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The role of predation by harp seals (*Pagophilus groenlandicus*) in the collapse and non-recovery of northern Gulf of St. Lawrence cod (*Gadus morhua*)

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1 ABSTRACT: A statistical catch-at-age model was developed to assess the effects of predation
2 by the northwest Atlantic harp seal population on northern Gulf of St. Lawrence cod by
3 estimating the relative importance of different sources of mortality that affected the stock
4 during a period of collapse and non-recovery. Cod recruitment at age 1 is modeled via a non-
5 linear stock-recruitment relationship based on total egg production and accounts for changes in
6 female length-at-maturity and cod condition. Natural mortality other than seal predation also
7 depends on cod condition used as an integrative index of changes in environmental conditions.
8 The linkage between seals and cod is modeled through a multi-age functional response that was
9 derived from the reconstruction of the seal diet using morphometric relationships and stomach
10 contents of more than 200 seals collected between 1998 and 2001. The model was fitted
11 following a maximum likelihood estimation approach to a scientific survey abundance index
12 (1984-2006). Model results show that the collapse of the northern Gulf of St. Lawrence cod
13 stock was mainly due to the combination of high fishing mortality rates and poor environmental
14 conditions in the early to mid-1990s contributing to the current state of recruitment overfishing.
15 The increase in harp seal abundance during 1984-2006 was reflected by an increase in predation
16 mortality for the young cod age-groups targeted by seals. Although current levels of predation
17 mortality affect cod spawning biomass, the lack of recovery of the NGSL cod stock seems
18 mainly due to the situation of very poor recruitment.

19 *Keywords : cod, harp seal, functional response, model, predation, recovery*

INTRODUCTION

21 During the early 1990s, there was an almost simultaneous collapse among most of the Atlantic
22 cod (*Gadus morhua* L.) fisheries in eastern Canada, leading to severe restrictions or moratoria
23 on commercial fishing (Myers et al. 1997, Rice et al. 2003). The northern Gulf of St. Lawrence
24 (NGSL) cod stock (NAFO divisions 3Pn4RS; Fig. 1) was historically the second largest cod
25 population in the Western Atlantic with a spawning stock biomass (SSB) of 380,000 t in the
26 early 1980s, supporting a fishery of more than 100,000 t in 1983 (MPO 2007). By the late
27 1980s, the population and fishery declined to such an extent that a moratorium was imposed
28 in 1994, leaving a SSB of only about 10,000 t. Subsequently, the population increased quite
29 rapidly and in 1997, a small fishery reopened, although the stock has remained at about 50%
30 of its minimum conservation biomass limit (B_{lim}) of about 80-90,000 t.

31 There has been considerable debate concerning the causes for the decline in cod abundance
32 and subsequent lack of recovery (Hutchings 1996, Shelton & Healey 1999, Shelton et al. 2006).
33 Overfishing has been identified as the main factor that contributed to the collapse due to several
34 interrelated common factors including underestimated discarding and misreporting (Myers et
35 al. 1996, Myers et al. 1997, Savenkoff et al. 2004). For the NGSL cod stock, it has been
36 suggested that changes in environmental conditions in the early 1990s also played a role in the
37 fishery collapse through a decline in condition, which consequently resulted in reduced growth
38 and recruitment (Lambert & Dutil 1997a, Dutil et al. 1999, Dutil & Lambert 2000). Poor
39 condition is also suspected to have led to an increase in natural mortality through starvation
40 and enhanced vulnerability to predation (Dutil et al. 1999). The current poor productivity
41 of the stock associated with ongoing fishing activity has been proposed as a major factor
42 explaining the lack of recovery for NGSL cod (Dutil et al. 2003, Shelton et al. 2006).

43 During the last 3 decades, there has been a marked increase in abundance of many pinniped
44 populations in the northwest Atlantic (Hammill et al. 1998, Hammill & Stenson 2005). Higher
45 natural mortality resulting from increased predation has also been put forward as a plausible
46 hypothesis explaining the collapse and failure of northwest Atlantic groundfish populations

47 to recover (Bundy 2001, DFO 2003, Rice et al. 2003). In particular, harp seals (*Pagophilus*
48 *groenlandicus*), the most abundant pinniped in the northwest Atlantic, have increased from
49 less than 2 million in the early 1970s to almost 6 million today (Hammill & Stenson 2005).
50 Harp seals summer in the Arctic, but migrate south along the Labrador coast in late autumn
51 to overwinter off northeastern Newfoundland and in the NGSL (Fig. 2). The majority of
52 the seals overwinter off the southeastern Labrador-northeastern Newfoundland coast (NAFO
53 Divisions 2J3KL), while approximately 25-33% of the harp seals move into the NGSL (Stenson
54 et al. 2002, 2003). Harp seals are generalist predators characterized by a diverse diet of fish
55 and invertebrates (e.g. Lawson et al. 1998, Hammill & Stenson 2000). Stomach samples
56 collected during the winter and spring in the NGSL indicate that their diet is highly variable
57 both seasonally and geographically with average of approximately 5% made up by cod, mainly
58 in the size range of 15-25 cm, although in some samples this proportion may be as high as 45%
59 (Hammill & Stenson 2004, Stenson & Hammill 2004).

60 The correspondence between the decline of NGSL cod and the large increase in harp seal
61 abundance (Fig. 3) has led to a hypothesis that harp seal predation may play a significant role
62 in the cod lack of recovery (McLaren et al. 2001, DFO 2003). Seal predation on cod has been
63 examined using bioenergetic models that derive cod annual consumption estimates by taking
64 into account seasonal changes in feeding and variability in seal abundance, distribution and
65 diet composition (Stenson et al 1997, Hammill & Stenson 2000, Stenson & Hammill 2004).
66 However, because of the variable nature of the diets, these models have assumed an average
67 proportion of cod in the diet, in spite of the marked changes in cod biomass that have been
68 documented over the last 3 decades (MPO 2007). Assumptions about how predation changes in
69 response to variations in prey abundance have been shown to be critical in predicting impacts
70 of predators on commercially important prey (Yodzis 1994, Mackinson et al. 2003). However,
71 individual and population level phenomena among predators and prey alike, such as search
72 image, difficulty in finding prey, aggregation, and dispersion of predator and prey, can lead
73 to significant departures from the linear assumptions about the relationship between prey
74 abundance and the proportion of prey in the diet (Koen-Alonso 2006). In addition, harp seals
75 can display strong preference for specific prey, although they have been shown to be neutrally

76 selective toward Atlantic cod (Lawson et al. 1998, Lindstrøm et al. 1998, Wathne et al. 2000).
77 Modeling non-linear processes of the functional response of harp seals to cod appears therefore
78 to be a key to modeling predation over a wide range of prey abundance such as observed in
79 NGSL cod.

80 As described above, hypotheses related to fishing, seal predation, and environment have been
81 proposed for the decline of NGSL cod but few studies have attempted to simultaneously address
82 these hypotheses to determine which processes were at play at different times and how syner-
83 gistic forcing could have led to the cod collapse. Here, a predator-prey model was constructed
84 for NGSL cod, with mortality partitioned into components caused by harp seal predation and
85 by other processes including an explicit influence of an integrated environmental forcer. The
86 impact of seals on cod was modelled through a non-linear multi-age functional response similar
87 to a multi-species functional response (Smout & Lindstrøm 2007). Cod condition was used as
88 an index for the cumulative and lagged effects of environmental conditions related to factors
89 such as food availability and temperature experienced by cod (Lambert & Dutil 1997a-b, Rätz
90 & Lloret 2003). Changes in cod condition were assumed to affect natural mortality (Dutil
91 & Lambert 2000) and fecundity, i.e. potential recruitment strength (Marteinsdottir & Begg,
92 2002). The main objective of the analysis was to determine the relative impact of fishing, seal
93 predation, and environment on the decline of NGSL cod by integrating these processes into a
94 single consistent modeling framework. Such a framework is essential to both understand what
95 happened to the stock and serve as a springboard for determining future states in the context
96 of cod recovery and providing advice on exploitation of NGSL harp seal and cod populations.

97 **COD POPULATION MODEL**

98 Cod population dynamics were modelled through a separable statistical catch-at-age analysis
99 (Fournier & Archibald 1982) without process error, where mortality was derived from three
100 sources; the fishery, harp seal predation, and natural mortality due to causes other than harp
101 seal predation (other predation, disease, virus, etc.). The Seal IMPact on Cod ABundance
102 (SIMCAB) model considers 13 age classes of cod from 1 to 13 and 100% mortality occurs at

103 end of the 13th age class for all cohorts. Following Pope’s approximation (1972), mortality
104 processes were modelled as successive steps, i.e. removals of cod by harp seal predation were
105 taken instantaneously at the beginning of the year, fishing removals were taken instantaneously
106 in the middle of the year, and residual mortality occurred between the 2 portions of the
107 year (Table 2, D9-D11). The NGSL cod fishery changed considerably in the last decades,
108 particularly after the 1994 moratorium when the winter mobile fishery dominated by otter
109 trawlers was closed (Fréchet et al. 2003a). To account for changes in fishing pattern, 2
110 periods of distinct fishery selectivity (partial recruitment) were considered: 1984-1993 and
111 1994-2006. For consistency with knowledge available on NGSL cod dynamics and because
112 of the convergence property of the virtual population analysis (VPA), the initial population
113 age-structure was based on cod numbers estimated in 1984 from VPA (MPO 2007) and the
114 model was run for the period 1984-2006. The parameters and variables used and the process
115 and observation equations are given in Tables 1 and 2 respectively.

116 **A condition-dependent reproduction function**

117 Recruitment, i.e. fish numbers at age 1, was modelled through a Beverton and Holt (1957)
118 stock-recruitment relationship based on an index of the reproductive potential of cod that
119 was developed from an estimate of total egg production of the stock (TEP) (Table 2, D3).
120 TEP accounted for yearly changes in cod numbers-at-age, sex ratio-at-age, the proportion of
121 mature females-at-age, fecundity-at-age through length-at-age and condition factor of the fish.
122 Previous studies have shown that length and condition factor have significant effects on the
123 fecundity of cod in the NGSL (Lambert & Dutil 2000, Lambert et al. 2000). This dependence
124 of fecundity on length and condition factor was used to develop a generalized linear model to
125 determine the variation in the fecundity between 1984 and 2006 (Lambert 2008).

126 Sex ratio and maturity ogive data were derived from winter surveys conducted with the MV
127 “Gadus Atlantic” (1984-1994) and spring samples available from the Groundfish Sentinel Fish-
128 eries Program (<http://www.osl.gc.ca/pse/en/>). Fulton’s condition factor (Fulton 1902) based
129 on length and weight data was also obtained from the winter surveys (1984-1994), the Sen-

130 tinal Fisheries Program (1995-2007), and supplementary research surveys conducted during
131 the pre-spawning period in 1994, 1995, 1997, and 1998 (Dutil et al. 2003, Fréchet et al. 2007).

132 **A multi-age functional response of harp seals to cod**

133 To account for effects of prey abundance on predation, assuming no predator interference in
134 the predation process, a multi-age functional response (FR), i.e. the number of cod consumed
135 per seal per year, was considered (Table 2, D4). The FR of seal to cod is comprised of a
136 maximum consumption rate for cod (J_{max}), attack rates-at-age (ζ_a), and age-structured cod
137 population. The shape parameter m of the FR determines the FR type, i.e. $m = 1$ corresponds
138 to a hyperbolic type 2 relationship and $m > 1$ corresponds to a sigmoid shaped type 3 response
139 (Koen-Alonso 2006).

