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Practical stakeholder-driven harvest policies for groundfish fisheries in British Columbia, Canada

Sean P. Cox [a,*], Allen Robert Kronlund [b]

[a] *School of Resource and Environmental Management, Simon Fraser University, 8888 University Drive, Burnaby, BC, Canada V5A 1S6*
[b] *Pacific Biological Station, Fisheries and Oceans, 3190 Hammond Bay Road, Nanaimo, BC, Canada V9T 6N7*

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ABSTRACT

Fisheries co-management processes must provide a mechanism for industry stakeholder involvement in decision-making, while also providing assurance that precautionary actions will be taken to conserve fish stocks into the future. We used a management strategy evaluation approach to evaluate practical harvest policies suggested by industry stakeholders in a co-managed fishery for sablefish (*Anoplopoma fimbria*) in British Columbia, Canada. These harvest policies included (i) data-based procedures that average recent catch limits with a smoothed research survey index of abundance and (ii) model-based procedures that rely on annual catch-at-age stock assessment modeling to estimate stock biomass. Both approaches attempt to implement constant exploitation rate harvest policies. We evaluated these procedures in four simulation scenarios that encompassed some important uncertainties related to current stock size and productivity. Both procedure types performed close to a perfect-information procedure in terms of catch, catch variability, and conservation, provided that exploitation rate policy parameters were set close to their optimal values. The smoothing function used in data-based procedures caused lags in which declines (increases) in catch limits extended for longer periods than declines (increases) in stock biomass. However, these lags did not create long-term adverse effects on performance. Model-based procedures generally performed better in terms of catch and inter-annual variability in catch. Interactions between harvest policy exploitation rates and stock assessment model biases caused similar lags as those of data-based procedures, although such biases also did not degrade performance severely. Our results, combined with empirical experience elsewhere suggest that data-based management procedures provide an appealing and practical means of setting annual catch limits either in the absence of an accepted model-based approach, or preferably, in combination with periodic stock assessment modeling. Such an approach provides transparency in a co-management process, while sacrificing little in terms of long-term conservation and utilization.

1. Introduction

One of the primary goals of fisheries management is to balance short-term harvest opportunities against the need to conserve fish stocks for future use. International agreements such as the Precautionary Approach to Capture Fisheries (FAO, 1995) reflect this goal by specifying that priority should be given to maintenance of the long-term productive capacity of the resource over short-term economic or social needs. Canada, which is signatory to the Precautionary Approach, has developed national policy that is consistent with maintenance of long-term productive capacity of fisheries resources (DFO, 2006). Fisheries management in Canada provides incentives to stakeholders in the form of harvest rights and involvement in decision-making over regulations. Such co-management situations give rise to at least two possible sources of conflict. First, there is a direct conflict between national policy, focused on conserving long-term production capacity, and stakeholder desires for greater short-term economic gain. Second, formal mechanisms for involving stakeholders in fishery decision-making processes remain undefined, despite significant financial and in-kind stakeholder contributions to the fisheries management system.

Management strategy evaluation (MSE) offers a potential vehicle for addressing both policy and process conflicts in fishery co-management. The management strategy evaluation process involves defining a set of operational objectives, identification of candidate management procedures (i.e., data collection, stock assessment, and harvest control rules), and a prospective evaluation of the procedures against the objectives (de la Mare, 1996). Operational objectives represent specific components of fisheries

* Corresponding author. Tel.: +1 778 782 5778; fax: +1 778 782 4968.
E-mail addresses: spcox@sfu.ca (S.P. Cox), allen.kronlund@pac.dfo-mpo.gc.ca (A.R. Kronlund).

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management policy. For example, a policy that requires maintenance of the productive capacity of the resource can be represented by objectives related to minimum spawning stock size, while economic policy components can be represented by both short- and long-term catch levels as well as inter-annual variability of catch. Stock assessment methods and harvest control rules represent the decision-making process. Traditionally, the stock assessment component of management procedures has been a scientific choice, and the long-term policy consequences of particular assessment model choices are rarely evaluated. Harvest control rules, which specify the catch limit as a function of quantities estimated in stock assessments, represent the mechanism for implementing fisheries harvest policies. The final prospective evaluation component of management procedures involves testing a range of plausible scenarios for the stock and fishery dynamics, typically by computer simulation. Involving stakeholders in the development of all management procedure components facilitates co-management of the process (Smith et al., 1999). Furthermore, where there is confidence in the process, management procedures are more likely to be followed faithfully, which increases the likelihood that long-term policy objectives will be met (Rademeyer et al., 2007; Hilborn et al., 2002).

Stock assessment models are often the most contentious component of fishery management procedures. The growing complexity of stock assessment models appears to lead to frustration among fishery managers and stakeholders (Cotter et al., 2004), which potentially limits the use of scientific advice and instead tends to focus discussion on the technical aspects of model fitting at the expense of how best to provide management advice. Evaluation of whole management procedures does not necessarily relieve the technical burden and can appear to stakeholders to be an even more complex and technical exercise. Nevertheless, advantages of the management strategy evaluation method include focused attention on meeting long-term precautionary management objectives, providing information about trade-offs associated with alternative fishery management procedures (Butterworth and Punt, 1999; Walters and Martell, 2004), and robustness testing under known uncertainties. Furthermore, by regarding the data collection, stock assessments, and harvest control rules as part of a common process, the management strategy evaluation approach allows comparisons among alternative procedures to be made based on both performance and overall management cost.

This paper compares the performance of relatively simple data-based fishery management procedures with model-based procedures as might be applied to the sablefish (*Anoplopoma fimbria*) fishery off British Columbia (B.C.), Canada. We develop data-based management procedures that set annual catch levels by combining the preceding year's catch limit with the recent average of fishery-independent surveys, thus eliminating the traditional annual stock assessment modeling component. In contrast, model-based harvest control rules set annual catch limits using the constant exploitation rate policy $C_t = U^{\mathrm{ref}} B_t$, where U^{ref} is a reference exploitation rate and B_{t} is an estimate of stock biomass from a statistical catch-at-age model. The model-based procedures attempt to mimic more elaborate management systems that depend on stock assessment modeling and more demanding data requirements. More complex model-based procedures should have a greater chance of providing for optimal harvest if they consistently produce unbiased estimates of stock size. All candidate management procedures are tested in a simulation feedback control loop against a known fishery-operating model representing the stock, observation, and fishery dynamics. Such an approach to testing harvest management procedures is well documented in the literature (e.g., Walters, 1986; de la Mare, 1996; Cooke, 1999; Punt and Smith, 1999; Butterworth, 2007).

2. Methods

2.1. Sablefish and the fishery and data for sablefish off British Columbia

Sablefish (*A. fimbria*) inhabit Pacific Ocean shelf and slope waters to depths greater than 1500 m, from central Baja California to the Bering Sea and Japan (Beamish and McFarlane, 1988). Spawning occurs from January to March along the continental shelf at depths greater than 300 m and larval sablefish are found in surface waters over the shelf and slope in April and May. Juveniles migrate inshore over the following 6 months and rear in near shore and shelf habitats until ages 2–5 when they migrate offshore and recruit to deeper waters where they become vulnerable to the offshore trawl, longline trap, and longline hook fisheries. Sablefish early growth is rapid with mature females reaching an average length of 55 cm, and a maximum of 70+ cm, in 3–5 years. The oldest fish aged to date in B.C. waters was 87 years.

A commercial fishery for sablefish has operated off B.C. since the late 19th century (McFarlane and Beamish, 1983). Since full development of the fishery in the 1960s, total annual landings have ranged from 2349 *t* to 7691 *t* with an annual average of approximately 4200 *t*. The targeted sablefish fishery has operated under an individual transferable quota (IVQ) system limited to 48 license holders since 1990. Sablefish industry stakeholders have collaborated with Fisheries and Oceans Canada in the management and monitoring of the fishery, and in the collection of stock assessment data through annual surveys and tagging programs since inception of the IVQ system. Landings data generally improve in quality over time and have been dockside validated since 1990. A trap fishery catch rate index (kg/trap) is derived from fishery logbook data beginning in 1979 and a coast-wide survey provides fishery-independent trap catch rates beginning in 1990. Fish ages are available from both the commercial fishery and the survey, although not for all years and sometimes with relatively low annual sample sizes. A tag-release and recapture program has been in place since 1991, with releases occurring during annual stock assessment surveys and recaptures obtained through both the targeted and non-targeted Canadian fisheries and via U.S. fisheries (e.g., Wyeth et al., 2007).

2.2. Management procedures and their evaluation

The following sections describe our simulation approach to developing and testing alternative management procedures for the sablefish fishery. The work was initiated following consultations with stakeholders (fishery managers and the sablefish industry association) to determine if management strategy evaluation would be feasible for sablefish. We begin by describing the operating model for the fishery, which we use as the "real world" in which candidate management procedures are tested. We develop four versions of this model to represent the key scenarios that we feel bracket stock conditions that are plausible at this time. Plausibility is determined by fitting these operating models to existing fishery and survey data. We then present two classes of "practical" management procedures that were suggested by sablefish industry stakeholders as potential methods for setting annual catch limits. Both procedure classes – data-based and model-based – consist of (i) a stock assessment step in which simulated data from the operating model are interpreted (or smoothed), and (ii) a decision step in which a constant exploitation rate harvest policy translates the assessment information into a catch limit. The two classes mainly differ in terms of the level of complexity involved in the stock assessment step.

