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Considerations on the Demise or Otherwise of Northern Cod

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ABSTRACT

Perspectives on the current status of northern cod depend upon the interpretation of recent, decadal changes in its population dynamics. In particular, the unexpected decline of northern cod in offshore areas in the early 1990's has yet to be resolved in terms of a comprehensive hypothesis that can account for all anomalous events. Three major hypotheses have been put forward to explain the apparent collapse viz. (1) STOCK MISMANAGEMENT HYPOTHESIS-----the stock has been in continuous decline since the mid-1980's because of high fishing mortalities that were not detected by offshore RV surveys and resulted in persistent overestimation of stock biomass levels, (2) CATASTROPHIC MORTALITY HYPOTHESIS----massive increases in natural mortality of idiopathic origin devastated the stock in the early 1990's, and therefore the offshore surveys correctly tracked stock dynamics and (3) MIGRATION CHANGE HYPOTHESIS-----a major change in offshore migrations occurred in the early 1990's such that the overwintering now occurs in the inshore; i.e. offshore RV surveys adequately measure offshore, but not total abundance of northern cod. In this communication we have examined a wide variety of indices, particularly from the inshore, in an attempt to evaluate and eliminate hypotheses that are inconsistent with available data. We have concluded, from the congruence in the inshore trap indices, inshore gillnet indices, the Spans offshore index, exploitation and population estimates from inshore and offshore taggings, that northern cod abundance was stable and consistent with the offshore surveys at least up to the early 1990's. The Stock Mismanagement Hypothesis can therefore be rejected on the grounds that it lacks an evidential basis. With respect to the Migration Change Hypothesis, this hypothesis can only be rejected if Sentinel catch-rates of northern cod are shown to be an invalid measure of biomass. We have concluded, from a consideration of potential biases in Sentinel catch-rates along eastern Newfoundland and Labrador and from comparisons with implied biomass equivalence of sentinel catch-rates in Div. 3Ps, that these inshore indices represent significant (and perhaps substantial) concentrations of cod and therefore the Migration Change Hypothesis cannot yet be rejected. An alternative hypothesis, along with supporting evidence, is presented which places Sentinel catch-rates in the context of an expansion of historical links between northern cod and a resurgent 3Ps cod stock.

RÉSUMÉ

L'appréciation de l'état actuel de la morue du nord est fonction de l'interprétation de changements récents, décennaux, de la dynamique des populations. Plus particulièrement, le déclin imprévu de la morue du nord survenu dans les zones hauturières au début des années 1990 reste à décrire par le moyen d'une hypothèse détaillée prenant en compte toutes les anomalies. Trois grandes hypothèses ont été formulées pour expliquer l'effondrement du stock : 1) l'hypothèse de la MAUVAISE GESTION DU STOCK - le stock est en déclin constant depuis le milieu des années 1980 à cause d'une mortalité par pêche importante non décelée par les relevés par navires de recherche effectués au large et qui ont donné lieu à une surestimation constante de la biomasse, 2) l'hypothèse d'une MORTALITÉ CATASTROPHIQUE - une augmentation massive de la mortalité naturelle d'origine inconnue a dévasté le stock au début des années 1990 de sorte que les relevés hauturiers ont fait correctement état de la dynamique du stock et 3) l'hypothèse d'un CHANGEMENT MIGRATOIRE - une importante modification de la migration côtière s'est produite au début des années 1990 de sorte que l'hivernage a maintenant lieu en zone côtière, et les relevés hauturiers par navires de recherche ont correctement mesuré l'abondance au large mais non l'abondance totale de la morue du nord. Nous examinons, dans la présente communication, une large gamme d'indices, notamment pour la zone côtière, dans le but d'évaluer les hypothèses et d'éliminer celles non conformes aux données obtenues. Nous concluons, à partir de la congruence des indices des pêcheries fixes côtières, de l'indice de la pêche côtière au filet maillant, de l'indice hauturier Spans et des estimations des niveaux d'exploitation et de population obtenues par des marquages en zones côtière et hauturière, que l'abondance de la morue du nord était stable et correspondait aux résultats des relevés hauturiers, du moins jusqu'au début des années 1990. L'hypothèse de la mauvaise gestion peut donc être réfutée en se fondant sur l'absence de preuves. En ce qui a trait à l'hypothèse d'un changement migratoire, cette hypothèse ne peut être réfutée que si l'on peut montrer que les taux de capture des pêches sentinelles de morue du nord constituent une mesure non valable de la biomasse. Nous concluons, après examen des biais possibles des taux de capture des pêches sentinelles le long de la côte est de Terre-Neuve et de celle du Labrador et de comparaisons des biomasses équivalentes déduites des taux de capture des pêches sentinelles dans la division 3Ps, que ces indices côtiers font état de concentrations appréciables (et peut-être importantes) de morues, de sorte que l'hypothèse d'un changement migratoire ne peut être rejetée. Une autre hypothèse, accompagnée d'éléments à l'appui, est aussi présentée. Elle fait appel aux taux de captures des pêches sentinelles dans le contexte d'une expansion des liens historiques entre le stock de morue du nord et un stock résurgent de 3Ps.

INTRODUCTION

The unexpected decline in the abundance of cod in offshore surveys in Div. 2J3KL, beginning in 1991 and continuing through the 1990's, has yet to be resolved in terms of a definitive explanation that can account for all observed anomalies. For example, cod abundance in the offshore surveys in 2J3KL declined precipitously after 1992, despite the closure of the commercial fishery, yet sentinel fishery catch rates of cod in inshore waters along eastern Newfoundland are above normal commercial levels. A variety of hypotheses have been proposed to explain the apparent collapse (see Lear and Parsons 1993, Hutching and Myers 1994, Myers et al. 1996 and others). These can be summarized as follows;

STOCK MISMANAGEMENT HYPOTHESIS

This hypothesis states that massive increases in fishing mortality after the mid-1980's caused a monotonic decline in northern cod abundance that was not detected by the offshore surveys either because of inappropriate statistical expression of survey abundance indices (Hutchings 1996), or because an inappropriate calibration model was used to convert survey abundance to population abundance (Myers and Cadigan 1995b). This hypothesis debunks the sudden timing aspects of the stock collapse and attributes the disappearance of northern cod to persistent errors in stock assessment procedures.

CATASTROPHIC MORTALITY HYPOTHESIS

This hypothesis states that a sudden increase in mortality, beginning in the early 1990's and persisting for several years caused the stock to collapse, (i.e. the offshore RV survey correctly tracked stock abundance changes). Given that the stock continued to decline in the absence of a commercial fishery, and considering that unrealistic levels of catch under-reporting would have been necessary to account for the declines in survey abundance (P. Shelton, pers. comm.), the stock collapse can be attributed to massive increases in natural mortality (M). An ancillary hypothesis is that such mass mortality was exclusive to the offshore area and that "inshore" stocks remain healthy.

MIGRATION CHANGE HYPOTHESIS

A sudden and sustained change in overwintering migration patterns to areas other than the offshore began in the early 1990's, which aliased the offshore survey results and the survivors exist elsewhere (i.e. the offshore survey correctly measured abundance within the survey area but no longer correctly tracks total stock abundance). The only likely overwintering refugia is the inshore area where anecdotal and sentinel fishery information indicate commercial concentrations of cod.

If management measures to reopen the northern cod fishery are to be successful, it is important to understand the precipitant causes of recent stock dynamic changes so that corrective action is appropriately focussed. Clearly, the strategy of a fishing moratorium, begun in 1992, has not had the expected results if the offshore surveys are accepted, and the time-frame for recovery may now be indeterminate. In the following analyses a simple approach will be taken to examine the pros and cons of the various hypotheses in an attempt to more clearly focus attention on the most likely scenario.

MATERIALS AND METHODS

The major sources of abundance data used to assess northern cod are derived from indices collected by research vessels during the fall-spring period when northern cod are overwintering on the offshore banks. These indices assume that the timing of the offshore migrations are relatively constant from year to year and that the proportion of cod which overwinter on the shelf is not only constant but form the bulk of the total population. In this analysis, we shall concentrate on alternative indices of abundance from the inshore area during the summer-fall period in order to assess coherence with population trends from the offshore surveys. The data sources include inshore catch and effort (Hutchings and Myers 1994, Murphy and Shelton 1997), tagging data (Taggart et al. 1995, Myers et al. 1997), harp seal stomach content data (Stenson et al. 1995) and inshore juvenile cod data (Schneider et al. 1997). Since there is very little in the way of direct forensic evidence for a disease-based natural mortality increase or for catastrophic mass mortality, we shall concentrate on the stock mismanagement and migration change hypotheses with the intent that their elimination will automatically endorse the natural mortality hypotheses.

RESULTS AND DISCUSSION

CONSIDERATION OF THE STOCK MISMANAGEMENT HYPOTHESIS

We shall first examine the stock mismanagement hypothesis i.e., that the collapse of northern cod was neither sudden nor unexpected, but began in the mid-1980's as a result of high levels of fishing mortality that were not detected by the surveys because of unexplained increases in efficiency of the survey trawls. This hypothesis is based on the analyses of Myers and Cadigan (1995a, 1995b) (but see also Hutchings and Myers 1994), who attempted to correct for age-correlated annual residual patterns in the standard calibration model (ADAPT) using an alternative catch-at age tuning model (the correlated error model). Estimates of population abundance by the correlated error model are shown in Fig.1 (from Table 3 of Myers and Cadigan 1995b) along with estimates from the random error ADAPT assessment in 1991 (Baird et al. 1991). The 1991 Assessment was chosen as a temporal reference because it estimated

stock abundance up to 1990, before the stock collapse was evident from the 1991 fall survey. Both models were calibrated with the fall survey data but the correlated error model used 1993 as the terminal year. Population estimates of the two models begin to diverge after 1986 and by 1991 the correlated error model shows a decline of about 75% from the 1985 level whereas the 1991 assessment indicated relative stability since 1985. Research vessel catchability ($Q=RV\ 3+/ SPA\ 3+$) implied by the two models are concomitantly different after 1986 (Fig. 2); estimates from the 1991 assessment are consistent with the assumptions of such a research survey i.e. Q fluctuates randomly without trend. The correlated error model, however, implies that a 3-fold increase in Q occurred between 1987 and 1991. Since this research survey is carried out with standard gear and rigorously observed sampling protocols, such an increase in Q , if real, could only occur by a massive and progressive change in cod behaviour/distribution beginning around 1986. For example, a higher proportion of cod may have remained near bottom or the relative proportion of the population in the survey area progressively increased after 1986 or, as Myers and Cadigan (1995b) suggest, the fish merely became more concentrated over time which (somehow) resulted in survey overestimates of abundance. From a statistical viewpoint however, the allocation of residual patterns by the correlated error model for 1990's is facultative since increases in M or underreporting of catches in those years will also account for the year effects in the fitting procedure both models show the same anomalous residual patterns when 1993 or later years are used as the terminal year. That is to say, Myers and Cadigan (1995b) chose changes in $RV\ Q$ as the explanatory variable since it was a natural consequence of additional published analyses rejecting other possibilities (eg. Myers and Cadigan 1995a).

EVIDENCE FROM THE TRAP INDEX

The differences in the population trends between the 1991 assessment model and that proposed by Myers and Cadigan (1995a, b) are so striking that they should be resolvable by referring to other indices of abundance for confirmation. One source, used by Hutchings and Myers (1994), to show that the decline had begun well before 1991 is inshore catch-rates. These are available from vessel log-books on a trip by trip basis for traps and gillnets for the period 1985-91 (Hutchings and Myers 1994). The authors claimed that, based on their analyses, trap catch-rates in 3KL decreased from 1986 to 1991 with the decline being clearly evident by 1989. Similar conclusions were reached in their analyses of gill-net indices. These data have been reviewed by Murphy and Shelton (1997) who point out that the original analyses by Hutchings and Myers (1994) contained confounding errors of inappropriate data inclusion that may have invalidated their conclusions. We shall concentrate mainly on the trap index since traps are known to measure recruitment variations (Pinhorn 1986, Rose 1992) of northern cod. Murphy and Shelton (1997) provide a breakdown of catch/effort sampling intensity for the trap index which shows that only 3 of the

10 area subdivisions of northern cod were sampled in 1985, 8 of 10 in 1986, 9 of 10 in 1987 and all unit areas were sampled from 1988 onwards. Sampling rates for the various unit areas are poor in the early years but increase overall for the period 1988-91. We have chosen to include, in the following analyses, the trap data for 1986 to 1991 but note that confidence limits for the first several years may be large because of sampling deficiencies. The revised catch-rate index for traps (Fig. 3) shows an upward trend for the period 1986-91, with estimated trap effort fluctuating in a random manner.

In order to ameliorate the deficiencies in sampling for the initial years of the trap index, we used a General Linear Model to extract the yearclass effect from the 1986-91 trap index for ages 4 to 7. This approach, in essence, removes the age effect and integrates measures of yearclass size across several years of sampling data. The model provided a good fit to the data and explained 88 % of the variance (Table 1). In Figure 4 we show plots of the standardized estimates for the size of the 1979-87 yearclasses in relation to age 3 estimates of the same yearclasses from the survey-based 1991 Assessment (top panel) and the correlated error model (lower panel). There is very good coherence between the inshore trap index and the 1991 SPA ; however, the trap indices of yearclass size are inconsistent with estimates from the correlated error model for those 7 yearclasses (1983-87) which are sensitive to terminal year input parameters.

An additional approach was used to independently verify the trap index of yearclass strength shown in Figure 4. Myers et al. (1993) have shown that yearclass success of northern cod is largely determined by environmental conditions (with salinity (SAL 1) as its proxy) and spawning stock size (S S B). They developed an empirical recruitment model for the 1962-88 yearclasses as follows;

$$\text{Log (Recr)} = - 40.43 + 0.43 \log (\text{S S B}) + 1.37 \text{ SAL } 1 \text{ (Rsqu = 0.68)}$$

We recalculated the above algorithm for the converged portion of the SPA (1962-82 yearclasses) and obtained a revised recruitment model as follows;

$$\text{LOG (RECr)} = -38.144 + 0.395 \log (\text{S S B}) + 1.32 \text{ SAL } 1 \text{ (Rsqu=0.70)}$$

The revised model was used to estimate the 1983 to 1987 yearclasses that would have been expected from empirical estimates of salinity and SPA estimates of S S B for those years (S S B estimates for 1983-87 were quite similar for both the correlated error model and the 1991 assessment). Estimates of expected yearclass success by the above model agree quite well with the trap indices (Fig. 5, top panel) but are inconsistent with the correlated error model estimates for the 1983 to 1987. In summary, there is good coherence between the inshore trap indices of yearclass size, those from the survey-based 1991

assessment, and expectations of yearclass success from the empirical recruitment model.

There is, however, another approach which treats total inshore trap catches as a simple but direct measure of northern cod abundance, in particular cod recruitment variations (Pinhorn 1986). Rose (1992) extended the analyses of Pinhorn (1986) as a test of the hypothesis that effective fishing effort by cod traps along eastern Newfoundland remains relatively constant from year to year because the number of trap berths are naturally limited in number and are used annually on a regular basis. Therefore, over a series of fishing seasons, catches will be a direct reflection of recruitment variations overlaid with naturally occurring fluctuations in trap catchability (i.e. Templeman's (1966) water mass hypothesis). That is to say, stock biomass in year t is a predictable function of trap catches (i.e. recruitment) in the previous several years. Rose (1992) used this principle to develop a predictive index of 4+ biomass in year t (B I O) that was a weighted average of trap catches in the previous 3 years (T I) as follows;

$$B I O = 3.19*(T I*exp. 1.568)$$

In Table 2 we have calculated Rose's trap index (TI) for the period 1985-90 and used the above predictive model to predict 4+ biomass levels for the years 1986 to 1991 for direct comparison with equivalent biomass estimates from the 1991 assessment (Baird et al. 1991), and the correlated error model. Clearly, the trap predictive index provides a close match to the biomass levels estimated from the 1991 assessment, including the projections from that assessment of stock biomass for 1991. On the other hand, the trap index is quite inconsistent with the estimates of 4+ biomass from the correlated error model of Myers and Cadigan (1995b).

It can be argued that the trap index, averaged over a lag period of 3 years, might mask a sudden decline in abundance in the late 1980's -early 1990's. This possibility was evaluated using the raw (i.e. direct year-to-year) relationship between annual trap catches and age 4 recruitment biomass (which underpins the trap index of Rose 1992) for the converged part of the SPA (i.e. up to 1985) to predict age 4 biomass for the period 1986-1991. The relationship for the calibration period (Fig. 6) is statistically significant ($Rsq=0.82$, $p=0.01$) and demonstrates that, at least up to 1985, trap catches were basically a simple function of recruitment variations. Did this simple relationship hold through to 1991 when the correlated model estimates show the stock to be undergoing a drastic decline? In Figure 7 (top panel) we show the fit of the 1986-91 points from an SPA calibrated by trap catch-at age data for the years 1978-91 (Baird et al. 1992). The 1986-91 paired comparisons in Figure 7 bracket the historical relationship and in particular the 1990 and 1991 points are very consistent with those for similar recruitment levels estimated from the converged SPA for the 1980's. On the other hand, the correlated error model shows the

1988-91 estimates to be anomalous with respect to the historical relationship (Fig. 7, lower panel) and implies that trap effort must have at least doubled during the late 1980's-early 1990's. There is certainly no evidence for such an increase in trap effort from the log-book data of Murphy and Shelton (1997), extrapolated to the total trap catch (Fig. 3). In fact, trap effort appears to have remained quite stable, consistent with the hypothesis of Rose (1992).

EVIDENCE FROM THE INSHORE GILLNET INDEX

Logbook estimates of gillnet catch and effort in inshore areas of 2J3KL are available for the years 1985-91 (Murphy and Shelton 1997). Sampling deficiencies noted above for the trap index apply equally for gillnets and we have therefore restricted our analyses to the years 1986-91. Gillnet catch-rates (annual mean catch per net-day) show a slight decline from 1986 to 1990 (Fig. 8, top panel), but a substantial decline in catch-rates occurred in 1991. Estimated inshore gillnet effort was relatively constant up to 1990 but dropped dramatically in 1991 when extreme coastal ice conditions in the summer of that year restricted effort particularly in 2J and 3K. Comparisons of gillnet catch-rates with age 5+ biomass from the 1991 Assessment and the correlated error model (Fig. 8, lower panel) indicates that the gillnet catch-rates are more consistent with the 1991 Assessment than with the correlated error model up to 1990 but that the correlated error model is more consistent with the 1991 catch-rate. The substantial drop in gillnet catch-rates in 1991 can be interpreted as either reflecting mass mortality of the older age-groups in 1991 or, given the extreme ice conditions, a major change in the distribution and/or catchability of the older cod in that year. The gillnet catch-rates do not reflect the magnitude of the biomass decline estimated by the correlated error model at least up to 1990, but sampling deficiencies are such that this conclusion must be verified from other data sources.

EVIDENCE FROM INSHORE AND OFFSHORE TAGGING EXPERIMENTS

The conclusion, from both theory (Rose 1992) and data (Fig. 3 and Fig. 8), that inshore effort has remained relatively constant during the 1980's can be resolved by examining a data source that is independent of the inshore commercial indices. We refer to tagging experiments which have been conducted in both the inshore and offshore areas of the northern cod distribution in the past several decades (Taggart et al. 1995). These can be used to provide indices of inshore exploitation rates as an independent estimate of inshore effort trends. We shall first examine exploitation indices from inshore tagging data.