140 The attack rates (ζ_a) were expressed relative to baseline attack rates (ζ_a^0) through a scaling
141 factor (ρ) that was estimated in the fitting process (see section 'Fitting the model'). Baseline
142 attack rates were derived from cod biomass requirements for the NGSL harp seal population,
143 cod abundance based on the matrix of numbers-at-age derived from virtual population analysis
144 (MPO 2007), that was extended to ages 1-2 based on preliminary runs of the model without
145 predation, and proportion and age-structure of cod in the seal diet in 1998-2001.

146 First, harp seal energetic requirements to maintain the seal population were estimated based
147 on the daily gross energy intake of an individual harp seal derived from the allometric Kleiber
148 equation that scales animal's metabolic rate to the 3/4 power of the animal's mass, and the
149 age-structure of the seal population (Hammill & Stenson 2004, Stenson & Hammill 2004). 25%
150 of the northwest harp seal population was assumed to move into the Gulf of St. Lawrence each
151 year to spend on average 150 days per year in its northern part (Stenson et al. 2002, 2003).
152 The biomass of cod consumed in 1998-2001 was then estimated based on the average energy
153 of cod in winter, set equal to 4.96 kJ g^{-1} (Lambert & Dutil 1997a), and on the annual average
154 proportion of cod in seals diet during November-March 1998-2001 (Table 3). The relative
155 energetic contribution of cod to seal's diet was based on recovered otoliths and accounted for

156 harp seal population residency in the areas 4Ra-c, 4Rd-3Pn, and 4S of the NGSL (Fig. 2;
157 Hammill & Stenson 2004). The annual number of cod consumed was then estimated based on
158 the mean mass of cod consumed by harp seals and reallocated between cod age-groups based on
159 information available from the analysis of seal stomach contents (Lawson et al 1998; Hammill
160 & Stenson 2004). Lengths of cod ingested were estimated from otoliths found in seal stomachs
161 and published fish length - otolith length regression equations (Hammill & Stenson 2004). Fish
162 lengths were based on 671 otoliths found in 140 seal stomachs collected during November to
163 March, 1989-2001 in the Gulf of St. Lawrence (Chassot et al. 2007). Harp seal size ranged
164 from 102-166 cm. Otoliths collected from stomach contents were assumed to represent the size
165 distribution of cod eaten. Proportions were estimated by fitting a mixture model of normal
166 distributions to the prey size-frequency histogram by applying a constraint on the variance of
167 cod size derived from DFO summer survey (Chassot et al. 2007).

168 The maximum consumption rate J_{max} was calculated based on the energy requirements of
169 the NGSL harp seal population in 1998-2001 and a maximum value of cod proportion in seal
170 stomachs derived from diet information available. Following the same approach as described
171 above, fish biomass required to maintain the NGSL seal population was first calculated. Based
172 on a maximum proportion of cod of 18.3% derived from the maximum proportions of cod in
173 seals diet observed since 1980 and the relative residency time in each area of the NGSL, the
174 potential maximum number of cod eaten per seal per year J_{max} was calculated. The baseline
175 attack rates-at-age were finally calculated from the number of cod-at-age consumed per seal
176 and cod numbers-at-age derived from the VPA in 1998-2001 (Table 2, D4). Since the baseline
177 attack rates were based on a number of cod eaten calculated from the mean weight of cod in
178 the reference period 1998-2001, the mass ratio $\left(\frac{\overline{W}_t}{W_0}\right)$, an index of cod weight, was included
179 to account for yearly fluctuations in the mean mass of cod when calculating the number of cod
180 predated by seals (Table 2, D5).

181 In order to illustrate the multi-age functional feeding response to changes in cod availability,
182 a large range of levels of cod abundance were simulated for 2 distinct age-structures of the
183 cod population. For each age-structure, cod-at-age consumed per seal per year were calculated

184 based on the model fitting results.

185 **Removals by natural mortality other than harp seal predation**

186 The rate of natural mortality excluding harp seal predation, i.e. residual mortality was mod-
187 elled as an age-dependent function, in the form of a decreasing exponential as a function of
188 age with a condition-dependent asymptote (Table 2, D6). The asymptote was modelled as a
189 decreasing linear function of cod condition based on results of laboratory experiments relating
190 natural mortality to cod condition (Dutil & Lambert 2000, Chassot et al. 2007). The condition
191 factor which is an indicator of the energy content of cod was used here as an integrative index
192 of environmental conditions (Lambert & Dutil 1997b). Alternative values for the asymptote
193 of the residual mortality curve, i.e. a constant value of 0.4 y^{-1} and a time-dependent natural
194 mortality derived from stock assessment models (MPO 2007), were considered to assess the
195 impact on the results through the Akaike Information Criterion (AIC; Akaike 1974).

196 **Effects of seal predation vs. fishing on the reproductive potential**

197 Because seal predation and fishing are mortality sources for different cod age groups, a unit of
198 mortality from each does not have the same impact on the population. Similarly to a multi-
199 fleet biomass-per-recruit analysis, effects of changes in harp seal predation vs. fishing mortality
200 were investigated by calculating the spawning stock biomass per recruit (SSB/R) for different
201 levels of mortality. Such an analysis focuses on the long-term survival of a recruit and does
202 not consider the linkage between the reproductive potential and recruitment at age 1. Based
203 on the mortality rates estimated and averaged over the period 2002-2006 and assuming all else
204 constant, we examined the relative importance of seal predation mortality and fishing mor-
205 tality on the long-term stock reproductive potential under two scenarios: a high productivity
206 environment and a low productivity environment. Considering a range of fishing mortality
207 multipliers from 0 to 2, cod SSB/R was calculated at equilibrium for distinct levels condition
208 because condition was assumed to affect the asymptote ϑ of the residual natural mortality in

209 the model. 3 levels of seal predation mortality were considered corresponding to 0.5, 1, and
210 1.5 times the average harp seal predation mortality estimated in 2002-2006. 2 scenarios of
211 conditions represented different environmental conditions for the cod population, with values
212 of $\vartheta = 0.26$ and $\vartheta = 0.35$ corresponding to a high and a low productivity regime scenario,
213 respectively.

214 **FITTING THE MODEL**

215 **Abundance and catch data**

216 Stratified random bottom-trawl surveys have been conducted in the NGSL annually, in summer
217 months, since 1984. A consistent time-series of numbers-at-age, accounting for the changes in
218 research vessels, was used as abundance indices for cod for the period 1984-2006 (Bourdages
219 et al. 2003, 2007, Chassot et al. 2007).

220 The catch-at-age matrix of cod (in numbers) was obtained from the last updated stock as-
221 sessment carried out in February 2007 (MPO 2007). All quantities caught and landed for
222 both commercial and recreational fisheries, excluding discards, were included in the data. In
223 the present analysis, reported landings were considered as the “true” catch despite potential
224 discarding and misreporting in the fishery (Fréchet et al. 2003b, Savenkoff et al. 2004).

225 **Statistical estimation approach**

226 Maximum likelihood methods were used to estimate SIMCAB parameters. The likelihood
227 function of the model was derived from assumptions on the statistical distribution of the 4
228 datasets acquired from independent sampling processes: total abundance index in number,
229 proportion-at-age for the abundance indices, total catch in number, and proportion of catch-
230 at-age (Fournier & Archibald 1982, Quinn & Deriso 1999) (Table 5).

231 A stochastic observation equation accounting for correlated errors among ages was considered
232 to link the abundance indices to the population model (Myers & Cadigan 1995) (Table 5, S1).
233 For some years, some abundance indices-at-age for very young (age 1) or old cod (ages 11-13)
234 were not observed during the surveys. For these years, total survey indices were considered as
235 right-censored data, providing less information to the likelihood component than real observa-
236 tions (Lawless 2003). This is true for the sum over ages of the logarithms of the survey indices
237 since these indices are extrapolated to the scale of the NGSL and are positive numbers greater
238 than 1. The observed likelihood of total surveys indices was then composed of a product of
239 densities and survival functions, where a survival function is defined as 1 minus the distribu-
240 tion function (Table 6, L1). The survival factor in the likelihood represents the integration,
241 over all its possible values, of the probability that a random total survey index is higher than
242 an underreported observation. It is a common way of including partial data information in a
243 statistical analysis, but rarely used in fisheries analysis (for a notable exception see Hammond
244 and Trenkel 2005).

245 Proportion-at-age for the abundance indices and catch data were assumed to follow dirichlet
246 distributions to account for correlations among age groups (Fournier & Archibald 1982) (Table
247 5, S2 and S3). Total catch observed were considered mainly dependent on the fishing year
248 and assumed measured with lognormal observation errors (Table 5, S4). The assumption of
249 log-normal error distributions for survey and catch data was checked through the residuals.

250 The maximum likelihood estimates were found by minimizing the negative sum of the log of
251 the 4 likelihood components (i.e. objective function) given in Table 6. Minimizations were
252 performed with the Nelder-Mead simplex non-linear optimization algorithm implemented in
253 the R package (R Development Core Team 2008). The parameter space θ to estimate included
254 37 parameters from 644 observations, an observation being a survey index or catch number
255 for a given age group in a given year: 2 for the TEP-recruitment relationship (R_{max} and
256 r), 2 for the FR (m and ϱ), 2 for the residual mortality curve (α and β), 1 for the survey
257 catchability (q), 6 for the selectivity of survey and fishing gears ($\gamma_s, \delta_s, \gamma_c^1, \delta_c^1, \gamma_c^2, \delta_c^2$), 23 for
258 the fishing mortality rates (F_t), and 1 for the standard deviation of the observation error in

259 the total survey data (ψ) (see Table 1 for definition). The maximum likelihood estimate of the
260 observation error variance in the total catch data (σ_c^2) was:

$$\sigma_c^2 = 2 \left(\sqrt{1 + \frac{1}{T} \sum_{t=1}^T \left(\log \frac{C_t^*}{C_t} \right)^2} - 1 \right) \quad (1)$$

261 Where the notations used are given in table 4.

262 Confidence statements about parameters were inferred using parametric bootstrap methods
263 accounting for bias (Efron & Tibshirani 1998). 100 bootstrap replicate samples with the same
264 dimensions as the datasets described above were generated by drawing from the probability
265 distributions characterised by the parameters estimated at the maximum likelihood. Replicate
266 samples were then submitted to the estimation procedure to obtain bootstrap replicates of the
267 estimator and calculate percentiles of the distribution.

268 **Sensitivity analysis**

269 To complete the bootstrap analysis that only considered uncertainty in the catch and survey
270 data, a sensitivity analysis was performed to account for uncertainty in some input parameters
271 of the predation model and harp seal population in the NGSL (Hammill & Stenson 2004,
272 Stenson & Hammill 2004). In a first step, the sensitivity of the results was examined by
273 independently increasing parameter values by 10% from a baseline value (values from the
274 standard run), re-fitting the model to observations for each sensitivity run, and assessing the
275 change between the new and baseline results through a measure of relative sensitivity (Stenson
276 & Hammill 2004, Mohn & Bowen 1996). The sensitivity to 5 parameters was investigated:
277 the proportion of the northwest Atlantic population of harp seals moving into the NGSL, the
278 average proportion of cod in seal's diet in 1998-2001, the maximum proportion of cod in seal's
279 diet, the average energy of cod, and the Kleiber multiplier used as a constant multiplicative
280 parameter in the Kleiber equation (Table 7). In order to summarize the model results and
281 because predation by harp seals was the main focus of the analysis, only results for mean cod

282 biomass removed by seals and mean predation mortality rate for ages 1-4 during 1984-2006
283 were presented. In a second step, the impacts of setting the value of the functional response
284 type m to 1, i.e. a hyperbolic type 2 relationship, were examined.

