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A preliminary evaluation of the performance of the Canadian management approach for harp seals using simulation studies.

Une évaluation préliminaire du rendement de l'approche de gestion canadienne des phoques du Groenland à l'aide d'études de simulation

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ABSTRACT

The Precautionary Approach (PA), provides a framework within the context of fisheries management which attempts to take into account the uncertainties related to the status of the resource when setting harvest levels. In 2003, Fisheries and Oceans Canada (DFO) adopted a precautionary framework for the management of seals in Atlantic Canada. This framework is PA-like in that it identifies limit and precautionary reference levels, and explicitly incorporates uncertainty in our estimates, but the behaviour of the framework under simulated uncertainty in the parameters has not been examined. For marine mammals two basic frameworks have been developed (International Whaling Commission (IWC), Potential Biological Removal (PBR)), and these Management Procedures have been tested. Although both frameworks provide a guide to the simulation approach, they are not appropriate for management within the Canadian context. The model used to assess the Northwest Atlantic harp seal examined the impact of extending the modelled population from 1960 back to 1952, adjusting the within-year correlation in reproduction among cohorts, and assessing how alternative methods of applying the management plan impact the estimated population. Simulations to examine the impact of an unknown mortality of young animals related to poor ice conditions were also conducted. Failure to consider ice-related mortality in an Assessment model has a significant impact on perceptions of the resource. The preliminary simulations indicate that the current management framework is robust to avoiding a decline in the resource below the spawner stock biomass limit reference point (B_{lim}), thus satisfying a conservation objective, but can result in harvest closures after 10-20 years if the entire harvest was taken in every year. Adjustments in the approach are needed if industry would like to see long-term stability in harvest levels as a management objective.

RÉSUMÉ

L’approche de précaution (AP) offre un cadre dans le contexte de la gestion des pêches qui tente de tenir compte des incertitudes associées à l’état de la ressource lors de l’établissement des taux de prélèvement. En 2003, le ministère des Pêches et des Océans (MPO) a adopté un cadre de précaution pour la gestion des phoques du Canada atlantique. Ce cadre a un rapport à l’approche de précaution dans la mesure où il définit des seuils de référence limites et de précaution et intègre explicitement aux estimations un facteur d’incertitude. Cependant, le comportement du cadre dans un contexte d’incertitude simulée des paramètres n’a pas été vérifié. Deux cadres de base ont été mis au point pour les mammifères marins (IWC (International Whaling Commission) et RBP (Retrait biologique potentiel)) et ces méthodes de gestion ont été mises à l’essai. Bien que les deux cadres puissent servir de guide pour l’approche de simulation, ils ne conviennent pas à la gestion dans le contexte canadien. Le modèle qui a servi à évaluer les phoques du Groenland de l’Atlantique Nord-Ouest a examiné l’effet d’élargir la population modélisée en reculant de 1960 à 1952, de rajuster la corrélation dans l’année de la reproduction des cohortes et d’évaluer l’incidence d’autres méthodes d’application du plan de gestion sur la population estimée. On a également procédé à des simulations pour examiner l’incidence d’une mortalité inconnue des jeunes animaux associée au mauvais état des glaces. Le fait de ne pas tenir compte de la mortalité causée par les glaces dans un modèle d’évaluation a une incidence importante sur la perception de la ressource. Les simulations préliminaires indiquent que le cadre de gestion en vigueur est robuste pour ce qui est d’éviter une diminution de la ressource sous la B_{lim} (point de référence limite pour la biomasse génitrice) répondant ainsi à un objectif de conservation, mais pourrait entraîner la fermeture de la récolte après 10 à 20 ans si toute la récolte était prise chaque année. L’approche doit être redéfinie si l’industrie souhaite adopter comme objectif de gestion la stabilité à long terme des seuils de récolte.

INTRODUCTION

Historically, resource management has involved management actions that have been decided through a process of brokerage and negotiation in which a variety of biological, economic, operational, and political factors were considered (Cooke, 1995). Within this framework, resource scientists are required to provide regular advice to managers based on biological assessments of the state of exploited species. Because the indices used are usually subject to considerable variance, assessment of the current state of the stock is subject to considerable uncertainty (Cooke 1999), which in the past has led managers to require proof that populations or resources were in difficulty before action was taken (Taylor et al. 2000). Unfortunately, by the time the extent of the damage could be determined, populations often had already suffered serious harm. The collapse of northwest Atlantic cod (*Gadus morhua*) stocks and many large whale populations are examples where traditional management approaches have failed (DFO 2003, Baker and Clapham 2004).

The Precautionary Approach (PA), provides a framework within the context of fisheries management which strives to be more cautious when information is less certain, does not accept the absence of information as a reason for not implementing conservation measures, and defines, in advance, decision rules for stock management when the resource reaches clearly stated reference points (Punt and Smith 2001). An important aspect of PA is that it involves stakeholders in the development of the management approach and is transparent. To date, initiatives to implement PA have been mostly driven by fishery science with its traditional emphasis on fish population dynamics and reference points with an emphasis on protecting the resource and the environment they live in (Hillborn et al. 2001). However, it has been argued that PA is much more than implementation of precautionary harvest control rules. It needs to also include a process that facilitates communication and fosters cooperation among the different sectors involved (Hillborn et al. 2001). In developing a PA framework, the objective of management is to achieve benefits, rather than to avoid disasters (DFO 2003). Within the ICES framework, the recommendation has been that for fisheries to be within safe biological limits, there should be a high probability that the spawning stock biomass remains above the limit reference point (B_{lim}). However, because of error of estimation, management actions should be taken before the limits are approached, if the limit is to be avoided with high probability. ICES has therefore defined precautionary thresholds at which management actions should be taken to ensure a high probability that the resource does not fall below the limit reference point (ICES 2001).

The key component within PA is to avoid the condition of irreversible harm to the resource. The approaches developed will depend in part on the current status and knowledge of the resource. Hence more aggressive risk tolerant approaches might be developed in cases where the knowledge on the status of the resource is considered to be good (Data Rich), whereas more adverse approaches might be developed for depleted or resources where the knowledge is considered poor (Data Poor)(Hammill and Stenson 2007). In this document, we restrict our discussion to the Data Rich case.

The United Nations *Agreement on Straddling and Highly Migratory Fish Stocks* (UNFA), which came into force in 2001, commits Canada to use the PA in managing straddling stocks as well as, in effect, domestic stocks. In 2003, the Privy Council Office, on behalf of the Government of Canada published a framework applicable to all federal government departments that set out guiding principles for the application of precaution to decision making about risks of serious or irreversible harm where there is a lack of full scientific certainty. In May 2006, DFO Science released a paper outlining the minimal requirements, from a science perspective, for a harvest strategy to be compliant with the Precautionary Approach (DFO 2006). The primary components of the generalized framework: Reference points and stock status zones (Healthy, Cautious and Critical). Harvest strategy and harvest decision rules and the need to account for uncertainty and risk when developing reference points and developing and implementing decision rules. The stock status zones are created by defining the Limit Reference Point (LRP) at the Critical:Cautious zone boundary, and an Upper Stock Reference Point (USR) at the Cautious:Healthy zone boundary and the Removal Reference for each of the three zones (Fig. 1). Under this framework, the USR is the stock level threshold below which removals must be progressively reduced in order to avoid reaching the LRP. Therefore at a minimum, the USR must be set at an appropriate distance above the LRP to provide sufficient opportunity for the management system to recognize a declining stock status and sufficient time for management actions to have effect. Secondly, the USR can be a target reference point (TRP) determined by productivity objectives for the stock, broader biological considerations and social and economic objectives for the fishery. A TRP is a required element under UNFA and in the FAO guidance on the application of the PA, as well as ecocertification standards based on it, such those of the Marine Stewardship Council and may also be desireable in other situations. Tailoring the generalized three-zoned decision framework for an individual stock and applying it involves a number of steps. The preferred approach is always to have reference points and harvest rules based on the best information available on stock biology and fishery characteristics while taking into account the limitations of the available data. With reference to the General Framework, a stock can be considered to be in the critical zone if the mature biomass, or its index, is less than or equal to 40% of $B_{\text{Maximum Sustainable Yield (MSY)}}$. In other words: *Biomass* $\leq$ *40%* $B_{MSY.}$ A stock can be considered to be within the cautious zone if the biomass, or its index, is higher than 40% of B_{MSY} but lower than 80% of B_{MSY}. In other words: *40%* B_{MSY} < *Biomass* < *80%* $B_{MSY.}$ A stock is considered healthy if the biomass, or its index, is higher than 80% of B_{MSY}. In other words: *Biomass* $\geq$ *80%* $B_{MSY.}$ This framework is in line with practices and standards used internationally, such as in New Zealand and U.S.A., and is consistent with the language found in various international agreements.

UNFA, FAO, and Canada have developed their frameworks around the concept of MSY. However, operationally, estimating MSY is difficult to determine due to uncertainties in quantifying population-related density-dependent changes and the impacts of poorly-understood ecosystem effects (ICES 2001). ICES (2001) has struggled with how to determine B_{lim} and many methods have being tried/used, including setting B_{lim} at a level termed B_{loss}, which is the lowest observed spawning stock size; at B_{20}, which is the spawning stock biomass corresponding to 20% of the unexploited biomass and also using some method based on stock-recruitment curve relationships. In another organization, the difficulties in using MSY was one of the factors that led to the International Whaling Commission (IWC) to reject their 'New Management Plan' approach in 1991 and to develop a new approach referred to as the 'Revised Management Plan' or 'RMP' (Cooke 1995). Under RMP, the IWC has adopted a different approach with the Catch Limit algorithm, which closed harvesting if the population fell below 54% of carrying capacity (K).

While PA has been applied to fisheries to minimize the probability of irreversible harm, the conservation movement has developed a parallel approach to minimize the risk of extinction with quite different objectives that are rooted in different types of tolerance to risk between the two disciplines. Initially, it could be considered that there was a difference between B_{lim} , which is concerned with management of commercial fisheries which might impair the status of the resource for continued harvesting (i.e., commercial extinction), and the levels identified in conservation biology which was concerned with the continued persistence of the species or designated unit (i.e., biological extinction or extirpation). However, conservation biology appears to have occupied this 'space' between levels necessary for continued economic viability of harvesting versus conservation from extirpation, through the introduction of control rules such as rate of decline over a specified time. For example, the Committee on Species and Endangered wildlife in Canada (COSEWIC) identifies that a species is Endangered if the population declined by 70% (http://www.cosewic.gc.ca/eng/sct0/assessment_process_e.cfm). However, using this approach it is possible that a population may be abundant, yet still be of conservation interest within the context of an Endangered species under these criteria. Although identification of possible concern is important to the conservation biology community, 'false alarms' can be disruptive to the fisheries management community (Rice and Legacé 2007).

The Northwest Atlantic harp seal is harvested commercially in Atlantic Canada and for subsistence purposes in Arctic Canada, and Greenland. Harp seals are also taken as bycatch in commercial fisheries. Until the early 2000's the harp seal commercial hunt was managed using a replacement yield approach, where the management objective was to adjust harvests so that over a 10 year period a constant population would be maintained. In 2001, the Eminent Panel on Seal Management concluded that use of replacement yield as a means of setting Total Allowable Catches for the Northwest Atlantic harp seal population had a high risk of leading to a decline in the population if the TAC was taken in full in every year (McLaren et al. 2001). This was because the replacement yield was established based on mean population size, which had a 50% probability that the population was higher than the mean, as well as a 50% probability that the true population size was less than the mean. The Panel recommended defining a set of control rules to be used to set the TAC and the way in which it can be taken, and a set of Reference Points that could be used to monitor the effectiveness of management and suggested that the probability that the exploited population will fall below a Limit Reference Point must be kept as low as possible. Following the Eminent Panel report, Fisheries Management requested that science establish an approach that incorporated control rules, and reference and precautionary levels that could be used to manage seals in Atlantic Canada. A framework was developed (Hammill and Stenson 2003a,b; 2007), presented to industry, and has been accepted as an approach to manage seals in Atlantic Canada since 2003 (DFO 2008a, b).

Ideally, the reference points would be selected after extensive simulation studies are completed. They must also be relatively easy to understand and acceptable to stakeholders. However, simulation studies are time-consuming and can result in delays in implementing the precautionary approach. For example, it took 12 years to complete the implementation trials under the International Whaling Commission Revised Management Plan for western North Pacific minke whales (Punt and Donovan 2007). However, frustration within the Scientific Committee led to the development of a framework on how the management procedure could be examined and indicated that evaluations could be completed within a much shorter period of 2 years (Punt and Donovan 2007). In the case of the Northwest Atlantic harp seal, a request for reference levels was made in 2002 during a period of intense harvesting, thus following the IWC approach, the simulation process could at best have been implemented prior to the 2006 hunting season, or at worst would still be ongoing throughout a decade of high levels of harvest. In developing the Atlantic seal management framework, it was felt that a relatively clear structure would be more acceptable to managers and stakeholders. The framework developed within the International Union for the Conservation of Nature (IUCN) and COSEWIC for assessing the status of populations (IUCN 2001, COSEWIC 2006) was identified as the basis for such an approach. Using the COSEWIC/IUCN framework, the first precautionary reference point, N_{Buf}, could be set at 70% (N70) of the maximum observed (or inferred) population size and $N_{Critical}$ could lie at 30% (N30) of the maximum. Setting N30 as a reference limit level would, in the case of harp seals, also be comparable to setting B_{loss} under ICES. Although this approach lacked a strong mathematical basis for its structure, it moved the debate away from a concept that is in itself controversial (i.e., MSY), and instead shifted the focus towards discussions surrounding benchmarks that were clearly defined, and in keeping with magnitudes of change in species abundance (30%, 50% and 70%) that are considered important enough to be of concern (Hammill and Stenson 2007, Stenson and Hammill 2008). Borrowing the framework from the conservation literature also served another purpose related to the challenges in managing fisheries, which is to force industry and managers to reduce harvesting in the face of resource declines. The failure of current practices in these sectors (and science methods of operation) to respond to decline in the resource are increasingly recognized as a major factor in the overexploitation of modern fisheries (Daw and Gray 2009). Therefore, if the framework was built within the concept of the conservation literature, a decline of 70% would result in a population of approximately 1.8 million animals, which would, by most measures, still be considered 'healthy', but would also trigger a review within the conservation biology community and provide an additional incentive for managers and industry to reduce harvesting. This framework was presented to stakeholders in 2003 and adopted into the management framework in that year (DFO 2003).

The commercial harvest of harp seals is limited through a management plan that outlines management objectives, catch levels, methods of hunting, and seasonal and regional closures (DFO 2006). Given the difficulties and challenges in identifying where precautionary and limit reference levels should be, the objective of this plan is to maintain an 80% probability that the population remains above a precautionary reference level, called N70, that is set at 70% of the largest population seen. Currently N70 is set at 4.1 million animals (DFO 2006, Hammill and Stenson 2007). If the population falls below N70, the catches are reduced to ensure an 80% probability that the population will return above N70 within 10 years (DFO 2006). A second precautionary reference level, set at N50, was established as a warning to managers that the resource was declining. The limit reference level has been set at N30, where commercial harvesting was to cease. The management approach currently in use by the Department fulfills some of the requirements for a PA approach that is the setting of management objectives, reference and precautionary levels and decisions as to what management actions should occur if the population falls below the precautionary level N70. However, to be complete, simulation testing of the framework is needed and if necessary, adjustments made to the approach (Hammill and Stenson 2007). Such testing is needed to ensure that the framework is able to balance mutually conflicting objectives which aim to maximize catches against minimizing the risk of substantial depletion putting the resource in jeopardy within the context of plausible bounds of error surrounding our understanding of the state of the resource (Butterworth 2007).

Here we examine the behaviour of the management framework and how it responds when the assumptions are not satisfied. These analyses do not include results from the 2008 assessment. They are based on data from the 2005 assessment, with catch and reproductive rate data extrapolated forward. The objective was to present the response of a model population subjected to changes in environmental, management, or model conditions. This represents a work in progress, since only a few scenarios have been examined to date. Further runs will be completed in the coming months.

MATERIALS AND METHODS

Northwest Atlantic harp seal population size is estimated using a two-parameter population model (Hammill et al. 2009). Two models are used. The first model, called here the fitting model, describes past changes in the population by incorporating information on removals from the population, including incidental catch from fisheries, struck and loss and non-reporting and age specific reproduction rates, fitted to independent estimates of pup production by adjusting the starting population and adult mortality rates to obtain an estimate of the population's trajectory over time. The fitting model produces a set of initial population (alpha) and adult mortality (M) estimates. This set of alpha and M is then used to project the trajectory of the population into the future (projection model) to evaluate the impact of different harvest or environmental factors on future trends in the population.

MODEL STRUCTURE

The basic model has the form : $n_{a,t} = ((n_{a-1,t-1} * w) - c_{a-1,t-1})\, e^{-(\gamma)m}$ (1)

for a =1

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

for $1 < a < A$,

$$n_{A,t} = \left[\left(n_{a-1,t-1} + n_{A,t-1}\right) e^{-M/2} - c_{a-1,t-1}\right] e^{-M/2} \tag{3}$$

for $a = A$, where A-1 is taken as ages A-1 and greater, and for $a = 0$;

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

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.
γ = a multiplier to allow for higher mortality of first year seals. Assumed to equal 3, for consistency with previous studies.
w = is the proportion of pups surviving an unusual mortality event arising from poor ice conditions or weather prior to the start of harvesting.
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).

The model is adjusted using the weighted sum-of-square difference between the estimated pup production from the model and the observed one from the surveys. The predicted values of pup production for the survey years are calculated using the equations presented above and their differences with the observed values is evaluated. The two parameters are optimized to minimize the weighted sum-of-square difference by iterative methods.

We included the uncertainty in the pregnancy rates and the pup production estimates in the fitting model by resampling them using Monte Carlo techniques. Both pregnancy rates and pup production data are resampled from normal distribution of known mean and standard error. For each Monte Carlo simulation, a new M and α were estimated and stored. The model was adapted to function within the programming language R.

DATA INPUT

Pup production estimates

The model is fit to 10 independent estimates of pup production (Table 1, Fig. 3) obtained by aerial surveys in 1952, and 1960 (Sergeant and Fisher 1960), mark-recapture methods in 1978, 1979, 1980 and 1983 (Bowen and Sergeant 1983,1985, Roff and Bowen 1983,1986) and aerial survey estimates for 1990, 1994, 1999 and 2004 (Stenson et al. 2002, 2003, 2005). There was considerable uncertainty surrounding the 1952 and 1960 estimates, therefore these were entered into the model with a 40% CV, which is more than twice as much variability as obtained from current aerial survey methods (mean=15%,SE=1.3, N=8)(Table 1).

Catches

Harvest levels from the Canadian commercial hunt, Greenland and Canadian subsistence harvests were corrected for unreported harvests (see below) and were incorporated into the model along with estimates of bycatch (Fig. 2)(Stenson 2005, Sjare et al. 2005). It was assumed that 99% of the YOY and 60% of the 1+ animals killed prior to 1983 were recovered. At this time, the harvest was dominated by the commercial whitecoat hunt and 1+ animals were killed in the whelping patch or in the moulting patch, hence losses were expected to be low. Since 1983, 95% of the YOY killed in the Canadian hunt and 50% of animals aged 1+ years and 50% of all animals killed in the Greenland and Canadian Arctic harvests were assumed recovered (Sjare and Stenson 2002).

