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Status and Recovery Potential of Porbeagle Shark in the Northwest Atlantic

État et potentiel de rétablissement de la maraîche dans l'Atlantique Nord-Ouest

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ABSTRACT

Abundance of porbeagle shark (*Lamna nasus*) in the Northwest Atlantic has declined and, in 2004, the population was listed as endangered by COSEWIC with the recommendation that it be listed under Schedule 1 of Canada's *Species at Risk Act*. If listed, activities that would harm the species would be prohibited and a recovery plan would be required. Before the listing decision is made, decisions on permitting incidental harm and in support of recovery planning need to be made. In support of this process, this document was prepared to evaluate the present status of porbeagle and to assess expected time frames for recovery under different management scenarios. Towards this objective, we used a forward projecting, age- and sex-structured life history model, fit to catch-at-length and catch per unit effort (CPUE) data to the end of 2004, to evaluate porbeagle population dynamics.

Four variants of the model are presented. One of the models is fit to standardized CPUE series for mature and immature porbeagle, whereas the other three models are fit to the un-standardized CPUE expressed as weight per hook. These later three models differ in their assumed productivity. The models place the present abundance in the range of 12.2% to 24.3% its size in 1961, and female spawner abundance at about 7% to 15% of its 1961 level. Estimated female spawner abundance in 2005 ranges between 77% and 92% of the female spawner abundance in 2002. The three models using the un-standardized CPUE indicate that total abundance is roughly unchanged since 2002, with corresponding increases in the abundance of immature fish.

Reproductive rates could not be reliably estimated with the models, so model scenarios with lifetime maximum reproductive rates in the range of 1.6 to 2.6 spawners/spawner, thought to be appropriate for this population, were investigated. All models indicate that this population can recover if levels of human-induced mortality are kept low. Although recovery rates vary among models, time scales are on the order of decades. Using deterministic population projections, the integrated CPUE models predict recovery to 20% of the unfished equilibrium biomass to take about one to three decades if human-induced mortality is kept to about 2% to 4% of the vulnerable biomass. Recovery to the level that produces MSY is expected to take much longer. Presently, a 5-year management (recovery) plan is in effect to the end of 2007 with a quota of 250t. Simulations based on the most pessimistic model (lowest starting abundance) indicate that managing under this plan until the end of 2007 will only slightly delay recovery in comparison with fishery closures implemented in 2005. This strategy would provide an opportunity for uncertainties in the estimates of population size and productivity to be resolved, including recent increases in CPUE that are not well predicted by the model, and discrepancies between the model and tagging data, the later suggesting higher survival rates than those predicted by the model.

RÉSUMÉ

L’abondance de la maraîche (*Lamna nasus*) dans l’Atlantique Nord-Ouest a diminué et, en 2004, la population a été désignée espèce en voie de disparition par le COSEPAC. Il était aussi recommandé de l’inscrire sur la liste de l’annexe 1 de la *Loi sur les espèces en péril*. L’inscription de la maraîche à cette liste entraînerait l'interdiction des activités qui lui seraient nuisibles et l'adoption d'un plan de rétablissement. Toutefois, avant que cette mesure ne soit prise, des décisions sont requises au sujet de l’autorisation des activités qui lui seraient nuisibles ainsi que de la planification du rétablissement. Dans le cadre de ce processus, le présent document a été préparé en vue d’évaluer l’état actuel de la maraîche et de déterminer le temps prévu du rétablissement selon différents scenarios de gestion. À cette fin, nous avons eu recours à un modèle de projection prospective du cycle biologique structuré selon l’âge et le sexe, qui utilisait les données sur les captures selon la longueur et les captures par unité d’effort (CPUE) jusqu’à la fin de 2004, de façon à évaluer la dynamique de la population de maraîche.

Quatre variantes du modèle sont présentées. Un des modèles utilise les séries chronologiques des CPUE standardisées pour les maraîches matures et immatures, tandis que les trois autres utilisent des CPUE non standardisées, exprimées en poids par hameçon. Ces trois derniers modèles sont fondés sur des scénarios de productivité hypothétique différents. Les modèles indiquent que la population est d’environ 12,2 % à 24,3 % de son effectif de 1961 et que l’abondance des femelles reproductrices a diminué, pour se situer à environ 7 à 15 % de son niveau de 1961. L’abondance des femelles reproductrices, en 2005, est d’à peu près 77 % à 92 % de celle de 2002. Les trois modèles qui utilisent les CPUE non standardisées révèlent que l’abondance totale est à peu près inchangée depuis 2002, avec des augmentations correspondantes de l’abondance des poissons immatures.

Les modèles n’ayant pas permis d’estimer les taux de reproduction de manière fiable, on a évalué des scénarios de productivité maximale de l’ordre de 1,6 à 2,6 géniteurs/géniteur, jugés appropriés pour cette population. Tous les modèles indiquent que la population peut se rétablir si les taux de mortalité de source anthropique sont faibles. Bien que le taux de rétablissement varie selon les modèles, le temps requis se mesure en décennies. À l’aide de projections déterministes de la population, les modèles de CPUE intégrées permettent d’estimer qu’il faudra de dix à trente ans, en supposant que la mortalité de source anthropique est maintenue aux environs de 2 % à 4 % de la biomasse vulnérable, pour obtenir un rétablissement à 20 % de la biomasse d’équilibre non exploitée. Le rétablissement à un niveau qui assure un PME devrait prendre beaucoup plus longtemps. Actuellement, un plan de gestion (de rétablissement) de cinq ans, en vigueur jusqu’à la fin de 2007, comprend un quota de 250 t. Des simulations basées sur le modèle le plus pessimiste (la plus faible abondance de départ) indique que la gestion conformément à ce plan d’ici la fin de 2007 ne retardera que légèrement le rétablissement par rapport aux fermetures de la pêche mises en œuvre en 2005. Cette stratégie donnerait la possibilité d’éliminer les incertitudes que contiennent les estimations de la population et de la productivité, notamment les augmentations récentes des CPUE qui ne sont pas bien prévues par le modèle, et les écarts entre le modèle et les données obtenues par marquage qui semble indiquer que le taux de survie est plus élevé que celui qui est prévu par le modèle.

INTRODUCTION

The porbeagle shark (*Lamna nasus*) is a large cold-temperate pelagic shark species of the family Lamnidae that occurs in the North Atlantic, South Atlantic and South Pacific oceans. The species range extends from Newfoundland to New Jersey and possibly to South Carolina in the west Atlantic, and from Iceland and the western Barents Sea to Morocco and the Mediterranean in the east Atlantic. It is the only large shark species for which a directed commercial fishery exists in Canadian coastal waters.

Fisheries management plans for pelagic sharks in Atlantic Canada established non-restrictive catch guidelines of 1500t for porbeagle prior to 1997 (O'Boyle et al. 1996). Because of the limited scientific information that was available at the time, abundance, mortality and yield calculations could not be made. Therefore, a provisional total allowable catch (TAC) of 1000t was set in place for the period 1997-1999, based largely on historic catches and the observation that recent catch rates had declined (O'Boyle et al. 1998).

In 1998, an intensive research program on all aspects of porbeagle biology and population dynamics was initiated at the Bedford Institute of Oceanography, Dartmouth, Nova Scotia. The research was carried out with the support and funding of the porbeagle shark fishing industry, and in collaboration with the Apex Predators Program of the U.S. National Marine Fisheries Service (NMFS). The combination of the BIO program, the industry support, and the NMFS collaboration greatly increased our understanding of porbeagle biology and population dynamics (Campana et al. 2002a,b, 2003; Jensen et al. 2002; Joyce et al. 2002; Natanson et al. 2002), and led to two analytical stock assessments of porbeagle (Campana et al. 1999, 2001). Based on those assessments, the Shark Management Plan for 2002-2006 reduced the TAC to 250t, a value that was thought to correspond with F_{msy} and was expected to allow for stock recovery.

In May 2004, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) designated the porbeagle as an endangered species, and recommended that it be listed under Schedule 1 of Canada's Species at Risk Act (SARA). If listed, activities that would harm the species would be prohibited and a recovery plan would be required. Before the listing decision is made, decisions on permitting incidental harm and in support of recovery planning need to be made.

The present document provides an up-to-date summary of current population status and recovery potential for porbeagle shark. The basis for the document is a statistical analysis of available data to the end of 2004 using a life history based, age-structured population model. The population model is used to evaluate both current population status and trends, potential recovery trajectories given various management options (specifically levels of human-induced mortality), and time frames for recovery. The document also identifies sources of human-induced mortality for which incidental harm permits could be considered. The approach used in this document is based on the framework for assessing incidental harm permits outlined in DFO (2004a).

Life History

Porbeagle sharks have low fecundity and late ages at sexual maturation. Age at maturity is about eight years in males and about thirteen years in females (Natanson et al. 2002). In the northwest Atlantic, mating occurs from September through November, and live birth occurs eight to nine months later (Jensen et al. 2002). Reproduction is thought to occur annually. Jensen et al. (2002) reports an average litter size of four young (range two to six). The life span of porbeagle is estimated to be between 25 and 46 years (Campana et al. 2002a; Natanson et al. 2002) and generation time is about 18 years (Campana et al. 2001). Porbeagle are thought to have a low natural mortality. Instantaneous natural mortality is estimated to be 0.10 for immature porbeagle, 0.15 for mature males, and 0.20 for mature females (Campana et al. 2001). Although these estimates are conditional on the gear selectivity assumed in their calculation, they are presently the best available for this population.

METHODS

The analyses presented in this document are adapted from those presented in Campana et al. (2001) and descriptions of the fisheries, management history and data are available therein.

Data

Data used in following analyses are those of Campana et al. (2001), updated to 2004.

Commercial landings

The commercial landings used here are the combined reported landings (all countries) for the northwest Atlantic (NAFO areas 3-6) from 1961 to 2004 (Table 1). All foreign data came from the Scotia-Fundy Observer Program (SFOP) or Newfoundland Observer Program (NFOP) and are thus considered accurate. All Canadian data came from pelagic longline logbook data cross-matched to landings. For the period from 1991 to present, these data are also considered to be relatively accurate, especially after 1996.

Landings rose from about 1,900t in 1961 to over 9,000t in 1964 and then fell to less than 1,000t in 1970 as a result of collapse of the fishery (Figure 1). Reported landings remained less than 500t until 1989, and then increased to a high of about 2000t in 1992. Landings since 1998 have been restricted by quota, and since 2001 have been in the range of 139t to 499t (218t in 2004). Most of the landings are from the directed porbeagle fishery (Table 2), although with recent quota reductions, the percent landings as bycatch has increased. Reported landings of porbeagle in fisheries outside the Scotia-Fundy region are lower and have been under 20t in three of the last five years (Table 3).

For this analysis the catch was apportioned to three areas: NF-Gulf = Gulf of St. Lawrence, area north of Laurentian Channel, plus NAFO Division 4Vn; Basin = Basins and inshore regions of Scotian Shelf; and the Shelf edge = area over and around the edge of the Scotian Shelf, plus the Gulf of Maine. Assignment of the catch to an area was based on the location of the reported catch for the years 1989 to 2004, and using the 1988 to 2002 averages for years prior to 1988 (Table 4). Nearly all directed landings in 2003 and 2004 were from the Basin and Shelf edge areas (Table 4).

Commercial catch rates (CPUE)

Catch-per-unit-effort (CPUE) is used as the primary index of abundance in this analysis. Calculations of porbeagle CPUE were based on porbeagle-directed longline catches, which account for virtually all historical catches. Initial examination of the catch rate data indicated that the major data sources could be categorized by country (Canada, Faroes), vessel CFV, season, and area fished.

Campana et al. (2001) disaggregated CPUE into those for immature and those for mature sharks; both calculated in terms of ln-transformed numbers per hook. A fork length equal to 200 cm is approximately midway between the lengths corresponding to 50% maturity in males and females, and is therefore a proxy for sexually mature porbeagles (Jensen et al. 2002). To calculate catch rate at length, the length composition was determined for each of the three subareas in each of three seasons (Jan-Mar, Apr-June, July-Dec) based on available measurements each year. Set-by-set catch rates in terms of weight were converted to numbers based on the mean weight of the length composition of the subarea-season-year cell, then apportioned according to the length frequency. Numbers above 200 cm FL were pooled within a set to form the index for mature sharks, while the remainder were pooled to form the index for immature sharks. Here, we consider both the standardized data and the un-standardized catch (weight) per hook as indices of abundance.

Box plots summarizing the three CPUE data sets are shown in Figures 2 to 4. Both the CPUE by weight data show little or no apparent trend (Figure 2), whereas the immature CPUE data show an increasing trend in recent years (Figure 3). A declining trend is evident in the mature CPUE data in all three areas (Figure 4).

CPUE time series are often standardized to correct for differences in the timing and gear used in the fishery (Maunder and Punt 2004) prior to being included in the assessment model. Alternatively, the standardization may be integrated into the assessment model, a method that has been shown to provide greater precision in biomass estimates than when the standardization is done prior to fitting the assessment model (Maunder 2001).

At least two issues exist with these CPUE data when deriving an index of abundance. First, the spatial distribution of the fishing effort has decreased markedly in the last few years (Figures 5.1 to 5.10). Coincidental with this change has been an increase in CPUE in 2003 and 2004 in the smaller area presently being fished, indicating either increased abundance, increased efficiency, a change in methods, or a change in the distribution of porbeagle in recent years. Second, there is little overlap in the vessels that took part in the fishery in the late 1980’s and 1990’s and those presently fishing (Tables 5.1 to 5.3). This issue creates difficulties separating year effects (changes in abundance) from vessel effects (changes in the fleet), and not all vessels fish with the same efficiency (Figure 6). Differences in catchability also exist among seasons (Figure 7).

To address these issues, we used two approaches to incorporate the CPUE data:

> 1) Following the approach of Campana et al. (2001), we developed a standardized CPUE time series by fitting a generalized linear model (GLM) with a gamma error distribution to the CPUE data at the set-by-set level. Year, season and vessel (CFV) were used as factors in the model and all interaction terms were included. Separate models were fit to the data from each subarea and for each mature or immature data series, for a total of six models. Models with CFV tended to outperform models using country (but not CFV) as a factor. For both countries, only vessels which fished more than 10 directed sets were included in the model. All terms were statistically significant. The estimated marginal means for Year for each subarea were used for fitting the population model. The resulting standardized CPUE time series are shown in Figure 8.
>
> 2) We integrated the CPUE by weight standardization into the assessment model. In this case the raw CPUE data were not standardized but were used directly in the population model. We fit several models (Table 6), starting with a simple model with a single catchability coefficient for all vessels in all areas in all seasons, then adding coefficients for area, CFV and season, and adding coefficients for combinations of these variables, in a stepwise fashion. This analysis was done with two weightings of the catch-at-length data (by changing the assumed sample size). Based on the Akaike Information Criterion, a model with separate catchability coefficients each vessel, in each area and in each season (each vessel, area and season combination is used as a separate index of abundance) was the best model and was retained for the analyses herein. Only vessels that fished in a season and area in three or more years were included in the analysis.

Catch at length

Campana et al. (1999, 2001) describe the porbeagle length data set and standardizations. Over 144,000 length measurements are available for known sex porbeagle, and more are available when sharks of unknown sex are included. To estimate the proportion of the catch by length, we assigned porbeagle to five cm length categories ranging from 65 to 285 cm total length. When fitting the model, we used sex specific data for the years: 1995 and 1998 - 2004 for the Basin region; 1988, 1989 to 1996, 1998 to 2000, and 2002 for the NF-Gulf region; and 1961, 1981, and 1990 to 2004 for the Shelf-Edge region. Observed proportions at length and sample sizes are shown in the Results section (Figures 13.1 to 13.8.

Tagging data

Descriptions of the porbeagle tagging programs are provided by Campana et al. (1999). Following Campana et al. (2001) we included only sharks less than 125 cm fork length at the time of tagging and assumed these sharks were either age-0 or age-1. Between 1980 and 1999, a total of 1083 porbeagle sharks in this size category were tagged, resulting in 121 recaptures (Table 7).

Population Modelling

The model used is that of Harley (2002), which we further adapted for this analysis. This model is the forward-projecting age- and sex-structured population dynamics model presented in Campana et al. (2001) and Harley (2002). Within this model, the population is projected forward from an equilibrium starting abundance and age distribution by adding recruitment and removing catches. A key assumption in the model is that the porbeagle population was at an unfished equilibrium at the beginning of 1961, when the directed commercial fisheries for porbeagle began. Model parameter estimates (e.g. selectivity parameters and catchability coefficients) are obtained by fitting the model to the available datasets using maximum likelihood. Two fisheries were included in the model of Harley (2002): the Scotian Shelf and Newfoundland-Gulf fisheries. The spawner-recruit (SR) function (a Beverton-Holt function) was parameterized using the unfished equilibrium recruitment (R_0) and steepness (h). For the model runs in Campana et al. (2001), steepness was calculated outside the model on the basis of the "well-defined reproductive parameters of porbeagle".

Adaptations of this model for the current study (described further below) include:

- Further splitting the shelf fishery into an inshore (Basin) and offshore (Shelf-Edge) fishery based on the idea that their selectivities are different (see Campana et al. (1999): Figures 9 to 11). The NF-Gulf fishery is retained.
- Integration of the CPUE analysis into the assessment model.
- Addition of a model component to include the tagging data.
- Addition of a model component to estimate reference points.
- Addition of a population viability analysis (PVA) to evaluate recovery trajectories.
- Reformulation of the SR model such that the parameters are the maximum rate at which female spawners produce age-1 recruits (α) and the asymptotic recruitment level (R_{asy}), with both parameters estimated within the model. The difference in this model and the runs in Campana et al. (2001) is that survival from birth to age-1 is not assumed to be known.

Population dynamics

Of primary interest is the number of fish in year t, of sex s and of age a, denoted $N_{t,s,a}$. The number of fish in each age class in the next year is given by an exponential decay model. Here, the total mortality rate is the sum of the sex and age specific instantaneous natural mortality rate ($M_{s,a}$) and the fishery (g) specific exploitation rate in each year, sex and age class ($u_{t,s,a}^{g}$):

$$N_{t+1,s,a+1} = N_{t,s,a} e^{-M_{s,a}} \prod_{g} (1 - u_{t,s,a}^{g}) .$$

Litter size is not thought to vary with age in porbeagle, so the spawner-recruit relationship is expressed in terms of the number of females rather than biomass. Using the letter F to denote the female sex category, the number of female spawners in year t (SSN_t) is a function of $N_{t,\mathrm{F},a}$ and the probability that a female fish of age a is mature at that age ($m_{\mathrm{F},a}$):

$$SSN_t = \sum_a N_{t,\mathrm{F},a} m_{\mathrm{F},a}$$

The life cycle is closed by modeling the number of age-1 fish of each sex in the year t+1 as a function of SSN_t using a Beverton-Holt spawner-recruit (Hilborn and Walters 1992) relationship:

$$N_{t+1,s,1} = \frac{\alpha SSN_t}{1+\dfrac{\alpha SSN_t}{R_{asy}}} e^{(\varepsilon_t - \sigma^2/2)} * 0.5$$

Here, α is the slope at the origin, and in the deterministic model is the maximum rate at which female spawners can produce age-1 recruits at low population sizes (Myers et al. 1999), and R_{asy} is the asymptotic recruitment level (expressed as the number of age-1 recruits). R_{asy} is the limit approached by R_t as S_t approaches infinity (Beverton-Holt models are often written in terms of the half saturation constant, K, which is related to R_{asy} by: $R_{asy} = \alpha K$). A 1:1 sex ratio at birth is assumed. Recruitment can vary around the fitted relationship though the log of a recruitment deviate for each year (ε_t), in which case a correction for transformation bias based on the standard deviation of the log recruitment deviate (σ) is applied to each deviate. As written, a lognormal error structure for recruitment (Myers et al. 1995) is assumed. In comparison with the other commonly used 2-parameter SR model, the Ricker model, the Beverton-Holt model has the advantage that R_{asy} can be rescaled and interpreted as an estimate of carrying capacity (Gibson and Myers 2003a, Myers et al. 2001), but is not a precautionary model selection because it typically provides estimates of α (and its related reference points) that are higher than those from the Ricker model (Gibson and Myers 2003b, Myers et al. 1999). Reference points provided herein are therefore not precautionary with respect to SR model selection.

Commercial fisheries

The commercial fisheries are included in the population dynamics through $u^g_{t,s,a}$. This term consists of two separable components: the gear (or fishery) and sex specific selectivity of the commercial fisheries $s^{g,s}_a$ and the exploitation rate of the fully exploited age class by each gear in each year, u^g_t :

$$u^g_{t,s,a} = s^{g,s}_a u^g_t$$

Selectivity was assumed to follow a double half Gaussian selectivity curve:

$$s_a^{g,s} = \left\{ \begin{array}{ll} \exp\left(\dfrac{-(a - s_{full}^{g,s})^2}{v_L^{g,s}} \right) & \text{if} \quad a \le s_{full}^{g,s} \\ \exp\left(\dfrac{-(a - s_{full}^{g,s})^2}{v_R^{g,s}} \right) & \text{if} \quad a \le s_{full}^{g,s} \end{array} \right\}$$

where *g* refers to the commercial fishery (Basin, Shelf-Edge or NF-Gulf). In this model, the age at which fish are fully selected by the fishery is denoted $s_{full}^{g,s}$. The steepness of the decline away from the age at full selectivity is governed by the *v* parameters for the left and right sides of the curve.

The fishery operates throughout much of the year, but for simplicity, we assume the catch is taken during a short time period half way through the year, an approximation attributed to Pope (Quinn & Deriso 1999). We assume that the total catch in each year by each fishery (C_t^g) is known without error. The exploitation rate (proportion of the vulnerable biomass removed) in each fishery in each year is then:

$$u_t^g = \frac{C_t^g}{\sum_{s,a} e^{-0.5 M_{s,a}} s_a^{g,s} N_{t,s,a} w_{s,a}}$$

where $w_{s,a}$ is the sex specific weight-at-age.

Initial conditions

We assume that the population is at an unfished equilibrium population and age structure at the start of the time period (1961). The calculation of the equilibrium population size is provided in the reference point section below.

Predictions from the model

Parameter estimates are obtained from the model by minimizing the discrepancies between the observed data and predictions from the model. Specifically, we want to obtain predictions of the annual CPUE of mature ($I_{t,large}^g$) and immature ($I_{t,small}^g$) sharks in the three fisheries, the predicted length composition of the catch in the three fisheries and the predicted number of tagged recaptures for each year.

Under the assumption that CPUE is proportional to abundance, the predicted CPUE's of mature and immature sharks are:

$$I_{t,immature}^g = q^g \sum_{s,a} e^{-0.5 M_{s,a}} s_{s,a}^g N_{t,s,a} \quad \text{for}\, a_f < 11 \text{ and } a_m < 12$$

and

$$I^{g}_{t,mature} = q^{g} \sum_{s,a} e^{-0.5M_{s,a}} s^{g}_{s,a} N_{t,s,a} \quad \text{for } a_f \geq 11 \text{ and } a_m \geq 12 .$$

Note that the ages do not correspond directly with the ages of 50% maturity. The mean length at maturity for male and female porbeagle is roughly 200 cm and the split in the data is on this basis. The ages above correspond to these lengths. These equations are modified when CPUE by weight is used by adding weight at age to the right-hand sum and by increasing the number of *q*'s (one for each boat in each area and each season).

Following Harley (2002), the sex specific predicted length composition in the catch $P^{g,s}_{t,l}$ is a function of the population age composition, the selectivity curves and the distributions of length at age:

$$P^{g,s}_{t,l} = \frac{\sum_{a} s^{g,s}_{a} f^{s}_{l|a} N_{t,s,a}}{\sum_{a} s^{g,s}_{a} \sum_{l} f^{s}_{l|a} N_{t,s,a}} .$$

Here, the sex-specific length proportion-at- age ($f^{s}_{l|a}$) is given by:

$$f^{s}_{l|a}(l^{s}_{a}, \sigma^{s}_{a}) = \frac{\delta}{\sqrt{2\pi}\sigma^{s}_{a}} \exp\left[\frac{-\left(x^{s}_{l} - l^{s}_{a}\right)^{2}}{2\left(\sigma^{s}_{a}\right)^{2}} \right]$$

where δ is the size of the length increment (5 cm in this analysis). Here we used the same growth analysis used in Campana et al. (2001) and Harley (2002), assuming a von Bertalanffy growth model to model the relationship between length and age, as well as a linear relationship between σ^{s}_{a} and l_a. Constants are provided in Table 8.

The tagged population

We assumed that the dynamics of the tagged population were identical to the untagged population. Denoting the number of tagged fish of age *a* that are alive in year *t* as $N^{\mathrm{T}}_{t,a}$, the number of tagged fish in the following year is:

$$N^{\mathrm{T}}_{t+1,a+1} = N^{\mathrm{T}}_{t,a} e^{-M_a} (1 - u_{t,a}) + R^{\mathrm{T}}_{t+1,a+1} (1 - k) ,$$

where $R^{\mathrm{T}}_{t,a}$ is the number of tagged fish of age *a* released in year *t* and *k* is the rate of tag loss or mortality associated with tagging assumed to occur shortly after tagging. The expected number of recaptures $T_{t,a}$ is then:

$$T_{t,a} = \xi N^{\mathrm{T}}_{t,a} e^{-0.5M} u_{t,a} .$$

Here, $u_{t,a}$ is the mean of the rates for the fisheries in the three regions and ζ is the reporting rate. Reporting rates of 0.9 were assumed for all years except 2003 and 2004 when values of 0.75 and 0.70 were assumed.

Likelihood equations

The model was fit to the data by minimizing an objective function (*O.F.V.*) that is the sum of the negative log likelihoods for the CPUE series (ℓ_{CPUE}), the tagging data (ℓ_{tag}) and length compositions in the catches ($\ell_{catch-comp}$). We used lognormal error structures for the CPUE time series, a Poisson error structure for the tagging data and a robust normal error structure (Fournier et al. 1990) for the proportions at length in the catch. For each fishery and size group, the log-likelihood for the CPUE component of the model is:

$$\ell_{CPUE}^{g,size} = -\sum_{1}^{n} \ln \sigma^{g,size} - \frac{1}{2}\log 2\pi - \sum_{t} \frac{(\ln \tilde{I}_t^{g,size} - \ln I_t^{g,size})^2}{2(\sigma^{g,size})^2} ,$$

where *n* is the number of observations in the series, $\sigma^{g,size}$ is the standard deviation of a normal distribution prior to exponentiation and $\tilde{I}_t^{g,size}$ is the observed CPUE index value in year *t*, region *g* and group *size* (mature or immature). We used a constant value of 0.3 of all $\sigma^{g,size}$ in this analysis. We also used the standard error of each estimate of $I_t^{g,size}$ as an estimate of $\sigma^{g,size}$, an approach that weights the contribution of each year differently based on the precision of the estimate. This alternative made little difference in the overall fits of the model so we retained the constant value of 0.3. This equation was appropriately modified when different grouping of the data were used.

From Harley (2002), for a given gear and sex category, the robust normal log-likelihood for proportions at length in the catch is:

$$\begin{aligned}\ell_{length-comp.}^{g,s} = 0.5\sum_{t=1}^{Y}\sum_{l=1}^{A}\log(2\pi(\varsigma_{t,l}^{s,g} + 0.1/A_l)) + \sum_{t-1}^{nyears} A_l \log(\tau_t^{s,g}) \\ -\sum_{t=1}^{Y}\sum_{l=1}^{A}\log\left[\exp\left\{\frac{-\left(\tilde{P}_{l,t}^{s,g} - P_{l,t}^{s,g}\right)^2}{2\left(\varsigma_{t,l}^{s,g} + 0.1/A_l\right)\tau_t^{s,g}}\right\} + 0.01\right] \quad ,\end{aligned}$$

where *Y* is the number with observed proportions at length, A is the number of length categories, $\tau_t^{s,g}$ is the sample size and $\varsigma_{t,l}^{s,g}$ is the variance. We set the maximum sample size at 3,000 to keep a few years with very large samples from dominating the fit, and used the variance of the predicted proportions (Fournier et al. 1990):

$$\varsigma_{t,l}^{s,g} = P_{t,l}^{s,g}(1 - P_{t,l}^{s,g}) .$$

We used a length-frequency distribution of the sexes combined for some years (see the Data section) with appropriate modifications to the above equations.

