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Stock assessment of the blue crab in Chesapeake Bay 2005: Final report

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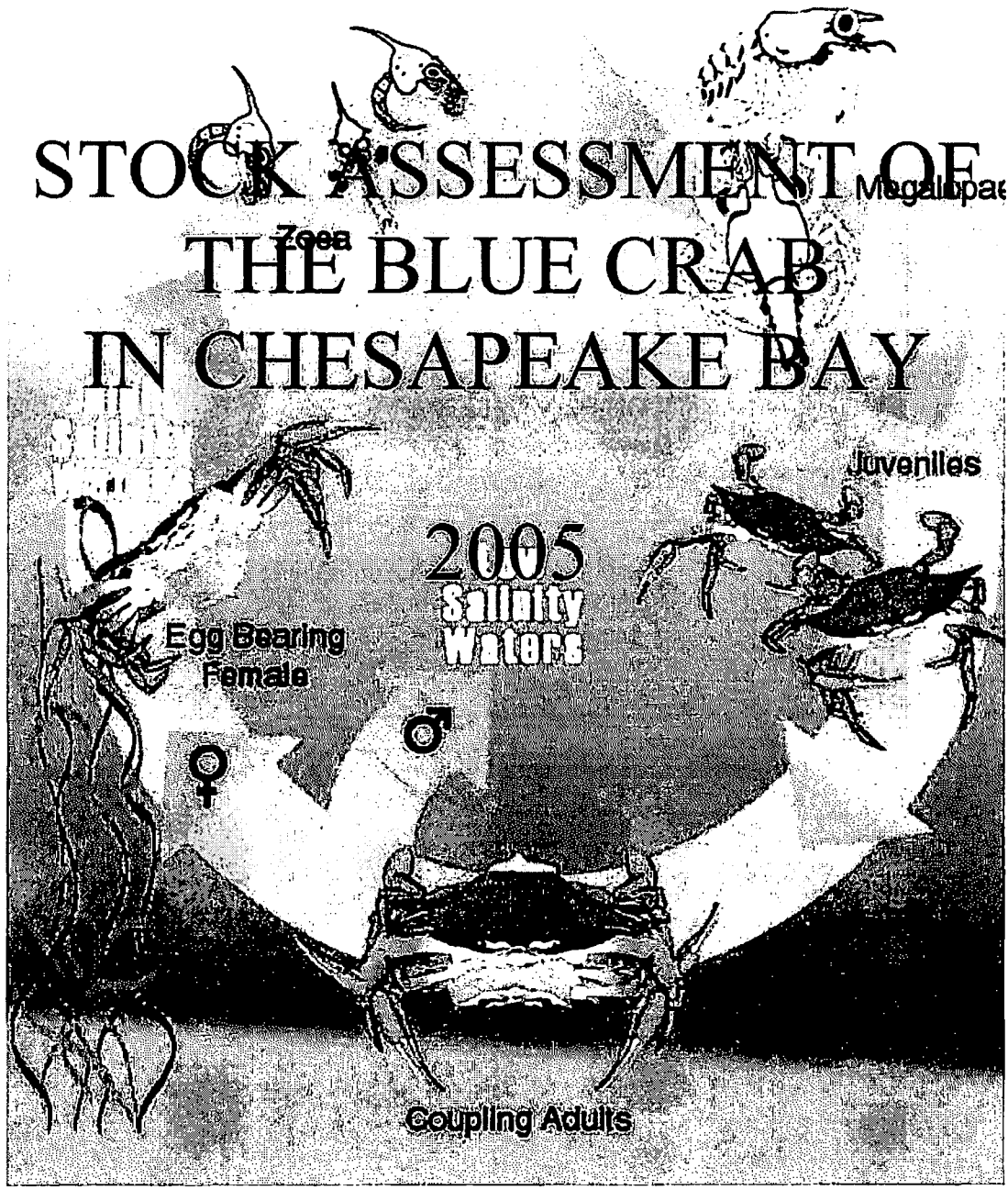
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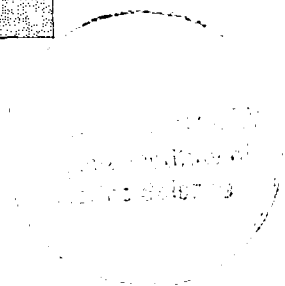
STOCK ASSESSMENT OF THE BLUE CRAB IN CHESAPEAKE BAY

2005
Salinity
Waters

Egg Bearing
Female

Juveniles

Coupling Adults



Stock Assessment of Blue Crab in Chesapeake Bay 2005

Final Report

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Executive Summary

The blue crab (*Callinectes sapidus*) is an icon for the Chesapeake Bay region. The commercial fisheries for blue crab in the Bay remain one of the most valuable fishery sectors in the Bay. Ecologically, blue crab is an important component of the Chesapeake Bay ecosystem. Thus, sound management to ensure the sustainability of this resource is critical.

The first bay wide assessment for blue crab was undertaken in 1995 and completed in 1997. It concluded that the stock was moderately to fully exploited and at average levels of abundance. Subsequent to this assessment concerns over the continuing status of blue crab were raised because of declines in abundance and harvests. In response to concerns from stakeholders, a Bi-State Blue Crab Advisory Committee was established in 1996. Work by this committee led to the establishment in 2001 of biomass and exploitation thresholds and an exploitation target reference points. Since 2001, the status of the blue crab stock has been updated annually and its status determined relative to the reference points. Over the ensuing years, the approach to determining the status of blue crab in the Chesapeake Bay has been modified, but a new inclusive assessment has not been conducted.

In 2003, we proposed and were funded to complete a thorough revision of the stock assessment for the blue crab in Chesapeake Bay. The following terms of reference were adopted to guide our assessment activities. We sought to (i) assess and quantify the life history and vital rates of blue crab in the Chesapeake Bay that are relevant to an assessment of the stock, (ii) describe and quantify patterns in fishery-independent surveys of blue crab abundance, (iii) describe and quantify patterns in catch and effort by sector and region, (iv) develop and implement assessment models for the Chesapeake blue crab fisheries, and (v) re-evaluate, and where necessary, update control rules for Chesapeake Bay blue crab fishery.

In conducting the assessment we sought to overcome some of the challenges that the biology of and fisheries for blue crab present. For example, uncertainties in estimates of natural mortality and growth dynamics produced concerns over the reliability of previous population models. Furthermore, interpretation of data on the historical harvest of blue crab in the Chesapeake has been made more difficult because of changes in the way in which harvests are reported to the individual jurisdictions. Thus, we see the following elements of the assessment we have developed as representing substantial advances that increase the chance of maintaining a sustainable blue crab fishery. We have re-evaluated estimates of natural mortality rates using both empirical and life history-based approaches. We have applied time series analysis to adjust historical landings for the known reporting changes. We developed a new assessment model that uses the corrected landings and data from all relevant fishery-independent surveys to understand changes in abundance and exploitation pressure of blue crab. Previous reference points were based on the rate of fishing mortality, F . Calculation of these reference points and the status of the stock relative to them required an estimate of the rate of natural mortality, M . Changes in the estimate of M would cause changes in the

reference points and of our understanding of the historical pattern of exploitation that had operated in the fishery. To overcome this problem, we developed an individual-based spawning potential per recruit model to estimate reference points based on the fraction of the vulnerable population that was harvested each year.

Our review of the biology of the blue crab in Chesapeake Bay supported the assumption that there was a single unit stock of blue crab in the bay. While there is likely to be exchange of individuals with neighboring populations (e.g., Delaware Bay), the data indicate that these are likely not a substantial or persistent feature of the dynamics of the Chesapeake Bay population. Our review of the life history of blue crab indicated that the most likely value for $M = 0.9$. This estimate is supported by direct, empirical estimates from tagging studies in Chesapeake Bay, by an analysis of life history patterns in the species generally, and by the relationship between the rate of total mortality and effort in Delaware Bay.

Our review of relevant fishery-independent surveys indicated that the blue crab population in Chesapeake Bay is likely at below average abundances. Although some indices have increased in the most recent years, the majority of indices still indicate the population is below its average abundance levels. In particular, the low abundance of spawning females in the lower Chesapeake Bay is worthy of close monitoring. We noted that changes to size-at-age conventions used to convert size-specific abundance information to age classes implemented since the last assessment have improved the ability of these surveys to track changes in the population. However, our understanding of the dynamics of the blue crab in Chesapeake Bay would benefit from a rigorous evaluation of the size structure data in these surveys.

We applied time series techniques to adjust for the effects of reporting changes on estimated landings. We found that the 1993 reporting change in Virginia and the 1981 and 1993 reporting changes in Maryland all significantly affected the estimates of landings. The reporting change in Virginia led to an average 20.3% decrease in estimated landings prior to 1993. The reporting changes in Maryland led to an 84% change in the estimated landings. Our reconstructed estimate of total baywide landings suggest that the average landings for 1945-2003 was $34,887 \pm 5,490$ MT. The reconstructed landings indicate that removals have been 16.7% higher than previously reported. The highest recorded baywide harvest was 47,719 MT in 1966. The lowest recorded baywide harvest was 21,539 MT in 2001. Landings in the three years 2000 – 2002 all set record lows for the time series.

We analyzed data from the winter dredge survey to estimate the proportion of the vulnerable population that is harvested each year. This was termed the exploitation fraction, μ . Estimates of μ have varied from 33% in 1991 to 71% in 1999. Current estimates of μ indicate that less than 50% of the vulnerable crabs are being caught each year. Importantly, the estimates of μ are independent of estimates of M , and will not change if estimates of M are changed by subsequent research. This is not the case with estimates of F , calculated from the same data. Thus, we recommend adoption of μ as the measure of fishing pressure for future assessments.

We developed an extension of the Collie Sissenwine Catch Survey model that permitted multiple fishery-independent time series to be used in assessing the population. The new catch-multiple survey (CMS) model utilized three fishery-independent surveys and the reconstructed commercial catch time series. Comparison of the predictions from the model with observed patterns of abundance and empirical estimate of exploitation fraction indicated that values of natural mortality $0.9 < M < 1.2$ were most likely. In all cases, the model indicated a disturbing pattern of exploitation in which the fraction exploited increases as abundance decreases. This compensatory pattern presents challenges to the sustainable management of the resource.

We used an individual-based model which captured the discrete nature of crab growth and the diverse sectors in the blue crab fishery to estimate exploitation fraction reference points using spawning potential per recruit criteria. The exploitation fraction threshold reference point was determined to be 53%, based on maintaining 10% of the virgin spawning potential. The exploitation fraction target reference point was determined to be 46%, based on maintaining 20% of the virgin spawning potential. We maintained the previously endorsed biomass threshold of the lowest abundance observed in fishery-independent time series. Based on these revised thresholds, we conclude that the blue crab stock is not overfished (i.e., it is at a higher level of abundance than the threshold), nor is it currently experiencing overfishing (i.e., the exploitation fraction is below the threshold). However, results indicate the stock did experience overfishing recently. As a result of this overfishing, the stock is currently at a relatively low level of abundance. Importantly, when exploitation fractions similarly to values currently observed have been seen in the past, abundances were substantial higher. This suggests potential for future short-term increases in abundance if the lower exploitation fractions are maintained.

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Terms of Reference

This work was funded by a grant from the NOAA Chesapeake Bay Office. The work had the following objectives that were adopted by the Blue Crab Assessment Group as the terms of reference (ToR) for the assessment. These were:

- ToR 1. Assess and quantify the life history and vital rates of blue crab in the Chesapeake Bay that are relevant to an assessment of the stock.
- ToR 2. Describe and quantify patterns in fishery-independent surveys.
- ToR 3. Describe and quantify patterns in catch and effort by sector and region
- ToR 4. Develop and implement assessment models for the Chesapeake blue crab fisheries
- ToR 5. Re-evaluate, and where necessary, update control rules for Chesapeake Bay blue crab fishery

1. Introduction

The blue crab (*Callinectes sapidus*) is one of fourteen swimming crab species in the genus *Callinectes*. Nine of the fourteen swimming crabs, including the blue crab are endemic to the western Atlantic basin, mainly in tropical and subtropical areas. The distribution of the blue crab is much wider than its conspecifics as it ranges from Uruguay to Massachusetts, with occasional records from Argentina to Nova Scotia (Williams 1974, Norse 1977). In addition to its endemic range, the species has become established as an exotic in the Mediterranean basin (Holthuis 1961).

Throughout this range, the blue crab is an important component of estuarine ecosystems (Baird and Ulanowicz 1989). Blue crabs are dominant and opportunistic benthic predators and scavengers (Eggleston et al. 1992). Their diets may include a wide range of taxa including bivalves, crustacea and fish (Hines et al. 1990, Mansour and Lipcius 1991). It is a dominant benthic predator and scavenger (Eggleston et al. 1992). Diets vary with crab size. Small crabs exploit thin-shelled bivalves and other invertebrates that are buried relatively shallowly in the sediments. Larger crabs can exploit thicker shelled bivalves and cannibalism is not uncommon (Dittel et al. 1995, Hines and Ruiz 1995). Thus, crabs may be keystone predators in the estuary, *sensu* Paine (1966), possibly playing a dominant role in structuring benthic communities throughout its range.

In addition to its ecological importance, the blue crab supports important commercial and recreational fisheries throughout much of its range. Blue crab has been

harvested since pre-colonial times. The commercial fishery started in earnest in the mid-nineteenth century (Cronin 1998). Commercial landings are regularly reported from coastal states from Texas to New York¹. Additional small landings are occasionally recorded in Connecticut and Rhode Island. Since 1950, annual landings of blue crab along the Gulf and Atlantic coasts have averaged 75,811 metric tonnes (MT) or more than 167×10^6 Lbs, with an average annual value of $\$57 \times 10^6$. In the 1950's, the Chesapeake Bay region represented almost 80% of the national landings. This figure has fallen steadily since then, so that now (2003), the Chesapeake Bay represents only approximately 30% of the national landings. Over the last decade, the top four states reporting crab landings have been North Carolina (22,252 MT), Louisiana (20,022 MT), Maryland (16,757 MT) and Virginia (14,974 MT). Thus, although in combination, the Chesapeake Bay remains the largest single source of blue crab harvest in the nation, based on landings the states of Maryland and Virginia have fallen to third and fourth place in national rankings.

Maryland, Virginia and the Potomac River Fisheries Commission are the management jurisdictions for blue crab in Chesapeake Bay. The management actions of the three jurisdictions are coordinated since all are signatories to the Chesapeake Bay Blue Crab Fishery Management Plan (FMP) (Chesapeake Bay Program 1997). The FMP provides recommendations for the management of commercial and recreational fishing of blue crab in the Bay. Its goal is "to manage blue crabs in the Chesapeake Bay to conserve the bay wide stock, protect its ecological value, and optimize the long-term utilization of the resource." Regulations and management actions are complementary across the jurisdictions, but recognize age-specific and sex-specific differences in utilization of the estuary by blue crab, and historical fishing patterns. The objectives and recommendations of the FMP focus on stabilizing the fisheries, limiting access to the fisheries, preventing increases in exploitation rates, monitoring the blue crab resource, improving enforcement, and developing better socioeconomic databases. The blue crab FMP adheres to the principles proposed for Chesapeake Bay FMPs that were developed by the Chesapeake Bay Program in 1998, in which precautionary management and protection of critical habitats are highlighted. This FMP is scheduled for revision in 2005 following implementation of the new Chesapeake 2000 Agreement (Chesapeake Bay Program 2000) which called for the implementation of ecosystem-based fishery management.

A Baywide stock assessment of blue crab had been lacking until NOAA's Chesapeake Bay Stock Assessment Committee (CBSAC) undertook the task. In 1997, the Technical Subcommittee (TSC) of CBSAC produced the first comprehensive stock assessment of blue crab in Chesapeake Bay (Rugolo et al. 1997). The CBSAC Assessment (Rugolo et al. 1997) indicated that stock abundance had been high in the

¹ Data from NOAA's Fishery Statistics and Economics Division, available online at <http://www.st.nmfs.gov/st1/>

1980s and had returned to average abundance in the following years (i.e., up to 1995). The CBSAC assessment noted a decrease in catch-per-unit-effort (CPUE) in the blue crab fishery since 1945 but no apparent declines in CPUE or fishing mortality rate since 1970. The TSC attributed these counter-intuitive results to gear saturation effects as the amount of commercial gear proliferated in recent years. The stock assessment also found that recruitments of young crabs had been above average since the 1970s. The stock was characterized as moderately to fully-utilized at the exploitation levels then occurring. The TSC recommended establishing and maintaining a fishing mortality rate reference point that insured escapement of at least 10% of the spawning stock that would be present in the absence of fishing. Although finding no cause for alarm, the TSC recommended no further increases in fishing effort or fishing mortality. The status of the stock has been updated regularly by the CBSAC since the assessment was completed².

Following the CBSAC assessment, Miller and Houde (1999) revisited the assessment of the blue crab fishery to develop threshold and target reference points. Miller and Houde recommended a hierarchy of target levels, designated to address sustainability, efficiency, and recovery scenarios. Targets were derived from 1) reported catches and effort in the commercial fishery, 2) statistics from fishery-independent surveys, and 3) knowledge of the biology of blue crab. Targets recommended included population sizes, catches, and effort levels, as well as reference fishing mortality rates. They were intended to be conservative and risk-averse and promote a sustainable and economically viable fishery, while protecting the ecological value of the blue crab in Chesapeake Bay. In the hierarchy, the first targeting level was one that designated population abundances and fishing mortality rates to ensure sustainability of the resource. Miller and Houde recommended a long term potential yield of ~36,000 MT (80×10^6 Lbs) and fishing mortality rates, $F < 0.9$. A second target level equivalent to $F=0.6$ was recommended to ensure that the maximum reproductive potential per crab would be obtained over the long term. A recovery target was also recommended of $F < 0.5$ to help build the stock in the case of recruitment overfishing. Some of the recommendations from the Miller and Houde assessment differed substantially from the earlier assessment as these authors interpreted the effects of a reporting change that occurred in Maryland in 1981 differently than had Rugolo et al. (1997). Fogarty and Miller (2004) demonstrated the impacts of reporting changes in the blue crab fisheries and argued that accounting for them would be important in future assessments.

In 1996, the Governors and Legislatures of Maryland and Virginia established the "Bi-State Blue Crab Advisory Committee" (BBCAC) to provide them with independent advice on the status and future trends of the blue crab fisheries. In 1998, BBCAC endorsed the findings of its technical work group that indicated that there were signs that the crab population was not in a healthy condition. Specifically BBCAC identified the following indicators of concern:

² The most recent update is available on line at <http://noaa.chesapeakebay.net/Fish/default.htm>

- Overall abundance for all age groups is down,
- Fishing mortality is increasing,
- Fishing effort is at near record levels,
- Spawning stock biomass is below the long-term average,
- The average size of crabs is decreasing,
- Fishery-independent surveys show a decreasing percentage of legal size crabs,
- The reproductive potential of the population may be comprised due to the reduced size of males and lack of mature females.

This consensus view motivated the development of a new management framework for the Chesapeake Bay blue crab fisheries (Miller 2001c). The recommendations recognized the need to distinguish between threshold and target reference points. Specifically, the framework identified biomass- and exploitation-based threshold reference points that bounded a zone of sustainable exploitation (Fig. 1). Within this zone of sustainable exploitation, researchers recommended a target exploitation rate that sought to double the current spawning potential of the blue crab population (Fig. 1). In making these control rules functional, empirical evidence and elementary per recruit analyses were combined to determine values for the threshold and target reference points. The abundance threshold reference point was determined to be the lowest standardized abundance (Z-score) that had been observed in the fishery-independent survey data. This was determined to be the value observed in 1968. The justification for this choice was that evidence was lacking to suggest that lower abundances could support a sustainable fishery. The fishing mortality rate threshold was determined from a standard spawning potential per recruit analysis. A value of F10% ($F=1.0$) was chosen based on previous precedence and because the value indicated was greater than the majority of fishing mortality rates that had been observed previously. The target reference point was chosen as F20% ($F=0.7$). This level was chosen as it was believed to be sufficiently far from the threshold reference points as to be detectably different, and would lead to an effective doubling of the spawning stock present in 2001.

Our understanding of the ecology of the blue crab and the fisheries it supports has continued to develop following the establishment of the threshold and target reference points described above. Considerable research effort has been invested in developing a greater understanding growth processes in blue crab, including the development of techniques to age crabs which offer hope for continued improvement in the future as well. Closer examination of the application of length-based approaches to estimating the rate of fishing mortality F raised concern regarding the sensitivity of this approach for blue crab. Accordingly, from 2002 annual updates on the status of the blue crab stock abandoned the length-based approach in favor of a direct enumeration method, based on ratio of the number of crabs caught in the different fisheries to the number available at the beginning of the season, estimated from the winter dredge survey. Finally, all management jurisdictions implemented conservation measures as a result of the BBCAC

recommendations. Thus, it is appropriate to re-asses the Chesapeake Bay blue crab stock to determine its current status and provide a foundation for evaluating management actions.

For this assessment we revisited all of the elements of the earlier assessments, together with assessment reports for other blue crab populations in the region (e.g., Delaware Bay (Helser and Kahn 1999, Kahn 2003), and North Carolina (Eggleston et al. 2004). In the sections that follow, we (i) review blue crab ecology as it relates to the assessment, (ii) describe the Chesapeake Bay fisheries and relevant fishery-dependent data, (iii) review the fishery-independent data, (iv) develop an assessment model of the Chesapeake Bay blue crab stock and (v) review reference points and provide a new exploitation fraction based reference point. Finally, we make recommendations for future assessment work, monitoring and management

2. Biology and Life History

2.1 Stock Structure

The blue crab ranges from Uruguay to Massachusetts (Williams 1974). Population structure within this range is uncertain. In 1994, McMillen-Jackson et al. (1994) used a protein electrophoretic approach to quantify the genetic variability in samples collected from Texas to New York. This research indicated moderate genetic structuring, with spatial patchiness of several loci evident throughout the range. However, the findings also indicated that a high level of regional gene flow acted to diminish population structure. Recently, these researchers have revisited the question of population structure within the blue crab using multiple genetic markers and restriction length fragment polymorphism analysis of mitochondrial DNA (McMillen-Jackson and Bert 2004). The genetic results indicated no clear split between Gulf of Mexico stocks and Atlantic coast stocks. However, there was, within the Atlantic coast a cline of genetic diversity, with the New York samples exhibiting significantly lower diversity than more southerly stocks. The authors inferred from these patterns a latitudinal expansion from a subtropical center of diversity. Furthermore, the maintenance of a cline in diversity suggests that local gene flow may be low or restricted. More recently, scientists at the Center for Marine Biotechnology have succeeded in developing genetic techniques to distinguish crabs from the Chesapeake from those from North Carolina (A. Plaice, COMB, pers. comm.). Thus, while there is no definitive evidence of genetic structuring indicative of separate populations, there is clear evidence of localized populations that experience limited gene flow between them. This is evidence for the existence of, at a minimum, a functionally separate Chesapeake Bay blue crab stock, that experiences only limited exchange of individuals with neighboring stocks.

Studies of larval distributions (see section 2.5.) provide further evidence for the presence of a “quasi-discrete” Chesapeake Bay stock (see Section 2.4.3). After being released, zoea move seaward, where they develop and return to enter estuaries as megalopae. While the precise details of the physical context and behavioral mechanisms employed by larvae to return to estuaries are not fully understood, what is known suggests that large scale exchanges of larvae are likely not typical. The prevailing oceanography of the regions suggests that only the Chesapeake Bay and more northerly populations (i.e., coastal bays and Delaware Bay) are sources of potential recruits to Chesapeake Bay. This suggests that population interchange is restricted. Furthermore, aspects of the physical environment and behavior of zoea suggests that the exchange is likely not a persistent feature of the dynamics of the different populations. Female crabs release zoea near the mouth of coastal Atlantic estuaries. Natunewicz and Epifanio (2001) found that zoea occur in distinct patches 0.5 – 2.5 km diameter in the vicinity of the mouth of Delaware Bay. Modeling studies by Garvine et al. (1997) indicated that some larvae return to Delaware Bay using upwelling-favorable wind events. However, these modeling studies also indicated that a not insignificant proportion of zoea are advected southward in a buoyancy driven coastal current. These larvae may represent

potential recruits to the Chesapeake Bay population. Studies of recruitment in the Chesapeake Bay stock indicate a similar picture to that found for Delaware. Roman and Boicourt (1999), found patches of zoea associated with the Chesapeake Bay plume front. In a numerical analysis Johnson and Hess (1990) estimated that only 13% of released zoea remained in the Chesapeake Bay and that the remaining zoea (87%) are advected out to sea. Johnson and Hess (op. cit.) calculated that 29% of the zoeal production returns to the Chesapeake Bay. It is important to note that these figures do not include zoeal mortality, which is likely to be substantial, and thus represent an upper bound.

From this review, we conclude that there is sufficient evidence to support the assumption that the blue crab population in the Chesapeake Bay comprises a unit stock, at least for assessment purposes. This does not imply that there is no exchange with or subsidy from neighboring populations; rather it assumes that the dynamics of the Chesapeake Bay population are determined from internal considerations, and not from subsidies or exchanges with other populations. Subsidies and exchanges do likely occur – we are simply assuming that they are not significant. However, we note that such subsidies and exchanges are likely to be more important when the size of the Chesapeake Bay population is small.

2.2 Growth

Blue crab growth have been widely studied and a recent summary of the physiology and ecology of growth is provided by Smith and Chang (in press). As with all crustaceans, crabs grow by molting (ecdysis). The molting process follows a well-described cycle in which five premolt stages, which prepare the organism for molting, are separated from postmolt and intermolt stages by ecdysis itself. Observations indicate that the molting cycle can be separated into a physiologically fixed time period required for the preparation for molting, molting and post molting processes occur, and a temporally variable intermolt period that can be affected by environmental and biotic factors.

There have been two principal approaches to describing growth in the blue crab: molt-process modeling and continuous growth modeling. The former provides an accurate representation of the physiological nature of growth, whereas the later provides a statistical representation of changes in size at age (Miller and Smith 2003). The first full molt-process model for blue crab was developed by Smith (1997). In this work, Smith developed a degree-day based formulation to incorporate the effects of temperature on the intermolt period. This approach accurately represented both the overwinter cessation of growth, and the impact of interannual variation in temperature on growth rates. The underlying model structure reflects the physiological controls on growth in having both a fixed and variable component to the overall intermolt period. The growth achieved per molt was estimated in Smith's model as a constant fraction of pre-molt size. In laboratory experiments, Brylawski (2002) reported an average growth per molt of $119.4\% \pm 7.5\%$ for Chesapeake Bay blue crabs. He found that growth per molt was not affected by sex or initial size over the range 25 - 70 mm carapace width (CW). In contrast, growth per molt was affected by temperature over the range 16 - 28 °C. In

parallel field enclosure studies, Bylawski reported a growth per molt of $122.4 \pm 7.6\%$. A typical trajectory from such a model is shown in Figure 2. More recently, Eggleston et al. (2004) used data from mark-recapture studies in the Sounds of North Carolina to parameterize a growth per molt model from empirical data collected in the field. Thus, it is not necessary to rely on laboratory or mesocosm generated data to parameterize such models.

A principal disadvantage of growth per molt models is that they are numerical simulations. The models do not yield simple parameter estimates that summarize growth and thus are difficult to include in stock assessment models (Miller and Smith 2003). One approach used by Smith (1997) and Eggleston et al. (2004) to overcome this difficulty was to use the simulation model to generate the growth trajectory, and then use a continuous model, such as the von Bertalanffy model, to describe the resulting data. However, this approach is at odds with the initial philosophical justification for the model (Miller and Smith 2003). A better approach is to incorporate the assessment model directly into the simulation. For example, Bunnell and Miller (in press) have extended Smith's approach to include consideration of mortality and yield (see Section 7.2). The Bunnell and Miller model performed well in evaluation studies which used the model to predict the size distribution of crabs from one winter to the next, given only the initial size distribution and the specific annual temperature cycle (Fig. 3).

The second approach to modeling growth is to apply traditional continuous time descriptions of growth such as the von Bertalanffy model. The application of these approaches to blue crab growth were reviewed by Miller (2001a). Several published models are available, although they differ in the nature of the underlying data used in estimating model parameters, and in the fitting algorithm itself. These models are described below and the parameter estimates are given in Table 1. In addition, Table 1 gives the predicted sizes at the onset of winter (December 15) for ages 0-2, assuming peak settlement occurs on July 15.

Two studies have developed growth curves based on assumptions about the maximum size and age of blue crabs. We note that parameter estimates from these models depend strongly on these assumptions, which limit their utility. Rothschild et al (1991) modified a von Bertalanffy function to account for the discrete nature of growth. Rothschild and colleagues considered the size of crabs L_m at a series specific molts, m . The expected time to a molt, t_m , can be derived from the traditional, continuous von Bertalanffy model as

$$t_m = -\frac{1}{K} \ln \left(\frac{L_\infty - L_m}{L_\infty} \right).$$

Subsequently, the intermolt period between successive molts can be written as

$$t_m - t_{m-1} = -\frac{1}{K} \ln \left(\frac{L_\infty - L_{m-1}}{L_\infty - L_m} \right)$$

This results in a series of estimates of intermolt period from the 0 stage to the final stage, ψ . From these simultaneous equations, K can be estimated as

$$K = \frac{1}{t_\psi - t_0} \ln \left(\frac{L_\infty - L_0}{L_\infty - L_\psi} \right)$$

To estimate K, Rothschild et al. (1991) set t_0 and L_0 equal to zero, t_ψ equal to six and set L_ψ to Churchill's (1919) estimate of 178 mm CW. L_∞ was estimated based on a life history invariant assumption that L_ψ is 95% of L_∞ . This resulted in estimates of the von Bertalanffy growth parameters of $K=0.506$ and $L_\infty = 187$ mm CW (Fig. 4).

In their review of the Chesapeake Bay blue crab stock, Rugolo et al. (1997) developed a von Bertalanffy model for blue crab based on estimates of the upper bounds of size at age and an assumed maximum carapace width of 260 mm at age 8. They generated von Bertalanffy parameter estimates of $K=0.587$, $t_0=0.115$ and $CW. = 262.5$ mm using the FISHPARM program (Table 1 & Fig. 4).

Other approaches to estimating growth in blue crabs have relied on empirical data collected in the field, mesocosms or in the lab. Rothschild et al. (1988) fitted a von Bertalanffy function to modal length classes collected in the commercial scrape fishery around Smith Island, MD. Their analysis involved fitting modal lengths on the date of collection and assumed a 12 month separation between modes. The resulting size-at-presumed age data were subjected to nonlinear regression. This analysis yielded von Bertalanffy parameters of $K=1.08$ and $CW. = 176$ mm (Table 1; Fig. 4).

Based on lipofuscin ageing, Ju et al. (2001) estimated von Bertalanffy model parameters for Chesapeake Bay blue crab. In this approach crab age was estimated from a biochemical assay that quantified the level of protein complexes called lipofuscin that accumulate in tissues over time. By measuring the lipofuscin level in non-dividing tissues, such as nervous tissue, Ju and colleagues were able to estimate physiological age. Validation studies have been conducted that permit the absolute level of lipofuscin to be correlated with chronological age based on crabs raised in raised in both the laboratory and in artificial ponds (Ju et al. 1999). Ju and colleagues have used this approach to estimate growth of crabs in the laboratory, in mesocosms and in the field. Laboratory reared crabs were held in individual containers at a constant 20°C and fed an artificial diet. Crabs raised in artificial ponds were held at ambient conditions and allowed to forage on naturally abundant prey. Analysis of the two rearing conditions yielded different estimates of CW. (lab: 180.9 mm, field: 240 mm), and markedly different estimates of the k (Fig. 4). The estimates were $K_{lab}=0.35$, $K_{field}=0.70$ (but with a significant t_0 offset = 0.11). We considered the estimates based on crabs reared in the laboratory too artificial to accurately describe growth of crabs in the field, and do not consider them in the assessment.

The final von Bertalanffy estimates for the Chesapeake Bay is derived from Smith's (1997) modeling of growth using the discrete molt-process model. Smith constructed growth patterns for blue crab using empirically derived estimates of growth per molt and inter-molt period as described above. He subsequently analyzed the output of the model to estimate von Bertalanffy parameters that best described the growth trajectory generated. His estimates are $K=0.64$, $CW_{\infty}=192$, and $t_0=0.31$. These parameters yield estimates of sizes at the onset of overwintering in the first, second and third years of 32.5, 107.5 and 147.6 mm CW.

More recently, Ju et al (2001) developed a seasonalized version of a von Bertalanffy model for Chesapeake Bay blue crab. The model is of the form

$$L_t = L_{\infty} \cdot \left[1 - \exp \left\{ -K' \cdot (t - t_0) - \left(\frac{CK'}{2\pi} \right) \cdot (\sin 2\pi(t - t_0) - \sin 2\pi(t - t_0)) \right\} \right]$$

This model induces seasonal sinusoidal variability in size at age, the magnitude of which is determined by the value of C and the period of oscillation (Table 2). Unlike traditional von Bertalanffy models, this form of the model does allow for a cessation of growth during winter (Fig. 5). However, the functional form of this equation does not readily lend itself to incorporation into assessment models.

Several studies in regions neighboring the Chesapeake Bay have used a von Bertalanffy description of blue crab growth. Helser and Kahn (1999) analyzed fishery-independent data from the Delaware Bay. Crabs were collected in a routine crab and juvenile finfish survey conducted by the Delaware Division of Fish and Wildlife. An August 1 recruitment to the survey date was assumed, based upon inspection of survey catches. Carapace width distributions from the survey were analyzed using MULTIFAN, a modal analysis package, to estimate von Bertalanffy parameters. Three significant age classes were detected in the data. The analysis generated three reasonable pairs of von Bertalanffy parameter estimates (Fig. 4). Recently Eggleston et al. (2004) developed a von Bertalanffy model for crabs in North Carolina based on an analysis of length frequency data. The parameter estimates so developed ($K=0.47$, $CW_{\infty}=216.9$, $t_0=-0.02$) indicated expected sizes at the onset of overwintering in the first, second and third years of 54.8, 115.6, 153.6 mm CW respectively (Table 1). Eggleston et al. also developed a molt process model for crabs in North Carolina based on a tag-recapture study. The estimates of growth per molt and intermolt period, together with assumed levels of variability were used in a molt process model to predict distributions of sizes at age. These were then used in a traditional von Bertalanffy model and yielded parameter estimates of $K=0.47$, $CW_{\infty}=216.9$, $t_0=-0.02$. The expected sizes at the onset of overwintering in the first, second and third years were 83.0, 163.9 and 202.5 mm CW respectively (Table 1).

The models differ substantially in predicted growth dynamics (Table 1 & Fig. 4). Overall the different models suggest an average value for K and CW_{∞} of 0.74 ± 0.22 and 206.92 ± 24.4 mm CW. For the assessment we chose to develop an estimate of von

Bertalanffy growth parameters using only empirically based studies – that is studies that used field data exclusively. These studies suggest an average value for K and CW of 0.82 ± 0.25 and 202.9 ± 24.1 mm CW. We note that there are only two strongly empirical set of parameter estimates for the Chesapeake Bay and that the average values are influenced heavily by the estimates from Delaware Bay (Helser and Kahn 1999).

Allometric relationships between carapace width and weight are needed for subsequent modeling exercises. Earlier studies estimated predictive relationships for weight based on a power function of carapace width (Newcombe et al. 1949). However, these data are based on crabs caught in the commercial fishery in the southern portion of the Chesapeake Bay. In order to utilize more recent data, we analyzed the relationship between total weight and carapace width for crabs collected in the field during CHESMAPP surveys (<http://www.fisheries.vims.edu/chesmmap/CmapTrawl.htm>), conducted between 2002-2004. These data suggested that there was no need to develop region specific weight-carapace width relationships. However, the data do suggest that the weight-size relationships for males and females differ. However, given that harvest data is not disaggregated over the entire period of record (see Section 5), we chose to use a single relationship, based on 3,423 observations, given by

$$W = 0.000842.CW^{2.422}$$

where weight is given in g and CW in mm.

2.3. Reproduction

2.3.1 Molt to maturity

Blue crabs reproduce sexually, and males and females are sexually dimorphic and exhibit different growth forms. Circumstantial evidence strongly suggests the presence of a terminal molt in female blue crab (Van Engel 1958, Abbe 1974). Limited physiological evidence suggests that the Y-organ does not degenerate as it does in other crabs that exhibit determinate growth, rather Smith and Chang (in press) speculate that in blue crab it is over production of MIH by the X-organ that enforces the terminal molt. As the Y-organ does not degenerate, female crabs maintain the physiological capacity to molt again under rare circumstances. Evidence for a terminal molt in males is less definitive than in females. There is some evidence for continued growth in males, particularly as most of the largest crabs collected are males. However, similarly to large females, large males form limb buds when they lose an appendage, and such males are often collected in the field suggesting that males molt infrequently at large sizes.

2.3.2. Age and size at maturity

Our limited ability to age blue crabs has precluded empirical development of maturity ogives for blue crab. However, recent evidence from attempts to develop large

scale aquaculture of blue crabs at the University of Maryland Marine Biotechnology Institute - Center of Marine Biotechnology, indicate that females can mature within their first year under ideal conditions. In the field, given the annual temperature cycle and typical megalopal settlement dates in August and September, it is unlikely that crabs could mature within their first year. It is more likely that they mature in the autumn of the following year when they are 12-18 months of age. Those that do not mature at this time, likely delay maturity for a following year, and mature when they are 24-30 months old. Hester et al. (1982) reviewed information on age at maturity in Chesapeake Bay. Their review suggested two production schedules: those females originally hatched in May reach maturity in 15 months and spawn at 24 –27 months of age, and those crabs originally hatched in August reach maturity in 21 months and spawn at 24 months. More recently, Hines et al. (2003) suggest that although females in different parts of the bay may mature at the same time, they differ in the timing of larval release (see Section 2.4.3).

Currently, there is insufficient knowledge to accurately describe maturity schedules for blue crabs, mainly due to difficulties in assigning ages. However, some research has quantified the probability of maturity as a function of crab size. Rothschild et al. (1991) produced maturity schedules based on 10-mm increments for crabs 60-180 mm CW. This size range presumably includes both age-1 and age-2 crabs. Rugolo et al. (1997) weighted size-based maturity schedules by estimates of the percentage of the population within each length class to generate the proportion mature by age. Their analysis resulted in estimates of 11% of age-1 crabs being mature, 92% of age-2 crabs being mature, and all crabs greater than age-2 being mature.

For this assessment we developed a size-dependent maturity ogive based on data from the winter dredge survey (see Section 3.2.4). Using these data, we calculated a female maturity ogive based on 5-mm size bins (N = 23 610 female blue crabs). A logistic function best described the maturity ogive, given as:

$$(5) \quad \Pr(\text{maturity}) = \frac{0.9994}{1 + \left(\frac{CW}{117.981} \right)^{-28.51}} \quad (r^2 = 0.99).$$

2.3.3. Mating and spawning periods and locations

Female blue crabs are only receptive during the period immediately following the post molt stage (see Section 2.3). Thus, all subsequent larval production results from sperm transferred by males during this single receptive period. Empirical evidence suggests intense competition among males for mating opportunities (Jivoff 1997, Kendall and Wolcott 1999). Males that mate frequently transfer less sperm which impacts the number of zoea released subsequently by mated females (Hines et al. 2003). Mating typically occurs from May – October (Hines et al. 2003). Mating pairs have been reported widely throughout the Chesapeake Bay system. Hines et al. (2003) found that

98% and 100% of mature females in the Rhode River and lower Bay held ejaculate stores, indicating a high level of mating success in the field.

Following mating, the behavior of inseminated females is believed to differ depending on their mating location (Hines et al. 2003, Aguilar et al. 2005). Females inseminated in the upper Bay in the summer probably overwinter and release larvae in the summer of the following year. In contrast, females inseminated in the lower Bay can release larvae in the same summer in which they were inseminated. Prior to hatching, ovigerous females migrate to the high salinity waters at the mouth of the Chesapeake Bay (Tankersley et al. 1998). Hatching occurs around nocturnal high tide and zoea are carried seaward on the ensuing ebb current.

2.3.4. Fecundity

Prager et al. (1990) conducted an extensive study of fecundity patterns in Chesapeake Bay blue crab. They found that fecundity level varied seasonally. Fecundity was low early in the season, peaked in mid season and declined at the end of the season (Prager et al. 1990). They concluded that fecundity was an increasing linear function of female carapace width, given by $\text{Fecundity (millions)} = -2.248 + 0.377 * \text{CW (cm)}$, $R^2 = 0.24$. The low R^2 value was partly due to a striking variability within a season, or may have arisen because of errors in estimation of carapace width. We note that fecundity estimates have not been updated since the Prager et al. study. These data were collected during a time of relatively high abundance. There is a potential that density-dependent changes in fecundity may have occurred in this species.

Hines et al. (2003) suggested that Chesapeake Bay females produce 1-3 broods (up to 9×10^6 eggs) per year and up to 6-7 broods (2.1×10^7 eggs) per lifetime. In contrast, Hines et al. (2003) suggested Florida crabs produced up to 6-7 broods (2.1×10^7 eggs) per year and up to 18 broods (5.4×10^7 eggs) per lifetime. In Florida, the last broods produced by lab-held female crabs were often infertile, indicating that females became sperm limited at the end of their lifetime. Experiments showed that male mating history affected female reproductive success, with females mated late in a sequence having only one third the brood hatching success of females mated early in the sequence.

2.4. Larvae

Larvae are transported out of the Chesapeake Bay and onto the coastal shelf (Roman and Boicourt 1999). Miller (2001b) used a size-based approach to estimate the mortality rate of this life history stage. Miller estimated that the probability that an individual survives the entire zoeal and megalopal period was 1.19×10^{-6} .

During their time at sea, zoea molt several times, before molting to the last larval stage, the megalopa, that reinvades the Chesapeake Bay. Time series of abundances of zoea and megalopae are available from the Chesapeake Bay Program's monthly

zooplankton monitoring program. These data have been analyzed by Lipcius and Stockhausen (2002).

2.5. Juveniles

The juvenile period is a critical life history stage for blue crabs. The importance of nursery habitats is widely reported (Etherington and Eggleston 2003, Etherington et al. 2003, Stockhausen and Lipcius 2003) – although recently the dominant paradigm of the critical role of seagrass as nursery habitat has been broadened to include a greater diversity of habitats. Van Montfrans et al. (pers. comm.) have documented predation of juvenile blue crab in seagrass beds by several fish including striped bass and red drum. Importantly, Etherington et al. found that mortality rates in seagrass habitats were equivalent to emigration rates, indicating that successful emigration to adult habitats is at least as critical a process as survival in the juvenile habitat.

2.6 Adults

A considerable amount is known about the feeding ecology (Mansour and Lipcius 1991) and the response to environmental parameters (Bell et al. 2003a, b) of adult blue crab. Research has also focused on assessing their role in structuring estuarine ecosystems (Hines et al. 1990). However, with regard to this stock assessment, the only feature of adult biology that is relevant is lifespan.

2.7 Lifespan

A substantial amount of attention has been focused on determining an appropriate longevity for blue crab. Owing to a lack of direct estimates of natural mortality rate (M ; see Section 2.9.), previous assessments for the Chesapeake Bay stock used longevity to estimate M . Thus, assumptions regarding lifespan directly affected the rate of natural mortality used in assessments and ultimately the biological reference points that were developed for the stock.

Ageing of most marine species is achieved by the analysis of hard parts – typically calcified structures that are retained throughout the life and that exhibit a regular structure that can be interpreted to estimate age. As a result of molting, the mode of growth exhibited by crabs and all other crustaceans, hard parts are not retained throughout an individual's life. Thus analysis of hard parts to estimate age is not possible in crabs. Recently, Ju et al. (1999) have perfected a spectrophotometric approach to quantify levels of lipofuscin in the nervous tissue of blue crab. Lipofuscin is one of a class of chemicals that accumulate in cells to protect the sub-cellular apparatus from damage from free radicals and other oxidizing agents. Thus, lipofuscin builds up in individual cells over time. Accordingly quantification of lipofuscin levels in cells has become the standard way of ageing cells. Traditionally, lipofuscin has been estimated

using fluorescence microscopy of stained sections of nervous tissues. Ju et al. (1999) developed a biochemical extraction approach that permits accurate quantification of lipofuscin levels while permitting higher sample throughput than the traditional approach. Ju et al. (2001) validated their ability to accurately age crabs up to 2 years of age by quantifying growth of crabs in aquaculture ponds. Subsequently, they applied the technique to crabs collected during a winter dredge survey conducted annually in Chesapeake Bay (Ju et al. 2003). The field application indicated the presence of four age modes in the Chesapeake Bay population in 1998 – 2000. Ju et al (2001, 2003) concluded that there were at least three ages (0, 1, and 2 yr old) crabs represented in the population. Their analyses also indicated the potential presence of a small number of age-3 crabs. Although this technique has not been fully validated, this work does provide evidence that life expectancy in crabs is at least 4 years under currently levels of exploitation.

Data from mark-recapture studies has also been interpreted to estimate blue crab life expectancy. In general, the growth process of blue crabs has presented challenges to the application of mark-recapture techniques to adult crabs. Common tags that attach to hard parts are usually lost or cause mortality during molting. Nonetheless, some tagging studies have been successful, particularly those focused on terminally-molted mature adult females.

Fischler (1965) employed mark-recapture techniques to estimate the abundance of blue crab in the Neuse River, and adjacent Pamlico Sound, NC. In 1958, Fischler tagged and released 2,949 female and 640 male crabs in the Neuse River and 1,350 female and 1314 male crabs in the adjacent Pamlico Sound. All crabs were tagged with an across the back carapace tag. Only crabs > 5" CW were tagged. Thus, all females that were tagged were mature, and thus would not molt again (see Section 2.4). Returns were obtained from commercial fishers, dealers and picking plants. Thirty-two percent of the tags in the Neuse River were returned. Of all returns reported, most occurred in 1958 (the year of release) and 1959. However, crabs were also recaptured in 1960 and 1961. Of the crabs recaptured in 1961, one was returned by a commercial fisher, the other was recovered from the crab in the ocean near Cape Lookout by a recreational fisher. If one assumes that the crabs were at least one year old when tagged, the recapture data indicate a life expectancy of four years, whereas if we assume the crabs were two years of age when tagged, a more likely estimate given their size at tagging, these data indicate a life expectancy of five years.

Recently, Lambert et al. (2005) conducted a mark-recapture study of female blue crab in the Chesapeake Bay as a part of a program to quantify the efficacy of a marine sanctuary for blue crab in the Chesapeake Bay. Lambert et al. (2005) used across the back strap tags to tag mature female crabs collected as a part of ongoing surveys in support of multispecies fisheries management. As with Fischler's (1965) study, Lambert et al. used the fact that females cease molting at maturity to overcome tag loss issues. Lambert et al. reported a very low survival rate of 2% for mature females. This would indicate that under current conditions, life expectancy of mature females in the exploited Chesapeake Bay stock is unlikely to be more than 3 years.

More controversially, John McConnaugha (Old Dominion University) in an unpublished report of a tagging study in the Chesapeake Bay included a tag return of eight years. This data point was used by Rugolo et al (1997) to estimate natural mortality and growth for blue crab.

In summary, neither lipofuscin-based nor mark-recapture approaches provide definitive evidence as to the life expectancy of blue crabs. However, the evidence suggests a lower limit of 4 years under light exploitation and an upper limit of perhaps 6 years. It is important to note that no evidence is available to determine the life expectancy of blue crab in an unfished condition, and fishing has probably truncated the age distribution of stocks (Hilborn and Walters 1992). However, given the uncertainty in longevity, we will also develop estimates of natural mortality from a range of approaches that do not require estimates of longevity.

2.8 Natural mortality

Although there is a substantial amount of literature about blue crab natural mortality as it relates to ecological questions (e.g., relative predation mortality as determined by tethering experiments), such data are collected at spatial and temporal scales too small to be incorporated into assessment models. We are aware of only two direct estimates of the natural mortality rate (M) for blue crab that are applicable at the scale of the stock. We note that these estimates, as well as those used in this assessment, are single, constant estimates for the stock as a whole. Consistent with past work on blue crab and the current level of knowledge about M for this species, we have assumed a fixed value of M (i.e., no size-, sex-, or age-dependence) in assessment models.

