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Units 1 and 2 Redfish Management Strategy Evaluation

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Foreword

This series documents the scientific basis for the evaluation of aquatic resources and ecosystems in Canada. As such, it addresses the issues of the day in the time frames required and the documents it contains are not intended as definitive statements on the subjects addressed but rather as progress reports on ongoing investigations.

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ABSTRACT

A Management Strategy Evaluation for the Atlantic redfish fishery in Fisheries and Oceans Canada management Units 1 and 2 in the Gulf of St. Lawrence and the Laurentian Channel was carried out. Three conservation objectives and four fishery objectives were formulated based on discussions in meetings between managers, scientists, industry members and other stakeholders. 18 performance metrics were defined to quantify how well candidate management procedures met each of the eight management objectives. A set of 18 operating models were formulated to evaluate the robustness of candidate management procedures to key uncertainties affecting the evaluation of management options. The operating models were age structured and fitted separately to data for the two redfish species, i.e., *Sebastes mentella* and *S. fasciatus*. A Bayesian approach to model fitting was adopted in which informative priors were applied for the steepness stock-recruit parameter for the Beverton-Holt stock recruit function, and historical deviates for cohorts that early research had concluded were strong ones. The models were fitted to time series of bottom trawl survey indices of abundance and length composition from Units 1 and 2, and length composition records of retained redfish catches in Units 1 and 2. Some of the key axes of uncertainty considered in formulating the operating models included uncertainty over fishery vulnerability-at-age, life history parameters, e.g., the rate of natural mortality and growth, the future time series of cohort strength, the magnitude and species composition of historical catches, and plausible values for the steepness stock-recruit function parameter. A set of six core operating models were judged to have higher credibility than the set of 12 stress test operating models which were still credible but according to technical and other considerations judged to be less so. Candidate management procedures were required to pass threshold values for performance metrics for the conservation objectives and were ranked according to the fishery performance metrics for the six core operating models. The management procedures (MPs) were model-free and included three key components. The primary one was a harvest control rule (HCR) which specified a catch limit based on the three-year trailing average of Unit 1 trawl indices. The second specified the start-year for the HCR which varied between 2018 and 2022. The third specified whether a maximum cap to catch limits was to be applied and what the cap should be in each future year. If the MP was capped, a ramp cap was initially applied from the initial year to some intermediate year in the 40-year horizon. Here, the cap was increased in successive years, with the anticipated (1) recruitment to the fishery of the large 2011-2013 cohorts and (2) increases in industrial capacity. A maximum cap was then applied at the end of the ramp. For the conservation objectives, capped MPs outperformed uncapped MPs under most of the core and stress test operating models. Average catches retained and interannual variability in catches were much higher for uncapped than for the capped MPs. Four of the uncapped MPs failed on one of the stock conservation objectives for one of the core operating models. Under the less optimistic core and stress test operating models, for example, historical discarding is lower than that in the base case, there were a number of MPs that passed all stock conservation objectives and provided average future catches of about 24 kt over the next 10-20 years. Projected median total allowable catches (TACs) over the next five years from most MPs were sufficiently low as to constitute low conservation risk to the stocks. Stock conservation objectives that aim to reach in 10 years and then maintain the SSB of both species in the Healthy Zone were met with high probability in all tested MPs. The abundant small fish (<22cm) associated with the 2011-2013 cohorts results in predicted catches from all candidate MPs failing to meet the requirements of the Small Fish Protocol in 2018 and 2019. Small fish are expected to remain abundant in the catch until 2020. Exploratory analysis indicates that the performance of MPs with respect to three conservation objectives was improved and total catches were increased in simulations that assumed that the two species of redfish were perfectly distinguished in fishery catches, enabling species specific TACs.

INTRODUCTION

BACKGROUND

Redfish inhabit cold waters along the slopes of banks and deep channels at depths ranging from 100 to 700 m. *Sebastes fasciatus* is typically found in shallower waters than *S. mentella*. In the Gulf of St. Lawrence and Laurentian Channel, *S. mentella* predominates in the main channels at depths ranging from 200 to 350 m. In contrast, *S. fasciatus* dominates at depths of less than 250 m, along the slopes of channels and on the banks, except in the Laurentian Fan where it inhabits deeper waters. Redfish generally live near the bottom. However, various studies have shown that these species reside near the bottom during the day, leaving the sea floor at night to follow their prey as they migrate (Brassard et al. 2017). Juvenile redfish feed mainly on various species of crustaceans, including several species of shrimp. The adult redfish diet is more diversified and includes fish (DFO 2018).

Redfish are a slow growing and long lived species. *S. fasciatus* grows slower than *S. mentella*, although this difference in growth rates becomes obvious only after 10 years of age. In both species, females grow faster than males after about 10 years of age. On average, it takes redfish seven to eight years to reach the 22 cm minimum commercial size (Gascon 2003). More details are provided in Appendix A.

Unlike many cold water marine fish species, fertilization in redfish occurs internally and females are ovoviviparous (Hamon 1972). Mating occurs in the fall, most likely between September and December, and the females carry developing embryos until they are expelled in spring at the larval stage when they are able to swim. Larval extrusion occurs from April to July, depending on the area and species (Ni and Templeman 1985). Mating and larval extrusion do not necessarily occur in the same locations. In the Gulf of St. Lawrence, *S. mentella* releases its larvae approximately three to four weeks earlier than *S. fasciatus*. The larvae develop in surface waters and juveniles gradually migrate to greater depths as they grow (Brassard et al. 2017).

Fishery Overview (also see Appendix A)

In the late 1950s, a directed redfish fishery developed in the Gulf of St. Lawrence and in the Laurentian Channel outside the Gulf. Prior to 1993, the redfish fishery was managed in three units, based on the Northwest Atlantic Fisheries Organization (NAFO) Divisions: 4RST, 3P and 4VWX Divisions. In 1993, these management units were redefined primarily to account for new knowledge and winter migration of redfish stocks from the Gulf to the Cabot Strait area. The resulting management units are as follows: Unit 1, consisting of divisions 4RST and comprising for the period January to May, subdivisions 3Pn4Vn; Unit 2, consisting of Divisions 3Ps4Vs, Subdivisions 4Wfgj and comprising for the period June to December, Subdivisions 3Pn4Vn; Unit 3, grouping divisions 4WdehklX. In the Gulf of St. Lawrence and the Laurentian Channel, Fisheries and Oceans Canada currently manages these two species as a single stock with two management Units (i.e., Units 1 and 2), although stocks assessments are species specific. This resource is managed mainly by an annual total allowable catch (TAC) which is not species specific. Other management measures (type of gear, area closures to protect fertilization and larval extrusion periods, observers, dockside monitoring, minimum size, bycatch monitoring, etc.) are also applied (Brassard et al. 2017, DFO 2018).

The redfish fishery in the Gulf of St. Lawrence was marked by intense exploitation episodes (1954-56, 1965-1976 and 1987-1992). Bottom and midwater trawls, especially in the early 1990s, are the most used fishing gear. Combined landings of both species and both units have dropped from over 100,000 t in the 1970s to less than 10,000 t since 2004. In Unit 1, the TAC for the redfish stock established under the new management modality, defined in 1993, was 60 000 t. After a rapid fall in landings in 1993 and 1994, a moratorium was declared in Unit 1 in 1995. Since 1999, the TAC has been maintained at 2000 t for the index fishery. Conservation measures include: application of a protocol for the protection of small fish (<22 cm), 100% dockside verification, mandatory hail reports upon departure and arrival, imposition of a level of observer coverage at sea and the application of a bycatch protocol. Closure periods have also been introduced: 1) to protect the mating (fall) and larval extrusion (spring) periods of redfish, 2) to minimize the harvest of Unit 1 redfish migrating from Subdivisions 3Pn4Vn in late fall and winter, and 3) to protect Atlantic cod reproduction (Div. 4RS). In addition, since the introduction of the index fishery in 1998, fishing is permitted only between longitudes 59° and 65° (W) at depths greater than 182 m (> 100 fathoms) to avoid incidental catch of Greenland Halibut, and an area has been closed in Division 4T since August 2009. There has been no moratorium on the commercial fishery in Unit 2 and the TAC has been 8500 t/year since 2006 (Brassard et al. 2017, DFO 2018).

Stock Overview

We provide in this section a brief overview and brief explanation of the definition and status of the redfish stocks in Units 1 and 2. Since there are actually two species of redfish caught in the fishery in these units, and the operating models developed include two single-species population dynamics models, one for each species, we will define two stocks for the purposes of the MSE, i.e., the population of *S. mentella* that occupies Units 1 and 2 as one of the stocks, and the population of *S. fasciatus* that occupies Units 1 and 2 as the other stock.

Species differentiation

More details are provided in Appendix A. An analysis of genetic variation at 13 microsatellite loci was performed on a total of 35 adult individuals (16 *S. mentella* specimens and 19 *S. fasciatus* specimens) harvested in the northwest Atlantic. The results suggest that Units 1 and 2 correspond to a single population of *S. mentella*, characterized by introgression from the other species (Valentin et al. 2014). This population is itself distinct from other *S. mentella* populations distributed in the northwest Atlantic. For *S. fasciatus*, the results suggest the presence of five populations in the northwest Atlantic. A first population of *S. fasciatus* is observed in the region corresponding to Units 1 and 2, excluding the southern margin of Unit 2. This population is characterized by introgression from the other species (Valentin et al. 2014). The *S. fasciatus* samples collected at the southern margin of Unit 2, including the mouth of the Laurentian Channel, belong to a second population of *S. fasciatus*. Its distribution extends along the slope of the continental slope, from the Grand Banks of Newfoundland (3LNO) to Nova Scotia (4W), which we will call "the Atlantic population of the slope of the continental slope". A third population of *S. fasciatus* has been identified in the eastern arm of Bonne Bay Fjord on the west coast of Newfoundland. Microsatellites also revealed the presence of a fourth genetic group in *S. fasciatus*. It groups together three samples (one in each of Units 1 and 2 and one in Unit 3) which, unlike the others, do not correspond to a spatially well-defined population on a regional scale. Analysis of additional samples would be required to document this group. Samples collected in the Gulf of Maine suggest the presence of a fifth genetically distinct population in this region. A detailed discussion is available in Valentin et al. (2014).

Three characteristics can be used to distinguish *S. mentella* from *S. fasciatus* in the Northwest Atlantic: the number of anal fin soft rays (NASR), the extrinsic muscle of the swim bladder (EMSB), and the genotype at the malate dehydrogenase (MDH-A *) locus. In the absence of information on microsatellites, the MDH-A * genotype has historically been considered the genetic criterion of reference. In general, *S. mentella* is characterized by the homozygous MDH-A * 11 genotype, a EMSB between ribs 2 and 3, and an NASR ≥ 8. *S. fasciatus* usually has the homozygous genotype MDH-A * 22, a EMSB between ribs 3 and 4 and NASR ≤ 7. It should be noted, however, that an NASR = 8 is not exceptional for this species; for example, almost 10% of *S. fasciatus* individuals from the Gulf of Maine and Unit 3 (regions where *S. mentella* is absent) present an NASR = 8 (Gascon 2003). The use of these three criteria (MDH-A *, NASR, EMSB) made it possible to describe the geographical distribution of species at the North Atlantic scale. In Units 1 and 2, it was found that *S. mentella* dominates the main channels, whereas *S. fasciatus* prefers shallower depths along channel slopes and on banks with the exception of the Laurentian cone where *S. fasciatus* dominates at all depths (Valentin et al. 2006). Excluding the Laurentian Fan area, data from the summer surveys of Units 1 and 2 indicate that the transition depth between the two species is around 300 m (DFO 2010).

Recruitment

Redfish perform internal fertilization and fertilization and are lecithotropic, that is, the larvae feed exclusively on the yolk of the egg. The copulation takes place in the fall, most likely between September and December. The spermatozoa are then kept in the physiological resting state inside the females until the maturity of the ovaries in February-March (Hamon 1972). Larval extrusion occurs from April to July, depending on the area and species, at the larval stage capable of swimming (Ni and Templeman 1985). Depending on the species and location, the timing of copulation and larval extrusion may vary. In the Gulf of St. Lawrence, *S. mentella* releases its larvae approximately 3 to 4 weeks earlier than *S. fasciatus*. The larvae develop in surface waters and the young migrate gradually to higher depths during their development. Absolute female fertility ranges from 3,330 to 107,000 larvae per female and increases with size of individual (Gascon 2003). Recruitment success in redfish is highly variable, with large year-classes being produced at irregular intervals, e.g., in Unit 1: 1946, 1956, 1958, 1974, 1980, 1985, 1988, 2003 and 2011-2013 (Appendix Table E.1). In addition, the 1985, 1988 and 2003 year-classes of *S. fasciatus*, which were very abundant at ages 2 to 4 from research survey data, were not subsequently detected and never significantly contributed to the fishery, potentially because they bore the genetic signature of the populations of the Grand Banks of Newfoundland that they joined later (Brassard et al. 2017).

COSEWIC and Recovery Potential Assessment

In 2010, the Committee on the Status of Endangered Wildlife in Canada (COSEWIC) designated *S. mentella* from the Gulf of St. Lawrence and Laurentian Channel as endangered and *S. fasciatus* in the Atlantic as threatened in both Units. The results of a recovery potential assessment for each of these populations, conducted in 2011, indicated that the spawning stock biomass of each of the two species was within the critical zone (McAllister and Duplisea 2011). According to 2011 estimates, DFO (2012) established benchmarks and concluded that the breeding stocks of *S. mentella* and *S. fasciatus* of Units 1 and 2 are in the critical zone below their respective limit reference point. The prospects for the redfish stocks of Units 1 and 2 are positive in the short term thanks to the strong cohorts of 2011, 2012 and 2013. These fish would begin to recruit significantly to the fishery from 2018 to 2020, which could lead to a rapid increase reproductive biomass (DFO 2018). Indeed, the abundance of juvenile redfish, largely dominated by *S. mentella*, has increased significantly in research surveys since 2013. In 2017 in the northern Gulf of St. Lawrence, i.e., Unit 1, the abundance of juvenile *S. mentella* and *S. fasciatus* is 60 times and 10 times higher than their respective average abundance for the period 1995-2015. The first strong cohort, that of 2011, had a modal size of 20 cm in the summer of 2017 (DFO 2018). Precautionary approach reference points for 2011 (DFO 2012) derived from a Bayesian model of excess production on biomass mature were revised in December 2015 with only minor changes (McAllister and Duplisea 2016). Based on these benchmarks, the status of both stocks improved. However mature biomasses were still in critical areas. In the summer 2017, the 2011 to 2013 redfish cohorts' modal size was 20 cm. If the anticipated growth of these cohorts continues, close to 50% of the individuals (59% biomass) of the 2011 cohort should be larger than 22 cm in 2018, the minimum commercial size. By 2020, 51% of the cohort (62% biomass) should be larger than 25 cm (DFO 2018).

MANAGEMENT STRATEGY EVALUATION

Management Strategy Evaluation (MSE) is a type of management approach that is focussed on developing and identifying Management Procedures (MPs) that can meet a prioritized set of fishery management objectives and can be expected to perform acceptably under a range of credible scenarios for the future dynamics of the fish stock (Smith et al. 1999; Rademeyer et al. 2007; Punt et al. 2014). The approach requires that fishery stakeholders meet with fishery managers to agree upon a set of fishery management objectives that typically include stock conservation objectives and fishery objectives relating to the economic value and stability of the fishery in future years. Performance metrics are defined which enable the quantification of how well different candidate MPs meet the different management objectives. Computer simulation models of the harvested fish stock (or stocks) and fishery are developed to project the future implementation of candidate MPs accounting for imperfections in the data collected in future years that will be used in harvest control rules (HCRs) that specify particular catch limits given the data fed into them. For example, it is common for indices of abundance to be applied in harvest control rules such that as stock abundance increases, catch limits can increase also and vice versa (Hicks et al. 2016; Jones et al. 2016). MPs include specifications for which data are to be collected and how the data are to be used in controlling future harvests. MPs may also include specifications to caps on catch limits that could be constant or vary over time (Hicks et al. 2016). The caps could be either minimum or maximum cap amounts.

To identify MPs that are robust to uncertainties in the future dynamics of the exploited populations and fishery behaviour, it is common practice in MSE for a number of different "operating models" (OMs) to be formulated that represent different scenarios for how the fish populations and fishery will behave in the future (Rademeyer et al. 2007). It may be the case that the ranking of candidate MPs using computed performance metrics may be sensitive to the OM applied to evaluate MP performance. It is generally agreed that a desirable attribute for a MP is for its performance to be acceptable across all of the OMs considered. If there is a subset of MPs that have acceptable performance across all OMs and all performance metrics, then a single MP can be chosen by ranking the MPs according to the most important performance metrics.

Once a MP is chosen, the MSE approach requires that this MP is adhered to in setting catch limits for a fixed number of years (Punt et al. 2014). While simulation evaluations performed may simulate up to two or three generations of the fish population of interest, it is common for the trial period for the MP to be much shorter and typically not more than about 5 years. At the end of the initial implementation phase, a retrospective analysis would take place in which the actual performance of the adopted MP would be evaluated. If the realized MP performance was found to be acceptable at the end of the initial implementation period, the fishery managers and stakeholders could then embark on a new MSE process to evaluate new candidate MPs and consider implementing a new MP in a new trial period.

Prior to implementation of a MP it is also common practice for a set of so-called exceptional circumstances to be identified in which a decision could be taken for the MP implementation to stop (Punt et al. 2014). These circumstances could include obtaining data outside of the range of those simulated in the MSE projections and catch limits also outside of the range of those formerly simulation tested (Rademeyer and Butterworth 2011). Should such exceptional circumstances arise and a MP be stopped, the management of the fishery would need to be reconsidered and re-evaluated before any new management actions could be taken.

Rationale

In this section we summarize why MSE was adopted for redfish, i.e., among other things, it offered a way to resolve the problem of competing models. Since 2011, model-based stock assessments have been carried out for Unit 1 and 2 redfish (e.g., McAllister and Duplisea 2011; Rademeyer and Butterworth 2015; Duplisea et al. 2016). In these assessments, it has proven difficult to fit the assessment models to bottom trawl survey stock biomass and survey and fishery length composition data. This is due to a variety of issues. For example, commercial catches are not sampled for species composition and assumptions are required to split the combined species catch records by species. There is potential for inaccuracy in the estimation of abundance and stock productivity if there exist systematic biases in the catch splitting by species. In addition, a recent study by Duplisea (2016) suggests that there appears to have been periods in the past in which there was considerable discarding of undersized redfish and the ratio of stock biomass killed to that retained and reported could be considerably higher than 1. Difficulties in species identification appears to have led to between-species contamination of trawl survey abundance indices by species and trawl survey length composition records by species, most especially for small redfish. The presence of large numbers of juvenile *S. fasciatus* from the Grand Banks stock in Units 1 and 2 has also led to contamination of bottom trawl survey abundance index and length composition records. In addition, the constant of proportionality for the bottom trawl survey indices of abundance, estimates of q, obtained from model fitting have been inexplicably large, especially for the bottom trawl survey index of abundance for *S. fasciatus* from Unit 2. There have been several attempts to address this issue (e.g., Rademeyer and Butterworth 2015) but until now, no credible mechanisms have been identified to explain why estimates of the Unit 1 and 2 fishery independent bottom trawl survey q are larger than 1 for *S. fasciatus* and *S. mentella*.

It is recognized that MSE could offer a more effective approach to the management of the Unit 1 and 2 redfish fishery. This focuses attention on the short and long-term objectives for the fishery and seeks input from the industry and other stakeholders on what the different groups want from the fishery. Requirements to meet stock conservation objectives are still maintained and addressed in the MSE analysis. The approach has in a number of instances eliminated conflicts between different stakeholder groups over models. New models that represent plausible hypotheses about stock dynamics and processes generating the data can be proposed and formulated. These can be incorporated in simulation evaluations that are applied to identify candidate MPs that can meet fishery and stock conservation objectives and are also robust to key uncertainties. For these reasons, different stakeholder groups were supportive of the adoption of a MSE approach for the Units 1 and 2 redfish fishery.

Key Challenges for assessment and MSE

- Landings are aggregates of two species
- Uncertainties in species splits in key data
- Contamination of data by 3LNO redfish stocks (*S. fasciatus*)
- Changes in gear over time series and space
- Multiple fleets
- Discarding – high in the 1980s
- Uncertainty over the strength of past strong cohorts

As mentioned above, there exist several challenges to address in a MSE framework that have plagued previous attempts at stock assessment of the two redfish stocks.

1. Landings have been the aggregate of the two main redfish species, *S. fasciatus* and *S. mentella*. The separation of landings records aggregated across the two species has been required for past attempts at stock assessment and for the MSE modeling. The estimates of species composition in fishery independent trawl survey records have been applied in the past to predict the species composition of historical catch biomass and length composition records. However, this has been problematic due to seasonal differences between when the survey has occurred and when the fishery has been prosecuted and potential seasonal differences in species composition in the locations surveyed. The uncertainty in the species composition in the historical catches and affects estimates of historical stock biomass, cohort strength and stock productivity, among other things for the two stocks and we describe below how this challenge has been addressed in the MSE.
2. Survey length composition data from Unit 1, and genetic analyses show that there may be contamination by 3LNO (Grand Banks) redfish stocks (S. *fasciatus*) in the Unit 1 and possibly records of Unit 2 trawl survey samples. In some years, this Grand Banks stock appears to use Unit 1 as a nursery area and migrate out of the nursery area before they recruit to harvestable sizes. This is problematic because there are occasional large Grand Banks cohorts apparent in trawl survey length composition records until about 18 cm which then vanish from the records prior to reaching harvestable sizes. Length structured models that have in the past been fitted to the survey length compositions have produced apparently spurious estimates of abundance and fishing mortality rates in trying to accommodate the length composition data (e.g., Duplisea et al. 2016). We describe further below how this challenge has been addressed in developing operating models for this MSE.
3. Commercial trawl gear has changed considerably over the past several decades. In some years, mid-water trawls were used more commonly. In the last few decades bottom trawl gear has been more commonly in use. Analysis of observer records carried out within the MSE and described below has suggested that the size distribution of fish retained has changed significantly over the history of the fishery. A multivariate statistical analysis also described below has suggested that blocks of years can be grouped into either two or three segments with similar size retention attributes. The MSE has adopted for now the hypothesis that the vulnerability at age to the fishery changed in 1994 and was different before and after this point in time.
4. Following the stock decline in the early 1990s, the redfish quota allocated to Unit 1 was markedly reduced and the remaining fishery has concentrated in Unit 2 since then. The age, stock and species composition has likely been different between Units 1 and 2. Yet insufficient data from the fishery has been available for this MSE to formulate a spatially structured population dynamics model for the two species.
5. There are a number of different fishing fleets that have captured redfish in Units 1 and 2. These can be characterized by smaller vessels that tend to fish nearer to shore than the offshore fleet of typically larger vessels. There are insufficient data to disaggregate the landings data by fleet and thus it has been necessary in this MSE to represent the fishery with a single averaged vulnerability function for each species within each of the time blocks.
6. There has been known discarding prior to the banning of discarding of redfish in Units 1 and 2 in 1995. The incidence of historical discarding has been documented in Duplisea (2016) and the findings of this study have been applied in this MSE to formulate different scenarios for discarding in historical years.
7. It is understood that in the early years of the Unit 1 and 2 redfish trawl fishery, some large cohorts, e.g. 1956, 1958, 1974, and 1980, have been identified (CAFSAC 1984; Gascon 2003; Valentin et al. 2015) (Appendix Table E.1). Prior to about 1980, these cohorts

occurred too early on for stock assessment analysis to provide reliable estimates of their strength. Nonetheless, specific years in which large cohorts were born have been identified (Table E.1). The paucity of data on the magnitude of cohort strength in these early years is problematic since it may be the case that different assumptions made about cohort strength in the early years of the fishery could create different estimates of stock productivity (e.g., stock-recruit parameters). The sensitivity of estimates of stock productivity to this source of uncertainty has been evaluated in this MSE and is reported below.

The OMs that have been formulated for the Unit 1 and 2 redfish MSE attempt to explicitly address most of the above mentioned issues and evaluate the robustness of different candidate MPs to some of these and other uncertainties.

Stakeholder Participation

A rebuilding plan for Units 1 and 2 redfish was initiated in 2014. A series of Working Group and technical meetings occurred 2016-2018 which included stakeholder participation. In December 2016 there was a conference call with Working Group members to agree to initiate the MSE process. In March 2017, October 2017, December 2017, and March 2018, Working Group meetings were held to initiate the development of the MSE. The final list of performance metrics and management procedures was agreed in March 2018. In May 2017, September 2017, October 2017, December 2017, and February 2018 – technical meetings were held to help foster the development of MSE models and the inputs to these models (Deith et al. 2021). Working group participants present at these different meetings have included academics and Provincial and Federal government research scientists, managers and biologists, representatives from harvesters/processors, indigenous groups and conservation organizations.

DATA SOURCES

The data used in the MSE included records of retained catch biomass that were initially not separated by species. The records were later on separated by species based on species splits in the fishery independent trawl survey records in Units 1 and 2. Records of fishery length composition in Units 1 and 2 were also used. These had to be split by species also using fishery independent trawl survey records from Units 1 and 2. Trawl survey abundance indices and length compositions were obtained for Units 1 and 2 from the DFO and GEAC surveys carried out in these different areas. For additional details see Appendix B.

BASE DATA

The base data used in the MSE included a fishery independent bottom trawl survey abundance index for the two species in Unit 1 which extended from 1984-2017. These were treated as relative indices of abundance in the estimation of OM parameters. The trawl survey gear used smaller meshes than the fishery and thus consistently captured smaller and younger fish than those typically captured and retained in the commercial fishery. The associated trawl survey length composition thus provided a fishery independent source of information on cohort strength. Attempts were carried out to remove the apparent large cohorts of Grand Banks *S. fasciatus* from the abundance index and length composition data from the Unit 1 trawl survey. The Unit 1 and Unit 2 trawl surveys are described in detail further below.

Abundance indices from the Unit 2 fishery independent trawl survey were available for fewer than half of the years from 2000 to 2016. These were also treated as relative indices of abundance in the estimation of operating model parameters. Length composition records for the two species from this trawl survey were also available and used for OM parameter estimation.

The 1960-2017 records of catch biomass retained in commercial landings in Units 1 and 2 were split into catch biomass by year for *S. mentella* and *S. fasciatus* using the species composition estimates in the fishery independent trawl survey data. Records of commercial length composition from Units 1 and 2 were also split into composition by species 1984-2016 also using the species split in the trawl survey length composition data. Composition data from 2017 was not included because the records had only been partially compiled for 2017 at the time of the analysis.

Fishery-Independent Data

Since 1984, DFO has conducted a multispecies research survey (groundfish and shrimp) in Unit 1 across the northern Gulf of St. Lawrence using a bottom trawl. The survey covers the waters of the Laurentian Channel and north of it, from the Lower Estuary to the west to the Straits of Belle Isle and Cabot East of the Area, Divisions 4R, 4S and the northern part of 4T of the Northwest Atlantic Fisheries Organization (NAFO) (Figure Appendix A.1). The study area is 116 115 km^2. Over the years, different vessels and fishing gear have been used. From 1984 to 1990, research surveys were conducted aboard the CCGS *Lady Hammond* using a Western IIA bottom trawl. From 1990 to 2005, the ship CCGS *Alfred Needler* and a trawl URI 81 '/ 114' were used. Finally, from 2004 to 2017, the *Teleost* and a bottom trawl Campelen 1800 are used for surveys. These vessels and gear combinations (CCGS *Lady Hammond*, CCGS *Alfred Needler,* and CCGS *Teleost*) were calibrated in in side-by-side comparative tows to establish conversion factors for about 20 species and thus extend the historical series of indices of abundance and biomass of redfish to 1984 to 2017 (Bourdages et al. 2007).

This survey uses a stratified random sampling plan. This technique involves subdividing the study area into more homogeneous strata. The study area is divided into 54 strata of which 52 have been visited each year. Cutting of these was done based on depth, NAFO divisions and substrate type. For this survey, an initial allocation of 200 trawling stations was allocated proportionally to the area of the strata, with a minimum of two stations per stratum. The positions of the stations were determined randomly within each of the strata. For each of the fishing lines, the catch was sorted and weighed by taxon and biological data collected on a sample of redfish: size, sex, number of anal fin soft rays, stomach, otoliths, and tissue samples. A detailed description of the fishing and sampling protocol and the calculation methods are presented in Bourdages et al. (2017). It should be noted that this sampling plan includes the range of redfish included in divisions 4RST, which corresponds to Unit 1.

Between 1997 and 2002, DFO also conducted surveys in Unit 2 with a Campelen 1800 survey trawl towed by the CCGS *Teleost*, with a 12.7 mm liner in the lower 7 m of the codend (Kulka and Atkinson 2016). However, these data are no longer used to assess the redfish stock. The time series for Unit 2 surveys has been replaced by an industry-funded survey that primarily used an Engel 170 trawl with a 30 mm liner in the lower 7 m of the codend, and a 21 m wingspread, towed by large “Cape” class commercial trawlers (45-50 m). This arrangement continued until 2014 when the vessel was switched to the 19 m M/V *Nautical Legend* and the gear was switched to a Campelen trawl with a 15.2 m wingspread. The industry survey began in 1997 and continues to the present (operated by the Groundfish Enterprise Allocation Council or GEAC with input on design from DFO; Kulka and Atkinson 2016). The industry survey is generally biannual although the last few surveys with accurate species composition estimates were completed in 2011 and 2016. Timing of the survey has also changed, as the first survey in 1997 was completed in December with the remainder of the surveys in August/September of each survey year.

Comparative trials between the DFO and GEAC Unit 2 survey gears were completed in August 2000 in order to convert the industry survey data into comparable Teleost-Campelen units

(Cadigan and Power 2010), and again in 2015 following the switch of vessel and gear used by GEAC to a configuration more similar to the DFO survey in Unit 1 (Kulka and Atkinson 2016).

Fishery-Dependent Data

As mentioned above 1984-2017 commercial catch-length data from Units 1 and 2 were split by species using the trawl survey species split for each of the years, using ort samplers and observer-at-sea data.

DATA MANIPULATIONS

Removal of Grand Banks Cohorts

One of the issues with understanding dynamics of Units 1 and 2 redfish is that *S. fasciatus* from NAFO area 3O (Grand Banks) sometimes use the Gulf of St Lawrence as a nursery areas for juveniles of 4 years old and less. After four years, individuals of these Grand-Banks cohorts leave the Units 1 and 2 area to take up adult residence along the shelf edge in NAFO area 3O. The Unit 1 survey will catch these cohorts from age 2 and when the length composition data shows stronger than average year classes from 1985, 1988 and 2003 for *S. fasciatus* and tracks them for up to three years after which they are no longer in the system. Genetics work (Valentin 2015) shows that these cohorts had the genetic signature of 3O fish and not Units 1 and 2 fish. The issue with these cohorts in modelling is that when survey length composition at young ages is used as one of the model inputs, the models try to explain the sudden loss of the cohorts from the system. Depending on the kind of model and its parameterisation, a model may try to generate unaccounted catch at young sizes, e.g., in both fishing mortality (F) and selectivity, or try to change natural mortality rate (M) at age (e.g., Rademeyer and Butterworth 2015). A state-space approach may inflate process and observation error.

Since the genetic work on various *S. fasciatus* cohorts has been thorough and well documented in the primary literature, it was deemed useful to remove these cohorts from survey length composition data. The method employed was relatively simple. Essentially the abundance in length classes between 7 and 17 cm (inclusive) were set equal to the mean abundance in those classes for years adjacent to the arrival and departure of the cohorts. For example, the 2003 year class is present from about 2005-2008 and between lengths of about 7 and 17 cm. Therefore the abundance for 7 cm individuals in all years from 2005 to 2008 was set equal to the mean abundance of 7 cm individuals in 2004 and 2009. Because the Grand Banks cohorts are mostly juveniles when present in the Units 1 and 2 area, it was not necessary to correct mature biomass estimates for Grand-Bank cohort removal. Also, due to indications of species contamination in the Unit 1 trawl survey length composition data for both species, i.e., *S. fasciatus* not being fully separated from the *S. mentella* length composition, apparent large Grand Banks cohorts were also removed from the Unit 1 trawl survey length composition data for *S. mentella*.

Following the adjustments to the Unit 1 survey composition data, the model could fit the survey length composition and survey biomass indices much better than previously (see results section). This was the case for both *S. mentella* and *S. fasciatus* and was especially so for years where Unit 1 trawl survey indices indicated relatively high abundance in the earlier and later parts of the time series. The population dynamics model could more consistently predict the survey length composition data for both species in these years compared to the years where the survey abundance indices had very low values. This is to be expected because where survey biomass was relatively high or about to increase substantially the survey biomass was made up mostly of larger-sized resident fish or a large resident cohort that was recruiting to the mature component of the resident population. Where survey biomass was low, resident adult biomass was much lower and it is conceivable that smaller Grand Banks cohorts could cause more unexplained variability in the length composition data.

Splitting Commercial Fishery Data by Species

The redfish fishery in Units 1 and 2 captures two redfish species, *S. mentella* and *S. fasciatus*, which are so similar in appearance that it is almost impossible to distinguish them from each other by a single visual examination. The unequivocal discrimination of the two species is only possible on the basis of microsatellite markers. To do this, a minimum of 4 loci seems sufficient to assign individuals to the species. However, the systematic use of microsatellites to determine the specific composition of catches would be costly and would require intensive logistics. It is therefore necessary to use a method that makes it possible to estimate the specific composition of the catches using information easily collected at sea. The number of anal fin soft rays (NASR) was chosen as a candidate because it is an easily identifiable meristic criterion whose pattern varies between the two species. This method makes it possible to deduce the proportion of each species in a given trawling station, from the distribution of NASR observed in this same trawling station. To do this, it was necessary to establish the theoretical distribution of NASR for each of the two species. The distribution of NASR by species and by unit was determined on the basis of 4 342 individuals harvested during the Multidisciplinary redfish program in Unit 1 program (in August, 1994-1997, n = 1 562) and in Unit 2 (in July-November, from 1995 to 1998, n = 2,780, Gascon 2003). First, individuals were assigned to the species based on the genotype at the malate dehydrogenase (MDH-A *) locus, considering heterozygotes to be *S. mentella*. Then, for each species, the redfish belonging to each class of NASR were counted. Thus, a count of 7 or fewer radii is characteristic of *S. fasciatus*, whereas *S. mentella* is usually associated with 8 or more rays. However, an 8-ray count can also be observed in *S. fasciatus*. These numbers were then converted to percentages.

In the DFO research survey, the number of NASRs is counted on up to 90 individuals and the observed frequency of NASR for each sample is calculated. Subsequently, the proportion of each species is estimated from the theoretical distributions of *S. mentella* and *S. fasciatus*, minimizing the squared difference between the observed and the theoretical distribution. This method, although extremely practical at sea, still has limitations. For example, enumeration of NASR is less accurate for individuals less than 15 cm.

Life history inputs to the operating models

Growth Rate

The growth rate estimates for *S. mentella* and *S. fasciatus* applied in McAllister and Duplisea (2011) were initially applied in the operating models for these two species in the current MSE. These were for both species 45.8 cm for L_∞ and 0.096 yr^{-1} for K, and -0.5 for t_0 from Saborido-Rey et al. (2004). In fitting the operating models to the data which included fishery independent length composition from both Units 1 and 2, it was found that for *S. fasciatus*, values for L_∞ 10% smaller (i.e., at 41.238 cm) and K 10% larger (i.e., at 0.106) provided improved fits to the data and were retained for *S. fasciatus* only.

It is conceivable that under the extremely large abundance of the 2011-2013 cohorts for both species, growth rate could become density dependent. This scenario was addressed in one of the stress test OMs for the MSE.

Natural Mortality

Natural mortality rates (M) for base case OMs have been set at 0.1 yr^{-1} for *S. mentella* and 0.125 yr^{-1} for *S. fasciatus*. This follows from previous modelling where these rates were taken from NAFO document for Flemish Cap (3M) redfish (McAllister and Duplisea 2011). Natural mortality in Canadian redfish has never actually been measured and value have been borrowed from other studies or estimated within models. In 1999 (DFO 1999) it was written that “the assumption that M=0.1 yr^{-1} for these stocks has been used for over 30 years” in reference to eastern Canadian redfish stocks, i.e., since 1969. M=0.04 yr^{-1} has been used for Gulf of Maine *S. fasciatus* (Miller et al. 2008) as this was the value that gave the best model objective function value and fewer retrospective problems. However, more recent estimates for M for Gulf of Maine *S. fasciatus* directly within a state-space catch at age model (Miller and Hyun 2017) showed M=0.14 yr^{-1}. It was suggested that M might be high for this stock because it is a stock on the southern edge of its range and outside of ideal survival conditions and that the high M estimate in the model may partial reflect unreported catches. A model for Barent’s Sea redfish estimated M at age as free parameters and M on ages 1-7 were on the order of 0.05 yr^{-1} with M increasing to > 0.4 yr^{-1} for age 14 but with an average of around 0.25 yr^{-1} for ages 8-13 (Planque et al 2012). The authors’ admitted that these freely estimated Ms are confounded with q and selectivity but viewed that as a strength as the model fitted the data better. There may also be a confounding with F since M increases with age contrary to life-history and ecological principles. It would appear that these estimates would not be applicable to Units 1 and 2 redfish.

Literature based methods for calculating M for these stocks based on various life-history measures like maximum age, growth rates, maximum size, mean age of maturity suggest that values between 0.08-0.15 yr^{-1} are not unsupported (Figure 1).

The literature based methods were chosen so that they have some applicability to redfish. For instance the Frisk method was developed for long-lived elasmobranch species with low growth rates and is a function of the Von Bertalanffy growth rate (Frisk et al. 2001). The Hoenig method based on maximum age is calculated with ages 50 and 75 years. The former being the oldest age observed in a particular ageing study involving the Units 1 and 2 stock (Campana et al. 2015) while the latter is the oldest age observed (Campana et al. 1990) where there is confidence in the age determination.

It is conceivable that under the extremely large abundance of the 2011-2013 cohorts for both species, the rate of natural mortality could become density dependent. This scenario was addressed in one of the alternative operating models for the MSE.

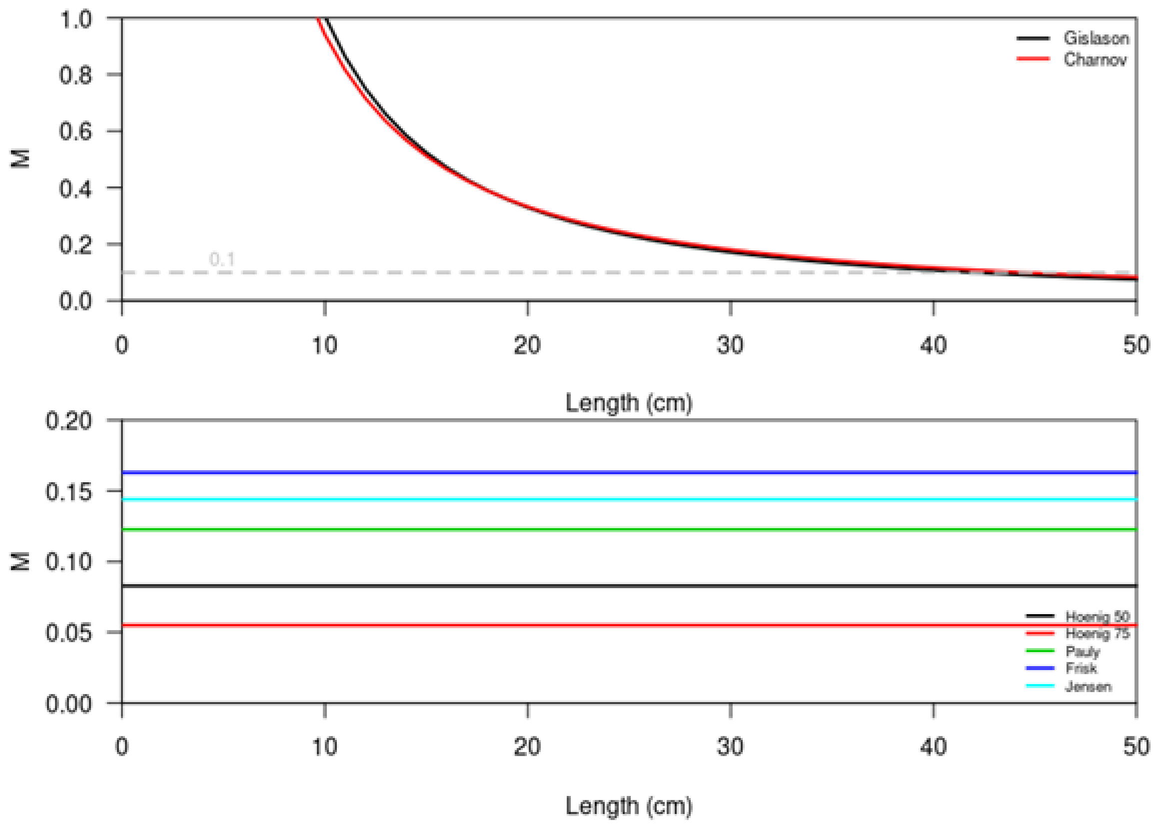

Figure 1. Literature-based estimates of natural mortality from life history characteristics that may have application to Units 1 and 2 redfish. Parameters used were k=0.096, L_∞=45.8 cm, average maximum age=50, maximum age observed=75, mean age at maturity=8, water temperature=3°C. Summaries of most of these methods can be found in Kenchington (2014) and Pardo et al. (2012).

Maturity

The relationship between maturity and length of an individual was derived from data presented in Gascon (2003), where 434 individuals from Unit 1 and 983 from Unit 2 were harvested between 1996 and 1999 species, age, stage of maturity and length were determined. The determination of the number of mature individuals of a given species is based on the proportion mature to length by species and sex according to a logistic curve. In mature females of both species, the shortest length at maturity was around 23 to 24 cm. The length range of female reproducers for the 1996-1999 period was much greater in the case of *S. mentella* than *S. fasciatus*. Indeed, while *S. mentella* females measured 24 to 47 cm in length, very few females of *S. fasciatus* were larger than 35 cm. Generally, males reach sexual maturity one to two years before females (*S. mentella*, 50% of males are mature at 9 years of age (22.8 cm) compared to 10 years (25.4 cm) of females, *S. fasciatus*, 50% of males are mature at 7 years (19.6 cm) compared to 9 years (24.1 cm) in females).

In DFO surveys, a sample of individuals is measured (maximum 200 individuals), sexed (30 individuals), and species identification is based on the number of soft rays of the anal fin. The proportion of mature individuals by species and sex is determined from the sample and extrapolated to the entire catch.

The equation of the logistic curve is in the form:

Mature proportion = $e^{(a + b * L)} / (1 + e^{(a + b * L)})$

S. fasciatus	female	*a* = -10.605	*b* = 0.441	*L50* = 24.1
S. fasciatus	male	*a* = -10.687	*b* = 0.545	*L50* = 19.6
S. mentella	female	*a* = -9.550	*b* = 0.377	*L50* = 25.4
S. mentella	male	*a* = -7.521	*b* = 0.330	*L50* = 22.8

These equations make it possible to determine the fraction of the stock that is mature according to the length of the individuals that compose it. Non-zero mature proportions are predicted for individuals smaller than the minimum size at which mature individuals were observed. Because of this these equations were adapted for the MSE. Based on Gascon (2003), the length of the smallest female being mature was identified (16 cm for *S. mentella* and 12 cm for *S. fasciatus*). All individuals falling under that threshold were designed as immature, whereas the equations were applied to all individuals longer than these minimal lengths to maturity.

Historically Strong Cohorts

Fishery independent bottom trawl survey length compositions are available 1984-2017. These enable estimation of cohort strength no earlier than about 1970. Research documents have indicated years prior to 1970 in which very large cohorts of Units 1 and 2 redfish were identified (CAFSAC 1984; Table E.1). We formulated informative prior distributions for recruitment deviates in years prior to 1970 that had been identified in the published literature to have very large cohorts. This prior distribution was based on a meta-analysis of stock-recruit datasets for stocks of *Sebastes* species obtained from the R.A.M. Myers Legacy database. See Appendix E for details on the method applied to formulate the prior distribution for recruitment deviates associated with a strong cohort.

OPERATING MODELS

BASE CASE MODEL DESCRIPTION

For the stocks of *S. fasciatus* and *S. mentella* in Units 1 and 2, age structured population dynamics models were formulated separately; both models included the same sets of equations for the various functional components (see Appendix D for details and the full set of equations). Given that there was a paucity of spatially structured data for the fisheries in Units 1 and 2, two spatially-aggregated population dynamics models were formulated, one for each fish stock. Also, given the paucity of data on captures for different components of the fishing fleet, the fraction captured and retained at age was modeled for the combination of fishing fleets operating in each year. The modeled abundances at age in each year thus represent the average state of each of these two populations in each year over the combination of the management Units 1 and 2.

A Beverton-Holt stock recruit function was applied to predict the expected number of age 1 fish produced by the spawning stock biomass in each year. This was due to the understanding that cannibalism and other analogous strong compensatory factors at least until recently have not been present in Units 1 and 2 redfish. The fraction maturing at age was represented by a logistic function. The logistic function parameters treated as fixed and known for each species and were obtained from a study of fraction maturing at age for *S. fasciatus* and *S. mentella* in the Gulf of St. Lawrence (Gascon 2003). However, the logistic functions from Gascon (2003) predicted positive fractions maturing for young ages for which no mature fish were detected in that study. Initial trials of the operating model in which this logistic function was applied appeared to over-predict spawning stock biomass in recent years due to this. Minimum ages at which mature fish were detected by Gascon (2003) were ages 4 and 6 years for *S. fasciatus* and *S. mentella*, respectively. Applying these minimum observed ages at maturity in the operating models together with the logistic function from Gascon (2003) for *S. fasciatus* and *S. mentella* avoided this problem.

Annual deviates from recruitment predicted by the stock recruit function, i.e., recruitment deviates, were modeled as lognormal random deviates. A von Bertalanffy growth function was applied to represent length at age with parameter values obtained from published reports on growth of the two species (McAllister and Duplisea 2011). Growth parameter values described in the growth rate section were used. Other estimates (e.g., McAllister and Duplisea 2011) produced implausible estimates for key parameters such as steepness. The former estimates of growth were therefore applied instead of the more recently estimated growth parameter values. But as noted above the estimates for *S. fasciatus* were updated based on fits of the operating model to the data for *S. fasciatus*. A constant fixed value was applied for the rate of natural mortality at age for the two species, i.e., 0.1 yr^{-1} for *S. mentella* and 0.125 yr^{-1} for *S. fasciatus* (see above section on literature sources for life history parameters).

MODELING FISHERY VULNERABILITY-AT-AGE

Vulnerability at age to fishing mortality was modeled using a logistic function for both species. Due to the known historical practice of discarding smaller sized redfish in this fishery, the retained catch was computed by applying a probability retained at age function which was represented by a logistic function (Appendix D). An offset parameter specified the mean difference in the a_{50} parameter for the probability retained and the a_{50} parameter for fishery vulnerability at age functions. Analysis of historical records of retained length frequency from observer coverage, has suggested that the size frequency distribution of redfish retained has shifted in historical years (below and Appendix D). The base case model used the most parsimonious result from this analysis which suggested that prior to 1994, the vulnerability at age was lower than that in 1994 and later years. To address the historical shift in fishery vulnerability, a random walk in fishery vulnerability parameters was tried. However, this failed to provide a stable fit of the model to the data. Fishery selectivity was thus assumed to be stationary over two fixed blocks of historical years, i.e., prior to 1994, and then from 1994 on. Additionally, a fixed value was applied to represent the ratio of catch biomass killed to catch biomass retained in each historical year (Figures 8 and 15). The values applied were formulated by an interview-based study on historical discarding practices (Duplisea 2016).

CONDITIONING THE OPERATING MODELS ON DATA

All of the alternative operating models considered that required changes to historical settings were each fitted to historical data including bottom trawl abundance indices and length compositions, and fishery length compositions, to obtain parameter estimates. It was assumed that in 1950, the abundance of both populations was close to the average unfished equilibrium state. Catch records extend back to 1960 and were small relative to catches in the 1980s and early 1990s. Extrapolations were made to extend the catch time series back to 1950 which presumed catches in 1950 were very small and increased gradually to those recorded in the early 1960s.

A Bayesian statistical catch-at-length stock reduction analysis framework was applied for parameter estimation (McAllister and Ianelli 1997). Parameters were estimated separately for each of the operating models for the two species. Informative prior distributions for some of the parameters, e.g., the Beverton-Holt steepness stock-recruit parameter (Forrest et al. 2010), were applied to facilitate parameter estimation. Each of the operating models was fitted to the available trawl survey biomass estimates for Units 1 and 2, the trawl survey length composition records for Unit 1, and the fishery length composition records combined from Units 1 and 2 from retained catches starting in 1984 (redfish species were impossible to distinguish in fishery operations). Fishery catch-length distributions were missing in Unit 1 but not Unit 2 for the years 1996-1999. This is partly due to the closure of the commercial fishery in Unit 1 but not Unit 2 for these years. The parameters that were estimated included the following: long-term average unfished mature stock biomass, the recruitment compensation ratio, the logistic parameters, and constants of proportionality for the Unit 1 and Unit 2 total survey biomass index, the constant of proportionality for the Unit 1 mature fish only survey biomass index (see Management Procedures section below), the logistic parameters for the fraction retained at age for the commercial fishery for years prior to 1994 and then 1995 and after, and age 1 recruitment deviates from 1947 to 2016.

Prior density functions for estimated parameters are shown in Appendix D. A uniform prior was applied for average unfished spawning stock biomass, B_0. The prior density function for the steepness parameter (which determines recruitment compensation) was obtained from Forrest et al. (2010). Prior to the years where Unit 1 trawl survey composition length data were available, scientific studies and management reports on redfish in the Gulf of St. Lawrence has pointed to evidence of large cohorts in a number of historical years (e.g., CAFSAC 1984; Gascon 2003; Table E.1). The studies reviewed data not used in the current analysis and were thus used to formulate priors for recruitment deviates for years prior to 1970. Years that were not identified to have produced large cohorts were assigned zero prior means for the natural logarithm of the recruitment multiplier. Years that had been identified to have produced large cohorts were given a nonzero, positive prior mean for the recruitment deviate (Appendix E). The prior mean for a year with a large cohort was formulated from an empirical-Bayes type meta-analysis of stock-recruit data for *Sebastes* species. The meta-analysis was carried out to quantify the mean value across *Sebastes* stocks for large recruitment deviates, i.e., recruitment multipliers (mR) with values at least 5, or values of the natural logarithm of mR (i.e., recruitment deviates) of at least 1.609. Stock-recruit data sets for *Sebastes* species were obtained from the Ram Myers legacy website and redfish stock assessment reports. Time series of recruitment and spawning stock biomass (SSB) data were compiled only for those stocks that did not show abrupt discontinuities in time series or time series of constant values (indicating lack of estimation or estimation failure). This left 19 data sets with only one of these being from a redfish stock, i.e., *S. fasciatus* in the Gulf of Maine. The mean value of large recruitment deviates was 2.1 with a standard deviation of 0.56. The average value for the standard deviation in the natural logarithm of recruitment multipliers was 0.84, with a standard deviation of 0.46. The prior mean thus for a large recruitment deviate was set at 2.1. The prior standard deviation for recruitment deviates was rounded up to 1.0, to allow for larger uncertainty due to the low representation of redfish in the *Sebastes* stock recruit data sets included in the meta-analysis. The likelihood functions applied were conventional ones for statistical catch-at-length stock assessment models (Appendix D). The Unit 1 survey and retained commercial catch length composition records were evaluated using multinomial likelihood functions. To give more weight to the survey composition records than the fishery composition records, an effective sample size of 25 was assigned to each annual survey composition record and 10 to each annual fishery composition record. The Unit 1 survey biomass and Unit 2 GEAC trawl survey biomass indices were evaluated using lognormal likelihood functions. The same error variance was applied to the Unit 1 and 2 survey biomass indices.

To check that the coding was done correctly, the operating model was coded up for *S. mentella* first in Excel and parameter estimation was carried out using Frontline Solver. The same model was built in ADMB and then fitted to the same data. After some debugging, practically identical parameter estimates were obtained, i.e., differences between posterior modal parameter estimates from ADMB and Excel were all less than 1%. ADMB was then applied to compute posterior standard deviations in parameters and posterior correlations between parameters. The above described population dynamics model for *S. fasciatus* was also coded up and fitted using ADMB. A positive definite Hessian matrix was obtained when calculating approximations of the posterior standard deviations and correlations between parameters, indicating that the function minimizations using ADMB were successful for both *S. fasciatus* and *S. mentella*. Posterior modal and standard deviations for the base case operating model parameters can be found in Tables 5 and 6 for the two species.

SIMULATION OF FUTURE POPULATION AND FISHERY DYNAMICS

It is considered good practice to represent parameter uncertainty for MSE with the use of Monte Carlo algorithms for Bayesian integration (e.g., Cox and Kronlund 2016). Numerous attempts were made over a two month period to apply the MCMC algorithms in ADMB to generate Markov Chains that could be used to form samples of parameters from the posterior distribution for operating model parameters. Markov Chains of parameter values from computer runs that took up to a few days were generated numerous times using ADMB software. However, despite trying numerous adjustments to the Markov Chain options in ADMB and very long run times, diagnostic analysis of the chains indicated that they were failing to represent the marginal posterior distribution. For example, plots of chains showed high instability in the chains and consistent posterior density functions could not be obtained from large sequences of parameter values from different segments of the Markov chains that were generated. The autocorrelation in parameter values in the chains was very high and it appeared that they remained in burn-in mode even after very long runs. This convergence failure of the Markov Chains tried could have resulted from the extremely strong but infrequent cohort signals in the length composition data (e.g., with point estimates of recruitment multipliers over 500x for 2011 for *S. mentella*), and poor quality of the commercial length composition data. These have been derived from splitting combined composition combined for the two species, and were derived only from sampling of retained catches. There were no composition data available for each species hauled aboard at sea.

To represent parameter uncertainty in the operating models, we used ADMB software estimates of the posterior modal values of parameters and the inverse Hessian formulation of the variance – covariance matrix for parameters. We used these ADMB outputs, i.e., posterior mode and posterior variance-covariance matrix obtained from the Hessian Matrix, to form multivariate normal (MVN) approximations of the joint posterior density functions of parameters. The posterior mode from ADMB function minimization was used to characterize the mean of the MVN, and the variance-covariance matrix of the MVN was taken from the Hessian Matrix computed at the posterior mode by ADMB software.

To carry out projections of the operating models in the evaluation of candidate management procedures, independent draws were taken from the multivariate normal distribution approximation of the joint posterior density function of parameters. In all of the fits of different operating models to the data, positive definite Hessian matrices were obtained. The posterior standard deviations on key parameters were plausible given the priors and fits to data obtained and the posterior correlations between parameters were never larger than about 0.9 in magnitude, indicating that the fitted models were not over-parameterized. Recognizing that the MVN approximation was a fairly crude approximation of the joint posterior, the parameter values drawn were applied first in a projection from 1950 to 2017. If the biomass projected did not survive to the present, due to the parameters drawn being too pessimistic, the draw was discarded. Only parameter draws that allowed computed stock biomass to survive to the present were retained for future projections.

Fisheries Selectivity Blocks

In order to define periods of similar redfish historical fishery selectivity, commercial landings data across gear, vessel size, month, and NAFO zones were compiled for Unit 1s and 2 between 1985 and 2015 (Appendix D). First, a principal components analysis was conducted on each of the four matrices describing fishery characteristics (gear, vessel, month, and zone). The two first principal components (PC), explaining between 89 and 99% of the variation in the matrices, were extracted and retained. This step allowed the representation of each matrix with the same number of variables because cluster analyses are sensitive to the number of variables included (Legendre and Legendre 2012). Therefore, the resulting 8 principal components were used into a *k*-mean clustering. This method iteratively creates groups aiming to partition *n* observations into *k* clusters in which each observation belongs to the cluster with the nearest mean. The SSI criterion was minimized and the clustering that seemed the most appropriate was 2 or 3 periods. The most important split was between 1993 and 1994, and a second was between 1989 and 1990. The characteristics that had the highest contrast in each period are as follows; between 1985 and 1989, bottom trawl was the gear that was the most utilized, whereas it was midwater trawl between 1990 and 1993. From 1994 until recent years, the moratorium in Unit 1 defined the last historical period, where landings decreased drastically. This analysis was conducted with *Sebastes* spp. and different ways to split the total commercial landings by species. In all cases, similar results were obtained.