These data have been analysed by Myers et al. (1997) who considered multi-year returns from tagging experiments that were conducted in inshore areas of 3KL during the fall-spring period. The authors considered these inshore over-wintering remnants to be local "inshore" populations and modelled inshore

tag returns (Discs, and Discs and Dangers combined) from each experiment over a 6 year recapture period to provide estimates of " inshore " exploitation rates. Myers et al. (1997) concluded that inshore exploitation rates on these stock components were extremely high and at unsustainable levels throughout the 1980's and early 1990's. It is important to remember, in the interpretation of these data, that the normal assumption of random mixing was not considered by the authors who were mainly interested in some measure of local exploitation rates. Therefore ,annual estimates of inshore exploitation rates from these experiments may not be representative in all cases. This can be best illustrated by comparing tag return rates from the first year (Fig. 9), when random mixing may not have occurred, with those from the same experiments during the second year (Fig. 10) when the assumption of random mixing was likely valid. Return rates during the first year (Fig. 9) were very high for several experiments conducted in 1963 (Myers et al. 1997, Table 4) that were not evident in the second year's returns from the same experiments (Fig. 10) or, indeed from returns in 1963 during the second year of the 1962 experiments. In general however, returns during the second year are linearly related to the initial numbers released as would be expected if exploitation rates are constant. This is also true for the first year returns excluding the 1963 experiments (Fig 11). That us to say, there is nothing unusual in these returns (1963 excepted) that would exclude them from being representative of inshore exploitation rates. In Figure 12 estimates of inshore exploitation rates from the base case model of Myers et al. (1997) are shown in comparison to raw return rates during the first year. First year returns are displayed since they are usually the simplest and best estimates from tagging data because tag return numbers are generally adequate and because fewer assumptions are necessary for the interpretation of the data than would be the case for long-term returns; for example the model of Myers et al. (1997) assumes that exploitation rates are constant, that there is no long-term tag loss, and that tag detectability and tag reporting rates remain unchanged over the 4-6 year return period. It is clear in Figure 12 that first year returns closely match the model estimates for the experiments conducted during the 1960's, the difference being that the model estimates are somewhat higher because they include model-derived adjustments for the effective tagging rate (i.e. the product of initial tagging mortality, reporting rate, detection rate, tag loss etc.). However, the model estimates for the experiments conducted in the 1979-90 period are several times higher than that inferred from first year returns and imply exploitation rates in the order of 35-40%. The difference, as Myers et al. (1997) point out, is a direct reflection of a major reduction in model estimates of the effective tagging rate, from about 80% in the base model estimates for the 1960's to 30-35% for the later experiments. Since other aspects of the tagging experiments have remained the same (tag type, application technique) the authors ascribe the difference mainly to changes in tag detection likely as a result of the decline in the salt cod industry and therefore the extent to which fish are individually handled. But, is this argument valid or are there other aspects of

these experiments that can account for the apparent discrepancy in the more recent experiments?

First of all, the argument that detection rate changed because of a decline in the salt cod fishery is likely fallacious. The salt cod fishery was already in substantial decline by the mid 1960's and cannot account for the more than 2-fold decline in detectability from the 1960's to the 1980's. Further, the diversion of product to frozen fish production assured individual handling of each fish on the filleting line so that detectability opportunities likely remained unchanged. Also, minimum estimates of detection rate can be inferred from maximum accrued tag return rates; Myers et al. (1997) provide estimates of a 78% return rate for an experiment conducted in Trinity Bay in 1991 during the first year of release. Returns from this experiment in subsequent years would have undoubtedly increased the total return rate had there been commercial fisheries in 1992 and beyond ie detection rates likely approach 100% and are unlikely to have changed significantly for the same tag types from the 1960's to the 1980's. Finally, mean survival rates during the first year, estimated as the ratio of year 2 returns to year 1 returns (Table 3 from Myers et al. 1997), are almost identical for the two tagging periods (34% for the 1960's versus 36 % for the 1980's) and contradict the major increases estimated by Myers et al. (1997). This is not to say that, occasionally, local stocklets are not subjected to high exploitation rates; this was obviously the case for a small tagging experiment conducted in Trinity Bay in 1991 in which local returns were extremely high (Myers et al. 1997). However, such exceptional cases are not representative of the overall inshore fishery. This means that the major increases in mortality estimated by Myers et al. (1997) for the 1980's are driven by differences in tag return rates beyond year 2 , and point to a likely violation of one of the several assumptions necessary to accept estimates from such long-term tag return data. In any event, tag returns beyond year 2 for the 1980's are extremely small (generally a tag or two per experiment, Fig. 13) and are likely insufficient for estimation purposes. We can therefore reject the modelled estimates of inshore exploitation rates by Myers et al. (1997) for the 1980's and consider only estimates from first and second year returns.

We shall consider only the 1970's onwards when, according to Myers et al. (1997), effective reporting rates were unchanged. Tagging experiments meeting the above criteria (as reported in Myers et al. 1997) were carried out in 1979, 1981, 1984 and also in 1990. We shall be consistent with Myers et al. (1997), and restrict our analyses to Disc/Dangler tags only, as these had the highest rates of return and were the only tag types to have been used over the entire time series. Given that tag types remain unchanged, that tag application procedures were carried out according to standard procedures (eg. see Lear and Rice 1987) ,and that reporting rates in this period were the same (Myers et al. 1997) the only variable (other than changes in M) that could significantly affect relative tag return rates for the 1979-90 tagging experiments is a change in

exploitation rate. That is to say, a plot of tag returns against tag releases should show returns from the 1990 experiment to be anomalously high, if inshore exploitation rates doubled by the early 1990's, as suggested by the correlated model in Figure 8. In Figures 14 and 15 we show plots of inshore tag returns versus tag releases for the various tagging experiments for year 1 and year 2 returns only (returns beyond year 2 were insufficient in number). A linear fit to the data provided a statistically significant relationship for both year 1 ($Rsq=0.86$, $p<0.01$) and year 2 ($Rsq=0.99$, $p<0.01$) returns. More particularly, returns from the 1990 experiment in 1990 (Fig. 14) and 1991 (Fig. 15) are consistent with return rates per unit tag release that were observed in the 1980-85 period. This is strong evidence that inshore exploitation rates were relatively constant during the 1980's and early 1990's.

EVIDENCE FROM OFFSHORE TAGGING EXPERIMENTS

A considerable number of tagging experiments were conducted in offshore areas of 2J3KL during the 1980's and 1990's. Details of these tagging experiments, including return statistics, are provided in Taggart et al. (1995). These offshore experiments permit tagged fish to become randomly mixed prior to and during their onshore migrations and should provide a more reliable index of exploitation rate than inshore taggings which sometimes may reflect only localized returns.

From these tag return statistics we have calculated an annual exploitation index defined as the simple ratio of tag returns during the first year to the number released ie the nominal recapture rate over a complete seasonal cycle of inshore and offshore fisheries. These return rates are shown by experiment in Figure 16 (top panel) for the period 1983 to 1992, and the lower panel of Figure 16 shows the annual exploitation index in relation to age 4+ fishing mortality rates from the 1991 SPA (Baird et al. 1991), and the correlated error model. The exploitation index has considerable variability from year to year and within years but, on an annual basis, is relatively stable from 1983 to 1990 and then declines during 1991-92. In a general sense the index is quite consistent with the estimates of fishing mortality from the 1991 Adapt but is totally inconsistent with the 3-fold increase in fishing mortality up to 1991 as estimated by the correlated error model.

Further, the offshore tagging experiments, in conjunction with the subsequent inshore fishery, can be used to provide estimates of northern cod biomass from 1990 to 1992 for comparison with the two divergent estimates of cod biomass from the 1991 assessment and the correlated error model. We have chosen only those tagging experiments which occurred after the winter offshore fishery in order to avoid associated fishing mortalities of tagged fish; the offshore releases also permit mixing of the tagged fish as they migrate towards the coast. For 1990, we have a single experiment conducted in June 17-18,

consisting of 1016 Disc & Dangler tags released on the north cape of Grand Bank. In 1991, we have 5 experiments conducted from May 16 to June 28, mainly on the northern part of 3L, totalling 3158 D & D releases and for 1992 we have a single experiment of 1839 releases on June 7-17, in the southern part of 3K. Tag returns from the subsequent inshore fishery in 1990, 1991 and 1992 can be used to provide estimates of population biomass in those years using the simple Peterson mark-recapture model (Ricker 1975);

$$\text{PopBio} = I C B * T * p * r // R$$

Where I C B = inshore catch biomass in the year of tagging

T = number marked

p = proportion of tagged cod which survive tagging and handling
= 0.87 (Lear and Rice 1987)

r = proportion of recaptures reported
= 0.72 (Lear and Rice 1987)

R = number of inshore recaptures in the year of tagging

Ignoring minor adjustments for initial tag recaptures and natural mortality, we have a total of 167 tag returns from an inshore catch of 117,000 tons in 1990 which provides a Peterson biomass estimate of 450,000 tons. Similarly, for 1991 we have a total of 178 tag returns from a catch of 61,000 tons giving a Peterson estimate of 675,000. For 1992 a total of 17 tags were returned from an inshore catch of 11,850 tons for a Peterson estimate of 800,000 tons (with obvious wide confidence limits). These mark-recapture biomass estimates are consistent with SPA biomass estimates from the 1991 Adapt for 1990 (about 600,000 tons of 4 + biomass, after removal of the 1990 offshore catch and natural mortality) and the 1992 adapt for 1991 (about 550,000 tons for the RV calibrated Adapt and about 800,000 tons for the trap calibrated Adapt). The mark-recapture estimates are, however, much higher than the similarly adjusted 4 + biomass estimates by the correlated error model for the same period (about 250,000 tons, 150,000 tons and 110,000 tons for 1990, 1991, and 1992 respectively).

EVIDENCE FROM SPANS BIOMASS INDEX

There is an additional index of northern cod which can also be used to evaluate whether or not northern cod were already severely depleted by 1991 as hypothesized by Myers and Cadigan (1995b). We refer to the SPANS index (Kulka et al. 1996) which is an alternative formulation of the offshore commercial catch-rate index. This method takes into account not only variation in stock density (i.e. catch-rate) but also changes in the spatial distribution of the stock which, together, provides a stock biomass index. This approach takes care of one of the major criticisms of mobile gear catch-rate indices viz. that high catch-rates can be maintained in a declining population as it concentrates itself into a

smaller and smaller spatial area. An initial version of this index was used as an ADAPT calibration tuner at the 1992 assessment of northern cod (Baird et al. 1992) and the final index given in Kulka et al. (1996), remains virtually unchanged. Estimates of cod abundance from the SPANS assessment (Baird et al. 1992, Table ___) are very consistent with that for the trap-calibrated model as well as the projection from the 1991 assessment calibrated by the 1990 fall survey (Fig. 17).

In conclusion, the coherence between the validated trap index, the inshore gillnet index, the SPANS index, expectations of recruitment success from an environmentally-dependent stock-recruit model, exploitation indices from the inshore and offshore taggings, and the population statistics derived from the offshore tagging data provides substantive proof that northern cod abundance was relatively stable, at least up to 1991, and that the offshore fall survey was timely in its depiction of population trends throughout the 1980's and early 1990's. The first hypothesis, that the decline began in the mid-1980's and was exaggerated by the time it was detected by the fall survey in 1992, can therefore be rejected on the basis that it lacks an evidential basis.

CONSIDERATION OF THE MIGRATION CHANGE HYPOTHESIS

We shall now concentrate on the second hypothesis, that a major progressive change in migration behavior occurred in the early 1990's resulting in cod remaining inshore during the entire year, thereby accounting for the high catch rates observed in the sentinel fisheries along eastern Newfoundland. In the absence of contemporary estimates of absolute abundance of cod in inshore areas during the summer fishery we are, by necessity, constrained to a detailed analysis of the sentinel catch-rates and to considerations of other, circumstantial evidence.

EVIDENCE FROM SENTINEL CATCH-RATES

We shall first examine the sentinel catch-rates since they provide the only empirical source of relative inshore abundance for the recent period (1996-97). These sentinel surveys are, in theory, designed to mimic the seasonal, spatial and depth distributions of the inshore commercial fishery and, as such, sentinel catch-rates can be directly compared with historical commercial catch-rates, after taking into account two potential biases. In the first instance, sentinel catch-rates may be high-biased because they do not reflect the seasonal depletion of biomass that would occur in a commercial fishery, if exploitation rates are substantial. In this case, mean catch-rates would be significantly lower than initial catch-rates at the beginning of the fishery. For example, if the seasonal exploitation rate is 50% then mean catch-rates over the entire season will be about 25% lower than initial catch-rates. Sentinel catch-rates could also be high-biased because they do not reflect the gear interference and gear competition

that reduces local catch-rates in a normal commercial fishery during periods of maximum effort. Such interference is most likely to affect movable fixed gear such as gillnets (Paloheimo and Dickie 1964, Rothschild 1977, Rose and Leggett 1989) which can be set across one another in areas of localized abundance. Cod traps, because of the limited number of berths which are fished annually on a traditional or draw basis, would be expected to be largely unaffected by gear interference but would be affected by the interception rate (ie, the seasonal depletion effect.....actually, a survival bonus). Since intense gear competition also reflects high fishing effort its effect will also be partially reflected in the seasonal exploitation rate. It should also be noted that catch-rates of such movable fixed- gears, like gill-nets, may also be biased upwards, relative to sentinel "control " fisheries, because of the "wolf pack" behaviour of mobile vessels as they interactively locate and converge on the most attractive concentrations of fish. In the following analyses we shall use only sentinel catch-rate statistics from the "control " traps and gill-nets which are truly passive in their capture of cod. Finally, we should put in perspective the expectation that, if the Migration Hypothesis is to be rejected, sentinel inshore catch-rates should approximately reflect the proportional reduction in biomass that has been observed in the offshore survey since 1990, after appropriate evaluation for potential biases. Total biomass in the offshore survey has dropped from about 1.2 million tons in 1990 to about 25,000 tons in 1996 (Murphy et al. 1997) , or roughly a factor of 50.

The bias introduced by seasonal depletion can be estimated from inshore exploitation rates during the commercial fishery. We shall choose the 1980's which is the base period for comparison of the relative level of current sentinel catch-rates. The simplest approach is to assume that the entire stock biomass is available to the annual inshore fishery (see Pinhorn 1984). In this scenario, the inshore exploitation rate can be approximated by the ratio of the annual inshore catch to SPA biomass (age 4+), appropriately reduced by the offshore catches during the January to April period (Lear et al.1986). From catch and SPA biomass estimates given in Baird et al. (1991), inshore exploitation rates averaged 13.3% for the converged period of the SPA (1980-85) and 12.0% for the period 1986-90; the overall inshore exploitation rate for the 1980's by this technique is 12.5%. This is almost certainly an under-estimate as tagging studies show that not all fish move inshore during the summer (Lear et al. 1986).

Estimates of inshore exploitation rates during the 1980's can also be estimated from the "inshore" tagging experiments described above, using the standard mark/recapture algorithm (Lear and Rice 1987);

$$U = R / (T * \exp [-t * M / 12] * p * r)$$

where U=exploitation rate
 R=first year inshore returns

T=number of tags applied

t=time, in months, between application and May 1

M=natural mortality rate

p=proportion of tagged cod which survive tagging and handling

r=proportion of recaptures reported

Using estimates of $p=0.87$ and $r=0.72$ from Lear and Rice (1987)
estimates of U for each of the tagging experiments from 1979-90 are as follows

TAGGING YEAR	U
1979	29.4%
1981	16.5%
1984	9.6%
1990	20.1%
Mean	18.9%

Thus, average inshore exploitation rates for the 1980's were in the range 8-19%, depending on whether offshore or inshore tagging experiments are used as a basis. Since we are evaluating the sentinel catch-rates for at least an order of magnitude difference with those reported for commercial fisheries in the 1980's, we need not consider seasonal depletion as a significant bias in such a comparison.

The bias in sentinel catch-rates introduced by gear interference is difficult to measure but can be inferentially assessed from direct comparisons of sentinel catch-rates with commercial rates from the 1997 commercial fishery in 3Ps. We shall refer to gillnets only as these are the standard sentinel gear used in subsequent comparisons. The preliminary estimate of the 1997 commercial catch-rates for gillnets in 3Ps is 57 kg per net-day (arithmetic mean for the May to October fishery (E. Murphy, pers. comm.). For the sentinel gillnets the estimate for the same period is 80 kg per net-day with confidence limits that overlap the commercial mean (B. Davis, pers. comm.). However, the commercial fishery in 3Ps in 1997 was brief and intensive and may not be truly representative of levels in a normal commercial fishery in which fishers have more time to locate and exploit the major concentrations. Given that we consider only "control" gillnets which are constrained to a single site, and that the interception probability of these nets are further reduced because only 3 gillnets are fished compared with 30-50 nets in the commercial fishery, it is likely that as additional data are collected, commercial catch-rates will be superior to those of sentinel gillnets.

We shall assess the 1996 and 1997 sentinel catch-rates of northern cod for traps and gill-nets (B. Davis, pers. comm.) for sustainability across years, gears (i.e. depth and age structure), areas and seasons. Spatial considerations are obviously important since the sentinel catch-rates are only density measures whereas we are more concerned with proportionality with historical biomass levels. Gillnets will be the major gear considered because of extended spatial and temporal distribution of that data series.

In Figure 18 we show the 1996 and 1997 mean (of weekly means for each bay) sentinel catch-rates for 2J3KL for traps, linetrawl and gillnets in comparison with average levels achieved in the commercial fisheries for these gears in the 1980's; similar data are shown for summer (weeks 24-40) sentinel data for the

3Ps stock also for comparison. It is clear that sentinel catch-rates of northern cod were sustained from 1996 to 1997 and that they are up to several times higher than levels characteristic of the historical commercial fishery; this is consistent with fishers views that sentinel catch-rates of cod would represent "good" catch-rates in the commercial context. Northern cod sentinel catch-rates are also roughly comparable to 3Ps sentinel catch-rates and show similar trends from 1996 to 1997. Sentinel gillnet catch-rates are also sustained throughout the summer season at levels above the commercial mean (8 fish per net-day) (Fig. 19). With respect to the spatial distribution from south to north, sentinel gillnet catch-rates are above mean commercial levels for all areas from Placentia Bay to White Bay but are well below normal levels for the Northern Peninsula and Labrador. The relative absence of northern cod in the northern part of its range is significant but may not be unusual for the recent decade as a southward shift in distribution of cod has been evident in the offshore survey data since the late 1980's during a period of cooler waters (Atkinson et al. 1997).

The other spatial dimension of the sentinel data that must be considered is the inshore-offshore distribution. Since the "control " gillnets and other sentinel gears are fished at traditional locations and in traditional depths it is assumed that sentinel catch-rates (i.e. densities of fish) are representative of cod abundance in summer fisheries, at least in the near-shore coastal zone. Insofar as the extent and abundance of cod away from the coast in summer is concerned, we have approached this problem from two perspectives: (a) examining seasonal trends in commercial catch-rates of offshore trawlers for relativities of summer (July to September) CPUE to winter CPUE when the bulk of the population are considered to be offshore, and (b) in the absence of common standard for commercial fishing effort, examining the spatial (coastal vs shelf) and depth distribution of tag returns during summer on the assumption that tag return rates reflect effort distribution and that effort distribution reflects cod distribution.

Offshore catch-rates by month are available as statistical output from General Linear Model estimates of trawler catch and effort statistics that traditionally formed part of the northern cod assessment. Monthly fishing power coefficients for the years 1983-91 (Baird et al. 1992) have been exponentiated and are shown by month in Figure 20. Trawler catch-rates in the offshore areas of 2J3KL are clearly highest in the winter and reach a nadir in July to October at a level approximately 25% of winter levels. Taken at face value, these data indicate that about 75% of the cod that overwinter on the offshore banks migrate inshore during the spring-summer period. Alternative estimates of the inshore migration rate can be roughly inferred from tagging data. As stated previously, inshore exploitation rates for 1980-85 can be estimated from the converged SPA on the assumption that all the survivors of the offshore fishery move inshore. The inshore exploitation rate is estimated to be 13% under this assumption. During this same period Lear and Rice (1987) have estimated inshore exploitation of

offshore-tagged cod to be 8%; differences in these two estimates reflect the degree to which offshore cod migrate inshore. In other words, $8/13$ or approximately 62% of these offshore cod moved inshore. For the inshore tagging experiments analysed above, the inshore exploitation rate is estimated to have been 18.5% for the same period. The ratio $18.5/13.0$ is a measure of the inshore tag density relative to what would have been expected had all the offshore survivors migrated inshore i.e., the inverse of this ratio, $13/18.5 = 70\%$, is another estimate of the inshore migration rate. Taken together, these indirect estimates imply that on average about 70% of the offshore cod are inshore during the traditional inshore fishery.