The first direct estimate is based on data from a tagging study of terminally-molted adult female crabs in Chesapeake Bay in 2002–2004 (Lambert et al. 2005). Annual survival estimates (S) are obtained from a Brownie model based on tag return data; the preliminary survival estimates are 0.055 for 2002 and 0.088 for 2003. We estimated M for adult female crabs by combining these estimates with the female-specific exploitation fractions for 2002 and 2003 (0.64, 0.55) using Baranov's catch equation (see Section 5.4.5). Given estimates of μ and Z [$Z = -\ln(S)$], the equation can be solved for F . Subtracting F from Z yields an estimate of M . The estimates, which only strictly apply to adult female crabs, are 0.94 for 2002 and 0.96 for 2003. Although the types of data used to generate the estimates of μ and Z differed from the ones we use here, this general approach was previously used in an assessment of the blue crab stock in Delaware Bay (Kahn and Helser 2005).

The second estimate was determined for the Delaware Bay stock by Helser and Kahn (1999). They used a different method to estimate M based on a regression of Z against fishing effort, where the y -intercept (zero effort) was considered an estimate of M (Ricker 1975). This approach resulted in an estimate for M of 0.84. Unfortunately, we cannot apply this method to the Chesapeake Bay stock because (1) the fishery effort data

do not provide sufficient contrast for such a regression, and (2) the reliability of fishery effort data collected under earlier reporting schemes is suspect.

Given limited direct information about M , we employed various indirect methods used in fisheries stock assessments to predict probable ranges of M for blue crab (Table 3). We felt that this was a prudent approach given the wide use of such methods and their intuitive and theoretical appeal. Additionally, the analyses provide a defensible range of estimates to include in assessment models. Each method yields estimates of M based on life history characteristics, such as growth parameters, lifespan, and age at maturity. We recognize that these empirical methods are often subject to large errors in predictions (Pascual and Iribarne 1993) and are only as good as the data upon which they were developed (Vetter 1988). Also, a notable problem with all of the methods employed here is that the relationships were developed primarily for fish, such that few of them include data on crustaceans specifically or invertebrates generally. Indeed, very few estimates of M exist for any crustaceans except shrimp and lobsters. In addition, many of the methods make use of parameter estimates for the von Bertalanffy growth model, and we recognize that the discontinuous nature of blue crab growth makes such a model only approximately correct (see Section 2.3). Following a review of the literature and currently available data, we compiled a range of values for various life history characteristics as inputs for these predictive methods (Table 3). As much as possible, we restricted the inputs to those values that were empirically-based. For example, we only used von Bertalanffy model parameter estimates (K , CW_{∞}) based on field data (those included in the empirically-based subset in Table 1). Below we discuss in some detail the various methods we employed to generate ranges of estimates for M .

The method used to estimate natural mortality for the Chesapeake Bay blue crab stock was perhaps the most controversial part of prior assessments (Rugolo et al. 1997, Miller and Houde 1999). The method used the standard assumption that M is inversely related to lifespan, and further assumed that 5% of the stock would survive to the life expectancy in the absence of fishing ($F = 0$). The method simplifies to $M \approx 3/\text{lifespan}$, where lifespan is measured in years. Rugolo et al. (1997) used a lifespan of eight years, resulting in an estimate for M of 0.375. Miller and Houde (1999) used the same method with a range of values for the lifespan (4-8 years), resulting in a range of estimates for M of 0.375 – 0.75. Hewitt and Hoenig (2005) discussed the history and mechanics of this method and showed that it has a poor theoretical foundation. Specifically, the arbitrary use of 5% survival to the estimated life expectancy is unnecessary, as the empirical method of Hoenig (1983) provided an analogous estimate and showed that approximately 1.5% of the stock survives to the life expectancy (resulting in $M \approx 4.22/\text{lifespan}$). Many crab biologists and fishermen have also expressed considerable skepticism about the estimate of an eight year lifespan for blue crab, raising further doubt about the veracity of the estimate of $M = 0.375$. Overall, the uncertainty about the lifespan of the species (section 2.7) makes the use of estimates of M based on lifespan inherently uncertain, and we consider it a positive step to have evaluated other methods for predicting M that do not rely on lifespan. Nonetheless, the range for M predicted by the method of Hoenig (1983) based on a lifespan of 4 to 6 years is consistent with the other indirect methods as well as the direct estimates (Figure 6).

Despite the uncertainty about the lifespan of blue crab, two methods that we used required estimates of the lifespan. We used a range of 4 to 6 years (see Section 2.7). The only method that relies solely on lifespan was the predictive regression model developed by Hoenig (1983), which regressed estimates of M on estimates of lifespan. This method was developed from an extensive dataset, including taxa as disparate as cetaceans and mollusks. Alverson and Carney (1975) presented another method that uses lifespan and the von Bertalanffy growth coefficient (K) to estimate M . They developed a theoretical model to predict the age at which a cohort would maximize its collective biomass (t_{mb}) if growth followed a von Bertalanffy model. They also developed a regression model based on 63 species of fish that allowed prediction of t_{mb} from lifespan. Since the model for predicting t_{mb} includes M , the model can be rearranged and solved for an estimate of M given estimates of lifespan and K . Taken together, these two methods provided estimates of M ranging from 0.30 to 1.35, but using an average value for K in Alverson and Carney produced a narrower combined range for M of 0.45 to 1.08 (Table 3; Figure 6A).

A second set of methods to estimate M in blue crab used what Charnov (1993) termed the "Beverton-Holt [life history] invariants" (B-H Invariants; first two methods in Table 3). Charnov reasoned that certain relationships among life history parameters were stable (invariant) within certain broad taxonomic boundaries because they arise from evolutionary trade-offs. Jensen (1996) derived the invariants directly from ecological theory, but the values he estimated for the parameters were somewhat different than those presented by Charnov (1993). Two of the three B-H Invariants treated by Charnov involve M . One relates M to the von Bertalanffy growth coefficient (K) (Charnov 1993; Jensen 1996), and the other relates M to the age at maturity (Charnov and Berrigan 1990, Charnov 1993, Jensen 1996). The values for K came from the empirically-based subset of growth estimates (Table 1) and the range for age at maturity (1-1.67 years) was taken from Van Engel (1958). The invariant based on K yielded estimates of M from 0.71 to 1.80, and the one based on age at maturity yielded estimates from 0.99 to 2.20 (Table 3; Figure 6A).

One of the most commonly used methods for predicting estimates of M for fish stock assessments are the regressions developed by Pauly (1980). His regressions relate M to estimates of K , L_{∞} , and W_{∞} from the von Bertalanffy growth model as well as an estimate of the average temperature a fish experiences in a year (T). We note that Charnov (1993) and Jensen (1996) showed that temperature added little to the predictive capabilities of the regressions, but we use them in their original form presented by Pauly. Maximum carapace widths (CW_{∞} ; treated as L_{∞}) were converted to maximum weights (W_{∞}) using the equation presented in Section 2.3. Predictions from the regressions based on carapace width yielded estimates of M from 0.91 to 1.72. The estimate of M based on averages of K and L_{∞} of 1.37, and predictions based on weight gave estimates of M from 0.82 to 1.52, with a value based on averages of K and W_{∞} of 1.22 (Table 3; Figure 6A).

Roff (1984) explored theoretically the evidence that life history parameters were consistently correlated among teleost fishes. He assumed these parameters had been adjusted over evolutionary time, due to trade-offs associated with growth, reproduction,

and survival, in order to maximize lifetime fecundity. Thus, the optimal age at maturity can be determined by its relation with K and M . Interestingly, the equation Roff (1984) developed is the same used by Alverson and Carney (1975) to solve for the age at which a cohort would maximize its biomass (t_{mb}), implying that the optimal age to mature is the same as the age at which a cohort would maximize its biomass. Roff (1984) showed that his model has reasonable predictive capability, approximately equal to Pauly's (1980) length-based approach. Roff and Alverson and Carney's methods gave estimates of M from 0.63 to 2.35, with a slightly more narrow range based on an average K of 0.84 to 1.94 (Table 3; Figure 6A).

Lastly, we considered a method that allowed prediction of M based on its inverse relation to body size (Lorenzen 1996). This is the only method we considered that could allow M to vary among individuals. However, we used the range of M produced by this method only in a way similar to the other approaches, and not as a way of introducing a more complex function for M . Through a quasi-theoretical approach, Peterson and Wroblewski (1984) predicted mortality due to predation in pelagic marine ecosystems to be a power function of body weight with an exponent of -0.25 . Their predictions matched observations well. Lorenzen (1996) took an empirical approach and related existing estimates of M to wet body weight for a large and diverse dataset for fishes of all sizes. He found that M in natural ecosystems (lakes, rivers and marine systems) could be explained by a significant power function of body weight. The relationship did not differ statistically among ecosystems or latitudinal zones. We used the joint relationship estimated for all natural ecosystems. We estimated crab weights for a range of carapace widths using the equation from Section 2.3. The method generated estimates of M from 0.52 to 1.19, with an estimate of 0.71 for crabs of average size (Table 3; Figure 6A). We note that for blue crab the results from this method were almost identical to the results from the more theoretical approach of Peterson and Wroblewski (1984).

Based on all of the predictive methods we used, the total range of possible estimates of M for blue crab was 0.30 to 2.35 (Figure 6A). In selecting a range of estimates to use in assessment models, we decided on 0.9 as the most likely value for M . This estimate is consistent with nearly all of the indirect predictions as well as the direct estimates for the Delaware Bay stock and adult females in the Chesapeake Bay. We also included boundary estimates of 0.6 and 1.2. We selected the value of 1.2 because it appeared to represent the parsimonious upper limit for M when considering various estimates of Z for the stock (knowing that F has been important over the period of record). Estimates of M greater than about 1.5 are probably unrealistic because (1) the 95% confidence intervals for preliminary estimates of Z based on tagging data (Lambert et al. 2005) include values as low as 1.9, and (2) fishery-dependent estimates of F (and thus Z) presented in Section 5.5 include values as low as 1.2 (Figure 6B). The value of 0.6 was selected because it generally represented the lower end of the ranges of M predicted by the indirect methods; only two methods gave estimates this low (Figure 6). Finally, although our review provides strong evidence that M for blue crab is substantially greater than 0.375, we included this estimate to provide comparisons with previous assessments. Such comparisons may be particularly necessary in management decisions.

In summary, the weight of evidence from empirical and life history analysis indicated that $M=0.9$ is the most likely value for the Chesapeake Bay blue crab stock. As a result, assessment model results and biological reference points are presented for $M=0.9$ first. However, we also present results for other levels of M (0.375, 0.6 and 1.2) to provide an indication of the sensitivity of the conclusions we draw to the value of M selected, and for comparison to findings of previous assessments.

3. Fishery-independent Data

3.1 Size-at-age Convention

Despite difficulties with ageing blue crabs, previous assessments have used size composition data from fishery independent surveys to develop estimates of abundance of blue crabs that are age 0, and age 1+ (Rugolo et al. 1997, Chesapeake Bay Stock Assessment Committee 2004). Blue crabs are assigned an age cohort based on current knowledge of growth and timing of recruitment. The correct size cut-offs for a single cohort are certainly influenced by such factors annual variations in growth rates, recruitment timing, and distribution. Considerable work has been undertaken to explore the consequences of alternative demarcations of size-at-age vectors (Chris Bonzek, VIMS pers. comm.). However, the size-based definitions of age-classes have not been rigorously and fully evaluated. In this assessment, we used the spatial, temporal and size thresholds (Table 4) that have been adopted by CBSAC in producing its annual status of the stock report (Chesapeake Bay Stock Assessment Committee 2004).

For all fishery independent surveys used in the assessment model, we examined the correlation between abundance of age-0 crabs and abundance of age 1+ crabs both within the same year, and with a one year lag. We assumed that a strong correlation between age 0 crabs in year i and age one-plus crabs in year $i+1$ indicated that the survey is effectively tracking cohorts. Data for all surveys were highly variable, so that significant relationships were difficult to discern from noise in the data.

3.2 Fishery-independent Survey Time Series

A strong point of Chesapeake Bay blue crab assessments is the abundant fishery-independent data that are available. For this assessment, data were analyzed from four fishery-independent surveys that differ in duration and geographical coverage (Table 5). The VIMS trawl survey, conducted for the past 49 years, is the longest-standing fishery-independent survey. It samples the southern portion of the Bay (Figure 7). The Calvert Cliffs pot survey is the second longest time series, covering 37 years. It is the most geographically restricted, sampling at one location near Calvert Cliffs, MD in the mid-Bay (Figure 7). The MD trawl survey, which is restricted to eastern shore sites and tributaries in Maryland waters of the Bay, has been conducted for the last 28 years (Figure 7). The winter dredge survey (WDS) has been conducted for 16 years and is the only Baywide survey (Figure 7). We analyzed the data from these four multi-year surveys.

These are not the only surveys available. For example, the US EPA Chesapeake Bay Program has been conducting zooplankton monitoring since 1985. This survey database includes records of blue crab megalopal abundance that were used by Lipcius and Stockhausen (2002) as recruitment index. Other fishery-independent studies considered, but not used because they were either too short in duration or too regional,

included a PEPCO survey conducted in the Potomac River in support of power plant operations, a trawl survey in the Rhode River and results from CHESFIMS and CHESMAPP – two multispecies surveys that collect, but do not target crabs.

3.2.1. VIMS Juvenile Fish and Blue Crab Trawl Survey

Since 1955, the Virginia Institute of Marine Sciences (VIMS) has conducted a trawl survey to monitor abundance trends in selected finfish and invertebrate species in the southern portion of Chesapeake Bay. Originally, the survey sampled only the York River, but it has expanded steadily. Currently, seven strata are recognized that cover an area from the mouth of the Bay to the VA/MD border, and up to the freshwater interfaces of the York, James and Rappahannock Rivers. Samples are collected monthly from about 60 stations within the strata. Both fixed and random station assignments have been employed. Blue crabs collected in the VIMS survey are enumerated, sexed and measured. Three age classes of crabs are recognized; age-0, age-1, and age-1+ (Table 4). However, the age-at-size matrix used in calculating the VIMS Survey Index is different from that used elsewhere (C. Bozek, pers. comm.). The size cut off for age-0 crab varies seasonally from 50 mm CW in September to 90 mm CW in the following July. Similar variation is included in the index for age-1+ (<125 mm CW in January, • 135 mm CW in July). An index for mature females was developed as well. The trawl used in the survey has changed over the survey time series. The most important changes were the addition of a tickler chain and a net liner in 1973 and 1979, respectively. We employed published calibration factors to account for changes in gear types in our analysis (Hata 1997).

Several indices were calculated from the VIMS survey database for both spring and fall periods. The longest time periods are available for indices calculated for the principal western shore tributaries (Rappahannock, York and James). Indices were calculated for age-0 crabs for the spring and autumn surveys (Fig 8). Abundance time series in all three tributaries showed similar patterns, although they differ in scale (Table 5). In general, surveys in the same season were more similar than surveys within the same system (Table 5). Spring survey abundances tended to be higher than those for the autumn. Peaks of abundance of age-0 crabs were apparent in the early 1970's, and mid-1980s (Fig. 8). Within the later period, three abundance peaks were present in the time series for each tributary (Fig. 8).

In previous assessments an aggregate VIMS trawl survey index was been employed (Table 6). This index comprised the age-0 crab abundance in the fall for all tributaries, the age-1+ abundance for the fall, and the mature female abundance for the summer. Only years for which the abundances for all three stages were available were included in the analyses. The 1956-2004 time-series mean CPUEs (crabs \cdot tow⁻¹) for the three classes were 5.14 ± 3.95 (age-0), 9.74 ± 7.31 (age-1+) and 2.15 ± 2.25 (mature females). The indices for each class were highly variable over the time series (Fig. 9A). There were periods of relatively high abundance in the early 1970's, and again throughout the 1980's. Indices for both the age-1+ and mature females have declined steadily since 1990 and have yet to recover to previous levels of abundance. Recent

levels of the age-0 index have also declined since the peak in the 1990, but have not remained consistently low, nor are they particularly low compared to the entire age-0 index time series.

These general patterns become clearer if the data are presented as standardized z-scores, where

$$z_i = \frac{U_i - \bar{U}}{SD_U}$$

where U_i is the catch per unit effort in a particular year (Fig. 9B). When viewed in this way, the consistent pattern of below average indices for age-1 and mature female crabs since 1990 is evident. Although recent low index scores are not time series minima, they the most protracted period of low index scores. In the two instances when age-1+ and mature female indices have reached such lows earlier in the time series, abundances had returned to previous levels within a six-year period. Thus the extended period of low abundance in the VIMS trawl survey indices are a cause for concern.

If a survey accurately reflects trends in population abundance, then abundances of age-0 individuals should be correlated with abundances of age-1 crabs in the subsequent year and with age-2 crabs two years later. To explore whether the VIMS survey data display this internal consistency we correlated aggregate age-0 abundance in year i against age-1+ abundance in year $i+1$ (Fig. 10). The correlation between age-0 crabs in year i and age-1+ crabs in year $i+1$ was slightly higher than for the unlagged case (Table 7), suggesting the index is successfully tracking cohorts within the population.

The VIMS trawl survey can be post-stratified to provide an index of mature female crabs on the spawning grounds (Lipcius and Stockhausen 2002). We analyzed abundance of the blue crab in the lower Chesapeake Bay spawning grounds over the 17-year interval ending in 2004. The approximate 80 % reduction in abundance of the spawning stock that occurred since 1992 has persisted through 2004 (Fig. 11). A more precise and detailed representation of the temporal sequence of the spawning stock decline is evident in the spawning stock trajectory over time (Fig. 11). Rather than a simple progressive reduction in the spawning stock, there appear to be three phases in the spawning stock trajectory, in which the spawning stock varies about some average abundance each phase before dropping to a lower average abundance in the subsequent phase. First, there was a phase of high abundance from 1989-1991 (Fig. 11A). Next, there was a phase of low abundance from 1992-1999 (Fig. 11B) subsequent to the initial population decline in 1992 (Lipcius and Stockhausen 2002). Finally, there was a third phase of even lower abundance from 2000-2004 (Fig 11B). The pattern in the spawning stock abundance as indexed by the VIMS is worthy of close monitoring.

3.2.2. Calvert Cliffs Pot Survey

A standardized pot survey of blue crabs in waters near Calvert Cliffs, MD has been conducted since 1968 (Abbe and Stagg 1996). The survey originally was initiated to determine the potential environmental impact of a nuclear power plant. Accordingly, sampling was specified at three stations: one adjacent to the power plant, and two at 4 - 8 km distances from the plant. Sampling is conducted using standard commercial crab pots, with 25 mm mesh, but without cull rings. Approximately 20 pots are fished at biweekly intervals at each site. Upon retrieval, crabs in each pot are measured (point to point), weighed and sexed. Data on catches from June - November are included in analyses. The smallest crab collected during the time series was 38 mm CW. However, more than 70% of the crabs collected were of legal size (127 mm CW) or greater. This indicates that the abundances of sub-legal crabs in this time series may be seriously biased. Thus, data for sub-legal crabs were not analyzed.

The CPUE data (crabs pot^{-1}) are tabulated (Table 8) and illustrated (Fig. 12A). Presumed ages were determined from carapace width measurements using the size at age criteria given above. The mean CPUE for age-1+ crabs for the total time series is 5.05 ± 2.65 crabs pot^{-1} . It is clear that the CPUE in 1981 was significantly higher than for the other years in the time series (Fig. 12B). CPUEs in 1984 and 1985 also were more than 1 SD above the mean.

Abbe and Stagg (1996) showed that the mean size of crabs caught in the Calvert Cliffs survey had declined between 1968 and 1995. The percentage of legal-sized males had declined from >50% of the catch in 1968 to <20% in 1995. The decline is evident even if calculated on absolute numbers. Correspondingly, the mean size of males had declined from >150 mm cw in 1968 to <125 mm in 1995. The decline in mean male size was well described by a linear regression with slope -0.663 mm yr^{-1} . The average size of female crabs captured in the survey declined from 1968-1995, and then increased through 2004 (Fig. 13). Since the late 1990's the average size of male crabs in the survey has in fact increased steadily, so that the average size in 2004, 135 mm CW, is in line with sizes of males reported early on in the survey time series. As significantly, the proportion of males greater than 127 mm CW (5") in the survey also abruptly increased in the late 1990's, paralleling changes in average size (Fig. 14). In the most recent years, approximately 60% of males caught were > 127 mm CW (Fig. 14). While this proportion is still below the historic values observed early in the time series, it is substantially higher than values observed in the mid 1990s. Changes in size of female crabs, have been less marked (Figs. 13 & 14).

The Calvert Cliffs pot survey only provides an index of age-1+ crabs. Accordingly, it is not possible to conduct lagged analyses to determine the degree to which the survey tracks abundance across year classes.

3.2.3. MD DNR Trawl Survey

Beginning in 1977, Maryland DNR instituted a trawl survey of Eastern Shore sites and tributaries, and the Patuxent River. The survey is conducted from May -

November. However, coverage is inconsistent temporally and spatially from year to year. In our analysis, we followed the approach of the CBSAC stock assessment and focused only on the Choptank River, Eastern Bay, and Pocomoke and Tangier Sounds. To reduce the impact of missing strata on the overall index, we again followed the CBSAC approach and calculated monthly average CPUEs (number \cdot tow⁻¹) for all regions combined, which were subsequently averaged to yield annual mean CPUE estimates for age-0, age-1 and age-2 male and female crabs (Table 9 and Fig. 15).

The MD trawl time series for age-0, age-1+ and mature female crabs were all highly variable (Table 10). Additionally, several breaks in the time series complicated the interpretation of the data. The time-series mean CPUEs (crabs \cdot tow⁻¹) for the three classes were 2.09 ± 2.44 (age-0), 8.95 ± 5.05 (age-1+) and 1.11 ± 0.74 (mature females). The time series included periods of above- and below-average abundances (Figure 15B). However, the periods differed among the presumed age classes and between the sexes. Age-0 crabs were generally at or below the time series mean from 1977 – 1988. Several years (1989 – 1993) of relatively high age-0 CPUE followed. Most recently, age-0 CPUE has again been at or below the time series mean. Patterns for age-1+ crabs were broadly similar, except this class and mature females show a period of relatively high abundance early in the time series from 1982 – 1987, and period of lower abundance from 1988 – 1993. However, we note that gaps in the time series between 1988- 1993 make this interpretation open to question.

There was not a strong relationship between age-0 CPUE and 1 year lagged age-1+ abundance at any lag examined (Table 10). The lagged response at one year (Fig. 16) was weaker than that in unlagged data (Table 10), indicating further work is required to refine size-at-age conventions. However, the committee felt that the survey still provided useful information on population trends.

3.2.4 Winter Dredge Survey

Crabs overwinter in bottom sediments of Chesapeake Bay and can be sampled effectively by a dredge. In 1989 a Baywide survey to estimate crab abundance that took advantage of this life history trait e survey was instituted. The winter dredge survey employs a stratified random design to estimate abundance by presumed age and sex. Except in 1989, the strata have been consistent during the fifteen years of the survey. The survey samples >1,200 randomly selected stations throughout the Bay from December - March. At each site a single 100 m tow of a 6' standard Virginia dredge (15 mm mesh bag) is made. All crabs in each tow are counted, measured and sexed. The gear samples crabs as small as 15 mm CW. CWs are converted to presumed ages, and abundance is expressed as crabs 1,000 m⁻² (Table 11).

As the only Baywide abundance index, the winter dredge survey provides a unique view of the crab population. Survey data have been used in several peer-reviewed papers to estimate the total abundance and mortality rate (Sharov et al. 2003); the efficacy of a marine sanctuary (Lipcius et al. 2003, Seitz et al. 2003) and most recently to quantify abundance and distribution (Jensen et al. in press a, Jensen and Miller in press b,

Jensen et al. in press c) of blue crabs. It is important to note that the precision of annual abundance estimates in this survey is unusually high (SD ~ 5-10%). Thus, the variability observed in the time series reflects true inter-annual variability in abundance rather than parameter uncertainty. Accordingly, the WDS is arguably the most valuable fishery-independent survey we possess to determine the population abundance of blue crabs in the Chesapeake Bay.

Evaluation of the raw survey CPUEs is informative (Table 11 and Fig. 17A). The estimated abundance of age-0 crabs has been more variable than for the other two presumed age classes (Fig. 17A). The time-series mean is 28.99 ± 15.22 crabs $1,000 \text{ m}^{-2}$. The age-0 time-series abundance was lowest in 1991 at 10.69 crabs $1,000 \text{ m}^{-2}$. The survey CPUE recovered in subsequent years, reaching a peak abundance of 52.16 crabs $1,000 \text{ m}^{-2}$ in 1996. However, since then, the survey CPUE of age-0 crabs has declined substantially and has remained approximately 1 SD below the survey mean since 1997 (Fig 17B). This suggests an ongoing pattern of low abundance of age-0 crabs. It should be noted that one deficiency of the WDS is that small, newly recruited crabs are not fully vulnerable to the dredge gear because of both size and their preference for shallow water habitat. Thus, it might be that the survey CPUE of age-0 crabs has declined because of a change in the timing of recruitment, or in a shift in their distribution.

The time-series mean CPUE for age-1+ crabs is 22.65 ± 10.79 crabs $1,000 \text{ m}^{-2}$. This age class of crabs was at highest abundance early in the time series. It declined to below average levels of CPUE in 1995 and has remained in that state ever since (Fig. 17B). The time-series mean CPUE for mature female crabs is 8.59 ± 4.07 crabs $1,000 \text{ m}^{-2}$. As with the pattern seen in the age-1 crabs, mature female CPUE has generally declined since the beginning of the survey. However, the pattern of mature female CPUE has been more variable than the other two life stages (Fig. 17B), although even here, abundances have generally been below the time series mean since 1996.

We analyzed the correlation structure among lagged abundance indices (Table 12). An analysis of the relationship between the abundance of age-0 crabs and the abundance of age-1+ crabs one year later indicates a weaker correlation in the lagged case than in the within year analysis (Table 12). The distribution of data suggested several outlier points (Fig 18), but overall, the fit was similar to that observed in VIMS trawl survey data. Neither the within year correlation, nor the lagged year correlation was significant, potentially indicating either the need to refine size-at-age conventions, or limited length of the time series.

The development of year- and boat-specific estimates of catchability coefficients (Volstad et al. 2000), has permitted expansion of survey CPUE's into absolute abundances. The abundance time series have been analyzed using two different statistical methods that have fundamentally different assumptions, but that have yielded similar conclusions. Sharov et al. (2003) used traditional design-based approaches to analyzing patterns of abundance. These calculations indicated that the total population of blue crab in Chesapeake Bay was approximately 1×10^9 in the early years of the time series (Fig. 19A). Crab abundance has declined relatively steadily and consistently since

1996 and currently stands at about 40% of the earlier figure. Jensen and Miller (in press b) used a geostatistical approach to analyze the same data. This approach indicated a higher initial abundance of perhaps 1.6×10^9 crabs, but similar current levels (Fig 19A). Regression of a fishery-dependent index of abundance from the dredge fishery against both fishery-independent estimates of abundance (Fig. 19B) suggests that the higher initial figures calculated by Jensen and Miller may be more appropriate.

The geostatistical analysis has the additional advantage that it is able to generate maps of the distribution of abundance (Jensen and Miller in press b). There is considerable interannual variability in the distribution of crabs in winter. In 1991, a year of high abundance overall, geostatistical analysis of winter distributions indicates that areas of highest abundance were in Tangier Sound and near the mouths of the Patuxent and Potomac Rivers (Fig. 20). In 2002, a year of low abundance, no particular areas of high density were apparent, although areas near the Honga River and in Tangier Sound were higher than background levels (Fig 21). Additionally it is possible to combine maps for all years and assess the average distribution of crabs (Fig. 22). This analysis indicates that the Honga River, Tangier Sound, and lower sections of the Rappahannock, York and James rivers are all important areas for the distribution of crabs in winter. Moreover, the analysis also indicates that different regions of the bay have likely been affected by the overall decline differently (Fig. 22B). For example, the central mainstem of the Bay has experienced a strong decline over the decade considered

3.2.5. Coherence in Combined Fishery-Independent Survey Time Series

We analyzed the fishery-independent time series to determine if there were correlations in abundance indices among the different time series. The primary tool for this evaluation was time-series analysis. This tool is restricted to relatively long time series (>30 observations) with extensive periods of overlap. Thus, we were only able to explore the potential cross-correlations for time periods in the VIMS trawl survey and the Calvert Cliffs pot survey indices.

Our first analysis explored the relationship between the abundance of age-1+ crabs in the VIMS trawl survey and the abundance of legal crabs in the Calvert Cliffs pot survey. The two surveys were significantly cross-correlated at 0 and 1 year lags. This result may reflect the expected lifespan of mature crabs in the exploited population in Chesapeake Bay. Higher order cross-correlations at 6-9 years also were significant but were difficult to interpret. We also calculated the cross-correlation between the abundance of age-0 crabs in the VIMS trawl survey and the abundance of legal crabs in the Calvert Cliffs survey. There was a significant correlation at a 1 yr lag, indicating that abundance of legal crabs as measured at Calvert Cliffs is significantly related to abundance of age-0 crabs measured in the VIMS trawl survey one year earlier. Higher order interactions at between 6-11 years also were present also, but not interpretable.

3.2.7. Combined indices

The control rules established by BBCAC and adopted by the jurisdictions to manage the blue crab fishery used a combined index calculated as the simple average of all standardized fishery independent surveys. For the period 1955-1968 the combined survey Z-score reflects just the VIMS trawl survey, for the 1968- 1976 period, the combined index reflects the VIMS and Calvert Cliffs surveys only and from 1977 – 1989 the index reflects the VIMS, Calvert Cliffs and the MD trawl survey. It is only from 1990 forward that the combined index reflects patterns in abundance from all four years.

The combined survey Z-score index has varied between ~ 1 SD below to about 2 SD above the mean over the course of the four surveys (Table 13 and Fig. 22). There are distinct periods of high and low abundance evident in the time series. For example, save for a limited number of good years, the combined Z-scores for all categories of crabs are below the time series means up until 1980. Time series lows for both age-0 and age-1+ were observed during this period. For age-0 crabs the time series low occurred in 1960 and was -1.299. For age-1+ crabs the time series low was also in 1960 (-1.33). However, these two estimates are based solely on the VIMS survey. If only combined Z-scores reflecting more than one survey are considered, the time series minima for age-0 and age-1+ crabs occurred in 1974 (-1.22) and 1968 (-1.13) respectively. Importantly, this later mark still serves as the overfished definition for the stock. Time series maxima were observed for age-0 crabs in 1990 (2.30) and age-1 crabs in 1971 (2.17). The time series minimum for mature female crabs occurred in 1961 (-0.92). For the complete time series, following an initial period of below average abundances, crab CPUES were at or above average between 1981 and ~ 1994. Following this period of average or above average abundances, the combined index declined for all three components of the population. It is important to note that the period of below average abundances observed since the mid 1990's is of similar extent, or longer than the period of below average abundances observed in the 1970's.

We analyzed the correlation structure among the different indices (Table 14). There was a significant relationship between the combined survey Z-score index of age-0 crabs in year i with the combined survey index of age-1+ crabs in year $i+1$ (Fig 23). A linear regression explained 21% of the variation in the data (Table 14). This indicates that the combined survey index appears to track the population adequately. The correlation of the lagged data was higher than the correlation between the age-0 and age-1+ crabs within the same year (Table 14).

4. Chesapeake Bay Fisheries

There is not a single fishery for blue crab in the Chesapeake Bay; rather there is a diversity of sectors, both recreational and commercial. Here we review the development and current status of the principal fisheries. Because management regulations vary among the jurisdictions, we present each jurisdiction separately. Information regarding the status and importance of recreational fisheries is so limited that we are unable to include them in the assessment.

Sette and Fiedler (1925) reported that the modern crabbing industry dates to 1873. Van Engel (1999) suggested that the growth of the commercial fishery at this time resulted from the successful development of methods for shedding and shipping crabs out of the region. The decline in landings in New York and New Jersey created a demand for crab meat that further encouraged development and expansion of the fishery. In response to the developing fishery, the States of Maryland and Virginia mandated oversight of the fisheries by their respective state agencies (Van Engel 1999). Virginia vested authority over the crab fishery in the Virginia Board of Fisheries in 1898. The situation in Maryland was more fluid until 1939 when the Maryland Commission of Fisheries was created.

During the early development of the crab fishery, crabs were harvested principally by dipnet, trotlines and scrapes (Van Engel 1999). Use of dredges to harvest overwintering crabs was limited to Virginia. Wire mesh crab pots were introduced in 1928 in Virginia, although they were not legalized in Maryland until 1941. Crab pots became the principal gear for hard crabs after World War II and remain so today.

The first regulations for the fisheries recognized gear, region and season differences. The establishment of a closed winter season occurred early in the history of the fishery. Local winter closures occurred in individual counties in Maryland as early as 1902 (Van Engel 1999), but it was not until 1930 in Maryland and 1932 in Virginia that the winter closure of the fishery was broadly enforced. Size limits on crabs were also established early on. The first successful implementation of size limits occurred in 1916, which Van Engel (1999) credited to a lack of relevant biological information and a focus on the oyster fishery prior to this date. Size limits on peeler crabs date to the 1920's. Perhaps the most important early regulation enacted were regulations to ban capture and possession of sponge crabs in 1916, although the duration of the ban has varied.

Data on the harvest from the fishery are available from as early as 1880. Many of these data were compiled by Sette and Fielder (1925) and Cronin (1987). The accuracy and potential biases in these early data are not fully understood. Since then several approaches to collecting data on the crab harvest, its characteristics and the effort expended to land the catch have been implemented, revised and modified. However, it is only in very recent years that attempts have been made to verify the level of compliance or accuracy of the various reporting systems. Accordingly, the reliability of indices of effort and harvest developed from the available time series remain an open question.

Rugolo et al. (1997), Van Engel (1999) and Fogarty and Miller (2004) all commented the need for caution in interpreting both the catch and effort time series (see Section 5).

4.1. Virginia

Commercial fishing for blue crab in the Commonwealth of Virginia is regulated by the Virginia Marine Resources Commission (VMRC). There are a variety of gear types that can be legally used to harvest crabs within the Commonwealth, but crab pots, peeler pots and dredges predominate. Crab pots can be fished in both the mainstem of Chesapeake Bay and in the tributaries. No person may place, set or fish more than a combined total of 500 hard crab pots in Virginia tidal waters. Peeler pots are fished on a more seasonal basis, and can be "baited" with live adult crabs. Crab dredges are restricted to the mainstem of the bay during winter months. The number of licenses in the dredge fishery is capped at 225. Season and time restrictions have been enacted, and differ among the different fishery sectors. Minimum size limits have been set for male hard crabs, immature female hard crabs, and soft and peeler crabs. No size limits exist for adult female hard crabs. Dark sponge (brown through black coloration) crabs must be returned to the water alive. There is a 17 barrel daily catch limit per vessel in effect for the crab dredge fishery. For a complete listing of regulations, see the VMRC website (<http://www.mrc.state.va.us/regindex.htm>).

A principal feature of blue crab management in Virginia has been the use of sanctuary areas in the lower Bay to protect females on the spawning grounds. The Virginia Blue Crab Spawning Sanctuary was established in 1941 and has expanded since that time, now covering 240,092 hectares from the Bay mouth to the VA/MD border (Lipcius and Stockhausen 2002, Lipcius et al. 2003). The sanctuary is closed to commercial harvest from 1 June to 15 September.

4.2. Maryland

The state of Maryland recognizes both commercial and recreational fishery sectors. Currently, in the commercial fishery Maryland prescribes seven legal methods for harvesting blue crab: scrapes and dredges, dipnets, trotlines, handlines, seines, bank traps and pots (<http://www.dnr.state.md.us/fisheries/regulations/regindex.html>). However, the fishery is dominated by the hard crab pot fishery, and the trotline fishery. There are numerous temporal and spatial regulations that limit when, where and how these gear types can be used. The hard crab pot fishery is seasonal as a result of both regulation and the life history of the crab. Like the pot fishery in Maryland, the trotline fishery is also seasonal. The trotline fishery is limited principally to the tributaries in Maryland in pot which pot fishing is banned. The amount of gear is regulated by the commercial license that each waterman holds. The limited crab catcher license permits individuals to land crabs only by employing up to 50 crab pots or deploy the prescribed amount of trotline. The "CB3" license permits waterman to employ up to 300 crab pots or the prescribed amount of trotline. Two additional licenses, the CB6 and CB9, increase the number of pots allowed to 600 and 900 respectively.

4.3 Potomac River Fisheries Commission (PRFC)

Under the *Maryland and Virginia Potomac River Compact of 1958* (Compact), fisheries in the Potomac River are managed by the Potomac River Fisheries Commission which is charged with the establishment and maintenance of a program to conserve and improve the fisheries resources in the river. The PRFC has established regulations limiting the number of pots that can be used in both the hard crab and peeler pot fisheries. Various size limits have also been established. See the PRFC website for a full listing of regulations (<http://www.prfc.state.va.us/index.htm>). The Potomac River is closed to crabbing from 1 December to 31 March each year..

5. Fishery-dependent Data

An accurate determination of the levels of total removals by the fisheries are central to the reliability of any assessment (National Research Council 1998). Accordingly, fisheries agencies have invested heavily in trying to get accurate estimates of the level of total removals (Fabrizio and Richards 1996). Typically, the most common concern is that of misreporting of the landings or missing entire sections of the landings. However, when dealing with fisheries that have a long history, it is often common to find changes over time in the way that removals have been reported. Adjusting for such reporting changes, when present, is an important consideration in developing accurate time series of removals (Fogarty and Miller 2004). The reporting systems for commercial crab landings in both Maryland and Virginia have undergone changes since 1929. Indeed, an area of controversy in prior blue crab assessments has been how such reporting changes were taken into account (Rugolo et al. 1997, Miller and Houde 1999). Thus, it is critical that we assess fully and where necessary adjust for the effects of these reporting changes. The principal reporting changes for the blue crab fisheries in VA and MD are as described below.

5.1. Reporting Changes

5.1.1. Virginia.

Through consultation with staff at VMRC, we identified three time periods that differ in how removals were estimated:

- 1956 - 1973. National Marine Fisheries Service was responsible for collecting data. Data were recorded by region (Chesapeake and Landings by State). Estimates are based on dealer reports by month subsequently aggregated by year. Landings are available by gear.
- 1973 – 1992. VMRC instituted a more detailed dealer-based reporting based system. A large, but haphazard sample of principal dealers was included in the survey. VMRC agents “picked” the principal dealers involved in the fishery. Each dealer provided a monthly report of the crabs sold to him by commercial fishers. The reports identify the gear, and region within the Chesapeake producing the landings. The reporting system was reviewed and critiqued in 1984-1985 by a group from Pennsylvania State University. Their report highlighted a lack of uniformity in data collection procedures and high variability in reported monthly landings among dealers. However, these deficiencies are balanced by the observation that the top 20 dealers handled 76.5% of the hard-shell catch, and 95% of the soft-shell trade. Moreover, reported landings of key dealers appeared stable over time. However, concerns over the reporting system lead Virginia to implement a mandatory reporting scheme in 1993. Knowledge of the impending change meant that

some of the principal dealers failed to report their removals data in 1992, and so estimates for this year are considered unreliable.

1993 – present. A mandatory, fisher-based reporting scheme was instituted in 1993. Fishers report daily catch and daily effort on standardized forms. Data reported is consistent with the Atlantic Coastal Cooperative Statistics Program standards. Data is checked for quality control on a routine and consistent basis and compliance and oversight procedures are in place to ensure accurate reporting. VMRC staff believe that the data are consistent from 1994/1995 onward, and are particularly reliable from 1997 onwards. However, VMRC staff remain skeptical over reported landings in 1993.

5.1.2. Maryland

Three reporting periods can be recognized in the data maintained by the Maryland Department of Natural Resources (MD DNR)

1929 – 1980. Prior to 1981, the MD DNR employed a self reporting system in which harvesters reported directly to the state. Data for the 1929-1980 period are available by month and by gear. Raw data are no longer available – all information is now held in computer files.

1981 – 1993. Concerns over the deficiencies in the self reporting system lead to a change in reporting in 1981. From 1981 – 1993, MD DNR employed a statistical survey to estimate removals. Commercial harvesters were stratified according to gear, participation and effort. A sample of volunteer harvesters was selected each month to provide detailed removals information to MDNR. Total removals were subsequently estimated by expanding data to total number of crabbers within license strata. Expansion assumes that people with a given license type that did not report, fished at a similar level to those that did report with that same license type.

1994 – present. In 1994, MD DNR implemented a mandatory reporting scheme. This scheme collects information on the removals by month, license type, gear, area fished, effort and market category. Concerns over continuing misreporting are addressed by the continued use of the expansion program used to calculate total removals for the 1981-1993 period.

5.2. Analytical Approach to Adjusting Reporting Changes

As a result of these numerous changes to the reporting schemes, we conducted analyses to assess the importance of each reporting change. Fogarty and Miller (2004) applied multivariate time series analysis to assess the impact of the 1981 reporting change in Maryland. Their approach was to assess the potential impacts of both underlying changes in crab abundance and the reporting change on estimated removals.

Fogarty and Miller's approach employed Box-Jenkins time series methodologies to (i) estimate a transfer function that quantified the relationship between crab abundance, indexed by a fishery-independent survey, and reported Maryland removals, (ii) estimate an step intervention model to account for the 1981 change, and (iii) estimate a combined model that incorporated the influence of both effects on reported removals. The overall model can be written as

$$c_t = \Theta + \beta(B)x_t + \omega(B)I_t + \frac{\theta_t(B)}{\phi_t(B)}z_t$$

where c is the catch, Θ is a constant, B is the backshift operator, β is an estimated parameter related to the importance of the fishery-independent survey abundance x , ω is an estimated parameter related to the impact of the intervention I which is a 0,1 variable whose value is 1 for years after the intervention and 0 in prior years, θ and ϕ are polynomial parameters related to a moving average and autoregressive time series model that is fit to the data so that the residuals from the model (z) are a pure white noise process. Model selection, i.e., whether transfer and intervention functions significantly improved model fit, was based on an Akaike's Information Criterion (AIC) statistic. In the original application, the Calvert Cliffs Pot survey was used (see Section 3.2.2 for details), and only the single 1981 intervention was modeled. The central conclusion was that both underlying abundance and interventions terms significantly improved model fit. Subsequently, an adjusted time series of removals was estimated using an intervention model applied to the removals time series. Other interventions were not considered in the original publication as insufficient data were available at the time to permit an evaluation of the 1994 reporting change.

The same approach outlined above can be generalized to consider multiple (j) surveys and multiple (k) interventions. The expanded model is

$$c_t = \Theta + \sum_{j=1}^r \beta_j(B)x_{j,t} + \sum_{k=1}^s \omega_k(B)I_{k,t} + \frac{\theta_t(B)}{\phi_t(B)}z_t$$

The approach to fitting is to first check for stationarity of the fishery independent time series. Where necessary the time series can be differenced or otherwise filtered to achieve stationarity. Subsequently, the cross-correlation between the fishery-independent, intervention and removal time series is estimated. Moving average and autoregressive terms of appropriate order are then added to the estimation process until the residuals are not significantly different from a pure white noise process. Resulting models, with and without the transfer and intervention functions, are compared using AIC and the model with the lowest AIC score is retained.

For analyses that follow, the Box-Jenkins approach of identification and estimation was implemented in SAS (SAS Corp, Cary, NC). The identification stage involved visual inspection of patterns in the autocorrelation (AF), inverse autocorrelation

(IAF) and partial autocorrelation (PAF) functions. Stationarity is a fundamental assumption of time series analysis. Stationarity of each time series was determined using the Augmented Phillips-Perron test for an autoregressive process (SAS Corporation, SAS v 8 Help Manual). We assessed whether transformation of the landings data was required prior to analysis using the %logtest macro (SAS Corporation, SAS v 8 Help Manual). Model fits were determined using AIC.

5.3. Reconstructed Commercial Landings

5.3.1. Virginia

Virginia commercial landings data were provided by Stephanie Iverson (VMRC, Newport News, VA). Raw monthly data, summarized by market category, gear type and water code were available for the period 1973 – 2003. Only annual totals were available for the period prior to 1973. The response variable in all analyses was the total annual landings of blue crab (metric tonnes, MT) for the period considered (Table 15, Fig. 24).

The average annual commercial landings in Virginia over the period 1945-2003 was $16,055 \pm 4,269$ MT ($=35.39 \times 10^6$ Lbs). Virginia commercial blue crab landings varied from a minimum of 7,791 MT (17.18×10^6 Lbs in 1958) – 28,559 MT (62.96×10^6 Lbs in 1966). Although, highly variable, there is no global trend evident in the time series (Fig 24). There is some indication of cycles in the time series of annual landings, with peaks in landings in 1950, 1966, 1984 and 1993. A rapid drop in landings, followed by an equally abrupt increase is apparent in 1992-93. When the first differenced (i.e., $L_{t+1} - L_t$) time series is examined, the landings anomalies for 1992-1993 become more evident (Fig. 25). As discussed above (see Section 5.1.1), VMRC has concerns regarding the validity of reported landings in these two years. Accordingly, to prevent undue influence of these two data points in analyses, all analyses were run with these two reported annual landings omitted. Subsequently, the most parsimonious model was used to estimate landings for these years.

A comparison of autoregressive models of the raw and log-transformed data indicated that transformation was necessary ($\mathcal{L}_{\text{raw}} = -201.863$; $\mathcal{L}_{\text{log}} = -202.740$). The autocorrelation function (AF) declined exponentially, and was significant for lags up to 2. The IAF and PAF functions declined smoothly. These patterns indicate an autoregressive function. There was no evidence in the IAF or the PAF that indicated significance of higher order lags, despite the appearance of cycles in the raw data.

We used the VIMS trawl age-1+ CPUE index to develop a transfer model. Because of changes in the spatial coverage of the VIMS survey, analysis was limited to the period 1968 – present. Inspection of the three life history stages in the VIMS trawl survey indicated that there was only a weak relationship between the age-0 CPUE and commercial landings (Fig. 26). There were stronger relationships between both the age-

1+ and mature female CPUEs and commercial landings (Fig. 26). We chose to use the age-1+ index as it was felt to be a more inclusive measure of crab abundance.

The simple transfer model with white noise errors fit the data well until the most recent years which it consistently over-estimated (Fig 27A). The AIC for the model was -18.38 (Table 16). Inspection of the AF, IAF and PAF suggested an autoregressive model of the first order (AR1). When autoregressive errors were included, the AIC for the model dropped substantially to -21.30 (Table 16). The model fit was improved for all but the most recent year (Fig 27B). An intervention term (separating pre 1993 and post 1993) added to the model was significant (Table 16) and improved model fit further (Fig 27C). The parameter estimate for the intervention was negative, indicating that the pre-1993 reporting system tended to over-estimate landings compared to the more recent, detailed and mandatory reporting system. The combined transfer function and intervention model did not yield any substantial improvement in the model AIC (Table 16). Parameter estimates indicated that the transfer function parameter was not significant. The intervention term remained significant in the final model.

The intervention model with autocorrelated errors was chosen as the basis for reconstructing the Virginia commercial landings. We used the parameter estimate for the intervention term to adjust the estimated landings prior to 1993. Note that we used the values for 1992 and 1993 predicted by the time series model in place of the raw estimates given the concern over the reliability of the original values for those years. The reconstructed time series is shown in Fig. 28. The average of the reconstructed 1945-2003 time series was $12,787 \pm 3,037$ MT. This represents a 20.3% decrease from the unadjusted values. The lowest adjusted landing was 5,923 MT (1958), and the highest adjusted landing was 22,714 MT (1966). The most recent landings (8,773 MT) are approaching the time series minimum, but have not exceeded the 1958 value. The period of decline apparent in the recent years in Figure 28 is of a similar magnitude to the declines in adjusted landings that occurred in the 1950's and in the mid 1960's, both in terms of duration and magnitude.

5.3.2. Maryland

The raw landings time series for Maryland are provided in Table 17 and shown in Fig 29. The abrupt increase in reported landings that occurred in 1981 is clear in this figure. Landings prior to 1981 averaged $11,130 \pm 2,703$ MT ($\sim 25 \times 10^6$ Lbs). In 1981, landings jumped substantially to 26,740 MT. After a period of relatively stable landings until the early 1990's landings subsequently have declined such that they are now equivalent to landings observed prior to 1981.

