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**Research Document 2003/068** — **Document de recherche 2003/068**

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# Harvest Simulations for 2003-2006 Harp Seal Management Plan.

# Plan de gestion du phoque du Groenland pour 2003-2006 à partir de Simulations de récolte.

By

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ISSN 1499-3848

**Abstract**

The northwest Atlantic harp seal population is currently estimated to number around 5 .2 million animals. The current large size of the herd coincides with a failure of northwest Atlantic cod stocks to recover from a period of intensive overfishing that resulted in closure of the Atlantic cod fishery at the beginning of the 1990s. Although the potential impact of harp seals on the recovery of Atlantic cod are inconclusive, the large size of the herd combined with improvements in market conditions have lead to requests for an increase in the Canadian allowable harvest level. Current management objectives to increase the economic return to the industry consider that a smaller population size is acceptable as long as it remains above the Precautionary Reference level ($N_{70}$), which at 70% of the estimated maximum population size is about 3.85 million animals. Owing to uncertainty associated with current estimates of population size, it was suggested that the lower 60% confidence limit serve as a metric to determine when $N_{70}$ had been attained. The width of the 60% confidence intervals increases as time since the last survey increases, reflecting an increase in uncertainty concerning the estimate of overall population size. Regular and frequent surveys are necessary to reduce the uncertainty surrounding these estimates. To determine the impact of various harvest levels on northwest Atlantic harp seals, a simplified Excel model incorporating uncertainty was constructed. The model results were similar to the model used previously to estimate abundance of this population to 2000. Harvests ranging from 75,000 – 500,000 over the next three years were examined for their potential impact on the northwest Atlantic population. Assuming that the age structure of the Canadian and Greenland harvests does not change, and that no changes occur in reproductive and natural mortality rates, the replacement yield for the current population of about 5.2 million animals is approximately 255,000 animals. With the exception of one run assuming a harvest of 75,000 animals, the scenarios assumed harvest levels that exceeded current estimates of replacement yield and consequently resulted in an overall population decline. Harvest levels of 275,000 animals result in only a slight change in abundance until 2009, after which the population begins to decline at a more rapid rate. Harvests as high as 500,000 animals for three years, followed by harvests of 275,000 animals per annum resulted in the population dropping rapidly to $N_{70}$ by 2009. Harvest scenarios that examined a variable take within a three year period had a similar impact on the population as fixed harvest levels that removed the same total number of animals during the three year period. Including additional uncertainty such as the variability surrounding the actual fraction of the established quota that is harvested, and reporting rates, increased the overall uncertainty around the modeled population estimates. Further simulation testing is needed to examine the performance of the model and the usefulness of using the 60% C.I. as a metric for population size in response to failures in model assumptions and additional uncertainty.

**Résumé**

Selon des estimations, la population de phoque du Groenland de l'Atlantique nord-ouest compte quelque 5,2 millions d'animaux à l'heure actuelle. Cette forte taille du troupeau coïncide avec l'échec des stocks de morue de cette écozone de se rétablir de la surpêche dont ils ont été l'objet, ce qui a mené à l'arrêt de la pêche de la morue au début des années 1990. Quoique l'impact potentiel du phoque du Groenland sur le rétablissement de cette espèce commercialement importante n'ait pas encore été confirmé, la grande taille du troupeau, ajoutée à une conjoncture plus favorable du marché, ont donné lieu à des demandes à l'effet que le total autorisé des captures dans les eaux canadiennes soit augmenté. Les objectifs de gestion actuels, qui visent à accroître les revenus de l'industrie, considèrent qu'une réduction de la taille de la population est acceptable tant que les effectifs demeurent au-dessus du niveau de référence préventif ($N_{70}$) qui, à 70 % de la taille maximale estimative de la population, se situe à environ 3,85 millions d'animaux. À cause de l'incertitude entourant les estimations courantes de la taille de la population, on a suggéré que la limite inférieure de l'intervalle de confiance à 60 % serve de point de référence pour déterminer lorsque $N_{70}$ a été atteint. L'étendue des intervalles de confiance à 60 % augmente au fur et à mesure que passe le temps depuis le dernier relevé, ce qui indique que l'incertitude entourant l'estimation de la taille globale de la population augmente. Des relevés fréquents et réguliers sont donc requis pour réduire l'incertitude entourant ces estimations. Afin de déterminer l'impact de divers niveaux de capture sur le phoque du Groenland de l'Atlantique nord-ouest, on a élaboré un modèle Excel simplifié incluant l'incertitude. Les résultats obtenus étaient semblables à ceux issus du modèle utilisé pour estimer l'abondance de cette population jusqu'en 2000. On a tenté d'établir l'impact sur celle-ci de prises se situant entre 75 000 à 500 000 animaux au cours des trois prochaines années. Si l'on suppose que la structure des âges des prises canadiennes et groenlandaises ne change pas, tout comme les taux de mortalité naturelle et de naissance, la production de remplacement pour la population actuelle de près de 5,2 millions d'animaux se situe à environ 255 000 animaux. À l'exception du passage du modèle à un niveau de prises de 75 000 animaux, les scénarios reposaient sur des niveaux supérieurs aux estimations actuelles de la production de remplacement et ont par conséquent résulté en une diminution de la population. À un niveau de prises de 275 000 animaux, l'abondance des phoques diminue légèrement jusqu'en 2009, après quoi elle diminue beaucoup plus rapidement. Des prises de jusqu'à 500 000 animaux pendant trois ans, suivies de prises de 275 000 animaux par an, font rapidement chuter la population à $N_{70}$ dès 2009. Les scénarios reposant sur des prises variables pendant trois ans ont la même incidence sur la population que des niveaux de prises fixes prélevant le même nombre total d'animaux pendant la même période. L'inclusion d'une autre source d'incertitude, comme la variabilité de la fraction réelle du quota fixé qui est récoltée, et des niveaux de rapport des prises donne lieu à une augmentation de l'incertitude générale entourant les estimations modélisées de la population. D'autres essais de simulation doivent être faits afin d'établir la performance du modèle et l'utilité de l'intervalle de confiance à 60 % comme mesure de la taille de la population en réponse aux faiblesses des hypothèses du modèle et à l'incertitude additionnelle.

## Introduction

The harp seal (*Phoca groenlandica*) is the most abundant pinniped in the northwest Atlantic. Total population size is estimated using a population model that incorporates information on pup production from aerial surveys, information on reproduction rates and known mortalities (including reported harvests in Canada and Greenland, estimates of the number of seals killed but not landed and the number of seals caught as bycatch in fishing gear). The most recent estimate of pup production was obtained from aerial surveys carried out in 1999. Incorporating this pup production estimate into the population model, resulted in an estimated population of 5.2 million animals (SE=600,000) in 2000 (Healey and Stenson 2000).

In addition to subsistence hunts in the Canadian Arctic and Greenland, northwest Atlantic harp seals are harvested commercially in the Gulf of St Lawrence and off the coast of northeast Newfoundland and Labrador. Harp seals are also known to be taken as incidental catches in a number of fisheries, particularly the lumpfish fishery in Newfoundland.

In recent years Canadian total allowable harvest levels have been established with the underlying management objective that the population would remain relatively constant, with no increase or decrease in the estimated population size (referred to as Replacement Yield or Replacement Harvest). In 1996, the Minister established a TAC of 250,000 for the Canadian commercial harvest. This was increased to 275,000 in 1997. Since 1996 the average catch has been 237,000 seals of which more than 90% were young of the year. Based upon the assessment carried out in 2000 the Canadian landed replacement harvest was estimated to be ~257,000, assuming all other sources of mortality remained constant.

The current size of the harp seal population has raised concerns about seal predation on the recovery of Atlantic groundfish stocks. Seals are major consumers of fish, shellfish and zooplankton in eastern Canada (Hammill and Stenson 2000; Bundy 2001; Stenson and Perry 2001; Morissette *et al.* In preparation). However, the potential impact of seals on the recovery is difficult to assess. Bundy (2001) suggested that harp seals were slowing the recovery of 2J3KL cod stocks, but her model simulations were dependent upon the assumptions that the harp seal population continued to grow and that the system was controlled by predation (i.e. top-down). The model failed to predict observed changes in the shrimp and large crustacean stocks. As well, it predicted a recovery of cod stocks to half of their pre-collapse levels by the year 2000, which has not been observed. More recently, Morissette *et al*. (In preparation) conclude that harp seals play an important role in maintaining ecosystem structure. Although the potential impact of harp seals on the recovery of Atlantic cod are inconclusive, the large size of the herd combined with improvements in market conditions have lead to requests for an increase in the Canadian allowable harvest level. Here, we examine different harvest scenarios and outline their potential impacts on the northwest Atlantic harp seal population.

## Materials and Methods

The impact of changes in Canadian commercial harvest levels on the future population trajectory of Northwest Atlantic harp seals was examined using a population model that incorporates information from aerial survey estimates of pup production, female age specific reproduction rates, the age structure and total number of removals from the population due to bycatch in fishing gear, the Canadian commercial and subsistence hunts and the Greenland subsistence hunt, to predict population changes.

The model we used (hereafter referred to as the 'Risk model') is similar to that described by Healey and Stenson (2000). Their model which was written in SAS, consists of two components, the first being the population dynamics model while the second involved a statistical model. This model works well, but the SAS framework as written is awkward for simulation testing. To simplify the process we adapted the original model to run within an EXCEL spreadsheet and incorporated uncertainty in the parameters using an EXCEL add in called @Risk (@Risk, Palisade Corporation 2000). Briefly, @Risk allows statistical distributions (*e.g.* Normal, Negative binomial, Triangle, Uniform) to be associated with parameters within the spreadsheet. The parameters can then be resampled repeatedly (Monte Carlo resampling) from within the distributions in order to estimate the impact variability in parameters such as reproductive rates, mortality rates, reported catches or estimates of struck and loss has upon estimates of population trajectories.

### Model structure

We begin by describing the basic model (Healey and Stenson 2000). We then describe how the model was modified, the data inputs and finally the simulation framework.

The basic model has the form:

$$n_{a,t} = \left(n_{a-1,t-1}e^{-(\gamma)m/2} - c_{a-1,t-1}\right)e^{-(\gamma)m/2} \quad (1)$$

for a =1

$$n_{a,t} = (n_{a-1,t-1}e^{-m/2} - c_{a-1,t-1})e^{-m/2} \quad (2)$$

for $1 < a < A$,

$$n_{A,t} = (n_{A-1,t-1}e^{-m/2} - c_{A-1,t-1})e^{-m/2} \quad (3)$$

for $a = A$, where $A$-1 is taken as ages $A$-1 and greater, and

$$n_{0,t} = \sum_{a=1}^{A} n_{a,t}P_{a,t} \quad (4)$$

for $a = 0$;

where $n_{a,1}$ = population numbers-at-age $a$ in year $t$,
$c_{a,t}$ = the numbers caught at age $a$ in year $t$,

$P_{a,t}$ = per capita pregnancy rate of age a parents in year $t$, assuming a 1:1 sex ratio. P is expressed as a Normally distributed variable, with mean and standard error taken from the reproductive data

$m$ = the instantaneous rate of natural mortality.

$\gamma$ = a multiplier to allow for higher mortality of first year seals. Assumed to equal 3 for consistency with previous studies.

$A$ = the 'plus' age class (i.e. older ages are lumped into this age class and accounted for separately, taken as age 25 in this analysis).

