion of oysters, no visible change was evident in the pie charts between the 1930s and 1880s and only one figure was given.

Striped Bass
2000s

Benthic Fauna
  Macrobenthos
  Crabs
Blue Crabs
  Horseshoe Crabs
Sand Shrimp
Shrimp
  Zooplankton
  Ctenophores
  Suspension Feeders
  Hard Clams
Forage Fish
Tropical Fish
Sea Robins
  Northern Kingfish
Drums & Croakers
  Flatfish-Other
Summer Flounder
  Winter Flounder
  Gadidae
  Adult Bluefish
  YOY Bluefish
  Detritus

Striped Bass
1980s
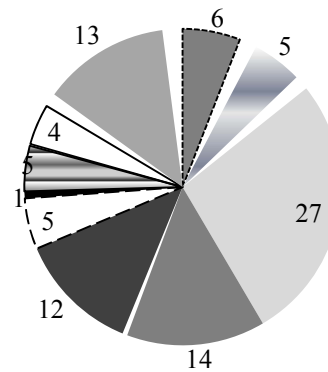
Benthic Fauna
  Macrobenthos
  Crabs
Blue Crabs
  Horseshoe Crabs
Sand Shrimp
Shrimp
  Zooplankton
  Ctenophores
  Suspension Feeders
  Hard Clams
Forage Fish
Tropical Fish
Menhaden
Sea Robins
  Northern Kingfish
Drums & Croakers
  Flatfish-Other
Summer Flounder
  Winter Flounder
  Gadidae
  Bluefish
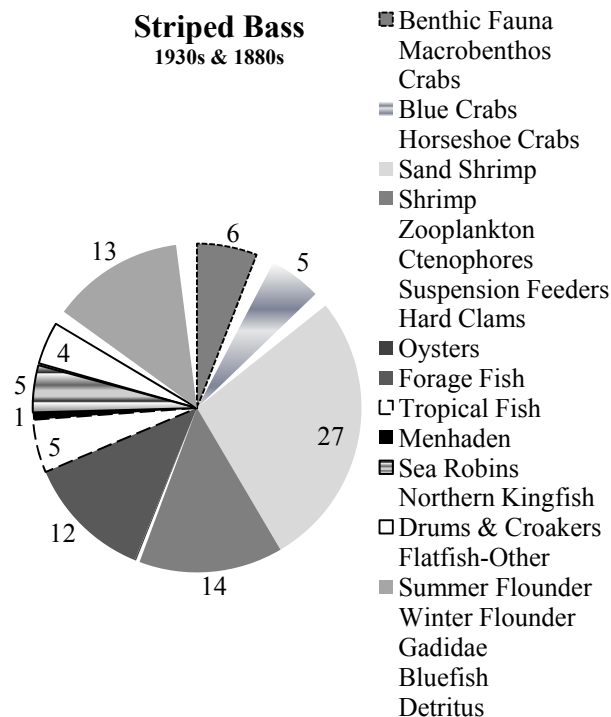 Detritus

Striped Bass
1930s & 1880s

Legend:
- Benthic Fauna
  Macrobenthos
  Crabs
- Blue Crabs
  Horseshoe Crabs
- Sand Shrimp
- Shrimp
  Zooplankton
  Ctenophores
  Suspension Feeders
  Hard Clams
- Oysters
- Forage Fish
- Tropical Fish
- Menhaden
- Sea Robins
  Northern Kingfish
- Drums & Croakers
  Flatfish-Other
- Summer Flounder
  Winter Flounder
  Gadidae
  Bluefish
  Detritus
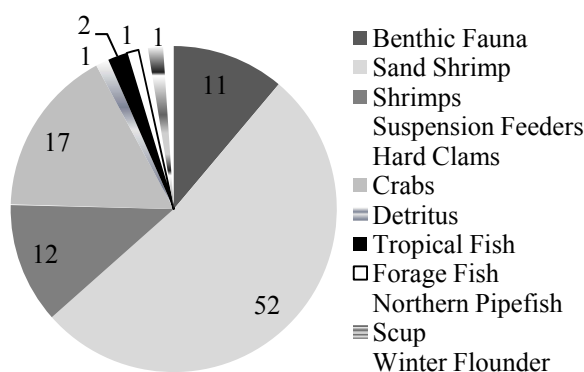
Sea Robins

Includes:
- *Prionotus carolinus* (Northern Searobin)
- *Prionotus evolans* (Striped Searobin)
- *Prionotus* sp.
- Unidentified Cottidae (Sculpins)

Parameter Estimates: Stock biomass was set to 0.1966 t/km^2 based on the benthic trawl survey of Frisk and Munch in 2007 (2008). Ecopath estimates for this grouping were obtained from Fishbase (Froese and Pauly 2005) using the assumptions of an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. Average values were used from both striped and northern sea robins and included a P/B = 0.45 and Q/B = 7.15. This P/B value was adjusted to 0.43 during the balancing procedure of the 2000s model. Hanlon (1983) estimated the sea robin biomass used in the 1980s model at 0.1221 t/km^2. Input with the Q/B (7.15) and P/B (0.43) estimates of the 2000s, the program was left to estimate the EE of searobins. As no biomass measures were available for the 1930s and 1880s systems, the EE estimated from the balanced 1980s model (0.986) was used with P/B and Q/B values for the 1930s and 1880s models.