Simulation of Future Recruitment

To enable the projection of credible future scenarios for recruitment, it was necessary to study the time series attributes of the estimates of historical recruitment deviates for *S. mentella* and *S. fasciatus*. Some of the key attributes of the time series of historical recruitment deviates were as follows: (1) There is a high positive correlation (about 0.74) between recruitment deviates for *S. fasciatus* and *S. mentella*. (2) The lag 1 autocorrelation in recruitment deviates is fairly high for both species, i.e., about 0.5 for *S. fasciatus* and 0.34 for *S. mentella*. (3) For both species there were no more than four large recruitment deviates (multipliers larger than 5) within a forty-year period. (4) For both species there could be up to two or three strong cohorts produced within a three-year sequence. (5) For both species, after a short sequence of strong cohorts had occurred, the next strong cohort occurred no sooner than eight years after the last strong cohort in the previous sequence of strong cohorts. Based on our study of recruitment deviate estimates we formulated a conditional non-parametric bootstrap procedure to generate future time series of recruitment deviates. To evaluate the sensitivity of simulation evaluation results to the method applied to simulate recruitment deviates, we formulated an alternative conditional parametric bootstrap protocol to generate time series of recruitment deviates. See Appendix D for a summary of the two alternative conditional bootstrap procedures used to generate future recruitment deviates.

IMPLEMENTATION OF MPS IN CLOSED-LOOP SIMULATION

The closed-loop simulation was coded up in R-statistical software. The software reads in vectors of parameter values drawn from an approximation of the joint posterior density function for operating model parameters. For each future simulation, a vector of forty recruitment deviates is also drawn using the conditional non-parametric bootstrap methodology described in Appendix D (and for one of the stress test operating models , a conditional parametric bootstrap).

Because the species composition of the Units 1 and 2 redfish catch is not assessed the implementation of future catches needed to account for uncertainty in the catch biomass by species killed and fishing mortality in each future year. The combined-species catch limit was split to a potential retained amount by species using a bootstrap of the range of historical values for the proportion of *S. fasciatus* in the catch retained and reported 2000-2017. To be consistent with the most recently assumed ratio of catch biomass killed to catch biomass retained and reported, the catch limit values split to species in each future year were multiplied by a factor of 1.1 to compute the total catch biomass killed by species. Harvest rates by species could then be computed by dividing the catch biomass killed by species by the vulnerable biomass by species (see Appendix D). Providing that the calculated catch biomass killed per species was less than 95% of the vulnerable biomass for the two species, it was assumed that the catch limit amounts specified by future candidate harvest control rules would be retained and reported by the fishing fleet with 110% of that biomass being killed and 10% but not retained, e.g., lost from nets, spoiled, or caught as unreported bycatch in other fisheries.

MAIN AXES OF UNCERTAINTY

Numerous sources of uncertainty were addressed in the MSE for the Units 1 and 2 redfish fishery; these were grouped into five main axes of uncertainty. We have listed these according to our understanding of their order of importance in determining the fate of the redfish fishery in future years and the performance of the candidate management procedures. There exist yet other types of uncertainty that were not addressed in this MSE, e.g., concerning ecological interactions between redfish and other fishery resources. However, within the current redfish MSE time was not available to develop credible operating models to represent these other types of uncertainty. For example, potential ecosystem interactions may be highly complicated and would require further in-depth research before credible and statistically rigorous models to represent them could be developed for the purposes of MSE.

Axis 1: the timing of future episodes of large cohorts. These have occurred with a frequency of about four times in four decades for both species but with as long as a thirty year gap between episodes of strong recruitment. While oceanographic factors have been hypothesized, understanding of the potential mechanisms that favour strong cohorts remain poorly understood and unpredictable. The occurrence of large cohorts is the main source of biomass production for the fishery and due to the low rates of natural mortality, whether or not one or more strong cohorts will appear within the next five or ten years will determine how fast the predicted redfish biomass surplus from the recent strong cohorts will be depleted.

Axis 2: uncertainty in the species split in historical catches. The lack of a direct measure of species composition in historical catches creates a scaling problem for the size and productivity of each of the two stocks in the past and in the future. Estimates of cohort strength and current stock biomass of the two redfish species are strongly determined by values assumed for the species split of the catch in historical and recent years.

Axis 3: key life history attributes of the two redfish species. Due to difficulties with obtaining age samples of redfish of the two species, there remain uncertainties in the rate of natural mortality of the two species and whether it can vary systematically with age, and whether it could follow a Lorenzen pattern in which it decreases inversely in proportion to body size. Growth estimates also remain uncertain for the two species. Direct analysis of trawl survey records suggest a smaller length infinity (L_∞) parameter than estimates for redfish in other parts of the North Atlantic and those supported in fitting the operating models to the historical survey length composition records. There also remains uncertainty over whether there could be density dependence in growth and the rate of natural mortality especially in highly abundant redfish cohorts.

Axis 4: the extent of fishery targeting of larger sized fish and discarding of smaller-sized fish, especially in the past. It is known that at certain times in the past considerable discarding of smaller sized fish occurred; this was documented in a recent interview-based study (Duplisea 2016). The magnitude and extent of discarding of small sized redfish in historical years remains poorly understood but Duplisea (2018) made some attempts to quantify some of these aspects of historical discarding. Uncertainty over historical discarding creates uncertainty over estimates of historical fishing mortality rates, and historical and current stock abundance and productivity. In addition, uncertainty exists over hypotheses about the relative vulnerability of older larger fish which could give rise to either dome-shaped or asymptotic vulnerability at age functions.

Axis 5: the magnitude of recruitment compensation. In the MSE literature, uncertainty over the recruitment steepness parameter, which indexes recruitment compensation, has often been found to be one of the most influential types of uncertainty affecting operating model predictions (Punt et al. 2014). The values assumed for the steepness parameter has for example strongly determined the ranking of management procedures and the extent of the trade-off between fishery catch objectives and stock conservation objectives (Edwards 2016). However, with Units 1 and 2 redfish, data suggest that recruitment compensation may be fairly low for both species and slightly higher for *S. fasciatus* and that uncertainty over the extent of recruitment compensation is dwarfed by uncertainty over how soon another strong cohort will materialize. It was found that by imposing a Lorenzen schedule for M for one of the core operating models which increased M for younger fish, this created more productivity and the fitting of steepness compensated with a lower value for steepness. By keeping M fixed and lowering steepness in the stress test, this forced lower productivity than seen in the base case and Lorenzen M case, for example, the u_{msy} estimate was lower for the low steepness scenario than the Lorenzen M scenario (see below).

OPERATING MODELS

Based on discussions at Working Group and technical group meetings, several alternative operating models were proposed to represent a limited set of credible scenarios under the five axes of uncertainty outlined above. The chief aim of applying different operating models within an MSE is to test the robustness of candidate management procedures to credible sources of uncertainty in the fishery of interest (Edwards 2016). The alternative models represent alternative hypotheses about attributes of the fish populations and fishery that exploits them and can represent different versions of what may be conceived to have happened in the past and will happen in the future, or keep the same representation of the past but entertain different scenarios for future fishery and population dynamics. In keeping with other recent MSE exercises (e.g., for Canadian Atlantic Pollock; Rademeyer and Butterworth 2011), the alternative operating models were grouped into two categories: core models and stress-test models. The rationale for this categorization of operating model types and details on the different operating models formulated are provided in the two following subsections.

Core Models

Candidate management procedures will be required to perform acceptably under this set of operating models if they are to be considered for implementation. Core models represent credible alternative hypotheses for how the fishery and stocks have behaved or will behave, are considered to represent the most important axes of uncertainty and are considered to be credible from the standpoint of scientists and stakeholders familiar with the fishery. For example, redfish in Units 1 and 2 have been observed to undergo long periods of low recruitment lasting up to a few decades, e.g., from 1958-1971 and 1980-2010 (Figures 9 and

16) giving credibility to the possibility that long drought in strong cohorts could occur following the one in 2011.

In most instances where it has been reliably estimated, natural mortality rate has been found to decay consistently with age and there is a well-understood ecological basis for this (Lorenzen 1996; 2000). Lorenzen-type natural mortality (Lorenzen-M) may occur also with *Sebastes* species such as redfish. It is of interest to consider Lorenzen-M for Units 1 and 2 redfish due to the very large 2011-2013 cohorts that have been detected since 2015 and the possibility that higher rates of natural mortality could cause cohort strength to decrease by the time they recruit to the fishery in 2018-2020.

We have formulated a conditional nonparametric bootstrap based on the attributes of recruitment deviates for the two redfish stocks in Units 1 and 2. However, it is more common to apply conditional parametric bootstrap approaches to simulate future recruitment deviates and it has been found that ranking of candidate MPs can be sensitive to how recruitment is simulated (Cox and Kronlund 2016; Szuwalski and Punt 2016). We have thus introduced another core operating model that applies a conditional parametric bootstrap to simulate future recruitment deviates (Appendix D).

A driving factor in the estimation of stock-productivity and abundance for Units 1 and 2 redfish is how catches are split by species. In using the trawl survey records of catch split, two alternative scenarios for catch split were formulated which were still credible based on the uncertainties in the species split in the trawl survey. The assumed time series of catch split by species may also affect the evaluation of performance of management procedures. Operating models based on these two additional scenarios for catch split thus formed two additional core operating models.

The alternative core operating models are expected to not differ strongly in their information content when fitted to the data. 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).

Stress-Test Models

These models are considered to be plausible alternative representations of fishery and stock behaviours but have less scientific credibility than the core and base case models. For example, with some alternative or additional process conjectured to have occurred in the past or to occur in the future, a particular mathematical specification for a model structure was formulated and tinkered with to obtain an acceptable fit to the data. However, with relatively little or no research conducted on the possible mechanisms and without sufficient data available to guide the development of the model component, the new candidate operating model remains purely speculative and will not have the same weight of credibility as one of the core models. For example in the scenario in which it is presumed that the M will increase substantially in coming years due to density-dependent processes, we formulated a model in which the rate of natural mortality doubled for the next twenty years for all redfish of ages 1 and up, and then returned to historical levels. This model implementation is highly arbitrary for several reasons. For example, it is unclear how much M in age 1+ redfish could increase under density-dependence as abundance increases with the large 2011-2013 cohorts, which ages could be affected, and for how long higher M could persist. Also, despite there being numerous statistical catch-at-age analyses of other *Sebastes* stocks, the co-authors' know of no instances for any *Sebastes* population where the rate of natural mortality of recruited fish (age 1+) varied measurably over time or was density dependent. Some evidence of cannibalism in Units 1 and 2 redfish has been obtained. Juvenile redfish have been observed in some mature redfish stomachs. This would suggest that a Ricker stock-recruit function could be considered as an alternative to the Beverton-Holt stock-recruit function but would not suggest that a doubling of natural mortality rates for all age 1+ fish for the next 20 years could be credible. For these reasons, the Unit 1 and 2 operating model with M doubled can only be considered to be a stress test model and not a core model.

Where dome-shaped vulnerability at age was proposed, there were no specific mechanisms in the Units 1 and 2 redfish fishery known to support this hypothesis. Larger fish for example could not be expected to be better at avoiding trawl nets or to dwell in places less frequented by the trawl fishery. Yet, while the model fitted the bottom trawl length compositions without overpredicting the right hand tail, the model tended to over-predict the fishery catch-length composition for both species and suggested the possibility of dome-shaped vulnerability or a growth curve that created fish at age larger than observed. Without any research to support plausible mechanisms for dome-shaped vulnerability, these operating models must also remain stress test models. This is so, even if these models happened to give lower AIC scores than the Core models.

It is desirable that candidate MPs that perform well under the Core set of models will also perform acceptably well under the Stress test set of models. If the best performing management procedures under the core model set do not perform acceptably well under the stress test models, this will be noted. Should it be discovered in the future that a stress test model becomes more credible than those in the core set and the MP applied had not performed acceptably under the stress test model and in practice, this may give rise to an instance of exceptional circumstances in which a new round of MSE analysis may be performed to identify a new MP which is found to perform acceptably under the updated set of core models.

Table 1. List of core, stress and sensitivity test models formulated for the Units 1 and 2 redfish MSE.

Model	Type	Description	Details
1	Core	Base Case	Model assumes fishery selectivity is logistic, and change in selectivity over time is described by two time blocks over the time series (early years to 1993, and 1994 to present). Simulated recruitment will show similar patterns to patterns seen in the past, using a nonparametric bootstrap of recruitment events from the historical time series. In projections, catch killed = 1.1*catch limit from HCR
6	Core	Reduced future recruitment	Simulations will assume there will be no strong cohorts for the next 20 years.
8	Core	Alternative M	This model will use a Lorenzen M function (where natural mortality rate, M, varies with fish size and is higher for smaller fish) instead of a single value.
9	Core	Alternate recruitment simulation method	Use a parametric bootstrap of historical recruitment for simulations, with the variance and autocorrelation coefficient estimated for recruitment events in the historical time series since 1970.

Model	Type	Description	Details
10	Core	Alternative catch split	Historical catch splits differ from base case – assume more *S. mentella.*
11	Core	Alternative catch split	Historical catch splits differ from base case – assume more *S. fasciatus.*
2	Stress	Alternative fishery vulnerability	Assume fisheries and survey vulnerability are dome-shaped, i.e., double-logistic for both species.
3	Stress	High future M	Future M is doubled for both species, for the next 20 years only. This is a way to examine density dependence during periods with strong cohorts.
4	Stress	Reduced future growth	Simulate a reduction in future growth of both species for the next 20 years by reducing the asymptotic length (L_∞) to a value 2/3 as large as in the base case, while assuming same value for K (L_∞, K are parameters in the von Bertalanffy growth equation). This is another way to examine density dependence during periods with strong cohorts.
5	Stress	Prolonged reduced future recruitment	Simulations will assume no strong cohorts for 40 years.
14	Stress	Alternative for catch killed: retained ratio	Use a different set of assumptions for the values for the ratio of catch killed to catch retained than the base case (+/- 0.5).
15	Stress	Alternative M	Reduce historical and future M by factor of 0.75 in both species.
16	Stress	Alternative M	Increase historical and future M by a factor of 1.25 in both species.
17	Stress	Alternative steepness	Assume that the steepness of the stock-recruitment relationship is higher than in base case by factor of 1.25.
18	Stress	Alternative steepness	Assume the steepness of the stock-recruitment relationship is lower than in base case by factor of 0.75, keeping M at the base case.
22	Stress	Alternative fishery Vulnerability	Fisheries vulnerability for 2017-2021 reverts back to the levels associated with the fishery prior to 1994.
23	Stress	High discarding rates 2018-2020 under OM1	Using the base case operating model 1, assume the ratio of catch biomass killed to catch biomass retained in 2018-2020 is 2 and then returns to 1.1 for 2021-2057.

Model	Type	Description	Details
24	Stress	High discarding rates 2018-2020 under OM3	Using the stress test operating model 3 (natural mortality rate doubles for next 20 years), assume the ratio of catch biomass killed to catch biomass retained in 2018-2020 is 2 and then returns to 1.1 for 2021-2057.
13	Sensitivity	Alternative for offset	Use a different value for offset in median age of fish killed and the median age of fish retained. (+0.5 to base, -0.5 to base).
19	Sensitivity	Alternative prior mean for strong cohorts	Assume a lower prior mean for strong historical cohorts (i.e., a lower prior means these strong cohorts will be smaller in this model than in the base case).
20	Sensitivity	Alternative prior mean for strong cohorts	Assume higher prior mean for strong historical cohorts (i.e., a higher prior means these strong cohorts will be larger in this model than in the base case).

Sensitivity Tests

To evaluate the sensitivity of parameter estimates in the base case operating models to the time series of data applied for estimation, a retrospective analysis was conducted for each species. In the retrospective analysis, the operating model was fitted to the data and parameters were re-estimated when one year of data was successively removed going back 10 years from the present. Estimates of annual recruitment, annual harvest rates, and spawning stock biomass were obtained and plotted.

To evaluate the sensitivity of parameter estimates to the type of likelihood function for the length composition data, a multivariate logistic likelihood function was tried in place of the multinomial likelihood function. Fits to the data similar to those obtained with the multinomial likelihood function could be obtained using the multivariate logistic likelihood function. However, the multivariate logistic function returned estimates of steepness close the upper bound of 1 for the parameter, e.g., 0.98 for *S. mentella* compared to no more than about 0.6 based on the multinomial likelihood function. We were unable to resolve why the multivariate logistic likelihood function gave such high estimates of steepness. We thus used the multinomial likelihood function for the analysis.

The different operating models considered that require new fits to the data test the sensitivity of estimates of parameters and stock status to the key axes of uncertainty outlined above. These included evaluations of the sensitivity of stock biomass reconstructions to different priors placed on historical strong cohorts, different inputted values for steepness and the rate of natural mortality, whether natural mortality is constant or declining with age, uncertainty over the species splits in the historical records of catch biomass, uncertainty over how time blocks are configured for vulnerability at age, the catch killed to catch retained ratio, the offset between the a50 parameters for catch retained and catch killed, whether vulnerability at age is dome-shaped or asymptotic, and whether discarding rates could increase when the strong cohorts start to recruit to the fishery.

Changes in area occupied by the stocks and interpretations of trawl survey indices

An unavoidable consequence of fitting stock assessment models to the Unit 1 and 2 trawl abundance indices (I_y) has been that the estimates of the constants of proportionality for the indices (q where (ignoring subscripts for each trawl survey) $I_y = qB_y$ and B_y is the stock biomass vulnerable to the trawl survey in year y, see Appendix D for details) for both species have ranged 0.3-4, with estimates of q for the Unit 2 trawl survey index being typically the largest for *S. fasciatus*. It has commonly been expected that estimates of constants of proportionality for swept area biomass obtained from fishery independent trawl surveys should be no larger than about 1 (Millar and Methot 2002; McAllister et al. 2010). Reasons why estimates of q have been larger than 1 for trawl survey indices of abundance in Units 1 and 2 redfish remain poorly understood. However, the larger than expected values for q appear to result from the following: 1) over the years with the largest catch biomass removals the annual decreases in the Unit 1 trawl survey index have been systematically larger than the reported annual catch biomass; for example, for *S. fasciatus* and *S. mentella*, in the six years with the largest annual decreases in the index in the late 1980s and early 1990s the annual decreases in the Unit 1 index were 1.1-2.3 times and 1.4-4.3, respectively, times the recorded annual catch biomass taken in those years, 2) the Unit 2 trawl index started in 2000 well after the largest decreases in the Unit 1 index occurred and when the abundance has appeared to be relatively stable and catch removals have been relatively small, and 3) the values for the Unit 2 trawl index have on average been considerably larger, e.g., for *S. fasciatus* and *S. mentella*, 10x and 5x the Unit 1 trawl index in the same years where both indices are available (i.e., from 2000-2016). See Figure 2 for plots of the reported retained catch biomass for *S. mentella* and *S. fasciatus* and the DFO trawl survey swept area biomass and GEAC swept area biomass time series for *S. mentella* and *S. fasciatus* Units 1 and 2.

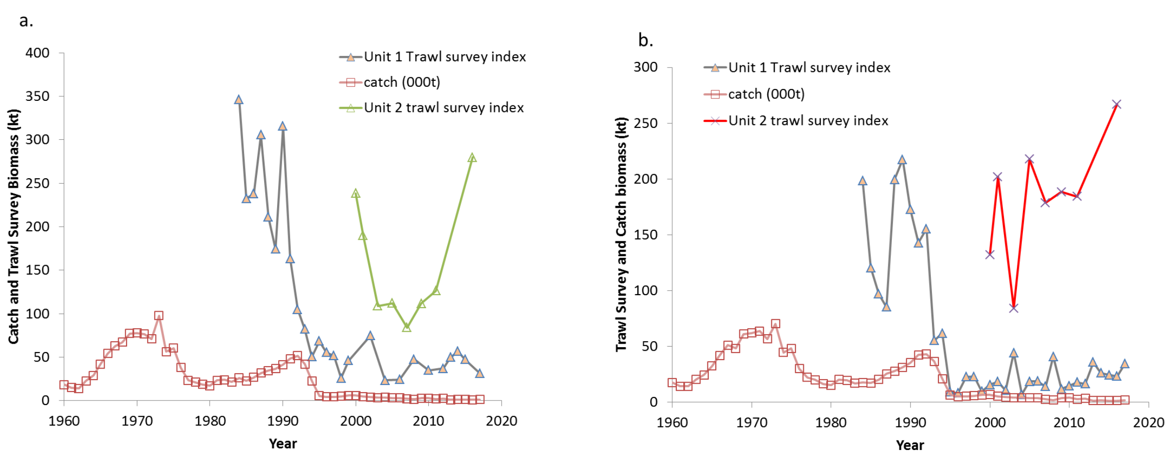

Figure 2. Reported retained catch biomass and DFO and GEAC trawl survey indices for the large fish portion of the survey catches (> 29 cm S. fasciatus and > 30 cm S. mentella).

The amplification of the q in the Unit 2 trawl survey index appears to come from using the two indices as relative indices of abundance and the 10-fold and 5-fold larger value for the Unit 2 index in years of overlap following the large drops in the unit 1 trawl index for *S. fasciatus* and *S. mentella*, respectively. For example, for *S. fasciatus*, the 10-fold higher value for the Unit 2 index than the Unit 1 index in the years of overlap of the two surveys would imply annual decreases in the Unit 2 index in the late 1980s and early 1990s 10x the annual decreases in the Unit 1 index. If the Unit 1 and Unit 2 trawl survey indices were both directly proportional to stock abundance as assumed, the ratios of the implied annual decreases in the Unit 2 index to annual catch biomass reported could therefore be expected to be about 10x the same ratio for Unit 1, thus inflating the estimate of q for the Unit 2 index by a factor several times that for Unit 1. See Appendix G for details on some exploratory analyses that had been carried out to improve understanding on why trawl survey q estimates for Units 1 and 2 redfish have been high and some suggested approaches to developing operating models and management procedures to address this issue. There was insufficient time available within the time frame of the MSE to complete the development of operating models and management procedures that could satisfactorily address the issue of high estimates of trawl survey q.

OBJECTIVES AND PERFORMANCE METRICS

Stock conservation objectives and fishery objectives were formulated in consultation with redfish stakeholders and fishery managers in redfish working group meetings in March 2017, October 2017 and December 2017.

STOCK OBJECTIVES

See Table 2 for descriptions of the three candidate stock objectives and associated performance metrics for application in the redfish MSE. Passing criteria are suggested for each of the stock performance metrics.

FISHERY OBJECTIVES

See Table 2 for descriptions of the six candidate fishery objectives and associated performance metrics for application in the redfish MSE. Passing criteria had been suggested for each of the fishery performance metrics. It was not possible to agree upon and justify in Working Group deliberations passing criteria for the fishery performance metrics. The suggested passing criteria for fishery performance metrics were therefore withdrawn in favour of ranking according to each of the fishery performance metrics instead.

Table 2. Stock and fishery objectives for the Units 1 and 2 redfish MSE. Note that objective 5 was removed in the final Working Group meeting in March 2018.

	Stock Objectives
1	Increase SSB of each of *S. mentella* and *S. fasciatus* above the lower reference point (LRP) and into the Healthy Zone in 10 years (95% probability).
Corresponding Performance Metrics	1a: Proportion of simulations where SSB of each species exceeds LRP in 10 years. PASS CRITERIA: 95% or higher 1b: Proportion of simulations where SSB of each species exceeds USR in 10 years. PASS CRITERIA: 95% or higher
2	Once in Healthy Zone, maintain SSB of each of S. mentella and S. fasciatus above the Critical Zone (95% probability), and in the Healthy Zone (75% probability).
Corresponding Performance Metrics	2a: Proportion of years the SSB of each species is above the LRP after 10 years. PASS CRITERIA: 95% or higher 2b: Proportion of years the SSB of each species is above the USR after 10 years. PASS CRITERIA: 75% or higher
3	Maintain exploitation rate U of S. mentella and S. fasciatus below Umsy, 50% probability.

	Stock Objectives
Corresponding Performance Metrics	3: Proportion of years where the ratio of U: Umsy by species < 1. PASS CRITERIA: 50% or higher

	Candidate Fishery Objectives
4	Maximize the number of years where fish < 22 cm represent < 15% of catch (and Small Fish Protocol is not triggered).
Corresponding Performance Metrics	4: Mean number of years where fish < 22 cm represent < 15% of catch. a) 5 years; b) All 40 years;
5	Maximize the number of years where fish < 25 cm represent < 15% of catch.
Corresponding Performance Metrics	5: Mean number of years where fish < 25 cm represent < 15% of catch.
6	Maximize the duration of high annual catch.
Corresponding Performance Metrics	6a: Average annual catch in a) 10-20, b) 10-40 years. 6b: Proportion of simulations where catch limit reaches or exceeds 40,000 tons by 2028 (i.e., after the large cohorts are expected to fully recruit to the fishery). 6c: Mean number of years where the catch limit is as large as or larger than 40,000 tons in the years 2028-2057. 6d: Proportion of years with > 2017 landings [2017 will be a reference year; note 2016 catch killed = approx. 4040 tonnes assuming 1:1 catch kill: retained ratio]
7	Maximize catch of large fish (>27 cm).
Corresponding Performance Metrics	7: Proportion of years where percentage of fish > 27 cm is > 80%. a) 5 years; b) All 40 years;
8	Maintain the stability of the fishery (annual changes in TAC are consistent with industrial capacity).
Corresponding Performance Metrics	8a: Percentage of years where recommended TAC is < 15% different from previous TAC. 8b: Average Annual Variation in TAC (percentage) during a) 10-20 years; b) 10-40 years.

REFERENCE POINTS

Given that the Unit 1 trawl survey has been taken in every year since 1984 and consistently shows periods with very different levels of stock biomass, biomass reference points were chosen based on indications from the Unit 1 trawl survey of years where stock biomass and fishery yields remained relatively high. Model-based reference points were not used as biomass reference points due to the large influence of cohort strength on stock biomass for Units 1 and 2 redfish and the preference among working group members to use times in the past when biomass levels were observed to be relatively high and stable. An empirical method was thus used to define reference points based on the 1984-2017 time series of survey data from Unit 1. For the Units 1 and 2 redfish stock assessment, the average mature biomass of the stock during the 1984-1990 and 1984-1992 reference periods were considered as proxies for B_{MSY} for *S. mentella* and *S. fasciatus,* respectively. Reference points set at 40% (Limit Reference Point, LRP) and 80% (Upper Stock Reference, or USR) of B_{ref} were 148 kt and 297 kt for *S. mentella*, and 132 kt and 263 kt for *S. fasciatus*. However, in the MSE calculations, the lower and upper biomass reference points were calculated within the operating models, using the average SSB values for these two sets of years for the two species. The LRP and USR used in the MSE were at 40% and 80% of the average SSB from 1984-1992 for *S. fasciatus* and 1984-1990 for *S. mentella*. The biomass reference points used in the MSE were thus model-based ones computed using those two historical time frames, and not the absolute values for the survey indices in those years. Performance metrics that used biomass reference points were thus computed based on stock status relative to stock status in those years, rather than the survey biomass obtained in those years.

Performance metrics that used the harvest rate at maximum sustainable yield, u_{msy}, in future years require the re-computation of u_{msy} based on draws of parameter values from their posterior distribution and when an operating model had non-stationarity in parameters that determined u_{msy}. So when the parameter values, e.g., M, changed in a future year, the u_{msy} would also need to be recomputed in that year.

MANAGEMENT PROCEDURES

Candidate management procedures included at least one of five possible components. The first component is a harvest control rule that specifies a catch limit from recently obtained trawl survey biomass estimates. The second component caps off the catch limit at some pre-specified value in particular years. Should the catch limit specified by the HCR exceed the cap specified for a particular year, then the catch limit would become the cap specified for that year. Otherwise, the catch limit from the HCR would be applied for that year. The values for caps in particular future years were determined from consultations with industry stakeholders. The third component is a status quo period of management before the year in which the harvest control rule is to be implemented. The catches taken in these years are either the average of the reported retained catches taken 2015-2017 (i.e., 2838 tons) or some pre-specified total catch amounts of interest to managers and industry members. The fourth component applies together with the HCR either a maximum allowed absolute or maximum allowed percentage change in catch limit between years. The fifth component is the implementation starting in 2018 of the small fish protocol whereby the catch limit is set to zero, if greater than 15% of the catch retained is composed of fish less than 22 cm long. Most of the candidate MPs evaluated included specifications for the first three above components. Relatively few of the MPs evaluated included the fourth and fifth components. The details of the HCR and cap specifications are outlined further below.

CLOSED-LOOP SIMULATION TESTING

For each candidate operating model, 1000 draws were taken from the approximation of the joint posterior density function of operating model parameters for that operating model. The parameters were vetted by projecting the model from past to present and used only if the stock survived to the present year. The operating model was projected 40 years into the future, i.e., from 2018 to 2057. To improve the elucidation of differences in performance metrics between candidate MPs, the same 1000 sets of recruitment deviates and draws of operating model parameters were applied within each operating model to compute performance metrics for the candidate management procedures. It took less than about eight hours of computing time to compute performance metrics for 20 candidate management procedures using 1000 simulations each.

HARVEST CONTROL RULE

The harvest control (HCR) used in each candidate MP uses a recommended catch limit that is calculated from a formula:

Equation 1. $\text{Catch Limit}_y = a + b\,(J_y - J_0) - \text{penalty}$

J_y is the ratio of 3 year trailing average survey biomass index (derived from fish > 29 cm for S. *fasciatus* and > 30 cm for S. *mentella*) to a reference index (1984-2016). The rationale for computing survey biomass indices from the larger-sized specimens is that the abundance indices and hence catch limits derived would be less sensitive to the appearance of very large cohorts. It is expected that the biomass indices and catch limits would only start to increase once the stock biomass of larger sized redfish that had much higher market values started to increase in biomass. An index that was computed from smaller sized redfish, e.g., > 22 cm, would in contrast be expected to surge up rapidly with the appearance of a large cohort and would result in large catch limits dominated by small sized redfish that would be much less valuable per ton than catch limits based on large sized redfish. Basing the survey index on large fish sizes would thus help to prevent growth overfishing and because natural mortality rates are fairly low for redfish, this would help to promote a longer lasting higher value fishery than one which harvested a large biomass of small sized low value redfish before they reached their highest economic value (see Licandeo et al. 2020 for details). The choice of the minimize sizes for computation of the bottom trawl survey indices was to some extent arbitrary. We had tried some smaller minimum sizes and found the performance was much worse for the set of MPs considered. 29 and 30 cm were considered for minimum sizes because redfish of these sizes and larger produce fillets with the highest market values (Jean Lanteigne, pers. comm.) and *S. mentella* has a lower natural mortality rate and a tendency for much larger recruitment events than *S. fasciatus*. By specifying a larger minimum size for *S. mentella* this could tend to reduce and delay the incidence of high harvest rates on *S. fasciatus* when much larger cohorts of *S. mentella* show up and also result in relatively little loss to natural mortality. Performance of MPs improved markedly with the larger minimum sizes of 29 and 30 cm for *S. fasciatus* and *S. mentella*. If yet larger minimum sizes had been set, it is expected that biomass would unnecessarily be lost to natural mortality. Clearly, a more systematic investigation of alternative minimum sizes for biomass index computation would be of interest.

The mean of recent survey values is defined as the trailing 3-year mean. For the catch limit for 2018, J_{2018} would take the mean of Unit 1 survey index values for 2015, 2016, and 2017. Both the recent and reference means are calculated using the geometric mean to dampen the effects of extreme survey values (as opposed using the arithmetic mean). Other parameters in Eq. 1 determine the relationship between the catch limit and J_y. The parameters a and b are parameters that set the scale of the catch limit. The parameter J_0 determines (1) catch limit reductions (as J_y- J_0 becomes small and eventually negative) and (2) the point on the J_y axis at which the catch limit is set to zero as survey index values and therefore J_y decline.

The penalty value is set to 0 when $J_y > J_0$, or to $c(J_y - J_0)^2$ when $J_y \leq J_0$.

The values of the base case HCR, i.e., HCR-1 were obtained by computing J_y using a range of values for 3-year trailing means and the mean for 1984-2016 and adjusting the parameter values a, b, and, c so that the catch limits specified by the resulting HCR would drop to zero at values for J_y slightly less than the historical minimum value for J_y, give catch limits less than retained catch biomass values in years with the highest J_y, and when applied in the base case operating model would give median values for harvest rates no more than about 75% of the u_{msy} for both species in the 40-year simulation horizon. The values applied for HCR-1 were 5 for a, 1.5 for b, 4 for c, and 1.5 for J_0.

Only species-specific trawl survey biomass indices from Unit 1 were applied in the harvest control rule. Indices from Unit 2 were not included. This was because the time series for Unit 1 indices extends back to 1984 and this trawl survey has been successfully executed every year since then and thus spans a much wider range of stock biomass sizes than the Unit 2 index. The Unit 2 survey in contrast began in 2000 and has been obtained once every two years, since then with a few missing years in recent years. The Unit 1 index also has considerably less interannual random variation in it compared to the Unit 2 index and thus lends itself better for application in a harvest control rule. The indices used to compute J_y are based on the base case selectivity functions estimated for the fraction of retained fish at age for the two species. Since records of catch retained reflect the component of the population of most interest to industry, it was appropriate to use an abundance index that was computed using the selectivity function for retained fish of each species, rather than the survey vulnerability or mature fish vulnerability functions.

CAPPED PROCEDURES

Caps to catch limits were implemented together with the HCRs in some of the management procedures. In capped procedures, there were two types of caps applied. The first was a ramp cap in which a specific cap was applied with the initiation of the HCR and the cap was increased a specific amount each year until the end of the ramp up period was reached. The second type of cap was a maximum cap which capped off the catch limits in all years following the end of the ramp up period. In each year where the HCR was applied, the catch limit from the HCR was implemented only if it did not exceed the cap specified for the year. If the catch limit exceeded the cap specified for that year the catch limit would be set to the cap for that year.

Table 3. Capped Management Procedures. All ramps increase the annual caps each year by 4,000 tonnes increments until the maximum cap is reached. An asterisk (*) marks Management Procedures where preliminary results were presented to the Working Group on December 13, 2017. Items in green have been identified as first priority, and items in yellow as second priority. Note that the ~ 3 kt per year was set at 2.838 kt, the average retained catch 2015-2017.

No.	Ramp Start	Ramp End	Ramp Start Cap	Max Cap	Notes on HCR
1*	2020	2027	14.5 kt	40 kt	Before ramp start year, assume ~3 kt per year.
4	2018	2025	14.5 kt	40 kt	Like 1, but starts 2 years earlier.

No.	Ramp Start	Ramp End	Ramp Start Cap	Max Cap	Notes on HCR
6*	2018	2029	17 kt	60 kt	-
8*	2020	2047	14.5 kt	120 kt	Before ramp start year, assume ~3 kt per year.
14	2020	2027	14.5 kt	40 kt	Like 1, but with an annual change of 15% maximum after 2028 (i.e., after the ramping period is complete).
16*	2018	2035	14.5 kt	80 kt	-
17*	2020	2037	14.5 kt	80 kt	-
18*	2018	2029	17 kt	60 kt	Like 6, but paired with Small Fish Protocol (SFP)at 22 cm (such that catch = 0 if SFP is violated) and a maximum increase or decrease in catch limit of 5,000 tonnes.
19*	2018	2020	20 kt	60 kt	-
22	2020	2031	17 kt	60 kt	Like 21, but paired with Small Fish Protocol at 22 cm (such that catch = 0 if SFP is violated) and a maximum increase or decrease in catch limit of 5,000 tonnes. Like 18, but delayed two years.
24	2022	2029	14.5 kt	40 kt	Like 1, but delayed two years.
34	2022	2029	14.5 kt	40 kt	Like 1, but with an annual change of 15% maximum after 2028 (i.e., after the ramping period is complete). Like 14, but delayed two years.
40	2020	2027	14.5 kt	40 kt	Before ramp start year, assume ~3 kt per year. Also with a minimum cap of 2.5 kt.

UNCAPPED PROCEDURES

Uncapped management procedures were evaluated where the main component was a harvest control rule that specified a catch limit as a function of the survey biomass values obtained in the previous three years (Equation 1). The uncapped management procedures varied in terms of the year where the HCR was to be implemented and in terms of whether a fixed catch limit would be applied and taken prior to the implementation of the HCR.

Table 4. Uncapped Management Procedures. Items in green have been identified as first priority, and items in yellow as second priority.

No.	HCR Start	Notes on HCR
12*	2020	Before start year, assume ~3 kt per year. Catch limit from the HCR formula is multiplied by 80% (a decrease of 20% and a more conservative procedure)
20*	2020	Before start year, 2018-2019, assume ~3 kt per year. HCR-1 starting in 2020 with no maximum cap.
25	2018	Set catch limits at 5 kt in 2018, 5 kt in 2019, 10 kt in 2020, 10 kt in 2021, then from 2022 onwards use the HCR-1 formula.
26	2022	Before start year, assume ~3 kt per year. Catch limit is from the HCR formula at 100% of the value Like 13, but delayed two years.
27	2022	Before start year, assume ~10.5 kt per year (i.e., current Units 1+2 TAC). Catch limit is from the HCR formula at 100% of the value
29	2022	Before start year, assume ~3 kt per year. Catch limit from the HCR formula is multiplied by 80% (a decrease of 20% and a more conservative procedure). Like 12, but delayed two years.
43	2018	Catch limits of 7.5, 10, 15 and 20 kt for 2018-2021, and the catch limits being set at 80% of the HCR 2022-2057.
44	2018	Catch limits of 5 kt for 2018-2019, and the catch limits being set at 80% of the HCR 2020-2057.
45	2018	Catch limits of 5 kt for 2018-2021, and the catch limits being set at the HCR 2022-2057.

RESULTS

OPERATING MODEL FIT AND DIAGNOSTICS

For both species, the base case operating models provided fits to the survey biomass and catch-length composition data as good or better as previous statistical catch-length models that had been applied to Unit 1 and 2 redfish stocks (Figures 3-16). Harvest rates varied considerably over the time series for both species and were as high as about 45-50% for both species in the 1990s. Spawning stock biomass showed a marked decline to the mid-1970s, a surge and then a marked decline in the 1990s. SSB has remained low for both species until the recent increase in the most recent year from the large cohorts from the year 2011 for both species (Figures 3 and 10). The models' fits to the Unit 1 trawl survey length composition data are better for early and late years in the time series (Figures 4 and 11). This is due to the large cohorts that appeared in the 1980s and then in 2011 and when larger cohorts appear, the model is better at predicting them, though for the 1980s it appeared to predict larger sizes than observed when the cohort got much older. The model was not able to fit the Unit 2 trawl survey length composition and fishery length composition for *S. mentella* very well, often over predicting the frequencies of older sized fish (Figure 6). In contrast, the model fit the Unit 2 trawl survey length and fishery length composition data for *S. fasciatus* noticeably better (Figures 12 and 13). The models' predictions of fractions of fish retained were quite variable over the time series for both species and after about 1995 predicted much higher fractions retained for both *S. mentella* and *S. fasciatus* (Figures 8 and 15). The fits to the survey biomass indices for both units showed no abnormalities for either Units 1 or 2 for both species (Figures 3, 10, 9, 16). Estimates of recruitment deviates showed relatively few episodes of large positive recruitment deviates punctuated by many years of relative small positive and negative recruitment deviates for both species and where large recruitment deviates occurred, they often occurred for both species at the same time (Figures 9 and 16).

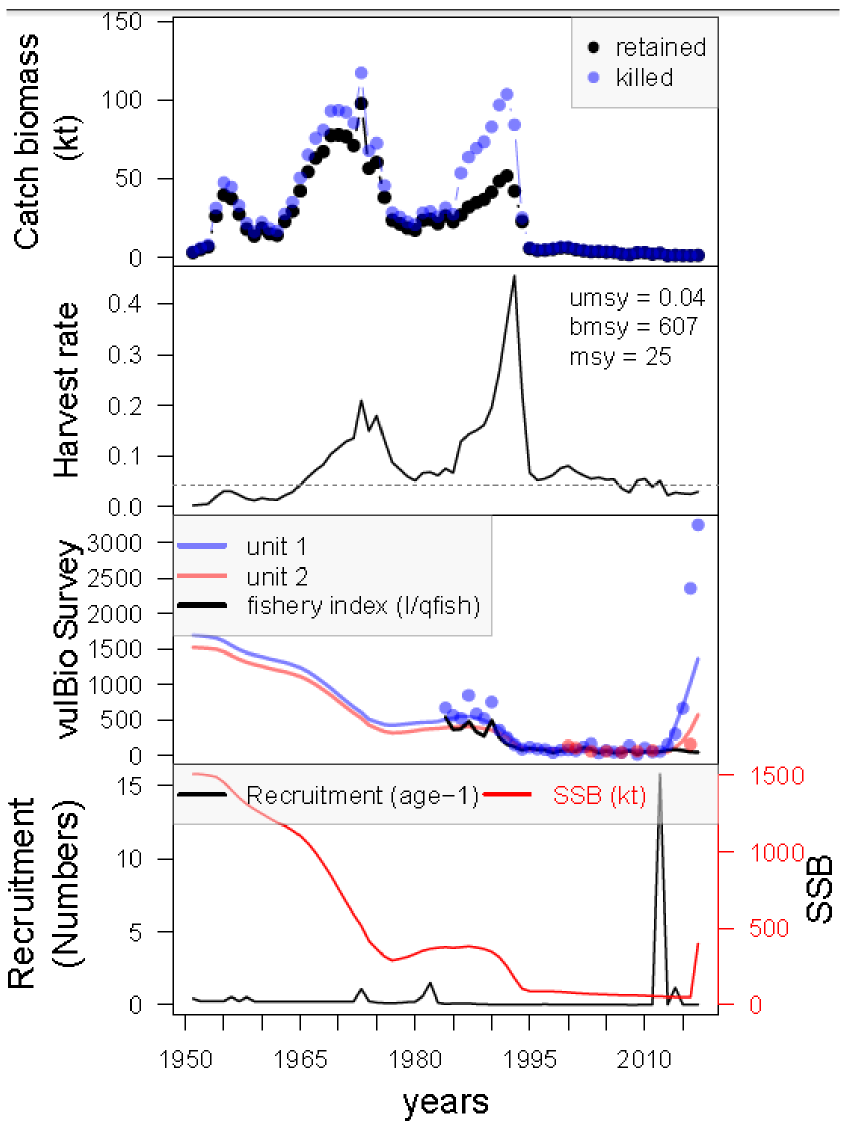

Figure 3. Plot of fits of the (top) catch biomass killed and retained, (middle top) harvest rate, (middle bottom) base case operating model fits to the Unit 1 and Unit 2 survey indices, and the Unit 1 index based on 30 cm+ fish and (bottom) age 1 recruitment and spawning stock biomass for S. mentella.

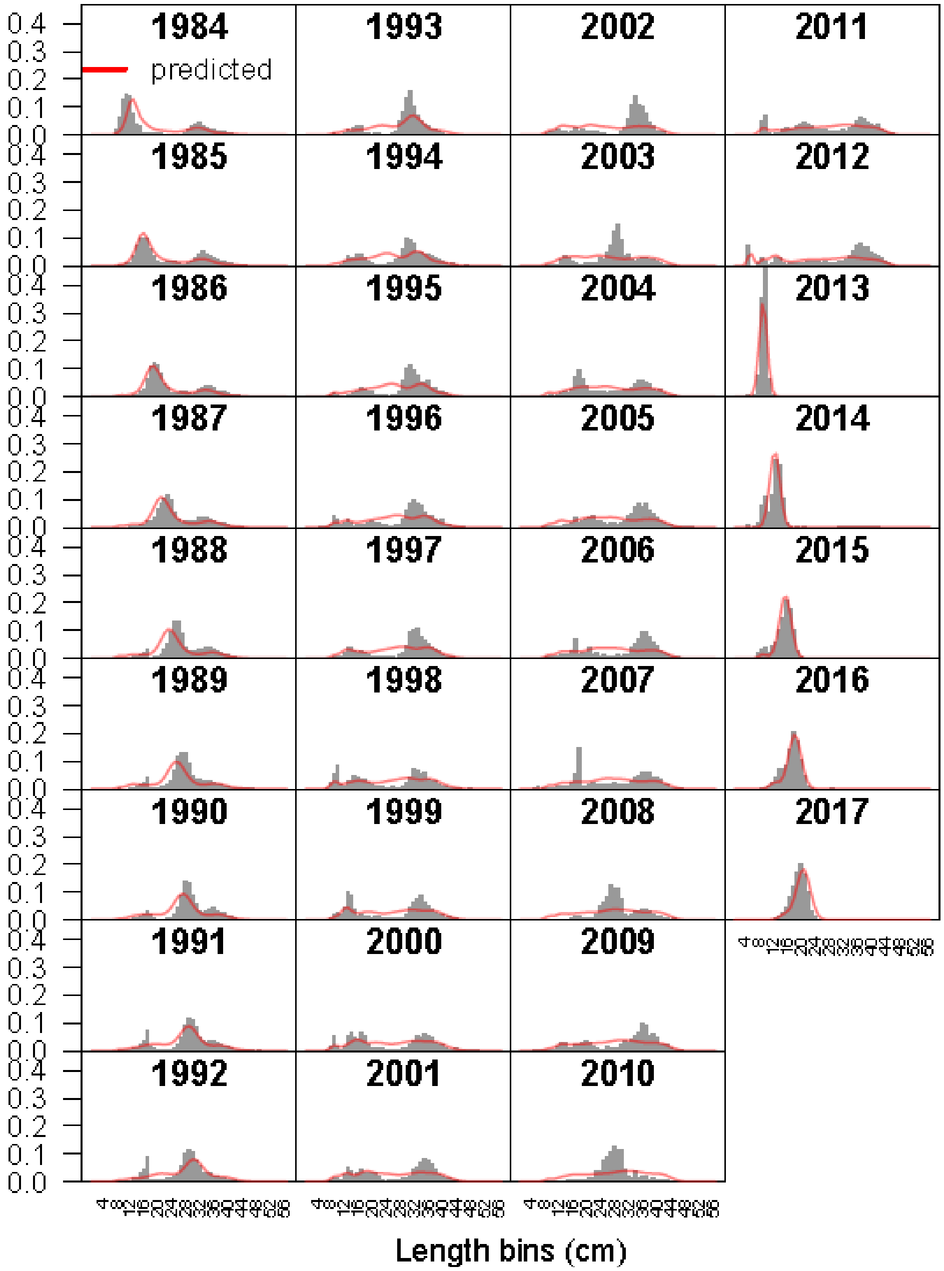

Figure 4. Fit of the base case operating model for S. mentella to Unit 1 survey composition data.

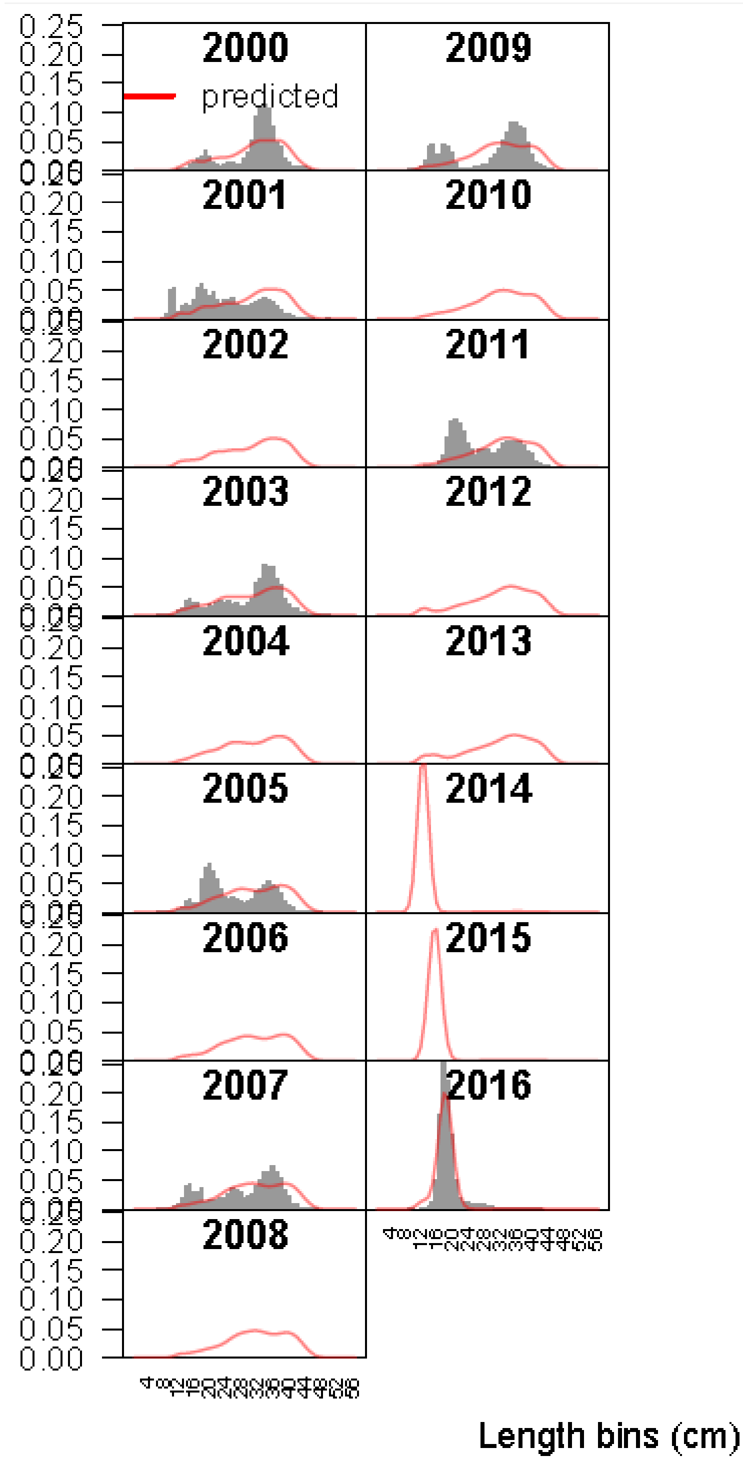

Figure 5. Fit of the base case operating model for S. mentella to Unit 2 survey composition data.

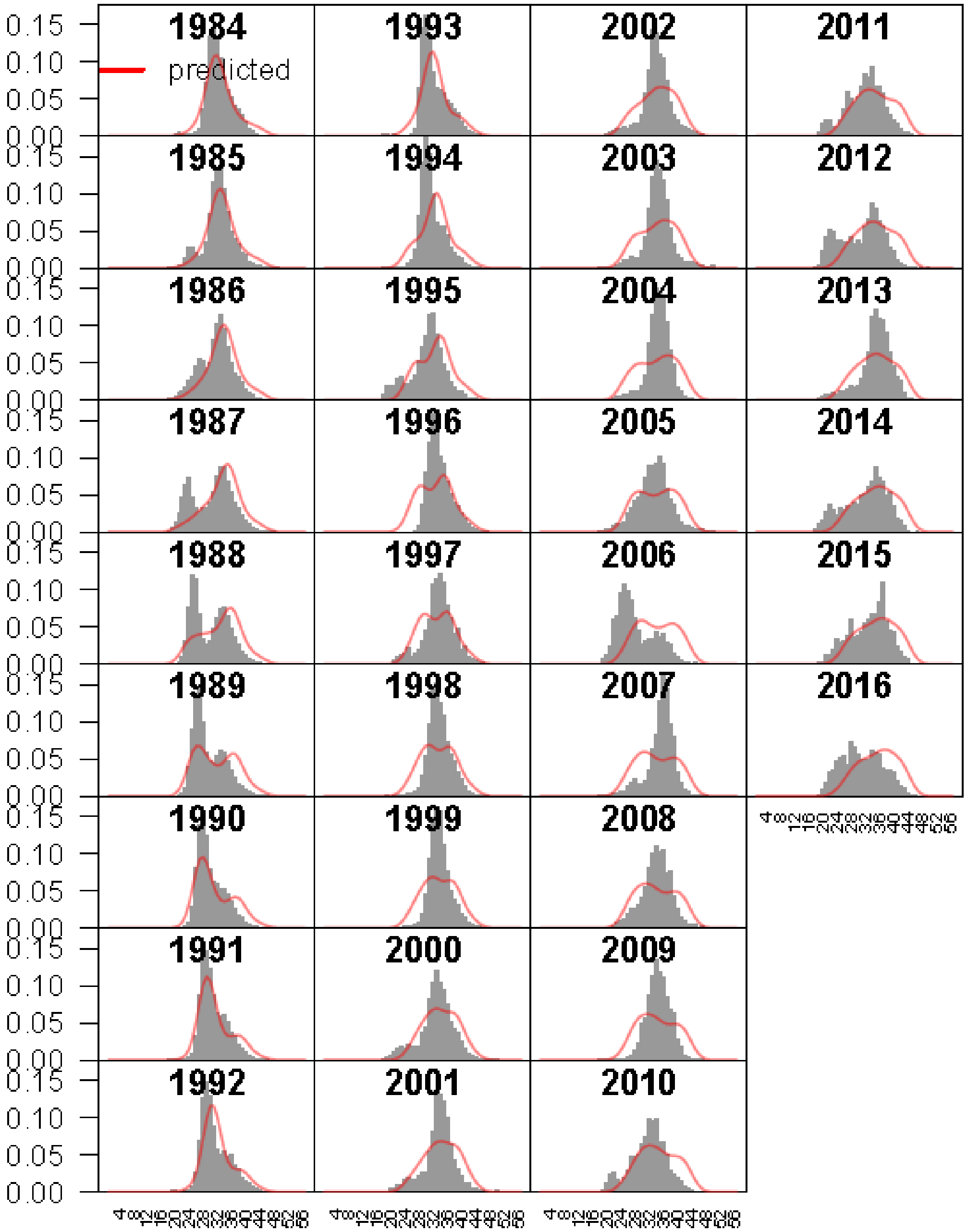

Figure 6. Fit of the base case S. mentella operating model to fishery length composition data.

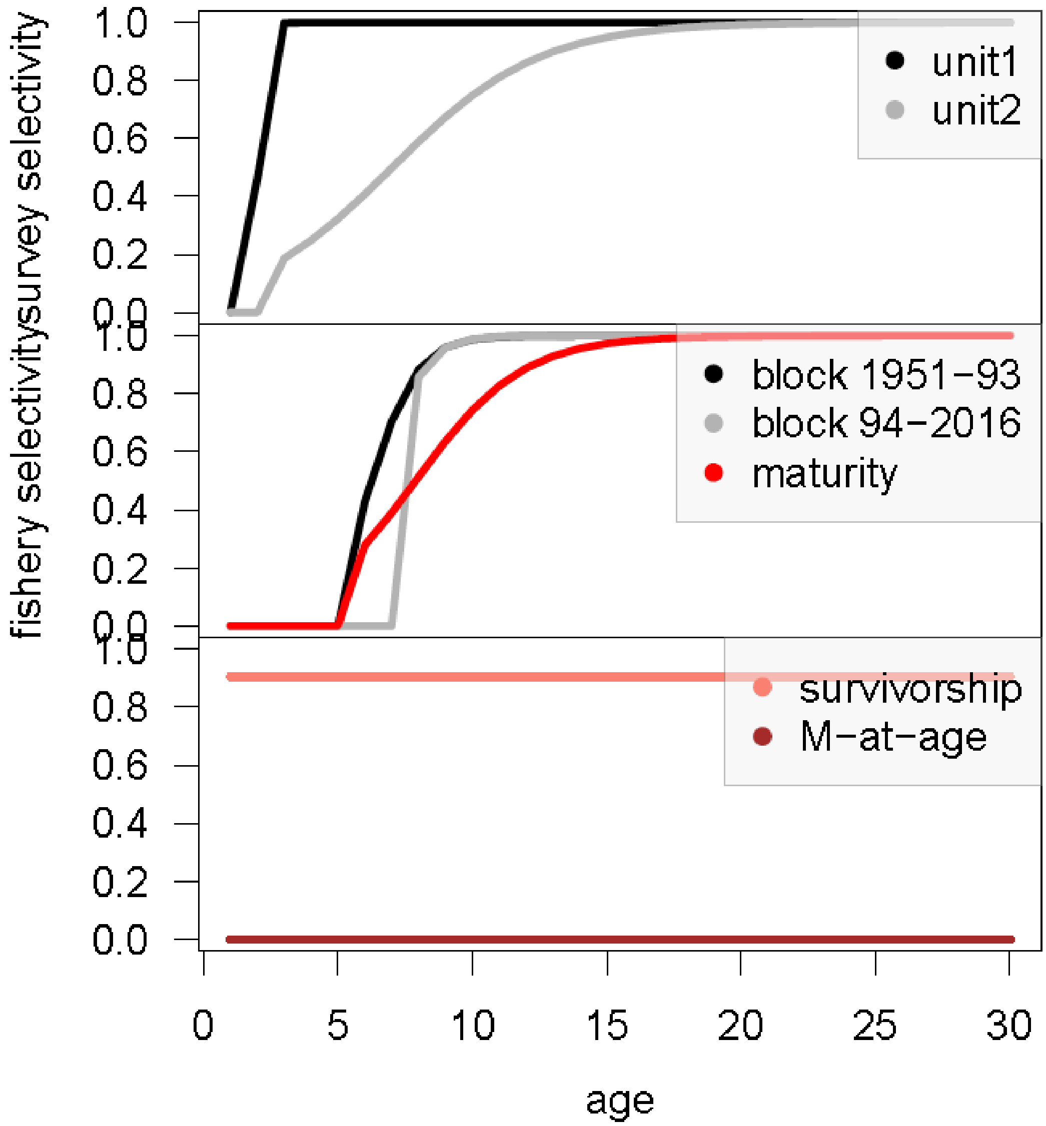

Figure 7. Plots of (top) vulnerability functions for the Units 1 and 2 trawl surveys, (middle) fishery vulnerability and fraction retailed functions, and fraction mature at age, and (bottom) natural survivorship and mortality rate at age for S. mentella in the base case model.