Table 1
Sablefish fishery-operating model for generating age-structured population dynamics, survey indices of relative abundance, and age-proportion data

Parameters	
T1.1	$\Theta = (B_0, \tau_1, \tau_2, q)$
Life history schedules	
T1.2	$l_a = L_\infty + (L_1 - L_\infty)e^{-k(a-1)}$
T1.3	$w_a = \exp(-23.6)l_a{}^{3.1}$
T1.4	$m_a = \dfrac{a^{\mu_1}}{a^{\mu_1} + \mu_2^{\mu_1}}$
Fishery selectivity	
T1.5	$\tilde{s}_{g,a} = \begin{cases} 0 & l_a < \bar{l} \\ \left(\dfrac{1}{1+e^{-\beta_{2,g}(l_a-\beta_{1,g})}}\right)\left(1-\dfrac{1}{1+e^{-\beta_{4,g}(l_a-\beta_{1,g}-\beta_{3,g})}}\right) & l_a \ge \bar{l} \end{cases}$
T1.6	$s_{g,a} = \dfrac{\tilde{s}_{g,a}}{\sum_a \tilde{s}_{g,a}}$
Unfished equilibrium recruitment	
T1.7	$\phi = \displaystyle\sum_{a=1}^{A-1} e^{-M(a-1)} m_a w_a + \dfrac{e^{-M(A-1)} m_A w_A}{1 - e^{-M}}$
T1.8	$R_0 = B_0/\phi$
T1.9	$N_{1,1} = R_0$
Initial population	
T1.10	$N_{a,1} = R_0 e^{-M(a-1)}, 2 \le a \le A-1$
T1.11	$N_{A,1} = R_0 e^{-M(A-1)}/(1-e^{-M})$
Age proportions in catch	
T1.12	$u_{g,a,t} = \dfrac{s_{g,a} N_{a,t}}{\sum_{a=1}^{A} s_{g,a} N_{a,t}}$
State dynamics	
T1.13	$\omega_t \sim N(0,1)$
T1.14	$N_{1,t} = \dfrac{4hR_0 S_{t-1}}{B_0(1-h) + (5h-1)S_{t-1}} \exp[\omega_t \sigma_R - 0.5\sigma_R^2]$
T1.15	$N_{a,t} = e^{-M}\left[N_{a-1,t-1} - \displaystyle\sum_{g=1}^{4} \dfrac{u_{g,a-1,t-1} C_{g,t-1}}{w_{a-1}}\right] \quad 2 \le a \le A-1$
T1.16	$N_{A,t} = e^{-M}\left[N_{A-1,t-1} + N_{A,t-1} - \displaystyle\sum_{g=1}^{4} \dfrac{(u_{g,A-1,t-1} + u_{g,A,t-1}) C_{g,t-1}}{w_A}\right]$
T1.17	$B_t^* = \displaystyle\sum_{a=1}^{A} s_{2,a} w_a N_{a,t}$
T1.18	$S_t = \displaystyle\sum_{a=1}^{A} m_a w_a N_{a,t}$
Survey and catch-at-age observations	
T1.19	$\delta_t \sim N(0,1), \varepsilon_{g,a,t} \sim N(0,1)$
T1.20	$I_t = qB_t^* \exp[\tau_1 \delta_t - 0.5\tau_1^2]$
T1.21	$x_{g,a,t} = \log u_{g,a,t} + \tau_{2,g}\varepsilon_{g,a,t} - \dfrac{1}{A}\displaystyle\sum_{a=1}^{A}[\log u_{g,a,t} + \tau_{2,g}\varepsilon_{g,a,t}]$
T1.22	$p_{g,a,t} = \dfrac{\exp[x_{g,a,t}]}{\sum_{a=1}^{A} \exp[x_{g,a,t}]}$

Beginning at the top, this table sequentially defines the population dynamics and monitoring observations. The parameters in T1.1 were estimated using a similar likelihood formulation as in Table 3. Model notation and parameter values are given in Table 2.

2.2.1. *Operating model*

We used an age-structured population dynamics model to construct scenarios for the "true" sablefish population in management strategy simulations (Table 1). Model notation and parameter settings are provided in Table 2. All operating model scenarios assume that the B.C. sablefish spawning stock was at unfished, deterministic equilibrium B_0 prior to directed fisheries in the mid 1960s. The models further assume that the B.C. population is closed to immigration and emigration. Equations T1.7–T1.11 initialize the population age composition to the unfished equilibrium. Simulated annual recruitment of age-1 fish (T1.14) is log-normally distributed about a Beverton–Holt stock-recruitment relationship. The unfished spawning biomass B_0 and steepness of the stock-recruitment relationship h determine the stock-recruitment relationship, and are therefore among the most important uncertainties in management strategy simulations (Butterworth and Punt, 1999; Walters and Martell, 2004).

The operating model simulates and appends research survey and catch-at-age observations to the existing sablefish monitoring dataset during each annual cycle. The research survey index of relative abundance (T1.20; kg/trap) is proportional to the biomass available to the survey gear (T1.17) with stochastic errors that are log-normal and corrected for bias by subtracting $0.5\tau_1^2$ from each observation. The bias correction is necessary here because simula-

Table 2
Notation for the operating model and catch-at-age stock assessment model

Symbol	Value	Description
Indices		
t	$\{1, 2, \ldots T\}$	Annual time step ($T = 82$)
a	$\{1, 2, \ldots A\}$	Age-class in years where ($A = 25$)
g	$\{1, 2, 3, 4\}$	Gear type index for trap fishery, survey, longline, and trawl, respectively
Model parameter**s**		
B_0	Table 4	Unfished spawning biomass (tonnes)
q	++	Catchability coefficient for research survey
τ_1	++ 0.26	Coefficient of variation for research survey abundance index
$\tau_{2,g}$	++, 0.66, 0.45	Standard error in observed proportions-at-age for $g = 1, 2$
$\hat{R}_t$	++	Age-1 recruitment in year t for catch-at-age model
$\hat{N}_{a,1}$	++	Initial abundance by age-class in catch-at-age model ($2 \le a \le A$)
h	0.45, 0.65	Recruitment function steepness
$\beta_{i,g}$	Fig. 1	Selectivity function parameters for gear g
σ_R	0.70	Standard error of log-recruitment deviations
M	0.08	Instantaneous natural mortality rate (yr^{-1})
L_∞	68.2	Asymptotic length (cm)
L_1	40.7	Length-at-age 1 (cm)
k	0.37	von Bertalanffy growth constant
μ_2	5	Age-at-50% maturity (yr)
μ_1	8	Maturity-at-age function power
Derived variables		
R_0		Unfished recruitment
$s_{g,a}$		Selectivity-at-age in fishery, g
m_a		Proportion mature-at-age
l_a		Length-at-age (cm)
w_a		Body mass-at-age (tonnes)
ϕ		Unfished equilibrium spawning biomass per recruit (tonnes)
State variables		
$N_{a,t}$		Number of age, a, fish in year, t
B_t^*		Biomass of fish vulnerable to research survey (tonnes)
$u_{g,a,t}$		Proportion of age, a, fish in harvestable population
S_t		Spawning biomass in year, t (tonnes)
Observations		
I_t		Research survey index value in year, t
$p_{g,a,t}$		Proportion of age, a, fish in gear, g, catch-at-age sample
n_g		Number of years with catch-at-age data for gear, g
Fishery controls		
$C_{g,t}$		Catch in fishery, g (tonnes)
$\bar{l}$	55	Minimum size limit in fisheries (cm). Does not apply to survey

Values in regular font are fixed in the operating model and bold values are fixed parameters that are common to both models. The (++) symbol indicates parameters estimated by the catch-at-age model

tion testing of data-based harvest policies requires that simulated future surveys have the same expected values as historical surveys for the same biomass levels. Fishery catch-at-age proportions and research survey catch-at-age proportions (T1.22) are modeled using multivariate-logistic random variables with gear-specific standard errors $\tau_{2,g}$ (Schnute and Richards, 1995).

Operating model parameters in T1.1 were estimated by fitting to gear-specific catch (1965–2006), trap fishery catch-per-unit-effort (CPUE; 1979–2006), research survey CPUE (1990–2002), fishery catch-at-age (1979–2002), and research survey catch-at-age (1988–2004). We used a penalized likelihood approach that was nearly identical to the one used in the catch-at-age assessment model (described below), except with additional likelihood components for fishery CPUE. Not all years were represented within the range of the two catch-at-age series. Natural mortality, length-at-age, maturity-at-age, and average selectivity-at-length function parameters were all estimated external to the operating model (Table 2).

2.2.1.1. Operating model scenarios. Candidate management procedures were tested against four operating models that highlight key uncertainties about the sablefish stock. The four operating models we chose for this paper result from setting two uncertain factors at two levels each. The first uncertain factor for B.C. sablefish – stock productivity – arises for two reasons. First, the fishery has taken a steady average catch since the 1970s while fishery and survey catch per unit effort have either remained steady or declined. Such a "one-way trip" (Hilborn and Walters, 1992) pattern does not allow us to easily distinguish between a large unfished biomass combined with low productivity and low unfished biomass combined with high productivity. Second, our estimates of stock productivity depend on what we assume about the natural mortality rate of sablefish.

