Management strategies for spasmodic stocks: a Canadian Atlantic redfish fishery case study

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Management strategies for spasmodic stocks: a Canadian Atlantic redfish fishery case study

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Abstract

There exist few recommendations for managing stocks with spasmodic recruitment, despite such stocks being not uncommon. Management procedures (MPs), developed for two rockfishes (Sebastes mentella and S. fasciatus) in Eastern Canada, are recommended for setting catch limits during high and low periods of abundance. A well-designed fishery-independent trawl survey is essential to provide advance warning of strong recruitment events and project future recruitment. Under an “inventory management” strategy, a more appropriate aim in spasmodic stocks may be to maximize the number of years with "good catches", instead of maximizing total catches as is traditionally considered in management strategy evaluation (MSE). Following a spasmodic recruitment event, an empirical harvest control rule based on larger fish delays the harvest of large cohorts by a few years, targets more commercially valuable fish sizes and reduces the risk of growth overfishing. Capped MPs produced longer periods of large catches than uncapped MPs. MPs allowed for low harvests during periods of low abundance, thus avoiding unnecessary hardship in the industry. MPs evaluated here could be good candidates for other stocks with similar or less extreme recruitment variability.
Introduction

Developing an understanding of population dynamics and identifying management approaches consistent with this understanding have been among the chief aims of fisheries management science. To facilitate the achievement of these aims, fish stocks have been categorized as steady-state, cyclical, irregular, and spasmodic (Caddy and Gulland 1983; Hilborn and Walters 1992). Spasmodically recruiting stocks (SRS) are typically the most variable of fish stocks; these stocks have been commonly characterized by infrequent and irregularly occurring episodes of strong recruitment, followed by long periods of weak recruitment. This recruitment pattern is known as “spasmodic recruitment” (Caddy and Gulland 1983; Spencer and Collie 1997; Caddy and Agnew 2005); spasmodic recruitment has been observed in a wide range of species groups including rockfishes, hakes, sardines, and mackerels (Templeman 1976; Spencer and Collie 1997; Roel and De Oliveira 2007; Needle 2016; He and Field 2017).

Spasmodically recruiting stocks can be quite productive once the strong cohort(s) enter the fishery because these cohorts are often much larger than the long-term average year-class strength. For example, for the Western horse mackerel a strong cohort was about 18 times higher than the long-term average (Roel and De Oliveira 2007). Especially in long-lived species, once a large recruitment occurs, the larger harvests can be sustained for several years, as has occurred for the Atlantic redfish fishery (Templeman 1976; CAFSAC 1984a). To maintain a viable fishery for as many years as possible before the next strong cohort appears, careful management planning is required (Caddy and Gulland 1983). By imposing high harvest rates on strong cohorts, the long-term biomass potential of these strong cohorts can be negatively impacted. Moreover, shortly following a strong year class, there is a predominance of small pre-recruit fish with little or no market value which are sometimes unavoidably captured by commercial fishing gear. Without appropriate regulation of fishing activities, strong cohorts can thus be at risk of growth overfishing, but management approaches also need to be designed to avoid high discarding rates.

A challenging aspect of SRS is that the steady-state assumptions of classical fishery models cannot be applied to these stocks (Spencer and Collie 1997). In particular, key recruitment parameters that
can influence the potential long-term yield, such as recruitment compensation (Hilborn and Walters 1992), are difficult to quantify even when long time series of stock-recruit data are available. For example, providing that recruitment can be estimated, management strategies that trade off short-term yields of smaller fish for delayed larger yields of larger fish could be reliably quantified for the short-term when a strong cohort appeared. However, due to the high year-class strength variability and their irregularity, the potential long-term benefits and trade-offs of different management approaches for such stocks will inevitably remain highly uncertain.

Management approaches for spasmodically recruiting stocks (SRS)

Despite long-standing awareness of the behaviours of SRS, there exist few guidelines for their management. Early studies suggested limiting the number of permanent participants to endure periods of low recruitment, and enabling flexible access to the fishery for periods of high productivity (e.g., additional licenses; Caddy and Gulland 1983). Two recent studies have focused on stocks with spasmodic recruitment and tested alternative management strategies using simulation (Roel and De Oliveira 2007; Needle 2016). However, the scope of these studies did not include formulating more general guidelines for managing SRS. For example, Needle (2016) used a rectilinear harvest control rule broadly used in the management of European fish stocks. This approach by itself however may be inadequate due to the lack of any additional measures to facilitate fleet adaptability to periods of building, high and low abundance characteristic of SRS. Roel and De Oliveira (2007) tested a series of harvest control rules (HCRs); an empirical vs. model-based approach. These authors concluded that either a fixed harvest rate or a constant catch strategy would maximize catch without increasing biological risks for Western horse mackerel. However, due to the paucity of fishery-independent data, a constant catch strategy was preferred.

While an empirical approach (i.e., a "slope strategy" in Roel and De Oliveira (2007)) is probably flexible enough for SRS, the traditional approach of maximizing the average annual catch throughout the "outburst" may be insensitive to the limits of realizable industrial capacity. Typically, during the "outburst", HCRs are designed to maximize annual yields. Without considering the potential for growth
overfishing and also the rapid liquidation of biomass during periods of abundance, this approach may also
generate hardship for the fishery in the long term. A better approach may be to develop harvest strategies
and rules that set harvest caps at agreed levels so that the annual yields produced by the strong cohorts
can be within the realizable capacities for industrial processing and marketing and spread over an
extended number of years. This management strategy represents an “inventory management” strategy
(Carl Walters com. pers.). Under this approach, fishing opportunities would be increased in a pre-agreed-upon way as stock biomass builds with the recruitment of a strong cohort. Management strategies would
also need to avoid high discarding rates during the “outburst”.

It is widely agreed that the management strategy evaluation (MSE) framework is the most
appropriate way to evaluate alternative strategies for achieving fishery and conservation objectives under
multiple sources of uncertainty (Punt et al. 2016). To evaluate and identify management strategies that
could be generally appropriate for stocks with highly spasmodic recruitment, we applied the MSE
framework to the redfish fishery in the Gulf of Saint Lawrence and Laurentian Channel (GSL-LCH), i.e.,
Management Units 1 and 2, on the East coast of Canada (Fig. 1).

*Atlantic redfish fishery as a case study*

The redfish fishery is composed of two species, Deepwater Redfish (*Sebastes mentella*) and
Acadian Redfish (*S. fasciatus*). These species have exhibited spasmodic recruitment dynamics (Valentin
et al. 2015). Similarly to other pairs of *Sebastes* species that are caught together, these two species are
indistinguishable based on external morphologic features (Ni 1981). Redfish are also slow-growing and
long-lived species (>65 years) (Campana et al. 1990). These species have sustained the Canadian Atlantic
redfish fishery for many decades (Fig. 2). The 1956, 1958, and 1970 cohorts were the first strong ones
recorded (Sévigny et al. 2007). After that, the last strong recruitment occurred in the early 1980s which
consisted mostly of *S. mentella*, but with a significant amount of *S. fasciatus* (Valentin et al. 2015). These
strong cohorts were subjected to high harvest rates. As time progressed, no further strong cohorts
appeared and the redfish stocks became less abundant in the fishing grounds, reaching very low levels for
the last two decades (i.e., 1995-2017). As a result, in 2010, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) designated *S. mentella* as endangered and *S. fasciatus* as threatened in both management Units (COSEWIC 2010). Despite of the very low spawning stock biomass (SSB), after >25 years of low recruitment, the Fisheries and Oceans Canada (DFO) trawl survey recorded strong cohorts belonging to the 2011, 2012, and 2013 year-classes. Recent DFO surveys also showed that these cohorts are growing in biomass (DFO. 2018a). These fish were expected to recruit significantly to the fishery from 2019 to 2020, which could lead to a rapid increase of SSB.

We present the results of the MSE for the redfish stocks to assess the trade-off between maximizing the long-term harvest under an inventory management strategy and keeping the risks of growth and recruitment overfishing acceptably low under multiple sources of uncertainty. The MSE needed to consider that the two target species cannot be distinguished on the commercial fishing grounds and have different productivities. Less abundant cohorts, as predicted for *S. fasciatus*, could be depleted faster and could limit fishing due to conservation objectives. Besides these issues, uncertainties about the population and fishery dynamics also needed to be considered. These are related to past and future trends in recruitment and productivity (e.g., recruitment compensation, natural mortality, and growth), data-related due to species identification and discarding, as well as changes in vulnerability and implementation error (see *Selection of uncertainties*).

**Materials and methods**

**Framework**

The first step in the MSE framework was to define the conservation and fishery objectives (Punt et al. 2016). These objectives were established via consultation with the redfish working group (RWG) (i.e., industry stakeholders, DFO fishery managers, non-government organizations, and scientists). The agreed-upon objectives had to be translated in terms of quantifiable measures known as performance metrics...
PMs enable the quantification of how well management procedures (MPs) meet the different management and fishery objectives. Another crucial aspect of the MSE approach is the identification of key sources of uncertainty (related to the biology of the stock, fishery, data collection, and analyses to determine a catch limit, CL) where the MPs are to be tested for robustness (Punt et al. 2016). The RWG identified five main axes of uncertainty (see “Selection of uncertainties”).

Conservation and fishery objectives and performance metrics

For conservation objectives 1 and 2, an empirical method was used to establish the reference points (RPs) for each species (Table A1). In particular, the RPs were based on the average simulated SSB of the operating models (OMs) for the periods when the biomass was high (i.e., 1984-1990 and 1984-1992 for *S. mentella* and *S. fasciatus*, respectively). These were considered proxies for the SSB<sub>msy</sub>, the SSB at maximum sustainable yield (MSY). The limit reference point (LRP) and upper stock reference (USR) were set at 40% and 80% for the reference periods mentioned above, respectively. The conservation objective 3, which used the harvest rate at MSY ($U_{msy}$) in future years, required the re-computation of $U_{msy}$ for each simulation. This was because the parameters values of the OMs were based on draws from their respective posterior distribution (see Operating model description). Also, when the OMs had parameters that changed in a future year and that determined $U_{msy}$ (e.g., natural mortality, vulnerability, and growth parameters), the $U_{msy}$ was recomputed in that year.

Many fishery PMs were based on the inputs of industry stakeholders. Of particular interest was maximizing catches of large fish (e.g., >27 cm), the average annual catch, and avoiding the small fish protocol (SFP) (Objectives 4-7; Table A1). The SFP is triggered in a sub-area of one of the management Units (i.e., Unit 1 or Unit 2) when fish <22cm represent more than 15% of the catch. This produces a temporary closure of that sub-area. For example, the industry was interested in evaluating the average retained catch (e.g., objective 6a) for the period 2028-2037 (10 years) and 2028-2057 (30 years) and the average number of years where the CLs are ≥40kt in the years 2028-2057 (e.g., objective 6c). Similarly, because the strong cohorts of 2011-2013 had not yet recruited to the fishery, the industry stakeholders
were interested in the length-based metrics for the stocks for the next five years (e.g., 2018-2022). For example, objective 4 examined the number of years when the SFP was triggered in the next five years. Passing criteria were suggested for each of the conservation PMs. It was not possible to agree upon and justify a passing criteria for objective 6 only (Maximize duration of high annual catch). As strong cohorts 2011-2013 are expected to fully recruit to the fishery by 2028, most of the fishery PMs were evaluated after this year.