We also examined the inshore-offshore distribution of tag returns in 2J3KL during July-September. The inshore has been defined as those statistical unit areas (30 km by 30 km) which impinge on the coast from St Mary's Bay to Labrador; the remainder define the offshore. The reference period is the 1980's, consistent with the comparisons for the catch-rate data. In Figure 21 we show the summer distribution of tag returns in the 1980's by unit area. Nearly 90% of the tag returns came from unit areas which were defined as inshore. The depth distribution of inshore tag returns (Fig. 22) indicates that nearly 80 % were caught at depths of 50 m or less. These tagging statistics suggest that, historically, the majority of northern cod are distributed in the coastal zone in the summer period and that within that zone the cod distribution was mainly in shallow water very near the coast.

In summary, estimates from a variety of sources indicate conclusively that historically, most of the biomass of commercial-sized northern cod was located within the coastal zone of eastern Newfoundland and Labrador during the summer period. The fact that northern cod have not been found in significant concentrations in offshore waters during the summer sentinel fishing period is therefore not a substantial bias. It is possible, however, that within the coastal area northern cod are now distributed in a much narrower band than during the 1980's. Anecdotal information from fishers indicate that occurrences of cod in near-shore shallow waters have increased in the last several years, suggesting that cod may have indeed become densely aggregated. However, sentinel catch-rates are several times higher than commercial levels during the 1980's ; this aspect provides compensatory leverage for a significant reduction in area occupied by cod in the coastal zone when biomass implications of sentinel catch-rates are considered.

Conversion of sentinel catch-rates (i.e. relative fish densities) of northern cod to biomass equivalents can be made if the catchability (Q =probability of capturing a fish in a single unit of effort) of that gear is known or can be estimated. Estimation of sentinel gillnet Q can be made if absolute estimates of cod density are available in an area sampled by the sentinel gear. Further, extrapolation of the derived Q to other areas sampled by that gear can be made,

providing the spatial distribution of the fishing effort is known and if the specified gear is fished in a standard fashion in the areas of comparison. Fortunately, both conditions can be met as sentinel gillnets are fished along the coasts of each of the bays and according to standard protocols; also, absolute estimates of cod density can be derived for Placentia Bay in 1997. Bratney and Cadigan (1998) provide a mark-recapture estimate of about 50,000 t for Placentia Bay in 1997. Assuming that sentinel gillnets randomly sample fishing depths to 120 m (B. Davis, pers. comm.) and that all of the estimated biomass was available to that gear, the catchability (Q) of sentinel gillnets during the summer fishery (weeks 24-40) in Placentia Bay in 1997 can be calculated from the following statistics;

AREA (sq.m) (< 120m)	BIOMASS (t) (tons)	AB. DENSITY (tons/sq.km)	REL. DENSITY (fish/net-day)	Q
3840	50,000	13.0	32.1	0.00247

The above estimates of Q was used to convert the bay by bay summer sentinel catch-rates of northern cod in 1997 to equivalent biomass levels using the following formula;

$$\text{BIOMASS} = (\text{CPUE} / \text{Q}) * \text{AREA}$$

BAY	AREA (sq. km)	CPUE (fish/ner-day)	EQUIV BIOMASS (tons)
SMBAY	2427	22.1	21,715
SSHORE	841	21.6	7,355
CBAY	1016	15.7	6,460
TBAY	2039	33.3	27,490
BBAY	2178	28.5	25,130
NDBAY	4085	17.6	29,110
WBAY	995	14.9	6,005

TOTAL EQUIVALENT N. COD BIOMASS 123,265 tons

The above estimate, as calculated, is an underestimate since it excludes cod north of White Bay (area estimates were not available) and any cod at depths greater than 120 m which did not contribute to sentinel catch-rates. Further, the mark-recapture estimate used to estimate Q underestimates the total population within the length range marked at a rate that is directly proportional to the length-specific selectivity of the gear used to obtain the marked sample. This selectivity bias could be substantial, if the population length frequency has a mode in the tails of the selection curve. For example, an acoustic estimate of a 40,000 tons (Rose and Lawson 1998) for a spawning

concentration of cod in statistical district 29 of Placentia Bay in May, 1997 provides a population biomass more than twice as large as the mark-recapture estimate, when extrapolated to the whole of Placentia Bay using the above technique with an estimate of commercial gillnet Q obtained from the spawning concentration concurrent with the acoustic estimate. Northern cod biomass calibrated by this estimate would be in the order of 250,00-300,00 tons. This points out how critical such point estimates of Q are to the above biomass extrapolations. In contrast to these estimates, an acoustic survey of 3KL in the fall of 1997 resulted in a provisional estimate of only 19,000 tons (Anderson et al. 1998). However, substantial potential biases relating to vessel avoidance (Anderson et al. 1998) and reduced detectability of cod in the bottom exclusion zone (G. Rose, pers. comm.) need to be resolved before the acoustic estimate can be accepted.

Nevertheless, taking into account the aforesaid qualitative considerations of the northern cod sentinel data and the above estimates of biomass equivalents, it is concluded that sentinel catch-rates represent significant, and perhaps substantial, biomass of northern cod and are inconsistent with the 50-fold drop in biomass that would have been expected from the trend in offshore surveys during the 1990's. The second hypothesis, that the drop in offshore abundance levels was caused by a major change in migration patterns, cannot yet be rejected. Neither can the the Migration Change hypothesis be accepted until further research, including a quantitative estimate of inshore biomass along eastern Newfoundland during the summer period, clarifies the contribution of local "Bay" stocks and seasonal immigration of cod from 3Ps. These issues are discussed below.

ANCILLARY EVIDENCE AND THE 3Ps LINK

If the Migration Change hypothesis is to be considered as a realistic possibility, then over-wintering of northern cod must occur in the inshore areas of Newfoundland. An alternative hypothesis that could also explain the relative strength of the sentinel catch-rates along eastern Newfoundland is a greater than normal northward extension into 3KL of a resurgent 3Ps cod stock. The 3Ps cod stock contains an inshore component (Templeman's so-called Avalon-Burin complex, Bratley 1996) which has a persistent but variable migration into southern 3L. At the time of its closure in 1993, at least one (1989) and probably two (1990) large yearclasses were about to enter the 3Ps cod stock (Shelton et al. 1996) and the survival benefits of the moratoria should have rapidly rebuilt abundance to levels at which the stock may have expanded its summer range. It need hardly be said that Atlantic cod are extremely migratory and that taggings in both 3Ps and 2J3KL have produced returns from nearly every cod stock in the northwest Atlantic. The following evidence will be presented in support of either one or both aforementioned hypotheses but will likely be insufficient to discriminate between them. We should point out that other possible explanations

are not meant to be excluded (eg. differential mortality of the offshore populations of northern cod).

- (1) The first piece of evidence indicating a likely stronger link between northern cod and 3Ps cod in the 1990's relates to the southward shift in the distribution of northern cod in the 2J3KL beginning in the late 1980's. Atkinson et al. (1997) have described this shift for the offshore areas but there is evidence that this southward shift was also occurring in the inshore area (Fig. 23). Approximately 50% of the inshore catch was landed in 3L up to 1988 but this had increased to nearly 70% in 1990 and to over 90% in 1992 i.e., by the early 1990's it can be inferred that the bulk of the inshore population of northern cod had moved to 3L, an area that in the past has had a significant interchange with 3Ps cod, in particular the Avalon-Burin complex. A cursory review of historical tagging experiments conducted in the October-March period from inshore areas of 3Ps (Taggart et al. 1995) shows persistent returns during the summer period from 2J3KL indicating that at least a portion of northern cod may have historically overwintered in 3Ps.
- (2) Further, seasonal catch-rates of sentinel gillnets in Placentia Bay increase by a factor of about 5 or more beginning around mid-October for all years from 1995 to 1997 (Fig. 24). There is no evidence for such a dramatic increase in the historical gillnet catch-rates (Shelton et al. 1996) for Placentia Bay in late fall, indicating that this massive influx of cod may be a recent phenomenon. One explanation is that an increased proportion of northern cod has shifted to overwintering areas in 3Ps.
- (3) Comparisons of yearclass structure of northern cod and 3Ps cod since the moratoria in 1993 (Lilly et al. 1998, Stansbury et al. 1998) shows that the same yearclasses (1989, 1990 and 1992) are dominant in both areas. In addition, age compositions of acoustically measured spawning concentrations of cod in Trinity Bay (Porter et al. 1998) and Placentia Bay (Rose and Lawson 1998) in the spring of 1997 are almost identical. Historically, similar age compositions would have been expected since the two stocks exhibit parallelism in yearclass success because of common environmental conditions (Myers et al. 1993); however, offshore surveys of age 2-4 cod in 2J3KL and 3Ps (Fig. 25) show a divergence in yearclass success after 1988 despite common environmental conditions, and both survey and mark-recapture data showing adequate levels of spawning stock for northern cod at least up to 1991. The apparent reduction in yearclass success of the 1989-92 yearclasses of northern cod evident in the offshore surveys relative to the success of these same yearclasses in 3Ps is difficult to reconcile, and particularly so, given the similarities of inshore yearclass compositions of these two stocks in the 1990's.

- (4) Mark-recapture estimates of northern cod abundance based on inshore catches in 1990 (450,000 tons), 1991 (675,000 tons) and 1992 (800,000 tons) do not show the declines in abundance that were evident in the 1991 and 1992 spring bottom-trawl surveys in 3L. Typically, the spring 3L survey indicated a stable biomass of 500,000 tons up to 1991 when biomass levels dropped to 173,000 tons and to 81,000 tons in 1992 (Murphy et al. 1997). This indicates that migration patterns changed in 1991 and that tag densities were being considerably diluted by existing populations of cod when the offshore-tagged cod from 3L reached inshore. This apparent shift in spring migrations of cod in 3L coincides with a major shift in the inshore migration patterns of capelin in 3L (Shackell et al. 1994). Prior to 1991 spring acoustic surveys in 3L consistently detected large aggregations of prespawning capelin enroute to coastal spawning beaches along eastern Newfoundland. These aggregations have not been detected in 3L at least up to 1996 (Carscadden 1997) but inshore indices of mature capelin have remained within normal range throughout the 1990's (see Carscadden 1997). This change in migration patterns of capelin in 3L has been attributed to temperature variations (Shackell et al. 1994). The current inshore migration routes of capelin remain unknown.
- (5) Tag returns from tagging experiments in Placentia Bay during the spring and summer of 1996 and 1997 also give evidence that a portion of these cod migrate into 3L. The seasonal pattern of recaptures from four batches of cod tagged at the bottom of Placentia Bay indicate a southward movement of fish down the east side of Placentia Bay during late spring and summer. A large portion of recaptures were reported close to the border with 3L. Due to the moratorium, fishing activity within 3L was minimal and restricted to Sentinel fishers; however, some tagged cod were recaptured in St. Mary's Bay and on the eastern Avalon as far north as Cape St. Francis.
- (6) There are other ancillary sources of information that support the hypothesis that migration patterns of northern cod have changed and that the inshore is now the key to defining current population levels. In the first instance, Schneider et al. (1997) compared the distribution of juvenile cod along eastern Newfoundland during the 1992-95 period with distributions from similar surveys for 1959-64 and found no evidence for contraction in habitat range. This was an unexpected result as both theory and observation would have predicted a major reduction in the range of juvenile cod, given the presumed virtual obliteration of the spawning stock in the 1990's. Further evidence in support of this anomalous observation is available from bycatch rates of juvenile cod in capelin traps in 3KL. These bycatches are primarily 2 and 3 year-olds and bycatch rates are generally higher in 3L than in 3K. In Figure 26 we have plotted the time

series for 1981-96. Excluding 1992 when there was no capelin fishery in Division 3L, bycatches of juvenile cod along eastern Newfoundland have remained at normal levels during the 1990's and for 1996 were the highest observed at 134 kg per haul. The 1996 bycatch level drops to 93 kg per tow with the removal of a single large catch but is still well above the historical average. Previous to the 1990's these bycatch rates were in general agreement with coarse trends evident from the offshore survey (Fig. 26). Additional information on the relative abundance of juvenile cod along eastern Newfoundland/Labrador is available from harp seal stomach contents which have been sampled from inshore areas since the early 1980's (Stenson et al. 1995). These are mainly age 0 to 2 year-old juvenile cod. Consumption levels of these very young cod have increased from the late 1980's and continue to remain high during the 1990's (Fig. 27, reproduced from Stenson et al. 1995). More recent samples from 1994 and 1995 seal stomach analyses continue to show high levels of juvenile cod consumption (G. Stenson, pers. comm). To summarize, evidence of juvenile cod abundance from disparate inshore sources are generally consistent in their depiction that these age-groups are within normal ranges of abundance which is in sharp contrast to the major reductions observed in the offshore surveys.

SUMMARY AND CONCLUSIONS

- (1) The hypothesis that the collapse of northern cod was due to undetected increases in fishing mortality beginning in the mid-1980's can be rejected as being inconsistent with nearly all indices of abundance and exploitation, both inshore and offshore. The only plausible hypotheses remaining are catastrophic mortality of idiopathic origin and migration change.
- (2) The hypothesis that northern cod have changed their migration patterns and are now concentrated in inshore areas cannot yet be rejected given the congruence of evidence from a variety of inshore sources. In particular, inshore sentinel catch-rates in 2J3KL almost certainly represent significant, and perhaps substantial, biomass of northern cod and are inconsistent with expectations from relative declines observed in the offshore. An alternative hypothesis, relating to an exaggeration of an historical link with 3Ps cod, is presented which may partly explain this anomaly.
- (3) Comparison of mark-recapture estimates of northern cod from inshore returns with offshore survey indices indicates that, if catastrophic mortality did occur, it was likely most severe in the short period following the abbreviated inshore fishery in 1992.

- (4) Quantification of biomass levels in inshore areas of 2J3KL during the traditional summer fishing period may be necessary if sentinel catch-rates are to be validated as biomass equivalents.

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Table 1. 1986-92 4+ population biomass of northern cod predicted from the trap Index model of Rose (1992) in comparison with estimates from the 1991 assessment and from the correlated error model. See text for explanation.

YEAR	TRAP INDEX	PREDICTED 4+ BIO	1991 ASSES. 4+ BIO	CORR. ERR. 4+ BIO
1986	39,300	1010	974	770
1987	37,600	940	895	730
1988	33,600	790	794	555
1989	35,900	875	743	385
1990	37,900	955	790	295
1991	43,330	1175	(1150)*	270
1992	41,100	1080	-----	180

* projection

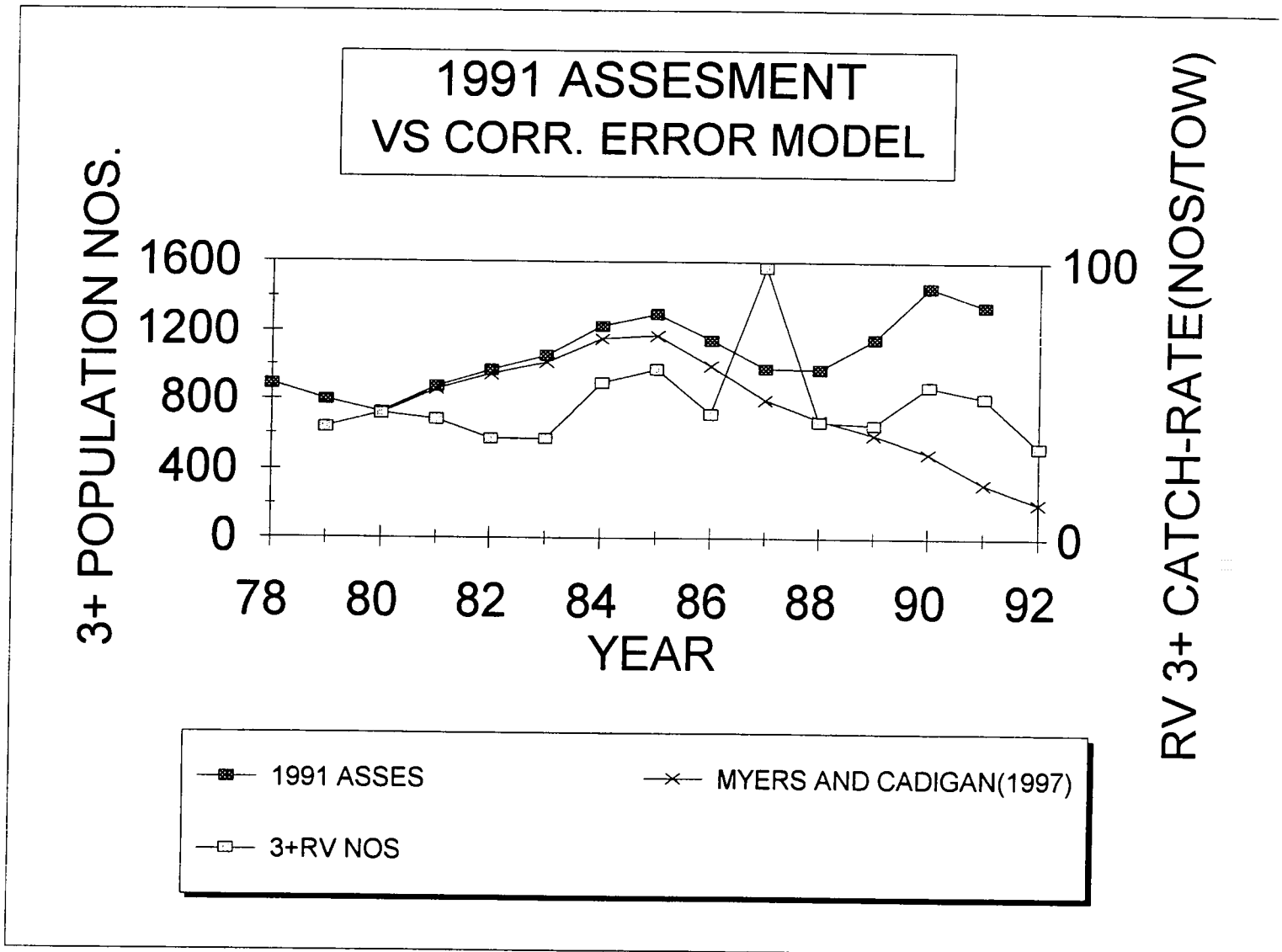
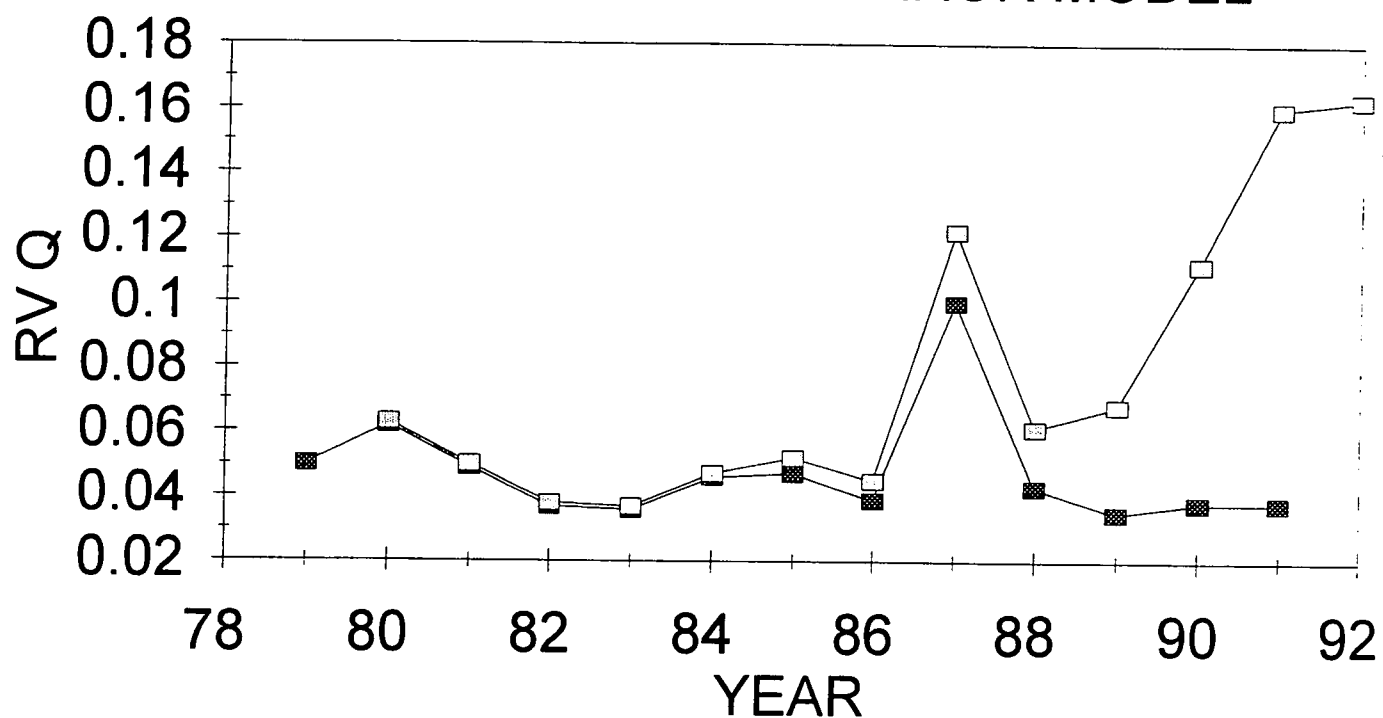


Fig. 1. Adapt population estimates from the 1991 assessment of northern cod (Baird et al. 1991) in comparison with similar estimates from the correlated error model (Myers and Cadigan 1995b) and the fall RV survey.

