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Sustainable fishing patterns for geoduc clam (*Panopea zelandica*) populations in New Zealand

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SUSTAINABLE FISHING PATTERNS FOR GEODUC CLAM (*Panopea zelandica*) POPULATIONS IN NEW ZEALAND

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1. INTRODUCTION

1.1 Overview

Geoducs (*Panopea zelandica*) are subtidal (3-20 m water depth) clams living beneath the sediment surface on clean silt or sandy substrates. A similar species, *Panopea abrupta*, supports the most important invertebrate fishery on the west coast of Canada (Harbo *et al.* 1992), and is also fished in the states of Alaska and Washington, USA.

In New Zealand, geoducs are also known as geoducks, king clams, and deepwater clams. The Maori name is hoehoe or hohehohe (Strickland 1990).

A small fishery has taken *P. zelandica* from Golden Bay since 1988. Preliminary studies (Breen 1991; Breen *et al.* 1991) examined growth rings in the shell and made estimates of growth and mortality rates. The rings have not been validated as annual rings, but similar rings in *P. abrupta* are well validated (Shaul & Goodwin 1982). An attempt to validate the ageing is described.

Growth and mortality rate estimates given here for *P. zelandica* are subject to confirmation of the rings as being annual. The importance of validation is well recognised (Beamish & MacFarlane 1983). At the same time, it is useful to explore the implications of preliminary results under the assumption that rings are annual. The method is a standard one, validated in a similar species living in a similar habitat, so the rings may well be annual. An attempted validation of the rings as annual rings from examination of modal progression is described.

This paper describes the results from a simple age-structured stochastic model used to examine the impacts of various constant-catch and pulse fishing strategies. The model was initialised from preliminary estimates of longevity, growth, natural mortality, recruitment and recruitment variability. Assumptions were then made about the incidental mortality of fishing on juveniles and the form of the stock-recruit relation. The fishing patterns examined were combinations of constant catches (annualised catches in the range 0.5-8.0% of the initial biomass, B_0) and rotation periods for pulse fishing (in the range 1-14 years).

The same model was used by Breen (in prep.) to estimate sustainable fishing patterns for *P. abrupta*.

For each set of 100 runs, statistics were calculated for 6 fishery indicators. Three indicators related to the population biomass and three to the ability of the fishery to take the target catch.

Appropriate fishing patterns could then be identified for the 6 indicators defined. Pulse fishing has no obvious biological benefits as part of a fishing pattern, but confers an economic advantage. Implications of pulse fishing are useful to explore, even if populations are not explicitly pulse-fished, because real fishing patterns are not uniform in space and time.

Sensitivity of one indicator is examined with respect to the major uncertainties: result indicate the major areas where further research would increase the effectiveness of management.

1.2 Description of the fishery

The fishery was described by Breen (1991) and Breen *et al.* (1991). Two species, *P. zelandica* and *P. smithae*, occur all around New Zealand, but only *P. zelandica* has appeared in the catches examined. Distributions of the species are not well described. *P. zelandica* is known from many sites all around New Zealand, but has been found in commercially abundant quantities in only a few places.

An experimental fishery began in Golden Bay under special permit in 1988 and continued for several years at a relatively low level (on the order of 100 t annually). The fishery is now closed pending the introduction of new species into the quota management system. No estimates of density or biomass have been made.

The biology of and fishery for the similar *P. abrupta* are described by Harbo *et al.* (1992) and Goodwin & Pease (1989).

2. MODEL AND MODEL PARAMETERS

2.1 Objectives of the model

The fishery for *P. zelandica* could be managed with a fixed catch strategy. Such a strategy is currently used to manage the fishery for British Columbia geoducs. For each fishing area, B.C. managers estimate the area of geoduc clam beds. Then they estimate initial biomass from the bed area, an assumed mean density for the region, a mean weight per individual, and the historic catch (Harbo *et al.* 1992). The allowable annual harvest is calculated by multiplying estimated initial biomass times a target harvest rate.

Under this approach, the two major problems for stock assessment are estimating the initial biomass and estimating the appropriate target harvest rate. Sloan (1985) discusses approaches to the first problem; this study deals solely with the second.

This study examines the consequences of both fixed catch level and "pulse fishing". In pulse fishing, populations are harvested in rotation every θ years at θ times the annual harvest rate. A possible economic advantage is that the fishery harvests from a higher density population, thus reducing fishing costs. A possible disadvantage of pulse fishing is that as θ becomes large, the target catch becomes an unrealistically high proportion of the remaining population. At high values of θ, the specified catch level cannot be obtained.

Even where pulse fishing is not conducted explicitly, populations could be fished erratically: heavy pressure might be applied to one area in one year, then moved elsewhere for some years before being returned to the original area. Examining the effects of pulse fishing gives clues to the prediction errors caused by assuming a constant annual harvest rate for any given population.

Determining the most appropriate fishing pattern depends heavily on the management goals defined for the fishery. In the absence of explicit fishery goals for *Panopea zelandica*, this study defines and uses fishery indicators such as mean biomass and mean catch based on recent stock assessments for other New Zealand species (eg. Annala 1992). These may or may not be the most appropriate indicators for the New Zealand geoduc fishery. The model described could easily be modified to incorporate other indicators considered important; or indeed another fishery management strategy.

2.2 Overview of the model

The model is a simple age-structured model written in Turbo Pascal[1]. It first simulates an initial population with an arbitrary value for mean recruitment. This population occupies an unspecified area, ie population density is not used or calculated. Stochastic recruitment around a Beverton-Holt stock-recruit function is simulated by specifying a coefficient of variation (CV) for annual recruitment, based on real data. When the initial population is established, past recruitments are treated stochastically; thus the initial population varies in size between runs because of recruitment variability.

The model contains a specified number of age classes or cohorts; sexes are not treated separately. Each year, after the simulated catch (if any) has been removed, the model simulates ageing and growth, natural mortality and recruitment. Numbers-at-age and weight-at-age are used to calculate cohort and total population biomass.

Each model run begins with a new stochastic initialisation and continues for a specified number of years. A set of runs comprises a specified number of runs, all made with the same parameter values but with recruitment varying between years and the pattern of recruitment varying between runs. Between sets of runs, the same string of random numbers is used to provide the same pattern of recruitment variability. Thus when similar sets of runs are compared, all the variation among sets has been caused by the changed parameters.

For each run, six fishery indicators are calculated. The mean and distribution of each indicator are calculated for each set of runs.

Where possible, parameter values were chosen after consideration of the published literature and analysis of unpublished data. Sensitivity of the results to the most uncertain

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[1] Borland International, Inc., 1800 Green Hills Road, P.O. Box 660001, Scotts Valley, California 95067-0001 USA. Mention of commercial products does not constitute endorsement.

parameters is reported. The major parameters are described below and summarised in Table 1.

2.3 Age and mortality

Panopea zelandica appears to be moderately long lived. Breen *et al.* (1991) report a mean 'age' of 13 years in market samples, based on ring counts, and individuals with up to 34 rings. A maximum age of 34 suggests a natural mortality rate of about 0.14 from the assumption that 1% of a cohort reaches maximum age. The model normally simulated 40 age classes. After 40 years all clams were assumed to have died.

In an attempt to validate the ageing, a sample of clams was taken from a site offshore of Collingwood in late March/early April 1993. The site had previously been sampled in mid March, 1991, and the site could be revisited reasonably accurately because GPS was available both times. In March 1991, 297 clams were collected; in 1993, 124 clams were collected. The ring count frequency distributions were compared between the two surveys, with the expectation that peaks in the distribution would have shifted by two if the rings were annual. Results (Breen, unpub. data) were consistent with the hypothesis that rings are annual, but a larger sample is required to validate the rings convincingly.

Ageing in the model population takes place each year according to:

1. $$N_{k+1,t+1} = N_{k,t} \exp(-M)$$

where k indexes cohort and t indexes year. The instantaneous rate of natural mortality, M, was assumed to be 0.15, based on the longevity. Breen *et al.* (1991) reported an estimate of 0.20, based on the catch curve; later ageing indicated so much temporal and spatial recruitment variability that this estimate has been rejected. The sensitivity of results to the assumed M was also explored.

2.4 Recruitment

The age of maturity is unknown for this species. Breen *et al.* (1991) found that all clams they examined, (lowest age 3, smallest size 61 mm) appeared to be mature. This would indicate an early age at maturity. In what follows, "biomass" means "recruited biomass", ie those animals 8 yrs and older.

The mean recruitment (the number of individuals $N_{1,t}$) for any time t was arbitrarily set at one million for a virgin population and populations greater than the initial state. For populations of less than the initial biomass, the expected recruitment was determined from a Beverton-Holt stock-recruitment function:

2. $$R_t = B_t \,/\, (\alpha + (\beta\, B_t))$$

where R_t is the expected recruitment to the population (at age 1), B_t is the population biomass (recruited to the fishery) in year t, and α and β are parameters. For convenience, the parameter "steepness" was used: this is the percentage of the virgin level of recruitment that occurs when the population is at 20% of initial biomass. Steepness (*s*) can vary from 0.2 (recruitment directly proportional to biomass) to 1.0 (recruitment constant at the initial level). The parameters are calculated from

3. $\alpha = (B_0 / N_{\bullet,0}) (1 - ((s - 0.2) / (0.8\ s)))$

4. $\beta = (s - 0.2) / (0.8\ s\ N_{\bullet,0})$

where $N_{\bullet,0}$ and B_0 are the numbers and biomass of the initial population B_0 respectively.

Nothing is known about steepness in *Panopea zelandica* populations. It is unrealistic to expect recruitment to remain high when biomass approaches zero, but bivalve molluscs often experience high recruitment from depressed spawning biomasses (eg. Hancock 1973). Some *Panopea zelandica* live at depths beyond those which the fishery can reach, at densities too low to be fished, and in substrates that are difficult to dig. The fished population is thus only part of the breeding population.

The steepness was assumed to be $s = 0.75$, and sensitivity to other values was examined.

Stochastic recruitments were calculated from

5. $N_{1,t} = \exp\ (\mu + (RND\ \sigma_R\ ))$

where *RND* is a random normal deviate (mean zero and standard deviation one) and μ and σ_R are calculated from

6. $\mu = \ln(R_t\ ) - (0.5\ \sigma_R^2)$

7. $\sigma_R^2 = \ln(CVRect^2 + 1)$

where *CVRect* is the coefficient of variation of recruitment.

Recruitment variability was estimated from a sample of geoducs taken from near Collingwood in March 1990 (Fig. 1). Past recruitment for the *k*th cohort was estimated from

8. $N_{1,t\text{-}k} = N_{k,t} \exp(M\ k)$

where *k* is age (ring count) and *t* is 1990. From the data in Fig. 1, $CVRect = 0.80$. A similar procedure applied to Canadian population data (Breen & Shields 1983) gave a *CVRect* of 0.6-0.9 depending on the range of ages used. The model assumed $CVRect = 0.80$.

Random numbers were obtained simply from the Pascal Random function. The first 1000 *RND*s used to generate recruitment was tested in several ways. The mean was tested against zero, the standard deviation was tested against one, the autocorrelation between successive *RND*s was tested against zero, and the distribution of "runs" of consecutive positive or negative RNDs was tested against the predicted binomial distribution. All tests were non-significant, so the simple random number function was accepted.

2.5 Growth

Growth of *Panopea zelandica* length appears to be rapid for about ten years, then becomes very slow (Breen 1991; Breen *et al.* 1991). The same pattern is reported for *P. abrupta* (eg. Goodwin 1976, Breen & Shields 1983, Harbo *et al.* 1983).

Growth in weight shows a more continuous increase (Fig. 2; see also Breen *et al.* 1991). Part of the increase in weight is due to an increase in shell weight with age, but the meat weight also continues to increase with age. Quotas are based on whole weight, so whole weight was modelled.

Two functions were explored: the von Bertalanffy equation based on weight and the Gompertz (see Ricker 1975):

9. $$Wt_k = W_\infty\,(1 - \exp(-K\,(k - t_0)))$$

10. $$Wt_k = W_0\,\exp(G\,(1 - \exp(-g\,k)))$$

where Wt_k is mean whole weight at age k. Both were fit to the data in Fig. 2 with FISHPARM (Saila *et al.* 1988). The better fit was obtained with (9), so that was used to model growth (Fig. 2).

2.6 Initialisation

For each set of runs, the model uses the same string of random numbers for stochastic recruitment if each set has the same runlength and number of runs.[2] At the beginning of each run, the model estimates the numbers-at-age as

11. $$N_{k,1} = \exp(\mu + (RND\ \sigma_R))\ \exp(-M\,(k - 1))$$

The constant μ is calculated using (6) with an expected recruitment, R_t, of one million. This procedure simulates stochastic variation in past recruitment to establish the initial population.

Initial biomass is then calculated and used to calculate the target catch from the specified quota, and in calculating some of the fishery indicators. B_0 varies between runs, depending on the mean strength of past recruitment.

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[2] Sets of runs are not be based on the same random number string, and thus cannot be directly compared, if either the number of years per run or the number of runs is varied between sets. Otherwise, sets of runs are directly comparable.

