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**CHESAPEAKE BAY FISHERIES:  
PROSPECTS FOR MULTISPECIES MANAGEMENT AND  
SUSTAINABILITY**

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# Chesapeake Bay Fisheries: Prospects for Multispecies Management and Sustainability

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## EXECUTIVE SUMMARY

Fishery resources in the Chesapeake Bay are currently managed as individual species. In this framework the potential effect of the harvest of a species on the ecosystem generally is ignored. However, the Chesapeake Bay supports a multispecies fishery that annually lands finfish and shellfish worth in excess of \$100 Million. Over the past 25 years the average annual commercial landing has been approximately 250,000 metric tonnes. Although menhaden and blue crabs represent ninety-five percent, by weight, of the commercial catch in the Bay, statistics show that 59 other species are also caught. The recreational sector also accounts for a large and diverse catch. Furthermore, there have been significant changes in the nature of the fishery. Over the last one hundred years landings of oysters have diminished greatly, and in their place, landings of blue crab have risen dramatically. Over the same time period landings of anadromous fishes, such as American shad have declined. In contrast, landings of menhaden have risen so that its fishery now accounts for over 80%, by weight, of the total catch.

The multispecies nature of the combined fisheries arises for both technical and biological reasons. Technical interactions, which arise when a fishery targets on one species but catches other species incidentally as bycatch, are present in the Chesapeake Bay. For example, 45 species are taken in poundnet fisheries and 53 species are taken in gillnet fisheries. Technical interactions are important considerations in fisheries management as they may limit the ability to regulate overall rates of fishing mortality. Biological interactions, which arise when a targeted species is an important link in a food web, also occur in the Chesapeake Bay. For example the removal of top predators (striped bass, bluefish and weakfish) may have significant impacts on the dynamics of the planktivore species, and thus the plankton community itself. Additionally, harvests of blue crab, spot and croaker have the potential to influence energy and nutrient exchanges between the benthic and pelagic food webs. To address these multispecies interactions several new approaches to fisheries management have been developed. These approaches implicitly account for intra-specific interactions. Ultimately, these approaches may be more compatible philosophically with the ecosystem-level management of the Chesapeake Bay's other natural resources.

We explored the need for and potential of multispecies approaches to the management of fisheries resources in the bay. The evidence suggests that adopting a multispecies approach would be advantageous. Many of the forces that lead to the adoption of multispecies management in other ecosystems are present in the Chesapeake, including concerns over extensive bycatches, and the presence of coupled population dynamics for several components of the ecosystem.

We reviewed multispecies approaches employed elsewhere in the U.S. and worldwide. We identified several broad classes of approach. The most direct approaches were descriptive involving graphical or multivariate statistical approaches such as principal components analysis and state-space time series analysis. These approaches are suitable to identify the extent and importance of the multispecies character of a fishery, but may have limited utility for management. Other approaches are more mechanistic. Examples include closed-form, and simulation models of interacting species, and more holistic models of the entire system. We suggest that multispecies models addressing technical interactions, and those involving descriptive rather than mechanistic approaches are most likely to be successful in the near-term. However, several approaches such as simulation modeling and multispecies virtual population

analysis seem unsuitable management tools for the Chesapeake Bay due to their high demand for data that is not currently available. New research, data collection and database development to correct these shortcomings are strongly recommended.

Our review indicates that several factors currently preclude adopting a multispecies approach in the Chesapeake Bay. Specific areas that must be addressed, which would improve current single-species management and develop the capability to explore the application of multispecies approaches include the need for:

- < systematic information on catch and effort for exploited stocks,
- < fishery-independent estimates of abundance for principal species in the bay,
- < basic life history information,
- < detailed knowledge of species interactions (especially predator - prey relationships),
- < effects of habitat alteration,
- < detailed understanding of multispecies models.

Adopting multispecies approaches to management would be a major shift away from traditional single species management and a major step toward fulfilling the ecosystem management goal of the Bay Program. We are not ready for this step today, but addressing the identified deficiencies will prepare us for multispecies management in the future.



# CHESAPEAKE BAY FISHERIES: PROSPECTS FOR MULTISPECIES MANAGEMENT AND SUSTAINABILITY

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## I. BACKGROUND.

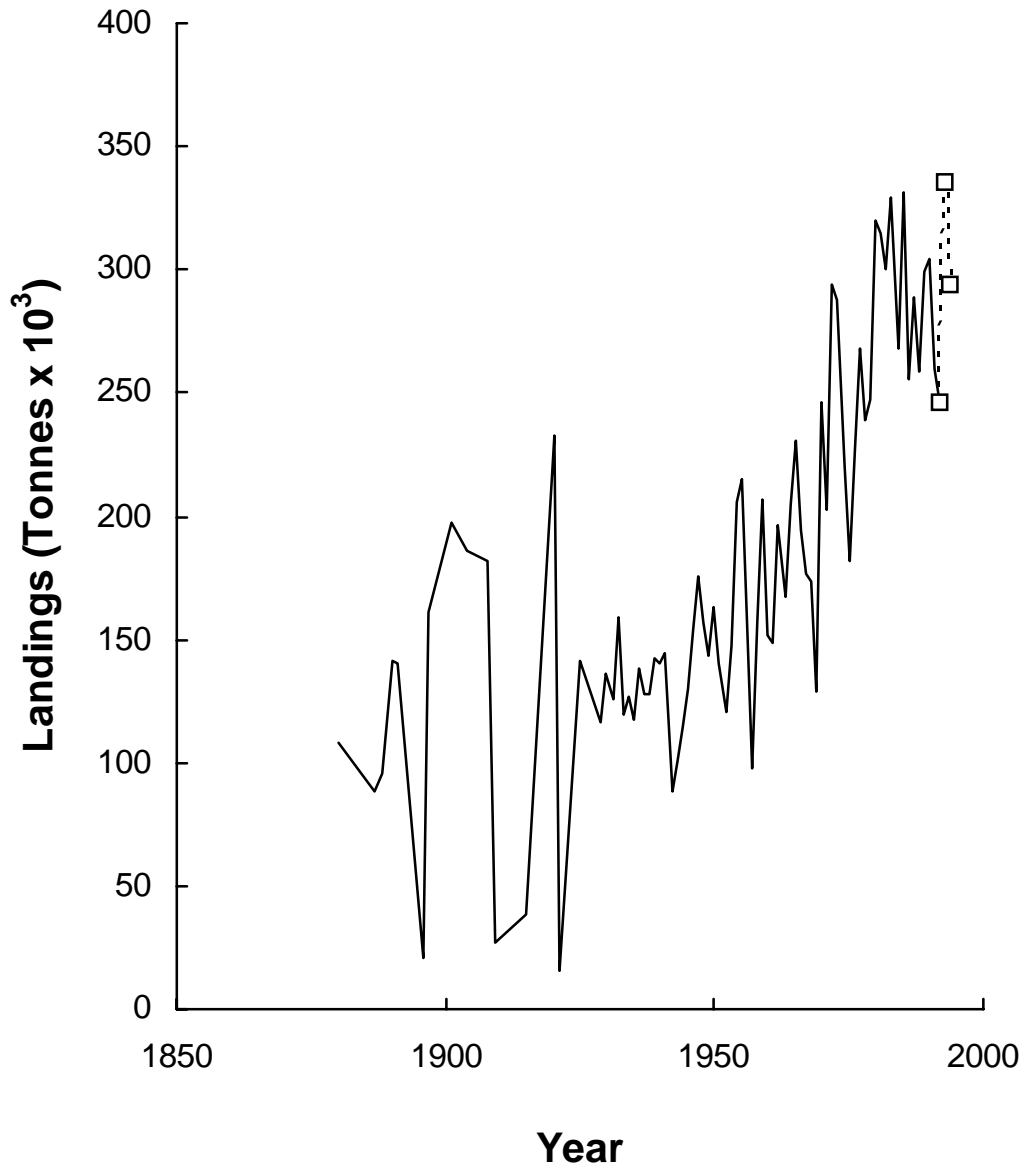
### A. Why a Multispecies Approach?

Fisheries in Chesapeake Bay contribute significantly to U.S. catches at the national and regional levels. Recent National Marine Fisheries Service (NMFS) statistics indicate that between 250,000 - 350,000 metric tonnes (t) of fish and shellfish are harvested annually from Chesapeake Bay waters (Fig.1), with a dockside value of more than \$100 Million. Maintaining the health of this fishery is an important but difficult task given the considerable interannual variability in catches of component species (Fig. 2). Moreover, individual species exhibit different, and sometimes opposite trends, further complicating analysis and understanding. The variability and trends illustrated in Figures 1 and 2 are not unique to Chesapeake Bay, but characterize fisheries in general (Hilborn and Walters 1992). Recent research has sought to include effects of species interactions and overall community structure into management practices by developing multispecies management models, thereby incorporating the complexity of multispecies dynamics into the regulatory process (Kerr and Ryder 1989; Appollonio 1994).

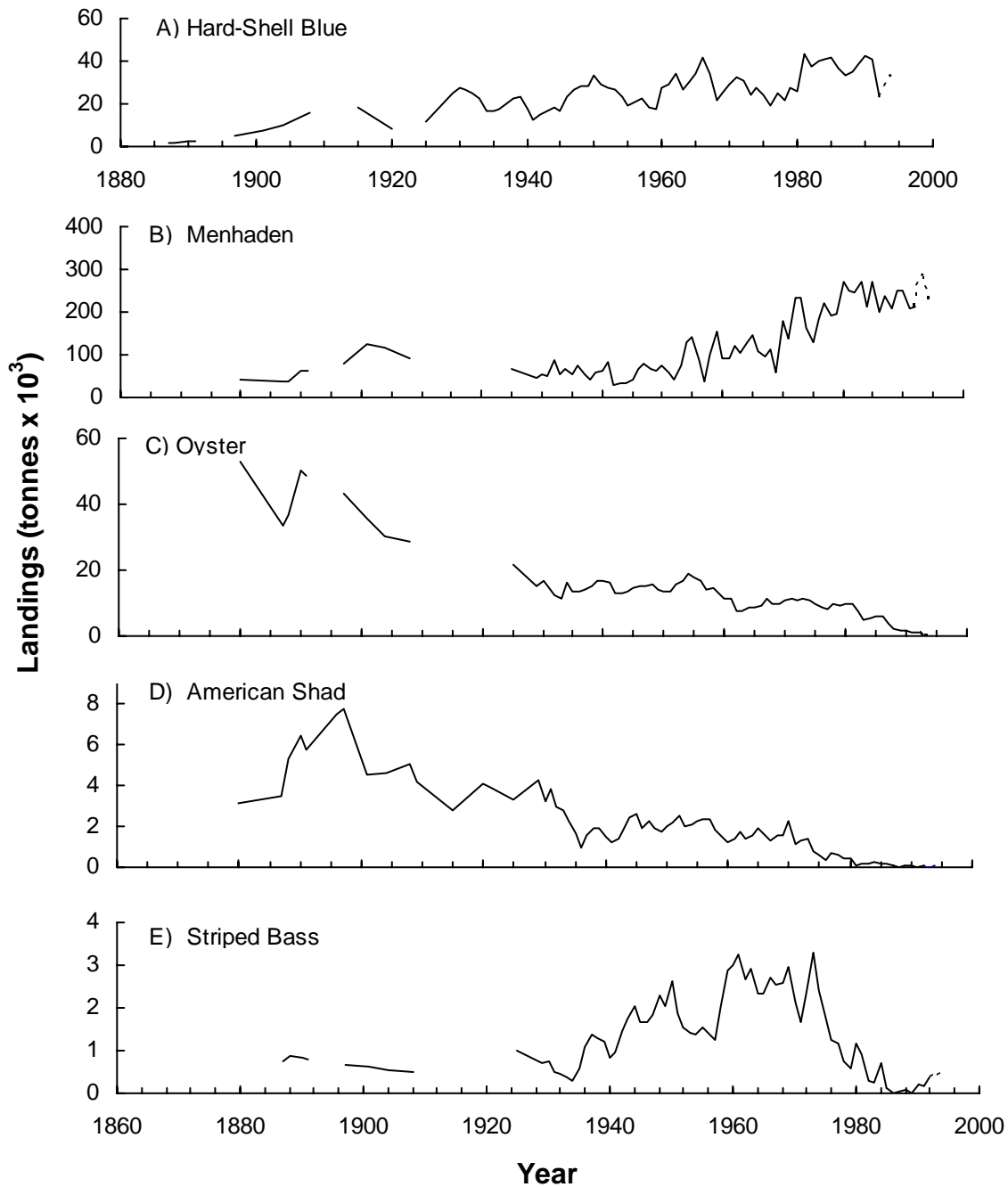
Multispecies approaches to fisheries management do not represent a single, unified methodology. However, all of the

approaches share a common attribute; all were formulated to account for the effects of removal of a single species by a fishery on other species in the system. As an example, when individuals of one species are removed by a fishery from an ecosystem, its predators presumably have less food, and its prey experience reduced mortality. These changes can alter the population dynamics of individual species and overall community structure. Moreover, fisheries are unlikely to exploit a single species. Thus, increased fishing on one species, may lead to increased incidental mortality, or bycatch, in other species that also are vulnerable to the particular fishing method. A variety of techniques can account for such interspecific interactions. Some are direct extensions of existing single-species approaches, such as the multispecies surplus production models (Sainsbury 1988, 1991). However, there are also techniques unique to the multispecies case, such as network analysis (Christensen and Pauly 1992).

The Chesapeake Bay is a complex ecosystem that supports living resources at several trophic levels and in diverse habitats (Baird and Ulanowicz 1989). Increases in nutrient loading during the recent past have been implicated in or have lead to dramatic, system-wide changes (Horton and Eichbaum 1991). The loss of submerged aquatic



**Figure 1.** Time series of total Chesapeake Bay commercial fishery landings (thousands of metric tonnes) from 1880 - 1992. Prior to 1956, Atlantic coast catches landed at Chesapeake ports were included in this total. Data for 1993-1994 are preliminary and are shown by the



**Figure 2.** Time series of commercial landings (thousands of metric tonnes) for selected individual species taken from the Chesapeake Bay. Prior to 1956, Atlantic coast catches landed at Chesapeake ports were included in this total. In combination the 5 species illustrated have represented between 95-99% of the total the Chesapeake Bay commercial catch. Data for 1993-1994 are preliminary and are shown by the dotted lines.



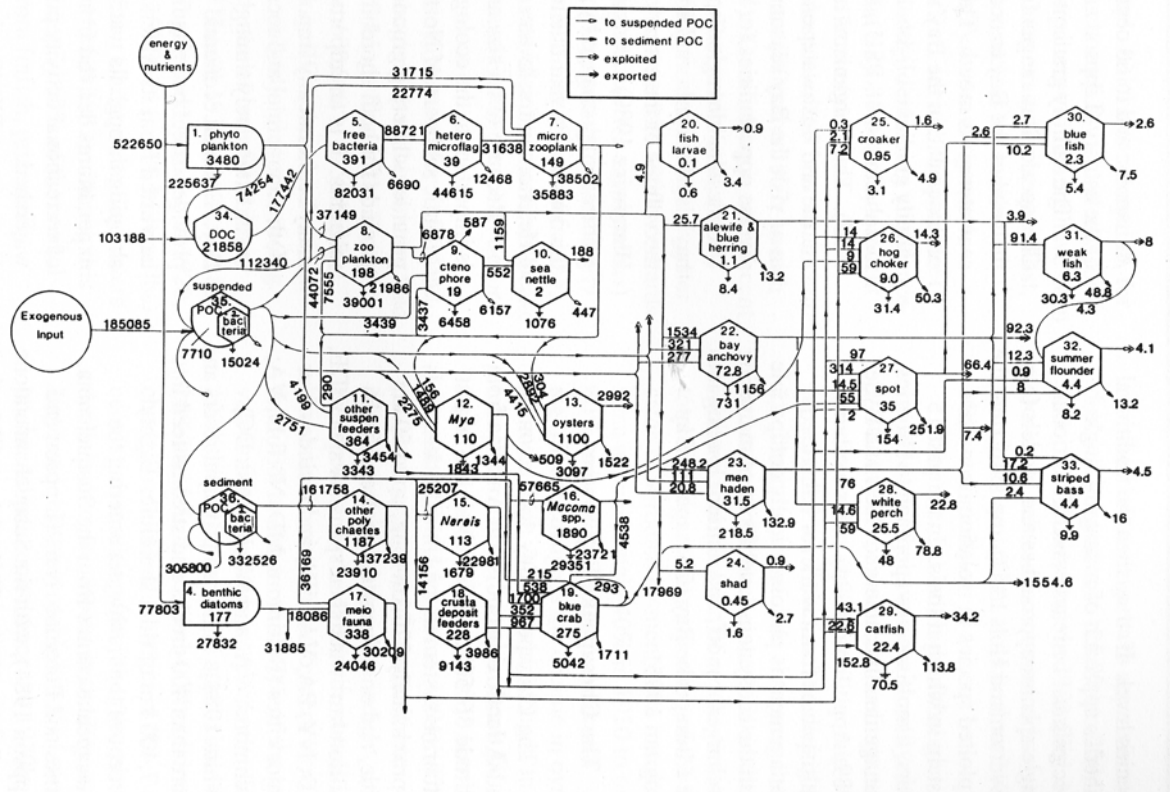
vegetation is one example. A second example is the increase in phytoplankton blooms and related incidence of anoxic waters in the Bay, caused by elevated levels of nitrogen and phosphorus (Boicourt 1992). Overall, the system-wide changes had important consequences for the distribution and productivities of living resources (Boynton et al. 1995). Currently, as nutrient controls are implemented throughout the Chesapeake watershed, system-level responses are likely to re-occur. Such system-level responses infer that living resources in the Chesapeake Bay are tightly linked and dependent upon one another, implying that a multispecies approach, which recognizes such dependencies, may have merit for fisheries management.

As with all ecosystems, multispecies interactions within the Chesapeake Bay may be driven both by trophic relationships within the ecosystem, and by forcing from outside, for example by nutrient loading (Carpenter et al. 1985). Research on the dynamics of the trophic network in the Chesapeake Bay is well advanced and is revealing a complex pattern of temporally and spatially explicit dependencies among predators and prey (Baird and Ulanowicz 1989; Brandt et al. 1995). This complexity results, in part, from substantial seasonal variation in community assemblages. For example, Baird and Ulanowicz (1989) indicated that control over the plankton community is exercised by crustacean zooplankton in the spring, but by the sea nettle (*Chrysaora quinquecirrha*) in the summer months. Moreover, strong seasonal migrations within the fish community produce clear seasonal differences in potential levels of piscivory and piscivore growth (Hartman and Brandt 1995a, b). In addition to the seasonal patterns in predator-

prey interactions, the linkage of benthic and pelagic food chains creates the potential for large seasonal shifts in nutrient cycling (Baird and Ulanowicz 1989; Baird et al. 1995; Bartleson and Kemp 1990; Kemp and Bartleson 1990).

Both exploited and unexploited species are involved in the extensive web of trophic interactions in Chesapeake Bay (Fig. 3). Among the exploited species in the Bay, these interactions may affect productive potentials. For example, bluefish (*Pomatomus saltatrix*) may compete with weakfish (*Cynoscion regalis*) and striped bass (*Morone saxatilis*) because all rely on bay anchovy (*Anchoa mitchilli*) and Atlantic menhaden (*Brevoortia tyrannus*) as prey (Hartman and Brandt 1995a). It is possible to trace many similar dependencies in Figure 3. Thus, because removal of individuals of one species by fishing has effects on other components of the ecosystem, a fishery may exercise broad control over the structure and productive capacity of the ecosystem. For example, Newell (1988) suggested that the reduction in oyster (*Crassostrea virginica*) populations in the Bay by fishing and habitat degradation (Rothschild et al. 1994) may have amplified the effects of increased nutrient loadings which, combined with loss of oysters, has led to profound changes in the living resources of Chesapeake Bay. Thus, management actions that target particular components of the ecosystem and fail to recognize interactions or dependencies may result in broad scale changes in ecosystem structure and productivity.

Overall, evidence suggests that fisheries, including those in Chesapeake Bay, may benefit from management at the system level, rather than at the individual



**Figure 3.** Energy flow network in the mesohaline area of the Chesapeake Bay during summer (from Baird and Ulanowicz 1989). Values are as follows: numbers associated with arrows are carbon flows in  $\text{mgAm}^{-2}\text{summer}^{-1}$ , numbers in the tops of boxes are identifiers, numbers in the bottoms of boxes are the biomass of carbon  $\text{mgAm}^{-2}$ .

species level. If true, then the traditional fisheries approach of managing single species must be reassessed. Traditional single-species approaches and models (e.g. Beverton and Holt 1957) implicitly treat the exploited species in isolation from the system in which it lives. In the past 15 years, a multispecies approach to fisheries management has emerged (Kerr and Ryder 1989; Appollonio 1994). This evolving multispecies framework for fisheries management is philosophically compatible with the ecosystem-level focus of managing discharges, runoff, and nutrient loading into the Chesapeake Bay (Chesapeake Bay Program 1995).

## **B. The Chesapeake Bay System.**

The Chesapeake Bay, located on the mid-Atlantic coast of North America from latitude 36°50' to 39°40' N, is the largest estuarine system in the United States. It is approximately 320 km long, up to 50 km wide, and on average 6.4 m deep. Its 16 Million km<sup>2</sup> watershed spans six states (DE, MD, NY, PA, VA, WV) and includes four major cities (Baltimore, MD; Norfolk, VA; Richmond, VA; and Washington, DC) (White 1984). Nineteen principal rivers and more than 400 creeks and streams feed into its 7,400 km of tidal shoreline. Eighty percent of the freshwater entering the Chesapeake comes from the Susquehanna, James and Potomac rivers (Lippson and Lippson 1984), with the Susquehanna alone contributing almost 50% of the total fresh water flow. The average flushing time of the Bay is about 42 days (Baird and Ulanowicz 1989). A semidiurnal tidal pattern occurs throughout the Chesapeake, ranging in magnitude from one meter in the southern Bay, to 30 centimeters near Annapolis, and to 60 centimeters at the head

of the estuary (Lippson and Lippson 1984).

The salinity gradient within the Chesapeake Bay ranges from freshwater at the head of the Bay to nearly 30 PSU seawater at its mouth. There are three major ecotones within the Bay's well-defined salinity gradient: oligohaline (0-6 PSU), mesohaline (6-18 PSU), and polyhaline (>18 PSU). Thus, organisms adapted to both marine and freshwater environments reside in or visit the Bay seasonally, providing diverse opportunities for fishery harvests. The mesohaline region spans 48% of the bay's surface area, and encompasses 47% of the volume of the estuary (Baird and Ulanowicz 1989). Longitudinal salinity gradients may change by as much as 5 PSU in the course of a year due to seasonal differences in freshwater flow.