285 **RESULTS**

286 **Age-structure of cod in seal's diet**

287 The size frequency histogram showed that 95% of cod found in harp seal stomachs were less than
288 38 cm (Fig. 4). The mixture model fitted well the cod size-frequency data as indicated by the
289 significance level for the goodness-of-fit test ($p > 0.05$), based on the chi-square approximation
290 to the likelihood ratio statistic (Du 2002). The model showed that cod consumed were young
291 cod, with ages 1-4 representing 26.5%, 48.5%, 18%, and 7% of the cod in the diet, respectively
292 (Fig. 4).

293 **Cod population dynamics**

294 The model fitted the survey and catch data well and the minimum value for the objective
295 function was 6,149.6, corresponding to an AIC value of 12,373 (Figs. 5-6). The variances of
296 the observation errors in the survey and catch data were low, the values of ψ and σ_c used as
297 proxies of coefficients of variation in the case of lognormal likelihoods being 5.0% and 2.7%,
298 respectively. Consequently, confidence intervals around parameters estimated by bootstrap
299 analysis were small (Table 1). Considering a constant asymptote of 0.4 y^{-1} and a variable
300 asymptote for the residual mortality curve led to higher values of AIC (12,583 and 12,696
301 respectively) than for a condition-dependent asymptote, indicating a better fit of the model
302 when residual mortality was related to cod condition.

303 SIMCAB estimates indicated a strong decline in cod abundance for all age-groups from 1984
304 to 1993, consistent with the collapse of the northern Gulf cod stock in the mid-1990s (Fig. 5).

305 Abundance indices for ages 4-8 then remained quite stable from the mid-1990s to the early
306 2000s, the stock showing no clear sign of recovery despite the moratoria on fishing implemented
307 in 1994-1996 and 2003 (Fig. 5). By contrast, abundance indices for cod aged 10-13 displayed
308 increasing trends from the mid-1990s to the 2000s; these age-groups representing a very small
309 proportion of the stock abundance but about 10% of the SSB on average for the period 1984-
310 2006 (Fig. 5). The declines in abundance were accompanied by a decrease in SSB from more
311 than 330,000 t in the early 1980s to a minimum of less than 30,000 t in 1994 and showed a
312 small increase thereafter. SSB estimates were thus below the conservation biomass limit of
313 80-90,000 t and indicated a clear case of recruitment overfishing for the stock.

314 The number of cod caught showed a strong decrease from the 1980s to the mid-1990s for ages
315 5-10 that represent the large majority of the biomass harvested (Fig. 6). SIMCAB predictions
316 appeared bumpy in the 1980s with catches underestimated in 1986 and 1988 for all age-groups
317 (Fig. 6). This mismatch between catch observed and predicted seems mostly due to the
318 exceptional and somewhat surprising abundance indices for ages 4-10 in 1987 (Fig. 5).

319 **Mortality components in different age-groups**

320 Total mortality divided into age-groups showed that sources and magnitude of mortality varied
321 in time and age (Fig. 7). Young cod aged 1-2 experienced high natural mortality, varying
322 between about 1.9 y^{-1} in the mid-1980s to more than 2.4 y^{-1} in the mid-1990s (Fig. 7a).
323 Predation mortality by harp seals only represented a low proportion of mortality for ages 1-2
324 throughout the period 1984-2006 but increased from about 6% in the 1980s to more than 12%
325 in the mid-2000s.

326 In addition to seal predation and residual natural mortality, cod aged 3-6 were subject to
327 fishing mortality, mainly in the 1980s (Fig. 7b). The decrease in fishing mortality from the
328 mid-1990s coincided with the change in fishing pattern, i.e. the commercial fishery after 1994
329 was only conducted by fixed gears (longlines, gill nets, and hand lines) targeting larger cod.
330 Residual natural mortality showed an increase from the 1980s to the mid-1990s associated with

331 lower condition during this period; the rates in the most recent years were similar to the levels
332 estimated in the mid-1980s. Seal predation mortality for ages 3-6 remained quite stable during
333 1984-2006 with its proportion in total mortality increasing to more than 20% in the 2000s due
334 to the decrease in fishing and residual natural mortality.

335 Cod aged 7-10 not preyed upon by harp seals showed high total mortality, particularly in the
336 mid-1980s and early 1990s with rates higher than 1.0 y^{-1} (Fig. 7c). The high interannual
337 variability in mortality rates was mainly due to the strong variations in fishing mortality that
338 represented more than 65% of the total mortality in some years (Fig. 7c). The low fishing
339 mortality values estimated in 1986 and 1988 could be due to the high abundance indices
340 observed in 1987 (Fig. 5). As for ages 3-6, residual natural mortality was the highest in the
341 mid-1990s, corresponding to a low-condition period for cod. Although fishing mortality rates
342 showed a decreasing trend in the 2000s, the values remained high for some years relative to
343 the low abundance of the stock, particularly in 2005.

344 **Harp seal predation**

345 Predation mortality rates for cod aged 1-4 showed a steady increase alongside the increase
346 in harp seal abundance from 0.1 y^{-1} in 1984 to more than 0.3 y^{-1} in 2000 (Fig. 8). The
347 rates then decreased to remain quite stable around 0.25 y^{-1} in the most recent years. Cod
348 biomass removed annually by seal predation was estimated to be around 10,000 t from 1985
349 to 1995. Removals then showed an increasing trend with a peak above 30,000 t in 2000 and
350 a temporally averaged median of 16,000 t in the 2000s. The high interannual variability in
351 biomass removed was mainly due to the variations in abundance of cod aged 1-2 that represent
352 the major age-groups preyed upon by harp seals.

353 The exponent of the multi-age functional response fitted to the data was estimated to be 2.13
354 (± 0.015), indicating a sigmoid shaped type 3 response characteristic of sharp changes in the
355 relative importance of the cod age-groups consumed as the result of a relatively small change in
356 their availability in the environment. The changes in biomass removed by seal predation were

357 then related to both changes in seal population energy requirements and changes in relative
358 cod abundance between age-groups. For instance, the predicted multi-age functional response
359 showed that cod age 2 would be the main target of harp seals when considering different levels
360 of cod abundance described by the 1984 population age-structure (Fig. 9). Considering an
361 age-structure similar to low abundance year of 1995, characterized by a high proportion of
362 cod aged 1 in the population, would lead to a very different response of seals with age-1 cod
363 becoming the major prey and the other age-groups quickly reaching their respective maximum
364 in seal consumption (Fig. 9).

365 **Sensitivity analysis**

366 The sensitivity analysis showed that the results of the model concerning harp seal predation
367 were robust to the assumptions made about the average energy of cod and the values of
368 contribution of cod to seal's diet (Table 7). Increasing the contribution of cod to seal's diet
369 in 1998-2001 did not affect the total biomass removed but led a relatively small increase in
370 predation mortality rates by decreasing the cod population. Although the increase in maximum
371 cod consumption led to an increase in the maximum rate of cod consumption J_{max} , it poorly
372 affected harp seal predation. Predation appeared quite sensitive to a change in the size of
373 the seal population and the Kleiber multiplier of the bioenergetic model (Table 7). Changes
374 in input parameters could be counter-intuitive due to the complexity and non-linearity of
375 the model; hence, the biomass removed by predation during 1984-2006 was decreased by an
376 increase in average energy of cod while it led to an increase in the predation mortality rates.
377 The relative sensitivity of predation rates could have a different sign than the sensitivity of
378 biomass removed since predation rates depend both on the cod biomass removed and cod
379 numbers-at-age.

380 Constraining the predation model to follow a hyperbolic type 2 relationship strongly affected
381 the results of the model and resulted in a lower quality of fit than for the standard run (AIC
382 = 12,697). Setting the shape parameter of the functional feeding response to 1 led to reduced
383 harp seal predation and increased cod numbers-at-age and spawning stock biomass.

384 **Effects of seal predation vs. fishing on the reproductive potential**

385 The SSB/R curves showed that the reproductive potential of the NGS cod was affected dif-
386 ferently by changes in seal predation and fishing mortality (Fig. 10). The distinct productivity
387 regimes represented by 2 levels of residual mortality modified the values of SSB/R, with low
388 productivity leading to lower values of SSB. In both cases of productivity regimes, the nearly
389 horizontal curves indicated that changes in SSB/R would be poorly modified by changes in fish-
390 ing mortality while variations in seal predation mortality would have a higher effect on SSB/R
391 (Fig. 10). This result was mainly due to the low fishing mortality rate of 0.15 y^{-1} estimated
392 during 2002-2006. Considering the harp seal predation mortality estimated in 2002-2006 and a
393 low residual mortality (i.e. a high productivity regime), the SSB/R was estimated to be equal
394 to 129 g (Fig. 10 - intermediate dashed line curve). In this case and for a fishing mortality
395 fixed at the level of 2000-2006, a decrease in the predation mortality rate from the mean value
396 of 0.26 y^{-1} in 2000-2006 to a level around 0.13 y^{-1} observed in the mid-1980s would allow
397 almost doubling the SSB/R (Fig. 10 - upper dashed line curve). Changes in fishing mortality
398 assuming constant predation mortality would be less beneficial to the SSB, a decrease from
399 the fishing mortality rate of 0.15 y^{-1} in 2002-2006 to a low rate of 0.03 y^{-1} increasing the
400 SSB from 129 g to 171 g per recruit. Considering a low productivity regime strongly decreased
401 the SSB/R that was estimated to be 73 g in 2000-2006, due to the higher residual mortality
402 decreasing the probability of young cod to reach maturity (Fig. 10 - intermediate solid line
403 curve). In this case, the SSB/R for a predation mortality of 0.13 y^{-1} and constant fishing
404 mortality would only be 131 g (Fig. 10 - upper solid line curve). For the 2002-2006 predation
405 mortality, a decrease in fishing mortality would slowly increase the SSB/R to a maximum of
406 97 g in the situation of a moratorium.

407

DISCUSSION

408 The SIMCAB statistical catch-at-age model was developed to assess the effects of predation
409 by the northwest Atlantic harp seal population on NGS cod by estimating the relative im-

410 portance of different sources of mortality that affected the stock during a period of collapse
411 and non-recovery. SIMCAB is a consistent modeling framework which combines the effects of
412 fishing, predation mortality and environment (through the condition index) on cod population
413 dynamics. SIMCAB showed that the increase in harp seal abundance during 1984-2006 re-
414 sulted in an increase in predation mortality for cod age-groups targeted by seals. Despite the
415 increasing importance of harp seal predation in cod mortality, the collapse of the NGS� cod
416 stock appears mainly due to the combination of high fishing mortality rates and poor environ-
417 mental conditions that affected the productivity of the population in the early to mid-1990s
418 and led to the current state of recruitment overfishing. SIMCAB results also show, however,
419 that current levels of predation mortality could increase stock rebuilding time by decreasing the
420 probability of fish to reach maturity. Although cod condition has improved in recent years, the
421 current lack of recovery of the NGS� cod seems due mainly to the very low spawner biomass,
422 driven both by the fishery inherently targeting larger fish and increased harp seal predation.
423 Within an ecosystem perspective, management scenarios based on a decline in seal population
424 to promote stock rebuilding should however consider environmental conditions, prey availabil-
425 ity for seals and other cod predators of the NGS� that all affect cod productivity and might
426 give rise to unexpected outcomes with a decline in seal abundance.