Pregnancy rates

The age specific pregnancy rates were based upon November-December 1954-2004 samples and so provide late-term pregnancy rates. All seals less than four year of age were considered immature, while seals eight years of age and older were considered fully-recruited to the breeding population and grouped together (Sjare and Stenson 2010). There are no reproductive data for many year-age combinations, and in some years the samples are quite small. A non-parametric regression estimator was used to estimate the expected pregnancy rates, although a different dataset was used compared to the assessment data (Stenson et al. 2010). Reproductive rates were smoothed from 1952 to 2005. From 2005 onwards, the 2005 smoothed rate was extrapolated forward. 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. Although pregnancy rates do vary between years, it is expected that a bad year will affect all age classes, therefore within-year reproductive rates were correlated across age classes.

Climate variability

Variable environmental conditions have likely had an impact on mortality rates among years. Sergeant (1991) identified 1969 and 1981 as particularly poor ice years and 1981 in particular may have resulted in substantial pup mortality in all areas. In the model, pup survival has been set at 0.75, 0.75, 0.94, 0.88, 0.75, 0.75, 0.95, and 0.8 for 1969, 1981, 1998, 2000, 2002, 2005, 2006, and 2007, respectively.

PROJECTION MODEL AND SIMULATIONS:

The modeling consists of two main steps. The first one is the fitting model (explained earlier) where multiple population matrices are created using Monte Carlo and the parameters M and α are estimated. This is done from 1952 or 1960 until the last year data are available. The second part of the model is the projection, where the population is projected into the future following different management plans. The projection model is based on the same equations as the fitting model but in this case, the parameter α is not used as it is associated with the initial population vector. The projection is instead started from the last year of the population vector estimated by the fitting model. The mortality rates (M) used for the projection are selected from the set of *M's* created with the fitting model. Data on pregnancy rates, seal removals and ice conditions are then extrapolated to complete the projection.

The pregnancy rate data are modified according to the simulation being examined. They are permitted to vary around a normal distribution. Ice-related mortality is selected randomly from the vector [0, 0, 0.1, 0.2, 0.3] assuming a uniform distribution, which gives a mean mortality rate from environmental conditions of 12%.

Finally, harp seal removals for the projection years are generated by summing the Canadian quotas, the Greenland and Arctic catches and the bycatch. From these four sources of mortality, only the Greenland catches are allowed to vary and follow a uniform distribution ranging from 70,000 to 100,000 catches per year. Canadian commercial catches are determined following the different management scenarios being tested. Bycatch and Arctic catches are believed stable at 10,500 and 1,000 respectively. In every case, the age structure is taken into account in calculating the amount of mortality within the population. In this study, it was assumed that the Canadian hunt consists of 90% of young of the year while the Greenland hunt is limited to 14% young of the year (Stenson 2009). If in one year Canada removes 200,000 animals and Greenland 100,000 seals from the population, 10% and 86% of that would be considered to be 1+ seals which would then be distributed uniformly within the 1+ age classes following the age structure of the population.

A struck and lost factor is added to the three different hunts to take into account the seal that are being killed but that are not recovered or reported. This struck and lost factor is calculated the same way it was to evaluate removals for the fitting model. A total of 95% of the YOY in Canada and 50% of every other animal (adult in Canada and all seal in Greenland and the Arctic) are considered to have been reported and the estimated morality adjusted accordingly.

Monitoring of the Canadian harvest occurs via a daily hail-in system, where fishermen report their catches on a daily basis. Dockside monitoring and comparison with hail-in tallies provides an incentive for hunters to provide accurate information on daily takes. Nonetheless, there is considerable capacity in the fleet, which might result in the TAC being exceeded before the fishery can be closed (Table 2). Known as implementation error, this uncertainty is included in the model as a multiplier applied against the Canadian reported catch, before correcting for unreported harvests (struck and loss), as a Uniform distribution with a minimum of 1.0 and a maximum of 1.1, for a mean of 1.05.

Two different approaches were used to assess the performance of the harp seal management framework. The first examined how the population model behaved if certain parameters were adjusted. This included factors such as the number of runs used to fit the model, or modifications to the correlation coefficients to examine their impacts on the CV associated with the output (Table 3). In the second approach, the objective was to determine how the management approach behaved under certain established conditions. For this component, two models were run in parallel, one model, mimicked the conditions that would be encountered during an assessment. Referred to as the 'Assessment model', it was assumed that for the Assessment model, inputs were known accurately (e.g., surveys were unbiased, there was no un-identified mortality, struck and loss and age compositions of the hunts were as identified in the assessment data). The second model, called the 'Reference model' was modified to examine the impacts of failures of certain assumptions e.g., impacts if there was early mortality of young of the year (YOY) due to poor ice conditions (M_{ice}). The Assessment model assumed that all inputs into the model were known without error. The model was run, and the Total Allowable Catch (TAC) that would still respect the management plan was determined using this model. To determine if errors in the assumptions did have an impact on the population, the TAC derived from the Assessment model, was incorporated into the Reference model, then an input variable was modified, for example, mortality rates on YOY were increased by 20%. The Reference model was then run with the TAC and elevated mortality. The pup production estimate obtained from the Reference model, was then used as a new pup survey estimate for the Assessment model. The Assessment model was then refitted using this new 'pup survey estimate' and a new TAC was derived. In this way different scenarios were examined.

The testing algorithm was as follows. Both the assessment and Reference models used the same inputs up to and including 2009. The 2009 Canadian commercial harvest was set at 200,000 animals. The Assessment model was projected forward to estimate the Total Allowable catch (TAC) in the absence of any problems with the inputs. This TAC was then inserted into the Reference model which was also modified according to the scenario being tested (e.g., assume that YOY were subjected to an unknown mortality of 20% (called M_{ice}) that occurred prior to the opening of the hunt). Using the 'sustainable' catch obtained from the assessment, the Reference model (modified to include an additional YOY mortality of 20%) was run to obtain a new estimate of pup production (and population) at the time of the next survey. The interval for a new survey was set at five years, and it was assumed that the average CV around the pup production estimate was 0.15 (the average of eight estimates completed between 1979 and 2004). This new estimate of pup production was then introduced into the Assessment model to extend the data set of abundance estimates. The Assessment model was then fitted to this new survey series and a new TAC was determined. The process was repeated for the next 'management plan'.

The estimated TAC had to respect the management plan over the timeframe being examined, i.e., the TAC was a constant level of harvest (assumed to be taken in full) during a five year block that maintained an 80% probability of the population being above N70. Only a single parameter was adjusted in each simulation. The simulations were allowed to proceed for 50 years (2009 to 2059).

RESULTS

Fitting the non-parametric regression estimator to the pregnancy rate data showed that reproductive rates among older animals were high throughout the 1960s, until the early 1970s, while reproductive rates for 4 and 5 year old females were low throughout this period (Fig. 3).

Among the latter age classes, reproductive rates increased from 1974 until about 1980, then rates among all age classes of animals five years old and greater declined to 1999. The low rates were extended from 2000 to 2060, the end of the simulation period for this study.

For the simulated population pup production increased from an estimated 572,493 (SE=20,292) in 1960 to 1,007,897 (SE=87,105) in 2005 then declined slightly to 994,954 (SE=110,252, CV=0.11) animals in 2009 (Fig. 4). The total population increased from 2.6 million (SE=100,000) animals in 1960 to 5.9 million (SE=427,375) in 2005 and then declined slightly to 5.8 million before increasing again to 6.0 million (SE=618,724, CV=0.10) in 2009 (Fig. 4). Adult mortality rate was 0.056 (SE=0.003. CV=0.054). The sensitivity of the model to changes in the initial population vector was examined by doubling the initial population size, halving the proportion of young of the year, or doubling the proportion of young of the year. Several measures were examined including M, the average proportion of pups in the population, deviations in number of pups or total population size from that generated using the original age vector, or average CV in numbers of pups or total population size (Table 4). None of these adjustments resulted in showed significant changes between initial population size, M, pup production or total population.

The model was extended back in time by fitting to reproductive rate data from the 1950s and the 1952 and 1960 aerial surveys. Struck and loss in the commercial hunt was set at 1% for pups until the end of the whitecoat hunt in 1983. The model did not converge until the struck and loss factor for the commercial hunt was reduced from 50% to 40% for adults, which reflects that the harvest at the time was directed primarily towards whitecoats, with large numbers of adults also being taken in the breeding or whelping areas. Struck and loss for both pups and adults on the ice were likely lower at this time. This adjustment was applied to Canadian commercial harvests from 1952 to 1983. Pup production was estimated at 616,581 (SE=21 618, CV=.035) in 1952, declined to 471,394 (SE=20,847) in 1960 and then increased to 992,143 (SE=112,507, CV=.11) in 2009. Total population size in 1952 was 2.8 million (SE=77,026, CV=0.028) declining to 2.4 million (SE=97,932) animals in 1960, then increasing to 6.0 million (SE=633,686, CV=0.10)(Fig. 4). Adult mortality rate was 0.056 (SE=0.003, CV=0.054). The proportion of pups in the population that started in 1952 was slightly higher than in the model that was initiated in 1960, but overall changes in the initial age vector had little impact on M, the CV of pup numbers or total population size, or deviations from the population generated using the initial population vector. For the remainder of this study, we used only the model that started in 1952.

The number of simulations had little effect on α and M (Table 5). In the previous version of the model, it was assumed that reproductive rates between cohorts within a single year were completely independent. We compared the impact of treating reproduction between year classes as a nearly independent variable (r=0.1) or as a highly correlated variable (r=0.85). The assumption is that if a cohort is likely to have a bad year, then several other cohorts will also have a bad season within the same year. Setting the correlation coefficient to 0.85 had little impact on the predicted estimate of pup production or total population size, but a stronger correlation increased the variance around the estimate (Fig. 5, Table 5). Therefore, the correlation coefficient among reproductive rates was set at 0.85, because biologically it seemed more logical that if one year class had a good year, then other age classes were more likely to have a good year as well.

The effect of varying harvests within a 5 year block was examined by taking the quota in full in Year 1, 0% of the quota in Year 2, the entire quota plus all of the carryover from the previous year in Year 3, then 50% of the quota in Year 4 and the full quota plus all of the carryover in Year 5 (Fig. 6). This was compared to 5-year blocks that took about the same quota each year over the entire 5-year block. An average of 6,000 more seals were taken annually under the variable harvest scenario, but this is not considered to be significant because harvests were usually only balanced to the nearest 5,000 or 10,000 animals (Fig. 5, Appendix 1).

The commercial harvest takes primarily young of the year (YOY: 90%-97%), but assessments are only carried out every 4-5 years. We examined the impact of a large harvest of 700,000 animals in the Canadian hunt in 2009 and a harvest of 600,000 from 2010 until 2014, followed by harvests of 200,000 animals in the northwest Atlantic population. An aerial survey that would be flown in 2009 is completed before the harvest starts. Modelling this scenario indicates that no change in pup production would occur until 2014, but no change in pup production would be observed until the 2019 survey (Fig. 7). Surveys flown more frequently could expect to start detecting a change in pup production in 2015 as pup production begins to decline, six years after the initial high harvest was taken. However, a modeling approach would indicate that the total population size would begin to decline in 2009.

MANAGEMENT SCENARIOS

Ice mortality

Variable ice conditions in Atlantic Canada suggest that natural mortality among young animals may be increasing. A factor for ice related mortality (M_{ice}) has been included in the assessment since 2003. In this scenario we compare the impact of not including M_{ice} in the Assessment model to the impact of assuming an annual mortality of YOY of 20% (M_{ice}=20%). This mortality occurs prior to the hunt. The Assessment model assumed that M_{ice}=0, and was used to set harvest quotas in 5-year blocks. This harvest was then incorporated into the Reference model, that had M_{ice}=20% and the model was projected forward to obtain a new survey estimate of pup production at the end of the 5-year block.

The Assessment model suggested that a constant harvest of 575,000 could be taken over the first 5-year block (2009-2014)(Fig. 8). After five years, this was reduced to 200,000 for the next 5-year block, and then the harvest was closed for a 10 year period (2020-2029), before re-opening for a brief five year period in 2030, and then closing again until the end of the simulation. The total harvest for the commercial hunt, assuming 90% YOY was 5,125,000 animals. The Assessment model attempted to fit to the change in pup production and the fit to the 2014 point was quite good (Fig. 8). However, in subsequent projections the quality of the fit of the model to the survey points decreased as the model attempted to fit to increasingly lower estimates of pup production. Throughout the time series, little change was observed in the starting population, but M in the Assessment model was observed to increase as the model attempted to fit to the lower estimates of pup production. This had the effect of reducing the fit to the peak estimates of pup production obtained from surveys 'flown' in 2004, 2009 and 2014 as the model was influenced by the increasing number of 'surveys' at lower pup production levels (Fig. 8). The reduction in harvests was in response to the decline in the L20 population size below N70 in the Assessment model (Fig. 9). When L20 first fell below N70, the reduction in harvests was sufficient to allow L20 from the Assessment model to recover to N70. Opening the harvest again in 2030, had the impact of causing an immediate decline in L20. In response, the commercial harvest was reduced, but in spite of tests of different commercial harvest levels, even reducing commercial harvests to zero, declines in L20 to levels as low as N50 were observed before recovering slowly and possibly moving above N70 by 2059. Throughout this period the 'real' population as predicted by the Reference model was observed to decline to near N50 (≈3 million animals), but did not decline further. The L20 and L2.5 (i.e., lower 95% C.I.) lines also declined below N70, but the L20 line remained above N50, indicating that the probability of dropping below N50 was less than 20%. After the mid-2030s the L2.5 line continued to decline and may have reached N30 at the end of the simulation period, indicating that there was a 2.5% chance that the population would hit N30 by 2059 (Fig. 9).

The previous simulation examined the impact of different harvest scenarios within 5-year blocks when choosing an acceptable harvest level for the next five years. However, earlier scenarios indicated that changes in pup production in response to a harvest would not be expressed in the population until five years after a new harvest regime had been introduced. We examined a variation on the M_{ice} scenario by continuing to fit the model every five years to new survey data, but with the constraint that harvests must respect the management plan throughout a 10 year projection into the future. In the first run, the Assessment model suggested that a constant harvest of 400,000 could be taken over the first 5-year block (2009-2014)(Fig. A1a). After five years, this was reduced to 375,000 for the next 5-year block, and then reduced in subsequent 5-year blocks until the hunt would be closed completely in 2040 and would remain closed until the end of the simulation period (Fig. A1a). The total harvest for the commercial hunt, assuming 90% YOY was 5,725,000 animals. The Assessment model fit to the new pup production patterns as observed in the first scenario (Appendix Scenario A1, Fig. A1a). Throughout this period the 'real' population mean and L20 (as predicted by the Reference model) were observed to decline to near N50 (≈3 million animals), but no further. The L2.5 line declined below N50, but did not reach N30 by the end of the simulation period (Fig. 10, Appendix A1b).

This scenario was repeated, with the period to evaluate whether L20 remained above N70 extended from 10 years to 20 years (see Appendix Scenario A2). The pattern of fit to the data was similar to that observed under the previous scenarios, but the commercial harvest was closed in 2034, after only 24 years into the simulation period (Fig. Appendix A2a). During that time a total of 7.1 million animals were removed. In spite of the closure of the commercial hunt, the mean, L20 and L2.5 continued to decline, with L20 falling below N50 by 2040 and L2.5 falling below N30 by 2055 (Fig. 11, Appendix A2b).

A new scenario kept the period for the projection to respect the objectives of the management plan to 10 years, but examined the impact if the plan called for a rule that maintained a 90% probability that the population remained above N70 (Appendix A3). The pattern of fit to the data was similar to that observed under the previous scenarios. Although the commercial harvest was not closed in 2034, as in the previous scenario, the catch was reduced dramatically to only 10,000 animals. The commercial harvest was increased to 200,000 (2045-2049), then declined to 20,000 animals for the remainder of the simulation period. Under this scenario a total of 6,800,000 animals were removed in the commercial catch (Appendix A3a). In spite of the closure of the commercial hunt, the mean, L10 and L2.5 continued to decline, with both L10 and L2.5 falling below N50 by about 2050. L2.5 continued to decline, but remained just above N30 at the end of the simulation period (Fig. 12, Fig. A3b).

A new scenario kept the period for the projection to respect the objectives of the management plan to 10 years, but examined the impact of changing the probability of staying above N70, from 80% to 70% (Appendix A4). The pattern of fit to the data was similar to that observed under the previous scenarios. In the first run, the Assessment model suggested that a constant harvest of 425,000 could be taken over the first 5-year block (2009-2014)(Fig. A4a). After five years, this was reduced to 350,000 for the next 5-year block, and then reduced to 80,000 until the hunt was closed in 2029, where it remained until the end of the simulation period. The total harvest for the commercial hunt was 5,525,000 animals. In spite of the closure of the commercial hunt, the mean, L30 and L5 continued to decline, below N50 at the time of the closure in 2029, and all indices falling below N30 by 2043 (Fig. 13, Fig. A4b).

DISCUSSION

The Precautionary Approach which might also be considered as 'better safe than sorry' was developed in Germany in the late 1960s as part of that country's new environmental protection laws considering the release of chemicals into the environment (Anon. 2004). A key feature of PA is the creation of benchmarks for different levels of concern, and in discussion with industry, the identification of specific actions that will occur if thresholds for these benchmarks are exceeded during the management plan. At the end of a plan, changes can be made prior to the next plan if stakeholders are in agreement. An extension on the PA is the management procedure approach, which differs from more traditional approaches by using simulation to determine if there is a reasonable chance that the management plan will reach its objectives given that there will be some uncertainty surrounding the current best estimate of the status of the resource (Butterworth 2007). The Management Procedure (MP) is compatible with PA and was first developed by the IWC during the 1980s (Butterworth 2007). The advantage of the MP is that it identifies the data required as well as the estimation method, and areas where the uncertainty is greatest which allows research to be focused on those areas. Using medium-term projections, the simulation testing framework provides an appropriate basis for evaluation of risk and provides a framework for interactions with stakeholders, in particular before exploitation begins so that various approaches can be evaluated and decided upon which reduces the need for short-term haggling and meetings to decide upon actions in the face of unexpected results (Butterworth 2007). If presented properly, the simulations provide stakeholders with an opportunity to observe the consequences of specific management actions on a 'virtual' population and not the resource, which can help to guide decision making. Disadvantages of the MP approach include a lengthy development to complete the simulations (Butterworth 2007, Punt and Donovan 2007) although once these are done non-productive scientific and political haggling are greatly diminished. The MP framework may be overly rigid, there can be a tendency to operate in a manner that considers that future monitoring efforts are not required, and arguments can ensue on whether or not certain hypotheses are sufficiently plausible to merit retention (Butterworth 2007).

The Revised Management Plan (RMP) developed by the IWC and the Potential Biological Removal (PBR) are two of the better-known examples of a MP being established to manage removals from marine mammal populations. The basis of the Revised Management Plan is a requirement for information on catches and estimates of absolute abundance to calculate acceptable harvest levels. Extensive simulation testing was undertaken under different scenarios to test how the approach satisfied three primary objectives: 1) catches remain as stable as possible, 2) the stock could not be depleted below some chosen level, and 3) obtain the highest continuing yield from the stock under consideration (IWC 1994). The IWC decided that the basic information required included information on catches and estimates of abundance, where absolute estimates of abundance were superior to indirect indices such as Catch Per Unit of Effort (Punt and Donovan 2007). The basic assessment model assumed density-dependence using a surplus production model in the form of the Pella-Tomlinson model (IWC 1994). The key to the RMP is the Catch Limit algorithm. IWC set up a control rule that harvests would fall to 0, if the stock size fell below 0.54 of carrying capacity (K)(Cooke 1999). If the stock size is above 0.54 K, then the TAC would be set at the 41.02 percentile of the marginal posterior distribution of the Limit, where the Limit would be equal to $3\mu(D-0.54)P$, where μ is a productivity parameter, D is a measure of depletion from K levels, and P is the population size (Butterworth and Best 1994).