Productivity can be represented in the operating models by adjusting value of the steepness of the stock-recruitment relationship h, which is defined as the fraction of the unfished recruitment that occurs when the spawning stock biomass is reduced to 20% of the unfished level. A steepness value near $h = 1.0$ means that expected recruitment is the same as unfished recruitment when the spawning stock is reduced to 20% of its unfished level. In an analysis of more than 700 stock-recruitment data sets, Myers et al. (1999) found that steepness averaged 0.69 over a wide range of fish families. Sablefish, which were included in the study, had the lowest steepness value in the entire study at $h = 0.26$. Our estimates of steepness based on fitting the operating model to the available data for B.C. sablefish are either $h = 0.49$ (S.E. = 0.11) or $h = 0.56$ (S.E. = 0.16) depending on assumptions about how well trap fishery CPUE reflects stock biomass (see next section). Therefore, we chose $h = 0.45$, 0.65 to bracket these values in the operating model.

The second factor distinguishing operating models is the current status of B.C. sablefish relative to average unexploited conditions. Similar to many stocks around the world, sablefish biomass estimates for the first two decades of commercial fishing depend strongly on fishery catch-per-unit-effort (CPUE). Obviously, there are clear dangers involved in using CPUE as an index of stock abundance under the assumption that it is linearly proportional to abundance (Hilborn and Walters, 1992), especially over long time periods such as 1970–2000 during which rapid evolution in fishing technology occurred. On the other hand, ignoring CPUE leaves a very short time-series of fishery-independent information that may provide unreliable estimates of unfished conditions and thus current stock status. As a compromise, we fitted two versions of the operating model to fishery CPUE in combination with surveys and catch-at-age. In the first, we assumed that CPUE is linearly proportional to exploitable biomass. Under this scenario, estimated 2007 spawning biomass is 29% and 31% of the deterministic unfished equilibrium for $h = 0.45$ and $h = 0.65$, respectively. In our second approach, we admitted the possibility that CPUE could remain high and stable (i.e., hyperstable, Hilborn and Walters, 1992) over a wide range of sablefish biomass. We implemented this assumption by treating hyperstability as a free parameter in the operating model. The estimated 2007 spawning stock biomass corresponding to this hyperstability assumption is 18% and 20% of the unfished level for $h = 0.45$ and $h = 0.65$, respectively (Table 4). Both hyperstability scenarios are important because industry stakeholders are skeptical about data from the early fishery due to the systematic biases associated with hyperstability as well as lack of consistency

Table 3
Likelihood function for fitting the statistical catch-at-age model to simulated survey and catch-at-age observations

Estimated parameters	
T3.1	$\Theta = (\hat{N}_1, \hat{R}, \hat{\tau}_1, \hat{\tau}_{2,1}, \hat{\tau}_{2,2}, \widehat{\log q})$
Residuals	
T3.2	$\xi_t = \log\left(\frac{I_t}{B_t^*}\right) - \widehat{\log q}$
T3.3	$\eta_{g,a,t} = \log p_{g,a,t} - \log u_{g,a,t} - \frac{1}{A}\sum_{a=3}^{A}[\log p_{g,a,t} - \log u_{g,a,t}]$
T3.4	$\omega_t = \log \hat{R}_t - \frac{1}{T}\sum_{t=1}^{T}\log \hat{R}_t$
Conditional maximum likelihood estimates	
T3.5	$\widehat{\log q} = \frac{1}{T}\sum_t \log\left(\frac{I_t}{B_t^*}\right)$
T3.6	$\hat{\tau}_1^2 = \frac{1}{T}\sum_{t=1}^{T}\xi_t^2$
T3.7	$\hat{\tau}_{2,g}^2 = \frac{1}{(A-3)n_g}\sum_{a=3}^{A}\sum_{t=1}^{n_g}\eta_{g,a,t}^2$
Log-likelihood	
T3.8	$\ell = T\log\hat{\tau}_1 + (A-3)\sum_{g=1}^{2}n_g\log\hat{\tau}_{2,g} + \frac{1}{2\sigma_R^2}\sum_t\omega_t^2$

between model results and personal experience during the 1970s and 1980s (e.g., biomasses in operating model fits appear too high in the 1970s). Both the data-based procedure and model-based using a catch-at-age model for stock assessments ignore data collected before 1992, and thus present opportunities to deal with these concerns. Conditioning the operating models on existing data also allowed us to maintain consistency between the historical data and the simulated future data, which is important for establishing the credibility of the management procedure approach with stakeholders. Furthermore, model-selection criteria based on these fits show that the four scenarios are essentially equally plausible (Table 4). We refer to the four scenarios as S1 (low depletion/low productivity), S2 (high depletion/low productivity), S3 (low depletion/high productivity), and S4 (high depletion/high productivity).

2.2.2. *Data-based management procedures*

Sablefish industry stakeholders requested that we examine a process for setting catch limits that "... reflects the fish on the grounds ..." perhaps by using only the most recent survey or fishery catch-per-unit-effort (CPUE). Initial trials with fishery CPUE had the expected negative consequences given that fishery catch rates appear hyperstable. Annual sablefish surveys provide a reasonable fishery-independent data source that we chose to consider for data-based procedures. Technically, most procedures for setting catch limits depend on statistics computed from fishery data, and as such can be defined as data-based management procedures. We narrow this definition for this paper, however, to include only procedures that make no assumptions about the biological dynamics of the fish stock. One such data-based procedure computes a catch limit according to the simple formula:

$$C_{T+1} = \lambda_1 C_T + (1-\lambda_1)\lambda_2 I_T^* \tag{1}$$

where C_{T+1} is the catch limit for year $T+1$, I_T^* is a statistic computed from a relative abundance survey of the stock, $0 \le \lambda_1 \le 1$ is the proportion of the projected catch limit that derives from the current one, and $\lambda_2 > 0$ is a harvest policy parameter that scales the units of the abundance index to the units of catch. In this paper, the statistic I_T^* is a 3-year moving average of the catch rate from relative abundance surveys. Eq. (1) is similar in appearance to the "hold-steady" harvest policy described and evaluated by Hilborn et al. (2002) for northeast Pacific rockfish (*Sebastes* spp.). However, Eq. (1) acts as a constant exploitation rate policy in contrast to Hilborn et al.'s formula, which is a constant escapement policy. The data-based harvest procedure is appealing because it is a direct calculation based on readily observable fishery statistics based on the strong assumption that survey catchability and selectivity are both constant over time.

The policy parameter λ_2 represents an average exploitation rate that is scaled by survey catchability; that is, $\lambda_2 = \bar{U}/q$ where $\bar{U}$ is a long-term average exploitation rate and q is survey catchability. To see why this is so, consider that Eq. (1) converges over an infinite time horizon to:

$$C_{T+1} = (1-\lambda_1)\lambda_2\sum_{i=0}^{\infty}\lambda_1^i I_{T-i}^* \tag{2}$$

If λ_1 and λ_2 are chosen so that a long-term sustainable catch is possible (i.e., the stock does not decline to extinction), then the averages of the survey index and the catch will converge to constants, which we can define as $\bar{I} = q\bar{B}$ and $\bar{C}$, respectively. The average survey index is assumed linearly proportional to the average biomass $\bar{B}$ as before. Factoring the survey average from the sum and noting that $(1-\lambda_1)\sum_{i=0}^{\infty}\lambda_1^i = 1$, we can solve for $\lambda_2 = \bar{C}/q\bar{B} = \bar{U}/q$. Thus, λ_2 is the key harvest policy parameter of the data-based procedure because it will determine the long-term average stock size and yield from the fishery.

The primary role of λ_1 is to reduce short-term fluctuations in catch by reducing the rate at which catch limits are adjusted in response to changes in the survey index. Historical values for the parameters of Eq. (1) were estimated from a multiple linear regression of annual catch limits C_T on C_{T-1} and I_{T-1}^*. The resulting values $\lambda_1 = 0.79$ and $\lambda_2 = 253$ were used to identify an upper limit on the range of data-based procedures because in preliminary simulation tests this procedure (i) always performed the worst in terms of depletion under all scenarios and (ii) leads to 40-year stock declines and fishery failure for two of the four operating model scenarios. Therefore, we examined combinations of $\lambda_1 = 0.20$, 0.50, 0.80 and $\lambda_2 = 120$, 150, 180, 210, 240 to represent both rapid to slow feedback responses to surveys and low to high average fishing mortality. It is important to note that initial depletion levels for operating model scenarios S1–S4 determine the effective exploitation rates of the data-based procedures because each initial depletion level implies a different catchability coefficient q. In particular, values of $\lambda_2 = 120$, 150, 180, 210, 240 equate to exploitation rates ranging from 0.04–0.08 for the low depletion

Table 4
Summary of operating model characteristics that define the scenarios for the management procedure simulations

Scenario	B_0	h	D_{2007}	U_{MSY}	D_{MSY}	MSY	ΔAIC
S1	152,970	0.45	0.178	0.046	0.402	2,946	2.84
S2	160,371	0.45	0.288	0.046	0.402	3,088	0.0
S3	148,032	0.65	0.203	0.083	0.346	4,340	2.18
S4	156,047	0.65	0.308	0.083	0.346	4,575	0.18

The first two variables–unfished spawning biomass (B_0) and recruitment steepness (h)–define a scenario. The remaining columns provide the spawning biomass depletion (D_{2007}) at the start of management procedure simulations, the exploitation rate at the maximum sustainable yield (U_{MSY}), spawning biomass depletion at MSY (D_{MSY}), and the MSY. The final column gives the differences in Akaike Information Criterion (ΔAIC) values for operating model fits to existing data using scenario S2 as the "best-fit" model.

scenarios (S1 and S3) and 0.03–0.06 for high depletion scenarios (S2 and S4).