2.7 Fishing

The fishery for *Panopea zelandica* may be seasonal. For *Panopea abrupta*, there is differential visibility of geoducs to divers at different seasons (Goodwin 1977, Turner & Cox 1981). Because of this probable seasonality, and because the instantaneous fishing mortality rates are relatively low, the fishery is simulated as a Type I fishery (Ricker 1975), in which fishing occurs at the beginning of the year and natural mortality occurs after fishing ends.

Fishing always takes place in model year 1. The *quota* is specified as a fixed percentage of B_0 to be taken annually. If fishing takes places each year, then the *target catch* is that proportion of B_0. Otherwise

12. $$\textit{target catch} = (\textit{quota}/100)\, B_0\, \theta$$

The finite rate of exploitation each year is calculated as

13. $$u_t = \textit{target catch} \;/\; B_t$$

and the population is reduced accordingly:

14. $$N_{k+1,t+1} = N_{k,t}\,(1 - u_t)\,\exp(-M)$$

Recruitment to the commercial fishery may begin at age 4 and appears substantial by age 8 (Breen *et al.* 1991). The model assumed knife-edged recruitment at age 8.

For large θ, u_t becomes relatively large (approaches or even exceeds 1.0). It is unrealistic to expect the fishery to remove a large fraction of the remaining population. Not all individuals are visible at any one time (Goodwin 1977, Turner & Cox 1981); fishing causes disturbance and individuals retract their siphons, becoming cryptic; and the fishery will probably abandon areas where low availability of clams causes a poor economic return (Harbo & Peacock 1983). The maximum realistic exploitation rate, u_{max}, is assumed to be 0.35. In other words, in the model, only 35% of the standing stock in the area fished could be taken by the fishery in a single year. This was arbitrary, and chosen based on the percentage of geoducs that "show" at any one time (see Goodwin 1977), and the disturbances created by fishing that cause other clams to withdraw their siphons.[3]

Sensitivity of the results to this assumption is examined below. Where the *target catch* exceeds $u_{max}\,B_t$, equation 14 is replaced with

15. $$N_{k+1,t+1} = N_{k,t}\,(1 - u_{max})\,\exp(-M)$$

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[3] Recent information from L. Goodwin (pers. comm.) suggests that u_{max} for *Panopea abrupta* may be higher than the 0.35 assumed here. That information is based on higher-density populations of larger individuals than in *P. zelandica*.

Some small individuals are either caught by mistake or are brought to the surface incidentally when larger individuals are removed. The survival rate of such animals may be low: small geoducs have difficulty re-burying themselves because digging liquefies the substrate, and they float on its surface, vulnerable to predation. This may be a significant issue, because small *Panopea abrupta* tend to associate with adults. Shaul & Goodwin (unpubl.) found that about 66% of juveniles in a study site in Washington State were found within 5 cm of adult siphons.

The model assumes incidental mortality on small clams to be half the exploitation rate (see 13) on recruited clams for ages 2 through 5. Sensitivity of the results to other values is also examined. The incidental mortality is modelled by

16. $$N_{k+1,t+1} = N_{k,t}\,(1 - 0.5u_t)\exp(-M); \quad 1 < k < 6$$

2.8 Run length

Ideally, the model should be run until the model population stabilises. However, a population with 40 age classes takes some time to reach equilibrium. In contrast, the time horizon should be realistic. An arbitrary compromise was made between suitably long and realistically short periods, and the length of runs described here was 50 years except for demonstration trajectories.

2.9 Number of runs per set

This is also a compromise, between the precision of estimates (higher with more runs) and the time required to make a set of runs (longer with more runs). In the results reported here, 100 was chosen as an appropriate number of runs per set.

2.10 Fishery indicators

Six indicators were defined in this study. For a set of 100 runs, the model records the mean, minimum, maximum and 95% confidence interval for each of the first four indicators. The final two indicators are simple percentages.

Indicator 1: The mean recruited biomass during the last 20 years of each run. This indicator reflects the relative health of the population at the end of a run; a mean was used to smooth stochastic variation.

Indicator 2: The mean catch taken during the run. This indicator reflects the value of a fishing pattern to the commercial fishery. It was calculated on an annual basis: total catch divided by (the number of times fished during the run times θ). This method of calculation was required for different periods of pulse fishing to be compared directly.

Indicator 3: The mean catch expressed as a percentage of the target catch. This indicator reflects the ability of the population to support the fishing pattern specified. Ideally, this indicator should be 100%.

Indicator 4: The minimum biomass observed during each run. This indicator provides another index of the health of the population.

Indicator 5: In a set of runs, the percentage of runs in which the biomass fell below 20% B_0 at any stage. The level of 20% B_0 is used as a reference level in New Zealand risk simulations (eg. Francis 1992) after being proposed by Beddington & Cooke (1983). This indicator is an index of 'risk to the population'.

Indicator 6. In a set of runs, the percentage of runs in which the fishery was unable to take at least 80% of the target catch at any stage during the run. This indicator is an index of 'risk to the fishery'.

In the model, the order of operations is: 1) update year, 2) reduce populations by fishing, if applicable, 3) calculate biomass, 4) simulate ageing, recruitment, natural mortality and growth, re-calculate biomass. The biomass used for indicators is therefore the recruited biomass at the end of the simulated year.

3. $F_{0.1}$ YIELDS

A standard reference level of fishing intensity is $F_{0.1}$ (see Hilborn & Walters 1992 for review). This is the value of instantaneous fishing mortality rate, F, where the slope of a plot of yield-per-recruit vs F is 10% of the slope at the origin.

$F_{0.1}$ was estimated with a spreadsheet using the standard parameter values shown in Table 1. Constant recruitment was assumed, and incidental mortality on juveniles was assumed to be one half the exploitation rate on adults, as in standard model runs. M was assumed to be 0.15, 0.10 and 0.20 to examine sensitivity of results to M.

4. RESULTS

4.1 Simple results

Figs. 3 and 4 show 8 biomass trajectories from sets of runs with *quota* = 2% and 5% B_0 for the basic parameters (Table 1) and with no pulse fishing. These are long runs of 150 years, made for illustration only. Both figures show the range of initial biomass B_0 (from about 400-600 t) caused by stochastic past recruitment, and the range of variation in individual biomass trajectories caused by stochastic variation in recruitment during the eight runs.

For the low quota of 2% B_0, the effect of fishing is hard to see in the background noise. For the higher quota of 5% B_0, fishing caused a clear decrease in population size, but there is a wide range of responses. In some runs, the population crashed to near zero after 30 to 130 years (Fig. 4); with a much longer run length the remaining runs might have crashed also. This illustrates the dilemma faced in choosing a suitable time horizon.

Figs. 5-10 shows the 6 fishery indicators as a function of *quota*. Each plot for the first four indicators shows the mean of the indicator, the estimated 95% distribution of the indicator (not to be confused with the 95% CL around the mean indicator value, which is much smaller) and the minimum and maximum values of the indicator observed in 100 runs.

Indicator 1 (mean biomass; Fig. 5) decreases with increasing *quota*. However, the distribution of results for a given *quota* is very wide.

Indicator 2 (mean catch; Fig. 6) increases steadily with increasing *quota* until about 5%B_0, then levels off; at higher *quota* it decreases. This change is caused by the parameter u_{max}, the maximum exploitation rate. At *quota*s around 5%, the population decreases to the point where *target catch* is greater than u_{max} times remaining biomass. At this stage, only part of *target catch* can be taken.

The mean of indicator 3 (mean catch as a percentage of target catch; Fig. 7) remains at 100% until a *quota* of about 4% B_0 and then declines. Indicator 4 (minimum biomass; Fig. 8) declines with increasing *quota* in the same manner as mean biomass (Fig. 5).

Risk to the population (Fig. 9) is zero until a quota near 2%, then rises steadily to 100% at quota levels around 7% B_0. Risk to the fishery (Fig. 10) is similar, but remains zero until about 3% B_0.

4.2 The effect of pulse fishing

Fig. 11 shows the *quota* vs θ parameter space for the mean of indicator 1 (mean biomass). Fig. 5 was essentially the front elevation of this response surface. As in Fig. 5, the mean biomass declines as *quota* increases. At *quota* from 0-2%, there is no change as θ increases (the contours are vertical), indicating that pulse fishing causes no change in mean biomass. Above 2% *quota*, mean biomass increases as θ increases. This effect is caused by the parameter u_{max} limiting catch at higher θ.

Mean catch, indicator 2, is shown in Fig. 12. The increased catch at higher *quota* is modified by increasing θ above a *quota* of 2% B_0. Mean catch as a percentage of the target catch (Fig. 13) remains at 100% until *quota* reaches about 2% B_0, then begins to fall at high θ values. It falls at progressively lower θ until just over 4% B_0 *quota*, by which time the mean catch is less than 100% of the target catch for all θ.

Minimum biomass (Fig. 14) shows a pattern similar to mean biomass (Fig. 11). Risk to the population (Fig. 15) is nil up to a *quota* of about 3% B_0, and remains low at high θ, where u_{max} protects much of the population during the rare fishing events. Risk is 90% or more at combinations of high *quota* (above 7% B_0) and low θ (less than 2 yr).

Risk to the fishery (Fig. 16) also varies from zero at low *quota* to 100% at high *quota*. For this risk, the shape of the transition is different from that of Fig. 15: the model fishery is unable to take the target catch at combinations of high *quota* and high θ.

In hindsight, a useful indicator might have been one that reflected the biomass of geoducs encountered by the fishery; this should be an increasing function of θ for a given *quota*.

This relation is shown for a *quota* of 4% B_0 in Fig. 17. The biomass encountered by the fishery at the beginning of the last period of fishing increased steadily as a function of θ; with $\theta = 14$ the biomass encountered by the fishery was more than one and a half times that encountered when there was no pulse fishing.

4.3 Sensitivity

Sensitivity of the model to uncertainties in major parameters was examined with the deterministic model (*CVRect* = 0.0). It was impractical to do this for all indicators and all combinations of θ and *quota*, so indicator 1 (mean biomass in the last 20 years) was chosen as the test output variable. θ was set at 1 (no pulse fishing), and the model was run over *quota*s from 0.2-4.0% B_0.

The 'standard' run against which changes were compared was that shown in Fig. 5, with 40 cohorts, $M = 0.15$, 50% incidental mortality on juveniles, $u_{max} = 0.35$, and steepness = 0.75. Each of the parameters just listed was then varied singly to two or three other levels.

The effect of varying maximum age was examined with 30, 40, and 50 cohorts (Fig. 18). Results were insensitive to this parameter.

The effect of varying M was examined with $M = 0.10$, 0.15, and 0.20 (Fig. 19). Sensitivity was highest at *quota*s around 5-6% B_0. At a *quota* of 6% B_0, the effect of doubling M from 0.10 to 0.20 resulted in more than doubled mean biomass.

The effect of varying steepness was examined with $s = 1.00$, 0.75, and 0.50 (Fig. 20). Again, sensitivity was highest when *quota*s were around 5% B_0.

Sensitivity to the incidental mortality on juveniles was varied through 0%, 25%, 50%, 75% and 100% of the adult exploitation rate (Fig. 21). The effect was significant. The higher the incidental mortality, the lower the mean biomass for any given *quota*.

The effect of varying u_{max} (the maximum allowed exploitation rate when the target catch is a high percentage of the available biomass) through 0.20, 0.35, 0.50 and 0.75 is seen in Fig. 22. There is no effect unless the *quota* is high enough to cause u_{max} to operate; the effect is relatively small unless u_{max} is quite small. Because u_{max} becomes operative at smaller *quota* levels when θ is higher (Fig. 11), this sensitivity was also examined at $\theta = 10$ (Fig. 23). After a *quota* of 2% B_0, larger values of u_{max} resulted in much smaller mean biomass levels.

Table 2 shows the percentage change in indicator 1 at one *quota* level. The *quota* 3% B_0 was chosen as a possibly realistic value for this species. There was a high sensitivity to natural mortality, incidental mortality on juveniles, steepness and the maximum exploitation rate (at high θ only).

4.4 Finding an appropriate *quota*

Developing appropriate management advice requires an explicit definition of management goals and standards. For instance, it may be desirable to one manager to maintain the population above 40% B_0 while another is content to use 20% B_0 as a reference level. The indicators used here, while reasonable, may not reflect the exact goals of managers for this fishery. The model can easily be adapted to provide different reference points.

Suppose for illustration that the management goals were 1) to maintain a population with expected biomass at least 40% B_0 to a 50-year time horizon, with 2) less than 20% risk of falling below 20% B_0, and 3) without more than a 40% risk to the fishery (the risk of not being able to take the target catch); finally 4) to obtain the maximum yield permissible within these other three constraints. These constraints are entirely arbitrary and for illustration only.

1) To maintain 40% B_0, Fig. 11 suggests a maximum *quota* of just under 5% B_0 at $\theta = 1$; higher *quota* would be permissible (at least under this constraint) at higher θ. The unacceptable area under this constraint is shown in Fig. 24.

2) To have less than 20% risk of the biomass falling below 20% B_0 requires (Fig. 15) a *quota* of less than 4% B_0 at $\theta = 1$, or alternatively a θ greater than 4. This constraint precludes all the area which is unacceptable under constraint 1.

