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A Stochastic Population Model for Managing the Atlantic Menhaden
Fishery and Assessing Managerial Risks

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A management-oriented population model for Atlantic menhaden is presented. The model includes an age-structure, a stochastic egg-recruitment relationship, density-dependent juvenile growth, age-dependent fishing mortality, and fecundity dependent upon size as well as age. The model was used to investigate three types of harvesting strategies: constant yearly catch policies, constant fishing mortality rate policies, and "egg escapement" policies which are defined in the paper. Because of stochastic recruitment, constant yearly catch policies appear unsuitable for managing Atlantic menhaden. Both other types of policies are suitable, but the "egg escapement" policies have higher long-term average catches. Using decision theory we investigated risks due to the randomness of recruitment and to the

estimation errors for the biological parameters in our model. The risks appear to be acceptable.

Key Words: stochastic population model, optimal harvest, risk, decision theory.

1. INTRODUCTION

The Atlantic and Gulf menhaden together support the largest U.S. commercial fishery measured in pounds landed, and, excluding shellfish, menhaden catches rank next to those of salmon and tuna in monetary value. Catches of Atlantic menhaden experienced a rapid decline after 1962, e.g. a total catch of 537 thousand metric tons in 1962 dropped to 193 thousand metric tons in 1967 (Nicholson, 1971). Recent catch data show that the Atlantic menhaden fishery has only partially recovered. We developed a simulation model of the Atlantic menhaden fishery to investigate various harvesting policies and their associated risks.

Beddington, Botkin, and Levin (1981) have noted that there are two roles for mathematical models of renewable resources: 1) understanding and suggestions, and 2) management. Models of the first type are often simple and analytically tractable, while models in the second category contain detailed structures which generally improve their predictive performance. In the past, most mathematical models have been of the first type.

We used a detailed, management-oriented model. To the fullest extent possible, we structured our model to reflect information from both a large data base collected from 1955 through 1979 by the National Marine Fisheries Service Laboratory at Beaufort, NC, and many suggestions from the staff there. An extensive analysis of these data is found in Deriso, et al. (1983) and Reish, et al. (1984), and our model used many of the estimates found in these papers.

Important features of menhaden biology were incorporated into the model, including an age-structure, density-dependent juvenile growth, age-dependent fishing mortality, size and age-dependent fecundity, and a stochastic egg-recruitment relationship. We investigated three classes of management policies: 1) constant yearly catch policies, 2) constant instantaneous fishing

mortality (F) policies, where the age-specific F's are kept fixed, and 3) policies we call "egg escapement" policies where the egg production by the mature stock is controlled, though not completely.

To analyze these policies and their associated risks, we used decision theory, the science of making decisions under conditions of uncertainty. Fisheries management entails uncertainty because the parameters in our models are estimated with error and because the actual fish populations usually exhibit random variations. Even if our model corresponded perfectly with the true population and all parameters were estimated without error, we might still face considerable uncertainty if the population had stochastic components. (Another cause of risk, uninvestigated here, is model misspecification.)

From decision theory, we borrowed the ideas of "utility function" and "opportunity loss". Although we did not make extensive use of formal decision theoretic methods, we found these tools very useful when measuring risk.

2. THE BIOLOGICAL MODEL

In the introduction, we mentioned the biological characteristics incorporated in the simulation model. Here we give a more extensive description of the model. Complete details can be found in Ruppert, et al. (1983) and Reish (1982).

The model predicts yearly catches of menhaden on the Atlantic coast. More complex (area-and season-specific) models than ours did not seem feasible, partly due to time and cost constraints, but largely because of inadequate data. Although a large amount of catch-effort and mark-recapture data are available, they do not provide accurate estimates of migration rates and area and season-specific natural mortality rates (Deriso et al., 1983).

The egg-recruitment data were well fit by the Beverton-Holt (1957) model

$$(2.1) \quad N_0 = 1/(\alpha+\beta/E) ,$$

where N_0 is young-of-the-year abundance in September, E is total egg production for the year, and α and β are parameters. For complete details of this analysis see Deriso et al. (1983) and Reish et al. (1983). Equation (2.1) is, of course, non-stochastic, but in the actual population recruitment is highly random even when egg production is held constant (Nelson et al., 1977). After further data analysis, it was found that a model fitting the random nature of the data is given by

$$(2.2) \quad N_0 = 1/(\alpha+\beta/E+\sigma\varepsilon)$$

where ε is a standard normal random variable, i.e., normal with mean 0 and standard deviation 1. See Reish et al. (1983) and Carroll and Ruppert (1982).

Model (2.2) is not entirely satisfactory since the denominator may be negative and, even when it is positive, N_0 is unbounded as the denominator approaches 0. Therefore, we decided to truncate the denominator in (2.2) so that it could never fall below $1/(2N_{0,max})$ where $N_{0,max}$ is the largest recruitment observed in the data. Thus, N_0 could never exceed $2N_{0,max}$, which seemed like a reasonable restriction.

Statistical analysis of menhaden weight data (Deriso et al., 1983) suggested that during the juvenile period prior to recruitment growth is density-dependent, but growth after recruitment is independent of density. Female fecundity is also likely to be density-dependent, since egg production is size-dependent (Higham and Nicholson, 1964).

Within the range of historic data, weight and egg production for a given cohort were modeled as the product of an increasing age-specific function and a decreasing linear function of its pre-recruitment density. Beyond the historic range of these juvenile densities, age-specific weight and egg production in the model decreased exponentially to one-half the lowest observed value. Further discussion and numerical details are in Ruppert, et al. (1983).

Using mark-recapture data, we obtained estimates of age-specific natural mortality rates. Differences between the age-specific estimates seem due only to statistical variability, so in the model a constant natural mortality rate was used (Reish et al., 1983). It will be denoted by m .

Fishing mortality (F) of course depended upon the harvesting policy, and these will be discussed in detail later. For each year, a policy determined, directly or indirectly, the instantaneous fishing mortality rate (F_2) and the exploitation fraction (μ_2) for age 2 menhaden. In the model, instantaneous fishing mortality rates for the other ages were determined so that the ratios of F_2 (for the constant F policy) or of μ_2 (for the constant

catch and "egg escapement" policies) to the other age-specific rates were kept equal to the corresponding estimated ratios (called R_i for age = i) determined for the actual population between 1971 and 1975. Figure 2.1 illustrates a yearly cycle of the model with a fishing mortality rate policy in use.

3. UTILITY FUNCTIONS AND OPPORTUNITY LOSSES

The utility functions in our study measured the value of a catch of a given amount (figure 3.1). Past research on fishing policies normally considered the value of a catch as proportional to its amount. Such a utility function is called "risk neutral" and is characterized by having a constant first derivative (or marginal utility).