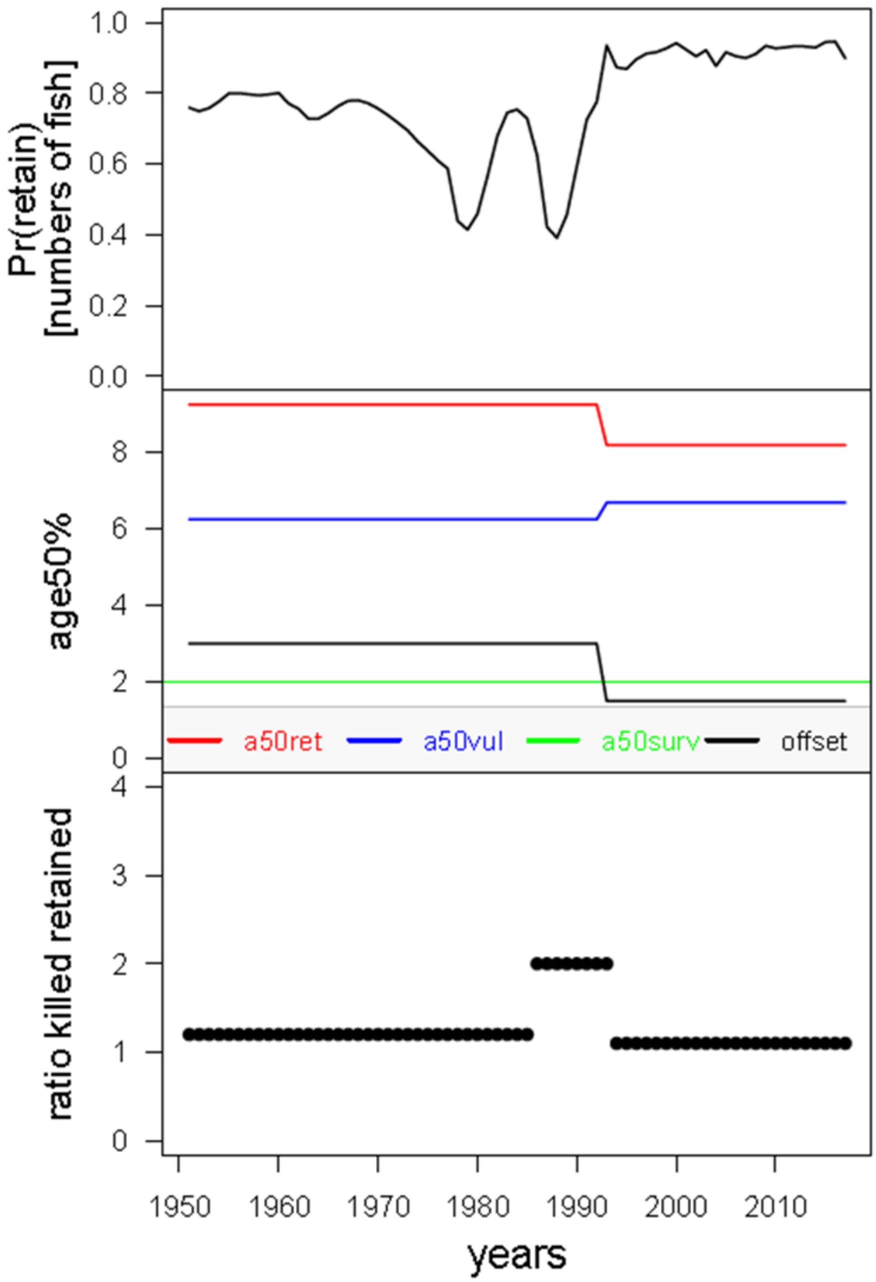

Figure 8. Plots of (top) estimates of the proportion of the catch that is retained, (middle) the estimated a50 parameter for the catch retained and catch killed, and Unit 1 survey, and the offset parameter, (bottom) the ratio of catch biomass killed to retained for S. mentella in the base case model.

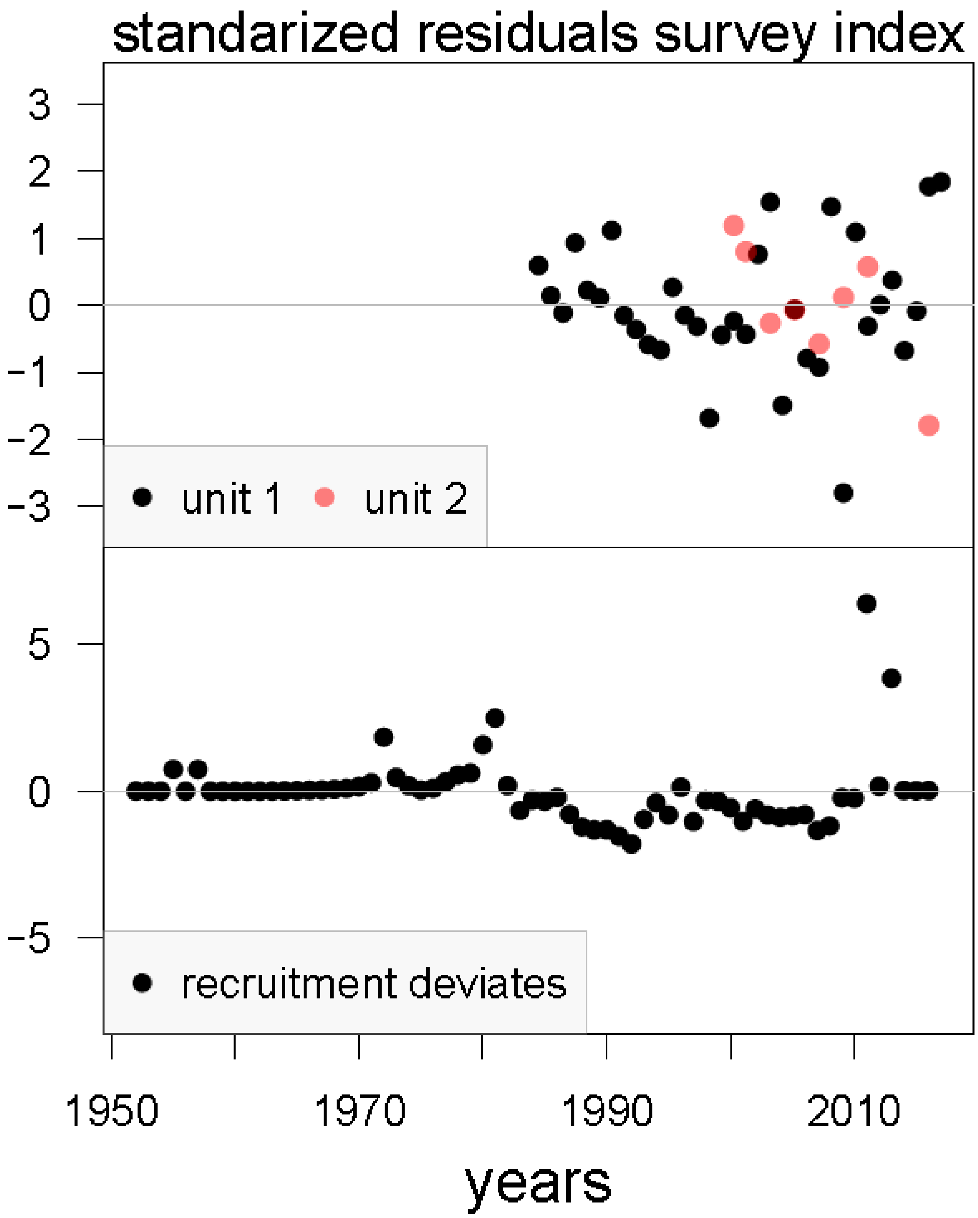

Figure 9. Plots of (top) residuals from the fit of the base case operating model to the Unit 1 and 2 survey biomass data and (bottom) stock-recruit deviates for S. mentella.

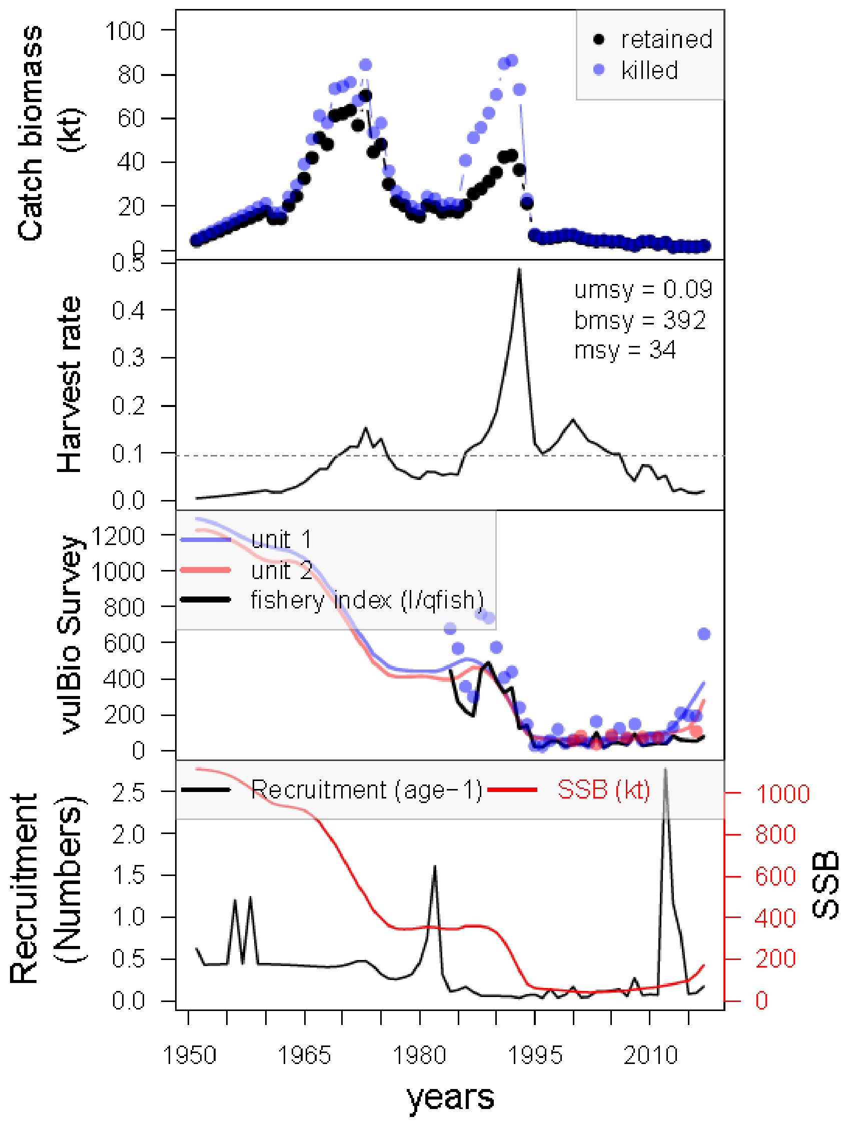

Figure 10. Plot of fits of the (top) catch biomass killed and retained, (middle top) harvest rate, (middle bottom) base case operating model fits to the Unit 1 and unit 2 survey indices, and the Unit 1 index based on 30 cm+ fish and (bottom) age 1 recruitment and spawning stock biomass for S. fasciatus.

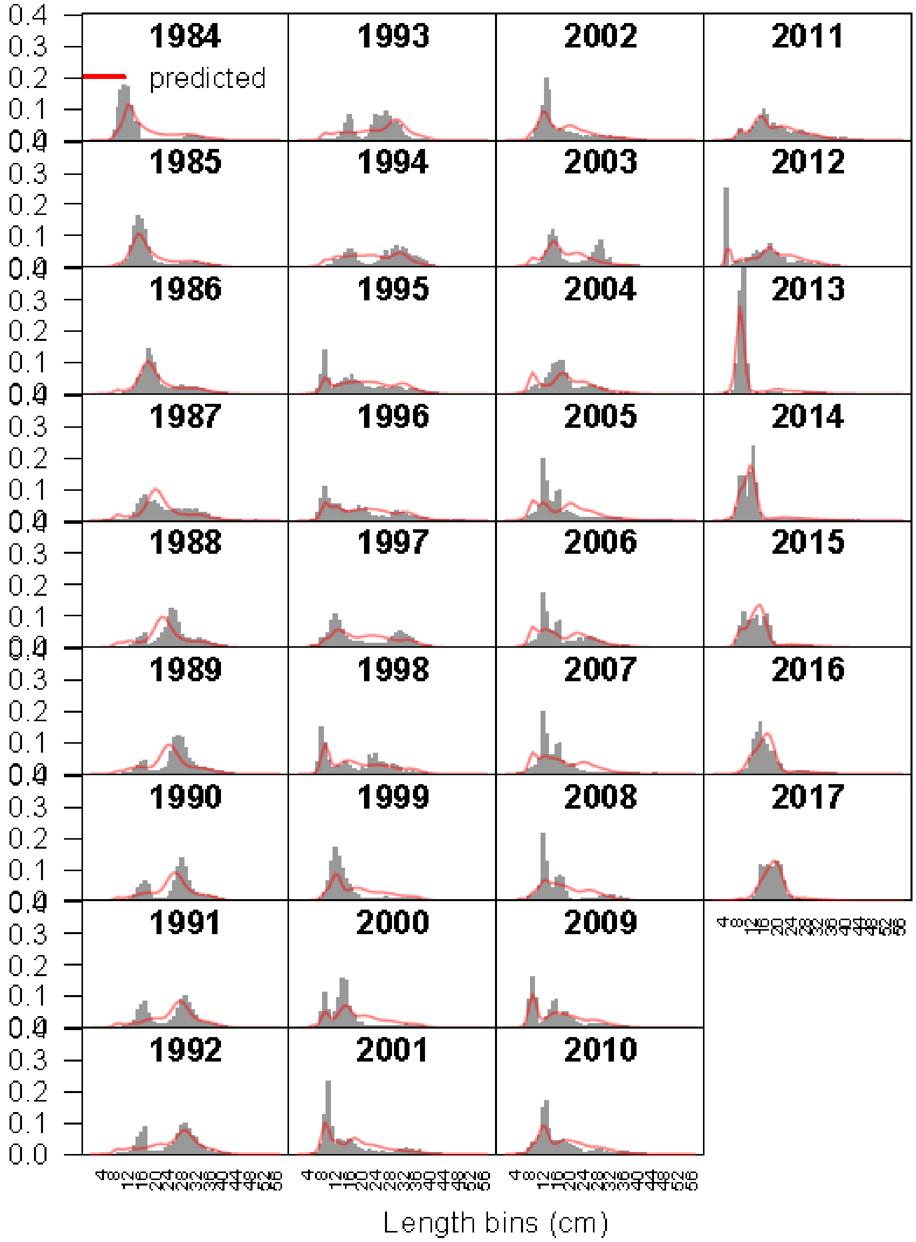

Figure 11. Fit of the base case operating model for S. fasciatus to Unit 1 survey composition data.

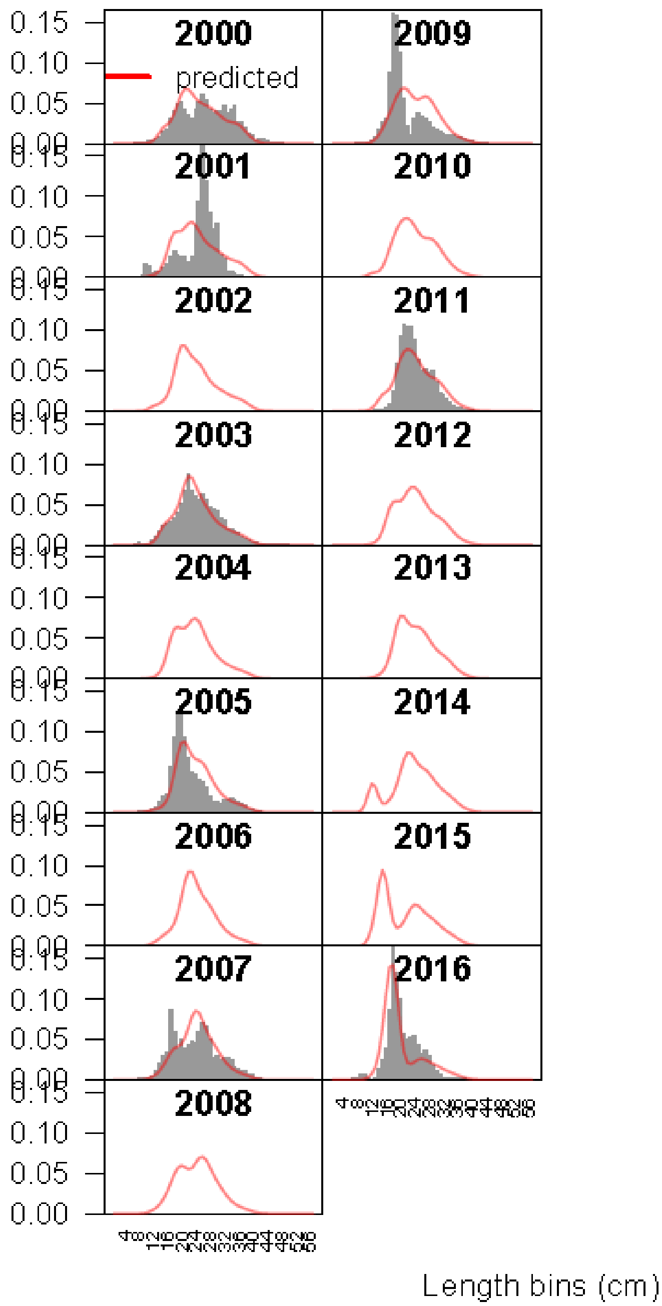

Figure 12. Fit of the base case operating model for S. fasciatus to Unit 2 survey composition data.

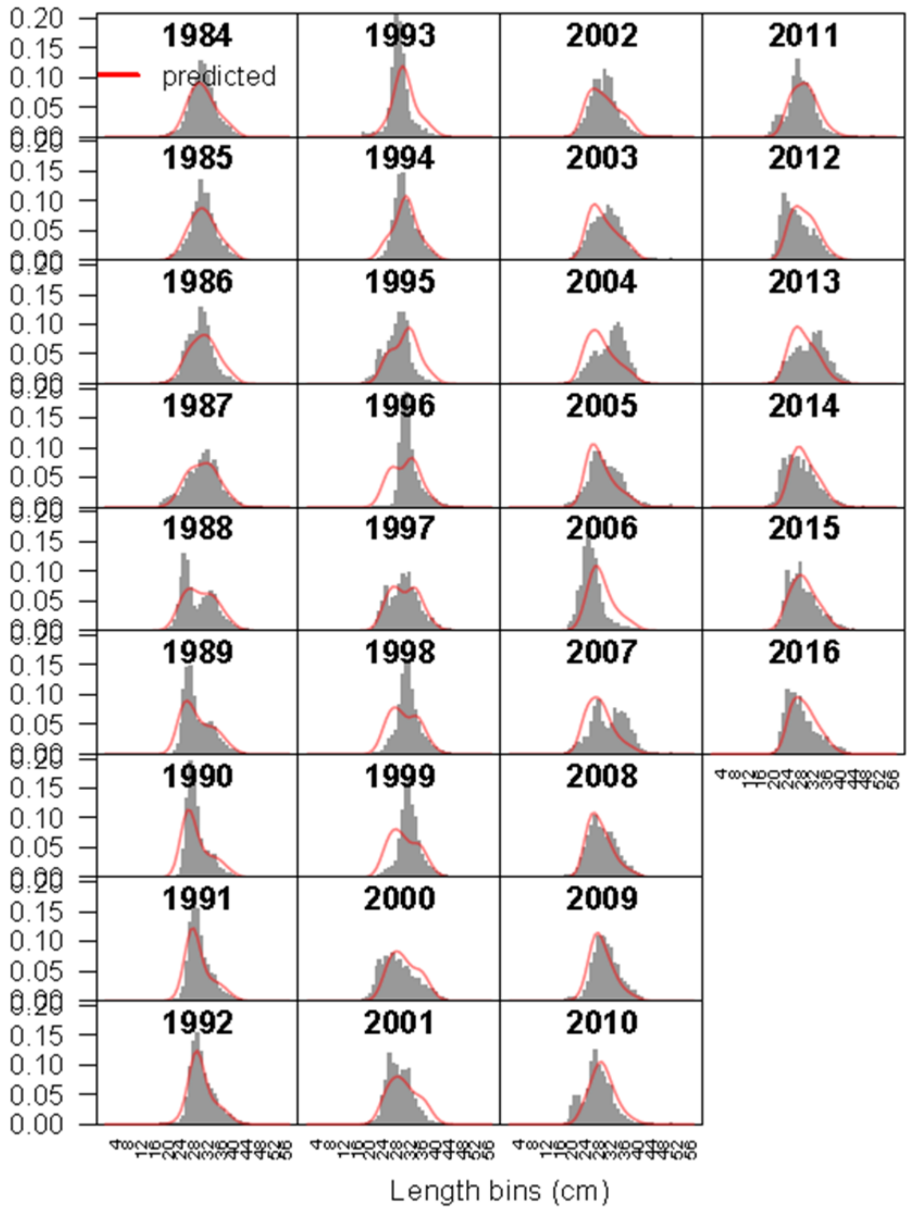

Figure 13. Fit of the base case S. fasciatus operating model to fishery length composition data.

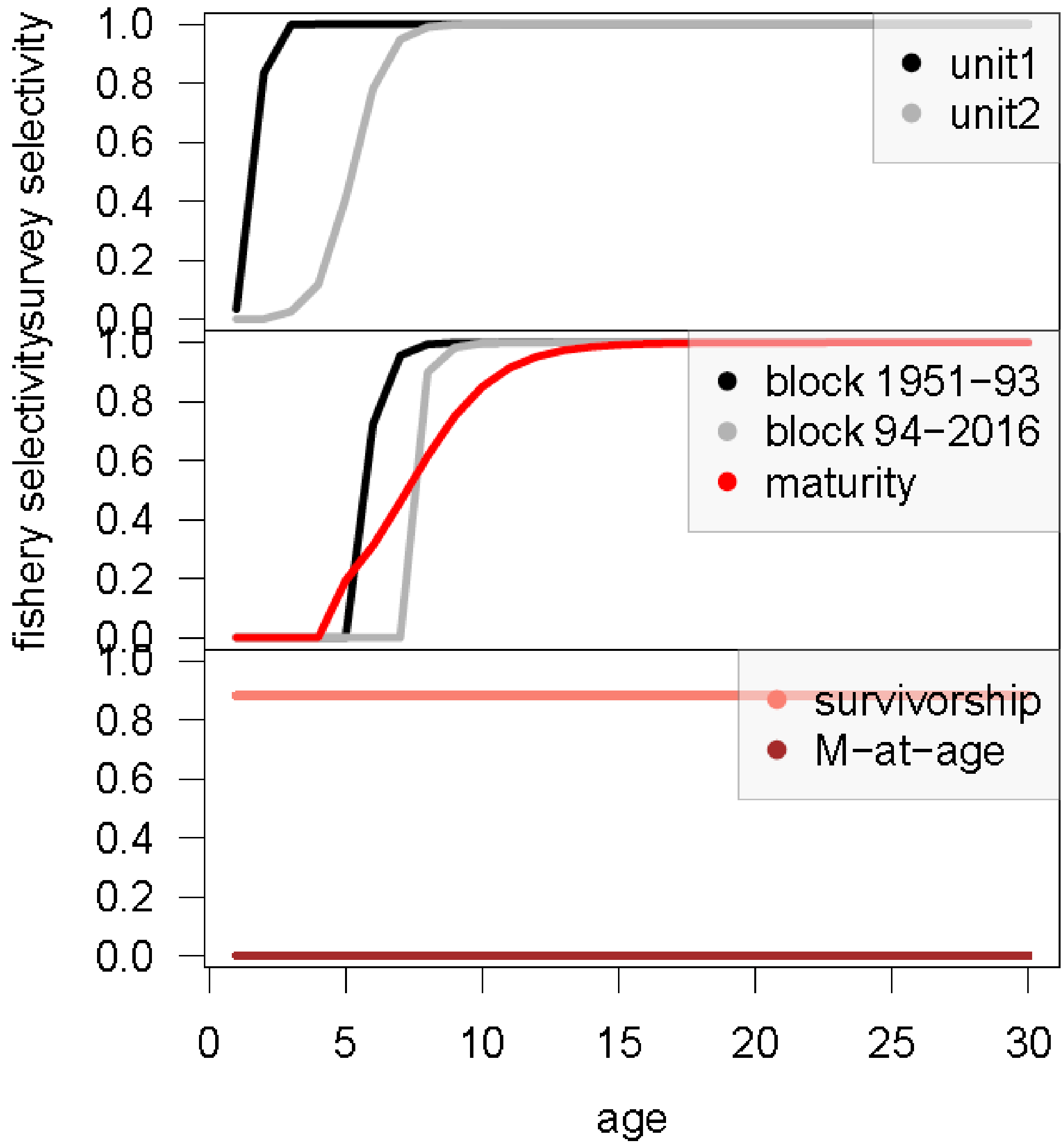

Figure 14. Plots of (top) vulnerability functions for the Units 1 and 2 trawl surveys, (middle) fishery vulnerability and fraction retailed functions, and fraction mature at age, and (bottom) natural survivorship and mortality rate at age for S. fasciatus in the base case model.

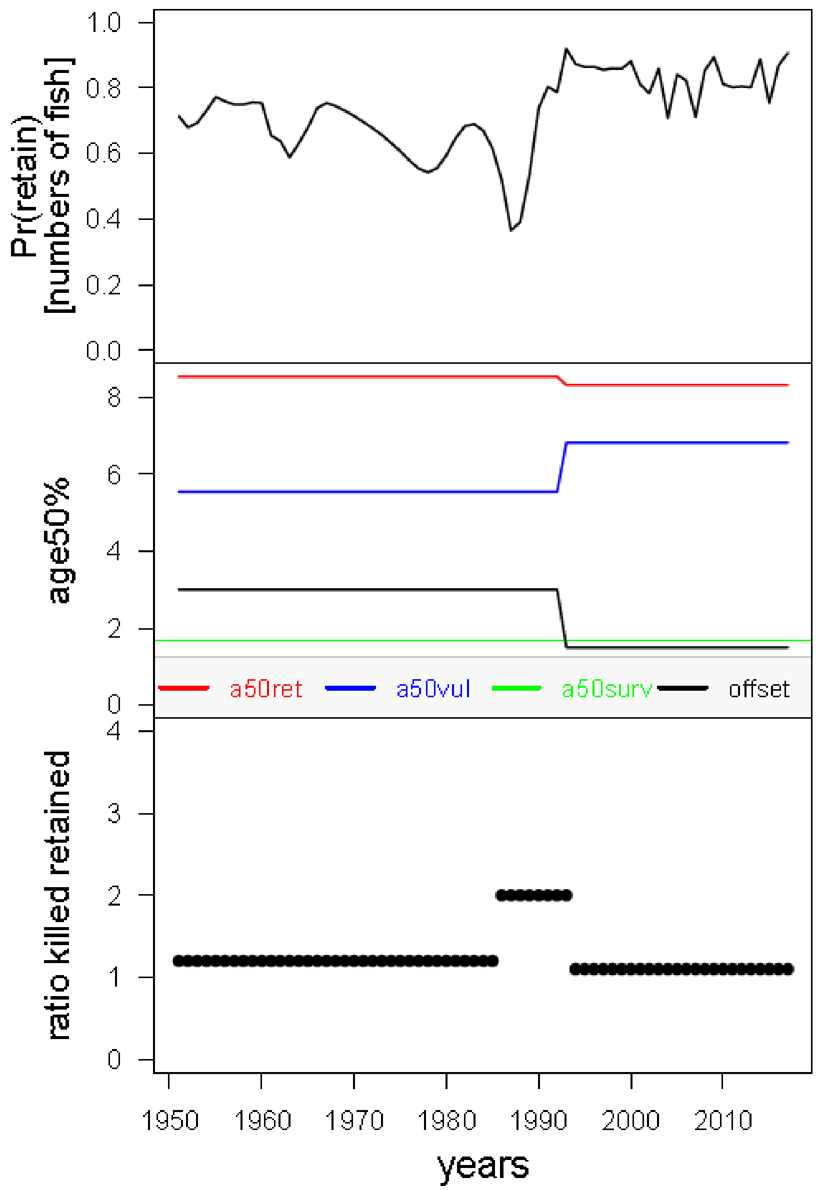

Figure 15. Plots of (top) estimates of the proportion of the catch that is retained, (middle) the estimated a50 parameter for the catch retained and catch killed, and Unit 1 survey, and the offset parameter, (bottom) the ratio of catch biomass killed to retained for S. fasciatus in the base case model.

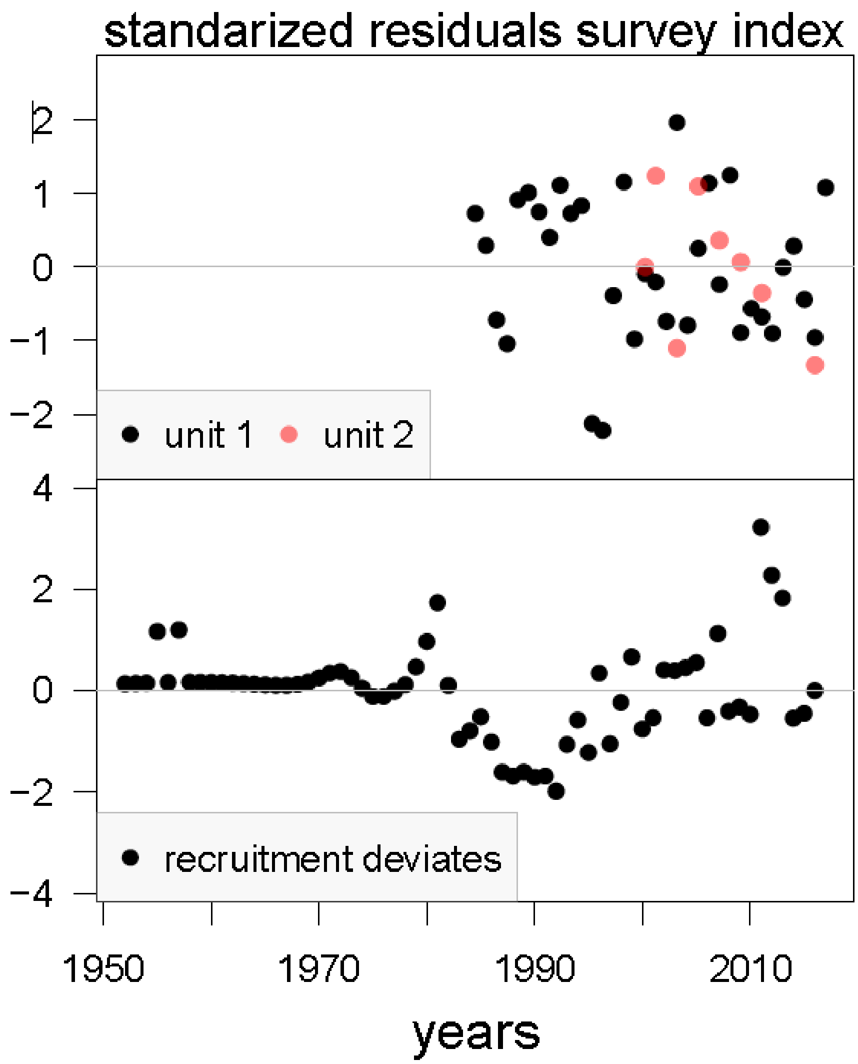

Figure 16. Plots of (top) residuals from the fit of the base case operating model to the Unit 1 and 2 survey biomass data and (bottom) stock-recruit deviates for S. fasciatus.

Retrospective analysis indicated that the estimates of parameters and key quantities of interest did not show strong retrospective patterns when the models were fitted to fewer years of data. The plots for each of the time series of estimates of spawning stock biomass, harvest rate, and age 1 recruits for the two species can be viewed in Figure 17 and Figure 18. A slight retrospective pattern can be seen in the plot of spawning stock biomass for *S. fasciatus*. Fits in 2007-2010 tend to over predict the harvest rates for 2010-2014. However, subsequent fits, i.e., in 2012-2017 show more consistent time series estimates of harvest rates for 2010-2014. Fits for *S. mentella* show no apparent retrospective pattern.

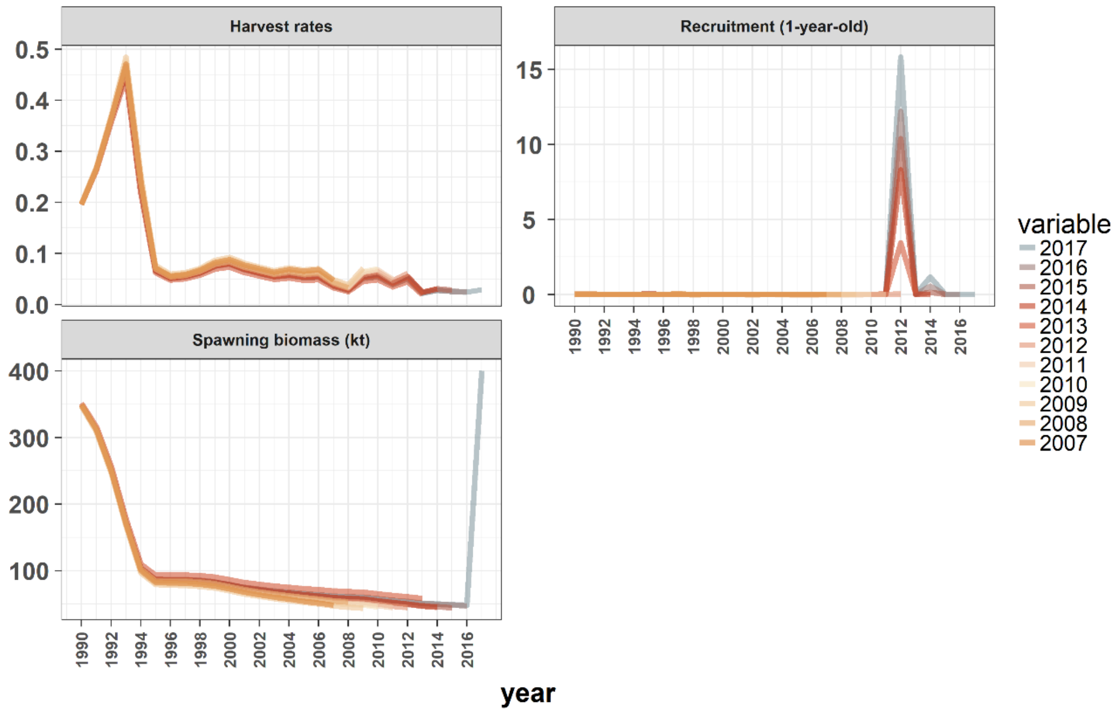

Figure 17. Plots of estimates of spawning stock biomass, harvest rates and age 1 recruitment, from retrospective analyses of the base case operating model for S. mentella.

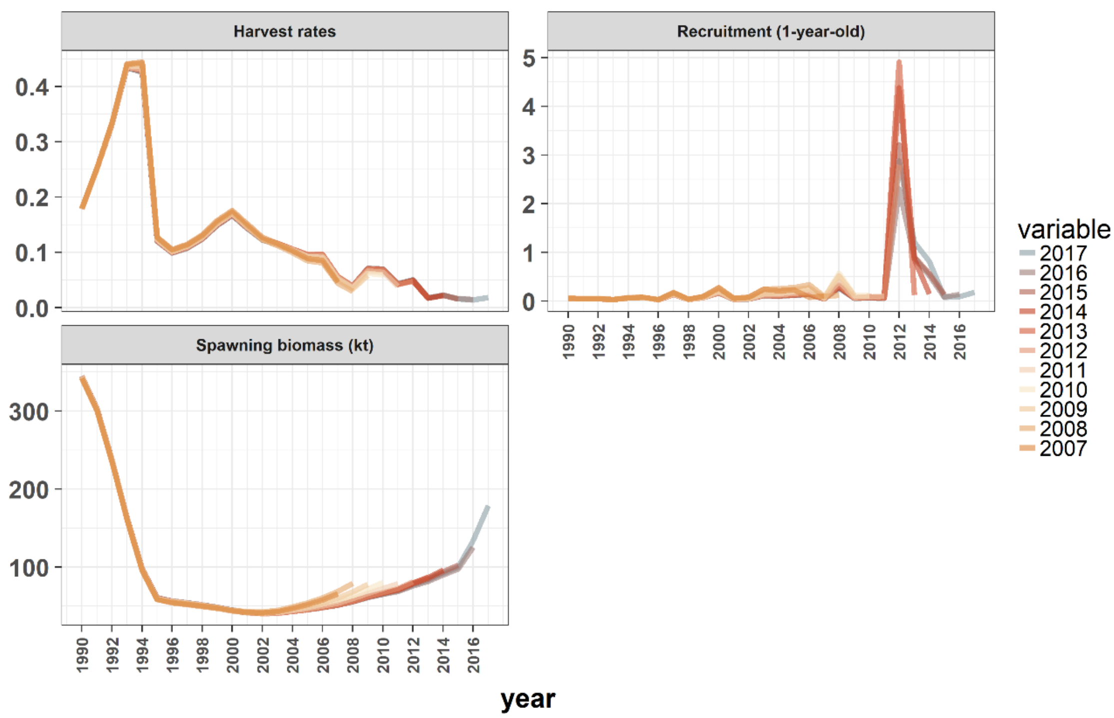

Figure 18. Plots of estimates of spawning stock biomass, harvest rates and age 1 recruitment, from retrospective analyses of the base case operating model for S. fasciatus.

PARAMETER ESTIMATES

Estimates of parameters for the base case operating models are listed in Tables 5 and 6 for *S. mentella* and *S. fasciatus*. Estimates of these quantities for other operating models that required fits to data are provided in Tables 7 and 8. AIC values and some of the associated parameter estimates for the alternative operating models that were fitted to data are shown also in Tables 7 and 8.

Table 5. Estimates of parameters (est) and key variables of interest for S. mentella from the base case operating model. Biomass values are in units of metric tonnes. Standard deviations (std) are shown for parameters that were estimated. See Table D.2 for definitions of the terms for the parameters and quantities in the left column.

parameter	est	std	parameter	est	std
ro	0.420	0.067	$pr(a50^{ret})_{mean}$	0.500	-
$ssbo$	1508.900	240.090	$pr(a50^{ret})_{sd}$	0.250	-
k	3.082	0.790	$pr(a50^{ret})$	1.000	-
h	0.435	0.063	$pr(recdevs)_{mean}$	2.100	-
q_{u1}	0.666	0.046	$pr(recdevs)_{sd}$	1.000	-
q_{u2}	1.745	0.226	cvl1	0.120	-
q_{u1}^{mature}	0.765	0.059	cvl2	0.050	-
$q_{u1}^{fishery}$	0.638	0.052	Linf	45.820	-
sd(index)$_{u1}$	0.469	-	vbk	0.096	-
sd(index)$_{u2}$	0.452	-	to	-0.500	-
sd(index^{mature})$_{u1}$	0.574	-	m	0.100	-
sd(index^{fishery})$_{u1}$	0.407	-	age50$_{mat}$	7.889	-
$age50_{u1}$	2.022	0.061	age50sd$_{mat}$	1.980	-
$age50sd_{u1}$	0.145	0.018	minage_{mat}	5.000	-
$age50[ls]_{u1}^{*}$	19.000	-	Min$age_{kill\text{-}ret}$	7.000	-
$age50sd[ls]_{u1}^{*}$	1.000	-	offset$_{2017}$	1.500	-
$age50_{u2}^{*}$	7.017	1.348	killratio$_{2017}$	1.100	-
$age50sd_{u2}^{*}$	2.731	1.038	$age50_{blk2}^{kill}$	6.684	-
$age50_{blk1}^{ret}$	9.245	0.679	ess$_{u1}$	25.000	-
$age50_{blk2}^{ret}$	8.184	0.700	ess$_{u2}$	10.000	-
$age50sd_{blk1}^{ret}$	0.866	0.216	ess$_{fishery}$	5.000	-
$age50sd_{blk2}^{ret}$	0.735	0.250	$negloglike$	3621.827	-
$age50[ls]^{ret*}$	19.000	-	$negloglike_{index_{u1}}$	56.273	-
$age50sd[ls]^{ret*}$	1.000	-	$negloglike_{index_{u2}}$	9.808	-
U_{msy}	0.041	-	$negloglike_{lengthf_{u1}}$	2777.400	-
msy	25.030	-	$negloglike_{lengthf_{u2}}$	275.758	-
ssb_{msy}	606.522	-	$negloglike_{lengthf_{ret}}$	502.588	-
u_{2018}	0.029	-	$npar$	82.000	
ssb_{2018}	400.490	-	AIC	7407.655	-
D_{ssb}	0.265	-			
sig_R	1.000	-			
$pr(h)_{mean}$	0.670	-			
$pr(h)_{sd}$	0.170	-			
$cv(index)_{u1}$	0.250	-			
$cv(index)_{u2}$	0.250	-			
$pr(q_{u1})_{mean}$	0.200	-			
$pr(q_{u2})_{mean}$	0.200	-			
$pr(q_{u1})_{sd}$	1.000	-			
$pr(q_{u2})_{sd}$	1.000	-			

Table 6. Estimates of parameters (est) and key variables of interest for S. fasciatus from the base case operating model. Standard deviations (std) are shown for parameters that were estimated. Biomass values are in units of metric tonnes. See Table D.2 for definitions of the terms for the parameters and quantities in the left column.

parameter	est	std	parameter	est	std
ro	0.623	0.123	$pr(a50^{ret})_{mean}$	0.500	-
$ssbo$	1115.500	220.800	$pr(a50^{ret})_{sd}$	0.250	-
k	6.522	2.084	$pr(a50^{ret})$	1.000	-
h	0.620	0.085	$pr(recdevs)_{mean}$	2.100	-
q_{u1}	0.533	0.034	$pr(recdevs)_{sd}$	1.000	-
q_{u2}	2.495	0.252	cvl1	0.120	-
q_{u1}^{mature}	0.525	0.037	cvl2	0.050	-
$q_{u1}^{fishery}$	0.444	0.036	Linf	41.238	-
sd(index)$_{u1}$	0.501	-	vbk	0.106	-
sd(index)$_{u2}$	0.382	-	to	-0.500	-
sd(index^{mature})$_{u1}$	0.519	-	m	0.125	-
sd(index^{fishery})$_{u1}$	0.516	-	age50$_{mat}$	7.256	-
$age50_{u1}$	1.676	0.109	age50sd$_{mat}$	1.580	-
$age50sd_{u1}$	0.201	0.034	minage_{mat}	4.000	-
$age50[ls]_{u1}^{*}$	18.000	-	Min$age_{kill\text{-}ret}$	7.000	-
$age50sd[ls]_{u1}^{*}$	1.000	-	offset$_{2017}$	1.500	-
$age50_{u2}^{*}$	5.224	0.363	killratio$_{2017}$	1.100	-
$age50sd_{u2}^{*}$	0.609	0.211	$age50_{blk2}^{kill}$	6.829	-
$age50_{blk1}^{ret}$	8.545	0.534	ess$_{u1}$	25.000	-
$age50_{blk2}^{ret}$	8.329	0.345	ess$_{u2}$	10.000	-
$age50sd_{blk1}^{ret}$	0.471	0.237	ess$_{fishery}$	5.000	-
$age50sd_{blk2}^{ret}$	0.534	0.237	$negloglike$	3483.668	-
$age50[ls]^{ret*}$	18.000	-	$negloglike_{index_{u1}}$	64.160	-
$age50sd[ls]^{ret*}$	1.000	-	$negloglike_{index_{u2}}$	7.008	-
U_{msy}	0.094	0.032	$negloglike_{lengthf_{u1}}$	2676.830	-
msy	33.746	6.565	$negloglike_{lengthf_{u2}}$	255.776	-
ssb_{msy}	391.676	95.085	$negloglike_{lengthf_{ret}}$	479.894	-
u_{2018}	0.019	0.002	$npar$	82.000	-
ssb_{2018}	171.500	20.100	AIC	7131.336	-
D_{ssb}	0.153	0.035			
sig_R	1.000	-			
$pr(h)_{mean}$	0.670	-			
$pr(h)_{sd}$	0.170	-			
$cv(index)_{u1}$	0.250	-			
$cv(index)_{u2}$	0.250	-			
$pr(q_{u1})_{mean}$	0.200	-			
$pr(q_{u2})_{mean}$	0.200	-			
$pr(q_{u1})_{sd}$	1.000	-			
$pr(q_{u2})_{sd}$	1.000	-			

Table 7. AIC results and some of the parameter estimates for the core and stress-test operating models that were fitted to data for S. mentella. Estimation results for models 9, 3, 4, and 5 are identical to model 1 because the modifications were done only to the future simulations. Biomass values are in kt.

Model	Type	Description	Bo	h	msy	u_{msy}	D_{2017}	#p	nllik	AIC	Notes
1	Core	Base Case	**1509**	**0.44**	**25**	**0.041**	**0.265**	**82**	**3621.8**	**7407.7**	-
8	Core	Lorenzen M	**1509**	**0.34**	**20**	**0.031**	**0.307**	**82**	**3681.0**	**7526.0**	-
10	Core	More *S. mentella* in catch split	**1714**	**0.45**	**29**	**0.040**	**0.272**	**82**	**3629.0**	**7422.1**	-
11	Core	Less *S. mentella* in catch split	**1287**	**0.43**	**21**	**0.041**	**0.260**	**82**	**3622.3**	**7408.6**	-
2	Stress	Domed fishery selectivity	**1488**	**0.46**	**29**	**0.081**	**0.158**	**82**	**3525.3**	**7214.6**	The parameters for only the ascending limb were estimated
15	Stress	Low M (0.075)	**1478**	**0.57**	**26**	**0.048**	**0.244**	**82**	**3618.7**	**7401.5**	-
16	Stress	High M (0.125)	**1612**	**0.33**	**20**	**0.030**	**0.280**	**82**	**3628.5**	**7420.9**	-
17	Stress	High steepness	**1374**	**0.54**	**29**	**0.058**	**0.296**	**82**	**3622.1**	**7408.4**	-
18	Stress	Low steepness	**1623**	**0.33**	**18**	**0.025**	**0.223**	**82**	**3621.9**	**7407.7**	-
13a	Stress	Lower offset	**1500**	**0.43**	**25**	**0.042**	**0.265**	**82**	**3619.7**	**7403.5**	-
13b	Stress	Higher offset	**1519**	**0.44**	**25**	**0.041**	**0.266**	**82**	**3623.7**	**7411.3**	-
14a	Stress	Lower catch killed: retained	**910**	**0.45**	**16**	**0.043**	**0.312**	**82**	**3619.8**	**7403.7**	-
14b	Stress	Higher catch killed: retained	**2110**	**0.43**	**35**	**0.041**	**0.247**	**82**	**3623.7**	**7411.4**	-
19	Stress	Low prior strong cohorts	**1582**	**0.43**	**26**	**0.041**	**0.253**	**82**	**3621.8**	**7407.7**	-
20	Stress	High prior strong cohorts	**1412**	**0.44**	**24**	**0.042**	**0.284**	**82**	**3621.8**	**7407.7**	-

Table 8. AIC results and some of the parameter estimates for the core and stress-test Operating Models that were fitted to data for S. fasciatus. Estimation results for models 9, 3, 4, and 5 are identical to model 1 because the modifications were done only to the future simulations. Biomass values are in kt.

Model	Type	Description	Bo	h	msy	u_{msy}	D_{2017}	#p	nllik	AIC	Notes
1	Core	Base Case	**1116**	**0.62**	**34**	**0.094**	**0.153**	**82**	**3483.7**	**7131.3**	-
8	Core	Lorenzen M	**1140**	**0.47**	**30**	**0.079**	**0.143**	**82**	**3464.8**	**7093.5**	-
10	Core	More *S. mentella* in catch split	**958**	**0.62**	**29**	**0.094**	**0.150**	**82**	**3488.2**	**7140.4**	-
11	Core	Less *S. mentella* in catch split	**1380**	**0.60**	**40**	**0.091**	**0.152**	**82**	**3495.4**	**7254.9**	-
2	Stress	Domed fishery selectivity	**1488**	**0.46**	**29**	**0.081**	**0.158**	**82**	**3525.3**	**7214.6**	The parameters for only the ascending limb were estimated
15	Stress	Low M (0.094)	**1183**	**0.73**	**32**	**0.090**	**0.140**	**82**	**3492.4**	**7148.8**	-
16	Stress	High M (0.125)	**1078**	**0.53**	**34**	**0.096**	**0.116**	**82**	**3478.4**	**7120.7**	-
17	Stress	High steepness	**807**	**0.78**	**29**	**0.129**	**0.218**	**82**	**3483.8**	**7131.5**	-
18	Stress	Low steepness	**1323**	**0.47**	**30**	**0.062**	**0.126**	**82**	**3483.6**	**7131.2**	-
12	Stress	Alternative selectivity blocks	-	-	-	-	-	-	-	-	-
13a	Stress	Lower offset	**1107**	**0.62**	**34**	**0.096**	**0.155**	**82**	**3484.2**	**7132.3**	-
13b	Stress	Higher offset	**1120**	**0.62**	**33**	**0.093**	**0.152**	**82**	**3483.4**	**7130.8**	-
14a	Stress	Lower catch killed: retained	**701**	**0.63**	**22**	**0.096**	**0.176**	**82**	**3481.0**	**7126.1**	-
14b	Stress	Higher catch killed: retained	**1533**	**0.62**	**46**	**0.093**	**0.144**	**82**	**3486.1**	**7136.1**	-
19	Stress	Low prior strong cohorts	**1192**	**0.61**	**36**	**0.092**	**0.144**	**82**	**3483.7**	**7131.5**	-

Model	Type	Description	Bo	h	msy	u_{msy}	D_{2017}	#p	nllik	AIC	Notes
20	Stress	High prior strong cohorts	**1009**	**0.63**	**31**	**0.097**	**0.169**	**82**	**3483.5**	**7131.1**	-

PERFORMANCE OF MANAGEMENT PROCEDURES

Plots of fishery and stock performance metrics for the 21 management procedures for the base case operating model can be seen in Figures 19-22. All of the candidate management procedure passed the LRP and USR performance metrics across all of the operating models 100% of the time (no plots shown except for base case). Under one of the core operating models, i.e., OM 10, all of the uncapped and four of the nine capped management procedures passed the harvest rate to harvest rate at MSY performance metric; MPs 20, 25-27, and 45 failed (Figure 23). Poorer performance on this performance metric resulted when catch splits favoured *S. mentella* in the past, and the catch killed to retained ratio was low.

All of the performance metrics failed the PMs on maximizing the number of years where fish < 22 cm represent < 15% of catch (and Small Fish Protocol is not triggered) in the next a) 5 years, b) 40 years (Figure 24). Capped MPs scored slightly higher than uncapped MPs on this PM. Start year had very little effect on performance. Scores were lower with high past and future M, or doubling of future M, and where catch splits in the past favoured *S. mentella*. Scores were higher when there were no strong cohorts in the future, when both future and past M was lower, and catch splits favoured *S. fasciatus*.

All management procedures failed the performance metrics related to maximizing the number of years where fish < 25 cm represent < 15% of catch over the next a) 5 years, b) 40 years.

In terms of average catch performance metrics, uncapped MPs ranked higher than uncapped MPs though less so if the uncapped HCR was reduced by 80% (Figures 25-26). In the capped MPs the scores increased with higher MP caps. The start year had no effect on this PM. Scores were higher under parametric recruitment deviates, low past and future M, and lower under high future M, and high past and future M, and the higher and lower steepness scenarios. The gain in average catch in the uncapped versus capped MPs was less well pronounced in the 10-40 year horizon than in the 10-20 year horizon (Figures 25-26).

In terms of the proportion of simulations where catch limits exceeded 40 kt by 2028 and the mean number of years were the catch limit exceeded 40 kt between 2028 and 2057, capped MPs ranked higher, especially ones with 15% maximum changes in catch limits allowed and ones with lower caps (Figures 27 and 28). Uncapped MPs with the HCR at 80% had lower average catches. Again there was no effect of start year. Scores were higher under the parametric bootstrap of recruitment deviates, and low past and future M and lower under high future and past and future M, and altered steepness.

In terms of the proportion of years were the catch limit exceeded landings in 2017, all MPs performed well, except for those with a built-in small fish protocol which did poorly under some of the stress test operating models (Figure 29).

In terms of maximizing the catch of large fish, in 5 and 40 years, capped MPs scored slightly higher, but were less robust than uncapped MPs (Figure 30). Scores were lower under high past and future M and doubled future M, and splits favouring *S. mentella* but higher under low past and future M, and splits favouring *S. fasciatus*.

In terms of maintaining stability of the fishery, capped MPs scored higher and were more robust that uncapped MPs (Figures 31 and 32). There was some sensitivity in scores to parametric recruitment and high future M.

In terms of key trade-offs between performance metrics there were five important ones. Average catches in 10-20 years were correlated with average catches in 10-40 years especially for capped MPs. However, for uncapped MPs the catches over 10-40 years were considerably lower (Figure 33).

There was a pronounced trade-off between average catch and the number of years where the small fish protocol would be avoided (Figure 34). Uncapped MPs had high average catches but fewer years with small fish protocols avoided. Capped MPs had lower average catches but more years with the small fish protocols avoided.

There was a pronounced trade-off between average catch and the proportion of years with large fish in the catch (Figure 35). Uncapped MPs had high average catch and lower proportions of years with large fish and vice versa for capped MPs.

There was a pronounced trade-off between the average catch and the average number of years where the catch limit exceeded 40 kt (Figure 36). Uncapped MPs had higher average catches but fewer years with catch limits > 40kt and vice versa for capped MPs. However, capped procedures with the small fish protocol and 5 kt maximum allowed changes had both low average catches and low numbers of years with catches > 40 kt.

There was also a pronounced trade-off between the average catch retained and the proportion of years where the catch limit changed less than 15% (Figure 37). Uncapped MPs had high average catches but lower proportions of years where the catch limit changed by less than 15% and vice versa for capped MPs.

To illustrate the sensitivity of some of the trade-offs to the operating model specification, we have shown five trade-off plots under operating model 3 where the rate of natural mortality doubles in the next 20 years. The average catches over 10-20 and 10-40 years are much lower than under the base case and the trade-off is less severe between the different MPs (Figure 38). The trade-off is also less severe between the years where the small fish protocol is not triggered and the average catch (Figure 39), the proportion of years where fish > 27 cm are > 80% of the catch and average catch (Figure 40), years where the TAC > 40 kt and average catch retained (Figure 41), and proportion of years where the change in TAC < 15% and the average catch retained (Figure 42).