The PBR approach was developed in the United States in response to the Marine Mammal Protection Act. There the management objective is to prevent populations from becoming depleted, where depletion is considered to have occurred if a population falls below its maximum net productivity level. The maximum net productivity level is a concept tied in with MSY and density-dependence where, depending on the form of the density-dependent relationships, the maximum net productivity level would fall between 50% and 85% of carrying capacity, although it is more likely to fall at the lower end of this range (Taylor and DeMaster 1993). Therefore, from the definitions it follows that the $PBR = N_{min},\text{-}1/2\ R_{max}\ F_R$ where: $N_{\min}$ = the minimum population estimate of the stock, R_{max} is the maximum rate of increase at a small population size, F is a recovery factor varying between 0.1 and 1.0 (Wade 1998). The PBR approach has also be subject to extensive simulation testing to examine how it behaves under different scenarios, with the objective that the population must have a 95% probability that it will not become depleted. The PBR formula is easy to use, and unlike the IWC approach which fits a model to an increasing series of catch and abundance estimates, to estimate allowable catches, the PBR uses a single point estimate to establish acceptable levels of removals. This avoids the need to identify trends in abundance before management decisions are taken, but increases the sensitivity of the approach to estimates from a single survey (offset to some extent by estimating $N_{\min}$).

Both of the above management procedures are well-known and as with any approach have their strengths, as well as their weaknesses. For both approaches, one of their strengths is that they have undergone extensive simulation trials to examine how they might behave under certain deviations in model assumptions. It is tempting to automatically apply one or both of these approaches to management of marine mammal stocks in DFO, since much of the hard work will already have been completed. However, both approaches rely heavily on being able to describe the behaviour of a population through a density-dependent modeling framework, which assumes that density-dependent mechanisms will act in a predictable manner. However, population regulation is not so easy to predict because of the complex interaction between intrinsic factors related to density-dependent factors and extrinsic factors involving environmental variability (de Little et al. 2007). Below carrying capacity, extrinsic factors are thought to be the primary drivers of population change, while close to K, intrinsic factors are considered to be the dominant mechanism to limit population growth (Fowler 1981). The lack of recovery in the St. Lawrence beluga whale population could be one example of where an expected density-dependent response has not occurred with the removal of harvesting. Another example would be the difficulties in understanding the density-dependence mechanisms observed among the different harp seal populations in the north Atlantic that appear to be mediated by environmental and/or food availability (see. Sjare and Stenson 2010, ICES 2009).

A second key component with the RMP is the closure to harvesting if the stock falls below 54% of K. However, the need for a decision rule to limit harvests at 54% of K within the Catch Limit Algorithm was made by the Commission based on the belief that such a rule was required to ensure that the RMP is 'at least as conservative as the NMP' (Butterworth and Best 1994). However, such a decision, given the three objectives of the IWC for the RMP, implies that the Commission considers that 0.54K is a population below which the risk of extinction to a stock is seriously increased if any catch is taken (Butterworth and Best 1994). This apparently contradicts the views expressed by the Scientific Committee at the time at which the New Management Plan was being formulated, which made it very clear that protection for stock management purposes was different from levels at which it would be in danger of biological extinction appears (Butterworth and Best 1994). Butterworth and Best (1994) also point out that in the development of the aboriginal whaling scheme, in 1987, the scientific committee agreed that the Bering-Chukchi-Beaufort Sea bowhead was likely above the minimum level, but this also implied agreement by the committee that populations of 0.23K and above are not placed under serious risk of extinction by limited catches.

The main challenge with the IWC approach is that the RMP was not developed in consultation with industry and it has never been implemented (Punt and Donovan 2007) so its acceptance and behaviour in a commercial setting are not known. It is not clear whether applying the RMP, as is, in other settings could occur, because the approach can be considered as so risk-averse that the only real scientific basis for questioning its immediate implementation is that it is so conservative that it will waste much of a potential harvest (Butterworth 1995).

Within the management of Atlantic seals the PBR framework has been used for populations that area considered Data Poor (e.g., hooded seals). The PBR approach requires only a point estimate and where other data are lacking, it provides an easy way to estimate removals that are unlikely to cause harm. However, the PBR was developed to address concerns about levels of incidental catches and the potential harm they may cause to a population, not to deal with commercial harvesting. Although the PBR is easy to determine, it can also be considered a black box approach that does not take into consideration other information. For example, it assumes that the removals of different age classes occur in proportion to their abundance in the population; it also makes an assumption about the rate of increase and relies on a single point estimate in its assessment. Failure to consider the age composition of the harvest can have important implications for the level of harvest that would be acceptable. We can examine this impact using a harp seal population of 7 million animals. In the Canadian harp seal hunt, YOY make up 97% of the catch. If this composition remains unchanged, and the remaining 3% are taken proportionally to their abundance, then a harvest scheme could remove 700,000 animals annually and still respect the plan over a 7-year period. However, if the proportion of YOY is reduced to 20%, which is approximately their weighting in the population, and all other variables remain unchanged (reproductive rates, Greenland catches), then the TAC would have to be reduced by 60% to 265,000 to respect the management plan, which would result in significant loss of animals to industry. Recently, situations have arisen where PBR does not appear to be conservative at all. Among white Sea harp seals, PBR removals result in a high probability of causing the population to decline while quotas identified using a more detailed approach maintain current populations (ICES 2009).

The challenge in developing a PA framework is identification of B_{lim}. In the United States, B_{lim} has been mandated within a legal framework, based on the concept of MSY. In the case of the IWC, the B_{lim} appears to have been selected by the Commission without consultation with industry, and is linked to carrying capacity (K). As outlined in the Introduction there have been difficulties in identifying B_{lim}, linked to MSY because of uncertainties in the underlying stock-recruitment relationship. This relationship is poorly understood for harp seals as well (McLaren et al. 2001, Sjare and Stenson 2010). Difficulties on agreeing on a suitable B_{lim} appear to be a major factor in the slow pace at which the PA is implemented. Although Canada ratified the United Nations Fish Stock Agreement in 1999, calling for the implementation of PA, the harp seal fishery is the only fishery implemented in the Department for which there is some semblance to a PA framework (although additional simulation testing is needed)(DFO 2005). The harp seal management framework was implemented in 2003. At that time, the Department had not developed the general precautionary framework. However, the two approaches are comparable. The General Framework is linked to the concept of MSY, where a stock would be considered healthy at levels ≥of B_{MSY}, unhealthy at levels of ≤40% of B_{MSY}, and a cautious zone would exist at levels in between. For marine mammals MSY is considered to be located somewhere between 50% and 80% of carrying capacity and is more likely to be at the lower end of this range (Taylor and DeMaster 1993, Wade 1998). Therefore the limit between healthy and cautious would be established between 50% and 85% of K, and the limit between healthy and unhealthy for marine mammals would be established somewhere between 20% and 34% of

K (40% of B_{MSY}). This overlaps very closely with the framework now in use for harp seals, where an upper Precautionary Level (N70) has been set at 70% of the maximum abundance observed, a proxy for K, and the limit reference point has been set 30% of the largest population observed (called N30). In addition, for harp seals a second precautionary level has been established at 50% of the maximum abundance (N50). Thus the framework used to manage harp seals is consistent with both the general departmental framework and UNFA as well.

Both the PBR and IWC frameworks identify approaches when carrying out testing of the management procedure framework. In this study, a series of relatively simple tests were carried out to examine the behaviour of the model with simple changes. As part of this component we examined the impact of changing the age composition of the starting population vector, fluctuations in harvest levels, and the time required to detect changes in the population. The model was relatively insensitive to changes in the starting population vector in 1960 and in 1952. This likely results from the effects of these changes occurring so far back within the modeled series to have little impact on current total abundance and pup production. Surprisingly, fluctuations in harvests, operating through a carryover also had little impact, although this did not examine the cumulative impacts that might result from a combination of high harvests and a year when ice mortality was unusually high. However, because the harvest applies to YOY, it is possible that their impacts are dampened because mortality amongst this group would be expected to be high anyway. Subsequent simulations examine the behaviour of the Management Procedure under reasonable deviations from assumptions related to the assessment or the status of the resource. This approach is time consuming and only a limited number of evaluations were possible.

We compared the impact of including ice related mortality of YOY in a Reference model to what would be expected in an Assessment model that failed to consider this source of mortality, to determine the impacts of the resulting management advice on the population. The scenarios were examined with the objective of maintaining an 80% probability that the population would remain above a precautionary threshold of N70. As subsequent assessments were completed, an increasingly poor fit by the model to the ‘survey’ data was observed. Under the various scenarios examined, failure to consider the effects of M_{ice}=20% resulted in population declines below the Precautionary threshold of N70. The best results were obtained if the period to evaluate setting the TAC was extended from the five year assessment period to extend to a 10-year period, and if the probability of staying above N70 was 80% or 90%. Although the population fell below N70 quite rapidly, and in many cases fell below the second Precautionary level of N50, the framework was very robust in maintaining the population above the B_{lim} more than 95% of the time. The scenarios examined harvesting within the context of respecting the management framework over a 5- or 10-year period. The best results were obtained if the recommended TAC was evaluated within the context of a 10-year framework, but over time it still resulted in a decline in harvest levels until closure of the fishery. Although this approach did indicate that the framework is in a conservation sense very robust to failures under the conditions of this scenario, closure of harvesting would not be satisfactory to stakeholders if the expectations were that markets were to remain strong (and open). This points to the need for some changes to the framework to allow to maintain harvesting over the 50-year projection period.

In addition to the failure to account for M_{ice} in the Assessment model, it was evident that high levels of frequently occurring, persistently poor, ice conditions would have an important impact on the trajectory of the northwest Atlantic harp seal population and this needs to be considered when modeling the trajectory of this population. In this study, unusual mortality due to poor ice conditions (M_{ice}), was applied only to the YOY. The effects of increased mortality were not observed until 10 years after the change in mortality conditions were initiated, which as identified above reflects a combination of factors including the time for YOY to reach sexual maturity and to begin contributing to the breeding component of the population, as well as the 5-year intervals between surveys. Thus our perceptions of the current trajectory in pup production numbers will be influenced most by harvest and mortality conditions encountered during the previous decade. Similarly, M_{ice} will have cascading effects as the impacts of multiple years of YOY mortality work their way through the population. Unusual mortality amongst older age classes was not examined here, but would be expected to act on the population much earlier.

The harp seal population is monitored using a population model that combines information on pup production, removals, reproductive rates and unusual mortality to provide information on the status of the stock. IWC has suggested that total abundance counts are the most effective means of following the population and are much superior to other indirect indices such as catch per unit of effort. For harp seals the primary tool to follow the population directly is the aerial survey to evaluate pup production. Although effective, changes in harvesting practices will not be reflected in pup production until at least five years after a new harvest regime has been implemented, as new cohorts enter into the breeding component of the population. If the population is only assessed every five years, then differences in the population may not be detected until as much as 10 years after changes have occurred. This might be reduced to five years if annual surveys were completed, but would not completely eliminate the lag. Using a population model that incorporates information on changes in reproduction and harvests translates the status of the resource into a single currency that not only allows for earlier detection of changes in the population, but also provides an index for the population that is easy for all stakeholders to understand. The fact that we have annual reproductive data to extrapolate the pup production value is critical. IWC do not have reproductive data for their species of concern and so must be more conservative and allow for possible changes that they do not monitor.

In this study we examined projections over a 50 year time frame. This is shorter than the 100 year timeframe examined by IWC and for development of the PBR. We selected this timeframe because of the shorter generation time for seals compared to large whales. Also, it fits within the context of a working lifespan for a biologist or fisherman, although it might be considered considerably longer than the career of a minister of the day. Throughout the simulation we kept all other variables constant. Although it might be unrealistic to assume that reproductive rates would remain low throughout the simulation period while a population declined, our objective was to examine how the Management Procedure might function under specified conditions.

Both the IWC and PBR frameworks outline several steps and scenarios that could be examined within the management procedure framework, which take considerable time to complete (Punt and Donovan 2007). Implementation of the Management Procedure framework for the management of Atlantic seals focused on setting clearly-identifiable benchmarks that were acceptable to stakeholders. In following this approach, the debate around appropriate levels for B_{lim} within the scientific community, which would have blocked its implementation, were avoided. As a result, PA-type management approach was established in 2003 and it remains the only PA-type fishery managed by the Department today. Seven years later, there does not appear to be any standard approach to setting B_{lim} and the separation between management under PA and management under Species at Risk concerns are not clear (Rice and Legacé 2007). It is evident that testing of the Management Procedure framework is needed and, as necessary, adjustments can be made in response to the findings from these simulations. In this study only a few simulations were examined.

LITERATURE CITED

Anonymous. 2004. The price of prudence. Economist January 22, 2004.

Baker, C.S., and Clapham, P.J. 2004. Modelling the past and future of whales and whaling. Trends Ecol. Evol., 19: 365–371.

Bowen, W.D. and Sergeant, D.E. 1983. Mark-recapture estimates of harp seal pup (*Phoca groenlandica*) production in the northwest Atlantic. Can. J. Fish. Aquat. Sci. 40(6):728-742.

Bowen, W.D. and Sergeant, D.E. 1985. A mark-recapture estimate of 1983 harp seal pup production in the northwest Atlantic. Northwest Atl. Fish. Organ. SCR Doc. 85/I/1, Ser., N935. 14p.

Butterworth, D.G. 1995. Additional Essays on Whales and Man, High North Alliance, An essay first presented at the Symposium on Scientific Management of Fisheries and Marine Mammals in Washington D.C., April 1994, Organised by the Science and Environment Policy Project.

Butterworth, D.S. 2007. Why a management procedure approach? Some positives and negatives. ICES J. Mar. Sci. 64:613-617.

Butterworth, D.G. and P.B. Best. 1994. The origins of the choice of 54% of carrying capacity as the protection level for baleen whale stocks and the implications thereof for management procedures. Rep. Int. Whal. Commn. 44:491-497.

Cooke, J.G. 1995. The International Whaling Commission's revised management procedure as an example of a new approach to fishery management. *In* Whales, Seals, Fish and Man, pp. 647–658. Ed. by A.S. Blix, L. Walløe, and Ø. Ultang.

Cooke, J.G. 1999. Improvement of fishery-management advice through simulation testing of harvest algorithms. ICES J. of Mar. Sci. 56:797-810.

Committee on the Status of Endangered Wildlife in Canada (COSEWIC). 2006. COSEWIC's assessment procedure and criteria. Available at http://www.cosewic.gc.ca/pdf/assessment_process_e.pdf

Daw, T. and T. Gray. 2009. Fisheries science and sustainability in international policy: a study of failure in the European Union's Common Fisheries Policy. Mar. Policy 29 (2005) 189–197

de Little, S.C., Bradshaw, C.J.A., McMahon, C.R. and Hindell, M.A. 2007. Complex interplay between intrinsic and extrinsic drivers of long-term survival trends in southern elephant seals. BMC Ecology 2007, 7:3 doi:10.1186/1472-6785-7-3

DFO. 2003. Proceedings of the zonal assessment meeting—Atlantic cod. DFO Can. Sci. Advis. Sec. Proceed. Ser. 2003/021. vi + 125 p. Available at: http://www.dfo-mpo.gc.ca/csas/Csas/proceedings/2003/PRO2003_021_E.pdf

DFO. 2005. Proceedings of the Meeting of the Science Working Group on the Precautionary Approach, Ottawa, Ontario, 20–21 October 2005. Fisheries and Oceans Canada. Canadian Science Advisory Secretariat. Proceedings Series 2005/027.

DFO. 2008a. Overview of the Atlantic Seal Hunt 2003 – 2005. Available at: http://www.dfo-mpo.gc.ca/fm-gp/seal-phoque/reports-rapports/mgtplan-plangest2003/mgtplan-plangest2003-eng.htm

DFO. 2008b. Overview of the Atlantic seal hunt 2006-2010. Available at: http://www.dfo-mpo.gc.ca/fm-gp/seal-phoque/reports-rapports/mgtplan-plangest0610/mgtplan-plangest0610-eng.htm

Fowler CW. 1981 Density dependence as related to life history strategy. Ecol. 62:602-610.

Hammill, M.O. and G.B. Stenson. 2003a. Harvest Simulations for 2003-2006 Harp Seal Management Plan. DFO Can. Sci. Advis. Sec. Res. Doc. 2003/068. 29 p. http://www.dfo-mpo.gc.ca/csas/Csas/Home-Accueil_e.htm

Hammill, M.O. and G.B. Stenson. 2003b. Application of the Precautionary Approach and Conservation Reference Points to the management of Atlantic seals: A Discussion Paper. DFO Can. Sci. Advis. Sec. Res. Doc. 2003/067. 23 p.

Hammill, M.O. and G.B. Stenson. 2007. Application of the Precautionary Approach and Conservation Reference Points to the management of Atlantic seals. ICES J. Mar. Sci., 64: 702–706.

Hammill, M.O., B. Ferland-Raymond, L.-P. Rivest and G.B. Stenson. 2009. Modelling Northwest harp seal populations: modifying an Excel model to R. CSAS Res. Doc. 2009/108.

Hilborn, R., J-J Maguire, A.M. Parma and A.A. Rosenberg. 2001. The Precautionary approach and risk management:can they increase the probability of success in fishery management. Can. J. Fish. Aquat. Sci. 58:99-107.

ICES 2001. Report of the study group on the further development of the precautionary approach to fishery management. ICES CM 2001/ACFM:11. 49 p.

ICES. 2009. Report of the Joint ICES/NAFOWorking Group on Harp and Hooded Seals (WGHARP). ICES Document CM 2009/ACOM: 17.

International Union for the Conservation of Nature (IUCN). 2001. IUCN Red List Categories and Criteria : Version 3.1. IUCN Species Survival Commission. IUCN, Gland, Switzerland and Cambridge, UK.

International Whaling Commission 1994. Annex H: The revised management procedure (RMP) for Baleen Whales. IWC 44:145-157.

McLaren, I.A., S. Brault, J. Harwood and D. Vardy. 2001. Report of the Eminent Panel on seal management. 146 p. Available at: http://www.dfo-mpo.gc.ca/fm-gp/seal-phoque/reports-rapports/mgtplan-plangest2001/EPSM-GEGP_e.pdf

Punt, A. and G.P. Donovan. 2007. Developing management procedures that are robust to uncertainty: lessons from the International Whaling Commission. ICES J. Mar. Sci. 64: 603-612.

Punt, A.E., and Smith, D.M. 2001. The gospel of maximum sustainable yield in fisheries management: birth, crucifixion and reincarnation. *In* Conservation of Exploited Species, pp. 41–66. Ed. By J. D. Reynolds, G. M. Mace, K. H. Redford, and J. G. Robinson. Cambridge University Press, Cambridge, UK.

Rice, J.C. and E. Legacé. 2007. When control rules collide: a comparison of fisheries management reference points and IUCN criteria for assessing risk of extinction. ICES J. Mar. Sci. 64:718-722.

Roff, D. and W.D. Bowen. 1983. Population dynamics and management of the Northwest Atlantic harp seal. Can. J. Fish. Aquat. Sci. 40: 919-932.