Management procedures

The MPs included an empirical (or data-based) HCR for each species to prescribe the CL which follows a “target-based” approach; i.e., the CL is adjusted up or down each year in relation to the extent to which the recent average of an index of abundance is above or below a target level (Fig. 3) (Rademeyer and Butterworth 2011). We used the DFO biomass survey conducted since 1984 in Unit 1 across the northern GSL using a bottom trawl (DFO 2018a), which generates an index for each species. An additional industry biomass survey (performed in Unit 2) was not used in the MP because it represented a much shorter time series (only beginning in 2000). The empirical HCR was based on the MP used for Western component Pollock (Rademeyer and Butterworth 2011). A full mathematical description of the HCR is in Table A5. In order to dampen the effects of extreme survey values, the HCR uses the ratio of the geometric mean of the trailing 3 years of the survey biomass index to a reference period (i.e., 1984-2017). The DFO biomass survey index was transformed into a “fishery index” by summing only the lengths of the population of most interest to the industry (i.e., ≥30 and 29 cm for S. mentella and S. fasciatus, respectively). Similarly, the simulated “fishery index” for the projections also included these lengths and stochastic components (Table A3.A20). The HCR parameters of the empirical MP (Table A4) were tuned using the historical “fishery index” so that the prescribed CLs would be similar to the observed retained catch biomass during periods of low abundance but dropping to zero if the “fishery index” reached a threshold (Fig. 3). Besides, the parameter that controls the rate of harvest (i.e. $\Omega^h$, Table A4) in the HCR
was also tuned so that the median values generated for the OM would produce no more than ~75\% of the $U_{\text{msy}}$ for both species in the 40-year simulation horizon.

Besides the HCR, the MP included six other components that were applied depending on the MP (see Table S4-S5 for the complete list of MPs). 1) A ramp cap may be applied, with an initial and maximum CL with starting and ending years (Fig. 4). This component generated two kinds of MPs, i.e., “capped” and “uncapped”. Note that the caps were applied to the two species combined because the fishery does not distinguish the between species (Table A5.B4-B5). After the cap implementation, the HCR added a “catch split implementation” which takes into account the error in species identification in the DFO surveys. The values corresponded to the proportion of $S. \text{fasciatus}$ in the catch retained and reported in 2000-2017 (Table A5.B6). After the “catch split implementation”, we applied a multiplier of 1.1 which takes into account “implementation error” due to catch biomass not retained (e.g., lost from nets, spoiled, or caught as unreported bycatch in other fisheries) to be consistent with the recent fishing practices (Table A5.B7). 2) A status quo period may be applied before the year in which the HCR starts (Fig. 4). During the status quo, it was assumed that the catch taken (i.e., not derived from the HCR) is either the average catch retained in 2015-2017 (i.e., 2,838 tons) or some pre-specified catch of interest to managers and industry members. 3) A maximum allowed CL or percentage of change in CL may be applied between years. 4) The SFP (starting in 2018) with CL set to zero may be applied if it is triggered fishery-wide in a given year (Table A5.B9). 5) A minimum cap may be applied. 6) A multiplier to the parameter that controls the rate of harvest may be applied but only in “uncapped” MPs (Table A5.B1). A suit of MPs was proposed and tested in the MSE framework (Table S4-S5). See Fig. 5 for a summary of the steps for calculating the realized CL under MP1.

**Conditioning the operating model**

The OM describes the most plausible dynamics of the current stocks and fisheries and is used to project the stocks into the future to assess the performance of the MPs (Rademeyer et al. 2007). We used the available historical data to obtain estimates of the model parameters for the OMs. The historical
information included the abundance indices and length compositions from the DFO survey (1984-2017) and the biannual industry survey (2000-2016) carried out in Units 1 and 2, respectively. The DFO and industry surveys generate information (e.g., indices and length compositions) by species based on the number of anal fin soft rays and also genetic analysis (Gascon 2003). The fishery information included the catch biomass retained (1960-2017) and the commercial length compositions from Units 1 and 2 (1984-2017). This information was split by species using the composition estimates from the DFO survey. More information about the survey sampling design and methods for splitting the commercial catches can be found in DFO (2018a).

Operating model description (base case)

In order to condition the OMs on data, an age-structured population dynamics model was used. The OMs for the two species included the same sets of equations. Due to the paucity of data, the fisheries in Units 1 and 2 were aggregated into a single fishery in the OM for each species. Similarly, the fishing fleet data in Units 1 and 2 also were combined (e.g., catches and retained length compositions). The survey information was not aggregated. A Bayesian statistical catch-at-length stock reduction analysis (SRA) framework was applied for parameter estimation (McAllister and Ianelli 1997). Parameters were estimated separately for each of the OMs for the two species (see Table A2, and Table S1-2-3 for a full description of the equations and parameters values for the base case). The model was coded in ADMB (Fournier et al. 2012). The parameters estimated included, for example, the long-term average of unfished recruitment, the recruitment compensation ratio, age-1 recruitment deviates from 1952 to 2017, constants of proportionality for the survey biomass, and selectivity parameters (see Table S1.C1). A Beverton-Holt stock-recruit model was used to predict the recruitment of age-1 fish (Table S1.C3). An informative prior distribution formulated in terms of the Beverton-Holt steepness stock-recruit parameter was applied (Table S2.D2; Table S3) (Forrest et al. 2010). We included informative priors for recruitment deviates in years where early reports on redfish have shown evidence of strong cohorts (CAFSAC 1984b; Sévigny et al. 2007) and the historical length composition could not inform the cohort strengths. An empirical-Bayes
A type meta-analysis of stock-recruit data was carried out to quantify the mean value across *Sebastes* stocks for large recruitment deviates (DFO 2018b). Informative priors on recruitment deviates were applied for three years (Table S2.D5) in the initial abundance at age 3 (Table S1.C2) and two early years (i.e., 1956 and 1958) (Table S1.C3).

The observed retained catch-at-length data were predicted using a fraction retained at age probability represented by a logistic function (Table S1.C4). Discarding data was not available; therefore, the age-at-50% vulnerability for the fishery was calculated by subtracting a time-varying offset parameter (Fig. S2), as an input vector to the estimated age-at-50% retained (Table S1.C5-C6). Two time-blocks were used, i.e., prior to 1994, and then from 1994 onward (Table S1.C5) (Fig. S1). In order to calculate the catch biomass killed, the observed catch biomass retained was multiplied by a time-varying input vector representing the ratio of catch biomass killed to catch biomass retained in each historic year (Table S1.C11) (Fig. S3). The values were formulated by an interview-based study on historic discarding practices (Duplisea 2018). It was assumed that in 1950, the initial abundance was close to the average unfished equilibrium state (Table S1.C2). A linear extrapolation was made to extend the catch time series back to 1951 from the early 1960s (Figs. 6-7). The likelihood functions applied were conventional ones for statistical catch-at-length stock assessment models (Table S2.D6-7) (McAllister and Ianelli 1997). The values that were applied for the coefficient of variation (CV) for survey indices, effective sample sizes for length composition likelihoods, and prior density functions can be found in Table A2 and S3.

In order to characterize parameter uncertainty in the OMs (Punt et al. 2016) in the conditioning and projections, we sampled the parameters from a multivariate normal (MVN) distribution using the posterior mode and posterior variance-covariance matrix obtained from the Hessian matrix. When the OM structure (e.g., a different vulnerability function) or input values (e.g., life history parameters) was modified, a new MVN was obtained for that OM (i.e., the modified model was again fitted to the data). One thousand draws were obtained for the key parameters for the OMs (i.e., indicated by a “^” symbol in
Table A2). The parameter values drawn were applied in the OMs from 1951 to 2017. If the modelled stock abundance dropped to zero in any year up to 2017, the draw was discarded.

Selection of uncertainties

The main sources of uncertainty were over how to characterize past and future recruitment and the productivity of *S. mentella* and *S. fasciatus*, species composition of historic catches, discarding, changes in vulnerability, and life history parameters (a full list of the OMs is in Table A6). The set of OMs was grouped into “core” and “stress-test” OMs. Here, the core OMs represent the most credible hypotheses for how the fishery and stocks have behaved or will behave and are credible from the standpoint of scientists and stakeholders. The stress-test OMs are plausible alternatives for fishery and stock behaviors but have less scientific credibility than the core models. Thus, in terms of acceptability (e.g., stakeholders) and support (data), the core and stress-test have different weights. Following the common practices in the MSE framework, a base case OM representing the most credible set of specifications for an OM was formulated (Rademeyer et al. 2007). To test the sensitivity of results to alternative inputted values and prior values (e.g., historical strong cohorts), a set of “sensitivity-test” OMs were also evaluated. We reported AIC values to provide an indication of how well the OM fitted the data (Tables S6-7); the AIC however was not (and should not) used to rank the plausibility of OMs. AIC measures only statistical performance; in contrast, working group knowledge of the fishery was taken to be more appropriate for judging plausibility (Guthery et al. 2005).

Present and future recruitment

It was important to obtain credible historic recruitment estimates for the redfish stocks and to develop a credible model to generate future recruitment. Future recruitment deviate sequences were simulated as they were observed in the past (Table A6.OM1). This was done using a nonparametric bootstrap which sampled from the set of historical recruitment deviate estimates (DFO 2018b). However, fishery stakeholders were interested in having MPs tested against a variety of scenarios for future
recruitment patterns (Table A6.OM5-6). We also developed a conditional parametric bootstrap protocol (Table A6.OM9).

In the case of long-lived species such as redfish, incomplete time-series of fishery-independent/dependent data limit the estimation of past cohort strengths (in terms of magnitude and frequency). Past strong redfish cohorts have been identified based on literature (e.g., 1952, 1980; CAFSAC 1984b). Unfortunately, earlier length compositions do not exist to estimate these cohort strengths. We examined the effect of including these early strong cohorts as informative priors as was described in the “Operating model description” section. We also examined their impact using alternative (higher and lower) prior means for a strong recruitment deviate for these strong cohorts (Table A6.OM19-20).

Species identification

Stock productivity determines the recovery of depleted stocks but also the magnitude of sustainable catches (Hilborn and Walters 1992). In the case of S. mentella and S. fasciatus, these species are difficult to tell apart and are not distinguished in commercial fishery data (DFO 2018a). This situation has important implications for the reconstruction of historical abundance trends, estimates of cohort strengths, and stock productivity. We addressed this issue applying different catch split hypotheses to the data (i.e., retained catch biomass and length compositions; Table A6.OM10-11).

The length compositions (e.g., from the DFO bottom trawl surveys) contain juveniles of S. fasciatus that do not contribute to the adult population to the Units 1 or 2 but instead to the Grand Banks stock (3LNO) (Valentin et al. 2015). As a result, the bottom trawl survey length compositions have produced spurious estimates of abundance and harvest rates in the model fitting (e.g., Duplisea et al. 2016). We applied a simple method to remove these cohorts from the bottom trawl survey length compositions (see Fig. S4 for details).
The credibility of the estimates of cohort strength and stock productivity are also compromised
due to past (in the 1980s and 90s) and present practices of discarding small fish (Duplisea 2018). As a
result, the commercial length compositions are quite uninformative in terms of cohort strengths,
vulnerability, and fishing mortality. Several OMs were developed to address this issue. For example, to
explore the hypothesis that discarding was higher or lower compared with the base case, we applied two
stress-test OMs, one with a 50% increase and another with a 50% decrease for the multiplier to calculate
the catch killed biomass. (Table A6.OM14). Following suggestions from industry participants, we
examined, using two other OMs, the hypotheses that high discarding rates may occur for 2018-2020
because the fishery cannot avoid the small fish from the 2011-2013 cohorts. We increased the
“implementation error” from 1.1 to 2 in the HCR but we reverted it back to 1.1 from 2021-2057 (Table
A5.B7). In two stress test OMs, we applied this scenario under the base case (Table A6.OM1) and under
high levels of natural mortality (Table A6.OM24).