Diet: Dietary trends were combined from two separate studies. Data for the striped sea robin, *Prionatus evolans*, was obtained from Shinnecock Bay and Port Jefferson Harbor (Sagarese 2009). Steimle et al. (2000) provided diet data for northern sea robins, *Prionatus carolinus*, in the Hudson-Raritan system. The unidentified fish

category from Steimle et al. (2000) was divided among the finfish found in Sagarese (2009): tropical fish, forage fish, northern pipefish, scup, and winter flounder. These inputs were used for the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters. However, as only 0.0202% of the total diet was altered, no visible change was noticed in the pie chart and only one figure was given.

Sea Robins

Pie chart legend:
- Benthic Fauna
- Sand Shrimp
- Shrimps
  Suspension Feeders
  Hard Clams
- Crabs
- Detritus
- Tropical Fish
- Forage Fish
  Northern Pipefish
- Scup
  Winter Flounder

Values shown: 11, 52, 12, 17, 2, 1, 1, 1

Summer Flounder

Includes:
- *Paralichthys dentatus*
- *Paralichthys* sp.
- Unidentified Paralichthyidae (Large-tooth Flounders)

Parameter Estimates: Frisk and Munch (2008) provided data that estimated summer flounder biomass for the 2000s model (0.1923 t/km^2). However, the considerable landings harvested from this species suggested this estimate was too low. Therefore, the 2000s biomass was left to be estimated by Ecopath. As three Ecopath parameters were required, initial estimates for this grouping were taken from Fishbase (Froese and Pauly 2005), Chesapeake Bay (Christensen et al. 2005), and a summer flounder mortality study (Weber 1984). The Fishbase estimate of Q/B (4.0) was calculated assuming an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. Weber (1984) placed summer flounder mortality in the 1980s at 0.88 (P/B). However, the persistence of summer flounder in fishery landings and noted declines in the stock suggested an increase in mortality had occurred between the 1980s and 2000s. Therefore, the 2000s model inputs included a P/B = 1.0. EE was estimated at 0.95 from Christensen et al.

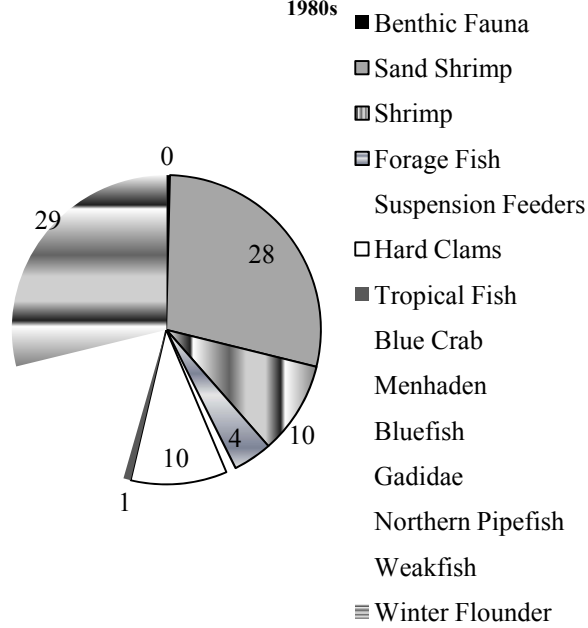(2005). The 1980s stock biomass (1.0215 t/km^2) was provided by data from Hanlon (1983). Using the mortality estimate from 1980s data in Weber (0.88) and Q/B from Fishbase (4.0), we allowed the Ecopath parameterization to calculate EE. The 1930s biomass measure (0.8758 t/km^2) was computed from the commercial trawling data of Dickinson (1938). A mortality estimate of summer flounder in the late 1950s (P/B = 0.6775) was given in Weber 1984 from data found in Poole 1962. Inputted with the Fishbase Q/B and Dickinson biomass, the Poole P/B estimate was scaled down to 0.4 during the balancing process owing to the low EE (when P/B = 0.678, EE = 0.095) being calculated by the program. However, even when mortality was placed at 0.4 for summer flounder, the EE still only measured at 0.161. Using qualitative trends given in Neville et al. (1938), the 1930s summer flounder biomass was scaled to give a 1880s abundance estimate (2.8706 t/km^2). The biomass was input with the Fishbase Q/B and 1930s EE estimate (0.161), leaving P/B for the model to estimate in the 1880s model.