*Initial age structure and model fitting*

In order to reconstruct the population prior to the period for which abundance estimates are available, Healey and Stenson (2000) assumed that the annual pup catch was a constant proportion ($s$) of the number of pups born ($s$=(1/exploitation rate)). Thus, for years prior to the first year for which pregnancy data were available (year $t_0$), the modification for predicting the numbers at age in the year $t_0$-1 is as follows:

$$n_{a,t0-1} = se^{-m(\gamma+a-1)}c_{0,t_0-a-1} - (\sum_{i=1}^{a-1} e^{-m(i-1/2)}c_{a-i,t_0-i-1}) - e^{-m(a-1+\gamma/2)}c_{0,t_0-a-1} \quad (5)$$

for a = 2,…,$A$,

$$n_{0,t_0-1} = sc_{0,t_0-1}, \quad (6)$$

and

$$n_{1,t_0-1} = se^{-(\gamma)m}c_{0,t_0-2} - e^{-(\gamma)m/2}c_{0,t_0-2}. \quad (7)$$

for $a$ = 1 to $A$, where $A$ is a terminal (rather than a plus) age ($A$=25 years in the formulations that follow). This equation is applied iteratively to go back in time and fill in the numbers-at-age matrix.

*Risk model*

For the simulations carried out here, the model was modified as follows: Since much of the harvest occurs before there is significant natural mortality among pups, Equations 1-3 were modified to apply mortality after the catches are removed. For example, Equation 1 was replaced by:

$$n_{a,t} = (n_{a-1,t-1} - c_{a-1,t-1})\ e^{-(\gamma)m} \quad (8)$$

where $m$ is the instantaneous rate of natural mortality as above. To capture some of the variability in natural mortality, m was modified in Equations 2,3 and 8 to: $m=n*b$, where $n$ is mortality and $b$ is a Normally distributed variable with Mean=1 and standard error of 0.015 for pups (age=0 years) and 0.0015 for adults (age =1+ years).

Recognizing that there is some variability associated with the age specific pregnancy rates, the mean age specific reproductive rate ($P_{a,t}$ ; Equation 4) incorporated into the model was defined as a Normally distributed variable, with the mean and standard error defined by the annual age specific mean reproductive rate and standard error determined by the smoothing procedure (obtained from Healey *et al* in prep; Table 2).

In 1998, 2000 and 2002, poor ice conditions were observed in the Gulf and it was suspected that pup mortality was higher than normal. Although it is difficult to quantify the difference in mortality during these years compared to 'normal' ice years, it was assumed that during 1998 and 2000, 25% of the pups died before harvesting began. Assuming that Gulf pups make up 30% of the total pup production this translated to an overall mortality of 8% prior to harvesting, (i.e. 92% of the pups survived). During 2002 we received reports of large numbers of whitecoats in the water and dead animals were found on beaches in Prince Edward Island, the Magdalen Islands and along the west coast of Newfoundland. Overall, ice conditions in 2002 were very similar to conditions observed in 1981. During that year pup mortality appeared to be extremely high and Sergeant (1991) noted that the 1981 year class was almost completely absent in subsequent age class samples collected during an early winter fishery at La Tabatiere. Assuming that the Gulf cohort in 2002 suffered extremely high mortality similar to what had been observed in 1981, we assumed that 75% of the Gulf pups died prior to harvesting which translates into an overall mortality of 25% for the population. For these years equation 8 becomes:

$$n_{a,t} =((n_{a-1,t-1} * w) - c_{a-1,t-1})\, e^{-(\gamma)m} \qquad (9)$$

where w=0.92, 0.92, 0.75 for 1998, 2000 and 2002 respectively.

The original model used Equations 4-7 to move backwards in time to reconstruct the population prior to the period when reproductive data were available. For the current exercise it was decided to limit the starting point for the population to 1960. To start the model it was assumed that pup production at the beginning of the simulation period (1961) was 800,000 animals. From this initial pup production, reproductive rates and catch data (Tables 1, 2), the age structure of the population was reconstructed using Equations 6 and 7. The model was fitted by adjusting the initial (1960) pup production and adult mortality rates to minimise the mean sum of squares (MSS) between the predicted pup production and the estimated pup production (weighted inversely proportional to their variance) obtained from independent surveys in 1990, 1994 and 1999 using Risk Optimizer (an Excel spreadsheet add-in from Palisade Corporation 2000).

RiskOptimizer uses genetic algorithms to search for optimal answers to simulation models (Palisade Corporation 2000). For some model inputs (*e.g.* reproductive rates) information is available to describe sample variability in our estimates (mean and standard error). To capture some of the variability in these parameters, single parameter values were replaced by statistical distribution functions with mean and standard error estimated from the available data.

The model randomly selects values for the initial population size and mortality rates, then re-samples (Latin Hypercube) values from the defined functions for each parameter (*e.g.* values for reproductive rates and mortality rates). Sampling was repeated 200 times (replicates) and from the 200 replicates the mean sum of squares (MSS) was calculated. These constitute a simulation. The MSS for the simulation was stored, new values for the initial population size and mortality rate chosen at random and 200 samples from the defined functions drawn to complete another simulation. After 51 simulations have been completed, the optimizer retains the 50 simulations with the smallest MSS. From this sample of 50 simulations the software selects combinations of initial population size and mortality rates. This exchange of values, called crossover is used to create new combinations (offspring) of initial population size and mortality rates which are used in a new simulation. The objective is to use the information gained from the previous simulations to try to focus the selection of new values towards values that would reduce the MSS. After crossover, the software allows for a random value, referred to as a mutation, to be generated for either the initial population size or the mortality rate. Mutations allow for new combinations to be introduced into the population of 50 retained simulations, which in turn may lead to the lowest MSS. A total of 1000 simulations were completed. The model inputs that generated the smallest MSS were retained.

<u>*Data Input*</u>

*Catches*

Reported landings vary considerably between years owing to a combination of market conditions and ice conditions that affect access to the herd. Catches prior to 1998 were taken from Stenson *et al*. (2000). Canadian commercial catches from 1998 – 2002 were obtained from the Department of Fisheries and Oceans (DFO) Statistics Branch. Greenland Home Rule provided Greenland catches up to 2000. As in previous studies, the reported catches were corrected for unreported harvest (Stenson *et al*. 2000). Greenland catches for 2001 and 2002 were assumed to be the same as in 2000. No recent information is available on catches in the Canadian Arctic. Therefore, they were assumed to remain constant at the level reported by Stenson *et al*. (2000). Catches since 1996 are presented in Table 1. The proportion of young of the year taken in the Canadian commercial harvest was provided by DFO. The age structure of older seals and seals caught in Greenland and the Canadian Arctic (Table 1) was assume to be the same as reported by Stenson *et al*. (2000).

The proportions of seals killed but not landed and the level of bycatch were assumed to be the same as given in Healey and Stenson (2000). It was assumed that 95% of the pups killed in the Canadian hunt and that 50% of animals aged 1+ years and 50% of all animals killed in the Greenland and Canadian Arctic harvests were recovered (Stenson *et al.* 2000). Thirty percent of pup production was assumed to occur in the Gulf and the Canadian harvest was assumed to consist of 92% young of the year (1997-2002 average).

Table 1. Estimated and reported landings from the Canadian Arctic, Canadian commercial and Greenland hunt.

| Year | Canadian Quota | Canada Arctic [1] | Canada commercial[2] Harvest | Greenland |
|---|---|---|---|---|
| 1996 | 250,000 | 4881 | 242,906 | 81,663 |
| 1997 | 275,000 | 4881 | 264,210 | 75,854 |
| 1998 | 275,000 | 4881 | 282,067 | 89,742 |
| 1999 | 275,000 | 4881 | 244,552 | 102,499 |
| 2000 | 275,000 | 4881 | 91,602 | 108,432 |
| 2001 | 275,000 | 4881 | 226,493 | 108,432[1] |
| 2002 | 275,000 | 4881 | 307,345 | 108,432[1] |

[1] estimated or assumed

[2] After the simulations were completed, the 2002 catch was revised upwards to 312,367.

*Pregnancy rates*

The age specific pregnancy rates were based upon samples obtained between 1954-1997. The raw data are presented in Sjare *et al.* (2000), adjusted for age at birth. All seals less than four year of age were immature while seals eight years of age and older were considered fully recruited to the breeding population and grouped together (Sjare *et al*. 2000). Samples were obtained between November and February and so provide late-term pregnancy rates. Age-specific sample sizes were highly variable with total annual sample sizes ranging from 11 to 258 seals. . The vast majority of samples were collected in the Newfoundland area.

Previous analyses have attempted to provide annual pregnancy rates from the available sampling data. Bowen *et al*. (1981) used annual smoothing (as opposed to smoothing by age as in this analysis) to ensure that for any given year the proportion mature increased with age in the event that the sampling predicted otherwise. An analysis by Shelton *et al*. (1992) attempted multi-linear regression, analysis of covariance, analysis of variance, and auto-regression models, and discovered that all methods were inadequate to predict the unknown pregnancy rates. More recent efforts to estimate pregnancy rates are based upon the method described in Shelton *et al*. (1996) (presented with some modifications in Warren *et al*. (1997)). For each age, successive contingency table analysis tests successive pregnancy sample data for significant changes in pregnancy rates, and the resulting rates are referred to as `harmonized' rates.

Healey *et al*. (In prep.) developed a nonparametric regression estimator to estimate the expected pregnancy rates. There are no data for many year-age combinations, thus these expectations have to be inferred from neighboring observations using a simple model. Assuming that for each age, the number of pregnant seals sampled in year $t$ (denoted as $Y_t$) from a total of $n_t$ is Binomially distributed, with mean $n_t p_t$ where $p_t$ is the probability that a seal was pregnant. With no further restrictions on $p_t$, the maximum likelihood estimate (mle) of $p_t$ is $y_t / n_t$ - the sample proportion of pregnant seals.

The sample proportion of pregnant seals may be quite dissimilar from year to year, but the population pregnancy rates are not expected to vary widely between years. Sample proportions may vary widely when the sample size is small, and this is compounded when there is considerable within-age population variability in sampled pregnancy rates. Another problem is estimating pregnancy rates in the years with no samples. These problems suggest that some reasonable model restrictions of the $p_t$'s are necessary, especially to infer $p_t$'s in years not sampled. Assuming that the $p_t$ must be a smooth function of $t$, the amount of smoothness will be determined by the available data. The statistical problem then is to estimate this function or, equivalently, to estimate $p_t$. Since it is not possible to estimate $p_t$ via maximum likelihood without specifying this function more exactly, a non-parametric approach is taken. Local averaging is a commonly used alternative to estimate $p_t$. The rationale for local averaging is as follows.

Define an ε-neighborhood of observations around some given year $t$ as $A_t=\{i : | t_i - t | \leq \varepsilon\}$. If ε is chosen small enough then it can be assumed that $p(t_i)=p(t)$ for all $i$ in $A_t$. In this case the mle for $p(t)$ is:

$$\hat{p}(t) = \frac{\sum_{i \in \mathrm{A_t}} \mathrm{y_i}}{\sum_{i \in \mathrm{A_t}} \mathrm{n_i}} . \qquad (10)$$

Only $y_i$'s with $t_i$ values within the ε - distance of $t$ have a full contribution to the estimate of $p(t)$. Other $y_i$'s have no contribution to the estimate. Another approach is to use a weight function designed so that the contribution of $y_i$ changes gradually according to the distance between $t_i$ and $t$. The weight function $W$ measures the distance between $t$ and $t_i$. The size of the neighborhood is determined by a bandwidth, $b$. The maximum local likelihood estimate is:

$$\tilde{p}(t) = \frac{\sum_i \mathrm{W}\left\{\frac{(t_i - t)}{b}\right\} \mathrm{y_i}}{\sum_i \mathrm{W}\left\{\frac{(t_i - t)}{b}\right\} \mathrm{n_i}} . \qquad (11)$$

The Gaussian weight function, $W(\mathrm{x}) \propto \exp(-\mathrm{x}^2/2)$, is used here, although other functions are commonly used. The Gaussian weight function defines elliptical neighborhoods in $t$. As $b \to 0$, the neighborhood includes just $t_i$.