Change in vulnerability

The commercial trawl gear has changed considerably over the past several decades (DFO 2018a).
We used two time-blocks to describe the changes in vulnerability. This hypothesis was represented in the
OM1 (base case). In a stress test OM, we examined a further hypothesis where the future fishery
vulnerability for 2017-2021 reverts back to the fishery vulnerability estimated for the first time-block
(Table A6.OM22). This represents the hypothesis that fishery vulnerability changes when strong cohorts
are not completely recruited (i.e., small fish are discarded) in contrast the vulnerability patterns that favor
the capture of larger fish in periods of low adult abundance and low recruitment.

Changes in fishing gear/vessels can further affect the vulnerability at age, which can be
asymptotic or dome-shaped depending on whether older fish are more or less vulnerable to the fishing
gear (Table A6.OM2). Stress test OM2 had dome-shaped vulnerability for both the fishery and the bottom
trawl survey. Changes in discarding practices also affect fishery vulnerability. In two other stress test
OMs, we used different input values for the offset vector to calculate the age-at-50% vulnerability (Table A6.OM13).

**Life history parameters**

Stakeholders were concerned about the potential density-dependent effect on individual growth and natural mortality rates ($M$) from the high predicted abundance for the 2011-2013 cohorts. Under this hypothesis, the strong cohorts 2011-2013 would generate lower biomass than anticipated due to low growth, cannibalism, lack of food, and intra- or inter-specific competition. In one stress test OM, the future low growth rates (e.g., for the next 20 years) were implemented by reducing the asymptotic length ($L_\infty$) to 2/3 of the base case value (Table A2) while retaining the other von Bertalanffy growth parameters (Table A6.OM4). For $M$, we examined several hypotheses as follows: in a core OM, the value for $M$ is higher for smaller fish (Table A6.OM8); in some additional stress test OMs, $M$ increases during periods with strong cohorts (e.g., for the next 20 years; Table A6.OM3), both historic and future $M$ are lower or higher than the values for $M$ specified under the base case (Table A6.OM15-16), and $M$ increases for the next 20 years and there are high discarding rates for 2018-2020 (Table A6.OM24).

Finally, in some other stress test OMs, we examined alternative values for the recruitment compensation parameter with respect to the base case (Table A6.OM17-18). When the parameter values were modified (e.g., increasing or decreasing $M$), the SRA model was re-run to condition the OM.

**Results**

**Operating model fit and diagnostics**

For both species, the base case OMs showed high harvest rates in the 1990s, as high as about 45-50%. After 1993, the harvest rates decreased gradually to be lower than 5% in the most recent years (Figs. 6-7). SSB showed a marked decline to the mid-1970s, an increase, because of the biomass generated from the strong cohorts in 1980, and a marked decline caused by the high harvest rates in the 1990s, likely targeting the 1980 cohorts. SSB remained low for both species until the last few years when the SSB
started to increase because of maturing fish from 2011-2013 cohorts. Survey indices for both species followed the estimated SSB pattern. Survey indices showed an increase prior to that in the SSB because the surveys included juvenile fish (particularly for DFO survey in Unit 1), which have increased in abundance in the surveyed area (DFO 2018a). Recruitment estimates showed spasmodic dynamics; a few strong large positive recruitment deviates were interrupted by many years of relatively small positive and negative recruitment deviates. The strong cohorts from 2011-2013 for S. mentella were the largest ever estimated for the redfish stocks. Also, the strong cohorts for both species occurred at the same time (Figs. 6-7; S10-11).

For both species, the survey length compositions fitted as well or better than in previous statistical catch-length models (e.g., Duplisea 2018). The OMs predicted very closely the length compositions when they belonged to large cohorts (i.e., the strong cohorts 2011-2013) and were able to predict the strong 1980 cohorts when they started to appear in the survey length compositions in 1984 (Figs. S4-5; S7-8). The OMs were not able to predict the retained length compositions for S. mentella very well, but did so for S. fasciatus (Figs. S6-S9). For both species, there were no concerning residual patterns for fits to the survey indices for either Units 1 or 2 (Figs. S10-11). A retrospective analysis showed no evidence of retrospective patterns for the time series of SSB, recruitment, and harvest rates (Fig. S12).

**Operating model outputs**

Estimates of parameters of the 15 OMs are listed in Tables S6-7 for S. mentella and S. fasciatus. $U_{\text{msy}}$ estimates for S. mentella were lower than those for S. fasciatus while the unfished spawning biomass was higher for S. mentella. The spawning biomass depletion in 2017 ($D_{2017}$) was higher for S. fasciatus than S. mentella. For the base case, $U_{\text{msy}}$, MSY, and $D_{2017}$ were 0.04, 25kt, 0.27 for S. mentella and 0.09, 34kt, and 0.15 for S. fasciatus.

**Management procedure performance**

**Conservation objectives (1-3)**
For both species, all 22 MPs passed conservation objectives under the base case OM1 (Table A1). When catch splits were biased towards S. mentella (OM10), the uncapped MPs 20, 25, 26, 27, and 45 failed to meet conservation objective 3 (e.g., maintain harvest rates below $U_{\text{msy}}$ for both species) for S. fasciatus. Also, we found that when the discarding rates were higher than those in the base case (OM14), two capped MPs (18 and 22) and the same uncapped MPs mentioned for OM10 failed to meet conservation objective 3 (Fig. S13).

Fishery objectives for avoiding SFP (4-5)

In the short term (five years), all the MPs failed to meet the fishery objectives of avoiding the 22 (i.e., SFP) and 25 cm fish (i.e., objective 4 and 5), under all of the OMs. The starting year of the HCR therefore had no impact on avoiding the SFP. Considering the 40 years of projections and the core OMs, only OM11 where the catch split was biased to S. fasciatus avoided the SFP for 22 cm fish. However, for OM1 the capped MPs 1, 4, 14, 24, 34, and 40 passed the PM criteria while for OM11 all the MPs passed (Fig. S14). All the MPs that passed the different PM criteria under all OMs had caps between 14, 5, and 40kt (Table S4). Under the stress-test OMs, most of the uncapped MPs did not pass the SFP except under the OMs that assumed no strong cohorts for the next 40 years (OM5), had lower discarding rates (OM14), and lower $M$ (OM15). However, some capped MPs (e.g., 1, 4, 14, 24, 34, and 40) passed the SFP under different levels of recruitment compensation (OM17-18) and higher discarding rates for 2018-2020 (OM23) (Fig. S14).

Fishery objectives for maximizing catch biomass and fish size (6-7)

Uncapped MPs had better performance in terms of retained catch biomass than capped MPs, but uncapped MPs showed more variability in annual catches (Fig. 8). Uncapped MPs showed lower variability for the first ten years after that the 2011-2013 cohorts were assumed entirely recruited (i.e., for the simulation years 10-20) compared with 10-40 years of simulations. During the simulation years 10-20, the retained catch biomass was ~75kt for uncapped MPs. For capped MPs, the retained catch biomass
increased with the MP cap (Table S4). All the capped MPs with 40kt caps reached the cap independently of the OMs except in OMs that assumed double $M$ or age-specific $M$ (OM3 and 8), dome-shaped vulnerability (OM2), and low growth (OM4) (Figs. 8 and S15). The highest retained catch biomass with capped MPs was under MP-16 with a cap of 80kt with ~60-65kt (depending on the core OM). The uncapped MPs with HCR-generated catch limits reduced by a factor 80% (i.e., MP12, 29, 43, and 44) produced slightly inferior catch biomass than those that did not have reduced catch limits. The OM with recruitment simulated with parametric bootstrapping (OM9) had a higher retained catch biomass in the 10-40 year horizon than in the 10-20 year horizon, especially with uncapped MPs. OMs with lower $M$ and higher recruitment compensation had a larger effect on the performance of uncapped MPs, generating ~85-90kt in the 10-20 year horizon. Note that this PM was not affected by the starting cap because of the horizon used to calculate the retained catch biomass.

Uncapped MPs with the SFP (such that CL = 0 if SFP is violated) and a maximum increase/decrease in catch limit of 5kt (i.e., MP18 and 22) had the highest proportions of simulations (i.e., >0.9) with CLs ≥40kt across all of the core OMs (except OM8) and most of the stress OMs (Fig. S16). Similarly, for the core OMs, uncapped MPs with maximum increase/decrease of 15% but without the SFP (i.e., MP14 and 34) also scored high (i.e., >0.8) except for the OM with no strong cohorts for the next 40 years (OM5). MP14 and 34 also scored high under the OM15 and 18 (Fig. S16).

MP18 and 22, on average, are expected to produce 30 years with CLs ≥40kt for most of the OMs (except for OMs 2, 3, 4, and 8). All the capped MPs under the OM9 showed ≥28 years with CLs ≥40kt (Fig. 9). Similarly, many MPs under OM15 and 18 also scored high (e.g., ≥26 years) (Fig. S17). There was a trade-off between the number of years with CLs ≥40kt and the average catch retained; uncapped MPs had high catches but with fewer years with CLs >40kt, especially when 100% of the slope for the HCR was applied (e.g., MP45). MP18 and 22 maximized the number of years with CLs ≥40kt and the average catch under most of OMs. All the MPs scored high under the OM9 (Figs. S18-19).
All MPs performed well (e.g., $\geq 0.9$) for the proportion of years with CL $\geq 4\text{kt}$ (i.e., landings in 2017) except for MP18 and 22 under OM2, 3, 4, 8, and 24.

All of the MPs failed to pass the criterion in which the proportion of years where large fish (27cm) comprise $>80\%$ of catch in the 5 and 40-year horizons. Some uncapped MPs had $\geq 0.5$ of the years with large fish under OM11 and 15.

*Fishery objectives for stable catches (8)*

All the MPs passed the PM criteria of changes in CL between years less than 15\%, but uncapped MPs scored lower than capped MPs (Fig. S20). Similarly, all MPs passed criteria (\leq 15\%) for the average annual variation between years (AAV), but uncapped MPs scored lower than capped MPs in the 10-40 and 10-20 year horizons.

**Discussion**

In this paper we examined key challenges to managing fisheries for stocks that show spasmodic recruitment. This recruitment pattern is found in *Sebastes* species and many other fishes such as hakes, sardines, capelins, mackerels, and anchovies (de Moor et al. 2011; Needle 2016; DFO 2018c). Strongly spasmodic recruitment can make fishery management particularly challenging. Under spasmodic recruitment and particularly when recruitment shows only a weak dependence on the parental stock, the classical assumptions of steady-state models with conventional patterns in recruitment variability (e.g., where the largest cohorts are no more than a few times the long-term average and the HCRs considered are often directly based on MSY-based reference points) cannot be applied here. In fact, there is growing evidence that this situation is quite common. For example, when examining patterns of variability in fish stocks, Spencer and Collie (1997) found that only three of the 30 stocks conformed to steady-state behaviour. Recent studies have shown that many fish stocks around the world demonstrate non-steady state behaviours, e.g., non-stationary stock-recruitment functions (Dorner et al. 2008; de Moor et al. 2011; Vert-pre et al. 2013; Minto et al. 2014; Britten et al. 2016).
The two redfish stocks examined in this paper are extreme examples of spasmodic recruiters; the MSE carried out on them provides information on how to sustainably manage fisheries for such stocks. We note characteristics of models and management strategies that are most promising for developing sustainable management of these stocks:

1. Management harvest control rules should be triggered only by biomass increase for sizes of fully mature individuals larger than just post-recruit sizes.
2. A simple empirical stock size indicator (e.g., bottom trawl survey biomass) has advantages both for modeling and for communication with stakeholders as the main indicator of stock status.
3. Capped harvest control rules provide stability in catch and can extend the number of years with large harvests following spasmodic recruitment events.
4. A range of recruitment modeling methods including parametric and non-parametric resampling ensures robustness of the management strategy to the main perceived uncertainty.