2.2.3. Model-based procedures

Although most industry stakeholders are skeptical of complex stock assessment models, some nevertheless agreed that candidate procedures should include the available commercial fishery and survey age composition data, a significant amount of which has been collected with industry support through the co-management process. At a minimum, industry stakeholders sought to determine whether the extensive catch sampling and aging programs required to support these collections are worth the effort and expense. Various age-structured stock assessments have been applied to B.C. sablefish in the past, but recent assessments have relied primarily on tag-recovery data and stock abundance indices (Haist et al., 2005).

Model-based procedures represent a more elaborate way to set annual catch limits. In contrast to data-based procedures, model-based procedures attempt to estimate annual recruitment and take the uncertainty associated with the observations directly into account. However, such approaches require many assumptions about the underlying fish population dynamics and observations, as well as the random variability of these processes. In some cases, strong assumptions about fish recruitment, growth, and mortality rates can lead to systematic trends in assessment biases (Walters, 2004).

The model-based procedures we consider each involve a constant exploitation rate strategy in which a point estimate of the catch limit for year $T+1$ is computed as:

$$C_{T+1} = U^{\mathrm{ref}} B_{T+1} \quad (3)$$

where B_{T+1} is the stock biomass projected to be present at the beginning of year $T+1$ and U^{ref} is a reference exploitation rate. The stock assessment model, used to estimate and project the stock biomass, is a statistical catch-at-age model. We implemented Eq. (3) using the range of exploitation rates $U^{\mathrm{ref}} = 0.04$, 0.06, 0.08, 0.10, where the first two values bracket U_{MSY} in the low productivity scenarios and latter two bracket U_{MSY} in the high productivity scenarios. The high values also correspond approximately to target fishing mortality rates used in Alaskan sablefish assessments (Hanselman et al., 2006).

For the purpose of comparing management procedure performance against performance based on the true optimal harvest rates, we developed a “perfect-information” procedure that computed catch limits based on (i) setting $U^{\mathrm{ref}} = U_{\mathrm{MSY}}$ based on the values given in Table 4 for each scenario and (ii) setting exploitable biomass B_{T+1} equal to the true operating model biomass at the beginning of year $T+1$. These perfect-information procedures attempt to highlight the effects of assessment errors on model-based performance and also provide reference trajectories for catch and biomass that facilitates comparisons among candidate procedures.

2.2.3.1. Catch-at-age stock assessment model. The model-based management procedures employed a statistical catch-at-age model for the stock assessment step. Catch-at-age stock assessment modeling is potentially appealing for stock assessment of sablefish for several reasons. First, age-composition changes over time may contain information about temporal trends in fishing mortality and recruitment. Indeed, this particular capability is among the main reasons why so many fisheries agencies attempt to use ageing data in assessments. Second, in contrast to the data-based approach, observed changes in fishery selectivity, as measured by the annual sablefish tagging program, can be accounted for in assessments as either fixed parameters or priors. Changes in fishery (and possibly

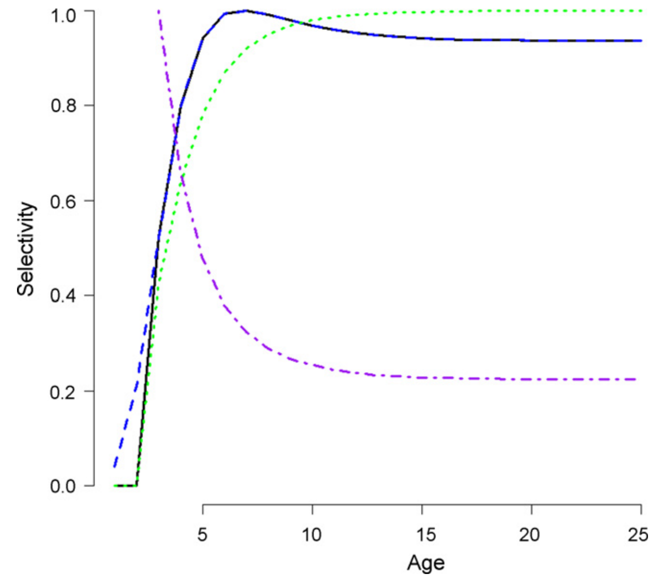

Fig. 1. Selectivity functions estimated from tag-recovery data for sablefish in the research trap survey (dashed) and commercial fisheries by trap (solid), longline (dotted), and trawl (dash-dot).

survey) selectivity can have profound influences on abundance estimates from age-structured models, especially when there are few data to distinguish between dome-shaped and asymptotic selectivity functions. An extensive industry-funded tag-recovery program for B.C. sablefish allows for direct estimation of length-based fishery selectivity from tagging (Fig. 1), and therefore potentially large improvements in age-structured assessment estimates, provided that the length-age relationship remains stationary over time. Finally, a catch-at-age assessment approach provides the ability to use shorter times-series (<20 years) of fishery-independent data.

The catch-at-age stock assessment model used in our model-based procedures considers survey relative abundance data, fishery catch-at-age, and survey catch-at-age from 1992 onwards because industry stakeholders expressed concerns about the quality of historical biomass estimates based on incomplete fishery dockside monitoring and logbook reporting during the 1970s and 1980s. Besides estimating historical biomass, the stock assessment model projects exploitable biomass one year into the future so that a catch limit in year $T+1$ can be calculated via the harvest rule (Eq. (3)). This projection first involved estimating the initial population composition $\hat{N}_{a,1}$ for ages $2 \le a \le A$ and annual age-1 recruitments $\hat{R}_t$ for years $1 \le t \le T$.

The catch-at-age stock assessment model follows operating model equations T1.2–T1.18 (omitting T1.7–T1.9 and T1.13) except for the following two substitutions. First, the initial population abundance in T1.10–T1.11 is replaced by:

$$N_{a,1} = \begin{cases} \hat{R}_1 & a = 1 \\ \hat{N}_{a,1} & 2 \le a \le A \end{cases} \quad (4)$$

Second, annual recruitment in T1.14 is replaced by

$$N_{1,t} = \begin{cases} \hat{R}_t & 1 < t \le T \\ \dfrac{1}{T}\displaystyle\sum_{i=1}^{i=T} \hat{R}_i & t = T+1 \end{cases} \quad (5)$$

where the second term represents the assumption that projected recruitment in year $T+1$ is equal to the historical average. Values for operating model gear-specific selectivity are provided to the simulated assessment and are assumed constant in the future for the purpose of this analysis. Clearly, full evaluation of sablefish management procedures should examine uncertainty in selectivity parameter values derived from tagging, further temporal changes perhaps due to density-dependent growth, and changes in selectivity as a function of sablefish abundance (e.g., changes in fishery targeting behaviour). Similarity between operating model and assessment likely leads to a "best-case" scenario for stock assessment model-based performance.

We used a penalized maximum likelihood approach for fitting the catch-at-age model to simulated survey observations of relative abundance and age proportions in fishery and survey catches (Table 3). The residual function (T3.2) for the relative abundance survey assumes a log-normal observation model of the same form as T1.20. Equations T3.5–3.6 provide the conditional maximum likelihood estimates (MLE) of log-survey catchability and survey variance, respectively. The likelihood function for the observed age proportions ($p_{g,a,t}$) is a multivariate-logistic, which we adopted because it does not over-weight age-proportion data in the manner of traditional multinomial likelihoods (Schnute and Richards, 1995). The age proportion residual calculation (T3.3) is done for trap fishery and trap survey age proportions and involves ages 3 to the plus group at age 25. Equation T3.7 gives the conditional MLE of the age proportion variances.

The final term in the total likelihood (T3.8) is the kernel of a $N(0,\sigma_R)$ prior on annual log-recruitment deviations from the estimated long-term average. Note that we provide this prior with the true recruitment standard deviation used in the operating model because the maximum likelihood approach cannot estimate process and observation error variances simultaneously. Although we could have chosen an errors-in-variables approach (Schnute and Richards, 1995), this would involve making an assumption about the ratio of process to observation errors, which adds another management procedure option and is thus beyond the scope of this paper. Similar to selectivity, future management strategy evaluation should examine whether this catch-at-age model is robust to mis-specification of the process error variance. This is especially important for sablefish because recruitment variances may be poorly estimated for species that are difficult to age, such as sablefish.

Once a catch limit is determined, it is then allocated among trap, longline, and trawl fisheries in the same proportion as occurred in 2006. This may not be realistic in the long-term due to the introduction of new regulations designed to reduce by-catch and to promote accountability for catch (Koolman et al., 2007). However, the choice is reasonable until patterns of catch distribution among gear sectors emerge from the new management regime.