Diet: Estimates of the dietary habits were provided by Sagarese (2009) from Shinnecock Bay and Port Jefferson Harbor. The 1980s model required a reduction in the gadidae and blue crab components during the balancing process. A further reduction in the gadidae component was applied to the 1930s model. Furthermore, the 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Summer Flounder

2000s — Benthic Fauna
Macrobenthos
Sand Shrimp
Shrimp
Mantis Shrimp
Squid
Forage Fish
Suspension Feeders
Hard Clams
Tropical Fish
Crab
Menhaden
Butterfish
Cunner
Gadidae
Northern Pipefish
Scup
Sea Robin
Weakfish
Winter Flounder
Detritus

Summer Flounder

1980s

- ■ Benthic Fauna
- ■ Sand Shrimp
- ▦ Shrimp
- ▨ Forage Fish
- Suspension Feeders
- ▢ Hard Clams
- ■ Tropical Fish
- Blue Crab
- Menhaden
- Bluefish
- Gadidae
- Northern Pipefish
- Weakfish
- ▤ Winter Flounder

Pie chart values: 0, 28, 10, 4, 10, 1, 29

Summer Flounder

1930s

- ■ Benthic Fauna
- ■ Sand Shrimp
- ▦ Shrimp
- ▨ Forage Fish
- Suspension Feeders
- ▢ Hard Clams
- ⊡ Oysters
- Tropical Fish
- Blue Crab
- Menhaden
- Bluefish
- Gadidae
- Northern Pipefish
- Weakfish
- ▤ Winter Flounder

Pie chart values: 0, 28, 10, 4, 8, 3, 29

Summer Flounder

1880s

- ■ Benthic Fauna
- ▣ Sand Shrimp
- ▦ Shrimp
- ▨ Forage Fish
-   Suspension Feeders
- ▢ Hard Clams
- ⊡ Oysters
-   Tropical Fish
-   Blue Crab
-   Menhaden
-   Bluefish
-   Gadidae
-   Northern Pipefish
-   Weakfish
- ≡ Winter Flounder

0

29   28

8   4   10

3
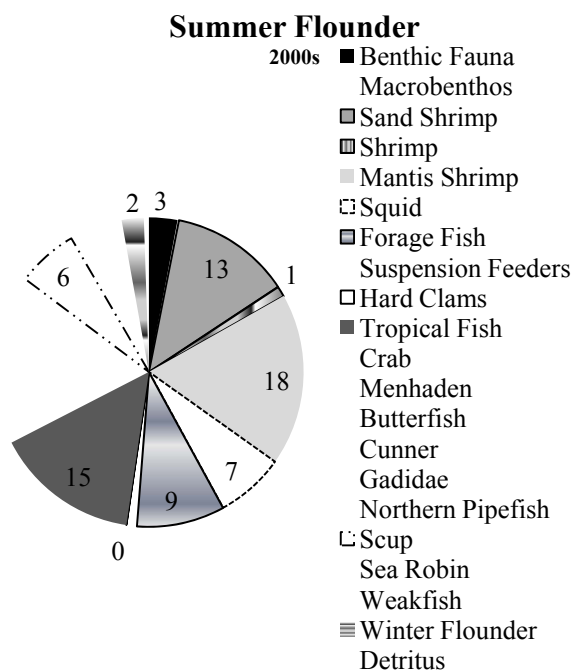
Weakfish

Includes:
- *Cynoscion regalis* (Gray Sea Trout)
- *Cynoscion* sp.

Parameter Estimates: The weakfish biomass was set to 0.1134 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). Length frequency da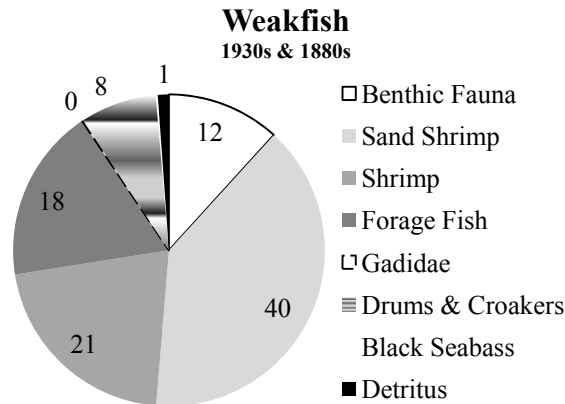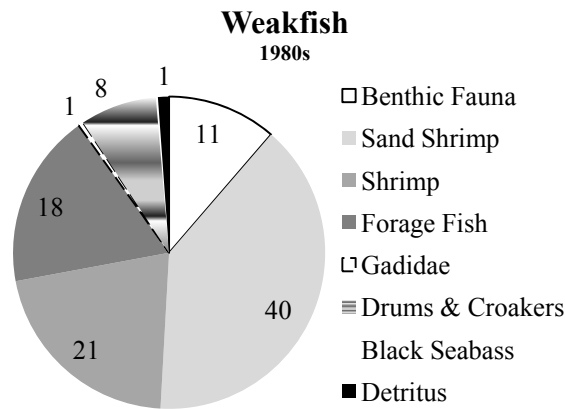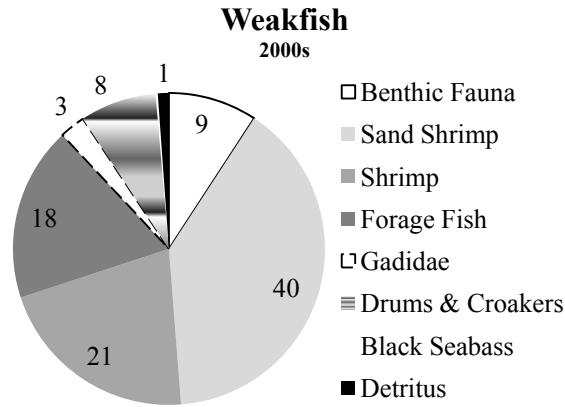ta estimated from catch records of Frisk and Munch (2008) provided a mortality measure of weakfish at 1.0666 (P/B). Input with an initial Q/B estimate of 4.1 from Fishbase (Froese and Pauly 2005), the calculated EE was not possible (EE > 1). Therefore, we set EE = 0.999 and allowed the program to calculate the mortality of the weakfish stock. Christensen et al. (2005) estimated an EE for weakfish at 0.88 in the Chesapeake, supporting a high estimate for this parameter. No additional studies were found that supplied data that could be transferred into a catch-at-age format. Furthermore, biomass estimates were available for the remaining three models. Therefore, with inputs of Q/B (4.1) and EE (0.999), P/B was left to be estimated by Ecopath for the remaining models. Hanlon (1983) measured the 1980s weakfish stock biomass at 0.1741 t/km^2. An estimate of the stock in the 1930s (2.8801 t/km^2) was supplied by data from Dickinson (1938) and Westman (1938). Scaled from the 1930s estimate, the 1880s model used a weakfish measure of 5.7601 t/km^2 based on trends provided by Nesbit (1938).