1) Harvest control rules responsive to the biomass of large-sized fish

We identified several desirable properties of an empirical approach to managing SRS such as redfish: First, a “target-based” empirical approach was computationally and conceptually more tractable than a model-based approach (i.e., simulation of a stock assessment in the closed-loop). OMs fitted to data already captured the strong 2011-2013 cohorts, which were projected and propagated with uncertainty in the OMs. Thus, the problem was reduced to finding MPs that best extended the harvest benefits of these strong cohorts (e.g., number of years with large catches and larger fish landed) while avoiding high discard rates. Second, the parameters of an empirical MP do not depend on MSY reference points. The HCR parameters were tuned in the base case OM using $U_{msy}$ estimates to avoid high harvest rates for the less abundant species (i.e., $S. fasciatus$). Third, a significant fraction of the catches will be small fish when strong cohorts are recruiting to the fishery. Applying an empirical MP based on a total
biomass index (e.g., the Unit 1 DFO bottom trawl survey) is problematic because it will be highly
"reactive" to the juvenile fish biomass. However, this survey index was modified to a "fishery index" by
including only larger fish (i.e., >30 and >29 cm for *S. mentella* and *S. fasciatus*, respectively) which
dampened the variability in CLs by reducing the effect of the strong cohorts. This "fishery index" also has
other benefits. It only "reacts" to changes in the surveyed biomass of large fish. More importantly, it
allows harvest of the strong cohorts a few years later when they have a larger commercial size, and
reduces the risk of growth overfishing. Basing the "fishery index" on large-sized fish of both species,
avoided high harvest rates on the stock component with weaker recruitment. In earlier simulations, when
the total biomass index was based on smaller fish, a strong cohort led to high CLs and high harvest rates
on the weaker stock.

2) *Use an empirical index to formulate harvest control rules*

For short-lived species that show a clear boom and bust recruitment dynamics (e.g., sardines and
anchovies) empirical MPs have been applied (e.g., de Moor et al., 2011; 2016). Model-based MPs also
have been applied with a rectilinear or empirical HCR (e.g., Roel and De Oliveira 2007; Hicks et al.,
2016; Needle 2016). Some species of gadoids, for example, as Pacific hake and haddock, appear to have
low average recruitment with occasional large year-classes (e.g., Needle 2016). However, it is not clear
whether a rectilinear HCR will outperform an empirical MP when the fishery targets strong cohorts; e.g.,
it should be sufficiently reactive when the biomass increases rapidly and should reduce fishing mortality
when the strong cohorts are exhausted. In the case of sardine and anchovy, de Moor et al., (2011; 2016)
showed that including a “two-tier” system with an empirical MP allowed large catch limits during
“booms” in abundance and decreased catch limits once the biomass started to decline, but still produced
reasonable stability for the industry and low risk of depletion.

In the case of spasmodically recruiting species caught together and that cannot be told apart on
the fishing grounds, the control parameters of the “target-based” MP had an important role in determining
the CLs and avoiding overfishing. *S. fasciatus* has high recruitment compensation but the strong 2011-
2013 cohorts were weaker for this species. Therefore, the risk of rapidly exhausting these cohorts is higher for *S. fasciatus* as preliminary simulations predicted. Tuning the control parameters so that the harvest rates were on average lower than the estimated $U_{msy}$ levels avoided the risk of overfishing for the less abundant species (e.g., *S. fasciatus*). The tuned parameters also prevented the SSB from falling below the LRP at the end of the 40-yr projection.

3) Capped harvest control rules

The ramp cap period in the MPs was a critical aspect identified to regulate CLs following large recruitment events because strong cohorts can generate a rapid increase in biomass mainly from immature fish. Note that most of the MPs evaluated in the literature impose a single cap (e.g., Hicks et al., 2016). A set of capped and uncapped MPs with a suite of rules and constraints (e.g., changes in rate of harvest in the HCR, maximum change in CL between years) were developed to extend the duration of high projected biomass and reduce the discarding of small fish (e.g., SFP, starting years for the MP, status quo, minimum caps). The suite of MPs exposed the trade-off of “inventory planning”. The aim was to maximize the number of years with "good catches" instead of maximizing total catches as is traditionally done in MSE. The notion of “good catches” (e.g., 40kt) was an industry objective that was rooted in its processing capacity and the market for redfish fillets. We showed that depending on the MP, stable catches of about 20-60kt could be caught in the next two decades while still avoiding overfishing. Uncapped MPs produced the largest catches, but capped MPs produced the longest periods of large catches (e.g., >40kt) and more stable catches than uncapped MPs. Capped MPs lowered the risk of growth overfishing but the use of a large fish “fishery index” and tuning the HCR parameters also helped. Stakeholders mentioned that the current market for redfish is limited (i.e., mostly sold for bait). The suite of capped MPs limited the future catches taken so that the industry could build the fishing processing capacity required and develop better access to markets for higher-valued redfish products.

HCRs within MPs commonly evaluated for the steady-state stocks are not suitable for SRS. It is interesting to note that if a threshold-type HCR had been applied (e.g., the 40-10 control rule used by the
Pacific Fishery Management Council), the Unit 2 fishery would have been closed also for more than 25
years (the Unit 1 fishery was closed in 1995, but a quota of 2kt was subsequently allowed to maintain an
index fishery). This hypothetical situation would have had caused even greater economic hardship.
Maintaining low catches under low stock sizes indicates that harvest rates fixed at low levels may be
suitable for SRS under periods of low abundance. Recent literature has shown that threshold-type HCR
produces unnecessary hardship to the industry in mixed-stock fisheries and when recruitment parameters
are nonstationary. In this situation, a better alternative is fixed harvest rates rules that allow fishing even
when the stock is low (e.g., Hawkshaw and Walters 2015; Walters et al. 2019).

Similarly, fixed escapement HCRs may not be suitable because they might not allow for
maximizing the number of years with large catches (instead it will maximize total catch, Hilborn and
Walters 1992) and it may produce undesirably high harvest rates. For example, before the current
management Units were established in the mid-1990s, the harvest rates were over 10%, peaking around
50% in 1993. Probably, the early scientific advice given did not consider the spasmodic dynamics of the
redfish stocks or was based on the rules of thumb, “take what you can when [abundance is] high” and
“fish [for] something else in bad periods”, that were suggested for these kinds of stocks at the time
(Caddy and Gulland 1983). Either way, as a result, the CLs applied to the last strong cohorts in the 1980s
exhausted these cohorts quickly.

Typical MPs used for steady-state stocks seem too “rigid” to accommodate the spasmodic stock
dynamics. For example, the reference harvest rate (or $F_{\text{target}}$) in the fixed harvest rate rules (or in
threshold-type rules) have little meaning when the stock changes rapidly from very low to very high
abundance owing to a single very abundant year class. Moreover, as these HCRs depend on estimates of
stock size (e.g., a model-based approach), the prescribed CLs would be too risky from a conservation
perspective, simply because the biomass estimates may be biased and imprecise (Hilborn and Walters
1992). A “target-based” HCR, when carefully tuned, is both more flexible and safer. For example, this
HCR still prescribes a CL when the stock is at low stock sizes with catches similar to those when the
stock was low, thus, accommodating the possibility that a long run of weak year-classes could occur in
the future for these stocks. However, the HCR can also protect the stocks when the “fishery index” drops
below a threshold (de Moor et al. 2011; de Moor and Butterworth 2016). When the stock is in an
abundant period, the capped MPs will extend the years with large catches. Our MPs followed the
empirical approach of the “boom” and “bust” dynamics of de Moor et al. (2011) and de Moor and
Butterworth (2016) but with a more extreme “booms” and more extended “bust” periods than is typical.
Roel and De Oliveira (2007) also used two-slope strategies (i.e., an empirical HCR) that included a
minimum CL to ensure that fishery closures are minimized.

4) Test harvest strategies against a variety of recruitment assumptions

For SRS, the projections of the strong cohorts require a careful study of the historical recruitment
pattern and reliable estimates of the cohort strengths. Long periods of weak recruitment have been
documented for S. mentella in other areas (Templeman 1976; Sévigny et al. 2007; Planque et al. 2012)
and in others Sebastes (e.g., Wetzel et al. 2017). According to our OMs, the estimated strong cohorts for
2011-2013 are the largest observed (for both species) in GSL-LCH. Preliminary projections showed that
including more than four strong cohorts (here defined as recruitment deviates greater than 2.5 in
logarithm space) in the projection horizon produced unrealistically high (combined) biomasses. We also
found that the typical approach of generating future recruitment from a stock-recruitment relationship
(i.e., expected recruitment plus error) or even selected ranges of years of historical recruitments (e.g.,
Punt and Methot 2005) produced future stock sizes which were not at all representative of what was seen
previously in these stocks.

We found that redfish stocks showed five key features in their recruitment dynamics. First, the
strong recruitments events for S. mentella and S. fasciatus were highly correlated: as one species had a
strong recruitment, so did the other (Figs. S10-11). Second, cohort strength was consistently higher for S.
mentella (e.g., the magnitude of the recruitment deviate was higher). The last strong cohort in 1980 also
contained both species, but S. mentella dominated according to genetics and clustering analysis (Valentin
et al. 2015). Third, we found that strong cohorts were produced in a pulse of no fewer than eight years apart following the previous strong cohort (same as Planque et al. 2012). Fourth, the large recruitment deviates occurred within no more than three consecutive years (or two according to Templeman 1976) but only two large deviates could occur within such sequences. Finally, for the period where cohort strengths could be estimated (i.e., 44 years), there were no more than four strong cohorts (Figs. S10-11). As such, we developed both parametric and non-parametric bootstrap approaches to mimicking these recruitment dynamics. Both methods had a similar performance in term of conservation and fishery objectives, though the non-parametric approach was preferred by the RWG; however, it has proven difficult to replicate redfish recruitment dynamics using a parametric approach. The parametric method seems to be too optimistic regarding the proportion of simulations and numbers of years with CLs >40kt across OMs and MPs. The non-parametric approach, although perhaps more realistic, is more complex to implement and is limited by the time series length of observations which cover less than about two generation times for the stock. It may, however, better replicate the particular recruitment dynamics of these stocks.

We used a Beverton-Holt stock-recruitment function to simulate future recruitment. It can be argued that a Ricker stock-recruitment function may be a better option, as it takes into account density-dependent effects at large stock sizes (e.g., Roel and De Olivera 2007). Also, instead of sampling recruitment deviates, a better option could be sampling directly from the historical recruitment (e.g., Punt and Methot 2005) to avoid using a stock-recruitment function. For example, the strong redfish cohorts can be produced at very low levels of SSB and after a long period of weak recruitment events. We used a stock-recruitment function mainly because DFO required MSY reference points to avoid overfishing. However, both concerns were addressed in OM5 and OM6 where no strong cohorts were simulated for the 40-years and the first 20 years of projections, respectively. Therefore, these OMs simulated 1) a strong density-dependent effect due to a large SSB (as in a Ricker function) and 2) no strong recruitments despite including a stock-recruitment function. Our MPs still were robust to these two OMs (both passed conservation objectives). Even if the non-parametric approach was not able to fully mimic the redfish
dynamics (e.g., OM1), the better performing MPs still avoided overfishing in this OM. Other studies with
SRS have also simulated the recruitment consistent with historical observations. For example, Needle
(2016) simulated one large recruitment in a random year within each 10-year simulation period, where the
large year-classes were separated by at least two years. Low recruitments were simulated by a log-normal
distribution about the geometric mean of low historical observations. de Moor et al., (2016) used a
hockey-stick stock-recruitment model with autocorrelation in recruitment deviates. For periods of low
recruitment, Roel and De Olivera (2007) used a Ricker model with autocorrelation and a cap so that
recruitment does not exceed the maximum estimated in the stock assessment. A large cohort was
generated about once in every 20 years as long as the SSB had as minimum historical threshold.