The choice of bandwidths is critical in smoothing. A bias-variance trade-off exists in determining the size of the bandwidths. A small bandwidth leads to an estimator with small bias, but large variance (i.e. erratic), while a large bandwidth leads to an estimator with large bias, but small variance (i.e. oversmooth). The data were used to choose a bandwidth, or the amount of smoothness, that minimizes a measure of prediction error. The measure used is Generalized Cross Validation. This is a common prediction error measure used in kernel smoothing and spline smoothing (Healey *et al.* In prep.), along with comparisons with other methods. The amount of smoothness that is useful will depend on age, so bandwidths were selected separately for each age.

*Harvest Simulations*

The output of the Risk model for the period 1960 – 2000 was compared to population trajectories predicted by the Healey and Stenson (2000) model (Fig. 1). Once the form of the model describing changes in the population up to 2003 had been established, the impact of different harvest scenarios on future population trajectories was examined. During these simulations it was assumed that the Greenland and Canadian Arctic harvests, age structure of the harvest, bycatch, levels of struck and lost and reproductive rates had not changed since 2000. Simulations (1000 runs) were completed by changing the size of the Canadian TAC starting with the 2003 harvest and observing its impact on the projected population 50 years into the future (software @Risk, Palisade Corporation 2000). During these simulations it was also assumed that no changes in natural mortality would occur (*e.g.* no additional bad ice years).

The harvest scenarios examined are summarized in Table 3.We also examined predicted changes in population size that would be expected if the allowable harvest was determined based upon an approach developed in the United States known as the Potential Biological Removals (PBR). Given that marine mammals are not harvested in the US, the PBR identifies the maximum permissible takes of marine mammals in commercial fisheries that would still allow a 95% probability that the population would increase to Optimum Sustainable population levels (Wade 1998). It is calculated using the formula:

$$PBR = N_{MIN} \frac{1}{2} R_{MAX} F_R \ , \qquad (12)$$

where: $R_{MAX}$ is the maximum rate of increase, set at 0.12 for seal populations. $N_{Min}$ is the minimum population estimate, which is calculated as the lower limit of the two-tailed 60% confidence interval of the log-normal distributed best abundance estimate, *i.e.* equivalent to the 20th percentile of the log-normal distribution (Wade 1998). $F_R$ is a recovery factor with a value set at ≤ 1.0 (Wade 1998).

*Reference Point ($N_{70}$)*

Marine mammals are long lived, have low annual reproductive rates, and delayed maturity (age 4 years or older). As a consequence, population growth rates are low and they recover very slowly from over-exploitation. The Canadian commercial hunt, which accounts for over 60% of the removals in this population, is directed primarily towards young of the year. Currently, population size is determined through a combination of pup surveys flown every five years, and annual collections of reproductive and catch data. Owing to the pattern of harvesting and the survey frequency, management impacts will not be detected until a considerable period of time after a management action was initiated. Marine mammal populations are vulnerable to over-exploitation, current harvests are high and there is considerable uncertainty in modelling population trajectories more than a few years into the future.

A forum was held in St. John's, Newfoundland (November 2002), to examine the application of Objective-Based Fisheries Management and the Precautionary Approach to the management of the Northwest Atlantic harp seal (DFO 2003). As a result of this meeting, Fisheries management has accepted the use of a series of reference points developed for 'Data Rich' species (Appendix 1; Hammill and Stenson 2003) for the management of harp seals. The first reference point was established at 70% of the largest estimated population. Called $N_{70}$ (referred to as Nconservation at the forum), this level would be established at 3.85 million for the Northwest Atlantic harp seal. The year in which the population reaches this reference point level was determined for each harvest scenarios examined.

For each of the different harvest scenarios examined we estimated the harvest levels that would be required to halt a population decline at the end of a particular management regime. Also, if the harp seal population declined to 3.85 million ($N_{70}$), we estimated the reduction in TAC that would be required to prevent any further decline. The Forum in Newfoundland did not attempt to establish a control rule that would define when a population had fallen below a reference point. Here we propose a control rule, recognizing that further simulations are required to examine the rule for robustness, to examine the impacts on model predictions of potential changes in population parameters and failure to satisfy model assumptions.

## Results

### Model

The model provided the best fit to the data with a 1960 pup production of 488,000 and an instantaneous mortality rate ($m$)= 0.058. The model provided population and pup production estimates that were very close to simulation results obtained by Healey and Stenson (2000)(Fig. 1). The population estimate provided in 2000 was 5.2 million, (Healey and Stenson 2000). Based on the reported level of harvest prior to 2002, the Northwest Atlantic harp seal population increased to 5.5 million (SE=580,000) animals in 2002. However, a combination of very high pup mortality and a very large harvest in 2002 appeared sufficient to halt this increase and the 2003 population was estimated to have declined slightly to around 5.3 million (SE=608,000)(Fig. 1).

### Reference Points

The population trajectories estimate changes in the predicted mean estimate of population size over time. Under this approach there is a 50% chance that the population is either above or below this estimate. Simulating the impacts of an annual Canadian harvest of 325,000 seals beginning in 2003 on the population, it is evident that the population immediately begins to decline. Using only the mean estimate, the population would not decrease to $N_{70}$ until 2018, 15 years from now (Fig. 2). To halt the decline, the Canadian harvest would need to fall from 325,000 to 40,000 animals (not shown). This approach fails to consider the increasing uncertainty associated with the projected population as time from the start of the simulation increases and hence could not be considered a risk-adverse approach. This increase in uncertainty is encompassed by the standard error around the population estimate, which in the absence of new survey information, increases over time. In the scenario with annual harvests of 325,000, it can be seen that when the estimate has fallen to $N_{70}$, there is approximately a 30%, 15% and 3% chance that the population will have fallen below $N_{50}$, $N_{critical}$ and to 0 respectively (Fig. 2). Using the lower confidence limit to determine when the population may have fallen below the reference point levels offers a more conservative approach to dealing with this uncertainty. An initial choice could be the lower 95% C.I. In the above scenario, the lower 95% C.I. would fall to $N_{70}$ by 2005, while the estimate would remain at 5.2 million or about current levels. It was felt that this level was too conservative. A less conservative approach, would be to use the lower 60% C.I., which assumes a 20% probability that the population is below the lower 60% C.I. This would result in a control rule that when the lower 60% C.I., reaches the reference level, then a more conservative harvesting approach would be required to halt the decline of the population and to ensure that the population remained or moved above the reference level. Using the lower 60% C.I. control rule, with the scenario of an annual harvest of 325,000 animals, the population would fall to $N_{70}$ in 2011. The mean estimate would fall from 5.2 million in 2003 to 4.8 million. At the same time, the probability that the population would have fallen below $N_{50}$ would only be about 5% (Fig. 2). To halt the decline, and to ensure an 80% chance that the population would begin to increase, harvests would have to fall from 325,000 to 125,000.

*Model Scenarios*

Figure 3shows the estimated impacts of harvests using the 2002 Total Allowable Catch (TAC, 275,000), a new estimate of Canadian Replacement Harvest and a lower level based upon the Potential Biological Removal (PBR) approach developed in the United States. The new estimate of Replacement yield, designed to keep the population stable was calculated to be 250,000. We assumed a constant Canadian harvest component of the PBR of 75,000 animals from estimates of population size in 2003. By comparing the population trends observed under these different harvest scenarios it is clear that although changes in total population can be observed relatively quickly (Fig 3a), there is a lag of 10-15 years before changes in harvest are reflected in pup abundance (Fig 3b). Thus, considerable changes in total abundance can occur following changes in harvest levels before they will be detected during subsequent pup surveys.

Under the harvesting strategies using a TAC of 275,000 or 250,000,the overall population size changes little from 2003 until 2009. After 2009, the population with a TAC of 275,000 begins to decline, while the population with a TAC of 250,000 remains stable. The population is expected to increase over the same time period under the PBR harvest scenario (Fig. 3a). However, in reality the PBR harvest level should increase as population size changes. By not adjusting the PBR, the growth rate of the population under this harvest strategy is overestimated. For pups, maintaining the current harvest of 275,000 would result in a decline in pup production that would not be seen until about 2015 (Fig. 3b).

Using the lower 60% confidence limit of the population estimate the predicted rate of decline in the population and pup production is greater owing to an increase in the variance around the estimate as one moves farther away from the last survey point (1999)(Fig. 4). The lower 60% confidence interval line for the new replacement yield calculation of 250,000 also shows a decline, but at a slower rate compared to the population trajectory predicted from a harvest of 275,000, again a result of the increase in the variance around the population estimate as the time since the last survey (1999) increases. A new survey would reduce this uncertainty.

*Transferable TAC*

We were also asked to examine the impact of establishing a three year management plan that will limit the total number of animals to be taken during the life of the management plan, but allowing quotas to vary by 10 or 20% between years. For example, a management plan with a Canadian TAC of 825,000 for the three years could result in annual harvests of 275,000 (the current TAC) in each of the three years or it could result in annual harvests of 300,000, 300,000 and 225,000 animals (~10% transfer to the first two years with subsequent decrease in final year of plan). We model this scenario as well as one in which there is a 20% transfer to the first two years (330,000, 330,000 and 165,000 animals). Two more scenarios, where the total harvest was 900,000 animals to be taken over three years as 330,000, 330,000 and 240,000 or as 360,000, 360,000 and 180,000 were also examined. In all cases, at the end of the three year period the harvest returned to a level of 275,000.

The scenarios where the total harvest does not exceed 825,000 (3 X 275,000) animals within a three year period are presented in Figure 5. Within the life of this management regime that encompasses the years 2003-2006, there was very little difference in impacts on the population between an annual harvest of 275,000 and a harvest regime with 10% (300,000, 300,000 and 225,000) transfer (Fig 5). In both cases the population is expected to decline slightly from about 4.8 million to about 4.7 million (lower 60%). A harvest allowing a 20% transfer (330,000, 330,000 and 165,000) resulted in a greater decline in the population to ~4.65 million during 2004 and 2005, but with a slight recovery to 4.7 million in 2006 as a result of the lower harvest in year 3 of the management plan (Fig 5). There seemed to be very little difference in the long-term impacts of the different management approaches after 2006, when harvests remained constant at 275,000. In all cases the $N_{70}$ level would be reached by about 2014 (Table 3). If the three year management plan with a total harvest of 825,000 and a 20% transfer was repeated over 100 years, the population estimates will be slightly lower than that obtained from an annual harvest of 275,000. $N_{70}$ would be reached by 2013, instead of 2014, and the resulting population trajectory shows a pattern of marked inter-annual fluctuations (Fig. 5; Table 3).

*Increased TAC*

Increasing the TAC to 300,000, 325,00, 350,000 and 500,000 for three years and then returning to a TAC of 275,000 all resulted in population declines (Figs 6-7). There was little difference between the estimates obtained by assuming a total three year harvest of 900,000 (3 x 300,000) with either a 10% (330,000, 330,000 and 240,000) or 20% (360,000, 360,000 and 180,000) transfer among years (Fig 6). The impact of an increase in the TAC to 300,000 per year with no transfer is not shown, but was very similar to the runs allowing transfer among years. Allowing an average harvest of 300,000 per year caused the population to decline much more rapidly compared to the current TAC. Under this scenario, $N_{70}$ was reached by 2014 (Table 3). If the harvest was at 300,000 for the full period of the simulation, the population would decline more quickly, reaching $N_{70}$ by 2012 (Table 3).