3) Risk to the fishery (the risk of not catching the target catch) is less than 40% at a *quota* of 3.5% B_0 or less when $\theta = 1$ (Fig. 16); at a lower *quota* if θ is greater than 1.

4) Fig. 24 shows the mean catch (as % B_0) within the remaining acceptable area of the parameter space. This indicates that the best mean catch available under the other three constraints is obtained when $\theta = 4$ and *quota* is just over 4% B_0.

A manager would then want to know how this conclusion is altered by the uncertainty around M and the other parameters examined in Figs. 18-23; she would want to do sensitivity analysis specific to the point chosen. The model described is easily adapted for this.

4.5 $F_{0.1}$ yields

$F_{0.1}$ was estimated to be in the range 0.079 to 0.107 as M varied from 0.10 to 0.20 (Table 3). At each level of assumed M, the deterministic recruited biomass when $F = F_{0.1}$ stabilised at about 45% of B_0. The equilibrium $F_{0.1}$ catch, expressed as a percentage of B_0 as above, varied from 3.13% B_0 at $M = 0.10$ to 4.15% B_0 at $M = 0.20$.

The $F_{0.1}$ yields can be compared with yields from the model comparing Table 3 with Fig. 19. At $M = 0.10$, the model yield that stabilises the population at 45% B_0 is about 3.75% B_0; the $F_{0.1}$ yield is 3.13%; the comparable model yield at $M = 0.20$ is about 5.25% B_0, compared with 4.15% from $F_{0.1}$. Thus using the model to estimate a yield that allows the population to remain at an expected 45% B_0 gives a higher estimate of yield than does using yield-per-recruit analysis. The difference between the two methods increases as M increases.

5. DISCUSSION

The modelling procedure described above provides a useful system for examining consequences of management alternatives in terms of defined management goals. The specific results above, and indicators used, should be viewed as examples. Managers for the *Panopea zelandica* fishery should consider their specific goals, re-define appropriate indicators if necessary, and obtain results from further modelling over the range of *quota* and θ values of interest.

5.1 Management implications

The results above suggest that a cautious approach to management is warranted. The sustainable yields for this species, with realistic management constraints, appear to be on the order of 2% to 4% of B_0. The relatively high age and slow growth of these clams mean that productivity is relatively low.

These estimates can be compared with those for *Panopea abrupta*, calculated with exactly the same method (Breen in prep.). The current fishing pattern for British Columbia is based on *quota* = 1.0% B_0 and $\theta = 3$ (Harbo *et al.* 1992). With this pattern, the expected mean biomass after 50 years is 70% B_0, and the minimum biomass is about 64% B_0. The model predicts very little risk either of the population falling below 20% B_0 or of the fishery being unable to take the target catch. Thus the current fishing pattern for *P. abrupta* appears extremely safe. It is quite conservative in the sense that a somewhat higher catch could be obtained without much increase in risk. The fishing pattern is robust, at least in terms of the mean biomass indicator, to all uncertainties except M. (Once appropriate indicators have been defined, managers should examine their sensitivity also.)

The work presented above considered the consequences of constant fishing patterns only. In the early phases of the fishery, a two-phase approach might be more appropriate. During the fishing-down phase, a higher than sustainable harvest rate could be used until the population approaches the desired equilibrium level. After that stage the appropriate safe and sustainable fishing pattern could be imposed.

The model assumes that only the target catch is taken. In British Columbia, some *Panopea abrupta* are darker and hence of lower quality; the darkness is a function of both substrate (Goodwin & Pease 1987) and age (Breen & Yamanaka, unpubl. data). Any wasteful removal of geoducs that are subsequently discarded would effectively increase the *quota* being taken.

As part of a fishing strategy for this species, pulse fishing appears to have limited benefits (at least, as reflected by the indicators examined; pulse fishing is currently useful in restricting the number of landing ports in Canada). In reality, populations are likely to be pulse fished in practice, or fished in patterns that vary between years. Pulse fishing has no impact on mean catches at low to moderate *quotas* (Figs. 12 & 13), but at high *quotas* pulse fishing reduces the ability of the fishery to take the target catch (Fig. 16). Pulse fishing does have the advantage that the industry operates on a higher biomass and perhaps at a lower cost (Fig. 17).

6. ACKNOWLEDGEMENTS

I thank Lynn Goodwin for allowing me to cite his unpublished data and commenting on the similar MS for *P. abrupta*; Owen Anderson for preparing the contour plots and Graham McKay for preparing Fig. 24; Owen Anderson and Keith Michael for diving assistance; Di Tracey, Claire Gabriel, Jacqui Greaves and Stephanie Kalish for cutting shells and counting rings; Terese Kendrick, Ray Maulder and Jacqui Grieves for technical and field assistance; and Drs. Chris Francis and John Annala for helpful comments on the MS.

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Table 1. Values for major parameters of the model. The "normal value" is the one used for the main results. "Other values" shows the values explored in sensitivity testing.

Parameter	Symbol	Normal Value	Other values
Natural mortality rate	M (yr^{-1})	0.15	0.10, 0.20
Maximum Age	*AgeMax* (yr^{-1})	40	30, 50
Age at recruitment		8	
Juvenile vulnerability ages		2-5	
von Bertalanffy parameters:	W_∞ (g)	259.3	
	K (yr^{-1})	0.226	
	t_0 (yr^{-1})	3.113	
CV of recruitment	*CVRect*	0.80	0.0
Expected recruitment at B_0		1 000 000	
Steepness	s	0.75	1.00, 0.50
Incidental juvenile mortality - % adult rate	*SLMort*	50%	0%, 25%, 75%, 100%
Maximum exploitation rate	u_{max}	0.35	0.25, 0.50, 0.75, 1.00
Quota	*quota* (%B_0)	0.5 to 8.0	
Period of rotation for pulse fishing	θ (yr)	1 to 14	
Runs per set	*nrun*	100	1
Runlength	*runlength* (yr)	50	150

Table 2. Sensitivity of the model results to changes in the major uncertain parameters. The table shows the percentage change in indicator 1 (mean biomass in the last 20 years of a model run) caused when the parameter being tested is changed from its original value (in bold). All changes are measured when the *quota* is 3.0% B_0.

Parameter	Value	% change in indicator 1
M	0.10	-14.3
	0.15	0.0
	0.20	9.1
Maximum age	30	2.6
	40	0.0
	50	-0.7
Incidental mortality as a percentage of the adult exploitation rate		
	0.00	13.4
	0.25	6.9
	0.50	0.0
	0.75	-7.6
	1.00	-15.9
Steepness	1.00	5.4
	0.75	0.0
	0.50	-12.9
Maximum exploitation rate u_{max} at $\theta = 1$		
	0.20	0.0
	0.35	0.0
	0.50	0.0
	0.75	0.0
Maximum exploitation rate u_{max} at $\theta = 10$		
	0.20	17.2
	0.35	0.0
	0.50	-2.2
	0.75	-2.2

Table 3. $F_{0.1}$ estimates made with three assumed values of M; the catch from a population at equilibrium with $F_{0.1}$ and the assumed M, expressed as a percentage of B_0; and the equilibrium pre-fishing population $B_{F0.1}$, expressed as a percentage of B_0.

Assumed M	$F_{0.1}$	Catch	$B_{F0.1}$
0.10	0.079	3.13% B_0	43.2% B_0
0.15	0.094	3.69% B_0	44.3% B_0
0.20	0.107	4.15% B_0	45.0% B_0

FIGURE CAPTIONS

Figure 1. The age structure (yr) of a clam population near Collingwood, sampled in 1991.

Figure 2. The age (yr) vs whole weight (g) relation from the population shown in Fig. 1. The solid line shows the growth functions used by the model.

Figure 3. Biomass trajectories from the model, using the standard parameters in Table 1 and a *quota* of 2% B_0. These trajectories are from runs of 150 years (most other results are based on 50-year runs).

Figure 4. Biomass trajectories from the model, using the standard parameters in Table 1 and a *quota* of 5% B_0.

Figure 5. Indicator 1 (mean biomass in the last 20 years of a run) as a function of *quota*. The heaviest line shows the mean of 100 runs of 50 years each; other lines show the minimum and maximum values observed in the 100 runs and the predicted 5% and 95% confidence limits of the distribution of results.

Figure 6. Indicator 2 (mean catch (t) over the 50 years of a run) as a function of *quota*.

Figure 7. Indicator 3 (the mean percentage of the target catch which was caught over the 50 years of a run) as a function of *quota*.

Figure 8. Indicator 4 (the minimum biomass observed during the 50 years of a run) as a function of *quota*.

Figure 9. Indicator 5 (the percentage of runs in which biomass fell below 20% B_0 at any time during the run) as a function of *quota*.

Figure 10. Indicator 6 (the percentage of runs in which the target catch was not taken at any point during the run) as a function of *quota*.

Figure 11. Indicator 1 (mean biomass) plotted as a function of both *quota* (x-axis) and the period of rotation for pulse fishing, θ (y-axis). Steepness = 0.75 and incidental juvenile mortality was 50%.

Figure 12. Indicator 2 (mean catch) (kg) plotted as a function of both *quota* (x-axis) and the period of rotation for pulse fishing, θ (y-axis).

Figure 13. Indicator 3 (mean catch as a percentage of target catch) plotted as a function of both *quota* (x-axis) and the period of rotation for pulse fishing, θ (y-axis).

Figure 14. Indicator 4 (minimum biomass) plotted as a function of both *quota* (x-axis) and the period of rotation for pulse fishing, θ (y-axis).

Figure 15. Indicator 5 (risk to the population) plotted as a function of both *quota* (x-axis) and the period of rotation for pulse fishing, θ (y-axis).

Figure 16. Indicator 6 (risk to the fishery) plotted as a function of both *quota* (x-axis) and the period of rotation for pulse fishing, θ (y-axis). Steepness = 1.0 and incidental juvenile mortality was zero.

Figure 17. A special indicator: the biomass encountered by the fishery at the beginning of the last period of fishing in a 50-year run, as a function of θ. This was calculated by the deterministic model (*CVRect* = 0.0) and with a *quota* of 4% B_0.

Figure 18. Sensitivity of indicator 1 (mean biomass) (B_0) to variation in maximum age. Results were obtained from the deterministic model (*CVRect* = 0.0) using the standard parameter values in Table 1.

Figure 19. Sensitivity of indicator 1 (mean biomass) (B_0) to variation in natural mortality rate, M. Results were obtained from the deterministic model (*CVRect* = 0.0) using the standard parameter values in Table 1.

Figure 20. Sensitivity of indicator 1 (mean biomass) (B_0) to variation in steepness. Results were obtained from the deterministic model (*CVRect* = 0.0) using the standard parameter values in Table 1.

Figure 21. Sensitivity of indicator 1 (mean biomass) (B_0) to variation in the amount of incidental mortality on juveniles (expressed as a proportion of the rate on adults). Results were obtained from the deterministic model (*CVRect* = 0.0) using the standard parameter values in Table 1 and $\theta = 1$.

Figure 22. Sensitivity of indicator 1 (mean biomass) (B_0) to variation in the maximum exploitation rate, u_{max}. Results were obtained from the deterministic model (*CVRect* = 0.0) using the standard parameter values in Table 1 and $\theta = 1$.

Figure 23. Sensitivity of indicator 1 (mean biomass) (B_0) to variation in the maximum exploitation rate, u_{max}. Results were obtained from the deterministic model (*CVRect* = 0.0) using the standard parameter values in Table 1 and $\theta = 10$.

Figure 24. The parameter space of θ vs *quota*, showing areas (shaded) where the management constraints discussed in the text are not met. Each shaded area is labelled with respect to the specific constraint. In the unshaded area at the left, the mean catch contours are shown.