Roff, D. and W.D. Bowen 1986. Further analysis of population trends in the Northwest Atlantic harp seal (*Phoca groenlandica*) from 1967 to 1985. Can. J. Fish. Aquat. Sci. 43:553-564.

Sergeant, D.E. and H.D. Fisher 1960. Harp seal population sin the western North Atlantic from 1950 to 1960 Fisheries Research Board of Canada, Arctic Unit. Circular No. 5.

Sergeant, D.E. 1991. Harp seals, man and ice. Can. Spec. Publ. Fish. Aquat. Sci. 114: 153 p.

Sjare, B. and G.B. Stenson. 2002. Estimating struck and loss rates for harp seals (*Pagophilus groenlandicus*) in the Northwest Atlantic. Mar. Mamm. Sci. 18: 710-720.

Sjare, B., and Stenson, G. B. 2010. Changes in the reproductive parameters of female harp seals (*Pagophilus groenlandicus*) in the Northwest Atlantic. – ICES Journal of Marine Science, 67: 304–315

Sjare, B., D. Walsh, S. Benjamins and G.B. Stenson. 2005. An Update on Harp Seal By-catch Estimates in the Newfoundland Lumpfish Fishery: 1999-2003. DFO Can. Sci. Advis. Sec. Res. Doc. 2005/049. Available at: http://www.dfo-mpo.gc.ca/csas/csas/publications/resdocs-docrech/2005/2005_049_e.htm

Stenson G. B. 2009. Total Removals of Northwest Atlantic Harp Seals (*Pagophilus groenlandicus*) 1952-2009. DFO Can. Sci. Advis. Sec. Res. Doc. 2009/112. Available at: http://www.meds-sdmm.dfo-mpo.gc.ca/csas/applications/Publications/publicationIndex_e.asp

Stenson, G.B., M.O. Hammill, M.C.S. Kingsley, B. Sjare, W.G. Warren and R.A. Myers. 2002. Is there evidence of increased pup production in northwest Atlantic harp seals, *Pagophilus groenlandicus*? ICES J. Mar. Sci. 59:81-92.

Stenson, G.B., M.O. Hammill, J.W. Lawson, J.-F. Gosselin, and T. Haug. 2005. 2004 pup production of harp seals (*Pagophilus groenlandicus*) in the Northwest Atlantic. DFO Can. Sci. Advis. Sec. Res. Doc. 2005/037. Available at: http://www.meds-sdmm.dfo-mpo.gc.ca/csas/applicationsCsas/Publications/publicationIndexResDocs-DocRech/2005/2005_037_e.asphtm

Stenson, G.B., L.-P. Rivest, M.O. Hammill, J-F. Gosselin and B. Sjare. 2003. Estimating Pup Production of Harp Seals, *Pagophilus groenlandicus*, in the Northwest Atlantic. Mar. Mammal. Sci. 19:141-160.

Stenson, G.B. and M.O. Hammill. 2008. Incorporating the Precautionary Approach into the provision of advice on Marine Mammals. DFO Can. Sci. Advis. Sec. Res. Doc. 2008/079. Available at: http://www.dfo-mpo.gc.ca/CSAS/Csas/Publications/ResDocs-DocRech/2008/2008_079_e.htm

Stenson, G.B., Hammill, M.O. and Healey, B. 2010. Reproductive rates of Northwest Atlantic harp seals, 1954-2007. DFO Can. Sci. Advis. Sec. Res. Doc. 2009/113.

Taylor, B .L., and D.P. Demaster 1993. Implications of non-linear density dependence. Mar. Mamm. Sci. 9360-37.

Taylor, B.L., Wade, P.R., De Master, D.P., and J. Barlow. 2000. Incorporating uncertainty into management models for marine mammals. Cons. Biol. 14:1243-1252.

Wade, P.R. 1998. Calculating limits to the allowable human-caused mortality of cetaceans and pinnipeds. Mar. Mamm. Sci. 14:1-37.

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The authors thank L.-P. Rivest and B. Ferland-Raymond, University of Laval, QC. for help with the model.

Table 1. Estimates of northwest Atlantic pup production from mark-recapture and aerial surveys.

Year	Estimate	SE
1952	578,000	231,200[2]
1960	301,000	120,428[2]
1978	497,000	34,000
1979	478,000	35,000
1980	475,000	47,000
1983	534,000	33,000
1990	577,900	38,800
1994	702,900	63,600
1999	997,900	102,100
2004	991,400	58,200
2009 [1]	997,998	149,700

[1] Estimated by running the model fit to time series beginning in 1960 and assuming a harvest of 200,000 animals in the Canadian commercial hunt in 2009, then assuming a CV of 0.15.
[2] SE estimated assuming a CV of 0.4.

Table 2. Quota and reported harvests in the Canadian commercial and Greenland subsistence hunts.

		Canada		Greenland
	Quota	Catch	%	
2003	325,000	289,512	89.1	68,499
2004	350,000	365,971	104.6	70,585
2005	319,517	329,829	103.2	91,361
2006	335,000	354,867	105.9	N/A
2007	270,000	224,745	83.2	N/A
2008	275,000	217,636	79.0	N/A
Average.	320,000	312,985	97.0	76,815

Table 3. Model scenarios examined in the study.

Base case trials	
Start 1960	
Start 1952	
Correlation coefficient	Examine r=0.1 or r=0.85
Constant vs versus variable harvest	Constant harvest in a 5 5-year block. Harvest in 5 5-year block with annual harvests of 100%, 50% 150%, 50%,150%
Change initial population vector	Original, half size of initial vector, half proportion of pups in vector, double proportion of pups in vector
Base run	Unbiased inputs
Variable harvests	Allow fluctuations in quotas and in catches, look for stability
Scenarios	
Impacts of ice	Set ice mortality to 20% in every year.
Bias surveys	Not yet examined
Change productivity	Not yet examined
Change struck and loss	Not yet examined

Table 4. Effects of modifying the Initial population size or age composition of the starting vector for the model fitted to 1960-2009 data (top). Average coefficient of variation in pup numbers (SE), total population size (SE), average proportion of pups in the estimated population (SE), average deviation (SE) from the original age vector defined as: $(Pup_{original}-Pup_{experimental})/Pup_{original}$ and average deviation for the total population (SE)(bottom) for models fitted to 1960 to 2009 data (bottom).

	Alpha	SD	CV	M	SD	CV
Original	0.,232	0,0.007	0,0.029	0,0.056	0,0.003	0,0.054
1/2 original	0,.465	0,0.013	0,0.029	0,0.056	0,0.004	0,0.064
Double proportion YOY	0,.232	0,0.007	0,0.028	0,0.056	0,0.003	0,0.053
1/2 proportion	0,.232	0,0.007	0,0.029	0,0.056	0,0.003	0,0.055

	Original	1/2 original	Double proportion YOY	1/2 proportion YOY
Pup CV	0.059 (0.024)	0.091 (0.132)	0.059 (0.024)	0.06 (0.024)
Total CV	0.046 (0.0195)	0.057 (0.034)	0.045 (0.0196)	0.046(0.02)
Proportion pup	0.177 (0.0203)	0.177 (0.02)	0.177 (0.0203)	0.177 (0.020)
Pup deviation		0 (0.004)	0 (0.002)	0 (0.001)
Total deviation		0 (.001)	0 (0)	0.002 (0.002)

Table 5. Changes in α and M with changes in the number of simulations run or the correlation coefficient between reproductive rates.

	α			M		
	Mean	SE	CV	Mean	SE	CV
r=0.1, 10 10,000 runs	0,0.251	0,0.007	0,0.026	0,0.056	0,0.003	0,0.051
r=0.1 1,000 runs	0,0.250	0,0.007	0,0.027	0,0.056	0,0.003	0,0.052
r=0.85, 10 10,000 runs	0,0.249	0,0.008	0,0.031	0,0.056	0,0.003	0,0.059
r=0.85 1,000 runs	0,0.250	0,0.008	0,0.032	0,0.056	0,0.003	0,0.061

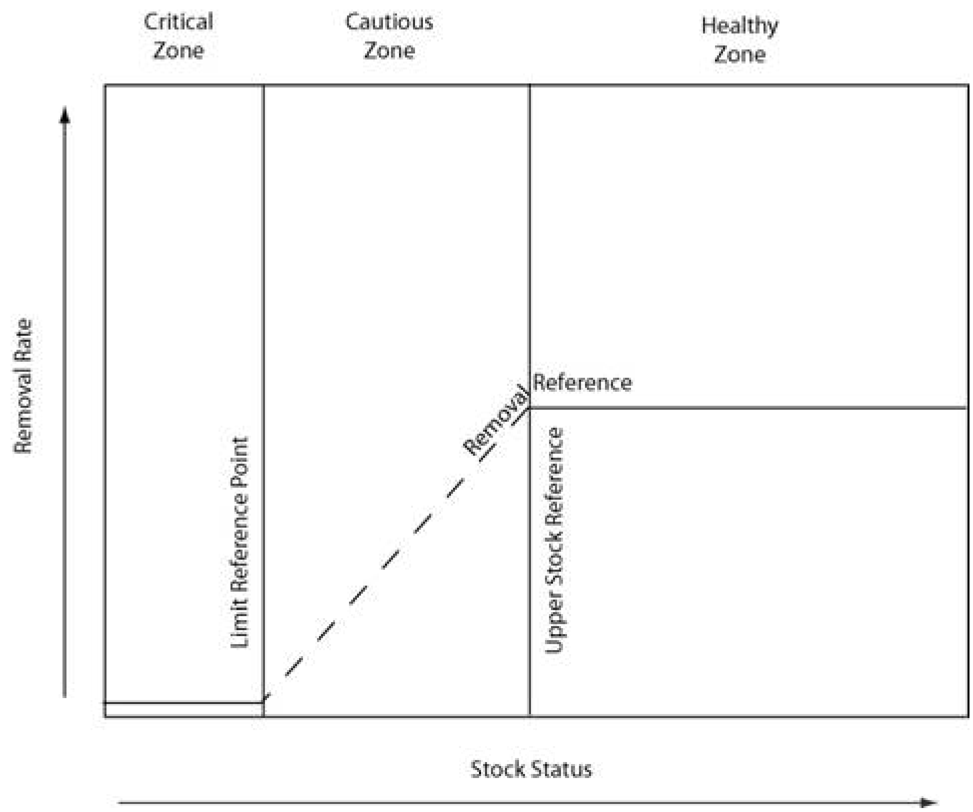


Figure 1. The general Precautionary Approach framework developed by Canada. The stock status zones are created by defining the Limit Reference Point (LRP) at the Critical:Cautious zone boundary, and an Upper Stock Reference Point (USR) at the Cautious:Healthy zone boundary and the Removal Reference for each of the three zones. The three-zoned diagram below shows these different elements.

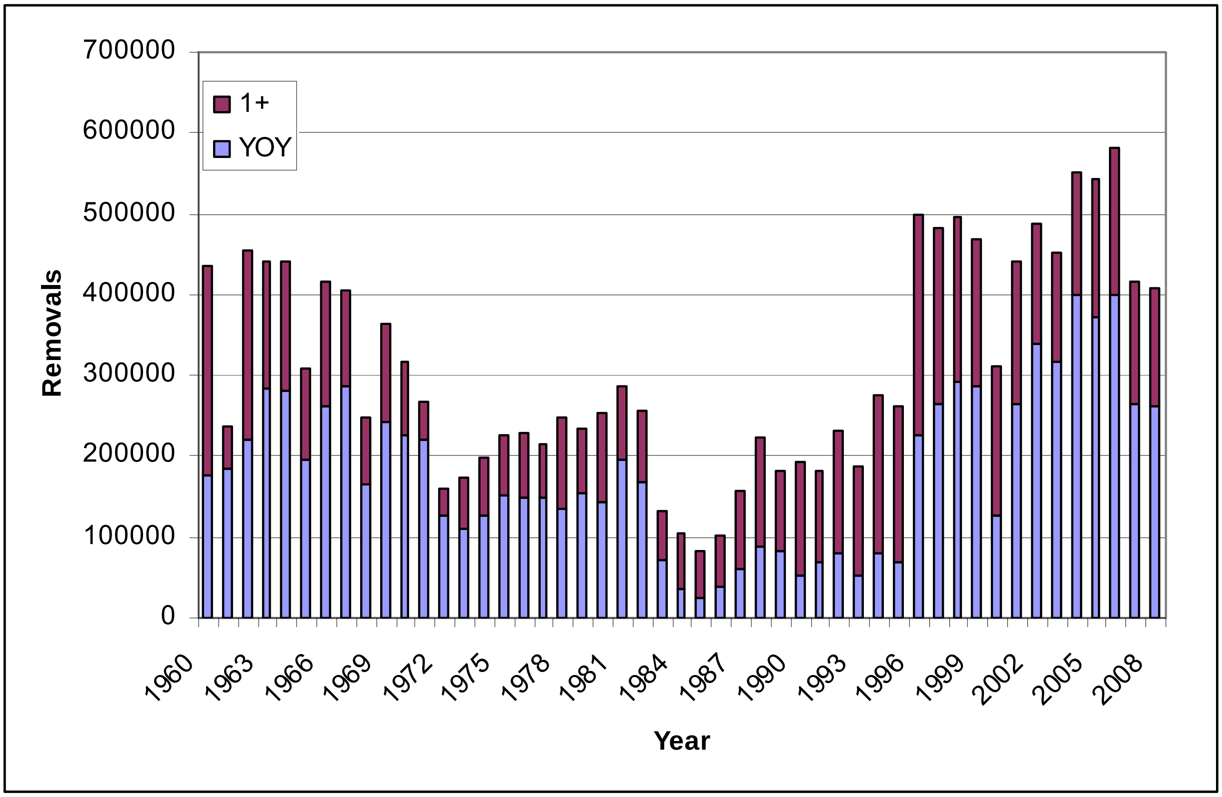


Figure 2. Total removals from the Northwest Atlantic harp seal population after correcting reported harvests for struck and lost, and mis-reporting of catches.

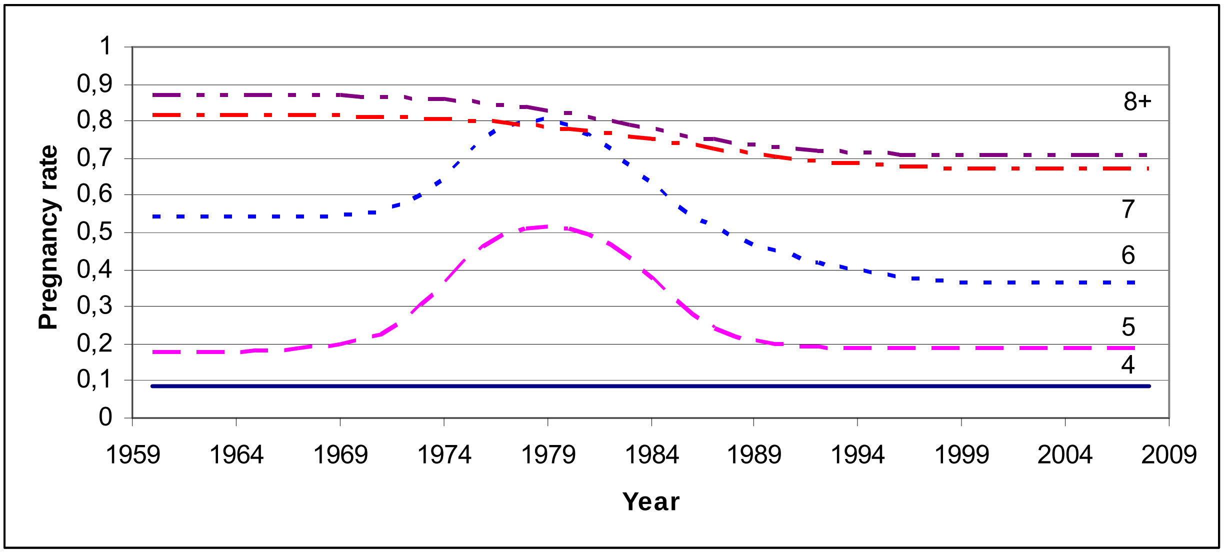
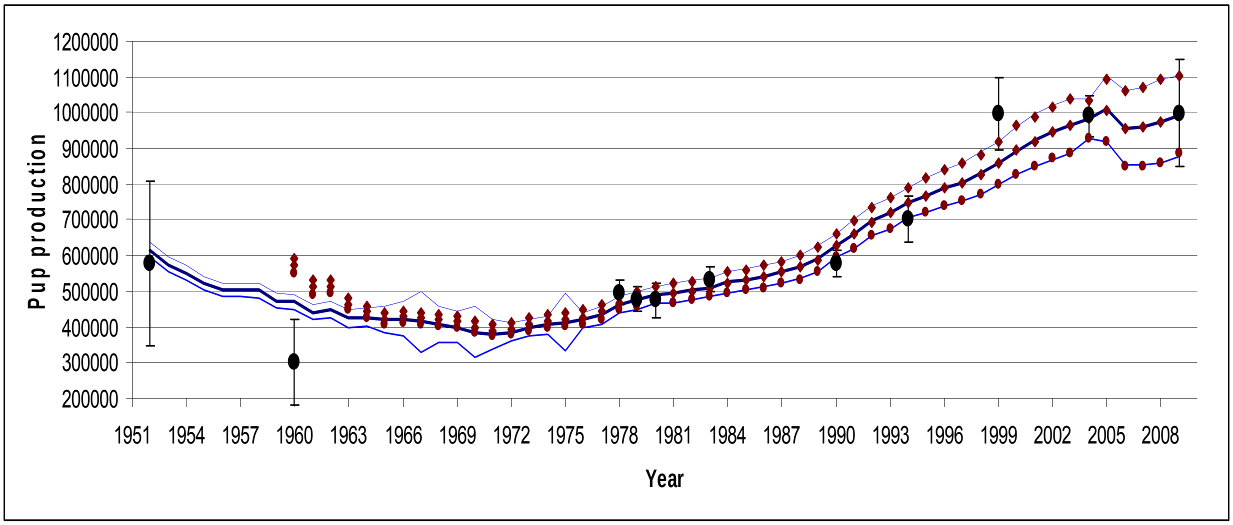


Figure 3. Age-specific reproduction rates after smoothing, used in the model fitting and projections.



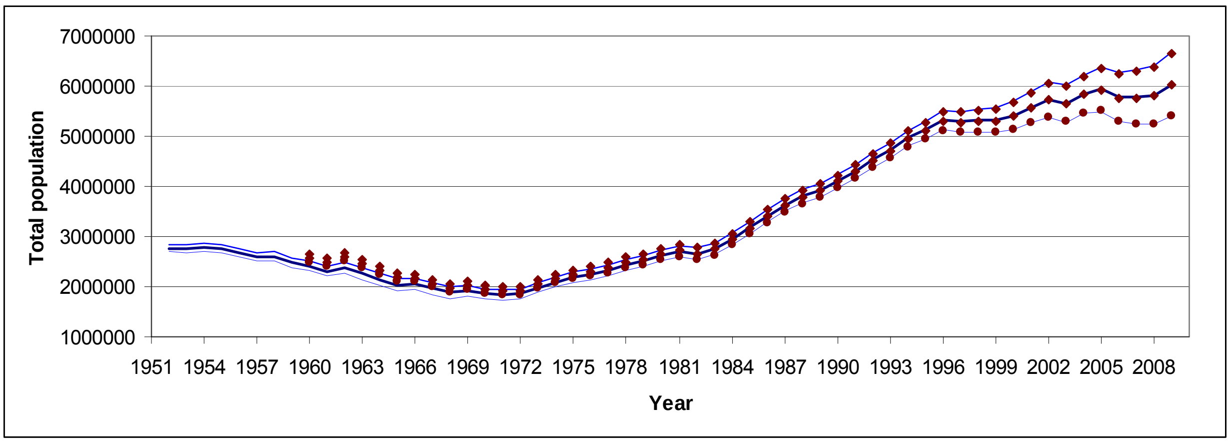


Figure 4. Estimated trends (Mean±SE) in pup production (top) and total abundance (bottom) of the northwest Atlantic harp seal population between 1952 and 2009 (lines) and between 1960 and 2009 (points). Points with error bars represent mark-recapture and aerial survey estimates of pup production.