427 **Multispecies modeling and functional response**

428 Separating the sources of mortality affecting animal populations is a major issue in ecology
429 and has a long history in fisheries science (Andersen & Ursin 1977). This is particularly
430 important in the context of multispecies management as predator control has been proposed
431 for some time as a beneficial tool for fisheries (Flaaten 1988, FRCC, 1999). Multispecies
432 models are useful tools to address the issues raised about the effects of marine mammals on
433 fisheries (Yodzis 1998). In particular, predator-prey models have been used to evaluate the
434 effects of fur seal (*Arctocephalus pusillus pusillus*) predation on hakes (*Merluccius capensis* and
435 *Merluccius paradoxus*) in the Benguela system (Punt & Butterworth 1995) and more recently
436 the effects of northwest Atlantic grey seal (*Halichoerus grypus*) predation on cod population
437 dynamics and causes of collapse and non recovery on the eastern Scotian shelf (ESS) have been

438 explored (Mohn & Bowen 1996, Fu et al. 2001, Trzcinski et al. 2006). In the Benguela system,
439 modeling the biological interaction between cape fur seals and the cape hakes suggested that
440 an initial reduction in seal numbers would be beneficial, but the resulting decrease in seal
441 consumption would also result in increased predation by a less commercially important hake
442 species *M. capensis* on the more commercially important hake *M. paradoxus*, resulting in little
443 net benefit (Punt and Butterworth 1995). Among northwest Atlantic grey seals, the first
444 modeling approach included predation mortality explicitly in cohort analysis and showed that
445 grey seals had little effect on the collapse of cod on the ESS (Mohn & Bowen 1996). By
446 contrast, more recent analyses suggested that increasing predation mortality since the 1990s
447 could affect the survival of immature cod and contribute to the failure of the ESS and southern
448 Gulf of St. Lawrence cod stocks to recover (Fu et al. 2001, Chouinard et al. 2005, Trzcinski
449 et al. 2006).

450 The SIMCAB model is an age-structured population dynamics model structurally similar to
451 the models used for grey seals on the ESS but it differs in several important points. A major
452 difference with the models from Mohn & Bowen (1996) and Fu et al. (2001) but addressed
453 by Trzcinski et al. (2006) concerns the linkage between cod abundance and seal consumption
454 through the functional feeding response. The form of the FR is a key issue in multispecies
455 modeling, especially at low abundance, because it defines how predators impact their prey as
456 a function of prey abundance (Yodzis 1994, Mackinson et al. 2003). Most multispecies models
457 such as the Multi-Species Virtual Population Analysis (MSVPA; Magnússon 1995) are based
458 on hyperbolic type 2 functional relationships between prey abundance and predation rates
459 (for a review see Pláganyi 2007). These models assume constant ration formulations where
460 per-capita consumption is set equal to the predator's required daily ration, consistent with
461 the interpretation that feeding selectivities are independent of prey abundance. A sigmoidal
462 functional response is however likely more appropriate when modeling generalist predators
463 (Magnússon & Pálsson, 1991) and recent analyses based on field data tend to support a type
464 3 relationship (Middlemas et al. 2006, Kempf et al. in press). In Ecopath with Ecosim
465 models (Walters et al. 1997), the functional form of interactions is based on the 'foraging
466 arena' concept that allows exploring alternative forms of functional response but biological

467 and ecological justifications about this concept remain controversial (Pláganyi 2007).

468 In the present analysis, cod age-groups were considered distinct prey, consistently with the
469 opportunistic behaviour of harp seals (Hammill & Stenson 2000) and the size-based structure
470 of marine food webs that regulates predation processes (e.g. Shin & Cury 2004). The multi-age
471 FR fitted to the data was a sigmoid shaped type 3 response, suggesting that the response of
472 seals to changes in cod abundance might be complex and highly non-linear, due to predator
473 preferences and potential for prey switching as a function of the relative abundance of prey
474 types in the environment. A sigmoidal shape for the FR would be consistent with the available
475 knowledge on the spatio-temporal and vertical distribution of cod that differs between cod
476 age-groups, juvenile cod being generally found in shallower waters and closer to the shore
477 than adult cod (e.g. Castonguay et al. 1999). Based on survey conducted on board the MV
478 “Gadus Atlantica” in winter from 1978 to 1994, young cod age-groups have been shown to
479 occupy different depths and areas in the NGS (DFO unpublished data), which could favor
480 the emergence of a type 3 FR. Cod migration and distribution patterns might also change in
481 time in relation to changes in environmental conditions affecting their habitat (Castonguay et
482 al. 1999).

483 The sensitivity analysis showed that a type 2 FR would be less consistent with both the data
484 and outputs from other predation models (Hammill & Stenson 2004, Duplisea & Hammill
485 2006). Although cod predation removals were of the same order of magnitude as those found
486 with models based on linear assumptions about seal consumption (Hammill & Stenson 2004,
487 Duplisea & Hammill 2006), the use of a FR led to different conclusions by accounting for
488 changes in cod age-groups abundance in time. Our results are consistent with Middlemas et
489 al. (2006) who provided empirical support for a type 3 FR of seals to Atlantic salmon in an
490 estuarine system in Scotland; however, in a multispecies or multi-age context, the interpretation
491 of the FR type is not straightforward as the changes in alternative prey abundance influence
492 the response of the predator to the availability of any prey (Smout & Lindstrøm 2007). This
493 will be important for harp seals who have been shown to show strong preference for pelagic
494 species such as capelin (*Mallotus villosus*) but are neutrally selective towards cod (Lawson et

495 al. 1998). Attempts to estimate statistical parameters of a FR have revealed the difficulty
496 of relating microscale observations of stomach contents to the effective feeding responses of
497 marine species at the macroscale (Pláganyi 2007). This is further complicated in the case of
498 harp seal because biomass estimates of important alternative prey such as capelin, sandlance
499 (*Ammodytes* sp), Arctic cod (*Boreogadus saida*) and krill (*Euphausidae* sp) are not available
500 for the NGSL. New data collection, experiments, and analyses such as conducted by Kempf
501 et al (in press) may help clarify harp seal sigmoidal feeding behaviour since modeling such
502 non-linear processes is key to explore the potential impact of management scenarios based on
503 a decline in seal population.

504 **Cod population dynamics**

505 Despite major differences with the current method used to assess the cod stock status (e.g.
506 no commercial data included in the analysis), SIMCAB estimates of cod abundance, fishing
507 mortality, and SSB accorded with the last assessment of the northern Gulf cod (MPO 2007).
508 Results were consistent with the collapse of the stock in the late 1980s and the current lack of
509 recovery (MPO 2007). In the present analysis, cod abundance in the initial year 1984 was based
510 on the numbers-at-age provided by VPA results, considering that the convergence property of
511 this method would give reliable estimates of abundance in the past (Jones 1961). Setting
512 the initial cod numbers helped in the estimation of the parameters linking cod abundance to
513 survey abundance indices, i.e. catchability and selectivity. Although a statistically 'optimal'
514 solution was obtained through the fitting process, the information provided to the model was
515 insufficient to distinguish between the level of residual mortality and magnitude of recruitment
516 at age 1, i.e. there could be a correlation between the maximum recruitment (R_{max}) and the
517 residual mortality parameters (α and β). Such a correlation could affect the absolute values of
518 residual mortality at age 1 but would not modify the model results for the other cod age-groups
519 for which information is provided through survey data.

520 A novel aspect of our cod population model is a stock-recruitment relationship based on egg
521 production, which for cod is considered a better measure of the true reproductive potential

522 of the stock than spawning biomass (Marshall et al. 2006). The high interannual variations
523 in abundance indices for cod aged 1-2 included both natural survival variability of cod larvae
524 and juveniles, and sampling noise associated with the research vessel gear that does not well
525 select very small cod (< 15 cm) predated by seals. In the context of trophic interactions
526 where small fish form the bulk of the diet of many predators, collecting data on fish larvae and
527 juveniles abundance appears as a key issue to explain the factors driving prerecruit survival
528 and recruitment.

529 In addition, total egg production in our model was derived from a statistical model that related
530 egg production to length-at-maturity and condition of mature females (Lambert 2008). Hence,
531 recruitment in the model accounted for changes in growth of cod that could be density and/or
532 temperature dependent (Swain et al. 2003), as well as changes in environment that could affect
533 egg productivity through cod condition. Natural mortality other than harp seal predation was
534 assumed age- and condition-dependent based on a comparative analysis between laboratory
535 feeding experiments and wild cod collected in the Gulf of St. Lawrence in the 1990s (Lambert
536 & Dutil 1997a). The residual natural mortality term for old ages was consistent with temporal
537 patterns estimated with virtual population analysis (Grégoire & Fréchet 2005). Accounting
538 for effects of condition on residual mortality led to a better fit of the model and suggested that
539 the decline in the condition and energy reserves of cod during the 1990s may have lowered the
540 productivity of the stock and contributed to its collapse (Lambert & Dutil 1997a).

541 In contrast to the VPA, SIMCAB allows the separation of seal predation mortality from fishing
542 mortality and other mortality through time and accounts for environmental influences on
543 mortality and cod fecundity. Modeling the mechanistic processes involved in predation is a
544 valuable alternative to the classic assumption of constant natural mortality rates to include
545 some ecosystem components in stock assessment methods (e.g. Lindstrøm et al. 2002) and
546 progressively move toward ecosystem-based fishery management (Pikitch et al. 2004). This
547 seems particularly critical for conducting stock projections to evaluate the expected effects of
548 fishery management rules (e.g. Hollowed et al. 2000) in a context where predation is suspected
549 to delay the NGSL cod stock recovery.

550 Cod recovery within an ecosystem perspective

551 The impact of changes in predation and fishing mortality on the SSB/R was assessed for
552 different values of residual natural mortality assumed here to represent different environmental
553 conditions. Diagnostics about a decline in seal predation were then shown to be dependent
554 on the productivity of the stock related to the environment (Dutil et al. 2003). Residual
555 mortality affecting young cod was shown to be more important than harp seal predation but
556 not explicitly modelled in the present analysis. Predation mortality has been shown to be the
557 dominant source of mortality for small cod (≤ 35 cm) in the 1990s and 2000s, representing
558 about 95% of total mortality (Savenkoff et al. 2006). In the NGSL, small cod have been shown
559 to be preyed upon by large cod through cannibalism, other large demersals (e.g. white hake
560 *Urophycis tenuis*), grey seals and boreal cetaceans including mysticetes such as minke whale
561 (*Balaenoptera acutorostrata*) and odontocetes such as white-beaked dolphins (*Lagenorhynchus*
562 *albirostris*) (Morissette et al. 2006, Savenkoff et al. 2004, 2006, 2007). While cannibalism
563 decreased in the last decades due to the decline in adult stock biomass, it still represented an
564 important component of mortality for small cod in the early 2000s (Savenkoff et al. 2007). A
565 high proportion of the total mortality of cod aged 3-10 was also not explained by harp seal
566 predation or fishing and attributed to residual mortality. This residual mortality includes both
567 a predation mortality component due to large cod predators such as grey seals and another
568 mortality component that could involve starvation, disease, and parasites (Savenkoff et al.
569 2004, 2006). In SIMCAB, mortality linked to these latter factors was related to cod condition
570 through the asymptote of residual mortality, assuming that fish in bad condition would have
571 little energy reserves to survive over the winter months or critical stages of their life cycle
572 (Lambert and Dutil 1997, Dutil et al. 1999).