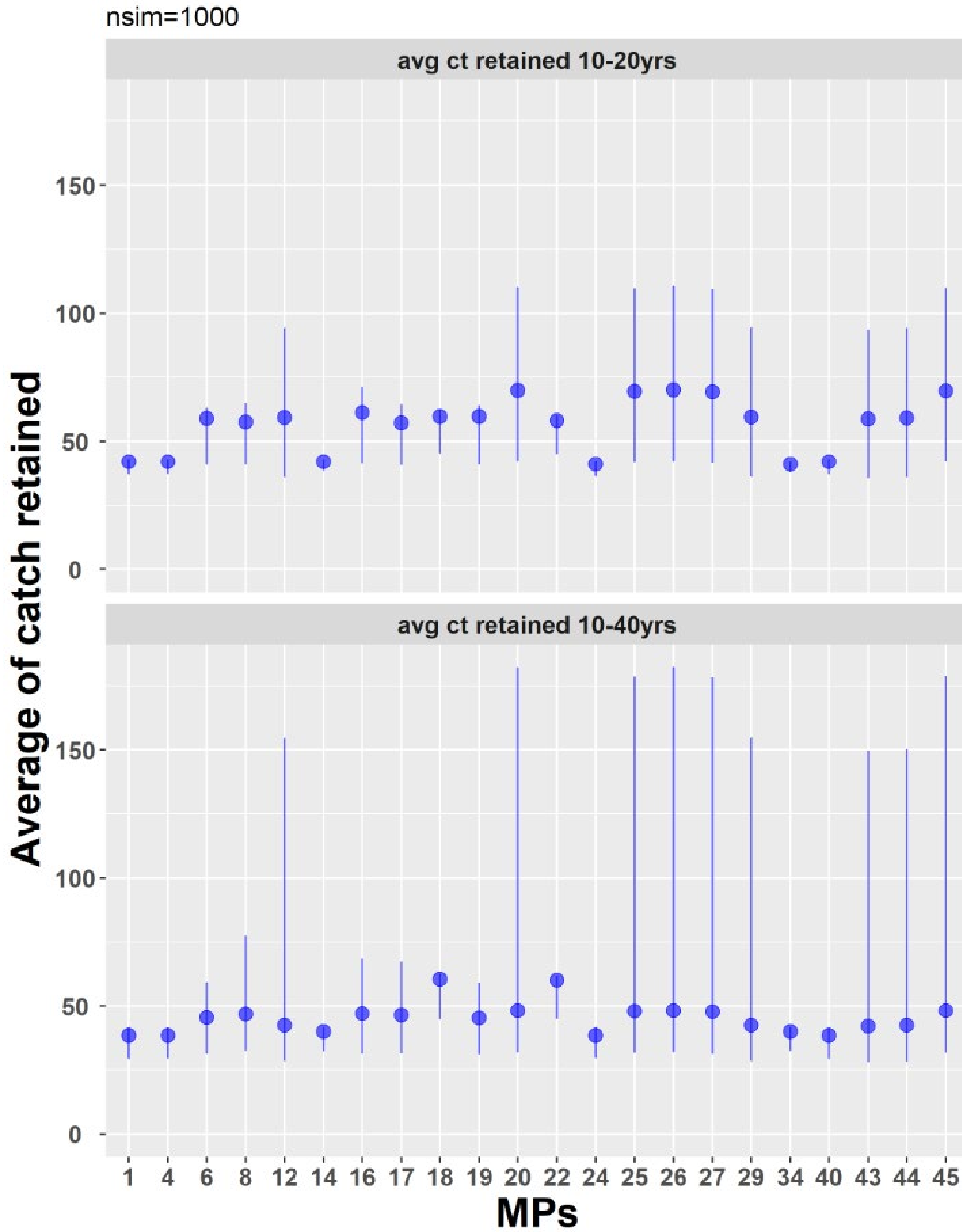

Figure 19. Average catch retained 10-20 years and 10-40 years for the 21 management procedures under the base case operating model. Dots show medians and bars shown interquartile intervals.

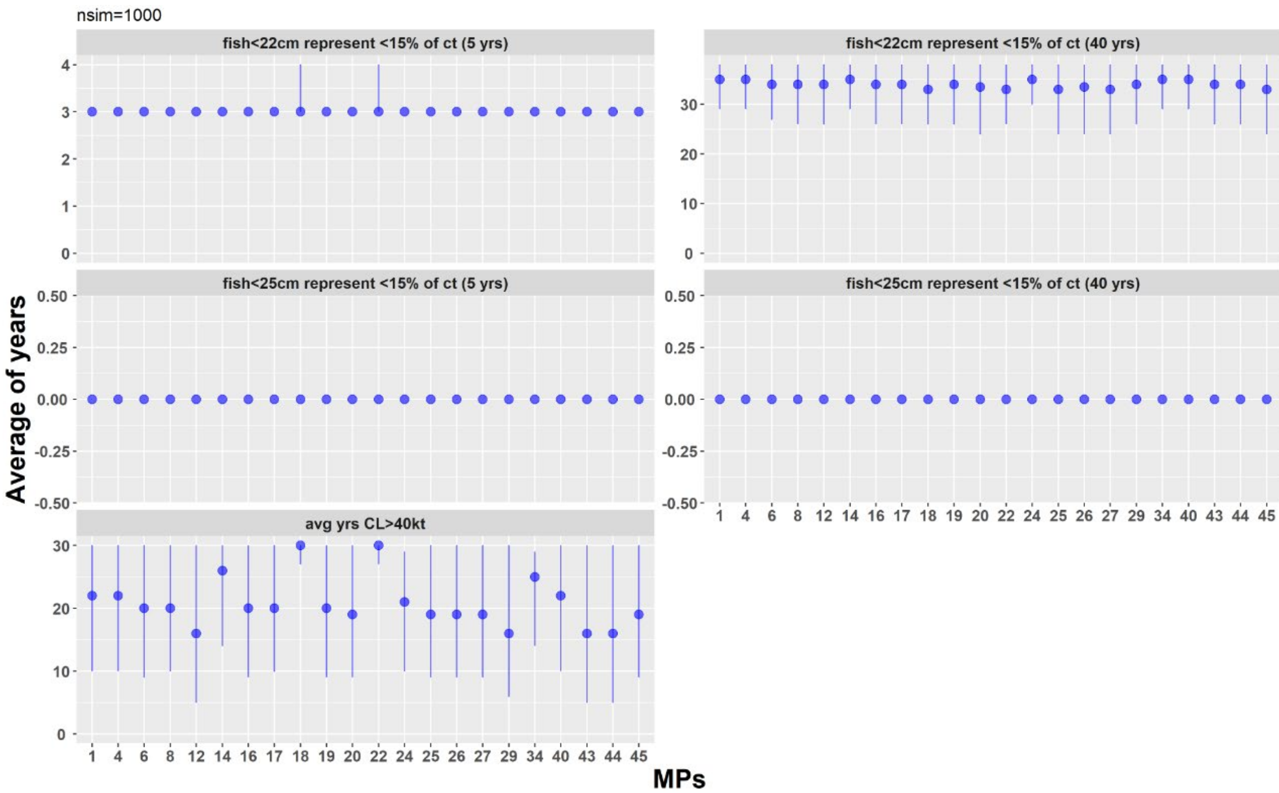

Figure 20. Plots of the average number of years where (1) fish < 22 cm represent < 15% of the catch in the first 5 years and over 40 years (2) fish < 25 cm represent <15% of the catch in the first 5 years and over 40 years, and (3) catch limits are the same as or greater than 40 kt over the 40 year horizon procedures under the base case operating model. Dots show medians and bars shown interquartile intervals.

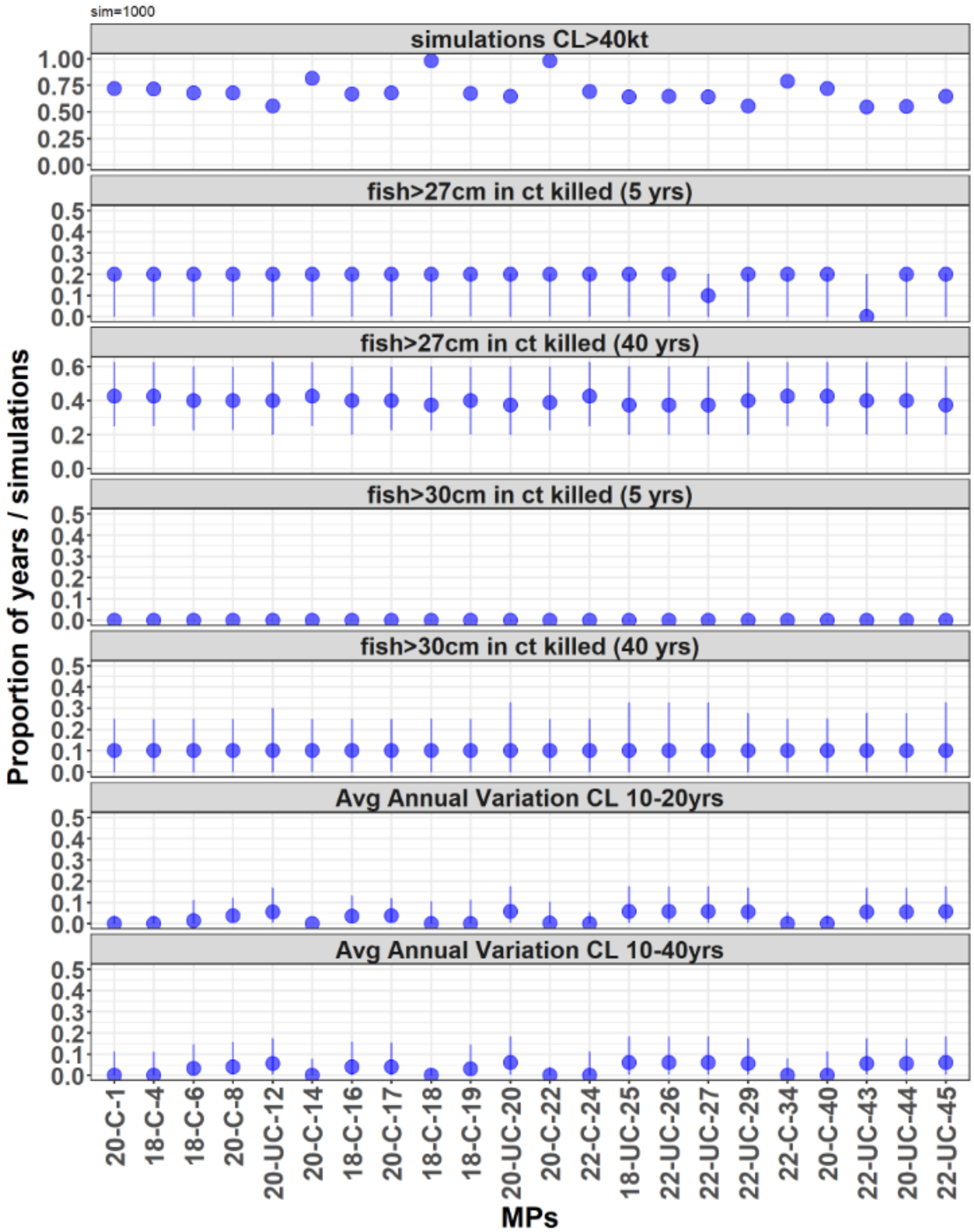

Figure 21. Plots of the average proportion of (1) years where the catch limit exceeded 40 kt, (2) years where the percentage of fish > 27 cm is > 80% of the catch in the first five years and 40 years, (3) years where the percentage of fish > 30 cm is > 80% of the catch in the first five years and 40 years, and (4) the average annual proportional variation in catch limits from 10-20 years and 10-40 years procedures under the base case operating model. Dots show medians and bars shown interquartile intervals.

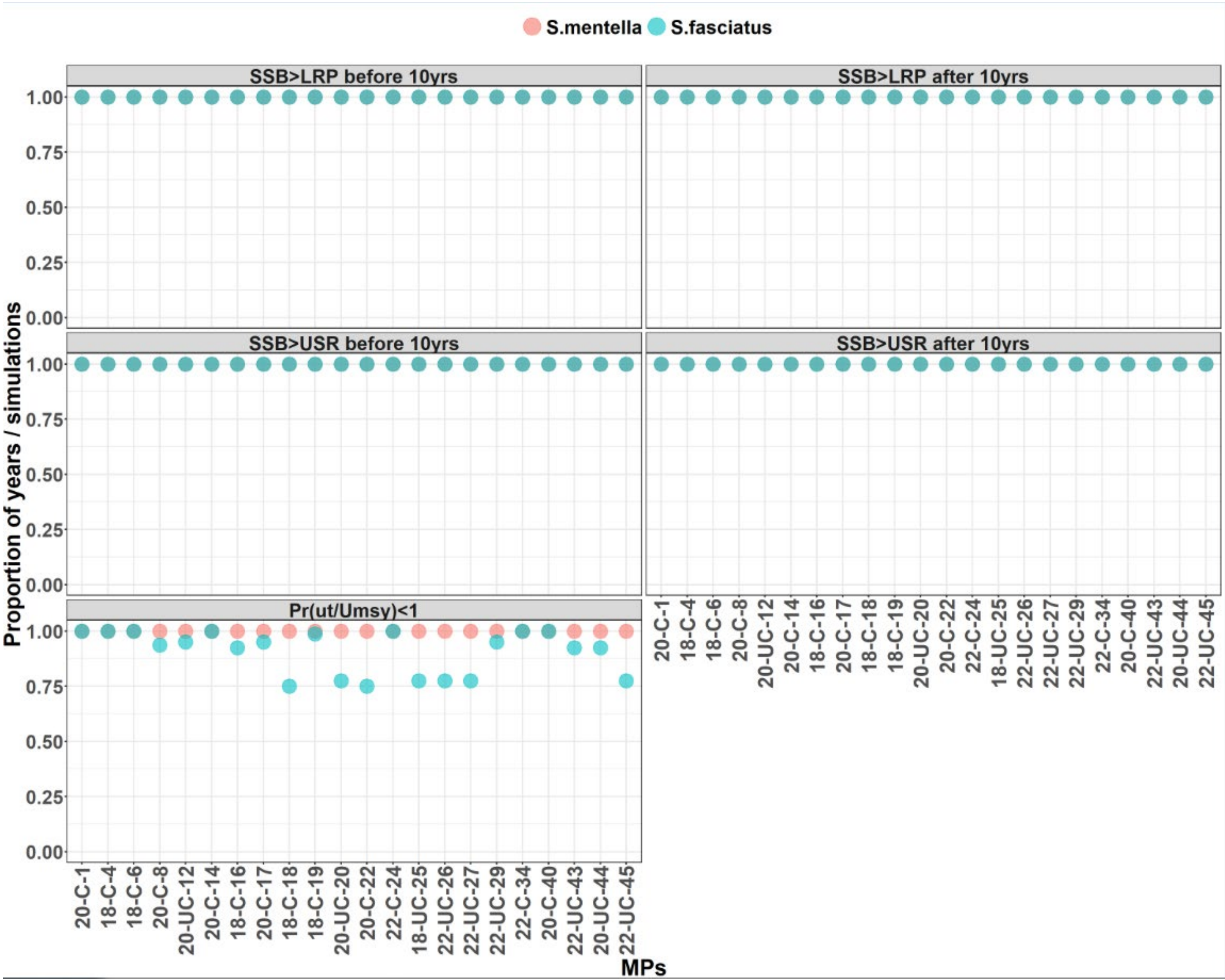

Figure 22. Plots of the mean proportion of years where (1) the SSB exceeds the LRP before 10 years, 2) the SSB exceeds the USR before 10 years, (3) the SSB exceeds the LRP after 10 years, (4) the SSB exceeds the USR after 10 years, and (5) the ratio of harvest rate to u_{msy} is less than 1 for S. fasciatus and S. mentella procedures under the base case operating model. Dots show medians and bars shown interquartile intervals.

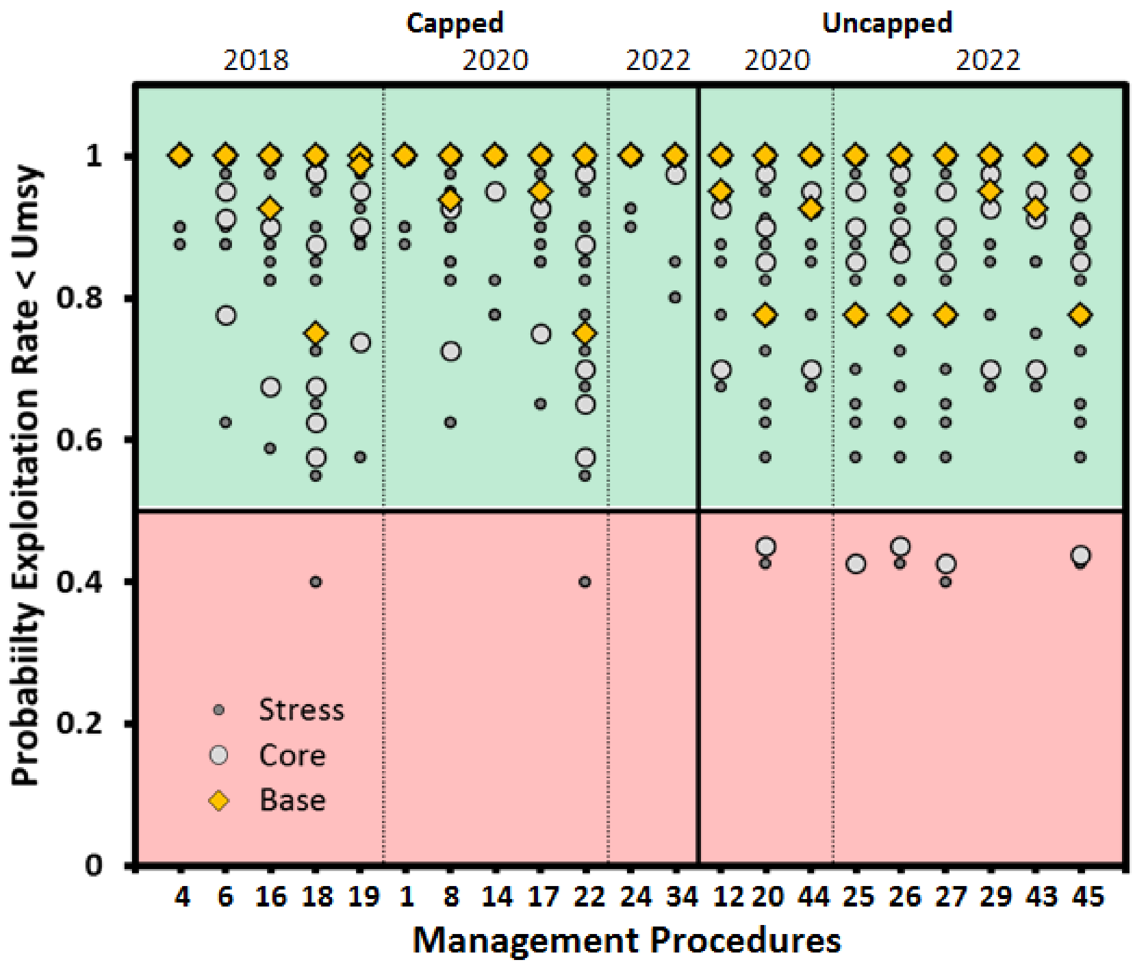

Figure 23. Plot of the probability that the harvest rate on each species remains below the u_{msy} for each of the management procedures under the core and stress test operating models. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models. The green area shows the range of values for suggested passing performance. Red shows the range of values for suggested failing performance.

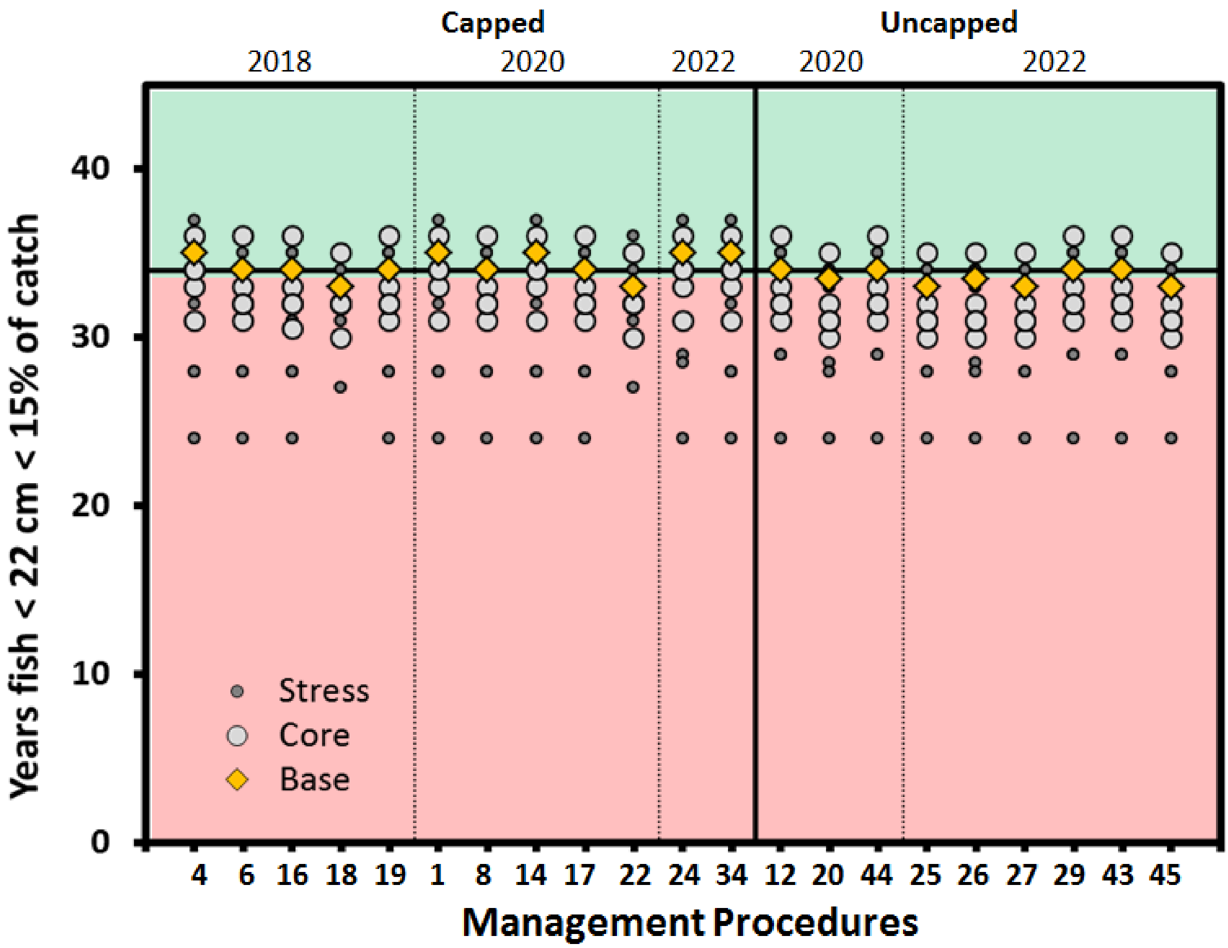

Figure 24. Plot of the number of years fish < 22 cm are less than 15% of the catch for each of the management procedures under the core and stress test operating models. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models. The green area shows the range of values for suggested passing performance. Red shows the range of values for suggested failing performance.

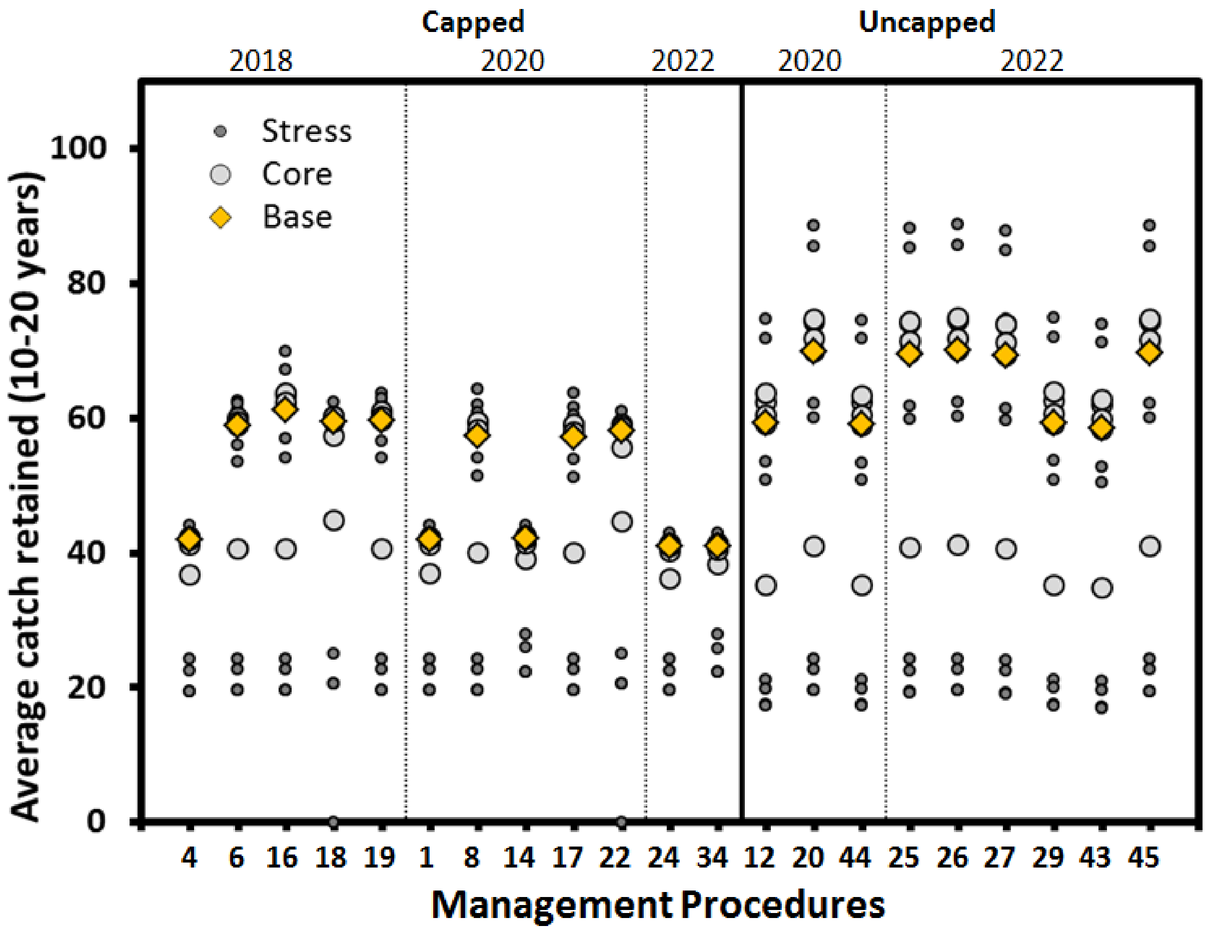

Figure 25. Average annual catch retained 10-20 years. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

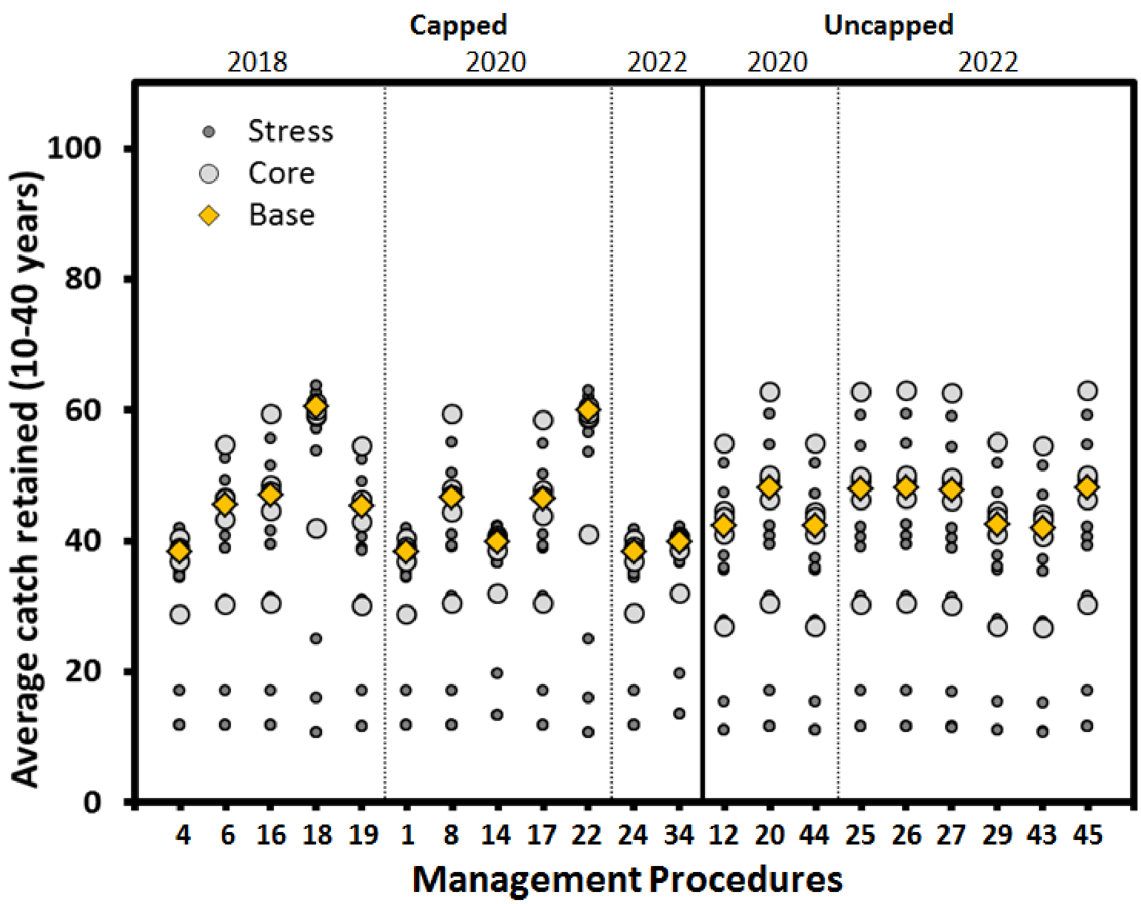

Figure 26. Average annual catch retained 10-40 years. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

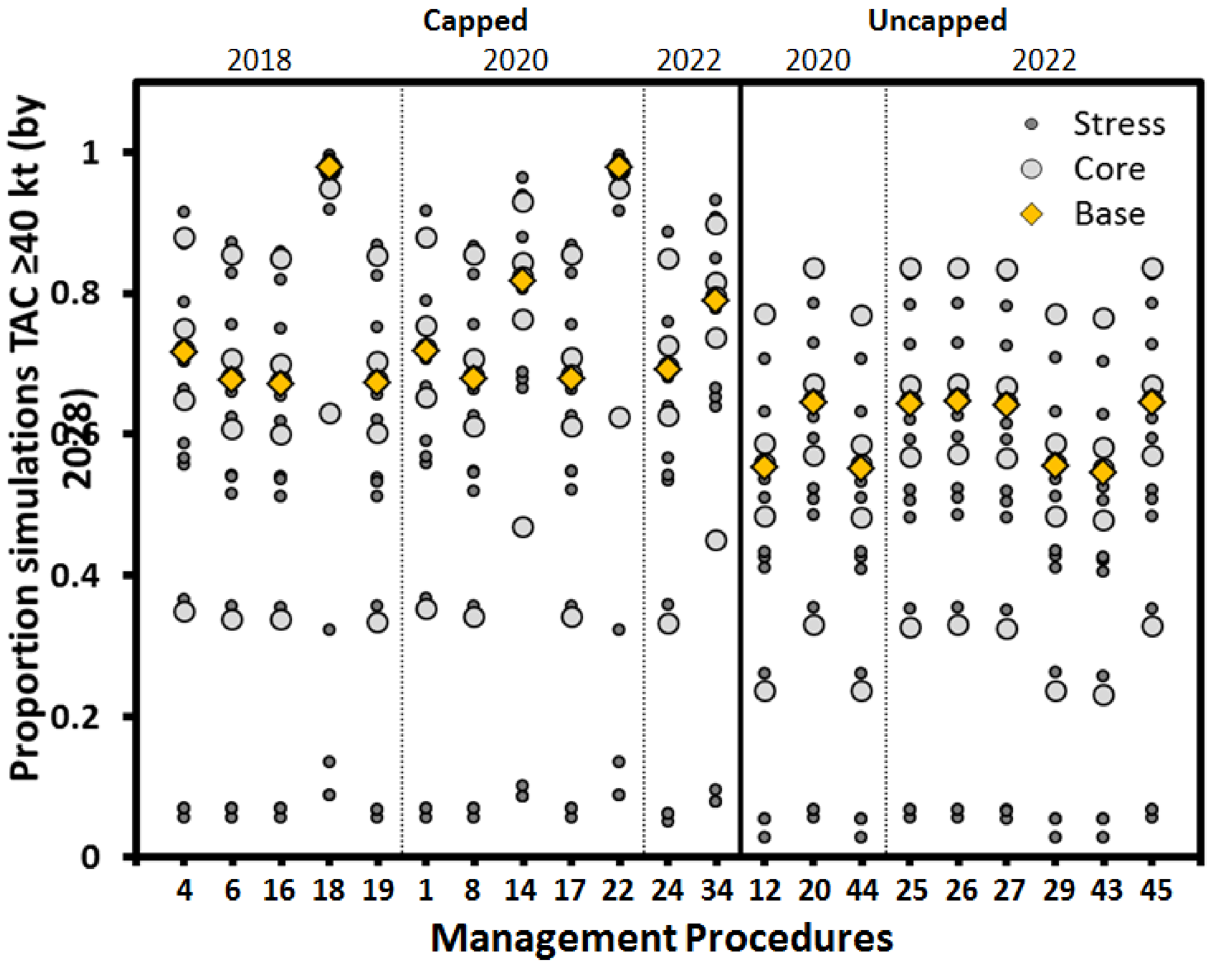

Figure 27. The proportion of simulations where catch limits were greater than or equal to 40 kt by 2028. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

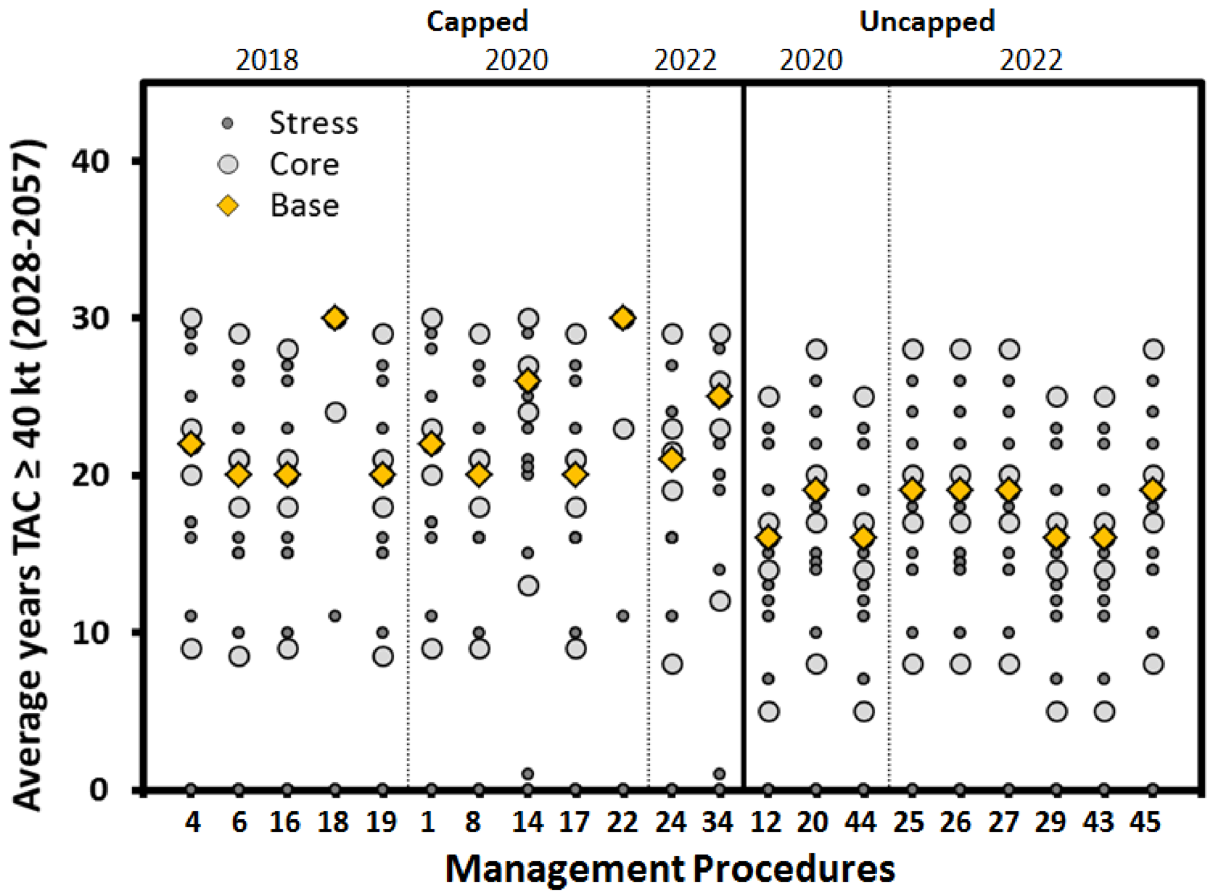

Figure 28. Mean number of years where catch limit ≥ 40 kt between 2028-2057. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

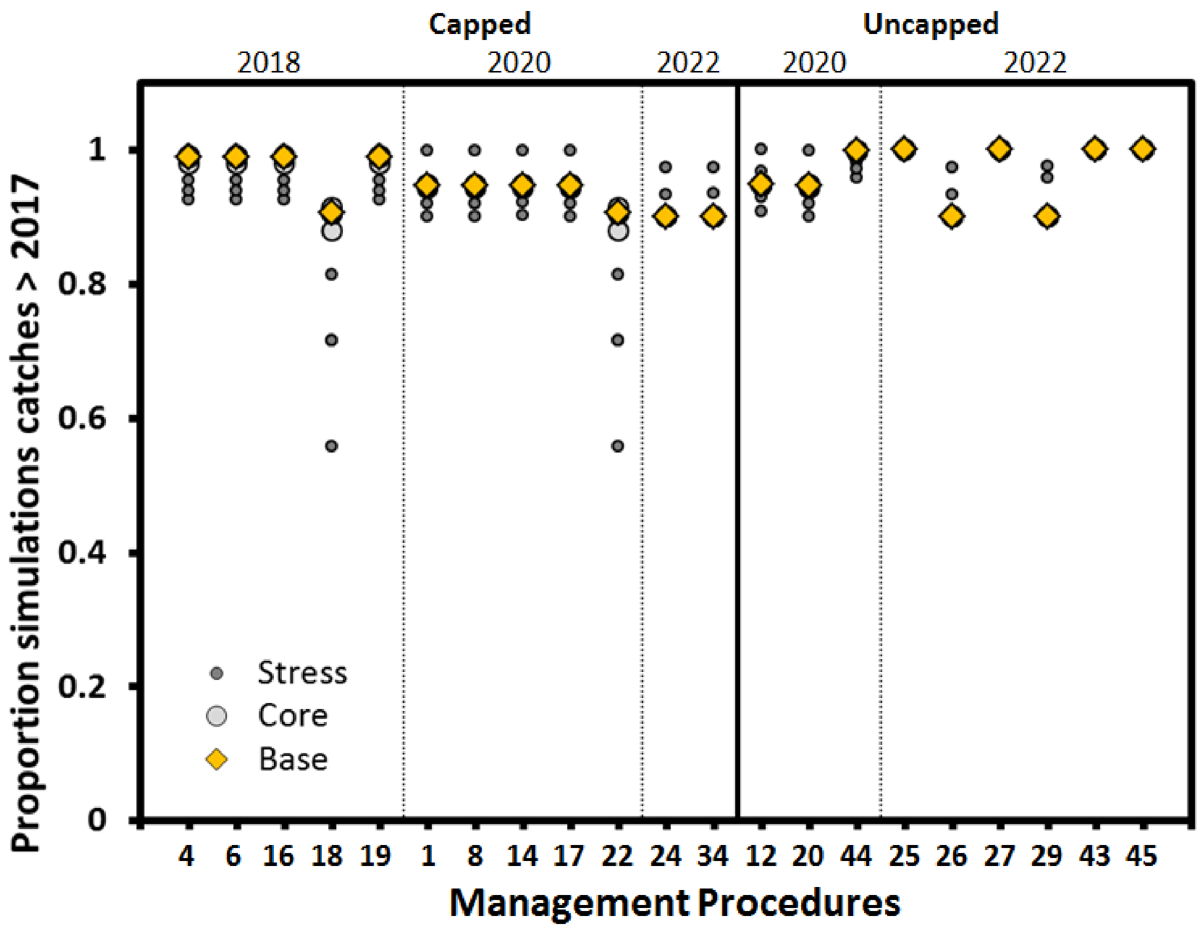

Figure 29. Proportion of years where landings exceed 2017. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

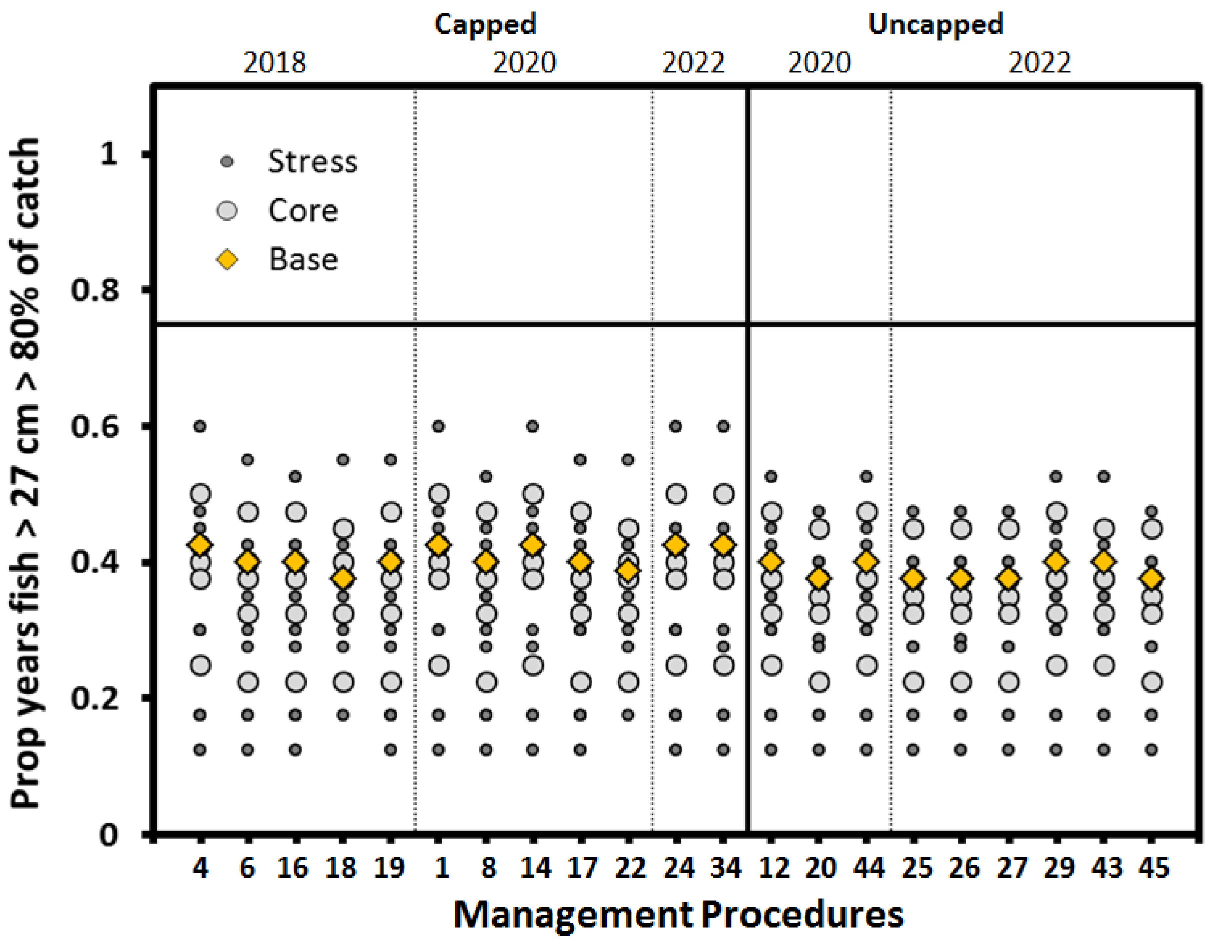

Figure 30. Proportion of years catch of large fish (>27 cm) in a) 5 years and b) 40 years. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

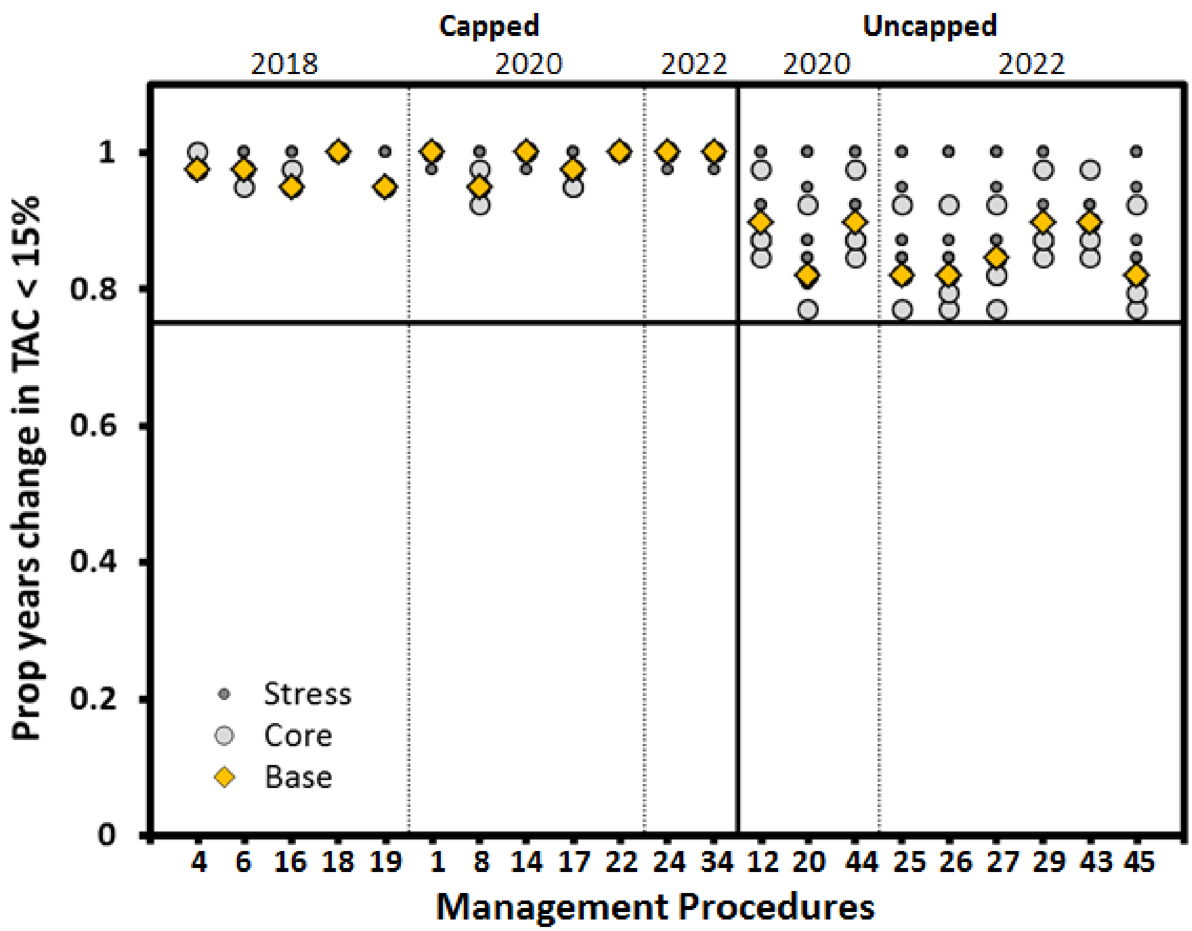

Figure 31. Proportion of years the change in catch is less than 15%. Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

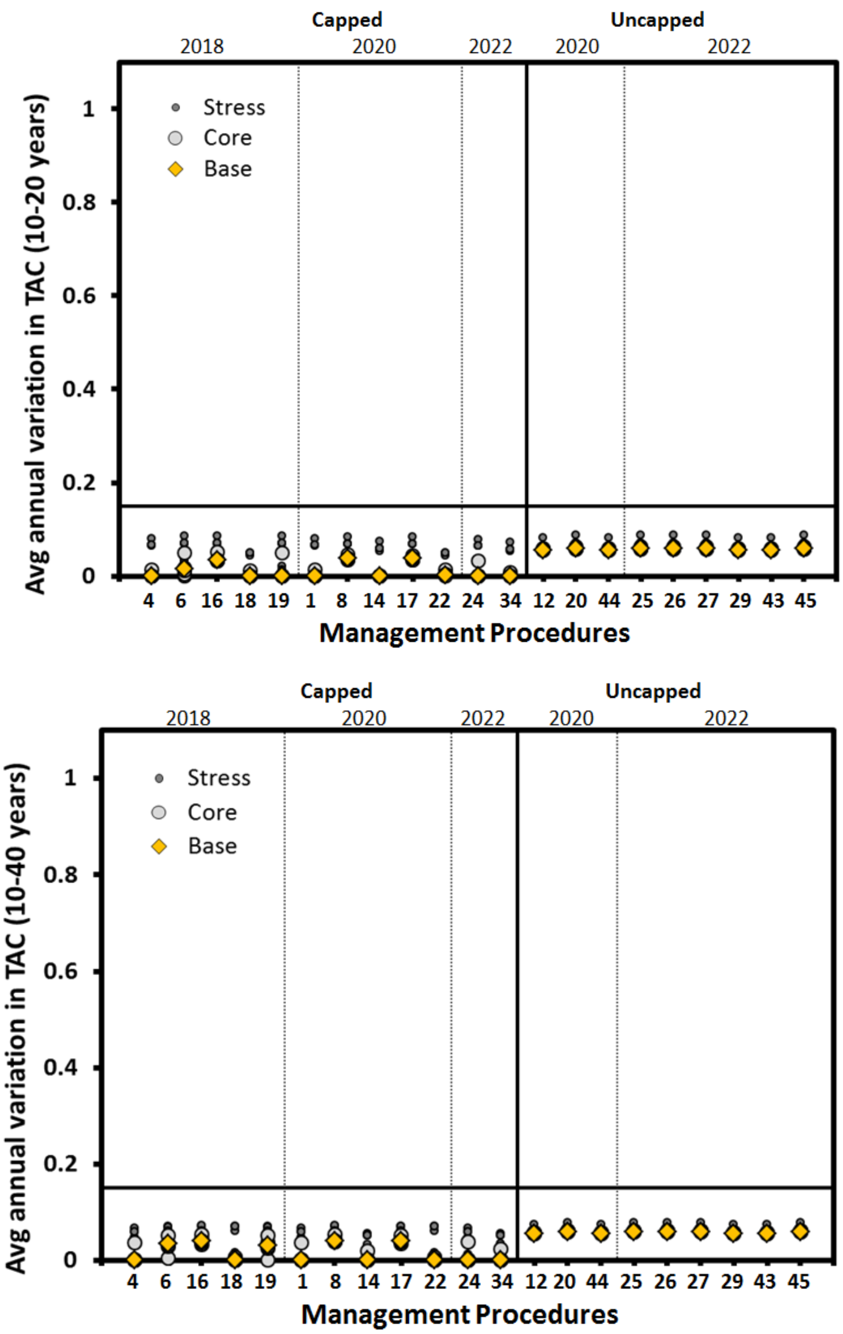

Figure 32. Average annual variation in catch 10-20 years (upper panel) and10-40 years (lower panel). Yellow diamonds show the results for the base case operating model. White circles show results for the core operating models other than the base case. Small grey dots show the results for the stress test operating models.

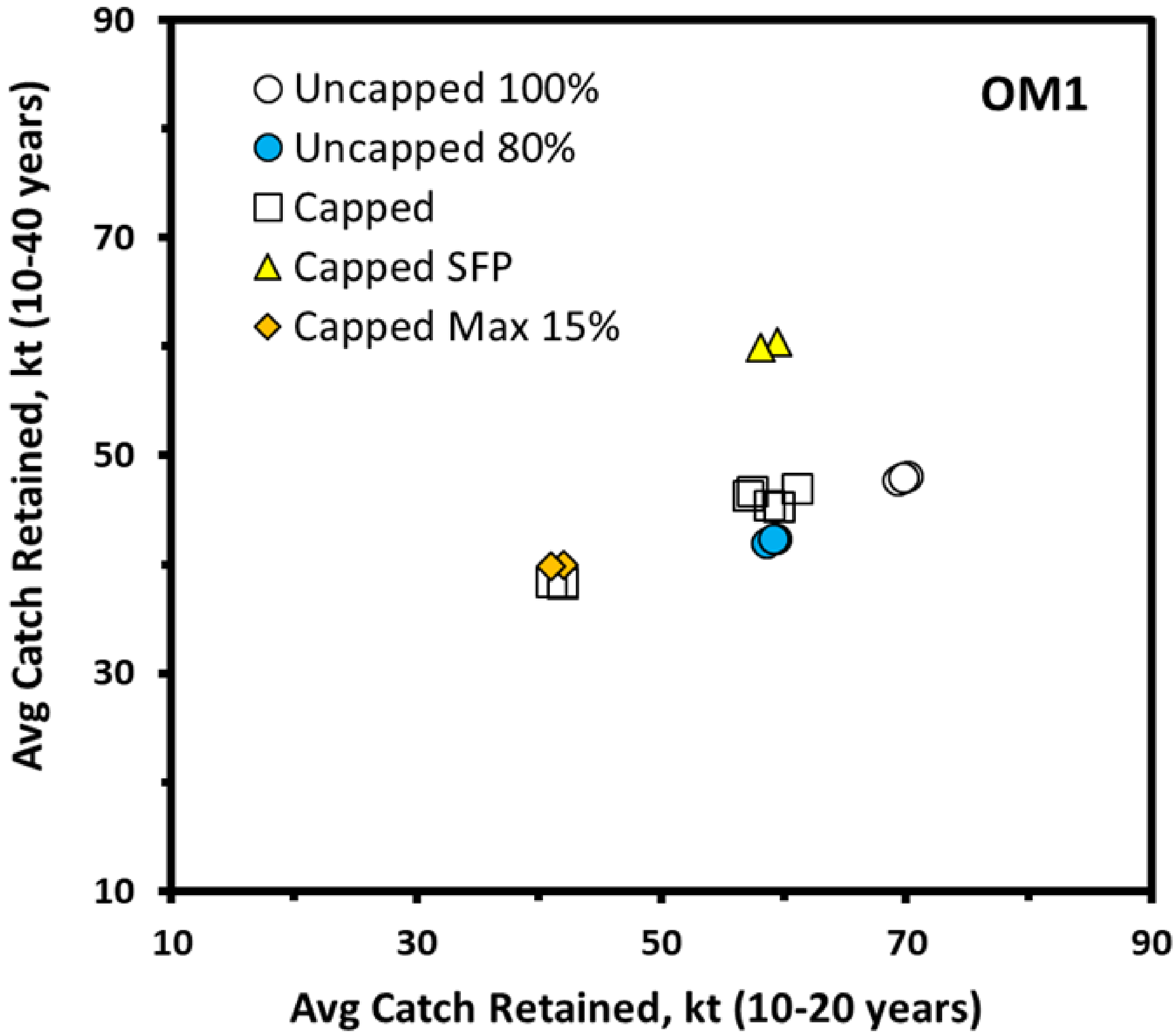

Figure 33. Trade off plot between average catch retained 10-40 years and average catch retained 10-20 years under the base case operating model.