Diet: Dietary trends were estimated from multiple studies. The predominant trends were estimated from the Hudson-Raritan estuary (Steimle et al. 2000). The lack of predation on black seabass and low presence of drums and croakers in other diet studies

necessitated the addition of a Chesapeake Bay diet study (Hartman and Brandt 1995). However, only the contribution of black seabass and drums and croakers were used from Hartman and Brandt. A reduction in the gadidae component was also required to balance the 2000 model. Further reductions in gadidae were necessary for the 1980s and 1930s model. The 1930s data was used in the 1880s model.

Weakfish
2000s

- Benthic Fauna
- Sand Shrimp
- Shrimp
- Forage Fish
- Gadidae
- Drums & Croakers
- Black Seabass
- Detritus

Weakfish
1980s

- Benthic Fauna
- Sand Shrimp
- Shrimp
- Forage Fish
- Gadidae
- Drums & Croakers
- Black Seabass
- Detritus

Weakfish
1930s & 1880s

- Benthic Fauna
- Sand Shrimp
- Shrimp
- Forage Fish
- Gadidae
- Drums & Croakers
- Black Seabass
- Detritus

Windowpane Flounder

> **Includes:**
>
> - *Scophthalmus aquosus*

Parameter Estimates: Windowpane biomass was set to 0.2329 t/km^2 based on a GSB benthic trawl survey conducted in 2007 (Frisk and Munch 2008). The initial estimates of P/B (0.47) and Q/B (6.2) for this grouping came from the Fishbase equation (Froese and Pauly 2005). We assumed an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. The 1980s measure of biomass at 0.1255 t/km^2 was provided by Hanlon (1983). Coupled with the P/B and Q/B values of the 2000s model, EE was left to be estimated. As no biomass estimates were available for the 1930s and 1880s, the 1980s EE measure of 0.3693 became a necessary input with P/B (0.47) and Q/B (6.2).

Diet: Dietary trends were estimated from Sagarese (2009) in Shinnecock Bay and Port Jefferson Harbor. This study was used for the 2000s and 1980s models. The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the 1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters. However, as only 0.017% of the overall windowpane diet was transferred, only one figure was given.

Windowpane Flounder
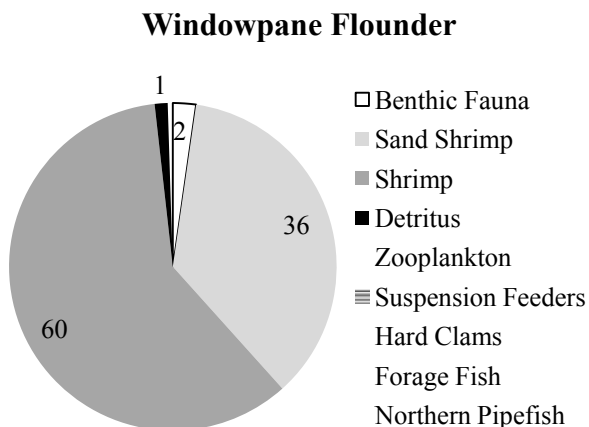
Winter Flounder

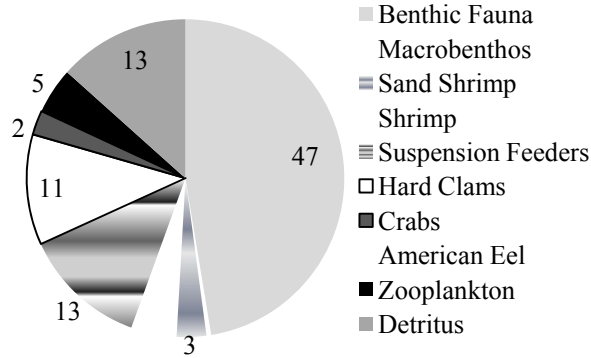**Includes:**
- *Pseudopleuronectes americanus*