The log likelihood for the tagging component of the model is:

$$\ell_{tag} = -\sum_{t,a} T_{t,a} + \sum_{t,a} \tilde{T}_{t,a} \ln T_{t,a} - \sum_{t,a} \ln(\tilde{T}_{t,a}!) ,$$

where ~ is again used to denote the observed data.

The final objective function is then:

$$O.F.V. = -\left(\sum_{g,sizes} \ell_{CPUE}^{g,sizes} + \sum_{g,s} \ell_{catch-comp.}^{g,s} + \ell_{tag} \right)$$

We programmed this model using AD Model Builder (Fournier 1996). AD Model builder uses the C++ auto-differentiation library for rapid fitting of complex non-linear models, has Bayesian and profile likelihood capabilities, and is designed specifically for fitting these types of models.

The production model and reference points

We modelled the population dynamics of porbeagle using two equations: a spawner-recruit relationship that expresses recruitment as a density dependent function of spawner biomass, and the replacement line, the slope of which is the inverse of the rate at which recruits produce replacement spawners. Here, an implicit assumption is made that all density-dependent processes occur between spawning and recruitment. The production model also includes a third component: a yield-per-recruit relationship. We used the selectivity curves for the Shelf-Edge fishery in the following analysis. All results are therefore specific to that fishery. Results would vary if other selectivity curves had been assumed.

The SR model was discussed in the previous section. We modelled the rate at which recruits produce spawners (the inverse of the slope of the replacement line) by calculating the number of spawners per recruit (SPR_F) as a function of fishing mortality (Shepherd 1982, Mace and Sissenwine 1993, Mace 1994):

$$SPR_F = 0.5 \sum_{1}^{a_{max}} \left[m_a e^{-\sum_{1}^{a-1} (M_{\mathrm{F},a-1} + F_{\mathrm{F},a-1}^{g})} \right] \quad ,$$

where $F_{\mathrm{F},a}^{g}$ is the age and gear specific fishing mortality rate for females. Note that the resulting reference points are specific to the selectivity assumed in the calculation.

The yield-per-recruit for a given *F* (YPR_F) is found similarly:

$$YPR_F = \sum_{s} \sum_{1}^{a_{max}} e^{-\left(0.5M_{s,a} + \sum_{1}^{a-1}(M_{s,a-1} + F_{s,a-1}^{g})\right)} \left(1 - F_{s,a}^{g}\right) w_{s,a} \ .$$

For a given value of *F*, the spawning biomass produced by the number of recruits in year *t* is $SSN = SPR_F \cdot R_t$. Equilibrium spawning biomasses and recruitment levels (denoted with asterisks) were found by solving this equation for R_t, and substituting the result in the spawner-recruit model (Quinn and Deriso 1999):

$$\frac{SSN^*}{SPR_F} = \frac{\alpha SSN^*}{1 + \dfrac{\alpha SSN^*}{R_{asy}}} \ .$$

The equilibrium spawning biomass (*SSB**) is then:

$$SSN^* = \frac{(\alpha SPR_F - 1) R_{asy}}{\alpha} \ ,$$

and the equilibrium number of recruits (*R**) is found by substituting the *SSN** in the spawner-recruit model:

$$R^* = \frac{\alpha SSN^*}{1 + \dfrac{\alpha SSN^*}{R_{asy}}} \ .$$

The equilibrium catch (*C**) is *R** multiplied by the yield-per-recruit for the given value of *F*:

$$C^* = R^* \cdot YPR_F \ .$$

Reference points from the spawner-per-recruit and yield-per-recruit analyses were found using a grid search across a set of *F*'s {0 to 2.0; increment of 0.0025}. We calculated YPR_F and SPR_F for each value of *F*, and reference points were then estimated by selecting the fishing mortality rate corresponding to the appropriate reference point criterion. The $\mathrm{SPR}_{x\%}$ reference points were found by selecting the fishing mortality rate where the SPR_F was x% that of $SPR_{F=0}$.

We estimated five reference points from the production model. The equilibium spawning biomass in the absence of fishing, SSN_{eq}, was estimated directly from the production model. A spawning biomass of 20% SSN_{eq} is sometimes used as a minimum threshold population size (Beddington and Cooke 1983, Goodyear 1993). $SSN_{20\%}$ was calculated as 20% the equilibrium female spawner abundance in the absence of fishing:

$$SSN_{20\%} = 0.2\frac{(\alpha SPR_{F=0} - 1)R_{asy}}{\alpha}$$

The grid searches were used to find the fishing mortality rate that produces maximum sustainable yield (F_{msy}), the corresponding spawner biomass that produces maximum sustainable yield (SSN_{msy}) and the fishing mortality rate that drives the population to extinction (F_{col}). We estimated F_{msy} by calculating C^* for each value of F, and selecting the value of F where C^* was maximized. SSB_{msy}, the spawning biomass at MSY, was found similarly. The equilibrium fishing mortality rate at which the population goes extinct, F_{col}, is determined by the slope of the SR relationship at the origin α, and is the value of F where $1/SPR_{\mathrm{F}} = \alpha$.

Population viability analysis

Under SARA, incidental harm permits can only be issued if the activity will not jeopardize the survival or recovery of the species. To assess recovery and timelines for recovery, recovery targets are required. Recovery targets have not yet been established for porbeagle. Here, we assess how differing levels of incidental harm (mortality associated with bycatch in fisheries not targeting porbeagle) affects the recovery timelines relative to two commonly used fishery reference points $SSN_{20\%}$ and $SSNmsy$. These are not recovery targets, but are reference points against which population growth can be evaluated.

Population viability analysis is an important tool which can be used to evaluate recovery potential, recovery trajectories and recovery times. In a PVA, a population dynamics model is used to determine how the probability of persistence is affected by current conditions and future perturbations (Beissinger and McCullough 2002). The goals of a PVA are to 1) determine the current viability of a population, 2) identify threats to persistence, and 3) provide a defensible structure for management and legal action. Typically, there are several other benefits of PVA such as identifying information gaps, and directing future research.

A disadvantage of PVA is that it is data intensive and the minimum data required is only available for a few species. For porbeagle, we have estimates of reproductive rates (as characterized via the spawner-recruit model), maturity schedules and mortality rates. However, we do not presently have estimates of variances for these life history parameters or their temporal autocorrelation, two factors than can effect recovery times and population viability. We therefore used two methods to evaluate how mortality associated with bycatch effects recovery potential and timelines.

First, we projected the population forward deterministically (no variability added) from the estimated 2004 population size and age-structure using the estimated life history parameters and an assumed bycatch rate. We used the selectivity parameters from the Shelf-Edge fishery for these simulations. Simulations were carried out for 17 levels of bycatch mortality (defined as the proportion of the vulnerable biomass taken as bycatch) ranging from 0.0 to 0.1. Population projections were 100 years in length.

Second, we used Monte Carlo simulations (random variability added) to the SR and natural survival component of the model, selecting variances that we consider would be high for a long-lived, slow-growing population such as porbeagle. The modification to the SR model component to add random variability (assumed lognormal) is:

$$R_t = \frac{\alpha SSN_t}{1+\frac{\alpha SSN_t}{R_{asy}}} \exp\left(\varepsilon_t \sigma - \frac{\sigma^2}{2}\right), \quad \text{where } \varepsilon_t \sim N(0,1)\,,$$

and σ is the standard deviation of a normal distribution prior to exponentiation, and ε_t is a random number drawn from a standard normal distribution separately for each year in the projection. The term $-\sigma^2/2$ corrects for the fact that the lognormal distribution does not have an expectation of 1 (Hilborn 2001). We assumed σ =0.2 for the analysis present here.

We added random variability to natural survival by first mapping the age-specific probability that a fish survives (a proportion) to the real line using a logistic transformation, assuming variability was normally distributed on the transformed scales, and then back-transforming to get the proportion that survive (Gibson and Myers 2003b). The first transformation is:

$$\text{logit}(s_a) = \log\left(\frac{s_a}{1-s_a}\right).$$

The random component was introduced on the logistic scale by drawing a random number from a normal distribution with mean of zero and a standard deviation of 0.2. This value was back-transformed to obtain the random s_a:

$$s_a = \frac{\exp(\text{logit}(s_a)+\varepsilon_a)}{1+\exp(\text{logit}(s_a)+\varepsilon_a)}, \quad \text{where } \varepsilon_\text{a} \sim N(0,0.2)\,.$$

Note that in both cases, the standard deviation selection was arbitrary, but chosen to reflect high variability. Two hundred population simulations were then carried out for each of the 17 levels of *u*. The same random deviates were used for each level of *u* to ensure that the results do not reflect variations in the random number series. Recovery probabilities were then calculated as the proportion of the simulations that recovered in a given time frame. These simulations were only carried out using the parameter estimates from Model 3.

RESULTS

Initial model fitting indicated that, as is often the case with these types of models, estimation of natural mortality was confounded with estimation of selectivity. Additionally, results differed depending on how CPUE was included, and α could not be successfully estimated when the integrated CPUE analysis was used. We are therefore presenting 4 models fit to the data, each representing a different scenario:

- Model 1: standardized CPUE for immature and mature porbeagle, obtained from the GLM, was used; constant *M*: *M*=0.1 and 0.2 for immature and mature porbeagle respectively.
- Model 2: integrated CPUE by weight; *M*= 0.1 and 0.2 for immature and mature porbeagle respectively; constant α =2.0 (lower productivity scenario).
- Model 3: integrated CPUE by weight; *M*= 0.1 and 0.2 for immature and mature porbeagle respectively; constant α =2.5 (intermediate productivity scenario).
- Model 4: integrated CPUE by weight; *M*= 0.1 and 0.2 for immature and mature porbeagle respectively; constant α =3.2 (higher productivity scenario).

Fits to the data

Based on the maximized likelihoods (Table 9), of the three models with the integrated CPUE, Model 4 is the most plausible scenario, followed Model by 3. Model 2 is the least plausible of these models. While Model 1 cannot be compared to the other models using the likelihood, because of the afore mentioned issues with the standardization, and because the length frequency information enters the model twice, it is the least preferred of the four models. The estimated selectivity of the three fisheries is roughly similar among the four models (Figure 9; Table 9), with similar parameter estimates for all four models (Table 9). Standard errors on all estimated parameters and derived values appear very low, possibly as a result of bounds placed on estimated parameters, and do not reflect the true uncertainty in the estimates.

Differences in fits to the data are also subtle among the models. The fitted CPUE from Model 1 shows a declining trend for mature porbeagle over the range of the data, although the CPUE for the early years is not predicted well by the model (Figure 10). For immature porbeagle, a declining trend is evident during the 1990’s, although the recent increases are not well fitted by the model. The predicted CPUE series for porbeagle are similar among the integrated models, although the data show considerable variability around the fitted relationship (Figures 11.1 to 11.5). No trend in the CPUE residuals is evident (Figure 12). Fits to the catch at length data are shown in Figures 13.1 to 13.8, and are again virtually indistinguishable among models. Although no trend is apparent in the catch at length residuals (Figures 14.1 to 14.3), the model apparently under-predicts the proportion of larger fish in early years as well as the proportion of smaller fish in the later years (Figure 15), although as shown in Figures 14.1 to 14.3, the magnitudes of the residuals are relative small.

Residual patterns for the tagging recaptures are also similar among models (Figure 16). In all models, the catch of younger (< age-4), tagged porbeagles is overestimated, whereas the catch of tagged, older porbeagles is underestimated. Comparison of the log likelihoods (Figure 16) indicates that models 2, 3 and 4 provide better fits than Model 1, probably due to the higher abundance during the late 1990's predicted by these models.

Population dynamics

Estimates of α could not be obtained from the models with the integrated CPUE, but was estimated to be 2.06 using Model 1 (Table 9). Values of 2, 2.5 and 3.2 were used in the integrated models, and are thought to span the range of plausible values for porbeagle (see Discussion). The estimate of the maximum lifetime reproductive rate ($\alpha * SPRF0$) from Model 1 is: 1.7 spawners per spawner, and the assumed values from Models 2 to 4 range from 1.6 to 2.6. As expected for sharks, these values are at the lower end of the range for fish populations (Myers et al. 1999), and are indicative of very low population growth rates.

Reference points

Estimates of F_{msy} from the four models range from 0.033 to 0.065 (Table 9), and of F_{col} from 0.069 to 0.134. The estimates of SSN_{msy} were similar among Models 1 to 3 (37,586 to 40,676 females), but lower from the highest productivity model (Model 4: 29,382 females). The spawning biomass per recruit reference points $F_{35\%}$ and $F_{45\%}$ met or exceeded F_{col} in all model runs and are not safe reference points for porbeagle fisheries. These reference points were calculated using the selectivities for the Shelf-Edge fishery, but given the similarity to the selectivities for the Basin fishery, reference points for the Basin would likely be close to the values for the Shelf-Edge. Reference points for the NF-Gulf fishery would likely differ, although little fishing is presently occurring in that region (Table 4).

Trends in abundance and exploitation

Trends in abundance are also roughly similar between the models (Figure 17). Estimates of the number of spawners in 1961 were highest from Model 2. All models suggest an increase in spawner abundance in the late 1970's and early 1980's, although the increase is small. The estimated total number of porbeagle also increases only slightly during the 1980's (Figure 17). The models with the integrated CPUE analysis (Models 2 to 4) have higher recent abundance than the model using the standardized CPUE (Model 1).

Prior to 1980, estimates of exploitation rate are also similar among the models (Figure 18), although the Model 1 (standardized CPUE) produces estimates of recent exploitation that are higher than the models with the integrated CPUE. All models estimate exploitation in the Basin region to be less than 2% in 2004 (Table 10). Estimates of the exploitation rate in 2004 for the Shelf-Edge fishery are similar among Models 2, 3 and 4 (0.037 to 0.39), but higher from Model 1 (0.074). With the exception of Model 1, these estimates are close to the values expected to produce MSY.

Population status

Estimates of the population size in 2005 (Table 11) are lowest for the model with the standardized CPUE (Model 1 - 94,309 fish) and highest for the model with the lowest productivity (Model 2- 195,230 fish). The estimated number of mature females from Models 2 and 3 are about 9,000 to 13,000 fish (Table 11), or about 15% of the population. The models indicate that the population is about 0.10 to 0.24% its size in 1961 (Table 11), and that female spawner abundance has declined to about 6% to 15.0% of its 1961 level. The models with integrated CPUE indicate that the reduced quotas from 2002 to 2004 have more or less halted the decline in population size, whereas the model using standardized CPUE indicates a slowed, but ongoing decline. The decline in female spawner abundance since 2002 has been greater than the decline in total number (Table 11), potentially as a result of higher exploitation on porbeagles in the late 1990's effecting present spawner abundance.

Estimates of the vulnerable biomass in 2005 differ depending on the assumed selectivity as well as among models (Table 12). Assuming the Shelf-Edge selectivity, the preferred models (integrated CPUE), place the vulnerable biomass in 2005 (mid-year) at just over 4,500t.

Recovery trajectories

All models indicate that the northwest Atlantic porbeagle population can recover if levels of human-induced mortality are kept low (Figure 19). Although recovery rates vary among models, time scales are on the order of decades. Models 1, 2 and 3 predict a slight decline in spawner abundance for the next few years, possibly as a result of high fishing mortality for juveniles in the late 1990's. The preferred integrated CPUE models indicate that, in the absence of human-induced mortality, recovery to $SSN_{20\%}$ should occur circa 2015. Using the Shelf-Edge selectivity, a human-induced mortality rate of 2% of the vulnerable biomass delays recovery slightly to about 2020. Estimated recovery times to SSN_{msy} vary depending on the assumed productivity. Based on Models 1 and 2, in the absence of human-induced mortality, recovery to SSN_{msy} is expected to occur about 2060 to 2070, whereas Model 4 predicts recovery in the early 2030's. A human-induced mortality rate of 4% of the vulnerable biomass is expected to delay recovery to SSN_{msy} to the 22nd century based on all models except Model 4. Model 4 provides the most optimistic scenario in part due to the higher assumed productivity but also to the lower estimated reference points.

Results of a stochastic PVA, based on Model 3, are shown in Figures 20 and 21. When random variability is added to recruitment and natural mortality, recovery times are slightly longer (compare with Figure 19). None of the simulated populations went extinct, suggesting (if the underlying dynamics are correct, e.g. no Allee effects) that the population is not so small as to be threatened by demographic stochasticity. At a human-induced mortality rate of 2%, time to recovery to SSN_{msy} varied by about three decades (Figure 20),and 90% of simulated populations recovered to SSN_{msy} by c.2075. At a human-induced mortality rate of 4%, about 30% of the populations did not recover to SSN_{msy} within 100 years. At human-induced mortality rates greater than about 7%, most of the simulated populations did not recover with 100 years (Figure 21).

A five-year recovery plan has been established for this population, covering the time period to the end of 2007. We evaluated the effects of continuing the plan for its duration on recovery times using deterministic projections based on Model 1. This model was chosen as the most pessimistic scenario of the four models presented. Under this scenario, the effect of continuing with the current management plan until 2007 is to delay predicted recovery times by less than five years (Figure 22). The effect would be less under the other models due to the larger population sizes estimated by these models.

Alternate model runs

The four models were run with various data weighting combinations, various combinations of fixed and estimated parameters, and various combinations of data without finding significantly improved or very different fits. Five other variations on the model that were examined are:

1) Estimating the 1961 population size without the assumption that the population was at an unfished equilibrium in 1961 (this is a key assumption, but one that is likely met). We tested the effect of this assumption by estimating a starting number of age-1 fish (1961 - constrained to be less than R_{asy}), assuming a unfished equilibrium age structure in 1961 and fitting the model using 1961 constants. This approach produced only very small differences than fitting the model assuming the population was at an unfished equilibrium in 1961.

2) Using length-based selectivity rather than age-based selectivity using Model 1. Hook retention selectivity is on the basis of size, not age, whereas contact selectivity may be on the basis of size or age. Maximised likelihoods were less for all four models when length-based selectivity was used, so we retained the age-based selectivity model.

3) Use of recruitment deviates. We fit the models estimating recruitment deviates for all years after 1986 using the Model 4 constants. In these fits, exploitation rates tended to be slightly higher, declines similar, and recovery rates more rapid. Total abundance in 2005 was estimated to be 265,000 individuals using this model. However, a temporal pattern was present in the deviates: positive for the seven of the last eight years, and nine negative deviates in the previous 10 years and the estimates may not be reliable as a result. In this case, the estimated female spawner abundance in 2005 was 12,300 fish; 0.14 times that of 1961 and 0.87 that of 2002. Given this scenario, recovery to $SSN_{20\%}$ occurs before 2020 and to SSN_{msy} around 2030 in the absence of human-induced mortality. The vulnerable biomass (Shelf-Edge fishery) in 2005 was estimated to be about 7,000t using this model. This was the most optimistic recovery scenario of the realistic fits.

4) Use of a plus group. The four models presented use a plus group to model the older age classes, a method that precludes the need to specify a maximum age. A consequence of including a plus group is that as mortality on the older age classes decreases, abundance in the older age classes increases. We investigated the use of the effect of the plus group by fitting the model using a maximum age of 25 and the Model 4 constants. Parameter estimates were only slightly different when this approach was used.

5) We modified the tagging component of the model by assuming that all tagged sharks were age-0, then 50% of tagged sharks were age-0 and 50% were age-1, and used both

the mean age-specific selectivity and the selectivity for the Shelf-Edge fishery only. None of these attempts improved the residual pattern.

DISCUSSION

The present management plan was developed based on the analysis of Campana et al. (2001). In that assessment the Paloheimo Z analysis was considered the most reliable of the three methods used to estimate fishing mortality rates in that assessment. Of the models presented here, Model 1 provides estimates of total mortality that are higher than their Paloheimo Z analysis (Table 13), whereas the other models provide estimates that are less. Model 4 has the closest agreement with the Paloheimo Z analysis of Campana et al. (2001).

Of the 4 models presented in this document, statistically, only three can be compared because the data and number of parameters are different for Model 1. This model is the least preferred because variability in the CPUE data is not carried forward to the population model and when it is used, the length frequency data enters into the model twice (once when calculating the number mature and immature and once when fitting the population model). For these reasons, Models 2, 3 and 4 are preferred. When the integrated CPUE model is used, α was not successfully estimated. For this reason, models were fit using three values of α thought to reasonably cover the range of plausible values. Of the integrated models, Model 4 (higher value of α) provided the best statistical fit, while Model 2 (lower α and higher M_{mat}) provided the worst fit. From the perspective of assessing the effects of human-induced mortality, the question of selecting a preferred alternative from Models 3 and 4 may be moot: as the higher productivity model (Model 4) provides lower estimates of current abundance which, when estimating a quota, offsets the higher reference fishing mortality rates obtained with this model.

The incorporation of the tagging data into the model was not entirely satisfactory, given the pattern in the residuals in relation to age. Campana et al. (2001) initially estimated fishing mortality using the tagging data with an assumption that the tag loss was 0.1. This assumption was initially built into the model, but the model fits didn’t come close to the data. We therefore chose to estimate an initial rate of tag loss and/or mortality. When this parameter is estimated, the fits intersect the data, but survival estimates from the model are still less than that suggested by the tagging data. Rather than assuming a one-time loss, tagging loss and/or mortality could occur slowly over time. This approach to incorporating the tagging data was suggested by Harley (2002). However, given that survival from tagging data is higher than from the other data, this approach would not alleviate the residual problem. The issue may result from the assumed selectivity (all regions combined for the tagging component), and the addition of a spatial component to the tagging data may therefore be a solution. Additionally, two age groups (age-0 and age-1) were initially tagged, whereas in the model a single age of tagging was assumed. Clarification of ages at the time of tagging may improve the fit. A third possibility that might improve the fit would be to use lower natural mortality rates for the mid-range age classes. Overall, the tagging data indicates higher abundance and lower total mortality than predicted by the models.

The values of α used in the integrated CPUE models compare favourably with published estimates of juvenile survival in sharks. If a mean litter size of 3.9 is assumed, a value of α of 2 equates to a survival rate of 0.51 between birth and age-1. Using a depletion method with a marked population, Gruber et al. (2001) estimated annual survival of juvenile lemon sharks to vary between 0.38 and 0.65. Most sharks in their study were marked at age-0 although some age-1 and age-2 sharks were also included. Our above assumed values include deaths at time of birth and onset of feeding that would not be a part of the Gruber et al. study, so a survival estimate to the lower end of their range is not implausible given the differences in our studies.

That said, the maximum intrinsic rate of increase (r_{max}) is low relative to estimates for some other sharks. Using the Leslie matrix method (Krebs 1985) and the demographic parameters from Models 2 and 4, r_{max} is estimated to be 0.032 and 0.061 respectively. These values bracket the value of 0.051 estimated by Campana et al (2001). Cortes (2002) estimated a lower value of r_{max} for porbeagle (0.022) due to differences in the assumed natural mortality and longevity. McAllister et al. (2001) derived priors for r_{max} for sandbar shark with medians in the range of 0.07 to 0.09 and for blacktip shark with a median of about 0.125. Smith et al. (1998) estimated r_{max} for several shark species, although due to methodological differences, their results and ours are not directly comparable (our estimate is low relative to their values for most other species). If productivity is being overestimated in our study, the results from Model 2 would be most conservative. Note however, that although a productivity scenario cannot be selected on the basis of model fit, the estimates of the vulnerable biomass is 2005 is similar among the integrated CPUE models.

Reference points calculated herein based on the production model do not account for uncertainty in the SR parameter estimates. Gibson and Myers (2004) found that when reference fishing mortality rates were calculated by maximizing the expectation of the catch (a decision-theoretic method that accounts for uncertainty in the SR parameter estimates), the resulting reference fishing mortality rates were lower, but on average produced higher yields, maintained larger population sizes and substantially reduced the risk of overexploiting the population. While the method used here is commonly used to estimate reference points, it is not precautionary if given uncertainty in the SR parameters.

When models that included recruitment deviates were fit to the data, slightly higher estimates of abundance were obtained. The temporal pattern in the deviates (positive for seven of the last eight years, and nine negative deviates in the previous 10 years) is indicative of temporal autocorrelation which has not been considered here. Using a simple two species model with environmental forcing, MacCall (2002) shows how changes in prey density as a result of environmental cycling can create long-term cycles in abundance of a predator species. If autocorrelation is present in the recruitment of porbeagle, it could either increase or decrease the time to recovery depending on the time scales of the cycles.

While the switch to a model with an integrated CPUE expressed in weight units has aided in reconciling differences between the model and Paloheimo Z results in the 2001 assessment, at least two issues are unresolved: the tagging data indicate that survival and abundance may be higher than predicted by the models and the high (but variable) CPUE in recent years is not fit well by the models. While resolution of these issues would substantially enhance the credibility of the models, these two lines of evidence are suggestive that management advice based on the predicted abundances from the models would be more precautionary.

As is the case with predictive models, the estimated recovery times are conditional on the model assumptions. Here, a key assumption is that the ocean ecosystem in the future will be similar enough to the conditions in the past such that the life history parameters used to estimate future population trajectories will still apply. If ecosystem conditions change markedly, recovery trajectories would also be expected to change.

While the overall prognosis in this document is that recovery times will be slow, the overall indications are, given the past performance of this fishery (slight recovery in the late 1970’s and early 1980’s), the modelling results and the limited number of human activities that effect porbeagle, that this population can recover if human-induced mortality is appropriately regulated.

Interpretation of results in the context of incidental harm permitting

DFO (2004b) poses a set of questions as a framework for assessing whether incidental harm permits should be issued. In this section, we use these questions as a guide for this discussion.

1. What is present/recent species trajectory?

All of our analyses indicate that the abundance of porbeagle in the northwest Atlantic declined during the late 1960’s, increased slightly during the late 1970’s and early 1980’s, and decreased again during the late 1900’s. Based on the integrated CPUE models, the analyses indicate that the decline in total abundance has either slowed or halted since the quota reductions in 2002. The abundance of spawners has continued to decline, and is probably due in part to high exploitation on juvenile porbeagle during the late 1990’s. Population size is expected to increase now that exploitation rates are lowered, but with a time lag as cohorts subjected to lower exploitation have time to mature. Analyses indicate that spawner abundance is expected to continue to decline during the next couple of years until recent cohorts mature.

2. What is the present/recent species status?

Our analyses indicate that the estimated number of mature females is in the range of 6,000 to 13,000 individuals, and in the range of 6% to 15% of its 1961 level. The total population size is thought to be about 10% to 24% its size in 1961 and about 92% to 123% its size in 2002. Spawner abundance in 2005 is about 73% to 90% its 2002 value.

These later results likely reflect the effects of higher exploitation on juvenile porbeagle in the late 1990's combined with a recent increase in juvenile abundance as a result of the reduced exploitation since that time. If these juvenile fish are provided the opportunity to mature and reproduce, the expectation is that the population will then increase.

3. What is the expected order of magnitude / target for recovery?

While recovery targets have not been established for porbeagle, in this document we used 20% of the equilibrium unfished spawner abundance and the equilibrium spawner adundance at MSY to evaluate time frames for recovery. From Model 3, these values are about 16,000 and 41,000 fish respectively. Spawner abundance in 2005 is estimated to be roughly 77% of $SSN_{20\%}$.