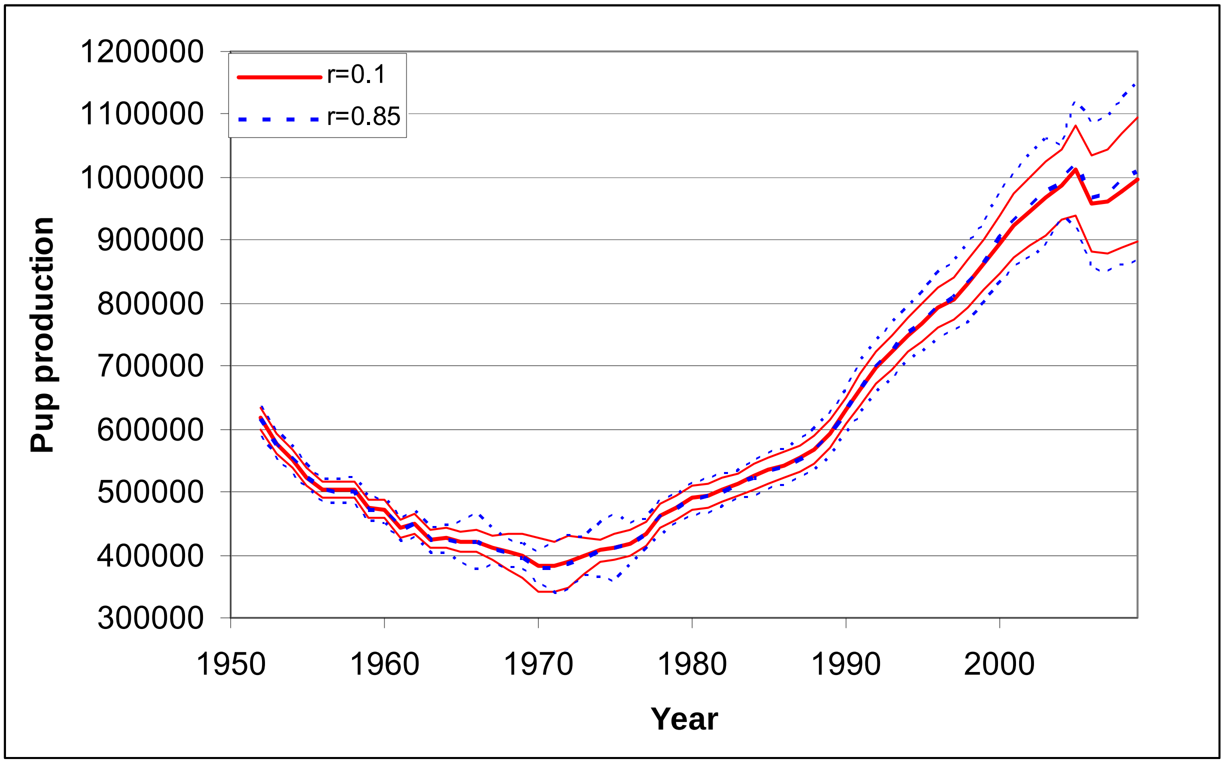


Figure 5. Effects of setting the correlation coefficient between reproductive rates among age classes at r=0.1 (solid line) or r=0.85 (dotted line) on pup production (mean±SE).

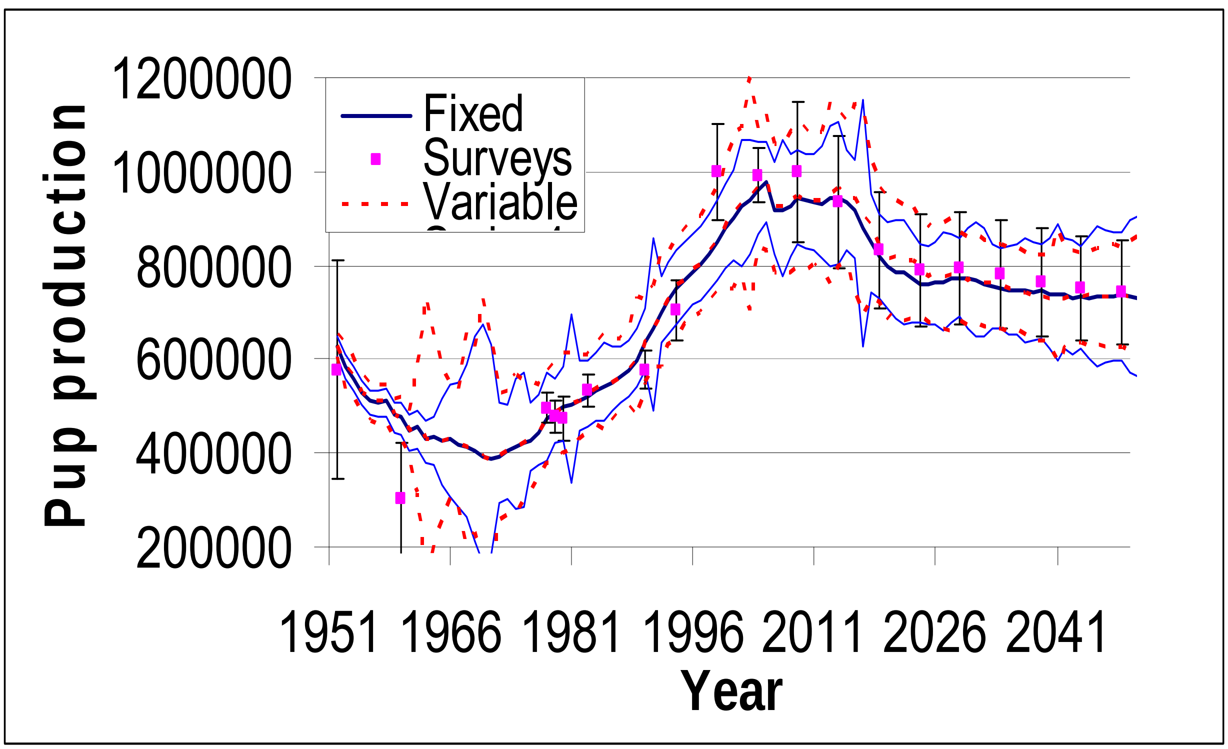
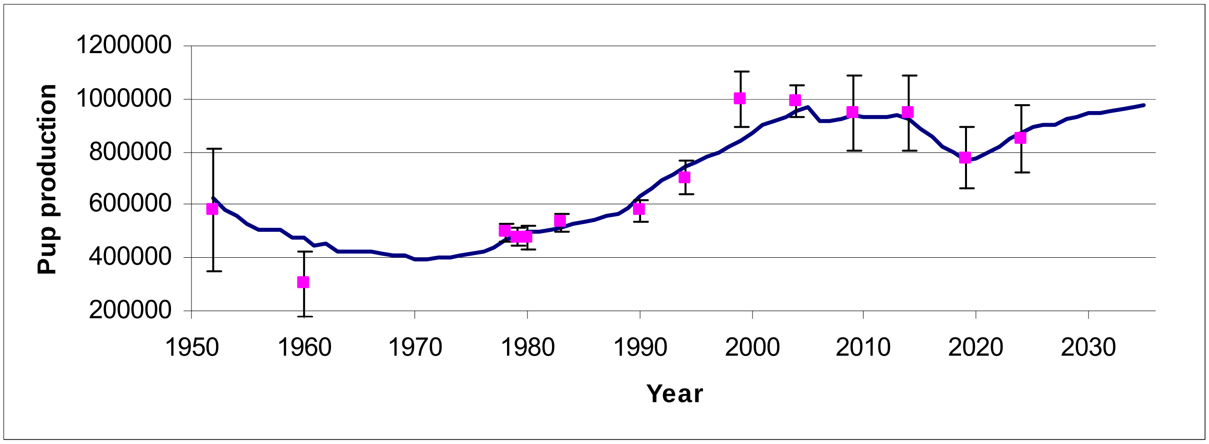


Figure 6. Comparison of pup production and total population subjected to a constant harvest and a variable harvest.



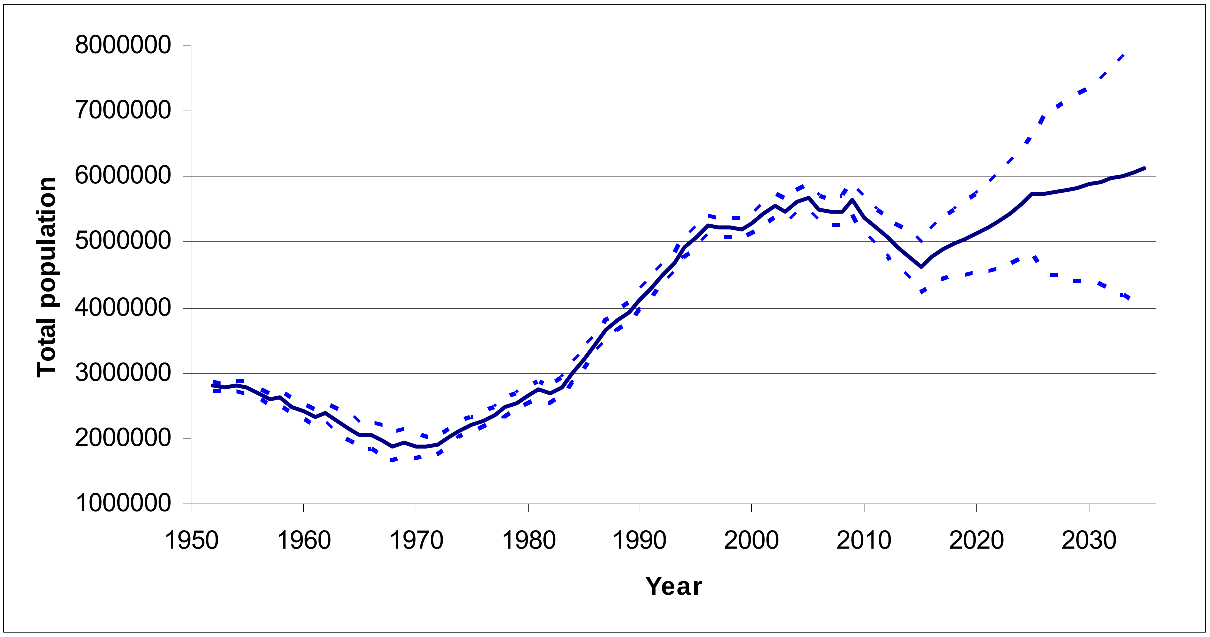


Figure 7. Change in pup production and population size when a harvest of 700,000 animals was applied in 2009 and 600,000 annually from 2010 until 2014 inclusive, then decreased to 200,000 per year.

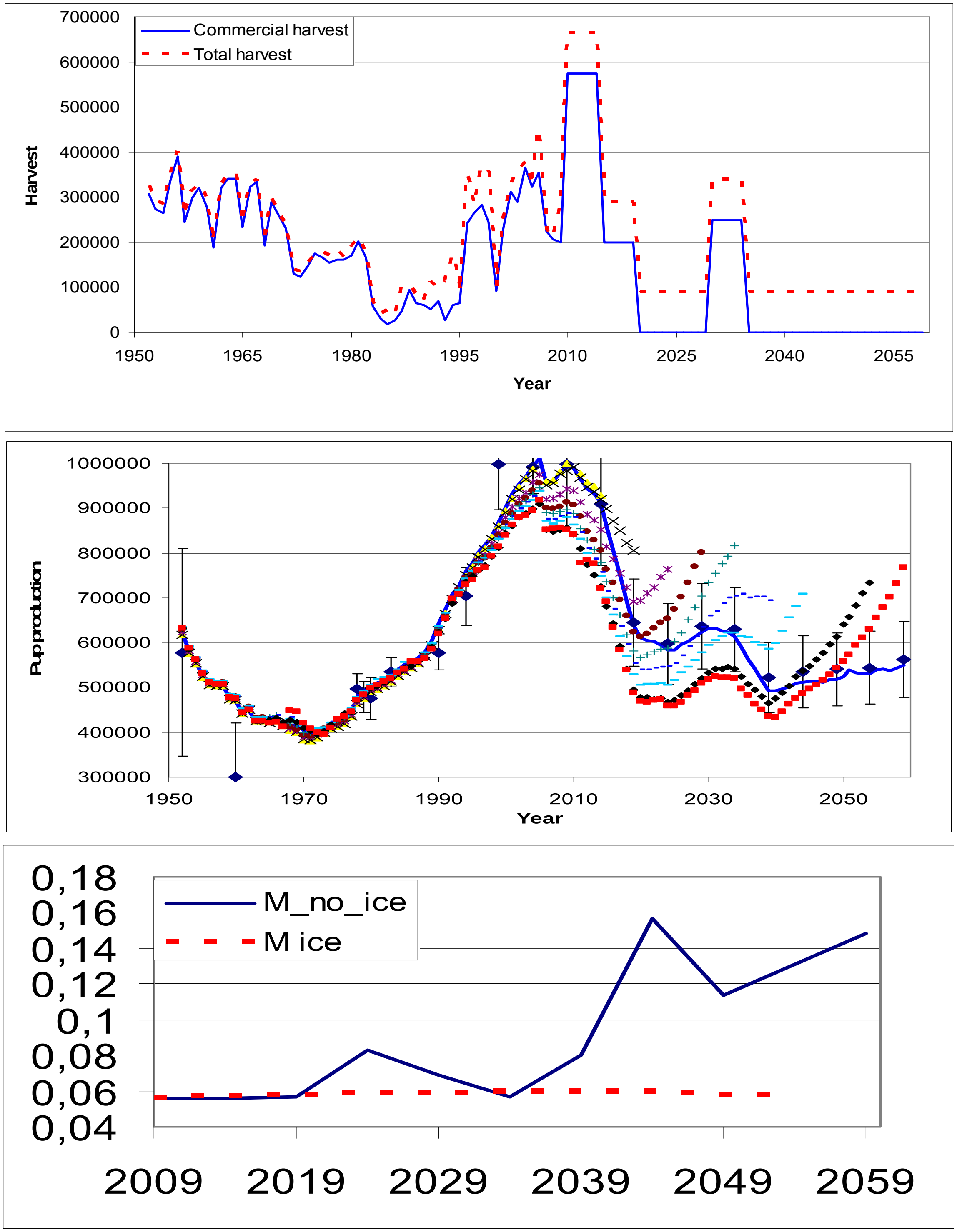


Figure 8. Harvest levels set in 5-year blocks under scenario 1, which assumed that M_{ice}=20% (top). Estimated pup production (middle) and total population (bottom) size. The solid line represents the true population (Reference model), the points represent runs every 5 years, assuming no M_{ice}.

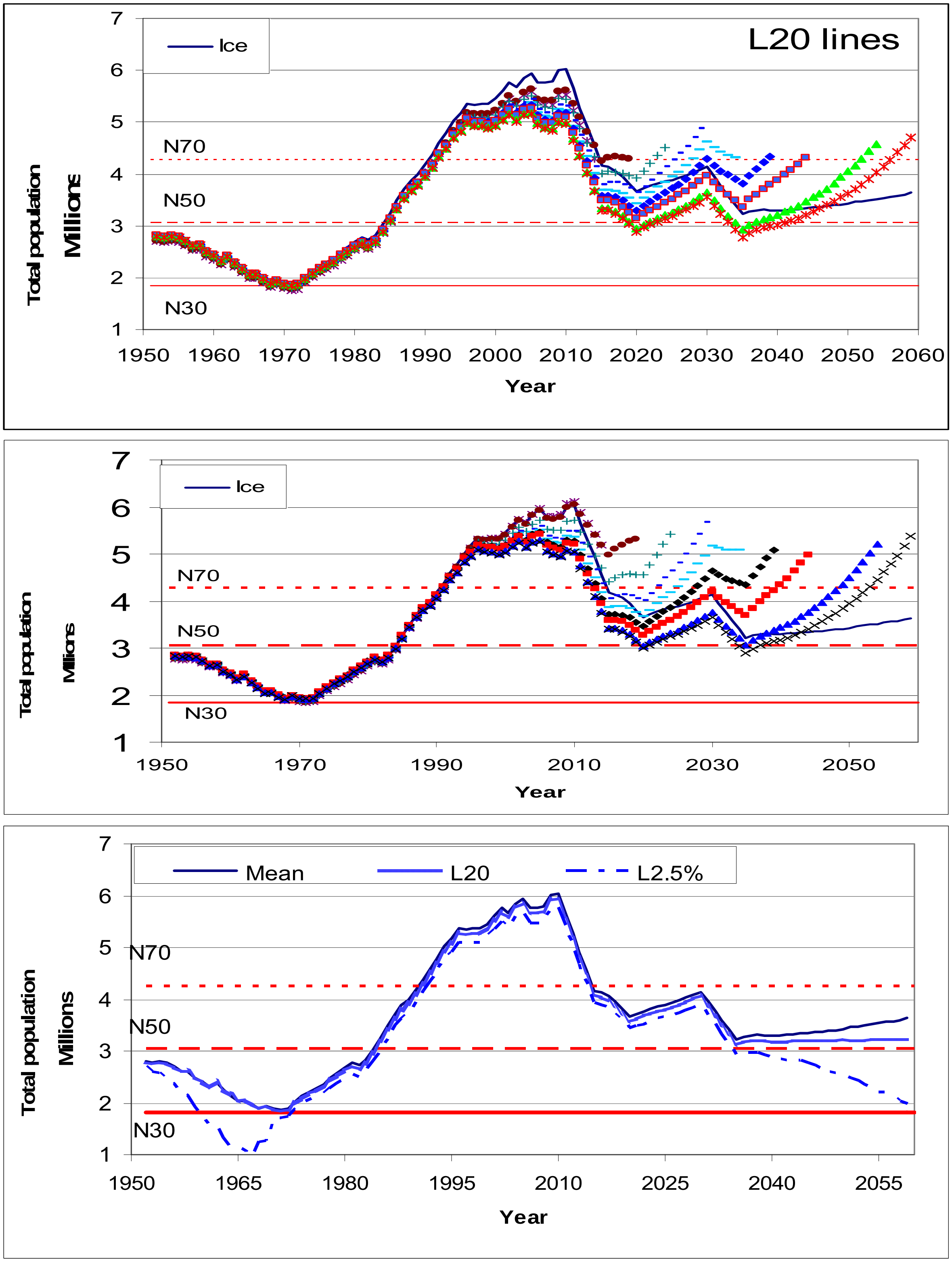


Figure 9. Changes in L20 as the Assessment model is refit in 5-year blocks (top), changes in total population size from the Assessment model (middle) and changes in the population as determined from the Reference model (bottom).