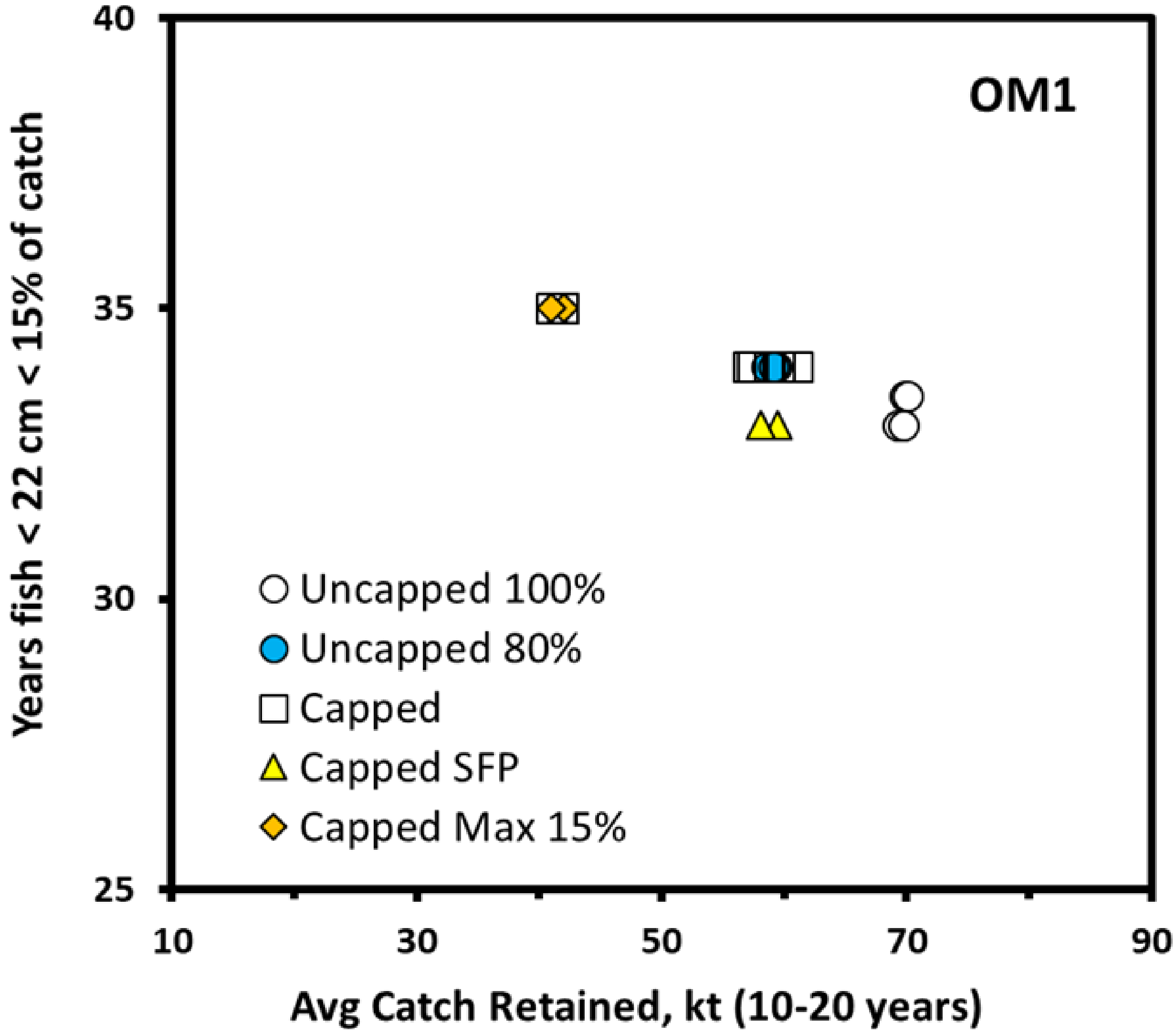

Figure 34. Trade off plot between average catch retained 10-20 years and years where fish <22 cm make up < 15% of the catch under the base case operating model.

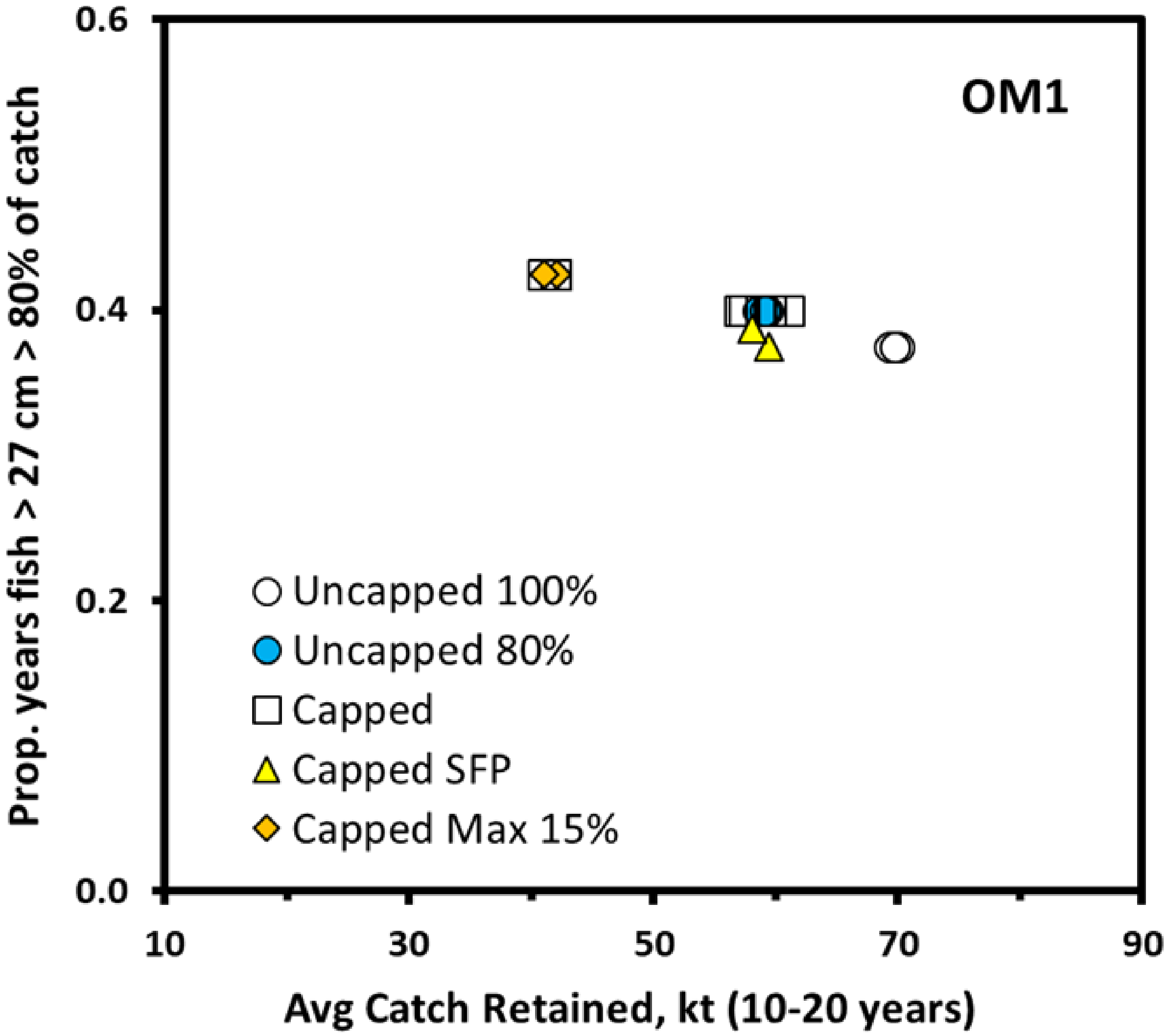

Figure 35. Trade off plot between average catch retained 10-20 years and years where fish >27 cm make up > 80% of the catch under the base case operating model.

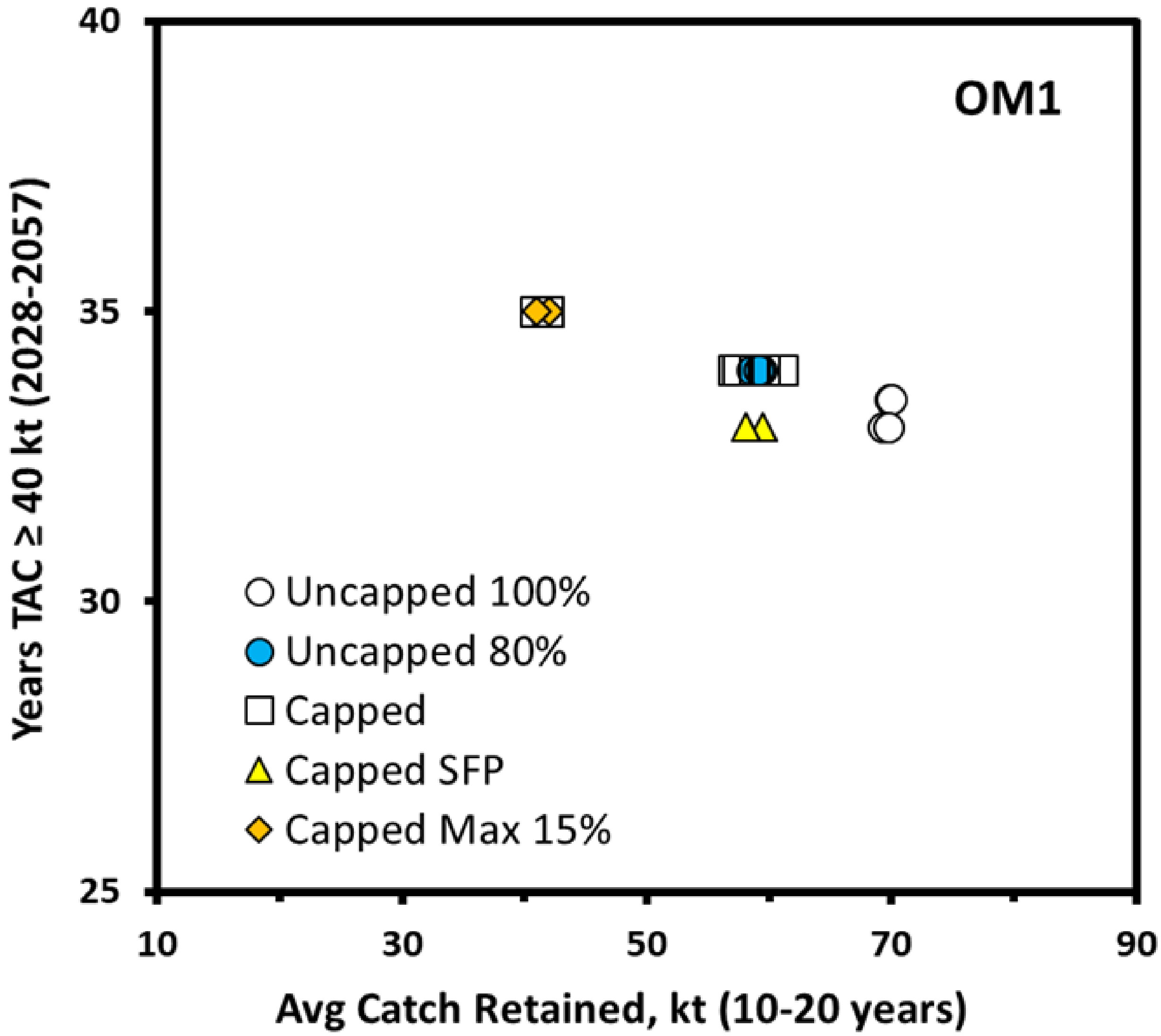

Figure 36. Trade off plot between average catch retained 10-20 years and average years where TAC ≥ 40 kt under the base case operating model.

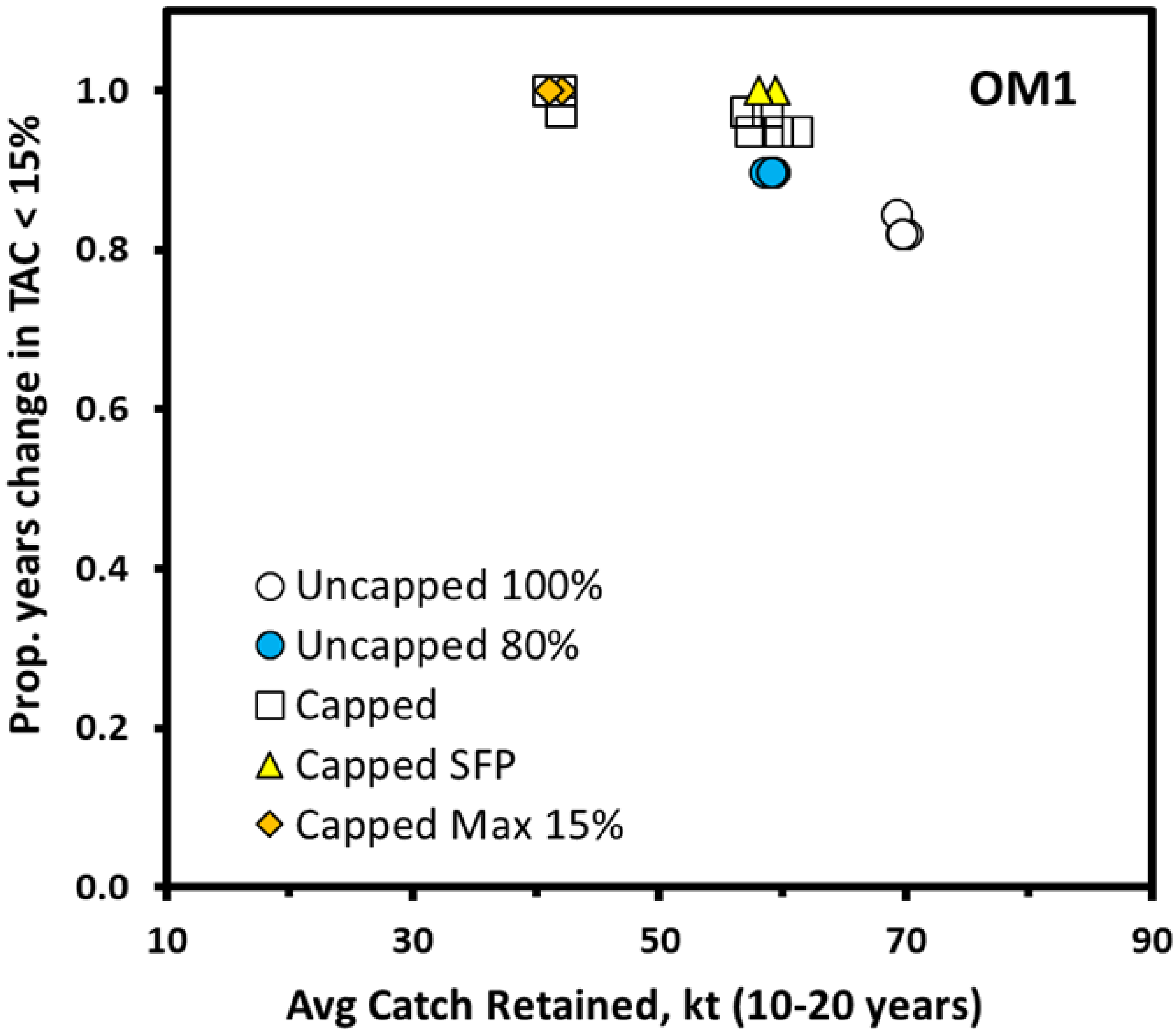

Figure 37. Trade off plot between average catch retained 10-20 years and proportion of years where the change in catch limit < 15% under the base case operating model.

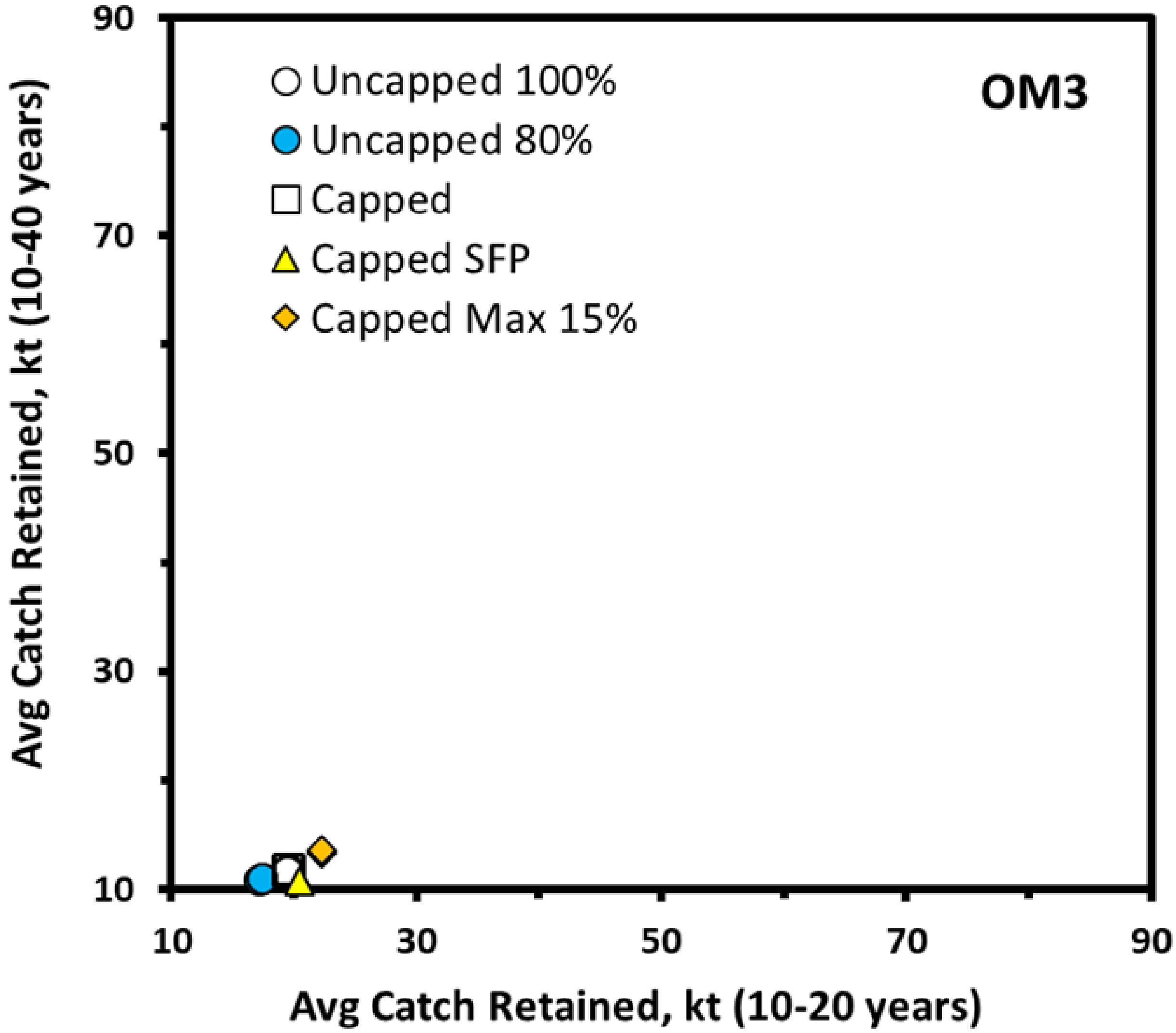

Figure 38. Trade-off plot between the average retained catch 10-40 years and average catch retained 10-20 years under the stress test operating model with a doubling of natural mortality over the next 20 years (OM3).

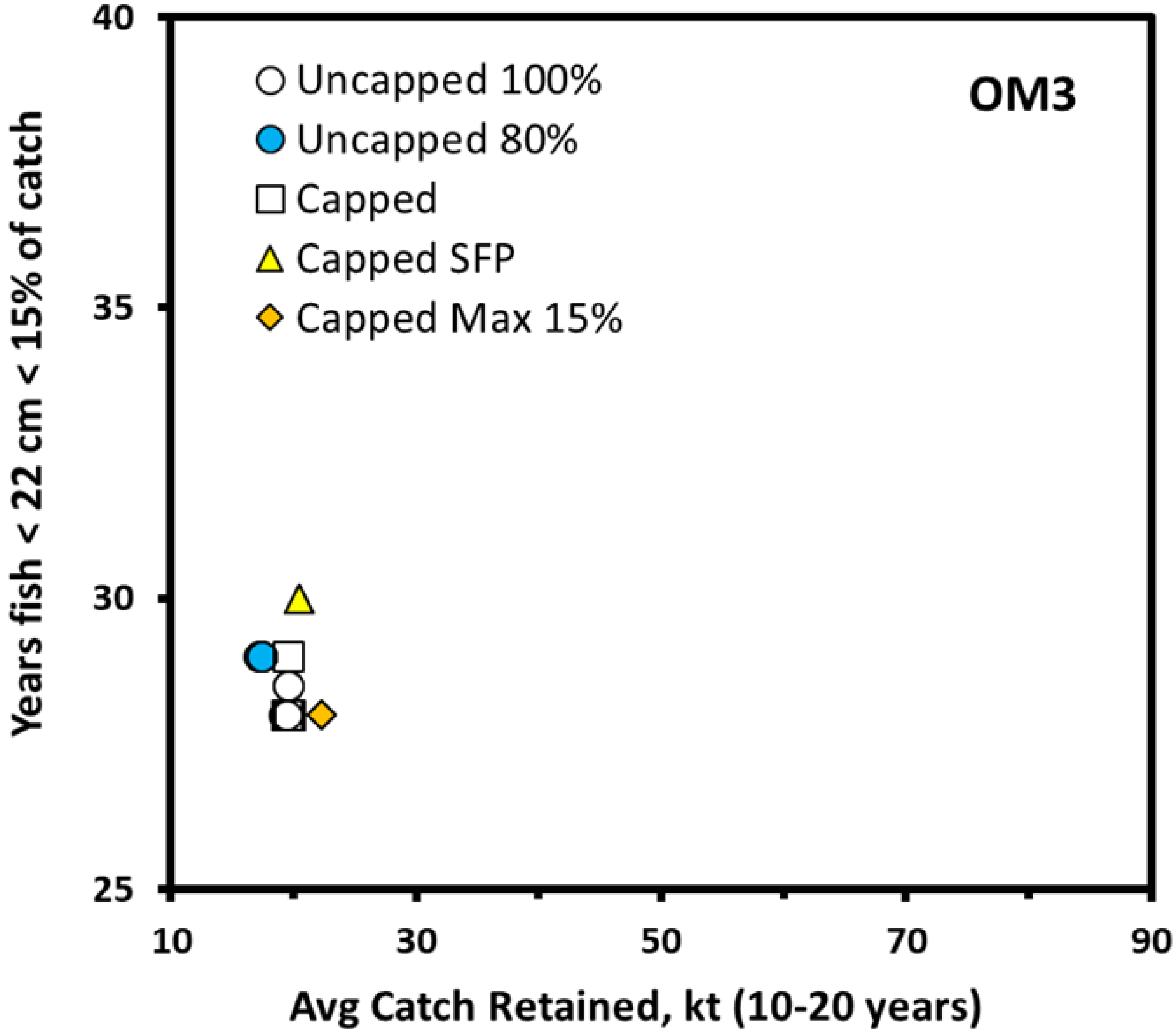

Figure 39. Trade-off plot between the average retained catch 10-20 years and the number of years where fish < 22 cm make up less than 15% of the catch under the stress test operating model with a doubling of natural mortality over the next 20 years (OM3).

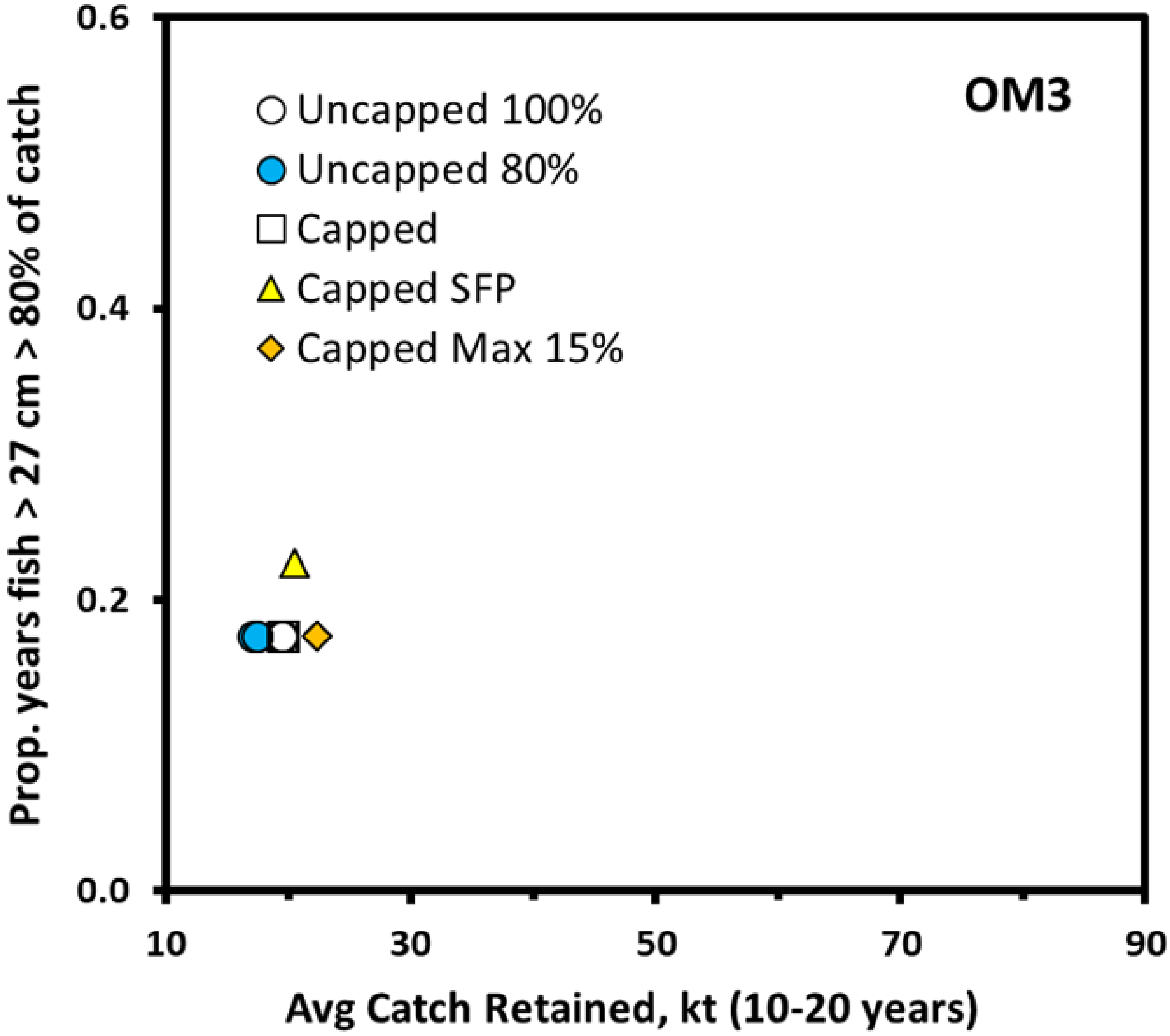

Figure 40. Trade-off plot between the average retained catch 10-20 years and the proportion of years where fish > 27 cm make up at least 80% of the catch under the stress test operating model with a doubling of natural mortality over the next 20 years (OM3).

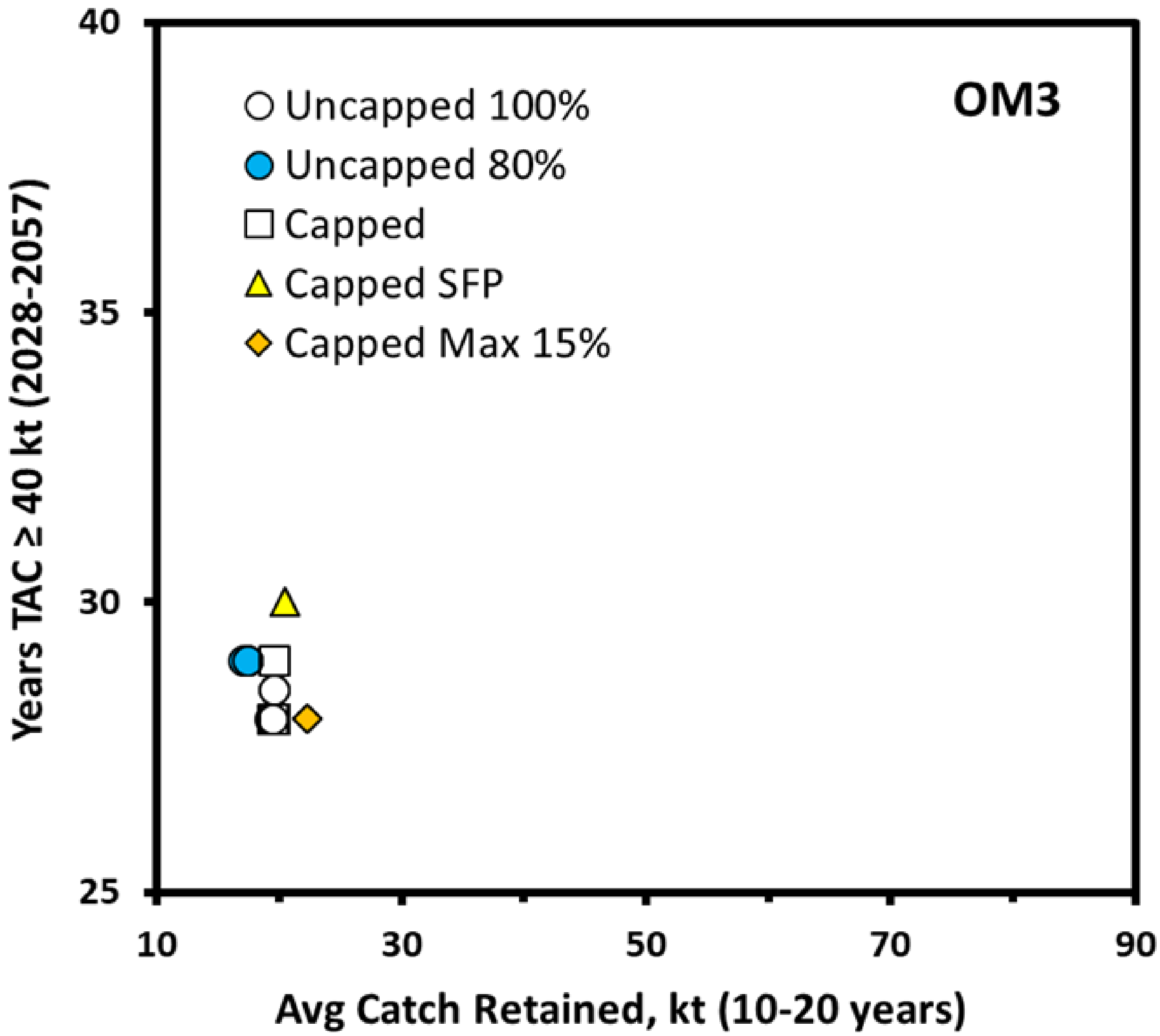

Figure 41. Trade-off plot between the average retained catch 10-20 years and the number of years the TAC is larger than or equal to 40 kt under the stress test operating model with a doubling of natural mortality over the next 20 years (OM3).

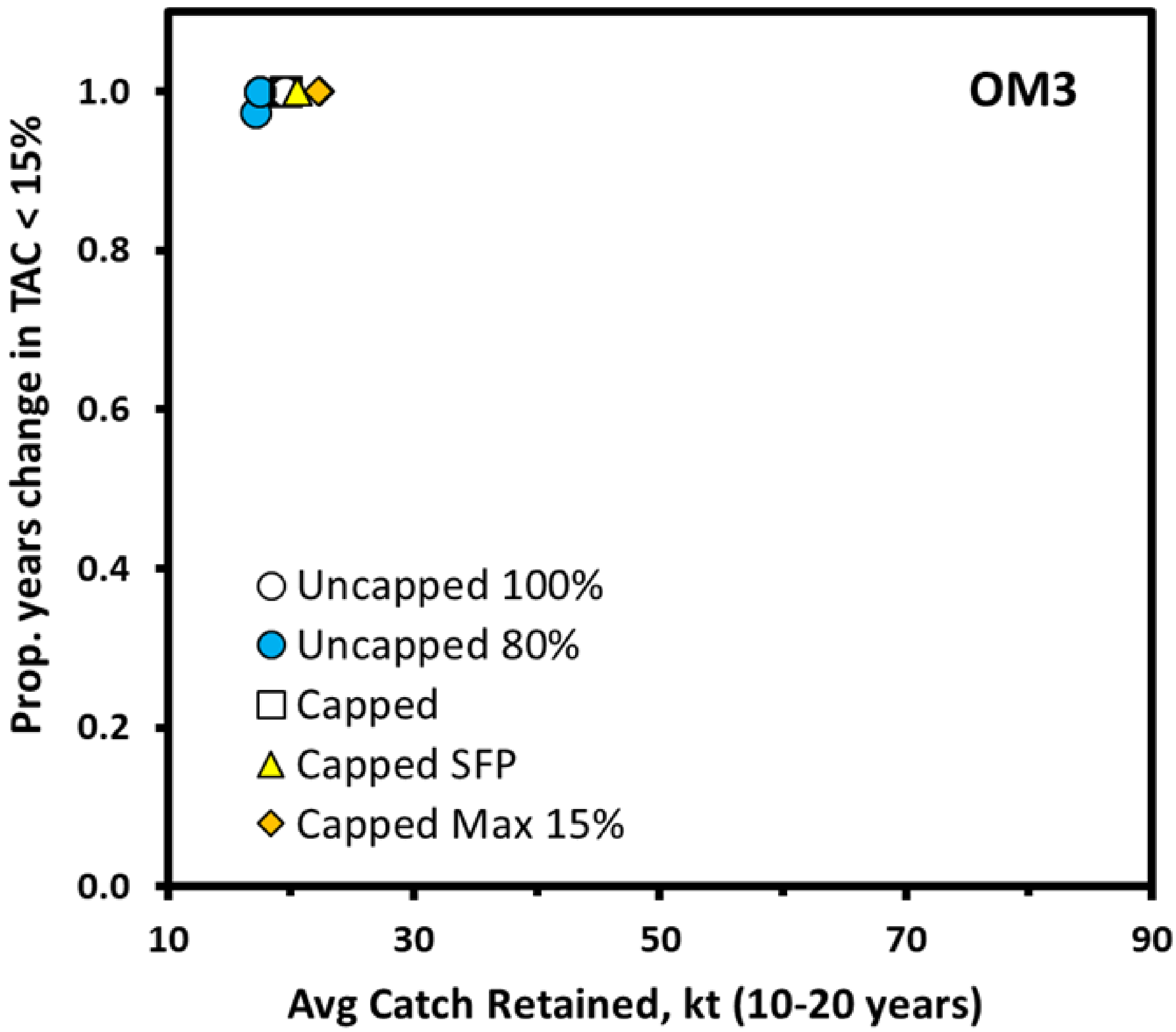

Figure 42. Trade-off plot between the average retained catch 10-20 years and the proportion of years where the change in TAC is less than 15% under the stress test operating model with a doubling of natural mortality over the next 20 years (OM3).

DISCUSSION

The stock assessments of Unit 1 and 2 redfish from 2011-2016 were plagued with data issues and failed to provide satisfactory fits to the data for a number of reasons (DFO 2016a). These included potentially large unreported discards prior to 1995, contamination of trawl survey biomass and length composition records for the Unit 1 trawl survey by stock components that utilize Unit 1 as nursery area but recruit to the Grand Banks before maturation, and changes in the types of trawl gear applied between earlier and later years of the fishery, among other things. Partly due to these issues, there was dissatisfaction with the stock assessment approach and this paved the way for the consideration and adoption of a MSE approach to the Unit 1 and 2 redfish fishery. The MSE was initiated in January 2017 with some initial modelling work and then officially in March 2017 with a series of working group meetings between fisheries managers, stakeholders, and fisheries scientists. This report documents the development and application of models to support the MSE approach for the Unit 1 and 2 redfish fishery and the results obtained from it.

The 18 different operating models formulated for the Unit 1 and 2 redfish MSE fitted the Unit 1 and 2 trawl survey index data and Unit 1 trawl survey length composition data much better than in previous stock assessments (e.g., Duplisea et al. 2016) but still not that well. It fitted the Unit 2 survey length composition and fishery length composition less well for *S. mentella* than for *S. fasciatus*. Considerably better fits to length composition data, particularly the fishery length composition, could only be obtained under what were judged to be unrealistic parameter estimates, e.g., an extremely low value (i.e., <0.25) for the steepness stock-recruit parameter.

The estimates of the constants of proportionality for survey q remained high for Unit 2 in all of the operating models developed and applied in this MSE. Despite numerous attempts to identify plausible hypotheses for the high estimates of survey q, and develop models to represent these hypotheses, we were unable to formulate an operating model that could satisfactorily fit the data and also avoid obtaining high estimates of Unit 2 survey q in the time available. Some exploratory stock assessment modeling in a spreadsheet format however could achieve fairly good fits to the trawl survey biomass indices and also provide estimates of q for both trawl surveys lower than 1. For example, extremely (unrealistic) high catches (e.g., with catch killed 16x the reported catches) in the late 1980s and early 1990s could bring the estimates of constant of proportionality for Unit 2 (q_2) to values less than 1.

Spatial plots of catch rates in the Units 1 and 2 bottom trawl surveys (Appendix G) however suggested the possibility of range contraction to Unit 2 when stock levels were at low abundance. Some trial versions of stock assessment modeling approaches that accommodated range contraction could also bring the estimates of q_2 to values less than 1 and also generate fairly good fits to the trawl survey abundance indices (Appendix G). It is thus recommended that in the next round of MSE modeling of Unit 1 and 2 redfish that sufficient time be allocated to developing operating models that represent range contraction in the two stocks and in how this affects the trawl survey stock biomass indices in Units 1 and 2.

Uncapped MPs 20, 25, 26, 27, and 45 failed one stock PM under a core model, i.e., OM10 in which lower fractions of *S. fasciatus* are assumed in the historical catches; these MPs and MPs 14 and 19 also failed under stress test OM14a which considers a lower ratio of catch biomass killed to retained. Both OM10 and OM14a which suggest lower catches, especially for *S. fasciatus*, predict lower absolute abundance for *S. fasciatus*, especially in the most recent years. Thus, these uncapped MPs which tend to prescribe larger catches than the other MPs will result in higher harvest rates and failure under the u_{msy} PM under these two operating models.

There were considerable differences between capped vs uncapped MPs in average catch retained, average inter-annual variation in catch. Larger average catches were obtainable with uncapped MPs but lower inter-annual catch variation obtainable with capped MPs.

There were trade-off between catch retained vs. catch variability and stock objectives for uncapped MPs. Larger catches could be obtained but high variability in catch limits could also be expected with uncapped MPs. These trade-offs were less severe for ramp-up and capped MPs.

The performance metric scores obtained for the MPs were sensitive to the following:

- Specifications for the maximum cap;
- But not the start year for the HCR;
- OM specifications – e.g., no strong recruitment for 40 years, lower catch killed ratio for *S. fasciatus*, M, steepness;

- Capped MPs with maximum allowed change of 15% performed better under OM1 but worse under OM3;
- Capped MPs with small fish protocols + Capped with lower slopes, allowed larger catches and better stock outcomes under OM3, OM5, and OM14a.

All operating models predicted high fractions of small fish in the catch through to 2020.

- Small fish protocol triggered 100% 2018-2019
- 20% of catch < 25 cm 2018-2021.

CONCLUSIONS

For several of the performance metrics, the performance of the capped and uncapped MPs showed no substantial differences. This is attributed partly to the effect of the strong 2011-2013 cohorts now entering the redfish fishery and the incorporation of a single generic HCR that specified catch limits based on survey biomass values obtained from the Unit 1 trawl survey. The presence or absence of maximum caps or adjustment of HCR catch limits by 0.8 mattered much more to candidate MP performance than the year in which the HCR was first implemented. Differences in MP performance related to trade-offs average catches retained 10-20 years in the future against the duration of high TACs (≥ 40 kt), TAC stability (chance that annual changes in TAC will be less than 15%), number of years where the small fish protocol will be avoided (fish < 22 cm < 15% of the catch), and the number of years with a high percentage of large fish in the catch (fish > 27 cm > 80% of catch). The strength of these trade-offs depends on the operating model examined.

While the stock conservation objective of reaching and maintaining the SSB in the healthy zone for both species was met in all tested MPs, some uncapped MPs, e.g., MP45 failed to maintain exploitation rates below u_{msy} for *S. fasciatus* with 50% probability in one core operating model (OM10a).

The abundant small fish from the 2011-2013 cohorts results in predictions that catches from all candidate MPs will invoke the Small Fish Protocol 2018 and 2019 and that there will be a high percentage of small (< 22cm) fish in catches until 2020. Even with the proportion of small fish in fishery catches greater than that permitted within the small fish protocol, the modeling predicted that this would not compromise the achievement of other conservation objectives.

RECOMMENDED IMPLEMENTATION PERIOD

Once a MP is chosen, this MP must be adhered to in setting catch limits for a fixed number of years (typically not more than 5). At the end of the implementation phase, a retrospective analysis should take place in which the actual performance of the MP is evaluated, and updates to MP design, OMs, simulation procedures, objectives, performance metrics, and exceptional circumstances are re-evaluated in a new MSE process.

An implementation period of five years is recommended. This duration will balance the period of time required to phase in the M, and the collection of sufficient fishery and survey data to assess the MP's performance. Assuming that an MP from the present MSE will be implemented in 2019, a second MSE process should be initiated in 2022. This may result in a new or revised MP to be implemented starting in 2023.

INFORMATION SUPPORT REQUIREMENTS

The implementation of an MP from the MSE requires annual updates of key information to inform the HCR and to evaluate whether the Exceptional Circumstances Protocol should be invoked. The necessary information is:

1. The biomass index for *S. mentella* (>30cm) and *S. fasciatus* (> 29cm) from the annual Unit 1 bottom trawl survey.

2. Data on length composition of catches in the surveys and in the fishery in Units 1 and 2, and the mature survey biomass of both redfish species in the Units 1 and 2 surveys.

The above two data components would need to be collected and analyzed appropriately to compute a combined species total annual catch limit in each year and to evaluate whether the exceptional circumstances protocol should be invoked.

EXCEPTIONAL CIRCUMSTANCES PROTOCOL

Exceptional circumstances are commonly defined in the MSE process, where a decision could be taken for the implementation of a MP to stop before the pre-determined implementation period comes to an end. These circumstances describe events that are sufficiently outside the range for which the MP in use has been tested against in simulation, such that confidence in MP performance may be reduced. Such circumstances include:

1. Survey Index Ratio

- Beginning in 2019, if the survey index ratio (J_y) for either *S. mentella* or *S. fasciatus* falls below 0.35 (i.e., lowest historical value) or is outside the 90% confidence interval for which the survey index ratio is projected to lie for core operating models.

2. Survey Biomass Data

- If the Unit 1 or Unit 2 mature survey biomass indices for either *S. mentella* or *S. fasciatus* fall below their historical lowest values (Unit 1: 1984-2017; Unit 2: 2000-2016), for two consecutive surveys.
- If the Unit 1 survey, which provides the survey index ratio (J_y) for the HCR, has either not taken place or has been substantially curtailed or changed for two consecutive years. While the Unit 2 survey data are used to evaluate mature survey biomass (above) and length composition (below), they are not used in the annual application of the HCR and there is therefore no exceptional circumstance for failure to complete that survey as planned.

3. Length Composition

- A substantial and unanticipated change in the catch length composition structure of the fishery or survey for either *S. mentella* or *S. fasciatus* in either Unit 1 or 2 (either truncated, or spread out). This could result from a significant unanticipated change in fishery or survey selectivity, density-dependent effects, emigration events or the presence of a previously unknown strong cohort. It should be noted that the MSE has already tested the robustness of the candidate MPs to scenarios reflecting potential changes in fishery selectivity (OM22) and density dependent effects (OM3, OM4). What constitutes a substantial and unanticipated change needs to be defined during the first year of implementation of the MSE.

4. Operating model assumptions

- An important change in the understanding of the life history or stock parameter assumptions in the core operating models in the MSE affecting management procedure performance. These may include:
  - The value for a parameter, e.g., for the rate of natural mortality or growth, is found to be significantly different from the ranges tested in the operating models or sensitivity tests.
  - A stress-test model becomes more credible than the core models, and the management procedure has not performed acceptably under this model.
  - No operating models have been developed that adequately address the specific biological change observed (e.g., significant spatio-temporal differences in stock distribution).

ANNUAL REVIEW AND REPORTING

The information required annually for the implementation of the MP from the current MSE, as regards the HCR and Exceptional Circumstances Protocol need to be peer-reviewed to ensure its accuracy and then published to ensure scientific integrity and transparency of the process. Annual Canadian Science Advisory Secretariat Science Responses, or an equivalent process, are recommended as the means by which to achieve annual review and reporting.

RESEARCH RECOMMENDATIONS

Implementing the collection of representative species composition data in fishery catch sampling is a high priority. Data on species composition in fishery catches will improve the fidelity of subsequent MSE processes in the future (i.e., their ability to correctly simulate stock and fishery dynamics) and would contribute to enhancing the sustainable management of *S. fasciatus*, while potentially allowing for higher overall catches of redfish if the species composition of catches can be estimated with high accuracy and precision and if the commercial fishery can reliably target *S. mentella*.

A number of uncertainties concerning important life history parameters were identified in the MSE and were represented using stress-test models. Research aimed at reducing these uncertainties would improve the fidelity of the MSE process. Notably, this includes data on the natural mortality and growth rate of *S. fasciatus* and *S. mentella*. Furthermore, the underlying equation used to model recruitment in the MSE was a Beverton-Holt stock-recruitment function. Given strong evidence of cannibalism in redfish, a Ricker stock-recruitment model could be considered for the 5-year review of the present MSE.

Preliminary analyses undertaken outside the MSE process based on both Unit 1 and Unit 2 surveys (2000-2017) suggest the possibility that Unit 2 densities could remain elevated even at lower stock levels and that as abundance increases, densities increase rapidly in Unit 1. This phenomenon, termed hyper-expansion, would result in a disproportionate increase in the Unit 1 survey index as abundance increases, leading to a risk of setting catch levels under the HCR that are biologically too high. Conversely, abundance declines would result in a disproportionate decrease in the Unit 1 survey index (hyper-depletion), resulting in catch levels under the HCR that cause foregone yield. Such a hypothesis was not simulated in the present MSE. Further research into the spatial distribution and dynamics of redfish prior to the evaluation phase of the present MSE is highly recommended. To support this research, and in anticipation of a possible modified MSE process following the five-year implementation period, the biennial Unit 2 survey should continue and ideally be made annual.

Given the importance of OM10 and OM11, which address uncertainties in the historical catch species-split uncertainty, further research into the feasibility of alternative methods to address the historical species-split in the commercial catches should be explored.

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APPENDIX A – OVERVIEW OF THE UNIT 1 AND 2 REDFISH FISHERY

HISTORY

A fishery for redfish began in the Gulf of St. Lawrence and the Laurentian Channel in the 1950s (DFO 2016a). Until 1993, the fishery was managed as three NAFO divisions, 4RST, 3P and 4VWX. However, in 1993 the fishery was divided into the newly created management Units 1, 2 and 3 to support the then-current understanding of stock structure in redfish, with Units 1 and 2 now considered to comprise the same stock (Kulka and Atkinson 2016; Figure 1). Thus, Unit numbers are used to manage this fishery, instead of NAFO divisions as for many other groundfish stocks in Atlantic Canada.

Unit 1 consists of what is also NAFO divisions 4RST, as well as 3Pn4Vn from January to May. Unit 2 consists of NAFO subdivisions 3Ps4Vs, 4Wfgj, and the subdivision 3Pn4Vn from June to December.

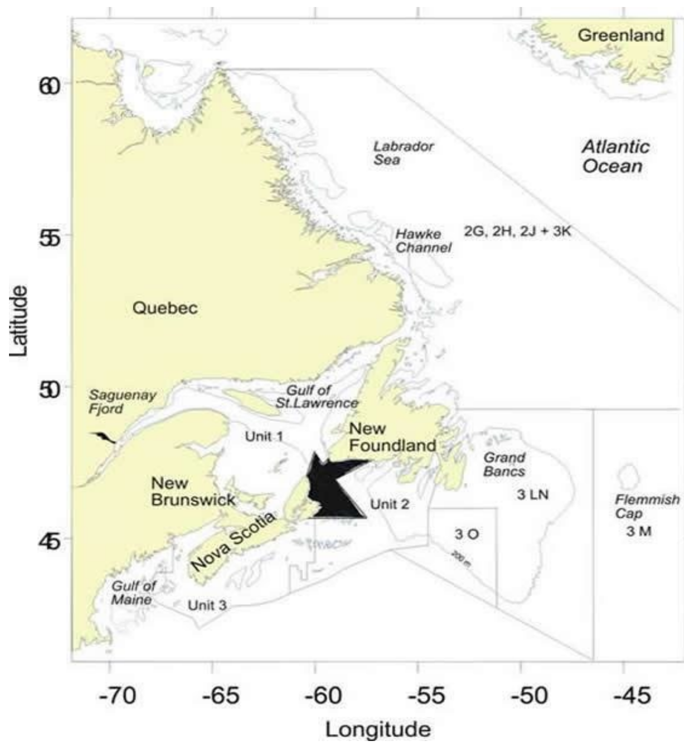


Figure A.1. Map of Unit 1 and Unit 2 redfish management areas in Atlantic Canada. The area in black (3Pn4Vn) is part of Unit 1 from January to May, and Unit 2 from June to December. Image from COSEWIC (2010).

Although never as profitable as cod, redfish aggregations are easily captured by trawl and therefore redfish can be profitably harvested even at low prices (Duplisea 2016). The redfish fishery underwent three periods of high exploitation (1954-1956, 1965-1976, and 1987-1992), after which landings dropped precipitously in 1993-1994 (Brassard et al. 2017). Landings and Total Allowable Catches (TAC) for the Units 1 and 2 redfish fishery have continued to show strong declines since 1993. With the creation of the Units, Unit 1 TAC was set at 60,000 t and Unit 2 28,000 t. Unit 1 has been under commercial moratorium since 1995, with an index fishery since 1998 which is currently permitted a TAC of 2,000 t. Unit 2 continues to support a commercial fishery, with a TAC of 8,500 t since 2010. There is also an international aspect to this fishery as France (through the islands of St. Pierre et Miquelon) has 3.6% of the current Unit 2 TAC, while indigenous harvest is minimal (DFO 2016b).

The Unit 1 index fishery is prosecuted by a single mobile gear fleet (< 19.81 m; DFO 2017). In Unit 2, the Integrated Fisheries Management Plan for 3Ps groundfish describes redfish TAC allocations among several different fleets: offshore vessels over 100' with fixed gear hold an allocation of 77% of the redfish TAC; 3.7% to nearshore fixed gear vessels < 65', 14.4% to nearshore mobile gear vessels < 65', and < 1% to midshore mobile gear vessels 65-100'.

According to the *Atlantic Fishery Regulations* (Government of Canada 1985), groundfish (including redfish) may be recreationally fished without a licence via hand-line or angling. Redfish may be fished commercially by all types of authorized trawls, with 90 mm diamond mesh in the cod end and the lengthening piece, and a minimum of 130 mm diamond mesh in the rest of the trawl. The maximum percentages of bycatch for redfish in Unit 1 are 5% for cod, 15% for Greenland halibut and 5% for other groundfish species in a given fishing trip (DFO 2017), and 10% bycatch for mobile gear fleets >65' in Unit 2 (DFO 2016a). In 1997 the Small Fish Protocol was adjusted to its current form for redfish, prohibiting the presence of fish < 22 cm at quantities greater than 15% of the catch (DFO 1997). Fishing effort in Unit 1 occurs from June 15 to October 31, and is subject to 100% dockside monitoring and 25% observer coverage with a reduction to 10% for vessels using Vessel Monitoring Systems (DFO 2017). The Unit 2 fishery operates July 1 – Oct 31 (fixed gear), or July 1-March 31 (mobile gear) and is subject to 100% dockside monitoring and 10% observer coverage for fixed or 5-20% for mobile gears (DFO 2016a). Various closure periods and areas are in effect to protect mating and larval extrusion periods, migrations in 3Pn4Vn, and to avoid either 4T Greenland Halibut or cod spawning in 4RS (DFO 2016a).

FISHERY GEAR

At present, the gears used in the Unit 1 redfish index fishery are bottom otter trawls (90 mm mesh size), while 20 gillnets (5 ½ mesh size) and bottom otter trawls (90 mm mesh size) are used in Unit 2 (DFO 2016a). Scottish seines comprised a large portion of the landings in Unit 1 between 2007 and 2014, and other gear types (traps, Danish seines, longlines and handlines) contribute small or negligible amounts to landings data (Brassard et al. 2017).

During the period of strong declines in redfish abundance in the 1980s and early 1990s, the redfish fishery went through a substantial change in gear used. The traditional bottom trawl was the principle gear used until the mid-1980s, but a midwater diamond otter trawl was introduced in the early 1980s that could yield double the landings and could be used around the clock instead of only during the day (at night, bottom trawls had too much bycatch; Duplisea 2016). However, at midwater depths, smaller juvenile redfish would be targeted, which may have resulted in high levels of discarding or diversion of small fish into fish meal in a way that was not reflected in the landings data of the time (Duplisea 2016). This midwater trawl contributed a substantial or even majority of landings in Unit 1 between 1987-1994, after which the moratorium was imposed, and played a similar role in Unit 2 between 1989-2000 (Brassard et al. 2017).

The switch of commercial gears from bottom trawl to midwater trawl in the mid-1980s led to increasing selectivity of the gear for smaller redfish (since juvenile fish occupy midwater depths and adult fish occur in deeper waters), but likely reduced bycatch that was considered problematic for bottom trawls, particularly at night (Duplisea 2016). Small redfish have limited commercial value (i.e., fish meal; Duplisea 2016) yet remain vulnerable to the fishing gear before reaching sexual maturity around 20-25 cm. To address conservation concerns, the Small Fish Protocol, first barring fish < 25 cm, then < 22 cm, was implemented in 1996 (DFO 1997).

Multiple stock assessment models have indicated that landings data from the 1980s and early 1990s are difficult to fit, a period of time that represents precipitous declines in redfish stock abundance and therefore an important driver of modelled fish population dynamics. In a recent series of interviews conducted with harvesters active during that time period, there is evidence that large numbers of small fish < 20 cm were in fact caught, but legally discarded or otherwise unreported and therefore not reflected in catch or catch-at-length data from the period (Duplisea 2016). Discard mortality is expected to be 100% due to barotrauma.

APPENDIX B – DATA

FISHERY-INDEPENDENT DATA

DFO has conducted annual summer multidisciplinary research surveys of groundfish and shrimp in Unit 1 (northern Gulf of St. Lawrence) since 1984 (McAllister and Duplisea 2016). Between 1984 and 1990, this survey was performed with a Western IIA bottom trawl from the vessel *Lady Hammond* (Brassard et al. 2017), from 1990-2004 with a URI 81'/114' shrimp trawl from the CCGS *Alfred Needler*, and thereafter a Campelen 1800 survey trawl with Rockhopper footgear on the CCGS *Teleost* (Bourdages et al. 2007). Comparative studies in 1990 allow all Unit 1 survey data to be converted to Teleost-Campelen Units (Bourdages et al. 2007; Brassard et al. 2017).

Between 1997 and 2002, DFO also conducted surveys in Unit 2 with a Campelen 1800 survey trawl towed by the CCGS *Teleost*, with a 12.7 mm liner in the lower 7 m of the codend (Kulka and Atkinson 2016). However, these data are no longer used to assess the redfish stock. The time series for Unit 2 surveys has been replaced by an industry-funded survey that primarily used an Engel 170 trawl with a 30 mm liner in the lower 7 m of the codend, and a 21 m wingspread, towed by large "Cape" class commercial trawlers (45-50 m). This arrangement continued until 2014 when the vessel was switched to the 19 m M/V *Nautical Legend* and the gear was switched to a Campelen trawl with a 15.2 m wingspread. The industry survey began in 1997 and continues to the present (operated by the Groundfish Enterprise Allocation Council or GEAC with input on design from DFO; Kulka and Atkinson 2016). The industry survey is generally biannual although the last few surveys were completed in 2011, 2014 and 2016. Timing of the survey has also changed, as the first survey in 1997 was completed in December with the remainder of the surveys in August/September of each survey year.

Comparative trials between the DFO and GEAC Unit 2 survey gears were completed in August 2000 in order to convert the industry survey data into comparable Teleost-Campelen Units (Cadigan and Power 2010), and again in 2015 following the switch of vessel and gear used by GEAC to a configuration more similar to the DFO survey in Unit 1 (Kulka and Atkinson 2016).

APPENDIX C – DATA MANIPULATIONS

SPLITTING DATA BY SPECIES

For assessment modelling of redfish in Unit 1 and 2 it is necessary to know how to attribute the overall *Sebastes* spp. catch to the two species, *S. fasciatus* and *S. mentella* present in the same and not distinguished in the commercial catch data. Previous work (McAllister and Duplisea 2012) split the aggregated commercial catch data into species from each of the Units by smoothing the mature biomass annual survey catch split proportion and applying this to the commercial data. For the catch at length models fitted in 2015 (Duplisea et al. 2016), it was also necessary to split the commercial catch at length by species. This was done by determining the survey split in each Unit by length and year and applied to the proportions in that Unit, year and length. This was tried both with smoothed composition data and raw composition data (Duplisea 2016).

The commercial catch data are available by total landings and composition of some of the landings. In some cases there are positions and depth associated with catch. None of these data are species split. They overall catch comes from the commercial ZIFF data which usually provides overall landings for redfish from the Unit 1 and 2 area but not necessarily other areas. The ZIFF database is populated on a regional basis with reporting from Quebec, Gulf, Maritimes and and Newfoundland regions. Each region updating the database according regional timelines and procedures. Catch length frequency is much more difficult to obtain for the whole stock. These are collected regionally and data need to be requested regionally. These data are based on port sampler reports which are all managed through individual regional programs.