Because of its size and location, the Bay is important to the ecology of the entire Atlantic seaboard of North America. It provides wintering grounds and breeding grounds for both shorebirds and neotropical migrants. It is an important nursery for many economically important fish species. Both commercial and recreational fishing occur extensively throughout the Chesapeake Bay. From an economic viewpoint the Bay provides direct benefits from the animals harvested from its waters, the passage of shipping through its waterways, and from expenditures derived from the wide variety of recreational activity pursued in its watershed.

## **C. Chesapeake Bay Commercial and Recreational Fisheries.**

Fisheries have always been important in the Chesapeake Bay (Fairbanks and Hamill 1932; Wharton 1957). Many species have been and are exploited for consumption, for industrial products such as fish oil and for

recreation. The patterns of fishing have been diverse and include examples of specialist, targeted fisheries such as the menhaden purse-seine fishery and more general, opportunistic efforts such as the poundnet fishery. Importantly, declines in abundance of fish within the Bay are not only a recent concern. The effects of dams on anadromous fish species has been a concern for over 200 years (Wharton 1957; Loesch and Atran 1994). Overfishing is not a recent problem either. Rothschild et al. (1994) present compelling evidence that the oyster was overfished from the late 1800's onwards. Atlantic sturgeon (*Acipenser oxyrinchus*) abundances had been greatly reduced early this century by the direct effects of fishing, and now are almost absent in the Chesapeake system (Horton and Eichbaum 1991; Secor 1995).

The increase in demand for finfish and shellfish at the beginning of this century lead to increases in fishing effort in the Bay. In 1880, more than 100,000 t (. 250 Million lbs) of fish and shellfish were landed from the Chesapeake and adjoining waters (Fig. 1), with a total value of over \$8 Million (Fairbanks and Hamill 1932). Between 1880 and 1930, landings varied, peaking in 1920, when possibly 250,000 t (. 600 Million lbs) was harvested from Chesapeake Bay waters (Fairbanks and Hamill 1932). During this period, approximately 60% of the annual harvest came from finfish, principally menhaden (Fig. 4). Shellfish, principally hard-shelled blue crab (*Callinectes sapidus*) and oysters, represented the remaining 40% (Fig. 4). Principal gears used were purse seines for menhaden, poundnets for shad, trotlines for crabs and tongs for oysters (Chowning 1990). Since the 1930's, total commercial landings have approximately doubled from almost 125,000 to 250,000 t

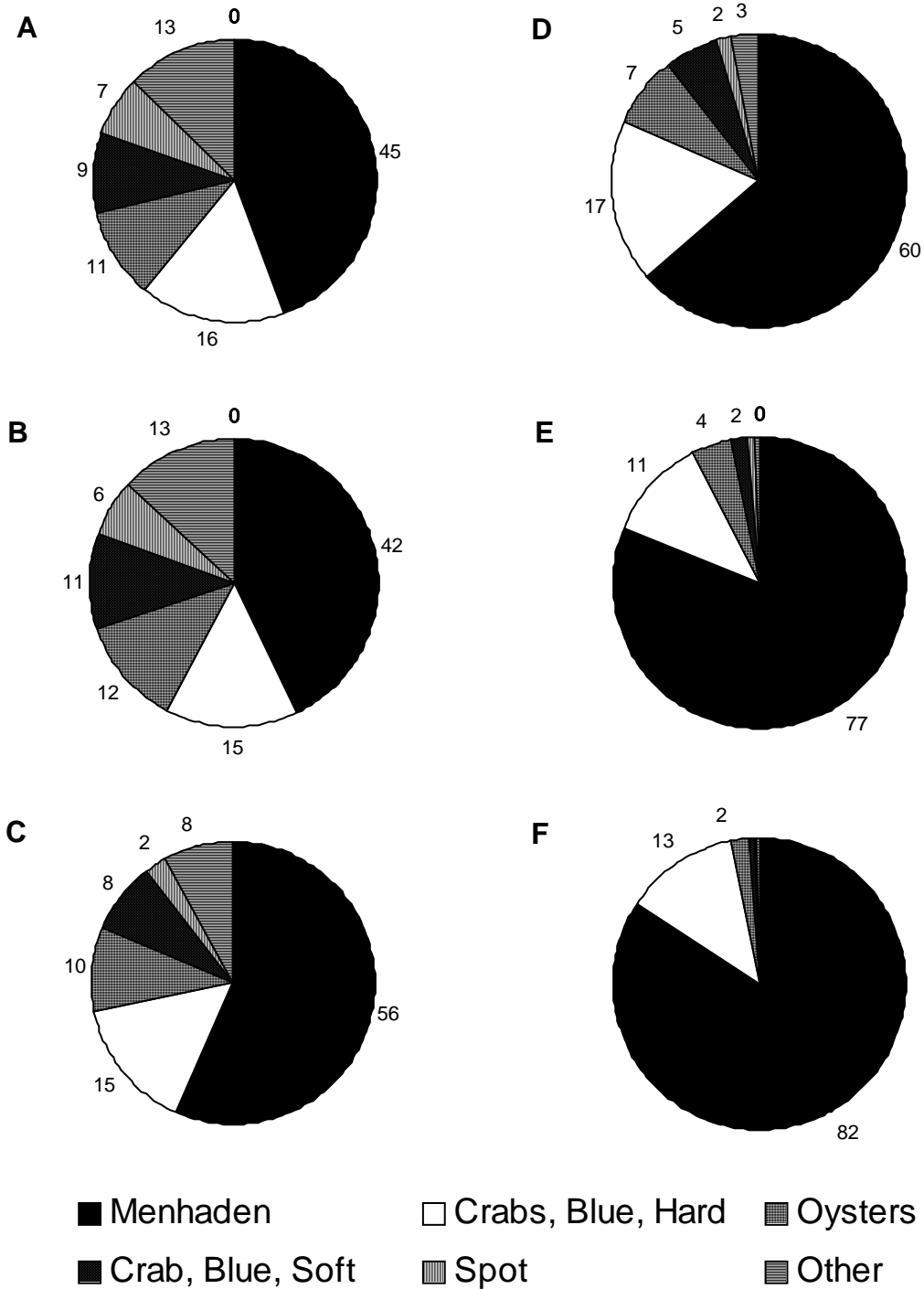
(. 300 - 600 Million lbs) in 1992 (Fig. 1). Dockside values of commercial harvests in the 1990's are valued at more than \$100 Million annually.

Commercial landings, although variable, clearly have risen steadily since 1930. In fact, commercial landings increased quite rapidly until the mid-1980's (approx. 2.9% yr<sup>-1</sup>) before stabilizing (Fig. 1). The most significant change in the commercial fisheries since 1930 is the increased contribution of menhaden to the total landings. The proportion of the total catch contributed by menhaden has doubled from approximately 40% in 1930 to over 80% today (Figs. 2B, 4).<sup>1</sup> A second significant change is the decline in oyster harvests (Figs. 2C, 4), which began in the late 19th century, but which have fallen from 15% of the total in 1930 to less than 1% today (Rothschild et al. 1994). The picture is even more dramatic if one considers value, rather than weight, of the two species. In the 1930's the values of harvested menhaden and oysters were less than 10% and more than 40% of the total value of Bay landings, respectively. In the 1990's these figures were approximately 20% and 5%, respectively. These dramatic changes reflect both changes in the ecosystem resulting from the precipitous declines of oyster stocks in the Bay and a large and relatively stable stock of menhaden on the Atlantic coast since the 1970's.

Recreational fisheries are of major

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<sup>1</sup>Menhaden landings for recent years are not reported directly in Chesapeake Bay fisheries statistics. For this report, approximate landings were derived from an analysis of total menhaden landings and Chesapeake Bay menhaden landings. See page 18 for explanation.



**Figure 4.** Average percent composition of Chesapeake Bay catch for the most abundant species for the decades A) 1930's, B) 1940's, C) 1950's, D) 1960's, E) 1970's and F) 1980's. Percent values are shown next to each segment.

importance in the Bay (Fig. 5). Since 1981, the National Marine Fisheries Service has annually surveyed the recreational fishery. A diverse array of marine, anadromous and freshwater species is caught, including striped bass, bluefish, weakfish, summer flounder (*Paralichthys dentatus*), croaker (*Micropogonias undulatus*), spot (*Leiostomus xanthurus*), white perch (*Morone americana*), and channel catfish (*Ictalurus punctatus*). NMFS data suggest that for the principal species, recreational landings are of the same magnitude or exceed the commercial harvest (Fig. 5A). A catch exceeding 2,500 tons has been landed in recent years. However, this is far lower than the catches taken in the early to mid 1980s. The sharp decline in bluefish landings in the late 1980's reflects a coast-wide decline, rather than one specific to the Chesapeake Bay. Additionally, effort in the recreational fishery is likely to increase. In the 15 years of the NMFS recreational survey the estimated number of annual fishing trips on the Bay has more than doubled from 2 - 4.5 million (Fig. 5B). Moreover, the catch and impact of the very large recreational fishery for blue crab are essentially unknown. Recreational landings of blue crab might be 10-25% of those in the commercial fishery (Chesapeake Bay Program 1996).

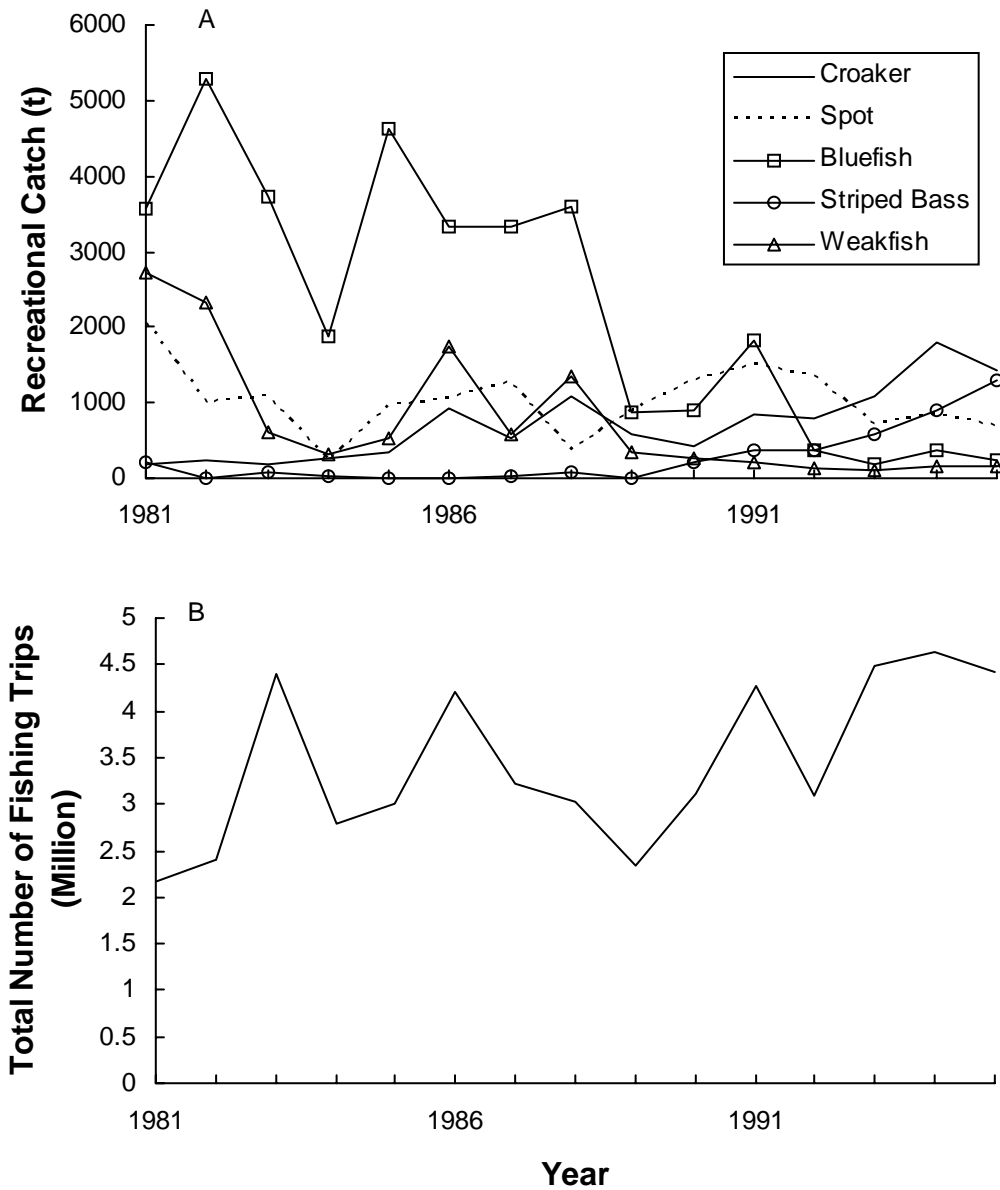
#### **D. Fisheries Management in the Chesapeake Bay.**

Both federal and state agencies have responsibility for managing fisheries within the Chesapeake Bay. Maryland and Virginia are responsible for regulations of fisheries within their territorial boundaries. However, the majority of stocks of individual species in the Bay span both jurisdictions. The

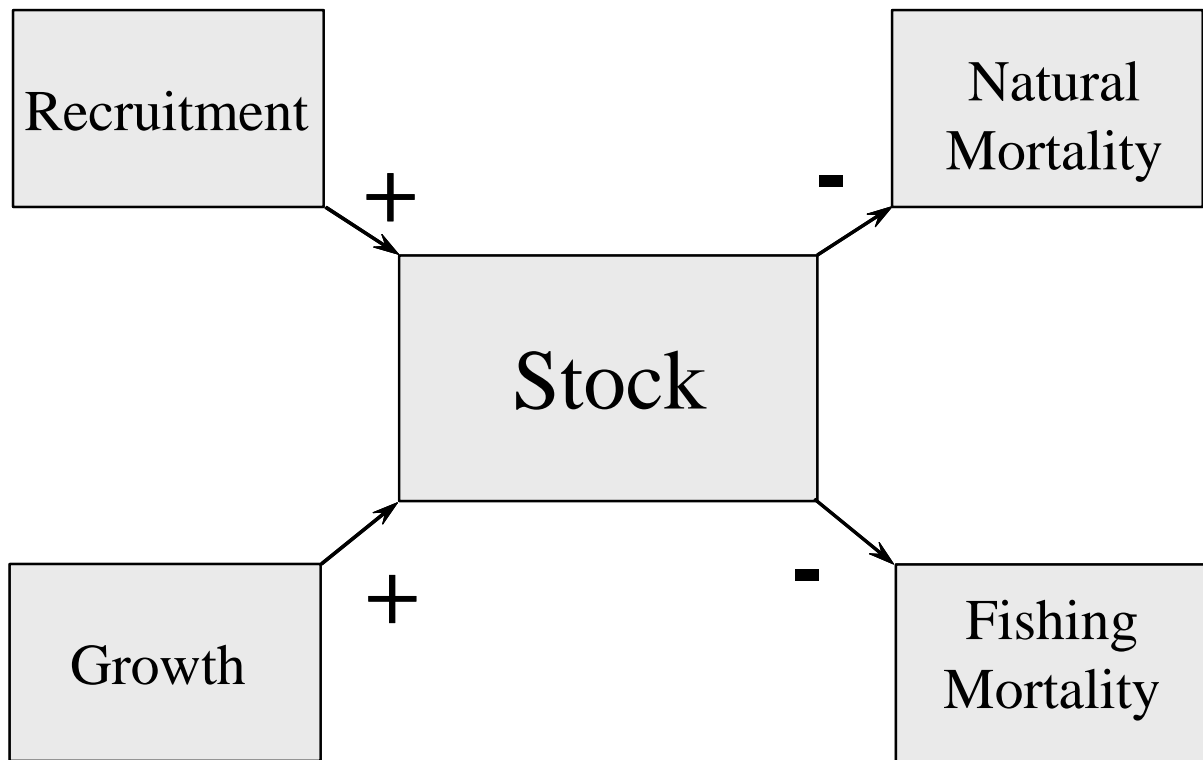
efforts of both states are now coordinated via the Chesapeake Bay Program, which oversees development of Fishery Management Plans (FMP) as specified in the 1987 Chesapeake Bay Agreement. Furthermore, if a species is migratory and spends some of its life in coastal or oceanic waters, it is subject to the jurisdiction of federally appointed bodies. In these cases, management is coordinated either individually or jointly by the Atlantic States Marine Fisheries Commission (coastal species within 3 miles of the coast), and the Mid-Atlantic Fishery Management Council (3 to 200 mile offshore zone).

All fisheries within the Bay are managed on an individual species basis. Richkus et al. (1992) provide an excellent summary of the approaches used. It is important to note that all management approaches identified by Richkus et al. (1992) rely on similar conceptual models. In all of the models applied, fluctuations in abundance of individual species result from changes in rates of four processes: recruitment, growth and natural and fishing mortalities (Fig. 6). Increases in the rates of recruitment and growth lead to increases in abundance and biomass. In contrast, increases in the rates of natural and fishing mortalities cause decreases in abundance and biomass.

Elaborations of the conceptual model (Fig. 6) have produced management models, such as the surplus production, spawner-recruit analyses and yield per recruit models, that were originally formulated by Schaefer (1954), Ricker (1954) and Beverton and Holt (1957), respectively. Recent efforts have expanded these themes to include age and spatial structure, stochastic processes and environmental covariates in the basic approaches (Gulland 1988). Principal

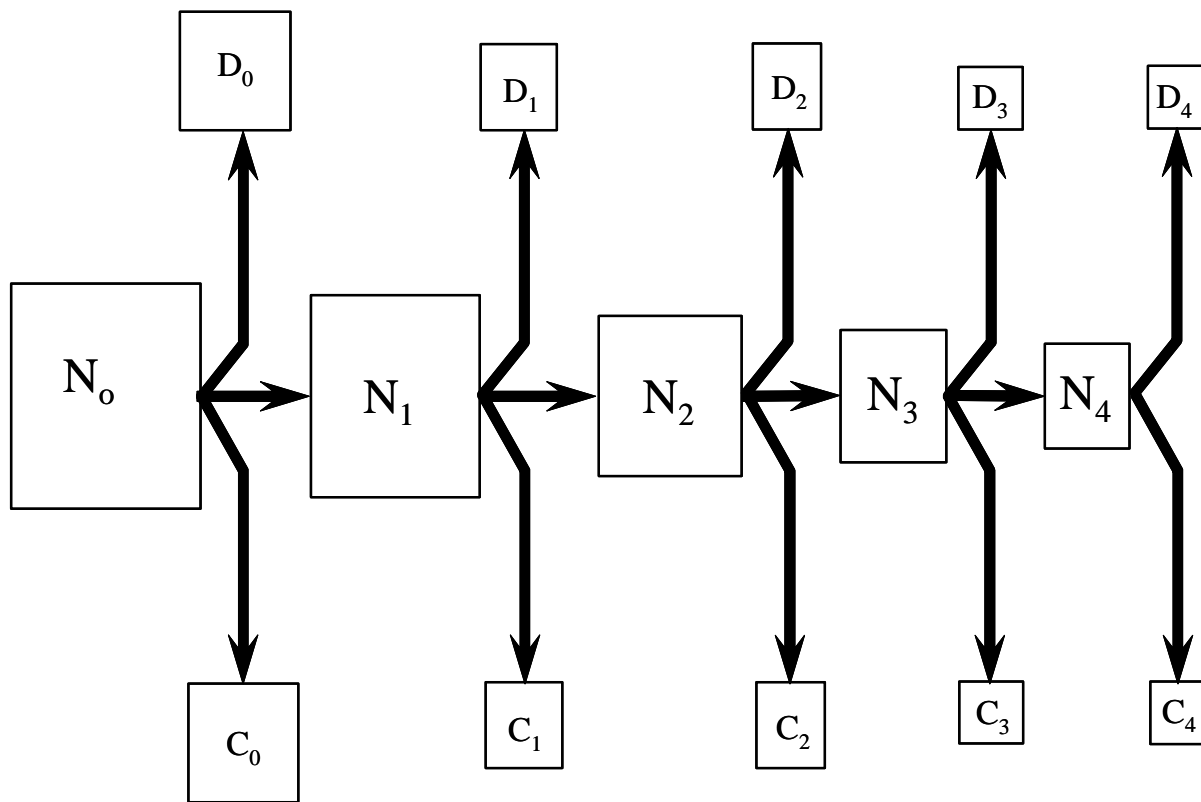


**Figure 5.** The importance of the recreational catch. A) time series of the estimated total recreational catch for five species. B) time series of the estimated total number of fishing trips. Data are the summed values from the NMFS marine recreational fishery survey for the State territorial waters of Maryland and Virginia. These figures include catches from all estuaries, enclosed bays and other marine waters not directly offshore.



**Figure 6.** Conceptual representation of processes regulating stock size. Signs on each flow arrow indicate whether processes act to increase or decrease stock size. Recruitment is defined as the number of young individuals entering the fishable stock annually. Natural mortality is the number of deaths from all causes other than fishing.





Virtual Population

**Figure 7.** Conceptual representation of single species VPA (after Sparre 1991). Boxes labeled  $N_0 - N_4$  represent the abundance of individuals in age classes 0 - 4. Boxes labeled  $C_i$  and  $D_i$  represent the associated losses due to fishing and natural mortality, respectively.

among these are the age-structured, cohort-based methods derived by Gulland (1965) and Pope (1972) that lead to virtual population analysis (VPA), which underlies much of modern marine fisheries management. The VPA is an elaborate accounting procedure. Essentially, VPA is an application of the basic conceptual model presented in Figure 6 to every age (or stage) of a fish's life history. We illustrate this conceptually in Figure 7. The "virtual population" is the summed catches and mortalities over all ages. If the natural mortality schedule is known, then the age-specific abundances and fishing mortality rates required to produce the observed catches can be hindcasted from the oldest age class to the youngest (Hilborn and Walters 1992). The application of a VPA or other model is dictated by both the biology of the species concerned and the availability of biological and fishery data to parameterize the models. Richkus et al. (1992) identify deficiencies in data as a principal factor limiting application of the more sophisticated models, such as VPA, to the Chesapeake Bay's natural resources.