Collingwood 1991 Samples

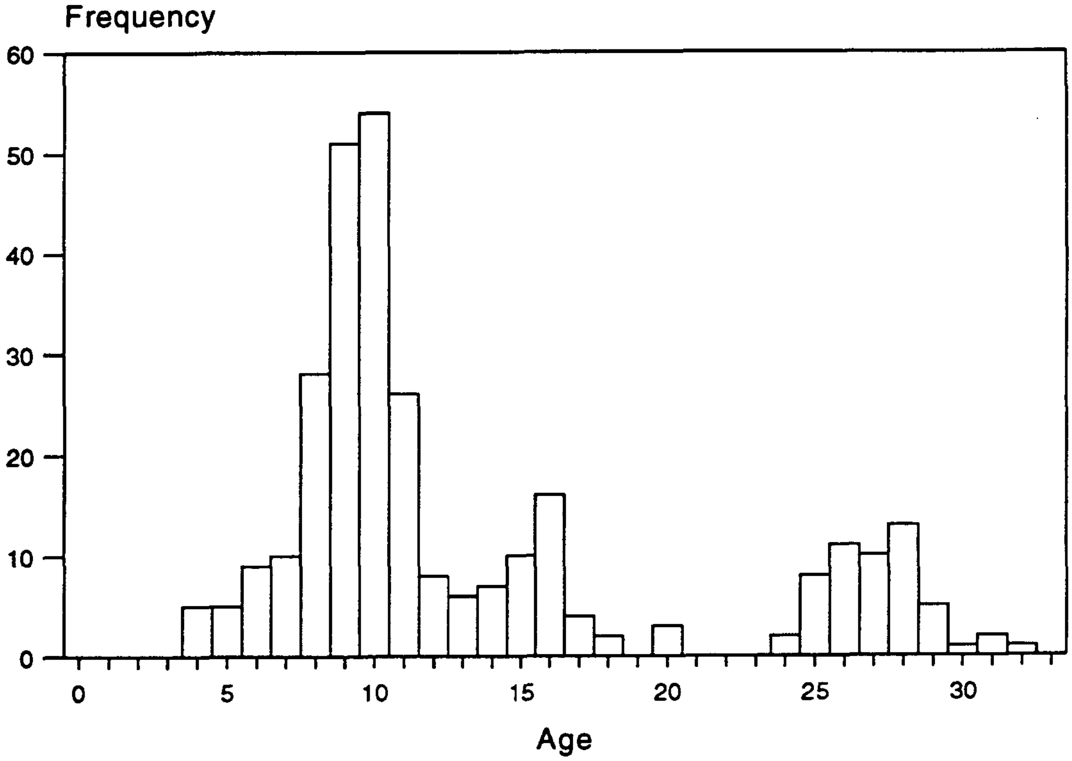
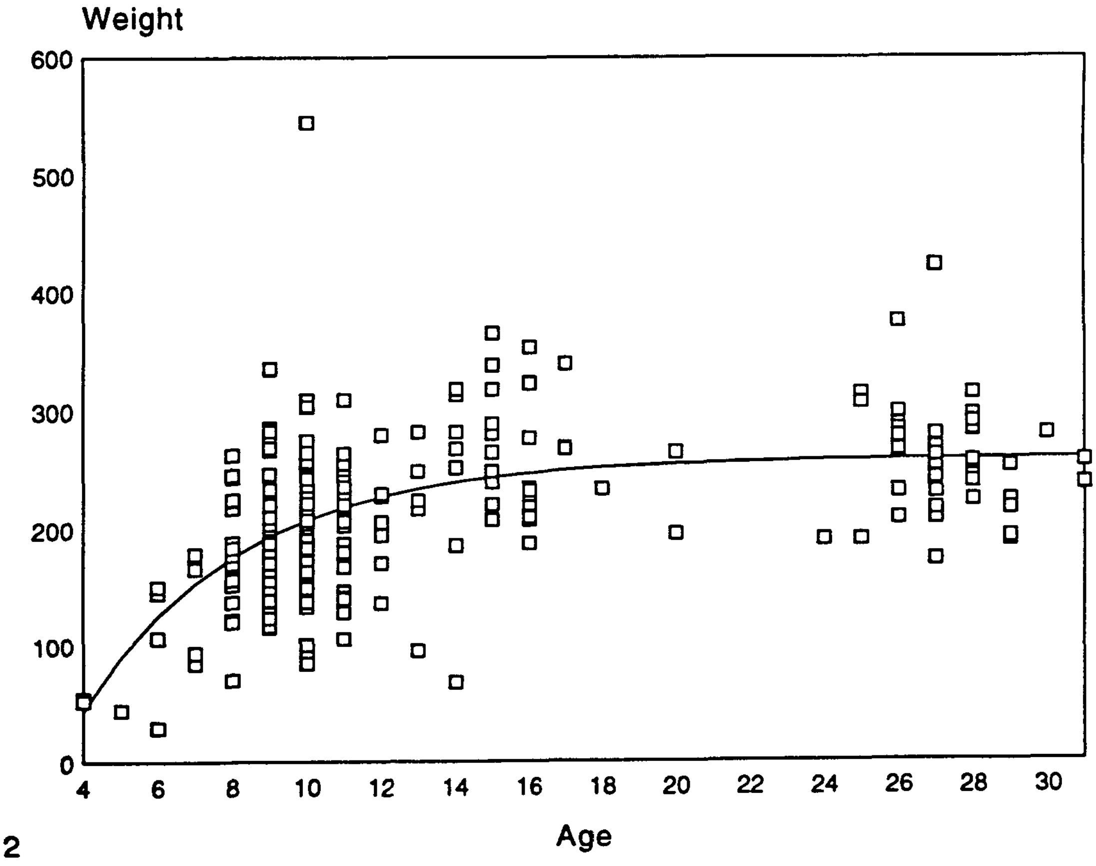


Figure 1



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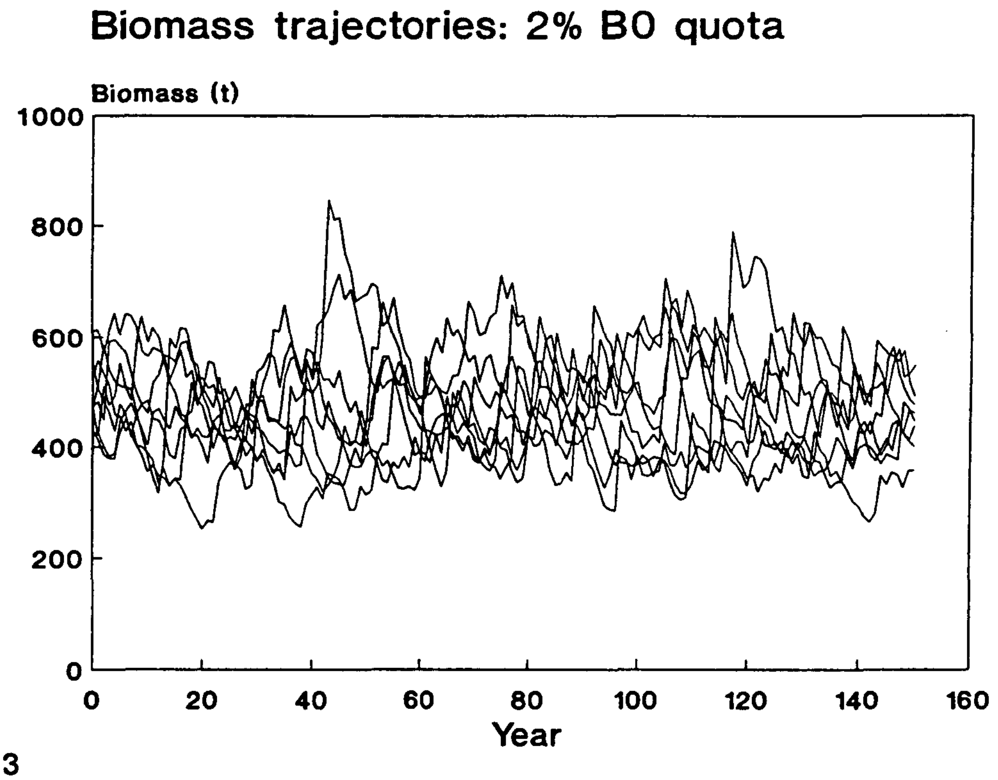

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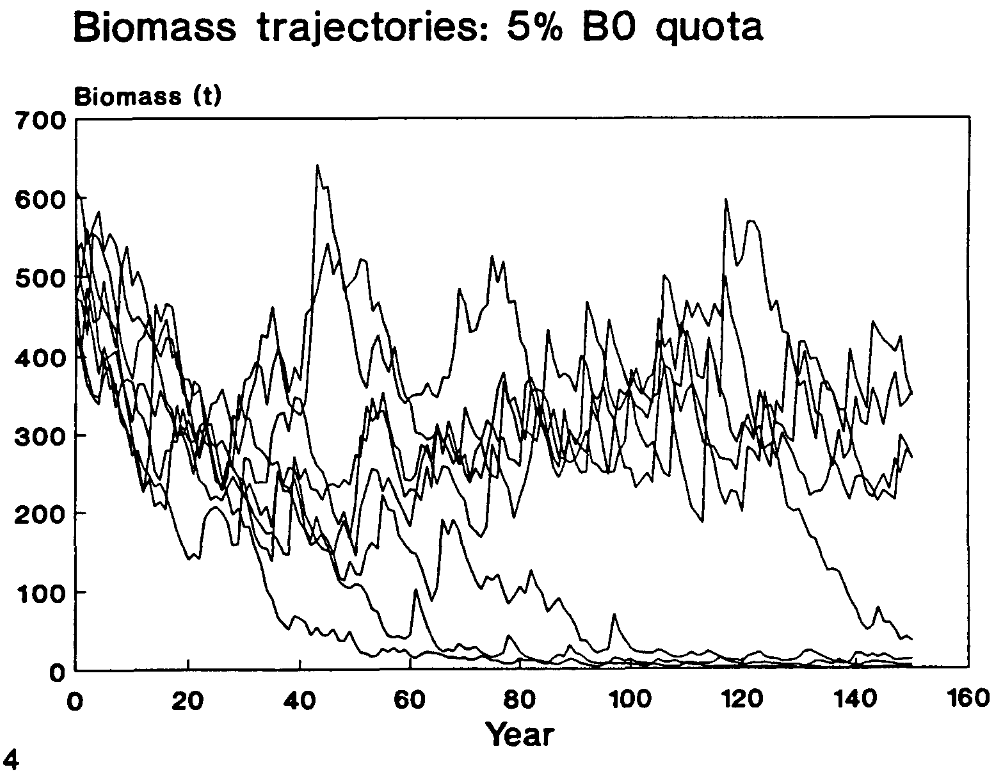


Figure 4

Indicator 1 (mean biomass)

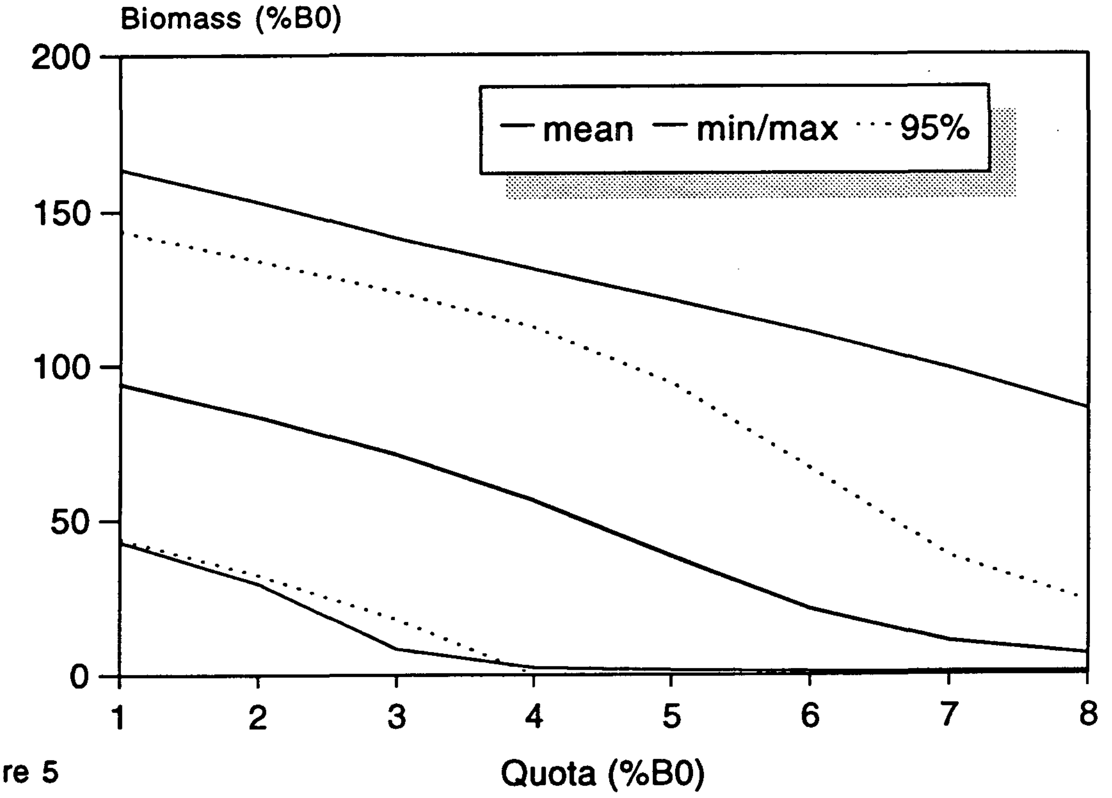
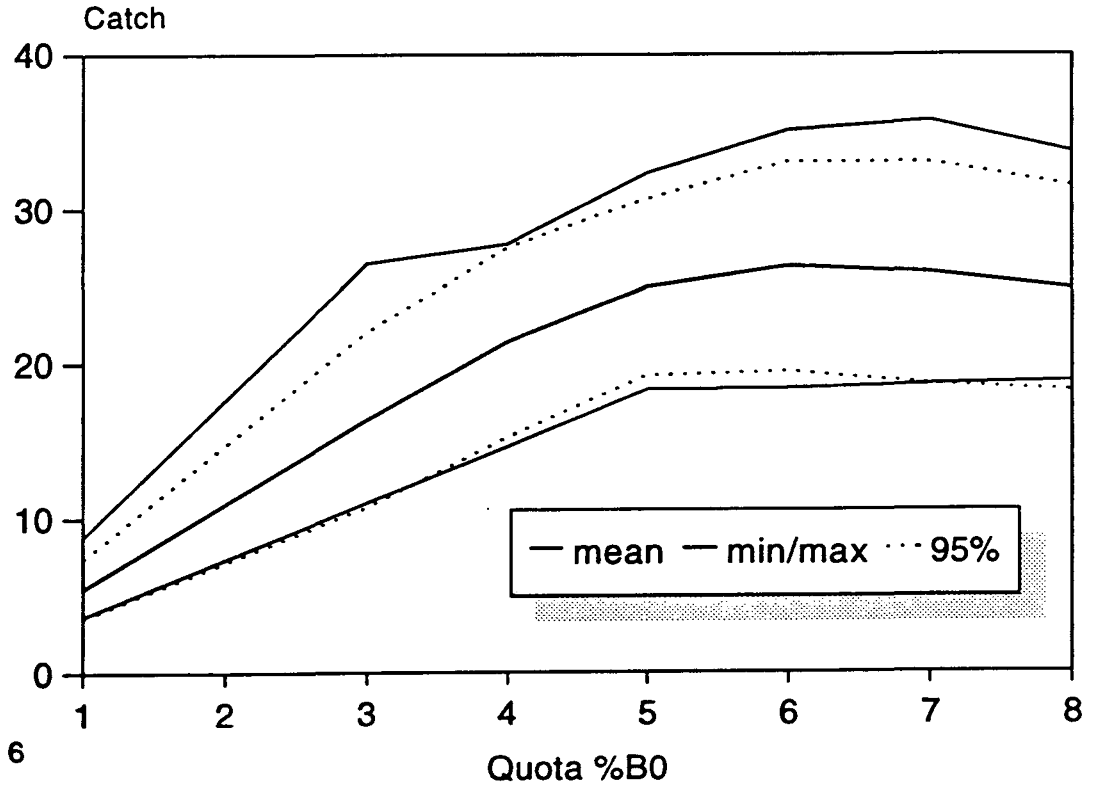