It is important to determine the contribution to the observed increase in landings throughout the 1990s of changes in underlying abundance during this period that are evident in the survey data (see Section 3.2.7) from changes that reflect the contribution of the reporting change. In the 1997 assessment, Rugolo et al. (1997) assumed that all of the change was the result of changes in underlying abundance. In their assessment Miller and Houde (1999) assumed the contrary, that all of the change resulted from the reporting

change. In a recent analysis of these data, Fogarty and Miller (2004) concluded that both a change in underlying abundance and the reporting change contributed to the change in abundance.

We have applied and extended the Fogarty and Miller (2004) approach to consider both the 1981 and the 1994 reporting changes in Maryland using the techniques described above. We again used the Calvert Cliffs fishery-independent survey as our index of abundance. We did this for two reasons. First, use of this time series allows the longest possible time series to be analyzed. Second, analysis of the only alternative, the Maryland trawl survey, indicates that this survey is not internally consistent, in that the abundance of year classes cannot be tracked in the population (see Section 3.2.3).

When Fogarty and Miller (2004) analyzed the Maryland commercial landings data they initially determined that the data for the period 1929 – 1980 were stationary and that the 1981-1994 data were similarly stationary. Since their analysis, nine data points have been added to the time series. These new data have caused the recent time series to become non-stationary. Stationarity is a principal assumption of time series analysis and breach of this assumption has serious consequences. For example, if we fit an intervention model to a decaying time series, the intervention term will be significant because of the pattern of decay, not because of some underlying shift. Accordingly, the analysis was conducted on a differenced time series (Fig. 30). Differencing removed the impact of the recent decline in landings. However, differencing still allows the potential impact of the 1981 and 1994 reporting changes to be examined. For example, first differenced estimates for these two years are both > 10 , whereas the remainder of the data fall between $-5 < d < 5$ (Fig. 30).

Time series analysis employed the differenced commercial landings and the differenced age-1+ abundance index in the Calvert Cliffs survey (see Section 3.2.2). A simple transfer function model did appear to capture the abrupt change in the differenced time series around 1981 (Fig 31A). However, the model fit was not compelling, either with white noise or autocorrelated error (Table 18). For both error structures considered, the mean term in the model was not significantly different from zero, as would be expected with a differenced time series (Table 18). Yet, the transfer function term was significant in both cases (Table 19), indicating that underlying abundance does play a role in predicting landings. There was evidence of the need to include an autoregressive error term. It is clear from inspection of the model fit, that a simple transfer function model fits the data well, except for the intervention years in 1981 and 1993 (Fig. 31A). A simple intervention model indicated a significant effect of the 1981 intervention, but not of the 1993 intervention. The simple intervention model did not fit the data well, as indicated by the AIC parameter (Table 18) or graphically (Fig. 31B). A combined transfer function – intervention model provided the best fit to the data (Table 18). Again, model parameters indicated that the 1981 intervention was significant while the 1993 change was not (Table 18). Inspection of the model fit indicates that the combined model does a better job of forecasting both the pattern and the magnitude of the interventions than either of the other two models (Fig. 31C).

We used the estimated intervention term parameter to reconstruct the Maryland commercial landings. The reconstructed Maryland landings are shown in Fig. 32. The mean of the adjusted annual 1945 – 2002 landings time series was $20,556 \pm 4842$ MT. This represents a 84.6% increase in the estimated average annual landings. The lowest adjusted landing was 9,180 MT (2000), and the highest adjusted landing was 26,743 MT (1981). The landings over the last five years have been below previous time series lows. The recent period of consistent declines apparent in Figure 32 is not evident in the earlier periods of the time series, suggesting the recent declines are a cause for concern.

5.3.3. Baywide

We present the combined, adjusted baywide landings for 1945-2002 in Table 19 and Figure 33. The average baywide annual landings for this period was $34,887 \pm 5,490$ MT. The reconstructed landings indicate that removals have been 16.7% higher than previously reported. The highest recorded baywide harvest was 47,719 MT in 1966. The lowest recorded baywide harvest was 21,539 MT that occurred in 2001. Landings in the last three years (2000 –2002) have all set record lows for the time series.

5.4 Estimates of Fishing Exploitation and Mortality

Estimates of exploitation can be generated based on the estimated number of crabs available at the beginning of the season and the total catch during the season.

5.4.1 Estimating Bay wide Catch in Numbers

Commercial harvest of hard crabs is generally reported to the Maryland Department of Natural Resources (MD DNR), Virginia Marine Resources Commission (VMRC) and the Potomac River Fisheries Commission (PRFC) in bushels. The harvest of peelers and soft crabs is reported in numbers. The three jurisdictions convert reported bushels of hard crabs into pounds using a standard conversion of 40 pounds per bushel. Although it has been shown that the 40- pound conversion for bushels to pounds of hard crabs is reasonably accurate (Stagg and Knotts 1991, Sharov and Volstad 2002), the average weight of individual crabs within a 40 pound bushel varies by year, sex and region. Therefore, annual, sex-specific estimates of mean carapace width from the Maryland and Virginia fishery independent trawl surveys are assumed to represent the mean size of crabs available to the fishery (Davis et al. 2001, Bonzek and Latour 2003). This mean size is then used to estimate the average weight for individual crabs using regression equations developed from Maryland trawl data pooled over years 1994 to 2004.

Males: $W = 21.45 - CW*0.927 + CW^2*0.014$ (df=16,372, $p < 0.0001$, $r^2 = 0.94$)
 Females: $W = 2.59 - CW*0.247 + CW^2*0.008$ (df=10,382, $p < 0.0001$, $r^2 = 0.95$)

The resulting weight estimate is then used to convert pounds to individuals.

In Maryland, crabs reported as 'mixed', which are bushels of either mixed gender crabs or less marketable males, are assumed to be best represented by the mean weight of females. In Virginia, crabs reported as 'unclassified' are apportioned into male and female harvest according to the ratio of reported female harvest to reported male harvest averaged from 1994 to 2003. This ratio is approximately 75% and is likely to slightly underestimate the harvest of females (Rob O'Reilly, VMRC, pers. comm., January 2005). No fishery-independent sampling is conducted in the Potomac River, so Maryland trawl estimates are applied to Potomac River crabs. The number of individual hard crabs harvested for each jurisdiction were summed and added to the number of peeler/soft crabs harvested to estimate the total number of crabs harvested from the Bay (Tables 20-22).

The recreational harvest, which has been estimated to be 5.3 to 8.5% of the total harvest (Ashford and Jones 2001, 2002) was not included in the estimate of Bay wide catch because the recreational harvest is not reported routinely, and has only been estimated via survey for the years 2001 and 2002. However, the effects of additional harvest on estimates of F are discussed below in the sensitivity analysis section.

5.4.2. Estimating abundance

The abundance of over wintering blue crabs in the Chesapeake Bay was estimated from the winter dredge survey (see Section 3.2.4 and Sharov et al. 2003). It is assumed that the estimated mean density of blue crabs in any year is representative of the entire distribution area for blue crabs in Chesapeake Bay. Absolute abundance is estimated by expanding crab density for every year to the total bay area, estimated at 9,812 km² by GIS. It should be noted that the dredge survey does not sample waters less than 1.5 m depth, that account for approximately 10% of the total bay area. These shoal waters were sampled with a limited number of stations in 1992 and 1993 using a small, modified dredge. Density estimates derived from these shallow water sites were not significantly different than those derived for the area deeper than 1.5 m (Rothschild et al. 1992).

5.4.3 Estimating Exploitation Fractions

Using estimates of total abundance developed from the winter dredge survey (1990-2003) and estimates of Bay-wide catch, the annual exploitation fraction for the Chesapeake Bay blue crab fishery can be calculated as:

$$\mu_i = \frac{C_i}{N_i}$$

where C_i is the total annual catch in numbers and N_i is the number of crabs available to the fishery (Sharov et al. 2003). This exploitation fraction is calculated for the entire

population, since we assume that all crabs sampled during the winter (N_t) will become vulnerable to the fishery at some point during the subsequent fishing season.

5.4.4. Estimating F

Once the exploitation fraction (μ) is calculated, it is possible to iteratively solve Baranov's catch equation for F given a presumed level of M. Specifically:

$$\mu = \frac{F}{F + M} (1 - e^{-(F+M)})$$

We estimated F assuming four levels of M: 0.375, 0.6, 0.9 and 1.2 (Table 23, Fig. 34). The differing levels of M produced varying estimates of F with the lowest values being associated with an M of 0.375 and the highest with an M of 1.2. As would be expected the trends in F for each value of M are identical with rates increasing steadily through the 1990's, peaking in 1999 and decreasing from the year 2000 through 2003.

5.4.5. Sensitivity of F Estimates to Error in Catch

Two major sources of uncertainty in the estimation of F using this technique are the accuracy of the reported harvest and the conversion of the reported harvest from pounds to numbers. Despite legal requirements to report harvest, there are fishermen in all jurisdictions who fail to report their harvest entirely, or who do not report their harvest accurately. In addition, the technique described above used to convert harvest from pounds to numbers does not account fully for the temporal and spatial variation in the size and weight of crabs accessible to the fishery (Tables 24-26). However, we consider it safe to assume that both sources of bias result in a consistent underestimation of the harvest and that the magnitude of this underestimation is not likely to vary significantly from year to year.

Estimates of F are highly sensitive to potential bias in catch estimates (Fig. 35). A 5% increase in harvest results in an increase in F of 8 to 10% depending on the level of M assumed. A 25% increase in harvest results in an increase in F of greater than 50%. However, for a given level of M, the overall trend in F from 1990 to 2003 is difficult to disrupt even if the magnitude of the bias in harvest varied among years (Fig. 36). The uncertainty in the absolute value of F, but the robustness of trend, lends credence to using the exploitation fraction as an empirical reference point in the management of the blue crab fishery.

5.4.6. Sex-specific Exploitation Fractions

The blue crab fishery is unique in that male and female crabs are marketed separately, so that sex-specific landings in all jurisdictions are recorded. In addition, abundance estimates can be calculated separately for the two sexes. This allows us to

track the exploitation fraction for males and females independently. The percentage of the baywide harvest that is soft and peeler has ranged between 4-9% between 1990 - 2003. The harvest of soft and peeler crabs is not reported separately for males and females, so we assumed a 50% sex ratio for this harvest. Ongoing fishery dependent monitoring by MD DNR supports this assumption.

In general, the exploitation fraction on female crabs is substantially higher than the exploitation fraction for males (Fig. 37). In addition, exploitation on female blue crabs has not declined at the same rate as for male blue crabs between the years 2000 and 2003. This is worthy of note, since trends in exploitation appear to be robust regardless of potential bias in estimates of catch. Some of the difference in exploitation between males and females could be explained by differing reporting rates. There is some evidence in Maryland that female harvest is more accurately reported than is the harvest of males. This difference would likely be due to fishermen being less apt to report more valuable male crabs sold as baskets directly to consumers than females sold to processors. However, simulation exercises indicate that the Bay wide male harvest would need to be increased by 40% in order for exploitation rates for the two sexes to be similar. It is highly unlikely that male harvest is underestimated to this degree. In addition, adding 40% to the male harvest, and then recalculating a combined (non sex-specific) exploitation fraction results in removals of greater than 100% in at least one year of the time series.

5.5 Harvest Prediction

It is also possible to use the abundance in the winter dredge survey to forecast the harvest in the next year. A simple linear regression model indicates a significant linear relationship between abundance in the winter dredge in year t and harvest in year t (Fig. 38). The regression explains 85% of the variation in the data and suggests that this may offer a mechanism by which managers can forecast upcoming harvests. Catch can be predicted from harvest using the simple relationships

$$C = qEN$$

where C is catch, q is catchability, E is fishing effort and N is abundance. We note that the strong linear relationship evident in Fig. 38 infers that the product qE has been a constant in the Chesapeake Bay fishery. In turn this implies that if effort has increased as is believed, catchability must have declined. This ability may be of great utility in planning.

5.6. Effort Data

Generally, the data currently available concerning effort expended in the commercial fisheries are not sufficiently reliable to be included in this assessment. However, effort data available in the winter dredge fishery in Virginia are considered

to be reliable (see Fig. 19A). We consider the approach taken by the CBSAC assessment (Rugolo et al. 1997) to characterize effort to be very coarse and not fully representative of the pattern of effort in the fisheries. The deficiencies in the effort data are partly due to the difficulty in assessing effort in passive gears like crab pots, and because of the lack of detailed monitoring systems. For example, there are no regulations on crab pot float marking that would allow verification of individual reports. Only recently have there been attempts to independently survey the number of pots deployed in order to verify watermen-based reporting. Maryland DNR has been funding work to conduct a detailed, spatially-explicit survey of effort in the blue crab pot fishery in Maryland. These survey results are too preliminary to be included in this assessment, but are a promising step forward. Currently data collection methods do require fishers to report both the number of pots fished, soak times and catches. However, the relationship between catch rates and effort has yet to be defined on a baywide basis. Perhaps the only clear pattern in effort is that participation in the soft crab fishery has increased over the last decade (Miller 2001d).

6. Assessment Model

Collie and Sissenwine (1983) developed an approach to estimating population size from survey-based measures of relative abundance. The technique divides a population into two stages: pre-recruited and fully-recruited. Time series of estimates of relative abundance of the pre-recruit and fully-recruited stages from the surveys are combined with estimates of catch to yield an expression which can be used to estimate the absolute abundance of the pre-recruit and fully-recruited stages in the population. Because age-structured models are notoriously difficult to fit to some crustacean stocks due to the difficulty in ageing crustaceans, their rapid growth and short life spans, an approach with only limited population structure is appropriate for these taxa. Thus, the Collie and Sissenwine approach has been widely used in assessments of crustacean fisheries including blue crab in Delaware Bay (Kahn and Helser 2005) and northern shrimp in Gulf of Maine (Cadrin et al. 1999).

However, the Collie-Sissenwine model (1983) is limited in that it utilizes a single, but complete series of relative abundance information data for the fully-recruited and pre-recruit stages. A common practice is to partition a single survey time series of relative abundances into a pre-recruit and a fully-recruited indices using size or age composition information (Cadrin et al. 1999). It is important to note, that for the Collie-Sissenwine model to be accurate, the partitioning of the single survey must itself be accurate. Incorrect inferences regarding membership of the pre-recruit and fully-recruited stages will tend to smooth trends in population dynamics, thereby under-estimating vital rates

In many fisheries, no single fishery-independent survey fully indexes population abundance. Often surveys are of limited geographic or temporal coverage. Yet, for many of these same fisheries, multiple fishery-independent surveys are available. However, taken together, these surveys may integrate a sufficient spatial or temporal domain to adequately index the entire population. Thus, expansion of the original Collie-Sissenwine method to permit multiple surveys would offer substantial advantages. However, no analytical approach yet exists to include additional survey time series.

There are two obvious, ad hoc, ways in which to combine multiple survey time series into the catch-survey estimation. First, if the data from the two independent surveys are in the same units and of the same length, the data could simply be combined as the average over the two data sets. This approach, however, may introduce additional bias or uncertainty in estimated parameters as the method assumes that the coefficients of variation in the two independent surveys are equal. A second approach would be to carry out two independent assessment models using each independent data set and then average the outputs from the two assessment models. Again, this may also introduce the same bias for the same reasons as averaging the data.

A better approach than these ad hoc methods is to fit a single population dynamics model to the data, where the relative contribution of each dataset to the objective function is properly weighted by the inverse of the observation errors. Here, we re-parameterize

the Collie-Sissenwine model to a form in which process errors are treated as parameters to be estimated from the data. We term this approach the Catch-Multiple Survey (CMS) model. We then apply the CMS model to the Chesapeake Bay blue crab fishery. The model incorporates much of the fishery-independent data available for the stock, and can generate estimated time series of abundances, and exploitation fractions.

6.1 Methods

6.1.1 Summary of the Collie-Sissenwine Model

The original catch-survey model developed by Collie and Sissenwine (1983) can be written as follows:

$$N_{t+1} = (N_t + R_t - C_t) \cdot e^{-M} \cdot e^{\epsilon_t}, \quad \text{Eq. 1}$$

assuming that the surveys are conducted just prior to the start of each fishing season t . N_t is the fully-recruited abundance in season t , R_t is the number of pre-recruits entering the fishing season, and C_t is the total number removed by the fishery in each fishing season. The instantaneous natural mortality rate (M) is assumed constant, and the process errors (ϵ_t) are assumed log-normal. Equation 1 is easily modified to account for time lags between the fishing season and the time of year that the survey was conducted. For example, if the survey is conducted six months prior to opening of the fishing season then equation 1 is re-expressed as:

$$N_{t+1} = \left[(N_t + R_t) e^{-\frac{M}{2}} - C_t \right] \cdot e^{\frac{M}{2}} \cdot e^{\epsilon_t}$$

Survey abundance indices are assumed to be proportional to N_t and R_t and to be measured with a lognormal error so that:

$$n_t = q_n N_t \cdot e^{v_t} \quad \text{Eq. 2}$$

and

$$r_t = q_r N_t \cdot e^{\delta_t} \quad \text{Eq. 3}$$

The n_t and r_t terms are the indices for fully-recruited and pre-recruit abundances, respectively. The q_n and q_r terms represent the catchability coefficient for each respective series and v_t and δ_t are measurement errors.

Estimated model parameters consist of q_n and time series of N_t and R_t . It is not possible to estimate both q_n and q_r because the relative difference in catchability of fully-

recruited to pre-recruits must be specified a priori. This is usually done by specifying a ratio, which may or may not vary with respect to fishing season, of pre-recruit catchability to fully-recruited catchability such that $q_r = q_n \lambda_t$. The estimation process proceeds by using a non-linear search routine to estimate N_t , R_t and q_n by minimizing the following residuals:

$$\nu_t = \log(n_t) - \log(q_n N_t)$$

Eq. 4

$$\delta_t = \log(r_t) - \log(q_r R_t) = \log(r_t) - \log(q_n \lambda_t R_t)$$

and the process error residuals are calculated by substituting equations 2 and 3 into equation 1 to yield:

$$n_{t+1} = \left(n_t + \frac{r_t}{\lambda_t} - q_n C_t \right) \cdot e^{-M} \cdot e^{\varepsilon_t}$$

Eq. 5

Solving Eq 5 for ε_t yields

$$\varepsilon_t = \frac{\log(n_{t+1})}{\log \left[\left(n_t + \frac{r_t}{\lambda_t} - q_n C_t \right) \cdot e^{-M} \right]}$$

Eq. 6

The overall objective function that is minimized is the weighted sum of sum of the residual errors given in equations 4 and 6.

6.1.2. Summary of the Catch-Multiple-Survey (CMS) Model

Our main objective is to be able to incorporate multiple survey datasets into the catch-survey approach. The approach is not limited to the model structure defined in equation 1: other more complicated age-structured or delay difference models can be substituted for equation 1.

The model structure remains the same as that defined in equation 1, but rather than estimate the actual number of fully-recruited individuals (N_t 's), we update N_{t+1} based on estimates of R_t and ε_t and the initial number of fully-recruited individuals (N_1). That is, the estimated parameter set changes from $\Theta_{CS} = (N_{t=1:n}, R_{t=1:n}, q_n)$ to $\Theta_{CMS} = (N_1, R_{t=1:n}, \varepsilon_{t=2:n}, q_n)$. The number of parameters estimated remains unchanged, and the only restriction on how many survey time series can be estimated from the population model is the length of the catch time series. This alternative set of estimated parameters provides an opportunity to integrate over the nuisance parameter q_n (Walters and Ludwig 1994) and to simultaneously fit the model to multiple sets of survey data.

6.1.3 Dealing With Nuisance Parameters

Given initial estimates of N_1 and $R_{t=1:n}$ and the observed removals (C_t), the number of fully-recruited individuals in each fishing season is first calculated using equation 1. It is not necessary to specify any values other than 0 for the ϵ vector initially. Thus, the first approach might be to set $\epsilon=0$, equivalent to a measurement error only model. However, it is necessary to estimate q_n in order to calculate q_r , given the relative selectivity λ or ratio of pre-recruit catchability to fully-recruited catchability. The catchability coefficient q_n is referred to as a nuisance parameter in that it scales the relative abundance observations to the same units as the estimated number of recruits N_t . In this way, q_n can be thought of as the slope of the regression line between n_t and N_t . In essence, only the trend information in n_t is being used to infer changes in N_t . Therefore, it is not necessary to actually estimate q_n and the maximum likelihood estimate of q_n can be substituted into equations 2 and 3.

For any given values of N_1 and sequence of R_t the maximum likelihood estimate for q_n is given by the slope of the regression assuming a zero intercept

$$q_n = \frac{\sum_{t=1}^n n_t N_t}{\sum_{t=1}^n N_t^2} \quad \text{Eq. 7}$$

Alternatively, and usually more computationally efficient, we can use the mean of the residuals between n_t and N_t , i.e.,

$$q_n = \exp\left[\frac{1}{n} \sum_{t=1}^n (\log(n_t) - \log(N_t))\right]. \quad \text{Eq. 8}$$

By substituting this estimate of q_n from equation 7 or 8 into equation 2 and its corresponding ratio in equation 3, it is no longer necessary to treat q_n as a parameter to be estimated.

To contrast this approach with the original Collie-Sissenwine catch-survey model, we now treat the process errors (ϵ) as estimated parameters and thus it is possible to use the relationships defined in equations 2 and 3 to compare multiple survey time series. Accordingly, we calculate the residuals for each survey time series j as:

$$\nu_{j,t} = \log(n_{j,t}) - \log(q_{n,j} N_{j,t})$$

Eq. 9

$$\delta_{j,t} = \log(r_{j,t}) - \log(q_{a,j} \lambda_{j,t} R_{j,t})$$

We employ a simplifying step in the calculation of the process error terms. The process error term is the difference between the observed n_{t+1} index and the predicted index based on estimates of N_t and R_t . Substituting these values into equation 1 yields:

$$\varepsilon_t = \log(n_{t+1}) - \log[(N_t + R_t - C_t) \cdot e^{-M}]$$

Further substitution of n_{t+1}/q_n for N_{t+1} results in the residual process error:

$$\varepsilon_{j,t} = \log\left(\frac{n_{j,t+1}}{q_n}\right) - \log[(N_t + R_t - C_t) \cdot e^{-M}]$$

now indexed over each survey time series j .

6.2 Implementation

The model described above was implemented in AD Model Builder (Otter Research, Sydney, BC). The code is provided in Appendix I. We used the reconstructed time series of annual commercial catches in Virginia and Maryland (see Section 5.3.3) in the model as catch time series (C_t). The model also simultaneously used three fishery-independent surveys to drive the predicted population dynamics: the VIMS trawl survey, the MD trawl survey and the winter dredge survey. The Calvert Cliffs survey was not included because it does not provide estimates of pre-recruit and fully-recruited abundance. The model estimated a time series of predicted pre-recruit abundances (R_t), and a time series of calculated fully recruited abundance (N_t). These calculations relied on a time series of observed catch, and an estimate of the initial abundance of fully recruited crabs (N_1). The initial value of N_1 was scaled to the maximum observed catch during the entire time series. Within the model, annual exploitation fractions, μ_t were estimated as the ratio of the observed catch and the predicted fully-recruited abundance. Estimated exploitation fractions were compared to the direct estimates calculated from the annual winter dredge survey abundance estimates and baywide commercial landings (see Section 5.4.)

6.2.1. Base Model Run

The model was run with a range of parameter estimates and the degree of fit to the observed data was assessed (Appendix II). Based on these simulations, a parameter set for the base run was selected. The degree of fit was calculated according to the likelihood value, L and the sum of squares of the observed minus the predicted exploitation fractions. We considered the base run of the model as that with $M = 0.9$ and ratios of survey selectivity (λ) for each survey set to 0.4 for both of the trawl surveys and 0.5 for the winter dredge survey. The ratios of observation to process errors were set at 0.65 for both trawl surveys and 0.4 for the winter dredge survey.

6.2.2. Additional runs

A full accounting of all model runs completed during the assessment is provided in Appendix II. An initial suite of model runs were conducted that varied the ratios of observation to process error were conducted to explore the response of model predictions to variations in these parameters. The final values used in the base run were based on both model fit and on the consensus that the winter dredge survey is more precise than both of the trawl surveys

Once values for the ratio of observation to process error were established, we conducted further model runs with different values of M based on the review in Section 2.8. Initially model runs were conducted with $M=0.375$ to provide a comparison of model results with using the natural mortality rate used in past assessments with the results of those assessments. In addition, we conducted model runs with $M=0.6$ and 1.2 which were felt to represent limits to the range of M credible.

6.3 Results

The results from the base run of the model provided an adequate fit to the observed survey time series (Table 27, Fig. 39). The abundance of fully-recruited crabs predicted by the model agreed well with the observed trends in the fishery-independent indices of abundance of fully-recruited crabs (Fig. 39). For example, the decline in abundance of fully-recruited crabs evident in all three surveys since 1990 is evident in the predicted abundance time series. Reflecting the trends in the trawl survey indices, the predicted abundances of fully-recruited crabs indicate a period of low abundance in the mid to late 1970's, followed by a population recovery by the early 1980s. Subsequently, there is a period of relative stability until 1990, from which point the time series exhibits a steady decline. The predicted time series shows the quick rebound following the decline in the 1970's, whereas the abundance has remained consistently low since 1993. Although, the predicted abundances of fully-recruited crabs have generally increased for the last four years, compelling signs of population recovery are yet to be apparent in the predictions. The time series of pre-recruit abundance predicted by the model is highly variable, and does not fully capture the variability evident in the fishery-independent surveys (Fig. 39). However, it does appear that recent recruitments have been lower than those observed 1980 – 1990.

The base model also produced reasonable estimates of exploitation fractions, although two of the 35 estimated μ 's were greater than 1 (in 1975 and 1998, Fig. 39). The model reliably estimates μ in the early 1990's and in the early 2000's, but does not fully capture the high μ 's observed in the mid 1990s. Finally, we examined the relationship between the abundance of fully-recruited crabs predicted by the model, and the predicted exploitation fraction. There was a strong negative relationship between predicted μ and predicted abundance of fully recruited crabs (Fig. 39). This pattern indicates that the blue crab fisheries act in a depensatory fashion – that is as abundance declines, the exploitation on the crabs remaining increases. Such a pattern in exploitation is not conducive to sustainability.

The overall model fit and estimated time series of abundances and exploitation fractions were sensitive to the value of M (Table 27). Model runs were conducted with what were believed to be credible limits to M , specifically $M=0.6$ and $M=1.2$. When $M=0.6$, the degree to which the model explained the observed data declined compared to the base run (Table 27, Fig. 40). The estimated values of survey catchabilities and the predicted exploitation fractions for this level of M were higher than for the base case (Fig. 40). Indeed, estimates of μ exceeded 1 in 10 of the 35 years, indicating unreliable predictions. However, the patterns in the predicted abundance of fully-recruited and pre-recruited crabs were broadly similar. The relationship between predicted μ and the predicted abundance of fully recruited crabs was still negative, indicative of higher exploitation fractions at lower population abundances. The model results for a run for the higher credible limit of M ($=1.2$) model did not change substantially over the results when $M=0.9$. The likelihood and the μ deviation statistic were relatively unchanged (Table 27). The general patterns in the abundances of fully-recruited and pre-recruit crabs were unchanged (Fig. 41), although estimated survey catchabilities were lower than in the base scenarios (Table 27). None of the predicted μ 's were > 1 (Fig. 41, and they were considerably lower than those estimated from the direct enumeration approach (Fig. 41). Nevertheless, the negative relationship between predicted μ and predicted abundance of fully recruited crabs was still evident.

When $M=0.375$, the abundance of fully-recruited crabs predicted by the model agreed well with the observed trends in the fishery-independent indices (Fig. 42). However, when $M=0.375$, the model predicted unrealistic exploitation fractions ($\mu > 1$) for much of the time series (Table 27). The inverse relationship between μ and the abundance of fully-recruited crabs was maintained.

The degree of model fit, as measured by the both the overall process error and the sum of squared residuals of the estimated minus the predicted values of μ , varied as a function of the level of M used in a simulation (Fig 43). The minimum process error occurred with an M greater than 1.2, whereas the minimum in the sum of squared residuals in exploitation fraction occurred with an M of approximately 1.05

6.4 Implications

The CMS model was able to reproduce the dynamics of fully-recruited and pre-recruit crabs observed in Chesapeake Bay. Model results suggest that current abundances of fully –recruited crabs are two-three times lower than those observed in earlier decades. All models indicate that the abundance of fully-recruited crabs declined throughout the 1990's. Equally, all models indicate that the abundances of fully-recruited crabs have increased in the most recent years. However, these increases are modest and are not indicative of a widespread recovery. The model with the most likely value of $M (=0.9)$ indicated that recent rates of exploitation increased rapidly in the early 1990's, reaching peak levels in 1996. Thereafter, exploitation fractions declined such that the model now indicates that 45-50% of the fully-recruited population is harvested each year. However, that the CMS model was not able to fully capture the observed time series of exploitation fractions. The discrepancies may reflect stage-specific variation in the actual value of natural mortality rates, inaccuracies in the partitioning survey catches into pre-recruit and fully-recruited stages and errors in the ratios of observation to process error.

A significant and consistent finding of the assessment model is the inverse relationship between predicted rates of exploitation and abundance (Figs. 39-42). In all models considered, the exploitation rate predicted by the model as required to yield the observed harvests increased as the abundance of fully-recruited crabs declined. This compensatory pattern of exploitation is likely to be a major challenge in managing for a sustainable fishery.

7. Biological Reference Points

The precautionary approach which guides fisheries' management under the Magnuson-Stevens Fishery Conservation and Management Act requires that only patterns of fishing that are compatible with sustainable exploitation be permitted. Using this principal, the BBCACs technical subcommittee (TSC) recommended the current control rule system for managing the bluecrab fishery (Fig. 1). Thresholds and a target were established to ensure a sustainable blue crab fishery in the Chesapeake Bay. In defining these thresholds and target, the TSC chose to compare the predictions of tactical fishery yield models with empirical evidence. The original exploitation reference points were developed from a traditional yield per recruit modeling approach. Additionally, an overfished reference point was recommended on purely empirical grounds. The reference point was selected as the abundance of age-1+ crabs in 1968, the lowest abundance on record as estimated by an aggregate fishery-independent survey index.

Here we present two sets of reference points for the blue crab stock: (i) the original BBCAC reference points but with revised estimates of M , and (ii) a new set of reference points based on an individual based model of spawning potential per recruit. The new reference points were developed because of concerns regarding the original reference points. The original BBCAC overfished reference point was calculated as the equally weighted average of the Virginia Trawl, Maryland Trawl, Calvert Cliffs Pot and winter dredge surveys. The equal weighting used to calculate the average implies that each survey is provided an equally reliable index of overall abundance. Since the BBCAC reference points were developed, concern over the weighting scheme has been expressed. Additionally, and perhaps more significantly, the overfishing reference point was developed using an age-based approach which determines the expected yield per recruit give rates of natural mortality, fishing mortality and the vector of age-specific vulnerabilities. This approach assumes that age, rather than size, is the principal determinant of vulnerability to the fishery and growth can be represented as a continuous function. Because these assumptions are likely to be violated to an important degree for the blue crab stock, we reconsidered the foundation of the BBCAC reference points. Accordingly, we present alternative reference points based on exploitation fractions developed from a size-dependent individual-based spawning potential per recruit model.

7.1 BBCAC Reference Points

In 2001, the BBCAC TSC made the following recommendations for the Chesapeake Bay blue crab fishery:

- The fisheries management agencies (MDNR, PRFC and VMRC) should adopt an overfishing threshold that preserves a minimum of 10 % of the spawning potential of an unfished stock.
- The fisheries management agencies should adopt an overfished threshold equivalent to the lowest stock estimate that can be shown to have

subsequently sustained a fishery, as recorded by fisheries-independent surveys (which occurred in 1968).

- The fisheries management agencies should adopt a target fishing mortality rate that preserves a minimum of 20 % of the unfished spawning potential.

The biomass reference point is based on the average abundance of age-1+ crabs in the four principal fishery-independent surveys (see Section 3.2). Survey Z-scores from each survey are averaged to yield a single abundance measure for age-1+ crabs. The threshold reference point was chosen as the 1968 abundance. The fishing mortality rate reference point was developed from a traditional Beverton-Holt style yield per recruit analysis. Since 2001, the annual stock status report issued by the NOAA Chesapeake Bay Office has used these reference points to assess the status of the stock.

The history of the blue crab stock relative to the BBCAC control rule is shown in Figure 44. For 2003, the stock was not overfished (i.e., it was above the biomass threshold), nor was the stock experiencing overfishing (i.e., it was below the fishing mortality rate threshold). However, the stock was experiencing rates of fishing higher than the target exploitation rate, and had experienced overfishing during the 1998-2002 period. The highest fishing mortality rate experienced during this period ($F=2.81$ in 1999) was more than double the overfishing threshold. Since reaching a peak in 1999, the fishing mortality rate declined rather steadily, though the stock abundance has not yet recovered to levels observed in the early 1990's when fishing mortality rates were equivalent to those observed in 2003. Therefore, we conclude that continued concern over the level of abundance in the stock is warranted.

7.2 Individual-based Per Recruit Reference Points

The reference points developed above use an estimate of natural mortality rate to convert either empirical or predicted values of exploitation fraction, μ , to estimates of F that are then compared against per recruit reference points. Consequently, errors in M are translated to errors in F . Equally, if estimates of M are changed, estimates of the past fishing history are changed. Thus, it would seem desirable to develop reference points based directly on μ , thereby avoiding making any inferences regarding M . Moreover, existing per recruit reference points are inherently age-based, whereas the pattern of exploitation in the blue crab fisheries is likely to be size-dependent than age dependent. Thus, a new approach to estimating per recruit reference points is needed.

7.2.1. Modeling approach

Recently, Bunnell and Miller (in press) developed an individual-based approach to spawning potential per recruit analysis (IBM SPR). The IBM SPR model has several advantages over conventional models: *i*) known individual variation in size and growth rate could be incorporated, *ii*) the underlying discontinuous growth pattern could be

simulated, and *iii*) the complexity of the fishery, where vulnerability is based on size, shell status (e.g., soft, hard), maturity, and sex, could be accommodated.

Full details of the modeling details are provided in Bunnell and Miller (in press), and are only summarized here. Briefly, blue crabs were modeled to grow discontinuously by molting, the magnitude and frequency of which was governed by temperature (Tagatz 1968, Leffler 1972) and crab size (Tagatz 1968, Fitz and Wiegert 1991). Intermolt period, or the time between molts, was represented by degree-days ($^{\circ}\text{C}$) and also was dependent on blue crab CW. Each day in the model, blue crabs accumulated degree-days until some threshold had been reached which, in turn, resulted in molting. Degree-days were calculated by subtracting 8.9, the physiological minimum temperature for blue crab growth (Smith 1997), from the mean daily water temperature. Molting did not occur when water temperatures were less than 8.9 $^{\circ}\text{C}$ (i.e., winter in the Chesapeake Bay) because degree-days will not accumulate. The shell status of blue crab was monitored (i.e., hard-shell, soft-shell, peeler) because the fishery regulations change with shell status. Blue crabs were assumed to be soft-shelled for the day of molting and the following day only (i.e., individuals returned to hard-shell status two days after molting). Blue crab shells were classified as “peelers” for approximately one week before molting.

Maturity of individual blue crabs was assigned only to females in this model, as maturity was assumed not to influence growth of males. Maturation was a function of time: females could mature between 1 April and 1 June or between 1 July and 1 October, corresponding with the two mating periods in the Bay (Hines et al. 2003). During each molt within those periods, a probabilistic, size-dependent function determined if an immature female reached maturity (see Section 2.3.2). Once a female matured, she remained a hard-shell crab and ceased molting for the remainder of her lifetime.

For each potential biological reference point considered, we simulated a cohort of 2000 juvenile, “super-individual” blue crabs through two years following the year of initial settlement. The model began on January 1 of year $x+1$ and continued through December 31 of year $x+2$, where x is the year of settlement. By assigning each super-individual an internal amount of 150 000 at the start of each simulation, we modeled a cohort of 300 million individuals, which is within the range (95 to 540 million) of new recruits estimated in the Chesapeake Bay during the first winter of life. Sizes for each super-individual in the cohort were drawn from a log-normal distribution with a mean of 27.2 mm CW and a standard deviation of 10.3, which is reflective of the size distribution sampled in the winter dredge survey. We assumed an initial 1:1 sex ratio for each cohort. Water temperatures were equal across both years, and equaled the mean daily water temperature at the VIMS pier for 1991 – 2002. Natural and fishing mortality rates were applied separately in the model.

Reference points were presented in terms of the exploitation fraction μ , calculated as the number of crabs harvested divided by the number of crabs in the cohort that were alive when the fishery began (April 1, year $x+1$). Spawning potential equaled the sum of the total numbers of eggs predicted to be spawned by females in the second year of the

simulation; no females were large enough to mature and spawn by September 15 of the first year of the model. Mature females were randomly assigned a spawning day between 15 May and 15 September. The spawning potential, or the number of eggs spawned, for each super-individual was the product of its size-based fecundity (millions of eggs = $-2.248 + 0.337(CW)$; (Prager et al. 1990) and its internal amount, a , (i.e., the number of individuals that were living) on the day of spawning. For each combination of fishing and natural mortality, we summed the spawning potential of each super-individual. We assumed that the virgin, unfished spawning potential occurred when μ or $F = 0.0 \text{ y}^{-1}$.

The IBM SPR could replicate results from traditional per recruit analysis when variability among individual growth trajectories was set to zero, and when only a single fishery was modeled. Indeed simulation results presented by Bunnell and Miller (in press) indicated that inclusion of individual variability in growth produced results that did not differ substantially from those developed in traditional per recruit analyses. Importantly, however, inclusion of multiple fisheries, each with different size-dependent and shell stage-dependent vulnerabilities did yield results that differed substantially from the traditional per recruit models.

7.2.2. IBM SPR Reference Points

We interpolated model results to create a contour plot of the proportion of the virgin, unfished spawning potential that is protected for a given level of natural mortality and fishing pressure (Figure 45). The estimates of $\mu_{20\%}$ estimate decreased with increasing levels of M . This pattern contrasts with that from a traditional per recruit analyses based on fishing mortality rates. In traditional analyses increases in estimates of M would have produced increases in $F_{x\%}$ reference points. There are two important reasons why the $\mu_{20\%}$ contours decline with increasing M : i) there is a considerable period of time when immature crabs are vulnerable to the fishery, and ii) μ -based reference points are based on numbers of crabs surviving and not on rates of mortality. When these conditions hold, given that the number that must survive to maturity to provide 20% of the virgin spawning potential is a fixed number, fewer immature crabs can be removed by the fishery. Thus, the permitted exploitation fraction must decline. We note that this pattern is not an artifact of the use of the IBM SPR. Indeed if we calculate μ -based reference points from the traditional per recruit analysis, $\mu_{20\%}$ reference points do decline as M increases. Equally, this is not a feature of all per recruit models, but rather reflects the exploitation of immature crabs.

We used the IBM SPR to estimate the threshold ($\mu_{10\%}$) and target ($\mu_{20\%}$) exploitation reference points for $M=0.9$ as 0.53 and 0.45 respectively (Table 28). We present results for other values of M for comparative purposes (Table 28).

7.3.3. Control Rule Plots

We developed two control rule plots based on the IBM SPR reference points. The first control rule plot used the empirical estimates of μ and abundance developed from the winter dredge survey (Fig 46). We view Figure 46 as the prime control rule for management. There is no requirement to estimate a value for M to plot the data on this control rule plot once the threshold and target exploitation fractions have been estimated. All that is needed are estimates of abundance from the winter dredge survey, and the subsequent number of crabs harvested by the fisheries in the following years. For this figure, we used the standardized survey Z-score for 1968 as the biomass limit. However, as the winter dredge time series lengthens, it is hoped that these data will replace the survey Z-score. Figure 46 is interpreted to mean that the stock is not currently overfished, nor is it experiencing overfishing. However, we note that the stock was experiencing overfishing from 1998-2002, and as a result is not at a lower abundance in years characterized by similar levels of μ observed in 2003. It appears from Figure 46 that the stock is at a relatively low level of abundance compared to periods when exploitation fractions were of a similar magnitude.

In order to provide a broader historical perspective for the performance of the fishery and for the current abundance level, we developed a second plot that used the estimate of abundance and μ predicted by the CMS model (see Section 6). When $M=0.9$ was used in the CMS to calculate μ and abundance, the results indicate that the stock is close to the target exploitation rate, but at a relatively low level of abundance (Fig. 47). This figure reinforces the concept that the current levels of abundance are relatively low compared to other periods when similar levels of μ were in operation. The model predictions also indicate that the Chesapeake Bay blue crab stock is not currently experiencing overfishing, nor is it overfished. However, the stock experienced overfishing between 1972-1976 and 1995-2001. One important distinction in this figure from the BBCAC plot, is that the lowest observed sustainable biomass was predicted to have occurred in 1975 and not 1968.

7.4 Summary

The conclusions regarding the current status of the stock are the same, regardless of the reference point framework chosen. Both the existing BBCAC reference point and the two μ -based reference point framework all indicate that the stock is not experiencing overfishing, nor is it overfished. However, all frameworks also indicate that the stock did experience overfishing for a period of time in the late 1990's early 2000's: the frameworks only differ in the exact years during which overfishing was deemed to have occurred. All frameworks also indicate that the current level of abundance believed to characterize the blue crab stock is lower than that observed previously when similarly levels of exploitation have occurred. This would suggest that, providing the system carrying capacity has not changed, crab abundance should increase in coming years if the exploitation pressure is maintained at current or lower levels. Additionally, this would suggest that the historical pattern of overfishing is responsible for the current low level of abundance. We note that in no framework is the exploitation pattern compatible with

target levels of exploitation. Continued regulation is necessary to ensure that target exploitation levels are achieved.

8. Conclusions and Recommendations

8.1. Conclusions

1. In 2003, the blue crab stock was not experiencing overfishing, nor was it overfished. However, exploitation rates were still higher than target exploitation rates.
2. The blue crab stock did experience a period of overfishing from 1998-2002, which was associated with below average abundances of crab. There is yet to be convincing signs of recovery from this period of low abundance.
3. Analyses indicate that the blue crab stock experienced overfishing during the early 1970's which was associated with low levels of abundance similar to those observed currently.
4. Analyses indicate that the blue crab stock was at relatively high levels of abundance during the 1980's, a period when exploitation rates were lower than those currently observed.
5. Independent analyses indicate that the most likely value for natural mortality (M) $\sim 0.9 \text{ yr}^{-1}$, based on an analysis of empirical results from Chesapeake and Delaware Bays, life history analysis and the stock assessment model developed herein. Values less than 0.6 or above 1.2 do not appear credible.
6. Data on the maximum age of crabs was reviewed, and the assessment determined that uncertainties associated with this value make it unsuitable as the sole foundation for calculation of natural mortality rate.
7. Fishery independent surveys were an important component of the analyses presented here. In our analyses of the principal fishery-independent surveys we found that the individual year effects did not mask the ability of the survey to track cohorts.
8. Despite the finding above, size-at-age is highly variable, and concern over assigning cohort membership is still warranted.
9. The survey of the spawning stock abundance indicates that the spawning stock is at a low level of abundance. Trends in spawning stock abundance are particularly worthy of monitoring.
10. Reporting changes have had a significant impact on the level of landings reported, and corrections must be applied to accurately represent the historical pattern of removals. Appropriately adjusted time series indicate that recent landings of blue crab in the Chesapeake Bay are at historical low levels.
11. Exploitation fractions, whether quantified through direct empirical approaches, or with the assessment model increased from 1990-1999, and those estimated by the model were high compared to those for the previous decade. Exploitation fractions have declined in more recent years.
12. There is a strong negative relationship between exploitation fraction and abundance, whether determined through direct empirical approaches, or with the assessment model, which indicates that a greater proportion of the population is harvested when abundance levels are low. This pattern does not promote sustainability.

13. The method for calculating fishing pressure was changed from one based on fishing mortality rate, F , to one based on exploitation fraction, μ . The advantage of the changes is that estimates of μ are not reliant on estimates of M .
14. A new μ -based overfishing reference point was developed as $\mu_{10\%}=0.53$. A parallel μ -based target reference point, $\mu_{20\%}=0.46$, was also developed.

8.2 Recommendations

1. Research that quantifies size-dependent, spatially-dependent and inter-annual patterns in natural mortality would greatly improve future assessments.
2. Reproductive information (e.g., maturity, fecundity and batch production) for blue crab was collected during a period of relatively high crab abundance. No recent estimates of reproductive parameters are available. Research that quantifies the pattern of maturation, the number of zoea released and the frequency of spawning would be helpful.
3. Future assessments would benefit for information on spatial and inter-annual variation in crab growth in Chesapeake Bay.
4. Fishery-independent surveys are critical to the assessment. Additional analysis of the survey time series to understand their coherence, and their ability to track population variation would be beneficial. A thorough evaluation of survey efficiency and options for enhancing their utility should be undertaken.
5. The monitoring of removals by the different fisheries has improved. However, we recommend that attention be given to ensuring that the biological characteristics of each fishery be quantified, and that the spatial and temporal distribution of the removals be quantified.
6. Reconstruction of landings was difficult because at the time that reporting changes were implemented little considerations was given to the cross-validation of alternative reporting systems. Should future changes in reporting systems be implemented, cross validation studies should be conducted.
7. Regional agencies should strive to ensure that high quality information on the temporal and spatial distribution of effort in the fisheries be collected to obtain reliable catch-per-effort estimates which may be useful in developing surplus production models.
8. Given the importance of the blue crab fishery to the Chesapeake Bay region, greater coordination of the management of fishery-independent and fishery-dependent data is recommended.
9. We recommend management be based on reference points developed from exploitation rates. A reexamination of reference points based on preserving 10% (threshold) to 20% (target) of the maximum spawning potential may be warranted.
10. Given the negative relationship between exploitation fraction and abundance, the use of an estimate of abundance as a trigger for action, above the overfished threshold, to limit fishing pressure should be considered.
11. Given differences in exploitation between male and female crabs, a sex-specific assessment of the blue crab stock, which might include individual sex-specific control rules, should be considered in the future.

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Table 1. Published estimates of von Bertalanffy growth parameters blue crab used in the assessment. Studies were categorized as to whether they were purely empirical or whether they relied on parameters estimated external to the study.

Source	Approach	Direct empirical estimation?	K	CW_x	t_0	Size at onset of winter when age (mm CW)		
						0	1	2
Rothschild et al. (1991)	Assumed max age and size ^{ab}	No	0.51	187.0	0.00	49.30	104.31	137.34
Rugolo et al. (1997)	Assumed max age and size ^a	No	0.59	262.5	0.01	76.68	159.18	205.06
Smith (1997)	Molt process model	No	0.64	191.9	0.31	32.51	107.85	147.58
Eggleston et al. (2004)	Molt process model	No	0.74	237.7	0.02	82.95	163.87	202.47
Rothschild et al. (1988)	Modal analysis ^b	Yes	1.08	176.0	0.00	83.94	144.74	165.38
Ju et al. (2001) Pond Mesocosm	Lipofuscin-based age and assumed max size ^a	Yes	1.09	240.0	0.40	47.01	175.11	218.18
Helser and Kahn (1999) Model 1	MULTIFAN	Yes	0.75	234.7	-0.10	95.86	169.12	203.72
Helser and Kahn (1999) Model 2	MULTIFAN	Yes	0.62	200.6	-0.15	74.60	132.82	164.14
Helser and Kahn (1999) Model 3	MULTIFAN	Yes	0.93	200.3	-0.15	100.58	160.96	184.78
Eggleston et al. (2004)	Modal analysis and assumed max size ^a	Yes	0.47	216.9	0.02	51.75	113.68	152.39
Overall average			0.74	206.9	0.05	69.52	143.16	178.10
Overall SD			0.22	24.4	0.21	22.91	26.73	28.36
Empirically-based average			0.82	202.9	0.00	75.62	149.40	181.43
Empirically-based SD			0.25	24.1	0.23	22.32	23.48	25.53

^a CW_x fixed and not used in averages

^b t_0 fixed and not used in averages

Table 2. Estimates of seasonalized von Bertalanffy parameter estimates for Chesapeake Bay blue crab

Experiment	K'	Cwinf	t0	C	ts	r2
Pond (Aug 1998 - Aug 1999)	1.71	207.5	0.31	1	0.01	0.9
Pond (Mar 1999 - Apr 2000)	1.19	207.5	0.15	0.94	0.01	0.93

Table 3. Predictive methods used to obtain the estimated ranges of M shown in Figure 6. Superscript letters indicate references and superscript symbols indicate parameter definitions or values listed below the Table.