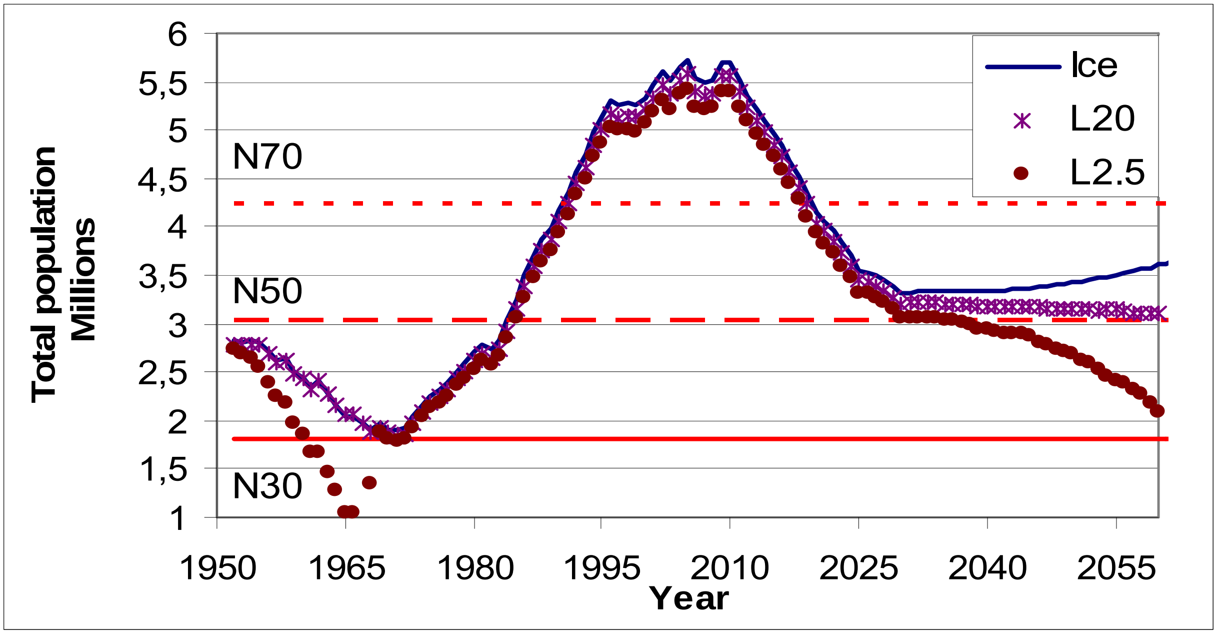


Figure 10. Changes in mean, L20 and L2.5 lines compared to N70, N50 and N30, assuming M_{ice}=20%, but the TAC was set to respect the management plan over a 10 year period for each projection.

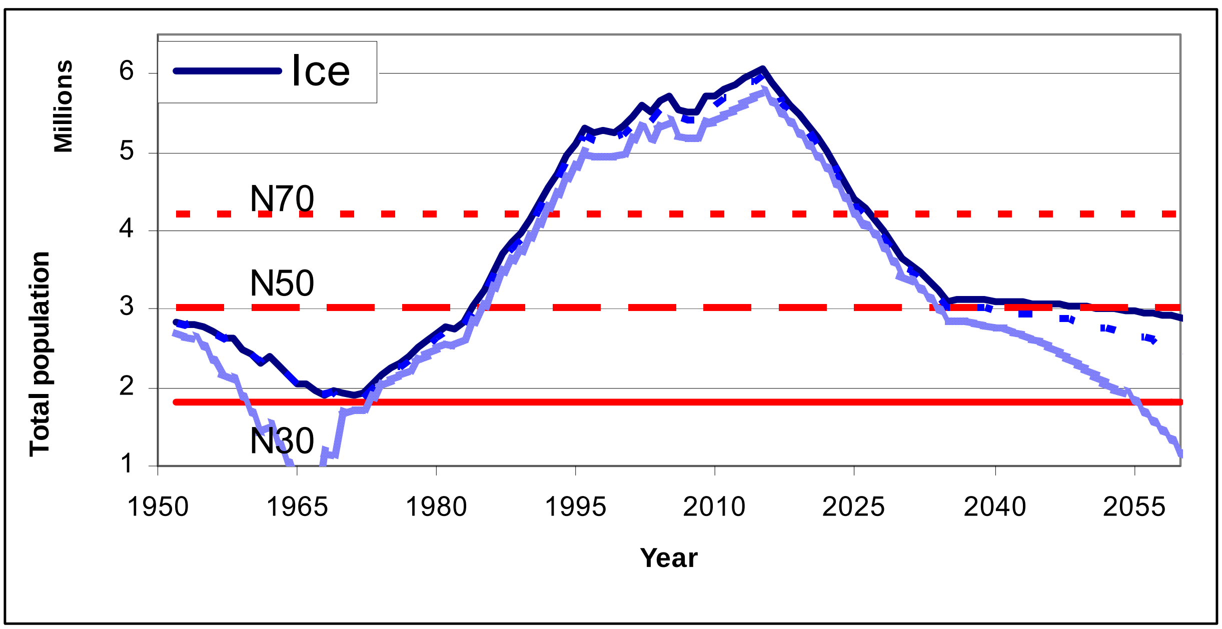


Figure 11. Changes in mean, L20 and L2.5 lines compared to N70, N50 and N30, assuming M_{ice}=20%, but the TAC was set to respect the management plan over a 20 year period for each projection.

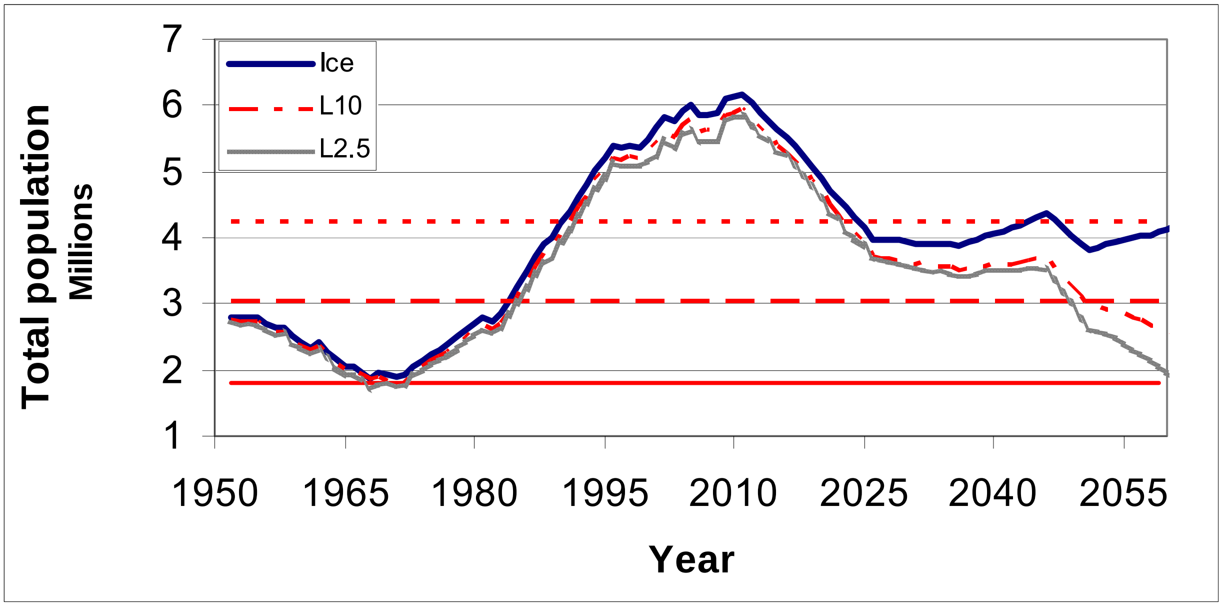


Figure 12. Changes in mean, L10 and L2.5 lines compared to N70, N50 and N30, assuming M_{ice}=20%, but the TAC was set to respect the management plan over a 10 year period for each projection with a 90% probability that the population was above N70.

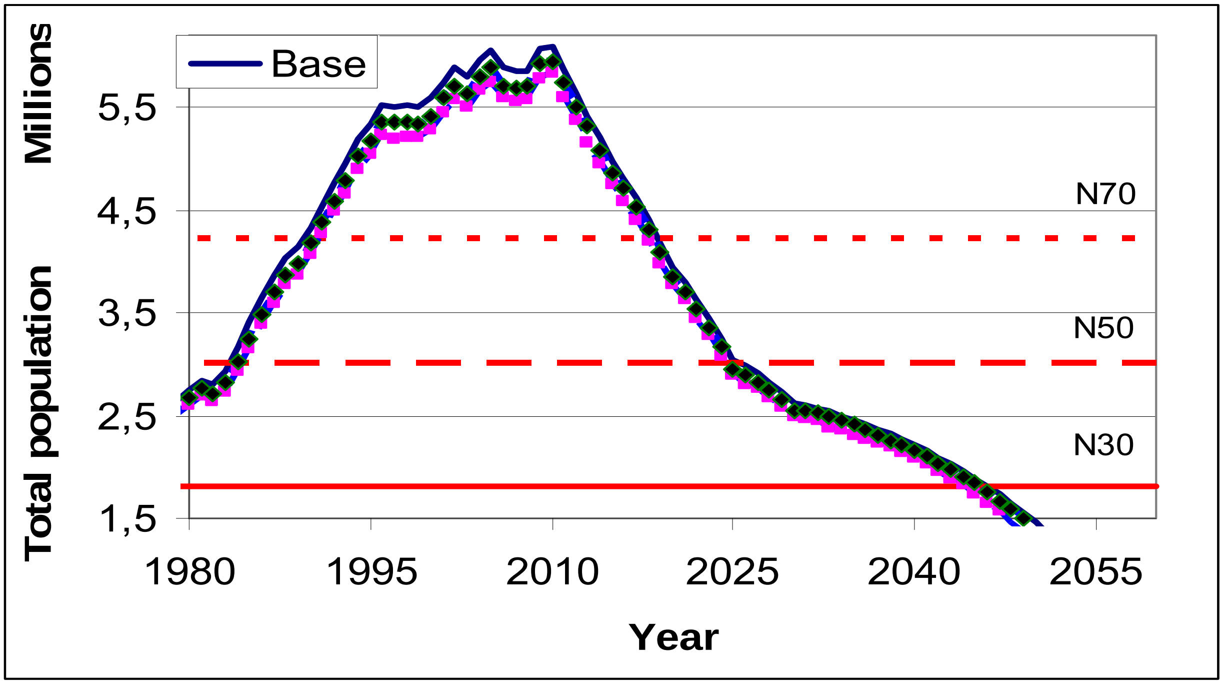


Figure 13. Estimated mean population (solid), L30 (diamond) and L5 (square) sizes. Where for L30, 70% of the runs were above this level, for L5, 95% of the runs were above this level.

APPENDIX A

A1: M_{ice}=20%, TAC set with an 80% probability that the population is above N70 for a 10-year projection. The first simulation examined the impact of different harvest scenarios within 5-year blocks when choosing an acceptable harvest level for the next five years. However, earlier scenarios indicated that changes in pup production in response to a harvest would not be expressed in the population until five years after a new harvest regime had been introduced. We examined a variation on the M_{ice} scenario by continuing to fit the model every five years to new survey data, but with the constraint that harvests must respect the management plan throughout a 10-year projection into the future. In the first projection, the Assessment model suggested that a constant harvest of 400,000 could be taken over the first 5-year block (2009-2014)(Fig. A1). After five years, this was reduced to 375,000 for the next 5-year block, and then reduced in 5-year blocks until the hunt would be closed completely in 2040 until the end of the simulation period. The total harvest for the commercial hunt was 5,725,000 animals. The Assessment model attempted to fit to the change in pup production and the fits to the 2014 and 2019 points were quite good. However, in subsequent projections the quality of the fit of the model to the survey points decreased as the model attempted to fit to increasingly lower estimates of pup production. Throughout the time series, little change was observed in the starting population, but M was observed to increase as the model attempted to fit to the lower estimates of pup production. This had the effect of reducing the fit to the peak estimates of pup production obtained from surveys 'flown' in 2004, 2009 and 2014 as the model was influenced by the increasing number of 'surveys' at lower pup production levels (Fig. A1). The reduction in harvests was in response to the decline in the L20 population size below N70 in the Assessment model. Under this scenario of setting harvest to a 10-year block, harvest levels were established based on the position of L20 relative to N70 (approximately 4.1 million animals) over a10-year period, with the starting point for the period being the most recent survey. In subsequent runs, when the model was refit to new survey data, L20 often fell below N70 during the period prior to the most recent survey (Fig. A2). Similar changes were also observed in the mean for the projections that did not include M_{ice}. Throughout this period the 'real' population mean and L20 as predicted by the Reference model were observed to decline to near N50 (≈3 million animals), but did not decline further. The L2.5 lines was observed to decline below N50, but did not reach N30 by the end of the simulation period.

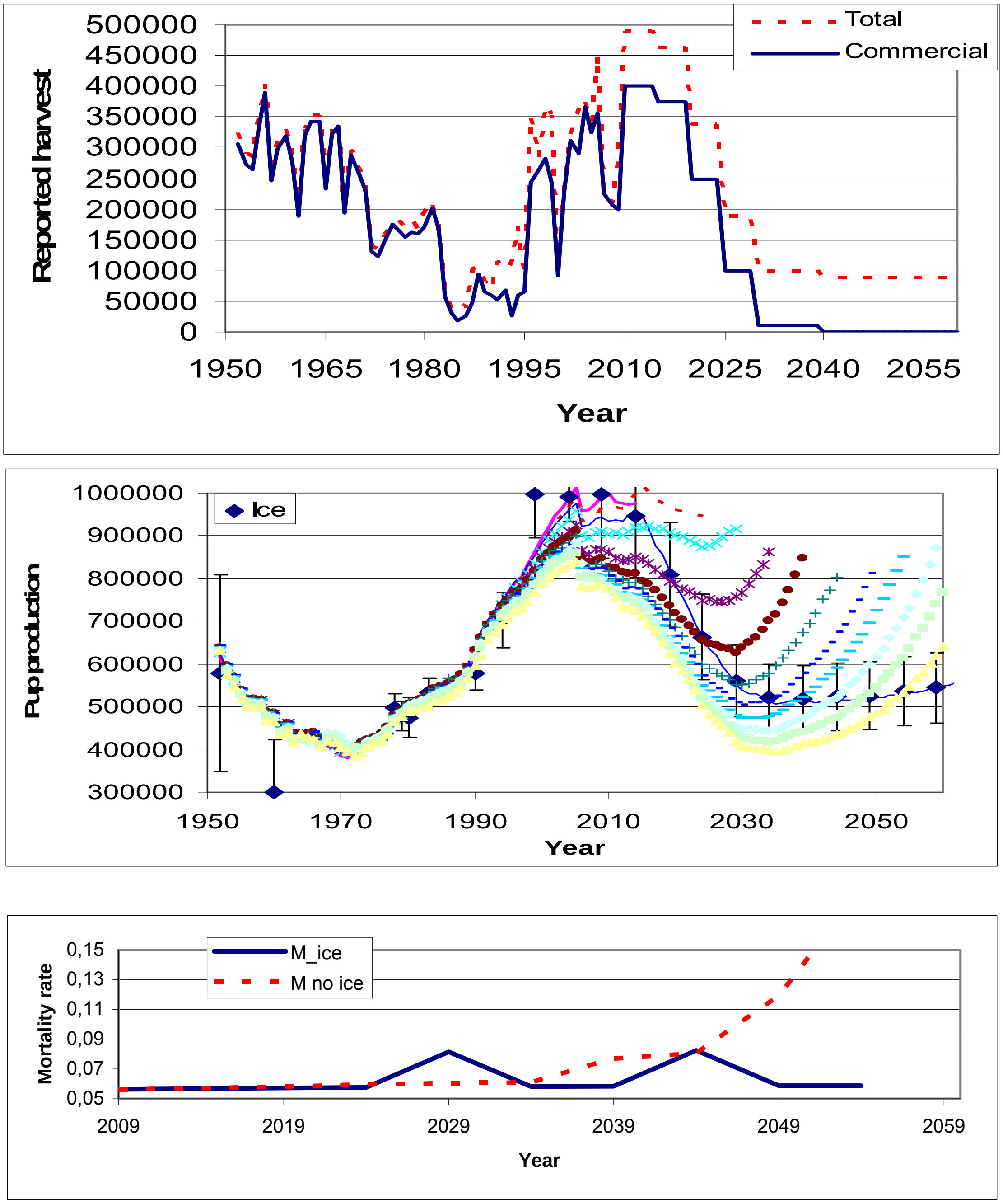


Figure A1a. Harvest levels were calculated to respect the plan for 10 years, but new assessments were completed in 5-year blocks, which assumed that M_{ice}=20%. Reported commercial and total removals (top), estimated pup production from the Reference model that included M_{ice} (solid line) and the Assessment model predictions (points)(middle). Estimated mortality rates from the Reference model (ice) and Assessment model (no ice)(bottom).

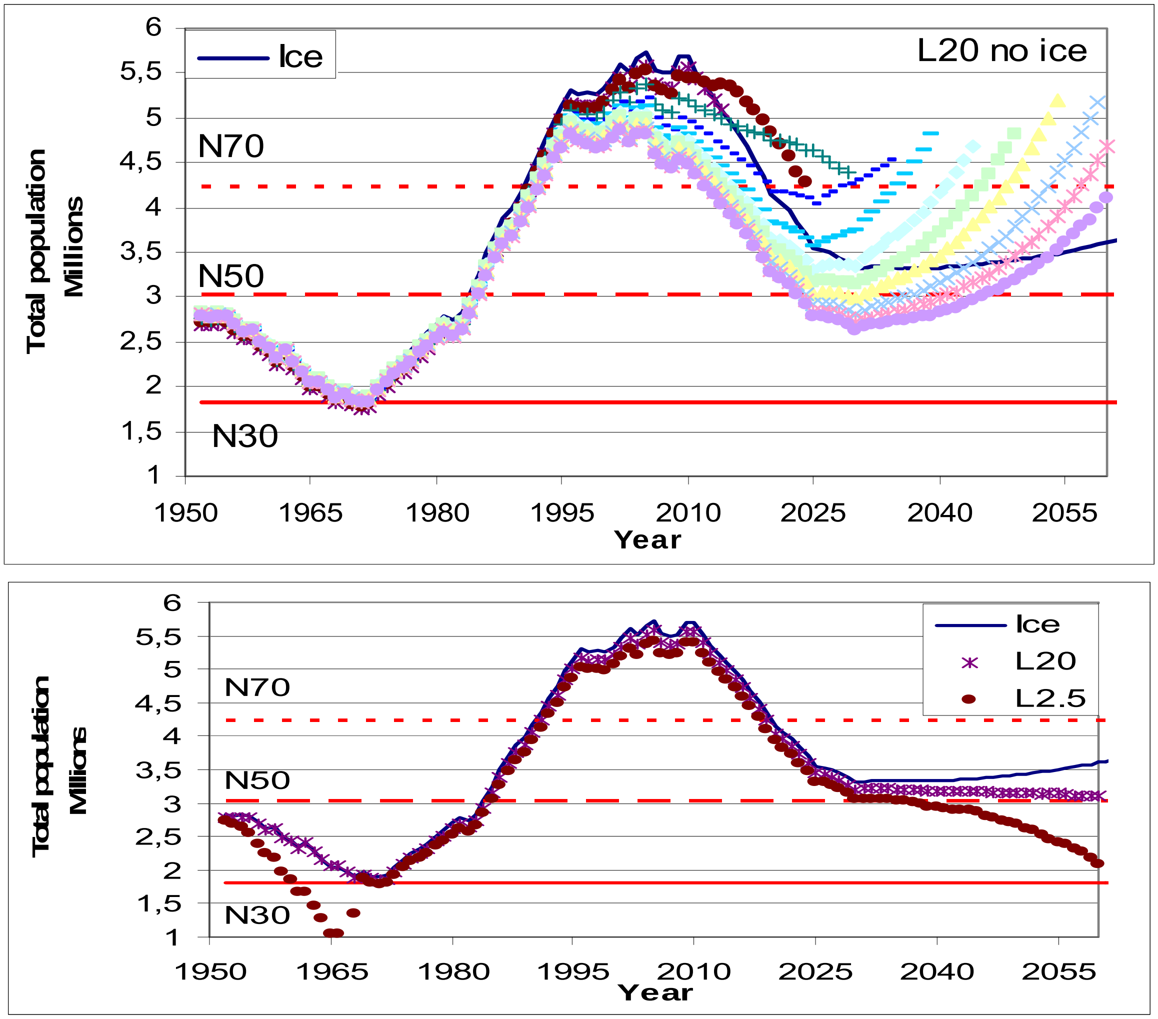


Figure A1b. Changes in L20 as the Assessment model is refit in 5-year blocks, but each projection estimates the TAC assuming a 10 year projection (top), and changes in the population as determined from the Reference model (bottom). Estimated pup production (mean±SD), L20 and L2.5, where for L20, 80% of the runs were above this level. At L2.5, 97.5% of the runs were above this level.