The main data for splitting catches comes from the Unit 1 and Unit 2 surveys. These surveys have been exploited in various ways to try to split the commercial catch into species with varying levels of dependence. Before using the surveys for these analysis, they were pre-treated to remove the cohorts which were not of Unit 1 and 2 origin and which disappeared from the area (see other reports on removing Grand Banks cohorts). Usually these disappearing cohorts will not affect commercial catch splits since the strong cohorts leave the system before recruitment; however, because smoothing is employed in some of the methods described below, estimation of cohort strength for some years could still be unduly affected.

Method 1: overall survey

The initial approach to catch data species splitting for the present modelling effort has been to follow the Duplisea et al. 2015 precedent. That is, split the overall catch biomass by the mature biomass split from the surveys in the respective Units; split the length frequency in each Unit based on year and length without applying a smoother to the survey proportion of each species at length by Unit and year. The method chosen to smooth the overall data was a quantile gam (R library qgam) with the 0.5 quantile chosen as the smooth.

High and low catch split alternatives

Alternative scenarios for split were developed based on variations of method 1. The baseline scenario for overall catch smooth was through the data median (q=0.5 in qgam). A high scenario was chosen through the 75th percentile (q=0.75) and a low scenario through the 25th percentile (q=0.25).

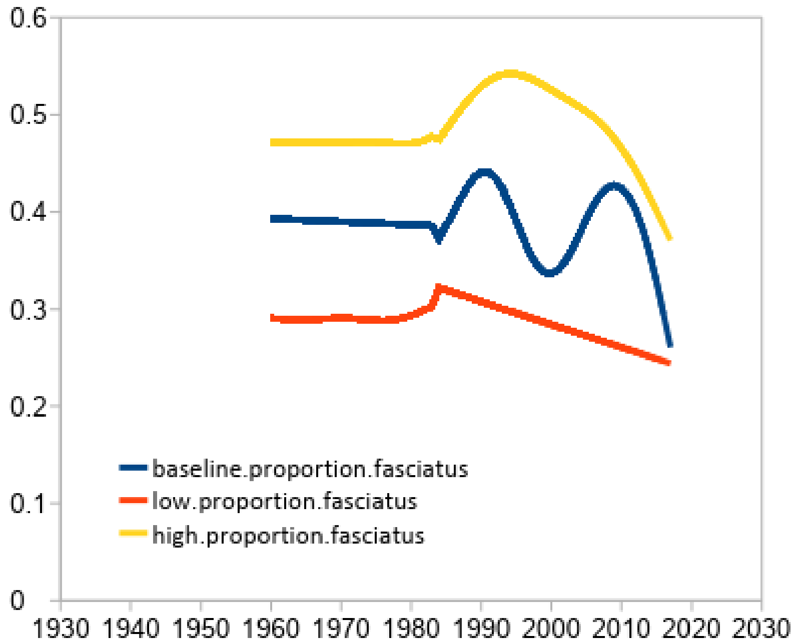


Figure C.1. Unit 1 catch smooth, baseline, high and low scenarios for proportion of S. fasciatus in the catch.

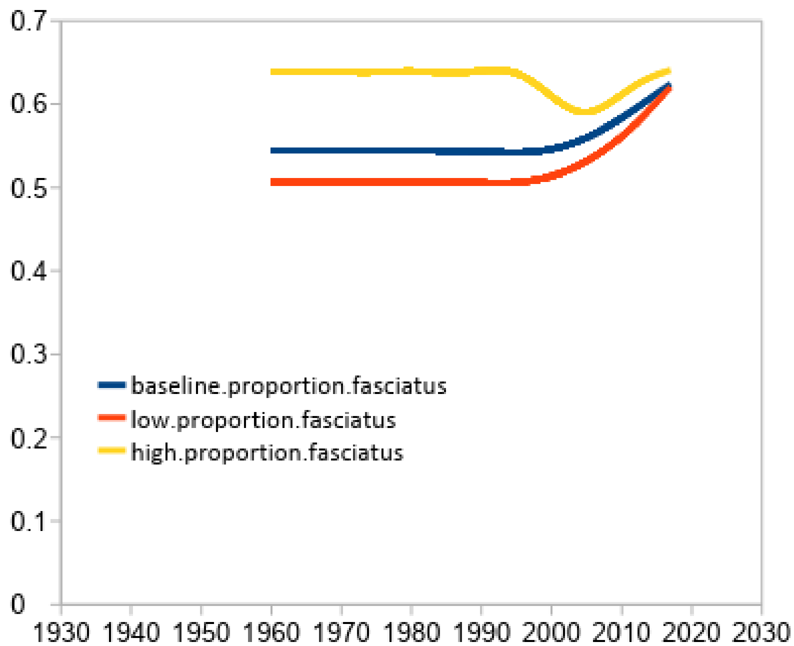


Figure C.2. Unit 2 catch smooth, baseline, high and low scenarios for proportion of S. fasciatus in the catch.

The composition was split in a similar but not identical way. The proportion of *S. fasciatus* at length and by year and Unit was determined from the survey and applied in a ‘raw’ form to the catch composition from that length, Unit and year. The smoother was actually fitted to logit transformed proportions to prevent smooths into negative values and back transformed.

Smoothing was given df=N and therefore data were not smoothed but this was necessary in order to apply smooths to catches that were not caught in the survey, i.e. it was important not to multiply a catch at length observed in the commercial catch by 0 simply because it was not caught in the survey. High and low scenarios were similarly calculated but for proportions halfway between the baseline and 100% for the high scenario and baseline and 0% for the low scenario. This prevented lines from each scenario crossing thus bracketing a range of values that encapsulated the baseline data (Figure C.3).

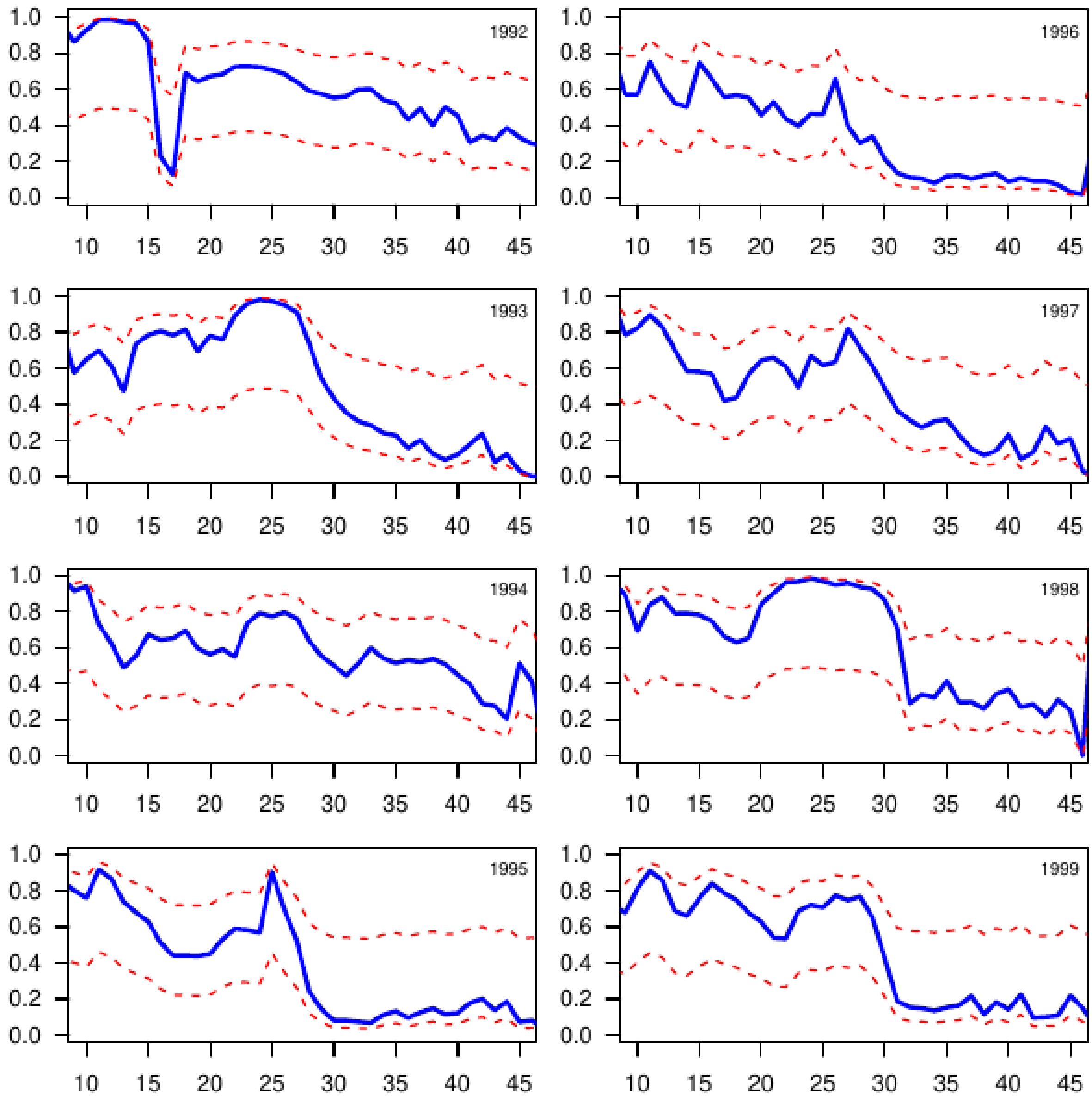


Figure C.3. An example of species split (proportion S. fasciatus) by length and year based on the survey – in this case Unit 1. The blue line is the raw proportion and the red dashed lines represent the high and low scenarios.

In all cases missing years are just filled in with means of the series.

Method 2: depth based splits

The two redfish species as adults have different depth preferences where *S. mentella* tends to prefer deeper water than *S. fasciatus*. This has been described in the survey data by the logistic relationship between proportion of *S. fasciatus* and water depth (m):

proportion.fasciatus= 1/(1+exp(-(L-alpha)/beta))

where alpha= 290 and beta=-54.35

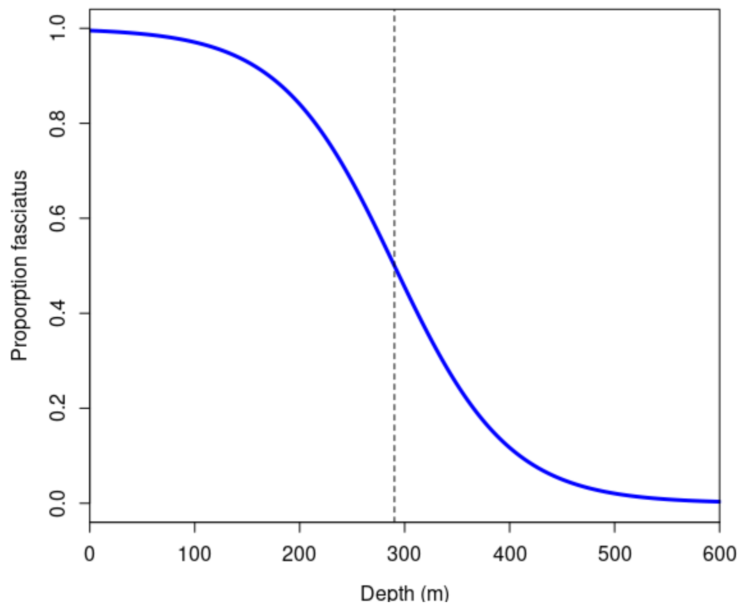


Figure C.4. Proportion of S. fasciatus vs depth fitted logistic relationship from Unit 1 survey data.

This relationship was determined from the survey data from Unit 1 since 1990.

Commercial catch data since 1986 sometimes is associated with a depth and/or a position for the catch. The R library marmap was used with the British Oceanographic Data Centre GEBCO 30 arc second data to determine depths based on position. In most cases the reported depth and depth based on position were not completely different. Some large discrepancies were further investigated and a selection of depths for species splitting were investigated and a choice was made to use either reported depth or depth based on position.

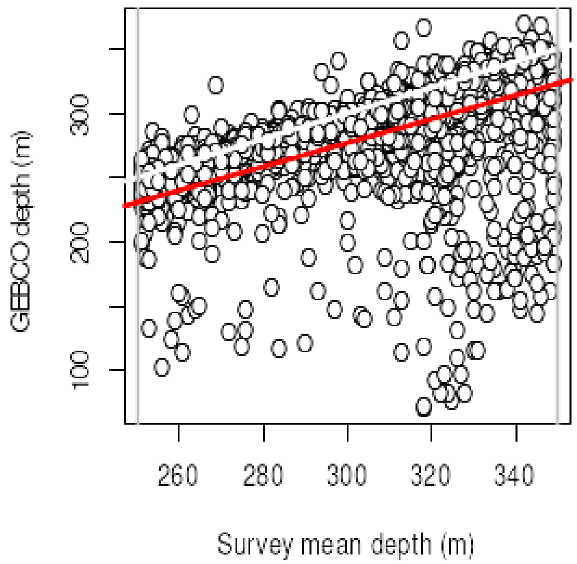


Figure C.5. depth based on position (GEBCO 30 arc second chart) vs reported catch depth over the range 250-350 m where the depth vs species composition relationship shows the greatest change. The red line is the fitted linear model with intercept forced through the origin and the white line is the 1:1 line.

Georeferenced fishing events appear in about half the reported catches. It was assumed that the proportion of each species determined from this subset of catches was the same as the total proportion.

The advantage of this kind of analysis is that it removes the dependence of the catch data on the surveys for splitting the catch. Some of the disadvantages are that there may be differences between the summer and winter vertical distribution of species and in some years <1995 the winter (Jan-May inclusive) catches were often about 50% of the catch.

We were unable to split the length frequency data so far based on this method. The length frequency data are stored in different places and each DFO region keeps that within region. The data with depth were not available to apply this method though we anticipate that it may be possible.

Method 3: survey in commercially-fished areas

A third method possible would be to apply the proportional split from the survey to catches from the same stratum. This method has not been applied yet because of difficulties associating a catch with a stratum and making the assumption that the stratum species split in the summer is the same as the stratum species split throughout the year. It is know that redfish migrate between the summer and winter periods and in fact the definition of the Unit 1 stock area expands to include NAFO zone 3Pn and 4Vn in winter owing to winter movement of fish out of the Gulf (Unit 1) in the winter.

This is a potential method that might be considered in later processes but owing to time constraints, it was not used here.

APPENDIX D – OPERATING MODELS

CORE MODELS

Candidate management procedures (MPs) will be required to perform acceptably under this set of operating models if they are to be considered for implementation. This set of models will need to represent credible alternative hypotheses for how the fishery and stocks have behaved or will behave. If the models are fitted to the same set of data, then they must be required to fit the data reasonably well. However, as explained above, AIC could not be used in this MSE to choose an operating model mainly due to the approach taken to weight different datasets when fitting models to data.

STRESS MODELS

These models are considered to be plausible alternative representations of fishery and stock behaviours but have less scientific credibility than the core and base case models. It is desirable that candidate MPs that perform well under the Core set of models will also perform acceptably well under the Stress test set of models. If the best performing management procedures under the core model set do not perform acceptably well under the stress test models, this is noted but is not sufficient in itself to lead to the rejection of an MP. Should it be discovered in the future that a stress test model becomes more credible than those in the core set and the MP applied had not performed acceptably under the stress test model and in practice, this may give rise to an instance of exceptional circumstances in which a new round of MSE analysis may be performed to identify a new MP which is found to perform acceptably under the updated set of core models.

SENSITIVITY ANALYSES

These models represent plausible (more accurately, not implausible) alternative representations of fishery and stock dynamics. However, if the statistical fitting of these models indicates that estimated parameters are similar to one of the core or stress models already formulated, then the candidate model will not be applied in simulation testing the candidate management procedures simply because they are effectively redundant. However, if the parameter estimates and apparent model dynamics are different from all of those models already in the core and stress test set of models, then this new model will be applied as another stress test type of model.

Table D.1. Core, sensitivity and stress-test Operating Models for the Units 1 and 2 redfish MSE process, following discussions at the December 12 and 14, 2017 Technical Meetings and the December 13, 2017 Working Group meeting in Halifax, NS.

Model	Type	Description	Details
1	Core	Base Case	Model assumes fishery selectivity is logistic, and change in selectivity over time (and offset) is described by two time blocks over the time series (early years to 1993, and 1994 to present). The catch killed:retained ratio is 1.2 from early years to 1985, 2 from 1986-1993, and 1.1 from 1994 to present. Simulated recruitment will be produced similar to what has been seen in the past, using a nonparametric bootstrap of recruitment events from the historical time series.
2	Core	Reduced future recruitment	Simulations will assume there will be no strong cohorts for the next 20 years.
3	Core	Alternative M	This model will use a Lorenzen M function (where natural mortality, M, varies with fish size and is higher for smaller fish) instead of a single value.
4	Core	Alternate recruitment simulation method	Use a parametric bootstrap of historical recruitment for simulations, with the variance and autocorrelation coefficient estimated for recruitment events in the historical time series since 1970.
5	Core	Alternative catch split	Historical catch splits differ from base case – assume more *S. mentella.*
6	Core	Alternative catch split	Historical catch splits differ from base case – assume more *S. fasciatus.*
7	Stress	Alternative fishery selectivity	Assume fisheries selectivity is dome-shaped or double-logistic for both species (and that selectivity decreases for both large and small fish).

Model	Type	Description	Details
8	Stress	High future M	Future M is doubled for both species, for the next 20 years only. This is a way to examine density dependence during periods of time with strong cohorts.
9	Stress	Reduced future growth	Simulate a reduction in future growth of both species for the next 20 years by reducing the asymptotic length (Linf) to a value 2/3 as large as in the base case, while assuming same value for K (Linf, K are parameters in the von Bertalanffy growth equation). This is a way to examine density dependence during periods of time with strong cohorts.
10	Stress	Reduced future recruitment	Simulations will assume no strong cohorts for 40 years.
11	Sensitivity	Alternative M	Reduce historical and future M by factor of 0.75 in both species.
12	Sensitivity	Alternative M	Increase historical and future M by a factor of 1.25 in both species.
13	Sensitivity	Alternative steepness	Assume the steepness of the stock-recruitment relationship is higher than in base case by factor of 1.25, provided this is not already realized in the above scenarios. This will be done provided that it has not already been realized in other identified models.
14	Sensitivity	Alternative steepness	Assume the steepness of the stock-recruitment relationship is lower than in base case by factor of 0.75, provided this is not already realized in the above scenarios. This will be done provided that it has not already been realized in other identified models.

Model	Type	Description	Details
15	Sensitivity	Alternative fishery selectivity	Use three time blocks to represent different periods of fisheries selectivity (early years to 1984, 1985-1993, and 1994 to present), and not two.
16	Sensitivity	Alternative for offset	Use a different value for offset in median age of fish killed and the median age of fish retained.
17	Sensitivity	Alternative for catch killed: retained ratio	Use a different set of assumptions for the values and/or time periods for the ratio of catch killed to catch retained than the base case.
18	Sensitivity	Alternative prior mean	Assume lower prior mean for strong historical cohorts (i.e., how strong these cohorts are assumed to have been – a lower prior means these strong cohorts will be smaller in this model than in the base case). Model will be done only if key parameters are estimated to be different.
19	Sensitivity	Alternative prior mean	Assume higher prior mean for strong historical cohorts (i.e., how strong these cohorts are assumed to have been – a higher prior means these strong cohorts will be larger in this model than in the base case). Model will be done only if key parameters are estimated to be different.
22	Stress test	Alternative vulnerability blocks	Vulnerability in 2017-2021 reverts back to that estimated for years up to 1994 and then returns to that estimated for 1995-2016. The earlier vulnerability favoured smaller fish and with the large 2011-2013 cohorts recruiting to the fishery 2018-2020, it is speculated that it will be difficult to avoid catching small redfish in the next few years.

Model	Type	Description	Details
23	Stress test	High discarding rates 2018-2020 under OM1	Using the base case operating model 1, assume the ratio of catch biomass killed to catch biomass retained in 2018-2020 is 2 and then returns to 1.1 for 2021-2057.
24	Stress test	High discarding rates 2018-2020 under OM3	Using the stress test operating model 3 (natural mortality rate doubles for next 20 years), assume the ratio of catch biomass killed to catch biomass retained in 2018-2020 is 2 and then returns to 1.1 for 2021-2057.

A list of the parameters, parameter values, variables and descriptions of the parameters and variables used in the base case operating model for Units 1 and 2 redfish is provided in Table D.2. A list of the equations applied in the operating models for these two stocks is provided in Table D.3. A list and description of the harvest control rule parameters are provided in Table D.4. Mathematical specifications for the model free harvest control rule are provided in Table D.5. Mathematical specifications for the population dynamics model components are provided in Table D.6. A list of the parameter estimated, residuals, and negative log-likelihood function (NLL) for fitting the operating model is provided in Table D.7. A list of the values inputted into the prior distribution functions and likelihood components in Table D.7 is provided in Table D.8.

Table D.2. Base case operating model for Sebastes Mentella (Sm) and Sebastes fasciatus (Sf).

Symbol	**Value** (*Sm; Sf*)	**Description**
Indices		
x	1; 2	Index for species: 1: *S. mentella 2: S. fasciatus*
a	$1,...,A$; $1,...,A$	Age-year class
t	$1951,\ldots,T$; $1951,\ldots,T$	Annual time step
l	$1,..,L$; $1,...,L$	Length classes (cm)

Symbol	Value (*Sm; Sf*)	Description
g	1,2,3,4; 1,2,3,4	Index for gear type *g*: 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)
p	-	Index for Management Procedures (MPs)
Model settings		
A	57; 57	Plus-group
T	2017; 2017	Last year before the management procedure starts (MP)(i.e., 2018)
T_2	2053; 2053	Year when the MP ends
L	57; 57	Number of size classes
blk_1	1951,…,1993;1951,…,1993	Vulnerability time-block 1
blk_2	1994,…, 2017;1994,…2017	Vulnerability time-block 2
Parameters		
L_∞	45.82[1]; 41.24*	Mean asymptotic length (cm)
k	0.096[1]; 0.106*	Growth parameter (cm/yr)
t_0	-0.5; -0.5	Theoretical age at length zero (yr)

Symbol	Value (*Sm; Sf*)	Description
a^w	0.00762; 0.00762	Scaling constant for weight-at-length (cm*gr)[*]
b^w	3.193; 3.193	Allometric factor[*]
M_a	0.10; 0.125	Instantaneous natural mortality at age (yr^{-1})[2]
$\hat{R}_0$	0.420; 0.623	Mean unfished recruitment (age-1) x [1e9]
$\hat{k}$	3.082; 6.522	Compensation ratio
h	0.435; 0.620	Steepness
$\hat{\omega}'_{a=3};\hat{\omega}''_t$	1952,…,T;1952,…T	Estimated deviates for age 3 in the initial abundance and recruitment deviates in year *t*
ω'''_t	$t>T,\ldots,T_2;t>T,\ldots,T_2$	Simulated recruitment deviates in year *t*
σ_R	1.0; 1.0	Standard deviation of the recruitment variation
Ω_1	7.889; 7.256	Age-at-50% maturity[3]
Ω_2	1.98; 1.58	Age-at-maturity slope[3]
$\hat{a}^{50}_{g=1,2}$	2.022, 7.017; 1.676, 5.224	Age-at-50% vulnerability for survey index *g*
$\hat{a}^{sd}_{g=1,2}$	0.145, 2.731; 0.201, 0.609	Age-at-vulnerability slope for survey index *g*
$\hat{a}^{50_{ret}}_{t\in blk_1,g=4}$	9.245; 8.545	Age-at-50% retained for index *g* and time block 1
$\hat{a}^{sd_{ret}}_{t\in blk_1,g=4}$	0.866; 0.471	Age-at-retained slope for index *g* and time block 1
$\hat{a}^{50_{ret}}_{t\in blk_2,g=4}$	8.184; 8.329	Age-at-50% retained for index *g* and time block 2

Symbol	**Value** (*Sm; Sf*)	**Description**
$\hat{a}^{sd_{ret}}_{t\in blk_2,\mathrm{g}=4}$	0.735; 0.534	Age-at-retained slope for index *g* and time block 2
$a^{sd}_{t\in blk_1,\mathrm{g}=4}$	$a^{sd_{ret}}_{t\in blk_1,\mathrm{g}=4}$	Age-at-vulnerability slope for index *g* and time block 1
$a^{sd}_{t\in blk_2,\mathrm{g}=4}$	$a^{sd_{ret}}_{t\in blk_2,\mathrm{g}=4}$	Age-at-vulnerability slope for index *g* and time block 2
$a^{50_{ls}}_{\mathrm{g}=1,2,3}$	19.0; 18.0	Age-at-50% vulnerability/retention for the left side of the double logistic
$a^{sd_{ls}}_{\mathrm{g}=1,2,3}$	1.0; 1.0	Age-at-vulnerability/retention slope for the left side of the double logistic vulnerability
$a^{vul\_ret}_{\min}$	7; 7	Minimum age-at-retention/vulnerability for projections
$a^{mat}_{\min}$	5; 4	Minimum maturity-at-age
u_{init}	0; 0	Initial exploitation rate for *t*=1951
U_{max}	0.95; 0.95	Maximum exploitation rate
$\hat{\tau}_{\mathrm{g}=1,2,3}$	0.469, 0.452, 0.407; 0.501, 0.382, 0.519; 0.501, 0.382, 0.516	Standard deviation for survey index *g*
$\hat{q}_{\mathrm{g}=1,2,3}$	0.666, 1.745, 0.638; 0.533, 2.495, 0.444; 0.533, 2.495, 0.444	Catchability coefficient for survey index *g*
$n_{\mathrm{g}=1,2,3,4}$	34, 8, 34, 33; 34, 8, 34, 33	Number of years with index data
Derived variables		
$\emptyset_0$	-	Unfished equilibrium spawning biomass per recruit

Symbol	Value (*Sm; Sf*)	Description
l_a		Survivorship-at-age per recruit
$\bar{L}_a$		Mean length-at-age (cm)
S_a		Natural survival-at-age (yr)
W_a		Weight-at-age (gr)
W_l		Weight-at-length (gr)
m_a		Proportion mature-at-age
SSB_0		Unfished spawning biomass (kt)
$a^{50}_{t,\mathrm{g}=4}$		Age-at-50% vulnerability for fishery in year *t*
$v_{a,g=4}$		Vulnerability-at-age, for index *g*
$v'_{a,t,\mathrm{g}=4}$		Vulnerability-at-age, in year *t* for the fishery (domed-shaped)
$v''_{a,\mathrm{g}=1,2}$		Vulnerability-at-age for the survey (domed-shaped)
$\Phi_x^{P(l\|a)}$	$f(\bar{L}_a, CV_a^L)$	Transition matrix for species *x*
C_t^k		Catch biomass killed in year *t* (kt)
$f_{a,t}^{ret}$		Fraction retained-at-age, *a*, in year *t* for the fishery
$u_{x,t}$		Harvest rate in year *t* for species *x*
$N_{x,l,t}$		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)

Symbol	Value (*Sm; Sf*)	Description
Observations		
$I_{x,t,\mathrm{g}=1,2,3}$		Observed survey biomass index *g* in year *t* (kt) for species *x*
$I'_{x,t,\mathrm{g}=3}$		Simulated survey fishery index (length -based) in year *t* (kt) for species *x*
C_t^{ret}		Catch biomass retained in year *t* for the fishery (kt)
$D_t^{kill_{ratio}}$	No species-specific	Catch biomass killed to catch biomass retained ratio in year *t*
a_t^{offset}	No species-specific	Offset in year *t* (ages)
$N_{l,t,\mathrm{g}=1,2}^{survey\_obs}$		Observed numbers-at-length for the survey in Units 1 and 2
$N_{l,t,\mathrm{g}=4}^{ret\_obs}$		Observed numbers-at-length (retained) for the fishery in Units 1 and 2
$ESS_{\mathrm{g}=1,2,4}$	25, 10, 5; 25, 10, 5;	Sample size for index *g*
$CV_{\mathrm{g}=1,2}$	0.25, 0.25; 0.25, 0.25	Coefficient of variation for survey index *g*
CV_a^L	$0.12_{a=1,}\ldots,0.05_A; 0.12_{a=1,},\ldots,0.05_A$	Coefficient of variation for length-at-age (linear increase)

[1] Saborido-Rey et al. (2004); *This study

[2] McAllister and Duplisea (2011); [3] Gascon (2003)

Table D.3. Equations for the operating model for Sebastes mentella (Sm) and S. fasciatus (Sf).

Life history schedules	
Natural survival-at-age	$S_a = e^{(-M_a)}$
Survivorship per recruit	$l_a = \begin{cases} 1 & a = 1 \\ l_a = l_{a-1}s_{a-1}(1 - v_{a-1}u_{init}) & 1 < a < A \\ l_A = l_{A-1}s_{A-1}(1 - v_{A-1}u_{init})/[1 - s_A(1 - v_A u_{init}] & a = A \end{cases}$
Mean length-at-age	$\bar{L}_a = L_\infty\left[1 - e^{(-k(a-t_0))}\right]$
Proportion mature-at-age	$m_a = \begin{cases} 0 & if\ a \le a_{\min}^{mat} \\ \dfrac{1}{1 + e^{-(a-\Omega_1)/\Omega_2}} & otherwise \end{cases}$
Weight-at-age	$w_a = a^w \bar{L}_a^{b^w}$
Vulnerability-at-age	$v_{a,t,\mathrm{g}} = \begin{cases} 0 & if\ a \le a_{\min}^{vul\_ret} \\ \dfrac{1}{1 + e^{-\left(a_{t,\mathrm{g}}^{50} - a\right)/a_{t,\mathrm{g}}^{sd}}}\ t \in blk_1, blk_2; \mathrm{g} = 4 & otherwise \end{cases}$
Stock-recruitment relationship	
Spawning biomass per recruit	$\emptyset_0 = \sum_{a=1}^{a=A} l_a m_a w_a$
Unfished spawning biomass	$SSB_0 = \hat{R}_0 \emptyset_0$
Beverton-Holt recruitment parameters	$\alpha = \frac{\hat{k}}{\emptyset_0}$ $k > 1$ $\beta = \frac{k-1}{SSB_0}$
Population dynamics	
Initial condition	$N_{a,1} = R_0 l_a$ $a \notin 3$ $N_{a,1} = R_0 l_a e^{\hat{\omega}'_a \sigma_{Rpr} - 0.5(\sigma_{Rpr})^2}$ $\omega'_a \sim \mathrm{N}(0,1)$ $a = 3$
Recruitment deviates	$\gamma_t = \begin{cases} \hat{\omega}''_t & t < T \\ \omega'''_t & otherwise \end{cases}$ $\omega'''_t \sim bootstrap(\hat{\omega}''_t, OM_n)$ see OMs section
Recruitment (*t* > 1; *a* = 1)	$N_{1,t} = \dfrac{\alpha SSB_{t-1}}{1 + \beta SSB_{t-1}} e^{\gamma_t \sigma_R - 0.5(\sigma_R)^2}$

Life history schedules	
Abundance dynamics ($t > 1$)	$N_{a,t} = \begin{cases} N_{a-1,t-1}S_{a-1}(1 - v_{a-1,t}u_{t-1}) & 1 < a < A \\ N_{A-1,t-1}S_{A-1}(1 - v_{A-1,t}u_{t-1}) & \\ +N_{A,t-1}S_A\left(1 - v_{A,t}u_{t-1}\right) & a = A \end{cases}$
Spawning biomass	$SSB_t = \sum_{a=1}^{A} N_{a,t}m_a w_a$
Vulnerable biomass killed	$VB_t^k = \sum_{a=1}^{A} N_{a,t}\, v_{a,t,\mathrm{g}}w_a \quad \mathrm{g} = 4$ $VB_t^k = \sum_{a=1}^{A} N_{a,t}\, v_{a,t,\mathrm{g}}w_a \quad t > T \quad v_{a,t,\mathrm{g}} \in blk_2$
Vulnerable biomass retained	$VB_t^{ret} = \sum_{a=1}^{A} N_{a,t}\, v_{a,t,\mathrm{g}}f_{a,t}^{ret}w_a \quad \mathrm{g} = 4$ $VB_t^{ret} = \sum_{a=1}^{A} N_{a,t}\, v_{a,t,\mathrm{g}}f_{a,t}^{ret}w_a \quad t > T \quad v_{a,t,\mathrm{g}}; f_{a,t}^{ret} \in blk_2$
Catch biomass killed	$C_t^k = u_t VB_t^k$
Catch biomass retained	$C_t^{ret} = u_t VB_t^{ret}$
Fishery index (length-based)	
Number-at-length	$N_{x,l,t} = \underset{1\times A}{N_{x,a,t}}\underset{A\times L}{\mathbf{\Phi}_x^{P(l\|a)}}$
Fishery index	$I'_{x,t,\mathrm{g}} = \hat{q}_{x,\mathrm{g}} \sum_{l\geq 30cm}^{L} N_{x,l,t}\, W_{x,l}e^{\varepsilon_t^x \hat{\tau}_{\mathrm{g}} - 0.5\hat{\tau}_{\mathrm{g}}^2} \quad \varepsilon_t^x \sim N(0,1) \;\; \mathrm{g} = 3, x = 1$ $I'_{x,t,\mathrm{g}} = \hat{q}_{x,\mathrm{g}} \sum_{l\geq 29cm}^{L} N_{x,l,t}\, W_{x,l}e^{\varepsilon_t^x \hat{\tau}_{\mathrm{g}} - 0.5\hat{\tau}_{\mathrm{g}}^2} \quad \varepsilon_t^x \sim N(0,1) \;\; \mathrm{g} = 3, x = 2$

Table D.4. Harvest control rule (HCR) parameters for Sebastes mentella (Sm) and S. fasciatus (Sf).

Symbol	**Value** *Sm; Sf*	**Description**
Ω_x^a	4; 4	Tuning parameter for HCR: intercept (kt)
Ω_x^b	2; 2	Tuning parameter for HCR: slope (kt/year)
b	No species-specific	Slope multiplier
Ω_x^c	1.2; 1.2	Penalty (kt/year) for HCR
J_{0x}	1.5; 1.5	Threshold parameter when $J_{x,t,\mathrm{g}}$ decreases at low biomass levels in the HCR
$J_{x,t,\mathrm{g}}$	-	Relative fishery index for the species *x*
$I_{x,t,\mathrm{g}=3}$	-	Historical fishery index for Unit 1 for species *x*; fish > 30 and 29 cm for *Sm* and *Sf*, respectively)
$CL_{t,p}^{cap}$	No species-specific	Catch biomass caps (kt) in time *t* for the MP=*p*.
$CL_{x,t}$	-	Raw catch limit (kt) derived from the HCR for species *x*
CL_t^c	-	Catch limit (kt) combined for *Sm* and *Sf.*
$CL_{x,t}^*$	-	Catch biomass (kt) after catch split implementation for species *x*
$CL_{x,t}^{kill_{ratio}}$	-	Catch biomass (kt) after implementation error for species *x*
π_x	See Table A5.B6	Proportion for catch split implementation for species *x*
$D_{prj}^{kill_{ratio}}$	1.1; 1.1	fish killed:retained ratio for the projections

Table D.5. Description for the model free (MF) harvest control rule (HCR) for Sebastes mentella (Sm) and S. faciatus (Sf).

	$t > T; g = 3$
Raw CL	$CL_{x,t} = \left[\Omega_x^a + b\Omega_x^b (J_{x,t,\mathrm{g}} - J_{0x})\right] - pen_x$
Trailing average for fishery index	$J_{x,t,\mathrm{g}} = \exp\left[\frac{1}{3}\sum_{t-2}^{t} \ln\left(I'_{x,t,\mathrm{g}}\right)\right] \Big/ \exp\left[\frac{1}{34}\sum_{t=1984}^{t=2017} \ln\left(I_{x,t,\mathrm{g}}\right)\right]$
Penalty	$pen_x = \begin{cases} 0 & if\ \ J_{x,t,\mathrm{g}} < J_{0x} \\ \Omega_x^c (J_{x,t,\mathrm{g}} - J_{0x})^2 & otherwise \end{cases}$
Combined CL	$CL_t^c = \sum_{x=1}^{2} CL_{x,t}$
Cap implementation	$CL_t^c = \begin{cases} CL_t^c & if\ \ CL_t^c \leq CL_{t,p}^{cap} \\ CL_{t,p}^{cap} & otherwise \end{cases}$
Catch split implementation	$\pi_{x\in 1,t} \sim Unif(0.39, 0.44)$ $\pi_{x\in 2,t} = 1 - \pi_{x\in 2,t}$ $CL_{x,t}^* = CL_t^c \pi_{x,t}$
Implementation error	$CL_{x,t}^{kill_{ratio}} = CL_{x,t}^* D_{prj}^{kill_{ratio}}$
Output control	$u_{x,t} = min\left(\frac{CL_{x,t}^{kill_{ratio}}}{VB_{x,t}^k}, U_{\max}\right)$
Small fish protocol	$u_{x,t} = \begin{cases} 0 & if\ small\ fish\ protocol \\ u_{x,t} & otherwise \end{cases}$

Table D.6. Description for the population dynamics model (OM1) for Sebastes mentella (Sm) and S. faciatus (Sf).

Parameters estimated	$\widehat{\Theta} = \begin{pmatrix} R_0, k, \omega'_{a=3}, \{\omega''_t\}_{t=1952}^{t=2017}, \{q_g, \tau_g\}_{g=1,2}, \{a_g^{50}, a_g^{sd}\}_{g=1,2} \\ \{a_{blk_1}^{50_{ret}}, a_{blk_1}^{sd_{ret}}, a_{blk_2}^{50_{ret}}, a_{blk_2}^{sd_{ret}}\}_{g=4} \end{pmatrix}$
Population dynamics	
Initial abundance $(t = 1951; u_{init} = 0)$	$N_{a,t} = \begin{cases} R_0 l_a e^{\widehat{\omega}'_a \sigma_{Rpr} - 0.5(\sigma_{Rpr})^2} & a \in 3 \\ R_0 l_a & a \notin 3 \end{cases}$
Recruitment	$N_{1,t} = \begin{cases} \dfrac{\alpha SSB_{t-1}}{1+\beta SSB_{t-1}} e^{\omega''_t \sigma_{Rpr} - 0.5(\sigma_{Rpr})^2} & if\ 1951, 1953 \\ \dfrac{\alpha SSB_{t-1}}{1+\beta SSB_{t-1}} e^{\omega''_t \sigma_R - 0.5(\sigma_R)^2} & otherwise \end{cases}$
Fraction retained for the fishery (*g* = 4)	$f_{a,t}^{ret} = \begin{cases} 0 & if\ t \in blk_1\ a \leq 5; t \in blk_2\ a \leq 7 \\ \dfrac{1}{1+e^{-(a-a_{blk1,g}^{50_{ret}})/\Omega_{blk1,g}^{sd_{ret}}}} & otherwise \\ \dfrac{1}{1+e^{-(a-a_{blk2}^{50_{ret}})/\Omega_{blk2,g}^{sd_{ret}}}} & \end{cases}$
Vulnerability for the fishery (*g* = 4)	$a_{t,g}^{50} = \begin{cases} a_{t,g}^{50_{ret}} - a_t^{offset} & t \in blk_1 \\ a_{t,g}^{50_{ret}} - a_t^{offset} & t \in blk_2 \end{cases}$ $a_{t,g}^{sd} = \begin{cases} a_{t,g}^{sd_{ret}} & t \in blk_1 \\ a_{t,g}^{sd_{ret}} & t \in blk_2 \end{cases}$
Logistic vulnerability for fishery (*g* =4)	$v_{a,t,g} = \begin{cases} 0 & if\ t \in blk_1\ a \leq 5; t \in blk_2\ a \leq 7 \\ \dfrac{1}{1+e^{-(a_{t,g}^{50}-a)/a_{t,g}^{sd}}} & g = 4 \end{cases}$
Vulnerability for the survey (total biomass)	$v_{a,g} = \begin{cases} 0 & a = 1 \\ \dfrac{1}{1+e^{-(a_g^{50}-a)/a_g^{sd}}} & otherwise \end{cases}$
Option for double logistic fishery (retained)	$v'_{a,t,g} = \dfrac{1}{1+e^{-(a-a_{t,g}^{50})/a_{t,g}^{sd}}} \left(1 - \dfrac{1}{1+e^{-(a-a_g^{50_{ls}})/a_g^{sd_{ls}}}}\right) \quad g = 4$ $v'_{a,t,g} = \dfrac{v'_{a,t,g}}{\max_a(v'_{a,t,g})}$

Option for double logistic vulnerability for survey	$v''_{a,g} = \frac{1}{1+e^{-(a-a_g^{50})/a_g^{sd}}}\left(1-\frac{1}{1+e^{-(a-a_g^{50_{ls}})/a_g^{sd_{ls}}}}\right)$ g = 1,2 $v''_{a,g} = \frac{v''_{a,g}}{\max_a(v''_{a,g})}$
Harvest rates	$u_t = \log^{\lim} U_{\max} + (1-\log^{\lim})\, C_t^k / VB_t^k$
Catch killed	$C_t^k = C_t^{ret} D_t^{kill_{ratio}}$
Vulnerable biomass killed	$VB_{t,g}^k = \sum_{a=1}^{A} N_{a,t}\, v_{a,t,g} w_a$ g = 4
Predicted vulnerable biomass for the survey	$VB_{t,g} = \sum_{a=1}^{A} N_{a,t}\, v_{a,g} w_a$ g = 1,2
Predicted vulnerable biomass for the fishery index	$VB_{t,g} = \sum_{a=10}^{A} N_{a,t}\, w_a$ g = 3
Catch-at-age (killed) for the fishery	$C_{a,t,g}^k = N_{a,t} v_{a,t,g} u_t$ g = 4
Predicted vulnerable numbers-at-length for the survey	$N_{l,t,g} = N_{a,t} v_{a,g} \mathbf{\Phi}^{P(l\|a)}$ g = 1, 2
Predicted proportions-at-length for the survey	$P_{l,t,g} = N_{l,t,g} / \sum_{1>5cm}^{L} N_{l,t,g}$ g = 1, 2
Observed proportions-at-length for the survey	$O_{l,t,g} = N_{l,t,g}^{survey\_obs} / \sum_{1\geq 5cm}^{L} N_{l,t,g}^{survey\_obs}$ g = 1, 2
Predicted numbers-at-length (retained) for the fishery	$N_{l,t,g}^{ret} = C_{a,t,g}^k f_{a,t}^{ret} \mathbf{\Phi}^{P(l\|a)}$ g = 4
Predicted proportions-at-length (retained) for the fishery	$P_{l,t,g}^{ret} = N_{l,t,g}^{ret} / \sum_{1\geq 18cm}^{L} N_{l,t,g}^{ret}$ g = 4
Observed proportions-at-length (retained) for the fishery	$O_{l,t,g}^{ret} = N_{l,t,g}^{ret\_obs} / \sum_{1\geq 18cm}^{L} N_{l,t,g}^{ret\_obs}$ g = 4

Table D.7. Residuals and negative log-likelihood function (NLL) for fitting the operating model for Sebastes mentella (Sm) and S. faciatus (Sf).

Description	Equation
Conditional likelihood estimates	$\mathrm{g} = 1, 2, 3$ $\eta_{t,\mathrm{g}} = \ln\left(I_{t,\mathrm{g}}\right) - \ln(VB_{t,\mathrm{g}})$ $\overline{\ln q_{\mathrm{g}}} = {}^{1}\!/_{n_g} \sum_{t\in \mathrm{g}} \eta_{t,\mathrm{g}}$ $q_{\mathrm{g}} = e^{\overline{\ln q_{\mathrm{g}}}}$ $\tau_{\mathrm{g}}^2 = \frac{1}{(n_{\mathrm{g}} - 2)} \sum_{t\in \mathrm{g}} \eta_{t,\mathrm{g}}^2$
Priors	
-	$h = \frac{k}{4 + k}$ $P_h = \frac{(h - \bar{h})^2}{2(\sigma_h)^2}$
-	$P_q = \sum_{\mathrm{g}} \frac{(q_{\mathrm{g}} - \bar{q}_{\mathrm{g}})^2}{2(\sigma_{q\mathrm{g}})^2}$ $\quad\quad \mathrm{g} = 1,2$
-	$P_{\omega'} = \frac{(\omega'_{a=3} - \bar{\omega}')^2}{2(\sigma_{Rpr})^2}$
-	$P_{\omega''} \begin{cases} \frac{\omega''_{1951} - \ln(\bar{\omega}''_{1951})^2}{2(\sigma_{Rpr})^2} \\ \frac{\omega''_{1953} - \ln(\bar{\omega}''_{1953})^2}{2(\sigma_{Rpr})^2} \\ \sum_{\substack{t\in\, 1952, \\ 1953:2017}} \frac{\omega''_t - \ln(\bar{\omega}'')^2}{2(\sigma_{Rpr})^2} \end{cases}$
Likelihood components	$L_\eta = \sum_{g\in 1,2} \sum_{t} \frac{(\eta_{t,\mathrm{g}})^2}{2(CV_{\mathrm{g}})^2}$ $L_{\ell'} = -\sum_{\mathrm{g}\in 1,2} \sum_{t} \sum_{l} (ess_{\mathrm{g}}\, 0_{l,t,\mathrm{g}} \ln p_{l,t,\mathrm{g}})$ $L_{\ell''} = -\sum_{\mathrm{g}\in 4} \sum_{t} \sum_{l} (ess_{\mathrm{g}}\, 0_{l,t,\mathrm{g}}^{ret} \ln p_{l,t,\mathrm{g}}^{ret})$

Description	Equation
NLL	$nll(\Theta) = L_\eta + L_{\ell'} + L_{\ell''} + P_h + P_q + P_{\omega'} + P_{\omega''}$

Table D.8. Parameter specifications for prior distributions in Table D.7 applied in fitting the operating models for redfish Sebastes mentella (Sm) and S. fasciatus (Sf).

Symbol	**Value** (*Sm;Sf*)	**Description**
$\overline{\omega}''$	2.1; 2.1	Mean prior for recruitment residuals
$\overline{\omega}'$	2.1; 2.1	Mean prior for residuals in initial abundance
σ_{Rpr}	1.0; 1.0	Standard deviation prior for recruitment residual and initial abundance
$\overline{q}_{\mathrm{g}=1,2}$	0.2; 0.2	Mean prior for catchability for survey index *g*
$\sigma_{q_{\mathrm{g}=1,2}}$	1.0; 1.0	Standard deviation prior catchability for survey index *g*
$\overline{h}$	0.67; 0.67	Mean prior for steepness
σ_h	0.17; 0.17	Standard deviation prior steepness

HISTORICAL FISHERY PERIODS AND FISH SIZE SELECTIVITY

In order to define periods of similar redfish historical fishery selectivity, commercial landings data across gear, vessel size, month and NAFO zones were collected for Unit 1 and 2 between 1985 and 2015. First, a principal component analysis was conducted on each of the four matrices describing fishery characteristics (gear, vessel, month, and zone). The two first principal components (PC), that explained between 89 and 99% of the variation in the matrices were extracted and retained. Thus each matrix represented the same number of variables because cluster analyses are sensitive to the number of variables included (Legendre and Legendre 2012). Therefore, the resulting 8 principal components were used into a k-mean clustering. This method iteratively creates groups aiming to partition n observations into k clusters in which each observation belongs to the cluster with the nearest mean. The SSI criterion was minimized and the clustering that seemed the most appropriate was 2 or 3 periods. The most important split was between 1993 and 1994, and a second was between 1989 and 1990. The characteristics that are mostly contrasting each period are that between 1985 and 1989, bottom trawl was the gear that was the most utilized, whereas it was midwater trawl between 1990 and 1993. From 1994 until recent years, the moratorium in Unit 1 defined the last historical period, where landings decreased drastically. This analysis was conducted with *Sebastes* spp. and different ways to split the total commercial landings by species. In all cases, similar results were obtained.

Figures D.1 and D.2 show the results of the cluster analysis with two and three groups, respectively. The boxplots illustrate the variance in the principal components representing fishery characteristics, namely landings across gear, vessel, month, and zone for each period. The minimum, lower quartile, median, upper quartile, maximum value and outliers are indicated.

When boxplots are not overlapping, it is indicative that this characteristic is different between periods. Principal components are composite variable compressing the variability present in the data, meaning that, for instance, a high value of the “sizePC” does not necessarily correspond to a large vessel size. Looking at the boxplots we can see that all periods diverge from each other in all landings characteristics: gear, size, month, and zone.

For each time period, fish size selectivity was estimated. To do so, length-frequency measured by observers at sea during 7489 fishing events in Units 1 and 2 between 1978 and 2017 were gathered. Five quantiles (Q10, Q25, Q50, Q75, and Q90) were computed for each fishing event to represent the distribution of captured fish length frequency. The values were averaged for each gear, in each Unit, for every year. The proportions of landings for each gear, in each Unit, for every year were also computed. These proportions were multiplied by their respective quantile (specific to gear, Unit and year) to weight their importance based on their contribution to the fishery. Therefore, if, for instance, most landings were harvested in Unit 2 for a specific year, the quantiles estimated in Unit 1 would contribute less to the results. Finally, the weighted quantiles were averaged for the duration of the different periods determined previously by the k-mean analysis (Table D.9). Once again, the time period selectivity was assessed with *Sebastes* spp, and different ways to split the total landings by species.

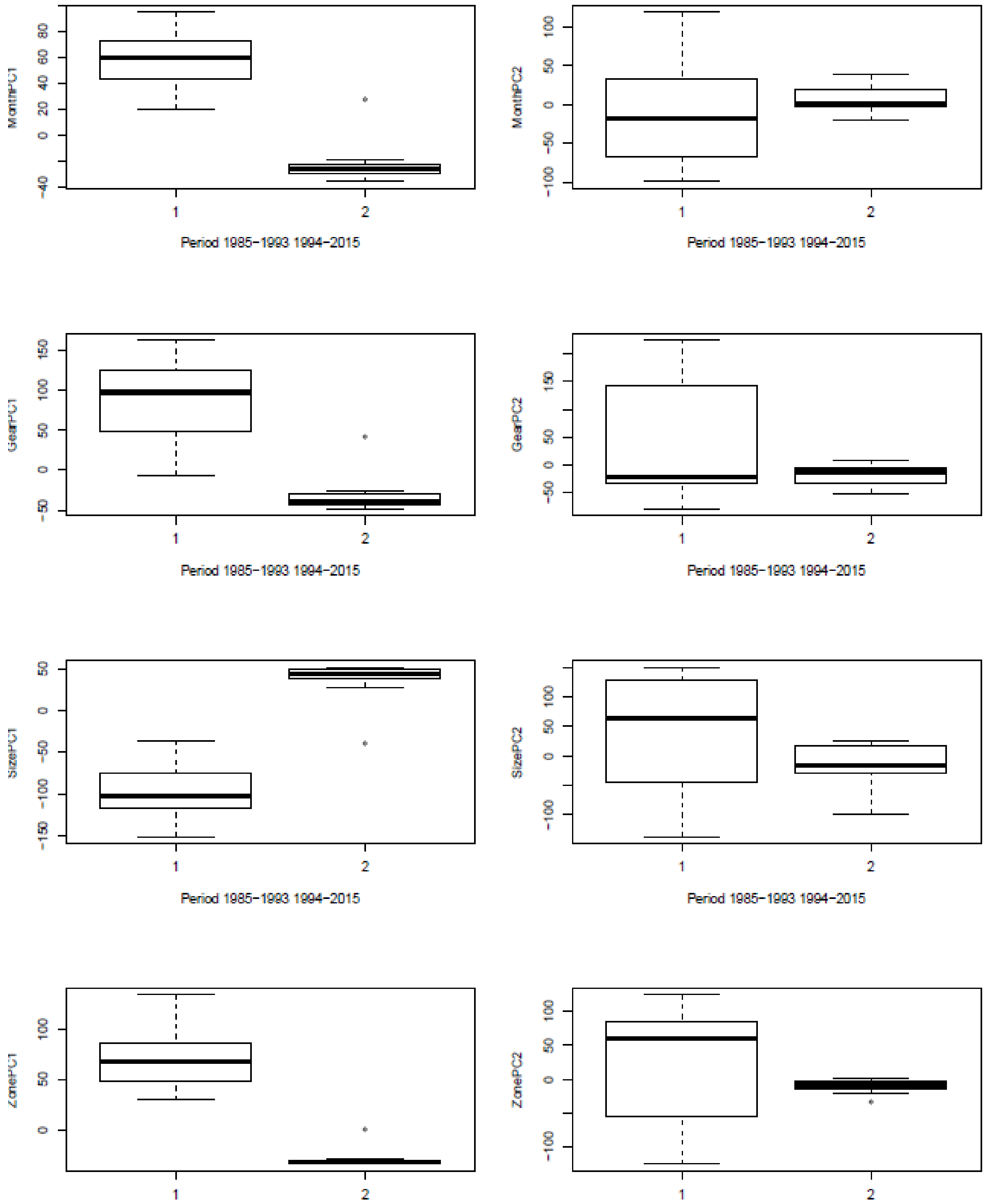


Figure D.1. Results of Cluster Analysis with two groups.

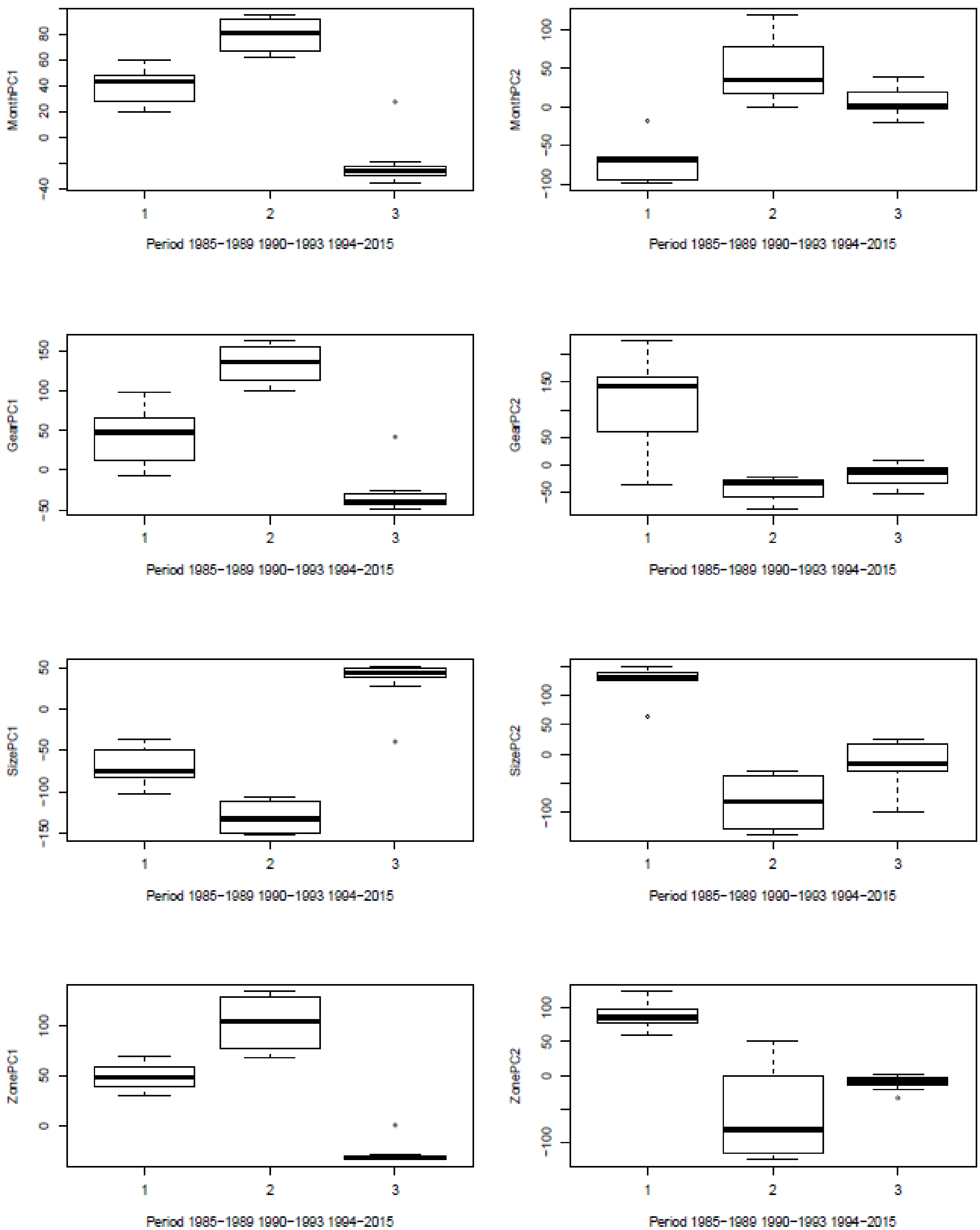


Figure D.2. Results of Cluster Analysis with three groups.