Method Name	Equations/Variables	Range of Inputs	Range of M	Data	Comments
Age at Maturity B-H Invariant	$M * t_m = X$, [*] where $X = 1.65^a, 1.75-2.2^{bc}$	$t_m = 1 - 1.67^{ns}$	0.99 – 2.20	- $X = 1.65$ theoretical; maximization of lifetime fecundity ^a - $X = 1.75-2.2$ from empirical regressions; > 25 species, high R^2	- constant M over lifespan - assumes fecundity increases with weight
Growth B-H Invariant	$M/K = X$, [§] where $X = 1.5^a$ or 1.65^c	$K = 0.47^o - 1.09^e$ (average = 0.82) [see Table 1]	0.71 – 1.80 (avg K : 1.23 – 1.35)	- $X = 1.65$ empirical [from Pauly (1980) ^k] $K = 0.04 - 4.92$ $M = 0.03 - 7.80$ $R^2 = 0.66$ - $X = 1.5$ theoretical; maximization of lifetime fecundity ^a	- constant M over lifespan - assumes fecundity increases with weight - Charnov ^c and Jensen ^a show that temperature adds little to Pauly (1980) model
A-C t_{max} and K^m	$M = (3K)/[e^{(K*(0.38*t_{max}))}-1]$, ^{§‡}	$K = 0.47^o - 1.09^e$ (average = 0.82) [see Table 1] $t_{max} = 4^i - 6^f$	0.30 – 1.35 (avg K : 0.45 – 0.99)	- theoretical - tested against 8 species from eastern Pacific $M \approx 0.1 - 1.7$ $R^2 \approx 0.9$	- constant M over lifespan - t_{max} used to get age at which max biomass occurs ($t_{mb} =$ $t_{max} * 0.38$), based on 63 species - K s not given for test species, but range from anchovies to tunas
Hoenig t_{max}^j	$M = e^{[1.44 - 0.982 * \ln(t_{max})]}$, [‡]	$t_{max} = 4^i - 6^f$	0.73 – 1.08	- empirical regression - 134 stocks, from mollusks to cetaceans $M = 0.01 - 2.0$ $t_{max} = 1 - 123$ $R^2 = 0.82$	- constant M over lifespan - age ranges used to estimate M varied among stocks - sample size for getting t_{max} not included in model
Pauly Length ^k	$\log M = -0.0066 - 0.279 * \log(L_\infty) +$ $0.6543 * \log K + 0.4634 * \log T$, ^{§‡}	$K = 0.47^o - 1.09^e$ (average = 0.82) $L_\infty = 17.6^d - 23.5^l$ cm (average = 20.29 cm) [see Table 1]	0.91 – 1.72 (avg K & L_∞ : 1.37)	- empirical regression - 175 stocks, 84 marine and freshwater species $R^2 = 0.72$ $T = 5 - 28$ $L_\infty = 4 - 1226$ cm	- constant M over lifespan - K is most important to regression; size and temperature less so

		$T = 16.5^{\text{e}}$		$K = 0.04 - 4.92$ $M = 0.03 - 7.80$	
Pauly Weight ^k	$\log M = -0.2107 - 0.0824 * \log(W_{\infty}) + 0.6757 * \log K + 0.4627 * \log T, \text{ }^{\text{f}}$	$K = 0.47^{\text{a}} - 1.09^{\text{c}}$ (average = 0.82) $W_{\infty} = 231^{\text{d}} - 464^{\text{f}}$ g (average = 326.2 g) [see Table 1] $T = 16.5^{\text{e}}$	0.82 – 1.52 (avg K & W_{∞} : 1.22)	- empirical, as above $R^2 = 0.71$ T,K,M as above $W_{\infty} = 0.4 - 1.4 \times 10^7$ g	- constant M over lifespan - K is most important to regression; size and temperature less so
Roff t_m and K ^h	$M = (3K) / [e^{(K * t_m)} - 1], \text{ }^{\text{g}}$	$K = 0.47^{\text{a}} - 1.09^{\text{c}}$ (average = 0.82) [see Table 1] $t_m = 1 - 1.67^{\text{ns}}$	0.63 – 2.35 (avg K: 0.84 – 1.94)	- theoretical - tested against 30 stocks [data mostly from Ni (1978) ^q] $R^2 = 0.47$ - poorer predictions for $M > 0.9$	- constant M over lifespan - compared favorably to Pauly (1980) length-based regression in predicting M for a subset of 17 stocks
Lorenzen Weight-Varying ^r	$M = 3.00 * w^{-0.288}, \text{ }^{\text{p}}$	$w = 25.0 - 450.0$ g (average = 150 g)	0.52 – 1.19 (avg w: 0.71)	- empirical regression (Thiel) on log-transformed data - 308 “stocks” - Thiel 90% CIs for parameters: $2.70 < 3.00 < 3.30$ $-0.315 < -0.288 < -0.261$	- size-dependent M - nearly identical predictions for blue crab as Peterson and Wroblewski (1984) ^q theoretical approach

^a t_m = age at maturity (years)

^b K = coefficient of von Bertalanffy growth model

^c t_{max} = maximum age (years)

^d L_{∞} = asymptotic length from the von Bertalanffy growth model (here, carapace width)

^e W_{∞} = asymptotic weight from the von Bertalanffy growth model

^f grand annual mean of VIMS pier water temperature (T) for 1990-2003

^g w = wet weight

Table 4. Summary of size, times and areas used in calculating fishery-independent crab abundance indices for the Chesapeake Bay.

	0			1+			Adult Females		
	Size	Month	Areas	Size	Month	Areas	Size	Month	Areas
Calvert Cliffs				>95	June - Aug	N/A	>=120	Sept	N/A
MD Trawl (revised during 2001 TSC meeting)	<=50mm	Sept & Oct	Pocomoke, Tangier, Choptank, Patuxent	>=51	June - Oct	not Potomac	All	Aug-Oct	not Potomac or Chester
VA Trawl	<=50mm <=60mm <=60mm	Sept - Nov	Upper and Lower Rivers	>=36mm >=51mm >=61mm	Aug - Oct	Upper and Lower Rivers	All	Aug-Oct	Lower Rivers & Ches. Bay
VA Trawl Spring Juvs.	<=60mm <=80mm <=90mm	May -July	Upper and Lower Rivers						

Table 5. Correlation structure among fishery-independent age-0 CPUE time series for different Virginia tributaries calculated from the VIMS trawl time series

		Rappahannock		York		James	
		Spring	Fall	Spring	Fall	Spring	Fall
Rappahannock	Spring	1.0000					
	Fall	0.2439	1.0000				
York	Spring	0.7066	0.4553	1.0000			
	Fall	0.2937	0.3428	0.1677	1.0000		
James	Spring	0.7271	0.2677	0.6418	0.1356	1.0000	
	Fall	0.2959	0.3245	0.1161	0.5986	0.3517	1.0000

Table 6. Geometric mean catch.tow-1 in the VIMS trawl survey. Criteria for determining cohort membership are provided in Table 4. For more detail see Bonzek et al.

Year	Survey			Zscore		
	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem
1955						
1956	8.205	9.862	0.968	0.776	0.016	-0.527
1957	8.804	4.002	0.551	0.928	-0.786	-0.712
1958	1.368	6.606	0.402	-0.953	-0.429	-0.779
1959	1.264	4.731	1.275	-0.979	-0.686	-0.391
1960			0.776	-1.299	-1.333	-0.612
1961	0.804	8.534	0.084	-1.096	-0.166	-0.920
1962	2.886	11.676	3.675	-0.569	0.264	0.676
1963	0.279	12.063	2.394	-1.228	0.317	0.106
1964	4.083	4.726	1.070	-0.266	-0.687	-0.482
1965	4.729	5.430	0.344	-0.103	-0.590	-0.804
1966	3.548	19.075	2.041	-0.402	1.276	-0.050
1967	0.654	3.992	0.899	-1.133	-0.787	-0.558
1968	6.240	4.149	2.217	0.279	-0.765	0.028
1969	1.038	17.576	1.189	-1.036	1.071	-0.429
1970	12.885	19.561	7.596	1.960	1.343	2.418
1971	8.867	36.642	10.886	0.943	3.679	3.880
1972	2.197	7.211	3.457	-0.743	-0.347	0.579
1973	2.377	5.505	0.830	-0.698	-0.580	-0.588
1974	0.319	2.534	0.828	-1.218	-0.986	-0.589
1975	0.866	1.813	0.195	-1.080	-1.085	-0.871
1976	1.637	2.542	0.619	-0.885	-0.985	-0.682
1977	5.354	8.413	0.997	0.055	-0.182	-0.514
1978	4.151	10.909	2.202	-0.249	0.159	0.021
1979	0.342	7.795	4.289	-1.213	-0.267	0.949
1980	11.969	15.863	3.785	1.728	0.837	0.725
1981	11.269	27.227	4.560	1.551	2.392	1.069
1982	5.110	14.613	3.842	-0.007	0.666	0.750
1983	10.287	18.137	4.035	1.303	1.148	0.836
1984	3.963	12.960	1.791	-0.297	0.440	-0.161
1985	5.529	12.835	2.826	0.099	0.423	0.299
1986	3.455	6.851	2.425	-0.425	-0.396	0.120
1987	3.678	7.561	3.304	-0.369	-0.299	0.511
1988	3.797	11.954	4.838	-0.339	0.302	1.193
1989	17.791	12.572	4.889	3.200	0.387	1.216
1990	12.404	29.679	8.018	1.838	2.727	2.606
1991	7.071	9.084	2.728	0.489	-0.090	0.255
1992	4.878	8.172	0.859	-0.065	-0.215	-0.575
1993	5.850	3.212	1.382	0.180	-0.894	-0.343
1994	2.716	2.817	0.882	-0.612	-0.948	-0.565
1995	11.326	6.626	0.391	1.565	-0.427	-0.784
1996	5.874	8.585	1.406	0.187	-0.159	-0.333
1997	4.382	9.702	0.881	-0.191	-0.006	-0.566
1998	8.416	5.412	0.368	0.830	-0.593	-0.794
1999	2.896	6.430	0.541	-0.566	-0.453	-0.717
2000	3.045	3.781	0.462	-0.529	-0.816	-0.752
2001	2.643	4.585	0.480	-0.631	-0.706	-0.744
2002	2.661	4.158	0.387	-0.626	-0.764	-0.785
2003	8.392	5.529	0.315	0.823	-0.577	-0.817
2004	4.244	4.046	0.369	-0.226	-0.780	-0.793
Average	5.136	9.745	2.154			
SD	3.954	7.310	2.250			

Table 7. Correlations between abundances of age-0, age-1+ and mature female crabs determined by the VIMS trawl survey in one year with the abundances of these stages lagged by 0, 1 or 2 years.

Correlation Matrice

		Year i		
		age-0	age-1+	mat fem
Year i	age-0		1	
	age-1+	0.491211		1
	mat fem	0.492668	0.493808	1

		Year i		
		age-0	age-1+	mat fem
Year i+1	age-0	0.225033		
	age-1+	0.523474	0.36153	
	mat fem	0.385717	0.493808	0.583658

		Year i		
		age-0	age-1+	mat fem
Year i+2	age-0	0.054514		
	age-1+	-0.08667	0.149999	
	mat fem	0.034708	0.210468	0.202045

Table 8. Mean catch.tow⁻¹ in the Calvert Cliff pot survey. The survey uses commercial crab pots and thus does not reliably index age-0 crabs. For more detail see Abbe and Stagg (1996)

Year	Survey			Zscore		
	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem
1955						
1956						
1957						
1958						
1959						
1960						
1961						
1962						
1963						
1964						
1965						
1966						
1967						
1968		1.100	0.090		-1.491	-1.250
1969		5.580	1.840		0.201	-0.605
1970		3.340	0.210		-0.645	-1.205
1971		6.800	3.420		0.662	-0.022
1972		4.720	1.750		-0.124	-0.638
1973		2.670	2.450		-0.898	-0.380
1974		5.400	2.720		0.133	-0.280
1975		4.420	4.270		-0.237	0.291
1976		3.940	1.820		-0.418	-0.612
1977		2.580	1.750		-0.932	-0.638
1978		3.570	2.840		-0.558	-0.236
1979		4.550	5.550		-0.188	0.763
1980		3.990	2.810		-0.399	-0.247
1981		16.490	13.490		4.321	3.690
1982		5.960	3.330		0.345	-0.055
1983		7.040	5.350		0.752	0.689
1984		8.150	8.570		1.172	1.877
1985		9.390	10.210		1.640	2.481
1986		5.480	4.660		0.163	0.435
1987		2.270	1.010		-1.049	-0.910
1988		3.760	3.590		-0.486	0.041
1989		5.710	4.700		0.250	0.450
1990		4.440	4.390		-0.230	0.336
1991		5.220	5.570		0.065	0.771
1992		5.680	1.910		0.239	-0.579
1993		8.630	4.740		1.353	0.465
1994		5.760	2.740		0.269	-0.273
1995		4.110	1.100		-0.354	-0.877
1996		4.960	1.440		-0.033	-0.752
1997		5.140	3.190		0.035	-0.107
1998		2.560	1.650		-0.940	-0.675
1999		4.020	3.350		-0.388	-0.048
2000		3.790	1.500		-0.475	-0.730
2001		2.870	1.300		-0.898	-0.804
2002		3.700	2.570		-0.509	-0.335
2003		4.130	3.390		-0.347	-0.033
2004						
Average		5.048	3.480			
SD		2.648	2.713			

Table 9. Mean catch.tow⁻¹ in the Maryland Trawl survey. Criteria for determining cohort membership are provided in Table 4.

Year	Survey			Zscore		
	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem
1955						
1956						
1957						
1958						
1959						
1960						
1961						
1962						
1963						
1964						
1965						
1966						
1967						
1968						
1969						
1970						
1971						
1972						
1973						
1974						
1975						
1976						
1977	0.838	15.083	1.314	-0.511	1.214	0.277
1978	0.220	2.208	0.346	-0.764	-1.334	-1.035
1979	0.274	3.117	0.603	-0.742	-1.154	-0.686
1980	0.578	0.904	0.267	-0.618	-1.592	-1.141
1981	0.613	6.579	1.430	-0.603	-0.469	0.434
1982	0.457	5.191	0.489	-0.667	-0.744	-0.840
1983	1.443	14.168	1.561	-0.263	1.033	0.612
1984	2.433	18.353	3.047	0.142	1.861	2.624
1985	1.434	13.914	1.663	-0.267	0.982	0.749
1986	1.624	15.036	1.357	-0.189	1.204	0.335
1987	0.911	9.761	1.444	-0.481	0.161	0.453
1988		8.394	0.587	-0.855	-0.110	-0.708
1989	3.330	11.134	0.282	0.510	0.432	-1.121
1990	11.683	9.559	1.313	3.932	0.120	0.275
1991	1.229	7.882	0.477	-0.351	-0.211	-0.857
1992	7.337	9.644	0.610	2.152	0.137	-0.677
1993	3.997	10.722	0.832	0.783	0.351	-0.376
1994	1.138	12.353	1.594	-0.388	0.673	0.657
1995	0.651	3.875	0.923	-0.588	-1.004	-0.253
1996	2.068	20.677	3.203	-0.007	2.321	2.835
1997	0.790	11.390	1.762	-0.531	0.483	0.884
1998	2.151	3.895	0.878	0.027	-1.000	-0.314
1999	2.077	8.068	1.100	-0.003	-0.174	-0.013
2000	1.443	3.821	0.458	-0.263	-1.015	-0.883
2001	1.769	4.708	0.628	-0.130	-0.839	-0.653
2002	2.072	5.191	0.736	-0.006	-0.744	-0.506
2003	1.666	6.022	1.056	-0.172	-0.579	-0.073
2004						
Average	2.086	8.950	1.110			
SD	2.441	5.053	0.738			

Table 10. Correlations between abundances of age-0, age-1+ and mature female crabs determined by the MD trawl survey in one year with the abundances of these stages lagged by 0, 1 or 2 years.

Correlation matrices

		Year i		
		age-0	age-1+	mat fem
Year i	age-0	1		
	age-1+	0.168425	1	
	mat fem	0.027017	0.2228	1

		Year i		
		age-0	age-1+	mat fem
Year i+1	age-0	0.160269		
	age-1+	0.101412	0.293183	
	mat fem	-0.126947	0.2228	0.255315

		Year i		
		age-0	age-1+	mat fem
Year i+2	age-0	0.607901		
	age-1+	0.023709	0.191791	
	mat fem	-0.142497	0.060714	0.106736

Table 11. Mean catch.tow⁻¹ in the winter dredge survey. Criteria for determining cohort membership are provided in Table 4. For more detail see Sharov et al. (2003)

Year	Survey			Zscore		
	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem
1955						
1956						
1957						
1958						
1959						
1960						
1961						
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1987						
1988						
1989	47.340	34.820	9.010	1.206	1.128	0.104
1990	36.430	49.110	18.590	0.489	2.452	2.455
1991	10.690	27.440	15.520	-1.203	0.444	1.701
1992	51.270	37.050	6.210	1.465	1.334	-0.583
1993	30.940	22.610	9.280	0.128	-0.004	0.170
1994	30.710	19.720	4.360	0.113	-0.271	-1.037
1995	51.820	24.680	10.070	1.501	0.188	0.364
1996	52.160	17.590	7.480	1.523	-0.469	-0.271
1997	16.870	19.540	8.270	-0.796	-0.288	-0.077
1998	22.760	9.170	4.730	-0.409	-1.249	-0.946
1999	13.900	16.590	10.710	-0.992	-0.561	0.521
2000	15.860	10.760	4.240	-0.863	-1.102	-1.066
2001	19.810	12.540	4.400	-0.603	-0.937	-1.027
2002	17.880	21.910	7.690	-0.730	-0.068	-0.220
2003	16.370	16.200	8.220	-0.829	-0.598	-0.090
2004						
Average	28.987	22.649	8.585			
SD	15.215	10.793	4.076			

Table 12. Correlations between abundances of age-0, age-1+ and mature female crabs determined by the winter dredge survey in one year with the abundances of these stages lagged by 0, 1 or 2 years.

Correlation Matrices				
Year i				
	age-0	age-1+	mat fem	
Year i	age-0	1.000		
	age-1+	0.491	1.000	
	mat fem	0.012	0.606	1.000

Year i				
	age-0	age-1+	mat fem	
Year i+1	age-0	0.227		
	age-1+	0.315	0.557	
	mat fem	0.500	0.606	0.220

Year i				
	age-0	age-1+	mat fem	
Year i+2	age-0	0.073		
	age-1+	0.130	0.787	
	mat fem	0.010	0.188	0.101

Table 13 Average survey Z-score for the four principal fishery-independent surveys.

Year	VIMS Zscore			CC Zscore			MD Trawl Zscore			WDS Zscore			Average		
	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem	Age-0	Age-1+	Mat Fem
1955															
1956	0.776	0.016	-0.527										0.776	0.016	-0.527
1957	0.928	-0.786	-0.712										0.928	-0.786	-0.712
1958	-0.953	-0.429	-0.779										-0.953	-0.429	-0.779
1959	-0.979	-0.686	-0.391										-0.979	-0.686	-0.391
1960	-1.299	-1.333	-0.612										-1.299	-1.333	-0.612
1961	-1.096	-0.166	-0.920										-1.096	-0.166	-0.920
1962	-0.569	0.264	0.676										-0.569	0.264	0.676
1963	-1.228	0.317	0.106										-1.228	0.317	0.106
1964	-0.266	-0.687	-0.482										-0.266	-0.687	-0.482
1965	-0.103	-0.590	-0.804										-0.103	-0.590	-0.804
1966	-0.402	1.276	-0.050										-0.402	1.276	-0.050
1967	-1.133	-0.787	-0.558										-1.133	-0.787	-0.558
1968	0.279	-0.765	0.028	-1.491	-1.250								0.279	-1.128	-0.611
1969	-1.036	1.071	-0.429	0.201	-0.605								-1.036	0.636	-0.517
1970	1.960	1.343	2.418	-0.645	-1.205								1.960	0.349	0.606
1971	0.943	3.679	3.880	0.662	-0.022								0.943	2.171	1.929
1972	-0.743	-0.347	0.579	-0.124	-0.638								-0.743	-0.235	-0.029
1973	-0.698	-0.580	-0.588	-0.898	-0.380								-0.698	-0.739	-0.484
1974	-1.218	-0.986	-0.589	0.133	-0.280								-1.218	-0.427	-0.435
1975	-1.080	-1.085	-0.871	-0.237	0.291								-1.080	-0.661	-0.290
1976	-0.885	-0.985	-0.682	-0.418	-0.612								-0.885	-0.702	-0.647
1977	0.055	-0.182	-0.514	-0.932	-0.638	-0.511	1.214	0.277					-0.228	0.033	-0.292
1978	-0.249	0.159	0.021	-0.558	-0.236	-0.764	-1.334	-1.035					-0.507	-0.578	-0.416
1979	-1.213	-0.267	0.949	-0.188	0.763	-0.742	-1.154	-0.686					-0.977	-0.536	0.342
1980	1.728	0.837	0.725	-0.399	-0.247	-0.618	-1.592	-1.141					0.555	-0.385	-0.221
1981	1.551	2.392	1.069	4.321	3.690	-0.603	-0.469	0.434					0.474	2.081	1.731
1982	-0.007	0.666	0.750	0.345	-0.055	-0.667	-0.744	-0.840					-0.337	0.089	-0.048
1983	1.303	1.148	0.836	0.752	0.689	-0.263	1.033	0.612					0.520	0.978	0.712
1984	-0.297	0.440	-0.161	1.172	1.877	0.142	1.861	2.624					-0.077	1.157	1.446
1985	0.099	0.423	0.299	1.640	2.481	-0.267	0.982	0.749					-0.084	1.015	1.176
1986	-0.425	-0.396	0.120	0.163	0.435	-0.189	1.204	0.335					-0.307	0.324	0.297
1987	-0.369	-0.299	0.511	-1.049	-0.910	-0.481	0.161	0.453					-0.425	-0.396	0.018
1988	-0.339	0.302	1.193	-0.486	0.041	-0.855	-0.110	-0.708					-0.597	-0.098	0.175
1989	3.200	0.387	1.216	0.250	0.450	0.510	0.432	-1.121	1.800	0.797	0.161	1.837	0.466	0.176	
1990	1.838	2.727	2.606	-0.230	0.336	3.932	0.120	0.275	1.133	1.679	1.326	2.301	1.074	1.136	
1991	0.489	-0.090	0.255	0.065	0.771	-0.351	-0.211	-0.857	-0.439	0.341	0.953	-0.100	0.026	0.280	
1992	-0.065	-0.215	-0.575	0.239	-0.579	2.152	0.137	-0.677	2.040	0.935	-0.160	1.375	0.274	-0.503	
1993	0.180	-0.894	-0.343	1.353	0.465	0.783	0.351	-0.376	0.798	0.043	0.193	0.587	0.213	-0.015	
1994	-0.612	-0.948	-0.565	0.269	-0.273	-0.388	0.673	0.657	0.784	-0.135	-0.405	-0.072	-0.035	-0.147	
1995	1.565	-0.427	-0.784	-0.354	-0.877	-0.588	-1.004	-0.253	2.073	0.171	0.250	1.017	-0.404	-0.406	
1996	0.187	-0.159	-0.333	-0.033	-0.752	-0.007	2.321	2.835	2.094	-0.267	-0.026	0.758	0.466	0.431	
1997	-0.191	-0.006	-0.566	0.035	-0.107	-0.531	0.483	0.884	-0.062	-0.146	0.071	-0.261	0.091	0.070	
1998	0.830	-0.593	-0.794	-0.940	-0.675	0.027	-1.000	-0.314	0.298	-0.786	-0.360	0.385	-0.830	-0.536	
1999	-0.566	-0.453	-0.717	-0.388	-0.048	-0.003	-0.174	-0.013	-0.243	-0.328	0.367	-0.271	-0.336	-0.102	
2000	-0.529	-0.816	-0.752	-0.475	-0.730	-0.263	-1.015	-0.883	-0.123	-0.688	-0.420	-0.305	-0.749	-0.696	
2001	-0.631	-0.706	-0.744	-0.898	-0.804	-0.130	-0.839	-0.653	0.118	-0.578	-0.400	-0.214	-0.755	-0.650	
2002	-0.626	-0.764	-0.785	-0.509	-0.335	-0.006	-0.744	-0.506	0.000	0.000	0.000	-0.210	-0.504	-0.407	
2003	0.823	-0.577	-0.817	-0.347	-0.033	-0.172	-0.579	-0.073	-0.092	-0.352	0.064	0.186	-0.464	-0.215	
2004	-0.226	-0.780	-0.793										-0.226	-0.780	-0.793

Table 14. Correlations between abundances of age-0, age-1+ and mature female crabs determined by the combined survey Z-score survey in one year with the abundances of these stages lagged by 0, 1 or 2 years.

Correlation Matrices

		Year i		
		age-0	age-1+	mat fem
Year i	age-0	1.000		
	age-1+	0.398	1.000	
	mat fem	0.370	0.841	1.000

		Year i		
		age-0	age-1+	mat fem
Year i+1	age-0	0.339		
	age-1+	0.464	0.248	
	mat fem	0.420	0.397	0.372

		Year i		
		age-0	age-1+	mat fem
Year i+2	age-0	0.168		
	age-1+	-0.085	0.079	
	mat fem	-0.075	0.144	0.168

Table 15. Annual commercial blue crab landings in Virginia.

Year	Landings			
	Pounds	Lbs x 10 ³	MT	MT x 10 ³
1945	19,203,000	19.203	8710.337	8.710
1946	26,458,000	26.458	12001.150	12.001
1947	34,151,000	34.151	15490.637	15.491
1948	42,203,000	42.203	19142.964	19.143
1949	40,061,000	40.061	18171.369	18.171
1950	46,079,000	46.079	20901.089	20.901
1951	38,679,000	38.679	17544.504	17.545
1952	33,162,000	33.162	15042.034	15.042
1953	32,775,000	32.775	14866.494	14.866
1954	32,205,000	32.205	14807.946	14.608
1955	26,980,000	26.980	12237.926	12.238
1956	24,715,000	24.715	11210.539	11.211
1957	23,748,000	23.748	10771.007	10.771
1958	17,176,000	17.176	7790.905	7.791
1959	18,927,000	18.927	8585.145	8.585
1960	36,768,000	36.768	16677.689	16.678
1961	40,418,000	40.418	18333.302	18.333
1962	49,520,000	49.520	22461.900	22.462
1963	42,950,000	42.950	19481.798	19.482
1964	46,720,000	46.720	21191.842	21.192
1965	45,619,000	45.619	20692.436	20.692
1966	62,961,000	62.961	28558.637	28.559
1967	56,040,000	56.040	25419.324	25.419
1968	45,647,000	45.647	20705.137	20.705
1969	35,665,000	35.665	16177.376	16.177
1970	41,372,000	41.372	18766.029	18.766
1971	47,288,000	47.288	21448.575	21.449
1972	46,795,000	46.795	21225.861	21.226
1973	33,847,910	33.848	15353.158	15.353
1974	37,129,039	37.129	16841.454	16.841
1975	30,502,168	30.502	13835.555	13.836
1976	22,460,778	22.461	10188.040	10.188
1977	34,009,015	34.009	15426.234	15.426
1978	31,619,024	31.619	14342.152	14.342
1979	35,540,172	35.540	16120.755	16.121
1980	31,762,956	31.763	14407.439	14.407
1981	37,228,761	37.230	16887.140	16.887
1982	40,235,124	40.235	18250.350	18.250
1983	39,256,787	39.257	17806.584	17.807
1984	43,047,778	43.048	19526.149	19.526
1985	35,370,850	35.371	16043.952	16.044
1986	31,341,791	31.342	14216.401	14.216
1987	27,501,843	27.502	12474.630	12.475
1988	31,601,117	31.601	14334.030	14.334
1989	38,679,153	38.679	17544.574	17.545
1990	49,331,059	49.331	22376.198	22.376
1991	41,257,141	41.257	18713.930	18.714
1992	21,367,595	21.368	9692.181	9.692
1993	48,744,182	48.744	22109.995	22.110
1994	33,185,265	33.185	15052.587	15.053
1995	30,963,351	30.963	14044.744	14.045
1996	31,988,271	31.988	14509.640	14.510
1997	36,727,978	36.728	16659.535	16.660
1998	30,646,866	30.647	13901.234	13.901
1999	30,202,769	30.203	13699.759	13.700
2000	28,545,309	28.545	12947.938	12.948
2001	24,519,850	24.520	11122.020	11.122
2002	26,079,399	26.079	11829.420	11.829
2003	19,341,210	19.341	8773.028	8.773
Average	35,395,197	35.395	16054.996	16.056
SD	9,413,375	9.413	4269.836	4.270
Min	17,176,000	17.176	7790.905	7.791
Max	62,961,000	62.961	28558.637	28.559
Range	45,785,000	45.785	20767.733	20.768

Table 16. Results of time series analysis of Virginia commercial landings involving transfer function, intervention and combined transfer and intervention models (see Section 5.3 for details of terminology).

Model	Parameter Estimate	p	Residual Error	AIC
Transfer with white noise error	$\mu = 2.40$ (0.095)	<0.0001	0.032	-18.38
	$\beta_0 = 0.158$ (0.043)	0.0002		
Transfer with AR1 error	$\mu = 2.56$ (0.121)	<0.0001	1.12	-21.3
	$\beta_0 = 0.563$ (0.179)	0.0017		
	$\phi_1 = 0.082$ (0.049)	0.098		
Intervention with white noise error	$\mu = 2.77$ (0.039)	<0.0001	0.07	11.99
	$\omega_0 = -0.207$ (0.092)	0.0245		
Intervention with AR1 error	$\mu = 2.82$ (0.067)	<0.0001	0.026	-23.54
	$\phi_1 = 0.531$ (0.168)	0.0016		
	$\omega_0 = -0.274$ (0.121)	0.0231		
Combined with white noise error	$\mu = 2.53$ (0.104)	<0.0001	0.0281	-22.13
	$\beta_0 = 119$ (0.043)	0.0055		
	$\omega_0 = -0.162$ (0.068)	0.0169		
Combined with white noise error	$\mu = 2.65$ (0.124)	<0.0001	0.026	-23.56
	$\phi_1 = 0.430$ (0.189)	0.0234		
	$\beta_0 = 0.073$ (0.049)	0.139		
	$\omega_0 = -0.218$ (0.105)	0.0386		

Table 17. Annual commercial blue crab landings in Maryland 1929 – 2002.

Year	Landings			
	Pounds	Lbs x 10 ⁶	MT	MT x 10 ³
1929	28,099,678	28.100	12745.803	12.746
1930	36,938,783	36.939	16755.155	16.755
1931	33,841,160	33.841	15350.096	15.350
1932	32,939,431	32.939	14941.079	14.941
1933	30,097,129	30.097	13651.832	13.652
1934	15,909,700	15.910	7216.521	7.217
1935	19,821,400	19.821	8990.838	8.991
1936	15,563,100	15.563	7059.305	7.059
1937	18,712,400	18.712	8487.804	8.488
1938	23,597,500	23.598	10703.649	10.704
1939	27,296,300	27.296	12381.397	12.381
1940	16,822,100	16.822	7630.378	7.630
1941	12,811,700	12.812	5811.291	5.811
1942	15,693,700	15.694	7118.545	7.119
1944	18,267,300	18.267	8285.910	8.286
1945	20,170,500	20.171	9149.187	9.149
1946	28,031,700	28.032	12714.969	12.715
1947	28,578,500	28.579	12962.993	12.963
1948	22,482,900	22.483	10198.075	10.198
1949	24,463,300	24.463	11096.369	11.096
1950	30,420,400	30.420	13798.465	13.798
1951	29,198,000	29.198	13243.994	13.244
1952	29,086,800	29.087	13193.554	13.194
1953	28,273,800	28.274	12824.784	12.825
1954	20,182,200	20.182	9154.495	9.154
1955	16,432,500	16.433	7453.659	7.454
1956	23,036,700	23.037	10449.274	10.449
1957	31,838,200	31.838	14441.569	14.442
1958	30,360,700	30.361	13771.386	13.771
1959	23,160,100	23.160	10505.248	10.505
1960	29,855,900	29.856	13542.412	13.542
1961	29,350,300	29.350	13313.076	13.313
1962	31,553,100	31.553	14312.249	14.312
1963	18,836,500	18.837	8544.095	8.544
1964	25,789,600	25.790	11697.969	11.698
1965	33,848,600	33.849	15353.471	15.353
1966	32,110,000	32.110	14564.855	14.565
1967	26,774,200	26.774	12144.576	12.145
1968	10,227,200	10.227	4638.981	4.639
1969	25,070,900	25.071	11371.972	11.372
1970	26,191,800	26.192	11880.404	11.880
1971	27,283,600	27.284	12375.636	12.376
1972	24,568,300	24.568	11143.997	11.144
1973	20,652,700	20.653	9367.910	9.368
1974	24,650,200	24.650	11181.146	11.181
1975	24,821,100	24.821	11258.665	11.259

1976	20,115,800	20.116	9124.376	9.124
1977	21,129,400	21.129	9584.137	9.584
1978	16,892,700	16.893	7662.402	7.662
1979	24,666,200	24.666	11188.403	11.188
1980	24,943,534	24.944	11314.200	11.314
1981	58,958,422	58.958	26743.098	26.743
1982	43,243,515	43.244	19614.934	19.615
1983	51,384,953	51.385	23307.829	23.308
1984	48,770,764	48.771	22122.053	22.122
1985	55,527,179	55.527	25186.712	25.187
1986	46,413,612	46.414	21052.866	21.053
1987	42,647,714	42.648	19344.683	19.345
1988	41,673,423	41.673	18902.752	18.903
1989	42,352,309	42.352	19210.690	19.211
1990	45,094,468	45.094	20454.512	20.455
1991	47,490,656	47.491	21541.405	21.541
1992	30,857,923	30.858	13996.922	13.997
1993	56,820,973	56.821	25773.567	25.774
1994	44,243,040	44.243	20068.311	20.068
1995	41,173,306	41.173	18675.903	18.676
1996	37,020,587	37.021	16792.261	16.792
1997	40,159,876	40.160	18216.218	18.216
1998	25,677,914	25.678	11647.309	11.647
1999	31,570,031	31.570	14319.929	14.320
2000	20,238,872	20.239	9180.201	9.180
2001	22,668,127	22.668	10282.093	10.282
2002	23,842,533	23.843	10814.794	10.815
2003				
Average	27,797,947	27.798	12608.940	12.609
SD	10,157,886	10.158	4607.541	4.608
Min	10,227,200	10.227	4638.981	4.639
Max	58,958,422	58.958	26743.098	26.743
Range	48,731,222	48.731	22104.117	22.104

Table 18. Results of time series analysis of Maryland commercial landings involving transfer function, intervention and combined transfer and intervention models (see Section 5.3 for details of terminology).

Model	Parameter Estimate	p	Residual Error	AIC
Transfer with white noise error	$\mu = 0.103$ (0.526)	0.815	9.416	174.66
	$\beta_0 = 1.034$ (0.154)	<0.0001		
Transfer with AR1 error	$\mu = 0.090$ (0.319)	0.777	7.44	167.84
	$\beta_0 = 1.00$ (0.158)	<0.0001		
	$\phi_1 = -0.475$ (0.159)	0.0028		
Intervention with white noise error	$\mu = -0.111$ (0.676)	0.869	14.64	190.6
	1981 $\omega_0 = 15.54$ (3.88)	<0.001		
	1993 $\omega_1 = -5.59$ (3.88)	0.15		
Intervention with AR1 error	$\mu = -0.259$ (0.388)	0.504	11.03	182.25
	$\phi_1 = -0.575$ (0.158)	0.0003		
	1981 $\omega_0 = 12.88$ (2.96)	<0.001		
	1993 $\omega_1 = -0.732$ (3.18)	0.808		
Combined with white noise error	$\mu = 0.0799$ (0.543)	0.883	9.37	176.32
	$\beta_0 = 0.868$ (0.202)	<0.0001		
	1981 $\omega_0 = 4.49$ (4.04)	0.265		
	1993 $\omega_1 = -3.29$ (3.15)	0.296		
Combined with white noise error	$\mu = -0.135$ (0.297)	0.649	6.61	165.7
	$\phi_1 = -0.598$ (0.155)	0.0001		
	$\beta_0 = 0.789$ (0.171)	<0.0001		
	1981 $\omega_0 = 6.58$ (2.64)	0.0127		
	1993 $\omega_1 = 0.938$ (2.438)	0.7		

Table 19. Reconstructed commercial fishery landings in Chesapeake Bay.

Year	MD Landings	VA landings	MD metric	VA metric	Total metric	MD adjusted	VA adjusted	Total adjusted
1929	28,099,678		12.746		12.746			0.000
1930	36,938,783		16.755		16.755	34.223		34.223
1931	33,841,160		15.350		15.350	32.683		32.683
1932	32,939,431		14.941		14.941	32.139		32.139
1933	30,097,129		13.652		13.652	30.714		30.714
1934	15,909,700		7.217		7.217	24.144		24.144
1935	19,821,400		8.991		8.991	25.783		25.783
1936	15,563,100		7.059		7.059	23.717		23.717
1937	18,712,400		8.488		8.488	25.010		25.010
1938	23,597,500		10.704		10.704	27.091		27.091
1939	27,296,300		12.382		12.382	28.634		28.634
1940	16,822,100		7.631		7.631	23.748		23.748
1941	12,811,700		5.811		5.811	21.794		21.794
1942	15,693,700		7.119		7.119	22.966		22.966
1943						22.831		22.831
1944	18,267,300		8.286		8.286	22.696		22.696
1945	20,170,500	19,203,000	9.149	8.710	17.860	23.424	6.623	30.047
1946	28,031,700	26,458,000	12.715	12.001	24.716	26.855	9.125	35.980
1947	28,578,500	34,151,000	12.963	15.491	28.454	26.968	11.778	38.746
1948	22,482,900	42,203,000	10.198	19.143	29.341	24.068	14.555	38.623
1949	24,463,300	40,061,000	11.097	18.171	29.268	24.832	13.816	38.648
1950	30,420,400	46,079,000	13.799	20.901	34.700	27.399	15.892	43.290
1951	29,198,000	38,679,000	13.244	17.544	30.789	26.709	13.340	40.049
1952	29,086,800	33,162,000	13.194	15.042	28.236	26.524	11.437	37.961
1953	28,273,800	32,775,000	12.825	14.866	27.691	26.020	11.303	37.323
1954	20,182,200	32,205,000	9.155	14.608	23.763	22.215	11.107	33.322
1955	16,432,500	26,980,000	7.454	12.238	19.692	20.379	9.305	29.684
1956	23,036,700	24,715,000	10.449	11.211	21.660	23.239	8.524	31.763
1957	31,838,200	23,746,000	14.442	10.771	25.213	27.097	8.190	35.286
1958	30,360,700	17,176,000	13.772	7.791	21.563	26.292	5.924	32.215
1959	23,160,100	18,927,000	10.505	8.585	19.091	22.890	6.528	29.418
1960	29,855,900	36,768,000	13.543	16.678	30.220	25.793	12.681	38.473
1961	29,350,300	40,418,000	13.313	18.333	31.647	25.428	13.939	39.368
1962	31,553,100	49,520,000	14.312	22.462	36.774	26.292	17.078	43.371
1963	18,836,500	42,950,000	8.544	19.482	28.026	20.389	14.813	35.202
1964	25,789,600	46,720,000	11.698	21.192	32.890	23.408	16.113	39.521
1965	33,848,600	45,619,000	15.354	20.692	36.046	26.929	15.733	42.662
1966	32,110,000	62,961,000	14.565	28.559	43.124	26.005	21.714	47.719
1967	26,774,200	56,040,000	12.145	25.419	37.564	23.450	19.327	42.777
1968	10,227,200	45,647,000	4.639	20.705	25.344	15.809	15.743	31.552
1969	25,070,900	35,665,000	11.372	16.177	27.550	22.407	12.300	34.707
1970	26,191,800	41,372,000	11.881	18.766	30.647	22.781	14.268	37.049
1971	27,283,600	47,286,000	12.376	21.449	33.824	23.141	16.308	39.449
1972	24,568,300	46,795,000	11.144	21.226	32.370	21.774	16.139	37.913
1973	20,652,700	33,847,910	9.368	15.353	24.721	19.863	11.673	31.537
1974	24,650,200	37,129,039	11.181	16.841	28.023	21.541	12.805	34.346

1975	24,821,100	30,502,168	11.259	13.836	25.094	21.484	10.520	32.003
1976	20,115,800	22,460,778	9.125	10.188	19.313	19.215	7.746	26.961
1977	21,129,400	34,009,015	9.584	15.426	25.011	19.539	11.729	31.268
1978	16,892,700	31,619,024	7.663	14.342	22.005	17.483	10.905	28.387
1979	24,666,200	35,540,172	11.189	16.121	27.309	20.874	12.257	33.131
1980	24,943,534	31,762,956	11.314	14.407	25.722	20.864	10.954	31.819
1981	58,958,422	37,229,761	26.744	16.887	43.631	29.579	12.840	42.418
1982	43,243,515	40,235,124	19.615	18.250	37.866	22.315	13.876	36.192
1983	51,384,953	39,256,787	23.308	17.807	41.115	25.873	13.539	39.412
1984	48,770,764	43,047,778	22.122	19.526	41.649	24.552	14.846	39.399
1985	55,527,179	35,370,850	25.187	16.044	41.231	27.482	12.199	39.681
1986	46,413,612	31,341,791	21.053	14.216	35.270	23.213	10.809	34.022
1987	42,647,714	27,501,843	19.345	12.475	31.820	21.370	9.485	30.855
1988	41,673,423	31,601,117	18.903	14.334	33.237	20.793	10.899	31.692
1989	42,352,309	38,679,153	19.211	17.545	36.756	20.966	13.340	34.306
1990	45,094,468	49,331,059	20.455	22.376	42.831	22.075	17.013	39.088
1991	47,490,656	41,257,141	21.542	18.714	40.256	23.027	14.229	37.256
1992	30,857,923	21,367,595	13.997	17.756	31.753	15.347	13.501	28.848
1993	56,820,973	48,744,182	25.774	13.127	38.901	26.989	13.127	40.116
1994	44,243,040	33,185,265	20.069	15.053	35.121	21.149	15.053	36.201
1995	41,173,306	30,963,351	18.676	14.045	32.721	19.621	14.045	33.666
1996	37,020,587	31,988,271	16.793	14.510	31.302	17.603	14.510	32.112
1997	40,159,876	36,727,978	18.217	16.660	34.876	18.892	16.660	35.551
1998	25,677,914	30,646,966	11.648	13.901	25.549	12.188	13.901	26.089
1999	31,570,031	30,202,789	14.320	13.700	28.020	14.725	13.700	28.425
2000	20,238,872	28,545,309	9.180	12.948	22.128	9.450	12.948	22.398
2001	22,668,127	24,519,850	10.282	11.122	21.404	10.417	11.122	21.539
2002	23,842,533	26,079,399	10.815	11.829	22.644	10.815	11.829	22.644
2003		19,341,210	0.000	8.773	8.773	0.000	8.773	8.773

Table 20. Conversion factors, harvest in pounds and harvest in numbers for Maryland 1990 to 2003. Sex-specific mean carapace widths (CW mm) are calculated annually from the Maryland trawl survey. The width is converted to a weight using regression equations derived from Maryland trawl data pooled over 1994 to 2004. The reported harvest in pounds is divided by the weight to estimate the number of crabs harvested. The peeler and soft crab (combined under peelers) is reported in numbers.

Year	Trawl				Reported landings and conversions to individuals								TTL Indiv
	Females		Males		Females		Males		Culls		Peelers		
	CW	Wt lbs	CW	Wt lbs	Pounds	Individuals	Pounds	Individuals	Pounds	Individuals	Pounds	Reported	
1990	150.51	0.32	147.33	0.42	11,745,370	36,406,460	23,699,395	57,070,267	8,233,711	25,521,567	1,416,002	6,796,810	125,795,104
1991	153.30	0.34	149.94	0.43	11,527,369	34,304,598	26,908,809	62,025,634	7,314,456	21,767,280	1,740,036	8,352,171	126,449,683
1992	149.12	0.32	144.49	0.40	9,435,594	29,856,317	15,805,025	39,958,809	4,414,173	13,967,424	1,203,133	5,775,038	89,557,589
1993	151.66	0.33	146.86	0.41	18,962,353	57,793,717	28,925,129	70,215,866	7,110,445	21,671,310	1,823,044	8,750,612	158,431,506
1994	152.10	0.33	146.56	0.41	16,377,036	49,596,893	19,926,218	48,619,569	4,420,628	13,387,614	1,582,313	7,595,103	119,199,179
1995	150.87	0.32	146.42	0.41	15,561,701	47,980,821	20,119,492	49,206,872	3,858,414	11,896,507	1,633,711	7,841,813	116,926,013
1996	147.65	0.31	141.88	0.38	15,969,519	51,655,515	15,562,956	41,190,528	3,744,482	12,112,021	1,743,627	8,369,410	113,327,473
1997	151.08	0.33	143.84	0.39	15,780,444	48,506,133	18,708,030	47,831,906	4,175,668	12,835,223	1,495,733	7,179,520	116,352,782
1998	158.02	0.36	148.95	0.43	9,043,267	25,167,894	12,532,229	29,367,920	2,870,725	7,989,381	1,231,351	5,910,487	68,435,682
1999	145.75	0.30	147.23	0.41	11,749,021	39,112,674	15,088,962	36,399,685	3,247,931	10,812,411	1,484,121	7,123,782	93,448,552
2000	146.64	0.30	147.55	0.42	7,935,916	26,063,355	9,212,751	22,102,289	1,726,567	5,670,440	1,363,638	6,545,463	60,381,547
2001	151.98	0.33	150.93	0.44	7,890,391	23,936,368	10,863,127	24,631,546	2,320,392	7,039,164	1,594,218	7,652,245	63,259,323
2002	142.28	0.28	151.89	0.45	9,025,077	31,703,604	11,426,412	25,505,432	2,215,327	7,782,079	1,175,717	5,643,442	70,634,557
2003	146.79	0.31	143.06	0.39	10,303,242	33,765,278	11,957,654	30,995,332	2,047,264	6,709,193	1,574,375	7,556,999	79,026,803

Table 21. Conversion factors, harvest in pounds and harvest in numbers for Virginia 1990 to 2003. Sex-specific mean carapace widths (CW mm) are calculated annually from the Virginia trawl survey. The width is converted to a weight using regression equations derived from Maryland trawl data pooled over 1994 to 2004. The reported harvest in pounds is divided by the weight to estimate the number of crabs harvested. The peeler and soft crab (combined under peelers) is reported in numbers.

Year	Trawl				Reported landings and conversions to individuals								
	Females		Males		Females		Males		Unclassified		Peelers		TTL Indiv
	CW	Wt lbs	CW	Wt lbs	Pounds	Individuals	Pounds	Individuals	Pounds	Individuals	Pounds	Reported	
1990	146.24	0.30	145.03	0.40	36,151,122	119,450,667	12,299,641	30,806,246			880,296	4,226,097	154,483,010
1991	146.22	0.30	145.23	0.40	29,800,565	98,498,678	10,139,000	25,307,222			1,317,576	6,325,377	130,131,278
1992	147.00	0.31	143.97	0.39	15,590,330	50,927,953	5,304,274	13,532,491			472,991	2,270,720	66,731,164
1993	143.40	0.29	141.63	0.38	36,049,027	124,443,863	14,748,979	39,205,933			1,672,261	8,028,137	171,677,933
1994	134.92	0.25	138.42	0.36	23,669,860	93,647,503	8,060,978	22,703,646			1,454,427	6,982,367	123,333,516
1995	139.37	0.27	141.40	0.37	22,019,762	81,002,263	7,190,787	19,194,225			1,752,802	8,414,796	108,611,284
1996	136.97	0.26	140.96	0.37	22,573,114	86,340,593	7,687,914	20,684,168			1,727,243	8,292,093	115,316,854
1997	137.31	0.26	140.50	0.37	25,528,564	97,090,967	9,084,367	24,644,035			2,115,047	10,153,850	131,888,852
1998	139.02	0.27	140.80	0.37	21,241,142	78,587,817	6,968,577	18,802,232			2,437,247	11,700,658	109,090,706
1999	139.17	0.27	141.84	0.38	21,217,608	78,304,226	6,854,668	18,155,403			2,130,513	10,228,099	106,687,728
2000	139.71	0.27	140.85	0.37	20,975,768	76,743,778	5,521,031	14,883,638			2,048,510	9,834,422	101,461,837
2001	142.52	0.29	142.92	0.38	16,291,748	57,018,282	5,826,752	15,141,282			2,401,350	11,528,325	83,687,889
2002	142.33	0.28	143.70	0.39	16,870,709	59,216,118	7,099,830	18,197,267			2,111,780	10,138,166	87,551,551
2003	143.92	0.29	142.84	0.38	14,359,708	49,170,036	5,179,193	13,476,172			1,599,052	7,676,678	70,322,886

Table 22. Maryland conversion factors, harvest in pounds and harvest in numbers for the Potomac River 1990 to 2003. Sex-specific mean carapace widths (CW mm) are calculated annually from the Maryland trawl survey. The width is converted to a weight using regression equations derived from Maryland trawl data pooled over 1994 to 2004. The reported harvest in pounds is divided by the weight to estimate the number of crabs harvested. The peeler and soft crab (combined under peelers) is reported in numbers.