4. What is the expected general time frame for recovery to the target?

All analyses indicate that this porbeagle population can recover even if human-induced mortality is greater than zero, but that the time horizon for recovery is sensitive to the amount of human-induced mortality. The integrated CPUE models predict recovery to $SSN_{20\%}$ is expected to take on the order of about 10 years in the absence of human-induced mortality, and to occur before 2020 if the human-induced mortlaity rate is 2% of the vulnerable biomass. These latter estimates are conditional on the assumed selectivity (Shelf-Edge) and recovery times given human-induced mortality will differ if a different selectivity is assumed.

5. What is the maximum human-induced mortality which the species can sustain and not jeopardise survival or recovery of the species?

Of the three integrated CPUE models presented, Model 2 is the least optimistic due to the lowest productivity assumed in this model. This model predicts that recovery will occur if human-induced mortality is less than 4% the vulnerable biomass. Under this model, recovery to SSN_{msy} is predicted to take over 100 years at exploitation rates of 2% the vulverable biomass. These estimates are again conditional on the assumed selectivity. Assuming the Shelf-Edge selectivity, Models 3 and 4 (which fit better than Model 2) predict that keeping the rate of human-induced mortality to less than 2% of the vulnerable biomass would be precautionary and would keep expected recovery times on the order of decades.

6. (a) What are the major potential sources of mortality/harm?

Throughout this document, we have focused on mortality associated with fisheries. Campana et al. (2003) states: "All lines of evidence indicate that fishing mortality is largely and solely responsible for the decline in population abundance since 1961." The COSEWIC listing document (COSEWIC 2004) also does not identify any factors other than fishing that may be either responsible for the decline or limiting recovery. COSEWIC (2004) discounts fisheries on food supplies as a factor in the porbeagle decline due to the diverse diet of porbeagle (Joyce et al. 2002). Under SARA, directed fisheries for endangered species cannot be permitted (DFO 2004a). Fisheries directed at other species that take porbeagle as bycatch are therefore the only known sources of human-induced mortality that need to be considered for incidental harm permitting.

6. (b) Do Canadian activities alone impact the species? For transboundary species that migrate in and out of Canadian waters, list all international activities that may impact the species.

Canadian activities are not the only activities that affect porbeagle, but they are the major ones. The Japanese tuna longline fishery catches porbeagle, but they are now excluded from the Canadian EEZ. The US commercial tuna and swordfish pelagic longline fisheries also catch porbeagle as bycatch (COSEWIC 2004; our Table 3). Recent reported landings from other countries are small relative to the Canadian landings (Table 1).

7. For those factors NOT dismissed, quantify to the extent possible the amount of mortality or harm caused by each activity.

Nearly all bycatch in Canadian waters is thought to occur in the Scotia-Fundy region, and bycatch from 2000 to present is provided in Table 2. The bycatch has ranged from 13.5t to 53.3t during this time period and has been higher in recent years.

8. Aggregate total mortality/harm attributable to all human causes and contrast with that determined in Question #5.

Assuming the Shelf-Edge selectivity, the preferred models (Models 3 and 4), place the vulnerable biomass in 2005 (mid-year) at just over 4,500t. The bycatch in 2004 was ~46t. Assuming the bycatch is taken with a selectivity similar to the Shelf-Edge directed fishery, at a rate of 1%, an appropriate level of bycatch would be 45t, and at twice this rate 90t. These values would also be appropriate if Model 1 (the least productive scenario) was assumed.

Mitigation of harm

Analyses presented herein indicate the current population is not so small that random factors will threaten the population. Although the recent trajectory of the stock is downwards, the expectation is that abundance will increase as spawner abundance increases due to maturity of juveniles, so that survival or recovery is not in jeopardy in the short term. The known sources of human-induced mortality (bycatch) are under management control and, assuming they can be monitored and enforced, is unlikely to increase during the permitting process. As a result, a low level of human-induced mortality will not leave the stock in worse shape at the end of the permitting period than it was at the start, and if appropriately controlled, will not "jeopardise the survival or recovery of the species" during the period of the permit.

Under SARA, before the competent Minister can issue an incidental harm permit, it will have to be shown that:

a) all reasonable alternatives to the activity that would reduce the impact on the species have been considered and the best alternative has been adopted
b) all feasible measures will be taken to minimize the impact of the activity on the species or its critical habitat or the residences of its individuals; and
c) the activity will not jeopardize the survival or recovery of the species.

In this document, we have quantified the extent to which incidental harm will affect recovery of porbeagle in the northwest Atlantic, but have not evaluated methods other than quota control through which mortality may be minimized. The effect of bycatch on the porbeagle recovery may be different than the effect predicted here if the selectivity used in the projections is different than that actually used in practice. However, given the predicted times to recovery in the absence of human-induced mortality, recovery times will still be on the order of decades irrespective of the gear selectivity.

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Table 1. Reported landings (metric tonnes) by country for NAFO areas 2 to 6. Canadian landings have been converted to live equivalent weight, which differs in some cases from the live weight recorded in the statistics.

Year	Canada	Faroe Is	France	Iceland	Japan	Norway	Spain	USSR	USA	Total
1961	0	100				1824				1924
1962	0	800				2216				3016
1963	0	800				5763				6563
1964	0	1214		7		8060				9281
1965	28	1078				4045				5151
1966	0	741				1373				2114
1967	0	589			36					625
1968	0	662			137	269				1068
1969	0	865			208					1073
1970	0	205			674					879
1971	0	231			221					452
1972	0	260				87				347
1973	0	269								269
1974	0									0
1975	0	80								80
1976	0	307								307
1977	0	295								295
1978	1	121								122
1979	2	299								301
1980	1	425								426
1981	0	344			3					347
1982	1	259			1					261
1983	9	256			0					265
1984	20	126			1	17				164
1985	26	210			0					236
1986	24	270			5			1		300
1987	59	381			16			0	12	468
1988	83	373			9			3	32	500
1989	73	477			9			3	4	566
1990	78	550			8			9	19	664
1991	329	1189			20			12	17	1567
1992	814	1149			7			8	13	1991
1993	920	465			6			2	39	1432
1994	1573				2				3	1578
1995	1348		7		4				5	1364
1996	1043		40		9				8	1100
1997	1317		13		2				2	1334
1998	1054		20		0				12	1086
1999	955				6					961
2000	899				0					899
2001	499				0					499
2002	229				0					229
2003	139				0					139
2004	218				0					218

Notes:

France data is from FAO Statistics (1998)
Northwest Atlantic Data for 1950 - 60 is from FAO (ICCAT Report of Shark Working Group, Miami, 26 - 28 February 1996)
Canada for 1961 - 90 is from NAFO
Canada for 1991 - 2002 is from DFO Zonal Statistics File, corrected to appropriate live equivalent weight.
Canada for 2003-2004 is from DFO MARFIS
Faroe Is for 1961 - 63 is from FAO (ICCAT Report of Shark Working Group, Miami, 26 - 28 February 1996)
Norway from 1961-86 is from NAFO
Northwest Atlantic Data for 1964 - 86 is from NAFO
Northwest Atlantic Data for 1987-2004 is from Scotia-Fundy & NF IOP (includes landings and discards)
Northwest Atlantic Data (US/ 1961 - 94) is from FAO (ICCAT Report of Shark Working Group, Miami, 26 - 28 February 1996)

Table 2. Porbeagle bycatch (kg) in Scotia-Fundy fisheries.

Year	2000	2001	2002	2003	2004
Groundfish fixed gear 45 - 65	997	789	958	2,400	2,031
Groundfish fixed gear <45	4,743	6,925	13,141	13,041	14,044
Groundfish inshore					56
Groundfish midshore	461	518	697	1384	101
Groundfish offshore	191	285		220	409
Groundfish unspecified	456	1,059	1,184	1,105	1,010
Total Groundfish	6,848	9,576	15,980	18,150	17,651
Directed porbeagle exploratory	870,741	476,703	172,001	86,059	172,520
Swordfish	5,482	9,582	18,939	29,160	22,155
Tuna	1,266	577	18,435	5,559	6,156
Herring		256			23
Total	884,337	496,694	225,355	138,928	218,505
Total from bycatch	13,596	19,991	53,354	52,869	45,985
Percent total from bycatch	2%	4%	24%	38%	21%

Table 3. Porbeagle catch (kg) in fisheries outside of Scotia-Fundy.

Year	2000	2001	2002	2003	2004
Newfoundland fixed gear	141	946	1,851	1,071	142
Newfoundland mobile gear			40		
Gulf (all gears)	18,976	1,192	11,566	2,565	12,876
Gulf (unspecified shark)[1]	8,378	6,945	8,799	5,156	2,894
USA (all gears)[2]	3,595	785	1,813	1,185	

[1] may include porbeagle
[2] landings only

Table 4. Proportion of the reported, directed porbeagle landings from each of three regions.

Year	Basin	NF-Gulf	Shelf-Edge
1988	0.03	0.33	0.64
1989	0.09	0.35	0.56
1990	0.32	0.25	0.43
1991	0.18	0.42	0.40
1992	0.12	0.49	0.39
1993	0.12	0.42	0.46
1994	0.20	0.27	0.53
1995	0.08	0.43	0.48
1996	0.14	0.33	0.54
1997	0.14	0.32	0.54
1998	0.08	0.34	0.58
1999	0.15	0.22	0.63
2000	0.17	0.39	0.44
2001	0.11	0.24	0.66
2002	0.43	0.22	0.35
2003	0.51	0.02	0.47
2004	0.20	0.02	0.78
Average	0.18	0.30	0.52

Table 5.1. Distribution of sets with reported catch (weight) and effort, by vessel and year, in the Basin area.

		YR																			
SUBAREA3	Vessel	1981	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	Total
BASIN	1											x	x								29
	2												x				x				3
	3											x	x	x	x						41
	4											x	x	x	x	x					95
	5							x	x	x	x	x	x	x	x	x					100
	6										x	x	x	x	x	x	x	x			77
	7												x								27
	8											x	x								6
	9											x	x								27
	10											x	x								14
	11										x			x							7
	12												x								2
	13									x		x	x	x							13
	14									x	x	x	x	x	x	x		x		x	52
	15										x					x					2
	16								x	x	x	x	x	x	x		x				55
	17													x	x	x	x	x	x	x	113
	18													x	x	x	x	x			109
	19													x	x	x		x	x		35
	20																	x			10
	21																		x	x	13
	22						x	x	x	x	x										114
	23	x		x	x	x	x	x	x												110
	24						x	x	x												65
BASIN	Total Sets	6		2	3	35	94	74	66	88	47	94	146	94	118	108	66	37	15	26	1119

Table 5.2. Distribution of sets with reported catch (weight) and effort, by vessel and year, in the NF-Gulf area.

		YR																			
SUBAREA3	Vessel	1981	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	Total
NFGULF	1															x					17
	2							x	x	x	x	x	x	x	x	x	x				395
	3												x								6
	4											x			x						9
	5								x	x	x	x	x	x	x	x	x	x			676
	6						x	x	x	x	x										241
	7	x	x	x	x	x	x	x	x												415
	8						x	x	x												202
NFGULF Total Sets		8	61	67	55	68	167	306	245	187	209	78	122	116	88	138	33	13			1961

Table 5.3. Distribution of sets with reported catch (weight) and effort, by vessel and year, in the Shelf Edge area.

		YR																			
SUBAREA3	CFV	1981	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	Total
SHELFEDG	1												x		x		x				23
	2																x				12
	3											x									1
	4										x	x	x	x	x	x					88
	5							x	x	x	x	x	x	x	x	x	x				656
	6													x	x	x	x	x	x		113
	7													x							5
	8												x	x							3
	9											x	x								8
	10													x							10
	11												x		x						17
	12										x	x		x							4
	13											x	x	x	x	x	x	x		x	198
	14															x					12
	15								x	x	x	x	x	x	x	x	x	x			467
	16																			x	3
	17													x	x	x	x				44
	18														x					x	11
	19																	x	x	x	12
	20																			x	6
	21						x	x	x	x	x										207
	22	x																			19
	23	x	x	x	x	x	x	x	x												500
	24						x	x	x												173
SHELFEDG Total Sets		86	59	69	62	58	134	203	192	199	171	220	218	265	270	175	147	22	8	34	2592

Table 6. Summary of model fits for different ways of including the CPUE data for two different weightings of the catch-length data. Within each weighting group, the model with the lowest AIC is the best model.

Model	Catch-length weighting	No. of parameters	Objective function value	Cpue neg. log. like	cpue sigma	AIC	Hessian?	alpha	SSN2005	VB2005 (shelf)	R0
single q	350	0	12,717	18,832	1.09	25,434	no	2.06	18,477	6,564	280,805
areas only added	350	2	12,598	18,722	1.09	25,200	no	3.27	13,333	6,176	140,307
cfv only added	350	14	11,560	17,661	1.07	23,148	no	3.6	10,016	5.510	134,908
seasons only added	350	2	12,469	18,594	1.08	24,942	yes	3.6	13,011	6,165	138,759
cfv+season added	350	16	11,524	17,633	1.07	23,080	no	2.04	13,733	5,219	270,991
CFV+area added	350	16	11,589	17,641	1.07	23,210	no	2.03	13,638	4,817	270,949
CFV+season+area	350	18	11,361	17,538	1.07	22,758	no	3.6	8,836	4,432	135,027
CFV +area+ season within area	350	22	11,298	17,469	1.07	22,640	yes	3.6	9,460	5,026	134,484
Area + CFV within area +season within area	350	36	11,149	17,112	1.06	22,370	n	3.6	9,099	4,887	136,139
individual q's	**350**	**52**	**10,669**	**16,852**	**1.06**	**21,442**	**yes**	**3.6**	**10,086**	**4,855**	**103,596**
single q	6500	0	15,543	18,814	1.09	31,086	no	2.27	16,917	6,324	235,828
areas only added	6500	2	15,530	18,807	1.09	31,064	no	2.25	16,900	5,864	237,244
cfv only added	6500	14	14,439	17,684	1.07	28,906	yes	3.45	10,389	5,144	140,181
seasons only added	6500	2	15,326	18,604	1.09	30,656	yes	3.6	12,871	6,103	139,201
cfv+season added	6500	16	14,368	17,622	1.07	28,768	yes	2.35	12,428	5,007	211,614
CFV+area added	6500	16	14,462	17,760	1.07	28,956	no	2.24	12,668	4,749	233,566
CFV+season+area	6500	18	14,270	17,589	1.07	28,576	yes	2.84	10,042	4,596	167,763
CFV +area+ season within area	6500	22	14,165	17,490	1.07	28,374	yes	3.6	9,311	4,766	136,529
Area + CFV within area +season within area	6500	36	14,005	17,124	1.06	28,082	no	3.6	8,984	4,721	136,392
individual q's	**6500**	**52**	**13,528**	**16,865**	**1.06**	**27,160**	**yes**	**3.24**	**10,788**	**4,798**	**112,920**

Table 7. Number of porbeagle sharks tagged in the USA and Canadian tagging programs between 1980 and 1999 and the number and timing of recaptures of porbeagle that were tagged when under 125 cm in fork length.

Year		Number Tagged		Year recaptured																			
Tagged		Total	<125cm FL	82	83	84	85	86	87	88	89	90	91	92	93	94	95	96	97	98	99	00	**Total**
USA	80		1																				0
	81		5																				0
	82		5			1																	1
	83		3																				0
	84		5																				0
	85		7																				0
	86		20					1		1				1									3
	87	83	25						1								1						2
	88	55	11								2				1								3
	89		1											1									1
	90		0																				0
	91		10										1										1
	92		16												1	1	1			1			4
	93	132	106													3	2	1	5	2	3		16
	94	135	131													2	10	6	6	4	5	2	35
	95	118	116															4	3		4		11
	96	44	37																2		1		3
	97	135	99																1		4	2	7
	98	91	53																		1	1	2
	99	151	148																		3	2	5
USA Total		944	799	0	0	1	0	1	1	1	2	0	1	2	2	6	14	11	17	7	21	7	94
Canada	94	40	40														1		3				4
	95	179	179															4	3	6	6		19
	96	37	37																1		1	1	3
	97	23	23																			1	1
	98	5	5																				0
Can. Total		284	284													0	1	4	7	6	7	2	27

Table 8. Constants used in the assessment model.

Component	Parameter	Females	Males
Growth (Von Bertalanffy)	L_∞	309.8	257.7
	t_0	0.061	0.080
	k	-5.90	-5.78
Growth variability (linear)	b_0	12.05	11.05
	b_1	4E-4	0.0048
Length to weight conversion	b_i	5E-5	5E-5
	b_{ii}	2.713	2.713
Maturity (logistic)	A_{50}	13	7
	a	-14.57	-6.07
	b	1.042	0.75
Age for splitting CPUE		11	12

Table 9. Parameter maximum likelihood estimates and reference points obtained from four models fit to the porbeagle data. The models differ in the assumed natural mortality rate. Reference points are calculated using the selectivity curves for the Shelf Edge fishery.

		Model 1	Model 2	Model 3	Model 4
Differing assumptions:		α estimated	α =2.0	α =2.5	α =3.2
Differing data		CPUE by number imm/mat, stand.	CPUE by weight, integrated	CPUE by weight, integrated	CPUE by weight, integrated
OFV		-3246.1[a]	13,358.7[a]	13,309.1[a]	13,260.6[a]
Spawner - Recruit	Rasym	256,610 (1,017)	284,920 (2,599)	198,970 (1,100)	151,830 (591)
	alpha	2.06 (0.129)	2[b]	2.5[b]	3.2[b]
	SPRF0	0.808	0.808	0.808	0.808
Selectivity (Basin)	B_SfullF	2 (bound)	2 (bound)	2 (bound)	2 (bound)
	B_SfullM	2.017 (0.263)	2.097 (0.009)	2.093 (0.84)	2.074 (0.748)
	B_varLestF	0.4579 (0.068)	0.455 (0.066)	0.45 (0.066)	0.444 (0.065)
	B_varLestM	1.852 (1.0657)	2.156 (0.447)	2.117 (3.448)	2.017 (2.959)
	B_varRestF	44.096 (3.812)	30.295 (2.304)	32.013 (2.565)	33.883 (2.794)
	B_varRestM	236.92 (23.984)	185.38 (1.77)	203.48 (32.728)	229.1 (36.876)
Selectivity (N.-Gulf)	N_SfullF	9.882 (0.201)	9.916 (0.222)	10.109 (0.226)	10.348 (0.232)
	N_SfullM	16.657 (0.463)	14.754 (0.513)	15.396 (0.682)	15.848 (0.006)
	N_varLestF	31.260 (1.941)	33.375 (2.288)	33.668 (2.255)	34.251 (2.249)
	N_varLestM	77.528 (4.861)	62.672 (5.084)	66.071 (6.629)	67.134 (1.224)
	N_varRestF	24.511 (2.282)	25.108 (2.641)	24.605 (2.7)	24.017 (2.794)
	N_varRestM	0.335 (0.890)	6.698 (3.121)	3.821 (3.424)	2.121 (0.261)
Selectivity (Shelf Edge)	E_SfullF	2.077 (0.004)	2 (bound)	2 (bound)	2 (bound)
	E_SfullM	13.396 (0.242)	13.057 (2.506)	13.526 (0.164)	13.615 (0.102)
	E_varLestF	0.434 (0.053)	0.49 (0.057)	0.469 (0.056)	0.455 (0.055)
	E_varLestM	187.35 (12.922)	177.38 (61.782)	178.57 (12.985)	171.72 (11.464)
	E_varRestF	60.923 (0.059)	52.94 (1.596)	56.241 (1.729)	60.196 (1.914)
	E_varRestM	0.100 (0.004)	0.277 (1.947)	0.1 (bound)	0.1 (bound)
Catchability (Basin)	logqCPUE	-15.187 (0.063)			
Catchability (N.-Gulf)	logqCPUE	-15.410 (0.061)	-8.573 (0.065)[c]	-8.527 (0.065)[c]	-8.473 (0.06)[c]
Catchability (Shelf Edge)	logqCPUE	-15.076 (0.058)			
Tag loss/mort.	proportion	0.700 (0.028132)	0.586 (0.039)	0.596 (0.038)	0.603 (0.037)
Reference Population Sizes	SSNeq	82,772 (328.32)	87,754 (800)	81,181 (448)	75,230 (293)
	20% SSNeq	16,554 (65.665)	17,551 (160)	16,236 (89)	15,046 (59)
	Req	102,440 (406.34)	108,610 (991)	100,470 (555)	93,107 (363)
	SSNmsy	37,586	40,663	40,676	29,382
Reference Fishing Mortality Rates	Fmsy	0.033	0.033	0.048	0.065
	Fcol	0.069	0.069	0.099	0.134
	F35%	0.142	0.153	0.153	0.145
	F45%	0.124	0.116	0.116	0.110

[a]only models 2, 3 and 4 are comparable; [b]constants (not estimated); [c]average q's for 53 CPUE series

Table 10. Estimates of recent exploitation rates (proportion of vulnerable biomass taken by the fishery).

	Year	Model 1	Model 2	Model 3	Model 4
Basin	2002	0.041 (0.002)	0.022 (0.001)	0.022 (0.001)	0.022 (0.001)
	2003	0.029 (0.002)	0.016 (0.001)	0.016 (0.001)	0.016 (0.001)
	2004	0.017 (0.001)	0.009 (0.000)	0.009 (0.001)	0.009 (0.000)
NF Gulf	2002	0.041 (0.003)	0.015 (0.001)	0.017 (0.001)	0.019 (0.001)
	2003	0.002 (0.000)	0.001 (0.000)	0.001 (0.000)	0.001 (0.000)
	2004	0.003 (0.000)	0.001 (0.000)	0.001 (0.000)	0.001 (0.000)
Shelf	2002	0.037 (0.002)	0.019 (0.001)	0.019 (0.001)	0.020 (0.001)
	2003	0.029 (0.001)	0.015 (0.001)	0.015 (0.000)	0.018 (0.000)
	2004	0.074 (0.004)	0.037 (0.002)	0.038 (0.001)	0.039 (0.001)

Table 11. Estimates of population size obtained from four models fit to the porbeagle data. See text for model descriptions.

		Model 1	Model 2	Model 3	Model 4
Differing assumptions: Differing data		α estimated CPUE by number imm/mat, stand.	α =2.0 CPUE by weight, integrated	α =2.5 CPUE by weight, integrated	α =3.2 CPUE by weight, integrated
1961	SSN	82,772 (328)	87,754 (800)	81,181 (448)	75,230 (371)
	N	876,150 (3,475)	928,880 (8,473)	859,300 (4,750)	796,310 (3,105)
1971	SSN	25,880 (262)	32,706 (452)	25,551 (335)	19,413 (259)
	N	368,280 (3,090)	429,580 (7,228)	371,480 (4,207)	318,660 (3,715)
1981	SSN	28,657 (254)	35,031 (439)	28,649 (338)	22,862 (257)
	N	318,890 (2,841)	385,650 (5,058)	342,620 (3,987)	302,760 (3,209)
1991	SSN	23,715 (266)	30,436 (436)	26,159 (362)	22,252 (294)
	N	323,830 (3,209)	397,370 (5,299)	375,110 (4,410)	355,190 (3,702)
2002	SSN	7,534.1 (297)	15,007 (512)	12,531 (426)	10,376 (355)
	N	102,390 (4,363)	198,040 (6,226)	190,300 (5,741)	184,450 (5,273)
2005	SSN	5,519.6 (290.52)	12,945 (540)	11,013 (436)	9,371 (371)
	N	94,309 (4,550.9)	195,230 (6,609)	190,520 (6,197)	187,960 (5,823)
2005/1961	SSN	0.066 (0.003)	0.148 (0.006)	0.136 (0.005)	0.120 (0.005)
	N	0.107 (0.005)	0.21 (0.007)	0.222 (0.007)	0.236 (0.007)
2005/2002	SSN	0.732 (0.010)	0.863 (0.008)	0.879 (0.006)	0.903 (0.006)
	N	0.921 (0.005)	0.986 (0.004)	1.001 (0.003)	1.019 (0.003)

Table 12. Estimates of the mid-year vulnerable biomass (metric tonnes) for 2005 from the four models and three fishery selectivities. Note that the vulnerable biomass is conditional on the selectivity and, given a selectivity is applicable to the entire population. The values do not apply separately to each region.

	Model 1	Model 2	Model 3	Model 4
Differing assumptions:	σ estimated	$\sigma = 2.0$	$\sigma = 2.5$	$\sigma = 3.2$
Differing data	CPUE by number imm/mat, stand.	CPUE by weight, integrated	CPUE by weight, integrated	CPUE by weight, integrated
Biomass removed using:				
Basin selectivity	2,476.7 (115.56)	4,645.3 (156.69)	4,663.2 (275.94)	4,720.8 (233.54)
NF Gulf selectivity	1,553.8 (107.05)	3,972.5 (160.85)	3,661.6 (154.81)	3,431.2 (124.66)
Shelf selectivity	2,299.7 (116.9)	4,626.3 (263.97)	4,526 (147.24)	4,502.1 (582.26)

Table 13. Comparison of total mortality rates obtained from the four models with those of Campana et al. 2001.

Age and Year	Campana et al. 2001	Model 1	Model 2	Model 3	Model 4
$Z_{3\text{-}5,\ 1999 \text{ to } 4\text{-}6,\ 2000}$	0.27	0.365 (0.009)	0.254 (0.005)	0.259 (0.004)	0.262 (0.004)
$Z_{6\text{-}8,\ 1998 \text{ to } 7\text{-}9,\ 1999}$	0.35	0.402 (0.009)	0.282 (0.005)	0.292 (0.005)	0.301 (0.005)
$Z_{6\text{-}8,\ 1999 \text{ to } 7\text{-}9,\ 2000}$	0.32	0.405 (0.011)	0.273 (0.005)	0.281 (0.005)	0.289 (0.005)
$Z_{9\text{-}12,\ 1998 \text{ to } 10\text{-}13,\ 1999}$	0.37	0.447 (0.011)	0.306 (0.006)	0.324 (0.006)	0.342 (0.006)
$Z_{9\text{-}12,\ 1999 \text{ to } 10\text{-}13,\ 2000}$	0.30	0.422 (0.011)	0.281 (0.005)	0.296 (0.005)	0.311 (0.005)

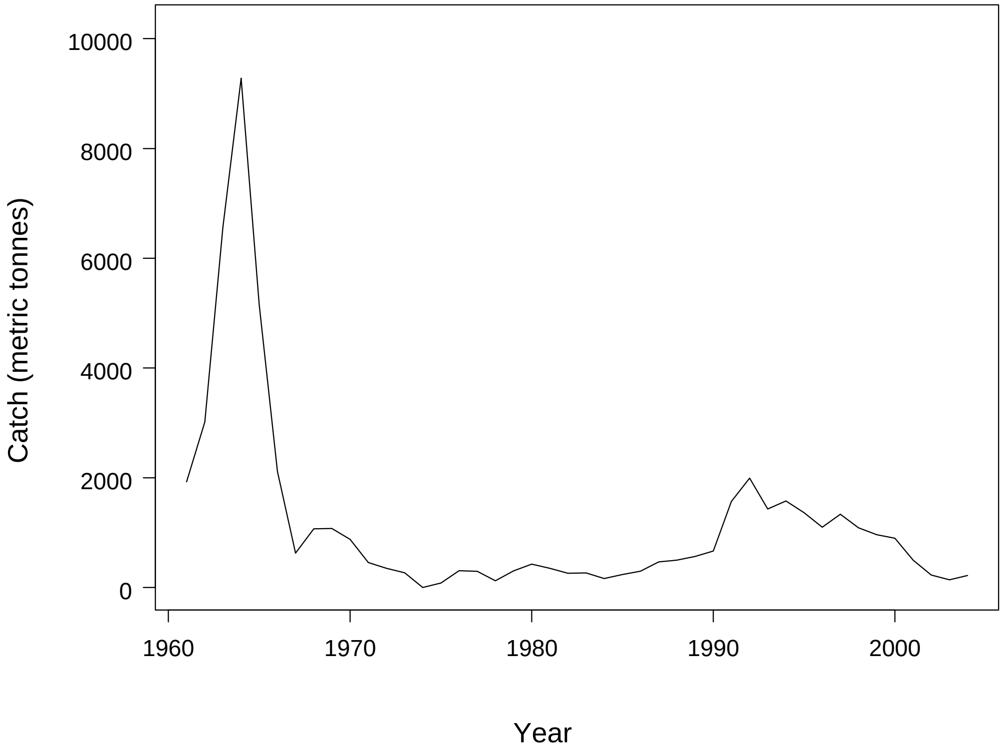


Figure 1. Porbeagle landings in northwest Atlantic (NAFO Areas 2-6) from 1961 to 2005.