2.2.4. *Performance measures*

Management procedures are typically evaluated based on three main performance categories: catch, catch variability, and conservation. The time-horizon over which performance statistics are computed is also important because trade-offs among the three main categories may change over time. Thus, performance statistics were computed for four non-overlapping time blocks consisting of 1–5, 6–10, 11–20, and 21–40 years into the future. Catch performance was summarized by the average annual catch during each period, while catch variability was summarized by the average absolute variation (AAV) in catch (Punt and Smith, 1999), i.e.:

$$\mathrm{AAV} = \frac{\sum_{t=t_1}^{t_2} |C_t - C_{t-1}|}{\sum_{t=t_1}^{t_2} C_t} \tag{6}$$

where t_1 and t_2 are, respectively, the first and last years of the time block. Stakeholders expressed concern during several management strategy evaluation workshops that inter-annual variability in catch limits greater than 15–20% would not be acceptable so we treated this level of variation as an initial AAV objective. Conservation performance was measured by the average spawning biomass depletion relative to the unfished equilibrium:

$$\bar{D} = \frac{1}{t_2 - t_1 + 1} \sum_{t=t_1}^{t_2} \frac{S_t}{B_0} \tag{7}$$

where S_t and B_0 are the operating model spawning biomass in year t (T1.18) and the unfished spawning biomass, respectively. We computed medians of the above statistics over 50 simulation replicates to summarize overall performance. These summaries are presented as median average catch, median average AAV, and median average depletion. We deemed 50 simulations to be adequate because we were mainly interested in the average trade-off relationships between catch and conservation, which are not strongly affected by the number of simulation trials beyond about 50. Further simulations will be required in the future as we examine more specific, probabilistic objectives.

3. Results

The transition from historical catch levels to those simulated from application of management procedures in the future was smooth for data-based procedures, but quite abrupt for model-based procedures. For example, all catch-at-age model-based procedures gave large immediate reductions in catch from approximately 4500 *t* in 2006 (actual outcome) to 1200–2000 *t* (simulated outcomes) in 2007. These changes from existing catch limits arise because the catch-at-age model estimates low biomass and depletion from the existing 1992–2006 data regardless of true initial depletion for the scenarios (discussed further below). It is unlikely that such changes would be acceptable to industry stakeholders given their preference for limiting annual variation in catch to less than 15–20% even though model-based procedures may ultimately perform better in the long-term. Therefore, we imposed a maximum 15% annual change constraint on both data-based and model-based catch limits over the first 5 years of the projection period. We applied this dampener to both procedure types and the perfect-information procedures to avoid confounding effects. The constraint was removed after 5 years so that catch limits were set to levels recommended by the procedures. Performance statistics for the first 5 years of the projections must therefore be interpreted with the understanding that the range of short-term outputs is often truncated by this constraint.

Although larger values of control parameter λ_1 were effective at reducing inter-annual fluctuations in catch, the main effects of this parameter were not substantial on long-term performance. For example, average differences in catch among procedures using $\lambda_1 = 0.20$, 0.80 and $\lambda_2 = 180$ ranged from approximately 11% in the short-term to less than 1% in the long-term. We therefore limited our discussion of results to data-based procedures using $\lambda_1 = 0.50$.

The maximum 15% change constraint ensured that all procedures met AAV criteria over the first 5 years. Expected catch variability as represented by the medians of AAV values were always lower than 15% over 11–20 and 21–40 year periods (Fig. 2). More aggressive harvest policies tended to cause higher fluctuations in catch for the model-based procedures. However, even the largest AAVs for any simulation for these procedures were less than 15%, except for the $U^{\mathrm{ref}} = 0.10$ model-based procedure under scenario S1. In this case, a few high AAV values occurred when the procedure caused rapid stock collapses and thus transition to zero catch at

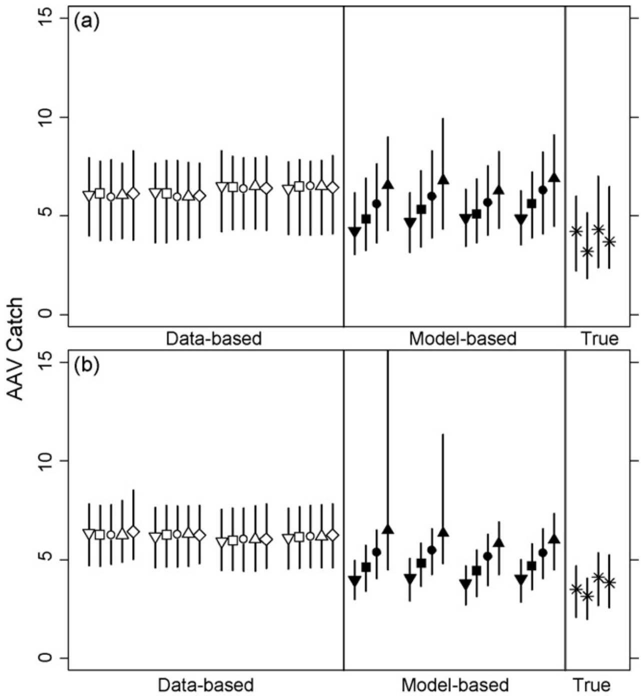

Fig. 2. Average annual variation in catch (AAV) obtained from data-based, model-based, and perfect-information (true) management procedure classes. Within each procedure class, statistics are grouped from left to right corresponding to operating model scenarios S1–S4. Within each scenario grouping, data-based procedures are ordered from left to right according to λ_2 = 120, 150, 180, 210, 240, model-based procedures according to U^{ref} = 0.04, 0.06, 0.08, 0.10, and perfect-information procedures according to U_{MSY} = 0.04, 0.04, 0.08, 0.08. Panels correspond to averaging windows (a) 11–20 years and (b) 21–40 years. Symbols represent medians, and bars indicate the 10th and 90th percentiles.

some point during the projection period. Median values of AAV for data-based procedures were less than 10% under all scenarios and projection periods (Fig. 2).

Over the range of scenarios we examined, model-based and data-based management procedures tended to trade-off catch and conservation in similar ways. This is evident by examining the trade-off between median average catch and median average depletion under each combination of management procedure and operating model scenario (Fig. 3). In the short-term (1–5 years), all trade-off relationships were relatively steep regardless of procedure type or scenario indicating that large reductions in average catch would provide only small improvements in stock depletion (Fig. 3a). This reflects the fact that the 2007 spawning biomasses in the operating models are below the level at which maximum sustainable yield is estimated to be achieved for all scenarios and recent catches remove most of the surplus production available for stock growth. The 15% constraint on catch changes further restricted potential increases in stock size even under the most conservative procedures. In the medium- and long-term, however, trade-off relationships became less severe mainly because most procedures caused increases in stock size under most scenarios (Fig. 3b–d). In general, data-based procedures (open symbols in Fig. 3) obtained lower median average catch for a given level of stock depletion during the first 20 years of the simulations (Fig. 3a–c). Lower catches taken by data-based procedures in the short-term tended to promote greater stock growth in the long-term (21–40 years), which ultimately caused the data-based trade-off contours to move higher along the depletion axis rather than the catch axis (Fig. 3c–d). In the long-term, data-based procedures obtained similar or greater catches at higher depletion levels, whereas model-based procedures (solid symbols in Fig. 3) obtained greater catches at lower depletion levels (Fig. 3d).

3.1. Scenario 1—low productivity/low initial depletion

All procedures caused further stock declines over the first 10 years of the projection period under this scenario (Fig. 3a–b). Such declines result because (i) recruitments in the years prior to implementation of management procedures (e.g., 2000–2006) were below the long-term average and catches during these years were not adjusted to compensate, and (ii) the maximum 15% annual change constraint maintains catch levels near the 2000–2006 average, which as mentioned above, appear to be greater than recent average surplus production. As a consequence both median average catch and median average depletion are considerably lower than 2007 levels by years 6–10. During this period, differences between data- and model-based procedures begin to arise, particularly for the more aggressive policies λ_2 = 210, 240 and U^{ref} = 0.08, 0.10, respectively (Fig. 3b, S1). The effects of persistent over-fishing become evident by 21–40 years when these aggressive policies obtain lower catches than less aggressive policies (Fig. 3d, S1) that promoted stock growth during earlier periods. Procedures λ_2 = 120 and U^{ref} = 0.04 closely tracked catch and conservation performance of the perfect-information procedure with U_{MSY} = 0.04 (Fig. 3a–d, S1), although both procedures obtained slightly lower depletion levels by year 21–40.

3.2. Scenario 2—low productivity/high initial depletion

All procedures again result in stock declines over the first 10 years for this scenario (Fig. 3a–b). Only the λ_2 = 120, 150 and U^{ref} = 0.04 procedures promoted stock recovery to the 2007 level by years 11–20. However, all data-based procedures except λ_2 = 240 allowed the stock to recover beyond the 2007 level by years 21–40. Similar to scenario S1, model-based procedures with U^{ref} = 0.08, 0.10 caused further declines in both median average depletion and catch. For this scenario, performance of λ_2 = 150 and U^{ref} = 0.04 closely tracked the perfect-information procedure with U_{MSY} = 0.04.

3.3. Scenarios 3 and 4—high productivity

Similar to the low productivity scenarios, the stock declined during the first 5 years for high productivity scenarios regardless of initial conditions (Fig. 3a). However, short-term declines were minor and were generally followed by stock increases within the 6–10 year period in both scenarios S3 and S4, and under all procedures (Fig. 3b). Model-based procedures tended to increase both catch and depletion simultaneously between the 6–10 and 21–40 year periods, while data-based procedures increased depletion slightly more than catch. As expected, performance of the model-based procedure with U^{ref} = 0.08 closely tracked the perfect-information procedure with U_{MSY} = 0.08 under both scenarios S3 and S4. The λ_2 = 180 data-based procedure provided similar median average catch at slightly higher depletion to the perfect-information procedure under scenario S3, while all data-based procedures obtained lower catch, but higher depletion than the perfect-information procedure under scenario S4.