Year	Trawl				Reported landings and conversions to individuals								
	Females		Males		Females		Males		Culls		Peelers		TTL Indiv
	CW	Wt lbs	CW	Wt lbs	Pounds	Individuals	Pounds	Individuals	Pounds	Individuals	Pounds	Reported	
1989	154.55	0.34	153.922	0.46	2,020,566	5,906,378	3,257,819	7,036,409			43,182	207,307	13,150,094
1990	150.51	0.32	147.329	0.42	1,975,720	6,124,028	3,185,512	7,670,998			63,350	304,129	14,099,155
1991	153.30	0.34	149.938	0.43	2,221,810	6,611,942	4,954,579	11,420,457			47,233	226,755	18,259,154
1992	149.12	0.32	144.489	0.40	2,202,288	6,968,528	3,550,815	8,977,294			56,973	273,514	16,219,336
1993	151.66	0.33	146.856	0.41	3,060,175	9,326,843	4,414,605	10,716,472			74,588	358,080	20,401,394
1994	152.10	0.33	146.555	0.41	2,707,467	8,199,404	3,207,907	7,827,229			56,332	270,437	16,297,070
1995	150.87	0.32	146.417	0.41	1,579,989	4,871,521	2,394,825	5,857,100			74,532	357,811	11,086,431
1996	147.65	0.31	141.878	0.38	2,386,307	7,718,826	3,224,573	8,534,488			76,928	369,313	16,622,627
1997	151.08	0.33	143.844	0.39	3,612,923	11,105,448	5,358,824	13,701,217			89,041	427,465	25,234,131
1998	158.02	0.36	148.947	0.43	1,688,190	4,698,322	3,506,240	8,216,494			92,401	443,596	13,358,412
1999	145.75	0.30	147.201	0.41	1,806,878	6,015,126	3,404,770	8,216,801			77,132	370,293	14,602,220
2000	146.64	0.30	147.55	0.42	821,184	2,696,955	1,217,055	2,919,834			82,978	398,358	6,015,147
2001	151.98	0.33	150.933	0.44	959,184	2,909,790	1,411,931	3,201,476	381,197	1,156,403	54,189	260,149	7,527,818
2002	142.28	0.28	151.89	0.45	1,044,756	3,670,055	1,506,324	3,362,337	304,957.6	1,071,266	31,910	153,193	8,256,850
2003	146.79	0.31	143.06	0.39	813,918	2,667,332	1,161,242	3,010,046			30,116	144,580	5,821,958

Table 23. Bay wide catch in individuals, estimated abundance from the winter dredge survey, the exploitation fraction (U) and fishing mortality rate (F) estimated at 4 levels of M.

Year	Catch (individuals)	N	U	F at M=0.375	F at M=0.6	F at M=0.9	F at M=1.2
1990	297,491,349	791,044,135	0.376	0.56	0.6620	0.7752	0.8999
1991	278,855,948	827,688,010	0.337	0.51	0.5721	0.6672	0.7717
1992	176,385,097	366,841,146	0.481	0.83	0.9439	1.1197	1.3161
1993	349,273,842	852,086,834	0.41	0.66	0.7458	0.8765	1.0213
1994	249,239,144	487,203,597	0.512	0.91	1.0414	1.2413	1.4656
1995	245,661,766	486,534,636	0.505	0.89	1.0197	1.2140	1.4319
1996	248,129,457	661,467,667	0.375	0.58	0.6597	0.7724	0.8967
1997	286,817,477	679,597,887	0.422	0.68	0.7774	0.9150	1.0676
1998	199,249,200	352,764,088	0.565	1.07	1.2320	1.4826	1.7665
1999	218,558,608	308,051,773	0.709	1.66	1.9814	2.4873	3.0777
2000	171,526,393	281,291,465	0.61	1.22	1.4203	1.7264	2.0762
2001	155,137,452	253,616,340	0.612	1.23	1.4290	1.7379	2.0908
2002	166,544,504	314,961,329	0.529	0.96	1.0997	1.3146	1.5564
2003	153,789,531	334,370,622	0.46	0.77	0.8820	1.0432	1.2227

Table 24. Statistics on bushels of crabs by market category measured at blue crab dealers in Maryland during a study by Stagg et al. (1988).

variable	month	Market category							
		#1 Males		# 2Males		Females		Mixed	
		mean	SE	mean	SE	mean	SE	mean	SE
number of crabs per bushel	July	89.2	1.6	109.4	1.3	121.5	n/a	104.3	2.2
	Aug	93.7	1.3	114.3	1.3	113	n/a	115	2
	Sept	88	1.8	106	2.2	104.7	n/a	101.1	2
	Oct	91.9	2.2	107.8	1.9	113.2	n/a	97.7	3.1
	total	89.5	1.5	109.2	2.2	115	2.3	101.5	3
bushel weight	July	40.2	0.2	39.6	0.2	41.5	n/a	36.7	2.4
	Aug	41.8	0.2	39.2	0.2	40.6	n/a	39.1	2.2
	Sept	38.8	0.2	37	0.3	36	n/a	35.6	2.1
	Oct	40.3	0.3	37.2	0.3	37.7	n/a	32.5	3.1
	total	40.3	0.4	38.7	0.6	39.2	0.7	36	1
carapace width, mm	July	150.2	0.26	136	0.19	147.3	0.24	136.8	0.33
	Aug	146.1	0.19	133	0.16	149.2	0.23	135.1	0.27
	Sept	147.2	0.26	136	0.29	145.6	0.25	137.9	0.43
	Oct	147.4	0.34	136	0.26	146.8	0.25	143.5	0.61
	total	146.71	0.13	134.52	0.1	146.01	0.11	136.81	0.15
crab weight lb	July	0.42	0.005	0.34	0.003	0.32	n/a	0.33	0.02
	Aug	0.42	0.004	0.33	0.002	0.34	n/a	0.32	0.018
	Sept	0.43	0.005	0.34	0.004	0.32	n/a	0.34	0.017
	Oct	0.42	0.007	0.33	0.004	0.32	n/a	0.32	0.026
	total	0.42	0.006	0.33	0.004	0.32	n/a	0.33	0.007

Table 25. Statistics on bushels of crabs by market category measured at blue crab dealers in Maryland during a study by Sharov and Volstad (2001).

variable	Month	market category							
		#1 Males		#2 Males		Females		Mixed	
		mean	SE	mean	SE	mean	SE	Mean	SE
number of crabs per bushel	June	97.75	3.77	n/a	n/a	106.33	3.87	84	11
	July	87.07	2.49	122.58	1.5	94.03	2.57	86	7.19
	Aug	89.6	2.04	103.96	1.7	106.92	2.37	105.56	5.85
	Sept	76.51	2.89	98.33	1.8	102.25	2.17	107.66	4.88
	Oct	88.85	2.14	96.5	0.9	95.75	1.74	90.39	7.58
	overall	86.77	1.47	93.73	3.46	98.07	1.91	93.16	2.69
bushel weight	June	n/a	n/a	n/a	n/a	N/a	n/a	n/a	n/a
	July	n/a	n/a	41.5	n/a	N/a	n/a	n/a	n/a
	Aug	36.29	1.09	n/a	n/a	38.5	0.71	n/a	n/a
	Sept	n/a	n/a	n/a	n/a	N/a	n/a	n/a	n/a
	Oct	40.5	1.84	n/a	n/a	43	0.75	n/a	n/a
	overall	37.09	1.46	41.5	1.5	40.5	1.38	n/a	n/a
carapace width, mm	June	154.6	0.68	n/a	n/a	155.54	0.71	158.12	0.7
	July	156.9	0.39	139.9	0.41	156.48	0.45	152.46	0.97
	Aug	151.6	0.36	141.58	0.61	158.19	1.68	153.33	0.57
	Sept	158	0.54	140.65	0.55	159.04	0.85	151.97	0.38
	Oct	163.2	0.44	145.4	0.45	159.06	0.49	151.22	0.62
	overall	156.27	0.13	143.59	0.17	154.5	0.12	154.47	0.16
crab weight lb	July	n/a	n/a	0.29	n/a	N/a	n/a	n/a	n/a
	Aug	0.37	0.03	n/a	n/a	0.31	0	n/a	n/a
	Sept	n/a	n/a	n/a	n/a	N/a	n/a	n/a	n/a
	Oct	0.54	0.04	n/a	n/a	0.43	0	n/a	n/a
	overall	0.41	0.03	0.29	0.01	0.37	0.03	n/a	n/a

Table 26. Statistics on bushels of crabs by market category measured at blue crab dealers in Maryland during a study by MD DNR (2004).

Variable	Month	1 Males		2 Males		Females	
		Mean	SE	Mean	SE	Mean	SE
Average Number of Crabs per Bushel	May	105.55	2.22			112.33	29.28
	June	85.42	2.81	110.50	7.23	120.25	4.82
	July	72.62	2.43	105.71	6.82	104.88	5.80
	Aug	83.28	3.82	98.00		119.33	17.85
	Sept	81.50	4.93	117.00	4.00	106.42	3.78
	Oct	66.00	4.02	101.50	7.50	91.25	6.16
Average Bushel Weight	May	37.82	0.52			34.78	8.32
	June	39.85	0.36	40.16	1.47	41.37	1.10
	July	38.54	0.38	41.24	0.79	41.66	0.58
	Aug	37.66	0.77	32.60		41.68	2.41
	Sept	38.91	0.83	39.68	0.37	38.93	1.02
	Oct	33.79	0.85	36.77	4.25	36.76	2.47
Average CW	May	149.38	0.68			152.15	1.74
	June	153.48	0.61	139.10	1.10	151.94	1.00
	July	160.49	0.66	141.94	1.17	158.45	1.20
	Aug	153.97	0.75	136.40	0.61	160.55	1.65
	Sept	159.54	0.98	139.41	0.82	160.39	0.92
	Oct	170.05	1.76	144.40	1.01	158.00	1.42

Table 27. Results from the catch multiple survey model applied to the Chesapeake Bay blue crab stock.

M	μ denominator	Likelihood	Process error	SSq Exploitation deviations	MD trawl			VIMS		WDS			
					Relative selectivity	Observation as P(total variance)	Adult catchability	Relative selectivity	Observation as P(total variance)	Adult catchability	Relative selectivity	Observation as P(total variance)	Adult catchability
0.375	n	-40.26	0.103	15.182	0.3	0.65	0.03121	0.3	0.65	0.0429332	0.5	0.4	0.126656
0.6	n	-38.31	0.0770084	3.8622	0.3	0.65	0.0226309	0.3	0.65	0.031235	0.5	0.4	0.0829461
0.9	n	-35.19	0.0656968	0.94371	0.3	0.65	0.016971	0.3	0.65	0.0233974	0.5	0.4	0.0583285
1.2	n	-27.15	0.0557927	0.89073	0.3	0.65	0.00766771	0.3	0.65	0.0105047	0.5	0.4	0.024103

Table 28 Biological Reference Points developed from IBM SPR.

M	μ 10%	μ 20%	Overfished threshold (Crabs x 10 ⁶)
0.375	0.875	0.685	89.9
0.6	0.64	0.575	116.7
0.9	0.53	0.455	166.4
1.2	0.42	0.365	378.4

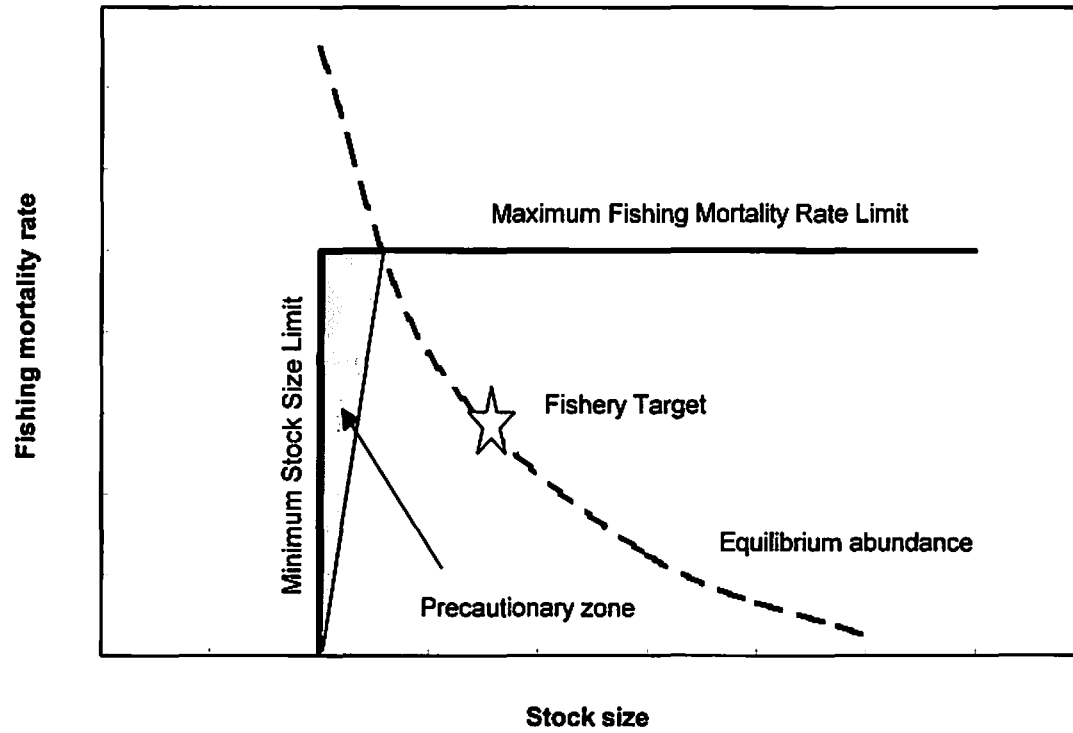


Figure 1. BBCAC control rules adopted for the Chesapeake Bay blue crab fisheries

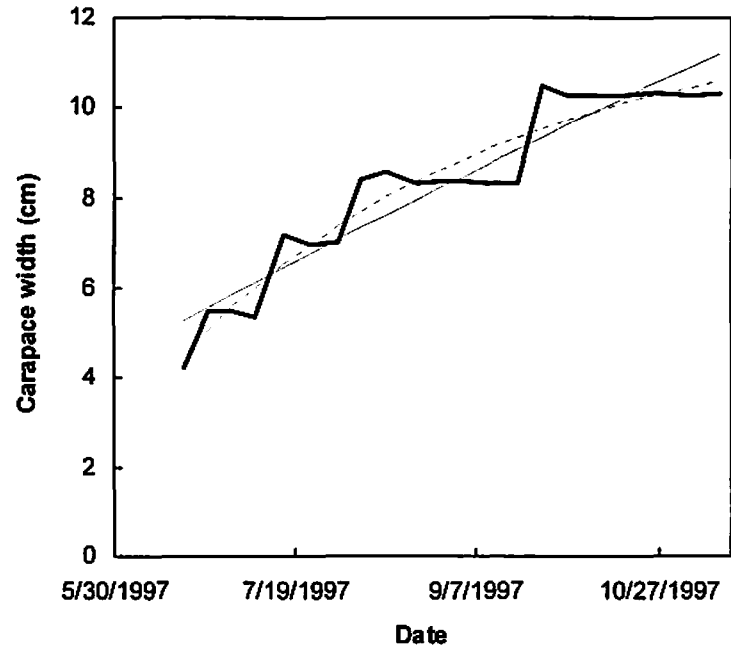


Figure 2. Observed growth trajectory of an individual crab and associated von Bertalanffy fits

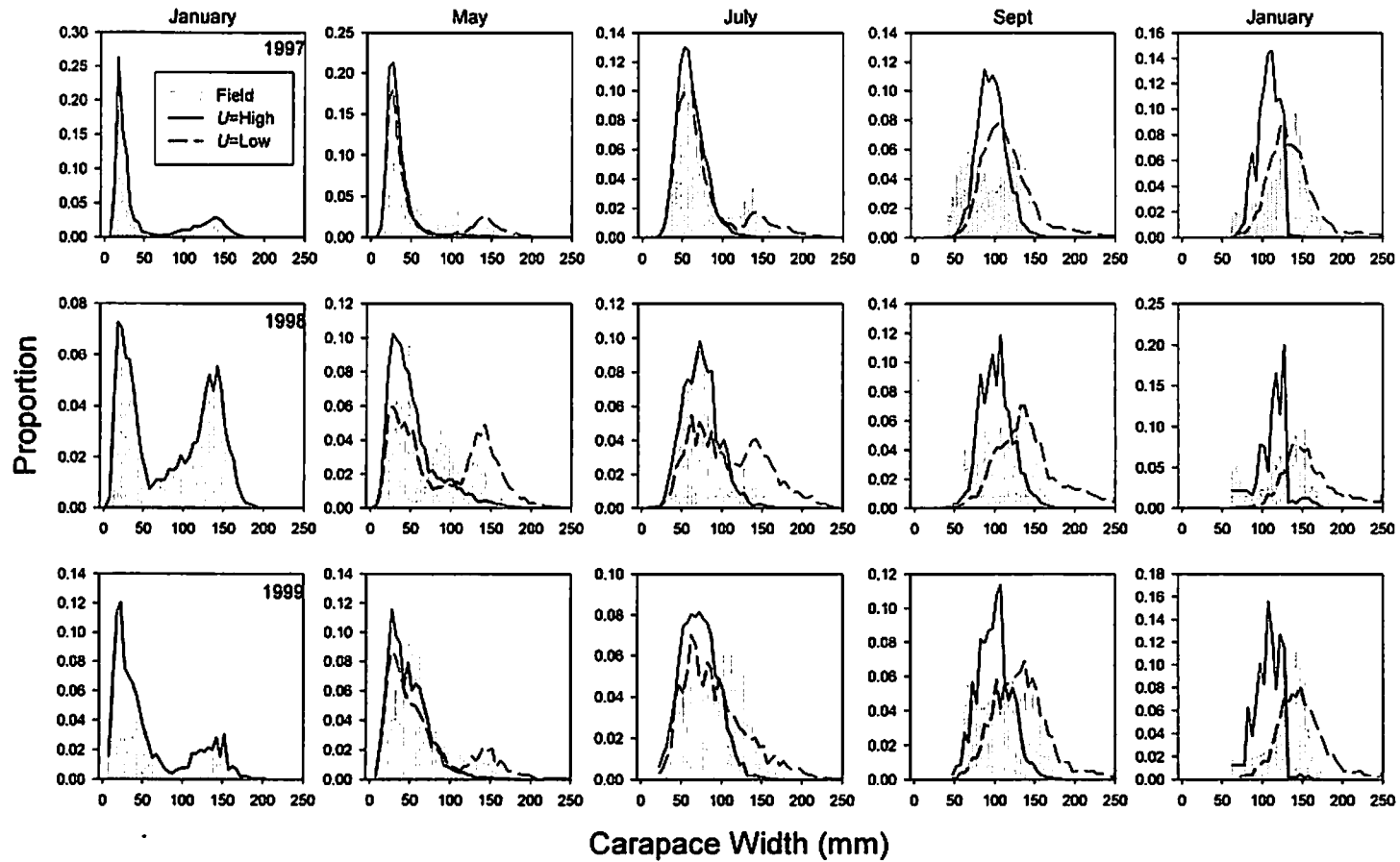


Figure 3. Comparison of observed size frequency distributions of blue crab in the Chesapeake Bay (solid) with size frequency distributions predicted in an individual-based model (Bunnell and Miller, in press).

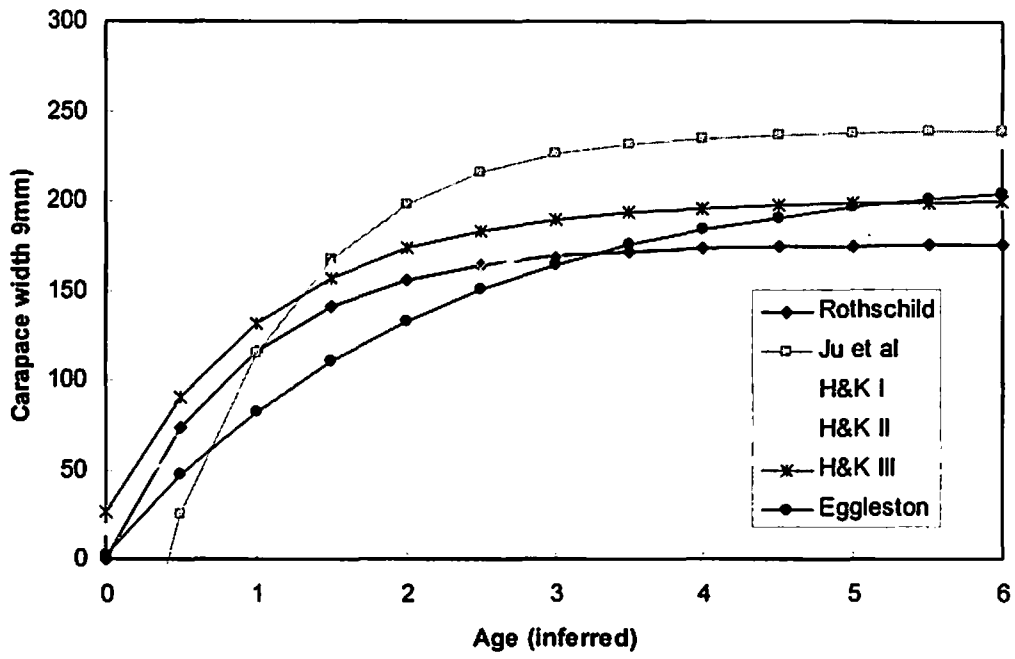


Figure 4. Published von Bertalanffy growth models for blue crab. Details of each study are provided in Section 2.3

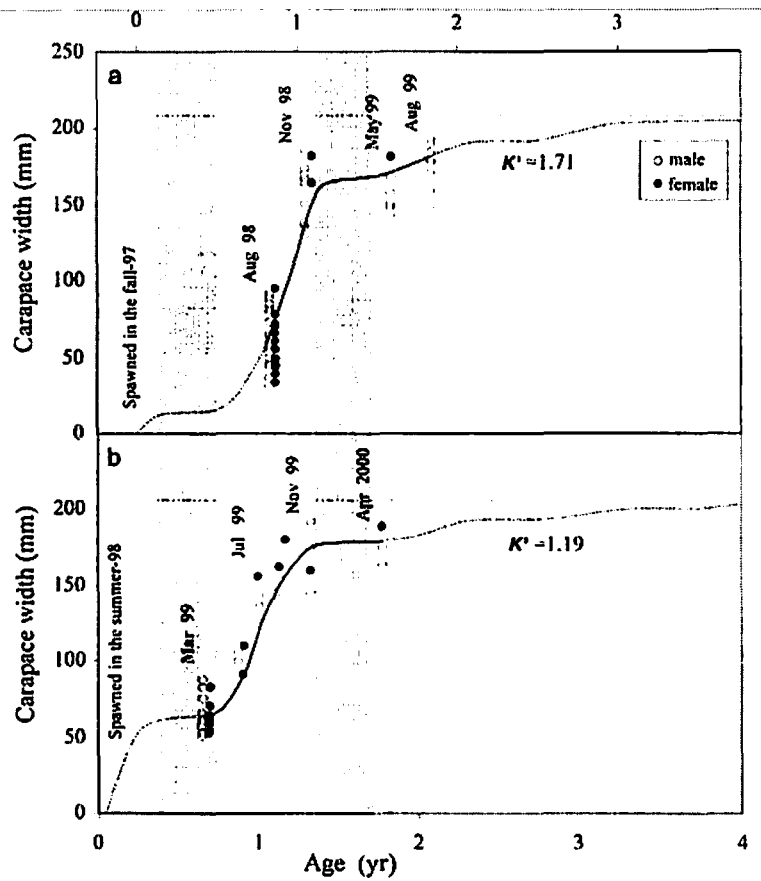


Figure 5. Seasonal form of the von Bertalanffy growth form developed for blue crab raised in aquaculture ponds (Ju et al., 2001). For details of the study, see Section 2.3

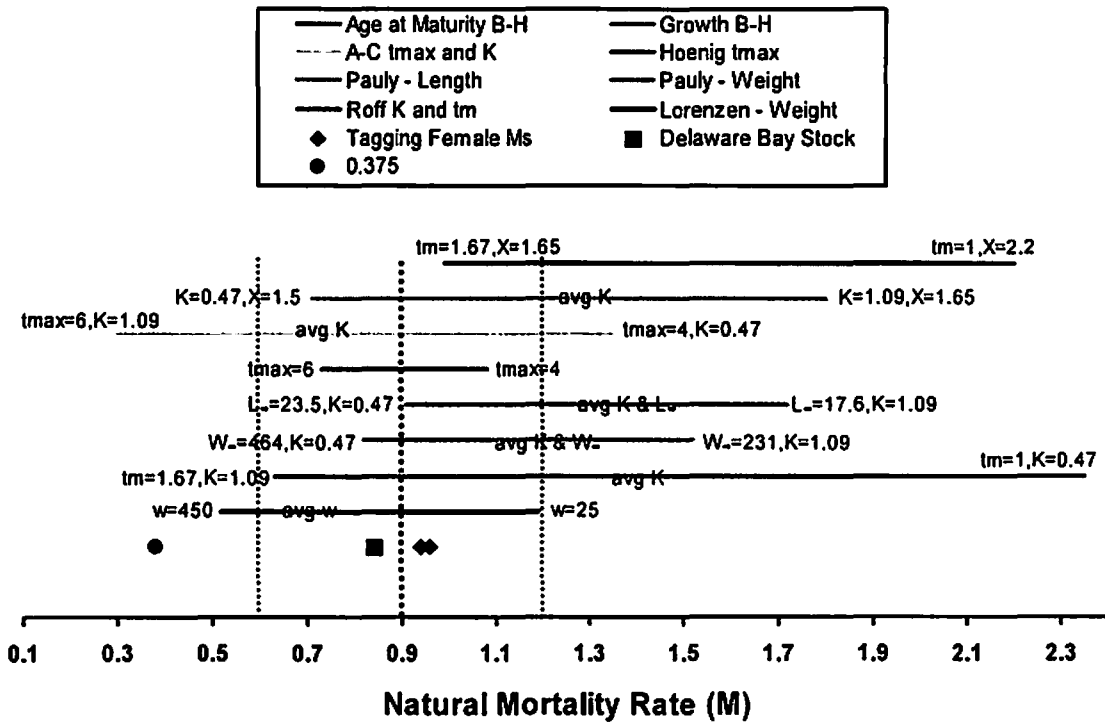


Figure 6A. Estimated ranges of M for blue crab given by the predictive approaches shown in Table 3. Landmark values are indicated by the overlaid text associated with the ranges. The vertical dashed lines indicate the range of values chosen for the assessment; 0.9 was selected as the most likely value. The square is the direct estimate of M for the Delaware Bay stock (0.84) and the diamonds are the direct estimates for adult females in the Chesapeake Bay in 2002 (0.94) and 2003 (0.96). For comparison, the circle is the estimate used by Rugolo et al. (1997; 1998), estimated by $M=3/\text{lifespan}$ with a lifespan of 8 years.

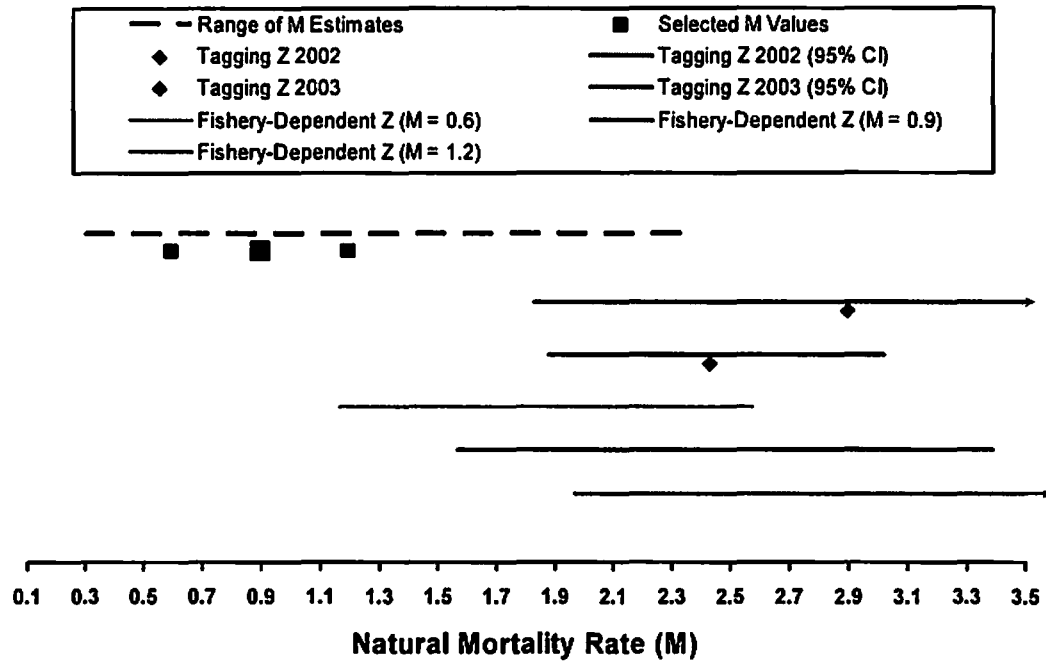


Figure 6B. Comparison of the selected range of values for M with various estimates of Z.

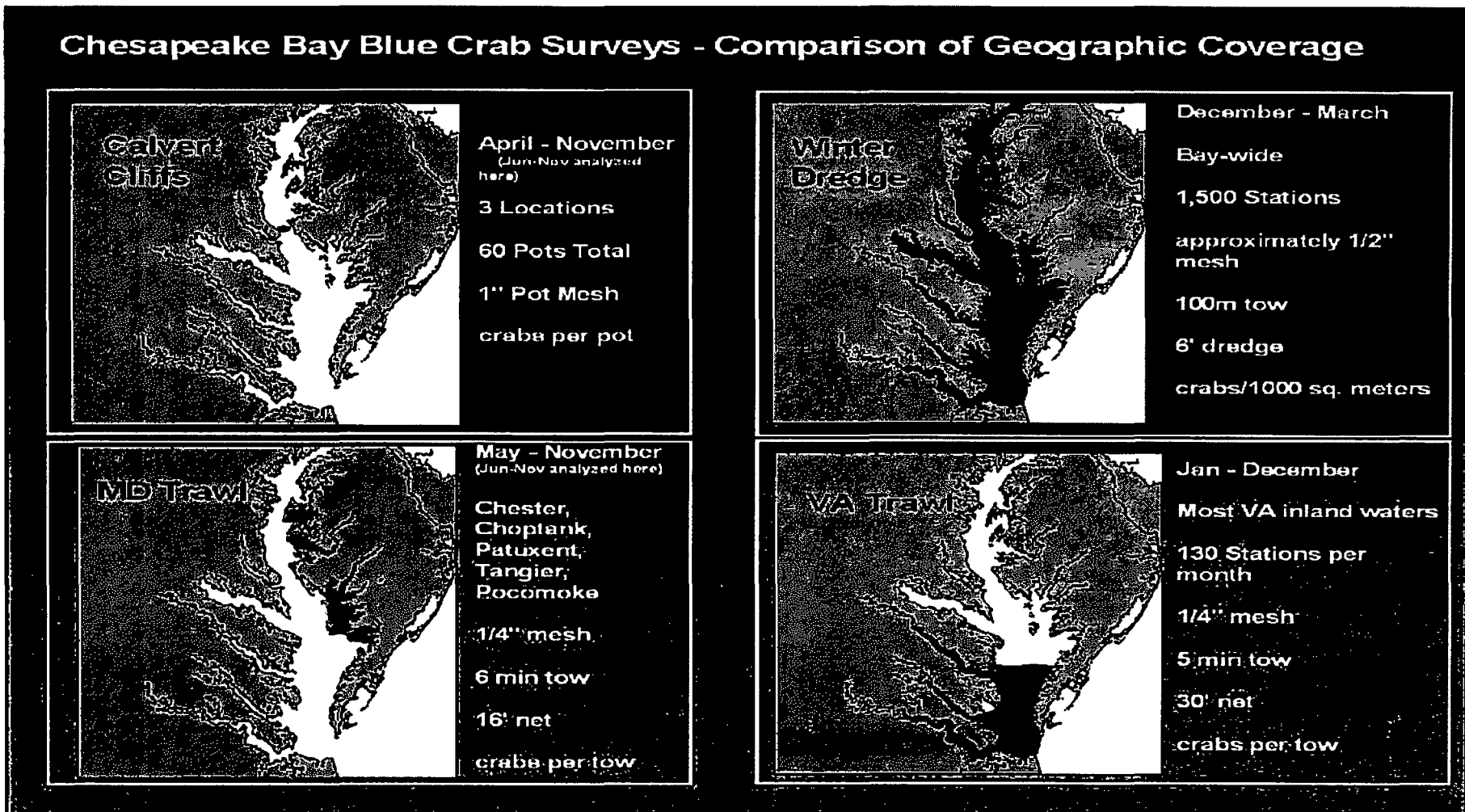


Figure 7. Coverage maps of the four principal fishery-independent surveys used in the assessment.

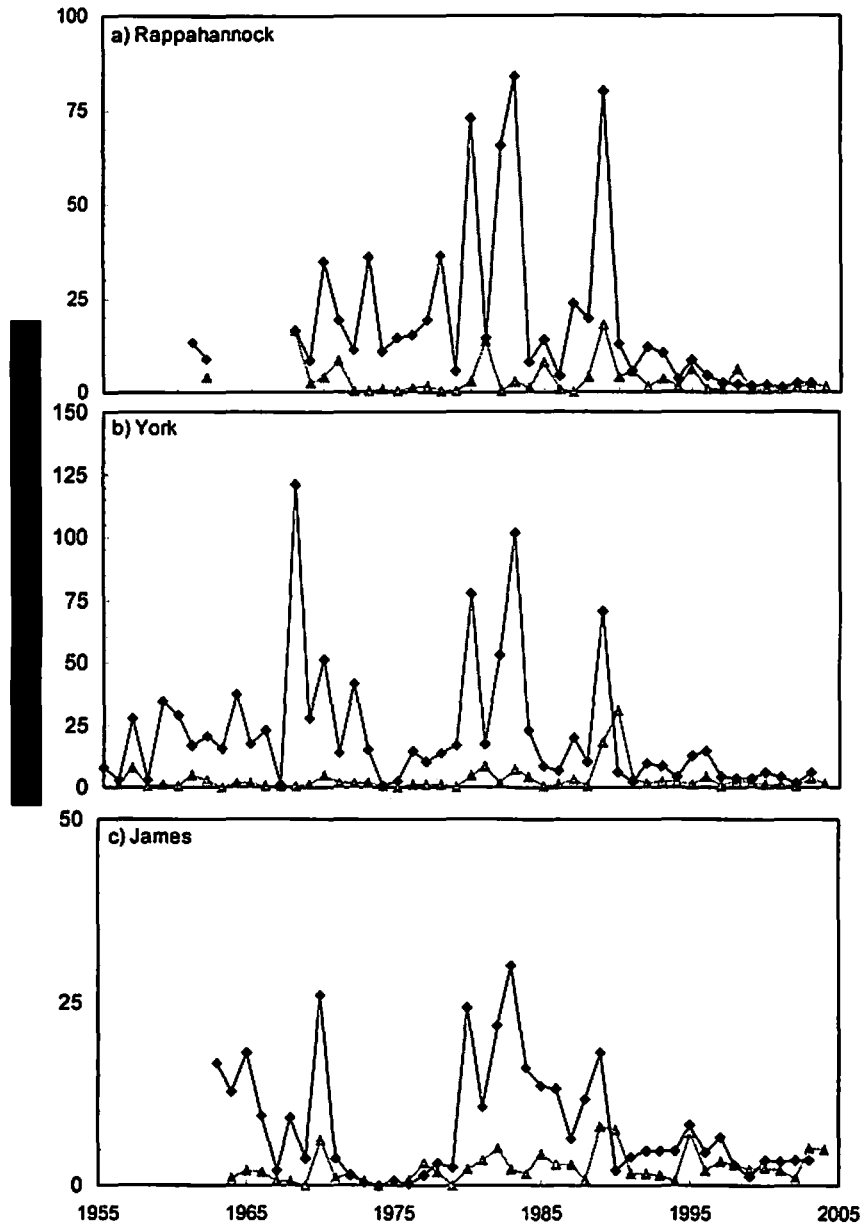


Figure 8. Time series of abundance of age-0 blue crabs in the principal western shore tributaries in the Virginia Trawl Survey for spring (blue, lagged for the year of settlement) and fall (red).

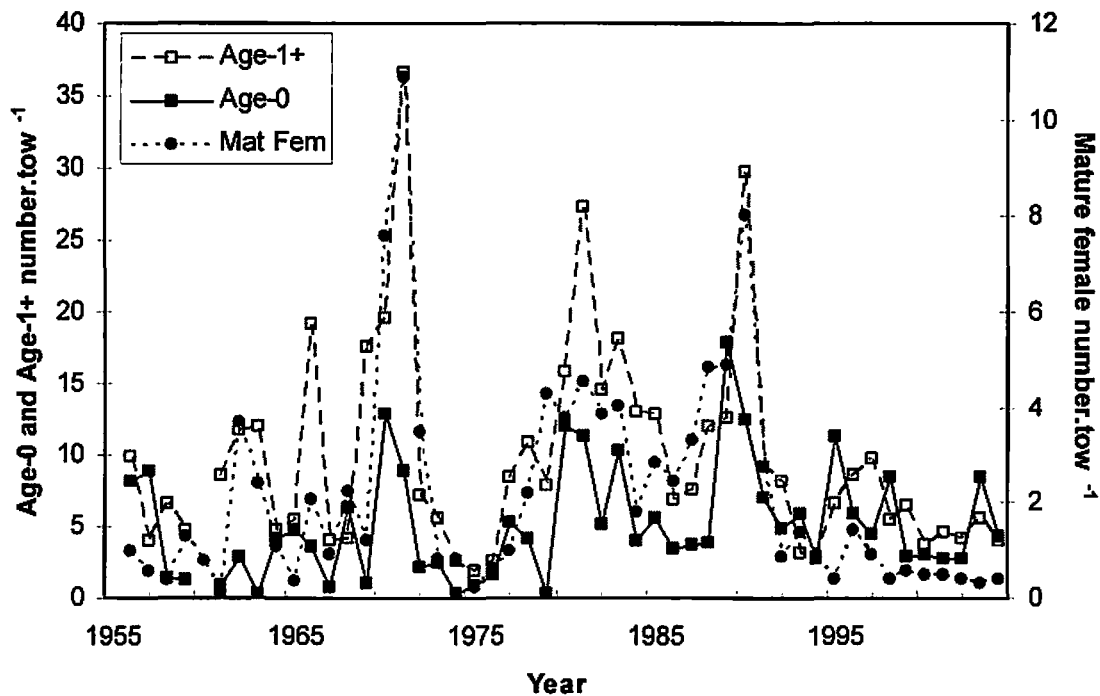


Figure 9A. Time series of abundance of blue crab of different life history stages in the Virginia Trawl Survey

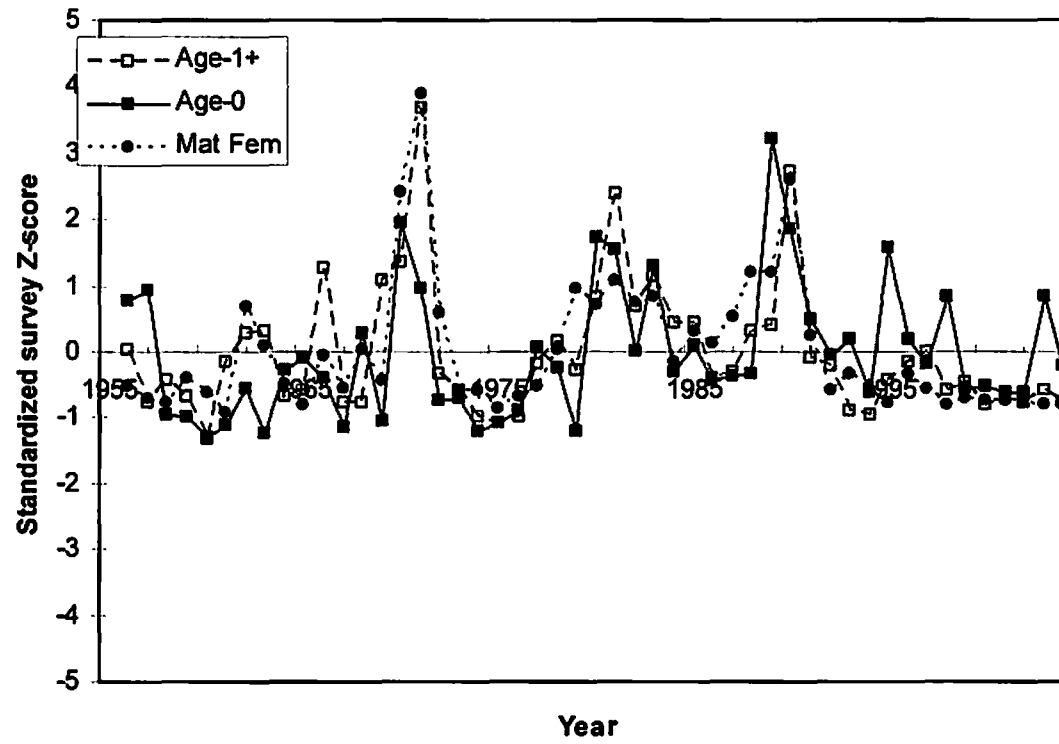


Figure 9B. Time series of standardized CPUE for the Virginia Trawl Survey.

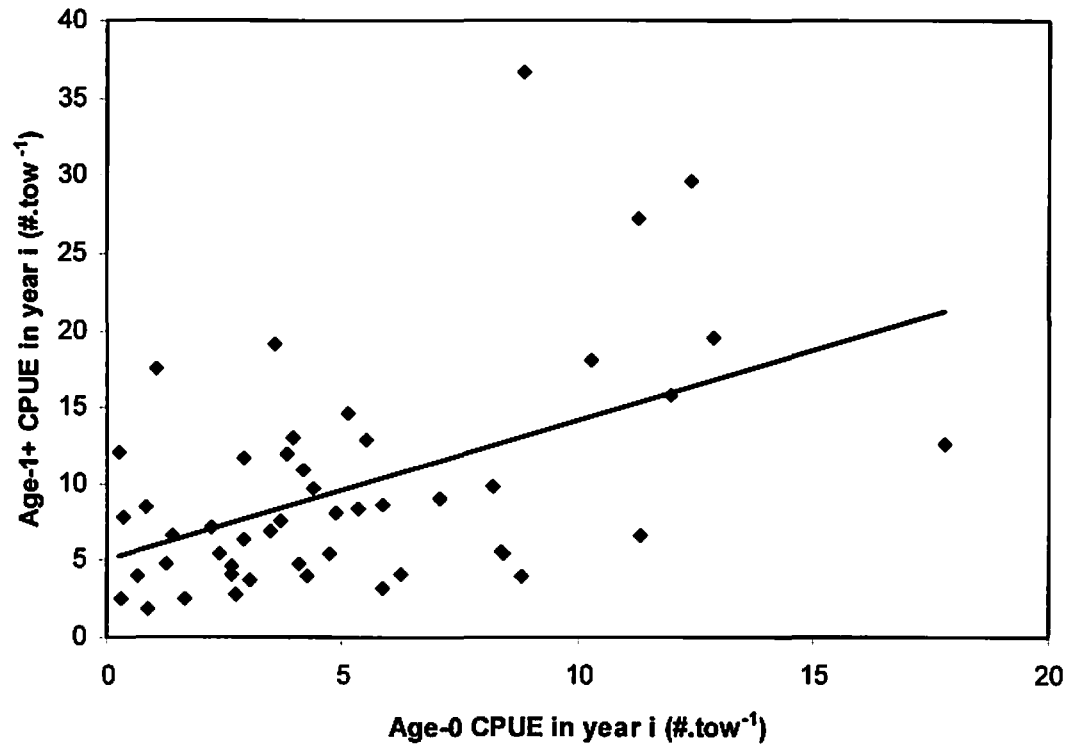


Figure 10. Relationship between abundance of age-0 crabs in year i and age-1+ crabs in year i+1 in the VIMS trawl survey.