5) Other considerations

Data quantity and quality (including survey and catch data e.g., discarding)

For the redfish fishery and SRS in general, an essential aspect was the availability of fishery-independent
abundance surveys to detect and monitor strong recruitment events. OMs fitted to only commercial length
compositions are uninformative for estimating future strong year classes. It can thus be argued that
effective management of SRS can be achieved more easily in data-rich situations as it is the case for
redfish stocks. In a data-limited situation, of course, potential strong cohorts could have been simulated as
a possible OM, but without the knowledge of when the cohort(s) was produced, the magnitude, and
whether both species produced strong cohorts at the same time. This approach could generate less
credibility and added uncertainty in the MSE process because the redfish species also have different
productivity.

Careful analysis of survey samples was essential to eliminate the Grand Banks cohorts. _S. fasciatus_
has generated strong year classes that disappear from research surveys after a few years from
Units 1 and 2 (Valentin et al. 2015). For this study, genetic analyses were fundamental to discriminate the
Grand Banks cohorts from those that will contribute to the redfish fishery. For example, when strong
cohorts are detected in fish <22cm in future surveys, genetic analyses can help determine their potential contribution to the fishery. Also, it is important to monitor strong cohorts to define PMs. DFO trawl surveys have tracked these strong cohorts since 2011-2013, which are expected to materialize in the fishery by 2028. As a result, stakeholders were interested in PMs after this period when they start to contribute to the catch biomass. Moreover, there were no strong cohorts before 2011-2013. Thus, it was inappropriate to evaluate PMs (e.g., average catch biomass) before 2028.

In times of very strong recruitment, high percentages of small fish in the catch and high discarding rates are possible before the fish grow into marketable sizes; catches with large percentages of small fish have recently been observed on the Unit 1 and 2 fishing grounds (DFO 2018a). For the redfish stocks, there were no data available to estimate the vulnerability at age and discard rates in the OMs. This likely resulted in the poor fit to the retained compositions, especially for *S. mentella*. Uncertainty in the vulnerability at age and discarding were addressed in the OMs (e.g., OM1, 13, 14, 22, 23, and 24). The results showed that in the long term the vulnerability at age and discarding had a low impact on the fishery and conservation objectives compared with changes in productivity parameters (e.g., individual growth rates, recruitment compensation, and $M$). For example, a change in vulnerability at age towards younger fish (i.e., OM22) for the less abundant species, *S. fasciatus*, still passed the conservation criteria. However, all of the OMs showed that the SFP would be triggered unavoidably for the next two years independently of the MPs. This finding agreed with the Unit 1 DFO bottom trawl survey where the biomass of *S. mentella* and *S. fasciatus* juveniles was 60 and ten times higher, respectively, than their mean biomass for 1995-2015.

The number of times that the SFP will be triggered will depend significantly on growth rates of the strong 2011-2013 cohorts. Current individual growth estimates have shown that the strong 2011-2013 cohorts are growing as expected (DFO 2018a). Projections showed that under the OM4 (i.e., low individual growth) the catch biomass retained would be significantly reduced. We believe that this is unlikely. Under this hypothesis, the last strong 1980 cohorts would not have produced the observed catch
biomass retained if these cohorts had suffered a strong density-dependent effect. A similar situation can be inferred by doubling $M$ as the fishery representatives had suggested. Monitoring the strong 2011-2013 cohorts will provide more information about density-dependent effect in SRS.

Stakeholder participation and buy-in

This work showed the importance of the MSE process in helping to refocus efforts from selecting the “best stock assessment model” for redfish stocks (e.g., McAllister and Dupleisea 2011; Dupleisea et al. 2016; Dupleisea 2018) to selecting MPs that can meet objectives and are robust to uncertainty (Rademeyer et al. 2007; Punt et al. 2016). This situation was facilitated because of the prospective strong 2011-2013 cohorts, which are expected to be fully recruited to the fishery by 2028 (DFO 2018a). This situation “forced” the stakeholders to focus on both generating management objectives for this fishery and acceptable trade-offs. After a series of meetings and workshops (in total nine), the stakeholders, at first skeptical, got to recognize the value of participating fully in the MSE process. As the process progressed, the RWG received more requests from the industry to assess new OMs and candidate MPs (e.g., we ended up with more than 17 OMs and 22 MPs). We believe that the MSE was facilitated by the dynamic, open, and participatory process of the MSE. The entire process of conditioning the OMs, MP and PM development, projections, and peer-review took one and a half years, with results in-progress presented at every meeting. As the models were being developed it was appropriate to communicate to stakeholders that the results as presented were provisional and would be updated with further analysis before upcoming meetings. Therefore, we recommend that to engage participation, analysts be prepared to present MSE results in-progress throughout the MSE process, rather than waiting until the MSE models are fully developed and results are finalized.

We opted for an empirical HCR instead of a model-based approach to prescribing TACs. Empirical MPs have the benefit that they are more transparent and easier to explain to the industry (Geromont and Butterworth 2015b) compared to model-based MPs. For example, over the meetings, an industry member noticed that the initial HCRs could not meet their objectives when the stocks were at
low abundance. The control parameters to the HCRs were revised and modified to address the concerns.

This situation would have been less easily resolved under a model-based approach.

**Conclusion: Recommendations for managing spasmodically recruiting stocks**

Only rather general and imprecise recommendations have been provided for managing stocks with spasmodic recruitment (SRS) (e.g., Caddy and Gulland 1983), despite being not uncommon. We thus offer based on our recent experience with Atlantic redfish, the following new recommendations.

1) Fishery-independent survey data are needed to estimate trends in abundance and cohort strength and model future recruitment. For example, well-designed fishery-independent trawl surveys that cover the known spatial extent of the stock(s) can enable reliable detection of strong recruitment events and monitoring of changes in abundance. The trawl survey should provide age or length composition samples of pre-recruits to give advance warning of strong recruitment events. It is less critical to attempt to gather age or length composition data from the fishery, since possible increases in discarding rates when stronger cohorts are recruiting to the fishery compromise the usefulness of these data.

2) Devoting resources and time to obtain credible records of harvested catch biomass (i.e., retained + discarded) is essential for stocks with spasmodic recruitment; adjusting catch biomass records for past discarding practices (e.g., Duplisea 2018) will reduce bias in reconstructions of stock biomass and productivity.

3) For SRS of longer-lived species such as gadoids and *Sebastes*, it may be advantageous to adopt an inventory management harvest strategy, which takes a longer-term view of the fishery when strong cohorts are predicted to recruit to the fishery. However, management procedures should also be designed to allow for continued low harvests during extended periods of low abundance. For example, in the Unit 1 and 2 Atlantic redfish fishery, the largest cohorts for both stocks originated from the lowest estimated abundances in the sixty year history of the fishery and while non-negligible harvests were allowed to be taken by industry.
4) Management procedures for long-lived species such as redfish could include the following features: (i) An empirical HCR based on fishery-independent abundance indices based on desirable fish sizes. This allows harvest of the strong cohorts a few years later and reduces the risk of growth overfishing (i.e., new recruits). (ii) Ramp caps in catch limits when abundance is building and a maximum cap when harvestable biomass has peaked. Also ramp-ups should be compatible with feasible expansions of industrial capacity and foreseeable marketing opportunities for the expanded harvests.

5) Bayesian approaches are suitable for fitting operating models for SRS because they provide formalized methods to include results from meta-analyses of key parameters from other similar stocks (e.g., Forrest et al. 2010) and anecdotal information about, e.g., trawl survey catchability (McAllister et al. 2010), historic discarding practices from fisher interviews (Duplisea 2018) and strong cohorts from early fishery management briefs.

6) It is critical to formulate operating models of multi-stock complexes targeted by a single fishery which simultaneously simulate the capture of the main fish stocks of interest to the fishery. This can help the analysis become more credible to industry and managers and allow the evaluation of trade-offs between the conservation of weaker stocks and fishery yields, especially following spasmodic recruitment events for one or more stocks.

7) Where scientific and managerial capacity are available and harvest potential is significant, we would recommend that fishery managers consider adopting MSE as a collaborative approach. Close collaboration between scientists, fishery managers, and industry stakeholders is essential for developing management procedures that enable well-controlled responses of industrial capacity to large systematic changes in harvestable biomass. In addition, this helps industry to establish and adapt effectively to better understanding of stock dynamics. For example, awareness of “boom-and-bust” stock dynamics can help industry to develop long-term plans to avoid overcapacity during periods of low productivity and achieve long-term profitable levels of capacity especially after large cohorts are detected.
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strong juvenile year classes in redfish (Sebastes spp.): the importance of species identity, population

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Fig. 1. Units 1 and 2 of the Canadian Atlantic redfish fishery. The overlapping region (Unit 1+2) constitutes NAFO divisions 4Vn and 3Pn and is included in Unit 1 from January-May, and Unit 2 from June-December. PEI (Prince Edward Island), NS (Nova Scotia). Figure was produced using the R software (R Core Team 2019).
Fig. 2. Historical landings (bars) and catch limits (CL) (lines) for *Sebastes* spp. for the management Units 1 and 2, off eastern Canada.
Fig. 3. Example of the harvest control rule (HCR) applied in the management procedures (MPs) for *Sebastes mentella* (and *S. fasciatus*). For each species, the raw catch limit (CL) increases or decreases depending on an index \( J_t \) which measures the level of the biomass of the DFO trawl survey index during the three last years (as a geometric mean) relative to a geometric mean for a reference period (i.e., 1984-2017) (Table B5.B1-B2). Here, \( J_t \) is called “the fishery index” because it only uses commercially sought after fish sizes for each species. The raw CL decreases linearly as \( J_t \) decreases, but the raw CL decreases faster when \( J_t \) reaches a threshold due to quadratic term in the HCR (B5.B3). The raw CL is zero at low biomass levels. The HCR was tuned so that the raw CLs were similar to the average historical catch retained (dot) when the stock was at low levels (e.g., 2007-2017). The arrow indicates the average \( J_t \) for the last decade. Also, the HCR was tuned to avoid overfishing in periods of high abundance. After calculating the raw CL, a series of rules determine the realized CL for each species (see Fig. 4). A full mathematical description of the HCR is in Table A5.
Fig. 4. Example for determining the catch limits (CL in kt) for each species. The lines are the raw CL for *S. mentella* (twodash) and *S. fasciatus* (long dash), CL caps (dotted), and the realized CL (solid gray) for MP1 (see Table S4 for *p* = 1) under the base case operating model. The raw CL for each species was obtained after applying the HCR (Table A5.B1-B3), and then the raw CLs were combined (A5.B4). After that, a CL cap is applied (A5.B5). For MP1, the CL cap has a ramp cap starting in 2020 (at 14.5kt) and ending in 2027 (at 40kt), followed by a maximum cap at the end of the ramp up period (i.e., 40kt). Note that when the combined CL exceeded the cap, the CL corresponds to the cap for that year. Also, note that this MP has a status quo for 2018-2019 (at 3kt). The catch biomass realized for each stock is obtained after catch split implementation (A5.B6) and implementation error (A5.B7). These last two steps are not shown in the plot.