The relatively small differences between model-based procedures with U^{ref} = 0.04, 0.08 and perfect-information procedures were caused primarily by catch-at-age model estimation and projection errors. Biomass estimation errors showed a characteristic retrospective bias pattern in which the stock was over-estimated during declines and under-estimated during stock increases (Fig. 4). Such a pattern arises because the procedures estimated the long-

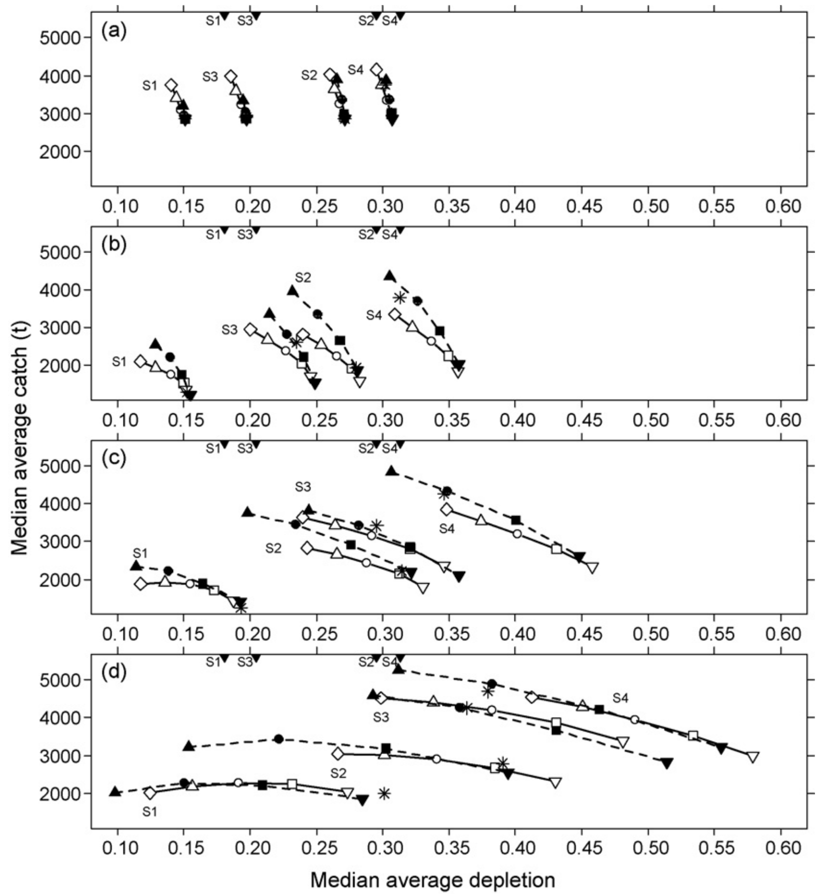

Fig. 3. Trade-offs between median average annual catch and median average depletion for model-based (dashed lines, solid symbols), data-based (solid lines, open symbols), and perfect information (asterisks) management procedure classes for operating model scenarios S1–S4. For each class and scenario combination, procedures are ordered from left to right as for Fig. 2. Panels represent the averaging windows (a) 1–5 years, (b) 6–10 years, (c) 11–20 years and (d) 21–40 years. The inverted triangles and labels along the top of each panel correspond to the spawning biomass depletion at the start of the first year of the projection period.

term average recruitment in the stock assessments as opposed to estimating a functional stock-recruitment relationship. Note that the catch-at-age stock assessment model also under-estimated biomass for the first few years under the high depletion scenarios S3 and S4. This occurs because the existing data (i.e., 1992–2006) are more consistent with low rather than high 2007 depletion. Estimation biases diminish to the point that they depend only on stock trajectory rather than initial conditions under scenarios S3 and S4.

3.4. Reduction of the management procedure set

The ultimate goal of management strategy evaluation is to identify a single management procedure that performs adequately with respect to the objectives across all plausible scenarios. Early in the process, however, the goal may simply be to reduce the set of candidate procedures to a manageable few. Although this task is usually accomplished by comparing performance against fixed objectives (e.g., MSY or B_{MSY}), it is also useful to compare procedure performance against a reference trajectory, especially where the initial conditions may be far from the objectives. In such cases, the transient approach to long-term objectives may be critical to adoption of a management procedure. In our analysis, we used the perfect-information catch and depletion trajectories as "ideal" transient paths for future fishery development (Walters, 1998) under each scenario. Simulation trajectory summaries for spawning biomass depletion using one data-based ($\lambda_2 = 150$) and one model-based procedure ($U^{ref} = 0.06$) are shown in Fig. 5. Corresponding trajectory summaries of catch are shown in Fig. 6. These particular procedures lead to long-term stock increases under all scenarios, while obtaining among the highest catches of those procedures promoting stock growth. Both procedures also provide similar conservation performance in the short-term and similar catch performance in the long-term to scenario-specific perfect-information procedures (Table 5). As expected, neither procedure clearly dominates over all scenarios and time periods. An exception occurs for scenario S2, where the $\lambda_2 = 150$ data-based procedure remains within 4% of the perfect-information procedure in both the short- and long-term; this arises because $\lambda_2 = 150$ provides an average exploitation rate very close to U_{MSY} for this scenario. Interestingly, long-term catch performance of the $\lambda_2 = 150$ procedure in the more productive scenario S3 is also closer to perfect-information than $U^{ref} = 0.06$ even though the latter is a closer approximation to U_{MSY}. This arises, in part, because the stock assessment model used in $U^{ref} = 0.06$ under-estimates biomass during rapid stock growth over the last 20 years of the projection period (e.g., Fig. 4, S3).

4. Discussion

Fishery co-management policies must provide a realistic mechanism for stakeholder involvement in decision-making. We used the management strategy evaluation approach to actively engage

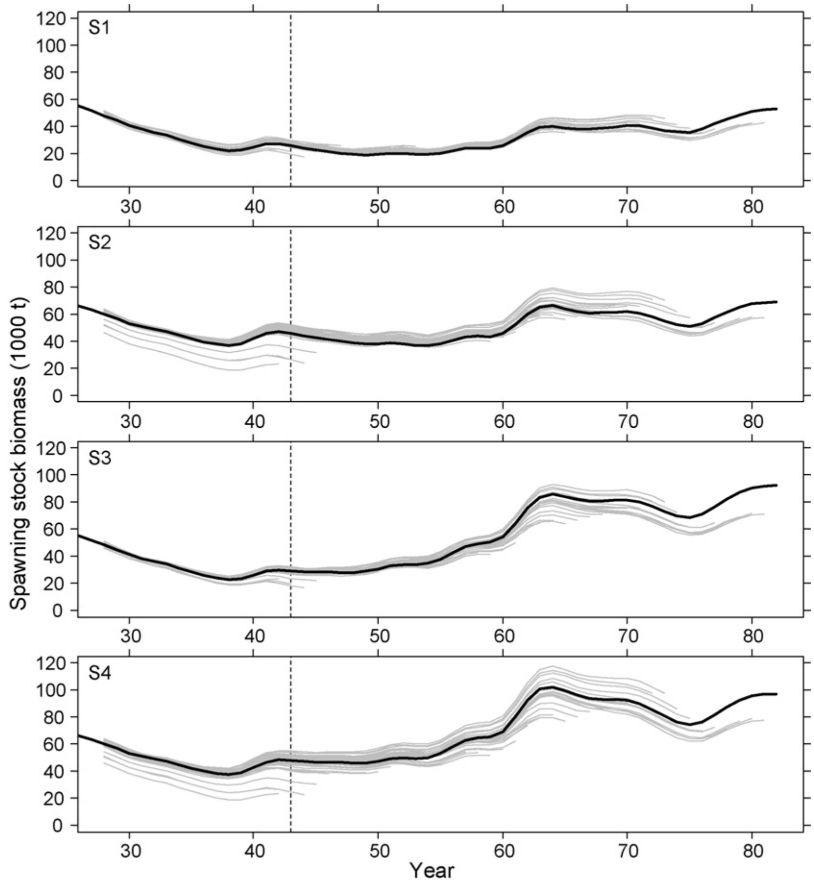

Fig. 4. Single simulation replicate example of retrospective biomass estimation for the $U^{ref} = 0.06$ model-based management procedure under scenarios S1–S4. Thick black lines show the true operating model spawning biomass for each scenario and gray lines indicate annual stock assessment estimates of the historical spawning biomass.

stakeholders in the development of fisheries management policies. Stakeholders provided compelling reasons for evaluating practical data-based methods for determining catch limits, as well as more elaborate methods based on modern catch-at-age analysis that use industry-supported fishery monitoring programs. Simulation testing of candidate management procedures indicated that both approaches could meet inter-annual catch variability criteria. The data-based procedures provided stable or increasing stock sizes in the long-term under most circumstances mainly because these procedures took lower catches in the short-term in response to recent declines in survey indices of abundance. Model-based procedures, on the other hand, tended to be more efficient in terms of short-term catch because these procedures estimated biomass each year and did not use previous catch limits directly as the data-based procedures did.

The parameter λ_2 of the data-based procedures defines a constant exploitation rate policy in much the same way as model-based policies use exploitation rates U^{ref}. Therefore, both procedure types are expected to have similar long-term expectations under equivalent exploitation rate policies. Our analyses show that this is indeed the case and that the two procedure classes trade-off catch and conservation in similar ways during the transient approach to these long-term expectations. Although this is not particularly surprising, it does point to the need for greater emphasis on setting better target harvest policies rather than what is perhaps the current trend toward developing more sophisticated stock assessment models that attempt to better estimate stock biomass (Cotter et al., 2004; Kell et al., 1999).