573 Based on the large database of cod stomachs collected by Fisheries and Oceans Canada dur-
574 ing numerous research and commercial fishing-vessel surveys since the mid-1980s, modelling
575 explicitly cannibalism within SIMCAB could help separating the components of residual mor-
576 tality for small cod. Following the one-way interaction modelled between harp seal and cod,
577 predation by other species than harp seals such as grey seals could also be included in the

578 model if suitable data are available. The use of ecosystem models such as mass-balanced and
579 end-to-end models (Travers et al. 2007) in complement to SIMCAB would allow to represent
580 the full complexity of the food web, to identify the major factors affecting cod mortality, to
581 include bottom-up effects of prey availability on production and mortality, and eventually to
582 compare simulation outputs for assessing the robustness of the results.

583 Long-term projections based on the SSB/R analysis did not consider issues of recruitment in
584 a context where the low reproductive potential of the stock associated with a low productivity
585 are major factors explaining its lack of recovery (Dutil et al. 2003, Shelton et al. 2006).
586 Projections performed with SIMCAB to estimate recovery time should also account for the
587 effects of condition on recruitment (Lambert et al. 2000, Marteinsdottir & Begg 2002) and
588 growth (Dutil et al. 1999) that would also modify the response of the cod population to
589 the implementation of management rules based on a decline in seal population. Considering
590 alternative projection scenarios based on different harvest control rules for the commercial cod
591 fishery and different environmental regimes is important to determine the utility of managing
592 seals to increase the recovery rate of the NGSL cod population.

593 CONCLUSION

594 The main objectives of this study were to examine the relative impacts of fishing, environmental
595 conditions and seal predation on the decline and recovery of the NGSL cod population within
596 a consistent modeling framework. Within this framework harp seals could play an important
597 role in the recovery of NGSL cod. This framework lends itself to adding additional components
598 which might result in different conclusions. For example, harp seals are generalist predators
599 and their impact will be affected by the availability of alternative prey. Elsewhere it has been
600 shown that incorporating even only a few other prey alternatives may result in different and
601 unexpected outcomes (Punt & Butterworth 1995, Morissette et al. 2006, Matthiopoulos et al.
602 2007). Therefore, although harp seals may have an important impact on recovery of NGSL cod
603 population, efforts to manage seals with this objective in mind may not achieve the expected

604 outcomes because of other components within the NGSL marine ecosystem.

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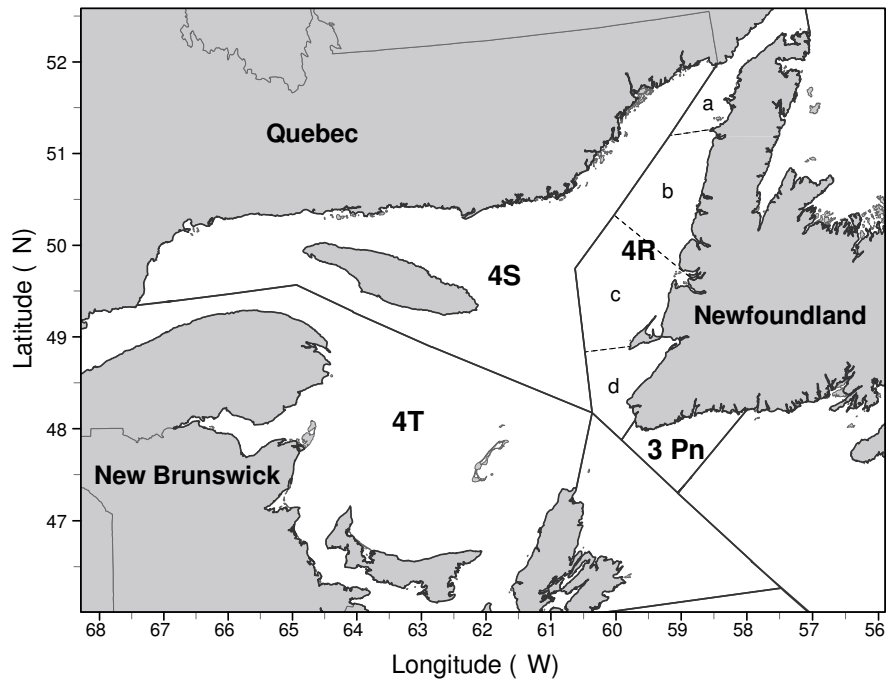


Figure 1: Location of the northern Gulf of St. Lawrence (NAFO divisions 4RS)

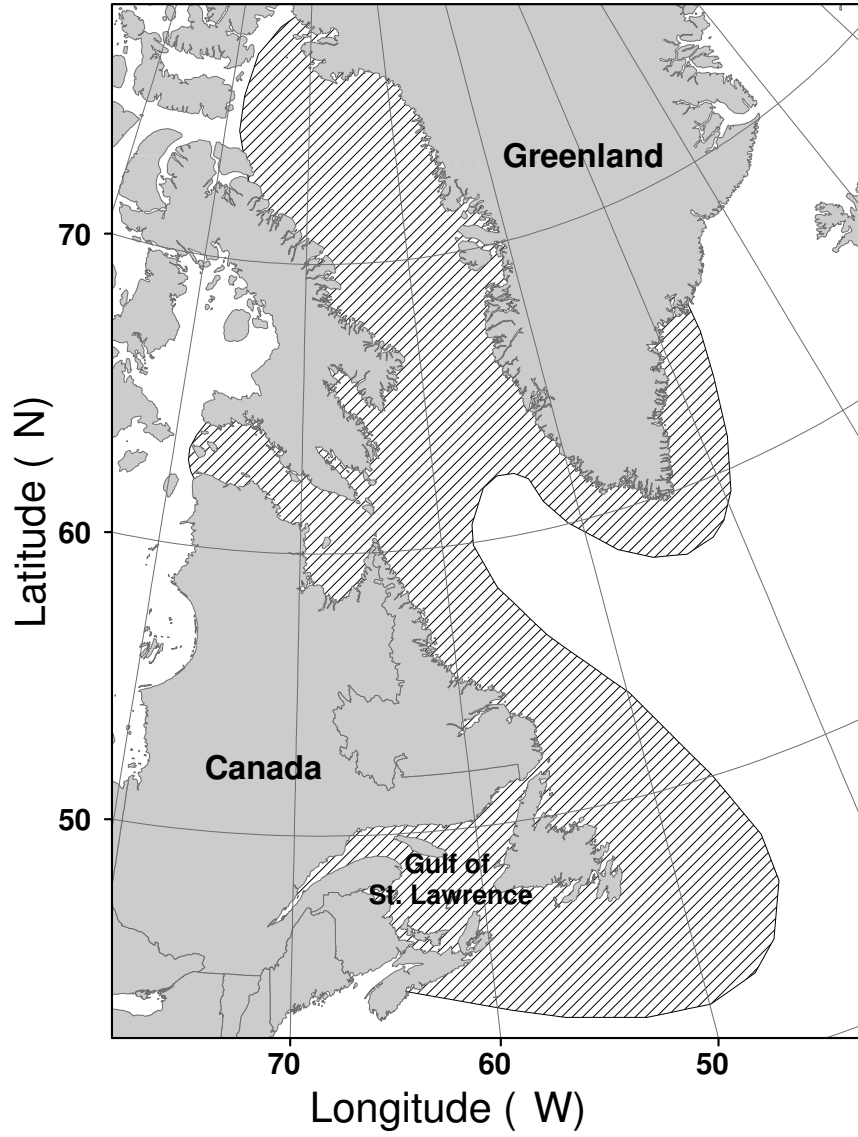


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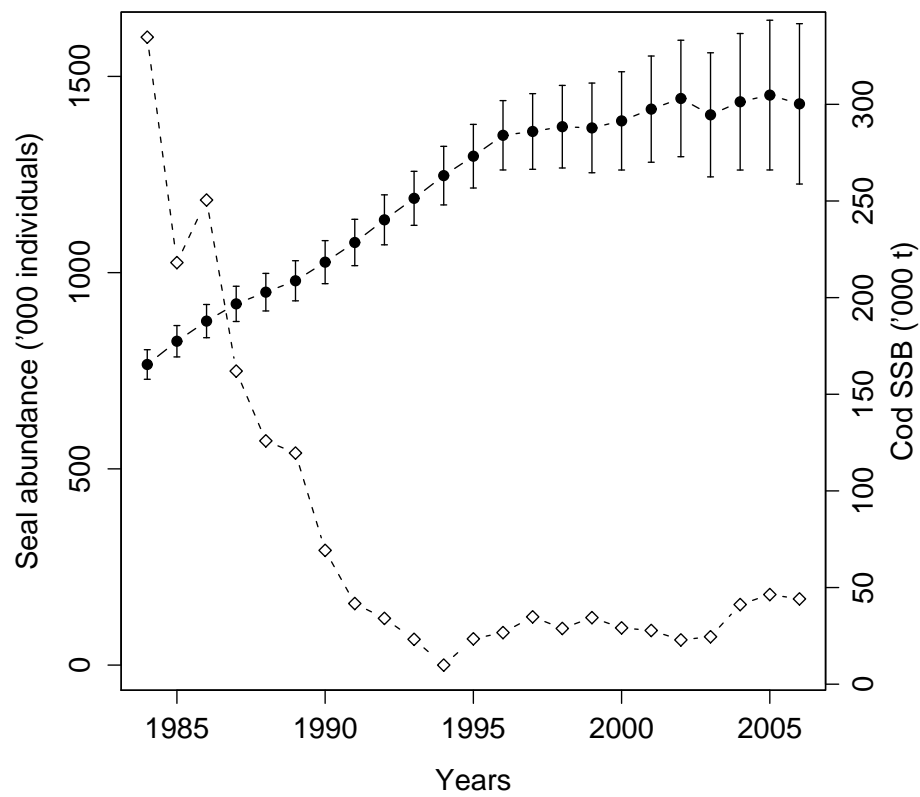