A total harvest of 925,000 animals over three years (325,000 per year or 350,000, 350,000, 225,000) and then a lower TAC of 275,000 animals was also examined (Fig 7). Little difference was observed between holding the harvest level constant and allowing some transfer. In both cases the population declined to $N_{70}$ by 2013 (Table 3). The estimated replacement yield at the end of the three year period would be 230,000.

Assuming a Canadian TAC of 350,000 seals for three years, followed by a TAC that returned to 275,000 resulted in a rapid drop in the population, with abundance falling from 5.3 million animals in 2003 to 4.5 million by 2006 (Fig 7). The rate of decline slows after this as the TAC of 275,000 begins to take effect, but the population is expected to reach $N_{70}$ by 2012 (Table 3). Maintaining the harvest at 350,000 throughout the simulation increases the rate of decline with $N_{70}$ being reached in only 7 years, in 2010.

Assuming a Canadian TAC of 500,000 seals for three years, followed by a TAC that returned to 275,000 resulted in a marked drop in the population, with abundance falling from 5.3 million animals in 2003 to 4.0 million by 2006 and to $N_{70}$ by 2009 (Table 3).

## Discussion

Past management of the Northwest Atlantic harp seal has used the Replacement Yield to achieve the underlying management goal of maintaining a stable population. The current estimate of Canadian Replacement Harvest using this model is 250,000. Therefore with the exception of the PBR removals, all of the scenarios examined here will lead to a decline in the harp seal population. Using the reference points presented at the 2002 St. John's Seal Forum (DFO 2003), the northwest Atlantic harp seal could fall to the level of the first reference level ($N_{70}$ = 3.85 million) as early as 2009 in the case of a three year harvest of 500,000, or as late as 2014 in the case of an annual harvest of 275,000 animals based on the use of the lower 60% confidence limit for the population trajectory. Once the population reaches $N_{70}$ a new management approach would be triggered with the goal of reducing the probability to less than 20%, that the population will continue to decline.

Approximately 10-15 years are needed before changes in the population due to an increased Canadian harvest can be recognized in the pup surveys used to monitor the population. The Canadian harvest is directed towards young of the year animals. However, harp seal females do not have their first pup until they are five years old. Therefore, any change in harvest levels will take at least five years before their impacts are reflected in the breeding population. Currently pup production surveys are conducted every 4-5 years and so, depending upon the timing of surveys, it could be 10 years before the reduced level of pup production could be detected by a survey. Given the precision of the surveys, it is likely that at least two surveys will be needed before the reduced level of pup production is confirmed. If harvest levels have been high and the population is declining rapidly, then even with significant reductions in harvest levels, further declines in the population will continue to occur for several years (demographic momentum) until the new (protected) cohorts enter into the reproductive component of the population. This delay between the decline in the population and when it would be detected means that the lower 60% confidence interval of the population estimates would decline to the $N_{70}$ level under the majority of these scenarios before changes in pup production may be quantified.

The simulations completed for this study were designed to examine the impacts of different management strategies on the northwest Atlantic harp seal population. The results from these simulations rely on certain assumptions being satisfied. In this study, we assumed that no changes in environmental (*e.g.* food availability, ice conditions), biological (*e.g.* reproductive and mortality rates) and sociological (*e.g.* age structure of the harvest) conditions occurred over time. For the short term, these assumptions are probably reasonable. However, it is difficult to predict how conditions will change over the longer term. Changes in the age structure of the harvest via a shift towards harvesting older animals will cause the population to decline more rapidly, with a consequent need to quickly reduce the harvest. Also, ice conditions in the Gulf of St. Lawrence have been very poor in recent years. If this trend continues seals may move to other areas or the mortality rates may increase. The latter would reduce the population and resulting in a need to reduce the Canadian TAC or Greenland harvest to prevent further declines in the population.

As the population declines it might be expected that density dependent changes in reproductive rates would occur. Unfortunately, density-dependent relationships are difficult to predict and the expected density dependent changes in population parameters for this species are not as convincing as has been suggested (McLaren 2001). Model simulations for this study, assumed no change in the relatively low reproductive rates (compared to the 1970s) currently experienced by this population. If density dependent changes do occur they will compensate somewhat for the changes in harvest and population changes will be less.

The model also assumed that there is no error in reporting of catches and that the entire TAC is taken. In reality this is unlikely. For example, the basic simulations were completed using a reported Canadian 2002 harvest of 307,000 animals. This total was later revised upwards to 312,000 animals, an increase of about 2% (Table 1). Furthermore, the catch statistics from 1995-2002 show that the TAC was not taken in all years (Mean=87%, SD=26%, Range 33-112 %; Table 1). Using these approximate values (harvest ranging from 90% of quota to 102%) as an example of the impact of errors in reporting or not taking the entire quota, resulted in a lower rate of decline in the population compared to the basic assumptions used in our simulations (2002 Canadian harvest=307,000, no mis-reporting and 100% of TAC harvested). In the model with variable reporting, and variable proportion of the TAC being taken, $N_{70}$ was not reached until 2012, a year later than observed in the basic simulation, but owing to the increase in variance, the $N_{70}$ reference level was reached at a higher mean population size for the variable reporting model compared to the basic fixed model (Fig. 9). Better information on the possible levels of mis-reporting of harvests would allow us to make more realistic predictions of future population trends.

We have attempted to account for some of the uncertainty in our knowledge concerning population trends by using the lower 60% confidence limit of the estimate. As a measure of the uncertainty associated with estimates of population size, the use of the lower 60% confidence limit encourages managers to be more cautious when establishing harvests to minimize approaching reference levels. It also has as a secondary effect to reduce demographic momentum. As the time since the last survey increases, the variance associated with the estimate also increases, which takes into account the increasing uncertainty associated with our understanding of population size. Therefore, it is important to maintain regular monitoring (aerial survey, reproductive data) of this population if we wish to have a reasonable idea of population changes. Also, more extensive simulations are needed to examine the sensitivity of the model to uncertainty in model assumptions, to incorporate additional sources of uncertainty (e.g. uncertainty associated with the size and age composition of the Greenland harvest) and to assess the performance of the lower 60% confidence limit as part of the control rule process of the Precautionary Approach.

Major changes (increases) in harvest levels in the short term will necessitate major reductions in harvests down the road in order to conserve the resource. These changes will be exaggerated if environmental deterioration (*e.g*. poor ice conditions) occurs. Also, major changes or continued increases in the Greenland harvest will also have an impact on the status of the stock. We have assumed that Greenland harvests remains at the 2000 level. However, they have been increasing since the mid 1970s (Stenson *et al*. 2000) and a continuation of this trend reduces the estimates of sustainable harvests in Canada. Unfortunately, in most years the harvest statistics from Greenland are normally at least two years out of date. Management decisions based on current estimates of natural mortality and the Greenland harvest could change drastically two years later when statistics are updated. If as a consequence there is a need to reduce harvests significantly from one year to the next, this could have significant commercial impacts. If a major reduction in the northwest Atlantic harp seal population occurs, this will have an important impact on the Canadian hunt, unless the role of Greenland becomes more formalized in the management process.

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Table 2. Mean reproductive rates and standard errors used in model simulations (Healey and Stenson unpublished data)

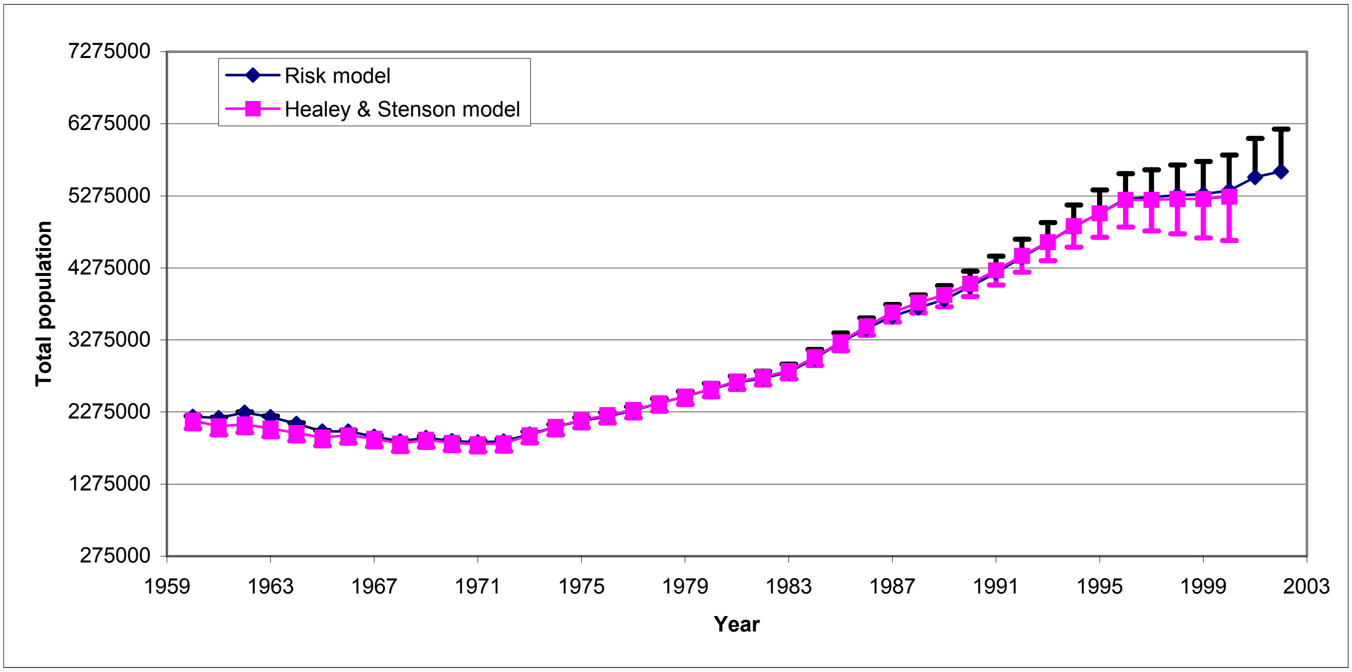
| Mean | Age | 1961 | 1962 | 1963 | 1964 | 1965 | 1966 | 1967 | 1968 | 1969 | 1970 | 1971 | 1972 | 1973 | 1974 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 4 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 |
| | 5 | 0.177 | 0.177 | 0.179 | 0.180 | 0.182 | 0.184 | 0.187 | 0.191 | 0.198 | 0.209 | 0.227 | 0.258 | 0.304 | 0.361 |
| | 6 | 0.543 | 0.542 | 0.542 | 0.542 | 0.543 | 0.543 | 0.544 | 0.545 | 0.547 | 0.551 | 0.558 | 0.573 | 0.601 | 0.645 |
| | 7 | 0.817 | 0.817 | 0.817 | 0.817 | 0.817 | 0.816 | 0.816 | 0.815 | 0.815 | 0.814 | 0.812 | 0.811 | 0.809 | 0.806 |
| | 8+ | 0.873 | 0.873 | 0.873 | 0.872 | 0.872 | 0.871 | 0.870 | 0.870 | 0.869 | 0.867 | 0.866 | 0.864 | 0.861 | 0.858 |
| S.E. | Age | 1961 | 1962 | 1963 | 1964 | 1965 | 1966 | 1967 | 1968 | 1969 | 1970 | 1971 | 1972 | 1973 | 1974 |
| | 4 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 |
| | 5 | 0.039 | 0.038 | 0.037 | 0.036 | 0.035 | 0.034 | 0.034 | 0.034 | 0.035 | 0.035 | 0.035 | 0.035 | 0.036 | 0.039 |
| | 6 | 0.048 | 0.047 | 0.045 | 0.044 | 0.043 | 0.042 | 0.042 | 0.042 | 0.042 | 0.043 | 0.044 | 0.044 | 0.043 | 0.043 |
| | 7 | 0.032 | 0.032 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 | 0.031 |
| | 8+ | 0.013 | 0.013 | 0.013 | 0.013 | 0.013 | 0.013 | 0.013 | 0.012 | 0.012 | 0.012 | 0.012 | 0.012 | 0.012 | 0.012 |
| Mean | Age | 1975 | 1976 | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 |
| | 4 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 | 0.086 |
| | 5 | 0.418 | 0.465 | 0.495 | 0.512 | 0.516 | 0.511 | 0.496 | 0.468 | 0.428 | 0.377 | 0.324 | 0.278 | 0.244 | 0.221 |
| | 6 | 0.699 | 0.751 | 0.786 | 0.803 | 0.804 | 0.791 | 0.765 | 0.725 | 0.677 | 0.626 | 0.581 | 0.544 | 0.514 | 0.490 |
| | 7 | 0.804 | 0.800 | 0.796 | 0.791 | 0.786 | 0.780 | 0.773 | 0.766 | 0.759 | 0.751 | 0.743 | 0.735 | 0.727 | 0.720 |
| | 8+ | 0.855 | 0.850 | 0.844 | 0.838 | 0.830 | 0.821 | 0.811 | 0.801 | 0.791 | 0.780 | 0.770 | 0.760 | 0.752 | 0.744 |
| S.E. | Age | 1975 | 1976 | 1977 | 1978 | 1979 | 1980 | 1981 | 1982 | 1983 | 1984 | 1985 | 1986 | 1987 | 1988 |
| | 4 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 |
| | 5 | 0.046 | 0.053 | 0.058 | 0.060 | 0.061 | 0.060 | 0.058 | 0.055 | 0.051 | 0.048 | 0.045 | 0.044 | 0.042 | 0.041 |
| | 6 | 0.047 | 0.053 | 0.059 | 0.063 | 0.063 | 0.062 | 0.061 | 0.060 | 0.062 | 0.066 | 0.068 | 0.068 | 0.066 | 0.063 |
| | 7 | 0.031 | 0.031 | 0.031 | 0.032 | 0.033 | 0.034 | 0.035 | 0.037 | 0.039 | 0.041 | 0.043 | 0.045 | 0.047 | 0.049 |
| | 8+ | 0.012 | 0.012 | 0.012 | 0.013 | 0.013 | 0.014 | 0.015 | 0.015 | 0.016 | 0.017 | 0.018 | 0.019 | 0.020 | 0.021 |