Both data-based and model-based procedure classes attempt to implement an exploitation rate policy by extracting signals of biomass change from noisy observations and adjusting catch limits accordingly. However, the signal processing step is done in very different ways. Data-based procedures employed a simple exponentially weighted moving average smoother, while the model-based procedure used a more efficient statistical catch-at-age fitting method. When both policy exploitation rates were set equal to U_{MSY} for a scenario, they closely followed the perfect-information procedure suggesting that smoothing or estimation had little impact on overall performance. For exploitation rates above or below U_{MSY}, persistent biomass estimation errors degraded model-based performance relative to perfect-information, although these effects were not substantial in this study probably because the simulated assessment model closely mirrored the operating model. In a more realistic situation in which model parameters such as selectivity, natural mortality, and recruitment variance are unknown, interactions between exploitation rate and biomass errors may cause serious problems (NRC, 1998; Walters, 2004). Data-based procedures, on the other hand, made no strong assumptions about the underlying stock dynamics (aside from the modeling process involved in obtaining λ_2) so that declining or increasing catches were the consistent result of relatively short-term changes in the survey moving average.

The ability of data-based procedures to meet catch and conservation objectives was sensitive to the scenario chosen because the stock biomass present when the procedure is implemented defines the harvest policy exploitation rate $\bar{U} = q\lambda_2$. Whereas model-based approaches treat catchability as a nuisance parameter, data-based

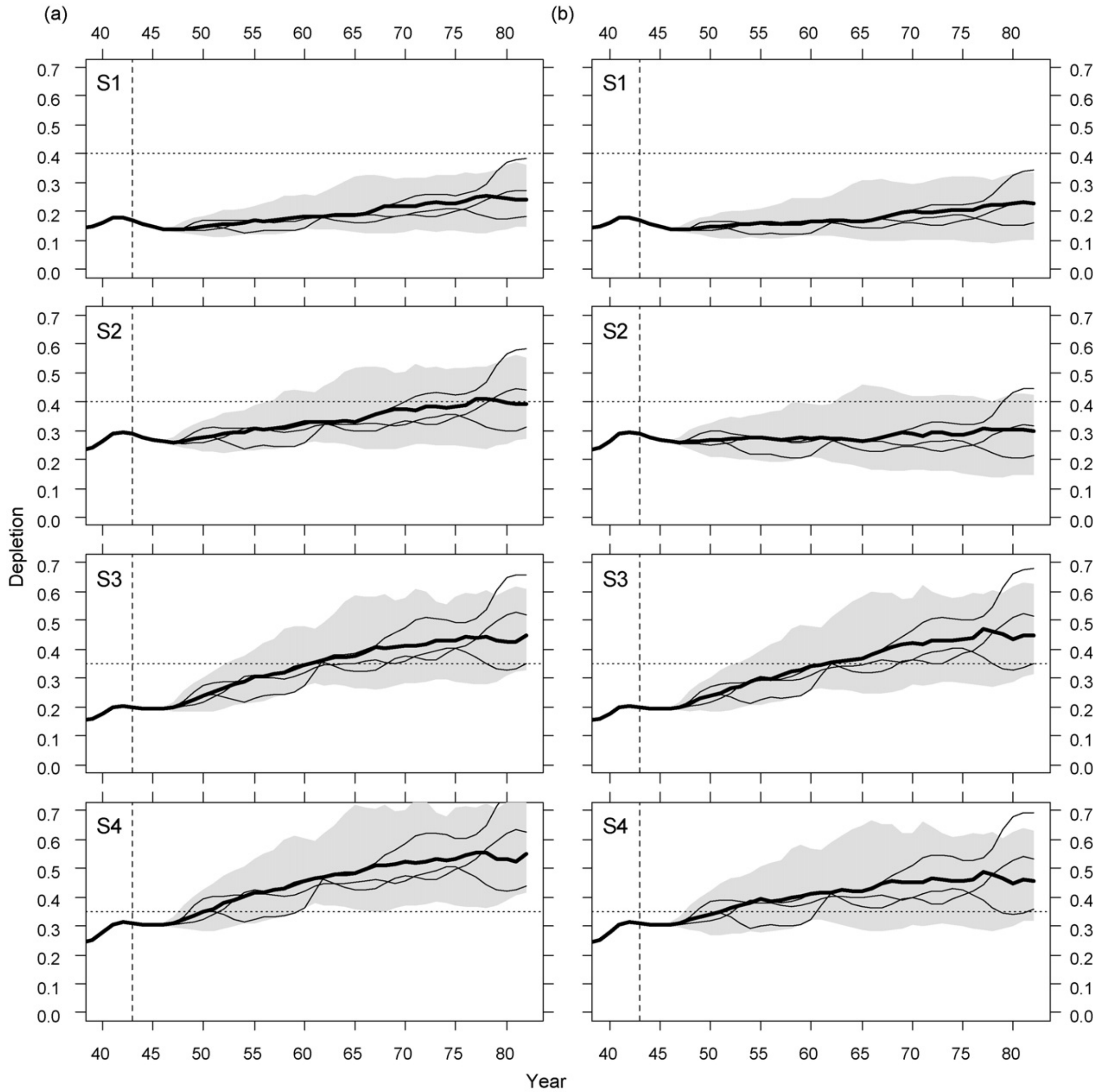

Fig. 5. Trajectories of spawning biomass depletion under (a) data-based $\lambda_2 = 150$ and (b) model-based $U^{\mathrm{ref}} = 0.06$ management procedures. Panels are arranged vertically corresponding to scenarios S1–S4, respectively. Vertical dashed lines indicate 2007 and dotted horizontal lines indicate depletion levels corresponding to B_{MSY}. Trajectories are summarized by the median (thick black line), three individual simulation replicates (thin black lines), and 5th to 95th percentiles (shaded area).

procedures incorporate catchability as an integral parameter in the procedure. Although this is perhaps a major weakness of data-based approaches, it seems that for a well-designed fishery-independent survey, this may be acceptable. On the other hand, changes in survey catchability and selectivity are not unrealistic given experiences in many fisheries (Harley et al., 2001; Parma, 2002). The potential problem with data-based procedures is that undetected increases (decreases) in catchability or selectivity are translated directly into unwanted increases (decreases) in exploitation rate on the stock. Similarly, shifts in survey selectivity toward younger ages will cause increases in catch limits independent of actual changes in the stock. Thus, continued stock assessment modeling, preferably in the form of ongoing management strategy evaluations, would be necessary to support data-based management procedures in TAC-managed fisheries. Such a combined approach is likely to appeal to both stakeholders, who benefit from a simple method for computing annual TACs, and to scientists and managers, who are provided the assurance that such procedures incorporate the best available information and are tested for robustness in closed-loop simulations.

Catch-at-age model-based procedures have clear advantages over simple data-based methods for setting catch limits when survey catchability and selectivity are subject to change—provided that other model assumptions are not strongly violated. First, catch-at-age models treat survey catchability as a nuisance parameter that simply scales the average of the surveys to the average population biomass. Patterns of exploitation are inferred partly from changes in age composition independent of fishery surveys and partly from long-term trends in the surveys. Therefore, considerable time would pass before short-term changes in survey