### **E. Species Interactions in Fisheries Models; The Development of Multispecies Approaches.**

Despite increases in sophistication, current fisheries modeling and management approaches, clearly show the heritage of work conducted during the first half of this century. Most importantly, practices continue to be dominated by a single-species approach which treat the harvested resource as largely isolated from the ecological community within which it lives. Effectively, management has focussed primarily on the interaction between stock size and fishing mortality and recruitment;

ignoring growth and natural mortality (Fig. 6). However, all exploited species exist in complex ecosystems, in which individual species interact with one another. These interactions exert influence on the harvested species through the processes of natural mortality, growth and recruitment illustrated in Figure 8. The single-species approach, which discounts, *a priori*, several potential controllers of stock size, may have serious limitations, especially when applied to tightly coupled ecosystems. Moreover, single-species approaches also ignore the fact that the harvest in many, even perhaps most, fisheries is not restricted to a single species. Many non-targeted species are taken as bycatch (i.e. a technical interaction), and on occasion, for example in tropical shrimp fisheries, can represent the major fraction of the total harvest (Alverson et al. 1994). Such biological and technical concerns are motivating the development of multispecies approaches to fisheries management.

Early attempts to incorporate interactions between species into fisheries management models can be found in Larkin's (1963) and Riffenburgh's (1969) research. However, the first clear statement of the need for a multispecies approach came in 1977, when the U.N. Food and Agriculture Organization convened a panel of experts to examine the need for a broader multispecies approach to management (FAO 1978). This report highlights the potential importance of multispecies management approaches for both biological and technical reasons. We summarize the features of a fishery that have motivated the development of multispecies approaches in Table 1.

Subsequent to the FAO (1978) report, fishery scientists began to explore the

Table 1. Characteristics of fisheries motivating a multispecies approach

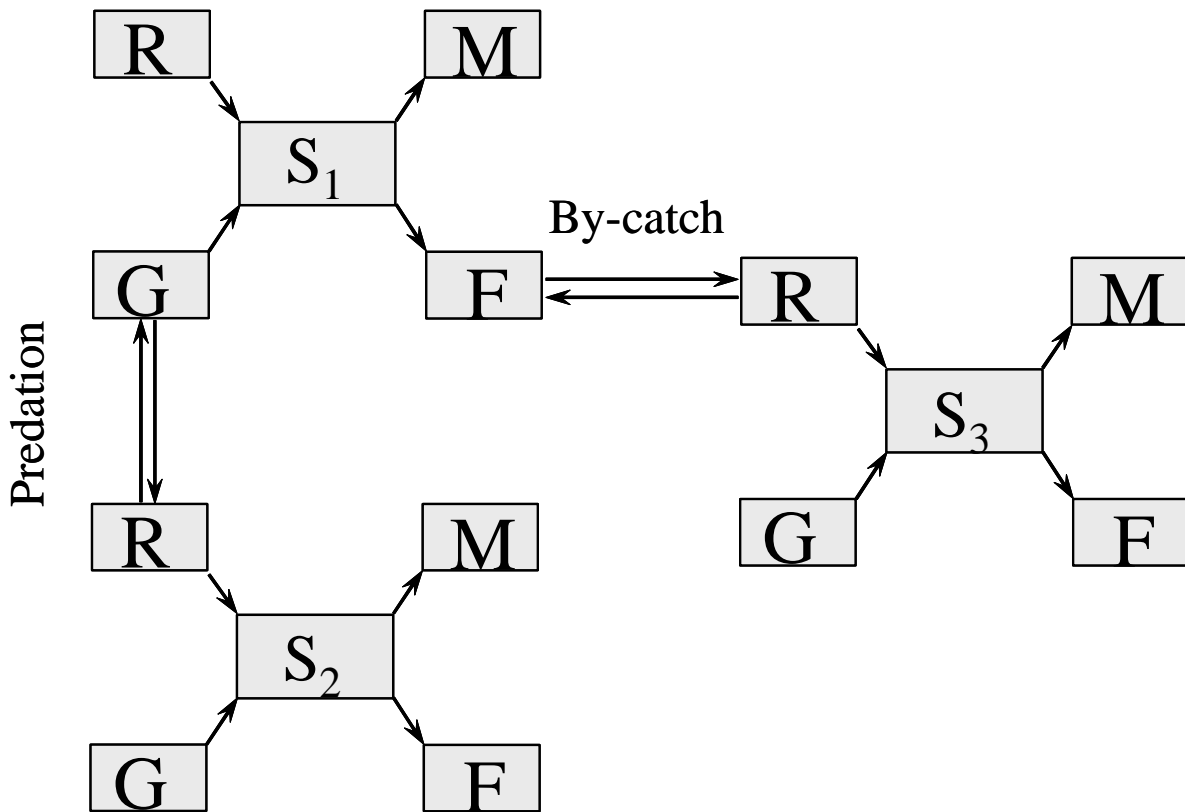
Category	Examples
Technical	Incidental bycatch of non-target species in targeted fisheries Use of non-specific fishing gear Exploitation of mixed species schools
Biological	Extensive predation mortality Competition among species for food Similar responses to environmental forcing

importance of multispecies approaches (Sissenwine and Daan 1991). Research promoted by the International Council for the Exploration of the Sea (ICES) was central in these developments. In 1977, Andersen and Ursin developed a detailed ecosystem model of the North Sea that, for the first time, permitted estimation of the integrated effects of predators and competitors on harvested species. Simultaneously, Daan (1973, 1975) recognized the potential importance of including interspecific interactions in understanding the dynamics of cod (*Gadus morhua*) in the North Sea. The combination of these two developments and the FAO report lead to the international “year of the stomach” in 1981, which sought to provide detailed spatial and temporal data on the trophic interactions within the North Sea through intensive stomach sampling (Sissenwine and Daan 1991).

Based upon the concepts in Andersen and Ursin’s (1977) simulation model, four groups independently developed management approaches that have come to be known as Multispecies Virtual Population Analysis (MSVPA). Conceptually, MSVPA extends the single-species approach by making the natural mortality and growth

components of Figure 6 functions of the abundances of predators and competitors, respectively. This gives rise to a linked system in which the effects of changes in abundance of a single species can be mapped onto changes in abundance of other species through its effects on its prey and competitors (Fig. 8). The only additional complexities are that each stock is divided into a series of age classes and many species of minor importance are combined into a single category.

Much of the early research on MSVPA techniques was applied to the North Sea groundfish fishery (Sissenwine and Daan 1991). ICES again played a prominent role. The wide application of MSVPA techniques has been limited because they are data intensive, and because they often produce counter-intuitive predictions of population behavior under heavy exploitation (e.g. Sparholt 1994). Moreover, because MSVPA models are complex, their mathematical behavior remains poorly characterized or understood (Magnusson 1995). Consequently, MSVPA models are often used heuristically to investigate probable behavior of fished populations under different exploitation strategies. To date, managers have been reluctant to accept them



**Figure 8.** Conceptual examples of biological and technical stock interactions for three stocks  $S_1$ ,  $S_2$  and  $S_3$ . Stocks  $S_1$  and  $S_2$  are linked by a predator-prey relationship (biological interaction) in which adults of  $S_1$  eat juveniles of  $S_2$ . Thus increased predation by  $S_1$  leads to increases in its growth rate and decreases in recruitment to  $S_2$ . Stocks  $S_1$  and  $S_3$  are linked by a by-catch issue (technical interaction) in which juveniles of  $S_3$  are a bycatch in a targeted fishery for  $S_1$ . Increased  $F_1$  leads to a reduced recruitment to  $S_3$ . R = recruitment; G = growth in weight; M = natural mortality; F = fishing mortality.

as a primary management tool (Brugge and Holden 1991). However, MSVPA models have been applied to several temperate fisheries (see examples in Pope and Macer 1991; Sparholt 1991).

## **F. Objectives.**

In this literature synthesis we explore the background of and justification for multispecies management. We consider and discuss possible advantages of the approach to ensure sustainable harvests of Chesapeake Bay's living resources. We review the approaches that have been adopted in other ecosystems to determine which multispecies approach might be appropriate in Chesapeake Bay and, in doing so, we identify gaps in our understanding that must be filled before multispecies approaches can be fully evaluated or adopted. Specifically, our objectives are to:

1. Analyze fishery-dependent and fishery-independent datasets to explore the degree to which species interactions can be identified and to determine if there is significant evidence of correlated patterns in abundance of important components of the Bay's fisheries resources.
2. Review patterns of interaction among species within the Chesapeake Bay ecosystem. We focus particularly on identifying resident species and seasonal visitors to the ecosystem that interact closely. We then explore the potential impact of these strong interactors on the fisheries within the Bay.
3. Review the concepts and approaches that underpin multispecies management. The review is conducted within a classification framework of multispecies approaches that recognizes descriptive, dynamical, multivariate and integral categories. We discuss the assumptions and data requirements of each approach.
4. Address the potential application of multispecies management in the Chesapeake Bay. Specifically, we assess whether the conditions that lead to multispecies approaches in other systems are present in the Bay and its fisheries. We identify which, if any, of these approaches is most appropriate for Chesapeake Bay, and identify gaps in our knowledge that currently preclude either the evaluation or application of multispecies management in the Bay.

## II. PATTERNS IN FISHERIES HARVESTS AND ABUNDANCES.

In this chapter we analyze the principle fisheries within the Chesapeake Bay to identify possible species interactions. As a preliminary step to assess the utility of a multispecies approach to Chesapeake Bay fisheries management, we explore the patterns in landings to identify the extent to which species landings are correlated.

### A. Data Sources and Preparation.

We examined landings data provided to NMFS by the Virginia Marine Resources Commission and Maryland Department of Natural Resources for the period 1880 - 1992. Landings data for 1993 -1994 are preliminary, due in part to statistical reporting changes in Virginia. Thus, while we have analyzed only data up to 1992, we illustrate landings for 1993-1994. We were unable to standardize catch to fishing effort because effort data are unsystematically collected and recorded for most species. Rothschild et al. (1981) identified the same problem 15 years ago. As a result, there is some uncertainty whether trends in the data reflect changes in overall population abundance or changes in fishing effort. To gain insight on trends in population abundances, we also analyzed survey data from the Virginia trawl survey series provided by the Virginia Institute of Marine Science. These research-survey data are standardized to effort and changes in catch-per-unit-effort reflect changes in underlying population abundances of some species.

Historically, landings records date to 1880 but are inconsistent until 1929. Changes in reporting and recording methods complicate analysis and interpretation of landings trends. Prior to 1956, landings

reported for Maryland and Virginia were aggregated Chesapeake Bay and Atlantic catches landed at ports in the two states. After 1956, the landings from the Bay and Atlantic sources were separated, and thus, from 1957 onwards harvest data are specific to the waters of the Bay. For some species, such as butterfish (*Peprilus triacanthus*) and scup (*Stenotomus chrysops*) which are caught almost exclusively in Atlantic waters, the distinction is not problematic since all harvests prior to 1956 represent ocean-side landings. However, for species which are caught both within the Bay and in the Atlantic, no simple partitioning is possible. A correlation analysis for principal Bay-specific and ocean-specific landings from 1957-1990 was attempted to partition landings prior to 1957. However, only the correlation for weakfish was significant ( $r = 0.76$ ,  $n = 21$ ). Thus, we were unable to remove this source of bias from our analysis.

There are difficulties in determining menhaden catches, the most abundant species landed by weight. Since 1977 only two firms have fished for menhaden, thus preventing NMFS from reporting the catch, because reporting it would allow each of the companies to calculate the catch of the other. Consequently, from 1977, there is an enormous increase in the landings of “unclassified finfish” in the data set, increasing by almost two orders of magnitude from under 3,000 to 200,000 t between 1976 and 1977. Obviously, the increase represents menhaden, and we have used data for “unclassified finfish” as a surrogate for menhaden from 1977 onwards. However, approximately 1% of this catch that we ascribe to menhaden is unclassified finfish other than menhaden and cannot be

separated in the statistics. Our analyses are not compromised by this small error. A second problem with menhaden landings records is a complete lack of data for menhaden in 1980-1983. To estimate the missing landings data for these four years we correlated menhaden landings for the Chesapeake region with the total U.S. menhaden catch for the period 1963-1977 (Fig. 9A). Menhaden landings from Chesapeake Bay were significantly correlated with U.S. menhaden landings:

$$\text{Bay Menh.} = 0.59 \text{ @ U.S. Menh.} - 11.4 \times 10^3, \quad r^2 = 0.57, \quad n = 13 \quad (1)$$

Overall, there was a satisfactory pattern in the residuals of the relationship between Chesapeake Bay menhaden and total U.S. menhaden landings (Fig. 9B). Consequently, we derived an estimate of Chesapeake Bay menhaden landings for the four years 1980 -1983 using U.S. menhaden landings in Equation 1.

No fisheries landings data were available for 1943. To fill this gap, we estimated landings for each species for this year as:

$$\text{Catch}_{1943} = \frac{\text{Catch}_{1942} + \text{Catch}_{1944}}{2} \quad (2)$$

where the subscripts refer to year.

The final data set, developed as described above, was analyzed to determine the presence and importance of multispecies patterns in the Chesapeake Bay. A necessary step, though not a direct objective of this literature synthesis, was an analysis of trends in landings of individual species. For brevity, we will only summarize them

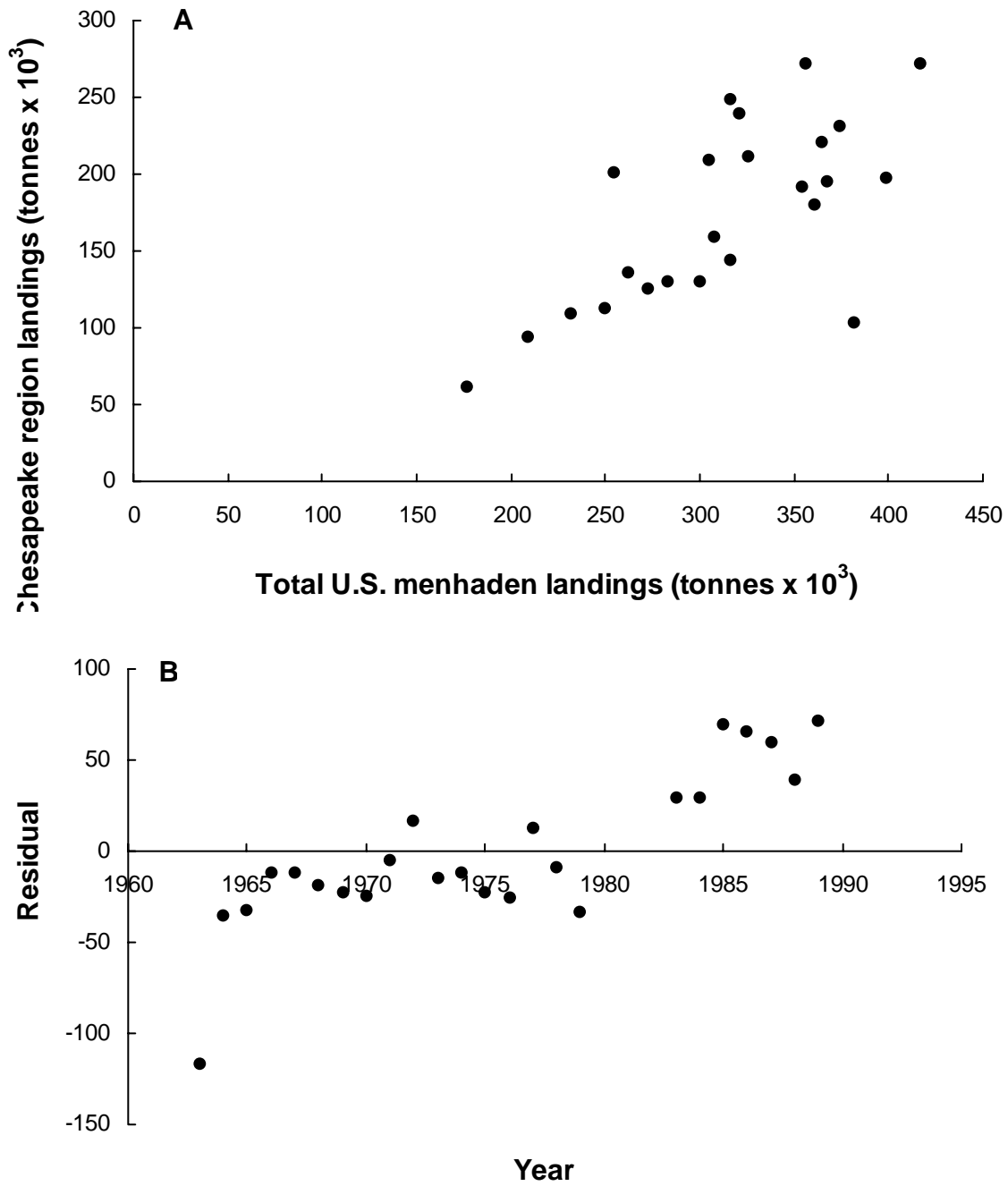
here. There is a strong increasing trend in total landings. However, when one considers fluctuations around this trend, total landings in the current year are strongly related only to landings in the previous year. Such time series are said to be autoregressive. This form of autoregressive relationship, with a clear relationship only to the year immediately prior, describes a time series that exhibits abrupt changes from year to year. The landings time series for menhaden, oysters, soft-shell blue crab and alosids (*Alosa* spp.) show similar patterns. In contrast, striped bass and hard-shell blue crab exhibit a different pattern. Landings for each of the latter two species show an autoregressive pattern in which current landings are best predicted from landings over the last three years. This form of autoregressive relationship describes a landings time series in which there are more gradual fluctuations, with distinct periods of above average, and below average landings.

In the following sections we present the analysis of multispecies patterns. Simple graphical techniques and multivariate statistical approaches were used to determine the extent of multispecies interactions. All statistical tests were conducted using SAS v6.10. Details of the statistical approaches are provided at the beginning of each section.

## **B. Multispecies Patterns.**

### 1. Graphical analysis.

To detect multispecies trends in the fish landings in the Chesapeake Bay, we developed plots of rank order of abundance in catches (Fig. 4) and time series of multispecies groupings (Figs. 10-17). No



**Figure 9.** Relationships between landings data used to estimate menhaden landings in 1980-1983. A) Relationship between Chesapeake Bay menhaden and total U.S. coastwide menhaden landings for 1963 - 1977. B) Annual residuals of regression relationships between Chesapeake Bay menhaden and total coastwide menhaden landings for 1963-1977.



formal statistical analysis was conducted on either the rank abundances or on the time series. The length of most time series precluded application of formal time series approaches. However, visual inspection was sufficient to detect the presence of general trends or striking coherence among time series.

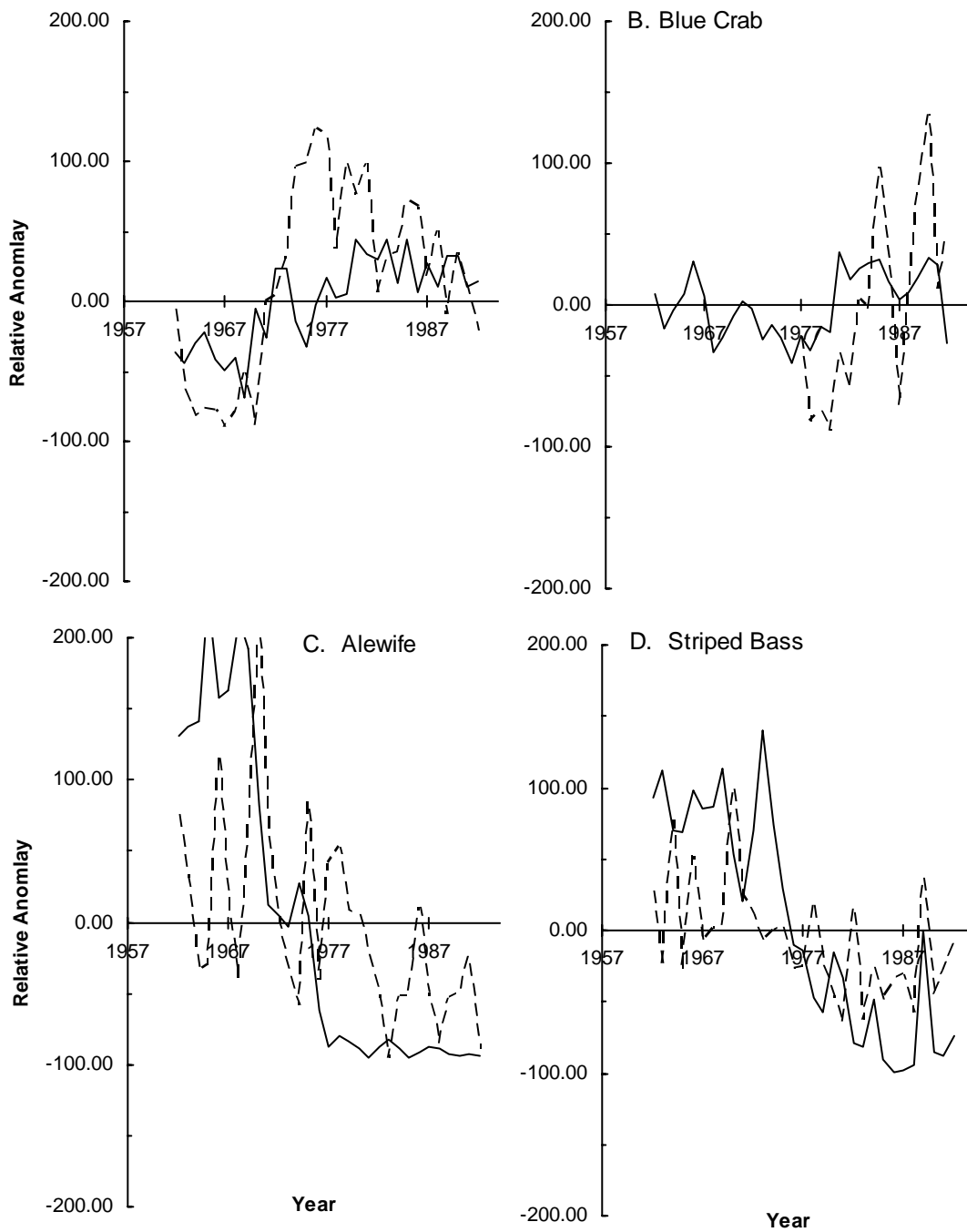
The rank order of commercial landings, which may index abundance in some cases (e.g. oyster), has remained stable since the 1930's (Fig. 4). Menhaden, blue crab, oyster, spot and alewife (*Alosa pseudoharengus*) have dominated the landings. However, the evenness of the catch has declined. In the 1930's, the landings of menhaden, blue crab, croaker, oyster, alewife, weakfish, and American shad combined to produce 95% of the total landings. By the end of the 1980s, menhaden and hard-shell blue crabs alone produced 95% of the landings. This pattern is driven by the increases in crab and menhaden catches (Figs. 2A, B) and concomitant declines in oyster landings (Fig. 2C).