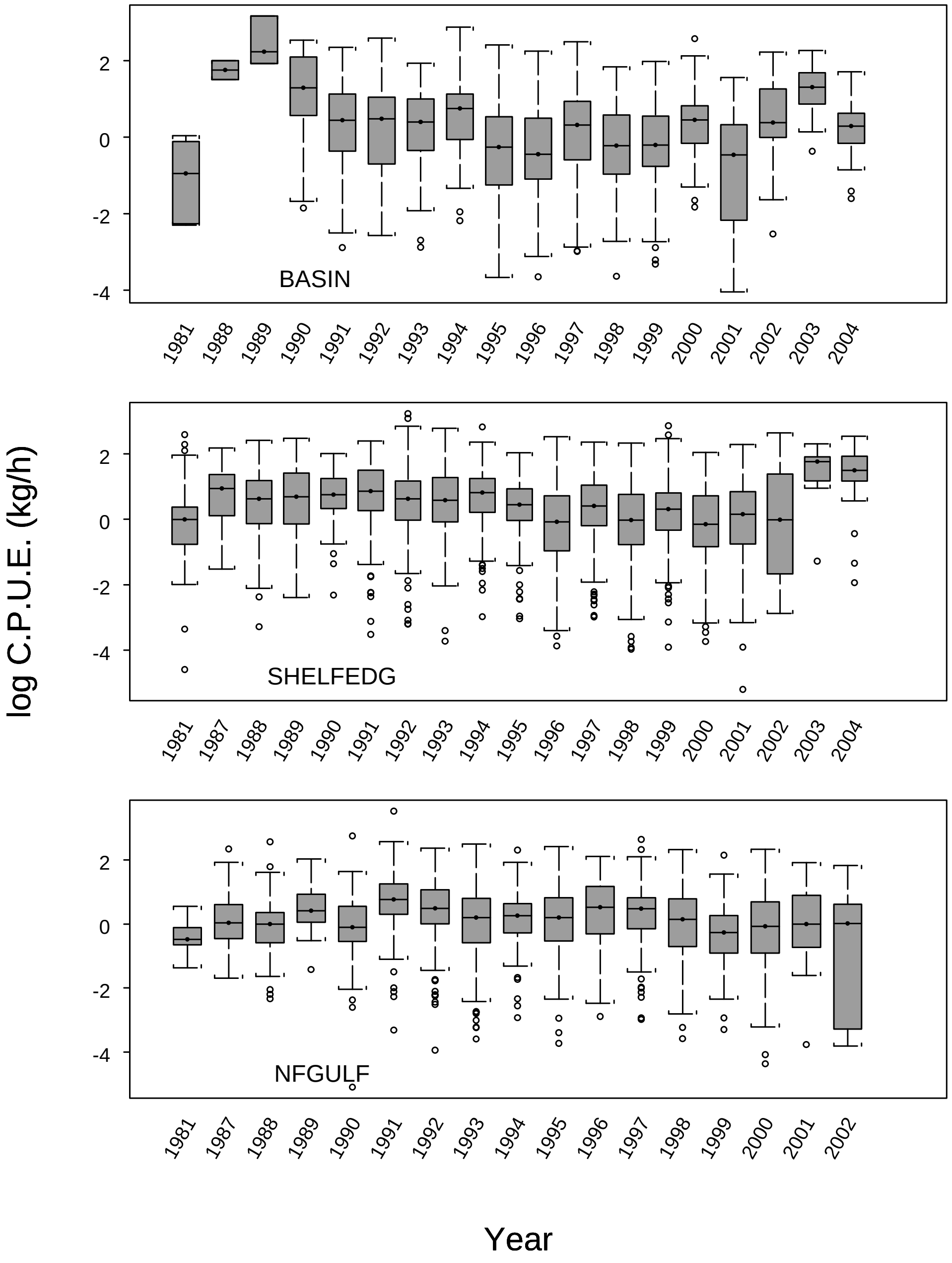


Figure 2. Box plots showing the catch per unit effort (kg/hook) for porbeagle in 3 areas from 1981 to 2004. Note that the years differ between graphs.

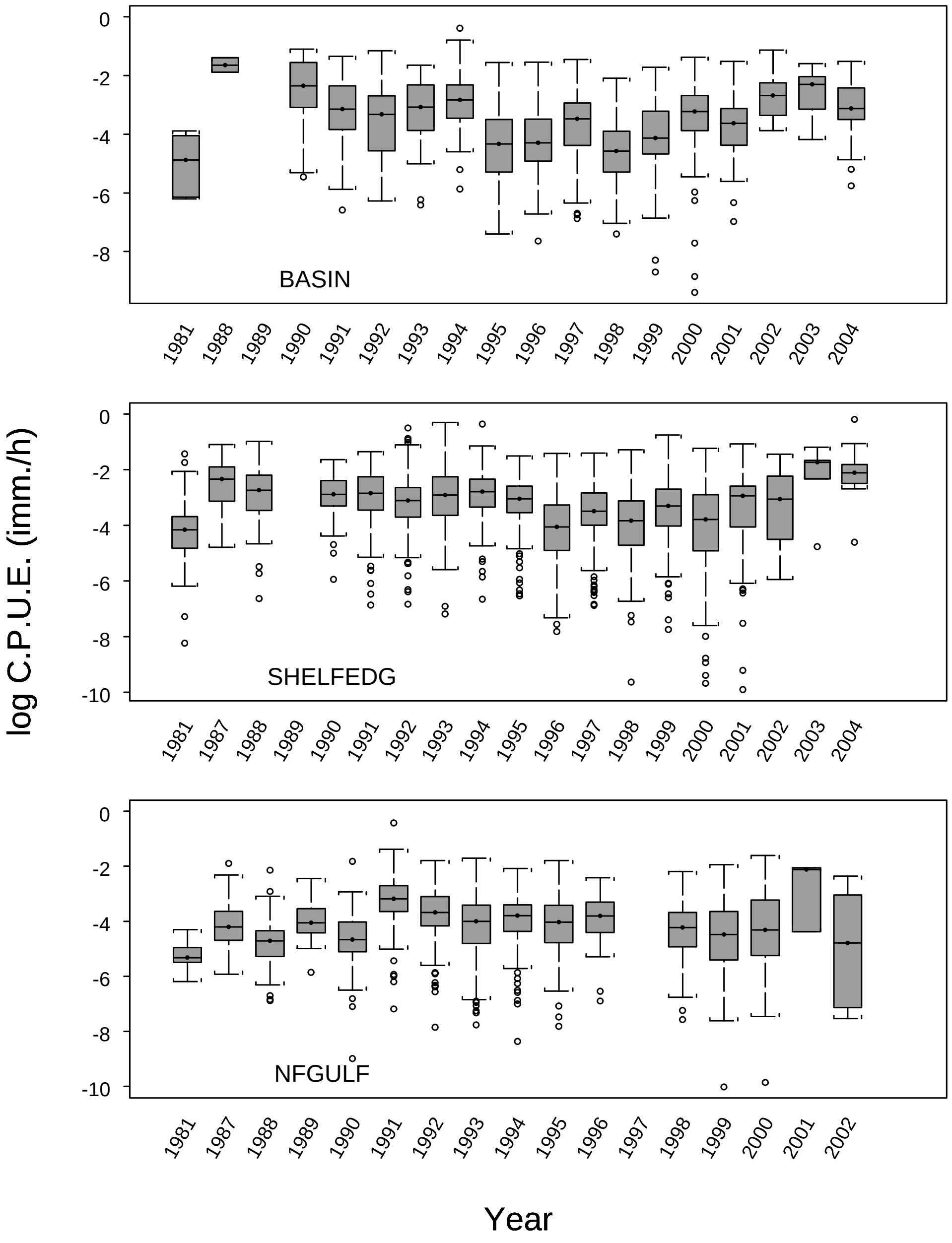


Figure 3. Box plots showing the catch per unit effort (number/hook) for immature porbeagle in 3 areas from 1981 to 2004. Note that the years differ between graphs.

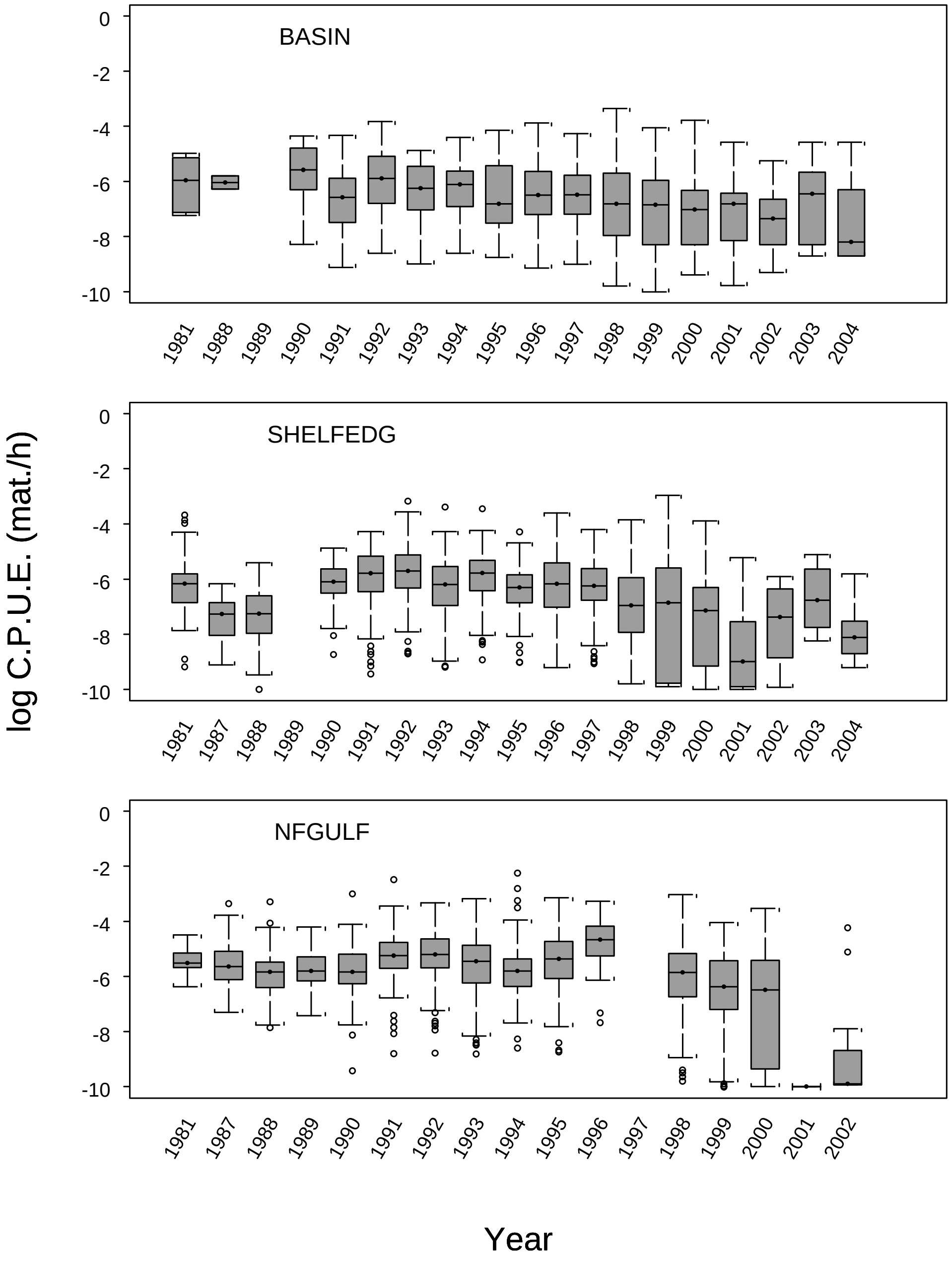


Figure 4. Box plots showing the catch per unit effort (number/hook) for mature porbeagle in 3 areas from 1981 to 2004. Note that the years differ between graphs.

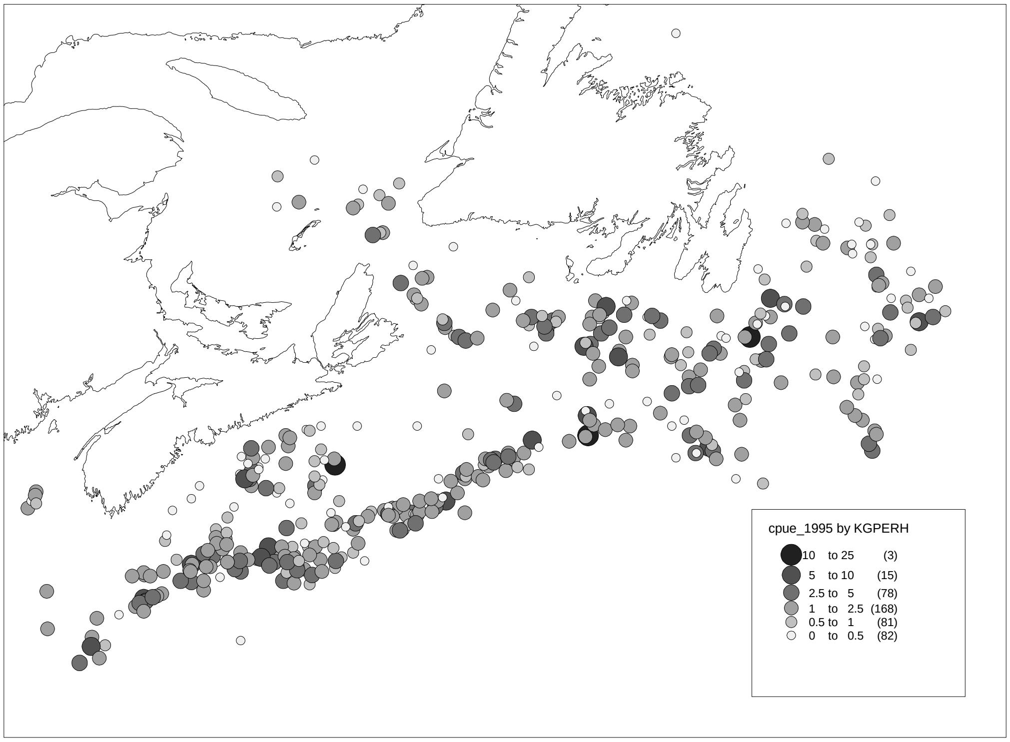


Figure 5.1. Distribution of CPUE in 1995.

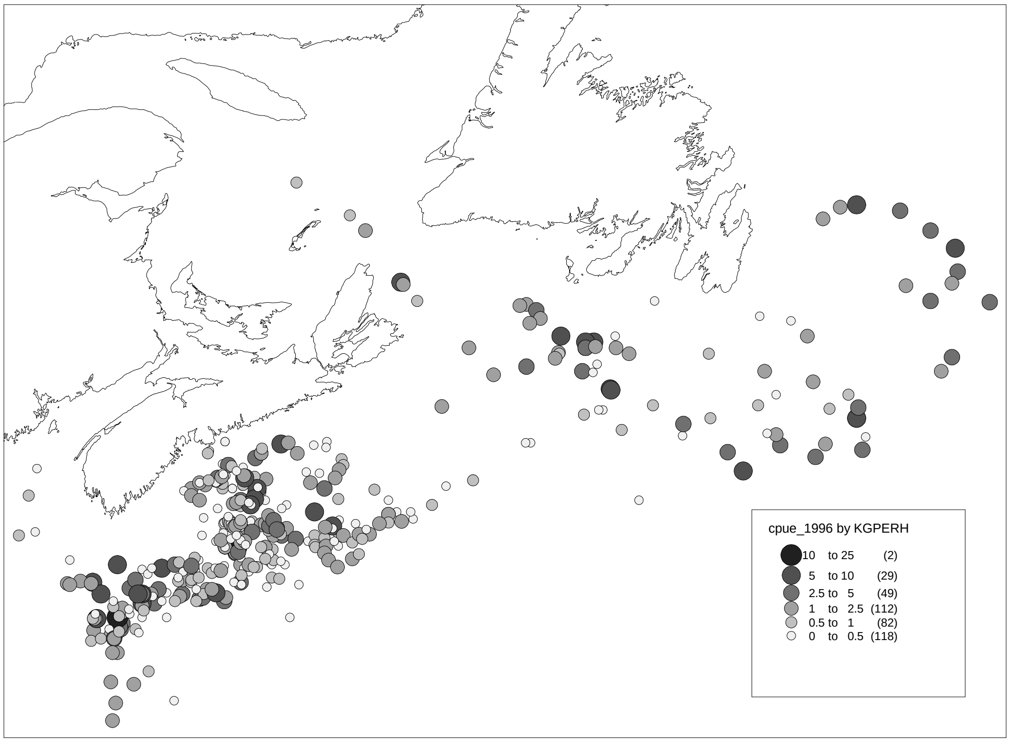


Figure 5.2. Distribution of CPUE in 1996.

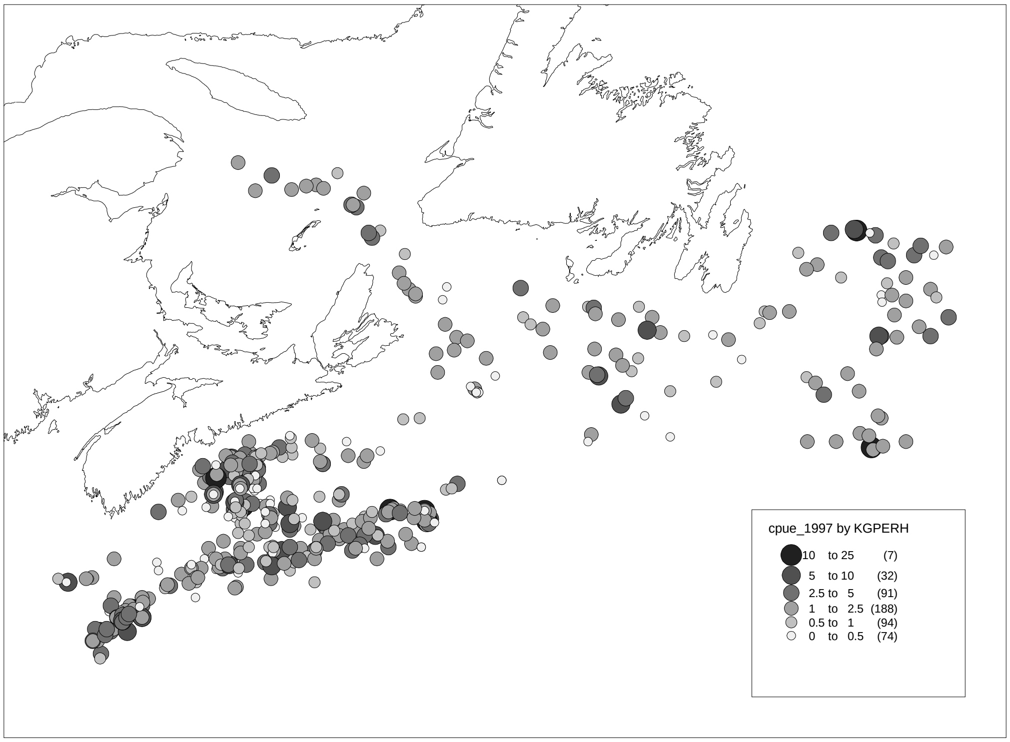


Figure 5.3. Distribution of CPUE in 1997.

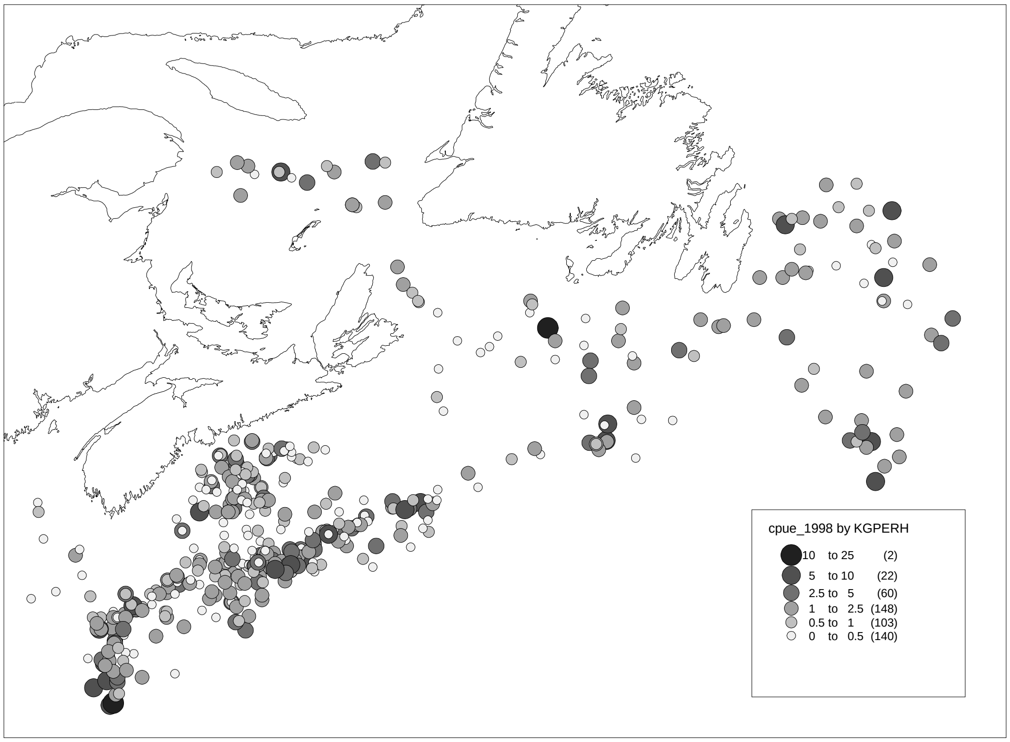


Figure 5.4. Distribution of CPUE in 1998.

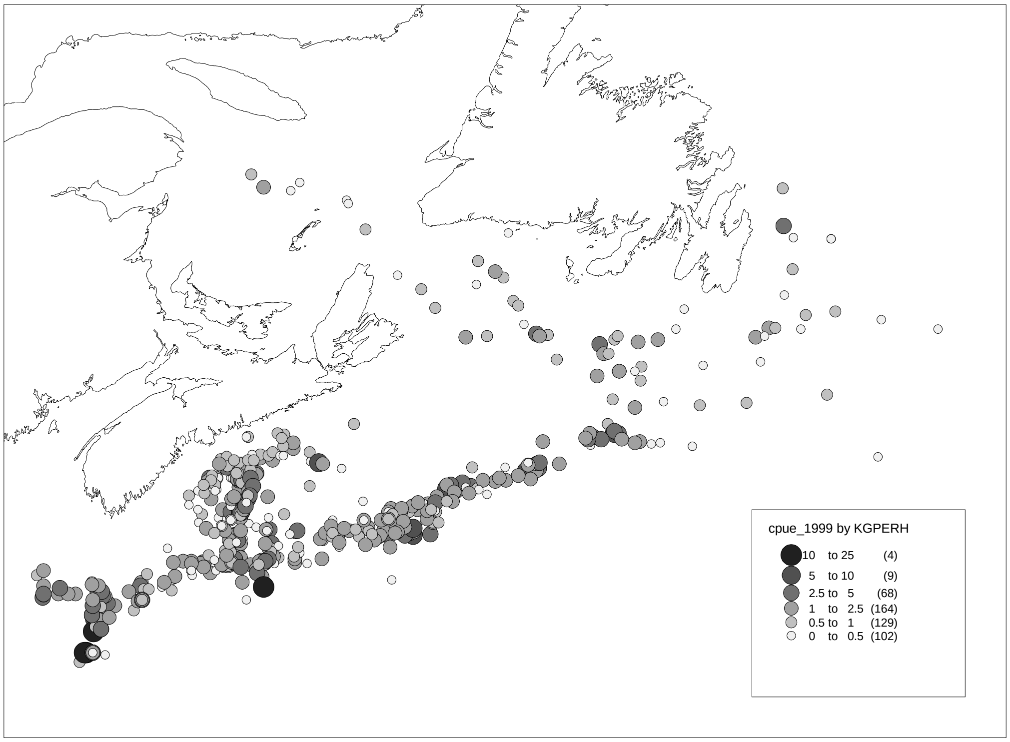


Figure 5.5. Distribution of CPUE in 1999.

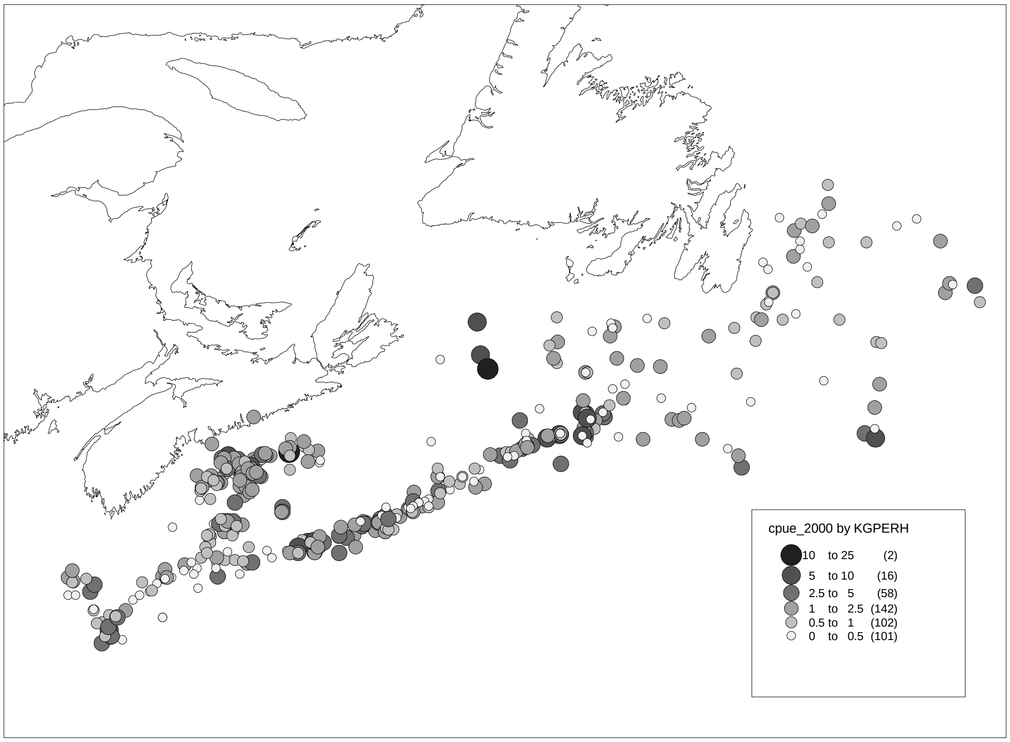


Figure 5.6. Distribution of CPUE in 2000.