Table 2 (continued).

| Mean | Age | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 4 | 0.086 | 0.086 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 | 0.085 |
| | 5 | 0.207 | 0.199 | 0.195 | 0.192 | 0.190 | 0.190 | 0.189 | 0.189 | 0.189 | 0.189 | 0.189 | 0.189 | 0.189 | 0.189 |
| | 6 | 0.469 | 0.450 | 0.434 | 0.420 | 0.409 | 0.399 | 0.390 | 0.383 | 0.377 | 0.372 | 0.367 | 0.367 | 0.367 | 0.367 |
| | 7 | 0.713 | 0.707 | 0.701 | 0.695 | 0.691 | 0.686 | 0.683 | 0.679 | 0.676 | 0.673 | 0.671 | 0.671 | 0.671 | 0.671 |
| | 8+ | 0.737 | 0.732 | 0.727 | 0.723 | 0.719 | 0.717 | 0.714 | 0.712 | 0.710 | 0.709 | 0.707 | 0.707 | 0.707 | 0.707 |
| S.E. | Age | 1989 | 1990 | 1991 | 1992 | 1993 | 1994 | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 |
| | 4 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 | 0.016 |
| | 5 | 0.039 | 0.038 | 0.038 | 0.038 | 0.038 | 0.039 | 0.041 | 0.042 | 0.044 | 0.046 | 0.049 | 0.049 | 0.049 | 0.049 |
| | 6 | 0.060 | 0.058 | 0.056 | 0.056 | 0.057 | 0.058 | 0.060 | 0.063 | 0.066 | 0.071 | 0.075 | 0.075 | 0.075 | 0.075 |
| | 7 | 0.052 | 0.054 | 0.056 | 0.058 | 0.059 | 0.061 | 0.063 | 0.064 | 0.066 | 0.067 | 0.069 | 0.069 | 0.069 | 0.069 |
| | 8+ | 0.022 | 0.022 | 0.023 | 0.024 | 0.024 | 0.024 | 0.025 | 0.025 | 0.026 | 0.026 | 0.026 | 0.026 | 0.026 | 0.026 |

Table 3. Harvest scenarios, estimated replacement yield after a three year harvest period, a new quota assumed following the current management plan and the estimated year that the population would decline to $N_{70}$ under the assumed harvest regime.

| Total harvest over 3 year plan ('000s) | Annual Harvest 2003-2005 ('000s) | Replacement harvest after 3 years (000s) | Harvest level in 2006 and beyond (000s) | Year $N_{70}$ reached |
|---|---|---|---|---|
| 825 | 275, 275, 275 | 250 | 275 | 2014 |
| | 300, 300, 225 | 235 | 275 | 2014 |
| | 330, 330, 165 | 235 | 275 | 2014 |
| | 330, 330, 165 | 235 | 3 year plan repeated | 2013 |
| 900 | 300, 300, 300 | 225 | 275 | 2014 |
| | 330, 330, 240 | 230 | 275 | 2013 |
| | 360, 360, 180 | 230 | 275 | 2013 |
| 925 | 325, 325, 325 | 230 | 275 | 2013 |
| | 350, 350, 225 | 230 | 275 | 2013 |
| | 300, 300, 300 | 225 | 300 | 2012 |
| 990 | 330, 330, 330 | 220 | 330 | 2011 |
| 1,050 | 350, 350, 350 | 215 | 275 | 2012 |
| | 350, 350, 350 | 215 | 350 | 2010 |
| 1,500 | 500, 500, 500 | 175 | 275 | 2009 |



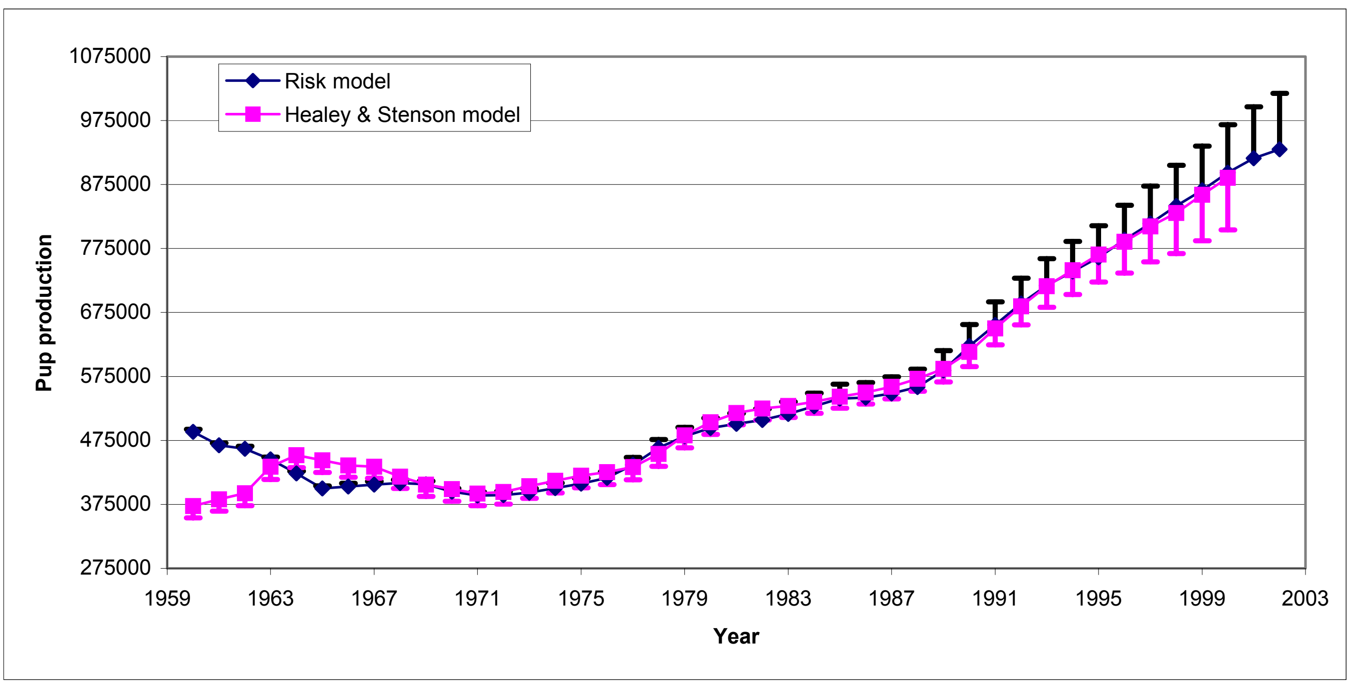




Figure 1. Pup production and total population trajectories of Northwest Atlantic harp seals from 1960 to 2002 using the model formulation in Healey and Stenson (2000) with mean and mean minus 1 SE and a modified formulation (mean and mean plus 1 SE.).

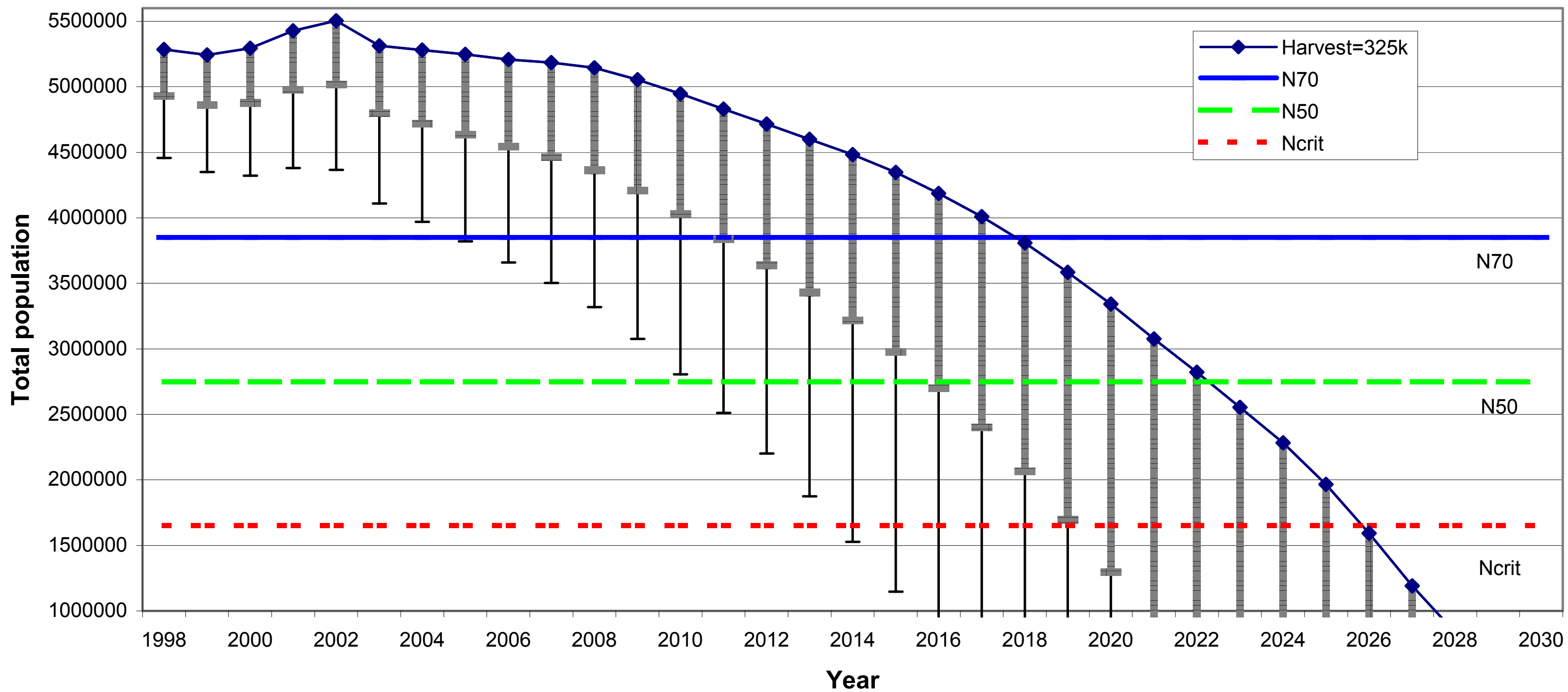



Figure 2. Simulated population trajectories (mean, Lower 60% C.I. and lower 95% C.I.) for a Canadian harvest of 325,000 seals per year. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