It is important to note that the historical landings pattern likely does not reflect underlying changes in the abundance of some species, but is a consequence of shifting effort that reflects changes in the marketplace. To assess the extent to which landings data reflected underlying abundances we compared the landings time series with equivalent fishery-independent time series collected by the Virginia Institute of Marine Sciences. To aid presentation, we present representative comparisons as anomaly plots in which the time series are plotted as percent deviations from the long term mean (Fig 10A-D). Direct comparison of these plots is difficult because the VIMS survey focuses on younger, pre-recruit fishes. Thus, for some species, the two time

series should be lagged to reflect age at recruitment. Moreover, the period of lag will be species-specific, preventing use of a general correction factor. However, for most of the principal species the gross pattern of the two time series is similar. We interpret this to mean that while the landings data are not equivalent to abundance, they do reflect underlying changes in abundance and are consequently, an adequate surrogate in our analyses.

To explore multispecies patterns in the data set, we grouped species that were reported in commercial landings into four broad categories. Some species were assigned to more than one category. The four categories were taxonomic, trophic, habitat, and habit. Groups were designated based upon two taxonomic categories: 1) families and 2) finfish vs. shellfish. Trophic group membership was determined by the predominant items in the diet. We defined three trophic groups (planktivore, piscivore, benthivore). Habitat groupings were based upon whether an animal was pelagic or demersal and with respect to the salinity range within which it occurs. Finally, three habit groups (resident, seasonal resident, occasional) were defined. Unless noted, data for menhaden were excluded from these analyses because they dominate most trends. Details of the classification are presented in Table 2.

Landings of non-menhaden finfish and shellfish have varied considerably (Fig. 11). Several features of this figure are noteworthy. Between 1920-1960 non-menhaden finfish and shellfish landings were of similar magnitude. Moreover, they demonstrate a high degree of coherence ( $r = -0.61$  at lag = 4 yrs. See Table 4 for details). However, since 1970, the two time series have diverged as the non-menhaden finfish



**Figure 10.** Anomaly plots for commercial landings time series (solid line) and VIMS trawl survey data CPUE (dotted line) for four species.

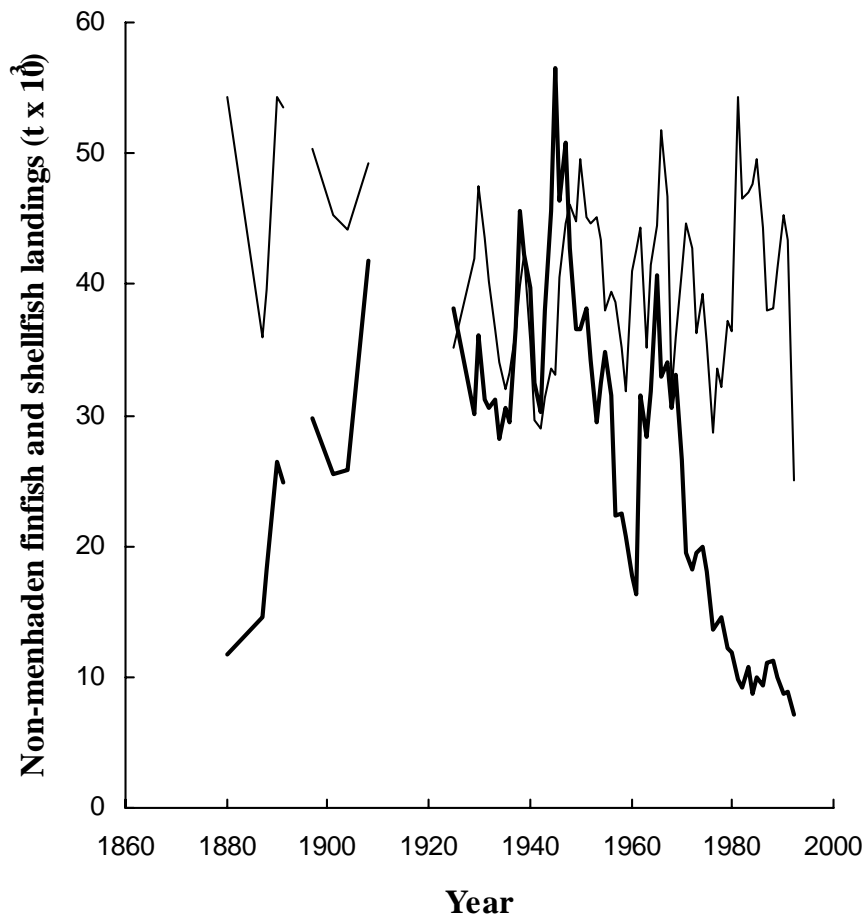
Table 2. Classification of common Chesapeake Bay species present in the commercial catch into taxonomic, trophic, and life history groups. Taxonomic groups were shellfish (S) and finfish (F), based upon taxonomic order. Trophic criteria were planktivore (P), benthivore (B), piscivore (C) and detritivore (D) based upon the primary dietary items. Life history groups were resident (R), seasonal (S) and occasional (O). Resident species were those which remained in the Bay for the entire year, and completed the bulk of their life history within the Bay. Blue crab were included in this group even though they do spend a brief period as larvae outside of the Bay. Seasonal species were ones that have an obligate estuarine phase, and prolonged periods outside of the Bay. Spot and croaker were included in this group as adults leave the bay system to spawn. The final group were occasional visitors, which were defined as ones that are caught in Bay waters, but are not obligate estuarine species

Common Name	Genera	Species	Fish/ Shellfish	Trophic Status	Habit	BAY spawner
Windowpane Flounder	Scophthalmus	aquosus	F	B	O	n
Sturgeon	Acipenser	oxyrhincus	F	B	S	y
Bowfin	Amia	calva	F	C	R	y
Eels, Common	Anguilla	rostrata	F	C	S	n
Summer Flounder	Paralichthys	dentatus	F	C	S	n
Crevalle	Caranx	crysos	F	C	O	y
Crappie	Pomoxis	spp.	F	C	O	y
Thread herring	Opisthonema	oglinum	F	P	O	n
Gizzard Shad	Dorosoma	cepedianum	F	P/D	R	y
Hickory Shad	Alosa	mediocris	F	C	S	y
Menhaden	Brevoortia	tyrannus	F	P	S	n
Alewives	Alosa	psuedoharengus	F	P	S	y
Am. shad	Alosa	sapidissima	F	P	S	y
Dolphinfish	Coryphaena	hipprus	F	C	O	n
Carp	Cyprinus	carpio	F	C/D	R	y
Pikes Or Pickerels	Esox		F	C	R	y
Red Hake	Urophycis	chuss	F	B	O	n
Pigfish	Orthopristis	chrysoptera	F	B	O	n
Catfish	Various		F	C	R	y
Tautog	Tautoga	onitis	F	B	O	n
Gar	Lepisosteus	osseus	F	C	R	y
Tripletail	Lobotes	surinamensis	F	C	O	n
Snapper	Various		F	C	O	n
Tilefish	Lopholatilus	chamaeleonticeps	F	B	O	n
Mullet, Striped	Mugil	cephalus	F	P	O	n
White perch	Morone	americana	F	C	R	y
Striped Bass	Morone	saxatilis	F	C	S	y
Yellow perch	Perca	flascens	F	C	R	y
Winter flounder	Psuedopleuronectes	americanus	F	B	S	y

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American plaice	Hippoglossoides	platessoides	F	B	O	n
Bluefish	Pomatomus	saltatrix	F	C	O	n
Drum, Black	Pogonias	cromis	F	B	O	y
Drum, Red	Sciaenops	ocellatus	F	B	O	n
Sea Trout, Spotted	Cynoscion	nebulosus	F	C	O	n
Croaker, Atlantic	Micropogonias	undulatus	F	B	S	n
Spot	Leiostomus	xanthurus	F	B	S	n
Weakfish	Cynoscion	regalis	F	C	S	n
Sea Basses	Centropristis	striata	F	C	O	n
Hogchoker	Trinectes	maculatus	F	B	R	y
Sheepshead, At	Archosargus	probatoccephalus	F	B	O	n
Sea Robins	Prionotus	carolinus	F	B	O	n
Clams, Razor	Ensis	directus	S	P	R	y
Clams, Hard	Mercenaria	mercenaria	S	P	R	y
Clams, soft	Mya	arenaria	S	P	R	y
Clams, Surf	Spisula	solidissima	S	P	O	n
Oyster	Crassostrea	virginica	S	P	R	y
Horseshoe Crab	Limulus	polyphemus	S	B	R	y
Crab, At, Rock	Cancer	irroratus	S	B	R	y
Crabs, Blue	Callinectes	sapidus	S	B	R	y
Lobster, American	Homarus	americanus	S	B	O	n
Shellfish, Other			S	B	R	y
Turtle, Green (Sea)	Chelonia	mydas	R	B	O	n
Turtle, Loggerhead (Sea)	Caretta	caretta	R	B	O	n
Turtles, Snapper	Chelydra	serpentina	R	B	R	y
Turtles, Unc			R	B	R	y

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**Figure 11.** Time series of Chesapeake Bay commercial landings by broad taxonomic categories of finfish (heavy line) or shellfish (light line). Note that data for menhaden have been removed from this time series. See Table 2 for taxonomic affiliations of the species used in the analysis.

catch has declined. Landings of non-menhaden finfish have fallen since the late 1940's from almost 60,000 t to less than 10,000 t currently (Fig. 11). This is due principally to decline in alewife landings. In contrast, total shellfish landings, while variable, remained relatively constant through the 1980's.

Within the broad finfish taxonomic group considerable variability exists (Fig. 12). For example, while alewife catches declined dramatically in the mid 1970's, catches of the other anadromous shads and river herrings (blueback herring, *Alosa aestivalis*, American shad, *A. sapidissima*, and hickory shad, *A. mediocris*) declined more gradually over the last one hundred years. Moreover, catches of a freshwater clupeid, the gizzard shad (*Dorosoma cepedianum*), have increased dramatically in recent years, although its landings are small relative to the historical alosid catches (Fig. 12).

Average landings of shellfish have remained approximately constant since 1920, at approximately 40,000 t (Fig. 11). However, the time series of landings of individual species in the group differ widely (Fig. 13). It is clear from Figure 13 that the overall stability of shellfish landings is caused by a replacement of oyster, which declined dramatically during this century, by hard-shell blue crab in the total shellfish landings. It is also evident from this figure that there is coherence between blue crab and both hard and soft clam landings. For example, the time series of both species show unusually low landings in 1963, 1968 and 1978.

We explored patterns in the time series of broad trophic groupings. With menhaden removed, the landings of planktivorous species declined steadily and quite rapidly over the period 1960-1992 (Fig. 14). This

reflects the continuing decline of anadromous clupeoid fishes and oysters. Landings of piscivorous species declined slowly. However, considering that a complete fishing moratorium was imposed on striped bass in Maryland from 1985-1989, the overall trend indicates relative stability in levels of total piscivore landings. Yet, the time series of landings of individual predator species is highly variable, especially for bluefish and weakfish (Fig. 15). Peak years of striped bass landings (1950s to 1970s) coincided with a period of low weakfish and bluefish catches. Moreover, commercial catches of bluefish only peaked in the mid to late 1970's when striped bass catches were declining rapidly and weakfish catches were at low levels. We do not suggest that there is necessarily any causal relationship in these patterns, because they are not simple replacements. We do, however, suggest that they may be driven by the same underlying mechanisms. Shifts in the dominance of piscivore species almost certainly do reflect a combination of varying natural abundances and the behavior of the fishery, which can shift its effort in accord with both availability of fish and marketing opportunities.

In contrast to the planktivores and piscivores, the probable abundances of benthivorous fishes, as indexed by landings, have fluctuated widely around a long-term mean, with no clear long-term trend (Fig. 14). The fluctuations represent variation in the harvests of component species, including spot, croaker and blue crab. With the exception of blue crab, landings of which have increased (Fig. 13), due most likely to increased effort, responses by other species of the benthivore group are more similar (Fig. 16). Catches of benthivore species, primarily croaker and spot, peaked in the 1940s and 1950s at levels exceeding 30,000

t and then declined. Landings of channel catfish have increased recently, although it is uncertain if this reflects an increase in their abundance.

The species are grouped by life history habits in Figure 17. The dramatic decline in catches of the seasonal, anadromous clupeoid fishes is clearly evident since the 1960's. Landings of resident species, primarily blue crab and oyster, have remained surprisingly stable, centered on a long term mean, although they have varied widely. As previously mentioned, the stability reflects the replacement of oyster by blue crab. Variations in resident species landings mostly reflect variability in the landings of hard-shell blue crab. Landings of the occasional species, represented primarily by scombrids have changed relatively little since 1963.

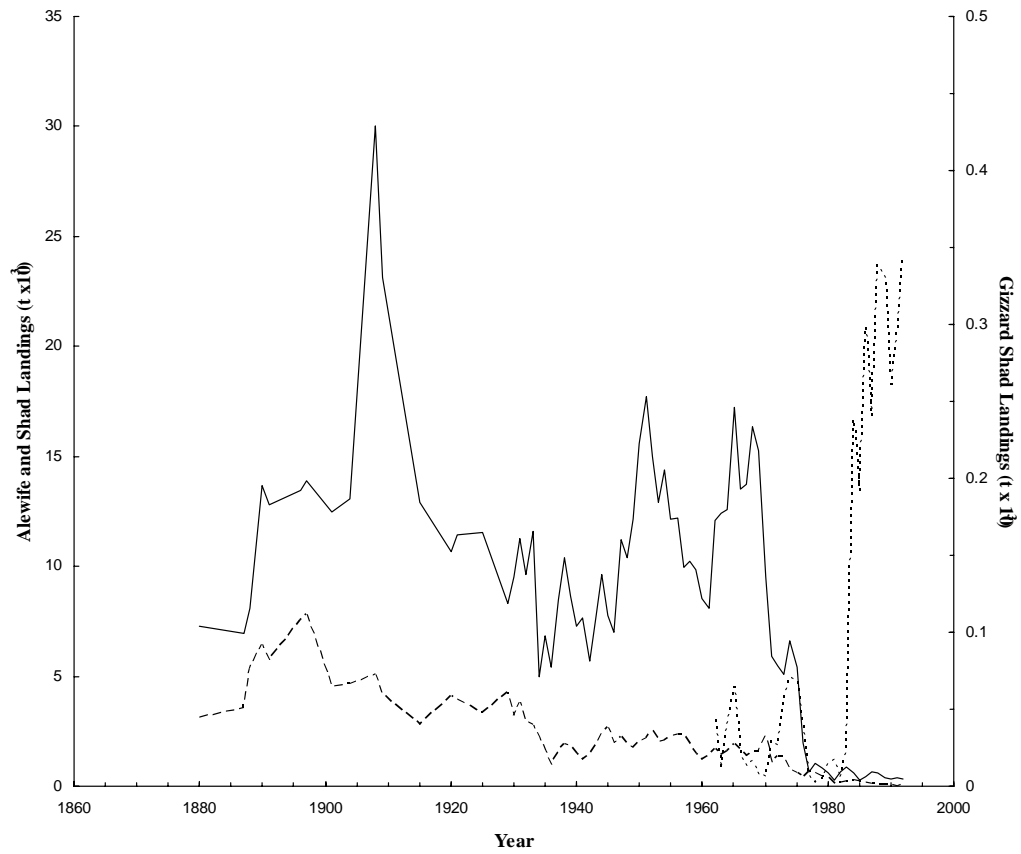
Overall, there is no direct or clear evidence in these summary plots of dramatic species replacements in the Bay, nor of complementary patterns in abundance. However, there is ample evidence of complex patterns of covariation in the landings time series of groups of species. The presence of covariation and its potential to affect harvests and overall well being of resources and fisheries should be considered by resource managers. To explore further the degree of covariation in time series of landings, more sophisticated methods are required.

## 2. Multivariate analyses.

A principle components analysis (PCA) was applied to investigate patterns in the landings of commercial species (Anderson 1984). This approach has been used previously for fisheries data sets to suggest significant groupings of species (Pepin 1990; Rothschild 1991). PCA is a statistical

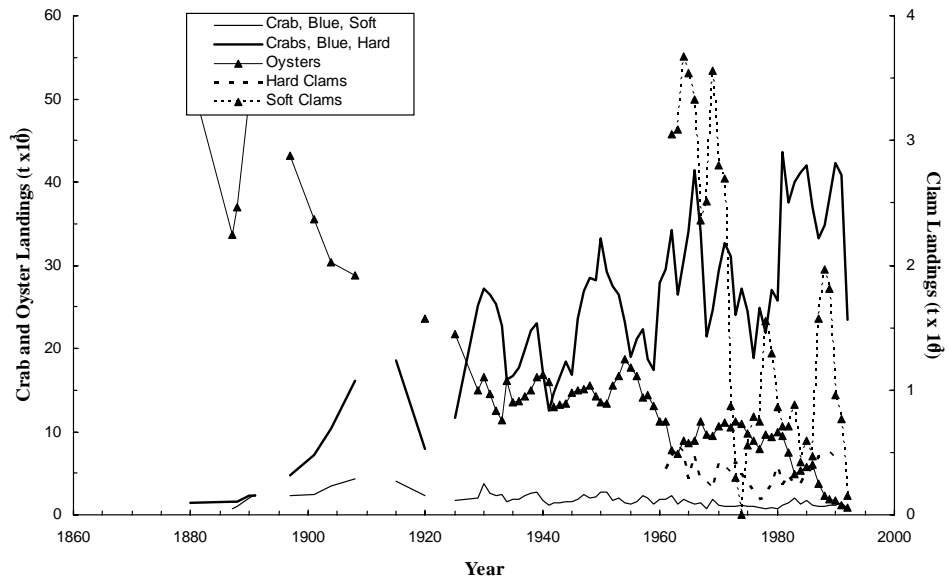
technique that uses the correlation in the data observed for all species to generate a small number of component groupings that account for most of the variability in the original data set. The components are weighted, linear combinations of the original data. PCA calculates the correlation or loading of each species on each component. Species with high absolute loadings on an individual component contribute most to the variation expressed by that component. Furthermore, each component is associated with a percentage of the total variation in the dataset. In this instance, we analyzed the landings anomalies, i.e. the % deviation of annual landings from the mean 1930-1992 landings. This transformation scales all time series equally, removing the effects of absolute amounts of fish landed. Consequently, the analysis will identify those groups that exhibit similar dynamics in their time series. The results are summarized in Table 3 and Figure 18.

The first principle component (PCA1) explained 52% of the variation, and was weighted heavily and positively toward hard-shell blue crabs and negatively toward croaker (Fig. 18). These two species exhibited patterns in their landings time series quite unlike any other species considered. Hard-shell crabs were the only species in the analysis that exhibited a generally increasing trend. The croaker time series is unique in that, except for a peak period in the mid 1940's, croaker landings have been consistently below the long-term mean. Most of the other species exhibited loadings close to 0 on PCA1. One group of fishes, bluefish, striped bass and white perch, that share similar patterns in their landings time series, all exhibit similar loadings on

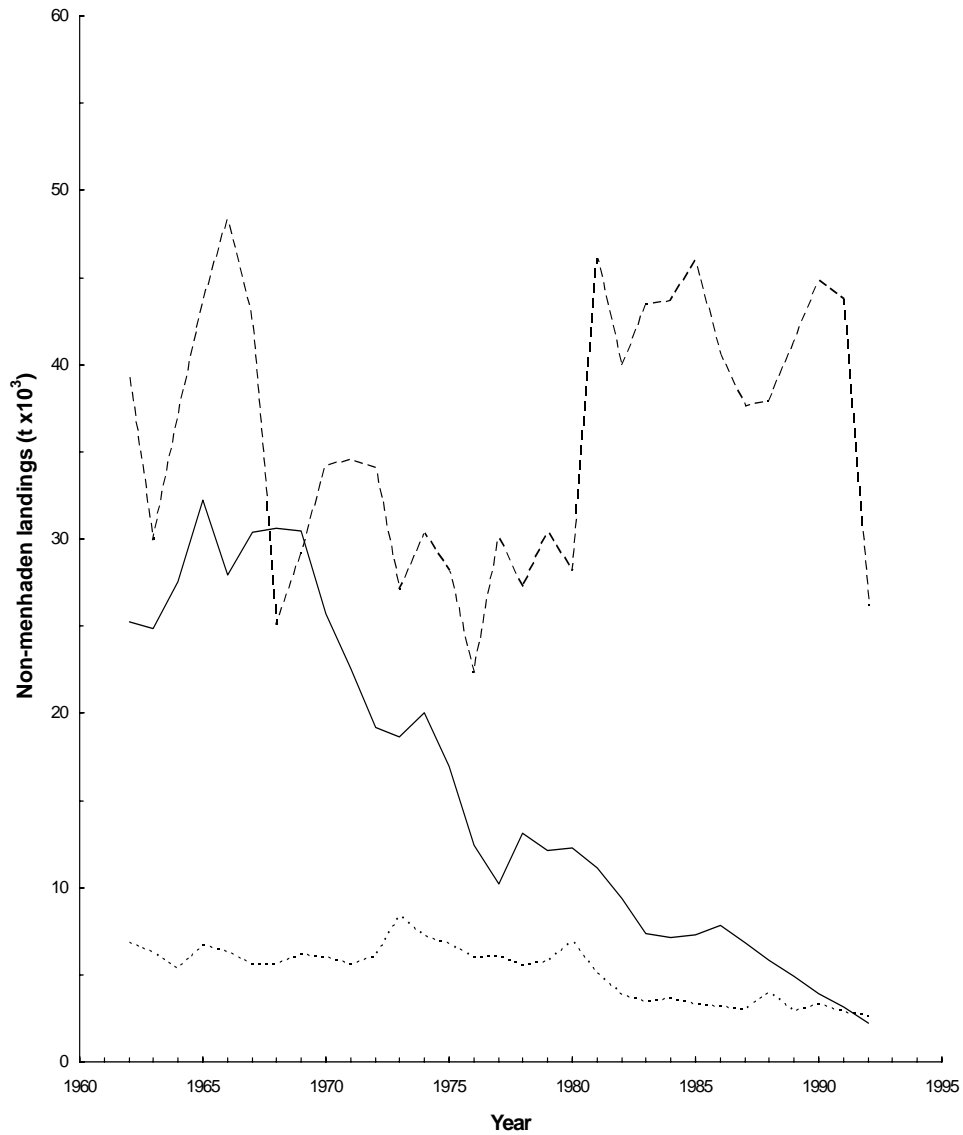


**Figure 12.** Time series of Chesapeake Bay commercial landings for alewives (solid line) other *Alosa* species (dashed line), and gizzard shad (dotted line).