Parameter Estimates: Although Frisk and Munch (2008) provided data that estimated winter flounder biomass for the 2000s model (0.1241 t/km^2), the landings harvested from this species suggested the Frisk and Munch (2008) estimate was too low. Therefore, the 2000s biomass was left to be estimated by the software. Initial Ecopath estimates for winter flounder were taken from the Fishbase website (Froese and Pauly 2005), Chesapeake Bay (Christensen et al. 2005), and two mortality studies by Poole (1966 and 1969). The Fishbase estimate of Q/B (6.3) was used for all four models and assumed an aspect ratio of 1.32 (Christensen et al. 2005), a carnivorous diet (supported by our diet data), and an average bay temperature of 15°C. The two Poole studies supplied P/B measures of winter flounder for the 1930s and 1960s and had no direct measure available for the 2000s. Therefore, the most recent mortality measure (0.72), from 1966, was used with the expectation that it would have to be increased to account for recorded stock declines and no drastic reductions in fishing effort. Similar to the rationale behind discarding our 2000s biomass estimate, the landings mortality on winter flounder coupled with the predation pressures seen in the dietary input suggested the initial EE estimate from Christensen et al. (2005) (0.9) may be too low. Therefore, it was maximized to 0.99. There was a relative agreement between the computed Ecopath biomass in the 2000s (0.2350 t/km^2) with that predicted from Frisk and Munch (2008). Winter flounder stock biomass in the 1980s was estimated by Hanlon (1983) at 8.7177 t/km^2. Although no direct measure of P/B was available, the 1980s winter flounder mortality was initially input at 0.72 based on measured values from the 1960s (Poole 1969). Q/B was set to 6.3, allowing the program to estimate the 1980s EE. The commercial trawling statistics seen in Dickinson (1938) measured stock biomass at 9.4214 t/km^2 in the 1930s. The 1937 mortality measure from Poole (1969) (0.78) was used for the 1930s P/B estimate. Using the Fishbase estimate of Q/B, EE was calculated with Ecopath. The qualitative trends given in Lobell (1938) allowed for the 1930s biomass to be scaled and provide an abundance estimate for the 1880s (18.1772 t/km^2). The EE from the 1930s (0.1130) was input with Q/B, leaving the P/B ratio to be estimated by the program.

Diet: Dietary trends for the 2000s model were combined from two separate studies. Data was obtained from Navesink, New Jersey (Stehlik and Meise 2000) and the Hudson-Raritan estuary (Steimle et al. 2000). The fish category from Stehlik and Meise (2000) was attributed to the American eel component as it was the only finfish identified in a winter flounder diet study (Kurtz 1975). The 1980s model used diet data obtained from the Great South Bay (Kurtz 1975). The 1930s and 1880s models, however, required the addition of an oyster grouping in their diet as these mollusks dominated the GSB shellfish community before the 1930s. As the 1930s represented an abundance of hard clams, 75% of the 1980s predation on hard clams were fed into the hard clam grouping while the remainder (25%) was assumed satisfied by oysters. Similarly, GSB in the

1880s experienced a wealth of oysters; therefore, 25% of the 1980s predation on hard clams was attributed to hard clams while the remaining 75% were fed into oysters.

Winter Flounder
2000s

- Benthic Fauna
  Macrobenthos
- Sand Shrimp
  Shrimp
- Suspension Feeders
- Hard Clams
- Crabs
  American Eel
- Zooplankton
- Detritus

Winter Flounder
1980s

- Benthic Fauna
  Macrobenthos
- Sand Shrimp
  Shrimp
- Suspension Feeders
- Hard Clams
- Crabs
  Blue Crab
  American Eel
- Detritus

Winter Flounder
1930s

- Benthic Fauna
  Macrobenthos
- Sand Shrimp
  Shrimp
- Suspension Feeders
- Hard Clams
- Oysters
- Crabs
  Blue Crab
- American Eel
- Detritus