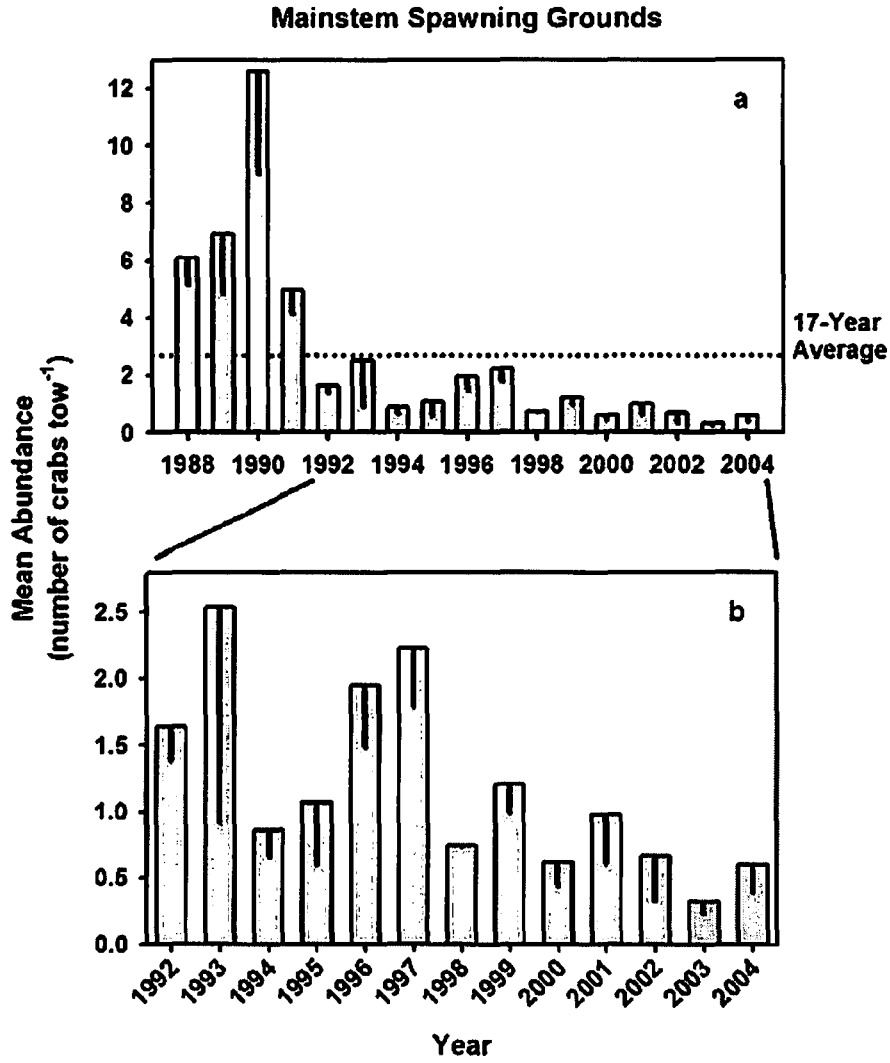


Figure 10. Time series of spawning stock abundance estimated from Virginia Trawl Survey catches on the spawning grounds. For details see Lipcius and Stockhausen (2002)

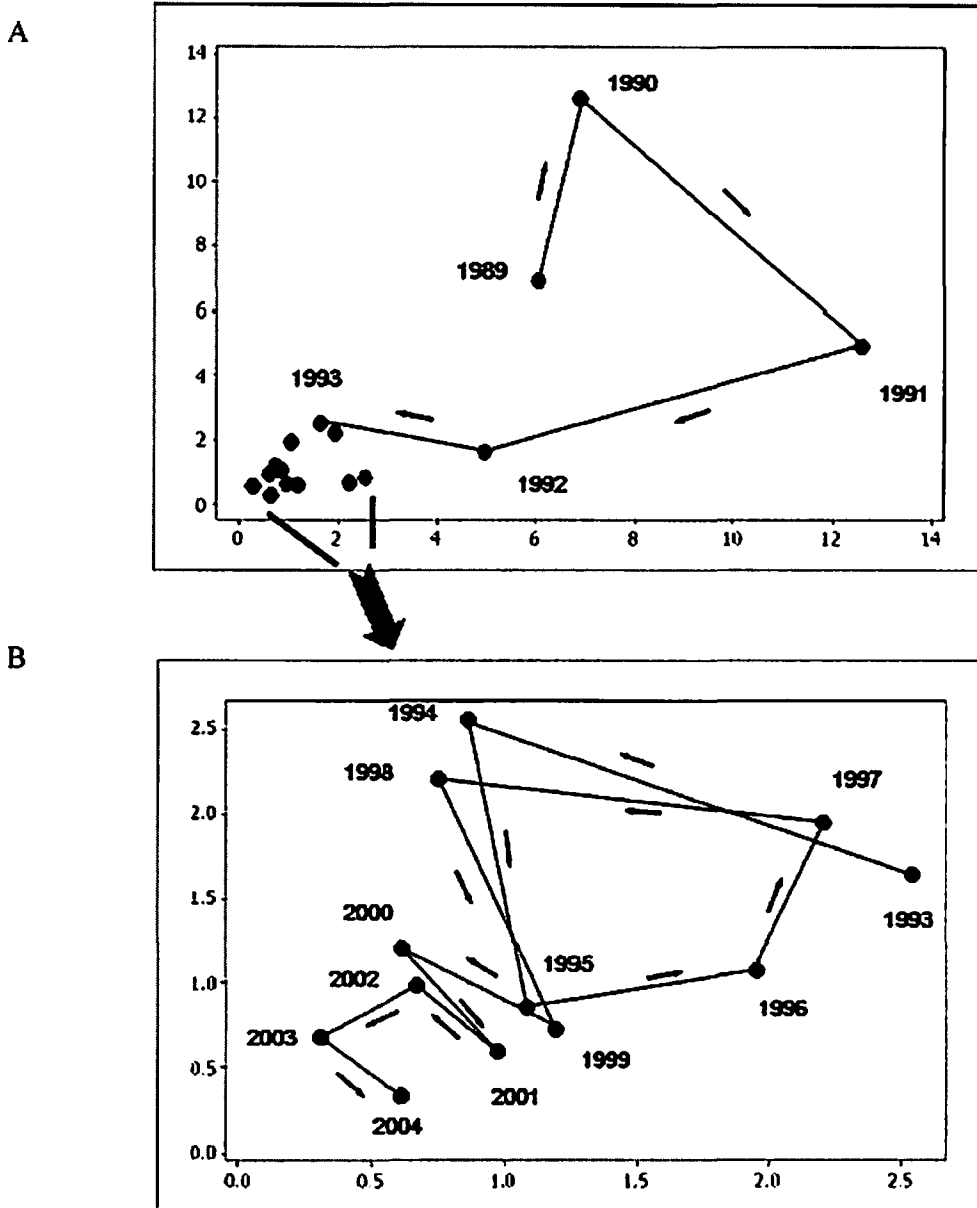


Figure 11. Phase plots of spawning stock abundances from A) 1989-2004, and B) 1994-2004).

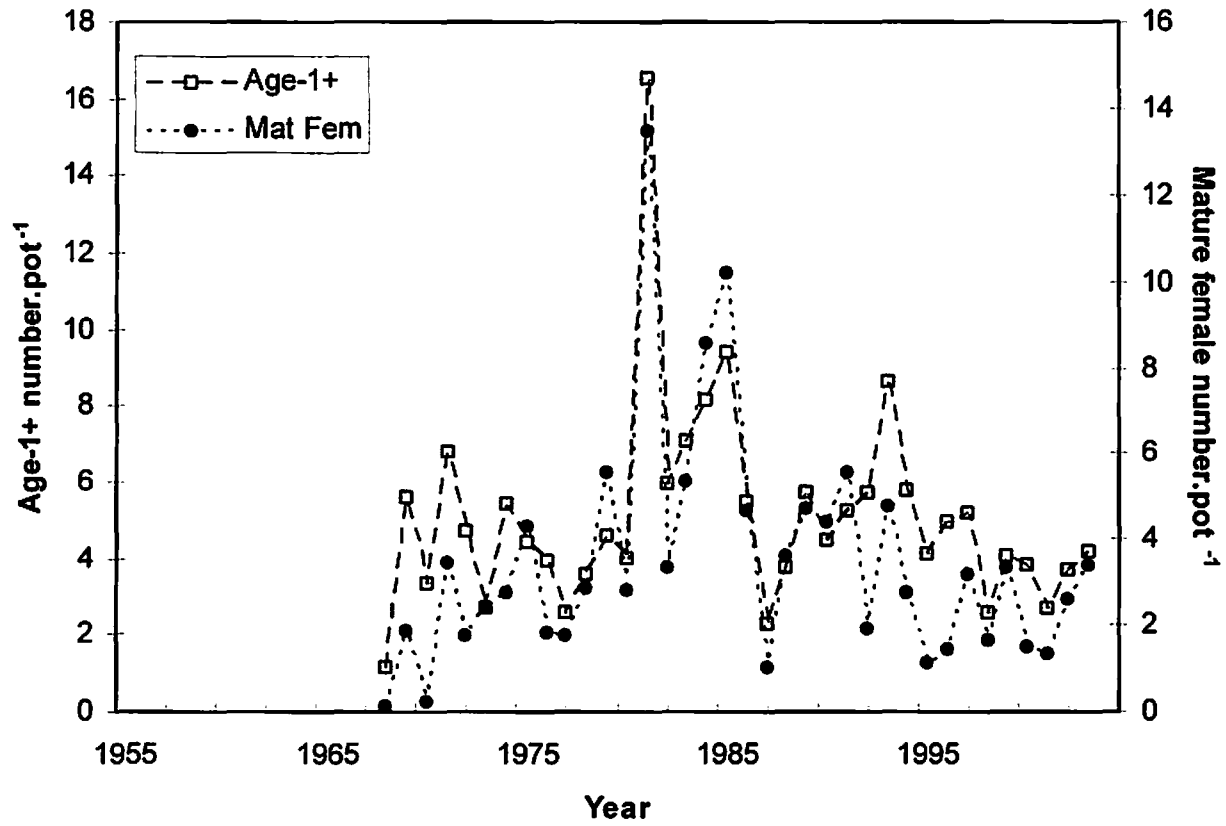


Figure 12A. Time series of age-1+ and mature female abundance estimated in the Calvert Cliffs pot survey.

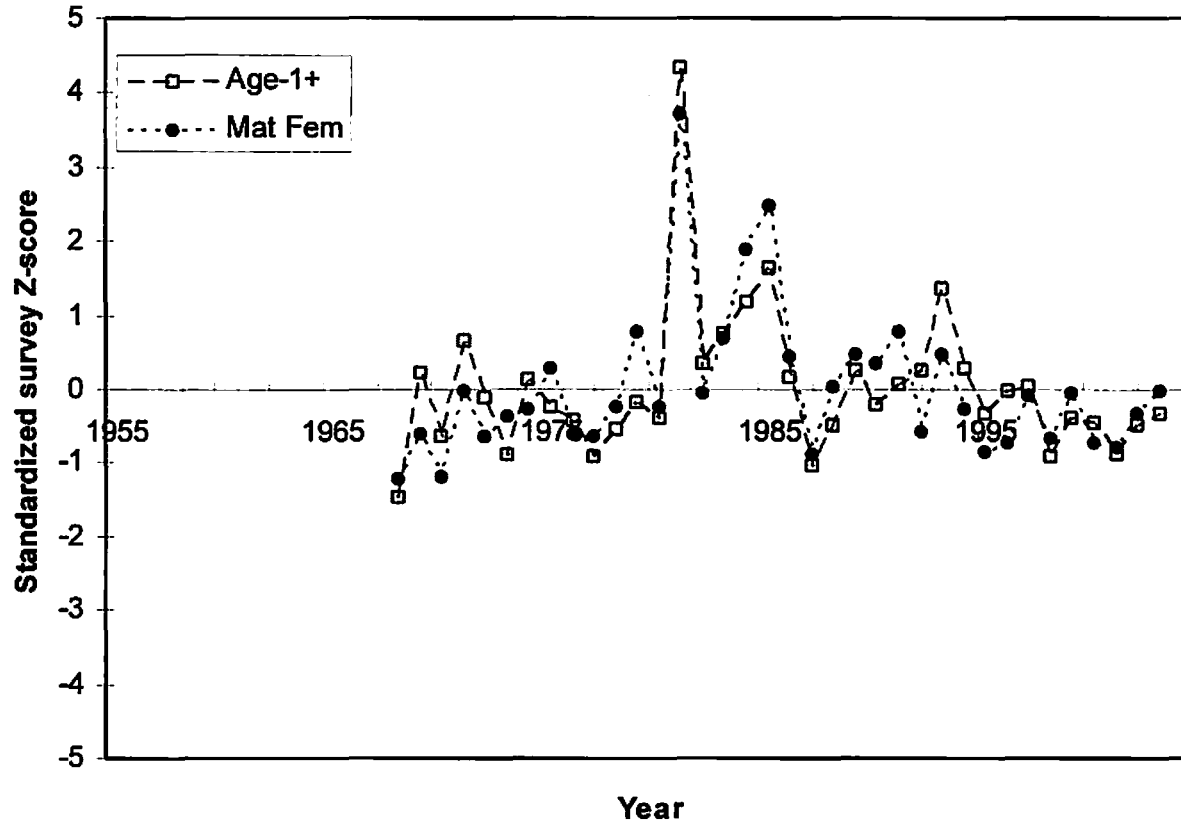


Figure 12B. Standardized time series of age-1+ and mature female abundances estimated from the Calvert Cliffs pot survey.

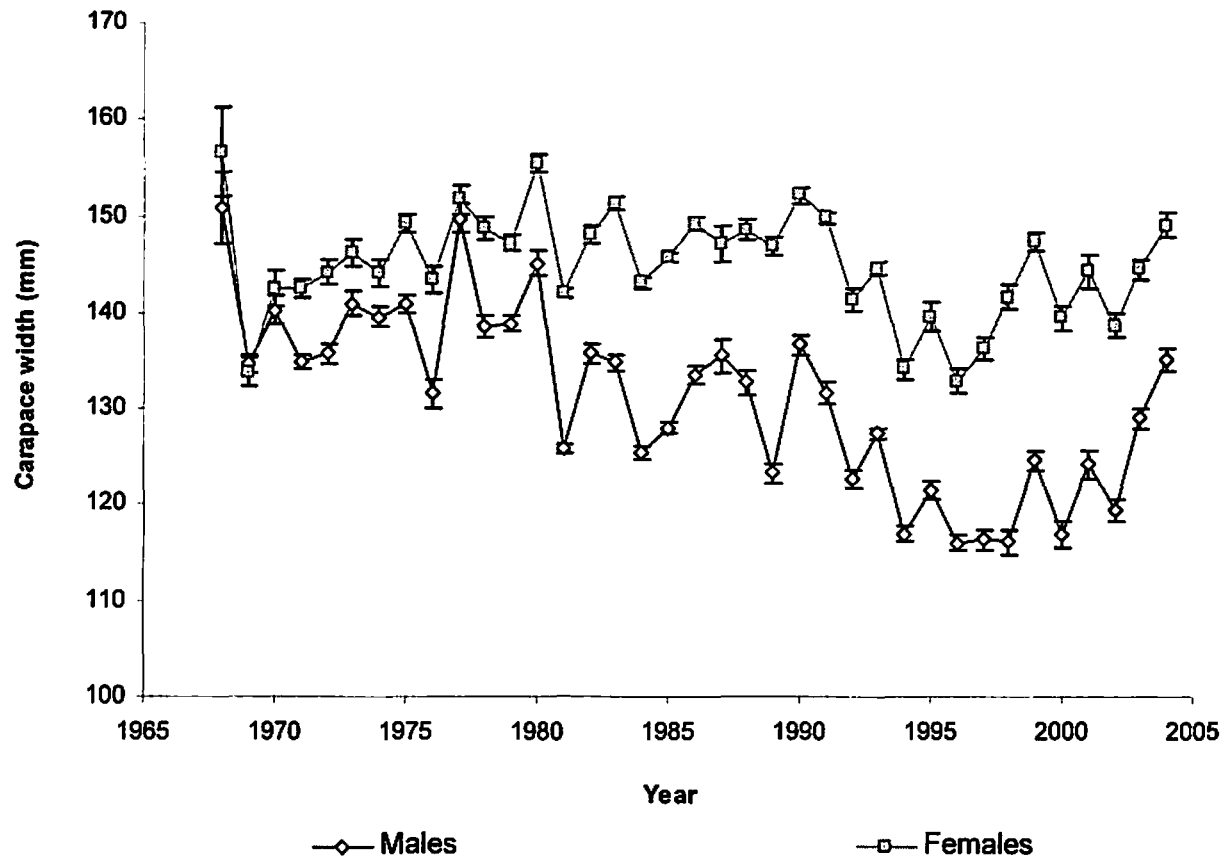


Figure 13. Average carapace width of blue crabs captured in the Calvert Cliffs pot survey from 1968 - 2004



Figure 14. Percentage of the total catch in the Calvert Cliffs pot survey that is > 5" by sex.

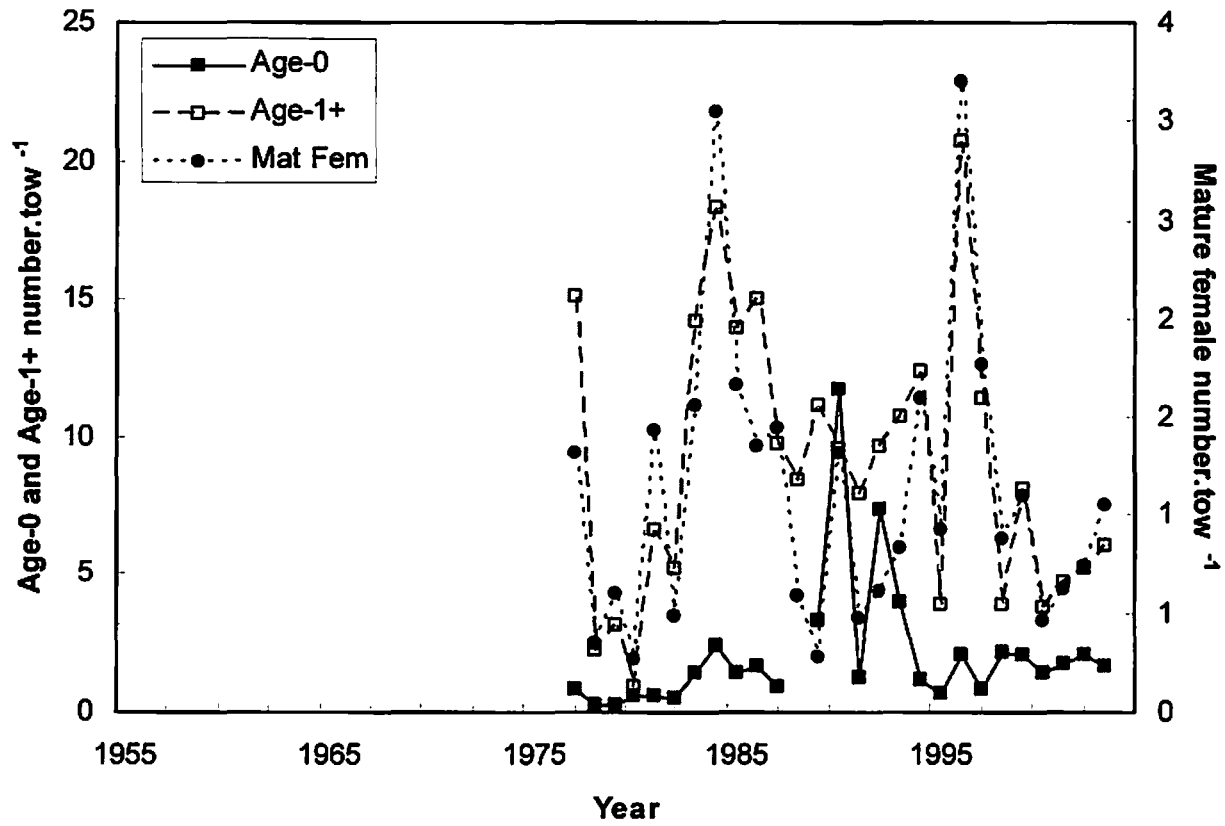


Figure 15A. Time series of age-0, age-1+ and mature female crabs from the Maryland trawl survey.

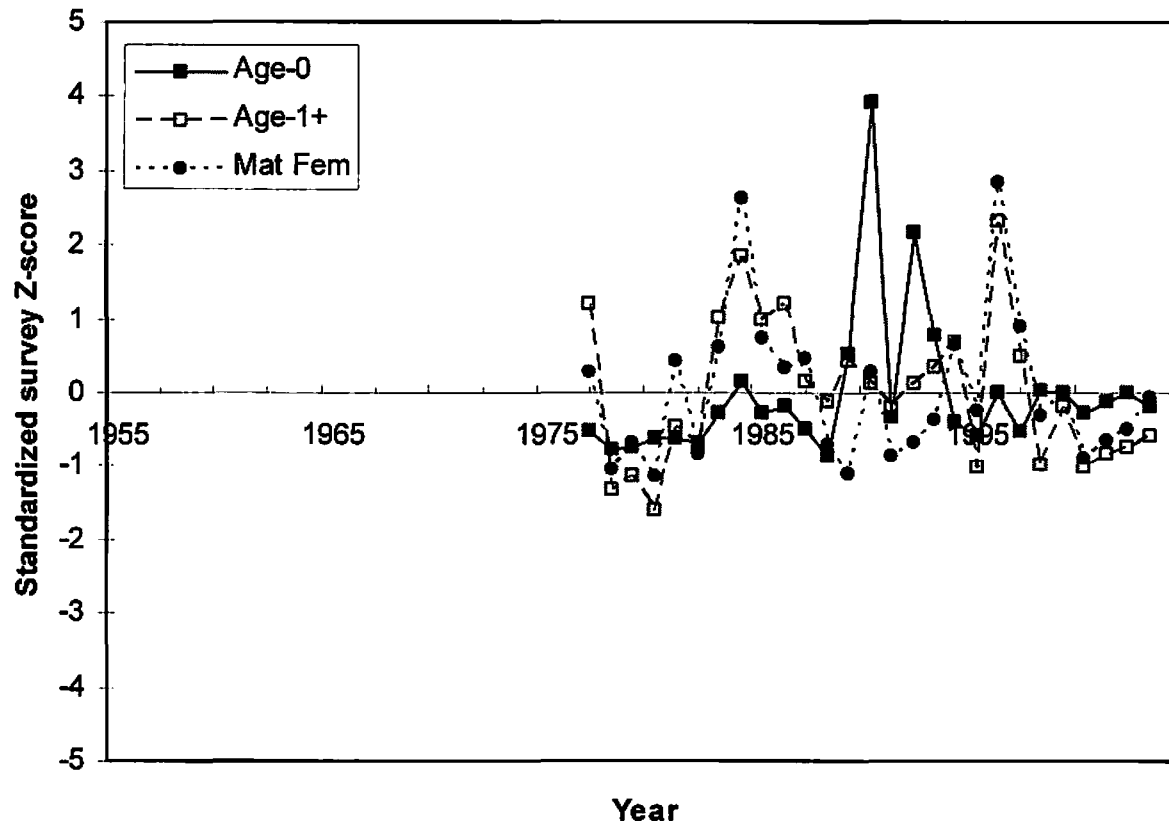


Figure 15B. Time series of standardized abundance of age-0, age-1+ and mature female crabs from the Maryland trawl survey

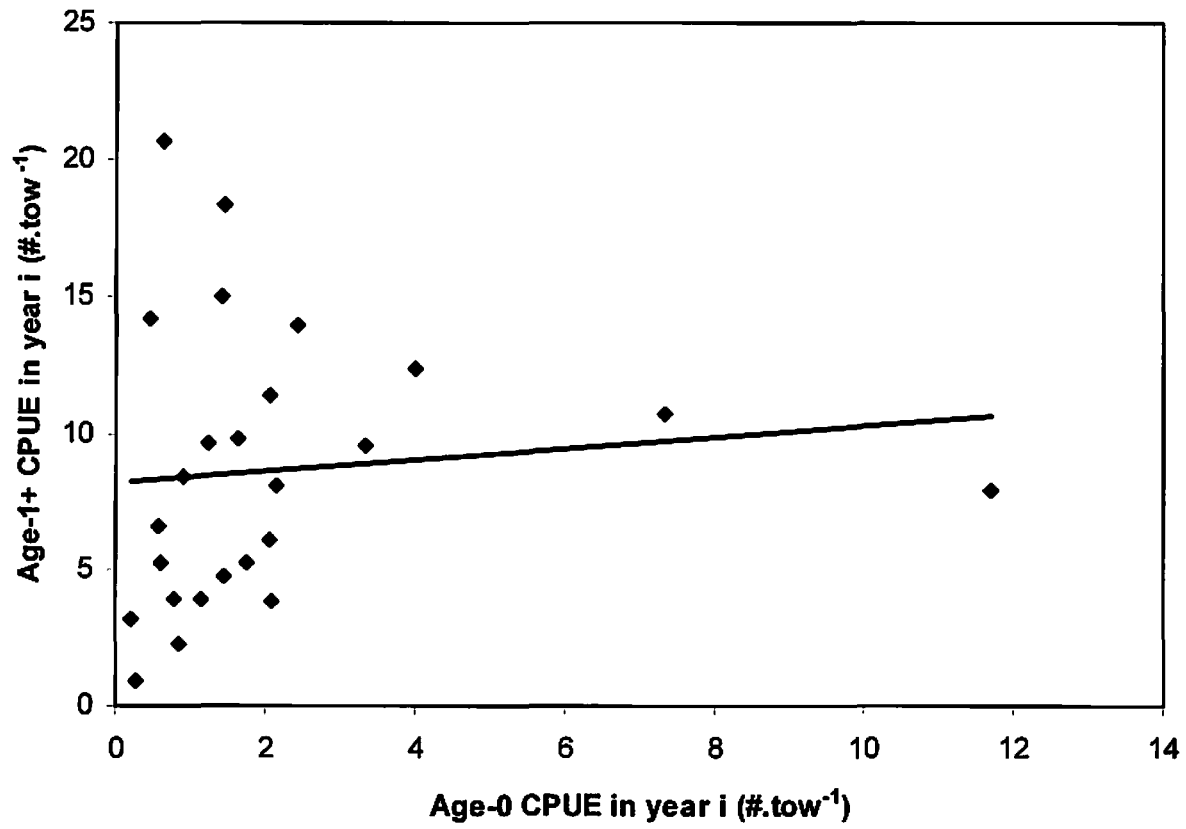


Figure 16. Relationship between abundance of age-0 crabs in year i with age-1+ crabs in year i+1 in the Maryland trawl survey.

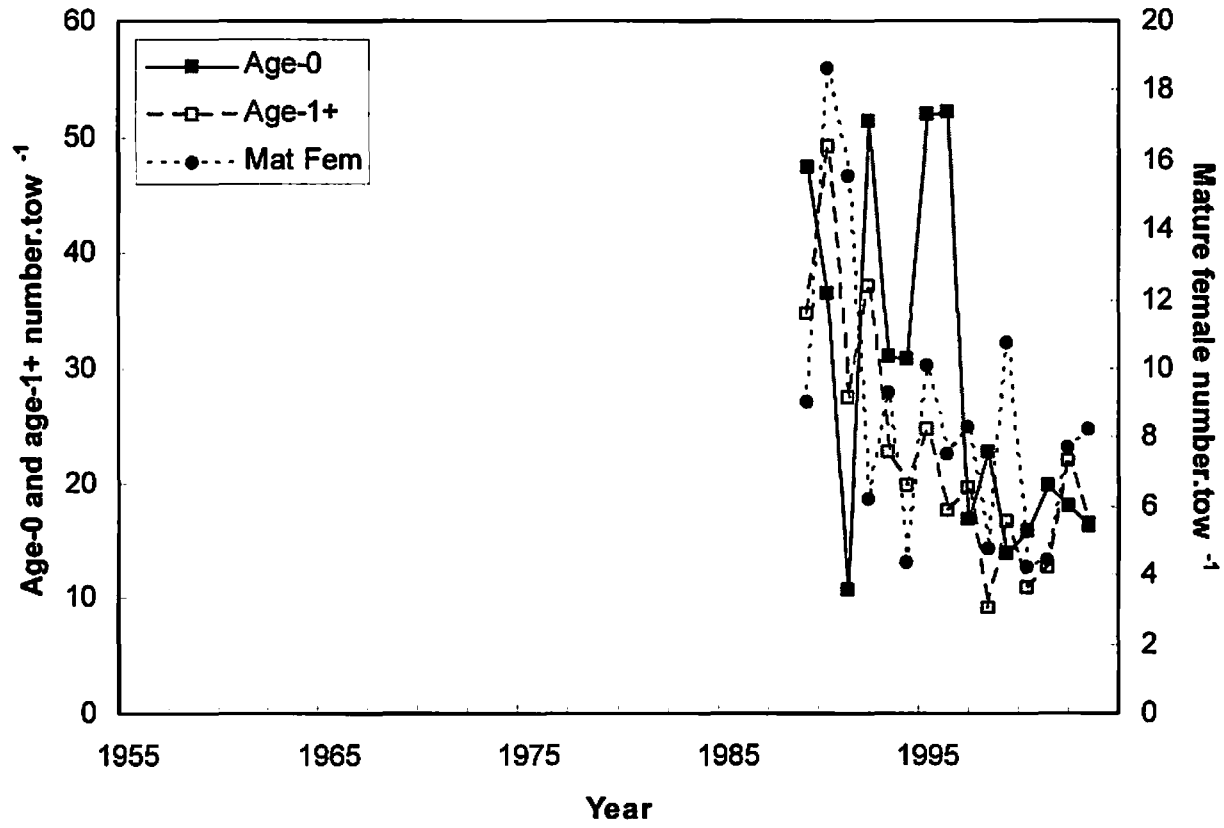


Figure 17A. Time series of catch per unit effort for age-0, age-1+ and mature female crabs in the winter dredge survey.

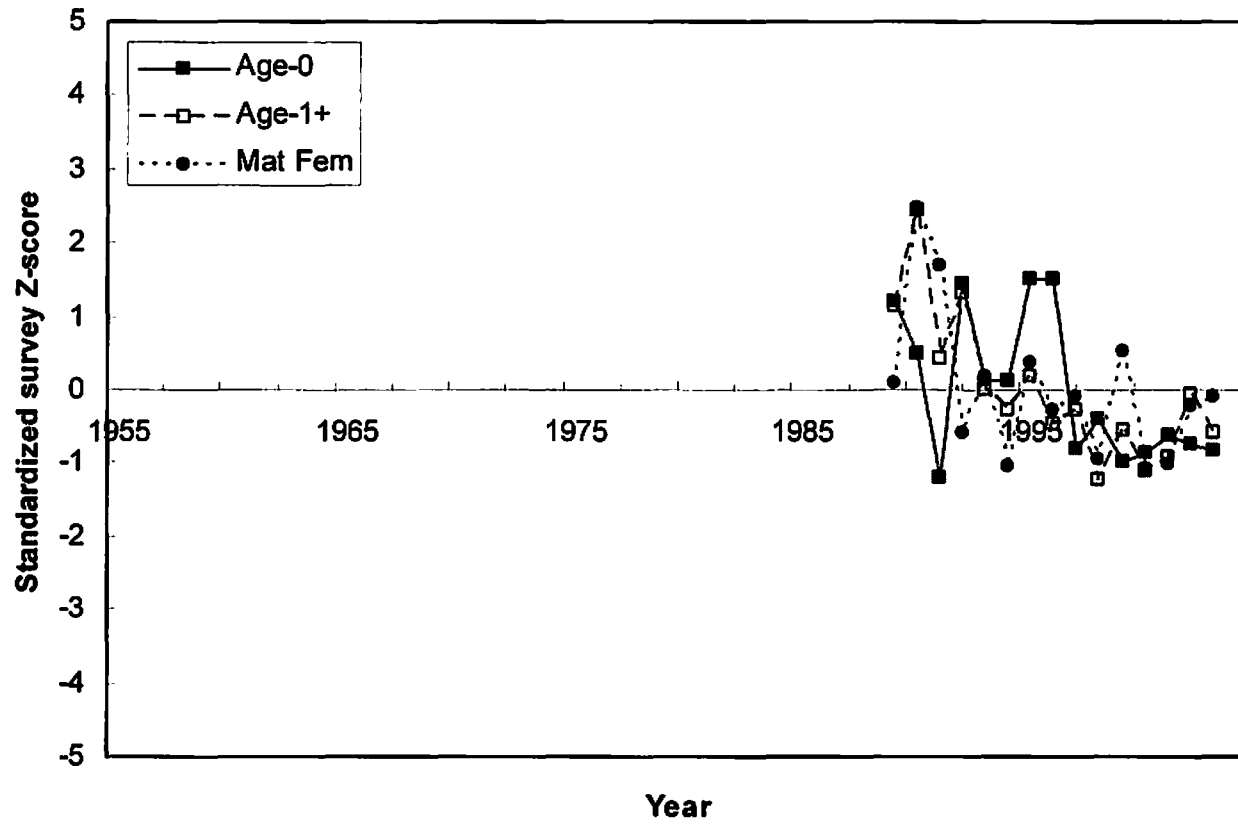


Figure 17B. Standardized residual catch.1000m2 in the winter dredge survey.

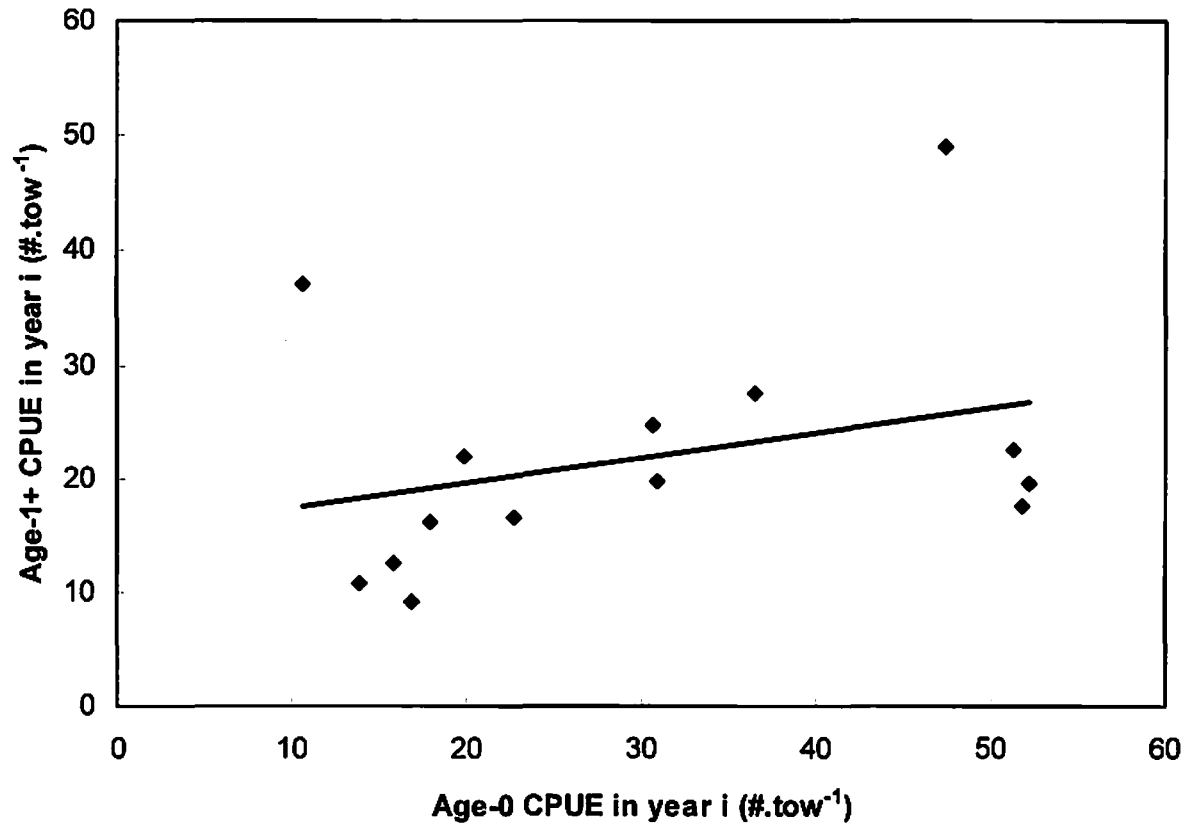


Figure 18. Relationship between abundance of age-0 crabs in year i with age-1+ crabs in year i+1 in the winter dredge survey.

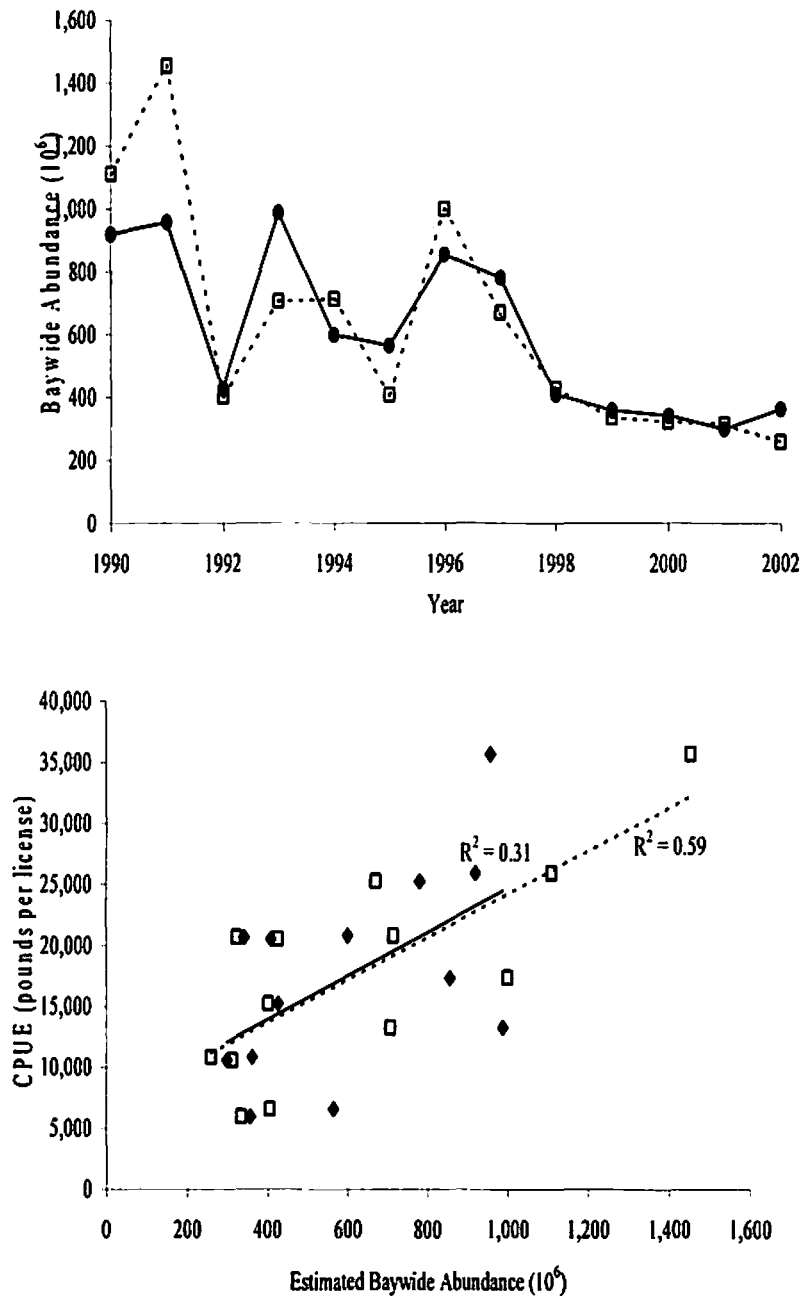
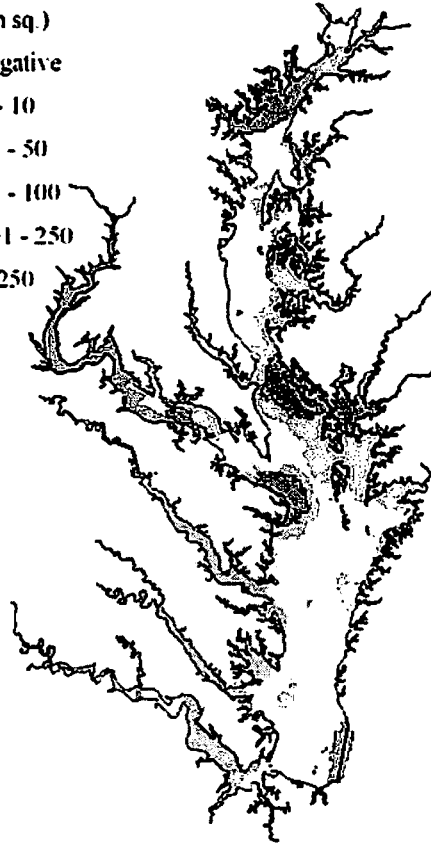
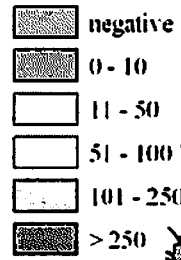


Figure 19. A) Time series of absolute abundance estimates developed by a design-based approach (solid symbols) and a geostatistical analysis (open symbols) and B) the relationship between commercial catch per license in the dredge fishery and both estimates of absolute abundance.

a. Blue crab density
(#/1000 m sq.)



b. Standard Error

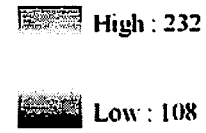


Figure 20. Distribution of blue crabs in winter 1990/1991. Maps are predictions of absolute abundance (left) and precision (right).

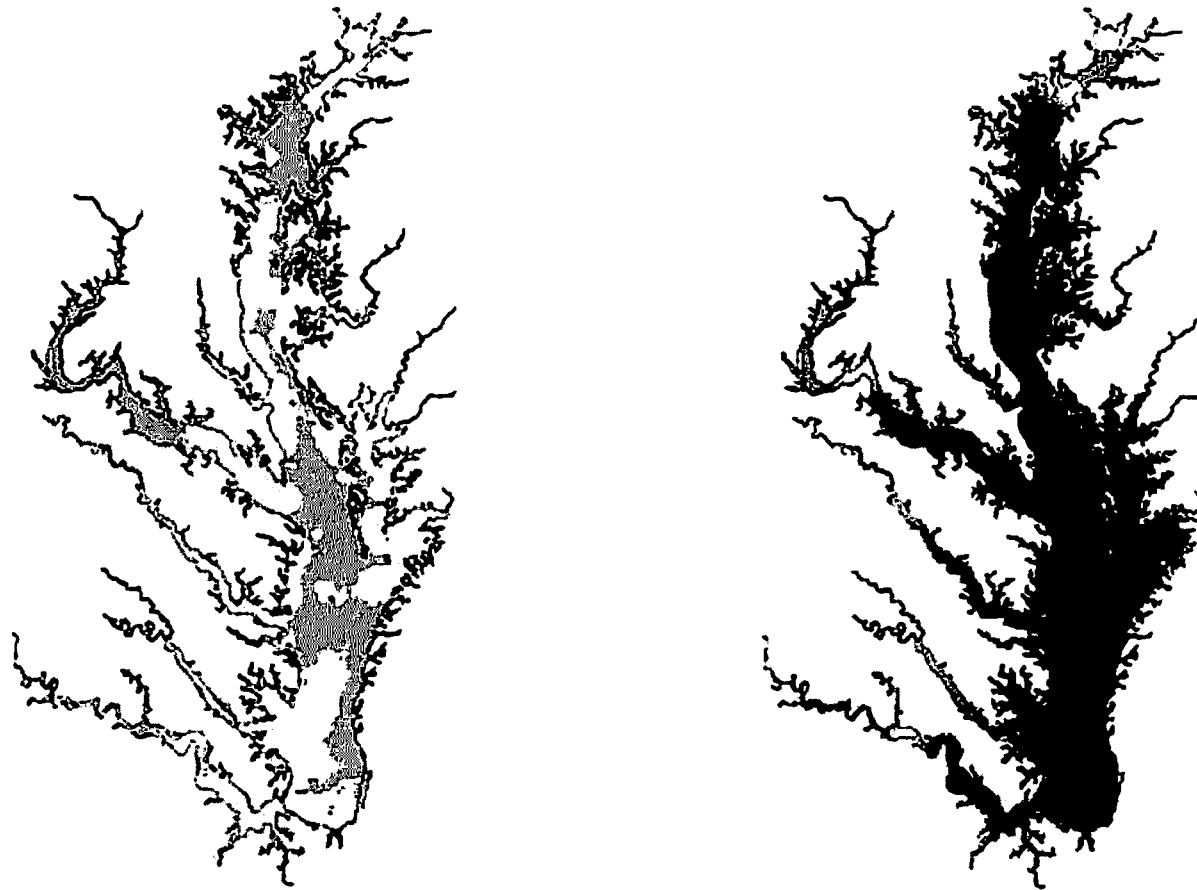


Figure 21 Distribution of blue crab in winter 2001/2002. Maps are predictions of absolute abundance (left) and precision (right).

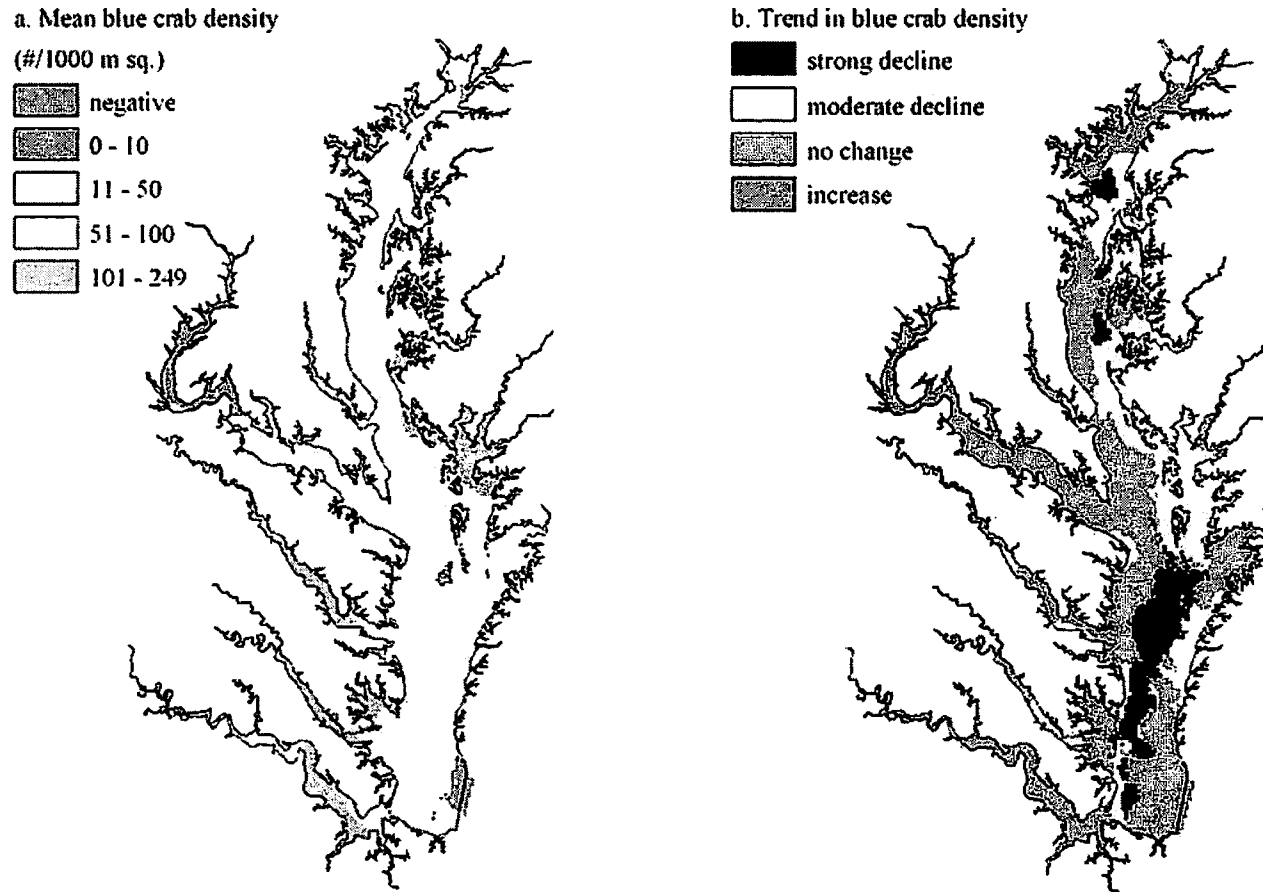


Figure 22. Spatial trends in blue crab A) mean abundance and B) change in abundance estimated from the winter dredge survey for the time period 1995-2003.

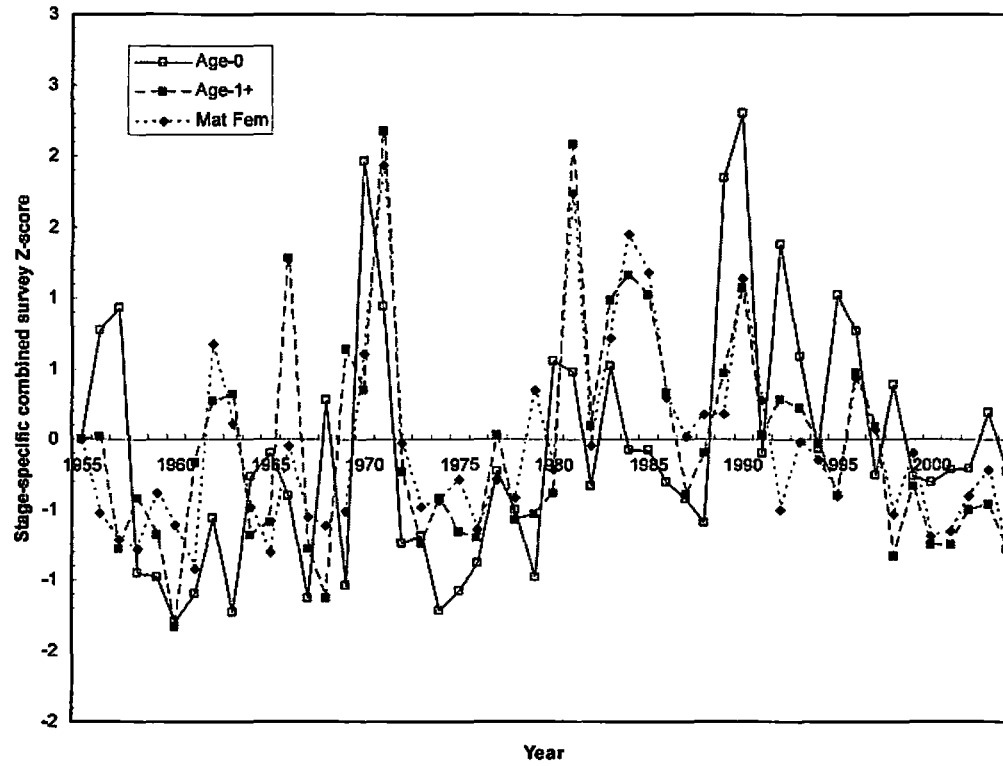


Figure 21. Combined survey Z-score for the four fishery-independent surveys.