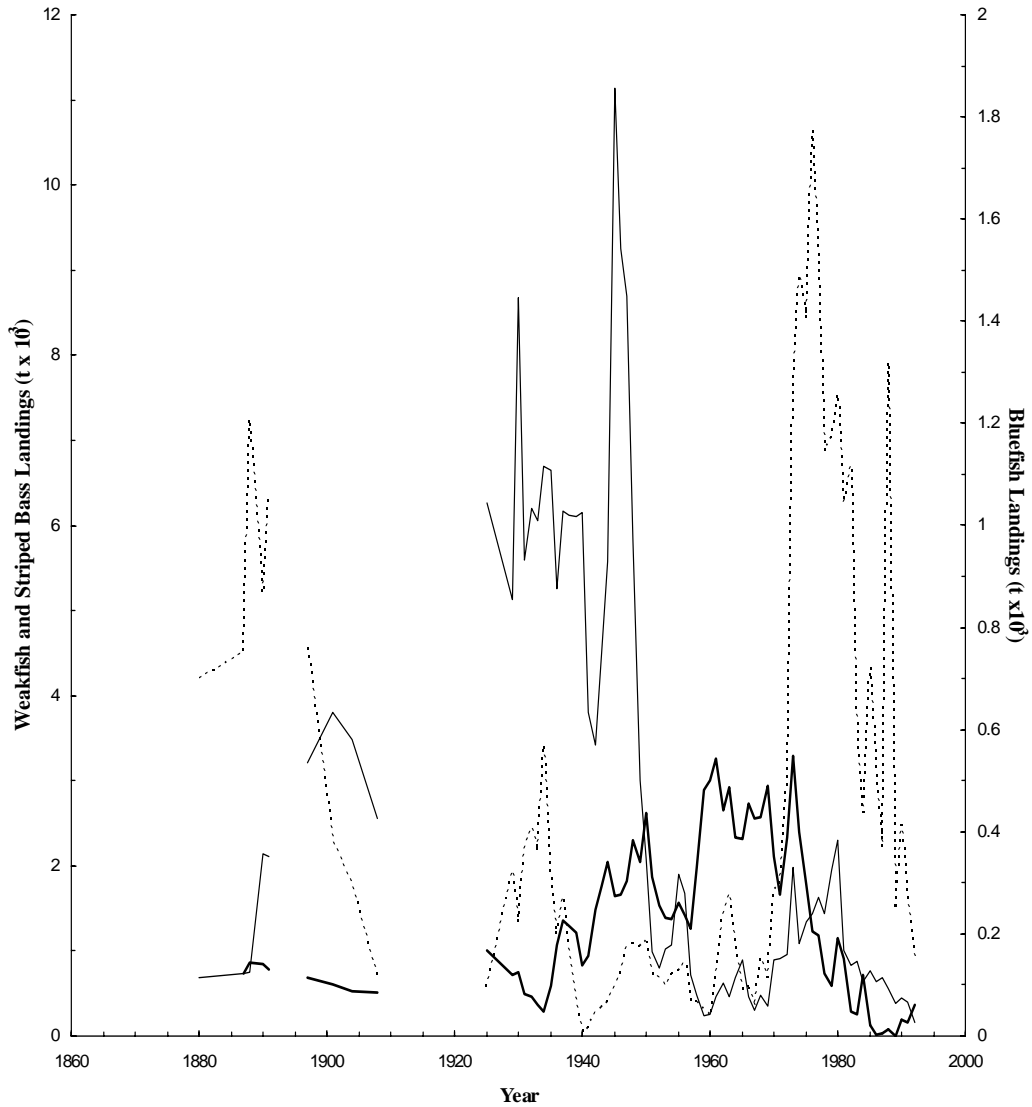




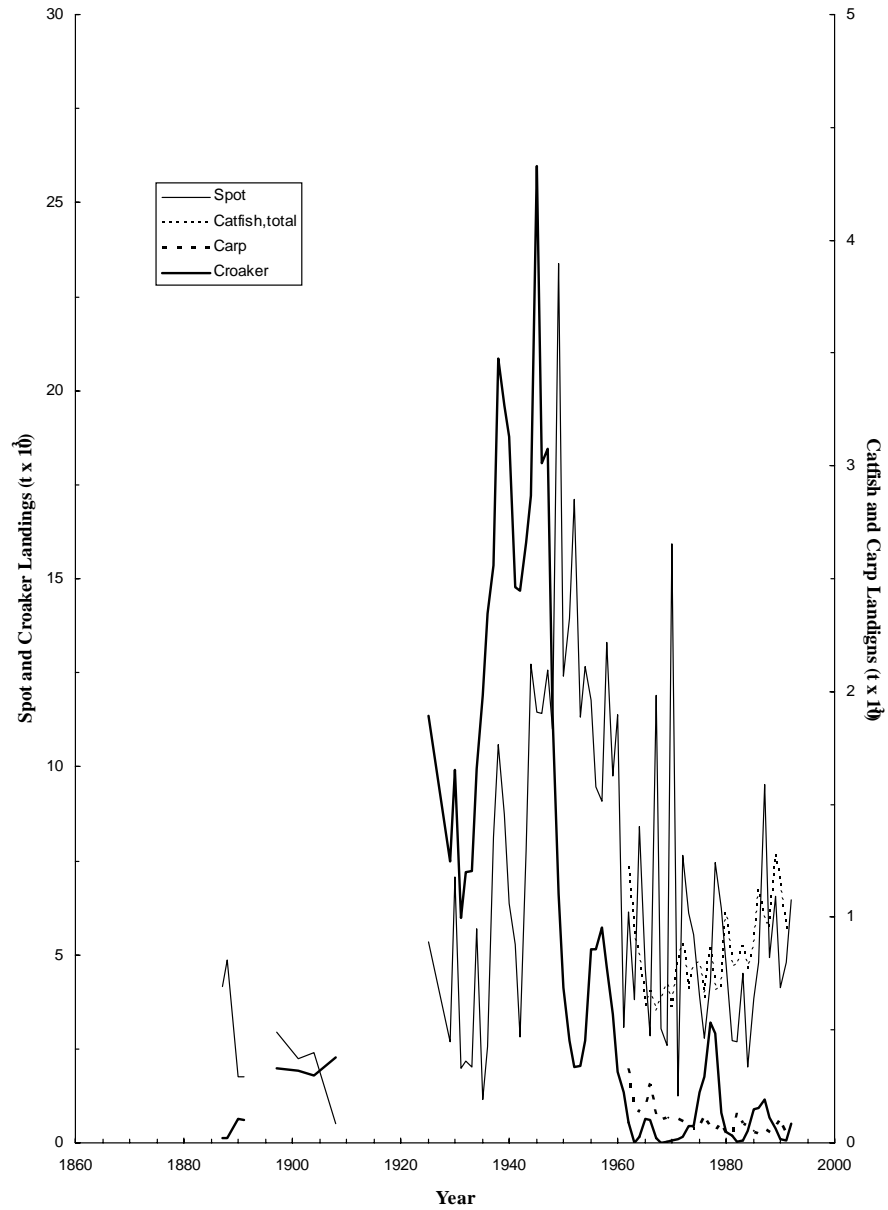
**Figure 13.** Time series of commercial landings for four components of the shellfish harvest from Chesapeake Bay (hard blue crab, soft blue crab, oyster, hard clam).



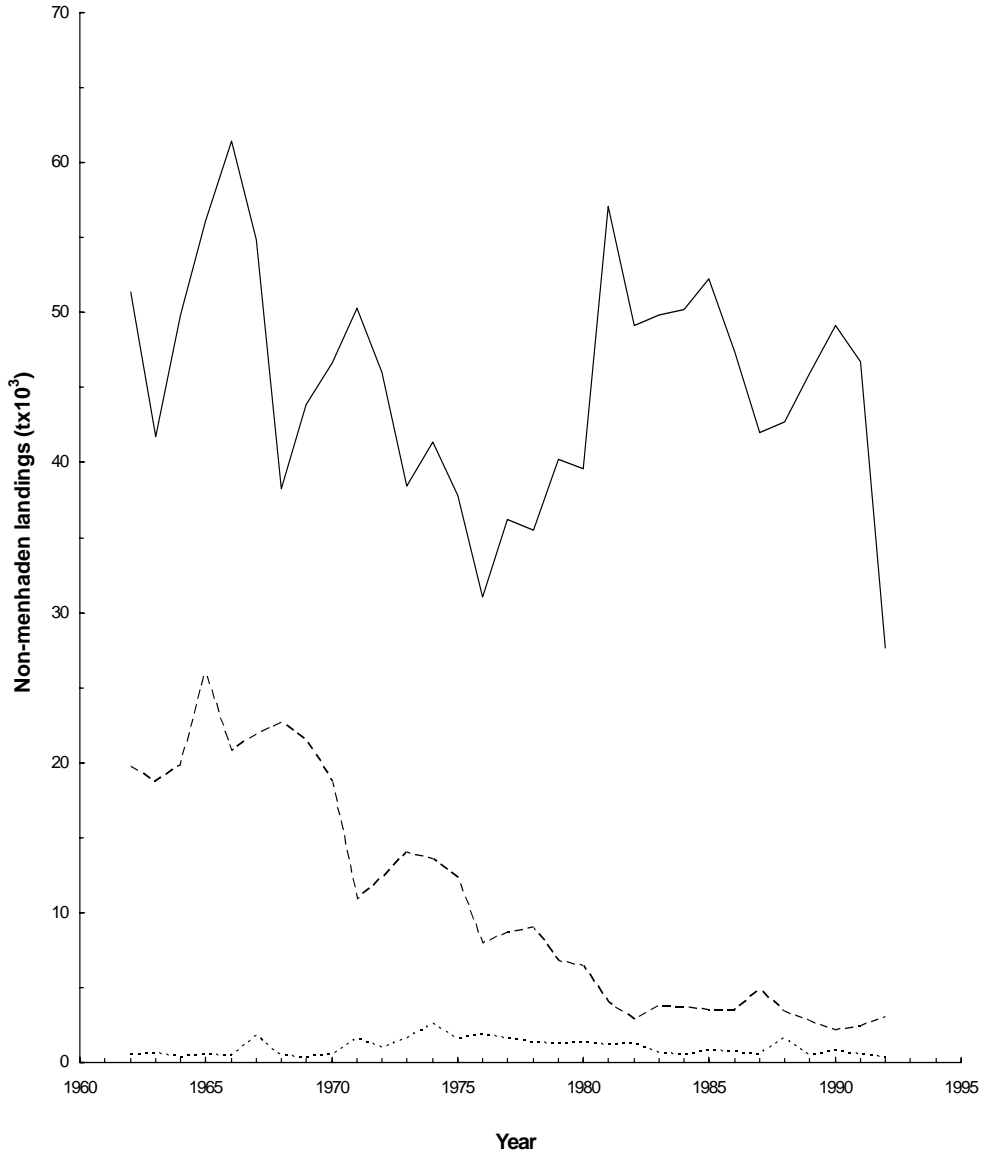
**Figure 14.** Time series of commercial landings by trophic category of planktivore (heavy line), benthivore (dashed line) or piscivore (dotted line) for both finfish and shellfish. Note data for menhaden have been removed from this time series. See Table 2 for details of



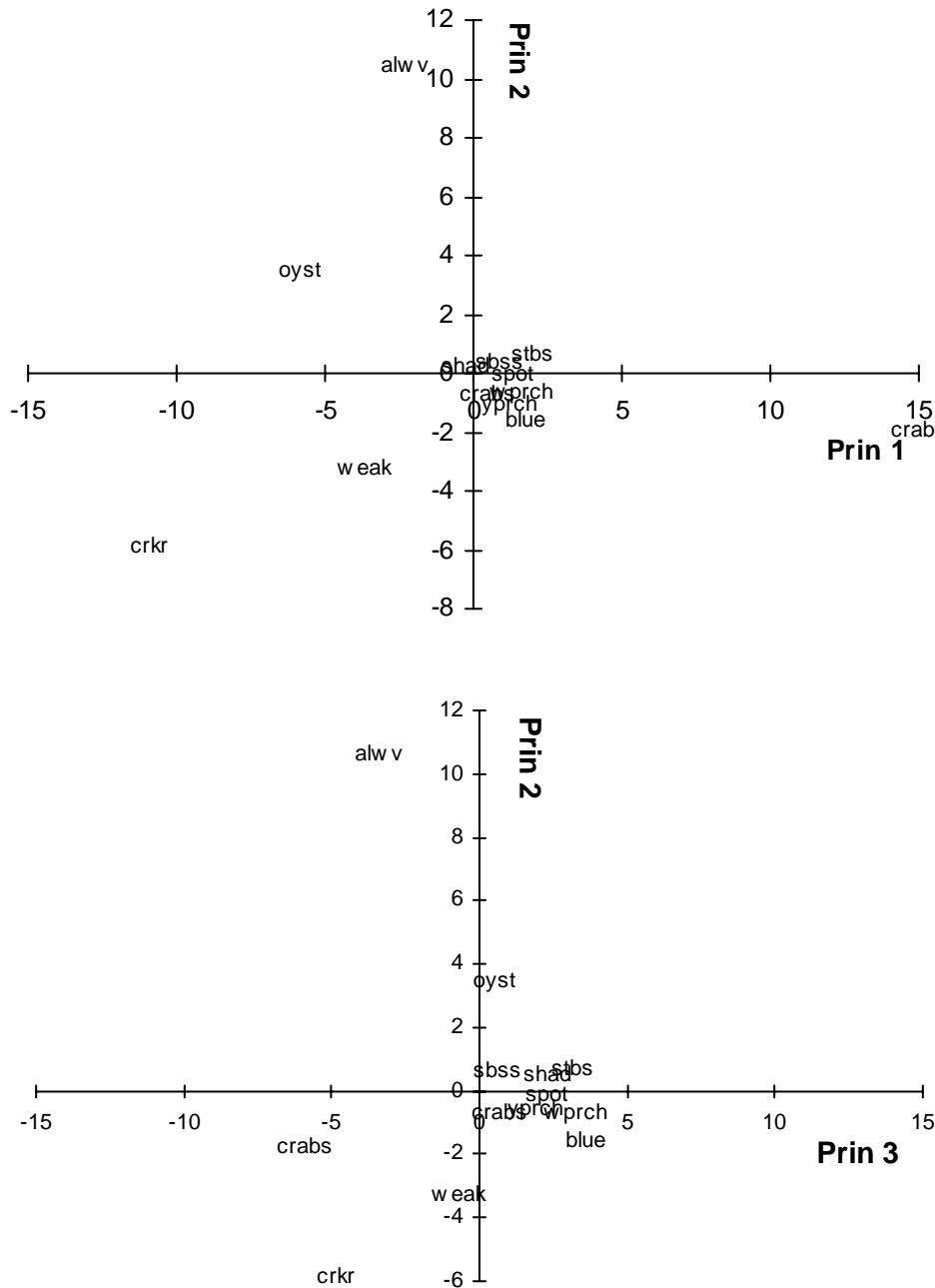
**Figure 15.** Time series of Chesapeake Bay commercial landings for three species within the piscivore guild, striped bass (heavy solid line), weakfish (solid line), and bluefish (dotted line).



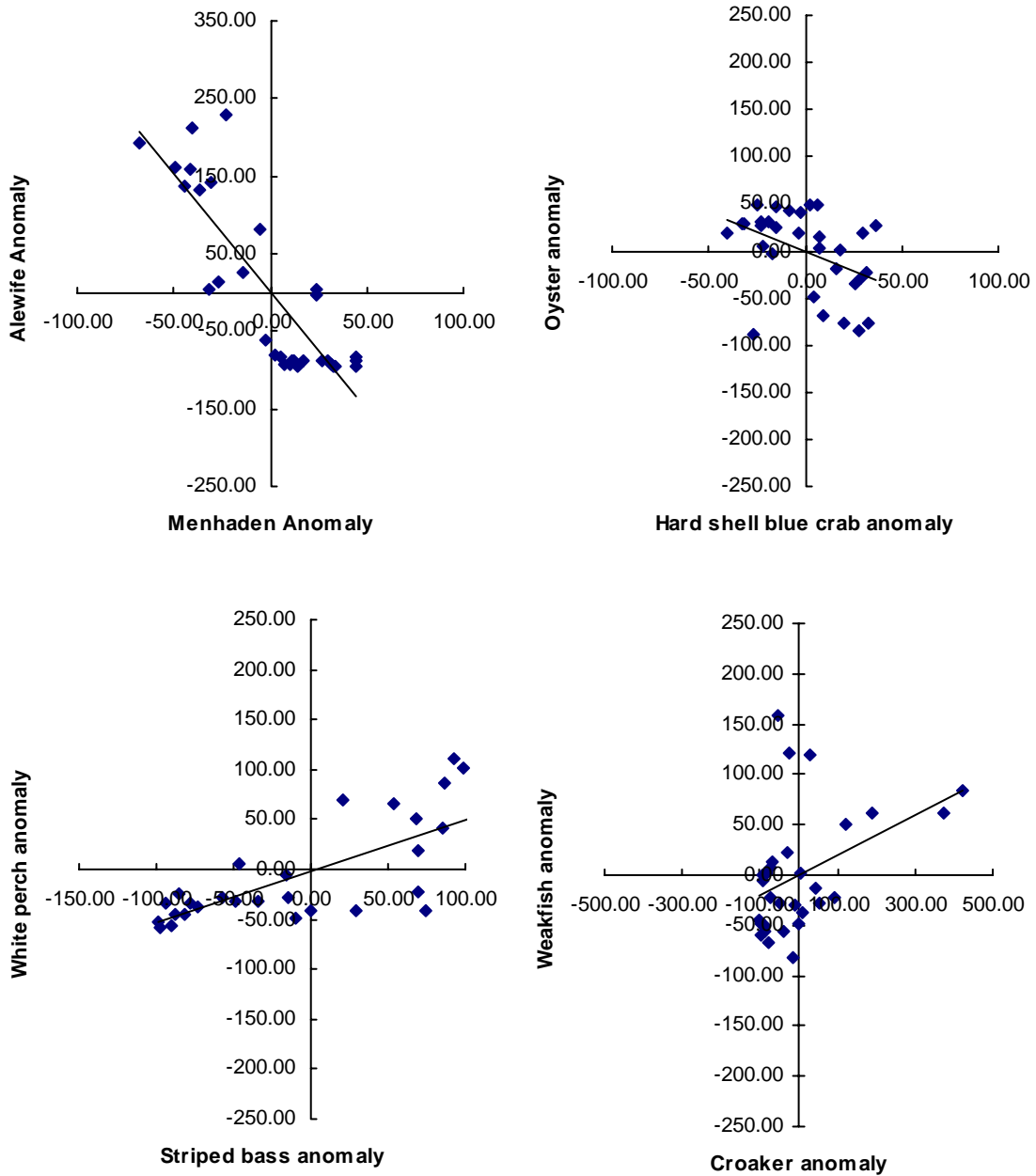
**Figure 16.** Time series of Chesapeake Bay commercial landings for four components of the benthivore guild; croaker, spot, carp, and catfish.



**Figure 17.** Time series of commercial landings by life history category of resident (solid line), seasonal (dashed line), or occasional (dotted line) species for both finfish and shellfish. Note that data for menhaden have been removed from this time series. See Table 2 for details of life history affiliations for species used in the analysis.



**Figure 18.** Results of Principal Components Analysis of landings anomalies for 13 species in the Chesapeake Bay commercial fishery during the period 1930-1992. Scatter plots of species loadings on the first three principal components axes. Species abbreviations are: alewife (alwv), bluefish (blue), hard-shell crab blue crab (crab), soft-shell blue crab (crabs), croaker (crkr), oyster (oyst), combined non-alewife alosids (shad), sea bass (sbss), spot (spot), striped bass (stbs), weakfish (weak), white perch (wprch), and yellow perch (yprch).



**Figure 19.** Examples of correlation structure between species pairs identified in the time series cross-correlation analysis. Data are plotted as the anomalies from the overall mean landing for each species.

Table 3. Summary of PCA analysis of time series for 13 principal species present in commercial landings from Chesapeake Bay.

A) Explained variation

PCA Axis	Eigenvalue	% variation explained	Cumulative variation explained
PCA1	33.22	51.9	51.9
PCA2	14.97	23.3	75.3
PCA3	9.44	14.7	90.0
PCA4	2.87	4.4	94.5
PCA5	1.91	2.98	97.5

B) Species Loadings

Species	PCA1	PCA2	PCA3
Alewife	-2.45	10.65	-3.58
Bluefish	1.75	-1.65	3.32
Soft Shell Crab	0.58	0.45	1.02
Hard Shell Crab	14.48	-1.85	-6.11
Croaker	-10.99	-5.95	-5.11
Oyster	-5.95	3.37	0.19
Black Sea Bass	0.60	0.17	1.12
Shad	-0.25	0.29	1.13
Spot	1.16	-0.11	1.80
Striped Bass	1.93	0.54	2.80
Weakfish	-3.69	-3.31	-1.08
White Perch	1.50	-0.65	2.22
Yellow Perch	1.32	-1.04	2.25



Table 4. Cross-correlation coefficients for landing time series. Data are not lagged, and cross-correlations may be higher for lagged values. Values associated with probability levels > 0.05 are shown as ns. All other values are significant at p# 0.05. To illustrate the nature of the stronger correlations, selected species pairings are plotted in Figure 19.

	Alewives	Bluefish	Blue Crab, Soft	Blue Crab, Hard	Croaker	Menhaden	Oysters	Weakfish	Shad	Spot	Striped bass	White Perch	Yellow Perch
Alewives	1												
Bluefish	-0.591	1											
Blue Crab, Soft	0.509	-0.466	1										
Blue Crab, Hard	ns	ns	ns	1									
Croaker	ns	-0.367	0.423	-0.605	1								
Menhaden	-0.706	0.595	-0.615	0.602	-0.642	1							
Oysters	0.592	-0.36	0.497	-0.651	0.596	-0.739	1						
Weakfish	ns	ns	0.495	-0.485	0.879	-0.566	0.554	1					
Shad	0.766	-0.517	0.651	-0.462	0.452	-0.787	0.762	0.563	1				
Spot	0.418	-0.365	ns	ns	ns	ns	0.423	ns	0.361	1			
Striped bass	0.63	ns	ns	ns	ns	-0.33	ns	ns	0.321	ns	1		
White Perch	0.647	-0.38	ns	ns	ns	ns	ns	ns	ns	0.31	0.74	1	
Yellow Perch	0.6	-0.618	0.555	-0.32	0.677	-0.74	0.51	0.615	0.651	ns	ns	ns	1

PCA1. All three of these species exhibited a peak in landings in the mid to late 70's until the early 80's, and then declined rapidly thereafter. No other distinct groupings could be identified on PCA1.