Most decision makers are thought to be normally "risk averse", i.e., to have decreasing marginal utilities (Thorp, 1971). This would mean that a unit of biomass in a small catch is worth more than the same unit in a larger catch. This makes sense from the viewpoint of a fishery biologist, a fishery manager, or a commercial fisherman.

When catch is an adequate index of abundance, then a fishery biologist would normally consider individual fish from small catches to have greater utility than those from large catches. The "reproductive burden" per fish increases as the population size decreases, so that individual fish in a small population have more value in terms of replenishing the stock. At low population levels, the manager becomes extremely cautious concerning the collapse of the fishery and is more adverse to risk taking.

A commercial fisherman generally has more time and money invested per fish in smaller catches. For example, a scarcity of Atlantic menhaden will increase the search time and number of purse seine sets needed to fill a vessel (Nicholson, 1971 and Deriso et al., 1983). Also, after a fisherman has caught enough to meet expenses and earn a reasonable profit, additional catches may have less value to him.

In decision theoretic problems one must choose between many courses of action, called decisions. In the context of this paper, a decision is a harvesting policy. Decision making involves risk because among the many possible "states of nature", we do not know which is the true one. If we

ignore uncertainty about biological parameters, such as m or the parameters α , β , and σ in model (2.2), then in our situation the state of nature is the set of values of the random variables ϵ in (2.2). Later, when we consider uncertainty about parameters, the state of nature is the set of ϵ values and parameter values.

When one knows the probability of each possible state of nature, then it is possible to calculate the expected utility for any decision. If there is only a finite number, say k , states of nature then the expected utility of a decision is $\sum_{i=1}^k P_i U_i$, where P_i is the probability of state of nature i and U_i is the utility of state of nature i when that particular decision is made. With continuously distributed states of nature, the sum is replaced by an integral, which in our case cannot be analytically evaluated and must be approximated by a Monte Carlo average, that is, an average over many simulations using computer generated pseudo-random numbers. There is general agreement that when expected utilities are known, then the decision with the greatest expected utility should be chosen. In our case, expected utilities are known when we ignore uncertainty about biological parameters.

Often, however, one cannot objectively evaluate the probability distribution of the states of nature, and one is also unwilling to subjectively assess these probabilities. For us, this situation occurs when we include biological parameters as part of the state of nature. In this case there is no consensus on what decision is best, since there is no natural criterion, like expected utility, to optimize. In this case, it is worthwhile to calculate "opportunity losses". For a fixed state of nature, the opportunity loss for any decision is the difference between this decision's utility and the maximal utility over all possible decisions. Some decision makers would rule out decisions which can have a large opportunity loss, even if the probability of a large loss is unknown.

We decided to focus upon harvesting over a twenty-five year horizon, so that the stochastic steady state for the population would surely be reached. The initial population vector was the estimated (by cohort analysis) population in 1976, the last year for which we had estimated abundances of all age classes. The criterion to be maximized was UT , which was defined as

$$(3.1) \quad UT = E\left\{ \sum_{i=1}^{25} U(C_i) \right\}$$

where E means mathematical expectation, C_i is the catch (in biomass) in the i th year, and U is a utility function. Notice that C_i depends upon the random recruitments in years 1 through i and upon the harvesting policy.

Because there was no single utility function that seemed natural, we worked with two rather different utilities, in the hope that any reasonable utility function would fall somewhere between these extremes. The first utility function was $U(x) = x$, which has routinely been used in fisheries research and which has a constant marginal utility. With this choice of U , our criterion became the maximization of total expected catch over twenty-five years, with no attention paid to year-to-year variability of the catches. Such a utility is probably not completely desirable from several viewpoints but it represents one extreme.

The second utility function was $U(x) = \log(x+1)$, which has a quickly decreasing derivative, $U'(x) = 1/(x+1)$. An optimal policy under this utility must maximize the geometric mean of the catch. Also, when this utility function is used a harvesting policy with high year-to-year variability will have a lower value of UT than a policy with the same total expected catch, but less variation in catch.

4. CONSTANT CATCH HARVESTING POLICIES

Perhaps the simplest harvesting policy is to take the same size catch, call it C^* , each year. When choosing C^* , one must balance the immediate reward of a large C^* with the possibility of extinction. Maximization of UT , as defined in section 3, is one means of choosing this balance, since $C_i = C^*$ prior to extinction and $C_i = 0$ after extinction, if it occurs. Using stochastic approximation (Ruppert et al., 1982), we found that UT is maximized with C^* equal to 325 and 340 metric tons for $U(x) = \log(x+1)$ and $U(x) = x$, respectively.

In the context of simulation, "extinction" is biological extinction, but for the actual population we can think of extinction as collapse of the fishery. Because economic collapse is so drastic, instead of maximizing UT one might prefer to control the extinction probability. In figure 4.1, the curve labeled "no phase in policy" is a plot of the extinction probability against 25 times C^* , which is the total catch if extinction does not occur.

Although the 1976 population size is not small compared to recent historic values, it is smaller than equilibrium population sizes under harvesting policies discussed later. To determine whether constant catch policies would be more successful if the population was first allowed to increase, we defined a "phase in" policy where the catch is C^* for years 4 to 25, but is only 25%, 50%, and 75% of C^* for years 1, 2, and 3, respectively. Thus, provided extinction does not occur, the total catch over 25 years will be 23.5 times C^* . In figure 4.1 we plot the extinction probability against 23.5 C^* for the "phase in" policy. From this figure, we see that with an average yearly catch of 400 metric tons (total catch = 10,000 metric tons) the extinction probability is almost 0.15 with "phase in" but almost 1.0 without.

5. CONSTANT FISHING MORTALITY RATE POLICIES

For certain simple fisheries models, the optimal policy keeps the instantaneous fishing mortality rate constant (Beddington and May, 1977). We will call a constant-F policy one where the age 2 fishing mortality rate, which we call F_2 , is fixed. For our particular model, this implies that all other age-specific fishing mortality rates are constant; see section 2. For populations with highly variable recruitment, a constant-F policy, which adjusts catches to accommodate fluctuations in the population seems preferable to constant catch policies.

We simulated the fishery 250 times for each value of F_2 from 0.1 to 7.0, incremented by 0.1, and plotted the Monte Carlo average utility as a function of F_2 . The graph for $U(x) = x$ is given in figure 5.1. This curve of average utility has been smoothed with a spline on the SAS GRAPH software package, and it includes 95% confidence limits for the expected utility. For greater detail only F_2 from 0.1 to 4.0 is plotted. The graph for $U(x) = \log(x+1)$ is somewhat similar, and both have a maximum at about $F_2 = 0.8$. We have estimated that during the mid 1970's, the menhaden fishery was operating with F_2 around 1.8, which is clearly suboptimal (Ruppert et al., 1983).