Table D.9. Fish size selectivity of Sebastes spp. described by five quantiles (cm) for each time periods determined by the cluster analysis (2 or 3 clusters scenarios).

	2 clusters scenario				
	Q10	Q25	Q50	Q75	Q90
Period 1985-1993	26.30	28.37	30.95	33.68	36.22
Period 1994-…	28.10	30.13	32.37	34.81	37.06

	3 clusters scenario				
	Q10	Q25	Q50	Q75	Q90
Period 1985-1989	25.75	28.34	31.40	34.24	36.73
Period 1990-1993	26.99	28.40	30.38	32.99	35.58
Period 1994-...	28.10	30.13	32.37	34.81	37.06

METHODS TO SIMULATE FUTURE RECRUITMENT DEVIATES

It is common practice in MSE to simulate future recruitment such that they retain attributes seen in estimates of historical recruitment obtained from fits of the operating models to historical data (Cox and Kronlund 2016; Szuwalski and Punt 2016). It is common to simulate recruitment using a stock recruitment function obtained from fitting the operating model to data and a non-parametric or parametric bootstrap of recruitment deviates based on the estimated time series of historical recruitment deviates (Cox and Kronlund 2016; Szuwalski and Punt 2016; Jones et al. 2016). It is also common to observe positive autocorrelation in estimates of historical recruitment deviates and to mimic this pattern in simulations of future recruitment by applying a lag-1 positive autocorrelation function in a parametric bootstrap, or to take draws of sequences of historical recruitment deviates with the use of a conditional non-parametric bootstrap (Smith et al. 1993; Cox and Kronlund 2016).

We thus examined the temporal patterns of estimates of historical deviates obtained from fitting the operating models to data to identify key attributes in the historical recruitment deviates and then based on the observed attributes in the estimated recruitment deviates formulated two alternative approaches to simulate future recruitment deviates. See Table D.10 for estimates of cohort strength for 1951-2016 obtained from fitting the base case operating model to data for *S. mentella* and *S. fasciatus*. We have focused our characterization of the attributes of historical age 0 recruitment deviates on years 1970-2013, i.e., the years in which there is sufficient information in the data to estimate the recruitment deviates. Recruitment deviates are first estimated for the 1951 cohort. However, the first cohort strengths that could be estimated based on data would be around 1970. The length composition data in the fishery independent trawl survey in Unit 1 began in 1984. Recruitment deviate estimates for both *S. mentella* and *S. fasciatus* show increased departures from their prior means starting around 1970, thus indicating that the information contained in length composition data provide cohort strengths back to the 70’s. In addition, the trawl survey selects fish with 50% or greater probability at age 2, and extends to 2017. Thus, the last cohort that could be estimated would be 2013, since a minimum of about two years of composition data are needed for cohort strength estimation (e.g., Cox and Kronlund 2016).

The time series of estimated recruitment deviates for both species appears to be highly structured (Figure 1). Firstly, there are a few brief periods in both time series when large and exceptionally large recruitment deviates (i.e., multipliers larger than about 5) for both species occur. These include the years 1972-73, 1980-1981, and 2011-2013. This suggests that conditions favouring unusually good juvenile survival occur infrequently for a few years for both species. The correlation in deviates across the two species was quite high, at 0.74 (p-value=0.001), also supporting the notion that the ecological conditions favouring good recruitment are similar for both species and co-occur in Units 1 and 2. The autocorrelation in recruitment deviates at lag 1 was positive at 0.32 (p-value=0.03) for *S. mentella* and 0.48 (p-value=0.001) for *S. fasciatus* for 1970-2013. Excluding the final three years, i.e., 2011-2013, that had extremely high recruitment deviates the autocorrelation increased to 0.66 (p-value=0.001) for *S. mentella*, but remained nearly the same at 0.45 (p-value=0.001) for *S. fasciatus*. This suggests that for the bulk of the 44-year time series, auto-correlation at lag 1 is positive and fairly high for both species. For both species, sequences of large positive recruitment deviates (multipliers of about 5 and larger) occurred no fewer than eight years following the previous strong cohort. In any forty year sequence, there were no more than four large recruitment deviates. Also there have been no more than three large deviates occurring within a three year sequence (i.e., two large deviates in a row for *S. mentella*, i.e., 1980-81, and three in a row for *S. fasciatus*, i.e., 2011-2013).

The observed spacing of sequences of large deviates and relatively low frequency of exceptional large historical deviates have prevented excessively large abundances from accumulating in the past. For example, the maximum estimated SSB was 800 kts for *S. fasciatus* and 1200 kt for *S. mentella*. We found that if two or more sequences of large deviates were simulated to occur within a short sequence of years, e.g., within a shorter span than previously observed, then unrealistically high stock biomass would accumulate within a few decades (e.g., spawning stock biomass of greater than 20 million tons). These attributes were considered in the formulation of a conditional nonparametric bootstrap and also a conditional parametric bootstrap.

Method 1: Conditional nonparametric bootstrap

The aim of this approach was to simulate future recruitment deviates that similar statistical attributes to those that were estimated for the years 1970-2013. First, a large cohort was defined using the fifth largest estimated recruitment deviate (i.e., in 1980 for *S. mentella* and 2013 *S. fasciatus*). The six key attributes that were observed for both species that this approach aimed to mimic were as follows:

(1) high correlation in recruitment deviates exists between the two species, e.g., when a high recruitment event occurs in one species, it is also likely to occur in the 2nd species, and vice versa,

(2) high autocorrelation in recruitment deviates occurs at lag one year for both species,

(3) sequences of large recruitment events can last no longer than three consecutive years and if three years, there can be a small recruitment between two large ones,

(4) conditions for exceptional recruitment (for up to three years) can occur no more than twice within 40 years (e.g., as was observed for 1980-1981 and 2011-2013),

(5) no more than four large cohorts (i.e., large and exceptionally large cohorts) can occur within 40 years,

(6) conditions for a strong recruitment event (which may occur over one to three years) can occur no sooner than 8 years after the previous instance of strong recruitment.

A conditional non-parametric bootstrap was identified as the Monte Carlo technique that would most easily incorporate all six attributes listed above. To allow for a sufficiently large population of unique draws, a non-parametric bootstrap requires a large number of potential replicate sequences and a relatively short sequence length. Our sequence of deviates from 1970-2013, had only 44 deviates. With segment size set at 5 years we obtained 39 unique candidate segments. 5 years is an arbitrary choice for segment size but still allowed for the representation of lag 1 autocorrelation and sequences of up to three large recruitment events to be drawn. To implement these six conditions we applied the following steps:

Step 1. For the 1st five years from 2018-2022, a time-series of five years that contains recruitment deviates for both species that are not large in magnitude is chosen at random and with replacement from the estimated sequences of deviates 1970-2013 (following condition 6).

Step 2: For the next five years, a time-series of five years with consecutive recruitment deviates for both species is chosen at random with replacement (this sequences may or may not include large (or exceptional) deviates).

Step 3. Repeat step 2 with a different random seed six more times.

Step 4. Join the eight sequences from Step 1-3 to create a single time-series of 40 years.

Step 5. Repeat Step 1-4, 10,000 times and for operating model projections select only those time-series that follow conditions 3-6 above.

Method 2: Conditional Parametric Bootstrap

We assumed that future recruitment deviates for the two species had the following properties:

(1) *S. mentella*

- Standard deviation in recruitment deviates (σ_R (sm)) 1.56
- Auto correlation coefficient at lag 1 year = 0.32

(2) *S. fasciatus*

- Standard deviation in recruitment deviates (σ_R (sf)) = 1.06
- Auto correlation coefficient at lag 1 year = 0.48

(3) Correlation between *S. mentella* and *S. fasciatus* recruitment deviates = 0.74

We applied the following steps to attempt to achieve the above three sets of conditions.

Step 1. For *S. fasciatus* (sf), simulate from a normal distribution that has a mean of zero and standard deviation 1.06, and using a lag 1 year autocorrelation function with an autocorrelation coefficient (rho(sf)) of 0.5 (rounding up from 0.48), generate a time series of forty recruitment deviates (e(y)). i.e., for years 2018-2057:

$$e(sf, y) = rho(sf) * e(sf, y\text{-}1) + rannorm * \sigma_R(sf) * Sqr(1 – rho(sf) * rho(sf))$$

where rannorm is a random normal deviate with a mean of zero and standard deviation of 1.

Step 2. Using a normal distribution with a mean of zero and standard deviation of 1.56, the across species correlation coefficient of 0.74, and time series of recruitment deviates generated for *S. fasciatus* from step 1, generate a time series of forty recruitment deviates from 2018 to 2057 for *S. mentella*, i.e., for years 2018-2057:

e(sm, y) = Sqr(0.5) * (corfm * σ_R(sm) / σ_R(sf) * e(sf, y) + rannorm * σ_R(sm) * Sqr(1 - corfm * corfm))

+ Sqr(0.5) * (rho(sm) * e(sm,y - 1) + rannorm * σ_R(sm) * Sqr(1 – rho(sm) * rho(sm)))

Note that when the above procedure was implemented the parameter corfm had to be adjusted to 1 and the rho(sm) parameter to 0.11 to achieve error deviates for *S. mentella* with a standard deviation of about 1.56, correlation between *S. mentella* and *S. fasciatus* error deviates of about 0.74, and an autocorrelation coefficient of about 0.32.

Step 3. Repeat steps 1 and 2 10000 times.

Table D.10. Estimates of cohort strength for S. mentella and S. fasciatus in Units 1 and 2 obtained from fitting the base case operating model to data.

Year	*S.mentella*	*S.fasciatus*	Year	*S.mentella*	*S.fasciatus*	Year	*S.mentella*	*S.fasciatus*
1951	1.0	1.1	1973	1.6	1.3	1995	0.4	0.3
1952	1.0	1.1	1974	1.2	1.0	1996	1.1	1.4
1953	1.0	1.1	1975	1.0	0.9	1997	0.4	0.3
1954	1.0	1.2	1976	1.1	0.9	1998	0.7	0.8
1955	2.1	3.2	1977	1.4	1.0	1999	0.7	1.9
1956	1.0	1.2	1978	1.7	1.1	2000	0.6	0.5
1957	2.1	3.3	1979	1.8	1.6	2001	0.4	0.6
1958	1.0	1.2	1980	4.8	2.6	2002	0.5	1.5
1959	1.0	1.2	1981	12.1	5.7	2003	0.4	1.5
1960	1.0	1.2	1982	1.2	1.1	2004	0.4	1.6
1961	1.0	1.2	1983	0.5	0.4	2005	0.4	1.7
1962	1.0	1.2	1984	0.7	0.5	2006	0.5	0.6
1963	1.0	1.1	1985	0.7	0.6	2007	0.3	3.1
1964	1.0	1.1	1986	0.8	0.4	2008	0.3	0.7
1965	1.0	1.1	1987	0.5	0.2	2009	0.8	0.7
1966	1.0	1.1	1988	0.3	0.2	2010	0.8	0.6
1967	1.0	1.1	1989	0.3	0.2	2011	589.9	25.1
1968	1.1	1.1	1990	0.3	0.2	2012	1.2	9.7
1969	1.1	1.2	1991	0.2	0.2	2013	46.5	6.2
1970	1.2	1.3	1992	0.2	0.1	2014	1.0	0.6
1971	1.3	1.4	1993	0.4	0.3	2015	1.0	0.6
1972	6.3	1.5	1994	0.7	0.6	2016	1.0	1.0

APPENDIX E: METHODOLOGY FOR DEVELOPING A PRIOR FOR RECRUITEMENT DEVIATES OF STRONG COHORTS

INTRODUCTION

Redfish stocks within the Gulf of St. Lawrence have been found to show evidence of large cohort size during historical periods prior to years for which fishery and survey data are available (Table E.1). To account for these large recruitment deviates within the current investigation, a meta-analysis using an empirical Bayes approach was conducted. This analysis quantified recruitment deviates observed in *Sebastes* spp. populations outside the current study area for use in generating a prior for mean deviation for the current investigation in years where it was identified that large cohorts were present in the data.

METHODS

The empirical Bayes meta-analysis of *Sebastes* spp. stock recruitment data was conducted utilizing data compiled from a literature review and the RAM Legacy Stock Assessment Database (Ricard *et al.* 2013). Of the available data, a total of nineteen stocks were included in the investigation (Table E.2). The remaining eighteen available datasets were excluded due to missing spawner or recruit data, presence of abrupt discontinuities in the time series, or presence of constant values (i.e. indicating estimation failure). Data was utilized for both Pacific and Atlantic *Sebastes* spp. stocks, with the majority of data (17 of 19 stocks) coming from Pacific stocks. Data was available for only two redfish stocks on the east coast. The mean length of time-series data utilized was 31.5 years.

Calculations of recruitment deviance were evaluated as follows. First estimates of expected recruitment through time for each stock was determined by evaluating fits of the data to the Beverton-Holt stock-recruitment function (equation 1) (Walters and Martell 2004). The form of the Beverton-Holt stock-recruitment function for population *i* is:

(1) $$R_{i,t+1} = \frac{\alpha_1 E_{i,t}}{1+\beta_1 E_{i,t}} e^{\sigma_t}$$

where α describes the slope of the function near the origin (i.e. the maximum survival rate of recruits), β is the scaling parameter of the stock-recruit function, $E_{i,t}$, is an index of the spawning stock biomass (measured as SSB), $R_{i,t+1}$ is the expected number of recruits, and $e_{i,t}$, is a normally distributed process error. Parameter estimation for Beverton-Holt stock-recruitment function was conducted using Solver in Excel (Microsoft Corporation 2010). The objective function was the negative log likelihood.

Once a time series of expected recruitment was determined, the scale of recruitment deviance was determined by calculating a recruitment multiplier (mR_t) by year t as (leaving out the stock indicator i):

(2) $$mR = e^{Et_t}$$

where e_t is the recruitment deviate in year t.

mR within each time series were then compared to an arbitrarily determined threshold deviance of 5, or values of in the natural logarithm of at least 1.609. The mean of each time series of mR was then calculated by stock. The grand mean and standard deviation (later converted to precision) of mR across populations was then calculated. These values were then used as an informative, non-negative positive prior for recruitment deviates in the operating model for years that were identified to have produced large cohorts.

RESULTS

Fits of each stock to the Beverton-Holt stock recruitment function are shown in figure 1 with stock specific estimates the parameters of the Beverton-Holt equation (α and β) and the standard deviation in the natural logarithm of the deviates (σ_r) shown in Table E.3. Of the nineteen stocks investigated, a total of ten were found to have large recruitment deviates. The number of years large recruitment deviates varied by stock from one to five, with the mean number of large recruitment deviates across stocks of 2.4. The largest mean recruitment deviate (3.33) was observed for North Pacific coast *S. flavidus*.

The mean value of large recruitment deviates across populations was 2.07 with a standard deviation of 0.56. The average value for the standard deviation in the natural logarithm of recruitment multipliers was 0.84, with a standard deviation of 0.46. The prior mean thus for a large recruitment deviate was set at 2.07. The prior standard deviation for recruitment deviates was rounded up to 1.0, to allow for larger uncertainty due to the low representation of redfish in the *Sebastes* spp. stock recruit data sets included in the meta-analysis.

REFERENCES

Microsoft Corporation. 2010. Microsoft Excel for Mac 2011. Version 14.7.7.

Ricard, D., Minto, C., Jensen, O.P. and Baum, J.K. 2013. Evaluating the knowledge base and status of commercially exploited marine species with the RAM Legacy Stock Assessment Database. *Fish and Fisheries* 13 (4) 380-398. DOI: 10.1111/j.1467-2979.2011.00435.x

Table E.1. Identification of years with strong Unit 1 and 2 redfish cohorts based on trawl survey and literature sources.

Source	Quality	Year class strength	Years available	Strong year classes	Notes	Reference
Gadus winter survey	High	yes but not by species	1978-1994	1974, 1980, 1985, 1988	biased in later years due to ice conditions	DFO survey data repository, Quebec Region
Hammond summer survey	High	yes	1984-1990	1974, 1980, 1985, 1988	Western IIa trawl	DFO survey data repository, Quebec Region
Alfred Needler summer survey	High	yes	1990-2005	1980, 1985, 1988, 2003	URI trawl	DFO survey data repository, Quebec Region
Teleost summer survey	High	yes	2004-2017	2003, 2011, 2012, 2013	Campelen shrimp trawl	DFO survey data repository, Quebec Region
Sentinel mobile survey	High	yes	1995-2017	2003, 2011, 2012, 2013	Engels trawl	DFO survey data repository, Quebec Region
Literature	High	no	1974-2013	1974, 1980, 1985, 1988, 2003	genetic confirmation of species	Valentin et al 2015
Literature	Medium	no	1945-1960	1956, 1958	mention of good year classes and in scientific reports	CAFSAC Document 1984

Table E.2. Description of Sebastes spp. stock-recruit datasets utilized in analysis.

Scientific name	Region	Region Code	Data collection agency	Time series length (yrs)	Time series range (yrs)	Source
S. variabilis	Gulf of Alaska	GoA	Alaska Fisheries Science Center	23	1977-1999	1
S. aleutianus	Eastern Bering Sea and Aleutian Islands	EBSAI	Alaska Fisheries Science Center	23	1977-1999	1
S. alutus	Gulf of Alaska	GoA	Alaska Fisheries Science Center	31	1977-2007	1
S. alutus	Eastern Bering Sea and Aleutian Islands	EBSAI	Alaska Fisheries Science Center	22	1977-1998	1
S. alutus	Eastern Pacific Coast	EPC	Northwest Fisheries Science Center	34	1956-1989	1
S. ruberrimus	Eastern Pacific Coast	EPC	Northwest Fisheries Science Center	33	1923-1955	1
S. carnatus	South Eastern Pacific Coast	SEPC	Southwest Fisheries Science Center	36	1965-2000	1
S. crameri	Eastern Pacific Coast	EPC	Northwest Fisheries Science Center	35	1928-1962	1
S. fasciatus	Gulf of Maine / Georges Bank	GoM	Northeast Fisheries Science Center	27	1981-2007	1
S. flavidus	North Eastern Pacific Coast	NEPC	Northwest Fisheries Science Center	36	1967-2002	1
S. goodei	South Eastern Pacific Coast	SEPC	Northwest Fisheries Science Center	35	1892-1928	1

Table E.2 continued. Description of Sebastes spp. stock-recruit datasets utilized in analysis.

Scientific name	Region	Region Code	Data collection agency	Time series length (yrs)	Time series range (yrs)	Source
S. jordani	Eastern Pacific Coast	EPC	Southwest Fisheries Science Center	22	1964-1985	1
S. levis	Southern California	SCAL	Northwest Fisheries Science Center	36	1900-1935	1
S. melanostomus	Eastern Pacific Coast	EPC	Northwest Fisheries Science Center	36	1950-1985	1
S. mystinus	California	CAL	Northwest Fisheries Science Center	36	1916-1951	1
S. paucispinis	South Eastern Pacific Coast	SEPC	Northwest Fisheries Science Center	56	1951-2006	1
S. polyspinis	Gulf of Alaska	GoA	Alaska Fisheries Science Center	45	1961-2005	1
S. polyspinis	Eastern Bering Sea and Aleutian Islands	EBSAI	Alaska Fisheries Science Center	22	1977-1998	1
S. mentella	Irminger Sea and adjacent areas	ISAA	International Council for the Exporation of the Sea	12	1990-2001	2

[1]. Ricard, D., Minto, C., Jensen, O.P. and Baum, J.K. 2013. Evaluating the knowledge base and status of commercially exploited marine species with the RAM Legacy Stock Assessment Database. *Fish and Fisheries* 13 (4) 380-398. DOI: 10.1111/j.1467-2979.2011.00435.x

[2]. ICES. 2017. Report of the North Western Working Group (NWWG), 27 April – 4 May 2017, Copenhagen, Denmark. ICES CM 2017/ACOM:08. 642 pp.

Table E.3: Stock specific estimates of α, β, and σ_r.

Scientific name	Region	Alpha	Beta	σ_r
S. variabilis	GoA	0.68	5.42×10^{-5}	1.05
S. aleutianus	EBSAI	5.27×10^{5}	586.72	0.62
S. alutus	GoA	5.09	8.40×10^{-5}	0.48
S. alutus	EBSAI	3.73×10^{10}	5.8×10^{-5}	0.55
S. alutus	EPC	8.18×10^{9}	4.05×10^{-5}	2.00
S. ruberrimus	EPC	7.10	0.00	0.62
S. carnatus	SEPC	813.14	4.96×10^{-3}	0.28
S. crameri	EPC	1.77	1.11×10^{-3}	0.54
S. fasciatus	GoM	2.00×10^{8}	6729.98	1.15
S. flavidus	NEPC	4.09×10^{-3}	0.00	0.76
S. goodei	SEPC	1.29×10^{8}	5684.36	3.35
S. jordani	EPC	66.00	1.45×10^{-4}	1.09
S. levis	SCAL	27.13	1.22×10^{-3}	0.98
S. melanostomus	EPC	3.14	1.08×10^{-5}	0.86
S. mystinus	CAL	2.80×10^{7}	4762.43	0.33
S. paucispinis	SEPC	0.29	1.54×10^{-4}	0.82
S. polyspinis	GoA	1.49×10^{4}	1.10	0.6
S. polyspinis	EBSAI	2.00	6.00×10^{-5}	0.63
S. mentella	ISAA	6.48	6.47×10^{-2}	0.13

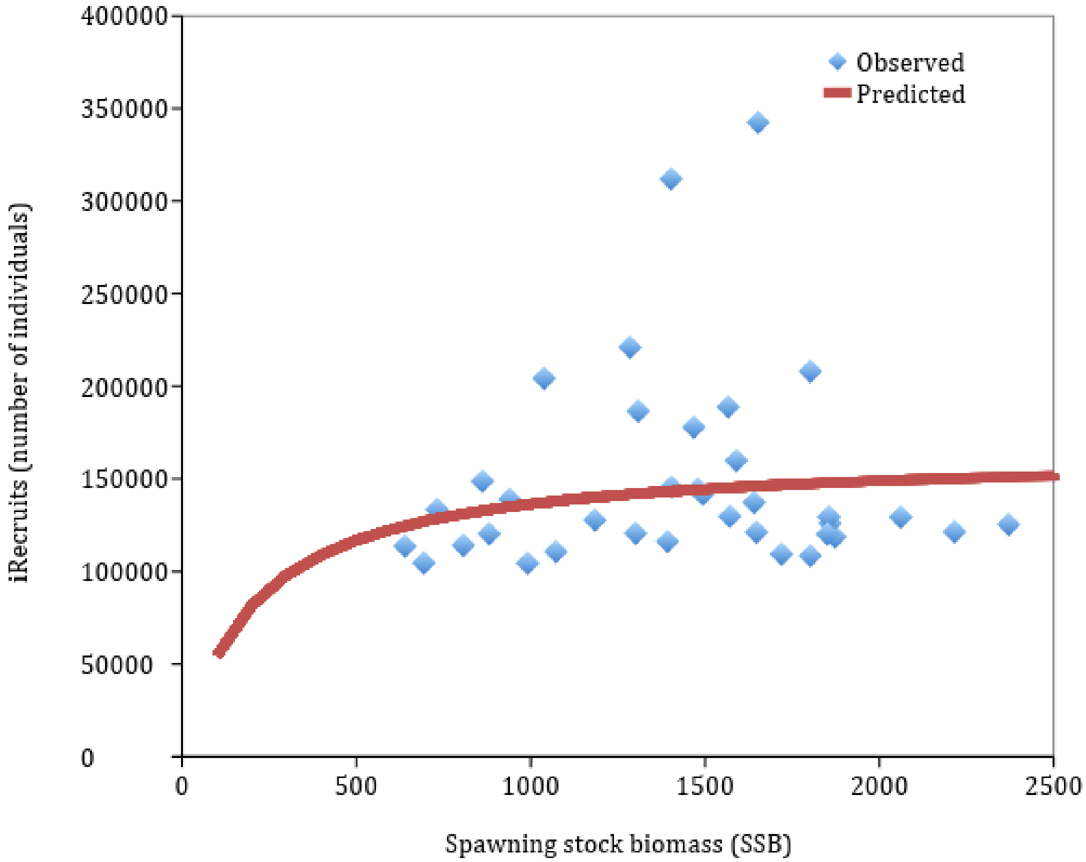


Figure E.1. Fits to stock-recruitment datasets obtained for 19 Sebastes spp. stocks under the assumption the Beverton-Holt (red line) stock-recruitment function.

APPENDIX F – OBJECTIVES AND PERFORMANCE METRICS

Table F.1. Candidate MSE Objectives, matched to the corresponding Performance Metrics, with passing criteria identified.

	Candidate Stock Objectives
1	Increase SSB of each of *S. mentella* and *S. fasciatus* above the lower reference point (LRP) and into the Healthy Zone in 10 years (95% probability).
Corresponding Performance Metrics	1a: Proportion of simulations where SSB of each species exceeds LRP in 10 years PASS CRITERIA: 95% or higher 1b: Proportion of simulations where SSB of each species exceeds USR in 10 years. PASS CRITERIA: 95% or higher
2	Once in Healthy Zone, maintain SSB of each of *S. mentella* and *S. fasciatus* above the Critical Zone (95% probability), and in the Healthy Zone (75% probability[SC2]).
Corresponding Performance Metrics	2a: Proportion of years the SSB of each species is above the LRP after 10 years. PASS CRITERIA: 95% or higher 2b: Proportion of years the SSB of each species is above the USR after 10 years. PASS CRITERIA: 75% or higher
3	Maintain exploitation rate U of *S. mentella* and *S. fasciatus* below Umsy, 50% probability.
Corresponding Performance Metrics	3: Proportion of years where the ratio of U:Umsy by species < 1. PASS CRITERIA: 50% or higher
4	Maximize the number of years where fish < 22 cm represent < 15% of catch (and Small Fish Protocol is not triggered).
Corresponding Performance Metrics	4: Mean number of years where fish < 22 cm represent < 15% of catch. a) 5 years; PASS CRITERIA: 85% or higher b) All 40 years; PASS CRITERIA: 85% or higher
5	Maximize the number of years where fish < 25 cm represent < 15% of catch.

	Candidate Stock Objectives
Corresponding Performance Metrics	5: Mean number of years where fish < 25 cm represent < 15% of catch. a) 5 years; PASS CRITERIA: 85% or higher b) All 40 years; PASS CRITERIA: 85% or higher
6	Maximize the duration of high annual catch. PASS CRITERIA: No pass/fail, but rank the performance of the Management Procedures/Harvest Control Rules
Corresponding Performance Metrics	6a: Average annual catch in a) 10-20, b) 10-40 years. 6b: Proportion of simulations where catch limit reaches or exceeds 40,000 tons by 2028 (i.e., after the large cohorts are expected to fully recruit to the fishery). 6c: Mean number of years where the catch limit is as large or larger than 40,000 tons in the years 2028-2057. 6d: Proportion of years with > 2017 landings [2017 will be a reference year; note 2016 catch killed = approx. 4040 t assuming 1:1 catch kill:retained ratio]
7	Maximize catch of large fish (>27 cm[SC3]).
Corresponding Performance Metrics	7: Proportion of years where percentage of fish > 27 cm is > 80%. a) 5 years; PASS CRITERIA: 50% or higher b) All 40 years; PASS CRITERIA: 75% or higher
8	Maintain the stability of the fishery (annual changes in TAC are consistent with industrial capacity).
Corresponding Performance Metrics	8a: Percentage of years where recommended TAC is < 15% different from previous TAC. PASS CRITERIA: 75% or higher 8b: Average Annual Variation in TAC (percentage) during a) 10-20 years; PASS CRITERIA: 15% or lower b) 10-40 years; PASS CRITERIA: 15% or lower

APPENDIX G: EXPLORATION OF A HYPOTHESIS ABOUT RANGE CONTRACTION IN UNIT 1 AND 2 REDFISH

In the Unit 1 and Unit 2 redfish MSE, there are three key assumptions about the swept area abundance indices obtained from the DFO and GEAC groundfish trawl surveys in those areas.

1. Each index for each species is directly proportional to abundance.

- Therefore the relative changes in the index can be expected to accurately represent the actual relative changes in stock abundance.

2. Each index can be treated as a relative abundance index.
3. If used on its own each index would accurately reflect actual relative changes in stock abundance.

- Therefore the Unit 1 trawl survey index for the two redfish species can be used as a relative abundance index in management procedures for both species.

Over the years in which the abundance index data are available for Units 1 and 2, it appears that adequate fits of the operating models to the trawl survey abundance indices are obtained for both species from Units 1 and 2, i.e., the model predicted biomass follows the trends seen in both indices for both species (Figures 3, 10). However, as noted above, the estimates of the constant of proportionality for the Unit 1 and especially the Unit 2 trawl survey indices are very high (i.e., larger than 1.0) for both species for most operating models (e.g., for the Unit 2 trawl survey index, q was about 1.7 for *S. mentella* and about 2.5 for *S. fasciatus* (Tables 4 and 5)).

Two hypotheses that specified alternative mechanisms for the high estimates of q for the trawl abundance indices for Unit 2 are as follows:

1. Catch under-reporting: Catches e.g. for 1988-1994 under-reported. If catches were substantially underreported in the period in which the abundance indices dropped most severely this would cause a model that used the under-estimates of catch to produce inflated estimates of q.

2. Range contraction: Absolute fish abundance depletes more in Unit 1 than Unit 2 when stock abundance decreases. If the stock area with the highest fish densities under all different levels abundance was in Unit 2, and the stock expanded its range into Unit 1 as abundance increased and then gradually left Unit 1 as the stock declined, this could give rise to hyperdepletion (or hyper-expansion) in the Unit 1 index. Without the availability of data from surveys taken in years prior to 2000 in Unit 2, the extent of depletion of stock biomass in Unit 2 prior to 2000 remains unknown. If there was a high degree of range contraction, the density of redfish in Unit 2 could remain hyperstable when the stocks either increased or decreased, even though the apparent abundance in Unit 1 would appear to increase markedly or decrease markedly depending on whether the stock was undergoing range contraction or range expansion.

We report below the results of several different analyses. In the first set of analyses, population dynamics models for *S. fasciatus* and *S. mentella* were fitted to the Unit 1 and 2 trawl survey abundance indices using the records of reported redfish catches and hypothesized ratios of catch biomass killed to catch biomass reported. To allow more rapid and thorough data analysis, a state-space semi-age-structured model, delay-difference model, was fitted to the abundance indices using Excel spreadsheets (Walters and Martell 2004; Yamanaka et al. 2018). This type of stock assessment model includes a Beverton-Holt stock recruit function, annual deviates from this function to allow estimation of cohort strength, the same growth and natural mortality rates, and the same median age at maturity as in the base case operating models. For a description of the particular approach to fitting the delay difference model to abundance indices and the equations applied see Yamanaka et al. (2018). The time series of reported landings for both species, the base case scenario for the ratio of catch killed to catch retained and prior for Beverton-Holt steepness were also applied. The delay difference models were fitted to different configurations of the Units 1 and 2 trawl survey biomass indices to explore the sensitivity of estimated trends in abundance to different interpretations of the data. In the initial fits, a vague prior for the constant of proportionality for the trawl survey indices, q, was implemented to allow unconstrained estimation of q. Fixed standard deviations for the likelihood function with values similar to those applied under the base case operating model were also applied when fitting the delay difference model.

Five different versions of the delay difference model were tried.

1. **Treat the abundance indices as relative abundance indices, i.e., $I_{i,y} = q_i B_y$.** where $I_{i,y}$ is the predicted index of abundance for trawl survey i, q_i is the constant of proportionality for the trawl survey i abundance index and B_y is the total stock biomass vulnerable to the survey in year y. The delay difference model was first fitted to the trawl data to examine the goodness of fit of the model to the data under the commonly made assumption that the indices can be treated purely as relative indices of abundance. The model predicted values for the survey biomass were back-projected to 1960 to show the model's predictions of the abundance indices relative to spawning stock biomass (SSB), should the Unit 2 trawl survey have been conducted all years back to 1960 when the stocks were assessed to have been much higher in abundance. This back projection of the predicted index allows an evaluation of the credibility that the indices from the Units 1 and 2 trawl surveys are directly proportional to total stock size under all levels of stock abundance. Large values for the model-predicted trawl survey indices for Unit 2 (e.g., $q_2 * B_y > 2$ million tons), especially in the earlier years of the fishery when stock size could have been much higher would call into question the credibility of the assumption that the indices are directly proportional to abundance.

2. **Consider the catch under-reporting hypothesis**. Estimate a bias factor, *b*, in the reporting of catches from 1985-1994 when the Unit 1 index depleted the most, i.e. for 1985-1994,

$c_y = b*ckr_y * c_{y,rep}$

where c_y is the catch biomass killed, ckr_y is the catch biomass killed to catch killed ratio for year y assumed in the base case operating model, *b* is the inverse of the average bias adjustment factor in catch reporting which is a multiple of the value for the fixed inputted value of ckr_y, and $c_{y,rep}$ is the total catch biomass reported in year *y* for the species evaluated. If the product of $b*ckr_y$ is excessively high, e.g., implying catches would need to be 10 or more times larger than the reported landings to explain the large drop in the Unit 1 index, then this would cast doubt on the hypothesis that underreporting of catches could cause the estimated values for q to be very high.

3. **Estimate a nonlinearity coefficient β for the Unit 1 trawl survey index**, i.e., $I_{1,y} = q_1 B_y^{\beta}$ where β is the nonlinearity coefficient for the Unit 1 trawl survey index. Estimates of β = 1 would imply that the index is directly proportional to abundance; estimates of β > 1 would predict the index to decline at a steeper rate than that for actual abundance and would be consistent with the hypothesis of hyperdepletion in the trawl index of abundance.

4. **Estimate a range contraction parameter, *rc*, for the Unit 1 trawl survey index of abundance**. One potential representation of a time dynamic value for the constant of proportionality, q_y, for the Unit 1 trawl survey that is driven by range contraction is given by: $q_y = q_z / (rc + (1 - rc) * B_y / (fBz * B_0))$ when B_y is < $fBz * B_0$

and

$q_y = q_z$ when $B_y \geq fBz * B_0$

where q_z is the constant of proportionality for stock sizes larger than ***fBz * B₀***, *rc* is the range contraction factor, $\boldsymbol{B_0}$ is the average unfished stock biomass vulnerable to the trawl survey, ***fBz*** is the fraction of average unfished stock biomass under which range contraction is predicted to occur, and B_y is the total population biomass that is potentially vulnerable to the trawl survey. In fitting the delay difference model to the data, the parameters *fBz* was fixed at different trial values and then fixed arbitrarily at 0.5, implying that range contraction occurs when stock size drops to less than half of the average unfished stock size. The index for Unit 2 was assumed to be a purely relative index of abundance. The index for Unit 1 was assumed to undergo range contraction with the value for q_y given by the above given dynamic equation for q_y.

5. Sum the Unit 1 and 2 indices for every year where there a Unit 1 and 2 index.

At a meeting held for a pre-peer review of the Units 1 and 2 redfish MSE in February 2018 it was suggested that to address the issue of anomalously high estimates of the constant of proportionality (q) for the swept area trawl indices in Units 1 and 2, a combined Unit 1-Unit 2 swept area biomass index (I_{1+2}) should be formulated from the Unit 1 and 2 trawl survey data sets. The idea being that should Units 1 and 2 encompass the full range of the stock area for both species, an index that combined the trawl survey records from both Units 1 and 2 should eliminate the potential range contraction (and expansion) attributes of the Unit 1 trawl survey index.

The Unit 1 and 2 trawl survey data could be combined starting in 2000 when both Unit 1 and 2 trawl survey indices become available. The Unit 1 index would remain as a separate relative index of abundance for the eight years in which the Unit 2 survey index was not available following 2000 (about once every two years). An approximation of the combined index would be the summation of the swept area biomass index for Unit 1 and Unit 2 for each year in which both indices were available. Apparently the survey gear and survey protocols have been practically identical in the DFO Unit 1 and GEAC Unit 2 trawl surveys since 2000. So, for example, for the year 2000, $I_{1+2,\,2000} \approx I_{1,\,2000} + I_{2,\,2000}$. Thus the value of the combined index for the two management Units could be expected to be larger than either one by itself in years where both indices are available, i.e.,

$I_{1+2,y} = I_{1,y} + I_{2,y}$

where $\boldsymbol{I_{1+2,y}}$ is the trawl survey index resulting from the summation of the index values for Units 1 and 2 in year y. We considered two options for fitting the model to a trawl survey index for the years 1984-1999 where only the Unit 1 index was available. The first option was to fit the model to the Unit 1 index for years 1984-2017 in which the Unit 2 index was not available, treating it as a relative abundance index. A second option was to impute a Unit 1+ Unit 2 trawl survey index for 1984-1999 by adding the mean for the Unit 2 index in years 2000-2016 to the annual observed Unit 1 index in each of the years 1984-1999. This assumes that within the time from 1984-2016 the abundance in Unit 2 remains on average hyperstable with changes in total stock abundance and that at least for years 1984-1999 the stock would increase into Unit 1 as the stock systematically increased and contract out of Unit 1 as the stock decreased. The proposition that for 1984-2016 the average abundance in Unit 2 remains hyperstable with changes in total stock sizes is purely speculative.

Here it was assumed that under hypothesized range contraction of the Unit 1 and 2 redfish stocks, the average stock biomass in Unit 2 remained relatively stationary from 1984-2016. We thus imputed a value for the Unit 2 stock biomass for the years 1984-1999 by taking the average value of Unit 2 stock biomass from 2000-2016, e.g., about 182 kt and 156 kt for *S. fasciatus* and *S. mentella*, respectively. We then added this averaged Unit 2 survey biomass value to the Unit 1 index for each year from 1984-1999 and then used the summed Unit 1 and 2 biomass for 2000-2016 where both indices were available.

The ability of the model to fit the abundance index data and resulting estimates of q for the abundance indices under both options for combining the Unit 1 and Unit 2 trawl indices was assessed.

In addition, spatial plots of average trawl survey catch rates in Units 1 and 2 were examined to check for the possibility of range contraction as the stocks declined and range expansion as the stocks have recently started to increase.

RESULTS

The state-space delay difference models that were fitted to the Units 1 and 2 trawl indices as relative indices fitted the data fairly well and give estimates of depletion for both species similar to those obtained from fitting the base case operating model (Figure G.1; Table G.1). The model fit estimates of the standard deviation, i.e., sigma, in the log deviates between observed and predicted values for both species for Units 1 and 2 were between about 0.3 and 0.45 and were not excessively high. However, the model predicted values for the Unit 2 trawl index in the early 1960s were very high, i.e., about 3.5 million tons for *S. fasciatus* and 1.8 million tons for *S. mentella*. These appear to be unrealistically high values for the expected swept area biomass values that could be obtained from Unit 2. These models where the trawl indices for Units 1 and 2 are treated separately as relative abundance indices are referred to below as the reference models for each species.

When a catch multiplier was estimated for *S. fasciatus* for the years 1985-1994, the model also fitted the data not too badly and the AIC decreased by about 10.5 Units from the reference model (Figure G.3, Table G.1). The estimates of the constants of proportionality for Units 1 and 2 decreased and were both less than 1 for Units 1 and 2 (Table G.1). However, the total amount of reported catch would need to be inflated would be by about 16 times, i.e., the multiplier was estimated at 8.07 when the catch biomass killed to retained ratio for these years was set at a value of 2.0 (Table G.1). While the approximated reported catches for *S. fasciatus* were about 42 kt and 43 kt in 1991 and 1992, the model-predicted catches of *S. fasciatus* approached 700 kt in 1991 and 1992.

When the delay difference model was fitted to the data for *S. fasciatus* and a nonlinearity coefficient was estimated, the model fitted the data slightly better with the AIC dropping by about 1.2 AIC Units from the reference model (Figure G.4). The hyperdepletion parameter was considerably larger than 1, i.e., at 3.28 (Table G.1). The estimate of q for Unit 2 was less than 1, i.e., at 0.81. The estimated depletion was at higher value than when the abundance indices were treated as relative indices, i.e., at 0.24 compared to 0.07 when the abundance indices were treated as relative (Table G.2). However, the estimation was unstable at the function minimized parameter estimates and this model could not be taken seriously.

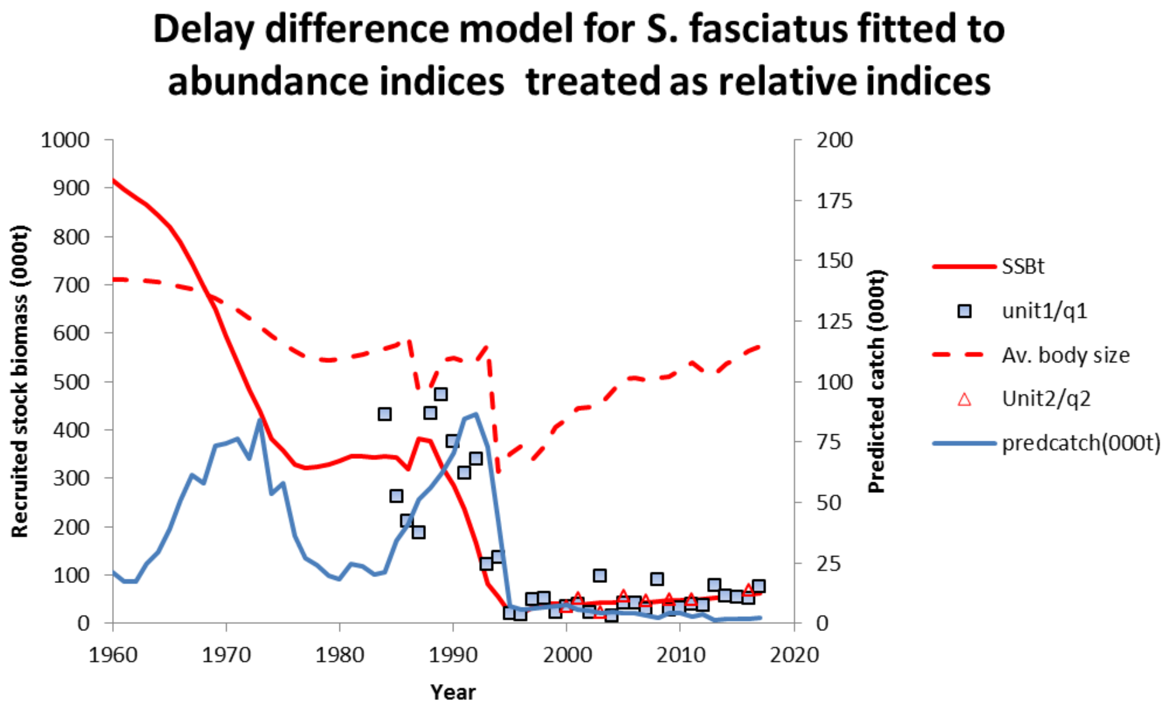
When a range contraction parameter was estimated when fitting the delay difference model to the data for *S. fasciatus*, a slightly better fit to the abundance indices was obtained with the AIC also dropping by about 1.2 Units from the reference model (Table G.1, Figure G.5). The estimate of q for Unit 2 was also less than 1, i.e., at 0.66. The estimate of depletion was also higher at 0.37 compared to when the indices were treated as relative. A stable fit to the data was obtained also.

When the delay difference model was fitted to the *S. fasciatus* index that was the summation of the swept area biomass values for the Unit 1 and Unit 2 trawl surveys and the Unit 1 index from 1984-2017 for years where there was no Unit 2 survey, the q for the Unit 2 index was still very high, i.e., at 3.08 and depletion estimates for both species still very low (Table G.1). The fits to the revised abundance indices obtained were similar to those obtained under the reference model but poorer to the Unit 1 index (Table G.1 and Figure G.6).

When the delay difference model of *S. fasciatus* was fitted to (1) a further revised Unit 1 trawl survey index (i.e., only for years after 2000 when there was no Unit 2 trawl survey) and (2) a trawl index that was the summation of the swept area biomass values for the Unit 1 and Unit 2 trawl surveys with extrapolation of the Unit 2 index for the years 1984-2000, the estimate of q for both indices was below 1 (Table G.1). The estimates of depletion are also higher for both species, i.e., 0.41 for *S. fasciatus* and 0.34 for *S. mentella*, compared to estimates under the reference models (Table G.1).

The resulting fit to the combined 1984-2016 Unit 1+2 index was much better than to either index by itself (Table G.1 and Figure G.7). For example the model fit CV was only 0.10 for *S. fasciatus* and 0.16 for S.mentella for the combined Unit 1+ 2 index compared to about 0.25-0.4 for fits to the separated Unit 1 and 2 indices (Table G.1). The estimate of q for the Unit 1 index dropped to 0.06 and 0.08 the estimate of q for the Unit 2 index dropped to 0.65 and 0.43, for *S. fasciatus* and *S. mentella*, respectively (Table G.1). Thus, combining the Unit 1 and 2 trawl survey indices serves to bring the q estimates for the Unit 1 and 2 trawl surveys to values less than 1 only if average values for the Unit 2 indices for 2000-2016 are extrapolated to the years 1984-1999 when the Unit 2 indices are not available and a combined Unit 1 and 2 index is formulated for 1984-2016.

a.



b.

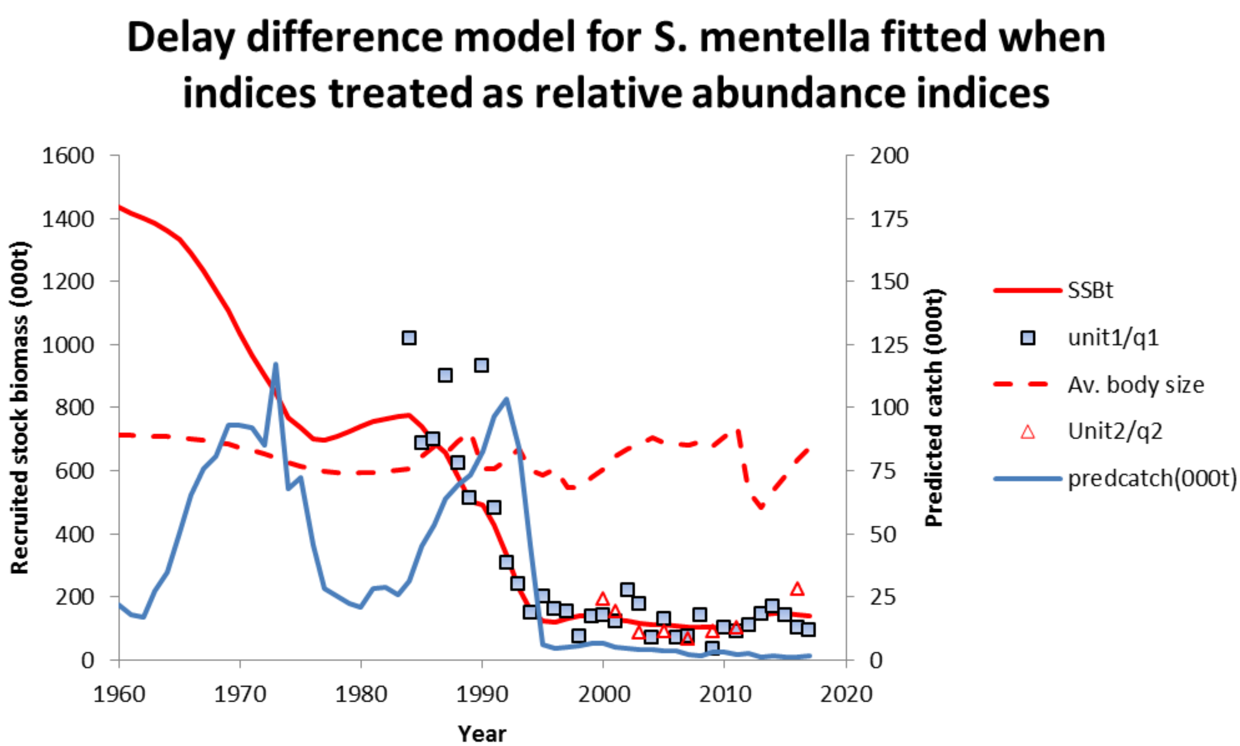
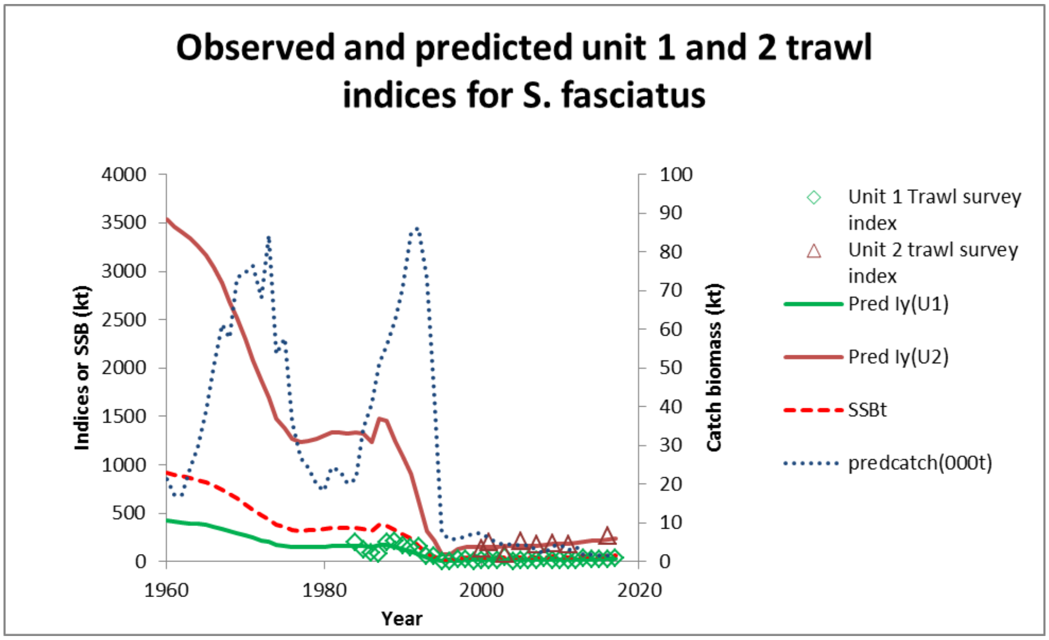


Figure G.1. Fits of a delay difference model to the Units 1 and 2 trawl survey indices for a. S. fasciatus and b. S. mentella. predcatch depicts the model-predicted catch biomass which is given by the reported catch biomass times the ratio of catch biomass killed to retained in the base case operating model.

a.



b.

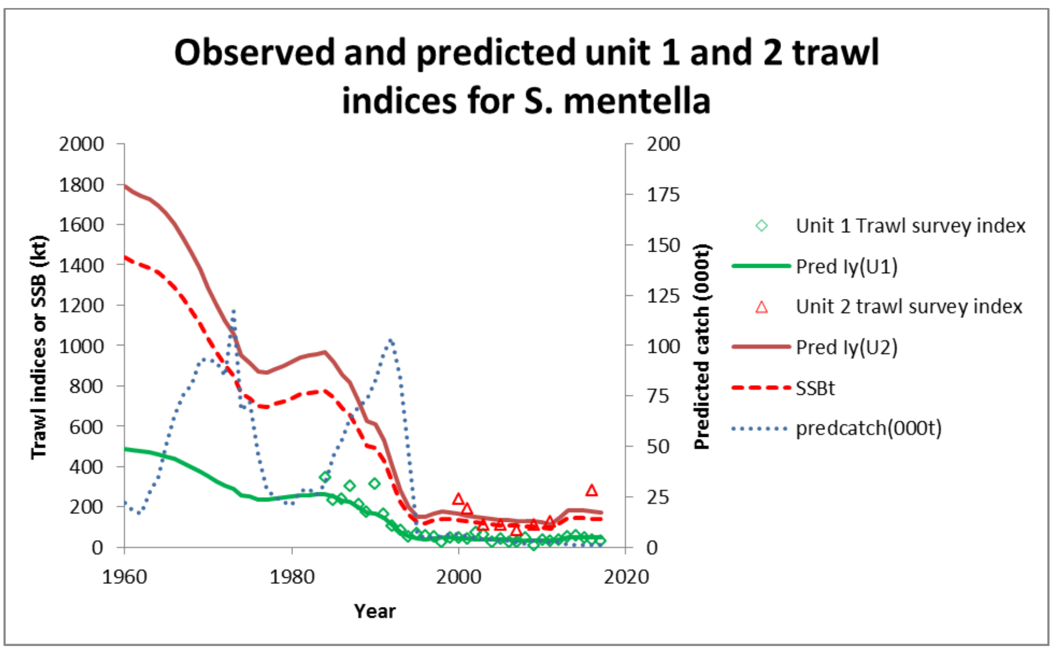


Figure G.2. Time trajectories of observed and predicted trawl indices of abundance for a. S. fasciatus and b. S. mentella.

Table G.1. Estimates of parameters obtained from fitting a delay-difference model (Walters and Martell 2004) to the Units 1 (U1) and 2 (U2) bottom trawl abundance indices. The base case catch biomass killed to retailed ratio was applied in all estimations. Biomass values are in kt. Bzero is the average unfished spawning stock biomass, steepness is the steepness parameter in the Beverton-Holt stock-recruit function, Depletion is the ratio of stock biomass in 2017 to Bzero, npars is the number of parameters estimated included the annual deviates from the stock-recruit function, sigma is the standard deviation in the lognormal likelihood function.

Run	Species	Key attributes	Bzero	Steepness	Depletion	q Unit 1	q Unit 2	Extra parameter estimated	npars	sigma Unit 1	sigma Unit 2 or Units 1+2	AIC	Comments
1	*S. fasciatus*	U1 and U2 index kept separate	917	0.48	0.067	0.459	3.86	#N/A	37	0.45	0.29	-0.45	Indices treated as relative
1	*S. mentella*	U1 and U2 index kept separate	1438	0.58	0.097	0.339	1.24	#N/A	37	0.35	0.30	#N/A	Indices treated as relative
2	*S. fasciatus*	U1 and U2 index kept separate	2247	0.50	0.146	0.075	0.67	8.07	38	0.38	0.28	-10.92	The extra parameter is a multiplier on reported catch biomass 1986-1994
3	*S. fasciatus*	U1 and U2 index kept separate	922	0.53	0.243	4.2E-07	0.81	3.28	38	0.42	0.30	-1.64	The extra parameter is a non-linearity coefficient for the Unit 1 index
4	*S. fasciatus*	U1 and U2 index kept separate	743	0.58	0.373	0.27	0.66	11.6	38	0.43	0.29	-1.69	The extra parameter is a range contraction parameter in which range contraction occurs with stock biomass < 0.5 B0
5	*S. fasciatus*	U1+2, U1, Option 1	890	0.36	0.080	0.524	3.08	#N/A	37	0.51	0.24	#N/A	No imputing of U1+2 index
5	*S. mentella*	U1+2, U1 , Option 1	1133	0.59	0.070	0.701	2.83	#N/A	37	0.28	0.25	#N/A	No imputing of U1+2 index
5	*S. fasciatus*	U1+2, U1, Option 2	1052	0.63	0.411	0.057	0.65	#N/A	37	0.38	0.10	#N/A	1984-99 U1+2 index imputed
5	*S. mentella*	U1+2, U1, Option 2	1701	0.62	0.341	0.084	0.43	#N/A	37	0.35	0.16	#N/A	1984-99 U1+2 index imputed

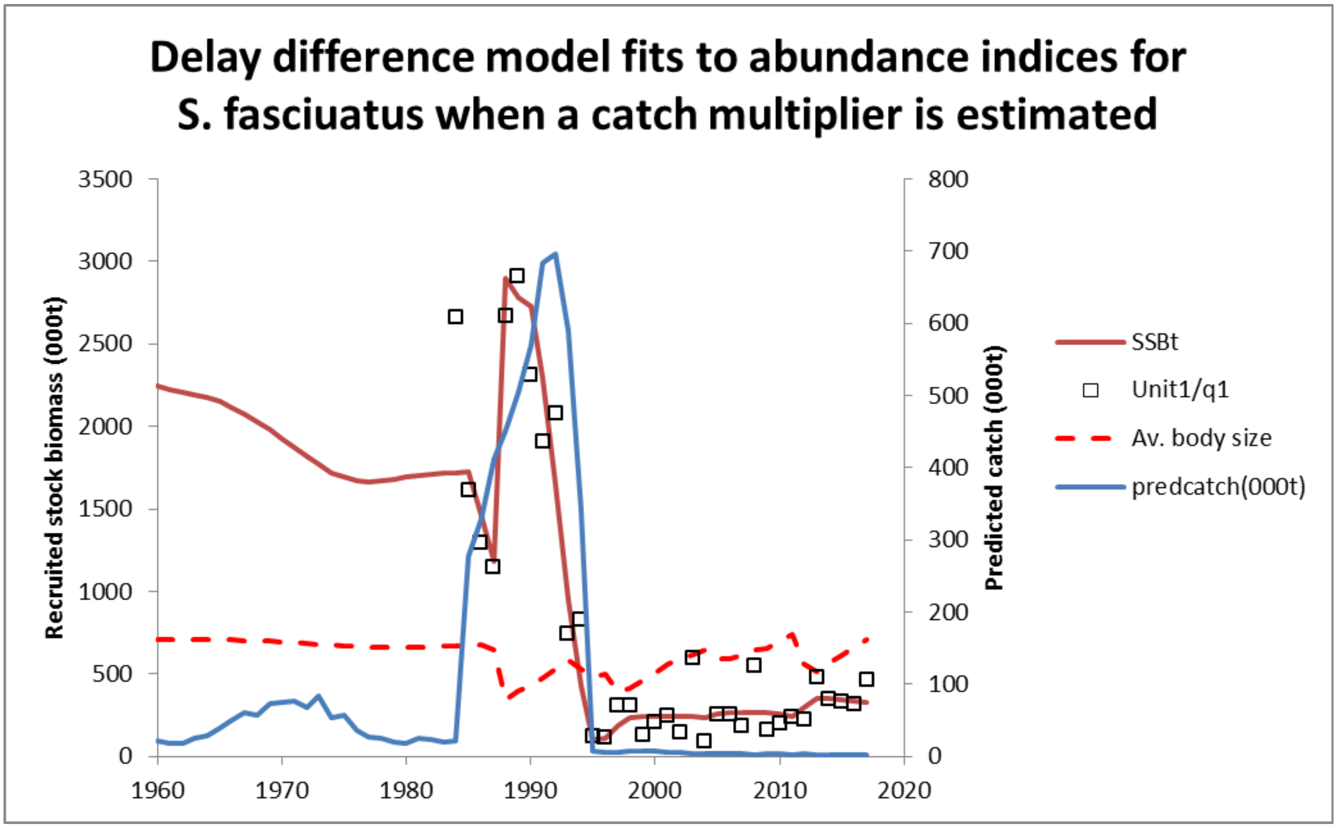


Figure G.3. Plot of the fit of the delay difference model to the Units 1 and 2 trawl survey indices for S. fasciatus when a catch multiplier parameter was also estimated for the years 1985-1994. The model predictions of average body size and predicted catch (predcatch) are also plotted.