Winter Flounder

1930s

- Benthic Fauna
  Macrobenthos
- Sand Shrimp
  Shrimp
- Suspension Feeders
- Hard Clams
- Oysters
- Crabs
  Blue Crab
- American Eel
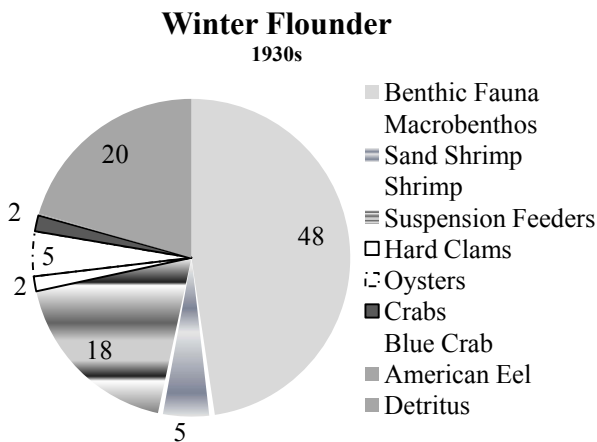- Detritus

Zooplankton

Includes:
- Copepods
  - Calanoidea
  - Harpacticoidea
  - Unidentified Copepods
- *Euphausia* sp. (Euphasids or Krill)
- Larvae
  - Decapoda Larvae
  - Crustacean Larvae (Zoea or Megalopa)
  - Osteichthyes Larvae
- Pteropods
  - *Clione* sp. (Sea Angels)
  - Cymbulidae
  - Thecostomata (Sea Butterflies)
- *Tomopteris* sp. (Planktonic Polychaetes)
- Unidentified Chaetognatha (Arrow Worms)

Parameter Estimates: The initial estimates for this grouping came from Delaware Bay (Frisk et al. in review). Estimates included P/B = 25.0 and EE = 0.9. The unassimilated consumption of this planktivorous grouping was also raised to 0.4, thought to be more accurate.

Diet: Dietary estimates were obtained from Frisk et al. (in review) from Delaware Bay in which phytoplankton was the sole dietary component of this grouping.

Phytoplankton

Parameter Estimates: The primary production of GSB was calculated from two assessments. Lively et al. (1983) calculated phytoplankton productivity in GSB at 450 t/km^2/year, attributing 85% of GSB's total production to these producers. Assuming the 1:9 wet weight conversion ratio from Pauly and Christensen (1995) is accurate, the production of GSB in the late 1980s was 4050 t/km^2/yr. Lonsdale et al. (1996) measured depth-integrated productivity sporadically throughout the year, providing data to estimate GSB production at 836 t/km^2/year. Using the ratio from Pauly and Christensen, a production measure of 7529 t/km^2/yr was calculated. Although these two studies only measured phytoplankton production, 85% of the production in GSB is supplied by this grouping (Lively et al. 1983). Unfortunately, these two studies only provided production estimates for two modeled periods, necessitating the recycling of one of these two measures through the other models. The measure from Lively et al. (1983) was the first estimate used in the models. As this measure provided the smallest production estimate and still proved drastically higher than that required to satisfy GSB consumers, the Lively PP value of 4050 t/km^2/yr was used for each model. The initial parameter estimates for this grouping came from the Delaware Bay (Frisk et al. in review) in which P/B was estimated at 60 and EE at 0.95. These measures were used for all four models.

Detritus

Includes:
- Animal Remains
- Setae (Bristles)
- Debris
- *Mytilus byssus*
- Shell Hash
- *Zostera* sp. (Eelgrass)
- *Spartina* sp. (Cord Grass)
- Sand/Mud
- Unidentified Algae/Plants
- Unidentified Insecta (Insects)
- Unidentified Arachnida (Spiders)

Parameter Estimates: Initial biomass of detritus was set to 1.0000 tonnes/km^2 based on the initial value used in Frisk et al. (in review) in Delaware Bay.

Appendix 2.2. Blue Crab Model

A fifth Ecopath model was constructed to forecast the potential changes in GSB should the current trend of increasing blue crab abundance continue. Using the balanced 2000s model, blue crab biomass was significantly increased and the model rebalanced to assess the impacts of this crustacean on the ecosystem.

Since the 1980s, our models showed more than a doubling in the blue crab stock biomass. Using this trend, blue crab abundance was increased to 1.0 t/km^2, up from 0.427 t/km^2 in the 2000s. P/B (1.4) and Q/B (4.0) outputs from the 2000s were also used. Furthermore, as an increase in blue crab biomass was expected to flow through the system given increases in various crustacean predators (skates, striped bass, and summer flounder), the EE from the 2000s model (0.95) was also input, preventing the model from impeding the flow of increased biomass by simply lowering the final EE.

The next step in compiling the blue crab model was determining which stocks were likely to be altered the most should blue crab biomass continue to increase. As the Ecopath routine balances loss processes against stock production, the two predominant loss processes (fishing and predation) were used as a baseline for the alteration of model parameters. First, fishery landings of blue crabs were assessed to determine the response of the GSB fisheries to the abundant crustacean. Since the doubling in biomass between the 1980s and 2000s was met with a 3.4% reduction in landings, no change was deemed necessary to blue crab harvest in the hypothetical model. Second, the dietary matrix was explored to select stocks that were susceptible to blue crab fluctuations. In this, biomasses of stocks that both feed upon and are consumed by blue crabs were

considered. Biomass was estimated by the program for stocks that contributed more than 1% to the total blue crab diet by weight or stocks that consumed more blue crabs than 1% of their total diet, requiring inputs of P/B, Q/B, and EE from the final 2000s model. While this method adopts uncertainty in the assumption that mortality rates (P/B) and dietary habits remain constant across these stocks, all three of these parameters (biomass, P/B, diet) are likely to change should stock abundances vary and with no direct measures available, uncertainty is unavoidable. As biomass fluctuations were the ultimate interest, the estimation of these parameters was the focus for groupings thought to be influenced the most. Conversely, stocks that are not directly dependent on the blue crab stock (indirect predation impacts) and are described as contributing less than 1% to the total blue crab diet or consume less blue crabs than 1% of their total diet were simulated using the final 2000s biomass, Q/B, and EE. P/B was estimated by the program for these groups.