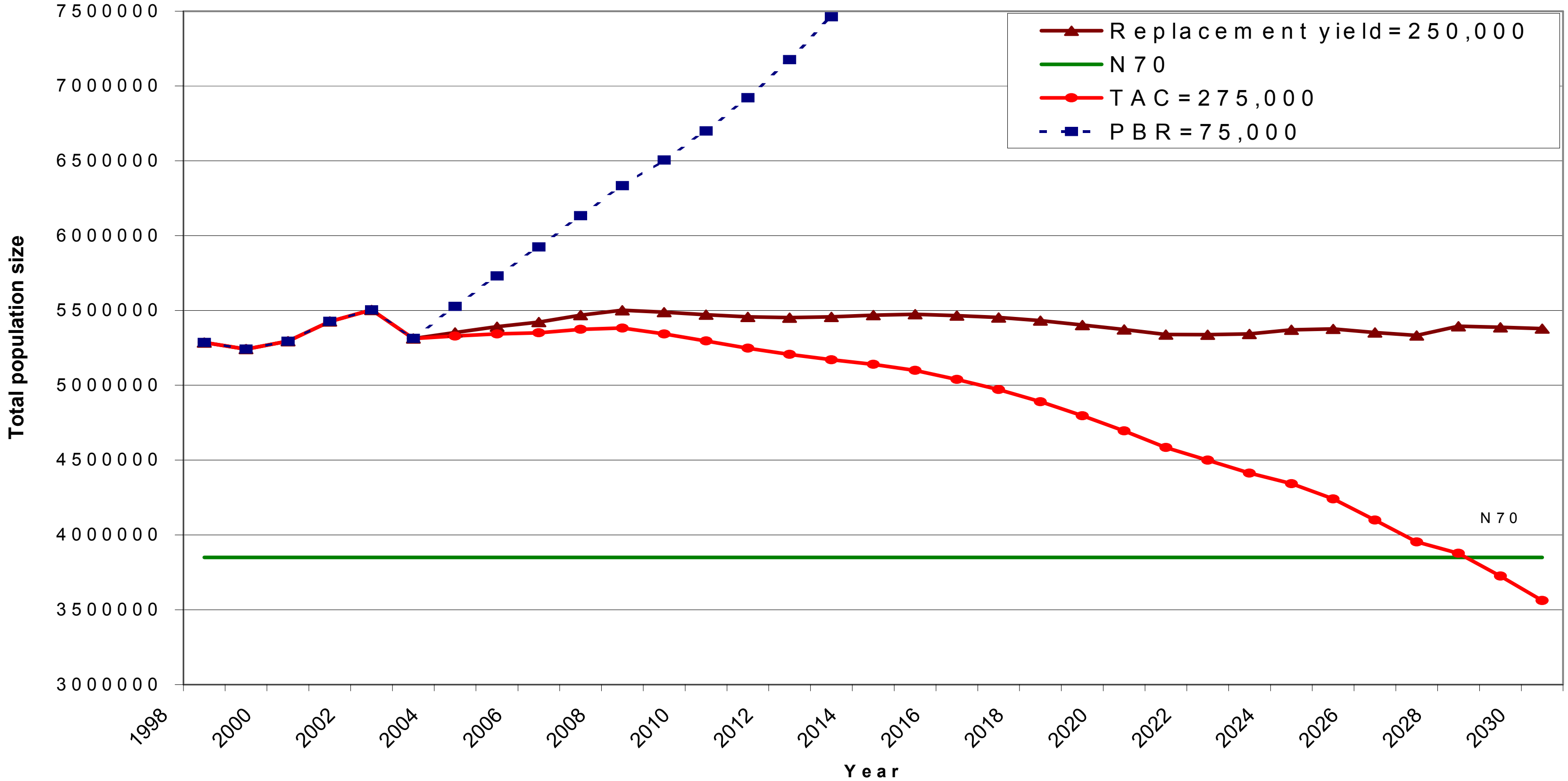



Figure 3a. Estimated mean population trajectories of NW Atlantic harp seals illustrating the impacts of PBR (75,000), current TAC (275,000) and new replacement yield calculations (250,000) harvest levels beginning in 2003. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

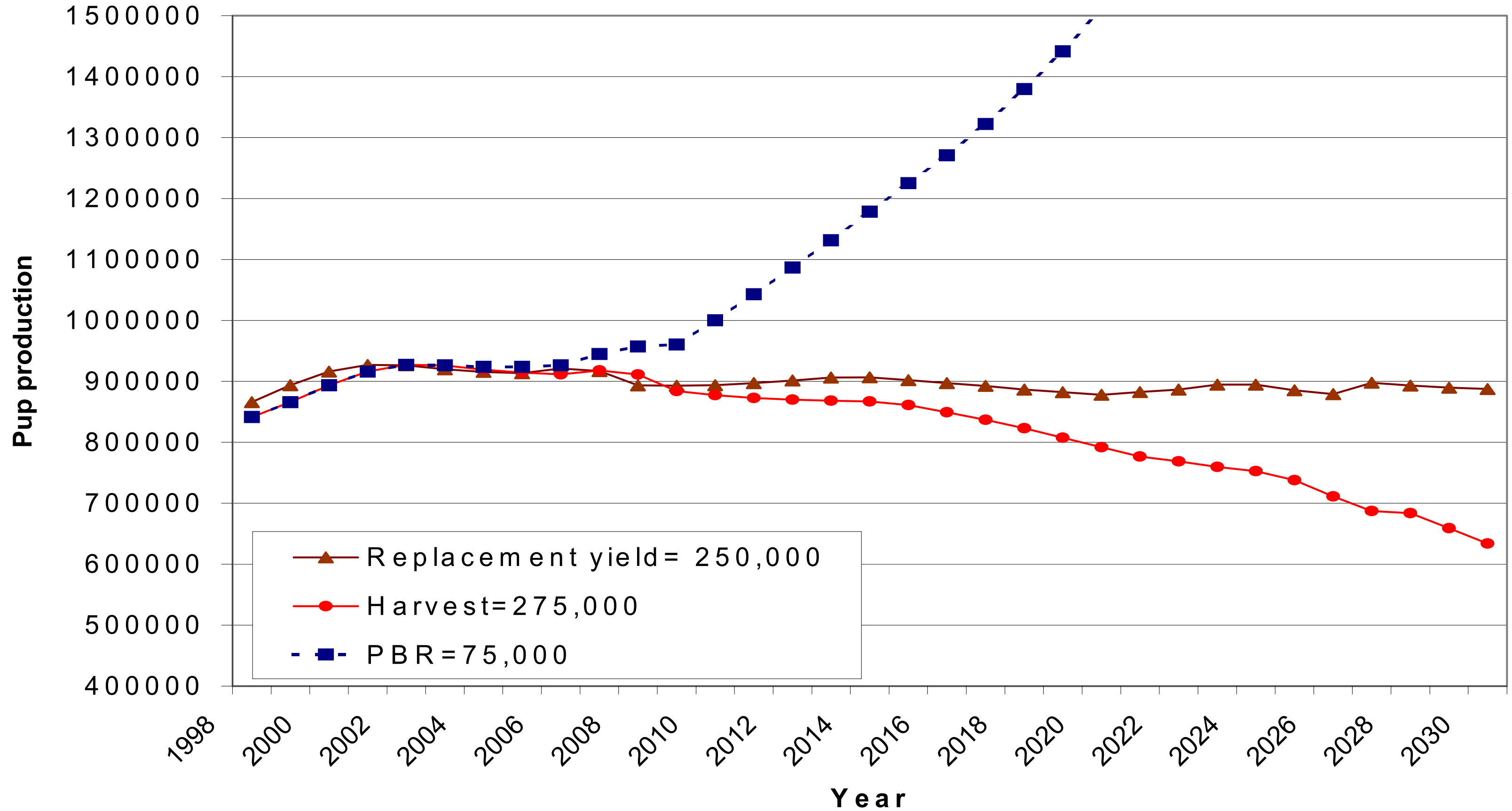



Figure 3b. Estimated mean pup production trajectories of NW Atlantic harp seals illustrating the impacts of PBR (75,000), current TAC (275,000) and new replacement yield calculations (250,000) harvest levels beginning in 2003. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