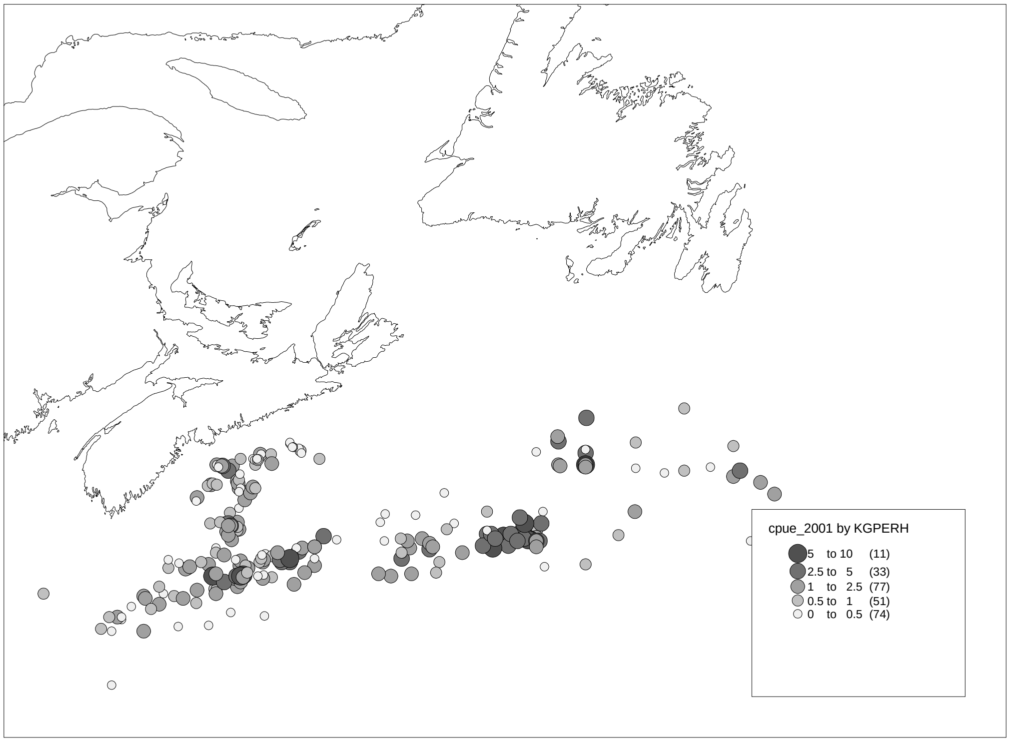


Figure 5.7. Distribution of CPUE in 2001.

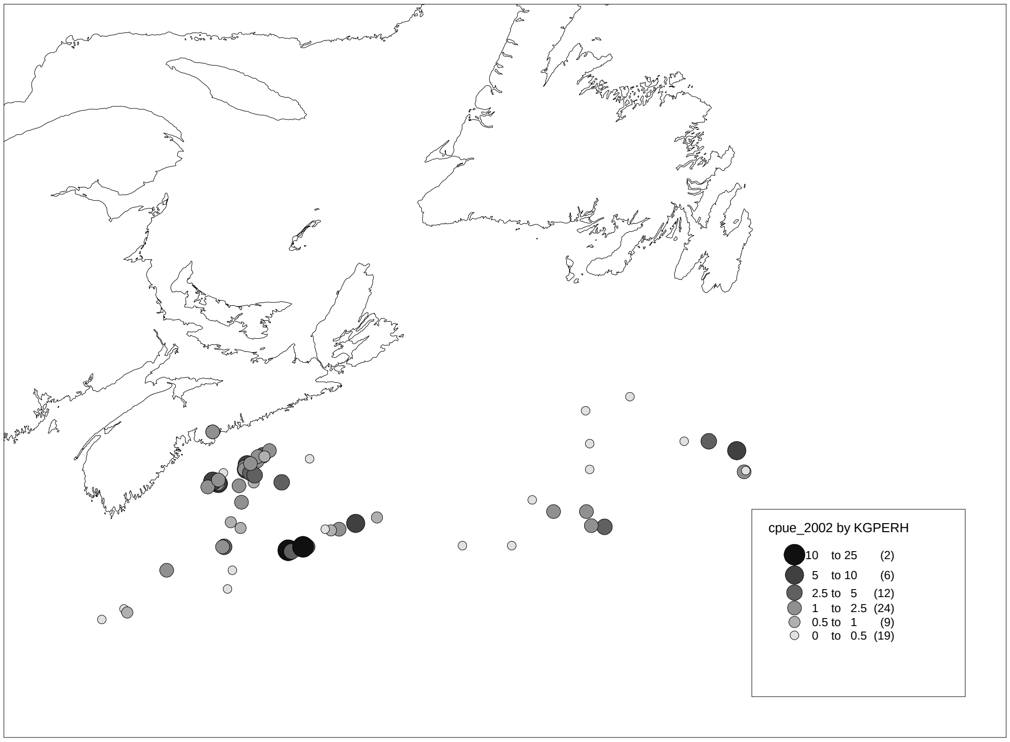


Figure 5.8. Distribution of CPUE in 2002.

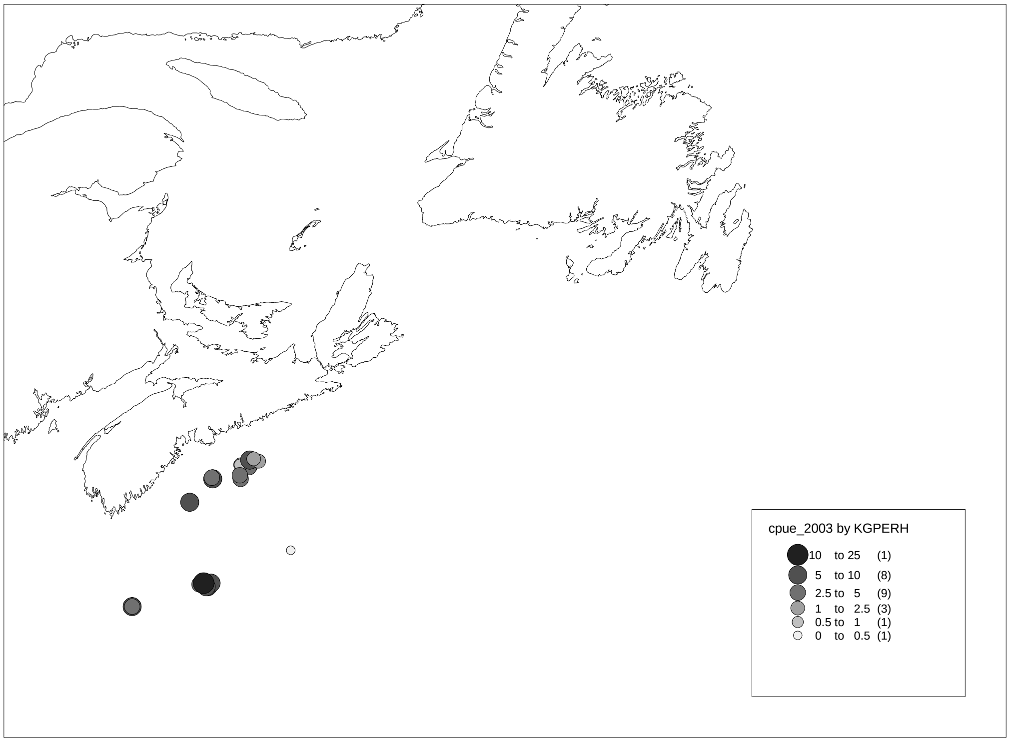


Figure 5.9. Distribution of CPUE in 2003.

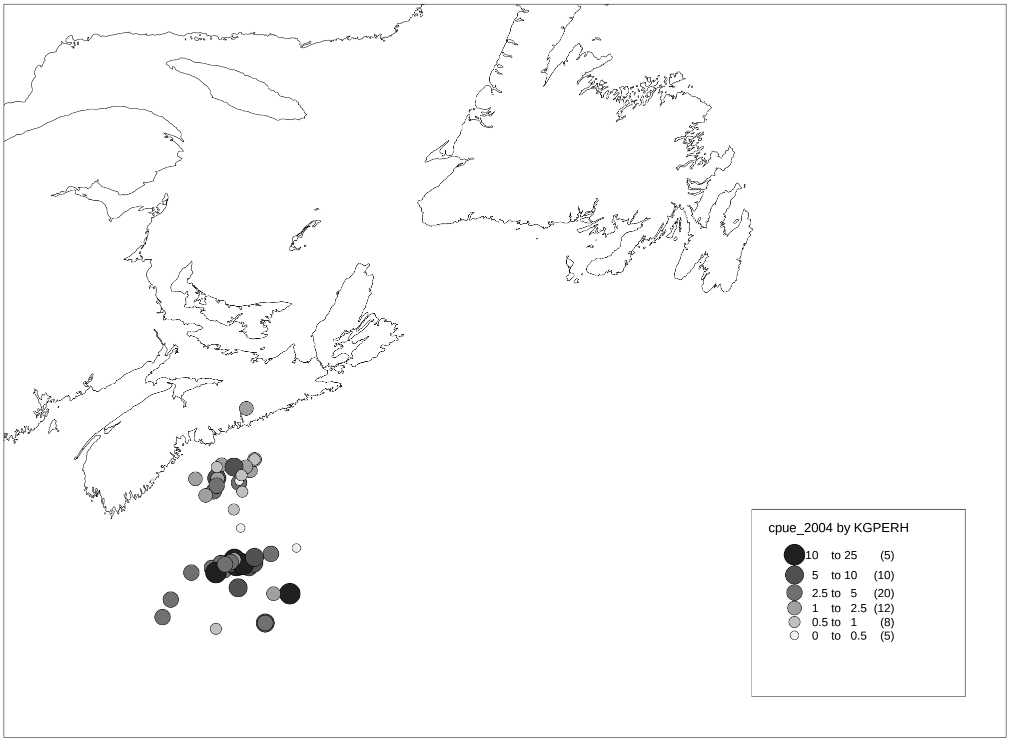


Figure 5.10. Distribution of CPUE in 2004.

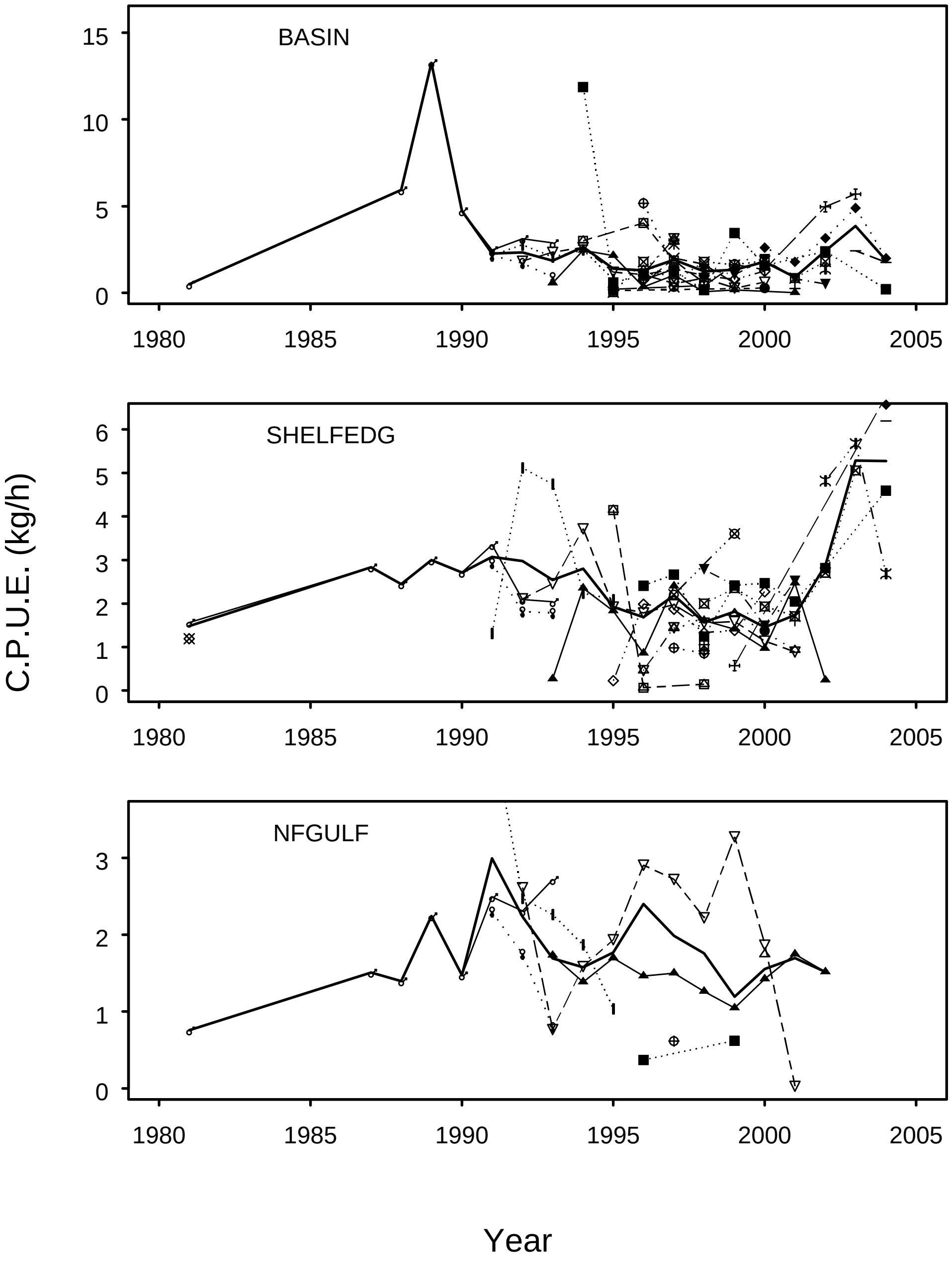


Figure 6. Comparison of the grand mean (solid line) of the CPUE (kg/hook) with the mean CPUE for each vessel.

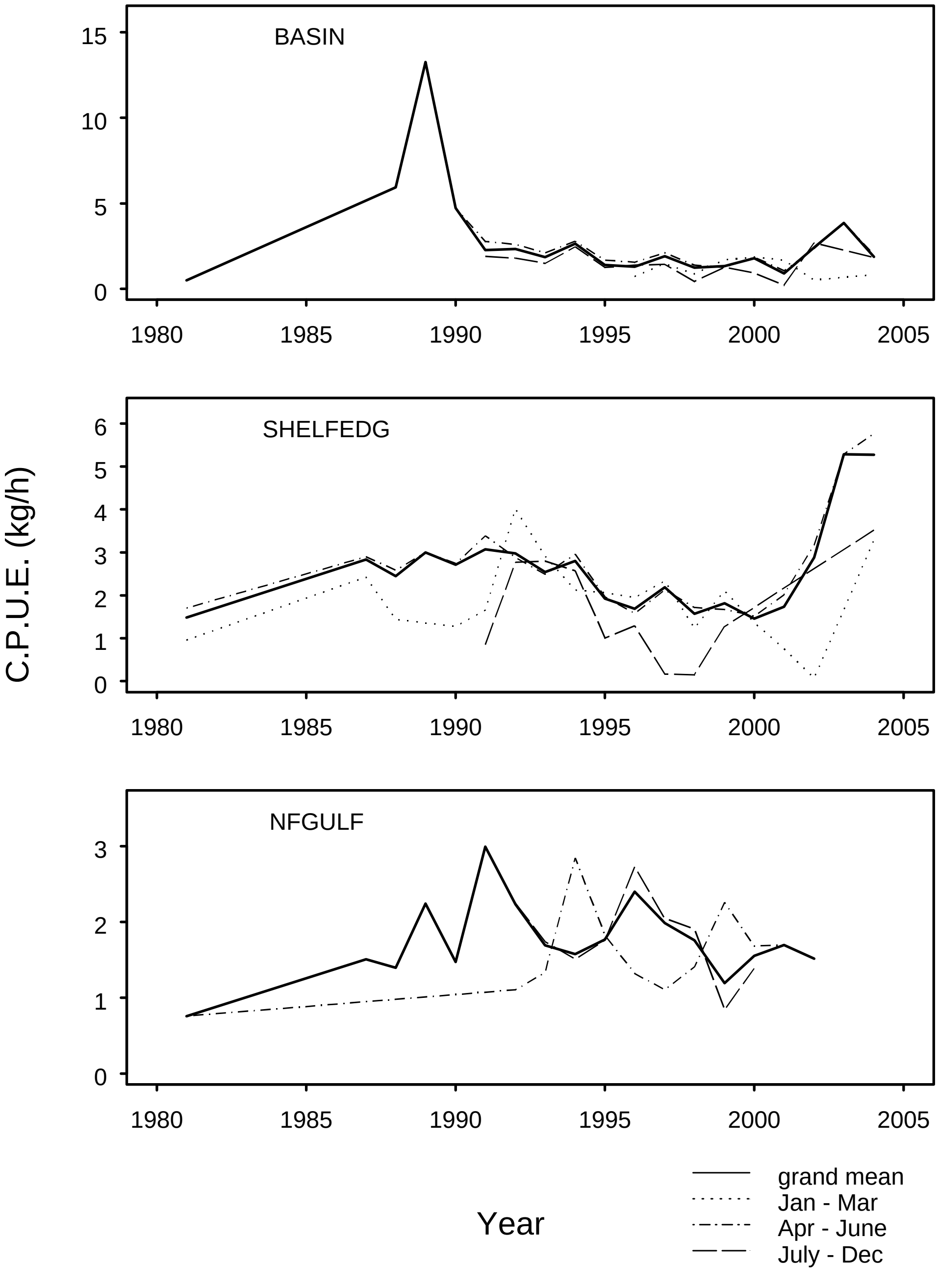


Figure 7. Comparison of the grand mean (solid line) of the CPUE (kg/hook) with the seasonal mean CPUE's.

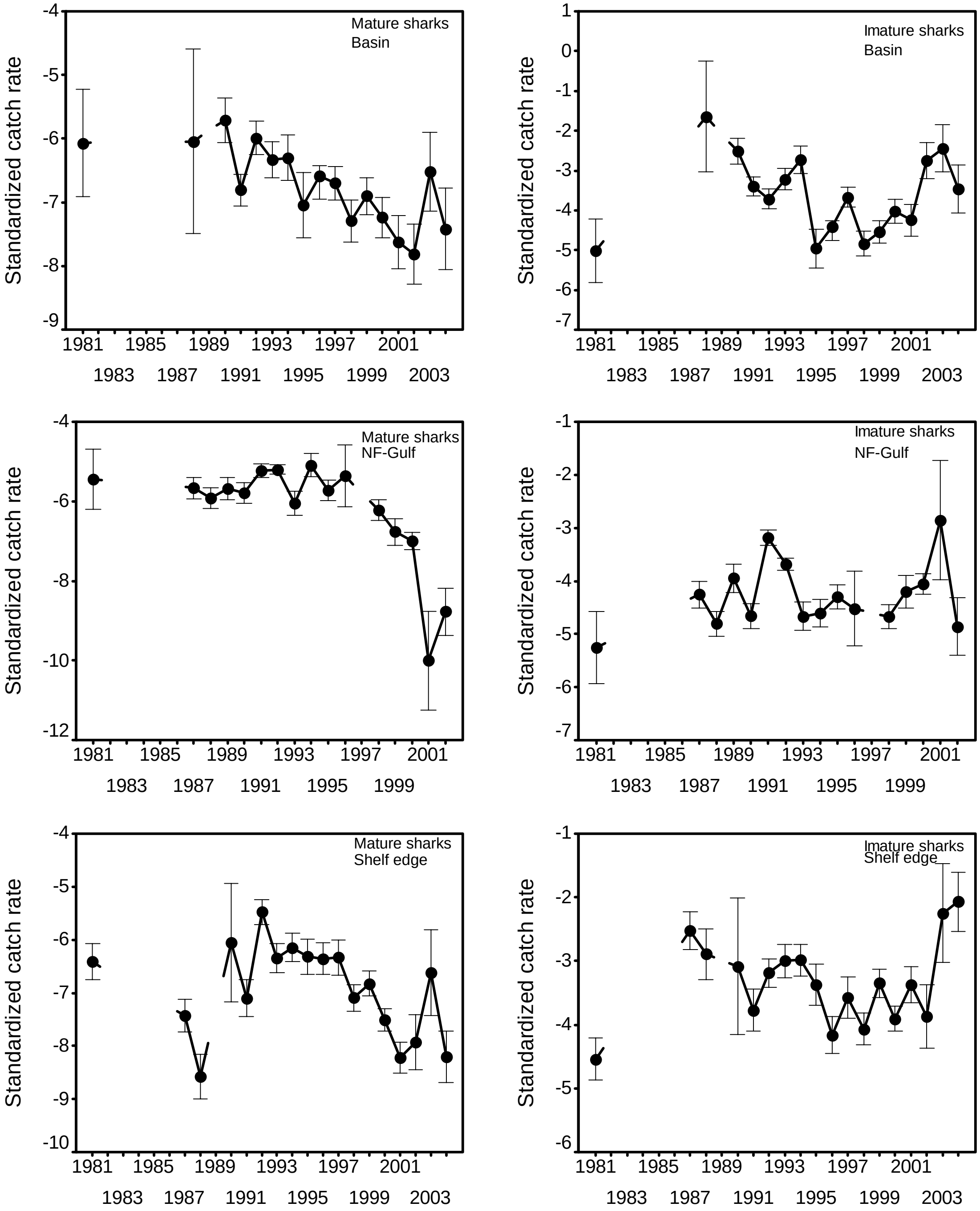


Figure 8. Standardized catch rate (log C.P.U.E) for mature and immature porbeagle in the 3 regions. Error bars are 95% confidence intervals.

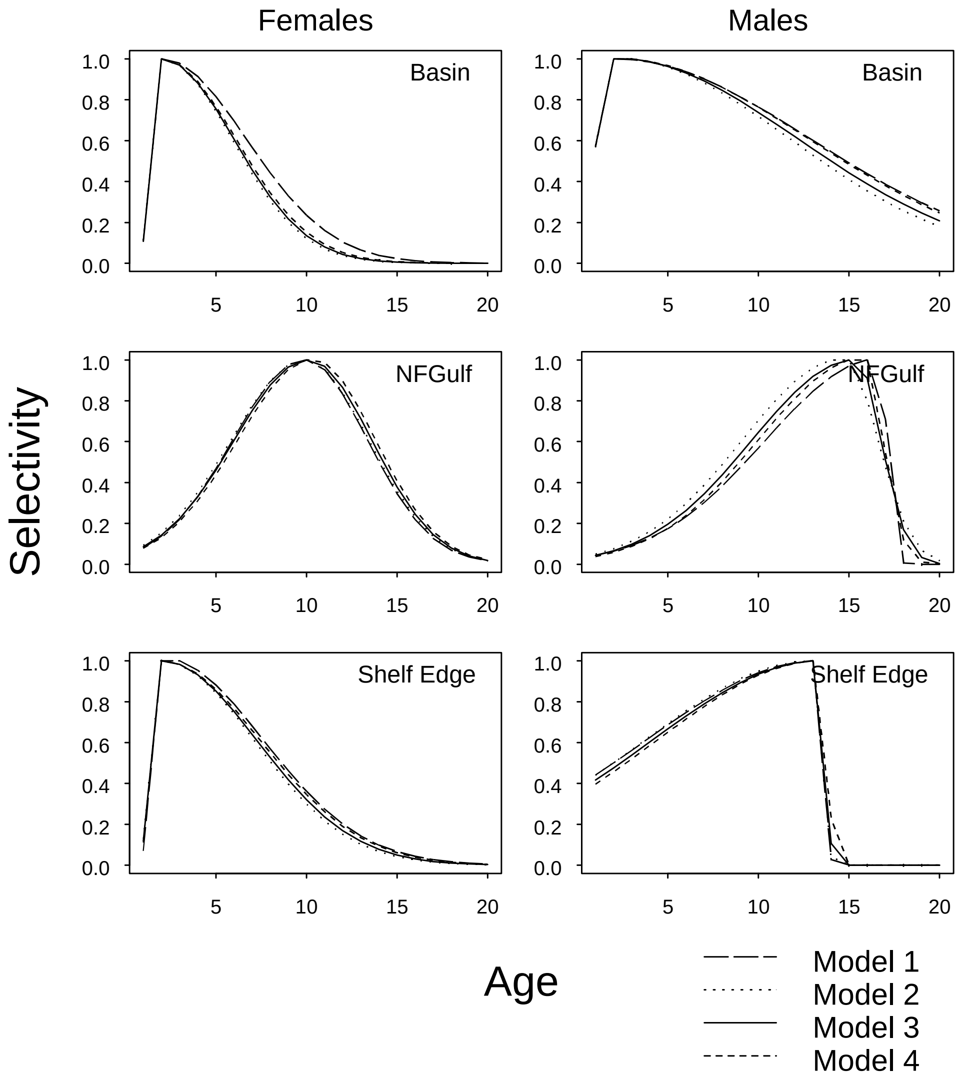


Figure 9. Estimated selectivity curves for porbeagle fisheries in three regions obtained from the four models (see text).

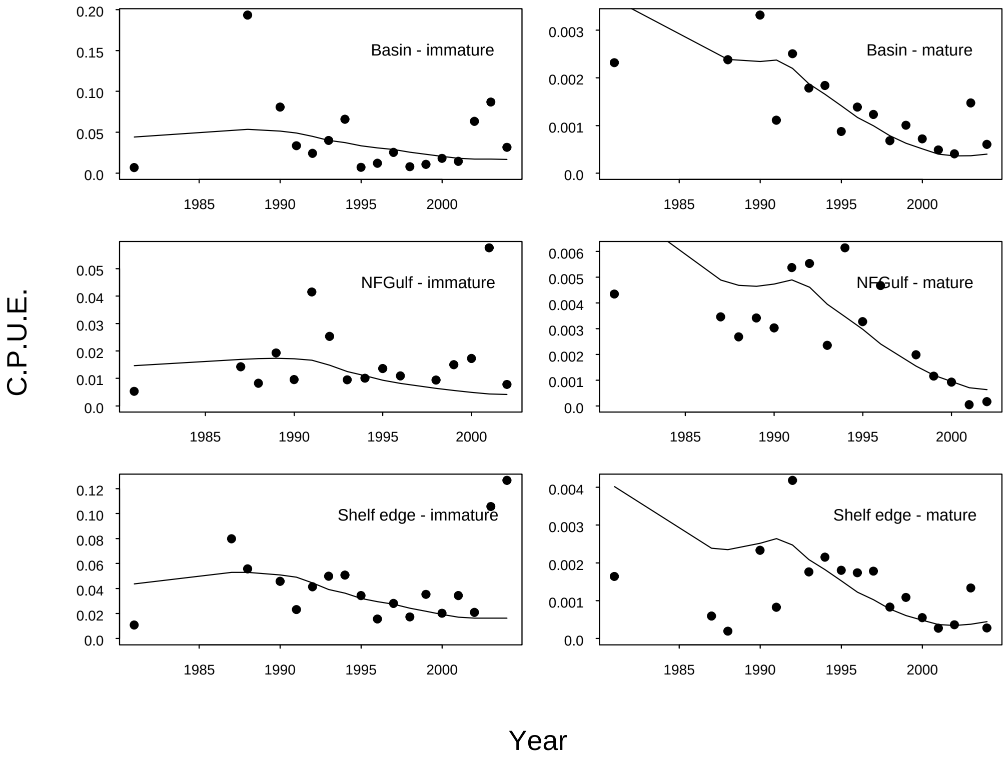


Figure 10. Observed (points) and fitted (lines) standardized (GLM) catch per unit effort for mature and immature porbeagle in each of three regions. The fit is from Model 1.