Figure 5

Indicator 2 (mean catch)



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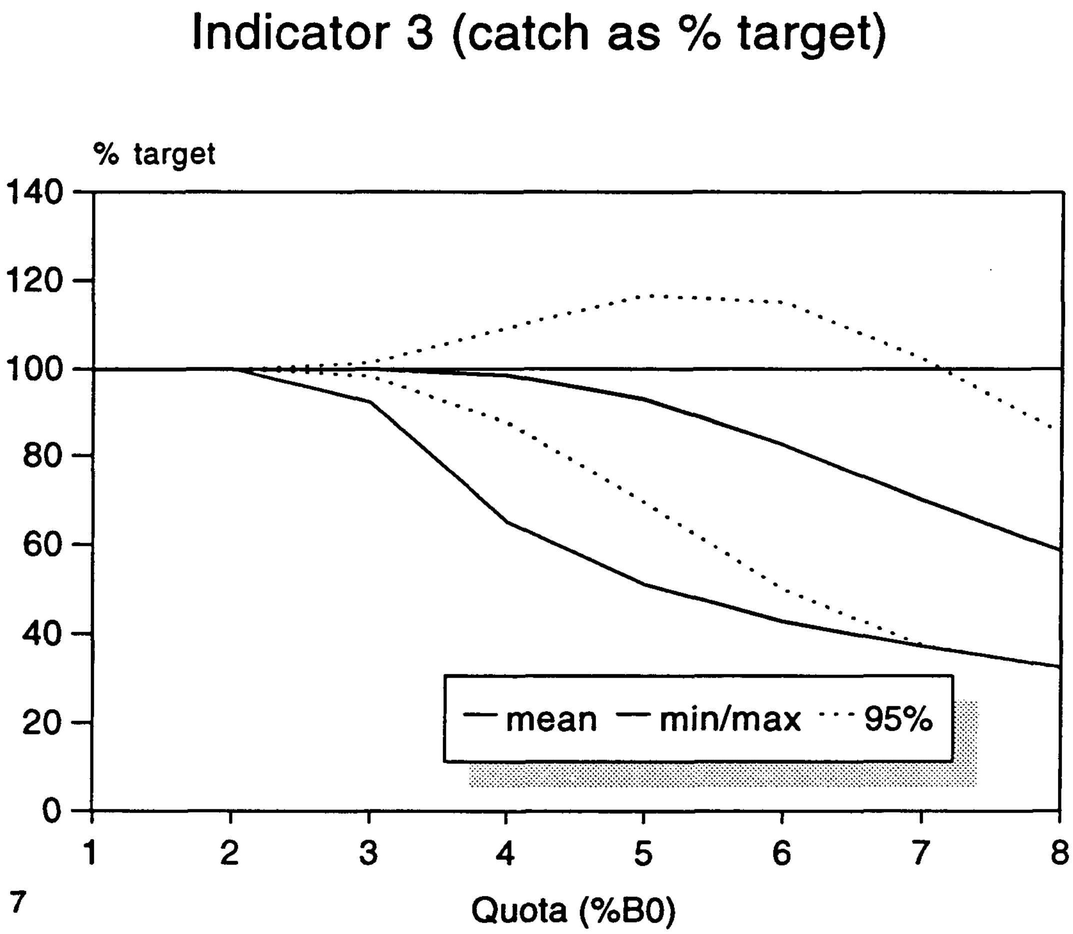

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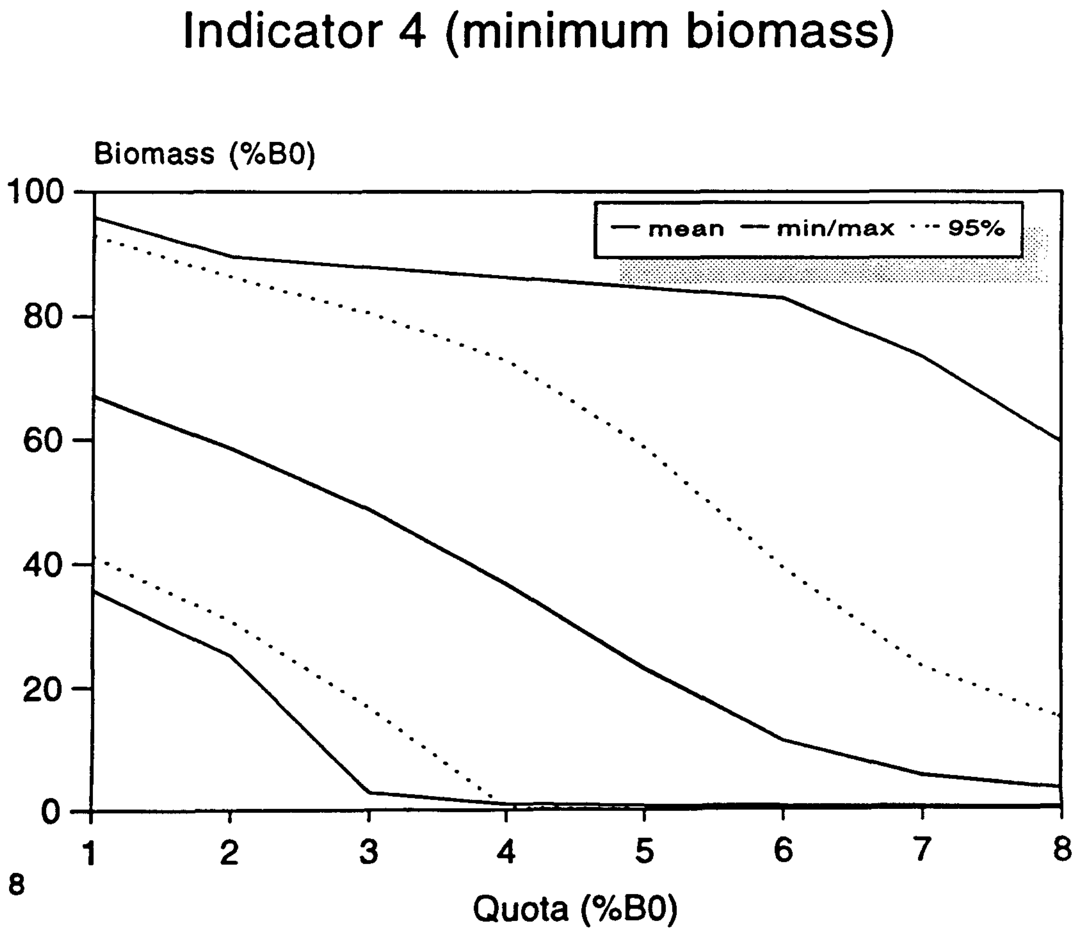

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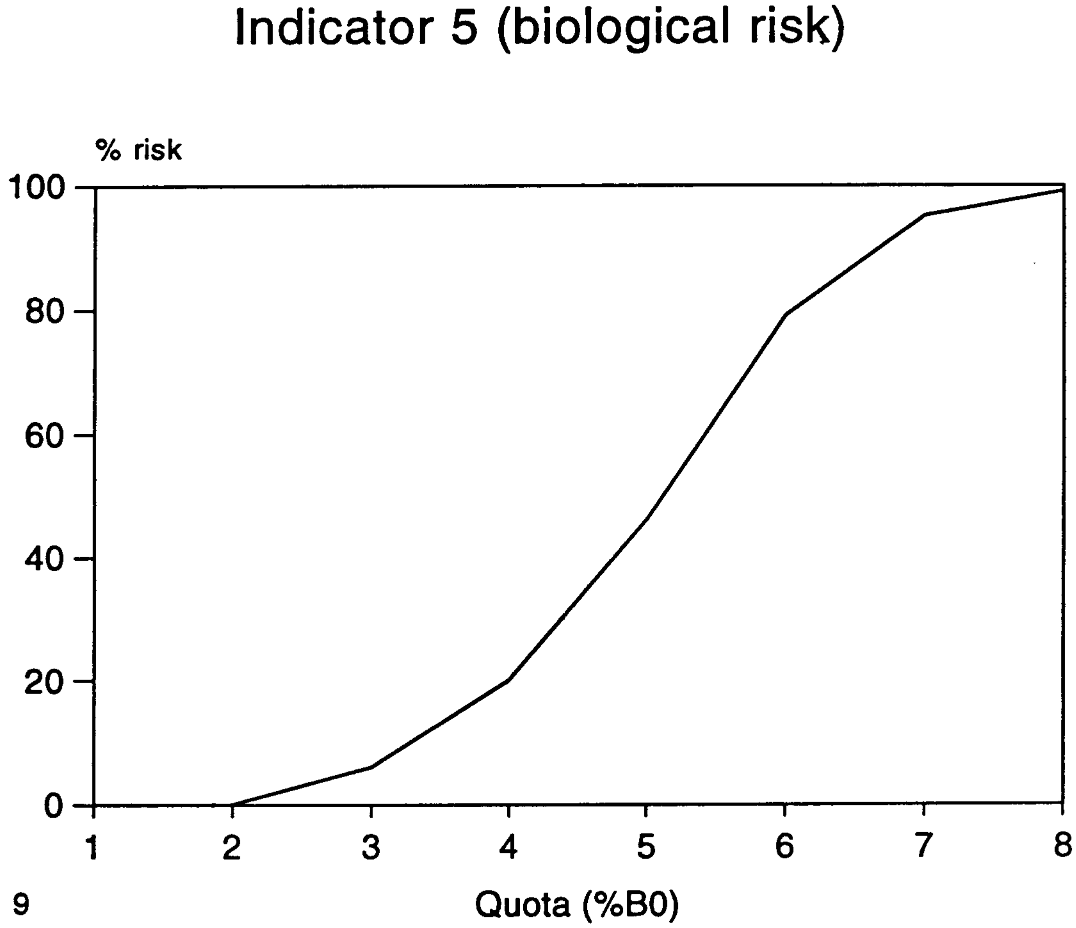

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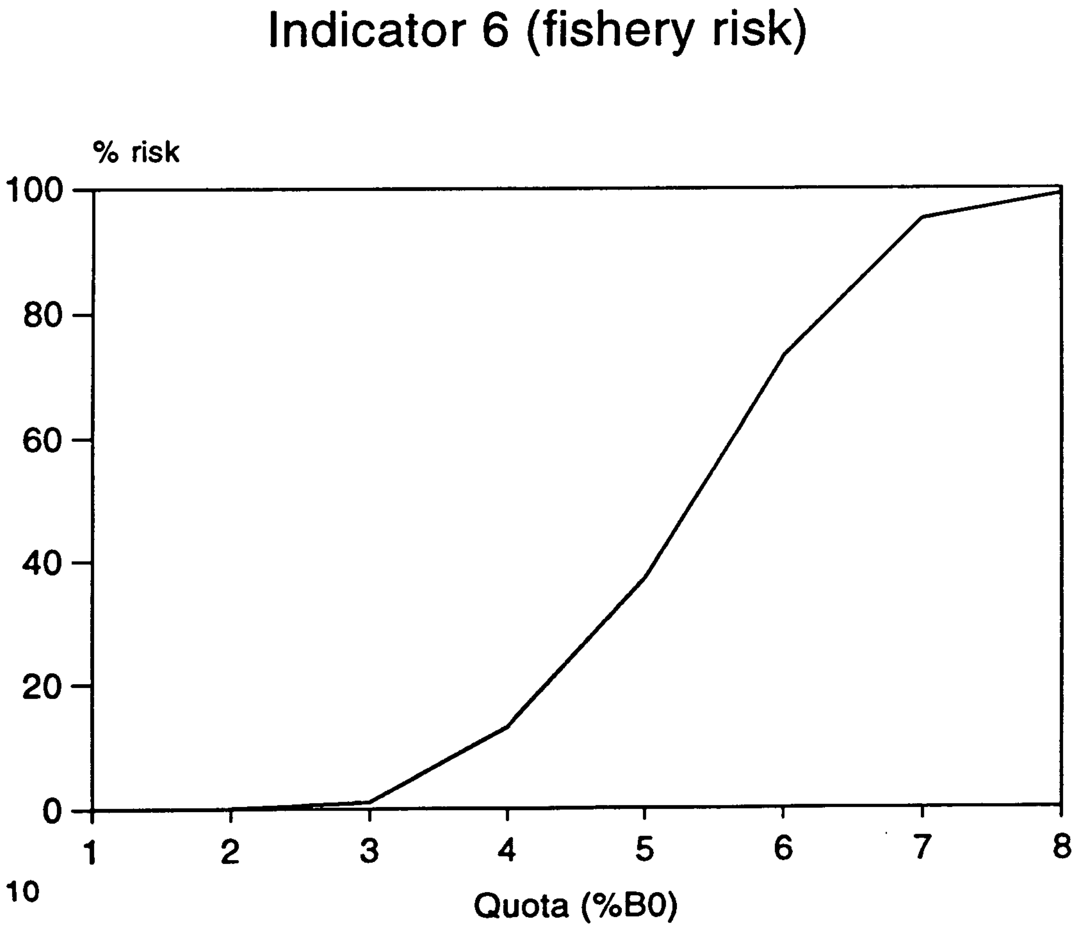