In figure 5.2 we plot the expected catch in biomass for years 1 to 25 for the $F_2 = 0.8$ policy. The results of 250 Monte Carlo repetitions were smoothed using splines, and 95% confidence limits for the expected catch are also given. Notice that the expected catch rises nearly linearly for the first six years, rises more slowly for years 7 to 10, and then stays relatively constant at about 450. A slight dip around year 20 is most likely just a Monte Carlo random occurrence. Similar graphs show that population size, recruitment, and egg production each increase for 8-10 years and then reach equilibrium

(Ruppert et al., 1983). Egg production increases considerably more than recruitment, as would be expected since the Beverton-Holt curve implies that recruitment is virtually independent of egg production, for large values of egg production. Catch and population size in biomass increase considerably more than catch and population size in numbers, since a major effect of reducing F_2 to 0.8 is to increase the average age of the population.

6. "EGG ESCAPEMENT POLICIES"

Constant-F policies work well because they adjust the catch size to the population size. Since the menhaden population's age structure, as well as its size, is random, a better policy might be to adjust the F's so that they are higher in years when older cohorts dominate. Such a policy can be achieved by basing catch in a given year upon potential egg production, which we define as the egg production (for all age classes combined) which would be obtained if there was no fishing during the year. Let C and E_p be catch and potential egg production for a given year.

A flexible class of harvesting policies can be obtained by letting

$$\begin{aligned} C &= A(E_p - B)^\theta && \text{if } E_p > B \\ &= 0 && \text{if } E_p \leq B, \end{aligned}$$

where A, B , and θ are parameters.

If $\theta = 1$ and $B > 0$, then by choosing A properly, one can keep actual egg production fairly constant (the so-called constant escapement policy). However, egg production cannot be kept exactly constant, since egg production per unit biomass varies with the random age structure.

Letting $\theta = 1$ results in great fluctuations in catches, whereas $\theta = 0$ results in constant catches. Having $0 < \theta < 1$ might be a good compromise between these extremes, so we investigated policies with $\theta = \frac{1}{2}$ as well as those with $\theta = 1$. The $\theta = 0$ policies have been discussed in section 4. For each value of θ and form both $U(x) = x$ and $U(x) = \log(x+1)$, we used stochastic approximation to find the values of A and B optimizing expected utility (Ruppert et al., 1982).

Figures 6.1 and 6.2 show the catch functions given by equation (6.1) with the optimal A and B values. The optimal constant catch policies are also shown. When $\theta = \frac{1}{2}$, the optimal policy depends very little upon the choice of U, but if $\theta = 1$ then A and B are markedly different for the two utility functions. When $U(x) = \log(x+1)$, the optimal policy dampens fluctuations in catch size by having $B < 0$ and the slope A relatively small.

For each of the four optimal egg escapement policies, ($\theta = \frac{1}{2}$ or $\theta = 1$ and $U(x)=x$ or $U(x) = \log(x+1)$) we plotted catch and population size (both in numbers of fish and biomass), recruitment, and potential and actual egg production against the simulation year. The four "egg escapement" policies and the $F_2 = 0.8$ policy behave somewhat similarly, but with a few notable differences. For the "egg escapement" policies, catch size stabilizes at higher levels than the $F_2 = 0.8$ policy. Also, the five policies have substantial differences in catch variability and in expected catch during years before equilibrium is reached. In table 6.1 means and standard deviations of catch (in biomass) are given for years 1 to 8 and year 15, the latter being typical of years 9 through 25. The "egg escapement" policy which is optimal for $\theta = 1.0$ and $U(x) = x$ approaches equilibrium very quickly and has a highly variable equilibrium catch, compared with the four other policies, which are similar among themselves, except that the $F_2 = 0.8$ policy has a considerably smaller expected equilibrium catch.

7. RISKS DUE TO PARAMETER ESTIMATION ERROR

Until now we have ignored risks due to estimation error. To assess these risks we estimated the standard errors for most estimates in the model. Then for each parameter we found the opportunity loss caused by estimation error.

Because of the large data base available, growth parameters were estimated relatively precisely, so we concentrated on four other parameters: m , the instantaneous natural mortality, and the parameters α , β , and σ in equation (2.2).

The parameter σ was estimated by regressing $1/N_0$ on $1/E$. Of course, E is estimated with error and this would cause σ to be overestimated, since variability in the estimation of E would be attributed to variability in recruitment. Errors in E also probably biased our estimates of α and β (Ludwig and Walters, 1981), but lacking knowledge of the structure of the errors we have not attempted bias corrections.

Estimates of m , α , β , and σ are in table 7.1. We calculated opportunity losses associated with estimation errors of magnitude equal to one standard deviation. Since m appears to be a crucial parameter, we also considered misestimation of m by two standard deviations. As σ is likely to be smaller than $\hat{\sigma}$, we calculated the opportunity loss if $\sigma = (0.7)\hat{\sigma}$. The value 0.7 was chosen simply because some definite value was needed. We did not know the real magnitude of overestimation of σ , but recruitment certainly does vary greatly, and we felt it unlikely that σ is much less than $(0.7)\hat{\sigma}$.

In table 7.2, the calculation of opportunity losses are given for the four optimal "egg escapement" policies. To illustrate one calculation, suppose we are using $U(x)=x$ and $\theta=1$ and we want to calculate the opportunity loss if m is two standard deviations above its estimate. The optimal values of A and B were calculated for $m = \hat{m}$ and $m = \hat{m} + 2 s(\hat{m})$ (where $s(\hat{m})$ is the standard error

of \hat{m}). From table 7.2 we see that if $m = \hat{m} + 2s(\hat{m})$ and the values of A and B which are optimal for this m are used, then the expected utility (as estimated by simulation) is 8453.5. However, we believe that $m = \hat{m}$, so we would use the values of A and B which are optimal for $m = \hat{m}$. Here the expected utility is 7732.8. The opportunity loss is $8453.5 - 7732.8 = 720.7$. This is the decrease in total utility (total catch over 25 years) caused by using the policy optimal for $m = \hat{m}$ when, in fact, $m = \hat{m} + 2s(\hat{m})$. The opportunity loss is less than 10% of the expected utility, which does not seem very big, considering that an estimation error of two standard errors has only about a 5% chance of occurring.