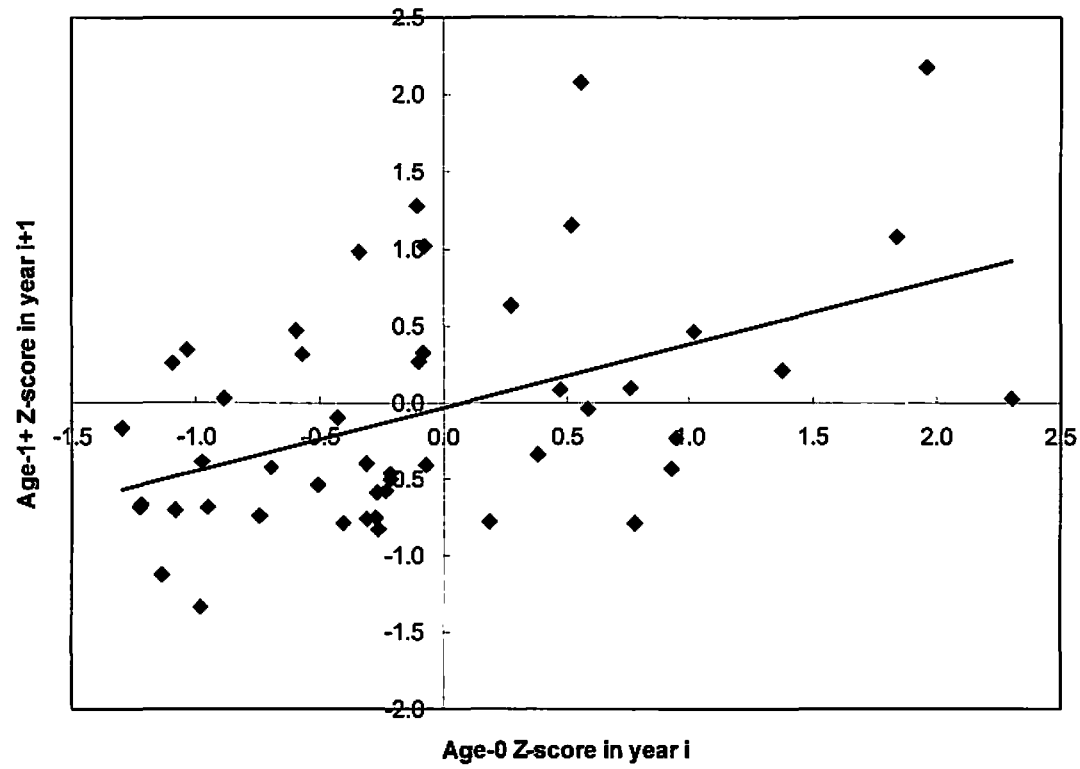


Figure 22 Relationship between combined survey abundance of age-0 crabs in year i with combined survey abundance of age1+ crabs in year i+1.

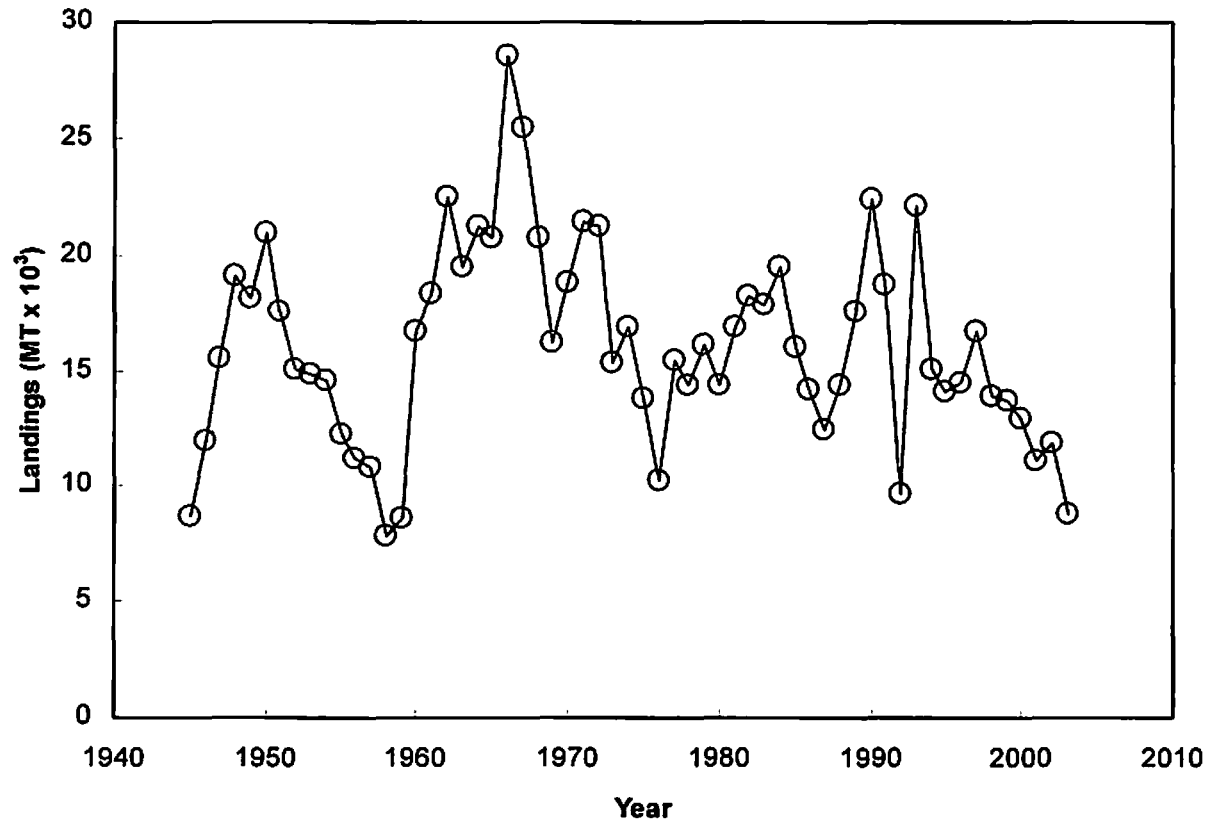


Figure 24. Commercial landings of blue crab in Virginia (MT x 10³) for the period 1946-2003.

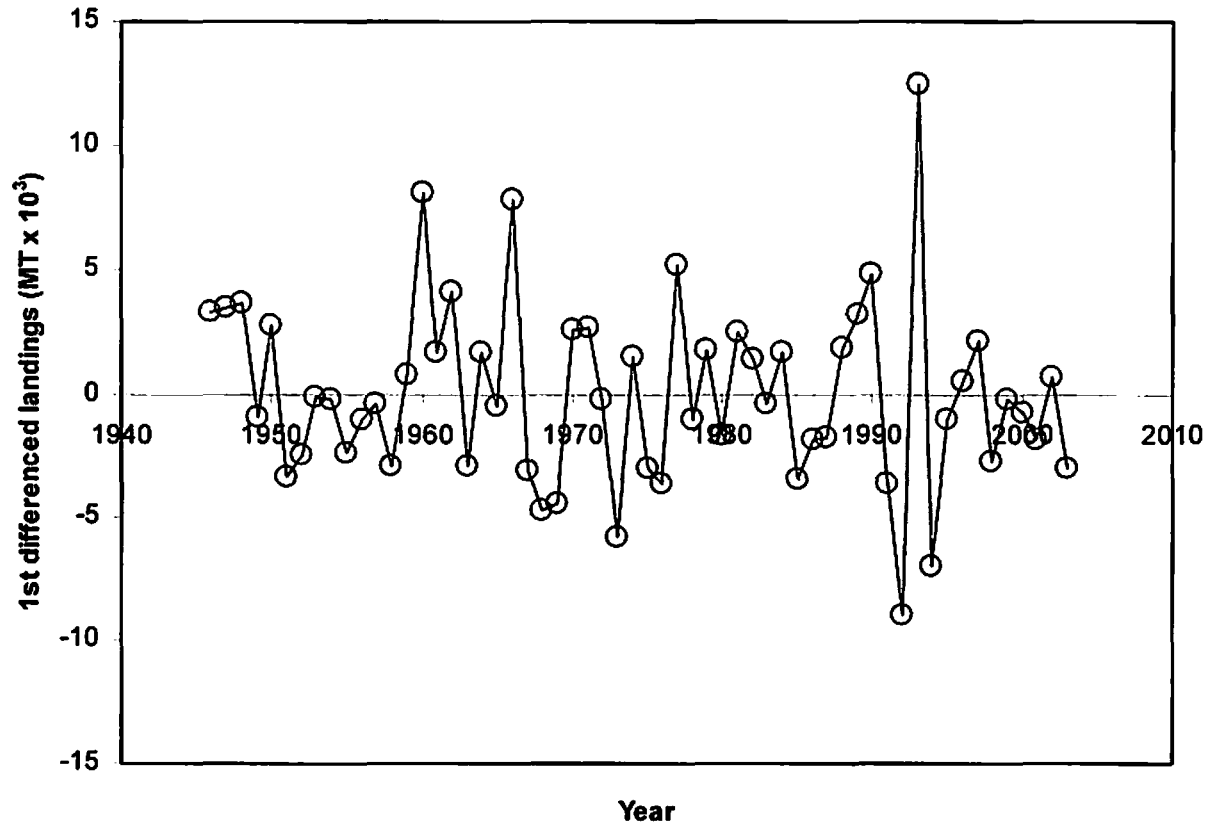


Figure 25. 1st differenced time series of Virginia commercial landings.

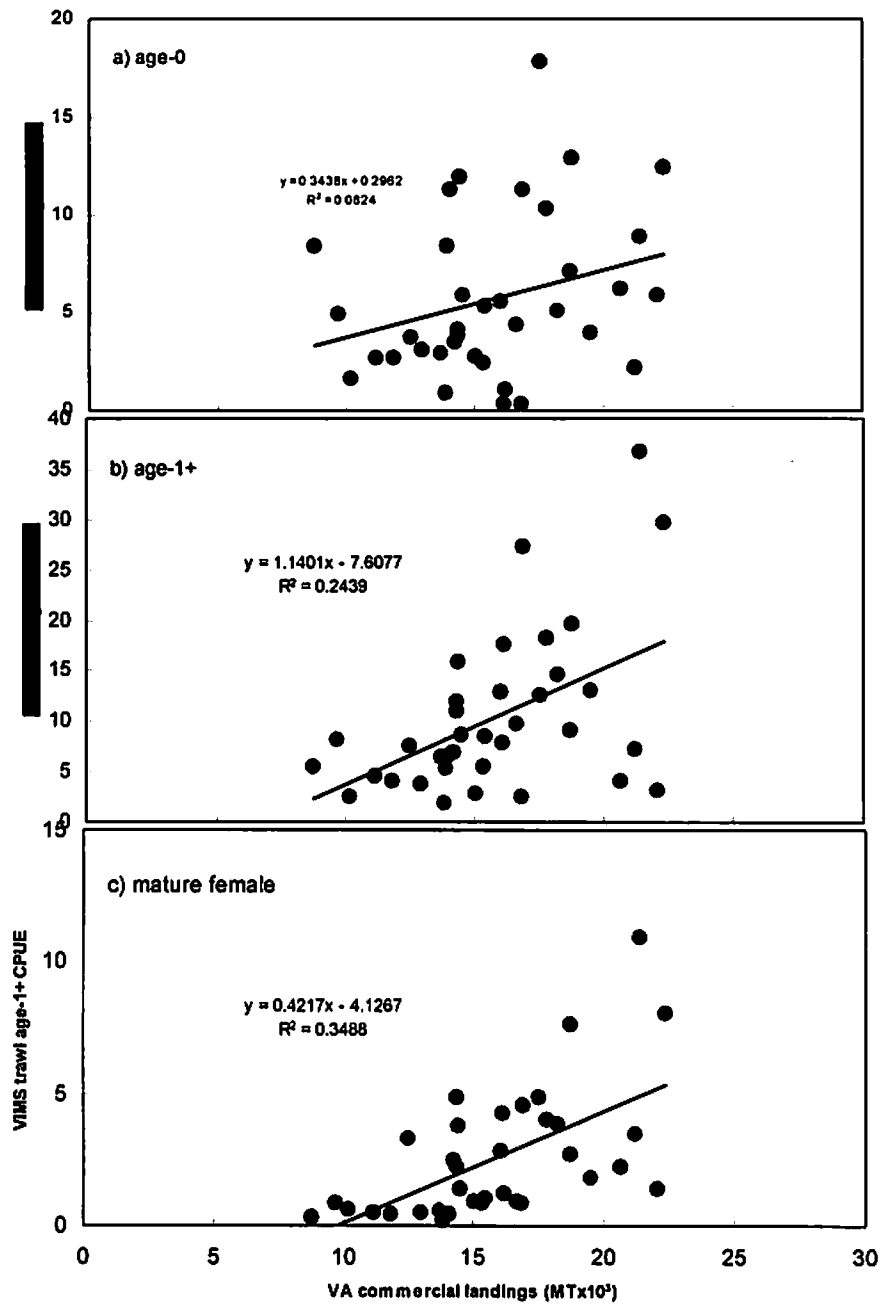


Figure 26. Relationships between VA commercial landings and VIMS trawl indices for a) age-0, b) age-1+ and c) mature female crabs

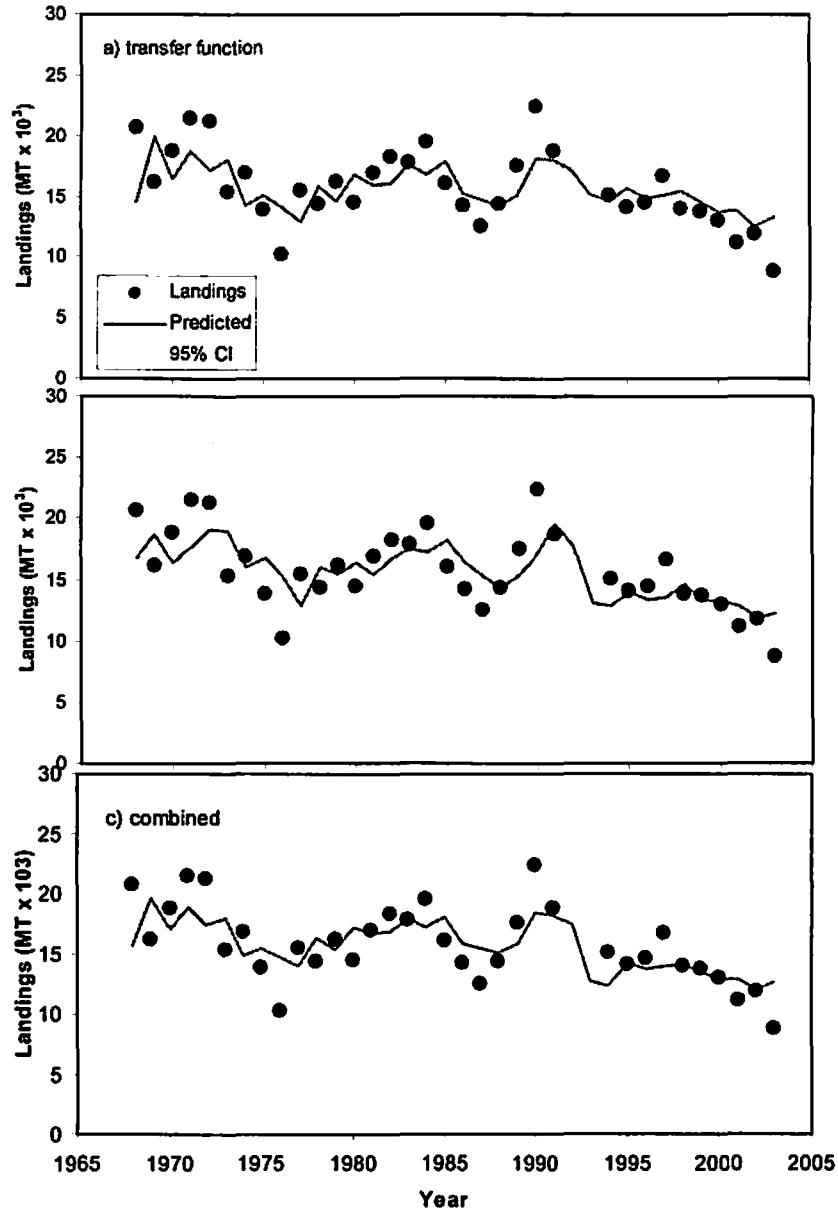


Figure 27. Time series analysis of Virginia commercial landings with autoregressive errors for a) a VIMS age-1+ transfer function, b) an intervention accounting for the 1993 reporting change, and c) a combined model.

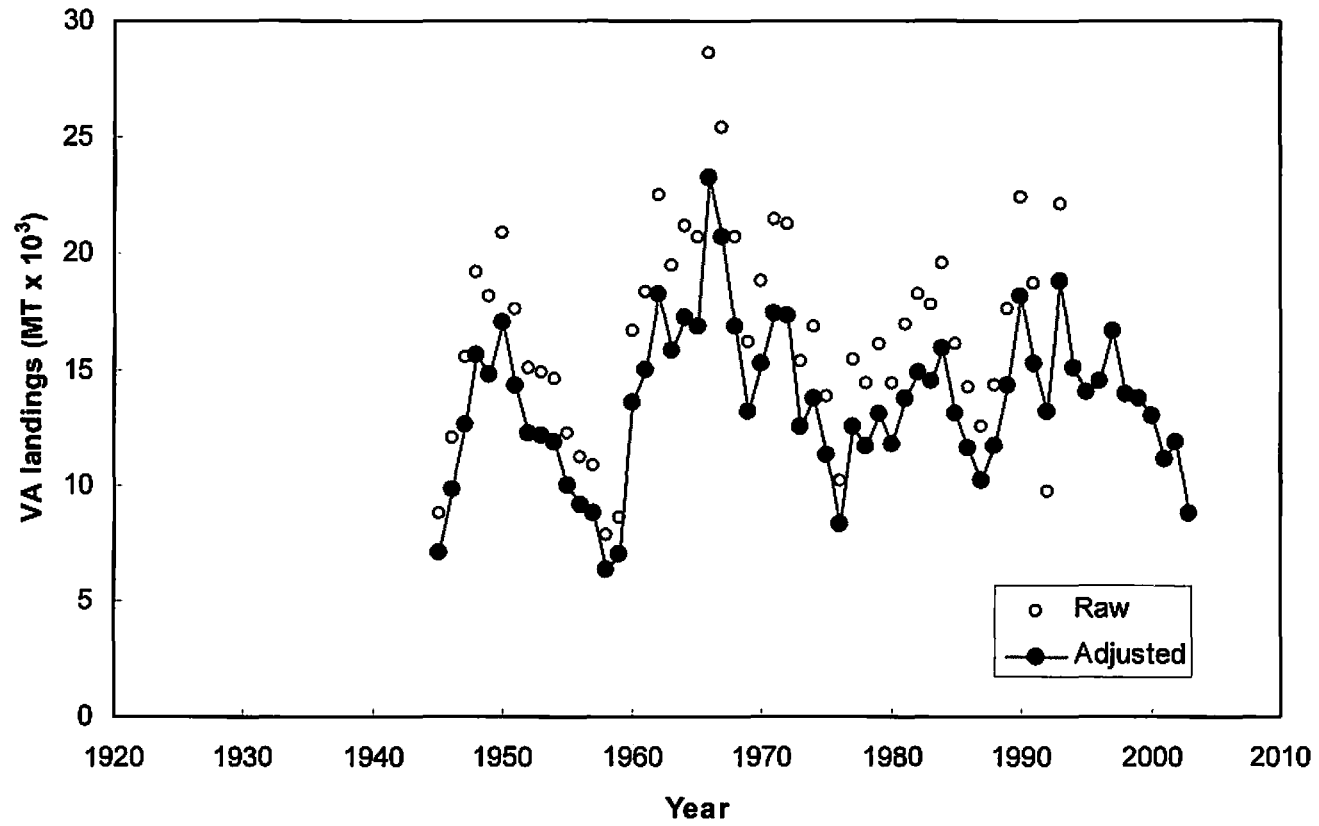


Figure 28. Adjusted estimates of Virginia commercial landings (MT x 10³) for the period 1945-2003.

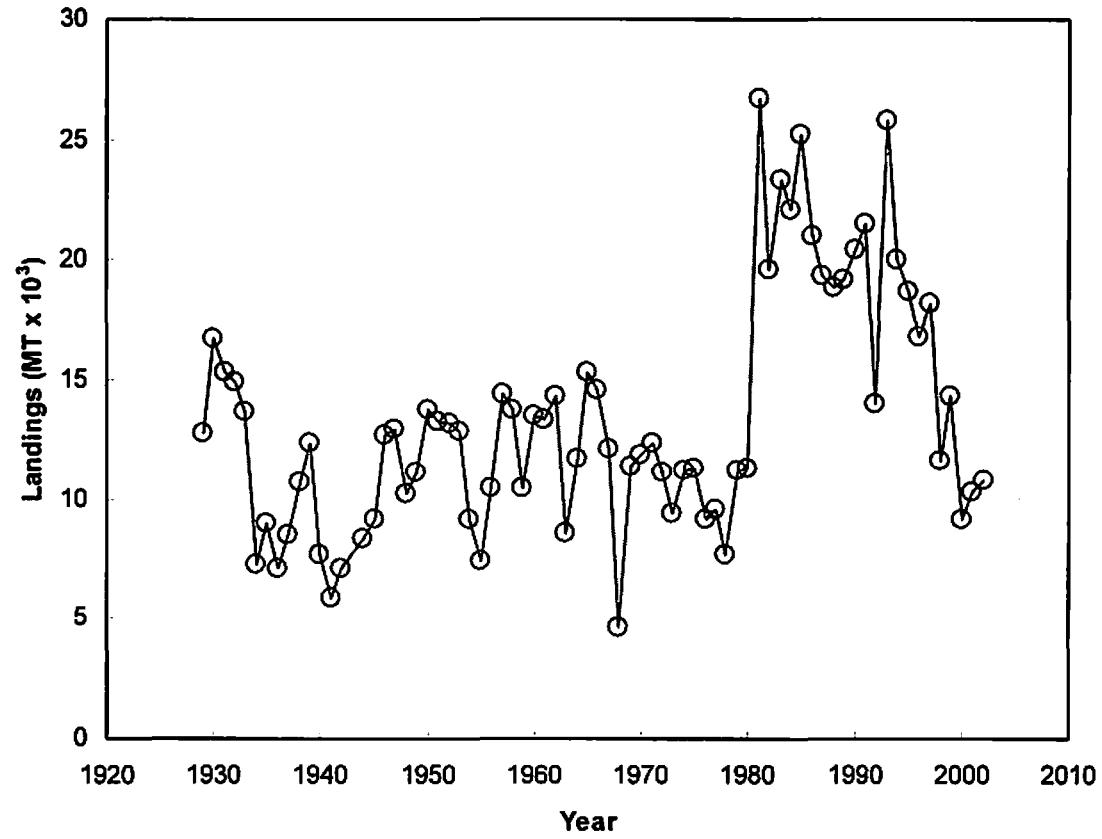


Figure 29. Annual Maryland commercial landings for the period 1929-2003.

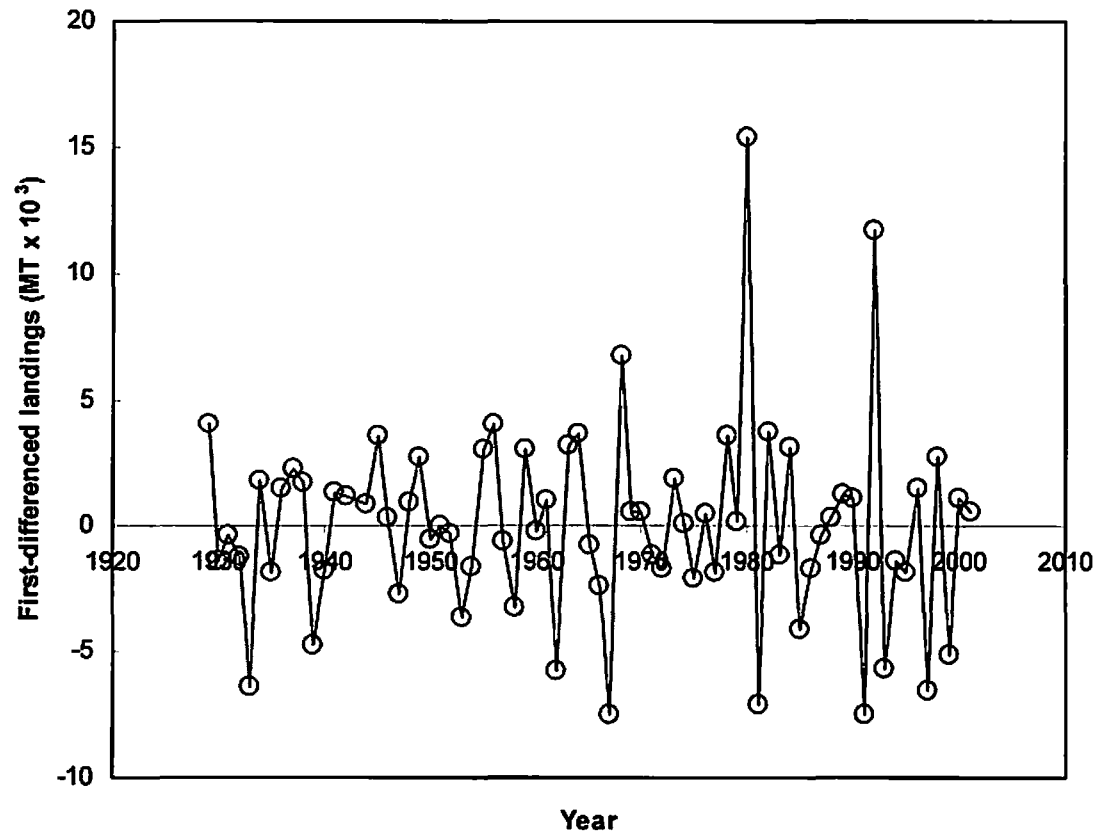


Figure30. First-differenced commercial landings time series for Maryland.

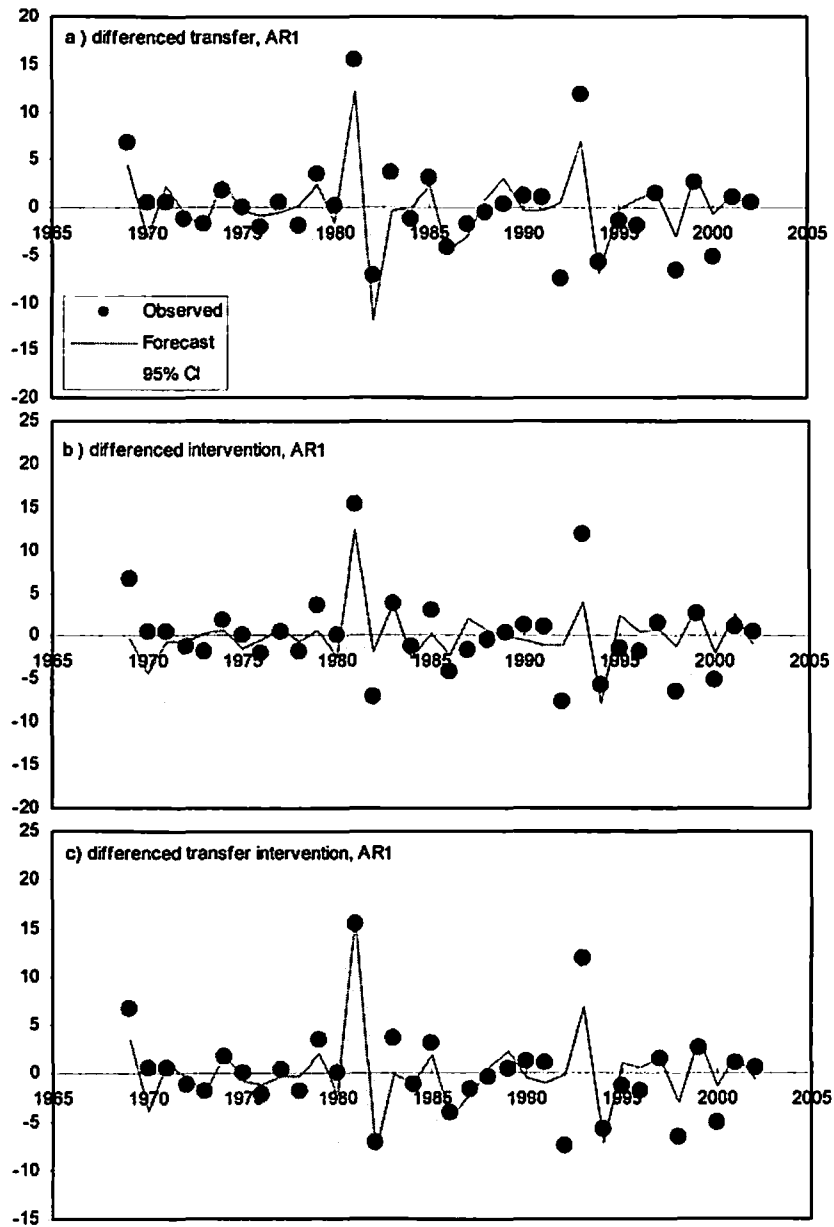


Fig 31 - Transfer function, intervention and joint time series models fit to differenced Maryland commercial landings.

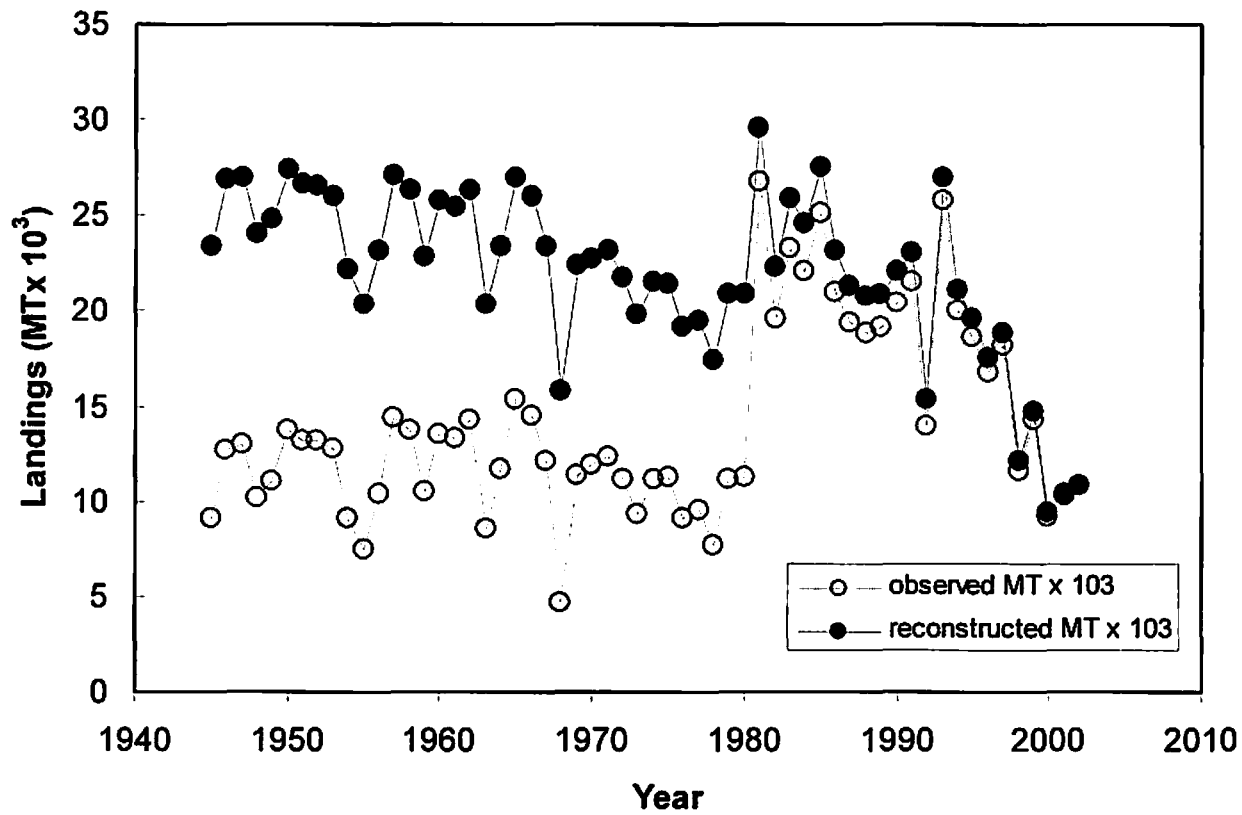


Figure 32. Reconstructed Maryland commercial landings. Landings reconstruction was based on the estimated impact of the 1981 and 1993 reporting changes.

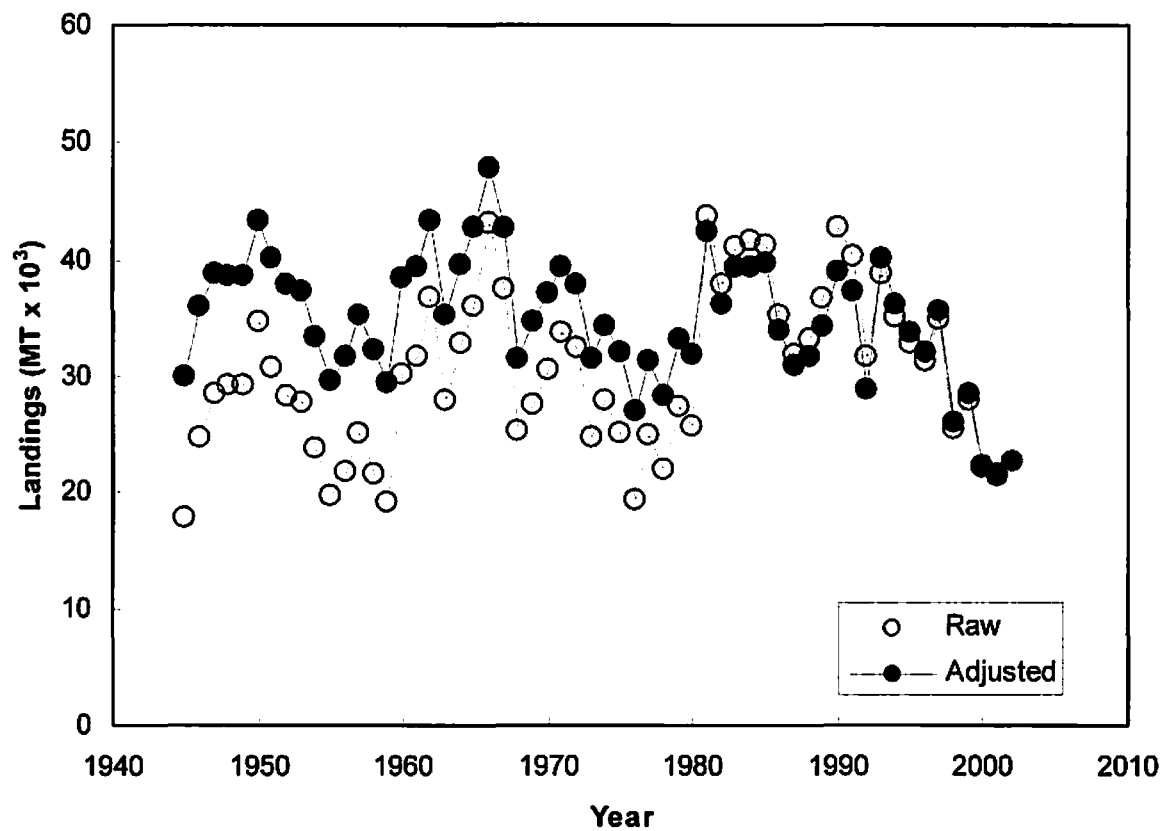


Figure 33. Reconstructed Baywide commercial landings for the period 1945-2003.

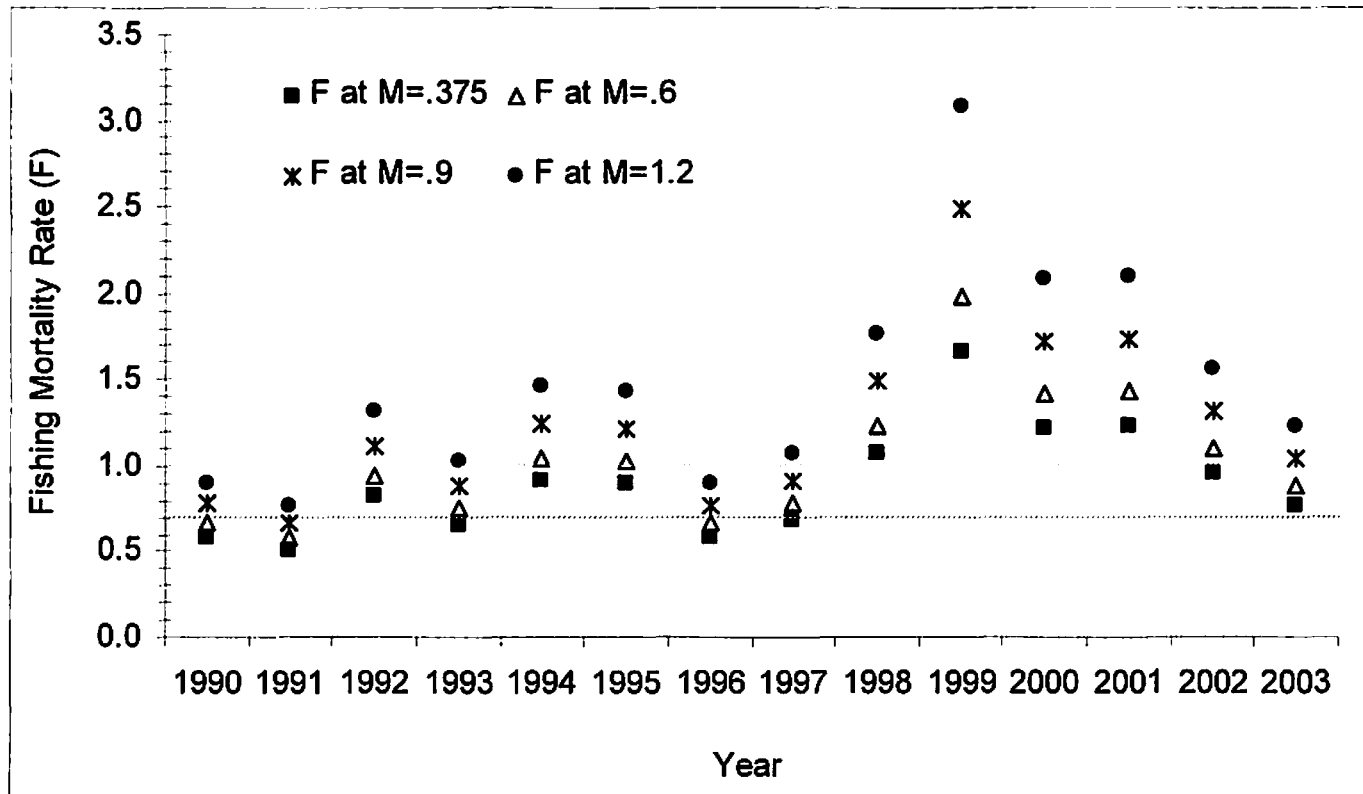


Figure 34. Fishing mortality rate (F) calculated at 4 different levels of M using harvest data and estimates of abundance from the winter dredge survey 1990 through 2003. The dotted lines represent the current (2005) target and threshold levels of F set at 0.7 and 1.0 respectively.

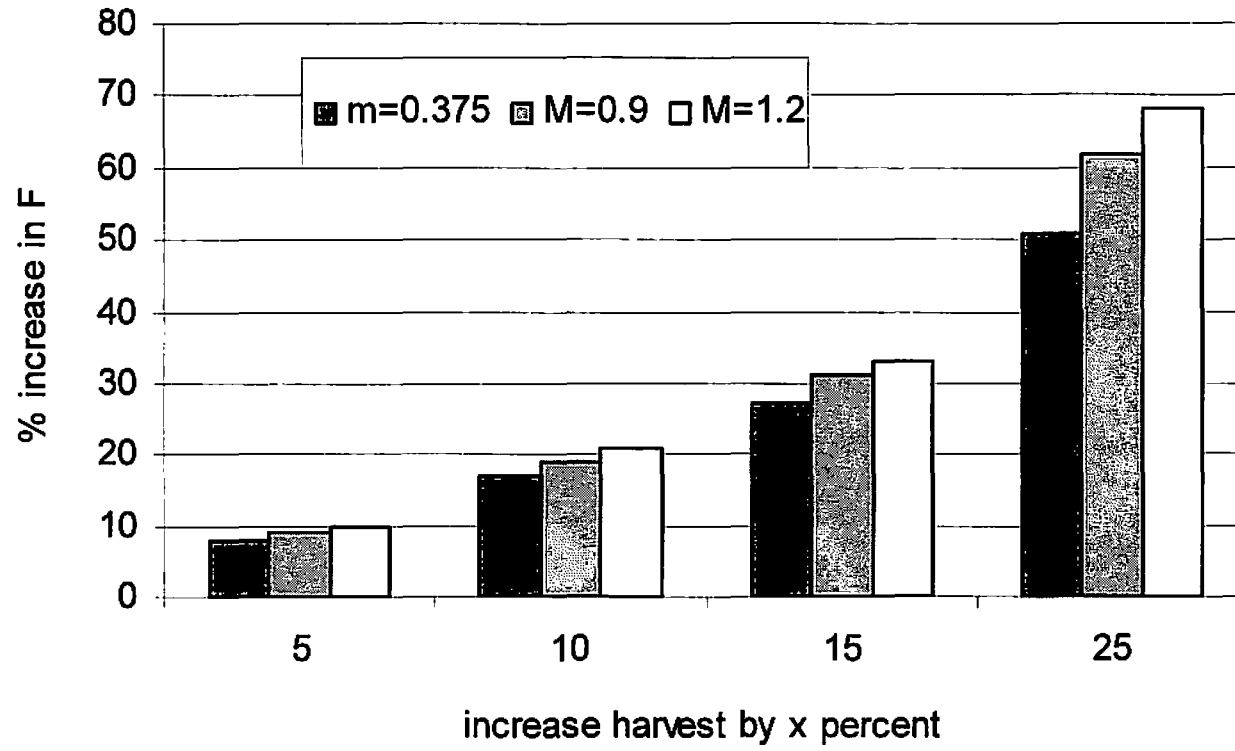


Figure 35. Results of a simulation examining the sensitivity of fishing mortality rates (F) to increases in estimated baywide harvest.

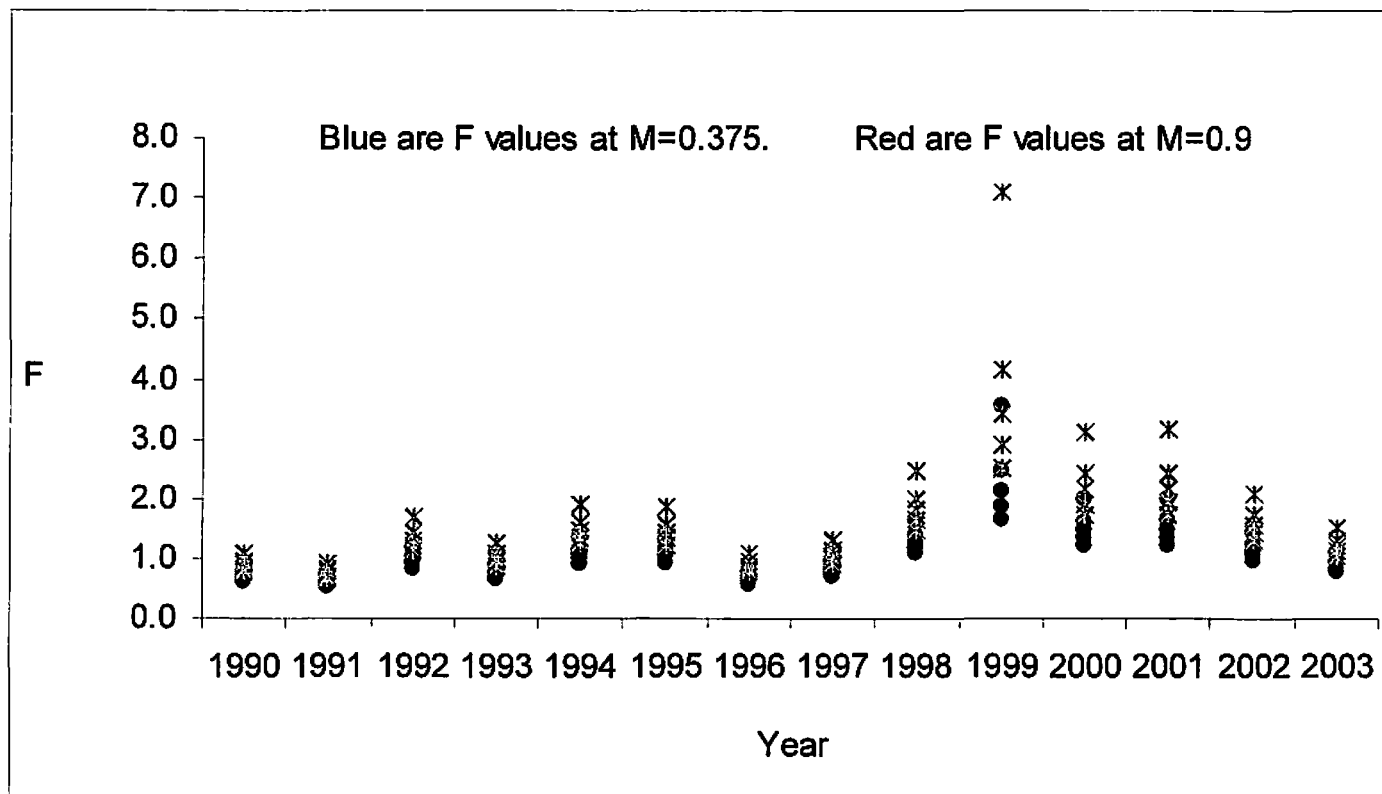


Figure 36. Ranges of F for reported harvest and for reported harvest increased by 5, 10, 15 and 25% at two levels of M. For each level of M, the lowest F represents reported catch and the highest F represents a 25% increase in catch. Despite the sensitivity of F to potential bias in estimates of catch, the overall trend in F, for a given level of M, would be difficult to disrupt.

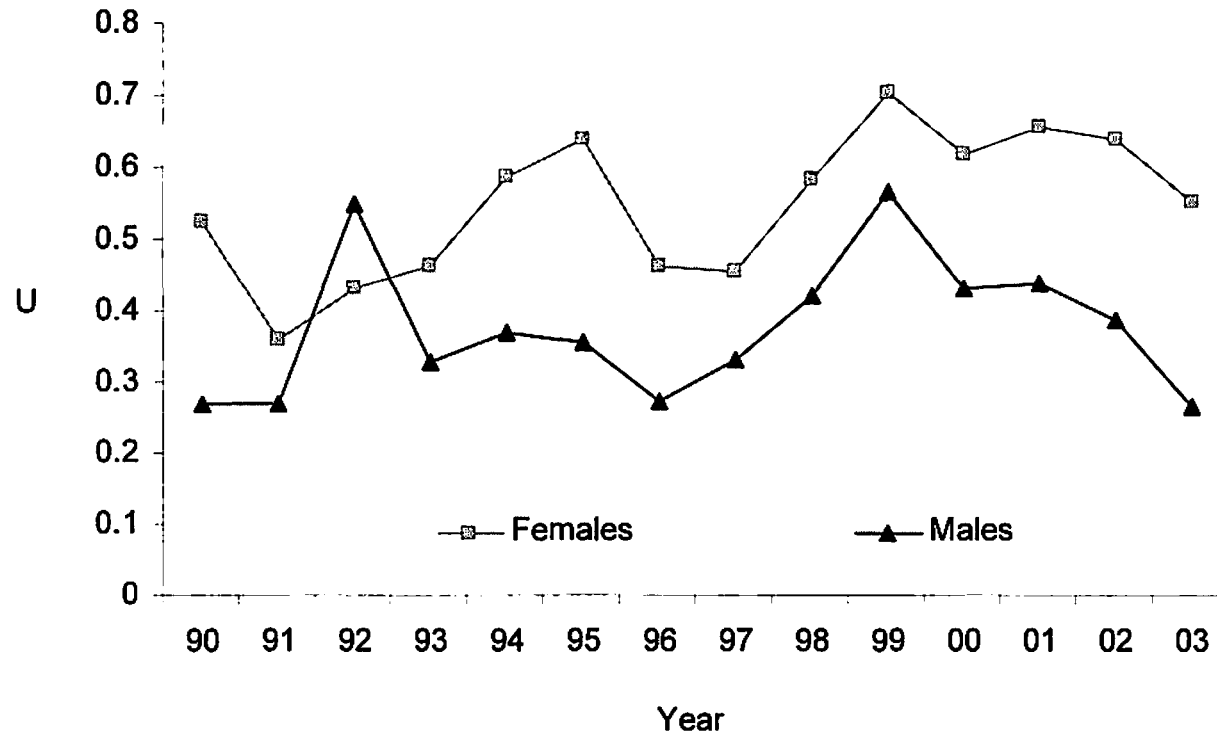


Figure 37. Baywide exploitation fraction (U) calculated separately for male and female blue crabs. Calculation assumes that the sex ratio of the peeler catch is 50:50.