RESEARCH VESSEL Q

1991 ASSESM. VS CORR. ERROR MODEL



—■— 1991 ASS. —□— C E MODEL

Fig. 2. Estimates of RV catchability (Q) implied from the 1991 Adapt assessment of northern cod in relation to similar estimates implied from the correlated error model.

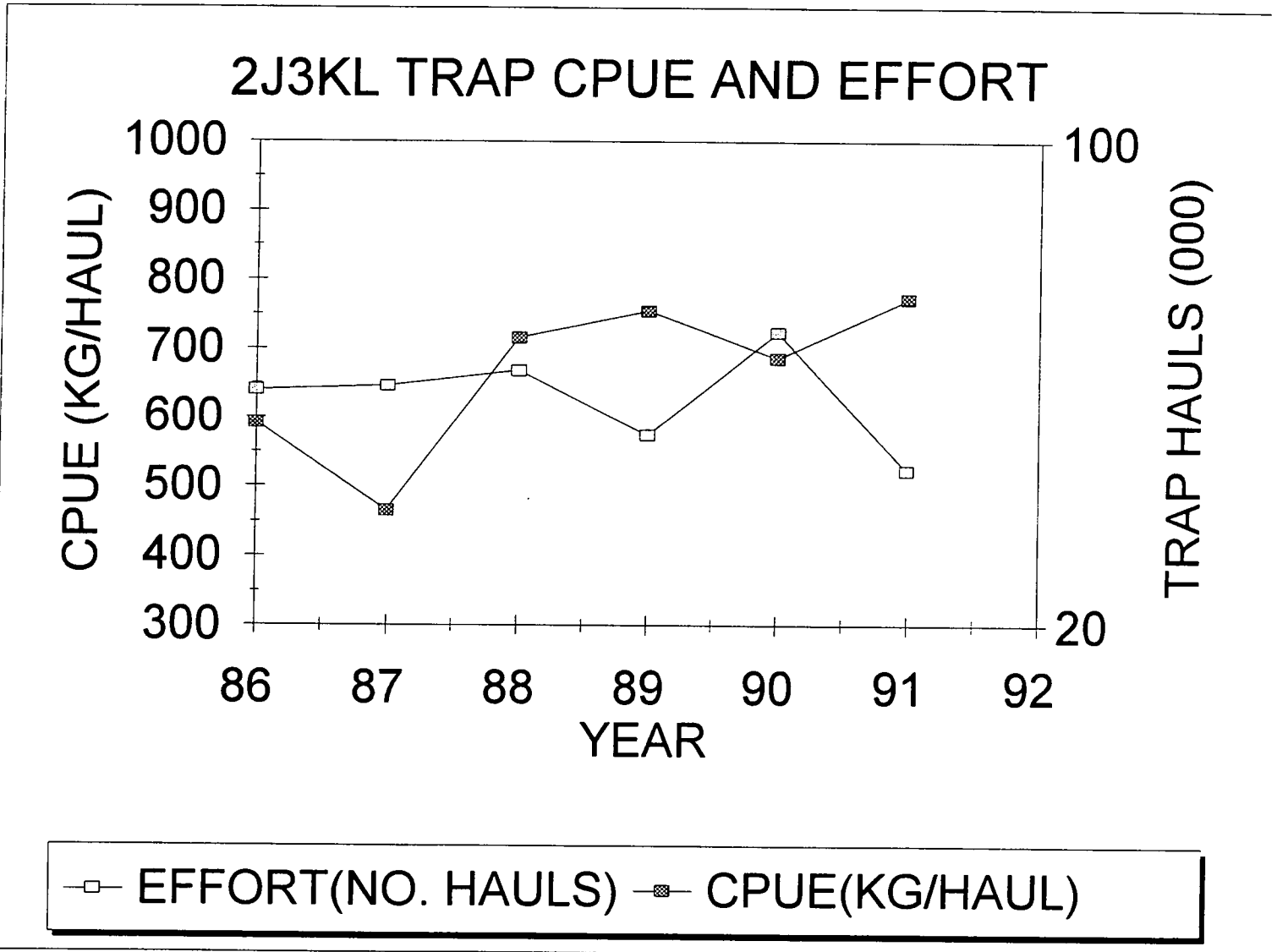


Fig. 3. Estimates of 2J3KL trap catch-rate (kgs per haul) and effort (extrapolated to total catches) from trap log-book statistics.

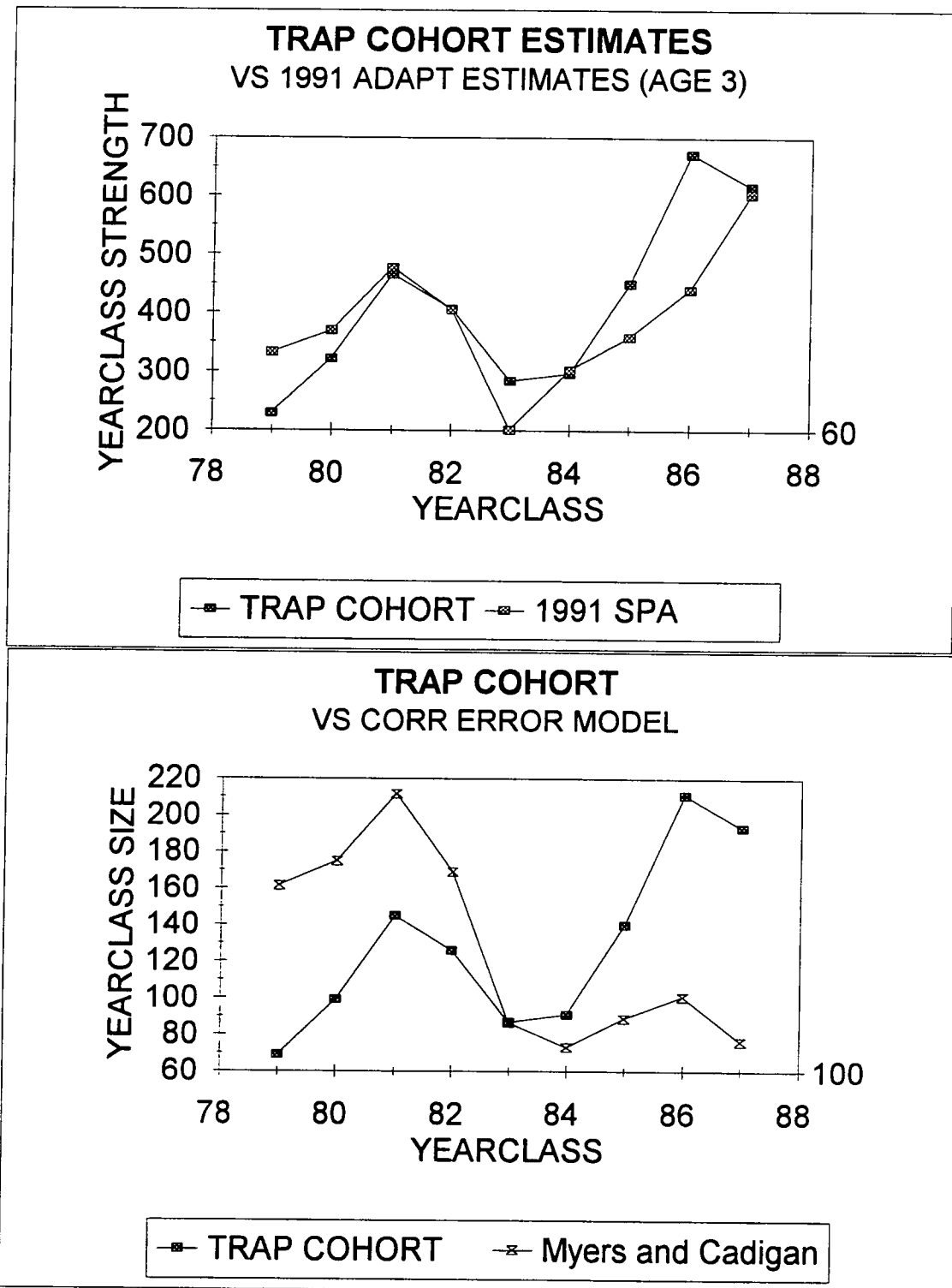


Fig. 4. Standardized estimates of the 1979-87 yearclasses from the trap index in relation to estimates of the same yearclasses by the 1991 assessment (Baird et al. 1991) (top panel) and the correlated error model (lower panel).

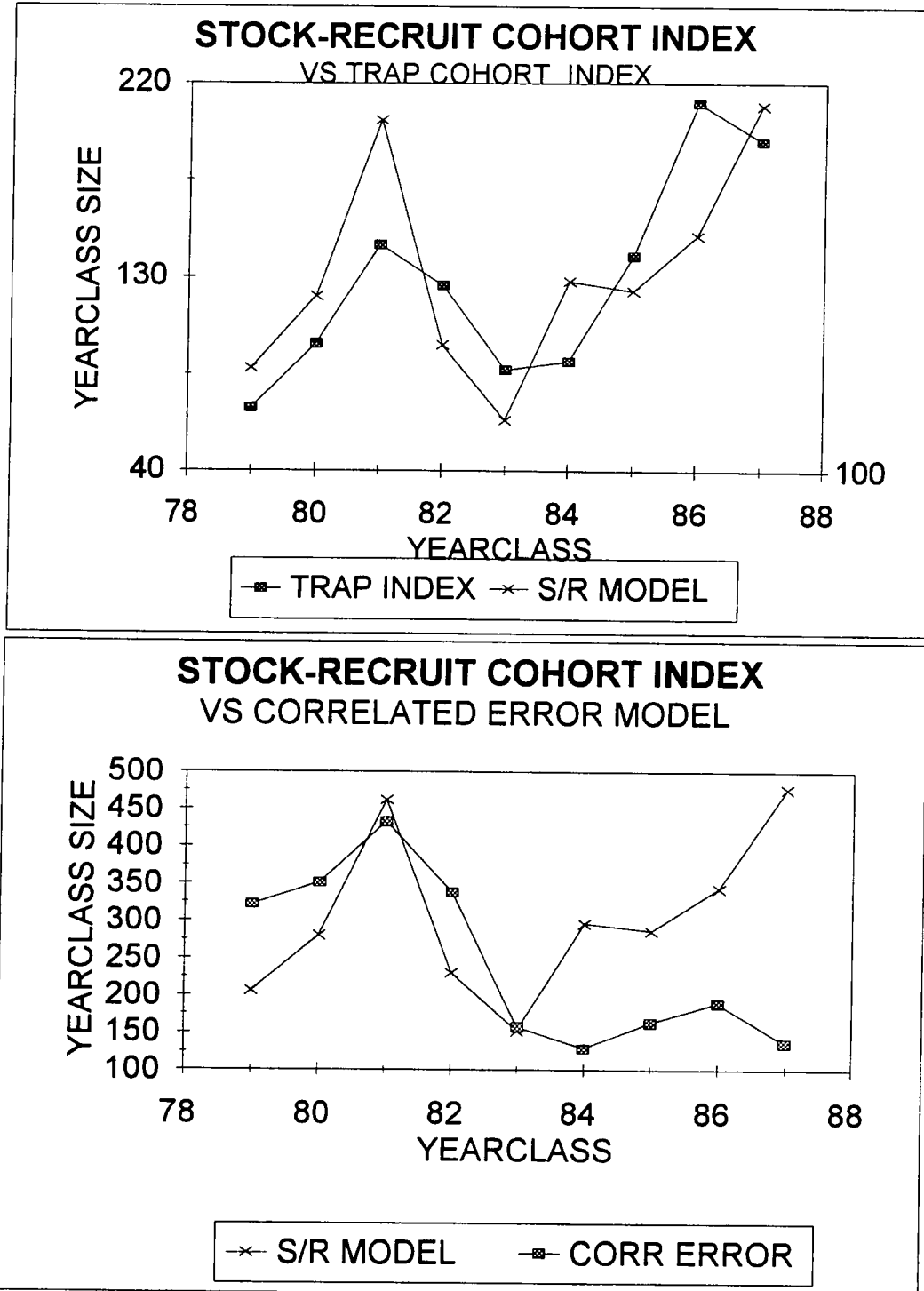


Fig. 5. Estimates of yearclass success by the empirical recruitment model of Myers et al. (1993) in relation to that estimated by the trap index (top panel) and the correlated error model (lower model).

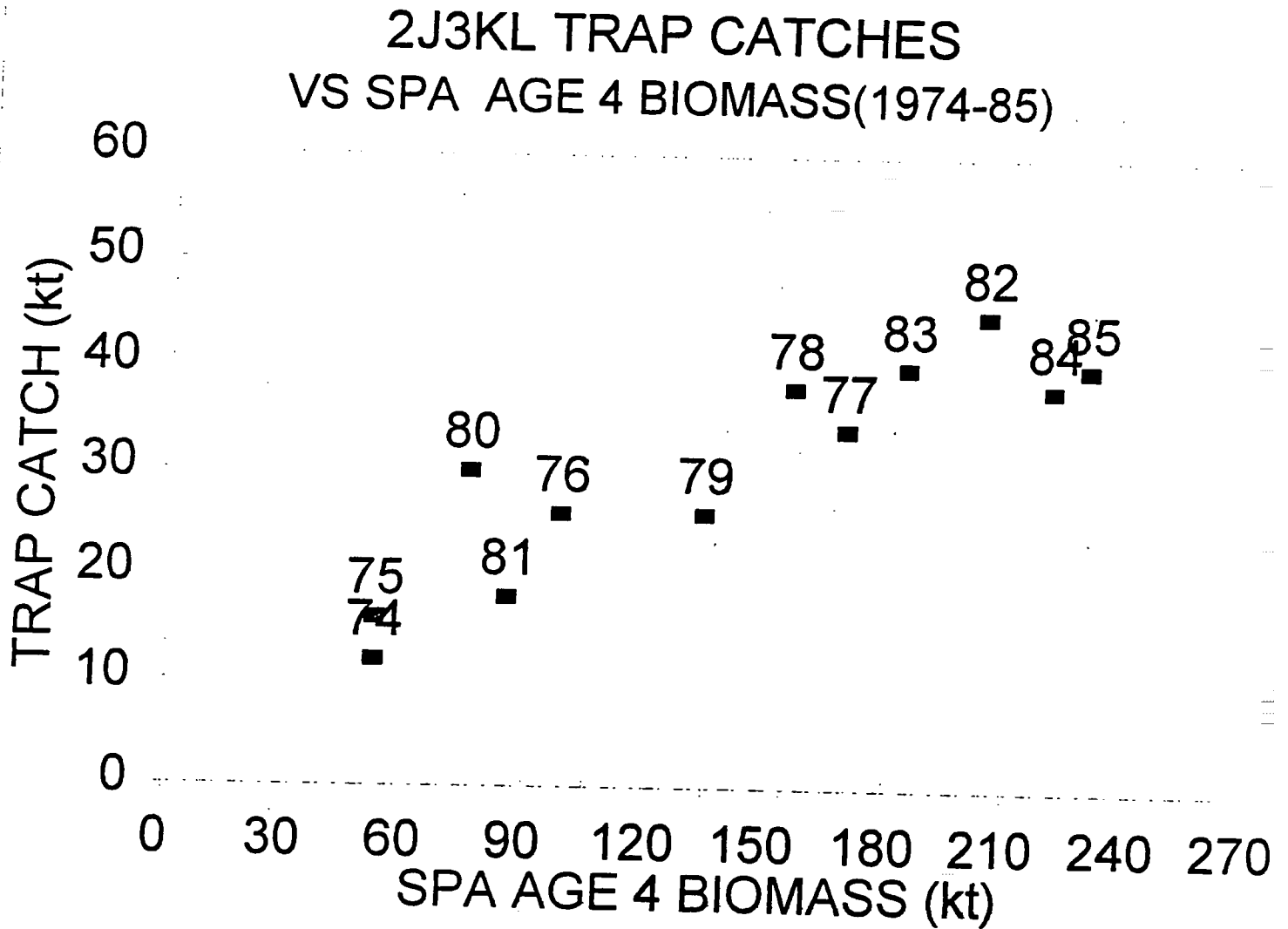


Fig. 6 . Relationship between annual trap catches of northern cod and age 4 SPA biomass for the years 1974-85 (SPA data from Baird et al. 1991).

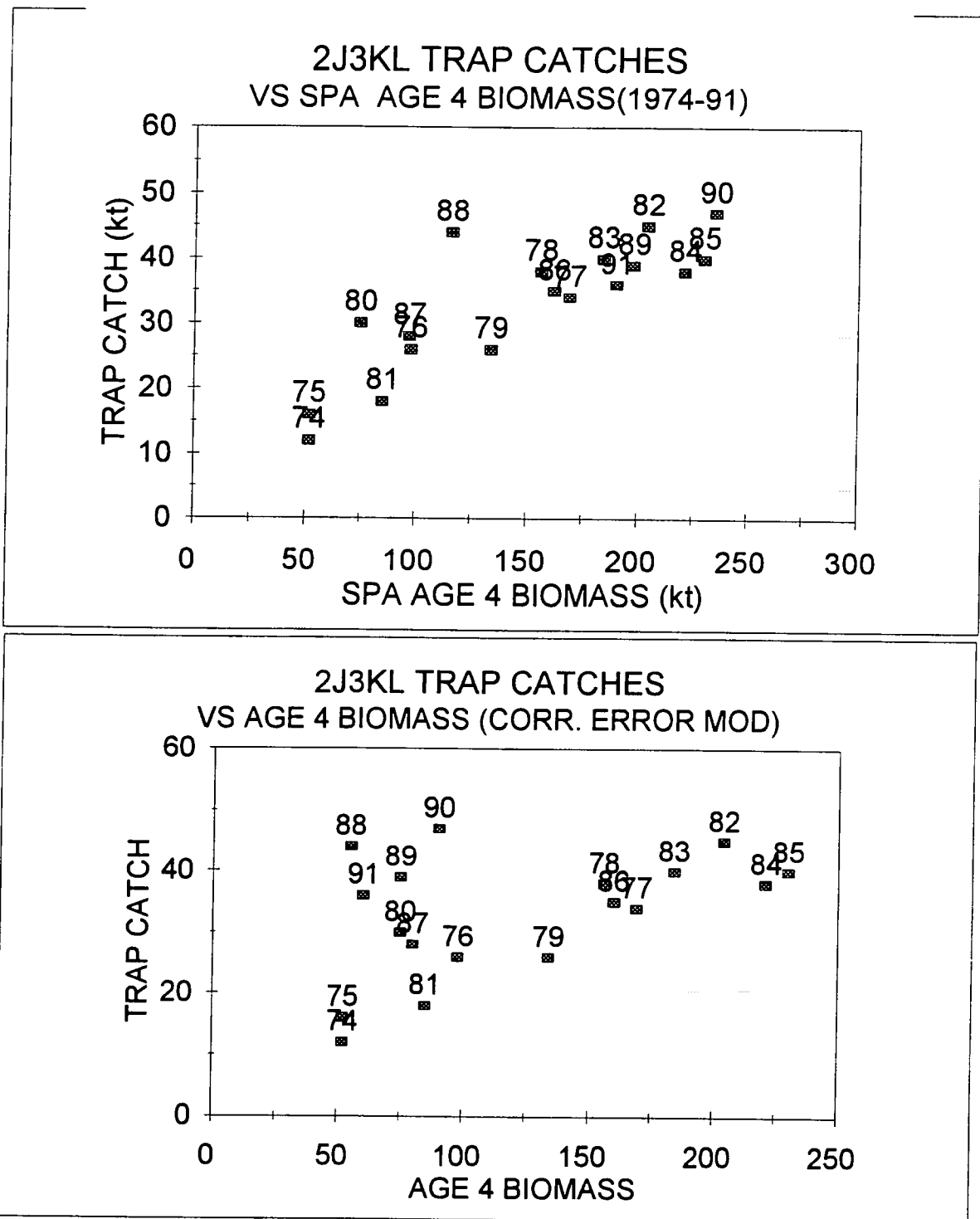


Fig. 7 . Relationship between annual trap catches of northern cod and age 4 SPA biomass updated to include the years 1986-91 from a trap-calibrated SPA (Baird et al. 1992) (top panel); the lower panel shows the relationship of annual trap catches with age 4 biomass estimates from the correlated error model.