Table 7.2 shows that opportunity losses are extremely small when $U(x) = \log(x+1)$, and moderately small when $U(x) = x$. The worst cases are when m is misestimated.

A sensitivity analysis was also made. By sensitivity we mean the change in the expected utility of a fixed policy due to a change in a parameter. This can be large. For example, let $U(x) = x$ and $\theta = 1$ and consider the policy which is optimal if all parameters are equal to our estimates. The expected utility is 12596.5 if $m = \hat{m}$ but only 7732.8 if $m = \hat{m} + 2s(\hat{m})$. That is a 39% change. Although estimation errors can only cause our "egg escapement" policies to be slightly suboptimal compared to what would be obtainable if the parameters were known, estimation errors do prevent us from predicting average future catches with great accuracy.

8. CONCLUSIONS

Our analysis suggests that Atlantic menhaden were much over-exploited in the mid-1970's when the fishery was operating with F_2 around 1.8, more than twice the fishing mortality ($F_2 = 0.8$) we found to be optimal for maximizing catch and logarithm of catch. We looked for reasons why this relatively low F_2 was optimal and found that to a large extent it is due to increases in yield per recruit accrued by allowing mean age in the stock to increase. Growth and natural mortality data show biomass of a cohort peaks between ages 2 and 3 years, whereas current fishing intensity allows few menhaden to survive even to age 3 years.

Policies that allow for increases in the long-term harvest are numerous. Constant fishing mortality and egg escapement based policies all work well. Constant catch policies where annual quotas remain at fixed level are not as desirable, because of high calculated extinction probabilities when the quota exceeds 400 thousand metric tons. Table 6.1 shows that catches are slightly lower during the 25 simulated fishery years for the constant fishing rate policy as compared to egg escapement based policies, with the highest total catch (table 7.2) obtained by essentially a constant escapement policy. But the constant escapement policy has a higher standard deviation of catch than the constant fishing rate and the compromise ($\theta = \frac{1}{2}$) policies. In fact, maximum geometric mean of catch (where we use a log utility function) is actually higher for the compromise ($\theta = \frac{1}{2}$) policy as compared to constant escapement because of the increased stability of catch.

Estimation errors for parameters in our model cause us to be uncertain as to the amount of catch obtainable under the optimal "egg escapement" policies. However, since opportunity loss is rather small if a mistake is made in parameter specification (table 7.2), we can be reasonably certain that following an optimal policy will result in nearly the best obtainable catch.

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Table 6.1. Mean and standard deviation of total catch. The first value is the mean and the second is the standard deviation. The population is approximately in equilibrium by year 9. Years 1 to 8 show the approach to equilibrium. Units in thousands of metric tons.

Policy	Year									
	1	2	3	4	5	6	7	8	15	
$F_2 = 0.8$	236 1	368 12	373 79	396 186	437 235	441 221	465 257	452 240	450 220	
$\theta = 1.0$ A = 2.36 B = 77.6 Optimal for $U(x) = x$	124 0	421 21	329 128	583 529	538 440	522 408	556 432	512 365	528 406	
$\theta = 1.0$ A = 1.21 B = -80.5 Optimal for $U(x) = \log(x+1)$	256 0	355 11	327 64	446 227	484 248	499 236	524 261	513 238	527 244	
$\theta = 0.5$ A = 12.0 B = 124 Optimal for $U(x) = x$	122 0	446 13	345 92	490 202	505 225	519 220	533 235	521 231	525 221	
$\theta = 0.5$ A = 11.0 B = 108 Optimal for $U(x) = \log(x+1)$	166 0	405 12	339 78	464 179	491 203	509 202	526 215	518 212	527 203	

Table 7.1. Estimates of biological parameters used in the stochastic simulation model

<u>Parameter</u>	<u>Notation</u>	<u>Estimate</u>	<u>Standard Deviation of Estimate</u>
Natural mortality rate	m	0.50200	0.0914
Constant in Beverton- Holt curve	α	0.00019750	0.000047
Slope in Beverton- Holt curve	β	3.17695	1.38619
Standard deviation in Beverton-Holt curve	σ	0.00012591	-

Table 7.2. Calculation of opportunity losses. The upper value is the expected utility for the policy which is optimal for the changed value of the biological parameter. The middle value is the expected utility for the policy which is optimal for the parameters equal to the estimates given in table 6.2. The lower value, the difference between the two, is the opportunity loss.

Change in parameters from estimates given in table 6.2	u(x)=x		u(x)=log(x+1)	
	$\theta=1$	$\theta=.5$	$\theta=1$	$\theta=.5$
None	12596.5 <u>same</u> 0	12501.3 <u>same</u> 0	153.145 <u>same</u> 0	153.250 <u>same</u> 0
m is 2 s.d.'s above estimate (m = .6848)	8453.5 7732.8 <u>720.7</u>	8267.7 7680.0 <u>587.7</u>	143.407 143.049 <u>0.358</u>	142.973 139.578 <u>3.395</u>
m is 1 s.d. above estimate (m = .5934)	10298.0 10078.6 <u>219.4</u>	10158.6 9984.8 <u>173.8</u>	148.260 148.068 <u>0.192</u>	148.327 147.483 <u>0.844</u>
m is 1 s.d. below estimate (m = .4106)	15543.7 15236.0 <u>307.7</u>	15530.1 15375.4 <u>154.7</u>	158.492 158.197 <u>0.295</u>	158.714 158.634 <u>0.080</u>
m is 2 s.d.'s below estimate (m = .3192)	19827.5 17956.6 <u>1870.9</u>	19654.1 18978.2 <u>675.9</u>	164.223 163.119 <u>1.104</u>	164.872 164.087 <u>0.785</u>
$\sigma = .7 \hat{\sigma}$ ($\sigma = .881392$)	10649.3 10630.2 <u>19.1</u>	10642.6 10637.6 <u>05.0</u>	150.353 150.254 <u>0.099</u>	150.429 150.247 <u>0.182</u>
α is 1 s.d. above estimate ($\alpha = .0002444$)	10199.4 10169.3 <u>30.1</u>	10159.1 10154.5 <u>04.6</u>	148.685 148.578 <u>0.107</u>	148.760 148.502 <u>0.258</u>
α is 1 s.d. above estimate ($\alpha = .0001506$)	16040.8 16006.9 <u>33.9</u>	15880.9 15831.7 <u>49.2</u>	158.745 158.698 <u>0.047</u>	158.928 158.788 <u>0.140</u>
β is 1 s.d. above estimate ($\beta = 4.5631$)	11962.8 11949.1 <u>13.7</u>	11918.8 11900.2 <u>18.6</u>	152.029 151.928 <u>0.101</u>	152.205 152.083 <u>0.122</u>
β is 1 s.d. below estimate ($\beta = 1.7908$)	13377.1 13322.7 <u>54.4</u>	13215.2 13149.4 <u>65.8</u>	154.693 154.397 <u>0.296</u>	154.823 154.452 <u>0.371</u>