The second component (PCA2) explained an additional 23% of the variation (Table 3). Alewife exhibited a high positive loading on PCA2, whereas croaker exhibited a strong negative loading on PCA2. The third component (PCA3) explained an additional 15% of the variation in the dataset. A group consisting of bluefish, striped bass, white perch and yellow perch all exhibited positive loadings on PCA3, while croaker and hard shell blue crab had negative loadings on this axis. In general, with the exception of the bluefish-stripped bass-white perch-yellow perch complex, PCA did not suggest any strong complementary patterns in the landing time series.

To explore the correlation pattern between individual pairs of species we used time series techniques to calculate the cross-correlation between pairs over time. Cross-correlation is the time series equivalent of the correlation coefficient in linear models. Trends in time series have to be removed

prior to cross-correlation analysis. This was achieved by using differencing or “pre-whitening.” A summary of the results is presented in Table 4. The analysis clearly identify species whose landings have generally declined and those whose landings have generally increased. For example, this is particularly clear in the highly significant, negative correlations between alewives and menhaden (Fig. 19A), and between hard-shell blue crab and oysters (Fig. 19B). However, there are some significant correlations among species whose landings do not show a clear trend in the time series. For example, the correlations between white perch and striped bass (Fig. 19C) and between weakfish and croaker (Fig. 19D) probably represent examples of biological interactions. Considering the top piscivores, i.e. bluefish, striped bass, and weakfish, there was no compelling evidence of a correlation structure that would suggest competition for similar prey resources (Table 4). The influence of heavy exploitation and major declines in landings of several species, and of the long-term increase in menhaden landings, complicated the detection of fine-scale patterns in cross correlations of the landings time series.

### III. SPECIES INTERACTIONS WITHIN CHESAPEAKE BAY.

Individual species perform only a few direct roles in a food web: species may be competitors, predators or prey. However, this simplified view focuses attention on species within particular trophic levels and on those immediately adjacent levels. In this section, we review current hypotheses on control and regulation of aquatic food webs, including a discussion of complex indirect trophic effects, to identify those species in Chesapeake Bay for which a multispecies approach may be warranted. We also identify biological factors, in addition to technical factors already identified, that have motivated the development of multispecies approaches in other ecosystems and which are present in the Chesapeake Bay (Table 5).

#### A. Overview of Food Webs.

Since the turn of the century ecologists have been depicting the flow of energy in ecosystems with diagrams of food chains and food webs. For example, in its simplest form a food chain between top predators (highest trophic level) and primary producers (lowest trophic level) shows the dependence of top predators on primary production, and explains changes in abundance through predator-prey interactions (Fig. 20). A classic example in the Chesapeake Bay is the pelagic food chain linking striped bass and other piscivorous fishes to phytoplankton through their predation on bay anchovy which feed

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Table 5. Assessment of the presence of factors motivating multispecies approaches in the Chesapeake Bay.

Category	Motivation	Example in Chesapeake Bay
Technical	Incidental bycatch of non-target species in targeted fisheries	Migrating alosids in coastal intercept fishery
	Use of non-specific fishing gear	Poundnet fishery
	Exploitation of mixed species schools	Gillnet fishery
Biological	Extensive predation mortality	Striped bass, bluefish and weakfish are all top predators
	Competition among species for food	Control of zooplankton abundances by gelatinous zooplankton
	Similar responses to environmental and biological forcing	Spot, croaker and crab recruitment patterns

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upon copepods, the predators of phytoplankton (Baird and Ulanowicz 1989). In such chains, species one trophic link below a specified species are its prey, and those one link above are its predators. Thus, in Figure 20, striped bass and weakfish are predators on bay anchovy. Species that share a common resource occupy the same trophic level and may affect each other's abundance through competition for resource. Consequently, striped bass and weakfish also are potential competitors. In competitive interactions, changes in species abundances can be attributed to one species' ability to exploit the resource more efficiently (due to feeding efficiency, reproductive rate, spatial overlap, etc.). If the resource is limited, a decrease in the weaker competitor's growth rate or abundance is expected. Competition may occur over different life stages, adding to a system's complexity.

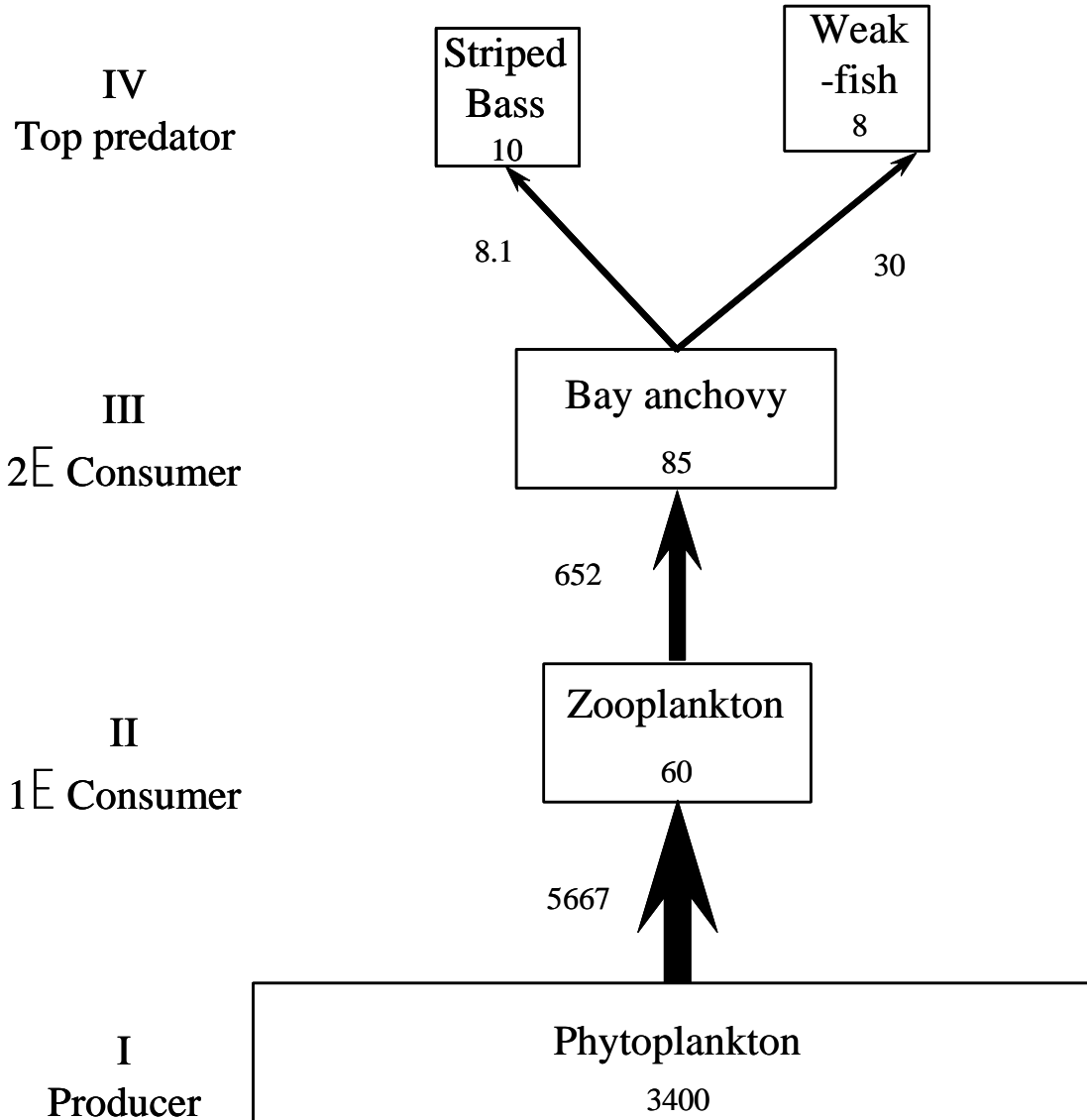
This classical view of trophic interactions, involving purely predation and competition, has been challenged by the recognition of additional classes of interactions termed intra-guild predation and indirect effects. A guild is a group of species that exploits the same resource in a similar way. Therefore, intra-guild predation is the "the killing and eating of species that use similar, often limiting, resources and are thus potential competitors" (Polis et al. 1989). Such complex interactions provide benefits to an organism directly through predation and indirectly by reducing competition on a similar resource. Most intra-guild predation occurs by generalist predators in a size-structured community. For this reason intra-guild predation probably is particularly important in aquatic systems in which size-dependent predation predominates. An intra-guild predator is usually larger than its prey and

has a broad diet, which may include juveniles of its own species. An excellent example of intra-guild predation in the Chesapeake Bay occurs between the two dominant jellyfishes, sea nettle and the ctenophore *Mnemiopsis leidyi*. While both the sea nettle and ctenophore feed upon zooplankton, the sea nettle also feeds voraciously upon the ctenophore (Feigenbaum and Kelly 1984; Purcell and Cowan 1995: see Fig. 3 in Baird and Ulanowicz 1989). The dynamics of these two gelatinous organisms are regulated by their competitive abilities and their relative abundances. Thus, intra-guild predation actually may insure the coexistence of these competing species via the differential effects of prey switching by the dominant species. A complex pattern of competitive interactions among species, such as among gelatinous zooplankton in Chesapeake Bay, has been used as justification for a multispecies approach in other systems (Table 5).

In traditional models of food webs such feedback is not considered and food web structure is controlled only by the amount of energy entering the system and by the efficiency of its transfer between trophic levels (Fig. 20). Indirect effects, which are highly likely in aquatic ecosystems, complicate the traditional view of control of food web structure in which the controlling process alternates between predation and competition at successive trophic levels ( Hairston et al. 1960). Complex, indirect interactions allow feedback between non-adjacent trophic levels. For example, one predator may cause shifts in the relative abundances of species at lower trophic levels. In turn, these changes at lower trophic levels may feed back to alter the relative abundances of predators. A specific example of the importance of indirect effects

in the Chesapeake Bay ecosystem involves

Nominal  
Trophic Level



**Figure 20.** An example of a Chesapeake Bay food chain. Values associated with arrows are rates of carbon flow between components (mg.m<sup>-2</sup>.summer<sup>-1</sup>) and values in each component box are biomasses of carbon (mg.m<sup>-2</sup>). Data are taken from Baird and Ulanowicz (1989).

the high summer rate of predation on zooplankton by bay anchovy, ctenophores, and sea nettles, which precipitates a decrease in the standing stock of zooplankton. However, the three zooplanktivores also interact, with the gelatinous zooplankters feeding on the egg and larval stages of bay anchovy (Govoni and Olney 1990; Cowan and Houde 1992, 1993; Purcell et al. 1994). These interactions produce complex, dynamic patterns of planktivore abundances during the summer. Moreover, the low zooplankton biomass is coincident with the annual summer peak in phytoplankton production and a peak in bacterial biomass and production (Malone et al. 1986; Harding 1994; Shiah and Ducklow 1994). Baird and Ulanowicz (1989) suggest that the low abundance of zooplankton frees the phytoplankton from predator-mediated control and makes available more phytoplankton production to support protozoan and bacterial production. They estimate that as much as 50% of net phytoplankton production is channeled and cycled through a microbial loop in the summer (as compared to 30% in fall, 28% in winter, and 25% in spring), rather than channeled through zooplankton and ultimately to fish.

Food web diagrams are useful tools to gain an understanding of organism interactions, but often do not represent the complexities of natural ecosystems fully. Food web diagrams generally do not depict the relative strength of species interactions or the spatial and temporal components of food webs (Paine 1989). For example, although striped bass in Chesapeake Bay feed mainly on menhaden and bay anchovy in the summer, their diet shifts to juvenile spot and Atlantic croaker in the winter (Setzler-Hamilton and Hall 1991). Furthermore, during the summer, differences

in the strength of striped bass predation on menhaden and bay anchovy may be related to their spatial overlap in the estuary, the timing of prey production and abundance, or predator preference (Hartman and Brandt 1995a, b).

Consequences of increased nutrient loading provide another example of the complexity of ecological interactions in the Chesapeake Bay, which are not accounted for in food webs. Excess nutrients lead to increased phytoplankton abundance and epiphytic growth on submerged aquatic vegetation (SAV), causing a drastic decline in SAV acreage during the 1960s and 1970s. This decline directly and indirectly influenced the productivity of many organisms that rely on SAV beds for shelter, nursery areas, or on the foraging potential associated with greater fish density and organism diversity in SAV beds (Funderburk et al. 1991). Increased nutrient loading also has led to increased duration and extent of anoxic conditions of water below the pycnocline in the mainstem of the Chesapeake Bay. This causes potentially complex effects upon the biota and species interactions in the Bay (Breitburg 1992; Smith et al. 1992; Breitburg et al. 1994). For example, anoxic waters, combined with temperature preferences, limit the distribution of striped bass and its prey, and decrease habitat area where potential for growth is positive (Brandt et al. 1992; Brandt and Kirsch 1993). Yet, food web diagrams do not account for such temporal, spatial and behavioral complexities, even though these features may be important controlling factors (Pimm and Kitching 1988).

## **B. The Chesapeake Bay Food Web.**

The diverse topography, extended salinity gradient, and dynamic tidal regime in the Chesapeake Bay provide a wide range of habitats that in turn support diverse biological communities. Because of seasonal patterns of abundances and the importance of seasonal migrants, there is extensive coupling between the different communities (Baird and Ulanowicz 1989). Shifts in habitat use by different life stages of fishes and invertebrates (e.g. striped bass and blue crabs) increase connections and linkages among habitats and communities. Recently, declines in SAV and oysters, two important components of the Chesapeake ecosystem, have had dramatic impacts. For example, at the turn of the century the oyster population in the Bay may have had the potential to filter more than half the carbon production on a daily basis but by the 1980's that figure had fallen to less than 1% (Newell 1988). Moreover, the decline of the oyster has led to the general decline in the oyster reef community which it supported (Kennedy 1991). Despite these declines, Nixon (1988), in a comparison of marine and freshwater systems, indicated that primary production in the mid-Chesapeake Bay was high, amounting to 335-780 g C m<sup>-2</sup> yr<sup>-1</sup>. Thus, the Bay remains a highly productive, if altered, ecosystem (Harding 1994). Much of its productivity, originally associated with benthic communities, apparently has shifted to the pelagic zone.

Strong seasonal patterns in migration and productivity are important to the ecology of the Chesapeake Bay. In general, biological activity in Chesapeake Bay rises in spring, peaks during summer, and declines through the fall and winter. This typical seasonal pattern is driven by nutrient inputs and increasing temperatures that trigger biological activity in the spring (Harding 1994). Phytoplankton biomass

peaks in spring, although production rates are highest in summer (Harding 1994). About 35% of net phytoplankton biomass is grazed during spring and summer (compared to 10% in fall and 14% in winter). As we have discussed previously, phytoplankton are grazed heavily by numerous herbivores. The herbivore community responds to the seasonal pulses in phytoplankton production, and itself exhibits seasonal pulses in abundance and complex internal dynamics.

Gelatinous organisms are believed to be important consumers in the Bay (Cowan and Houde 1992, 1993; Purcell et al. 1994; Purcell and Cowan 1995). Although present year-round, abundance of the ctenophore *Mnemiopsis leidyi* peaks in the early summer and fall. It feeds mainly on zooplankton, along with microzooplankton and suspended detrital matter. Medusae of the sea nettle appear in summer and feed on ctenophores, zooplankton, fish eggs and fish larvae. The sea nettle feeds higher on the food chain than blue crab, oyster, anchovy, or American shad, and may control recycling of carbon in the planktonic community during summer and early fall (Baird and Ulanowicz 1989).

In the mesohaline region of the Chesapeake Bay, the biomass and productivity of suspension-feeding fish peaks in summer. Seasonally abundant menhaden are major consumers of phytoplankton and resident bay anchovy are major consumers of zooplankton (Funderburk et al. 1991). In a recent study of the relative abundance, biomass and production of bay anchovy Wang and Houde (1995) suggested that early life stages of this species are major consumers of zooplankton in Chesapeake Bay. They estimated that when larval and juvenile anchovy are included, total annual production of anchovy reaches 233,000 t in the upper to mid-Bay



regions. Based upon projections of a bioenergetics model, Luo and Brandt (1993) noted that during fall months bay anchovy alone may consume a substantial proportion of the total zooplankton production. Moreover, both striped bass and menhaden are important prey for piscivorous fish.

Benthic communities in Chesapeake Bay also show complex spatial and temporal variation. The Chesapeake Bay has a well developed benthic community dominated by deposit feeders such as *Nereis succinea*, other polychaetes, tellinid bivalves, amphipod crustaceans, and other meiofauna (Baird and Ulanowicz 1989). These deposit feeders utilize a diverse flora of sediment-bound bacteria as a major food source. Several species of clams, including the soft clam *Mya arenaria*, are important filterers of phytoplankton and zooplankton, and hence link the benthic and pelagic food chains. They are also an important prey item in blue crabs and several fishes, further strengthening their role as a bridge between pelagic and benthic communities.

As with the pelagic community, the macrobenthic community biomass and productivity show distinct seasonal patterns, with declines by late fall due to predation by blue crabs and nekton as well as cumulative effects of physical stress (low temperatures and low dissolved oxygen levels). Moreover, the extent to which benthic and pelagic systems exchange energy via predator-prey interaction varies seasonally and spatially. Predation on benthic invertebrates by age 0+ striped bass represents one example of coupling. Strong year classes of striped bass and other benthic-feeding fishes may transfer large amounts of the energy flowing through the benthic community to the pelagic realm, directly through their consumption of benthos and indirectly through predation

upon them by larger pelagic predators.

Several environmental factors influence the complex biotic interactions summarized above. As the ecotones in the Chesapeake Bay are principally defined by salinity, the extent and timing of spring runoff can affect the spatial extent of the oligo-, meso-, and polyhaline zones. Years with above average runoff may be characterized by more southerly distributions of many seasonally-resident fish species. Moreover, the intensity and timing of individual runoff events can be important. Secor and Houde (1995) suggest that recruitment in striped bass may be controlled by the pattern of runoff. The extent of anoxia maybe an important regulator of biological interactions (Breitburg 1990). Coutant and Benson (1990) argued that a spatial “squeeze” may be suffered during summer by large predator fish (e.g. striped bass) in the Chesapeake Bay as they try to balance feeding opportunities with thermal optima and avoid anoxia. Kemp and Boynton (1992) documented the importance of oxygen dynamics in controlling the extent of benthic-pelagic interactions. They discussed both the direct effect of anoxia on benthic organisms and the indirect effects of this reduced productivity on the pelagic system, presumably a result of eutrophication.

### **C. Multispecies Interactions in Chesapeake Bay.**

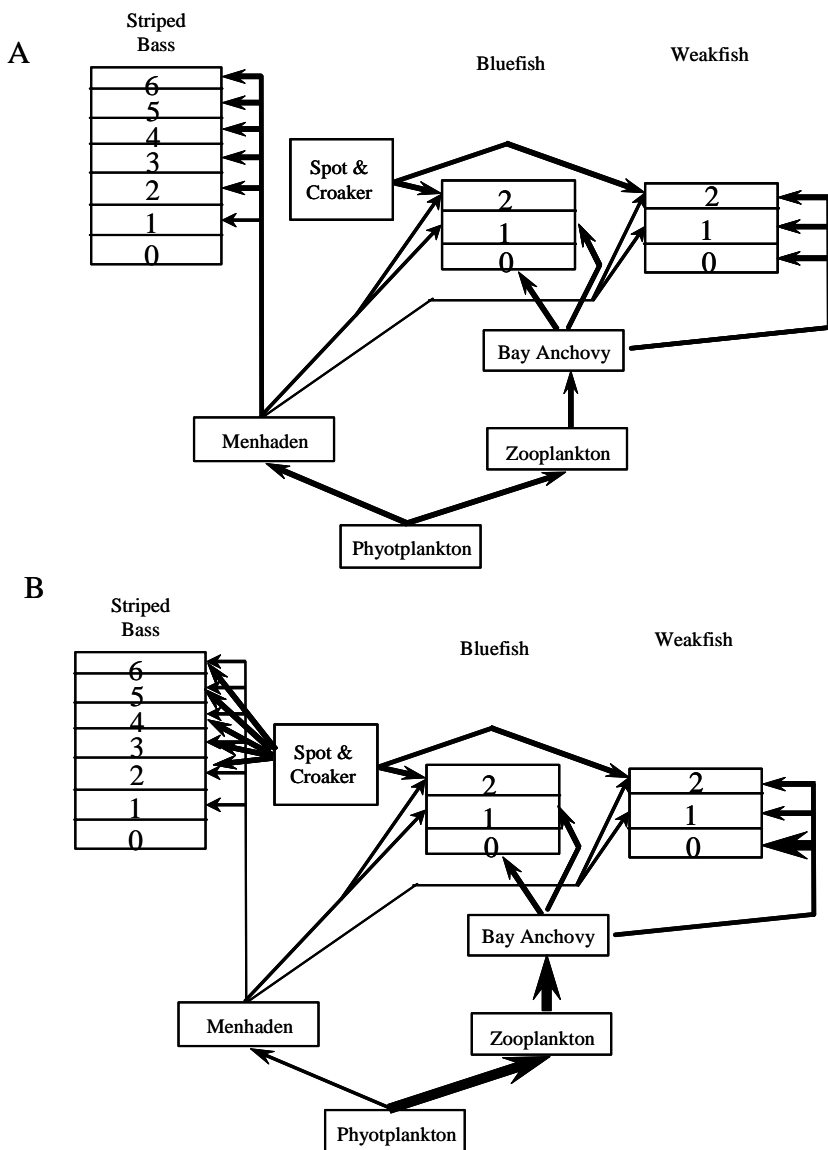
In detailed studies, Hartman and Brandt (1995a, b) examined the seasonal diet patterns and the predatory demand of striped bass, bluefish and weakfish. Bay anchovy dominate the diets of age 0+ weakfish and bluefish, accounting for up to 90% of the dietary intake of this age class. Older individuals of both species utilize bay

anchovy less, relying primarily on menhaden. Spot and croaker may also be important components of the diet. In contrast, the diet of age 0+ striped bass is dominated by invertebrates. But, older striped bass shift to a menhaden-dominated diet, such that menhaden represent fully 70% of the diet of age 6+ striped bass. Thus, with the exception of age 0+ striped bass, the three piscivores exploit the same prey species. Moreover, in comparing predicted predatory demand and the concentration of prey in the field, Based upon bioenergetics models, Hartman and Brandt (1995a, b) concluded that potential demand was often greater than supply, with the difference between the two increasing over the course of the year. Yet, detailed information on the actual biomasses of each predator and the extent of spatial overlap is required before Hartman and Brandt's (1995a, b) conclusions can be fully substantiated. However, the three top piscivores in Chesapeake Bay likely compete for limited food resources, particularly as the seasons progress. Thus there is a clear potential for complex dynamics among the three predators and two prey species. The interactions among these species represent an example of a motivation for developing multispecies fisheries in other systems (Table 5).