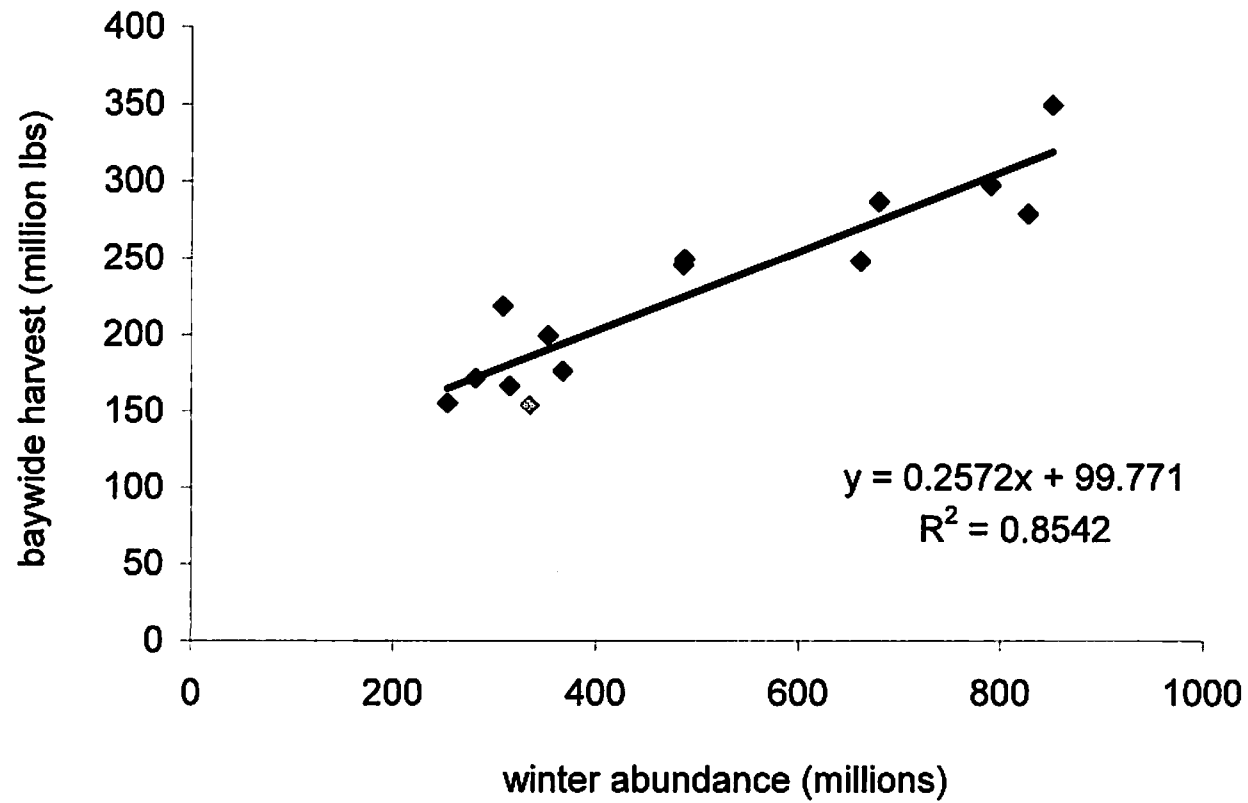


Figure 38. Relationship between the abundance of overwintering crabs of all age classes measured during the winter dredge survey from December through March, and the harvest occurring during the subsequent fishing season occurring from April through December. The red dot is the 2003 data point.

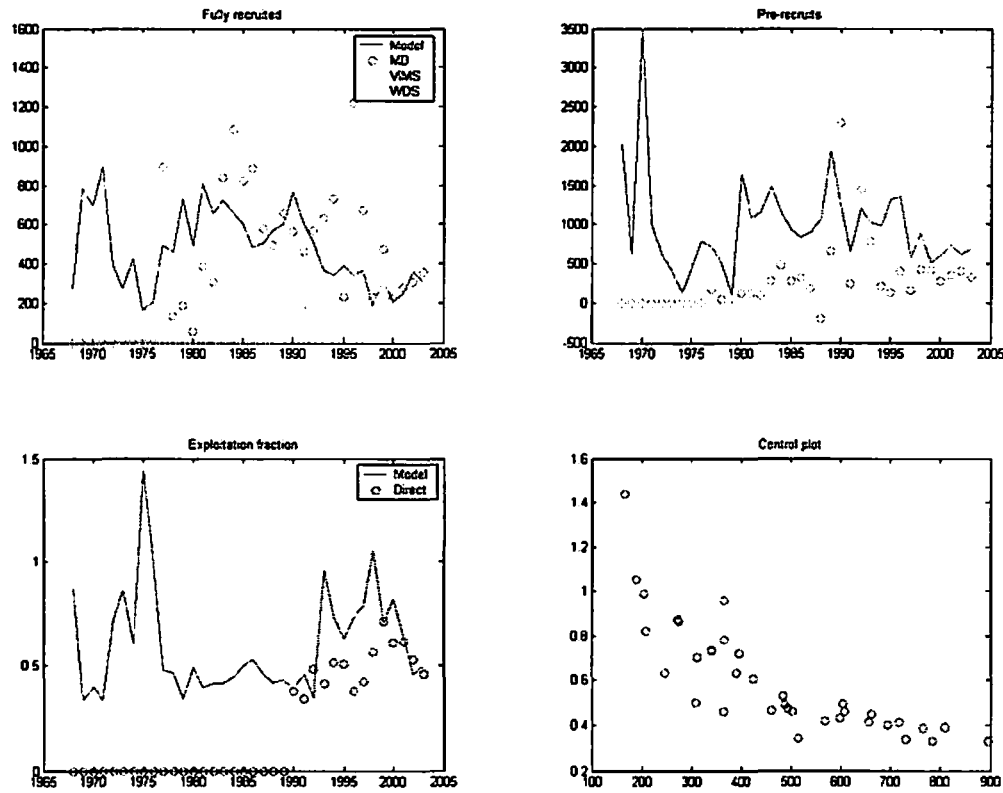


Figure 39. Results for the Catch-multiple survey model for the base simulation for $M=0.9$. A) the predicted abundance of fully recruited crabs (solid line) and the scaled abundance in each survey (symbols), B) the predicted abundance of pre-recruited crabs (solid line) and the scaled abundance in each survey (symbols), C) the predicted (solid line) and the observed (symbols) exploitation fractions and D) the relationship between exploitation fraction (y) and the predicted fully-recruited abundance (x).

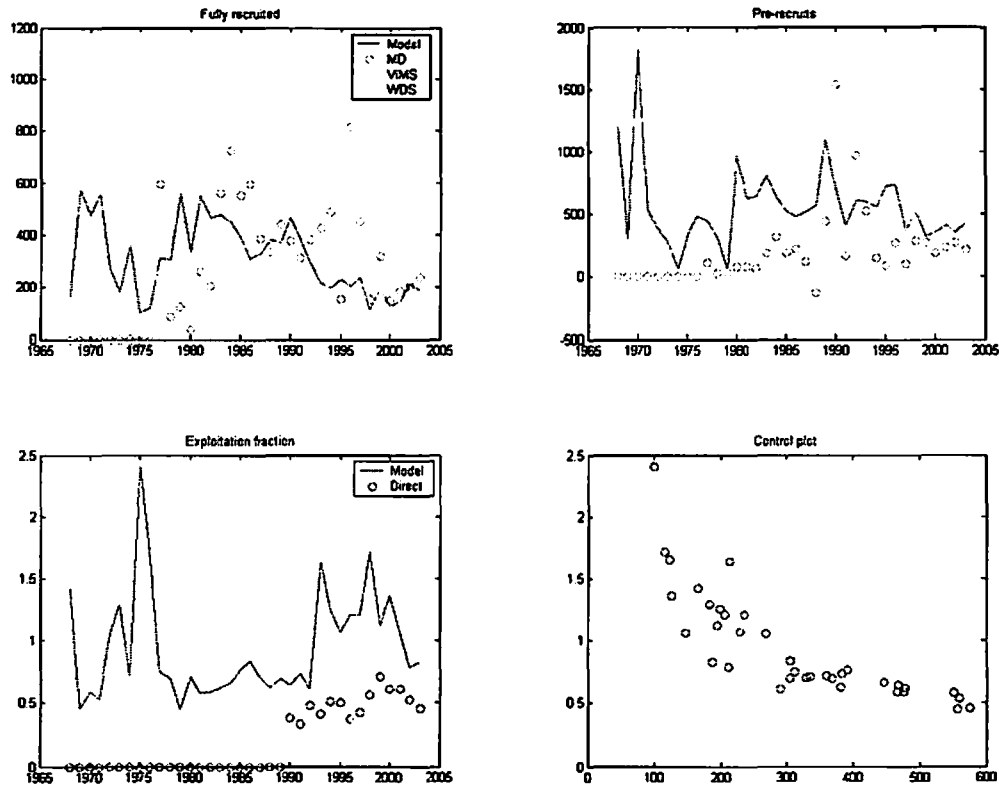


Figure 40. Results for the Catch-multiple survey model for the base simulation for $M=0.6$. A) the predicted abundance of fully recruited crabs (solid line) and the scaled abundance in each survey (symbols), B) the predicted abundance of pre-recruited crabs (solid line) and the scaled abundance in each survey (symbols), C) the predicted (solid line) and the observed (symbols) exploitation fractions and D) the relationship between exploitation fraction (y) and the predicted fully-recruited abundance (x).

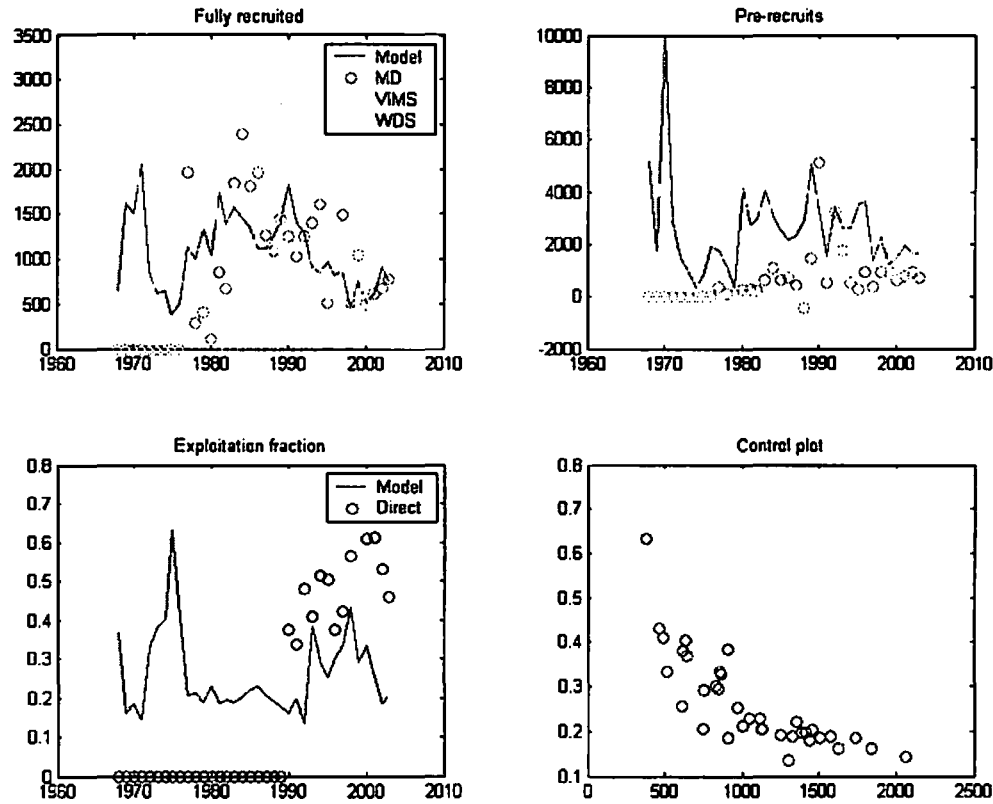


Figure 41. Results for the Catch-multiple survey model for the base simulation for $M=1.2$. A) the predicted abundance of fully recruited crabs (solid line) and the scaled abundance in each survey (symbols), B) the predicted abundance of pre-recruited crabs (solid line) and the scaled abundance in each survey (symbols), C) the predicted (solid line) and the observed (symbols) exploitation fractions and D) the relationship between exploitation fraction (y) and the predicted fully-recruited abundance (x).

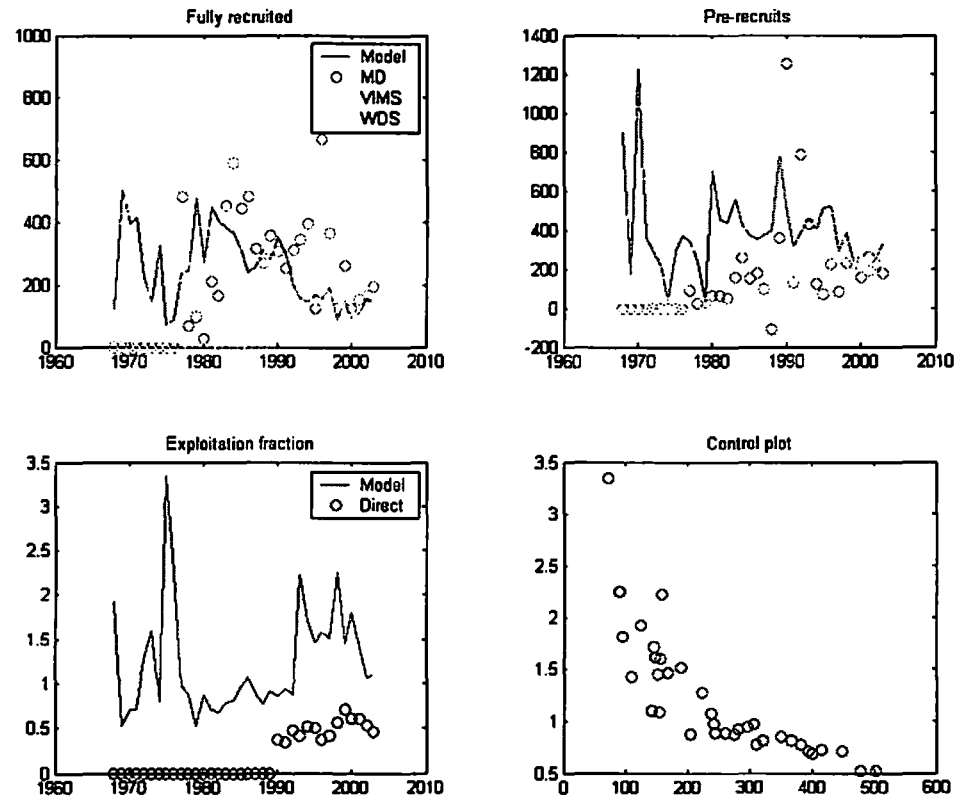


Figure 42. Results for the Catch-multiple survey model for the base simulation for $M=0.375$. A) the predicted abundance of fully recruited crabs (solid line) and the scaled abundance in each survey (symbols), B) the predicted abundance of pre-recruited crabs (solid line) and the scaled abundance in each survey (symbols), C) the predicted (solid line) and the observed (symbols) exploitation fractions and D) the relationship between exploitation fraction (y) and the predicted fully-recruited abundance (x).

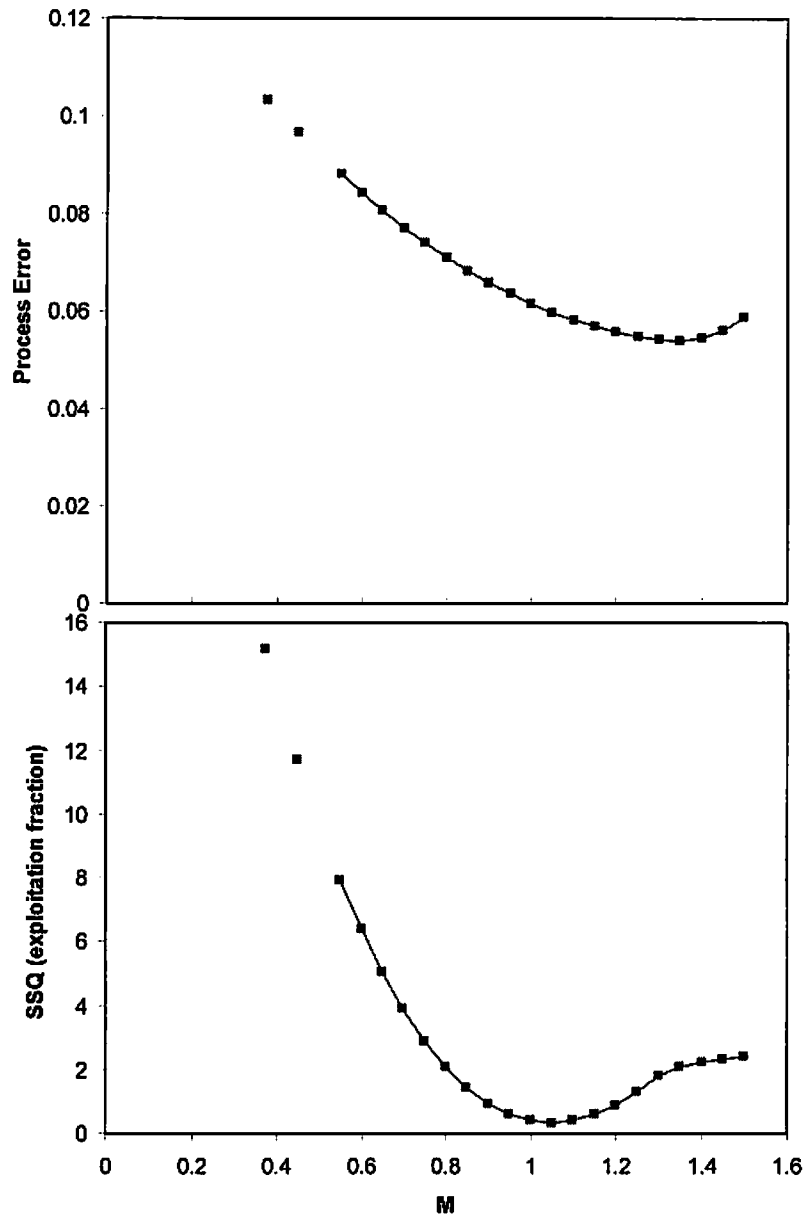


Figure 43. Change in estimated process error (top) and the sum of squared residuals of estimated minus predicted values of exploitation fraction (bottom).

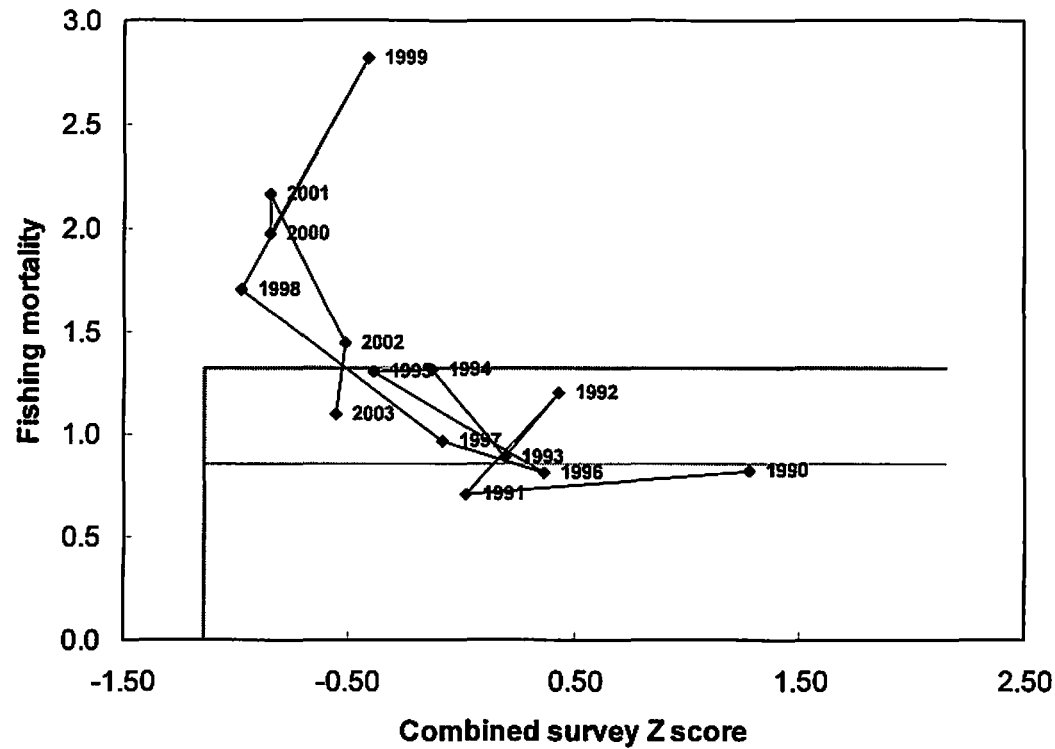


Figure 44. Conventional BBCAC reference points for the Chesapeake Bay blue crab fishery. Plotted are estimates of abundance based on the combined survey Z-score for CPUE and fishing mortality rate calculated from the winter dredge survey exploitation fraction estimates using Baranov's catch equation and $M=0.9$. Shown in red are the overfished biomass reference point (1968 combined survey Z-score) and the overfishing limit and target calculated from a per recruit analysis with $M=0.9$.

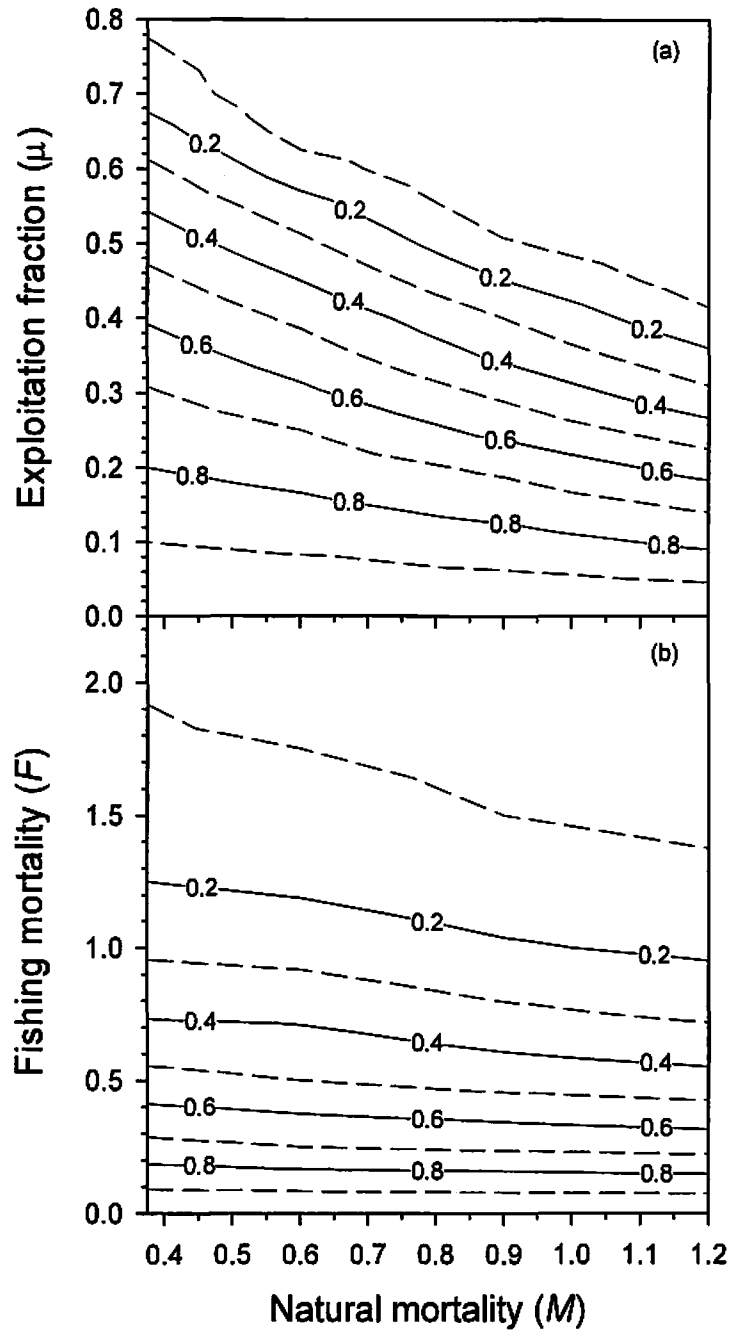


Figure 45. Contour plot of the spawning-potential per-recruit as a function of natural mortality and A) exploitation fraction and B) fishing mortality rate.

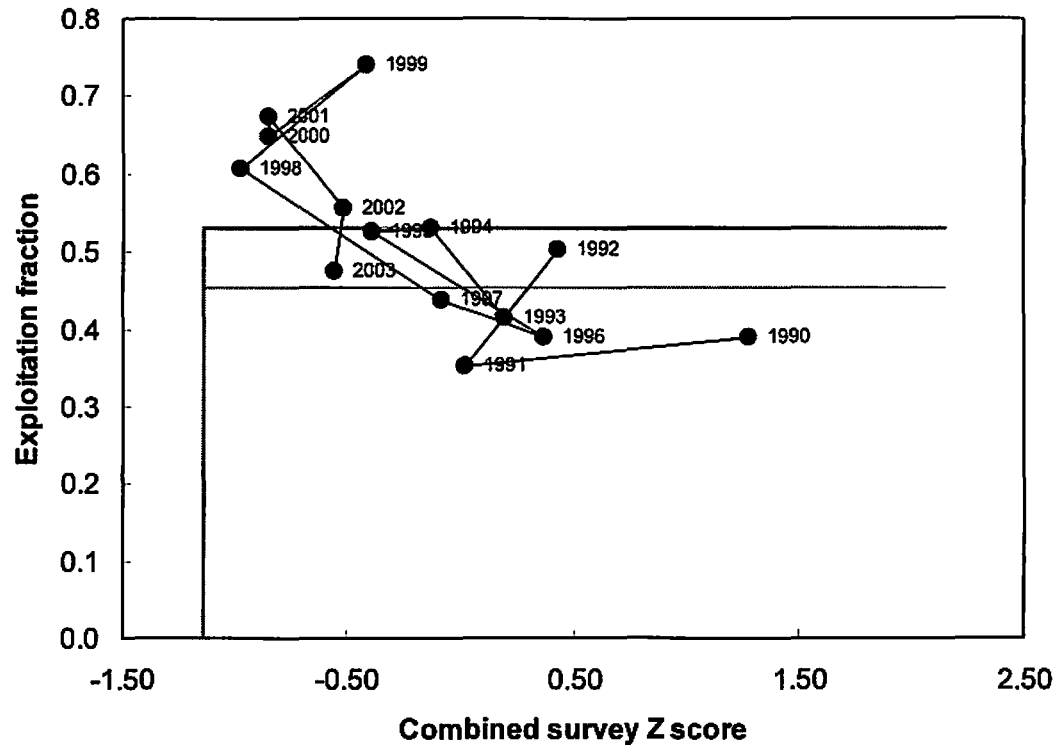


Figure 46. Exploitation fraction reference points for the Chesapeake Bay blue crab fishery. Plotted are estimates of abundance based on the combined survey Z-score for CPUE and the exploitation fraction estimates from the winter dredge survey. Shown in red are the overfished biomass reference point (1968 combined survey Z-score) and the overfishing threshold and target reference points calculated from an IBM SPR analysis with $M=0.9$.

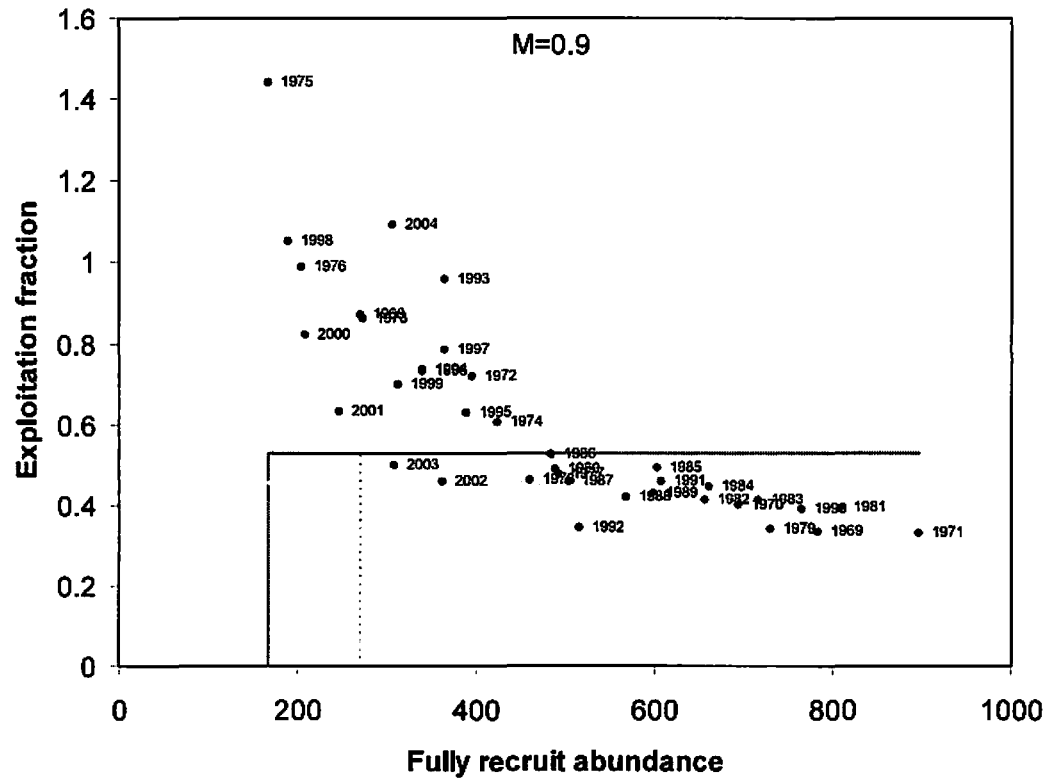


Figure 47. Historical pattern of exploitation fraction and abundance predicted by the stock assessment model for M=0.9. Shown on the plot are the threshold reference points (red solid lines) and the target exploitation rate (green solid line). Also show on the figure is the overfished definition based on the 1968 abundance (red dotted line).

Appendix I

The code for the catch-multiple survey model given as 1) The input data file and 2) The ADMB code for the catch-multiple survey model.

1) Input Data File

```
#MCS_02.dat

#M
1.2

#Fishery Month
6
#Catch Years
1968 2003
#Observed Catches (numbers millions - adjusted for aggregation)
236.4809357 260.131417 277.6822682 295.6692575 284.1569658 236.3664636
    257.4263932 239.865908 202.0712899 234.3562537 212.7629447
    248.3146325 238.4820141 317.9256192 271.2558407 295.393832
    295.2937287 297.407952 254.9978099 231.2571345 237.5291186
    257.1212061 297.491349 278.855948 176.385097 349.273842
    249.239144 245.661766 248.129457 286.817477 199.2492
    218.558608 171.526393 155.137452 166.544504 153.789531

#####
#####

# f_switch == 1 use just adults in mu, ==2 use both adults and
recruits in mu
1

#####
#####
#Observed exploitation fractions

#mu_switch - mu_switch =1 includes mu in obj function, mu_switch=0 does
not include
0

#muyrs
1990
2003

# estimates of mu from winter dredge
0.376 0.337 0.481 0.41 0.512 0.505 0.375 0.422 0.565 0.709 0.61 0.612
    0.529 0.46

#####
#####
#nsets (1=MD Trawl, 2=VA Trawl, 3=Winter Dredge)
```

```

3
#relative selectivity
.3 .3 .5
#Survey Years (Start)
1977 1968 1989
#Survey Years (end)
2003 2003 2003

#Survey months
7 8 1

#Proportion of total error due to measurement errors
0.65 0.65 0.4

#Juvenile index
#MD Trawl Age-0
0.838161466 0.220002454 0.273959361 0.577966404 0.613488458 0.456580726
1.442834038 2.433357152 1.433722245 1.624156695 0.910648497 -1
3.330324427 11.68339368 1.228791303 7.336799733 3.997026895
1.137537526 0.650950317 2.068125274 0.790198574 2.151057343
2.077317276 1.442942848 1.769304306 2.072050804 1.66591647

#VA Trawl Age-0
6.239784086 1.038294575 12.88531883 8.866668329 2.197434384 2.377291891
0.318632253 0.865552067 1.6373425 5.353982017 4.151022771
0.341552088 11.96851229 11.2688621 5.110171821 10.28658746
3.96323093 5.528867558 3.455118611 3.677970658 3.797123415
17.79092346 12.40353061 7.070781362 4.877781552 5.84979765
2.715897257 11.32588119 5.874106885 4.382366963 8.416255249
2.89630844 3.045241591 2.642898565 2.661300779 8.391752711

#WDS Age-0
47.34 36.43 10.69 51.27 30.94 30.71 51.82 52.16 16.87 22.76 13.9 15.86
19.81 17.88 16.37

#Adult index
#MD Trawl age-1+
15.08274579 2.207753285 3.116877282 0.904109602 6.579024935 5.191383085
14.16775951 18.35264905 13.91359703 15.03570983 9.761260386
8.393684133 11.13355347 9.558742745 7.881852811 9.643541433
10.72232716 12.35280206 3.874636083 20.67742133 11.39041733
3.895424694 8.06830504 3.820817754 4.70801844 5.191465756
6.022407635

#VA Trawl age-1+
4.149481555 17.57585915 19.56106848 36.64225308 7.211312842 5.505333827
2.533806181 1.813408842 2.542395976 8.413205483 10.90927387
7.794555736 15.86317507 27.2274816 14.61320527 18.13652439
12.95953708 12.83504554 6.850855333 7.561408773 11.95411892
12.57229806 29.6791512 9.084116607 8.172381664 3.211522844
2.816836068 6.62623903 8.58538325 9.702318315 5.411580381
6.430492246 3.780810999 4.584801395 4.158327899 5.529122445

```



```
#WDS age-1+
34.82 49.11 27.44 37.05 22.61 19.72 24.68 17.59 19.54 9.17 16.59 10.76
12.54 21.91 16.2
```

```
#####
#####
```

```
#Other unpaired data for adults
```

```
#n_y
```

```
1
```

```
#yrs
```

```
1968
```

```
2003
```

```
#survey month
```

```
8
```

```
#proportion of error due to measurement error
```

```
0.5
```

```
#Calvert Cliffs age-1+
```

```
1.1 5.58 3.34 6.8 4.72 2.67 5.4 4.42 3.94 2.58 3.57 4.55
3.99 16.49 5.96 7.04 8.15 9.39 5.48 2.27 3.76 5.71 4.44
5.22 5.68 8.63 5.76 4.11 4.96 5.14 2.56 4.02 3.79 2.67
3.7 4.13
```

2) ADMB File

```

//*****
// Programmer: Steve Martell
// Project Name: Blue crab CMS model
// Date:
// Version: 4
// Comments: Modified by Tom Miller to include N=N+R
//
//*****/
DATA_SECTION
  init_number m;

  init_number fm;          //Fishery month
  !!fm=fm/12;

  //Read in Catch data
  init_int scyr;
  init_int ecyr;
  init_vector ct(scyr,ecyr);
  int nobs;
  !!nobs=ecyr-scyr;
  //!ct*=0;

  init_int f_switch      //switch to include adults (1) or
adults and prerecruits in mu (2)
  // read in estimated exploitation fractions

```

```

    init_int mu_switch;           // switch to include
    exploitation fraction in fvec (1=in, 0=out)
    init_int smuyr;
    init_int emuyr;
    init_vector mu(smuyr,emuyr);
    int nmui;                     // number of obs of exploitation
fraction

    !! nmui=(emuyr-smuyr);
    !!cout<<"Mu_switch "<<mu_switch<<endl;

//read in paired survey data
init_int nsets;                 // Number of surveys
init_vector reisel(1,nsets);
init_ivector sSyr(1,nsets);
init_ivector eSyr(1,nsets);
init_vector sm(1,nsets);
init_vector p(1,nsets);        //sigma/tau
init_matrix yt_r(1,nsets,sSyr,eSyr);
init_matrix yt_a(1,nsets,sSyr,eSyr);
!!sm=sm/12;
!!!cout<<"Survey start year"<<endl<<sSyr<<endl;
!!!cout<<"End start year"<<endl<<eSyr<<endl;

//read in unpaired adult survey data
init_int ny;
init_ivector syt(1,ny);
init_ivector eyt(1,ny);
init_vector usm(1,ny);
init_vector mode_pry(1,ny);
init_vector cv_pry(1,ny);
init_matrix yt(1,ny,syt,eyt);
!!usm=usm/12;

PARAMETER_SECTION
    init_bounded_number log_no(0.0001,10,1);
    //init_bounded_vector sel(1,nsets,0.0001,2,2);
    init_bounded_vector kappa(1,nsets,0.0001,3,1);           //total
variance
    init_bounded_vector log_rt(scyr,ecyr+1,0.0001,10,1);
    init_bounded_vector nu(scyr+1,ecyr,-15,15,1);

    !!kappa=0.065;

    !!log_no=log(max(ct)/0.007);

    !!log_rt=log(exp(log_no)/exp(-m)-exp(log_no));

sdreport_number sd_no;

objective_function_value f;

```

```

number tau;

vector na(1,nsets);
vector nr(1,nsets);
vector ny(1,nsets);

vector q_a(1,nsets); // selectivity for adults in
survey
vector q_r(1,nsets); // selectivity for recruits in
survey

vector nt(scyr,ecyr+1); // fully recruited abundance
vector rt(scyr,ecyr+1); // recruit abundance
vector ft(scyr,ecyr); // exploitation fraction
vector ft2(scyr,ecyr); // exploitation fraction based
on n+r
vector mu_dev(smuyr,emuyr); // exploitation fraction
deviations
vector Nt(scyr,ecyr+1); // total population
abundance

matrix epsilon(1,nsets,sSyr,eSyr); //adult observation
errors
matrix delta(1,nsets,sSyr,eSyr); //recruit observation errors
matrix p_nu(1,nsets,sSyr+1,eSyr); //predicted process errors

```

PROCEDURE_SECTION

```
//-----MAIN-----
```

```

pop_dynamics();
//cout<<"OK after pop_dynamics"<<endl;
exp_dev();
//cout<<"OK after exploitation deviations"<<endl;
observation_errors();
//cout<<"OK after observation_errors"<<endl;
process_errors();
//cout<<"OK after process_errors"<<endl;
calc_objective_function3();
//cout<<"OK after calc_objective_function3"<<endl;

```

```
// _____
```

FUNCTION pop_dynamics

```

int i;
nt.initialize();
nt(scyr)=exp(log_no);
rt=exp(log_rt);

//process error var.
tau=min(elem_prod((1.-p),kappa));
sd_no=exp(tau);
//cout<<"tau = "<<tau<<endl;

for(i=scyr;i<=ecyr;i++)
{

```

```

        nt(i+1) = ((nt(i)+rt(i))*exp(-m*fm) - ct(i))*exp(-m*(1.-
fm));

        if(i>scyr) {
            nt(i+1) *= exp(sqrt(tau)*nu(i));
        }

        Nt(i) = nt(i)+rt(i);
    }
    switch(f_switch)
    {
        case 1:                // use just adults
            ft = elem_div(ct, nt(scyr, ecyr));
            break;
        case 2:                // use both adults and
recruits                       ft = elem_div(ct, Nt(scyr, ecyr));
                                break;
    }

// _____
FUNCTION exp_dev
    int i;
    mu_dev.initialize();

    for(i=smuyr; i<=emuyr; i++)
    {
        mu_dev(i) = mu(i) - ft(i);
    }

FUNCTION dvariable get_q_MLE(dvector& y, dvar_vector& n)
{
    //compute conditional MLE for q_a
    int i;
    dvariable q;
    dvariable xy;
    dvariable x2;

    xy=0.;
    x2=0.;

    for(i=y.indexmin(); i<=y.indexmax(); i++)
    {
        if(y(i)>0) {
            xy+=y(i)*n(i);                // maintain
running totals                   x2+=n(i)*n(i);
        }
    }
    q=xy/x2;
    return(q);
}

```

```

FUNCTION observation_errors
  int i,j;
  //Modify to accomodate missing data  DONE

  epsilon.initialize();
  delta.initialize();
  q_a.initialize();
  q_r.initialize();

  na.initialize();
  nr.initialize();

  for(j=1;j<=nsets;j++)
  {

    //MLE for q_a
    q_a(j)=get_q_MLE(yt_a(j),nt(sSyr(j),eSyr(j)));
    q_r(j)=q_a(j)*relsel(j);
    //q_r(j)=q_a(j)*sel(j);

    //do adult indices first

    for(i=sSyr(j);i<=eSyr(j);i++)
    {
      if(yt_a(j,i)>0){
        epsilon(j,i)=log(yt_a(j,i))-log(q_a(j))-
log(nt(i)*exp(-m*sm(j)));
        na(j)+=1.;
// sample size
      }
    }

    //now do juvenile indices

    for(i=sSyr(j);i<=eSyr(j);i++)
    {
      if(yt_r(j,i)>0){
        delta(j,i)=log(yt_r(j,i))-log(q_r(j))-
log(rt(i)*exp(-m*sm(j)));
        nr(j)+=1.;
// sample size
      }
    }

    //if(yt_a(j,i)>0)epsilon(j,i)-=sum(epsilon(j))/na(j);
  }

  //cout<<"e = "<<sum(epsilon)<<endl;

```

```

//cout<<"d = "<<sum(delta)<<endl;

FUNCTION process_errors
//For each index, calculate the expected process errors
//based on relative abundance indices and pop dy model.

//Modify to accomodate missing data DONE

int i,j;
p_nu.initialize();
ny.initialize();

for(j=1;j<=nsets;j++)
{
    for(i=sSyr(j);i<eSyr(j);i++)
    {
        if(yt_a(j,i+1)>0&&yt_a(j,i)>0&&yt_r(j,i)>0){
            p_nu(j,i+1)=log(yt_a(j,i+1)/q_a(j));
            p_nu(j,i+1)-=log((nt(i)+rt(i))*exp(-m*fm)-
ct(i))*exp(-m*(1.-fm)));
            ny(j)+=1.;
        }
    }
    //cout<<p_nu(j)<<endl;
    //cout<<"p_nu = "<<sum(p_nu)<<endl;
}
//cout<<"p_nu = "<<sum(p_nu)<<endl;

FUNCTION calc_objective_function3
//Modify to accomodate missing data DONE(changes in sample size
na)

int j;
dvar_vector fvec(1,nsets+1);
fvec.initialize();

for(j=1;j<=nsets;j++)
{
    fvec[j]=0.5*(na(j)+nr(j)+ny(j))*log(kappa(j))
            +0.5*(na(j)+nr(j))*log(p(j))
            +0.5*ny(j)*log((1-p(j)))
            +1./(2.*kappa(j))

    *(norm2(epsilon(j))/p(j)+norm2(delta(j))/p(j)+norm2(p_nu(j))/(1.-
p(j)));
}

if(mu_switch==1) {
    fvec[nsets+1] =(1./(2*(nmu-
1)))*log(norm2(mu_dev));
}

```

```

    }
    cout<<"fvec"<<fvec<<endl;
    f=sum(fvec);

REPORT_SECTION
    int i;
    report<<"Nt"<<endl<<nt<<endl;
    report<<"Rt"<<endl<<rt<<endl;
    report<<"nsets"<<endl<<nsets<<endl;
    report<<"sSyr & eSyr"<<endl<<sSyr<<endl<<eSyr<<endl;
    report<<"Adult Survey"<<endl;
    for(i=1;i<=nsets;i++){
        report<<yt_a(i)/q_a(i)<<endl;
    }
    report<<"Recruit Survey"<<endl;
    for(i=1;i<=nsets;i++){
        report<<yt_r(i)/q_r(i)<<endl;
    }

    //fishing mortality rate
    report<<"Ft"<<endl<<ft<<endl;
    report<<"Ct"<<endl<<ct<<endl;
    report<<"Ft2"<<endl<<ft2<<endl;
    report<<"Exp. Deviations"<<endl<<mu_dev<<endl;

    //residuals
    report<<"delta"<<endl<<(delta)<<endl;
    report<<"epsilon"<<endl<<(epsilon)<<endl;
    report<<"p_nu"<<endl<<(p_nu)<<endl;
    report<<"nu"<<endl<<nu<<endl;

    report<<"q_a"<<endl<<q_a<<endl;
    report<<"tau = "<<tau<<endl;

```

Appendix II

Summary of CMS Model runs

Run	M	Rel selectivities			P(measurement error)			Fswitch	Mu_Switch	L	Process Error	SSq Mu_dev
		MD Trawl	VA Trawl	WDS	MD Trawl	VA Trawl	WDS					
1	0.9	0.5	0.5	0.5	0	0	0	1	0	did not converge		
2	0.9	0.5	0.5	0.5	0.25	0.25	0.25	1	0	-22.537	0.111578	0.5685
3	0.9	0.5	0.5	0.5	0.5	0.5	0.5	1	0	-30.672	0.0726539	0.4167
4	0.9	0.5	0.5	0.5	0.75	0.75	0.75	1	0	-29.9	0.0337372	0.3602
5	0.9	0.5	0.5	0.5	0.65	0.65	0.4	1	0	-29.9	0.0337372	0.3602
5	0.9	0.5	0.5	0.5	0.6	0.6	0.4	1	0	-33.069	0.0806083	0.3564
6	0.9	0.5	0.5	0.5	0.65	0.65	0.4	1	0	-33.179	0.0793732	0.346
7	0.9	0.5	0.5	0.5	0.65	0.65	0.35	1	0	-33.795	0.0847009	0.3462
8	0.9	0.4	0.4	0.5	0.65	0.65	0.4	1	0	-35.407	0.0715073	0.4697
9	0.9	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-35.19	0.0656968	0.9437
10	0.6	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-39.319	0.084192	6.3743
11	1.2	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-27.149	0.0557927	0.8907
12	0.375	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-40.257	0.103411	15.1823
13	0.4	0.3	0.3	0.5	0.65	0.65	0.4	1	0			
14	0.45	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-40.156	0.0966483	11.7051
15	0.5	0.3	0.3	0.5	0.65	0.65	0.4	1	0			
16	0.55	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-39.688	0.0881425	7.9295
17	0.65	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-38.86	0.0804759	5.0224
18	0.7	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-38.312	0.0770084	3.8622
19	0.75	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-37.674	0.073797	2.8832
20	0.8	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-36.943	0.0708434	2.0759
21	0.85	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-36.117	0.0681452	1.432
22	0.95	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-34.157	0.0634907	0.6038
23	1	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-33.011	0.061518	0.4053
24	1.05	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-31.746	0.0597695	0.3417
25	1.1	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-30.353	0.0582369	0.4062
26	1.15	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-28.824	0.0569129	0.5916
27	1.25	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-25.32	0.0548735	1.2967
28	1.3	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-23.331	0.0541536	1.8
29	1.35	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-2.11E+01	0.0538838	2.0845
30	1.4	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-1.83E+01	0.0545354	2.2046
31	1.45	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-1.51E+01	0.0561724	2.3029
32	1.5	0.3	0.3	0.5	0.65	0.65	0.4	1	0	-1.14E+01	0.058861	2.3816