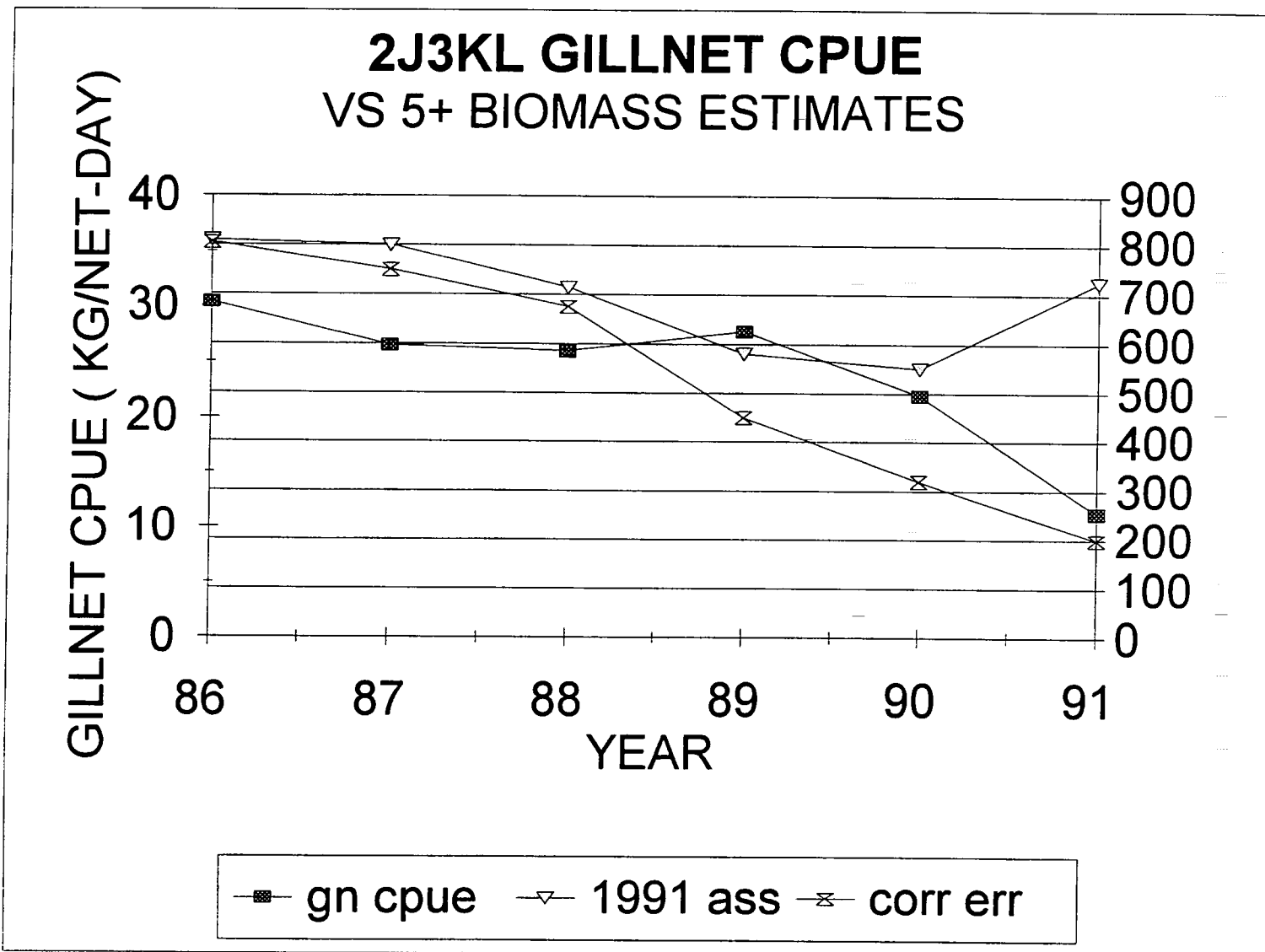


Fig. 8. Trends in annual inshore gillnet catch-rate and effort for the years 1986-91.

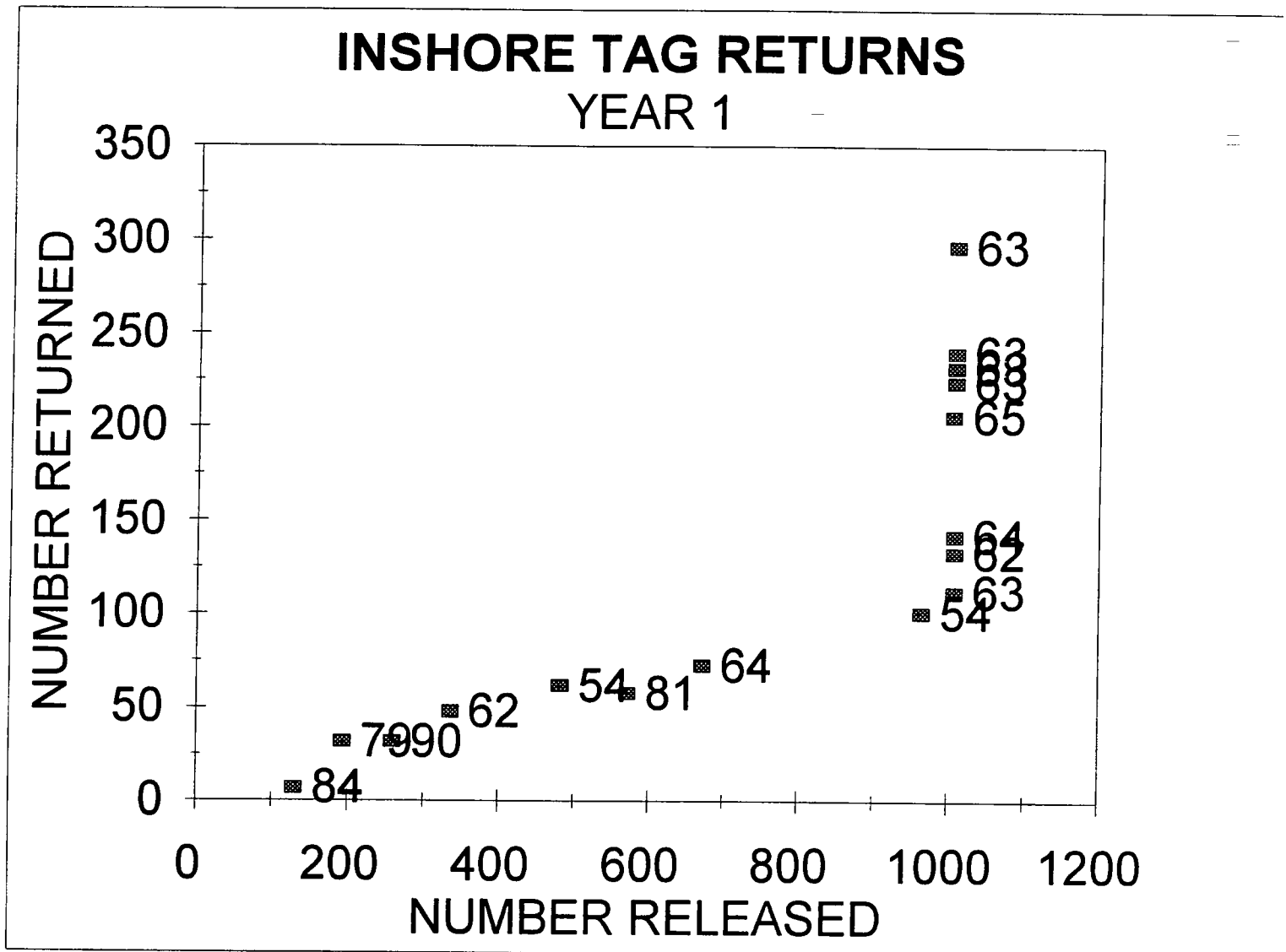


Fig. 9. Relationship between year 1 tag returns and the number of tag releases for all "inshore" tagging experiments from 1954-90 (data from Myers et al. 1997b).

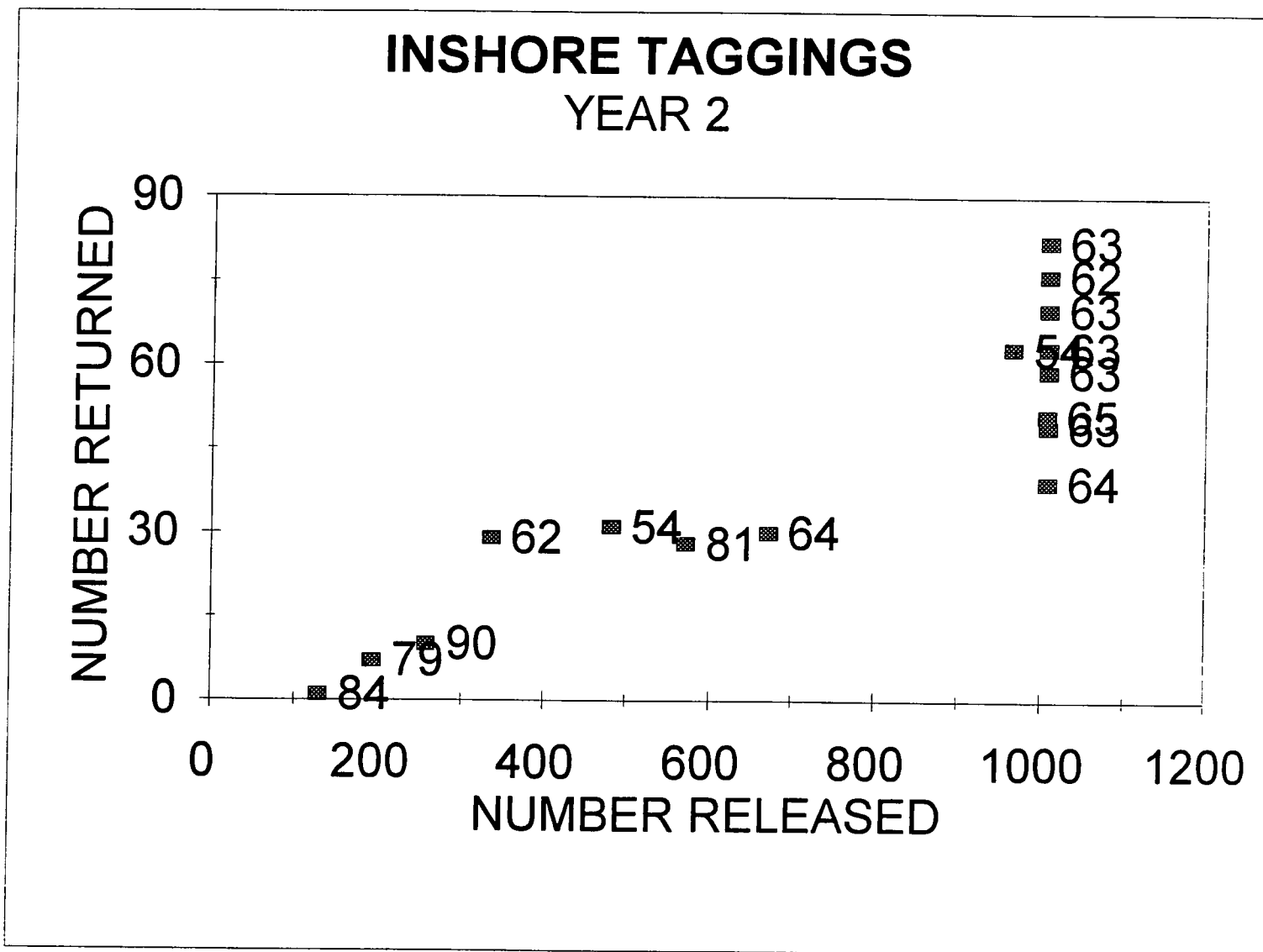


Fig. 10. Relationship between year 2 tag returns and the number of tag releases for all "inshore" tagging experiments from 1954-90 (data from Myers et al. 1997b).

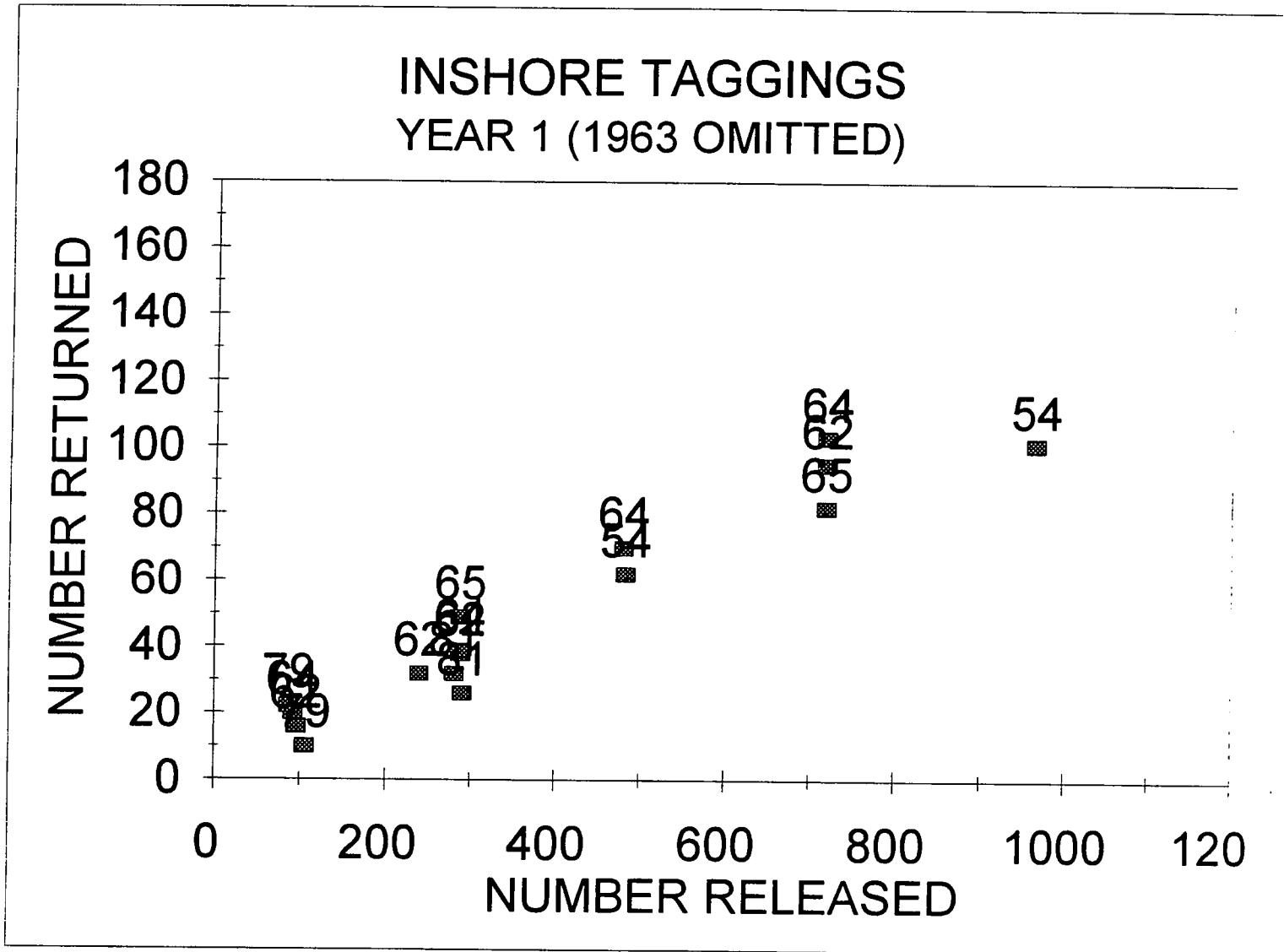


Fig. 11. Relationship between year 1 tag returns and the number of tag releases for all "inshore" tagging experiments from 1954-90 (data from Myers et al. 1997b) (1963 experiments excluded).

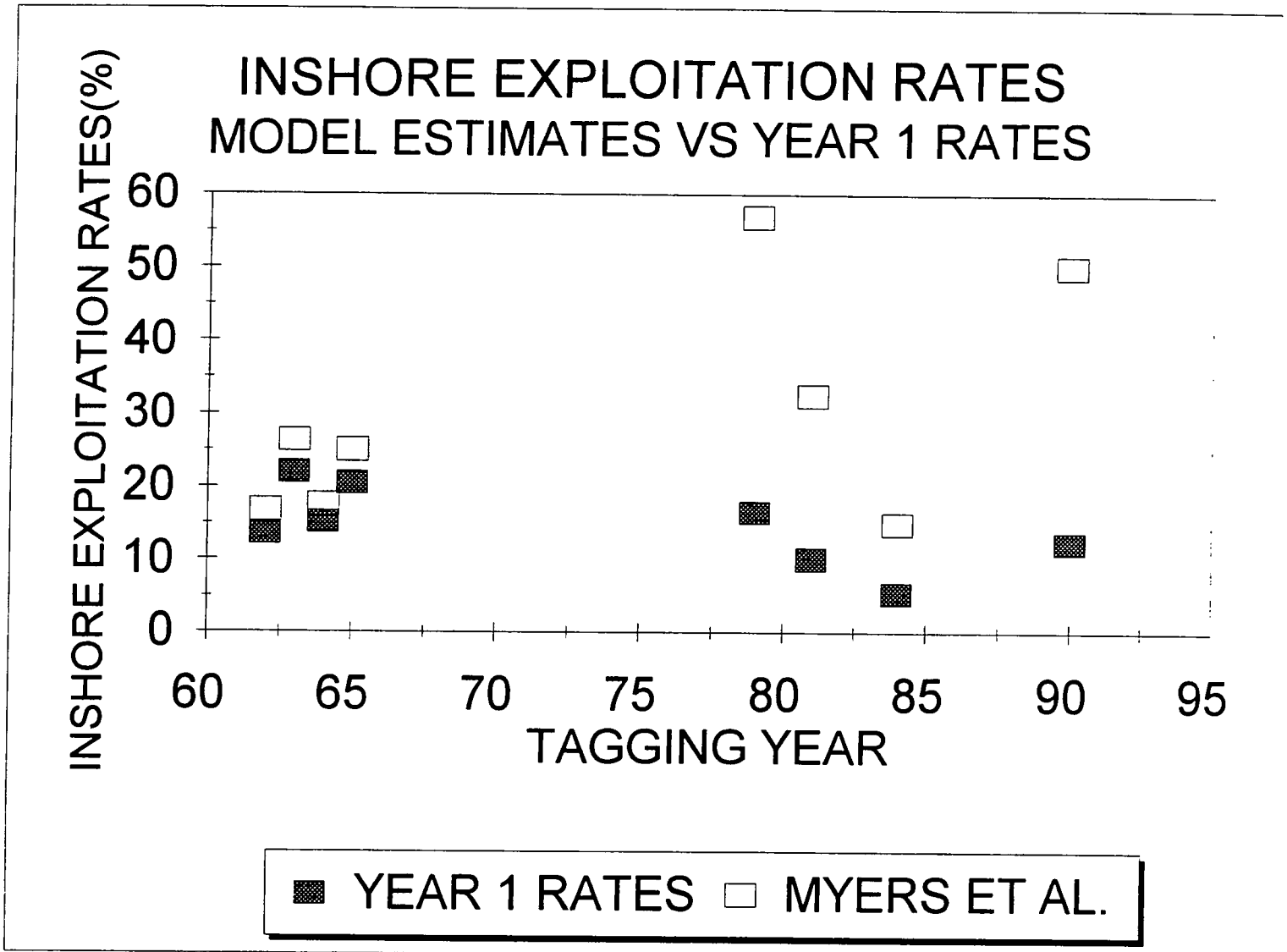


Fig. 12. Inshore exploitation rates estimated from the multi-year tagging model of Myers et al. 1997) in relation to simple estimates from the first year returns.