The final blue crab model showed a biomass increase across multiple Ecopath groupings utilized as prey by blue crabs (Table A2.2-1): suspension feeders (+3.56%), hard clams (+3.51%), benthic fauna (1.50%), zooplankton (+1.40%), and sand shrimp (+0.90%). These groups make up a substantial portion (59.7%) of the blue crab diet (Appendix 2.1). Due to the mass-balance assumption in Ecopath, the increase in blue crab prey biomass was calculated to satisfy the elevated predation pressures from blue crab. Other prey stocks showed enlarged mortality rates (P/B): spider crabs (+2.13%) and the generic crab grouping (+5.65%).

Predators of blue crabs also showed fluctuations under the availability of prey. A slight increase in bluefish biomass was also recorded (+0.000013%), owing to the usage

of blue crabs as a prey component. Although individual bluefish consume an immense quantity of biomass ($Q/B = 15.4$), the small percentage that blue crabs compose the overall diet (2.86%) coupled with the omnivorous nature of bluefish minimized the potential stock increase. Conversely, other predators that utilize blue crabs as a primary dietary component (more than 1%) like striped bass, skates, and blackfish showed no measured variations to either P/B or biomass.

The results of the blue crab model indicate the trophic impact of an enlarged blue crab stock in GSB may be minimal. Only small variations in abundance were observed between the 2000s model and that calculated in the trial, the largest of which measured a difference of 3.56%. Furthermore, the mixed trophic impact of specific groupings showed limited variation (Figure A2.2-2) from the 2000s model. However, the omnivorous nature of blue crabs (Laughlin 1982) creates uncertainty in this finding. The groupings included in the diet of blue crabs for this study represent only a fraction of potential prey items. If other edible stocks become more accessible, blue crabs may alter their dietary habits and transition their predation pressure to other groupings. In this, other stocks may be less resilient to blue crab predation. However the proportion of available production utilized by the system increased in this model (Table A2.2-2), suggesting a blue crab increase may increase the trophic efficiency of GSB.

Table A2.2-1. Final Parameters (Biomass, P/B, Q/B, EE, P/Q) for the Blue Crab Model.

Group Name	2000 Model - Blue Crab				
	Trophic Lvl	Biomass	P/B	Q/B	EE	P/Q
Benthic Fauna	2.00	7.65	4.58	24.36	0.859	0.188
Macrobenthos	3.14	0.57	1.00	1.65	0.804	0.605
Sand Shrimp	2.08	1.59	3.75	24.90	0.999	0.151
Shrimp	2.34	2.19	3.30	16.10	0.860	0.205
Mantis Shrimp	3.01	0.22	1.36	7.43	0.950	0.182
Squid	2.29	0.05	2.74	7.00	0.990	0.392
Forage Fish	3.08	1.30	1.29	15.00	0.999	0.086
Suspension Feeders	2.09	70.68	0.30	2.18	0.931	0.138
Hard Clams	2.09	1.85	3.73	5.10	0.999	0.731
Tropical Fish	3.11	0.38	1.27	7.97	0.999	0.159
Crabs	2.95	7.61	1.46	8.50	0.963	0.172
Blue Crab	3.20	1.00	1.40	4.00	0.950	0.350
Horseshoe Crabs	2.97	0.06	0.60	3.00	0.500	0.200
Spider Crabs	2.55	0.10	2.34	15.63	0.950	0.150
Skate	3.55	0.82	0.15	4.10	0.983	0.037
American Eel	2.01	0.27	0.22	3.70	0.500	0.059
Menhaden	2.06	1.02	1.10	28.00	0.990	0.039
Black Sea Bass	3.17	0.02	0.30	6.90	0.614	0.043
Blackfish	3.78	0.19	1.21	3.10	1.000	0.391
Bluefish	3.62	0.80	1.06	15.40	0.999	0.069
Butterfish	2.58	0.03	0.80	5.50	0.142	0.145
Cunner	3.12	0.01	0.48	7.50	0.990	0.064
Dogfish	3.52	0.10	0.05	4.77	0.950	0.010
Drums & Croakers	3.04	0.27	0.40	3.90	0.497	0.103
Flatfish-Other	3.28	0.01	0.74	7.03	0.950	0.105
Gadids	3.16	0.01	1.94	3.66	0.999	0.530
Northern Kingfish	3.77	0.07	0.30	5.90	0.538	0.051
Northern Pipefish	3.27	0.01	1.30	6.90	0.999	0.188
Oyster Toadfish	3.49	0.00	0.36	6.20	0.058	0.058
Scup	3.08	0.27	0.60	5.50	0.999	0.110
Striped Bass	3.73	2.30	0.49	2.45	0.900	0.200
Sea Robins	3.30	0.20	0.43	7.15	0.980	0.060
Summer Flounder	3.80	1.79	1.00	4.00	0.950	0.250
Weakfish	3.40	0.11	1.33	4.10	0.999	0.325
Windowpane Flounder	3.22	0.23	0.47	6.20	0.051	0.076
Winter Flounder	2.94	0.24	1.50	6.30	0.999	0.238
Zooplankton	2.00	1.66	25.00	90.91	0.900	0.275
Phytoplankton	1.00	4050.00	60.00		0.001	
Ctenophores	2.50	0.15	8.80	35.20	0.900	0.250
Detritus	1.00	1.00			0.001	