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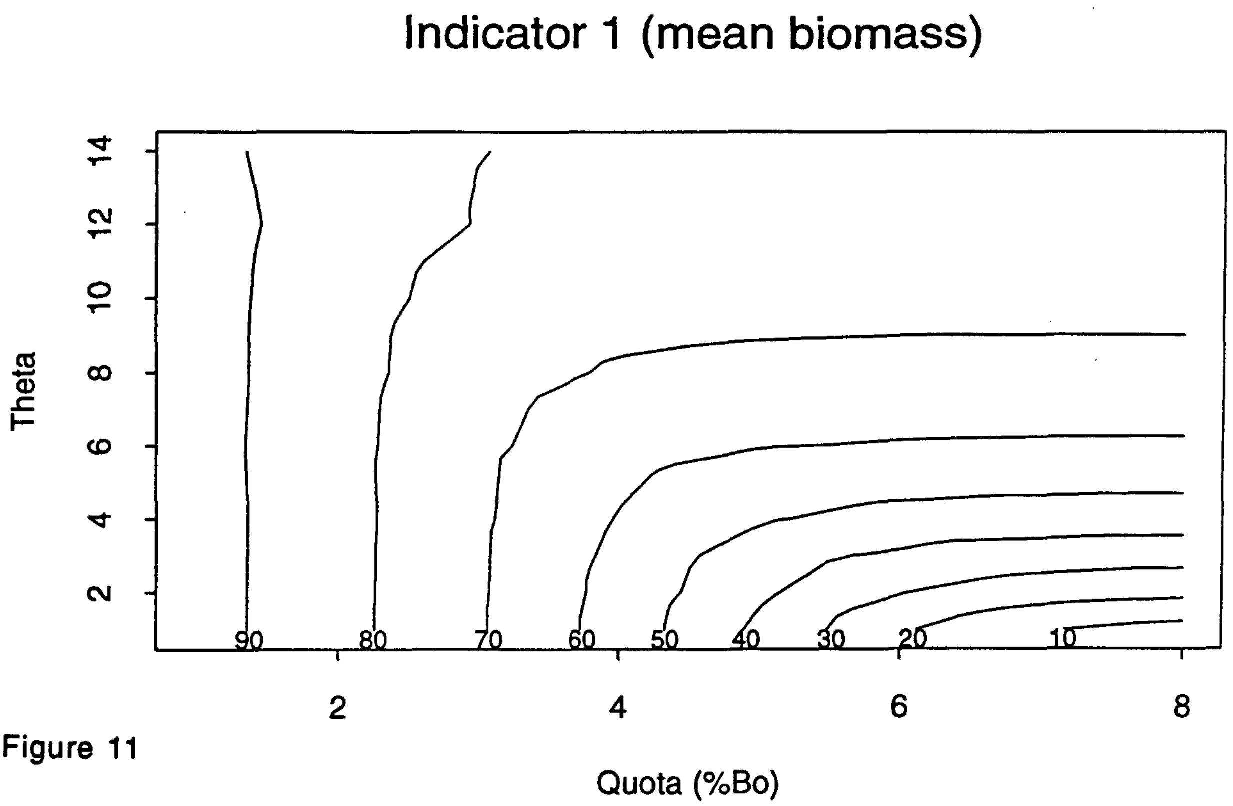

Indicator 1 (mean biomass)

Figure 11

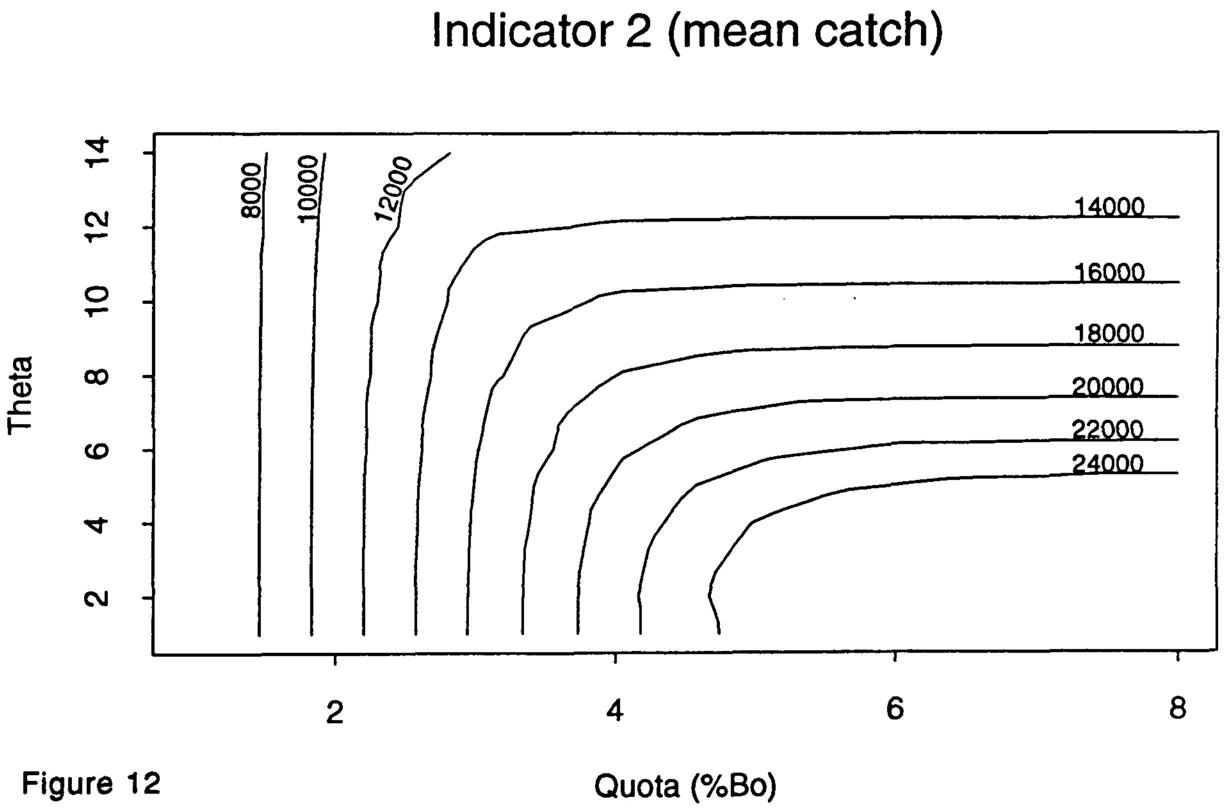
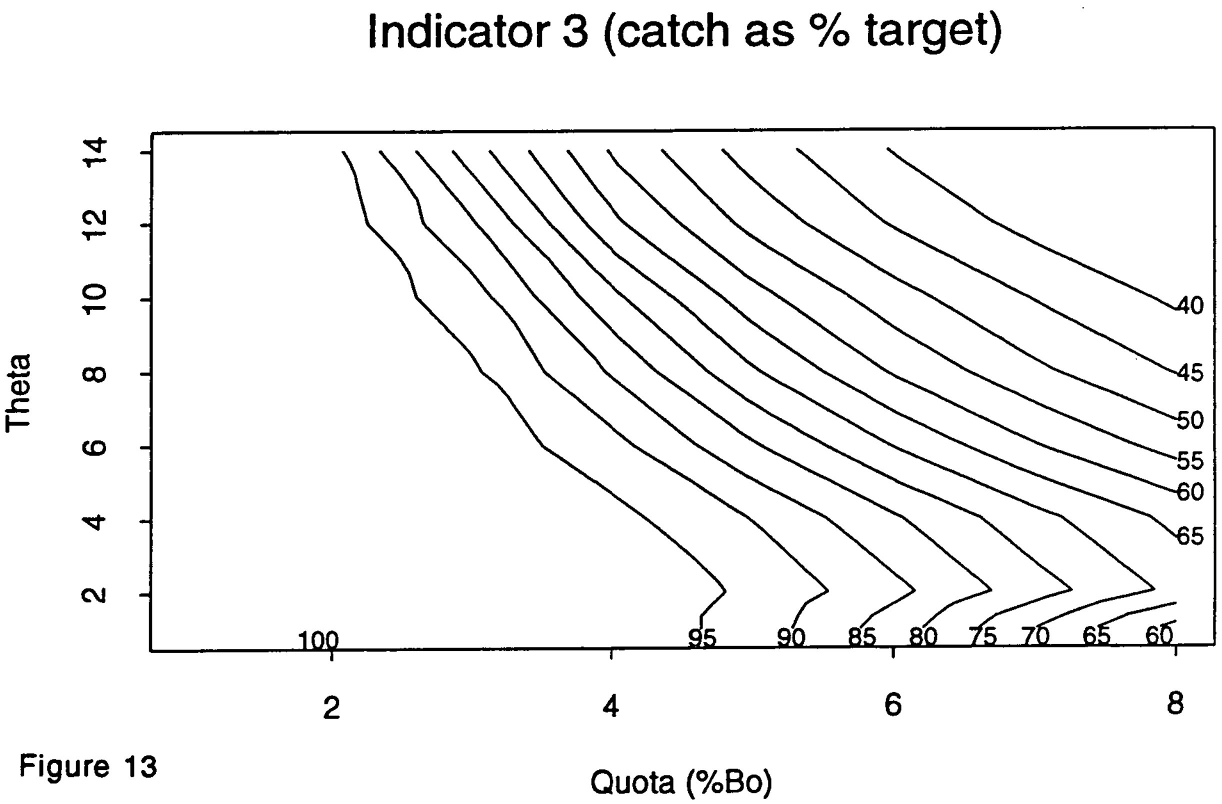

Indicator 2 (mean catch)

Figure 12



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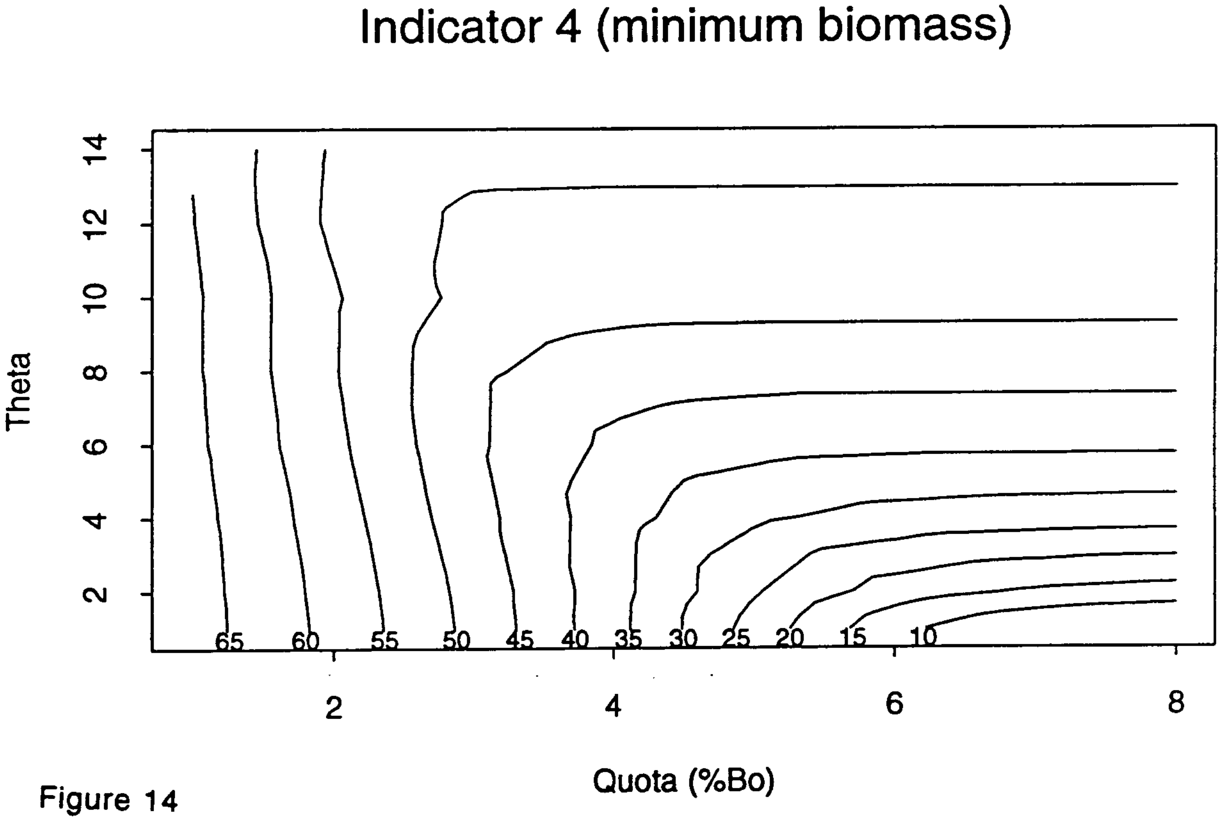