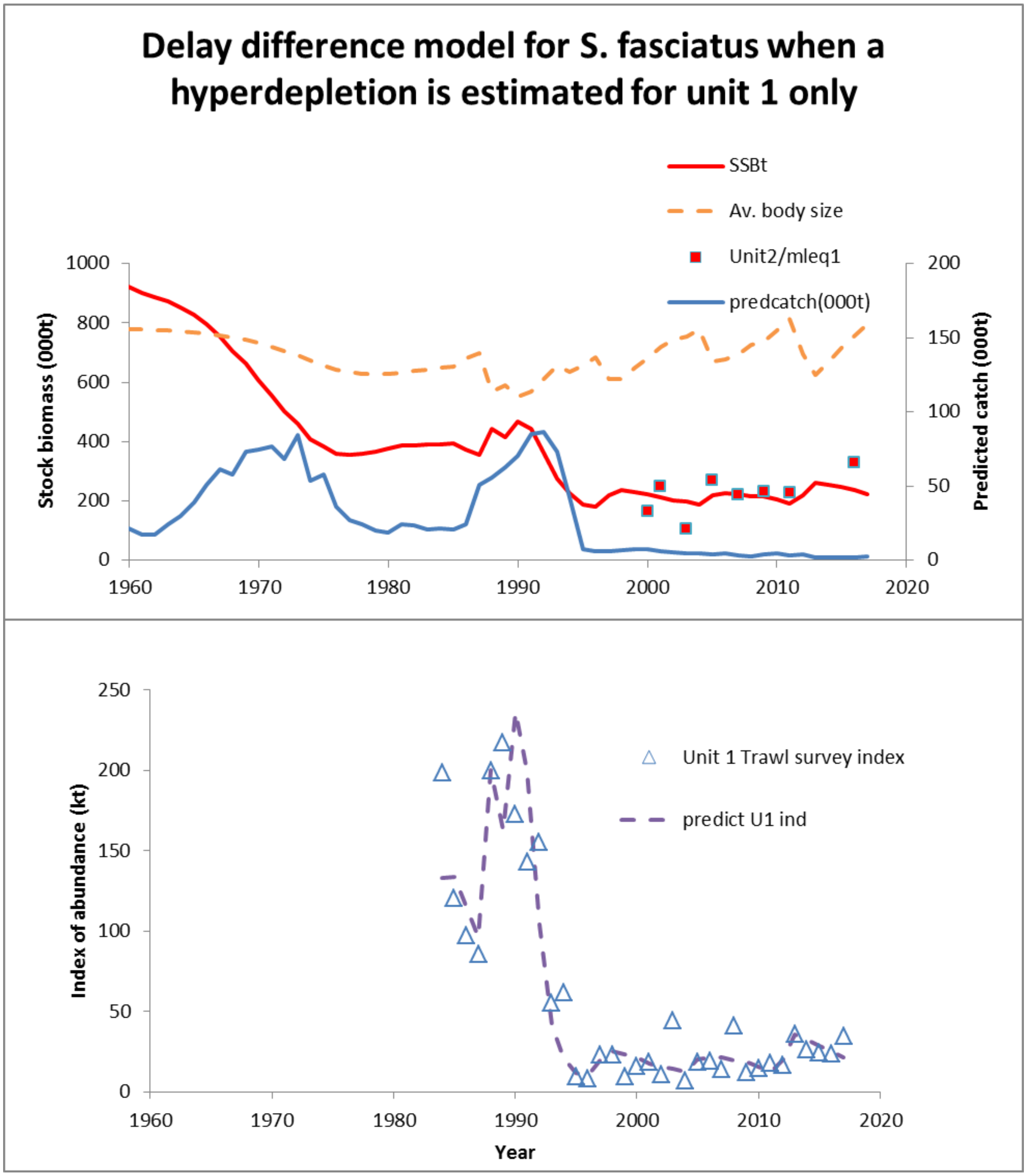


Figure G.4. Plot of the fit of the delay difference model to the Units 1 (bottom panel) and 2 (top panel) trawl survey indices for S. fasciatus when a hyperdepletion parameter was also estimated. The model predictions of spawning stock biomass (SSBt), average body size and predicted catch (predcatch) are also plotted in the top panel.

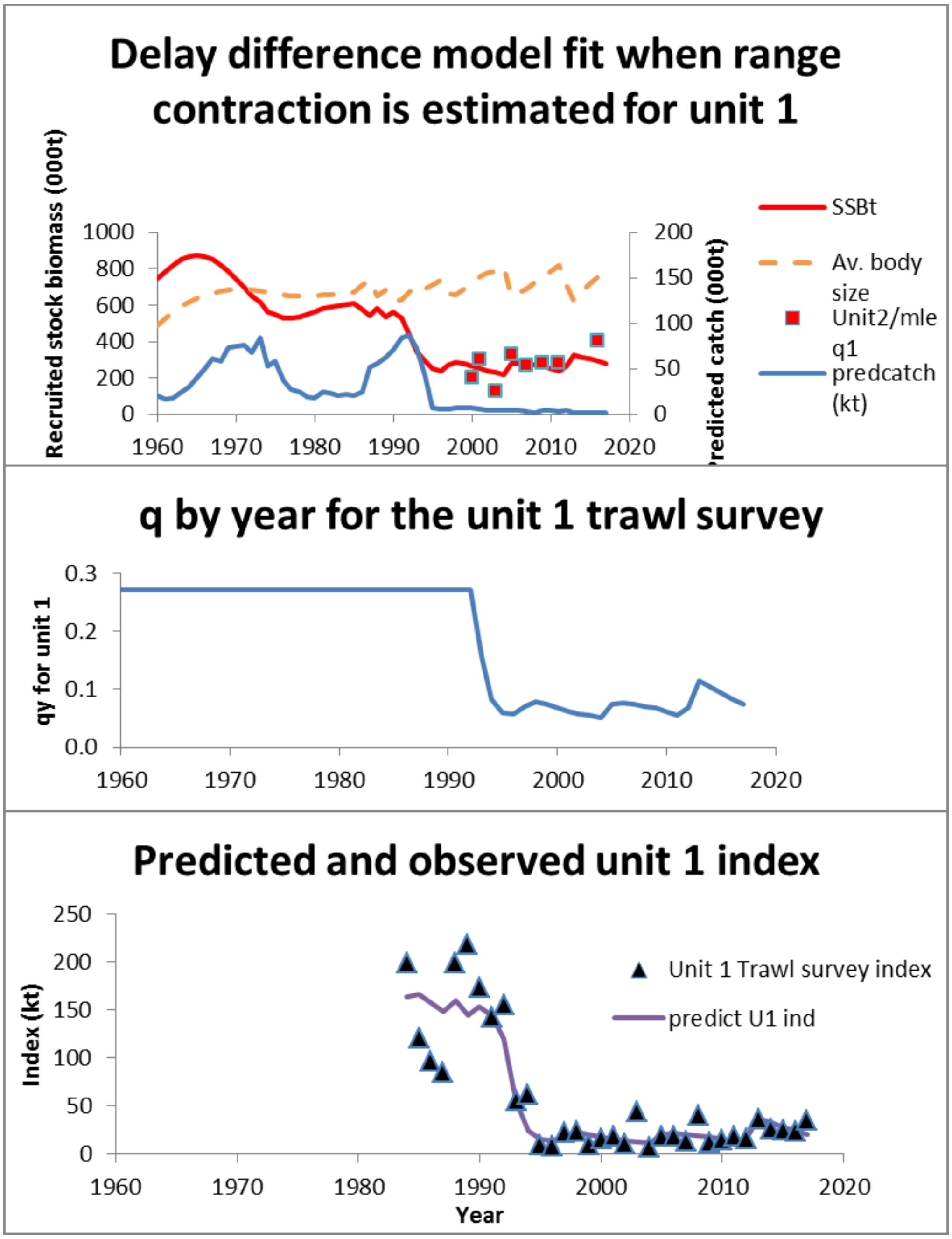


Figure G.5. Plot of the fit of the delay difference model to the Units 1 (bottom panel) and 2 (top panel) trawl survey indices for S. fasciatus when a hyperdepletion parameter was also estimated. The middle panel shows the model predictions of the constant of proportionality for the Unit 1 trawl survey. The model predictions of spawning stock biomass (SSBt), average body size and predicted catch (predcatch) are also plotted in the top panel.

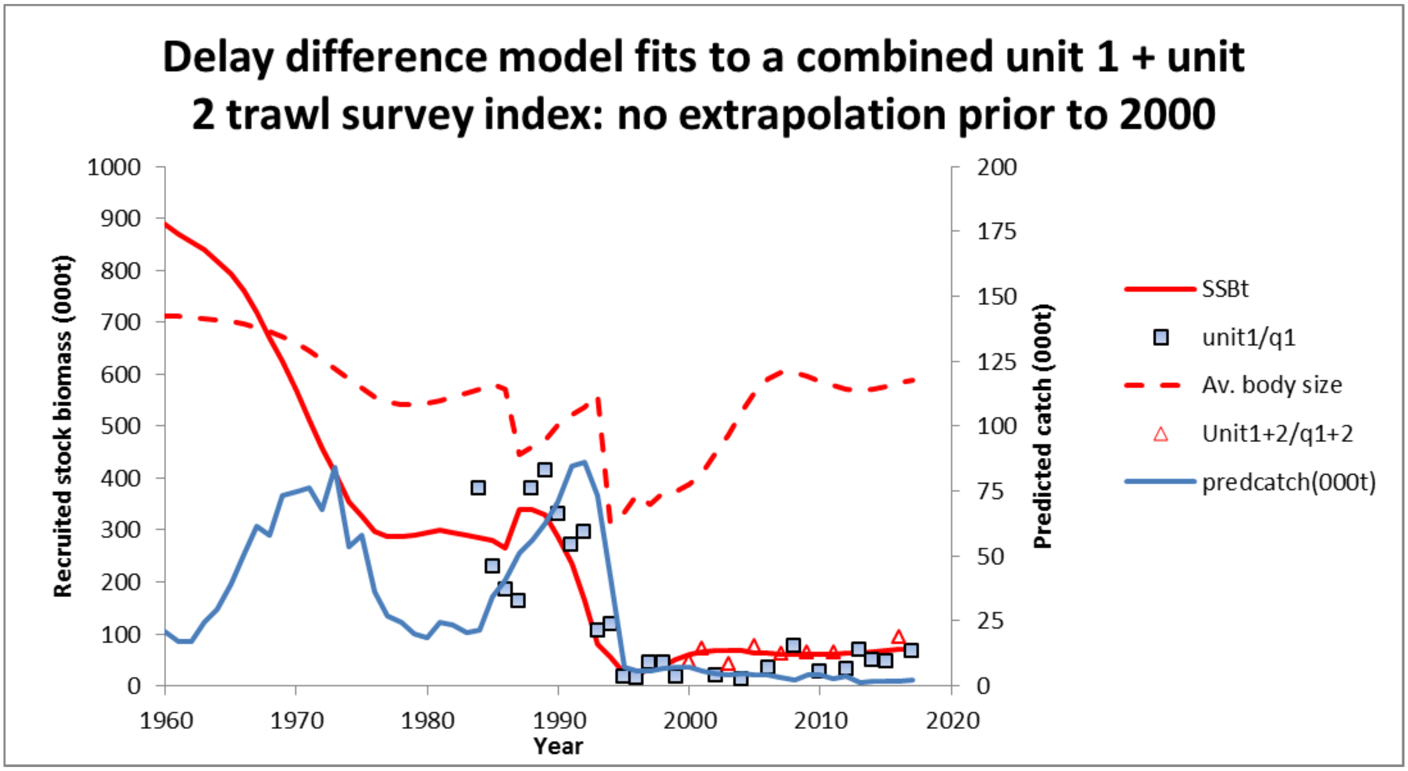
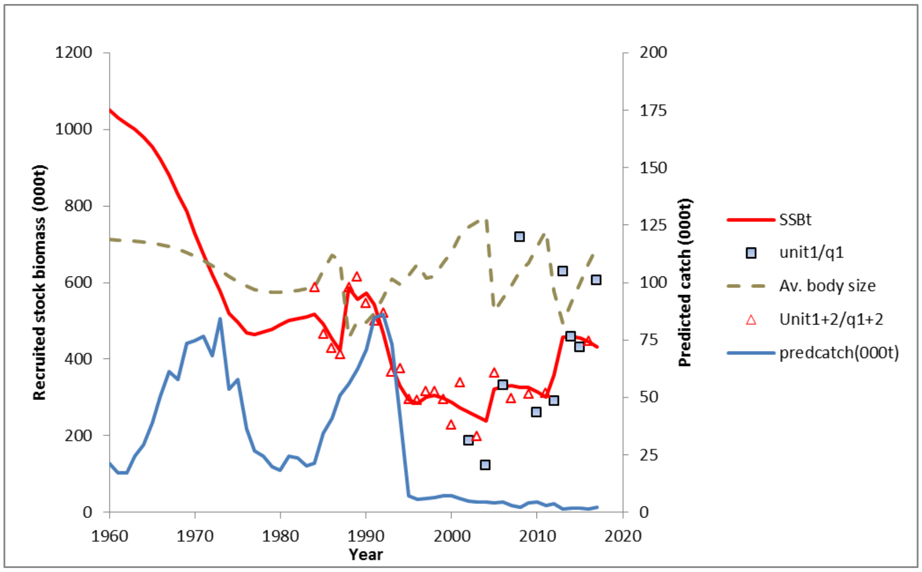


Figure G.6. Plot of the fit of the delay difference model to (1) a revised Unit 1 index 1984-2017 (excluding years where there is a Unit 2 trawl survey index) and (2) a combined Unit 1 and Unit 2 trawl survey index 2000-2016 for S. fasciatus without any extrapolation of the Unit 2 index for years prior to 2000.

a.



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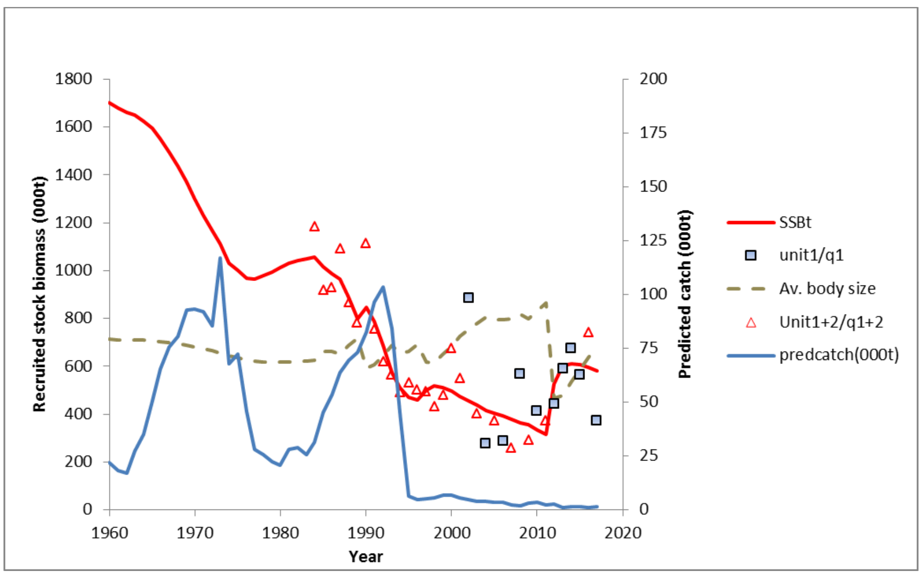

Figure G.7. Plot of the fit of the delay difference model to a revised Unit 1 index and combined Unit 1 and Unit 2 trawl survey index for a. S. fasciatus and b. S. mentella 1984-2016 with extrapolation of the Unit 2 index for years prior to 2000 to extend the combined index back to 1984.

We investigated also spatial patterns in the Unit 1 trawl survey data to evaluate the plausibility of the hyperdepletion hypothesis. The occupancy-abundance (OA) relationship for Unit 1 Sebastes spp was examined in the trawl survey data 1990-2017. Different minimum sizes were considered ranging from 22 cm (minimum fishery catchable size) to 27 cm (preferred fishery size). All of these OA relationships were positive with significant slopes. For example, the slope was estimated at 0.16 with a p value of 0.002 and R^2 of 0.31 for the fitted linear model for lengths ≥ 22cm (Figure G.8). A positive slope indicates that as the population increases in abundance, it spreads out and occupies more space and vice versa. The relatively low R-squared values suggest that abundance is not the most important descriptor of occupancy but the significant range contraction with decreases in abundance potentially makes redfish more susceptible to fishery depletion at low stock size.

Plots of spatial catch rates over blocks of years from 1984-2017 (Figures G.9-G.12) are not inconsistent with the hypothesis of range contraction in Unit 1. The spatial distribution of fairly high catch rates of mature redfish in Unit 1 is well spread across Unit 1 in years of high abundance, i.e., prior to 1995, but becomes confined more to the eastern portion of Unit 1 in years of lower abundance following 1995 (Figures G.3 and G.5). The spatial extent of high catch rates of mature redfish in Unit appears to systematically decrease from the mid-1980s to 2000 and then increase in the final block, 2011-2017. When the stocks are low in abundance from 2000 onwards, the spatial distribution and survey catch rates remain fairly stable in unit 2 for mature *S. mentella* and *S. fasciatus* where survey records are available from 2000-2016 (Figures G.3 and G.5).

DISCUSSION AND CONCLUSIONS

Observed decline in the trawl survey index of abundance for both redfish species in Unit 1 were larger than the catch biomass removed suggests that either the reported catches underrepresent the actual catches during the period of severe depletion in the late 1980s and early 1990s (e.g, Duplisea 2018) or that the trawl survey index in Unit 1 could be hyperdepleting, i.e., depleting proportionally more than the actual depletion in stock size. The above reported analysis suggested that actual catches would need to be about 16 times the reported catches for *S. fasciatus* from 1988 to 1994 to predict the declines seen in the Unit 1 index and also give estimates of q less than 1. While it is known that there was some fairly substantial underreporting of catches in this period (i.e., based on Duplisea 2018, expert judgement put this at a factor of 2), it appears to be implausible that actual catch biomass could be about 16 times the reported catch biomass in this period. However, discarding of small fish while those fish still have considerable potential somatic growth before recruitment is a factor that could appear to inflate the degree of under-reporting of recruited fish size latter and clearly small fish discards and under reporting was a dominant feature of the fishery in the 1980s and 1990s (Duplisea 2018).

The above reported analyses which evaluated the range contraction hypotheses however showed plausible fits of the model to the data under this hypothesis in each of the three different approaches to testing for range contraction that were considered. Also, the positive correlation between occupancy and area occupied in the Unit 1 trawl survey and the apparent systematic changes in the spatial extents of high trawl survey catch rates of mature redfish in Unit 1 and lack thereof in Unit 2 are consistent with the range contraction hypothesis.

If range contraction had occurred and affected only the Unit 1 trawl index, this may be problematic for estimation of stock biomass trends because the fitting of the operating models to the Unit 1 and 2 trawl indices could cause the operating models to estimate more severe declines than have actually taken place and also cause the estimates of stock productivity to be lower than they actually are. The model for how the index is related to abundance would also be incorrect in the base case operating model since it would fail to predict hyper-depletion in the index as the stock dropped and hyper-expansion of the index as the stock increased and instead incorrectly presume that the index reliably tracked stock size. This potentially inaccurate representation of the behaviour of the index could cause poor performance of management procedures that assumed that the index was instead directly proportional to abundance. For example, as the stocks increased, the Unit 1 index would increase disproportionately more than the actual rate of increase of the stock, cause the catch limits specified by the adopted MP to increase excessively and lead to further overfishing and further unanticipated stock depletion within the next two decades.

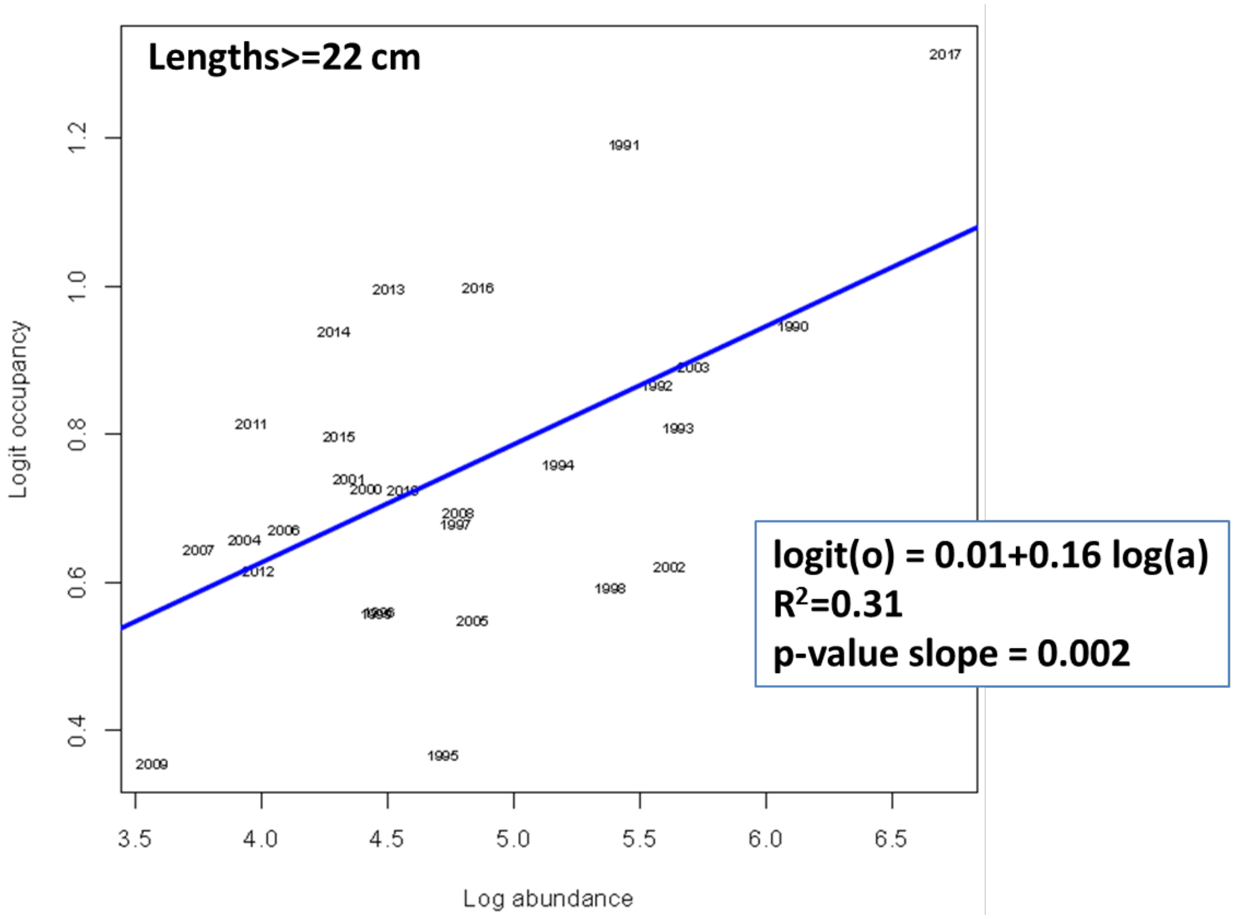

Figure G.8. Occupancy-abundance of Unit 1 redfish (Sebastes spp) from the DFO survey. Linear model fits are presented on each graph showing how occupancy changes with abundance for different size components of the redfish population. The minimum size considered was 22 cm which is a fully recruited fish to the Unit 1 survey gear.

As indicated above, one potential mechanism for hyperdepletion in a trawl survey index is a change in how the stock is distributed spatially as the stock depletes from high abundance to low abundance. This is often known as range contraction (Walters and Martell 2004). This hypothesis would imply that the stock is broadly distributed when it is abundant but as stock declines its spatial range contracts to a much smaller area. So when the stocks were abundant up until the mid-1980s, they occupied both Units 1 and 2 and when they began to deplete the relative occupancy of Unit 1 declined more so than in Unit 2; yet, from 2000 on when the Unit 2 trawl survey indices are available, the relative abundance in both Units 1 and 2 have remained relatively stable. This could predict the hyperdepletion in the Unit 1 trawl survey index and still allow the Unit 2 index to stay much less depleted. Since the trawl survey indices from Units 1 and 2 are based on 30 cm+ fish and Unit 2 has been surveyed only once every two years in recent years, it is too soon to evaluate whether the Unit 2 index will remain hyperstable and the Unit 1 index will show signs of hyper-expansion as stock size for both species increases.

Another possible cause of hyperdepletion in the Unit 1 trawl survey index is that the redfish population in Unit 1 may become less vulnerable to the Unit 1 trawl survey as abundance declines. For example, as the Units 1 and 2 redfish stocks declined, they could occupy with higher frequency untrawlable areas or areas where trawl survey gear works less efficiently in Unit 1, which could include deeper or more rugged zones within Unit 1. However, if this were to be the case, the same behaviour would be expected of redfish inhabiting Unit 2 where so far there exists no evidence of either hyperdpletion or hyperexpansion. It therefore appears to be more credible that range contraction is the main mechanism responsible for the apparent hyperdepletion of the Unit 1 trawl survey indices for *S. mentella* and *S. fasciatus*.

The range contraction hypothesis could explain why the Unit 1 trawl survey index for both species dropped rapidly in the 1990s with the annual drops being much larger than the estimated catch removals. And it could explain why the Unit 2 swept area biomass from the trawl survey has shown much higher stock biomass estimates than the Unit 1 index in the same years since 2000, despite the area of Unit 2 being similar to that of Unit 1.

When range contraction affects one survey with a very long time series which covers the period from before and then after the stock decline and range contraction occurred and there exists a second trawl survey that started after the range contraction occurred but in an adjacent area at the heart of the range contraction, this may cause difficulties for the stock assessment, as have been previously witnessed.

An important question is how to deal with the issues with the trawl survey data and try to produce the most accurate assessment using all of the data.

One approach that was explored in this Appendix was to keep the two abundance indices separate and model hyperdepletion in the Unit 1 index but no range contraction in the Unit 2 index. We tried to do this two different ways using the delay-difference population dynamics model (Walters and Martell 2004). In the first version, we included a non-linearity coefficient, β, to predict hyperdepletion in the Unit 1 survey index. This model also fitted the trawl survey indices and also predicted severe hyperdepletion, with the β estimated at 3.3 ($\beta = 1$ implies linearity, $\beta > 1$ implies hyperdepletion). However, the estimation result was numerically unstable. In a second version, we modelled q for Unit 1 to change as stock size relative to average unfished stock size changed. This used just one additional parameter to determine the extent of range contraction. This model version fitted the data and predicted severe range contraction. In both the hyperdepletion and range contraction models we put a strong prior on the constant of proportionality for the Unit 2 survey index, with a mean at 0.5 and standard deviation of 0.25. The estimated values for q were both less than 1 when the two alternative models were fitted to the data for *S. fasciatus*. One of these models (with the range contraction term) predicted high Bzero and low steepness and the other (with the non-linearity term) predicted lower Bzero and higher steepness. Both models predicted that the stocks did not decline nearly as much as under the base case model that ignored the possibility of range contraction and assumed that the Unit 1 index could be treated as a relative index of abundance.

The second approach to addressing the issue of range contraction of the stocks is to produce a single abundance index that combines the trawl indices from the two surveys in Units 1 and 2. The only way this could be got to work without producing anomalous results (i.e., estimates of q larger than 1) was by extrapolating a mean value for the Unit 2 index for the years 1984-1999 when only the Unit 1 trawl index was available and summing the mean value for the Unit 2 index with the Unit 1 index for those years. As mentioned above the assumption that abundance is hyperstable in Unit 2 is a big assumption and should be treated only as one potential scenario if this approach were to be considered for the development of a new operating model. However, the estimates of q were all less than 1, the best fits to the data were obtained using this approach and the estimation results were numerically stable.

If there is to be another MSE for this stock, it may be appropriate to include alternative operating models that predict the effects of range contraction in the Unit 1 survey index, with either a model with explicit range contraction for Unit 1 or a model that combined the Units 1 and 2 indices and made assumptions about the degree of hyperstability of abundance and the Unit 2 trawl index. New candidate management procedures could be formulated that can avoid the unintended consequences of range expansion and contraction by e.g., being based on the summation of the Unit 1 and Unit 2 trawl survey indices. Further research could be conducted that analyzes various available data to improve understanding of the potential for range contraction in Units 1 and 2 redfish.

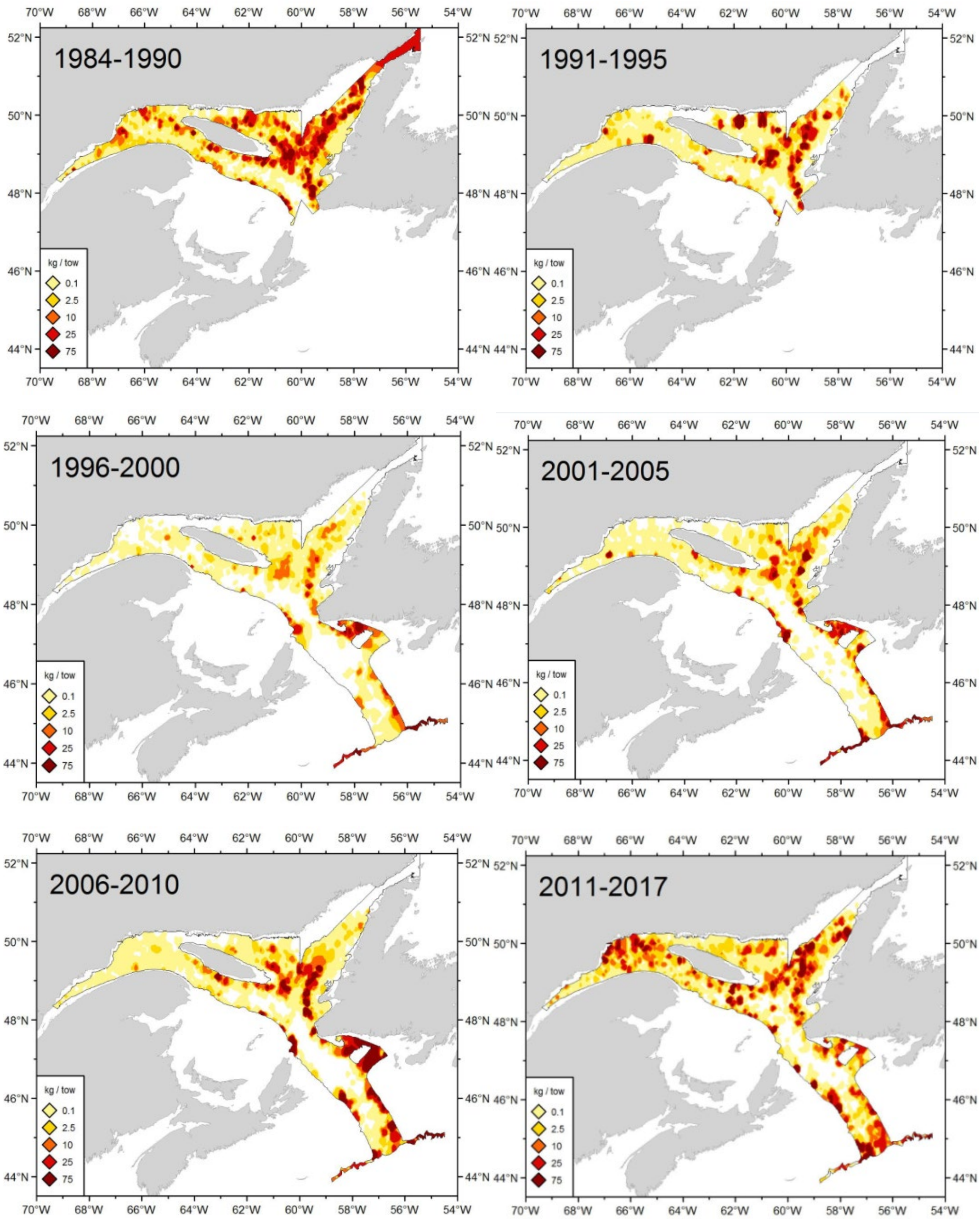

Figure G.9. DFO Unit 1 trawl survey and GEAC Unit 2 trawl survey catch rates for immature S. fasciatus averaged for 1984-1990, 1991-1995, 1996-2000, 2001-2005, 2006-2010, 2011-2017.

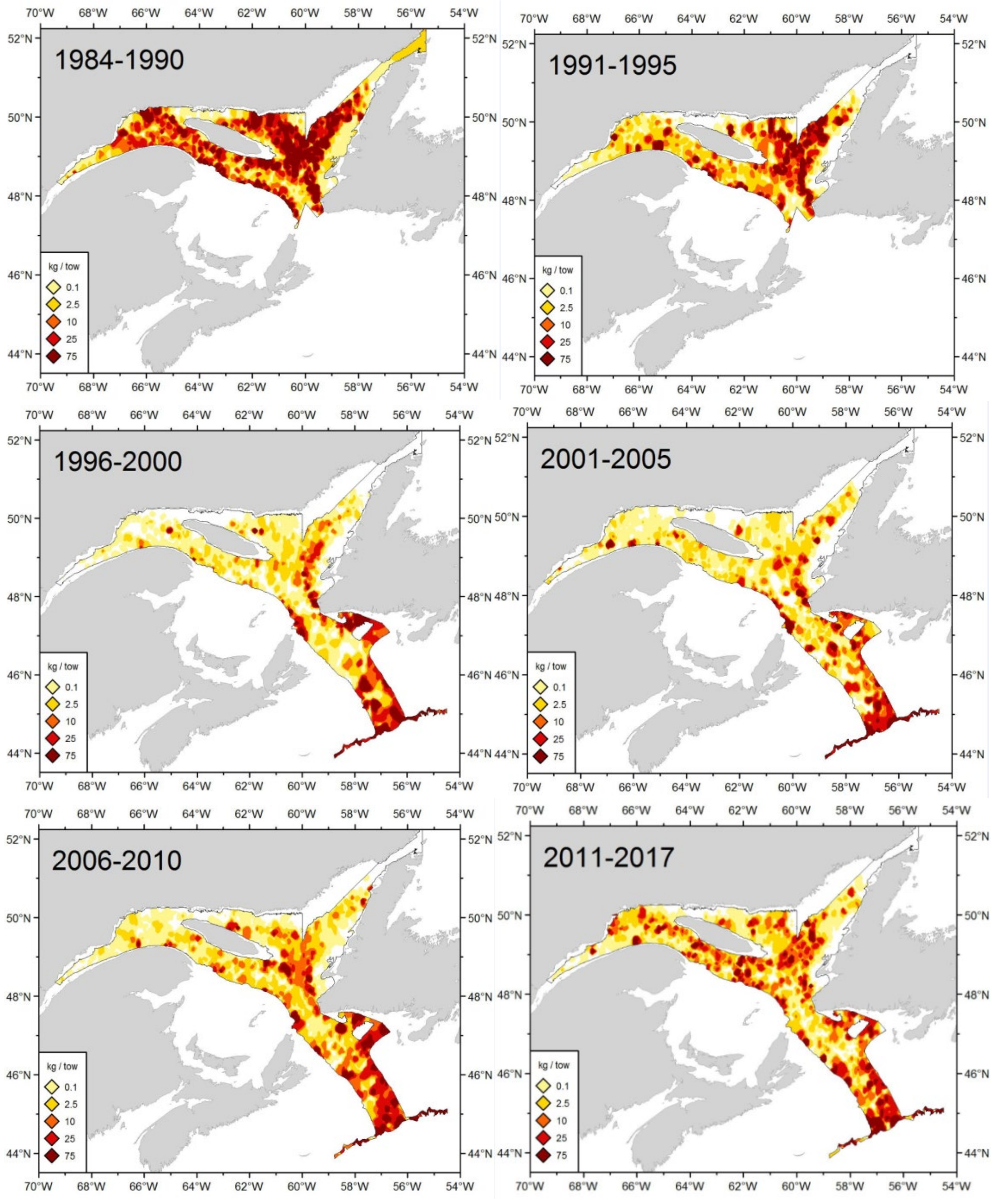

Figure G.10. DFO Unit 1 trawl survey and GEAC Unit 2 trawl survey catch rates for mature S. fasciatus averaged for 1984-1990, 1991-1995, 1996-2000, 2001-2005, 2006-2010, 2011-2017.

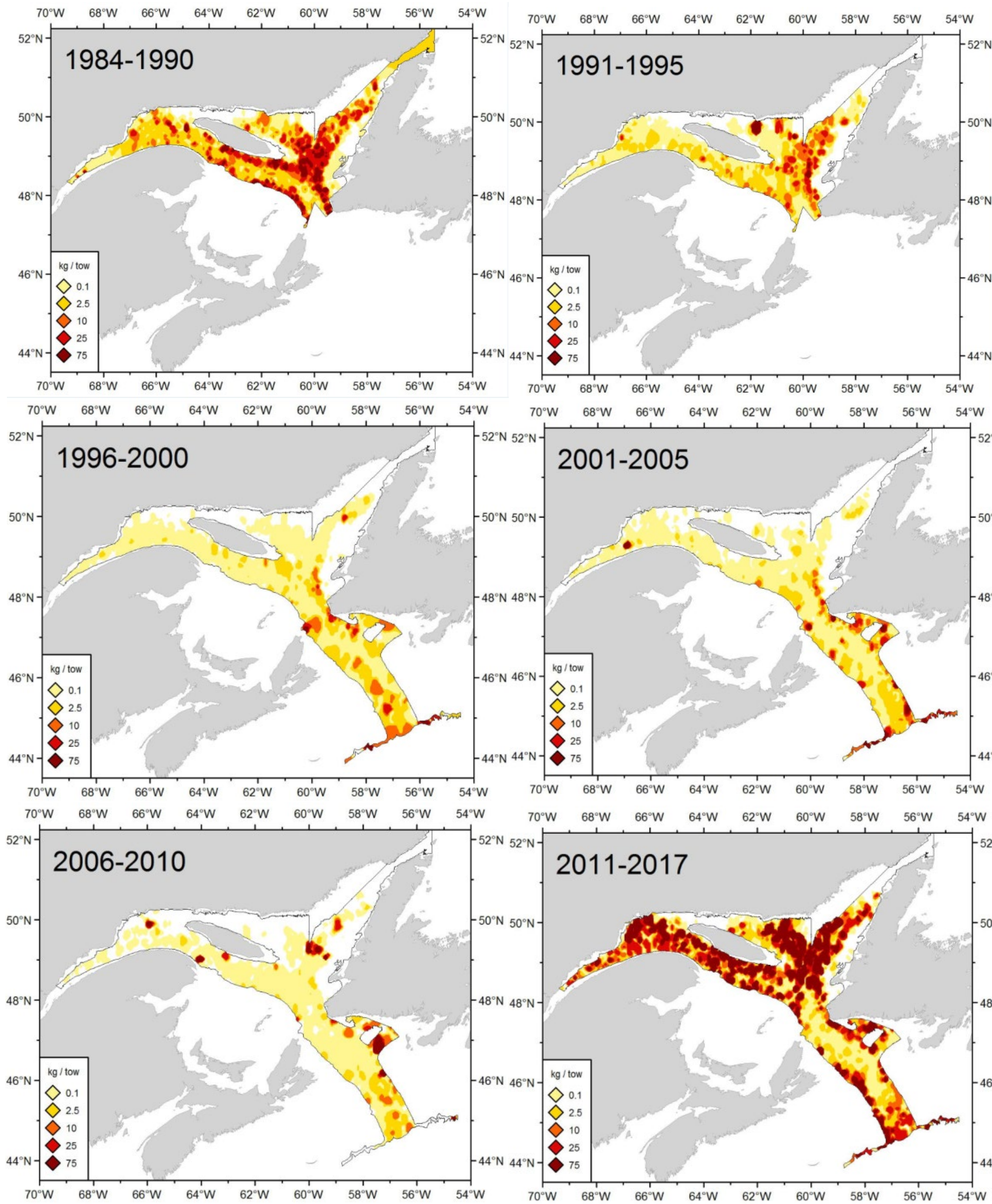

Figure G.11. DFO Unit 1 trawl survey and GEAC Unit 2 trawl survey catch rates for immature S. mentella averaged for 1984-1990, 1991-1995, 1996-2000, 2001-2005, 2006-2010, 2011-2017.

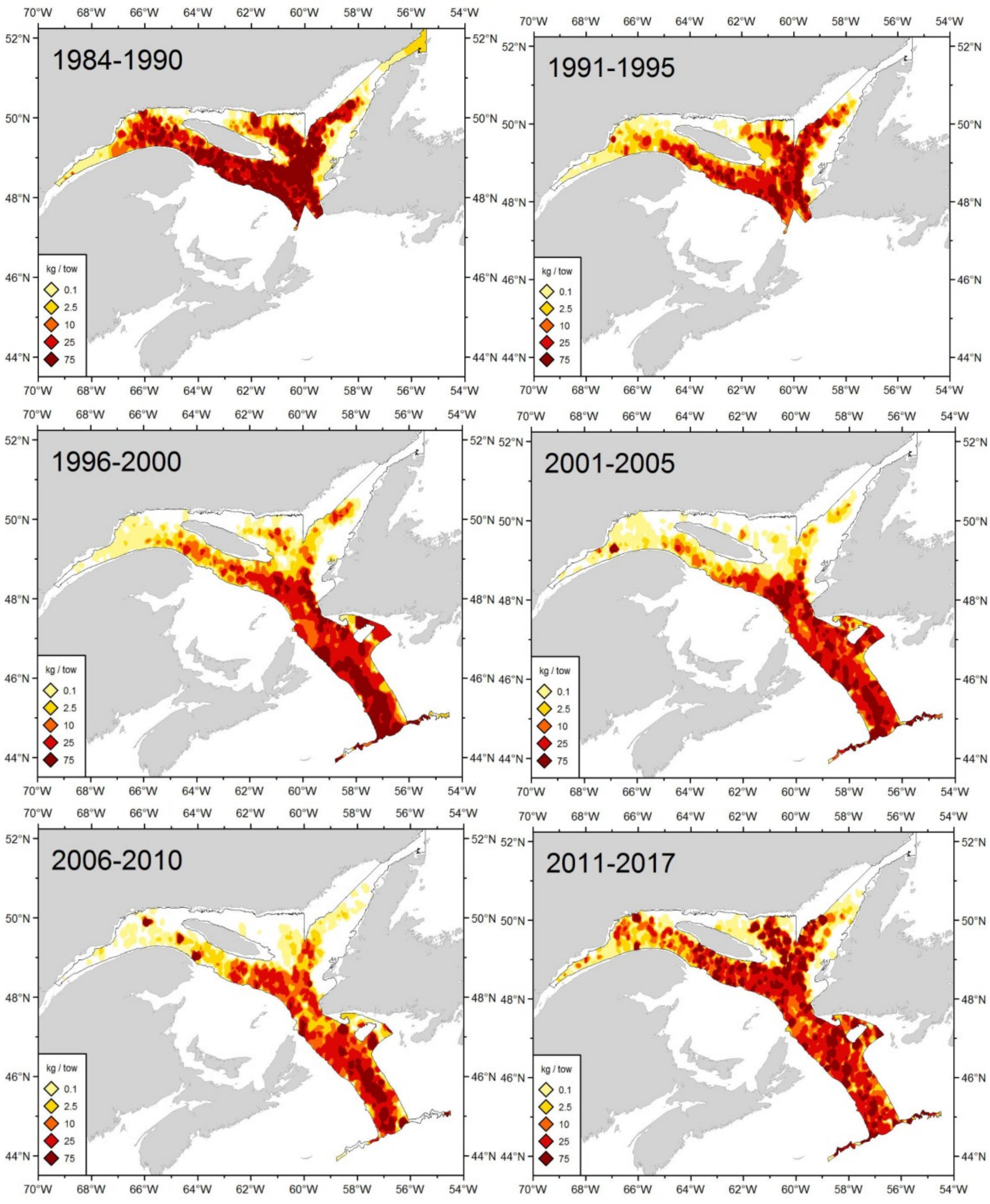

Figure G.12. DFO Unit 1 trawl survey and GEAC Unit 2 trawl survey catch rates for mature S. mentella averaged for 1984-1990, 1991-1995, 1996-2000, 2001-2005, 2006-2010, 2011-2017.

APPENDIX H: EVALUATION OF MANAGEMENT PROCEDURES WITH CATCH LIMITS IMPLEMENTED BY SPECIES

During the March 2018 meeting of the redfish Working Group, stakeholders requested an evaluation of some additional management procedures (MPs) in which the species composition of catches was accurately assessed and catch limits were specified for each species. The interest was in evaluating the extent to which total catch limits could be increased without compromising conservation objectives by setting catch limits by species. To evaluate these types of MPs it was assumed that catches were split perfectly by species in future years and that catch limits for each species would be obtained by applying the harvest control rule to the Unit 1 trawl survey abundance indices for each redfish species. It was also assumed that the fishery would catch 1.1 times the specified catch limit for each species in each future year. Previously specified MPs 43, 44 and 45 were modified to implement separate catch limits by species and the new corresponding MPs were called MPs 46, 47 and 48.

Table H.1. Specifications for three management procedures that specified catch limits (CL) by species.

No.	HCR Start	Notes on MP and implementation
46	2018	Catch limits of 7.5, 10, 15 and 20 kt for 2018-2021, and the CL being set at 80% of the HCR 2022-2057 for each species.
47	2018	Catch limits of 5 kt for 2018-2019, and the CL being set at 80% of the HCR 2020-2057 for each species.
48	2018	Catch limits of 5 kt for 2018-2021, and the CL being set at the HCR 2022-2057 for each species.

Each HCR was tuned separately for each species similarly as the HCR developed for the combined species MPs. The slope of the HCR was adjusted to meet conservation constraints under the base case operating model assuming perfect implementation. This was so that that $u_y < u_{msy}$ in greater than 95% of the future years but was as close to u_{msy} as possible for each species.

RESULTS

Under MPs 46-48, catch limits and catches were projected to be higher than those obtained under the corresponding MPs 43-45 (Figures. H.1.-H.3, Table H.1). This was obtained without compromising the conservation objectives of the core operating models. Average catches were larger for the entire 40-year time series for all three MPs that specified catch limits by species. For MPs 46 and 47 the average increases in catches were by a factor of 1.4-1.9 times the catch biomass values that could be obtained from MPs 43 and 44, with average annual catch biomass up to 58 and 60 kt larger in the late 2020s (Table H.1). This would amount to 984 and 982 kt more catch biomass in the next forty years under MPs 46 and 47, respectively compared to that obtainable under MPs 43 and 44. For MP 49, the average increases in catches were by a factor of 1.4-2.0 times the catch biomass values that could be obtained from MP45 with average annual catch biomass up to 76 kt larger in the late 2020s (Table H.1). This would amount to 1124 kt more catch biomass in the next forty years under MP 48 compared to that obtainable under MP 45.

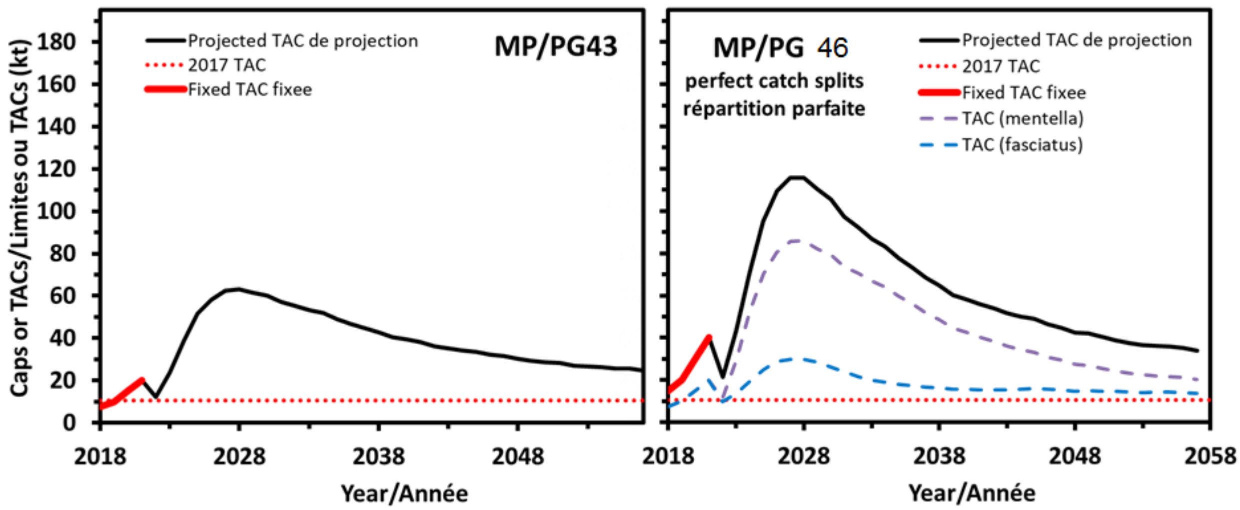

Figure H.1. Plots of the average total allowable catches (TACs) under management procedures 43 and 46 under the base case operating model.

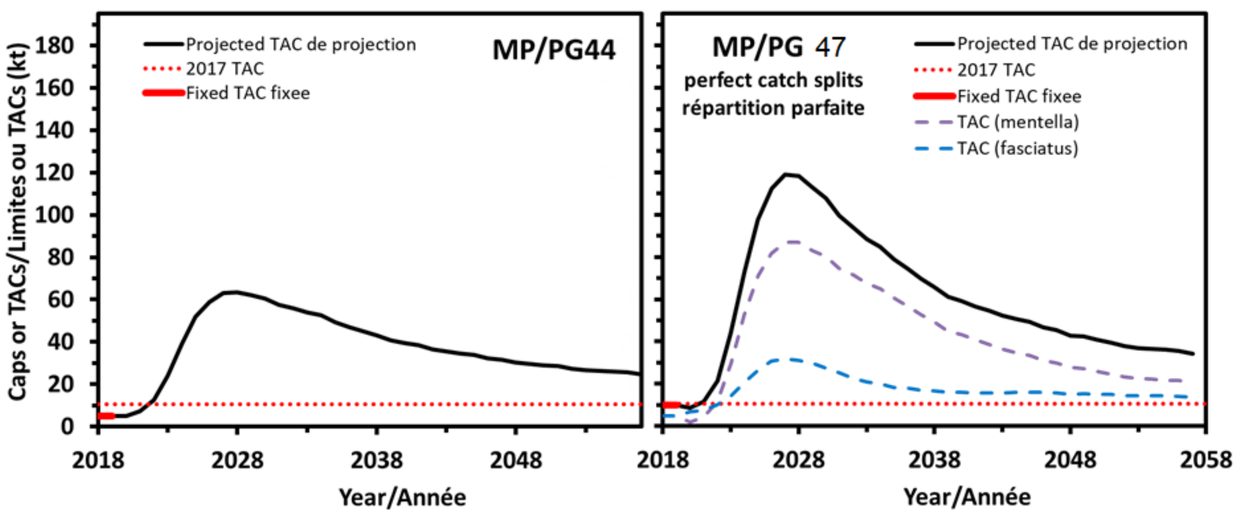

Figure H.2. Plots of the average total allowable catches (TACs) under management procedures 44 and 47 under the base case operating model.

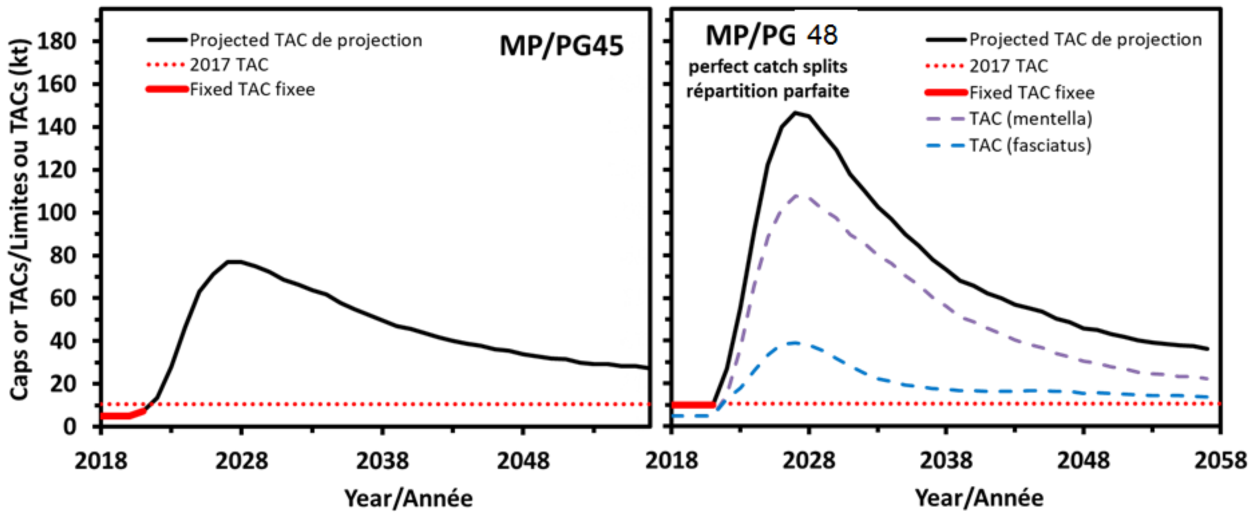

Figure H.3. Plots of the average total allowable catches (TACs) under management procedures 45 and 48 under the base case operating model.

CONCLUSIONS

MPs 46-48 that set catch limits by species met all conservation objectives under all core operating models and gave annual catches 1.4-2.0 x that under MPs without catch splitting by species. Being able to implement MPs that specify catch limits by species could allow larger catches almost immediately and still meet conservation objectives compared to MPs that specify catch limits lumped across the two species. However, the simulations performed assumed perfect implementation of MPs without any error in assessing the species composition of the catches and the computations were done only for the core operating models. Given that it is likely that there will be some error in assessing the species composition in future catches, should this be attempted, the expected increases in catch biomass from an MP that specified a catch limit by species and still met the conservation objectives could be smaller than those reported in this analysis.

Table H.1. Average improvements in catch limits from MPs that specified catch limits by species.

MP TAC by species	MP TAC combined	Average improvement in TAC	Maximum annual improvement in TAC	Total gain in TAC over 40 years
46	43	1.4-1.9x	58 kt / year	984 kt
47	44	1.4-1.9x	60 kt / year	982 kt
48	45	1.4-2.0x	76 kt / year	1124 kt