A2: M_{ice}=20%, TAC set with an 80% probability that the population is above N70. This simulation was the same as the previous one, but extended the period where the plan was projected to respect the plan from 10 years to 20 years. In the first projection, the Assessment model suggested that a constant harvest of 320 000 could be taken over the first 5-year block (2009-2014)(Fig. A2a). After five years, this was reduced to 315,000 for the next 5-year block, and then reduced in 5-year blocks until the hunt would be closed completely in 2034 until the end of the simulation period. The total harvest for the commercial hunt was 7,100,000 animals. The Assessment model attempted to fit to the change in pup production and the fits to the 2014 and 2019 points were quite good. However, in subsequent projections the quality of the fit of the model to the survey points decreased as the model attempted to fit to increasingly lower estimates of pup production. Throughout the time series, little change was observed in the starting population, but M was observed to increase as the model attempted to fit to the lower estimates of pup production. This had the effect of reducing the fit to the peak estimates of pup production obtained from surveys 'flown' in 2004, 2009 and 2014 as the model was influenced by the increasing number of 'surveys' at lower pup production levels (Fig. A2a). The reduction in harvests was in response to the decline in the L20 population size below N70 in the Assessment model. Under this scenario of setting harvest to a 10-year block, harvest levels were established based on the position of L20 relative to N70 (approximately 4.1 million animals) over a10 year period, with the starting point for the period being the most recent survey. In subsequent runs, when the model was refit to new survey data, L20 often fell below N70 during the period prior to the most recent survey (Fig. A2). Similar changes were also observed in the mean for the projections that did not include M_{ice}. Throughout this period the 'real' population mean and L20 as predicted by the Reference model were observed to decline to near N50 (≈3 million animals), but did not decline further. The L2.5 lines was observed to decline below N50, but did not reach N30 by the end of the simulation period.

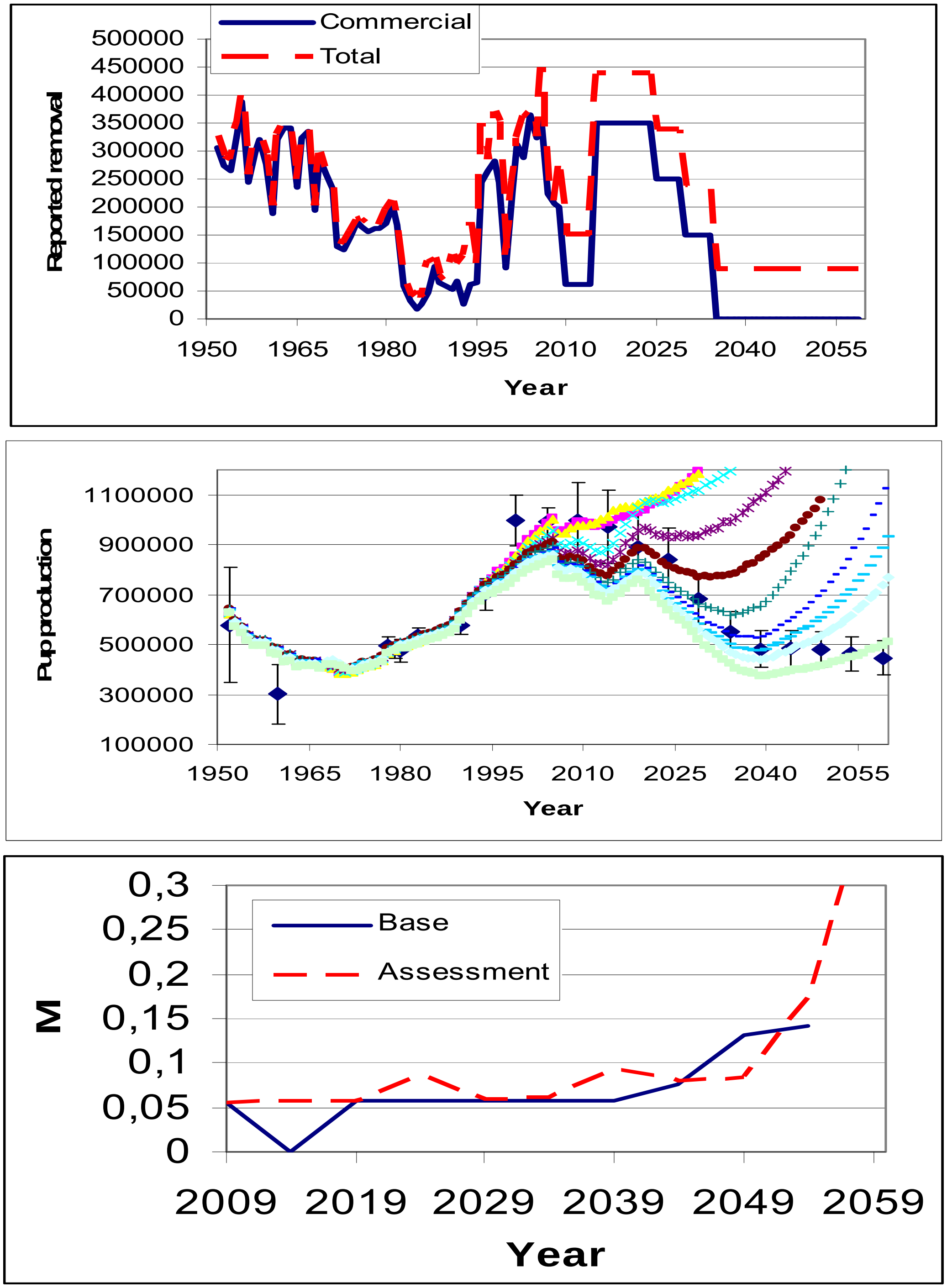
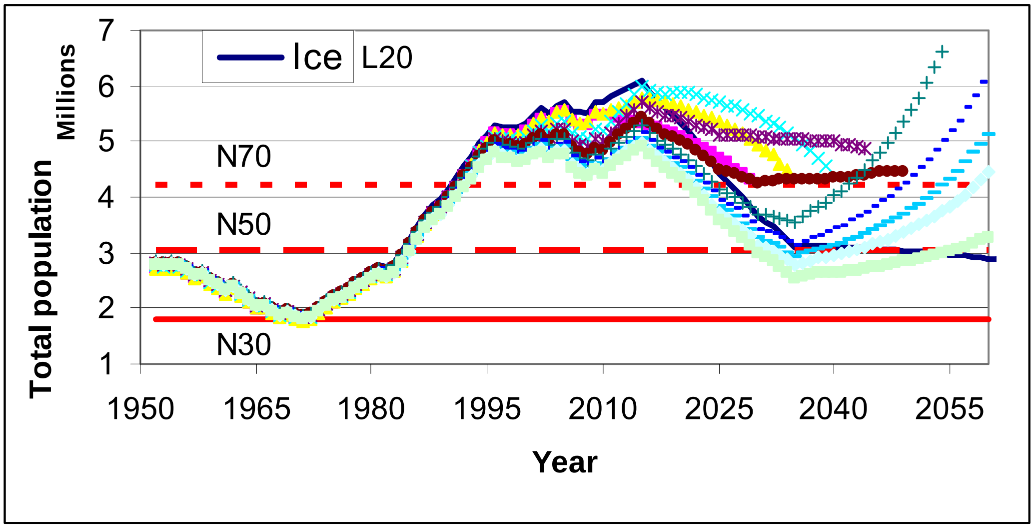


Figure A2a. Harvest levels were calculated to respect the plan for 10 years, but new assessments were completed in 5 year blocks, which assumed that M_{ice}=20%. Reported commercial and total removals (top), estimated pup production from the Reference model that included M_{ice} (solid line) and the Assessment model predictions (points)(middle). Estimated mortality rates from the Reference model (ice) and Assessment model (no ice)(bottom).



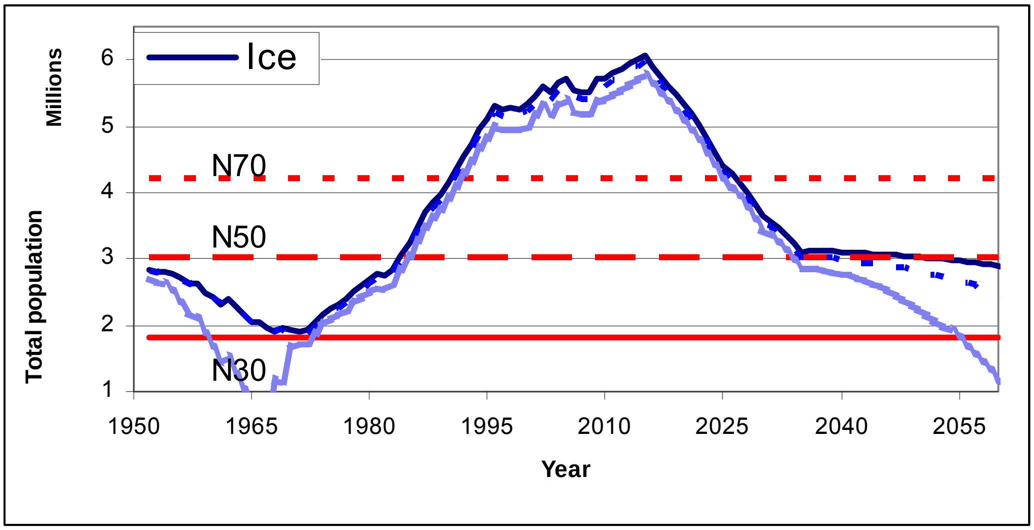


Figure A2b. Changes in the Reference population (solid line) and the Assessment L20 as the Assessment model is refit in 5-year blocks, but each run estimates the TAC assuming a 20 year projection (top), and changes in the population as determined from the Reference model (bottom). Estimated mean population (solid), L20 (dotted) and L2.5 (thickened) sizes. Where for L20, 80% of the runs were above this level, and for L2.5, 97.5% of the runs were above this level.

A3: M_{ice}=20%, TAC set to respect the management plan for 10 year projection, with a 90% probability that the population remains above N70. This simulation examined the effects of setting the probability of respecting the plan to 90% (instead of 80%) for a period of 10 years. In the first projection, the Assessment model suggested that a constant harvest of 350 000 could be taken over the first 5 year block (2009-2014)(Fig. A3a). After five years, this was reduced to 306,000 for the next 5-year block, and then reduced in 5-year blocks to 10,000 animals for a decade (2035-2044), increased to 200 000 for a brief 5 year period, then reduced to 20,000 until the end of the simulation period. The total harvest for the commercial hunt was 6,800,000 animals. The Assessment model attempted to fit to the change in pup production and the fits to the 2014 and 2019 points were quite good. However, in subsequent projections the quality of the fit of the model to the survey points decreased as the model attempted to fit to increasingly lower estimates of pup production. Throughout the time series, little change was observed in the starting population, but M was observed to increase as the model attempted to fit to the lower estimates of pup production. This had the effect of reducing the fit to the peak estimates of pup production obtained from surveys 'flown' in 2004, 2009 and 2014 as the model was influenced by the increasing number of 'surveys' at lower pup production levels (Fig. A3a). The reduction in harvests was in response to the decline in the L20 population size below N70 in the Assessment model. Under this scenario of setting harvest to a10-year block, harvest levels were established based on the position of L20 relative to N70 (approximately 4.1 million animals) over a10-year period, with the starting point for the period being the most recent survey. In subsequent runs, when the model was refit to new survey data, L20 often fell below N70 during the period prior to the most recent survey (Fig. A2). Similar changes were also observed in the mean for the projections that did not include M_{ice}. Throughout this period the 'real' population mean and L20 as predicted by the Reference model were observed to decline to near N50 (≈3 million animals), but did not decline further. The L2.5 line was observed to decline below N50, but did not reach N30 by the end of the simulation period.

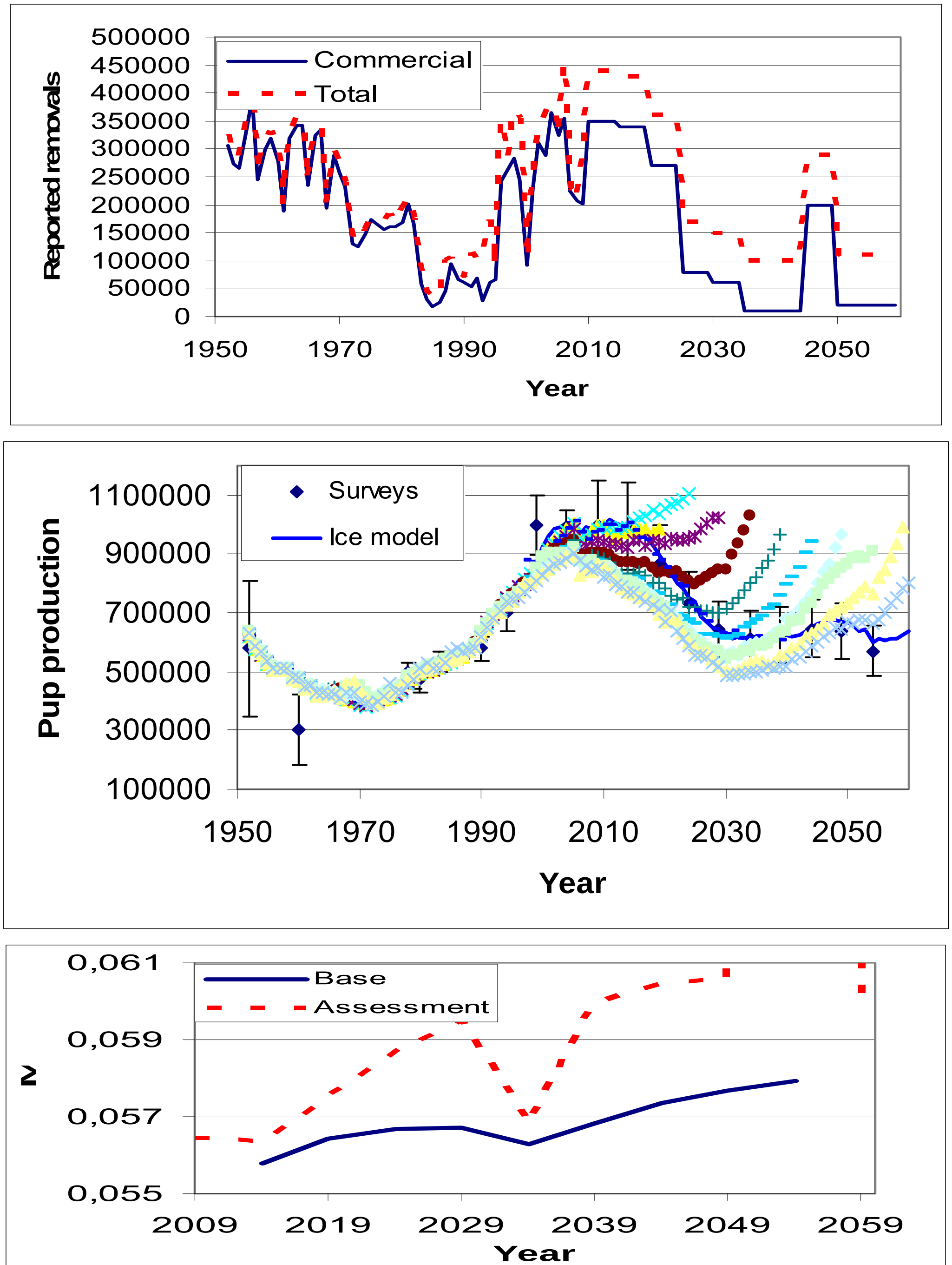


Figure A3a. Harvest levels were calculated to respect the plan for 10 years, with a 90% probability that the population remains above N70. New assessments were completed in 5-year blocks. Reported commercial and total removals (top), estimated pup production from the Reference model that included M_{ice} (solid line) and the Assessment model predictions (points)(middle). Estimated mortality rates from the Reference model (ice) and Assessment model (no ice)(bottom). The Assessment model had a maximum of 0.18 (not shown).

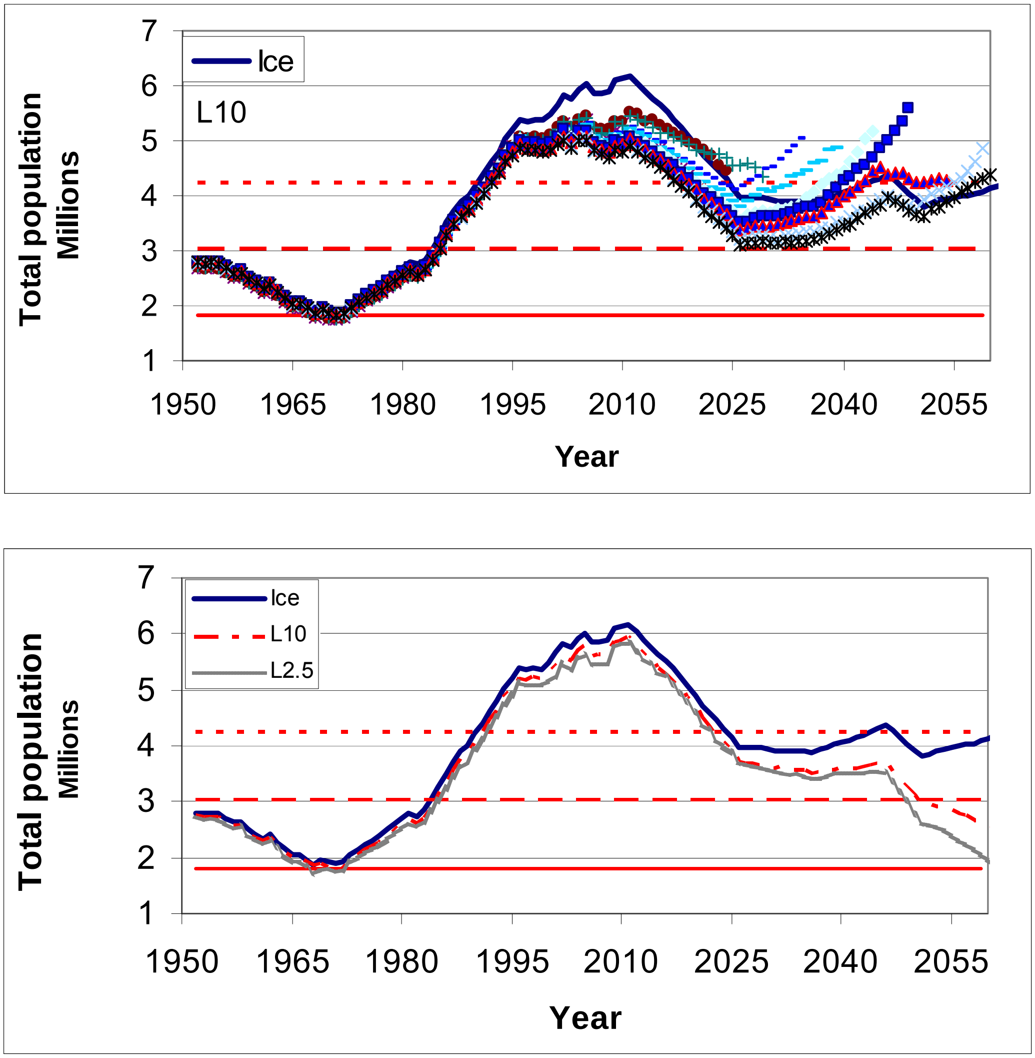


Figure A3b. Changes in the base population (solid line) and the Assessment L10 as the Assessment model is refit in 5-year blocks, but each run estimates the TAC assuming a 10 year projection (top), and changes in the population as determined from the Reference model (bottom). Estimated mean population (solid), L10 (dotted) and L2.5 (thickened) sizes. Where for L10, 90% of the runs were above this level, and for L2.5, 97.5% of the runs were above this level.