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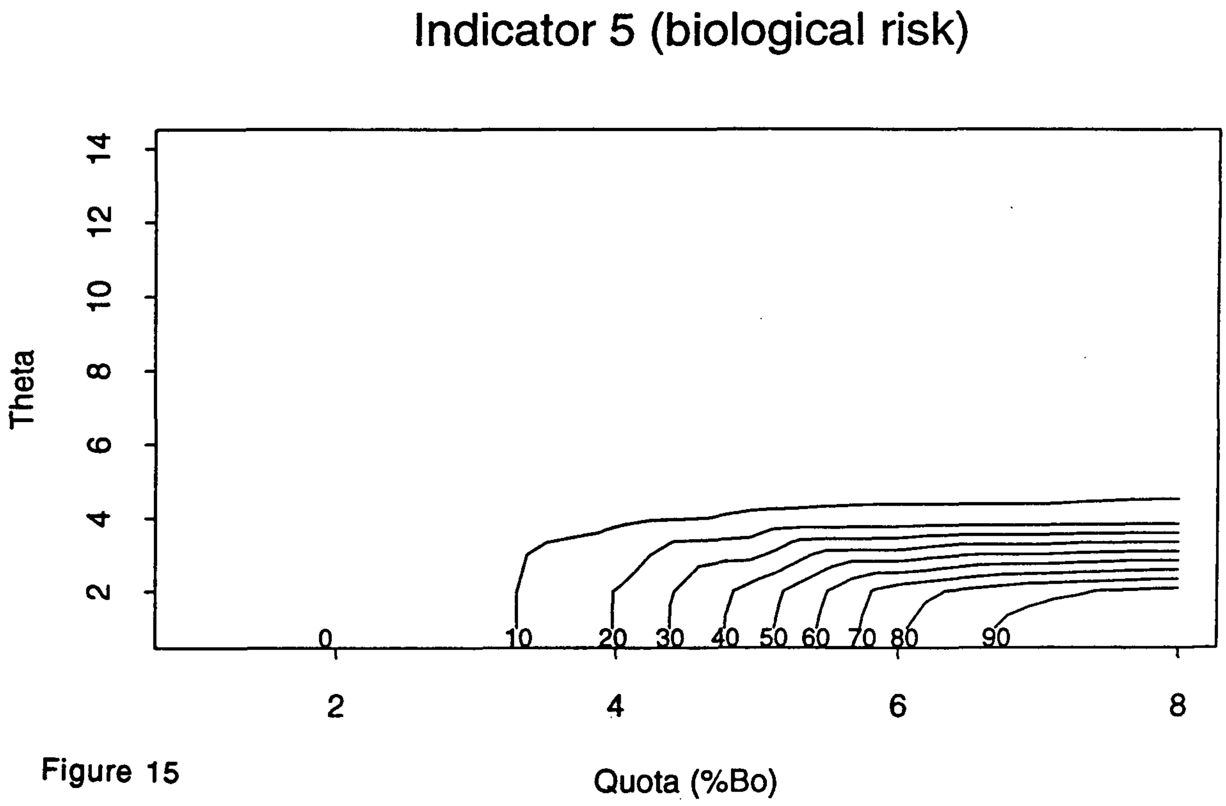

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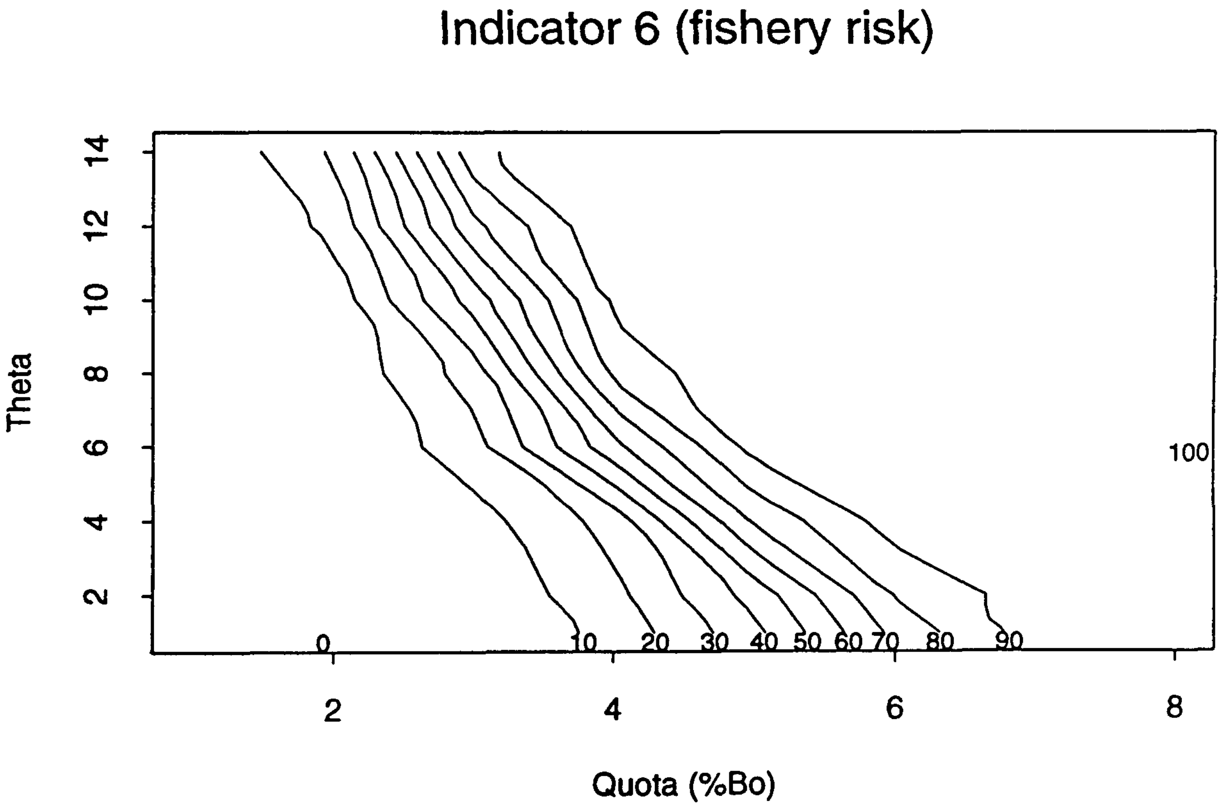

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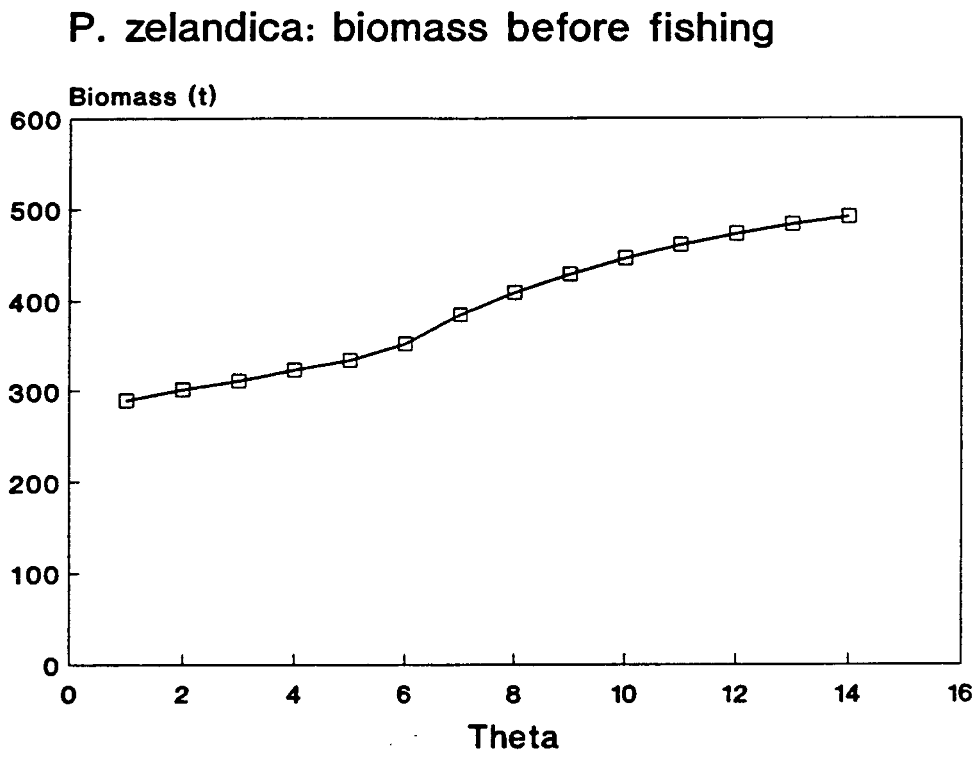

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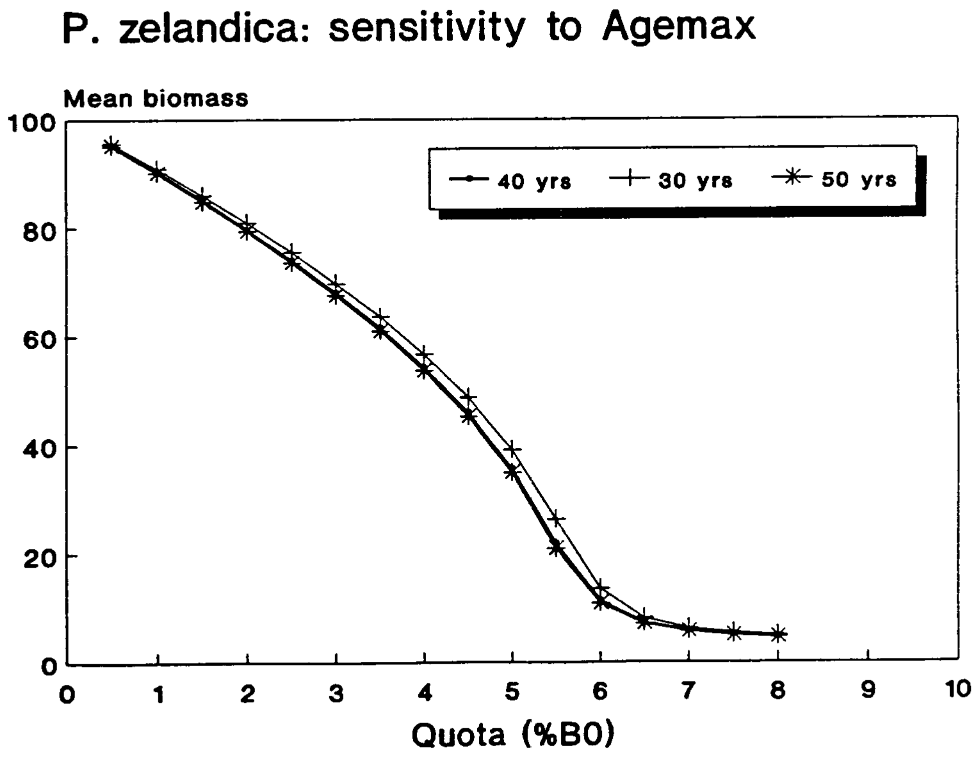