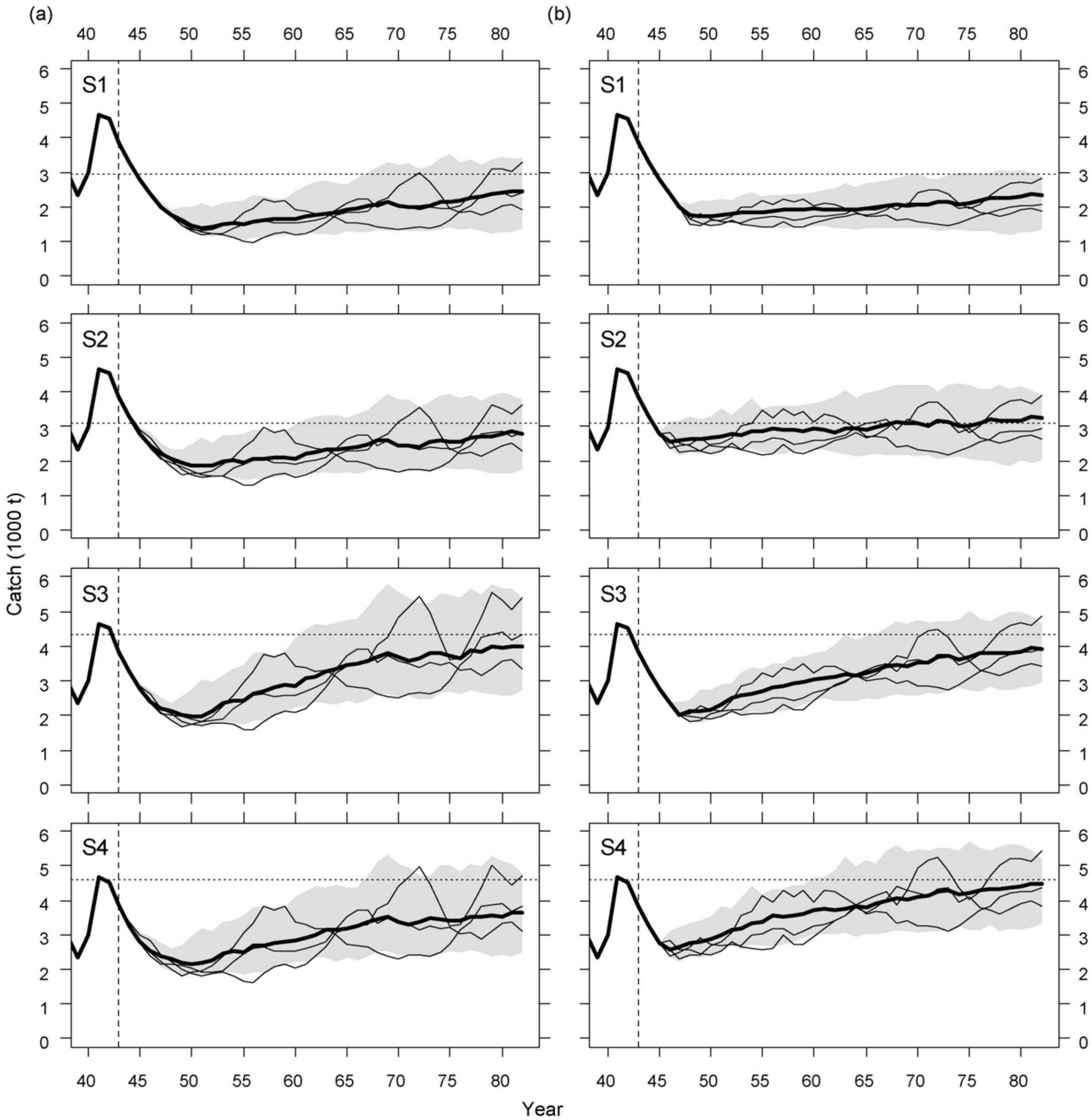

Fig. 6. Trajectories of annual catch under (a) data-based $\lambda_2 = 150$ and (b) model-based $U^{\mathrm{ref}} = 0.06$ management procedures. Panels are arranged vertically corresponding to scenarios S1–S4, respectively. Vertical dashed lines indicate 2007 and dotted horizontal lines indicate MSY. Trajectories are summarized by the median (thick black line), three individual simulation replicates (thin black lines), and 5th to 95th percentiles (shaded area).

Table 5
Performance of data-based $\lambda_2 = 150$ and model-based $U^{\mathrm{ref}} = 0.06$ procedures relative to the perfect-information procedures for each scenario

Scenario	Procedure	6–10 years			21–40 years		
		$\bar{C}$	AAV	$\bar{D}$	$\bar{C}$	AAV	$\bar{D}$
S1	$\lambda_2 = 150$	**0.18**	**−0.42**	−0.02	0.13	0.79	**−0.23**
	$U^{\mathrm{ref}} = 0.06$	0.35	−0.56	−0.02	**0.11**	**0.32**	−0.31
S2	$\lambda_2 = 150$	**−0.01**	**1.33**	**−0.01**	**−0.04**	0.98	**−0.02**
	$U^{\mathrm{ref}} = 0.06$	0.37	1.54	−0.04	0.14	**0.53**	−0.22
S3	$\lambda_2 = 150$	−0.21	**0.49**	0.02	**−0.09**	0.45	**0.18**
	$U^{\mathrm{ref}} = 0.06$	**−0.15**	0.56	0.02	−0.14	**0.08**	0.19
S4	$\lambda_2 = 150$	−0.41	**1.38**	0.12	−0.25	0.61	0.41
	$U^{\mathrm{ref}} = 0.06$	**−0.23**	1.41	**0.10**	**−0.10**	**0.23**	**0.22**

Statistics are relative differences from perfect-information results computed for short-term (years 6–10) and long-term (years 21–40) periods. Values shown in bold font are closest to the perfect-information result for the scenario.

catchability affected catch limits. Furthermore, it is also possible that retrospective patterns would appear in the catch-at-age predictions (Mohn, 1999), thus potentially providing early warning that survey catchability was potentially changing. Second, catch-at-age models can use independent estimates of survey or fishery selectivity to account for potential changes in these processes over time (Myers and Hoenig, 1997). However, catch-at-age models are not without problems. For example, very slow changes in survey catchability and selectivity can possibly go undetected for long periods leading to persistent estimation biases (Walters, 2004). For example, our simulated catch-at-age assessments showed retrospective biases that ultimately caused over-fishing during stock declines and under-fishing during stock increases. Such patterns occurred despite several strong similarities between the operating model and assessment model such as constant catchability, known selectivity and recruitment variance, and similar model structures.

It therefore follows that management strategy evaluations should routinely simulate the actual estimators to be used in management procedures rather than assuming that assessment errors will be unrelated to the stock trajectory.

Our results for data-based fishery management procedures generally reflect the experiences of other simulation studies on the performance of empirical management procedures. Where data- and model-based procedures have actually been compared against the same operating models, data-based management procedures generally give slightly lower average annual catch and higher inter-annual catch variability than model-based procedures (Rademeyer et al., 2007). Our results for data-based procedures agreed in general with Hilborn et al. (2002) who showed that a constant escapement form of data-based rule for long-lived west coast rockfish (*Sebastes* spp.) gave comparable results to model-based procedures (although they did not simulate an actual stock assessment). Their case, like ours, pointed to the difficulty of specifying exploitation rates or data-based targets for TAC-managed fisheries. In particular, long-term declines were likely where initial choices for exploitation rates were based on biased biomass estimates. Data-based procedures have been developed and adopted for Namibian hake and South African west coast rock lobster (Rademeyer et al., 2007) and New Zealand rock lobster (Bentley et al., 2005), although procedures for hake and rock lobster in South Africa were both considered interim in the absence of better quality data for future model-based procedures.

We assumed the sablefish stock is closed to immigration and emigration, even though we know of strong evidence to the contrary. Long-term tagging studies show considerable long-range movement of some tagged sablefish, and also remarkable site fidelity particularly for adult fish (Beamish and McFarlane, 1988; Kimura et al., 1998). Redistribution of sablefish throughout the northeast Pacific appears to mainly be driven by both movement of adults (Kimura et al., 1998) and by emigration of juveniles from nearshore waters and inlets. Although the stock structure and spatial dynamics of sablefish are not particularly clear at this time, the information that exists should be used to construct plausible scenarios for further testing of management procedures because spatial processes may have profound effects on both data-based and model-based outcomes (Punt, 2003). At-sea discarding is also a potentially serious process impacting the performance of management procedures. Estimates of sablefish discard rates in directed fisheries and as by-catch were not available for this study. However, stakeholders, managers, and scientists involved with the sablefish fishery all agree that evaluation of discarding impacts on management procedure performance is a high priority.

Feedback advice to stakeholders as a result of this initial management strategy evaluation process has included the need to consider further precautionary modifications to all management procedure classes. For example, none of the candidates we presented here actually comply with Canada's national policy on the precautionary approach to fisheries management, which requires that harvest control rules be divided into critical, cautious, and healthy stock status zones (DFO, 2006). In the critical zone, stock conservation considerations prevail and management actions must be consistent with stock recovery. In the cautious zone, stock conservation and economic considerations are balanced to reflect the stock trajectory and position in the zone. For instance, if the stock biomass is increasing and closer to the healthy zone, economic considerations may receive greater emphasis. When stocks are assessed as healthy, economic considerations prevail provided that they are consistent with long-term conservation objectives. The actions prescribed in each zone can be reflected in both data-based and model-based management procedures by adjusting the exploitation rate parameters (i.e., λ_2 or U^{ref}) in pre-defined ways depending upon the zone. The management strategy evaluation approach we describe is well-suited to defining both the stock status reference points that delineate the zones and the actions within each zone that best meet national and stakeholder objectives.

This work represents an initial step toward a more extensive management strategy evaluation process that addresses some of the key uncertainties identified above as well as a wider range of candidate management procedures that address both stakeholder and government requirements. B.C. sablefish stakeholders have had an important role in contributing to this process by helping to initiate projects examining stock structure and migration, changes in the stock distribution with changes in abundance, fishery and survey selectivity, impacts of discarding, and the biology of sablefish in mainland B.C. inlets. In addition to their scientific value, such programs have helped the management strategy evaluation process by introducing stakeholders to notions of alternative scenarios, projections, and sensitivity analysis, which are important concepts in management strategy evaluation. Ultimately, development of data-based management procedures may prove to be the critical link that connects stakeholders to precautionary fisheries management policy. Such procedures are now integral to the management strategy evaluation process for sablefish because stakeholders can easily calculate catch limits into the near future under alternative scenarios about the sablefish stock and scientific surveys.

5. Conclusion

The choice of implementing a specific fishery management procedure involves a compromise among possible candidates that may perform differently under equally plausible, yet contrasting scenarios (i.e., operating models). Stakeholder involvement in the management strategy evaluation process helped to develop practical data-based and model-based fishery management procedures that address particular industry concerns. Thus, industry stakeholders are in a better position to make the necessary compromises and trade-offs compared to situations where complex management procedures are defined outside the co-management arena. Furthermore, iterative refinement and testing of these procedures against known uncertainties provides a formal mechanism for fishery co-management in which stakeholders have a central role in decision-making, essentially deciding on the process by which catch limit decisions will be made. Developing management procedures in this way also addresses national precautionary fishery management policy directives by requiring precise statements of how harvests are to be adjusted in response to departures from operational objectives.

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