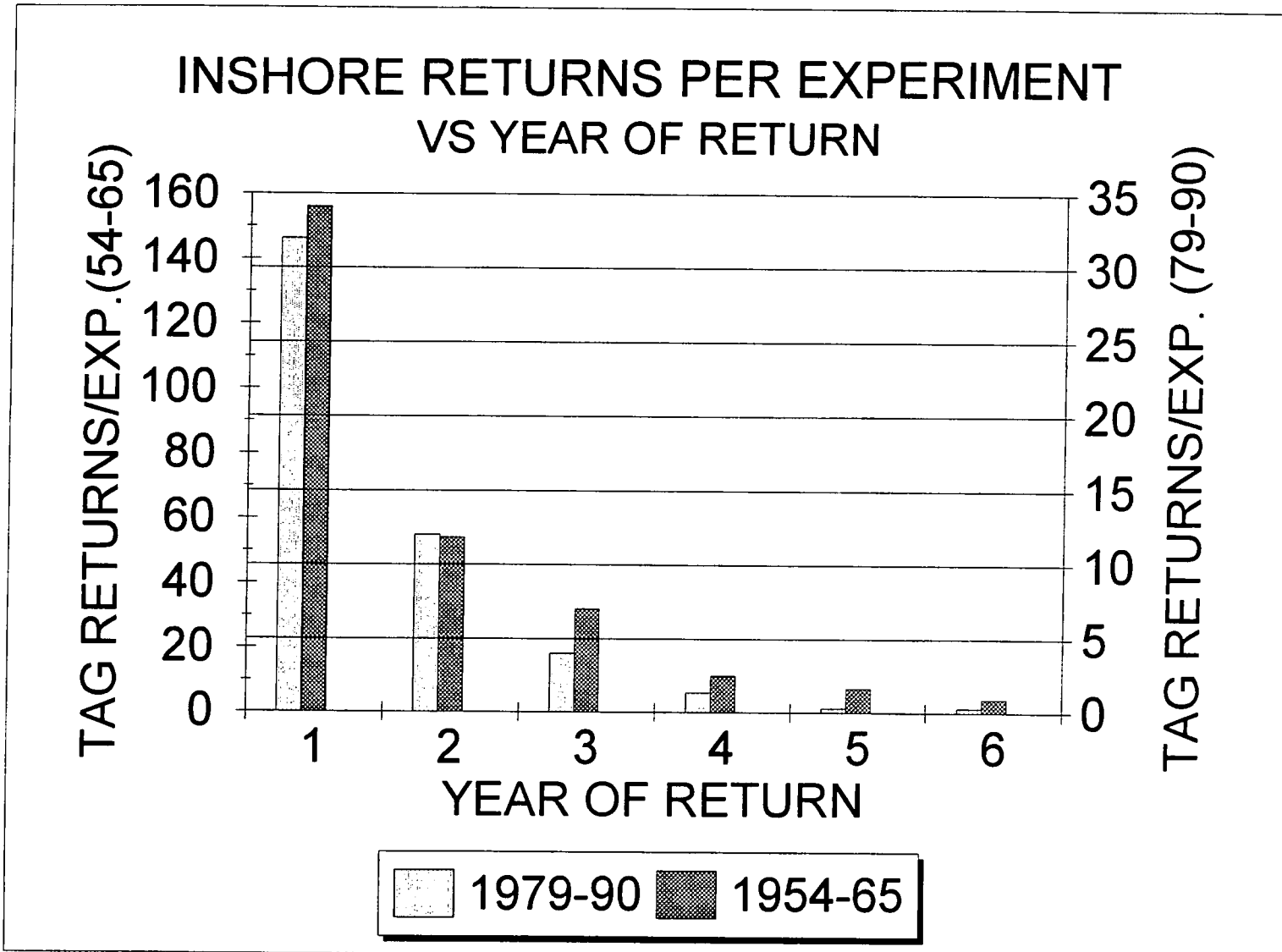


Fig. 13. Mean tag returns per experiment for each year at liberty for the historical tagging experiments (1954-65) in relation to mean tag return rates for 1979-90.

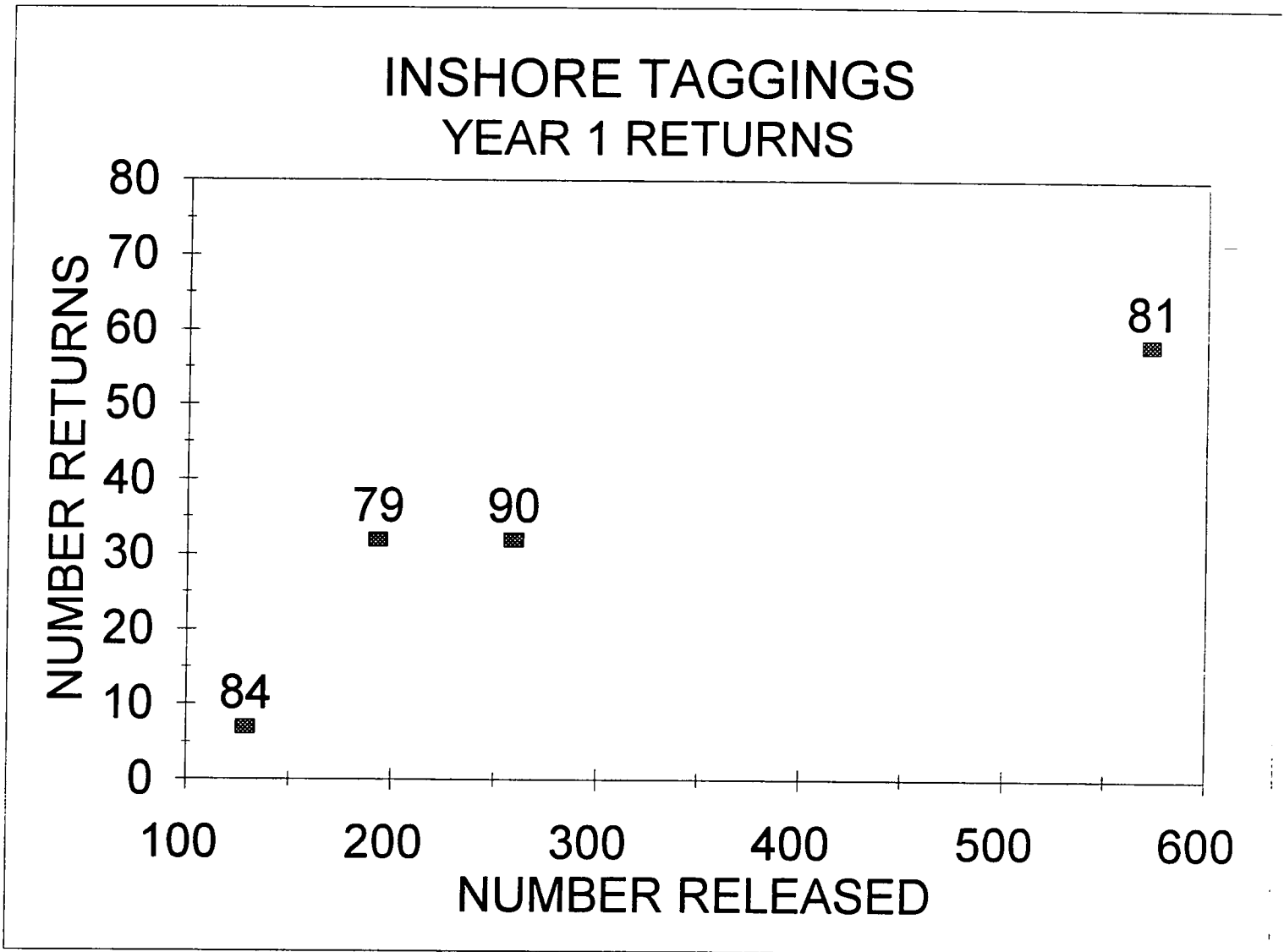


Fig. 14. Relationship between year 1 tag returns and the number of tags released, for the 1979, 1981, 1984, and 1991 tagging experiments (data from Myers et al. 1997).

INSHORE TAGGINGS YEAR 2 (ANNUAL)

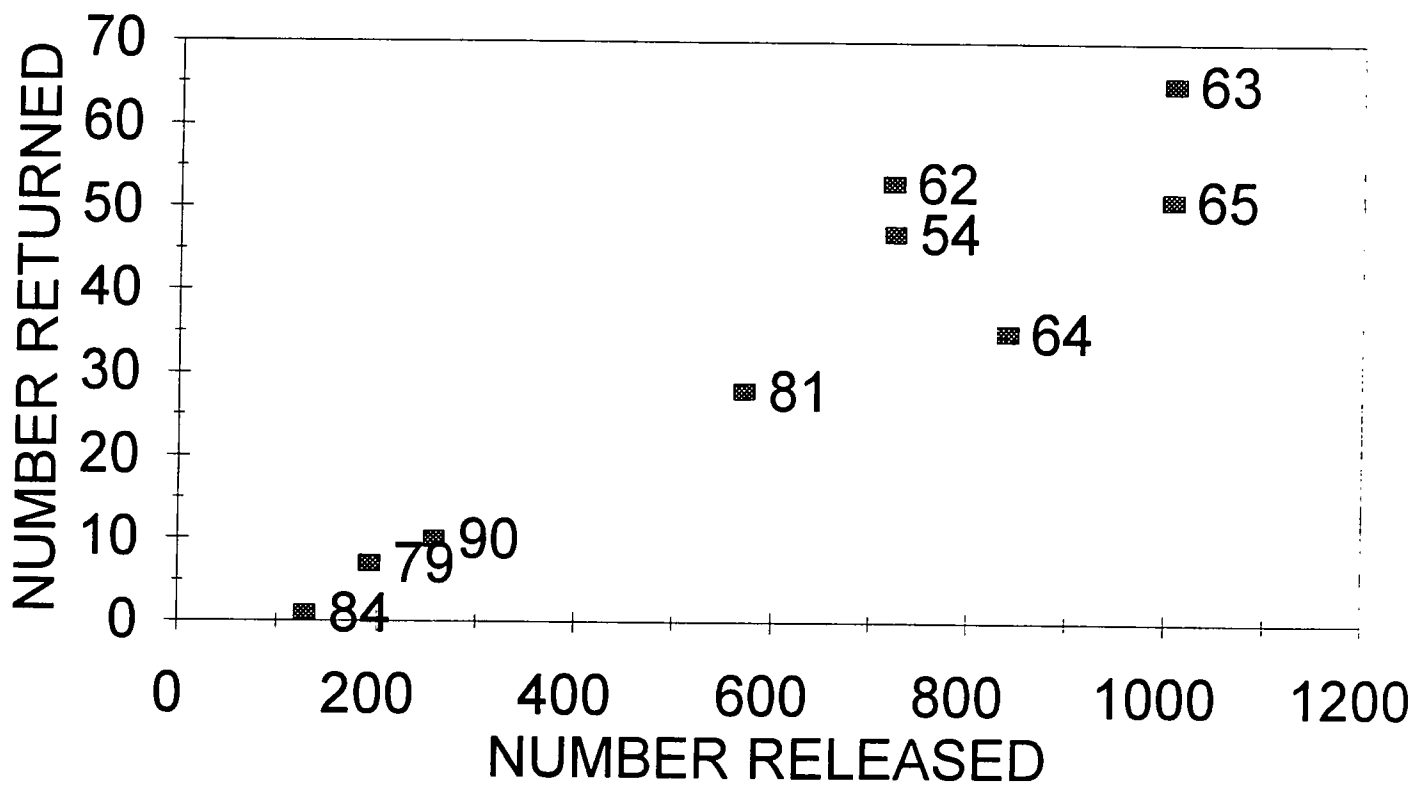


Fig. 15 . Relationship between year 2 tag returns and the number of tags released for the same tag experiments referenced in Figure 9.

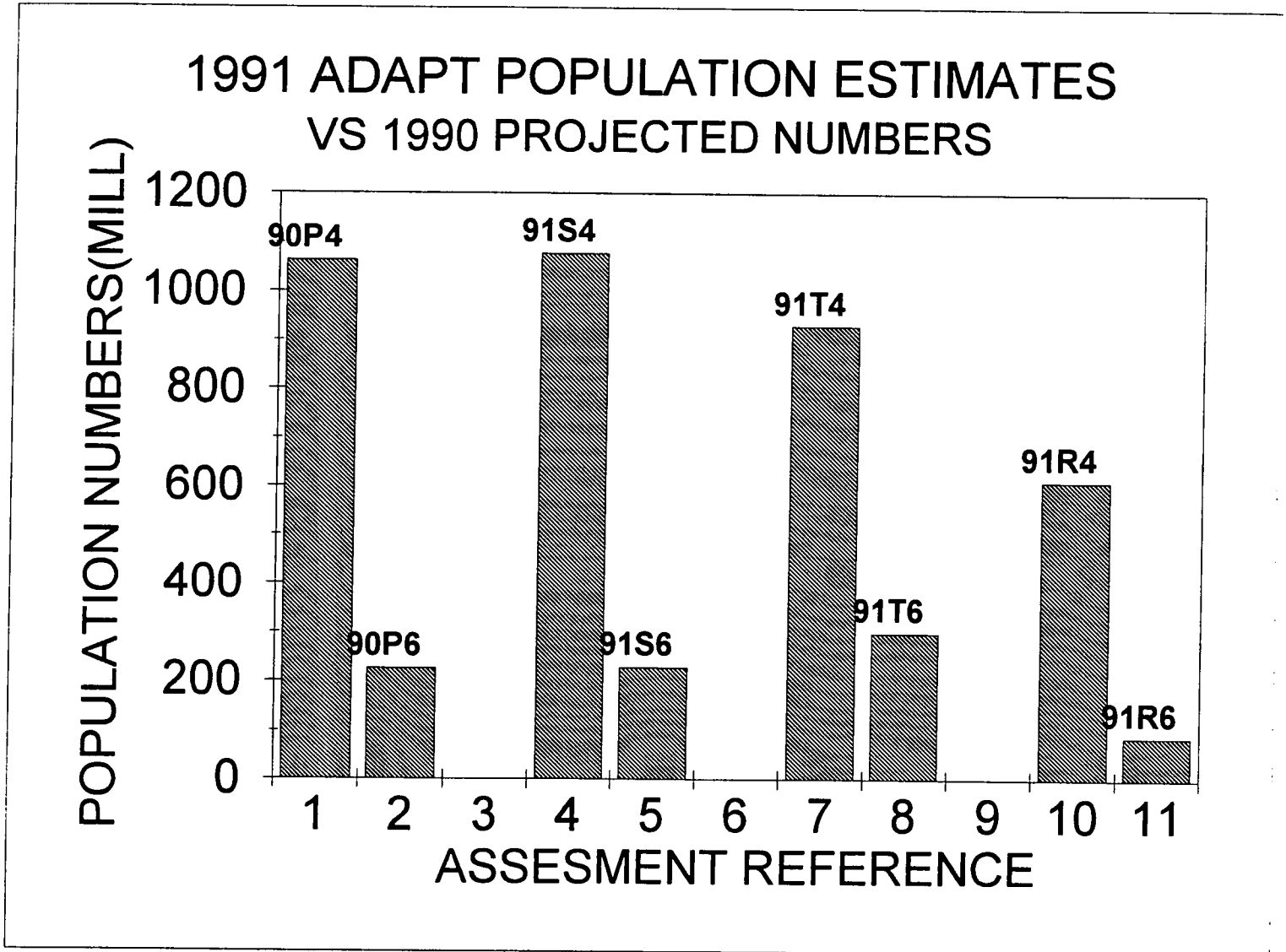


Fig. 16. Adapt 4+ and 6+ 1991 population numbers projected from the 1991 Adapt assessment of northern cod (90 P4 and 90 P6 respectively) and similar Adapt estimates for 1991 from the 1992 assessment calibrated from SPANS (91 S4 and 91 S6) and traps (91 T4 and 91 T6).

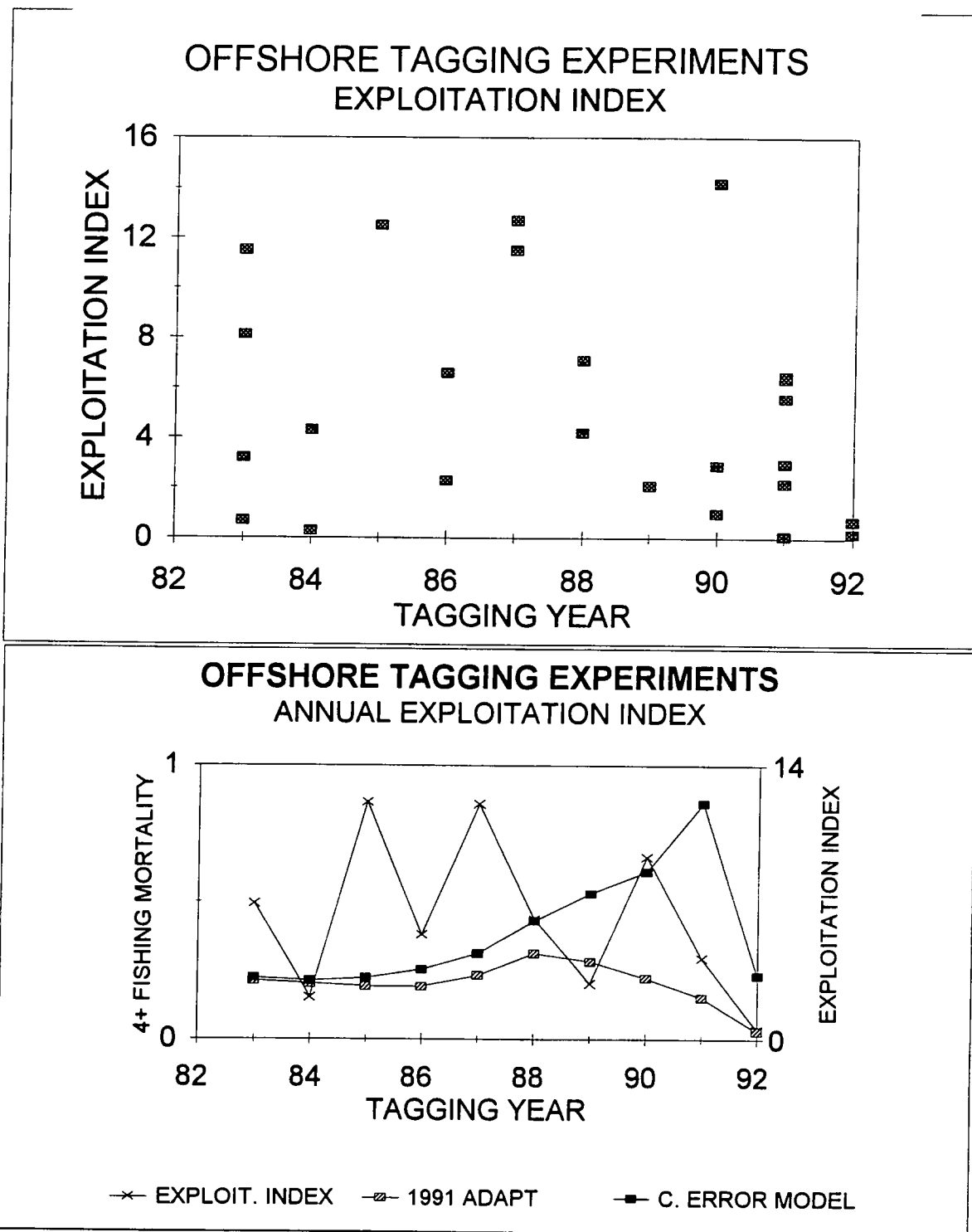


Fig. 17. First year exploitation indices of northern cod as derived from offshore tagging experiments for the period 1983-92 (top panel); the bottom panel shows annual means in comparison with age 4+ fishing mortality from the 1991 Adapt assessment and from the correlated error model.