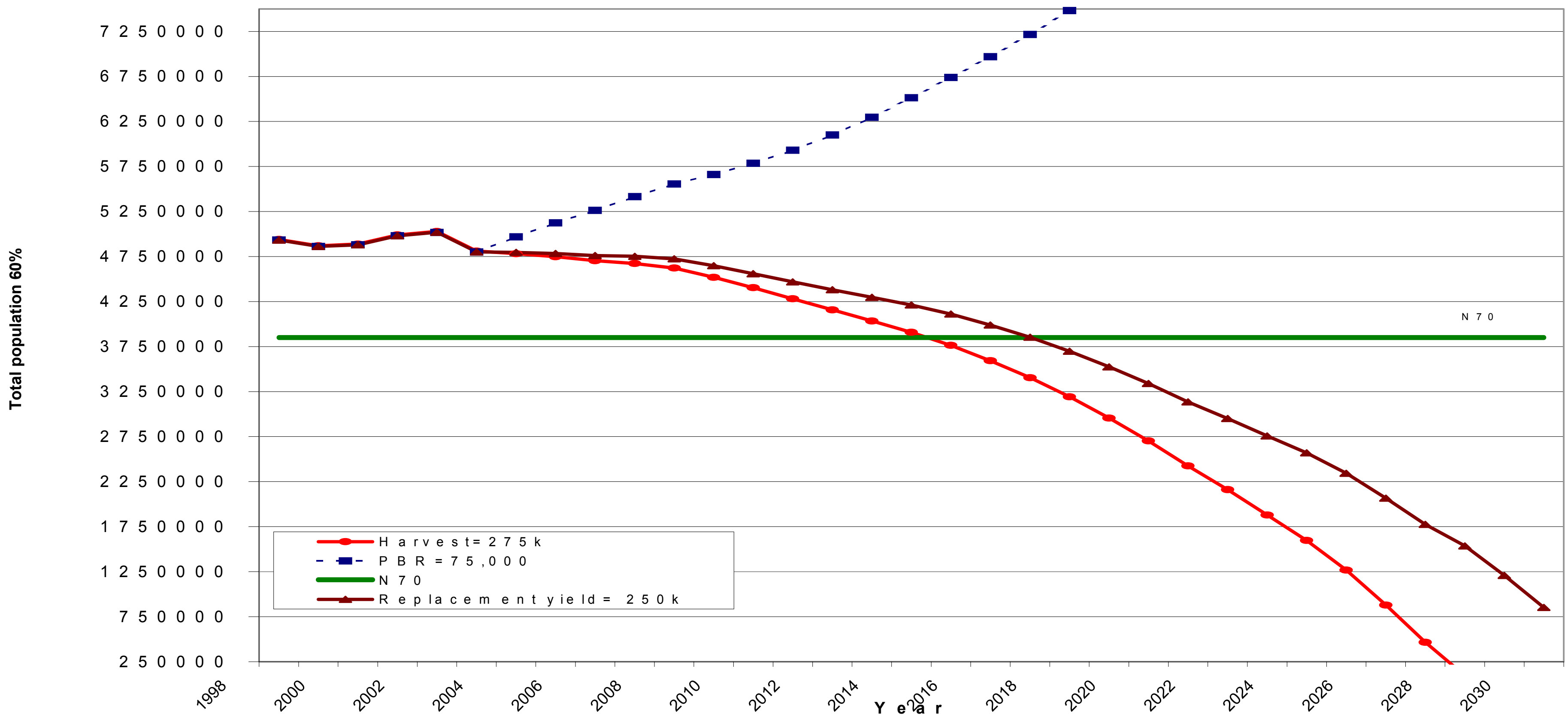



Figure 4. Estimated lower 60% C.I. population trajectories of NW Atlantic harp seals illustrating the impacts of PBR (75,000), current TAC (275,000) and new replacement yield calculations (250,000) harvest levels beginning in 2003. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

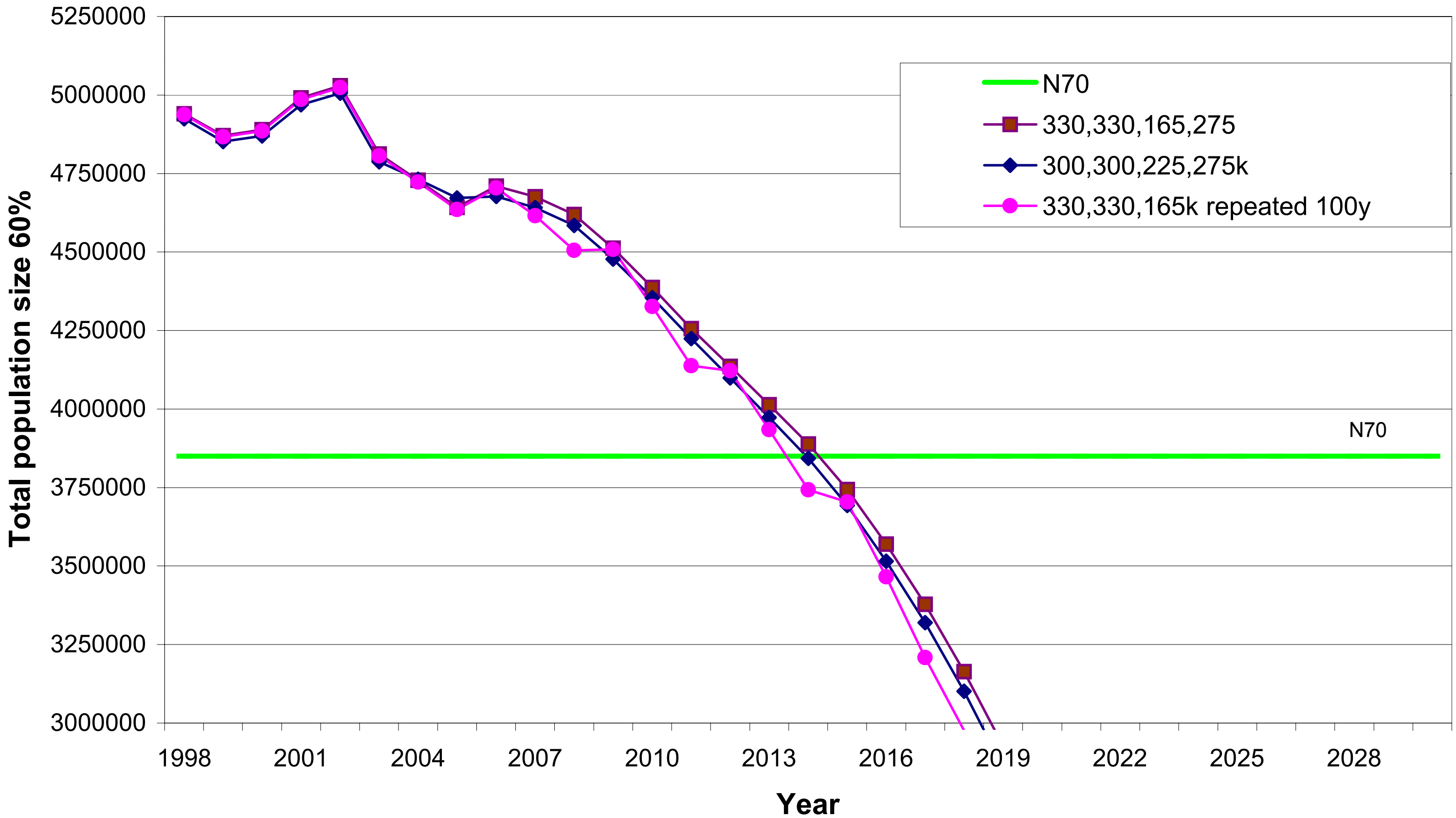



Figure 5. Estimated lower 60% C.I. population trajectories of the NW Atlantic harp seal population under a three year management plan based upon an annual TAC of 275,000 with a 10% transfer among years (300,000, 300,000, and 225,000) and a 20% transfer (330,000, 330,000, and 165,000) for the years 2003, 2004, and 2005. Harvest in 2006 and beyond are assumed to remain at 275,000. A third plot maintains a three year pattern of 330,000, 330,000, and 165,000 for the entire simulation. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

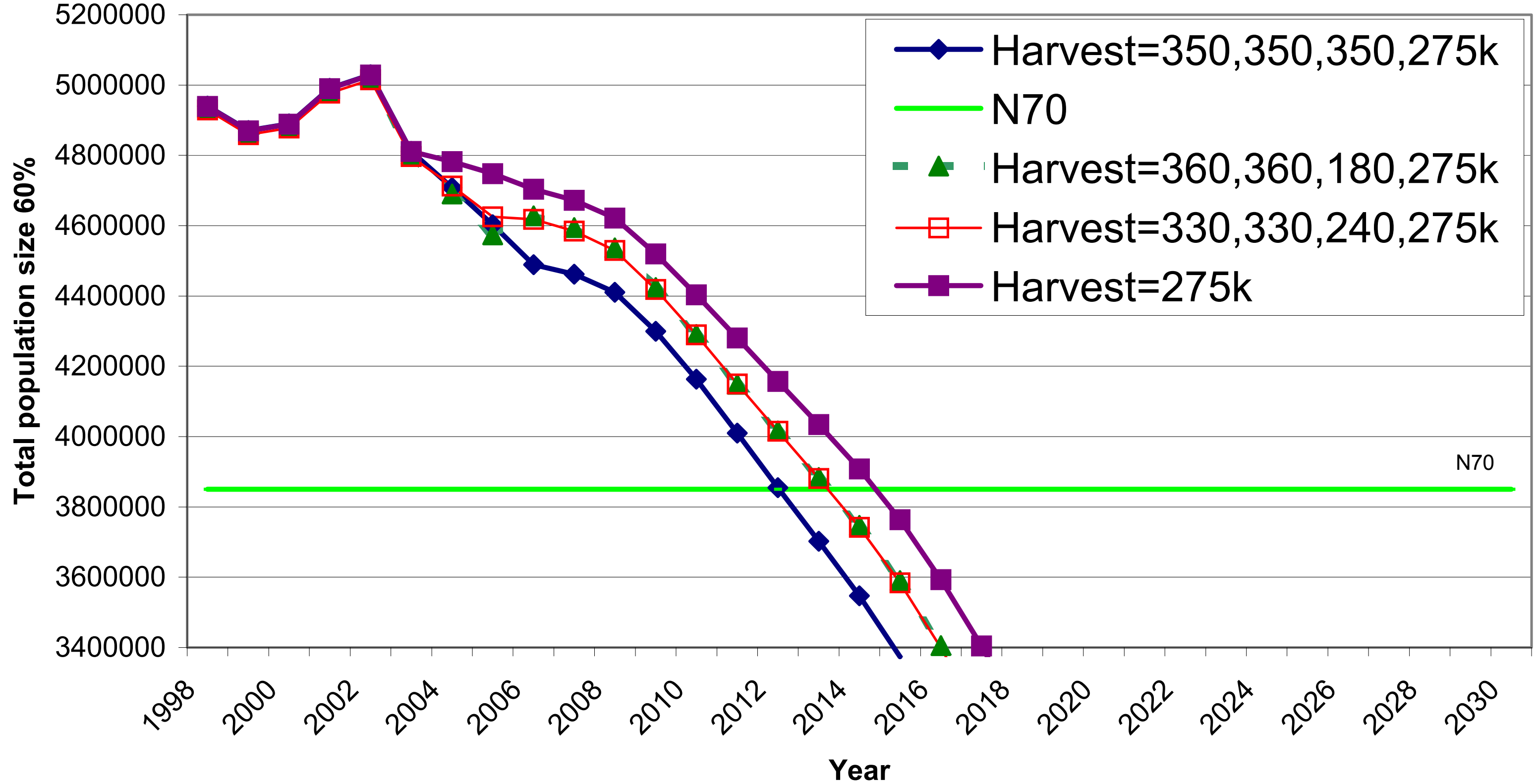



Figure 6. Estimated lower 60% C.I. population trajectories of the NW Atlantic harp seal population under a three year management plan based upon an annual TAC of 300,000 with a 10% transfer among years (330,000, 330,000, and 240,000) and a 20% transfer (360,000, 360,000, and 180,000) for the seasons of 2003, 2004, and 2005. Also a constant harvest of 350,000 for 3 years and the base run (275,000) are illustrated. Harvest in 2006 and beyond are assumed to remain at 275,000. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