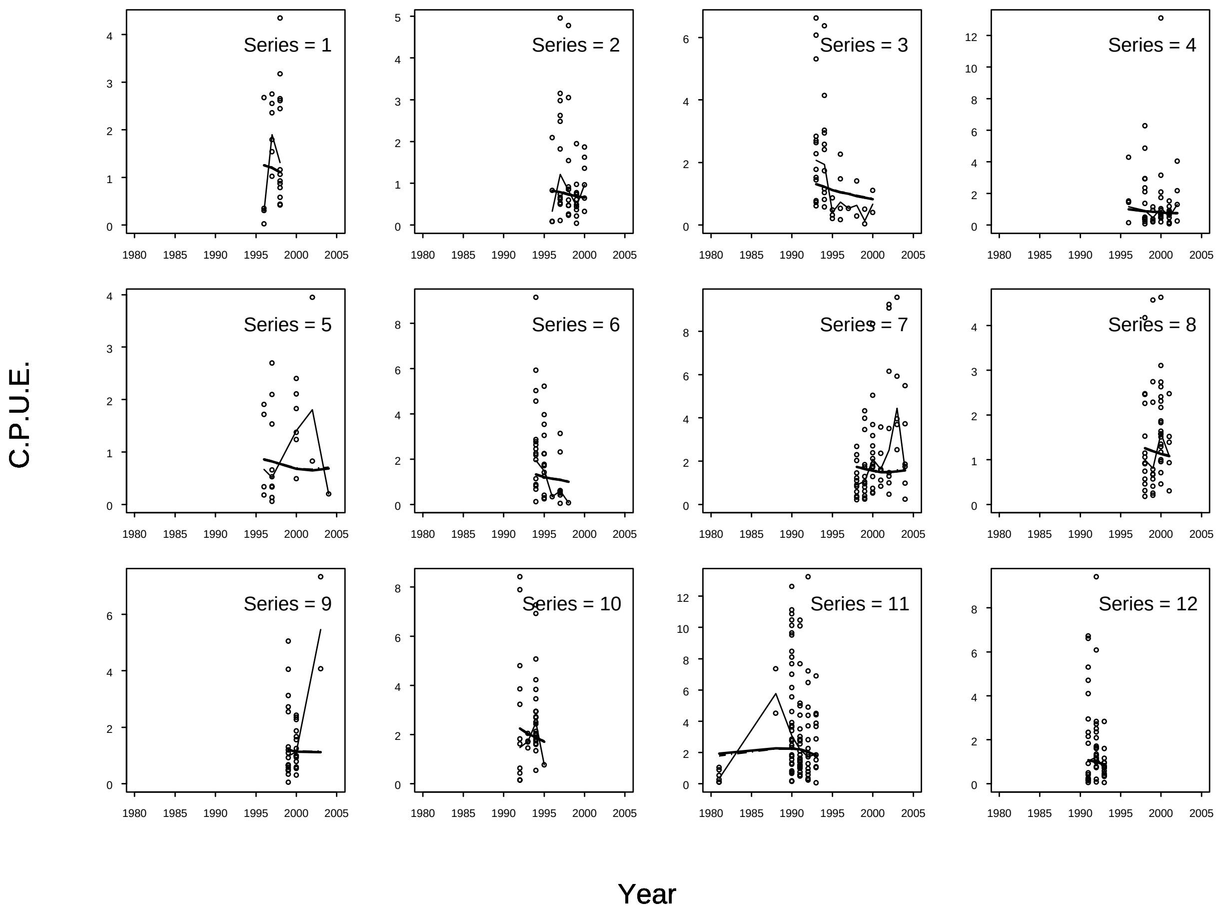


Figure 11.1. Observed (points) and fitted (lines) catch per unit effort by weight for each porbeagle CPUE series (a single vessel fishing in a single area during a single season). The light line is the annual geometric mean of the data. The dark line is the model fit obtained from Model 3.

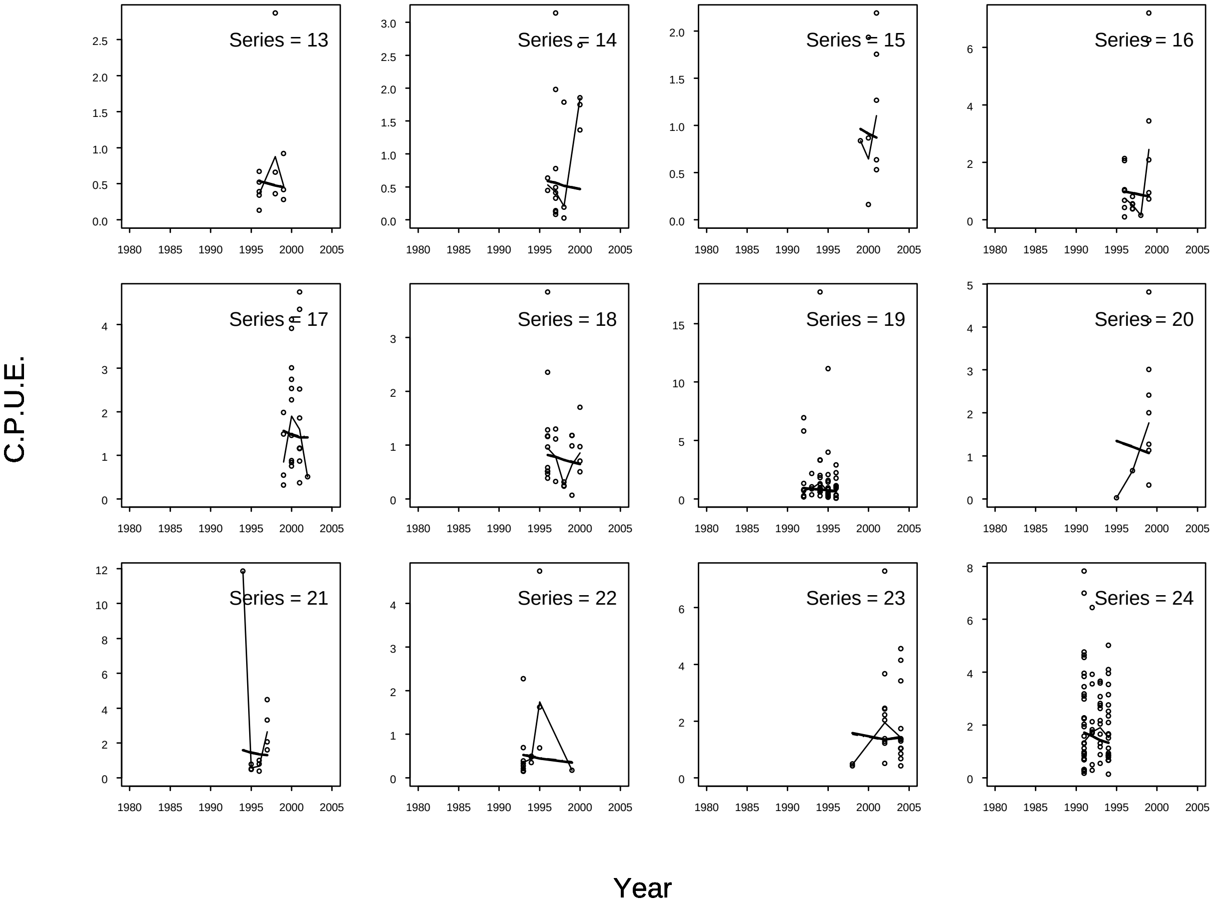


Figure 11.2. Observed (points) and fitted (lines) catch per unit effort by weight for each porbeagle CPUE series (a single vessel fishing in a single area during a single season). The light line is the annual geometric mean of the data. The dark line is the model fit obtained from Model 3.

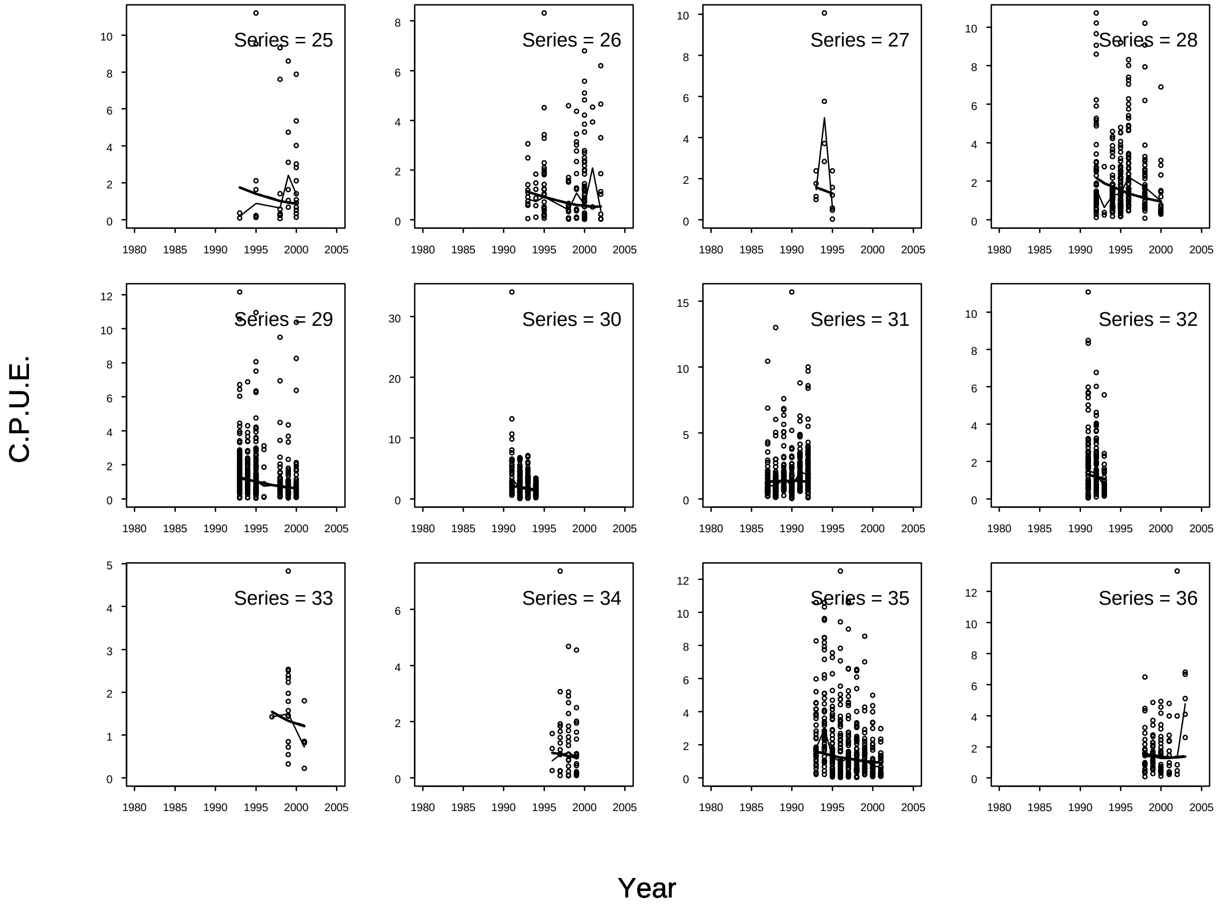


Figure 11.3. Observed (points) and fitted (lines) catch per unit effort by weight for each porbeagle CPUE series (a single vessel fishing in a single area during a single season). The light line is the annual geometric mean of the data. The dark line is the model fit obtained from Model 3.

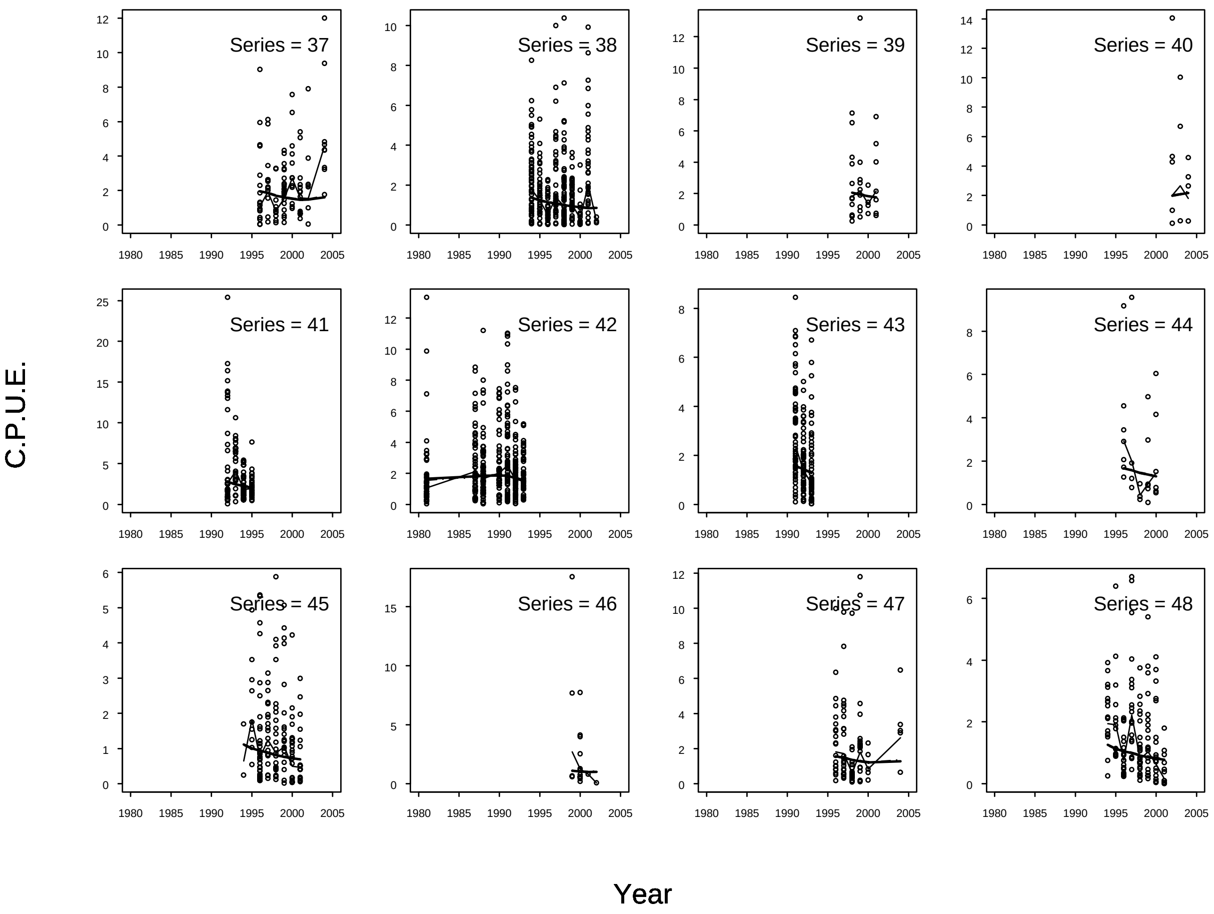


Figure 11.4. Observed (points) and fitted (lines) catch per unit effort by weight for each porbeagle CPUE series (a single vessel fishing in a single area during a single season). The light line is the annual geometric mean of the data. The dark line is the model fit obtained from Model 3.

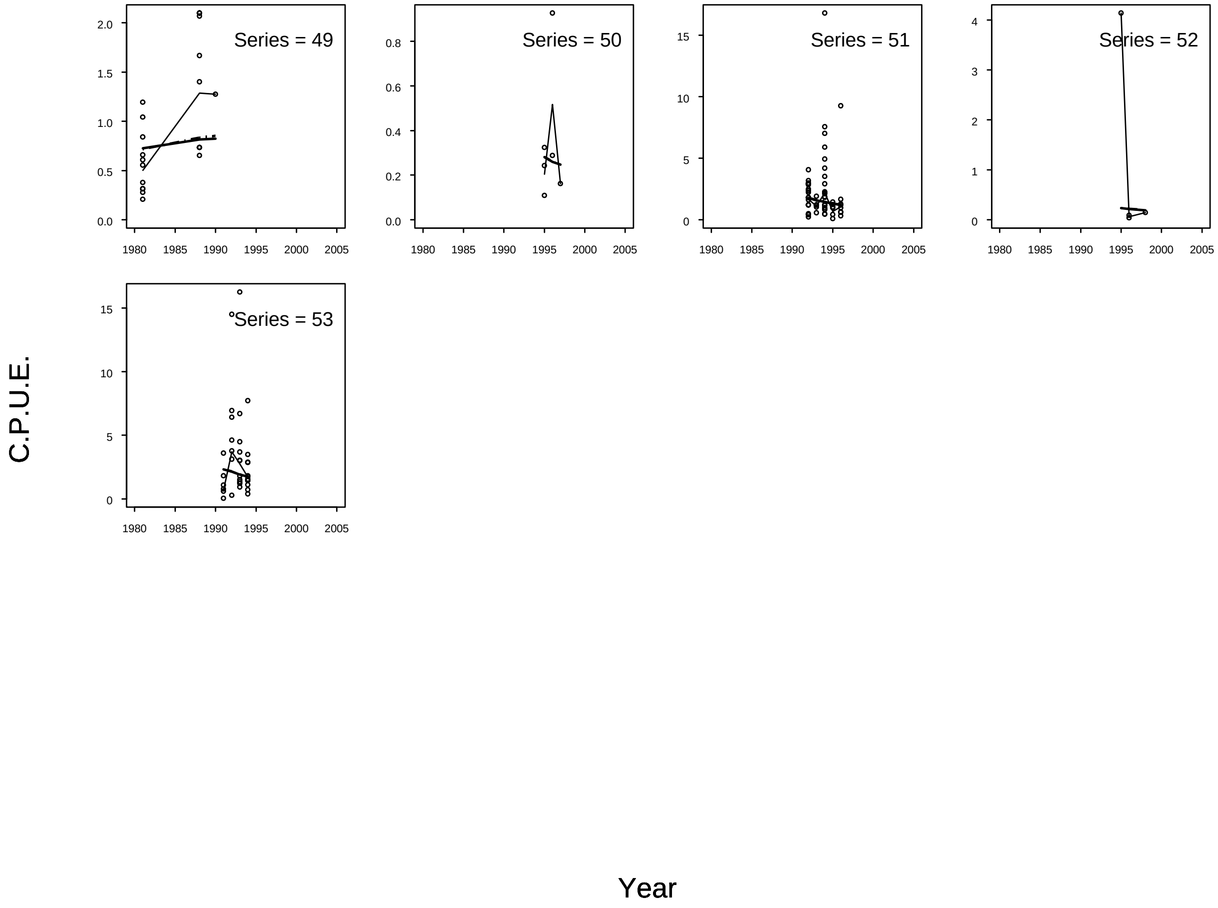


Figure 11.5. Observed (points) and fitted (lines) catch per unit effort by weight for each porbeagle CPUE series (a single vessel fishing in a single area during a single season). The light line is the annual geometric mean of the data. The dark line is the model fit obtained from Model 3.

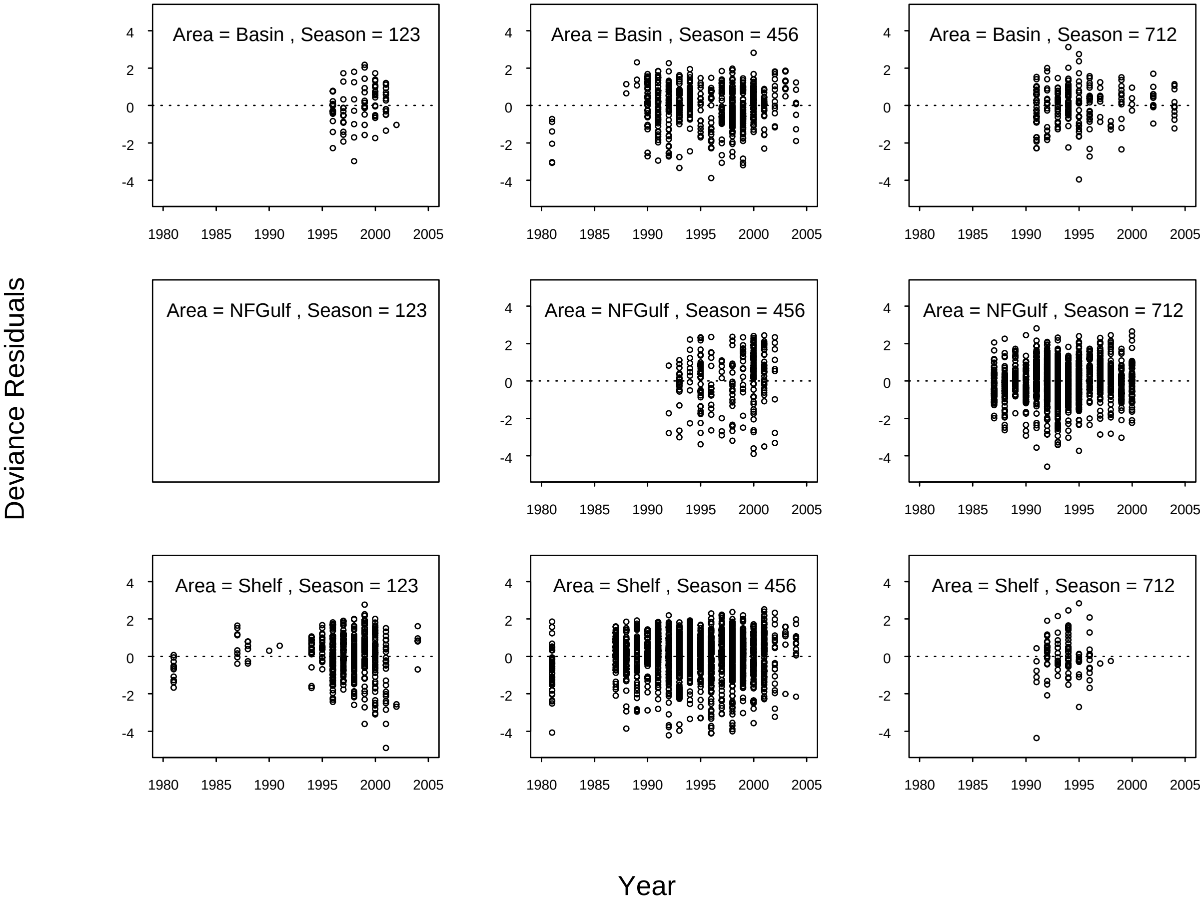


Figure 12. Deviance residuals for the fits to the CPUE data by season and area, obtained from Model 3.

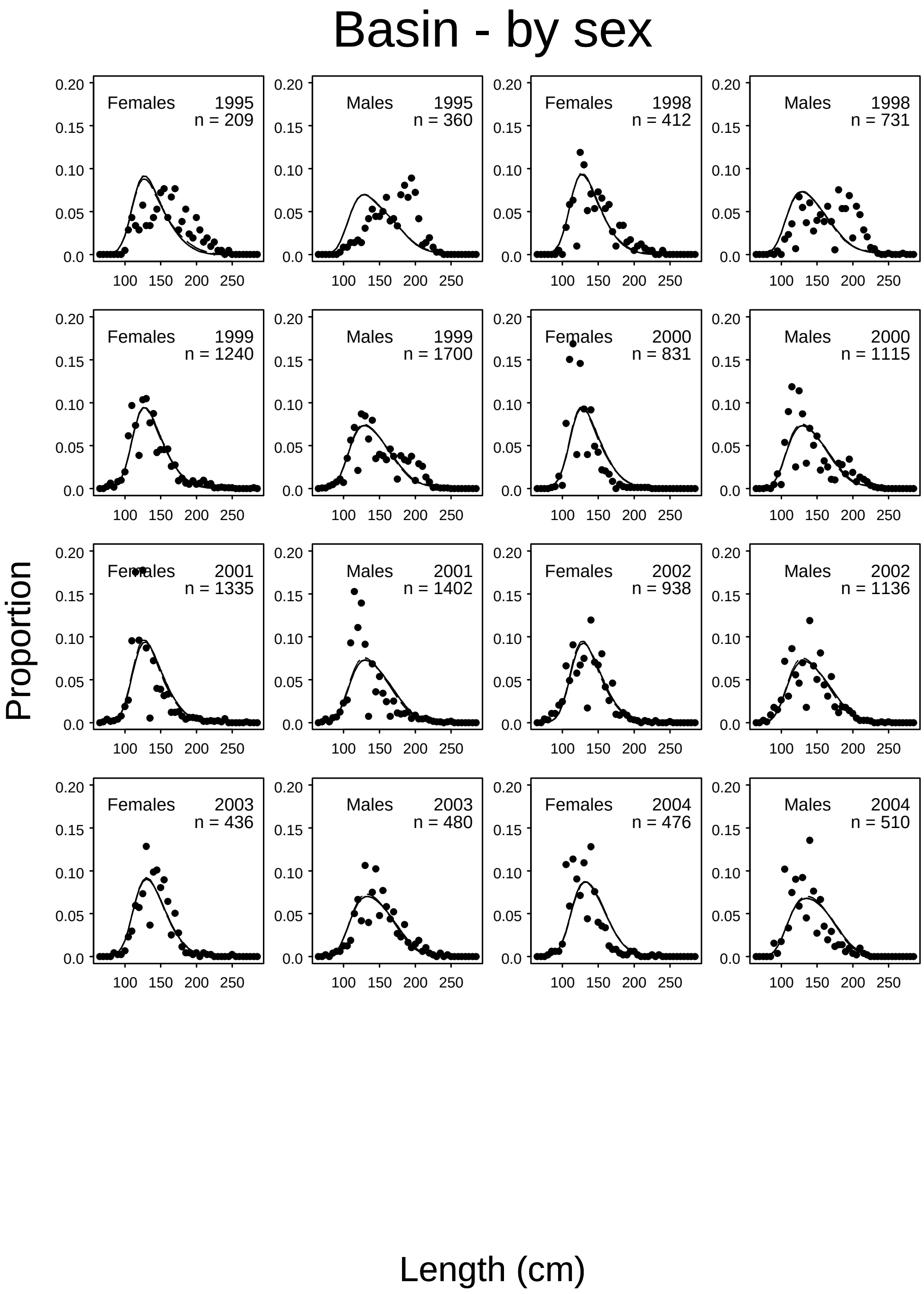


Figure 13.1. Observed (points) and fitted (lines) catch-at-length proportions by sex in the Basin region. Line symbolism is the same as Figure 9. All fits are virtually identical.

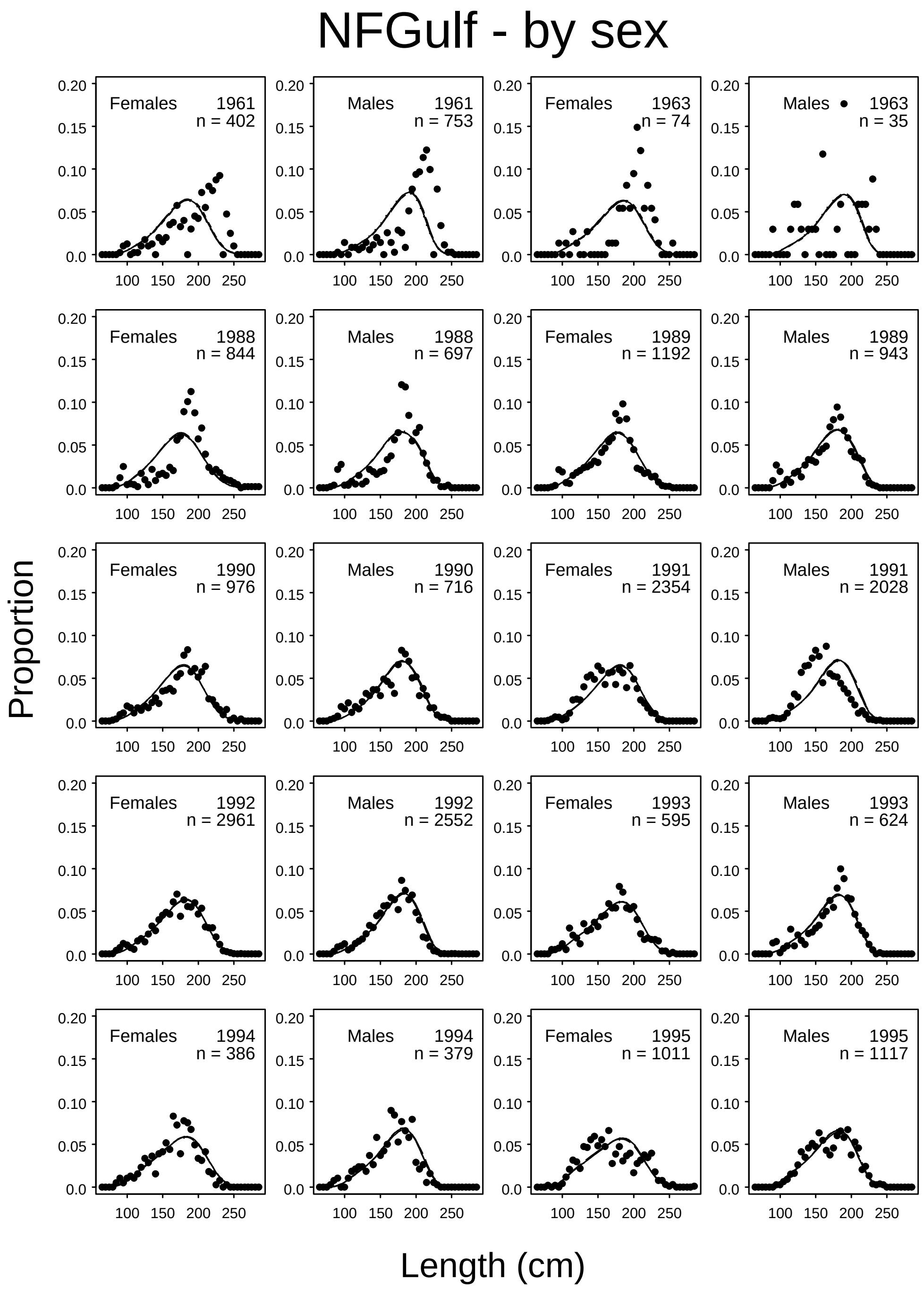


Figure 13.2. Observed (points) and fitted (lines) catch-at-length proportions by sex in the NF-Gulf region up to 1998. Line symbolism is the same as Figure 9. All fits are virtually identical.