Figure 18

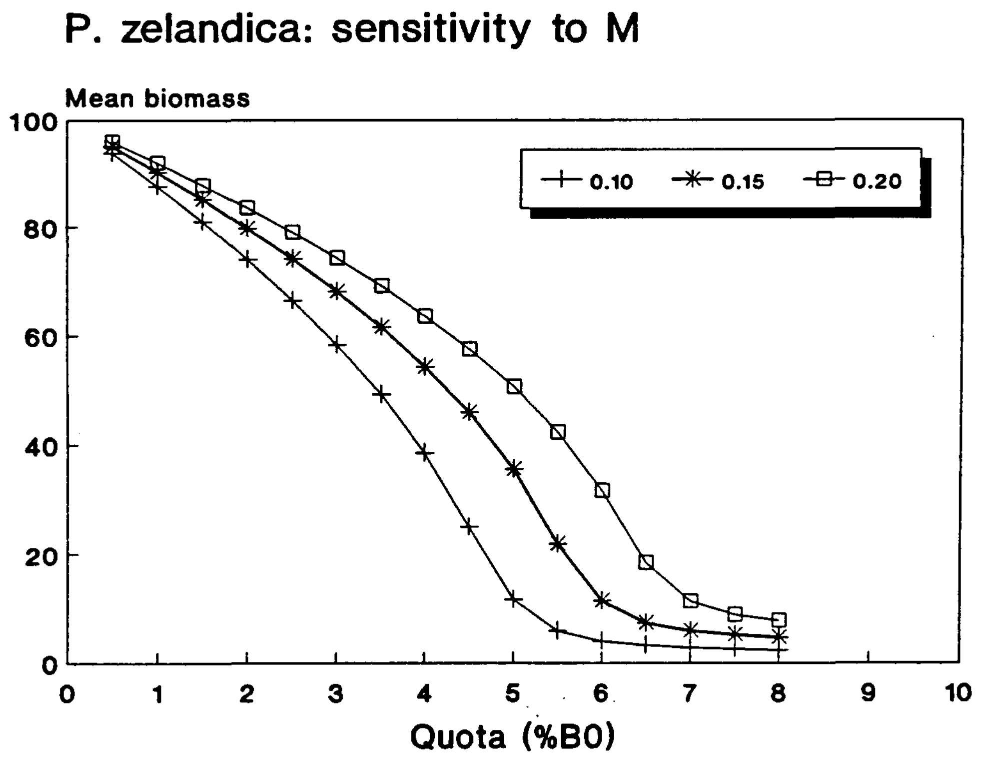

P. zelandica: sensitivity to M

Figure 19

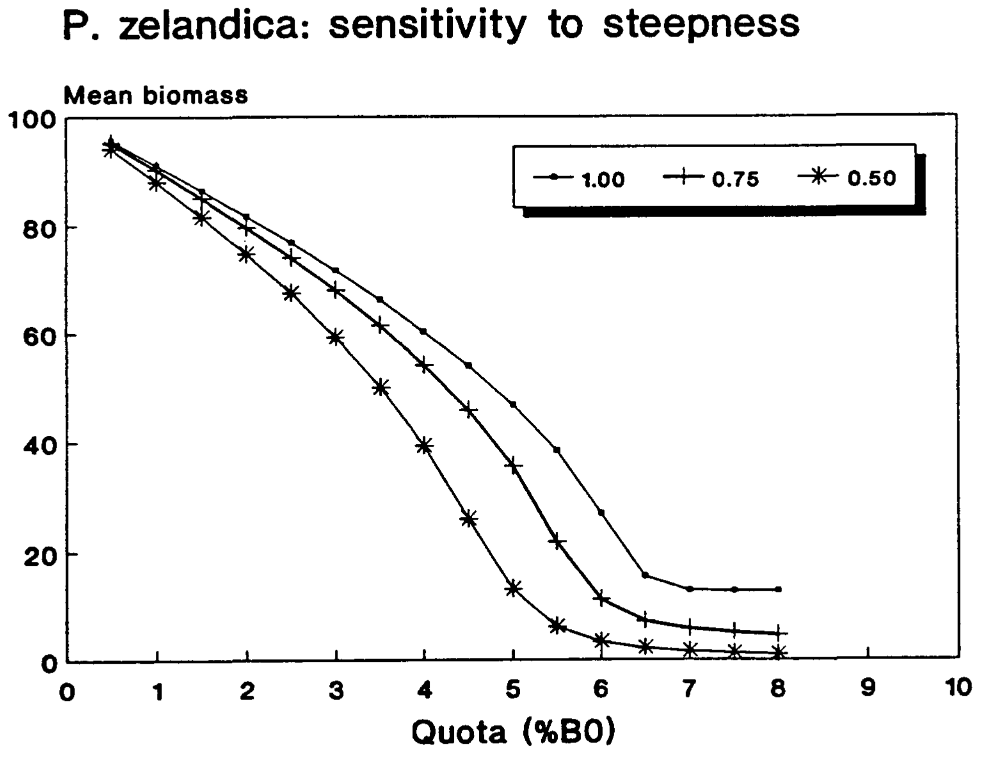
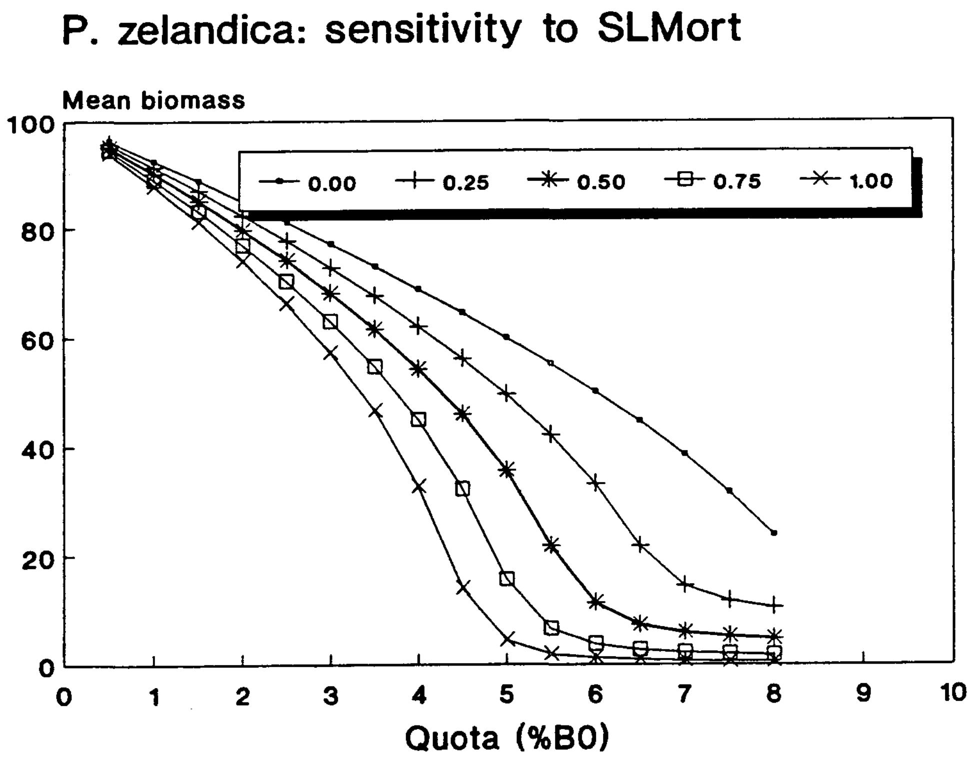

P. zelandica: sensitivity to steepness

Figure 20



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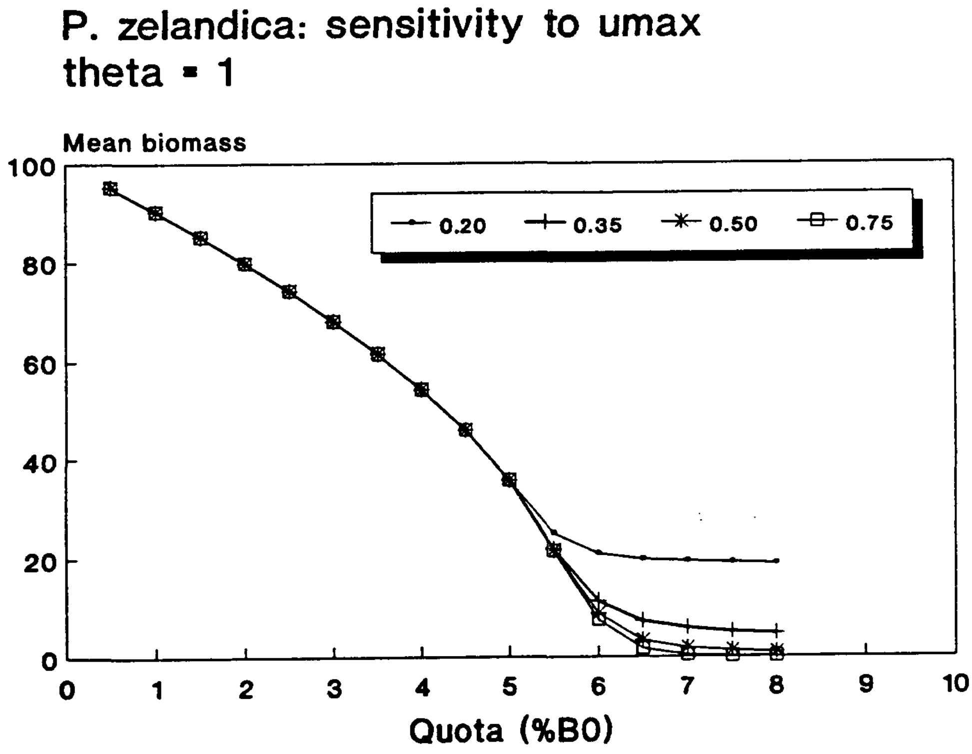

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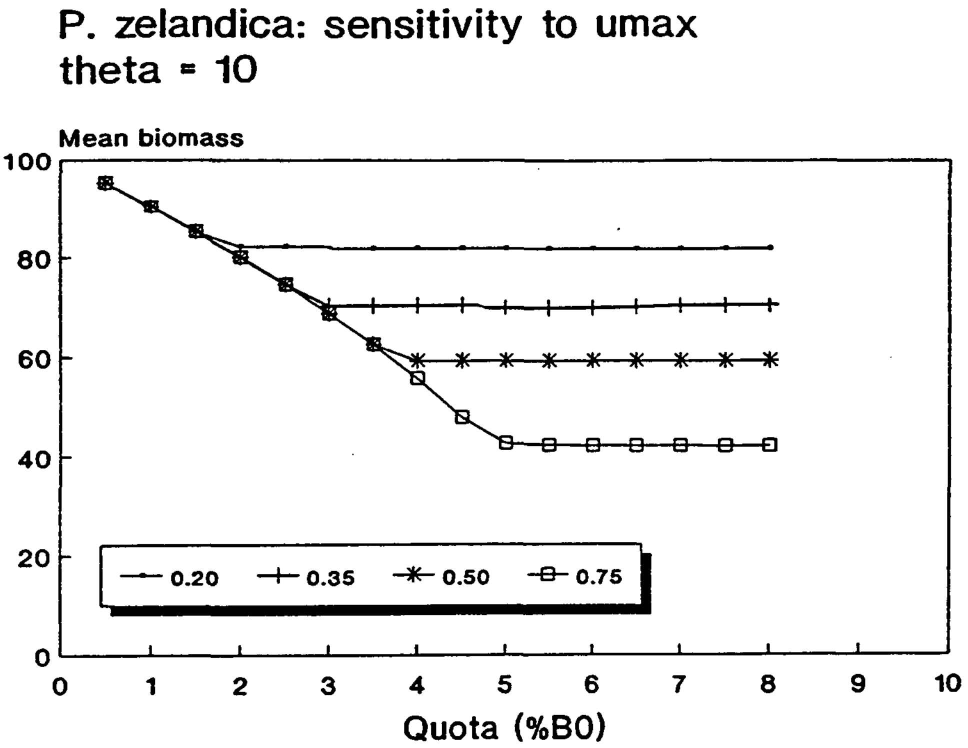

Figure 23

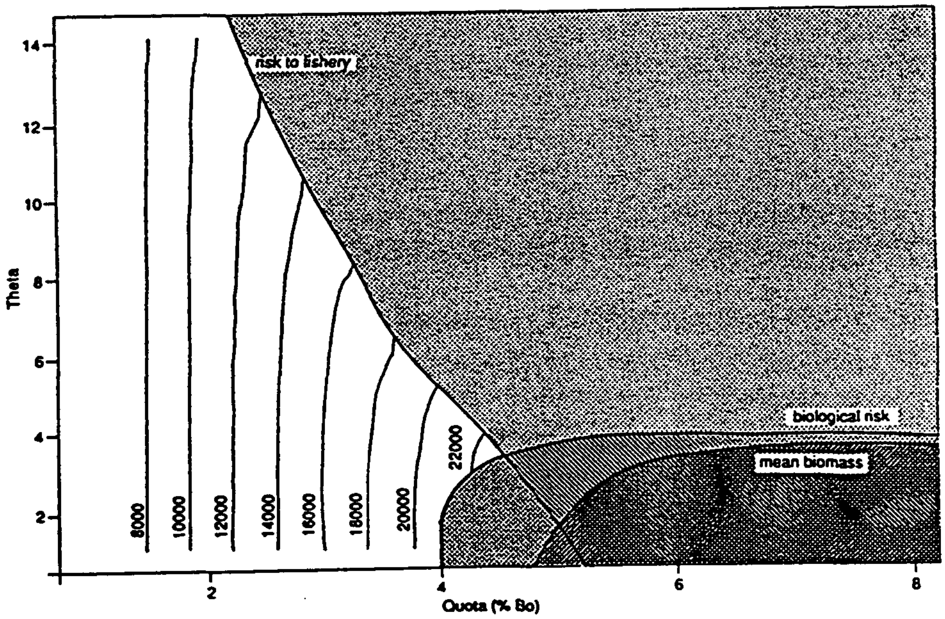

Figure 24