Figure 2.1. Flowchart of a year of the stochastic simulation model with a constant fishing mortality rate policy in use. c_i is a constant contribution of eggs in age class i . b_i is the effect of density dependence on egg production (E). $N_{0,i}$ is the abundance prior to recruitment of a cohort age i . Likewise, z_i is a constant average weight (W) in age class i and U_i is the effect of density dependence on growth. The recruitment function is the inverse form of the Beverton-Holt equation 2.2 ($s=\sigma$). R_i is the average instantaneous fishing mortality for years 1973-1975 in age i divided by the average instantaneous fishing mortality for the same period in age 2. $R_i=1$ when $i=2$; $R_i < 1$ when $i < 2$; and $R_i \sim 1$ when $i > 2$.

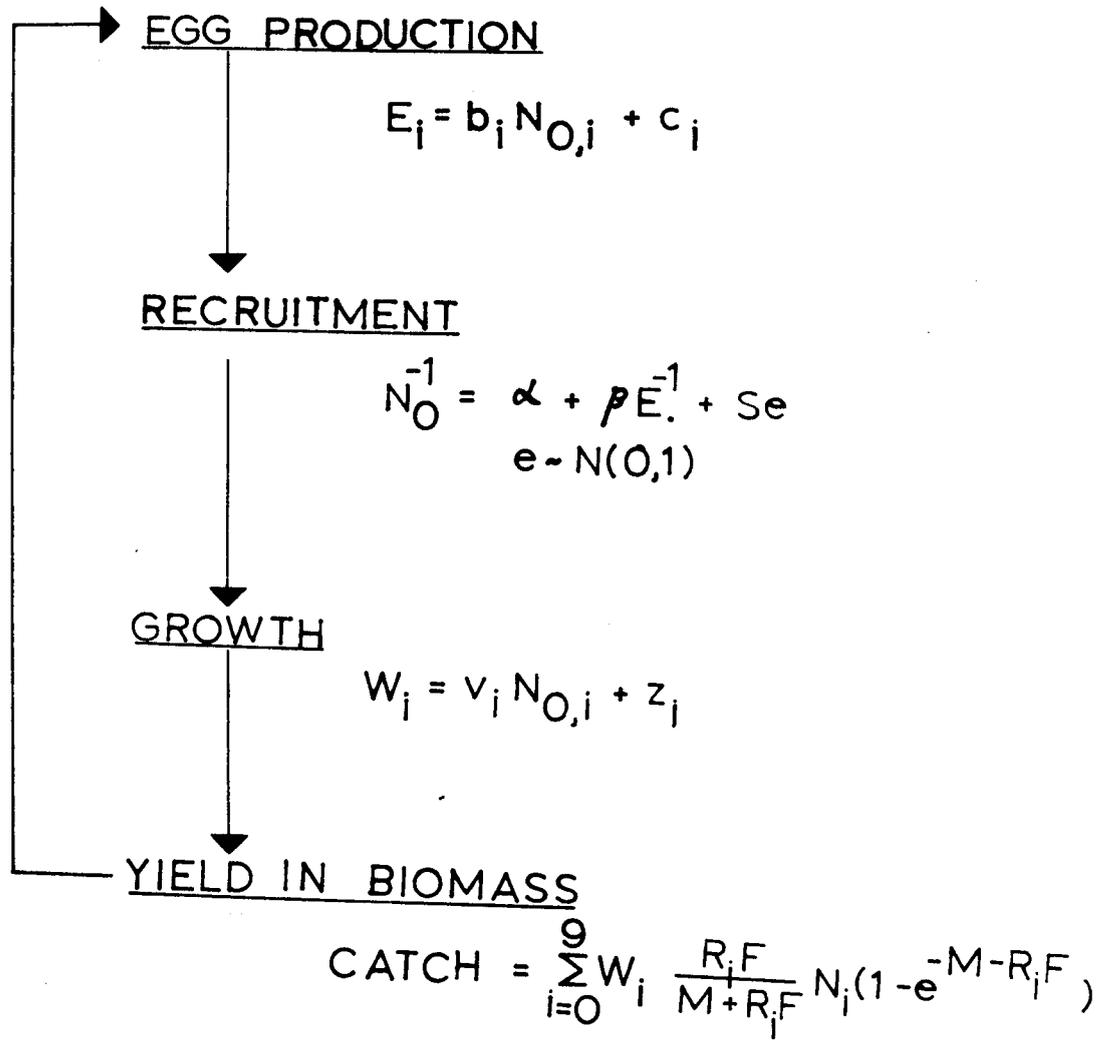


Figure 2.1

Figure 3.1. Three examples of utility functions. Under the risk neutral utility, the value of the catch is proportional to its amount. The risk averse, risk neutral, and risk preferring (or prone) utility functions have decreasing, constant, and increasing marginal utility functions (first derivatives), respectively.