We offer the following hypothetical example of the potential for multispecies interactions in the Chesapeake Bay. In a "normal" year most energy flows into the three top predators via two routes (Fig. 21A). Striped bass rely principally on the phytoplankton - menhaden pathway, whereas weakfish and bluefish rely more on the bay anchovy pathway. In addition,

weakfish and bluefish depend more upon spot and croaker and so are coupled to the benthic food chain. However, when menhaden abundance is low the pattern changes (Fig. 21B). A lack of menhaden may cause striped bass diets to shift to include more spot and croaker. This shift will increase competition with weakfish and bluefish. Hypothetically, in years of low menhaden abundance, the reduced grazing pressure on phytoplankton potentially may increase zooplankton abundance, leading to enhanced growth and survival of bay anchovy. In turn, increases in bay anchovy may lead to increased production of weakfish and bluefish, thus further increasing potential competition among the top predators. Consequently, we suggest that the major fishery for menhaden in Chesapeake Bay, combined with natural fluctuations menhaden abundance, potentially can alter routes of energy flow among top predators in Chesapeake Bay and possibly affect their relative productivities.

Intense seasonal effects and variation in recruitment of predators combine to complicate potential interactions. Bluefish, weakfish, and summer flounder, all are resident in the Bay during spring, summer and fall but are mostly absent in winter. Some striped bass are present year-round. Recruitments of bluefish and weakfish are dependent upon processes largely external to the Bay. Thus, not only are the dynamics of top predators potentially affected by processes in the Bay, but they also are heavily influenced by processes that occur outside the Bay. Hence, the dynamics of species and interactions among species within the Bay reflect the outcome of



**Figure 21.** Conceptual example of complex interactions in the Chesapeake Bay piscivore guild (striped bass, bluefish, weakfish) in A) normal years when sufficient energy flows through the bay anchovy and menhaden pathways and B) years when menhaden abundance is reduced leading to increased competition for spot and croaker by the top piscivores. Numbers in boxes represent age classes of the predators. The importance of the interaction with respect to energy flow is indicated by the weight of the arrow.

combined external and internal natural forces as well as directed fishing effort on predators and prey.

Although the suite of piscivores have been most intensively studied, they are not the only example of multispecies interactions within the Bay's ecosystem. We have already indicated potential feedback among the two gelatinous zooplankton and their prey. An additional example of an interacting group is the spot, croaker, flounder and blue crab complex. All four species principally eat clams and polychaetes. Moreover, each species has an obligate offshore larval stage and all rely on specific oceanographic conditions to transport larvae back into the Bay. Thus, all four might be expected to show similar trends in abundance. Counteracting this tendency is the fact the species are also potential competitors (Baird and Ulanowicz 1989) and a particularly strong year class of

one species, could be expected to negatively impact the other three. A recognition that species may exhibit common responses to external forcing, such as with the four species complex discussed here has been used to justify multispecies approaches in other systems (Table 5).

This brief review of the Chesapeake Bay food web has emphasized the importance of species interactions and indirect trophic effects in complex communities. Observed patterns are often surprising or counter-intuitive. If the management goal of the Chesapeake Bay Program is to insure sustainability of living resources while maintaining viable fisheries, then an understanding of multispecies interactions that control and modify structure and productivity of biological communities becomes an important requirement for successful long-term fisheries management.

#### IV. MULTISPECIES MANAGEMENT: CONCEPTS AND APPROACHES.

We have shown that all of the factors used to justify multispecies approaches in other systems are present in Chesapeake Bay (Table 5). In this section, we review evidence in support of multispecies management, briefly discuss the principal methods to achieve it, and point out some of the problems associated with its application.

##### A. The Need for a Multispecies Approach.

The justification for multispecies approaches falls into two broad categories: biological and technical (Table 6).

##### 1. Biological interactions: yield, stability and resilience.

A common motivation for multispecies management is that the knowledge that sum of the maximum sustainable yields for individual species is less than the estimated maximum sustainable surplus production from the community. The principal reason underlying this fact is that the major predators of fish in the ocean are other fish (Sissenwine 1984, 1986; Bax 1991). As the transfer of energy from one trophic level to the next is not efficient, and on average, only about 8-10% of the energy is transferred, a substantial proportion of the estimated total biological production supports the metabolic costs and growth of other fishes (Pauly and Christensen 1995). By accounting for these interactions in our efforts to manage fisheries, we may be able to increase the aggregate catch. For example, when we harvest a predator, we presumably also may be able to harvest the biomass of its prey that would have supported the predators removed from the system. In an extreme example, if we overharvest a predator to the point of

population collapse, a very much larger theoretical yield of its prey (perhaps 8 to 10 times the maximum sustainable yield of the predator) could be harvested from the ecosystem (May et al. 1979). Single-species management ignores the possibility of such feedbacks between the species. Thus, although single-species approaches are conservative, and limit the risk of collapse of individual species, they are unlikely to optimize yields of component species simultaneously.

Multispecies approaches may also help preserve the stability and resilience of a system and hence its ability to sustain long-term exploitation. Ideally, an exploited system should exhibit "global stability," such that it will always tend to return to its pre-exploited state after perturbation by harvests (Beddington 1986). In all exploited systems, short-term fluctuations in the abundances of individual species necessarily lead to different rates of exploitation on the component species. But, if the exploited system is globally stable, such annual variations will be damped out in the long term to produce predictable and sustainable long term harvests (Beddington 1986).

However, it is by no means certain that ecosystems are or remain globally stable even in an unexploited state (May 1972). For example, between 1940 and 1960 the estimated biomass of sardine (*Sardinops sagax caerulea*) off California declined by two orders of magnitude (MacCall 1990). Simultaneously, the biomass of northern anchovy (*Engraulis mordax*) increased by a similar factor. This species replacement dramatically affected the fishing economy along the California coast (MacCall 1990). It initially was believed that the species replacement was driven by heavy

exploitation of the sardine which lead to its population collapse, and that in the absence of fishing pressure the dominance of sardine would have continued. However, in analyses of the deposition rates of fish scales in anoxic sediments, the two species were demonstrated to have undergone dramatic transitions such as the one observed in the 1950's for at least 2,000 years (Soutar and Isaacs 1974; Baumgartner et al. 1992). The fluctuation in the 1950's probably was not a result solely of exploitation, but was more likely a result of shifts in ocean circulation patterns, perhaps abetted by heavy fishing on sardine. Other examples include fisheries in Europe and Asia. Historical landings of herring in Norway have fluctuated widely prior to the recent periods of extreme exploitation (Daan 1980) indicating natural variability. The Japanese sardine (*Sardinops sagax melanosticta*) yielded more than 1 million tonnes annually to a fishery in the 1930's before collapsing and essentially providing no yield until its recovery during the 1970's and 1980's, when annual catches peaked at more than 5 million tonnes annually (Kawasaki 1992, Kawasaki and Omori 1995; Watanabe et al. 1996). Furthermore, Crawford (1991) and Kawasaki (1992) have shown that fluctuations of sardine stocks (*Sardinops* spp), and possibly associated pelagic species, are coherent globally and have hypothesized that major shifts in ocean climate drive the observed changes in community structure and species dominance.

Even were stability the norm, exploitation clearly can shift communities away from stable equilibria. For example, harvests of cod and haddock from the fishing grounds in the northwest Atlantic once seemed limitless. However, in the face of prolonged over-exploitation the groundfish

stocks collapsed (Hutchings and Myers 1994; Walters and Maguire 1996). On George's Bank the commercially valuable groundfish species have been replaced by a suite of commercially less desirable species, dominated by skates and dogfish (Rothschild 1991). Similar shifts in community structure also have been reported in the South Atlantic (Gulland and Garcia 1986). In the Gulf of Thailand, small, short-lived fishes and squids came to dominate the community after large piscivore fishes were overexploited (Pauly 1988). Managing fisheries in the face of these complex patterns of species replacements and interactions clearly requires consideration of multispecies approaches.

## 2. Technical interactions: bycatch.

In addition to the biological reasons for adopting a multispecies approach, technical ones also must be considered (Table 6). The principal motivation for these models is that even though a single species may be the target of a fishery, other species are inevitably caught in the process. This is termed the bycatch. Pauly (1995) noted that about one third of the annual world marine landings are discarded as bycatch. Although fishing techniques have improved, non-target species inevitably are caught as bycatch in targeted fisheries (Alverson et al. 1994; Alverson and Hughes 1995). In some cases, such as the menhaden purse-seine fishery in Chesapeake Bay and the Mid-Atlantic Bight, the bycatch is minimal (Austin et al. 1994). In others, such as the penaeid shrimp fisheries, it is a substantial problem (Alverson et al. 1994). Often the species caught as bycatch are juvenile stages of other commercially important species. For example, Goodyear (1995) has suggested that the red snapper (*Lutjanus campechanus*) fishery in the Gulf

of Mexico cannot recover from its overfished state until a substantial reduction in bycatch of snapper juveniles by the shrimp fishery is achieved. Thus, increased effort directed at one species may be detrimental to sustained harvests of other species (Fig. 8). Table 5 presents examples of technical interactions that may be important in the management of Chesapeake Bay living resources.

The need to account for significant non-target landings is obvious. Initial attempts to account for technical interactions involved allocating effort and, hence, partitioning fishing mortality into components for each species (Alverson and Hughes 1995). However, multispecies management may provide better ways to account for such technical interactions. One further important difference between technical and biological motivation for multispecies approaches is that data required to evaluate and account for technical interactions typically is already collected while data on biological interactions are not (Pikitch 1988). Consequently, multispecies approaches directed at problems arising from technical interactions may seem more tractable than ones designed to account for biological interactions.

## **B. Multispecies Approaches.**

Just as the motivation for multispecies approaches can be separated into two broad categories, so too can their application. Concern over both technical interactions and biological interactions have motivated multispecies approaches. Approaches to the former are similar in structure to single-species approaches. Laurec et al. (1991) and Murawski et al. (1991) provide examples of multispecies models involving technical interactions. The basic thrust of these

models is to partition allocation of effort onto both target and non-target fisheries. Once achieved, total effort can be regulated to minimize the impact of the fishery on non-target organisms. Alternatively, gear can be modified to reduce the effective effort on non-target species. However, in spite of complexity, multispecies models that deal with technical interactions are of the same philosophical lineage as single-species approaches. This is not the case for multispecies models aimed at accounting for biological interactions. While the motivation for multispecies approaches is often the same, approaches that emphasize biological interactions recognize that harvests of individual species do not occur in an ecological vacuum. Consequently, the approaches adopted can be widely different (Table 6).

In a review of multispecies analyses for biological interactions, Kerr and Ryder (1989) suggested that the range of approaches adopted can be usefully categorized into four groups: descriptive multivariate, dynamical multivariate, multivariate systems, and integral systems (Table 6). We use these categories to document the range of approaches in multispecies management. Below we present an overview of each category.

### **1. Descriptive multivariate approaches**

In this category Kerr and Ryder (1989) include those techniques that principally describe the pattern of species interactions observed in empirical datasets. Hence, these approaches are not mechanistic descriptions of species interactions: rather, they are statistical or graphic description of

Table 6. Motivations, advantages and disadvantages of the principal multispecies approaches

Objective	Category	Motivation	Advantage	Disadvantage	Example
Technical Interaction		Effects of catch of incidental species in targeted fishery	<ol style="list-style-type: none"> <li>1. Most easily adapted from single species data</li> <li>2. Produce management alternatives</li> </ol>	<ol style="list-style-type: none"> <li>1. Do not include biological interactions</li> </ol>	Laurec et al. 1991; Murawski et al. 1991
	Descriptive Multivariate	Description of dynamics of species in multispecies assemblages	<ol style="list-style-type: none"> <li>1. Empirical, using existing data</li> <li>2. Diversity of simple approaches</li> <li>3. Combining biological and environment variables</li> </ol>	<ol style="list-style-type: none"> <li>1. Limited forecasting ability</li> <li>2. Data intensive</li> <li>3. Not readily adaptable to management</li> </ol>	Tyler 1971; Salla and Erzini 1987; Rothschild 1991.
	Dynamical Multivariate	Prediction of dynamics of multispecies assemblages	<ol style="list-style-type: none"> <li>1. Predictive</li> <li>2. Adaptable to management use</li> <li>3. Exploration of alternative harvesting strategies</li> <li>4. Assessment of system stability</li> </ol>	<ol style="list-style-type: none"> <li>1. Data intensive, often requiring non-standard data</li> <li>2. Propagation of parameter error</li> <li>3. Counter-intuitive results</li> <li>4. Poor ability to include environmental forcing</li> </ol>	Daan 1987; Christensen and Pauly 1992
Biological Interactions	Multivariate System	Prediction of aggregate properties of assemblages	<ol style="list-style-type: none"> <li>1. Predictive</li> <li>2. Extension of existing techniques</li> <li>3. Use aggregate data</li> </ol>	<ol style="list-style-type: none"> <li>1. Insensitive to dynamics of individual species, and species interactions</li> <li>2. Limited ability to include environmental forcing</li> </ol>	Ralston and Polovina 1982; Sainsbury 1988, 1991; Hightower 1990, Brander 1977, 1991
	Integral Systems	Mechanistic understanding of assemblage dynamics	<ol style="list-style-type: none"> <li>1. Derived from fundamental properties of ecosystems</li> <li>2. Flexible</li> </ol>	<ol style="list-style-type: none"> <li>1. Limited management ability</li> <li>2. Require non-standard data</li> <li>3. Untested, emerging technique</li> </ol>	Pope et al. 1994; Heath 1995.



the observed dynamics (Table 6). This is often the initial effort in the assessment of whether multispecies approaches are required in a particular system. Our literature synthesis has been largely dependent upon descriptive approaches.

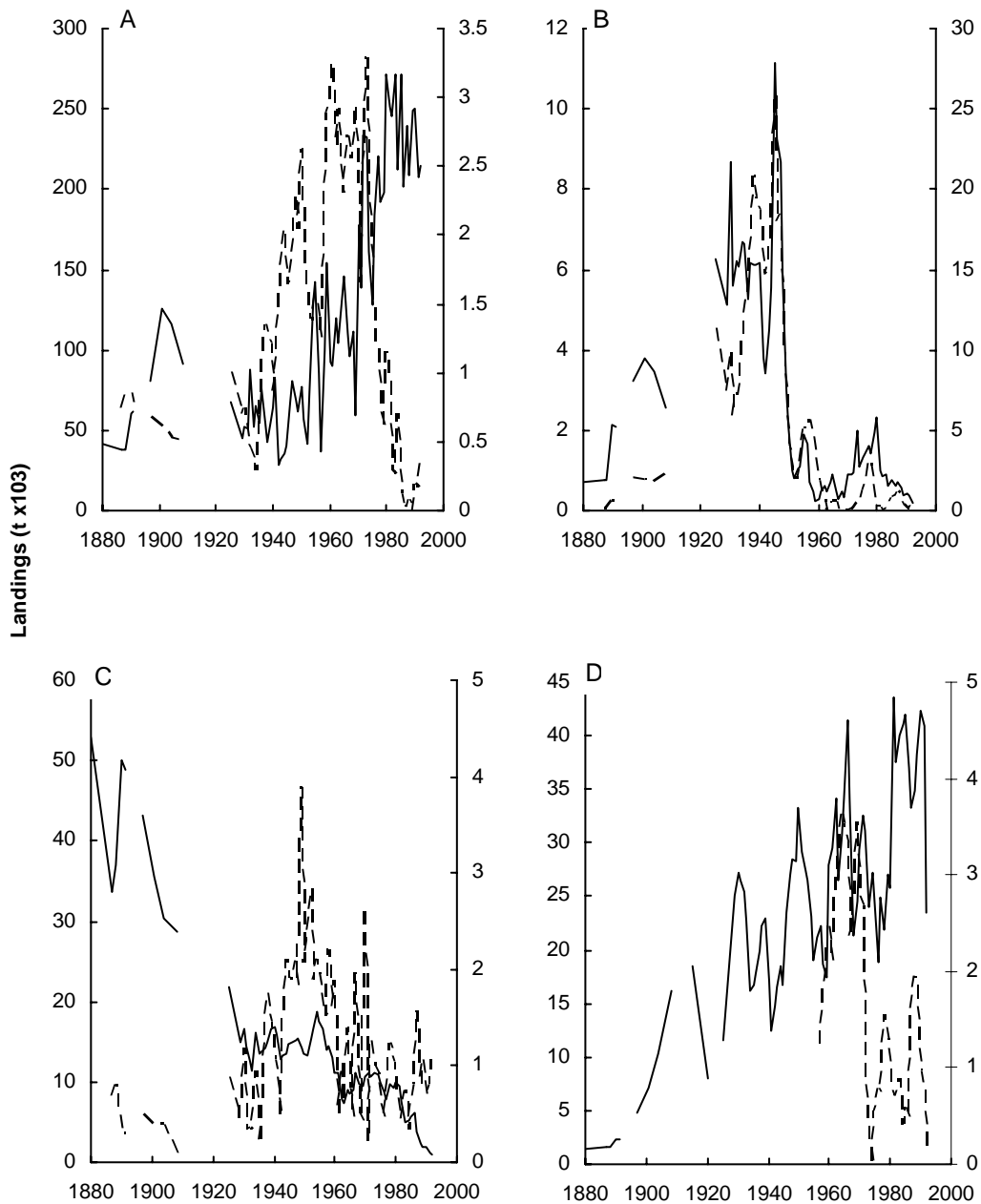
Within the overall category, several analytical tools have been used. One may simply plot abundance times series, and visually inspect for species which show variations in abundance that are consistently in- or out-of-phase. Rothschild et al.'s (1981) summary of trends in the abundance of commercially important species within the Chesapeake Bay is a good example. Rothschild et al. (1981) plot species that show similar trends in abundance on the same graph. Thus, 1) American shad (*Alosa sapidissima*) and river herrings (*A. aestivalis* and *A. pseudoharengus*), 2) menhaden and striped bass, and 3) weakfish and Atlantic croaker were plotted on separate panels because each species pair exhibited the same temporal trend over the time period originally considered. We have updated the Rothschild et al. plots to include recent data (Fig. 22). Some trends illustrated by Rothschild et al. (1981) have been maintained but others have diverged when data for the period 1980-1992 are included in the plots. Similar approaches were used by Tyler (1971) in his analysis of fish communities along the eastern seaboard of North America, and by Regier and Hartman (1973) in their analysis of successional patterns of species in the Laurentian Great Lakes. These simple techniques still have a role in modern multispecies research. They are an essential first step before more detailed analysis (Shelton 1992).

Statistical analyses approaches also have been used. Gabriel and Murawski (1985), Mahon (1985) and MacPherson and Gorda (1992) all used cluster analysis to

identify groups of species that exhibit the same trends in abundance. In these analyses, species that are within the same statistical cluster exhibit similar dynamics. Species from adjacent clusters share more similarities in their dynamics than do species from distant clusters. Kerr and Ryder (1989) note that ordination of species within cluster groups may primarily illustrate responses to physical forcing factors rather than to biological controlling agents.

Principal components analysis (PCA) has been used to determine patterns of similarity in abundances among species. The traditional approach is to produce scatter plots of species loadings against pairs of principal component axes (Koslow 1984; Pepin 1990; Fig. 18, this report). These plots are then inspected for groupings; species within the same group exhibit similar dynamics. Koslow (1984) used PCA to analyze 14 recruitment time series in northwest Atlantic fish stocks and concluded that large-scale physical forcing, rather than local biological interactions, predominately regulate recruitment to these fisheries. Pepin (1990) also used PCA to conclude that recruitment variability in North Sea fishes was independent of changes in the abundance of plankton over the long term. In a different approach, Rothschild (1991) plotted the PCA eigenvectors themselves to show how abundance of George's Bank stocks had varied over time. He found that the temporal pattern of abundance was dominated by the patterns of variation in three species: spiny dogfish (*Squalus acanthias*), winter skate (*Raja ocellata*) and haddock (*Melanogrammus aeglefinus*).

Time series approaches have been used to analyze patterns in multispecies interactions. One of the first applications



**Figure 22.** Landings time series through 1992 for selected species (updated from Rothschild et al. 1981). Ordinates on the left side of each panel are associated with the species plotted as a bold line, ordinates on the right side of each panel are associated with the species plotted as a dotted line. Species plotted are A) menhaden (bold) and striped bass (dotted), B) weakfish (bold) and croaker (dotted), C) oyster (bold) and spot (dotted) and D) hard shell blue crabs (bold) and soft clams

was Saila and Erzini's (1987) Markov model of multispecies systems in the Gulf of Thailand and Gulf of Maine. Saila and Erzini (1987) concluded that, under heavy exploitation, a system configuration dominated by squid and other non-valuable finfish would continue to dominate, and that the system is most likely to return to a squid-dominated state after any disturbance.

More sophisticated time-series approaches have been applied recently to the multispecies management issue. Criddle (1991) proposed a state-space, time-series approach to model the dynamics of the North Pacific groundfish complex of walleye pollock (*Theragra chalcogramma*), Pacific halibut (*Hippoglossus stenolepis*), Pacific cod (*Gadus macrocephalus*), yellowfin sole (*Limanda aspera*) and red king crab (*Paralithodes camtschatica*). This approach yielded accurate predictions of the short-term dynamics of this system. However, the application of multivariate time series is not a well established procedure in ecological or fisheries science, and its uses have been infrequent to date (Chatfield 1989). The principal drawback of the approach is that it is very data-intensive. Consequently, long and detailed time series are required to develop a model, and still retain portions of the data for model evaluation.

All of the approaches described above are essentially empirical. They have limited forecasting capabilities, given the potential for large environmental, biological or anthropogenic perturbations that can occur annually or on longer time scales. However, these descriptive approaches have yielded important insights into the regulation and stability of multispecies fisheries (Table 6).

## 2. Dynamical multivariate.

The dynamical multivariate approach includes techniques that extend knowledge

of single-species dynamics to predict the multispecies case (Kerr and Ryder 1989 - Table 6). This approach assumes that the dynamics of multispecies systems are predictable from a detailed knowledge of the dynamics of system components. Thus, multispecies assemblages are assumed not to possess emergent properties. Within this broad area are included both closed-form analytical approaches (May et al. 1979; Shepherd 1988) and simulation models (Andersen and Ursin 1977).

Multispecies systems that can be analyzed analytically are, of necessity, relatively simple. Typically such systems include few species with well defined trophic relationships. May et al. (1979) used simple Lotka-Volterra predator-prey models to analyze the influence of harvest levels on the dynamics of the interaction between baleen whales and krill in the Southern Ocean ecosystem. They concluded that the maximum sustainable krill yield can be taken at intermediate krill harvest levels, and high levels of whale harvest. Competition models from basic ecological theory also have been applied to multispecies fisheries. Silvert and Crawford (1988) developed a model of two competing species, that explained species replacements in the south-east Atlantic when the individual species were exploited by separate fleets (i.e. no technical interaction). In a later development, Shelton (1992) developed a model for two competing species and a single fleet exploiting both species. Both Silvert and Crawford (1988) and Shelton (1992) conclude that man's exploitation in multispecies systems is equivalent to indirect competition.