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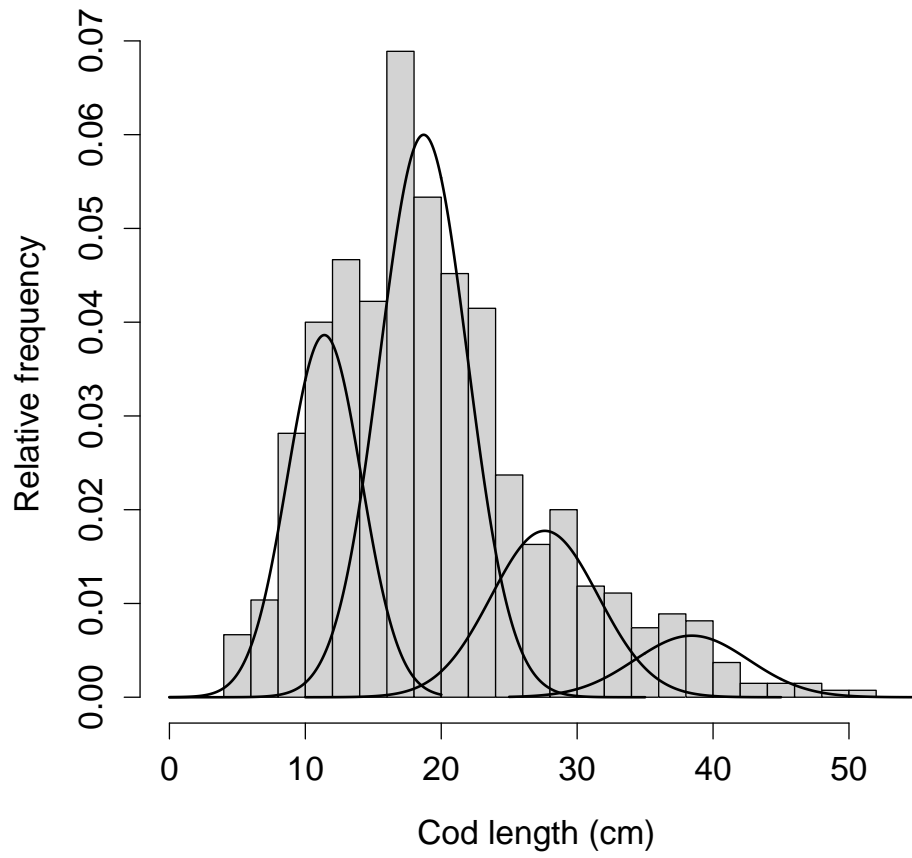


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Cod abundance indices

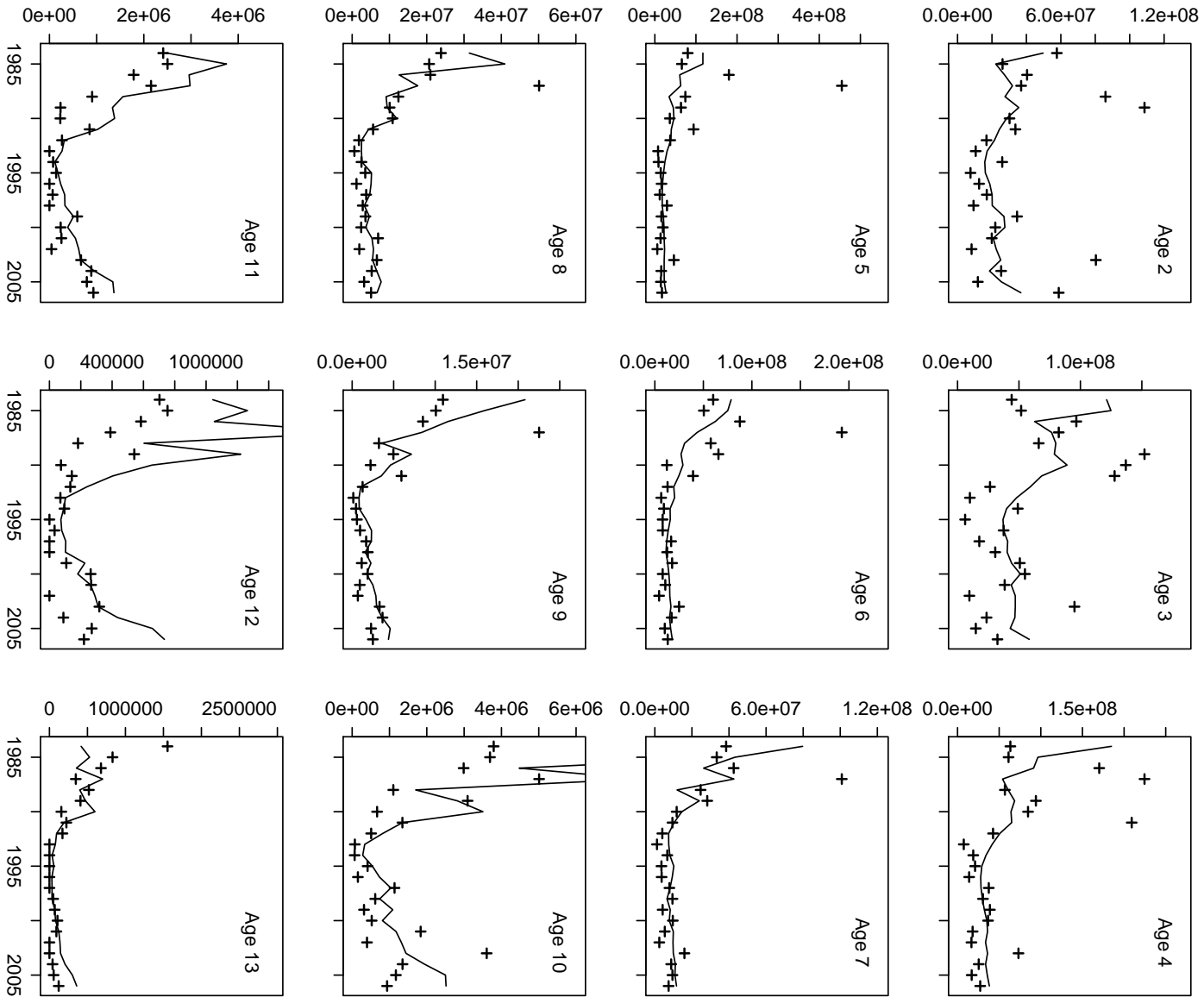


Figure 5: Survey numbers-at-age observed (cross) and predicted by the SIMCAB model (solid line). Scales were selected for readability along y-axes

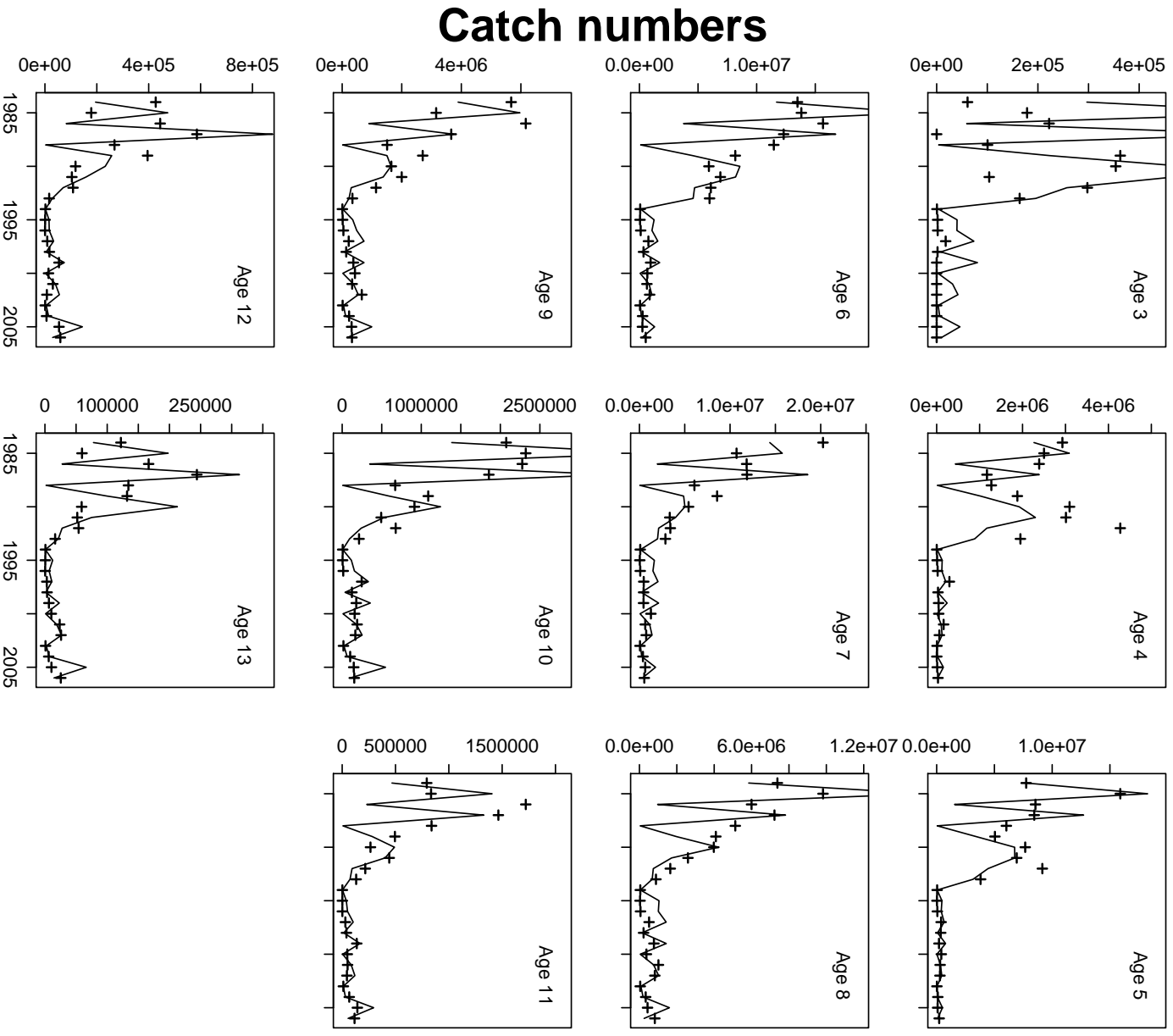


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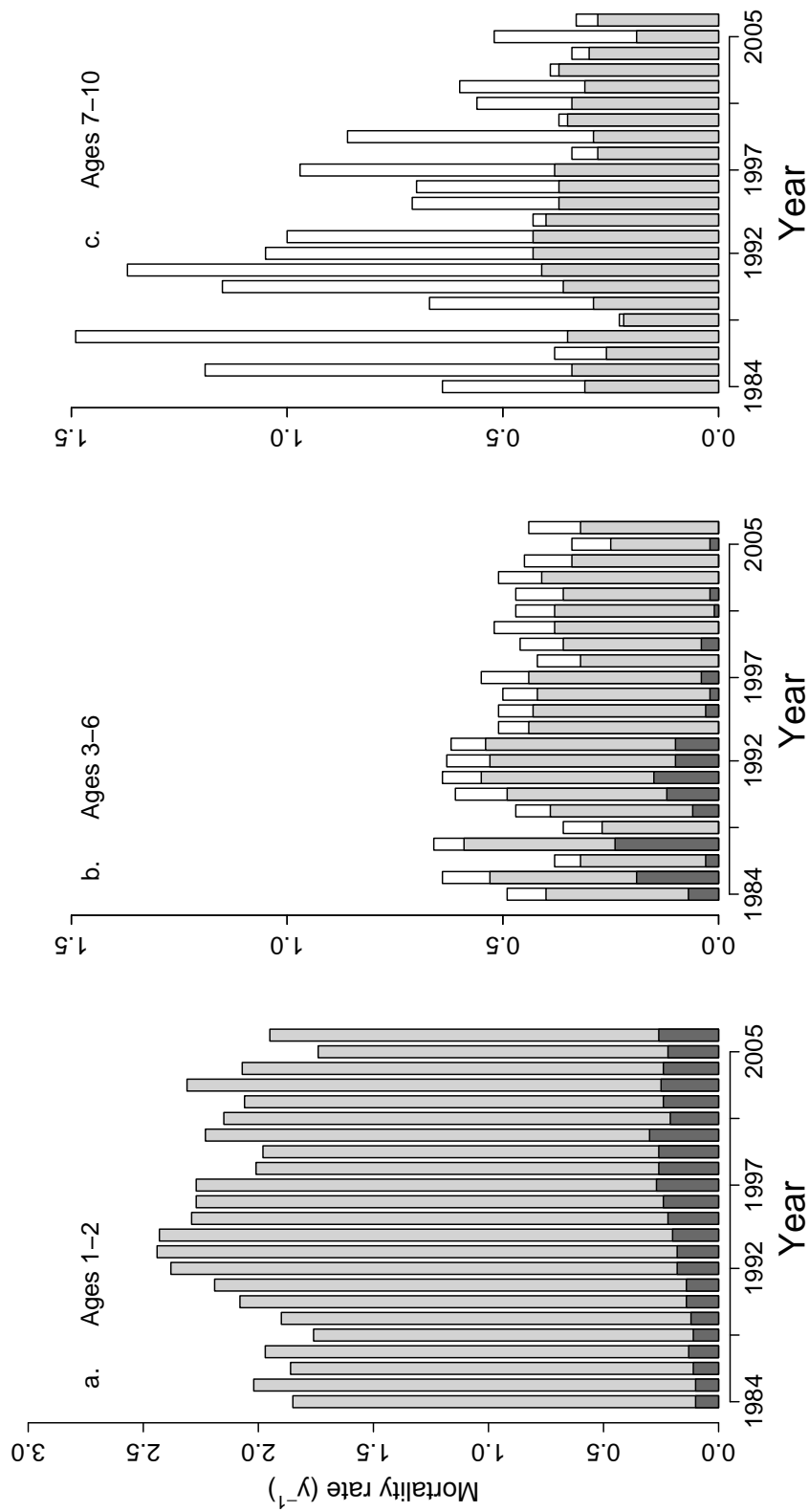