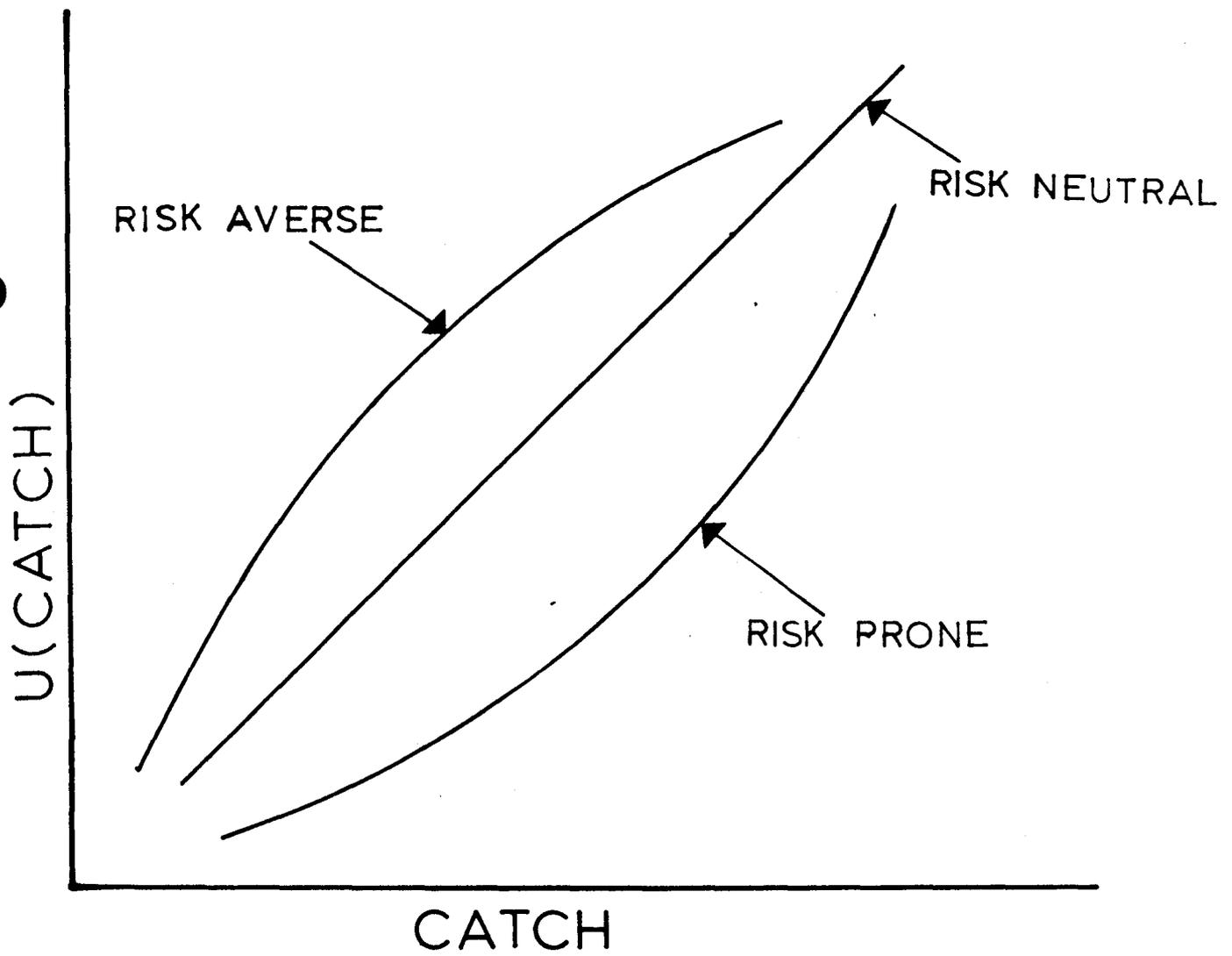


Figure 3.1

Figure 4.1. The probability of extinction for the constant catch policies.

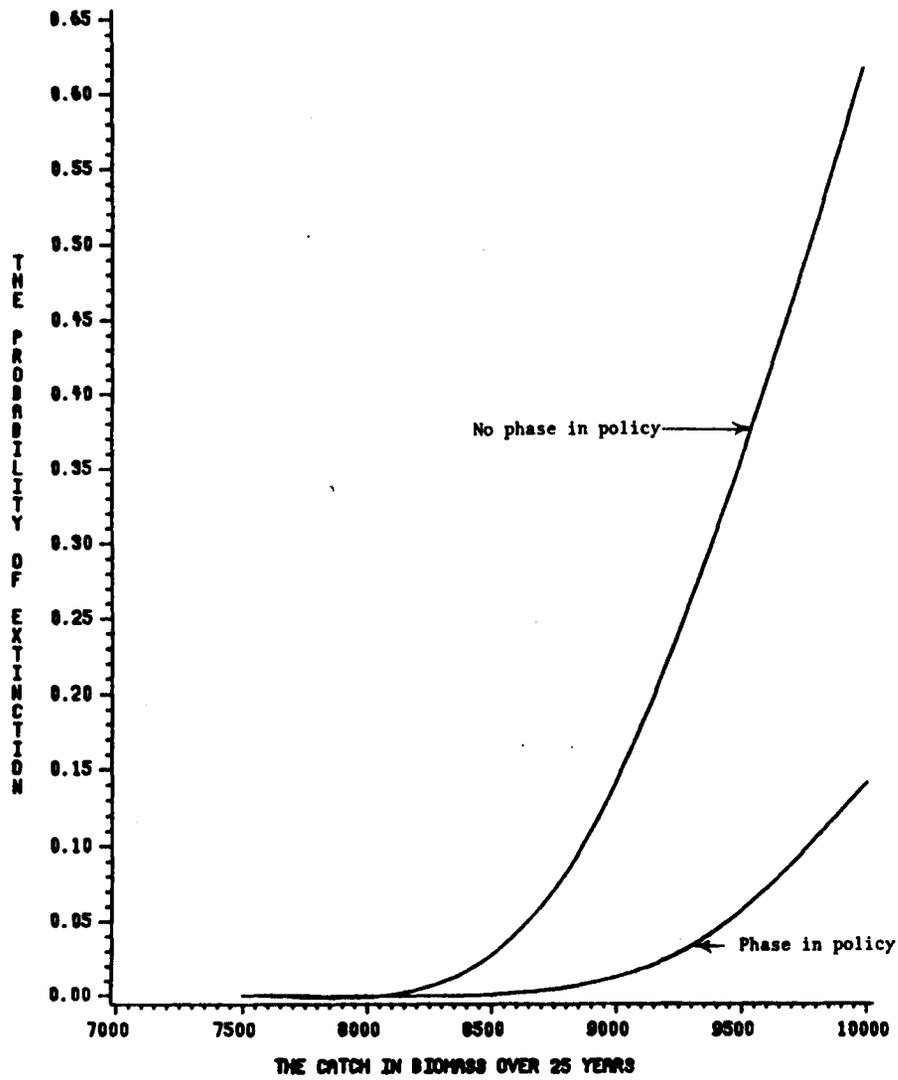


Figure 4.1

Figure 5.1. Catch in biomass plotted against the yearly fishing mortality rate. The estimated mean and 95% confidence intervals for the mean are given. The unit of catch is thousands of metric tons.