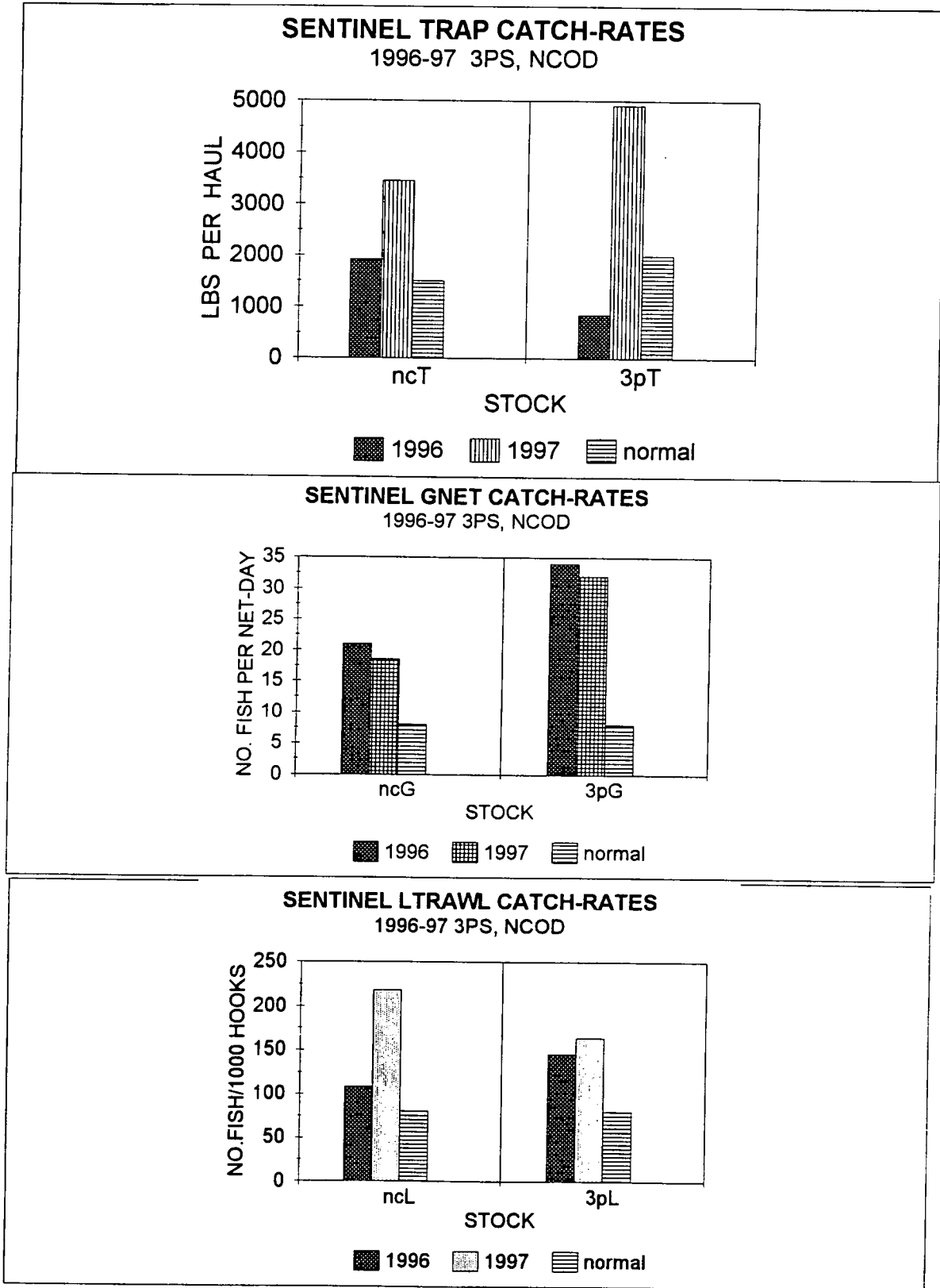


Fig. 18. Sentinel catch rates of northern cod and 3Ps cod by gear for 1996 and 1997. Mean commercial catch-rates are shown for comparison.

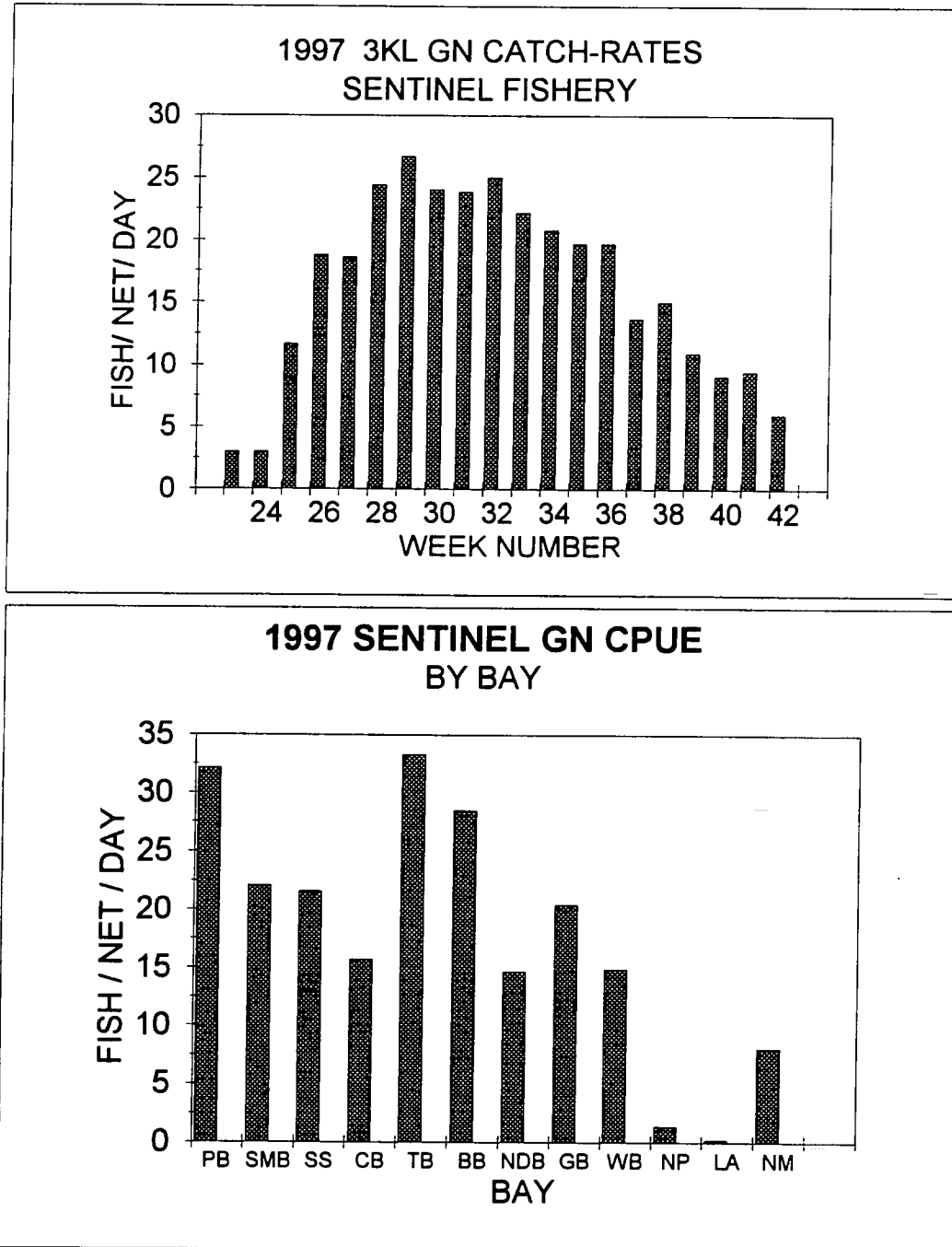


Fig. 19. Seasonal (weekly) catch-rates of sentinel "control" gillnets for 2J3KL cod for 1997 (mean of weekly means for all areas combined) (top panel); and mean catch-rates of the same "controls" by area for 1997 (lower panel).

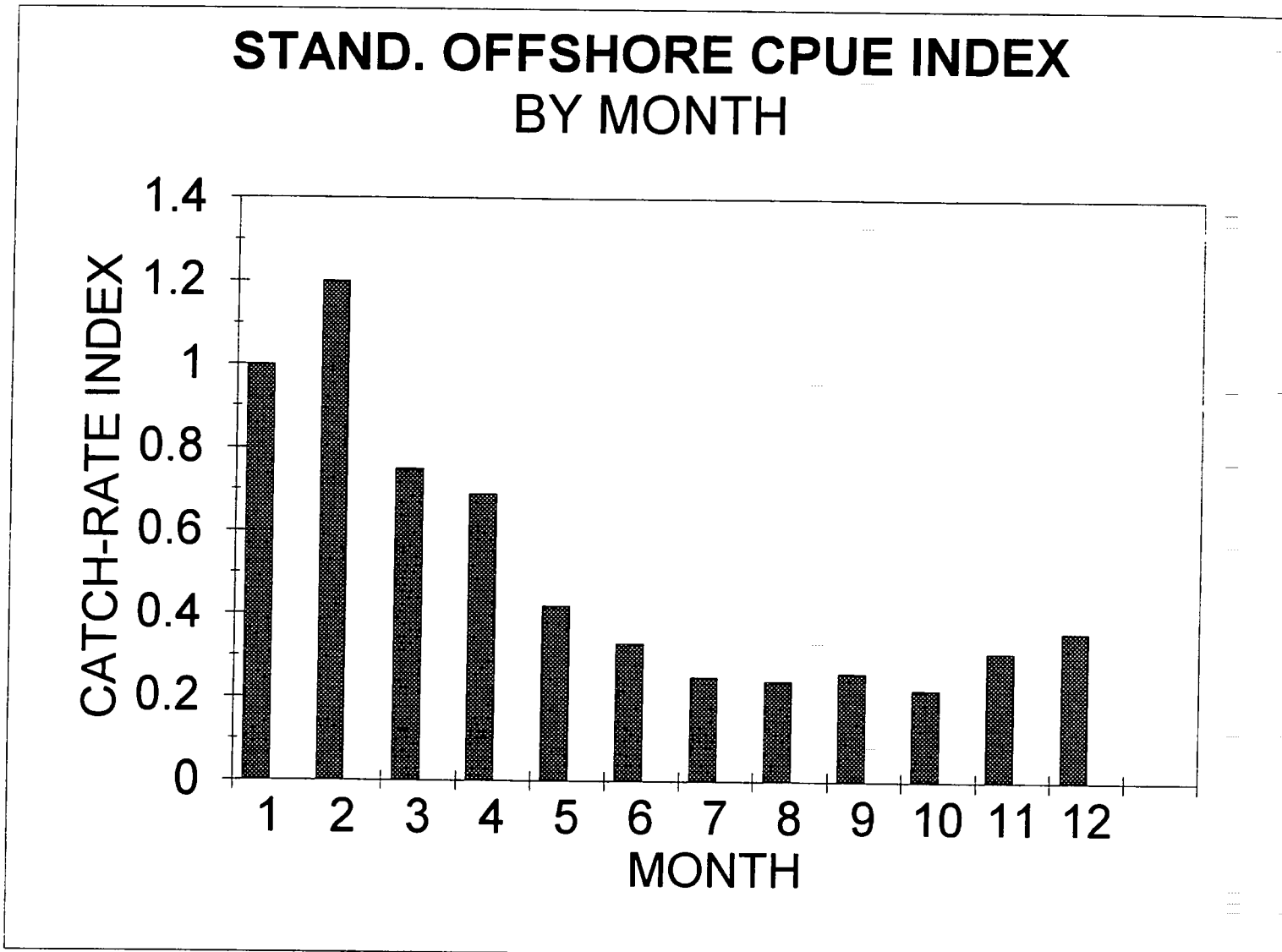


Fig. 20. Monthly fishing power coefficients of Canadian trawlers fishing for northern cod in 2J3KL for 1983-91 (from Baird et al. 1992).

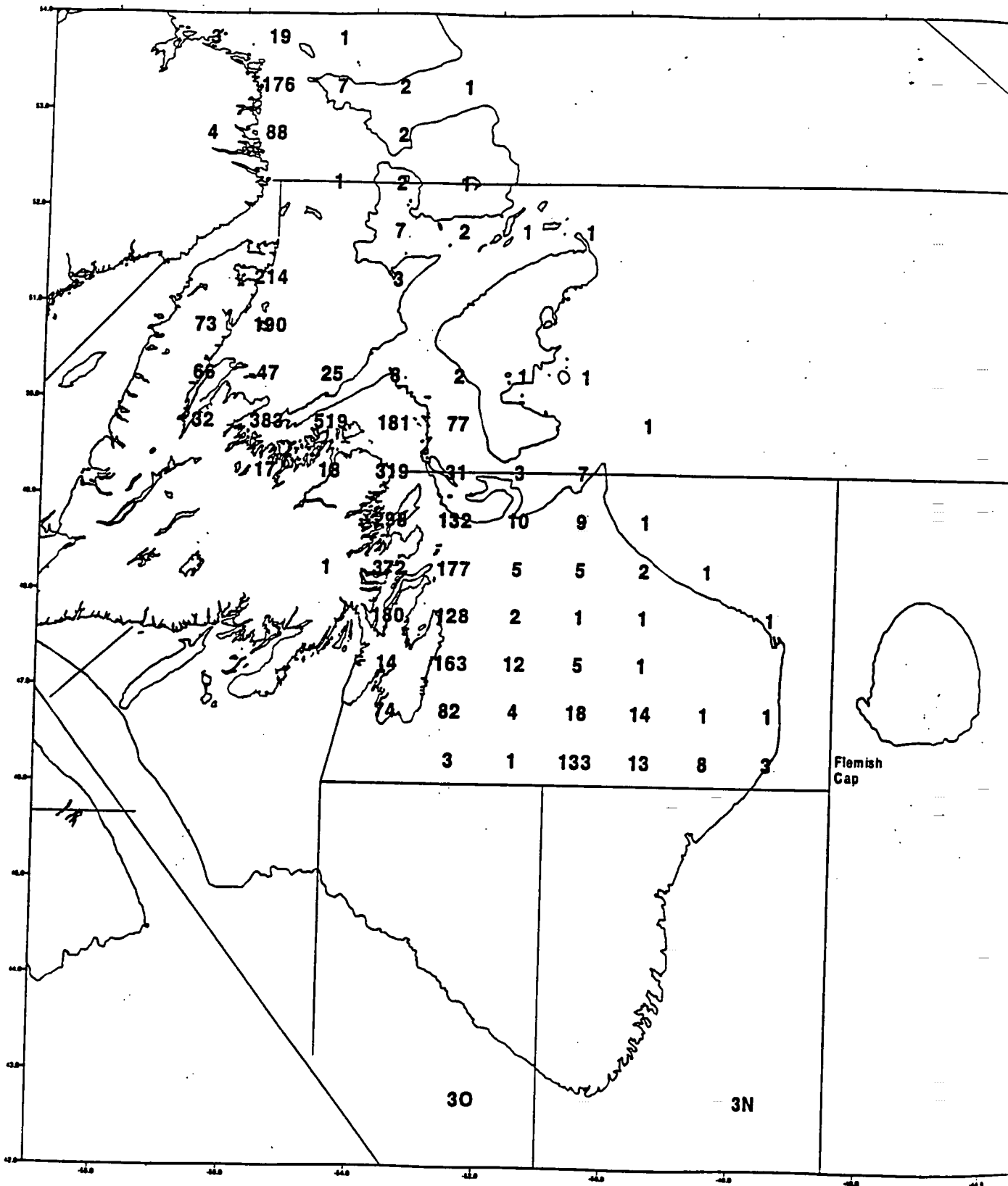


Fig. 21. Spatial distribution of tag returns of northern cod during the summer period (July-September) for 1980-90.

DEPTH DISTRIBUTION OF TAG RETURNS 1980-89 COASTAL ZONE

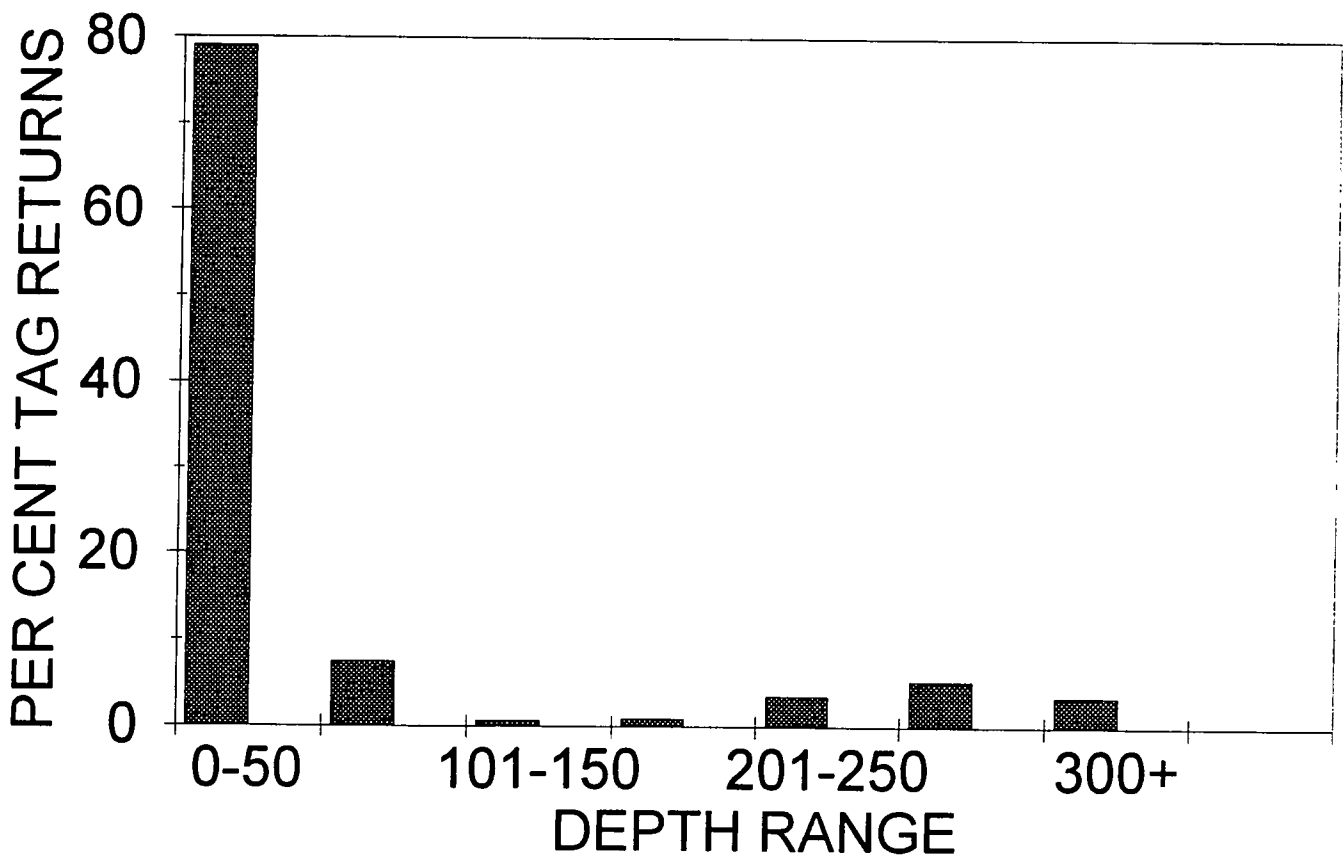


Fig. 22. Depth distribution of tag returns of northern cod from the inshore fisheries for the summer period, 1980-90.