In research more directly related to fisheries management models, Shepherd (1988) examined the dynamics of a system of single-species, yield-per-recruit models in

a mixed species fishery with predator-prey interactions. He found that, with respect to parameterizing the models, that a lack of knowledge of the dynamics of the mathematical system is a smaller constraint than the lack of comparable biological knowledge. Moreover, increases in the variety of fleets exploiting the multispecies system has more impact on the overall model dynamics than does increasing the number of interacting species. Interestingly, Shepherd (1988) also concluded that the multispecies dynamics modeled were more sensitive to the parameters of underlying stock-recruitment relationships than to the effects of predation.

Attempts have been made to extend Ricker's spawner-recruit approach and Beverton and Holt's yield-per-recruit models to multispecies situations. Ricker (1958) considered the problem of simultaneous exploitation of three stocks in a graphical analysis. Based upon spawner-recruit relationships, he showed that a mixed-stock maximum sustainable yield (MSY) was only possible at levels that would drive one of the stocks to extinction. However, Ricker (1958) assumed that the three stocks must have the same replacement size. Paulik et al. (1967) extended Ricker's analysis to consider the case when the three species differed in their stock-recruitment functions and found that the risk of extinction of individual species increased when the species differed substantially in abundance or productivity. Murawski (1984) developed a multispecies yield-per-recruit model for the groundfish complex of George's Bank. He considered the effect of increasing mesh size on the multispecies yield per annual recruitment as a function of fishing effort. He concluded that larger mesh sizes produced larger total yields at higher sustained fishing efforts. However, these

models suffer from the same shortcoming as those noted by Shepherd (1988) - the lack of knowledge or inclusion of biological interactions leads to uncertainty in accuracy and precision (Table 6).

Network analysis is another form of dynamical multivariate approach that quantifies the relationship among elements in complex food webs. Baird and Ulanowicz (1989) have applied network analysis to the Chesapeake Bay. In this approach the multispecies system of interest is represented as a series of components linked by flows of energy. Components may be individual species, or trophic aggregations (Ulanowicz 1996). In general, network analysis maps the flow of energy through an ecosystem, by assigning values to 1) the energy inputs, 2) all rates of flow between elements in the system, 3) the rates of dissipation of energy, usually through respiration, and 4) any exports of energy from the system (Ulanowicz and Kay 1991; Christensen and Pauly 1992). Application of such models yields estimates of productivity and efficiency for individual components and entire systems. One emerging modeling framework for this approach is ECOPATH II (Christensen and Pauly 1992), which has been widely applied in tropical marine and freshwater systems to estimate total fisheries yields (Christensen and Pauly 1993).

Another dynamical multispecies approach is based upon simulation modeling of trophic interactions. This approach is exemplified by Andersen and Ursin's (1977) model, which modeled the principal competitive and predatory interactions for all of the major species within the North Sea ecosystem. However, the complexity of such efforts insures that a full understanding of a system's dynamics is unlikely. Thus, while such simulation models do permit exploration of the integrated effects of

fishing on the ecosystem, the large number of unsubstantiated assumptions required are a cause for concern. Simulation models do not often lend themselves to confident exploration of management strategies (Sissenwine and Daan 1991).

Independently, Pope (1979) and Helgason and Gislason (1979) proposed a modeling approach that incorporated the complexity of the fish community, yet was sufficiently simple to give management agencies a degree of confidence in its predictions. The approach, multispecies virtual population analysis (MSVPA), is an extension of the VPA technique commonly applied to single-species systems. Although this new approach was simpler than the full ecosystem model proposed by Andersen and Ursin (1977), it still required voluminous dietary data for the complete range of ages and sizes of principal fish species. These data were provided by a “model-driven” program to collect stomach data from fishes over the widest possible range of species and sites in the North Sea (Daan 1987). Together, the development of the modeling framework and the implementation of the stomach-analysis program provided the impetus for application of MSVPA to the North Sea (Sissenwine and Daan 1991, Pope 1991).

Specific applications of MSVPA have been reviewed thoroughly by Sparre (1991) and Magnusson (1995). For diverse fish communities, MSVPA is very data- and computer-intensive, and is often dependent upon initializing assumptions of parameter estimates. Fortunately, experience has shown that results are largely insensitive to the initial parameter estimates of relative abundances and size distributions (Magnus and Magnusson 1987; Rice et al. 1991). In contrast, the results are very sensitive to assumptions regarding the partitioning of the

total consumption among the suite of predators (Helgason and Gislason 1979; Gislason and Helgason 1985; Pope 1979; Sparre 1991), and on the assumption that predator ingestion rates can be accurately predicted from stomach analysis surveys (Hilden 1988). Furthermore, the temporal and spatial scales over which prey abundance and predator diets are estimated also has been shown to be important (Anon. 1992). To date, MSVPA models have been developed for several prominent fisheries, including the North Sea (Daan 1987), Baltic Sea (Sparholt 1991), George’s Bank (Overholtz et al. 1991), and the Benguela upwelling system (Blinov 1991).

### 3. Multivariate system.

Techniques included in this approach rely upon aggregating information on individual species to produce data for fishery yield, trophic efficiency and productivity (Table 6). Kerr and Ryder (1989) note that this approach is exemplified by the FAO’s multispecies, surplus production model (FAO 1978). The multispecies and single-species models do not differ in their mathematical form. Both are of the form:

$$\begin{pmatrix} \text{New} \\ \text{Biomass} \end{pmatrix} = \begin{pmatrix} \text{Old} \\ \text{Biomass} \end{pmatrix} - \begin{pmatrix} \text{Natural} \\ \text{Mortality} \end{pmatrix} + \text{Growth} \quad (3)$$

Typically the middle three terms of the right-hand-side of the equation are combined to represent the “surplus production.” The management objective is to adjust catch to maximize surplus production. In single-species models, each term of the equation is applied to each of the species. In the multispecies form, the biomasses of all species are summed to create an aggregate

value for the ecosystem. Similarly, aggregate values for surplus production and catch also are produced. The analysis proceeds, as in the single species case, by plotting surplus production against stock biomass and estimating the multispecies biomass at which surplus production is maximized. To be successful at the ecosystem level, productivity must be a conservative property, or perhaps the averaging processes effectively damp out variability. Brander (1988) argued that the improved fit of the aggregate model may result from the fact that the approach is based upon long-term steady state conditions and thus implicitly includes interactions among species.

Application of aggregate surplus production models to the Gulf of Thailand fishery has been very successful (FAO 1978). Brander (1977, 1989) has applied aggregated, multispecies surplus production models to determine acceptable quotas for the Irish Sea and Bristol Channel demersal fisheries. Although his model fit the data extremely well, it was never used in a management context. The demersal fishery off the northwest coast of Australia has been managed using an aggregate surplus production model (Sainsbury 1988, 1991). The model was used to explore the limiting role of habitat availability. The aggregated model was one of four different multispecies models developed to determine management strategies. The other models all required more precise formulations of the nature of inter-specific interactions. When the models were compared over a seven-year period, all models produced similar qualitative results. But, the aggregated model indicated that higher sustainable yields were possible than the non-aggregated models.

Hightower (1990) compared the yield predictions of single-species models and a

multispecies model based upon aggregate biomass to develop harvesting policies for the multispecies rockfish (*Sebastes spp*) fishery of the northwest Pacific coast of North America. He found that the predicted variance in yield was lower for multispecies models than for single-species models, suggesting that the multispecies model was effective in accommodating interannual variations in relative abundances of the five targeted species. A final example of the application of an aggregated multispecies model is Ralston and Polovina's (1982) exploration of the deep-sea longline fishery off Hawaii. Cluster analysis of catch-per-unit-effort produced 3 groups of species characterized by different depths of capture. Ralston and Polovina compared the predictions of models at varying levels of aggregation. Surplus production models were fit to individual species of the three groups suggested by the cluster analysis. The fit of data to the aggregated species model was much better for both spatially explicit and spatially aggregated data in each case.

#### 4. Integral Systems.

A final category of multispecies models considers the properties of the ecosystem to be emergent, i.e. they are not determined by the interactions at lower organizational levels (Dickie et al. 1987 - Table 6). This assumption may be justified as the total yield from a mixed species fishery is often relatively conservative, despite the high variability of yields of individual species (Regier and Hartman 1973; Ryder et al 1974; Sutcliffe et al. 1977; Holden 1978; Pauly and Christensen 1995). Furthermore, Humphreys (1979) detailed analysis of production and respiration rates suggests that the ratio of these two is constant within taxonomic groups. If such

predictable ecosystem-level, emergent properties are common, they could be the basis of multispecies management strategies. To our knowledge, no direct management application of this approach has been attempted (Table 6).

Several general, organizing principles have been suggested as being potentially important for management, such as the allometry of mortality with respect to size (McGurk 1986), and niche breadth (Pearre 1986). Here, we review the potential application of one particular property, the distribution of organism sizes in aquatic systems, and its relationship to multispecies management. Indeed, Dickie et al. (1987) suggest that body-size distributions may be “one of the most sensitive criteria for biological management effectiveness.” Many biological processes show clear allometric patterns (Peters 1983). However, it was not until Sheldon et al.’s (1972, 1973) analyses that clear allometric patterns were demonstrated at the ecosystem level. Sheldon et al.’s analysis demonstrated that, in an ecosystem, organism concentrations declined exponentially and predictably as organism size increased. Not only do abundance patterns demonstrate regularities at the ecosystem level, but so too does biomass (Sprules et al. 1983, 1991; Schwinghammer 1981, 1985; Schwinghammer et al. 1986). These initial analyses and ideas were related to potential fisheries production by Dickie and colleagues (Dickie et al. 1987; Boudreau and Dickie 1989, 1992). However, the size-spectra had considerable variability, making it uncertain if the spectra were conservative.

The properties invoked by these ecosystem-level models are now being applied to specific problems in fisheries ecology (Table 6). For example, Pope et al. (1994) have used size-spectra theory to

explain regularities in spawning time of marine fishes. Heath (1995) adopted these concepts and applied them to a particular fjord system, Loch Linnhe in Scotland. By accounting for net import and export to the Atlantic, Heath could show that the measured size spectra demonstrated repeatable patterns in agreement with the Sheldon et al. (1972, 1973) original suggestions. Investigations of trophic interactions at the ecosystem level, and application of size-spectra analysis and modeling, presently are the basis of NSF-sponsored research on Chesapeake Bay that seeks to quantify secondary production and its relationship to both nutrient inputs and the coupling of biological processes to physical features (Boynton et al. 1994).

### **C. Practical Considerations For Multispecies Approaches.**

The intuitive biological appeal of multispecies approaches is undeniably attractive (Gulland et al. 1991). Current, single-species approaches largely fail to incorporate or consider interactions between species that are important determinants of stock dynamics. Why have fishery management agencies not moved toward adoption of what Gulland et al. (1991) termed “the Holy Grail of perfect fishery management.” There are several reasons.

Dealing with uncertainty has become increasingly important in fishery management (Hilborn and Walters 1992). The uncertainty inherent in biological systems limits our ability to exploit species to their estimated maximal sustainable levels (Larkin 1963). Management already is hampered by a lack of reliable information and the adoption of multispecies models can only make the situation more difficult (Table 6). As with any mathematical system the

more complex a model becomes, the more likely that input errors propagate within the model to produce erroneous results. Thus, in general the most parsimonious model is preferred. As we have seen, while they are intuitively pleasing, all multispecies approaches that account for biological and technical interactions are more complex than their single-species equivalents (Table 6). Consequently, process error is a concern in multispecies models. Single-species models are simply likely to be more robust. Within multispecies approaches, those aimed at accounting for technical interactions are most similar to single-species models and we suspect that, at least in the foreseeable future, technical interaction models will be adopted most widely.

A second issue that limits the applicability of multispecies models is their utility (Brugge and Holden 1991). In this aspect also, biologically- and technically-motivated models differ substantially. Technical interaction models deal with problems that managers face directly (e.g. bycatch problem). Thus, the terms of the model are directly translatable into current management practices. In sharp contrast, biologically motivated models often force the manager to consider new issues. Moreover, these models often predict outcomes that are beyond the scope of management control. For example, in many organisms, abundances are regulated during early life stages; yet, managers have no control over survival at this stage.

Another factor which limits the utility of multispecies models, and also

differentiates them from the single-species approaches, is the time scale at which they operate. Single-species models operate basically at the annual level. Quotas can be and are adjusted annually or even within a season. Multispecies models incorporate entire life cycles and are inherently multi-year models. It is uncertain if multispecies models would be flexible or robust enough to account for mid-season corrections in effort allocation or interannual differences in fleet activity. Brugge and Holden (1991) suggested that multispecies models will be used best as exploratory tools which permit managers to explore the possible outcomes of alternative strategies prior to application of traditional, single-species management.

Finally, multispecies models will only be adopted if they permit us to address effectively fundamental questions. In a pessimistic note, Gulland et al. (1991) suggested that multispecies models will be adopted only if they produce management predictions in which there are no clear losers. Certainly, technical interaction models may allow us to limit the negative impacts of bycatch by regulated effort on the target species. However, the advantages of biologically-motivated multispecies models are less certain. For example, they may well suggest that increased harvest of prey species is attainable, in the face of reduction in abundance of predators by other fisheries. Many doubt that such fine-scale and directed management is possible in multispecies fisheries (Gulland et al. 1991) which are subject to environmental uncertainties and poorly understood biological interactions.

## V. THE POTENTIAL FOR MULTISPECIES APPROACHES IN CHESAPEAKE BAY

The commercial fisheries of Chesapeake Bay, like most regional fisheries

throughout the world, target a relatively few species which, because of their abundance,



traditional food value, high economic value, or recreational importance, dominate the overall catch. Yet, many species are landed, if only in minor quantities. And, some species that are not fished, yet are abundant forage species such as bay anchovy, are important contributors to production of economically valuable piscivores as well as key elements in the Bay's food web. Single-species fishery management plans (FMPs) in Chesapeake Bay do not consider the biological or technical interactions that potentially can control productivity of individual species. In this regard, Chesapeake FMPs are no different than those elsewhere in the world. Yet, the possibility of increasing productivity, optimizing economic return, and insuring sustainability is leading fishery scientists and managers to consider broad ecosystem approaches to fisheries management. Ecosystem management is a goal of the Chesapeake Bay Program and, as such, requires consideration of multispecies management of its fisheries.

Menhaden dominate the landings from Chesapeake Bay and all but 2-3% of menhaden landings are from purse-seine catches. Purse seines, which are highly selective for menhaden (Austin et al. 1994), are restricted in their use to Virginia waters of Chesapeake Bay; consequently, the geographical area of the Bay's major (by weight) fishery is limited by regulation. Other gears take more diverse catches. For example, NMFS commercial statistics for 1992 indicated that 61 species were landed in Bay fisheries. Poundnets accounted for 45 species and gillnets for 53 species. Recreational fishermen also account for a diverse catch, although they too target a few desirable species. It is clear that the Bay supports a multispecies fishery. However, it is uncertain if multispecies approaches will

improve overall resource management in the short term because of inadequate knowledge of biological interactions at the ecosystem level. Research aimed at understanding such interactions is much desired as the Bay Program moves toward full implementation of an ecosystem approach in living resources management.

Do the forces that are leading towards multispecies approaches elsewhere exist in the Chesapeake Bay? Several lines of evidence suggest that they do (Table 5). Evidence comes from both technical and biological interactions. Although the purse-seine menhaden fishery is quite free of bycatch, it accounts for >65% of the annual, coast wide landings of menhaden (Ahrenholtz 1991; Ahrenholtz et al. 1987). There is concern, although no evidence at this time, that the fishery competes with piscivore predators (e.g. bluefish, weakfish, striped bass) for the menhaden resource. Hartman and Brandt (1995a, b) have clearly shown that menhaden constitute a major fraction of the diet of piscivores in Chesapeake Bay and that the three major species potentially compete for prey. We have indicated how these interactions potentially can have complex and sometimes counter-intuitive effects on overall abundances (Fig. 21). In another example, concern has been expressed recently over the predatory impact of the recovering striped bass population on the stock of blue crabs. Available scientific evidence for this interaction is weak (Booth and Gary 1993; Goshorn and Casey 1993), but possibly significant. Historically, it is certain that blue crab and striped bass coexisted in high abundances in Chesapeake Bay. However, blue crab catches, fishing effort, and fishing mortality rates have increased markedly over the past 60 years. It is conceivable that mortality of prerecruit blue crabs from

predation by striped bass and other predators now constitutes a significant control on abundance of fishable crabs. In this example, man and striped bass compete for a limited supply of blue crabs, in which the "bycatch" of blue crabs in the striped bass diet generates the interaction.

The fisheries in the Bay are managed by complex sets of largely independent regulations in individual FMPs. No formal procedures exist in which the effects of changing regulations for one species are considered with respect to potential effects on other species. As we pointed out, this insufficiency was a motivating factor that is leading towards imposition of multispecies approaches in the New England and other fisheries. In this consideration, preliminary steps that include workshops to define the specific issues, research programs to address the issues, and development of multispecies models to understand implications are required before multispecies management might be adopted in the Bay.

What species or species groups are the "strong interactors" and thus likely to be central to a multispecies approach? While it is unlikely that a complete, predictive understanding of the food-web dynamics will be achieved for the Chesapeake Bay, it is already apparent that some species are more likely to interact than others and therefore may be critical in a multispecies plan. For example, the suite of top piscivores clearly is linked by their common reliance on menhaden and bay anchovy prey. Moreover, blue crab, because of its reliance and probable impact on benthic prey, as well as its possible control through predation on it by predatory fishes, is a candidate for inclusion in multispecies planning. The complex of alosid species (shads, river herrings), and other planktivore fishes (menhaden, bay anchovy, gizzard shad) also

are potential strong interactors. Oysters and menhaden form a species pair that depends upon filtration of phytoplankton to meet nutritional demands. To what extent do oysters and menhaden substitute for or complement each other in the Bay's trophic structure and dynamics? Finally, species with no apparent direct link to harvestable fisheries, e.g. jellyfishes which are major consumers of zooplankton and ichthyoplankton, may indirectly limit fisheries productivity and thus are "strong interactors," deserving to be included in planning for management of multispecies fisheries in Chesapeake Bay.

What multispecies approach might be most appropriate, and what advantages would be gained through its adoption in the Chesapeake Bay? As discussed earlier, the intensely seasonal nature of the Bay's food web complicates making a choice. Large portions of the life cycle of many species in the Bay's fisheries are being driven by processes external to the Bay. For example, spawning and early life dynamics of menhaden, spot, and croaker occur offshore and away from the influence of the Bay's estuarine environment. Moreover, age-specific and season-specific migratory behaviors of many species that reside seasonally in the Bay occur on a coast wide scale (e.g. menhaden, bluefish, weakfish, striped bass). This suggests that techniques such as the MSVPA will be difficult to apply. Application of such techniques will require information from a broad region of the mid-Atlantic Bight in addition to information from the Bay. We suggest that, initially, descriptive multivariate techniques and multivariate system approaches may be most successful (Kerr and Ryder 1989). Supporting this suggestion, much of the data on commercial species that is required to implement a multispecies surplus production

model already exists. However, there remain serious deficiencies in the data base that must be addressed before any multispecies management approach could be envisaged.

We identify the following areas as principal impediments to implementation of multispecies approaches in the Chesapeake Bay, and as areas where research is needed to guide the Bay Program toward decisions regarding the advisability of multispecies management.

1. Systematic information on catch and effort for exploited stocks. Rothschild et al. (1981) previously identified this concern. For many of the Bay's commercial stocks, effort data for both commercial and recreational components of the catch are unavailable or of unknown quality. A major assumption in the recent CBSAC stock-assessment of blue crab was the level of the recreational harvest. Until commercial and recreational landings and associated effort data are collected consistently, the ability to infer trends or changes in abundance will be compromised. Moreover, improvements in this area are required even if multispecies approaches are not adopted. Such data are fundamental to effective, long-term single-species management.
2. Fishery-independent estimates of abundance for the principal species in the Chesapeake Bay. A major problem in identifying multispecies patterns in the landings data is the lack of reliable time series of abundances for many species. In some cases, landings may be a poor indicator of abundance. It is critical to

obtain Baywide abundance information on key species such as the heavily exploited menhaden and the unexploited bay anchovy, which almost certainly dominate the numbers and biomasses of fish in the bay. Some of the shortcomings in abundance estimates are currently being addressed through increased use of hydroacoustic assessments in Baywide research programs. It is unlikely that any multispecies approach can be fully implemented until the abundances of major target and non-target species are known.

3. Basic life history information. There remains a lack of basic knowledge on the life histories of many harvested organisms in Chesapeake Bay, which impedes moving confidently towards multispecies management. For example, it is difficult to know if restricting crab pots to water depths less than 40' will conserve female blue crabs, as has been suggested by some, because too little is known about the behavior, pathways or cues that control migrations of blue crabs. We generally know little about causes of variability in recruitment and abundances of many species in the Bay community. Better information is required on stage-specific mortality rates and on bioenergetics relationships for key fisheries species and for organisms that support productivity of the harvested species. It is especially important to gain a better comprehension of predator-prey relationships and competitive interactions that can exercise control over species abundances, productivity, and

- potential yields.
4. Effects of habitat alteration. Loss, degradation, or alteration of habitat can lead to changes in community structure and productivity. There is an insufficient knowledge of effects of habitat changes in Chesapeake Bay through habitat destruction (e.g declines in SAV and oyster reefs) or habitat deterioration brought about by deliberate alteration, the presence of toxicants, or disease organisms. In the case of anadromous fishes, the productivity of this multispecies group may have been permanently damaged by dams and blockages on spawning tributaries, although a major effort by the Bay Program is underway to alleviate these conditions. Understanding the spatial extent of suitable habitats of key harvested species and of strongly interacting species is critical for long-term multispecies management.
5. Lack of detailed understanding of multispecies models. Much of traditional single-species management relies on an extensive body of experience in applying traditional models to a wide variety of fisheries. Despite the long tradition, a high degree of uncertainty remains. In the case of multispecies models, experiences are few and knowledge of their behavior and probability of success are limited. Consequently, their utility is more uncertain in the eyes of managers than are more familiar single-species models. The uncertainties should not deter efforts to improve and understand the multispecies models. In the long term, fisheries management regionally and globally seems certain to move toward multispecies approaches that are compatible with ecosystem management goals.

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