*Fig. 5.* Flow chart illustrating steps to calculate the realized catch limit (CL) for *Sebastes mentella* (S.m) and *S. fasciatus* (S.f.) under management procedure #1 (MP): (i) HCR, (ii) combined CL, (iii) capped CL and population and fishery processes represented in the operating model (OM). $J_t$ is the “fishery index” calculated from the DFO trawl survey data (see *Fig. 3*). Note that the OM simulates the catch biomass (CB) removed by species. In the catch split, the CB split by species was drawn randomly from the set of species split ratios observed for 1984-2017 in the Unit 1 bottom trawl survey (DFO has used meristic counts to distinguish between the two species). When simulating implementation error, the catch biomass discarded dead by species in excess of the total CL is taken into account.
Fig. 6. Plots (from top to bottom) for the base case operating model (OM1) for *S. mentella* of the 1) catch biomass killed and retained, 2) harvest rates, 3) observed (dots) and predicted fits (lines) for the Units 1 and 2 survey indices, and 4) age 1 recruitment (long dash) and spawning stock biomass (SSB)(solid line). The shaded bands are 95% confidence intervals.
Fig. 7. Plots (from top to bottom) for the base case operating model (OM1) for *S. fasciatus* of the 1) catch biomass killed and retained, 2) harvest rates, 3) observed (dots) and predicted fits (lines) for the Units 1 and 2 survey indices, and 4) age 1 recruitment (long dash) and spawning stock biomass (SSB)(solid line). The shaded bands are 95% confidence intervals.
Fig. 8. Average catch retained (in kt) for the 10-20 and 10-40 year horizon for the 21 management procedures (MPs) under the core operating models (see Table A6). Dots show medians and bars shown the 90% confidence intervals. The label on the x-axis indicates: the assigned number for the MP, capped (C) or uncapped MPs, and the starting year for the HCR (see Tables S4-5).
Fig. 9. Average number of years with catch limits (CL) $\geq$40kt in the 2028-2057 year horizon for the 21 management procedures (MPs) under the core operating models (see Table A6). Dots show medians and bars shown the 90% confidence intervals. The label on the x-axis indicates: the assigned number for the MP, capped (C) or uncapped MPs, and the starting year for the HCR (see Tables S4-5).
Table A1. Conservation and fishery objectives and their respective performance metrics (PMs) agreed by the redfish working group. The parenthesis indicates the probability of passing criteria for the PM. CL (Catch Limit).

<table>
<thead>
<tr>
<th>Type</th>
<th>Objective</th>
<th>Performance metric(s)</th>
</tr>
</thead>
</table>
| Conservation | 1) Increase spawning stock biomass (SSB) for each species above the lower reference point (LRP) and upper stock reference (USR) in 10 years | a) Proportion of simulations where SSB > LRP for each species (>0.95)  
    |               |                                                                           | b) Proportion of simulations where SSB > USR for each species (>0.95)                 |
|              | 2) Once in USR (healthy zone), maintain SSB of each species above LRP (critical zone) and healthy zone after 10 years | a) Proportion of simulations where SSB > LRP for each species (>0.95)  
    |               |                                                                           | b) Proportion of simulations where SSB > USR for each species (>0.75)                 |
|              | 3) Maintain the harvest rate ($U_t$) of each species below the harvest rate at maximum sustainable yield ($U_{msy}$) all 40-years | Proportion of years where $U_t$ is below $U_{msy}$ (>0.5)                              |
| Fishery      | 4) Maximize the number of years where the small fish protocol (SFP) is not triggered | Average years where fish <22cm represent <15% of catch over:  
    |               |                                                                           | a) Simulation years 2018-2022 (0.85)  
    |               |                                                                           | b) all 40-years (0.85)                                                              |
|              | 5) Maximize the number of years where the SFP is not triggered            | Average years where fish <25cm represent <15% of catch over:  
    |               |                                                                           | a) Simulation years 2018-2022 (0.85)  
    |               |                                                                           | b) all 40-years (0.85)                                                              |
|              | 6) Maximize duration of high annual catch                                | a) Average annual catches over:  
    |               |                                                                           | i. Simulation years 2028-2037 (10 years)  
    |               |                                                                           | ii. Simulation years 2028-2057 (30 years)  
    |               |                                                                           | b) Proportion of simulations where CLs ≥ 40kt by 2028-2057  
    |               |                                                                           | c) Average number of years where CLs ≥ 40kt between 2028-2057  
    |               |                                                                           | d) Proportion of years with 4kt (avg. catch retained in 2015-2017)                  |
|              | 7) Maximize catch of large fish (>27cm)                                  | Proportion of years where large fish comprise >80% of catch over:  
    |               |                                                                           | a) Simulation years 2018-2022 (>0.5)  
    |               |                                                                           | b) all 40-years (>0.75)                                                             |
|              | 8) Maintain fishery stability between years                              | a) Proportion of years where CLs changes less than 15% between years (>0.75)  
    |               |                                                                           | b) Average annual variation (AVV) in CL over:  
    |               |                                                                           | i. Simulation years 2028-2037 (<0.15)  
    |               |                                                                           | ii. Simulation years 2028-2057 (<0.15)                                              |
Table A2. Base case operating model (OM1) for redfish *Sebastes mentella* (*Sm*) and *S. fasciatus* (*Sf*).

Note that we omitted subscript $x$ because both species have the same sets of equations. However, we included the subscript $x$ to clarify the differences between the two species for some equations. The parameter with a “$\hat{\cdot}$” symbol indicates that the value was estimated in the OM (see Table S1), otherwise it was pre-specified.

<table>
<thead>
<tr>
<th>Symbol</th>
<th>Value (<em>Sm</em>; <em>Sf</em>)</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>$x$</td>
<td>1; 2</td>
<td>Index for species: 1: <em>S. mentella</em> 2: <em>S. fasciatus</em></td>
</tr>
<tr>
<td>$a$</td>
<td>$1,\ldots, A; 1,\ldots, A$</td>
<td>Age-year class</td>
</tr>
<tr>
<td>$t$</td>
<td>$1951,\ldots, T; 1951,\ldots, T$</td>
<td>Annual time step</td>
</tr>
<tr>
<td>$l$</td>
<td>$1,\ldots, L; 1,\ldots, L$</td>
<td>Length classes (cm)</td>
</tr>
<tr>
<td>$g$</td>
<td>$1,2,3,4; 1,2,3,4$</td>
<td>Index for gear type: 1: Survey Unit 1 (total biomass or length compositions) 2: Survey Unit 2 (total biomass or length compositions) 3: “fishery index” for Unit 1 (see text for details) 4: Index for fishery (combined length compositions retained for Units 1 and 2)</td>
</tr>
<tr>
<td>$p$</td>
<td>See Tables S4-5</td>
<td>Index for Management Procedures (MPs)</td>
</tr>
<tr>
<td>$A$</td>
<td>57; 57</td>
<td>Plus-group</td>
</tr>
<tr>
<td>$T$</td>
<td>2017; 2017</td>
<td>Last year before the management procedure starts (MP) (i.e., 2018)</td>
</tr>
<tr>
<td>$T_2$</td>
<td>2053; 2053</td>
<td>Year when the MP ends</td>
</tr>
<tr>
<td>$L$</td>
<td>57; 57</td>
<td>Number of size classes</td>
</tr>
<tr>
<td>$blk_1$</td>
<td>1951,\ldots, 1993; 1951,\ldots, 1993</td>
<td>Vulnerability time-block 1</td>
</tr>
<tr>
<td>Parameters</td>
<td></td>
<td></td>
</tr>
<tr>
<td>$L_\infty$</td>
<td>45.82; 41.24</td>
<td>Mean asymptotic length (cm)$^1$</td>
</tr>
<tr>
<td>$k$</td>
<td>0.096; 0.106</td>
<td>Brody growth parameter (cm/yr)$^1$</td>
</tr>
<tr>
<td>$t_0$</td>
<td>-0.5; -0.5</td>
<td>Theoretical age at length zero (yr)</td>
</tr>
<tr>
<td>$a^w$</td>
<td>0.00762; 0.00762</td>
<td>Scaling constant for weight-at-length (cm*gr)$^2$</td>
</tr>
<tr>
<td>$b^w$</td>
<td>3.193; 3.193</td>
<td>Allometric factor$^2$</td>
</tr>
<tr>
<td>$M_a$</td>
<td>0.10; 0.125</td>
<td>Instantaneous natural mortality at age (yr)$^{-1}$</td>
</tr>
<tr>
<td>$\hat{R}_0$</td>
<td>0.420; 0.623</td>
<td>Mean unfished recruitment (age-1) x $[1e9]$</td>
</tr>
<tr>
<td>$\hat{c}$</td>
<td>3.082; 6.522</td>
<td>Compensation ratio</td>
</tr>
<tr>
<td>$h$</td>
<td>0.435; 0.620</td>
<td>Steepness</td>
</tr>
<tr>
<td>$\hat{\omega}_a^{3}, \hat{\omega}_a^{8}$</td>
<td>1952,\ldots, $T$; 1952,\ldots, $T$</td>
<td>Estimated deviates for age 3 in the initial abundance and recruitment deviates in year $t$</td>
</tr>
</tbody>
</table>
### Simulated Recruitment

- $\omega^m_t$: $t > T, ..., T_2; t > T, ..., T_2$
  - Simulated recruitment deviates in year $t$

### Recruitment Variation

<table>
<thead>
<tr>
<th>Symbol</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\sigma_R$</td>
<td>1.0; 1.0</td>
</tr>
<tr>
<td>$\Omega_1$</td>
<td>7.889; 7.256</td>
</tr>
<tr>
<td>$\Omega_2$</td>
<td>1.98; 1.58</td>
</tr>
</tbody>
</table>

### Age-at-50% Maturity

- $\hat{a}^m_{g=1,2}$: 2.022, 7.017; 1.676, 5.224
  - Age-at-50% vulnerability for survey index $g$

### Age-at-Vulnerability Slope

- $\hat{a}^{sd}_{g=1,2}$: 0.145, 2.731; 0.201, 0.609
  - Age-at-vulnerability slope for survey index $g$

### Age-at-50% Retention

- $\hat{a}^{sd}_{ret, g=1,2}$: 9.245; 8.545
  - Age-at-50% retained for index $g$ and time block 1

### Age-at-Vulnerability Slope for Survey Index

- $\hat{a}^{sd}_{g=1,2}$: 0.866; 0.471
  - Age-at-vulnerability slope for survey index $g$ and time block 1

### Age-at-50% Retention for Survey Index

- $\hat{a}^{sd}_{ret, g=1,2}$: 8.184; 8.329
  - Age-at-50% retained for index $g$ and time block 2

### Age-at-Vulnerability Slope for Survey Index

- $\hat{a}^{sd}_{g=1,2}$: 0.735; 0.534
  - Age-at-retained slope for index $g$ and time block 2

### Derived Variables

#### Survival-at-Age

- $L_a$: Mean length-at-age (cm)
- $s_a$: Natural survival-at-age (yr)
- $w_a$: Weight-at-age (gr)
- $w_l$: Weight-at-length (gr)

#### Age-at-Fishery

- $\hat{v}_{a,g} = 4$: Fig. S1
  - Vulnerability-at-age, for index $g$

- $\hat{v}_{a,50,t,g} = 4$: Fig. S1
  - Age-at-50% vulnerability for fishery in year $t$