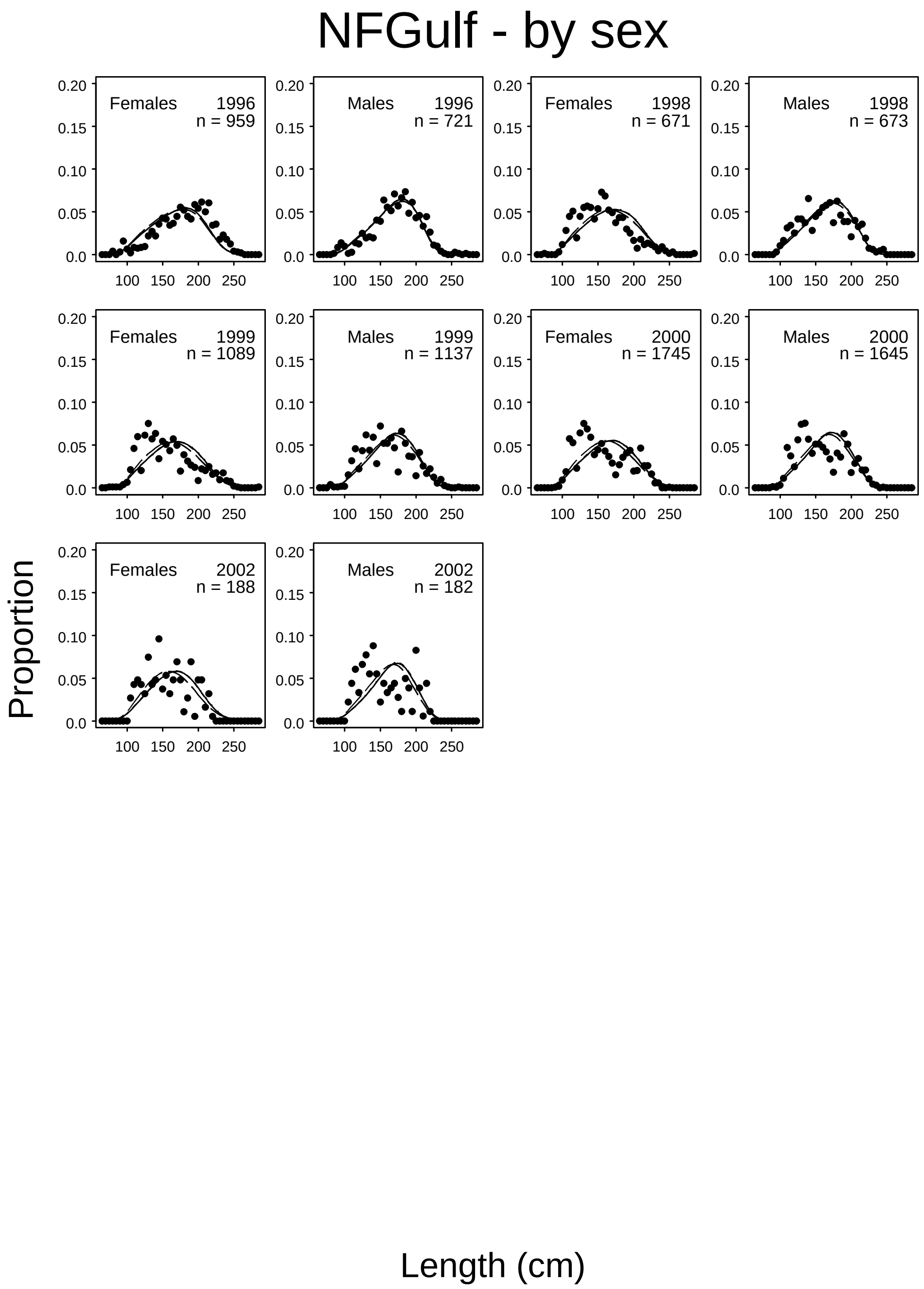


Figure 13.3. Observed (points) and fitted (lines) catch-at-length proportions by sex in the NF-Gulf region after 1998. Line symbolism is the same as Figure 9. All fits are virtually identical.

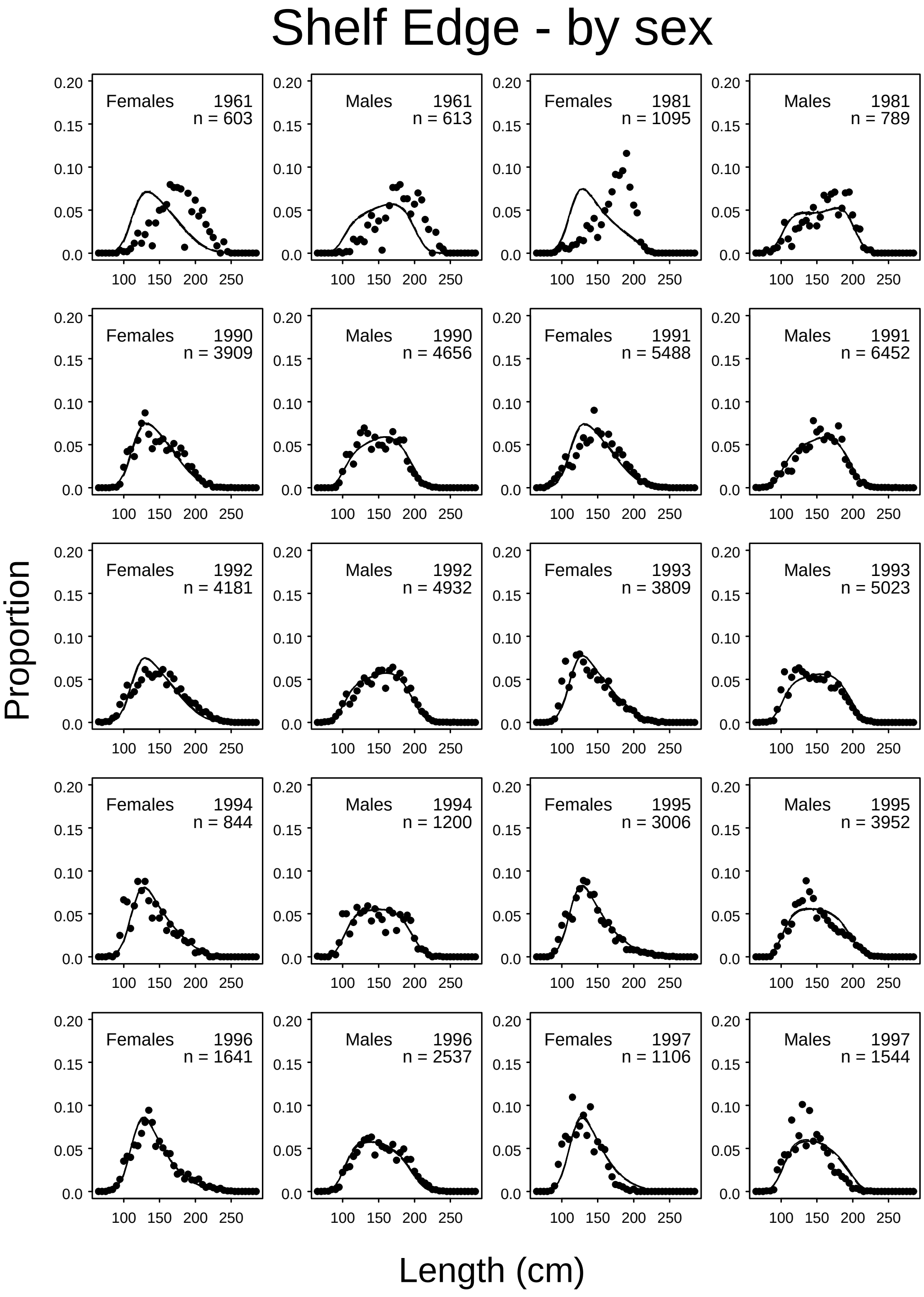


Figure 13.4. Observed (points) and fitted (lines) catch-at-length proportions by sex in the Shelf-Edge region. Line symbolism is the same as Figure 9.

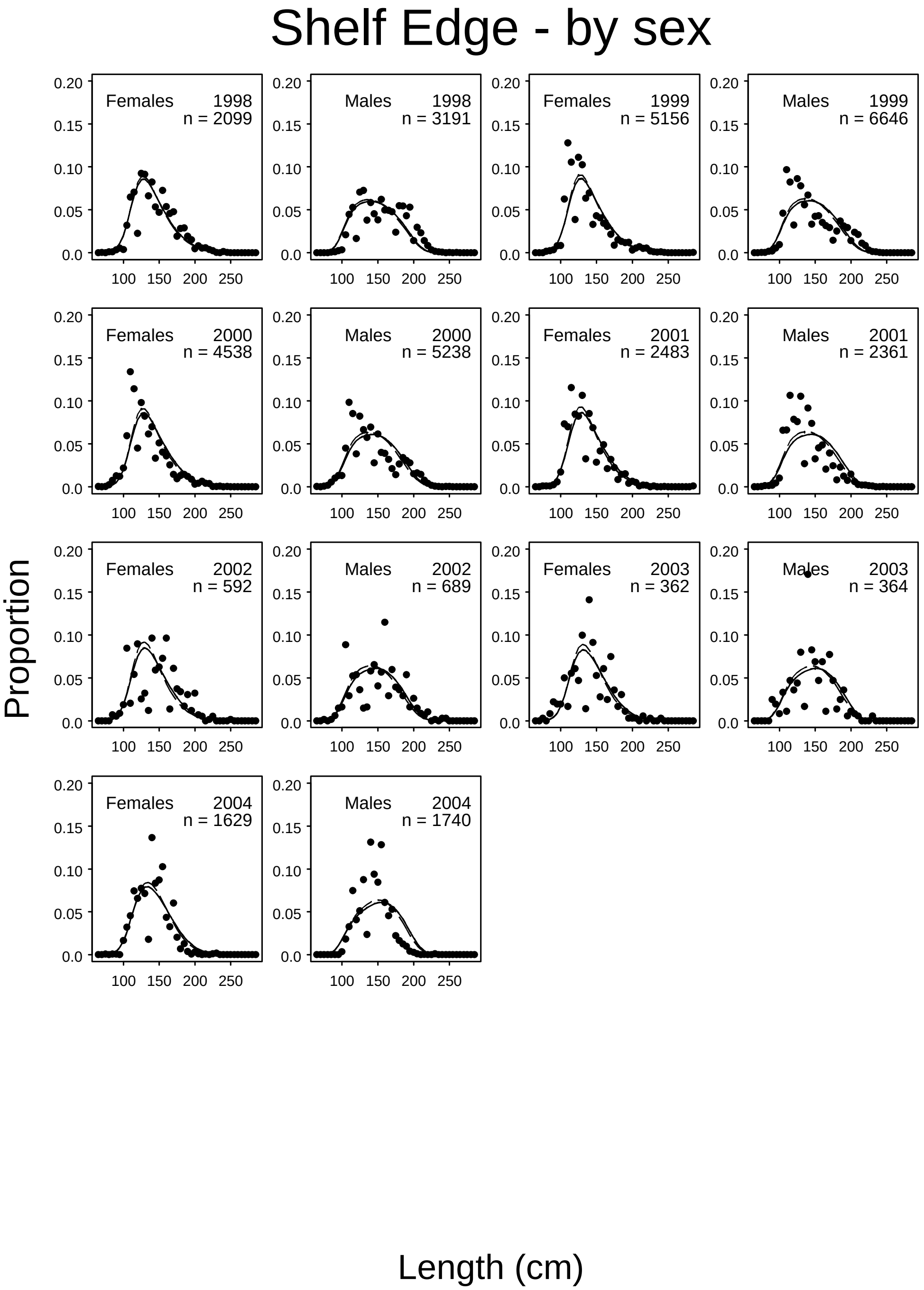


Figure 13.5. Observed (points) and fitted (lines) catch-at-length proportions, sexes combined in the Shelf-Edge region. Line symbolism is the same as Figure 9. All fits are virtually identical.

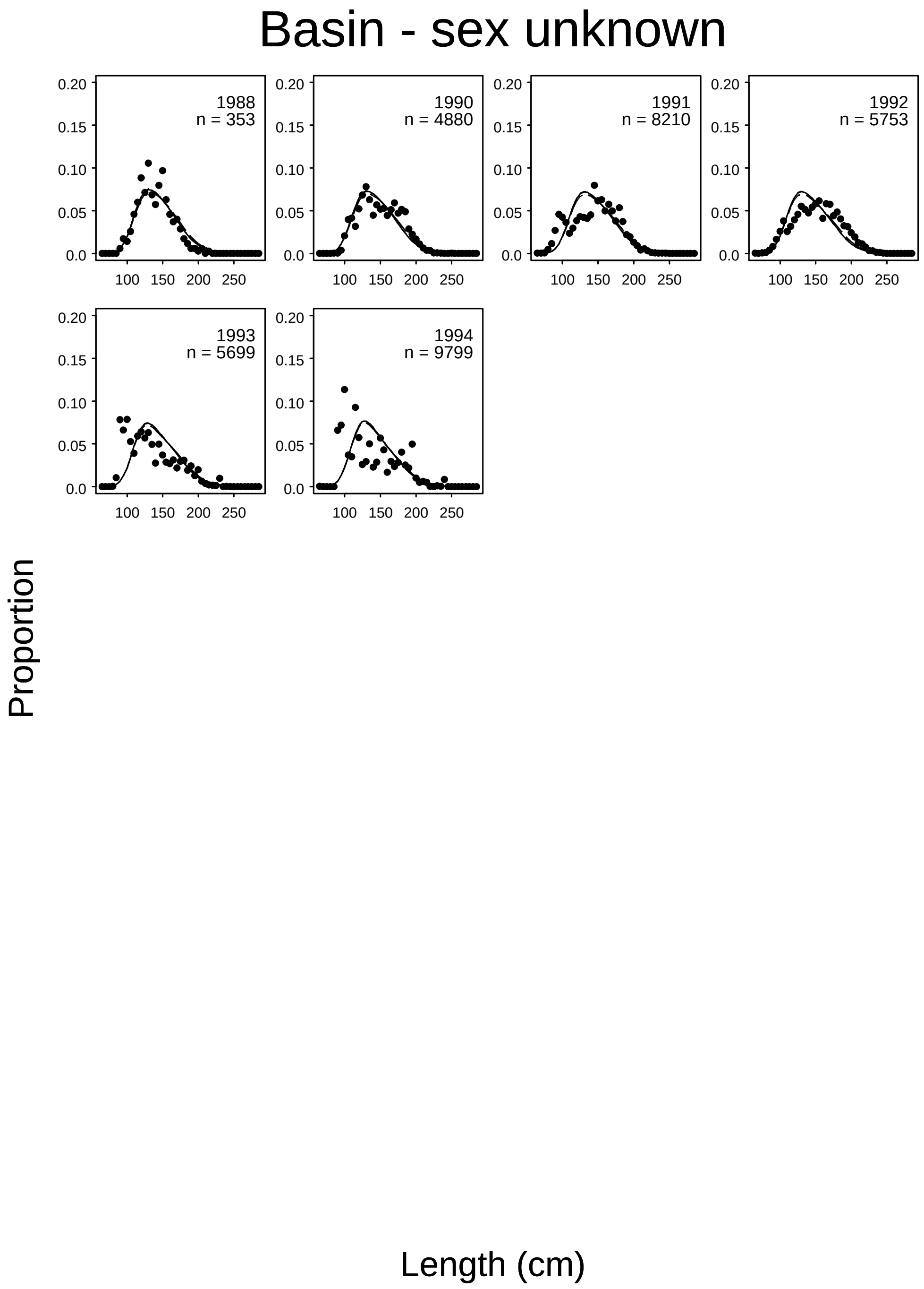


Figure 13.6. Observed (points) and fitted (lines) catch-at-length proportions, sexes unknown in the Basin region. Line symbolism is the same as Figure 9. All fits are virtually identical.

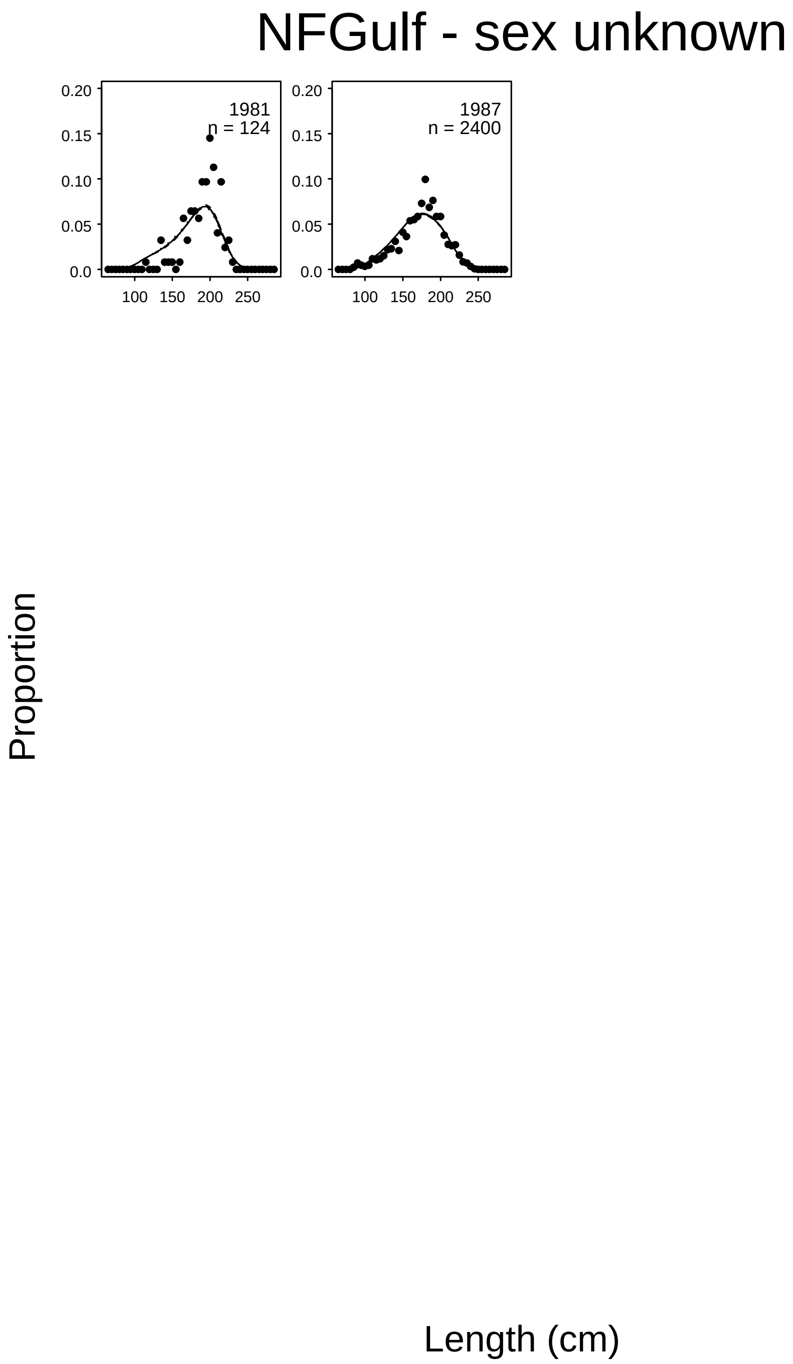


Figure 13.7. Observed (points) and fitted (lines) catch-at-length proportions, sexes unknown, in the NF-Gulf region. Line symbolism is the same as Figure 9. All fits are virtually identical.

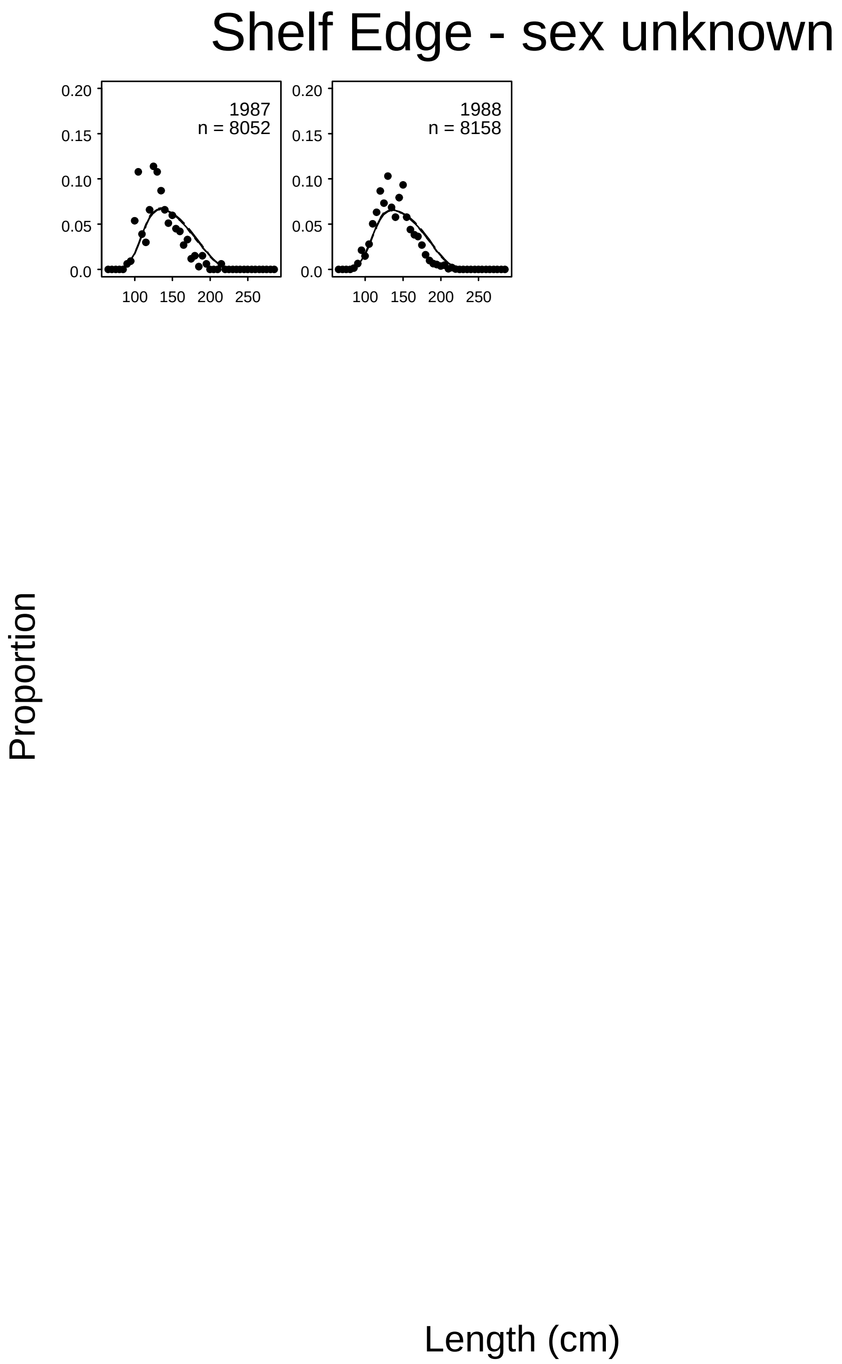
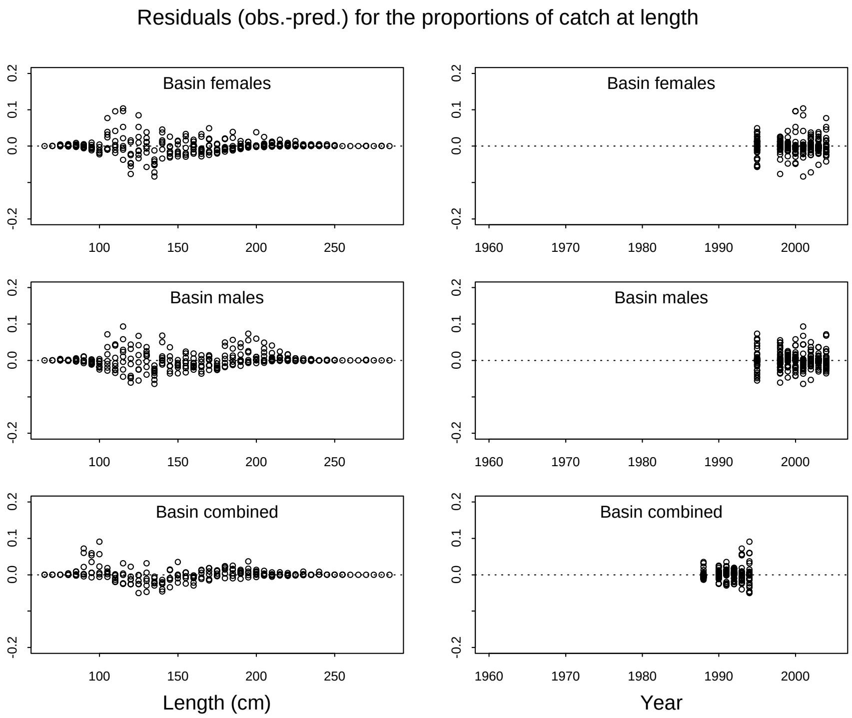


Figure 13.8. Observed (points) and fitted (lines) catch-at-length proportions, sexes combined in the Shelf-Edge region. Line symbolism is the same as Figure 9. All fits are virtually identical.