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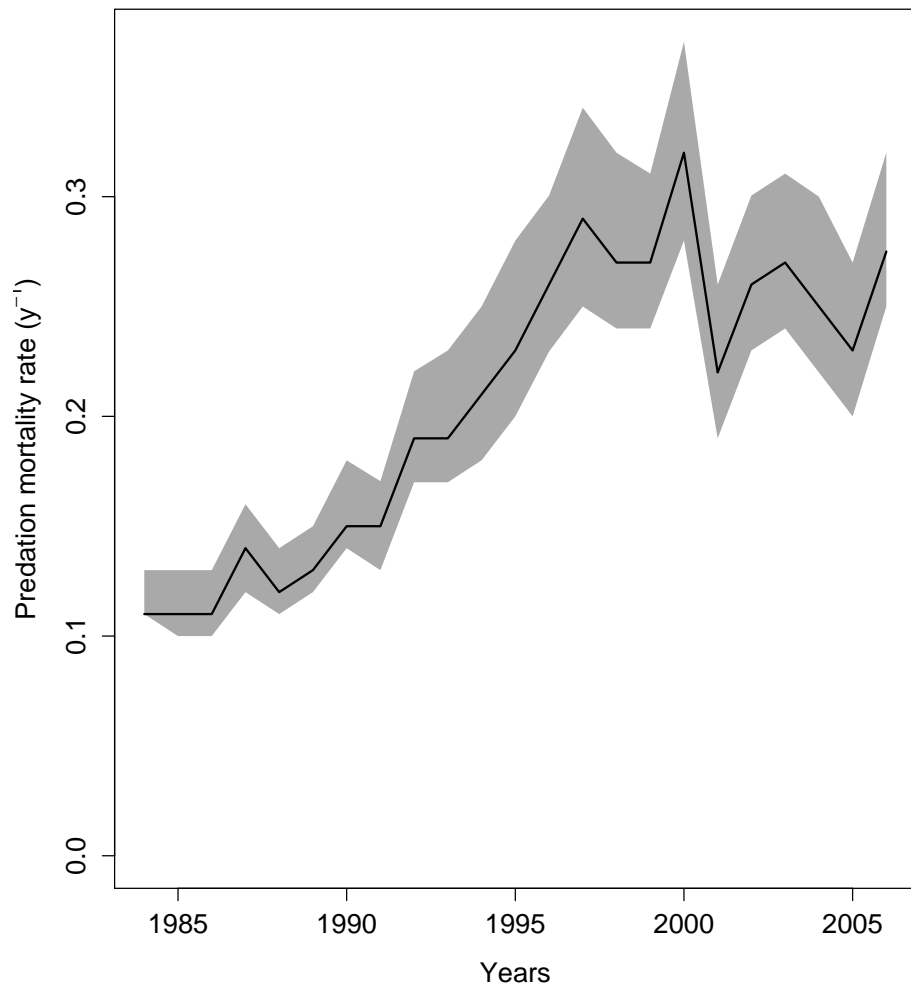


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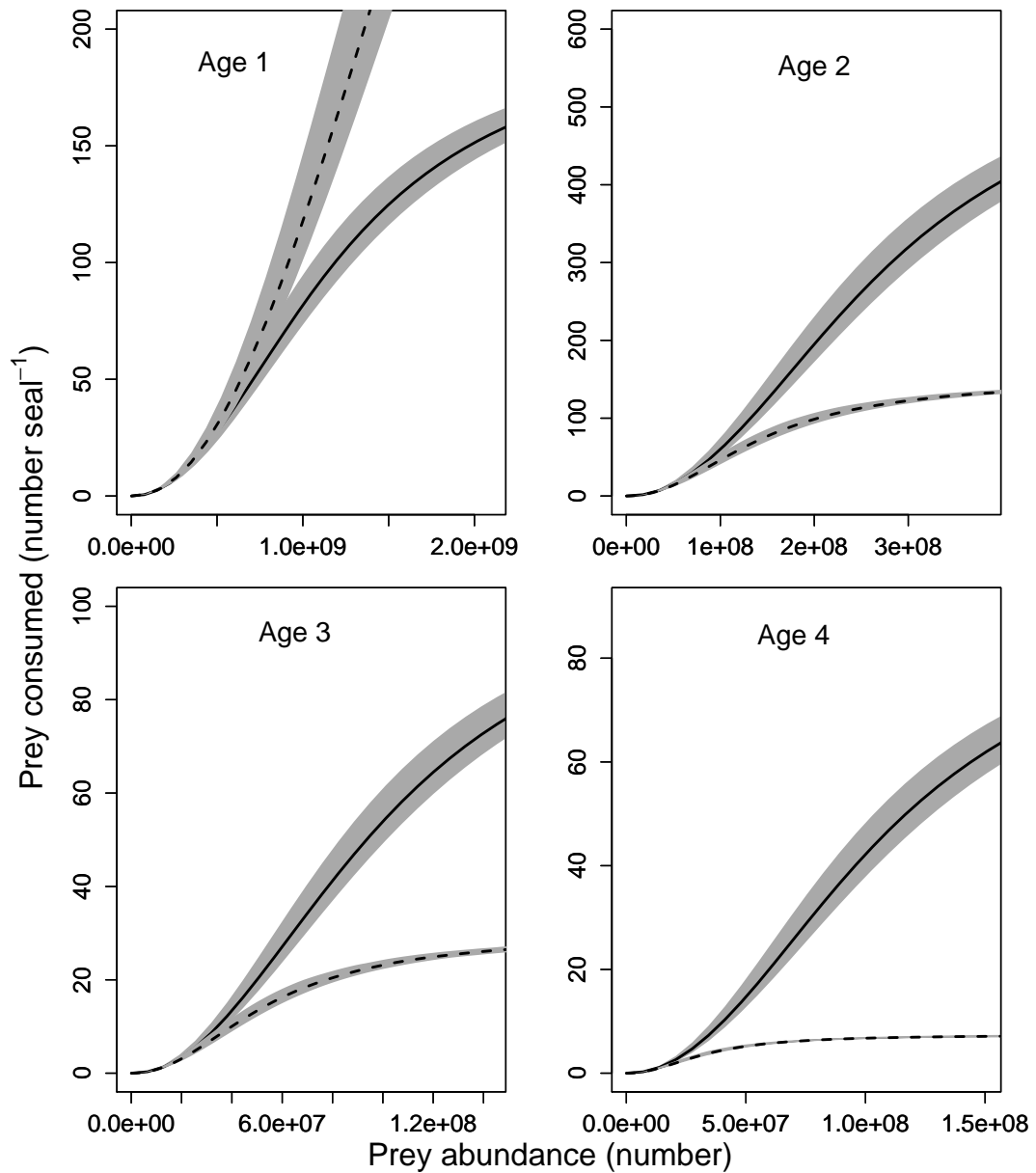


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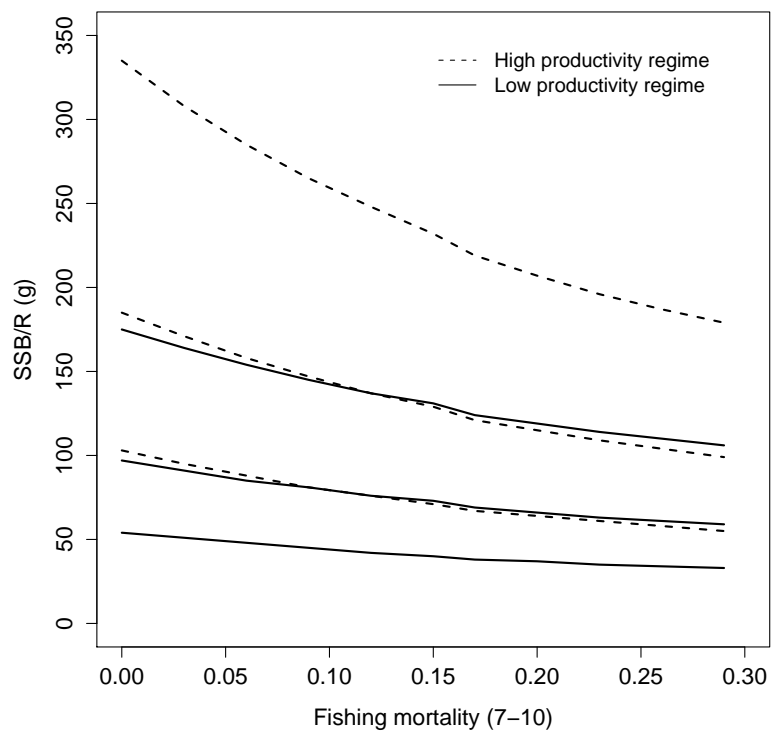


Figure 10: Spawning stock biomass per recruit (SSB/R) as a function of fishing mortality for 3 different levels of harp seal predation and 2 distinct productivity regimes (see text for details)