#### Other Parameters

- $\phi_0$: Unfished equilibrium spawning biomass per recruit
- $l_a$: Survivorship-at-age per recruit
- $SSB_0$: Unfished spawning biomass (kt)
- $n_g=1,2,3,4$: 34, 8, 34, 33; 34, 8, 34, 33
- $U_{max}$: Maximum exploitation rate
- $u_{init}$: Initial exploitation rate for $t=1951$
- $\hat{t}_g = 1,2,3$: 0.469, 0.452, 0.407; 0.501, 0.382, 0.519; 0.501, 0.382, 0.516
  - Standard deviation for survey index $g$
- $\hat{q}_g = 1,2,3$: 0.666, 1.745, 0.638; 0.533, 2.495, 0.444; 0.533, 2.495, 0.444
  - Catchability coefficient for survey index $g$

#### Minimum Age-at-Vulnerability/Retention

- $a_{m_{min}}$: Minimum age-at-retention/vulnerability for projections

<table>
<thead>
<tr>
<th>Parameter</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\omega^m_t$</td>
<td>$t &gt; T, ..., T_2; t &gt; T, ..., T_2$</td>
</tr>
<tr>
<td>$\sigma_R$</td>
<td>1.0; 1.0</td>
</tr>
<tr>
<td>$\Omega_1$</td>
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<tr>
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</tr>
<tr>
<td>$\hat{a}^{sd}_{ret, g=1,2}$</td>
<td>9.245; 8.545</td>
</tr>
<tr>
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<tr>
<td>$\hat{a}^{sd}_{g=1,2}$</td>
<td>0.735; 0.534</td>
</tr>
<tr>
<td>$\hat{a}^{sd}_{ret, g=1,2}$</td>
<td>0.866; 0.471</td>
</tr>
<tr>
<td>$\hat{a}^{sd}_{g=1,2}$</td>
<td>19.0; 18.0</td>
</tr>
<tr>
<td>$\hat{a}^{sd}_{g=1,2}$</td>
<td>1.0; 1.0</td>
</tr>
<tr>
<td>$a_{m_{min}}$</td>
<td>7; 7</td>
</tr>
<tr>
<td>$n_g=1,2,3,4$</td>
<td>34, 8, 34, 33; 34, 8, 34, 33</td>
</tr>
<tr>
<td>$\phi_0$</td>
<td>Unfished equilibrium spawning biomass per recruit</td>
</tr>
<tr>
<td>$l_a$</td>
<td>Survivorship-at-age per recruit</td>
</tr>
<tr>
<td>$L_a$</td>
<td>Mean length-at-age (cm)</td>
</tr>
<tr>
<td>$s_a$</td>
<td>Natural survival-at-age (yr)</td>
</tr>
<tr>
<td>$w_a$</td>
<td>Weight-at-age (gr)</td>
</tr>
<tr>
<td>$w_l$</td>
<td>Weight-at-length (gr)</td>
</tr>
<tr>
<td>$m_a$</td>
<td>Proportion mature-at-age</td>
</tr>
<tr>
<td>$SSB_0$</td>
<td>Unfished spawning biomass (kt)</td>
</tr>
<tr>
<td>$\hat{a}_{50,t,g} = 4$</td>
<td>Fig. S1</td>
</tr>
<tr>
<td>$\hat{v}_{a,g} = 4$</td>
<td>Fig. S1</td>
</tr>
</tbody>
</table>
\[ V'_{a,t,g=4} \] Vulnerability-at-age, in year \( t \) for the fishery (domed-shaped)

\[ V''_{a,g=1,2} \] Vulnerability-at-age for the survey (domed-shaped)

\[ \Phi^p_{x}(L_a) \] Transition matrix for species \( x \)

\[ C^k_t \] Fig. 6; Fig. 7 Catch biomass killed in year \( t \) (kt)

\[ f^{ret}_{a,t} \] Fraction retained-at-age, \( a \), in year \( t \) for the fishery

\[ u_{x,t} \] Fig. 6; Fig. 7 Harvest rate in year \( t \) for species \( x \)

\[ N_{x,t} \] Number-at-length, \( l \), in year \( t \) for species \( x \)

\[ SSB_t \] Spawning biomass in year \( t \) (kt)

\[ VB_{x,t}^k \] Vulnerable biomass killed in year \( t \) (kt) for species \( x \)

\[ VB_{t}^{ret} \] Vulnerable biomass retained in year \( t \) (kt)

Observations

\[ l_{x,t,g} = 1,2,3 \] Fig. 6; Fig. 7 Observed survey biomass index \( g \) in year \( t \) (kt) for species \( x \)

\[ I'_{x,t,g=3} \] Simulated survey fishery index (length-based) in year \( t \) (kt) for species \( x \)

\[ C^{ret}_t \] Fig. 6; Fig. 7 Catch biomass retained in year \( t \) for the fishery (kt)

\[ D_{kill, ratio}^{t} \] No species-specific (Fig. S3) Catch biomass killed to catch biomass retained ratio in year \( t \)

\[ a^{off, set}_t \] No species-specific (Fig. S2) Offset in year \( t \) (ages)

\[ N_{survey, obs}^{t|g} = 1,2 \] Figs. S4-5; Figs. S7-8 Observed numbers-at-length for the survey in Units 1 and 2

\[ N_{ret, obs}^{t|g} = 4 \] Fig. S6; Fig. S9 Observed numbers-at-length (retained) for the fishery in Units 1 and 2

\[ ess_g = 1,2,4 \] 25, 10, 5; 25, 10, 5; Sample size for index \( g \)

\[ CV_g = 1,2 \] 0.25, 0.25; 0.25, 0.25 Coefficient of variation for survey index \( g \)

\[ CV_{a} \] 0.12 \(_a=4\), ..., 0.05 \(_a=4\); 0.12 \(_a=4\), ..., 0.05 \(_a=4\) Coefficient of variation for length-at-age (linear increase)

Table A3. Equations for the base case operating model (OM1) for redfish \( Sebastes mentella \) (Sm) and \( S. fasciatus \) (Sf). Note that we omitted subscript \( x \) because both species have the same sets of equations.
However, we included the subscript $x$ to clarify the differences between the two species for some equations.

<table>
<thead>
<tr>
<th>Life history schedules</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>(A1) Natural survival-at-age</td>
<td>$s_a = e^{-M_a}$</td>
</tr>
<tr>
<td>(A2) Survivorship per recruit</td>
<td>$l_a = \begin{cases} 1 \ l_{a-1} &amp; 1 &lt; a &lt; A \ l_{a-1} \cdot s_{a-1} (1 - v_{a-1} \cdot s_{a-1}) / [1 - s_{a-1} (1 - v_{a-1} \cdot s_{a-1})] &amp; a = A \end{cases}$</td>
</tr>
<tr>
<td>(A3) Mean length-at-age</td>
<td>$\bar{L}_a = L_0 [1 - e^{-k(a - \eta_0)}]$</td>
</tr>
<tr>
<td>(A4) Proportion mature-at-age</td>
<td>$m_a = \begin{cases} 0 &amp; \text{if } a \leq a_{\text{mat}}^{\text{min}} \ 1 &amp; \text{otherwise} \end{cases}$</td>
</tr>
<tr>
<td>(A5) Weight-at-age</td>
<td>$w_a = a^{w} L_a^{b^w}$</td>
</tr>
<tr>
<td>(A6) Vulnerability-at-age</td>
<td>$v_{a,t,g} = \begin{cases} 0 &amp; \text{if } a \leq a_{\text{vul}}^{\text{min}} \ 1 / \left[ 1 + e^{-\left(\frac{a_{\text{vul}} - a}{a_{\text{vul}} - a_{\text{vul}} + g}</td>
</tr>
</tbody>
</table><p>ight)^2} \right] &amp; \text{otherwise} \end{cases}$ |
| Stock-recruitment relationship |  |
| (A7) Spawning biomass per recruit | $\phi_0 = \sum_{a=1}^{a_{\text{ad}}} l_a m_a w_a$ |
| (A8) Unfished spawning biomass | $SSB_0 = \hat{R}<em>0 \phi_0$ |
| (A9) Beverton-Holt recruitment parameters | $\alpha = \frac{\kappa}{\phi_0}$, $\beta = \frac{\kappa - 1}{SSB_0}$ |
| Population dynamics |  |
| (A10) Initial condition | $N</em>{a,1} = R_0 l_a$ for $a \neq 3$ |
|  | $N_{a,1} = R_0 l_a e^{-\theta_a^2 \sigma_{\theta_a} - 0.5(\sigma_{\theta_a})^2}$, $\theta_a \sim N(0,1)$ for $a = 3$ |</p>
(A11) Recruitment deviates
\[
\gamma_t = \begin{cases} 
\hat{\omega}_t'' & t < T \\
\omega_t'' & \text{otherwise}
\end{cases}
\]
\[
\omega_t'' \sim \text{bootstrap}(\hat{\omega}_t'', \text{OM}_t)
\]

See OMs section

(A12) Recruitment
\[
(t > 1; \ a = 1)
\]
\[
N_{1,t} = \frac{\alpha \text{SSB}_{t-1} - e^{\gamma \sigma x - 0.5(\sigma x)^2}}{1 + \beta \text{SSB}_{t-1}}
\]

(A13) Abundance dynamics
\[
(t > 1)
\]
\[
N_{a,t} = \begin{cases} 
N_{a-1,t-1} s_{a-1} (1 - v_{a-1,t} u_{t-1}) & 1 < a < A \\
N_{a-1,t-1} s_{a-1} (1 - v_{A-1,t} u_{t-1}) & a = A
\end{cases}
\]

(A14) Spawning biomass
\[
\text{SSB}_t = \sum_{a=1}^{A} N_{a,t} m_a w_a
\]

(A15) Vulnerable biomass killed
\[
V_{B_i} = \sum_{a=1}^{A} N_{a,t} v_{a,t,g} f_{a,t} w_a
g = 4
\]
\[
(t > T) \quad v_{a,t,g} \in \text{blk}_2
\]

(A16) Vulnerable biomass retained
\[
V_{B_i} = \sum_{a=1}^{A} N_{a,t} v_{a,t,g} f_{a,t}^{ret} w_a
g = 4
\]
\[
(t > T) \quad v_{a,t,g} ; f_{a,t}^{ret} \in \text{blk}_2
\]

(A17) Catch biomass killed
\[
C_i = u_i V_{B_i}
\]

(A18) Catch biomass retained
\[
C_i = u_i V_{B_i}^{ret}
\]

Fishery index (length-based)

(A19) Number-at-length
\[
N_{x,i,j} = N_{x,a} \Phi^{P(1/a)}_{x} A_{x} L
\]