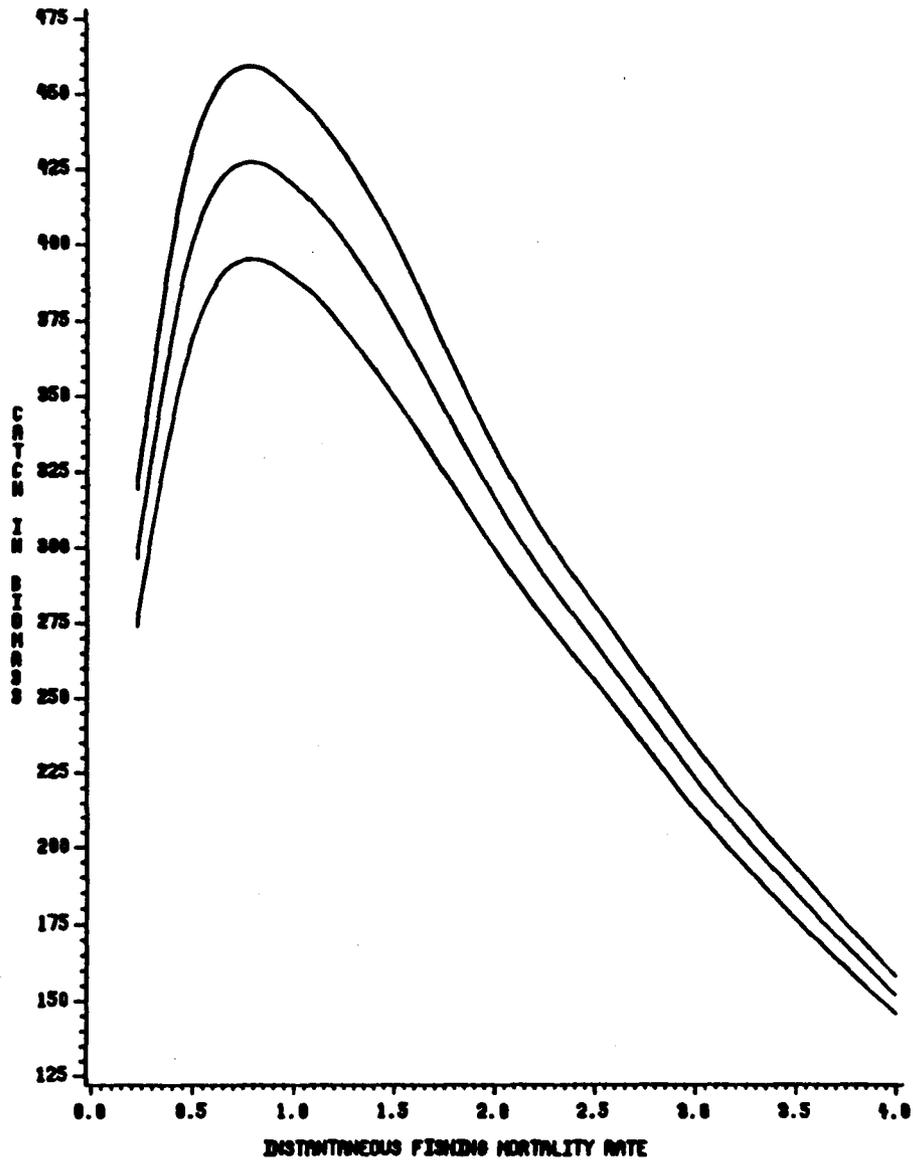


Figure 5.1

Figure 5.2. Catch in biomass plotted against year. The optimal constant fishing mortality rate ($F_2 = 0.8$) policy was in effect. The estimated mean and 95% confidence limits for the mean based on 250 Monte Carlo trials are given. The unit of catch is thousands of metric tons.

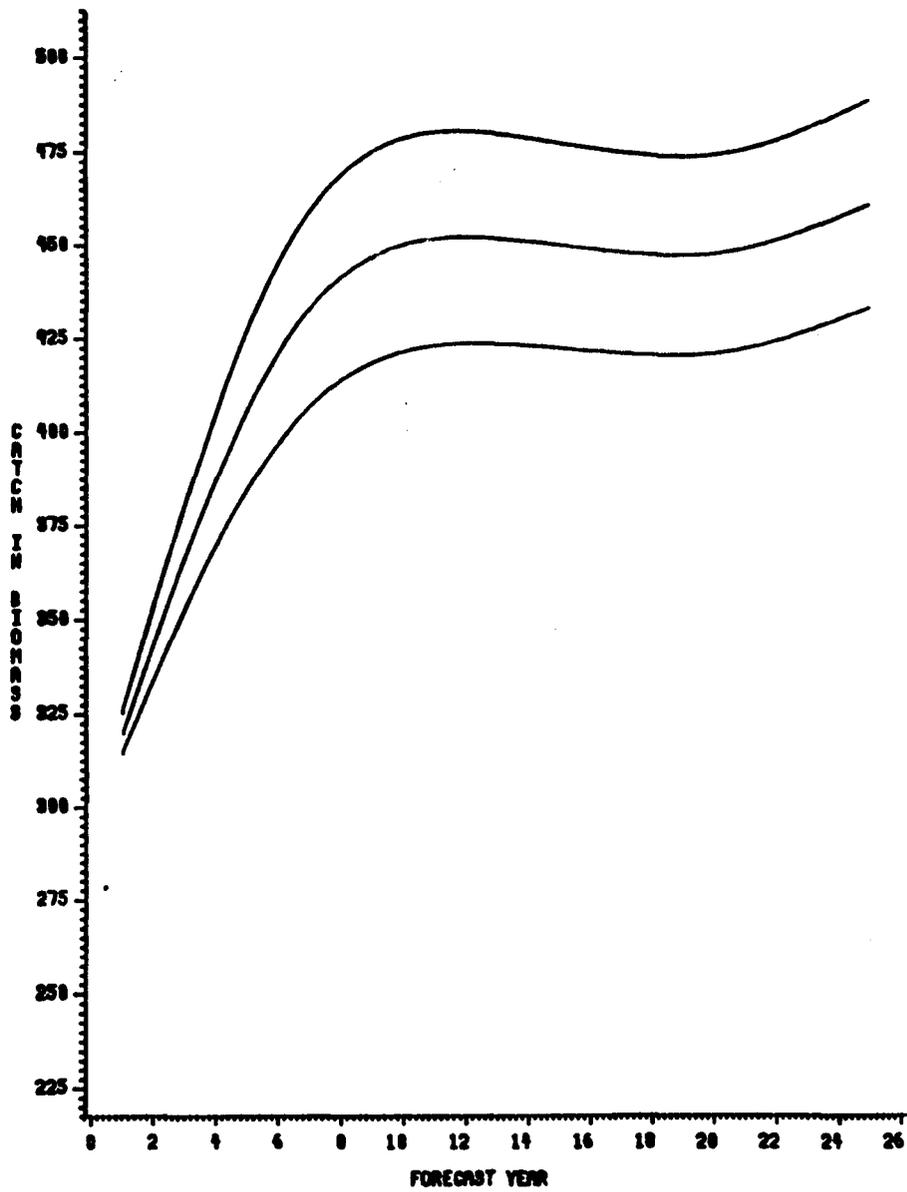


Figure 5.2

Figure 6.1. Optimal catch functions for "egg escape-
ment" policies when the utility function
is $U(x)=x$. The unit of catch is thousands
of metric tons.