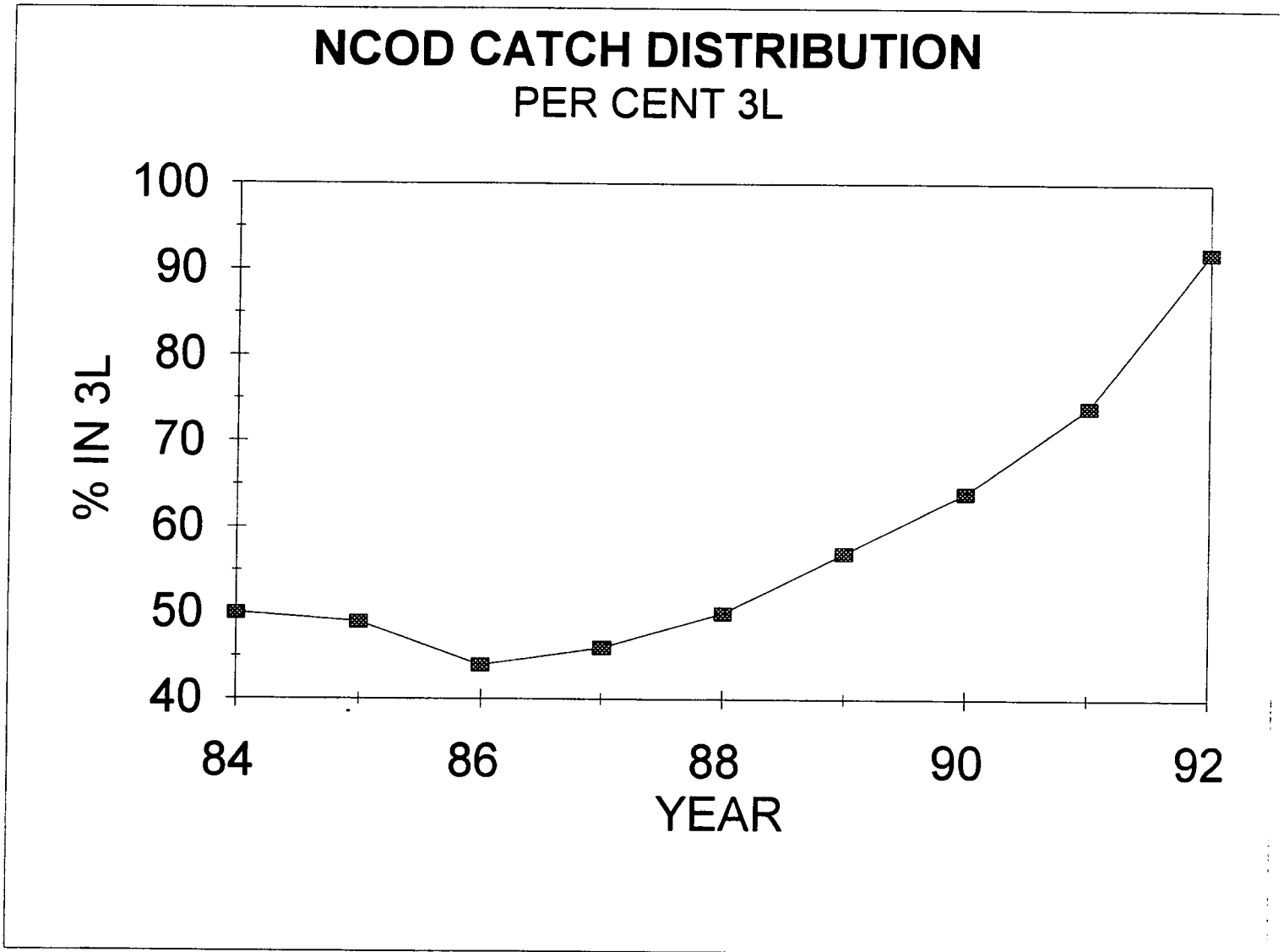


Fig. 23. Div. 3L inshore catch of northern cod as a per cent of total inshore catches.

Average CPUE (Number of Fish per Net) for 5 1/2" Gillnet
 All Fishers Combined NAFO Div: 3PS

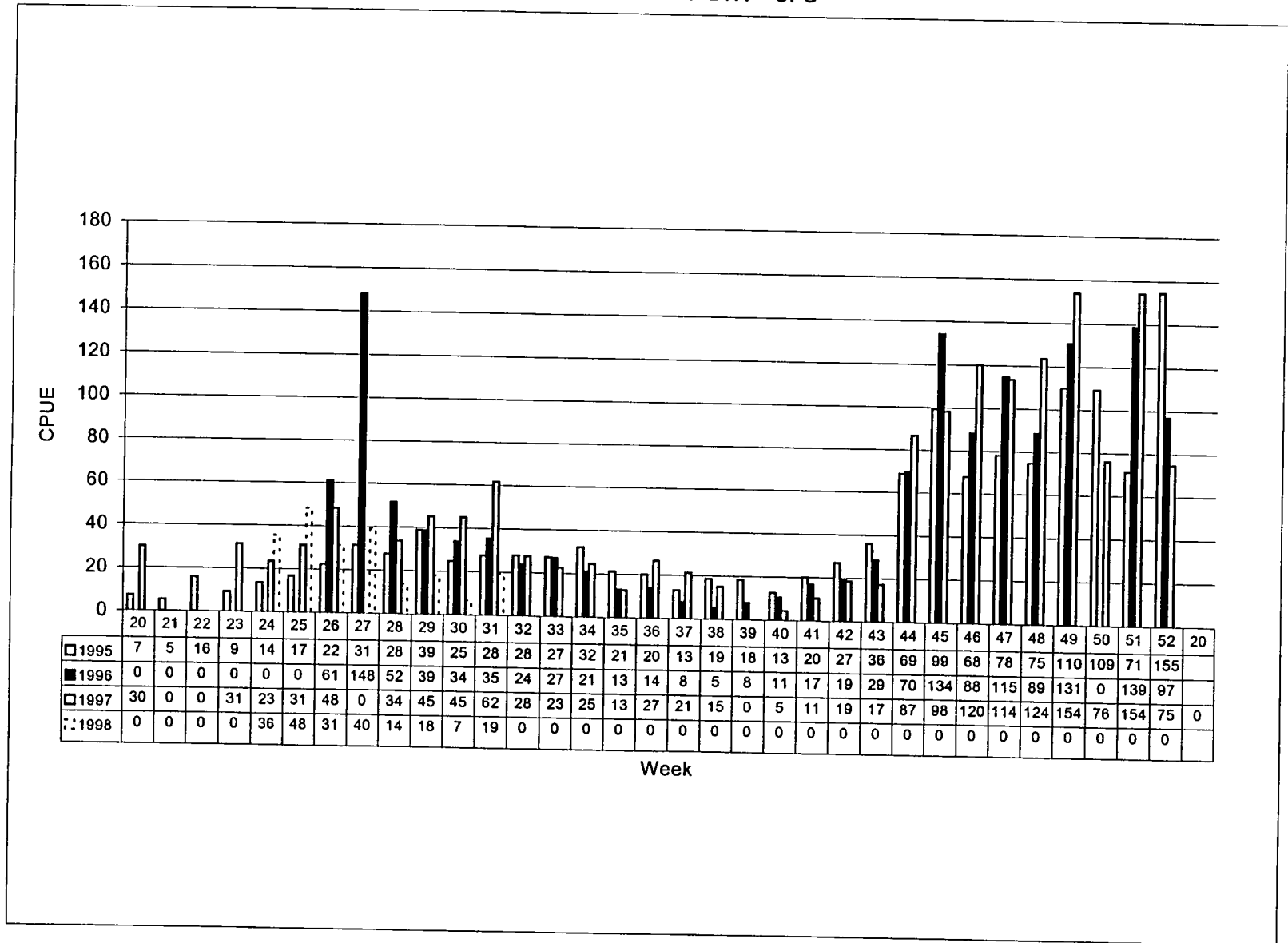


Fig. 24. Seasonal (weekly) catch-rates of sentinel gillnets in Placentia Bay for 1995-97 (from Davis 1998).

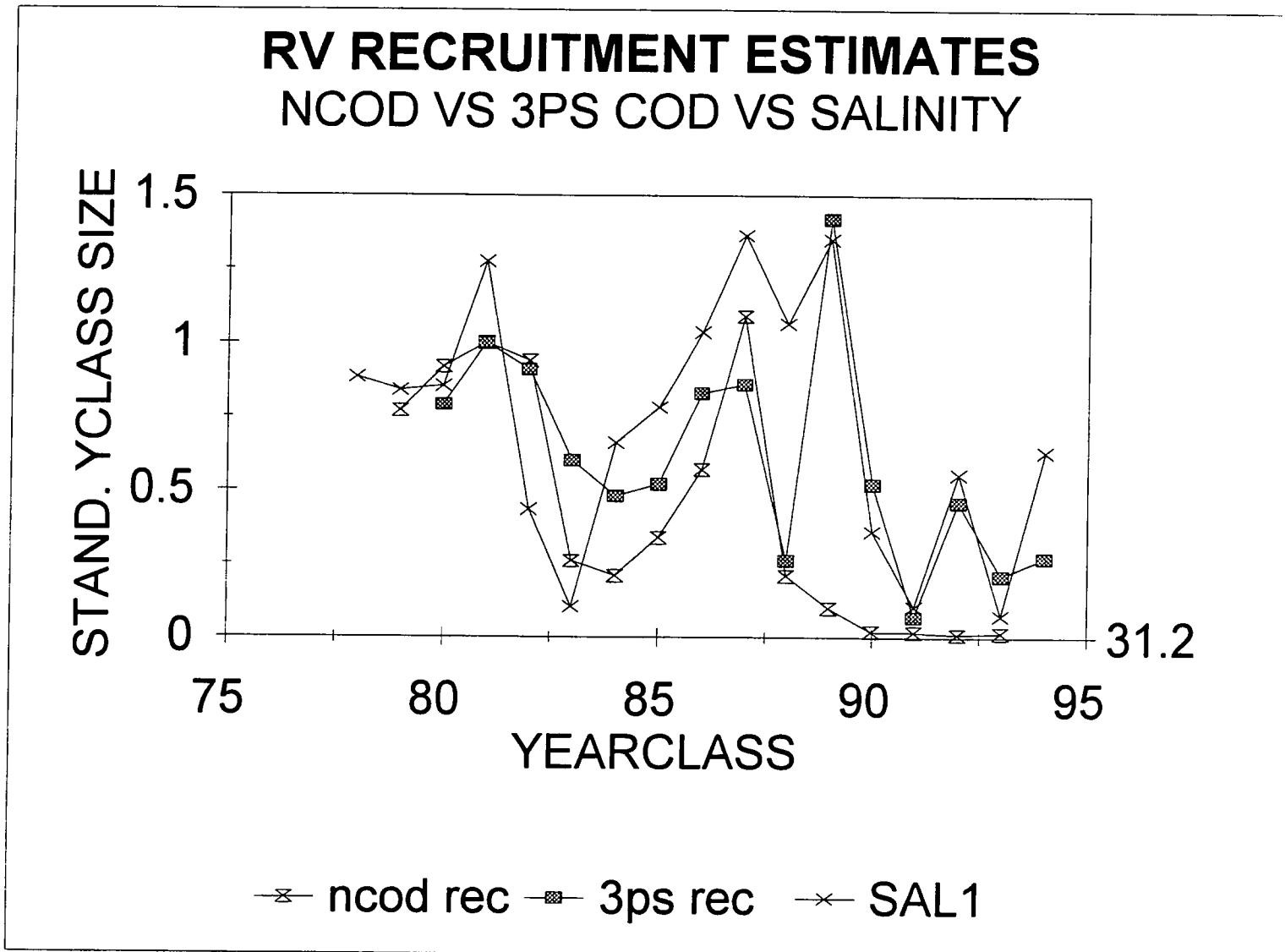


Fig. 25. Relative yearclass strength of northern cod and 3Ps cod as estimated by multiplicative models from offshore RV survey data for ages 2-4.

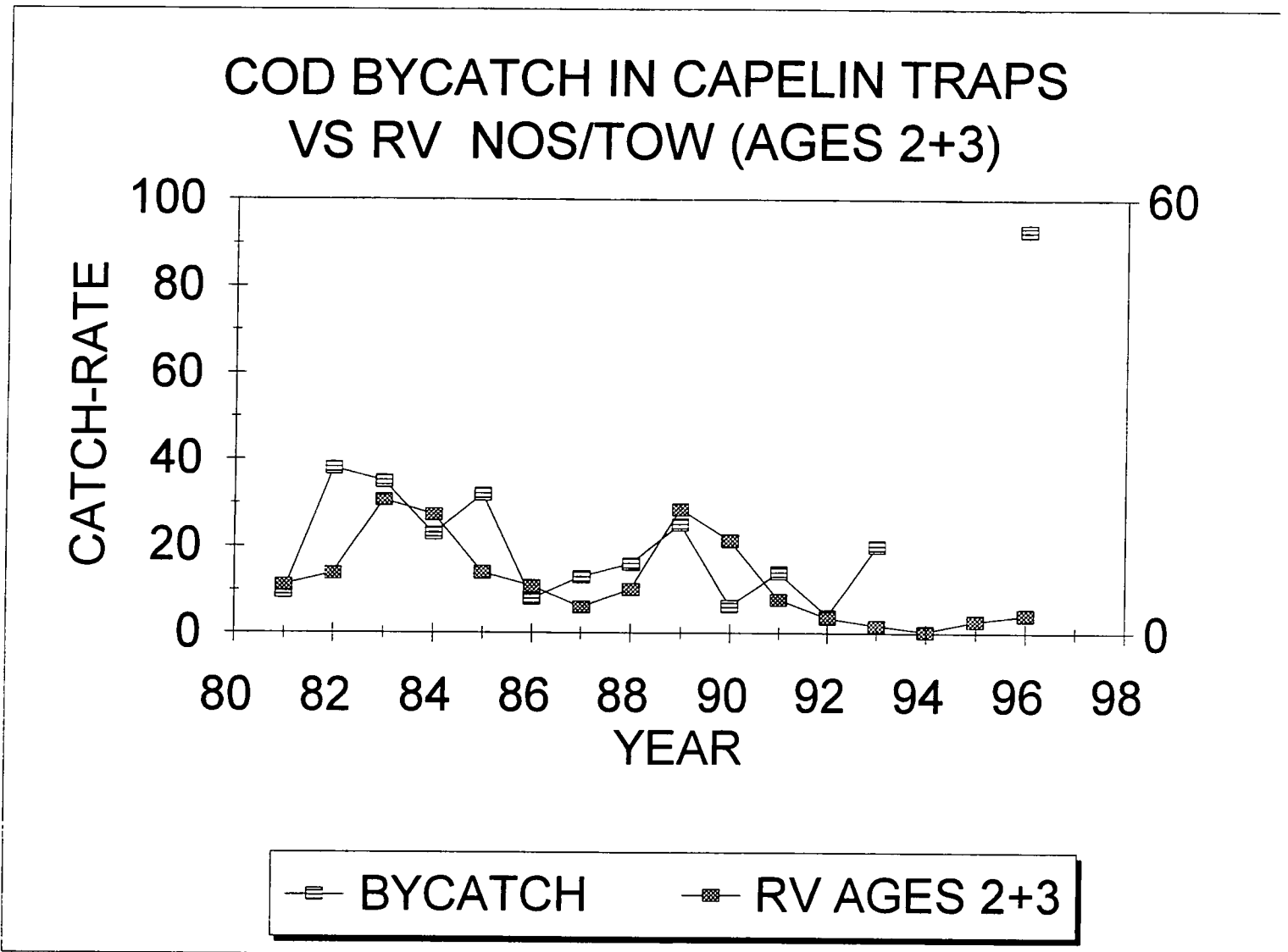


Fig. 26. Mean bycatch of cod (kg per haul) in capelin traps in Div. 3KL for the period 1981-96 in relation to mean catch-rates of age 2 + 3 cod in the fall RV survey.

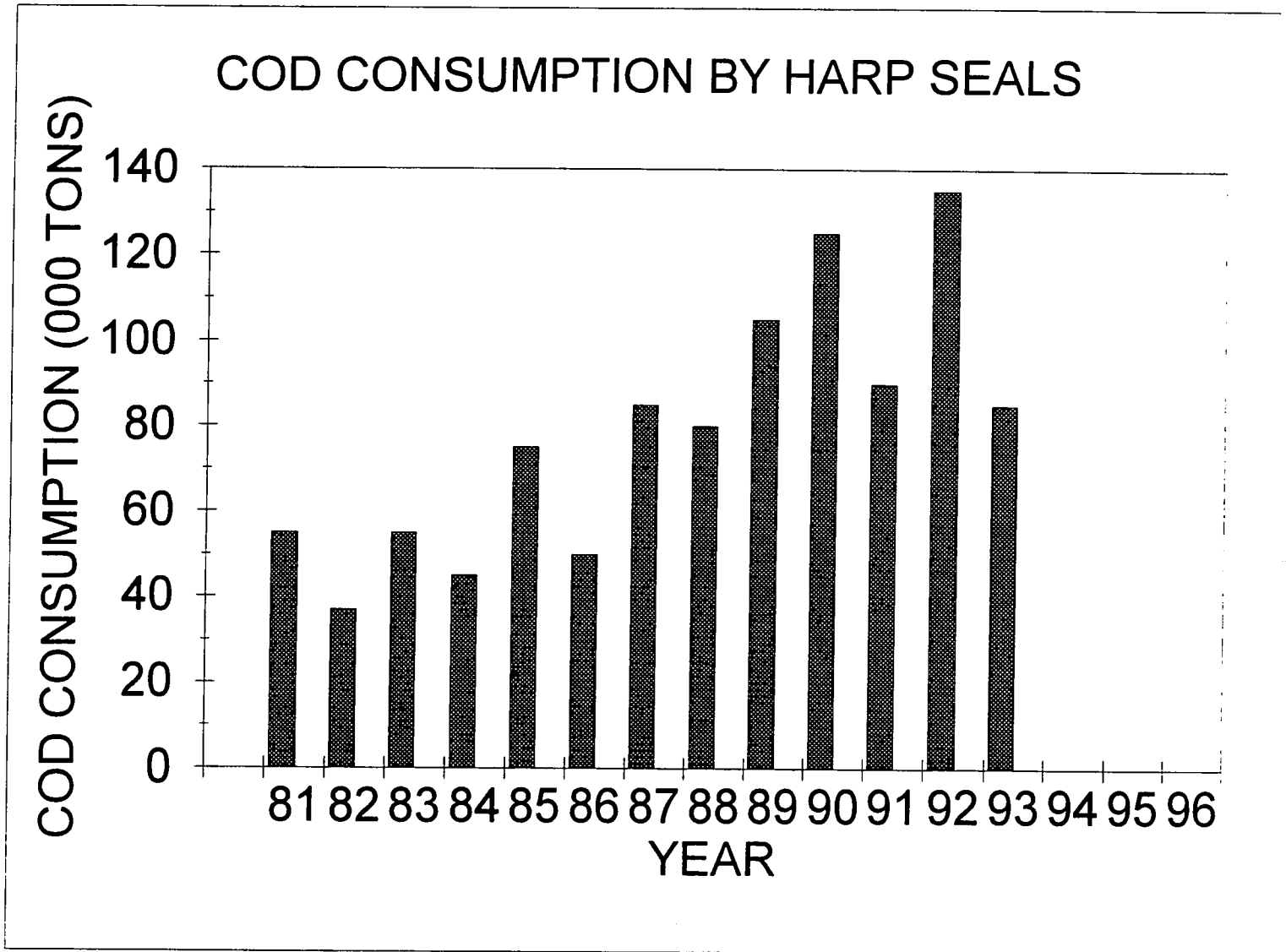


Fig. 27. Annual estimates of cod consumption by harp seals 1981-93 (data from Stenson et al. 1995).