(A20) Fishery index
\[
I'_{x,i,j} = q_{x,g} \sum_{L \geq 30\text{cm}} N_{x,j} w_{x,j} e^{\varepsilon x - 0.5\varepsilon^2} e_x \sim N(0,1) \quad g = 3, x = 1
\]
\[
I'_{x,i,j} = q_{x,g} \sum_{L \geq 29\text{cm}} N_{x,j} w_{x,j} e^{\varepsilon x - 0.5\varepsilon^2} e_x \sim N(0,1) \quad g = 3, x = 2
\]
<table>
<thead>
<tr>
<th>Symbol</th>
<th>Value $Sm; Sf$</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>$\Omega^a$</td>
<td>4; 4</td>
<td>Tuning parameter for HCR: intercept (kt)</td>
</tr>
<tr>
<td>$\Omega^b$</td>
<td>2; 2</td>
<td>Tuning parameter for HCR: slope (kt/year)</td>
</tr>
<tr>
<td>$b$</td>
<td>No species-specific</td>
<td>Slope multiplier</td>
</tr>
<tr>
<td>$\Omega^c$</td>
<td>1.2; 1.2</td>
<td>Penalty (kt/year) for HCR</td>
</tr>
<tr>
<td>$J_{0x}$</td>
<td>1.5; 1.5</td>
<td>Threshold parameter when $J_{x,i,g}$ decreases at low biomass levels in the HCR</td>
</tr>
<tr>
<td>$J_{x,i,g}$</td>
<td></td>
<td>Relative fishery index for the species $x$</td>
</tr>
<tr>
<td>$I_{x,i,g=3}$</td>
<td></td>
<td>Historic fishery index for unit 1 for species $x$; fish &gt; 30 and 29 cm for $Sm$ and $Sf$, respectively</td>
</tr>
<tr>
<td>$CL_{i,p}^{cap}$</td>
<td>No species-specific</td>
<td>Catch biomass caps (kt) in time $t$ for the $MP=p$.</td>
</tr>
<tr>
<td>$CL_{x,t}$</td>
<td></td>
<td>Raw catch limit (kt) derived from the HCR for species $x$</td>
</tr>
<tr>
<td>$CL_{x}$</td>
<td></td>
<td>Catch limit (kt) combined for $Sm$ and $Sf$.</td>
</tr>
<tr>
<td>$CL_{x,t}^*$</td>
<td></td>
<td>Catch biomass (kt) after catch split implementation for species $x$</td>
</tr>
<tr>
<td>$CL_{x,t}^{killratio}$</td>
<td></td>
<td>Catch biomass (kt) after implementation error for species $x$</td>
</tr>
<tr>
<td>$\pi_x$</td>
<td>See Table A5.B6</td>
<td>Proportion for catch split implementation for species $x$</td>
</tr>
<tr>
<td>$D_{prj}^{killratio}$</td>
<td>1.1; 1.1</td>
<td>fish killed:retained ratio for the projections</td>
</tr>
</tbody>
</table>
**Table A5.** Description for the empirical (target-based) harvest control rule (HCR) for redfish *Sebastes mentella* (Sm) and *S. fasciatus* (Sf).

<table>
<thead>
<tr>
<th>Equation</th>
<th>Description</th>
</tr>
</thead>
<tbody>
<tr>
<td>$i &gt; T; g = 3$</td>
<td>$t &gt; T$; $g = 3$</td>
</tr>
<tr>
<td>(B1) Raw CL</td>
<td>$CL_{t,i} = \left[ \Omega_x^a + b\Omega_x^b(J_{x,t,g} - J_{0x}) \right] - pen_x$</td>
</tr>
<tr>
<td>(B2) Trailing average for fishery index</td>
<td>$J_{x,t,g} = \exp \left[ \frac{1}{3} \sum_{i=2}^{t} \ln(I_{x,i,t,g}) \right] \exp \left[ \frac{1}{34} \sum_{i=1984}^{t=2017} \ln(I_{x,i,t,g}) \right]$</td>
</tr>
<tr>
<td>(B3) Penalty</td>
<td>$pen_x = \begin{cases} 0 &amp; \text{if } J_{x,t,g} &lt; J_{0x} \ \Omega_x^c(J_{x,t,g} - J_{0x})^2 &amp; \text{otherwise} \end{cases}$</td>
</tr>
<tr>
<td>(B4) Combined CL</td>
<td>$CL^*<em>i = \sum</em>{x=1}^{2} CL_{x,t}$</td>
</tr>
<tr>
<td>(B5) Cap implementation</td>
<td>$CL^<em>_i = \begin{cases} CL^c &amp; \text{if } CL^c \leq CL^cap \ CL^</em><em>i \pi</em>{x,t} &amp; \text{otherwise} \end{cases}$</td>
</tr>
<tr>
<td>(B6) Catch split implementation</td>
<td>$\pi_{x=1,t} \sim Unif(0.39, 0.44)$</td>
</tr>
<tr>
<td></td>
<td>$\pi_{x=2,t} = 1 - \pi_{x=2,t}$</td>
</tr>
<tr>
<td></td>
<td>$CL^<em>_i = CL^</em><em>i \pi</em>{x,t}$</td>
</tr>
<tr>
<td>(B7) Implementation error</td>
<td>$CL^{kill_ratio}_i = CL^*_i D^{kill_ratio_proj}$</td>
</tr>
<tr>
<td>(B8) Output control</td>
<td>$u_{x,t} = \min \left( \frac{CL^{kill_ratio}_i}{VB^*<em>{x,t}}, U</em>{max} \right)$</td>
</tr>
<tr>
<td>(B9) Small fish protocol</td>
<td>$u_{x,t} = \begin{cases} 0 &amp; \text{if } \text{small fish protocol} \ u_{x,t} &amp; \text{otherwise} \end{cases}$</td>
</tr>
</tbody>
</table>
Table A6. Core, stress, and sensitivity test operating models (OMs) formulated for the redfish management strategy evaluation (MSE). Full description of the base case operating model (OM1) is given in Tables S1-2-3.

<table>
<thead>
<tr>
<th>OM</th>
<th>Type</th>
<th>Description</th>
<th>Concerns addressed</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>Core (base case)</td>
<td>Logistic fishery selectivity, in two-blocks (1951-1993 and 1994-2017). Future recruitment as seen in the past, simulated with a non-parametric bootstrap using the recruitment estimates from 1970-2016. HCR uses a future fish killed:retained ratio ($D_{pr}$) equal to 1.1</td>
<td>The base case was developed based on the current state of knowledge about redfish stocks in Units 1+2</td>
</tr>
<tr>
<td>6</td>
<td>Core</td>
<td>Like the base case but no strong cohorts for the 20 years of projections</td>
<td>It may be possible that after the strong cohorts of 2011-2013 there will be no more strong cohorts for the next 20 years. As a result, the expected projected biomass would be lower than expected</td>
</tr>
<tr>
<td>8</td>
<td>Core</td>
<td>Like the base case but the $M$ is age-specific following a Lorenzen function ($M$ is higher for smaller fish) for historic and future projections</td>
<td>The strong cohorts 2011-2013 may introduce density-dependent effects due to cannibalism and juvenile natural mortality</td>
</tr>
<tr>
<td>9</td>
<td>Core</td>
<td>Like the base case but future recruitment is simulated with a parametric bootstrap, with variance and autocorrelation parameters generated from the recruitment estimates from 1970-2016</td>
<td>The future recruitment is unpredictable so the recruitment dynamics may not follow the observed recruitment in the past</td>
</tr>
<tr>
<td>10-11</td>
<td>Core</td>
<td>Like the base case but the historic catch biomass retained and species composition of landings is biased towards either <em>S. mentella</em> (OM10) or <em>S. fasciatus</em> (OM11)</td>
<td>The catch split is based on DFO survey carried out in Unit 1. This assumes that the species composition in Unit 1 is similar to Unit 2</td>
</tr>
<tr>
<td>2</td>
<td>Stress</td>
<td>Like the base case but the historic and future fishery and survey vulnerability are dome-shaped</td>
<td>The vulnerability may not be asymptotic due to temporal gear/vessel changes or spatial distribution of fish/fleet</td>
</tr>
<tr>
<td>3</td>
<td>Stress</td>
<td>Like the base case but future natural mortality ($M$) is doubled for the first 20 years of projections</td>
<td>The strong cohorts of 2011-2013 may introduce density dependent effects due to cannibalism and juvenile natural mortality. As a result, the expected projected biomass would be lower than expected</td>
</tr>
<tr>
<td></td>
<td>Stress</td>
<td>Like the base case but for the first 20 years of projections, the growth rate is reduced by reducing the asymptotic length ($L_\infty$) to 2/3 as large as in the base case, while fixing the other von Bertalanffy growth parameters</td>
<td>The strong cohorts of 2011-2013 may introduce density-dependent effects on individual growth rates via, for example, competition and food limitation. As a result, the expected projected biomass would be lower than expected</td>
</tr>
<tr>
<td>---</td>
<td>---</td>
<td>---</td>
<td>---</td>
</tr>
<tr>
<td>5</td>
<td>Stress</td>
<td>Like the base case but no strong cohorts for the 40 years of projections</td>
<td>It may be possible that after the strong cohorts of 2011-2013 there will be no more strong cohorts for the next 40 years. As a result, the expected projected biomass would be lower than expected</td>
</tr>
<tr>
<td>14</td>
<td>Stress</td>
<td>Like the base case but the historic fish killed:retained ratio ($D_{prj}^{kill:ret}$) is either increased or decreased by 50%</td>
<td>No quantitative information was available to estimate discarding rates. The catch biomass retained would be more or less to the reported landings</td>
</tr>
<tr>
<td>15-16</td>
<td>Stress</td>
<td>Like the base case but historic and future $M$ is lower by a factor of 0.75 (OM15) or higher by a factor of 1.25 (OM16)</td>
<td>$M$ was input parameter in the OMs. This productivity parameter has a large impact on management strategy performance. It has been recommended to consider this uncertainty in any MSE (Punt et al., 2014)</td>
</tr>
<tr>
<td>17-18</td>
<td>Stress</td>
<td>Like the base case but the historic and future steepness ($h$) is lower by a factor of 0.75 (OM18) or higher by a factor of 1.25 (OM17) respect to the base case</td>
<td>$h$ was input parameter in the OMs. This productivity parameter has a large impact on management strategy performance. It has been recommended to consider this uncertainty in any MSE (Punt et al., 2014)</td>
</tr>
<tr>
<td>22</td>
<td>Stress</td>
<td>Like the base case but the fishery vulnerability for 2017-2021 reverts back to the estimated for the first time-block (1951-1993)</td>
<td>It is possible that the estimated fishery vulnerability for the first time-block is a better presentation for the period when the strong cohorts 2011-2013 are still not recruited</td>
</tr>
<tr>
<td>23</td>
<td>Stress</td>
<td>Like the base case but the $D_{prj}^{kill:ret}$ was increased from 1.1 to 2 from 2018-2020</td>
<td>The historic high discarding in the late 1980s would occur again in 2018-2020 due to the strong cohorts 2011-2013 but the discarding will return to 1.1 for 2021-2057</td>
</tr>
<tr>
<td>24</td>
<td>Stress</td>
<td>Like the base case but the $D_{prj}^{kill:ret}$ was increased from 1.1 to 2 from 2018-2020 and the future $M$ is doubled for the first 20 years of projections (OM3)</td>
<td>The historic high discarding in the late 1980s would occur again in 2018-2020 due to the strong cohorts of 2011-2013 but the discarding will return to 1.1 for 2021-2057. Moreover, the strong cohorts of 2011-2013 may introduce density dependent effects</td>
</tr>
<tr>
<td></td>
<td></td>
<td>due to cannibalism and juvenile natural mortality for the first 20 years of projections</td>
<td></td>
</tr>
<tr>
<td>---</td>
<td>---</td>
<td>---</td>
<td></td>
</tr>
<tr>
<td>13</td>
<td>Sensitivity</td>
<td>Like the base case but it uses an alternative input offset vector (a_{i}^{\text{offset}}) to calculate the historic age-at-50% vulnerability for fishery. The offset was increase/decrease 0.5 respect to the base case.</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>No information was available to estimate the vulnerability-at-age. It was estimated indirectly from age-at-50% retained minus a time-varying offset. The offset take into consideration changes in fishing vulnerability due to discarding</td>
<td></td>
</tr>
<tr>
<td>19-20</td>
<td>Sensitivity</td>
<td>Like the base case but it assumes a lower (OM19) and higher (OM20) prior mean for the strong historic cohorts respect to the base case</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>No data were available to estimate earlier strong cohorts. These historic strong cohorts were included as informative priors. Priors have an impact in model estimates</td>
<td></td>
</tr>
</tbody>
</table>