Table 1: Parameters and variables used in SIMCAB. NS: not shown

Notation	Definition	Origin	Value	Equation
B	Cod biomass (t)	Calculated	NS	D1
N	Cod numbers (number)	Calculated	NS	D1-D5,D9,D11-D12
w_a	Cod weight-at-age (t)	Fixed	NS	D1-D2
SSB	Spawning stock biomass (t)	Calculated	NS	D2
TEP	Total egg production (number)	Calculated	NS	D3,D8
ξ	Sex ratio	Fixed	NS	D3
ϕ	Proportion of maturing females	Fixed	NS	D1,D3
f	Fecundity (number of eggs cod ⁻¹)	Fixed	NS	D3
A	Last age-group	Fixed	13	D3
λ	Functional response of harp seals to cod (number seal ⁻¹ year ⁻¹)	Calculated	Fig. 9	D4,D5
ζ	Attack rate (number seal ⁻¹ year ⁻¹)	Calculated	NS	D4
ζ^{ref}	Reference attack rate (number seal ⁻¹ year ⁻¹)	Calculated	NS	D4
ρ	Scaling factor of the attack rates	Estimated	0.021 (0.003)	D4
J_{max}	Maximum consumption rate (number seal ⁻¹ year ⁻¹)	Calculated	996	D4
m	Shape parameter defining the functional response type	Estimated	2.13 (0.015)	D4
ρ	Number of feeding days spent by seals in Gulf each year	Fixed	150	D5

Table 1: Parameters and variables used in SIMCAB (continued)

Notation	Definition	Origin	Value	Equation
\overline{W}_t	Cod mean weight for ages targeted by seals (t)	Calculated	NS	D5
\overline{W}_0	Cod mean weight for ages targeted by seals in 1998-2001 (t)	Calculated	288e-6	D5
S	Seal numbers (number)	Fixed	Fig. 3	D5
M	Residual natural mortality rate (y^{-1})	Calculated	Fig. 7	D6
α	Intercept of the M curve (y^{-1})	Estimated	5.03 (0.013)	D6
β	Slope of the M curve ($y^{-1} \text{ a}^{-1}$)	Estimated	0.90 (0.002)	D6
ϑ	Asymptote of the M curve (y^{-1})	Calculated	NS	D6
F	Fishing mortality rate (y^{-1})	Estimated	Fig. 7	D7
$s_{a,c}$	Partial recruitment	Calculated	NS	D7
γ_c^1	Shape parameter of the partial recruitment (1984-1993)	Estimated	1.95 (0.007)	D7
δ_c^1	Age at which 50% of the individuals are vulnerable to fishing gear (1984-1993)	Estimated	5.39 (0.006)	D7
γ_c^2	Shape parameter of the partial recruitment (1994-2006)	Estimated	1.64 (0.017)	D7
δ_c^2	Age at which 50% of the individuals are vulnerable to fishing gear (1994-2006)	Estimated	6.46 (0.016)	D7
R	Recruitment (number)	Calculated	NS	D8
R_{max}	Maximum number of recruits produced (number)	Estimated	1.9e9 (6.9e6)	D8
r	TEP needed to produce recruitment equal to $R_{max}/2$ (number of eggs)	Estimated	5.3e11 (1.3e11)	D8
N'	Cod numbers in the middle of the year (number)	Calculated	NS	D9-D10
P	Cod numbers predated by the seals (number)	Calculated	NS	D9

Table 1: Parameters and variables used in SIMCAB (continued)

Notation	Definition	Origin	Value	Equation
C	Fishery catch (number)	Calculated	Fig. 6	D10-D11,D14-D15
I	Abundance index (number)	Calculated	Fig. 5	D12
$\varsigma_{a,s}$	Survey selectivity-at-age	Calculated	NS	D12
q	Survey catchability	Estimated	1.38 (0.076)	D12
γ_s	Shape parameter of the survey selectivity	Estimated	2.99 (0.028)	D12
δ_s	Age at which 50% of the individuals are vulnerable to the survey gear	Estimated	2.72 (0.007)	D12
$p_{a,t,s}$	Proportion of number-at-age in the survey	Calculated	NS	D13
$p_{a,t,c}$	Proportion of catch-at-age	Calculated	NS	D15

Table 2: Deterministic process and observation equations in the SIMCAB model. a and t index age and time respectively

State moments	
(D1)	$B_t = \sum_{a=1}^A w_a N_{a,t}$
(D2)	$SSB_t = \sum_{a=1}^A w_a N_{a,t} \phi_{a,t}$
(D3)	$TEP_t = \sum_{a=1}^A N_{a,t} \xi_{a,t} \phi_{a,t} f_{a,t}$
Mortality components	
(D4)	$\lambda_{a,t} = \frac{J_{max} \zeta_a N_{a,t}^m}{J_{max} + \sum_i \zeta_i N_{i,t}^m}$ with $\zeta_a = \varrho \zeta_a^0$
(D5)	$P_{a,t} = \frac{\lambda_{a,t} N_{a,t}}{\left(\frac{\bar{W}_t}{W_0}\right) S_t \frac{\rho}{365}}$
(D6)	$M_{a,t} = \vartheta_t + \alpha \exp\left(\frac{-a}{\beta}\right)$
(D7)	$F_{a,t} = \zeta_{a,c}^k F_t$ with $\zeta_{a,c}^k = \frac{1}{1 + \exp(-\gamma_c^k (a - \delta_c^k))}$
Process functions	
(D8)	$R_t = \frac{R_{max} TEP_{t-1}}{r + TEP_{t-1}}$
(D9)	$N'_{a,t} = (N_{a,t} - P_{a,t}) \exp(-M_{a,t}/2)$
(D10)	$C_{a,t} = N'_{a,t} (1 - \exp(-F_{a,t}))$
(D11)	$N_{a+1,t+1} = (N'_{a,t} - C_{a,t}) \exp(-M_{a,t}/2)$
Observation functions	
(D12)	$I_t = \sum_{a=1}^A q \zeta_{a,s} N_{a,t}$ with $\zeta_{a,s} = \frac{1}{1 + \exp(-\gamma_s (a - \delta_s))}$
(D13)	$p_{a,t,s} = \frac{I_{a,t}}{\sum_{a=1}^A I_{a,t}}$
(D14)	$C_t = \sum_{a=1}^A C_{a,t}$
(D15)	$p_{a,t,c} = \frac{C_{a,t}}{\sum_{a=1}^A C_{a,t}}$

Table 3: Proportion of Atlantic cod in harp seal diet weighted by their relative residency in each area during the period November-March. The year 1998 refers to the period November 1997-March 1998 and similarly for the other years

Year	Sample size			Proportion
	4Ra-c	4Rd-3Pn	4S	
1998	27	21	-	0.0506
1999	28	22	24	0.0538
2000	25	39	29	0.0382
2001	30	16	-	0.0377

Table 4: Parameters and variables used in the stochastic equations and likelihood components. a , t , s , and c index age, year, survey, and catch respectively

Notation	Definition	Equation
$I_{a,t}^*$	Observed abundance index	L1
$J_{a,t}^*$	log-observed abundance index	S1
T_1	Number of uncensored data, with $T_1 = \text{Card}(\bar{I})$	L1
Φ	Cumulative standard normal distribution	L1
ψ	Standard deviation of the observation error in the survey data	L1
$n_{t,s}^*$	Observed total number of fish taken by the survey vessels	L2
$p_{a,t,s}^*$	Observed proportion of number-at-age, with $p_{a,t,s}^* = \frac{I_{a,t}^*}{\sum_{a=1}^A I_{a,t}^*}$	L2
$p_{a,t,s}$	Predicted proportion of number-at-age	L2
T	Number of years	L2-L4
Γ	Gamma distribution	L2-L3
$p_{a,t,c}^*$	Observed proportion of catch-at-age	L3
$p_{a,t,c}$	Predicted proportion of catch-at-age	L3
$n_{t,c}^*$	Total catch sampled to establish age-proportions	L3
C_t^*	Observed total catch	L4
C_t	Predicted total catch	L4
σ_c	Standard deviation of the observation error in the catch data	L4

Table 5: Definitions used to extend the deterministic model in Table 2 to a stochastic model. iid: independent and identically distributed; \sim : distributed as; \mathcal{N} : normal distribution; *Dir*: Dirichlet distribution

(S1)	$J_t^* = \sum_{a=1}^A J_{a,t}^* \stackrel{iid}{\sim} \mathcal{N} \left(\sum_{a=1}^A \log I_{a,t}(\theta), \psi^2 \right)$	$\psi^2 = A\sigma^2 + A^2\tau^2$
	where $J_{a,t}^* = \log(I_{a,t}^*) = \log(I_{a,t}) + \epsilon_{a,t} + \eta_t$	$\epsilon_{a,t} \stackrel{iid}{\sim} \mathcal{N}(0, \sigma^2)$ $\eta_t \stackrel{iid}{\sim} \mathcal{N}(0, \tau^2)$
(S2)	$p_{a,t,s}^* \stackrel{iid}{\sim} \text{Dir}(p_{1,t,s}(\theta), \dots, p_{A,t,s}(\theta), n_{t,s}^*)$	
(S3)	$p_{a,t,c}^* \stackrel{iid}{\sim} \text{Dir}(p_{1,t,c}(\theta), \dots, p_{A,t,c}(\theta), n_{t,c}^*)$	
(S4)	$\log C_t^* \stackrel{iid}{\sim} \mathcal{N} \left(\log C_t(\theta) - \frac{\sigma_c^2}{2}, \sigma_c^2 \right)$	

Table 6: Likelihood components of the SIMCAB model. The generic notation θ represents the set of parameters to estimate

Equation	Likelihood
(L1)	$L(\{I_{a,t}^*\} \theta) = \underbrace{\frac{1}{(2\pi)^{\frac{T_1}{2}} \psi^{T_1}} \exp \left\{ - \sum_{t \in \bar{I}} \frac{\left(\sum_{a=1}^A \log \frac{I_{a,t}^*}{I_{a,t}(\theta)} \right)^2}{2\psi^2} \right\}}_{\text{real observations}} \underbrace{\prod_{t \in I} \left\{ 1 - \Phi \left(\frac{\sum_{a=1}^A \log \frac{I_{a,t}^*}{I_{a,t}(\theta)}}{\psi} \right) \right\}}_{\text{censored observations}}$
(L2)	$L(\{p_{a,t,s}^*\} \theta) = \prod_{t=1}^T \Gamma(n_{t,s}^* + 1) \frac{\prod_{a=1}^A (p_{a,t,s}^*)^{n_{t,s}^* p_{a,t,s}(\theta)}}{\prod_{a=1}^A \Gamma(n_{t,s}^* p_{a,t,s}(\theta) + 1)}$
(L3)	$L(\{p_{a,t,c}^*\} \theta) = \prod_{t=1}^T \Gamma(n_{t,c}^* + 1) \frac{\prod_{a=1}^A (p_{a,t,c}^*)^{n_{t,c}^* p_{a,t,c}(\theta)}}{\prod_{a=1}^A \Gamma(n_{t,c}^* p_{a,t,c}(\theta) + 1)}$
(L4)	$L(\{C_t^*\} \theta) = \frac{1}{(2\pi)^{\frac{T}{2}} \sigma_c^T} \exp \left\{ - \frac{1}{2\sigma_c^2} \sum_{t=1}^T \left(\log \frac{C_t^*}{C_t(\theta)} - \frac{\sigma_c^2}{2} \right)^2 \right\}$

Table 7: Relative sensitivity (%) of biomass removed (BR) by seals and predation mortality rate for cod aged 1-4 ($M_p(1-4)$) to a 10% increase in input parameters

Parameter	Base value	BR	$M_p(1 - 4)$
% in the NGSL	25	67	129
% cod contribution to diet 1998-2001	4.53	1	25
% cod maximum contribution to diet	18.3	-6	-2
Average energy of cod (kJ g^{-1})	4.96	-13	40
Kleiber multiplier (kJ)	293	67	129