A4: M_{ice}=20%, TAC set to respect the management plan for 10 year projection, with a 75% probability that the population remains above N70. This simulation examined the effects of setting the probability of the population staying above N70 to 75% (instead of 80%) for a period of 10 years. In the first projection, the Assessment model suggested that a constant harvest of 425,000 could be taken over the first 5 year block (2009-2014)(Fig. A4a). After 5 years, this was reduced to 350,000 for the next 5-year block, and then reduced to 80 000 until the hunt was closed in 2029, where it remained until the end of the simulation period. The total harvest for the commercial hunt was 5,525,000 animals. The Assessment model attempted to fit to the change in pup production and the fits to the 2014 and 2019 points were quite good. However, in subsequent projections the quality of the fit of the model to the survey points decreased as the model attempted to fit to increasingly lower estimates of pup production. Throughout the time series, little change was observed in the starting population, but M was observed to increase as the model attempted to fit to the lower estimates of pup production. This had the effect of reducing the fit to the peak estimates of pup production obtained from surveys 'flown' in 2004, 2009 and 2014 as the model was influenced by the increasing number of 'surveys' at lower pup production levels (Fig. A4a). The reduction in harvests was in response to the decline in the L20 population size below N70 in the Assessment model. Under this scenario of setting harvest to a 10 year block, harvest levels were established based on the position of L20 relative to N70 (approximately 4.1 million animals) over a10-year period, with the starting point for the period being the most recent survey. In subsequent runs, when the model was refit to new survey data, L20 often fell below N70 during the period prior to the most recent survey (Fig. A2). Similar changes were also observed in the mean for the projections that did not include M_{ice}. Throughout this period the 'real' population mean and L20 as predicted by the Reference model were observed to decline to near N50 (≈3 million animals), but did not decline further. The L2.5 line was observed to decline below N50, but did not reach N30 by the end of the simulation period.

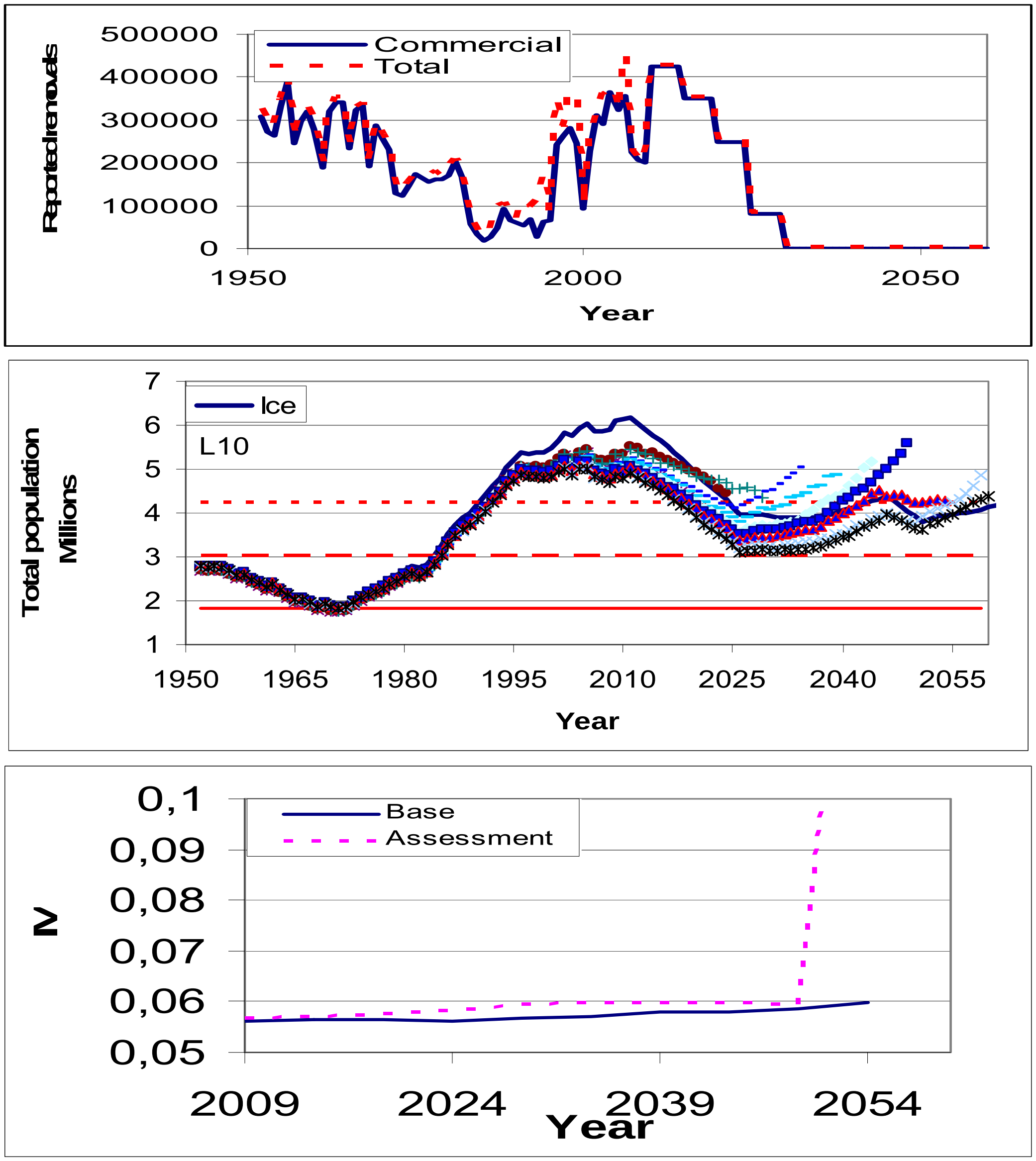


Figure A4a. Harvest levels were calculated to respect the plan for 10 years, with a 90% probability that the population remains above N70. New assessments were completed in 5-year blocks. Reported commercial and total removals (top), estimated pup production from the Reference model that included M_{ice} (solid line) and the assessment model predictions (points)(middle). Estimated mortality rates from the Reference model (ice) and Assessment model (no ice)(bottom). The Assessment model had a maximum of 0.2 (not shown).

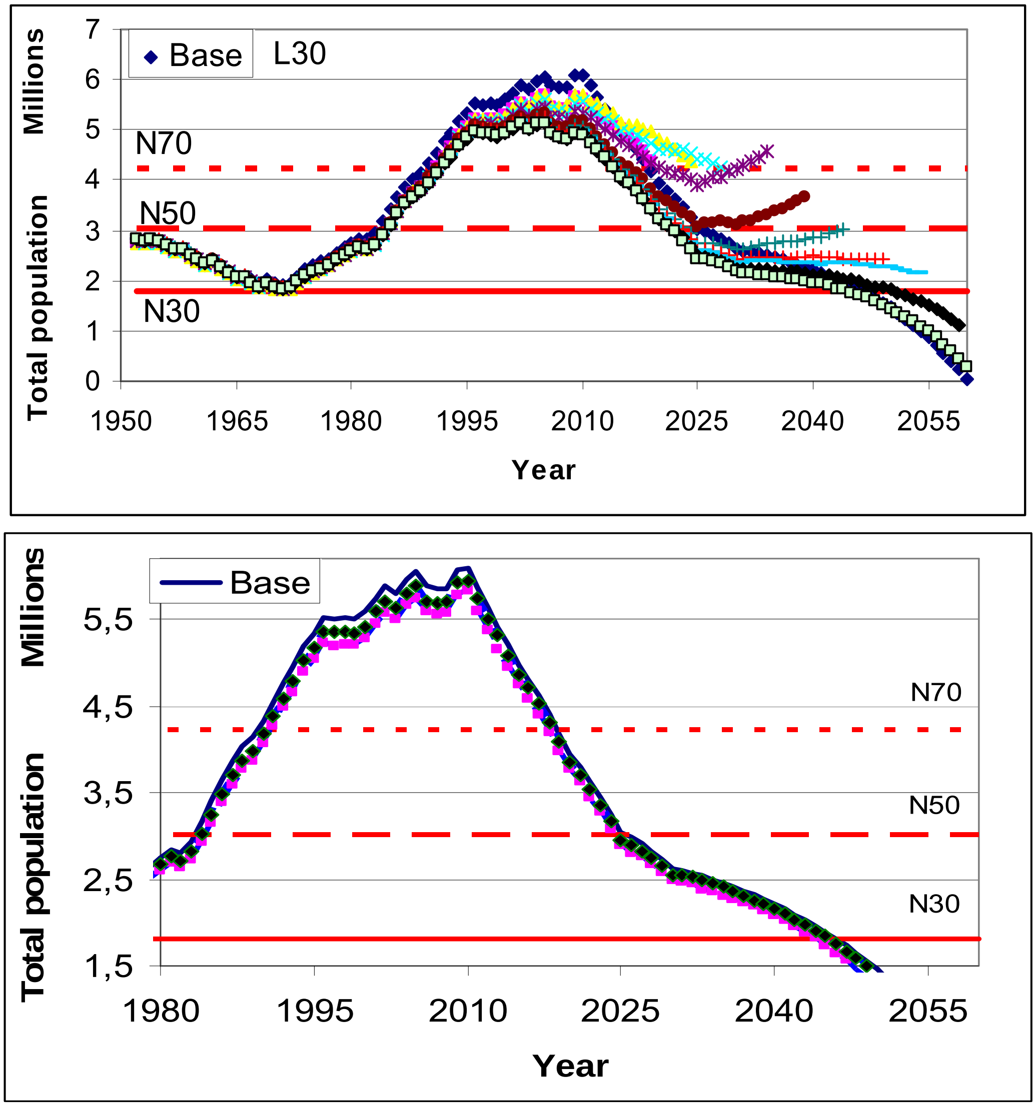


Figure A4b. Changes in the Reference population (solid line) and the Assessment L30 lines as the Assessment model is refit in 5-year blocks, but each run estimates the TAC assuming a 10-year projection (top), and changes in the population as determined from the Reference model (bottom). Estimated mean population (solid), L30 (diamond) and L5 (square) sizes. Where for L30, 70% of the runs were above this level, for L5 95% of the runs were above this level.

Appendix 1. Removals from the population evaluated in 5-year blocks assuming a constant harvest is taken in each year and a variable harvest, where the harvest is 100%, 50%, 150%, 50%, and 150%.

	Constant harvest						Variable harvest	
Year	arctic	greenland	canpup	can1+	by1plus	bypup	canpup	can1+
1952	1784	16400	198063	109045	0	0	198063	109045
1953	1784	16400	197975	74911	0	0	197975	74911
1954	1784	19150	175034	89382	0	0	175034	89382
1955	1784	15534	252297	81072	0	0	252297	81072
1956	1784	10973	341397	48013	0	0	341397	48013
1957	1784	12884	165438	80042	0	0	165438	80042
1958	1784	16885	140996	156790	0	0	140996	156790
1959	1784	8928	238832	81302	0	0	238832	81302
1960	1784	16154	156168	121182	0	0	156168	121182
1961	1784	11996	168819	19047	0	0	168819	19047
1962	1784	8500	207088	112901	0	0	207088	112901
1963	1784	10111	270419	71623	0	0	270419	71623
1964	1784	9203	266382	75281	0	0	266382	75281
1965	1784	9289	182758	51495	0	0	182758	51495
1966	1784	7057	251135	72004	0	0	251135	72004
1967	1784	4242	277750	56606	0	0	277750	56606
1968	1784	7116	156458	36238	0	0	156458	36238
1969	1784	6438	233340	55472	0	0	233340	55472
1970	1784	6269	217431	40064	15	53	217431	40064
1971	1784	5572	210579	20387	99	391	210579	20387
1972	1784	5994	116810	13073	141	480	116810	13073
1973	1784	9212	98335	25497	107	358	98335	25497
1974	1784	7145	114825	32810	41	141	114825	32810
1975	1784	6751.5	140638	33725	66	219	140638	33725
1976	1784	11956	132085	32917	169	923	132085	32917
1977	1784	12866	126982	28161	296	1281	126982	28161
1978	2129	16638	116190	45533	538	2381	116190	45533
1979	3620	17544.5	132458	28083	511	2799	132458	28083
1980	6350	15255	132421	37105	263	2454	132421	37105
1981	4672	22973.5	178394	23775	382	3539	178394	23775
1982	4881	26926.5	145274	21465	343	3442	145274	21465
1983	4881	24784.5	50058	7831	458	4504	50058	7831
1984	4881	25828.5	23922	7622	425	3683	23922	7622
1985	4881	20785	13334	5701	632	4225	13334	5701
1986	4881	26098.5	21888	4046	1042	7136	21888	4046
1987	4881	37859	36350	10446	1978	11118	36350	10446
1988	4881	40414.75	66972	27074	1391	7154	66972	27074
1989	4881	42970.5	56346	8958	799	9457	56346	8958
1990	4881	45526.25	34402	25760	921	2700	34402	25760
1991	4881	48082	42382	10206	615	9074	42382	10206
1992	4881	50637.75	43866	24802	6507	18969	43866	24802
1993	4881	56319	16401	10602	7596	18876	16401	10602
1994	4881	59684	25223	36156	11374	35881	25223	36156

	Constant harvest						Variable harvest	
Year	arctic	greenland	canpup	can1+	by1plus	bypup	canpup	can1+
1995	4881	66297.5	34106	31661	6754	13641	34106	31661
1996	4881	73947	184856	58050	18436	10765	184856	58050
1997	1804	68815.5	220476	43734	5328	13541	220476	43734
1998	719	81272	251403	31221	1070	3571	251403	31221
1999	368	93117	237644	6908	6361	9750	237644	6908
2000	280	98458.5	85035	7020	1632	9715	85035	7020
2001	405	85427.5	214754	11739	4903	14572	214754	11739
2002	715	66734.5	297764	14603	3837	5492	297764	14603
2003	715	66149	280174	9338	1881	3486	280174	9338
2004	715	70585.5	353553	12418	3832	8494	353553	12418
2005	715	91695.5	319127	4699	3217	8351.8	319127	4699
2006	715	92210	346426	8441	3217	8351.8	346426	8441
2007	715	81446.5	221488	3257	3217	8351.8	221488	3257
2008	715	81446.5	206171	265	3217	8351.8	206171	265
2009	715	85000	180000	20000	3217	8351.8	180000	20000
2010	715	85000	405000	45000	3217	8351.8	405000	45000
2011	715	85000	405000	45000	3217	8351.8	202500	22500
2012	715	85000	405000	45000	3217	8351.8	607500	67500
2013	715	85000	405000	45000	3217	8351.8	202500	22500
2014	715	85000	405000	45000	3217	8351.8	607500	67500
2015	715	85000	270000	30000	3217	8351.8	270000	30000
2016	715	85000	270000	30000	3217	8351.8	135000	15000
2017	715	85000	270000	30000	3217	8351.8	405000	45000
2018	715	85000	270000	30000	3217	8351.8	135000	15000
2019	715	85000	270000	30000	3217	8351.8	405000	45000
2020	715	85000	135000	15000	3217	8351.8	180000	20000
2021	715	85000	135000	15000	3217	8351.8	90000	10000
2022	715	85000	135000	15000	3217	8351.8	270000	30000
2023	715	85000	135000	15000	3217	8351.8	90000	10000
2024	715	85000	135000	15000	3217	8351.8	270000	30000
2025	715	85000	180000	20000	3217	8351.8	180000	20000
2026	715	85000	180000	20000	3217	8351.8	90000	10000
2027	715	85000	180000	20000	3217	8351.8	270000	30000
2028	715	85000	180000	20000	3217	8351.8	90000	10000
2029	715	85000	180000	20000	3217	8351.8	270000	30000
2030	715	85000	157500	17500	3217	8351.8	180000	20000
2031	715	85000	157500	17500	3217	8351.8	90000	10000
2032	715	85000	157500	17500	3217	8351.8	270000	30000
2033	715	85000	157500	17500	3217	8351.8	90000	10000
2034	715	85000	157500	17500	3217	8351.8	270000	30000
2035	715	85000	157500	17500	3217	8351.8	135000	15000
2036	715	85000	157500	17500	3217	8351.8	67500	7500
2037	715	85000	157500	17500	3217	8351.8	202500	22500
2038	715	85000	157500	17500	3217	8351.8	67500	7500
2039	715	85000	157500	17500	3217	8351.8	202500	22500
2040	715	85000	135000	15000	3217	8351.8	135000	15000
2041	715	85000	135000	15000	3217	8351.8	67500	7500
2042	715	85000	135000	15000	3217	8351.8	202500	22500

	Constant harvest						Variable harvest	
Year	arctic	greenland	canpup	can1+	by1plus	bypup	canpup	can1+
2043	715	85000	135000	15000	3217	8351.8	67500	7500
2044	715	85000	135000	15000	3217	8351.8	202500	22500
2045	715	85000	135000	15000	3217	8351.8	135000	15000
2046	715	85000	135000	15000	3217	8351.8	67500	7500
2047	715	85000	135000	15000	3217	8351.8	202500	22500
2048	715	85000	135000	15000	3217	8351.8	67500	7500
2049	715	85000	135000	15000	3217	8351.8	202500	22500
2050	715	85000	108000	12000	3217	8351.8	112500	12500
2051	715	85000	108000	12000	3217	8351.8	55800	6200
2052	715	85000	108000	12000	3217	8351.8	169200	18800
2053	715	85000	108000	12000	3217	8351.8	55800	6200
2054	715	85000	108000	12000	3217	8351.8	169200	18800
2055	715	85000	112500	12500	3217	8351.8	117000	13000
2056	715	85000	112500	12500	3217	8351.8	58500	6500
2057	715	85000	112500	12500	3217	8351.8	175500	19500
2058	715	85000	112500	12500	3217	8351.8	58500	6500
2059	715	85000	112500	12500	3217	8351.8	175500	19500