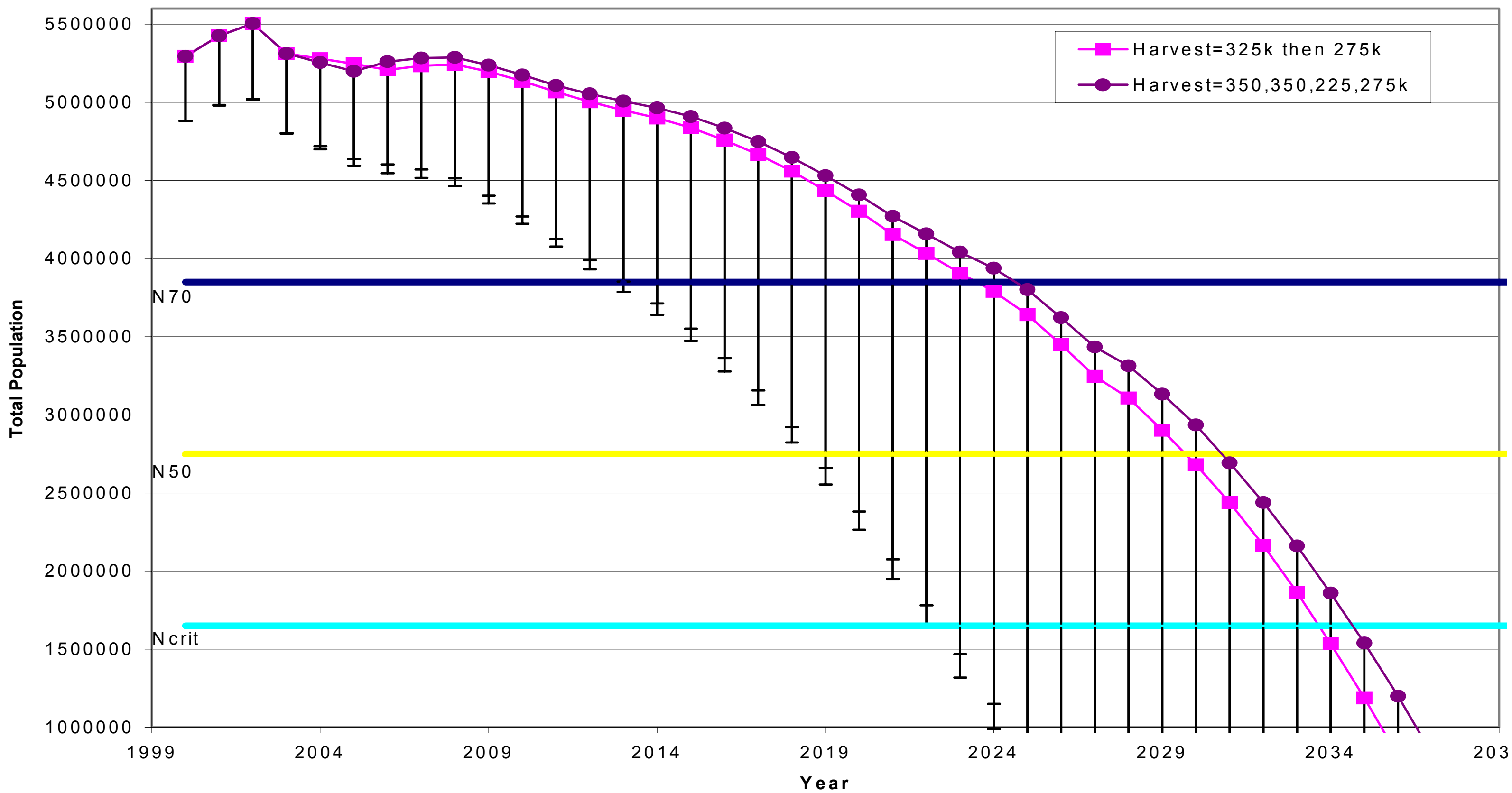



Figure 7. Mean and estimated lower 60% confidence limit population trajectories for NW Atlantic harp seal population under harvesting regimes of 325000, 325,000, and 325,000 seals, followed by an annual quota of 275,000 seals, or an annual harvest of 350,000, 350,000 and 225,000 seals followed by an annual quota of 275,000. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

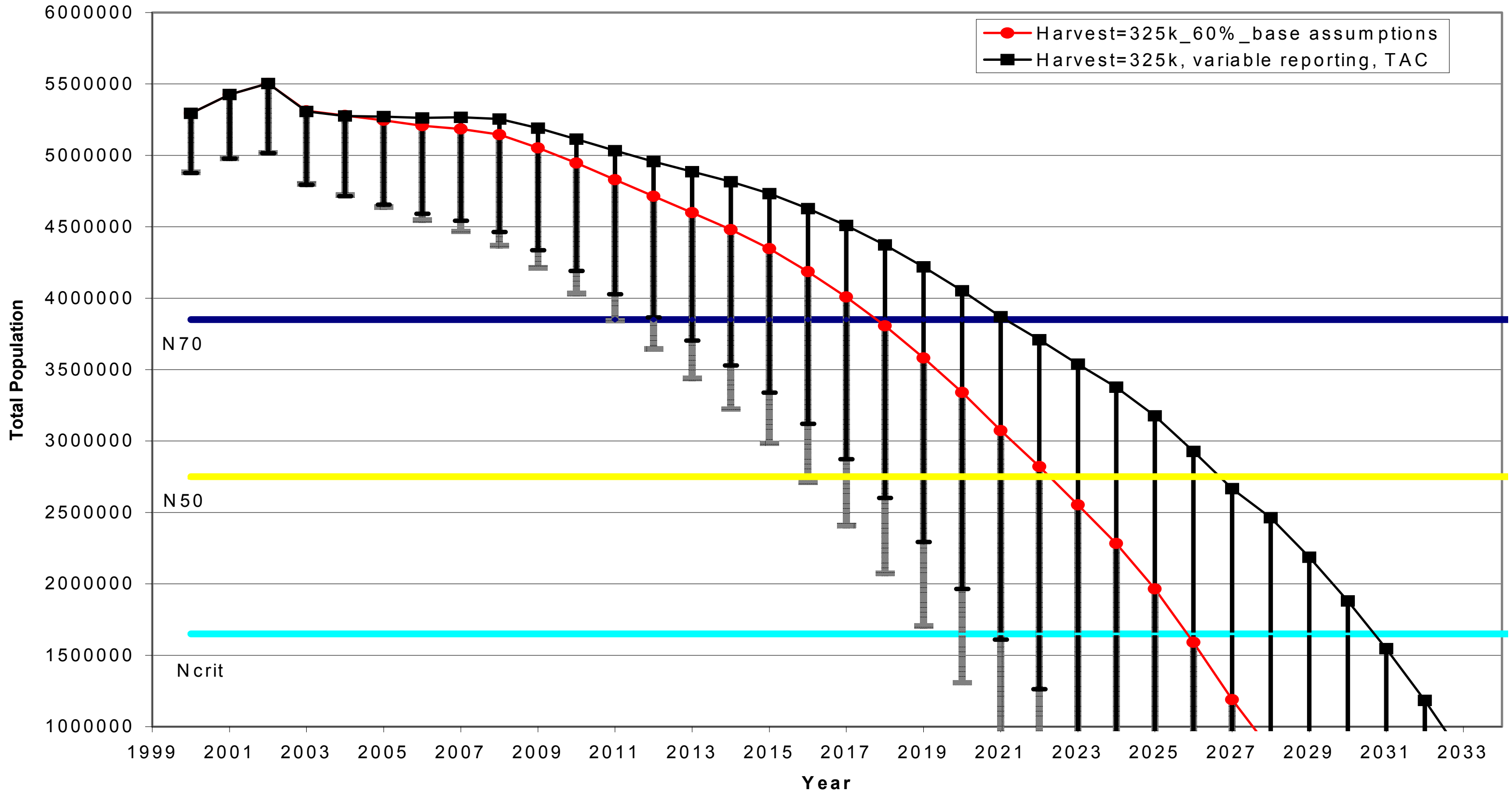
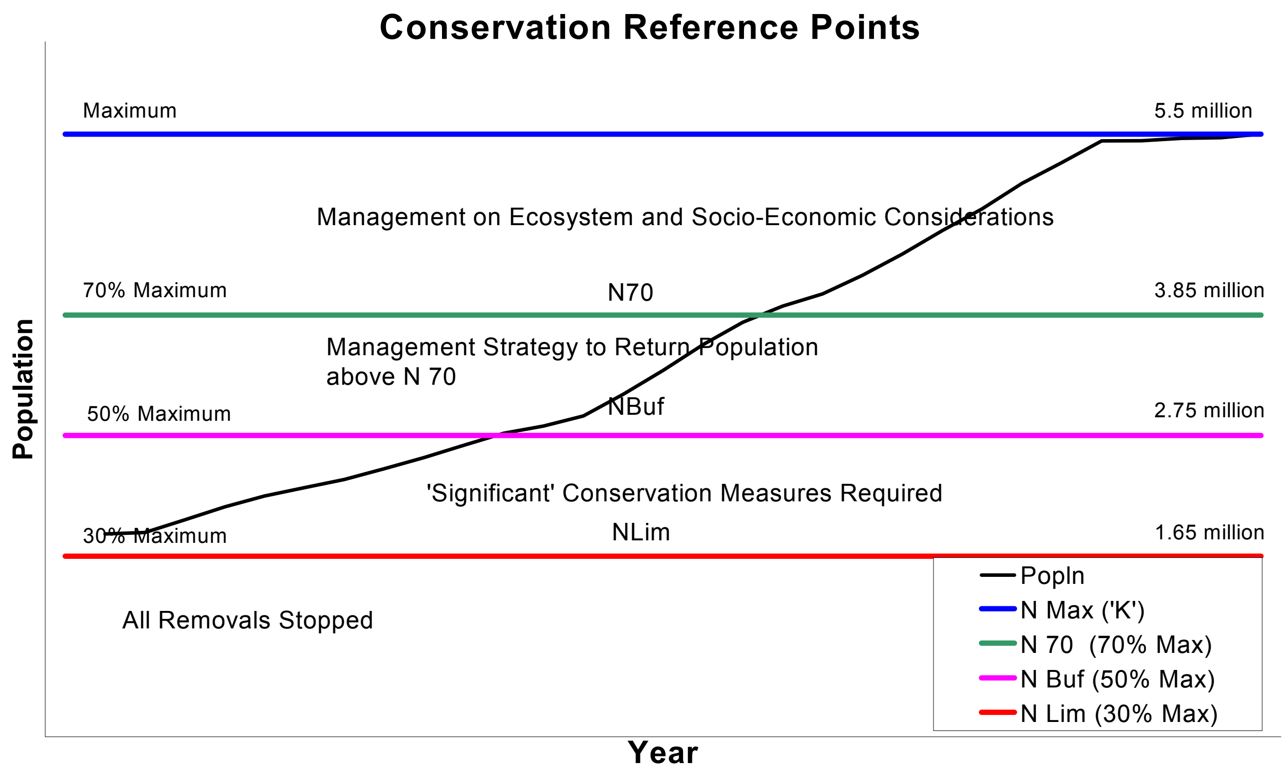



Figure 8. Predicted lower 60% C.I. population trajectory for 2 forms of the @Risk model. Both models assume a constant TAC of 325,000 beginning in 2003. The base form assumes a 2002 harvest of 307,000, no mis-reporting and the entire TAC is taken each year. The variable reporting and variable proportion TAC model assumes a 2002 harvest of 312,000, 90-102% of landings are reported and an average of 87% (SD=26) of the TAC is taken. The predicted trajectories assume that no changes in model parameters or assumptions occur over the period of the projection.

Appendix 1. Reference Points Discussed at Seal Forum, Nov 2002, St. John's, Newfoundland (Hammill and Stenson 2003)




Reference points for incorporating the Precautionary Approach into seal management. The points are set as percentages of the maximum or pristine population size, whichever is higher (i.e.. the largest harp seal population known is 5.5 million). Consequently $N_{70}$ (referred to as Nconservation at the St. John's forum) is set at 70% of the maximum or 3.85 million. The next lower precautionary limit is $N_{Buf}$ (previously referred to as Nminimum), which could also be referred to as $N_{50}$, set at 50% of the maximum population or 2.75 million. The lowest reference point, $N_{lim}$ (also referred to as Ncritical) is set at 30% of the maximum or 1.65 million. As the population moves below each successively lower limit level, harvest levels should be more conservative to reduce the risk that the decline will continue.