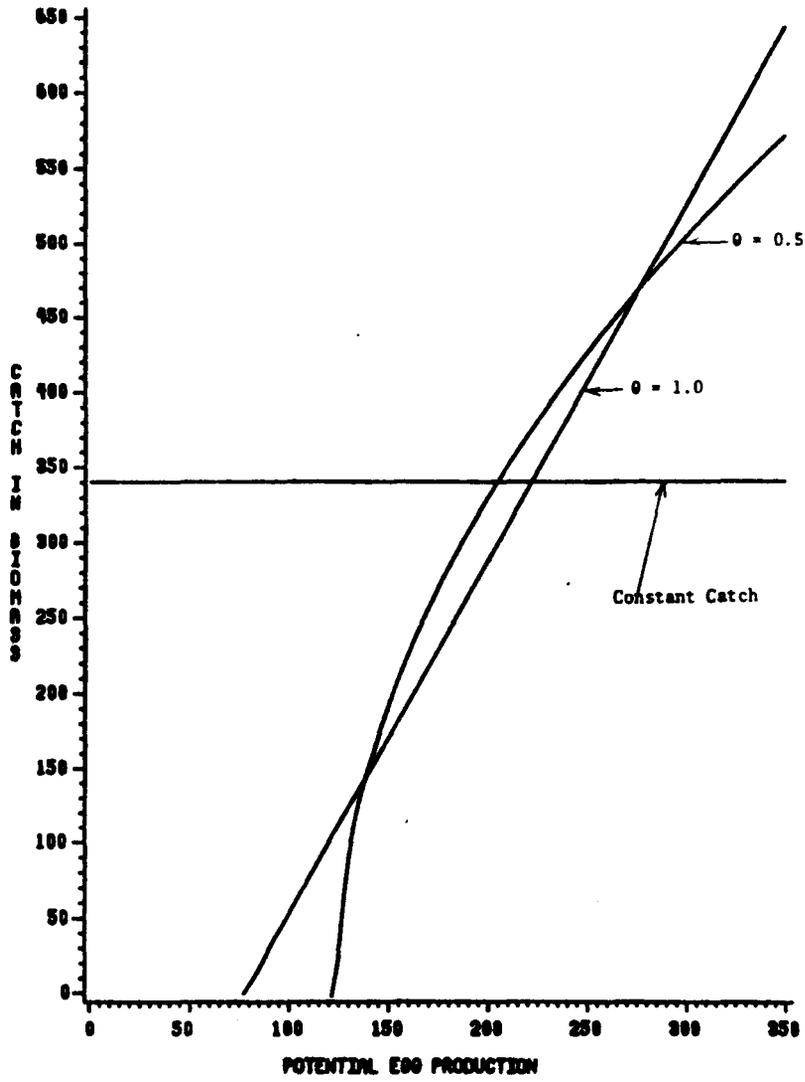


Figure 6.1

Figure 6.2. Optimal catch functions for "egg escape-
ment" policies when the utility function
is $U(x) = \log(x+1)$. The unit of catch is
thousands of metric tons.

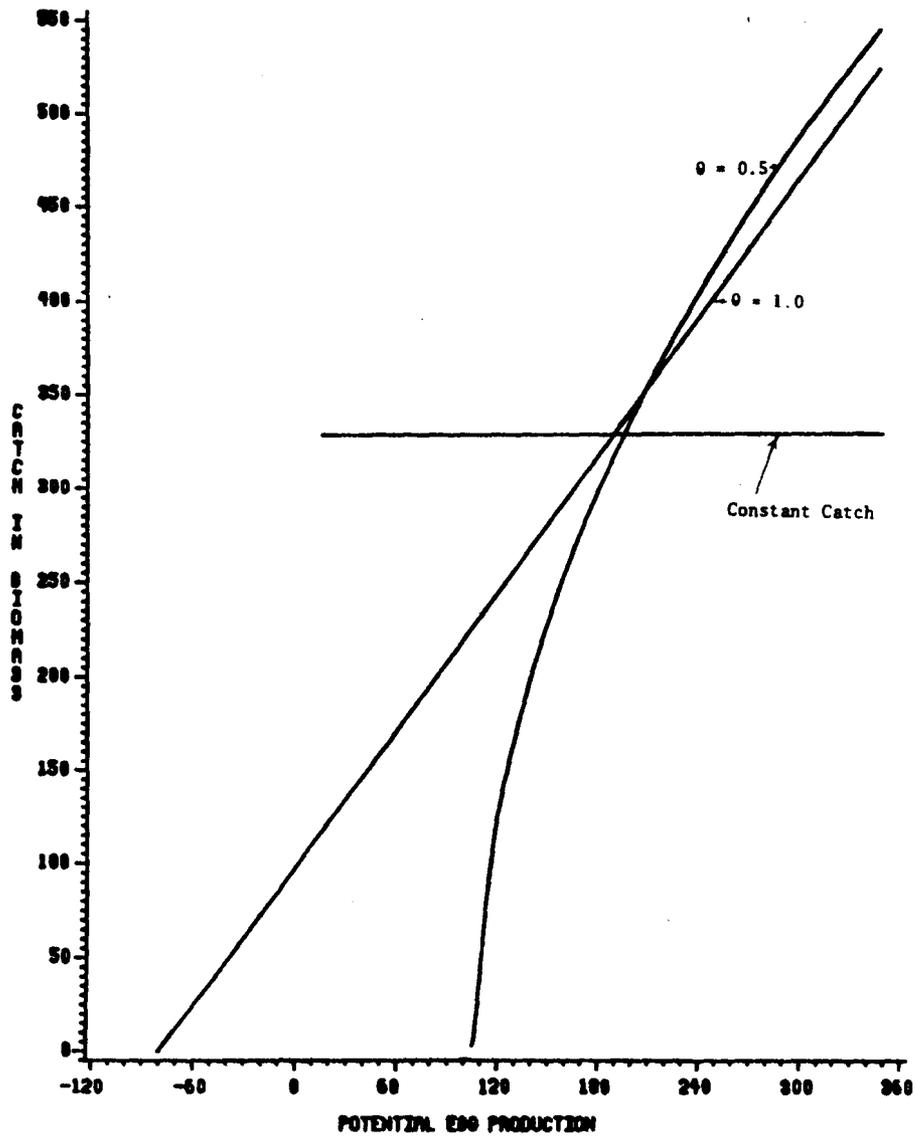


Figure 6.2