Table A2.2-2. Summary Statistics for the Blue Crab Ecopath Models.

Statistic	2000 BC Model
Total System Production (ΣP)	243141.60
Production Required	499.30
Net Primary Production (PP)	243000.00
Total Respiration (RESP)	423.01
Net System Production	242577.00
Total System Throughput (TST)	486713.30
Total Biomass w/out Detritus (B)	4155.83
Connectance Index	0.19
System Omnivory Index (SOI)	0.32
Total Consumption (ΣQ)	744.87
Total Flows to Detritus	242891.30
Total Exports (EXP)	242654.10
Mean Trophic Level of Catch	3.59
Total Catch	4.46
Gross efficiency (Catch/PP)	0.00

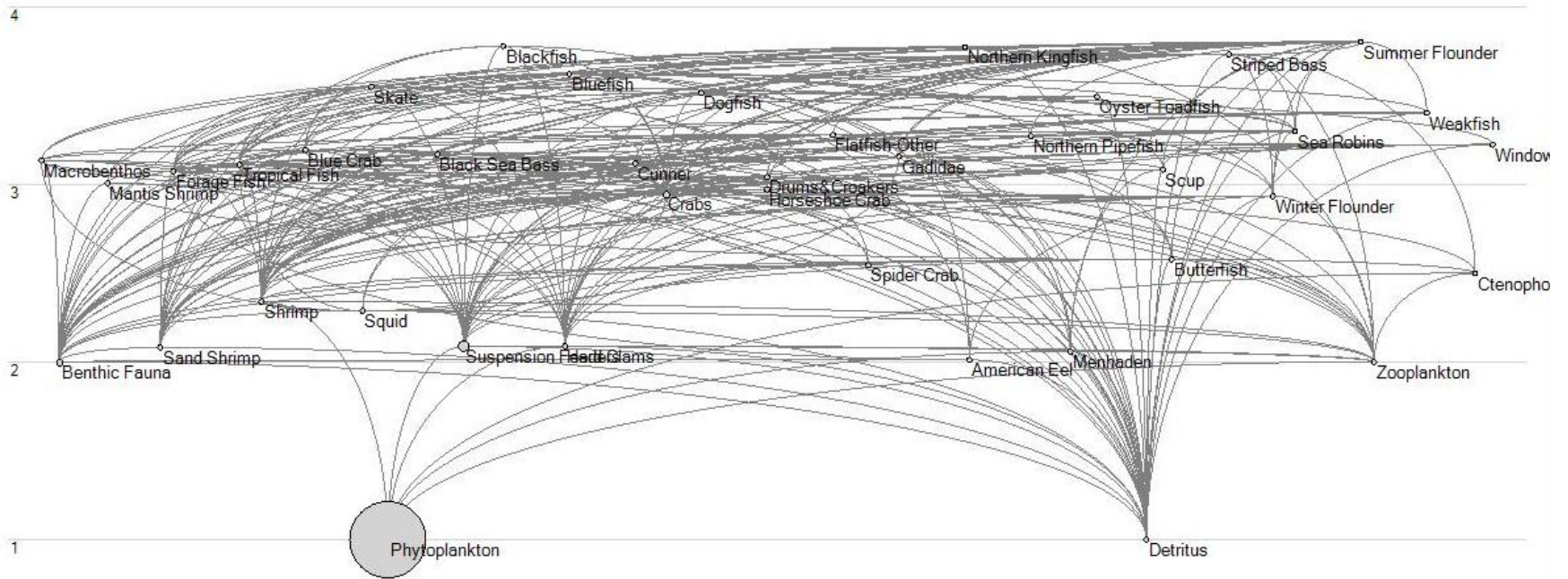Figure A2.2-1. Flow Diagram of the Blue Crab Model.

Figure A2.2-2. Total Ecosystem Impact (ε) vs. Keystoneness for the Blue Crab Model. Stocks with high values of ε and KS represent groups that impose large influences on the ecosystem.