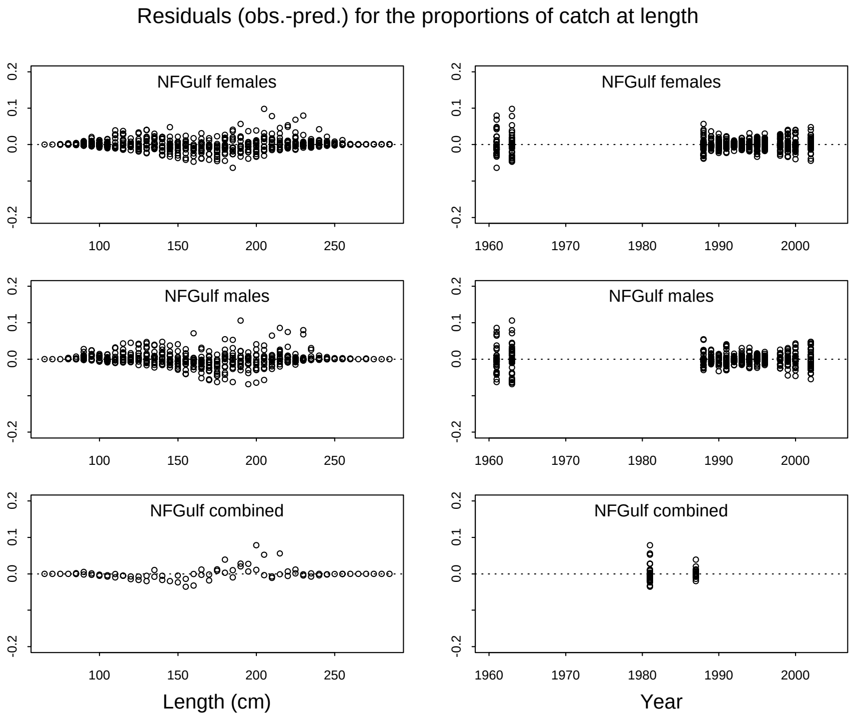

Residuals (obs.-pred.) for the proportions of catch at length

Basin females

Basin males

Basin combined

Figure 14.1. Raw residuals for fits to the catch-at-length data by length and by year for the Basin region obtained from Model 3.


Residuals (obs.-pred.) for the proportions of catch at length

NFGulf females

NFGulf males

NFGulf combined

Figure 14.2. Raw residuals for fits to the catch-at-length data by length and by year for the NF-Gulf region obtained from Model 3.

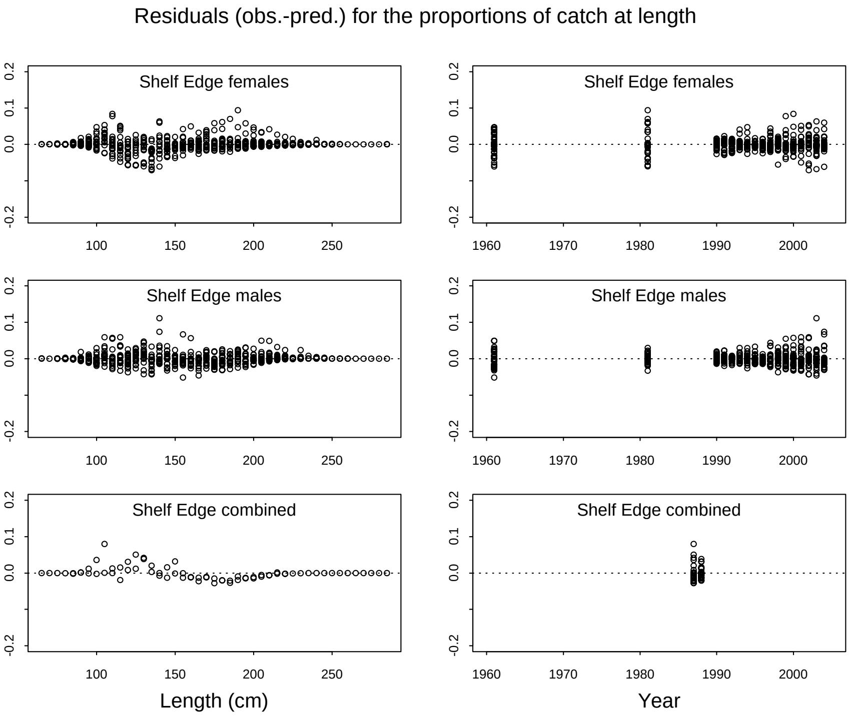


Figure 14.3. Raw residuals for fits to the catch-at-length data by length and by year for the Shelf-Edge region obtained from Model 3.

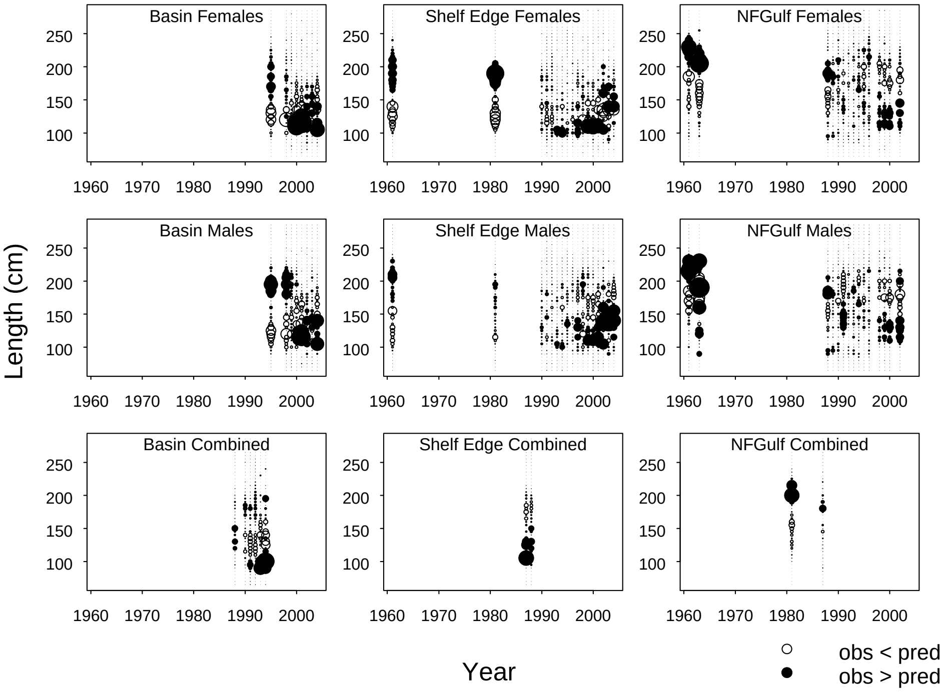


Figure 15. Raw residuals for fits to the catch-at-length data for the three regions obtained from Model 3.

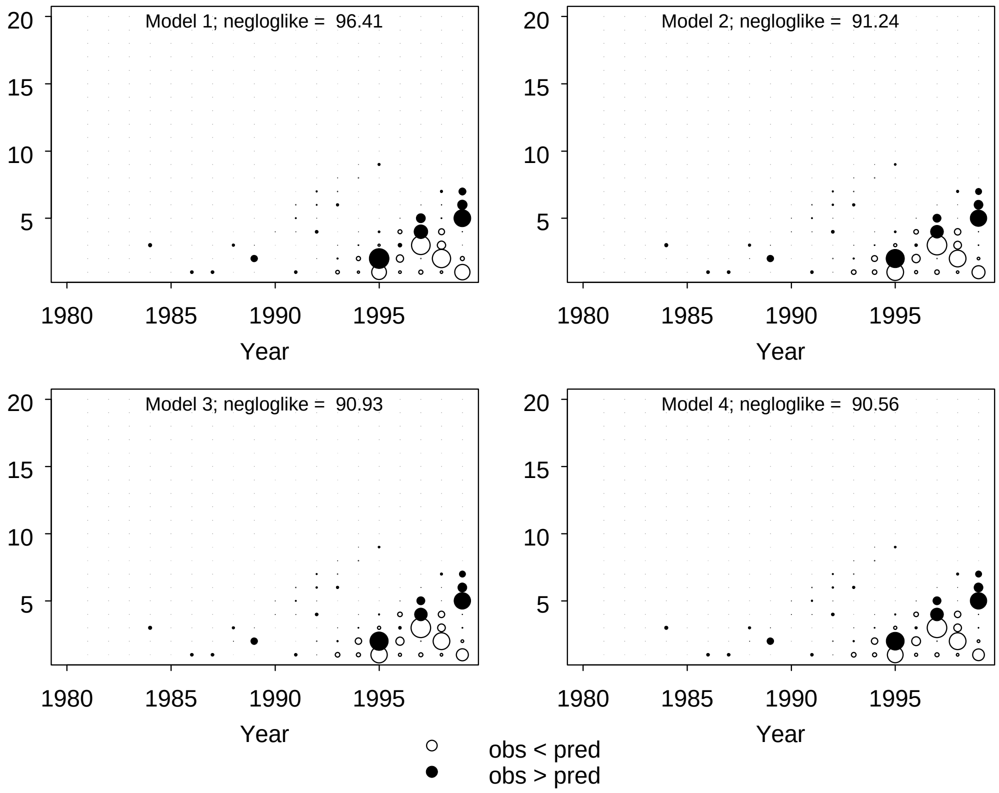


Figure 16. Residuals (observed tag recaptures - predicted tag recaptures) for the tagging component of the model. Points are proportional in size to the size of the residual (max. 12.5).

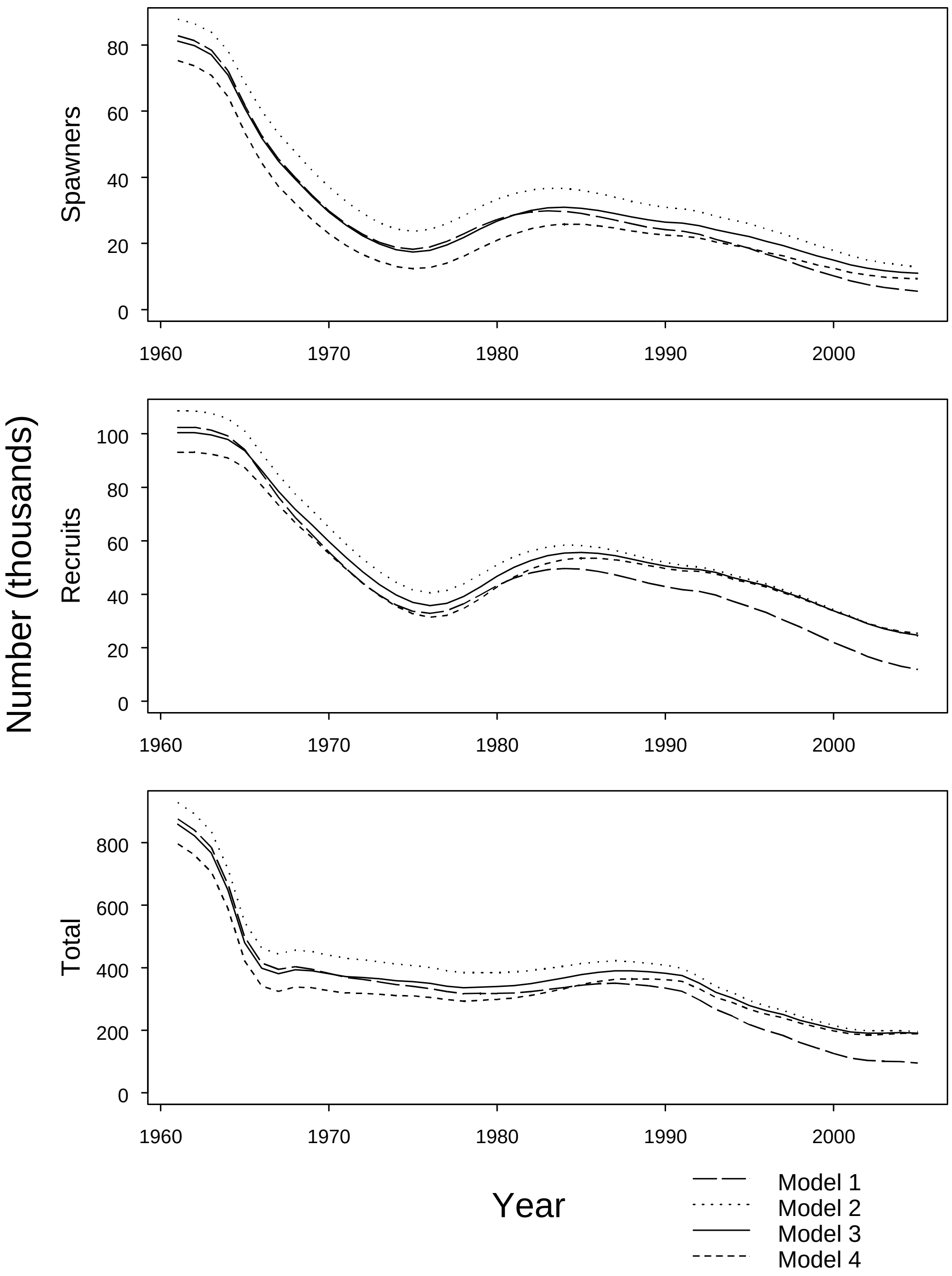


Figure 17. Comparison of the predicted time series for female spawner abundance, recruitment at age-1 and total number from each of the four models.

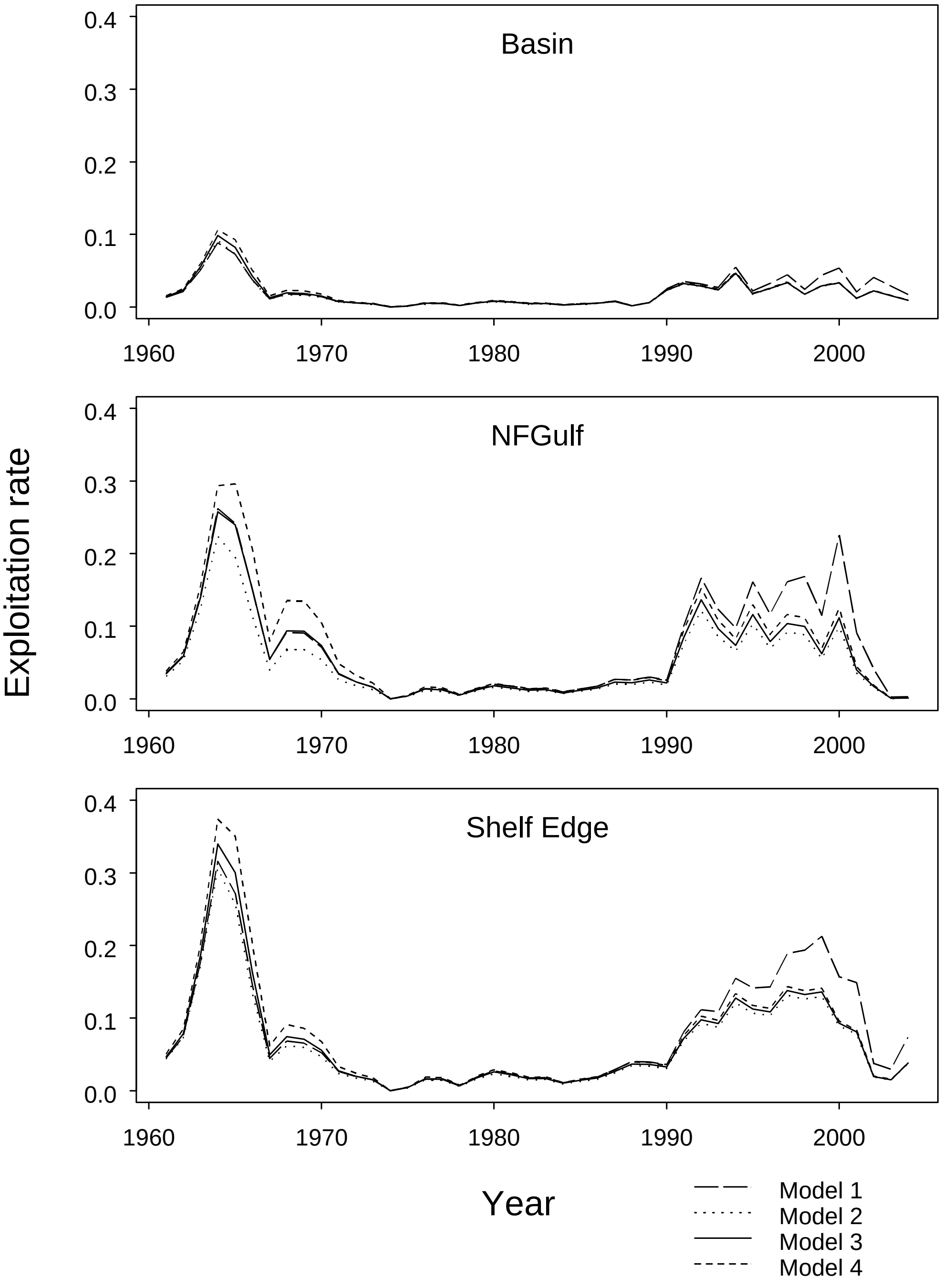


Figure 18. Comparison of the predicted time series of exploitation rates (proportion of the vulnerable biomass) from each of the four models. Line symbolism is the same as Figure 13.

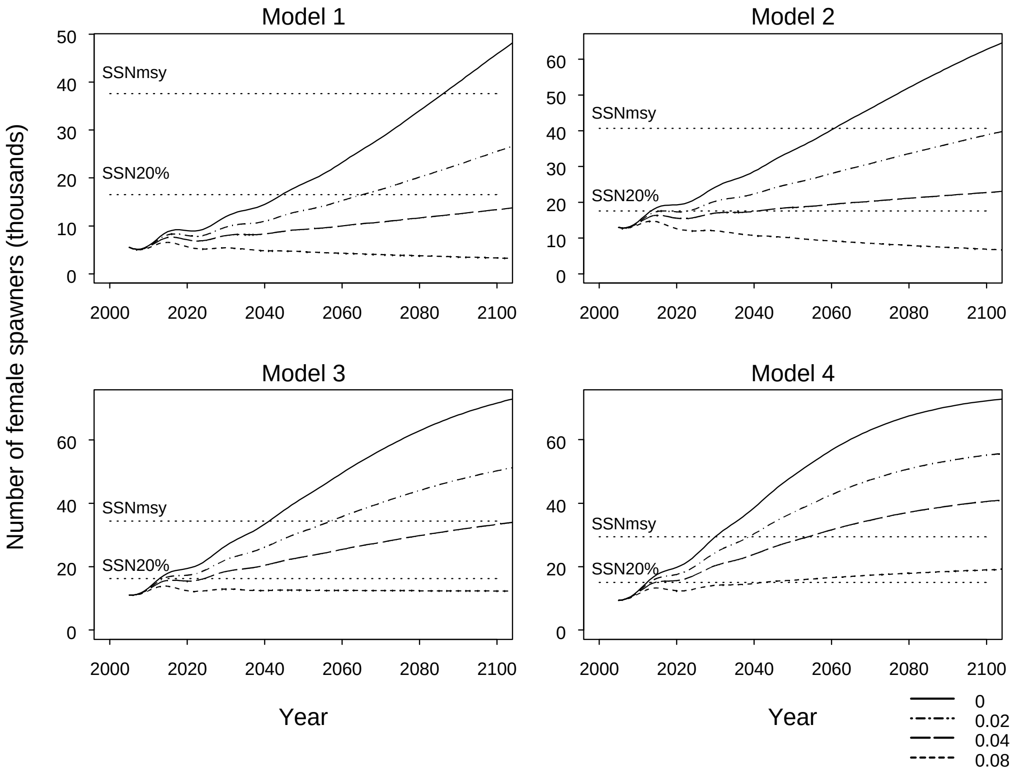


Figure 19. Comparison of the recovery targets and recovery trajectories obtained from each of the four models (see text for description) assuming 4 different exploitation rates (and the Shelf-Edge selectivity). Population projections begin in 2005 from the abundance by age and sex predicted by the model, and are projected deterministically using the life history parameters obtained from the model and the assumed exploitation rate. The Shelf-Edge selectivity is used in each case.

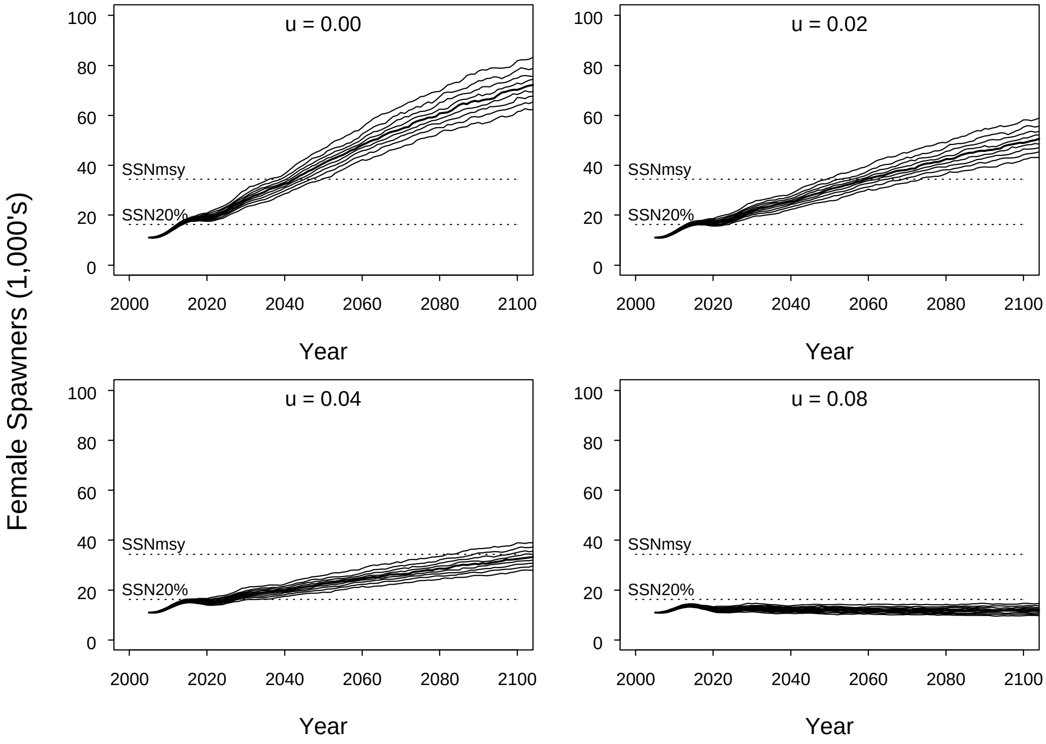


Figure 20. Recovery trajectories for porbeagle obtained from the population viability analysis under four exploitation scenarios. Each plot summarizes the results of 200 Monte Carlo population simulations with random variability added to reproduction and survival (see text for explanation. The lines connect the quantiles of the population size in each year from low (bottom line = 0.1) to high (top line = 0.9). The Shelf-Edge selectivity was used to model exploitation.

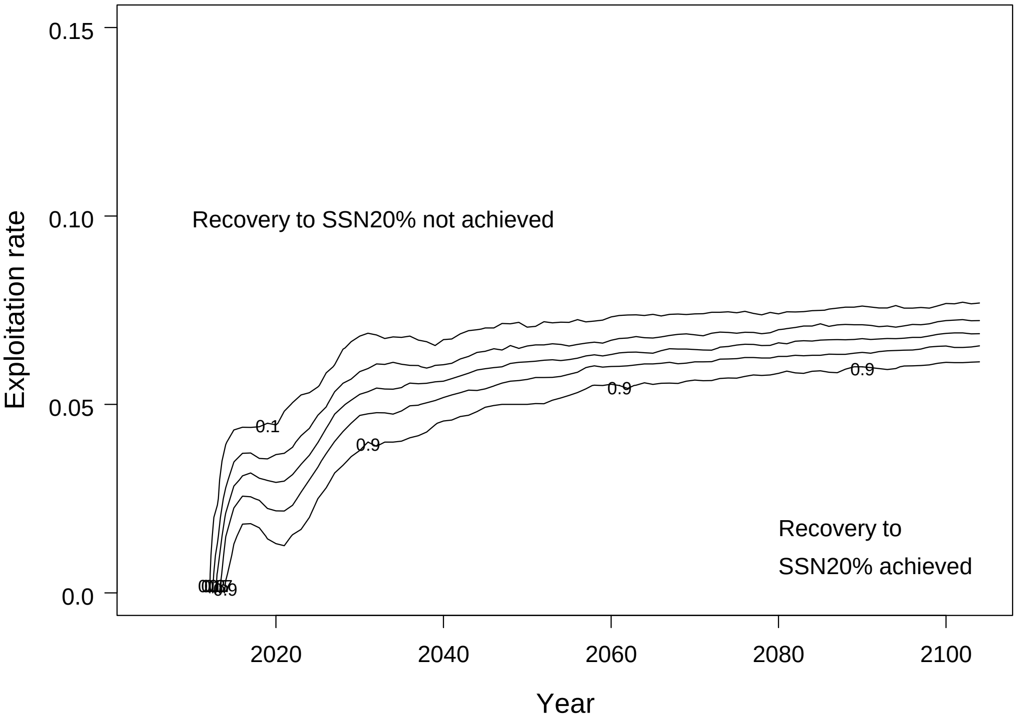


Figure 21. Contour plot showing the how recovery to $SSN_{20\%}$ is influenced by exploitation rate. Lines represent the contours of the probability of recovery from high (bottom line = 0.9) to low (top line = 0.1). Recovery probability is calculated as the proportion of 200 Monte Carlo population simulations that were above $SSN_{20\%}$ within the given time. The Shelf-Edge selectivity was used to model exploitation.

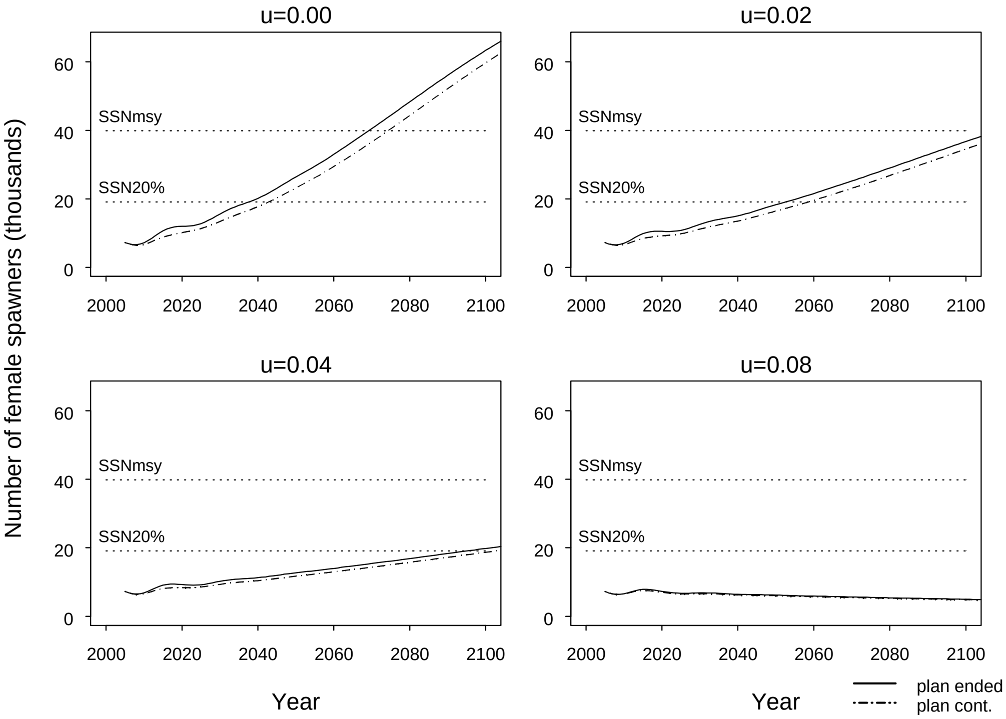


Figure 22. Comparison of the predicted recovery trajectories under two management scenarios based on Model 1. The solid line indicates the recovery trajectories if the current management 5 year plan ended at the end of 2005 (3 years into the plan) and porbeagle were managed under SARA after that time. “u” is the exploitation rate corresponded to different levels of human-induced mortality. The dashed line indicates the recovery trajectory if the current management plan was allowed to continue for the duration of the plan (ending at the end of 2007), and management of the population was under SARA after that time. Population projections begin in 2005 from the abundance by age and sex predicted by the model, and are projected deterministically using the life history parameters obtained from the model and the assumed exploitation rate. The Shelf-Edge selectivity is used for the projections.